An Evaluation of the Nitrate-Nitrogen Loading to Groundwater from Two Unsewered Subdivisions near Stevens Point, Wisconsin

by

Peter D. Arntsen

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APPROVED BY THE GRADUATE COMMITTEE OF:

Dr. Byron H. Shaw, Committee Chairman
Professor of Soil and Water Science

Dr. David L. Ozsvath
Associate Professor of Geology and Water Science

Dr. James A. Bowles
Professor of Soil Science

Abstract/Executive Summary

The impact of residential development on groundwater quality is a topic of concern in Wisconsin due to the urbanization of the wellhead recharge areas of municipal water supply wells. In an effort to better understand these impacts, two unsewered subdivisions in Central Wisconsin have been the focus of extensive research and monitoring. Analysis of water samples obtained from approximately 250 monitoring wells and 100 private wells gives a view of groundwater quality as it passes beneath the subdivisions. With this information, it was possible to document the quality of groundwater recharged from subdivision sources. The results were, in turn, used to (1) test the accuracy of certain computer simulation programs, (2) better define the variables that contribute to their results, and (3) assess the validity of using these models as a basis for land use development plans.

The computer simulation package used in this study is the BURBS program (Hughes and Pacenka, 1985) created at Cornell University. This program uses 18 different variables to predict the nitrogen concentration of the groundwater recharge from residential areas. The factors considered are those pertinent to nitrogen and water budgets.

Questionnaire results obtained from the subdivision homeowners early in the study, in addition to follow-up information, were used to supply values for variables such as lawn fertilization rates and number of people per household. Both subdivision areas were mapped and digitized into pcARCINFO (a geographic information system package) and areal values for land use characteristics were calculated using this

program. Other values such as nitrogen leached from turf and in wastewater were estimated from field research and literature sources.

Several runs of the BURBS program with differing scenarios were compared to actual field data with the intent of determining the accuracy of the program's output and refining the estimated values to make the model more applicable to in-field conditions.

Chemistry results of groundwater samples obtained from the monitoring wells showed that the water quality originating from areas with little human influence is generally quite good in the study areas (e.g., concentrations of nitrate-N, chloride, and sodium were around 1 mg/L and concentrations of phosphate-P below 0.005 mg/L). However, much higher concentrations of these chemicals were detected in the groundwater that originated from agricultural fields, septic system drainfields, and fertilized lawns (e.g., concentrations of nitrate-N > 30 mg/L, chloride > 60 mg/L, sodium > 50 mg/L, and phosphate-P > 5 mg/L). Considerable variations in groundwater quality was noted within and downgradient of the subdivisions. Seasonal and yearly variability were also observed (due to varying amounts of groundwater recharge).

Once the values for the variables used in the BURBS model were accurately defined, the model yielded results similar to those obtained using data measured in the field. The agreement between the two methods was better for one subdivision than the other, primarily because one of the monitoring networks was more effective at monitoring the groundwater originating from its respective subdivision.

The calibrated model was used to estimate the average housing density required so as not to exceed the 10 mg/L nitrate-N drinking water standard in the groundwater recharge. The calculated housing density for one subdivision was 1.1 homes/hectare; the housing density for the other was 1.7 homes/hectare. The primary difference between the two subdivisions was the relative amount of natural land use within the two subdivisions (recharge from natural lands will tend to dilute contaminants originating from within the subdivisions).

Simulations were also run assuming the same conditions as one of the subdivisions with the exception that less recharge occurs (due to finer-textured soils and/or routing surface runoff out of the subdivision). Under these conditions the housing density required to be below the 10 mg/L nitrate-N standard was 0.5 houses/hectare.

The BURBS nitrogen and water mass balance model can be a useful tool for the planning of subdivision developments. The program does not, however, provide information regarding the nitrate-N concentration at any particular location; therefore, it can not be used for locating drainfields and water-supply wells to assure the latter are not impacted by the former. Locating the wells should be done by evaluating the positions of potential contaminant sources and the groundwater flow patterns in the subdivision.

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1.0 Introduction

1.1 Project Introduction

Beginning in the spring of 1987, two subdivisions near Stevens Point, WI have been the subject of a groundwater research project undertaken by Dr. Byron Shaw and graduate students at the University of Wisconsin-Stevens Point. A parallel study was done by a U.W.-Madison graduate student. In total there were approximately 250 monitoring wells placed throughout the two subdivisions that have been sampled and analyzed for several inorganic and some organic chemicals. Besides the monitoring wells, approximately three-quarters of the private wells have been sampled at least once for various inorganic chemicals. Other work that has been done in the subdivisions include: (1) a survey of homeowners designed to gather background information and to learn of practices that may effect the groundwater was taken to characterize residential habits; (2) land-use and sampling location maps of both subdivisions have been created and digitized into the geographic computer program pcARCINFO; (3) information on aquifer characteristics have been gathered using literature sources, a Ph. D. thesis, and field studies; and (4) analyses for several organic compounds have been run on water samples from select sites. This information helped describe how groundwater is influenced by suburban land-use. Nitrate-N was the primary indicator used to characterize subdivision impacts. Other inorganic analyses were performed to help identify and trace contaminant plumes and to help evaluate other subdivision impacts.

The BURBS computer model is a relatively simple spreadsheet designed to characterize nitrogen loading from residential sources. The variables used in the computer model were defined using field data and literature sources. The results from the BURBS model were used as a point of comparison for the chemistry results observed in the field.

1.2 Project Justification

Much of the Central Wisconsin area is located on a glacial outwash sand plain, which has little capacity to attenuate potential contaminants to groundwater. Many of the residents in the Central Wisconsin area live outside of areas served by municipal sewer and water. Most of these homes use on-site waste disposal systems (generally conventional septic systems consisting of a separation tank and soil adsorption field) and obtain water from shallow, driven-point wells. Published research from several authors indicate that in unsewered residential areas, the density of septic system drainfields is the primary influence on groundwater quality (Bicki and Brown, 1991; Yates, 1985; Perkins, 1984). Because the wells are frequently screened in the upper portion of the aquifer, private wells are often highly susceptible to groundwater contamination from nearby sources (such as septic systems).

The primary guidelines used in determining the depth of the private wells are obtaining an adequate supply of water and passing the drinking water standards for bacteria and nitrate-N. The high transmissivity of the aquifer allows the well's screen to be placed within ten feet of the water table—just deep enough to accommodate water table fluctuations.

Locating a water-supply well on a lot in Wisconsin must adhere to codes established by the Department of Natural Resources, which requires (among other things) that the well can be no closer than 25 feet from a septic tank and 50 feet from a soil absorption unit (drainfield). This may not offer adequate protection in sandy aquifers because nitrate-N and other contaminant concentrations may remain high for hundreds of feet downgradient of drainfields (Robertson, et al., 1991; Walker, et al., 1973).

Private drinking water wells are not the only water supplies threatened by subdivision land-use activities. Water departments generally place municipal wells near the outskirts of the city. If the well-head recharge area is extensively developed with unsewered subdivisions, the municipal water supply may show the cumulative effects of hundreds of drainfields. In these instances, if the city water supplies are to be kept safe for human consumption, the subdivisions must be developed so as to minimize impacts on the groundwater.

Lakes, rivers, and wetlands are also subject to influence by contaminants in groundwater. Elevated levels of nitrate-N and phosphate can contribute to excessive weed and algae growth and cause premature eutrophication of these surface waters. DeWalle and Schaff (1980) reported that both groundwater and surface water quality has gradually decreased in the a river basin in Washington State. The cause was attributed to an increase in unsewered residential areas in the basin.

1.3 Description of Study Areas

The study areas are two subdivisions (Jordan Acres and Village Green) located near the city of Stevens Point in Portage County Wisconsin. The area is in the north-central portion of the Central Wisconsin sand plain. A map showing the general location of the study areas is presented in Figure 1. A land use map showing the locations of the subdivisions is presented in Figure 2. The subdivisions are similar in terms of hydrologic characteristics, property values, family incomes, lot sizes, and residential practices. They differ in upgradient land use, subdivision size, housing distribution, and process of development.

Jordan Acres is the smaller of the two subdivisions, with an area of about 50 acres. At the start of the study fifty-nine (59) of the sixty-three (63) available lots were developed. Although the oldest home in the subdivision was constructed in 1960, most of the homes in the subdivision were built between the years of 1970 and 1977. The age distribution of the homes across the subdivision is fairly uniform (i.e., the homes in one portion of the subdivision are not of a significantly different age than the homes in another part of the subdivision). A land use map for Jordan Acres is shown on Figure 3a; the well locations and identifications are shown on Figure 3b.

Village Green is a much larger subdivision (approximately 160 acres). When the study began, approximately 70% of the subdivision was developed with single-family homes and related land uses. The remaining 30% was primarily undeveloped wooded areas. Similar to Jordan Acres, most of the homes in the subdivision were constructed in the 1970s. Unlike Jordan Acres, however, the development was not uniform. The southwest portion of the subdivision (except for the wooded area) was developed first, with the average year for home construction around 1972. The northwest portion was developed slightly later, with the average year of construction around 1973. The average year for building the homes in the southeast portion of the subdivision was about 1978. Although much of the northeast portion of the subdivision was still undeveloped at the start of the study, most of the homes that are present were built around 1980. A land use map of the Village Green subdivision is shown on Figure 4a; the well locations and identifications are shown on Figure 4b.

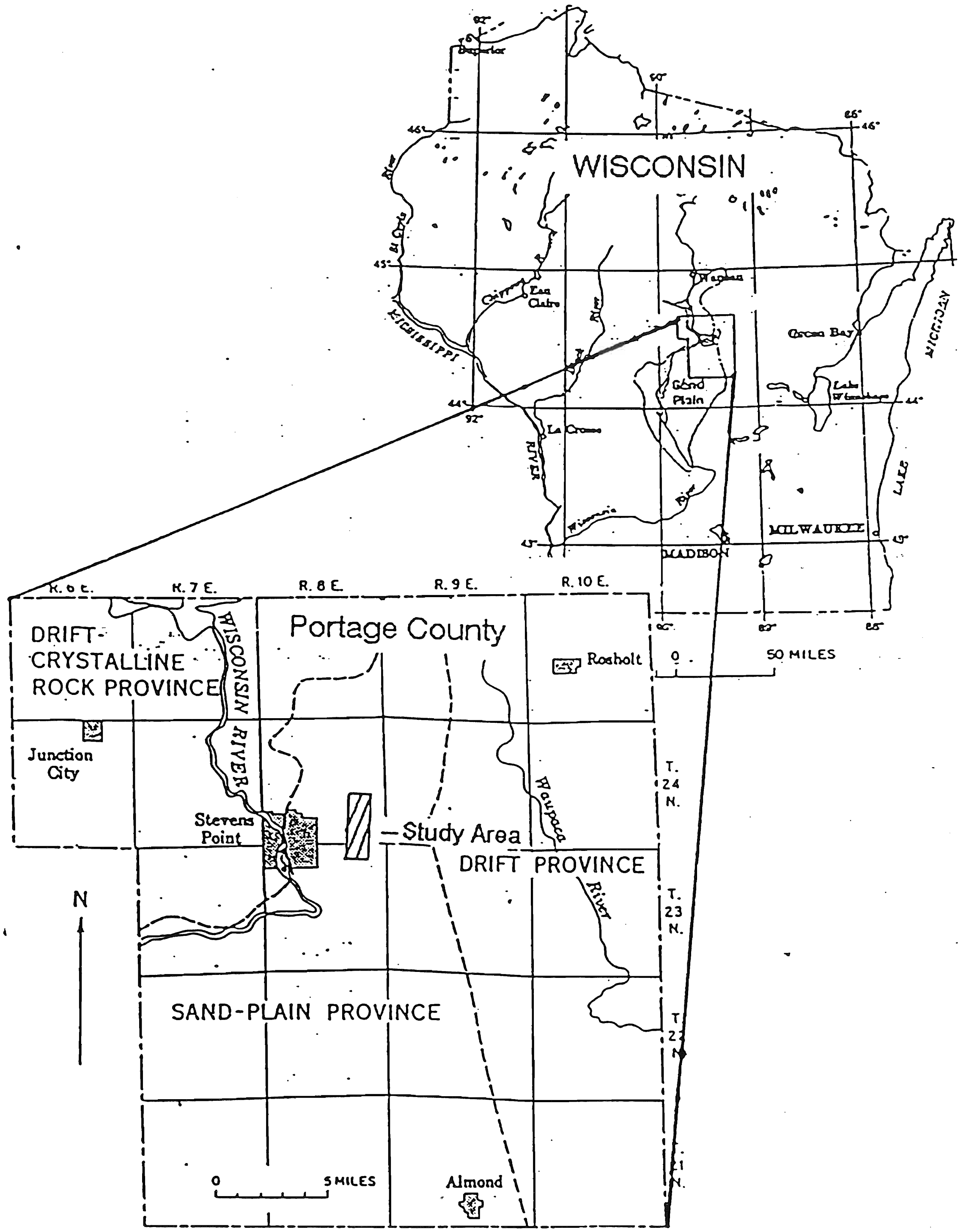

Figure 1: Location of the two focus subdivisions near Stevens Point in Portage County, Wisconsin (from Holt, 1965).

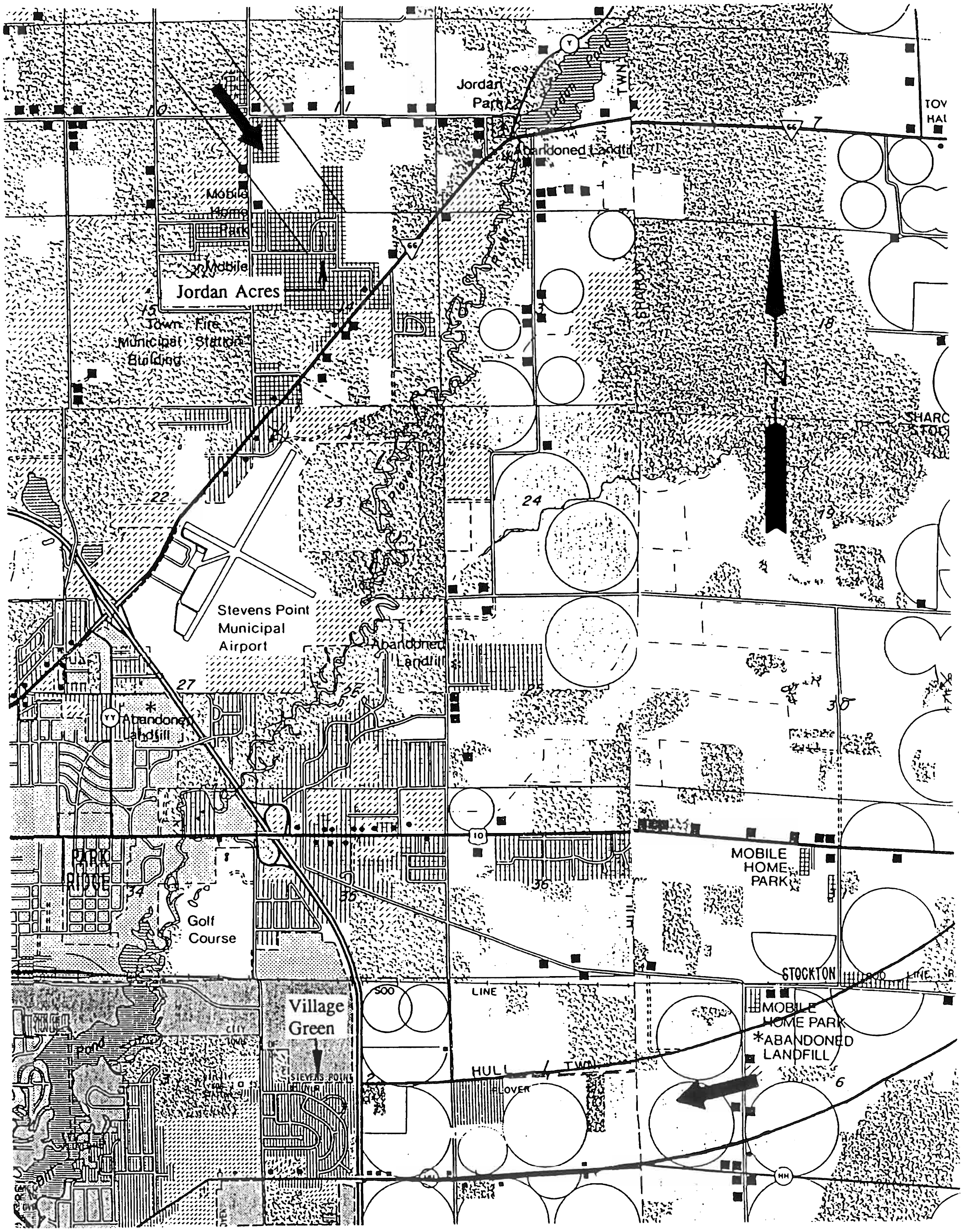

Figure 2: Land use map for the area near the study subdivisions. Groundwater flow directions are indicated.

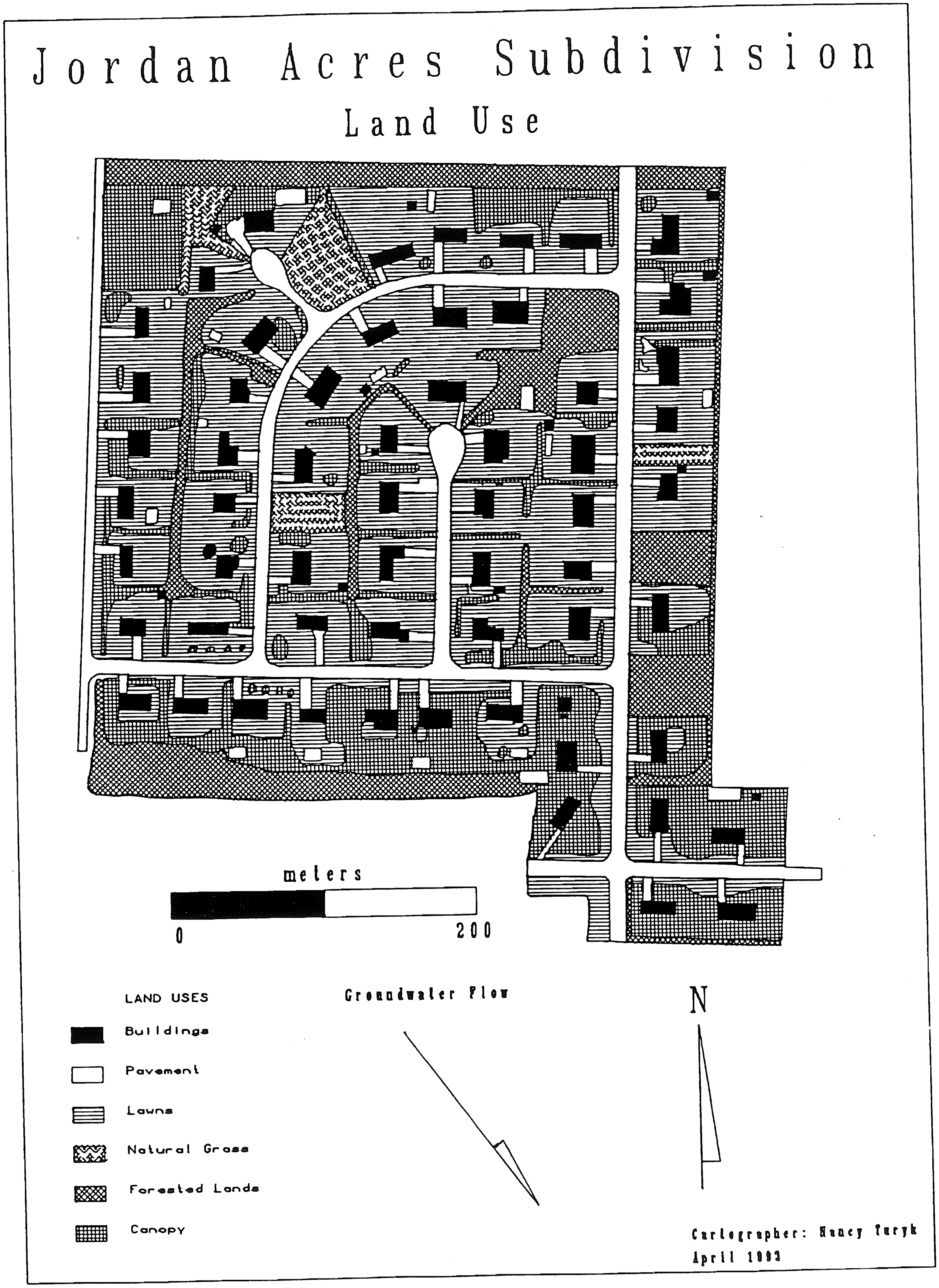

Figure 3a: Land use map of the Jordan Acres subdivision.

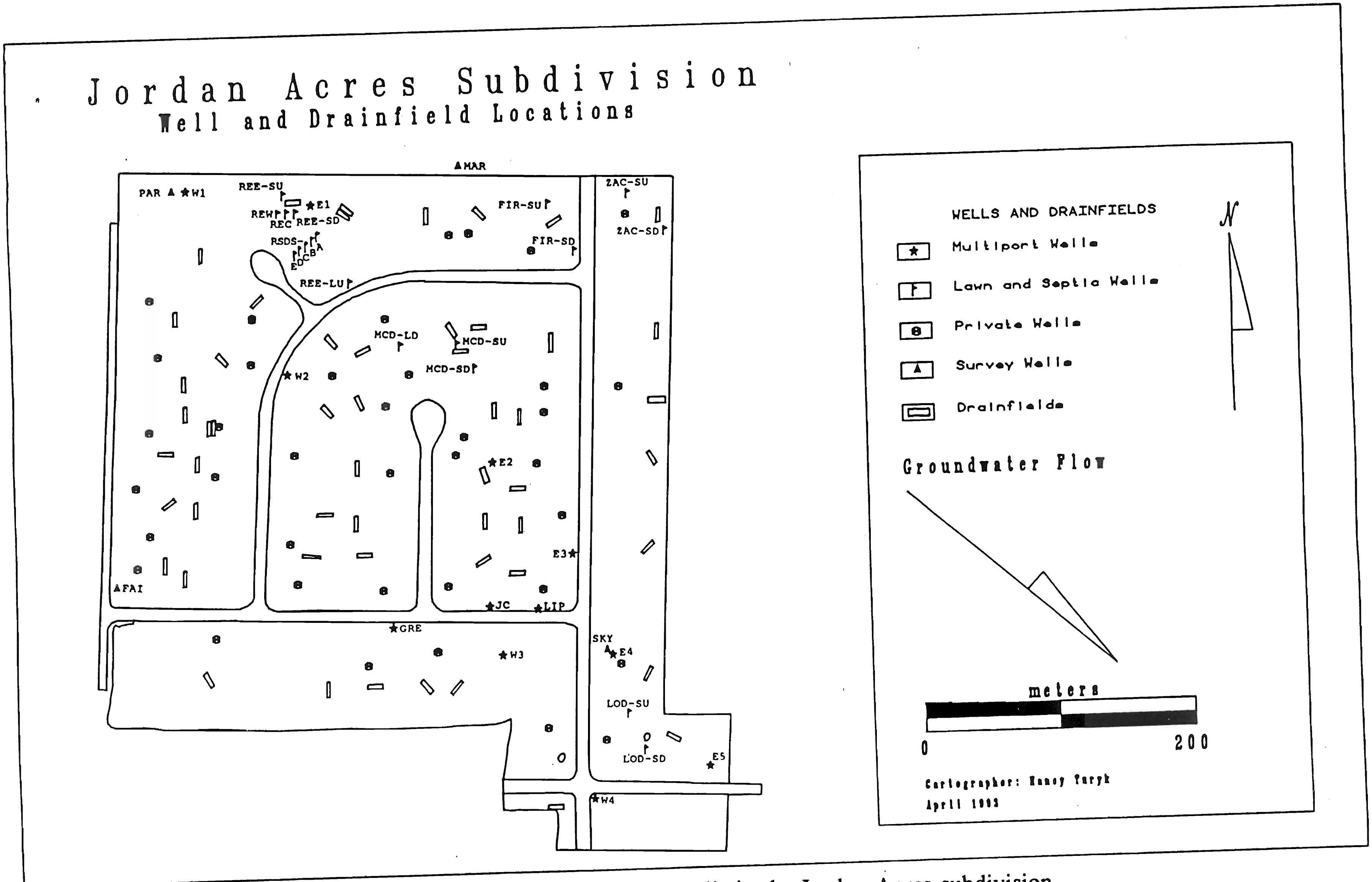

Figure 3b: Location and identification of the monitoring wells in the Jordan Acres subdivision.

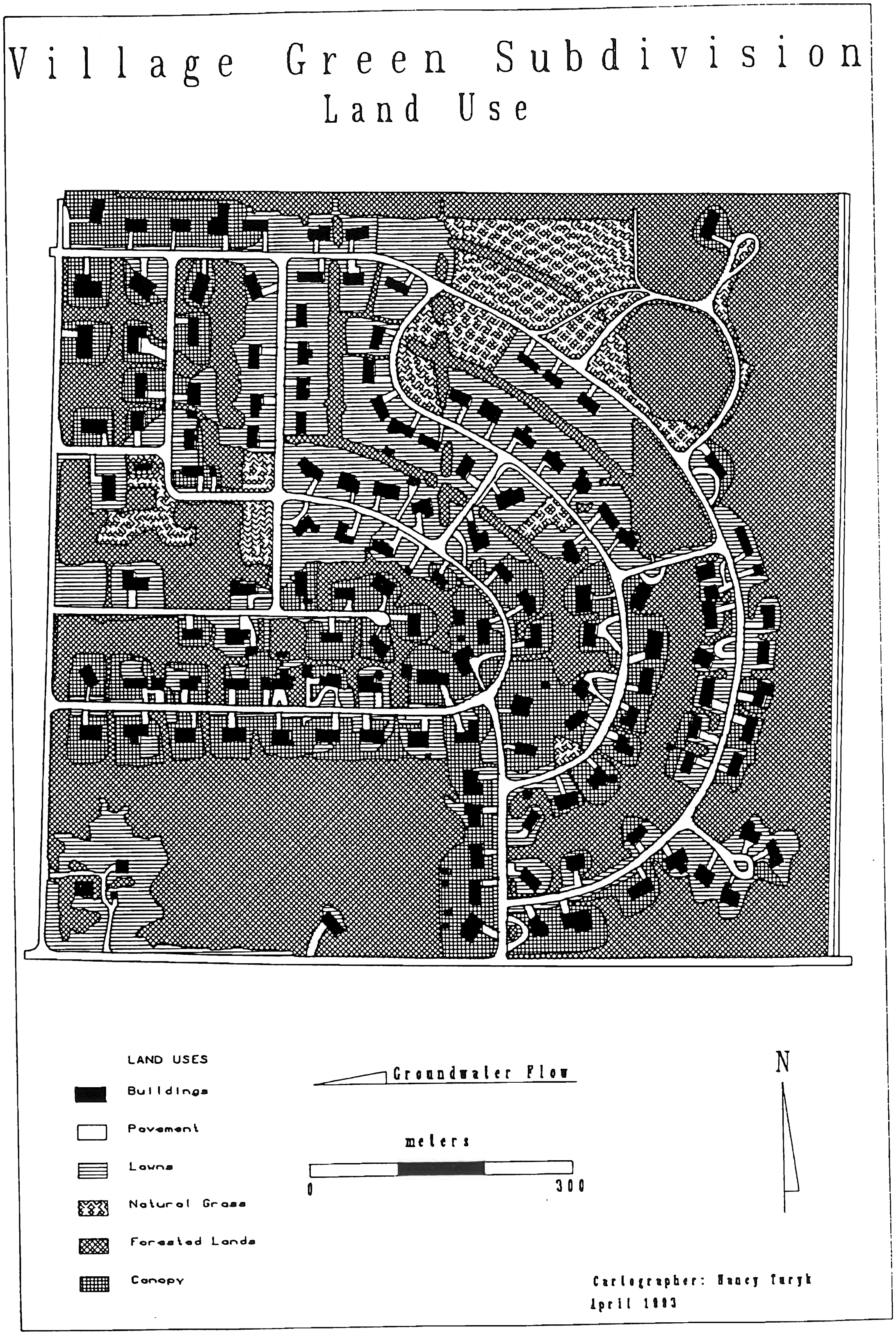

Figure 4a: Land use map of the Village Green subdivision.

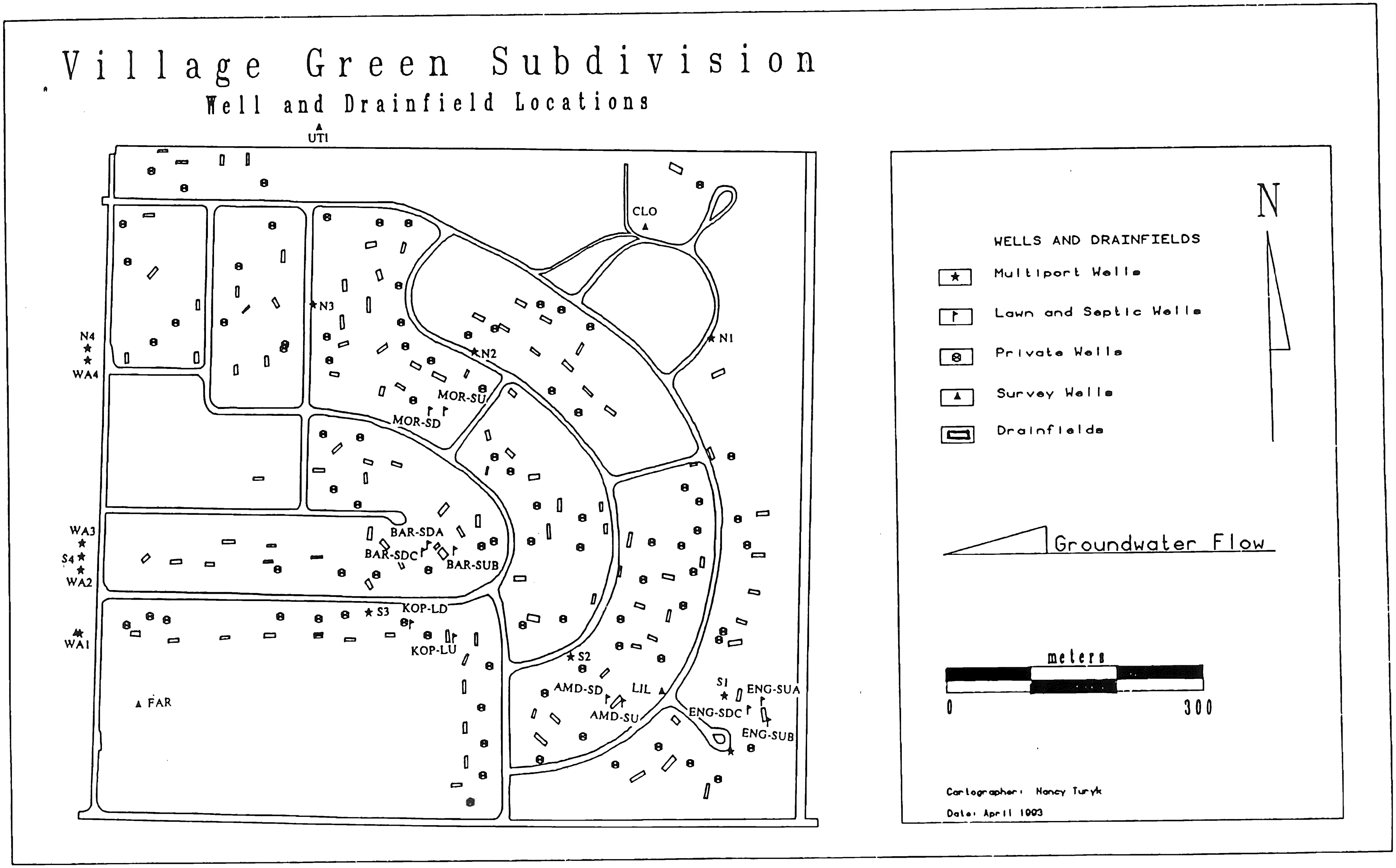

Figure 4b: Location and identification of the monitoring wells in the Village Green subdivision.

1.3.1 Hydrology and Topography

The city of Stevens Point is located in the east central portion of the Central Wisconsin River drainage basin (Devaul and Green, 1971). The Wisconsin River flows from the north to the south along the western edge of the city; the Plover River flows from the northeast to the southwest near the eastern edge of the city (see Figure 2).

Because Stevens Point is located on a glacial outwash sand plain, the topography is quite flat. A glacial end moraine several miles east of town serves as the surface water and groundwater divide for the eastern boarder of the drainage basin.

Jordan Acres is located three miles northeast of Stevens Point; Village Green is about one mile east of the city (see Figure 2). Neither site has significant topographic relief, thus runoff is minimal. Because the subdivisions are not served by storm sewers (not even culverts under the roads), runoff collects in the depressions next to the roads where it is allowed to infiltrate into the ground and contribute to groundwater recharge.

1.3.2 Geology

The geology beneath the subdivisions consists of sandy glacial outwash deposited over Precambrian crystalline bedrock. The average thickness of the unconsolidated deposits is generally less than 30 meters; however, a buried glacial river valley is present east of Stevens Point, and it has been reported that both subdivisions are located over this valley (Devaul and Green, 1971; Trotta and Cotter, 1973; Brown, et al., 1992). Wells installed up to twenty-one meters deep in each subdivision did not encounter bedrock.

The Soil Conservation Service has classified the soils in both subdivisions as part of the Plainfield-Friendship association, which is characterized by excessively drained to moderately well drained, nearly level sandy soils that formed in deep sandy deposits (SCS, 1978). This characterization was supported by information obtained while installing the monitoring wells. Grain-size analyses of soil samples obtained from the upper 15 meters of the geologic profile revealed that this material is composed of 96.5% to 99.7% medium and coarse sands (Harmsen, 1989). Samples obtained at greater depths had higher percentages of coarse sand and fine gravel.

1.3.3 Hydrogeology

As mentioned above, Stevens Point is near the east-central edge of the Central Wisconsin River watershed (Devaul and Green, 1971). Regional surface water and groundwater flow is generally to the south, with variations to the east and west, depending on the relative location of the Wisconsin River. Local groundwater flow directions are affected by wetlands, lakes, ponds, tributaries to the Wisconsin River, and other geologic factors.

The primary aquifer in the area east of the Wisconsin River consists of thick deposits of glacial outwash sand. The high hydraulic conductivity of this aquifer allows well yields to range from 50 gallons per minute (gpm) to over 1000 gpm, depending on the saturated thickness.

Jordan Acres is near the upgradient end of a relatively small local watershed.

The local groundwater divide is located approximately two miles northwest of the subdivision. Land use between the groundwater divide and the subdivision is approximately equally divided between non-irrigated agriculture, residential development; and natural (undeveloped) areas (see Figure 2).

Village Green is near the downgradient end of a much larger watershed. The groundwater divide occurs approximately seven miles east of the subdivision. The principal land use between the groundwater divide and the subdivision is center-pivot irrigation dedicated to potato and other vegetable production. There is also a significant amount of woodlands, a few residential areas, and one beef feedlot (see Figure 2).

Water table elevation data for each subdivision are presented in Tables 1 and 2; water table contours are shown on Figures 4 and 5 (see Appendix A for additional groundwater elevation data). Groundwater flow directions and hydraulic gradients remained relatively consistent throughout the study period. The hydraulic gradient in Jordan Acres was approximately 0.0026 meters/meter on August 28, 1989; the hydraulic gradient in Village Green was approximately 0.0020 meters/meter on January 16, 1990. Groundwater flow beneath Jordan Acres appears to be towards the southeast. The direction of flow beneath Village Green appears to be towards the northwest on the upgradient end of the subdivision, but is directly west at the downgradient end of the subdivision. [The cause of this bend in groundwater flow direction may be a greater amount of groundwater recharge occurring near the intersection of the county and state highways.]

Well Identification	Well Coordinates		Well-Top Elevation (meters)	Depth to Water (meters)	Water Table Elevation (meters)
	X-Cor.	Y-Cor.			
FIR-SU	38.99	21.40	340.80	6.91	333.89
FIR-SD	33.66	20.18	340.80	7.02	333.78
LOD-SU	35.37	8.30	340.42	7.36	333.05
LOD-SD	35.80	7.36	341.28	8.31	332.97
MCD-SU	30.94	17.72	340.57	6.86	333.71
MCD-SD	31.38	17.10	340.48	6.82	333.65
MCD-LD	29.61	17.66	340.84	7.07	333.77
REE-SU	26.68	21.56	340.10	5.95	334.14
REE-SDW	26.58	21.07	340.05	5.92	334.13
REE-SDC	26.78	21.09	339.93	5.81	334.12
REE-SD	27.00	21.10	340.02	5.92	334.10
REE-LU	28.35	19.29	339.83	5.90	333.93
ZAK-SU	34.89	21.67	341.10	7.29	333.81
ZAK-SD	35.83	20.72	340.92	7.20	333.72
E1	27.32	21.32	339.94	5.85	334.09
E3	33.86	12.31	340.20	6.96	333.24
E4	34.91	9.77	340.73	7.51	333.21
GRE	29.68	10.41	340.07	6.74	333.33
LIP	33.09	10.91	340.42	7.23	333.18
W2	26.92	16.92	339.61	6.06	333.55
W3	32.29	9.76	340.70	7.26	333.44
FAI	23.08	11.35	340.51	6.84	333.67
MAR	30.79	22.31	340.32	6.30	334.01
PAR	24.02	21.62	341.16	6.93	334.23
SKY	34.78	9.88	340.44	7.41	333.03

Table 1: Jordan Acres groundwater elevation data; water levels measured on August 28, 1989.

Well Identification	Well Coordinates		Well Top Elevation (meters)	Depth to Water (meters)	Water Table Elevation (meters)
	X-Cor.	Y-Cor.			
AMD-SD	27.18	6.13	334.63	7.49	327.14
AMD-SU	27.70	6.01	334.87	7.70	327.17
BAR-SDA	21.42	11.28	333.41	6.64	326.78
BAR-SDC	31.87	5.75	333.39	6.63	326.77
BAR-SUB	32.53	5.33	334.09	7.28	326.81
ENG-SDC	31.87	5.75	334.79	7.37	327.42
ENG-SUA	32.32	6.03	334.70	7.26	327.45
ENG-SUB	32.53	5.33	334.78	7.30	327.49
MOR-SD	21.51	15.81	333.63	6.94	326.70
MOR-SU	22.00	15.79	333.80	7.08	326.73
LC	31.21	4.36	334.30	6.91	327.39
N4	10.18	18.01	332.94	6.87	326.07
S1	31.03	6.21	334.81	7.46	327.35
S2	25.97	7.53	333.97	6.95	327.02
S3	19.39	9.05	333.72	7.08	326.64
S4	9.91	10.96	333.14	7.09	326.05
WA1	9.82	8.43	333.14	7.08	326.06
WA2	9.88	10.52	332.93	6.86	326.06
WA3	9.93	11.41	333.16	7.09	326.07
WA4	10.18	17.60	332.99	6.93	326.06
CLO	28.58	22.00	334.47	7.53	326.93
FAR	11.76	6.04	333.62	7.41	326.21
LIL	28.92	6.35	334.88	7.66	327.22
UTI	17.95	25.43	334.68	8.33	326.35

Table 2: Village Green groundwater elevation data; water levels measured on January 16, 1990.

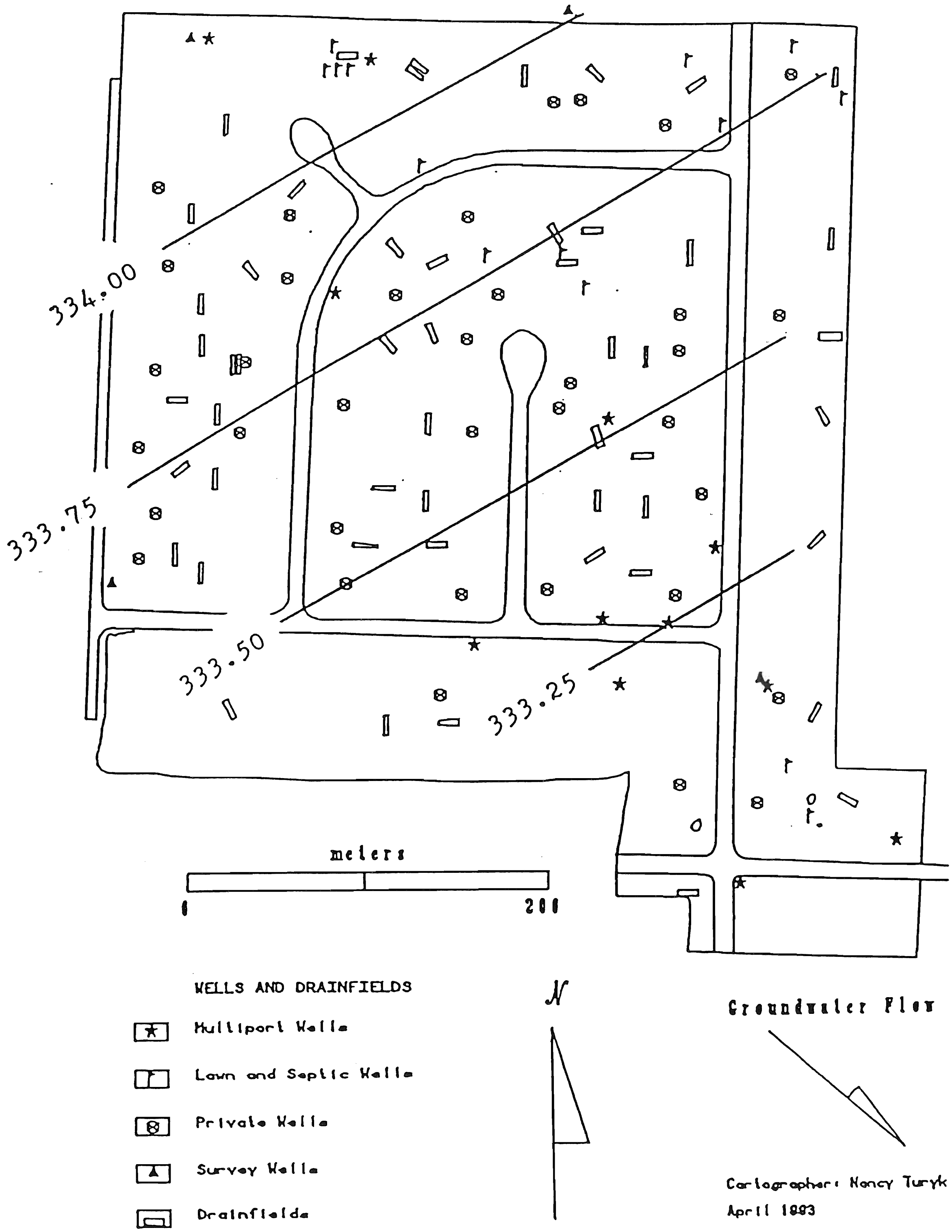

Figure 5: Water table contours beneath Jordan Acres subdivision; water levels measured on August 28, 1989.

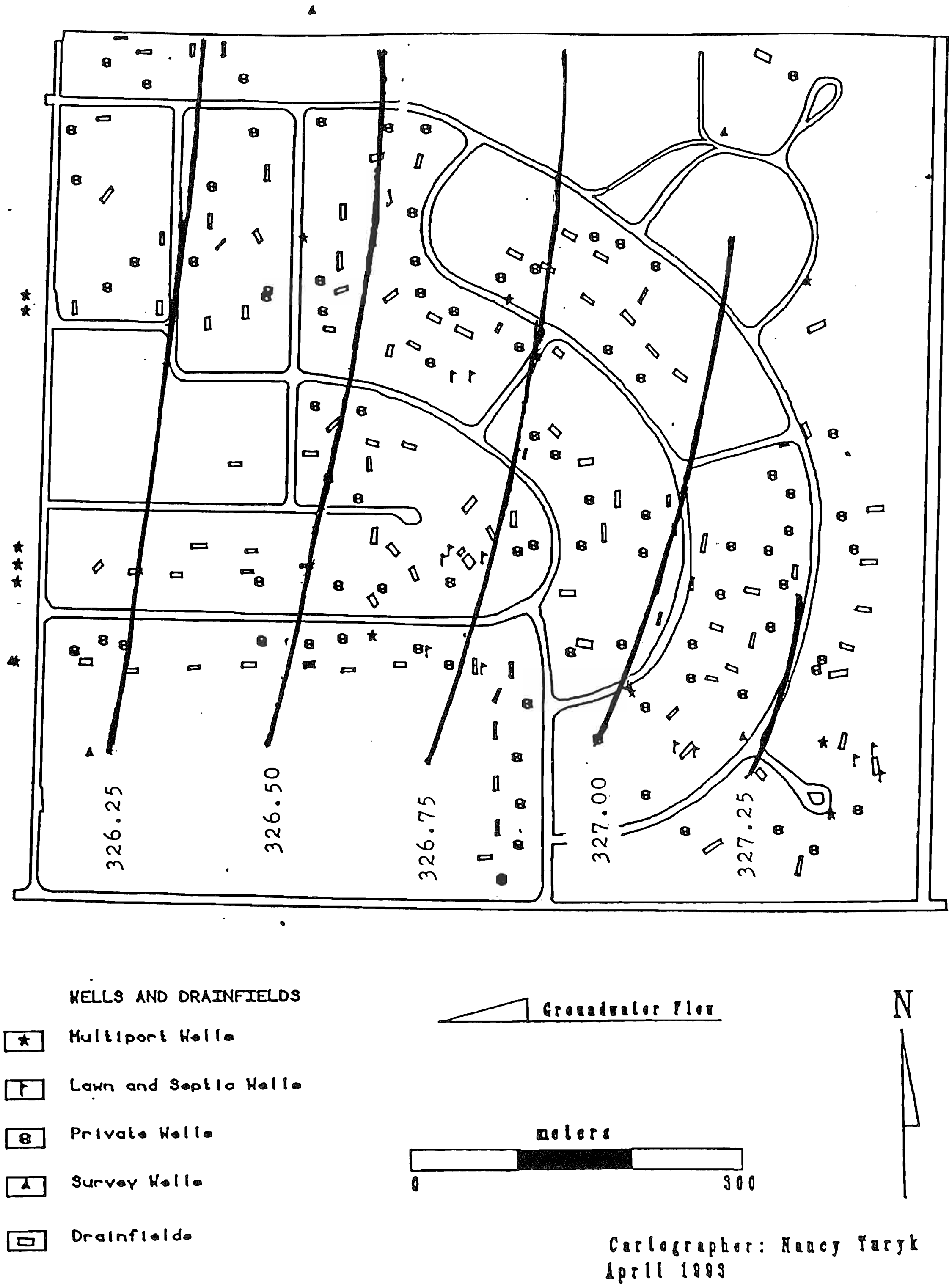

Figure 6: Water table contours beneath Village Green subdivision; water levels measured on January 16, 1990.

