PORT EDWARDS
GROUNDWATER PRIORITY WATERSHED

GROUNDWATER RESOURCE AND AGRICULTURAL PRACTICE EVALUATION

prepared by

George J. Kraft, Director
Will Stites, Associate Scientist
David Mechenich, Data Management Specialist
Jennifer Balma, Research Assistant

Central Wisconsin Groundwater Center
Cooperative Extension Service
College of Natural Resources, University of Wisconsin – Stevens Point

June 28, 1995

ACKNOWLEDGMENTS

This project owes great thanks to the vegetable growers of the Town of Port Edwards, particularly B&D Farms and RDO Farms. Randy Fleischauer, Ron Kohlman, Sam Ourada, and Chuck Von Einem adjusted their farming operations to help us get into the fields and collect samples. They provided places for us to work and keep equipment, and countless times they helped us solve equipment failures, freezeups, getting stuck in the mud and the snow, and all the other problems of fieldwork. RDO Farms and B&D Farms are exemplary models of growers working to find solutions to groundwater problems. There couldn't have been a Port Edwards Groundwater Priority Watershed project without their help and support. Thanks also to Jasper Eglund for helping us out of some difficulties.

We also want to thank the many other citizens of the Town of Port Edwards who graciously agreed to let us install monitoring wells on their property, and visit the wells for sampling and water level measurements, and those who helped out in various other ways.

Diana Berge, who worked on public-education aspects of the Port Edwards Groundwater Priority Watershed project, contributed ideas and enthusiastic help. Jo Carlson and Jeanne Commette provided excellent help at the Groundwater Center. Wood County Emergency Government staff provided our storage shed and help with burying, then finding and digging up wells, and the Land Conservation Department helped out continually with miscellaneous needs. Our colleagues at the University of Wisconsin (both Madison and Stevens Point) – Jean Bahr, Larry Binning, Walter Hall, Sam Kung, Fred Madison, Byron Shaw, Jeff Wyman – were helpful in collecting core samples, designing BMPs, well drilling, and with advice.

Scott Provost, who worked as a student then as a technician early in the project, was outstanding in tenacity, intelligence, and practicality. Our other student employees and technicians, Heather Barlow, John Bhend, Russ Biebl, Kirk Hackbarth, Cheryl Janowiak, Dave Johnson, Steve Osesek, Joe Pasqualucci, Scott Rebholz, Brian Weigel, Bob Wiza, and many others who worked briefly on special projects, deserve praise for their dedication and effort.

CONTENTS

List of Figures iv

List of Tables vi

List of Appendices vii

List of Archives viii

Executive Summary ix

Chapter 1. Introduction 1-1

Chapter 2. Geology 2-1

Chapter 3. Hydrogeology 3-1

Chapter 4. Groundwater Flow and Particle Tracking Modeling 4-1

Chapter 5. Groundwater Quality Conditions in the Port Edwards Groundwater Priority Watershed 5-1

Chapter 6. Agricultural Practices and Groundwater Conditions Under Four Irrigated Vegetable Fields 6-1

Chapter 7. Conclusions 7-1

FIGURES

1.1 Location of Study Area 1-4

1.2 Study Area Physical Features 1-5

2.1 Study Area Physical Features 2-2

2.2 Locations of Borehole Sites 2-6

2.3 Locations of Borehole Sites in Sections 5 and 6 2-7

2.4 Aquifer-bottom Elevation Contours 2-8

2.5 Contour Map of Depth to New Rome Member 2-18

2.6 Elevation Contours of New Rome Surface 2-19

2.7 Contour Map of Thickness of New Rome Member 2-20

2.8 Geologic Cross-sections ..... 2-21 – 2-27

2.9 Contour Map of Elevation of Second Silty Lacustrine Unit 2-28

2.10 Contour Map of Aquifer Saturated Thickness Above New Rome Member 2-29

2.11 Depth profiles of pH, loss on ignition, and organic matter content, Kimball Ave. and Creamery Rd. borehole 2-31

2.12 Loss on ignition vs. content of particles smaller than sand, Kimball Ave. and Creamery Rd. borehole 2-32

2.13 Depth profiles of phosphorus and potassium, Kimball Ave. and Creamery Rd. borehole 2-33

2.14 Particle-size classes as determined by sieving, Kimball Ave. and Creamery Rd. borehole ..... 2-34 – 2-36

2.15 Particle-size class variations with depth, Kimball Ave. and Creamery Rd. borehole 2-37

2.16 Cumulative particle-size plots for four samples from Kimball Ave. and Creamery Rd. borehole 2-39

2.17 Particle size distributions from four depth intervals in an irrigated field ..... 2-41 – 2-42

3.1 Monitoring well network 3-7

3.2 K, as determined from specific capacity, variation with pumping rate 3-12

3.3 Groundwater flow in the Port Edwards Groundwater Priority Watershed 3-15

3.4 Water-table fluctuations and monthly precipitation in the Port Edwards Groundwater Priority Watershed, September 1992 through March 1994 3-16

3.5 Water table configuration on seven dates ..... 3-17 – 3-23

3.6 Map of where gradients were observed 3-24

3.7 Water budget, Port Edwards Groundwater Priority Watershed 3-26

4.1 Finite-difference grid for the three-layer and one-layer models ..... 4-2 – 4-4

4.2 Areas where bedrock elevations were changed to improve model calibration 4-7

4.3 Area where hydraulic conductivity was changed to improve model calibration 4-8

4.4 One-layer calibrated vs. measured heads 4-9

4.5 Three-layer calibrated vs. measured heads 4-10

4.6 Flow lines generated by PATH3D 4-13

4.7 Flow lines determined from equipotential map 4-14

4.8 Advective solute plume migration from all irrigated fields ..... 4-15 – 4-18

4.9 Location of vertical profiles from Fields 1 and 2 4-19

4.10 Vertical cross-section of plume beneath Field 1 4-20

4.11 Vertical cross-section of plume beneath Field 2 4-21

5.1 Location of monitoring wells 5-3

5.2 pH at three depth intervals ..... 5-8 – 5-10

5.3 Highest NO3-N concentration at monitoring well locations with respect to plumes originating from irrigated fields 5-11

5.4 NO3-N concentrations at three depths ..... 5-12 – 5-14

5.5 Location of sites where wells were sampled for pesticides 5-16

5.6 Analyte ranges and means from agriculturally impacted and non-impacted groundwater at two depths ..... 5-18 – 5-19

5.7 Location of groundwater flowpath cross-sections ..... 5-20 – 5-21

5.8 Flowpath cross-sections ..... 5-23 – 5-28

6.1 Location of the four intensive-study fields 6-2

6.2 Intensive-study fields in Sections 5 and 6, showing multilevel piezometer sites and approximate groundwater flow direction 6-3

6.3 Rainfall and irrigation, 1992 6-7

6.4 Rainfall and irrigation, 1993 6-8

6.5 Construction of multilevel piezometers (schematic), showing port numbering order 6-9

6.6 Contours of chloride and nitrate and profiles of atrazine concentrations at MLPs throughout the sampling period ..... 6-17 – 6-19

6.7 Hypothetical contour plot of chloride concentrations at a multilevel piezometer affected by annual applications of KCl fertilizer 6-27

6.8 Contours of chloride concentration at multilevel piezometers throughout the sampling period ..... 6-29 - 6-32

6.9 Nitrogen fertilizer applications to Field 2 in 1992 and 1993 6-46

6.10 Effect of crop mature root depth on simulated recharge calculated by the PRZM model 6-48

6.11 Effect of crop mature root depth on simulated nitrate leaching calculated by the PRZM model 6-50

6.12 Effect of soil water-holding capacity on simulated recharge calculated by the PRZM model 6-51

6.13 Effect of soil water-holding capacity on simulated nitrate leaching calculated by the PRZM model 6-53

TABLES

1.1 Monthly total precipitation (inches), 1988–1993 1-7

1.2 Growing season length (days above freezing), 1988–1993 1-8

1.3 Degree days above 65°F, 1988–1993 1-8

2.1 Summary of geological materials found in boreholes in the Port Edwards Groundwater Priority Watershed 2-12

2.2 Estimated d_x and nonuniformity indices of samples from the Creamery Road and Kimball Avenue site 2-38

2.3 Median particle diameters of samples taken from irrigated fields 2-40

2.4 Median particle diameters of samples from irrigated fields 2-43

2.5 Comparison of sub-sand-size particle results from sieve analyses with hydrometer analyses for Ap horizon samples from irrigated fields 2-43

2.6 Analysis of variance for median praticle diameter in different depth intervals in irrigated fields 2-43

2.7 Porosity of C horizon samples collected with the vibra-core method 2-45

3.1 Hydraulic conductivities measured in the Wisconsin Central Sand Plain 3-2

3.2 Wisconsin Central Sand Plain hydraulic conductivities as estimated by grain-size analysis and permeameter measurements 3-3

3.3 Slug test inputs and results 3-11

3.4 Statistics for hydraulic conductivity estimated from specific capacity capacity 3-11

3.5 Statistics for hydraulic conductivity estimated from wells pumped at different rates 3-13

5.1 Analytical methods and holding times for inorganic and organic analyses 5-4

5.2 Summary of composite inorganic monitoring data 5-5

5.3 Summary of organic monitoring data from PEGPW monitoring network 5-15

5.4 Summary of pesticide detects sampled at multilevel piezometers located under study fields 5-17

6.1 Summary of 1992 field practices 6-4

6.2 Summary of 1993 field practices 6-5

6.3 Dates of sampling rounds and analyses performed in addition to routine panel 6-11

6.4 Average results from upgradient MLPs of fields 1, 2 and 3 vs. average of all in-field MLPs 6-12

6.5 Mean concentrations of NO_3-N throughout sampling period, weighted for MLP screen length. 6-13

6.6 Summary of pesticide detects for 256 ports sampled, fall 1992, spring 1993, and late fall – early winter 1993 6-14

6.7 Pesticide detections and average concentrations in MLPs, November-December 1993 ... 6-15

6.8 Fertilizer N inputs to vegetable fields, 1992 and 1993 6-20

6.9 Estimated harvested N from the four vegetable fields in 1992 and 1993 6-21

6.10 Inputs, outputs, and potentially leachable N from four vegetable fields, 1992 and 1993 .. 6-24

6.11 Mean concentrations of NO_3-N throughout sampling period, weighted for MLP screen length, and simple NO_3-N loading estimate for 1992 and 1993 recharge. 6-24

6.12 Mass of NO3-N at Field 2 MLP 1 for the 1992 crop year by the method of N loading through a water year 6-34

6.13 Mass of NO3-N at Field 2 MLP 2 for the 1992 crop year by the method of N loading through a water year 6-34

6.14 Mass of NO3-N at Field 2 MLP 3 for the 1992 crop year by the method of N loading through a water year 6-35

6.15 Mass of NO3-N at Field 2 MLP 4 for the 1992 crop year by the method of N loading through a water year 6-35

6.16 Mass of NO3-N at Field 2 MLP 5 for the 1992 crop year by the method of N loading through a water year 6-36

6.17 Mass of NO3-N at Field 2 MLP 6 for the 1992 crop year by the method of N loading through a water year 6-36

6.18 Summary of 1992 N loading by the method of measuring N loading through a water year 6-37

6.19 Mass of NO3-N at Field 2 MLP 1 for the 1993 crop year by the method of N loading through a water year 6-38

6.20 Mass of NO3-N at Field 2 MLP 2 for the 1993 crop year by the method of N loading through a water year 6-39

6.21 Mass of NO3-N at Field 2 MLP 3 for the 1993 crop year by the method of N loading through a water year 6-39

6.22 Mass of NO3-N at Field 2 MLP 4 for the 1993 crop year by the method of N loading through a water year 6-40

6.23 Mass of NO3-N at Field 2 MLP 5 for the 1993 crop year by the method of N loading through a water year 6-40

6.24 Mass of NO3-N at Field 2 MLP 5 for the 1993 crop year by the method of N loading through a water year 6-41

6.25. Summary of 1993 N loading by the method of measuring N loading through a water year 6-41

6.26 Comparison of water balance as estimated by PRZM and by the WISP + Ritchie method 6-44

6.27 Nitrogen loading from PRZM simulations with selected values for mature rooting depths and soil water-holding capacity 6-47

6.28 Field moisture capacity and wilting point values in A, B, and C horizons used in sensitivity analysis, with recharge amounts calculated by PRZM 6-49

6.29 Sensitivity of NO_3 loading calculation to soil water-holding capacity 6-52

6.30 Nitrate-N loading in irrigated fields in the Port Edwards Groundwater Priority Watershed as determined by four methods 6-56

6.31 N loading to groundwater as estimated by three methods for Field 2 6-57

APPENDICES

1 Details of vibra-core sampling A1-1

2 Record of borehole logs A2-1

3 Location and construction of boreholes A3-1

4 Field records from growers A4-1

5 Pesticide analytes and detection/ quantitation limits A5-1

6 Pesticide detections in multilevel piezometers within irrigated fields A6-1

4 References References-1

ARCHIVES

Archives are voluminous collections of data that are maintained at the Central Wisconsin Groundwater Center. They are available for inspection there, or information from them can be furnished upon request.

1 Database containing information on Central Wisconsin Groundwater Center boreholes and wells, and private wells, in the project area.

2 Database containing monthly hydraulic head measurements from Central Wisconsin Groundwater Center wells in the project area.

EXECUTIVE SUMMARY

The objectives of this evaluation with respect to the Port Edwards Groundwater Priority Watershed Project were to

- describe its hydrogeology
- measure the prevailing groundwater quality and assess the groundwater impacts of major land uses
- measure and compare nitrate loading to groundwater under conventional practices and best management practices (BMPs)
- determine which, if any, pesticides leach under conventional practices and BMPs
- evaluate the utility of a leaching model to simulate nitrate loading to groundwater.

These objectives are intended to help answer the following questions regarding groundwater quality sustainability and land-uses in the Wisconsin Central Sand Plain (WCSP):

- what quality of groundwater results from various land uses and land-management strategies in the WCSP?
- does existing BMP agriculture allow profitable farming and still adequately protect groundwater?
- can BMPs be improved to enhance groundwater protection?
- if the BMP approach is not the answer, what alternative strategies might result in acceptable quality groundwater?

GEOLOGY, SOILS, AND HYDROGEOLOGY OF THE PRIORITY WATERSHED

Geologic setting

The study area is generally flat, with a regional southwestward slope of 0.1%. Surface elevation varies from about 970 to 1000 ft above mean sea level.

Five major geologic units are present, but only the Pleistocene unit is important from a groundwater standpoint. The Pleistocene unit belongs to the Big Flats Formation and is generally 60-100 ft thick. It consists of well-sorted, low-organic-matter sands with one or more interbedded silty and clayey subunits. The uppermost silty/clay subunit is The New Rome Member of the Big Flats Formation. Within the priority watershed, the New Rome Member is typically 2-5 ft thick and is found 13 to 39 ft below the surface. Other silty subunits have not been named.

Soils

Soils in the watershed are predominantly Plainfield sand and loamy sand, with smaller amounts of Friendship, Meehan, and Newson loamy sands. Except for the top 1 ft, they contain over 95% well-sorted sand, and the median grain size is medium (250-500 μm) to coarse (500-1000 μm). The 0-1 ft depth interval contained 4-9% silt + clay. Organic-matter content below the top layer of soil was low, averaging 0.05%.

Hydrogeology

Aquifer boundaries

The aquifer base coincides with the base of Pleistocene deposits. The escarpment at the east edge of the watershed drops to wetlands, creeks, or the backwaters of the Wisconsin River which form the eastern boundary of the flow system. The boundaries of the flow system extend well beyond the study area to the north and west.

Hydrogeologic parameters

Hydraulic conductivity (K) was determined by slug tests and by analyzing specific-capacity test data from well records. Slug-test Ks had a geometric mean of 4.0×10^{-4} m s^{-1}. The average K estimated from specific capacity data was 5.2×10^{-4} m s^{-1}, but Ks estimated from wells pumping at greater than 3800 L min^{-1} were substantially higher, 8.6×10^{-4} m s^{-1}. Because this value (8.6×10^{-4} m s^{-1}) reflects the largest available scale of measurement, it should be most characteristic of basin-scale K. This value is close to the 9.0×10^{-4} m s^{-1} found in the model calibration by Bradbury et al. (1992) for the nearby Buena Vista Basin.

Hydraulic conductivities calculated by the various methods are not entirely comparable because K increases with the scale of measurement. For this reason, we selected 9×10^{-4} m s^{-1} as most representative of basin-scale K.

We measured an aquifer porosity of 40%.

Hydraulic heads were measured monthly and were used to infer horizontal and vertical gradients. Groundwater flows generally from northwest or west toward the southeast or east. The water table fluctuated by about 1.2 m during the study period, rising sharply during the spring of 1993 due to record precipitation, but receding again by March 1994. Flow patterns and horizontal gradients were generally consistent throughout study period, even with fluctuations in the water table.

Generally, vertical gradients were small, less than 0.04, and were related to the presence of a silty lacustrine layer between screens.

Water budget

Annual precipitation in the WCSP averages 80 cm. Under forest, grassland, and non-irrigated cropland, evapotranspiration is about 53 cm, leaving about 25 cm of precipitation as groundwater recharge. Evapotranspiration is greater on irrigated cropland, decreasing net groundwater recharge to 15 cm.

We calculated a steady-state groundwater budget for the watershed which consists of 1.9×10^4 m^3 d^{-1} of recharge, 2.2×10^4 m^3 d^{-1} of underflow entering from the west and north, and 4.2×10^4 m^3 d^{-1} of underflow leaving through the east and south.

GROUNDWATER FLOW AND PARTICLE TRACKING MODELS

Groundwater flow models were constructed using the computer code MODFLOW. Both one-layer and three-layer models were constructed. Calibrated heads were within 2 ft of measured heads at target locations, and most were within 1 ft. Overall, the model appeared to be a good representation of field conditions.

Particle tracking models were constructed using PATH3D to plot flow lines, simulate advective contaminant transport, and help interpret monitoring data. The one-layer MODFLOW simulation was used as the basis for PATH3D calculations of flow lines in plan view. PATH3D's groundwater flow lines closely matched flow lines drawn from equipotential maps. This tends to verify the accuracy of the flow and particle-tracking models. Results of particle tracking modeling showed approximately 54% of the watershed is underlain by plumes from irrigated agricultural fields.

A vertical profile of advective transport was traced with PATH3Dfor two fields in the northern part of the watershed, using the three-layer MODFLOW model as input. The profiles showed particle paths converging downgradient of fields. Predicted vertical particle paths were highly dependent upon New Rome vertical hydraulic conductivity, but this feature may be artificial.

GROUNDWATER QUALITY

Inorganic chemistry

The groundwater chemistry of the watershed is the Ca-Mg-HCO_3 type, with Cl, Na, SO_4, and NO_3 present as other major ions. The pH ranged from 4.9 to 8.7, and alkalinity averaged 26 mg L^{-1} $CaCO_3$. Outside of fields, Cl concentrations ranged from < 1.0 to 59 mg L^{-1} and NO_3-N from <0.2 to 26.4 mg L^{-1}. Beneath fields, the peak Cl concentration reached 119 mg L^{-1} and NO_3-N, 81 mg L^{-1}. In general, the groundwater was similar to that of the Wisconsin Central Sand Plain as a whole, but with lower concentrations of dissolved carbonates. In two instances, low NO_3-N concentrations at depth apparently were due to denitrification. However, denitrification did not seem to be a prevalent mechanism. Both pH and alkalinity

tended to increase with depth, from north to south, and from west to east. Nitrate concentrations often decreased with depth below the water table. Reducing conditions were frequently encountered at depth, and also near the water table in the northwest part of the project area, an area with shallow groundwater and poorly drained soils.

Groundwater in and upgradient of four vegetable fields in the north part of the Priority Watershed was monitored in detail. Groundwater was slightly more acidic within fields (pH 5.0, vs. 5.4 upgradient), and solute content was greater. Chloride, Na, K, Ca, Mg, and Al concentrations were substantially greater within fields, but Fe concentrations were less. The average NO_3-N concentration within the fields was 22.4 mg L^{-1}, compared to 1.4 mg L^{-1} upgradient.

Pesticides

Pesticides were detected at all sampling locations within the four intensively monitored fields. The same pesticides were found as outside the fields, plus de-isopropyl atrazine. Atrazine exceeded its Wisconsin preventive action limit (PAL) more frequently than the other pesticides. Metolachlor, carbofuran, and alachlor also exceeded their PALs occasionally. Atrazine and metolachlor exceeded their Wisconsin enforcement standards (ESs) at a few locations.

Pesticides were detected in wells at six monitoring locations outside but near irrigated vegetable fields. These were metolachlor, metribuzin, alachlor, carbofuran and atrazine with its metabolite de-ethyl atrazine. We were not able to test for all pesticide residues. For instance, alachlor ESA was probably present, but was not detectable with the analytical method used. Alachlor, de-ethyl atrazine, and metolachlor each was detected once above its PAL. None exceeded its ES. The lack of ES exceedences indicated that recent farming practices generally did not pollute groundwater above the pesticide ES very far from fields.

Land-use impacts

Plumes from irrigated fields underlay 54% of the basin, according to flow and particle-tracking models. Plume thicknesses, as indicated by monitoring data, ranged from about 3 to 15 m depending on flow conditions and the number of fields that lie along the same flow line. The plumes had a distinctive chemistry compared to other groundwater, usually exhibiting elevated concentrations of NO_3, Ca, Cl, K, Mg, Na, and SO_4, as well as occasional pesticides. All NO_3-N concentrations exceeding 8 mg L^{-1} were associated with plumes from irrigated vegetable fields. In some cases where several fields lay along a flow line, the entire saturated thickness exceeded the NO_3-N standard of 10 mg L^{-1}. We found no evidence that other land uses in the watershed significantly impacted groundwater. However, some common land uses absent from the

watershed, such as small-lot unsewered subdivisions and non-irrigated agriculture, may have important groundwater impacts in other parts of the WCSP.

NITRATE LOADING TO GROUNDWATER UNDER IRRIGATED FIELDS

Estimate of potentially leachable N

Potentially leachable N was calculated using a budget approach. Estimates ranged from 133 to 237 kg ha^{-1}, or 40-82% of total N input, depending on crop, yield, weather, and cultural treatments. The greatest leachable N percentages were found for sweet corn in both years, due to poor yields and meager harvested N output. The greatest amounts of leachable N were found with potato in 1993, primarily because excessive rain compelled heavy N fertilizer application.

Simple calculation from NO_3-N concentration and recharge

The average NO_3-N concentration in shallow groundwater below the four fields and the amount of recharge in both years were used to derive simple estimates of NO_3 loading to groundwater. These estimates were 76-95 kg ha^{-1} in 1992 and 97-122 kg ha^{-1} in 1993. For the two years and four fields, the estimated annual loading was 112 kg ha^{-1}, 42% of total N input.

Loading through a water year

Detailed concentration profiles were analyzed to quantify annual NO_3 loading to groundwater beneath one field. Estimated NO_3-N loading to groundwater in the 1992 water year, when sweet corn was grown, was 178 ± 31 kg ha^{-1}. For the 1993 water year, the estimate of NO_3-N loading under potato was 260 ± 53 kg ha^{-1}.

Fertilizer-N leaching model

A nitrogen leaching model was constructed for one field using the computer code PRZM to simulate fertilizer N leaching. For the 1992 sweet corn crop, the PRZM simulation indicated 42 kg ha^{-1} of NO_3-N leached to groundwater out of 250 kg ha^{-1} fertilizer N. This was less than the estimates by other methods, because only direct leaching of fertilizer N was included. PRZM's calculation of crop N uptake was combined with data on N removed in harvest to indicate 140 kg N ha^{-1}, or 56% of the applied fertilizer N, became available for leaching.

For the 1993 potato crop, the PRZM-simulated fertilizer NO_3-N loading was 193 (north field-half) or 216 (south) kg ha^{-1}. Residue N was not important with potato.

Comparison of methods for estimating N loading

The four methods of estimating NO_3-N loading evaluated different types of loading. Generally, they agreed fairly well, except that the result based on NO_3-N concentrations and recharge was low. Nitrate concentrations in the lower part of the sampled zone may be diminished by mixing with groundwater that originated upgradient from the fields, leading to artificially low averages. In 1992, the N budget and the water year method agreed perfectly. The N leaching model indicated less loading to groundwater, but this would be expected since it did not consider N sources other than fertilizer. The percentage of N input found to leach was nearly the same for all three methods. In 1993, the comparisons among methods differ between high and low-input regimes. Where N input was less , the N budget predicted less NO_3-N loading than the water year method, and the leaching model predicted almost the same as the water year method even though it ignores non-fertilizer N inputs. With more N input, the N budget estimate was again low compared to the water year method. The leaching model predicted less N loaded to groundwater than the water year method, as expected.

Efficacy of model

The PRZM N leaching model gave results for potato that agreed reasonably well with the other methods. The sweet corn results approached those of the other methods only when additional data (on crop residue N) were considered. PRZM provided useful results with a modest requirement for input data.

EFFECTIVENESS OF BMPS

The cooperating growers in this study followed practices close to UW-Extension recommended BMPs, yet groundwater beneath the fields was still substantially impacted by agrichemicals. The mean NO_3-N concentration in the top 3 m of groundwater below the four fields was 22.4 mg L^{-1}. Estimates of NO_3-N loaded to groundwater below sweet corn in 1992 were 178 kg ha^{-1} (140 kg ha^{-1} if only fertilizer N is considered). Below potato in 1993 the estimates ranged from 164 to 291 kg ha^{-1}, depending on N input and the method used to estimate loading. In addition, we detected seven pesticide residues in groundwater beneath the fields. Atrazine residues and metolachlor exceeded their Wisconsin ESs, and carbofuran and alachlor exceeded their PALs.

Still, BMPs brought some groundwater quality improvements through reducing or substituting agrichemical inputs. A decrease in N application of 15 or 60 kg ha^{-1} reduced NO_3-N loading by roughly similar amounts. Also, discontinuing atrazine applications resulted in a decrease in both the detection frequency and concentration of atrazine residues.

The use of current BMPs alone will not prevent further degradation of WCSP groundwater. A combination of new BMPs, crop improvement, different crops or rotations, and perhaps institutional controls may be needed.

If BMPs cannot protect groundwater, other strategies are needed. Stites and Kraft (1992) considered two hypothetical strategies for meeting the groundwater NO_3 standard, assuming that 40% of applied N leaches as NO_3, and the groundwater column beneath a point is completely mixed. The first strategy would implement a rotation such that the average groundwater recharge over the rotation meets standards. Potato, the most profitable crop, could be grown every 5-6 years, but no N could be applied during the rest of the rotation. The other strategy would continue the present rotation, but restrict the fraction of land allowed in production, so that average groundwater quality in an area would meet standards. Under this strategy, 30% of the basin could be in the current rotation, if no NO_3 leaches from the remaining land.

The apparent failure of BMPs to protect groundwater and the restrictiveness of the hypothetical strategies indicate that new ideas are needed to maintain profitable agriculture and protect groundwater. Some ideas could include improved crop varieties that would take up nutrients more efficiently and resist pests, improved pesticide compounds or techniques, alternative economically viable crop rotations, or economic incentives to permit Wisconsin vegetable growers to remain competitive while setting cropland aside at least part of the time.

CHAPTER 1

INTRODUCTION

PROJECT PURPOSE, DESCRIPTION, AND OBJECTIVES

The Port Edwards Groundwater Priority Watershed (PEGPW) Project was implemented by the Wisconsin Department of Natural Resources and County of Wood with the goal of maintaining or improving groundwater quality in sandy agricultural areas. This Groundwater Resource and Agricultural Practice Evaluation supports the primary and secondary objectives of the Project, particularly the following (from Statement of Objectives, Wisconsin Department of Natural Resources, 1991):

- "to implement best management practices within the project area which are necessary to achieve groundwater quality standards or maintain groundwater quality in areas which currently have water of better quality than groundwater standards."
- "to test the practicality of application of best management practices..."
- "to determine how knowledge gained from the project can be applied to broader geographic areas."
- "to determine how current agricultural practices affecting water quality in the project area compare to known conventional agricultural practices from a water quality standpoint."
- "to develop and use methodology for the estimation of the potential amount of improvement in groundwater quality from implemented management practices."

In a larger context, this evaluation addresses groundwater nonpoint pollution by agrichemicals in the Wisconsin Central Sand Plain (WCSP) and other sandy areas of the Midwest. Agrichemicals are a major source of nonpoint groundwater pollution (Hallberg, 1986) and an issue of "great public concern" (US Congress Office of Technology Assessment, 1990). Agrichemical leaching is a particular problem in some parts of the US (Johnson and Parker, 1993), such as the Wisconsin Central Sand Plain (WCSP; e.g., Rothschild, 1982; Kraft, 1990; Harkin et al., 1986; Hindall, 1978). The federal Maximum Contaminant Level of 10 mg L^{-1} NO_3-N is exceeded in more than 30% of domestic wells in some WCSP townships (Central Wisconsin Groundwater Center database, 1993), compared to a national exceedence rate only 2.4% (USEPA, 1990). The pesticide atrazine has been detected in over half the wells tested in one WCSP county (Portage County; R. Schmidt, Portage County Planning and Zoning Office, 1995. Pers. commun.), whereas the nation-wide detection rate is only 4.2% for all pesticides (USEPA, 1990). Other pesticides identified in WCSP groundwater include aldicarb, atrazine, metribuzin, metolachlor, and ethylene dibromide (WDNR, 1993). Besides health concerns, agrichemical groundwater pollution has economic impacts: high nitrate has

forced two WCSP municipalities to add costly treatment systems, and may cause devaluation of affected real estate.

Some basic information needs to manage groundwater quality in the WCSP include:

- the quality of groundwater that results from various land uses and land-management strategies in the WCSP;
- whether existing "best management practice" (BMP) agriculture allows profitable farming and still adequately protects groundwater;
- how to improve BMPs to enhance groundwater protection;
- if BMPs are not the answer, what alternative strategies might result in acceptable quality groundwater.

One of the goals of this study is to provide this information with an eye toward ultimately improving groundwater quality.

OBJECTIVES

The objectives of the Groundwater Resource and Agricultural Practice Evaluation are to:

1. describe the hydrogeology of the Priority Watershed Area.
2. measure the prevailing groundwater quality and assess the impacts of major land uses.
3. measure and compare nitrate loading to groundwater under conventional practices and BMPs.
4. determine which, if any, pesticides leach under conventional practices and BMPs.
5. evaluate the utility of a leaching model to simulate observed nitrate loading to groundwater.

OVERVIEW OF TASKS

An outline follows of the tasks implemented to meet the objectives:

Describe study area geology:

Review existing geological information

Drill and log boreholes to determine types and distribution of sediments

Analyze sediments

Describe study area hydrogeology:

Determine hydraulic properties of sediments

Determine groundwater flow patterns

Construct a groundwater flow model

Examine basin-scale water-quality and impacts of prevailing land uses:

Construct a particle-tracking transport model

Describe overall watershed groundwater quality

Measure changes in water quality along selected flow paths

Describe variations in groundwater quality under different land uses

Monitor water quality beneath irrigated fields to measure N loading under conventional and BMP agricultural practices

Construct and evaluate a computer leaching model to simulate NO_3-N loading to groundwater

Reconnaissance survey of agricultural pesticide leaching

Sample groundwater under irrigated fields to establish which pesticides leach under conventional and BMP agricultural practices

This report covers the period of study from June, 1991 through April 30, 1994.

STUDY AREA

The study area covers about 33 km^2 of the Wisconsin Central Sand Plain, the 12 western sections of the Town of Port Edwards (T21 N, R8 E.), about 1 km west of the city of Nekoosa (Figures 1.1, 1.2). The area is generally level, except for a terrace escarpment along its eastern edge that drops 30-35 ft to the backwaters and channels associated with the Wisconsin River. Otherwise, the regional slope dips southwest at about 0.001. Surface elevation ranges from 1000 to 970 ft MSL.

Irrigated agriculture is one major land-use, occupying 638 ha (about 22% of the total). Most of the irrigated land is used for a vegetable rotation, but one irrigated dairy operation is also present. Other major land-uses include forest, nonirrigated agriculture, meadow, or meadow returning to a natural state. A small but important land-use is residential use. About 200 residences are located in or within 1/2 km of the study area. The proximity of residential and agricultural land uses has resulted in conflict due to the presence of agrichemicals in groundwater.

Wisconsin Central Sand Plain

The WCSP is an extensive area characterized by level topography and a mantle of glaciofluvial and glaciolacustrine sand and gravel ("outwash") that ranges in thickness from a few to more than 40 m . The outwash overlies less permeable Precambrian crystalline rock, silty and clayey reworked residuum (Clayton, 1986) and/ or Cambrian sandstone.

WCSP glacial deposits exhibit heterogeneous lithology, stratigraphy, and grain size distribution due to differences in source material and depositional environments. Though largely composed of coarse grained

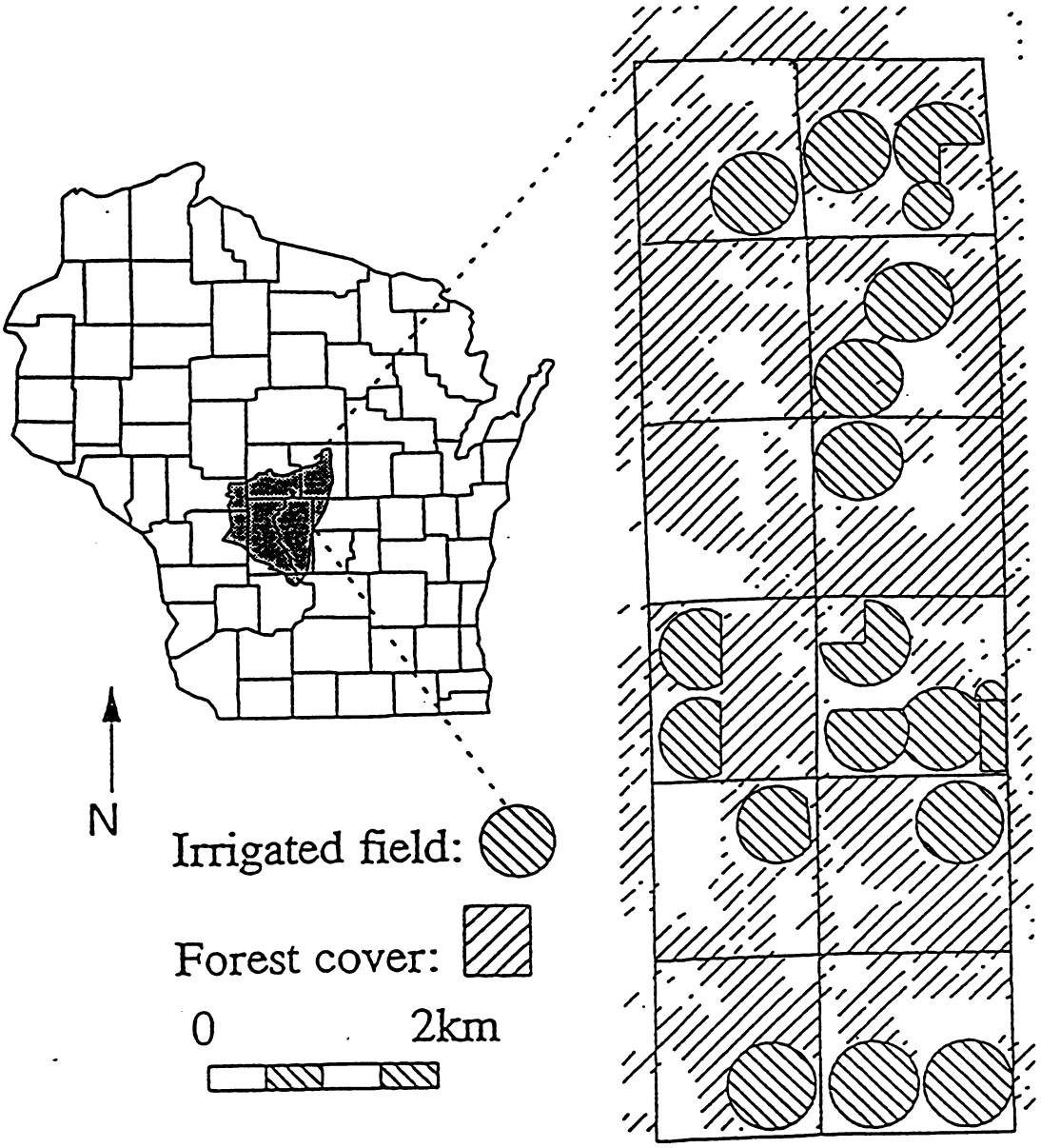


Figure 1.1 Location of study area

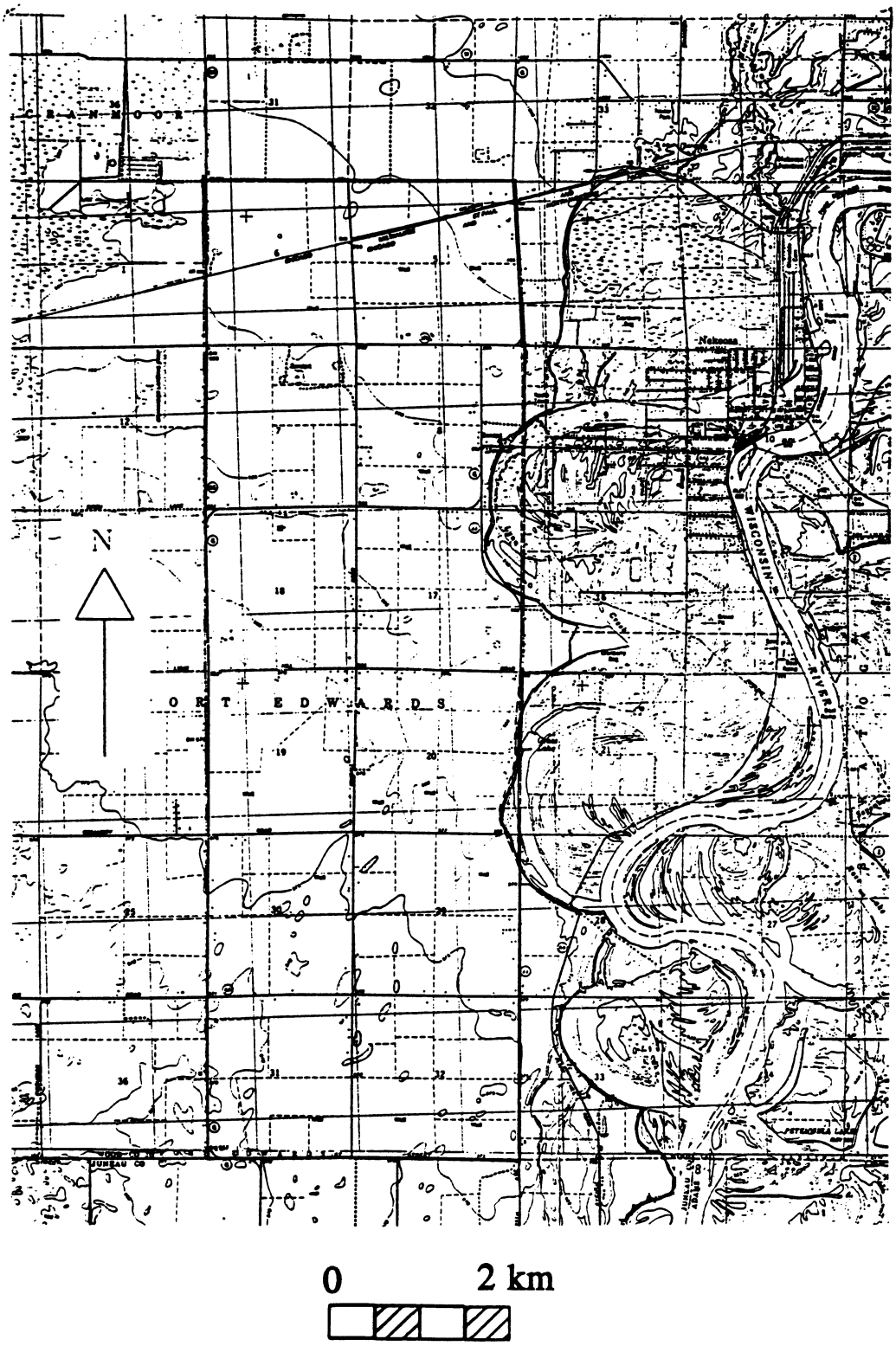


Figure 1.2 Study area physical features

materials, silty and clayey lacustrine beds are interspersed between coarser-grained units, especially in the western WCSP.

The glacier that deposited the outwash moved from the east and northeast across crystalline rock, sandstone, dolomite, and limestone, marking its furthest advance by the terminal moraines that flank the sand plain to the east. Though the tills associated with the glaciation are rich in carbonate fragments, the outwash is 90-95% quartz with most of the remainder derived from crystalline rock. Evidence suggests that carbonates are more abundant close to the moraine and decrease to the west, apparently due to the paucity of carbonate minerals in the sand-sized fraction of the glacial debris and their abundance in the coarser and finer fractions (Clayton, 1986). Coarser materials containing more carbonates probably were deposited closer to the moraines as high-energy water flowed westward from the glacier, keeping the fines suspended.

Natural groundwater chemistry is the Ca-Mg-HCO_3 type; however, NO_3 from anthropogenic sources has supplanted HCO_3 in some places. Hardness and alkalinity range from less than 25 to 250 mg L^{-1} and pH from 5.0 to 8.2 (Holt, 1965; Summers, 1965; Karnauskas, 1977; Rothschild, 1982). Typical NO_3–N values are less than 1 to 40 mg L^{-1}. Due to geology and groundwater flow patterns, pH, Ca, Mg, and alkalinity are highest close to the moraine and at deeper levels below the water table. Dissolved Fe and limited dissolved oxygen data suggest that groundwater is generally aerobic in the eastern WCSP and in shallow groundwater, but is oxygen-depleted in portions of the western WCSP and in deep groundwater (Holt, 1965; Summers, 1965; Kraft, 1990; Chambers, oral commun.; Bradbury, written commun.). Groundwater temperature is about 10 ± 1.5°C.

Climate and hydrologic budget

According to Bartelme et al., 1977, the climate of Wood County is typical of the center of a continent in the middle latitudes. Winters are long, cold, and snowy; summers are warm and occasionally humid; and spring and fall are sometimes short and are mixtures of summer and winter. An average of 9 days a year have temperatures of 90° F or higher while an average of 32 days a year have temperatures of 0°F or lower. Heat growth units during the growing season average 2240 above a 50°F threshold. The average growing season length (last freeze in spring to first in fall) is 133 days. Annual precipitation averages 31.3 inches, with a 10% probability of less than 24.6 or more than 40.1. Approximately 60% of the annual precipitation falls in May through September.

The hydrologic budget for sandy areas of central Wisconsin has been discussed by Weeks et al. (1965) and Holt (1965). They estimate that, of the 30-31 inches of precipitation, 20–22 inches goes to evapotranspiration, 9–10 inches to groundwater recharge, and 1 inch to direct runoff.

Climate during period of study

A summary of climatic data for the period 1988–1993 is given in Tables 1.1–1.3. Data in the tables are for Wisconsin Rapids, the closest reporting station to the Priority Watershed. Total yearly precipitation ranged from 24.06 inches in 1988 to an unusually wet 38.98 inches in 1993. 1988 and 1989 were drought years; the years since have had normal or above-normal precipitation. The growing seasons (last freeze in spring to first freeze in fall) throughout this period were close to the normal 133 days, with the exception of 1992, when the growing season was only 97 days. Heat growth units during the growing season above a 65°F threshold range from 190 days in 1992, a cool year, to 844 days in 1988. The heat units in 1993 were near average. To summarize the years of the study, 1992 had a short, cool growing season, and 1993 was very wet.

Table 1.1. Monthly total precipitation (inches), 1988–1993.

	1988	1989	1990	1991	1992	1993
January	1.55	0.43	0.56	0.96	0.63	1.25
February	0.32	0.58	0.74	0.39	0.48	0.22
March	1.72	2.34	2.16	2.90	2.47	1.25
April	2.34	0.98	4.36	1.97	2.77	5.58
May	0.61	8.82	5.53	3.52	2.36	5.92
June	1.13	1.01	1.20	5.60	1.29	7.76
July	2.30	2.65	3.80	6.73	3.81	5.61
August	4.03	2.95	1.84	6.88	2.55	4.55
September	5.24	1.24	2.99	2.88	8.11	3.25
October	1.44	4.06	2.48	1.99	1.15	1.57
November	2.61	1.38	4.39	0.63	3.89	0.99
December	0.77	0.31	1.36	1.74	2.11	1.03
TOTAL	24.06	26.75	31.41	36.19	31.62	38.98

Source: USEDS, 1988-1993.

Table 1.2. Growing season length (days above freezing), 1988–1993.

	1988	1989	1990	1991	1992	1993
Dates	5/25-10/3	5/12-9/23	5/11-10/2	5/4- 9/19	5/27-9/1	5/19-9/24
Length	131	134	144	138	97	128
Deviation from average of 133 days	-2	1	11	5	-36	-5

Source: USEDS, 1988-1993.

Table 1.3. Degree days above 65°F, 1988–1993.

	1988	1989	1990	1991	1992	1993
November–March: zero in all six years.						
April	0	0	23	7	0	0
May	55	10	1	76	25	27
June	181	61	73	158	55	56
July	304	189	103	152	43	118
August	271	93	86	135	62	155
September	33	7	48	49	5	0
October	0	1	0	0	0	0
TOTAL	844	361	334	577	190	356
Deviation from average (421 degree-days)	423	-60	-87	156	-231	-65

Source: USEDS, 1988-1993.

CHAPTER 2

GEOLOGY

Study area geology was investigated to obtain information needed for (1) understanding and modeling groundwater behavior and (2) providing a basis for comparing the PEGPW with sandy areas elsewhere in the WCSP, Wisconsin, and the Midwest. The objective of the geologic investigation was to describe aquifer sediment characteristics and their distribution, characterize the presence and distribution of aquitards, and describe the lower boundary of the aquifer.

BACKGROUND

Topography

The study area is generally flat except at a terrace escarpment along the eastern edge (Figure 2.1). The terrace escarpment is about 30-35 ft high and has a slope of 25% or more. Slopes over most of the area are 0.1 to 0.5%, and the local relief consists of subdued rises and swales differing by less than 3-5 ft. The regional slope is southwestward, with a gradient of 0.001. Surface elevation varies from about 970 to 1000 ft MSL (feet above mean sea level).

Geologic units

The geology of the area can be divided into five major units:

- Precambrian crystalline rock
- Cambrian sandstone
- Hillslope sediment
- Pleistocene deposits
- Recent deposits.

Precambrian crystalline rock is the predominant bedrock in southern Wood County, but is overlain by Cambrian sandstone in places (Clayton, 1991). Batten (1989) indicates the project area is in a region where the Cambrian sandstone pinches out. Both bedrock units are insignificant aquifers in terms of the present study, although they are used for water supplies in other parts of Wood County. Since the hydraulic conductivities of both rocks are likely several orders of magnitude less than that of the Pleistocene aquifer, bedrock units are treated as aquitards in this study.

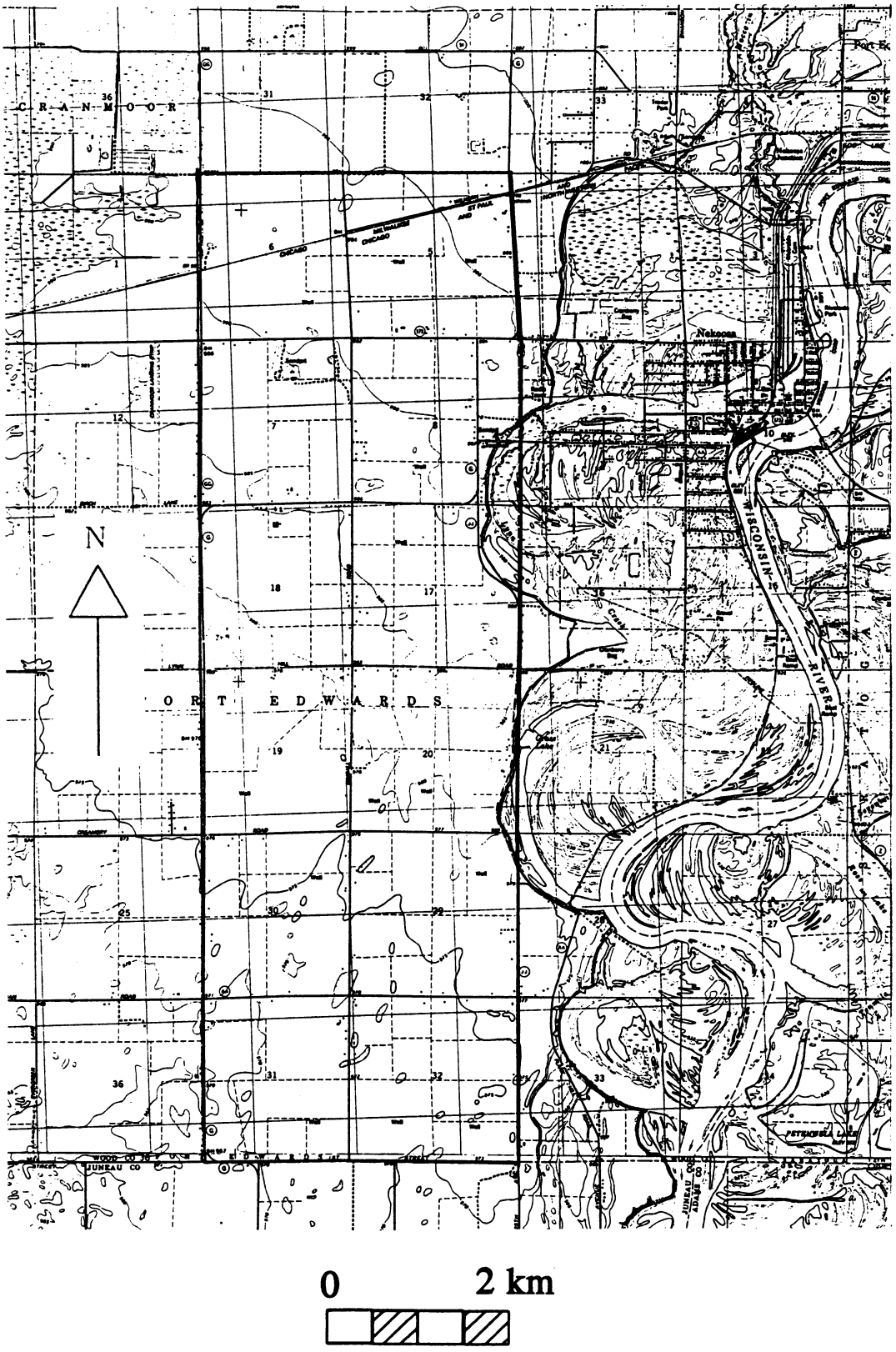


Figure 2.1 Study area physical features

Hillslope sediments are composed of residues of local rock decomposition that were reworked and transported by wind, water, soil creep, and ice-wedge formation during Pleistocene and pre-Pleistocene times (Clayton, 1991). Hillslope sediments usually contain greater than 50% silt and clay and less than 5% gravel. Where present, they are typically 9-12 ft thick and overlie the Precambrian rock in Wood County (Clayton, 1991). Though Clayton (1991) mentions this unit only in reference to the central and northern parts of Wood County where Pleistocene units are thin or lacking, we found the unit within the project area between the Pleistocene and bedrock units. Due to its fine texture, the hydraulic conductivity of this unit is substantially less than that of the Pleistocene aquifer, and it can also be treated as an aquitard.

The Pleistocene materials in the Port Edwards Groundwater Priority Watershed are mapped as off-shore and near-shore deposits of Glacial Lake Wisconsin, assigned to the Big Flats Formation (Attig et al., 1988). These consist largely of sand and gravel deposits containing one or more interbedded subunits of silt and clay. Brownell's (1986) description of the Big Flats sands states that they are moderately well sorted with mean grain size between 175 and 375 μm, pebbles are practically absent, and the content of non-quartz grains is typically less than 1%. An extensive silt and clay subunit occurring 15-30 ft below the surface has been named the New Rome Member of the Big Flats Formation. One or two other discontinuous fine layers have been found below the New Rome Member, but less is known about them (Clayton, 1991).

Recent deposits consist of aeolian sand, terrace alluvium, and peat. Aeolian sand originated during the final retreat of Glacial Lake Wisconsin, when exposed sand in the lakebed was subject to wind transport. In the southern part of Wood County, the aeolian sand is thin, averaging about 1.5 ft (Clayton, 1991). Alluvium has collected on postglacial terraces and meanders of the Wisconsin River along the east edge of the project area. Peat has formed in the low portions, such as the oxbows in the east parts of Sections 8 and 17 (Figure 1.2). The river bottom area is small and not representative of the Wisconsin central sand plain as a whole, and is not specifically addressed by the Port Edwards Groundwater Priority Watershed project.

Soils

The principal soils mapped in the project area are the excessively-drained Plainfield sand and loamy sand, moderately well-drained Friendship loamy sand, somewhat poorly-drained Meehan loamy sand, and poorly-drained Newson loamy sand. All four are included in a single soil catena (Bartelme, 1977), that is, they developed in identical parent materials with different drainage characteristics. "Drainage", as used in soil classification, describes the depth and duration of saturation within the soil profile. Soils with different characteristics will develop in the same parent material as a result of being wetter or drier.

Plainfield is by far the most extensive soil. The wetter three soils are found mostly in the northwest part of the project area (sections 6, 7, and 18).

Pleistocene history

The following outline of Pleistocene history is based on Clayton (1986, 1987, 1991). Most Pleistocene deposits in the study area probably originated during the late Wisconsin period, although some may have accumulated in early- or pre-Wisconsin times. Early- and pre-Wisconsin deposition could include the clayey and silty layers occurring below the New Rome member. The Pleistocene not only brought new materials into the study area, but modified older sediments through solifluction and other processes of the permafrost climatic regime.

During the late Wisconsin period, the Green Bay Lobe of the ice sheet advanced within about 25 km east of the project area. Farther south, the glacier reached the east end of the Baraboo Hills and formed an ice dam which blocked the flow of southward-flowing meltwater, forming Glacial Lake Wisconsin. The lake covered large parts of Adams, Juneau, Portage, and Wood Counties, including the project area, and smaller parts of several other counties. The lake may have formed and drained episodically during the Wisconsin Glaciation, as the ice dam was periodically breached and re-constructed. Sediment-laden meltwater flowing into Glacial Lake Wisconsin deposited sand and gravel as well as the New Rome and other fine-textured layers on the lake bottom. These lake sediments are the most conspicuous feature of the present-day landscape.

When the glaciers receded and the ice dam was breached a final time, an ancestral Wisconsin River carrying meltwater southward cut a channel through the lakebed sediments, and left terraces well above the present-day river level. Later, the river cut a deeper channel and removed the early terraces in places. Surficial sediments were reworked by wind in postglacial time. In most of the Wood County portion of the lake plain, including the project area, this aeolian sand is less than 1.5 ft thick, and any dunes are inconspicuous. Mostly after the time of dune formation, organic soils formed in perennially wet areas. There are some small occurrences of peat in the floodplain sediments at the east edge of study area, but these are outside the focus of the Port Edwards project.

GEOLOGY INVESTIGATION - DESCRIPTION AND METHODS

Use of existing records

Logs for production wells in the project area were located, compiled, and entered into an electronic database (Archive 1).

Boring and borehole logging

A total of 70 boreholes was drilled at 47 sites. Most boreholes (Figures 2.2 and 2.3) were installed in conjunction with monitoring-well installation, which was done in three phases (Chapter 3). Twelve of the boreholes reached the aquifer base. Sediment from one borehole was sampled at 5-ft. intervals and analyzed. Most of the samples were sieved to determine sand size fractions (three samples – the topsoil and two with less than 50% sand – were not sieved). pH, loss on ignition, and organic matter as determined by titration were also determined.

A truck-mounted drilling rig was used, usually with a 4-inch hollow-stem auger (in the north part of the project area, a 4-inch solid-stem auger was sometimes used). The auger was drilled to the required depth, with the bottom end of the hollow-stem auger closed off with a steel or PVC plug. A log was kept of the materials encountered based on cuttings brought up by the auger, sounds which could indicate gravel or consolidated rock, and the force needed to advance the auger. Silty layers often revealed themselves by the increased pressure needed to advance the drill, but this characteristic was difficult to distinguish. Therefore the interpretation of the depth of such layers was sometimes uncertain.

When the auger reached the required depth, a well was usually installed. The auger was then withdrawn one 5-ft section at a time. The cuttings on the auger flights were examined and used to verify and amplify the log. The easiest material to identify at this stage was silt or clay, since it stuck to the auger.

Porosity measurements

A vibra-corer was used to collect minimally disturbed cores at five sites around the watershed. Cores were collected on July 22 and 23, 1993. Five sites scattered around the watershed area had been chosen (Figure 2.2). The site in Section 20 was on the scarp at the east edge of the project area, where deeper glacio-lacustrine sediments were accessible due to the lakebed having been cut away by later erosion processes.

The vibra-core technique (Thompson et al., 1992?) is based on forcing a 74-mm inside diameter aluminum irrigation pipe into the subsurface, and then pulling it back out with a core of sediment held inside. A modified concrete vibrator powered by a gasoline engine is clamped to the pipe, and the vibration allows the pipe to be pushed into the ground with muscle power. A chain hoist was used to withdraw the cores.

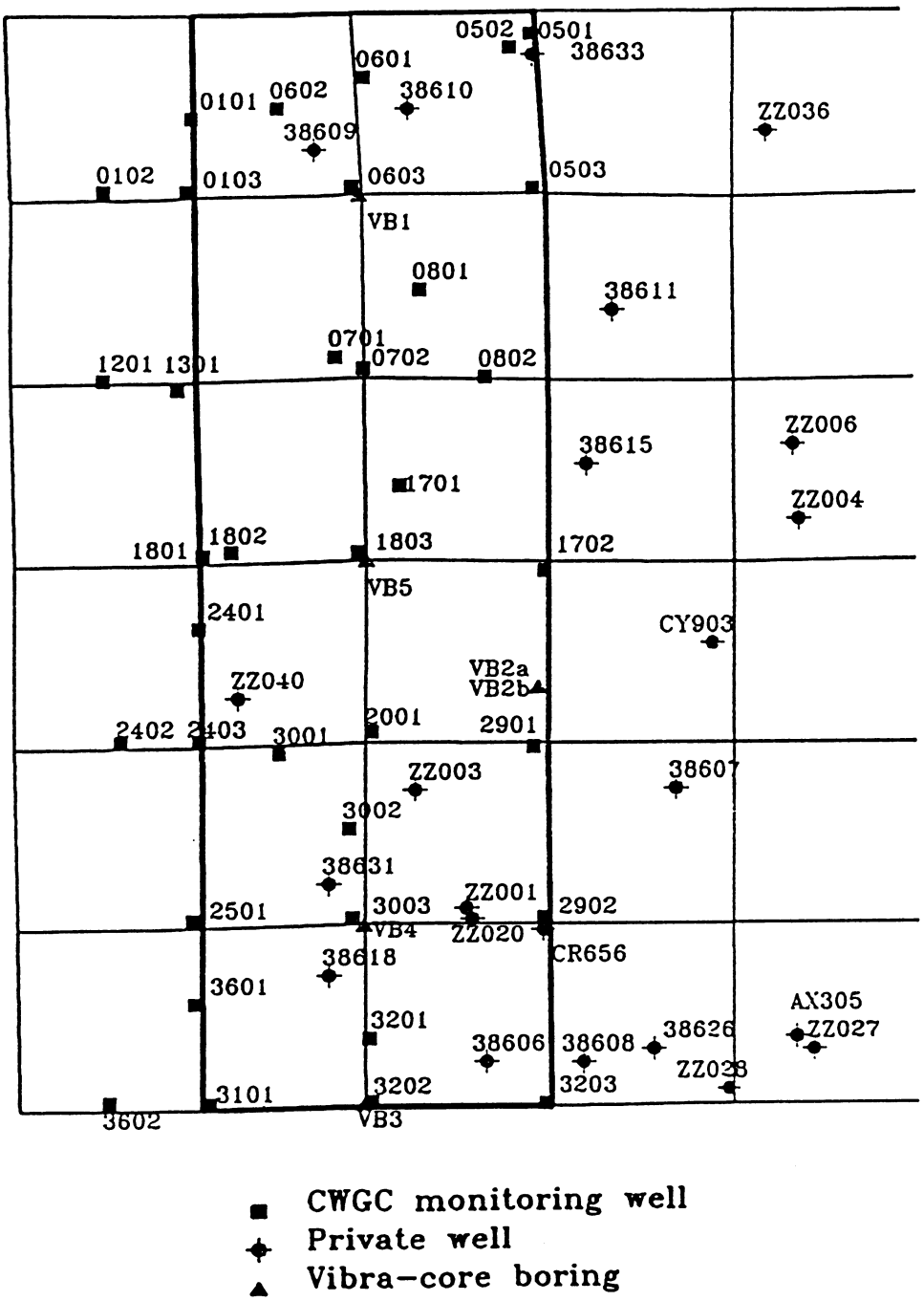


Figure 2.2 Locations of borehole sites, Port Edwards Groundwater Priority Watershed

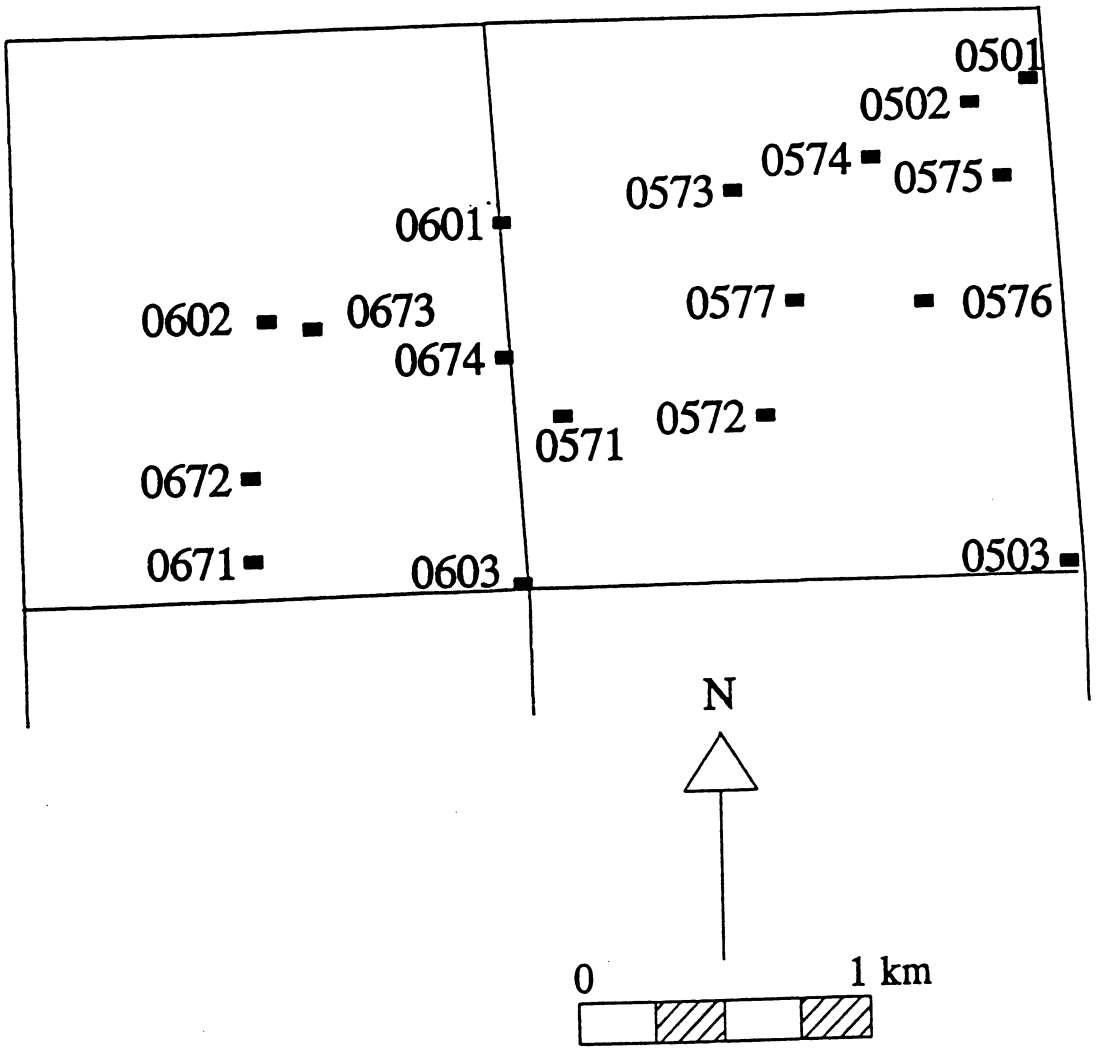


Figure 2.3 Locations of borehole sites in sections 5 and 6, Port Edwards Groundwater Priority Watershed

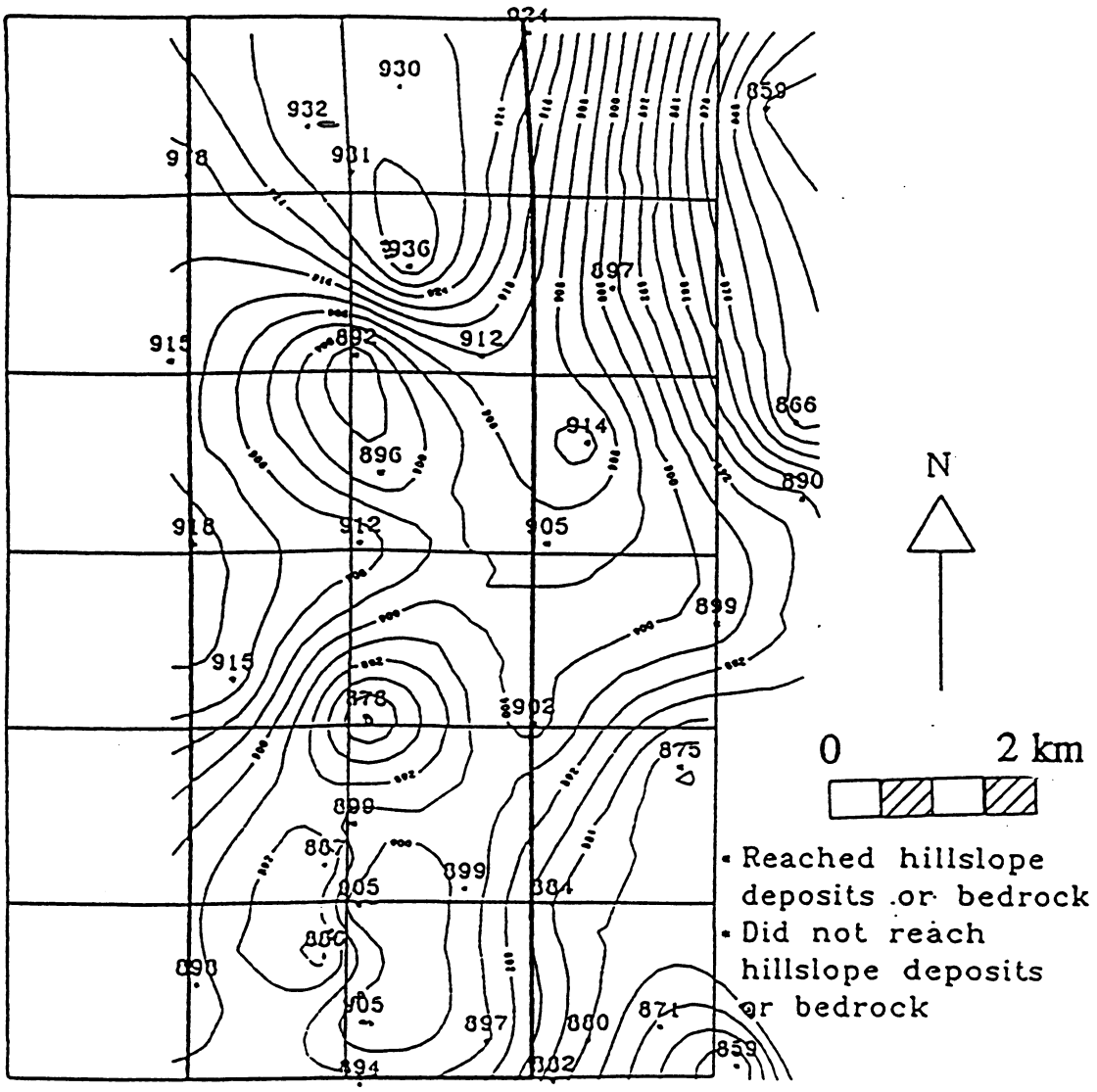


Figure 2.4 Aquifer-bottom elevations, Port Edwards Groundwater Priority Watershed

Measurements were taken to allow compaction of cores to be calculated. Details of sampling are included in Appendix 1.

The cores were analyzed in February 1994. They were frozen, then placed in a wooden clamp that also served as a saw guide. A circular saw, with the depth of cut set just sufficient to penetrate the pipe, was pushed along one side of the core, then the core was turned 180° and sliced again. The halves of the pipe were warmed to free them from the frozen sediment inside, and were lifted off the core. Horizonation and texture were described, and the core was photographed. The core was placed in a miter box and was cut into sections according to visible horizons; sometimes horizons were subdivided for convenient handling. The sections were measured along the longest and shortest side, and if any material had been lost, the lost volume was estimated. Underestimation of lost material would translate into an overestimate of porosity. Volume of the core section, Vc, was calculated as $\pi r^2(l_1 + l_2)/2$, where r is the inside radius of the irrigation pipe (37.0 mm), and l_1 and l_2 are the longest and shortest lengths of the core section. The sections were placed in paper bags and air dried. They were then weighed and oven-dried at 105° C for 24 hours or more.

The volume occupied by solids in each core section was determined by water displacement. A 1-L cylinder of the type used for soil-particle-size sedimentation analysis was pre-weighed. Approximately 500 mL of water was added, and the cylinder + water combination was weighed (for silty sediment samples, the water contained 1% sodium hexametaphosphate to help disperse the unitary particles; the density of this solution was taken as equal to that of pure water). Sediment was added; in most cases, each core section had to be divided among 2-4 cylinders. After soaking overnight to disperse aggregated particles, the sediment was agitated with a plunger to free any air, and the plunger was rinsed into the cylinder with a wash bottle. The wash bottle was weighed before and after rinsing so that the weight of water added in rinsing could be included. The cylinder was weighed again, water was added to the 1-L mark and the cylinder was weighed for the last time. The water volume, V_w, (mL) was assumed numerically equal to its weight (g) as determined by the differences among weighings. Particle volume, V_p, was calculated as 1000 mL - V_w.

The particle volume and weight were totaled for all the cylinders used for a given core section, and the pore volume, V_a, was calculated as $V_t - V_w$., where V_t is total core volume. Then porosity was given by $f = V_a \div V_t$.

Another way to estimate porosity is based on the assumption that the density of sediment particles, ρ_p, was 2.65. This is the density of quartz; densities of common minority minerals, namely feldspars, are also quite close to this value, e.g. 2.63 for albite, 2.56-2.63 for microcline, and 2.55-2.63 for orthoclase (Weast, 1980). The weight of the sediment is essentially the weight of the particles, M_p, so that $V_p = M_p \div \rho_p$. This implies $f = 1 - (M_p \div \rho_p V_t)$, where M_p is the weight of the dry sediment. This way was

judged more reliable than the particle-volume method. The latter depends on accurately measuring the core volume, several successive weighings, and on complete dispersion of particle aggregates. The density method depends on core volume and only one weight.

To avoid underestimating porosity, we corrected for compaction that took place during core collection. A single average compaction value was used to correct all of the porosities, rather than applying individual compaction results to particular cores.

Compaction, K, was defined as $(L_0 \div L_1) - 1$, where L_0 and L_1 are the undisturbed and final core lengths. We can assume all of the compaction took place in the pore volume; the particles themselves would not be compressed measurably. The corrected porosity would be $f_{in\ situ} = V_{a,}$ in situ $\div V_{t,\ in\ situ}$. $V_{a,\ in\ situ} = V_a + (V_{t,\ in\ situ} - V_t)$, since $(V_{t,\ in\ situ} - V_t)$ is the volume lost during compaction. $V_{t,\ in\ situ} = (K + 1)V_t$, and $V_a = fV_t$, so $V_{a,\ in\ situ} = (f + K)V_t$. $V_{t,\ in\ situ} = (K + 1)\ V_t$, therefore $f_{in\ situ} = (f + K) \div (K + 1)$.

Measured compaction among C horizon samples varied between -4.3% and 9.9%, with a mean of 1.9%. This value is far less than the 17% mean compaction found by Thompson et al. (1992?), and even the maximum compaction we observed is below their average value. Their higher average appears to be due to the fact that much of their sampling was conducted in highly compressible materials.

How sensitive was the corrected porosity to the choice of a compaction value? For a measured porosity of 0.392, the in-situ porosity would be calculated as 0.403 based on 1.9% compaction and 0.447 based on 9.9% compaction. Comparison of our porosity with literature values that are generally lower suggests compaction was not seriously underestimated, so we have chosen 0.40 as our best estimate of aquifer porosity.

Samples collected in irrigated fields

Soil samples were collected April 19-20, 1993, in the four irrigated fields in the north part of the study area that were monitored for agricultural groundwater impacts. These samples were collected using a 3-inch bucket auger at depth intervals of 0-1,, 1-3, 3-5, 5-7, and 7-10 ft below the surface. A riffle splitter was used in the field to reduce the sample volume to about 500-700 mL. Samples were not taken from below the water table, so the sampling depth was curtailed at some locations.

Geologic representations

Boring log interpretations were used to represent study area geology through cross sections and contour maps. The contouring software package SURFER was used to construct maps describing

- New Rome Member thickness,

- depth to the New Rome Member,
- elevation of the top of the New Rome Member,
- elevation of the top of a second lacustrine unit, and
- elevation of the Pleistocene (aquifer) base.

Kriging was used as the interpolation method. Data from the borings installed during this study as well as boring logs for production wells were used as inputs. The Pleistocene base was taken as either (1) the top of the hillslope deposits or (2) the bedrock surface, if no hillslope deposits were encountered. Because few boreholes contacted hillslope deposits or bedrock, the Pleistocene base was sometimes estimated from the deepest penetration into Pleistocene deposits at a particular location. The selection of which deepest penetration data to include was subjectively based on a comparison to adjacent data points. For example, the lower of two values would be chosen to reflect the aquifer bottom elevation at adjacent locations. Because of the uncertainty inherent in using borehole data when the aquifer bottom was not reached, the map of the aquifer bottom elevation is necessarily approximate.

Sediment analyses

Detailed information on particle size distributions comes from analyses of samples taken from a borehole near the center of the Port Edwards Groundwater Priority Watershed project area, and samples collected in the four intensively studied irrigated fields in the northern part of the project area.

In 1991, 17 samples from a borehole near the intersection of Creamery Road and Kimball Avenue (location code 2001, Figure 2.4) were collected and sent to the University of Wisconsin Soil and Plant Analysis Laboratory. Most of the samples were sieved to determine sand size fractions (three samples, the topsoil and two with greater than 50% silt + clay, were not sieved). The samples were also analyzed by hydrometer to determine the contents of silt and clay. Drilling continued at this site to 98 ft, but samples below 78 ft were not analyzed in the laboratory.

Soil samples collected in the irrigated fields were brought to the lab, and approximately 100-g subsamples were taken by quartering for sieve analysis. Forty of the 111 samples were sieved using a motorized shaker. The following sieve numbers and size classes were used: no. 10 - gravel, >2 mm; no. 18 - very coarse, 1-2 mm; no. 35 - coarse, 0.5-1 mm; no. 60 - medium, 0.25-0.5 mm; no. 140 - fine, 0.1-0.25 mm; no. 270 - very fine, 0.05-0.1 mm. Material that passed the no. 270 sieve was collected and measured as silt and clay. There was some concern that the sieve analysis might not be accurate for surface soil samples, with their greater organic-matter content. We thought that silt and clay particles might tend to stick together and be counted with sand fractions. Therefore, nine of the sieved samples were sent to the University of

Wisconsin Soil and Forage Laboratory at Marshfield for hydrometer analysis, however, chemical pre-treatment to remove organic matter and iron oxides was omitted.

RESULTS OF GEOLOGIC INVESTIGATION

Description of PEGPW geology

Locations of borings used in this study are shown in Figures 2.2 and 2.3. Lithology encountered in borings is summarized in Table 2.1, and detailed in Appendix 2. Pleistocene materials were encountered from the surface down to contact with bedrock or hillslope deposits at elevations from 958 to less than 835 ft MSL based on water-supply well reports, or from 939 to less than 882 ft MSL based on Groundwater Center borehole logs (Figure 2.4). They consist primarily of well-sorted medium sand, with up to two interbedded layers of silty and clayey materials. Total thickness was generally 60 to at least 100 ft.

Table 2.1. Summary of geological materials found in boreholes in the Port Edwards Priority Watershed.

Location		Depth		Elevation	
code	Material	from	to	from	to
0101	Sand	0	26	990.72	964.72
0101	Silt/Clay	26	30	964.72	960.72
0102	Sand	0	26	986.87	960.87
0102	Silt/Clay	26	28	960.87	958.87
0102	Sand	28	30	958.87	956.87
0103	Sand	0	29	986.72	957.72
0103	Silt/Clay	29	31	957.72	955.72
0103	Sand	31	53	955.72	933.72
0501	Sand	0	31	995.16	964.16
0501	Silt/Clay	31	33	964.16	962.16
0502	Sand	0	25	997.00	972.00
0502	Silt/Clay	25	27.5	972.00	969.50
0502	Sand	27.5	50	969.50	947.00
0502	Silt/Clay	52	60	945.00	937.00
0502	Sand	60	68	937.00	929.00
0503	Sand	0	21	993.05	972.05
0503	Silt/Clay	21	23	972.05	970.05
0503	Sand	23	25	970.05	968.05
0503	Silt/Clay	25	31	968.05	962.05
0503	Hillslope	69	80	924.05	913.05
0503	Sand	31	69	962.05	924.05
0573	Sand	0	24	992.91	968.91
0573	Silt/Clay	24	30	968.91	962.91

Table 2.1. Summary of geological materials found in boreholes in the Port Edwards Priority Watershed (continued).

Location code	Material	Depth from	Depth to	Elevation from	Elevation to
0573	Sand	30	32	962.91	960.91
0576	Sand	0	27	990.86	963.86
0576	Silt/Clay	27	28	963.86	962.86
0601	Sand	0	26	992.72	966.72
0601	Silt/Clay	26	28	966.72	964.72
0601	Sand	28	29	964.72	963.72
0602	Sand	0	19	993.00	974.00
0602	Silt/Clay	19	25	974.00	968.00
0602	Sand	25	33	968.00	960.00
0602	Silt/Clay	33	36	960.00	957.00
0602	Sand	36	54	957.00	939.00
0603	Sand	0	25	989.57	964.57
0603	Silt/Clay	25	29	964.57	960.57
0603	Sand	29	63	960.57	926.57
0671	Sand	0	29	987.37	958.37
0671	Silt/Clay	29	33	958.37	954.37
0671	Sand	33	37	954.37	950.37
0671	Silt/Clay	37	38	950.37	949.37
0671	Sand	38	65	949.37	922.37
0671	Silt/Clay	65	70	922.37	917.37
0674	Sand	0	20	991.76	971.76
0674	Silt/Clay	20	26	971.76	965.76
0674	Sand	26	37	965.76	954.76
0701	Sand	0	21	987.00	966.00
0701	Silt/Clay	21	26	966.00	961.00
0702	Sand	0	27	986.52	959.52
0702	Silt/Clay	27	29	959.52	957.52
0702	Sand	29	60	957.52	926.52
0702	Silt/Clay	60	61	926.52	925.52
0702	Sand	61	94	925.52	892.52
0801	Sand	0	13	986.01	973.01
0801	Silt/Clay	13	15	973.01	971.01
0801	Sand	15	21	971.01	965.01
0801	Silt/Clay	21	30	965.01	956.01
0801	Sand	30	56	956.01	930.01
0801	Hillslope	56	60	930.01	926.01
0802	Sand	0	30	988.24	958.24
0802	Sand	0	36	988.21	952.21
0802	Silt/Clay	30	35	958.24	953.24
0802	Sand	36	41	952.21	947.21
0802	Sand	35	85	953.24	903.24
0802	Silt/Clay	41	43	947.21	945.21

Table 2.1. Summary of geological materials found in boreholes in the Port Edwards Priority Watershed (continued).

Location code	Material	Depth from	Depth to	Elevation from	Elevation to
0802	Sand	43	74	945.21	914.21
0802	Hillslope	74	75	914.21	913.21
1201	Sand	0	18	980.31	962.31
1201	Silt/Clay	18	20	962.31	960.31
1201	Sand	20	22	960.31	958.31
1301	Sand	0	31	981.86	950.86
1301	Silt/Clay	31	34	950.86	947.86
1301	Sand	34	50	947.86	931.86
1301	Silt/Clay	50	53	931.86	928.86
1301	Sand	53	61	928.86	920.86
1301	Hillslope	61	69	920.86	912.86
1301	Bedrock	69	70	912.86	911.86
1701	Sand	0	26	984.00	958.00
1701	Silt/Clay	26	29.5	958.00	954.50
1701	Sand	29.5	90	954.50	894.00
1702	Sand	0	34	985.59	951.59
1702	Silt/Clay	34	37	951.59	948.59
1702	Sand	37	83	948.59	902.59
1702	Hillslope	83	86	902.59	899.59
1801	Sand	0	55	978.57	923.57
1801	Silt/Clay	55	60	923.57	918.57
1802	Sand	0	13	977.77	964.77
1802	Silt/Clay	13	24.5	964.77	953.27
1803	Sand	0	21	982.99	961.99
1803	Silt/Clay	21	24	961.99	958.99
1803	Sand	24	31	958.99	951.99
1803	Silt/Clay	31	33	951.99	949.99
1803	Sand	33	75	949.99	907.99
1803	Hillslope	75	76	907.99	906.99
2001	Sand	0	98	974.92	876.92
2401	Sand	0	55	976.83	921.83
2401	Silt/Clay	55	60.5	921.83	916.33
2401	Sand	60.5	68	916.33	908.83
2401	Hillslope	68	75	908.83	901.83
2402	Sand	0	19	972.43	953.43
2402	Silt/Clay	19	20	953.43	952.43
2402	Sand	20	21	952.43	951.43
2403	Sand	0	26	975.98	949.98
2403	Silt/Clay	26	28	949.98	947.98
2403	Sand	28	29	947.98	946.98
2501	Sand	0	7	969.30	962.30
2501	Silt/Clay	7	13	962.30	956.30
2501	Sand	13	14	956.30	955.30
2901	Sand	0	20	980.21	960.21

Table 2.1. Summary of geological materials found in boreholes in the Port Edwards Priority Watershed (continued).

Location code	Material	Depth from	Depth to	Elevation from	Elevation to
2901	Silt/Clay	20	21	960.21	959.21
2901	Sand	21	39	959.21	947.21
2901	Silty sand ?	39	42	941.09	938.09
2901	Sand	42	79.5	938.09	900.59
2901	Hillslope	79.5	80	900.59	900.09
2902	Sand	0	29	976.39	947.39
2902	Silt/Clay	29	33	947.39	943.39
2902	Sand	33	39	943.39	937.39
2902	Silt/Clay	39	43	937.39	933.39
2902	Sand	43	89	933.39	887.39
3001	Sand	0	59	976.31	917.31
3001	Silt/Clay	59	61	917.31	915.31
3001	Sand	61	65	915.31	911.31
3001	Silt/Clay	65	66	911.31	910.31
3001	Sand	66	70	910.31	906.31
3001	Hillslope	70	75	906.31	901.31
3002	Sand	0	19	975.68	956.68
3002	Silt/Clay	19	26	956.68	949.68
3002	Sand	26	79	949.68	896.68
3002	Hillslope	79	81	896.68	894.68
3003	Sand	0	14	971.41	957.41
3003	Silt/Clay	14	20	957.41	951.41
3003	Sand	20	66	951.41	905.41
3003	Hillslope	66	67	905.41	904.41
3101	Sand	0	28	966.13	938.13
3101	Silt/Clay	28	30	938.13	936.13
3101	Sand	30	33	936.13	933.13
3201	Sand	0	23	970.41	947.41
3201	Silt/Clay	23	26	947.41	944.41
3201	Sand	26	68	944.41	902.41
3201	Hillslope	68	70	902.41	900.41
3202	Sand	0	10	970.29	960.29
3202	Silt/Clay	10	11	960.29	959.29
3202	Sand	11	75	959.29	895.29
3202	Sand	75	76	895.29	894.29
3203	Sand	0	70	971.29	901.29
3203	Silt/Clay	70	73	901.29	898.29
3203	Sand	73	89	898.29	882.29
3601	Sand	0	22	969.76	947.76
3601	Silt/Clay	22	29	947.76	940.76
3601	Sand	29	45	940.76	924.76
3601	Silt/Clay	45	47	924.76	922.76
3601	Sand	47	75	922.76	894.76

Table 2.1. Summary of geological materials found in boreholes in the Port Edwards Priority Watershed (continued).

