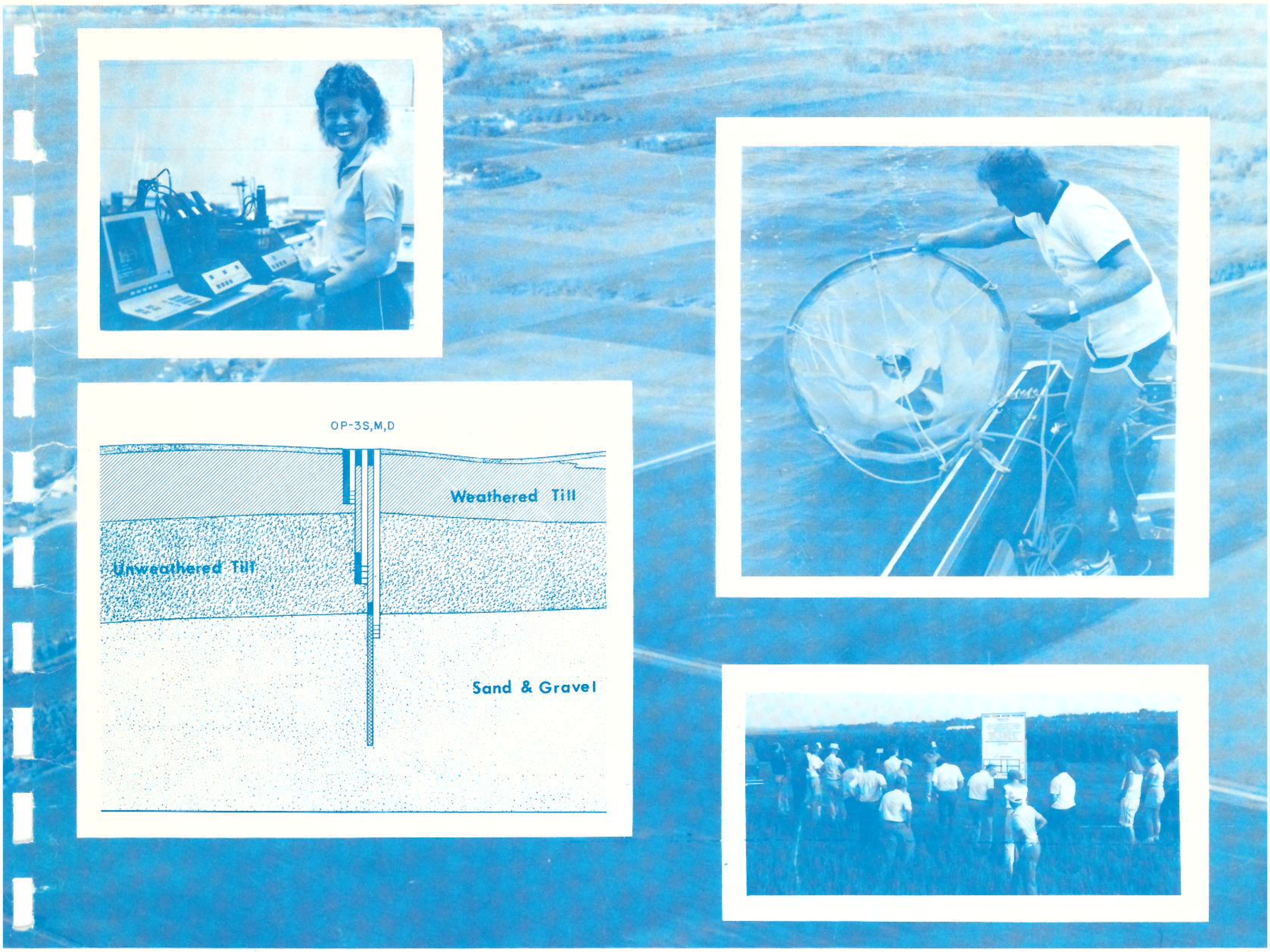
OAKWOOD LAKES - POINSETT

Rural Clean Water Program Comprehensive Monitoring and Evaluation Technical Report

Project 20

1989



SOUTH DAKOTA

May 1990

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COVER PHOTOGRAPHS

Upper left: Research Assistant performing laboratory analyses on water samples.
Upper right: Zooplankton sampling with a meter net.
Lower left: Geological cross-section of a field site.
Lower right: A field tour located at the Agricultural Chemical Leaching Study site.
Background: Two field monitoring sites near Lake Poinsett.

May 1990

ACKNOWLEDGEMENTS

The Oakwood Lakes-Poinsett Rural Clean Water Program/Comprehensive Monitoring and Evaluation Project is a cooperative special study. The following are contributing agencies: Office of Water Quality, Division of Geological Survey, South Dakota Department of Water and Natural Resources; Water Resources Institute, Station Biochemistry, Cooperative Extension Service, Wildlife and Fisheries, Plant Science Department, Biology Department, South Dakota State University; U.S. Environmental Protection Agency; Agricultural Stabilization and Conservation Service, Economic Research Service, Soil Conservation Service, U.S. Department of Agriculture.

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1.0 EXECUTIVE SUMMARY

1.0.1 Project Overview

The South Dakota Oakwood Lakes-Poinsett Rural Clean Water Program (RCWP), located in the glacial lakes region of east central South Dakota, includes a Comprehensive Monitoring and Evaluation (CM&E) project. The project area is over 106,000 acres in size, and the major land use is row crop agriculture. Corn, soybeans, alfalfa and small grains are the major crops produced.

The identified water quality problems within the Oakwood Lakes-Poinsett project area are: 1) elevated nitrate levels in certain areas of the Big Sioux aquifer that exceed the Environmental Protection Agency (EPA) drinking water standard of 10 milligrams per liter as nitrogen, and 2) high nutrient levels in the lakes through delivery by the intermittent tributaries. The aquifer has very little soil cover, making it susceptible to contamination from land surface activities.

The goal of the South Dakota Oakwood Lakes-Poinsett project is to reduce the amount of total nitrogen, pesticides, water, sediment borne contaminants, and animal waste from entering the ground and surface waters. To accomplish this goal, it was proposed to implement three Best Management Practices (BMPs), (conservation tillage, fertilizer management and pesticide management) and to construct waste management systems on 10 livestock operations. To date, 46,006 acres are under contract and 42,840 acres are implemented. This includes acres under conservation tillage, acres in fertilizer and pesticide management and this is 72% of the project's goals.

The goal of the CM&E project is to monitor the effects and evaluate the impacts to ground water and surface water from the implementation of agricultural best management practices. Specifically, the monitoring is conducted to determine if the BMPs recommended for prevention of surface water pollution will change the amount of nitrogen and pesticides input to the ground water system. These BMPs were recommended for prevention of surface water pollution, but the effects on ground water were unknown.

The monitoring for the CM&E is designed to 1) conduct site specific ground water monitoring in real field operations to determine ambient water quality and any changes in ground water quality due to BMP implementation; 2) conduct monitoring in the vadose zone on field sites and on an experimental plot to evaluate the effects of tillage practices and cropping on the percolating water; and 3) conduct monitoring of the Oakwood Lakes System in an effort to estimate the effects of BMPs on the lake water quality.

1.0.2 Summary of Ground Water Monitoring and Evaluation

Samples have been collected from monitoring wells since 1984. Years 1984 through 1986 had above normal precipitation and years 1987 through 1989 had below normal precipitation, resulting in above normal and below normal water levels respectively for each three year period. The drought period has resulted in 23 monitoring wells going dry.

Since May, 1984, 2,708 ground water samples have been collected and analyzed for nitrate as N. Except for one sample collected in 1989, NO_3-N concentrations greater than 5 mg/l are found only at depths less than 20 feet below the water table. Three geozones (shallow weathered till, shallow sand and gravel, and alternating sand/silt) continue to display elevated nitrate as N levels. Median nitrate as N concentrations in several geozones, analyzed without the unfarmed OP-site, indicate reduced levels during the last three years. The reduced NO_3-N concentrations appear to correlate with the period of below normal precipitation.

Pesticide samples have been collected from 68 monitoring wells since 1984. A total of 129 samples of 1142 collected have detected pesticides in 31 wells. Alachlor and 2,4-D represent approximately 60% of the detections since the project began. Since installation in 1987, seven pesticides have been detected in 48 samples from the epoxy resin monitoring wells. May through September continue to account for most (75%) of the pesticide detections. Pesticides have been detected in every month except March.

Pesticide samples were collected in surface water runoff during a major precipitation event and resulted in 16 detections from 17 samples. This evidence indicates overland flow as a possible mechanism of transport of pesticides and may help to explain why 65% of the detections in monitoring wells at a site do not have a history of use. Three of four pesticides detected in the runoff water also did not have a history of use at the BL-site where collection occurred.

1.0.3 Summary of Surface Water Monitoring and Evaluation

Comprehensive analysis and modeling to describe the Oakwood Lakes system as a whole was scheduled for 1989, but extension of monitoring activities to gain additional runoff data delayed that activity until 1990. A draft report of OLSS data analysis will be prepared by September 30, 1990, with a final report in December 1990.

Although precipitation was again below normal in 1989 normal snow pack just prior to the snowmelt was higher than in 1987 and 1988. Significant rain fell on several occasions during 1989. Three runoff events affected all tributaries to varying degrees after the snowmelt runoff had ceased.

The total suspended solids load to the lake system carried by six monitored tributaries was 97,567.8 kg in 1987, 46,262 kg in 1988 and 165,398 kg in 1989. The lake system trapped 43%, 33% and 100% of the suspended solids that was contributed by six monitored tributaries in 1987, 1988 and 1989 respectively.

All of the basins had a net gain of total PO_4-P in all three years of study. The lake system as a whole trapped 70%, 80% and 100% of the total PO_4-P that was contributed by six monitored tributaries in 1987, 1988 and 1989 respectively. The Oakwood Lakes system is operating as a phosphorus sink. The total PO_4-P load to the lake system carried by six monitored tributaries was 1,419 kg in 1987, 993 kg in 1988 and 2,528 kg in 1989. This represents an annual tributary total PO_4-P load to the Oakwood Lake system of .16 g/m^2/year in 1987, 0.11 g/m^2/year in 1988 and 0.28 g/m^2/year in 1989.

Although the lake system is operating as a nitrogen sink, nitrogen was trapped less efficiently by the lake system than phosphorus. The lake system as a whole trapped 46% of the tributary load in 1987, 44% in 1988 and 100% in 1989. The total nitrogen as N load to the lake system carried by six monitored tributaries was 14,016 kg in 1987, 5,095 kg in 1988 and 12,400 kg in 1989. This represents an annual tributary total nitrogen as N load to the Oakwood Lake system of 1.57 g/m^2/year in 1987, 0.57 g/m^2/year in 1988 and 1.39 g/m^2/year in 1989.

All of the tributaries supply significant amounts of nitrogen and phosphorus to the lake system. The highest flow weighted mean concentrations of total nitrogen as N and total PO_4-P were observed at Loomis Creek in all three years. These concentrations indicate a severe case of pollution in the Loomis Creek watershed.

Organic nitrogen concentrations and total PO_4-P in 1987, 1988 and 1989 indicate that all of the Oakwood Lakes continue to be hypereutrophic. Mean concentrations were lower in 1989 than in either 1987 or 1988 and ranged from .089 ppm at station L-3 to .145 ppm at station L-1. A decline in total PO_4-P concentrations was observed as water moved through the system in all three years.

Oxygen stratification occurred in the Oakwood lakes in 1987, 1988 and 1989 but was ephemeral. Stratification occurred during periods of no wind, warm weather and high algal biomass. Anoxic conditions at the sediment water interface are probably occurring during these events which would facilitate the release of phosphorus from the sediments.

Thirty-six sediment cores were obtained from Oakwood Lakes during March, 1989. Two cores from each site were chemically analyzed to determine the phosphorus content of the sediment. A laboratory experiment was conducted to investigate the extent of phosphorus release from the sediment to the overlying water during aerobic and anaerobic conditions.

The highest sediment concentrations of total, inorganic and organic phosphorus were observed in West Oakwood Lake followed by Johnson Lake and East Oakwood Lake. The phosphorus concentration profiles of the incubated sediment cores indicate anaerobic conditions enabled the release of phosphorus from a sediment depth of 45 cm.

Anaerobically incubated cores released significantly more Total PO_4-P (0.305 g/m^2) than the aerobically incubated cores (0.080 g/m^2). Based on a sediment surface area of 7.12 x 10^6 m^2, the potential sediment phosphorus load to the Oakwood Lakes system is 2,041.0 kg Total PO_4-P during anaerobic conditions and 467.76 kg Total PO_4-P during aerobic conditions.

We collected 38,180 fish representing 10 species. Bullheads dominated the catch in total number (86%). Northern pike, yellow perch, and walleye accounted for 2.5% of the total number.

About 90% of the fish in the Oakwood lakes can be classed as zooplanktivorous. The cascading trophic interaction theory suggests that because there is a negative relation between the densities of zooplanktivorous fish and zooplankton, then the zooplankton community might be too small to control the phytoplankton community.

Nitrate was the dominant form of nitrogen in OLSS land wells. Mean nitrate concentrations were above 1 ppm in only five wells. Total dissolved PO_4-P concentrations observed in most land wells surrounding the Oakwood Lakes fall in the .02 to .06 ppm range although other wells are much higher (.12 ppm). Ground water is probably a significant contributor of phosphorus to the system.

1.0.4 Summary of Vadose Zone Monitoring and Evaluation

Drainage water was collected from samplers installed in the soil 2', 4', 6', and 15' below the land surface for the first year in 1989 from 6 significant leaching/rainfall events. The "wetter" plots (those which had oats planted the previous year) were the only treatments where macropore drainage water was moving at low tensions. Alfalfa plots and the previous corn plots were mined of water enough that "low tension" water could not extract a sample. After the 5th rainfall event and a total of over 10 inches in almost 11 weeks, there was movement of water to the ground water. However, the rate was very slow and there was not sufficient time when the matric potential was above -100 cm of tension to collect any sample at all. Surprisingly, nitrate as N concentrations increased in the ground water beneath the alfalfa plots after the last rainfall on July 18th. This indicates that there is sufficient available nitrates in the soil with that type of crop that can be released into the ground water.

Water reached deeper in the soil profile quicker for the no-till (NT) tillage treatment compared to a moldboard plow (MP) treatment after a significant rainfall event. The concentrations of nitrate as N also were higher deeper in the profile for the no-till treatment.

Pesticides were detected less frequently deeper in the soil and more frequently closer to the surface. There was no significant difference between the number of pesticide detections between the NT and the MP treatments at any depth.

Some of the major pesticide detections were from chemicals that had not been applied for at least 3 years (atrazine was not applied for 7 years and it was detected). The pattern of detections for those non-applied chemicals was consistent with depth. A random Mass Spectrometry analysis is being run on those samples that had a high detection of a given pesticide to "weed" out false positives, if they exist.

1.0.5 Major Project Findings

A comprehensive report on the Oakwood Lakes-Poinsett RCWP project will be made in 1991. The following outline of major project findings has been included to summarize the information that is being produced by this project. The findings or observations for each project activity are presented as a basis for recommended actions to protect water resources or reduce water contamination. In many cases a complete analysis of the data has not been made and recommendations are based upon judgments and inferences made from the monitoring data.

I. Surface Water Quality Monitoring/Data Analysis

A. Findings

* The Oakwood Lakes are hypereutrophic (overproductive). They continue to produce excessive algal blooms and suffer from winterkill which both impair the designated beneficial uses.

* All of the Oakwood Lakes tributaries carry an excess load of both nitrogen and phosphorus (levels that would maintain eutrophic conditions).

* The Oakwood Lakes trapped between 70% (1987) and 100% (1989) of the phosphorus entering the lakes through surface water sources.

* Ground water entering the Oakwood Lakes is much lower in nutrients than surface water sources.

* Analysis of Oakwood Lakes sediment indicates a potential to release 0.248 g/m^2 of phosphorus. This is approximately equal to the annual surface water loading of .221g/m^2/year measured in 1987 and 1988.

* The presence of feedlots in a small tributary resulted in phosphorus concentrations in water delivered to the lake, 10 times higher than in a small tributary without feedlots.

* Correlation of BMP implementation with tributary water quality was difficult because record keeping of BMP implementation was by farm unit instead of by watershed unit.

* Preliminary data analysis indicates that BMP implementation by itself may extend the life of the Oakwood Lakes but may not result in measurable water quality improvements in the lakes.

B. Recommendations

* In future land treatment projects where water quality monitoring will be used to evaluate effectiveness of the project, records of BMP implementation by watershed unit could be more efficiently used for evaluation of water quality impacts due to the implementation.

* Water quality and land treatment teams should work closely in the early stages of a project to insure that proposed BMPs are appropriate to improve or protect the targeted resource, whether surface water or ground water.

* Seepage meters should be used to determine ground water movement through lake beds, but land wells or in-lake wells provide better water quality data.

* Assessment of internal loading from sediments should be made before water quality benefits are calculated for lakes based on watershed treatment alone.

II. Soil Water Quality Monitoring/Data Analysis

A. Findings

* Concentrations of two pesticides (alachlor and atrazine) exceeded the EPA drinking water proposed maximum contaminant levels (MCL) five times (2%); three times occurring at 2 ft. below the surface on moldboard plow (MP) treatments and twice occurring at 6 ft. below the surface on MP treatments.

* Concentrations of two pesticides (terbufos and trifluralin) exceeded the EPA drinking water health advisory five times (2%); once at 2 ft., three times at 4 ft., and one time at 15 ft. under both tillage treatments.

* Twenty five percent (25%) of the pesticide detections were chemicals that had not been applied to the soil for at least three years prior to sample collection. This indicates that residual agricultural chemicals may persist in the soil environment and may be released years later. Verification of these chemicals by gas chromatography/mass spectrophotometer (GC/MS) is forthcoming.

* On land where soil water was not mined by previous crops, significant increases in the water table were detected.

* Electronic tensiometers (transiometers) are required to measure the response of percolating soil water at various depths for a given rainfall event.

* The newly designed soil water monitoring/collection system enabled the sampling of macropore water as the leaching event occurred.

B. Recommendations

* Deep rooted crops, such as alfalfa, may be used: 1) to retrieve leached nutrients if the subsurface material allows root penetration, and 2) to minimize the leaching potential of agricultural chemicals by mining soil water.

* Shallow-rooted, early maturing crops (such as cereal grains) with low transpiration demand, allow the water leaching potential to increase as precipitation exceeds evapotranspiration.

III. Ground Water Quality Monitoring/Data Analysis

A. Findings

* Of 2411 water samples collected from monitoring wells ranging in depth from 7 to 65 feet, concentrations of nitrate as N did not exceed 5 mg/l at depths greater than 20 feet below the water table.

* Three geological settings (geozones) have been identified with consistently high concentrations of nitrate as N: 1) shallow sand and gravel with thin topsoils, 2) sand/silt alternating layers, and 3) shallow weathered till.

* Concentrations of nitrate as N found in the water samples are statistically greater under farmed sites than an unfarmed site.

* The fate of pesticides in the ground water is currently unknown, however, 84% of pesticide detections were one time events with no detections the following month indicating rapid degradation or dilution below detection limits.

* Approximately 64% of pesticide detections occurred in May through August which correlates well with time of application or shortly thereafter.

B. Recommendations

* Non-parametric statistical methods are valuable as an analysis technique because ground water quality sample populations are typically not normally distributed.

* Future CM&E type projects should have equal representation of all agricultural practices in the experimental design, even if it requires financial incentives to farmers that are not using BMPs to obtain access for water quality monitoring.

* The classification method of "geozones" is useful to aggregate and reduce variability of water quality data when a variety of hydrogeologic environments are being monitored.

* Nested wells (two or more at the same location screened at different depths) are valuable for documenting changes in water quality with depth and response to geologic stratigraphy.

* Twice a month or weekly sampling for pesticides during the months of May through August will assist in gaining a better understanding of the fate and transport in the ground water system.

* Pesticide samples should be verified using GC/MS to help eliminate false positives.

IV. Effectiveness of BMPs

A. Findings

* Concentrations of nitrate as N in the ground water show no statistically significant differences between conventional tillage and conservation tillage in seven years of monitoring.

* Infiltrating water on a no-tilled field reached 2 ft., 4 ft., and 6 ft. depths below the ground surface from a rainfall-creating-leaching event faster than from a moldboard plowed field for similar soil profile water contents. This is believed to be caused by better macropore development open to the surface on the no-tilled land.

* When high amounts of nitrogen fertilizers are broadcast at the soil surface (200#/acre as N) for corn, higher concentrations of nitrate as N are evident in percolating waters at 4 ft. and 6 ft. below the ground surface for the no-till treatment compared to a moldboard plow treatment.

* Based on data from one growing season, the number of pesticide detections in percolating soil water does not indicate differences or trends between tillage systems consistent with depth.

* There were twice as many pesticide detections in water samples collected at the 4 ft. depth for the no-till (NT) treatment compared to moldboard plow treatments, with seven out of ten pesticide maximum concentrations occurring on the no-till treatment.

* At 2 ft., 6 ft. and 15 ft., the number of pesticide detections from soil water samples were greater for the MP treatment compared to the NT treatment with eight out of ten pesticide <u>maximum</u> concentrations occurring on the MP treatment. This does not support the same trend of nitrate concentrations with depth between tillage treatments.

B. <u>Recommendations</u>

* To reduce shallow aquifer contamination, fertilizer management should be applied when farming over sand and gravel aquifers with overlying weathered till or thin topsoils.

* A combination of high fertilization and no-till management would not be recommended in areas where thin soils overlie shallow unconfined aquifers used for domestic drinking water purposes. An evaluation needs to be made on a site specific basis to determine the impacts that this may have on surface versus ground water quality.

V. Denitrification Monitoring

A. <u>Findings</u>

* Denitrification rates are the highest for corn and alfalfa (45-80 kg/ha/yr) and the lowest for grass (1-30 kg/ha/yr).

* Denitrification rates are the highest at the beginning and at the end of the growing season.

* Small grain denitrification rates are between 15 and 50 kg/ha/yr.

* The existing NTRM (a nutrient-tillage-residue management model) inadequately predicted the movement of nitrates to ground water mainly because the unsaturated flow submodel does not account for macropore water and solute movement.

B. Recommendations

* Event-based monitoring, particularly from rainfall events, must take place to gain an understanding of the mechanism for nutrient movement to the saturated zone.

1.1 Introduction/Project Background

1.1.1 Project Overview

The South Dakota Oakwood Lakes-Poinsett Rural Clean Water Program (RCWP) includes a Comprehensive Monitoring and Evaluation (CM&E) project. The project area is over 106,000 acres in size, and the major land use is row crop agriculture. Corn, soybeans, alfalfa and small grains are the major crops produced.

The identified water quality problems within the Oakwood Lakes-Poinsett project area are: 1) elevated nitrate levels in certain areas of the Big Sioux aquifer that exceed the Environmental Protection Agency (EPA) drinking water standard of 10 milligrams per liter as nitrogen, and 2) high nutrient levels in the lakes through delivery by the intermittent tributaries. The aquifer has very little soil cover, making it susceptible to contamination from land surface activities.

The goal of the South Dakota Oakwood Lakes-Poinsett project is to reduce the amount of total nitrogen, pesticides, water, sediment borne contaminants, and animal waste from entering the ground and surface waters. To accomplish this goal, it was proposed to implement three Best Management Practices (BMPs), (conservation tillage, fertilizer management and pesticide management) and to construct waste management systems on 10 livestock operations. To date, 46,006 acres are under contract and 42,840 acres are implemented. This includes acres under conservation tillage, acres in fertilizer and pesticide management. This is 72% of the project's goals.

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The monitoring for the CM&E is designed to 1) conduct site specific ground water monitoring in real field operations to determine ambient water quality and any changes in ground water quality due to BMP implementation; 2) conduct monitoring in the vadose zone on field sites and on an experimental plot to evaluate the effects of tillage practices and cropping on the percolating water; and 3) conduct monitoring of the Oakwoods Lakes System in an effort to estimate the effects of BMPs on the lake water quality.

1.1.2 Project Area Description

The Oakwood Lakes-Poinsett RCWP is located in east-central South Dakota in the counties of Brookings, Hamlin and Kingsbury. The Prairie de Coteau region, in which the RCWP is located, is distinguished by numerous lakes and sloughs, and is drained by the Big Sioux River and its tributaries. Geologically, this area is typified by collapsed (stagnation) drift glacial morphology. Glacial sand and gravel aquifers are frequently formed by such processes and underlie much of the study area. The topography ranges from nearly level to severely undulating and is characterized by deep silty, loamy, well drained soils.

The project area is characterized by a continental type climate with large variations in seasonal and daily temperatures. Cold winters and hot summers are the norm with temperature extremes of less than -20 degrees Fahrenheit in winter and greater than 100 degrees Fahrenheit in the summer. Average annual temperature is in the lower 40s, and the average annual precipitation is less than 22 inches.

The approximate population of the watershed is 2500 people. The major industry in the area is row crop agriculture with the exception of several small recreation-related businesses around the lakes. The seasonal population, with people living adjacent to the lakes, is about 4000. Although livestock production is a major enterprise, the individual operations are scattered and small.

Lake Poinsett and Oakwood Lakes serve as the focal point for many recreational activities for Watertown, Brookings, Huron and Madison. These cities are major population centers within eastern South Dakota and contain many of the estimated 175,000 people living within a 50 mile radius of the lakes.

The Big Sioux aquifer underlies a portion of the project area. This shallow glacio-fluvial aquifer produces water for domestic, municipal, agricultural and industrial uses within a ten county area. These ten counties comprise over 30% of the state's population, and the aquifer is the primary domestic water source for this population.

1.1.3 Project History

Several implementation activities preceded and were instrumental in initiating the RCWP. The lakes were originally selected as "208" Water Quality Study Areas to determine existing problems and to develop alternative solutions. Later the lakes and watersheds were approved as "208" Project areas to continue monitoring and to begin installing Best Management Practices (BMPs) for non-point source pollution reduction. To carry out these activities, the Department of Water and Natural Resources (DWNR) signed a $30,000 "208" funded contract with the Brookings, Hamlin, and Kingsbury County Conservation Districts. Portions of the two watersheds are located within these districts. The funds were used to hire a coordinator to do the monitoring, to contact farmers and to attempt to secure project funding for implementation.

The cooperating Districts, the Soil Conservation Service (SCS), and DWNR, worked with the Agricultural Stabilization and Conservation Service (ASCS) to develop the RCWP application which was approved and funded in October 1981. The Comprehensive Monitoring and Evaluation (CM&E) project and funding was approved in October 1982.

DWNR is under contract with ASCS with SCS concurrence, to conduct the CM&E water quality monitoring and evaluation. DWNR has sub-contracted with the Water Resources Institute at the South Dakota State University to conduct the Oakwood Lakes System Study, the Agricultural Chemical Leaching Study, the soils monitoring at the field sites, and the laboratory analysis of the ground water samples. Station Biochemistry at SDSU is also under contract to perform the pesticide scans for the ground water quality samples.

1.1.4 Water Quality Problems

A complex of lakes exists within the project area. The larger of these are: Lake Poinsett-7,868 acres/average depth of 10 feet; Oakwood Lakes--2,184 acres/average depth of 8 feet; and Lake Albert--2,400 acres/average depth of 4 feet. Smaller lakes partially within the project area are Lake St. John, Dry Lake, and Thisted Lake. Dry Lake drains directly into Lake Poinsett while Lake St. John and Thisted Lake drain to Lake Albert which drains into Lake Poinsett. Lake Poinsett has been hydraulically connected to the Big Sioux River through the Lake Poinsett diversion ditch. The Lake Poinsett drainage area (32,452 acres) is 83% cropland. The Oakwood Lakes drainage area (52,856 acres) is 50% cropland.

A national eutrophication study conducted by the EPA in 1977 ranked 31 lakes in South Dakota according to eutrophic state. With one being the least eutrophic and thirty-one being the most eutrophic, Lake Poinsett ranked 19th, East Oakwood ranked 20th, and West Oakwood ranked 22nd. Water quality data generated by this study indicated that nutrient rich water (high nitrogen and phosphorus concentrations) was entering the lakes from several intermittent streams.

A "208" surface water quality study was conducted in the project area in 1977 and 1978. The SCS estimated 1.6 acre-feet/year of sediment was deposited in Lake Poinsett and 3.6 acre-feet/year in Oakwood Lakes which is insignificant compared to the storage capacity of these lakes. Water quality data from the "208" study indicated (as did the eutrophication study) that high nutrient levels in the lake placed it in an eutrophic to hypereutrophic state. According to the study the high levels of nutrients were being introduced by way of the intermittent streams and the Big Sioux diversion ditch.

Additional surface water sampling from 1982 through 1984 indicated that the lakes are phosphorus limited based on the typical 15:1 nitrogen to phosphorus ratio. A trophic state index (tsi) of greater than 50 had been calculated for Lake Poinsett (based only on total phosphorus) which indicated that the lake was hypereutrophic (SD DWNR, 1985).

Non-bedrock aquifers in eastern South Dakota consist of glacial outwash deposits from the Pleistocene Epoch glaciation. Within the project area surficial deposits are of the Wisconsin stage of glaciation, specifically the Cary and Iowan substage till, loess, and outwash. The outwash of greatest thickness is the Cary outwash. Morphology of the outwash is that of a valley train outwash which forms along the margin of a glacier as it melts and recedes. This results in long, relatively thin sinuous deposits of sand and gravel of variable thickness and aerial extent. There are three outwash aquifer systems in the area, referred to as the surficial, intermediate and basal aquifers. Connections between these aquifers occur but are not reliably known. The Big Sioux aquifer is the surficial aquifer associated with the Big Sioux River and its tributaries, and is of greatest importance in the project area because of its high quality water, accessibility, and thickness. The Oakwood Lakes-Poinsett project is not directly concerned with the other two aquifers of greater depth.

Thirty-two percent of the state's population lives within the Big Sioux River basin. Within the basin, the Big Sioux aquifer supplies water to 94% of the total number of public wells. Domestic and stock use water needs are almost entirely supplied by ground water resources. Since the lakes are not suitable for drinking and the deeper underlying aquifers are of a poor water quality, the Big Sioux aquifer is the most extensively used source of water within the Big Sioux River basin.

The Big Sioux aquifer is found at a relatively shallow depth with the overlying soils being thin and/or highly permeable. The potential for aquifer contamination is great and from existing data, is underway. From the DWNR, Office of Water Quality files, water samples from 861 private wells in Brookings and Hamlin counties showed 27 percent of the wells had nitrate levels in excess of the EPA's standard for drinking water of 10 milligrams per liter (mg/L) nitrate as nitrogen in the early 80's. Nitrate concentrations ranged from less than 0.1 to greater than 120 mg/L as nitrogen. Sampling of domestic wells during the Big Sioux Aquifer Water Quality Study (1984) showed excess nitrates in similar proportions as those reported by DWNR.

1.1.5 Overall Project Goals

The goal of the Oakwood Lakes-Poinsett RCWP project is to improve and protect the surface and ground water quality of the project area by the application of selected BMPs. The overall goals of the RCWP project are to:

a. Reduce the amount of total nitrogen and pesticides entering ground water and surface water by assisting with fertilizer and pesticide management on 70,000 and 65,000 critical acres, respectively (see 1986 Oakwood Lakes Poinsett annual progress Report for the definition and delineation of critical areas).

b. Reduce the amount of water and sediment borne pollutants entering waterways and lakes by applying or maintaining conservation tillage on 65,000 critical acres.

c. Reduce the amount of animal waste entering waterways, lakes and ground water by applying waste management systems on 10 livestock operations.

1.1.5.1 CM&E objectives

The objective of the CM&E program is to monitor the effect and to evaluate the impact on ground water and surface water quality of selected BMPs that have been implemented by the RCWP. Several investigations, each with its own set of objectives that relate to the main objective, are underway to describe the various parts of the system.

Water quality monitoring objectives are to quantitatively describe the movement of water and nutrients within the natural system of a working farm field shown diagrammatically on Figure 2. This data will then be used to evaluate the effects conservation tillage, fertilizer management, and pesticide management (BMPs) have had on the system in terms of nitrogen and pesticide in the ground water, and pesticide, sediment and nutrient delivery to the lakes.

There are two levels of concerns: 1) are documented water quality problems due to farming operations? and 2) will the implementation of conservation tillage (BMP 9), fertilizer management (BMP 15), and pesticide management (BMP 16) have an effect on nitrogen and other nutrients, and pesticide levels in the hydrologic system?; if so, will the effect be positive or negative (i.e., will conservation tillage increase or decrease water related contaminant transport to the subsurface and directly or indirectly to the surface)?

1. Field Site Monitoring

Monitoring at field sites is being conducted to determine the effects of BMPs on ground water at several farmed fields and one unfarmed field. The aquifer is quite susceptible to contamination and concentrations of nitrate in the ground water are exceeding the recommended drinking water limit of 10 mg/L as nitrogen. Specific objectives for field site monitoring are as follows:

a) Test the idea (null hypothesis) that conservation tillage, fertilizer management, and pesticide management have no effect on ground water quality. Then, the goal of the project is to document changes in the water quality as a result of the BMP implementation in order to disprove this idea (null hypothesis).

b) Edge of field runoff monitoring of surface water is being conducted at some field sites to document concentrations of nutrients and pesticides leaving the ground water field sites. This data will be used to account for water and nutrients on field sites and to provide a basis for comparison of nutrients and pesticides in the runoff water with ground water quality data.

c) The soil profile monitoring in fields above the nested monitoring wells is to provide supporting information of vertical movement of water to the ground water from precipitation events. Soil sampling and analysis detail the chemical setting in the soil profile and allow evaluation of effects to the ground water.

2. Agricultural Chemical Leaching Study

Any water reaching the ground water table (under natural conditions) must first pass through the soil profile. The ACLS will study the movement of, and quality of, the water percolating through the soil profile. The specific objectives for the ACLS are as follows:

a) To determine the differences in vertical fluxes of nitrates, organics, tracers, and water to specific depths between moldboard plow (MP) and no-till (NT) tillage systems for a corn-oats rotation.

b) To develop an event-actuated monitoring/control system for collecting low tension unsaturated soil water (characteristic of water held in macropores), during or after, precipitation/leaching events.

c) To determine the differences in response time of a given precipitation event to the change in matric potential (measurement of soil moisture) at specific depths between NT and MP tillage systems on oats and corn.

d) To develop continuous and more complete unsaturated soil-water quality data relative to land use practice, soil physical characteristics, and soil moisture conditions.

3. Oakwood Lakes System Study

The goal of the OLSS project is to determine if the application of BMPs in agricultural watersheds will affect water quality in shallow, hypereutrophic prairie lakes. Documented surface water quality problems appear to be attributable to non-point source agricultural activities. Specific objectives for the OLSS are as follows:

a) To produce annual sediment, phosphorus and nitrogen budgets for the Oakwood Lakes System.

b) To produce an estimate of internal nutrient sources, both physical and biological.

c) To develop a predictive equation that will estimate the potential for improved trophic state resulting from changes in the nutrient budget.

4. Agricultural Non-Point Source Model (AGNPS)

The objective of the AGNPS watershed modeling is to predict the effects of BMP implementation on excess nutrient loading to the surface waters within the CM&E project area. The model allows an examination of water quality under various management schemes over the entire project area or in sub-watersheds. The AGNPS model will be calibrated by comparing AGNPS output data with results from the OLSS watershed monitoring program.

5. Nitrogen Tillage Residue Management Model (NTRM)

The original objective of NTRM was to predict the effect of BMP implementation on nitrogen transport into the ground water within the CM&E project area. The NTRM model simulates the soil-water-crop continuum based on climate, soil type, soil moisture, land use, management practices, vegetative cover, and fertilizer application. It estimates volumes and concentrations of leachate from the root zone entering the ground water. Denitrification studies were to be used to obtain the amount of nitrogen lost to the atmosphere, and the values obtained were to be put into a denitrification submodel to balance the applied nitrogen with its fate. The unsaturated flow submodel, which does not account for preferential flow, does not realistically simulate the real world water movement. Consequently, the model outputs from the different conditions may not simulate what is happening in the field with any real degree of certainty.

6. Nutrient Budget Denitrification Study

There are many pathways for nitrogen to follow as it cycles through the environment. Two such avenues are the input of nitrogen through nitrogen fixation and mineralization and the output of nitrogen through denitrification. The objective of the denitrification study is to determine the nitrogen fixation and denitrification on the field sites. Another objective is to determine typical rates of denitrification under various croppings and tillages so that they can be applied project wide.

1.2 Project Methodology

1.2.1 Implementation

Given the range of questions to be answered by project assessment, the CM&E portion of this project requires several approaches which will be integrated to the extent possible. Statistical analysis of chemical and physical water quality and quantity data will yield information regarding the on-site effectiveness of BMP implementations and the status of the Oakwood Lakes System. Hydrologic modeling will relate the site specific information to the entire project area and the other lakes and aquifers which are affected by BMP implementation. Economic modeling will evaluate the cost-effectiveness of an adopted implementation strategy. Given the limitations of climatic variation, project length, land use conditions and conservation treatment options in the project, modeling will allow the study of relative levels of effectiveness of BMPs and project plan alternatives.

1.2.2 Monitoring

To evaluate and discern the effects of BMPs, the system of water, nutrient, sediment, and pesticide cycling must be adequately described. This entails the monitoring of a physical and a chemical system. Six broad categories of monitoring arise from the pursuit of these objectives: ground water, surface water, soil profile, runoff, climate, and land use monitoring.

Ground Water

To accomplish the project objectives, the ground water monitoring approach is site specific. The site specific approach is desirable for two reasons: 1) the size of the project (106,000 acres) is too large to monitor as an entire unit; and 2) it increases the probability of detecting land use affected changes in water quality. The specific sites are seven fields 10-80 acres in size. Six of the sites are farmed fields and one is unfarmed. Of the six farmed fields, two are without BMPs.

Ground water monitoring characterizes the physical setting of the system by drilling and installing monitoring wells and conducting in-situ aquifer testing. Placement of the well within the monitoring site was an important consideration. Ground water environments are dynamic systems complicated by the aspect of three dimensions. Horizontal well placement must account for up-gradient chemistries, on-site conditions, and determination of resulting water quality down-gradient of the site. Vertical placement of wells is coordinated with horizontal placement so stratification of contaminants and gradients in the vertical direction can be ascertained over the entire site. Vertical placement must also account for the geological stratification. Well sampling defines the chemical setting and continues as input to the evaluation of trends and comparisons throughout the remainder of the project.

BMPs were implemented on the field sites prior to monitoring, so an experiment designed to compare with and without conditions is not possible. To discern effects attributable to BMPs, it is necessary to examine sites in terms of trends in water quality that may be developing due to the implementation of BMPs. The adopted analytical procedure makes comparisons between fields with BMPs, fields without BMPs, and fields that have never been farmed, in a pseudo paired watershed approach.

Surface Water

The OLSS study will meet project objectives of evaluating the water quality impacts of BMP implementation by determining hydrologic and nutrient budgets for the Oakwood lakes and assessing the current trophic state of the lake.

In-lake monitoring was designed to produce data for use in a stepwise multiple regression analysis to determine factors that best explain variability in dependent variables such as chlorophyll a, total phosphorus, algal biomass and Secchi disc transparency. The strategy also includes determining current trophic state, in-lake water quality, and monitoring of algae, zooplankton and fish populations. This empirical lake model will be used to determine the role of nutrients and other factors in controlling in-lake water quality.

One of the most important factors that determine the state of a lake is the hydrologic and nutrient budgets. The OLSS monitoring strategy includes the direct measurement of surface water, atmospheric and ground water flux through the lake system with evaporation derived from local weather records. Instrumentation to accomplish this includes gauging stations at all tributaries, and recording rain gauges around the lake.

Mass balance equations derived from hydrologic and nutrient budgets will be used to determine the degree to which the lake is acting as a nutrient sink. Input-output models will be used to determine if nutrient reductions that have resulted from BMP implementation will be likely to alter the trophic state of the Oakwood Lakes.

The importance of directly measuring surface water contributions to a lake budget has long been recognized but ground water contributions are often determined by indirect methods which are subject to substantial error. Direct measurement of ground water flux through the Oakwood Lakes was considered to be an important part of the OLSS monitoring strategy because of the presence of extensive sand and gravel aquifers adjacent to the lakes. CM&E field site monitoring indicated elevated concentrations of nitrate as nitrogen (NO3-N) in shallow ground water that were attributed to agricultural practices.

Instrumentation to measure ground water movement includes land wells, in-lake wells and lake bottom seepage meters. Land wells and in-lake wells were sampled to determine quality of ground water entering the Oakwood Lake System.

The Agricultural Nonpoint Source Pollution (AGNPS) model was used to estimate the percentage reductions in nutrient loadings due to BMP implementation. The OLSS monitoring strategy also included collection of water quality data in several watersheds to assess the reliability of AGNPS estimates.

Runoff monitoring at applicable field sites utilizes stage recorders and flumes to measure flow volumes and automatic sampling devices to determine water quality. Topographic base maps of the field and control sites have been developed to determine contributing land areas.

Soil Profile

In an effort to make the predictive evaluation more precise, the agricultural chemical leaching study (ACLS) was initiated in April 1988. The ACLS is a research farm site where 15 plots are being studied and described under controlled conditions. One fertilizer level is being applied, two tillage practices are being employed, three crops are grown, and pesticide application is being strictly regulated.

The expanded soil profile monitoring will include pesticide and nitrate soil water extractions and analysis on a leaching-event-actuated automatic extraction system. By using more sensitive instruments which enable more timely data, a better correlation between tillage practices, soil water content, and soil water quality may be obtained. It is the ability to detect the contact time between soil and input water and the rate of movement of soil water to the ground water along with water quality parameters that provides information relative to the interaction of tillage, pesticides, and crops.

The soil water monitoring includes the use of transiometers (an electronic tensiometer), volumetric soil water monitoring, and the use of infiltrometers to determine the rate of intake of water on the soil. Vadose zone water quality monitoring will detail the chemical setting in the soil profile and allow predictive evaluation of effects to the ground water.

Climatic

Monitoring of climatic conditions identify physical inputs to the system by measuring precipitation amounts and intensity, solar radiation, soil and air temperature, and wind speed and direction. Sampling and analysis of precipitation will establish atmospheric chemical inputs.

Land Use

Land use monitoring describes the induced inputs and subsequent outputs due to farming activities on the sites. Land use histories will be collected and continuing operations will be tracked and recorded.

1.2.3 Evaluation

The comprehensive monitoring and evaluation of the Oakwood Lakes-Poinsett RCWP is designed to describe the cause and effect relationship between agricultural management practices and the quality of the ground and surface water resources of the study area. Each project component adds a piece of evidence which contributes to the evaluation of the impacts of BMPs on the quality of the water resources over the project area.

The CM&E ground water monitoring activities are site specific to determine the impact of BMPs on the ground water at the field sites. Field site monitoring data will be used to analyze the influence of BMPs on ground water quality. This evaluation will be used to estimate the impact of BMPs on the water resources of the project area.

The NTRM model was to be used to estimate movement of water and solutes to the ground water. However, the model could not satisfactorily simulate the processes and reproduce field data because the unsaturated flow submodel does not account for macropore flow. Therefore, the model will not be used to evaluate the effectiveness of BMPs on a project-wide basis.

Tributary monitoring before and after BMP implementation to determine water quality effects was inappropriate because BMP installation preceded the two year OLSS monitoring period by several years. Evaluation of BMP impact on tributary water quality will be made to the extent that land treatment data is available. Project evaluation will incorporate monitoring of water quality and quantity, biological surveys, sediment nutrient surveys, land use modeling and, lake modeling to evaluate BMP impacts. The strategy relies upon the development of nutrient, sediment and hydrologic budgets for the Oakwood lakes system for comparison to reductions in loadings estimated by watershed models. Identification of factors that predict in lake water quality and the use of input-output lake models to estimate the potential for BMPs to improve trophic state will complete the evaluation.

Surface hydrologic modeling provides the basis for assessing NPS problem areas and environmental responses to BMPs for those season situations and storm events most closely associated with surface water quality problems. The information developed from a series of model runs encompassing a range of land uses and BMP treatments will be a primary basis for evaluating the relative effectiveness of BMPs over the project area.

The Economic Research Service used a multiple objective resource allocation model to analyze relationships between hydrologic and economic objectives as well as associated impacts and trade-offs for a range of planning situations with competing objectives. These land use and management options influence hydrologic parameters and provide the key link between hydrologic and economic models. Model solutions will express agricultural income and related hydrologic responses.

2.0 ANALYSIS AND EVALUATION OF FIELD SITE MONITORING

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2.1 Introduction

The CM&E ground water monitoring approach is site specific because the size of the project (106,000 acres) is too large to allow detailed monitoring as an entire study area, and the selected sites increase the chances of detecting land use affected changes in water quality. The seven sites are fields 10-80 acres in size. Six of the sites are farmed fields and one is unfarmed (OP-Site). The unfarmed site was cropped until approximately 1970 to 1974. Of the six farmed fields, one (JW-site) is a control site (i.e., farmed without the BMPs which are being evaluated: BMP 9 conservation tillage, BMP 15 fertilizer management, BMP 16 pesticide management). Since 1977, the VC-site changed from conservation tillage to conventional tillage as a result of a change in the land operator.

Ground water monitoring began at the field sites in 1984, and has continued to date. Locations of the field sites and the Master site (see section 3.0 for the discussion of the Master site) are shown in Figure 2-1 (Oakwood Lakes-Poinsett Rural Clean Water Program Comprehensive Monitoring and Evaluation, project map). Field sites are labeled on figure 2-1 by the owners initials. Extremely wet conditions in the area precluded test hole drilling and monitoring well installation in the fall of 1985 or in 1986, so drilling and final ground water instrumentation was completed in the fall of 1987. CM&E ground water monitoring at the seven field sites consists of 114 monitoring wells at 60 locations, 113 of which were installed for the project. One well was installed by the Department of Water and Natural Resources Division of Water Rights prior to the CM&E project.