Harmsen (1988) used data from several of the monitoring wells to evaluate the presence of vertical hydraulic gradients and concluded that there are both upward and downward vertical gradients present within the aquifer. However, considering the small magnitude of the head differences (generally near the precision limits of the measuring instrument), it is difficult to draw any firm conclusions based on the data (except perhaps that mixing of the groundwater appears to occur within the aquifer).

Pumping test data published by several authors and summarized by Bradbury et. al (1992) indicate that the hydraulic conductivity of the outwash material is in the range of 0.065 cm/sec to 0.179 cm/sec (185 ft/day to 508 ft/day). Data from slug tests performed in the outwash material near the Stevens Point municipal water-supply wells reported by Brown et al. (1992) indicate a hydraulic conductivity of around 0.077 cm/sec to 0.092 cm/sec (220 ft/day to 260 ft/day). Data obtained from municipal wells installed to serve two suburbs of Stevens Point (Whiting and Plover), as reported by the consulting engineers (Brown, 1980 and Donohue, 1989) indicate that the hydraulic conductivity of the outwash material is around 0.083 cm/sec (235 ft/day). Slug tests performed in the subdivisions by Harmsen (1989) yielded results of 0.02 cm/sec to 0.07 cm/sec (57 ft/day to 198 ft/day). For the purposes of this study, the hydraulic conductivity was assumed to range from 0.045 cm/sec (130 ft/day) to 0.085 cm/sec (240 ft/day). Assuming a 30-meter aquifer thickness, the transmissivity of the aquifer in both subdivisions is calculated to be between 1200 m^2/day and 2200 m^2/day (13,000 ft^2/day and 24,000 ft^2/day).

Specific yield and effective porosity values (which were considered to be equal for the purposes of this study) were also reported by some of the researchers discussed above. The values ranged from 0.20 (Born et al., 1988) to 0.35 (Bradbury et al., 1992). A value of 0.30 was used for this study.

The groundwater flow velocity beneath the subdivisions can be calculated by multiplying the hydraulic conductivity times the hydraulic gradient and dividing by the effective porosity. Using this equation and the values given above, the groundwater flow velocity beneath Jordan Acres is calculated to be in the range of 0.34 to 0.64 meters per day; the flow rate beneath Village Green is calculated to be in the range of 0.26 to 0.49 meters per day.

The groundwater flow time beneath the subdivision is calculated by dividing the length of the flow path by the average linear groundwater flow velocity. The flow path in Jordan Acres is 360 m; the flow path in Village Green is 850 m. Given the velocities from above, the flow time beneath Jordan Acres is calculated to be between 1.5 to 3 years; the flow time beneath Village Green is calculated to be between 5 and 9 years.

1.4 Water Budget

1.4.1 Precipitation

Daily precipitation data over a ten-year span were obtained from the weather station at the Stevens Point wastewater treatment plant. Monthly and yearly totals are presented in Table 3 and Figure 7. It is believed that this is the time period over which the water sampled in the monitoring wells entered the groundwater flow system. The three driest years (1987 to 1989) were the time period when much of the sampling was performed; thus the shallow water samples from this time of lower recharge may be representative of higher than average contaminant concentrations due to lesser amounts of dilution. Conversely, the three years prior to 1987 were relatively wet; thus the samples obtained from the ports deeper into the aquifer and earlier in the study may be representative of lower contaminant concentrations due to more dilution by recharge water.

Three-day precipitation totals from May 1987 through December 1989 are presented graphically in Figure 8. Water table elevation data from one of the survey wells in Jordan Acres are included on the same graph to show how the water table in the subdivisions were affected by precipitation events. As would be expected, water tables tend to be higher in late spring and early summer (due to snow melt and spring rains) and lower during winter (due to frost virtually eliminating recharge to the aquifer).

1.4.2 Evapotranspiration

The average amount of annual evapotranspiration (ET) in Portage County, as reported by Holt (1965), is 51 centimeters (20 inches) of water. This value was determined during studies at the Hancock and Marshfield Field Stations located in neighboring counties. A study on the Little Plover River Basin yielded an estimate of 53 centimeters (21 inches) of ET. Because plant growth is expected to remain relatively constant between years, it was assumed that similar losses occurred in the subdivisions during the study period. This assumption may not be valid in cases where the summer is exceptionally dry and plants are not able to realize their maximum transpiration potential, or if the precipitation occurs in patterns different from what is usual for the area.

	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
JAN	7.09	1.57	2.54	2.79	1.65	1.52	1.17	1.73	3.51	1.07	2.82
FEB	0.91	10.06	0.66	4.78	4.37	3.05	2.29	0.18	0.76	1.35	1.60
MAR	2.74	1.07	4.37	3.81	3.58	7.85	4.83	5.54	4.70	6.17	7.26
APR	5.89	13.84	9.60	3.38	9.86	3.76	6.30	5.23	6.38	1.80	5.79
MAY	8.03	3.05	8.51	11.48	6.22	9.35	8.61	5.05	3.18	21.64	10.80
JUN	9.02	8.86	4.95	3.58	16.10	6.12	8.38	10.52	2.39	3.23	16.41
JUL	5.41	8.76	21.82	7.14	12.17	6.07	14.55	11.56	8.99	6.30	7.16
AUG	21.79	8.53	6.91	12.83	16.41	11.79	9.78	6.58	11.18	9.35	11.99
SEP	17.42	7.92	9.04	13.11	12.88	12.34	21.16	9.96	9.70	8.15	8.20
OCT	5.13	8.33	6.65	7.57	14.30	4.98	6.53	3.51	4.55	10.46	5.08
NOV	0.61	1.45	10.64	8.18	10.85	12.88	3.12	7.39	6.30	3.23	2.11
DEC	2.77	2.26	7.59	4.47	5.59	4.85	1.12	4.88	1.98	0.81	5.51
TOTALS	86.82	75.72	93.29	83.11	113.97	84.56	87.83	72.11	63.60	73.56	84.73

Table 3: Monthly and yearly precipitation data (in centimeters) for the years 1980 through 1990 (source: Stevens Point wastewater treatment plant).

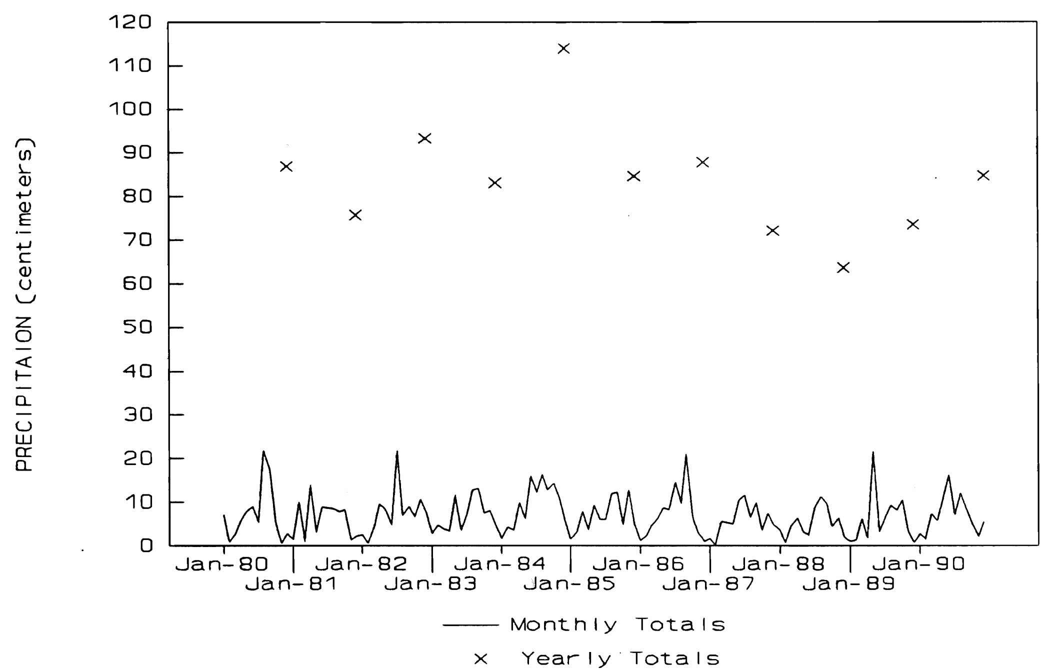

Figure 7: Monthly and yearly precipitation data for the years 1980 through 1990 (source: Stevens Point wastewater treatment plant).

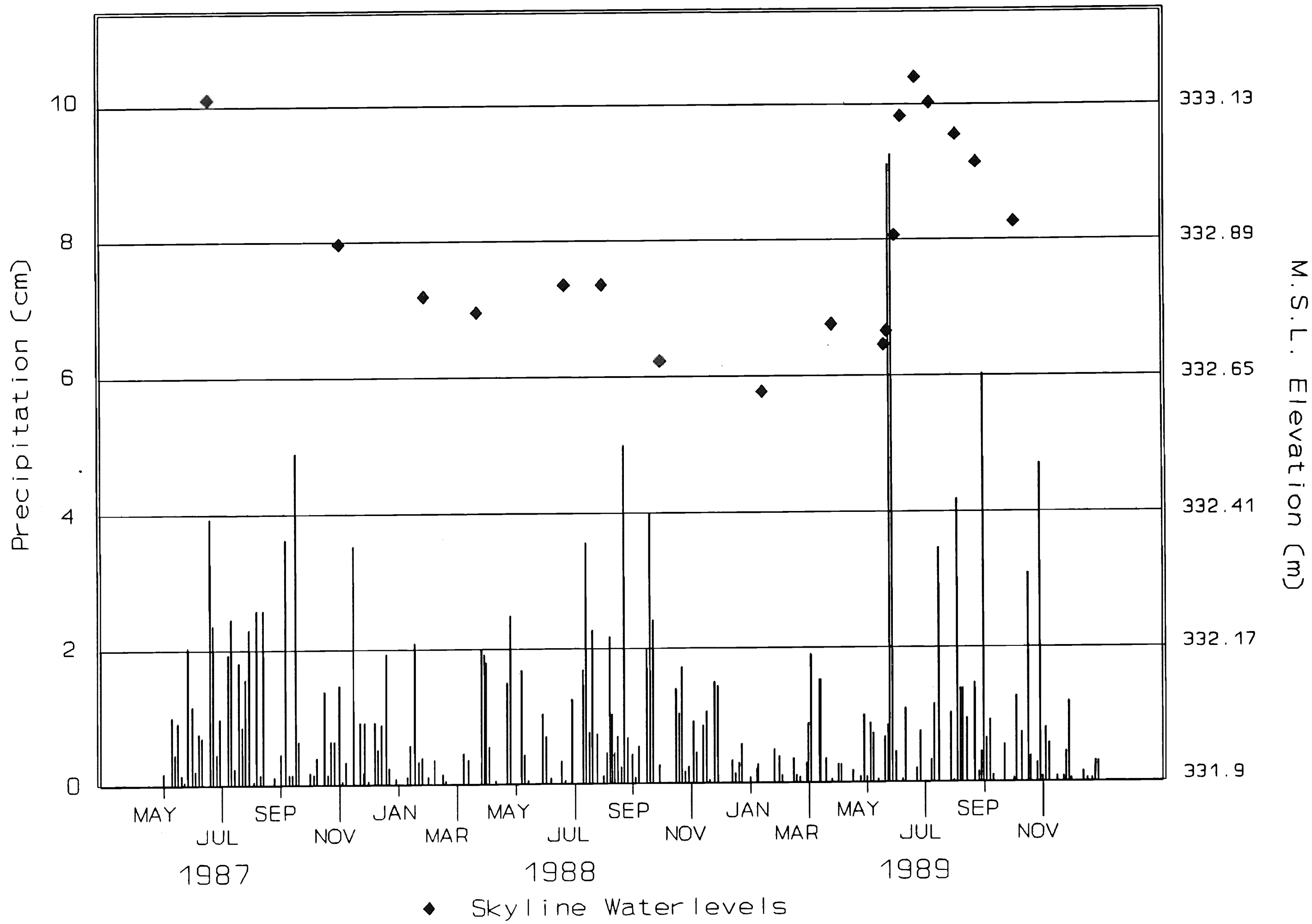

Figure 8: Three-day precipitation amounts for the time between May 1987 and December 1989 (source: Stevens Point wastewater treatment plant) and water table elevations for the survey well SKY in Jordan Acres.

1.4.3 Runoff

Water runoff is not considered to be significant in the subdivisions due to the flatness of the topography and the coarse soils. While the water will collect in the lower areas next to roads, there are no drainage culverts to direct overland flow out of the subdivisions. The small amount of water which may move off site is balanced by water moving into the area. Village Green may even have a positive overall runoff effect (more runoff entering the subdivision than leaving) considering that the highways to the east and south are at a slightly higher elevation than is the subdivision and runoff from these roads is routed into the edge of the subdivision.

1.4.4 Recharge

Because runoff was considered to be nil and the amount of water lost due to human influences was considered to be insignificant, groundwater recharge in the subdivisions was estimated to be equal to the yearly precipitation less yearly evapotranspiration. Assuming fifty-three centimeters (21 inches) of ET, recharge equals yearly precipitation minus 53 centimeters for the non-impervious areas. Added recharge occurs as runoff from roofs, roads, and other impervious areas enter adjacent porous soil areas.

2.0 Methods

2.1 Wells

In order to study the groundwater beneath the subdivisions, it was necessary to measure its various characteristics such as flow velocity and water chemistry. The groundwater chemistry was characterized by obtaining samples from several different types of monitoring wells.

2.1.1 Survey Wells

The survey wells were the first to be installed and were used to determine the direction of groundwater flow in the subdivisions. Wells were constructed using 3.05- or 6.1-meter (10- or 20-foot) lengths of 3.18-centimeter (1.25-inch) diameter schedule 40 PVC pipe. Screens were 30.5-cm (one-foot) long PVC with a 0.025-cm (0.01-inch) slot width. Lengths were connected using 40 or 80 gauge glue-joint couplings. Wells were placed so as to be located in the upper 3 meters (10 feet) of the aquifer. This placement was deemed adequate for accurately determining water table elevations while ensuring that the screens would remain below the water table during yearly fluctuations. The annular space around the wells was backfilled using drill cuttings. A bentonite seal at least 0.61 meters (two feet) thick was placed at the surface around each well. A 0.9- or 1.2-meter (3- or 4-foot long), 15.2-cm (6-in) diameter steel culvert with locking cap was placed around the well with at least half of its length below ground. The locations of the survey wells are shown on Figures 9 and 10.

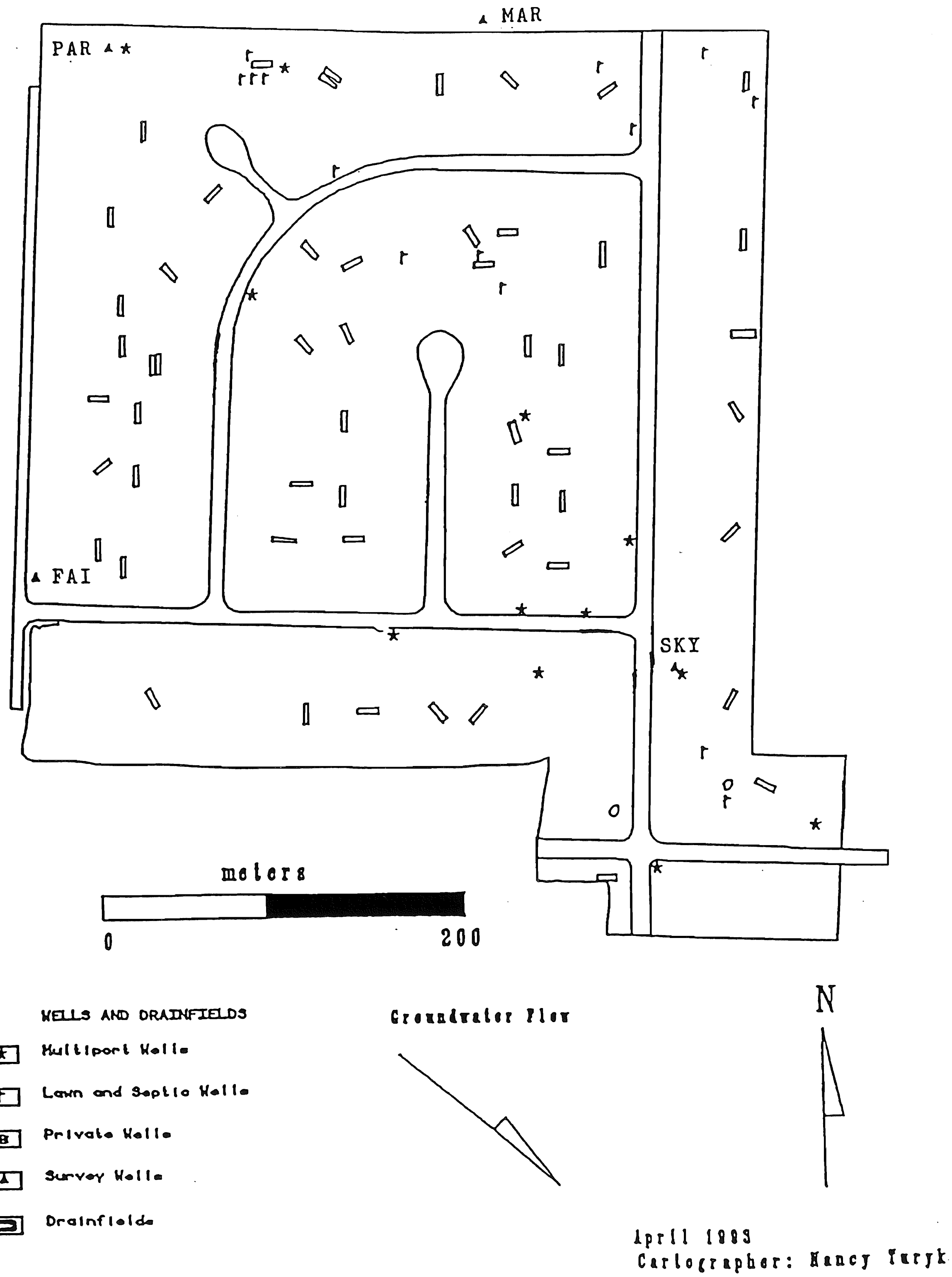

Figure 9: Location and identification of the survey wells in Jordan Acres.

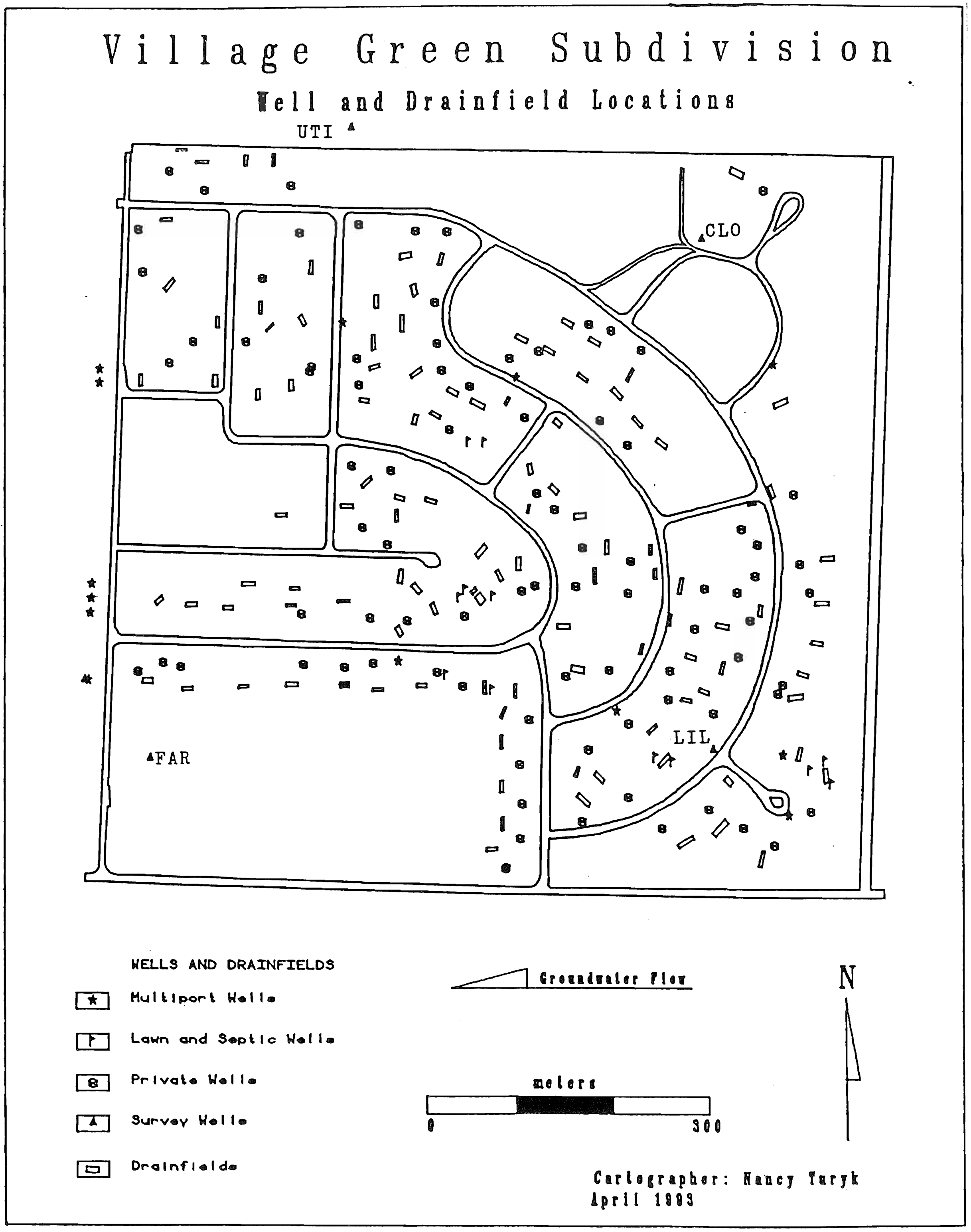

Figure 10: Location and identification of the survey wells in Village Green.

2.1.2 Multiport Wells

The multiport wells were the primary ones used to obtain samples for evaluating groundwater quality. They were constructed using 6.1-meter (20-foot) lengths of 1.3-cm (0.5-in.) PVC pipe with slip couplings and glue joints. A 1.5-meter (5-foot) long PVC screen with 0.025-cm (0.01-in.) slot width was placed in the middle so as to intercept the water table. These pipes were generally either 13.7 meters (45 foot) or 21.4 meters (70 foot) long and served as the spine for the polypropylene tubes. The actual sampling wells were made of lengths of 0.32-cm (1/8-in.) I.D. polypropylene tubing that had holes drilled in the lower 15.2 cm (6 inches) and nylon or typar mesh wrapped around the holes to serve as the screen. These tubes were attached to the PVC spine using nylon reinforced strapping tape and terminated at 0.76-meter, 1.5-meter, or 3.0-meter (2½-, 5-, or 10-foot) intervals into the aquifer. These wells were backfilled, sealed, and covered as described above. The location of the multiport wells are shown on Figures 11 and 12.

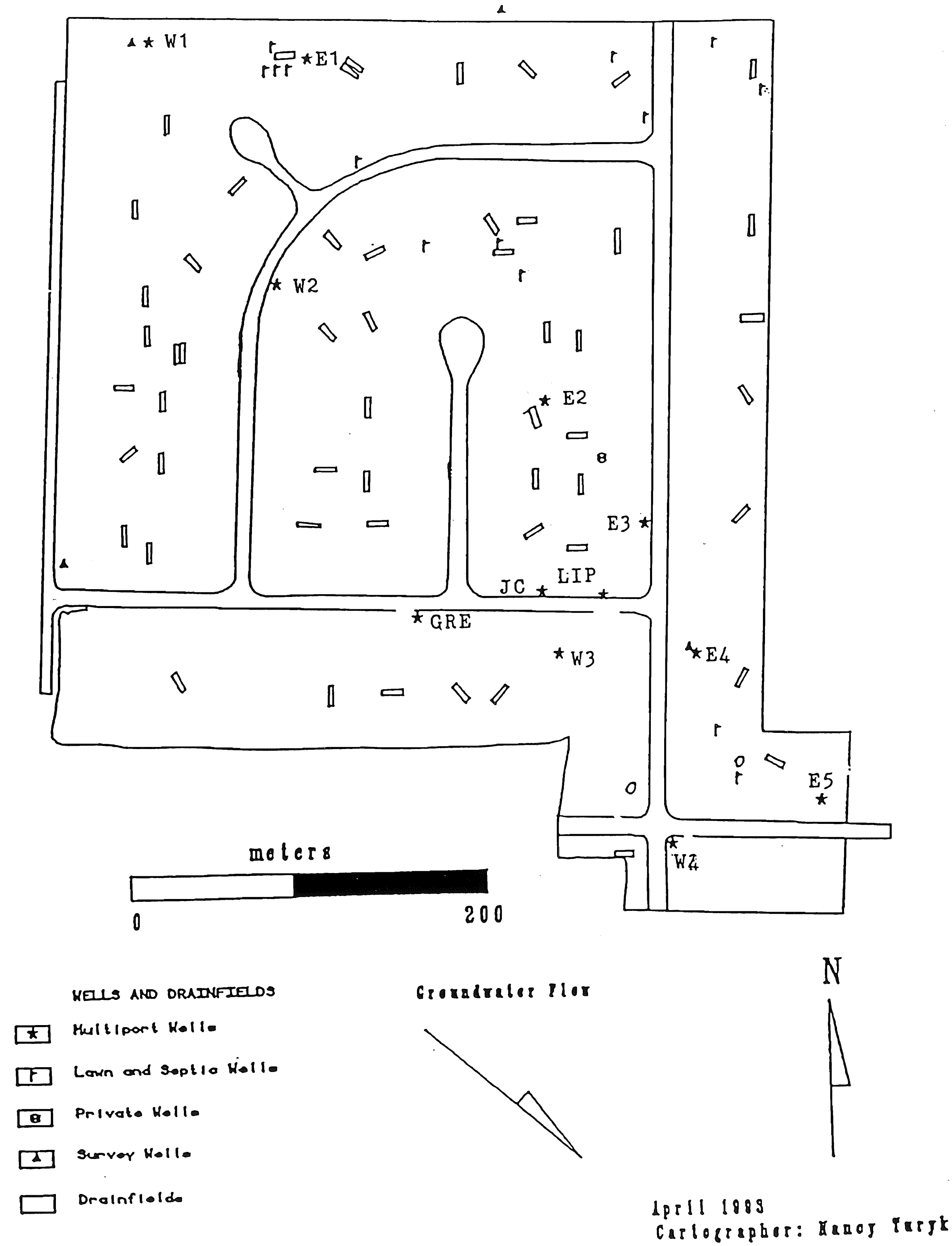

Figure 11: Location and identification of the multiport wells in Jordan Acres.

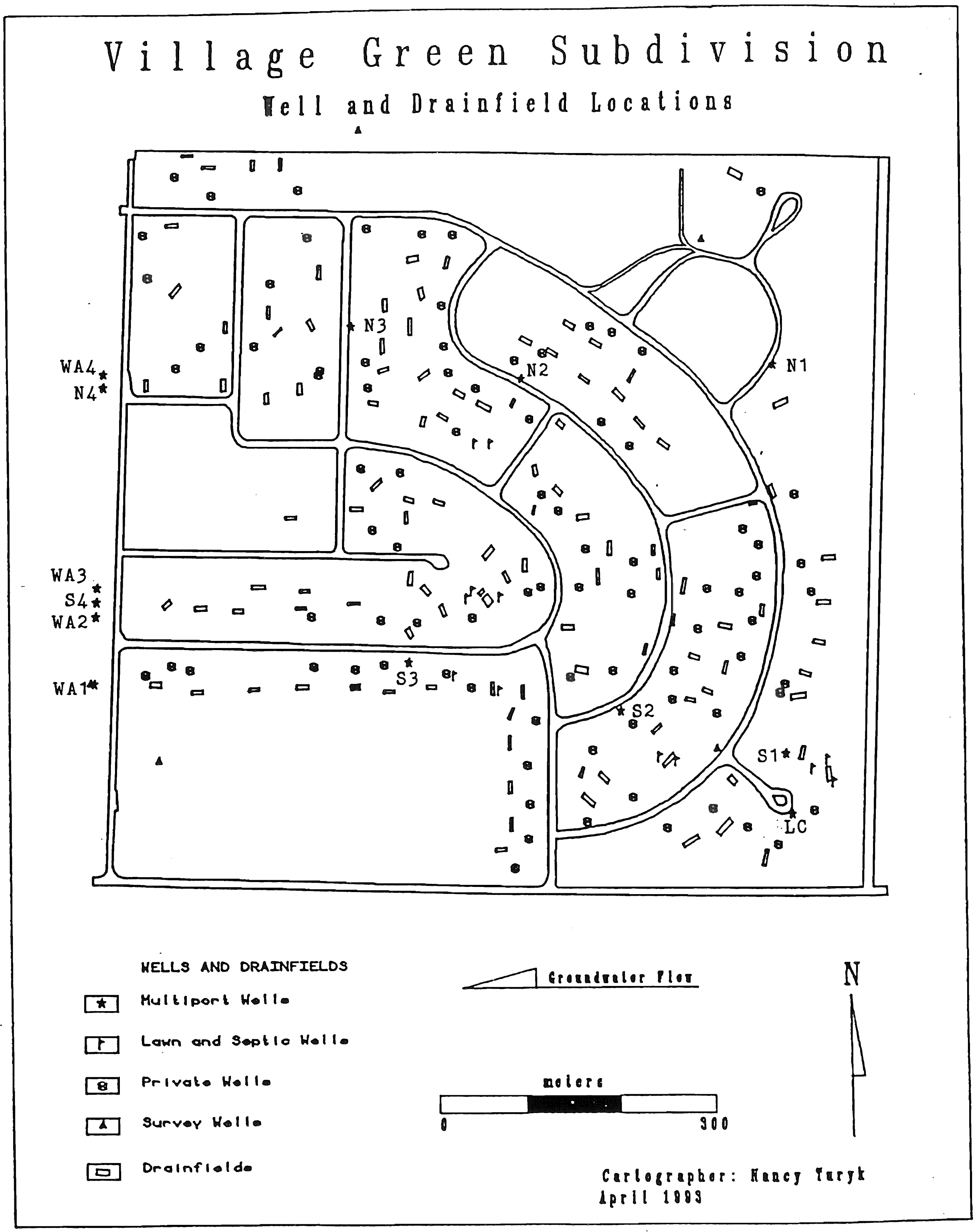

Figure 12: Location and identification of the multiport wells in Village Green.

2.1.3 Lawn and Septic Wells

Single-depth wells and multiple-depth well nests were used to monitor groundwater chemistry up- and downgradient of lawns and septic systems. The single-depth wells were constructed of 3.2-cm (1.25-in.) PVC threaded pipe and threaded couplings. Screens were 0.92-meter (3-foot) long threaded PVC with 0.025-cm (0.01-in.) slot widths. These wells were installed so that the lower 0.61 meters (2 feet) of screen was below the water table and the upper 0.31 meters (1 foot) was above. This was to enable interception of a fluctuating water table. Several of these wells did not have the culverts installed but instead a hole was drilled through the PVC cap and the top of the well and a long-shank lock installed. These wells were otherwise completed as discussed above.

Most of the wells installed to monitor the impact of septic system drainfields in Jordan Acres appear to have missed the contaminant plumes. In an effort to better define the water quality downgradient of a drainfield, two well nests were installed in addition to the two single-depth wells already present near one drainfield in Jordan Acres (identified as REW and REC). The additional wells were made of 1.9-cm (3/4-in.) threaded schedule 40 PVC pipe. Screens were .31-meter (1-foot) long threaded PVC with 0.25-cm (0.010-in.) slot width. They were installed in groups of three in order to sample discreet depths near the water table. One screen terminated near the water table. The next screen started at the base of the previous well, the final screen began 15 cm (6 in.) below the bottom of the middle screen. Each of the individual wells had a PVC slip cap; each nest was surrounded by a capped, 0.31-meter (1-foot) length of 15-cm (6-in.) diameter PVC acting as a protective cover. These wells we finished the same as the others.

The locations of the lawn and septic study wells are shown on Figures 13 and 14.

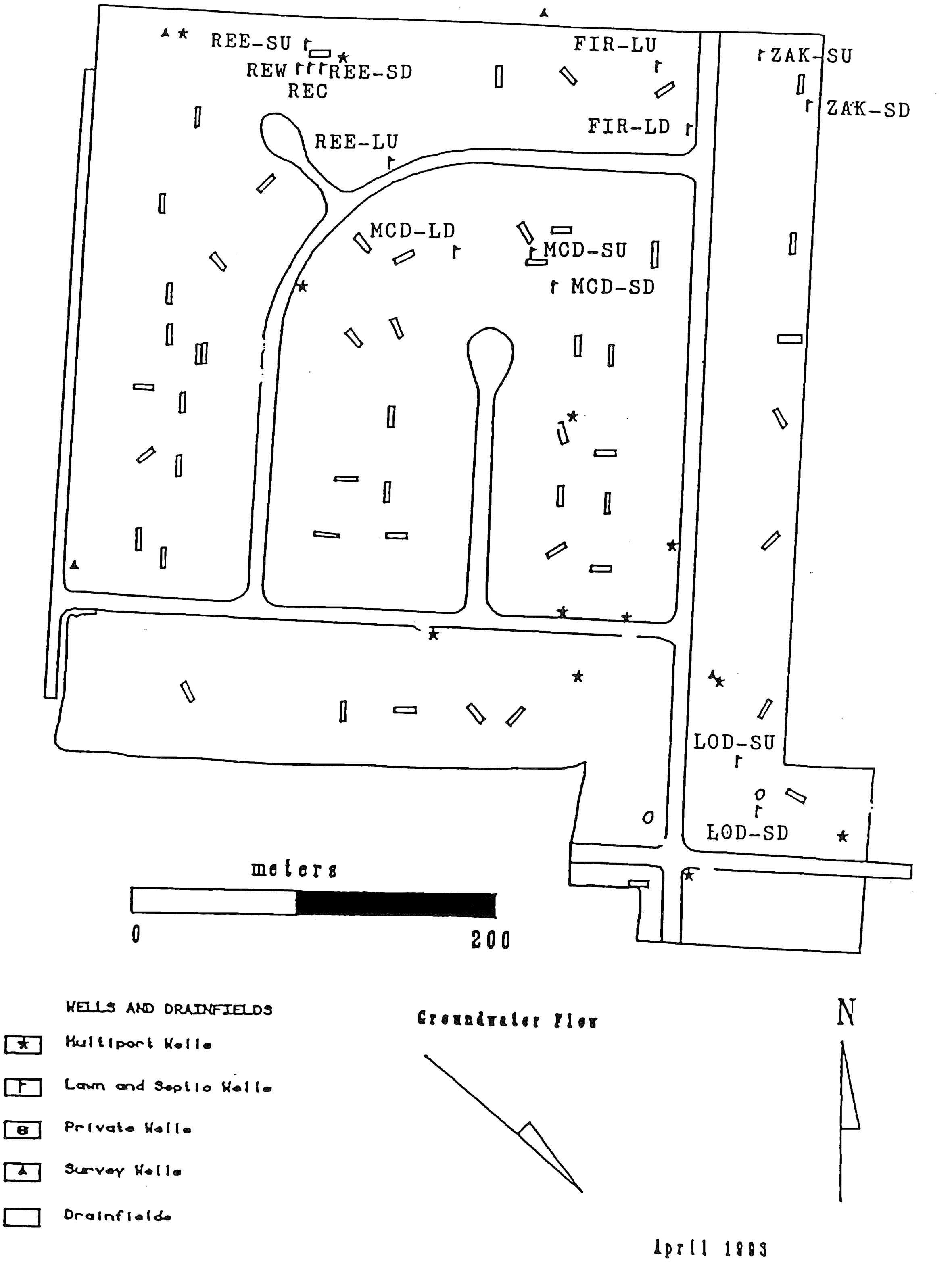

Figure 13: Location and identification of the lawn and septic wells in Jordan Acres.

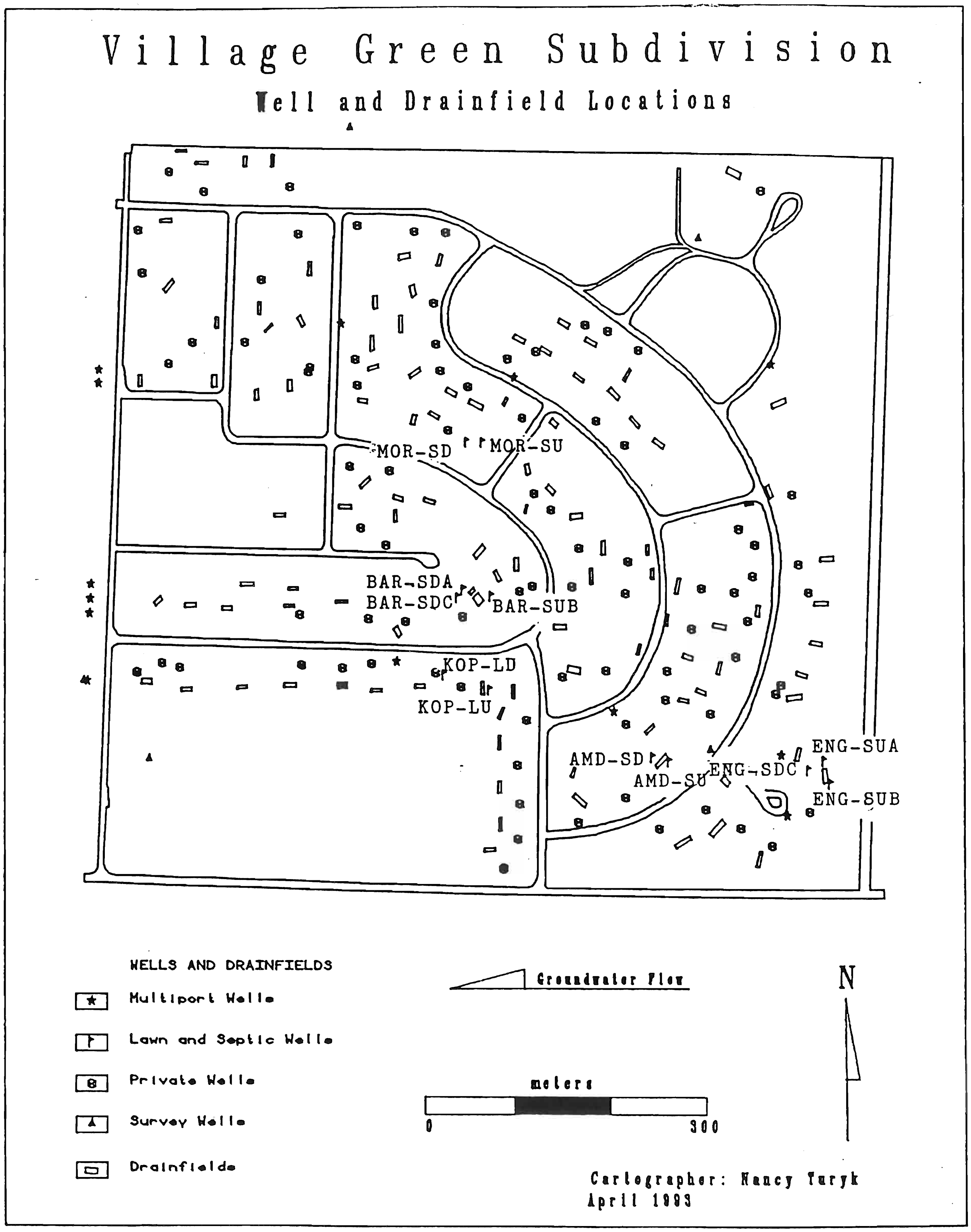

Figure 14: Location and identification of the lawn and septic wells in Village Green.

2.2 Well Installation

All wells except for the multiport wells were installed using a trailer-mounted drilling rig and 7.6-cm (3-in.) O.D. solid-stem augers. The multiport wells were installed using either a truck-mounted rig and 8.3-cm (3¼-in.) I.D. hollow-stem augers, or a truck-mounted rig using 10.8-cm (4¼-in.) I.D. hollow-stem augers. The trailer mounted rig is owned and operated by the Environmental Task Force Lab at the University of Wisconsin-Stevens Point. The truck mounted rig using 8.3-cm I.D. augers is owned by the Wisconsin Geological and Natural History Survey (WGNHS) and was operated by graduate students; the other truck-mounted rig is owned and operated by the WGNHS.

All wells except the multiports were developed by pumping water from the well (using either a gasoline-operated centrifugal pump or an electric peristaltic pump), and surging the well with the influent line. Wells were surged and purged until the pump discharge was clear. The multiport wells were developed by alternately withdrawing and injecting water and/or air using a peristaltic pump until sediment-free discharge was obtained.

2.3 Sampling

Sampling was carried out by the graduate students using either pumps or bailers. Field activities included measuring the depth to water, analyzing for pH and conductivity, sampling, and filtering. Sample containers were appropriate for the analyses performed. Samples were stored on ice until delivered to the laboratory. Sampling was performed, in general, on a quarterly basis.

2.3.1 Sample Acquisition

The peristaltic pump used to obtain groundwater samples was a Cole-Parmer, dual-headed, 12-volt DC electric pump. The pumping lines (the only wetted part) were silica tubing.

The multiport wells were sampled by attaching one of the pump's influent lines directly to the individual tubes, then withdrawing the water by vacuum. Because the pump had two separate pumping heads, two wells were frequently pumped at the same time. To sample the other types of wells, a length (or two) of 0.64-cm (¼-in.) O.D. polypropylene tubing was lowered into the well and the sample thus withdrawn with the pump.

The wells were purged prior to sampling by removing at least three times the volume of the well, or until constant temperature and conductivity readings were obtained.

Field pH and conductivity measurements were obtained by directing the pump effluent into the appropriate measurement container. The water was allowed to flow over the instrument's detector until a constant reading was obtained, at which time the value was recorded in a field notebook or on a data sheet.

After the pH and conductivity measurements were obtained, the samples were filtered. Filtering was accomplished by using a Gelman in-line filtering cartridge and 0.45 micron filters. At least 200 mls of water was allowed to pass through the filter prior to obtaining the sample. The filtered sample was discharged directly into a 250 ml Nalgene sample bottle or other suitable sample container.

Occasionally a bailer was used when sampling the lawn and septic wells, especially if VOCs were to be analyzed. The bailers were made of 1.5-meter (5-foot) lengths of 2.54-cm (1-in.) diameter Teflon or Schedule 40 PVC with a ball check-value in the bottom. The bailer was lowered into the well using a length of nylon rope. Three times the well volume was purged prior to obtaining the sample.

2.4 Chemical Analysis

Groundwater sample analyses were performed by the Environmental Task Force lab at the University of Wisconsin-Stevens Point (Wisconsin lab certification #750040280).

Nitrate-N, chloride, and reactive phosphorous (as PO_4-P were analyzed using a Technicon Autoanalyzer. Nitrate-N analysis used a sulfanilamide complex read at 520 nm (QuikChem Method No. 10-107-04-1-A.) Chloride analysis used a ferricyanide ion read at 480 nm (QuikChem Method No. 10-117-07-1-A). Reactive phosphorous analysis used a phosphomolybdenum complex read at 880 nm (Industrial Method No. 329-74 W/B).

Sodium analyses were performed using a Varian AA475 Atomic Absorption spectrophotometer read at 589.0 nm.

Analyses for alkalinity and total hardness were performed using techniques described in Standard Methods for the Examination of Water and Wastewater (APHA et al., 1985).

Relative fluorescence was measured using a Baird-Atomic Fluoripoint. The excitation scan was set at 355 nm and the emission was set at 425 nm.

The pH and specific conductance were measured in the field using a Corning electrode meter (pH) and a YSI conductivity cell.

2.5 Maps

Maps were created based on information obtained from airphotos, county parcel and development records, and direct in-field observations.

Parcel maps were obtained from the county land record office. From these the general layout and size of the subdivisions and private lots were determined. The individual parcels were numbered and maps were made from which to begin to define land use characteristics.

During home interviews conducted in the subdivisions, one of the objectives of the interviewer was to obtain or make a sketch of the lot. Information included on these sketches included the position of the home and other impermeable areas, the positions of the well(s) and septic system, and other site specific items such as gardens, doghouses, etc. This information was recorded on the individual parcel maps.

Air photo slides of the subdivisions were obtained from the Portage County Agricultural Stabilization and Conservation Service office. These slides were projected onto 86 cm x 56 cm (34 in. x 22 in.) copies of the parcel maps and the air photo information was traced thereon. Maps showing the precise positioning of roads, houses, woodlands, and other land use data were thereby created.