Location code	Material	Depth from	Depth to	Elevation from	Elevation to
3602	Sand	0	12	966.97	954.97
3602	Silt/Clay	12	28	954.97	938.97
3602	Sand	28	39	938.97	927.97
AX305	Sand	0	62	955.00	893.00
CR656	Sand	0	18	973.00	955.00
CR656	Silty	18	37	955.00	936.00
CR656	Sand	37	61	936.00	912.00
CY903	Sand	0	22	985.00	963.00
CY903	Silt/Clay	22	32	963.00	953.00
CY903	Sand	32	86	953.00	899.00
ZZ001	Sand	0	72	971.00	899.00
ZZ003	Sand	0	56	975.00	919.00
ZZ004	Sand	0	18	940.00	922.00
ZZ004	Silt/Clay	18	46	922.00	894.00
ZZ004	Hillslope	46	50	894.00	890.00
ZZ004	Bedrock	50	70	890.00	870.00
ZZ006	Sand	0	66	932.00	866.00
ZZ020	Sand	0	72	971.00	899.00
ZZ027	Sand	0	66	960.00	894.00
ZZ028	Sand	0	91	950.00	859.00
ZZ036	Sand	0	22	955.00	933.00
ZZ036	Silt/Clay	22	37	933.00	918.00
ZZ036	Sand	37	96	918.00	859.00
ZZ036	Bedrock	96	104	859.00	851.00
ZZ040	Sand	0	60	975.00	915.00
38606	Sand	0	83	972.00	889.00
38607	Sand	0	104	979.00	875.00
38608	Sand	0	93	973.00	880.00
38609	Sand	0	11	992.00	981.00
38609	Silt/Clay	11	15	981.00	977.00
38609	Sand	15	60	977.00	932.00
38610	Sand	0	63	993.00	930.00
38611	Sand	0	92	989.00	897.00
38615	Sand	0	73	987.00	914.00
38618	Sand	0	81	967.00	886.00
38626	Sand	0	103	974.00	871.00
38631	Sand	0	20	971.00	951.00
38631	Silt/Clay	20	30	951.00	941.00
38631	Sand	0	84	941.00	887.00
38631	Hillslope/Bedrock	84	84	887.00	887.00

Table 2.1. Summary of geological materials found in boreholes in the Port Edwards Priority Watershed (continued).

Location code	Material	Depth from	Depth to	Elevation from	Elevation to
38633	Sand	0	25	996.00	971.00
38633	Silt/Clay	25	35	971.00	961.00
38633	Sand	35	72	961.00	924.00
38633	Hillslope	72	72	924.00	924.00

The top of the uppermost silty and clayey layer, the New Rome Member of the Big Flats formation, occurs 13 to 39 ft below the surface at elevations between 941 and 974 ft MSL (Figures 2.5 and 2.6). Its thickness is greatest in Sections 8 and 18, reaching 9 ft in one area. More typically it is 3 to 4 ft thick (Figure 2.7). Two discontinuities were observed in the New Rome, one in the west-central region, and the second in the extreme southeast (Figures 2.8B, C, D). The New Rome topography undulates and has significant lows in the east-central and west-central regions.

Beneath the New Rome Member at elevations of 920 to 935 ft MSL, a discontinuous second lacustrine unit was detected (Figure 2.9). Its thickness is 2–3 ft, but it is apparently absent in the northwest, east-central, and southwest regions of the study area.

The aquifer bottom (bedrock or hillslope deposits) marks the base of the Pleistocene unit and Pleistocene aquifer. The surface is irregular, with a minimum of 100 ft of relief occurring in 3 km of horizontal distance. Because the bedrock or hillslope deposits represent a former erosional surface, aquifer bottom contours would be expected to show a dendritic drainage pattern. However, many more observations would be needed to establish such a pattern. Rather, the aquifer bottom appears as an undulating surface, with highs in the north and lows in the south and east.

Aquifer Extent

The aquifer consists primarily of sandy and gravelly Pleistocene deposits. Its vertical extent is from the water table down to the top of the hillslope deposits or bedrock. The saturated thickness of the aquifer above the New Rome Member ranges from -18 to 26 ft, where the negative value indicates the water table is below the New Rome Member. The maximum saturated thickness above the New Rome Member occurs in the northwest region of the study area and the minimum in the southeast (Figure 2.10). The New Rome and the deeper lacustrine unit may be locally confining and affect groundwater flow and solute transport.

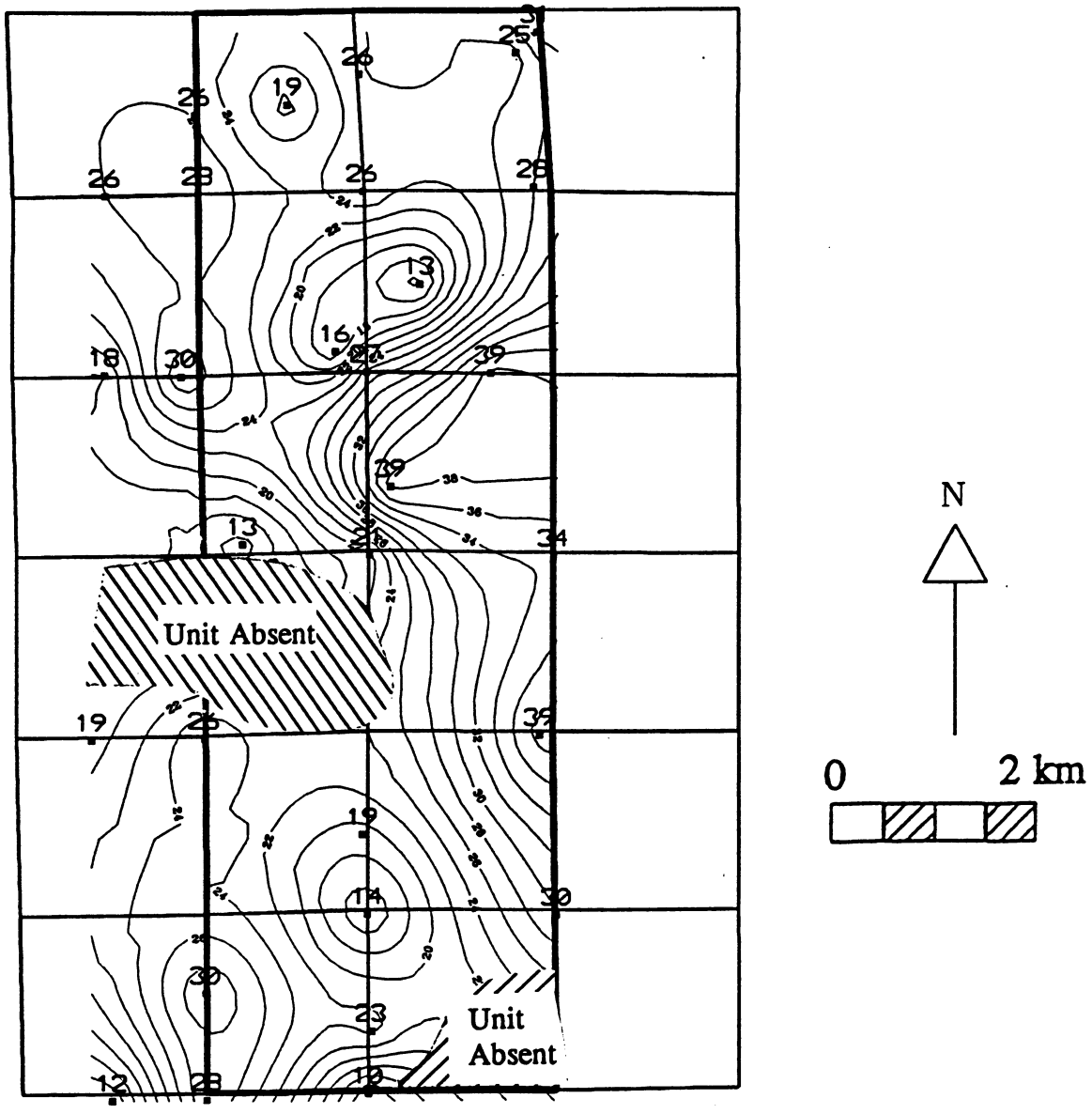


Figure 2.5 Depth to New Rome Member, ft

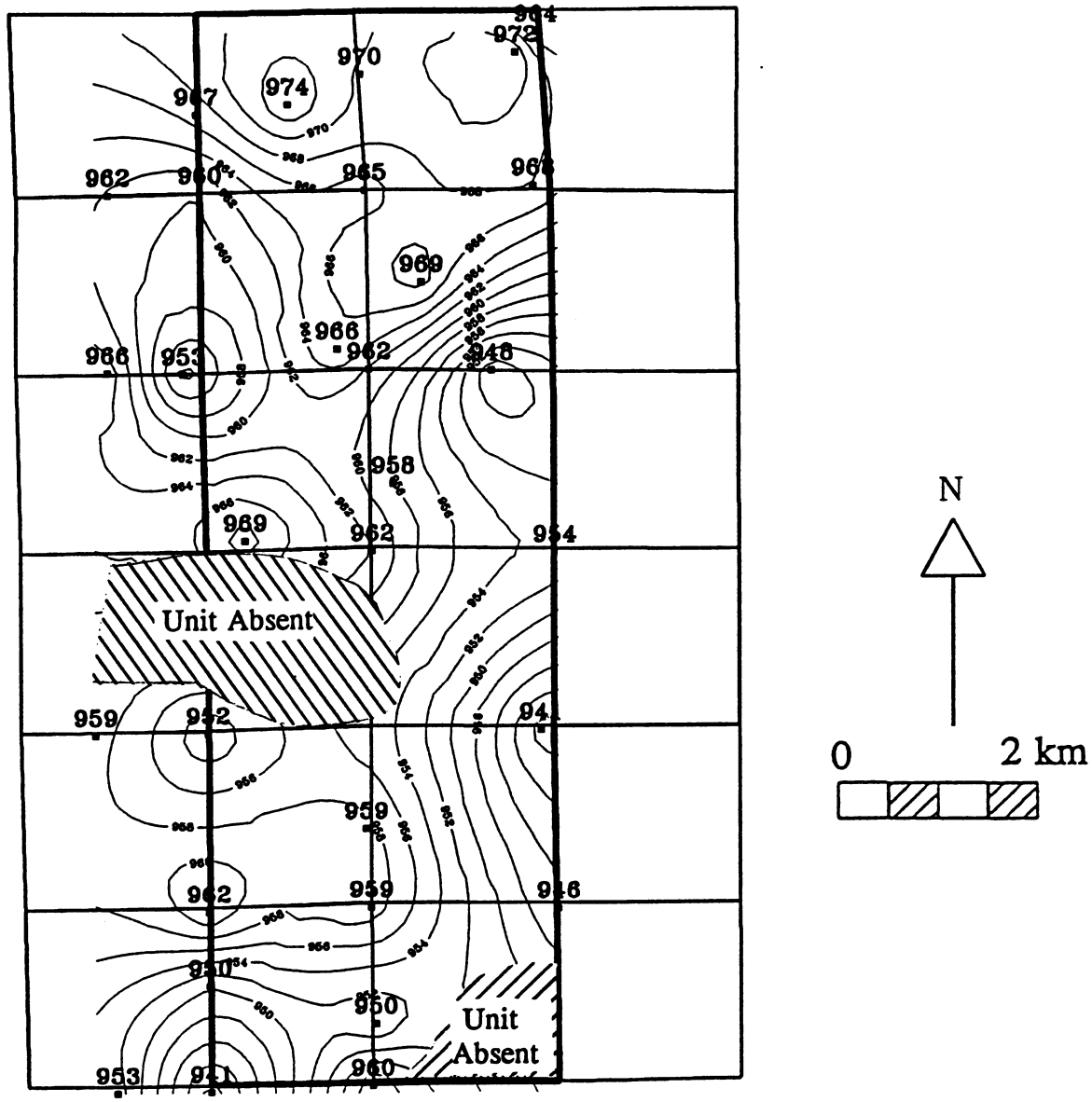


Figure 2.6 Elevation of New Rome surface, ft AMSL

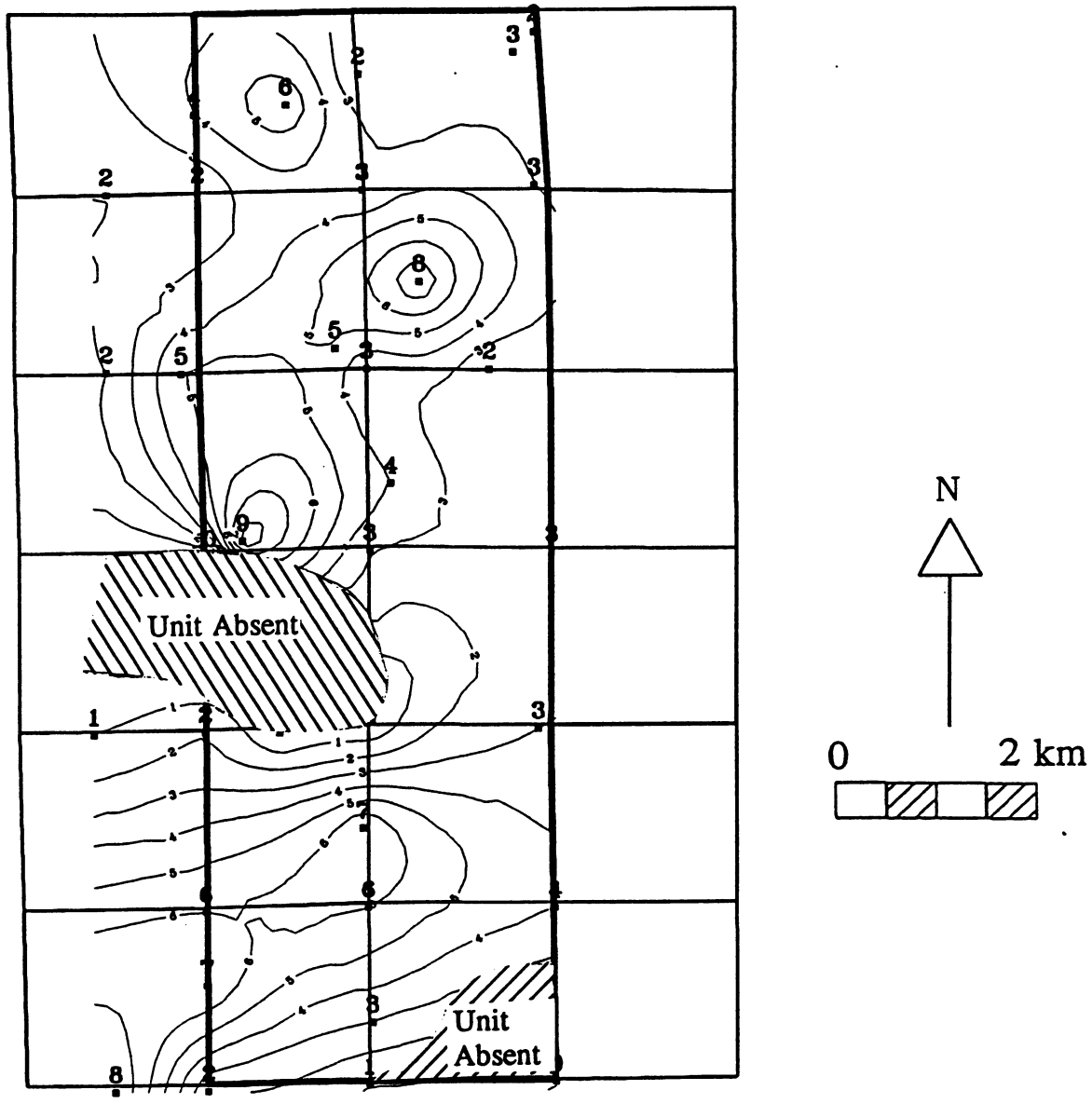


Figure 2.7 Thickness of New Rome Member, ft

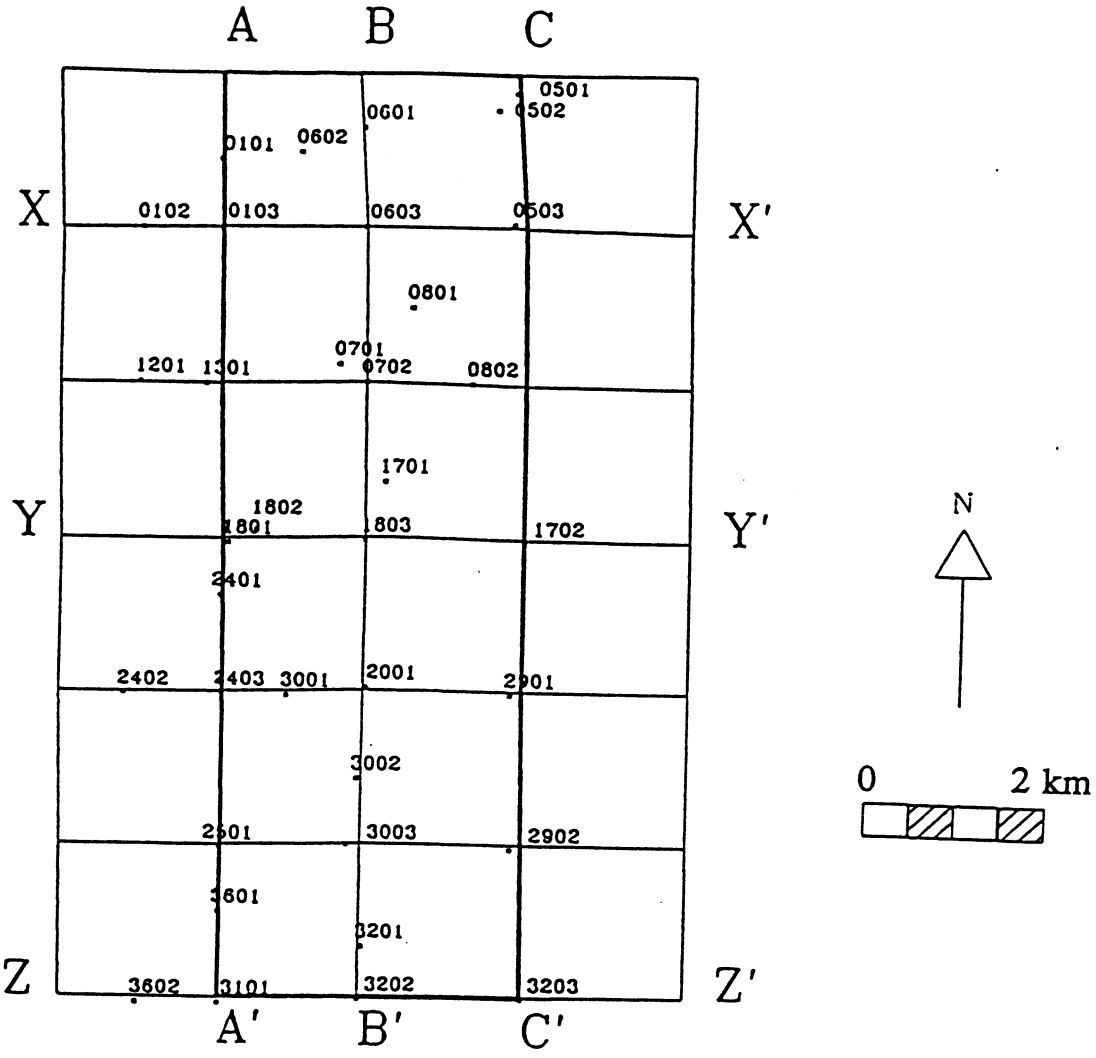


Figure2.8A Index to geologic cross-sections, Port Edwards Groundwater Priority Watershed

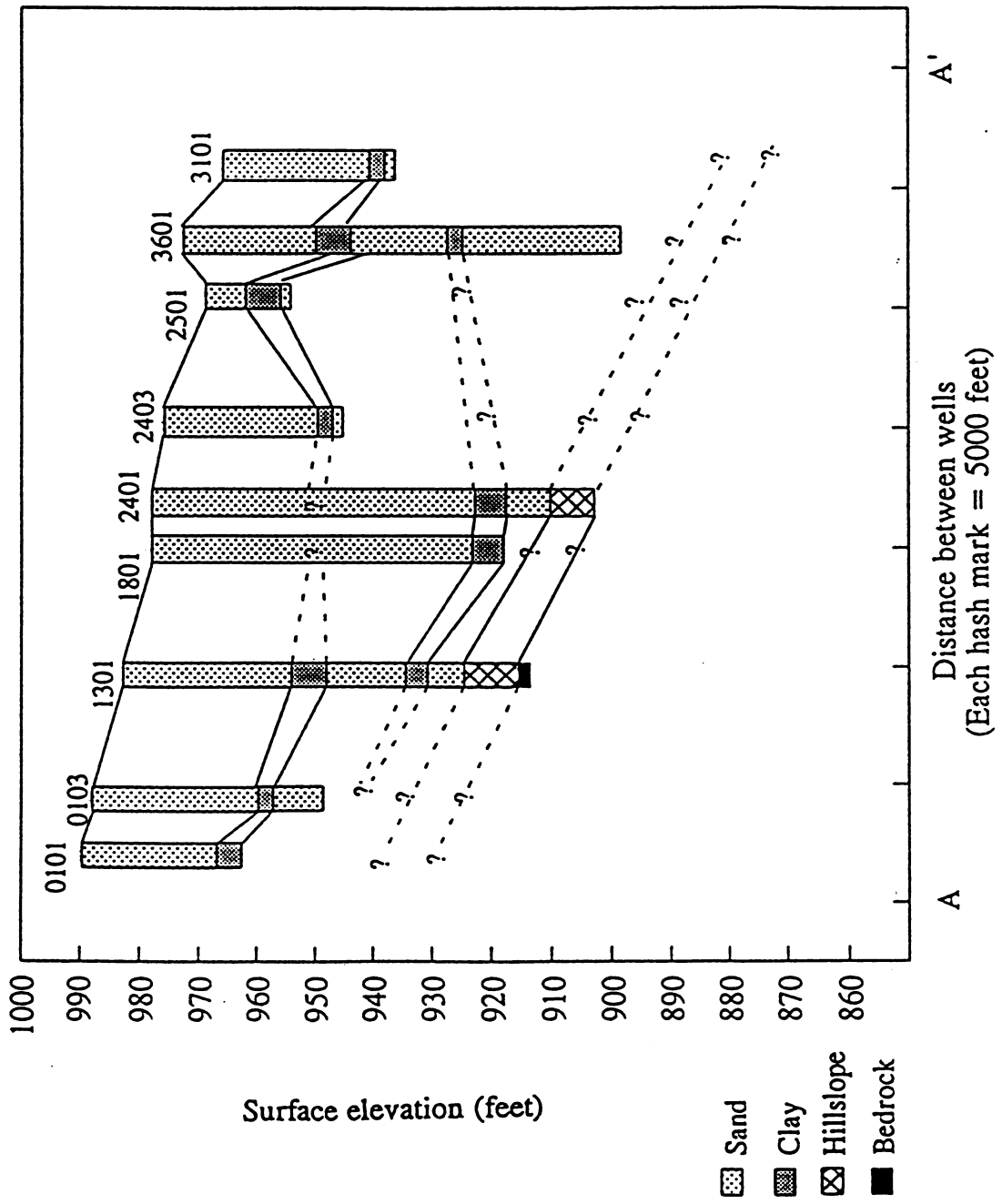


Figure 2.8B Geologic cross-section A-A', Port Edwards Groundwater Priority Watershed

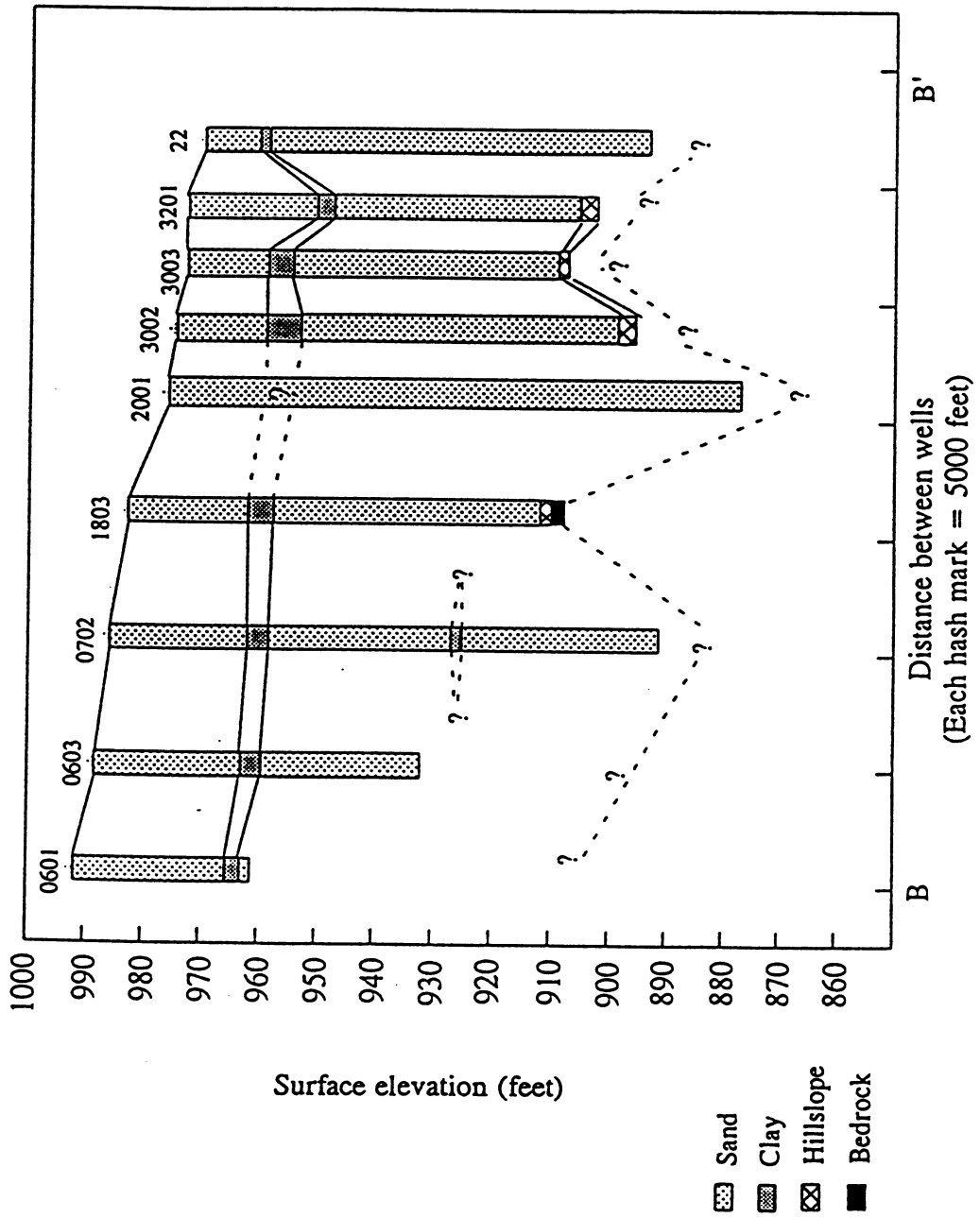


Figure 2.8C Geologic cross-section B-B', Port Edwards Groundwater Priority Watershed

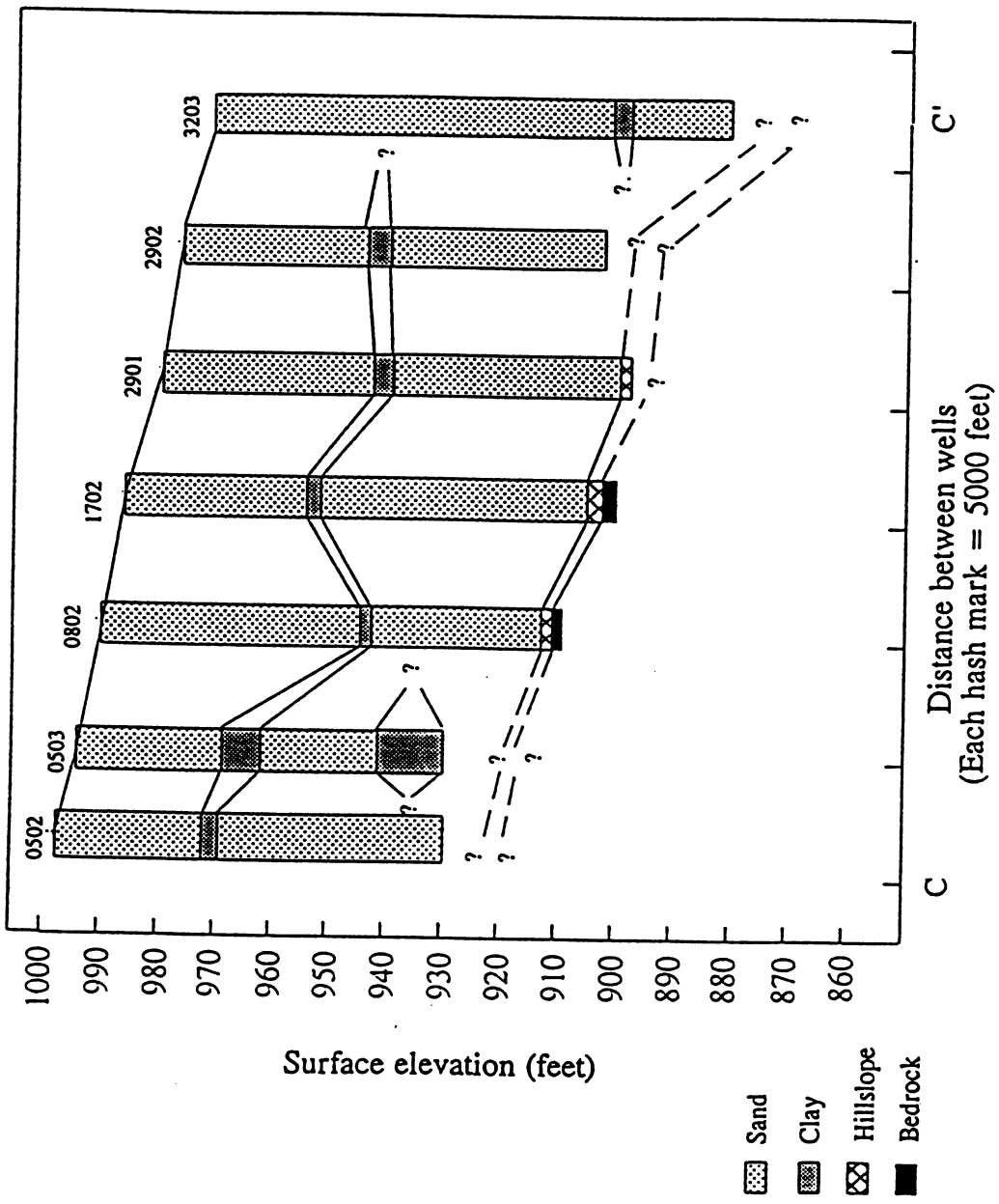


Figure 2.8D Geologic cross-section C-C', Port Edwards Groundwater Priority Watershed

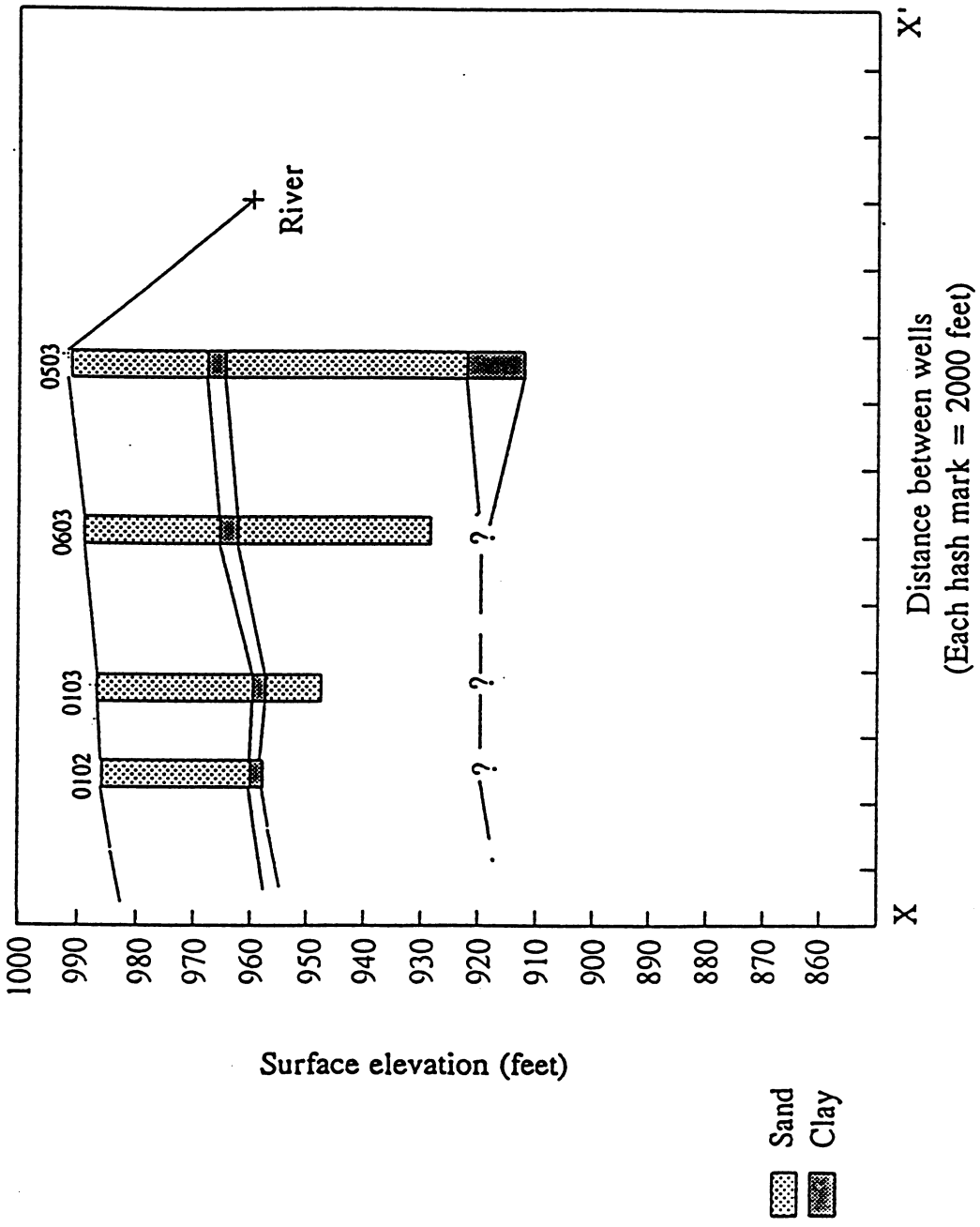


Figure 2.8E Geologic cross-section X-X', Port Edwards Groundwater Priority Watershed

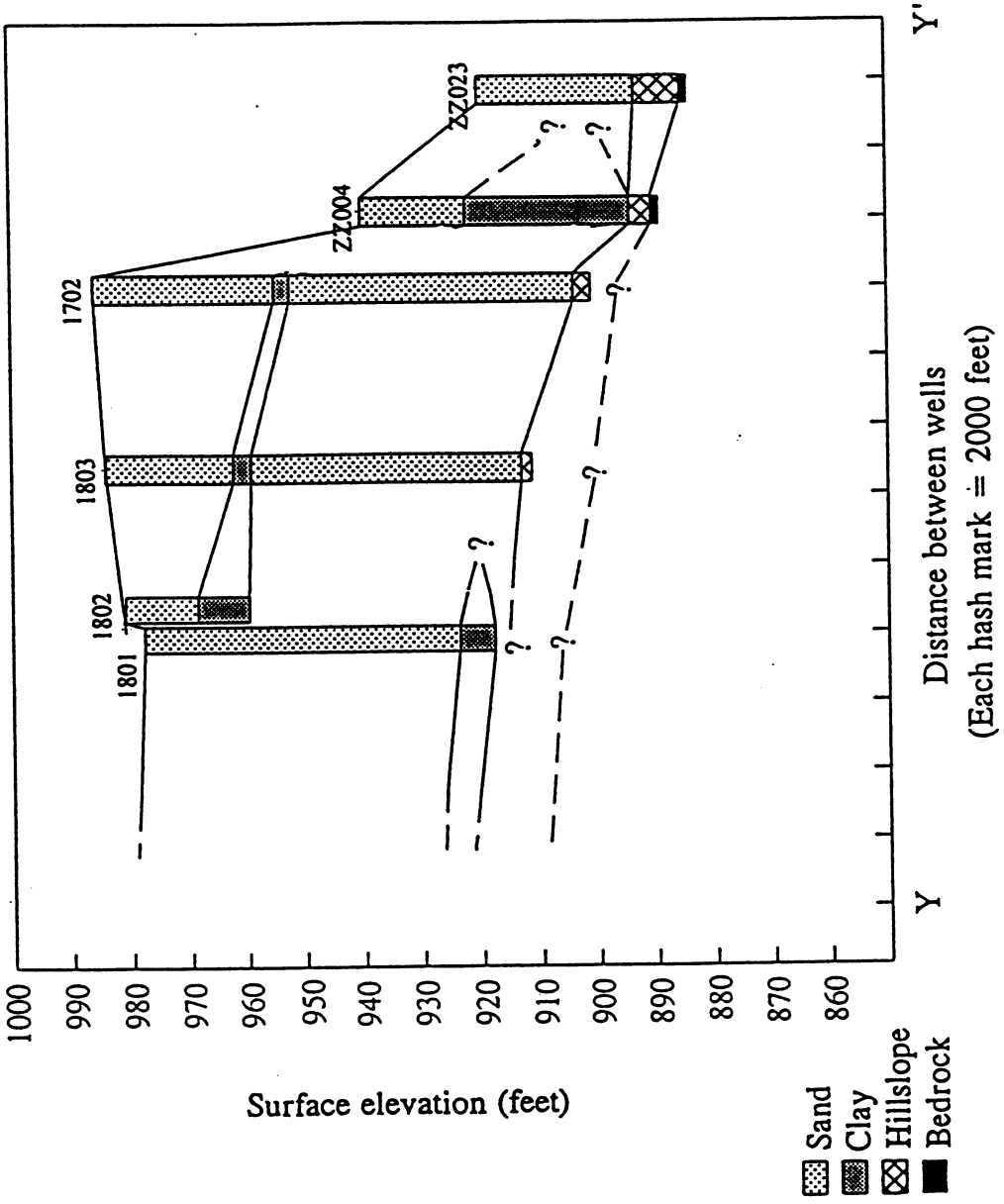


Figure 2.8F Geologic cross-section Y-Y', Port Edwards Groundwater Priority Watershed

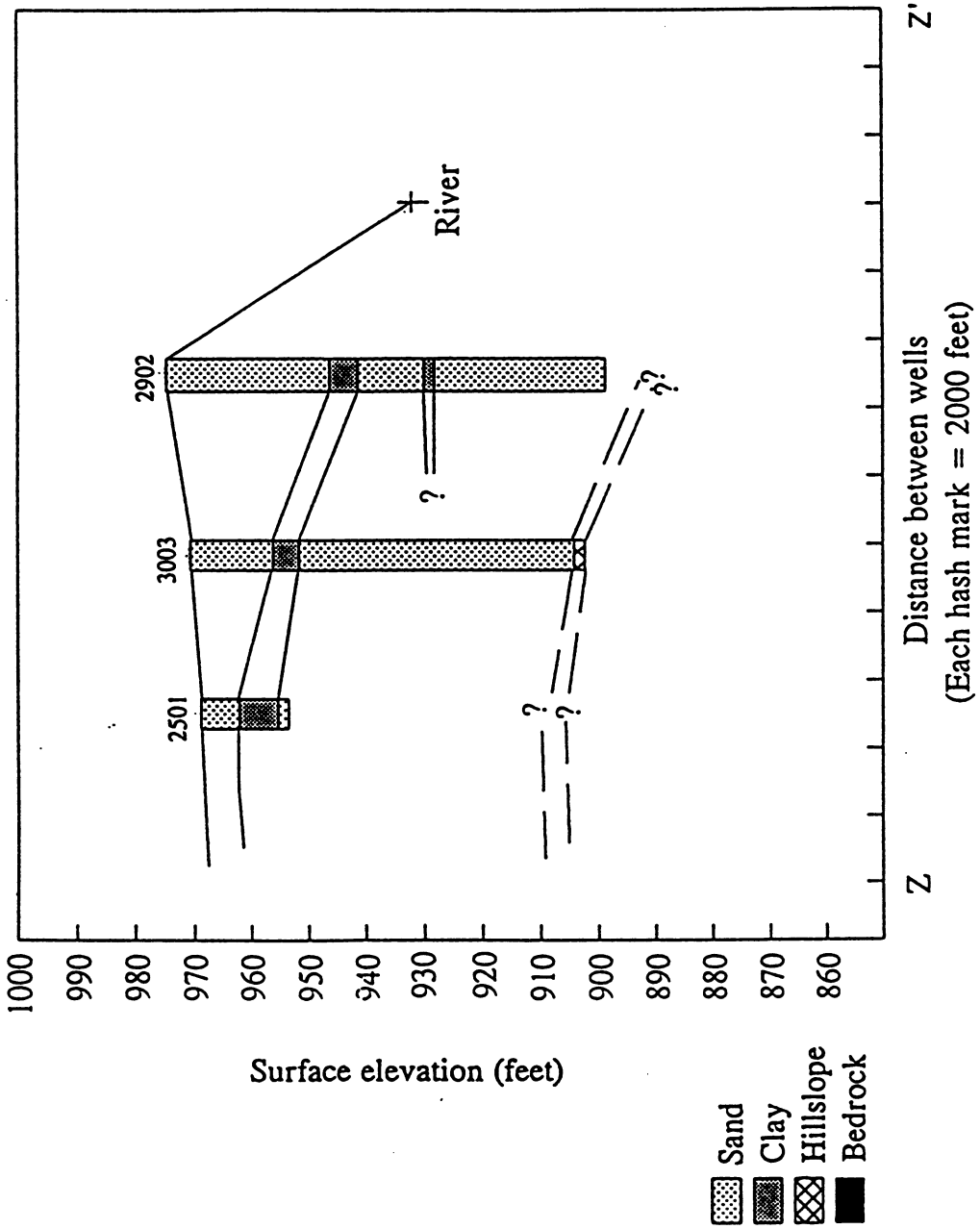


Figure 2.8G Geologic cross-section Z-Z', Port Edwards Groundwater Priority Watershed

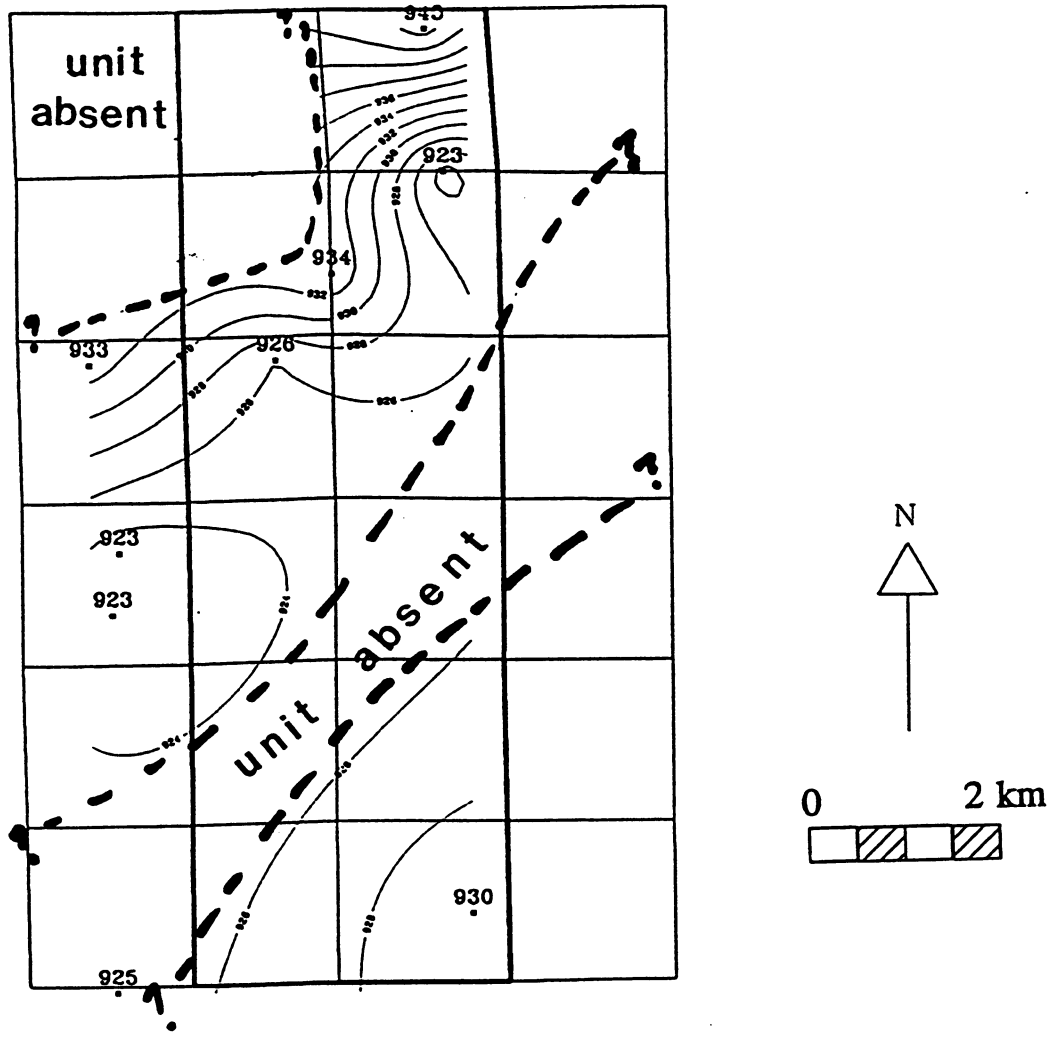


Figure 2.9 Elevation of lower silty lacustrine unit, ft AMSL

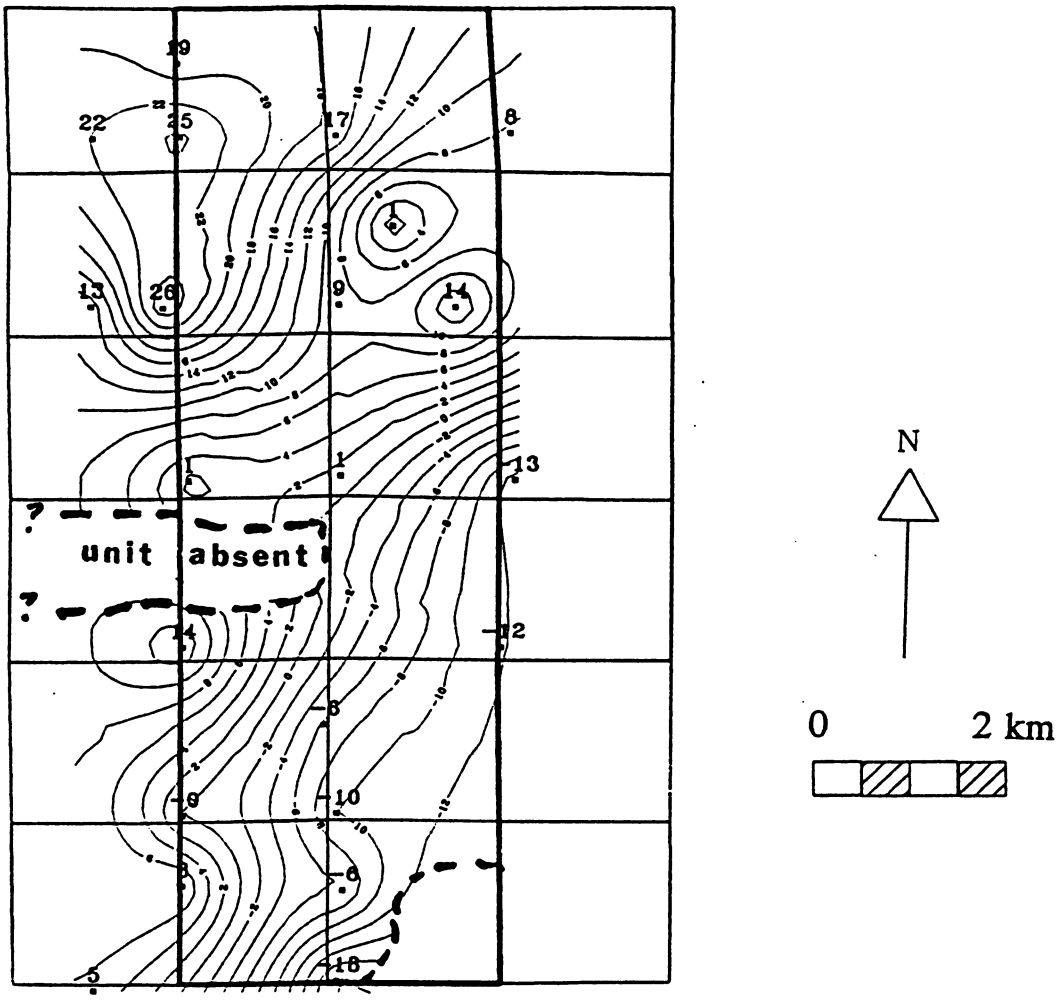


Figure 2.10 Aquifer saturated thickness above New Rome Member, ft AMSL

Sediment characteristics

Chemistry

Figure 2.11 shows profiles of pH, loss on ignition, and organic matter as determined by titration for samples from the Creamery Road and Kimball Avenue borehole, location code 2001. The pH was slightly below neutral near the surface. It increased sharply to 8.1 between 28 and 33 ft depth. No obvious textural change accompanied the pH discontinuity, and it may reflect simply different land uses in the areas contributing recharge to these layers. The loss-on-ignition profile contains four peaks that coincide with the samples containing more than 4% silt plus clay. The loss on ignition could represent water driven off by heating of clay minerals. This seems to be borne out by Figure 2.12, which shows a correlation between the amount of fines and the loss on ignition. However, the relationship only holds for the samples with low contents of fines, which could indicate a different mineralogy in the silty layers than in the sands. The organic-matter contents were quite low, averaging 0.05%, in all but the surface layer.

Figure 2.13 shows phosphorus and potassium concentrations. These concentrations are extractable rather than total amounts. The potassium peaks correspond with layers having greater contents of fines, while phosphorus is more uniform and does not appear related to texture.

Color

Colors at the Creamery Road and Kimball Avenue borehole were generally described as shades of yellowish brown. The exceptions were some red material, possibly banded, found at 26 ft depth, and gray-blue hues in the samples taken from 68 and 73 ft depth. The reddish bands were just above the depth where pH increased suddenly (as shown in Figure 2.11) and the gray-blue zone was below the stratum with the greatest silt and clay contents.

Grain-size analysis

Results of sieve analysis of samples from the site near the intersection of Creamery Road and Kimball Avenue are shown in Figure 2.14, and hydrometer results are in Figure 2.15. Figure 2.14 shows that medium sand predominates in most of the samples. The sands were well sorted, with the dominant size class containing at least 40% of the total material, and at least 70% of material in the two main size classes. Three samples stand out from the others, as seen in Figure 2.15. Sample 1010, at a depth of 43 ft, was 88% sand, 8% silt, and 4% clay by hydrometer analysis. The USDA texture class is still "sand", but this content of finer particles could reduce hydraulic conductivity considerably. Sample 1014, 63 ft below the surface, was 34% sand, 52% silt and 14% clay, with a USDA texture of "silt loam". Sample 1017, 78 ft below the surface, was

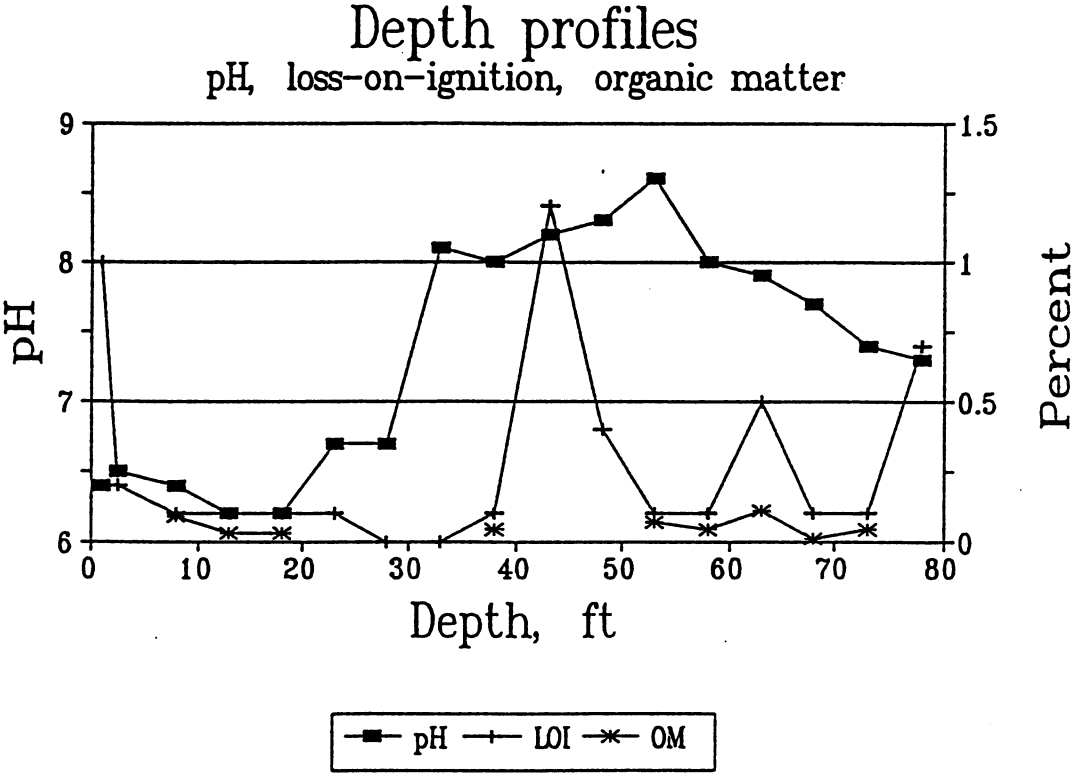


Figure 2.11 Depth profiles of pH, loss on ignition, and organic-matter content, Kimball Ave. and Creamery Rd. borehole.

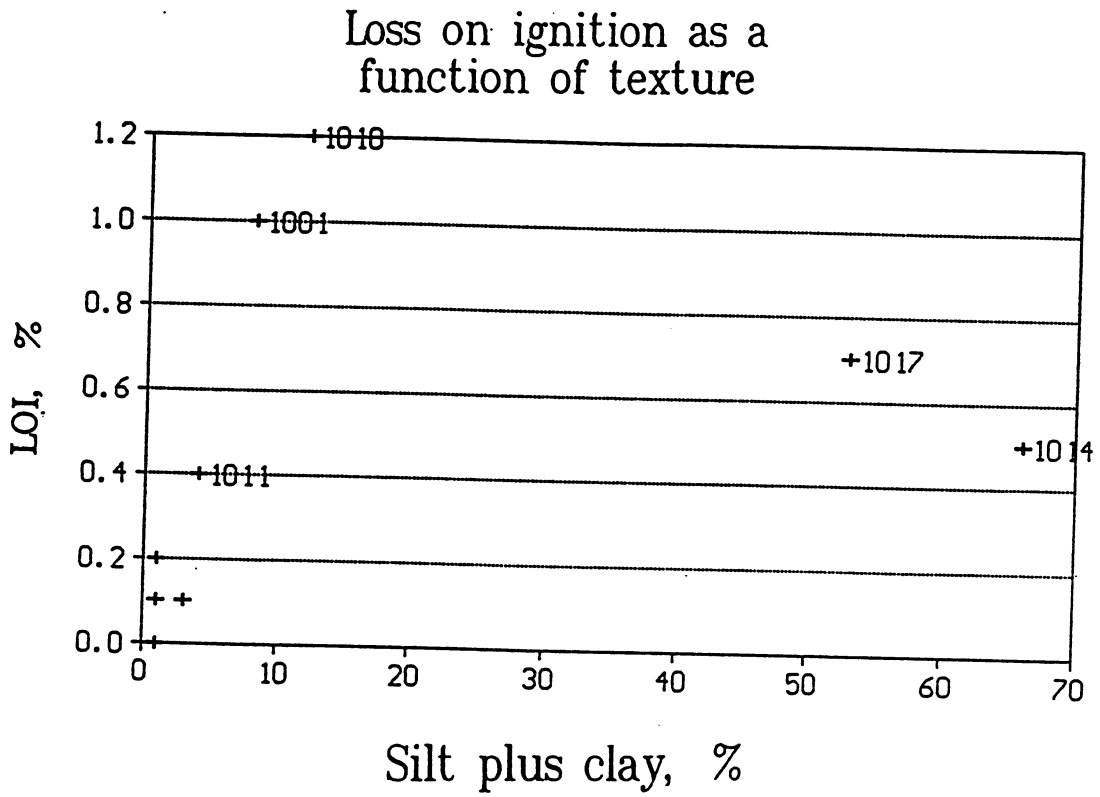


Figure 2.12 Loss on ignition vs. content of particles smaller than sand, Kimball Ave. and Creamery Rd. borehole.

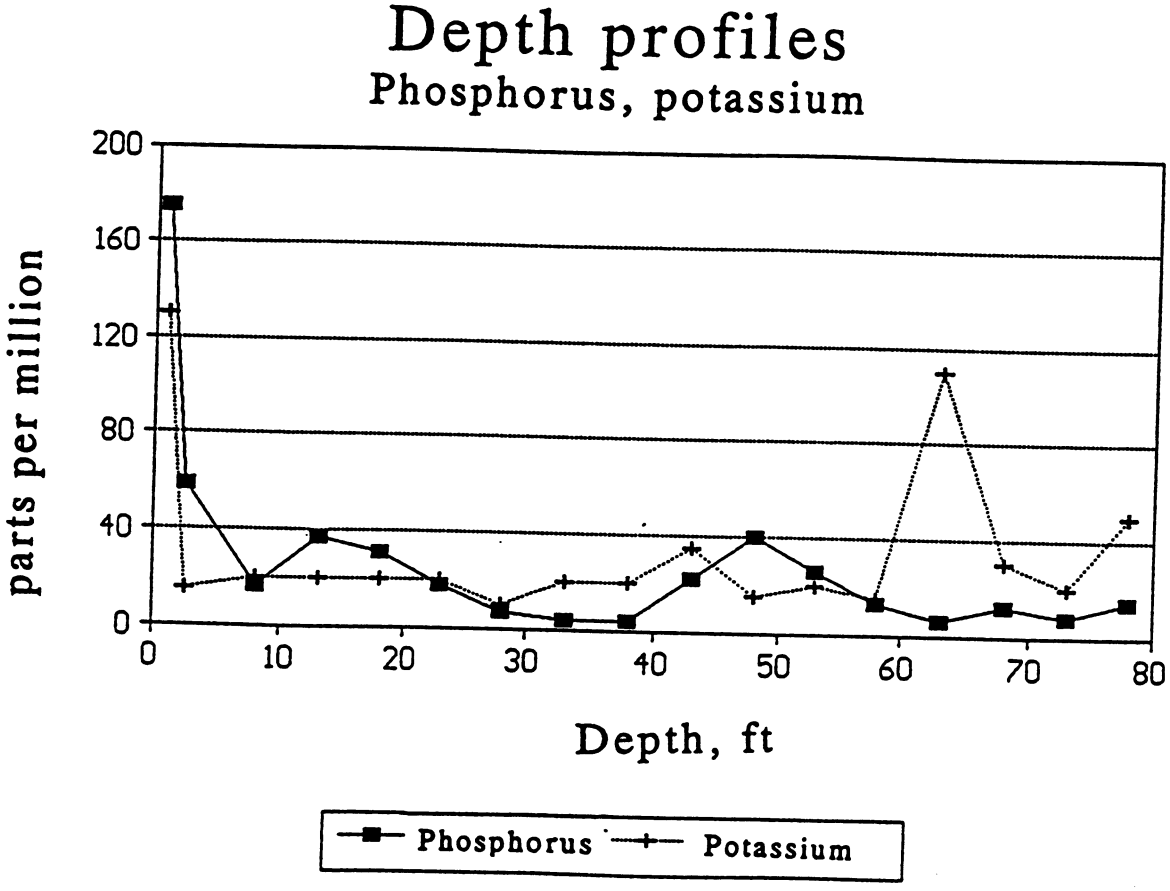


Figure 2.13 Depth profiles of readily desorbed phosphorus and exchangeable potassium, Kimball Ave. and Creamery Rd. borehole.

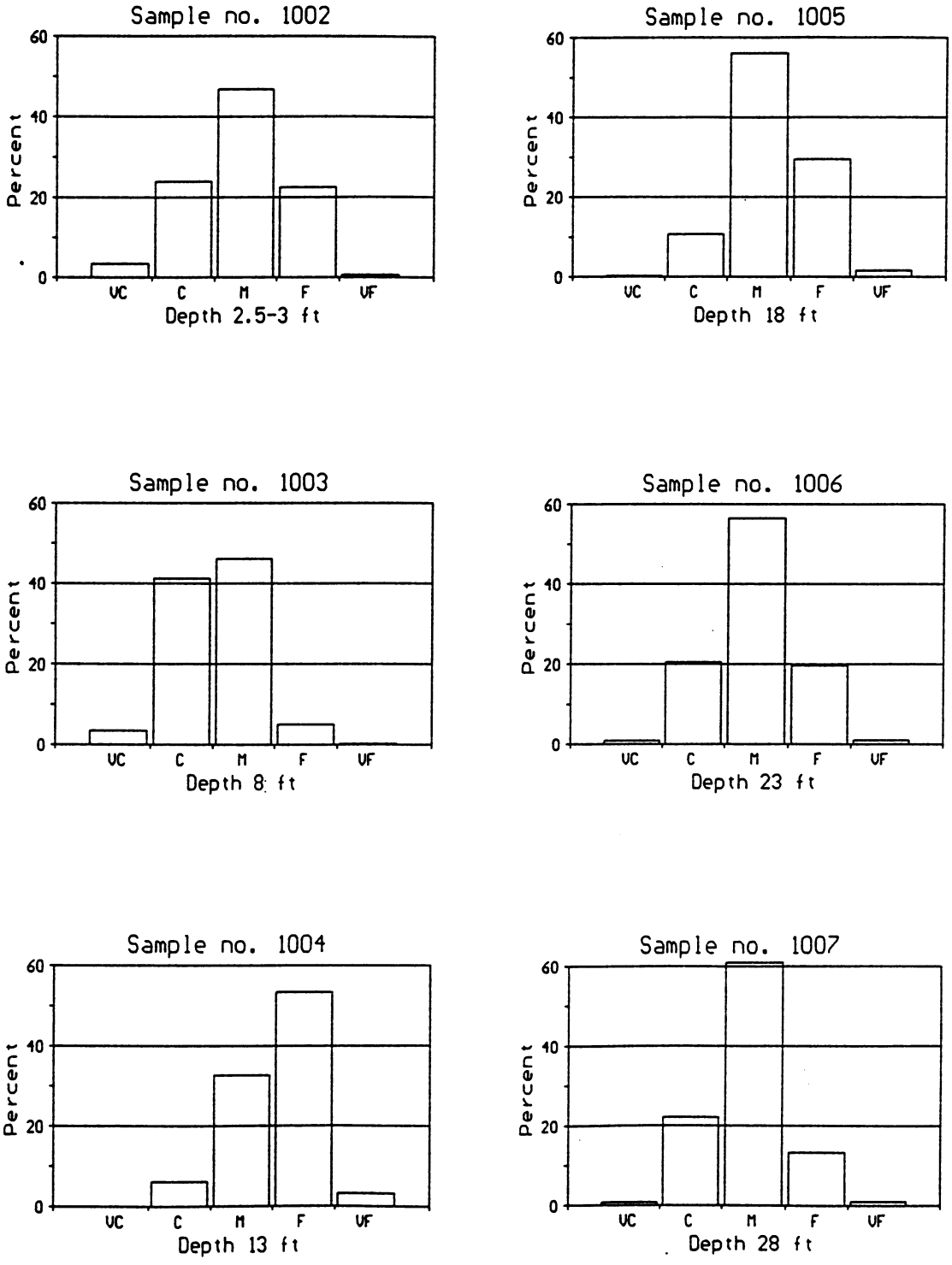


Figure 2.14 Particle-size classes determined by sieving of samples from Kimball Ave. and Creamery Rd. borehole.

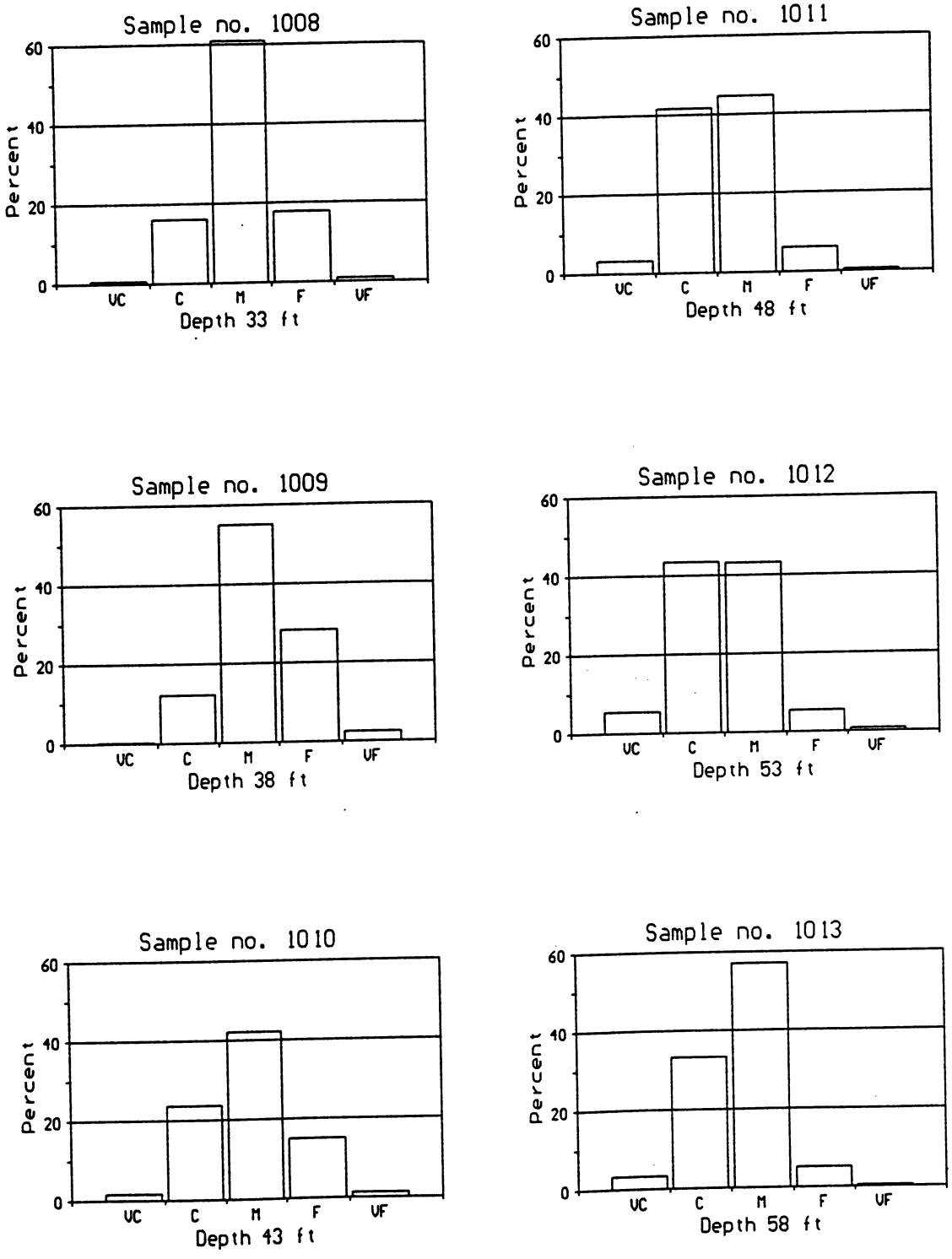
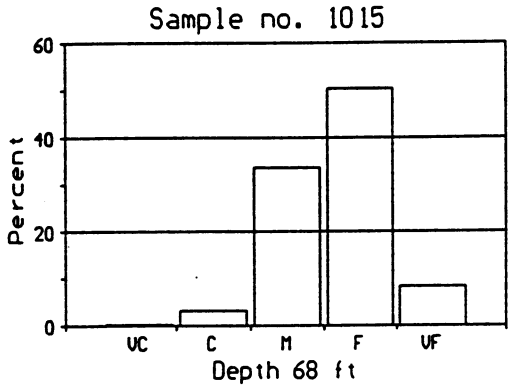


Figure 2.14 Particle-size classes determined by sieving of samples from Kimball Ave. and Creamery Rd. borehole (continued).

**Sample no. 1014 was not sieved;
it contained only 34% sand.**



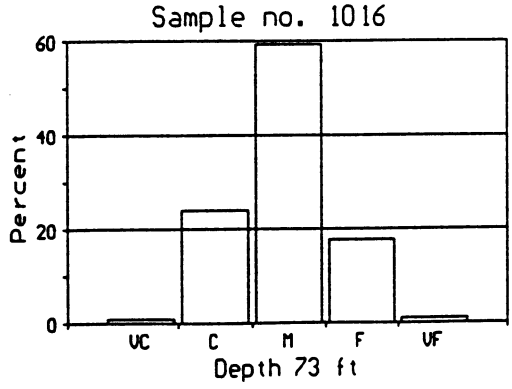


Figure 2.14 Particle-size classes determined by sieving of samples from Kimball Ave. and Creamery Rd. borehole (continued).

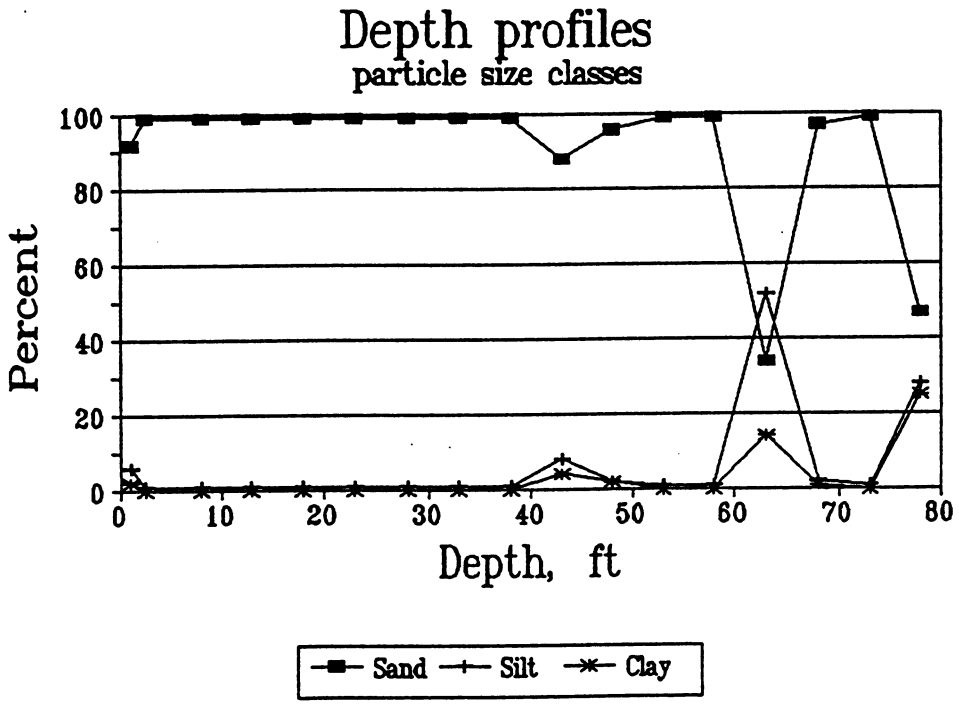


Figure 2.15 Particle size class variations with depth from Kimball Ave. and Creamery Rd. borehole.

"loam", with 47% sand, 28% silt, and 25% clay. These silt loam and loam layers, if extensive, would strongly impede subsurface water movement. According to the drilling log, the New Rome layer was not found, unless it was represented by a reddish layer of sand somewhat finer than the adjacent layers at about 28 ft.

The coefficient of nonuniformity, U, is defined as the ratio d_{60}/d_{10}, where x% of the sediment is smaller than d_x. Coefficients of nonuniformity were estimated for the samples which had been sieved by inspecting graphs such as Figure 2.16 to estimate d_{60} and d_{10}. The median grain size d_{50} was also estimated this way. The results are shown in Table 2.2. With one exception, the samples fell into a narrow range of U, from 1.8 to 2.8. The exception is sample no. 1010, with U = 12.6. The four distributions shown in Figure 2.16 were chosen because sample no. 1002 had the greatest value of U (2.8), no. 1013 had the smallest value (1.8), no. 1008 was closest to the mean (2.3), and no. 1010 was exceptional (12.6). Except for no. 1010, it is hard to distinguish much difference in the graphs. Jumikis (1984) states that sands typically have U in the range of 10 to 20, which implies that the sand from this borehole is distinctly uniform or well sorted. This differs from Brownell's (1986) description of Big Flats sand deposits as moderately well sorted.

Table 2.2. Estimated d_x and nonuniformity indices of samples taken from the Creamery Road and Kimball Avenue site.

Sample	d_{60}	d_{10}	d_{50}	U
1002	0.83	0.30	0.71	2.8
1003	1.20	0.50	0.91	2.4
1004	0.50	0.24	0.42	2.1
1005	0.73	0.27	0.60	2.7
1006	0.80	0.32	0.70	2.5
1007	0.83	0.41	0.73	2.0
1008	0.77	0.34	0.70	2.3
1009	0.71	0.28	0.61	2.5
1010	0.78	0.06	0.67	12.6
1011	1.10	0.46	0.91	2.4
1012	1.15	0.51	0.98	2.3
1013	0.94	0.51	0.83	1.8
1015	0.46	0.18	0.41	2.6
1016	0.82	0.40	0.74	2.1

Typical results of particle-size analyses of samples collected in the intensively studied fields are shown in Figure 2.17. Except for the 0-1 ft interval, these samples contained well over 95% sand, and the median grain size was in the medium (250-500 μm) to coarse (500-1000 μm) size range. The steep distribution curves indicate well-sorted sediment. Graphs like those in Figure 2.17 were inspected to estimate a median

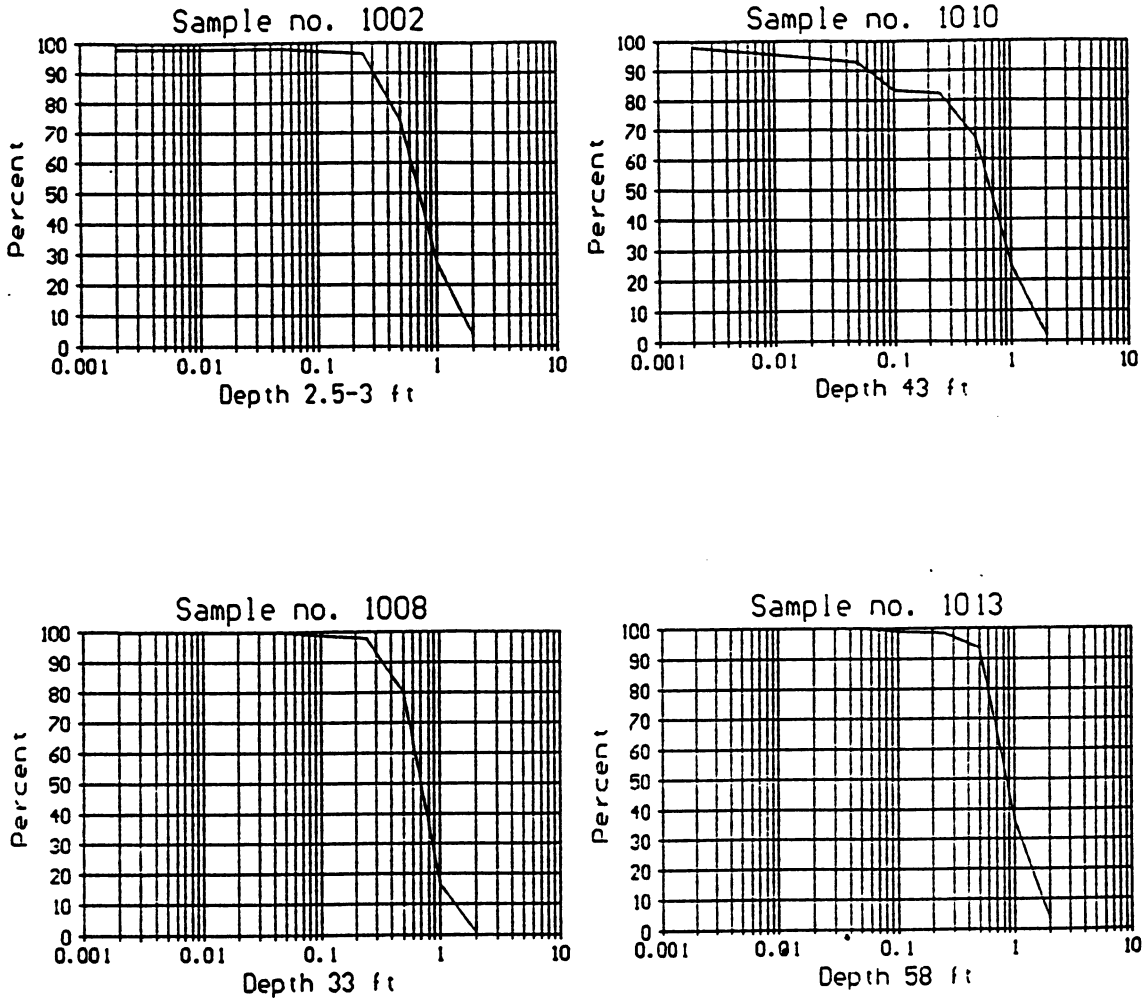


Figure 2.16 Cumulative particle-size plots for four samples from Kimball Ave. and Creamery Rd. borehole.

particle size for each of the 40 sieved samples; the results are shown in Table 2.3, and averages for the depth intervals are in Table 2.4. The largest median particle diameter among these samples was 1300 μm, and the smallest was 700 μm. Brownell (1986) reported mean grain sizes between 175 and 375 μm for Big Flats sand.

The 0-1 ft depth interval contained significant amounts of silt and clay, as shown in Table 2.5. Contrary to our expectation, the sieve analyses showed greater silt + clay contents than the hydrometer analyses. Evidently, shaking was more effective at breaking down composite particles than was dispersion in the hydrometer solution of sodium hexametaphosphate.

Table 2.3. Median particle diameters (mm) of samples taken from the irrigated fields, estimated from particle-size distribution graphs.

Field	Site	Depth	Median diam.	Field	Site	Depth	Median diam.
1	1	0-1	0.70	2	3	5-7	0.83
1	1	1-3	0.88	2	5	5-7	0.83
1	1	3-5	0.90	2	6	3-5	1.10
1	2	0-1	0.83	2	6	5-7	1.00
1	2	1-3	1.00	2	6	7-10	1.20
1	2	3-5	0.91	3	1	3-5	1.10
1	2	5-7	0.74	3	2	1-3	0.85
1	3	0-1	0.80	3	2	5-7	0.80
1	3	1-3	0.81	3	3	5-7	1.05
1	3	3-5	0.82	3	4	3-5	1.00
1	3	5-7	0.88	3	5	0-1	0.78
1	3	7-8	0.98	3	5	3-5	1.30
1	4	0-1	0.74	3	5	5-7	0.83
1	4	1-3	0.90	3	6	1-3	0.85
1	4	3-5	0.74	4	1	1-3	0.92
1	4	5-7	0.83	4	1	3-5	0.84
1	5	0-1	0.72	4	2	0-1	0.77
1	5	1-3	0.82	4	3	0-1	1.05
1	5	3-5	1.00	4	6	0-1	0.73
1	6	0-1	0.71				
1	6	1-3	0.87				
1	6	3-5	1.10				

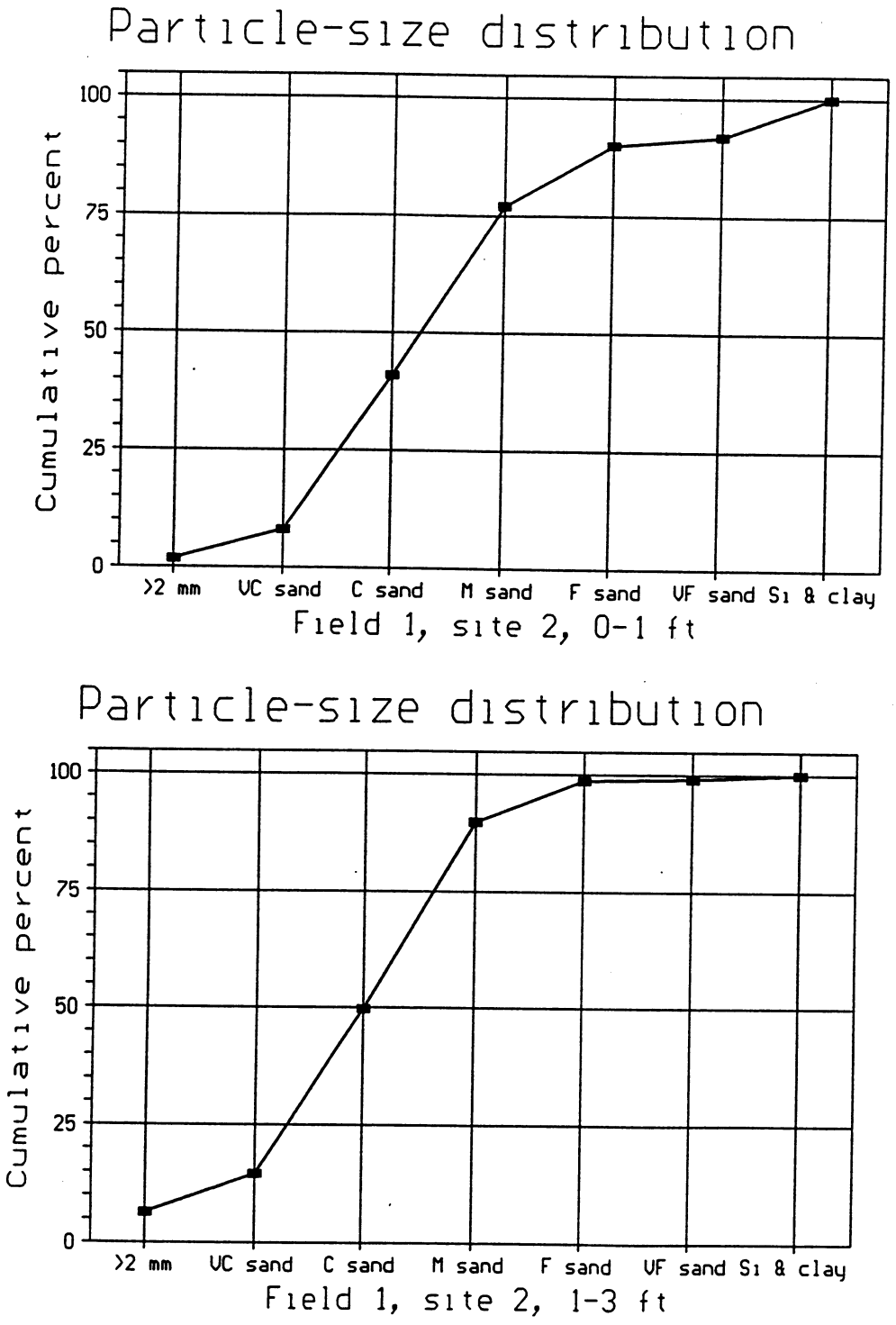


Figure 2.17 Particle size distributions from four depth intervals in an irrigated field.

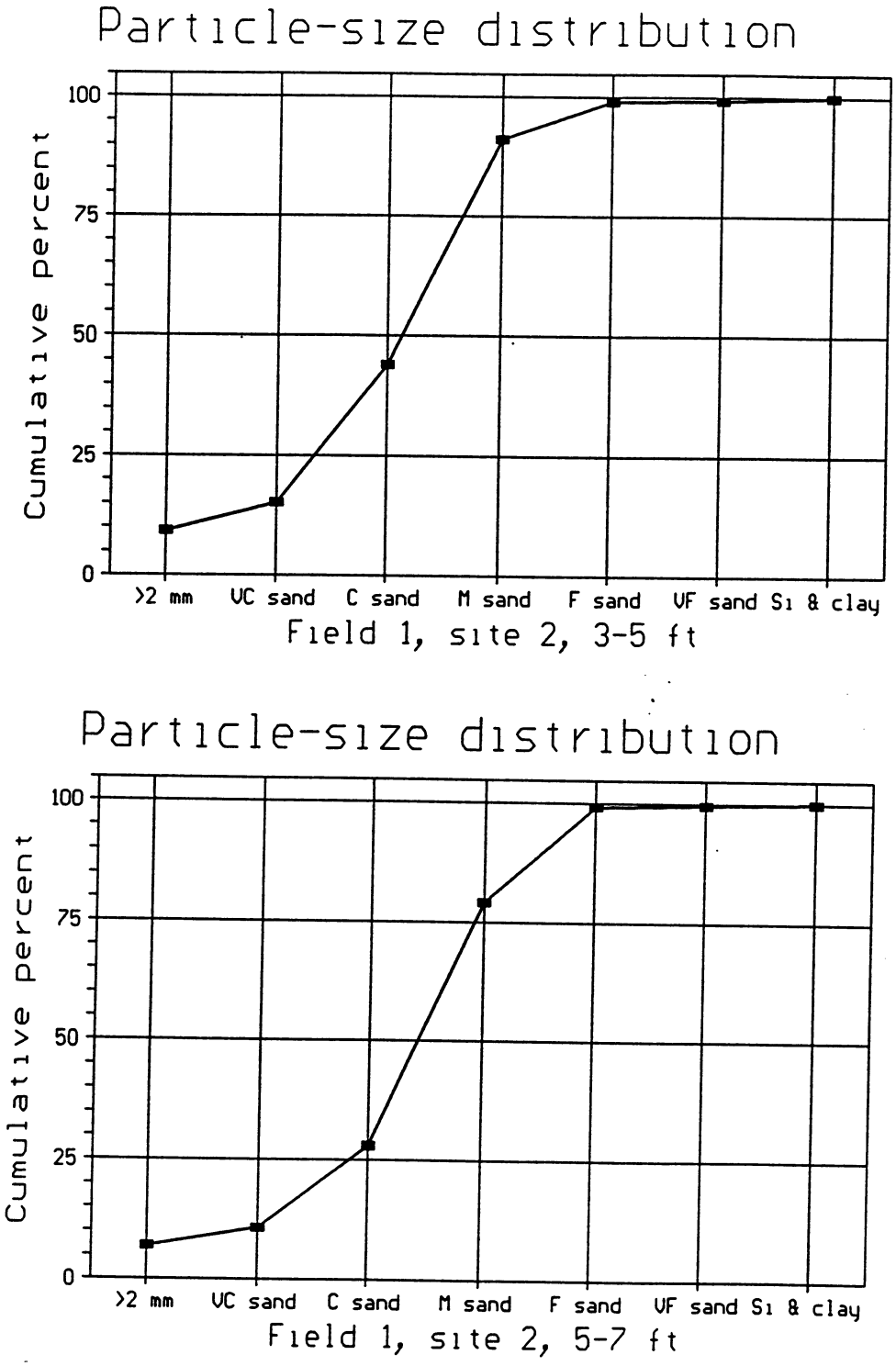


Figure 2.17 Particle size distributions from four depth intervals in an irrigated field (continued).