2.2 Methodology

Monitoring at the sites includes land use, ground water, soil profile, runoff water, and climatic data. The monitoring includes the collection of both chemical and physical parameters. The chemical monitoring is primarily water sample analysis and is designed to determine if water quality changes are occurring with time or with changes in land use practices. Physical monitoring includes recording land use, chemical use, water levels, flow volumes and other physical measurements. The physical monitoring will provide the foundation for explaining the mechanism of change and be the basis for projecting future trends, and discussing areal generalization.

Oakwood Lakes - Poinsett Project
Rural Clean Water Program
Comprehensive Monitoring & Evaluation

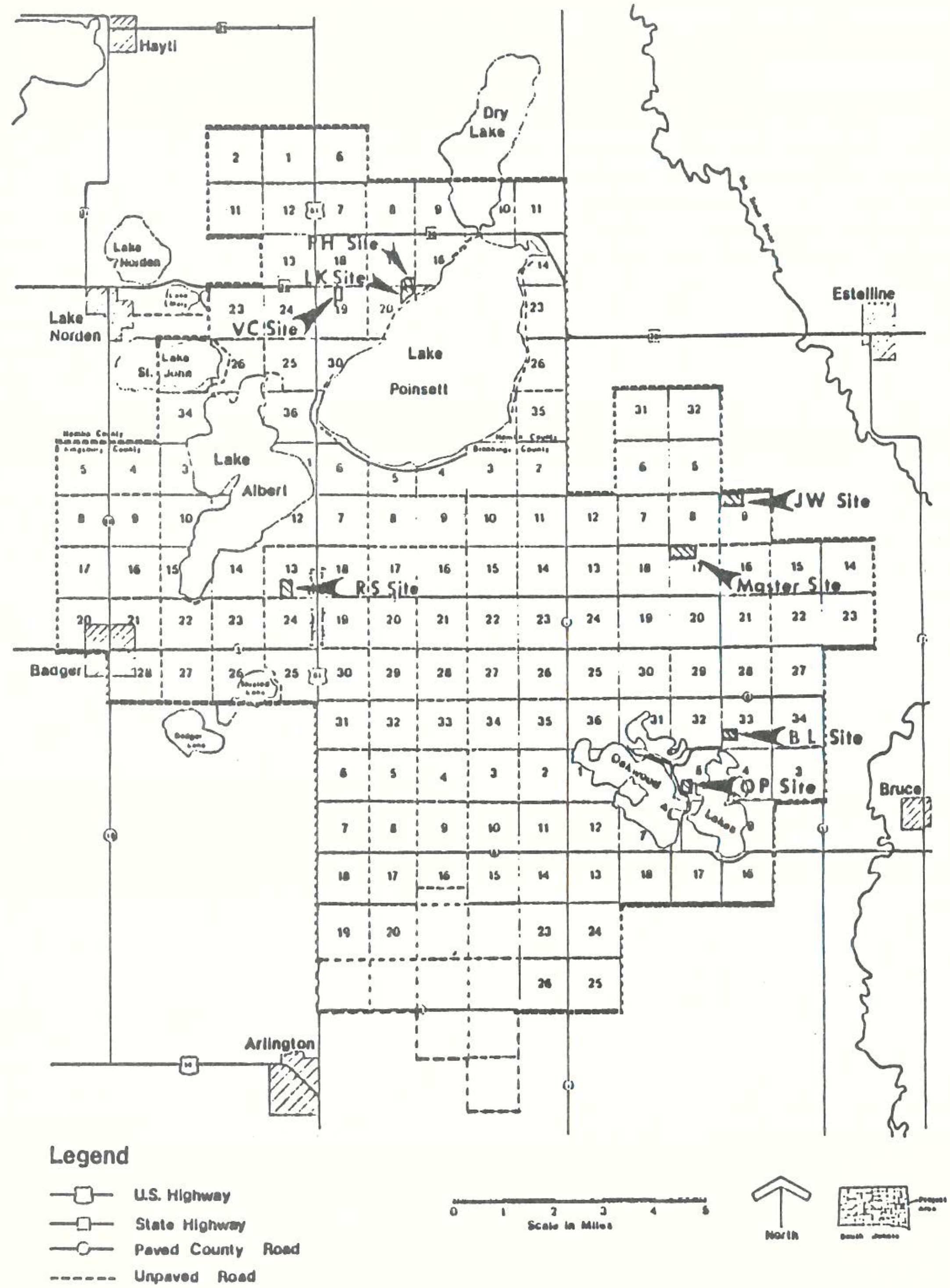


Figure 2-1 Oakwood Lakes - Poinsett Project area map.

2.2.1 Land Use

Land use monitoring is a major link in the monitoring system and an important information source for the final evaluation of the effects of BMPs. Land use histories at the field sites (section 2.5) have been recorded based on information provided by the land owner or operator. The farming operations during the current year were documented at interviews with the land operators. Land use information will also be collected for as much of the project area as feasible.

2.2.2 Hydrogeology

Monitoring the ground water system is accomplished through monitoring wells. In any ground water monitoring project, well installation and construction procedures, and horizontal/vertical placement are crucial considerations. Improper installation, construction, or placement can yield data which are inaccurate, misleading or incapable of providing information needed to evaluate results. Once installed, a monitoring well is a fixed sampling point.

Project monitoring wells were installed with a hollow flight auger drill rig so that: 1) no drilling fluid which could affect water chemistries was needed; 2) semi-undisturbed split spoon samples could be obtained to document the geology; 3) accurate well placements were possible in collapsing or caving formations and 4) sand/gravel pack and bentonite used to seal the annular space could be placed at the proper interval.

The majority of the monitoring wells are two-inch diameter polyvinyl chloride (PVC) casing with three-foot or five-foot, # 18 slot, commercial PVC well screens joined to the casing with threaded couplers. Five wells were constructed in November 1987 of fiberglass reinforced epoxy resin (ER) casing with three-foot slotted intervals, for pesticide detection verification. Epoxy resin material has been found to adsorb less synthetic organic compounds than other materials, and does not leach compounds into the water it contacts (Collins, A.G. and Johnson, A.I., 1988). Two-inch diameter casing was used because it is large enough to accept most sampling equipment yet small enough to be relatively easy to purge of standing water prior to sampling. The length of well screen was chosen so discrete samples, as opposed to integrated samples across an entire aquifer thickness, could be collected. This was significant because chemistries in ground water vary vertically.

A sand/gravel pack was placed around the well screen to increase the efficiency of the intake and to retain particulates which would otherwise pass through the well screen. Natural sand/gravel pack was developed in unconsolidated sand and gravel formations when the materials collapsed around the well screen as the auger was removed. A bentonite seal was placed above the well screen to isolate the screen from water outside the desired sampling interval. All monitoring wells were sealed from the ground surface to three feet below the ground surface. Wells finished in till or silty clay formations were sealed from one foot above the screen to the surface with bentonite. The annulus was sealed because backfilling with drill cuttings with a permeability potential orders of magnitude higher than the undisturbed soils could create an avenue for water movement.

Ground water environments are complicated three dimensional dynamic systems. Monitoring wells are horizontally placed, whenever feasible, to monitor for upgradient, on-site, and downgradient water quality and used to determine hydraulics of the site. Vertical placement of wells (well nests or groups of two or more wells at the same location screened at different depths) was coordinated with horizontal placement to document vertical variations of contaminants and vertical hydraulic gradients over the entire site. Vertical placement of wells was also directed in response to geologic stratigraphy.

2.2.3 Water Chemistry/Water Level Measurements

Ground water sampling at all field site wells was conducted quarterly from January to August and bi-monthly thereafter during 1989. Monthly samples were collected at selected wells on each site. During the months of June and July a bi-weekly schedule was used. Laboratory analysis of the bi-monthly water samples includes nitrate, nitrite, ammonia, organic nitrogen, total dissolved solids, pH, electrical conductivity, and dissolved oxygen. In addition to the above parameters, the monthly/bi-weekly laboratory analysis included a scan for 26 pesticides (complete list of pesticides in section 2.3.3.4, Table 2-2), total dissolved phosphorus, chloride, sulfate, total dissolved iron, total dissolved potassium, and total hardness.

The South Dakota Department of Water and Natural Resources (DWNR) has prepared a Quality Assurance (QA) plan according to the EPA recommendations and guidelines. The plan outlines QA activities and requirements for sampling procedures of all components including ground water, surface water, and soil samples, and for analytical procedures of all parameters. The QA plan also includes Quality Assurance/Quality Control documents prepared by the contractual laboratories.

Ground water sampling was conducted in a manner to assure a representative sample and to preserve the integrity of the sample prior to delivery to the laboratory. Samples were obtained with a pneumatic bladder pump, a variable capacity double check valve PVC bailer, or a peristaltic pump. The sample collection method depended on accessibility of the well, the weather sensitivity of the equipment, the specific capacity of the well, and the ability of the well to accept a sampling device. Before obtaining a well sample, the well was purged of standing water. Time for purging was determined at each pumpable well by pumping water through a cell until temperature and conductivity stabilized. In wells that must be bailed, the well was either bailed dry and sampled during recharge, or three to five well volumes were removed prior to sampling.

Pesticide samples have been collected from five epoxy resin wells (installed in 1987), since April of 1988. The epoxy resin wells were installed at the same location and depth as existing shallow PVC wells. The epoxy resin and PVC wells are sampled at the same time but using different techniques. The epoxy resin wells are sampled with a polytetraflouroethylene (PTFE) (Teflon-

Teflon is a registered trademark) cleaned with acetone and distilled water. The epoxy resin well is purged using the "clean" Teflon bailer. The sample is collected in an acetone and sample rinsed glass bottle. Sample results from the PVC and epoxy resin are compared to determine if the PVC materials and/or sampling techniques are a source of adsorption or desorption.

Sample containers were new or laboratory cleaned polyethylene bottles for most parameters, and glass with foil or teflon lids for the pesticides. Samples were kept cold until delivery to the laboratory and were always delivered within 24 hours of collection. A complete description of Standard Operating Procedures for ground water sampling is available upon request from DWNR.

The water level in each sampled well was measured prior to the well being purged. Water level measurements in the wells were taken weekly during frost free seasons and alternate weeks after frost. Flow directions and horizontal gradients were determined from the water level measurements at the shallow wells. Vertical gradients were calculated at well nests. Baildown and/or slug tests were conducted at wells to determine hydraulic conductivity. Geologic cross-sections were constructed from drilling logs to determine stratigraphy and most probable routes of contaminant transport.

2.2.4 Field Site Runoff

Surface water runoff samples were collected at the BL-site and analyzed to compare results with detections and concentrations of parameters in the ground water. Also, surface water results may help to define input mechanisms of agriculture chemicals into the ground water.

Samples were collected in one liter polyethylene or polypropylene bottles. Except for two grab samples, collection was done using an ISCO automatic sampler. Flows were determined by using the procedure for tributaries described in section 5.2.2.

2.2.5 Geozones

Due to the variety of hydrogeologic environments that were being monitored, different sample populations were encountered. A classification method was devised to aggregate and reduce the variability of the data. The method created "Geozones" which characterized each monitoring well by the geologic stratum in which it was screened, the depth of the well screen, and the thickness of overlying fine-grained material in the case of sand and gravel stratum. The geozone classifications are project specific and not intended to encompass all possible geologic environments, but the methodology used and type of classifications produced may be applicable to other ground water investigations. Eleven geozones have been classified for the CM&E project.

Depth at which the well is screened is expressed in two ways: 1) depth below ground surface (depth b.g.s.) for wells screened in glacial till; and 2) depth below the water table (depth b.w.t.) for wells screened in sand and gravel.

Figure 2-2 (Geozones) illustrates the geozone abbreviations displayed on a diagrammatic cross section. The cross section is not a cross section based on the drilling information from a single site but represents a composite of several real stratigraphic sequences found throughout the project area. The water table in the diagram is not intended to suggest flow directions, only the typical relative position or depth of the water table within the geozones. Below is an explanation of the geozone criteria, the number of project wells in each geozone, and the common abbreviation used throughout this report. (Till is glacial till, unsorted silty clay with a small percent of sand and gravel.)

Geozone Abbreviations and Definitions:

WTLT15 Weathered till (brown color) or silty clay (reworked till) with the screened interval of the well at a depth b.g.s. of less than or equal to 15 feet (ft.) (19 wells)

WTGT15 Weathered till or transition zone (greenish brown zone interpreted as a transition zone between the weathered and the unweathered till) with the screened interval at a depth b.g.s. of greater than 15 ft. (11 wells)

UT Unweathered till (gray color). (7 wells)

SC Silty clay aquitard located between an upper and a lower aquifer system. (2 wells)

SS-A Alternating layers of thinly bedded fine sand and silt. (4 wells)

NOTE: Overlying soil material refers to all silt and clay rich sediments overlying a sand and gravel layer. It includes silt and/or clay loams, and in some cases, glacial till.

SGLT5LT10 Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval less than or equal to 10 ft below the water table. (30 wells)

SGLT5GT10 Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval greater than 10 ft below water table. (15 wells)

SG5-15LT10 Sand and gravel with greater than 5 ft and less than or equal to 15 ft of overlying soil material, with the screened interval less than or equal to 10 ft below the water table. (2 wells)

SG5-15GT10 Sand and gravel with greater than 5 ft and less than or equal to 15 ft of overlying soil material, with the screened interval greater than 10 ft. below the water table. (3 wells)

SGGT15 Sand and gravel with greater than 15 ft of overlying soil material. (5 wells)

SG-UA Sand and gravel located under an aquitard as the lower unit of a two aquifer system. (16 wells)

The initial placement and subsequent distribution of wells within the geozones was targeted towards the shallower depths. The majority of the sand and gravel underlying aquifer geozone (SG-UA) wells penetrate saturated conditions at ten feet or less below the ground surface. This bias was intentional from the outset and has proven to be advantageous since the highest concentrations of chemicals associated with agricultural activity have been observed at the shallower depths.

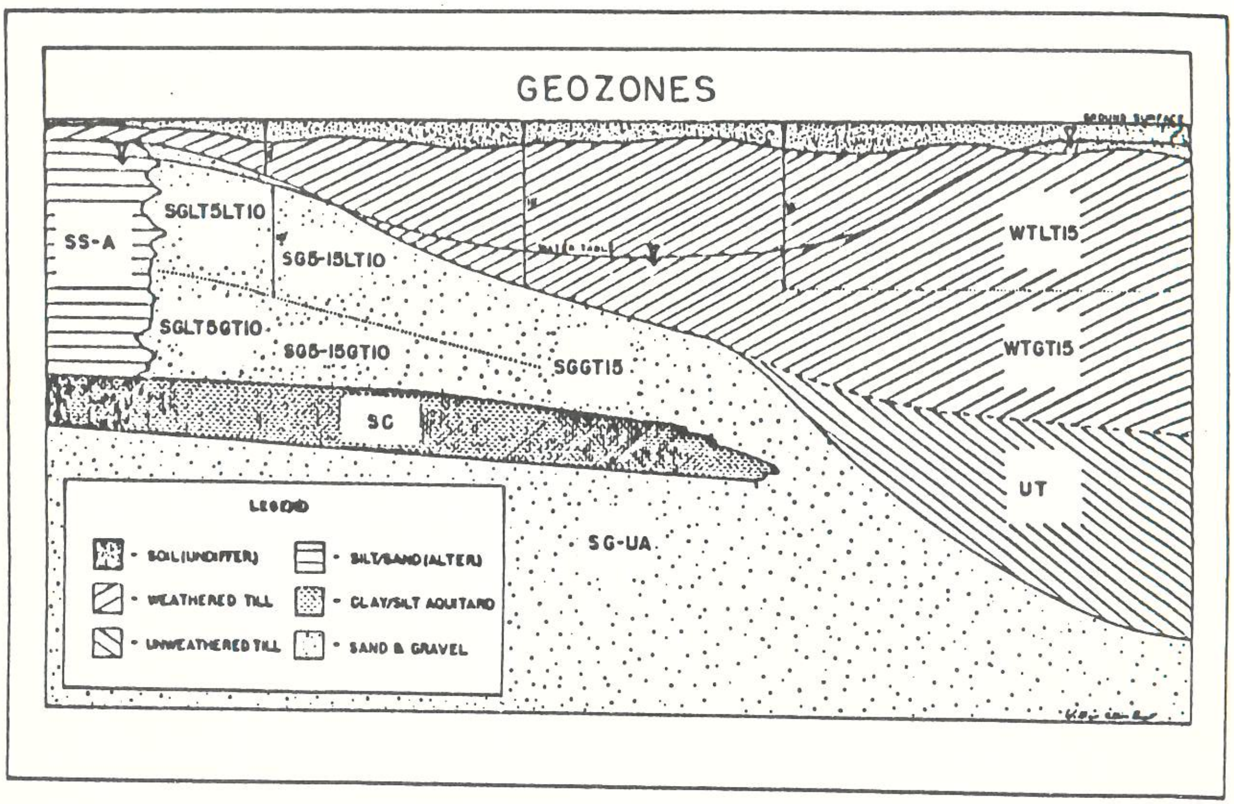


Figure 2-2 Geozone cross-section.

2.3 Results

2.3.1 Precipitation

Precipitation in the project area as a whole is best described by information from the U. S. Weather Bureau station in Castlewood (personal communication, Bill Lytle, State Climatologist). Lines of equal precipitation trend roughly in a southeast to northeast direction, and Castlewood is located just outside the project area to the northeast.

The intent of the following precipitation description is to indicate general trends during project monitoring. Different databases and techniques of computing estimated precipitation were employed in past years to describe site specific precipitation. Comparison of estimated versus actual precipitation at the Master Site indicates that below normal rainfall during the past two years has rendered the distance weighted mean method inaccurate.

Records of precipitation for Castlewood are complete from 1906 to present. Figure 2-3 (Annual Precipitation Castlewood Station) illustrates the total precipitation from 1906 through 1989. Average precipitation at the Castlewood station is 21.41 inches per year indicated by the horizontal line on the graph.

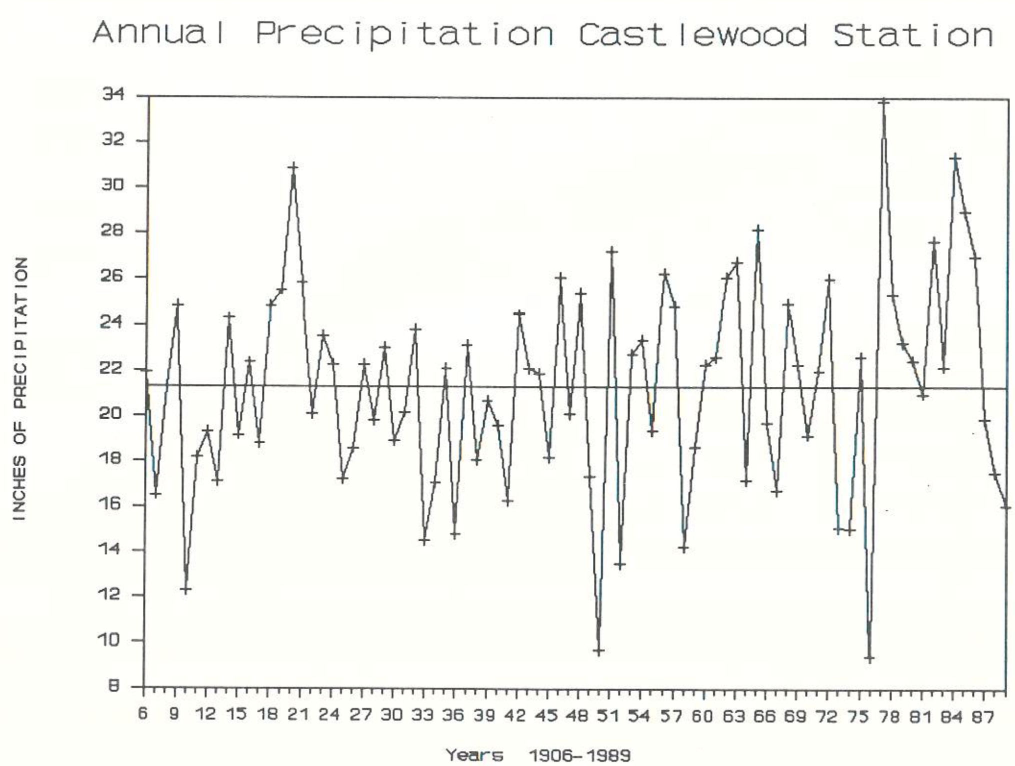


Figure 2-3. Annual precipitation for Castlewood Station.

For ten years prior to 1987, precipitation has been above or near average; 1981 was just below average. This is the longest period of above average precipitation for Castlewood on record. Total annual precipitation for 1987, 1988, and 1989 was approximately 7%, 18%, and 25% below normal, respectively. This is the third consecutive year (1984-1989) the project area has experienced below normal precipitation.

2.3.2 Hydrogeology

Ground water level measurements are the basis from which hydrogeology interpretations begin. By relating the depth to water from each well to a common datum, a water table or potentiometric map can be drawn to indicate the direction of ground water flow. By examining water levels through time the occurrence of a significant recharge event may be determined. Recharge via infiltration from the surface is the avenue for surface applied contaminants to the ground water.

Though recharge to ground water can be from surface water sources, the primary source is precipitation. Precipitation may be lost to evapotranspiration prior to reaching the ground water table. As precipitation increases above normal, infiltration increases and ground water levels rise. The degree of ground water rise is buffered by the degree of soil moisture deficit in the unsaturated zone above the water table.

Since December 1983, 16,129 water level measurements have been collected from the project monitoring wells. Hydrographs (water levels versus time) have been plotted for each well. A representative well has been chosen from each site, and the hydrographs for those wells are presented in section 2.5.

The hydrographs indicate that the water levels have been declining since May 1986. These declining water levels are the result of three years of below normal precipitation, and as of December 31, 1989, 23 of the 114 monitoring wells have gone dry. The smaller peaks on the hydrographs are the result of spring and sometimes fall recharge events. These spring and fall events were greater on the ascending than on the descending limb probably because the soil moisture content on the descending limb was lower resulting in less water reaching the saturated zone.

Note that the water elevation axis of the hydrographs (Y axis) is not at the same scale for all the graphs. Typically the rise and fall associated with recharge events has been of a higher magnitude in the wells completed in glacial till than in the wells completed in sand and gravel. Higher magnitude change has also been true for the decline in water levels since May 1986; glacial till wells have declined approximately 10 to 12 feet while sand and gravel wells have declined approximately 6 to 8 feet.

Water table maps are also presented with the fact sheet for each site in section 2.5. Data from June, 1989 were chosen as representative of the typical water table configuration. Shallow water table well data were used to construct the water table maps.

Water table maps were evaluated for changes through time during 1987 and 1989. Except for minor fluctuations, the water table configurations were basically stable.

2.3.3 Ground Water Chemistry

2.3.3.1 Nitrates

Analyses of nitrate as nitrogen concentrations for all samples collected from May 1984 through December 1989 include all data except data from wells known to be contaminated by activities not related to agricultural cropping practices. Two well nests (4 wells) have had samples which displayed consistently elevated nitrate as nitrogen (NO_3-N) concentrations. Wells PH-23S and PH-23D have most likely been contaminated by the pumping of septic tank overflow into a dry well which is immediately upgradient of these wells. Wells RS-1S and RS-1D have had elevated NO_3-N concentrations due to their position downgradient of an inactive livestock holding area. Therefore, analysis of "all data" excludes data from these wells due to the point-source nature of contamination.

From May 1984 through December 1989, 2,708 ground water samples were collected and analyzed for NO_3-N. Most of the data analysis focuses on nitrate since approximately 70% of all nitrogen in the samples is in the form of NO_3-N. In general, the less total nitrogen the smaller the nitrate percentage. The dominant nitrogen species in samples with low nitrate percentages is either organic or ammonia nitrogen.

The results of NO_3-N analyses were plotted versus depth below water table of sample collection (Figure 2-4). Depth below the water table where the sample was collected was used instead of depth below the ground surface because depth below the water table results in an equilibration of depths which would otherwise reflect differences in the depth to water. The plot includes 2,469 samples from 110 wells taken between May 1984 and December 1989.

The results of the plot are nearly the same as in 1988 with the exception of one point (5.82 mg/l) below 20 feet. Other than this one exception, the plot indicates that NO_3-N concentrations greater than 5 mg/l were found only at shallow depths of less than 20 ft below the water table. Nitrate as Nitrogen concentrations have not exceeded 0.2 mg/l at depths greater than 30 feet. Though the placement of wells is biased toward shallower depths, the data include 180 points which exceed the 20 foot depth. In 1989, 41 samples were collected from a depth greater than 20 feet below the water table.

When a sample population is not normally distributed, the median is a better measure of central tendency (than the mean) because it is not sensitive to extreme values (Crawford, C.G., 1984). All the geozone NO_3-N concentrations were analyzed for normality (Shapiro-Wilk or Kolmogorov-Smirnov test) using the Statistical Analysis System (SAS release 5.18) and had nonnormal distributions . It is typical for ground water quality sample populations to not be normally distributed. The number of samples and statistical results are shown on Table 2-1.

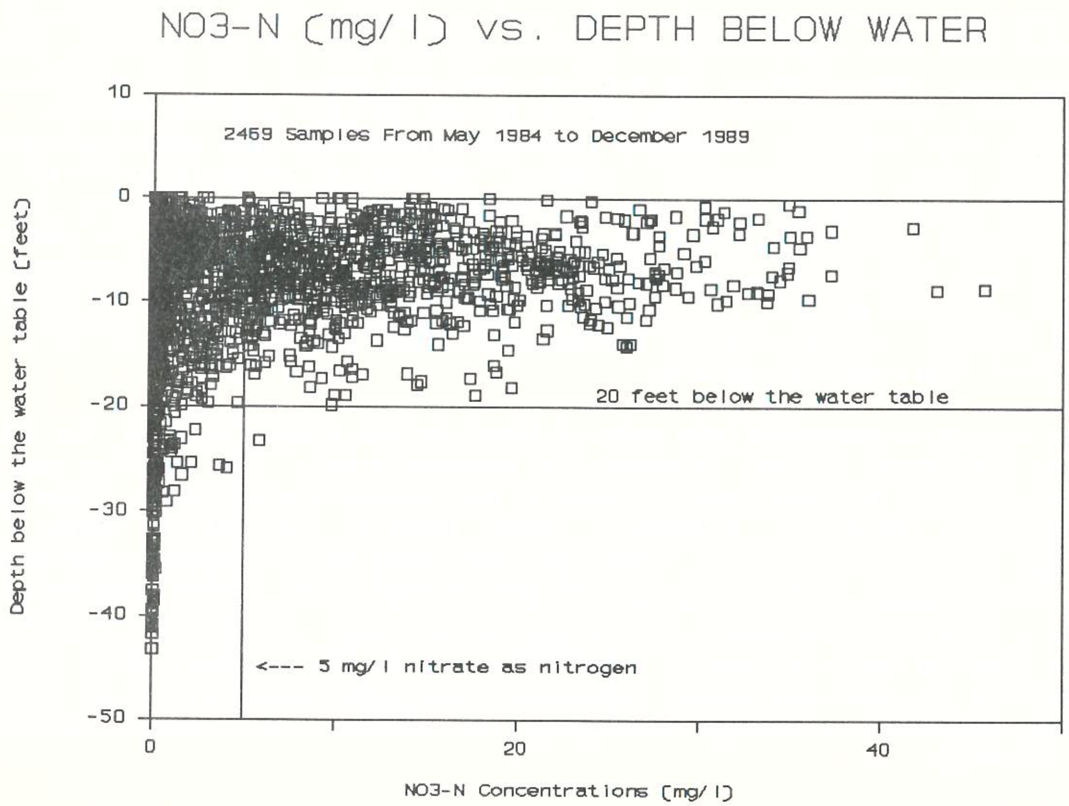


Figure 2-4. NO_3-N concentration (mg/l) versus depth below water table.

Table 2-1. Geozone Statistical Data

GEOZONE	N-SPECIES	MIN	MAX	MED	N	MEAN
SC	NO3	1.600	23.500	5.380	35	9.027
SG5-15GT1	NO3	0.000	2.680	0.100	37	0.616
SG5-15LT1	NO3	0.000	0.200	0.040	49	0.051
SGGT15	NO3	0.000	4.070	0.030	106	0.257
SGLT5GT10	NO3	0.000	27.000	0.285	330	3.255
SGLT5LT10	NO3	0.000	45.700	5.445	658	7.463
SG-UA	NO3	0.000	34.910	0.140	333	4.853
SS-A	NO3	0.020	26.200	2.780	109	6.554
UT	NO3	0.020	4.020	0.370	150	0.425
WTGT15	NO3	0.090	28.000	2.080	271	6.107
WTLT15	NO3	0.000	35.450	7.995	390	9.798

The median NO_3-N values for each geozone are shown on a bar graph (Figure 2-5). The total number of data points is 2,469 representing samples collected from May 1984 to December 1989. The plot of median NO_3-N concentrations revealed that the highest median concentrations were in the shallow horizons (Figure 2-5), specifically the shallow sand and gravel with thin topsoils (SGLT5LT10), the sand/silt alternating layers (SS-A), and the shallow weathered till (WTLT15). Samples from the silty clay (SC) also indicated relatively high concentrations. However, the median value from the SC geozone was based on 35 samples from only two wells. One of the two wells, PH-21, has exceptionally high median concentrations ranging from 9.3 mg/l in 1985 to 22 mg/l in 1989 and are probably more reflective of the immediately overlying shallow sand and gravel. The samples from deeper strata (sand and gravel and tills) display almost four to twenty fold lower concentration of NO_3-N throughout the sampling period.

Median nitrate concentrations were plotted on bar graphs by year for each geozone beginning in 1984 (Figures 2-6 through 2-10). The OP-site was eliminated from the data set to examine the farmed fields only. Five geozones were plotted; SGLT5LT10, SGLT5GT10, SG-UA, WTLT15, and WTGT15. These five geozones have the most wells, were the most frequently sampled, and displayed measurable change in NO_3-N concentrations through time. These geozones also follow a logical progression for ground water flow: Water and nutrients move from a shallow sand and gravel (SGLT5LT10) to slightly deeper sand and gravel (SGLT5GT10) to sand and gravel separated from upper aquifers by an aquitard (SG-UA). Likewise, a shallow weathered till (WTLT15) to a deeper weathered till (WTGT15).

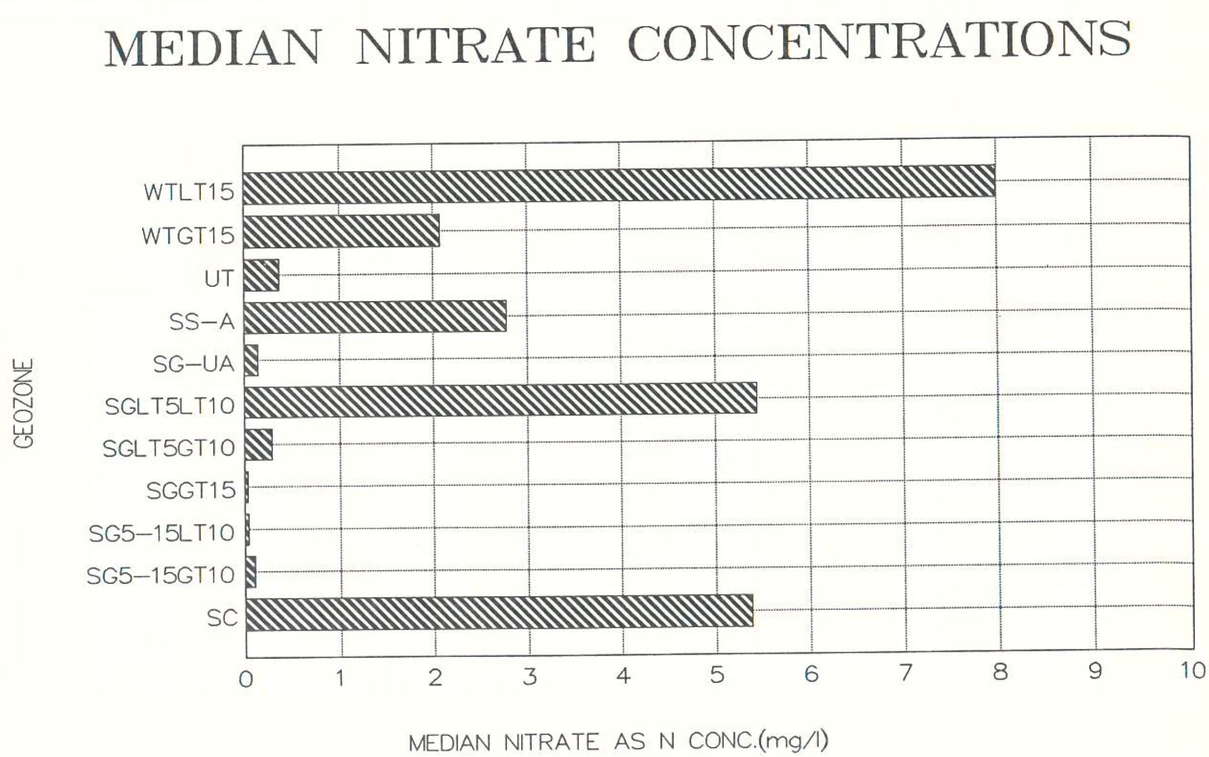


Figure 2-5. Median NO_3-N concentration by geozone.

The bar graph plots indicate that 1989 had the lowest median concentrations of NO_3-N for three geozones (SGLT5LT10, SG-UA, and WTLT15) since monitoring began in 1984. The shallow sand and gravel geozone (SGLT5LT10) median NO_3-N concentrations (Figure 2-6) have declined continously from greater than 8 mg/l in 1986 to approximately 2 mg/l in 1989, indicating an approximate 4 fold decrease. Concentrations of NO_3-N in the deeper sand and gravel geozone (SGLT5GT10 - Figure 2-7) have not changed substantially since 1985 but the 1987 results indicate a substantial increase in median concentrations (from less than 0.3 to greater than 2.8 mg/l).

The median concentrations of samples from the SG-UA geozone (Figure 2-10) indicate a sharp decrease in 1988 and 1989. The median concentration in 1985 was 4.75 mg/l; in 1989 the median concentration was approximately 34 times less at 0.14 mg/l.

The median NO_3-N concentrations in the shallow weathered till (WTLT15) geozone (Figure 2-9) have reduced continously since 1986. Median concentrations in 1986 were more than 3 times greater than in 1989. The deeper conterpart, geozone WTGT15, has remained fairly stable with median NO_3-N concentrations fluctuating between about 1.5 mg/l and 2.5 mg/l (Figure 2-8). In 1989 the median concentration was 1.9 mg/l.

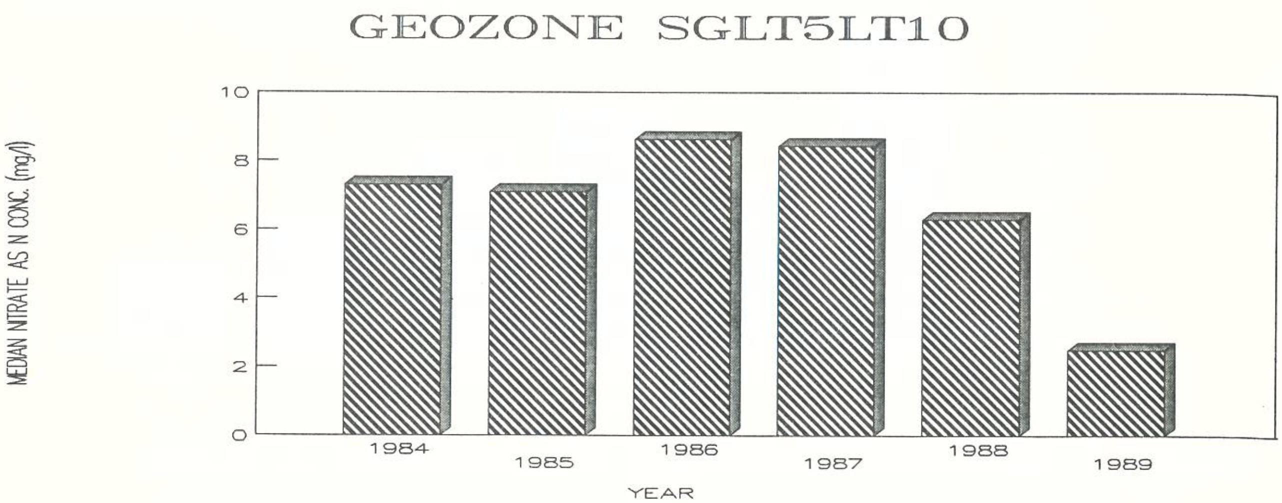


Figure 2-6 Median NO_3-N Concentrations in Geozone SGLT5LT10 w/o OP-Site

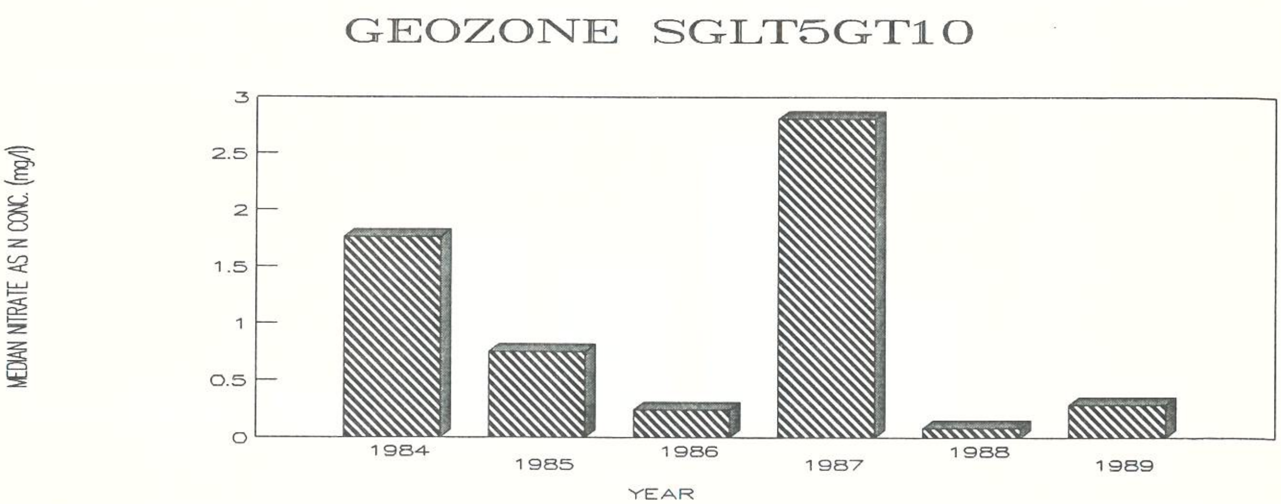


Figure 2-7 Median NO_3-N Concentrations in Geozone SGLT5GT10 w/o OP-Site

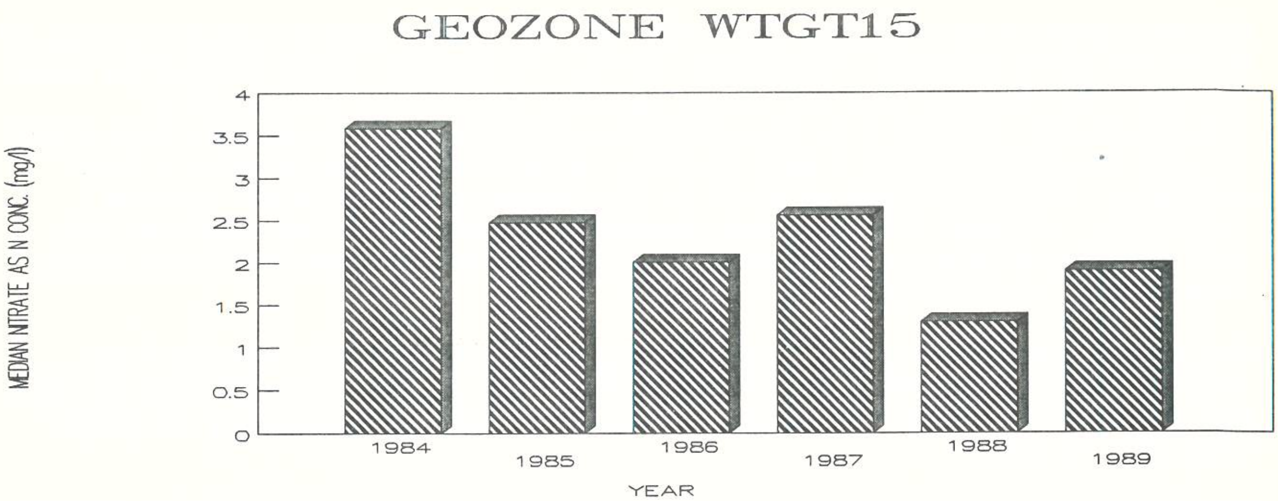


Figure 2-8 Median NO_3-N Concentrations in Geozone WTGT15 w/o OP-Site

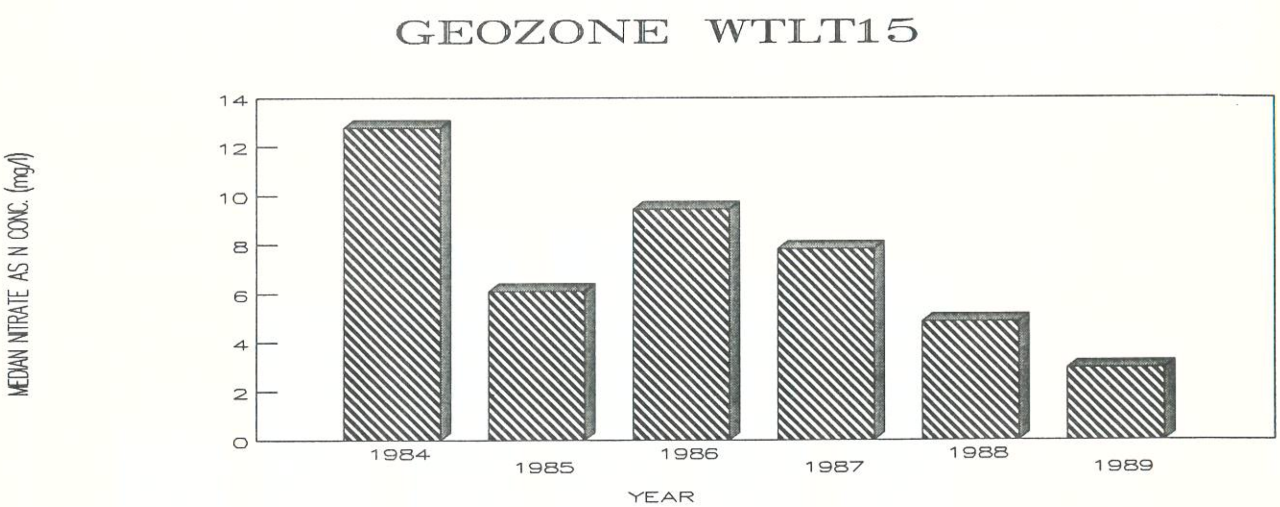


Figure 2-9 Median NO_3-N Concentrations in Geozone WTLT15 w/o OP-Site

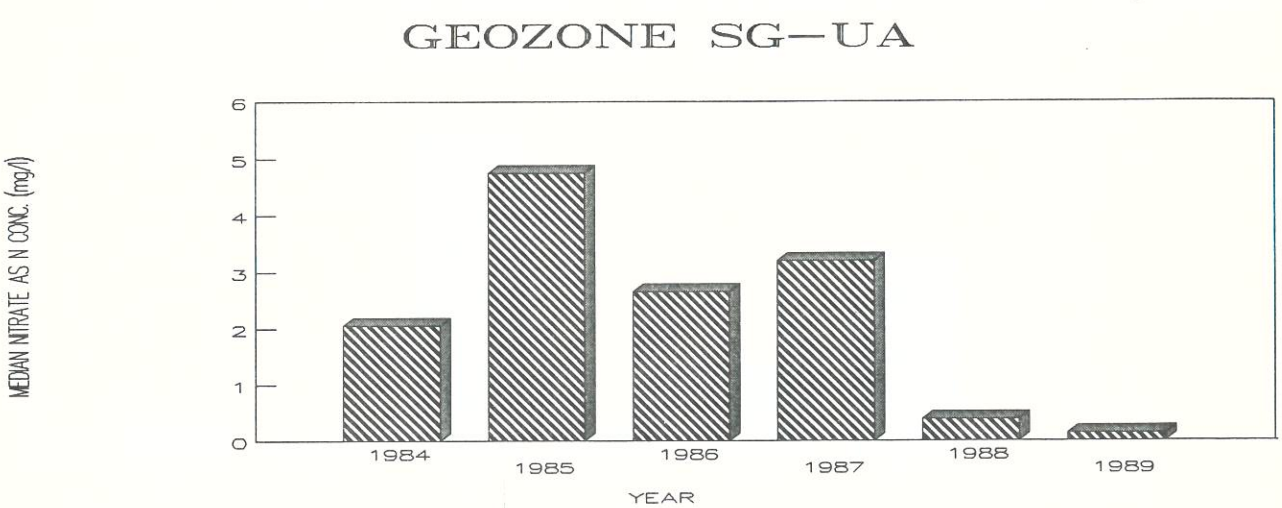


Figure 2-10 Median NO_3-N Concentrations in Geozone SG-UA w/o OP-Site

VC-Site

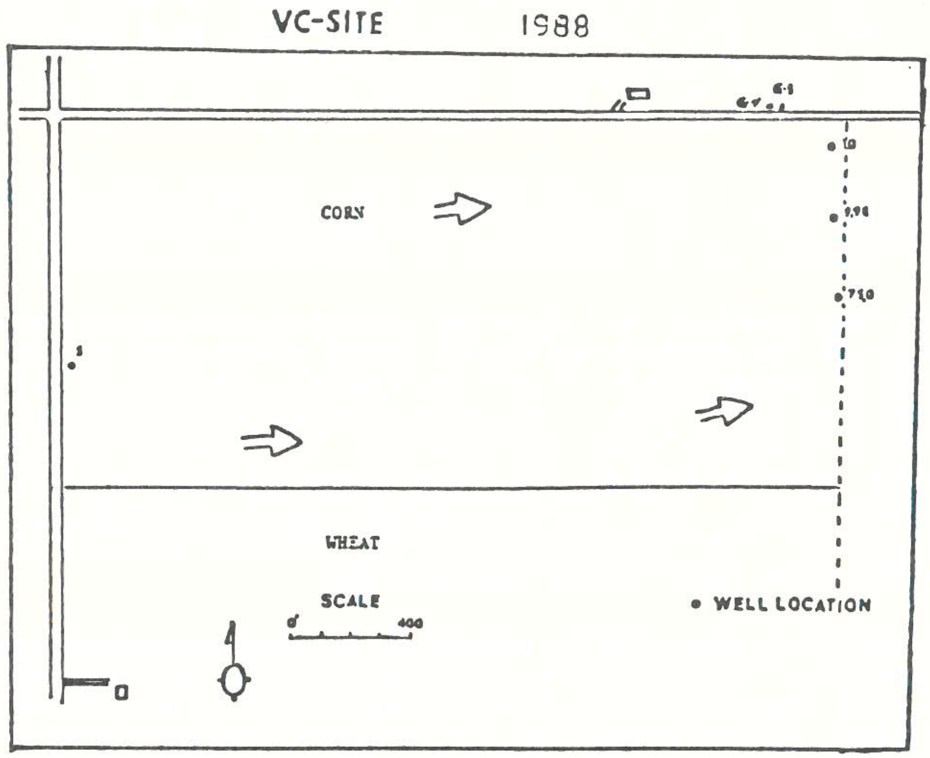
The VC-site was chosen for an in depth analysis because BMPs changed in 1987 from conservation tillage to conventional tillage due to a change in operators. Eight wells were monitored, and samples from seven wells were analyzed for NO_3-N. Two of the wells (VC-7 and VC-10) were sampled on a monthly basis. All of the wells are placed in sand and gravel at various depths ranging from 12 to 29.5 feet. Ground water flow is to the east-northeast.