Equipped with the more detailed maps, researchers went into the subdivisions to obtain additional cartographic data and to clarify information obtained from the airphotos. Monitoring wells, private wells and septic-system drainfields were added to the maps. In addition, land use data could be verified, added, or corrected. Land use characteristics were categorized in seven groups: lawns, canopied lawns (those with substantial tree cover), woods, natural grass, houses (and miscellaneous buildings), driveways, and roads. For water and nitrogen mass-balance purposes, the land use was classified as lawn (turf), natural (undeveloped), or impervious (buildings and roads).

After the field maps were completed, the information was digitized using the pcARCINFO geographic information package by the UWSP Geography department. This GIS was used to create the final maps of the subdivisions and also to calculate the areal data used in the study.

2.6 Nitrogen Budget

The first step in determining the nitrogen loading to groundwater is to define and quantify the influencing factors as accurately as possible. These factors include demographic data such as number of persons per dwelling, housing density, and home water use; climatic data such as precipitation and evapotranspiration; and nitrogen sources such as wastewater and lawn fertilizers. The BURBS computer program was used to calculate the nitrogen loading from the subdivision including the theoretical average concentration of nitrate-N in the groundwater recharge originating from within the subdivision.

The mass of nitrogen that has been added to the groundwater from subdivision sources can be calculated if the concentration of nitrogen in the groundwater originating from the subdivision and the total volume of this water is known. The nitrate-N concentration was calculated using chemistry data obtained from monitoring wells installed in and downgradient of the subdivisions. The value was estimated by averaging the concentrations from groundwater samples obtained from the wells representative of water originating from subdivision sources. A value for the total volume of groundwater impacted by the subdivision was also determined, in part, by using the water chemistry from monitoring wells.

The depth of the aquifer that contained water originating from within subdivision was estimated in two ways. One was based on comparing the quality of the groundwater entering at the upgradient end of the subdivision to the quality of the water entering the aquifer as recharge from the subdivision. By obtaining water samples from several depths within the aquifer throughout the subdivision, the depth at which the two "plumes" meet can be determined. For the other method, the total amount of water recharged from the subdivision during the groundwater flow time through the subdivision was calculated, then the portion of the aquifer represented by this volume was determined.

The first method described above assumes that the chemistry of the groundwater flowing into the subdivision is measurably different from the chemistry of the water originating from within the subdivision. Furthermore it assumes that the water moves in distinct plumes with minimal mixing. It also assumes that the water from farther upgradient is flowing at a greater depth within the aquifer than water originating nearer the monitoring location.

In addition to the nitrate-N previously mentioned, the water entering the subdivision was monitored over a three-year period for several inorganic chemical species including pH, specific conductance, total hardness, alkalinity, chloride, sodium, reactive phosphate, ammonia, dissolved oxygen, chemical oxygen demand, and relative fluorescence. These data allowed characterization of the upgradient water. These same species were monitored at various depths throughout the subdivision. The dominant factor influencing the depth below the water table of a particular unit of groundwater is the amount of recharge deposited over this unit after it had entered the aquifer. This will cause the groundwater near the water table on the upgradient end of the subdivision to be at some greater depth below the water table at the downgradient end. The water below this depth will be representative of water recharged upgradient to the subdivision; water above this depth will be representative of water recharging from the within subdivision. Unfortunately, the exact depth could not be easily defined due to factors such as mixing in the aquifer (due to dispersion and the effects of the pumping wells), differential groundwater recharge, and the limitations of the sampling network; however, it is believed the approximate depth of subdivision impact was defined with reasonable accuracy.

The total volume of recharge method uses the assumption from above (i.e., groundwater at the upgradient end of the subdivision is deeper in the aquifer when it reaches the downgradient end of the subdivision). The depth of the upgradient water upon reaching the downgradient end of the subdivision will be approximately equal to the thickness of the aquifer represented by the recharge water that has occurred during the time period over which it took the said upgradient water to reach the downgradient end of the subdivision. The aquifer thickness this volume represents can be calculated by dividing the height of recharge water by the porosity of the aquifer.

The concentration of nitrate-N in the groundwater was determined by periodic sampling and analysis of water from wells located in those parts of the aquifer deemed to be affected by subdivision sources. This allowed the nitrogen characteristics of the groundwater to be documented.

3.0 Results

3.1 Field Data

The groundwater was monitored by obtaining water samples from wells installed within and downgradient of the subdivisions. The types of wells include (1) private drinking water wells (mostly shallow driven-point wells), (2) single-depth observation wells, and (3) multiple-depth well nests. The time period over which the wells were monitored varied widely between the wells; however, with the exception of a few of the private wells, all the samples were obtained between May 1987 and February 1991. The number of samples obtained from each of the wells also varied considerably. Several of the private wells were sampled only once during this period, whereas some of the multiport wells were sampled as many as 18 times.

Although nitrate-N was the primary chemical used for evaluating subdivision impacts, other chemicals including chloride, sodium, phosphate, total hardness, alkalinity, and relative fluorescence were also monitored. For the purposes of this report, the results of nitrate-N, chloride, sodium and phosphate analyses were the primary chemicals used for interpreting subdivision impacts.

In the Central Wisconsin area, "natural" background chloride concentrations are generally below 5 mg/L and frequently below analytical detection limits. The most common sources of chloride in the groundwater are animal wastes (especially human), deicing salts, and agricultural fertilizers. As such, chloride can be used as an indicator of human impacts on groundwater. Chloride is generally considered to be conservative in groundwater systems, and Alhajjar et al. (1990) concluded that it was the best chemical property to serve as an indicator of septic system impacts on groundwater quality (as compared with specific conductance, pH, and relative fluorescence).

As with chloride, natural sodium concentrations in the groundwater are generally low, and elevated concentrations are generally due to animal wastes and road salts. Agricultural fertilizers generally do not contain sodium. As a cation, sodium, while relatively unreactive, is likely to be somewhat retarded in the vadose zone and aquifer due to cation exchange. Adsorption appears to minimal in these sandy outwash soils, as evidenced by consistent sodium to chloride ratios in contaminant plumes.

Phosphate is often considered to be immobile in groundwater systems (Rea and Upchurch, 1980), mainly due to adsorption and precipitation reactions with the geologic matrix and available cations (Reneau et al., 1989). However, phosphate has been detected in the groundwater in areas of sandy soils and high water tables (Childs et al., 1974), and Brown et al. (1980) concluded that phosphate adsorption varies greatly between soils. Cogger (1988) suggests that phosphate movement is generally associated with a soils finite capacity to attenuate the ion and soil-column studies performed by Sawhney (1977) showed that once a soil's phosphate adsorption capacity was exceeded, phosphate concentrations in the effluent will increase. The primary impact of high phosphate levels is the impact on surface waters, where it can cause weed growth, algae blooms, and contribute to surface water eutrophication (Cogger, 1988).

3.1.1 Private Wells

Tinker (1991) used data from private water-supply wells in five subdivisions in Wisconsin to characterize nitrate-N concentrations in the groundwater and concluded that septic systems and lawn fertilizers cause increased nitrate-N concentrations in the groundwater on the downgradient end of the subdivisions. Miller (1972) observed elevated nitrate-N concentrations in groundwater samples from private wells in two study areas in Delaware. The source of the nitrate-N was attributed primarily to septic tank discharges. The author concluded that if the trend of developing unsewered subdivisions around larger metropolitan areas continue, the city will be surrounded by a ring of groundwater unsuitable for water-supply purposes.

Well characteristics (e.g., well depth, screened interval, etc.) are frequently unknown for private wells; because the well screens may be placed so as to avoid contaminated groundwater, the reliability of using chemistry data from private wells is somewhat suspect. Because both monitoring wells and private wells were tested during this investigation, a comparison will be made between the two in terms of duplicity of data.

Most of the private wells in the subdivisions were sampled at the beginning of the study. Several of these wells were sampled again two years later. Chemistry data from other sampling events were available for many of the private wells. In all, fifty-two of the fifty-six private wells in Jordan Acres were sampled at least once during the study period; ninety-one of one hundred and thirty in Village Green. The average NO_3-N concentration for Jordan Acres was 6.9 mg/L; the average for Village Green was 11.3.

3.1.1.1 Jordan Acres

The location of the private water-supply wells in Jordan Acres are shown in Figure 15. Private-well groundwater chemistry data from Jordan Acres are presented in Appendix B. The relative nitrate-N concentration (low, moderate, and high) is indicated for those wells from which chemistry data were available. The estimated groundwater flow path through the subdivision is indicated.

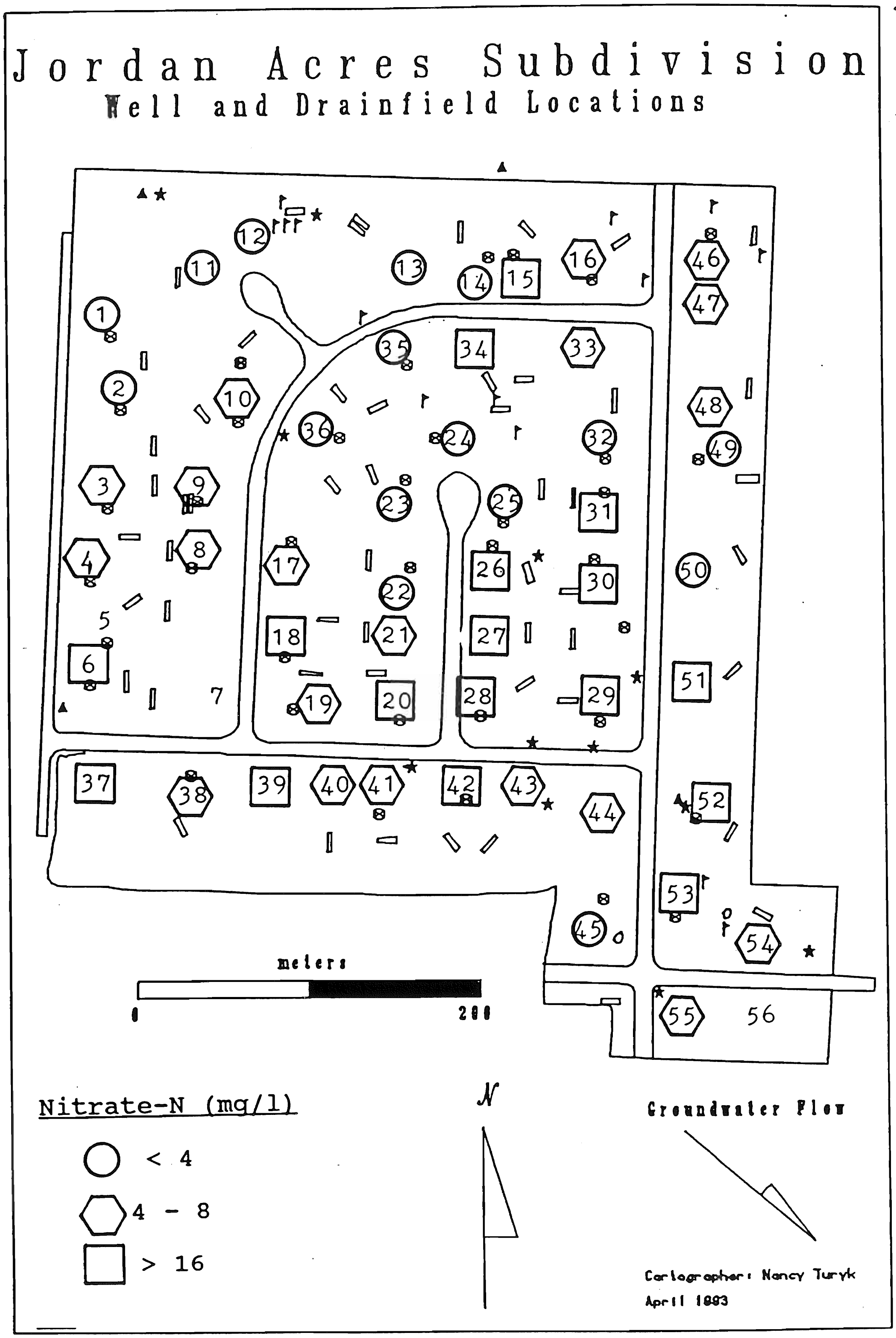

Figure 15: Location, identification, and relative nitrate-N concentration for the private wells in Jordan Acres.

3.1.1.1.1 Upgradient Water Quality

The private well water quality in Jordan Acres is quite good in terms of inorganic chemistry, especially at the upgradient end of the subdivision. Nitrate-N and sodium concentrations were generally less than 5 ppm, and chloride concentrations were around 10 ppm. Phosphate was not detected in any of the upgradient wells. Data from the private wells considered to be upgradient are presented in Table 4, the locations of the wells are shown on Figure 15.

Well Location	# of Samples	Monitoring Period	NO_3-N	Cl	Na	PO_4-P
1	1	May '87	2.8	8.0	3.9	<0.002
10	3	May '87 - June '89	5.1	13.3	2.1	<0.002
11	1	May '87	3.4	12.0	1.4	<0.002
12	3	May '87 - July '90	3.8	12.0	3.3	<0.002
13	4	June '87 - June '89	1.7	12.0	2.0	<0.002
14	1	May '87	3.2	14.0	2.0	<0.002
35	5	Nov. '85 - May '87	1.4	10.2	3.9	<0.002
36	3	June '87 - June '89	3.7	10.7	2.8	<0.002
Average			3.1	11.5	2.7	<0.002

Table 4: Jordan Acres upgradient private water-supply well groundwater chemistry data (in mg/L).

3.1.1.1.2 Downgradient Water Quality

As the groundwater flows beneath the subdivision, the concentration of inorganic contaminants appears to increase. Wells considered to be at the downgradient end of the subdivision had an average nitrate-N concentration in excess of 8 mg/L and approximately 30% of the downgradient wells that were sampled exceeded the 10 mg/L nitrate-N standard. Chloride and sodium concentrations showed similar increases, and phosphate was detected at significant concentrations (>0.01 mg/L) in 5 of the private wells. Downgradient private well chemistry is presented in Table 5; well locations are shown on Figure 15.

Well Location	# of Samples	Monitoring Period	NO_3-N	Cl	Na	PO_4-P
20	6	Apr. '84 - June '89	9.1	17.4	7.75	<0.002
28	2	May '87 - June '88	16.3	21.5	14.3	<0.002
29	6	Nov. '85 - June '89	9.1	19.8	28.6	3.65
41	2	May '87 - Aug. '89	7.7	19.0	1.9	<0.002
42	1	May '87	11.0	16.0	63	<0.002
43	4	May '87 - June '89	6.8	13.3	8.4	1.01
44	5	Nov. '85 - June '89	5.4	10.6	1.6	0.100
45	1	May '87	3.9	10.0	3.2	<0.005
51	1	May '87	10.0	24.0	6.5	0.095
52	4	May '87 - June '89	8.6	15.3	12.75	<0.002
53	5	Nov. '85 - June '89	11.5	16.2	14.3	0.049
54	6	Nov. '85 - June '89	4.5	22.2	11.8	<0.002
55	5	Nov. '85 - June '89	7.0	18.8	7.2	<0.002
Average			8.5	17.2	13.9	0.376

Table 5: Jordan Acres downgradient private water-supply well groundwater chemistry data (mg/L).

Not all of the downgradient private wells showed higher concentrations of inorganic contaminants. Several had chemistry similar to the upgradient wells. Those that had the higher concentrations appeared to be those directly (or nearly so) downgradient of one or more septic-system drainfields. Conversely, those with apparently no drainfields upgradient had lower contaminant levels. As can be seen by comparing Table 5 with Figure 15, it appears that there is a correlation between wells having elevated nitrate-N concentrations and having septic system drainfields directly upgradient of them. Although the locations of the drainfields may not be exact, the general locations are believed to be accurate.

3.1.1.2 Village Green

The location of the private water-supply wells in Village Green are shown in Figure 16. Private well groundwater chemistry data from Village Green is presented in Appendix C. As with Jordan Acres, the relative nitrate-N concentrations are indicated for those wells from which chemistry data were available. The estimated groundwater flow paths leading to the wells are indicated.

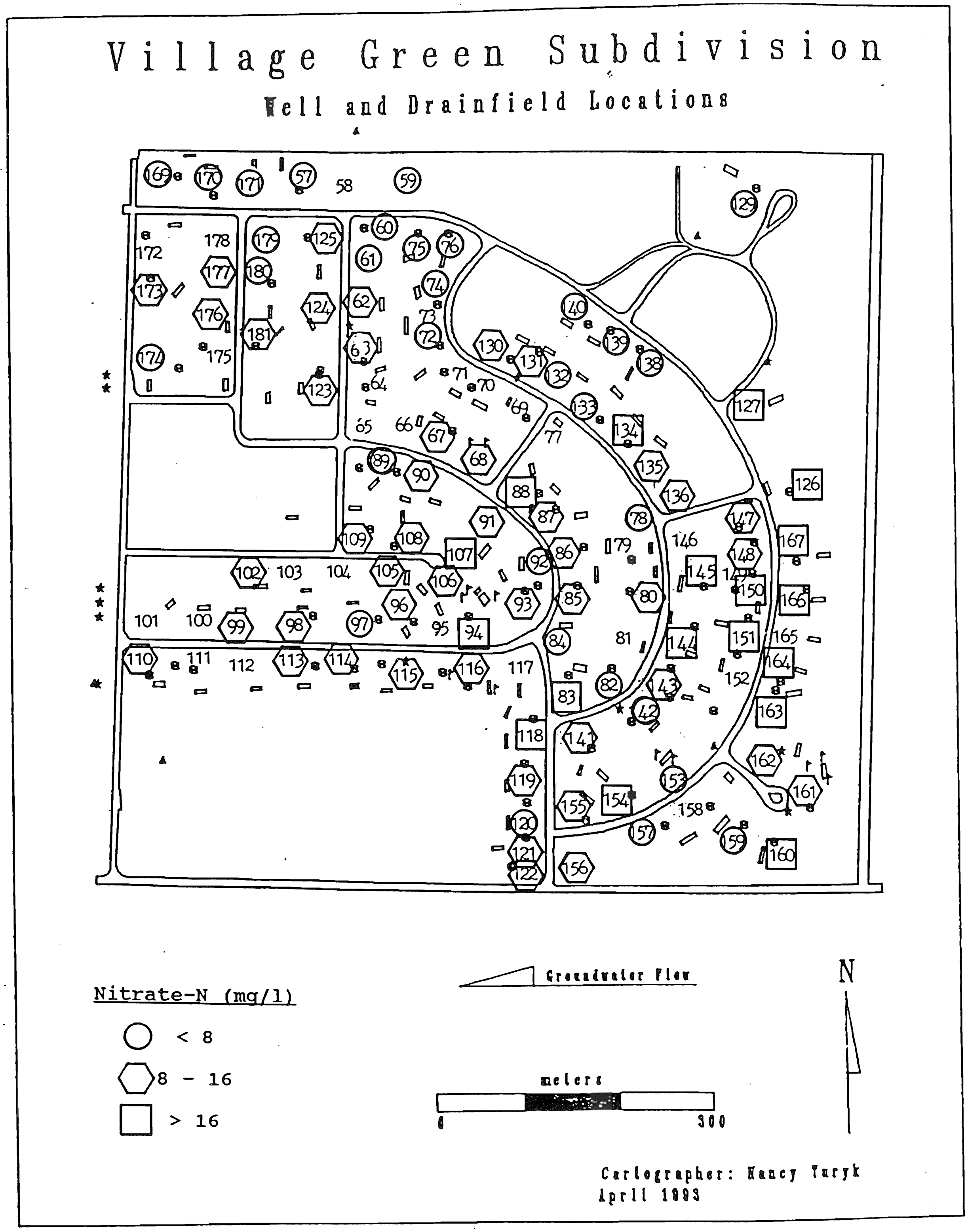

Figure 16: Location, identification, and relative nitrate-N concentrations for the private wells in Village Green.

3.1.1.2.1 Upgradient Water Quality

The chemistry results of water samples obtained from wells considered to be on the upgradient end of Village Green are summarized in Table 6. The location of the wells are shown on Figure 16. Note that all but three of the wells have an average nitrate-N concentration in excess of the Enforcement Standard (10 mg/L) established by the Wisconsin Department of Natural Resources. These three wells are downgradient of a wooded areas that appear to provide for low nitrate-N groundwater recharge. The elevated nitrate-N levels are believed to be due to the extensive irrigated agriculture upgradient of the subdivision. The higher chloride and sodium concentrations are believed to be due to fertilizers and animal wastes and also to road salt applied to S.T.H. "51".

Well Location	# of Samples	Monitoring Period	NO_3-N	Cl	Na	PO_4-P
126	1	May '87	16.3	80	39	<0.005
127	5	Mar. '86 - June '89	16.5	51	19	<0.005
129	3	Jan. '86 - May '87	1.7	31	18	<0.005
138	1	June '87	3.5	27	13	<0.002
139	1	May '87	2.0	18	5.1	<0.005
140	2	Jan. '85 - Jan. '86	2.0	6.5	--	<0.002
147	1	June '87	13.8	48	24	<0.002
149	3	June '87 - June '89	14.4	50	28	<0.002
150	1	June '97	18.5	58	19	<0.002
151	1	June '87	20.5	57	24	<0.002
160	2	July '86	17.4	41	22	0.005
161	5	June '87 - June '89	15.3	53	27	0.064
162	4	Jan. '86 - June '89	15.0	68	25	0.067
163	1	May '87	19.9	34	27	<0.005
164	3	Jan. '86 - June '89	18.0	72	23	<0.002
166	2	Jan. '86 - Mar. '88	17.8	53	NA	0.005
167	1	June '87	17.2	88	38	<0.002
Average			13.5	49	21	0.006

Table 6: Village Green upgradient private water-supply well groundwater chemistry data (in mg/L).

3.1.1.2.2 <u>Downgradient Water Quality</u>

Water quality data for the private wells considered to be on the downgradient end of the subdivision are presented in Table 7; their locations are shown on Figure 16. Because most of the private wells included in Table 7 were sampled on only one occasion, the conclusions based on the data are somewhat speculative; however, there are several observations to note.

Well Location	# of Samples	Monitoring Period	NO_3-N	Cl	Na	PO_4-P
98	1	May '87	11.2	95	47	<0.002
99	2	Jan. '86 - May '87	9.3	53	38	<0.002
102	1	June '87	15.6	34	33	1.40
110	1	June '87	10.2	11	18	0.005
113	4	May '87 - June '89	8.7	28	22	0.300
169	1	June '87	2.5	23	5.5	<0.002
170	1	June '87	3.2	24	9.0	<0.002
173	1	Jan. '86	8.0	21	--	0.012
174	2	Jan. 86 - June '87	6.5	15	13.5	0.069
176	1	Jan. '86	14.5	14	--	0.095
177	1	Jan. '86	10.5	16	--	0.028
Average			9.1	30	23	0.173

Table 7: Village Green downgradient private water-supply well groundwater chemistry data (in mg/L).

The difference in inorganic chemistry in the downgradient wells as compared to the upgradient wells is not as pronounced as in Jordan Acres; however, nitrate-N and chloride concentrations in the downgradient wells are considerably lower (9.1 mg/L and 30 mg/L downgradient vs 13.5 mg/L and 49 mg/L upgradient) and significant phosphate concentrations were detected in more of the wells (6 of 11 downgradient vs 2 of 14 upgradient) at higher concentrations.

The data from the Village Green downgradient private wells are similar to the data from the Jordan Acres downgradient private wells; average nitrate-N concentrations in both subdivisions are about 9 mg/L, phosphate was detected at relatively high concentrations in several wells, and with the exception of a few wells in Village Green (private wells 98, 99, and 102), the average sodium and chloride concentrations are also similar. Note that the wells with the lowest nitrate-N concentrations (private wells 169 and 170) are located in the far northwest corner of the subdivision, which has relatively little upgradient development.

3.1.2 Multiport Wells

The multiport wells, installed to provide a view of groundwater quality over the area and with depth, are likely to give the best indication of the overall water quality beneath the subdivision. The shallower ports reflect the water quality from sources immediately upgradient of the well, be it a lawn, septic system, woods, etc. The deeper ports sample water that has been in the flow system longer, thus the contaminants have had a chance to disperse and mix, thereby averaging the various subdivision recharge sources. The water quality in the deepest ports is presumed to be that of water originating upgradient of the subdivision.

3.1.2.1 Jordan Acres

The multiport wells in Jordan Acres are shown on Figure 17, as is the groundwater flow path to each of the wells. Groundwater chemistry data from the multiport wells in Jordan Acres are presented in Appendix B.

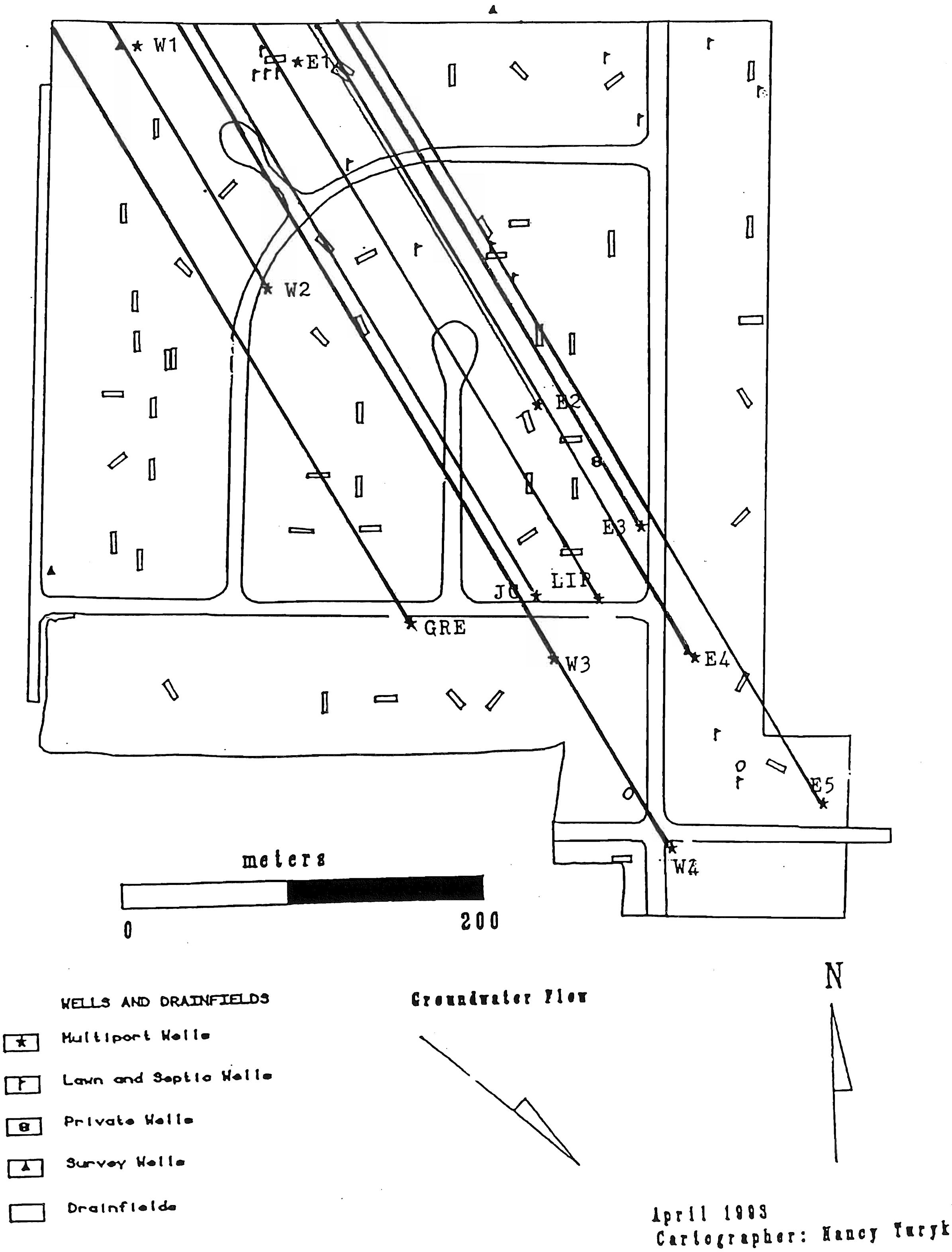

Figure 17: Location and identification of the multiport wells in Jordan Acres. Inferred groundwater flow paths to the wells are indicated.

3.1.2.1.1 Upgradient Water Quality

The water chemistry data from the two upgradient multiport well nests are presented in Table 8. The chemistry is virtually the same as the results obtained from the upgradient private wells—low concentrations of nitrate-N, chloride, sodium, and phosphate. The source of these low levels of contaminants is likely to be the low-intensity agricultural land use upgradient of the subdivision.

Well Location	Well Port	# of Samples	Monitoring Period	NO_3-N	Cl	Na	PO_4-P
E1	22	15	Sep. '87 - Jan. '91	2.4	12.5	2.5	0.002
	25	15	Sep. '87 - Jan. '91	2.7	11.5	2.9	<0.002
	30	15	Sep. '87 - Jan. '91	3.2	11.1	2.5	<0.002
	35	15	Sep. '87 - Jan. '91	3.7	10.3	2.9	<0.002
	40	15	Sep. '87 - Jan. '91	4.1	9.3	3.6	0.002
	45	15	Sep. '87 - Jan. '91	4.4	8.9	4	0.002
	55	15	Sep. '87 - Jan. '91	5.6	10.3	8.4	0.002
	65	15	Sep. '87 - Jan. '91	1.2	2.9	2.6	<0.002
Depth weighted average at E1				3.4	9.6	3.7	<0.002
W1	22	18	July '87 - May '90	3.6	9.7	3.7	<0.002
	25	18	July '87 - May '90	4.9	13.3	5.8	<0.002
	30	18	July '87 - May '90	5.7	12.4	8.5	<0.002
	35	18	July '87 - May '90	6.1	11.5	8.5	<0.002
	40	18	July '87 - May '90	6.4	10.6	8.4	<0.002
	45	17	July '87 - May '90	5.2	7.9	3.9	<0.002
Depth weighted average at W1				5.3	10.9	6.5	<0.002
Average concentrations for all upgradient ports				4.2	10.2	4.9	<0.002

Table 8: Jordan Acres upgradient multiport well groundwater chemistry data (mg/L).

It appears that there is a trend with depth in inorganic chemistry concentrations; nitrate-N and sodium concentrations increase (from an average of around 2.5 mg/L to over 5 mg/L) and chloride concentrations decrease (from averaging over 12 mg/L to less than 10 mg/L). The magnitude of the changes is rather small, thus it is difficult to draw any firm conclusions, but it is likely that "clean" recharge water from the wooded area immediately upgradient of the subdivision is contributing to the low values found near the water table. The higher concentrations in the deeper ports are attributed to the low intensity agriculture and residential housing that occur between the groundwater divide and Jordan Acres subdivision. The deepest port at well E1 shows little impact from any human sources.

Nitrate-N data for all the dates that water samples were obtained from E1 are presented in Table 9. The data are presented graphically in Figures 18a & b. The water chemistry from this well location is considered to be representative of groundwater entering at the upgradient end of the subdivision and as such shows the variability and trends in the upgradient groundwater quality.

Sample Date	Sampling Port							
	E1-22	E1-25	E1-30	E1-35	E1-40	E1-45	E1-55	E1-65
09/24/87	1.5	1.8	3.0	4.5	5.5	7.0	7.8	1.8
11/02/87	1.5	2.0	3.5	4.5	5.5	6.8	7.5	2.2
01/20/88	2.0	2.4	3.5	4.5	6.0	6.6	7.5	4.0
03/29/88	1.8	2.5	3.0	4.2	6.0	6.8	6.5	4.0
05/24/88	2.0	2.5	3.2	4.5	5.2	6.8	6.8	2.2
07/27/88	2.0	2.5	3.5	4.0	3.5	5.0	6.5	0.8
10/12/88	2.2	2.5	3.5	3.5	2.5	3.5	5.8	0.5
03/29/89	3.0	3.0	3.5	3.0	2.5	2.5	3.5	1.5
06/28/89	3.0	3.0	3.0	2.8	2.5	2.2	7.0	< 0.2
08/28/89	3.0	3.0	2.8	2.5	2.8	2.5	6.0	0.5
10/28/89	2.5	3.2	3.0	3.0	3.5	2.5	6.0	0.8
01/08/90	3.0	3.0	3.2	3.5	3.8	3.5	4.5	0.5
05/22/90	2.7	3.0	2.7	3.5	3.2	2.5	3.2	< 0.2
08/13/90	2.7	3.1	3.0	4.1	3.8	2.6	3.6	< 0.2
01/12/91	3.4	3.0	3.0	4.1	5.4	5.5	2.5	0.3
Average	2.42	2.70	3.16	3.75	4.11	4.42	5.65	1.31

Table 9: Groundwater nitrate-N concentrations (in mg/L) for all sampling dates from the ports at well E1 in Jordan Acres.

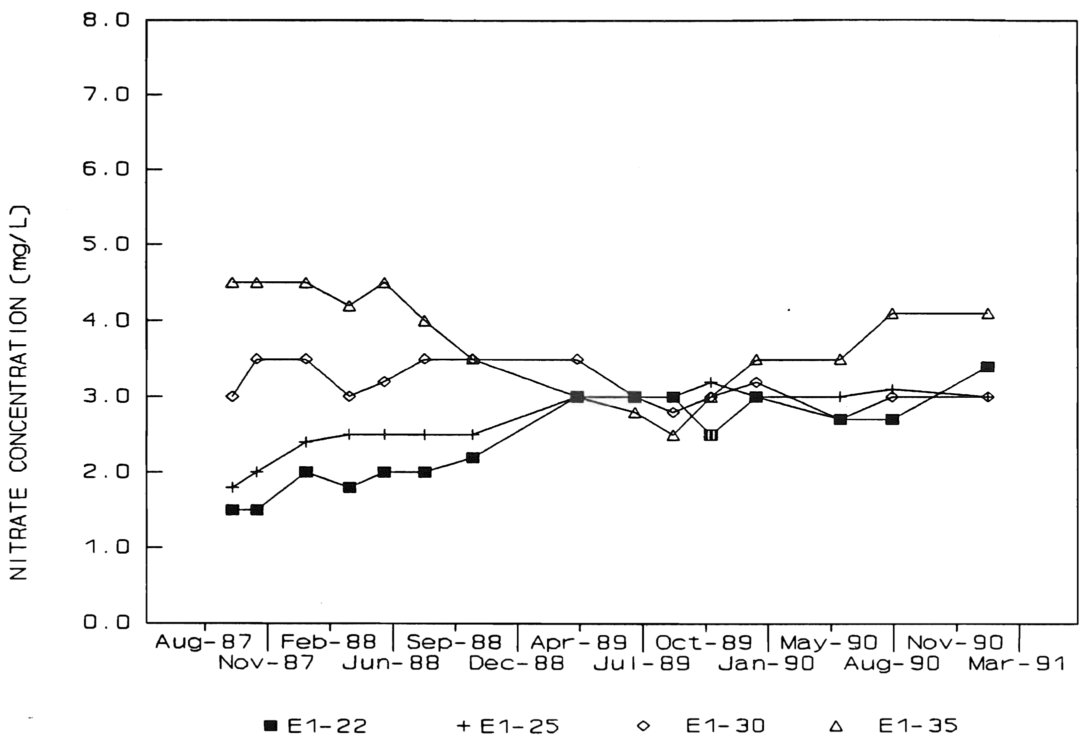

Figure 18a: Plot of nitrate-N concentration vs. time for well E1 in Jordan Acres.

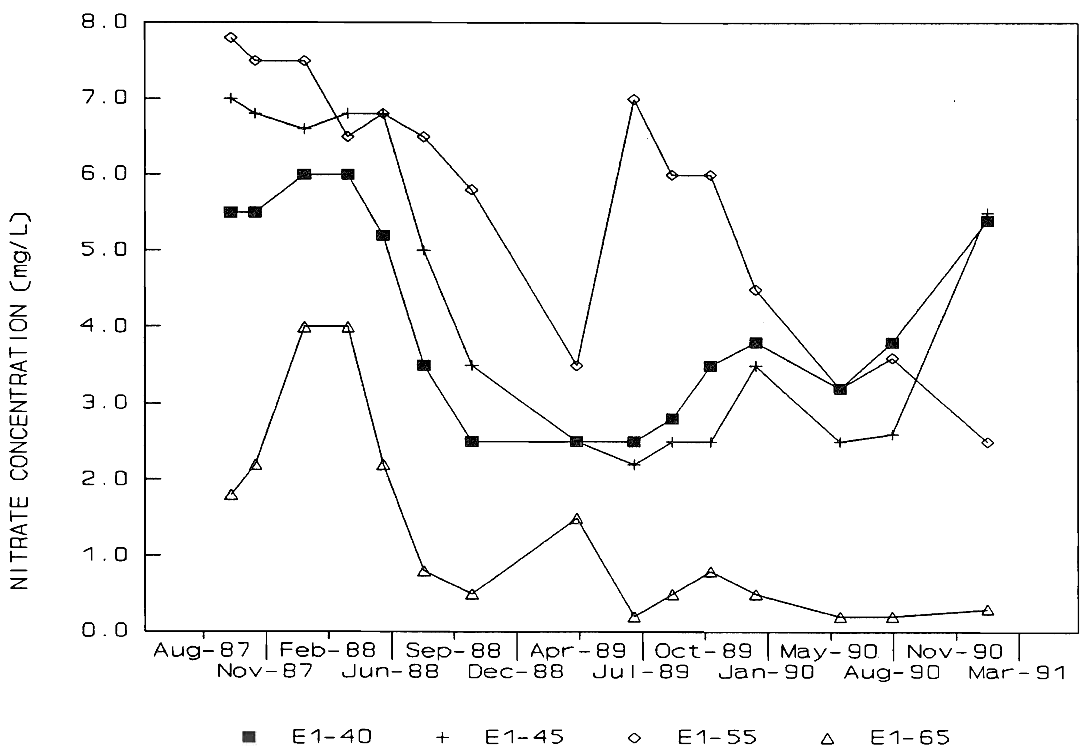

Figure 18b: Plot of nitrate-N concentration vs. time for well E1 in Jordan Acres.

The average nitrate-N concentrations in the shallowest ports (E1-22 and -25) increased from around 1.5 mg/L to over 3.0 mg/L, possibly due to lesser amounts of recharge occurring during the dry years of 1987 through 1989. The concentrations in the third port (E1-30) had remarkably consistent concentrations of around 3.0 mg/L, and reflects an average of the ports immediately above and below.

The concentrations in port E1-35 started at 4.5 mg/L in September 1987, decreased to below 3.0 mg/L by August 1989, then increased back up to over 4.0 mg/L in January 1991. The dip in nitrate-N concentrations can conceivably be attributed to additional dilution water recharged during the wet year of 1984. If this is true, then it would suggest that groundwater present at 3 to 4.5 meters below the water tale entered the groundwater flow system about five years previously.

The deeper ports (E1-40, -45, and -55) not only have higher average nitrate-N concentrations, but the concentrations also show more variability between sampling dates. The variability is likely to be due to the differing chemical characteristics of the recharge water occurring from agricultural fields. The nitrate-N concentrations from port E1-65 were generally quite low (below 1.0 mg/L) except during a few of the samples obtained earlier in the study, at which time concentrations as high as 4.0 mg/L were detected.

3.1.2.1.2 Groundwater Quality Through The Subdivision

The multiport wells were installed in an attempt to monitor the groundwater quality along two transects parallel with groundwater flow. The locations of the wells and the groundwater flow paths leading to them are shown on Figure 17 (p. 57). Wells included in the "East Transect" are labeled E-1, E-2, E-3, E-4, and E-5. Wells included in the "West Transect" are labeled W-1, W-2, W-3, and W-4. Groundwater chemistry data from the wells are summarized in Tables 10 and 11.

The approximate location of most of the septic system drainfields in the subdivision are included on Figure 17, as are other land use characteristics. Aside from the drainfields, it appears that lawns are the primary land use for groundwater recharge and potential impact on groundwater quality.

Well Location	Well Port	Monitoring Period	# of Samples	NO_3-N	Cl	Na	PO_4-P
E1	22	Sep. '87 - Jan. '91	15	2.4	12.5	2.5	0.002
	25	Sep. '87 - Jan. '91	15	2.7	11.5	2.9	<0.002
	30	Sep. '87 - Jan. '91	16	3.2	11.1	2.5	<0.002
	35	Sep. '87 - Jan. '91	15	3.7	10.3	2.9	<0.002
	40	Sep. '87 - Jan. '91	15	4.1	9.3	3.6	0.002
	45	Sep. '87 - Jan. '91	15	4.4	8.9	4.0	0.002
	55	Sep. '87 - Jan. '91	15	5.6	10.3	8.4	0.002
	65	Sep. '87 - Jan. '91	15	1.2	2.9	2.6	<0.002
Depth weighted average				3.4	9.6	3.7	0.000
E2	22	Sep. '87 - Aug. '89	8	2.9	4.8	3.9	0.011
	25	Sep. '87 - May '90	11	3.4	6.5	4.7	0.002
	30	Sep. '87 - May '90	11	2.2	7.5	4.0	0.003
	35	Sep. '87 - May '90	11	1.7	13.0	1.9	0.009
	40	Sep. '87 - May '90	11	2.0	13.0	1.6	0.002
	45	Sep. '87 - May '90	11	2.7	10.5	1.5	0.019
Depth weighted average				2.5	9.2	2.9	0.008
E3	22	July '87 - Aug. '89	4	1.4	14.5	16.2	0.002
	25	July '87 - May '90	14	2.7	19.3	12.1	<0.002
	30	July '87 - May '90	14	2.8	6.8	3.7	0.002
	35	July '87 - May '90	14	1.6	6.5	2.8	0.003
	40	July '87 - May '90	14	0.9	8.5	1.6	<0.002
	45	July '87 - May '90	14	2.3	13.1	1.5	<0.002
Depth weighted average				2.0	11.5	6.3	0.000
E4	22	July '87 - Aug. '89	15	6.3	30.1	18.2	0.120
	25	July '87 - May '90	15	6.6	17.1	11.6	0.117
	30	July '87 - May '90	15	3.9	6.9	3.5	0.004
	35	July '87 - May '90	15	4.9	11.3	2.6	<0.002
	40	July '87 - May '90	15	4.6	12.1	3.3	<0.002
	45	July '87 - May '90	14	5.4	10.5	2.4	<0.002
Depth weighted average				5.3	14.7	6.9	0.039
E5	30	Sep. '87 - Jan. '91	14	11.4	29.9	21.6	<0.002
	35	Sep. '87 - Jan. '91	14	12.3	29.6	18.7	<0.002
	40	Sep. '87 - Jan. '91	14	8.2	12.1	6.7	<0.002
	45	Sep. '87 - Jan. '91	14	9.3	13.0	7.2	<0.002
	50	Sep. '87 - Jan. '91	14	8.1	12.9	5.6	<0.002
	60	Sep. '87 - Jan. '91	13	7.3	12.2	4.0	<0.002
	70	Sep. '87 - Jan. '91	13	3.0	9.2	2.4	<0.002
Depth weighted average				8.5	17.0	9.5	<0.002

Table 10: Jordan Acres East transect groundwater chemistry data in (mg/L).

Well Location	Well Port	Monitoring Period	# of Samples	NO_3-N	Cl	Na	PO_4-P
W1	22	July '87 - May '91	18	3.6	9.7	3.7	<0.002
	25	July '87 - May '91	18	4.9	13.3	5.8	<0.002
	30	July '87 - May '91	18	5.7	12.4	8.5	<0.002
	35	July '87 - May '91	18	6.1	11.5	8.5	<0.002
	40	July '87 - May '91	18	6.4	10.6	8.4	<0.002
	45	July '87 - May '91	17	5.2	7.9	3.9	<0.002
Depth weighted average				5.3	10.9	6.5	<0.002
W2	22	Sep. '87 - Aug. '89	8	2.9	4.8	3.9	0.011
	25	Sep. '87 - May '90	12	7.8	14.8	7.7	0.207
	30	Sep. '87 - May '90	11	5.7	13.7	3.8	0.004
	35	Sep. '87 - May '90	11	3.3	11.2	2.2	0.002
	40	Sep. '87 - May '90	11	2.7	8.3	2.1	0.003
	45	Sep. '87 - May '90	11	3.5	6.9	4.1	0.002
Depth weighted average				4.3	10.0	4.0	0.038
W3	22	June '89	1	52.5	60.0	23.5	<0.002
	25	Sep. '87 - May '90	15	6.8	24.8	12.0	<0.002
	30	Sep. '87 - May '90	15	8.4	13.5	9.1	<0.002
	35	Sep. '87 - May '90	14	7.9	15.6	3.9	<0.002
	40	Sep. '87 - May '90	14	6.0	15.2	3.9	<0.002
	45	Sep. '87 - May '90	14	3.8	10.5	3.7	0.023
Depth weighted average				6.58	15.9	6.5	0.002
W4	25	Sep. '87 - May '90	14	12.1	42.7	26.8	0.158
	30	Sep. '87 - May '90	14	5.5	34.2	13.7	0.010
	35	Sep. '87 - May '90	13	8.8	17.0	2.3	<0.002
	40	Sep. '87 - May '90	13	5.9	14.1	2.0	<0.002
	45	Sep. '87 - May '90	13	5.5	14.2	6.7	0.017
Depth weighted average				7.6	24.4	10.3	0.036

Table 11: Jordan Acres West transect groundwater chemistry data (in mg/L).