Table 2.4. Median particle diameters of samples taken from the irrigated fields, averaged for the sampled depth intervals.

Depth interval, ft.	Median grain diameter, mm	n
0-1	0.783	10
1-3	0.878	9
3-5	0.983	11
5-7	0.866	9
7+	1.090	2

Analysis of variance among the depth intervals showed that significant differences existed (Table 2.6), but the differences showed little clear pattern. As a rule of thumb, the median grain size was smaller in the surface layer than in deeper layers. Clearer results might result from collecting samples on the basis of morphological horizons instead of arbitrary depths.

Table 2.5. Comparison of sub-sand-size particle results from sieve analyses with hydrometer analysis for A_p horizon samples taken from the irrigated fields.

	Marshfield hydrometer results			Silt + clay,
	%silt	%clay	Sum	from sieving
Field 1, site 1	9	9	18	7.7
Field 1, site 2	9	7	16	7.0
Field 1, site 3	6	8	14	8.4
Field 1, site 4	7	8	15	8.5
Field 1, site 5	8	9	17	8.7
Field 3, site 5	6	8	14	6.0
Field 4, site 2	6	6	12	4.8
Field 4, site 3	4	8	12	3.8
Field 4, site 6	5	7	12	7.7

Table 2.6. Analysis of variance for median particle diameter in different depth intervals in the irrigated fields.

Source	DF	SS	MS	F
Between groups	4	0.296	0.074	5.505
Within groups	36	0.484	0.013	p=0.0015
Total	40	0.779		

Porosity

A summary of the porosity results appears in Table 2.7. The mean ± standard deviation found for C horizon material was 0.392 ± 0.077. Correcting for compaction brought the result to 0.403 ± 0.079.

Weeks (1969) reported porosity of 0.32, based on samples from one boring, where porosity ranged from 0.28 to 0.36, with a mean ± standard deviation of 0.32 ± 0.03, and an oral communication from C.B. Tanner that the porosity of soils in the area ranges from 0.32 to 0.38. Our corrected porosity is 22% greater than the Weeks's.

Stoertz (1985) measured porosity at several depths from 0 to 6 m below the surface at a site near Wisconsin Rapids. Her four samples below the zone of strong pedogenic changes had a porosity of 0.389 ± 0.015 (mean ± standard deviation). Our mean porosity is 3.5% greater than this.

The likeliest source of error in our procedure is loss of material when the cores were opened. However, to explain the discrepancy between 0.403 and 0.32 through error alone would require that about 14 kg of sediment, corresponding to about 8.8 L of core, had been lost and not accounted for. The estimated material lost during core opening was 2.0 L, which would weigh about 3.2 kg.

CONCLUSIONS

The study area is a generally flat deposit of glacio-lacustrine sand, gravel, and silty materials. The regional slope is southwestward with a gradient of 0.001. There are five major geological units. These are the bedrock, consisting of Precambrian crystalline rock, Cambrian sandstone, hillslope sediments, Pleistocene materials, and thin Recent deposits of aeolian sand. Soils are Plainfield sand or loamy sand, and less well-drained members of the same soil catena.

The Pleistocene sediments probably originated during the late Wisconsin period, and consist primarily of well-sorted medium sand, with up to two interbedded layers of silty and clayey materials. The top of the uppermost silty and clayey layer, the New Rome Member of the Big Flats formation, occurs 13 to 39 ft below the surface. The New Rome member is interrupted in places. A discontinuous second lacustrine unit was detected beneath the New Rome Member.

The Pleistocene sands are coarser and better sorted than Brownell's (1986) description of Big Flats sand. The porosity, as determined from minimally disturbed core samples, was 0.403 ± 0.079.

Table 2.7. Porosity of C horizon samples collected with the vibra-core method.

Site	Drive no.	section	Porosity by displacement	Porosity by density	Porosity by density, corrected for compaction
VB1	1	2	0.305	0.339	0.361
VB1	1	3	0.352	0.374	0.395
VB1	2	2	0.354	0.378	0.399
VB1	2	3	0.358	0.381	0.402
VB2b	1	1	0.410	0.431	0.450
VB2b	1	2	0.341	0.360	0.382
VB2b	2	1	0.491	0.512	0.528
VB2b	2	2	0.374	0.391	0.411
VB2b	2	3	0.369	0.391	0.411
VB2b	3	1	0.397	0.420	0.439
VB2b	3	2	0.374	0.392	0.412
VB2b	3	3	0.416	0.427	0.446
VB2b	4	1	0.388	0.405	0.425
VB2b	4	2	0.376	0.400	0.420
VB2b	4	3	0.326	0.349	0.371
VB3	1	2	0.345	0.370	0.391
VB3	1	3	0.385	0.399	0.419
VB3	2	2	0.321	0.354	0.376
VB3	2	3	0.310	0.340	0.362
VB3	3	2	0.368	0.387	0.408
VB3	3	3	0.327	0.347	0.369
VB3	3	4	0.344	0.352	0.374
VB3	4	2	0.353	0.368	0.389
VB3	4	3	0.359	0.379	0.400
VB3	5	3	0.408	0.420	0.439
VB3	5	4	0.371	0.384	0.405
VB4	1	5	0.365		0.404
VB4	2	2	0.471	0.488	0.505
VB4	2	3	0.476	0.492	0.509
VB4	4	3	0.392	0.412	0.432
VB4	4	4	0.341	0.359	0.381
VB5	2	3	0.344	0.389	0.410
VB5	3	3	0.374	0.386	0.407
VB5	3	3	0.383	0.397	0.417

CHAPTER 3

HYDROGEOLOGY

INTRODUCTION

Purpose and scope

The investigation of study area hydrogeology was performed to predict the behavior, transport, and fate of contaminants entering the groundwater system. The objectives of the investigation were to define:

- the boundaries of the groundwater system;
- water table elevations and their temporal fluctuation;
- groundwater flow paths;
- hydrogeologic properties of aquifer sediments; and
- study area hydrology.

Tasks undertaken to fulfill these objectives included:

- defining the study area geology (Chapter 2);
- reviewing available literature;
- installing a network of monitoring wells throughout the study area;
- measuring hydraulic heads on an approximately monthly basis;
- estimating aquifer hydraulic conductivity.

PREVIOUS WORK ON WCSP HYDROGEOLOGY

Hydraulic conductivity

Hydraulic conductivity (K) of the WCSP aquifer has been estimated using pumping tests, specific capacity data, basin-scale model calibration, permeameter tests, and grain size analyses (Tables 3.1 and 3.2). Pumping tests were done by Holt (1965) and by Weeks et al. (1965) about 40 km northeast of the Port Edwards project area, and by Weeks and Stangland (1971) about 30 km northeast. Specific capacity data were used by Karnauskas (1977) in the Nepco Lake watershed about 5 km east of the study area, by Rothschild (1982) for 266 high-capacity irrigation wells in a large area stretching from about 8 km east to 50 km east of the project area, and by Batten (1989) for irrigation wells in southeastern Wood County. Since the study area contains a large fraction of the irrigation wells in southeast Wood County, Batten's results probably include wells in the project area. Slug tests were done by Bradbury et al. (1992) in T21-23N, R6-9E; the bulk of their work was in T22N, R6-7E about 24 km east of the project area. Kraft (1990) also performed slug tests about 32 km northeast of the project area. Bradbury et al. (1992)

Table 3.1. Hydraulic conductivities measured in the Wisconsin Central Sand Plain.

Test type	Source	Location	Number of tests	Geometric mean (10^{-4} m s^{-1})	Range (10^{-4} m s^{-1})
Pumping	Holt (1965)	Almond-Plover	4	8.8	8.0–10.4
	Weeks et al. (1965)	Little Plover R. Basin	1	8.2	—
	Weeks and Stangland (1971)	T19–23N, R7–9E	5	8.2	4.6–17.6
	All pumping		10	8.5	4.6-17.6
Specific capacity	Karnauskas (1977)	Nepco Lake (T22N R6E)	4	7.3	6.5–9.0
	Batten (1989)	Southeast Wood County	42?	7.4 [1]	5.5–9.9
	Rothschild (1982)	East of PEGPW	266	6.4	<0.3–169 [4]
	All specific capacity		270 [2] 312 [3]	6.4 [2] 6.5 [3]	3.9–16.8 [2] <0.3–169 [4]
Slug	Bradbury et al. (1992)	Buena Vista Basin (T22N R6–9E)	44	2.2	0.03–9.5
	Kraft (1990)	West central Portage County, T23N R7–8E	4	5.3	4.3–7.0
	All slug tests		48	2.4	0.03–9.5

[1] Not known to be a geometric mean

[2] Not including Batten's (1989) result

[3] Including Batten's result, assuming it is the geometric mean of 42 wells. This estimate comes from his reported value of 1.08×10^6 gal d^{-1} pumped from irrigation wells with the reported average flow of 645 gpm, and the assumption that each such well runs 10 days per year.

[4] Five of the highest values, including the maximum of this range, were rejected as "anomalously high"

Table 3.2. Hydraulic conductivities measured in the Wisconsin Central Sand Plain by grain-size analysis and permeameter measurements.

Method	Fine to medium, very well to moderately sorted	Medium, usually with fine gravel, moderately well sorted	Medium to coarse, moderately sorted, sometimes with gravel	Gravelly coarse, poorly sorted; medium, some gravelly, moderately sorted
		Brownell (1986) grain-size analyses		
Masch and Denny (1966)	1.0×10^{-4} m s^{-1} 28 ft/day	1.5×10^{-4} m s^{-1} 43 ft/day	1.1×10^{-4} m s^{-1} 31 ft/day	1.7×10^{-4} m s^{-1} 48 ft/day
Rose and Smith (1957)	7.3×10^{-4} m s^{-1} 210 ft/day	13×10^{-4} m s^{-1} 370 ft/day	58×10^{-4} m s^{-1} 1600 ft/day	16×10^{-4} m s^{-1} 450 ft/day
Hazen (1893)	2.4×10^{-4} m s^{-1} 68 ft/day	3.4×10^{-4} m s^{-1} 96 ft/day	3.9×10^{-4} m s^{-1} 110 ft/day	4.5×10^{-4} m s^{-1} 130 ft/day
		Stoertz (1985) grain-size analyses		
Masch and Denny (1966)	1.6×10^{-4} m s^{-1} 45 ft/day			
		Stoertz (1985)		
Permea-meter	1.5×10^{-4} m s^{-1} 43 ft/day			

calibrated the hydraulic conductivity in a mathematical model of the Buena Vista Basin to measured heads and a water balance. The calibration was optimized at 9×10^{-4} m s^{-1}.

Stoertz (1985) and Brownell (1986) used published formulae to infer K from grain-size analyses in Glacial Lake Wisconsin sands. Stoertz also measured K in a permeameter using repacked samples. These methods, considered less reliable (Shepherd, 1989) than pumping or slug tests, gave the results shown in Table 3.2. The disagreement among the different grain-size analysis methods reduces our confidence in their applicability. The Rose and Smith method provides the best agreement with hydraulic conductivities derived by other methods. The other grain-size methods and the permeameter generally gave smaller hydraulic conductivities than Rose and Smith or *in-situ* methods.

The Ks calculated by the various methods are not entirely comparable. The average for each method has a range from 1.5 to 9×10^{-4} m s^{-1}. This disparity is likely due to two factors. First, as Bradbury et al. (1992) point out, *ex-situ* methods such as permeameter tests and use of grain-size formulae are not as trustworthy as *in-situ* methods. Second, Bradbury and Muldoon (1990) note that Wisconsin Central Sand Plain Ks increase with the scale of measurement, in the following order: grain size analysis, permeameter tests, piezometer slug tests, specific capacity tests, pumping well / observation well tests, and flow model calibration. They explain that these methods involve progressively larger volumes of aquifer material, which presumably increases the chance of encountering sparsely distributed but highly conductive lenses.

Anisotropy

Weeks (1969) and Rothschild (1982) measured the anisotropy of WCSP sediments using a technique based on drawdown in partially penetrating wells:

Location	Ratio K_h/K_v
Hancock	4
Bancroft	2
Plover	7
Mosinee (alluvium)	2
PT-279 (near an end moraine)	20
Rothschild's Field B	5

Larger horizontal/ vertical anisotropies are associated with more pronounced sedimentary stratification. Greater stratification near the moraine probably explains the large anisotropy at the PT-279 site, which Weeks felt was not typical of the Central Sands. The outwash at Bancroft is much siltier than at the other sites, and Weeks stated that the calculated anisotropy there is less reliable than the others. Thus, the ratios at Hancock, Plover, Field B, and perhaps Mosinee should be indicative of those that might be found in the study area. The geometric mean of the four values is 4.1. The relative homogeneity of the project area sediments suggests their anisotropies are likely to be close to the range quoted by Freeze and Cherry (1979) for laboratory measurements of fluvial sediments: 2 to 10. The sediments in the project area are relatively uniform sands, that is, strongly expressed stratification of layers with great textural contrast is seldom found (except within distinct silt beds). Therefore, the anisotropy might be toward the lower end of the range. We selected a value of 2 for flow model construction. Flow calculations are seldom very sensitive to anisotropy.

Specific storage

Specific storage is the volume of water released by a unit volume of aquifer in response to a unit decrease in head (Freeze and Cherry, 1979). Holt's (1965) four pumping tests in the Wisconsin central sand plain indicated specific storages of 0.15, 0.14, 0.15, and 0.20. Weeks and Stangland (1971) measured values of 0.15, 0.20, 0.16, and 0.18. The mean ± standard deviation of these eight values is 0.17 ± 0.02.

Porosity

The porosity of WCSP sand and gravel deposits has been estimated at several locations. Nine samples taken from 0 to 30 m below surface in the Little Plover Basin averaged 0.32 porosity (range = 27.7 to 35.7, standard deviation 0.03; Holt, 1965). Porosities of soils near Hancock ranged from 0.32 to 0.38 (C.B. Tanner, 1957; written communication cited in Holt, 1965). Stoertz (1985) measured porosity of five sediment samples taken 0-6 m below the surface at a site south of Wisconsin Rapids. The mean was 0.38 ± 0.02. Except for the surface sample, Stoertz found no trend of porosity with depth. The mean porosity found in this investigation was 0.40 ± 0.08 (Chapter 2).

MONITORING WELL NETWORK

Installation and description

The monitoring well network (Figure 3.1) installed for this study consisted of three types of wells:

1. 109 wells installed at 25 locations for the purpose of describing basin-scale groundwater conditions. These wells were constructed of 1- or 2-inch PVC well casings with 1- to 5-ft long screens, installed in nests of 1–5 wells.
2. 13 wells at 11 locations around four intensively instrumented fields in the north part of the study area, for the purpose of defining groundwater flow characteristics around those fields (Chapter 6).
3. 30 multilevel piezometers (MLPs), consisting of up to 24 screens over a length of up to 4.1 m (13.4 ft) installed in and around the four intensively monitored fields.

Information on construction and location of wells is given in Appendix 3 and Archive 1.

Wells were installed in three phases. During the first phase (June and July 1991), 30 wells were installed at 25 locations on a grid pattern with the immediate purposes of (1) measuring water-table elevations to allow estimation of the direction of groundwater flow, and (2) obtaining samples and descriptions of subsurface materials. One water-table observation well was installed at each site, and at five of the sites (location codes 0603, 0702, 1803, 2001, and 3003; Figure 3.1) a deeper well was placed in order to monitor the vertical head gradient.

In the second phase, (September, 1991), 13 wells were installed at 11 locations around the edges of the four intensive-study fields in sections 5 and 6. Their primary purpose was to provide a more detailed picture of groundwater depth and flow directions in that region, in order to plan the network of MLPs to be installed within the four fields. Again, a water-table observation well was installed at each site, and a deeper well was also placed at two of the sites (location codes 0572 and 0674, Figure 3.1).

In the third phase (May and June, 1992), 79 well screens were installed at 21 locations downgradient from examples of areally significant land uses. Nine new sites were instrumented during this phase, and new screens were added at 12 phase-1 sites.

All wells constructed in phases 1 and 2 were 2-inch schedule 40 flush-threaded PVC (polyvinyl chloride) casings with 5 ft (1.5 m) long, 0.008-inch (0.2-mm) slotted screens. Phase 3 wells were sometimes 2-inch wells, but generally 1-inch schedule 40 flush-threaded PVC. The slot width was unchanged, and screen lengths were 1 ft (0.3 m), 2 ft (0.6 m) or 5 ft. One-inch wells were installed in bundles of two to five in a single borehole.

Drilling methods are described in Chapter 2. Wells were installed by advancing the hollow-stem auger to the required depth, then filling the stem with tap water to equalize hydrostatic pressure. This prevented

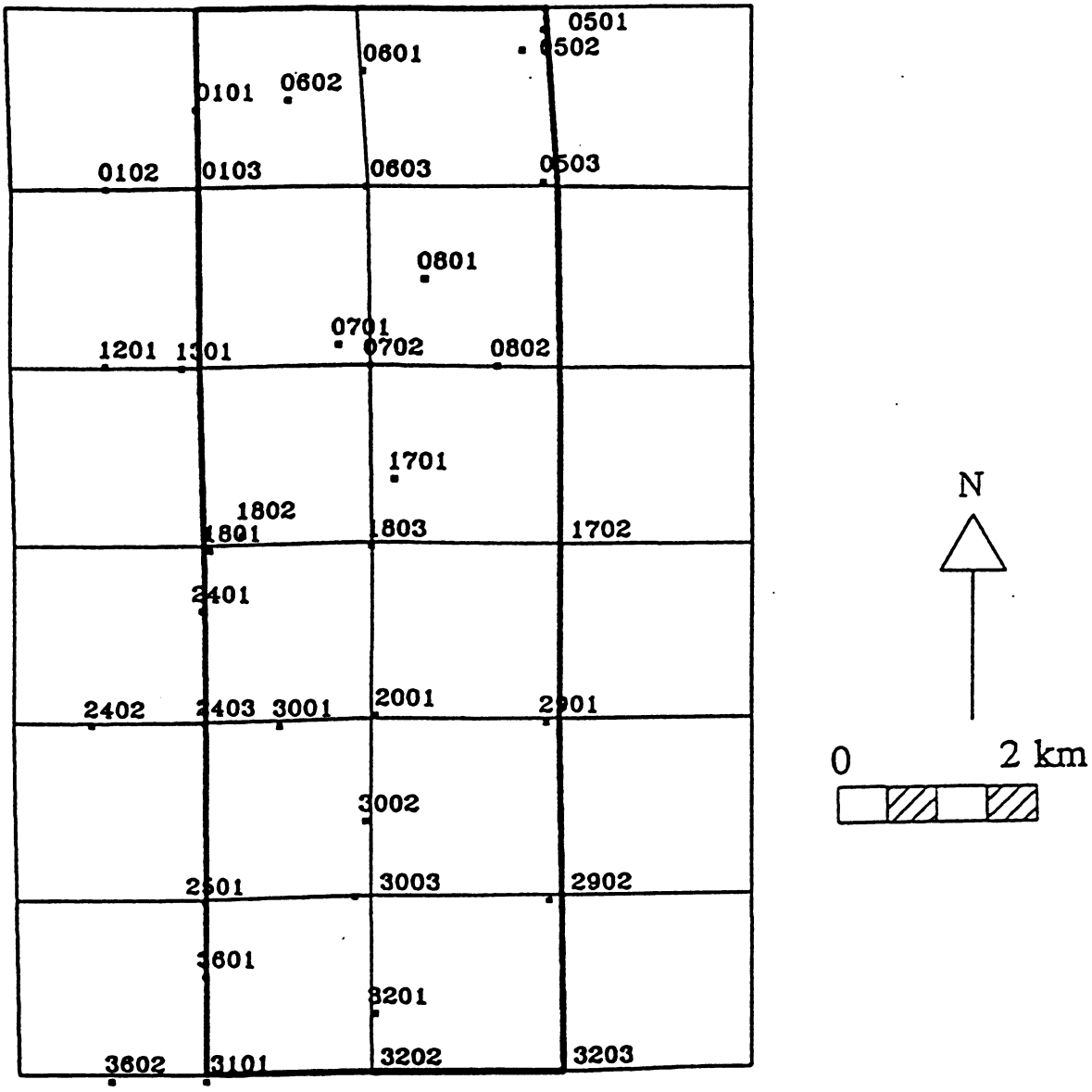


Figure 3.1. Monitoring well network.

sand from surging in when the plug at the auger bottom was removed. The plug was knocked out using a steel rod, and then casing and screen were emplaced through the hollow auger stem. The auger was then withdrawn one 5-ft section at a time, and the height of the well was checked each time an auger section was removed. Once the auger was completely removed, the height of the well was adjusted, and the borehole was backfilled with clean sand cuttings. A steel cover with padlock was installed into the upper part of the hole, and the fill was tamped into place.

In a few cases in phase 1 it proved feasible to install wells in holes drilled with 4-inch solid-stem augers. The borehole was drilled to the required depth, the augers were withdrawn, and the pre-assembled 2-inch well was inserted. Otherwise, the procedure was the same as with the hollow-stem auger.

HYDRAULIC CONDUCTIVITY MEASUREMENTS

Methods

Hydraulic conductivity by slug tests

Slug testing is a method of determining K from a single well. The procedure involves inducing a sudden change in the water level in a well, and then measuring the rate at which the water level recovers to its equilibrium level (Freeze and Cherry, 1979, p. 339). We performed slug tests on 12 wells at 7 sites throughout the study area. Changes in water levels were induced using a slug of 1990 cm^3 . Water level recovery was recorded using a pressure transducer and datalogger. Hydraulic conductivity was calculated by the method of Bouwer and Rice (1976), using the equation

$$K = \frac{r_c^2 \ln(R_e/r_w)}{2L} \frac{\ln(y_o/y_t)}{t}$$

where

r_c = inner screen radius
r_w = radial distance between the well center and the undisturbed aquifer
R_e = effective radius
L = screen length
t = time
y = vertical distance between the water level in the well and the equilibrium water level
y_o = y at time = 0
y_t = y at time = t.

The quantity $\ln(y_o/y_t)$ in the above equation varies linearly with t, so the right fraction,

$$\frac{\ln(y_o/y_t)}{t}$$

is determined by regressing ($\ln y_o/y_t$) on t. The remaining terms are constants dependent on well construction. R_e is not known outright, but rather is evaluated as part of the larger term ($\ln R_e/r_w$), where

$$\ln \frac{R_e}{r_w} = \left[\frac{1.1}{\ln(L_w/r_w)} + \frac{A + B \ln\left[(H - L_w)/r_w\right]}{L/r_w}\right]^{-1}$$

and

H = Water table to aquifer bottom or confining layer thickness

L_w = Water table to screen bottom depth

A and B are empirically derived factors obtained graphically from Bouwer and Rice (1976).

Three trials were performed both before and after each well was pumped for approximately 5 minutes at 0.15 L s^{-1}. Hydraulic conductivities were calculated for each trial at each well. Outliers were eliminated, and remaining values were averaged to produce one estimate per well.

Hydraulic conductivity from specific capacity data

Specific capacity (C_s) is a measure of well productivity defined as $C_s = Q/s$, where Q is the pumping rate of a well and s is the water level drawdown while pumping. Normally, C_s is measured some time after drawdown has apparently stabilized, although drawdown actually continues to increase slowly during pumping. The drawdown term includes both the hydraulic head in the aquifer at the well screen boundary as well as head loss created by turbulent flow of water through the screen and into the pump intake (Freeze and Cherry, 1979, p. 314). Specific capacity data, or at least pumping rate and drawdown data, are commonly available from well construction reports.

K can be estimated from specific capacity data. Bradbury and Rothschild (1985) discuss the methodology in detail, and present a FORTRAN computer code which facilitates computations. We re-coded that program into dBase since production well construction information for the Port Edwards project was already being managed with dBase. The required inputs into the program are:

1. pumping rate

2. length of pumping test
3. static water depth
4. water depth when pumping
5. well screen length
6. well radius
7. aquifer storage coefficient
8. saturated aquifer thickness
9. well loss constant.

Items 1-6 were obtained from well construction reports. An estimate of 0.2 was used for the aquifer storage coefficient (item 7), consistent with that used by Bradbury and Rothschild and commonly used for Wisconsin Central Sand Plain sandy sediments. Since the calculation is relatively insensitive to the storage coefficient, an exact value is not crucial (Bradbury and Rothschild, 1985). Saturated aquifer thickness (item 8) was estimated by either (1) subtracting static water depth from total water depth, as reported in well construction reports, or (2) from a map of saturated thickness as presented in Chapter 2. The greater of the two values was used as saturated thickness. Well losses were set to zero, again consistent with Bradbury and Rothschild.

Results

Estimates from slug tests

Inputs to the Bouwer and Rice equation (1976) and slug test results are listed in Table 3.3. Ks from slug tests had a geometric mean of 4.0×10^{-4} m s^{-1} a range of 1.2 to 18.3×10^{-4} m s^{-1}, and a ± one log standard deviation of 1.9 to 8.6×10^{-4} m s^{-1}. There was no apparent difference between Ks measured before and after wells were pumped.

Estimates from specific capacity tests

A total of 74 construction reports were located for production wells within the project area. Of these, 29 were unusable due to missing data, well completion in an underlying formation, or construction that precluded use of the method (e.g., wells constructed without screens). The K of the remaining 45 wells averaged 5.2×10^{-4} m s^{-1} (Table 3.4) and displayed considerable scatter. (All statistics assume that K is log-normally distributed; geometric means are reported). We observed no apparent relationships between specific capacity derived K and location or screen position in the aquifer. However, K is apparently higher in wells having

higher pumping rates during the specific capacity test (Table 3.5, Figure 3.2). This phenomenon may be due to the effect of larger aquifer volumes being sampled by higher pumping rates (Bradbury et al. 1992, discussed previously). We tested the hypotheses that K increases with increasing screen length or with total volume of water pumped. However, plots of K vs. screen length or total volume pumped display considerably more scatter than the comparison to pumping rates. We conclude that any apparent relationship of K to screen length or volume pumped is only due to a cross-correlation with pumping rate.

Table 3.3. Slug test inputs and results. For all wells, R_c = 0.084 ft, r_w = 0.04 ft, L_e = 4.75 ft, A = 2.8, B = 1.

Location	Well no.	ln $(y_0/y_t)/t$	H (m)	ln(R_e/r_w)	L_w (m)	K (10^{-4} m/s)
0603	EU762	0.5	7.4	2.5	2.7	2.6
	EU601	0.9	7.4	2.9	5.6	5.8
0103	EU654	0.8	8.2	2.5	2.9	4.4
	EU655	0.3	18.9	3.0	14.9	1.8
	EU605	0.1	8.2	3.6	8.1	1.2
0702	EU659	1.3	4.5	2.9	3.8	8.8
	EU660	0.5	14.8	2.9	10.7	3.1
1301	EU658	0.5	8.8	2.4	2.4	2.5
	EU608	2.8	8.8	2.9	6.4	18.3
1803	EU766	0.6	18.0	2.7	8.2	4.0
2001	EU770	1.2	20.2	2.2	1.7	6.0

Table 3.4. Statistics for hydraulic conductivity estimated from specific capacity.

Criterion	Value
Sample size	45
Geometric mean	5.2×10^{-4} m s^{-1}
Interval ± one log standard deviation	$0.3 \times 10^{-4} - 10 \times 10^{-4}$ m s^{-1}
Range	$0.6 \times 10^{-4} - 17 \times 10^{-4}$ m s^{-1}

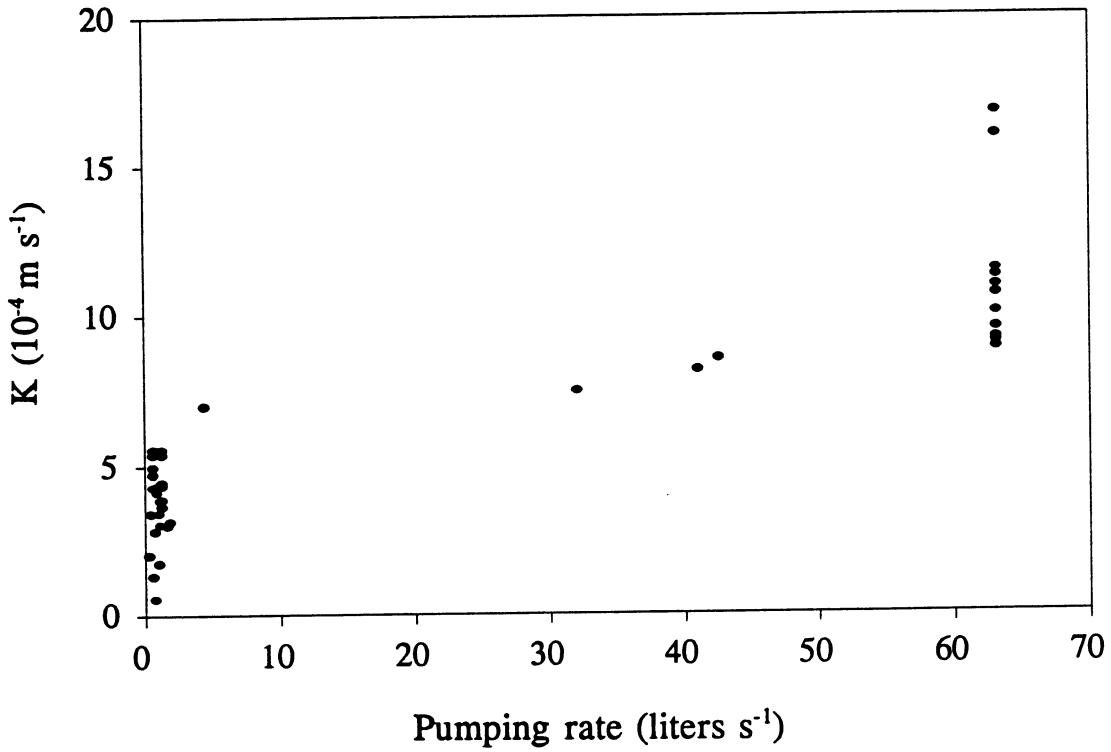


Figure 3.2. K, as determined from specific capacity, variation with pumping rate.

Table 3.5. Statistics for hydraulic conductivity estimated from wells pumped at different rates.

Pumping rate	< 500 gpm	> 500 gpm	≥ 1000 gpm
Sample size	30	15	12
Geometric mean (10^{-4} m s^{-1})	4.3	7.7	8.6
± one log standard deviation	2.3 – 8	4.8 – 12	5.5 – 13
Range	0.58 – 11	3.8 – 17	3.8 – 17

Discussion

The geometric mean of slug test-derived Ks in the PEGPW (4.0 x 10^{-4} m s^{-1}) was greater than that measured in the nearby Buena Vista Basin (2.2 x 10^{-4} m s^{-1}; Bradbury et al., 1992), but about the same as found near Plover (5.3 x 10^{-4} m s^{-1}; Kraft,1990). We observed an anomalously high value of 18.3 x 10^{-4} m s^{-1} at well EU608, which probably reflects a rare, high-conductance lens. Though this value is almost double the largest value reported by Bradbury et al. (1992), other atypically high slug test-derived Ks have been measured in the WCSP. For instance, Brown et al. (1992) measured values up to 40 x 10^{-4} m s^{-1} near the Stevens Point wellfield.

The geometric mean of specific capacity-derived Ks (5.2 x 10^{-4} m s^{-1}) was greater than that of slug tests (4.0 x 10^{-4} m s^{-1}; Tables 3.3 and 3.4). This is likely due to a scaling effect, whereby larger hydraulic conductivities are measured in tests that include larger aquifer volumes, due to the inclusion of sparse but highly conductive lenses (Bradbury and Muldoon, 1990).

Specific capacity-derived Ks are 20% lower or 30% higher than the average of other reported WCSP values (6.5 x 10^{-4} m s^{-1}, Table 3.1), depending if the PEGPW overall average or high pumping rate average is used for comparison. We observed that specific capacity-derived K was higher in wells with higher pumping rates, averaging 8.6 x 10^{-4} m s^{-1} in wells pumped at ≥ 3800 L min^{-1} (1000 gallons min^{-1}; Table 3.5). We hypothesize that this is due to the scaling effect discussed above. Since previous studies used data exclusively from high capacity wells, PEGPW values from wells pumping ≥ 3800 L min^{-1} are probably the best to use for comparison purposes.

The K measured in wells pumping at ≥ 3800 L min^{-1} (8.6 x 10^{-4} m s^{-1}) reflects the largest available scale of measurement, and hence should be most characteristic of the basin scale. This value is close to the 9.0 x 10^{-4} m s^{-1} found in the model calibration of the Buena Vista Basin (Bradbury et al., 1992).

HYDROLOGY OF THE PORT EDWARDS GROUNDWATER PRIORITY WATERSHED

Hydraulic head measurements and groundwater flow patterns

Methods

Hydraulic heads were determined from monthly depth-to-water measurements by subtracting the depth to water in each well from the elevation of the well casing. Depth to water was measured with a popper attached to a measuring tape or with an electronic water-level indicator.

Water table maps were created with the contouring software SURFER for selected months with the most complete sets of data. The maps were constructed using kriging as the interpolation method. Plots of hydraulic head versus time were constructed to explore temporal variations in water levels. These plots were made for selected wells along a north-south transect of the study area. We also explored the occurrence, magnitude, and temporal variation in vertical hydraulic gradients. Vertical hydraulic gradients were calculated at each well nest by dividing the head difference in adjacent wells by the distance between screen midpoints.

Results and discussion

Monthly hydraulic head measurements are available in Archive 2. Contouring of these data revealed that groundwater in the study area flows southeast and east toward the Wisconsin River and its backwaters (Figure 3.3). Flow was southeastward in the western part of the watershed, and became more eastward as it approached the river. Horizontal gradients were approximately 0.002 in the west and 0.005 in the east.

The water table fluctuated by about 1.2 m during the study (Figure 3.4). Water levels declined about 0.6 m from September 1992 to March 1993, reflecting the cessation of groundwater recharge during winter months. From March to June of 1993, the water table rose sharply about 1.2 m, coincident with extreme spring precipitation. The water table receded from this peak by March 1994. Flow patterns and horizontal gradients were generally consistent throughout study period, even with the fluctuating water table (Figure 3.5).

Vertical gradients were calculated from monthly head measurements. Detectable vertical gradients were observed at well locations 0502, 0503, 0602, 0603, 0801, 1803, and 3203 (Figure 3.6). Generally, the gradients were small, less than 0.04, and were related to the occurence of a lacustrine layer between screens. However, at location 0801, a gradient of 0.1 was observed across the New Rome and in the sand unit below the New Rome. An anomalously large gradient of –0.4 was calculated between two deep wells screened in the same sand unit at location 3203, however, we are unable to verify the authenticity of this

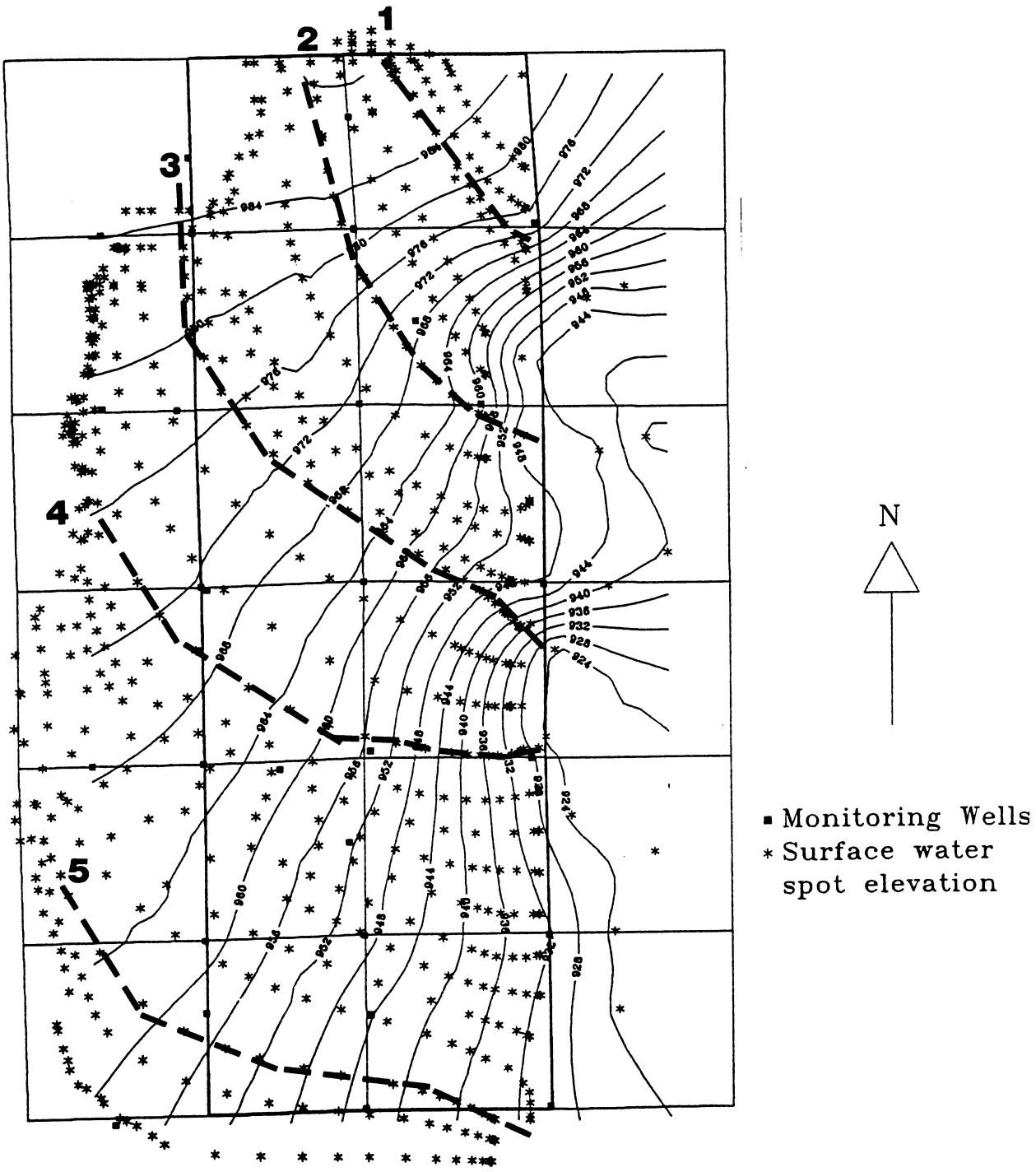


Figure 3.3 Groundwater flow in Port Edwards Groundwater Priority Watershed.

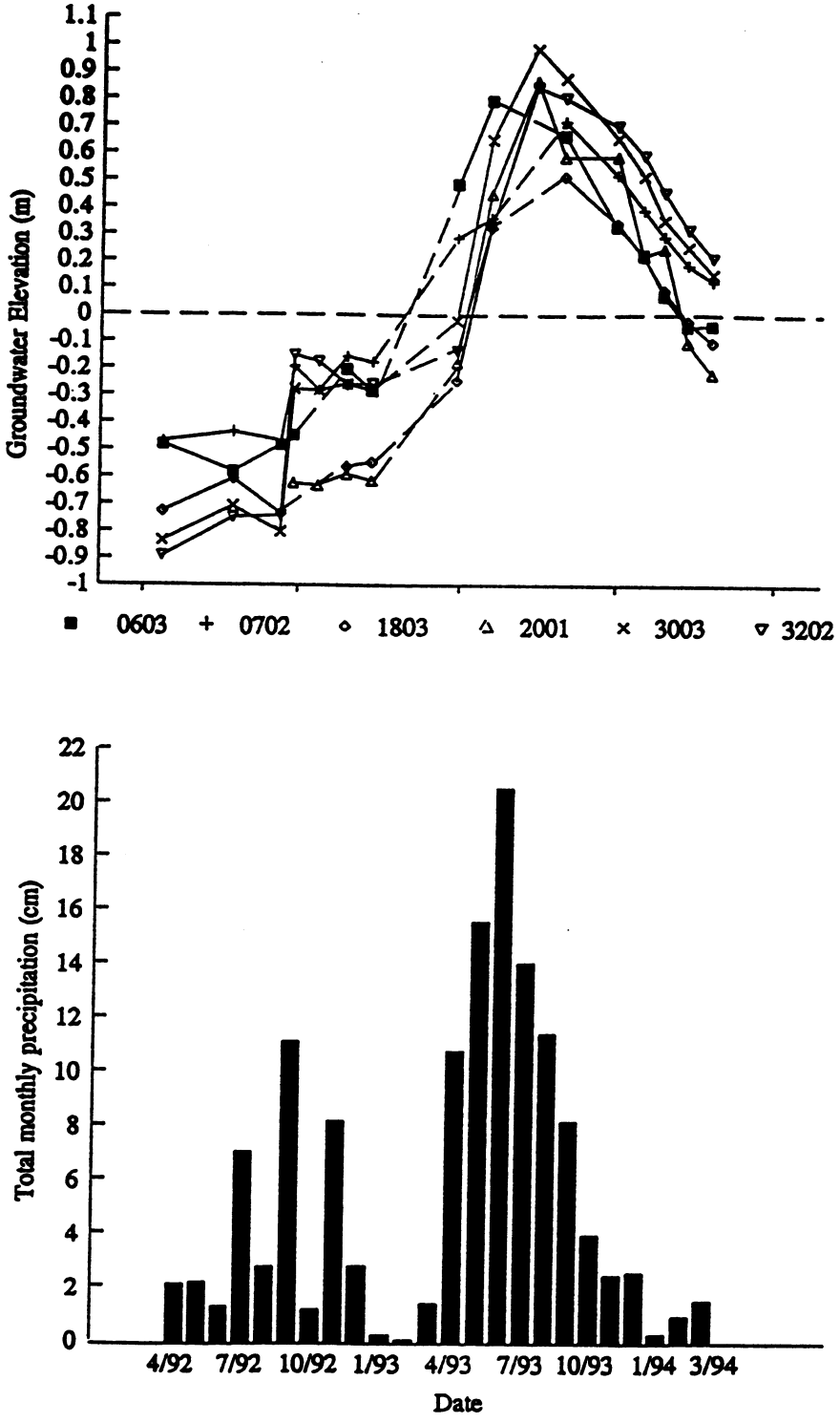


Figure 3.4. Water-table fluctuations and monthly precipitation in the Port Edwards Groundwater Priority Watershed, September 1992 through March 1994.

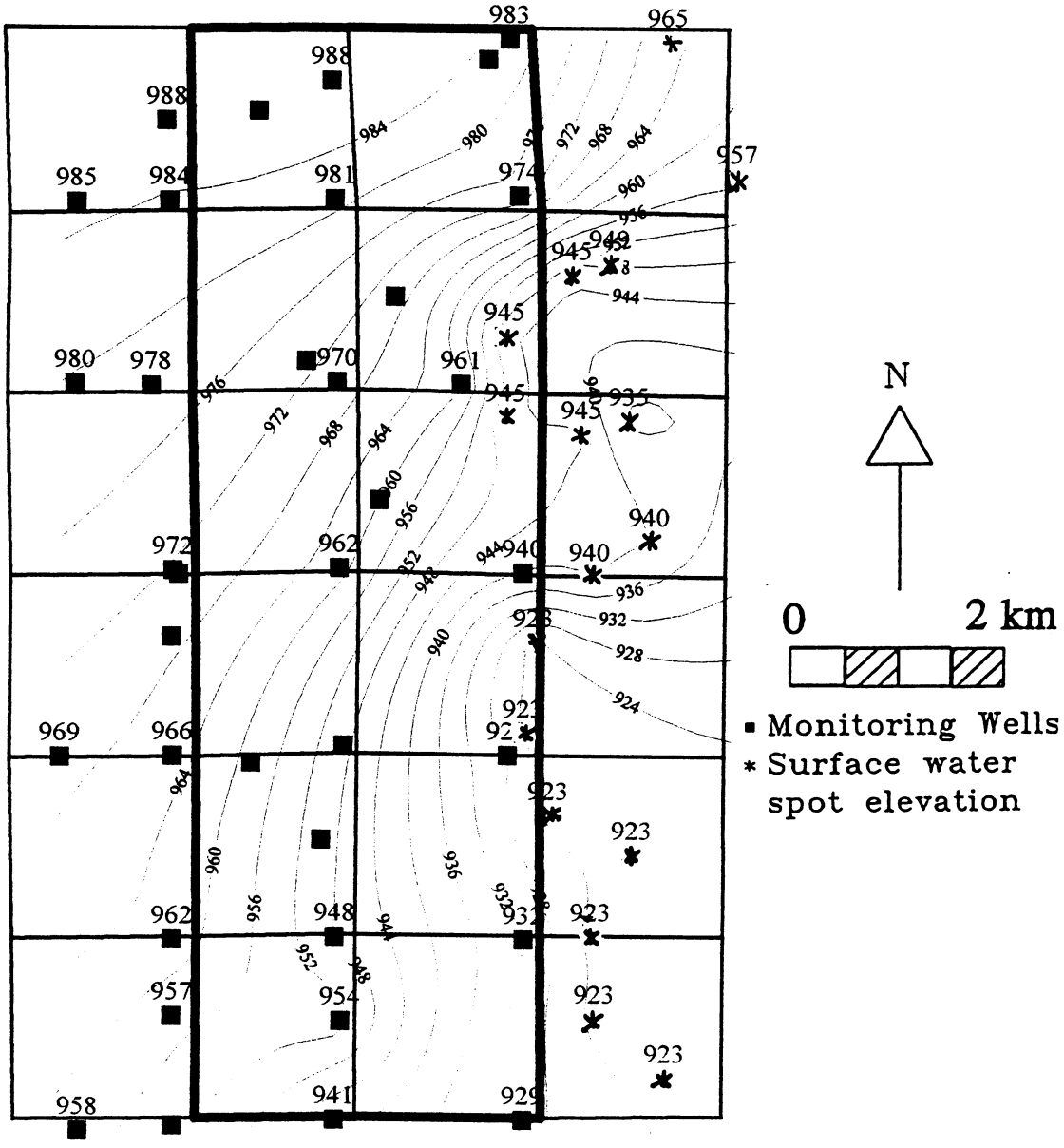


Figure 3.5A. Water table configuration, September 1992.

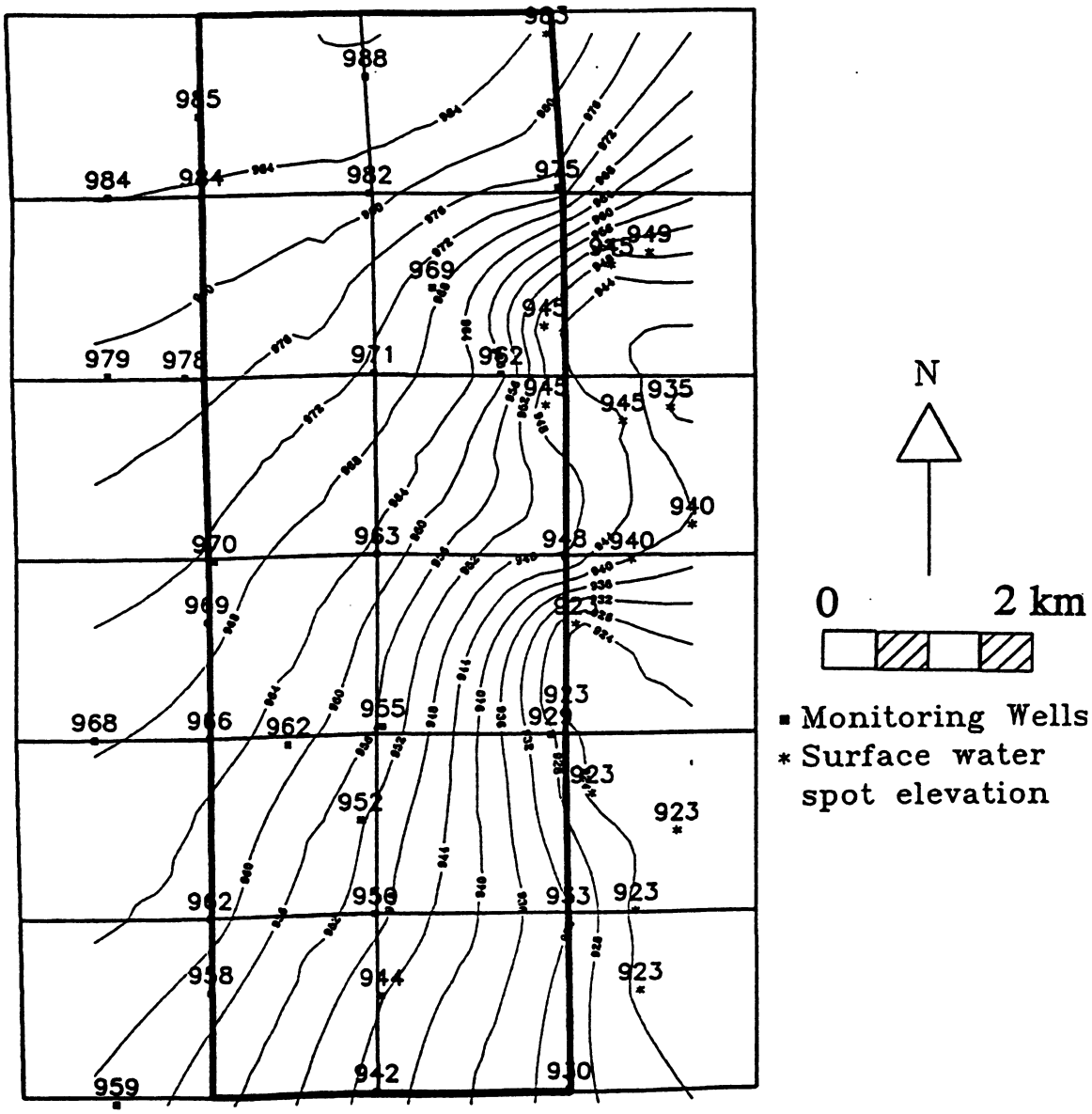


Figure 3.5B. Water table configuration, January 1993.

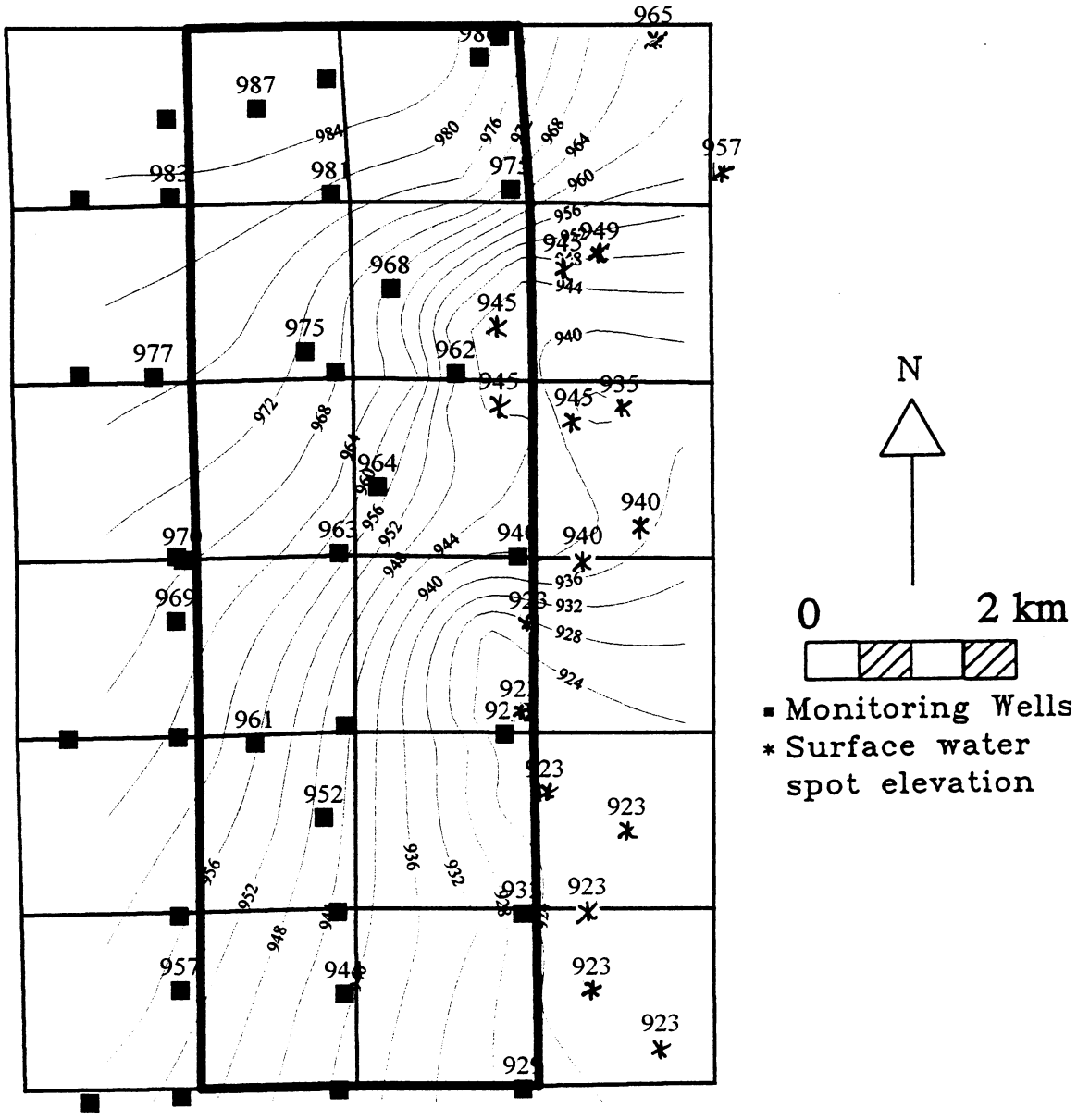


Figure 3.5C. Water table configuration, March 1993.

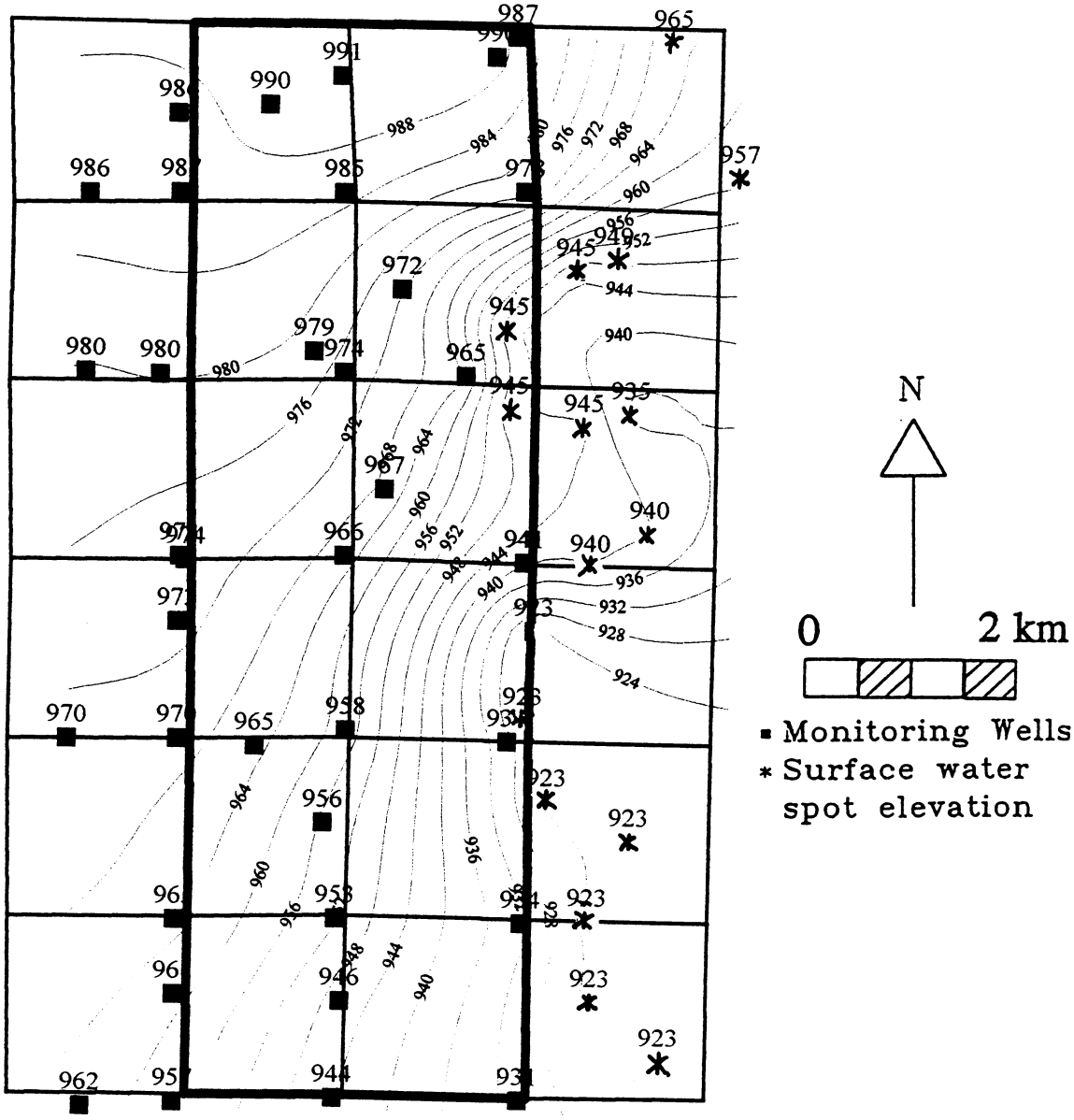


Figure 3.5D. Water table configuration, June 1993.

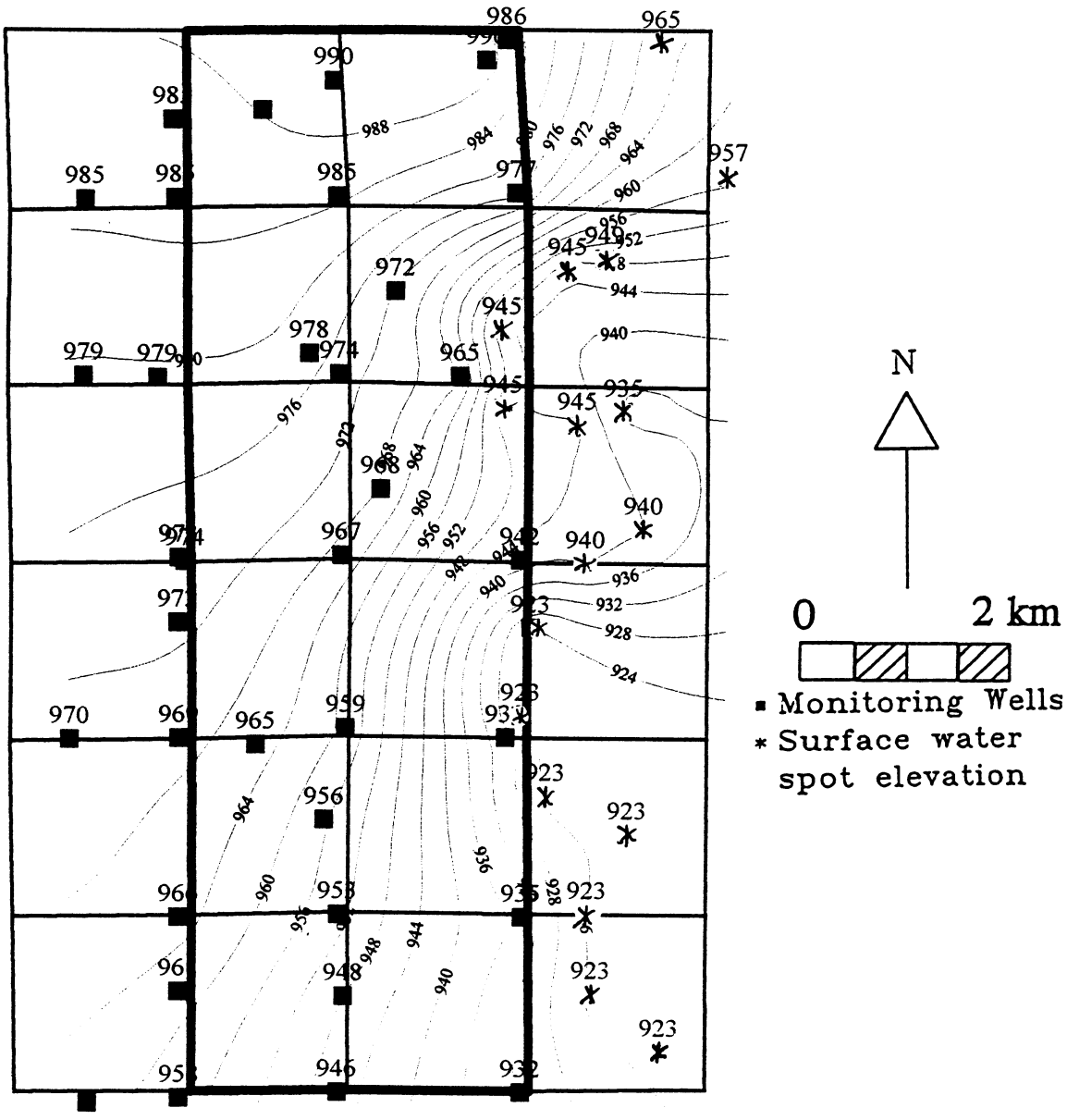


Figure 3.5E. Water table configuration, September 1993.

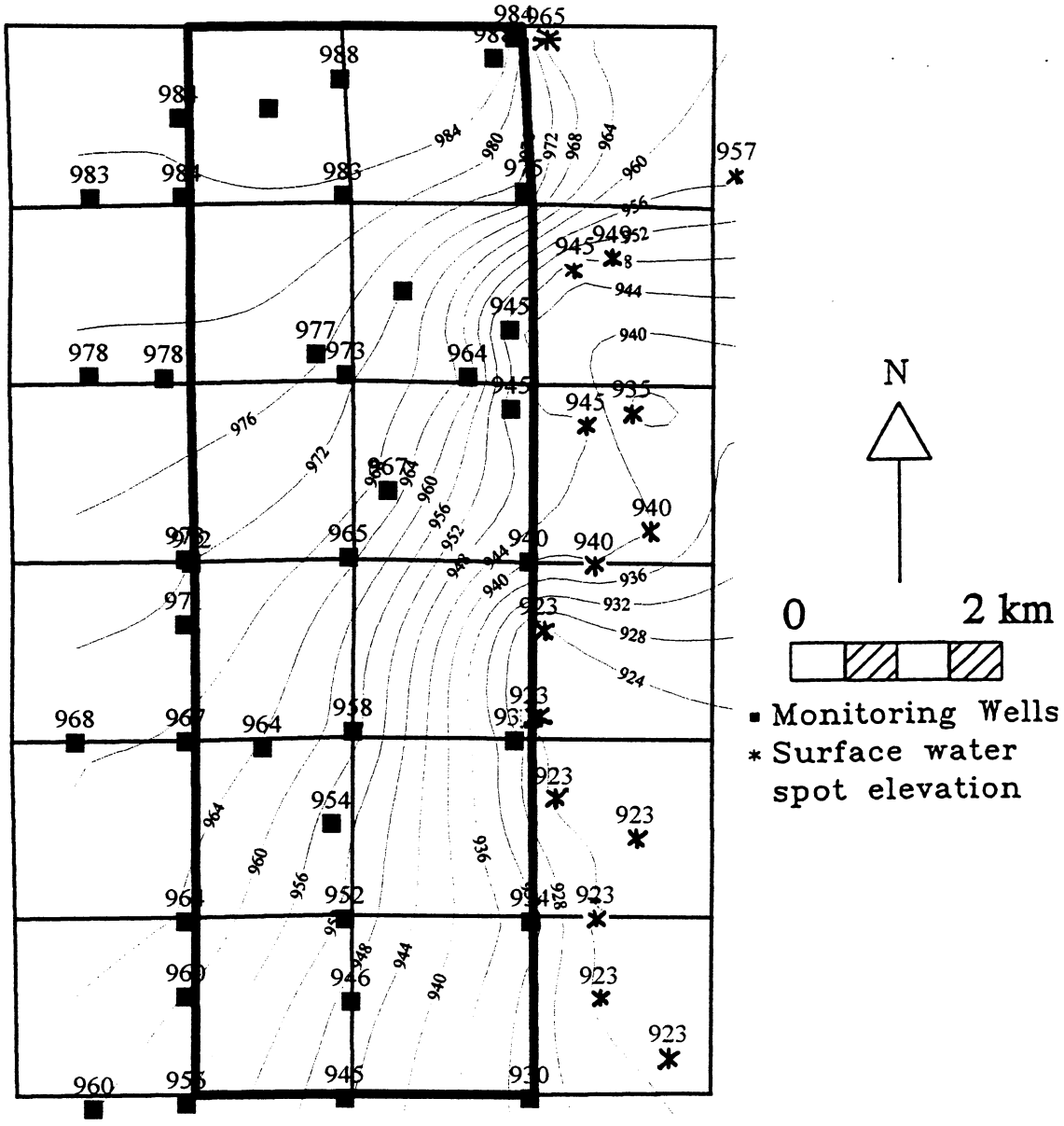


Figure 3.5F. Water table configuration, January 1994.

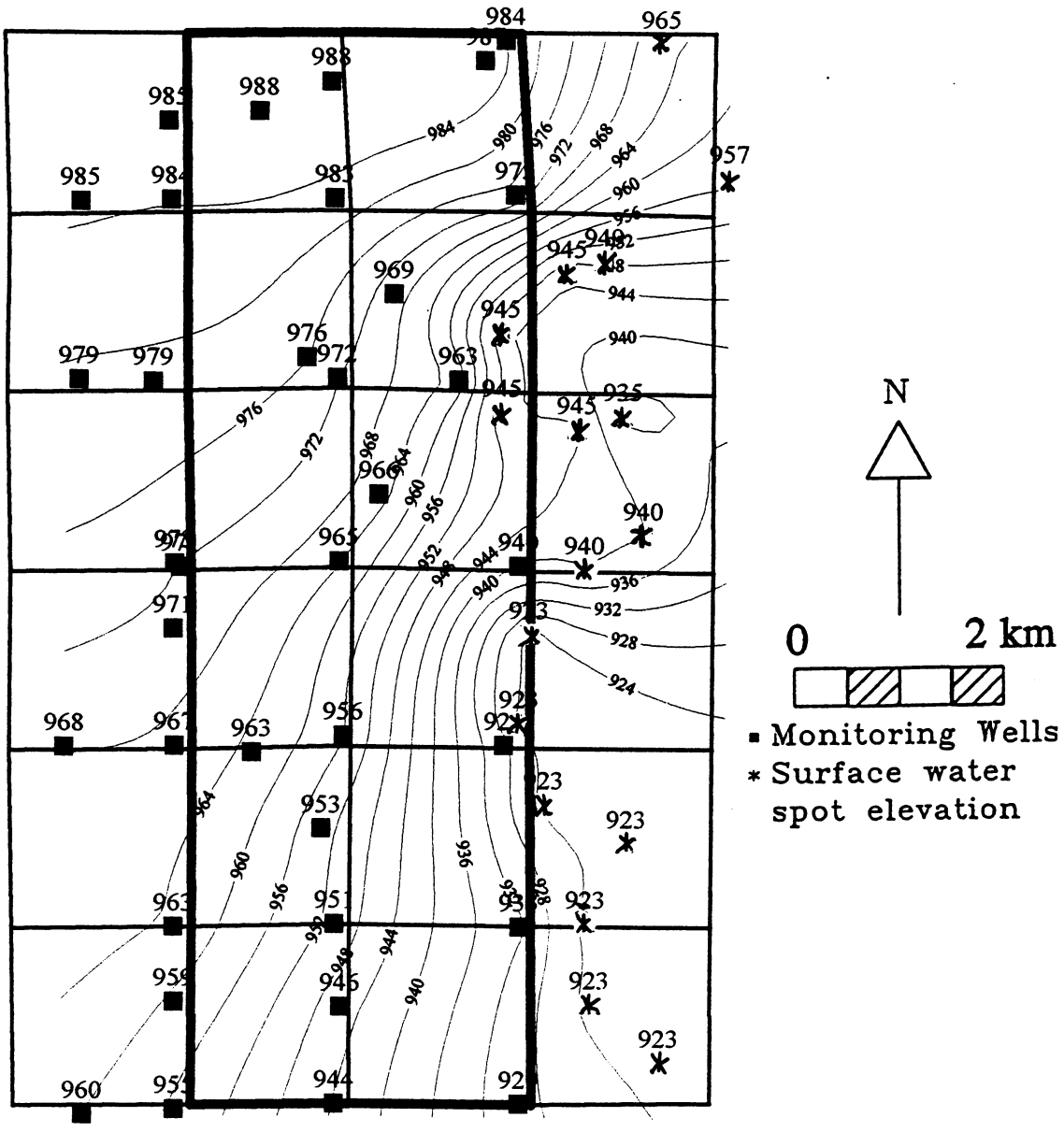


Figure 3.5G. Water table configuration, March 1994.

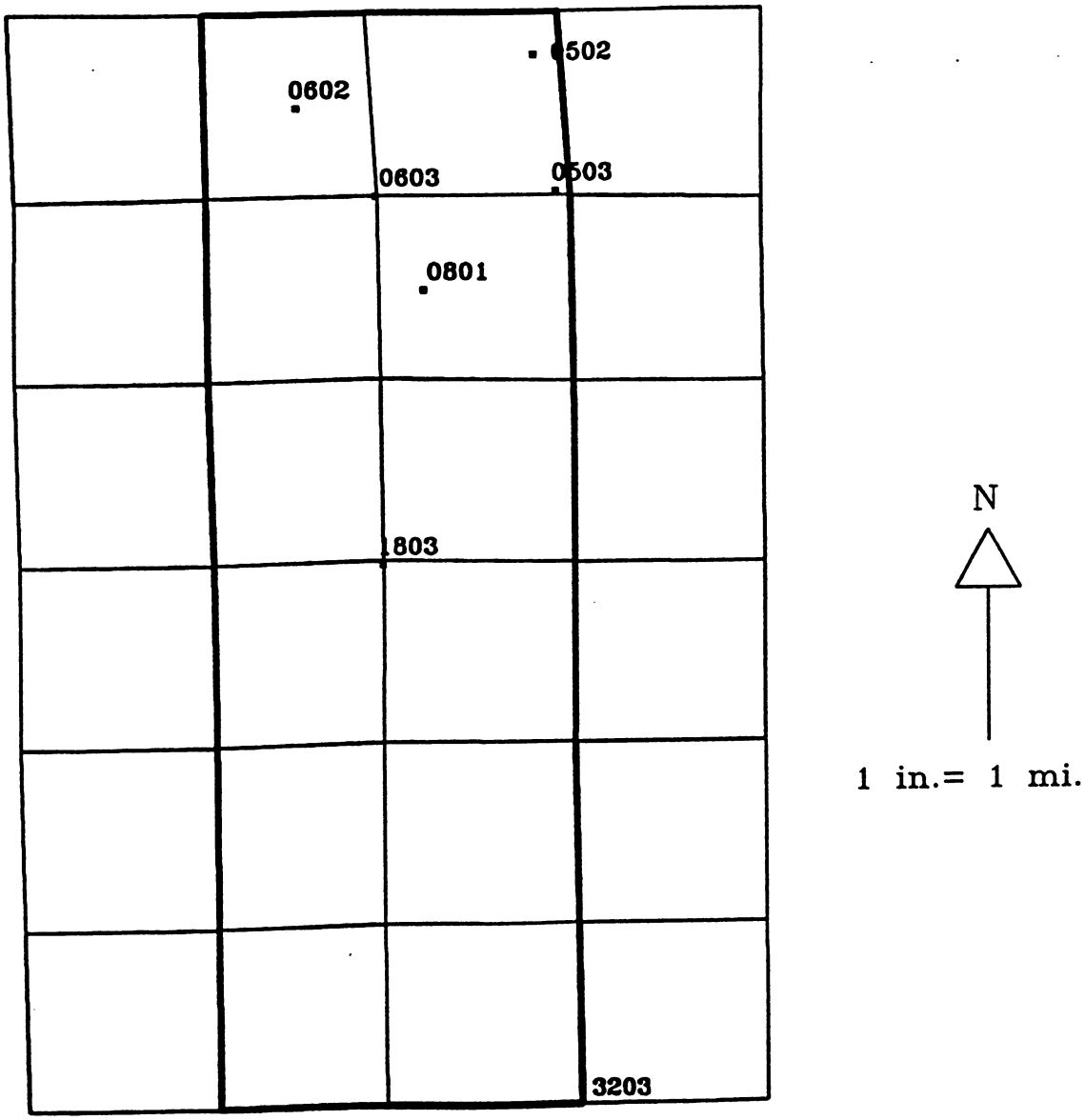


Figure 3.6. Map of where gradients were observed.

value. Frost heaving or other disturbances which might affect casing elevation could have produced an erroneous result.

Water budget

The water budget for PEGPW groundwater can be partitioned into recharge, underflow in, and underflow out. Recharge is equal to precipitation minus surface runoff and evapotranspiration. Annual precipitation in the WCSP averages 80 cm (Batten, 1989; Weeks et al., 1965). Surface runoff is minor, amounting to only about 2.5 cm, and evapotranspiration losses are dependent on land use. Under forest, grassland, and non-irrigated cropland, evapotranspiration is about 53 cm, leaving about 25 cm of precipitation as groundwater recharge (Weeks, 1965). Evapotranspiration is 10 cm higher under irrigated cropland, which decreases groundwater recharge to 15 cm. In summary, groundwater recharge is 15 cm yr^{-1} under irrigated cropland and 25 cm yr^{-1} in other areas. Total annual recharge was obtained by multiplying the area of irrigated or nonirrigated land uses by their respective recharge rates and then summing the result.

Underflow entering and underflow leaving the study area was derived by using the calibrated one-layer groundwater flow model discussed in Chapter 4. For purposes of obtaining underflow contributions to the water budget, constant head boundaries were placed along the boundaries of the PEGPW. The computer code MODFLOW provides a flow summary for all constant head cell, so underflow into the system was calculated by summing the flow in cells along the western and northern boundaries, while the underflow out of the system was calculated by summing flows along the eastern and southern boundaries.

The resulting steady-state water budget (Figure 3.7) consists of 1.9×10^4 m^3 d^{-1} of recharge, 2.2×10^4 m^3 d^{-1} of underflow entering from the west and north, and 4.2×10^4 m^3 d^{-1} of underflow leaving through the east and south.

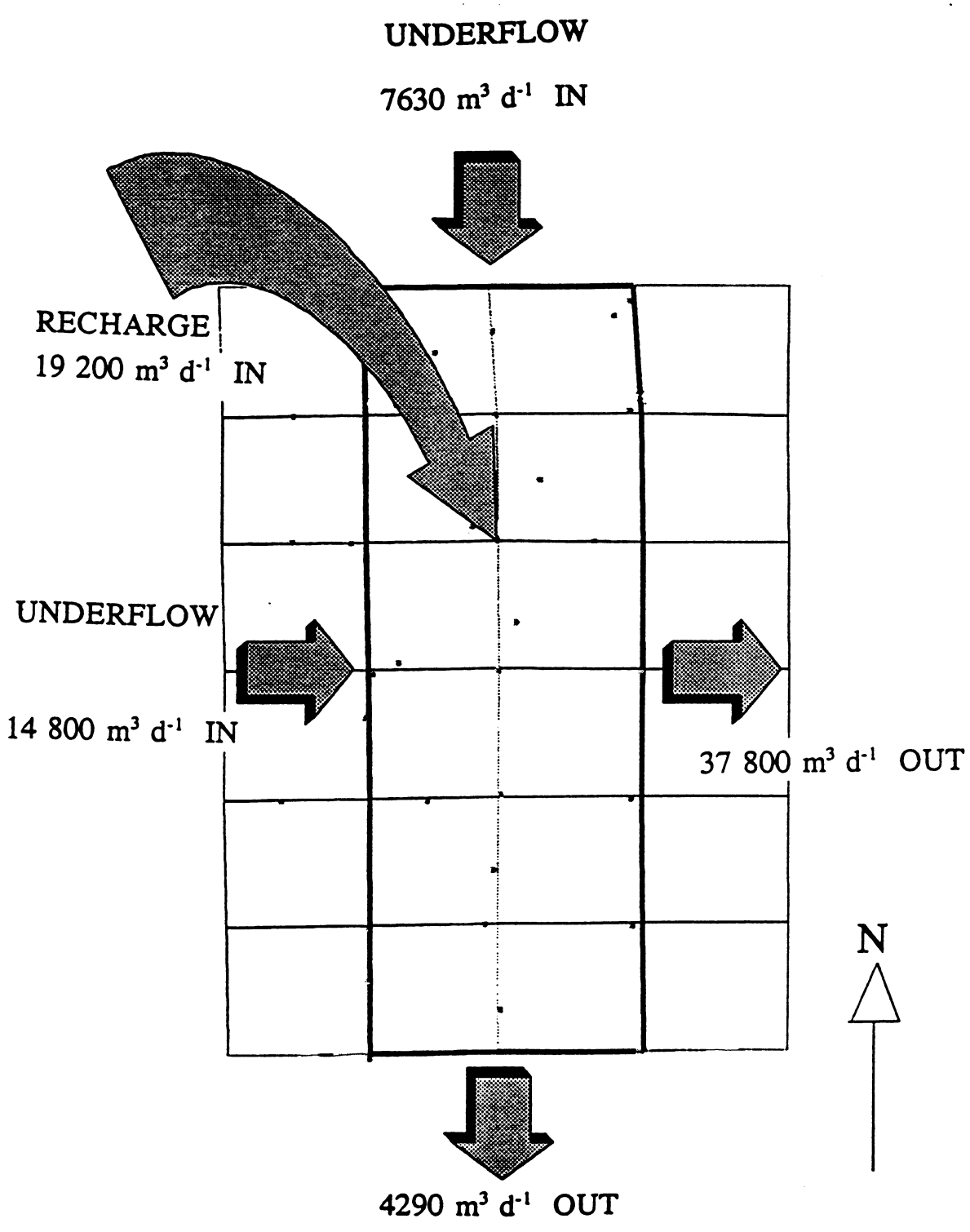


Figure 3.7. Water budget, Port Edwards Groundwater Priority Watershed.

CHAPTER 4

GROUNDWATER FLOW AND PARTICLE TRACK MODELING

Groundwater flow models were constructed to help describe and understand the hydrogeology of the Port Edwards Groundwater Priority Watershed. Particle tracking models were constructed to plot flow lines, simulate advective contaminant transport, and assist in monitoring data interpretation.

GROUNDWATER FLOW MODELING

Flow models were constructed using MODFLOW (McDonald and Harbaugh, 1988), a modular three-dimensional finite difference computer code. Both one-layer and three-layer steady-state models were constructed.

Model inputs

Discretization

The finite-difference grid for both the one- and three-layer models consists of 38 north-south columns and 33 east-west rows, and extends approximately 2.5 km beyond the boundaries of the study area (Figure 4.1A-C). The cell length along the row direction is variable. Cell width is 402 m in the west, decreasing to 201 m and then finally to 100 m. Cell length along the column direction is constant at 402 m. The one-layer model treats the aquifer as vertically homogeneous within each cell, while the three-layer model incorporates a layer each for the upper sand unit, the New Rome member, and the lower sand unit. Vertical dimensions of the layers were based on the geology maps presented in Chapter 2.

Hydraulic conductivity

A hydraulic conductivity of 9.0×10^{-4} m s^{-1} (240 ft day^{-1}) was adopted for the sand units based on field measurements (Chapter 3) and a model calibration in the nearby Buena Vista groundwater basin (Bradbury et al., 1992). The anisotropy ratio (K_h/K_v) was estimated at 2. For the finer-textured New Rome Member, the horizontal hydraulic conductivity was initially approximated as 10^{-6} m s^{-1} , based on generic values for material of this texture (Freeze and Cherry, 1979). An anisotropy ratio of 10 was assumed. In the one-layer model, the hydraulic conductivity of each cell was estimated as the thickness-weighted average of the three units in the cell.

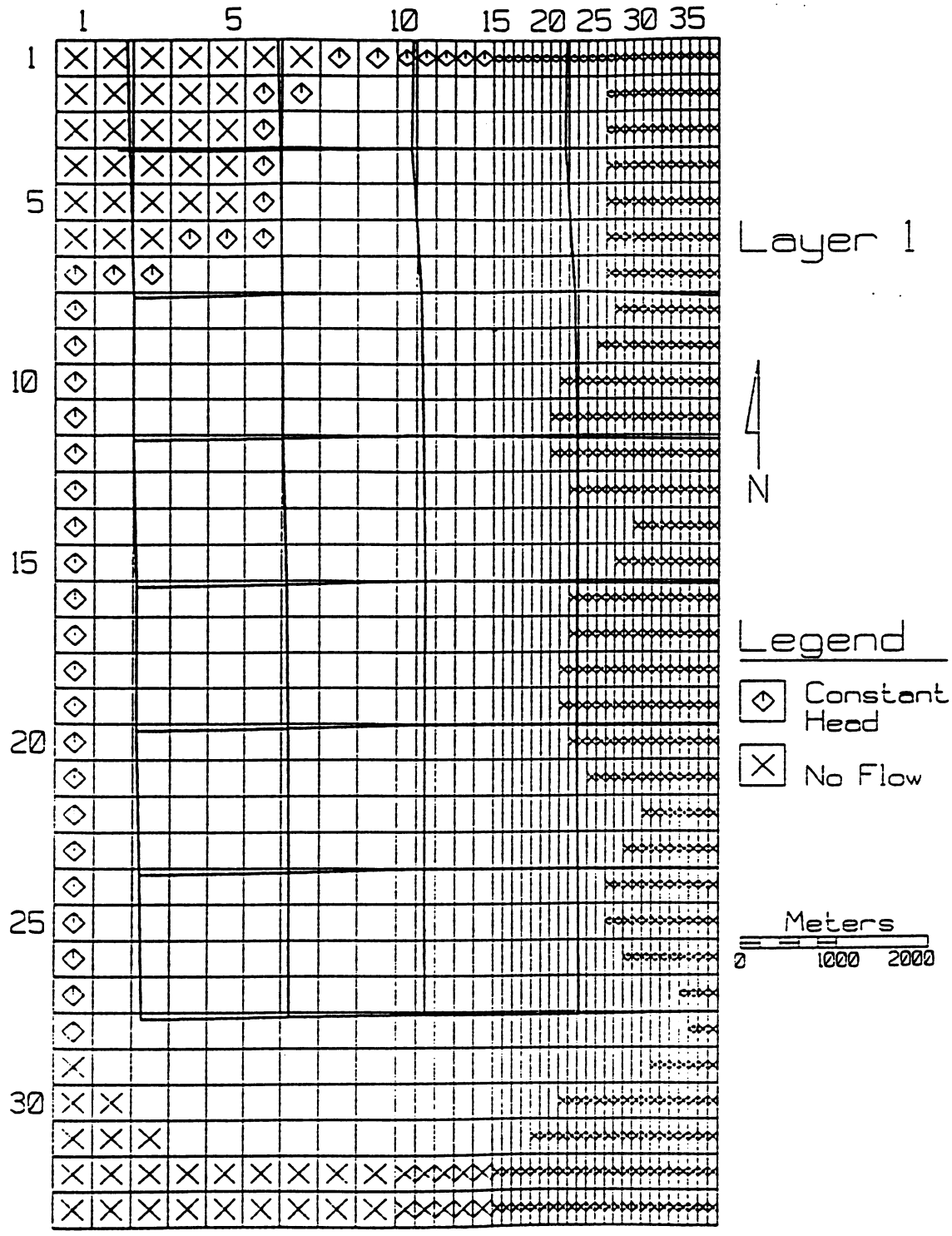


Figure 4.1A Finite-difference grid for layer 1 of the three-layer model.