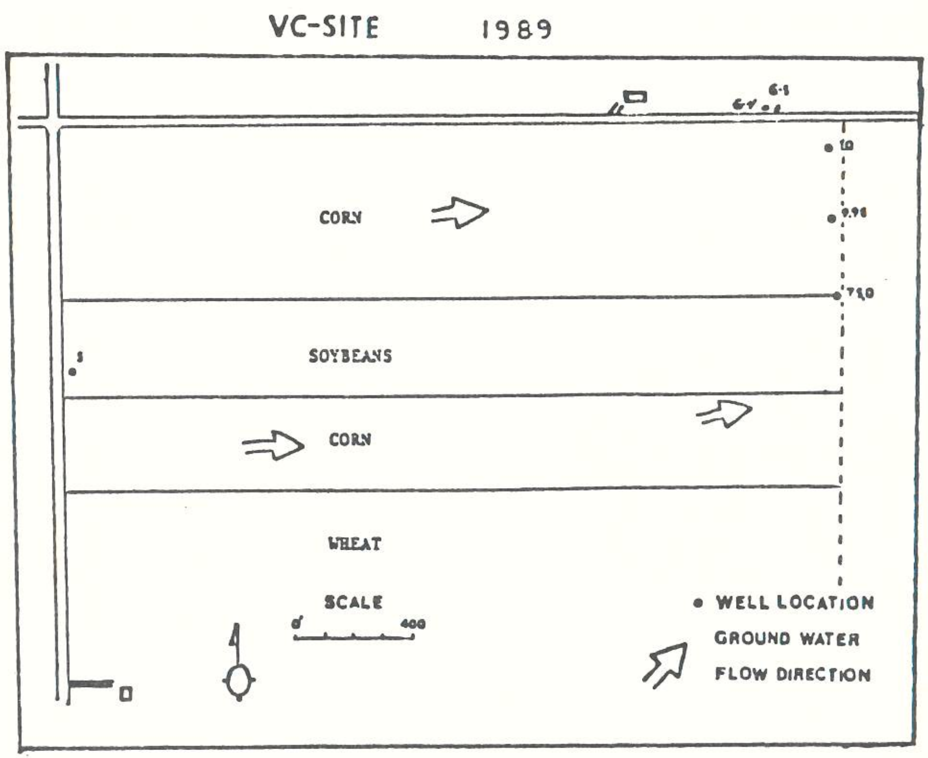
The site was planted in wheat and corn in 1988; soybeans and corn were planted in 1989 (Figure 2-11). The corn was fertilized with 28-28-0 at an application rate of 100 lbs/acre and no fertilizer was applied to the soybeans.

Relative to crop location, well VC-7 was located in corn during 1988 and at the NE corner of the soybeans and the SE corner of the corn in 1989 (Figure 2-11). The NO_3-N concentrations at well VC-7 shows a substantial decrease from 1988 to 1989 (see Figure 2-12). The median concentration decreased from 12.8 mg/l to 0.95 mg/l between years 1988 and 1989. The minimum values were 10.4 and 0.36 mg/l, and the maximum values were 15.65 and 11.28 mg/l for years 1988 and 1989 respectively. Well VC-7 is screened between 16.5 and 21.5 feet below the ground surface. The average depth of the bottom of the screen was 4.37 feet below the water table during 1988 and 1989.

Wells VC-9 and VC-10 show similar reductions in NO_3-N concentrations in 1989, but the magnitude of change not as great as well VC-7 (see Figures 2-13 and 2-14). The wells are screened at 14 - 17 feet and 9 - 12 feet for VC-9 and VC-10 respectively. In the deeper wells at the site, the NO_3-N concentrations have been quite low since monitoring began with all values less than 0.2 mg/l.

Figure 2-11 VC-site crop locations during 1988 and 1989





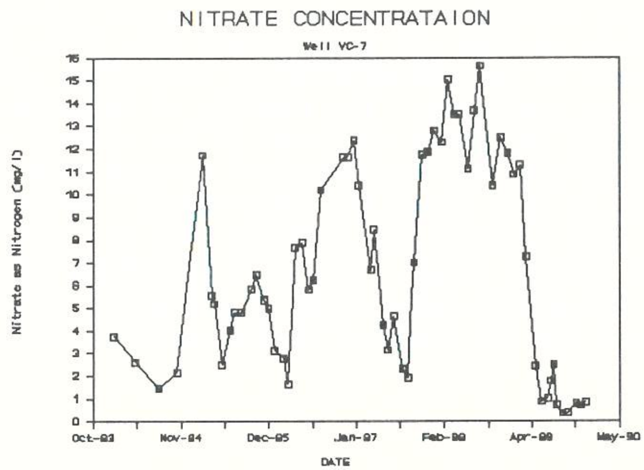


Figure 2-12. NO_3-N concentration vs time from well VC-7

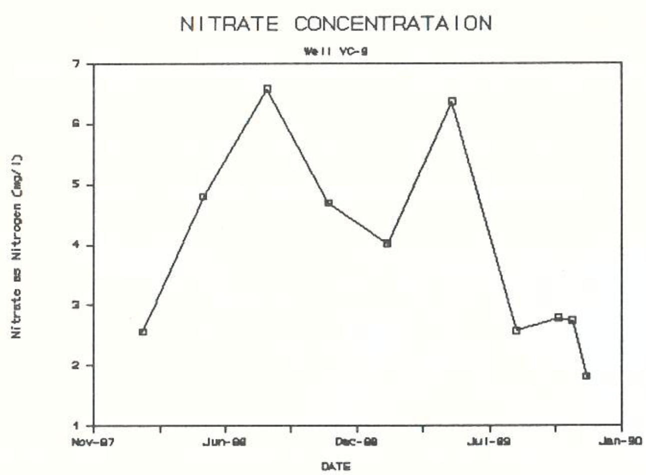


Figure 2-13. NO_3-N concentration vs time from well VC-9

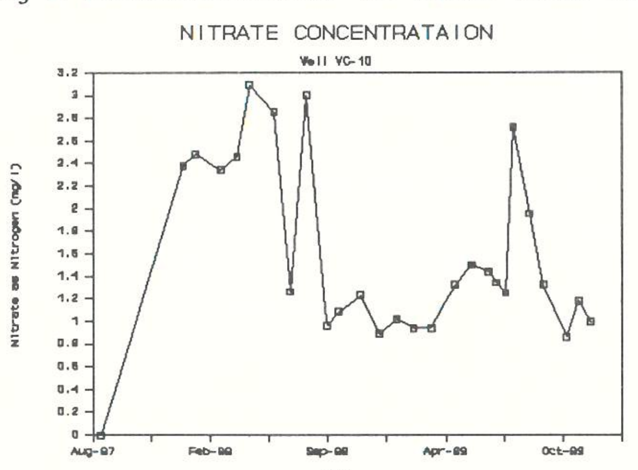


Figure 2-14. NO_3-N concentration vs time from well VC-10

2.3.3.2 Chloride

Analyses of chloride concentrations for all samples collected from May 1984 through December 1989 include all data except data from the previously mentioned wells known to be contaminated by activities not related to agricultural cropping practices. The results of chloride analysis were plotted vs depth below the water table as in the nitrate plot (Figure 2-15). A total of 1418 concentrations were plotted ranging from 0 to 68.5 mg/l at depths from 0 to 43.21 feet below the water table. Chloride concentrations tend to be higher than nitrates so a cutoff line of 10 mg/l was used rather than 5 mg/l on the nitrate graph. The plot indicated that chloride concentrations in the ground water display a similar trend; concentrations greater than 10 mg/l are not found at depths greater than 20 feet (5 exceptions).

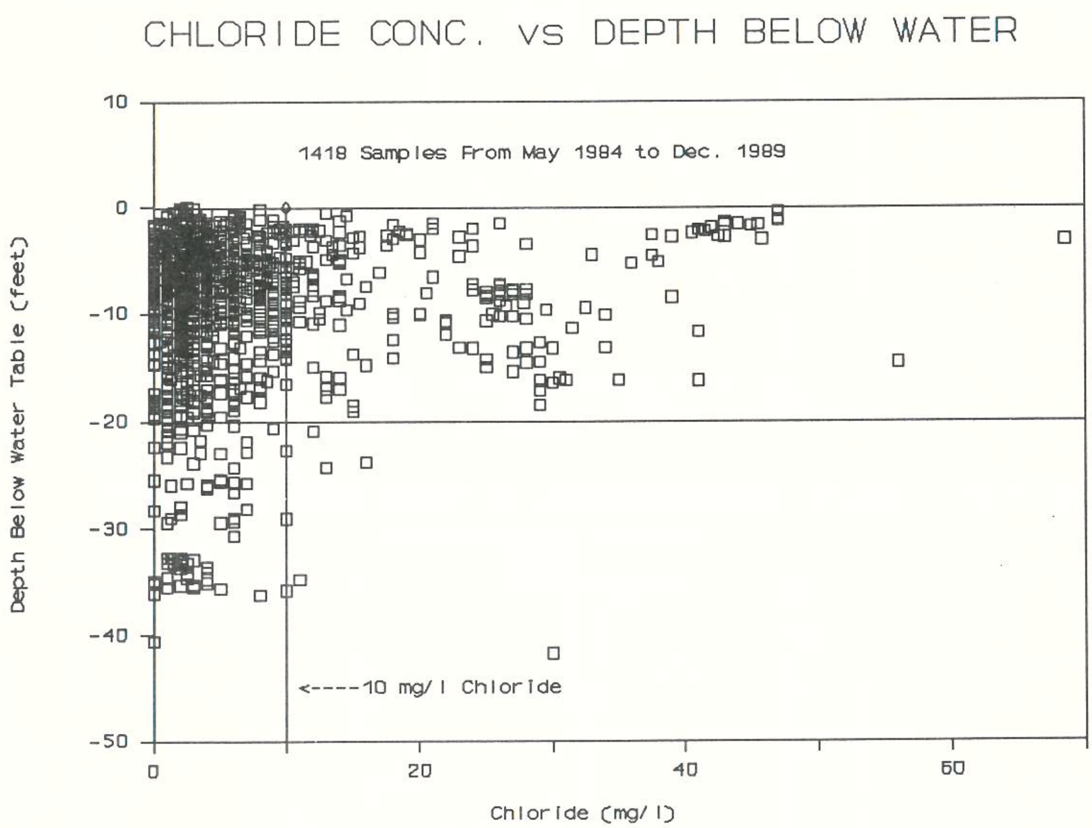


Figure 2-15. Chloride concentration vs depth below water table

2.3.3.3 Dissolved Oxygen

Dissolved oxygen is measured in each monitoring well with an in-situ probe after purging and sampling. Dissolved oxygen versus depth below the water table was plotted and is presented in Figure 2-16. The plot indicates that dissolved oxygen in the ground water decreases to less than 1 mg/l as the 20 feet below water table depth is approached.

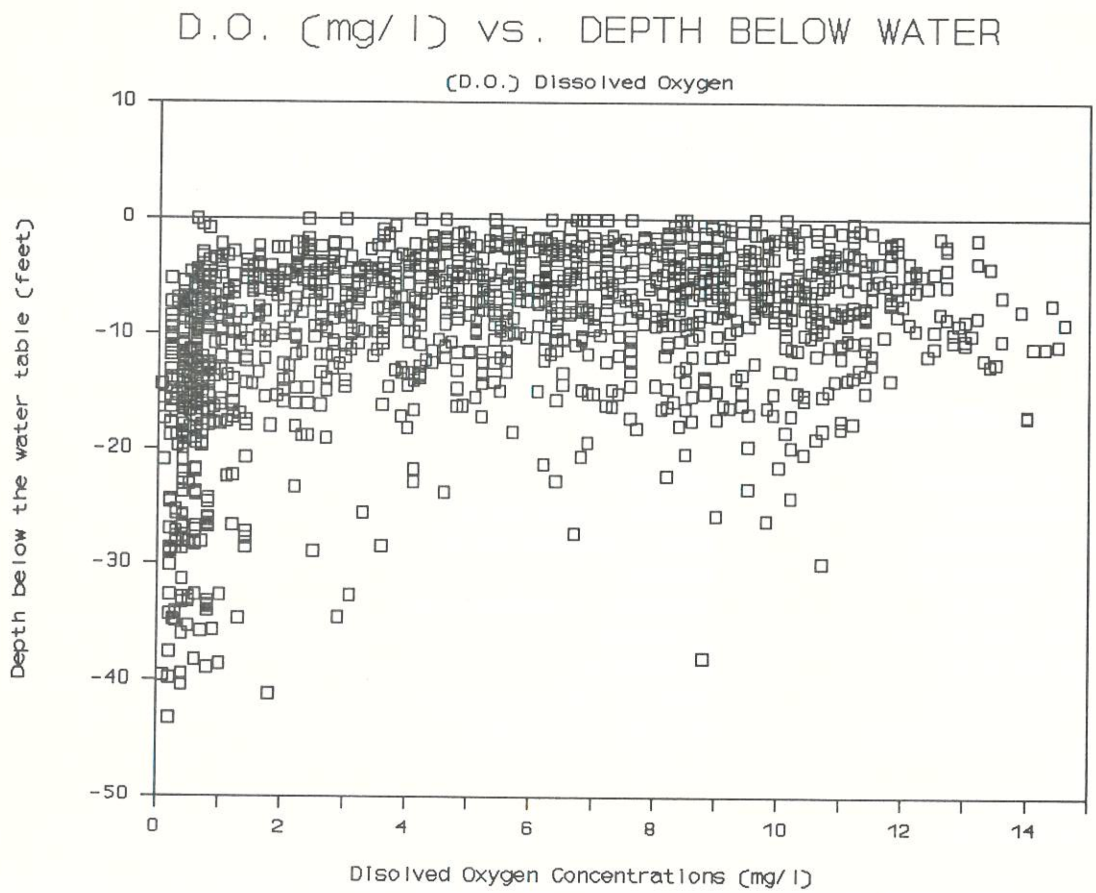


Figure 2-16. Dissolved Oxygen (D.O.) versus depth below water table.

2.3.3.4 Pesticides

Since May 1984, ground water samples have been collected and analyzed for pesticides. With a few exceptions, samples have been collected for pesticide analysis on a monthly basis since 1985. In 1984, four sets of samples were collected in May, July, August and October. Samples were not collected in the months of January and February of 1985. In 1986, samples were not collected in the months of September and October. Samples were collected every month in 1987 and 1988, and in 1989, samples were collected every month except in June and July when sampling was on a bi-weekly schedule.

The term pesticide in this discussion includes both herbicides and insecticides. Samples are laboratory extracted (approximately 500 milliliters) and scanned for 22 pesticides with electron capture gas chromatography techniques. When the analysis detects a pesticide, the sample is analyzed a second time with a different column (thermionic nitrogen-phosphorus selective detector). Selected samples were verified by using gas chromatography, mass spectrometry (GC/MS) in an attempt to eliminate false positives. Conservatively, detection limits are in the range of 0.10 - 0.01 parts per billion. The pesticide scan includes the chemicals in Table 2-2.

Table 2-2. Pesticides Tested via Scan

Ambien (chloramben)	atrazine
Banvel (dicamba)	Bladex (cyanazne)
Counter (terbufos)	Dual (metolachlor)
Dyfonate (fonofos)	endrin
Eradicane (EPTC)	Furadan (carbofuran)
Ramrod (propachlor)	lindane
Sutan (butylate)	parathion
Tordon (picloram)	Sencor (metribuzin)
Treflan (trifluralin)	Thimet (phorate)
toxaphene	2,4-D
methoxychlor	
Lasso (alachlor)	

The pesticides used on the field sites are presented in Table 2-3. With the exception of Basagran and MCPA, the pesticides historically used on the field sites are included in the pesticide scan.

Table 2-3. Pesticides Used On Field Sites

Banvel (dicamba)	Ramrod (propachlor)
Basagran	Sencor (metribuzin)
Furadan (carbofuran)	Treflan (trifluralin)
Lasso (alachlor)	2,4-D
MCPA	Dual (metolachlor)

In 1984, samples were collected from 33 different wells on approximately a quarterly basis. Starting in 1985, samples have been collected from 22 to 34 wells (shallow, representative wells at each monitoring site) on a monthly or bi-weekly basis. As of December, 1989, 1142 ground water samples have been collected from 68 different monitoring wells, and 129 positive detections from

31 different wells have been recorded. Pesticides have been detected in 10.5% of the samples and 46% of the wells.

Two samples which tested positive for Lindane in 1987 are believed to have been collected from wells that were contaminated by cattle. The cattle had been treated with an insecticide and were rubbing against the monitoring wells (knocking off the caps and in one case breaking the well). It is theorized the water samples were contaminated during the sampling process.

Table 2-4 summarizes the pesticide detections. All concentrations are in parts per billion (ppb).

Table 2-4. Pesticides Detected 1984-1989
(1142 samples from 68 wells, 129 detections from 31 wells)

Pesticide	# of Detect	% of Detect	Median (ppb)	Range (ppb)
Alachlor (Lasso)	49	41.8	0.220	0.03 - 3.44
2,4-D	22	18.5	0.440	0.04 - 6.70
Metribuzin (Sencor)	16	13.45	0.09	0.02 - 0.14
Metolachlor (Dual)	11	9.24	0.18	0.03 - 0.96
Trifluralin (Treflan)	3	2.52	0.047	0.02 - 0.38
Dicamba (Banvel)	3	2.52	0.120	0.07 - 0.12
Lindane	2	1.68	0.05	0.04 - 0.06
Picloram (Tordon)	2	1.68	0.46	0.42 - 0.50
Atrazine	3	2.52	0.34	0.07 - 5.4
Bladex (cyanazine)	2	1.68	0.465	0.19 - 0.74
Dyfonate	8	6.72	0.032	0.14 - 0.05
Parathion	8	6.72	0.285	0.04 - 13.6

The majority of the pesticide concentrations approach detection limits of the techniques and equipment. Examining the data for continuity revealed that 85% of the detections were one time events with no detection in that well in the following sample (typically four weeks later). On a year by year basis the pesticides detected are presented in Table 2-5.

Table 2-5. Pesticide Detections by Year

Pesticides Detected 1984
(55 samples from 33 wells, 3 detections in 3 wells)

Pesticide	# of Detect	% of Detect	Median (ppb)	Range (ppb)
2,4-D	3	100	0.45	0.42 - 0.80

Pesticides Detected 1985
(125 samples from 22 wells, 23 detections in 9 wells)

Pesticide	# of Detect	% of Detect	Median (ppb)	Range (ppb)
Alachlor (Lasso)	20	87	0.315	0.03 - 3.09
Trifluralin (Treflan)	1	4.3	-	0.02
Atrazine	1	4.3	-	5.4
Metribuzin (Sencor)	1	4.3	-	0.02

Pesticides Detected in 1986

(161 samples from 24 wells, 14 detections in 10 wells)

Pesticide	# of Detect	% of Detect	Median (ppb)	Range (ppb)
Alachlor (Lasso)	6	43	0.17	0.07 - 0.43
2,4-D	7	50	0.43	0.13 - 6.7
Metribuzin (Sencor)	1	7	-	0.03

Pesticides Detected in 1987

(219 samples from 26 wells, 5 detections in 5 wells)

Pesticide	# of Detect	% of Detect	Median (ppb)	Range (ppb)
Alachlor (Lasso)	2	40	1.87	0.30 - 3.44
2,4-D	1	20	-	0.18
Lindane (suspected cattle contamination)	2	40	0.05	0.04 - 0.06

Pesticides Detected in 1988

(277 samples from 34 wells, 35 detections in 18 wells)

Pesticide	# of Detect	% of Detect	Median (ppb)	Range (ppb)
Alachlor (Lasso)	12	34	0.07	0.036 - 1.38
2,4-D	9	26	0.65	0.04 - 5.16
Metribuzin (Sencor)	4	11	0.085	0.08 - 0.124
Metolachlor (Dual)	5	14	0.35	0.28 - 0.96
Trifluralin (Treflan)	2	6	0.21	0.047 - 0.38
Dicamba (Banvel)	2	6	0.095	0.07 - 0.12
Picloram (Tordon)	1	3	-	0.5

Pesticides Detected in 1989

(305 samples from 35 wells, 49 detections in 20 wells)

Pesticide	# of Detect	% of Detect	Median (ppb)	Range (ppb)
Alachlor (Lasso)	9	18.4	0.390	0.06 - 0.14
2,4-D	2	4.1	0.290	0.13 - 0.45
Metribuzin (Sencor)	10	20.4	0.092	0.06 - 0.14
Metolachlor (Dual)	6	12.2	0.090	0.024 - 0.18
Dicamba (Banvel)	1	2.0	-	1.55
Picloram (Tordon)	1	2.0	-	0.42
Bladex (Cyanazine)	2	4.1	0.465	0.19 - 0.74
Dyfonate	8	16.3	0.032	0.05 - 0.14
Parathion	8	16.3	0.270	0.04 - 13.6

Three pesticides detected in 1989 which have not previously been detected since the project began are Dyfonate, an insecticide to control rootworms, Parathion, an insecticide for grasshopper control, and Bladex, a broadleaf herbicide. No history of use for these pesticides has been reported by any of the site owners or operators.

Samples for pesticide analysis have been collected from wells ranging in depth below ground surface from 8.5 feet to 65.5 feet. Pesticide concentrations may be affected by transport through the soils, so the depth below ground surface is used when referring to the depth of a well. This is in contrast to depth below the water table used during the nitrate discussion. An analysis of depth below water table for pesticides has also been completed, however, for this report (Figure 2-17). The average depth of the sampled wells was 18.5 feet. The average depth of wells in which pesticides have been detected was 17.7 feet.

Figure 2-17a indicates the depth below the water table and the concentrations of pesticide detections. Approximately 95% of the detections are found in the upper 20 feet where the majority of pesticide samples have been collected. Figure 2-17b presents the range of depths below the water table where pesticides have been sampled.

Samples for pesticide analysis have been collected from every geozone. Figure 2-18 (Geozones With Pesticides Samples) presents the data as a pie diagram. The shallow sand and gravel with thin topsoil (SGLT5LT10) has been sampled more than the other geozones with the samples from the sand and gravel geozones as a group constituting approximately 65% of the total number of samples.

Ground water samples from eight of the eleven geozones have had pesticide detections (Figure 2-19). Samples from the SC, SGGT15, and SG5-15GT10 geozones have not had any pesticide detections. The shallow weathered till (WTLT15) samples have had the most pesticide detections, 29.5% of all pesticide detections. The glacial tills as a group constitute approximately 45% of the samples which detected pesticides.

Figure 2-20 presents the frequency of pesticide detections with time. With the exception of 1987, which had very few detections, it appears that the time when pesticides are most likely to be detected clusters around July. It is apparent that 1988 and 1989 have had a relatively large number of detections. Years 1988 and 1989 constitute 62.8% of all detections and 50% of the total samples collected since sampling began. Year 1989 comprised more pesticide detections than any previous single year, however, 28 more pesticide samples were collected during 1989 than in 1988.

Figure 2-21 presents in a pie chart format, the months when pesticides were detected. Samples collected in May through August comprise 64.1% of samples which detected pesticides. Samples collected during the five month period of May through September represent 75% of the samples which detected pesticides.

Figure 2-22 present pesticide detections by date and water level elevation for well JA-1M at the RS-site. There has been a history of repeated detections of Lasso and Sencor at this site. Records indicate that Lasso has been used regularly since 1981, but the origin of Sencor remains unclear. Other pesticides which have been applied include 2,4-D, Banvel, MCPA, Furadan, and Basagran but not as extensively as Lasso. During the wet years in 1985 and 1986, Lasso was detected 12 times. Since 1986, it has been detected only twice, once in 1988 and once in 1989.

Fiberglass reinforced epoxy resin (ER) monitoring wells were installed at five sites in 1987. The ER wells were installed at the same location and depth as an existing PVC well to determine if the PVC material was having an effect on pesticide concentrations. They were installed November 1987 at the OP (well OP-5E), VC (well VC-9E), RS (well JA-2E), BL (well BL-11E) and JW (well JW-7E) sites. Well JA-2E has been dry since installation. One sample was collected from well BL-11E before water levels dropped and samples could no longer be obtained. Well OP-5E was sampled six times before it went dry in November 1988. The remaining two wells VC-9E (sampled 22 times) and JW-7E (sampled 19 times) continue to be sampled. In total, 48 samples have been collected from the ER wells.

Seven samples of 48 collected from the ER wells have detected pesticides. Alachlor was detected in wells JW-7E and VC-9E, and trifluralin (Treflan) was detected in well OP-5E. The alachlor concentrations were the highest ever detected at the respective sites. The trifluralin had not been detected at the OP site previous to this, and the only other detection at the OP-Site that year (1988) was alachlor. Metolachlor (Dual) was detected for the first time (August, 1989) in VC-9E; it was not detected in the adjacent well VC-9. Dual was applied on the field site on May 10, 1989. Dyfonate (Fonofos) and Parathion, two pesticides which have not previously been detected, were found in well JW-7E. The adjacent well, JW-7S, also detected these pesticides on the same date in approximately the same concentrations.

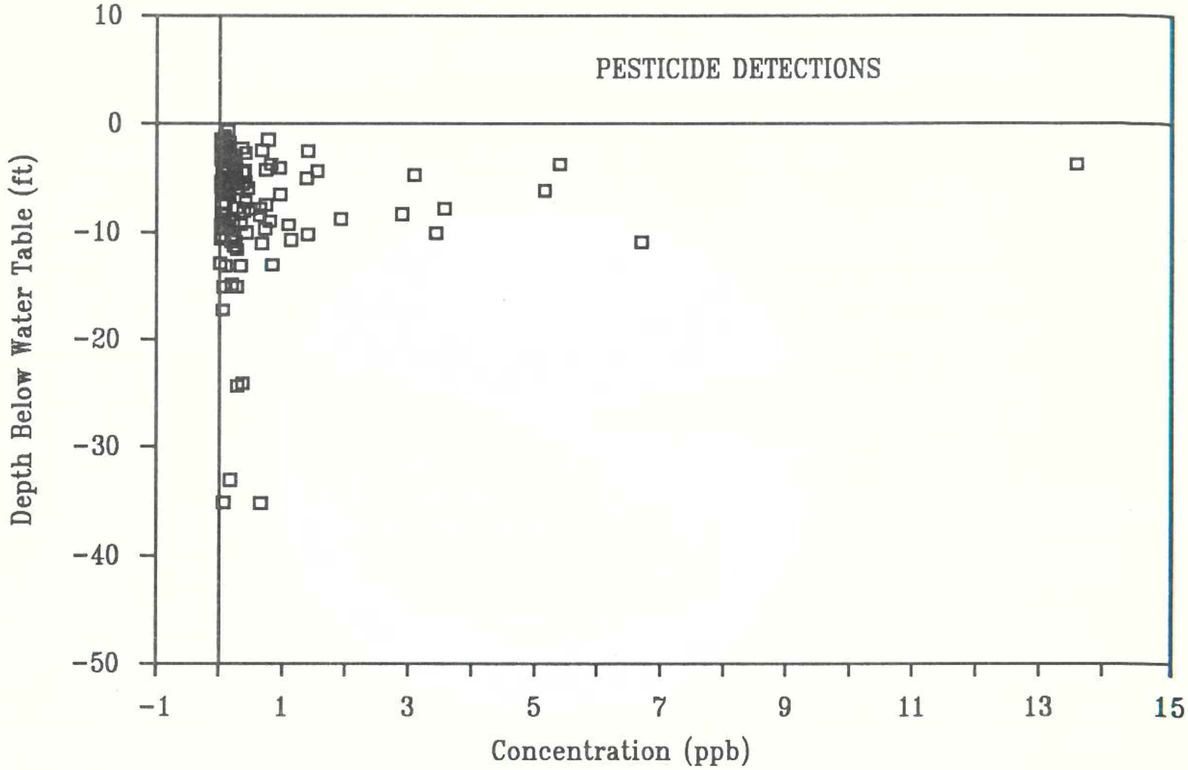


Figure 2-17a. Depth below water table where pesticides were detected

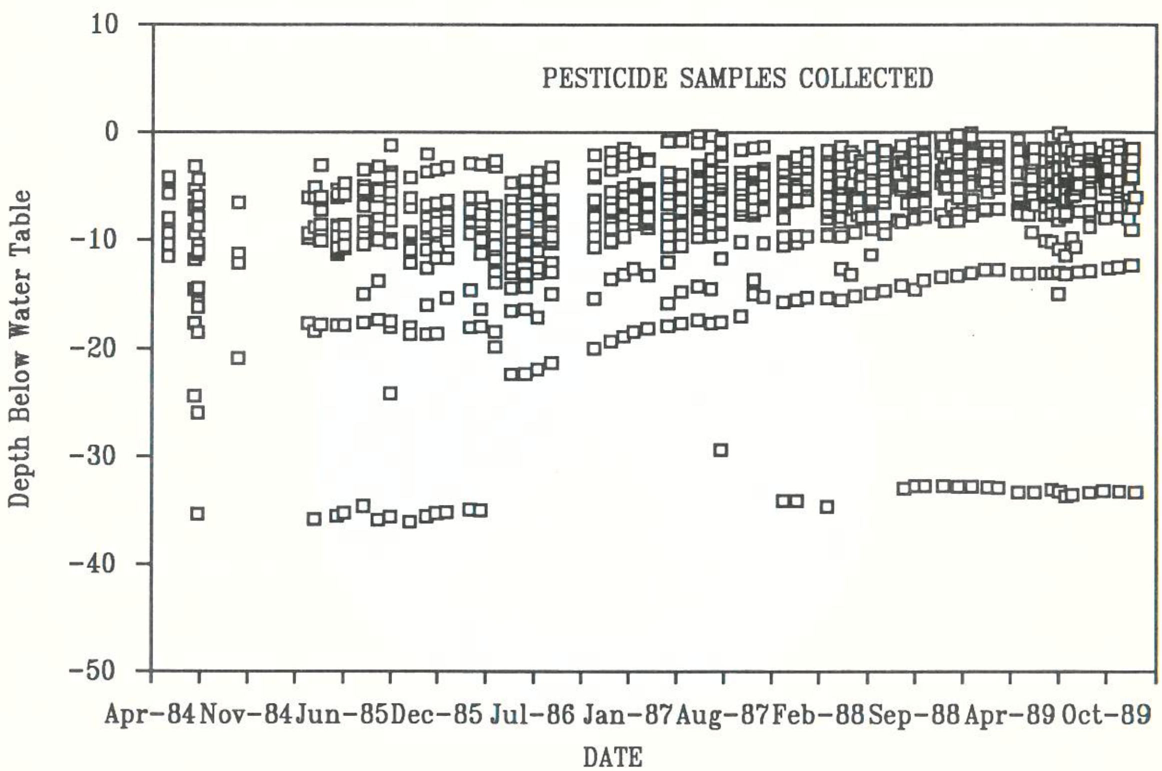


Figure 2-17b. Depth below water table where pesticides were sampled

Geozones with Pesticide Samples

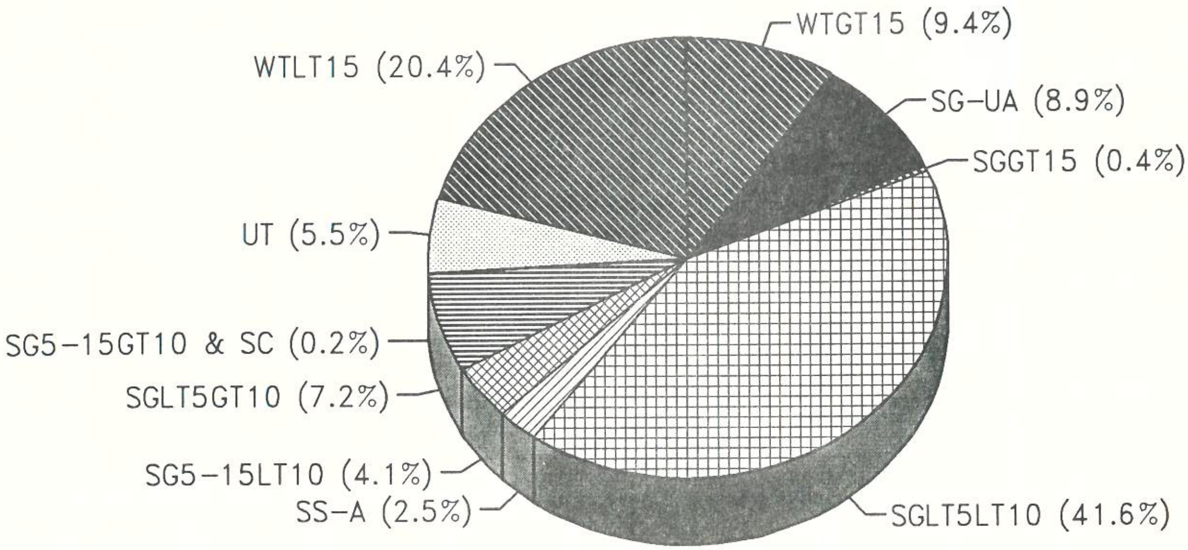


Figure 2-18. Geozones where pesticides were sampled.

Geozones with Pesticide Detections

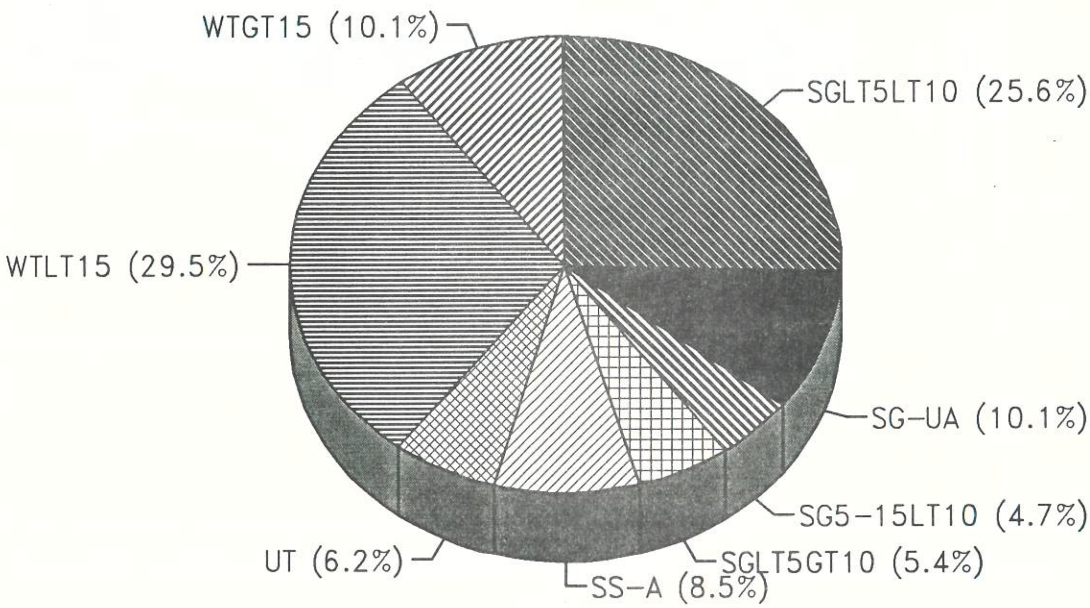


Figure 2-19. Geozones where pesticides were detected.

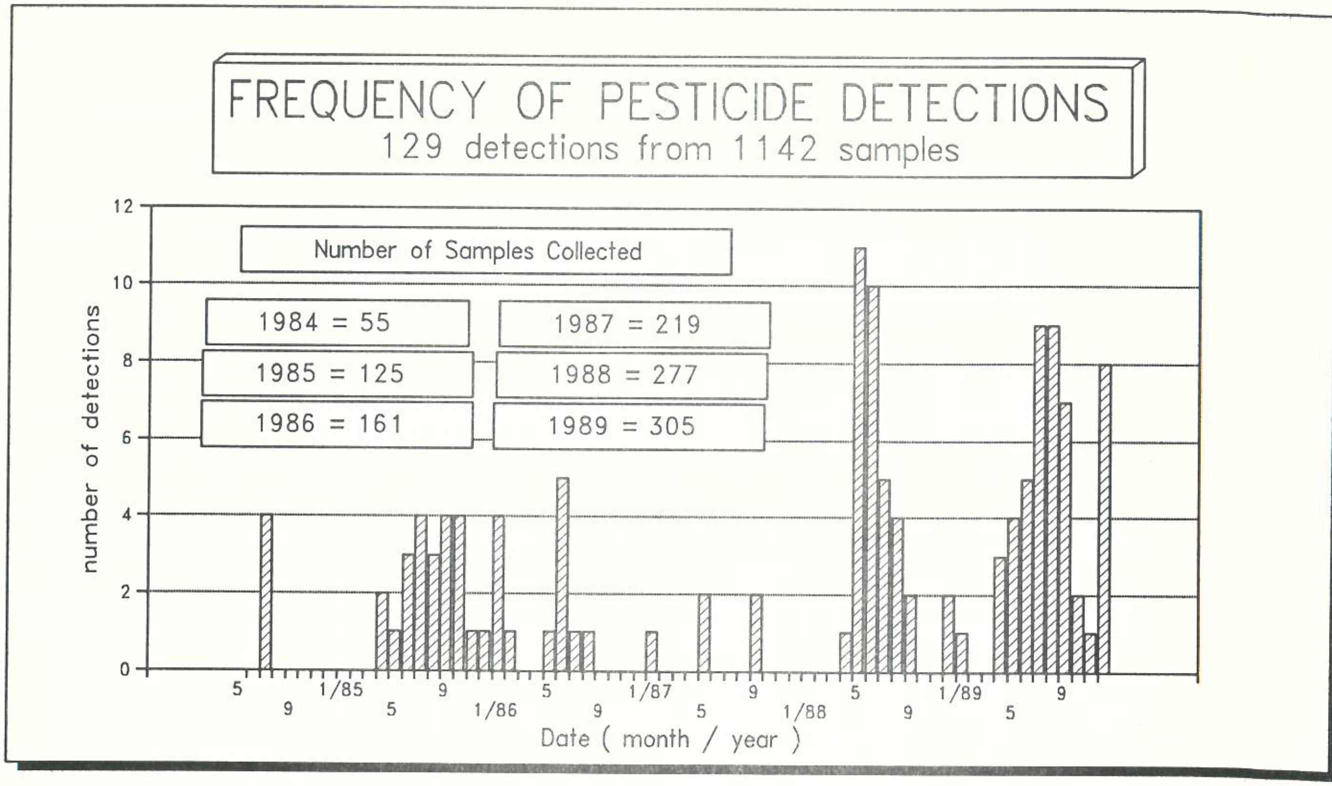


Figure 2-20. Frequency of Pesticide detections.

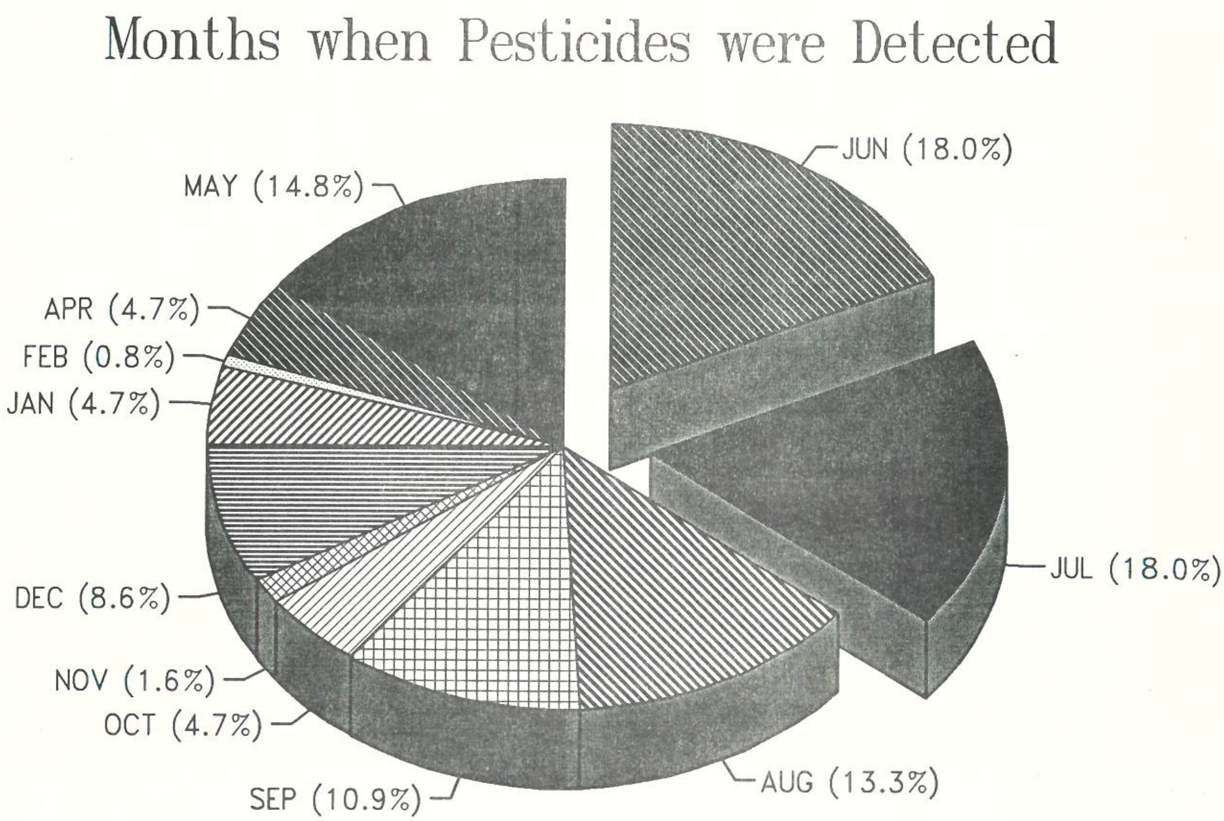


Figure 2-21. Months when pesticides were detected.