The chemistry in the ports at E1 was discussed in section 3.1.2.1.1. The ports E2 and E3 exhibit some of the lowest chemical concentrations of any of the wells included in this study. Average nitrate-N concentrations were generally between 1.0 mg/L and 3.0 mg/L; average chloride concentrations were generally below 15 mg/L and frequently below 10 mg/L; and average sodium concentrations were below 5 mg/L except at the shallowest two ports at well E3 (E3-22 and -25). There are virtually no drainfields upgradient of E2 (the chemicals from the drainfield by E1 will be diluted to insignificant concentrations during the flow time to reach the well) and the plume from the drainfield that was mapped as being directly upgradient of E3 appears to completely miss the well. Only one of the lawns along the flow path is considered to be intensively managed, thus minimizing the impact of lawn fertilizers. The elevated chloride and sodium concentrations detected in the shallower ports at E3 are probably due to salt applied to roads in the subdivision. The chemistry from the deeper ports is similar to that in the shallower ports at E1 (which was attributed to the wooded area). Neither of these two well locations appears to show the effect of the agricultural land in any of their ports.

Phosphate was detected in several of the ports at E2. The average concentrations from the shallower ports are due to high concentrations detected on one or two sampling occasions, which may indicate introduced contaminants during sampling. However, the deepest port had consistently high concentrations of phosphate-P. The source of phosphate is unknown.

The shallower ports at E4 and E5 have relatively high concentrations of nitrate-N, chloride, and sodium (see Table 10). The concentrations are primarily attributed to drainfields. E4 is downgradient of several drainfields just south of E2 and also the two east of W2; E5 appears to be impacted by those drainfields plus the two downgradient of E4. Road salt is also likely to be impacting the wells.

The chemistry from the deeper ports at well E5 (E5-50 and E5-60) may be reflecting the effects of relatively contaminated water from the subdivision (especially drainfields) mixing with the "clean" groundwater flowing into the subdivision from upgradient. The data from the deepest port (E5-70) is similar to the chemistry in the shallower ports at E1 (average nitrate-N concentrations around 3 mg/L, chloride around 10 mg/L, sodium around 2.5 mg/L).

Based on the chemistry profile, the maximum depth of subdivision impact at well E5 may be around 9 meters (30 ft). Note that this depth is based primarily on the nitrate-N concentrations in the deeper ports; the chloride and sodium (and phosphate-P) concentrations are similar to the concentrations at most of the other ports considered to be unaffected by the subdivision. The only ports that are considered to be definitely impacted by the subdivision are E5-30 and E5-35. The depth of subdivision impact will be discussed further hereinafter.

In general, the groundwater from the wells included in the west transect had higher concentrations of the inorganic chemicals than the wells included in the east transect. This is true even at the furthest upgradient well (W1). The higher values may be due to the natural variability of groundwater chemistry; however, it may also suggest that the "plume" of groundwater entering the subdivision along the west transect exhibits a greater impact from agricultural sources. This may be due to the location of upgradient agricultural areas relative to the subdivision. The east edge of the agricultural land roughly coincides with the western edge of the subdivision, thus the west half of the subdivision is slightly closer to the agricultural areas. The position of the agricultural field relative to the subdivision can be seen on Figure 2 (p. 7).

The shallowest port at W2 has low average concentrations of nitrate-N (2.9 mg/L), chloride (4.8 mg/L), and sodium (3.9 mg/L), which suggest little impact from subdivision sources. The relatively average high nitrate-N concentrations at W2-25 and W2-30 (7.8 mg/L and 5.7 mg/L) may be due to one or both of the drainfields south and east of W1, but the data are generally similar to the chemistry observed at the intermediate depths in the upgradient wells. Likewise the chemistry at the deeper ports can be attributed to upgradient groundwater. The phosphate concentrations in the shallowest two ports at W2 are relatively high, possibly due to fertilizers applied to the lawn in which the well is located. The high phosphate concentrations were consistent for all sampling events.

The drainfields directly downgradient of W2 may be contributing to the higher average nitrate-N concentrations (6 mg/L to 8 mg/L) observed in the middle ports at W3 (W3-25, -30, -35, -40); however, the concentrations are not dissimilar to those observed at the intermediate ports at W1 and W2. The chloride and sodium concentrations in the shallowest two ports (W3-25 and W3-30) are likely to be at least partially due to inputs from the subdivision. Except for the phosphate data, the chemistry at the deepest port at W3 (3.8 mg/L nitrate-N, 10.5 mg/L chloride, 3.7 mg/L sodium) is virtually the same as the shallowest port at W1. This may suggest that the depth of subdivision impact is less than the upper 25 feet of the aquifer; however, again the magnitude of the differences is too small to draw any firm conclusions.

The cause of the high phosphate concentration at the deepest port at W3 is unknown, but phosphate was generally at or below detection limits for all the sampling dates except for the last two, at which time the concentrations were 0.1 mg/L and 0.2 mg/L.

The shallowest port at W4 (W4-25) is believed to be impacted primarily by the dry well serving the residence on the northwest corner of the intersection by the well. The elevated average chloride and sodium concentrations (34.2 mg/L and 13.7 mg/L) in the next port down (W4-30) are also likely to be due to contaminants from the dry well or road salt. Because the average sodium concentrations in ports W4-35 and W4-40 are so low (around 2 mg/L), it is difficult to directly attribute the chemistry to subdivision sources, although lawn fertilizers will tend to contribute nitrogen and chloride but not sodium. The chemistry in the deepest port (W4-45) is suggestive of the upgradient groundwater quality and appears to show little impact from the subdivisions.

Vertical profiles showing chemistry data with depth at the wells along the East transect are presented in Figures 19a & b. Figure 19a shows the nitrate-N concentrations; Figure 19b shows the ratio of chloride to sodium. The ratio of chloride concentration to sodium concentration in the groundwater was considered to be a potential tracer in the groundwater. Using the atomic weights of the elements (35.5 for chloride, 23.0 for sodium), the ratio of dissolved ions from common salt (NaCl) is approximately 1.5. Fertilizers frequently contain chloride (from KCl), but little sodium, thus there will be a higher chloride to sodium ratio in water showing fertilizer impacts. Based on data from Shaw and Turyk (1992) and from this study (see section 3.1.3.3), water impacted by septic systems tends to have similar concentrations of sodium and chloride and thus ratios closer to one (1). Therefore, chloride to sodium ratios in the groundwater recharged from subdivision sources are likely to be lower than those showing agricultural impacts.

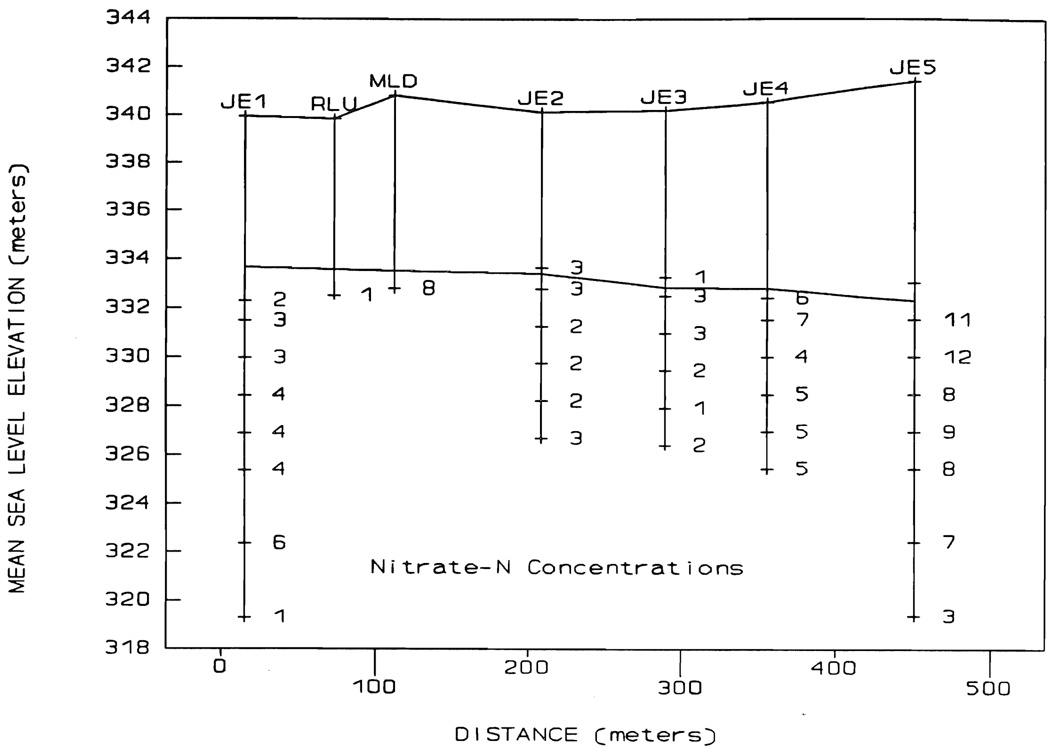

Figure 19a: Vertical profile showing the average nitrate-N concentration at each port for the wells included in the East transect in Jordan Acres.

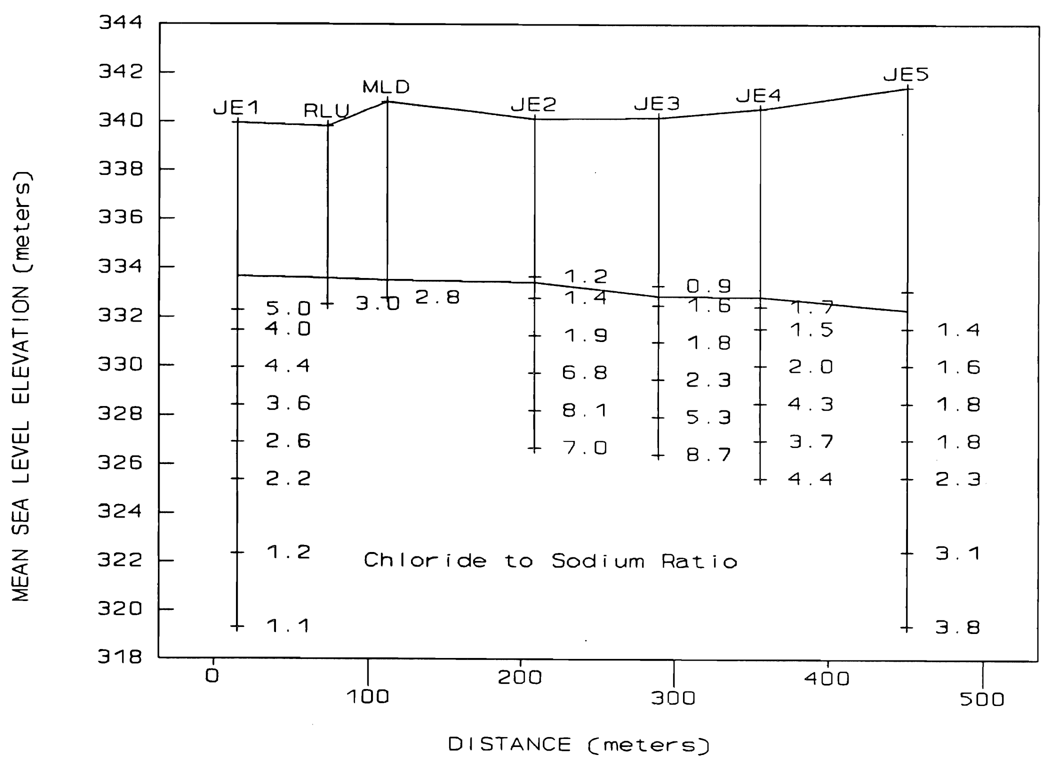

Figure 19b: Vertical profile showing the average chloride to sodium ratio at each port for the wells included in the East transect in Jordan Acres.

3.1.2.1.3 Downgradient Water Quality

In order to calculate the nitrogen and water budgets based on field data, it was necessary to define the portion of the aquifer impacted by recharge from the subdivision. It is assumed that there are three zones of water chemistry within the aquifer; water that originated from within the subdivision, a mixture of subdivision-recharged water and water originating from upgradient of the subdivision, and water that originated solely from upgradient of the subdivision. The chemistry data from the downgradient multiport wells (summarized in Table 12) are used to defined these zones. The locations of the wells are shown on Figure 17 (p. 57).

Comparing the chemistry data in the downgradient wells with the data from the upgradient wells, several well ports appear to be obviously impacted by subdivision sources. Other ports are likely to be reflecting a mixture of recharge from impacted and non-impacted areas. The deepest ports appear to be totally unaffected by subdivision impacts.

Well Location	Well Port	Monitoring Period	# of Samples	NO_3-N	Cl	Na	PO_4-P
GRE	25	Aug. '89 - May '90	5	8.7	51.6	29.9	0.019
	30	Aug. '89 - May '90	5	10.1	23.4	12.4	0.005
	35	Aug. '89 - May '90	5	6.8	14.4	3.7	<0.002
	40	Aug. '89 - May '90	5	6.9	15.8	2.4	0.004
	45	Aug. '89 - May '90	5	2.6	14.0	2.5	0.003
	50	Aug. '89 - May '90	4	2.3	13.0	2.3	0.003
	60	Aug. '89 - May '90	4	0.3	4.5	2.0	0.002
	70	Aug. '89 - May '90	4	0.3	6.0	2.7	0.007
			Depth weighted average	4.8	17.8	7.2	0.005
W3	25	Sep. '87 - May '90	15	6.8	24.8	12	<0.002
	30	Sep. '87 - May '90	15	8.4	13.5	9.1	<0.002
	35	Sep. '87 - May '90	14	7.9	15.6	3.9	<0.002
	40	Sep. '87 - May '90	14	6.0	15.2	3.9	<0.002
	45	Sep. '87 - May '90	14	3.8	10.5	3.7	0.023
			Depth weighted average	6.6	15.9	6.5	0.003
W4	25	Sep. '87 - May '90	14	12.1	42.7	26.8	0.158
	30	Sep. '87 - May '90	14	5.5	34.2	13.7	0.010
	35	Sep. '87 - May '90	13	8.8	17.0	2.3	<0.002
	40	Sep. '87 - May '90	13	5.9	14.1	2.0	<0.002
	45	Sep. '87 - May '90	12	5.0	9.4	2.3	<0.002
			Depth weighted average	7.5	23.5	9.4	0.032
JC	25	Mar. '89 - May '90	7	7.3	28.4	14	0.027
	30	Mar. '89 - May '90	7	7.4	18.1	9.9	0.026
	35	Mar. '89 - May '90	7	7.1	16.0	2.6	0.014
	40	Mar. '89 - May '90	7	8.7	15.7	6.1	0.013
	45	Mar. '89 - May '90	5	6.2	19.0	8.4	0.006
			Depth weighted average	7.3	19.4	8.2	0.017

Table 12: Jordan Acres downgradient multiport well groundwater chemistry data (in mg/L).

Well Location	Well Port	Monitoring Period	# of Samples	NO_3-N	Cl	Na	PO_4-P
LIP	25	Mar. '89 - May '90	5	32.2	43.4	52.7	0.052
	30	Mar. '89 - May '90	5	5.7	16.6	11.0	0.016
	35	Mar. '89 - May '90	5	5.5	18.2	17.8	0.003
	40	Mar. '89 - May '90	5	6.1	18.0	8.4	<0.002
	45	Mar. '89 - May '90	5	8.2	15.4	4.5	<0.002
	50	Aug. '89 - May '90	4	8.2	11.8	5.3	<0.002
	60	Aug. '89 - May '90	4	3.4	3.8	2.1	<0.002
	70	Aug. '89 - May '90	3	-0.2	2.7	1.8	<0.002
Depth weighted average				8.6	16.2	13.0	0.008
E3	22	July '87 - Aug. 89	4	1.4	14.5	16.2	0.002
	25	July '87 - May '90	14	2.7	19.3	12.1	<0.002
	30	July '87 - May '90	14	2.8	6.8	3.7	0.002
	35	July '87 - May '90	14	1.6	6.5	2.8	0.003
	40	July '87 - May '90	14	0.9	8.5	1.6	<0.002
	45	July '87 - May '90	13	2.3	13.1	1.5	<0.002
Depth weighted average				2.0	11.5	6.3	<0.002
E4	22	July '87 - May '90	15	6.3	30.1	18.2	0.120
	25	July '87 - May '90	15	6.6	17.1	11.6	0.117
	30	July '87 - May '90	15	3.9	6.9	3.5	0.004
	35	July '87 - May '90	15	4.9	11.3	2.6	<0.002
	40	July '87 - May '90	15	4.6	12.1	3.3	<0.002
	45	July '87 - May '90	14	5.4	10.5	2.4	<0.002
Depth weighted average				5.3	14.7	6.9	0.039
E5	30	Sep. '87 - May '90	14	11.4	29.9	21.6	<0.002
	35	Sep. '87 - May '90	14	12.3	29.6	18.7	<0.002
	40	Sep. '87 - May '90	14	8.2	12.1	6.7	<0.002
	45	Sep. '87 - May '90	14	9.3	13.0	7.2	<0.002
	50	Sep. '87 - May '90	14	8.1	12.9	5.6	<0.002
	60	Sep. '87 - May '90	13	7.3	12.2	4.0	<0.002
	70	Sep. '87 - May '90	13	3.0	9.2	2.4	<0.002
Depth weighted average				8.5	17.0	9.5	<0.002

Table 12 (continued): Jordan Acres downgradient multiport well groundwater chemistry data (in mg/L).

The chemistry in the shallowest two ports at well GRE (GRE-25 and GRE-30) are obviously impacted by subdivision sources, as evidenced somewhat by the higher average nitrate-N concentrations (8.7 mg/L and 10.1 mg/L), but more so by the relatively high average concentrations of chloride and sodium (chloride = 51.6 mg/L and 23.4 mg/L; sodium = 29.9 mg/L and 12.4 mg/L). The water in the middle ports (GRE-35 and GRE-40) is also likely to be showing impacts from subdivision sources (average nitrate-N around 7 mg/L, average chloride around 15 mg/L); however, the average sodium concentrations are lower than would be expected if the contaminants were from septic systems, thus the water may have recharged from lawns and/or be showing the effects of mixing with upgradient water. The deeper middle ports (-45 and -50) show little if any subdivision impacts, while the deepest ports (GRE-60 and GRE-70) appear to be completely unaffected by subdivision contaminant sources (average nitrate-N = 0.3 mg/L, average chloride = 4.5 mg/L and 6 mg/L, average sodium = 2.0 mg/L and 2.7 mg/L).

The water chemistry at JC is similar at all of the ports (see Table 12) except for the low average concentration of sodium at JC-35 (2.6 mg/L). Wells JC-25 and JC-30 have relatively high average sodium and chloride concentrations (chloride = 28.4 mg/L and 18.1 mg/L; sodium = 14 mg/L and 9.9 mg/L), which suggests that these ports are monitoring the groundwater from subdivision sources. The average nitrate-N and chloride concentrations in the deeper ports are higher than was detected in the upgradient wells, which may indicate that the deeper ports are at least partially impacted by recharge from the subdivision. The source of the phosphate detected the ports is unknown but is likely to be due to subdivision impacts. The fact that the chemistry is similar for all the ports raises the question of whether the well is allowing preferential mixing of the groundwater in the vicinity of the well.

The high average concentrations of nitrate-N (32.2 mg/L), chloride (43.4 mg/L), sodium (52.7 mg/L), and phosphate (0.052 mg/L) at LIP-25 indicate that this port is impacted by a septic system drainfield, probably the one serving the house on the lot in which the well is located. The elevated average sodium and chloride concentrations in the next two ports (LIP-30 and LIP-35) are also likely to be due to drainfields (the two directly upgradient of the previously mentioned drainfield). The chemistry in the ports LIP-40, -45, and -50 (nitrate-N 6.1 to 8.2 mg/L, chloride 11.8 to 18.0 mg/L, sodium 4.5 to 8.3 mg/L) is likely to be due in part subdivision sources, perhaps from the septic systems near the upgradient end of the subdivision. As with the deepest ports at GRE, LIP-60 and LIP-70 are considered to be free of water recharged from the subdivision.

Observations regarding the chemistry in the other wells was discussed in the preceding section.

As with the wells along the East Transect, vertical profiles of the downgradient multiport wells showing average nitrate-N and average chloride to sodium ratios are presented in Figures 20a & b. The profile is generally perpendicular to groundwater flow.

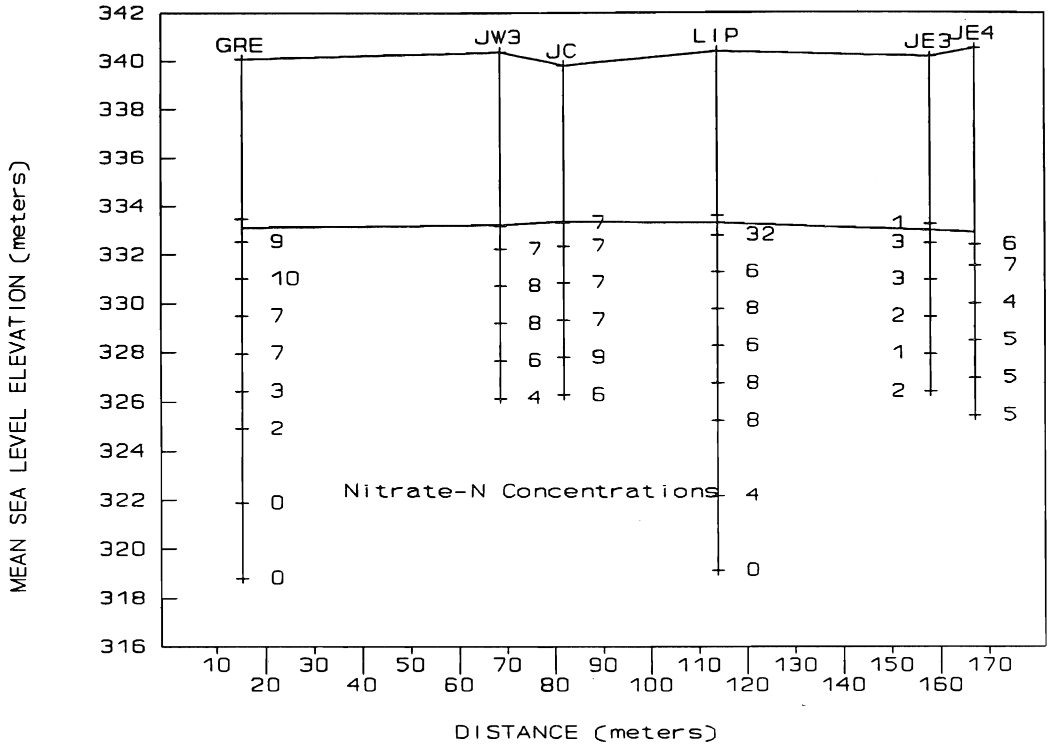

Figure 20a: Vertical profile showing the average nitrate-N concentration at each port for the wells included in the downgradient cross section in Jordan Acres.

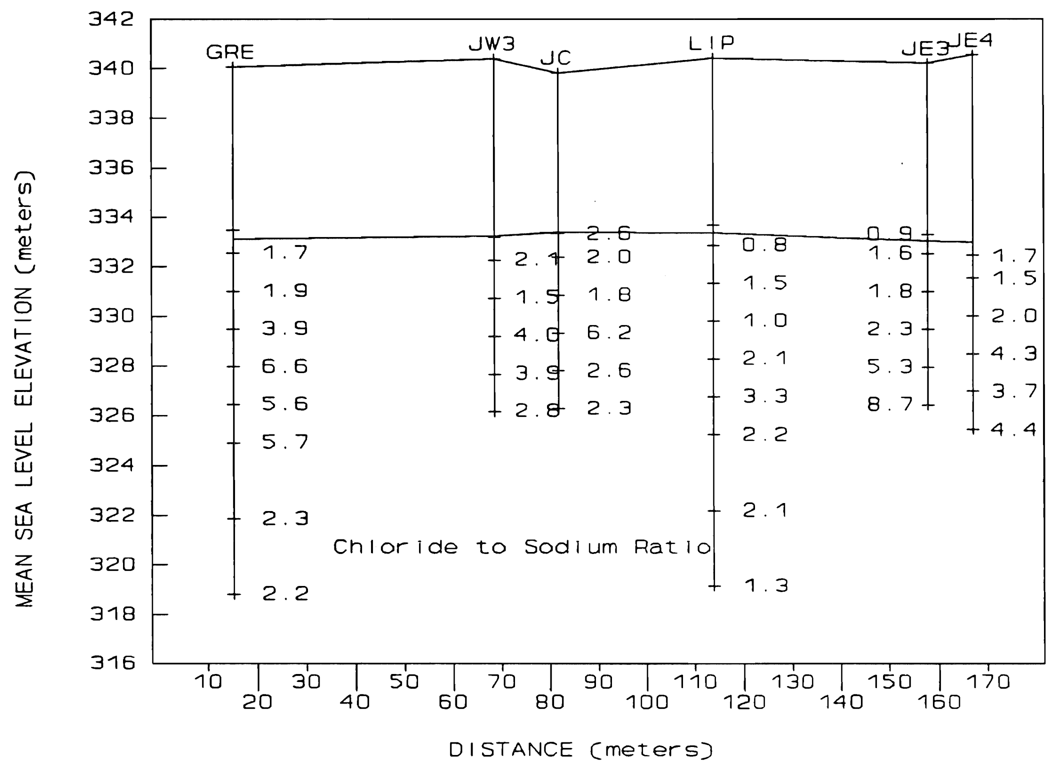

Figure 20b: Vertical profile showing the average chloride to sodium ratio at each port for the wells included in the downgradient cross section in Jordan Acres.

The nitrate-N concentrations for all the dates that samples were obtained from the ports at well East 5 are presented in Table 13. The data are presented graphically in Figure 21. It can be seen that although the nitrate-N concentrations in the shallower ports at well E1 were very consistent, the nitrate-N concentrations in the shallower ports at well E5 were much more variable. Conversely, the deeper ports at well E1 had more variability in the nitrate-N concentrations than the shallower ports, whereas the concentrations in the deeper ports at well E5 were more consistent than the shallow ports. This suggests that the chemical characteristics of the groundwater due to recharge from the subdivision is not uniform.

Sample Date	E5-30	E5-35	E5-40	E5-45	E5-50	E5-60	E5-70
09/24/87	3.8	8.0	6.5	4.0	4.5	6.3	4.5
11/02/87	5.0	7.5	8.0	4.5	4.2	6.5	4.5
01/20/88	9.2	7.8	11.2	6.5	4.8	8.5	4.2
03/29/88	9.0	8.5	11.5	13.2	6.5	9.5	3.2
05/24/88	9.2	8.4	11.5	7.0	8.5	10.5	3.0
07/27/88	10.0	6.8	11.2	7.5	5.0	9.8	3.0
10/12/88	5.0	5.2	11.5	10.5	5.8	9.5	2.5
03/29/89	9.5	24.5	7.8	10.0	10.8	6.5	2.4
06/27/89	27.5	15.5	5.7	12.2	12.0	7.0	2.2
08/29/89	13.0	10.2	7.8	13.0	13.5	8.8	2.5
10/28/89	15.2	20.5	7.5	9.0	12.8	< 0.2	2.5
01/15/90	15.5	7.5	4.5	8.5	8.5	7.2	2.5
03/28/90	10.0	20.0	6.0	10.5	8.0	-	-
05/23/90	17.8	21.5	4.5	13.5	9.0	4.5	2.5
Average	11.41	12.28	8.23	9.28	8.14	7.29	3.04

Table 13: Groundwater nitrate-N concentrations (in mg/L) for all sampling dates from all ports at well E5 in Jordan Acres.

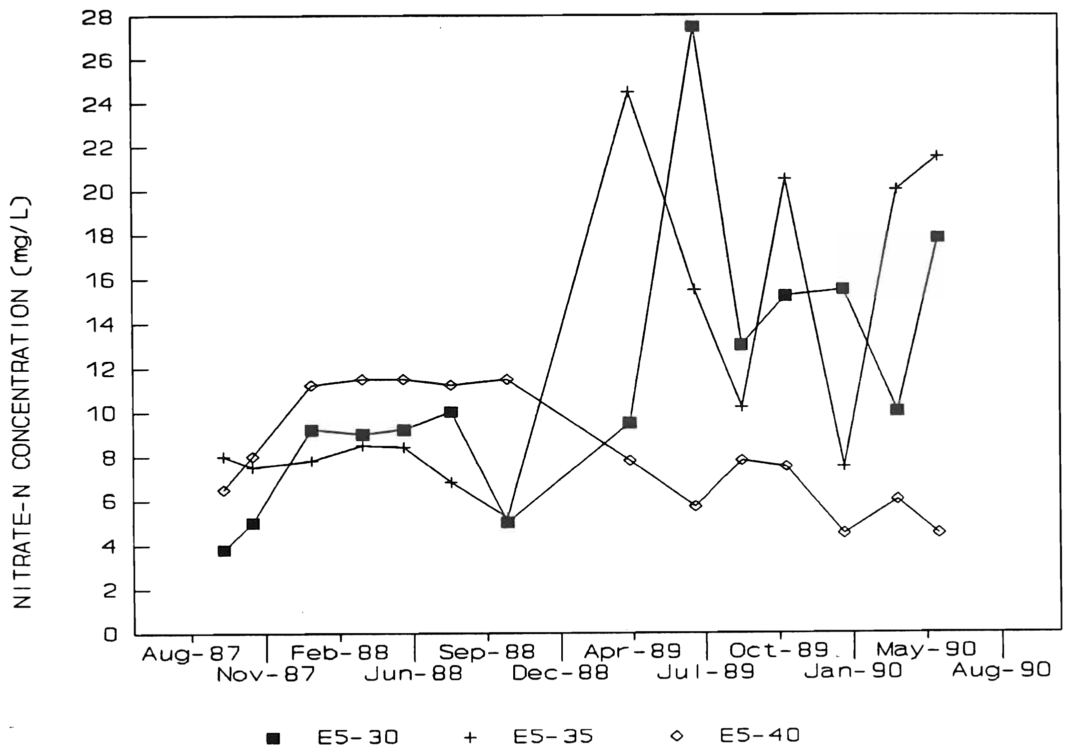

Figure 21a: Plot of nitrate-N concentration vs. time for well E5 in Jordan Acres.

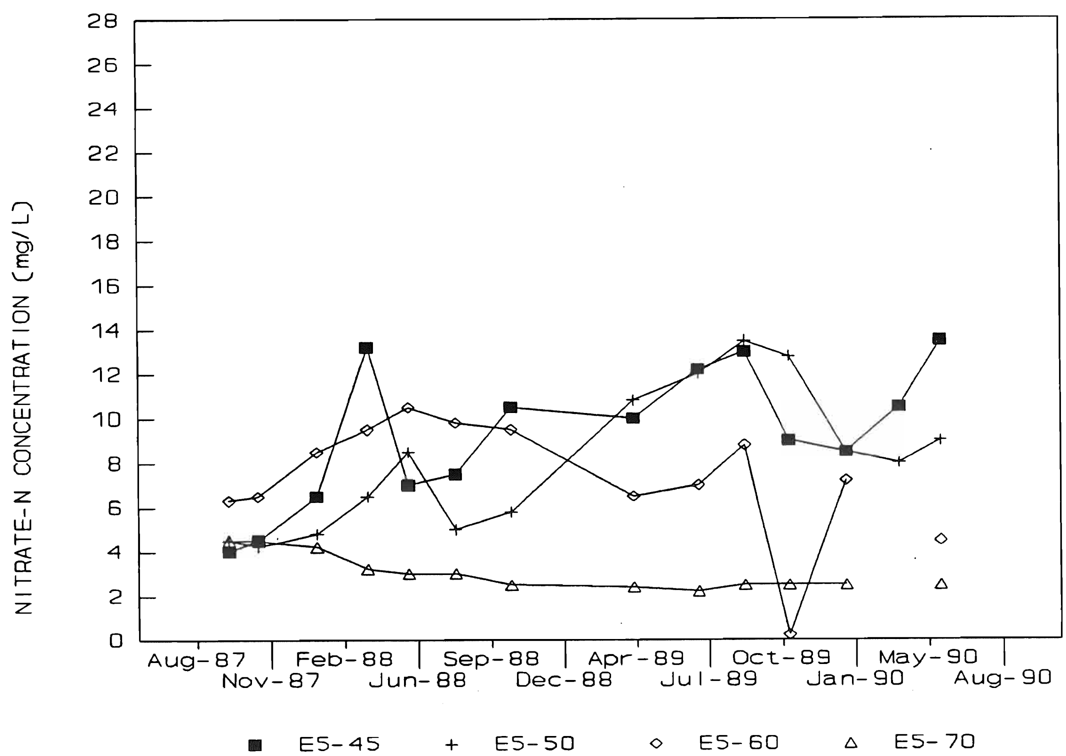

Figure 21b: Plot of nitrate-N concentration vs. time for well E5 in Jordan Acres.

3.1.2.1.4 Depth of Subdivision Impact

As stated previously, the groundwater chemistry in the downgradient multiport wells was used to define the three zones in the aquifer relevant to subdivision recharge (subdivision-only, mixing zone, and upgradient-only). Graphs showing chemistry data verses depth at the downgradient wells are presented at the end of this section (Figures 22 a through j). The chemistry data are presented in Table 12 and/or Appendix B. The estimated depths separating the zones are included on the graphs.

The top of the aquifer was defined as being equal to the average water table elevation at well E3 (333.1 m M.S.L.). This elevation is slightly lower than the shallowest port at this well from which chemistry data are available; however, it is likely that the actual elevation at this port is somewhat lower than presented in the graphs (i.e., the port was installed deeper than was calculated).

The line separating the bottom of the subdivision-only zone and the top of the mixing zone was defined at 331.8 meters. It was positioned based on the observation that (in general) the chemical concentrations in the ports above this line are noticeably different from the ports below this line (especially on the graphs of chloride, sodium, alkalinity, total hardness, and pH). The thickness of the subdivision-only zone is calculated to be 1.3 meters (333.1 m - 331.8 m).

The line defining the bottom of the mixing zone (top of the upgradient-only zone) is at an elevation of 323.6 and was positioned because it appeared that the chemistry data from the deepest ports at GRE and LIP were noticeably different from the ports above. The thickness of the mixing zone is calculated to be 8.2 meters (331.8 m - 323.6 m).

The ports considered to be in the mixing zone, especially the -30, -35, -40, and -45 ports (about 3 to 8 meters below the water table), all have similar chemistry concentrations. This suggests that the groundwater is mixed uniformly throughout this portion of the aquifer. While dispersion is considered to be the primary mixing influence in the aquifer, the water-supply wells will also contribute to plume mixing by creating sporadic localized variations in groundwater flow. It is believed that many of the private wells in the subdivisions were also installed in the upper 3 to 8 meters of the aquifer. A survey of subdivision homeowners indicated that the average depth of the private wells in the subdivision is around 8.5 meters (28 feet); the minimum and maximum reported values were 4.0 meters (13 ft) and 13 meters (42 ft). Most people were uncertain of the actual depth of their well and the reported average depth may be a little too shallow. According to information obtained from local plumbing contractors, the bottoms of private water-supply wells are generally installed at least 3 meters (ten feet) below the top of the water table. In Jordan Acres the water table is about 6 meters deep, thus the well points are likely to be installed at a depth closer to 9 meters. It is worth noting that during the summer of 1989, the third consecutive year of relatively dry weather (and low water table), a discussion with a homeowner revealed that the well serving his residence had to be driven deeper because it was going dry (pumping air and loosing its prime).

The total volume of water that originated from within the subdivision is considered to include all the water in the "subdivision-only" zone and a fraction of the water in the mixing zone. The thickness of the subdivision-only water is 1.3 meters;

the thickness of the mixing zone is 8.2 meters.

The fraction of the water in the mixing zone that originated from subdivision sources is difficult to define. Ideally, assuming the subdivision is contributing water with relatively high chemical concentrations and the upgradient water has low chemical concentrations, then the chemical profile will have a uniform slope of high to low concentrations with depth. Half the water in this zone will have originated from the subdivision and the other half from upgradient. However, because the private wells are considered to be significant mixing agents in the upper portion of the aquifer, and because the average well depth is deemed to be below the depth of subdivision-only impacted water, it is likely that more of the water in the mixing zone will be upgradient water. Thus the percentage of subdivision-water in the mixing zone will be less than 50%. Determination of the actual percentage of subdivision-water in the mixing zone is rather speculative, but a value of 25% is considered to be reasonable and yields results for the water balance similar to those predicted by the BURBs program (see section 3.2).

The discharge of subdivision water is calculated by multiplying the equivalent area discharging subdivision water by the aquifer's hydraulic conductivity and the hydraulic gradient. The equivalent depth of subdivision water is 3.4 m ((333.1 - 331.8) + 0.25 x (331.8-323.6)). The width of the section is 180 m. The hydraulic conductivity is estimated to be in the range of 39 m/day to 73 m/day. A typical hydraulic gradient in this subdivision is 0.0026 m/m. These values yield a range of discharge volumes of 62 m^3/day to 120 m^3/day (23,000 m^3/year to 43,000 m^3/year).

The average nitrate-N concentration from the upper two ports at all the downgradient wells was used to define the average nitrate-N concentration of the water recharged from the subdivision. These are the ports considered to be in the "subdivision-only" zone of the aquifer. The average nitrate-N concentration of these ports is 9.0 mg/L. Therefore, given this average concentration of nitrate-N and the range in annual discharge volumes given above, the total mass of nitrogen in the groundwater discharge from the subdivision ranges from 200 kg/yr to 390 kg/yr.

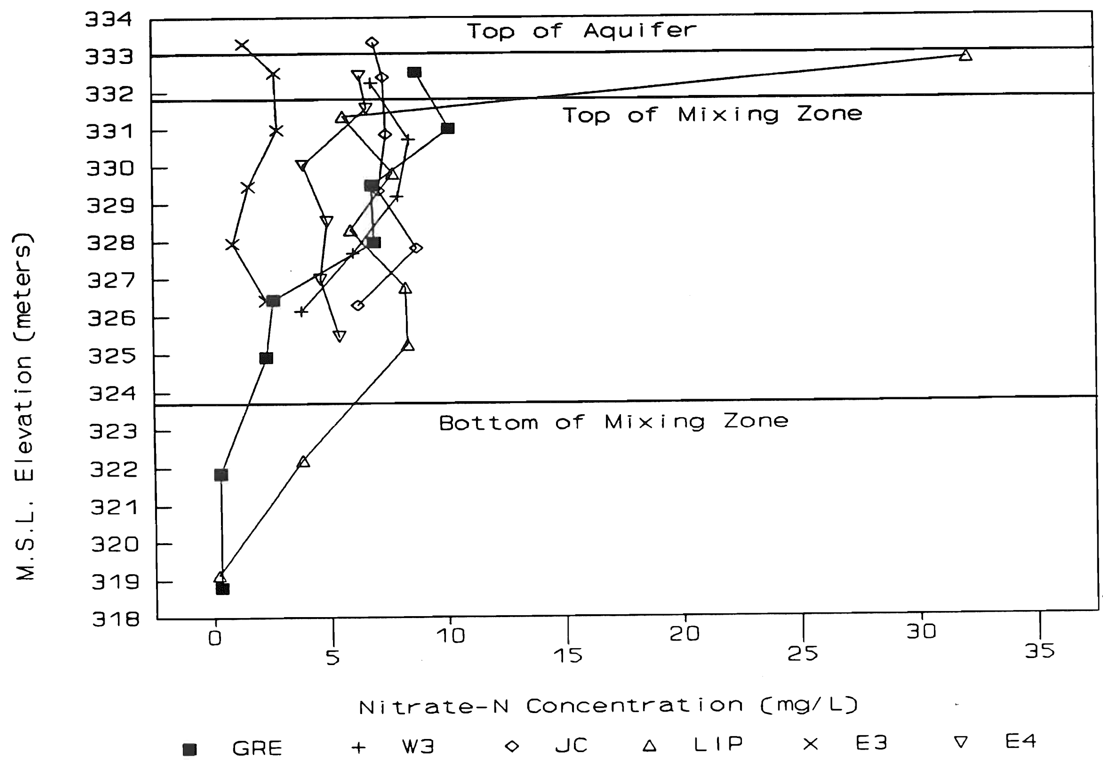

Figure 22a: Graph of average nitrate-N concentrations vs. elevation at the Jordan Acres downgradient wells.

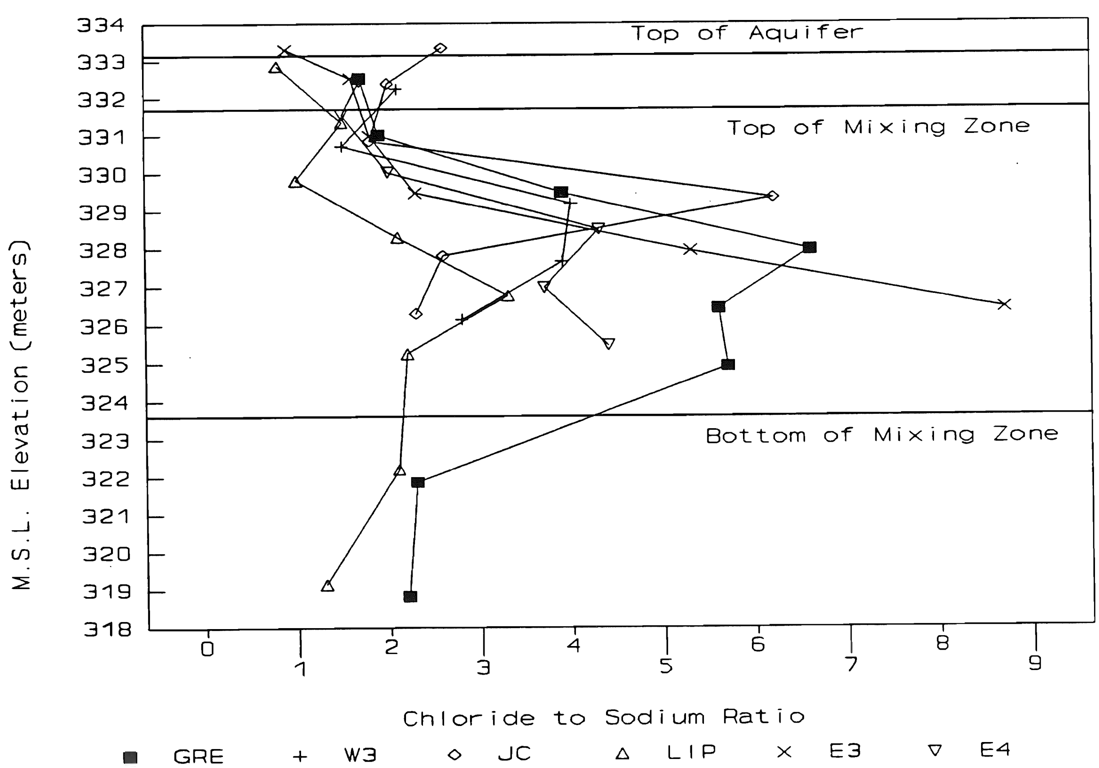

Figure 22b: Graph of average chloride to sodium ratios vs. elevation at the Jordan Acres downgradient wells.

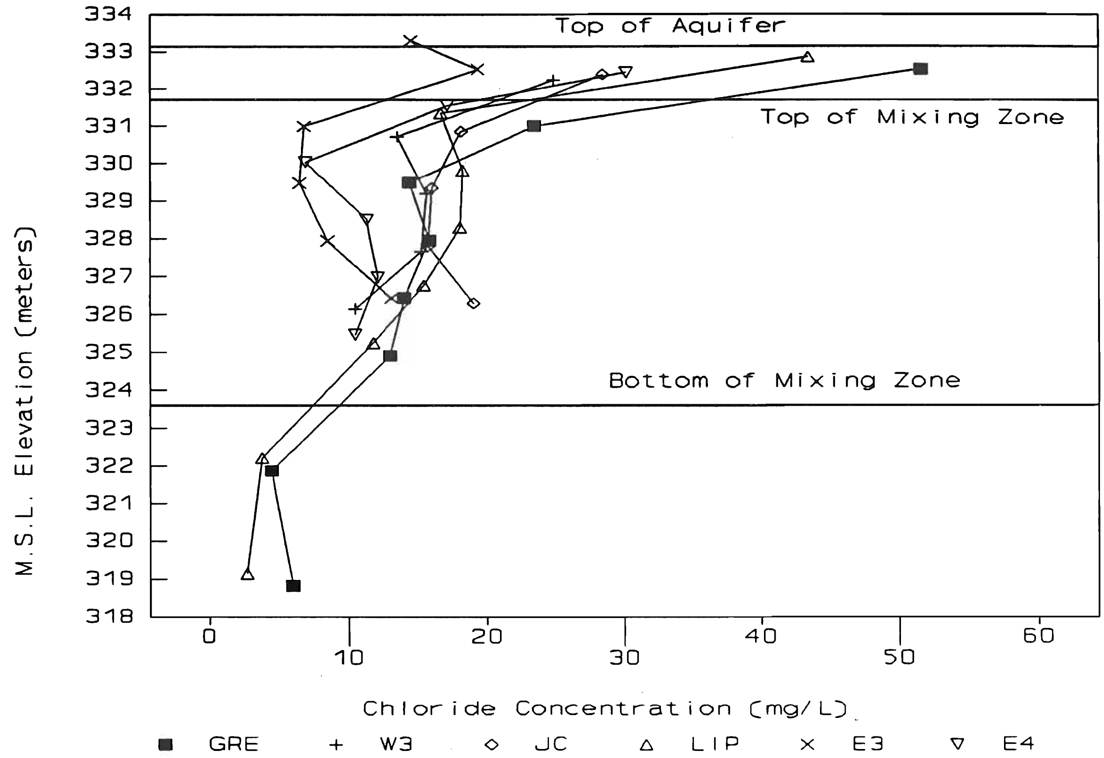

Figure 22c: Graph of average chloride concentrations vs. elevation at the Jordan Acres downgradient wells.