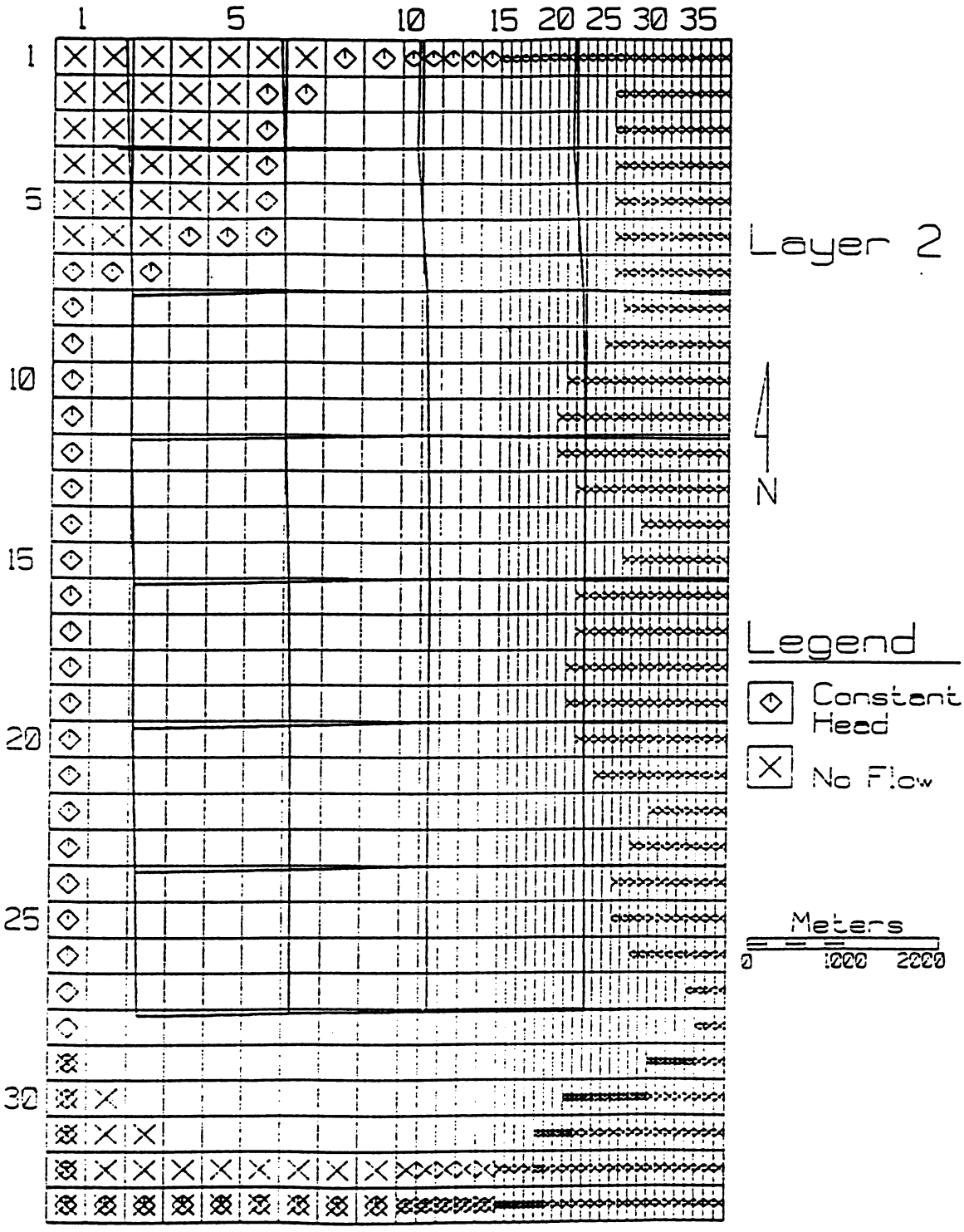


Figure 4.1B Finite-difference grid for layer 2 of the three-layer model.

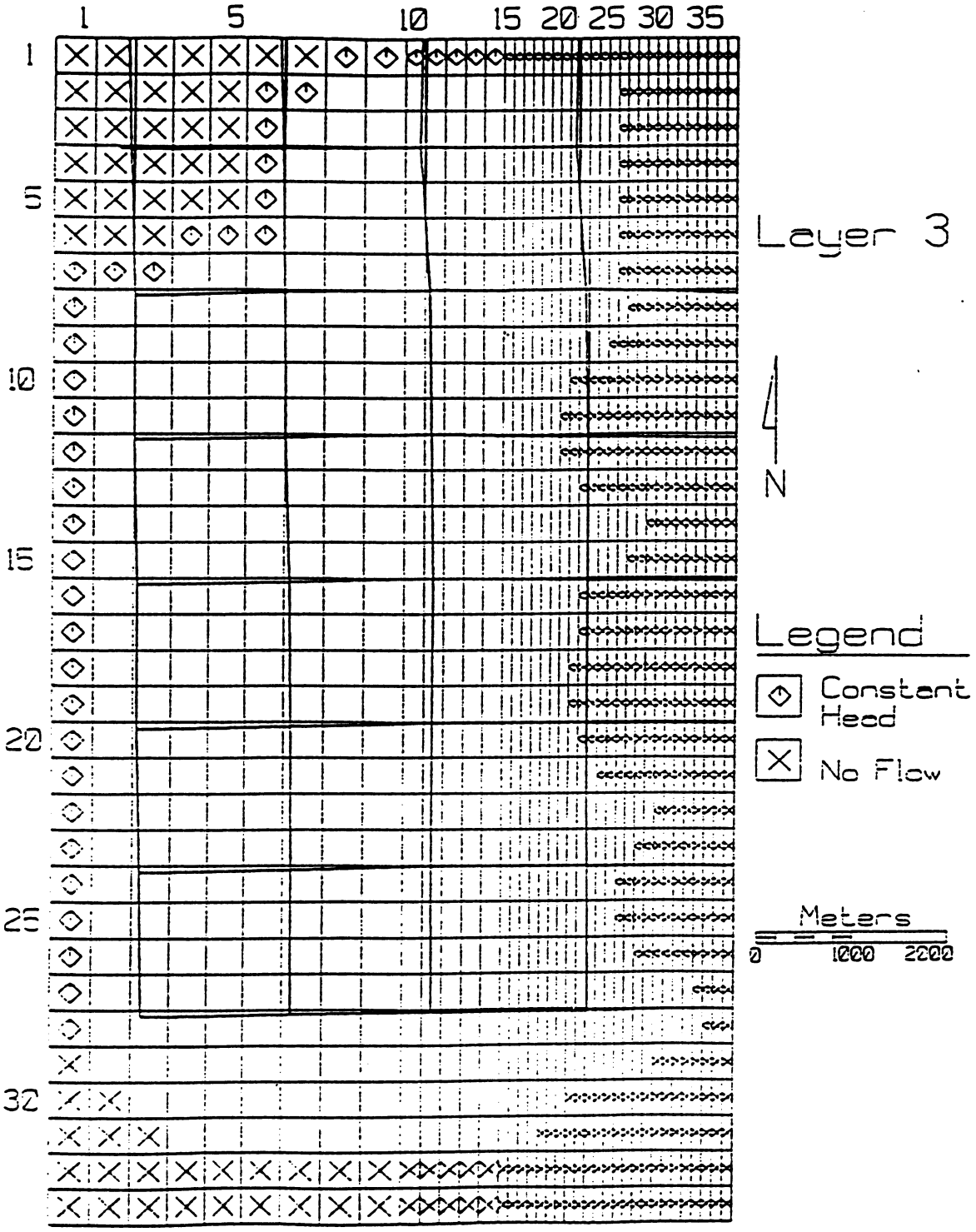


Figure 4.1C Finite-difference grid for the one-layer model and layer 3 of the three-layer model.

Boundary Conditions

Boundary conditions were constant head along the north, west, and east, and constant flux (no-flow) in the south (Figure 4.1). Constant head values were assigned from measured or extrapolated head measurements, or from elevations of natural water bodies.

Other Inputs

Geologic inputs, such as the elevations of the bottom and top of each geologic unit, were constructed as numerical arrays based on the contour maps presented in Chapter 2. A recharge rate of 0.23 m yr^{-1} was used. This is the area-weighted average of recharge on irrigated lands (0.15 m yr^{-1}) and all other lands (0.25 m yr^{-1}), based on Weeks (1965).

In the three-layer models, arrays (VCONT arrays) were needed to characterize the vertical conductance between layers. The values for each cell in the arrays were calculated using the formula

$$VCONT = \frac{1}{\frac{(lay1_{top} - lay1_{bot})/2}{K_{lay1}} + \frac{(lay2_{top} - lay2_{bot})/2}{K_{lay2}}}$$

where

$lay1_{top}$ = Elevation of top of the upper layer

$lay1_{bot}$ = Elevation of bottom of the upper layer

K_{lay1} = Vertical hydraulic conductivity of upper layer

$lay2_{top}$ = Elevation of top of the lower layer

$lay2_{bot}$ = Elevation of bottom of the lower layer

K_{lay2} = Vertical hydraulic conductivity of lower layer

Calibration of one-layer model

The one-layer model was calibrated to heads measured in January 1993, as these data appeared to best represent steady-state conditions. The calibration strategy was to first adjust model inputs in which we had the least confidence. If these adjustments had to be unrealistic in order to achieve a successful match, we then adjusted the next input in which we had least confidence. We sought to avoid making adjustments in patchwork fashion when a more global change could be performed.

When the one-layer model was first run, the overall match was fair except for problematic areas in

output heads were too high. Since bedrock elevation is the data in which we had least confidence, bedrock elevations were reduced in the south-central and southeastern portions of the study area to decrease modeled heads, and increased in the northeast to increase heads there (Figure 4.2). A change in bedrock by 6.1 m (20 ft) either way resulted in a approximately 0.3 m (1 ft) change in hydraulic head. To fine-tune the match, nearby boundary heads in the northeast were also adjusted. In the area of well locations 0503 and 0801, alterations of bedrock and boundary heads alone still did not yield a satisfactory head match. We then elected to increase hydraulic conductivity in this vicinity to achieve a match to measured heads (Figure 4.3).

Calibration and sensitivity of three-layer model

The three-layer model used boundary conditions and aquifer bottom elevations from the calibrated one-layer model. The increased hydraulic conductivity in the vicinity of well locations 0503 and 0801 was carried through to the three-layer models by increasing the hydraulic conductivity of the sand units accordingly. A K_v of 10^{-7} m s^{-1} (0.0283 ft d^{-1}) was initially assumed for the New Rome.

Model parameters were reasonably constrained with the exception of New Rome vertical hydraulic conductivity. Therefore, we performed a sensitivity analysis to assess uncertainty in the New Rome vertical hydraulic conductivity. The sensitivity analysis involved altering the vertical hydraulic conductivity for the New Rome Member to 10^{-9}, 10^{-7}, 10^{-6}, and 10^{-5} m s^{-1} (0.000283, 0.0283, 0.283, and 2.83 ft d^{-1}). These were incorporated into the model changing the VCONT arrays to allow for different New Rome vertical hydraulic conductivities.

Results and discussion

Calibrated heads were within 0.6 m of measured heads (Figures 4.4, 4.5), with the largest discrepancies in the south-central portion of the study area. Sensitivity analyses indicate New Rome K_v is greater than 10^{-9} m s^{-1}. When the model incorporated a K_v of 10^{-9} m s^{-1}, heads in the layer below the New Rome increased dramatically over those observed in the field. Based on the sensitivity analysis and generic values for materials of this texture, the K_v is likely in the range of 10^{-6} to 10^{-8} m s^{-1} .

Overall the model appears to be a fairly good representation of actual field conditions. The greatest uncertainty is where the hydraulic conductivity was adjusted in the calibration procedure near well locations 0503 and 0801. These adjustments may or may not be reasonable. Some uncertainty also exists regarding bedrock elevations. Additional field work would be required to ascertain field-truth. The adjustments made throughout the remainder of the study area seemed to be reasonable,

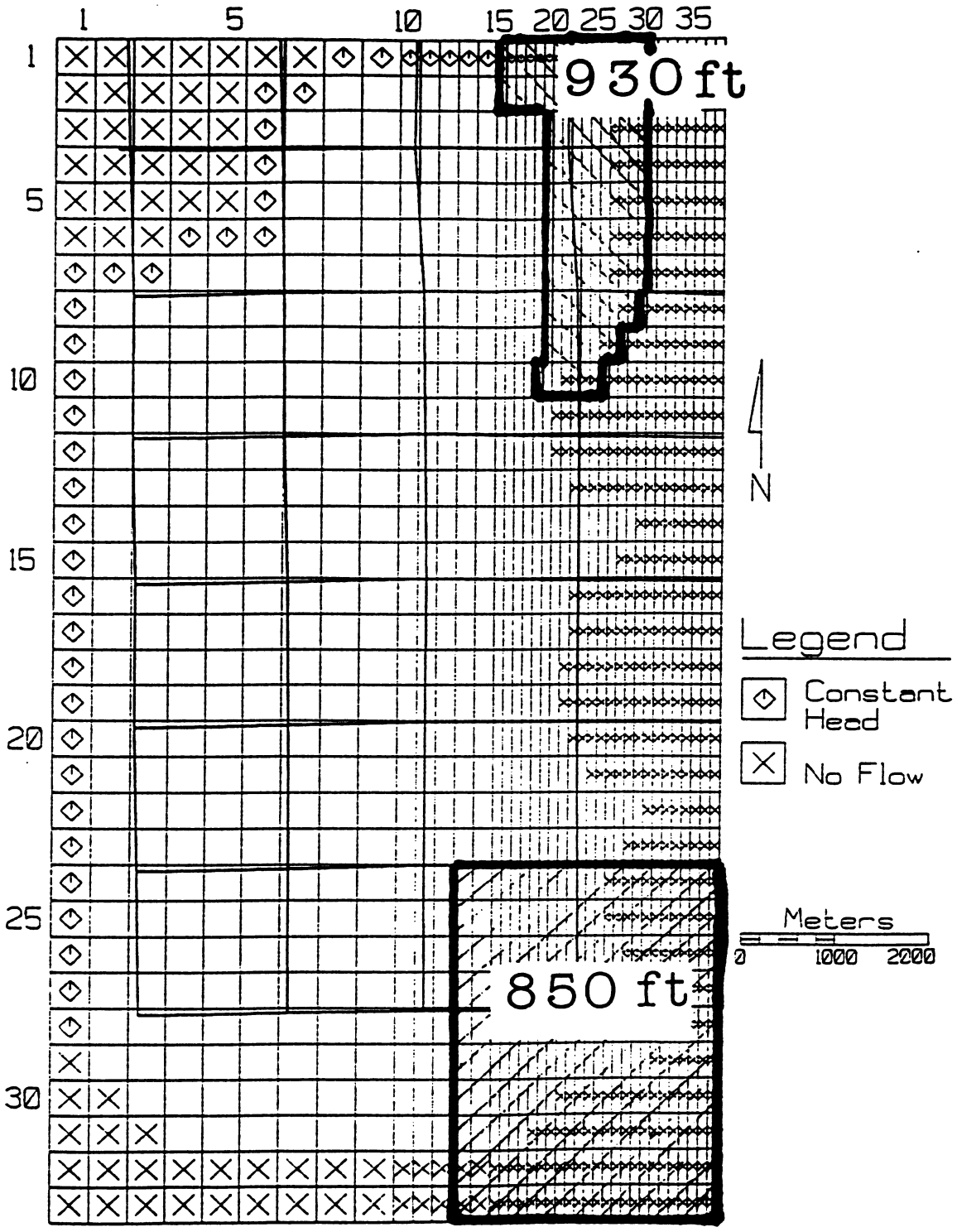


Figure 4.2 Areas where bedrock elevations were changed to improve model calibration.

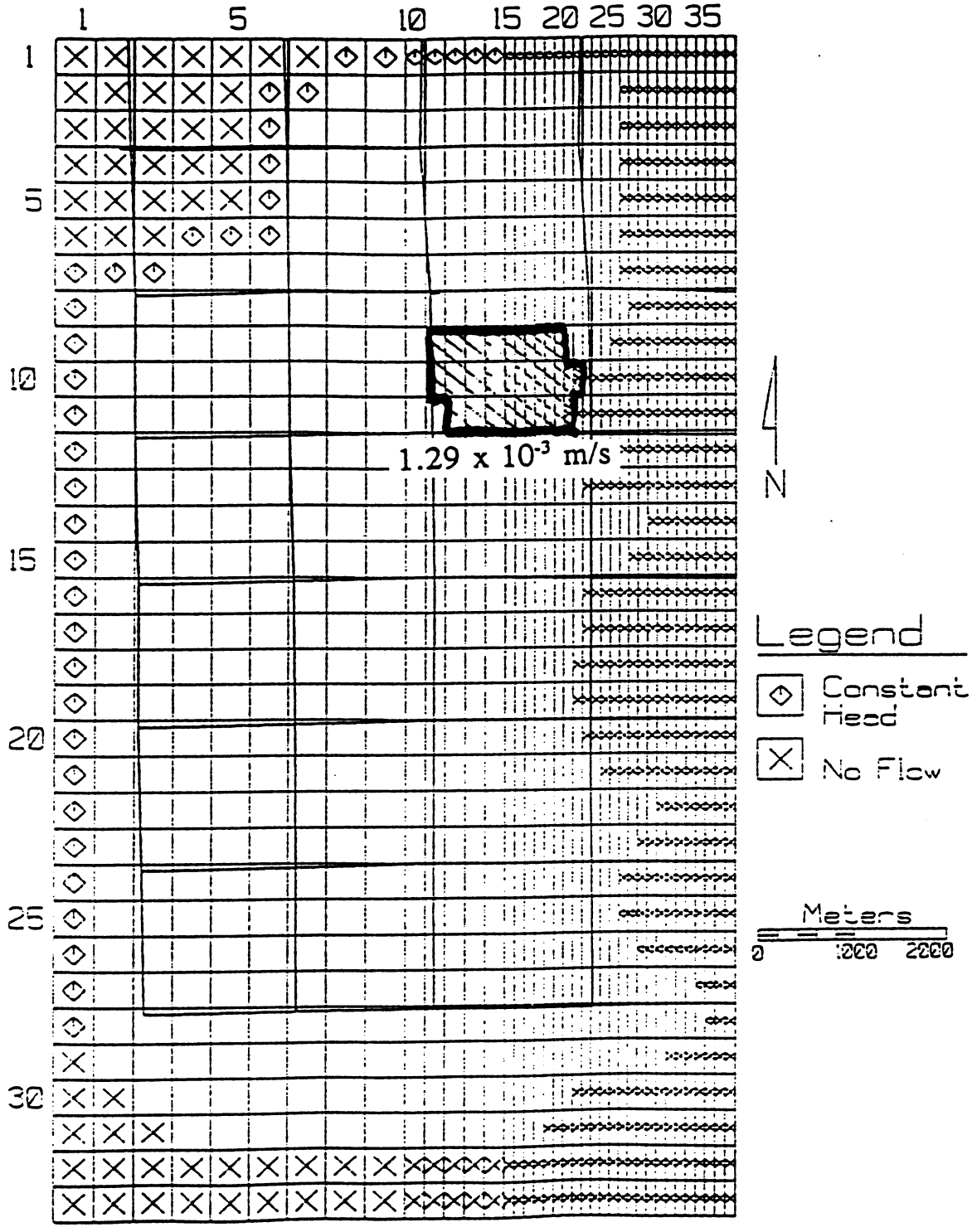


Figure 4.3 Area where hydraulic conductivity was increased to improve model calibration.

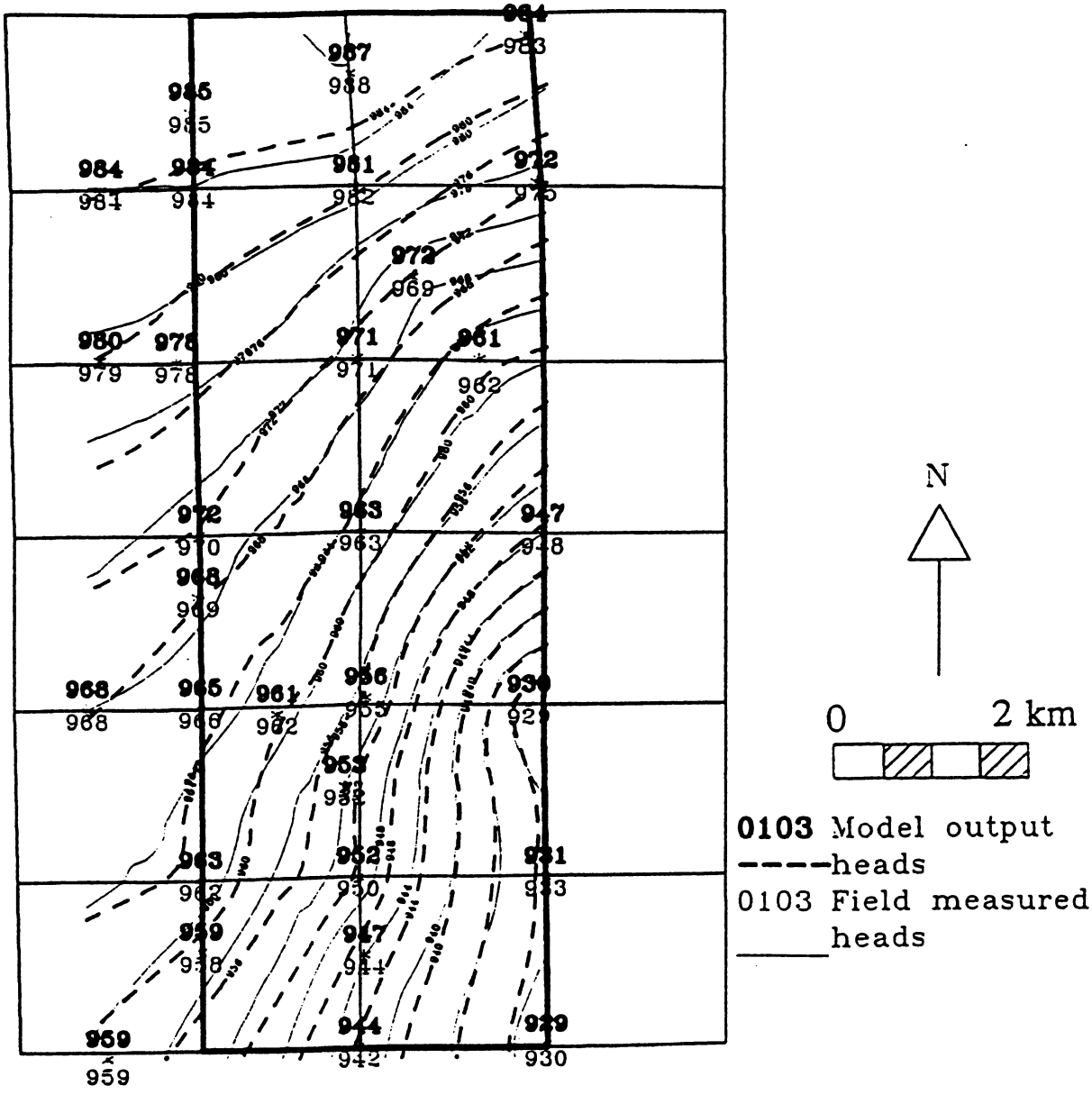


Figure 4.4 One-layer calibrated vs. measured heads.

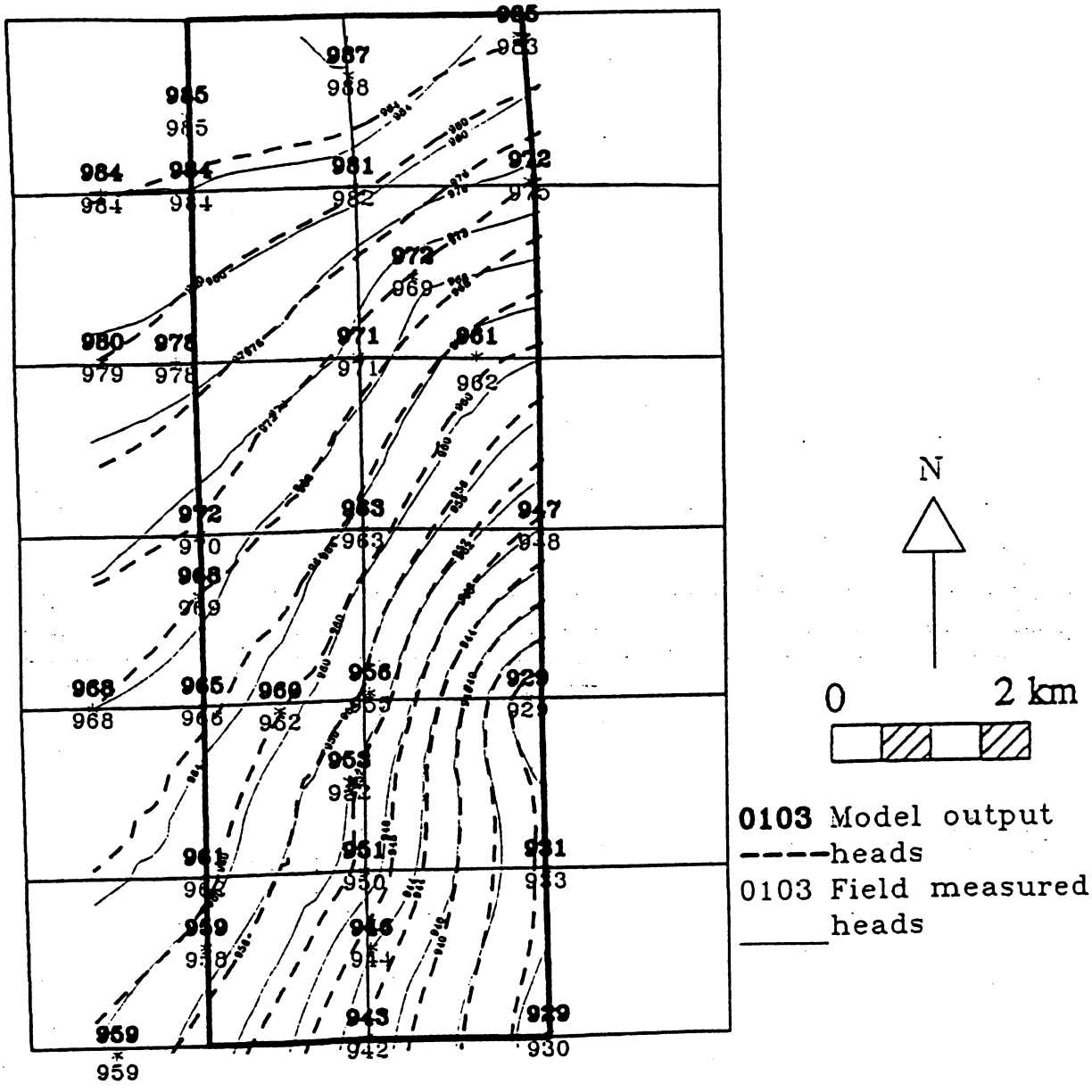


Figure 4.5 Three-layer calibrated vs. measured heads with $K_v = 10^{-7}$ m s^{-1}.

closely matching the measured field data. A longer period of record will be needed before the model can be validated (Anderson and Woessner, 1992; p. 253).

PARTICLE TRACK MODELING

Particle tracking models trace the path of groundwater movement through a flow system. The particle-tracking computer code used in this study is the FORTRAN-77 based PATH3D (Zheng, 1989). PATH3D is able to follow a fictitious fluid particle of known origin along a flowpath forward in time, or it can be used to determine the origin of a particle by going backward in time. It can be run in either steady-state or transient mode. The code consists of two components: a velocity interpolator that converts a hydraulic field into a velocity field, and a numerical solver that calculates particle movement and position. PATH3D uses the head solution and input files directly from MODFLOW. In addition, it requires layer thicknesses, porosity, and an input file of the x, y, and z coordinates of particle origins. Output consists of the x, y, z, and time coordinates along each particle's path. Particle coordinates can be plotted to display pathlines. PATH3D was used in this study to

- delineate horizontal flowlines across the PEGPW;
- delineate flow paths from fields to discharge zones; and
- illustrate a vertical profile of flowpaths from two fields.

Descriptions of simulations

Flowline delineation was performed beginning at twenty-six points selected along a north-south transect along the eastern edge of the study area. PATH3D was run backwards in time to trace flow paths back towards their origins. The one-layer MODFLOW simulation was used.

Flow from fields to discharge zones was delineated using 12-16 particles to outline the boundary of each field. PATH3D was run forward in time until particles had reached a model boundary. Again, the one-layer MODFLOW simulation was used. Output from this simulation is useful to determine how large an area is impacted by each field and in interpreting monitoring results.

A vertical profile of advective transport was traced for Fields 1 and 2, located in the northern part of the PEGPW. PATH3D simulations utilized the three-layer MODFLOW simulations. Two runs were made to incorporate New Rome vertical hydraulic conductivities of 10^{-7} and 10^{-8} m s^{-1}. We attempted vertical particle tracking for other fields as well. However, an apparent code error precluded us from doing so.

A vertical profile of advective transport was traced for Fields 1 and 2, located in the northern part of the watershed. PATH3D simulations utilized the three-layer MODFLOW simulations. Two runs were made to incorporate New Rome vertical hydraulic conductivities of 10^{-7} and 10^{-8} m s^{-1}. We attempted vertical particle tracking for other fields as well. However, an apparent code error precluded us from doing so.

Results and discussion

Groundwater flow lines delineated by PATH3D (Figure 4.6) matched closely the flow lines drawn subjectively from equipotential maps (Figure 4.7). This provided some intuitive verification of the accuracy of flow and particle-tracking models.

Figure 4.8A shows the amount of land influenced by plumes from irrigated agricultural fields, as predicted by PATH3D. Approximately 54% of PEGPW was underlain by the plumes. Figures 4.8B-D illustrate the plumes emanating from individual fields. These figures also illustrate how the plumes behaved as they moved toward the discharge zone, the Wisconsin River and its backwaters.

Groundwater flow generally converges as it approaches discharge zones, so if dispersion is ignored, one would expect solute plumes to contract in the dimension perpendicular to regional groundwater flow. This phenomenon was displayed by the plumes emanating from most fields in the PEGPW, e.g., Fields 1, 2, and 3 (Figure 4.8A). However, PATH3D simulations indicated that plumes from Fields 6, 15, 17, 18, and 20 displayed the opposite behavior, increasing in size as they approached the discharge zone. This was due to divergence in groundwater flow brought about by the sinuosity of the discharge zone.

Figures 4.9-4.11 illustrate a vertical profile of plume movement from Fields 1 and 2 incorporating New Rome K_vs of 10^{-7} and 10^{-8} m s^{-1}. These profiles illustrate convergent paths of particles originating up- and downgradient of each field. Predicted particle paths were highly dependent upon New Rome K_v. We doubt that this truly mimics reality, but further examination is beyond the scope of this project.

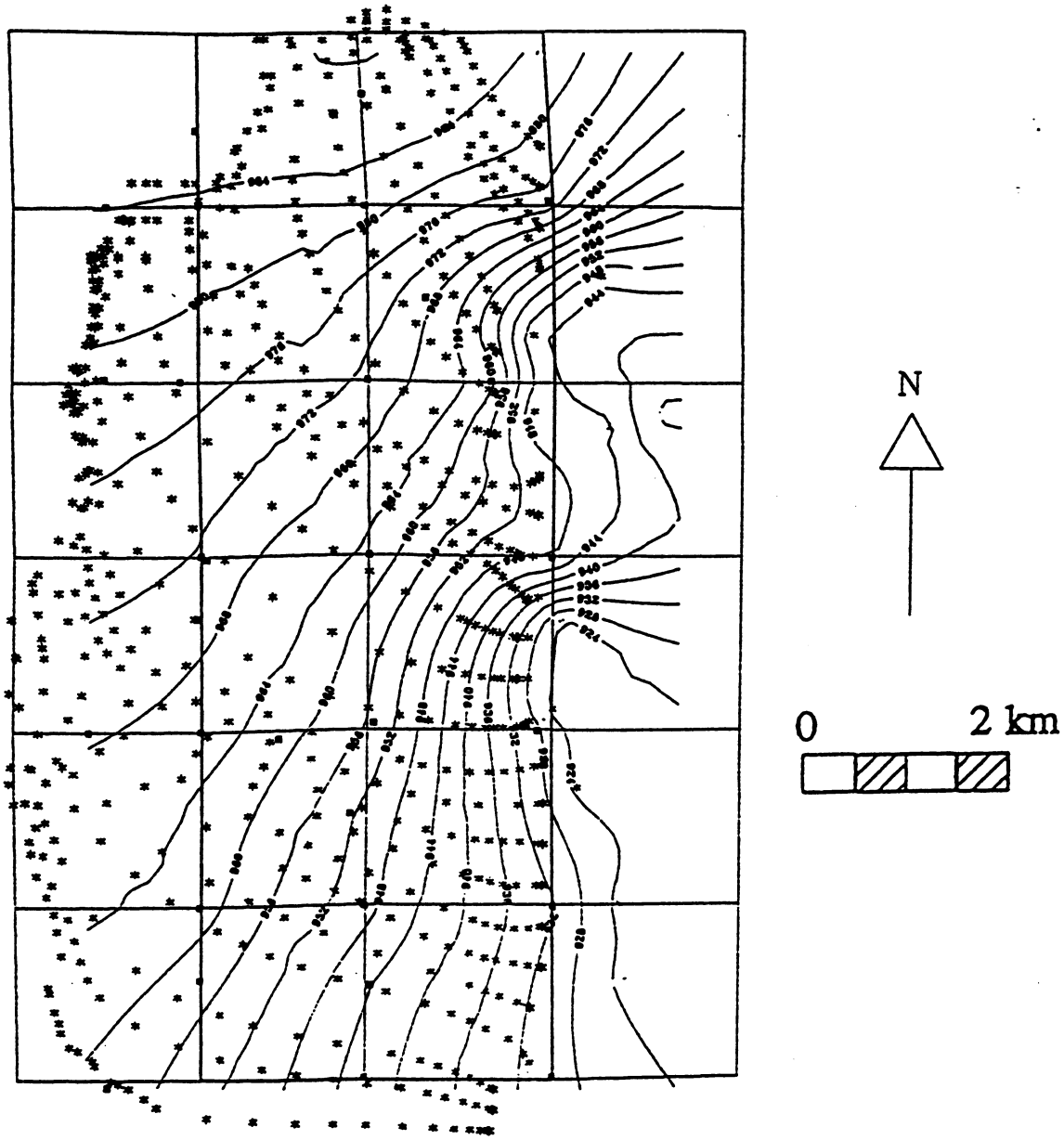


Figure 4.6 Flow lines generated by PATH3D.

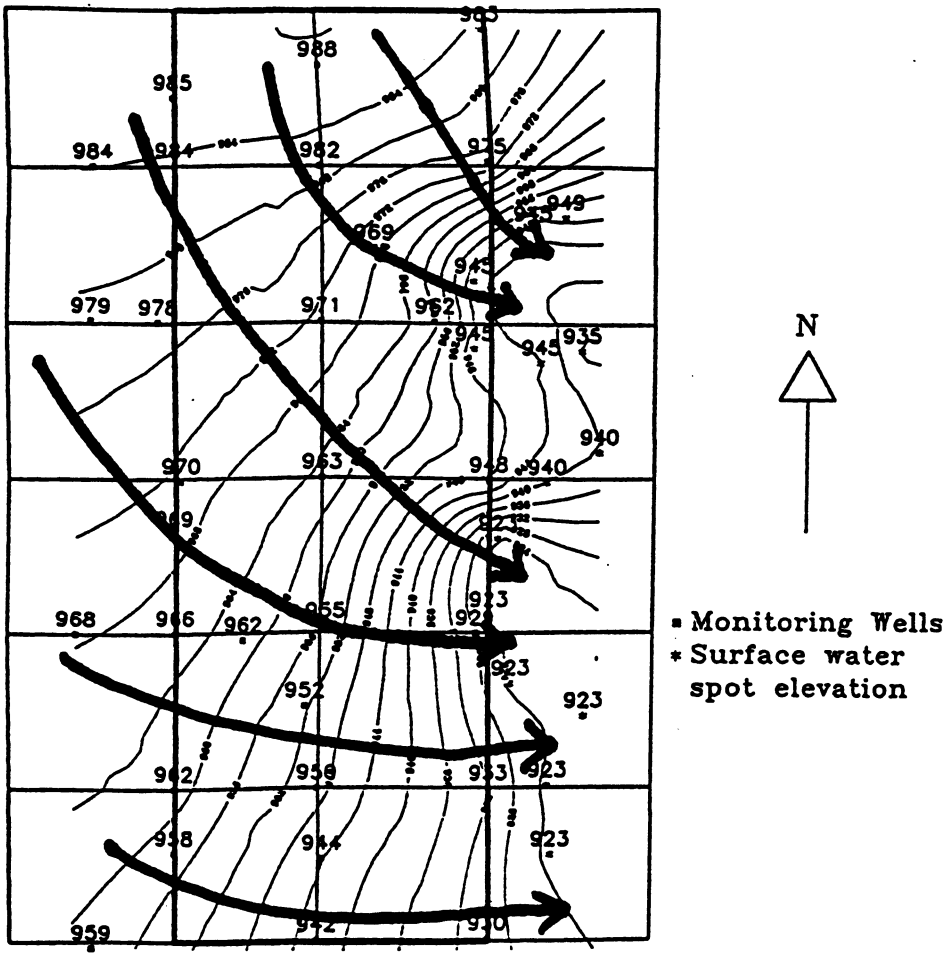


Figure 4.7 Flow lines determined from equipotential map.

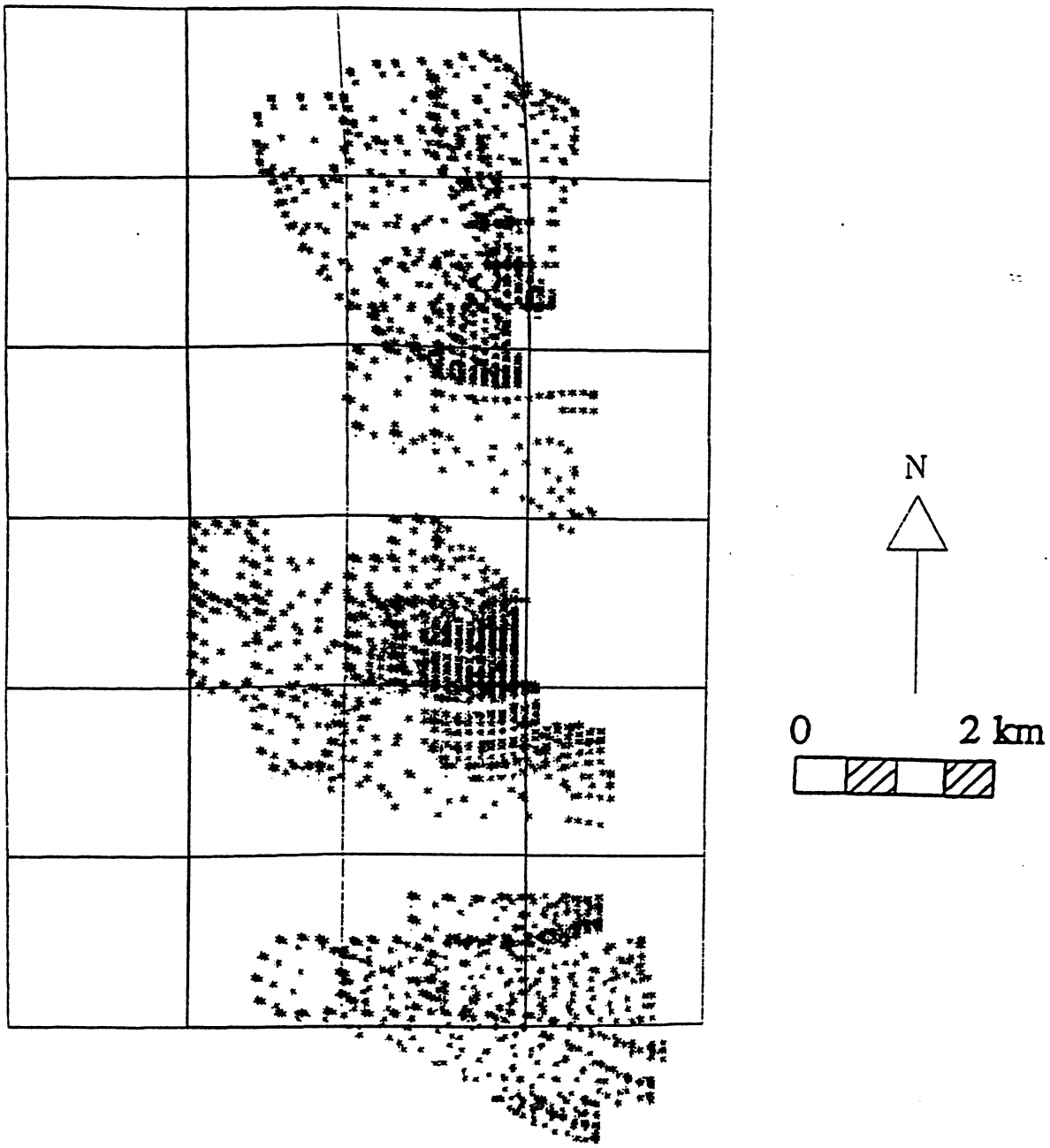


Figure 4.8A Advective solute plume migration from all irrigated fields (see text).

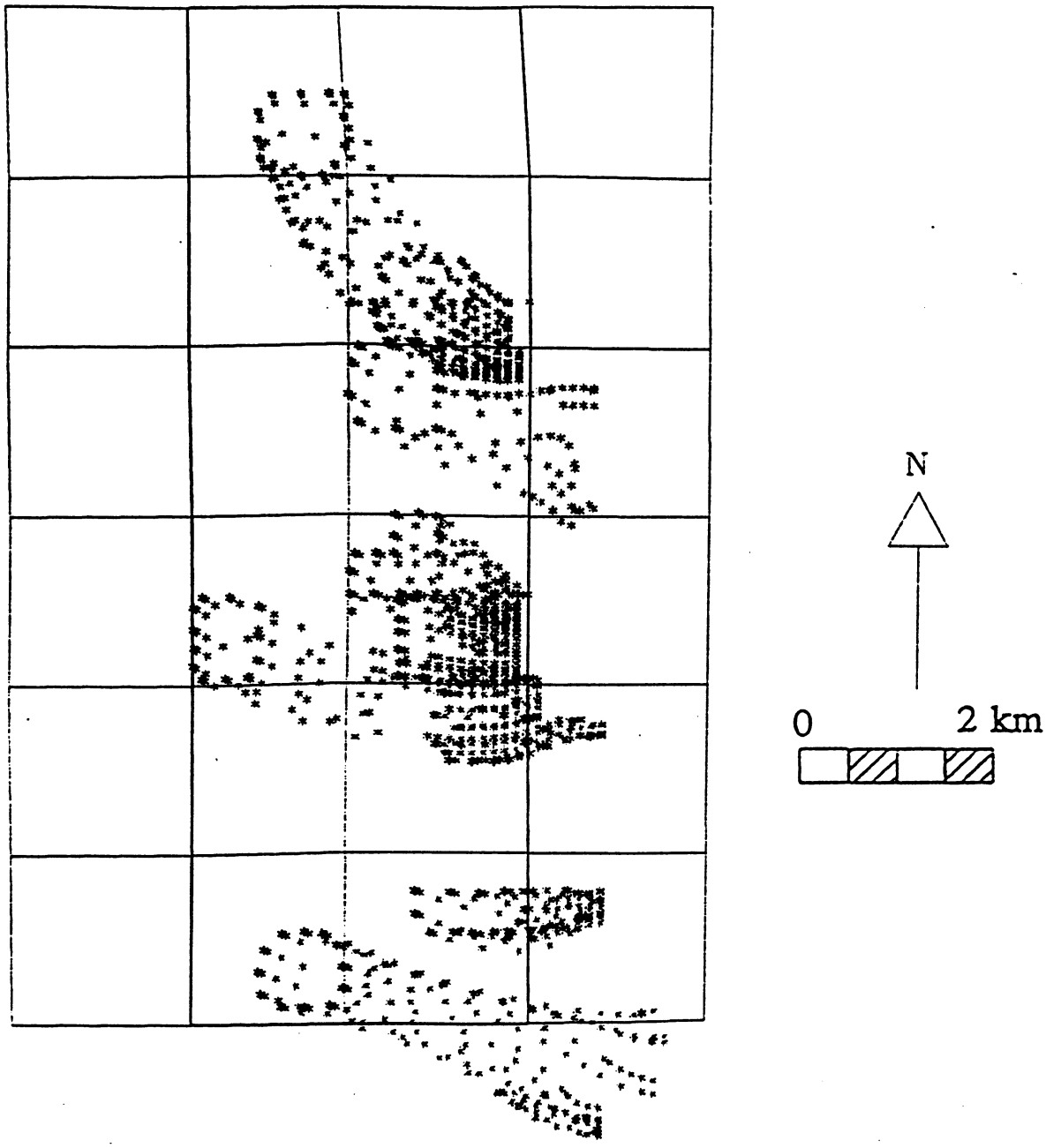


Figure 4.8B Advective solute plume migration from Fields 1, 5, 6, 8, 9, 11, 16, 17, and 19.

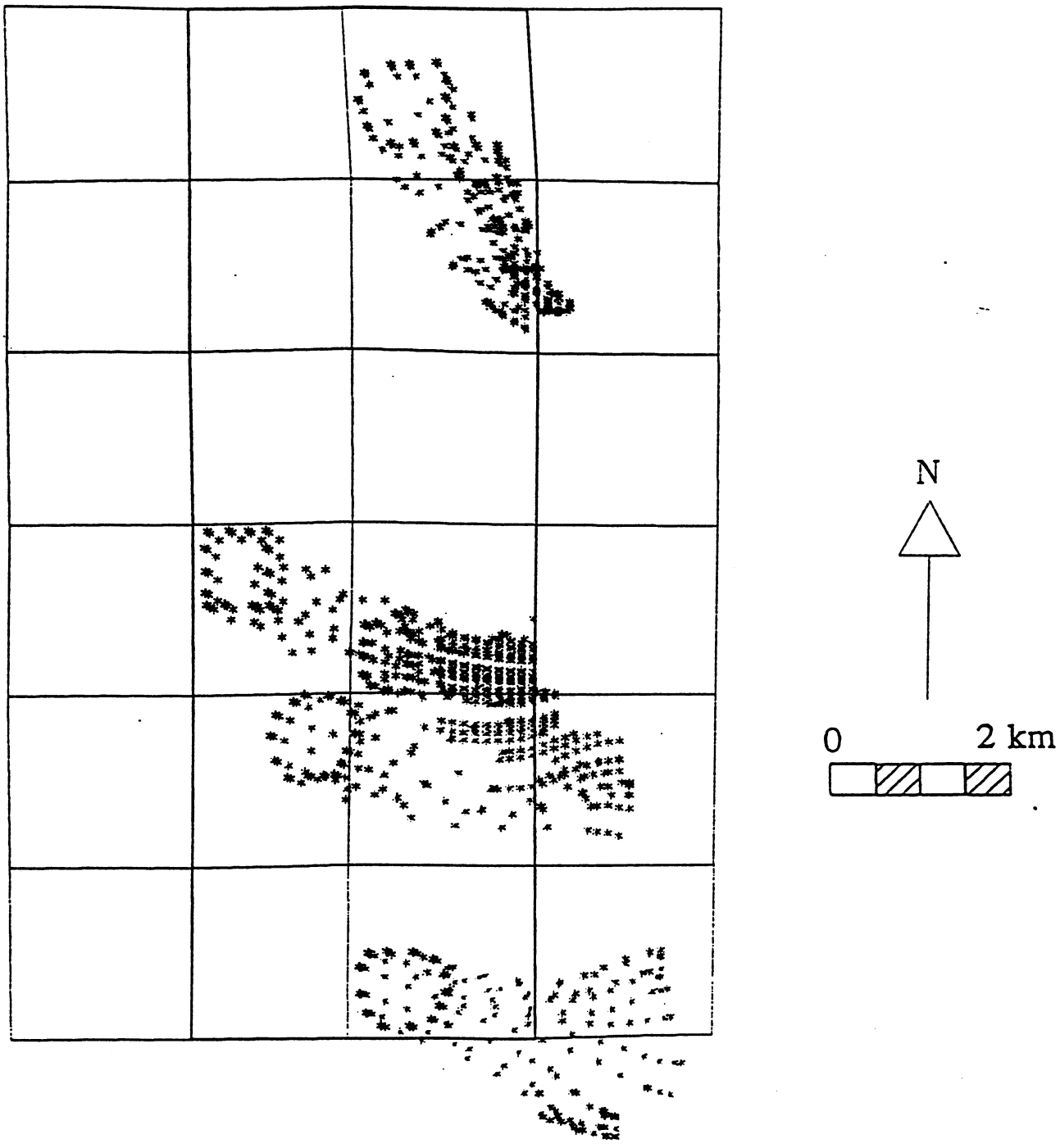


Figure 4.8C Advective solute plume migration from Fields 2, 4, 7, 10, 15, and 18.

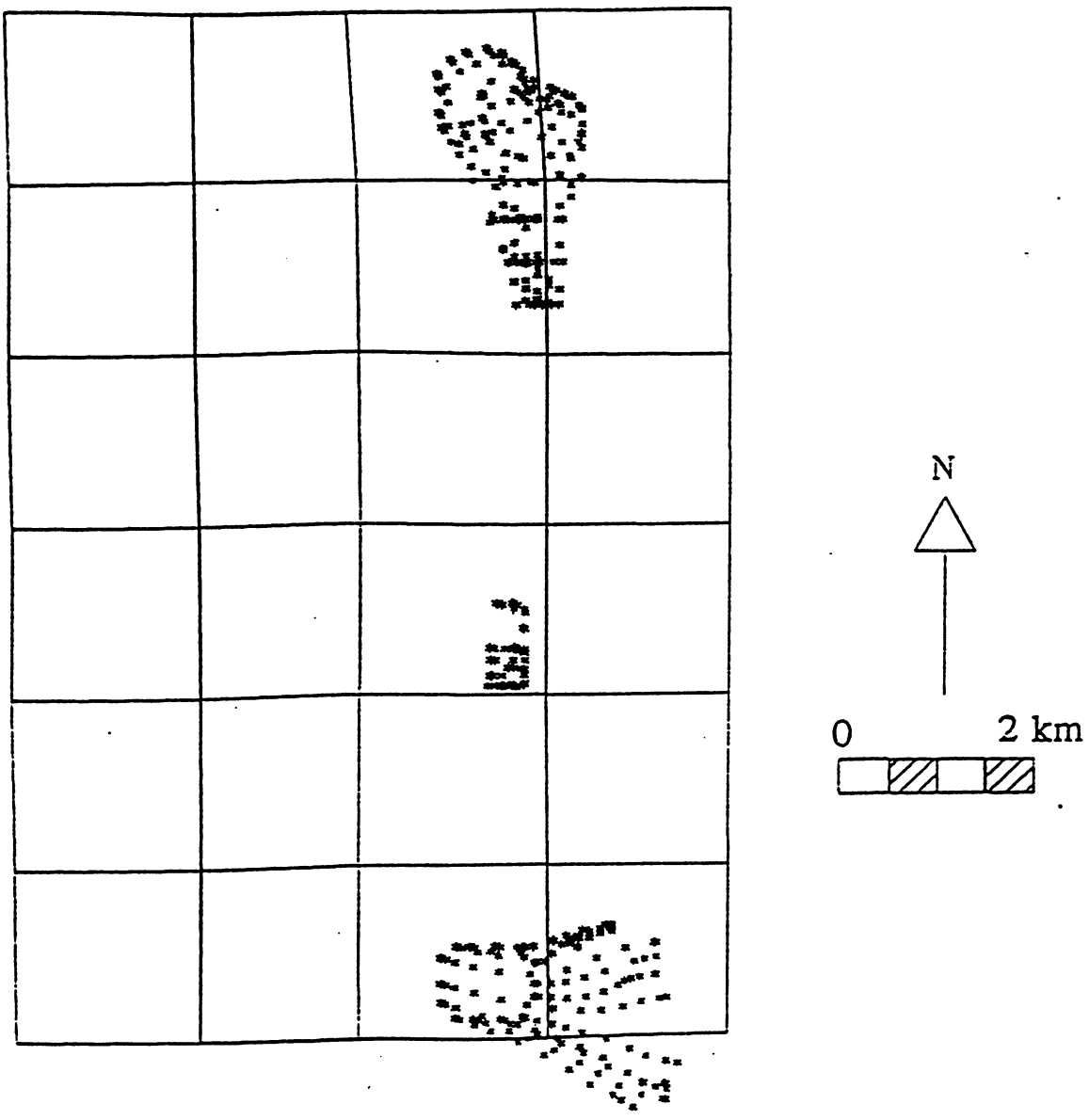


Figure 4.8D Advective solute plume migration from Fields 3, 12-14, and 20.

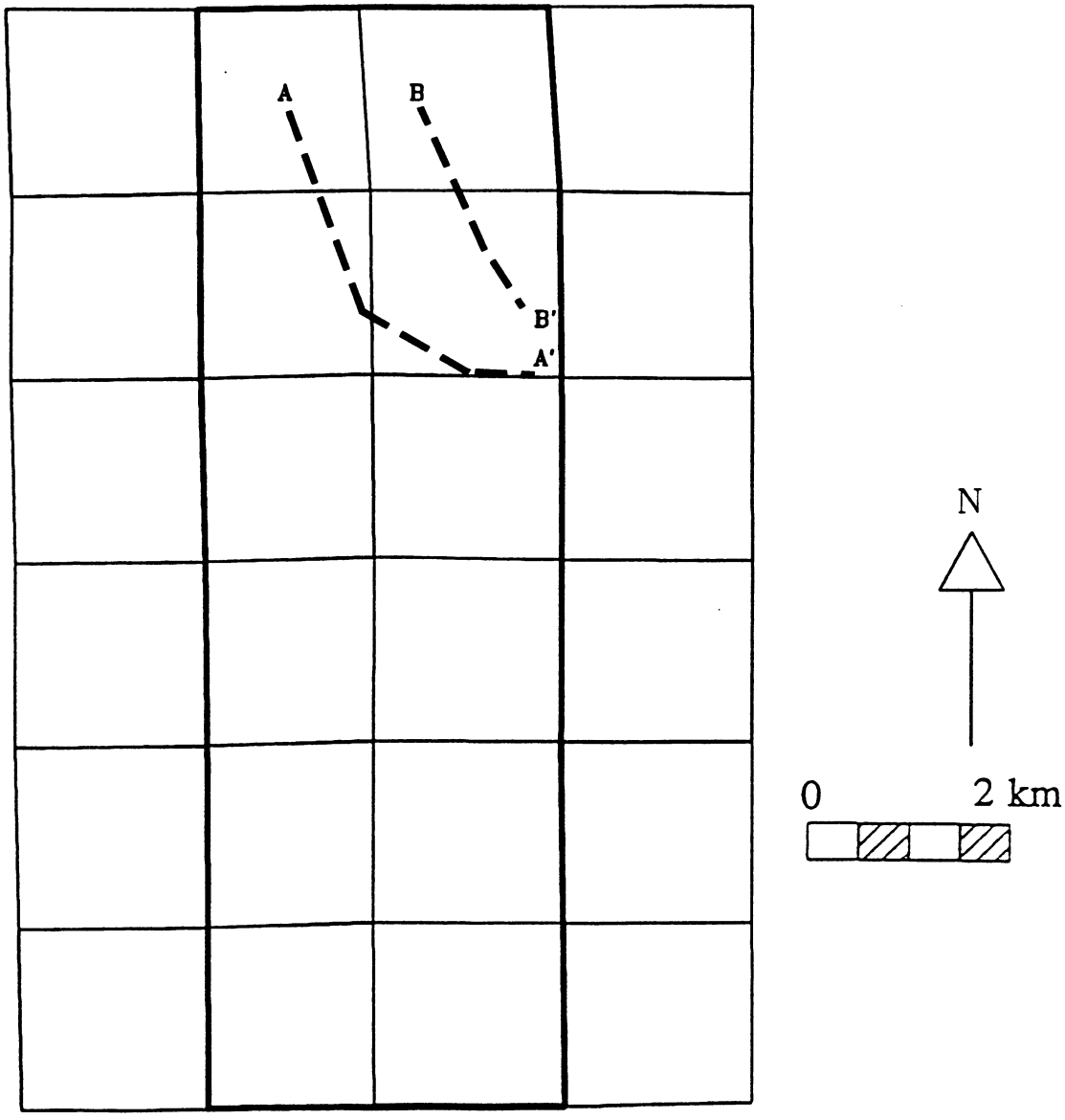


Figure 4.9 Location of vertical profiles from Fields 1 and 2.

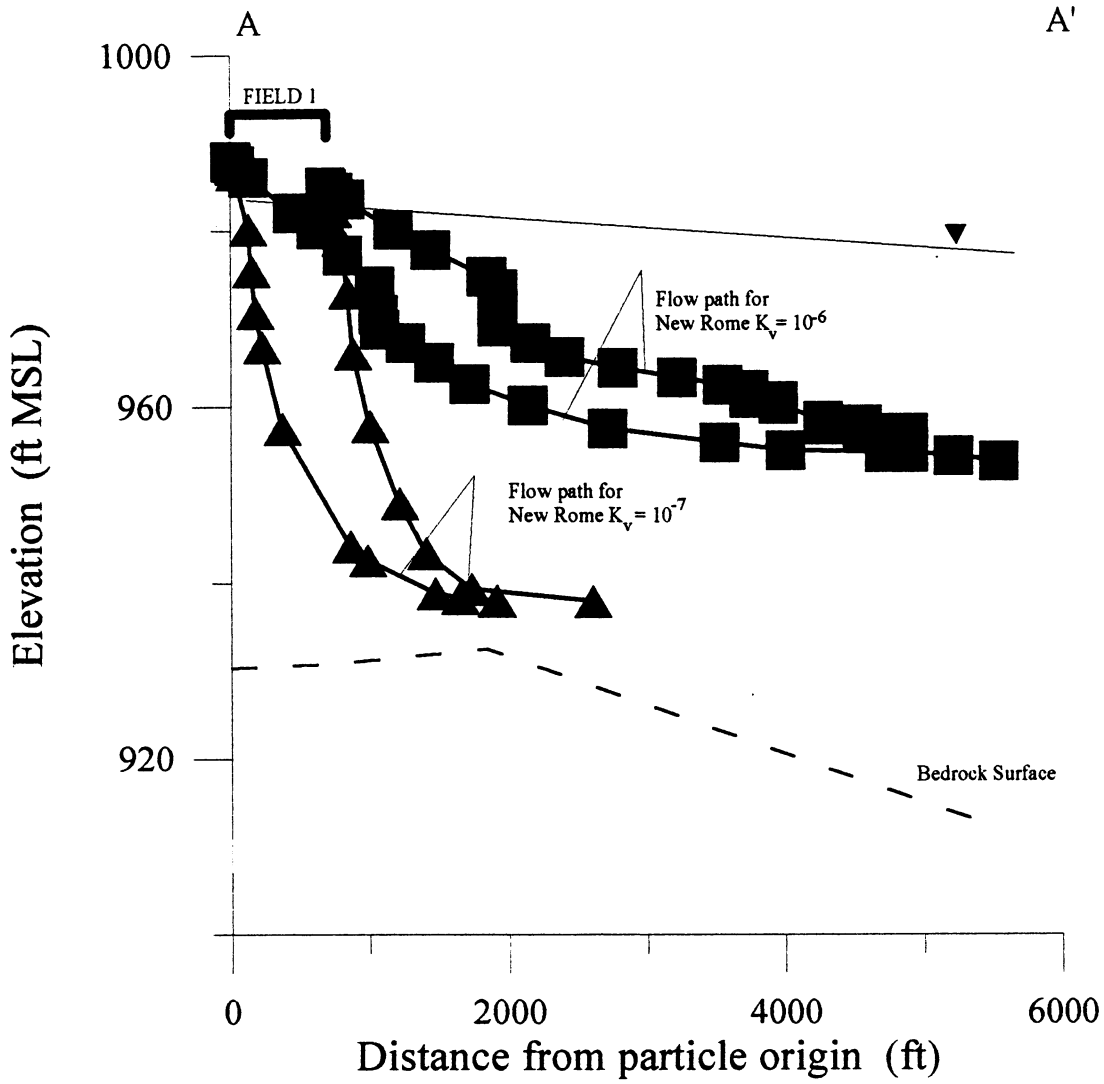


Figure 4.10 Vertical cross-section of plume beneath Field 1.

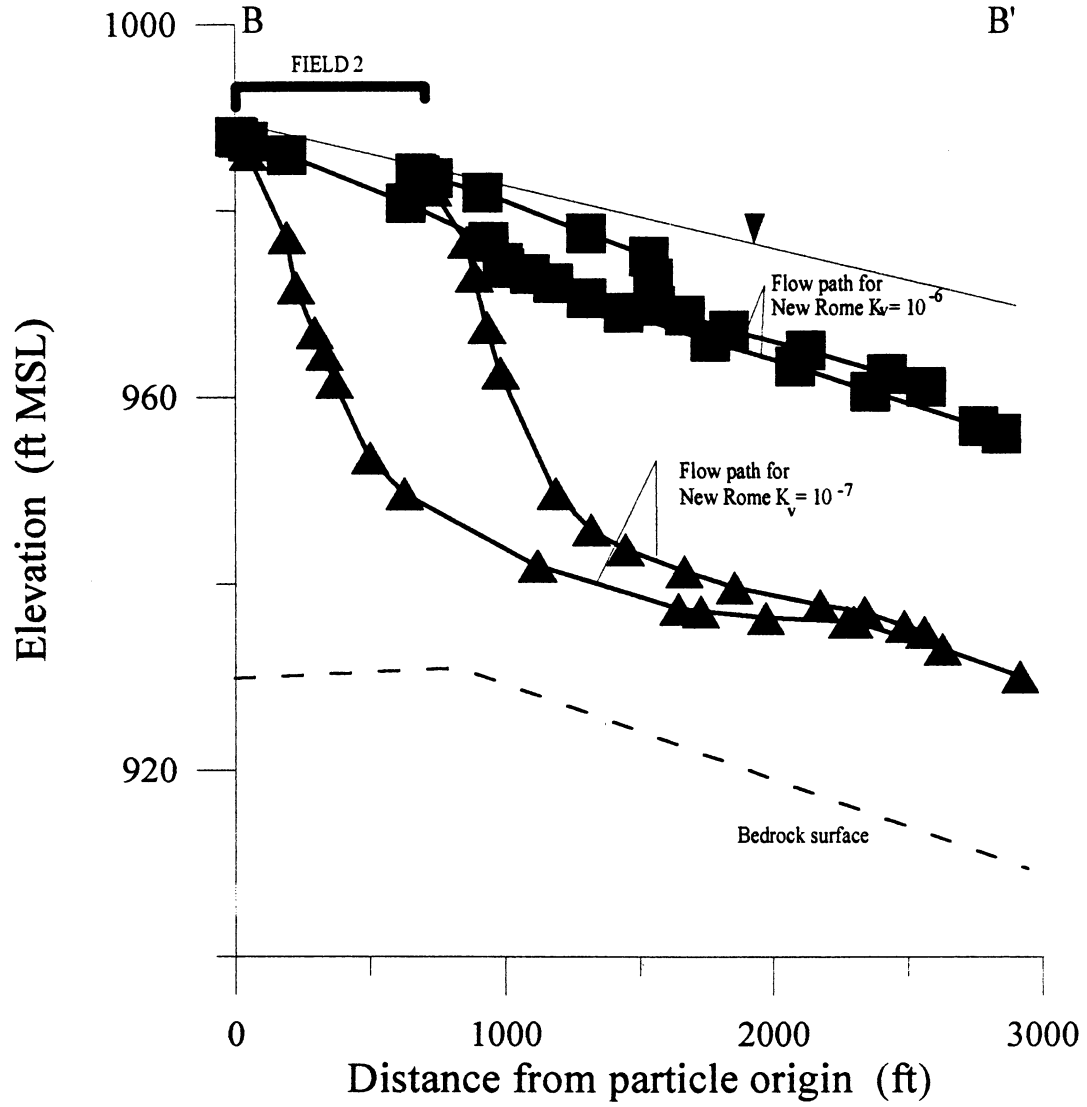


Figure 4.11 Vertical cross-section of plume beneath Field 2.

CHAPTER 5

GROUNDWATER QUALITY CONDITIONS IN THE PORT EDWARDS GROUNDWATER PRIORITY WATERSHED

The groundwater quality of the Port Edwards Groundwater Priority Watershed was investigated to characterize prevailing conditions and to assess the impacts of major land uses, particularly irrigated vegetable agriculture.

METHODS

Groundwater was sampled from 92 monitoring wells at 21 locations throughout the watershed in August 1992, March 1993, August 1993, and March 1994. Sampling procedures consisted of well purging, sample collection, sample preservation, and field analysis. Procedures generally followed the guidelines in Lindorff et al. (1987), and complied with NR 149, Wisconsin Administrative Code. Wells were purged with a centrifugal pump where the water table was less than about 7 m deep, and otherwise with a Waterra inertial pump[1]. Samples were collected with either a peristaltic pump or a Waterra pump, depending on depth to water table. Water samples were preserved or analyzed in the field, depending on the type of analysis for which they were bound, according to one of five procedures:

(1) pH, temperature, specific conductance, dissolved oxygen (DO). Measured in the field according to procedures in Table 5.1.

(2) Al, Ca, Fe, K, Mg, Mn, Na, SO_4. Samples were field-filtered through a 0.45-μm filter into 250-mL bottles containing approximately 0.25 mL of concentrated nitric acid.

(3) Cl, NO_3. Samples were field-filtered through a 0.45-μm filter into 250-mL bottles containing approximately 0.25 mL concentrated sulfuric acid.

(4) Alkalinity. Samples were filtered but not preserved.

(5) Pesticides. Samples were neither filtered nor preserved and were collected in 1-L glass bottles.

All samples were cooled in ice or in a refrigerator until analysis could take place.

Analytical methods and holding times are listed in Table 5.1. Metals were analyzed by either atomic absorption (AA) or inductively coupled plasma emission spectrophotometer (ICP). Method references (Table 5.1) are for AA; ICP analyses used similar preparation procedures, but performed quantitations similar to

---

[1]Waterra Pump Ltd. 77 Mourat, Suite 101, Toronto Ontario M6K3E3 Canada

APHA Method 305. Methods for detecting pesticides changed during the study. Method 507 was used for the August 1992 sampling, and a modified Method 8270 was used for the second and third samplings. The change in procedures was done to permit carbamate detection. The procedures were able to detect the following pesticides:

- vernolate (Vernam)
- propachlor
- propazine
- metolachlor (Dual)
- oxadiazon
- butylate (Sutan)
- molinate (Ordam)
- trifluralin (Treflan)
- de-ethyl atrazine
- metribuzin (Sencor)
- isopropalin (Paarlan)
- hexazinone (Velpar)
- alachlor (Lasso)
- carbofuran
- benfluralin (Balan)
- simazine
- profluralin (Tolban)
- pendimethalin (Prowl)
- EPTC (Eptam)
- pebulate (Tillam)
- cycloate (Ro-Neet)
- atrazine
- terbacil
- bromacil
- oxyfluorfen (Goal)
- diazinon
- de-isopropyl atrazine
- cyanazine

RESULTS

Inorganic analyses

Overview

Ninety-two monitoring wells at 21 locations (Figure 5.1) throughout the watershed were sampled on three or four dates. Analyses were almost always consistent for individual wells over the different sampling rounds, hence, analyte concentrations in individual wells are adequately represented by an average over the rounds.

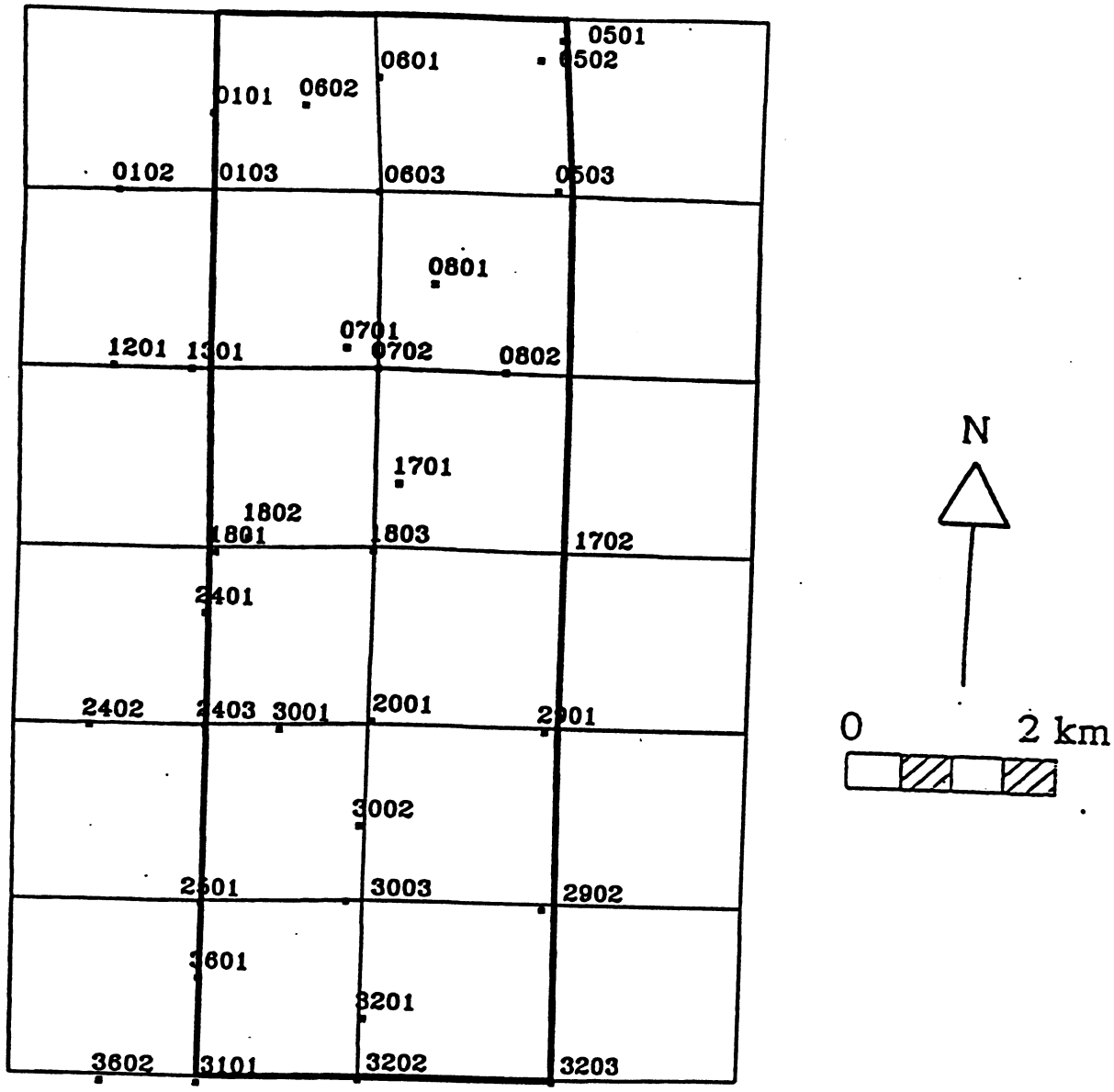


Figure 5.1 Port Edwards Groundwater Priority Watershed monitoring well locations.

Table 5.1 Analytical methods and holding times.

Parameter	Method	Max. hold time
pH	Potentiometric	N/A
Specific conductance	Electrometric	N/A
Temperature	Thermistor or mercury thermometer	N/A
Dissolved oxygen	Walton-Day et al., 1990; White et al., 1990.	N/A
Alkalinity	Titration. Method 403 in APHA, 1985.	14 d
Chloride	Automated colorimetry. Lachat Instruments, QuikChem Method 10-117-07-1-A. August, 1986.	28 d
Nitrate plus nitrite N	Automated colorimetry. Lachat Instruments, QuikChem Method 10-107-04-1-A. April, 1989.	28 d acidified, else 48 h
Calcium	AA1 or ICP2. APHA3 Method 311A.	14 d
Magnesium	AA or ICP. APHA Method 318A.	14 d
Sodium	AA or ICP. APHA Method 325A.	14 d
Potassium	AA or ICP. APHA Method 322A.	14 d
Iron	AA or ICP. APHA Method 215A.	14 d
Sulfate	Automated colorimetry. Technicon Industrial Systems, Industrial Method 118-71W/B. November, 1978.	7 d
Manganese	AA or ICP. APHA Method 229A.	14 d
Aluminum	AA or ICP. APHA Method 306A.	14 d
Pesticides	Method 507 Modified Method 8270.	(4)

1 AA - quantitation by atomic absorption.
2 ICP - quantitation by inductively coupled plasma emission spectrophotometry.
3 APHA, 1985
4 Extraction within 14 days after collection; analysis within 40 days after extraction.

Additional groundwater quality data are available from the four intensively monitored fields in the northern part of the watershed (Chapter 6). Thirty multi-level piezometers are located there, six in each field plus six upgradient of the fields. Each piezometer consisted of 19-25 screens. Sampling rounds were conducted at one- to two-month intervals. We wished to include these data in characterizing groundwater quality without excessive bias due to the large number of sampling points. To avoid this bias, we treated monitoring data obtained from the multi-level piezometers as follows:

- Data from multi-level piezometers in each field were averaged and treated as one well per field.
- Data from upgradient multi-level piezometers associated with Field 2 (numbers 7 and 8) were averaged and treated as one well.

- Data from upgradient multi-level piezometers F1W7 and F3W7 were individually averaged and treated as two wells.

This scheme resulted in a total of 99 monitoring points; 92 from the watershed monitoring network, four from in-field MLPs, and three from upgradient MLPs.

Data summary and charge balance

Table 5.2 summarizes inorganic analyses. Statistics in this table were calculated from sampling-round averaged analyte concentrations in each monitoring well. The mean, median, and standard deviation were calculated from values above the detection limit.

Table 5.2. Summary of composite inorganic monitoring data (see text).

			Range of detects				
Analyte1	No. wells with detects	No. wells sampled	Min.2	Max.	Mean	Median	Std. dev.
Al	99	99	0.03	5.30	0.29	0.10	0.69
Ca	99	99	0.04	56.9	18.6	15.8	12.6
Fe	76	99	-0.02	16.4	1.81	0.09	3.70
K	99	99	0.04	24.8	2.29	1.15	3.93
Mg	99	99	0.04	21.9	7.17	6.10	5.15
Mn	74	99	-0.01	0.90	0.11	0.08	0.14
Na	99	99	0.04	17.5	3.43	2.90	2.27
Cl	72	100	-1.00	58.7	9.51	8.33	12.9
NO_3-N	80	100	-0.20	26.4	5.52	2.70	7.88
SO_4	99	99	5.20	43.7	15.6	12.5	7.86
Alkalinity (meas.)	99	99	2.25	107	26.0	22.5	20.6
Alkalinity (calc.)	35	99	8.09	109	18.8	35.0	30.3
Conductivity	90	90	63.0	443	202	187	102
Diss. Oxygen	83	85	-0.05	6.70	2.40	2.20	1.66
pH	90	90	4.89	8.70	6.76	6.89	0.93

1 All concentration values mg L^{-1}, except pH (standard units), conductivity (μS/cm), and alkalinity (mg L^{-1} $CaCO_3$ equivalent).
2 Negative value means not detected at absolute value of number.

A charge balance was calculated for each sample with complete analyses. The charge balance error is calculated as (Freeze and Cherry, 1979)

$$\% \; CBE = \frac{\Sigma(cations) - \Sigma(anions)}{\Sigma(cations) + \Sigma(anions)} * 100$$

where

% CBE = percent charge balance error

(cations) = sum of milliequivalents of Al, Ca, Fe, K, Mg, Mn, Na

(anions) = sum of milliequivalents of Cl, NO_3, SO_4, and HCO_3/CO_3 alkalinity

About 60% of samples were within ± 5% CBE, a tolerance considered desirable by Freeze and Cherry (1979). The anion sum was generally lower than the cation sum. We suspect that missing alkalinity was the cause. Alkalinity measurements in low-alkalinity samples are subject to error (APHA, 1985). Others working in the WCSP (Fathulla et al., 1988; Brasino, 1986) have experienced similar problems with charge balances and also concluded that low alkalinities were the problem. While the % CBE is sometimes large, the numerical amount of missing alkalinity is small. Therefore, we have elected to use measured alkalinities in the remainder of this report rather than calculated alkalinities (cf. Fathulla et al., 1988).

Description of groundwater chemistry

The groundwater chemistry of the watershed was the Ca-Mg-HCO_3 type, with Cl, Na, SO_4, and NO_3 as other major ions. Generally, these analytes had concentrations of less than 30 mg L^{-1}. Minor ions included K, Fe, Mn, and Al. These usually had concentrations of less than 10 mg L^{-1} and frequently below 1 mg L^{-1}.

Groundwater chemistry was similar to that previously reported for other parts of the WCSP (Rothschild, 1982; Hindall, 1978; Batten, 1989). The pH range of 4.89 to 8.70 slightly exceeded the overall WCSP range of 5.0 to 8.2 reported by Rothschild (1982). Average alkalinity (26 mg L^{-1} $CaCO_3$), Ca (18.6 mg L^{-1}), and Mg (7.17 mg L^{-1}) concentrations were lower than averages reported for the WCSP (164 mg L^{-1} alkalinity as $CaCO_3$, 33 mg L^{-1} Ca, and 16 mg L^{-1} Mg; Hindall, 1978). The low concentration of dissolved carbonates in the groundwater may be due to a paucity of carbonate minerals in aquifer sediments. Evidence suggests carbonate minerals are common in the WCSP alluvial deposits near the moraine, but decrease in abundance westward from the moraine (Kraft, 1990; Clayton, 1986). The Rothschild (1982) and Hindall (1978) studies were closer to the moraine, and would have encountered groundwater in contact with more carbonate-rich aquifer sediments.

Average concentrations of Na and K and the range of Fe virtually coincided with those reported for the entire WCSP (Rothschild, 1982). Batten (1989) included data in his study of Wood County from a University of Wisconsin-Extension drinking water program that includes the watershed, and found NO_3 concentrations ranging from <0.2 to 31.5 mg L^{-1}, and Cl concentrations ranging from <1 to 70.5 mg L^{-1}. These compare closely with our results, although Batten's Cl range is higher.

Spatial and depth relationships

Some parameters, including pH, alkalinity, DO, Fe, Mn, and NO_3-N, were related to depth below the water table and location in the study area. The pH generally increased with depth below the water table, a trend also reported by Rothschild (1982). Exceptions to this trend arose in wells screened in the vicinity of the New Rome Member or other lacustrine layers, where pH tended to be lower. The pH in a given depth-below-water-table range varied areally as well (Figures 5.2A-C), generally increasing from north to south and from west to east. An exception to this general trend occurred at site 0801, a site that had a tendency toward other aberrant behavior (Chapter 4).

Alkalinity increased with depth, from 2.3 - 30 mg L^{-1} at the water table to 34 - 106 mg L^{-1} at depth, mimicking pH trends. Reducing conditions, indicated by elevated Fe and Mn and low dissolved oxygen, were frequently found at depth throughout the watershed as well as close to the water table in the area in the north-western part of the watershed containing poorly drained soils. Oxidizing conditions were found at depth at site 3203. DO was 5.5 mg L^{-1} and Fe and Mn were not detectable at a depth of 14 m below the water table. This was apparently due to steep vertical flow near the location which brings more oxic water to depth, and a lack of electron donors (e.g. organic materials) in sediments.

Elevated NO_3-N (>8 mg L^{-1}) concentrations were always associated with plumes from irrigated vegetable fields (Figure 5.3). Nitrate-N concentrations generally decreased with depth below the water table (Figures 5.4A-C), and were frequently <1 mg L^{-1} at depths >10 m. Causes of low NO_3 concentrations at depth may be:

(1) groundwater at a given location and depth entering the aquifer as recharge on non-agricultural lands, sometimes from outside the watershed;

(2) groundwater at the location and depth entering the aquifer on lands that are currently in agricultural use, but originated at a time prior to cultivation or conversion to vegetable production.

(3) denitrification occurring in the aquifer.

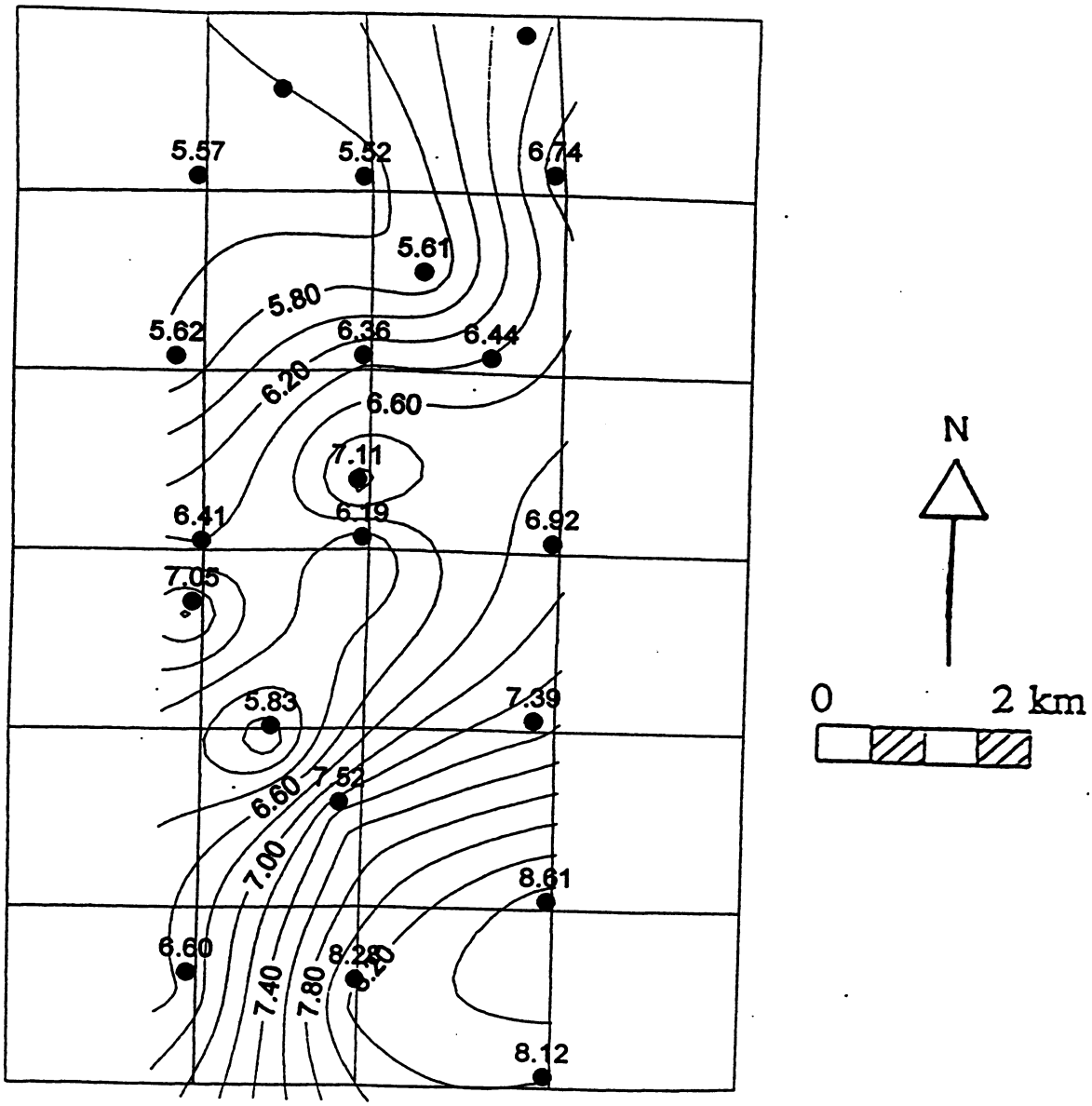


Figure 5.2A pH at depths less than 3 m below the water table.