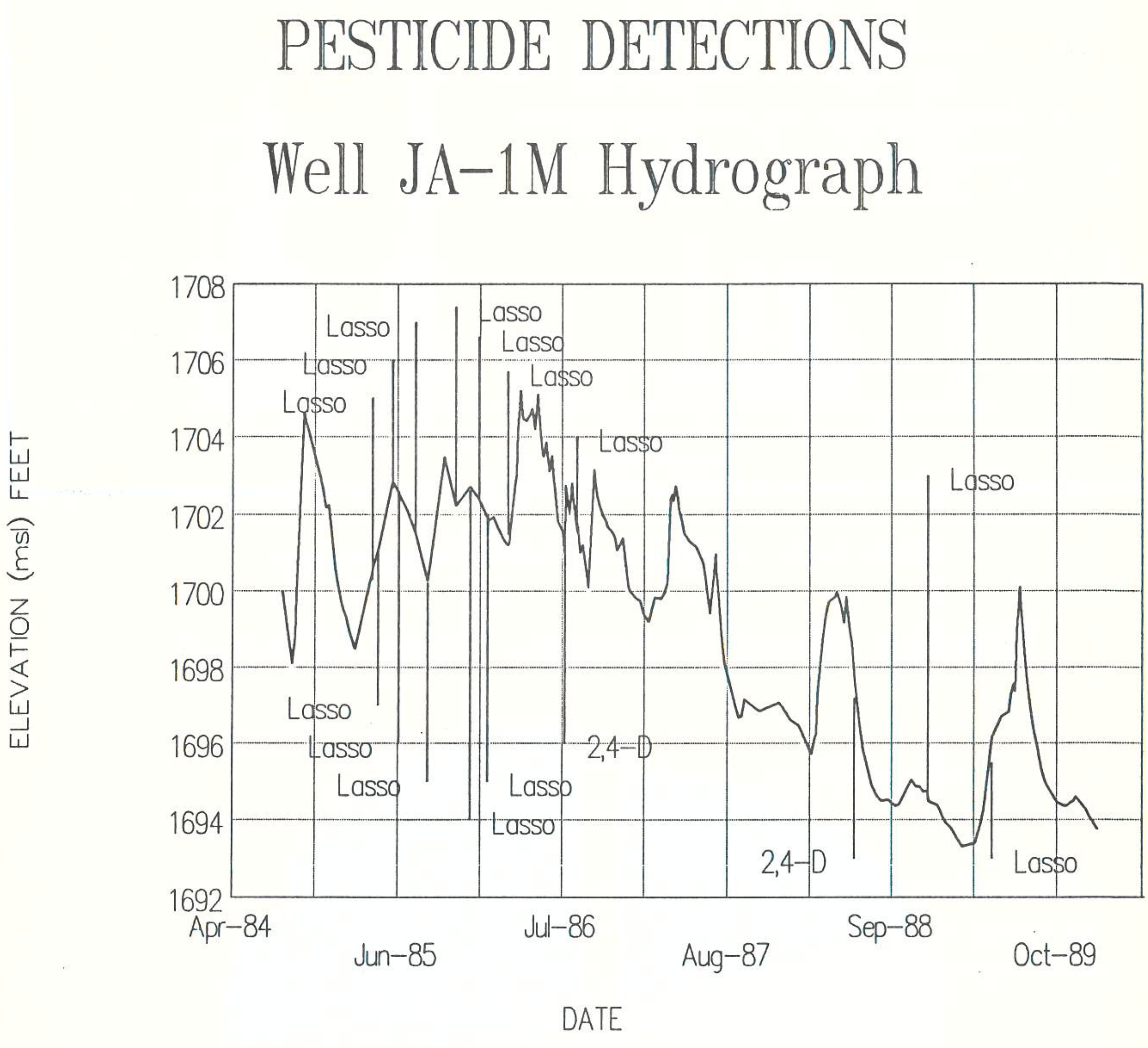


Figure 2-22. Pesticide detections in well JA-1M (RS-site). Vertical lines on graph indicate pesticide detection date and elevation of water table at time of detection.

2.3.4 Field Site Runoff Water Chemistry

Surface water samples were collected at the BL-site at the sharp crested weir during snow melt and major precipitation events during the spring and summer of 1989. The samples were analyzed for the following parameters:

1. nitrate
2. nitrite
3. ammonia
4. organic nitrogen
5. chloride
6. sulfate
7. suspended solids
8. total solids
9. hardness
10. total phosphorus
11. orthophosphorus (soluble reactive P)
12. pesticide scan (collected on 7/11/89)

The results of the water quality analysis are summarized in Table 2-6. Surface water chemistry is considerably different than ground water; organic nitrogen concentration is much higher while nitrate as nitrogen is lower. Average total phosphorus concentration is also much higher in the runoff than ground water.

Plots of nitrate, suspended solids, and total phosphorus concentrations with time of samples collected on July 11, 1989 during a major precipitation event are presented in Figures 2-23 and 2-24. Initial nitrate - nitrogen concentrations are relatively high (16.2 mg/l) and gradually reduce to 1.5 mg/l over an 18 hour period (Figure 2-23). Suspended solids follow the same basic pattern with high initial concentrations reducing to near zero after the 18 hour period (Figure 2-24). Initial total phosphorus concentrations were about 5.5 mg/l and then remained in the 1 to 2 mg/l range throughout most of the sampling period (Figure 2-23).

A plot of NO_3-N with time is presented in Figure 2-25 for wells BL-12S and BL-12D which are located near the sharp crested weir and screened in sand and gravel. The plot indicates that nitrates increased sharply between May 22 and August 7 in both the shallow and in the deeper well. Three major precipitation events occurred between these dates and water ponds in the field near these wells. The concentration of NO_3-N peaked at 9.51 mg/l on October 11. After peaking, the concentration declined to approximately two milligrams per liter on December 13 in both the shallow and deep wells.

A total of four pesticides (Bladex, Lasso, Dyfonate, and Treflan) were detected in runoff samples collected on July 11, 1989. Pesticides were detected in 16 of 17 water samples. Table 2-7 is a summary of the results. Actual concentrations may be higher since samples were collected in plastic containers and some may have been lost due to adsorption. Approximately 84% of the detections were Lasso and Bladex while Dyfonate and Treflan made up 5.4% and 10.8% percent respectively (Figure 2-26). Bladex is the only one of these pesticides which was detected in the ground water at the BL-site during 1989.

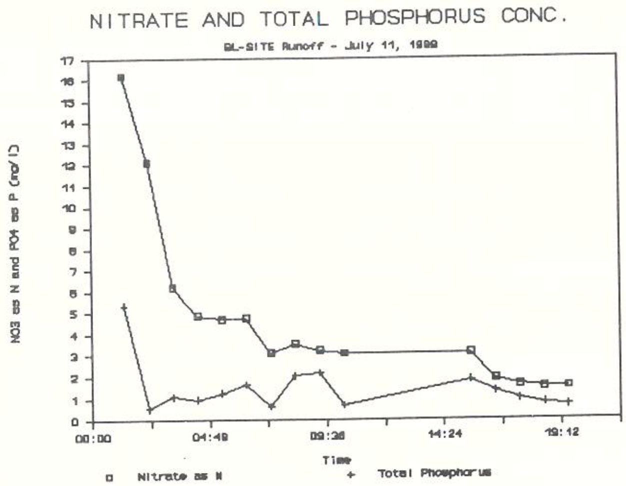


Figure 2-23. Total PO_4-P and NO_3-N in BL-site runoff samples

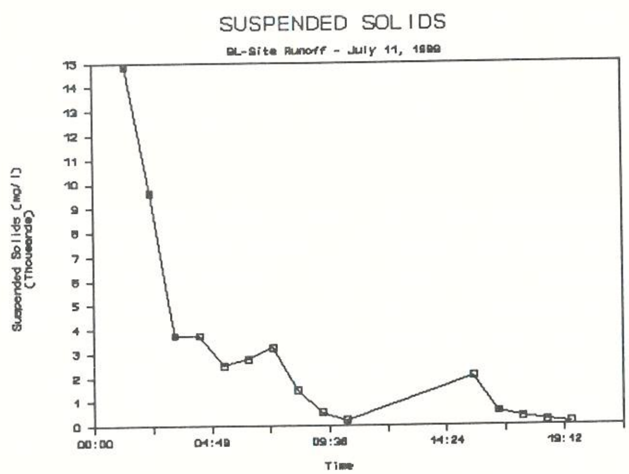


Figure 2-24. Suspended solids in BL-site runoff samples

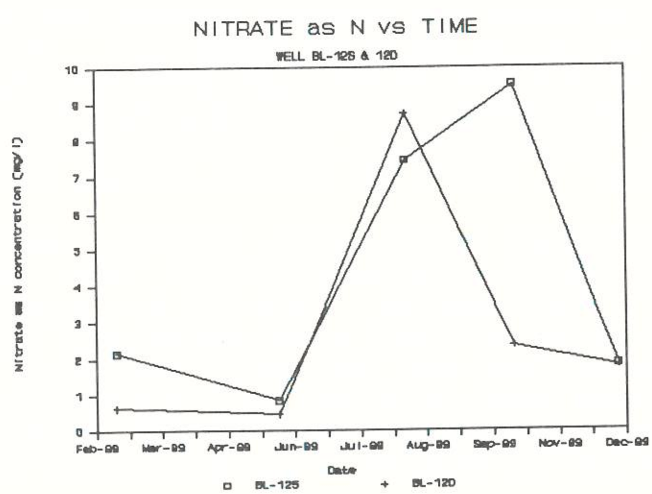


Figure 2-25. NO_3-N concentrations in 1989 water samples from BL-12 wells at BL field site.

Table 2-6. Selected Field Site Runoff Water Quality Results For 1989 (mg/l)

	Suspended Solids	Total Dissolved Solids	Ortho Phos.	Total Phos.	Nitrate as N	Nitrite as N	Organic Nitrogen	Ammonia as N
Min.	21	48	0.12	0.17	1.24	0.012	1.8	0.04
Max.	14877	6237	1.06	5.35	18.88	0.7	62	6.36
Mean	2764.71	746.65	0.40	1.19	4.44	0.10	14.87	1.95

Table 2-7. Results of Pesticide Analysis of Field Site Runoff (ppb)

Pesticide	Min	Max	Mean
Lasso	0.06	0.74	0.523
Bladex	0.13	1.1	0.533
Dyfonate	0.054	0.093	0.074
Treflan	0.049	0.19	0.102

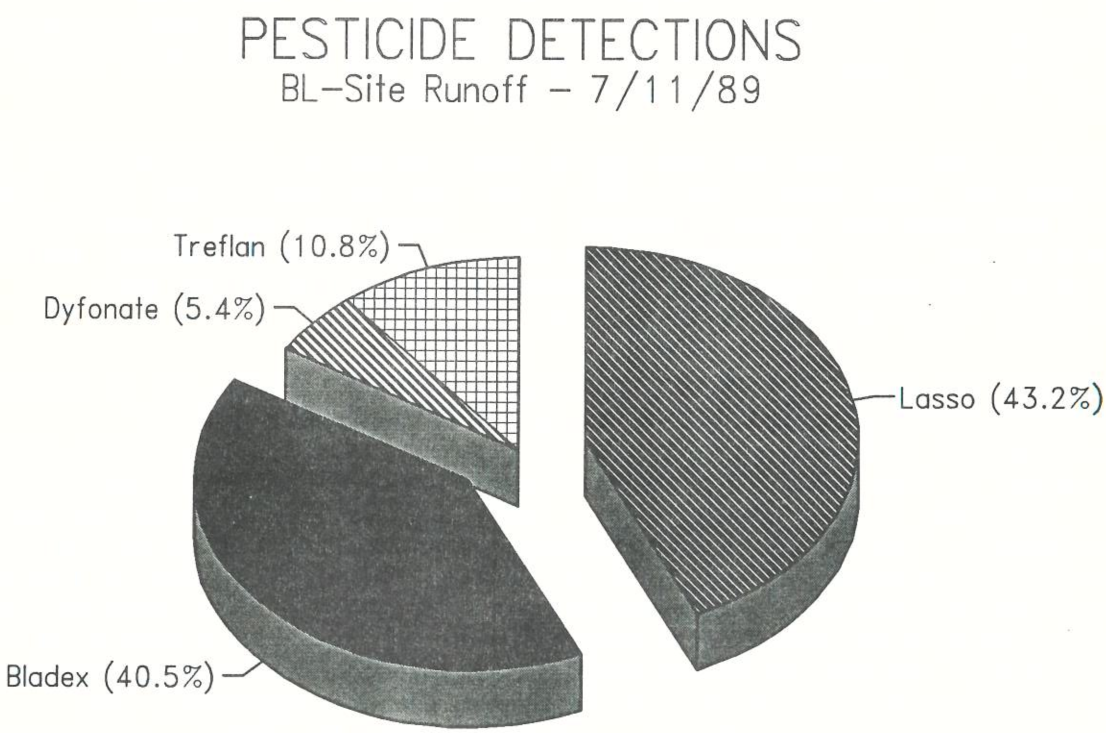


Figure 2-26. Pesticide detections in runoff at BL-site on July 11, 1989

2.4 Discussion

2.4.1 Hydrogeology

In response to increased precipitation in 1984 through 1986 water levels rose to a peak in May of 1986. Precipitation has been below normal in 1987, 1988, and 1989, and water levels which were abnormally elevated, have dropped dramatically. The water level drop has caused 23 monitoring wells to go dry. Three of the wells were epoxy resin wells installed in the fall of 1987 for pesticide detection verification.

To collect discrete samples near the top of the water table, the monitoring wells could not be screened deeply into saturated conditions. This resulted in shallow wells which went dry because long term declines in water levels were experienced. Though this is unfortunate, many of the wells are nested, and the deeper well can now be used for sampling.

The advantage of the changes in water levels recorded during the term of the project is the opportunity to evaluate nitrogen and other chemical parameter response to both a rising and falling ground water table. It does, however, make it more difficult to determine whether changes in chemical concentrations are the result of land use practices or major changes in precipitation.

2.4.2 Nitrates

The depth at which NO_3-N concentrations greater than 5 mg/l were found is less than 20 feet below the water table. The high (as high as 47.5 mg/l) NO_3-N concentrations attest to the amount of nitrogen that can enter the ground water. In the time since nitrogen has been applied in this area (approximately 20 to 30 years) it can be calculated to have had the ability to move to greater depths than it has (see 1987 Annual RCWP Progress Report - Project 20 for calculations of travel distances).

The plot of NO_3-N concentrations versus depth below the water table has been presented for four years, 1986, 1987, 1988, and now 1989 (1986, 1987, and 1988 Annual RCWP Progress Report - Project 20). Since 1986 when it was first presented over 1,700 additional water samples have been collected, and the 20 feet below the water table critical depth for 5 mg/l NO_3-N has been maintained, with only the one exception in 1989.

The reason why nitrates have not traveled to greater depths may be attributed to denitrification. Denitrification is the biological reductive transformation of nitrate or nitrite to gaseous nitrogen. Denitrification has been shown to take place in a shallow sandy ground water environment (Trudell, M. R., Gillham, R. W., Cherry, J. A., 1986) similar to the sandy aquifers in South Dakota. The carbon source necessary for denitrification may be provided by infiltrating water. Trudell, et al. (1986) states that "During recharge events when surface soil increases to near saturation, it is possible that organic carbon dissolved in the surface soil zone could be transported to the water table." Though denitrification is an anaerobic process it has been shown to take place in low (0.7 mg/l) oxygen or even apparently well oxygenated conditions (Focht D.D., and Verstraete, W., 1977). As the plot of dissolved oxygen versus depth below the water table illustrates (Figure 2-8), oxygen decreased rapidly at about the same depth that nitrate reduction began.

There seems to be no reason to preclude the possibility of denitrification limiting the depth to which nitrate can travel.

The median NO_3-N plots indicate that nitrates as nitrogen in the ground water on the RCWP/CM&E project area have been decreasing for the last three years. Below normal precipitation during the last three years of the project correlates with the decrease in NO_3-N concentration reduction. Since nitrates enter the ground water through infiltration, it follows that less NO_3-N would reach the ground water with fewer precipitation events. If NO_3-N is being adsorbed in the soil profile NO_3-N spikes should be evident during 1990 precipitation events if sampling is conducted during and after the event. If less NO_3-N is reaching the ground water and denitrification is occurring, NO_3-N should be decreasing in place.

The results of the analysis of the VC-site implies that rapid changes in water quality due to land use change may be possible. Reduction in nitrate as nitrogen concentrations at the VC site, especially in well VC-7, appears to be the result of less nitrogen applied due to the soybean crop. Wells VC-9 and VC-10 may not be as affected by the change because of the greater distance from the soybean field; the changes in NO_3-N concentrations seem to reflect this.

2.4.3 Chloride

The analysis of chloride concentrations with depth raises some new questions to the fate of nitrate as nitrogen in the ground water. Chloride, a conservative water quality parameter, seems to follow the same trend as NO_3-N with most concentrations greater than 10 mg/l found less than 20 feet below the water table (Figure 2-15). Most chloride concentrations in the ground water can be attributed to naturally occurring processes. Chloride occurs naturally in topsoil and deeper formations and increases with high mineral content; it is highly soluble in water and leaches into the ground water. Agricultural sources of chloride include fertilizer (KCl) and some pesticide compounds, however the amounts would be low relative to naturally occurring sources. Since chloride reduction at depth cannot be attributed to denitrification, it is possible that nitrate as nitrogen may be moving deeper into the profile with continued loading of nitrogen into the system. Continued monitoring of nitrates in the ground water under agricultural areas will be needed to quantify this possibility.

2.4.4 Pesticides

The most commonly detected pesticides were alachlor and 2,4-D, accounting for 55% of the pesticide detections. These pesticides were also used most frequently. Alachlor is used on corn and soybeans while 2,4-D is used only on corn. Many crop rotations throughout the project area include corn and soybeans, increasing the probability that one of these two pesticides would be used on the field sites.

Sampling for pesticides in ground water has taken place primarily in the shallow areas of the saturated strata. The analysis of geozones where pesticides were detected indicates that pesticides are most prevalent in the shallow zones. Approximately 66% of the pesticide samples were collected from the shallow sand and gravel and shallow glacial till geozones which constitute about 60% of the samples which detected pesticides.

In the last two years pesticide detections have increased in the sand and gravel geozones, in contrast to 1984 through 1987, when the majority of detections were in the glacial tills. Presently, the detections are approximately equal in both geozone types.

The presence of pesticides in the tills has been tentatively linked to the presence of cracks, joints and fractures known to exist in the tills (Connell, D. E., 1984). The fractures would allow rapid and direct access for infiltrating water to the ground water where samples were collected. Glass, Steenhuis, and Paralange (1988) found that the wetting front could move through sands as fingers rather than a pulse-like front. Finger structures were more distinct in dry soils and could serve as pathways for flow, transporting water and contaminants faster than the wetting front theory. This theory is a possible explanation for the increased pesticide detections in the sand and gravel geozones in 1988 and 1989.

Although pesticides have been detected in every month but March, the majority (75%) of the pesticide detections occur in a five month period of the year of May through September. Pesticide application takes place, in order of most frequent to least, in June, May, July, and April. The highest number of detections occur in the months of June and July (Figure 2-21). This indicates approximately one month lag time between application and detection.

The Lasso detections in well JA-1M (RS-site) may be a function of precipitation since Lasso has been applied nearly every year, and detections were more numerous during the wet years. Figure 2-22 indicates only 3 pesticides were detected since 1987, and the detections appear to be during a recharge (2 on the descending and 1 on the ascending limb). The pesticides may be degrading before reaching the ground water or adsorbed in the vadose zone during the drought period.

The detection of Metolachlor (Dual) in the epoxy resin well VC-9E seems reasonable since Dual was applied in the spring of 1989. The lack of a detection in the adjacent PVC well VC-9 suggests the possibility of adsorption to the PVC although a different sampling technique was employed for sampling the PVC monitoring well. There have been 3 detections in the ER well (VC-9E) since installation without a similar detection in the adjacent (VC-9) PVC well on the same date. Since installation of the epoxy resin wells in 1987, no pesticides have been detected in a PVC well without a similar detection in the adjacent ER well.

The detection of Dyfonate is questionable in some of the wells since the concentrations are very close to detection limits in the lab, and no history of use was reported. However, the detection on 8/14/89 in both JW-7E and the adjacent well (JW-7S) suggests validity. Dyfonate was detected in concentrations of 0.020 and 0.033 parts per billion in wells JW-7S and JW-7E respectively. Since each well was sampled using different equipment and methods (see explanation of sampling techniques for epoxy resin and PVC monitoring wells in section 2.2.3) cross contamination during sampling seems unlikely.

The detection of parathion at the JW-site on 9/15/89, with no history of use reported, also seems to be valid since it was detected in JW-7S and JW-7E in concentrations of 0.30 and 0.38 parts per billion respectively. Parathion was detected again at the JW-site on 12/5/89 and 12/12/89 in two different PVC wells. More data needs to be collected to determine the transport mechanism.

The fact that 85% of the pesticide detections were one time events seems to indicate that the majority of the pesticides detected drop below detection limits within four weeks. The concentrations of the pesticides could be decreasing due to dilution, degradation, or transport away from the sampling point.

2.4.5 Field Site Runoff

Organic nitrogen and total phosphorus found in surface water includes both dissolved and particulate material whereas analysis of ground water only includes the dissolved form. Higher concentrations are generally found in runoff than ground water because of this. The low concentrations of ammonia and nitrite - nitrogen in the ground water indicate most organic nitrogen is either converted to nitrates or adsorbed in the soil profile during infiltration. The higher average NO_3-N found in the ground water is further evidence of the conversion process. Nitrate - nitrogen is either carried off site in the runoff or infiltrates through the soil profile into the ground water since it is highly mobile. Higher average total phosphorus in the runoff compared to ground water also indicates an adsorption process within the vadose zone.

During a major precipitation event, NO_3-N on the ground surface is either carried away in the runoff or infiltrates into the ground water. After a period of time the concentrations are reduced to much lower levels as indicated by Figure 2-23. Suspended solids follow a similar pattern with relatively high initial concentrations gradually reducing to near zero at the end of the storm. The sediment load should normally be higher at the beginning of storm event because of soil particles previously loosened by natural processes or tillage practices, and impact by rain drops. The initial flows will therefore carry the most sediment. Phosphorus is often attached to soil particles, and Figure 2-23 indicates high initial total phosphorus concentration stabilizing in approximately 2 hours from the beginning of the storm event.

It appears that much of the NO_3-N will infiltrate quite rapidly into the ground water, especially in sand and gravel, as NO_3-N concentrations increased substantially in well BL-12S and BL-12D between June and August (see Figure 2-25) when three major precipitation events occurred (June 26, 2.0 inches, July 11, 6.3 inches, and July 18, 2.55 inches). Nitrogen was applied on the corn, which is upgradient of the BL-12 wells, in the first half of May. Soybeans, planted between the corn and the BL-12 wells, did not receive any fertilizer so it appears that runoff from the precipitation event could be a significant mechanism for infiltration of NO_3-N in this area of the BL-site.

There are four pesticides which have a history of use on the BL-site: MPCA, Basagran, Banvel, 2,4-D, and Lasso. Only one of these (Lasso) was detected in the runoff samples. The other three (Bladex, Dyfonate, and Treflan) do not have a history of use, however, Bladex was detected in a ground water sample collected on July 24, 1989. These results are similar to the pattern of pesticide detections in ground water where 65% of detections are not supported by a history of use. The presence of pesticides in field runoff water and the evidence of infiltration at well BL-12 indicates that overland flow may be a mechanism of pesticide transport. Because of topography, the BL-site is probably less affected by off-site surface water runoff than the other farmed field sites in the project.

2.4.4 Recommendations

Since the project is entering the final year of monitoring, major changes in monitoring strategy would not be effective from a statistics standpoint. Minor changes such as sampling schedules, precipitation data collection, and runoff monitoring will be employed.

Data analysis on a site by site basis will be completed. Emphasis will be placed on the VC-site because of the change in farming practices since 1987. A separate analysis could be done on above normal and below normal precipitation years at each site in an attempt to differentiate between BMPs and climatic conditions. Specific wells with a history of pesticide detections and/or high nitrate concentrations will also receive more intensive analysis.

Pesticide samples will be collected every two weeks in the months of May through August. This schedule should assist to gain a better understanding of pesticide fate and transport in the ground water system. Quality assurance should be stepped up with more duplicate and spiked pesticide samples sent through the lab. Sampling for pesticides will be done in deeper wells when detections are confirmed in a shallow adjacent well, and calculations indicate ground water movement downward, to increase understanding of transport times and degradation; some deeper wells have detected pesticides but sampling has been primarily in the shallow geozones.

To better evaluate NO_3-N movement into the ground water, samples will be collected from selected wells during and after major precipitation events. Samples could be collected at predetermined intervals either by bailing, pumping, or with the use of an automatic sampler. Information gathered from this investigation should provide some insight into transport times if analysis of the data collected indicate NO_3-N spikes when concentrations are plotted with time.

The source of many pesticide detections remain unclear with 65% having no history of use at the site. Sites which receive surface water runoff from adjacent fields will be investigated closely. An attempt will be made to collect land use data (specifically pesticide use) from land owners or operators farming near the field sites where potential from runoff exists. An investigation of chemicals used by counties for weed control will also be performed; many of the monitoring wells are located close to roads and may be subject to contamination from this source.

2.5 Site Data Summaries

Following are summaries of pertinent data for each site.

BL SITE Fact Sheet

Total area: 145 acres

Monitored: 40 acres

26	acres tilled
12	acres left in grass (hayed)
2	acres water (wetland)

General Geology: Two to three ft. of topsoil over 2 ft of clayey silt. Underlying the silt is usually 2 to 5 ft of poorly sorted sand and gravel which is absent in the northeast corner of the site and thickens to the west and south. Beneath the sand and gravel is 5 to 15 ft of brown weathered silty clay, little sand and gravel, glacial till overlying unweathered, gray glacial till. Varying thicknesses (0 ft to 30 ft) of a greenish brown transition till is found between the weathered and unweathered till. Under much of the site, at a depth of 40 to 45 ft is a sand and gravel unit 5 to 10 ft thick.

Precipitation (weighted mean calculation): Average 21.04" / year

1984	31.95"	52% over average
1985	25.37"	21% over average
1986	31.91"	52% over average
1987	21.64"	2% over average
1988	17.85"	15% under average
1989	20.84"	1% under average

Monitoring Wells: 19 wells at 8 locations (21 wells, 9 locations when W.R. & OLSS wells are used)

Average well depth: 17.2 ft. Range: 8 ft. to 43 ft.

Average Depth to water in 1989 (below ground surface): 10.62 ft.
Range: 3.86 ft. to 21.32 ft.

NO3-N concentrations: 451 samples average concentration 9.32 mg/l
(concentrations in mg/l)

YEAR	# SAMPLES	MEAN	MIN	MAX	MEDIAN
1984	21	8.38	0.08	30.95	5.28
1985	67	8.39	0.00	35.45	4.56
1986	83	11.51	0.00	35.00	9.85
1987	89	11.11	0.01	29.60	13.35
1988	83	8.30	0.02	27.50	3.17
1989	108	7.70	0.02	26.04	2.37

Geozones: Seven (7) geozone present on site.

WTLT15: Weathered till (brown color) or silty clay, reworked till with the screened interval of the well at a depth b.g.s. of less than or equal to 15 ft.

WTGT15: Weathered till or transition zone (greenish brown zone interpreted as a transition zone between the weathered and the unweathered till), with the screened interval at a depth b.g.s. of greater than 15 ft.

UT: Unweathered till (gray color).

SS-A: Alternating layers of thinly bedded fine sand and silt.

SGLT5LT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of less than or equal to 10 ft.
SGLT5GT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of greater than 10 ft.
SGGT15: Sand and gravel with greater than 15 ft of overlying soil material.

NO3-N Summary By Geozone: (Concentrations in mg/l)

	#WELLS	#SAMPLES	MEAN	MIN	MAX	MEDIAN
WTLT15	5	119	17.56	3.32	35.45	16.25
WTGT15	6	151	9.45	0.17	28.00	3.14
UT	2	63	0.56	0.16	4.02	0.45
SS-A	1	22	10.03	0.84	19.00	10.05
SGLT5LT10	1	17	14.86	5.95	27.75	15.60
SGLT5GT10	2	60	2.93	0.03	19.75	0.32
SGGT15	1	19	0.07	0.00	00.52	0.04

Pesticides: 175 Samples taken from 14 Wells with 19 Detections
Pesticides Detected (in ppb)

Pest.	# Samp.	# Wells	Ave Conc.	Median	Range	Year(s)
Lasso	5	3	0.27	0.160	0.030 - 0.820	1985,86,88
2,4-D	5	3	0.98	0.430	0.110 - 3.570	1986,88,89
Sencor	1	1	0.02	-	0.020	1985
Atrazine	1	1	5.40	-	5.400	1985
Lindane	2	2	0.05	0.050	0.050 - 0.050	1987
Tordon	1	1	0.50	-	0.500	1988
Banvel	2	2	0.09	0.095	0.095	1988
Bladex	1	1	0.19	-	0.190	1989
Dual	1	1	0.80	-	0.800	1989

Note: # Samp. = the number of samples which detected the pesticide name; # Wells = the number of wells in which the named pesticide was found, etc.

Land Use: Conservation Tillage (greater than 30% residue left on field at planting)

1983	Corn	60 lbs/ac 8-32-0 Spring	4.8 lbs./acre N
		3 qt./ac Lasso (alachlor)	
		1/4 pt./ac 2,4-D	
1984	Corn	70 lbs/ac 8-32-0 Spring	(total below)
		100 lbs/ac Anhyd. Summer	87.6 lbs./acre N
		7 lbs/ac Lasso (alachlor)	
		1/4 pt./ac 2,4-D	
		1/4 pt./ac Banvel (dicamba)	
1985	Oats	100 lbs/ac 30-12-0 Spring	30.0 lbs./acre N
		1/2 pt./ac MPCA	
	Corn just north of monitored area		

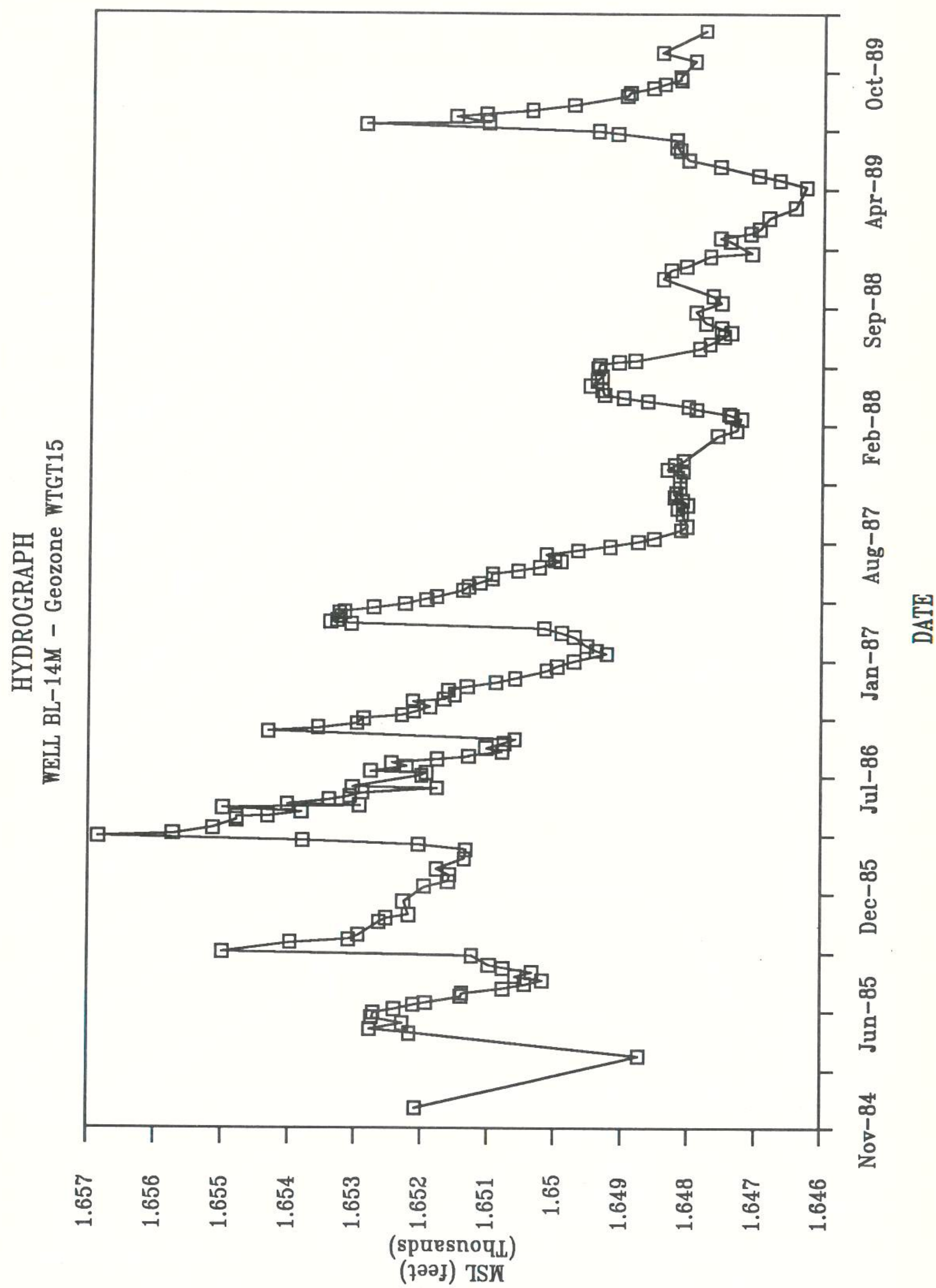
Fact Sheet continued

1986 Corn 75 lbs/ac 8-32-0 Spring (total below)
80 lbs/ac Anhyd. Summer 71.6 lbs./acre N
Corn south of well nest BL-15; Soybeans north of BL-15
3 qt./ac Lasso (alachlor)
1/4 pt./ac 2,4-D
1/4 pt./ac Banvel (dicamba)
Soybeans 0.0 lbs./acre N
3 qt./ac Lasso (alachlor)
1 qt./ac Basagran

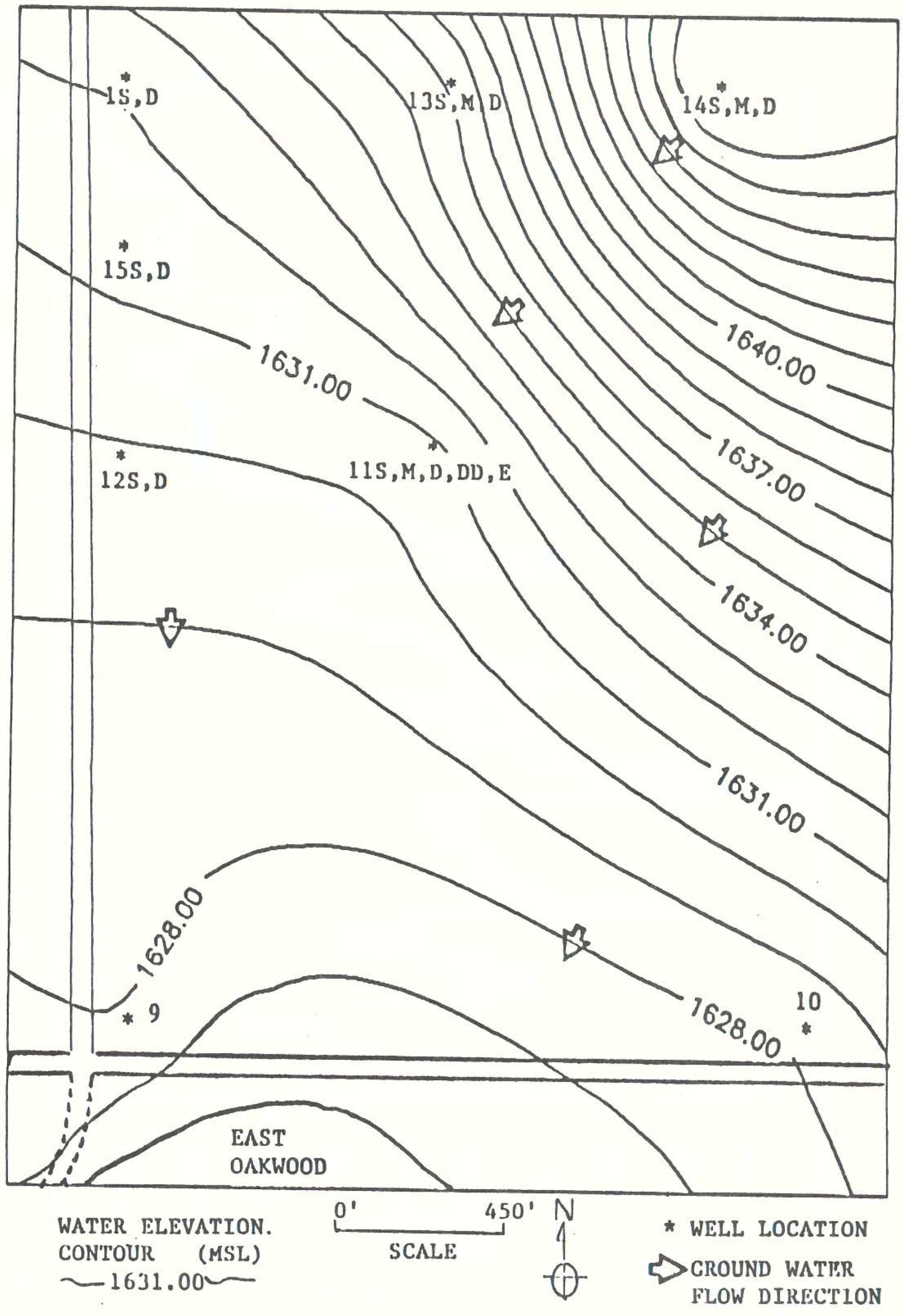
1987 Soybeans 0.0 lbs./acre N
Soybeans south of well nest BL-11; Corn north of BL-11
3 qt./ac Lasso (alachlor)
Corn 80 lbs/ac 8-32-0 Spring (total below)
80 lbs/ac Anhyd. Summer 72.0 lbs./acre N
3 qt./ac Lasso (alachlor)
1/4 pt./ac 2,4-D
1/4 pt./ac Banvel (dicamba)
Anhyd. = Anhydrous Ammonia

1988 Corn (drilled) 60 lb/ac 18-35-0 mid-May (total below)
80-90 lb/ac Anhyd. July 1 83.0 lbs./acre N
(side-dressed)
3/8 pint/ac Banvel mid-June
1/8 pint/ac 2,4-D mid-June
1 quart/ac LASSO banded at planting
South of E-W line Bl-15 well nest.
Soybeans 100 lb/ac 29-14-0 mid-April 29.0 lbs/acre N
M.C.P.A. 1/2 pint/ac Spring
North of E-W line BL-1 to BL-14 well nests. Chiseled (2) after harvest (fall).
Set Aside (remainder) Planted with rows of Sorghum. Sprayed mid-July with 1 pint/acre 2,4-D.

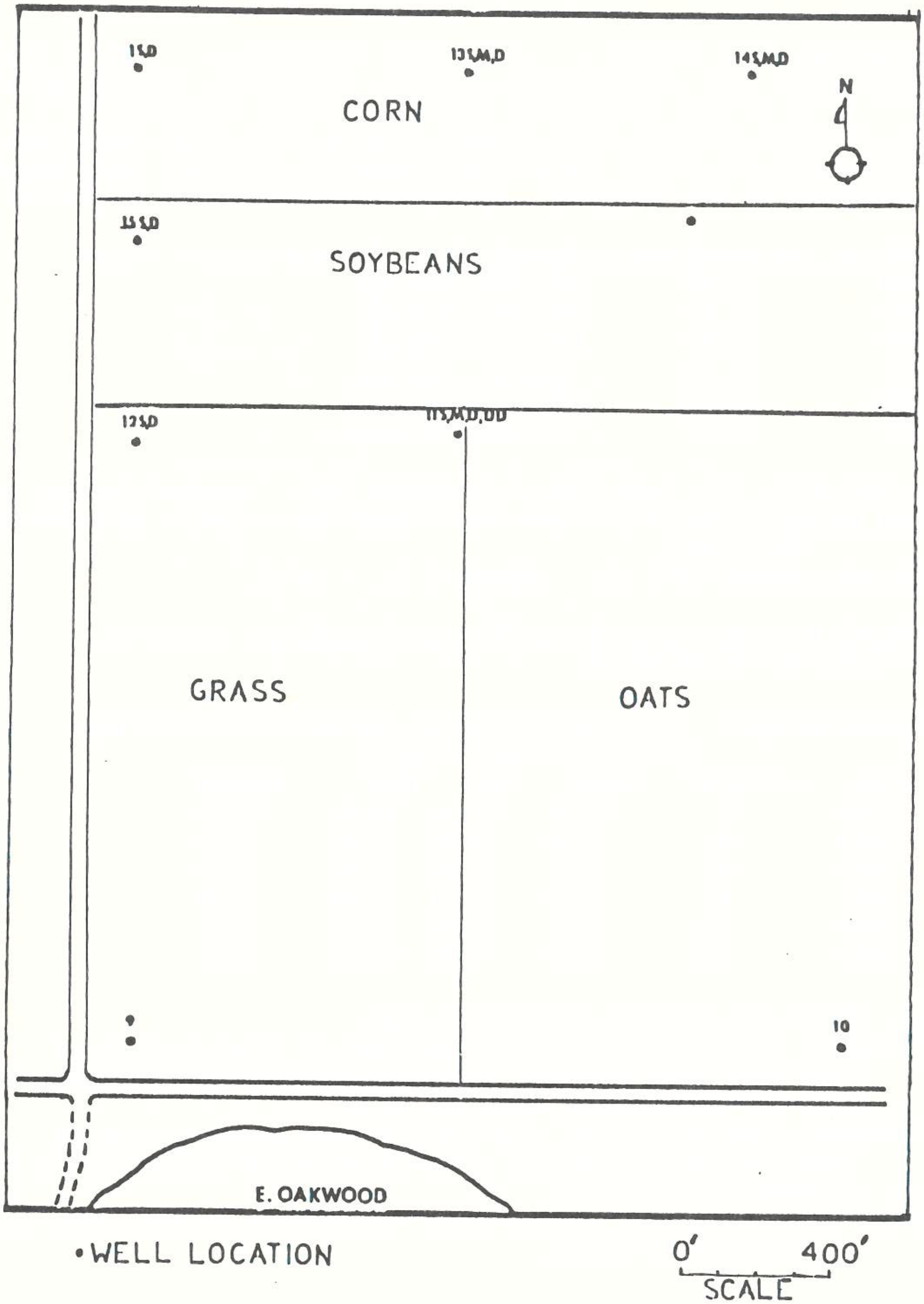
1989 Corn 120 lb/ac 46-0-0 early May 55.2 lbs/acre N
1 quart/ac LASSO banded at planting
1/4 pint/ac 2,4-D mid-June
1/2 pint/ac Banvel mid-June
Soybeans 1 quart/ac Treflan May 21
1/3 quart/ac Basagran
Oats 65 lb/ac 29-14-0 mid April 18.85 lbs/acre N
1/2 pint/ac MPCA June 1


HYDROGRAPH
WELL BL-14M - Geozone WTGT15
MSL (feet)
(Thousands)
1.657
1.656
1.655
1.654
1.653
1.652
1.651
1.65
1.649
1.648
1.647
1.646
Nov-84
Jun-85
Dec-85
Jul-86
Jan-87
Aug-87
Feb-88
Sep-88
Apr-89
Oct-89
DATE

BL-SITE WATER TABLE MAP 6-7-89



BL-SITE 1989 LAND USE MAP



JW SITE Fact Sheet

Total area: 75 acres 75.0 acres tilled
Because of high precipitation in 1985, 86, & 87
35 - 25 acres tilled
40 - 50 acres too wet

General Geology: Thin topsoil 1 - 2 ft thick overlying 2 - 3 ft of light brown silty clay (occasionally absent). Underlying the top sediments is 10 - 17 ft of sand and gravel, glacial outwash, underlain by 9 - 17 ft (24 ft thick on south west corner) of gray silty clay, underlain by a second sand and gravel outwash unit about 40 ft thick (only penetrated completely at one location). Beneath above sequences is glacial till.

Precipitation (weighted mean calculation): Average 21.20" / year

1984	31.64"	49% over average
1985	26.55"	25% over average
1986	30.81"	45% over average
1987	21.30"	approx. average
1988	17.59"	17% under average
1989	19.91"	6% under average

Thirteen new wells (13) installed in November 1987; sampled January 1988.
Monitoring Wells: 19 wells (13 new) at 9 locations (6 new); includes Epoxy Resin well.
Average well depth: 20.5 ft. Range: 8 ft. - 50 ft.

Average Depth to water in 1989 (below ground surface): 10.20 ft.
Range: 2.64 ft. to 40.06 ft.

NO3-N Concentrations: 340 samples average concentration 2.53 mg/l
(Concentrations in mg/l)

YEAR	# SAMPLE	MEAN	MIN	MAX	MEDIAN
1984	1	10.24	10.24	10.24	10.24
1985	20	4.84	0.03	11.85	4.13
1986	40	3.11	0.00	12.85	1.10
1987	48	2.57	0.01	9.88	0.89
1988	112	2.24	0.00	11.59	0.45
1989	119	2.13	0.00	14.35	0.94

Geozones: Three (3) geozones present on site.

SGLT5LT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of less than or equal to 10 ft.

SGLT5GT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of greater than 10 ft.

SG-UA: Sand and gravel located under an aquitard as the lower unit of a two aquifer system.

NO3-N Summary By Geozone: (Concentrations in mg/l)

	#WELLS	#SAMPLES	MEAN	MIN	MAX	MEDIAN
SGLT5LT10	9	222	3.77	0.00	14.35	1.96
SGLT5GT10	4	64	0.10	0.00	1.50	0.03
SG-UA	5	54	0.27	0.00	6.05	0.08

Pesticides: 165 Samples taken from 4 Wells with 10 Detections

Pesticides Detected (ppb)

Pest.	# Samp.	# Wells	Ave Conc.	Median	Range	Year(s)
Lasso	3	3	0.79	0.37	0.05 - 1.38	1985,88
2,4-D	2	2	0.70	0.70	0.45 - 0.95	1988,89
Treflan	1	1	0.38	-	0.38	1988
Banvel	1	1	1.55	-	1.55	1989
Sencor	1	1	0.06	-	0.06	1989
Dyfonate	2	2	0.027	0.027	0.02 - 0.033	1989

Note: # Samp. = the number of samples which detected the pesticide name;
Wells = the number of well in which the named pesticide was found, etc.