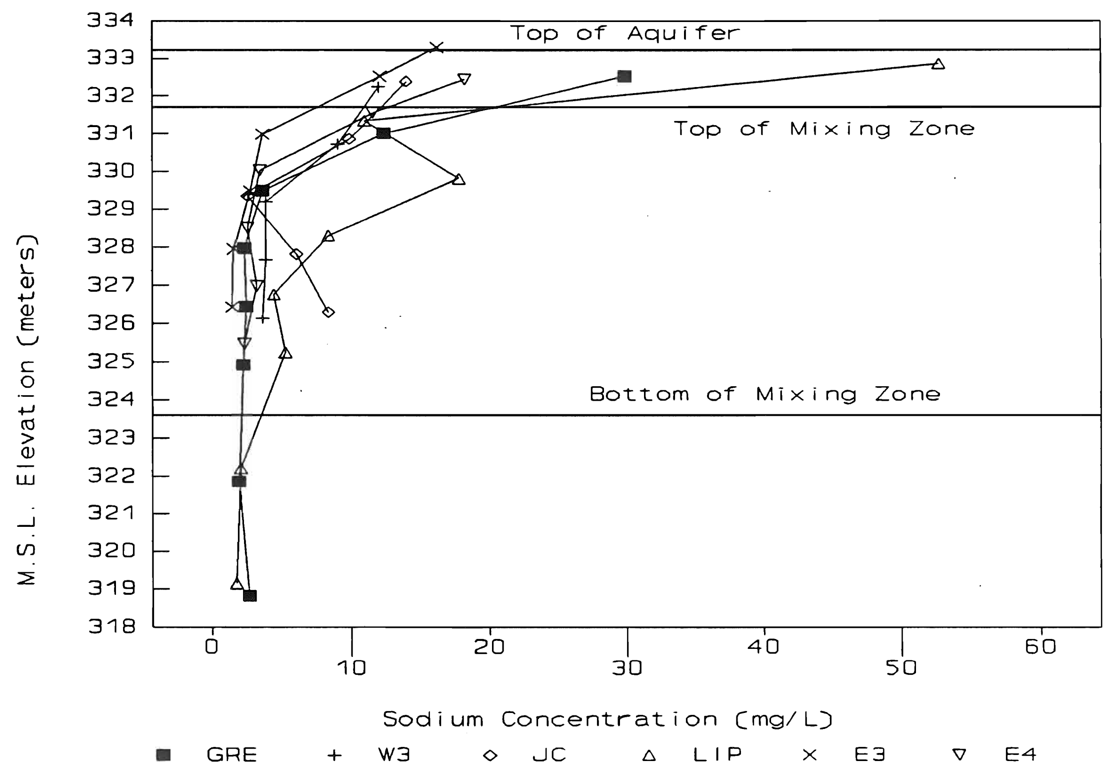

Figure 22d: Graph of average sodium concentrations vs. elevation at the Jordan Acres downgradient wells.

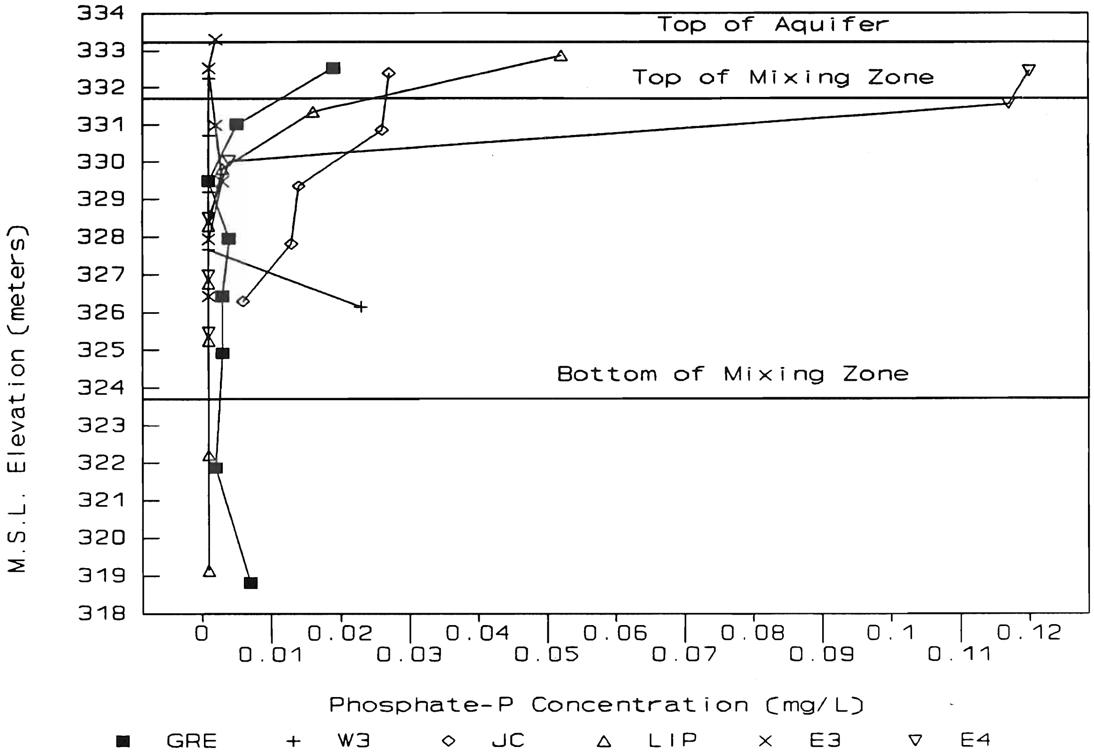

Figure 22e: Graph of average phosphate concentrations vs. elevation at the Jordan Acres downgradient wells.

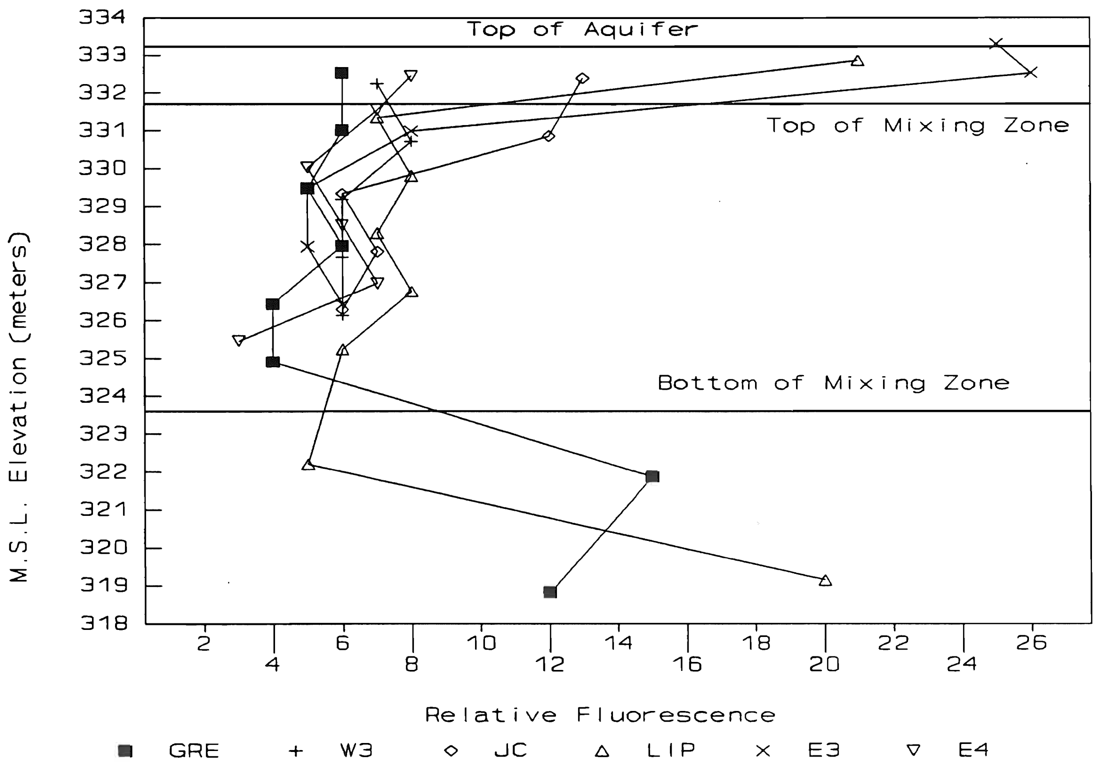

Figure 22f: Graph of average relative fluorescence vs. elevation at the Jordan Acres downgradient wells.

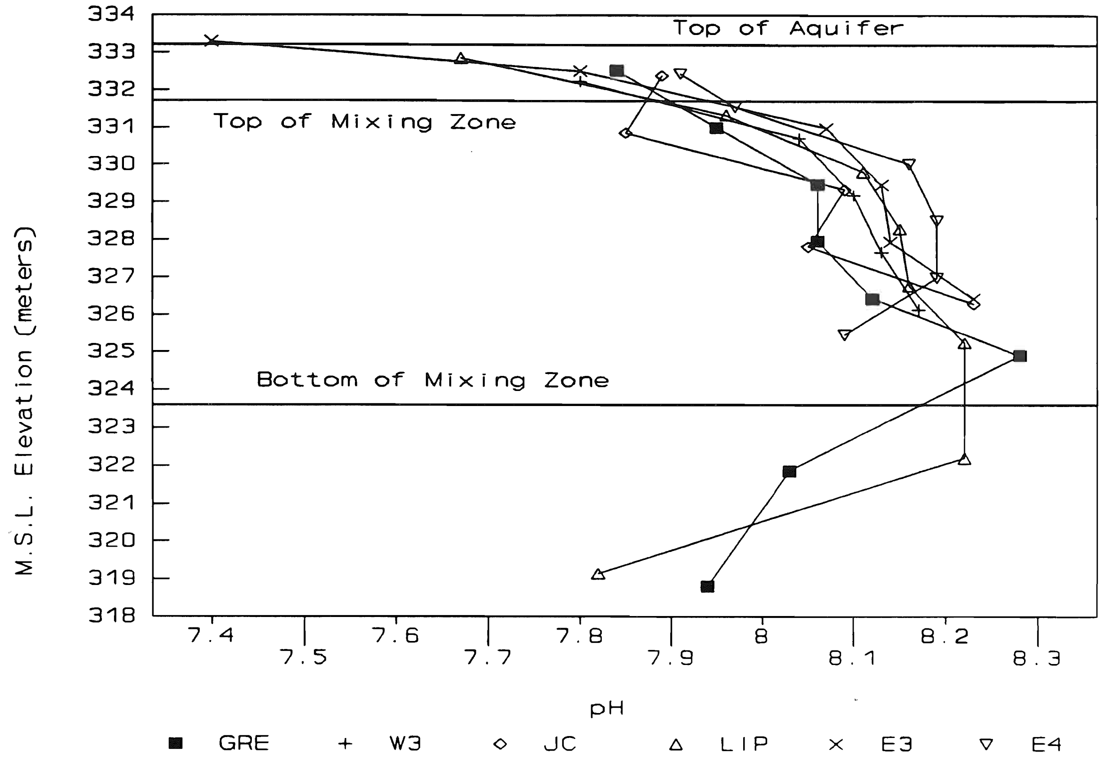

Figure 22g: Graph of average pH vs. elevation at the Jordan Acres downgradient wells.

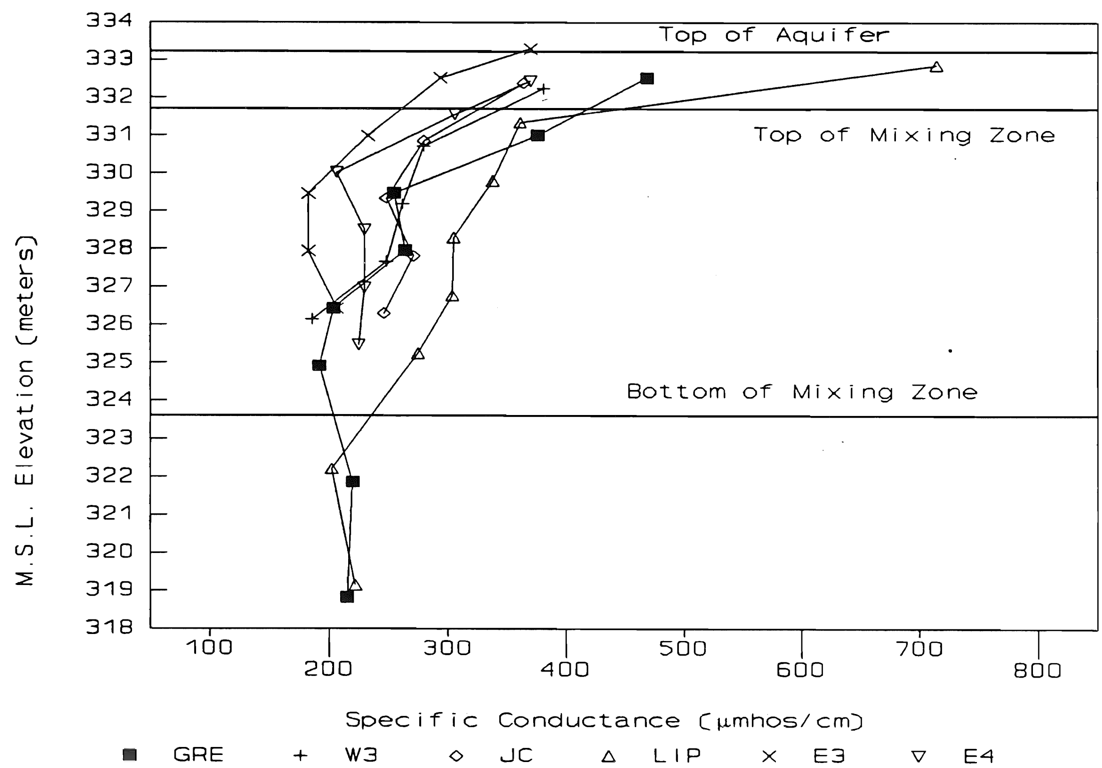

Figure 22h: Graph of average specific conductance vs. elevation at the Jordan Acres downgradient wells.

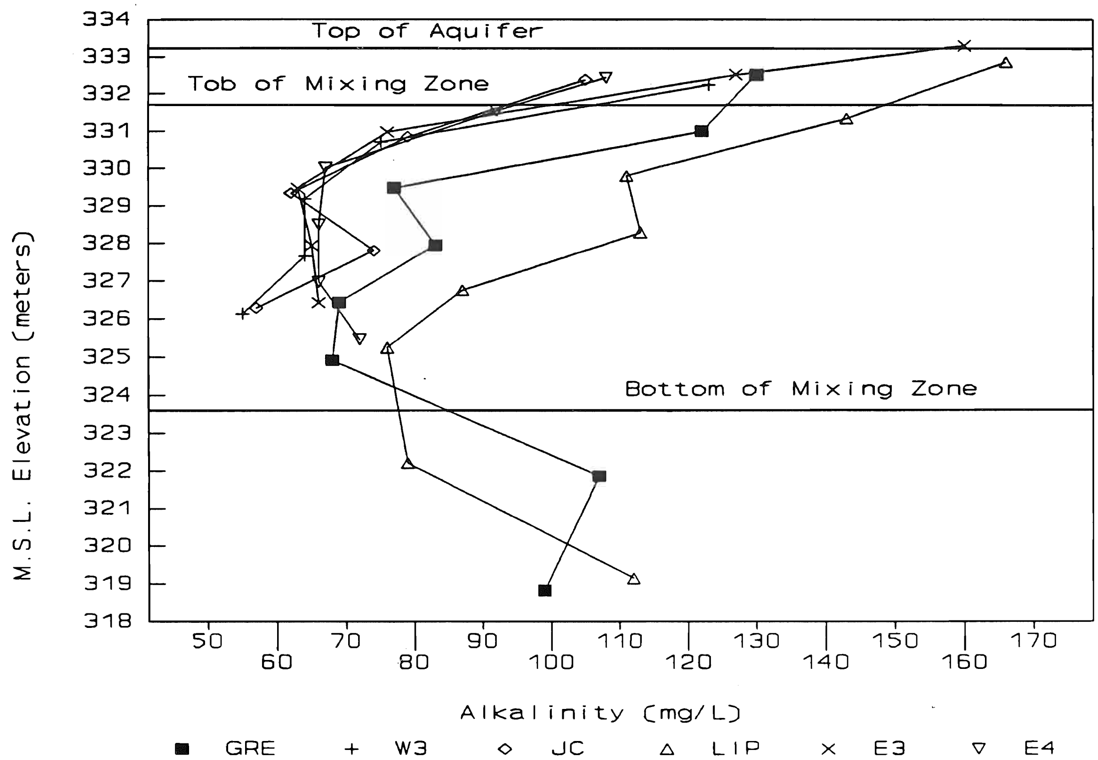

Figure 22i: Graph of average alkalinity vs. elevation at the Jordan Acres downgradient wells.

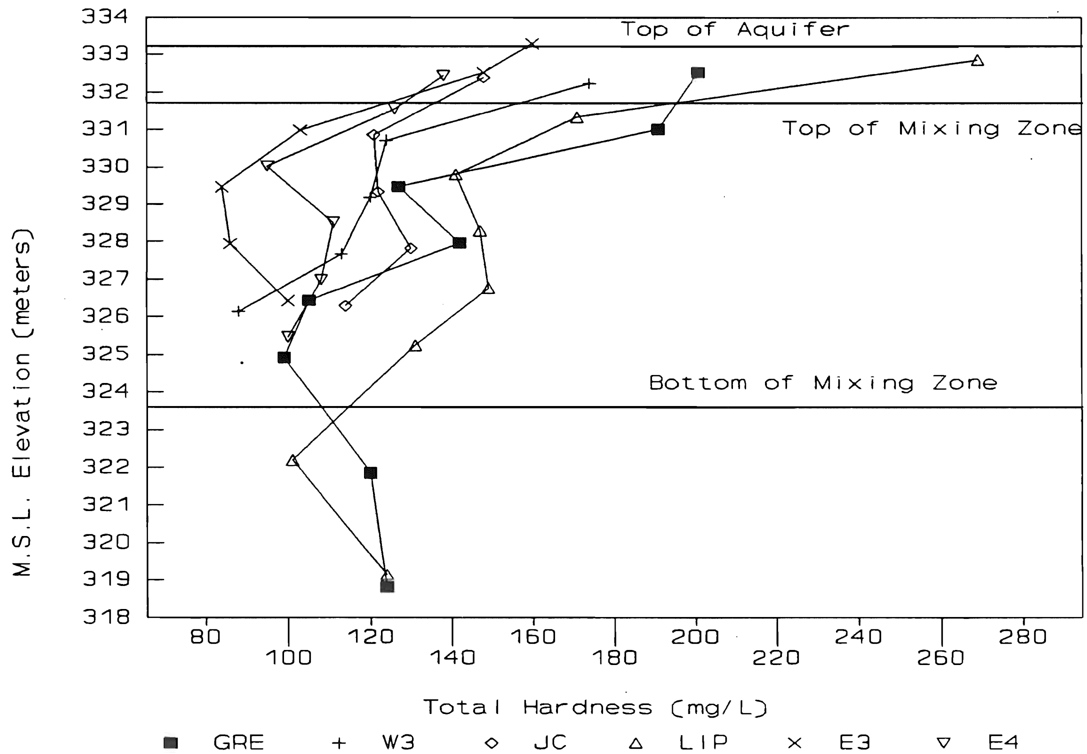

Figure 22j: Graph of average total hardness vs. elevation at the Jordan Acres downgradient wells.

3.1.2.2 Village Green

The location and identification of the multiport wells and the groundwater flow paths leading to the wells are presented in Figure 23. Groundwater chemistry data from the multiport wells in Village Green are presented in Appendix C.

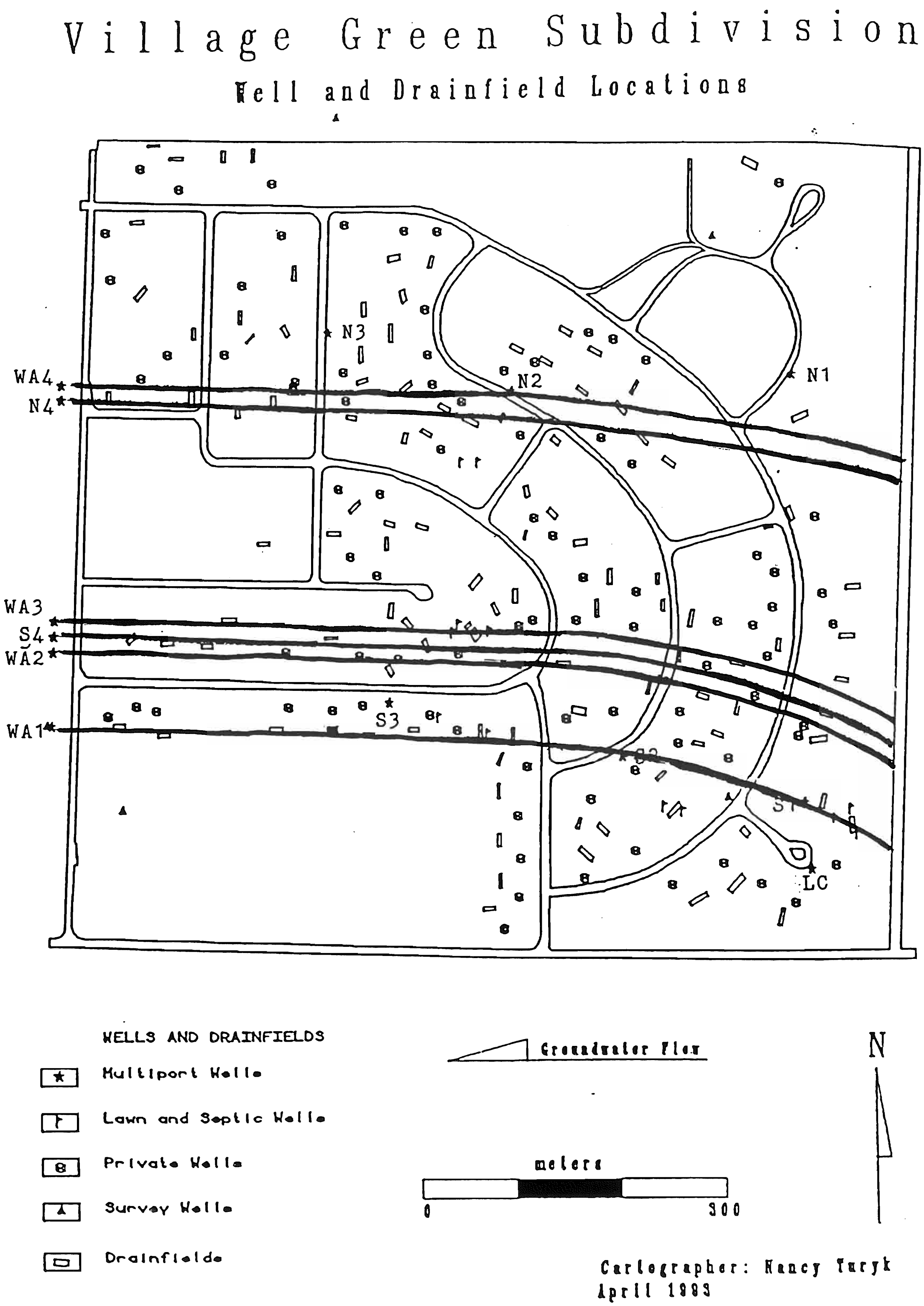

Figure 23: Location and identification of the multiport wells in Village Green. Inferred groundwater flow paths to the wells are indicated.

3.1.2.2.1 Upgradient Water Quality

The upgradient multiport wells in Village Green clearly show the effects of agricultural fertilizers (see the chemistry data in Table 14). The shallower ports have relatively low average nitrate-N concentrations (low in the Village Green multiport wells is considered to be less than 10 mg/L) and the concentrations increase (to over 30 mg/L) with depth down to 50 feet or so, where they again decrease (to around 20 mg/L). The chemistry in the shallow ports probably reflects the water quality of recharge from wooded, highway, and agricultural land uses; the middle ports are likely to be representative of uniform agricultural impacts; and the deepest ports show a mixture of water quality from wooded and agricultural areas.

The shallowest port at well nest LIL shows the diluting effects of the wooded area between the well and the highway. The average nitrate-N concentration (2.6 mg/L) is quite low and appears to be unaffected by agricultural practices. Likewise the average chloride and sodium concentrations (17.7 mg/L and 16.5 mg/L) are much lower than the shallower ports in the other upgradient wells. All of the chemical concentrations are markedly higher in the next port down (LIL-25) and are considered to represent a mixture of water from the three recharge areas (agricultural, highway, and woods). Continuing with depth (LC-30), nitrate-N concentrations have increased and sodium has begun to decrease, reflecting that more of the water was recharged in agricultural areas. The very high average nitrate-N concentrations (> 35 mg/L) and low sodium concentrations (< 8 mg/L) in ports LC-35 through LC-60 indicate that the water in the wells originated from agricultural areas. The deepest port at LC has lower nitrate-N and chloride concentrations, which may suggest lesser agricultural impacts.

Well N1 reflects similar trends as observed at LC. Nitrate-N concentrations are relatively low in the upper ports and they increase with depth, sodium concentrations are high in the shallow ports and decrease with depth. There is, however, no dramatic decrease in nitrate-N and chloride in the deepest port as was seen in well nest LIL.

There is no significant wooded area upgradient of well S1, as is the case at LC and N1, thus there is no buffer zone between the road impacted groundwater and the groundwater recharged from subdivision sources. High average concentrations of nitrate-N (24.0 mg/L), chloride (77.8 mg/L), sodium (54.4 mg/L), and phosphate-P (0.452 mg/L) in the shallowest port at S1 indicate an impact from at least one drainfield upgradient of the well. The next lower port also shows impacts from the drainfield(s). Because of this the ports are not useful for monitoring upgradient water quality. The chemistry in the middle ports (S1-30 and S1-35) is likely to be reflecting a mixture of water from road and agriculture sources. Only the deepest port (45 feet) appears to be impacted solely by agricultural sources (NO_3-N = 33.2 mg/L, Cl = 50.1 mg/L, Na = 6.5 mg/L).

Well Location	Well Port	# of Samples	Monitoring Period	NO_3-N	Cl	Na	PO_4-P
LC	22	3	Aug. '89 - May '90	2.6	17.7	16.5	0.007
	25	3	Aug. '89 - May '90	7.8	46.3	21.8	0.011
	30	3	Aug. '89 - May '90	11.6	42.3	10.2	0.002
	35	3	Aug. '89 - May '90	23.7	35.3	4.4	<0.002
	40	3	Aug. '89 - May '90	29.7	40.3	4.7	<0.002
	45	3	Aug. '89 - May '90	36.7	45.7	5.8	<0.002
	50	3	Aug. '89 - May '90	38.5	48.7	7.9	0.002
	55	3	Aug. '89 - May '90	35.5	46.3	7.8	0.003
	60	3	Aug. '89 - May '90	24.0	27.0	4.0	0.010
	70	3	Aug. '89 - May '90	12.2	18.7	3.6	0.015
Depth weighted average at LC				22.2	36.8	8.7	0.005
N1	25	18	Sep. '87 - June '90	8.1	35.7	22.9	<0.002
	30	18	Sep. '87 - June '90	16.4	57.4	22.3	0.002
	35	19	Sep. '87 - June '90	23.7	34.1	9.8	0.003
	40	17	Sep. '87 - June '90	25.6	40.1	7.9	0.002
	50	15	Sep. '87 - June '90	30.4	50.5	5.3	<0.002
	60	15	Sep. '87 - June '90	25.6	37.2	4.4	<0.002
	70	15	Sep. '87 - June '90	21.9	30.6	3.9	0.022
Depth weighted average at N1				21.7	40.8	10.9	0.005
S1	22	15	Sep. '87 - Mar. '90	24.0	77.8	54.4	0.452
	25	17	Sep. '87 - Mar. '90	13.0	66.6	46.8	0.004
	30	15	Sep. '87 - Mar. '90	19.0	46.3	19.5	0.005
	35	15	Sep. '87 - Mar. '90	21.9	42.4	16.1	<0.002
	40	15	Sep. '87 - Mar. '90	26.0	40.9	10.7	<0.002
	45	15	Sep. '87 - Mar. '90	33.2	50.1	6.5	0.002
Depth weighted average at S1				22.9	54.0	25.7	0.078
Average concentrations for all upgradient wells				22.6	39.7	10.3	0.003

Table 14: Village Green upgradient multiport well groundwater chemistry data (in mg/L).

The variability in the Village Green upgradient groundwater quality is demonstrated by the nitrate-N data from N1, which are presented in Table 15 and Figures 24a & b. In contrast to Jordan Acres, the deeper ports at the Village Green upgradient wells have more consistent chemistry as compared to the shallower ports. The extensive irrigated agriculture upgradient of Village Green acts as a consistent recharge area for the deeper ports, whereas the variability in the shallower ports is due to seasonal changes in the mixing of the recharge water from the cropland, highway, and woodland.

	N1-25	N1-30	N1-35	N1-40	N1-50	N1-60	N1-70
09/29/87	2.5	16.5	17.2	18.0	31.0	30.8	19.8
11/10/87	4.0	14.4	20.0	21.0	32.5	30.0	21.0
02/01/88	9.8	14.0	16.5	25.5	32.0	28.0	21.2
03/22/88	11.5	17.8	17.5	28.8	32.2	27.5	22.5
06/01/88	10.5	13.8	15.0	25.5	30.0	27.0	20.5
08/23/88	3.5	13.5	18.5	23.0	29.8	27.0	29.0
11/10/88	6.5	14.0	30.0	26.0	28.0	26.8	19.5
02/01/89	10.5	14.0	27.0	28.0	31.5	26.2	21.0
05/03/89	13.5	18.2	33.0	27.8	30.0	23.2	21.0
06/26/89	-	20.5	30.0	30.8	34.0	24.5	23.0
08/31/89	7.8	21.0	31.5	26.0	29.2	23.5	20.8
01/03/90	6.5	24.8	24.2	24.2	26.8	22.5	22.0
03/22/90	11.8	21.2	16.8	26.5	27.0	22.5	22.8
06/07/90	9.0	18.8	21.0	27.5	28.8	23.0	22.8
Average	8.26	17.32	22.73	25.61	30.20	25.89	21.92

Table 15: Groundwater nitrate-N concentrations (in mg/L) for all sampling dates from all ports at well N1 in Village Green.

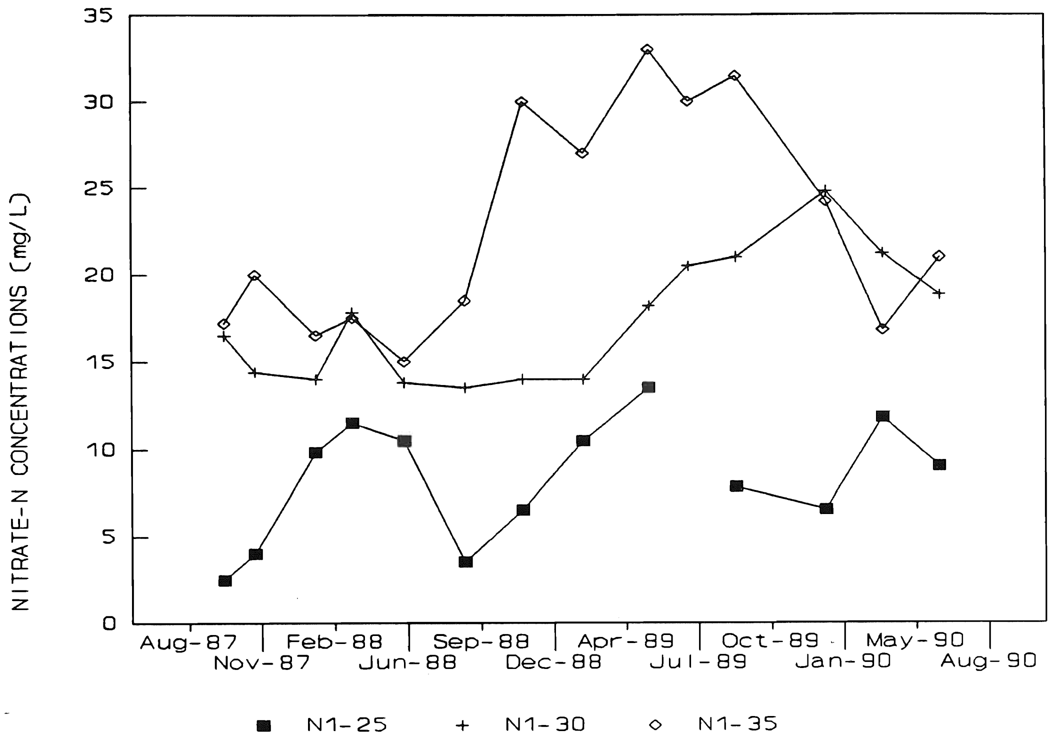

Figure 24a: Plot of nitrate-N concentrations vs. time for well N1 in Village Green.

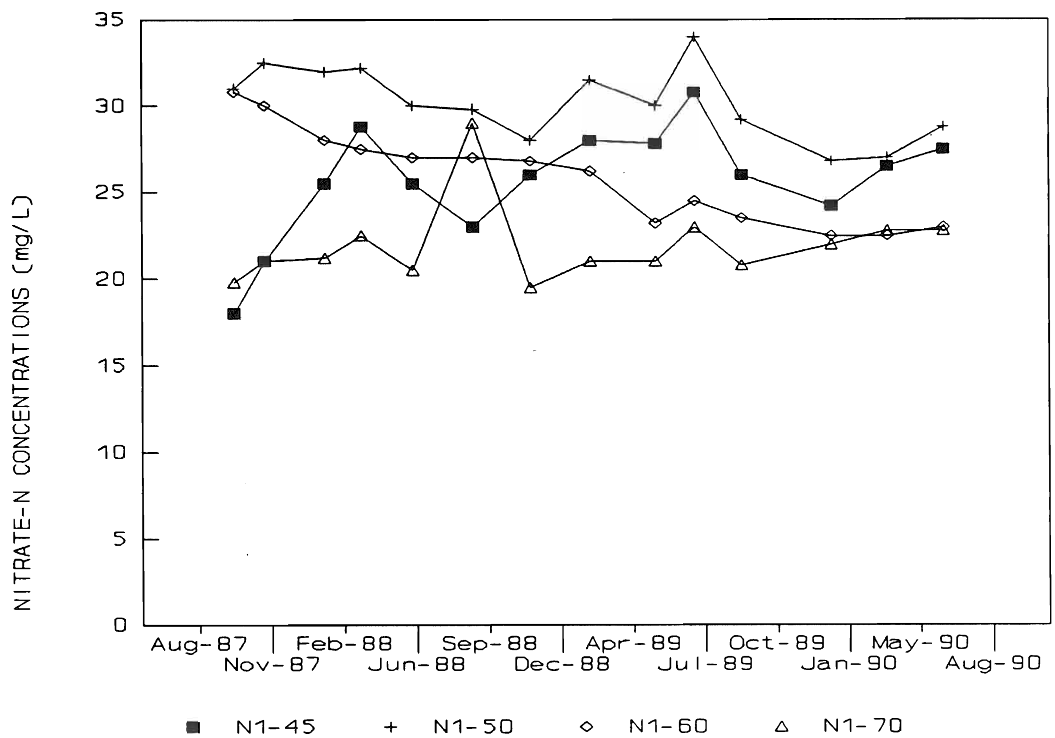

Figure 24b: Plot of nitrate-N concentrations vs. time for well N1 in Village Green.

3.1.2.2.2 Groundwater Quality Through The Subdivision

As in Jordan Acres, several of the multiport wells in Village Green were installed so as to monitor the same general flow path of groundwater as it moves through the subdivision. Average inorganic chemical concentrations obtained from multiport wells installed along the North transect are presented in Table 16; data from the wells included in the South transect are presented in Table 17. The locations of the wells are shown on Figure 23 (p. 90).

There are three primary groundwater recharge areas that are considered to be affecting the groundwater chemistry in the multiport wells installed in the Village Green subdivision. They are agricultural land, the permeable area along a four-lane divided highway, and the subdivision itself. Groundwater from agricultural sources is characterized by high (>20 mg/L) nitrate-N concentrations, high (>35 mg/L) chloride concentrations, and low (<10 mg/L) sodium concentrations. Water recharged from near the highway will tend to have relatively low concentrations of nitrate-N and high concentrations of sodium and chloride; however, the sodium and chloride will be discharged primarily in the spring (after the ground thaws and the snow-melt with dissolved road salt can infiltrate to the aquifer). Thus concentrations of these ions will tend to exhibit seasonal fluctuations. As discussed previously, the chemistry from subdivision sources will vary depending on the specific land use in the contributing area. These three source areas for groundwater recharge help to track the groundwater movement through the subdivision.

The profile of groundwater quality at the upgradient end of the subdivision

Well Location	Well Port	Monitoring Period	# of Samples	NO_3-N	Cl	Na	PO_4-P
N1	22	Aug. '88 - Nov. '88	2	9.7	12.5	2.2	0.009
	25	Sep. '87 - June '90	18	8.1	35.7	22.9	<0.002
	30	Sep. '87 - June '90	18	16.4	57.4	22.3	<0.002
	35	Sep. '87 - June '90	19	23.7	34.1	9.8	0.003
	40	Sep. '87 - June '90	18	25.6	40.1	7.9	0.002
	50	Sep. '87 - June '90	15	30.4	50.5	5.3	<0.002
	60	Sep. '87 - June '90	15	25.6	37.2	4.4	<0.002
	65	Sep. '87 - June '90	15	21.9	30.6	3.9	0.022
			Depth weighted average	20.2	37.3	9.8	0.004
N2	22	Sep. '87 - Mar. '90	11	8.6	32.0	9.8	0.010
	25	Sep. '87 - Mar. '90	12	7.4	20.8	6.3	0.006
	30	Sep. '87 - Mar. '90	11	11.9	36.7	16.9	0.016
	35	Sep. '87 - Mar. '90	12	19.2	55.8	22.2	0.004
	40	Sep. '87 - Mar. '90	12	20.8	52.7	17.1	0.002
	45	Sep. '87 - Mar. '90	13	22.1	49.8	12.3	0.004
			Depth weighted average	15.0	41.3	14.1	0.007
N3	22	Sep. '87 - Aug. '89	12	11.3	18.4	5.0	0.002
	25	Sep. '87 - Aug. '89	11	8.4	21.6	9.4	0.011
	30	Sep. '87 - Mar. '89	12	9.2	15.9	7.6	0.013
	35	Sep. '87 - Aug. '89	11	13.2	30.0	14.7	0.004
	40	Sep. '87 - Mar. '89	12	14.8	44.3	13.8	0.002
	45	Sep. '87 - Mar. '89	11	10.5	55.2	31.1	<0.002
			Depth weighted average	11.2	30.9	13.6	0.005
N4	22	Sep. '87 - Feb. '91	16	10.5	55.2	31.1	<0.002
	25	Sep. '87 - Feb. '91	16	12.0	32.8	23.3	0.010
	30	Sep. '87 - Feb. '91	16	14.9	26.8	18.4	0.062
	35	Sep. '87 - Feb. '91	16	12.3	23.4	14.4	0.019
	40	Sep. '87 - Feb. '91	16	12.3	32.9	11.5	<0.002
	50	Sep. '87 - Feb. '91	16	20.4	32.4	8.4	<0.002
	60	Sep. '87 - Feb. '91	16	19.8	26.6	5.4	0.008
			Depth weighted average	14.6	32.9	16.1	0.013

Table 16: Village Green North transect groundwater chemistry data (in mg/L).

Well Location	Well Port	Monitoring Period	# of Samples	NO_3-N	Cl	Na	PO_4-P
S1	22	Sep. '87 - Mar. '90	15	24.0	77.8	54.4	0.452
	25	Sep. '87 - Mar. '90	17	13.0	66.6	46.8	0.004
	30	Sep. '87 - Mar. '90	15	19.0	46.3	19.5	0.005
	35	Sep. '87 - Mar. '90	15	21.9	42.4	16.1	<0.002
	40	Sep. '87 - Mar. '90	15	26.0	40.9	10.7	<0.002
	45	Sep. '87 - Mar. '90	15	33.2	50.1	6.5	0.002
Depth weighted average				22.9	54.0	25.7	0.077
S2	22	Sep. '87 - Mar. '90	13	16.4	21.5	13.4	0.010
	25	Sep. '87 - Mar. '90	13	11.5	17.1	10.6	<0.002
	30	Sep. '87 - Mar. '90	12	12.4	59.0	29.5	<0.002
	35	Sep. '87 - Mar. '90	12	21.8	57.6	24.6	<0.002
	40	Sep. '87 - Mar. '90	12	24.1	58.1	15.0	0.004
	45	Sep. '87 - Mar. '90	12	27.5	49.2	6.8	0.002
Depth weighted average				19.0	43.8	16.7	0.002
S3	22	Sep. '87 - Mar. '90	12	5.3	37.8	14.7	<0.002
	25	Sep. '87 - Mar. '90	12	5.1	37.1	14.7	0.038
	30	Sep. '87 - Mar. '90	12	8.8	37.5	16.1	0.003
	35	Sep. '87 - Mar. '90	12	21.7	54.3	21.8	<0.002
	40	Sep. '87 - Mar. '90	11	23.1	44.9	8.8	0.004
	45	Sep. '87 - Mar. '90	12	24.0	28.0	4.5	0.005
Depth weighted average				14.7	39.9	13.4	0.008
S4	22	Sep. '87 - Feb. '91	16	16.8	48.1	36.1	0.032
	25	Sep. '87 - Feb. '91	16	17.8	38.8	30.3	0.090
	30	Sep. '87 - Feb. '91	16	22.5	44.0	40.5	0.027
	35	Sep. '87 - Feb. '91	16	18.3	45.5	30.2	<0.002
	40	Sep. '87 - Feb. '91	16	17.6	53.6	25.5	<0.002
	45	Sep. '87 - Feb. '91	16	19.9	51.4	22.7	0.003
Depth weighted average				18.8	46.9	30.9	0.025

Table 17: Village Green South transect groundwater chemistry data (in mg/L).

was discussed above. In general, the shallowest ports are impacted by woods and/or the highway, the middle ports are impacted by the highway and agriculture, the deepest ports are impacted only by agricultural sources.

At N2, the relatively low average concentrations of nitrate-N and sodium (< 10 mg/L each) in the shallowest two ports suggest that the water originated from within the subdivision. The flow path leading to the wells indicates the chemistry is due to recharge from both lawns and drainfields. Average nitrate-N concentrations increase with depth (to over 20 mg/L), reflecting the greater influence from agricultural areas. Sodium concentrations increase (due to highway impacts), then decrease with depth; however, they do not get as low in the deepest port (N2-45) as in the deeper ports at the upgradient wells; thus this port appears to be monitoring groundwater in the highway/agricultural mixing zone.

Well N3 shows a similar trend as observed at N2. The upper three ports appear to be affected by subdivision recharge, as evidenced by the relatively low average concentrations of nitrate-N (<12 mg/L), chloride (<22 mg/L), and sodium (<10 mg/L). Again the flow path indicates both lawn and septic impacts. It is difficult to distinguish the combination of factors influencing the chemistry in the deepest three ports. The high average sodium concentration (31.1 mg/L) in the deepest port suggests salt impact from the highway. This in turn suggests that the water in ports N3-35 and N3-40 originated from within the subdivision. Because the area immediately downgradient of the highway in the north part of Village Green is mostly undeveloped, the recharge would be expected to have low concentrations of inorganic chemicals. Contrary to expectations, these two ports have relatively high average concentrations of nitrate-N, chloride, and sodium (see Table 16). This may indicate that these ports are showing the effects of the drainfields located south and west of well N1.

At well N4, the average nitrate-N concentrations were between 10 and 15 mg/L in ports N4-22 through N4-40; the concentrations increase to around 20 mg/L in the deepest two ports. Average sodium concentrations exhibit a gradual decrease with depth over the entire profile (from 31.1 mg/L to 5.4 mg/L). Except for a high average concentration from the shallowest port (55.2 mg/L), chloride concentrations remain fairly constant (between 23 to 33 mg/L) throughout the profile. The nitrate-N and sodium concentrations in N4-50 and N4-60 are representative of agricultural water and thus the ports are considered to be unaffected by the subdivision. This would suggest that the water immediately above this zone should have a high concentration of sodium due to highway impacts; however, this does not appear to be the case. It is possible that the heart of the plume of highway-impacted water passes between N4-40 and N4-50, or perhaps misses this well nest completely due to localized recharge of highway runoff. The shallower ports are considered to be impacted only by the subdivision, but it is difficult to tell where the highway and agricultural impacts start. This is because the chemistry due to a mixture of agricultural and highway recharge is similar to the chemistry due to the combination of subdivision sources (i.e., drainfields, lawns, and roads).

The chemistry data from well S2 is similar to that observed at N2 except that the shallowest two ports show a greater impact by septic systems (as indicated by average nitrate-N concentrations of 16.4 and 11.5 mg/L). The middle ports are likely to be influenced by a combination of agricultural and highway water, while the deepest port is monitoring only agricultural recharge.

There appears to be a distinct change in groundwater quality at well S3 between the -30 and -35 ports: above is mostly subdivision water, below is mostly agricultural water. The relatively low average nitrate-N concentration (8.8 mg/L) in S3-30 indicates little agricultural impact, whereas the high average nitrate-N concentration (21.7 mg/L) in S3-35 suggests substantial influence by agricultural recharge. The chemistry in the -35 port could also be showing the impact from drainfields at the upgradient end of the subdivision (perhaps those impacting the shallow ports at S1). The deepest two ports appear to be monitoring agricultural water (as evidenced by the low sodium concentrations), which is unusual because only the deepest port at S2 was considered to be impacted solely by agricultural water.