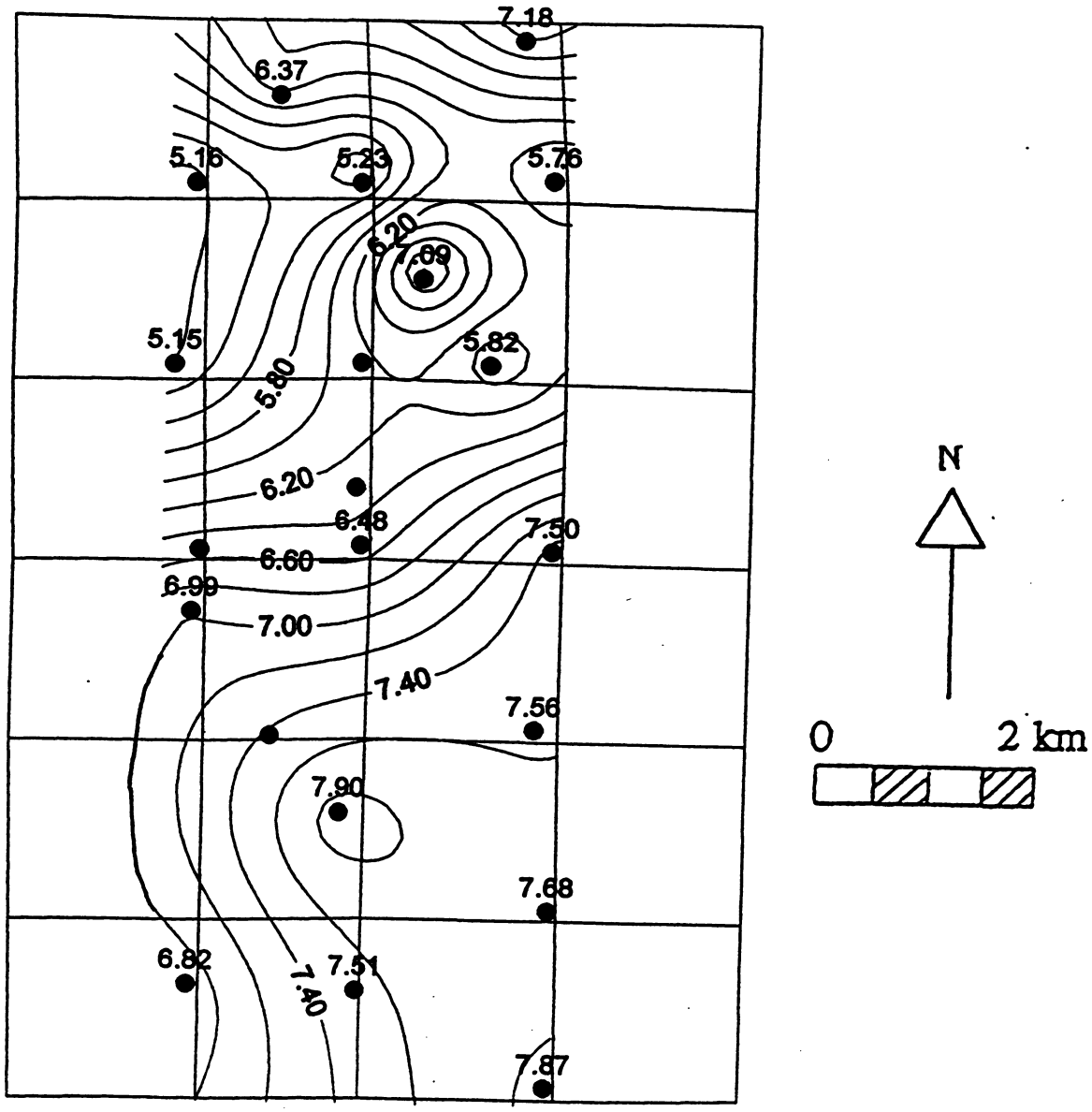


Figure 5.2B pH at depths between 3 and 10 m below the water table.

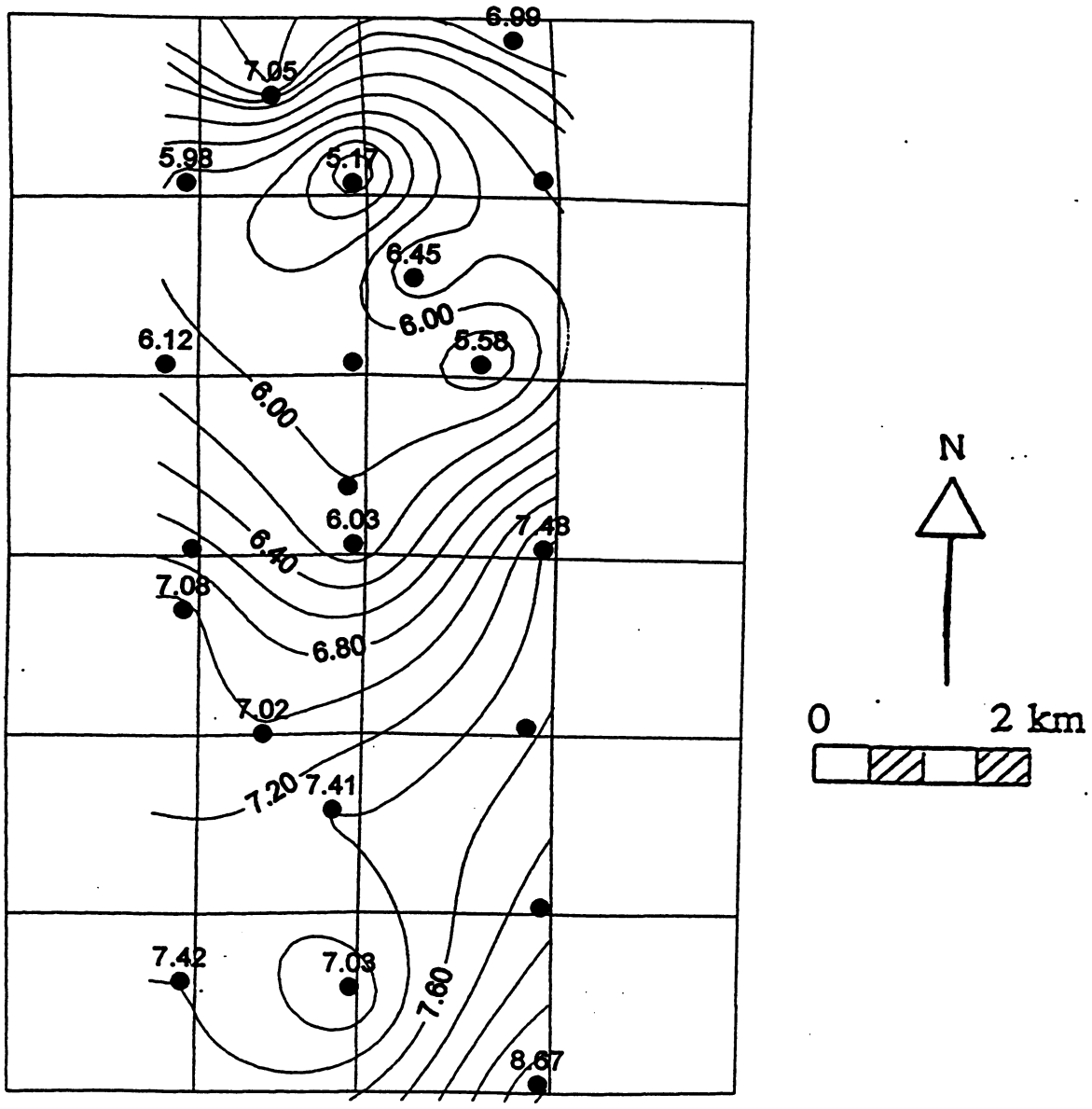


Figure 5.2C pH at depths greater than 10 m below the water table.

RESULTS FROM FIELDS

Table 6.1. Summary of 1992 field practices.

Field	Crop	Yield (Mg ha^{-1})	Nutrient Applications		Other Applications†	Irrigations	
			No.	Total NPK (kg ha^{-1})		No.	Total (cm)
1	sweet corn	20	7	214N, 12P, 109K	bromoxynil metolachlor(W) alachlor(E) atrazine(W)	14	16.6
2	sweet corn	25	7	250N, 12P, 109K	atrazine metolachlor(W) alachlor(E)	13	15
3	sweet corn	10	6	184N, 28P, 165K	alachlor(W) atrazine(E) alachlor(E)	?	13
4	potato	45.4 (NE½) 46.5 (SW½)	12	279N, 74P, 118K	carbofuran (SW) metribuzin permethrin (NE) mancozeb triphenyltin hydroxide chlorothalonil diquat	?	26

† Field-half of application is indicated where application was not to entire field.

Table 6.2. Summary of 1993 field practices.

Field	Crop	Yield (Mg ha^{-1})	Nutrient Applications		Other Applications †	Irrigations	
			No.	Total NPK (kg ha^{-1})		No.	Total (cm)
1	potato	33.6	5	333N, 54P, 311K	metolachlor metribuzin chlorothalonil carbofuran methamidophos diquat sulfur esfenvalerate	12	11.8
2	potato	46.4	6	north 297N, 54P, 389K south 357N, 54P, 389K	metolachlor metribuzin carbofuran sulfur esfenvalerate chlorothalonil methamidophos diquat	12	12.8
3	sweet corn	8.0	9	189N, 29P, 65K	alachlor(W) atrazine(E) alachlor bromoxynil(W) alachlor(E)	?	?
4	sweet corn	9.2	8	northeast 188N, 29P, 65K southwest 173N, 25P, 65K	alachlor atrazine	?	?

† Field-half of application is indicated where application was not to entire field.

Table 6.4. Average results from upgradient MLPs of fields 1, 2 and 3 vs. average of all in-field MLPs. Measurements are in mg L^{-1}, unless noted.

Parameter[1]	Upgradient MLPs	In-field MLPs
pH	5.4	5.0
conductivity	7.0[1]	33 [1]
alkalinity	4.9	2.8
nitrate-N	1.2	21
chloride	2.0	21
iron	1.5	0.02
sodium	1.8	2.9
potassium	0.59	16
calcium	5.3	26
magnesium	1.3	11
aluminum	0.20	0.69
manganese	0.08	0.09

[1] mS m^{-1}

Table 6.6. Summary of pesticide detects for 256 ports sampled, fall 1992, spring 1993, and late fall – early winter 1993.

Pesticide[1]	ES[2]	PAL[3]	Detect. lim.	Detects	Maximum[4]	Mean[4]	Median[4]
Alachlor	2	0.2	0.1	13	1	0.45	0.4
Metolachlor	15	1.5	0.4[5]	110	157	6.67	0.7
Metribuzin	250	50	0.2[5]	169	24.6	1.48	0.6
Atrazine	(atrazine and its		0.2[5]	134	2.5	0.44	0.3
de-ethyl	metabolites are		0.1	195	3.6	0.51	0.4
de-isopropyl	summed)		0.2[5]	12	0.4	0.27	0.2
Total	3	0.3			6.4	0.74	0.5
Carbofuran	50	10	0.1[5]	89	20.6	1.98	0.9

[1] All measurements µg L^{-1}.
[2] DNR Enforcement Standard.
[3] DNR Preventative Action Limit.
[4] Of MLPs with detects.
[5] Detection limits for method 507, used in the first sampling round, were metolachlor 0.6, metribuzin 0.1, atrazine 0.1; alachlor, de-isopropyl atrazine and carbofuran were not detectable.

Table 6.8. Fertilizer N inputs to vegetable fields, 1992 and 1993.

Field	Year	Fertilizer N (kg ha^{-1})
1	1992	214
	1993	333
2	1992	250
	1993	297 N; 357 S*
3	1992	184
	1993	189
4	1992	279
	1993	188 N; 173 S*

*N, north field-half; S, south field-half

Table 6.10. Inputs, outputs, and potentially leachable N from four vegetable fields, 1992 and 1993.

Field	Year	Crop[1]	N_{input} (kg ha^{-1}) [2]	N_{output} (kg ha^{-1}) [3]	N_{pl} (kg ha^{-1})	N_{pl} as % of N_{input}
1	1992	C	259	94	165	64
	1993	P	387	150	237	61
2	1992	C	295	117	178	60
	1993	P	351 N; 411 S*	187	164 N; 224 S*	47 N; 55 S*
3	1992	C	229	51	178	78
	1993	C	234	42	192	82
4	1992	P	333	190 NE; 200 SW*	143 NE; 133 SW*	43 NE; 40 SW*
	1993	C	233 NE; 218 SW*	47	186 NE; 171 SW*	80 NE; 78 SW*

[1] C = sweet corn; P = potato
[2] fertilizer + precipitation + dry deposition + crop seed + mineralization
[3] crop harvest + gaseous losses
* field half: N north, S south, NE northeast, SW southwest

Table 6.31. N loading to groundwater as estimated by three methods for Field 2. The N budget predicts potentially leachable N from total inputs, and the leaching model only considers fertilizer N inputs. (N) and (S) refer to field-halves.

	Crop and year			
	Sweet Corn, 1992		Potato, 1993	
Method	Percent of N input	kg N ha^{-1}	Percent of N input	kg N ha^{-1}
N budget (potentially leachable)	60	178	47 (N) 55 (S)	164 (N) 224 (S)
NO_3 loading through a water year	60	178	57 (N) 71 (S)	198 (N) 291 (S)
Leaching model (percent of fertilizer N)[1]	56	140	65 (N) 61 (S)	193 (N) 216 (S)

[1] Includes crop residue N

Table 5.4. Summary of pesticide detects sampled at multi-level piezometers located under study fields (Chapter 6).

			Range of detects			
Analyte6	No. wells with detects	No. ports sampled	Min.	Max.	Mean	Median
Alachlor	13	256	-0.10	1.10	0.50	0.40
Atrazine	134	256	0.10	2.50	0.40	0.30
-de-ethyl	13	256	0.10	3.60	0.50	0.40
-de-isopropyl	12	256	0.20	0.40	0.30	0.20
Carbofuran	2	26	0.10	20.60	2.00	0.90
Metolachlor	110	256	0.60	157.80	6.70	0.70
Metribuzin	169	256	0.10	24.60	1.50	0.60

6 All concentrations $\mu g\ L^{-1}$.

Table 5.3. Summary of organic monitoring data from PEGPW monitoring network.

Analyte3	No. wells with detects	No. wells sampled	Range of Detects				
			Min.	Max.	Mean	Median	Std. dev.
Alachlor	1	26	-0.10	0.53	-	-	-
Atrazine	14	34	-0.10	0.25	0.14	0.13	0.09
Carbofuran	2	26	-0.10	0.01	0.01	0.01	0.00
De-ethylAtrazine	6	26	-0.10	0.35	0.14	0.11	0.12
Metolachlor	3	36	-0.60	2.53	1.16	0.87	1.24
Metribuzin	10	36	-0.10	1.98	0.46	0.20	0.58

3 All concentration values $\mu g\ L^{-1}$.

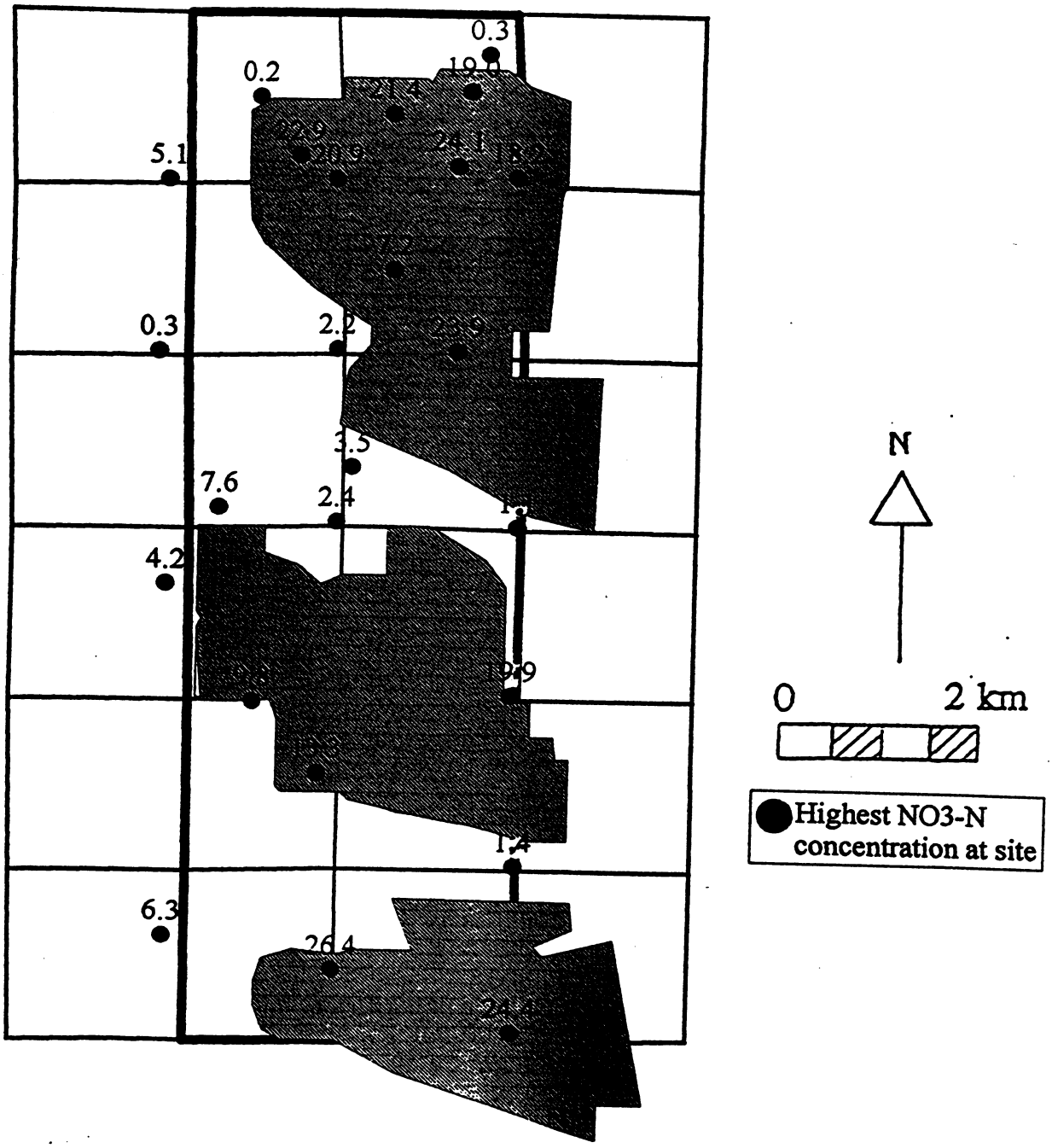


Figure 5.3 Greatest NO_3-N concentrations at monitoring well locations with respect to plumes originating from irrigated fields.

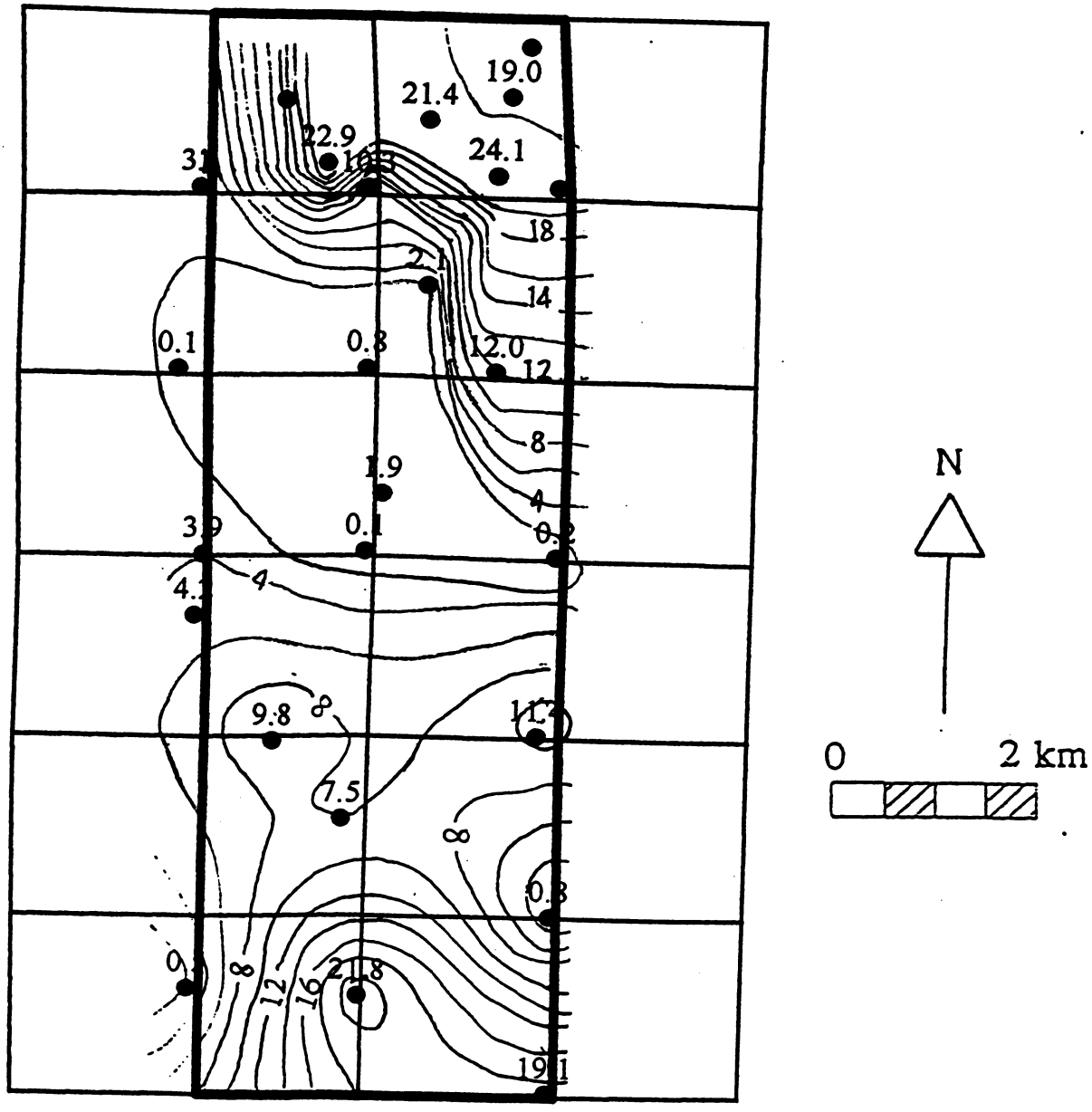


Figure 5.4A NO_3-N concentrations at depths less than 3 m below the water table.

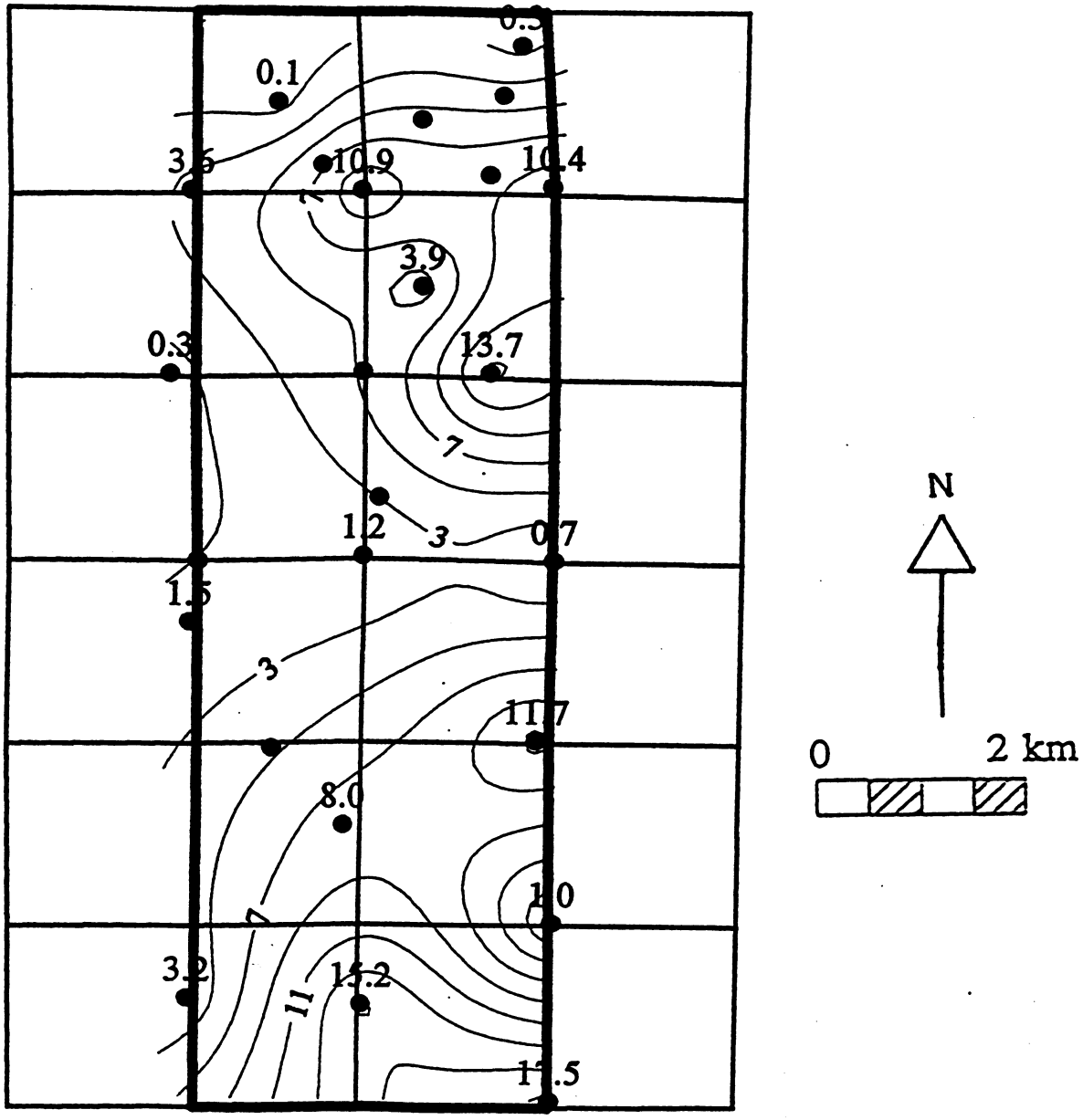


Figure 5.4B NO_3-N concentrations at depths between 3 and 10 m below the water table.

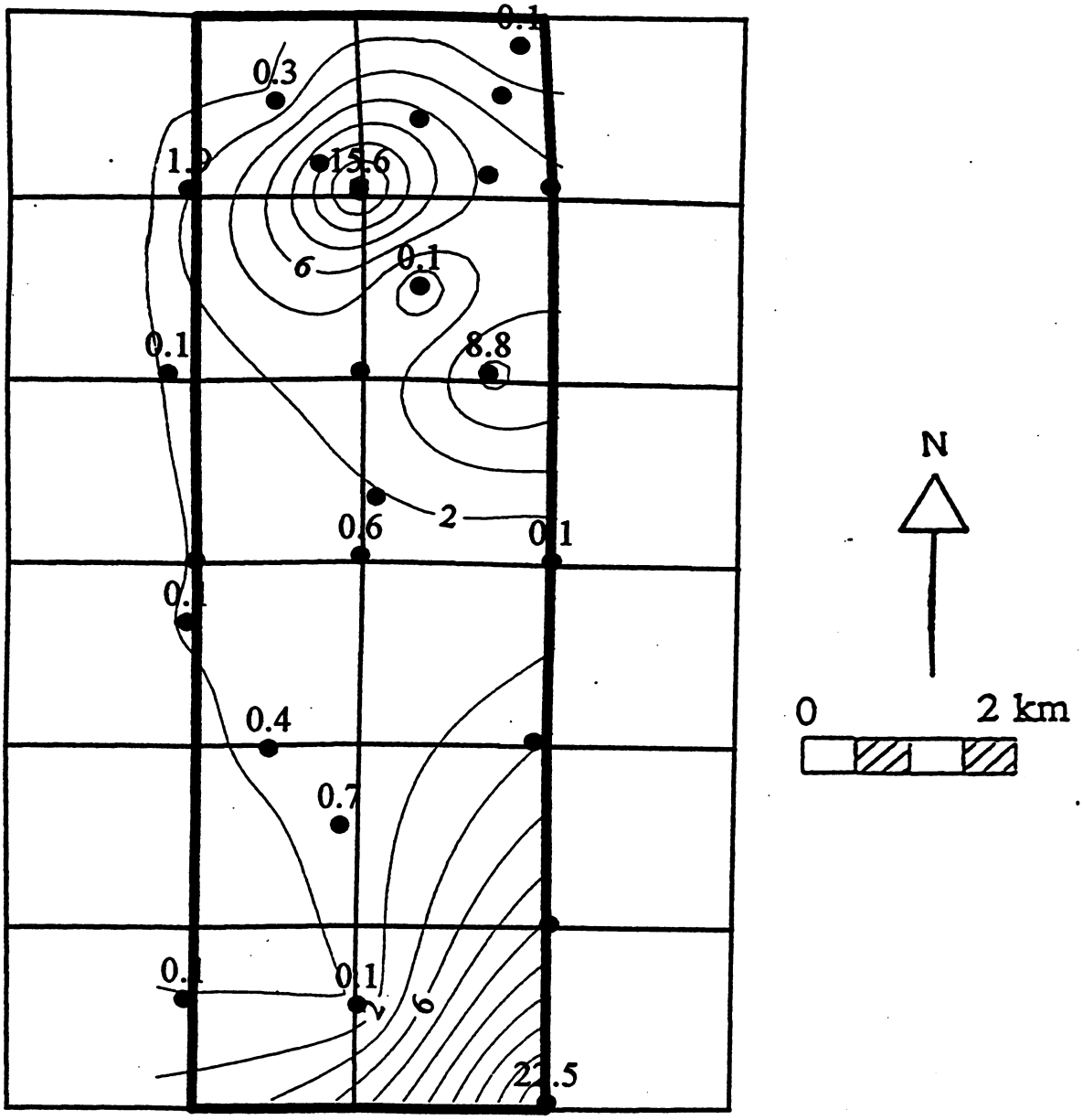


Figure 5.4C NO_3-N concentrations at depths greater than 10 m below the water table.

Denitrification did not seem to be a dominant mechanism controlling the fate of NO_3, but probably occurred at sites 0801 and 3601 (see Discussion). High NO_3-N concentrations sometimes occurred at depth, and were observed at sites 0603, 0802, and 3203. The entire saturated thickness at these sites was high in NO_3. This situation apparently resulted where groundwater flow paths traversed several irrigated vegetable fields without reducing conditions to facilitate denitrification.

Organic analyses

Summary of areawide data

Forty-two monitoring wells at 13 sites were analyzed for organic parameters (Figure 5.5, Table 5.3). Due to high analytical cost, only wells with a reasonable possibility of containing a pesticide were chosen for sampling. The first round of pesticide sampling was the broadest. Wells in succeeding rounds were selected on the basis of detects from earlier rounds. Pesticides were detected in wells at six sites located immediately downgradient of irrigated vegetable fields: 0603, 0802, 2901, 3001, 3201, and 3203. The pesticides detected were metolachlor, metribuzin, alachlor, carbofuran and atrazine with its metabolite de-ethyl atrazine. Alachlor, de-ethyl atrazine, and metolachlor were each detected once above their respective Wisconsin preventative action limits (PAL). None exceeded the enforcement standard (ES).

Table 5.3. Summary of organic monitoring data from PEGPW monitoring network.

			Range of Detects				
Analyte3	No. wells with detects	No. wells sampled	Min.	Max.	Mean	Median	Std. dev.
Alachlor	1	26	-0.10	0.53	-	-	-
Atrazine	14	34	-0.10	0.25	0.14	0.13	0.09
Carbofuran	2	26	-0.10	0.01	0.01	0.01	0.00
De-ethyl Atrazine	6	26	-0.10	0.35	0.14	0.11	0.12
Metolachlor	3	36	-0.60	2.53	1.16	0.87	1.24
Metribuzin	10	36	-0.10	1.98	0.46	0.20	0.58

3 All concentration values $\mu g\ L^{-1}$.

Summary of in-field data

Pesticide analyses were conducted at three multi-level piezometers in fall, 1992, from the intensively monitored fields (Chapter 6); at those three plus a fourth in spring, 1993; and, at those four plus fourteen others in late fall/early winter 1993. Pesticides were detected in all sampled multi-level piezometers (Table

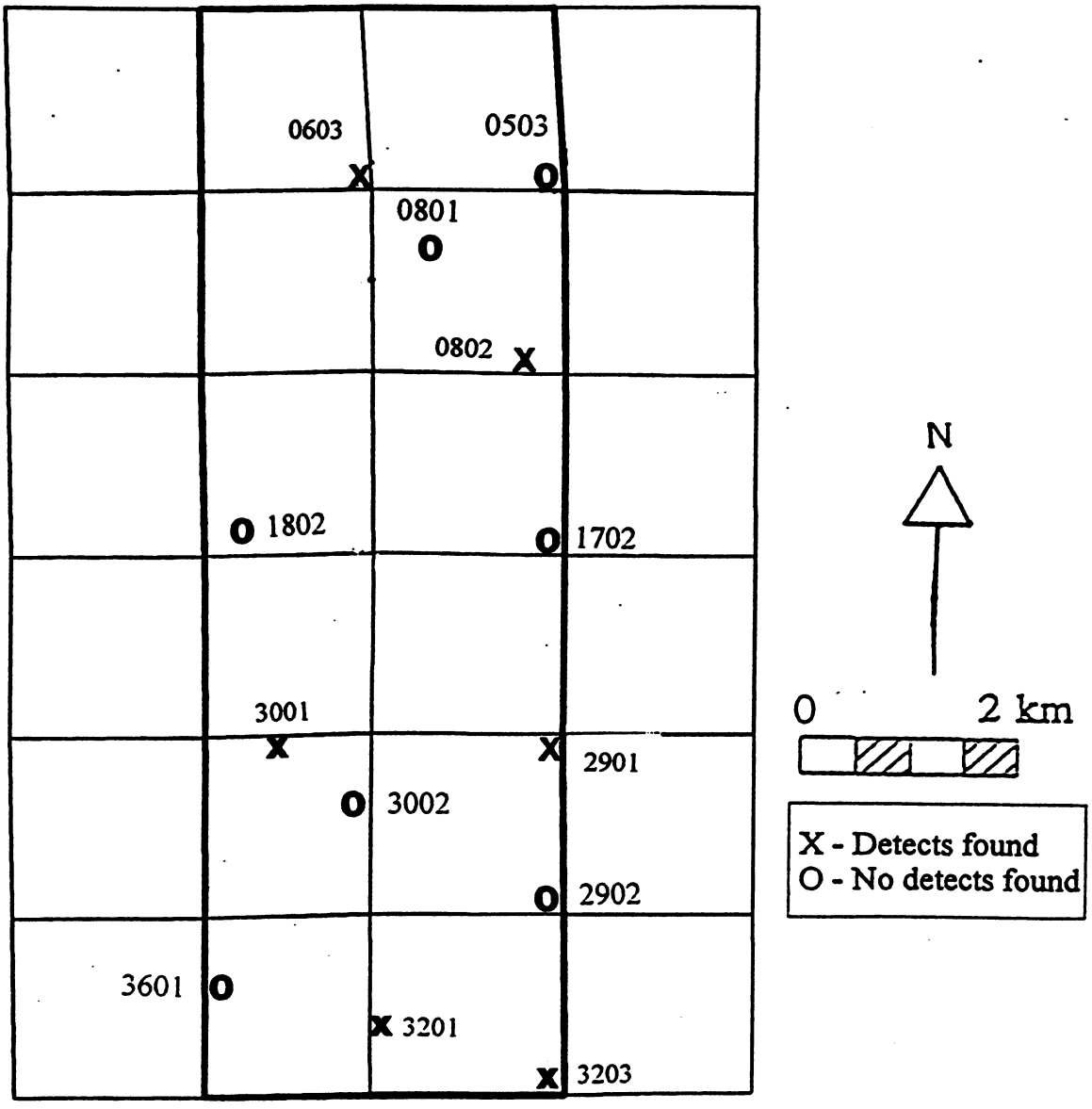


Figure 5.5 Sites where wells were sampled for pesticides.

5.4). Some detects were above the PAL, and a few exceeded the ES. Atrazine was the pesticide which exceeded the PAL the most often (at 15 of 18 MLPs tested), with metolachlor, carbofuran, and alachlor each exceeding their PAL at 8, 3, and 1 MLPs, respectively. Atrazine exceeded the ES at 3 MLPs; metolachlor, at one MLP.

DISCUSSION

Chemical characteristics of impacted and non-impacted groundwater

The differences in chemistries of agriculturally impacted versus non-impacted groundwater are illustrated by comparing wells with <2.0 mg L^{-1} NO_3-N ("non-impacted") to those with >10 mg L^{-1} NO_3-N ("impacted") (Figures 5.6A-B). Sites where denitrification was evident (0801 and 3601) were not included in this analysis. Agriculturally impacted groundwater within 3 m of the water table (Figure 5.6A) exhibited markedly higher concentrations of Ca, Cl, K, and Mg as well as elevated levels of Na and SO_4. The differences at depths >10 m below the water table (Figure 5.6B) exhibited a similar but muted pattern.

Table 5.4. Summary of pesticide detects sampled at multi-level piezometers located under study fields (Chapter 6).

			Range of detects			
Analyte6	No. wells with detects	No. ports sampled	Min.	Max.	Mean	Median
Alachlor	13	256	-0.10	1.10	0.50	0.40
Atrazine	134	256	0.10	2.50	0.40	0.30
-de-ethyl	13	256	0.10	3.60	0.50	0.40
-de-isopropyl	12	256	0.20	0.40	0.30	0.20
Carbofuran	2	26	0.10	20.60	2.00	0.90
Metolachlor	110	256	0.60	157.80	6.70	0.70
Metribuzin	169	256	0.10	24.60	1.50	0.60

6 All concentrations μg L^{-1}.

Water quality cross-sections

Six cross-sections were selected to examine groundwater quality trends and evolution (Figures 5.7A and B). Five were along groundwater flow paths delineated by PATH3D (Chapter 4) (Figure 5.8A-E), and one was roughly perpendicular to groundwater flow along the eastern edge of the watershed (cross-section A-A', Figure 5.8F). All flowpath cross-sections traversed irrigated fields except for cross-section 3 (Figure 5.8C).

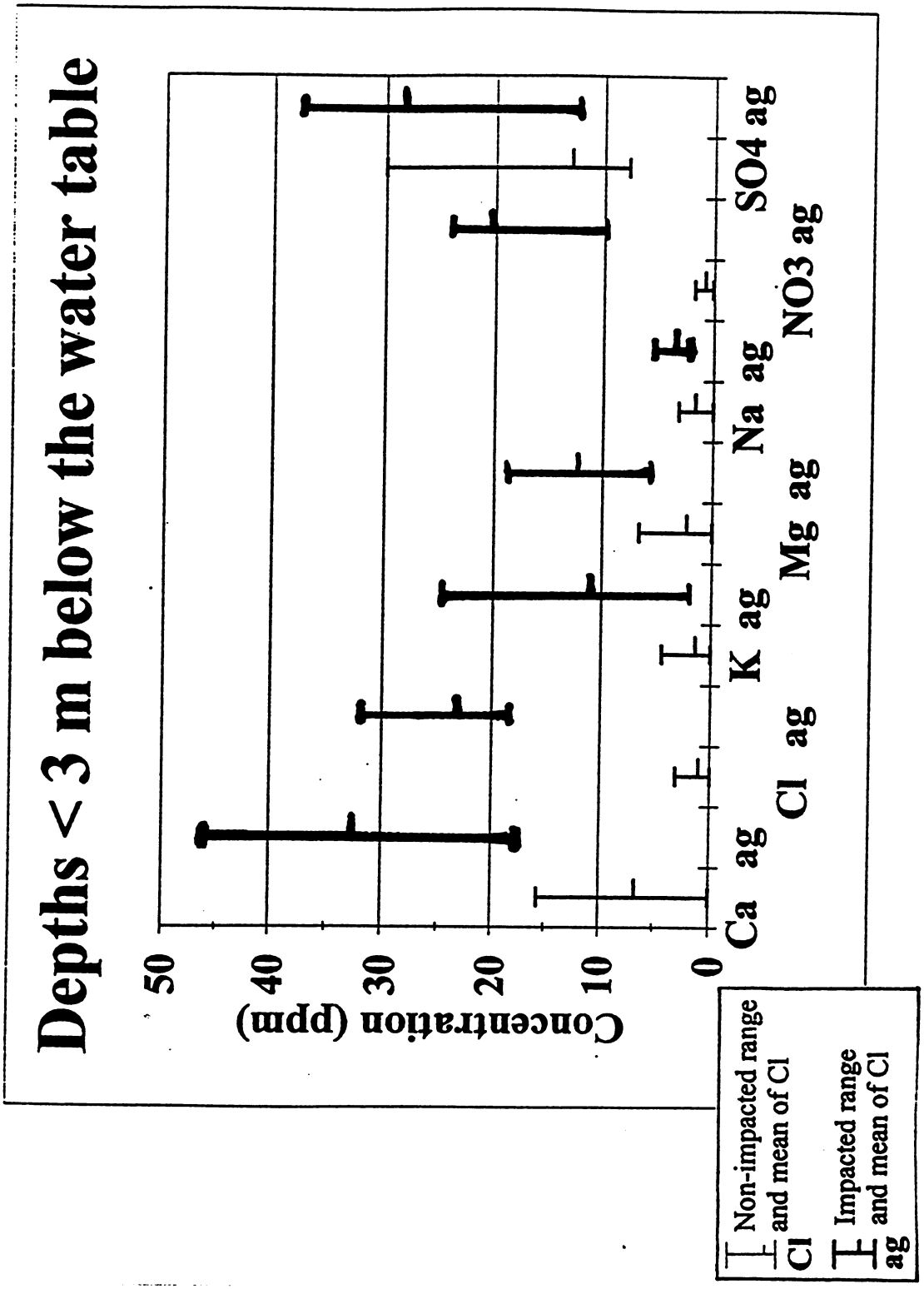


Figure 5.6A Analyte ranges and means from agriculturally impacted and non-impacted groundwater at depths less than 3 m below the water table.

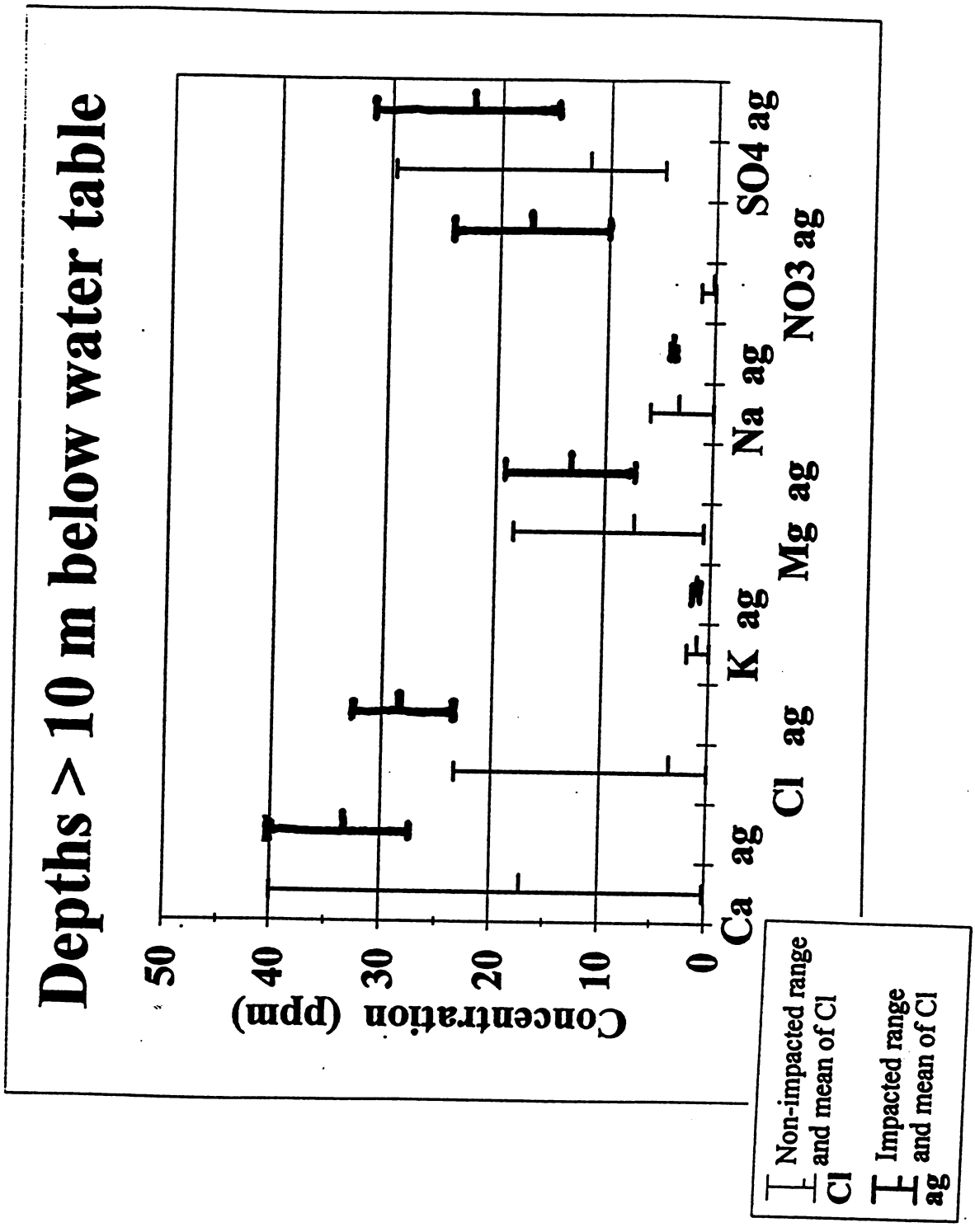


Figure 5.6B Analyte ranges and means from agriculturally impacted and non-impacted groundwater at depths greater than 10 m below the water table.

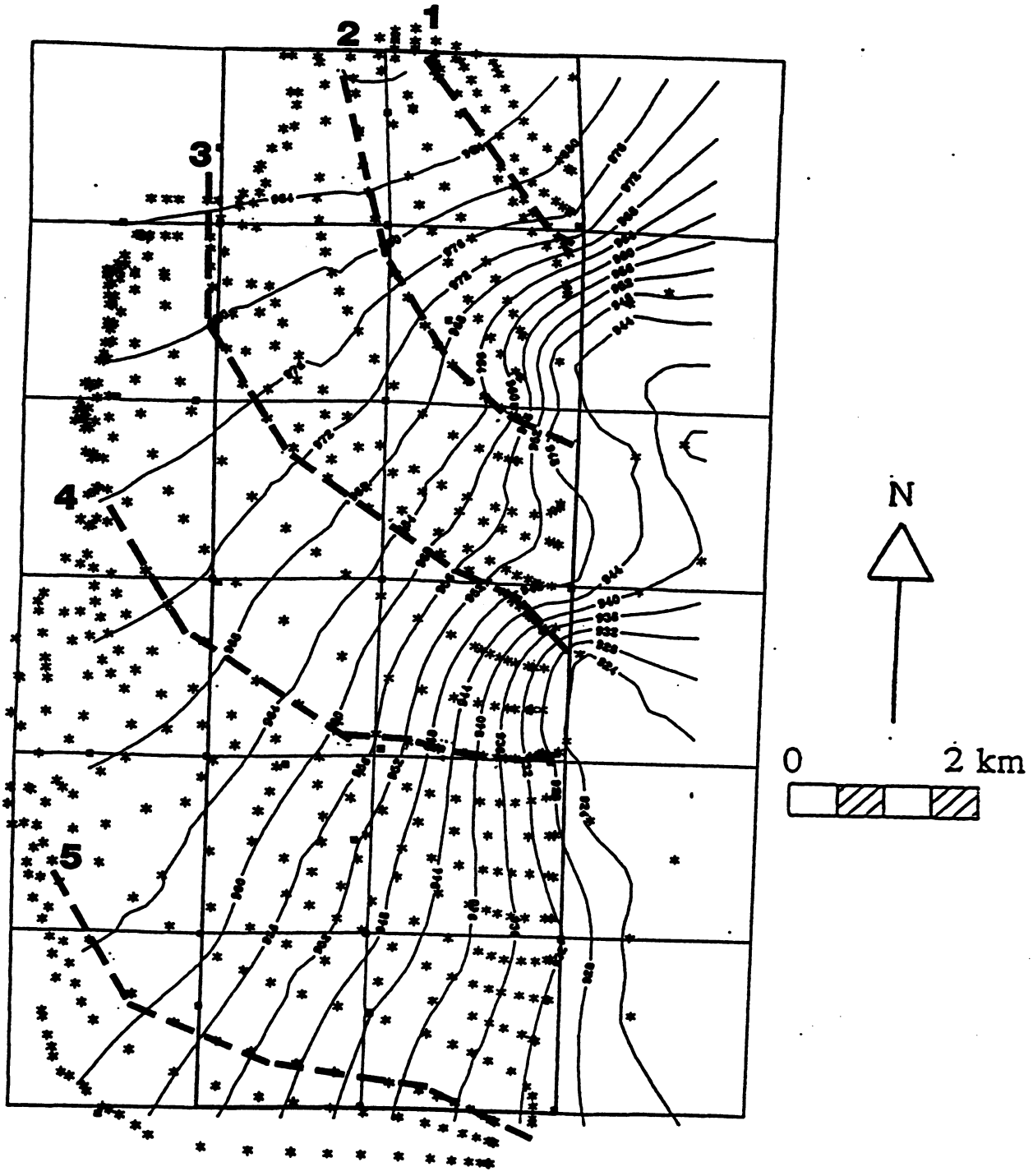


Figure 5.7A Locations of groundwater flow path cross-sections 1-5.

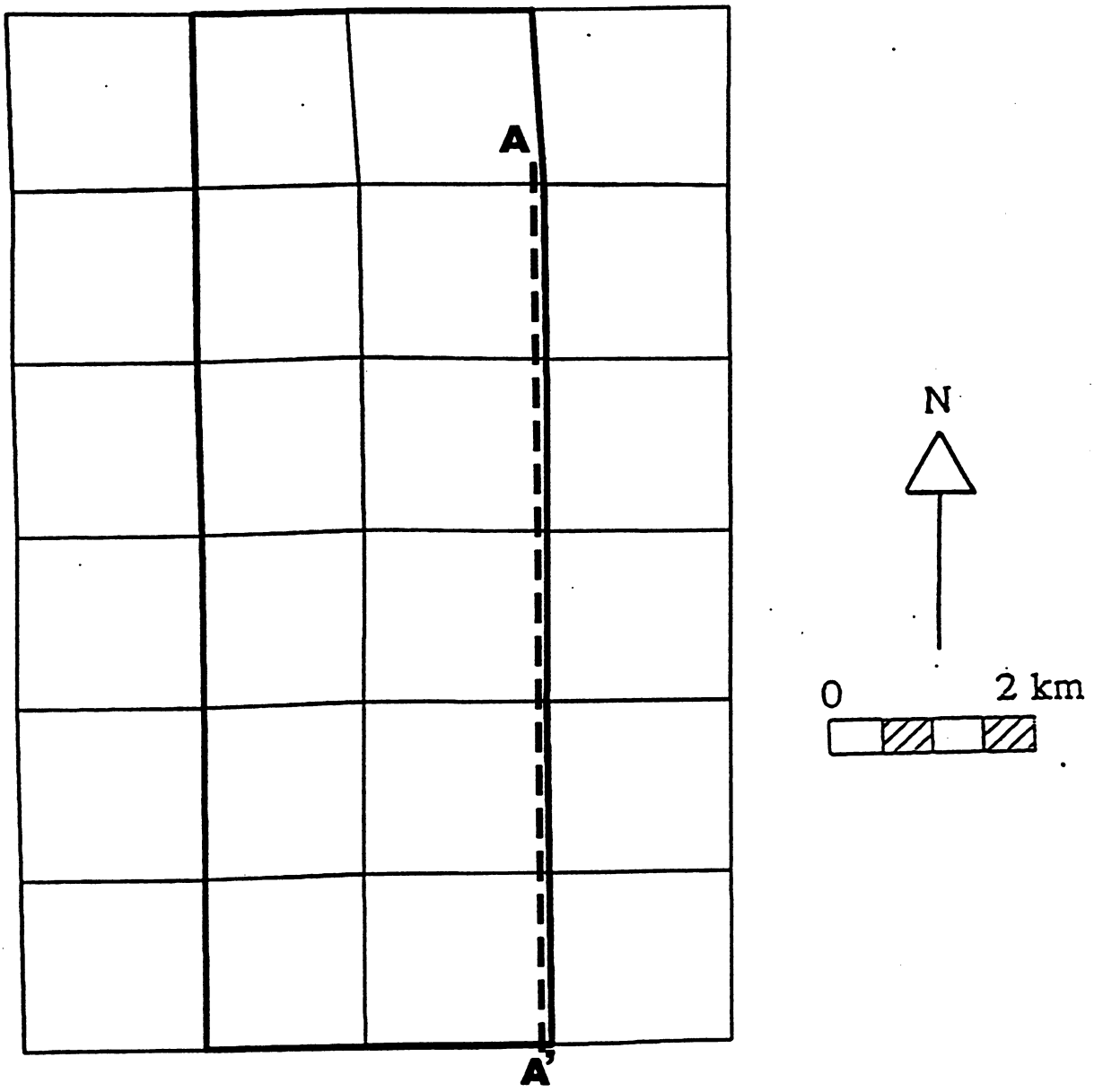


Figure 5.7B Locations of groundwater flow path cross-sections.

The upgradient-most site along each flow path, with the exception of site 3601, reflected groundwater not impacted by agricultural land uses. These demonstrated slightly elevated concentrations of NO_3-N (typically <0.2 to 6 mg L^{-1}) and Cl (<1 to 10 mg L^{-1}) at shallow depths, and were probably affected by septic systems and road salt. Nitrate and Cl tended to diminish at depth except at site 3601, where Cl at 18 mg L^{-1} occurred 17.7 m below the water table (well EU599).

Nitrate-N and Cl increased dramatically where groundwater flow paths traversed irrigated fields. The thicknesses of plumes from individual fields ranged from >3 m to 12 m (site 0603, cross-section 2; Figure 5.8B). Where plumes from two or more fields overlapped, the plume thickness was greater, (e.g., site 3203, cross-section 5; Figure 5.8E).

Denitrification was evident at site 0801 (cross-section 2, Figure 5.7B). This site is downgradient of irrigated vegetable land use. Well EU664 contained elevated Ca, Cl, and Mg, characteristic of groundwater originating under agriculture. However, little NO_3 was detected. The reducing conditions required for denitrification are indicated by high Fe (7.03 mg L^{-1}) and Mn (0.9 mg L^{-1}) concentrations. (Dissolved oxygen was indicated by field analyses as >1 mg L^{-1}, but we believe those were in error.) Denitrification may also have occurred at site 3601. High Cl (18 mg L^{-1}), Ca (40 mg L^{-1}), and Mg (15 mg L^{-1}) concentrations were observed there, but again, no NO_3. Analyses indicated little dissolved oxygen (<1 mg L^{-1}) and elevated Fe (0.55 mg L^{-1}) and Mn (0.16 mg L^{-1}) concentrations.

Although evidence of denitrification was apparent at sites 0801 and 3601, the phenomenon was not extensive. Since NO_3 was always detected in shallow groundwater near irrigated fields, denitrification must have occurred in the aquifer rather than in soils. The erratic occurrence of denitrification may be due to a patchy distribution of organic-rich aquifer sediment that promotes denitrifying conditions.

Cross-section 3 (Figure 5.8C) exemplifies an agriculturally non-impacted groundwater flow path. With the exception of the upgradient-most monitoring site, located near a residential area and a state highway, the NO_3-N and Cl concentrations remain very low, sometimes below detection limits.

Cross-section 6 (Figure 5.8F) is roughly perpendicular to groundwater flow along the eastern boundary of the watershed. High NO_3 and Cl concentrations were only found where the cross-section intersected plumes originating from irrigated vegetable fields (Figure 5.3). The vertical zone containing high NO_3 and Cl concentrations was thickest, up to 15 m thick, where flow paths traversed several fields. In places, this zone occupied the aquifer's entire saturated thickness. For example, at site 3203 (Figure 5.8E) NO_3-N and Cl concentrations remained above 19 and 29 mg L^{-1}, respectively, at all but one well throughout the water column. Figure 5.8F also contains two locations (sites 1702 and 2902) where NO_3 and Cl concentrations were low. Both of these points were in small gaps between plumes originating from fields.

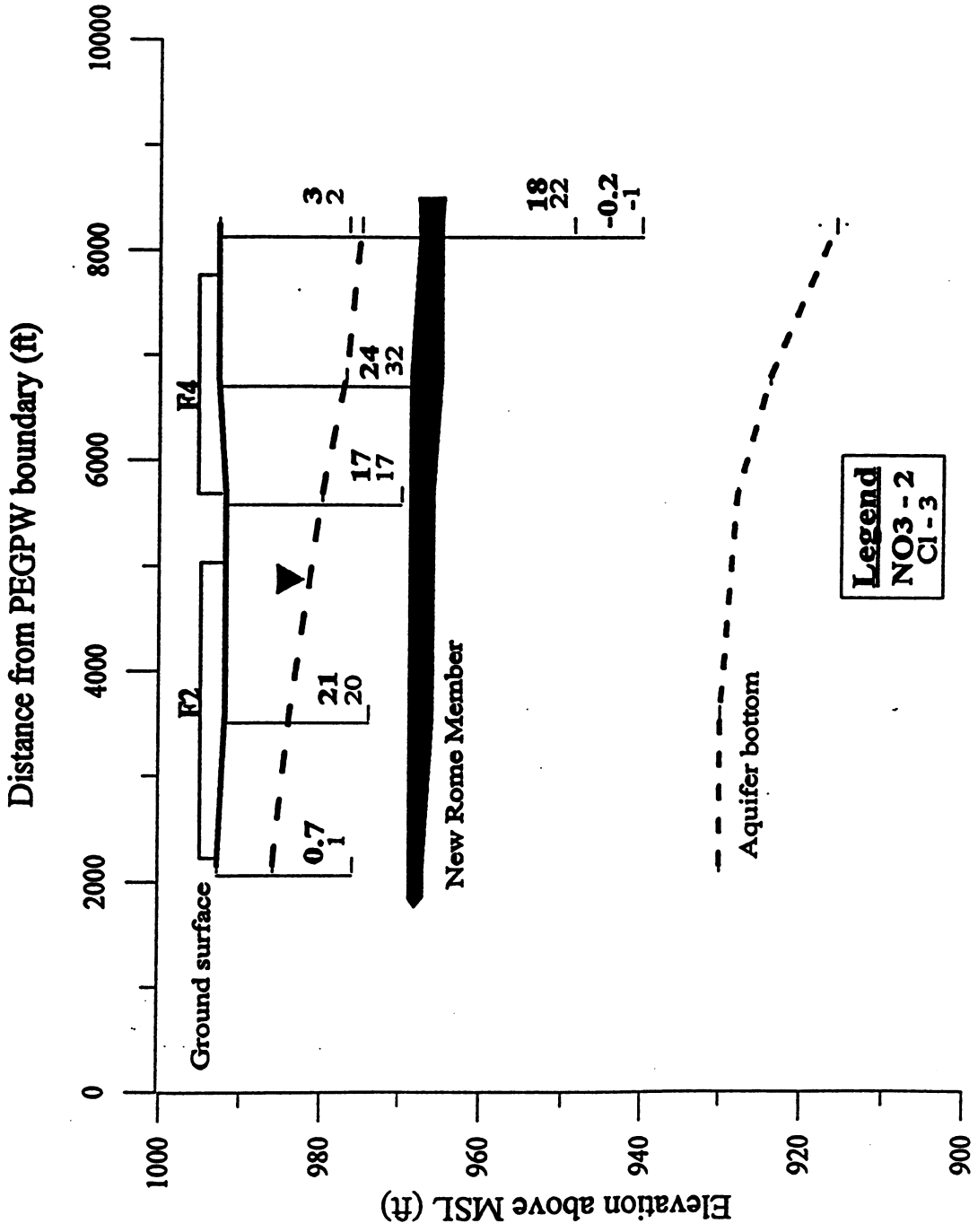


Figure 5.8A Flow path cross-section 1.

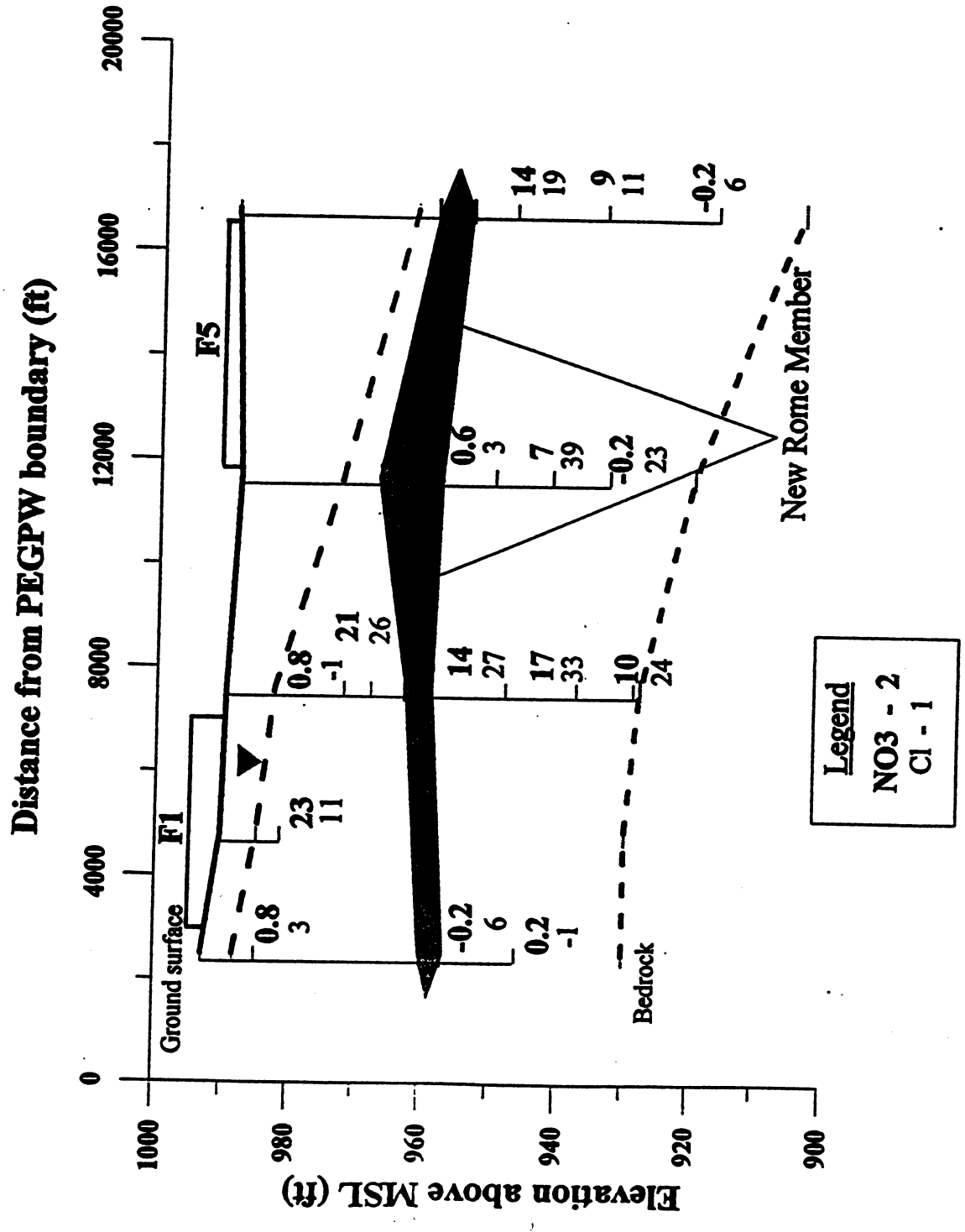


Figure 5.8B Flow path cross-section 2.

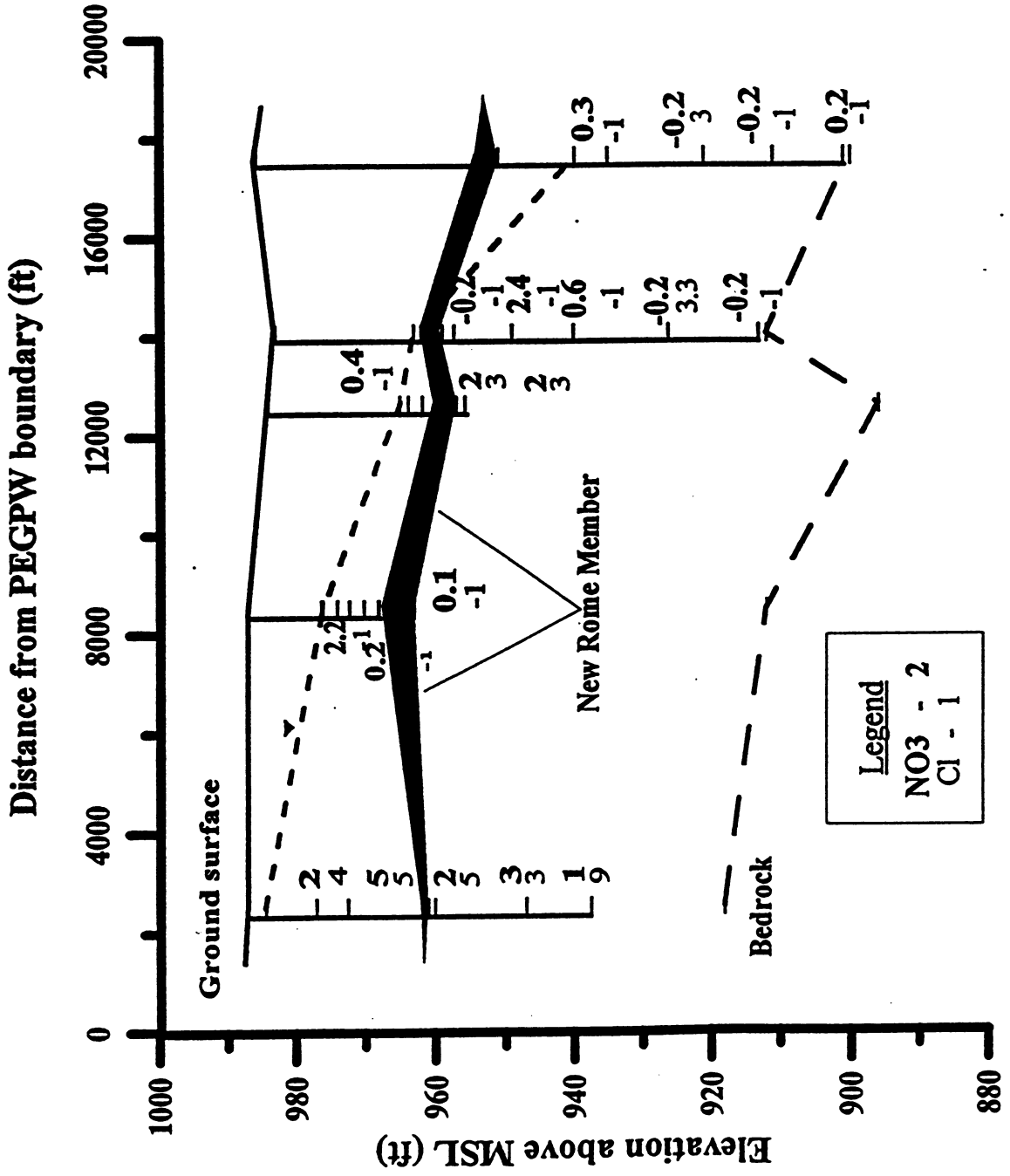


Figure 5.8C Flow path cross-section 3.

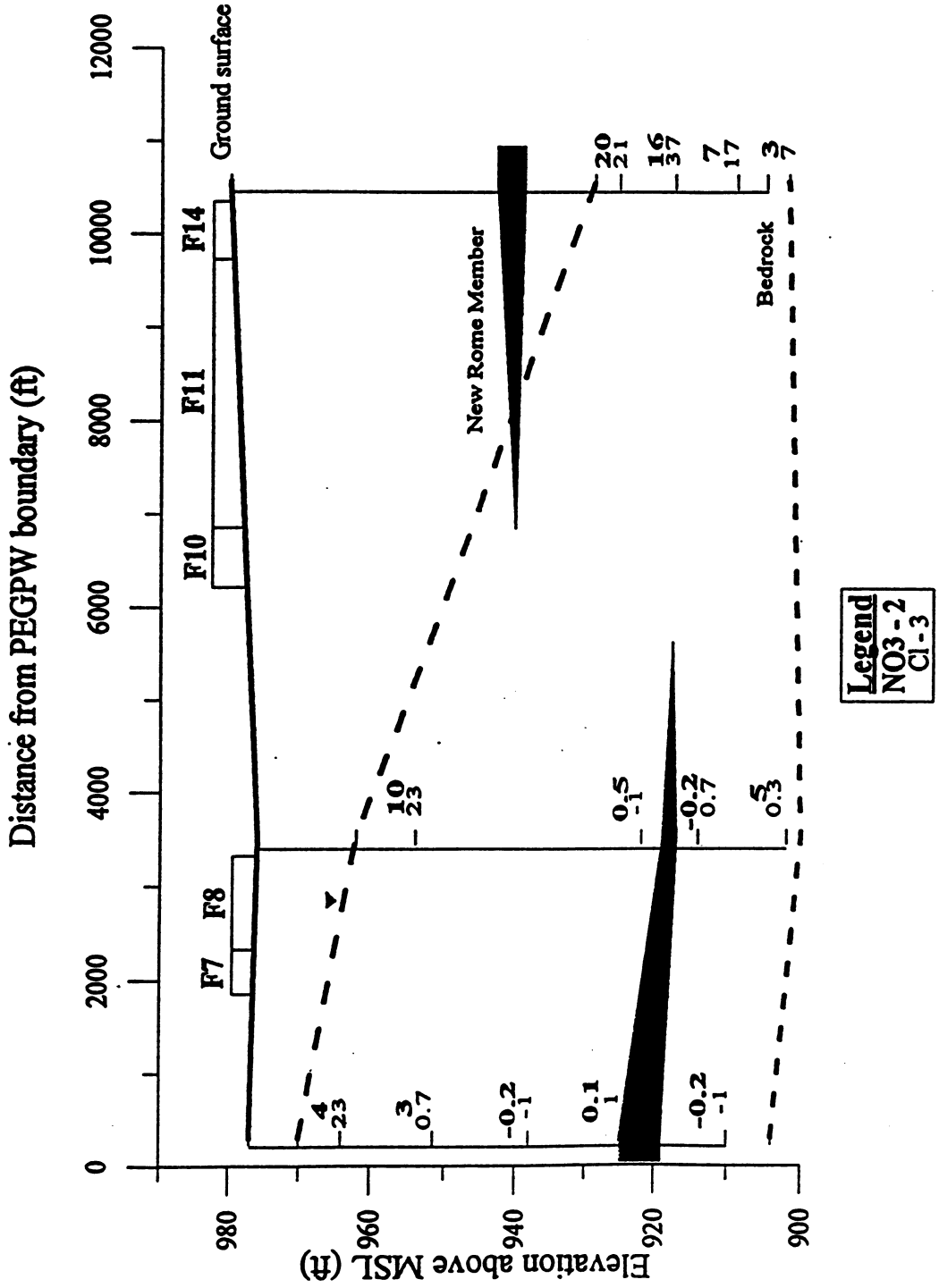


Figure 5.8D Flow path cross-section 4.

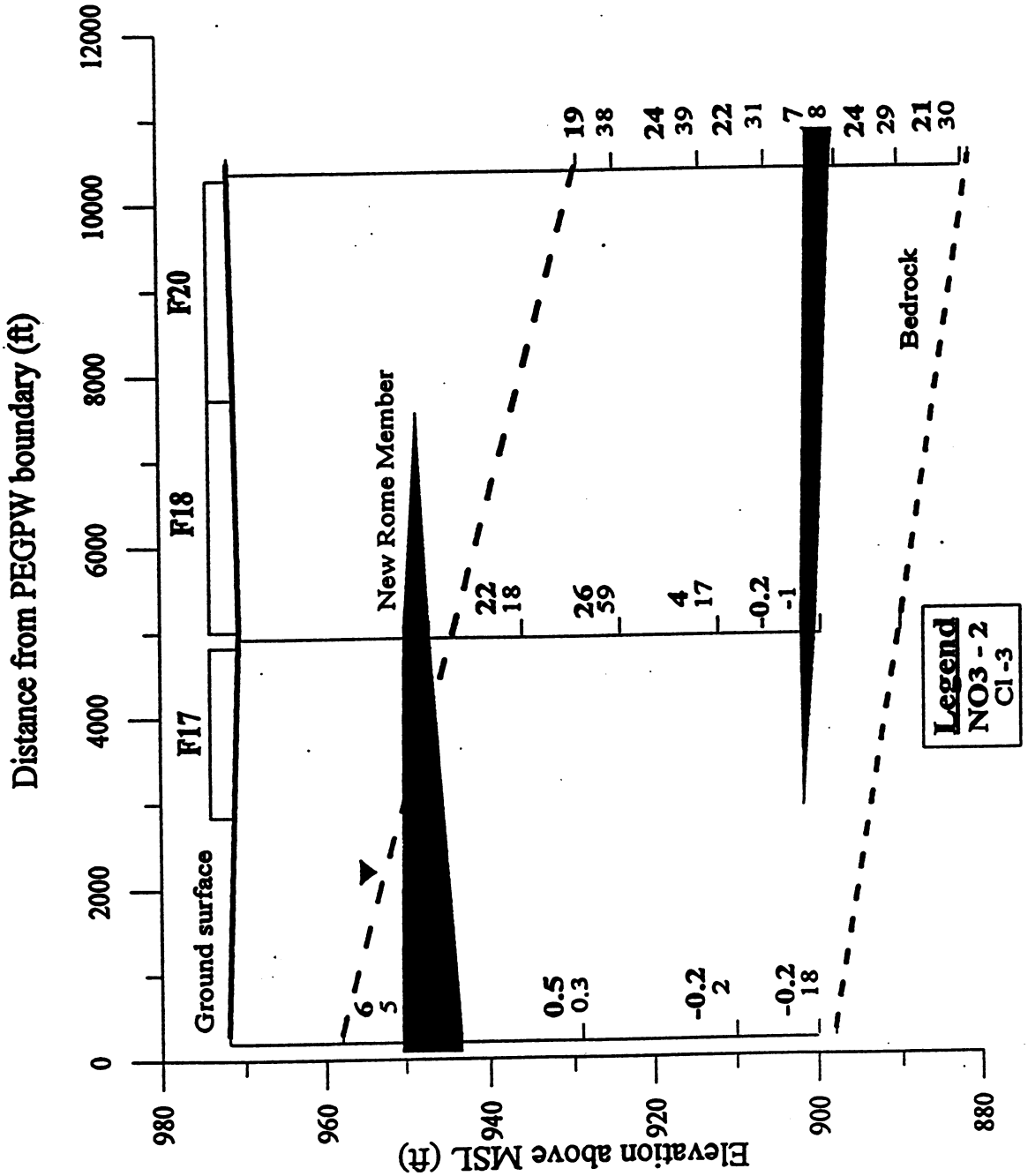


Figure 5.8E Flow path cross-section 5.

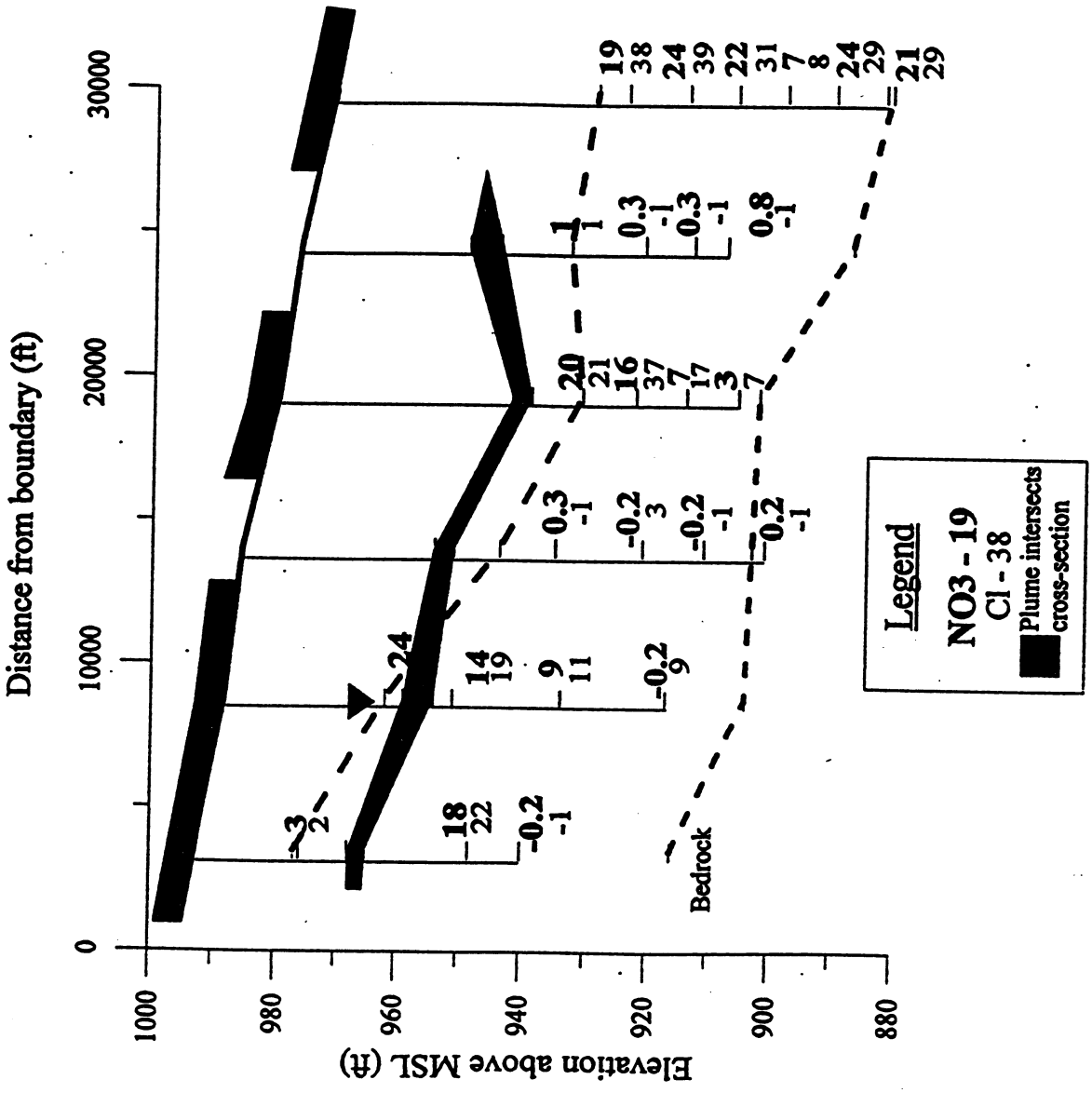


Figure 5.8F Flow path cross-section 6.

CONCLUSIONS

The groundwater chemistry of the Port Edwards Groundwater Priority Watershed was the Ca-Mg-HCO_3 type, with Cl, Na, SO_4, and NO_3 as other major ions. The pH ranged from 4.89 to 8.70, and alkalinity averaged 26 mg L^{-1} $CaCO_3$. Both pH and alkalinity tended to increase with depth, from north to south, and from west to east. Reducing conditions were frequently encountered at depth, and also near the water table in the northwest part of the watershed, an area with shallow groundwater and poorly drained soils. In general, the watershed had groundwater similar to that of the Wisconsin Central Sand Plain as a whole, but lower concentrations of dissolved carbonates.

Nitrate-N concentrations ranged from <0.2 to 26.4 mg L^{-1}, except directly under fields (Chapter 6). Nitrate-N concentrations of >8 mg L^{-1} were always associated with plumes emanating from irrigated vegetable fields. Chloride, another ion associated with agriculture, ranged from <1.0 to 58.7 mg L^{-1}. Nitrate-N frequently, but not always, decreased with depth below the water table. In two instances, low NO_3 concentration at depth was apparently due to denitrification. However, denitrification did not seem to be prevalent.

Flow and particle tracking models (Chapter 4) indicated plumes from irrigated fields underlie 54% of the watershed. Monitoring data indicated that plumes originating under a single field had vertical thicknesses ranging from >3 m to 12 m. Plumes traversing several fields sometimes polluted the entire saturated thickness of the aquifer with NO_3-N concentrations greater than 10 mg L^{-1}.

Chemistries of agriculturally impacted and non-impacted groundwaters were different. Besides NO_3-N, agriculturally impacted groundwater exhibited markedly higher concentrations of Ca, Cl, K, and Mg as well as elevated levels of Na and SO_4.

Pesticides were detected in wells at six monitoring locations outside but near irrigated vegetable fields. The pesticides detected were metolachlor, metribuzin, alachlor, carbofuran and atrazine with its metabolite de-ethyl atrazine. Alachlor, de-ethyl atrazine, and metolachlor each was detected once above its Wisconsin PAL. None exceeded its ES. The same pesticides plus alachlor were detected under fields (Chapter 6). Some concentrations were above the PAL, and a few exceeded the ES. Atrazine exceeded the PAL the most often (at 15 of 18 MLPs tested) and exceeded the ES as well.

We are fairly confident that our off-field monitoring well locations sampled some of the worst cases with respect to pesticides. The lack of enforcement standard exceedences indicated that recent farming practices generally did not pollute groundwater above the pesticide ESs very far from fields. However, we were not able to test for all pesticide residues. For instance, our methods are unable to detect alachlor ESA, found widely in groundwater. Since alachlor was detected in groundwater, it is probable that ESA was also present.

CHAPTER 6

AGRICULTURAL PRACTICES AND GROUNDWATER CONDITIONS UNDER FOUR IRRIGATED VEGETABLE FIELDS

INTRODUCTION

Purpose and scope

Four vegetable fields located in the north of the Priority Watershed (Figure 6.1) were studied with the goals of

- evaluating loading rates of NO_3 to groundwater;
- determining the effectiveness of "best management practices" (BMPs) in protecting or maintaining acceptable groundwater quality;
- comparing groundwater quality under fields to that of "background" or "natural" conditions;
- determining which, if any, pesticides leach under conventional practices and BMPs;

Three study fields are 42-54 ha and the fourth field is 12 ha (Figure 6.2). The geology underlying the fields consists of 18-27 m of sand interspersed with the New Rome silty/clayey sub-unit (Chapter 2). The New Rome layer is about 1 m thick and occurs about 6-8 m beneath the four fields. Soils are the Plainfield sand and Friendship loamy sand typical of the Priority Watershed and much of the WCSP. Groundwater flow is generally from the northwest to southeast (Figure 6.2). Depth to groundwater follows the same trend, occurring at 1.5 m in the northwest and 5 m in the southeast. Saturated hydraulic conductivity is estimated at 9×10^{-4} m s^{-1}. Lateral groundwater velocity is about 0.3 m d^{-1}.

CROPPING HISTORY AND INPUTS

History of study fields before study period

The exact years these fields were brought into irrigated production is not known. Air photos in the Wood County soil survey (Bartelme, 1977) show none were irrigated in 1974, and ASCS photos show that Fields 1-3 were irrigated by 1978 (Jerry Storke, pers. commun. 1995). Field 4 is irrigated from the same well as Field 3, so we suspect irrigation was added shortly after 1978. In 1974, Field 1 was forest, Field 2 was about 40% forest and 60% non-irrigated cropland, Field 3 was about 75% forest and 25% non-irrigated cropland, and Field 4 was non-irrigated cropland. The crop history of Fields 1 and 2 for some years prior to the project is 1991 sweet corn (*Zea mays* L.), 1990 potato (*Solanum tuberosum* L., var. Russet Burbank), 1989 sweet corn, 1988 sweet corn, and 1987 pea (*Pisum sativum* L.). Field 3's crop history is 1991 potato, 1990 soybean (*Glycine max* L.), 1989 pea, 1988 potato, and 1987 pea. For Field 4 the crop history is 1991 soybean, 1990 sweet corn, 1989 potato, 1988 sweet corn, and 1987 pea.

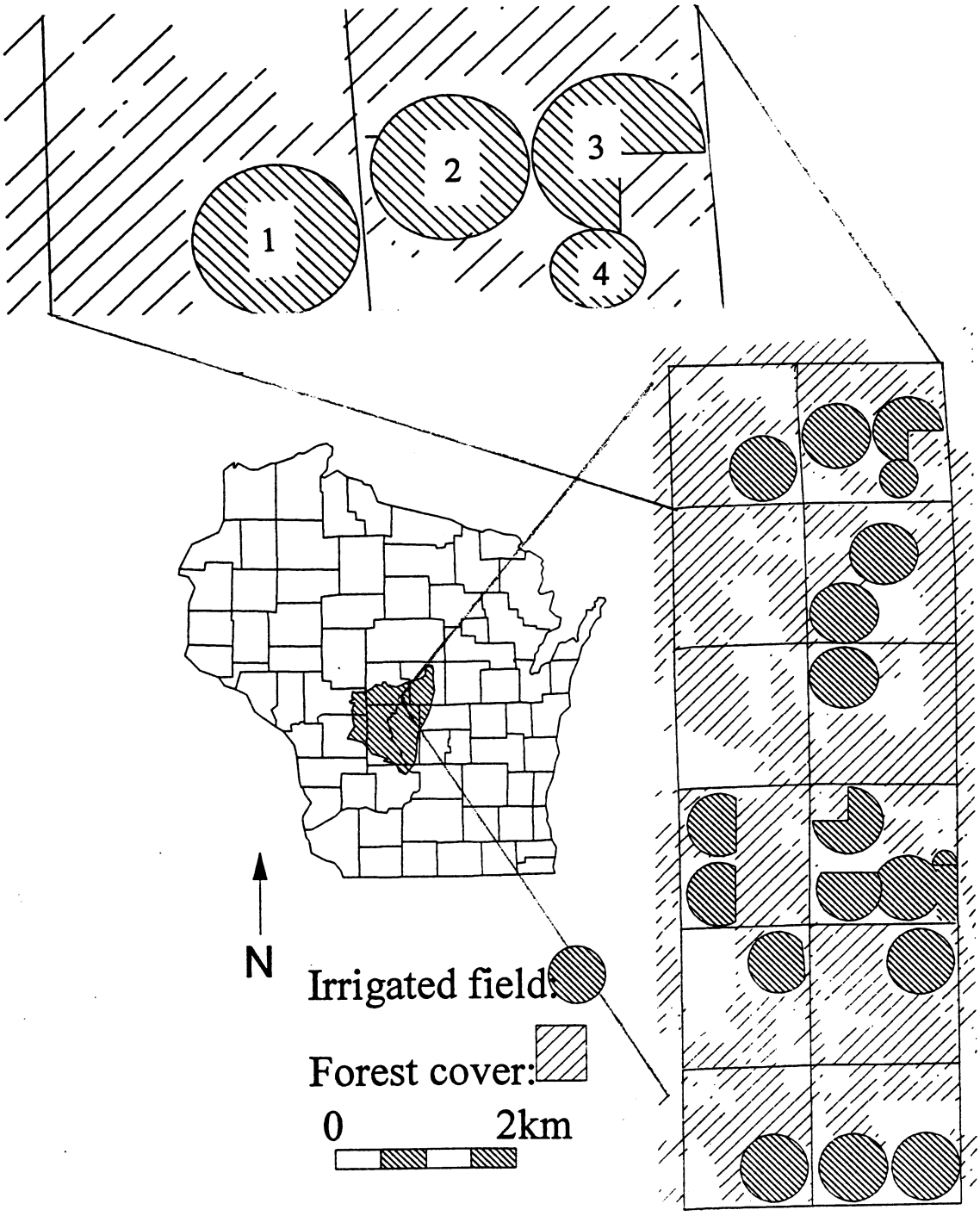


Figure 6.1 Location of the four intensive-study fields.