Land Use: Conventional Tillage (moldboard plow, less than 30% residue left on field at planting)

1983 Corn 100 lbs/ac 20-20-10 Summer — 20.0 lbs./acre N
7 lbs/ace Ramrod (propachlor)

1984 Oats (north 2/3 of field) — (total below)
Barley (south 1/3 of field) — 0.00 lbs./acre N
1/2 pt./ac 2,4-D

1985 West half too wet; only east half tilled
6 tons/ac manure Winter — (total below)
Corn 0 lbs/ac — 12.0 lbs./acre N *
1/2 pt./ac 2,4-D
1/8 pt./ac Banvel (dicamba)

1986 West half too wet; only east half tilled
6 tons/ac manure Winter — (total below)
Corn 0 lbs/ac — 12.0 lbs./acre N *
1/2 pt./ac 2,4-D
1/8 pt./ac Banvel (dicamba)

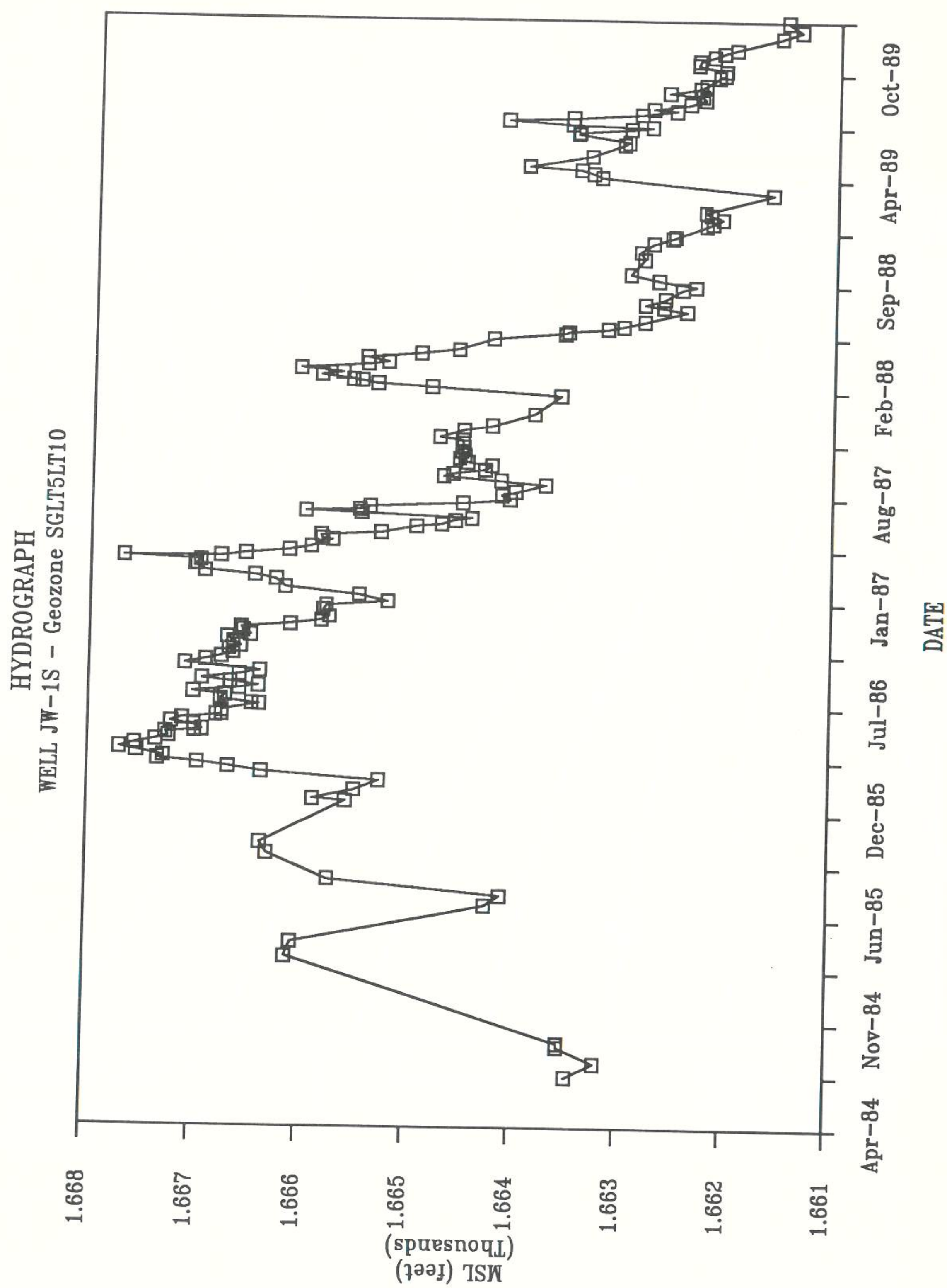
1987 West half too wet; only east half tilled
6 tons/ac manure Winter — (total below)
Corn 0 lbs/ac — 12.0 lbs./acre N *

1988 Corn ? ton/ac cow manure Winter — ??.? lbs./acre N*
(2/3 of west 1/3 of field site)
Oats ? ton/ac cow manure Winter — ??.? lbs./acre N*
(east 2/3 of east half; 1/3 south 1/3; 1/3 of west 1/3;
moldboard plow portion too wet past years)
1/4 pint/acre 2,4-D
1/8 pint/acre Banvel

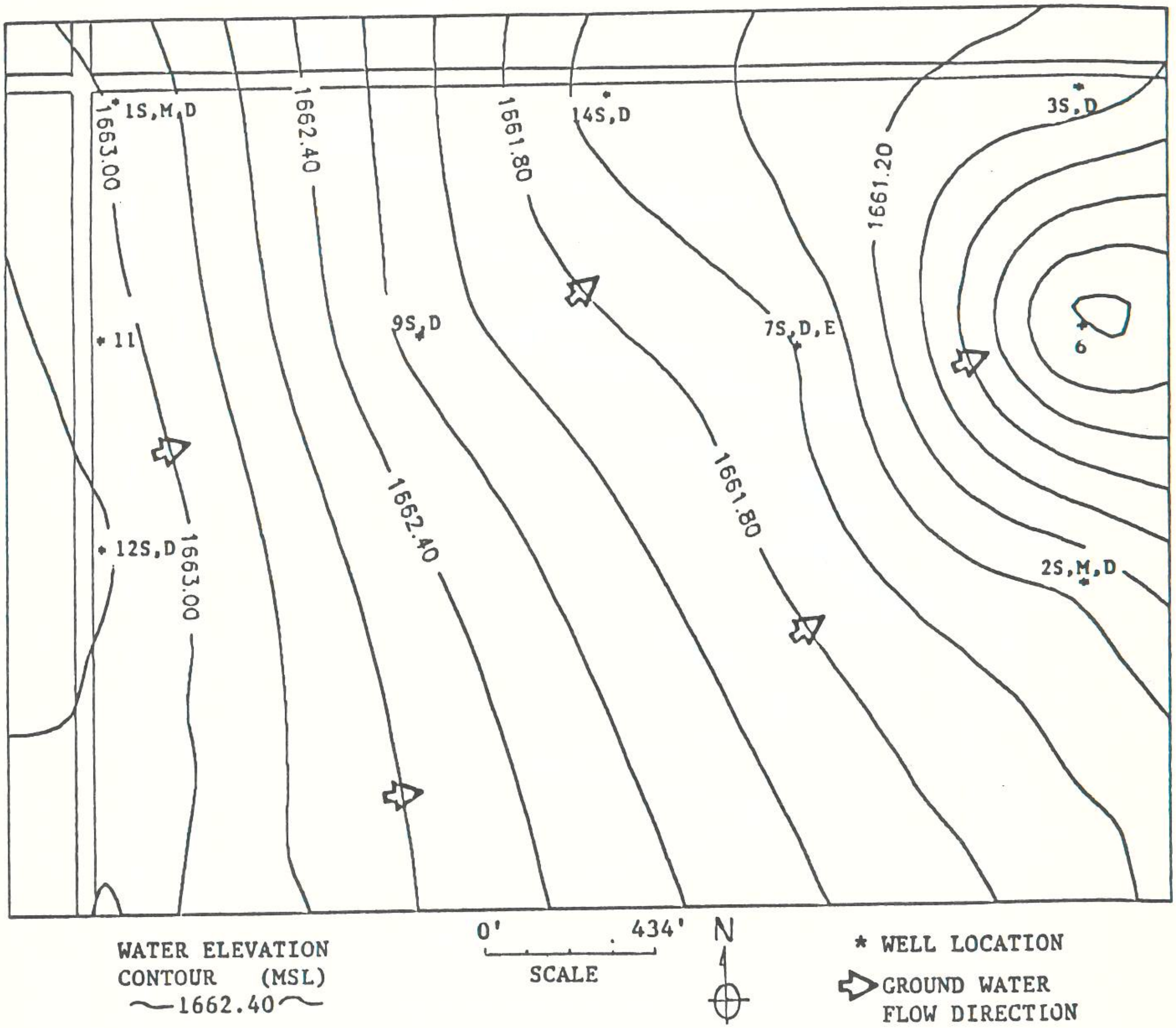
1989 Corn/Sorghum ? ton/ac cow manure Winter
Roundup March 23 (only on part of field)
Corn ? ton/ac cow manure Winter
2.5 pint/acre Atrazine May 23
1/4 pint/acre 2,4-D May 23
1/8 pint/acre Banvel May 23

1989	Barley	No chemicals applied
	Oats	No chemicals applied

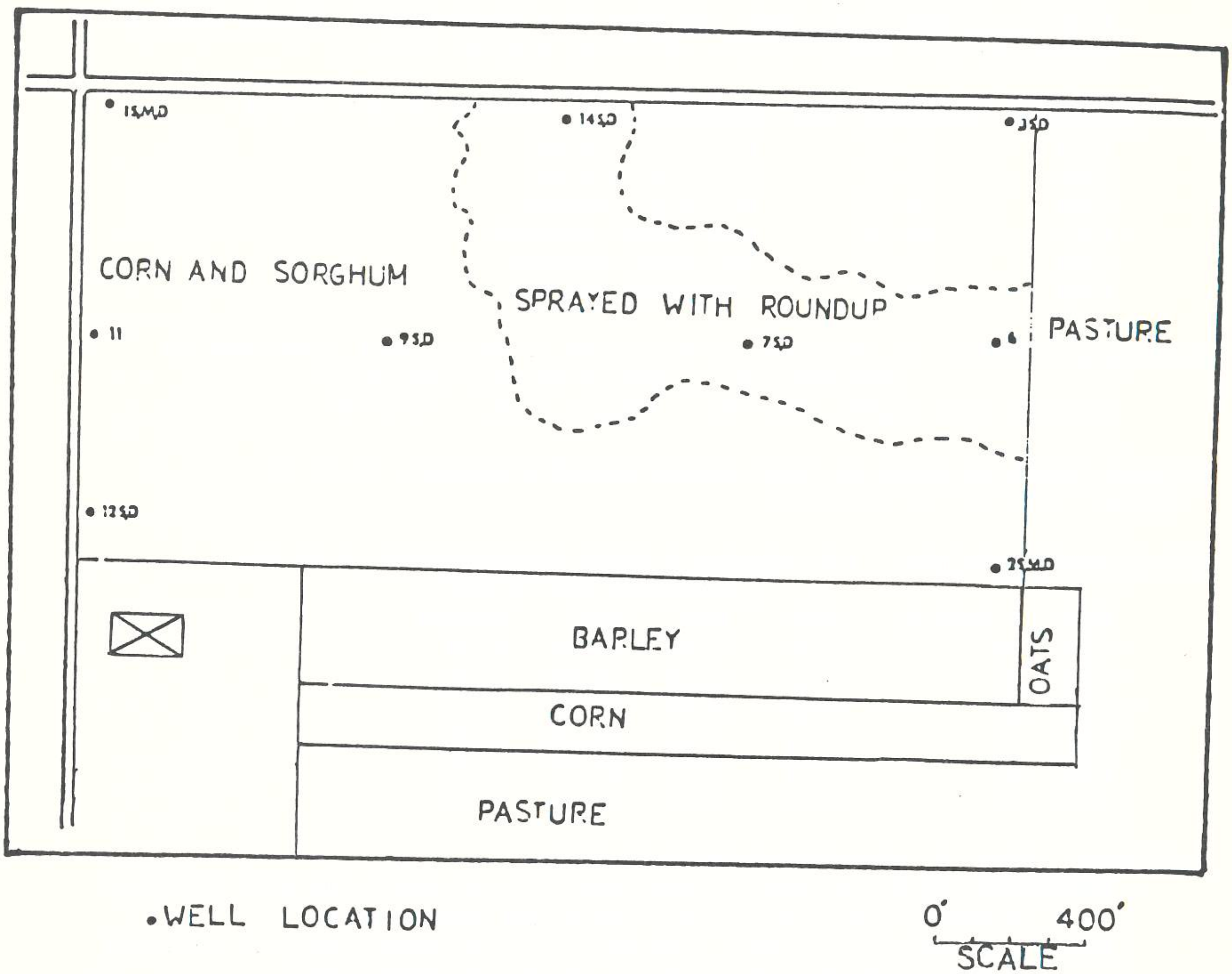
* Nitrogen content of manure determined from SCS handbook of average values, South Dakota Technical Guide, notice SD-142 pp. 633-634.


HYDROGRAPH
WELL JW-1S - Geozone SGLT5LT10
MSL (feet)
(Thousands)
1.668
1.667
1.666
1.665
1.664
1.663
1.662
1.661
Apr-84
Nov-84
Jun-85
Dec-85
Jul-86
Jan-87
Aug-87
Feb-88
Sep-88
Apr-89
Oct-89
DATE

JW-SITE WATER TABLE MAP 6-7-89



JW-SITE 1989 LAND USE MAP



LK SITE Fact Sheet

Total area: 43 acres
18.5 acres tilled
8 acres grass (often too wet to plant)
16.5 acres alfalfa
Fall of 1987 land leased to Hamlin County Equipment; moldboard plowed wet acreage and alfalfa acreage. All 43 acres planted in 1988.

General Geology: One to three feet of silty clay topsoil over 6.5 - 15 ft. of sand and gravel outwash (upper aquifer). Upper sand and gravel aquifer separated by 3.5 - 10 ft. of silty clay (absent in some places) from lower sand and gravel aquifer 12 - 20 ft thick. Whole site is underlain at a depth of 42 - 46 ft by glacial till.

Precipitation (weighted mean calculation): Average 21.26" / year

1984	31.49"	47% over average
1985	28.03"	31% over average
1986	36.00"	68% over average
1987	20.98"	1.5% under average
1988	17.49"	18% under average
1989	18.72	12% under average

Monitoring Wells: 21 wells at 13 locations (3 wells in 2 locations on adjacent site, used in LK site analysis)
Average well depth: 18.7 ft. Range: 7 ft. - 29.5 ft.

Average Depth to water in 1989 (below ground surface): 13.20 ft.
Range: 1.70 ft. to 26.96 ft.

NO3-N concentrations: 445 samples average concentration 6.07 mg/l
(Concentrations in mg/l)

YEAR	# SAMPLES	MEAN	MIN	MAX	MEDIAN
1984	19	5.20	0.00	15.00	4.34
1985	69	9.12	0.00	35.50	6.65
1986	85	7.60	0.00	27.00	6.15
1987	92	6.04	0.00	23.38	5.58
1988	82	4.79	0.00	24.77	1.95
1989	98	3.85	0.00	19.38	1.88

Geozones: Four (4) geozones present at site.

SGLT5LT10: Sand and gravel less than 5 ft of overlying soil material screened less than 10 ft. below the water table

SGLT5GT10: Sand and gravel less than 5 ft of overlying soil material screened greater than 10 ft. below the water table

SS-A: Alternating layers of thinly bedded sand and silt

SG-UA: Sand and gravel located under an aquitard as the lower unit of a two aquifer system

NO3-N Summary By Geozone: (Concentrations in mg/l)

	# WELLS	# SAMPLES	MEAN	MIN	MAX	MEDIAN
SGLT5LT10	6	163	8.92	0.11	35.50	7.75
SGLT5GT10	6	109	6.75	0.00	27.00	5.84
SS-A	2	73	5.76	0.02	26.20	0.20
SG-UA	4	100	0.90	0.00	16.63	0.06

Pesticides: 145 Samples taken from 5 Wells with 15 Detections

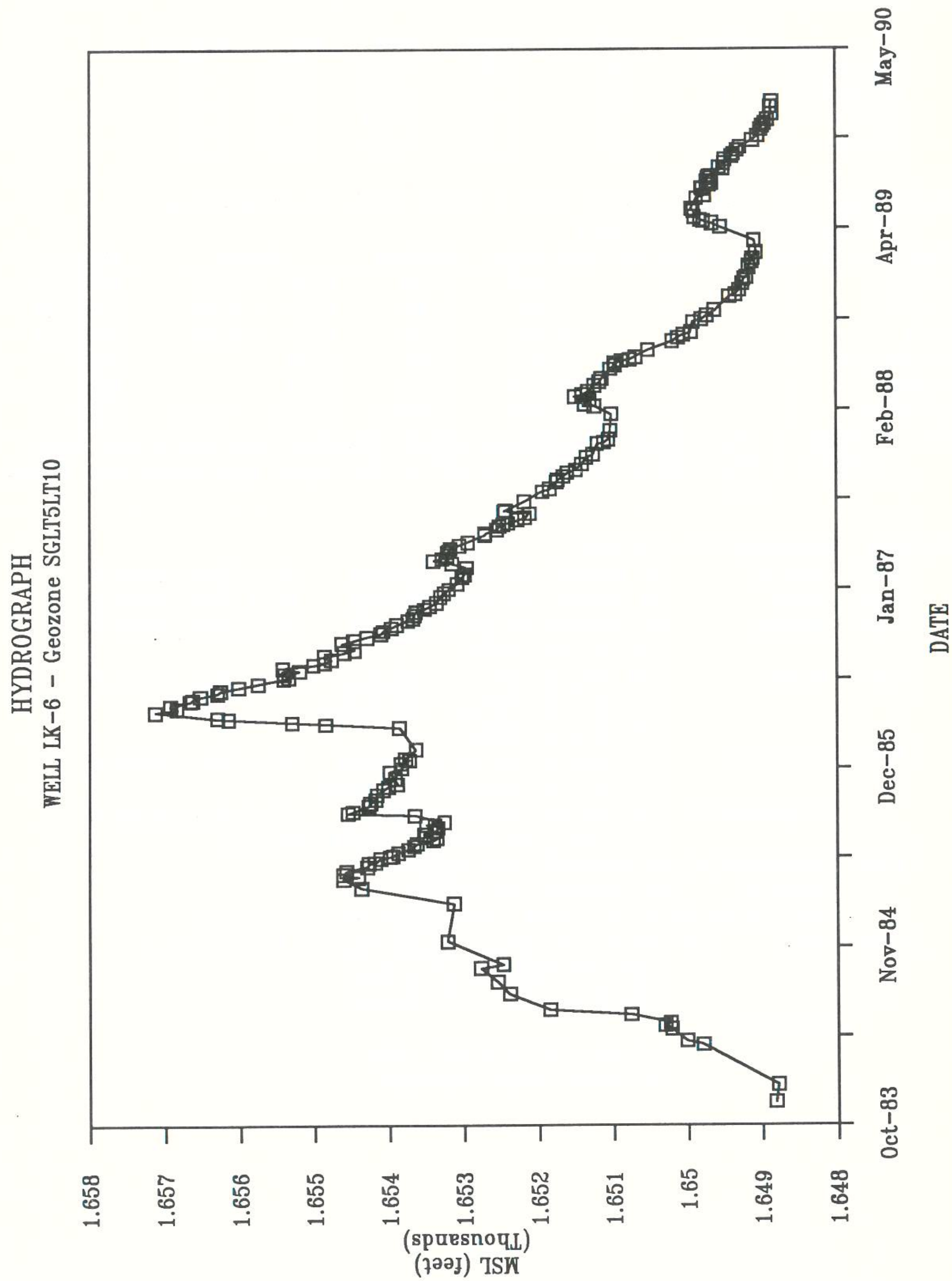
Pesticides Detected (ppb)

Pest.	# Samp.	# Wells	Ave Conc.	Median	Range	Year(s)
Lasso	4	3	0.248	0.10	0.05 - 0.74	1988,89
2,4-D	3	3	1.48	1.40	0.13 - 2.90	1986,88
Dual	3	3	0.493	0.35	0.17 - 0.96	1988,89
Parathion	2	2	0.040	0.04	0.04	1989
Bladex	1	1	0.740	0.74	0.74	1989
Dyfonate	2	2	0.036	0.0365	0.036 - 0.037	1989

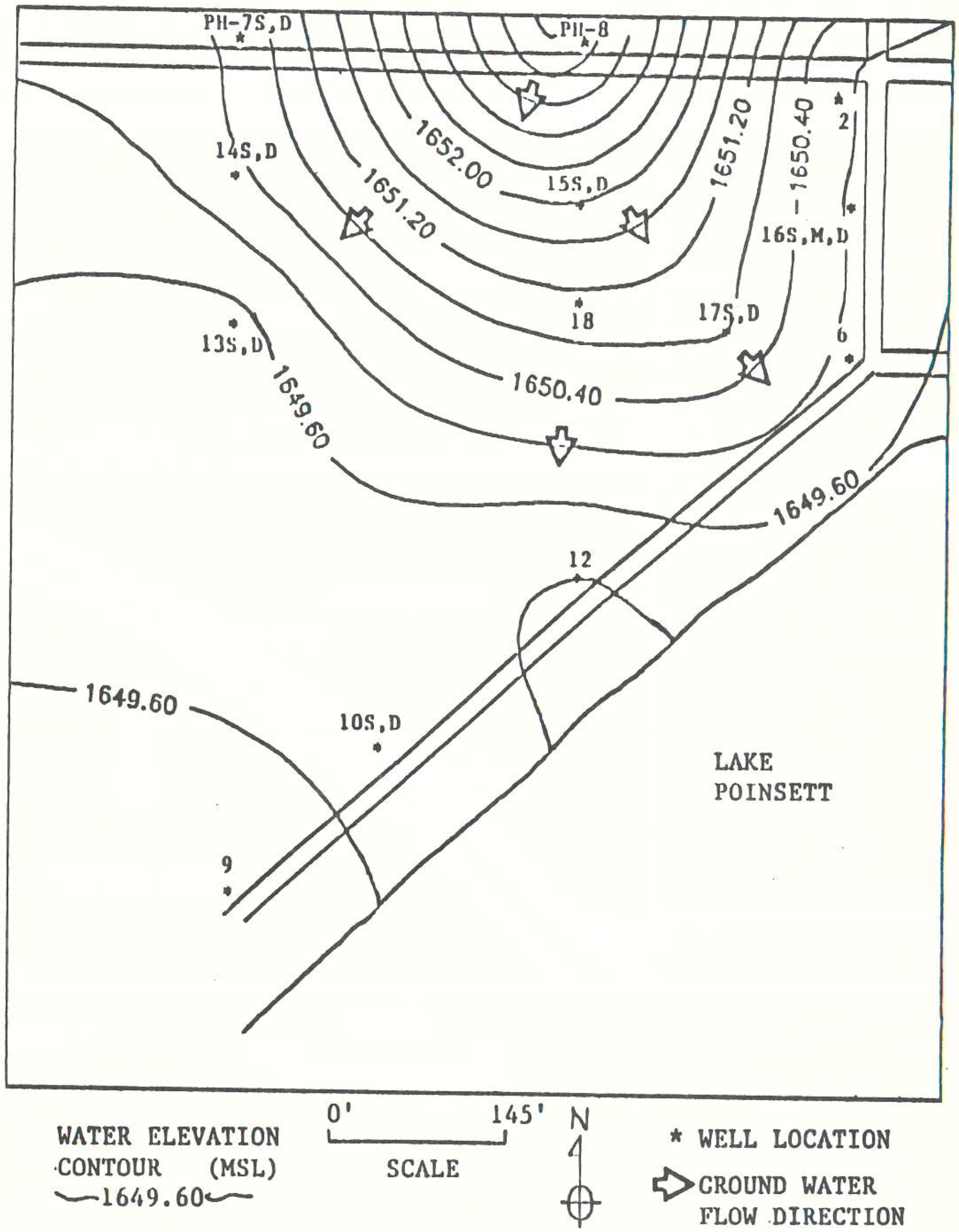
Note: # Samp. = the number of samples which detected the pesticide name; # Wells = the number of well in which the named pesticide was found, etc.

Land Use: Conservation Tillage

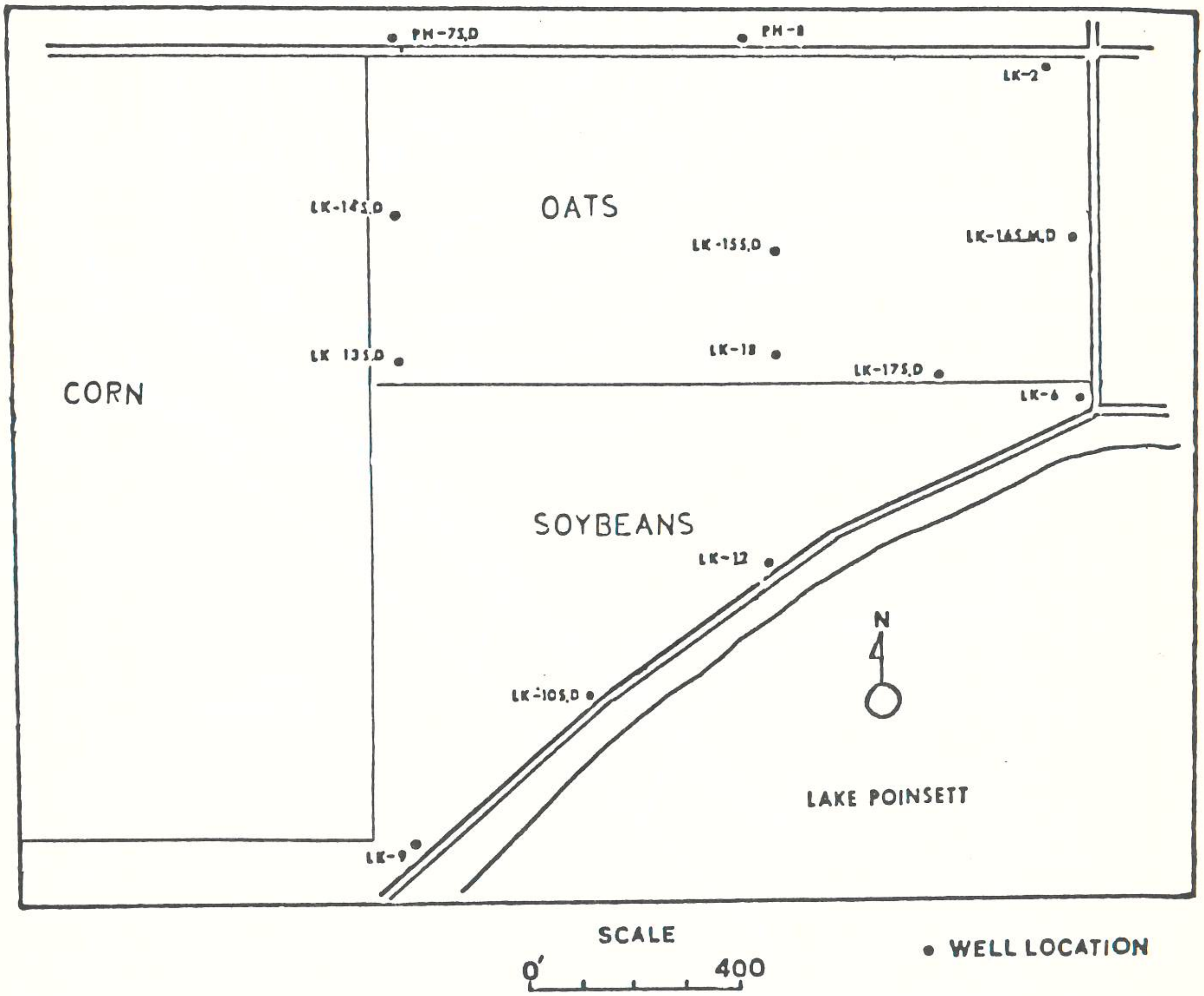
1983	Barley	80 lbs/ac 24-24-0 Spring	19.2 lbs./acre N
1984	Oats	80 lbs/ac 28-14-0 Spring	22.4 lbs./acre N
1985	Barley	80 lbs/ac 24-24-0 Spring	19.2 lbs./acre N
		1/8 pt./ac MCPA	
1986	Wheat	100 lbs/ac 28-14-0 Spring	28.0 lbs./acre N
		1/3 pt./ac Banvel (dicamba)	
		1/8 pt./ac MPCA	
1987	Wheat(n)	100 lbs/ac 20-20-0 Spring	20.0 lbs./acre N
		1/3 pt./ac Banvel (dicamba)	
	Soybeans(s)	100 lbs/ac 18-46-0 Spring	18.0 lbs./acre N
		1 1/2 pt./ac Treflan (trifluralin)	
1988	Soybeans(n)	No records available	
	Corn(s)	No records available	
	(n):north half of field (s):south half of field		
1989	Oats	100 lbs/ac 28-28-0 early May	28 lbs./acre N
		2 pt./ac MCPA early May	
		Plowed mid to late August	
	Soybeans		
		1.5 pt./ac Treflan May	
		Chiseled after harvest	


HYDROGRAPH
WELL LK-6 - Geozone SGLT5LT10
MSL (feet)
(Thousands)
1.658
1.657
1.656
1.655
1.654
1.653
1.652
1.651
1.65
1.649
1.648
Oct-83
Nov-84
Dec-85
Jan-87
Feb-88
Apr-89
May-90
DATE

LK-SITE WATER TABLE MAP 6-7-89



LK-SITE 1989 LAND USE MAP



OP SITE (State Park) Fact Sheet

Total area: 40 acres
Monitored: 17 acres — 0 acres tilled; 17 acres grass (left natural)

General Geology: Topsoil 1 - 3 ft thick overlying 2 - 4 ft of clayey silt overlying 1 - 12 ft of sand and gravel. The silt and/or sand layers are absent in certain areas of the site. Underlying the upper sediments (which may or may not be present) is 10 - 15 ft of brown silty clay little sand and gravel, weathered glacial till which is underlain by 15 - 30 ft of gray unweathered glacial till. Beneath the entire site at a depth of 30 - 45 ft is sand and gravel.

Precipitation (weighted mean calculation): Average 21.04" / year

1984	31.95"	52% over average
1985	25.37"	21% over average
1986	31.91"	52% over average
1987	22.49"	7% over average
1988	18.64"	11% under average
1989	21.72"	3% over average

Monitoring Wells: 17 wells at 7 locations (includes Epoxy Resin well)
Average well depth: 23.9 ft. Range: 11 ft. - 48.5 ft.

Average Depth to water in 1989 (below ground surface): 12.71 ft.
Range: 6.53 ft. to 24.30 ft.

NO3-N concentrations: 396 samples average concentration 2.12 mg/l
(Concentrations in mg/l)

YEAR	# SAMPLES	MEAN	MIN	MAX	MEDIAN
1984	4	1.38	0.04	3.65	0.91
1985	40	1.27	0.00	14.23	0.06
1986	80	1.99	0.00	16.72	0.17
1987	104	2.13	0.00	14.88	0.22
1988	78	2.20	0.00	15.48	0.18
1989	90	2.59	0.00	16.24	0.10

Geozones: Seven (7) geozones present on site.

SGLT5LT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of less than or equal to 10 ft.

SG5-15LT10: Sand and gravel with greater than 5 ft and less than or equal to 15 ft of overlying soil material, with the screened interval at a depth w.t. of less than or equal to 10 ft.

SG5-15GT10: Sand and gravel with greater than 5 ft and less than or equal to 15 ft of overlying soil material, with the screened interval at a depth w.t. of greater than 10 ft.

SGGT15: Sand and gravel with greater than 15 ft of overlying soil material.

SG-UA: Sand and gravel located under an aquitard as the lower unit of a two aquifer system.

OP SITE (State Park)   Fact Sheet Continued

WTLT15: Weathered till (brown color) or silty clay, reworked till with the screened interval of the well at a depth b.g.s. of less than or equal to 15 ft.

UT: Unweathered till (gray color).

NO3-N Summary By Geozone: (Concentrations in mg/l)

	#WELLS	#SAMPLES	MEAN	MIN	MAX	MEDIAN
SGLT5LT10	2	49	1.95	0.62	3.20	2.06
SG5-15LT10	1	49	0.05	0.00	0.20	0.04
SG5-15GT10	2	37	0.61	0.00	2.68	0.10
SGGT15	3	63	0.40	0.00	4.07	0.03
SG-UA	2	66	0.03	0.00	0.16	0.02
WTLT15	3	74	9.14	0.01	16.72	12.15
UT	3	57	0.29	0.02	0.88	0.25

Pesticides: 168 Samples taken from 4 Wells with 15 Detections

Pesticides Detected (ppb)

Pest.	# Samp.	# Wells	Ave Conc.	Median	Range	Year
Lasso	8	4	0.813	0.485	0.07 - 3.09	1985-89
2,4-D	1	1	0.41	-	0.41	1986
Treflan	1	1	0.047	-	0.047	1988
Tordon	1	1	0.42	0.42	0.42	1989
Dual	1	1	0.18	0.18	0.18	1989
Dyfonate	3	3	0.033	0.03	0.02 - 0.05	1989

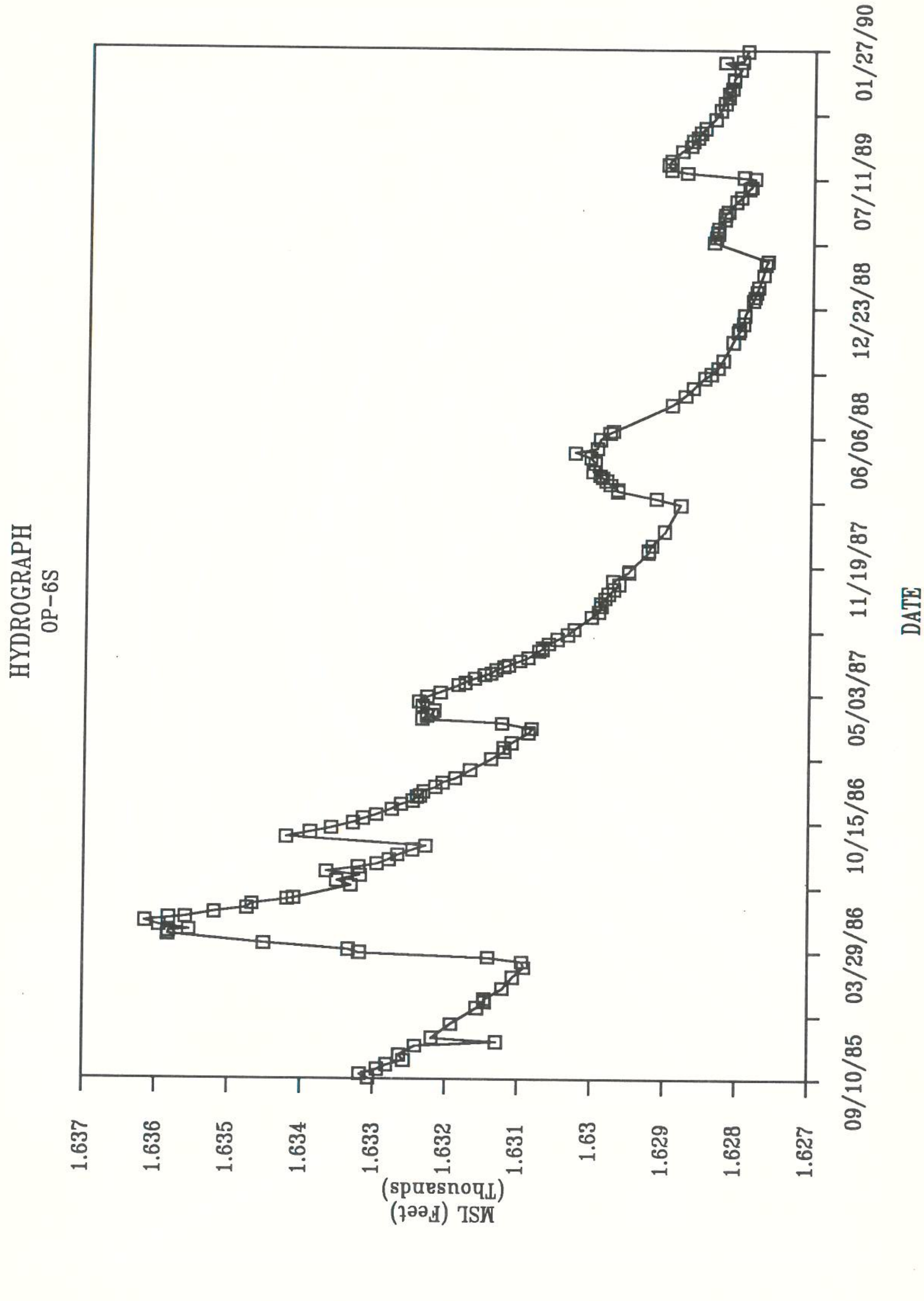
Note: # Samp. = the number of samples which detected the pesticide name;
Wells = the number of well in which the named pesticide was found, etc.

Land Use: Unfarmed

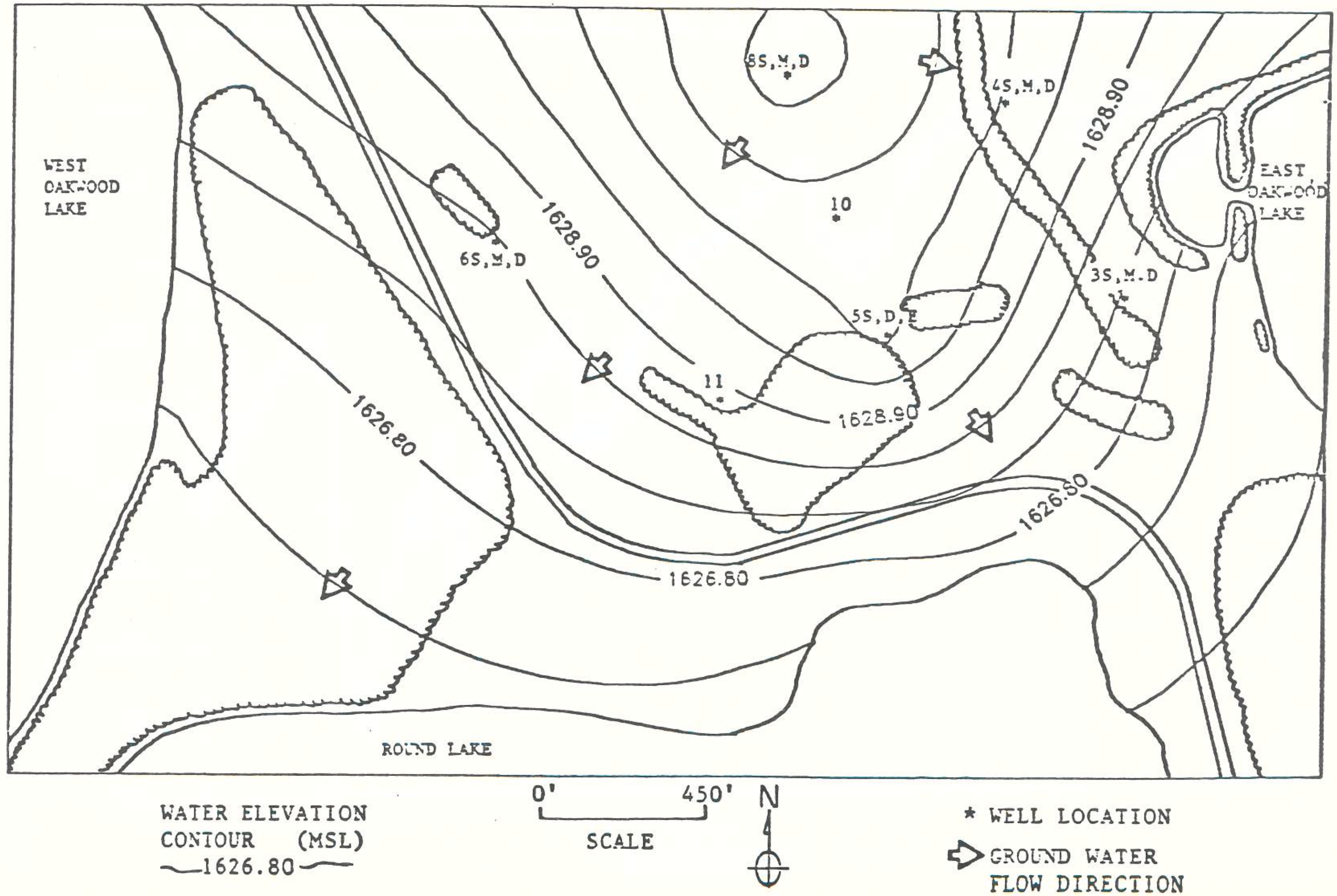
Farmed until 1974 (or 1976).

Planted with brome and other native grasses and left alone since then.

Pesticide Application - sprayed with Tordon 22-K and 2,4-D by both the Divisions of Wildlife/Parks and Recreation to control leafy spurge and canada thistle. Parks and Recreation Division applied 6 quarts/acre 2,4-D in late May/early June, 1988, and 3 pints/acre Tordon and 4 quarts/acre 2,4-D in October, 1988. Parks and Recreation Division applied mixture of 2 gallons of Tordon and 2 gallons of 2,4-D ester in 110 gallons of water to area SW of hedge row between wells OP-6 and OP-11 on the NW side of the road in the first week of June, 1989 and again on September 14, 1989. On September 12, 1989 a mixture of 2 gallons of 2,4-D and 1 pint of Tordon in 100 gallons of water was applied on the Fish and Game land near Round Lake.


HYDROGRAPH
OP-6S
MSL (Feet)
(Thousands)
1.637
1.636
1.635
1.634
1.633
1.632
1.631
1.63
1.629
1.628
1.627
09/10/85
03/29/86
10/15/86
05/03/87
11/19/87
06/06/88
12/23/88
07/11/89
01/27/90
DATE

OP-SITE WATER TABLE MAP 6-7-89



PH SITE (monitored portion) Fact Sheet

Total area: 160 acres
Monitored: 80 acres 80 acres tilled

General Geology: Thin silty clay topsoil over 8 - 13 ft. of sand and gravel glacial outwash (upper aquifer). Upper sand and gravel aquifer separated by 6 - 10 ft. of silty clay (absent in some places) from lower sand and gravel aquifer 12 - 20 ft thick. Whole site is underlain at a depth of 42 - 46 ft by glacial till.

Precipitation (weighted mean calculation): Average 21.26" / year

1984	31.49"	47% over average
1985	28.03"	31% over average
1986	36.00"	68% over average
1987	20.98"	<1% under average
1988	17.49"	18% under average
1989	18.72"	12% under average

Monitoring Wells: 15 wells at 10 locations (2 wells not sampled due to point source contamination)
Average well depth: 22.8 ft. Range: 11.5 ft. to 32 ft.

Average Depth to water in 1989 (below ground surface): 16.93 ft.
Range: 11.75 ft. to 22.45 ft.

NO3-N Concentrations: 324 samples average concentration 11.80 mg/l

YEAR	# SAMPLES	MEAN	MIN	MAX	MEDIAN
1984	15	8.42	0.90	37.25	5.12
1985	38	9.71	0.10	33.80	8.13
1986	65	10.36	0.00	45.70	8.78
1987	71	15.66	0.04	41.75	15.05
1988	71	13.73	0.01	34.91	15.38
1989	64	8.86	0.01	27.50	4.61

Geozones: Five (5) geozones present on site.
SGLT5LT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of less than or equal to 10 ft.
SGLT5GT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of greater than 10 ft.
SS-A: Alternating layers of thinly bedded fine sand and silt.
SC: Silty clay aquitard located between an upper and a lower aquifer system.
SG-UA: Sand and gravel located under an aquitard as the lower unit of a two aquifer system.

PH SITE (monitored portion) Fact Sheet

Total area: 160 acres
Monitored: 80 acres 80 acres tilled

General Geology: Thin silty clay topsoil over 8 - 13 ft. of sand and gravel glacial outwash (upper aquifer). Upper sand and gravel aquifer separated by 6 - 10 ft. of silty clay (absent in some places) from lower sand and gravel aquifer 12 - 20 ft thick. Whole site is underlain at a depth of 42 - 46 ft by glacial till.

Precipitation (weighted mean calculation): Average 21.26" / year

1984	31.49"	47% over average
1985	28.03"	31% over average
1986	36.00"	68% over average
1987	20.98"	<1% under average
1988	17.49"	18% under average
1989	18.72"	12% under average

Monitoring Wells: 15 wells at 10 locations (2 wells not sampled due to point source contamination)
Average well depth: 22.8 ft. Range: 11.5 ft. to 32 ft.

Average Depth to water in 1989 (below ground surface): 16.93 ft.
Range: 11.75 ft. to 22.45 ft.

NO3-N Concentrations: 324 samples average concentration 11.80 mg/l

YEAR	# SAMPLES	MEAN	MIN	MAX	MEDIAN
1984	15	8.42	0.90	37.25	5.12
1985	38	9.71	0.10	33.80	8.13
1986	65	10.36	0.00	45.70	8.78
1987	71	15.66	0.04	41.75	15.05
1988	71	13.73	0.01	34.91	15.38
1989	64	8.86	0.01	27.50	4.61

Geozones: Five (5) geozones present on site.