The concentrations of nitrate-N, chloride, and sodium were relatively constant for all the ports at well S4; average nitrate-N concentrations ranged from 16.8 mg/L to 22.5 mg/L, chloride from 38.8 mg/L to 54.3 mg/L, and sodium from 22.7 mg/L to 40.5 mg/L. Significant concentrations of phosphate in the upper three ports suggest subdivision recharge; however, it is unclear if the chemistry in the water from the lower ports is due to subdivision sources, road salts, lawn fertilizers, or a combination of the three.

Based on the groundwater flow patterns in the Village Green subdivision, the wells included in the south transect appear to be the most representative of how the groundwater quality changes as it passes through the subdivision. As with the east transect in Jordan Acres (see section 3.1.2.1.2) vertical profiles showing average nitrate-N concentrations and chloride to sodium ratios help demonstrate the chemical changes in the aquifer across the subdivision. The profiles are presented in Figures 25a & b.

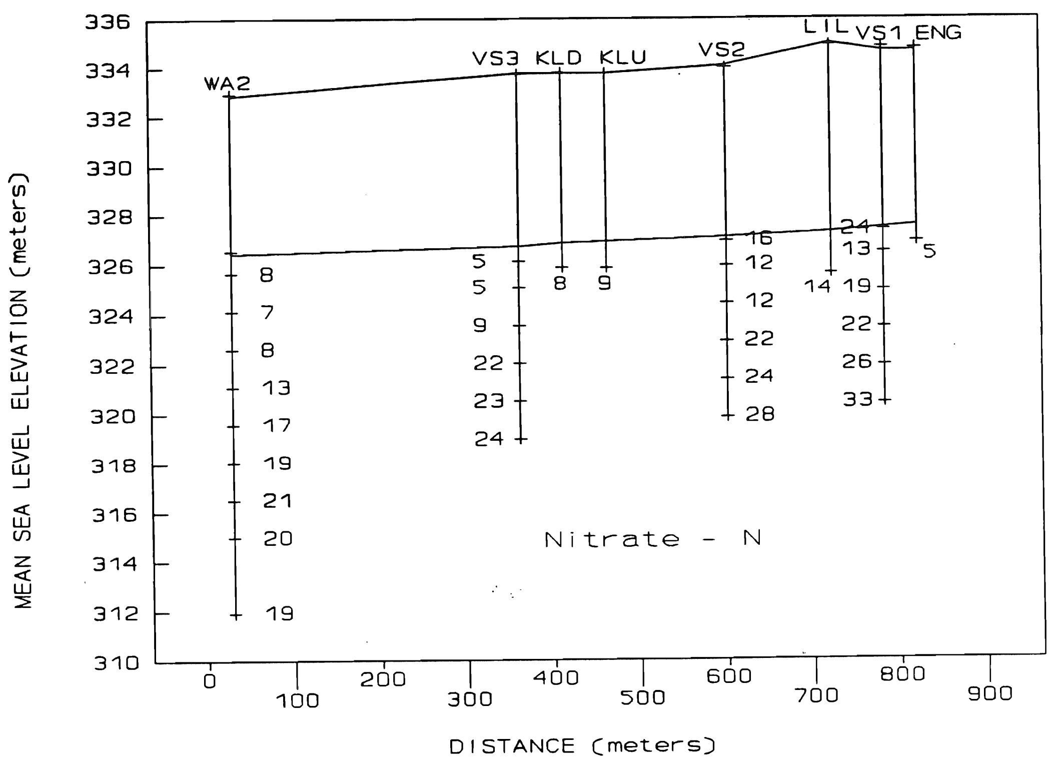

Figure 25a: Vertical profile showing the average nitrate-N concentration at each port for all the wells included in the South transect in Village Green.

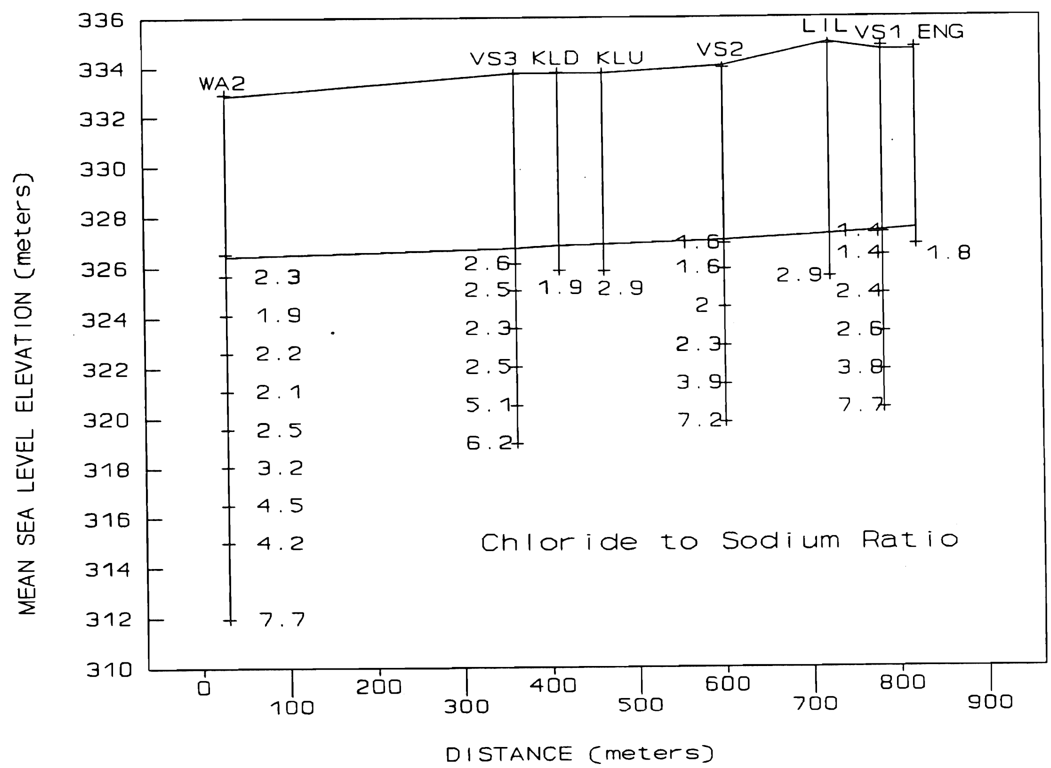

Figure 25b: Vertical profile showing the average chloride to sodium ratio at each port for all of the wells included in the South transect in Village Green.

3.1.2.2.3 Downgradient Water Quality

Six wells were installed perpendicular to the groundwater flow direction at the downgradient end of Village Green. These wells provide a view of the groundwater chemistry with depth as it discharges from beneath the subdivision. The average chemistry results for the ports at these wells are presented in Table 18.

Tracing the groundwater flow path leading to the wells (see Figure 23, p. 90) helps to explain the origin of the groundwater chemistry monitored in the ports. For the groundwater flow path through the southern half of the subdivision, the lots on the western (downgradient) half of the subdivision are laid out parallel with groundwater flow. The lots on the eastern (upgradient) half of the subdivision are laid out somewhat perpendicular to groundwater flow. All of the development in the north half of the subdivision is laid out generally perpendicular to groundwater flow. The positioning of the downgradient multiport wells in the southern half of the subdivision is such that they are likely to be impacted by either many drainfields (e.g., WA1 and S4) or very few (e.g., WA2). The downgradient multiport wells in the north half of the subdivision are influenced by a more even distribution of drainfields.

The two downgradient wells in the northern half of the subdivision (N4 and WA4) are only fifteen meters away from each other; however, the shallower ports exhibit somewhat different chemical characteristics. N4 has slightly higher nitrate-N concentrations and lower chloride and sodium concentrations (see Table 18). At WA4, it is likely that the higher chloride and sodium concentrations, and the greater depths at which they are present, are due to salt applied to the road running east-west

Well Location	Well Port	Monitoring Location	# of Samples	NO_3-N	Cl	Na	PO_4-P
N4	22	Sep. '87 - Feb. '91	16	10.5	55.2	31.1	<0.002
	25	Sep. '87 - Feb. '91	16	12.0	32.8	23.3	0.010
	30	Sep. '87 - Feb. '91	16	14.9	26.8	18.4	0.062
	35	Sep. '87 - Feb. '91	16	12.3	23.4	14.4	0.019
	40	Sep. '87 - Feb. '91	16	12.3	32.9	11.5	<0.002
	50	Sep. '87 - Feb. '91	16	20.4	32.4	8.4	<0.002
	60	Sep. '87 - Feb. '91	16	19.8	26.6	5.4	0.008
	70	Sep. '87 - Feb. '91	16	19.2	24.0	3.8	0.019
Depth weighted average				15.2	31.8	14.5	0.014
WA4	22	Aug. '89 - Feb '91	6	6.4	67.3	49.2	<0.002
	25	Aug. '89 - Feb '91	7	13.0	64.3	35.5	<0.002
	30	Aug. '89 - Feb '91	7	9.5	55.1	27.4	0.010
	35	Aug. '89 - Feb '91	7	11.3	41.9	20.4	<0.002
	40	Aug. '89 - Feb '91	7	11.2	27.9	22.2	<0.002
	45	Aug. '89 - Feb '91	7	14.5	37.0	19.9	<0.002
	50	Aug. '89 - Feb '91	7	16.5	37.9	13.2	<0.002
	55	Aug. '89 - Feb '91	7	17.7	30.3	9.1	<0.002
	60	Aug. '89 - Feb '91	7	18.1	25.3	5.0	0.003
	70	Aug. '89 - Feb '91	7	16.1	22.0	2.8	0.011
Depth weighted average				13.4	40.9	20.5	0.002
WA3	22	Aug. '89 - Feb '91	6	5.7	63.7	35.1	0.144
	25	Aug. '89 - Feb '91	6	12.3	67.7	22.3	0.195
	30	Aug. '89 - Feb '91	6	16.4	34.0	22.1	0.077
	35	Aug. '89 - Feb '91	6	16.2	38.3	21.7	0.033
	40	Aug. '89 - Feb '91	6	17.1	47.2	19.8	0.006
	45	Aug. '89 - Feb '91	6	19.5	50.3	15.5	<0.002
	50	Aug. '89 - Feb '91	5	21.2	46.0	9.3	<0.002
	60	Aug. '89 - Feb '91	6	19.8	48.8	3.4	0.008
	70	Aug. '89 - Feb '91	6	22.8	31.7	4.6	0.016
Depth weighted average				12.7	50.9	25.3	0.112

Table 18: Village Green downgradient multiport well groundwater chemistry data (in mg/L).

Well Location	Well Port	Monitoring Location	# of Samples	NO_3-N	Cl	Na	PO_4-P
S4	22	Sep. '87 - Feb. '91	16	16.8	47.1	36.1	0.032
	25	Sep. '87 - Feb. '91	16	17.8	38.8	30.3	0.090
	30	Sep. '87 - Feb. '91	16	22.5	44.0	40.5	0.027
	35	Sep. '87 - Feb. '91	16	18.3	45.5	30.2	<0.002
	40	Sep. '87 - Feb. '91	16	17.6	53.6	25.5	<0.002
	45	Sep. '87 - Feb. '91	16	19.9	51.4	22.7	0.003
Depth weighted average				18.8	46.7	30.9	0.025
WA2	25	Aug. '89 - Feb '91	6	8.3	54.0	23.4	<0.001
	30	Aug. '89 - Feb '91	6	6.5	22.0	11.3	<0.002
	35	Aug. '89 - Feb '91	6	8.4	28.7	13.0	<0.002
	40	Aug. '89 - Feb '91	6	13.3	35.7	17.3	<0.002
	45	Aug. '89 - Feb '91	6	16.5	59.7	24.3	<0.002
	50	Aug. '89 - Feb '91	6	19.0	62.5	19.6	<0.002
	55	Aug. '89 - Feb '91	6	21.1	50.8	11.3	<0.002
	60	Aug. '89 - Feb '91	6	19.9	73.5	17.7	0.002
	70	Aug. '89 - Feb '91	6	18.7	30.2	3.9	0.015
Depth weighted average				14.6	46.3	15.8	<0.002
WA1	22	Aug. '89 - Feb '91	7	13.8	53.0	33.1	<0.002
	25	Aug. '89 - Feb '91	7	17.0	25.9	19.6	<0.002
	30	Aug. '89 - Feb '91	7	13.8	24.6	18.7	0.002
	35	Aug. '89 - Feb '91	7	11.5	26.5	18.4	<0.002
	40	Aug. '89 - Feb '91	7	11.6	33.4	17.1	<0.002
	45	Aug. '89 - Feb '91	6	14.8	57.8	18.6	<0.002
	50	Aug. '89 - Feb '91	7	24.2	48.3	6.2	0.002
	60	Aug. '89 - Feb '91	7	21.8	27.6	3.2	0.011
	70	Aug. '89 - Feb '91	7	17.5	22.1	2.7	0.017
Depth weighted average				16.2	35.5	15.3	0.002

Table 18 (continued): Village Green downgradient multiport well groundwater chemistry data (in mg/L).

directly upgradient of the well. The runoff water from the road will also tend to have lower nitrate-N concentrations. The middle ports at both well locations have similar concentrations of nitrate-N, chloride, and sodium, and in the deepest ports the chemistry is almost the same. The ports that appear to define the upper zone of agricultural water (as evidenced by average sodium concentrations of under 10 mg/L) are N4-50 and WA4-55. Although it is difficult to tell the depth at which subdivision recharge begins to affect the groundwater quality, it appears to be less than nine meters into the aquifer.

Four wells (WA1, WA2, S4, and WA3) are monitoring the portion of the subdivision modeled for the nitrogen and water budgets. These wells are considered to be the most representative of the groundwater originating from a relatively well defined portion of a subdivision. WA1 appears to have a string of drainfields directly upgradient. WA2 appears to have no drainfields directly upgradient of it for about half the length of the subdivision, then the drainfield distribution becomes fairly uniform. S4 has many drainfields upgradient in the near half of the subdivision, then a more uniform drainfield distribution. WA3 appears to have only a few drainfields upgradient before reaching the area of relatively uniform drainfield distribution.

In order to better assess the variability in groundwater quality over a relatively short distance perpendicular to groundwater flow, multiport wells WA2 and WA3 were placed 50 feet on either side of S4. The shallower three ports at these wells show the effects of varying upgradient drainfield densities. WA2 has some of the lowest nitrate-N concentrations (<9 mg/L) of any of the sampling locations in the entire subdivision. WA3 has a low average nitrate-N concentration in the shallowest port (5.7 mg/L), but the concentrations quickly increase with depth. The shallowest ports at S4 have uniformly high average nitrate-N concentrations (around 17 mg/L). WA2 also has somewhat lower concentrations of chloride and sodium in its shallower ports than do the other two wells. WA2 is the well that is most likely to be impacted by recharge occurring in the vicinity of the east-west road upgradient of the three wells. It also appears to have few drainfields in the flow path. Because the shallower ports at WA2 have lower chloride and sodium concentrations than the other two wells, the higher concentrations of these species in the ports at WA3 and S4 is likely due to the impacts of drainfields.

The middle ports of the three wells (the deepest ports at S4) exhibit similar chemistry; average nitrate-N concentrations are between 15 and 20 mg/L, average chloride concentrations are between 35 and 55 mg/L, and average sodium concentrations are around 20 mg/L. The concentrations are slightly lower in the ports at WA2 and slightly higher in the ports at S4, but overall the chemistry is comparable. These ports are likely to be monitoring the relatively uniform distribution of subdivision impact sources (e.g., drainfields). There is also likely to be some mixing with upgradient water originating from agricultural sources.

Although it appears that all of the ports at S4 are impacted by subdivision recharge, the deepest three ports at WA3 and the deepest port at WA2 appear to show the impacts of agricultural recharge only (as evidenced by average sodium concentrations below 10 mg/L and average nitrate-N concentrations around 20 mg/L).

Assuming that the ports immediately above this zone reflect a mixture of agricultural and highway recharge, the maximum depth of subdivision impact is estimated to be the upper six to nine meters of the aquifer.

WA1 is set off by itself and seems to be impacted by a long row of drainfields immediately upgradient, then by a more scattered drainfield distribution nearer the upgradient end of the subdivision. The shallower ports at this well show the effects of the drainfields. The average nitrate-N concentrations in the upper four ports are fairly high (14 to 17 mg/L). The high average chloride and sodium concentrations (53.0 mg/L and 33.1 mg/L) at WA1-22 can be attributed to salt applied to the adjacent road and nearby intersection; the average concentrations in the -25, -30, and -35 ports are similar to each other (chloride around 25 mg/L and sodium around 19 mg/L) and may be reflecting uniform subdivision impacts (e.g., drainfields). As the depth increases, average nitrate-N concentrations decrease slightly (to around 12 mg/L) and average chloride concentrations increase, perhaps indicating a change in the chemistry due to the change in upgradient drainfield densities. The low sodium concentrations in the deepest three ports define the upper boundary of the agricultural-impacted groundwater. As before, assuming the next port up is monitoring ag/highway impacts, the maximum depth of impact by subdivision sources appears to be about six meters. The apparent bend in the groundwater flow pattern observed at the upgradient end of the subdivision (see Figure 6, p.18) may effectively lessen the length of subdivision impacting the flow path (i.e., the highway impacts are at a lesser distance upgradient of WA-1 than for the other downgradient wells).

Profiles showing the average nitrate-N concentrations and chloride to sodium ratios for the downgradient multiport wells are presented in Figure 26a & b.

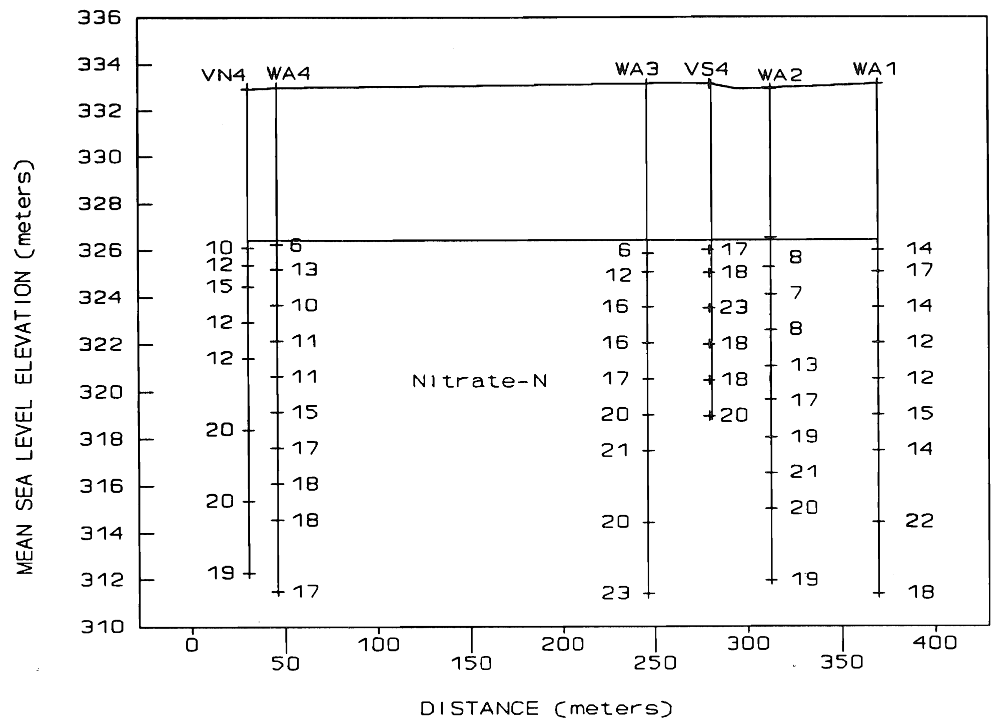

Figure 26a: Vertical profile showing the average nitrate-N concentration at each port for all of the wells included in the downgradient cross section in Village Green.

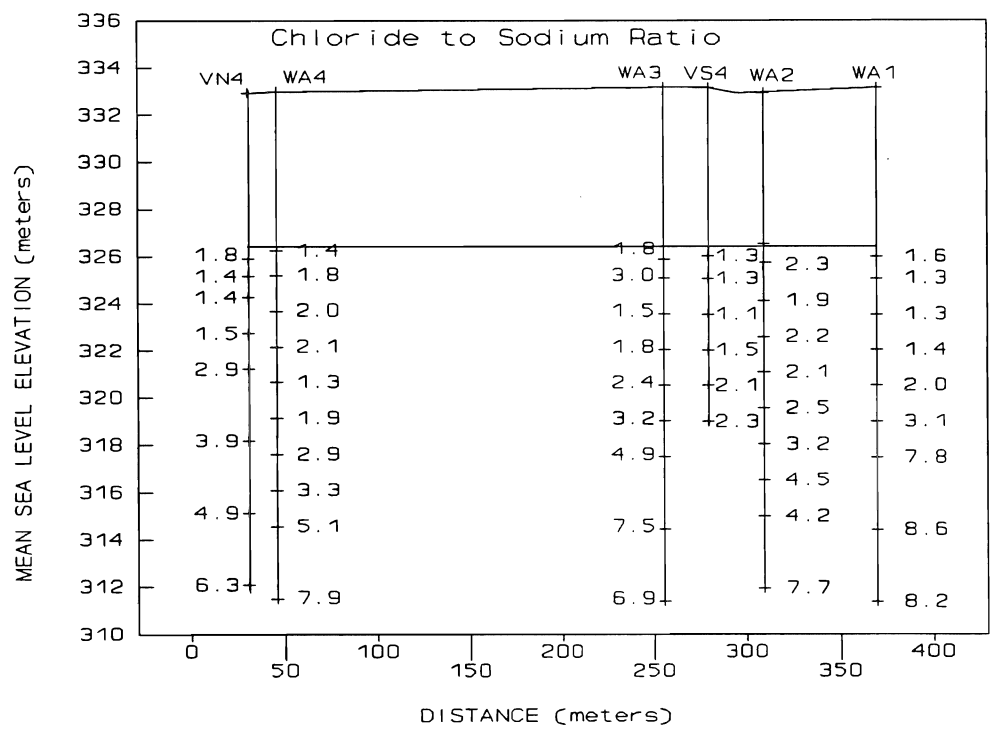

Figure 26b: Vertical Profile showing the average chloride to sodium ratio at each port for wells included in the downgradient cross section in Village Green.

The variability in the nitrate-N concentrations at the downgradient end of Village Green is demonstrated by data from well N4. Nitrate-N concentrations for all sampling dates are presented in Table 19; the data are presented graphically in Figures 27a & b.

The nitrate-N concentrations in the shallower ports at well N4 appear to show a trend of increasing concentrations over the duration of the monitoring period. The trend is not apparent at S4 or the other downgradient wells. The increase in concentrations may be due to the effects of lesser amounts of groundwater recharge (hence dilution water) during the early portion of the study period. A more likely explanation is that this well is reflecting the effects of increasing development in the northwest portion of the subdivision. As with the upgradient wells, the deeper ports have more consistent nitrate-N concentrations than do the shallower ports.

Sample Date	N4-22	N4-25	N4-30	N4-35	N4-40	N4-50	N4-60	N4-70
09/29/87	6.2	7.5	8.0	6.5	8.5	20.5	18.8	17.5
11/10/87	5.5	4.6	5.5	7.2	11.2	21.5	19.0	18.0
02/01/88	7.8	6.0	5.5	9.5	15.5	24.0	20.5	19.2
03/22/88	7.5	5.0	4.5	12.0	20.0	22.0	20.0	19.2
06/01/88	5.8	7.3	5.5	8.0	17.0	21.0	20.0	20.3
08/23/88	6.0	10.0	17.0	8.0	9.5	20.0	20.5	19.5
11/10/88	9.5	8.5	13.8	9.5	10.0	19.2	19.2	18.5
02/01/89	8.5	14.5	18.8	8.5	10.0	20.0	19.5	19.2
05/03/89	10.0	12.5	21.5	13.5	11.5	20.0	20.0	19.2
06/27/89	13.2	17.0	22.0	20.0	11.2	20.0	19.8	19.5
08/30/89	4.5	11.5	14.8	11.2	9.2	20.5	20.0	19.5
11/05/89	10.0	14.8	17.5	11.5	12.0	20.0	20.0	20.0
01/02/90	11.5	13.8	23.8	21.0	13.5	20.5	21.0	20.8
06/07/90	27.5	22.2	22.8	16.2	14.0	20.0	19.5	18.5
09/28/90	20.7	21.7	15.6	18.8	10.0	19.2	19.8	20.1
02/03/91	14.1	15.5	21.6	15.5	13.8	18.0	19.3	18.8
Average	10.52	12.03	14.89	12.31	12.31	20.40	19.81	19.24

Table 19: Nitrate-N concentrations for all sampling dates from the ports at well N4 in Village Green.

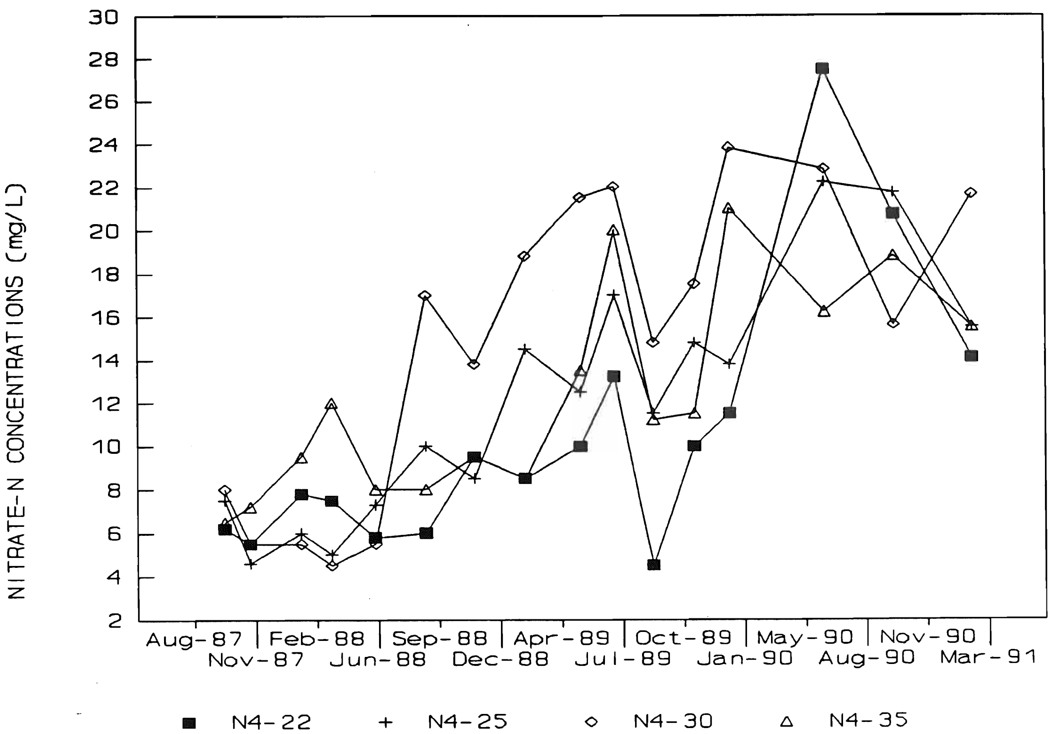

Figure 27a: Plot of average nitrate-N concentrations vs. time for well N4 in Village Green.

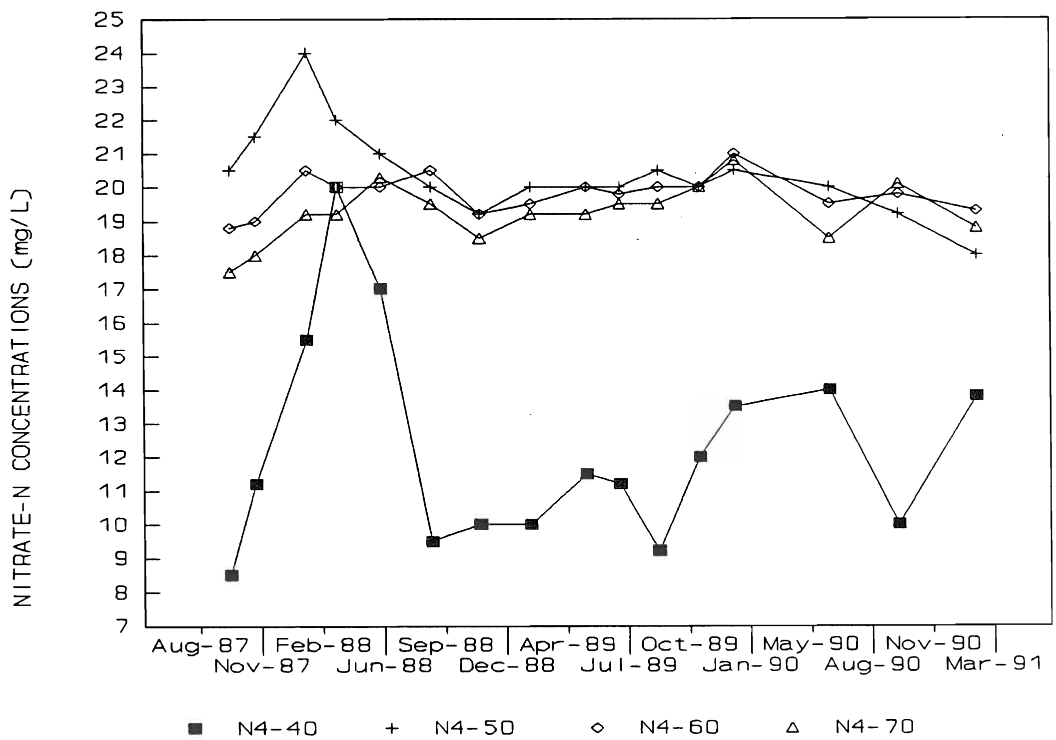

Figure 27b: Plot of average nitrate-N concentrations vs time for well N4 in Village Green.

3.1.2.2.4 Depth of Subdivision Impact

As with Jordan Acres, the groundwater chemistry in the downgradient multiport wells was used to define three zones in the aquifer relevant to subdivision recharge (subdivision-only, mixing zone, and upgradient-only). Graphs showing chemistry data verses depth at the downgradient wells are presented as Figures 28 a through j. The chemistry data are presented in Table 18 and/or Appendix C. The estimated depths separating the zones are included on the graphs.

The average groundwater elevation at the downgradient end of Village Green was calculated to be 326.2 meters M.S.L. and was thus used to define the top of the aquifer.

The top of the mixing zone (bottom of the subdivision-only zone) was defined at 321.5 m. The groundwater chemistry above this elevation is highly variable, but the variability is consistent with that expected from the various sources within the subdivision. Below this elevation the chemistry values begin to change consistently with depth. These observations are best demonstrated by the graphs of nitrate-N, chloride to sodium ratio, sodium, and relative fluorescence. The thickness of the subdivision-only zone is 4.7 meters (326.2 m - 321.5 m).

The bottom of the mixing zone (top of the upgradient-only zone) was set at an elevation 314 meters. This elevation is just below the second deepest port at most of the downgradient wells. It appears that the trend of consistently changing chemistry ended near this elevation. The thickness of the mixing zone is calculated to be 7.5 meters (321.5 m - 314.0 m).

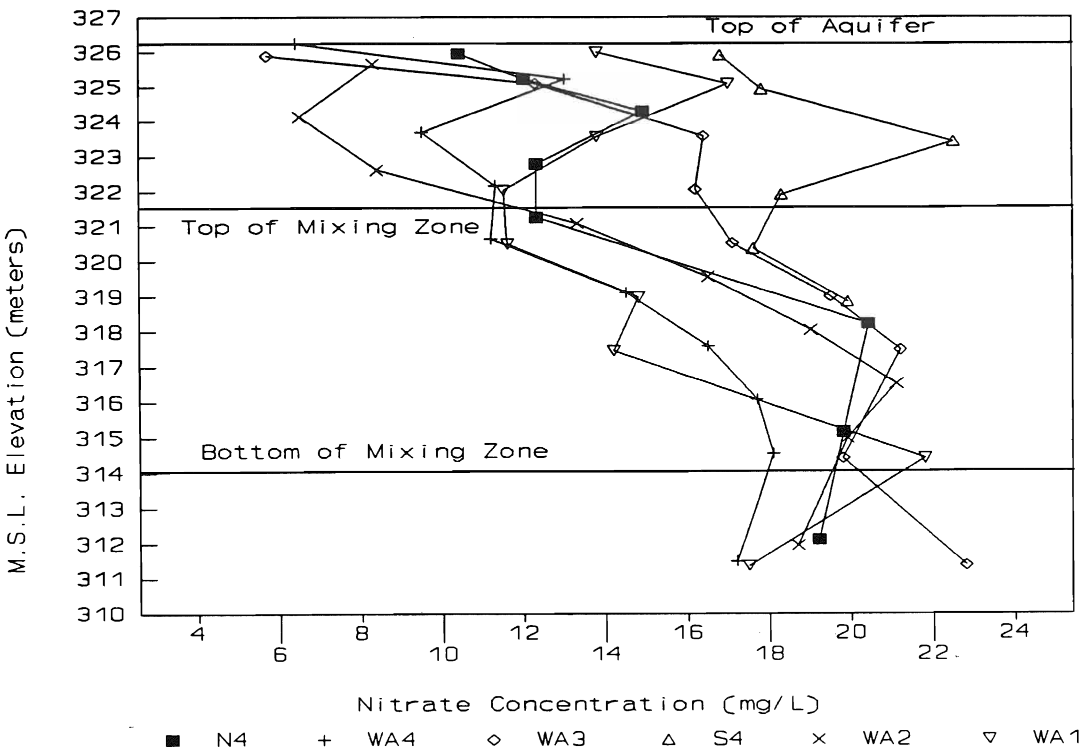

Figure 28a: Graph of average nitrate-N concentrations vs. elevation at the Village Green downgradient wells.

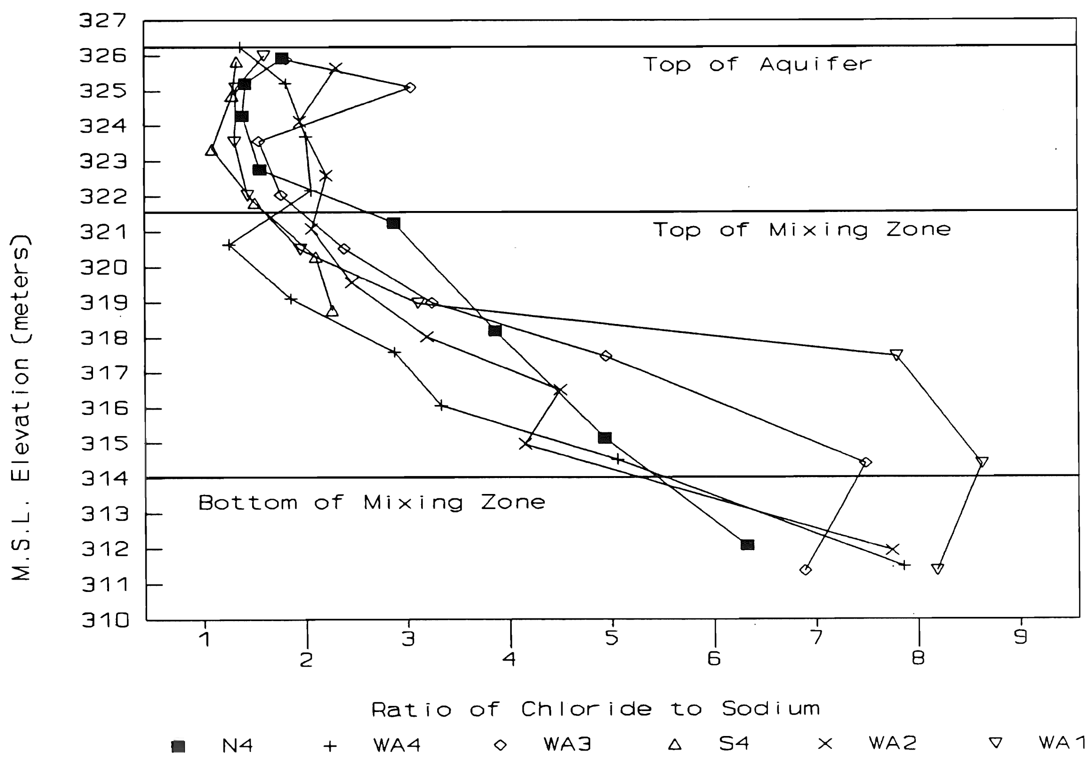

Figure 28b: Graph of average chloride to sodium ratios vs. elevation at the Village Green downgradient wells.

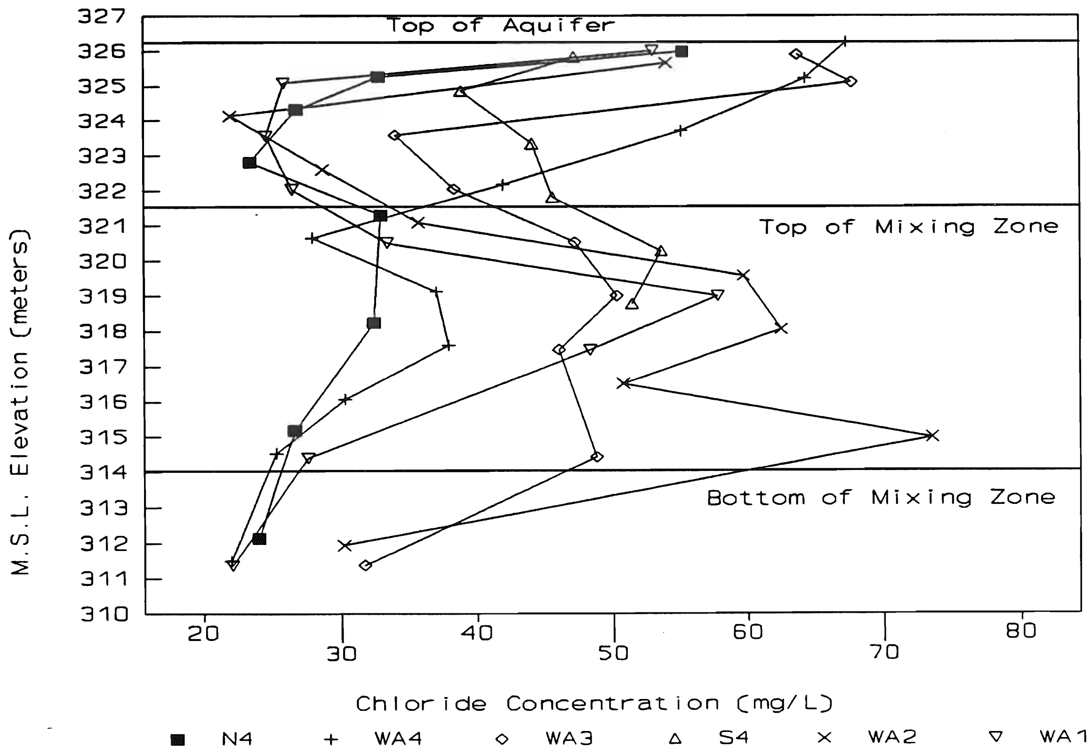

Figure 28c: Graph of average chloride concentrations vs. elevation at the Village Green downgradient wells.

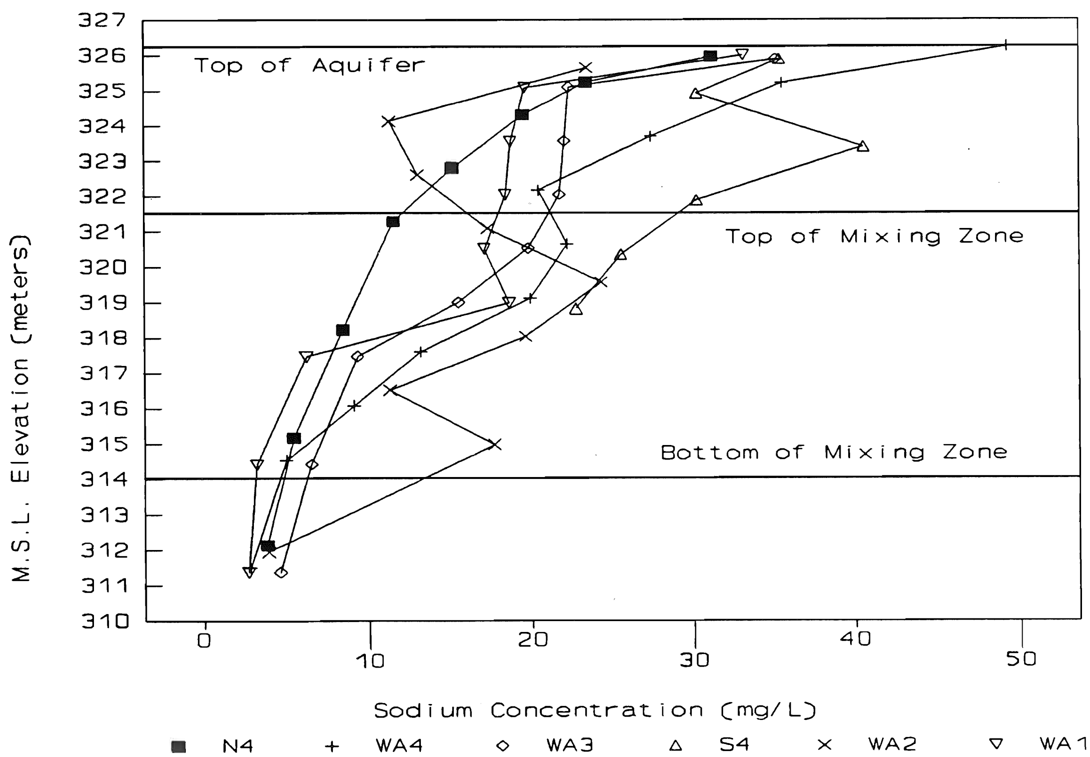

Figure 28d: Graph of average sodium concentrations vs. elevation at the Village Green downgradient wells.

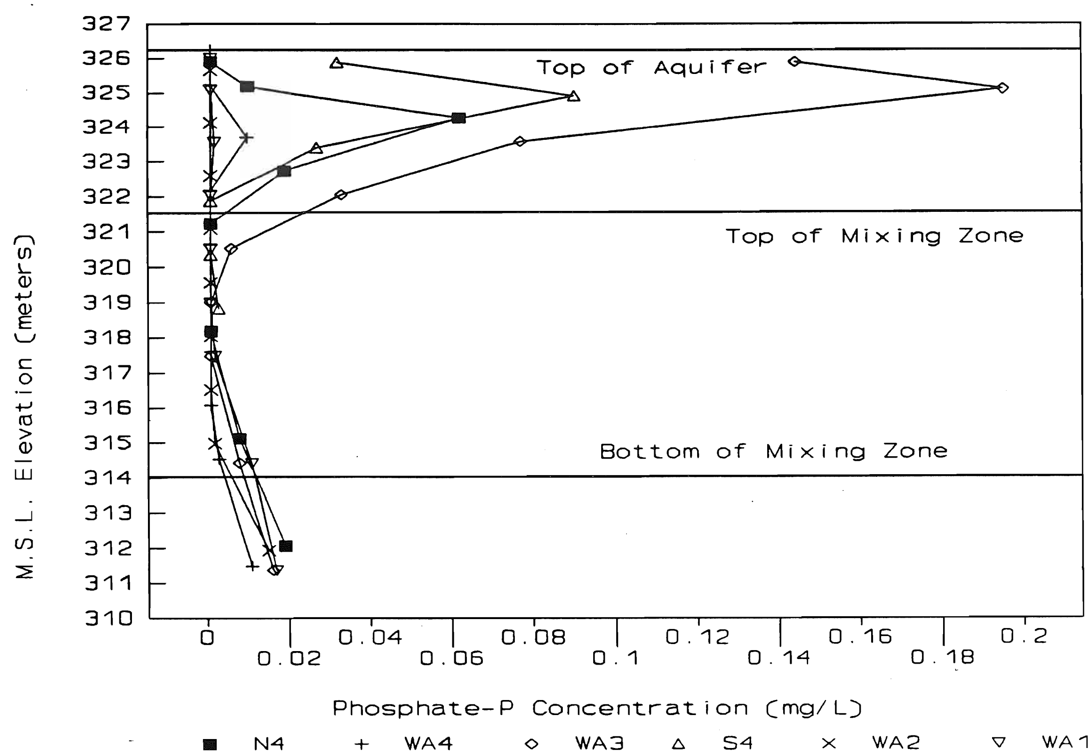

Figure 28e: Graph of average phosphate concentrations vs. elevation at the Village Green downgradient wells.