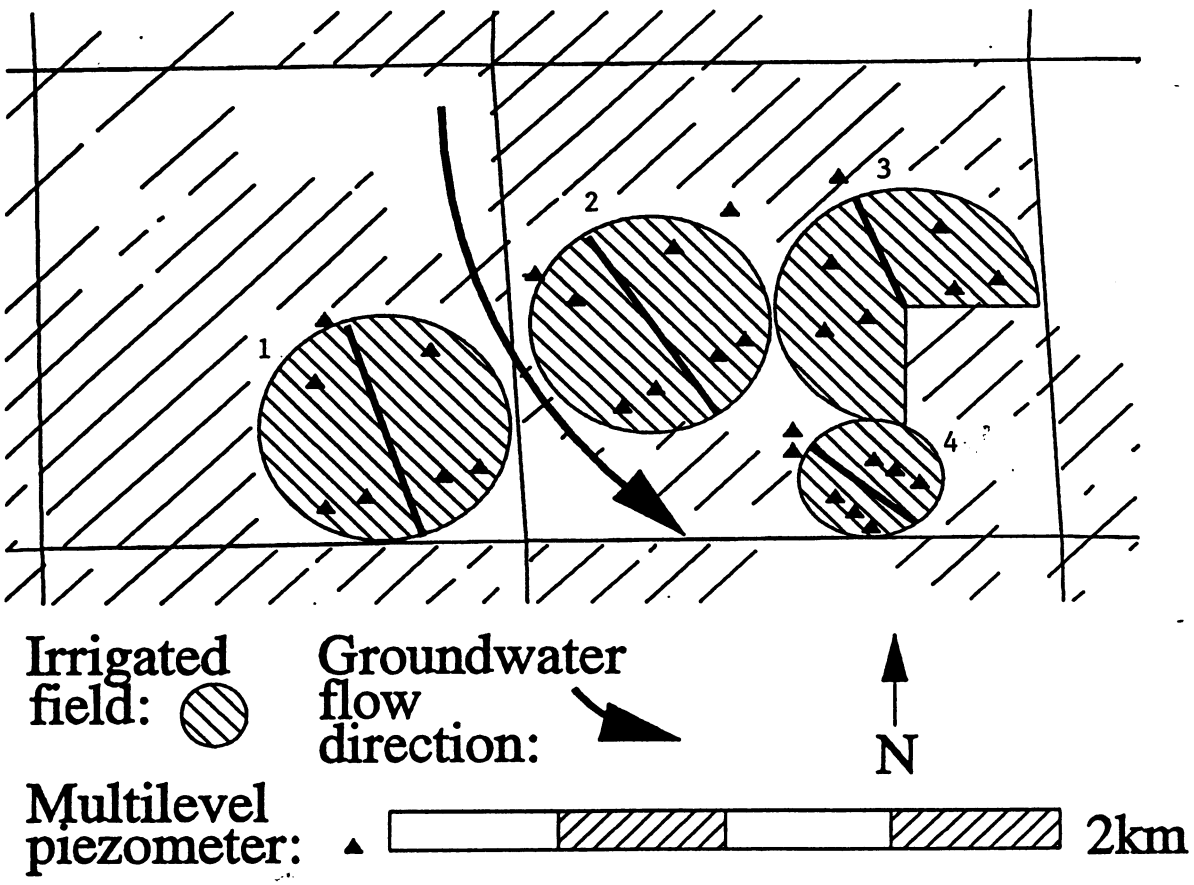


Figure 6.2 Intensive-study fields in Sections 5 and 6, showing multilevel piezometer sites and approximate groundwater flow direction.

Agricultural practices during study period (1992-1993)

During 1992, sweet corn was grown in Fields 1, 2 and 3, and potato in Field 4. Fertilizer amendments were split into 6-12 applications. These began with KCl applied 2 weeks to 2 months prior to a general NPK + micronutrient fertilizer. Additional fertilizer, mainly N, was applied at intervals throughout the remainder of the growing season. Fields 1, 2, and 3 received herbicide applications, while Field 4 received herbicide, insecticide, and fungicide applications. Sweet corn received an average of 15 cm of irrigation and potato received 26 cm. Table 6.1 summarizes 1992 field inputs.

Table 6.1. Summary of 1992 field practices.

			Nutrient Applications			Irrigations	
Field	Crop	Yield (Mg ha^{-1})	No.	Total NPK (kg ha^{-1})	Other Applications†	No.	Total (cm)
1	sweet corn	20	7	214N, 12P, 109K	bromoxynil metolachlor(W) alachlor(E) atrazine(W)	14	16.6
2	sweet corn	25	7	250N, 12P, 109K	atrazine metolachlor(W) alachlor(E)	13	15
3	sweet corn	10	6	184N, 28P, 165K	alachlor(W) atrazine(E) alachlor(E)	?	13
4	potato	45.4 (NE½) 46.5 (SW½)	12	279N, 74P, 118K	carbofuran (SW) metribuzin permethrin (NE) mancozeb triphenyltin hydroxide chlorothalonil diquat	?	26

† Field-half of application is indicated where application was not to entire field.

During 1993, potato was grown in Fields 1 and 2, and sweet corn in Fields 3 and 4. Fertilizer was split into 5-9 applications. Again, KCl was applied 1 month to 1 week before the starter fertilizer, with additional fertilizer N applied at intervals throughout the growing season. Herbicides were applied to corn, and potato received insecticides and fungicides along with herbicides. Irrigation records for Fields 1 and 2 list 12 irriga-

tions totaling 12-13 cm of water. Table 6.2 summarizes 1993 inputs. More detailed records of agricultural practices are given in Appendix 4.

Table 6.2. Summary of 1993 field practices.

			Nutrient Applications			Irrigations	
Field	Crop	Yield (Mg ha^{-1})	No.	Total NPK (kg ha^{-1})	Other Applications †	No.	Total (cm)
1	potato	33.6	5	333N, 54P, 311K	metolachlor metribuzin chlorothalonil carbofuran methamidophos diquat sulfur esfenvalerate	12	11.8
2	potato	46.4	6	north 297N, 54P, 389K south 357N, 54P, 389K	metolachlor metribuzin carbofuran sulfur esfenvalerate chlorothalonil methamidophos diquat	12	12.8
3	sweet corn	8.0	9	189N, 29P, 65K	alachlor(W) atrazine(E) alachlor bromoxynil(W) alachlor(E)	?	?
4	sweet corn	9.2	8	northeast 188N, 29P, 65K southwest 173N, 25P, 65K	alachlor atrazine	?	?

† Field-half of application is indicated where application was not to entire field.

HYDROLOGY OF STUDY FIELDS - 1992 AND 1993

Precipitation records were gathered and adjusted as described in the section of this chapter on nitrogen leaching modeling. Irrigation records were supplied by RDO Farms (Fields 1 and 2) and B&D Farms (Fields

3 and 4). RDO provided detailed irrigation records, with timing and amount of water applied. B&D does not record their irrigation in as much detail; they are more inclined to use informal judgment about when the crop needs water. Precipitation and irrigation are shown in Figure 6.3 for 1992 and Figure 6.4 for 1993.

Field 2 received 78 cm of rainfall plus 15 cm of irrigation in 1992, and 99 cm of rainfall plus 13 cm of irrigation in 1993 (Table 6.1 and 6.2). Field 1 received essentially the same, but with an additional 1-2 cm of irrigation water during these two years. In 1992 Field 3 received about 13 cm of irrigation and Field 4, about 25 cm. The timing and exact amounts of irrigations on Fields 3 and 4 are not known for 1993.

Run-on and runoff are unusual in this setting because the fields are flat and the soil is highly permeable. However, these may have occurred during the flood times in 1993, where standing water on Field 1 was continuous with water standing in low land and the highway ditch to the south. We cannot say how the flooding affected the overall water balance. It is also possible that some water runs on or off the fields during spring thaws in some years.

Detailed calculations to estimate evapotranspiration were made for Field 2 using WISP (Wisconsin Irrigation Scheduling Program; Curwen and Massie, 1990) for periods when crop cover were present, and the method of Ritchie (1972) for periods with no crop cover. Combining evapotranspiration estimates with rainfall, irrigation, and change in soil water storage allows estimation of groundwater recharge. In 1992, 93 cm of precipitation plus irrigation were received. The combination of WISP and Ritchie methods calculated 46 cm of ET and 1 cm of water added to soil storage. This leaves 46 cm of groundwater recharge. In 1993, precipitation plus irrigation was 112 cm, ET was 50 cm, and there was no change in soil storage, leaving a calculated groundwater recharge of 62 cm.

GROUNDWATER MONITORING

Instrumentation of fields

Each field was divided along a groundwater flow line, so that conventional practices could be implemented on one side and BMPs on the other. Three multilevel piezometers (MLPs) were installed in each field half, with one or two additional MLPs upgradient of each field (Figure 6.2). The MLPs were installed using a hollow stem auger in November 1991. MLPs consisted of 18 polypropylene tubes of 6-mm inside diameter attached to a 32 mm (1¼ in. nominal) PVC pipe which terminates in a nylon-mesh screen. The screens were 15, 20 or 40 cm long, providing continuous screening over a length of 340 cm (Figure 6.5).

Rainfall in 1993 was much greater than average, causing the water table to rise above the top of the highest screens in the MLPs, and making it impossible to sample groundwater at the top of the saturated zone. In order to restore full sampling capability, additional screens were installed in March 1994 adjacent to the existing MLPs. The new screens were numbered 19 through 25, and were constructed essentially like the

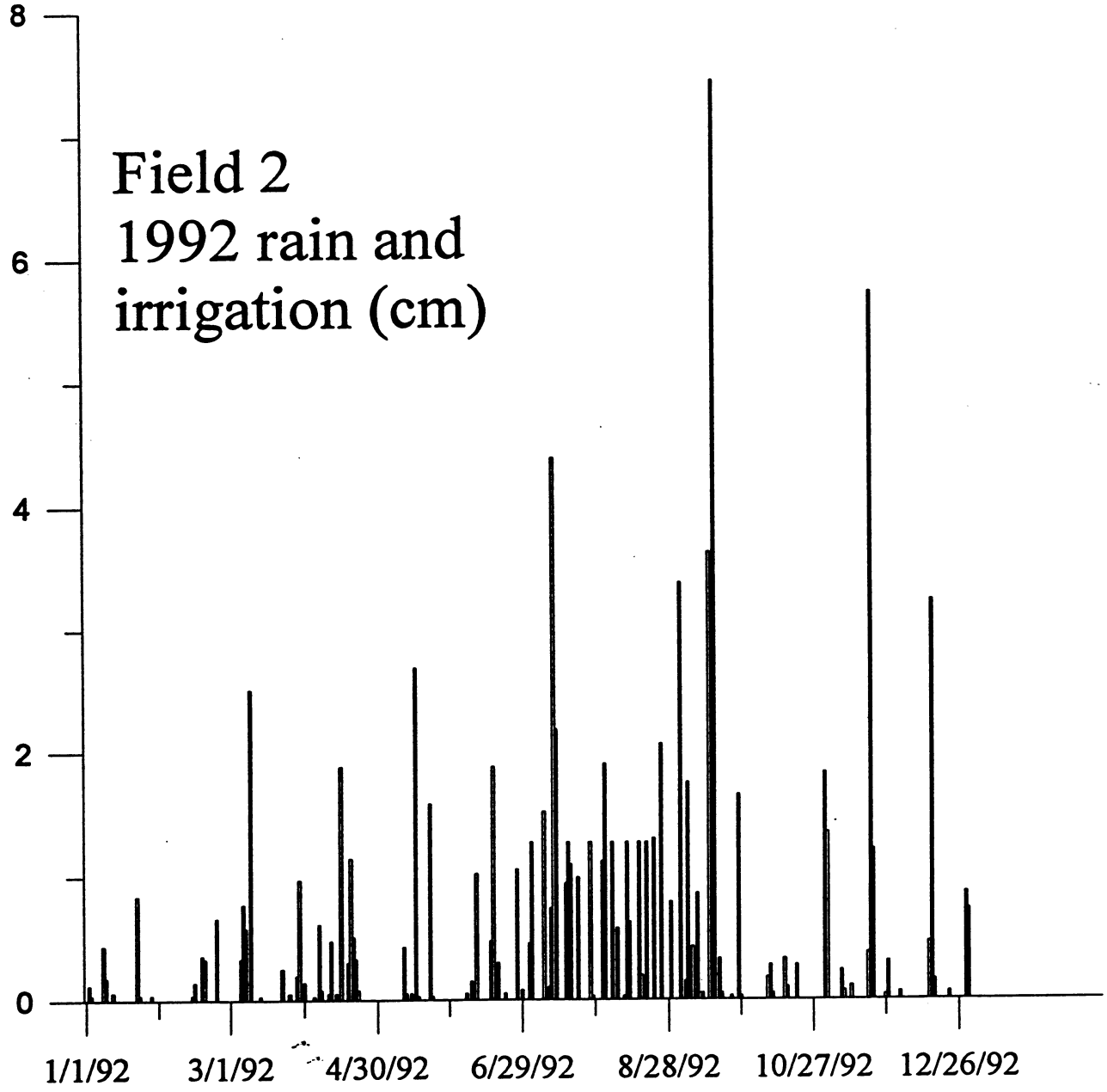


Figure 6.3 Rainfall and irrigation, 1992.

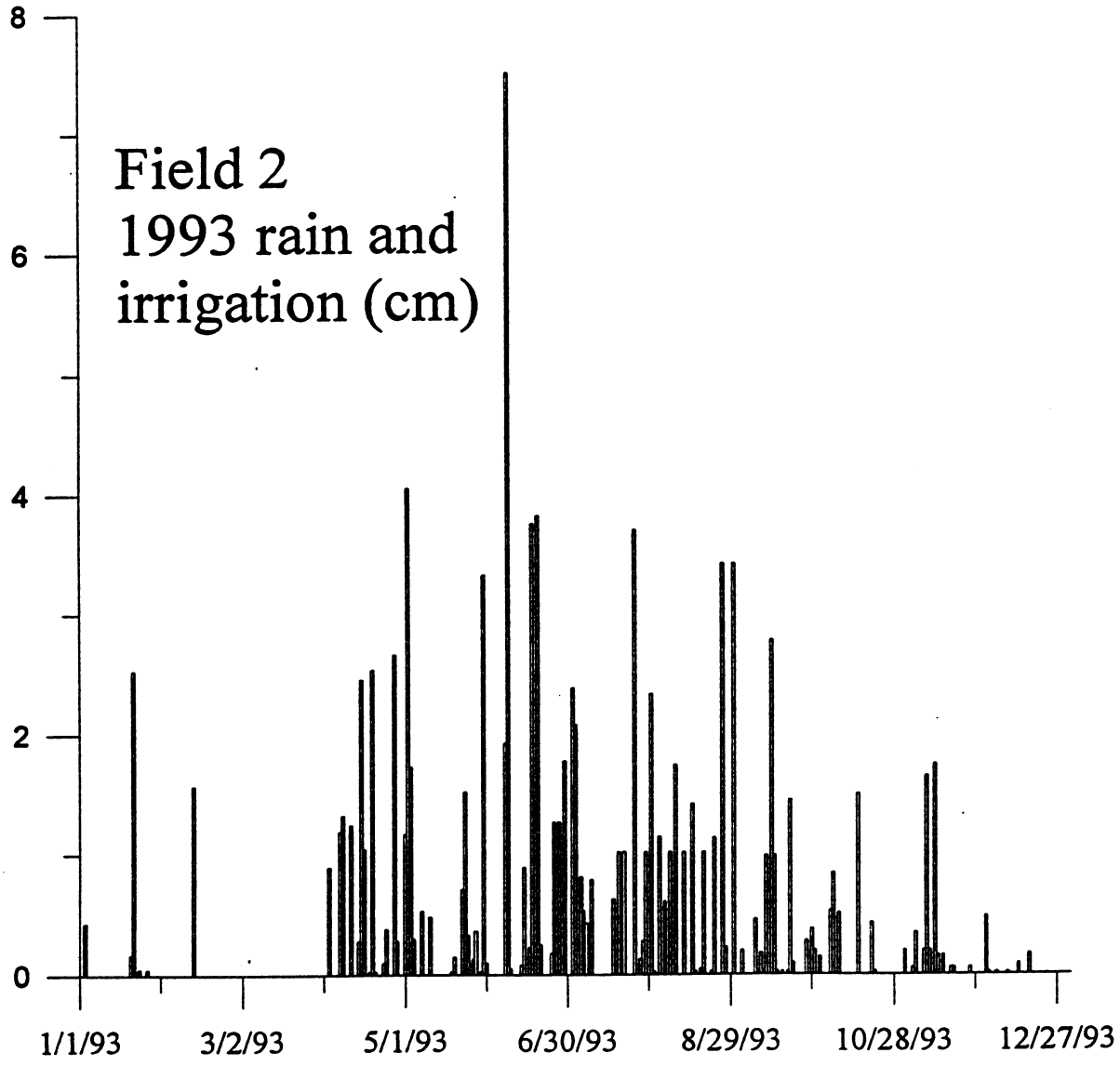


Figure 6.4 Rainfall and irrigation, 1993.

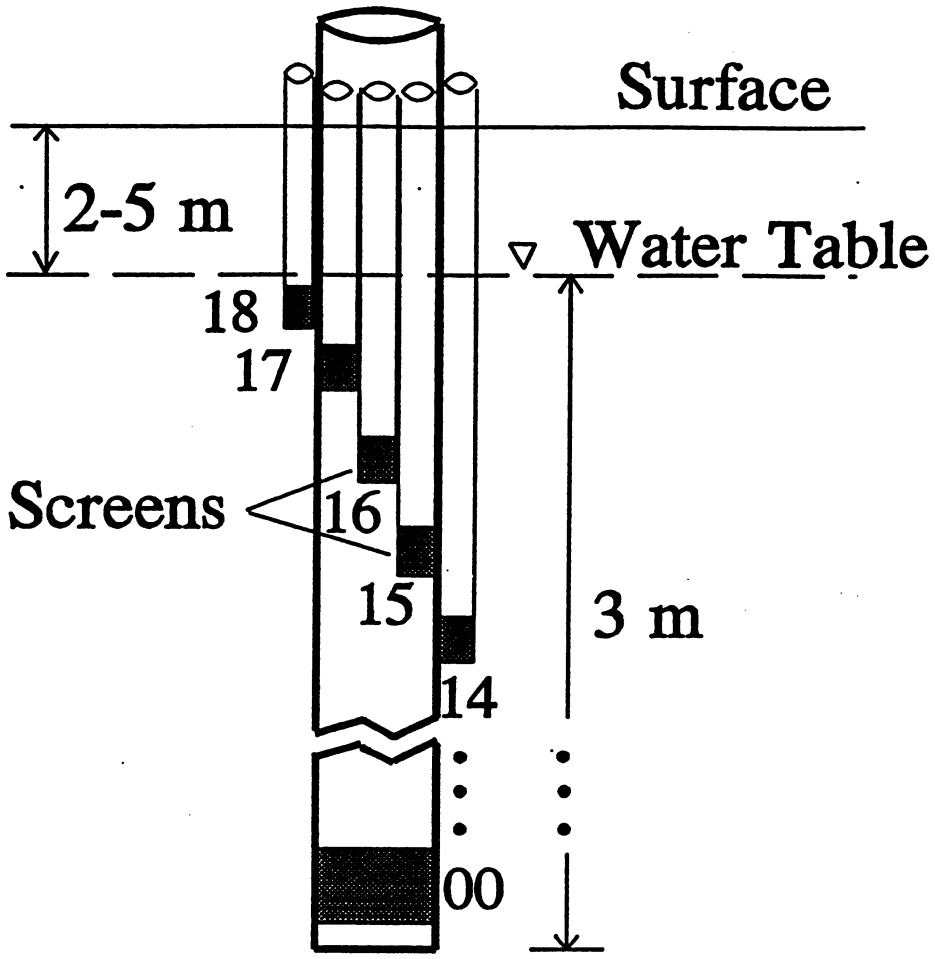


Figure 6.5 Construction of multilevel piezometers (schematic), showing port numbering order. The vertical dimension is compressed.

original MLPs. The only noteworthy difference is that the new screens were not contiguous; they were 15 cm long and were placed at 25-cm intervals. The bottom of screen no. 19 was at the same depth as the bottom of screen no. 18 on the original sampler. This increased the sampling resolution at that depth, since screen no. 18, which had initially been designed as an interceptor for a possibly rising water table, was 40 cm long (compared to 15 cm for most of the other screens).

Groundwater sampling and analyses

Groundwater sampling was conducted at 2-week to 2-month intervals depending on season. Samples were analyzed for NO_3, Cl, pH, conductivity and temperature each sampling round, and for additional inorganic parameters and pesticides periodically. Frequently, only alternate ports of each MLP were sampled. Sampling and analytical procedures were similar to those described in Chapter 5. Differences from those procedures will be described here.

Samples were collected with a peristaltic pump. Before the sample was collected, tubes were purged until at least three well volumes were removed. Samples were analyzed immediately for pH, temperature and conductivity with field meters. Temperature measurements were not reflective of groundwater conditions, but were used to standardize conductance to the standard 25°C basis.

Laboratory analyses were done at the Environmental Task Force Laboratory at the UW – Stevens Point. Inorganic methods were explained in Chapter 5. Pesticides were analyzed by gas chromatography (EPA Method 507; USEMSL, 1988) during fall 1992, and by gas chromatography/ mass spectroscopy (modified EPA method 8270; USEMSL, 1988) thereafter. A complete listing of pesticide analytes and their detection limits is presented in Appendix 5. More details of the analyses are documented in Kraft and Stites (1992).

Inorganic analyses

By summer 1994, 23 sampling rounds were completed (Table 6.3). In addition to the routine analyses, metals analyses (Ca, Mg, K, Na, Fe, Mn, and Al) were conducted at eight MLPs during Round 1, 10 MLPs during Round 7, and four MLPs during Round 14; alkalinity at all MLPs during round 1; and sulfate along with the metals in rounds 1 and 14.

Table 6.3. Dates of sampling rounds and analyses performed in addition to routine panel.

Round	Sampling dates	Analyses†
		1992
1	January 15-February 6	Alkalinity, SO_4, metals‡ (8)
2	March 23-April 6	
3	April 20-24	
4	May 27-June 10	
5	June 24-July 9	
6	July 9-23	
7	July 28-August 10	Metals (10)
8	August 28-September 9	
9	September 18-28	Pesticides (3)
10	October 19-November 9	
11	November 30-December 9	
12	Dec. 28-January 11 (1993)	
		1993
13	February 11-25	
14	March 27-April 8	SO_4, metals (4), pesticides (4)
15	June 10-16	
16	July 6-14	
17	August 5-14	
18	September 11-17	
19	October 15-November 3	Alkalinity, SO_4, metals (18), pesticides (8)
20	December 1-22	Pesticides (8; 2 ports at F3W7; 1 port at F2W8)
		1994
21	January 5-February 21	
22	March 5-19	
23	April 4-27	

†in addition to routine analyses, these analyses were done on the number of MLPs given in parentheses.
‡Al, Ca, Fe, K, Mg, Mn, Na

Because upgradient MLPs of Fields 1, 2, and 3 had similar groundwater chemistries, and because they were downgradient of mostly forested recharge areas, the results from these were grouped and considered representative of non-agriculturally affected groundwater near the study fields. Results from MLPs upgradient of Field 4 were not included with other upgradient MLPs since they are influenced by upgradient agricultural activities.

Data from the 24 in-field MLPs indicated marked differences in chemistry and agrichemical content when compared with data from the upgradient MLPs. This can be seen when averages of all in-field results

are compared to averages from the MLPs upgradient of Fields 1, 2 and 3 (Table 6.4). The average upgradient pH was 5.4 compared to 5.0 in-field. Solute content increased markedly in-field as indicated by a specific conductance increase from 7.5 mS m^{-1} upgradient to 34 in-field. Nitrate, Cl, Na, K, Ca, Mg, and Al concentrations also increased at in-field MLPs relative to upgradient MLPs. The in-field Fe concentration decreased.

Table 6.4. Average results from upgradient MLPs of fields 1, 2 and 3 vs. average of all in-field MLPs. Measurements are in mg L^{-1}, unless noted.

Parameter[1]	Upgradient MLPs	In-field MLPs
pH	5.4	5.0
conductivity	7.0[1]	33 [1]
alkalinity	4.9	2.8
nitrate-N	1.2	21
chloride	2.0	21
iron	1.5	0.02
sodium	1.8	2.9
potassium	0.59	16
calcium	5.3	26
magnesium	1.3	11
aluminum	0.20	0.69
manganese	0.08	0.09

[1] mS m^{-1}

Mean NO_3-N concentrations in the 3-m sampled zone are shown in Table 6.5. These were computed by averaging over the 23 sampling dates from January 1992 through April 1994. Each observation was weighted according to the length of MLP screen from which the sample was taken. The average NO_3-N concentrations for MLPs within the four fields are 24.4, 22.4, 19.3, and 23.4 mg L^{-1}. The mean NO_3-N concentration for the six MLPs upgradient of Fields 1-3 was 1.4 mg L^{-1}.

A lower 1993 fertilizer-N application on the southwest half of Field 4 (173 kg ha^{-1}, Table 6.2) compared to the northeast half (188 kg ha^{-1}) apparently reduced NO_3-N concentrations in groundwater. The mean weighted NO_3-N concentration for June 1993 through April 1994 in the northeast three MLPs was 27.5 mg L^{-1}, and in the southwest three MLPs it was 22.1 mg L^{-1}. The difference is statistically significant (p = 0.002; t test). Nitrogen fertilizer applications were uniform on Fields 1-3, and differences between field-halves' NO_3 concentrations were small and nonsignificant.

Table 6.5. Mean concentrations of NO_3-N throughout sampling period, weighted for MLP screen length.

Field	MLP	[NO_3-N]	Field	MLP	[NO_3-N]
1	1	25.3	3	1	18.8
1	2	19.5	3	2	17.6
1	3	23.4	3	3	19.0
1	4	24.0	3	4	18.2
1	5	28.5	3	5	22.4
1	6	25.4	3	6	19.9
1	7*	0.5	3	7*	2.7
1 (mean)	1 - 6	**24.4**	3 (mean)	1 - 6	**19.3**
2	1	24.0	4	1	29.6
2	2	17.0	4	2	26.7
2	3	23.7	4	3	25.6
2	4	24.6	4	4	19.5
2	5	20.5	4	5	18.2
2	6	24.7	4	6	21.1
2	7*	0.8	4	7*	17.7
2	8*	0.8	4	8*	17.5
2 (mean)	1 - 6	**22.4**	4 (mean)	1 - 6	**23.4**

*Upgradient MLP, not included in field average.

Pesticide results

Pesticide analyses were done at three MLPs in fall 1992, at those three plus a fourth in spring 1993; and at those four MLPs plus fourteen others in late fall/early winter 1993. Pesticides were detected at all MLPs. These included metolachlor, metribuzin, alachlor, carbofuran, and atrazine with its de-ethyl and de-isopropyl metabolites. Atrazine exceeded Wisconsin's preventive action limit (PAL) at 15 of 18 MLPs, more frequently than the other pesticides. Metolachlor exceeded its PAL at eight MLPs, carbofuran at three, and alachlor at one. Atrazine exceeded the enforcement standard (ES) at three MLPs; metolachlor at one. A summary of the results is shown in Table 6.6, and detailed data are in Appendix 6.

We compared pesticide analyses on the conventional and BMP halves of the four fields to determine if differences between treatments (Table 6.1 and 6.2) affected groundwater quality. Results from sampling round 19 (fall/early winter, 1993; Table 6.7) were used for comparisons since it was the most comprehensive.

Field 1 had more MLP ports with metribuzin and carbofuran detections in the east half, as well as higher concentrations of the same. However, this pattern does not correspond to known differences in applications of these materials. Metolachlor was applied only on the west half in 1992 and on the entire field in 1993.

This may explain why it was detected in more MLP ports in the east half during the late fall sampling of 1993.

Table 6.6. Summary of pesticide detects for 256 ports sampled, fall 1992, spring 1993, and late fall – early winter 1993.

Pesticide[1]	ES[2]	PAL[3]	Detect. lim.	Detects	Maximum[4]	Mean[4]	Median[4]
Alachlor	2	0.2	0.1	13	1	0.45	0.4
Metolachlor	15	1.5	0.4[5]	110	157	6.67	0.7
Metribuzin	250	50	0.2[5]	169	24.6	1.48	0.6
Atrazine	(atrazine and its		0.2[5]	134	2.5	0.44	0.3
de-ethyl	metabolites are		0.1	195	3.6	0.51	0.4
de-isopropyl	summed)		0.2[5]	12	0.4	0.27	0.2
Total	3	0.3			6.4	0.74	0.5
Carbofuran	50	10	0.1[5]	89	20.6	1.98	0.9

[1] All measurements $\mu g\ L^{-1}$.
[2] DNR Enforcement Standard.
[3] DNR Preventative Action Limit.
[4] Of MLPs with detects.
[5] Detection limits for method 507, used in the first sampling round, were metolachlor 0.6, metribuzin 0.1, atrazine 0.1; alachlor, de-isopropyl atrazine and carbofuran were not detectable.

In Field 2, the most prominent difference was in average carbofuran concentration, although there was no known difference in application of this material.

In Field 3, atrazine was applied only to the east half in both 1992 and 1993. Atrazine was detected in seven MLP ports in the east half and none in the west half. De-ethyl atrazine was detected in more ports and in greater concentrations in the east half. Both the mean and the maximum concentrations of summed atrazine and progeny products in each MLP differed significantly between the halves of Field 3, strongly suggesting that a real improvement in groundwater quality was achieved.

The southwest half of Field 4 had greater concentrations of de-ethyl atrazine and metribuzin, and less carbofuran than the northeast half. Again, there is no explanation for this in the records of pesticide applications. The great difference between field-halves in metolachlor concentrations resulted from an anomalous strong pulse at MLP 5, where the peak concentration reached 157 $\mu g\ L^{-1}$. The reason for this pulse is not known, but the smooth concentration profile involving all 12 sampling ports indicates it is not likely to be an artifact.

De-ethyl atrazine was reported at one port in each of two upgradient MLPs, at concentrations of 0.1 and 0.6 $\mu g\ L^{-1}$. The 0.6 $\mu g\ L^{-1}$ value in particular is not close to the detection limit. Neither of these detections is likely to indicate carryover from another sample, since other samples collected at the same time and place

Table 6.7. Pesticide detections and average concentrations in MLPs, November-December 1993. Field halves are considered separately. Halves sometimes received contrasting pesticide treatments.

		Atrazine		De-ethyl atrazine		De-isopropyl atrazine		Alachlor		Metolachlor		Metribuzin		Carbofuran	
Field-half	screens sampled	no. detects	avg. concentration.	no. detects	avg. concentration.	no. detects	avg. concentration.	no. detects	avg. concentration.	no. detects	avg. concentration.	no. detects	avg. concentration.	no. detects	avg. concentration.
1 W	24	16	0.72	17	0.95	0		0		4	0.90	7	0.73	3	1.43
1 E	22	13	0.57	19	0.76	2	0.20	0		14	0.62	22	3.90	15	4.33
2 W	24	20	0.26	24	0.46	3	0.33	0		10	0.46	22	0.86	5	0.18
2 E	24	19	0.34	24	0.60	0		0		11	0.67	22	1.20	5	0.46
3 W	24	0		9	0.10	0		0		7	1.26	8	0.48	14	0.28
3 E	24	7	0.24	24	0.27	0		11	0.52	3	0.47	3	0.20	11	0.45
4 SW	23	17	0.58	23	0.80	6	0.27	2	0.10	19	30.45	23	2.44	1	0.20
4 NE	24	13	0.32	23	0.38	1	0.20	0		15	4.33	24	0.78	16	2.44

were clean, and the sample collected immediately before these MLPs contained at most 0.1 $\mu g\ L^{-1}$ de-ethyl atrazine. Possible explanations could be if atrazine were sprayed far enough from the field edge to affect upgradient MLPs, or if sampling equipment came in contact with a contaminated surface.

Patterns of solute plume arrival and vertical advection.

Downward advection of solute plumes was observed at most MLPs. Figure 6.6A shows Cl peaks and minima trended downward with time. Figure 6.6B shows the same for NO_3-N, but the concentration contrasts tend to be less distinct. Figure 6.6C shows atrazine concentration profiles from the three pesticide sampling rounds at Field 2, MLP 3. Since pesticide sampling was less frequent, the patterns are not as clear as the NO_3-N but the concentration pulse arriving at the top of the sampled interval in the second round appears to have moved downward by the third round, possibly with some dispersion, and is being followed by another pulse.

NITRATE LOADING I: POTENTIALLY-LEACHABLE N FROM N BUDGET

An N budget was estimated for the four study fields for 1992 and 1993, using an approach similar to that of Meisinger and Randall (1991), focusing on N available for leaching:

$$N_{pl} = N_{input} - N_{output} - \Delta N_{st},$$

where N_{pl} = potentially leachable N, and

ΔN_{st} = change in stored N.

N_{input} and N_{output} are those N components that enter and exit the field between the crop canopy and soil root zone. The N_{output} component excludes N_{pl}. N_{pl} is an estimate of N potentially available for leaching to groundwater in the long term.

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

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

The inputs considered in development of the budget were fertilizer, symbiotic and non-symbiotic microbial fixation, irrigation water, precipitation, dry deposition, mineralization and crop seed. Outputs were harvested crop, ammonia and denitrification losses, soil erosion and runoff, and miscellaneous gaseous losses.

Nitrogen inputs

Fertilizer

Fertilizer-N inputs from growers' records are documented in Table 6.8.

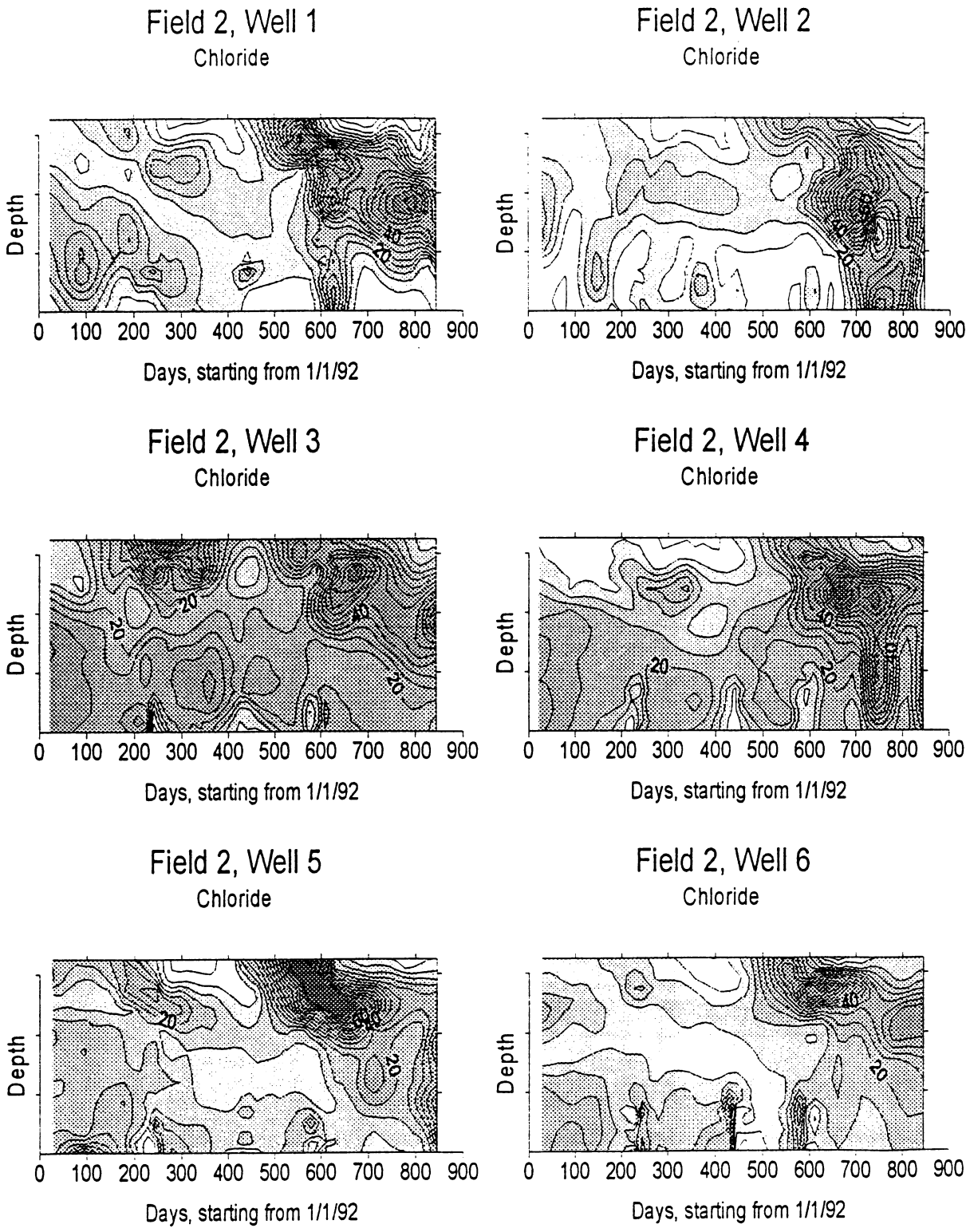


Figure 6.6A Contours of chloride concentration at multilevel piezometers throughout the sampling period. Darker shades represent greater Cl concentrations.

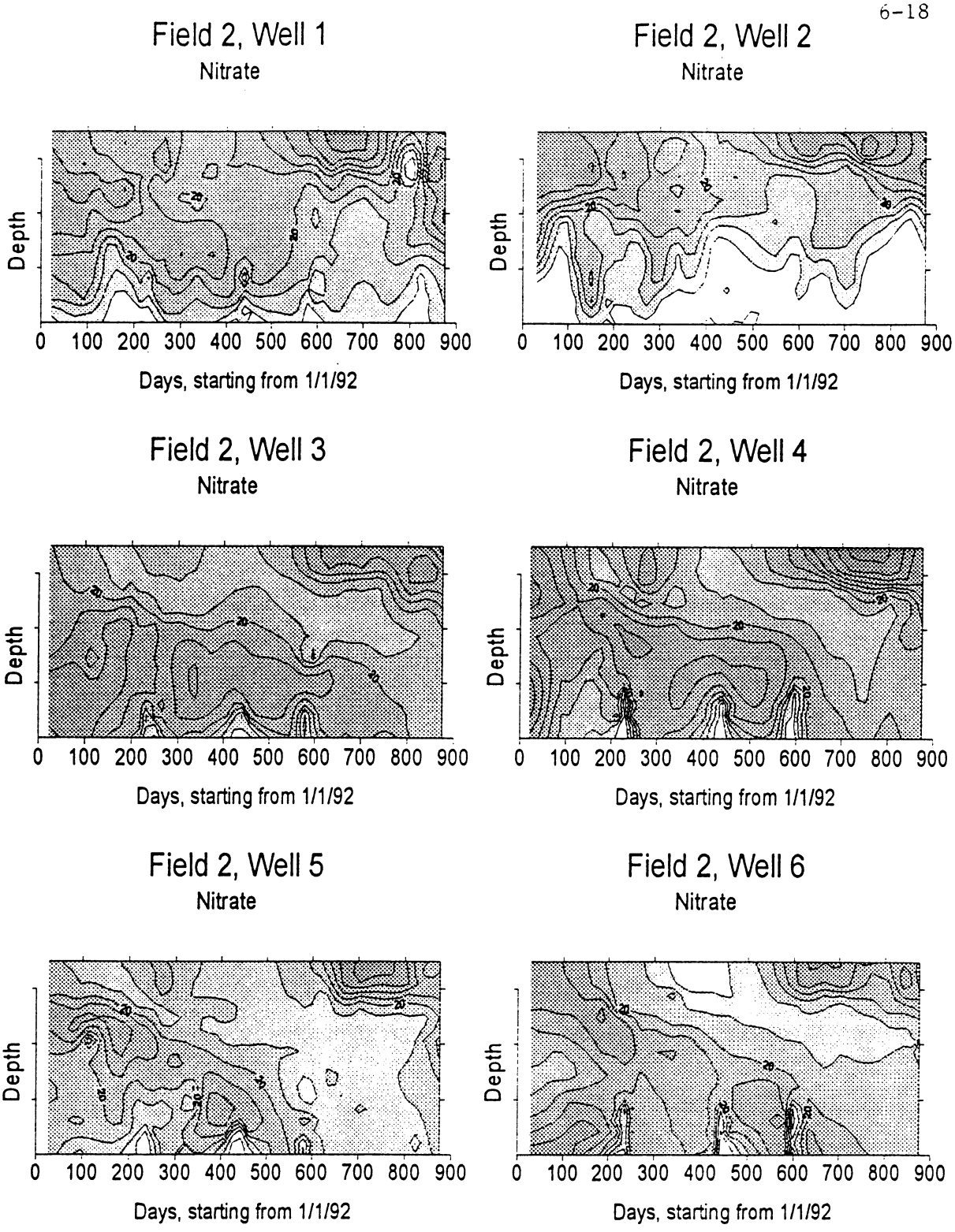


Figure 6.6B Contours of nitrate concentration at multilevel piezometers throughout the sampling period. Darker shades represent greater NO_3 concentrations.

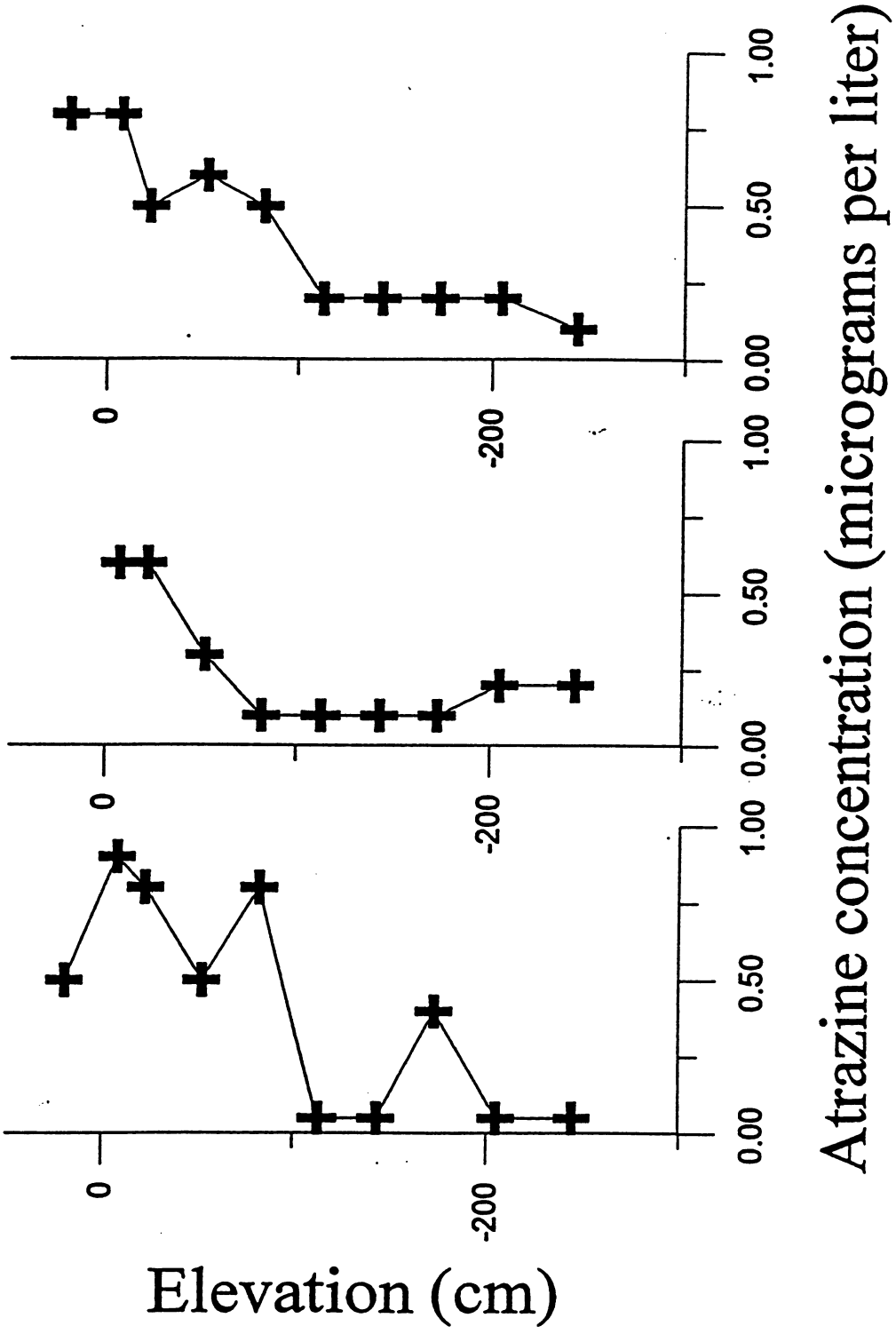


Figure 6.6C Concentration profiles of atrazine at Field 2 MLP 3 on three sampling dates.

Microbial fixation

Symbiotic and non-symbiotic N fixation were negligible for the crops and conditions of 1992 and 1993, and were disregarded.

Table 6.8. Fertilizer N inputs to vegetable fields, 1992 and 1993.

Field	Year	Fertilizer N (kg ha^{-1})
1	1992	214
	1993	333
2	1992	250
	1993	297 N; 357 S*
3	1992	184
	1993	189
4	1992	279
	1993	188 N; 173 S*

*N, north field-half; S, south field-half

Irrigation water

This N input can be estimated from the quantity of water applied and its N content. Limited sampling of irrigation water indicated a concentration of 0.4 mg L^{-1} NO_3-N. With yearly irrigation inputs ranging from 12 to 26 cm, N inputs from irrigation would range between 0.4 and 0.9 kg ha^{-1}, too low to be important. Other N forms (organic, NO_2, and NH_4) are not found in Wisconsin central sand plains groundwater at concentrations exceeding NO_3-N except near contaminant sources such as manure storage facilities and fertilizer spills, neither of which is a concern for these fields.

Atmospheric deposition

Schepers and Mosier (1991) present a national contour map of precipitation N (wet deposition of NO_3-N plus NH_4-N) constructed with data obtained from the National Atmospheric Deposition Program which shows that Wood County receives about 5.6 kg N ha^{-1} per year (5 lb of N/acre). However, annual average Wisconsin precipitation N was reported at 15 kg N ha^{-1} by Andraski and Bundy (1990) and about 20 kg N ha^{-1} by Hoeft et al. (1972). These estimates are supported in work in the upper Midwest by MacDonald et al. (1992). Therefore, 15 kg N ha^{-1} yr^{-1} was used as an estimate of precipitation N. Meisinger and Randall (1991) and Schepers and Mosier (1991) state that dry-deposition N is generally assumed to be about the same as precipitation N. In that case, annual dry deposition in the study area would also supply 15 kg N ha^{-1}.

Mineralization

Oberle and Keeney (1990), in a study of N release from soil organic matter on Plainfield loamy sand, indicate a net mineralization of 45 kg N ha^{-1} during the growing season. However, neither wet nor dry deposition were taken into account in the study. Therefore, mineralization of 15 kg N ha^{-1} (45 kg ha^{-1} minus our estimate of 30 kg N ha^{-1} for wet + dry deposition) was assumed.

Crop seed

This input can be estimated from the seeding rate and N content of seed (Meisinger and Randall, 1991, Table 5-4). Based on average seeding rates, corn seed would add less than 0.3 kg N ha^{-1}; potato seed, about 6-12 kg N ha^{-1} (Meisinger and Randall, 1991). Since potato seed N is comparable to wet or dry deposition, it was included in the N budget at 9 kg N ha^{-1}.

Nitrogen outputs

Harvested crop

Harvested N is estimated from crop yield and literature-reported crop N content. For potato, Saffigna et al. (1977) report N content of 2.8 g N kg^{-1} while Legg and Meisinger (1982) report 3.4 g N kg^{-1}. Meisinger and Randall (1991) report an average of 4.0 g N kg^{-1} with a range of 3-5 g N kg^{-1}. For sweet corn, Meisinger and Randall (1991) estimate harvested N content at 4.3 g N kg^{-1}, with a range of 3.8-4.8 g N kg^{-1}. Estimates of harvested crop N showing the average and range of harvested N based on Meisinger and Randall (1991) are shown in Table 6.9.

Table 6.9. Estimated harvested N from the four vegetable fields in 1992 and 1993.

Field	Year	Crop	Crop Yield (Mg ha^{-1})	Harvested N (kg ha^{-1})	
				average	range
1	1992	sweet corn	20	86	76-96
	1993	potato	34	140	102-170
2	1992	sweet corn	25	110	95-120
	1993	potato	46	180	138-230
3	1992	sweet corn	10	43	38-48
	1993	sweet corn	8.0	34	30-38
4	1992	potato	45 NE; 47 SW*	180 NE; 190 SW*	135-225 NE; 141-235 SW*
	1993	sweet corn	9.0	39	34-43

*NE northeast field-half; SW southwest field-half.

Fertilizer volatilization

When ammonium or ammonium-yielding fertilizers are not incorporated, N can be lost to ammonia volatilization (Blackmer, 1987). However, Saffigna et al. (1977) claim that ammonia volatilization will not occur on Plainfield soils due to the soil's acidity. Additionally, Oberle and Bundy (1987) report that 2.5 mm of rainfall within 4 days after urea application will significantly limit ammonia loss. As the study fields were irrigated regularly, ammonia volatilization will be assumed negligible.

Denitrification

Saffigna et al. (1977) suggest that denitrification is nonexistent in these soils. However, the spring of 1993 was exceptionally wet; perhaps conditions for denitrification existed. For purposes of the N budget, denitrification losses will be assumed negligible.

Soil erosion losses and surface runoff

Nitrogen may be lost by erosion as soil particles are removed from fields by wind or water. These losses were not significant.

Miscellaneous gaseous losses

Meisinger and Randall (1991) suggest that an approximation of these losses would equal a 1% loss of the total N inputs. Included in these losses are: N_2O evolution during nitrification, decomposition of nitrous acid, and reactions of nitrous acid with soil minerals and organic constituents. A loss of 1% of N_{input} will be assumed.

Gaseous losses from crops and crop residues

Information is limited concerning the gaseous loss of N as ammonia plus volatile amines during plant senescence. Estimates of 2-8% of total aboveground plant N are given for this loss (Meisinger and Randall, 1991). For purposes of accounting for this loss in the N budget, the following will be assumed:

- above-ground plant-N gaseous loss is 5%;
- above-ground plant-N for potato is 80 kg ha^{-1}; and
- above-ground plant-N for sweet corn, not including the harvested portion, is 118 kg ha^{-1}.

Total above-ground plant N content was based on Olson and Kurtz (1982). The mean harvested N (62 kg ha^{-1}) was subtracted from the above-ground sweet corn N. Therefore, the N loss associated with senescence will be taken as 4 kg N ha^{-1} for potato and 6 kg N ha^{-1} for sweet corn.

Little information is available regarding possible volatile loss of N from crop residue in this setting, but evidence indicates it is probably a small loss. In the case of potato, the amount of N contained in residue is small, so that any such loss would be minor. Sweet corn provides considerable amounts of residue from which N might be lost as ammonia or other less-soluble gases. We expect ammonia would be largely retained in the field for the same reasons that fertilizer volatilization should be minimal, and because chisel-plowing was usually done shortly after sweet corn harvest, which would virtually eliminate ammonia volatilization from the buried portion of the residue. Would N be lost in other, less-soluble gases? The literature indicates this is not likely. Wagger et al. (1985) recovered 100% of residue ^{15}N from incorporated sorghum and wheat residues after one growing season. Parker (1962) measured N content of field corn residue as it decomposed on the soil surface. While the total mass of the residue decreased 50% in 140 days, the mass of N recovered in the residue actually increased slightly. Sweet corn and field corn residues are reasonably analogous, so we conclude gaseous loss of N from sweet corn residue was minimal.

Results

Using the above equation, the potentially leachable N (N_{pl}) was determined for each field for the years 1992 and 1993 (Table 6.10). N_{pl} ranged from 133 to 237 kg ha^{-1}, or 40-82% of N_{input}. The greatest N_{pl} percentages were found for sweet corn in Fields 3 and 4 in both years, due to poor yields and little harvested N output. The greatest N_{pl} amounts were found with potato in 1993 on Field 1 and the south half of Field 2, due to the large amounts of fertilizer N applied and – on Field 1 – poor yield.

NITRATE LOADING II: CALCULATION FROM NO_3-N CONCENTRATION AND RECHARGE

A simple estimate of the NO_3-N loading to groundwater can be calculated from the average NO_3-N concentration in the shallow groundwater beneath the fields, assuming this concentration reflects recent agricultural practices. The estimated NO_3-N loading equals average NO_3-N concentration (Table 6.5) multiplied by groundwater recharge. Recharge calculated by PRZM (described later in this chapter) for Field 2 was used for all four fields. Calculated recharge was 39.1 cm under sweet corn in 1992 and and 61.0 cm under potato in 1993; the different recharge amounts are the only basis for the differences in N loading between the two

Table 6.10. Inputs, outputs, and potentially leachable N from four vegetable fields, 1992 and 1993.

Field	Year	Crop[1]	N_{input} (kg ha^{-1})[2]	N_{output} (kg ha^{-1})[3]	N_{pl} (kg ha^{-1})	N_{pl} as % of N_{input}
1	1992	C	259	94	165	64
	1993	P	387	150	237	61
2	1992	C	295	117	178	60
	1993	P	351 N; 411 S*	187	164 N; 224 S*	47 N; 55 S*
3	1992	C	229	51	178	78
	1993	C	234	42	192	82
4	1992	P	333	190 NE; 200 SW*	143 NE; 133 SW*	43 NE; 40 SW*
	1993	C	233 NE; 218 SW*	47	186 NE; 171 SW*	80 NE; 78 SW*

[1] C = sweet corn; P = potato
[2] fertilizer + precipitation + dry deposition + crop seed + mineralization
[3] crop harvest + gaseous losses
* field half: N north, S south, NE northeast, SW southwest

years in Table 6.11. Calculated recharge could be somewhat different in Fields 1, 3, and 4 when crops grown on them differed from Field 2, because potato transpires less than sweet corn. The 1993 N loading in Table 6.11 for fields 1, 3, and 4 may be greater than if field-specific recharge amounts had been used. However, as shown later below in the N-leaching model section, recharge is rather insensitive to crop rooting depth, and thus can be expected to be fairly insensitive to the crop species.

Table 6.11. Mean concentrations of NO_3-N throughout sampling period, weighted for MLP screen length, and simple NO_3-N loading estimate for 1992 and 1993 recharge. The greater loading in 1993 is a result of the greater recharge in that year. Added N refers to fertilizer and atmospheric inputs but not soil mineralization.

		N loading			
	[NO_3-N]	kg ha^{-1}			% of added N
Field	mg L^{-1}	1992	1993	Mean	(both years)
1	24.4	95	149	122	40
2	22.4	88	137	112	35
3	19.3	76	118	97	52
4	23.4	92	143	117	44
Mean	22.4	88	137	112	42

The simple estimates of NO_3-N loading were 76-95 kg ha^{-1} for the four fields in 1992, a fairly typical year, and 97-122 kg ha^{-1} for 1993, a wet year (Table 6.11). For the two years and four fields, estimated loading was 112 kg ha^{-1}, 42% of total N input.

Fifteen kg ha^{-1} less N fertilizer was applied on the southwest half of Field 4 in 1993 (Table 6.2) than on the northeast half. Concurrently, a statistically significant 5.4 mg L^{-1} reduction in NO_3-N in groundwater was observed. Using the simple loading method described above, apparently 33 kg ha^{-1} less NO_3-N loading resulted from the 15-kg ha^{-1} reduction in fertilizer N. The discrepancy probably results from imprecision in concentration and recharge estimates. Better quantitation of the effect of reduced N inputs would require more intensive study.

NITRATE LOADING III: LOADING THROUGH A WATER YEAR

In order to quantify a single year's NO_3 loading to groundwater beneath a field, the NO_3 concentration profile with depth needs to be known, and the top and bottom of a NO_3 plume need to be identified. Once these are known, the following equation can be used to determine the NO_3 per unit area in a groundwater plume:

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

with m = NO_3 mass loaded per year

a = depth of top of plume

b = depth of bottom of plume

θ = aquifer porosity

C = NO_3 concentration

z = depth below an arbitrary datum.

The top and bottom of a plume can be distinguished if some "marker" exists in the groundwater. Previous work in the WCSP has shown that conservative contaminants leach to groundwater during the year of application and that the contaminant plumes are bounded by bands of "clean" water. (Saffigna and Keeney, 1977; Manser, 1983; Bruce, 1985; Harkin et al., 1986). At the four PEG study fields, a Cl pulse from KCl fertilizer applied before planting may serve as a marker for delineating upper and lower recharge boundaries from a water year.

Methods

Consider a cropping system with annual applications of KCl fertilizer at the soil surface, with the simplifying assumption of one-dimensional vertical water flow. For a sampling point *A* at the water table, a water year begins with arrival of a Cl pulse as water affected by the early-spring application of KCl fertilizer arrives. The Cl concentration then decreases as new recharge water arrives, eventually reaching a minimum. About one year later, the cycle repeats as water influenced by the new year's KCl application arrives at the water table and the Cl concentration begins to increase again. As recharge water moves downward in the aquifer, a sampling point *B* some distance below the water table would exhibit a similar pattern, but delayed. The deeper the sampling point, the longer the delay. If the sequence of Cl concentration profiles is displayed on a time-depth iso-concentration plot, the result would resemble Figure 6.7. On day 0, a Cl pulse from an earlier year is visible in the lower part of the sampled range. After a little more than 100 days, a new Cl pulse begins to appear at the water table. Around day 230, the peak concentration appears. As time passes, the pulse moves downward through the saturated zone. The complete band of water corresponding to a year of recharge would extend from the bottom of such a Cl band to the bottom of the next incoming band above it. Note that in the figure, no attempt has been made to show dispersion of the Cl pulse as it moves deeper, and the timing is arbitrary.

Relaxing the one-dimensional assumption introduces a complication. If the unsaturated zone is homogeneous, flow will still be vertical, but as soon as a parcel of water reaches the saturated zone, it will begin to move not just downward, but also horizontally. A parcel of water near sampling point *B* some distance below the water table would not have reached the water table near point *A*, but rather some distance upgradient from *A*, and its point of origin on the soil surface would not be the same as the origin for water collected at *A*. Therefore, any spatial variability in fertilizer application or infiltration would tend to scramble the pattern seen in Figure 6.7. Further complications are to be expected because the flow medium is not homogeneous. This has several manifestations. The travel time from the surface to the water table can vary from place to

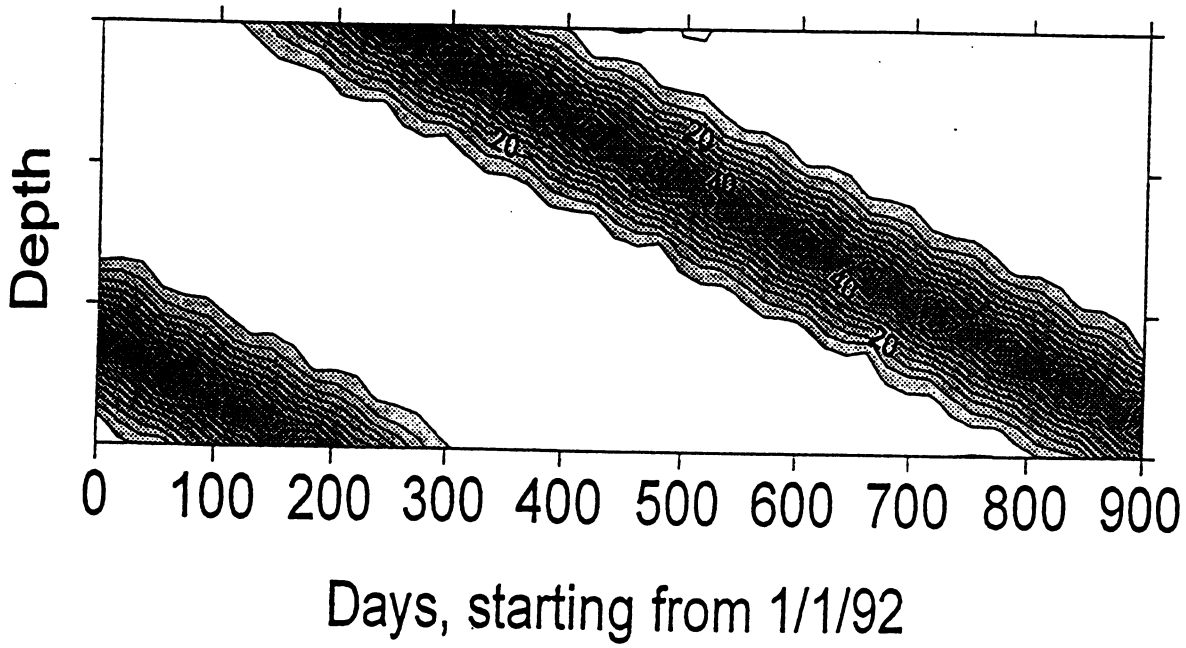


Figure 6.7 Hypothetical contour plot of chloride concentrations at a multilevel piezometer affected by annual applications of KCl fertilizer. Darker shades represent greater Cl concentrations.

place, so the Cl pulse may not reach the water table synchronously. Funnel flow (Kung, 1990) may cause water infiltrating the surface at a given place to reach the water table at different places from time to time depending on the water content of the vadose zone or the intensity of recharge. And flow paths in the saturated zone may not be parallel, so that mixing of water from various origins may take place.

Field 2 was selected for a trial of this method. Groundwater was collected from MLPs 23 times from January 1992 to April 1994. We attempted to identify horizontal bands of water between the arrival of successive annual Cl pulses as described above, using contour plots shown in Figure 6.8. When such a band was tentatively identified, individual Cl concentration profiles were examined to see if the Cl pulse persisted and maintained its identity through several sampling episodes. When a good candidate was found, a spreadsheet macro was used to integrate the amount of NO_3-N in the band.

Integrations were done for repeated dates from the uppermost sampling point down to the depth corresponding to the beginning of the water year (as defined by the Cl pulse). The amount of NO_3 indicated by these integrations was expected to increase with time as more of the year's water is available for sampling. When the next year's Cl pulse arrives, the upper boundary of the water year is defined, and the measured amount of NO_3 should be at its maximum and should remain constant thereafter. This was not always the case; sometimes the maximum amount of NO_3 was seen before the complete water year was available for sampling.

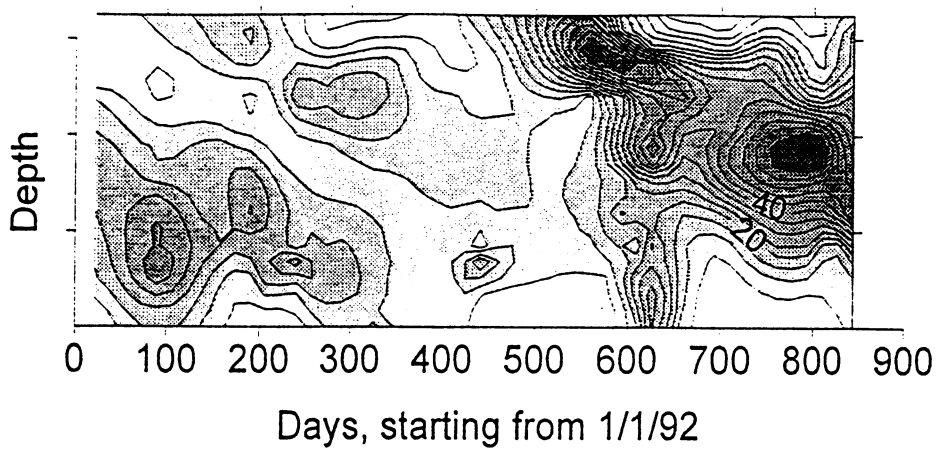
Results

In the discussion below, reference should be made to Figure 6.8 for Cl-profile features.

1992 water year

Results of the procedure are shown in Table 6.12 for the first water year at MLP 1, corresponding to the 1992 sweet corn crop. The 1992 Cl pulse appeared to be fully below the water table shortly after day 200 (Figure 6.8). The peak NO_3 mass, ordinarily indicating that all 1992 water was in the sampled zone, was reached by day 365 with 206 kg N ha^{-1}. As noted above, the greatest mass of NO_3 would not be expected until all of a year's recharge has arrived in the saturated zone. However, Cl from 1993 was not clearly present until around day 450. At that time, the mass of NO_3-N in the 1992 water had decreased to 188 kg ha^{-1}. By day 560-600, the 1992 Cl pulse seemed to have descended below the sampling depth, and the amount of 1992 N in the sampled zone was decreasing. Our best estimate of NO_3-N loading is the mean of the 3 values from day 365 through day 456: 190 kg ha^{-1}.

At MLP 2, the Cl pulse for 1992 was fully developed by about day 250 (Figure 6.8). As at MLP 1, the peak NO_3 mass was reached at day 365 (Table 6.13), with 221 kg N ha^{-1}, after which the mass decreased.



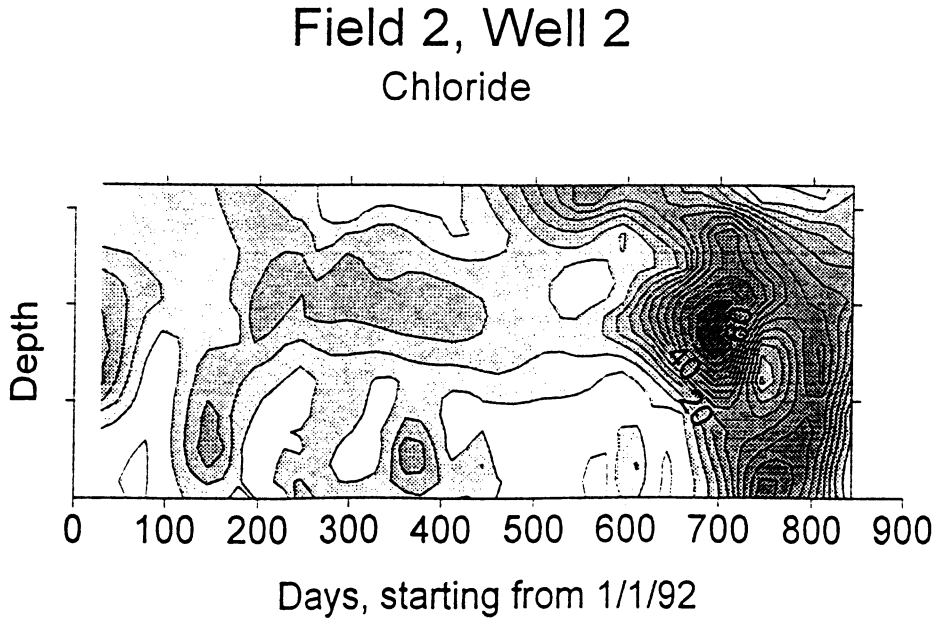
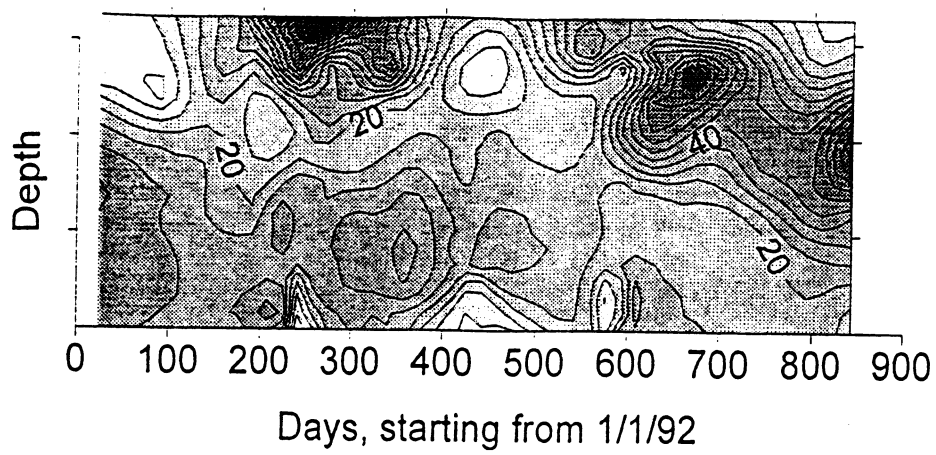


Figure 6.8 Contours of chloride concentration at multilevel piezometers throughout the sampling period. Darker shades represent greater Cl concentrations.



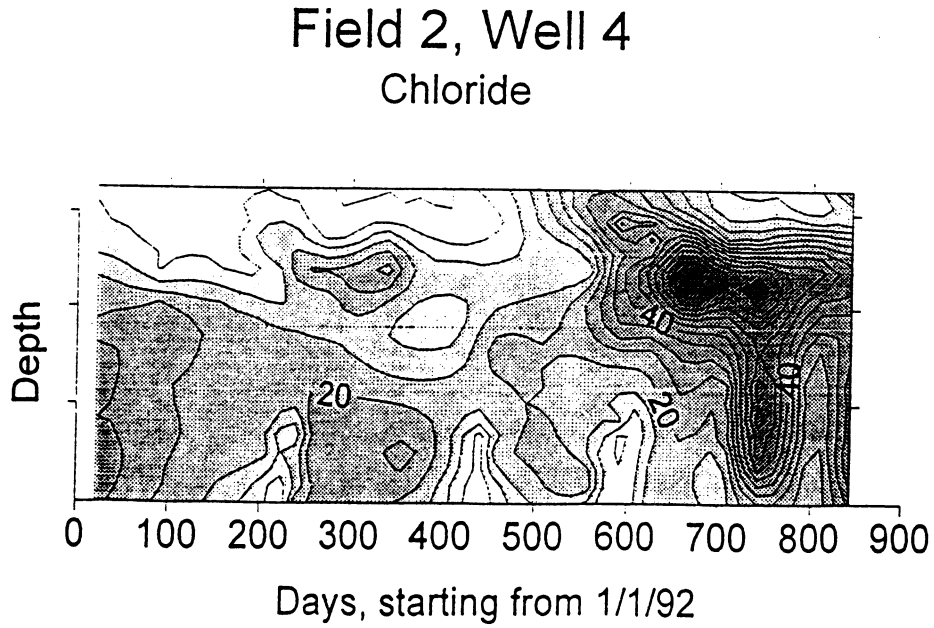
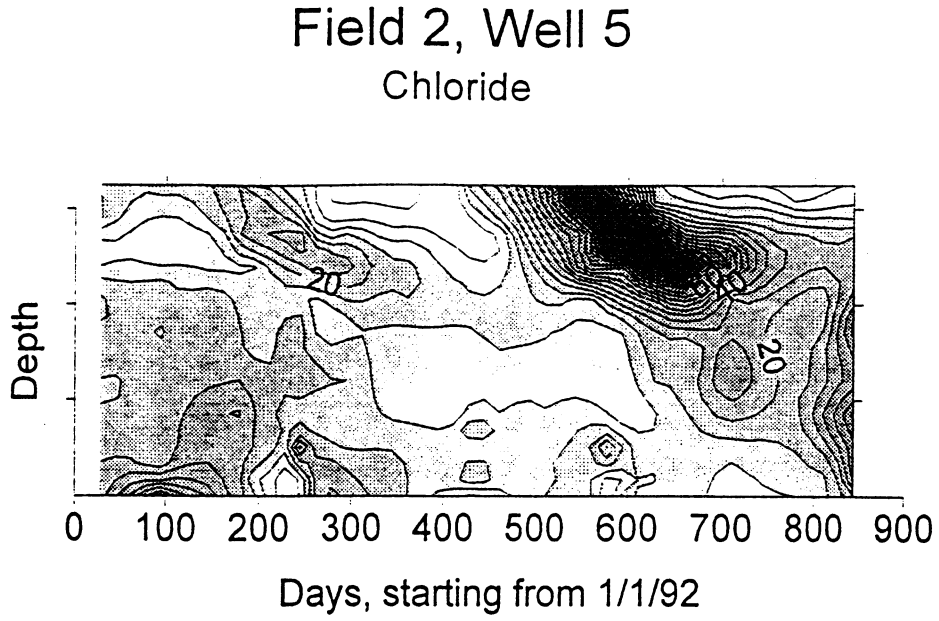


Figure 6.8 Contours of chloride concentration at multilevel piezometers throughout the sampling period. Darker shades represent greater Cl concentrations (continued).



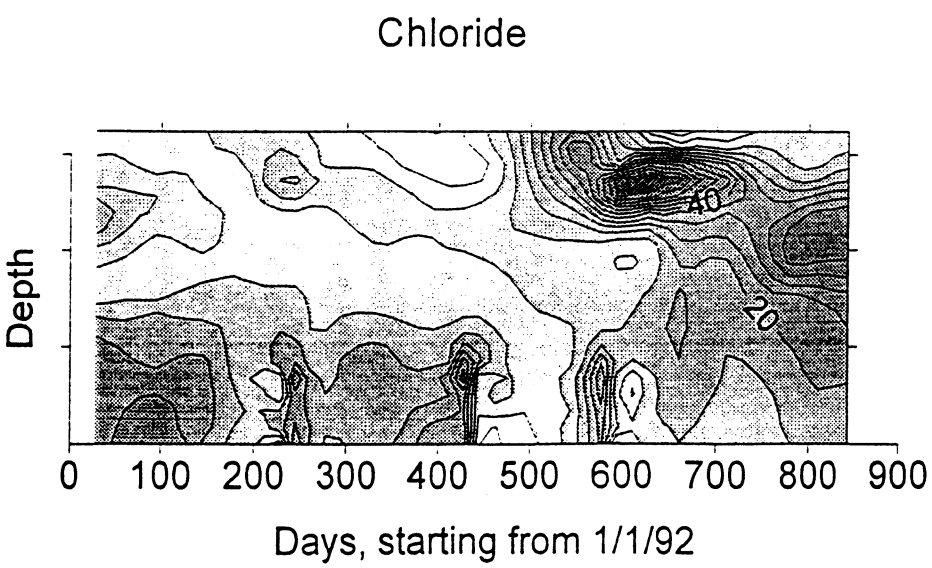
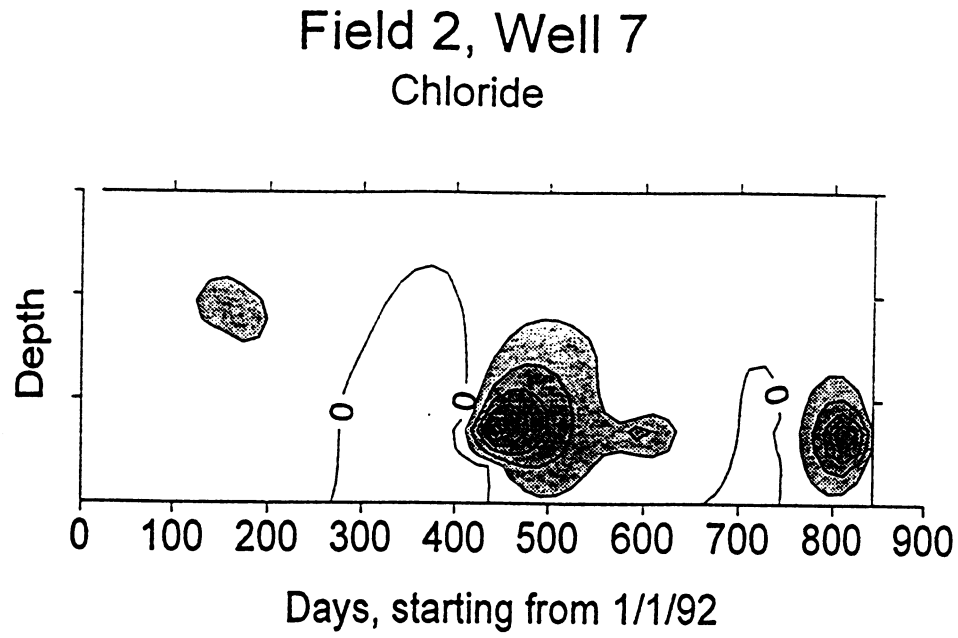


Figure 6.8 Contours of chloride concentration at multilevel piezometers throughout the sampling period. Darker shades represent greater Cl concentrations (continued).



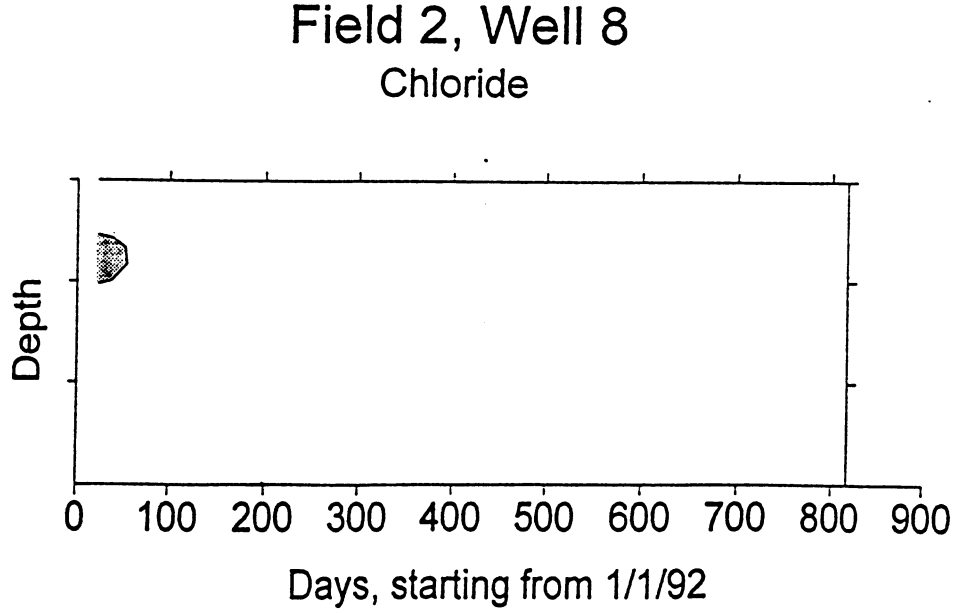


Figure 6.8 Contours of chloride concentration at multilevel piezometers throughout the sampling period. Darker shades represent greater Cl concentrations (continued).

The decrease in the measured NO_3 mass may be due to mixing of 1992 water with older water in the lower part of the sampled zone. Already, the 1992 Cl pulse was possibly passing the bottom of the MLP. The peak NO_3 mass occurred long before the arrival of 1993 Cl at the water table around day 450. By then, the mass of NO_3-N in the 1992 water was only 80 kg ha^{-1}. The fact that NO_3 masses starting on day 248 were not much less than the peak value indicates most NO_3 leaching occurred in the early part of the 1992 water year. The irregularities in the Cl pattern make it difficult to determine when the first good estimate of 1992 N loading can be computed. Therefore, the four values from day 248 through day 365 were averaged, giving an estimate of 200 kg ha^{-1}.

The 1992 Cl pulse was harder to distinguish at MLP 3. The arrival of low-Cl water at the water table was evident after day 400, early in 1993. The arrival of 1993 Cl coincided with the peak mass of NO_3-N, 216 kg N ha^{-1}, on day 459 (Table 6.14).

The pattern at MLP 4 is much like that at MLP 2 (Figure 6.8). The peak NO_3 mass was essentially reached on day 456 with 182 kg N ha^{-1} (Table 6.15), concurrent with the arrival of 1993 Cl. The NO_3-N mass remained practically constant in the next sampling round, after which the bottom of the 1992 water disappeared below the MLP. The mean of the two complete NO_3-N masses is 184 kg ha^{-1}.

At MLP 5, 1993 Cl showed up on day 416, which is when the peak NO_3-N mass of 144 kg ha^{-1} was observed (day 416, Table 6.16). In the next sampling round, the NO_3-N mass decreased to 133 kg ha^{-1}, after which the bottom of the 1992 water band descended below sampling depth. The mean of the two complete sums was 139 kg ha^{-1}.

The arrival of 1993 Cl at MLP 6 was obvious on day 458, but the mass of NO_3-N at that date was only 93 kg ha^{-1} (Table 6.17). The peak, 175 kg ha^{-1}, was not reached until day 592, more than four months later. Afterwards, the NO_3 mass decreased slowly and rather steadily. The bottom of the plume is difficult to track, but the NO_3-N mass remains reasonably constant through day 705, after which there was clear downward movement of the plumes. The mean of the NO_3-N masses from day 458 through day 705 is 142 kg ha^{-1}.

To summarize, the masses of NO_3-N found in 1992 water at the six MLPs were 190, 200, 216, 184, 139, and 142 kg ha^{-1} (Table 6.18). The mean of these amounts is 178 and the standard deviation is 31.

Table 6.12. Mass of NO_3-N at Field 2 MLP 1 for the 1992 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1992 N loading.

Sampling round	Date	Day	Screens	Depth range	NO_3-N, kg ha^{-1}
8	9/2/92	246	10 - 16	22.5 - 112.5	128
9	9/21/92	265	9 - 18	-40 - 127.5	186
10	10/23/92	297	8 - 17	7.5 - 142.5	122
11	12/2/92	337	8 - 18	-40 - 142.5	171
12	12/30/92	365	6 - 18	-40 - 172.5	**207**
13	2/19/93	416	4 - 17	7.5-205	**176**
14	3/31/93	456	2 - 16	22.5 - 245	**188**
16	7/12/93	559	0 - 14	52.5 - 300	161
17	8/14/93	591	0 - 8	142.5 - 300	106

Table 6.13. Mass of NO_3-N at Field 2 MLP 2 for the 1992 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1992 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
8	9/4/92	248	6 - 16	22.5 - 172.5	121
9	9/21/92	265	6 - 18	-40 - 172.5	**187**
10	10/23/92	297	2 - 18	-40 - 245	**218**
11	12/2/92	337	2 - 18	-40 - 245	**174**
12	12/29/92	364	2 - 18	-40 - 172.5	**221**
13	2/20/93	417	4 - 17	7.5 - 205	86
14	4/3/93	459	2 - 16	22.5 - 245	80
16	7/6/93	553	0 - 14	52.5 - 300	77
17	8/14/93	592	0 - 12	82.5 - 300	45

Table 6.14. Mass of NO_3-N at Field 2 MLP 3 for the 1992 water year by the method of N loading through a water year. Two possibilities are shown for round 12 due to the difficulty of identifying the bottom of the 1992 water. Bold numbers indicate numbers that appropriately estimate 1992 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
7	8/5/92	218	12 - 16	22.5 - 82.5	45
8	9/2/92	246	12 - 16	22.5 - 82.5	45
9	9/21/92	265	10 - 16	22.5 - 112.5	60
10	10/24/92	298	10 - 17	7.5 - 112.5	70
11	12/2/92	337	12 - 17	7.5 - 82.5	39
12	12/30/92	365	10 - 17	7.5 - 112.5	66
12 alternate	12/30/92	365	8 - 17	7.5 - 142.5	95
13	2/19/93	416	8 - 17	7.5 - 142.5	108
14	4/3/93	459	2 - 17	7.5 - 245	**216**
16	7/7/93	554	2 - 12	82.5 - 245	148
17	8/14/93	591	0 - 8	22.5 - 245	140

Table 6.15. Mass of NO_3-N at Field 2 MLP 4 for the 1992 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1992 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
6	7/14/92	196	12 - 17	7.5 - 82.5	56
7	8/3/92	216	12 - 16	22.5 - 82.5	54
8	9/2/92	246	12 - 16	22.5 - 82.5	45
9	9/22/92	266	12 - 18	-40 - 82.5	109
10	10/24/92	298	10 - 17	7.5 - 112.5	90
11	12/3/92	338	10 - 17	7.5 - 112.5	79
12	12/31/92	365	10 - 17	7.5 - 112.5	79
13	2/19/93	416	10 - 17	7.5 - 112.5	73
14	3/31/93	456	4 - 17	7.5 - 205	**182**
16	7/7/93	554	2 - 14	52.5 - 245	**186**
17	8/14/93	591	2 - 16	22.5 - 245	164

Table 6.16. Mass of NO_3-N at Field 2 MLP 5 for the 1992 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1992 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
5	7/1/92	183	12 - 17	7.5 - 82.5	60
6	7/13/92	195	12 - 17	7.5 - 82.5	57
7	8/5/92	218	10 - 16	22.5 - 112.5	88
8	9/2/92	246	8 - 12	82.5 - 142.5	63
9	9/22/92	266	11 - 18	-40 - 97.5	103
10	10/23/92	297	10 - 18	-40 - 112.5	112
11	12/3/92	338	6 - 17	7.5 - 172.5	125
12	12/31/92	365	6 - 17	7.5 - 172.5	130
13	2/19/93	416	4 - 16	22.5 - 205	**144**
14	4/3/93	459	4 - 16	22.5 - 205	**133**
16	7/7/93	554	1 - 8	142.5 - 265	73

Table 6.17. Mass of NO_3-N at Field 2 MLP 6 for the 1992 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1992 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
8	9/2/92	246	12 - 16	22.5 - 82.5	66
9	9/23/92	267	11 - 17	7.5 - 97.5	82
10	10/23/92	297	10 - 17	7.5 - 112.5	99
11	12/3/92	338	10 - 18	-40 - 112.5	121
12	12/31/92	365	10 - 18	-40 - 112.5	106
13	2/19/93	416	10 - 17	7.5 - 112.5	75
14	4/2/93	458	8 - 17	7.5 - 142.5	**93**
16	7/7/93	554	3 - 12	82.5 - 225	**131**
17	8/14/93	592	0 - 10	112.5 - 300	**175**
18	9/16/93	625	0 - 10	112.5 - 300	**166**
19	10/23/93	662	0 - 10	112.5 - 300	**144**
20	12/5/93	705	0 - 10	112.5 - 300	**144**
21	2/20/94	782	0 - 8	142.5 - 300	108

Table 6.18. Summary of 1992 N loading by the method of measuring N loading through a water year.