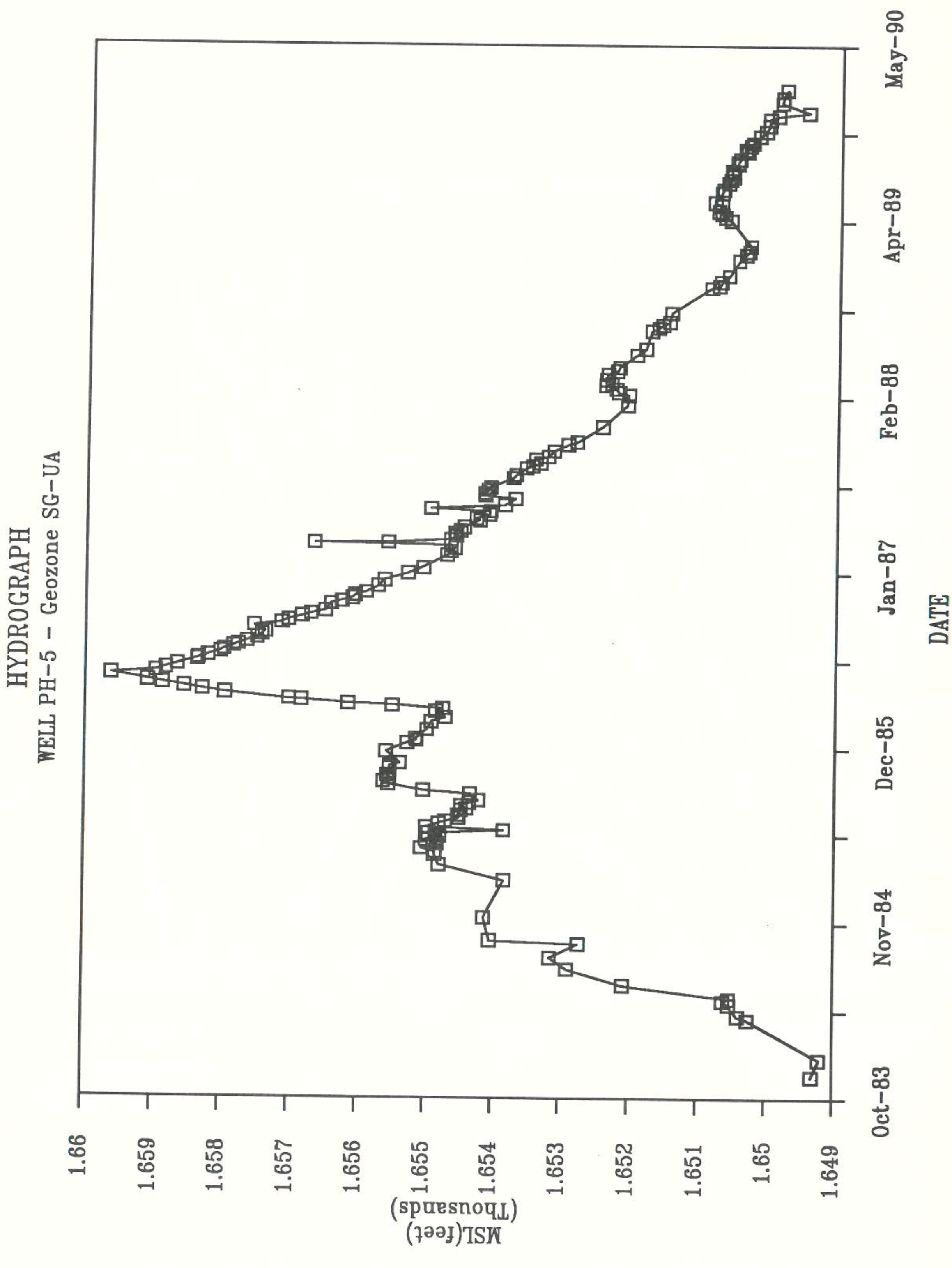
SGLT5LT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of less than or equal to 10 ft.

SGLT5GT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of greater than 10 ft.

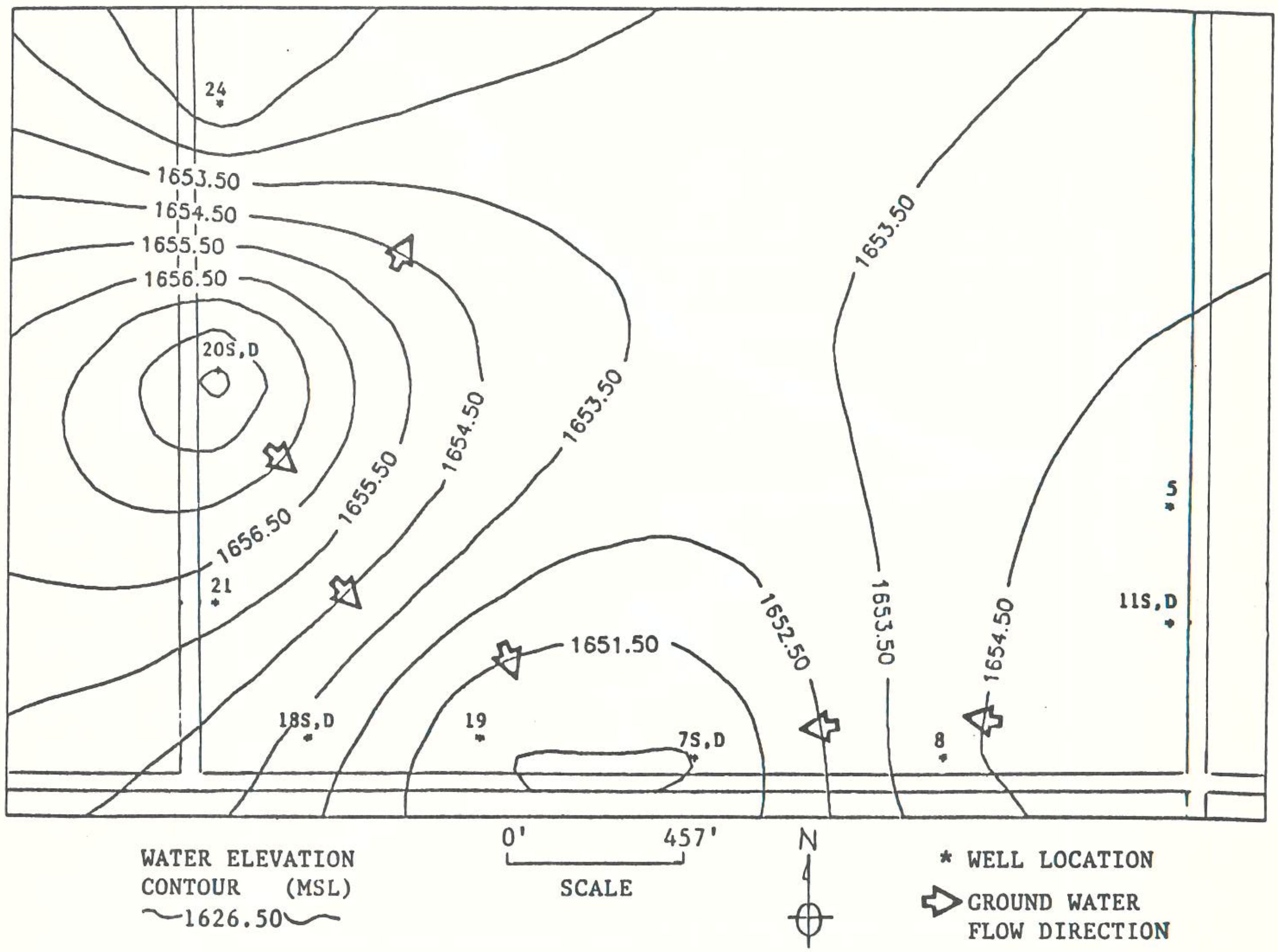
SS-A: Alternating layers of thinly bedded fine sand and silt.

SC: Silty clay aquitard located between an upper and a lower aquifer system.

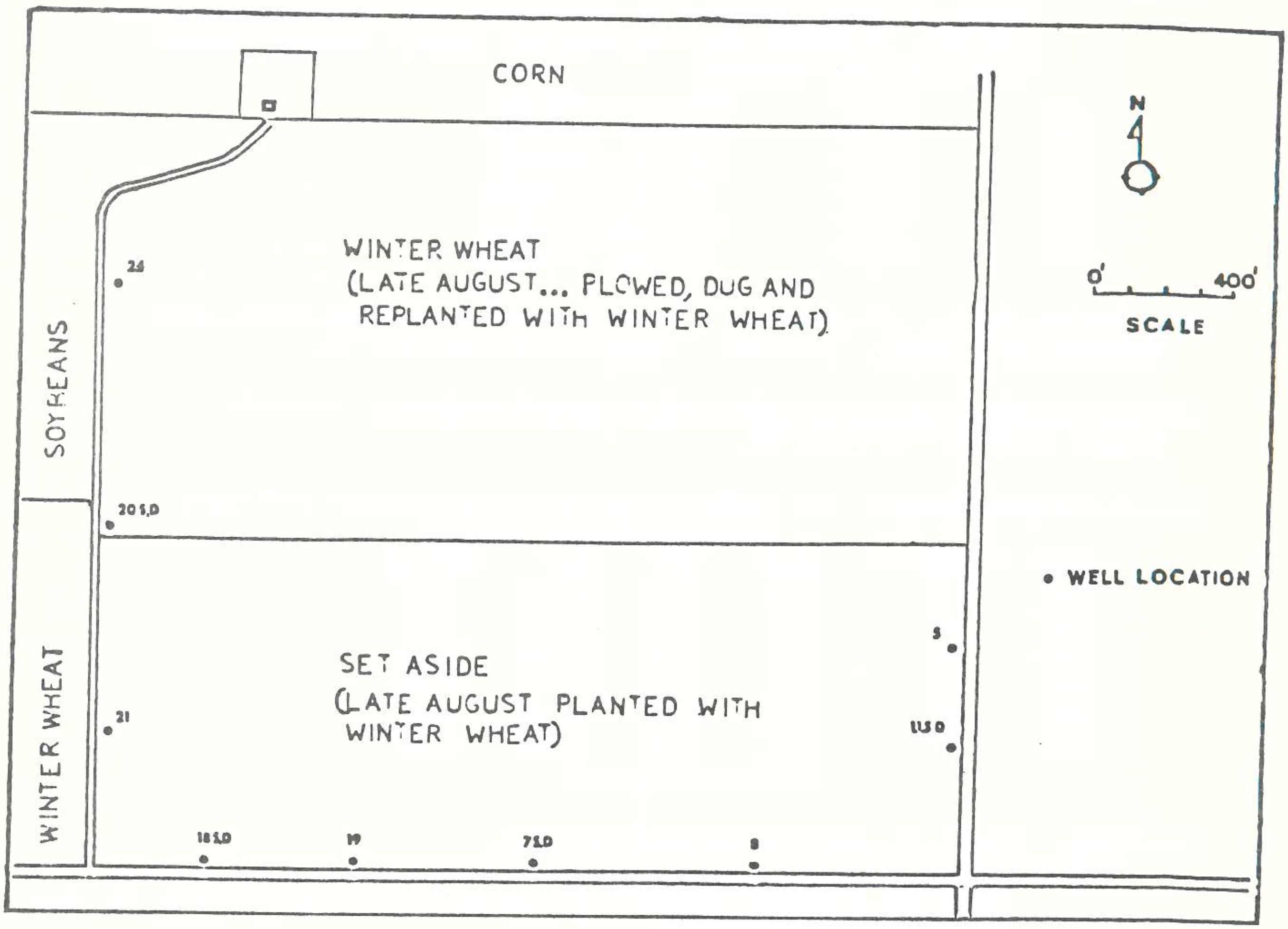
SG-UA: Sand and gravel located under an aquitard as the lower unit of a two aquifer system.


HYDROGRAPH
WELL PH-5 - Geozone SG-UA
MSL(feet)
(Thousands)
1.66
1.659
1.658
1.657
1.656
1.655
1.654
1.653
1.652
1.651
1.65
1.649
Oct-83
Nov-84
Dec-85
Jan-87
Feb-88
Apr-89
May-90
DATE

PH-SITE WATER TABLE MAP 6-7-89



PH-SITE 1989 LAND USE MAP



RS SITE Fact Sheet

Total area: 47 acres

43.2 acres tilled
1.5 acres grass waterway until 1988
2.3 acres observed to be too wet to farm

General Geology: Two to three feet of silty clay topsoil over 8 - 10 ft of brown turning to gray very silty clay trace sand and gravel (ablation till?; loess?) reworked till. Underlying these sediments is 4 - 14 ft of brown weathered silty clay little sand and gravel glacial till, underlain by gray unweathered glacial till to depths exceeding 75 ft.

Precipitation: Average 21.41" / year @ Castlewood Station (insufficient record at other stations to calculate long term weighted mean average) (weighted mean calculation [yearly totals] versus Castlewood for average)

1984	36.04"	68% over average
1985	27.15"	27% over average
1986	32.52"	52% over average
1987	21.84"	1% under average
1988	18.07"	16% under average
1989	20.98"	2% under average

Monitoring Wells: 17 wells at 7 locations (include Epoxy Resin well)
Average well depth: 15.7 ft. Range: 8 ft. to 29.5 ft.

Average Depth to water in 1989 (below ground surface): 9.91 ft.
Range: 4.80 ft. to 14.27 ft.

NO3-N Concentrations: 346 samples average concentration 3.74 mg/l
(Concentrations in mg/l)

YEAR	# SAMPLES	MEAN	MIN	MAX	MEDIAN
1984	13	2.99	0.04	16.55	1.65
1985	54	4.02	0.03	13.15	4.11
1986	74	5.48	0.10	13.88	5.51
1987	73	4.67	0.14	10.77	5.36
1988	69	2.83	0.00	6.66	3.39
1989	63	1.52	0.08	5.42	1.24

(no RSBY samples)

Geozones: Three (3) geozones present on site.

WTLT15: Weathered till (brown color) or silty clay, reworked till with the screened interval of the well at a depth b.g.s. of less than or equal to 15 ft.

WTGT15: Weathered till or transition zone (greenish brown zone interpreted as a transition zone between the weathered and the unweathered till), with the screened interval at a depth b.g.s. of greater than 15 ft.

UT: Unweathered till (gray color).

NO3-N Summary By Geozone: (Concentrations in mg/l)

	#WELLS	#SAMPLES	MEAN	MIN	MAX	MEDIAN
WTLT15	9	196	5.37	0.00	16.55	5.06
WTGT15	5	120	1.90	0.09	10.63	0.94
UT	2	30	0.42	0.03	1.55	0.30

Pesticides: 197 Samples taken from 11 Wells with 44 Detections

Pesticides Detected (in ppb)

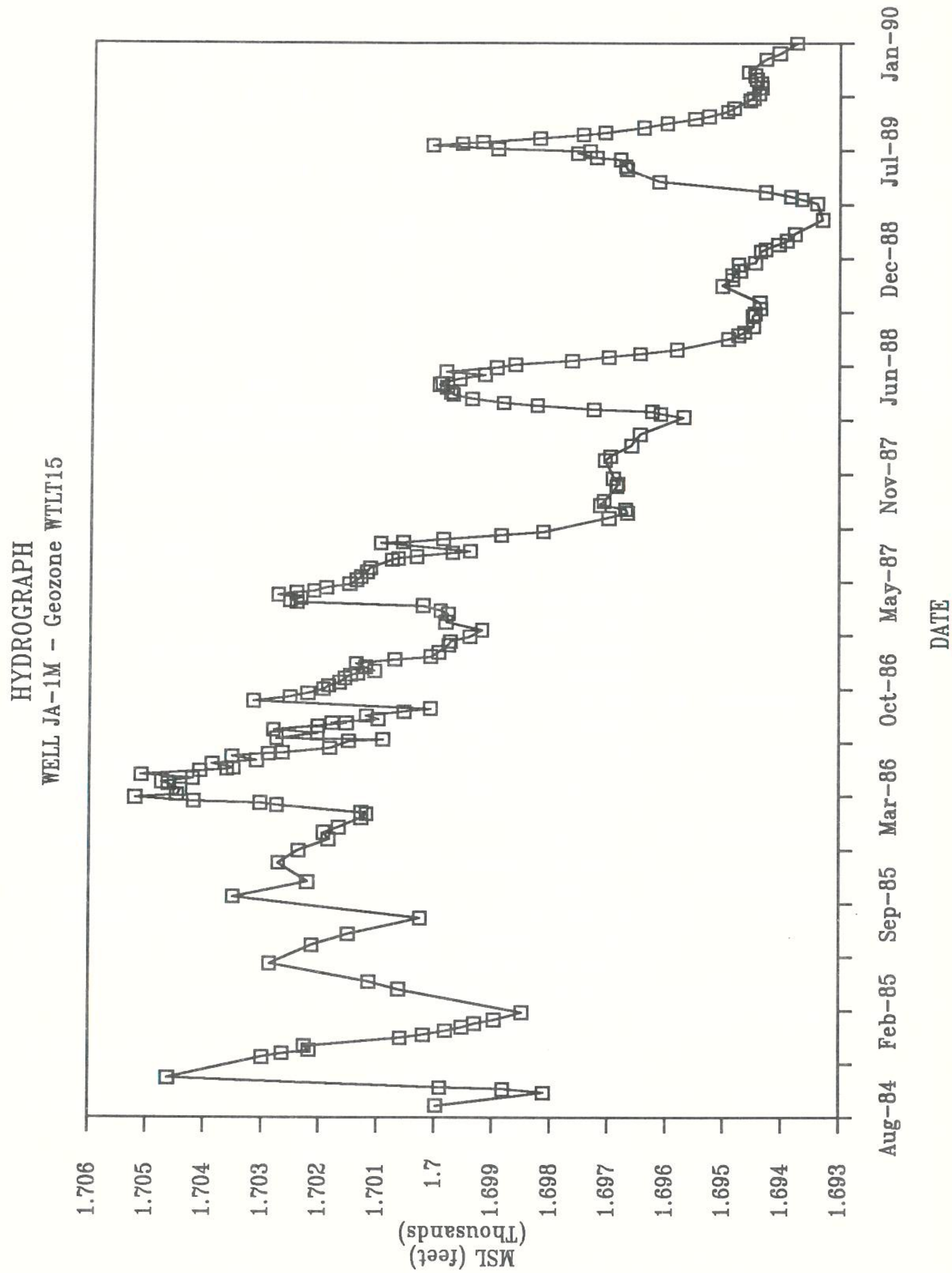
Pest.	# Samp.	# Wells	Ave Conc.	Median	Range	Year(s)
Lasso	19	3	1.21	0.28	0.036 - 3.44	1985,86,87,88
2,4-D	6	4	2.27	0.61	0.23 - 6.70	1984,86,88
Sencor	15	2	0.09	0.09	0.03 - 0.14	1986,88,89
Dual	1	1	0.10	0.10	0.10	1989
Atrazine	2	2	0.185	0.205	0.11 - 0.26	1989
Parathion	1	1	0.68	0.68	0.68	1989

Note: # Samp. = the number of samples which detected the pesticide name;
Wells = the number of well in which the named pesticide was found, etc.

Land Use:

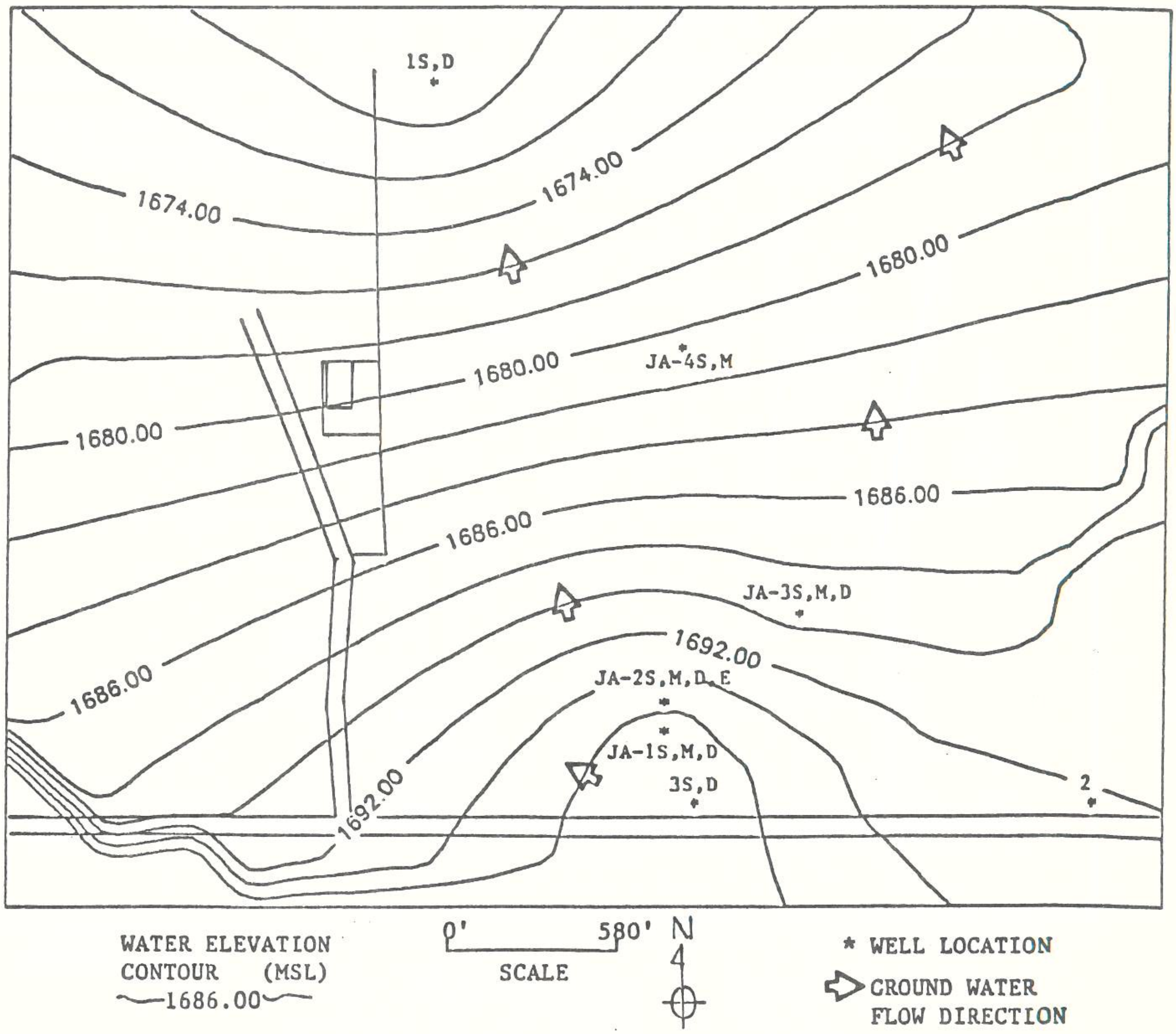
1983	Corn	100 lbs/ac Anhyd. Summer	(total below)
		100 lbs/ac 18-46-0 Summer	100.0 lbs./acre N
		8 lbs/ac Lasso (alachlor)	
		? pt./ac 2,4-D	
		Commercial application aerial spraying	
		Furadan (carbofuran)	
1984	Soybeans	80 lbs/ac 18-46-0 Summer	14.4 lbs./acre N
		8 lbs/ac Lasso (alachlor)	
1985	Corn	80 lbs/ac 18-46-0 Summer	(total below)
		100 lbs/ac Anhyd. Summer	96.4 lbs./acre N
		8 lbs/ac Lasso (alachlor)	
1986	Soybeans (s)	0 lbs/ac	(total below)
		8 lbs/ac Lasso (alachlor)	
	Buckwheat (n)	0 lbs/ac	0 lbs./acre N
1987	Corn	80 lbs/ac 18-46-0 Summer	(total below)
		100 lbs/ac Anhyd. Summer	96.4 lbs./acre N
		8 lbs/ac Lasso (alachlor)	
	(n):north 2/3 of field (s):south 1/3 of field		
1988	Wheat (planted in mid-April)		
		75 lbs/ac 46-0-0	34.5 lbs./acre N
		0.5 pint/acre M.C.P.A.	
1989	Soybeans	50 lbs/ac 18-46-0	9.0 lbs./acre N
		? lbs/ac Basagran Spring	
		8 lbs/ac Lasso Spring	

*Anhyd. = Anhydrous Ammonia

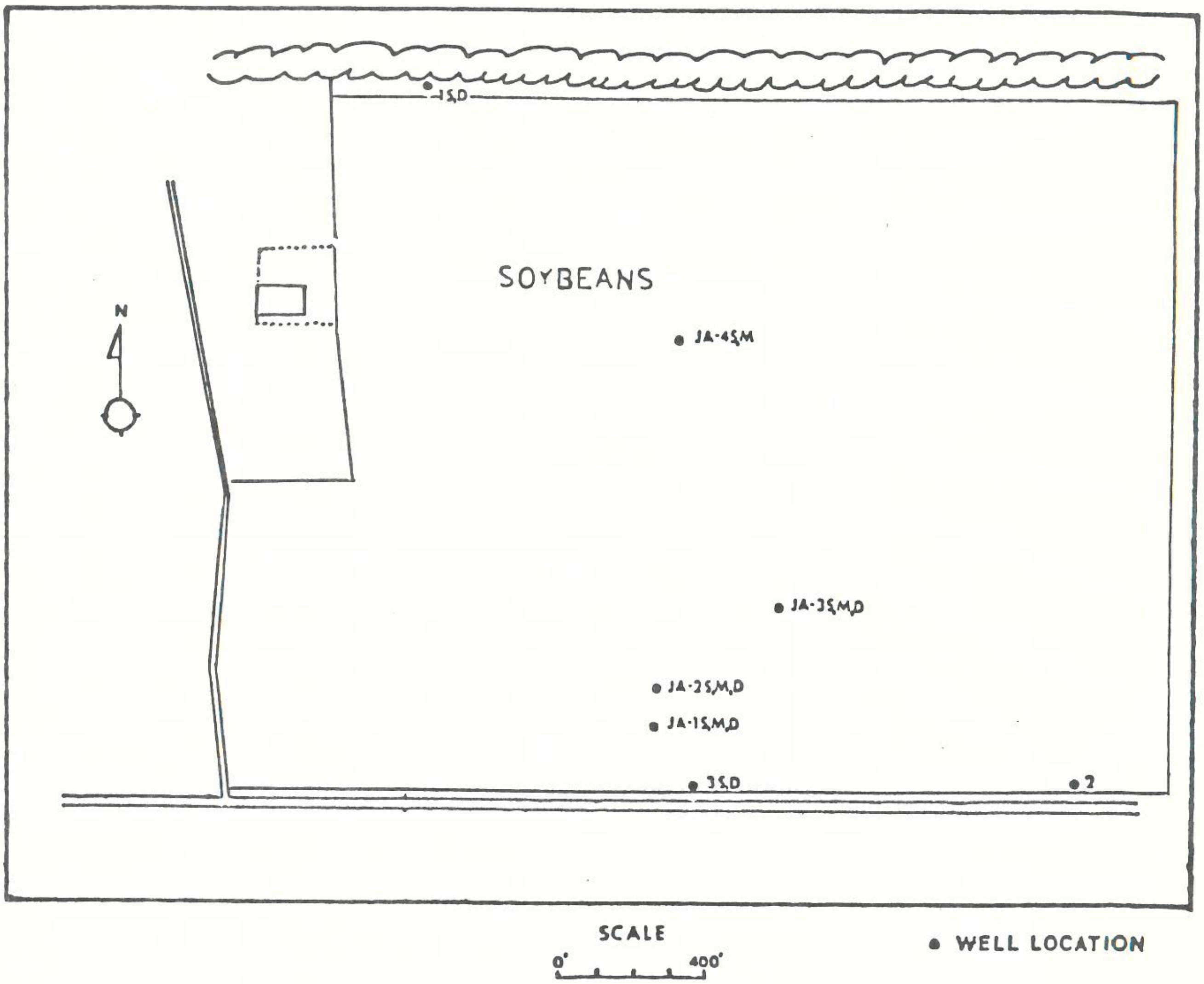


HYDROGRAPH
WELL JA-1M - Geozone WTLT15
MSL (feet)
(Thousands)
1.706
1.705
1.704
1.703
1.702
1.701
1.7
1.699
1.698
1.697
1.696
1.695
1.694
1.693
Aug-84
Feb-85
Sep-85
Mar-86
Oct-86
May-87
Nov-87
Jun-88
Dec-88
Jul-89
Jan-90
DATE

RS—SITE WATER TABLE MAP 6—29—89



RS-SITE 1989 LAND USE MAP



VC SITE (monitored portion) <u>Fact Sheet</u>

<u>Total area:</u> 140 acres
<u>Monitored:</u> 10 acres 10 acres tilled

<u>General Geology (of the 140 acres):</u> Topsoil 1 - 2 ft thick overlying 0.5 - 2.5 ft of very silty clay. Under the upper sediments: in the north and northeast sides of the site is 6 - 50 ft of sand and gravel underlain by glacial till (silty clay with some to little sand and gravel); in the north west corner is glacial till to depths exceeding 50 ft; on the west side is 17 ft of glacial till underlain by sand and gravel.

<u>Precipitation (weighted mean calculation):</u> Average 21.26" / year

1984	31.49"	47% over average
1985	28.03"	31% over average
1986	36.00"	68% over average
1987	20.57"	<1% under average
1988	17.06"	20% under average
1989	18.51"	13% under average

Five (5) new wells installed in November 1987; sampled Jan. 1988.
<u>Monitoring Wells:</u> 8 wells (5 new) at 5 locations (2 new)
<u>Average well depth:</u> 19.5 ft. <u>Range:</u> 12 ft. - 29.5 ft.

<u>Average Depth to water in 1989 (below ground surface):</u> 13.57 ft.
<u>Range:</u> 9.37 ft. to 20.50 ft.

<u>NO3-N Concentrations:</u> 161 samples average concentration 3.14 mg/l

YEAR	# SAMPLES	MEAN	MIN	MAX	MEDIAN
1984	12	0.87	0.00	3.76	0.07
1985	17	3.62	0.01	11.73	4.80
1986	18	3.83	0.00	11.65	2.22
1987	21	4.04	0.00	12.38	2.32
1988	43	4.49	0.01	15.65	2.38
1989	50	1.73	0.00	11.28	1.03

Geozones: (3)

SGLT5LT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of less than or equal to 10 ft.

SGLT5GT10: Sand and gravel with less than or equal to 5 ft of overlying soil material, with the screened interval at a depth w.t. of greater than 10 ft.

SGGT15: Sand and gravel with greater than 15 ft of overlying soil material.

VC SITE (monitored portion) <u>Fact Sheet Continued</u>

<u>NO3-N Summary By Geozone:</u> (Concentrations in mg/l)

	#WELLS	#SAMPLES	MEAN	MIN	MAX	MEDIAN
SGLT5LT10	3	105	4.78	0.00	15.65	2.78
SGLT5GT10	3	31	0.03	0.00	0.16	0.02
SGGT15	1	24	0.04	0.00	0.20	0.02

<u>Pesticides:</u> 76 Samples taken from 2 Wells with 2 Detections

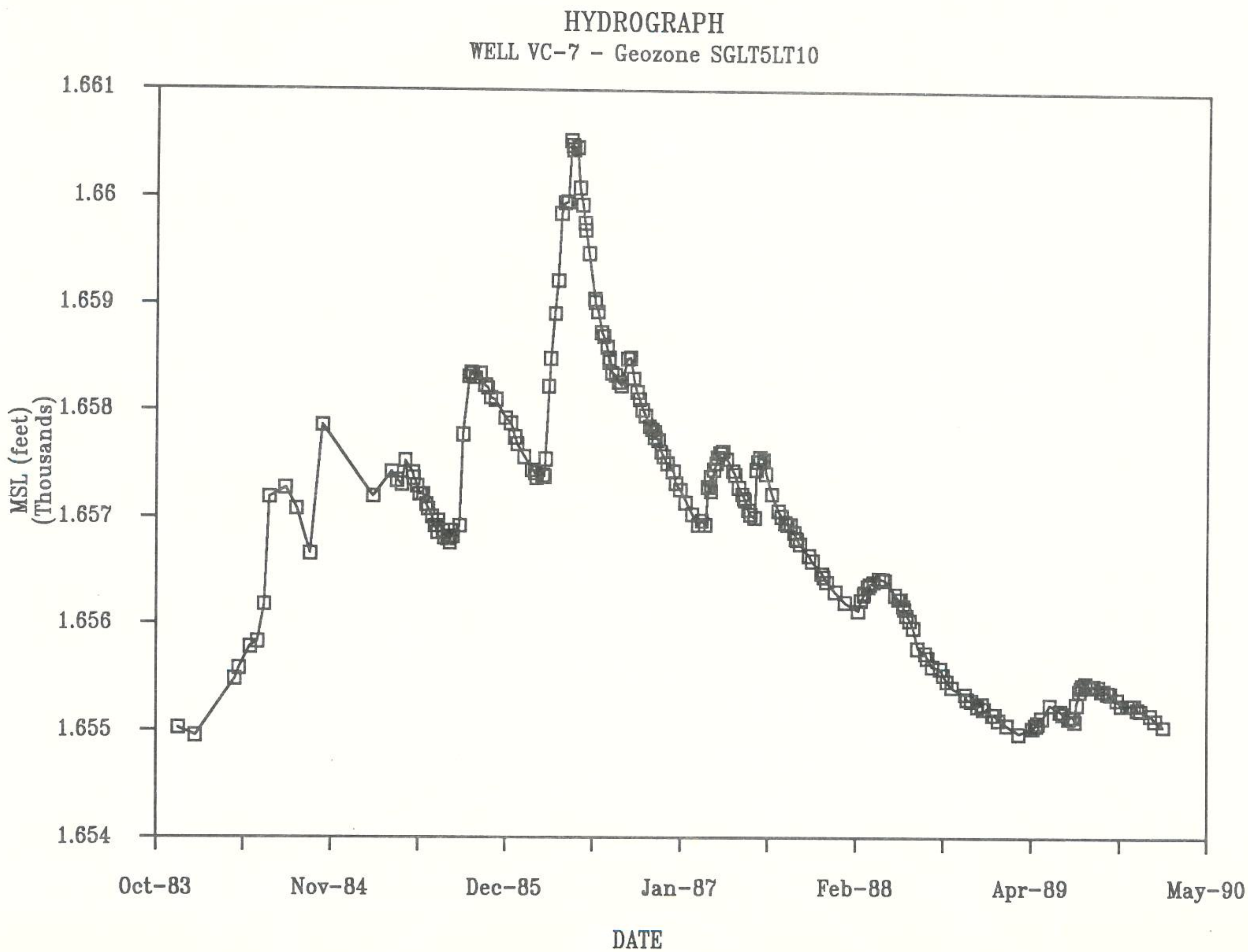
Pesticides Detected (in ppb)

Pest.	# Samp.	# Wells	Ave Conc.	Median	Range	Year(s)
Lasso	3	2	0.122	0.07	0.036 - 0.026	1988,89
Dual	2	2	0.053	0.053	0.29 - 0.077	1989
Parathion	1	1	0.05	0.05	0.05	1989

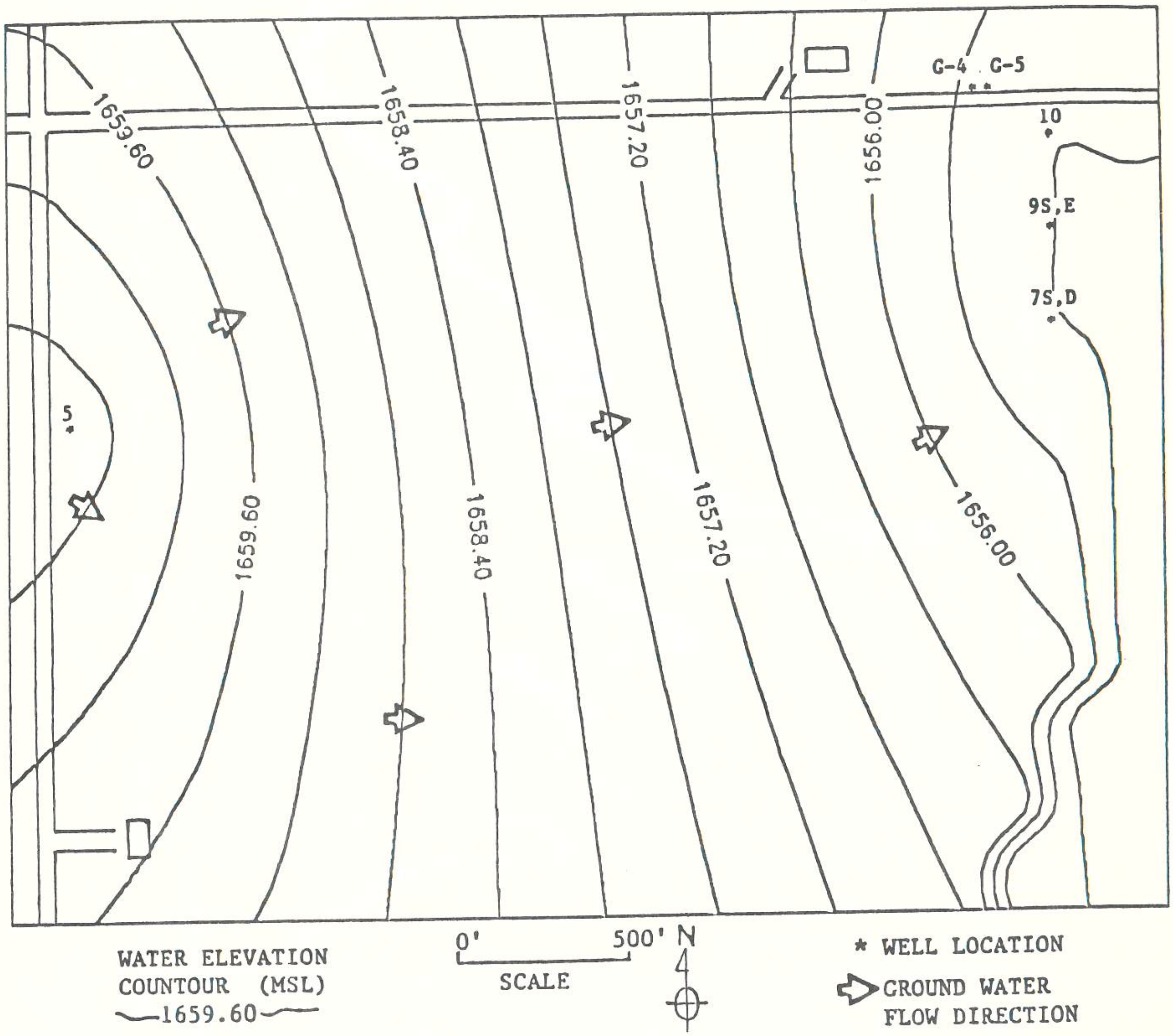
Note: # Samp. = the number of samples which detected the pesticide name;
Wells = the number of well in which the named pesticide was found, etc.

<u>Land Use:</u>

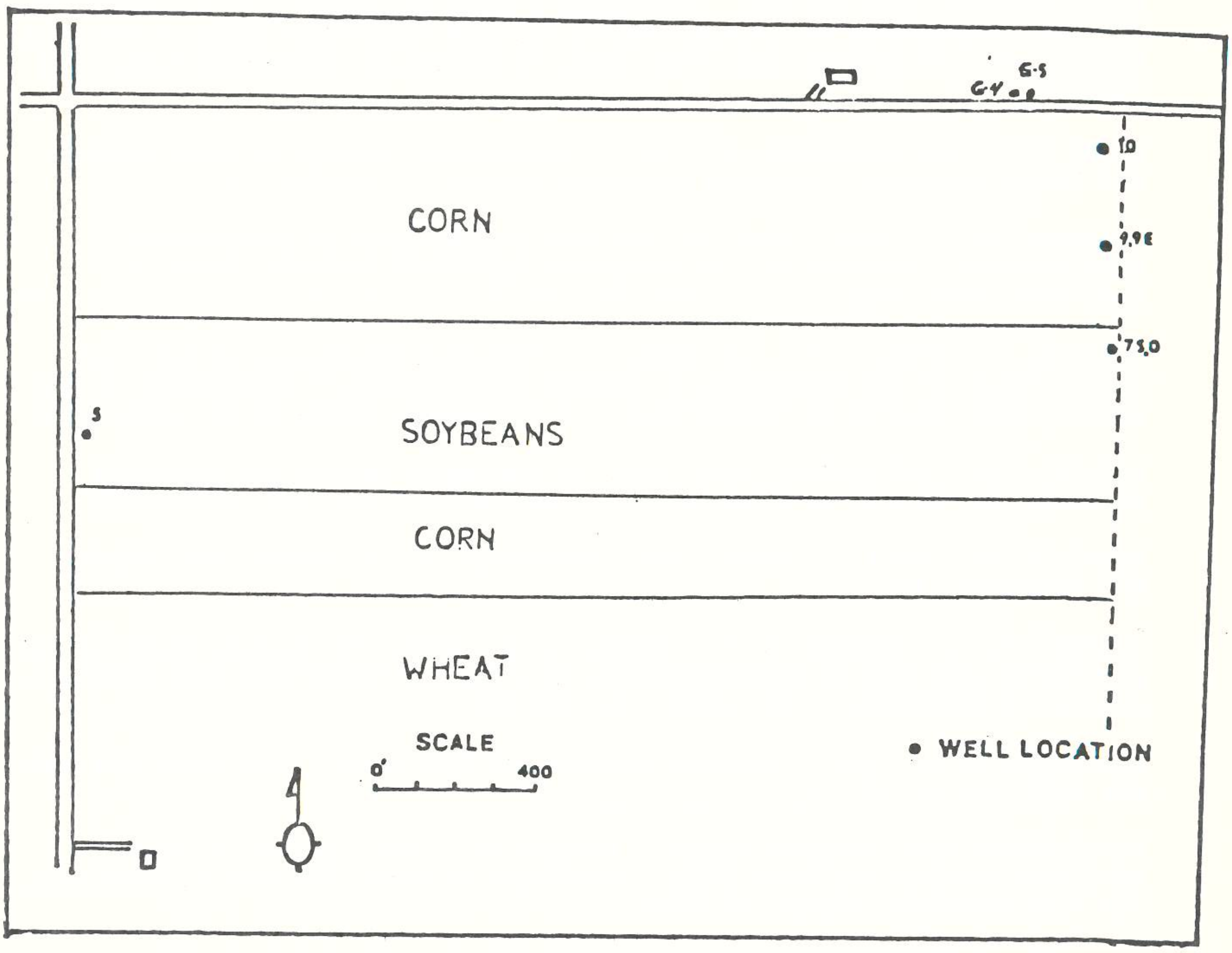
1983	Rye	70 lbs/ac 18-46-0 Fall	12.6 lbs./acre N
1984	Wheat	75 lbs/ac 18-46-0 Fall	13.5 lbs./acre N
1985			0.0 lbs./acre N
1986	Corn	100 lbs/ac 46-0-0 Spring	46.0 lbs./acre N
NOTE: new farmer starting in 1987			
1987	Winter grain	? lbs/ac ..... Fall	?... lbs./acre N
		? lbs/ac Banvel	
1988	Corn	100 lbs/ac 28-28-0 Spring	28.0 lbs./acre N
	Wheat		
1989	Corn	100 lbs/ac 28-28-0	28.0 lbs./acre N
	Wheat	MCP (spray) June 1	
	Soybeans	1.5 pints/ac Dual (banded) May 10	


HYDROGRAPH
WELL VC-7 - Geozone SGLT5LT10
MSL (feet)
(Thousands)
1.661
1.66
1.659
1.658
1.657
1.656
1.655
1.654
Oct-83
Nov-84
Dec-85
Jan-87
Feb-88
Apr-89
May-90
DATE

VC-SITE WATER TABLE MAP 6-7-89



VC-SITE 1989 LAND USE MAP



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3.0 VADOSE ZONE MONITORING - AGRICULTURAL CHEMICAL LEACHING STUDY

Author: J. H. Bischoff

3.1 Introduction

Agricultural chemicals have become a public concern in recent years because agricultural pesticides and nitrates are being detected in ground water that is used as a drinking water source. The United States Department of Agriculture (USDA) found one in five Americans may drink water contaminated from agricultural sources (Nielsen and Lee, Oct. 1987). The key to controlling chemical leaching is understanding the chemical and water transport processes through the unsaturated zone, although little field research has been conducted on the transport of chemicals from the soil surface to groundwater (Jury et al., 1987).

Agricultural chemical transport through the unsaturated zone depends on both water movement and chemical movement. Factors including tillage, crop type, crop growth stage, rainfall amounts and intensity, soil texture and structure affect water movement in unsaturated soil. Chemical movement depends primarily on water movement and the amount that a chemical is retarded in a particular soil. Since water movement influences chemical movement, the first step in understanding agricultural chemical leaching is to understand water movement in unsaturated soils and the impact that tillage or crop may have on the quality and rate of water movement.

Water movement in unsaturated soil is complicated by preferential flow paths or macropores in undisturbed soil. These pores allow rapid movement (Richard and Steenhuis, 1988) of water to specific depths, depending on amount, intensity, and duration of a rainstorm, while movement into the micropores (redistribution) occurs at a much slower rate. Many other studies have noted the effects of macropores on water movement through the soil profile (Steenhuis et al., 1989; Andreini and Steenhuis, 1988; Everts and Kanwar, 1988a; Sollins and Radulovich, 1988; White, 1985; Beven and Germann, 1982).

An evaluation early in 1987 of the Oakwoods/Poinsett Rural Clean Water Project (RCWP) was made to determine where improvements could be made in the design and implementation of best management practice (BMP) effectiveness strategy with the technology and project time available. Transiometers had been tested in other projects (Trooien, et al, 1984; Steele, 1987; Djakaria, 1989) with excellent success in monitoring the changes in matric potential with depth. In an effort to have more complete control of the land use practices and to monitor pesticides closer to the source, the Water Resources Institute at Brookings, South Dakota (with funding from the USDA RCWP in Washington, DC), started an agricultural chemical leaching study (ACLS) in 1988, to monitor and collect drainage water samples from the soil profile during rainfall events. The main goal of the study was to determine the effects of crop and tillage management on nutrient and pesticide and water movement through the unsaturated zone, which may be keys to help minimize the transport of water and chemicals to groundwater in shallow vulnerable aquifers. To achieve this goal, an event-actuated soil water sampling system was designed to automatically collect soil water samples for chemical analysis at various positions in the soil profile (Bjorneberg and Bischoff, 1989; Bjorneberg, 1989).

This is a discussion of the ACLS progress which was begun in 1988, as a part of the effort to determine the effects of BMP's on ground water quality for the Oakwood/Poinsett project. It includes an evaluation of the data collected from December, 1988 through January, 1990.

3.2 Methods and Materials

The leachate monitoring/collection system installed at the Master Site is a 0.75 acre) site (Figure 3-1) that had been previously used for a fertility study. Fifteen access holes were installed on plots that had received medium and high fertilizer rates during the previous study. (The previous study incorporated the objectives of two departments into one site. The Plant Science Department used a corn/oats rotation on moldboard plow, chisel plow, and no-till (minimum till) to determine if different tillage practices required different soil nutrient recommendations, based on soil tests. The Water Resources Institute used the site to determine if there were differing water contents in the soil profile for the different tillage systems on the same corn/oats plots). Two tillage practices, no-till (NT) and moldboard plow (MP), are used on three crops: corn, oats and alfalfa. Corn and oats are rotated while alfalfa is grown continuously. Three access holes were installed on each tillage and crop thereby comprising three subplots as opposed to three replications.