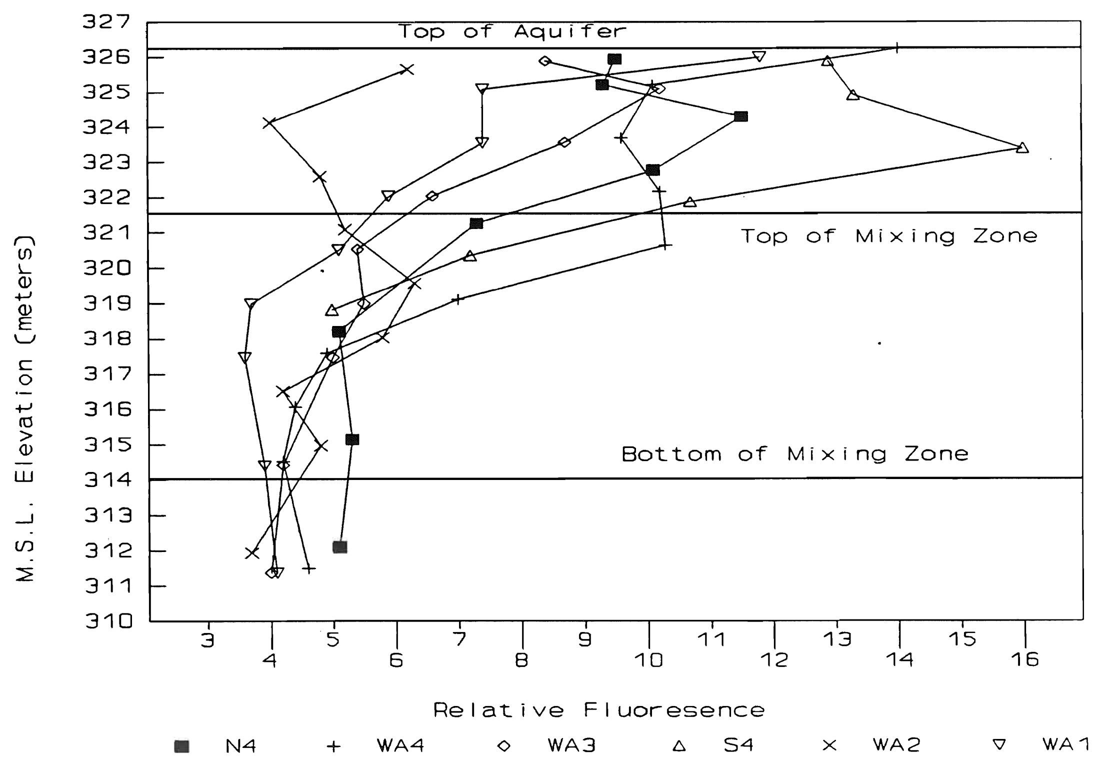

Figure 28f: Graph of average relative fluorescence vs. elevation at the Village Green downgradient wells.

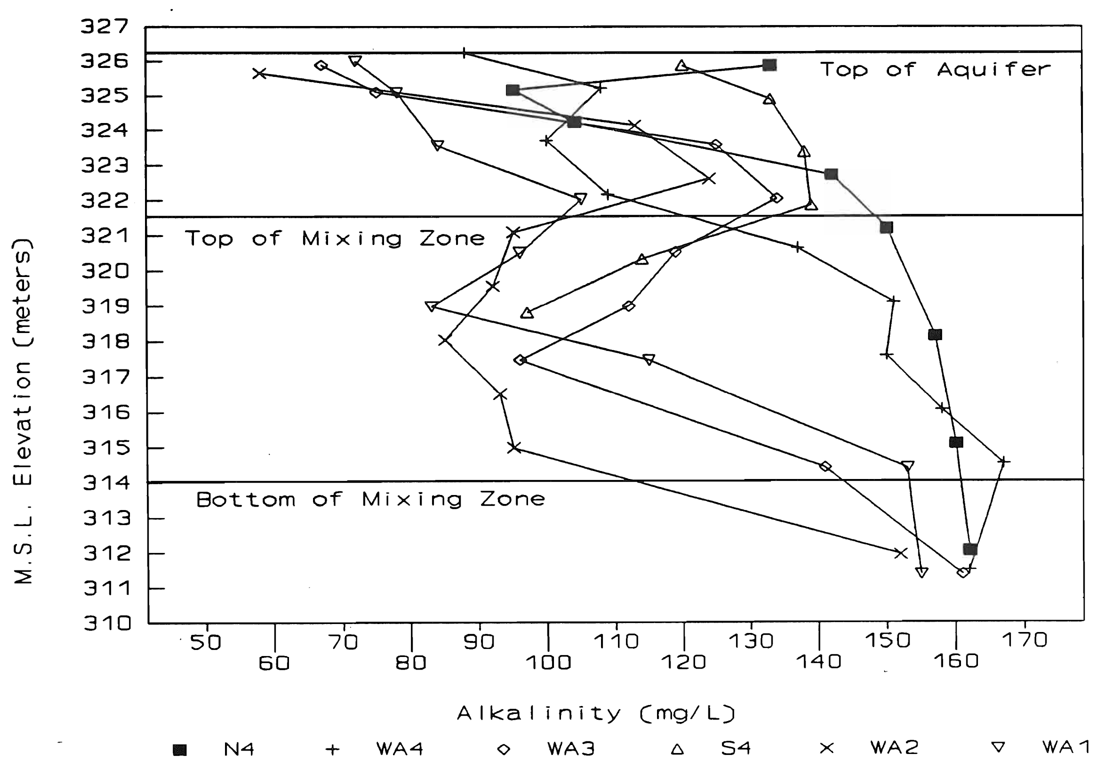

Figure 28g: Graph of average alkalinity vs. elevation at the Village Green downgradient wells.

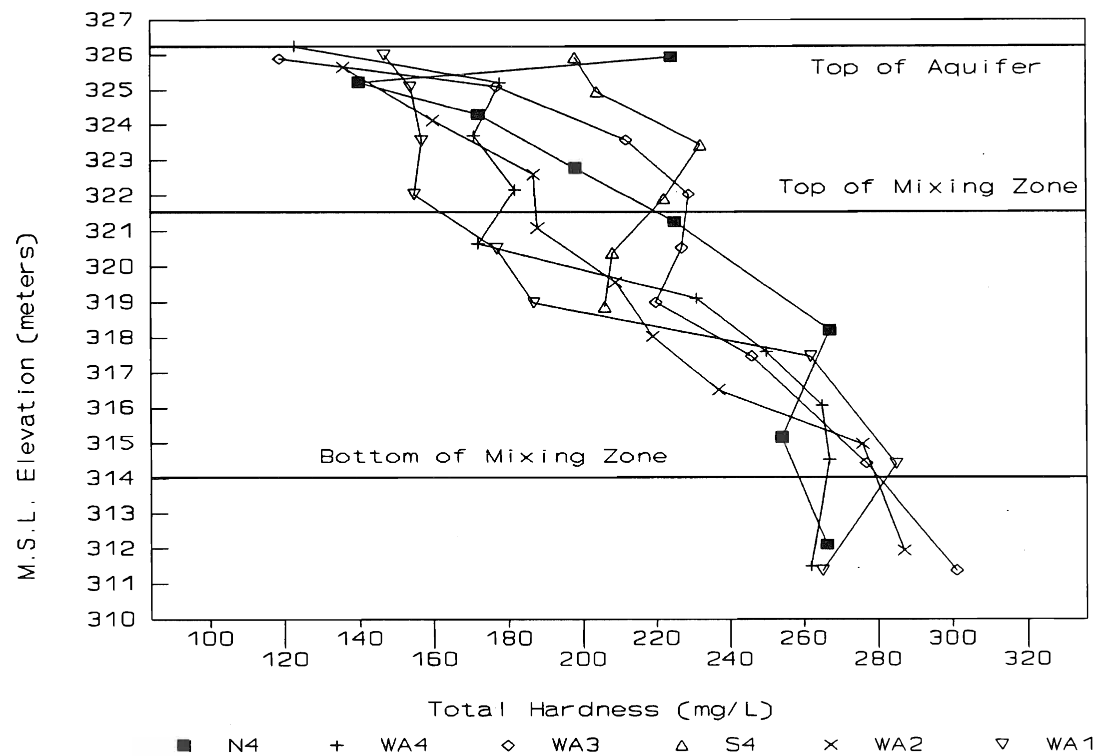

Figure 28h: Graph of average total hardness vs. elevation at the Village Green downgradient wells.

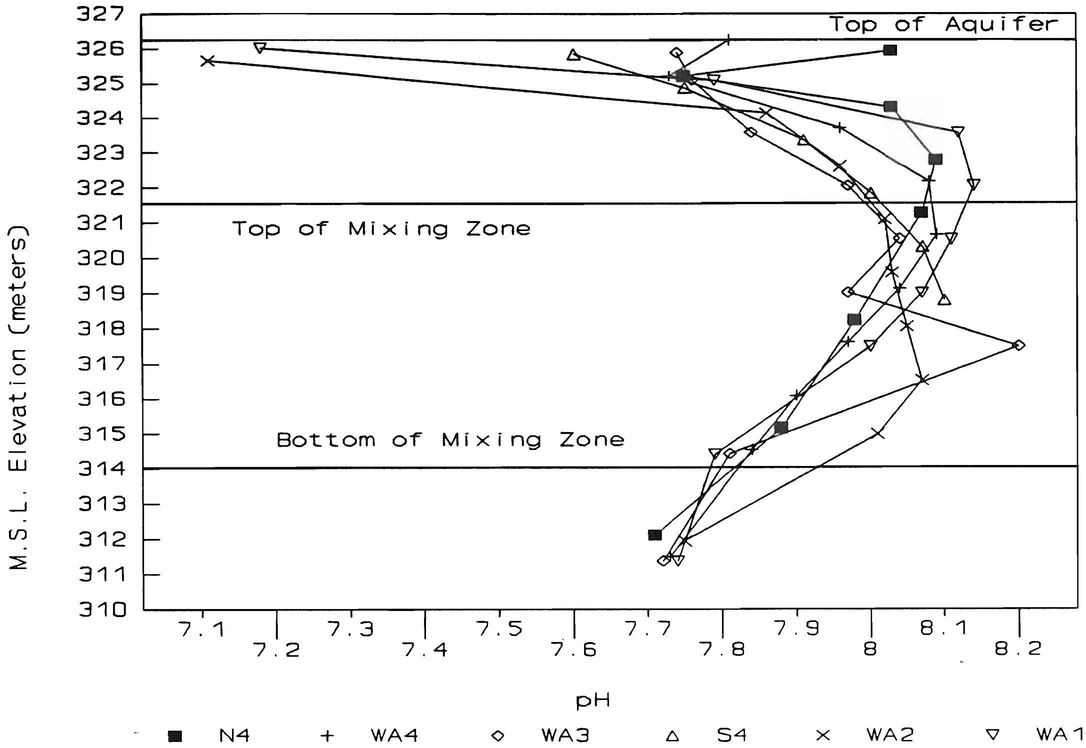

Figure 28i: Graph of average pH vs. elevation at the Village Green downgradient wells.

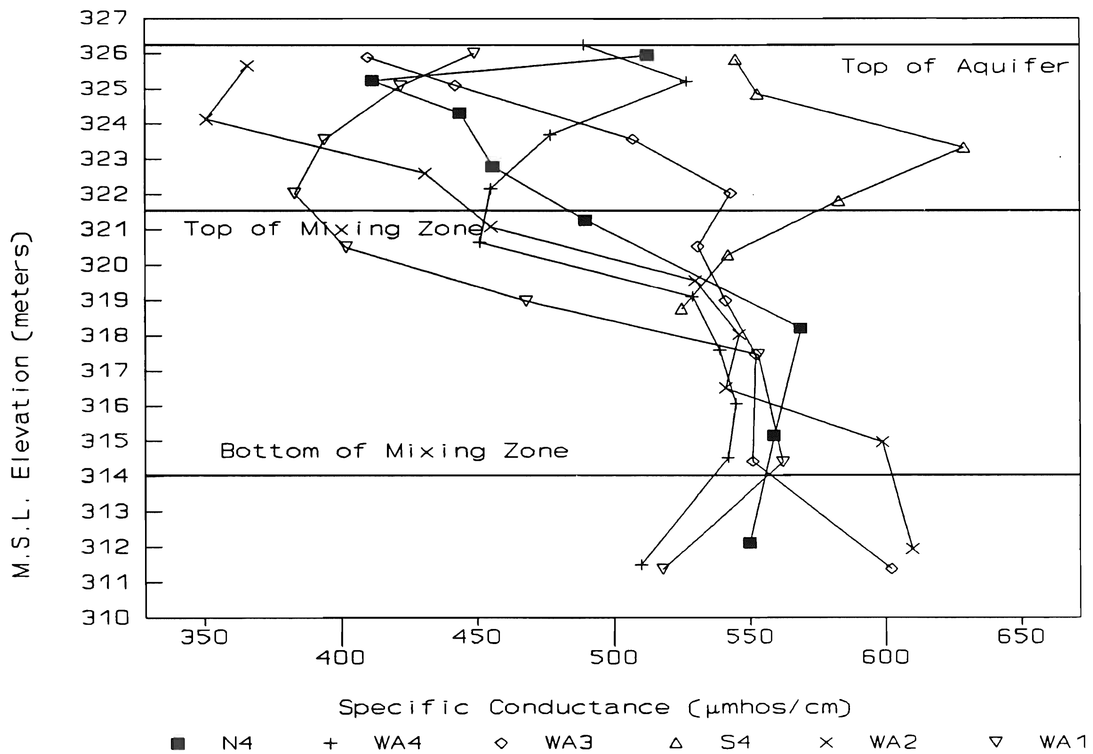

Figure 28j: Graph of average specific conductance vs. elevation at the Village Green downgradient wells.

The private wells are once again considered to be the primary mixing agents within the aquifer. The private wells in Village Green may be installed at a shallower depth into the aquifer than in Jordan Acres because of the high nitrate-N concentrations present at depth. Also, the thickness of the zone of subdivision-only water is greater in Village Green than in Jordan Acres. These two factors will cause a greater percentage of the water in the mixing zone to have originated from within the subdivision (as compared with Jordan Acres), although it is still expected that most of the water will have originated from upgradient. The fraction of subdivision-recharged water in the mixing zone was estimated to be 40%, thus the equivalent thickness of subdivision water in the mixing zone is calculated to be 3 meters (0.4 x 7.5 m).

Based on an equivalent subdivision thickness of 7.7 meters (4.7 m subdivision-only + 3 m equivalent portion of mixing zone), a cross-sectional width of 180 meters, a hydraulic conductivity ranging from 39 to 73 m/day, and a hydraulic gradient of 0.0020, the annual volume of subdivision discharge is between 39,000 and 74,000 m^3.

The nitrate-N concentrations for the upper 4 ports at wells WA1, WA2, WA3, and S4 were used to calculate the average nitrate-N concentration of the subdivision recharge. The average concentration is 13.6 mg/L. Therefore, given a yearly discharge volumes from above, the total mass of nitrogen discharged from the subdivision is in the range of 530 to 1000 kg/year.

3.1.3 Location-Specific Wells

3.1.3.1 Groundwater Quality from Natural Areas

The groundwater originating from "natural" areas (areas of minimal human impact) on the Central Wisconsin sand plain has been monitored by several wells installed and sampled by the Environmental Task Force lab - UWSP (unpublished data). Results from the chemical analysis of this water indicate that the levels of nitrate-N, chloride, sodium and phosphate are quite low, often near or below the detection limits for these compounds.

Several wells in the study area appeared to be monitoring the quality of the water originating from natural areas. The chemistry from these wells is summarized in Table 20. The locations of the wells are shown on Figure 29 (REE-SU, REE-LU, E1, and W1) or Figure 30 (FAR and UTI). A land use map of Jordan Acres is shown on Figure 3a (p. 8); a land use map of Village Green is shown on Figure 4a (p. 10).

Well Location	Well Point	# of Samples	Monitoring Period	NO_3-N	CL	NA	PO_4-P
REE	SU	12	Oct. '88 - Jan. '91	0.7	3.4	1.3	<0.002
REE	LU	13	June '88 - Aug. '90	0.8	4.5	1.5	<0.002
E1	22	15	Sep. '87 - Jan. '90	2.4	12.5	2.5	0.002
W1	22	18	July '87 - May '90	3.6	9.7	3.7	<0.002
FAR	SW	4	June '88 - Aug. '89	<0.2	0.5	0.8	<0.002
UTI	NW	4	June '88 - Aug. '89	0.6	<1.0	0.8	0.002
Average				2.0	6.8	2.4	<0.002

Table 20: Groundwater chemistry data (in mg/L) from wells showing little impact by man-made sources.

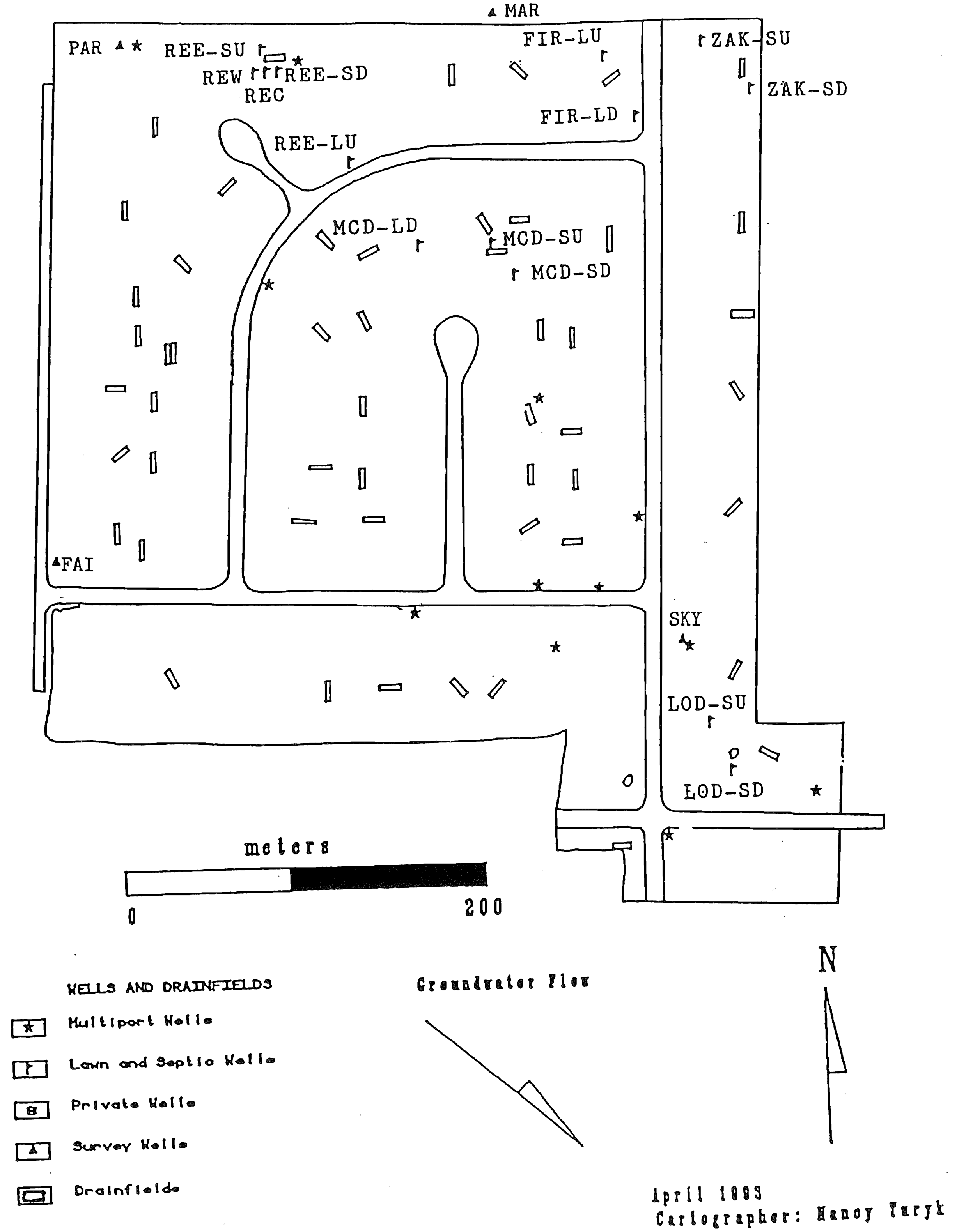

Figure 29: Location and identification of the survey wells and the lawn and septic wells in Jordan Acres.

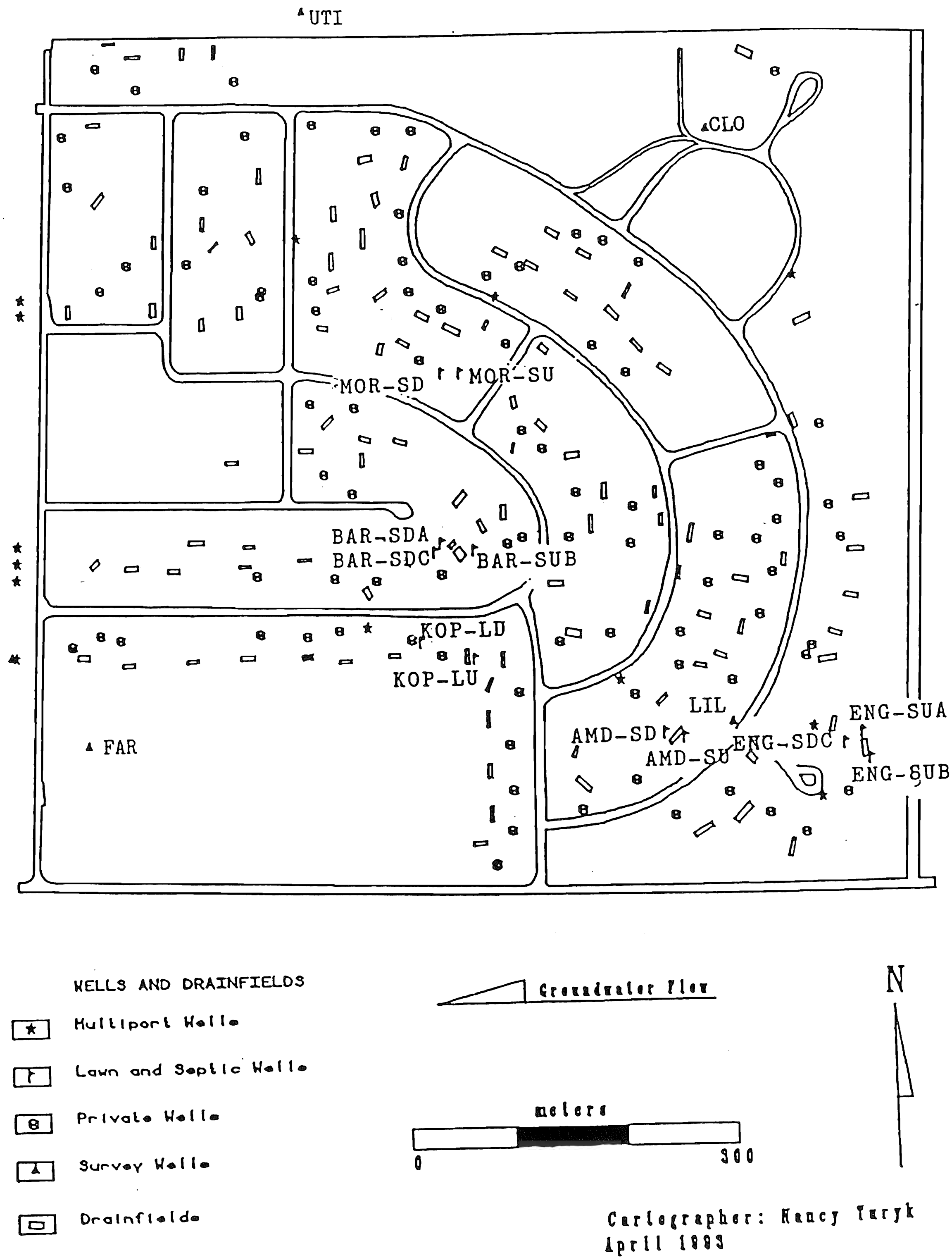

Figure 30: Location and identification of the survey wells and the lawn and septic wells in Village Green.

The first four wells listed in the table are located near the northwest corner of Jordan Acres. Although the inorganic chemical concentrations are relatively low, they are higher than many samples obtained from wells monitoring other natural areas. The woods is rather small (ten acres or so) and upgradient is agricultural land, thus crop fertilizers may account for the elevated concentrations of nitrate-N and chloride.

The last two wells are in Village Green. Their positions are such that they are near the downgradient end of the subdivision; however, they have undeveloped, nonagricultural land for their recharge areas.

The chemistry in these eight wells is believed to be representative of best-case water quality. They define the baseline for what would be expected if there were no major human impacts on the groundwater. As such, these are the values to which the chemistry data from the other wells should be compared. Any increase in the concentrations of the chemicals of concern can be attributed to man-made sources.

The nitrate-N, chloride, and sodium concentrations for all of the sampling dates at two of the wells in Jordan Acres are presented in Table 21. The data show nitrate-N concentrations consistently around 1.0 mg/L, chloride concentrations varying from less than 1 mg/L to 7 mg/L, and sodium concentrations between 1 and 2 mg/L. These results demonstrate the consistently low concentrations of these species that would likely occur in this area if there were no human impacts.

Sample Date	REE-LU			REE-SU		
	NO_3-N	Cl	Na	NO_3-N	Cl	Na
06/30/88	0.5	7	2.0	--	--	--
08/03/88	0.5	6	0.5	--	--	--
10/04/88	--	--	--	<0.2	3	1.5
10/20/88	0.8	7	1.5	0.5	4	1.0
01/18/89	0.8	5	2.5	1.2	6	2.0
03/31/89	1.0	7	1.0	1.0	5	1.6
05/26/89	1.2	6	1.6	--	--	--
06/13/89	--	--	--	1.0	5	1.6
08/08/89	1.5	3	1.0	1.8	3	1.0
09/08/89	0.8	5	1.5	0.8	5	1.5
10/26/89	1.2	<1	1.5	0.2	<1	1.0
01/08/90	1.0	3	1.5	0.5	3	1.0
02/14/90	0.5	3	1.4	--	--	--
05/17/90	< 0.2	4	1.6	<0.2	5	1.2
08/13/90	0.5	4	1.0	1.2	4	1.3
01/12/91	--	--	--	0.3	<1	1.2
Average	0.8	5	1.5	0.74	3	1.3

Table 21: Nitrate-N, chloride, and sodium concentrations (in mg/L) for all sampling dates at wells REE-LU and REE-SU in Jordan Acres.

3.1.3.2 Lawn Impacted Wells

In order to evaluate the impacts of lawns on groundwater quality, some of the wells were installed so that they had lawns as their primary recharge areas. Four wells from Jordan Acres and one from Village Green were determined to be representative of the water originating from lawn areas. Average inorganic chemistry data from these wells are shown in Table 22. The locations of the first four wells are shown on Figure 29, the fifth on Figure 30.

Well Location	Well Point	# of Samples	Monitoring Period	NO_3-N	CL	NA	PO_4-P
FIR	SD	11	July '88 - Jan. '90	4.0	13.3	5.1	<0.002
MCD	LD	12	June '88 - Aug. '90	7.8	14.3	5.1	<0.002
E2	22	8	Sep. '87 - Aug. '89	2.9	4.8	3.9	0.011
E3	25	14	July '87 - May '90	2.7	19.3	12.1	<0.002
S3	22	12	Sep. '87 - Mar. '89	5.3	37.8	14.7	<0.002
Average				4.5	17.9	8.2	0.002

Table 22: Groundwater chemistry data (in mg/L) from wells impacted primarily by lawns.

The nitrate-N concentrations from these wells are of greatest interest because road salt may be influencing the concentrations of chloride and sodium. E2 appears to be the only well sampling groundwater that is relatively unaffected by sodium and chloride.

The nitrate-N data is similar to data obtained when monitoring the groundwater beneath the Village of Park Ridge (a residential area near Stevens Point that is served by sanitary sewers but not municipal water) (Environmental Task Force, unpublished data). The water quality presented above and the data from the sewered residential area may have some nitrogen impact from wastewater—either from plume mixing in the subdivisions or from leaking sewers in the residential area.

The relatively high average phosphate concentration in E2 was due to detects of 0.05 mg/L and 0.03 mg/L measured in March and June of 1989. The cause of these higher phosphate levels is not known but may be due to lawn fertilizers.

MCD LD is the well that appears to be showing the influence of a common subdivision practice—intensive lawn care. The well is located near one of the greenest and best manicured lawns in the subdivision. Information obtained from the homeowner indicates that the lawn is fertilized four times per year, watered every other day, and mowed weekly. This level of care, while rather rigorous, is not unusual on the sand plain.

The nitrate-N, chloride, and sodium concentrations at two wells (REE-LU and MCD-LD) monitoring the groundwater quality up- and downgradient of an intensively managed lawn is presented in Table 23. A graph of the nitrate-N data is presented in Figure 31a; a graph of the chloride to sodium ratios is presented in Figure 31b. The nitrate-N, chloride, and sodium concentrations were higher in the well downgradient of the lawn (MCD-LD) than in the upgradient well. The higher chloride and sodium concentrations may be due in part to salt applied to the road present between the two wells, but the nitrate-N concentrations are primarily attributed to lawn fertilizers. Fluctuations in chloride concentrations coincide with the changes in nitrate-N concentrations; when nitrate-N concentrations are high, chloride concentrations are also high. Sodium concentrations show similar variations, but there were a few sampling occasions (e.g., October 1988 and August 1989) when nitrate-N and chloride concentrations were relatively high and sodium concentrations were relatively low.

At MCD-LD, the higher nitrate-N concentrations were generally detected in the late summer and fall and the lower concentrations detected in late winter and early spring; however, the second highest nitrate-N concentration (14.2 ppm) was from a sample obtained in January 1990, and the lowest concentration (1 ppm) was measured in a sample from June 1988.

Sample Date	REE-LU			MCD-LD		
	NO_3-N	Cl	Na	NO_3-N	Cl	Na
06/30/88	0.5	7	2.0	1.0	6	2.0
08/05/88	0.5	6	0.8	3.5	9	2.3
10/20/88	0.8	7	1.5	9.8	13	3.0
01/18/89	0.8	5	2.5	7.8	16	5.5
03/31/89	1.0	7	1.0	5.8	9	3.0
05/26/89	1.2	6	1.6	2.2	10	4.1
08/08/89	1.5	3	1.0	9.8	23	2.0
09/08/89	0.8	5	1.5	14.4	30	11.5
10/26/89	1.2	<1	1.5	13.0	27	7.0
01/08/90	1.0	3	1.5	14.2	11	8.5
02/14/90	0.5	3	1.4	4.8	8	6.9
05/17/90	< 0.2	4	1.6	7.0	9	4.8
Average	0.83	5	1.5	7.8	14	5.1

Table 23: Groundwater chemistry data (in mg/L) from two wells monitoring the groundwater upgradient and downgradient of an intensively managed lawn.

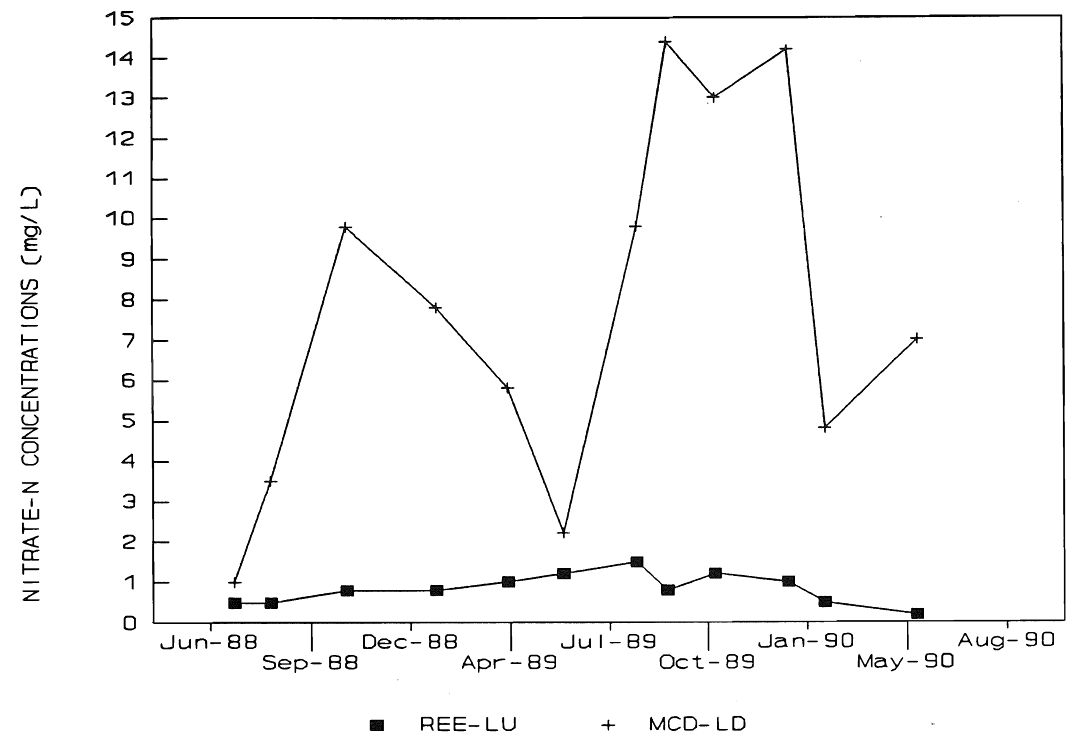

Figure 31a: Plot of groundwater nitrate-N concentration vs. time for well REE-LU and MCD-LD in Jordan Acres. One well is upgradient of a lawn; the other is downgradient.

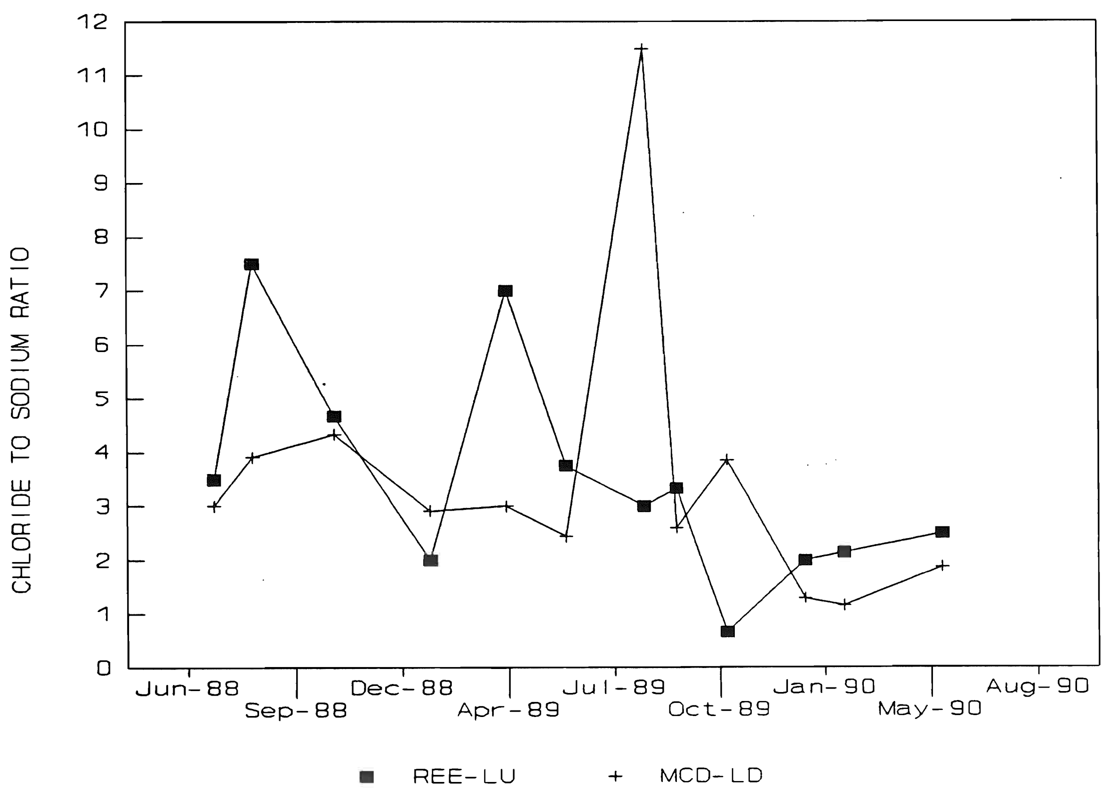

Figure 31b: Plot of groundwater chloride to sodium ratio vs. time for well REE-LU and MCD-LD in Jordan Acres. One well is upgradient of a lawn; the other is downgradient.

3.1.3.3 Septic System Impacted Wells

Several wells were installed directly up- and downgradient of septic system drainfields. Chemistry data from the wells that demonstrate the impact of septic systems are summarized in Table 24a. Results of the other inorganic analyses (pH, specific conductance, alkalinity, total hardness, and relative fluorescence) performed on samples from the wells monitoring drainfield impacts are presented in Table 24b. The locations of the first two wells (in Jordan Acres) are identified on Figure 29 (p.123), the rest (in Village Green) on Figure 30 (p.124).

Well Location	Well Point	# of Samples	Monitoring Period	Distance from Drainfield	NO_3-N	Cl	Na	PO_4-P
REE	SU	12	Oct. '88 - Jan. '91	Upgradient	0.7	3	1.3	<0.002
REC	SDM	11	Sep. '89 - Jan. '91	2-3 meters	37.9	35	22.5	<0.002
ENG	SUA	9	Oct. '88 - Mar. '90	Upgradient	8.5	80	48.6	<0.002
S1	22	15	Sep. '87 - Mar. '90	15 meters	24.0	77	54.4	0.452
AMD	SU	9	Oct. '87 - Mar. '90	Upgradient	2.9	44	12.0	0.004
AMD	SD	9	Oct. '88 - Mar. '90	2-3 meters	33.7	130	108	6.987
BAR	SUB	9	Oct. '88 - Mar. '90	Upgradient	16.6	65	28.6	0.004
BAR	SDA	9	Oct. '88 - Mar. '90	2-3 meters	36.9	69	68.6	6.503
MOR	SU	7	Oct. '88 - Mar. '90	Upgradient	5.6	22	16.5	<0.002
MOR	SD	7	Oct. '88 - Mar. '90	10 meters	19.2	51	40.7	3.486

Table 24a: Groundwater chemistry data (in mg/L) from wells installed directly up- and downgradient of septic system drainfields.

Well Location	Well Point	# of Samples	Monitoring Period	pH S.U.	Spec. Cond. µmho/cm	Alk. mg/L	Total Hard. mg/L	Rel. Fluor.
REE	SU	12	Oct. '88- Jan. '91	8.09	226	104	125	5
REC	SDM	11	Sep. '89 - Jan. '91	7.53	668	116	279	34
ENG	SUA	9	Oct. '88 - Mar. '90	7.60	495	95	159	11
S1	22	15	Sep. '87 - Mar. '90	7.23	655	92	172	36
AMD	SU	9	Oct. '88 - Mar. '90	7.44	317	87	135	7
AMD	SD	9	Oct. '88 - Mar. '90	7.12	968	101	221	35
BAR	SUB	9	Oct. '88 - Mar. '90	7.24	560	116	202	10
BAR	SDA	9	Oct. '88 - Mar. '90	6.65	838	139	255	125
MOR	SU	7	Oct. '88 - Mar. '90	7.15	315	101	132	16
MOR	SD	7	Oct. '88 - Mar. '90	7.08	597	138	198	31

Table 24b: Groundwater chemistry data (as indicated) from wells impacted primarily by septic system drainfields.

The first two wells listed in Table 24 (REE-SU and REC-SDS) are the ones that most clearly demonstrate the effects that drainfields can have on groundwater quality. The chemistry from REE-SU is considered to be virtually free of human impacts, as was discussed in Section 3.1.3.1. In the upgradient well the average concentrations of nitrate-N and sodium were around 1 mg/L and the average chloride concentration was less than 4 mg/L, thus the high average concentrations of these species (48 mg/L NO_3-N, 35 mg/L Cl, 20 mg/L Na) in the downgradient well are attributed solely to effluent from the drainfield. Similar upgradient/downgradient relationships are also seen at the other well pairs; however, the other downgradient wells all have high phosphate concentrations in addition to high concentrations of the other species.

The data also show that, in general, wells closer to a drainfield tend to have higher concentrations of contaminants than those farther away.

The concentrations of chloride and sodium from AMD-SD are much higher than were detected at any of the other wells monitoring drainfield impacts. These high concentrations are attributed to the effects of a water softener thought to be present in the house.

Comparing the upgradient and downgradient data, groundwater impacted by septic systems has a lower pH, and a higher specific conductance, total hardness, and relative fluorescence.

The chemistry in the septic-impacted wells is similar to or higher than the results reported by other authors. Robertson et al. (1991) installed relatively intensive monitoring networks downgradient of two septic drainfields in Ontario, Canada. Average concentrations from sampling ports considered to be in the "plume core" were determined to be: NO_3-N = 33 and 39 mg/L; Cl = 24 and 38 mg/L; Na = 86 and 45 mg/L; and PO_4-P = 4 and 0.01 mg/L. As with this study, one of the areas (the former values) was downgradient of an agricultural area which was impacting groundwater quality.

The variability of the inorganic contaminant concentrations in wells monitoring septic system impacts is demonstrated by the nitrate-N concentrations measured in three wells installed in Village Green and presented in Table 25. It appears that the chemistry shows considerable fluctuation; however, nitrate-N exceeded the 10 mg/L drinking water standard on all sampling occasions. This indicates that septic systems are a source of year-round groundwater contamination. MOR-SD was installed farther from the drainfield then the other two wells, hence has lower concentrations of nitrate-N (and other species).

Sample Date	AMD-SD	BAR-SDA	MOR-SD
10/21/88	40.8	35.0	19.5
01/11/89	34.5	36.5	14.0
04/05/89	39.5	44.5	12.5
06/07/89	30.0	43.2	31.0
07/05/89	36.5	39.5	--
08/03/89	38.0	40.5	17.0
01/03/90	28.2	23.5	20.0
03/20/90	35.0	29.5	20.6
Average	35.31	36.53	19.23

Table 25: Nitrate-N concentrations (in mg/L) for all sampling dates from wells AMD-SD, BAR-SDA, and MOR-SD. The wells are impacted primarily by septic system drainfields.

3.2 Water Budgets Based on Hydrologic Data

The volume of water recharged to the aquifer can be determined based solely on hydrologic data. It is calculated by multiplying the amount of groundwater recharge (as a height) times the area in which the recharge occurs.

The quantity of recharge from vegetated areas within the subdivisions is considered to be equal to the amount of water added as precipitation less the amount of water lost due to evapotranspiration (runoff is considered to be negligible). It was assumed that ninety percent of the precipitation falling on impervious surfaces that does not immediately evaporate goes to groundwater recharge (see Section 3.4.1), thus the volume of recharge from impervious areas is equal to 81% of precipitation (assuming 19% evapotranspirates). The equivalent height of groundwater recharge is calculated by multiplying the amount of groundwater recharge from each land-use area times the percentage of the area in each land use.

The aquifer thickness at the downgradient end of the subdivision that is represented by the volume of recharge occurring from the subdivision is equal to the equivalent height of water recharged from the subdivision during the groundwater flow time beneath the subdivision divided by the specific yield of the aquifer.

The factors and results of the water budgets (for both Jordan Acres and Village Green) determined based on hydrologic data are presented in Table 26. Note that the values are for only the portion of subdivisions (termed cuttings) impacting the monitoring wells.

	Jordan Acres Cutting	Village Green Cutting
Width of cross section (m)	180	180
Length of flow path (m)	360	850
Fraction of area with vegetation	0.78	0.76
Fraction of area which is impervious	0.22	0.24
Average yearly precipitation (cm)	78	83
Average yearly recharge from vegetated areas: precipitation - 53 cm evapotranspiration (cm)	25	30
Average yearly recharge from impervious areas: 81% of precipitation (cm)	63	67
Total yearly equalized recharge (cm)	33	39
Average volume of water recharged from subdivision (m^3/year)	21,000	60,000
Average linear groundwater flow velocity - medium (m/day)	0.49	0.37
Average flow time beneath subdivision (years)	2.2	6.9
Total equalized recharge occurring during flow time beneath subdivision (cm)	77	280
Specific yield of aquifer	0.3	0.3
Equivalent aquifer thickness of recharge at downgradient end of subdivision (m)	2.6	9.3

Table 26: Water budget factors and results for Jordan Acres and Village Green based on hydrologic data.

3.3 Summary of Nitrogen and Water Budget Results Determined Using Field Data

The variables and results of nitrogen and water budgets using subdivision field data are presented in Table 27.

The areal values used in the budget calculations (width of cross section and length of flow path) were based on data obtained from only a portion of the subdivisions (termed cuttings). The area included in the Jordan Acres cutting is shown on Figure 32; the Village Green cutting is shown on Figure 33.

The depth of subdivision impacted water was estimated based on the chemistry data obtained from the downgradient multiport wells and was discussed in sections 3.1.2.1.4 and 3.1.2.2.4 above.

The average linear groundwater flow velocities for both subdivisions were determined based on a range in hydraulic conductivity from 0.045 cm/sec to 0.085 cm/sec, an effective porosity of 0.30, and hydraulic gradients of 0.0026 (Jordan Acres) and 0.0020 (Village Green). The discharge volumes were calculated based on these hydrogeologic characteristics and the cross sectional area impacted by the subdivision.

The average nitrate-N concentrations were calculated from those ports at the downgradient multiport wells that were determined to be monitoring the groundwater recharged from subdivision-only sources.

Characteristic	Jordan Acres Cutting	Village Green Cutting
Width of cross section (m)	180	180
Length of flow path along cutting (m)	360	830
Equivalent depth of subdivision impacted water based on chemistry data (m)	3.4	7.7
Equivalent depth of subdivision impacted water based on hydrologic data (m)	2.6	9.3
Area of cross section discharging groundwater from the cutting (m^2)	610	1400
Average linear groundwater flow velocity - low (m/day)	0.34	0.26
Average linear groundwater flow velocity - medium (m/day)	0.49	0.37
Average linear groundwater flow velocity - high (m/day)	0.64	0.49
Discharge of subdivision impacted groundwater from cutting - low (m^3/year)	23,000	39,000
Discharge of subdivision impacted groundwater from cutting - medium (m^3/year)	33,000	57,000
Discharge of subdivision impacted groundwater from cutting - high (m^3/year)	43,000	74,000
Average nitrate-N concentration of groundwater leaving cutting (mg/L)	9.0	13.6
Mass of nitrogen in discharge from cutting - low (kg/year)	200	530
Mass of nitrogen in discharge from cutting - medium (kg/year)	300	770
Mass of nitrogen in discharge from cutting - high (kg/year)	390	1000
Groundwater flow time across cutting - slow (years)	2.9	9.0
Groundwater flow time across cutting - medium (years)	2.0	6.3
Groundwater flow time across cutting - fast (years)	1.5	4.8
Average yearly precipitation (cm)	78	83
Average volume of water recharged based on hydrologic data (m^3/year)	21,000	60,000
Average volume of water recharged assuming no drainfields, no impervious areas, and recharge = annual precipitation - 53 cm (m^3/year)	16,000	46,000

Table 27: Results of nitrogen and water budget calculations based on field data obtained from Jordan Acres and Village Green subdivisions.