MLP	Date	N-loading estimate
1	12/30/92	207
	2/19/93	176
	3/31/93	188
	mean	190
2	9/21/92	187
	10/23/92	218
	12/2/92	174
	12/29/92	221
	mean	200
3	4/3/93	216
4	3/31/93	182
	7/7/93	186
	mean	184
5	2/19/93	144
	4/3/93	133
	mean	139
6	4/2/93	93
	7/7/93	131
	8/14/93	175
	9/16/93	166
	10/23/93	144
	12/5/93	144
	mean	142
1992 mean		178

1993 water year

Potato was grown in 1993. It was a year of repeated heavy rains and abnormal recharge. In some cases, the recharge from the early part of the year may have sunk below the sampled zone before the last of the 1993 water arrived at the uppermost samplers. Therefore, in some cases the peak mass of NO_3 can be used as an estimated lower limit for the amount of NO_3 leached during that water year.

Figure 6.8 shows 1993 Cl arriving at MLP 1 before day 500. The downward course of the pulse looks ambiguous after day 600, but examination of single concentration profiles indicated that the tongue of Cl at the bottom of the sampled zone shortly after day 600 probably was not 1993 Cl. (The interpolations performed by the contouring software were sometimes imperfect.) The arrival of 1994 Cl was apparent on days 803 and 844. Therefore, the peak NO_3-N mass of 253 kg ha^{-1} on day 752 and the two later masses, 143 kg ha^{-1} on day 803 and 201 kg ha^{-1} on day 844 (Table 6.19), are legitimate estimates of 1993 N loading. The mean of these three is 199 kg ha^{-1}.

At MLP 2 it is clear that 1994 Cl did not arrive at the top sampling screen until after 1993 recharge began leaving at the bottom. The peak mass of NO_3-N found was on day 706 (Table 6.20), 208 kg ha^{-1}. The mass remained close to that value in the next two sampling rounds. The mean of these three values, 197 kg N ha^{-1}, is the best estimate of NO_3 loading to groundwater at MLP 2.

The peak NO_3-N mass, 305 kg ha^{-1}, appeared at MLP 3 on day 843 (Table 6.21). Chloride from 1994 had not appeared in the uppermost screens at the end of sampling, so the actual mass of NO_3 in 1993 recharge must have been at least 305 kg N ha^{-1}.

The peak NO_3-N mass at MLP 4 also appeared on day 843 (Table 6.22) and was 320 kg ha^{-1}, close to that in MLP 3. Later sampling revealed that 1994 water had not yet appeared, so again the true amount of NO_3 in 1993 water may have been greater than indicated here.

At MLP 5, the mass of NO_3-N reached 252 kg ha^{-1} by the end of sampling (Table 6.23). This is again a possibly low estimate of NO_3 loading for 1993.

The situation at MLP 6 was much the same, with a peak NO_3-N mass of 288 kg ha^{-1} on day 844 (Table 6.24), and no indication of 1994 recharge by the end of sampling.

Summarizing 1993 (Table 6.25), the best estimates of NO_3-N loading for the six MLPs were 199, 197, 305, 320, 252, and 288 kg ha^{-1}. But only the first two values included all the recharge belonging to 1993. The mean of all observations was 260 kg ha^{-1} with a standard deviation of 53 kg ha^{-1}. The mean of the two complete observations was 198 kg ha^{-1}.

Table 6.19. Mass of NO_3-N at Field 2 MLP 1 for the 1993 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1993 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
16	7/12/93	559	14 - 18	-40 - 52.5	106
17	8/14/93	592	14 - 18	-40 - 52.5	105
18	9/17/93	626	14 - 18	-40 - 52.5	137
19	10/22/93	661	4 - 18	-40 - 205	207
20	12/4/93	704	2 - 18	-40 - 245	205
21	1/21/94	752	0 - 18	-40 - 300	**253**
22	3/13/94	803	0 - 16	22.5 - 300	**143**
23	4/23/94	844	0 - 16	22.5 - 300	**201**

Table 6.20. Mass of NO_3-N at Field 2 MLP 2 for the 1993 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1993 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
17	8/14/93	592	12 - 18	-40 - 82.5	98
18	9/17/93	626	could not identify bottom of water-year band		
19	10/20/93	658	4 - 18	-40 - 205	174
20	12/6/93	706	0 - 18	-40 - 300	**208**
21	1/12/94	743	0 - 18	-40 - 300	**198**
22	3/19/94	809	0 - 18	-40 - 300	**184**
23	4/23/94	844	0 - 17	7.5 - 300	90

Table 6.21. Mass of NO_3-N at Field 2 MLP 3 for the 1993 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1993 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
16	7/7/93	554	12 -18	-40 - 82.5	72
17	8/14/93	592	8 - 18	-40 - 205	117
18	9/16/93	625	6 - 18	-40 - 172.5	161
19	11/3/93	673	4 - 18	-40 - 205	175
20	12/5/93	705	4 - 18	-40 - 205	180
21	1/13/94	744	0 - 18	-40 - 300	239
22	3/13/94	803	0 - 18	-40 - 300	300
23	4/22/94	843	0 - 20	-50 - 300	**305**

Table 6.22. Mass of NO_3-N at Field 2 MLP 4 for the 1993 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1993 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
16	7/7/93	554	14 - 18	-40 - 52.5	59
17	8/14/93	592	6 - 16	22.5 - 172.5	108
18	9/16/93	625	4 - 16	22.5 - 205	131
19	10/27/93	666	1 - 18	-40 - 265	227
20	12/5/93	705	1 - 18	-40 - 265	243
21	1/13/94	744	0 - 18	-40 - 300	267
22	3/13/94	803	0 - 18	-40 - 300	308
23	4/22/94	843	0 - 20	-50 - 300	**320**

Table 6.23. Mass of NO_3-N at Field 2 MLP 5 for the 1993 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1993 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
14	4/3/93	459	16 - 17	7.5 - 22.5	8
16	7/7/93	554	8 - 18	-40 - 142.5	105
17	8/14/93	592	6 - 18	-40 - 172.5	118
18	9/16/93	625	4 - 18	-40 - 205	133
19	10/27/93	666	2 - 18	-40 - 245	203
20	12/5/93	705	1 - 18	-40 - 265	204
21	2/21/94	783	0 - 18	-40 - 300	224
22	3/20/94	809	0 - 18	-40 - 300	235
23	4/24/94	844	0 - 20	-50 - 300	**252**

Table 6.24. Mass of NO_3-N at Field 2 MLP 6 for the 1993 water year by the method of N loading through a water year. Bold numbers indicate numbers that appropriately estimate 1993 N loading.

Sampling round	Date	Julian day	Screens	Depth range	NO_3-N, kg ha^{-1}
16	7/8/93	554	12 - 18	-40 - 82.5	74
17	8/14/93	592	10 - 18	-40 - 112.5	107
18	9/16/93	625	10 - 18	-40 - 112.5	125
19	10/23/93	662	10 - 18	-40 - 112.5	104
20	12/5/93	705	10 - 18	-40 - 112.5	113
21	2/20/94	782	8 - 18	-40 - 142.5	119
22	3/13/94	803	6 - 18	-40 - 172.5	148
23	4/24/94	844	2 - 20	-50 - 245	**288**

Table 6.25. Summary of 1993 N loading by the method of measuring N loading through a water year.

MLP	Date	N-loading estimate
1	1/21/94	253
	3/13/94	143
	4/23/94	201
	mean	199
2	12/6/93	208
	1/12/94	198
	3/19/94	184
	mean	197
3	4/22/94	305
4	4/22/94	320
5	4/24/94	252
6	4/24/94	288
1993 mean		260
Mean of complete water year, evident only at MLPs 1 and 2		198
Mean in north half of field		198
Mean in south half of field		291

The north and south halves of Field 2 received different amounts of N fertilizer in 1993 (Table 6.2). The north half, which contained MLPs 1 and 2, received 297 kg N ha^{-1} compared to 356 kg N ha^{-1} in the south half, which contained MLPs 3-6. The mean NO_3 loading for the two north MLPs was 198 kg N ha^{-1}, and in the four south MLPs it was 291 kg N ha^{-1}. The greater N loading found in the south MLPs was in spite of

the fact that the north MLPs showed complete 1993 recharge profiles and all the south-half profiles were incomplete. The difference in N loading corresponds to the different amounts of N fertilizer applied.

NITRATE LOADING IV: FERTILIZER-N LEACHING MODEL

A nitrogen leaching model was constructed to determine how much fertilizer N leaches to groundwater during the year of application. Fertilizer contributed about 85% of the N inputs to the study fields.

Leaching of N (primarily NO_3) to groundwater is a complex process involving biochemical transformations among different forms of N, uptake of N by plants or microorganisms, and nonuniform, unsaturated soil water transport. If all of the processes are understood, and if the data needed to characterize the system are available, it is possible in principle to design a comprehensive mathematical model that could simulate NO_3 leaching and describe where, when, and how much NO_3 will enter the groundwater under a set of agricultural practices. In practice, the processes affecting NO_3 leaching (especially the rates of biological reactions) are not fully understood, and the complete data needed to characterize the system are not available. However, a less-than-comprehensive model with much smaller data requirements could provide a less exact but still informative picture of NO_3 leaching under specified conditions. Even if the results are inaccurate, the model can be helpful in identifying what factors affect leaching, and can give an idea of which are most critical (Anderson and Woessner, 1992, p. 4).

The model PRZM (Pesticide Root Zone Model, Mullins et al., 1994) was selected to evaluate the feasibility of using such a simplified model to simulate NO_3 leaching under irrigated vegetable fields in the Port Edwards Groundwater Priority Watershed. PRZM is a one-dimensional computer model of water movement and chemical behavior in soil. It was designed to model pesticide transport in and immediately below the root zone. The movement of NO_3 can be handled by the same processes that are used for pesticides.

Methods

Water transport

PRZM uses a simple and non-mechanistic method to track water movement in soil. The soil is divided into layers (called compartments) whose thickness is chosen by the user. Adjacent compartments can be grouped into horizons, and each horizon is assigned a field moisture capacity and wilting point. Daily water inputs and outputs are computed for each compartment. If the resulting water content is greater than the field moisture capacity, then the excess water passes to the layer below. Layers are not allowed to lose water once they have reached the wilting point. The water-movement computation proceeds from the top down, so that a large application of water at the surface could traverse the entire modeled profile in one day. If the water content is between the wilting point and the field capacity, water is available for evaporation and transpiration

(ET). PRZM's authors cite Stewart et al. (1976) to the effect that this water-movement scheme is most accurate for sandy soils (such as the soil in the project area) and least accurate for fine-textured soils.

Plant roots are considered to extract moisture from compartments following a "triangular distribution", so that water is most easily removed from the top compartment, and progressively less easily from lower compartments. Roots grow linearly from the crop emergence date to the maturity date, and can extract water down to the bottom of the current day's root depth.

PRZM calculates evaporation of rain intercepted by the crop canopy, soil evaporation, and plant transpiration. Evaporative demand is based preferably on pan evaporation input. (If pan evaporation data are not given, PRZM uses an empirical formula based on other weather inputs.)

Daily pan evaporation was supplied as an input to PRZM. It was calculated from solar radiation and temperature data from automated weather stations near the study area. A station was installed in the study area in June 1992, and data from this station were used whenever possible. When data were missing, records from the nearby Cranmoor station were used, and if these too were missing, records from the Hancock experimental farm were used. These stations measure precipitation with a tipping bucket gauge, which is not accurate in winter. For the months of November through March, precipitation records from the automated stations were corrected by a factor which forced each monthly total to agree with a total from Hancock that was quality-controlled by the US National Oceanographic and Atmospheric Administration (NOAA). That is, if the total precipitation recorded at Nekoosa in January was 0.4 cm, while the NOAA reported 0.6 cm at Hancock, then the Nekoosa precipitation for each day in January would be multiplied by 1.5. The Priestley-Taylor equation we used to estimate pan evaporation was (Monteith and Unsworth, 1990, p. 196),

$$\lambda E = \alpha \frac{\Delta R_n}{\Delta + \gamma},$$

where λ = heat of vaporization of water, evaluated each day at the average temperature; joule $gram^{-1}$

E = evaporation flux of water; gram m^{-2}

α = proportionality constant. Bland (1994, pers. commun.) recommended 1.28 for Wisconsin (Priestley and Taylor used 1.26); dimensionless

Δ = slope of the graph of saturation water vapor pressure with temperature, evaluated each day at the average temperature; pascal K^{-1}

R_n = net effective solar radiation, taken as measured solar radiation multiplied by (1 - albedo); joule m^{-2}

γ = psychrometer constant, evaluated each day at the average temperature; pascal K^{-1}.

In order to test the reasonableness of PRZM's evaporation calculations, PRZM results were compared to those of the Wisconsin Irrigation Scheduling Program (WISP; Curwen and Massie, 1990). WISP is designed as an aid for farmers. It calculates a soil water budget similar in concept to that of PRZM, but with a simpler model of soil water processes. Because WISP is only intended for use with an actively growing crop, it does not estimate evaporation from bare soil. WISP cannot be used to estimate evaporation during the months outside the growing season.

To extend our comparison outside the growing season, we estimated evaporation by adapting a FORTRAN program of Ritchie (1972), which models soil-surface evaporation based on precipitation, temperature, solar radiation, and surface albedo. The results of the comparison for the two-year modeling period are shown in Table 6.26. In 1992, PRZM calculated about the same off-season evaporation as the Ritchie method did, and 32% more ET than WISP did during the growing season. In 1993, calculations during the growing season were similar for the two models, but PRZM calculated 40% less evaporation outside the growing season.

Table 6.26. Comparison of water balance as estimated by PRZM and by the WISP + Ritchie method. "Net precipitation" includes irrigation. Small changes in profile wetness caused ET + recharge to differ slightly from net precipitation. All values are in cm.

			Evaporation + transpiration				
	Net precipitation	Method	Growing-season	Off-season	Total	Recharge	
1992	93.18	WISP + R.	25.65	20.34	45.99	45.88	18
	36.6	PRZM	33.81	18.98	52.79	39.08	15
1993	111.89	WISP + R.	25.22	25.14	50.36	61.53	24
	44	PRZM	26.86	15.17	42.03	69.83	27.1

The point of the water budget was to estimate movement of water out the bottom of the root zone. PRZM calculated 15% less recharge than did extended WISP in 1992, and 13% more in 1993. We feel that this was a good degree of agreement, considering the differences in the two methods, and indicated that PRZM was generating reasonable output.

Nitrate Movement

The model assumed that all fertilizer N arrives as, or is promptly converted to, NO_3. We modeled NO_3 as a conservative solute. Water that moved into roots carried NO_3 along with it; the root was passive with respect to NO_3 uptake. This is roughly consistent with Barber (1984, p. 96), where 80% of NO_3 uptake by corn roots occurs by mass flow. We also assumed the fields were at a steady state with respect to N

storage in the soil. Mineralization of soil organic N is an important process, but in a field at steady state, overall mineralization is balanced by immobilization, so there is no net input from soil organic matter. The amount of fertilizer NO_3 remaining in the simulated profile on 31 Dec. of both years was negligible.

Soil and Crop Parameters

Five cm was the thickness chosen for soil layers. This thickness was recommended by the program authors, with the suggestion that numerical dispersion would approximate typical values of hydrodynamic dispersion. Sixty layers gave a total profile thickness of 300 cm. The top 30 cm was modeled as the Ap horizon, with a field moisture capacity (FMC) of 8.9% (all moisture contents are volumetric) and wilting point (WP) of 5.5%. The B horizon comprised the 20 cm of layers 7 through 10, with FMC of 8.2% and WP of 3.5%. The C horizon was the last 250 cm; its FMC was 5.3% and its WP was 3.0%.

The simulation was designed to mimic agricultural practices for 1992 and 1993 in Field 2. Field 2 was chosen partly because better irrigation records were available than for Fields 3 and 4, and the 1993 flooding was especially severe in Field 1. Field 2 also had problems with flooding that year, and fertilizer was applied differently to the north and south halves of this field because of difficulty gaining access to the field after some heavy rains. Therefore, both of the fertilizer regimes used in 1993 were modeled.

In 1992, sweet corn was planted. The emergence date used by PRZM was May 28, and the maturity and harvest dates were both set at Oct. 8. The mature root depth was set at 150 cm (Jackson et al., 1987). Irrigation was applied 13 times, totaling 15 cm of water, and N-fertilizer timing and amounts are shown in Figure 6.9.

The 1993 crop was potato. Its emergence date was set at May 10, maturity was Sept. 9, when the vines were killed, and harvest was Sept. 30. Although rainfall was excessive throughout most of the growing season, irrigation was used to apply chemicals. Water was applied 12 times, totaling 13 cm. We selected 60 cm as the mature root depth for potato (Jackson et al., 1987). Nitrogen-fertilizer applications for 1993 are also shown in Figure 6.9.

Results

Nitrogen Loading

For the 1992 sweet corn crop, the PRZM simulation indicated 42 kg ha^{-1} of fertilizer N leached to groundwater (Table 6.27) and 208 hg ha^{-1} was taken up by the plant. The 42 kg ha^{-1} is substantially less than the estimates made by other methods in this chapter, because the simulation only considered fertilizer N and ignored other sources of potentially leachable N (crop residue, organic matter mineralization, etc.). Subtracting the N removed in harvest, 110 kg ha^{-1} (Table 6.9), leaves 98 kg ha^{-1} of residue N that is potentially

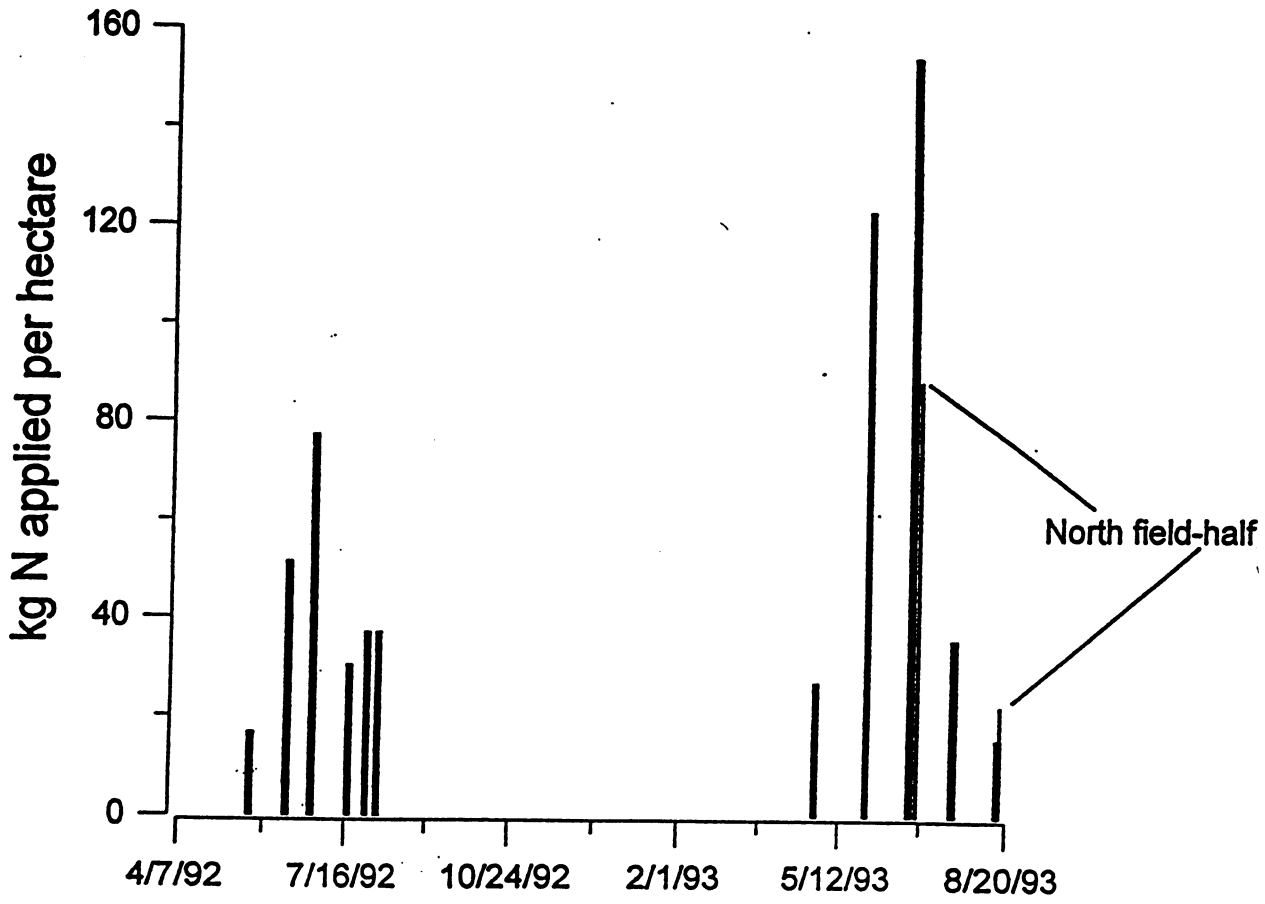


Figure 6.9 Nitrogen fertilizer applications to Field 2 in 1992 and 1993.

leachable. The sum of PRZM's leached fertilizer N and potentially-leachable residue N is 140 kg ha^{-1}, or 56% of applied fertilizer N.

Table 6.27. Nitrogen loading from PRZM simulations with selected values for mature rooting depths and soil water-holding capacity.

	Crop		N applied ($kg\ ha^{-1}$)	Fertilizer N leached	
				$kg\ ha^{-1}$	%
1992	Sweet Corn		280	42 140[1]	15 56[1]
1993	Potato	N½	336	193	57
		S½	395	216	55

[1]Including potentially leachable crop residue N

For the 1993 potato crop, the PRZM-simulated fertilizer NO_3-N loading was 193 (north field-half) or 216 (south) kg ha^{-1}. Crop N uptake calculated by PRZM was 103 (north) or 140 (south) kg ha^{-1}. This is somewhat less than the 180 kg ha^{-1} estimate of harvested N which was presented in the N budget earlier in this chapter. The discrepancy between PRZM-simulated N uptake and estimated harvested N may indicate that the harvested potato N concentration in the N budget was too large. We used 4.0 g N kg^{-1}, an average reported by Meisinger and Randall (1991) for a range of 3-5 g N kg^{-1}. However, Saffigna et al. (1977) reported 2.8 g N kg^{-1} for potato in Wisconsin. Perhaps a lower number for harvested N in potato would be more appropriate in the N budget.

Sensitivity Analyses

Sensitivity analyses were conducted to explore the effects of some parameters that were not known with great certainty, to wit rooting depths, soil moisture-holding capacities (FMC and WP), and initial (i.e. 1 Jan. 1992) soil moisture content. We expected that the appropriate initial soil wetness would be FMC throughout the profile, since Wisconsin tends to have rainy, cool autumns. To confirm this, PRZM was run with a very dry profile (drier than WP) on 1 Jan. 1992 and again with a wet profile (wetter than FMC). In both cases, the soil was at FMC throughout on 1 Jan. 1993. We took this as confirmation that FMC is the normal condition for Wisconsin on Jan. 1 as modeled in PRZM.

The mature corn root depth was varied from 40 cm to 240 cm while other input parameters were kept constant. Likewise, potato mature root depth was varied from 20 cm to 120 cm. The effect on recharge is shown in Figure 6.10. At the selected corn rooting depth of 150 cm, recharge was 39.1 cm. Reducing mature rooting depth to 40 cm led to recharge of 47.9 cm. Increasing the mature root depth to 240 cm brought

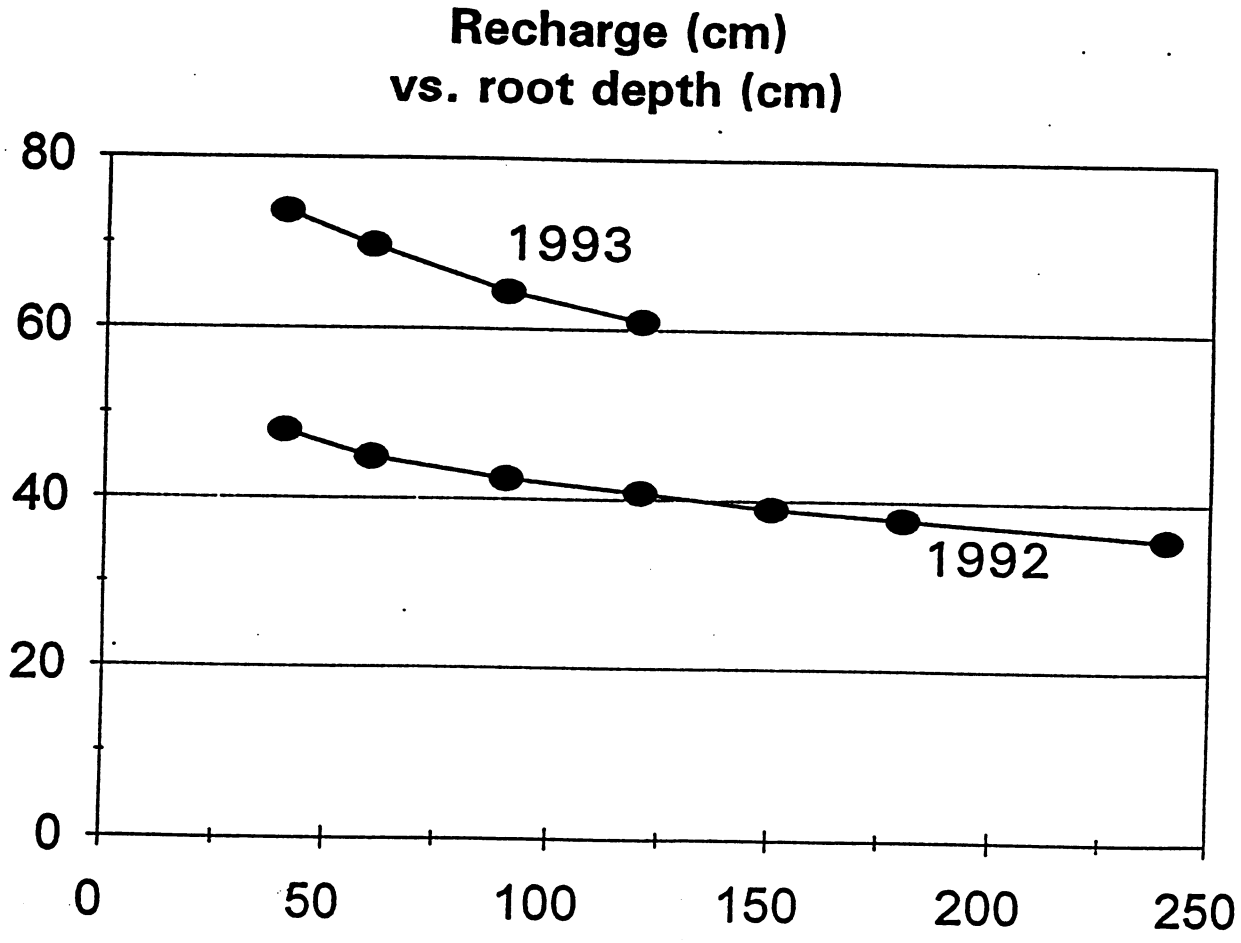


Figure 6.10 Effect of crop mature root depth on simulated recharge calculated by the PRZM model.

recharge to 35.8 cm. Thus as corn root depth increased by a factor of 6, recharge decreased by a factor of 1.34. For potato, PRZM calculated 61.0 cm of recharge with a 60-cm mature root depth, and recharge decreased to 54.1 cm with a mature root depth of 120 cm, or increased to 73.7 cm with a mature root depth of 20 cm. As potato mature root depth increased by a factor of 6, recharge decreased by a factor of 1.36.

The effect of changing crop root depth on NO_3 loading is shown in Figure 6.11. As corn mature root depth increased from 40 cm to 240 cm (sixfold), NO_3 loading decreased by a factor of 3.1, but most of the variation in NO_3 loading is at unrealistically small mature root depths, and there is very little change in the range from 90 cm to 240 cm mature root depth. The fact that N was applied at the surface, and plants are modeled as taking water preferentially from the upper part of the soil, tends to minimize the sensitivity of the NO_3 leaching calculation to rooting depth. Calculated NO_3 loading decreased by a factor of 1.75 (from 277 to 158 kg ha^{-1}) as potato mature root depth increased from 20 cm to 120 cm.

Soil water-holding capacity was varied by changing both FMC and WP in concert; their values were increased or decreased by 25%. The selected values and the high- and low-capacity values are shown in Table 6.28. Increasing soil water-holding capacity by 25% caused calculated recharge to decrease by 15% in 1992 and by 7% in 1993 (Figure 6.12). Decreasing water-holding capacity 25% increased recharge by 16% in 1992 and 10% in 1993.

Table 6.28. Field moisture capacity and wilting point values in A, B, and C horizons used in sensitivity analysis, with recharge amounts calculated by PRZM.

Soil water-holding capacity	FMC (v/v)	WP (v/v)	1992 recharge (cm)	1993 recharge (cm)
Selected	A 0.089 B 0.082 C 0.053	A 0.055 B 0.035 C 0.030	39.1	61.0
High	A 0.111 B 0.102 C 0.066	A 0.069 B 0.044 C 0.038	33.4	56.6
Low	A 0.067 B 0.062 C 0.040	A 0.041 B 0.026 C 0.022	45.4	66.8

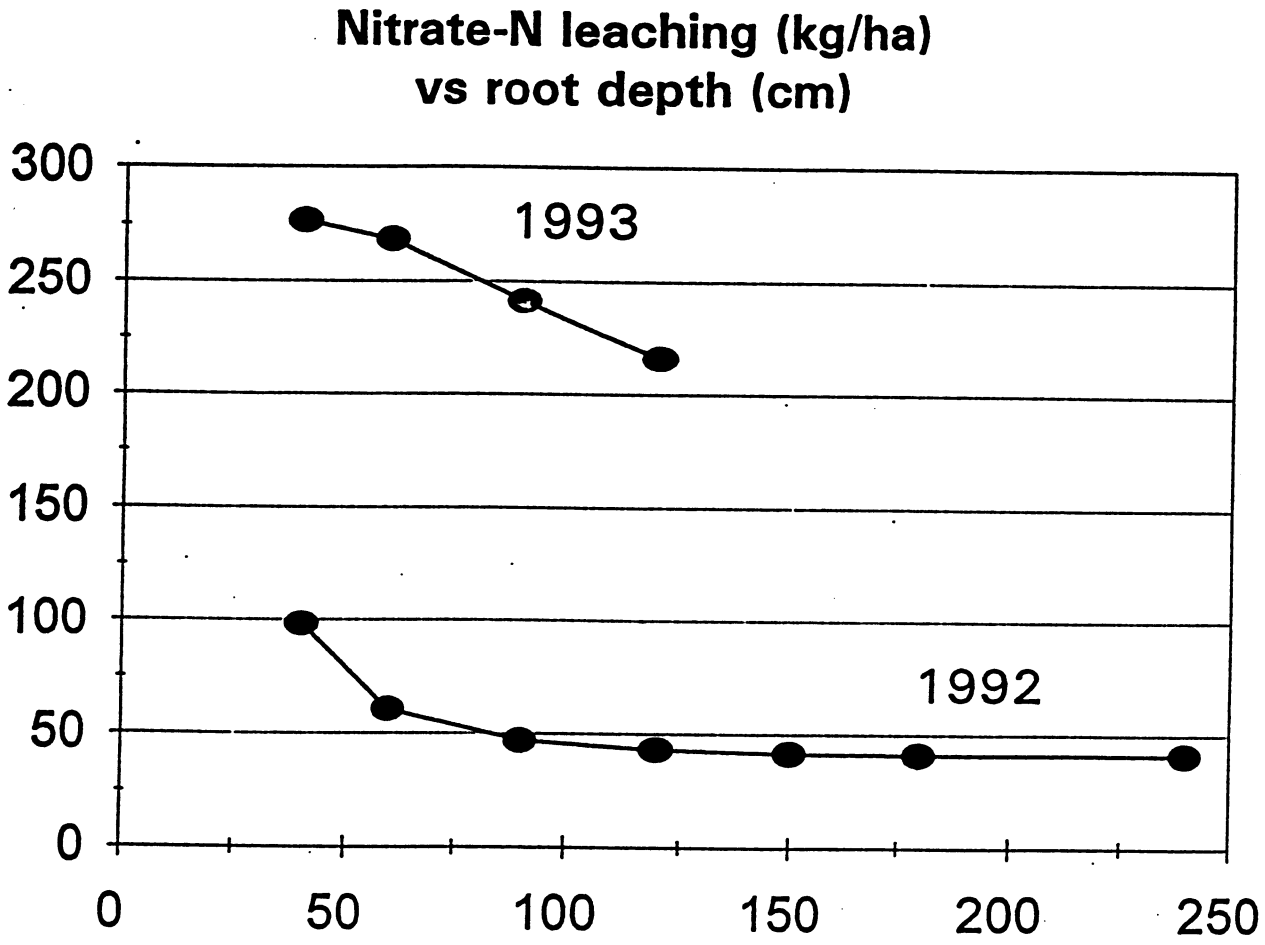


Figure 6.11 Effect of crop mature root depth on simulated nitrate leaching calculated by the PRZM model.

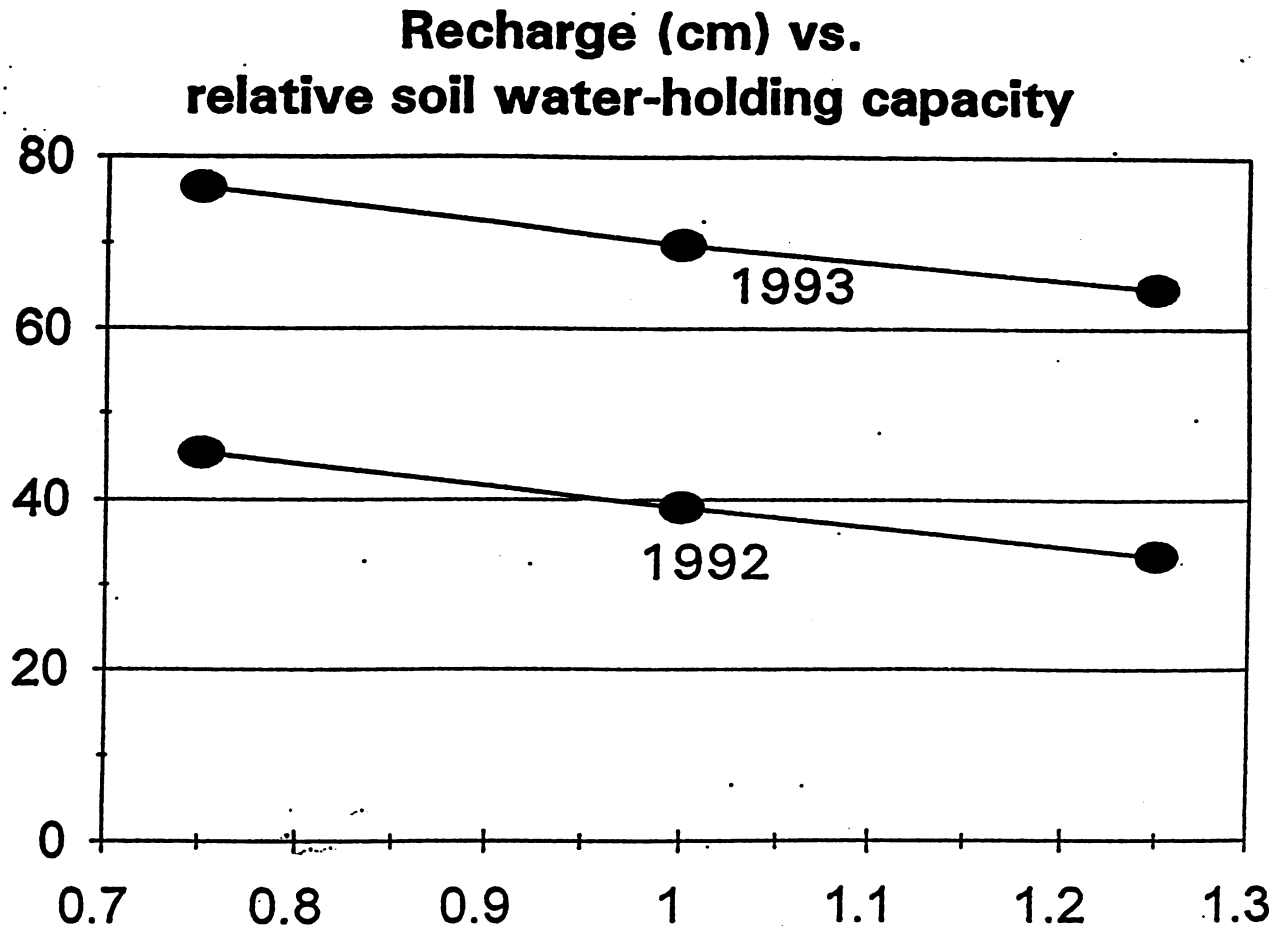


Figure 6.12 Effect of soil water-holding capacity on simulated recharge calculated by the PRZM model.

Table 6.29 and Figure 6.13 show the effect of varying soil water-holding capacity on the PRZM estimate of NO_3 loading in 1992 and 1993. The water-holding capacity was varied through a range from 0.75 to 1.25 times the selected value, which led to a change in calculated NO_3 loading of only 1-2% in 1992, but 10-11% in 1993.

Under the weather conditions during the period simulated, both recharge and NO_3 leaching were not very sensitive to crop root depth, but somewhat more sensitive to soil properties within the range considered. Simulated recharge was relatively insensitive to rooting depth, because in PRZM, even the deepest root zones transmit all water in excess of FMC in one day. Irrigation kept the profile close to FMC throughout the growing season.

Table 6.29. Sensitivity of NO_3 loading calculation to soil water-holding capacity. Nitrate amounts are kg N ha^{-1}.

Soil water-holding capacity	1992 NO_3 loading	1993 NO_3 loading
Selected	41.6	215.9
High	40.7	193.8
Low	42.4	240.0

CONCLUSIONS

Inorganic groundwater quality

In-field vs. upgradient groundwater quality differed markedly in the upper 3 m of the saturated zone. Groundwater was slightly more acidic at in-field MLPs (5.0 vs. 5.4) and solute content was greater. Nitrate, Cl, Na, K, Ca, Mg, and Al concentrations were substantially greater at in-field MLPs compared to upgradient MLPs, but Fe concentrations were less. The average NO_3-N concentration for MLPs within the four fields was 22.4 mg L^{-1}, compared to 1.4 mg L^{-1} upgradient.

Pesticide monitoring

Pesticides were detected at all in-field MLPs. The pesticides found were metolachlor, metribuzin, alachlor, carbofuran, and atrazine with its de-ethyl and de-isopropyl metabolites. Atrazine exceeded Wisconsin's preventive action limit (PAL) more frequently than the other pesticides. Metolachlor, carbofuran, and alachlor also exceeded their PALs at a few MLPs. Atrazine exceeded the enforcement standard (ES) at three MLPs, metolachlor at one.

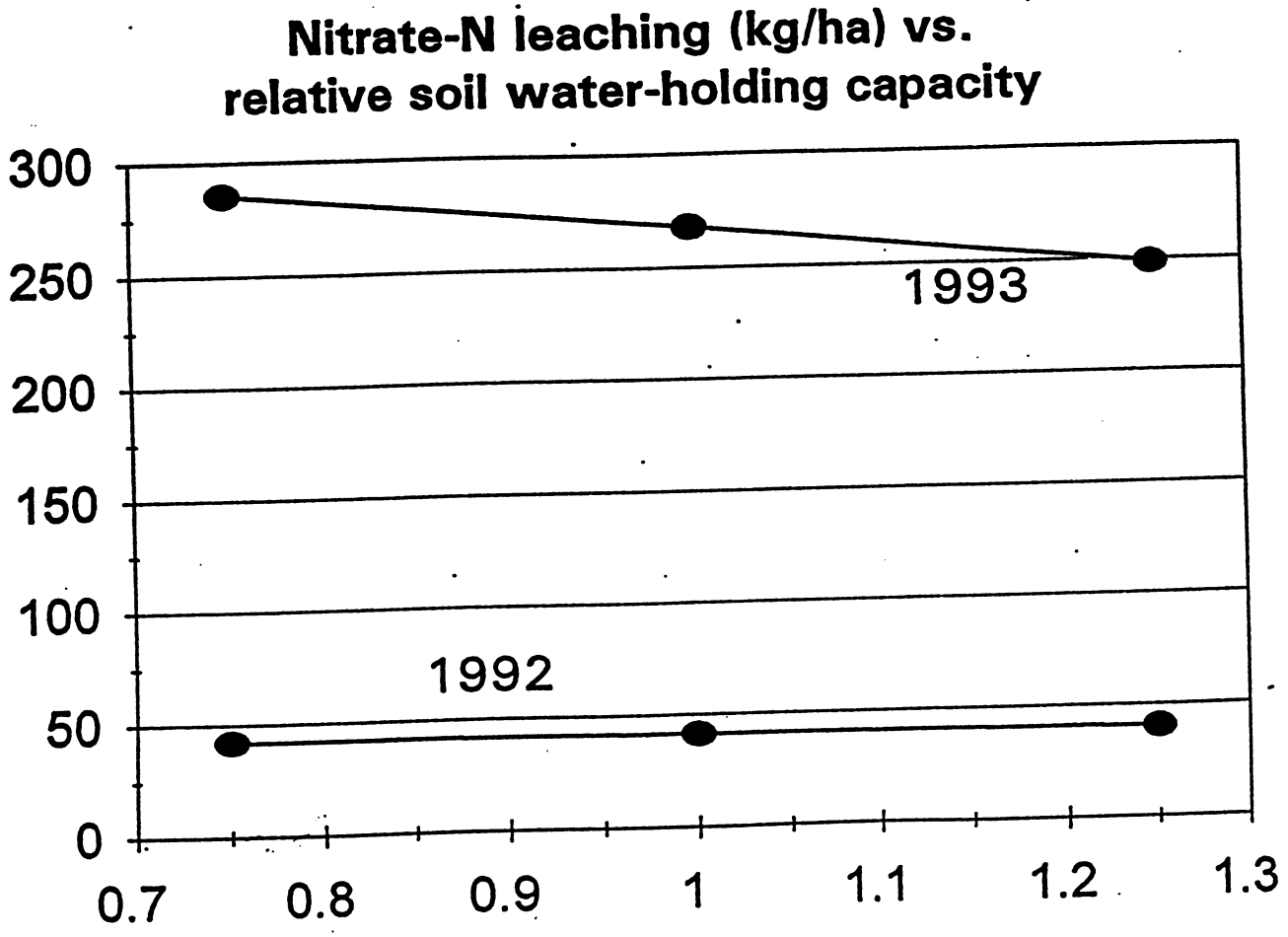


Figure 6.13 Effect of soil water-holding capacity on simulated nitrate leaching calculated by the PRZM model.

Atrazine was applied only to the east half of Field 3 in both 1992 and 1993. Atrazine and de-ethyl atrazine were detected in more MLP ports in the east half than in the west half. Atrazine and residue concentrations were significantly less in the west half, strongly suggesting that stopping atrazine application improved groundwater quality.

Patterns of solute plume arrival and vertical advection.

We observed the breakthrough of solute plumes at the water table on roughly annual cycles at most MLPs. Cl peaks and minima trended downward with time; for NO_3, the patterns were less distinct. Pesticides seemed to show similar behavior, but the patterns are more obscure due to less frequent sampling.

Nitrate loading estimates

Potentially leachable N from N budget

Potentially leachable N ranged from 133 to 237 kg ha^{-1}, or 40-82% of total N input (fertilizer, microbial fixation, irrigation water, precipitation, dry deposition, mineralization and crop seed). The greatest leachable N percentages were found for sweet corn in Fields 3 and 4 in both years, due to poor yields and meager harvested N output. The greatest amounts of leachable N were found with potato in 1993 on Field 1 and the south half of Field 2, primarily because excessive rains compelled heavy N fertilizer applications.

Simple calculation from NO_3-N concentration and recharge

Simple estimates of NO_3 loading calculated from the average NO_3-N concentration in shallow groundwater below the four fields and estimated recharge for the two years were 76-95 kg ha^{-1} in 1992 and 97-122 kg ha^{-1} in 1993. For the two years and four fields, the estimated annual loading was 112 kg ha^{-1}, 42% of total N input.

A statistically significant 5.4 mg L^{-1} reduction in NO_3-N in groundwater was observed on the southwest half of Field 4, which received 15 kg ha^{-1} less fertilizer N in 1993. Calculating N loading from NO_3 concentration and recharge, the 15 kg ha^{-1} reduction in input appeared to result in a 33 kg ha^{-1} decrease in N loading to groundwater. The overestimate probably reflects imprecision in concentration and recharge estimates.

Loading through a water year

Detailed concentration profiles were analyzed to quantify annual NO_3 loading to groundwater beneath Field 2. Estimated NO_3-N masses loaded to groundwater in the 1992 water year at the six MLPs were 190, 200, 216, 184, 139, and 142 kg ha^{-1}. The mean was 178 and the standard deviation was 31.

For the 1993 water year, the best estimates of NO_3-N loading for the six MLPs were 199, 197, 305, 320, 252, and 288 kg ha^{-1}. The mean estimate was 260 with a standard deviation of 53. The mean of the two complete observations was 198.

The north half of Field 2 received 60 kg ha^{-1} less N fertilizer than the south half in 1993. With this method, the mean NO_3-N loading for the two north MLPs was 93 kg ha^{-1} less than in the four south MLPs.

Fertilizer-N leaching model

A nitrogen leaching model was constructed using the computer code PRZM for Field 2 to simulate fertilizer N leaching to groundwater as NO_3 and to evaluate the ability of a simple model to simulate NO_3 leaching. All N was assumed to be in the NO_3 form and to move conservatively with water. These assumptions are close to reality. The model ignored all N inputs other than fertilizer. Fertilizer provided 79-87% of total N input, according to the N budget.

For the 1992 sweet corn crop, the PRZM simulation indicated 42 kg ha^{-1} of NO_3-N leached to groundwater out of 250 kg ha^{-1} fertilizer N. This was less than the estimates of N loading by other methods, because only direct leaching of fertilizer N was considered in the model. PRZM's calculation of 208 kg ha^{-1} crop N uptake, combined with data on N removed in harvest, indicated that 98 kg ha^{-1} of potentially leachable residue N remained on the field. The sum of PRZM's leached fertilizer N and potentially leachable residue N is 140 kg ha^{-1} of potentially leachable N originating as fertilizer, or 56% of the applied fertilizer N.

For the 1993 potato crop, the PRZM-simulated fertilizer NO_3-N loading was 193 (north field-half) or 216 (south) kg ha^{-1}. Residue N was not important with potato.

Sensitivity analyses explored the effects of rooting depths, soil moisture-holding capacities, and initial soil moisture content. Field moisture capacity was confirmed as the normal initial soil moisture condition on Jan. 1 as modeled in PRZM.

Recharge and NO_3 loading were fairly insensitive to crop root depth, because when rainfall occurs, PRZM simulates transport of any water in excess of FMC past the root zone in one day. Irrigation kept the profile close to FMC throughout the growing season, so the soil's ability to store rainfall was limited.

The model was more sensitive to soil water-holding capacity. Varying the soil water-holding capacity by 25% changed calculated recharge by 15-16% in 1992 and 7-10% in 1993. The same change in water-holding capacity led to a change in calculated NO_3 loading of only 1-2% in 1992, but 10-11% in 1993.

Comparison of methods for N loading

The four methods of estimating NO_3-N loading (Table 6.30) evaluate different types of loading . The N budget and the method of N loading through a water year both evaluated the total amount of NO_3-N that

leaches to groundwater. Therefore, they should be comparable, at least in soil/groundwater systems that respond rapidly, such as the study site. The simple method based on NO_3-N concentrations and recharge also evaluated the total amount of NO_3-N that leaches to groundwater, but is more appropriately used as a longer-term average and cannot evaluate NO_3 loaded to groundwater from a given growing season with much certainty. The PRZM model only considered direct leaching of fertilizer N, and ignored other N inputs. This is not a serious deficiency, since fertilizer constituted 79-87% of total N input. However, the simulation did not

Table 6.30. Nitrate-N loading (potential loading in the N budget method) in irrigated fields in the Port Edwards Groundwater Priority Watershed as determined by four methods. N, S, NE, and SW refer to field-halves.

				NO_3-N leached	
Method	Crop	Year	Field	kg ha^{-1}	% of N input
N budget	Sweet corn	1992	1	165	64
			2	178	60
			3	178	78
		1993	3	192	82
			4	178	79
	Potato	1992	4NE	143	43
			4SW	133	40
		1993	1	237	61
			2N	164	47
			2S	224	55
Average NO_3-N concentration and recharge	Sw. corn, potato[1]	1992-93	all	122	42
N loading through a water year	Sweet corn	1992	2	178	60
	Potato	1993	2N	198	57
			2S	291	71
					% of fertilizer N
PRZM model	Sweet corn	1992	2	42	17
				140[2]	56[2]
	Potato	1993	2N	193	65
			2S	216	61

[1]Four fields, two years; three field-years of potato and five of sweet corn.
[2]Includes potentially leachable crop residue N.

account for potentially leachable N in crop residue, and predicted much less NO_3 leaching than the other methods. Adding in the crop-residue N made the PRZM simulation comparable to the other methods.

Loading estimates by the four methods agree fairly well, except for the low estimates given by the method based on average NO_3-N concentrations and recharge. Nitrate concentrations in the lower part of the sampled zone may be diminished by mixing with groundwater that originated upgradient from the fields, leading to artificially low averages.

Comparisons between the other three methods are shown in Table 6.31 for Field 2. The method based on NO_3-N concentrations and recharge is excluded because partitioning of observations between the two crop management years (or water years) is not possible. The N budget and the water year method agreed perfectly for sweet corn in 1992. The N leaching model indicated less N loading to groundwater, but this would be expected since it did not consider N sources other than fertilizer. The percentage of N input that leached was nearly identical for all three methods. For the 1993 potato crop, the comparisons differ between field halves. On the north half, where total N input was 351 kg ha^{-1}, the water year and the leaching model results were quite similar, even though the leaching model ignores non-fertilizer N inputs. The N budget predicted 30-34 kg ha^{-1} less NO_3-N loading. On the south half, which received an additional 60 kg ha^{-1} of fertilizer N, the N budget estimate is again low compared to the water year method. The low estimates given by the N budget method could indicate that too large a value was used for the N concentration in harvested potato. That possibility was also suggested by the fact that the harvested N used in the budget was greater

Table 6.31. N loading to groundwater as estimated by three methods for Field 2. The N budget predicts potentially leachable N from total inputs, and the leaching model only considers fertilizer N inputs. (N) and (S) refer to field-halves.

	Crop and year			
	Sweet Corn, 1992		Potato, 1993	
Method	Percent of N input	kg N ha^{-1}	Percent of N input	kg N ha^{-1}
N budget (potentially leachable)	60	178	47 (N) 55 (S)	164 (N) 224 (S)
NO_3 loading through a water year	60	178	57 (N) 71 (S)	198 (N) 291 (S)
Leaching model (percent of fertilizer N)[1]	56	140	65 (N) 61 (S)	193 (N) 216 (S)

[1] Includes crop residue N

than the total crop uptake calculated in the PRZM simulation. In the south half, the leaching model predicted less N loaded to groundwater than the water year method, as would be expected.

Efficacy of model

The PRZM model gave results for potato that agreed reasonably well with the other methods. The sweet corn results approached those of the other methods only when additional data (on crop residue N) were considered. PRZM provided useful results with a modest requirement for input data. Results of the sensitivity analysis suggest that a more exact representation of soil water movement could help refine the N leaching calculations, because model results were somewhat sensitive to soil properties.

Comparison of N loading results to literature

The most appropriate available studies for comparison to our sweet corn results concern field corn. Tyler and Thomas (1977, Figure 2) show that with 168 kg ha^{-1} fertilizer N, leaching losses in a Kentucky silt loam were 55-65 kg ha^{-1} (33-39% of applied N) in the second year of corn . Leaching was less during the first year because N was being stored in the soil profile. Reddy and Reddy (1993) found that N losses in a sandy loam soil increase greatly as N application rates increase. With fertilizer applications of 200 kg N ha^{-1}, crop uptake was 45% (including unharvested portions), 15% of the applied N remained in the soil, and 40% was not recovered – most likely it leached below the root zone. The Reddy and Reddy study was a ^{15}N experiment, so only the N added in the experiment was measured. With soil N at steady state, the 15% of ^{15}N remaining in the soil would be offset by mineralization of previously stored N, and the entire 55% of N not taken up by the crop would be available for leaching. Crop residues remaining in the field could further increase the leaching potential as they decay and release N. Fermanich et al. (1993) report NO_3-N loading below the root zone from May through September of 58 kg ha^{-1} (29% of applied N) from field corn at Arena, Wisconsin. Nitrogen leaching was very sensitive to irrigation treatments, and the 58 kg ha^{-1} was measured with an irrigation treatment that caused 3/4 as much recharge as PRZM calculated in Field 2. To be compared with our results (Table 6.31), their crop-residue N and N leaching after the growing season (not known) should be included.

With potato, Milburn et al. (1990) report 2.5% to 22% of fertilizer N in tile drainage water with N application rates from 110 to 200 kg ha^{-1}. They point out that some N may have escaped the drainage system. Also, the presence of tile drains suggests that denitrification may have occurred. Perillo et al. (1993) present data based on suction samplers showing NO_3-N loading of about 75 kg N ha^{-1}, or 25% of fertilizer N, with an application rate of 300 kg N ha^{-1} at a Minnesota site similar to the WCSP. Saffigna et al. (1977) measured 120-215 kg N ha^{-1} in lysimeter drainage, which is 39-72% of the fertilizer + irrigation inputs of

235-350 kg N ha^{-1}. Their experimental treatments, which took great care to minimize N leaching, lost 39% (first year) and 51% (second year) of fertilizer + irrigation N while the conventional treatments lost 57% and 72%. Hill (1986) measured N loading below a sandy loam soil profile for three years. Seventy-eight to 220 kg N ha^{-1} or 45-100% of fertilizer + irrigation N leached below the profile. The great variability results partly from N stored in the soil profile at the end of the first season which leached during the second season. Hill's three-year average loading, 134 kg N ha^{-1}, is 70% of fertilizer + irrigation N.

The leaching percentages estimated by various methods for sweet corn in Table 6.31, 56-82% of N inputs, can be compared roughly with field studies showing leaching losses of 33-39% (Tyler and Thomas, 1977), 55% (Reddy and Reddy, 1993), and 29% (Fermanich et al., 1993) of fertilizer N input. The numbers are not strictly comparable, because different growing conditions, inputs, and outputs are included in various studies or methods. The field measurements most comparable to our potato results, 40-71% of N inputs, are 57-72% (Saffigna et al., 1977), and 70% (Hill, 1986). The variability of N leaching as a percentage N input in the potato studies is much less than the variability of kg ha^{-1} of N loading.

BMP effectiveness

The cooperating growers in this study followed practices close to UW-Extension recommended BMPs, yet groundwater beneath the fields was still substantially impacted by agrichemicals. The mean NO_3-N concentration in the top 3 m of groundwater below the four fields was 22.4 mg L^{-1}. Estimates of NO_3-N loaded to groundwater below sweet corn in 1992 were 178 kg ha^{-1} (140 kg ha^{-1} if only fertilizer N is considered). Below potato in 1993 the estimates ranged from 164 to 291 kg ha^{-1}, depending on N input and the method used to estimate loading. We also detected seven pesticide residues in groundwater beneath the fields. Atrazine residues and metolachlor exceeded their Wisconsin ESs, and carbofuran and alachlor exceeded their PALs.

Still, BMPs brought some groundwater quality improvements by reducing or substituting agrichemical inputs. A decrease in N application of 60 kg ha^{-1} on the north half of Field 2 in 1993 reduced NO_3-N loading by a roughly similar amount. Fifteen kg ha^{-1} less N fertilizer on half of Field 4 in 1993 resulted in a roughly similar decrease in N loading through a statistically significant 5.4 mg L^{-1} reduction in NO_3-N concentration. Also, discontinuing atrazine applications on the west half of Field 3 resulted in a statistically significant decrease in both the detection frequency and concentration of atrazine residues compared to the east half.

The use of current BMPs alone will not prevent further degradation of WCSP groundwater. A combination of new BMPs, crop improvement, different crops or rotations, and perhaps institutional controls may be needed.

CHAPTER 7

CONCLUSIONS

GEOLOGY, SOILS, AND HYDROGEOLOGY OF THE PRIORITY WATERSHED

Geologic setting

The study area is generally flat, with a regional southwestward slope of 0.1%. Surface elevation varies from about 970 to 1000 ft above mean sea level.

Five major geologic units are present, but only the Pleistocene unit is important from a groundwater standpoint. The Pleistocene unit belongs to the Big Flats Formation and is generally 60-100 ft thick. It consists of well-sorted, low-organic-matter sands with one or more interbedded silty and clayey subunits. The uppermost silty/clay subunit is The New Rome Member of the Big Flats Formation. Within the priority watershed, the New Rome Member is typically 2-5 ft thick and is found 13 to 39 ft below the surface. Other silty subunits have not been named.

Soils

Soils in the watershed are predominantly Plainfield sand and loamy sand, with smaller amounts of Friendship, Meehan, and Newson loamy sands. Except for the top 1 ft, they contain over 95% well-sorted sand, and the median grain size is medium (250-500 μm) to coarse (500-1000 μm). The 0-1 ft depth interval contained 4-9% silt + clay. Organic-matter content below the top layer of soil was low, averaging 0.05%.

Hydrogeology

Aquifer boundaries

The aquifer base coincides with the base of Pleistocene deposits. The escarpment at the east edge of the watershed drops to wetlands, creeks, or the backwaters of the Wisconsin River which form the eastern boundary of the flow system. The boundaries of the flow system extend well beyond the study area to the north and west.

Hydrogeologic parameters

Hydraulic conductivity (K) was determined by slug tests and by analyzing specific-capacity test data from well records. Slug-test Ks had a geometric mean of 4.0×10^{-4} m s^{-1}. The average K estimated from specific capacity data was 5.2×10^{-4} m s^{-1}, but Ks estimated from wells pumping at greater than 3800 L min^{-1} were substantially higher, 8.6×10^{-4} m s^{-1}. Because this value (8.6×10^{-4} m s^{-1}) reflects the largest

available scale of measurement, it should be most characteristic of basin-scale K. This value is close to the 9.0×10^{-4} m s^{-1} found in the model calibration by Bradbury et al. (1992) for the nearby Buena Vista Basin.

Hydraulic conductivities calculated by the various methods are not entirely comparable because K increases with the scale of measurement. For this reason, we selected 9×10^{-4} m s^{-1} as most representative of basin-scale K.

We measured an aquifer porosity of 40%.

Hydraulic heads were measured monthly and were used to infer horizontal and vertical gradients. Groundwater flows generally from northwest or west toward the southeast or east. The water table fluctuated by about 1.2 m during the study period, rising sharply during the spring of 1993 due to record precipitation, but receding again by March 1994. Flow patterns and horizontal gradients were generally consistent throughout study period, even with fluctuations in the water table.

Generally, vertical gradients were small, less than 0.04, and were related to the presence of a silty lacustrine layer between screens.

Water budget

Annual precipitation in the WCSP averages 80 cm. Under forest, grassland, and non-irrigated cropland, evapotranspiration is about 53 cm, leaving about 25 cm of precipitation as groundwater recharge. Evapotranspiration is greater on irrigated cropland, decreasing net groundwater recharge to 15 cm.

We calculated a steady-state groundwater budget for the watershed which consists of 1.9×10^4 m^3 d^{-1} of recharge, 2.2×10^4 m^3 d^{-1} of underflow entering from the west and north, and 4.2×10^4 m^3 d^{-1} of underflow leaving through the east and south.

GROUNDWATER FLOW AND PARTICLE TRACKING MODELS

Groundwater flow models were constructed using the computer code MODFLOW. Both one-layer and three-layer models were constructed. Calibrated heads were within 2 ft of measured heads at target locations, and most were within 1 ft. Overall, the model appeared to be a good representation of field conditions.

Particle tracking models were constructed using PATH3D to plot flow lines, simulate advective contaminant transport, and help interpret monitoring data. The one-layer MODFLOW simulation was used as the basis for PATH3D calculations of flow lines in plan view. PATH3D's groundwater flow lines closely matched flow lines drawn from equipotential maps. This tends to verify the accuracy of the flow and particle-tracking models. Results of particle tracking modeling showed approximately 54% of the watershed is underlain by plumes from irrigated agricultural fields.

A vertical profile of advective transport was traced with PATH3D for Fields 1 and 2, located in the northern part of the watershed, using the three-layer MODFLOW model as input. The profiles showed particle paths converging downgradient of fields. Predicted vertical particle paths were highly dependent upon New Rome vertical hydraulic conductivity, but this feature may be artificial.

GROUNDWATER QUALITY

Inorganic chemistry

The groundwater chemistry of the watershed is the Ca-Mg-HCO_3 type, with Cl, Na, SO_4, and NO_3 present as other major ions. The pH ranged from 4.9 to 8.7, and alkalinity averaged 26 mg L^{-1} $CaCO_3$. Outside of fields, Cl concentrations ranged from < 1.0 to 59 mg L^{-1} and NO_3-N from <0.2 to 26.4 mg L^{-1}. Beneath fields, the peak Cl concentration reached 119 mg L^{-1} and NO_3-N, 81 mg L^{-1}. In general, the groundwater was similar to that of the Wisconsin Central Sand Plain as a whole, but with lower concentrations of dissolved carbonates. In two instances, low NO_3-N concentrations at depth apparently were due to denitrification. However, denitrification did not seem to be a prevalent mechanism. Both pH and alkalinity tended to increase with depth, from north to south, and from west to east. Nitrate concentrations often decreased with depth below the water table. Reducing conditions were frequently encountered at depth, and also near the water table in the northwest part of the project area, an area with shallow groundwater and poorly drained soils.

Groundwater in and upgradient of four vegetable fields in the north part of the Priority Watershed was monitored in detail. Groundwater was slightly more acidic within fields (pH 5.0, vs. 5.4 upgradient), and solute content was greater. Chloride, Na, K, Ca, Mg, and Al concentrations were substantially greater within fields, but Fe concentrations were less. The average NO_3-N concentration within the fields was 22.4 mg L^{-1}, compared to 1.4 mg L^{-1} upgradient.

Pesticides

Pesticides were detected at all sampling locations within the four intensively monitored fields. The same pesticides were found as outside the fields, plus de-isopropyl atrazine. Atrazine exceeded its Wisconsin preventive action limit (PAL) more frequently than the other pesticides. Metolachlor, carbofuran, and alachlor also exceeded their PALs occasionally. Atrazine and metolachlor exceeded their Wisconsin enforcement standards (ESs) at a few locations.

Pesticides were detected in wells at six monitoring locations outside but near irrigated vegetable fields. These were metolachlor, metribuzin, alachlor, carbofuran and atrazine with its metabolite de-ethyl

atrazine. We were not able to test for all pesticide residues. For instance, alachlor ESA was probably present, but was not detectable with the analytical method used. Alachlor, de-ethyl atrazine, and metolachlor each was detected once above its PAL. None exceeded its ES. The lack of ES exceedences indicated that recent farming practices generally did not pollute groundwater above the pesticide ES very far from fields.

Land-use impacts

Plumes from irrigated fields underlay 54% of the basin, according to flow and particle-tracking models. Plume thicknesses, as indicated by monitoring data, ranged from about 3 to 15 m depending on flow conditions and the number of fields that lie along the same flow line. The plumes had a distinctive chemistry compared to other groundwater, usually exhibiting elevated concentrations of NO_3, Ca, Cl, K, Mg, Na, and SO_4, as well as occasional pesticides. All NO_3-N concentrations exceeding 8 mg L^{-1} were associated with plumes from irrigated vegetable fields. In some cases where several fields lay along a flow line, the entire saturated thickness exceeded the NO_3-N standard of 10 mg L^{-1}. We found no evidence that other land uses in the watershed significantly impacted groundwater. However, some common land uses absent from the watershed, such as small-lot unsewered subdivisions and non-irrigated agriculture, may have important groundwater impacts in other parts of the WCSP.

NITRATE LOADING TO GROUNDWATER UNDER IRRIGATED FIELDS

Estimate of potentially leachable N

Potentially leachable N was calculated using a budget approach. Estimates ranged from 133 to 237 kg ha^{-1}, or 40-82% of total N input, depending on crop, yield, weather, and cultural treatments. The greatest leachable N percentages were found for sweet corn in both years, due to poor yields and meager harvested N output. The greatest amounts of leachable N were found with potato in 1993, primarily because excessive rain compelled heavy N fertilizer application.

Simple calculation from NO_3-N concentration and recharge

The average NO_3-N concentration in shallow groundwater below the four fields and the amount of recharge in both years were used to derive simple estimates of NO_3 loading to groundwater. These estimates were 76-95 kg ha^{-1} in 1992 and 97-122 kg ha^{-1} in 1993. For the two years and four fields, the estimated annual loading was 112 kg ha^{-1}, 42% of total N input.

Loading through a water year

Detailed concentration profiles were analyzed to quantify annual NO_3 loading to groundwater beneath Field 2. Estimated NO_3-N loading to groundwater in the 1992 water year, when sweet corn was grown, was 178 ± 31 kg ha^{-1}. For the 1993 water year, the estimate of NO_3-N loading under potato was 260 ± 53 kg ha^{-1}.

Fertilizer-N leaching model

A nitrogen leaching model was constructed for one field using the computer code PRZM to simulate fertilizer N leaching. For the 1992 sweet corn crop, the PRZM simulation indicated 42 kg ha^{-1} of NO_3-N leached to groundwater out of 250 kg ha^{-1} fertilizer N. This was less than the estimates by other methods, because only direct leaching of fertilizer N was included. PRZM's calculation of crop N uptake was combined with data on N removed in harvest to indicate 140 kg N ha^{-1}, or 56% of the applied fertilizer N, became available for leaching.

For the 1993 potato crop, the PRZM-simulated fertilizer NO_3-N loading was 193 (north field-half) or 216 (south) kg ha^{-1}. Residue N was not important with potato.

Sensitivity analyses showed that field moisture capacity was the normal initial soil moisture condition on Jan. 1, and that model results were rather insensitive to crop root depth.

The model was more sensitive to soil water-holding capacity. Varying water-holding capacity by 25% changed calculated recharge by as much as 16%, and changed calculated NO_3 loading by up to 11%.

Comparison of methods for estimating N loading

The four methods of estimating NO_3-N loading evaluated different types of loading. Generally, they agreed fairly well, except that the result based on NO_3-N concentrations and recharge was low. Nitrate concentrations in the lower part of the sampled zone may be diminished by mixing with groundwater that originated upgradient from the fields, leading to artificially low averages. In 1992, the N budget and the water year method agreed perfectly. The N leaching model indicated less loading to groundwater, but this would be expected since it did not consider N sources other than fertilizer. The percentage of N input found to leach was nearly the same for all three methods. In 1993, the comparisons among methods differ between high and low-input regimes. Where N input was less , the N budget predicted less NO_3-N loading than the water year method, and the leaching model predicted almost the same as the water year method even though it ignores non-fertilizer N inputs. With more N input, the N budget estimate was again low compared to the water year method. The leaching model predicted less N loaded to groundwater than the water year method, as expected.

Low estimates by the budget method could indicate that too large a value was used for the N concentration in the harvested potatoes. That possibility was also suggested by the fact that the harvested N used in the budget was greater than the total crop uptake that PRZM calculated.

Efficacy of model

The PRZM N leaching model gave results for potato that agreed reasonably well with the other methods. The sweet corn results approached those of the other methods only when additional data (on crop residue N) were considered. PRZM provided useful results with a modest requirement for input data.

EFFECTIVENESS OF BMPS

The cooperating growers in this study followed practices close to UW-Extension recommended BMPs (Binning, L. 1995. pers. commun.), yet groundwater beneath the fields was still substantially impacted by agrichemicals. The mean NO_3-N concentration in the top 3 m of groundwater below the four fields was 22.4 mg L^{-1}. Estimates of NO_3-N loaded to groundwater below sweet corn in 1992 were 178 kg ha^{-1} (140 kg ha^{-1} if only fertilizer N is considered). Below potato in 1993 the estimates ranged from 164 to 291 kg ha^{-1}, depending on N input and the method used to estimate loading. In addition, we detected seven pesticide residues in groundwater beneath the fields. Atrazine residues and metolachlor exceeded their Wisconsin ESs, and carbofuran and alachlor exceeded their PALs.

Still, BMPs brought some groundwater quality improvements through reducing or substituting agrichemical inputs. A decrease in N application of 15 or 60 kg ha^{-1} reduced NO_3-N loading by roughly similar amounts. Also, discontinuing atrazine applications resulted in a decrease in both the detection frequency and concentration of atrazine residues.

The use of current BMPs alone will not prevent further degradation of WCSP groundwater. A combination of new BMPs, crop improvement, different crops or rotations, and perhaps institutional controls may be needed.

If BMPs cannot protect groundwater, other strategies are needed. Stites and Kraft (1992) considered two hypothetical strategies for meeting the groundwater NO_3 standard, assuming that 40% of applied N leaches as NO_3, and the groundwater column beneath a point is completely mixed. The first strategy would implement a rotation such that the average groundwater recharge over the rotation meets standards. Potato, the most profitable crop, could be grown every 5-6 years, but no N could be applied during the rest of the rotation. The other strategy would continue the present rotation, but restrict the fraction of land in produc-

tion, so that average groundwater quality in an area would meet standards. Under this strategy, 30% of the basin could be in the current rotation, if no NO_3 leaches from the remaining land.

The apparent failure of BMPs to protect groundwater and the restrictiveness of the hypothetical strategies indicate that new ideas are needed to maintain profitable agriculture and protect groundwater. Some ideas could include improved crop varieties that would take up nutrients more efficiently and resist pests, improved pesticide compounds or techniques, alternative economically viable crop rotations, or economic incentives to permit Wisconsin vegetable growers to remain competitive while setting cropland aside at least part of the time.

APPENDIX 1

CORE COLLECTION FOR POROSITY DETERMINATIONS

July 22, 1993. Personnel: Jean Bahr, Diana Berge, Cheryl Feind, Jennifer Gearing, Brian Hess, George Kraft, Will Stites. First site: NW corner of highway 173 and Kimball Ave., next to well location 0603.

First core: collected soil material from 1.0 to 4.05 ft (30 to 123 cm) below surface. There is no information on compaction, but this soil material is not representative of the aquifer, so its porosity is of less interest.

Second core: collected sediment from 46 to 96 inches (117 to 244 cm) below the surface. The top 4 inches (10 cm) was reported to be sloughed material caused by insertion of the second piece of pipe. The pipe was advanced 50 inches (127 cm), and at the finish contained 53½ inches (136 cm) of sediment. Assuming the 4 inches of sloughed material plus the 50 inches of new sediment was compacted into 53½ inches, the compaction was 0.9%.

Third core: we tried using a full 30-ft (9.1-m) length of pipe. Although the hole had just been advanced to 96 inches (244 cm), this third pipe could only be inserted to 55½ inches (141 cm), due to slumping of the sediment below the water table. Therefore the top 40½ inches (103 cm) of this core is waste. The pipe was advanced 41½ inches (105 cm), i.e. negligible progress beyond the second core.

Based on this experience, we felt it was impossible to make significant progress beyond the water table.

Second site, stop 1: on a logging road leading obliquely down the bank where the glacial lake bed was cut away by the glacial or postglacial Wisconsin River. This site is just east of highway JJ in Section 20 about ¼ mile north of Creamery Rd., across from Francis Elsen's driveway. Our first stop at this site was about 12-15 ft (3.5-4.5 m) below the surface of the highway.

We dug into the road surface with a shovel to clear away the gravel that had been added. It appeared we were on an outcrop of the New Rome member (Big Flats formation). The silty material was conspicuously laminated, brown, quite dry to the touch, and its structure was extremely friable. This outcrop may have been associated with a decrease in the slope of the road; minimal grading had been done to create the road. The shoveled hole was about 16 inches (41 cm) deep.

First core: the pipe was advanced 19 inches (48 cm). We used a graduated 1-inch PVC pipe, plugged at the bottom and resting on the sediment inside the aluminum pipe, as a rod to gauge of the advancement of the pipe relative to the top of the core. This pipe settled into the top of the core during vibration, and so did not provide a useful measure of core compaction. (We abandoned the PVC rod technique after taking this core.) There was apparently no compaction of this core, as far as could be determined by the position of that gauging rod.

Second core: the pipe was advanced 18¾ inches (48 cm).

Second site, stop 2: this was farther down the logging road, past a break where the slope steepened, and below the New Rome layer.

A pilot hole was dug with a shovel, about 16 inches (41 cm) deep.

First core: the pipe was driven 36 inches (91 cm) below the pilot hole, with ½ inch (1 cm) of compaction. This implies 1.4% compaction in the core.

Second core: we started with a 120-inch (305-cm) pipe; after we inserted it to the depth the first core reached, there were 4 inches of disturbed soil in the pipe. The pipe was advanced 36 inches (91 cm), with 1 inch (2.5 cm) of compaction. This implies 2.8% compaction in the core. The depth reached was 84 inches (213 cm).

Third core: we started with a 144-inch (366-cm) pipe. This was pushed down to 83 inches (211 cm) below surface, with 6 inches (15 cm) of sloughed material collecting in the bottom. Then the core was advanced 46 inches (117 cm), to 129 inches (328 cm) below the surface. The length of core in the pipe then was 49¼ inches (125 cm). The 6 inches of sloughed material plus the 46 inches of advancement should have given a 52-inch (132-cm) core, thus the compaction was 3¾ inches (9.5 cm), or 7.2%. The assumption of uniform compaction in a core with so much sloughed material is hard to support, and that will make interpretation difficult.

Fourth core: a 216-inch (549-cm) pipe was pushed down to 104 inches (264 cm) below surface; thus the first 25 inches (64 cm) of this core is sloughed material. The pipe was advanced 42½ inches (108 cm). Initial empty pipe length was not measured, so there is no way to calculate compaction in this core; however it is doubtful how meaningful such a computation would be, given the result with the third core. Final advancement in this hole was 145½ inches (368 cm).

Third site: NE corner of County Line Rd. and Kimball Ave., near well site 3202.

First core: a 6-inch (15-cm) pilot hole was dug, and a pipe was advanced 37 inches (94 cm). Compaction was ½ inch (1 cm), or 1.4%. The depth reached was 45 inches (114 cm).

Second core: a pipe was inserted to 45 inches (114 cm). Seven inches (18 cm) of sloughed material collected in the bottom of the pipe. The pipe was advanced 21¼ inches (54 cm), with 2¾ inches (7 cm) of compaction. Again, the assumption of uniform compaction throughout the core comes into question. The depth reached was 76 inches (193 cm). Judging from a sketch made when the core was opened, 2.1 inches (5.3 cm) of the compaction occurred in the sloughed material, leaving 0.67 inch (1.7 cm), or 3.2%, of compaction taking place within the core per se.

Third core: the pipe was inserted to 59 inches (150 cm), thus the top 27 inches (69 cm) of this core consists of disturbed material. It was advanced to 101 inches (257 cm) below surface. Once again, the initial empty pipe length was not measured, so there is no way to calculate compaction in this core, but the value of such a calculation is doubtful anyway. Upon withdrawal, the bottom 7 inches (18 cm) of the pipe was empty.

Fourth core: the top 7 inches (18 cm) or more of this core consists of sloughed material from the previous core. This pipe was inserted to 98 inches (249 cm) below surface, then advanced 25 inches (64

cm). Compaction in the core was ½ inch (1 cm), or 2.0%. When the core was pulled, the bottom 4 inches (10 cm) was lost. The depth reached so far is 123 inches (312 cm).

Fifth core: the pipe was inserted to 98 inches (249 cm), the same starting depth as the fourth core. It was advanced 57½ inches (146 cm), to 155½ inches (395 cm). A core 63 inches (160 cm) long was collected with none lost from the bottom. The top 25 inches (64 cm) or more consists of sloughed material from the previous core.

July 23, 1993. Personnel: Diana Berge, Cheryl Feind, Jennifer Gearing, Scott Provost (site 5 only); Will Stites.

Fourth site: NW corner of County AA and Kimball Ave., near well site 3003. A pilot hole about 6 inches deep was shoveled.

First core: the pipe was advanced 43½ inches (110 cm), ending 51 inches (130 cm) below the surface. Compaction in the core was 4½ inches (11.4 cm) or 10.3%. Note: the lower part of this core was lost during the process of opening the core.

Second core: the pipe was inserted to 51 inches (130 cm) below surface, the same depth reached with the first core, whereupon it contained 10 inches (25 cm) of sloughed material. It was advanced 60 inches (152 cm), to 110 inches (279 cm) below surface. There was no compaction in this core. We were forced to use the vibrator to pull the core; probably as a result, 26½ inches of sediment was missing from the bottom. Loss of sample from the bottom was also promoted by the coarseness of the sediment; it was a mix of coarse sand and fine gravel. The pipe was slightly dented at the bottom, probably from impact against gravel particles.

Third core: the initial insertion depth was not recorded, so compaction cannot be calculated. The depth reached was 89½ inches (227 cm) below surface, less than the 110 inches (279 cm) reached by the second core. This core served to clear out part of the sloughed material which fell from the previous core, and to recover a disturbed sample of the coarse sediment that was lost.

Fourth core: the pipe was inserted to 79½ inches (202 cm) below surface, less than the depth reached by the previous core. Initial level of material inside the pipe was not recorded, so it is not possible to determine compaction. But since all the material recovered in this core is heavily disturbed, that is of no concern. The pipe was advanced to 112 inches (284 cm) below surface, or slightly beyond the second core. Upon withdrawal, 26 inches (66 cm) of sediment was lost from the bottom of the core. The remaining core was 44½ inches (113 cm) long.

Fifth site: NW corner of Lynn Hill Rd. and Kimball Ave., near well site 1803. A pilot hole about 6 inches (15 cm) deep was dug.

First core: the pipe was advanced to 30½ inches (77 cm) below surface, and 4 inches (10 cm, 13.1%) of compaction occurred.

Second core: the pipe was inserted to 30½ inches (77 cm) below the surface, with only 1 inch of sloughed material at the bottom. It was advanced 46½ inches (118 cm), to 77 inches (196 cm) below

surface. The data indicate *expansion* of the core by 2 inches (5 cm), for a compaction of -4.3%. This probably has more to do with the precision of the measurements than actual core expansion. When the core was withdrawn, 7 inches (18 cm) of sediment was missing from the bottom.

Third core: the pipe was inserted to 69 inches (175 cm) below surface, 8 in (20 cm) less than the previous core reached. Six inches (15 cm) of sloughed material was present in the bottom of the pipe. It was advanced to 124½ inches (316 cm) below surface. The measured compaction was -½ inch (-1 cm, -0.4%). When the core was pulled, 24½ inches (62 cm) was lost from the bottom.