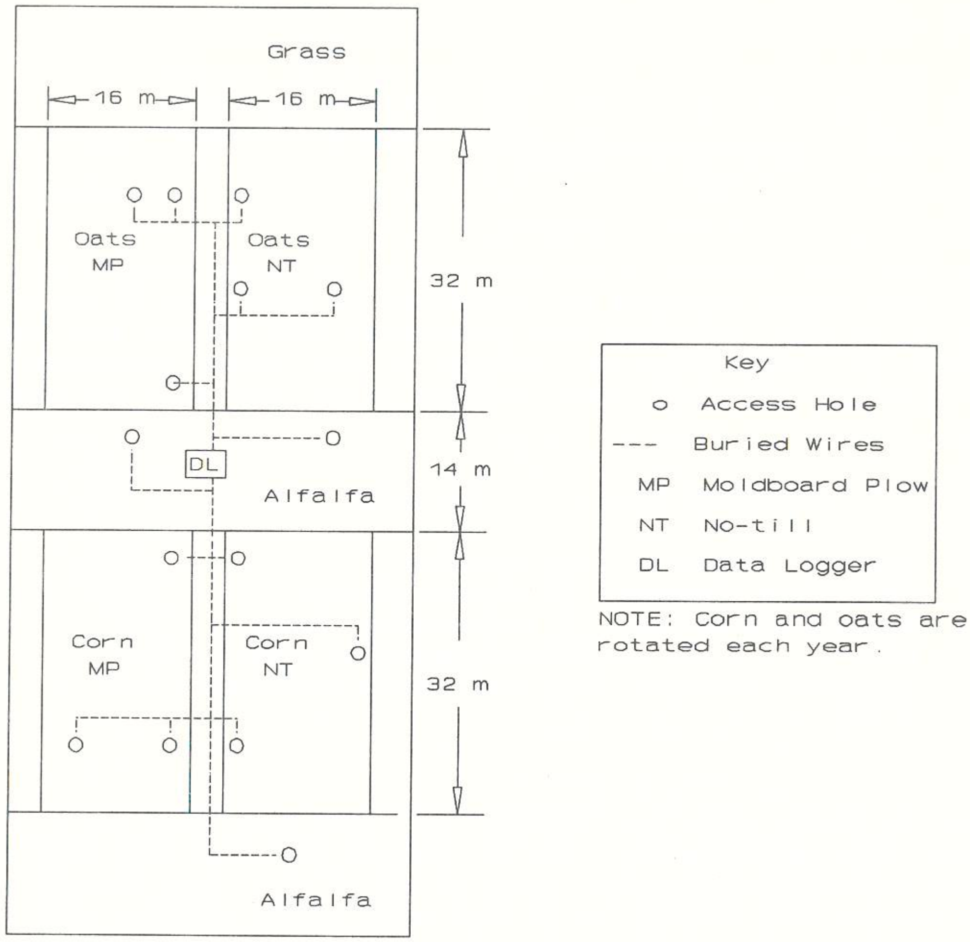


Figure 3-1. Experimental site showing location of access holes and crop and tillage practices.

Ammonium nitrate and di-ammonium phosphate fertilizers were dropped on the soil surface with a drop spreader. The corn, oats, and alfalfa treatments received 200, 100, and 0 pounds/acre of nitrogen, respectively. The MP tillage treatment had surface incorporation of the fertilizer with a spike-tooth harrow, and the NT tillage treatment received no incorporation.

Pesticides were surface-sprayed in 1989, with a tank mix sprayer with equal amounts between tillages. Counter (terbufos) was put down as a granule with the planter for both tillage treatments. All pesticides were applied at label rates. There was no incorporation of any of the pesticides applied. The pesticides that have been used on this study are listed in Table 3-1. Alachlor, Metolachlor, Dicamba, Terbufos, and 2,4-D were used on the corn treatment. Carbofuran was used for control of grasshoppers in July on the alfalfa treatment. Banvel, MCPA, and 2,4-D were used on the oats treatment.

Table 3-1. Pesticides used in the research project.

Alachlor (Lasso)	Carbofuran (Furadan)
Dicamba (Banvel)	Metolachlor (Dual)
MCPA	Terbufos (Counter)
2,4-D	

Pesticides that were scanned in the analysis of the water samples are listed in Table 3-2. Preliminary scanning for several known pesticides of the ACIS ground water samples in 1988 indicated that pesticides that had not been applied for several years were being detected. Because of this, these pesticides (Treflan, atrazine, Dyfonate, and Prowl) were included in the scan with the known applied pesticides.

Table 3-2. Pesticides scanned in the water samples.

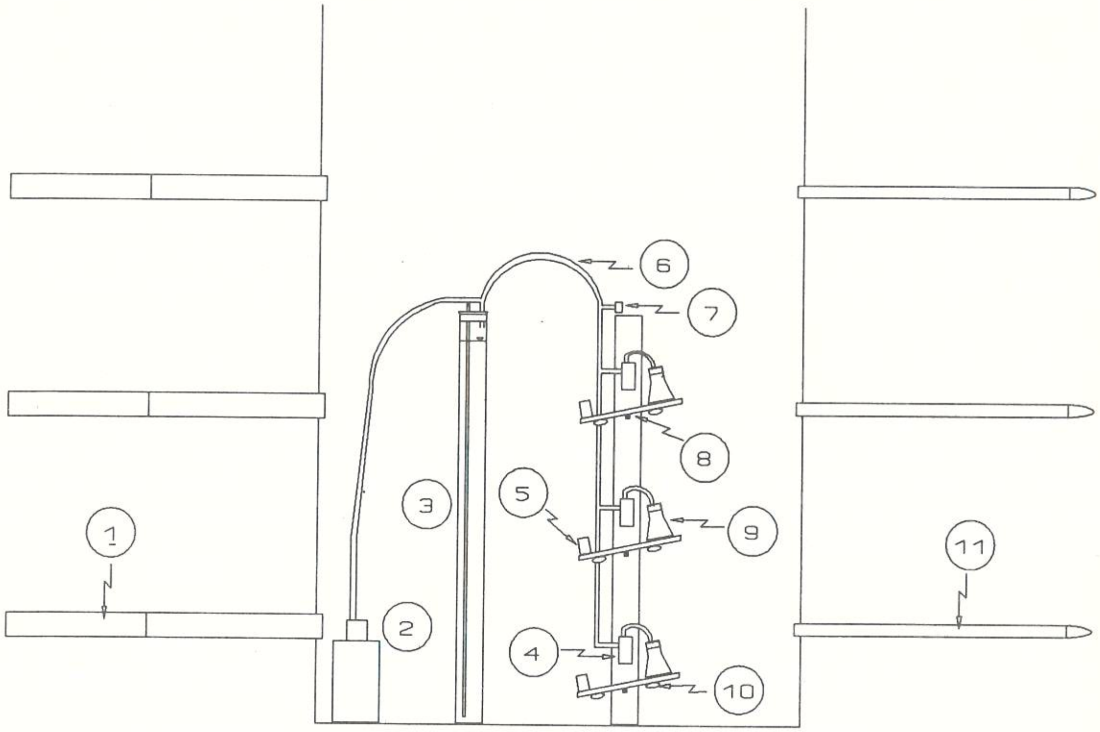
Alachlor (Lasso)	Atrazine
Carbofuran (Furadan)	Dicamba (Banvel)
Fonofos (Dyfonate)	Metolachlor (Dual)
MCPA	Pendimethalin (Prowl)
Terbufos (Counter)	Trifluralin (Treflan)
2,4-D	

The soil at the Master site is a Poinsett silt loam soil (fine-silty, mixed Udic Haploboroll) with approximately 3 to 5.5 feet of silt loam soil overlying a one foot coarse silt-alternating-sand layer on top of clay loam glacial till. The weathered glacial till extends to 25 feet below ground surface with another 25 feet of unweathered glacial till below that. A 20 foot thick confined aquifer of sand and gravel occurs beneath the unweathered till.

The entire monitoring system was designed around the treatments shown in Figure 3-1. Each location for monitoring was identified, and a 4 foot diameter access hole was drilled 7 feet below the ground surface for housing the matric potential sensors and sampling system. All essential components of the system in an access hole are represented in Figure 3-2. Transiometers are read on an hourly basis year round. A neutron access tube was installed near each access hole. Readings of volumetric water content at 6 inch depth increments are taken on the average once every five days.

Water samplers were installed at 2, 4, 6, and 15 feet depths below the ground surface. Each sampler was made from 0.5 micron pore diameter sintered stainless steel, which has approximately 400 cm of water bubbling pressure. Sintered stainless steel tubes, 2.5 in. in diameter by 30 in. long, were installed laterally from the access hole. Transiometers were also installed laterally near each water sampler to determine the matric potential.

When a rain event begins, a tipping bucket rain gage starts a datalogger recording the rain depth in a moving 10 minute sum. If the matric potential at the 2 ft. depth in each access hole is above -200 cm, a suction pump is activated to collect any potential drainage water moving through the soil profile.



1. Soil Water Sampler
2. Suction Pump
3. Suction Regulator
4. Solenoid Valve
5. Counter Weight
6. Suction Tube
7. Suction Transducer
8. Tipping Switch
9. Sample Bottle
10. Mercury Switch
11. Transiometer

Figure 3-2. Access hole with key components for automatic monitoring and low tension suction pump activation.

After the water sample is obtained, it is placed in a cooler for transport to the lab. Aliquots are poured from the main sample for electrical conductivity (EC), pH, and nitrate analysis. Nitrate analyses are determined using the Brucine-sulfate method within 24 hours of collection. Pesticide extractions are also done within 24 hours of collection and are analyzed on a HP 5890 Series II gas chromatograph with electron capture.

3.3 Results and Discussion

The discussion of the results will be divided into three categories: 1) soil water flow, 2) nitrates, and 3) pesticides. Total rainfall in 1987, 1988, and 1989 was 20 in. (6.7% below normal), 17.5 in. (17.9% below normal), and 21 in. (near normal), respectively. High water use crops tended to mine the soil water from the soil profile in 1988, thus lowering the leaching potential for these fields. Low water use crops, such as oats, planted in 1988, allowed maintenance of the subsoil water in the soil profile. Drainage water samples were collected in 1989 (the first operational season after instrument installation) only from the 1988 oats plots (north plots). All the unsaturated water quality samples were taken from the north plots (6 different access holes) which were planted to corn in 1989. The saturated water samples were collected on a monthly basis, and after significant rainfall events, from all fifteen access holes. Because the water table position varied tremendously between treatments, the samples were collected from depths varying from 2 to 12 feet below the water table.

3.3.1 Soil Water Flow

There were six significant leaching events in 1989 that created water to move to the water table for corn treatments (oats previous year). The amount and duration of the leaching events are given in Table 3-3. There was no recorded runoff during any of the rainfall events for any of the treatments. The recorded daily rainfall amounts are shown in Figure 3-3.

Table 3-3. Significant rainfall events that created leaching water to move to the water table for the spring and summer 1989.

DATE	DEPTH (mm)	(in.)	DURATION(hrs.)
4/26/89	16	0.63	3.25
4/28/89	18	0.65	8.4
6/17-18/89	41	1.61	18.0
6/25-26/89	42	1.65	5.5
7/10-11/89	88	3.46	21.0
7/18/89	56	2.20	7.0

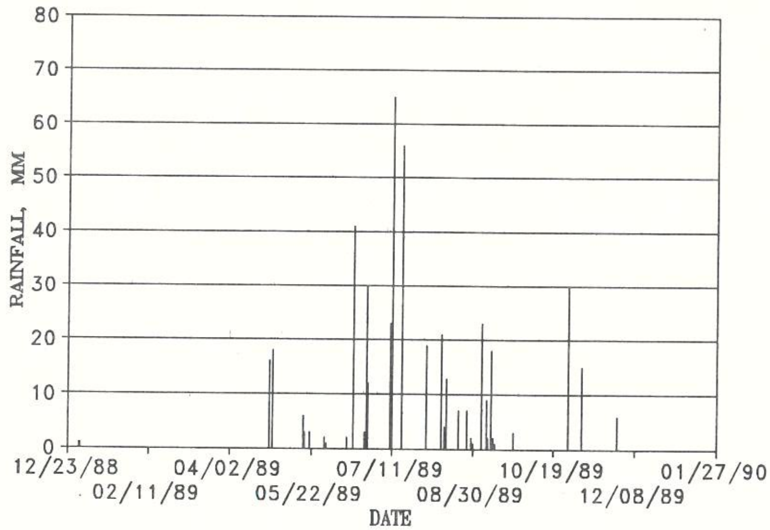


Figure 3-3. 1989 daily rainfall amounts recorded at the Agricultural Chemical Leaching Study site.

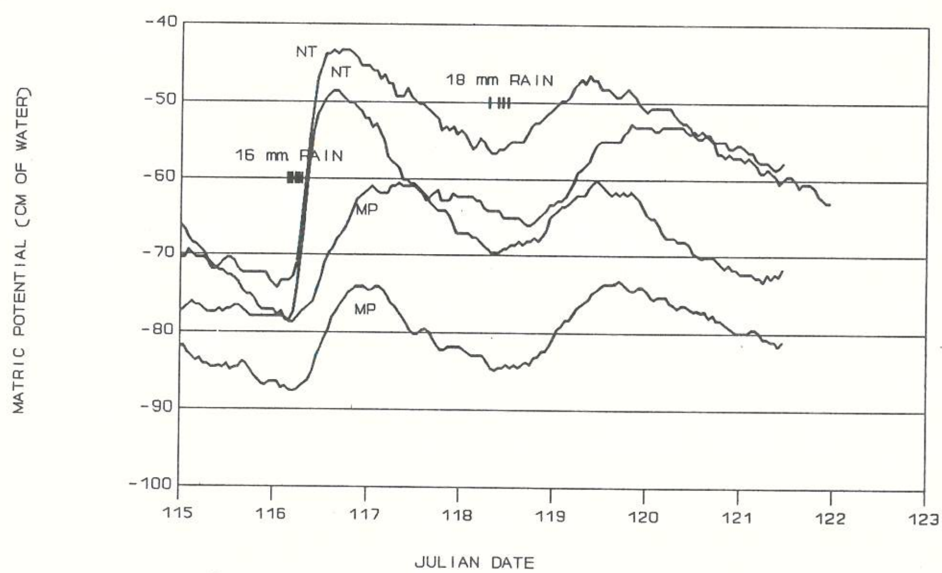


Figure 3-4. Matric potential response to two rainfall events in April for moldboard plow (MP) and no-till (NT) treatments 2 ft. below ground surface.

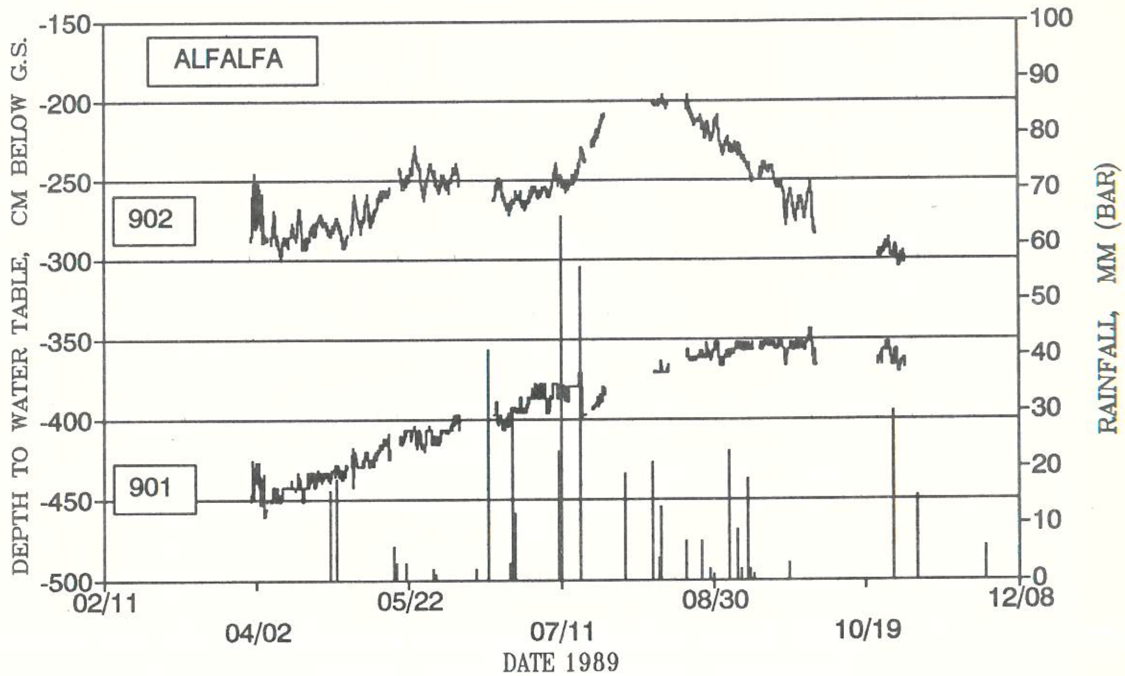


Figure 3-5. The water table position for two replications on alfalfa.

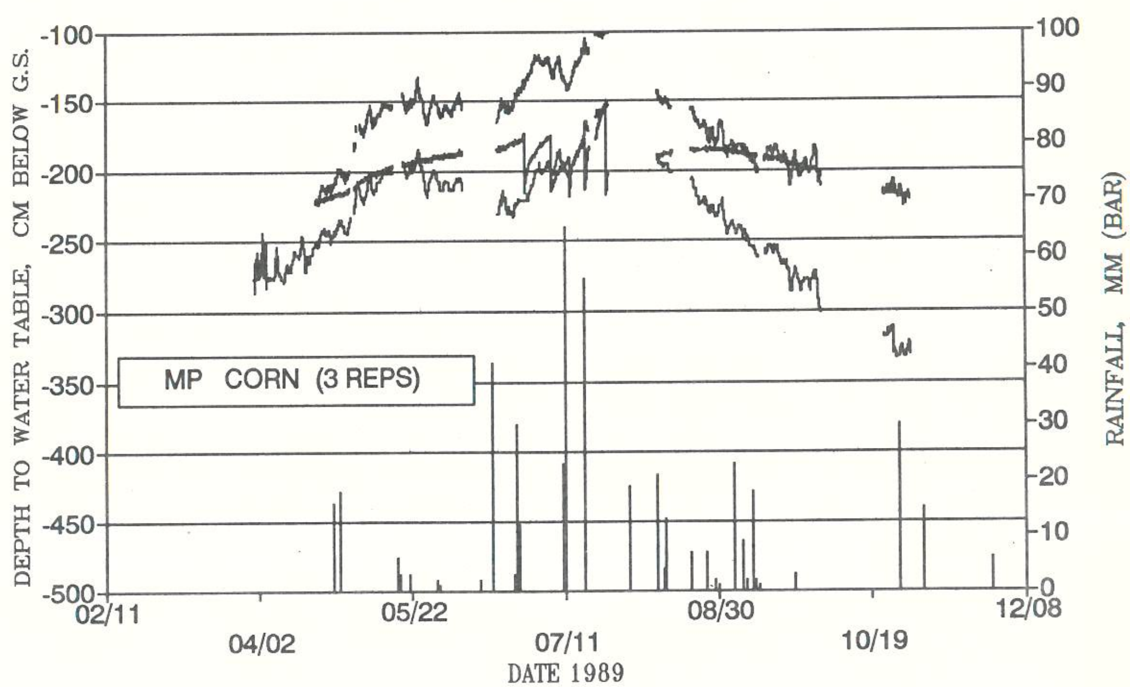


Figure 3-6. The water table position for 3 replications on moldboard plow corn.

These various leaching events affected the water movement across tillage and crop quite dramatically. As an example, prior to the 16 mm rainfall which fell on April 26, there was very little rain, and the soil surface was beginning to dry out. Figure 3-4 traces the changes that took place in the matric potential at 2 ft. as a result of a 16 mm and an 18 mm rainfall event two days apart. Notice the distinct difference in the response time of the percolating water between the NT and MP treatments. It appears that there was rapid movement to that depth and then redistribution into the soil matric with time. After the second rainfall two days later, the same type of phenomena did not occur. There are three possible explanations for this: 1) the soil macropores were closed down somewhat by the wetting of the soil from the first rain, 2) the soil water content was high enough in the macropores to reduce the flow, and 3) the intensity of the storm was much less for the second rain event. Water moved through the soil to the 6 ft. depth and began to raise the water level above 6 ft. There was a significantly higher rate of water table rise for NT compared to MP, but the amount of water leaching through the soil was very similar for both treatments, perhaps a slight increase for the NT.

Comparisons between the water table position throughout the year for the various treatments can be seen in Figures 3-5 through 3-8. Each of the treatments appeared to allow water to move through the soil profile to the water table after the frost in the ground was gone, regardless of water content and cropping regime. A corresponding increase in the nitrates for that same period of time indicates that there was movement of nutrients deeper in the profile. This increase in nitrates into the ground water beneath the alfalfa plots was not expected but was consistent for all replications. Concentrations of nitrate as nitrogen increased from 4 ppm to 6-8 ppm on the alfalfa as a result of the final two rainfall events that fell on July 10 and July 18. Plot #903 was significantly drier at 6' than the other two alfalfa replications (Figure 3-9) and did not show as great a nitrate concentration increase in samples collected from the saturated zone (Figure 3-19). The air entry level of the ceramic cup at 6' was exceeded on plot #903 because the soil was so dry and subsequently, continous readings were interupted. It was continually recharged and replaced, but the matric potential did not rise above -800 cm of water.

Figure 3-10 is a typical trace of the matric potential at 6 ft. below the ground surface for two replications on plots with oats on it. The cyclical pattern of the trace is the result of barometric pressure fluctuations.

Volumetric water content measurements, taken with a neutron scattering instrument, at different depths and at different times indicates that there was an increase in water content at 54" for plot 903 (Figure 3-11) and an increase in water content at 54", 60", and 66" for plot 901 (Figure 3-12) as a result of the two rainfall events that fell on July 10 and July 18. This verifies that water did move into the lower profile of the alfalfa plots and had the capability of carrying nitrates with it into the ground water.

The response in matric potential changes between tillage systems for the corn as a result of rainfall, was very similar for all 6 leaching events.
Figures 3-13 through 3-17 are graphs of the matric potential changes with depth for a 4.2 cm rainfall event that took place on June 26. The soil at the 2 ft. and 4 ft. depths for the alfalfa treatment (Figure 3-13) was drier than -1 bar

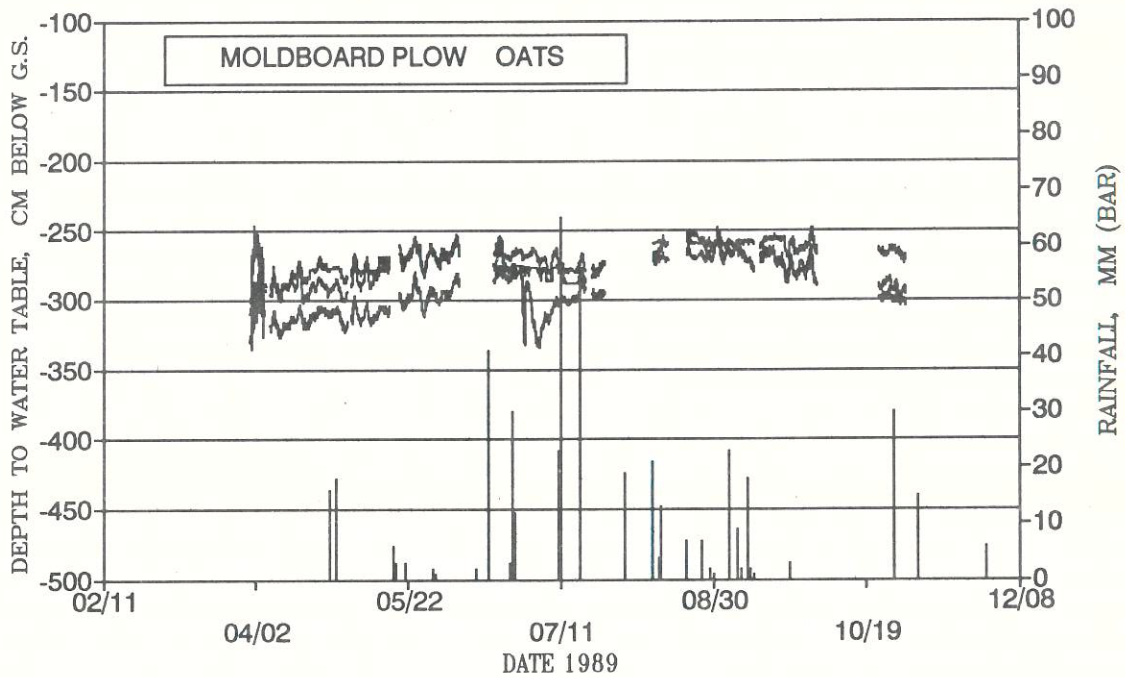


Figure 3-7. The water table position with time for three replications of moldboard plow oats.

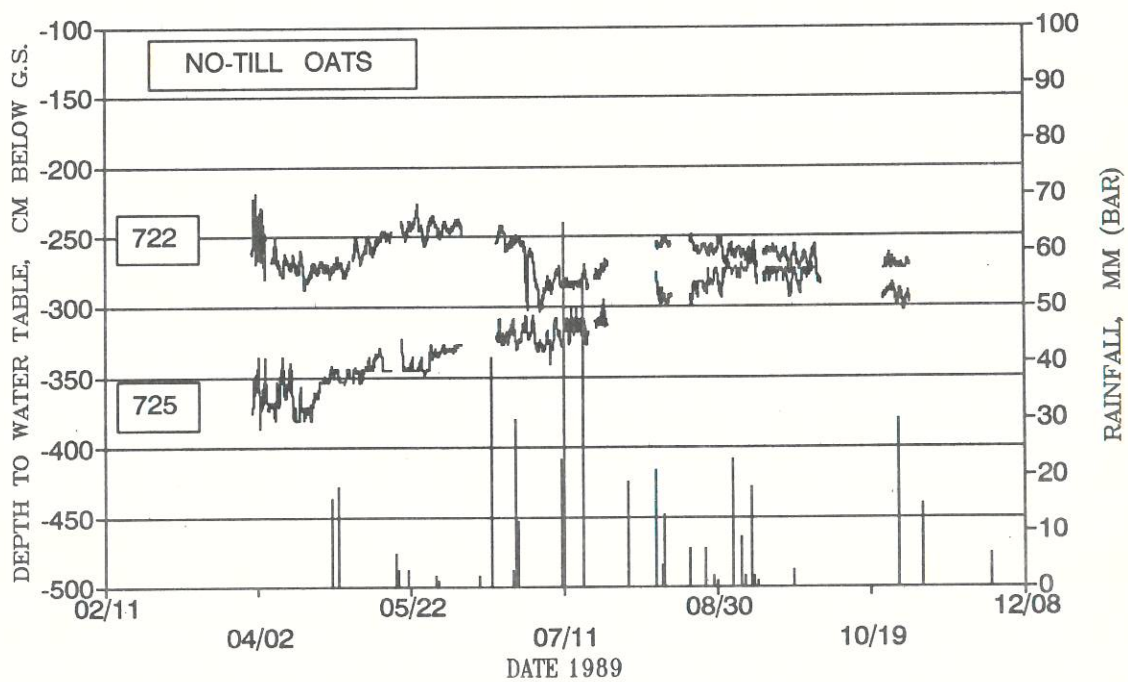


Figure 3-8. The water table position with time for two replications of no-till oats.

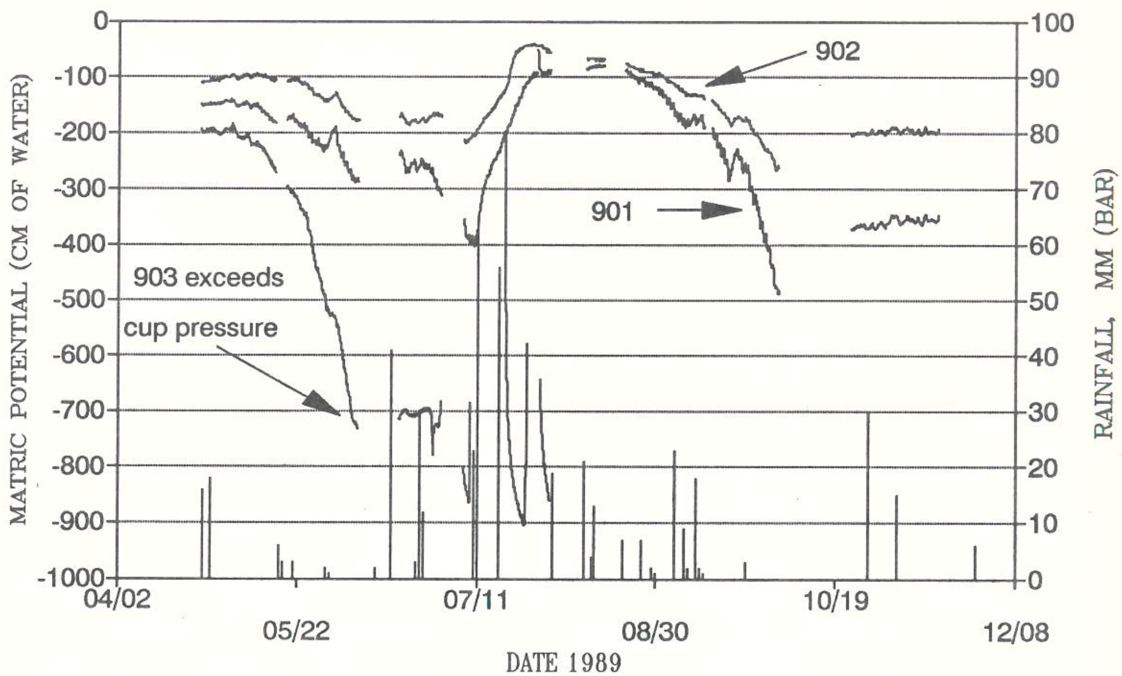


Figure 3-9. The matric potential at 6 ft. for three replications of alfalfa.

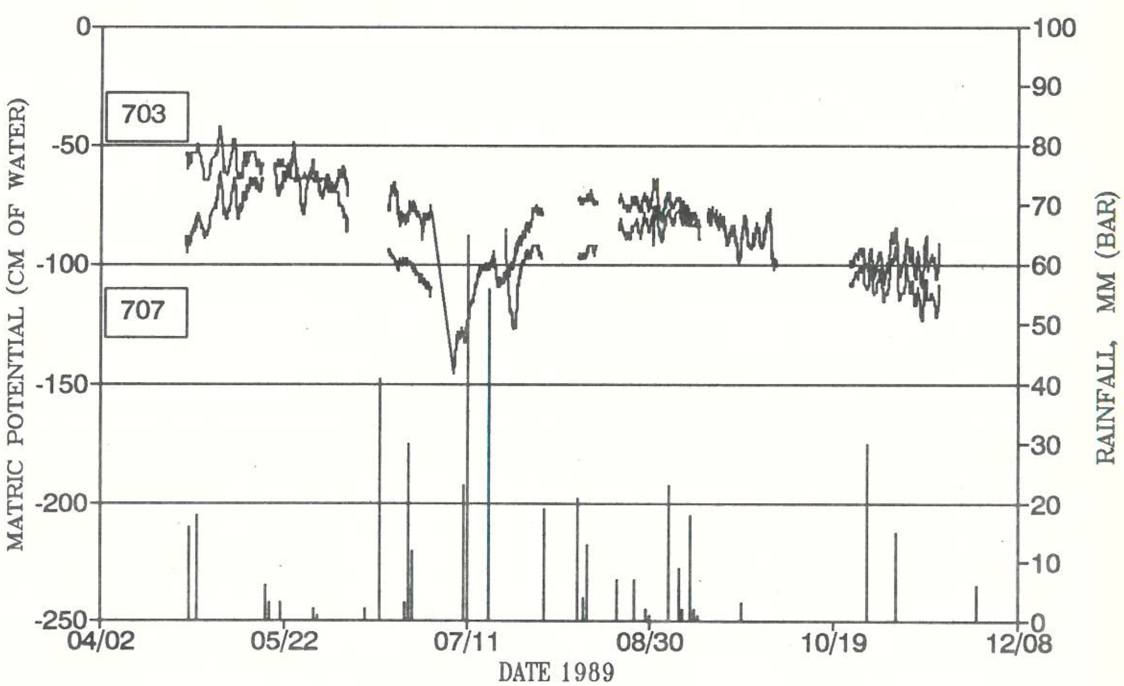


Figure 3-10. The matric potential at 6 ft. for two replications of moldboard plowed oats.

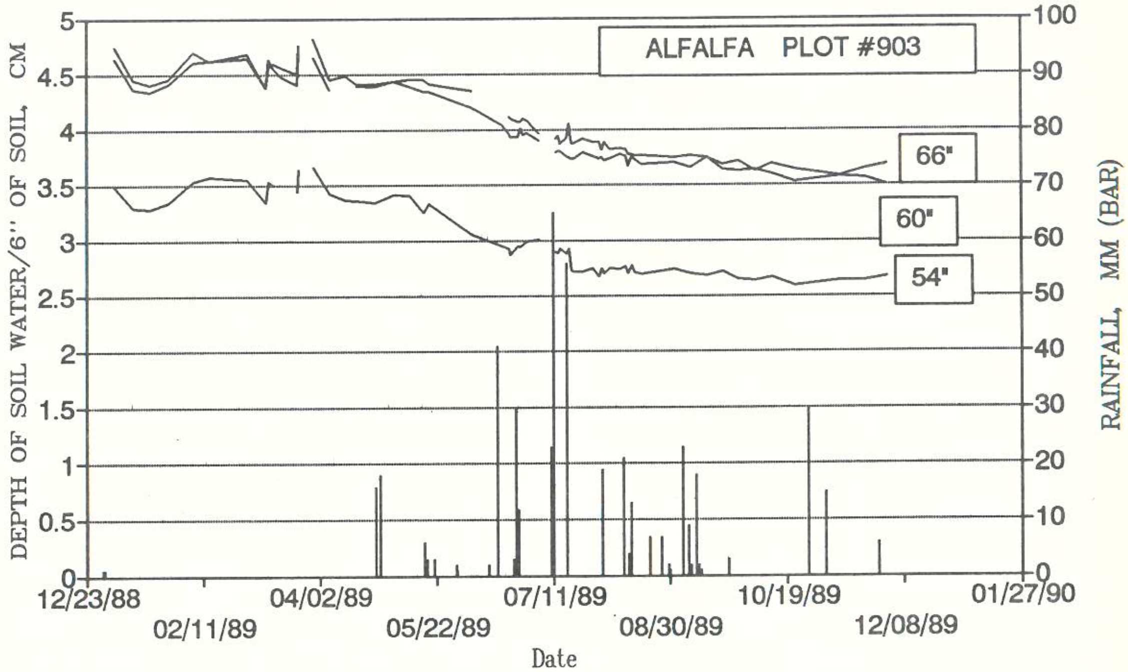


Figure 3-11. The volumetric water content of three different depths with time for the driest alfalfa plot #903

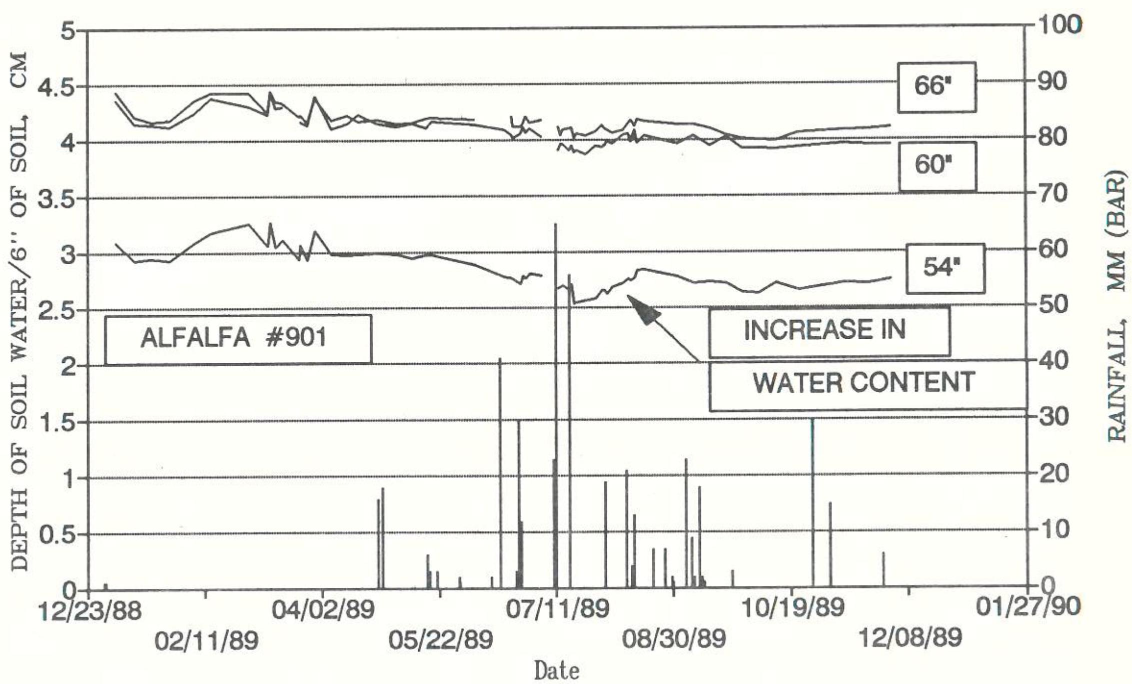


Figure 3-12. The changes in volumetric water content with time for three depths on alfalfa plot # 901.

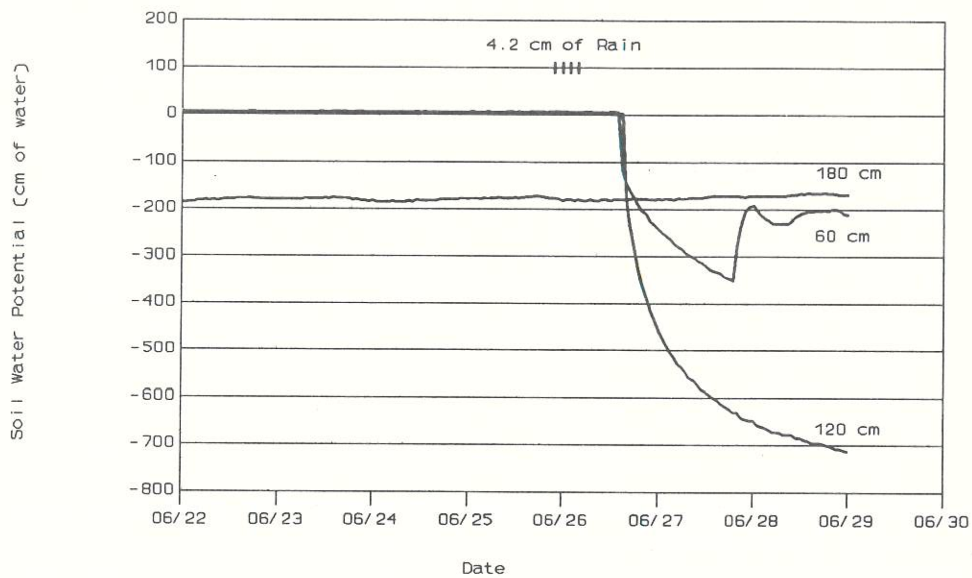


Figure 3-13. Effects of rain on leaching water for an alfalfa plot, 1989. (The 2 ft and 4 ft depth ceramic cups were recharged after the rainfall event.)

(bubbling pressure of the ceramic cup). After the rainfall event occurred, the cups were recharged and placed back into the soil. There was only a slight change in the matric potential at 2 ft. two days after the rainfall. The water never reached the 4 ft. or the 6 ft. depth. This is verified by the neutron probe readings which were taken June 23, 26, 27, 28, and 30.

Neither the MP or NT oats soils had any leaching water moving to the water table as a result of that rainfall event. The matric potential never changed at 2 ft., 4 ft., and 6 ft. depth (Figures 3-14 and 3-15).

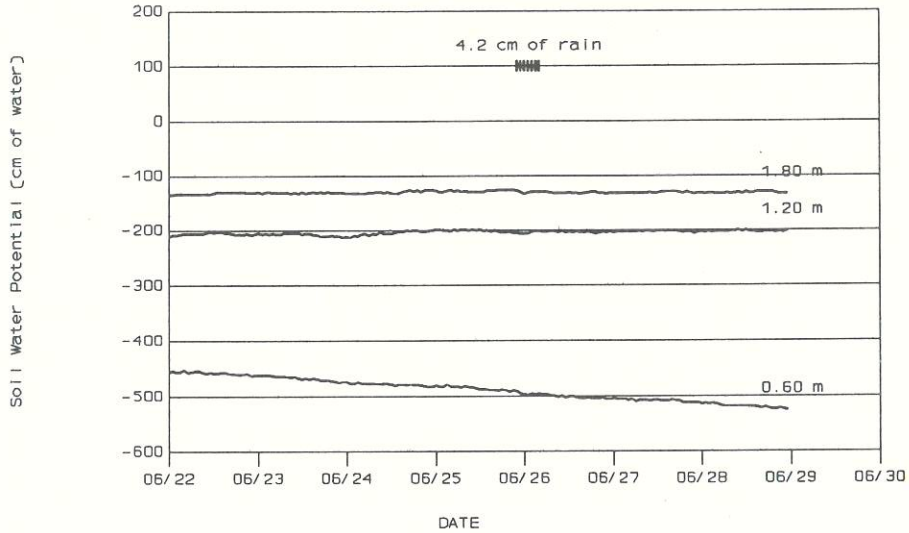


Figure 3-14. Effects of rain on leaching water for a NT oats plot (corn last year).

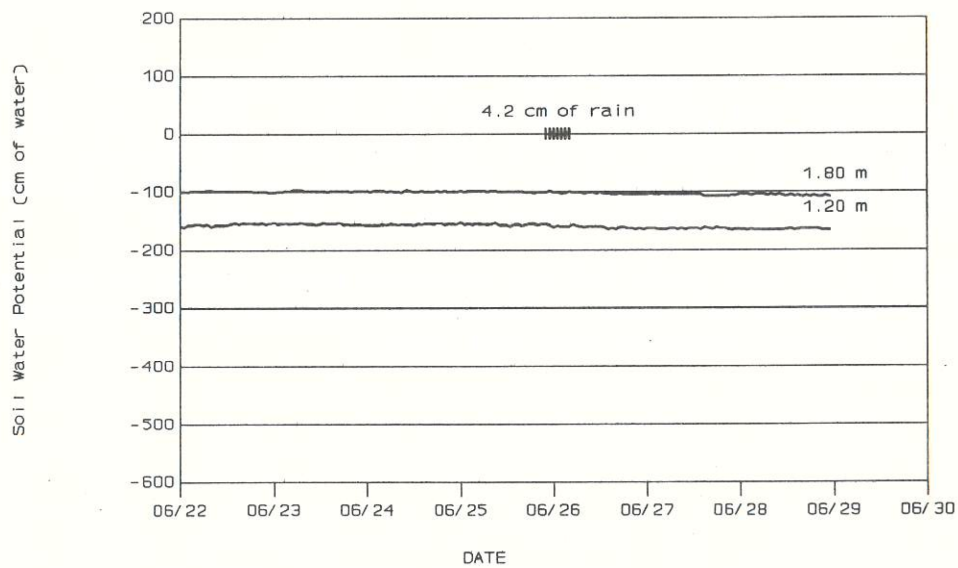


Figure 3-15. Effects of rain on leaching water for a MP oats plot.

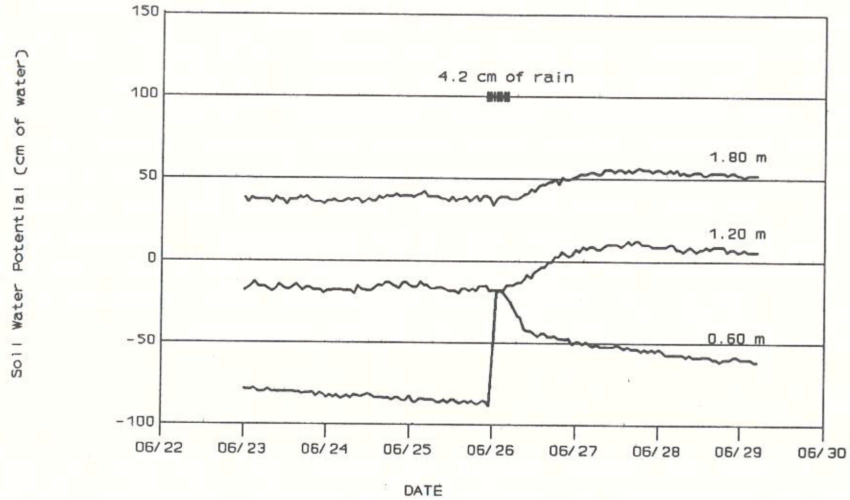


Figure 3-16. Effects of rain on leaching water for NT corn treatment, 1989.