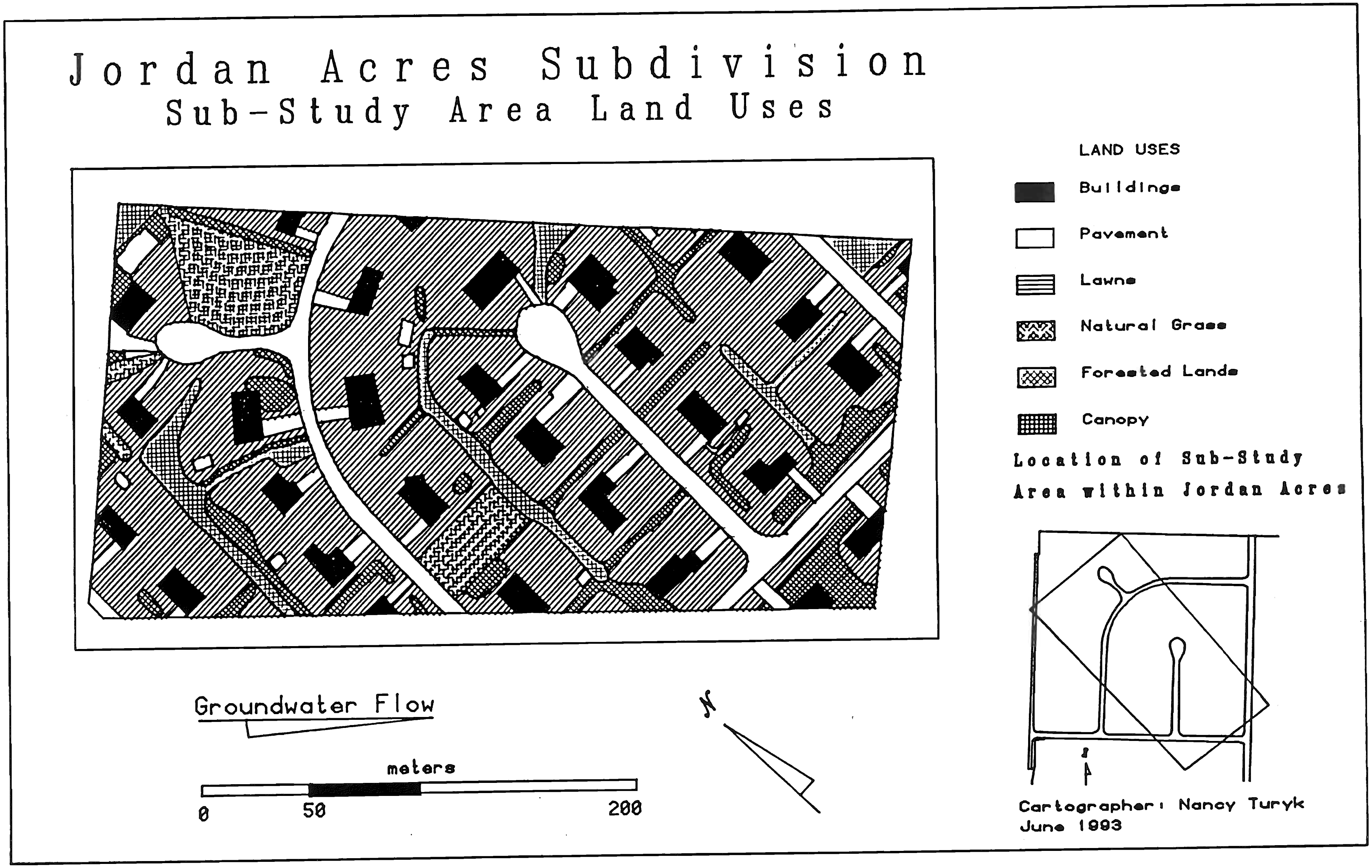

Figure 32: Portion (termed cutting) of Jordan Acres used for nitrogen and water mass balance calculations.

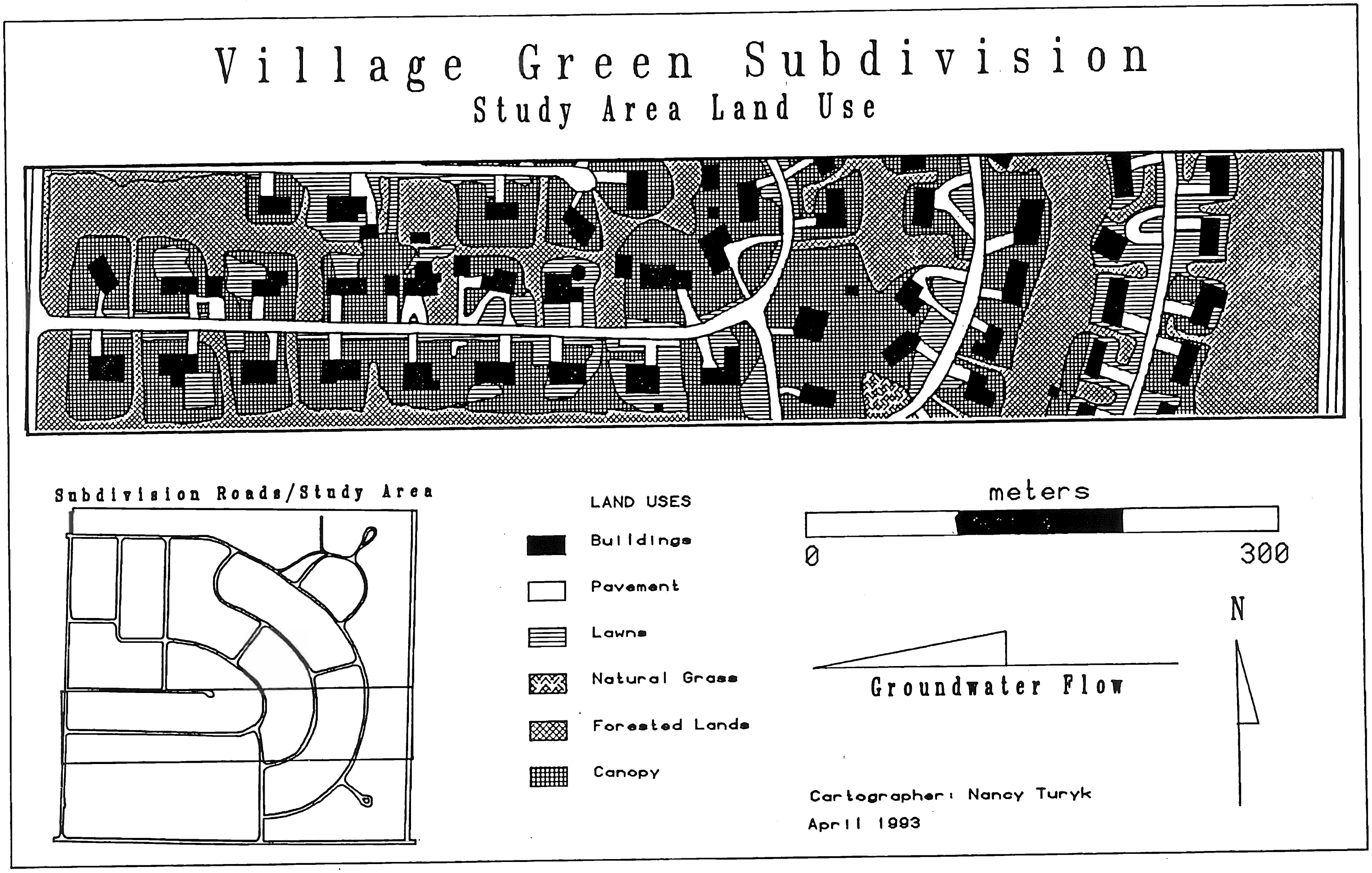

Figure 33: Portion (termed cutting) of Village Green used for nitrogen and water mass balance calculations.

The mass of nitrogen discharged from the cuttings were calculated using the average nitrate-N concentrations and the volume of discharge (mg/L x m^3/year x 0.001 = kg/year).

The groundwater flow times across the cuttings were calculated using the length of the subdivision and the range in average linear groundwater flow velocities (meters x days/meter x 1/365 = years).

The average yearly precipitation was calculated based on the average amount of precipitation that occurred over the groundwater flow time beneath the subdivision for a time period preceding and during the study (Jordan Acres: 1986 to 1990; Village Green: 1981 to 1990).

The volume of water recharged based on hydrologic data includes the water recharged from both vegetated and impervious areas.

The final volume of water presented in Table 27 represents how much water would be recharged to the aquifer under natural conditions (i.e., if there were no human impacts). This volume is included for comparing to the other water-budget values. The volume of recharge from a subdivision is expected to be greater than the amount from an equal area of natural land because more of the water that falls on impervious surfaces (assumed to be 81% of precipitation in the study areas) will recharge to the groundwater, as compared with about 25% from vegetated areas. This is discussed in section 3.4.1 below.

The effective aquifer thicknesses calculated using the hydrologic data (JA = 2.6 meters, VG = 9.3 meters) are reasonably close to those determined using the chemistry data (JA = 3.4 meters, VG = 7.7 meters); however, the differences are worthy of comment.

In the case of Jordan Acres, the hydrologic budget result suggests that the effective aquifer thickness is considerably less than was determined using the chemistry data, which in turn suggests that the elevated concentrations of various chemical species detected in some of the deeper ports were in fact not due to subdivision impacts but rather the natural variability in the upgradient groundwater chemistry. Both of the effective aquifer thicknesses indicate that only the shallowest two or three ports at the downgradient multiport wells are likely to be significantly influenced by subdivision recharge (i.e., most of the multiport well ports in Jordan Acres are not monitoring subdivision recharge). Also, the average annual volume of recharge that was calculated using the hydrologic data is less than the lowest volume of discharge calculated using the groundwater flow velocity. If the aquifer thickness calculated using the hydrologic data is used in calculating the yearly discharge of subdivision-impacted groundwater, the yearly discharge would be in the range of 17,000 m^3/year to 33,000 m^3/year. The yearly volumes of subdivision recharge calculated using the hydrologic budget and the BURBs model both fall near the middle of this range of discharge volumes.

The results of both water-budget methods are relatively more similar for Village Green than for Jordan Acres. The effective aquifer thickness calculated using the hydrologic method (9.3 meters) is somewhat higher than that determined using the chemistry data (7.7 meters), which may suggest that the portion of the aquifer affected by the subdivision was underestimated. The annual volume of recharge calculated using the hydrologic method (60,000 m^3/year) falls well within the range of discharge volumes (39,000 m^3/year to 74,000 m^3/year) calculated using the chemistry method. If the larger effective aquifer thickness is used to calculate the discharge volumes, the range is 47,000 to 89,000 m^3/year.

The average nitrate-N concentrations of subdivision-recharge water are similar for both subdivisions (JA = 9.0 mg/L; VG = 13.6 mg/L). The reason why the concentration in Village Green is higher than that in Jordan Acres is likely to be due to the higher concentration of nitrate-N in the upgradient groundwater, which contributes to the water used in the subdivision (and returned as recharge) having a higher concentration of nitrate-N.

Note that both nitrate concentrations are close to the 10 mg/L groundwater enforcement standard; one is slightly lower and the other is somewhat higher. This emphasizes the primary focus of this study; do unsewered subdivisions adversely affect groundwater quality? Based on these results, the answer is yes. Not only does this suggest that water from private wells screened in a subdivision-impacted zone within the aquifer are likely to be unsafe to drink (using 10 mg/L nitrate-N as the standard for safety), but if the well-head area of a municipal water-supply well is dominated by subdivision land use, an entire community's drinking water supply can be jeopardized.

3.4 BURBS

BURBS is a nitrogen and water mass balance program created at Cornell University (Hughes and Pacenka, 1985). It was developed to estimate the average concentration of nitrate-N in the recharge water originating from urban areas. The variables used in the model are:

1) Fraction of land in turf.
2) Fraction of land which is impervious.
3) Average persons per dwelling
4) Housing density.
5) Precipitation rate.
6) Water recharged from turf.
7) Water recharged from natural land.
8) Evaporation from impervious surfaces.
9) Runoff from impervious recharged.
10) Home water use per person.
11) Nitrogen concentration in precipitation.
12) Nitrogen concentration in water used.
13) Turf fertilization rate.
14) Fraction of nitrogen leached from turf.
15) Fraction of wastewater N lost as gas.
16) Wastewater fraction removed by sewer.
17) Nitrogen per person in wastewater.
18) Nitrogen removal rate of natural land.

Each of these variables is discussed and model input values are defined below.

The areas that were modelled are the sections (termed cuttings) of the subdivisions that are impacting selected downgradient multiport wells. The monitoring networks were not randomly spaced across the subdivisions, consequently the data are more representative of a part of the subdivisions than the entire subdivision. Because a goal of this project was to compare BURBS predictions with field monitoring values, it was necessary to define the BURBS parameters in terms of the conditions impacting the monitoring network. Thus while the demographic-type variables were defined using averages for the entire subdivision, the areal-type variables were based on specific land use within the cutting areas.

On site waste disposal is the primary source of nitrogen loading to groundwater from a subdivision. Once the model variables were accurately defined, simulations were run to evaluate the effect of doubling and halving the housing density (hence septic system density). Relative amounts of land use areas (i.e., turf, natural, and impervious) were adjusted to accommodate the increased (decreased) amount of impervious area associated with more (fewer) houses in a given area. For these simulations, the area of houses and driveways were doubled (halved) and the area of turf and natural land were reduced (increased) by a like amount in proportion with their baseline areas. The amount of road area was kept constant.

A number of runs were made to calibrate the model in terms of the amount of nitrate-N leached from lawn fertilizers. For these simulation runs, the amount of wastewater removed by sewer was set at 1.00 so as to eliminate wastewater impacts from the simulation results. The leaching values ranged from 0.05 to 0.40. The leaching value considered to be most representative of observed in-field conditions was the one that yielded a nitrate-N concentration most similar to the concentrations measured in water samples of wells impacted solely by lawns (approximately 4.3 mg/L NO_3-N).

Several runs were made so as to demonstrate the effect of precipitation amounts on groundwater nitrate-N concentrations. Wet years and dry years were simulated.

Once the mass balance variables were accurately defined for the conditions in the subdivisions, several runs were made to demonstrate how soil type and reduced groundwater recharge effect nitrate-N concentrations in groundwater.

3.4.1 Variable Definition

The values used for the model variables are discussed below. The units of measure used for the variables and results of the BURBs model are defined in English units; however, they have been converted to metric units for the discussions herein.

The fraction of land in turf, impervious, and natural ground covers in the cuttings were calculated by pcARCINFO from the respective portions of the subdivision maps. Maps of the cuttings are presented in Figures 32 and 33 (p. 139 and p.140). In the simulations where the housing density was varied, the land use percentages were modified to account for the differing amount of impervious area occupied by residences. For these simulations, the fraction of impervious area was divided into roads and residences (including buildings and driveways). The residential impervious area was modified by the change in housing density (doubled or halved) but the road area was kept the same. The fractions of land in turf and natural were modified to account for the change in impervious area. The land use fractions used in the simulations are summarized in Table 28.

	Jordan Acres Cutting			Village Green Cutting		
	Baseline (BL)	1/2 x BL	2 x BL	Baseline (BL)	1/2 x BL	2 x BL
Fraction of Land in "Natural" Conditions	0.12	0.13	0.10	0.35	0.39	0.28
Fraction of Land in Turf	0.66	0.72	0.55	0.41	0.45	0.33
Fraction of Land that is Impervious (residential)	0.13	0.07	0.26	0.15	0.08	0.30
Fraction of Land that is Impervious (roads)	0.09	0.09	0.09	0.09	0.09	0.09

Table 28: Relative amounts of turf, natural, and impervious areas used in the BURBS simulations for the Jordan Acres and Village Green cuttings.

The average number of persons per dwelling was 2.97 for Jordan Acres and 3.53 for Village Green. These values were determined by surveying a portion of the subdivision occupants. Approximately 50% of the homes in Jordan Acres and 35% of the homes in Village Green were surveyed.

The housing density for each of the scenarios was calculated using the total number of houses in each area of interest and dividing by the total area of the cutting. The value for Jordan Acres was 3.7 homes per hectare; Village Green was 2.9 homes per hectare. These values include roads, vacant lots, natural areas, and public lands—actual lot sizes are approximately 0.2 hectares.

The BURBS model considers the housing density to be equivalent to septic system drainfield density. It further assumes that the drainfields are evenly distributed throughout the subdivision. Observed in-field conditions indicate that some of the wells potentially get impacted by many drainfields, while others get impacted by few or none. To simulate this variability, the drainfield density was doubled in certain scenarios and halved in other scenarios. This variable was also set to zero in order to simulate the conditions for those wells not impacted by any septic systems.

Precipitation data are presented in section 1.4.1. As presented in Table 27, the calculated groundwater travel time beneath the Jordan Acres cutting ranges from 1.5 to 2.9 years; the travel time beneath Village Green could range from 4.8 to 9.0 years. The earliest groundwater samples analyzed for this study were obtained in the summer of 1987. Water in the wells that originated from within the subdivisions could have recharged any time during the groundwater flow period across the subdivision, thus based on an average groundwater flow velocity, the time period for Jordan Acres is 1985 to 1990 (78 cm) and Village Green is 1981 to 1990 (83 cm).

In order to demonstrate how fluctuations in precipitation can affect groundwater quality, the precipitation amount from relatively wet years and dry years were used in several simulations. The values that were chosen were the wettest and driest years over the time span used to determine average precipitation amounts (64 cm and 88 cm for Jordan Acres; 64 cm and 114 cm for Village Green).

As discussed in section 1.4.4, the water recharged to the groundwater was assumed to be equal to the total amount of precipitation less fifty-three centimeters of evapotranspiration. This relationship was assumed for both turf and natural land.

The evaporation from impervious surfaces was set at 10%, as recommended by the BURBS documentation.

The runoff from impervious surfaces going to recharge was not defined with a great deal of certainty due to the complexity of influencing factors. For example, rain that lands on rooftops is diverted to eaves troughs, where it is discharged to the ground in specific locations. Water that runs off of roads will tend to infiltrate through the soil close to the road. The water from impervious surfaces will be subject to some evapotranspiration; however, the localized area receiving the runoff water will quickly become saturated, thus facilitating water movement through the vadose zone and into the aquifer. Because the water-holding capacity of the soils in the subdivisions is very low and can quickly be exceeded by a precipitation event, the additional runoff from impervious surfaces is available to recharge the aquifer. For modelling purposes it was assumed that all the water not evaporating from impervious surfaces goes to groundwater recharge. Because this recharge water will have low nitrate-N levels, it will tend to lower average nitrate-N concentrations (by dilution) but not significantly effect nitrogen loading.

The volume of water used, per person, in the subdivisions was estimated after considering several sources. The US Environmental Protection Agency estimates that the per capita rate of water use is 170 liters (44 gallons) per day (USEPA, 1980). A survey to determine home water use in the City of Stevens Point (conducted for this study) yielded a result of 270 liters/person/day. This estimate may be high because of the uncertainty of the actual number of persons per household (assumed to be 3). It has also been suggested that homeowners with septic systems are more conscientious of their water use than those on city water and thus tend to be more conservative in terms of water use. A water meter installed on the well at a residence in the Jordan Acres subdivision indicates that the two adult occupants have consistently used approximately 190 liters (50 gallons) of water per day over a twelve month period. Data obtained from an investigation monitoring fifteen septic systems in nearby rural homes indicate that home water use is closer to 130 liters (35 gallons) per person per day. A value of 150 liters/person/day (40 gal./per./day) was used in the simulations.

The Environmental Task Force Lab - UW Stevens Point frequently tested for the nitrogen concentration in precipitation throughout the 1980s (unpublished data). The average nitrate-N concentration determined from these data was 0.25 mg/L.

Private well data from many of the homes in the subdivision were used to calculate an average nitrate-N concentration in the water used in the subdivisions. The average for Jordan Acres was 6.9 mg/L; the average for Village Green was 11.3 (see section 3.1.1.1 and 3.1.1.2).

The turf fertilization rate used in the model simulation was based on data obtained from subdivision homeowners. The survey results indicated that 74% all respondents used the amount specified by the manufacturer, 18% used more than was specified, 6% used no fertilizers, and 2% didn't read the bag (Mechenich, et. al., 1991). The survey also revealed that the overall fertilizer application rate was 1.6 times per year (1.8 times for users). A value of 3.3 kg/100 m^2 (1.6 lbs/1000 ft^2) was used in modelling both subdivisions (assuming a recommended application rate of 2.0 kg/100 m^2 applied 1.6 times per year).

Petrovic's (1990) review of relevant research revealed that although the amount of nitrogen leached from fertilized turf grass was highly variable, it was generally revealed that less than 5% was leached to groundwater. The exceptions were in areas where the fertilizers were applied in excessive amounts and/or the turf was over watered. The studies also showed that fertilizer leaching was greater in sandy soils than in finer-textured soils. The BURBs variable definitions cite a Long Island study that indicated up to fifty percent of lawn fertilizers used in sandy soils leached to groundwater. Because field data from lawn impacted groundwater were available (see section 3.1.3.2), the value for this variable could be calibrated using a range of values. For calibrating purposes it was assumed that all of the nitrogen in wastewater was removed by sewers.

Studies have shown that in well-aerated sandy soils, the amount of nitrogen in wastewater lost as a gas is negligible (Hantzsche and Finnemore, 1992; Walker, et al., 1973). This conclusion was supported by studies of private waste disposal systems in a nearby subdivision (Shaw and Turyk, 1993). The value for this variable used in the simulations was 0.

The subdivisions are unsewered, thus the wastewater fraction removed by sewer was set at 0 (except when used to calibrate the fertilizer leaching variable as discussed above).

The amount of nitrogen per person in wastewater has been fairly well documented. A value of 4.5 kg-N/person/year (10 lbs-N/person/year) was reported by Walker et.al. (1973). This value was also found for 15 septic systems in the

Stevens Point area (Shaw and Turyk) and is also specified by the EPA. Samples from a septic tank serving two adults in Jordan Acres contained 60 and 89 mg/L of total Kjeldahl nitrogen in the wastewater. The daily water use by this household was measured to be 190 l/person/day, thus the annual nitrogen loading rate is estimated to be 4.4 to 6.3 kg/person/year. A value of 4.5 kg/person/year was used for modelling purposes.

The nitrogen removal rate of natural land was set at 0.9 as recommended by BURBS documentation but it is negligible in model simulations because of the low nitrogen concentration in precipitation.

Values for the variables used in the BURBS model are summarized in Table 29.

Variable	Jordan Acres	Village Green
Fraction of land in turf - baseline	0.66 *	0.41 *
Low upgradient drainfield density	0.72	0.45
High upgradient drainfield density	0.55	0.33
Fraction of land that is impervious	0.22 *	0.24 *
Low upgradient drainfield density	0.16	0.17
High upgradient drainfield density	0.35	0.39
Average persons per dwelling	2.97 *	3.53 *
Housing density (#/hectare)	3.7 *	2.9 *
Low upgradient drainfield density	1.8	1.4
High upgradient drainfield density	7.4	5.7
Precipitations rate (cm/year)	78 *	83 *
Dry year	64	83
Wet year	88	114
Water recharged from turf (cm/year)	yearly precipitation - 53	
Water recharged from natural land (cm/year)	yearly precipitation - 53	
Evaporation from impervious surface (fraction)	0.1 *	0.1 *
Runoff from impervious recharged (fraction)	0.9 *	0.9 *
Home water use per person (liters/day)	150 *	150 *
Nitrogen concentrations in precipitation (mg/L)	0.25 *	0.25 *
Nitrogen concentration in water used (mg/L)	6.9 *	11.3 *
Turf fertilization rate (kg/100 m^2)	0.78 *	0.78 *
Fraction of nitrogen leached from turf (fraction)	varied from 0.05 to 0.40	
	0.25 *	0.25 *
Fraction of wastewater N lost as gas (fraction)	0 *	0 *
Wastewater fraction removed by sewer (fraction)	0 *	0 *
Fertilizer-leaching simulations	1	1
Nitrogen per person in wastewater (kg/year)	4.5 *	4.5 *
Nitrogen removal rate of natural land (fraction)	0.9 *	0.9 *

* Used for baseline model run

Table 29: Values for the variables used in the BURBs simulation for Jordan Acres and Village Green.

3.4.2 Simulation Results

The results of the various BURBS simulations are summarized in Table 30. Complete variable definitions and simulation results are presented in Appendix D.

The predicted nitrate concentration for the baseline values are higher for Jordan Acres (17.2 mg/L) than for Village Green (13.7 mg/L). This is because BURBs predicts more nitrogen leached (67 kg/acre/yr vs 61 kg/acre/yr) and less water recharged (39 cm/yr vs 45 cm/yr) for Jordan Acres. Jordan Acres has a higher rate of nitrogen loading because there is a greater amount of area in turf (66% vs 41%) and higher housing density (3.7 houses/hectare vs 2.9 houses/hectare) in Jordan Acres even though Village Green has a greater number of persons per dwelling (3.5 vs 3.0) and a higher nitrate-N concentration in the water used (11.3 mg/L vs 6.9 mg/L). Village Green has a larger amount of water recharged primarily because of the greater amount of precipitation used in the simulations (83 cm vs 78 cm) but also because of the greater amount of impervious area (24% vs 22%).

The fertilizer leaching results for Village Green are 30% to 35% lower than those for Jordan Acres. This is because Jordan Acres has a higher percentage of its land use as turf, whereas Village Green has more natural and impervious areas. Recharge from the non-turf areas acts as a diluting influence, thereby decreasing the overall nitrate-N concentrations.

Study Area and Simulation Conditions	Average NO_3 in Recharge (mg/L)	Nitrogen Leached (kg/ha/yr)	Water Recharged (cm/year)
Jordan Acres Cutting			
Baseline Values	17.2	67	39
High Upgradient Drainfield Density (7.4 dwellings/hectare)	23.7	120	50
Low Upgradient Drainfield Density (1.8 dwellings/hectare)	12.3	42	34
Wet Year (88 centimeters of Precipitation)	13.8	68	49
Dry Year (64 centimeters of Precipitation)	26.1	67	26
No Drainfield Impacts and:			
- 5% of fertilizer leaches	0.9	3.0	33
- 10% of fertilizer leaches	1.7	5.7	33
- 20% of fertilizer leaches	3.3	11	33
- 25% of fertilizer leaches	4.1	14	33
- 30% of fertilizer leaches	4.9	16	33
- 40% of fertilizer leaches	6.5	22	33
Village Green Cutting			
Baseline Values	13.7	61	45
High Upgradient Drainfield Density (5.7 dwellings/hectare)	20.0	110	57
Low Upgradient Drainfield Density (1.4 dwellings/hectare)	9.1	35	39
Wet Year (114 centimeters of Precipitation)	8.3	61	74
Dry Year (64 centimeters of Precipitation)	23.2	61	26
No Drainfield Impacts and:			
- 5% of fertilizer leaches	0.6	2.1	39
- 10% of fertilizer leaches	1.0	3.8	39
- 20% of fertilizer leaches	1.8	7.1	39
- 25% of fertilizer leaches	2.2	8.7	39
- 30% of fertilizer leaches	2.7	10	39
- 40% of fertilizer leaches	3.5	14	39

Table 30: BURBs simulation results for the Jordan Acres and Village Green cuttings.

None of the wells in Village Green had average nitrate-N concentrations lower than 5.3 mg/L (at S3-22). This may be because all the wells may be impacted by drainfields to a certain degree; however, the higher nitrate-N concentrations may be due to mixing with the high concentrations of nitrate-N in the upgradient groundwater. Also, because the lawns are irrigated from wells within the subdivision, water with relatively high nitrate-N concentrations is brought up from at depth and recharged at the water table (i.e., lawn fertilizers are not the only source of nitrate-N from turf areas).

Results of the Jordan Acres BURBS simulations that compare fertilizer leaching rates were evaluated to determine the amount of leaching that occurs within the subdivisions. Because the average nitrate-N concentration of wells monitoring lawn areas was 4.3 ppm (see section 3.1.3.2 above), the leaching rate for the baseline value used in the simulations was set at 25%. Results from Jordan Acres were used because most of the wells used to monitor lawn impacts were in that subdivision. The 4.3 mg/L concentration is also close to average for Park Ridge, a sewered subdivision adjacent to Stevens Point (ETF unpublished data). For Jordan Acres, the 25% leaching rate accounts for about 20% of the total nitrogen budget for the baseline simulation; for Village Green the 25% rate accounts for 13%.

Precipitation amounts greatly affect groundwater nitrogen concentrations. In wet years, there will tend to be more water available to dilute the nitrogen inputs; in dry years, less dilution will occur and nitrate-N levels will be higher. Precipitation extremes may have a short-term impact on groundwater quality and account for some of the variability found in shallow wells. Precipitation extremes could have a noticeable effect on groundwater quality if the conditions persist for several years.

The results of varying the drainfield density, in addition to the results from simulations that assumed no drainfield impacts, supports the observations and conclusion of several other authors (Yates, 1985; Perkins, 1984) that septic system drainfields are the primary cause of elevated nitrate-N concentrations in the groundwater beneath unsewered subdivisions. Note that in Jordan Acres, even at a relatively low drainfield density (1.8 homes/hectare) BURBS predicts nitrate-N concentrations in excess of the enforcement standard for nitrate-N of 10 ppm. This suggests that in subdivisions with sandy aquifers, the average area allotted per house should not be less than 0.5 hectare (this area includes roads, parks, etc.). In Village Green, the low drainfield density simulation yielded a result below the 10 ppm standard. The area for this simulation was 0.7 hectares. Simulations were run to determine the housing density that would be needed in Village Green and Jordan Acres to achieve a 10 mg/L nitrate-N concentration in the recharge. The housing densities are 1.7 dwellings/hectare in Village Green and 1.1 dwellings/hectare in Jordan Acres.

Several simulations were run varying housing density (and land use fractions) for each subdivision. Figure 34 shows the relationship between housing density and simulated nitrate-N concentrations in groundwater recharge for Jordan Acres and Village Green subdivisions. The primary reason for the differences between the two subdivisions is that precipitation amounts used for the two subdivisions differed by

5.1 cm, which resulted in less recharge and therefore less dilution in the Jordan Acres simulations. The higher percentage of lawn area in Jordan Acres resulted in more fertilizer leaching (hence greater nitrogen loading to the aquifer). This was largely offset by a slightly higher number of people per household in Village Green (which also increases nitrogen loading).

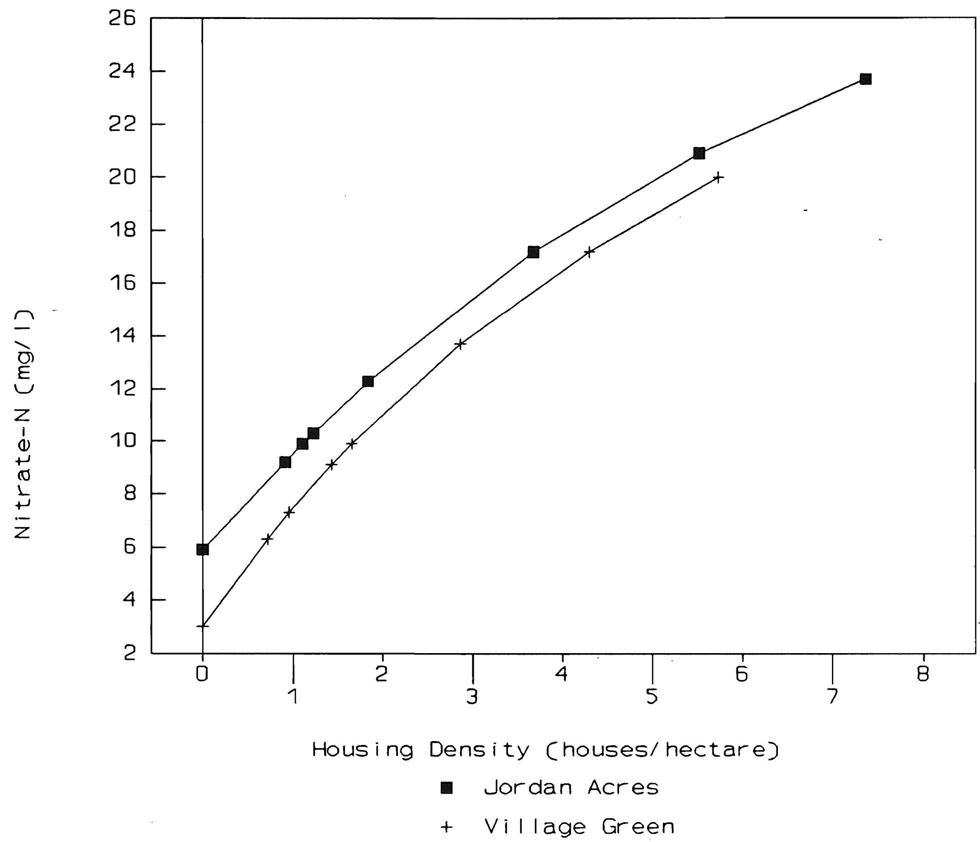

Figure 34: Graph showing relationship between housing density and simulated nitrate-N concentration using the BURBS model for Jordan Acres and Village Green subdivisions.

3.4.2.1 Simulation for Heavier-Textured Soils

In addition to the simulations run for Village Green and Jordan Acres, several runs were made changing the routing of runoff water and reducing groundwater recharge to 10 cm/yr, which is likely to be closer to the statewide average for groundwater recharge. The simulations are considered to be indicative of what might be expected in areas of heavier textured soils (loams as opposed to sands) and/or greater slope. The Village Green values were used for the variables except annual recharge from vegetated areas was reduced from 30.0 cm (11.8 in.) to 10.2 cm (4.0 in.), and recharge from runoff was reduced from 90 percent to 12 percent. Also, because the fraction of fertilizer that leaches from fine-textured soils tends to be less than in sandy soils (see Petrovic, 1990) the value for this variable was reduced from 0.25 to 0.05. The results of the simulations varying housing density in areas with heavier soils are presented graphically in Figure 35; model outputs are included in Appendix D. The baseline simulation resulted in a nitrate-N concentration of 34.9 mg/L, compared to 13.7 mg/L for the Village Green baseline values. Lot size to achieve a nitrate-N concentration of 10 mg/L increased from 0.6 hectares/home to 2.0 hectares/home.

The results of these simulations suggest that subdivision designs that maximize local groundwater recharge will provide maximum dilution of nitrogen inputs from septic systems (and lawns). These scenarios indicate that fertilizer leaching in sandy soil areas, while a significant part of the nitrogen budget, is effectively diluted by high recharge amounts. Decreased recharge with a similar percent of fertilizer that leaches will result in much higher nitrate-N concentrations reaching groundwater from lawns. More research is needed to evaluate nitrogen losses from lawns and also groundwater recharge amounts for different soil types in Wisconsin.

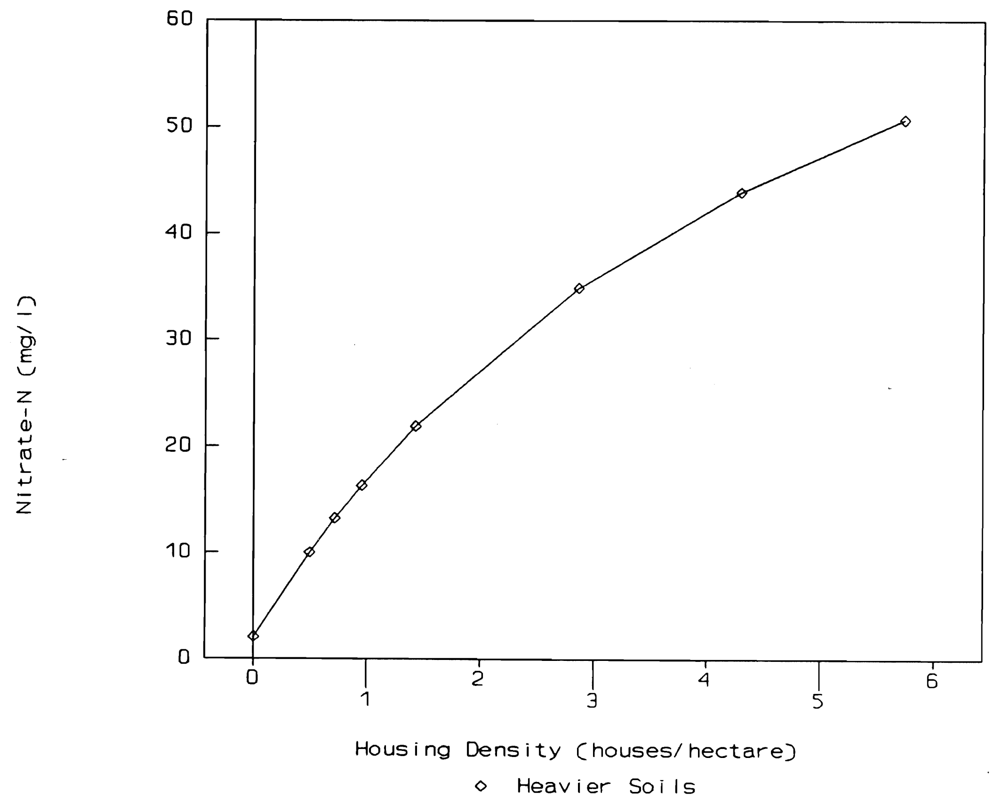

Figure 35: Graph showing the relationship between housing density and simulated nitrate-N concentration in heavy soils using values for Village Green subdivision.

4.0 Comparison of the Results of the Nitrogen and Water Budgets Determined by Two Separate Methods

The nitrogen and water budget results determined using the BURBS computer program and the results based on field data are presented in Table 31. Three scenarios of the field data are presented for comparing to the results of the baseline simulations of the BURBS model.

Budget Results	Average NO_3 in Recharge (mg/L)	Nitrogen Leached (kg/yr)	Water Recharged (m^3/yr)
Jordan Acres			
BURBS: Baseline values.	17.2	440	25,000
Field data: Low groundwater flow velocity	9.0	210	23,000
Field data: Medium groundwater flow velocity	9.0	300	33,000
Field data: High groundwater flow velocity	9.0	390	43,000
Water recharged based on hydrologic data			21,000
Water recharged assuming no impervious areas			16,000
Village Green			
BURBS: Baseline values	13.7	930	68,000
Field data: Low groundwater flow velocity	13	530	39,000
Field data: Medium groundwater flow velocity	13	770	57,000
Field data: High groundwater flow velocity	13	1000	74,000
Water recharged based on hydrologic data			60,000
Water recharged assuming no impervious areas			46,000

Table 31: Nitrogen and water budget results for Jordan Acres and Village Green cuttings. Results were calculated using values from both the BURBs computer program and field data.

For Jordan Acres, there is a dramatic difference between the nitrate-N concentration predicted by the BURBS model (17.2 mg/L) and the results determined from the field data (9.0 mg/L). Which value is likely to be closer to the actual concentration is a matter of debate. Although the concentration predicted by the BURBs model is higher than might be expected, the value based on the field data is considered to be low. Most of the multiport sampling ports were installed below the zone of subdivision-impacted groundwater (see section 3.3). The ports that were in this zone appeared to miss most of the septic plumes. Because of this, the multiport monitoring network in Jordan Acres is not considered to be adequate for assessing the overall groundwater quality originating from the subdivision. Note that the amount of water recharged predicted by the BURBs model and that calculated using hydrologic data (25,000 and 21,000 m^3/yr respectively) are similar to the field data results with a low groundwater flow velocity (23,000 m^3/yr). The results from the field data assuming higher groundwater flow velocities are considerably higher (33,000 to 43,000 m^3/yr). This again suggests that the effective aquifer thickness (3.4 m) used in the field data calculations was too large.

For Village Green, the BURBs and field data nitrate-N concentrations are virtually the same (13.7 mg/L and 13.6 mg/L respectively). The water recharged calculated by each of the three models (BURBS, field data, and hydrologic data) are also in the same range (see Table 31). This is considered to indicate that each of the models are reasonably accurate for this subdivision.

The results based on the field data are likely to be more accurate for Village

Green than for Jordan Acres. This is because a greater number of ports in the Village Green monitoring network are in the zone of subdivision-impacted groundwater (thus there are a greater number of data points) and because the downgradient wells are distributed more uniformly perpendicular to groundwater flow (thereby being more likely intercept areas of differing subdivision impacts). The monitoring network in Village Green is considered to accurately characterize the water quality originating from the subdivision.

5.0 Conclusions and Recommendations

Some of the conclusions that can be drawn from this study are discussed below.

1. Groundwater originating from unsewered subdivisions can have nitrate-N concentrations in excess of the 10 mg/L drinking water standard established by the Wisconsin Department of Natural Resources.

2. Residential lawns and septic system drainfields are point sources of pollution to groundwater, with septic systems generally having a greater impact than lawns.

3. Septic systems appear to be the primary cause of elevated nitrate-N concentrations in unsewered subdivisions (assuming that the groundwater has a low concentration of nitrate-N as it enters the subdivision). Nitrate-N concentrations in excess of 48 mg/L were detected in shallow groundwater monitoring wells installed downgradient of septic system drainfields. Relatively high concentrations of several other inorganic chemicals were also detected in the wells including chloride (>133 mg/L), sodium (>108 mg/L), and phosphate (>6.9 mg/L). Elevated concentrations were detected in wells installed 5 to 50 feet downgradient of drainfields.

4. The data from a well installed downgradient of a well-managed lawn suggest that residential lawn practices can cause elevated nitrate-N concentrations (<10 mg/L) in the groundwater downgradient of the lawn area.

5. Phosphate was detected in several of the wells throughout the subdivisions. In most cases the phosphate was obviously due to septic system drainfields. This indicates that sandy soil can become saturated with phosphorus originating from septic system drainfields within 10 to 20 years and result in significant leaching of this chemical. Concentrations of PO_4-P ranging from 1 to 11 mg/L were found in the groundwater downgradient of four septic systems. Phosphate-P concentrations between 0.02 and 0.2 mg/L were found at depths of up to 5 meters below the water table at the downgradient end of one of the unsewered subdivisions.

6. Fluorescing compounds originating from on-site waste disposal systems can be reliable indicators of septic system impacts on groundwater quality. Wells installed immediately upgradient and downgradient of septic system drainfields showed three- to twelve-fold increases of relative fluorescence in the downgradient wells.

7. The concentrations of nitrate-N, chloride, and sodium in the groundwater of the Central Wisconsin sand plain that is mostly unaffected by humans are very low (near 1 mg/L).

8. Inorganic chemistry data from private water-supply wells can give a reasonably accurate view of the groundwater quality impacts from subdivision sources, provided the wells are installed in the portion of the aquifer where the impacts occur. However, because these wells will tend to be installed so as to yield water with low contaminant concentrations, the data are likely to be lower than the actual average chemistry concentrations. Also, considering the variability of the inorganic chemistry observed in water samples obtained from the monitoring wells installed in the study subdivisions, the number of water-supply wells in a subdivision is likely to be insufficient to accurately demonstrate the spacial variability of the groundwater quality.

9. Agricultural practices (e.g., frequent fertilization and irrigation) on the Central Wisconsin sand plain can cause groundwater nitrate-N concentrations to exceed drinking water standards. Nitrate-N concentrations over 30 ppm were frequently detected in wells downgradient of several miles of center-pivot irrigation fields. The average nitrate-N concentration from wells monitoring the groundwater from these agricultural sources was over 20 ppm, which is more than twice the safe drinking water standard.

10. The BURBS nitrogen and water budget program is a useful tool for predicting the average nitrate-N concentration in the groundwater recharged from residential areas. However, the values used as input variables must be defined with a high degree of certainty in order for the model to yield accurate results. Also, the results give no indication of the nitrate-N concentration at any particular location in the aquifer.

11. Nitrogen and water mass balance models can yield accurate estimates of the average nitrate-N concentration in the groundwater recharged from subdivision sources. However, they do not yield information pertaining to the probability of any particular well being impacted. Assuming low background nitrate-N concentrations, the primary factor governing the likelihood of a water-supply well exceeding drinking water standards for this compound is the well's position relative to nearby drainfields. Placement and screen depth of water-supply wells in subdivisions need to be carefully considered relative to septic system location and groundwater flow to prevent unwanted recycling of wastewater.

12. Housing densities of less than 1.1 to 1.6 homes/hectare (depending on specific conditions) would be needed to keep the average nitrate-N concentration of the groundwater recharge originating from unsewered subdivisions below the 10 mg/L drinking water standard in areas of sandy soils in Central Wisconsin. Lower housing densities would be required in areas with less groundwater recharge. Note that the values are for average housing densities and not actual lot sizes. Average lot sizes would be considerably smaller.

13. Due to the recharge of most of the runoff water from roads and roofs, groundwater recharge from within a subdivision on sandy soils is considerably greater than from adjacent field and woods. This results in greater dilution of septic system (and other) contaminants. Subdivision designs that maximize local groundwater recharge will tend to provide maximum dilution of nitrogen inputs from the subdivision (e.g., septic systems and lawns). More research is needed to better evaluated the amount of groundwater recharge from subdivision areas.

14. It is impractical to propose a set number of monitoring wells required to adequately characterize the groundwater quality beneath unsewered subdivisions. Three multiport wells installed approximately 15 meters apart and sampling the aquifer at 1.5 meter intervals appeared to provide an accurate view of the groundwater quality originating from approximately 260 meters of upgradient subdivision land use. A monitoring network installed to assess the impact of a land use on groundwater quality must be carefully planned so as to obtain samples from the impacted zone. Factors to be considered include groundwater flow direction, distribution of impacting areas, and depth of impact.

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