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
0101	EU653	1	Sand	0	26	990.72	964.72
0101	EU653	2	Clay	26	30	964.72	960.72
0102	EU656	1	Sand	0	26	986.87	960.87
0102	EU656	2	Clay	26	28	960.87	958.87
0102	EU656	3	Sand	28	30	958.87	956.87
0103	EU607	1	Sand	0	28	987.29	959.29
0103	EU607	2	Clay	28	29	959.29	958.29
0103	EU607	3	Sand	29	42.4	958.29	944.89
0103	EU654	1	Sand	0	29	986.72	957.72
0103	EU654	2	Clay	29	31	957.72	955.72
0103	EU654	3	Sand	31	53	955.72	933.72
0103	EU655	1	Sand	0	29	986.89	957.89
0103	EU655	2	Clay	29	31	957.89	955.89
0103	EU655	3	Sand	31	47	955.89	939.89
0501	EU651	1	Sand	0	31	995.16	964.16
0501	EU651	2	Clay	31	33	964.16	962.16
0502	EU667	1	Sand	0	25	997.00	972.00
0502	EU667	2	New Rome	25	27.5	972.00	969.50
0502	EU667	3	Sand	27.5	50	969.50	947.00
0502	EU667	4	Less developed clay ?	52	60	945.00	937.00
0502	EU667	5	Sand	60	68	937.00	929.00
0503	EU622	1	Sand	0	21	993.05	972.05
0503	EU622	2	Clay	21	23	972.05	970.05
0503	EU622	3	Sand	23	25	970.05	968.05
0503	EU622	4	Clay	25	29	968.05	964.05
0503	EU622	5	Sand	29	30	964.05	963.05
0503	EU622	6	Clay, perhaps silty sand	30	38	963.05	955.05
0503	EU622	7	Sand	38	54	955.05	939.05
0503	EU622	8	Soft rock or hard gravel	54	0	939.05	993.05
0503	EU763	1	Sand	0	29	993.18	964.18
0503	EU763	2	Clay	29	31	964.18	962.18
0503	EU763	3	Sand	31	69	962.18	924.18
0503	EU763	4	Clay	69	80	924.18	913.18
0573	EU459	1	Sand	0	24	992.91	968.91
0573	EU459	2	Clay	24	30	968.91	962.91

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
0573	EU459	3	Sand	30	32	962.91	960.91
0576	EU640	1	Sand	0	27	990.86	963.86
0576	EU640	2	Clay	27	28	963.86	962.86
0601	EU652	1	Sand	0	26	992.72	966.72
0601	EU652	2	Clay	26	28	966.72	964.72
0601	EU652	3	Sand	28	29	964.72	963.72
0602	EU669	1	Sand & gravel	0	19	993.00	974.00
0602	EU669	2	New Rome	19	25	974.00	968.00
0602	EU669	3	Sand & gravel	25	33	968.00	960.00
0602	EU669	4	Clay	33	36	960.00	957.00
0602	EU669	5	Sand & gravel	36	54	957.00	939.00
0603	EU604	1	Sand	0	25	989.57	964.57
0603	EU604	2	Clay ?	25	29	964.57	960.57
0603	EU604	3	Sand	29	63	960.57	926.57
0603	EU762	1	Sand	0	28	989.23	961.23
0603	EU762	2	Clay	28	30	961.23	959.23
0603	EU762	3	Sand	30	31	959.23	958.23
0671	EU631	1	Sand	0	29	987.37	958.37
0671	EU631	2	Clay	29	33	958.37	954.37
0671	EU631	3	Sand	33	37	954.37	950.37
0671	EU631	4	Clay	37	38	950.37	949.37
0671	EU631	5	Sand	38	65	949.37	922.37
0671	EU631	6	Clay	65	70	922.37	917.37
0674	EU634	1	Sand	0	20	991.76	971.76
0674	EU634	2	Clay	20	26	971.76	965.76
0674	EU634	3	Sand	26	37	965.76	954.76
0701	EU590	1	Sand	0	16	987.00	971.00
0701	EU590	2	Silty sand	16	21	971.00	966.00
0701	EU590	3	Silt	21	26	966.00	961.00
0702	EU660	1	Sand	0	27	986.52	959.52
0702	EU660	2	Clay	27	29	959.52	957.52
0702	EU660	3	Sand	29	60	957.52	926.52
0702	EU660	4	Clay	60	61	926.52	925.52
0702	EU660	5	Sand	61	93	925.52	893.52
0702	EU660	6	Sand	93	94	893.52	892.52

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
0801	EU664	1	Sand	0	3	986.01	983.01
0801	EU664	2	Gravel	3	4	983.01	982.01
0801	EU664	3	Sand	4	13	982.01	973.01
0801	EU664	4	Clay	13	15	973.01	971.01
0801	EU664	5	Sand	15	21	971.01	965.01
0801	EU664	6	Clay	21	30	965.01	956.01
0801	EU664	7	Dense sand	30	33	956.01	953.01
0801	EU664	8	Sand	33	56	953.01	930.01
0801	EU664	9	'Dense stuff'	56	60	930.01	926.01
0802	EU625	1	Sand	0	30	988.24	958.24
0802	EU625	2	Clay	30	35	958.24	953.24
0802	EU625	3	Sand	35	85	953.24	903.24
0802	EU761	1	Sand	0	36	988.21	952.21
0802	EU761	2	Sand	36	41	952.21	947.21
0802	EU761	3	Clay	41	43	947.21	945.21
0802	EU761	4	Sand	43	74	945.21	914.21
0802	EU761	5	Clay above rock	74	75	914.21	913.21
1201	EU657	1	Sand	0	18	980.31	962.31
1201	EU657	2	Clay	18	20	962.31	960.31
1201	EU657	3	Sand	20	22	960.31	958.31
1301	EU611	1	Sand	0	29	981.98	952.98
1301	EU611	2	Clay	29	37	952.98	944.98
1301	EU611	3	Sand	37	50	944.98	931.98
1301	EU611	4	Clay	50	53	931.98	928.98
1301	EU611	5	Gravel	53	61	928.98	920.98
1301	EU611	6	Gravelly clay?	61	69	920.98	912.98
1301	EU611	7	Sandstone ?	69	0	912.98	981.98
1301	EU658	1	Sand	0	31	981.86	950.86
1301	EU658	2	Clay	31	34	950.86	947.86
1301	EU658	3	Sand	34	35	947.86	946.86
1701	EU585	1	Sand	0	13	984.00	971.00
1701	EU585	2	Lens (dense stuff)	13	0	971.00	984.00
1701	EU585	3	Light clay	26	29.5	958.00	954.50
1701	EU585	4	Sand	29.5	90	954.50	894.00
1702	EU630	1	Sand	0	34	985.59	951.59

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
1702	EU630	2	Clay	34	37	951.59	948.59
1702	EU630	3	Sand	37	83	948.59	902.59
1702	EU630	4	Soft sandstone ?	83	86	902.59	899.59
1702	EU767	1	Sand	0	7.5	985.70	978.20
1702	EU767	2	Gravel	7.5	8.5	978.20	977.20
1702	EU767	3	Sand	8.5	75	977.20	910.70
1702	EU767	4	Clay above rock	75	76	910.70	909.70
1801	EU764	1	Sand	0	42	978.57	936.57
1801	EU764	2	Sand	42	55	936.57	923.57
1801	EU764	3	Clay	55	59	923.57	919.57
1801	EU764	4	Clay	59	60	919.57	918.57
1802	EU595	1	Sand	0	13	977.77	964.77
1802	EU595	2	Silty sand to clay	13	24.5	964.77	953.27
1803	EU628	1	Sand	0	21	983.00	962.00
1803	EU628	2	Clay	21	29	962.00	954.00
1803	EU628	3	Sand	29	70	954.00	913.00
1803	EU628	4	Clay/dense stuff	70	71	913.00	912.00
1803	EU766	1	Sand	0	10	982.99	972.99
1803	EU766	2	Sand and gravel	10	15	972.99	967.99
1803	EU766	3	Sand	15	21	967.99	961.99
1803	EU766	4	Clay	21	24	961.99	958.99
1803	EU766	5	Sand	24	31	958.99	951.99
1803	EU766	6	Clay	31	33	951.99	949.99
1803	EU766	7	Sand	33	75	949.99	907.99
1803	EU766	8	Clay above rock	75	76	907.99	906.99
2001	EU769	1	Ap soil horizon	0	1	974.92	973.92
2001	EU769	2	Sand	1	2.75	973.92	972.17
2001	EU769	3	Sand	2.75	8	972.17	966.92
2001	EU769	4	Sand	8	13	966.92	961.92
2001	EU769	5	Sand	13	18	961.92	956.92
2001	EU769	6	Sand	18	23	956.92	951.92
2001	EU769	7	Sand	23	28	951.92	946.92
2001	EU769	8	Sand	28	33	946.92	941.92
2001	EU769	9	Sand	33	38	941.92	936.92
2001	EU769	10	Sand	38	43	936.92	931.92

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
2001	EU769	11	Sand	43	48	931.92	926.92
2001	EU769	12	Sand	48	53	926.92	921.92
2001	EU769	13	Sand	53	58	921.92	916.92
2001	EU769	14	Sand	58	63	916.92	911.92
2001	EU769	15	Sand	63	68	911.92	906.92
2001	EU769	16	Sand	68	73	906.92	901.92
2001	EU769	17	Sand	73	78	901.92	896.92
2001	EU769	18	Sand	78	81	896.92	893.92
2001	EU769	19	?	81	87	893.92	887.92
2001	EU769	20	?	87	98	887.92	876.92
2401	EU616	1	Sand	0	55	976.83	921.83
2401	EU616	2	Clay ?	55	60.5	921.83	916.33
2401	EU616	3	Sand	60.5	68	916.33	908.83
2401	EU616	4	Clay ?	68	75	908.83	901.83
2402	EU682	1	Sand	0	19	972.43	953.43
2402	EU682	2	Clay	19	20	953.43	952.43
2402	EU682	3	Sand	20	21	952.43	951.43
2403	EU681	1	Sand	0	26	975.98	949.98
2403	EU681	2	Clay	26	28	949.98	947.98
2403	EU681	3	Sand	28	29	947.98	946.98
2501	EU683	1	Sand	0	7	969.30	962.30
2501	EU683	2	Clay	7	13	962.30	956.30
2501	EU683	3	Sand	13	14	956.30	955.30
2901	EU563	1	Sand	0	39	980.09	941.09
2901	EU563	2	Silty sand ?	39	42	941.09	938.09
2901	EU563	3	Sand	42	79.5	938.09	900.59
2901	EU563	4	'Dense stuff'	79.5	80	900.59	900.09
2901	EU768	1	Sand	0	20	980.21	960.21
2901	EU768	2	Clay	20	21	960.21	959.21
2901	EU768	3	Sand	21	33	959.21	947.21
2901	EU768	4	Sand	33	34	947.21	946.21
2902	EU566	1	Sand	0	30	976.34	946.34
2902	EU566	2	Clay	30	35	946.34	941.34
2902	EU566	3	Sand	35	47	941.34	929.34
2902	EU566	4	Clay	47	48	929.34	928.34

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
2902	EU566	5	Sand	48	78	928.34	898.34
2902	EU686	1	Sand	0	18	976.39	958.39
2902	EU686	2	Sand & gravel	18	19	958.39	957.39
2902	EU686	3	Sand	19	39	957.39	937.39
2902	EU686	4	Silty clay	39	43	937.39	933.39
2902	EU686	5	Sand	43	88	933.39	888.39
2902	EU686	6	Sand	88	89	888.39	887.39
3001	EU620	1	Sand	0	59	976.31	917.31
3001	EU620	2	Slightly dense layer	59	61	917.31	915.31
3001	EU620	3	Sand	61	65	915.31	911.31
3001	EU620	4	Slightly dense layer	65	66	911.31	910.31
3001	EU620	5	Till-like material ?	70	75	906.31	901.31
3002	EU580	1	Sand	0	19	975.68	956.68
3002	EU580	2	Clay	19	26	956.68	949.68
3002	EU580	3	Sand, fine gravel	26	79	949.68	896.68
3002	EU580	4	Dk.gray, like A horiz.so	79	81	896.68	894.68
3003	EU685	1	Sand	0	14	971.41	957.41
3003	EU685	2	Clay	14	20	957.41	951.41
3003	EU685	3	Sand	20	66	951.41	905.41
3003	EU685	4	Clay	66	67	905.41	904.41
3101	EU689	1	Sand	0	28	966.13	938.13
3101	EU689	2	Clay	28	30	938.13	936.13
3101	EU689	3	Sand	30	32	936.13	934.13
3101	EU689	4	Sand	32	33	934.13	933.13
3201	EU575	1	Sand	0	23	970.41	947.41
3201	EU575	2	Clay	23	26	947.41	944.41
3201	EU575	3	Sand	26	60	944.41	910.41
3201	EU575	4	Gravel	60	61	910.41	909.41
3201	EU575	5	Sand	61	68	909.41	902.41
3201	EU575	6	'Tough stuff'	68	70	902.41	900.41
3202	EU688	1	Sand	0	10	970.29	960.29
3202	EU688	2	Clay	10	11	960.29	959.29
3202	EU688	3	Sand	11	75	959.29	895.29
3202	EU688	4	Sand	75	76	895.29	894.29
3203	EU687	1	Sand	0	70	971.29	901.29

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
3203	EU687	2	Clay?	70	73	901.29	898.29
3203	EU687	3	Sand	73	88	898.29	883.29
3203	EU687	4	Sand	88	89	883.29	882.29
3601	EU599	1	Sand	0	22	969.76	947.76
3601	EU599	2	Clay	22	29	947.76	940.76
3601	EU599	3	Sand	29	45	940.76	924.76
3601	EU599	4	Clay	45	47	924.76	922.76
3601	EU599	5	Sand	47	59	922.76	910.76
3601	EU599	6	Gravel	59	65	910.76	904.76
3601	EU599	7	Sand	65	75	904.76	894.76
3602	EU690	1	Sand	0	12	966.97	954.97
3602	EU690	2	Silty sand?	12	28	954.97	938.97
3602	EU690	3	Sand	28	38	938.97	928.97
3602	EU690	4	Sand	38	39	928.97	927.97

The following private wells were in or near the Port Edwards Groundwater Priority Watershed. In the "Loc. code" colu is shown the code that appears on Figure 2.3. In a few cases, these are Unique Well Numbers. In the UWN column, a CWGC serial number is shown, whose first two digits are the civil section where the well is located.

ZZ036	P0400100	1	Sand & gravel	0	22	955.00	933.00
ZZ036	P0400100	2	Silt	22	37	933.00	918.00
ZZ036	P0400100	3	Hardpan	37	96	918.00	859.00
ZZ036	P0400100	4	Soft rock (granite)	96	104	859.00	851.00
38610	P0500100	1	Sand	0	63	993.00	930.00
38633	P0500200	1	Coarse sand	0	5	996.00	991.00
38633	P0500200	2	Coarse sand & gravel	5	25	991.00	971.00
38633	P0500200	3	Grey clay	25	35	971.00	961.00
38633	P0500200	4	Gravel & coarse sand	35	72	961.00	924.00
38633	P0500200	5	Bluegreen clay	72	72	924.00	924.00
38609	P0600100	1	Sand	0	11	992.00	981.00
38609	P0600100	2	Sandy clay	11	15	981.00	977.00
38609	P0600100	3	Fine sand	15	19	977.00	973.00
38609	P0600100	4	Coarse sand & gravel	19	26	973.00	966.00
38609	P0600100	5	Fine sand w/ pebbles	26	60	966.00	932.00
38611	P0800100	1	Sand & gravel	0	92	989.00	897.00

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
ZZ004	P1600100	1	Sand	0	18	940.00	922.00
ZZ004	P1600100	2	Clay	18	46	922.00	894.00
ZZ004	P1600100	3	Soft rock	46	50	894.00	890.00
ZZ004	P1600100	4	Granite	50	70	890.00	870.00
ZZ006	P1600200	1	Top soil	0	2	932.00	930.00
ZZ006	P1600200	2	Sand & gravel	2	66	930.00	866.00
38615	P1700100	1	Sand & gravel	0	73	987.00	914.00
CY903	P2000700	1	Sand & gravel	0	22	985.00	963.00
CY903	P2000700	2	Clay	22	32	963.00	953.00
CY903	P2000700	3	Sand & gravel	32	86	953.00	899.00
ZZ040	P2400100	1	Sand	0	60	975.00	915.00
38631	P2500200	1	Coarse sand	0	5	971.00	966.00
38631	P2500200	2	Coarse sand & gravel	5	20	966.00	951.00
38631	P2500200	3	Coursesand,grav,grey clay	20	25	951.00	946.00
38631	P2500200	4	Grey clay	25	30	946.00	941.00
38631	P2500200	5	Coarse & medium sand	30	40	941.00	931.00
38631	P2500200	6	Medium & fine sand	40	45	931.00	926.00
38631	P2500200	7	Coarse sand & gravel	45	84	926.00	887.00
38631	P2500200	8	White clay & sandstone	84	84	887.00	887.00
38607	P2900100	1	Medium sand	0	5	979.00	974.00
38607	P2900100	2	Coarse sand & gravel	5	10	974.00	969.00
38607	P2900100	3	Med. sand & gravel	10	15	969.00	964.00
38607	P2900100	4	Medium sand	15	55	964.00	924.00
38607	P2900100	5	Coarse sand	55	65	924.00	914.00
38607	P2900100	6	Medium sand	65	70	914.00	909.00
38607	P2900100	7	Coarse sand	70	95	909.00	884.00
38607	P2900100	8	Medium sand	95	104	884.00	875.00
ZZ001	P3000100	1	Sand	0	72	971.00	899.00
ZZ003	P3000200	1	Sand	0	56	975.00	919.00
ZZ020	P3000300	1	Sand	0	72	971.00	899.00
38606	P3100100	1	Fine sand	0	5	972.00	967.00
38606	P3100100	2	Fine sand & gravel	5	25	967.00	947.00
38606	P3100100	3	Fine sand	25	30	947.00	942.00
38606	P3100100	4	Fine sand & gravel	30	40	942.00	932.00
38606	P3100100	5	Coarse gravel	40	55	932.00	917.00

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
38606	P3100100	6	Fine sand	55	60	917.00	912.00
38606	P3100100	7	Fine sand & gravel	60	75	912.00	897.00
38606	P3100100	8	Coarse gravel	75	83	897.00	889.00
38608	P3200200	1	Sand and gravel	0	93	973.00	880.00
38626	P3200300	1	Sand	0	10	974.00	964.00
38626	P3200300	2	Sand & gravel	10	70	964.00	904.00
38626	P3200300	3	Med. sand-Fine gravel	70	103	904.00	871.00
ZZ027	P3300100	1	Sand & gravel	0	66	960.00	894.00
ZZ028	P3300200	1	Sand & gravel	0	91	950.00	859.00
AX305	P3300500	1	Sand	0	62	955.00	893.00
38618	P3600100	1	Sand & gravel	0	81	967.00	886.00

The following private wells were in or near the Port Edwards Groundwater Priority Watershed. When a Wisconsin Unique Well Numbers exists, it is listed. Otherwise, a CWGC serial number is shown whose first 2 digits indicate the civil section where the well is located.

P0300100	1	Surface soil	0	3	970.00	967.00
P0300100	2	Sand	3	17	967.00	953.00
P0300100	3	Silty sand	17	24	953.00	946.00
P0300100	4	Sand w/ small gravel	24	37	946.00	933.00
P0300100	5	Sand w/ some sm. grav.	37	46.5	933.00	923.50
P0300200	1	Sand	0	16	970.00	954.00
P0300200	2	Decomposed granite	16	18	954.00	952.00
P0300200	3	Granite	18	60	952.00	910.00
P0300300	1	Surface soil	0	2	990.00	988.00
P0300300	2	Clay & sand mixed	2	10	988.00	980.00
P0300300	3	Fine sand	10	42	980.00	948.00
P0300300	4	Coarse gravel	42	45	948.00	945.00
P0400100	1	Sand & gravel	0	22	955.00	933.00
P0400100	2	Silt	22	37	933.00	918.00
P0400100	3	Hardpan	37	96	918.00	859.00
P0400100	4	Soft rock (granite)	96	104	859.00	851.00
P0400200	1	Sand & gravel	0	85	990.00	905.00
P0500100	1	Sand	0	63	993.00	930.00
P0500200	1	Coarse sand	0	5	996.00	991.00

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
	P0500200	2	Coarse sand & gravel	5	25	991.00	971.00
	P0500200	3	Grey clay	25	35	971.00	961.00
	P0500200	4	Gravel & coarse sand	35	72	961.00	924.00
	P0500200	5	Bluegreen clay	72	72	924.00	924.00
	CF622	1	Sand	0	41	992.00	951.00
	P0500400	1	Top soil & sand	0	2	993.00	991.00
	P0500400	2	Sand	17	35	976.00	958.00
	P0500400	3	Granite	35	35	958.00	958.00
	P0600100	1	Sand	0	11	992.00	981.00
	P0600100	2	Sandy clay	11	15	981.00	977.00
	P0600100	3	Fine sand	15	19	977.00	973.00
	P0600100	4	Coarse sand & gravel	19	26	973.00	966.00
	P0600100	5	Fine sand w/ pebbles	26	60	966.00	932.00
	DS503	1	Sand & gravel	0	57	992.00	935.00
	P0700100	1	Sand	0	24	988.00	964.00
	P0700100	2	Sand & light clay/silt	24	33	964.00	955.00
	P0700100	3	Sand & gravel	33	45	955.00	943.00
	AX316	1	Sand	0	46	990.00	944.00
	P0800100	1	Sand & gravel	0	92	989.00	897.00
	P0800200	1	Sand	0	53	987.00	934.00
	P0800300	1	Sand & gravel	0	45	992.00	947.00
	P0800400	1	Sand & gravel	0	61	992.00	931.00
	AT207	1	Sand & gravel	0	47	991.00	944.00
	CM259	1	Sand	0	60	991.00	931.00
	CO501	1	Sand & gravel	0	49	991.00	942.00
	P0800800	1	Sand	0	85	920.00	835.00
	P0800800	2	Clay	85	129	835.00	791.00
	P0800800	3	Granite	129	140	791.00	780.00
	CH777	1	Sand	0	25	980.00	955.00
	CH777	2	Clay	25	28	955.00	952.00
	CH777	3	Sand	28	42	952.00	938.00
	CH777	4	Clay	42	59	938.00	921.00
	CH777	5	Soft shaley rock	59	60.75	921.00	919.25
	CH777	6	Granite	60.75	62	919.25	918.00
	CH777	7	Soft rock	62	64	918.00	916.00

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
	CH777	8	Granite	64	84	916.00	896.00
	CH496	1	Clay-sand mix	0	25	948.00	923.00
	CH496	2	Sandstone	25	80	923.00	868.00
	CH496	3	Granite	80	112	868.00	836.00
	P1500100	1	Sand	0	23	958.00	935.00
	P1500100	2	Granite	23	75	935.00	883.00
	P1500200	1	Sand	0	23	920.00	897.00
	P1500200	2	Gravel	23	25	897.00	895.00
	P1500200	3	Granite	25	158	895.00	762.00
	P1500300	1	Sand	0	27	920.00	893.00
	P1500300	2	Clay	27	34	893.00	886.00
	P1500300	3	Rock blue granite	34	78	886.00	842.00
	P1600100	1	Sand	0	18	940.00	922.00
	P1600100	2	Clay	18	46	922.00	894.00
	P1600100	3	Soft rock	46	50	894.00	890.00
	P1600100	4	Granite	50	70	890.00	870.00
	P1600200	1	Top soil	0	2	932.00	930.00
	P1600200	2	Sand & gravel	2	66	930.00	866.00
	P1700100	1	Sand & gravel	0	73	987.00	914.00
	CR279	1	Sand & gravel	0	27	984.00	957.00
	CR279	2	Sandy clay	27	33	957.00	951.00
	CR279	3	Sand & gravel	33	45	951.00	939.00
	AT279	1	Sand & gravel	0	44	984.00	940.00
	CF621	1	Sand	0	57	984.00	927.00
	CR685	1	Sand	0	55	984.00	929.00
	P1800100	1	Sand	0	58	994.00	936.00
	P1800100	2	Decomposed granite	60	103	934.00	891.00
	P1800100	3	Granite	60	103	934.00	891.00
	P1900100	1	Sand	0	30	980.00	950.00
	P1900100	2	Clay	30	46	950.00	934.00
	P1900100	3	Granite	46	300	934.00	680.00
	AT296	1	Sand & gravel	0	23	981.00	958.00
	AT296	2	Light silt	23	27	958.00	954.00
	AT296	3	Sand	27	46	954.00	935.00
	AT296	4	Sand & gravel	46	49	935.00	932.00

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
	AE294	1	Sand	0	26	981.00	955.00
	AE294	2	Clay	26	29	955.00	952.00
	AE294	3	Sand	29	48	952.00	933.00
	DN521	1	Coarse & med. sand	0	5	977.00	972.00
	DN521	2	Coarse sand & gr.	5	18	972.00	959.00
	DN521	3	Med sand, gravel & clay	18	27	959.00	950.00
	DN521	4	Coarse & med. sand	27	60	950.00	917.00
	DN521	5	Sandstone	60	65	917.00	912.00
	P1900500	1	Sand & gravel	0	100	965.00	865.00
	P2000100	1	Sand	0	15	979.00	964.00
	P2000100	2	Sand clay	15	25	964.00	954.00
	P2000100	3	Sand	25	40	954.00	939.00
	P2000100	4	Sand, gravel	40	75	939.00	904.00
	P2000100	5	Clay & sand	75	90	904.00	889.00
	P2000100	6	Sand & gravel	90	110	889.00	869.00
	P2000200	1	Sand-gravel	0	111	976.00	865.00
	P2000300	1	Sand	0	62	986.00	924.00
	P2000400	1	Sand & clay	0	7	983.00	976.00
	P2000400	2	Sandrock	7	50	976.00	933.00
	P2000400	3	Soft rock	50	65	933.00	918.00
	CR278	1	Sand & gravel	0	26	982.00	956.00
	CR278	2	Silty sand	26	30	956.00	952.00
	CR278	3	Sand & gravel	38	53	944.00	929.00
	AU944	1	Sand & gravel	0	65	985.00	920.00
	CY903	1	Sand & gravel	0	22	985.00	963.00
	CY903	2	Clay	22	32	963.00	953.00
	CY903	3	Sand & gravel	32	86	953.00	899.00
	AX303	1	Sand	0	25	985.00	960.00
	AX303	2	Clay	25	31	960.00	954.00
	P2100200	1	Sand	0	45	1000.00	955.00
	P2100200	2	Clay	45	55	955.00	945.00
	P2100200	3	Soft rock	55	70	945.00	930.00
	P2200100	1	Sand & gravel	0	30	935.00	905.00
	P2200100	2	Clay	30	34	905.00	901.00
	P2200100	3	Granite	34	85	901.00	850.00

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
	P2400100	1	Sand	0	60	975.00	915.00
	P2500100	1	Sand & gravel	0	61	969.00	908.00
	P2500200	1	Coarse sand	0	5	971.00	966.00
	P2500200	2	Coarse sand & gravel	5	20	966.00	951.00
	P2500200	3	Coarse sand, grav, grey	20	25	951.00	946.00
	P2500200	4	Grey clay	25	30	946.00	941.00
	P2500200	5	Coarse & medium sand	30	40	941.00	931.00
	P2500200	6	Medium & fine sand	40	45	931.00	926.00
	P2500200	7	Coarse sand & gravel	45	84	926.00	887.00
	P2500200	8	White clay & sandstone	84	84	887.00	887.00
	P2900100	1	Medium sand	0	5	979.00	974.00
	P2900100	2	Coarse sand & gravel	5	10	974.00	969.00
	P2900100	3	Med. sand & gravel	10	15	969.00	964.00
	P2900100	4	Medium sand	15	55	964.00	924.00
	P2900100	5	Coarse sand	55	65	924.00	914.00
	P2900100	6	Medium sand	65	70	914.00	909.00
	P2900100	7	Coarse sand	70	95	909.00	884.00
	P2900100	8	Medium sand	95	104	884.00	875.00
	P2900200	1	Sand	0	79	976.00	897.00
	P3000100	1	Sand	0	72	971.00	899.00
	P3000200	1	Sand	0	56	975.00	919.00
	P3000300	1	Sand	0	72	971.00	899.00
	P3100100	1	Fine sand	0	5	972.00	967.00
	P3100100	2	Fine sand & gravel	5	25	967.00	947.00
	P3100100	3	Fine sand	25	30	947.00	942.00
	P3100100	4	Fine sand & gravel	30	40	942.00	932.00
	P3100100	5	Coarse gravel	40	55	932.00	917.00
	P3100100	6	Fine sand	55	60	917.00	912.00
	P3100100	7	Fine sand & gravel	60	75	912.00	897.00
	P3100100	8	Coarse gravel	75	83	897.00	889.00
	AT266	1	Sand & gravel	0	40	974.00	934.00
	CO507	1	Sand & gravel	0	46	971.00	925.00
	AT267	1	Sand & gravel	0	47	974.00	927.00
	P3200100	1	Sand	0	15	973.00	958.00
	P3200100	2	Sand & gravel	15	20	958.00	953.00

Loc. code	UWN	Segment no.	Material	Depth from	Depth to	Elev from	Elev to
	P3200100	3	Sand	20	25	953.00	948.00
	P3200100	4	Sand, gravel	25	40	948.00	933.00
	P3200100	5	Sand	40	45	933.00	928.00
	P3200100	6	Sand & gravel	45	60	928.00	913.00
	P3200100	7	Gravel & granite	60	94	913.00	879.00
	P3200200	1	Sand and gravel	0	93	973.00	880.00
	P3200300	1	Sand	0	10	974.00	964.00
	P3200300	2	Sand & gravel	10	70	964.00	904.00
	P3200300	3	Med. sand-fine gravel	70	103	904.00	871.00
	AT263	1	Sand & gravel	0	48	970.00	922.00
	CR656	1	Sand	0	18	973.00	955.00
	CR656	2	Silty	18	37	955.00	936.00
	CR656	3	Sand	37	61	936.00	912.00
	P3300100	1	Sand & gravel	0	66	960.00	894.00
	P3300200	1	Sand & gravel	0	91	950.00	859.00
	P3300300	1	Sand	0	81.5	975.00	893.50
	P3300400	1	Sand	0	62.5	975.00	912.50
	AX305	1	Sand	0	62	955.00	893.00
	P3300600	1	Sand & gravel	0	46	1000.00	954.00
	AX326	1	Sand	0	27	1000.00	973.00
	AX326	2	Clay	27	32	973.00	968.00
	AX326	3	Sand	32	51	968.00	949.00

Appendix 3. Construction details for wells installed by the Port Edwards Groundwater Priority Watershed. Lengths and elevations are feet, casing diameters inches.

Site	WUWN	Casing Length	Screen Length	Casing Diam.	Typical Water Depth	Surface Elev'n	Well-top Elev'n
0101	EU653	8.35	5.0	2	7.5	990.72	993.27
0102	EU656	5.00	5.0	2	4.3	986.87	988.43
0103	EU605	24.68	5.0	2	4.4	986.80	989.30
0103	EU606	14.75	5.0	1	4.4	987.06	989.56
0103	EU607	37.40	5.0	1	4.5	987.29	989.59
0103	EU654	6.71	5.0	2	4.7	986.72	988.49
0103	EU655	46.40	5.0	2	4.9	986.89	988.63
0501	EU651	13.68	5.0	2	13.9	995.16	997.22
0502	EU665	38.36	5.0	1	11.6	997.00	999.33
0502	EU666	49.49	5.0	1	11.6	997.00	999.33
0502	EU667	57.60	5.0	1	11.9	997.00	999.33
0503	EU621	40.90	5.0	1	19.8	993.05	994.77
0503	EU622	49.46	5.0	1	20.0	993.05	994.76
0503	EU763	18.39	5.0	2	17.6	993.18	995.02
0571	EU635	10.00	5.0	2	11.5	991.14	993.16
0572	EU691	42.99	5.0	2	14.4	991.57	993.70
0572	EU692	12.31	5.0	2	13.3	991.15	993.59
0573	EU459	10.00	5.0	2	11.3	992.91	995.68
0574	EU638	12.72	5.0	2	13.7	992.24	995.24
0575	EU639	14.56	5.0	2	15.3	991.18	994.18
0576	EU640	14.90	5.0	2	15.4	990.86	993.86
0577	EU641	11.88	5.0	2	13.4	990.59	993.07
0601	EU652	8.03	5.0	2	8.4	992.72	996.06
0602	EU668	29.60	5.0	1	6.7	993.00	995.00
0602	EU669	44.50	5.0	1	7.1	993.00	995.00
0603	EU601	19.45	5.0	2	9.2	989.54	991.64
0603	EU602	39.30	5.0	1	9.2	989.59	991.89
0603	EU603	50.10	5.0	1	9.1	989.52	991.82
0603	EU604	57.95	5.0	1	9.7	989.57	991.87
0603	EU762	10.37	5.0	2	10.3	989.23	992.08
0671	EU631	8.14	5.0	2	8.8	987.37	990.37
0672	EU632	8.62	5.0	2	9.4	989.45	992.45
0673	EU719	7.14	5.0	2	8.0	989.33	992.56
0674	EU633	8.56	5.0	2	10.5	991.79	994.34
0674	EU634	34.16	5.0	2	11.0	991.76	994.38
0701	EU586	10.78	2.0	1	13.0	987.00	988.92
0701	EU587	12.84	2.0	1	13.0	987.00	988.92
0701	EU588	14.75	2.0	1	13.0	987.00	988.92
0701	EU589	16.81	2.0	1	13.0	987.00	988.91
0701	EU590	18.68	2.0	1	13.0	987.00	988.92
0702	EU659	18.99	5.0	2	15.6	983.85	985.77
0702	EU660	43.00	5.0	2	17.0	986.52	988.12
0801	EU661	19.30	5.0	2	15.2	986.22	988.27
0801	EU662	35.36	5.0	1	16.2	976.26	978.31
0801	EU663	43.47	5.0	1	16.6	985.95	988.00
0801	EU664	51.45	5.0	1	16.6	986.01	988.06
0802	EU623	38.53	5.0	1	28.8	988.17	991.27
0802	EU624	52.58	5.0	1	28.8	988.16	991.29
0802	EU625	69.23	5.0	1	29.0	988.24	991.25
0802	EU761	28.38	5.0	2	28.6	988.21	990.51
1201	EU657	6.11	5.0	2	4.4	980.31	983.11

Appendix 3 (continued). Construction details for wells installed by the Port Edwards Groundwater Priority Watershed. Lengths and elevations are feet, casing diameters inches.

Site	WUWN	Casing Length	Screen Length	Casing Diam.	Typical Water Depth	Surface Elev'n	Well-top Elev'n
1301	EU608	20.12	5.0	2	5.0	981.83	983.73
1301	EU609	39.42	5.0	1	4.8	981.98	983.46
1301	EU610	45.85	5.0	1	4.6	981.98	983.50
1301	EU611	54.37	5.0	1	5.0	981.98	983.42
1301	EU658	8.12	5.0	2	7.0	981.86	984.82
1701	EU581	20.67	2.0	1	21.5	984.00	986.40
1701	EU582	22.59	2.0	1	21.6	984.00	986.48
1701	EU583	24.66	2.0	1	21.6	984.00	986.44
1701	EU584	26.53	2.0	1	21.6	984.00	986.41
1701	EU585	28.61	2.0	1	21.6	984.00	986.44
1702	EU600	61.58	5.0	1	48.0	985.60	987.76
1702	EU629	71.44	5.0	1	47.9	985.57	987.73
1702	EU630	81.12	5.0	1	47.8	985.59	987.76
1702	EU767	48.29	5.0	2	44.4	985.70	988.25
1801	EU764	10.00	5.0	2	13.9	978.57	981.47
1802	EU591	6.47	2.0	1	8.7	977.77	978.92
1802	EU592	8.06	2.0	1	8.7	977.78	978.93
1802	EU593	10.05	2.0	1	8.7	977.77	978.95
1802	EU594	12.16	2.0	1	8.7	977.78	978.93
1802	EU595	14.55	2.0	1	8.7	977.77	978.97
1803	EU626	31.71	5.0	1	22.2	982.93	985.66
1803	EU627	54.35	5.0	1	22.2	982.92	985.65
1803	EU628	67.67	5.0	1	22.2	983.00	985.74
1803	EU765	24.00	5.0	2	23.7	983.87	986.27
1803	EU766	41.00	5.0	2	23.6	982.99	986.13
2001	EU769	68.07	5.0	2	19.8	974.92	975.63
2001	EU770	15.79	5.0	2	20.0	974.92	975.25
2401	EU612	9.87	5.0	1	8.7	976.83	978.60
2401	EU613	22.15	5.0	1	8.3	976.83	978.60
2401	EU614	36.90	5.0	1	8.1	978.00	979.87
2401	EU615	50.03	5.0	1	8.4	976.82	978.69
2401	EU616	63.68	5.0	1	9.0	976.83	978.70
2402	EU682	7.87	5.0	2	6.8	972.43	975.43
2403	EU681	13.34	5.0	2	12.8	975.98	978.48
2501	EU683	9.94	5.0	2	10.3	969.30	972.01
2901	EU561	59.38	5.0	1	52.6	980.39	982.02
2901	EU562	67.46	5.0	1	52.6	980.08	981.73
2901	EU563	76.04	5.0	1	52.7	980.09	981.74
2901	EU768	51.98	5.0	2	53.7	980.21	982.81
2902	EU564	57.95	5.0	1	46.0	976.31	978.64
2902	EU565	66.18	5.0	1	46.0	976.34	978.67
2902	EU566	70.82	5.0	1	46.0	976.34	978.66
2902	EU686	46.20	5.0	2	46.5	976.39	979.11
3001	EU617	20.14	5.0	2	17.2	976.40	979.40
3001	EU618	51.36	5.0	1	16.1	976.28	978.25
3001	EU619	59.37	5.0	1	16.1	976.28	978.22
3001	EU620	70.68	5.0	1	16.1	976.31	978.28
3002	EU576	28.88	5.0	1	25.7	975.68	978.39
3002	EU577	33.71	5.0	1	25.7	975.67	978.37
3002	EU578	48.62	5.0	1	25.7	975.66	978.36
3002	EU579	60.94	5.0	1	25.7	975.68	978.38
3002	EU580	78.21	5.0	1	25.7	975.68	978.39
3003	EU684	29.71	5.0	2	24.6	971.21	973.49
3003	EU685	55.00	5.0	2	24.7	971.41	973.62

Appendix 3 (continued). Construction details for wells installed by the Port Edwards Groundwater Priority Watershed. Lengths and elevations are feet, casing diameters inches.

Site	WUWN	Casing Length	Screen Length	Casing Diam.	Typical Water Depth	Surface Elev'n	Well-top Elev'n
3101	EU689	14.52	5.0	2	15.9	966.13	968.95
3201	EU572	29.64	5.0	1	28.3	970.39	972.51
3201	EU573	41.45	5.0	1	28.3	970.40	972.47
3201	EU574	53.26	5.0	1	28.4	970.40	972.52
3201	EU575	65.98	5.0	1	28.3	970.41	972.47
3202	EU688	31.80	5.0	2	31.5	970.29	973.04
3203	EU567	54.41	5.0	1	44.0	971.68	973.43
3203	EU568	62.21	5.0	1	44.0	971.69	973.45
3203	EU569	70.71	5.0	1	44.0	971.69	973.45
3203	EU570	78.30	5.0	1	44.0	971.68	973.45
3203	EU571	86.58	5.0	1	43.6	971.69	973.45
3203	EU687	44.73	5.0	2	45.0	971.29	974.27
3601	EU596	18.00	5.0	2	12.7	969.95	972.65
3601	EU597	38.10	5.0	1	14.8	969.78	972.28
3601	EU598	57.78	5.0	1	14.4	969.77	972.27
3601	EU599	67.68	5.0	1	15.0	969.76	972.26
3602	EU690	10.35	5.0	2	10.4	966.97	968.71

APPENDIX 4

FIELD RECORDS FROM GROWERS

1992

FIELD 1

Crop Sweet corn, variety *Excellency*
Plant 5/18/92
Harvest 9/8-11/92
Yield 9 US ton/acre

Nutrient applications

3/2	81K (163 lb of 0-0-60)
5/18	154 lb starter: 15N 11P 16K 19S 10Mg 0.2B 2Zn urea
6/9	46N urea
6/24	69N urea
7/17	22N ((1000gal÷133A)*10.24lb/gal*28%) liquid ammonium nitrate
7/28	25N liquid ammonium nitrate
8/4	14N liquid ammonium nitrate
total	191N 11P 97K

Pesticide applications

5/22	Dual (metolachlor), 1.5 pint, west ½
5/22	Lasso (alachlor), 3 pint, east ½
6/20	Buctril (bromoxynil), 1.2 pint
7/16	Aatrex (atrazine), ¾ lb, west ½

Irrigations

6/10	0.4 inch	7/28	0.5 inch	
6/28	0.4 inch	8/2	0.5 inch	
7/3	0.5 inch	8/6	0.5 inch	
7/8	0.5 inch	8/11	0.5 inch	
7/10	0.5 inch	8/16	0.5 inch	
7/17	0.2 inch	8/19	0.5 inch	
7/18	0.5 inch	8/22	0.5 inch	total=6.5 inches

FIELD 2

Crop Sweet corn, variety *Excellency*
Plant 5/18/92
Harvest 9/8-11/92
Yield 11 US ton/acre

Nutrient applications

3/2	81K (163 lb of 0-0-60)
5/20	154 lb starter: 15N 11P 16K 19S 10Mg 0.2B 2Zn urea
6/9	46N urea
6/24	69N urea
7/17	27N ((1000gal÷108A)*10.24lb/gal*28%) liquid ammonium nitrate
7/28	33N liquid ammonium nitrate
8/4	33N liquid ammonium nitrate
total	223N 11P 97K

Pesticide applications

5/24	Aatrex (atrazine), 1.5 pint
5/24	Dual (metolachlor), 1.5 pint, west ½
5/24	Lasso (alachlor), 3 pint, east ½

Irrigations

6/10	0.4 inch	8/2	0.5 inch	
6/27	0.4 inch	8/5	0.5 inch	
7/3	0.5 inch	8/11	0.5 inch	
7/8	0.5 inch	8/16	0.5 inch	
7/17	0.2 inch	8/19	0.5 inch	
7/18	0.5 inch	8/22	0.5 inch	
7/27	0.5 inch			total=6 inches

FIELD 3

Crop	Sweet corn
Plant	6/14/92
Harvest	10/5/92
Yield	4.5 US ton/acre

Nutrient applications

6/1	100K (200 lb of 0-0-60)
6/14	23N 25P 47K (285 lb of 8-20-20)
7/10	32N (100 lb of 32-0-0, irrig.)
7/23	38N (120 lb of 32-0-0, irrig.)
8/10	32N (100 lb of 32-0-0, irrig.)
8/24	38N (120 lb of 32-0-0, irrig.)
total	164N 25P 147K

Pesticide applications

6/26	Bullet (atrazine + alachlor), 5 pint, east ½
	Arena (alachlor), 3 pint, west ½

Irrigations

total≈5 inches
Detailed records were not kept by the grower.

FIELD 4

Crop	Potatoes, variety *Russett Burbank*
Plant	5/6/92
Harvest	10/6/92
Yield	405 cwt, NE½;415 cwt, SW ½

Nutrient applications

4/5	199K (400 lb of 0-0-60), 34S 88Ca (200 lb of cal-sul)
5/6	38N 66P 55K (475 lb of 8-32-14)
5/27	82N (240 lb of 34-0-0), 41S 53Ca (240 lb of cal-sul)
6/10-30	101N 44Ca (630 lb of 16-0-0-7Ca, split in 4 weekly irrig. applications)
7/1-30	50K (500 lb of 0-0-12, split in 4 weekly irrig. applications)
7/15	28N (90 lb of 32-0-0, irrig.)
total	249N 66P 105K 75S 185Ca

Pesticide applications

5/27	Furadan (carbofuran), 1 qt, banded on SW½
5/28	Sencor (metribuzin), ⅓lb, ground sprayed
6/20	Ambush (permethrin), 8 oz, aerial (NE½)
6/30	Sencor, ⅓lb, ground sprayed
7/4	Ambush, 8 oz, ground sprayed
7/4-8/24	Penncozeb (mancozeb), 1 lb, ground sprayed 7 times
8/14	Super Tin (triphenyltin hydroxide), 4 oz, ground sprayed with Penncozeb
8/31	Bravo (chlorothalonil), 1 pt, ground sprayed
9/18	Diquat (vine killer), 1 pt

Irrigations

total≈10 inches
Detailed records were not kept by the grower.

1993

FIELD 1

Crop Potatoes, variety *Russett Burbank*
Plant 5/1-16/93
Harvest 10/5-11/93
Yield 300 cwt/A

Nutrient applications

3/24	249K (500 lb of 0-0-60)
4/26	350 lb starter:24N 48P 48K 11S 8Mg 0.35B 2Zn urea
6/1	110N urea
6/27	138N urea
7/20	25N ((1150gal÷133A)*10.24lb/gal*28%) liquid ammonium nitrate
total	297N 48P 277K

Pesticide applications

5/17	Dual (metolachlor), 1.5 pint + Sencor (metribuzin), 0.45 lb
6/26	Sulfur, 3 lb + Asana (esfenvalerate), 8 oz
6/28	Sencor, 0.5 lb
7/8	Bravo (chlorothalonil), 1 pint
7/15	Bravo, 1⅓ pint + Sulfur, 3 lb
7/22	Bravo, 1⅓ pint + Sulfur, 3 lb
7/28	Bravo, 1⅓ pint + Furadan (carbofuran), 1 qt
8/4	Bravo, 2⅛ pint + Monitor (methamidophos), 1 qt
8/11	Bravo, 2⅛ pint + Sulfur, 3 lb
8/18	Bravo, 2⅛ pint
8/25	Bravo, 2⅛ pint
9/1	Bravo, 2⅛ pint
9/8	Bravo, 2⅛ pint
9/15	Bravo, 2⅛ pint
9/19	Diquat, 1 pint
9/24	Diquat, 1 pint

Irrigations

6/14	0.2 inch	7/29	0.4 inch	
6/25	0.5 inch	8/3	0.4 inch	
6/27	0.5 inch	8/7	0.4 inch	
6/29	0.5 inch	8/12	0.4 inch	
7/20	0.2 inch	8/19	0.4 inch	
7/21	0.4 inch	8/23	0.3 inch	total=4.6 inches

FIELD 2

Crop	Potatoes, variety *Russett Burbank*
Plant	4/30-5/1/93
Harvest	9/28-10/2/93
Yield	414 cwt/A

Nutrient applications

3/24	299K (600 lb of 0-0-60)
4/26	350 lb starter:24N 48P 48K 11S 8Mg 0.35B 2Zn urea
5/28	110N urea
6/21-27	138N (south ½) urea
6/28	79N ((1300gal÷54A)*10.24lb/gal*32%) (north ½)*
7/19	32N ((1200gal÷108A)*10.24lb/gal*28%)*
8/14	20N ((7gal/A)*10.24lb/gal*28%) (north ½)*
8/14	14N ((5gal/A)*10.24lb/gal*28%) (south ½)*
total	265N,north½; 318N,south½ 48P 347K

*=liquid ammonium nitrate

Pesticide applications

5/12	Dual (metolachlor), 1.5 pint + Sencor (metribuzin), 0.45 lb
5/16	Furadan (carbofuran), 1 qt/A, applied on field perimeter and down field center (W - E)
6/26	Sulfur, 3 lb + Asana (esfenvalerate), 8 oz.
6/28	Sencor, 0.5 lb
7/8	Bravo (chlorothalonil), 1 pint
7/15	Bravo, 1⅓ pint + Sulfur, 3 lb
7/22	Bravo, 1⅓ pint + Sulfur, 3 lb
7/28	Bravo, 1⅓ pint
8/4	Bravo, 2⅛ pint + Monitor (methamidophos), 1 qt
8/11	Bravo, 2⅛ pint + Sulfur, 3 lb
8/18	Bravo, 2⅛ pint
8/25	Bravo, 2⅛ pint
9/1	Bravo, 2⅛ pint
9/8	Bravo, 2⅛ pint
9/11	Diquat, 1 pint
9/16	Diquat, 1 pint

Irrigations

6/14	0.2 inch	7/29	0.4 inch	
6/25	0.5 inch	8/3	0.4 inch	
6/27	0.5 inch	8/7	0.4 inch	
6/29	0.5 inch	8/12	0.4 inch	
7/19	0.4 inch	8/19	0.4 inch	
7/21	0.4 inch	8/23	0.5 inch	total=5 inches

FIELD 3

Crop Sweet corn, variety *Rival*
Plant 4/30-5/5/93
Harvest
Yield 3.58 US ton/acre

Nutrient applications

4/25	104K (200 lb of 0-0-62.5)
4/30-5/5	14N 26P 58K (280 lb of 5-21-25)
6/3	8N (28% through irrigator)
6/15	31N (")
6/16	31N (")
6/28	34N (")
7/12	41N (")
7/12	2N 7S (27 lb of 8-0-0-26S)
7/16	8N (28% through irrigator)
total	169N 26P 58K 7S

Pesticide applications

5/7	Bullet (atrazine + alachlor), 2 qt + Arena (alachlor), 1 pt, east ½ (53acres) Arena (alachlor), 2 qt, west ½ (50acres)
6/16	Buctril (bromoxynil), 1.5 pt, west

Irrigations

Irrigation was not recorded by the grower.

FIELD 4

Crop Sweet corn, variety *Rival*
Plant 5/5/93
Harvest
Yield 4.1 US ton/acre

Nutrient applications

4/25	104K (200 lb of 0-0-62.5)
5/5	14N 26P 58K (280 lb of 5-21-25), north ½ 0N 22P 58K (280 lb of 0-18-25), south ½
6/2	17N (28% through irrigator)
6/14	64N (")
6/29	37N (")
7/14	12N (")
7/14	4N 13S (50 lb 8-0-0-26S)
7/18	20N (28% through irrigator)
total	168N 26P 58K 13S north ½ & 154N 22P 58K 13S south ½

Pesticide applications

5/8 Bullet (alachlor + atrazine), 2 qt
Arena (alachlor), 1 pt

Irrigations

Irrigation was not recorded by the grower.

APPENDIX 5

PESTICIDE ANALYTES AND DETECTION LIMITS

University of Wisconsin-Stevens Point
College of Natural Resources
Environmental Task Force Lab
Stevens Point, WI 54481-3897 (715) 346-3209

MODIFIED EPA METHOD 8270

SAMPLE NAME:	EU 687	DATE SAMPLED:	08-Mar-93
SAMPLE SITE:	Port Edwards	DATE RECEIVED:	09-Mar-93
SAMPLE NUMBER:	77-93-1	DATE EXTRACTED:	12-Mar-93
SAMPLE MATRIX:	Groundwater	DATE ANALYZED:	19-Mar-93

All concentrations and levels of quantitation (LOQ) are reported in parts per billion (ug/L).

COMPOUND	LOQ	CONCENTRATION
EPTC	0.1	ND
Butylate	0.1	ND
Pebulate	0.1	ND
Molinate	0.2	ND
Cycloate	0.1	ND
Trifluralin	0.2	ND
Terbacil	0.8	ND
Bromacil	0.5	ND
Isopropalin	0.1	ND
Oxyfluorfen	0.1	ND
Hexazinone	0.5	ND
Vernolate	0.3	ND
Propachlor	0.4	ND
Benfluralin	0.4	ND
Simazine	0.5	ND
Propazine	0.4	ND
Profluralin	0.3	ND
Pendimethalin	0.3	ND
Oxidiazon	0.3	ND
Metolachlor	0.4	ND
Metribuzin	0.2	ND
Atrazine	0.2	ND
De-Ethyl Atrazine	0.1	ND
De-Isopropyl Atrazine	0.2	ND
Alachlor	0.1	ND
Carbofuran	0.1	ND
Cyanazine	0.2	ND

ND = This compound was not detected.
Note =

6/30/93

University of Wisconsin-Stevens Point
College of Natural Resources
Environmental Task Force Lab
Stevens Point, WI 54481-3897 (715) 346-3209

ENVIRONMENTAL TASK FORCE LABORATORY
NITROGEN AND PHOSPHOROUS CONTAINING PESTICIDE REPORT

EPA METHOD 507

METHOD DETECTION LIMITS - As determined by Code of Federal Regulations, No. 40, part B, Appendix 136. Definition and Procedure for the Determination of the Method Detection Limit - Revision 1.11. 1986.

All concentrations are reported in parts per billion (ug/L).

TRADE NAME	*OTHER NAMES*	*MDL*
Vernam	Vernolate	0.5
*Propachlor	**********	0.5
*Balan	Benfluralin	0.4
Simazine	**********	0.5
Propazine	**********	0.5
*Tolban	Profluralin	0.5
Dual	Metolachlor	0.6
*Prowl	Pendimethalin	0.5
*Oxadiazon	**********	0.5
Eptam	EPTC	0.3
Sutan	Butylate	0.1
Tillam	Pebulate	0.2
Ordram	Molinate	0.3
Ro-Neet	Cycloate	0.2
*Treflan	Trifluralin	0.2
Atrazine	**********	0.1
De-ethyl Atrazine	**********	NA
Terbacil	**********	0.2
Sencor	Metribuzin	0.1
Bromacil	**********	0.3
*Paarlan	Isopropalin	0.1
*Goal	Oxyfluorfen	0.1
Velpar	Hexazinone	0.2
Diazinon	**********	0.1
Lasso	Alachlor	NA

**Indicates that this compound is not covered by EPA Method 507.*

NA = = >NOT AVAILABLE

APPENDIX 6

PESTICIDE DETECTIONS IN MULTILEVEL PIEZOMETERS WITHIN IRRIGATED FIELDS

Field	Well	Port	Date	Atrazine	De-ethyl atrazine	De-isopropyl atrazine	Alachlor	Metolachlor	Metribuzin	Carbofuran
1	6	02	09/20/92	<0.1	<0.1	ND	ND[1]	<0.6	0.3	ND
1	6	04	09/20/92	0.3	<0.1	ND	ND	<0.6	0.6	ND
1	6	06	09/20/92	<0.1	<0.1	ND	ND	<0.6	0.5	ND
1	6	08	09/20/92	0.5	0.8	ND	ND	<0.6	<0.1	ND
1	6	10	09/20/92	0.5	<0.1	ND	ND	<0.6	<0.1	ND
1	6	12	09/20/92	<0.1	<0.1	ND	ND	<0.6	<0.1	ND
1	6	14	09/20/92	0.8	<0.1	ND	ND	<0.6	<0.1	ND
1	6	16	09/20/92	<0.1	<0.1	ND	ND	<0.6	<0.1	ND
1	6	17	09/20/92	0.5	<0.1	ND	ND	0.6	<0.1	ND
1	6	18	09/20/92	0.4	<0.1	ND	ND	<0.6	<0.1	ND
2	3	02	09/21/92	<0.1	<0.1	ND	ND	1.0	0.5	ND
2	3	04	09/21/92	<0.1	<0.1	ND	ND	1.2	0.6	ND
2	3	06	09/21/92	0.4	<0.1	ND	ND	1.1	0.5	ND
2	3	08	09/21/92	<0.1	<0.1	ND	ND	1.6	0.7	ND
2	3	10	09/21/92	<0.1	<0.1	ND	ND	0.9	0.4	ND
2	3	12	09/21/92	0.8	<0.1	ND	ND	2.0	0.4	ND
2	3	14	09/21/92	0.5	<0.1	ND	ND	1.2	0.4	ND
2	3	16	09/21/92	0.8	0.9	ND	ND	1.2	1.2	ND
2	3	17	09/21/92	0.9	<0.1	ND	ND	1.0	<0.1	ND
2	3	18	09/21/92	0.5	<0.1	ND	ND	<0.6	<0.1	ND
3	4	02	09/26/92	0.4	<0.1	ND	ND	<0.6	<0.1	ND
3	4	04	09/26/92	0.4	<0.1	ND	ND	<0.6	<0.1	ND
3	4	06	09/26/92	0.4	<0.1	ND	ND	<0.6	<0.1	ND
3	4	08	09/26/92	<0.1	<0.1	ND	ND	<0.6	<0.1	ND
3	4	10	09/26/92	<0.1	<0.1	ND	ND	<0.6	<0.1	ND
3	4	12	09/26/92	<0.1	<0.1	ND	ND	<0.6	<0.1	ND
3	4	14	09/26/92	<0.1	<0.1	ND	ND	<0.6	<0.1	ND
3	4	16	09/26/92	<0.1	<0.1	ND	ND	<0.6	<0.1	ND
1	6	02	03/28/93	<0.2	<0.1	<0.2	<0.1	<0.4	0.3	<0.1
1	6	04	03/28/93	<0.2	<0.1	<0.2	<0.1	<0.4	0.3	<0.1
1	6	06	03/28/93	0.2	<0.1	<0.2	<0.1	<0.4	0.2	<0.1

[1]ND = not determined

Field	Well	Port	Date	Atrazine	De-ethyl atrazine	De-isopropyl atrazine	Alachlor	Metolachlor	Metribuzin	Carbofuran
1	6	08	03/28/93	0.3	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	10	03/28/93	0.3	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	12	03/28/93	0.3	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	14	03/28/93	0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	16	03/28/93	<0.2	0.4	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	18	03/28/93	<0.2	0.4	<0.2	<0.1	<0.4	<0.2	<0.1
2	3	02	04/03/93	0.2	<0.1	<0.2	<0.1	0.4	0.9	0.4
2	3	04	04/03/93	0.2	0.1	<0.2	<0.1	0.4	0.8	0.2
2	3	06	04/03/93	<0.2	0.1	<0.2	<0.1	<0.4	0.6	<0.1
2	3	08	04/03/93	<0.2	0.1	<0.2	<0.1	<0.4	0.4	<0.1
2	3	10	04/03/93	<0.2	0.1	<0.2	<0.1	<0.4	0.2	<0.1
2	3	12	04/03/93	<0.2	0.2	<0.2	<0.1	<0.4	0.2	<0.1
2	3	14	04/03/93	0.3	0.3	<0.2	<0.1	0.7	0.2	<0.1
2	3	16	04/03/93	0.6	0.5	<0.2	<0.1	1.0	0.3	<0.1
2	3	17	04/03/93	0.6	0.5	<0.2	<0.1	0.5	0.2	<0.1
3	4	02	03/27/93	<0.2	0.1	<0.2	<0.1	0.4	0.6	12.5
3	4	04	03/27/93	<0.2	0.1	<0.2	<0.1	<0.4	0.2	12.7
3	4	06	03/27/93	<0.2	0.1	<0.2	<0.1	0.4	0.3	12.8
3	4	08	03/27/93	<0.2	0.1	<0.2	<0.1	0.4	0.3	2.8
3	4	10	03/27/93	<0.2	0.1	<0.2	<0.1	0.4	0.3	0.2
3	4	12	03/27/93	<0.2	0.1	<0.2	<0.1	0.4	0.3	<0.1
3	4	14	03/27/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	0.2
3	4	16	03/27/93	<0.2	0.2	<0.2	<0.1	<0.4	<0.2	0.5
3	4	17	03/27/93	<0.2	0.2	<0.2	<0.1	<0.4	<0.2	0.2
4	3	02	04/08/93	<0.2	0.1	<0.2	<0.1	11.6	3.7	0.2
4	3	04	04/08/93	<0.2	0.1	<0.2	<0.1	8.6	2.9	1.9
4	3	06	04/08/93	<0.2	0.1	<0.2	<0.1	7.0	1.8	2.9
4	3	08	04/08/93	<0.2	0.1	<0.2	<0.1	5.2	1.9	3.7
4	3	10	04/08/93	<0.2	0.2	<0.2	<0.1	4.2	1.8	3.2
4	3	12	04/08/93	0.2	0.4	<0.2	<0.1	1.2	1.4	2.1
4	3	14	04/08/93	0.3	0.4	<0.2	<0.1	0.7	1.2	1.6
4	3	16	04/08/93	0.5	0.4	<0.2	<0.1	<0.4	2.6	0.3
4	3	17	04/08/93	0.7	0.4	<0.2	<0.1	<0.4	4.7	0.2
1	3	00	12/01/93	<0.2	0.2	<0.2	<0.1	0.4	0.6	<0.1
1	3	01	12/01/93	<0.2	0.2	<0.2	<0.1	0.5	0.7	<0.1
1	3	02	12/01/93	<0.2	0.2	<0.2	<0.1	<0.4	0.6	<0.1
1	3	04	12/01/93	0.2	0.4	<0.2	<0.1	<0.4	0.2	<0.1
1	3	06	12/01/93	<0.2	0.3	<0.2	<0.1	<0.4	0.3	0.2

Field	Well	Port	Date	Atrazine	De-ethyl atrazine	De-isopropyl atrazine	Alachlor	Metolachlor	Metribuzin	Carbofuran
1	3	10	12/01/93	0.2	0.4	<0.2	<0.1	0.2	7.6	3.2
1	3	14	12/01/93	<0.2	0.4	<0.2	<0.1	<0.4	7.4	0.7
1	3	16	12/01/93	<0.2	0.4	<0.2	<0.1	0.4	12.0	2.0
1	3	17	12/01/93	<0.2	0.4	<0.2	<0.1	<0.4	24.6	4.8
1	3	18	12/01/93	<0.2	0.3	<0.2	<0.1	0.4	20.3	20.6
1	4	00	12/01/93	<0.2	<0.1	<0.2	<0.1	0.7	1.1	<0.1
1	4	01	12/01/93	0.2	<0.1	<0.2	<0.1	0.5	0.8	<0.1
1	4	02	12/01/93	0.2	0.3	<0.2	<0.1	0.4	0.6	0.1
1	4	04	12/01/93	0.2	<0.1	<0.2	<0.1	<0.4	0.4	<0.1
1	4	06	12/01/93	0.2	0.5	<0.2	<0.1	<0.4	0.3	0.2
1	4	08	12/01/93	0.3	0.5	<0.2	<0.1	<0.4	0.2	0.2
1	4	10	12/01/93	0.5	1.0	<0.2	<0.1	0.5	0.5	2.7
1	4	12	12/01/93	0.7	1.5	<0.2	<0.1	0.6	0.6	4.0
1	4	14	12/01/93	0.8	1.6	<0.2	<0.1	0.5	0.6	3.3
1	4	16	12/01/93	1.3	1.9	<0.2	<0.1	0.7	1.0	4.6
1	4	17	12/01/93	1.4	2.1	0.2	<0.1	1.9	3.0	11.3
1	4	18	12/01/93	1.2	1.9	0.2	<0.1	1.0	2.3	7.0
1	5	00	12/03/93	<0.2	0.3	<0.2	<0.1	<0.4	0.5	<0.1
1	5	01	12/03/93	0.2	0.3	<0.2	<0.1	<0.4	0.2	<0.1
1	5	02	12/03/93	0.4	0.5	<0.2	<0.1	<0.4	<0.2	<0.1
1	5	04	12/03/93	0.5	0.7	<0.2	<0.1	<0.4	<0.2	<0.1
1	5	06	12/03/93	<0.2	0.4	<0.2	<0.1	<0.4	0.2	<0.1
1	5	08	12/03/93	0.4	0.4	<0.2	<0.1	<0.4	<0.2	<0.1
1	5	10	12/03/93	1.2	1.5	<0.2	<0.1	0.5	<0.2	<0.1
1	5	12	12/03/93	0.8	0.9	<0.2	<0.1	0.4	<0.2	<0.1
1	5	14	12/03/93	0.4	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	5	16	12/03/93	0.4	0.3	<0.2	<0.1	<0.4	<0.2	<0.1
1	5	17	12/03/93	0.7	0.5	<0.2	<0.1	0.8	1.1	1.2
1	5	18	12/03/93	0.6	0.7	<0.2	<0.1	1.9	2.7	2.8
1	6	00	12/04/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	01	12/04/93	<0.2	<0.1	<0.2	<0.1	<0.4	0.2	<0.1
1	6	02	12/04/93	<0.2	<0.1	<0.2	<0.1	<0.4	0.2	<0.1
1	6	04	12/04/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	06	12/04/93	0.4	0.8	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	08	12/04/93	1.1	1.8	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	10	12/04/93	1.4	2.5	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	12	12/04/93	1.6	2.6	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	14	12/04/93	1.1	1.5	<0.2	<0.1	<0.4	<0.2	<0.1

Field	Well	Port	Date	Atrazine	De-ethyl atrazine	De-isopropyl atrazine	Alachlor	Metolachlor	Metribuzin	Carbofuran
1	6	16	12/04/93	0.3	0.5	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	17	12/04/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
1	6	18	12/04/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	0.3
2	3	00	11/03/93	0.4	0.5	<0.2	<0.1	0.6	0.8	0.7
2	3	01	11/03/93	0.2	0.3	<0.2	<0.1	0.5	0.6	0.7
2	3	02	11/03/93	<0.2	0.1	<0.2	<0.1	<0.4	0.4	0.2
2	3	04	11/03/93	0.2	0.3	<0.2	<0.1	0.4	0.6	0.4
2	3	06	11/03/93	0.2	0.4	<0.2	<0.1	<0.4	0.5	0.3
2	3	08	11/03/93	0.2	0.4	<0.2	<0.1	<0.4	0.5	<0.1
2	3	10	11/03/93	0.2	0.4	<0.2	<0.1	<0.4	0.4	<0.1
2	3	12	11/03/93	0.5	0.9	<0.2	<0.1	0.5	0.7	<0.1
2	3	14	11/03/93	0.6	0.9	<0.2	<0.1	0.6	0.9	<0.1
2	3	16	11/03/93	0.5	0.6	<0.2	<0.1	0.8	1.6	<0.1
2	3	17	11/03/93	0.8	1.2	<0.2	<0.1	1.1	3.3	<0.1
2	3	18	11/03/93	0.8	1.1	<0.2	<0.1	1.6	5.3	<0.1
2	4	00	10/27/93	<0.2	0.3	<0.2	<0.1	0.5	0.5	<0.1
2	4	01	10/27/93	<0.2	0.3	<0.2	<0.1	0.4	0.4	<0.1
2	4	02	10/27/93	<0.2	0.2	<0.2	<0.1	<0.4	0.3	<0.1
2	4	04	10/27/93	0.2	0.4	<0.2	<0.1	<0.4	0.3	<0.1
2	4	06	10/27/93	0.3	1.2	<0.2	<0.1	0.4	0.5	<0.1
2	4	08	10/27/93	0.3	0.7	<0.2	<0.1	<0.4	0.3	<0.1
2	4	10	10/27/93	0.2	0.8	<0.2	<0.1	<0.4	<0.2	<0.1
2	4	12	10/27/93	<0.2	0.6	<0.2	<0.1	<0.4	<0.2	<0.1
2	4	14	10/27/93	0.2	0.7	<0.2	<0.1	<0.4	0.5	<0.1
2	4	16	10/27/93	0.2	0.7	<0.2	<0.1	<0.4	1.4	<0.1
2	4	17	10/27/93	0.2	0.7	<0.2	<0.1	<0.4	2.3	<0.1
2	4	18	10/27/93	0.2	0.8	<0.2	<0.1	<0.4	4.3	<0.1
2	5	00	10/27/93	<0.2	0.1	<0.2	<0.1	<0.4	0.2	0.1
2	5	01	10/27/93	0.2	0.1	<0.2	<0.1	<0.4	0.3	0.2
2	5	02	10/27/93	0.2	0.2	<0.2	<0.1	<0.4	0.3	0.2
2	5	04	10/27/93	<0.2	0.2	<0.2	<0.1	<0.4	0.2	0.2
2	5	06	10/27/93	0.2	0.4	0.4	<0.1	0.4	0.2	0.2
2	5	08	10/27/93	0.2	0.5	<0.2	<0.1	0.4	0.2	<0.1
2	5	10	10/27/93	0.2	0.8	<0.2	<0.1	<0.4	<0.2	<0.1
2	5	12	10/27/93	0.4	1.1	<0.2	<0.1	0.4	0.2	<0.1
2	5	14	10/27/93	0.4	0.7	<0.2	<0.1	0.4	0.2	<0.1
2	5	16	10/27/93	0.3	0.7	<0.2	<0.1	<0.4	0.6	<0.1
2	5	17	10/27/93	0.4	1.0	<0.2	<0.1	<0.4	1.0	<0.1

Field	Well	Port	Date	Atrazine	De-ethyl atrazine	De-isopropyl atrazine	Alachlor	Metolachlor	Metribuzin	Carbofuran
2	5	18	10/27/93	0.5	1.7	0.2	<0.1	<0.4	1.9	<0.1
2	6	00	10/23/93	0.2	0.3	<0.2	<0.1	0.6	0.4	<0.1
2	6	01	10/23/93	0.3	0.3	<0.2	<0.1	0.7	0.4	<0.1
2	6	02	10/23/93	0.2	0.2	<0.2	<0.1	0.5	0.3	<0.1
2	6	04	10/23/93	<0.2	0.2	<0.2	<0.1	0.4	0.3	<0.1
2	6	06	10/23/93	<0.2	0.2	<0.2	<0.1	<0.4	0.2	<0.1
2	6	08	10/23/93	0.2	0.3	0.4	<0.1	<0.4	<0.2	<0.1
2	6	10	10/23/93	0.2	0.5	<0.2	<0.1	<0.4	0.2	<0.1
2	6	12	10/23/93	0.2	0.5	<0.2	<0.1	0.4	0.3	<0.1
2	6	14	10/23/93	0.2	0.3	<0.2	<0.1	0.4	0.6	<0.1
2	6	16	10/23/93	0.2	0.3	<0.2	<0.1	<0.4	1.6	<0.1
2	6	17	10/23/93	0.2	0.2	<0.2	<0.1	<0.4	3.6	<0.1
2	6	18	10/23/93	0.2	0.2	<0.2	<0.1	<0.4	5.7	<0.1
2	8	16	12/08/93	<0.2	0.6	<0.2	<0.1	<0.4	<0.2	<0.1
3	3	00	10/18/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	0.1
3	3	01	10/18/93	<0.2	0.1	<0.2	1.1	<0.4	<0.2	<0.1
3	3	02	10/18/93	<0.2	0.1	<0.2	0.4	<0.4	<0.2	0.2
3	3	04	10/18/93	<0.2	0.3	<0.2	0.3	<0.4	<0.2	<0.1
3	3	06	10/18/93	<0.2	0.1	<0.2	0.2	<0.4	<0.2	0.2
3	3	08	10/18/93	<0.2	0.2	<0.2	0.6	<0.4	<0.2	0.2
3	3	10	10/18/93	<0.2	0.2	<0.2	0.6	<0.4	<0.2	0.2
3	3	12	10/18/93	<0.2	0.2	<0.2	0.7	<0.4	<0.2	0.1
3	3	14	10/18/93	<0.2	0.2	<0.2	0.7	<0.4	<0.2	<0.1
3	3	16	10/18/93	<0.2	0.3	<0.2	0.3	<0.4	<0.2	<0.1
3	3	17	10/18/93	<0.2	0.4	<0.2	0.3	<0.4	<0.2	<0.1
3	3	18	10/18/93	0.2	0.6	<0.2	0.5	<0.4	<0.2	<0.1
3	4	00	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	0.9
3	4	01	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	1.4
3	4	02	10/16/93	0.2	0.1	<0.2	<0.1	<0.4	<0.2	1.0
3	4	04	10/16/93	0.2	0.1	<0.2	<0.1	0.4	0.2	0.5
3	4	06	10/16/93	0.2	0.2	<0.2	<0.1	0.5	0.2	<0.1
3	4	08	10/16/93	0.3	0.4	<0.2	<0.1	0.5	0.2	<0.1
3	4	10	10/16/93	<0.2	0.2	<0.2	<0.1	<0.4	<0.2	<0.1
3	4	12	10/16/93	<0.2	0.3	<0.2	<0.1	<0.4	<0.2	<0.1
3	4	14	10/16/93	<0.2	0.4	<0.2	<0.1	<0.4	<0.2	0.2
3	4	16	10/16/93	<0.2	0.4	<0.2	<0.1	<0.4	<0.2	<0.1
3	4	17	10/16/93	0.3	0.6	<0.2	<0.1	<0.4	<0.2	<0.1
3	4	18	10/16/93	0.3	0.7	<0.2	<0.1	<0.4	<0.2	<0.1

Field	Well	Port	Date	Atrazine	De-ethyl atrazine	De-isopropyl atrazine	Alachlor	Metolachlor	Metribuzin	Carbofuran
3	5	00	10/16/93	<0.2	<0.1	<0.2	<0.1	<0.4	0.2	0.4
3	5	01	10/16/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	5	02	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	0.2	<0.1
3	5	04	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	0.2
3	5	06	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	0.3
3	5	08	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	0.3
3	5	10	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	0.3
3	5	12	10/16/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	0.3
3	5	14	10/16/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	5	16	10/16/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	5	17	10/16/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	5	18	10/16/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	6	00	10/17/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	0.2
3	6	01	10/17/93	<0.2	<0.1	<0.2	<0.1	0.4	<0.2	0.3
3	6	02	10/17/93	<0.2	<0.1	<0.2	<0.1	0.7	0.3	0.2
3	6	04	10/17/93	<0.2	<0.1	<0.2	<0.1	1.2	0.5	0.2
3	6	06	10/17/93	<0.2	0.1	<0.2	<0.1	2.5	1.0	0.3
3	6	08	10/17/93	<0.2	0.1	<0.2	<0.1	2.1	0.8	0.3
3	6	10	10/17/93	<0.2	<0.1	<0.2	<0.1	1.4	0.6	0.2
3	6	12	10/17/93	<0.2	0.1	<0.2	<0.1	0.5	0.2	0.4
3	6	14	10/17/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	6	16	10/17/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	6	17	10/17/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	6	18	10/17/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	7	06	12/10/93	<0.2	0.1	<0.2	<0.1	<0.4	<0.2	<0.1
3	7	08	12/10/93	<0.2	<0.1	<0.2	<0.1	<0.4	<0.2	<0.1
4	2	00	12/11/93	<0.2	0.3	<0.2	<0.1	4.1	1.4	2.3
4	2	01	12/11/93	<0.2	0.1	<0.2	<0.1	1.5	0.6	1.7
4	2	03	12/11/93	<0.2	0.1	<0.2	<0.1	0.4	0.3	<0.1
4	2	04	12/11/93	<0.2	<0.1	<0.2	<0.1	0.4	0.2	<0.1
4	2	06	12/11/93	<0.2	0.2	<0.2	<0.1	<0.4	0.2	0.6
4	2	08	12/11/93	0.2	0.4	<0.2	<0.1	0.6	0.5	1.3
4	2	10	12/11/93	0.4	0.4	<0.2	<0.1	<0.4	0.6	1.1
4	2	12	12/11/93	0.4	0.5	<0.2	<0.1	<0.4	0.7	<0.1
4	2	14	12/11/93	0.5	0.9	<0.2	<0.1	<0.4	0.9	<0.1
4	2	16	12/11/93	0.2	0.5	<0.2	<0.1	<0.4	0.4	<0.1
4	2	17	12/11/93	0.2	0.7	<0.2	<0.1	<0.4	0.3	<0.1
4	2	18	12/11/93	0.3	0.9	<0.2	<0.1	<0.4	0.3	<0.1

Field	Well	Port	Date	Atrazine	De-ethyl atrazine	De-isopropyl atrazine	Alachlor	Metolachlor	Metribuzin	Carbofuran
4	3	00	12/15/93	<0.2	0.4	0.2	<0.1	11.5	1.5	3.8
4	3	01	12/15/93	<0.2	0.2	<0.2	<0.1	6.4	0.9	4.1
4	3	02	12/15/93	<0.2	0.1	<0.2	<0.1	5.4	0.7	3.8
4	3	04	12/15/93	<0.2	0.1	<0.2	<0.1	9.5	0.7	2.8
4	3	06	12/15/93	<0.2	0.1	<0.2	<0.1	12.4	0.8	4.1
4	3	08	12/15/93	<0.2	0.1	<0.2	<0.1	5.9	0.6	3.6
4	3	10	12/15/93	0.2	0.3	<0.2	<0.1	4.0	0.9	3.2
4	3	12	12/15/93	0.2	0.4	<0.2	<0.1	1.5	1.1	1.6
4	3	14	12/15/93	0.3	0.8	<0.2	<0.1	0.9	1.8	1.2
4	3	16	12/15/93	0.3	0.4	<0.2	<0.1	0.4	1.4	0.4
4	3	17	12/15/93	0.4	0.4	<0.2	<0.1	<0.4	1.0	<0.1
4	3	18	12/15/93	0.5	0.4	<0.2	<0.1	<0.4	0.9	0.3
4	4	00	12/15/93	<0.2	0.8	<0.2	<0.1	2.7	2.2	0.2
4	4	01	12/15/93	0.3	1.0	<0.2	<0.1	3.5	2.8	<0.1
4	4	04	12/15/93	0.2	0.5	<0.2	<0.1	4.2	1.9	<0.1
4	4	06	12/15/93	0.4	0.9	0.3	<0.1	3.4	2.8	<0.1
4	4	08	12/15/93	0.3	0.6	<0.2	<0.1	1.0	1.8	<0.1
4	4	10	12/15/93	0.4	0.7	<0.2	<0.1	0.5	2.0	<0.1
4	4	12	12/15/93	0.9	1.4	<0.2	<0.1	0.6	3.4	<0.1
4	4	14	12/15/93	0.7	0.9	0.2	<0.1	<0.4	1.7	<0.1
4	4	16	12/15/93	1.4	1.8	0.3	<0.1	<0.4	0.7	<0.1
4	4	17	12/15/93	2.5	3.6	0.3	0.1	<0.4	1.1	<0.1
4	4	18	12/15/93	1.4	2.1	0.2	0.1	<0.4	0.7	<0.1
4	5	00	12/17/93	<0.2	0.3	<0.2	<0.1	2.1	1.2	<0.1
4	5	01	12/17/93	<0.2	0.3	<0.2	<0.1	9.3	1.3	<0.1
4	5	02	12/17/93	<0.2	0.3	<0.2	<0.1	36.1	2.2	<0.1
4	5	04	12/17/93	<0.2	0.2	<0.2	<0.1	149.0	5.5	<0.1
4	5	06	12/17/93	<0.2	0.2	<0.2	<0.1	157.0	5.9	<0.1
4	5	08	12/17/93	0.2	0.3	<0.2	<0.1	112.0	6.0	<0.1
4	5	10	12/17/93	0.2	0.3	<0.2	<0.1	45.7	4.3	<0.1
4	5	12	12/17/93	0.2	0.4	<0.2	<0.1	24.1	3.2	<0.1
4	5	14	12/17/93	0.2	0.6	0.3	<0.1	13.4	2.1	<0.1
4	5	16	12/17/93	0.2	0.4	<0.2	<0.1	5.2	1.4	<0.1
4	5	17	12/17/93	0.2	0.4	<0.2	<0.1	4.7	1.0	<0.1
4	5	18	12/17/93	0.2	0.4	<0.2	<0.1	4.1	0.9	<0.1

REFERENCES

Anderson, Mary P. and William W. Woessner. 1992. Applied of groundwater modeling. Academic Press, New York.

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

APHA (American Public Health Association). 1985. Standard Methods for the Examination of Water and Wastewater, sixteenth edition. American Public Health Association, Washington, D.C.
Attig, J.W., L. Clayton, and D.M. Mickelson. 1988. Pleistocene stratigraphic units of Wisconsin 1984-87. Information Circular 62, Wisconsin Geological and Natural History Survey, Madison, Wisconsin.

Barber, S.A. 1984. Soil nutrient bioavailability. Wiley, New York.

Bartelmē, Robert J. 1977. Soil survey of Wood County, Wisconsin. U.S. Dept. of Agriculture Soil Conservation Service, Washington, D.C.
Batten, W.G. 1989. Hydrogeology of Wood County, Wisconsin. Information Circular 60, Wisconsin Geological and Natural History Survey, Madison, Wis.

Blackmer, A.M. 1987. Losses and transport of nitrogen from soils. p. 85-103. *In* F.M. D'Itri and L.G. Wolfson (ed.) Rural groundwater contamination. Lewis Publishers, Inc., Chelsea, MI.

Bouwer, Herman, and R.C. Rice. 1976. A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resour. Res. 12: 423-428.

Bradbury, K.R., J.M. Faustini, and M.W. Stoertz. 1992. Groundwater flow systems and recharge in the Buena Vista Basin, Portage and Wood Counties, Wisconsin. Wisconsin Geological and Natural History Survey Info. Cir. 72. Madison, Wisconsin.

Bradbury, K.R. and M.A. Muldoon. 1990. Hydraulic conductivity determinations in unlithified glacial and fluvial materials. p. 138-151. *In* Nielsen, D.M. and A.I. Johnson (ed.) Ground Water and Vadose Zone Monitoring. ASTM STP 1053. American Society for Testing and Materials, Philadelphia, Pa.

Bradbury, K.R. and E.R. Rothschild. 1985. A computerized technique for estimating the hydraulic conductivity of aquifers from specific capacity data. Ground Water 23:240-246.

Brasino, J.S. 1986. A simple stochastic model predicting conservative mass transport through the unsaturated zone into groundwater. Ph.D. thesis, University of Wisconsin, Madison.

Brown, T., G. Disher, and J. Gardner. 1992. Wellhead protection program and monitoring system design; Stevens Point, Wisconsin. Environmental Monitoring Systems Laboratory. Office of Research and Development. USEPA. Las Vegas, Nevada.

Brownell, James R. 1986. Stratigraphy of unlithified deposits in the central sand plain of Wisconsin. M.S. thesis, Department of Geology and Geophysics, University of Wisconsin, Madison.

Bruce, J.W. 1985. The fate of aldicarb residue in groundwater at a Central Sand Plain farm, Waushara County, Wisconsin. M.S. thesis, University of Wisconsin, Stevens Point.

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

Clayton, Lee. 1987. Pleistocene Geology of Adams County, Wisconsin. Information Circular 59, Wisconsin Geological and Natural History Survey, Madison, Wis.

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

Curwen, David, and Leonard R. Massie. 1990. Wisconsin Irrigation Scheduling Program. University of Wisconsin – Extension Integrated Pest Management Program, Madison, Wisconsin.

CWGC (Central Wisconsin Groundwater Center). 1993. Groundwater database. Central Wisconsin Groundwater Center. University of Wisconsin - Stevens Point.

Fathulla, R.N., F.A. Jones, J.M. Harkin, and G. Chesters. 1988. Distribution of aldicarb residues in the sand and gravel aquifer of central Wisconsin. 1. Relationship between aldicarb residue concentrations and groundwater chemistry. In A. Marani (ed.) Advances in Environmental Modelling. Elsevier, Amsterdam, the Netherlands.

Fermanich, K.J., B. Lowery, W.L. Bland, K.M. McSweeney. 1993. Water, atrazine, and nitrate flux below the root zone of a sandy soil. p. 92-99. *In* Agricultural research to protect water quality. Proc. Conf. SCSA, Minneapolis, MN. 21-24 Feb. 1993. Vol. 1.

Follett, R.F., and Walker, D.J. 1989. Groundwater quality concerns about nitrogen. p. 1-22. *In* R.F. Follett (ed.) Nitrogen management and groundwater protection. Elsevier Science Publishers B.V., The Netherlands.

Freeze, R. Allan, and John A. Cherry. 1979. Groundwater. Prentice-Hall, Englewood Cliffs, N.J.

Hallberg, G.R. 1986. Overview of agricultural chemicals in ground water. In Proceedings of the agricultural impacts on groundwater conference. Aug. 11-13, Omaha Nebraska. Water Well Journal Publishing Co., Dublin, OH.

Harkin, J.M., F.A. Jones, R.N. Fathulla, E.K. Dzantor, and D.G. Kroll. 1986. Fate of aldicarb in Wisconsin groundwater. p. 219-254. *In* W.Y. Garner, R.C. Honeycutt, and H.N. Nigg (ed.) Evaluation of pesticides in groundwater. ACS Symp. Series 315, Am. Chem. Soc., Washington, DC.

Hazen, A. 1893. Some physical properties of sands and gravels with special reference to their use in filtration. Mass. State Board of Health, 24th Ann. Rep., p. 553.

Hill, A.R. 1986. Nitrate and chloride distribution and balance under continuous potato cropping. Agriculture, Ecosystems and Environment 15:267-280.

Hindall, S.M. 1978. Effects of irrigation on water quality in the sand plain of Central Wisconsin. Information Circular 36, Wisconsin Geological and Natural History Survey, Madison, Wisconsin.

Hoeft, R.G., D.R. Keeney, and L.M. Walsh. 1972. Nitrogen and sulfur in precipitation and sulfur dioxide in the atmosphere in Wisconsin. J. Environ. Qual. 1:203-208.

Holt, C.L.R., Jr. 1965. Geology and water resources of Portage County Wisconsin. U.S. Geological Survey Water-supply Paper 1796, U.S. Government Printing Office, Washington, D.C.

Jackson, Gary, Dennis Keeney, Dave Curwen, and Bruce Webendorfer. 1987. Agricultural management practices to minimize groundwater contamination. Interdepartmental Report, Environmental Resources Center, University of Wisconsin Extension, Madison, Wisconsin. (p. 58).

Johnson, T.G. and J.C. Parker. 1993. A model of nitrate leaching from agricultural systems in Virginia's northern neck. Bull. 179. Virginia Water Resources Research Center, Blacksburg.

Jumikis, Alfreds R. 1984. Soil Mechanics. Robert E. Krieger Publishing Co., Malabar, Florida.

Karnauskas, Robert J. 1977. The hydrogeology of the Nepco Lake watershed in central Wisconsin with a discussion of management implications. M.S. Thesis, Department of Geology and Geophysics, University of Wisconsin, Madison.

Kraft, George J. 1990. Fate of aldicarb residues in a groundwater basin near Plover, Wisconsin. Ph.D. thesis, Department of Soil Science, University of Wisconsin, Madison.

Kraft, G.J. and W. Stites. 1992. Port Edwards Groundwater Priority Watershed Project - groundwater resource and agricultural practice evaluation. Central Wisconsin Groundwater Center (unpublished).

Kung, K-J. S. 1990. Preferential flow in a sandy vadose zone: 2. Mechanism and implications. Geoderma 46:59-71.

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

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

Legg, J.O. and J.J. Meisinger. 1982. Soil nitrogen budgets. p. 503-566. *In* F.J. Stevenson (ed.) Nitrogen in agricultural soils. ASA, CSSA, SSSA, Madison, WI.

Lindorff, David E., Jodi Feld, and Jack Connelly. 1987. Groundwater Sampling Procedures Guidelines. Wisconsin Department of Natural Resources PUBL WR-153-87. Wisconsin DNR, PO Box 7921, Madison, Wis. 53707.

MacDonald, N.W., A.J. Burton, H.O. Liechty, J.A. Witter, K.S. Pregitzer, G.D. Mroz, and D.D. Richter. 1992. Ion leaching in forest ecosystems along a Great Lakes air pollution gradient. J. Environ. Qual. 21:614-623.

Manser, R.J. 1983. An investigation into the movement of aldicarb residue in groundwater in the central sand plain of Wisconsin. M.S. thesis, University of Wisconsin, Madison.

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

Milburn, P., J.E. Richards, C. Gartley, T. Pollock, H. O'Neil, and H. Bailey. 1990. Nitrate leaching from systematically tiled potato fields in New Brunswick, Canada. J. Environ. Qual. 19:448-454.

Monteith, J.L., and M.H. Unsworth. 1990. Principles of Environmental Physics. 2. ed. Edward Arnold, London.

Mullins, J.A, R.F. Carsel, J.E. Scarbrough, and A.M. Ivery. 1994. PRZM-2, A Model for Predicting Pesticide Fate in the Crop Root and Unsaturated Soil Zones: User's Manual for Release 2.

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

Oberle, S.L. and D.R. Keeney. 1990. Factors influencing corn fertilizer N requirements in the Northern U.S. corn belt. J. Production Agriculture 3:527-534.

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

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

Perillo, C.A., S.C. Gupta, C.J. Rosen, J.F. Moncrief, and H.H. Cheng. 1993. Susceptibility of Minnesota glacial outwash soils to nitrate leaching III. Validation of SUBSTOR computer model to predict potato yield and nitrate leaching. p. 675-677. *In* Agricultural research to protect water quality. Proc. Conf. SCSA, Minneapolis, MN. 21-24 Feb. 1993. Vol. 2.

Reddy, G.B., and K.R. Reddy. 1993. Fate of nitrogen-15 enriched ammonium nitrate applied to corn. Soil Sci. Soc. Am. J. 57:111-115.

Ritchie, J.T. 1972. Model for predicting evaporation from a row crop with incomplete cover. Water Resour. Res. 8:1204-1212.

Rose, H.G., and H.F. Smith. 1957. Particles and permeability, a method of determining permeability and specific capacity from effective grain size. Water Well J., March 1957.

Rothschild, Edward R. 1982. Hydrogeology and contaminant transport modeling of the central sand plain, Wisconsin. M.S. thesis, Department of Geology and Geophysics, University of Wisconsin, Madison.

Saffigna, P.G. and D.R. Keeney. 1977. Nitrate and chloride in ground water under irrigated agriculture in Central Wisconsin. Ground Water 15:170-177.

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

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

Shepherd, Russell G. 1989. Correlations of permeability and grain size. Ground Water 27:633-638.

Stewart, B.A., D.A. Woolhiser, W.H. Wischmeier, J.H. Caro, and M.H. Fere. 1976. Control of Water Pollution from Cropland, Volume II: An Overview. U.S. Environmental Protection Agency, Athens, GA. EPA-600/2-75-026b.

Stites, W.D. and G.J. Kraft. 1992. Pilot project for groundwater quality management in the Wisconsin Cetral Sand Plain. p. 35. *In* Abstracts, Amer. Water Resour. Assoc. Wisconsin Section Conference, LaCrosse, Wisconsin, 12-13 March, 1992.

Summers, W.K. 1965. Geology and ground-water resources of Waushara County, Wisconsin. U.S. Geol. Survey Water-Supply Paper 1809-B. U.S. Government Printing Office, Washington, D.C.

Tanji, K.K. and S.K. Gupta. 1978. Computer simulation modeling for nitrogen in irrigated croplands. p. 79-130. *In* D.R. Nielson and J.G. MacDonald (ed.) Nitrogen in the environment- Vol. 1: Nitrogen behavior in field soil. Academic Press, Inc., New York.

Thompson, Todd A., Charles S. Miller, Paul K. Doss, Linda D.P. Thompson, and Steven J. Baedke. 1992?. Land-based vibracoring and vibracore analysis: tips, tricks, and traps. Indiana Geological Survey Occasional Paper 58.

Tyler, Donald D. and Grant W. Thomas. 1977. Lysimeter measurements of nitrate and chloride losses from soil under conventional and no-tillage corn. J. Environ. Qual. 6:63-66.

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

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

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

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

Weeks, E.P. 1969. Determining the ratio of horizontal to vertical permeability by aquifer-test analysis. Water Resour. Res. 5:196-214.

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

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

White, Art F., Maria L. Peterson, and Ray D. Solbau. 1990. Measurement and interpretation of low levels of dissolved oxygen in ground water. Ground Water 28:584-590.

White, R.E. 1985. A model for nitrate leaching in undisturbed structured clay soil during unsteady flow. J. Hydrology 79:37-51.

Zheng, Chunmiao. 1989. PATH3D version 2.0 user's manual. S.S. Papadopoulos and Associates, Rockville, Md.