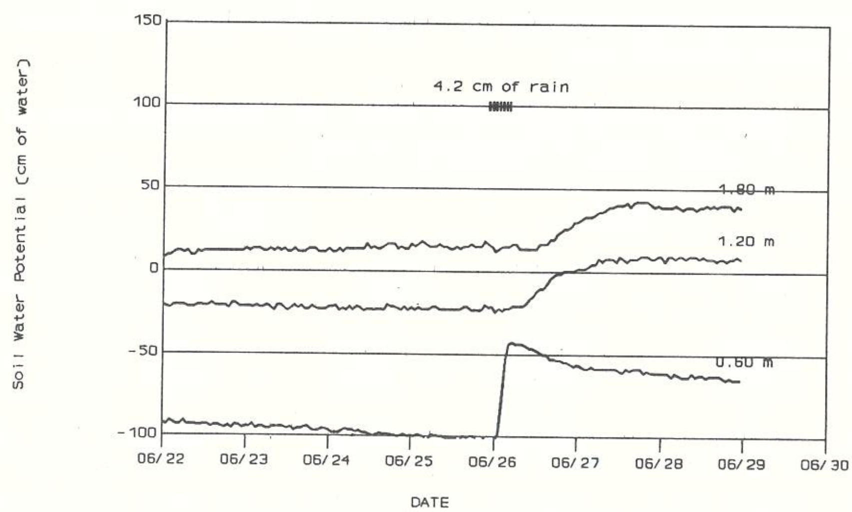


Figure 3-17. Effects of rain on leaching water for MP corn treatment, 1989.

Because the soil profile was wetter for the corn treatments (Figures 3-16 and 3-17), leaching water moved from the surface through the soil to the water table and raised the water table 20-25 cm. If the average drainable porosity of the soil profile at the 6 ft. depth were 5%, the depth of water leached through the soil profile would be equal to 1.0-1.25 cm. This water is not in a sand and gravel aquifer that is used for domestic drinking water purposes, so the water and nutrients will probably be used up by plant roots penetrating to that depth and mining them. The real question and concern is whether the mechanisms for macropore flow and subsequent solute transport follow this same pattern on shallow, coarser-textured soils with sand and gravel subsoils. Perhaps, the magnitudes are different, but the mechanism for transport as a result of tillage may be the same. In that same scenario, what would be the effect of injecting the soil fertilizer into the soil for a NT treatment? Would the same results apply? Does conservation tillage (chisel plow, ridge till) fall closer to the NT or MP effects? These are definitive questions that need to be answered with further studies addressing specifically these concerns.

3.3.2 Nitrates

Now that the fluxes of water based on crop and tillage effects have been determined, it is necessary to evaluate the solute transport mechanism. Table 3-4 is a summary of the mean concentrations of nitrate as N and the standard deviations from water samples collected from 3 depths and 5 leaching events. There is a significant difference in the concentrations of nitrate as N between NT and MP at the 2 ft. level both before and after fertilizer application (May 12). It may appear from the means that the MP treatment may have higher concentrations than the NT at the 4 ft. and 6 ft. depths. However, because of missing samples and high variability between subplots, there are no significant differences. When trends are plotted at the 6 ft. depth (Figure 3-18), the NT corn shows increasing concentrations of nitrate as N with time compared to the MP, with the exception of one MP subplot. This plot had fertilizer spilled into the access hole during application and the higher concentrations may be the result of this short-circuiting.

Table 3-4. Mean and Standard Deviation of Concentrations of Nitrate as N (ppm) from 5 leaching events.

	2 feet		4 feet		6 feet	
Date	NT	MP	NT	MP	NT	MP
05/04/89	26±8	76±28	14±4	19±3	17±3	----
06/22/89	41±11	173±30	20±2	24±0	20±1	27±5
06/28/89	45±14	152±27	40±10	88±56	38±14	54±51
07/12/89	63±6	162±25	73±34	55±3	49±22	40±30
07/18/89	38±14	157±43	83±52	92±49	73±29	40±22

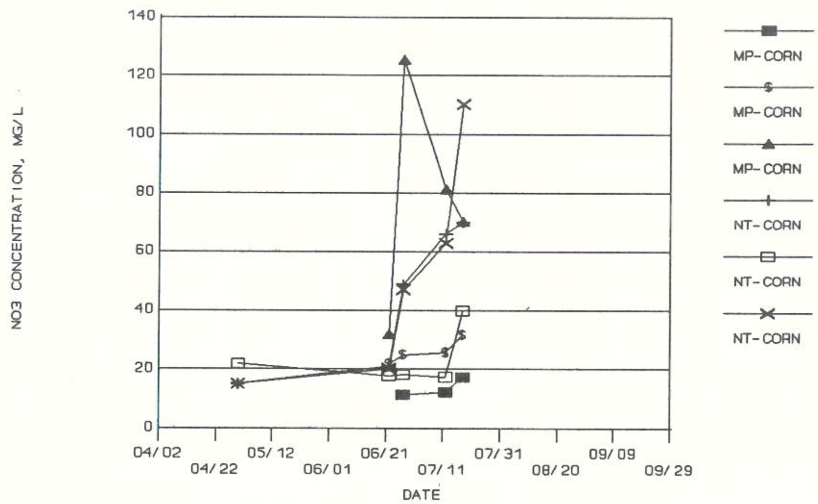


Figure 3-18. Concentrations of nitrate as N with time for the 6 ft. depth for NT and MP corn treatments.

Figures 3-19 through 3-23 are the concentrations of nitrate-nitrogen from the saturated samples plotted with time and with the rainfall events that took place over the year. As the graphs are viewed there is a clear indication that the concentrations of nitrate-nitrogen increased after the last two rainfall events occured on July 10 and July 18. This indicates that the soil profile reached field capacity and finally could not hold any more water and it leached to the subsurface water table. The only plots that did not show this type of trend were the drier plots (#903 on the alfalfa). All the plots decrease in concentrations of nitrate after the July rainfall peak, and then level off. There appears to be a gradual increase in concentration of nitrate-nitrogen over the winter months that increases right after the frost goes out of the soil.

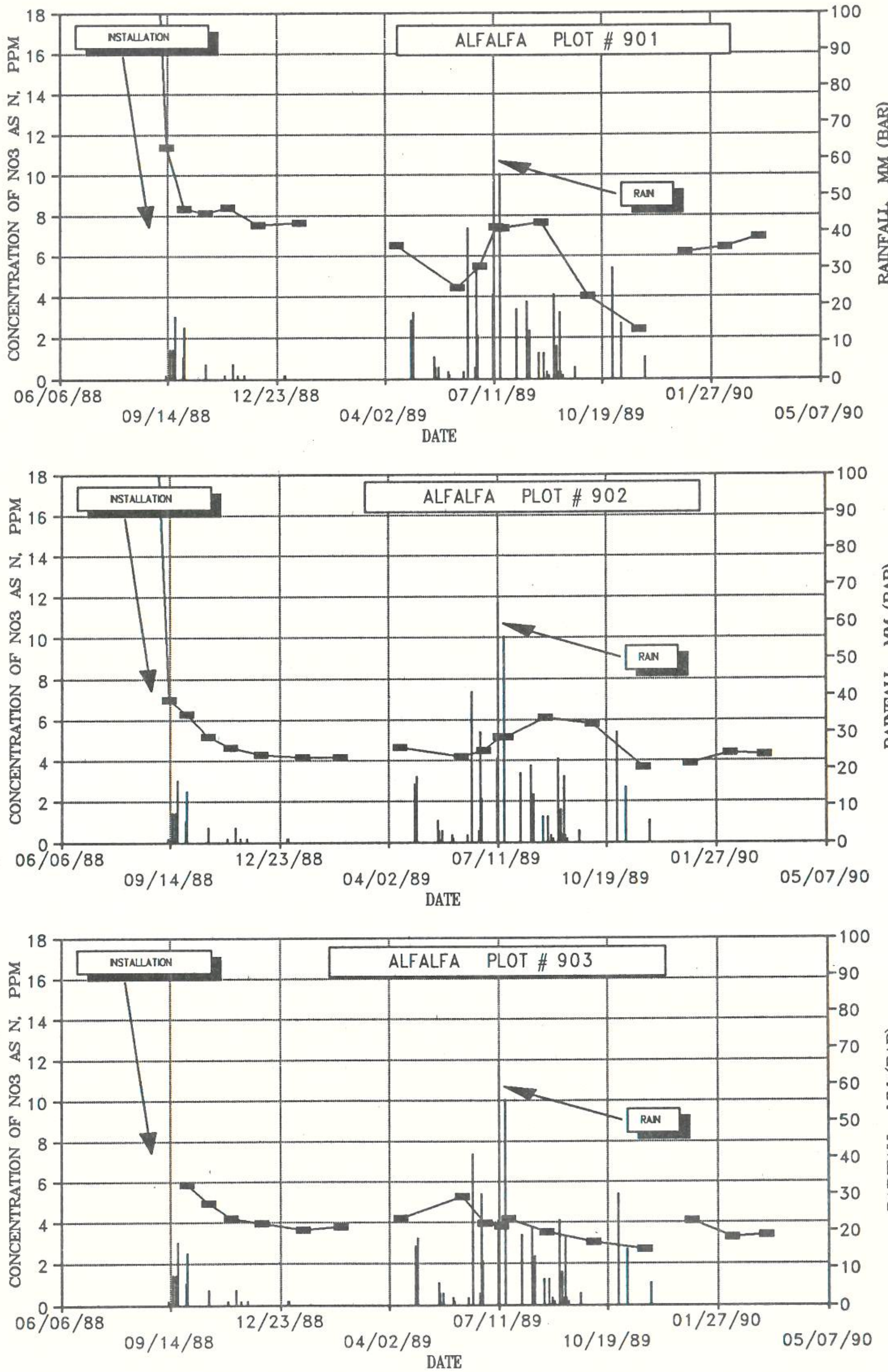
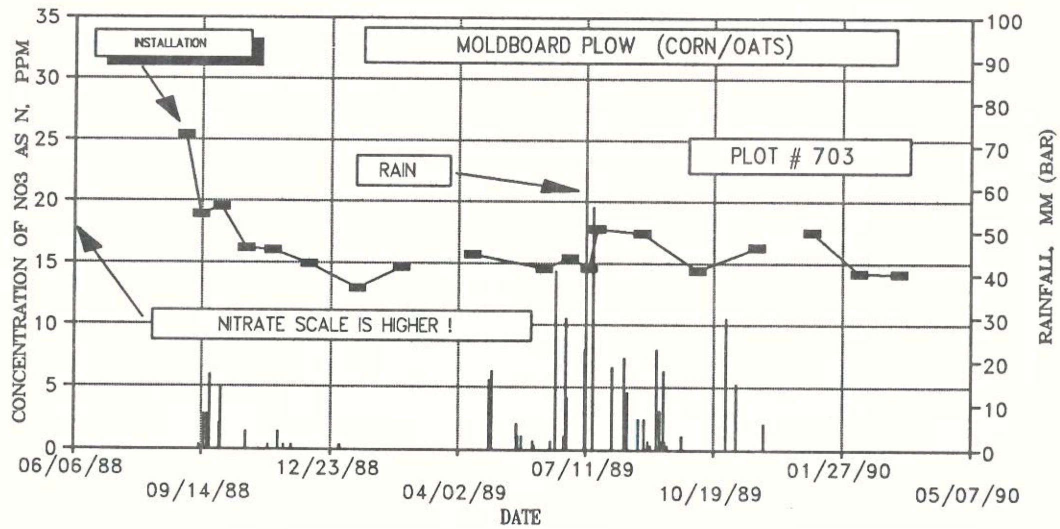
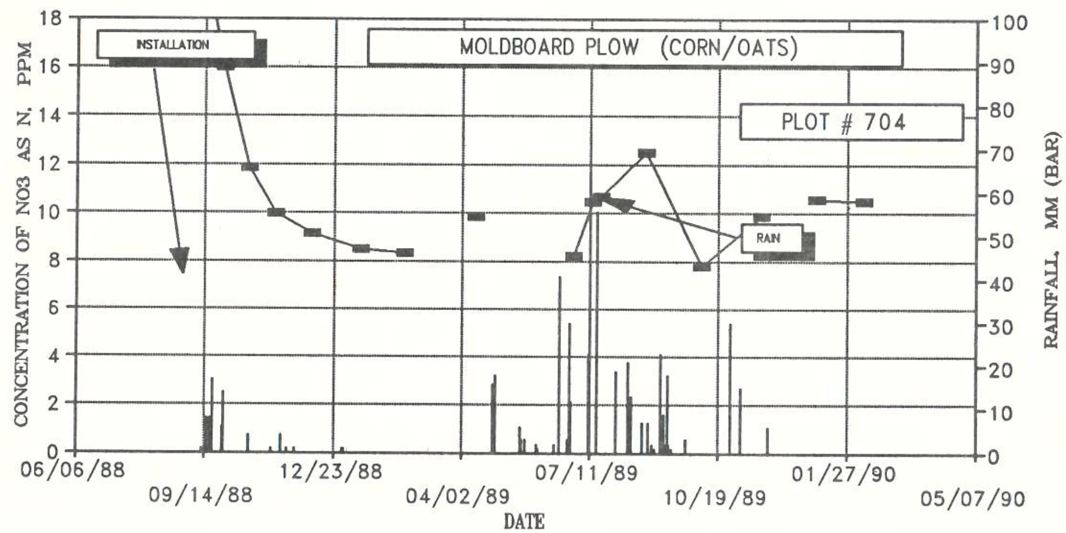


Figure 3-19. The changes in nitrate-nitrogen concentrations with time from the saturated layer for three replications of alfalfa.





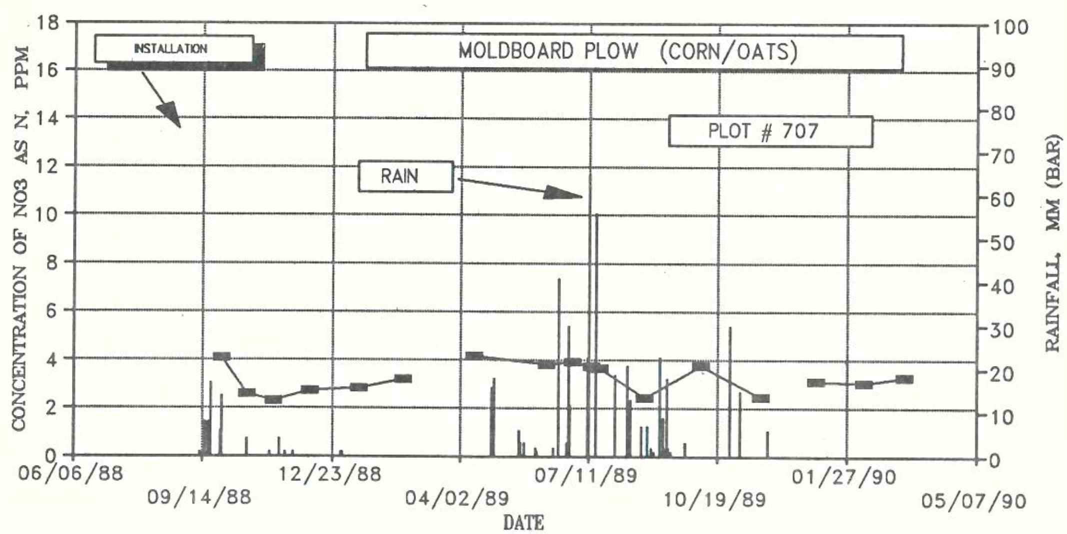


Figure 3-20. The changes in nitrate-nitrogen concentrations with time from the saturated layer for three replications of MP oats following corn.

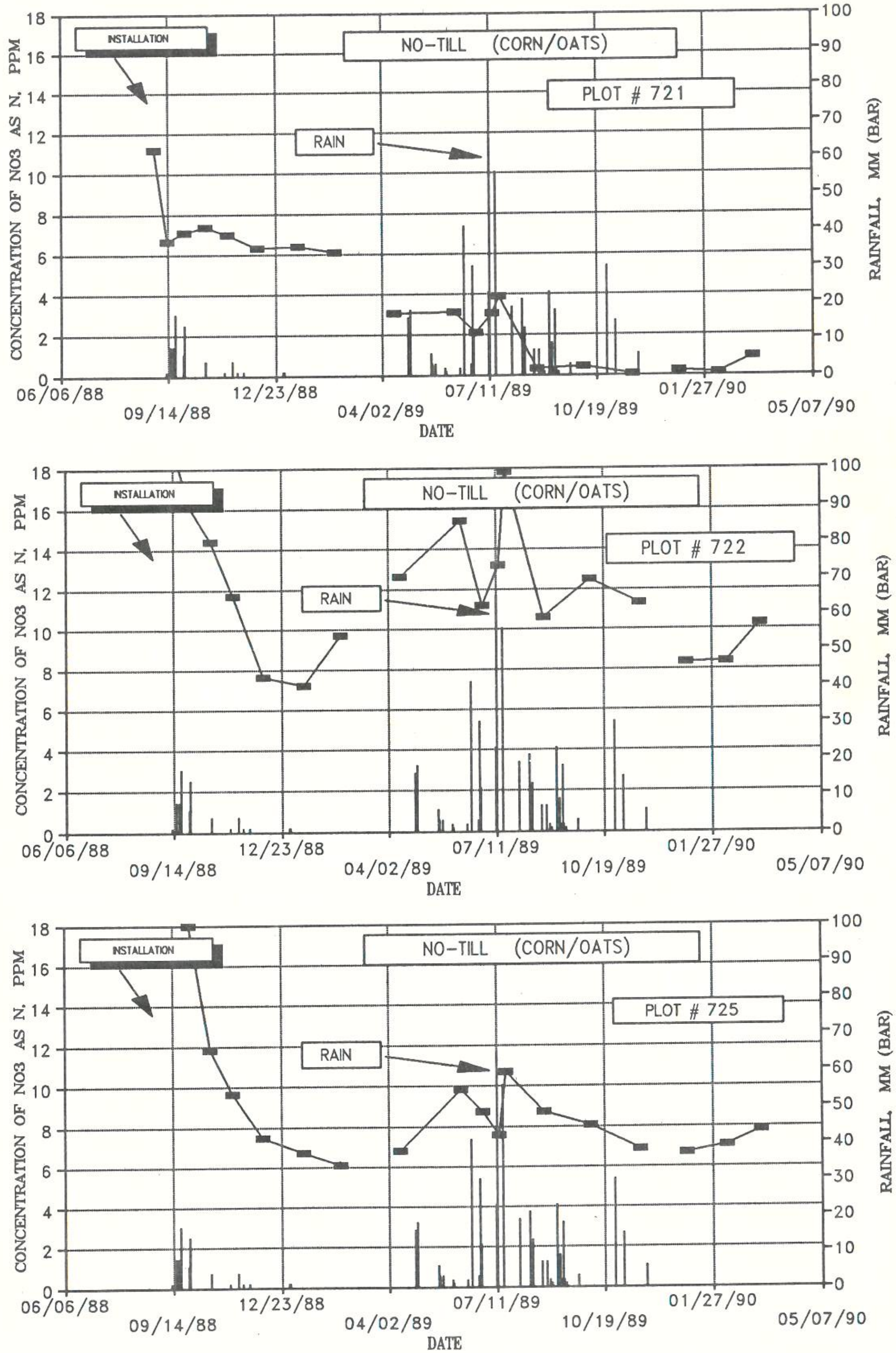


Figure 3-21. The changes in nitrate-nitrogen concentrations with time from the saturated layer for three replications of NT oats following corn.

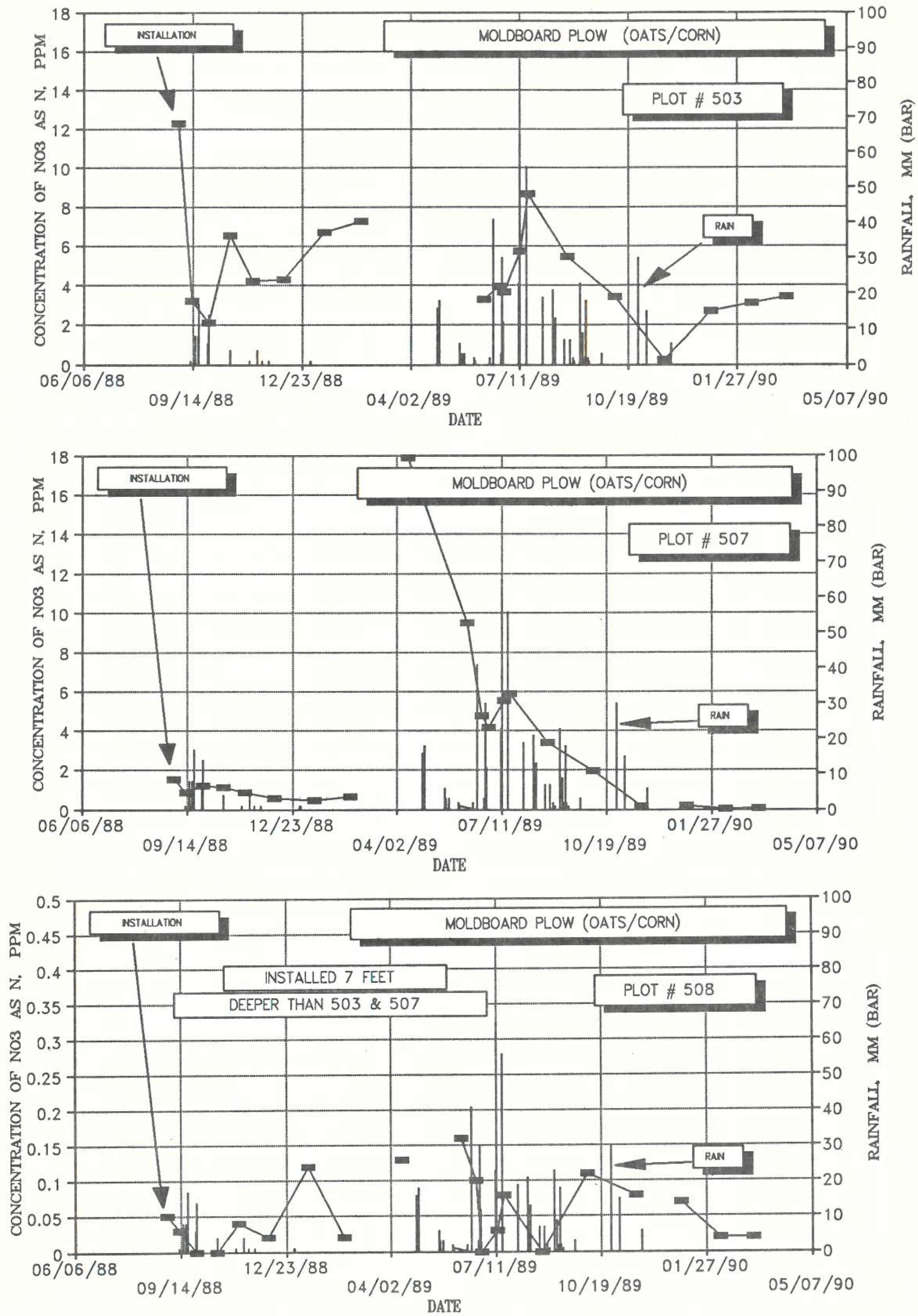


Figure 3-22. The changes in nitrate-nitrogen concentrations with time from the saturated layer for three replications of MP corn following oats.

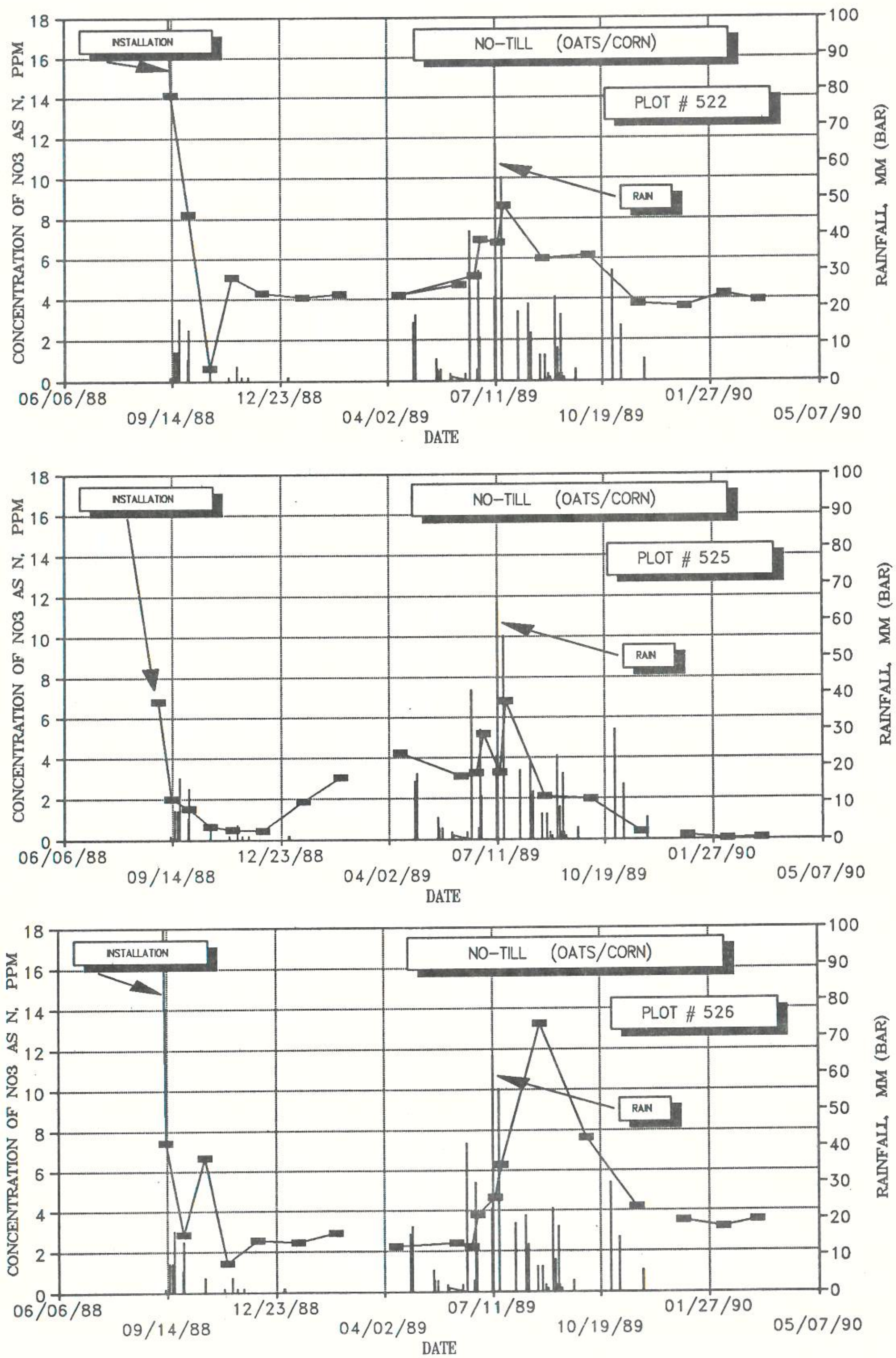


Figure 3-23. The changes in nitrate-nitrogen concentrations with time from the saturated layer for three replications of NT corn following oats.

3.3.3 Pesticides

Pesticide detections were more numerous in samples collected from the upper soil profile (unsaturated) than from samples collected from wells (saturated). The average number of pesticides detected per sample collected was the highest at the 2 ft. depth (and the lowest at the 15 ft. depth (Figure 3-24). The number of pesticide detections decreases as the depth below ground surface increases.

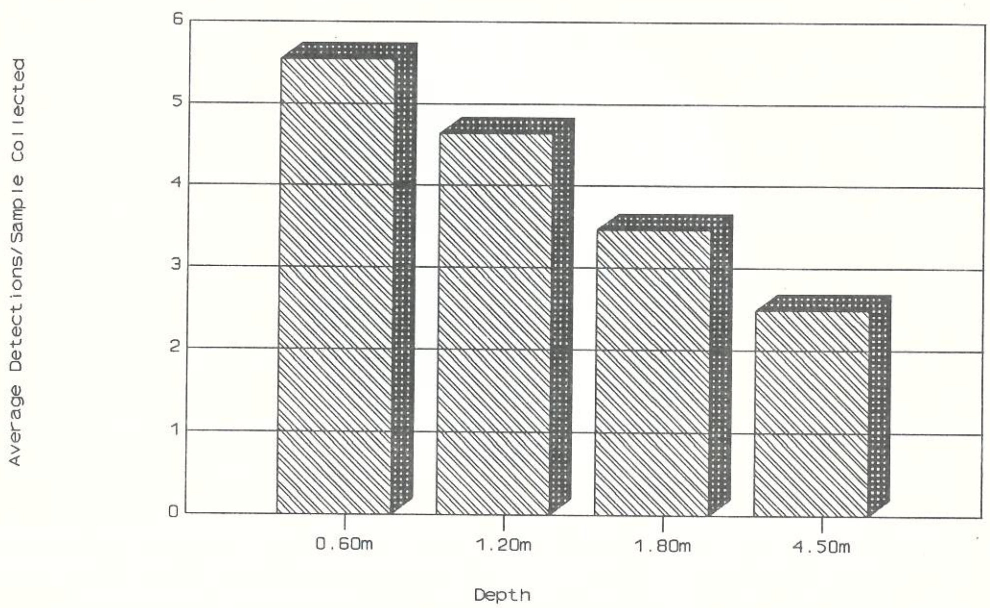


Figure 3-24. Average number of pesticides detected per sample collected by depth. 76 samples; 265 detections; 4 events; 11 pesticides analyzed.

The distribution of pesticide detections with time and by pesticide for all depths (Table 3-5) indicate that fonofos (Dyfonate) and trifluralin (Treflan) were the most frequently detected pesticides. When the deep samples (15 ft.) were removed from the analysis (Table 3-6), the same two pesticides were the most frequently detected. These two pesticides have not been applied for at least three years prior to sample collection. This indicates that pesticides that were thought to degrade quite rapidly in the soil may persist for years. Verification of some of the higher detections using GC/MS is forthcoming.

Figures 3-25 through 3-29 are the distribution of pesticides detected by pesticide for each of the 4 depths and with all depths combined. Figures 3-30 through 3-31 are the numbers of pesticide detections by date for the top three depths and for all depths. The largest number of pesticides detected per sample collected occured before the chemicals were applied. This may indicate that freezing and thawing may release some chemicals tied up on the organic matter into the soluble phase that is subject to leaching.

There appears to be no real difference between tillage systems on the number of pesticides detected at 2 ft. below the land surface (Figure 3-32). There are, however, twice as many pesticide detections at 4 ft. for the NT compared to the MP treatments. The maximum pesticide concentrations for the NT treatment are higher at 4 ft. than for the MP treatment in 7 out of the 10 pesticides tested.

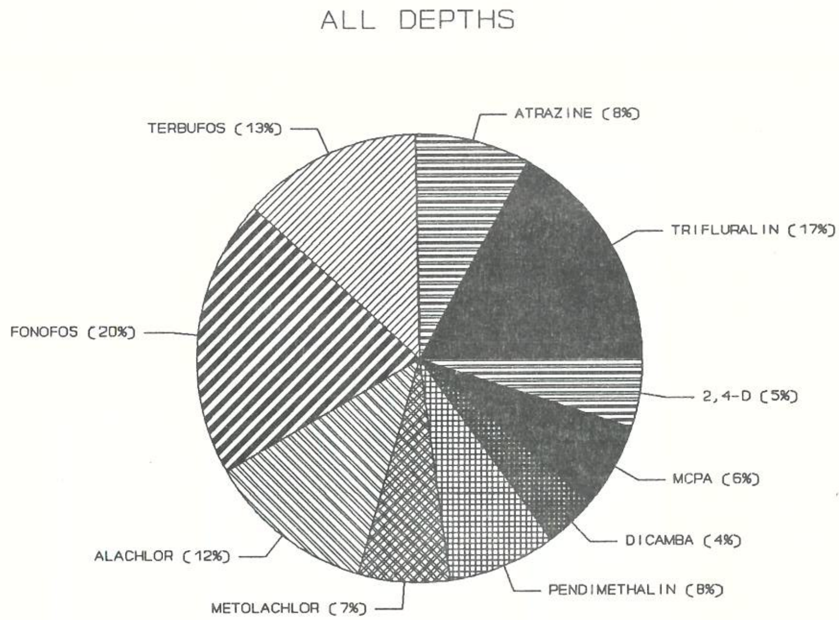


Figure 3-25. Distribution of 265 pesticide detections analyzed from 76 water samples collected from all depths (2, 4, 6, and 15 ft) for 4 leaching events.

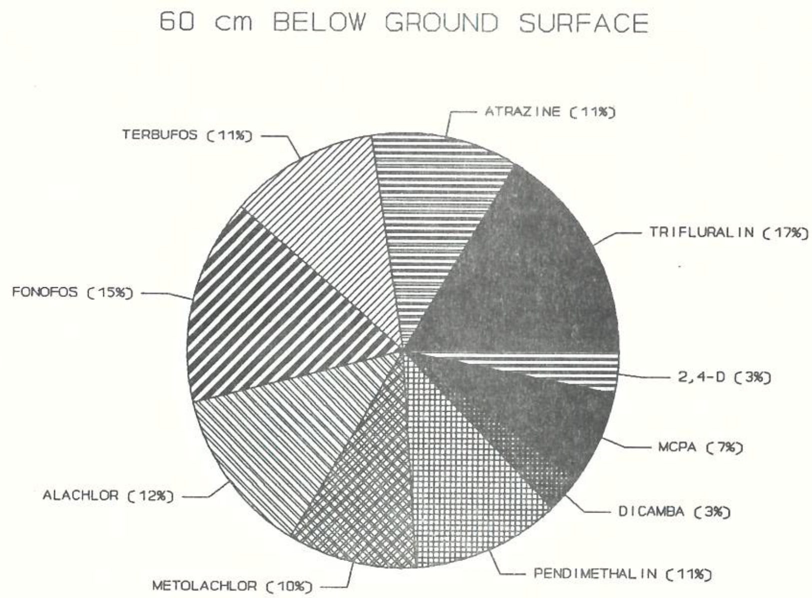


Figure 3-26. Distribution of 72 pesticide detections analyzed from 13 water samples collected 2 ft. below grade from 4 rainfall events (5/4, 6/22, 6/28, 7/12).

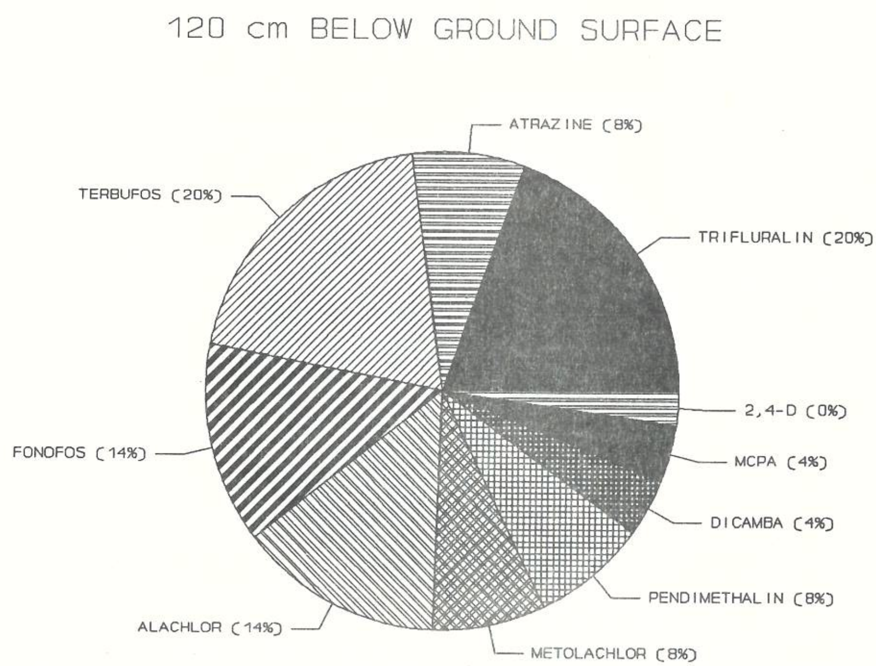


Figure 3-27. Distribution of 51 pesticide detections analyzed from 11 samples collected 4 ft below grade from 4 rainfall events (5/4, 6/22, 6/28, 7/12).

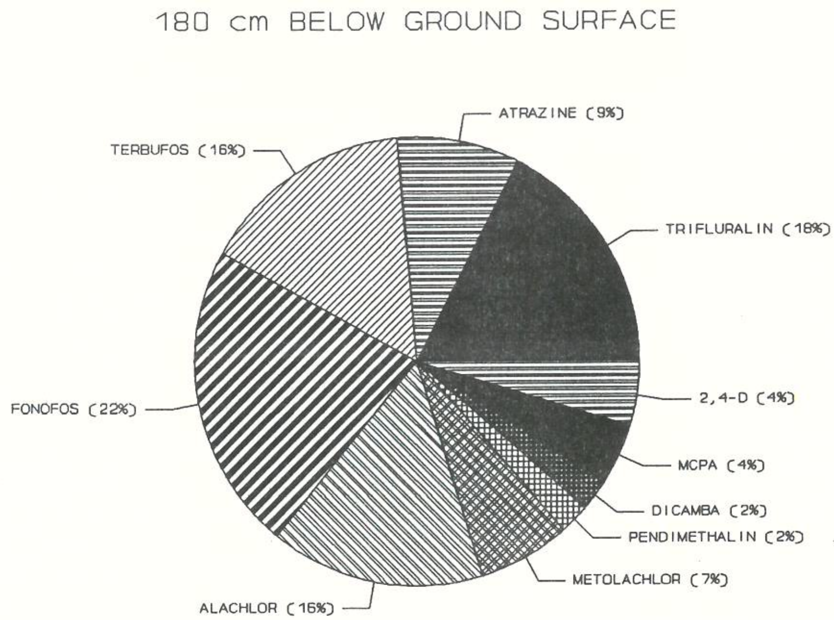


Figure 3-28. Distribution of 45 detections analyzed from 13 samples collected 6 ft below ground surface from 4 rainfall events (5/4, 6/22, 6/28, 7/12).

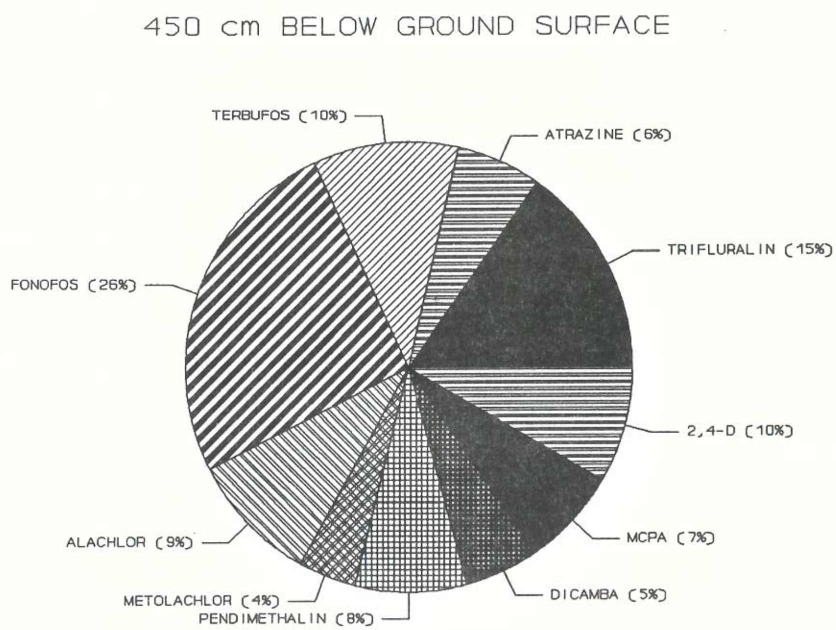


Figure 3-29. Distribution of 97 pesticide detections from 39 water samples collected 15 ft below grade from 4 rainfall events (5/4, 6/22, 6/28, 7/12).

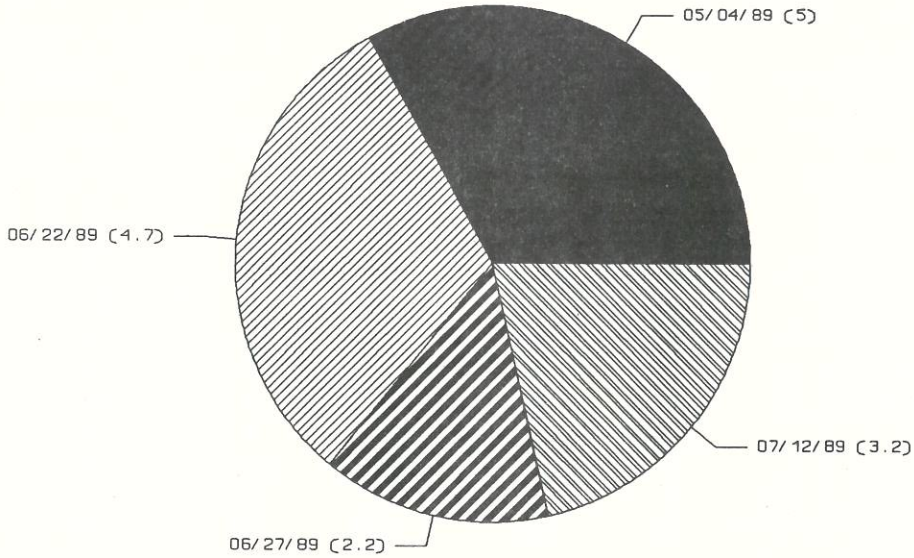


Figure 3-30. Average number of pesticides detected per water sample collected by date from 3 depths (2, 4, & 6 ft).

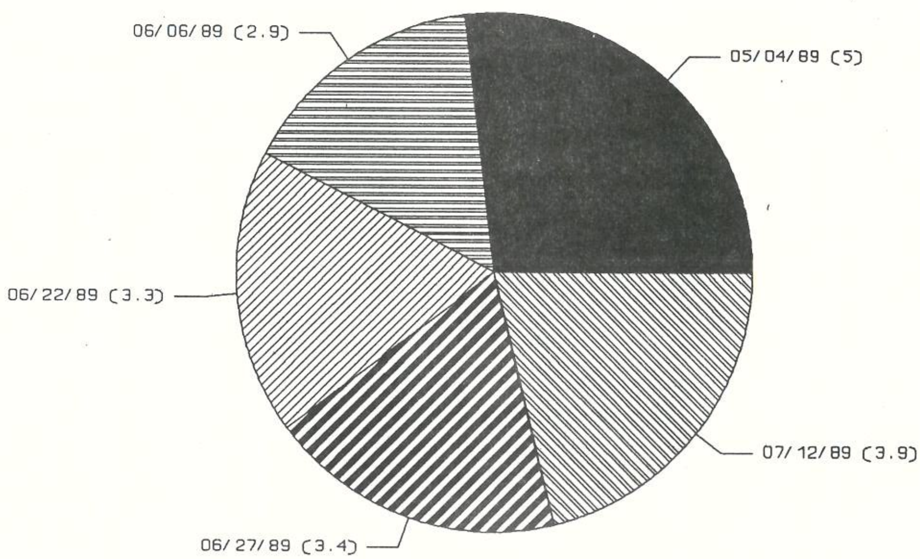


Figure 3-31. Average number of pesticides detected per water sample collected by date for all depths (2, 4, 6, & 15 ft).

Table 3-5. Number of detections of pesticides determined from water samples collected from 4 depths (2 ft., 4 ft., 6 ft., & 15 ft.).

DATE	5/04	06/06	06/22	06/27	07/12	TOTAL
No. of water samples	5	16	16	29	10	76
Treflan	4	6	8	18	9	45
Atrazine	0	1	5	11	5	22
Terbufos	3	3	4	16	9	35
Fonofos	3	10	11	24	5	53
Alachlor	3	5	6	14	4	32
Metolachlor	0	2	5	9	2	18
Pendimethalin	1	0	6	9	5	21
Dicamba	2	5	3	0	0	10
MCPA	5	7	4	0	0	16
2,4-D	4	8	1	0	0	13
TOTAL	25	47	53	101	39	265
Average Detections per sample	5	2.9	3.3	3.5	3.9	

Table 3-6. Number of detections of pesticides determined from water samples collected from 3 depths (2 ft., 4 ft., & 6 ft.).

DATE	05/04	06/06	06/22	06/27	07/12	TOTAL
No. of water samples	5	0	10	29	10	54
Treflan	4	0	6	13	7	30
Atrazine	0	0	4	8	4	16
Terbufos	3	0	4	11	7	25
Fonofos	3	0	9	12	4	28
Alachlor	3	0	6	10	4	23
Metolachlor	0	0	5	7	2	14
Pendimethalin	1	0	5	3	4	13
Dicamba	2	0	3	0	0	5
MCPA	5	0	4	0	0	9
2,4-D	4	0	1	0	0	5
TOTAL	25	0	47	64	32	168
Average detections per sample	5	0	4.7	2.2	3.2	

Table 3-7. Maximum concentrations of pesticides (nanograms per liter) detected by depth for no-till and moldboard Plow tillage treatments.

	60 cm		120 cm		180 cm		450 cm	
	NT	MP	NT	MP	NT	MP	NT	MP
Trifluralin	1224	1650	6702	80	38	41	301	128
Atrazine	1257	7809	1468	0	357	3963	1154	720
Terbufos	348	270	1431	185	69	251	224	168
Fonofos	387	333	328	590	232	1512	134	63
Alachlor	506	2492	1274	431	87	3281	367	86
Metolachlor	709	161	207	0	0	256	189	95
Pendimethalin	910	4604	762	128	0	1725	408	230
Dicamba	278	859	500	259	31	0	226	52
MCPA	529	58	157	440	108	0	236	550
2,4-D	65	708	0	378	40	3759	246	505

There appears to be no real definitive difference between the number of pesticide detections for NT compared to MP tillage (Figure 3-32). The only really interesting difference was that the NT treatment had twice as many detections as the MP for the 4 ft. depth. In addition, the maximum concentrations for the NT were substantially higher than the MP with 7 out of 10 maximum pesticide detections being on the NT treatment. Conversely, at the 6 ft. depth the MP treatment had 8 out of 10 maximum pesticide concentrations. At the present time, there appears to be no real differences that stand out between tillage systems regarding the number of pesticide detections. Further sample collection may be required before anything substantial may be derived from the data.

The pesticide detections were grouped according to date, to determine if the length of time after application may have a bearing on the number of detections. No significant trends were seen.

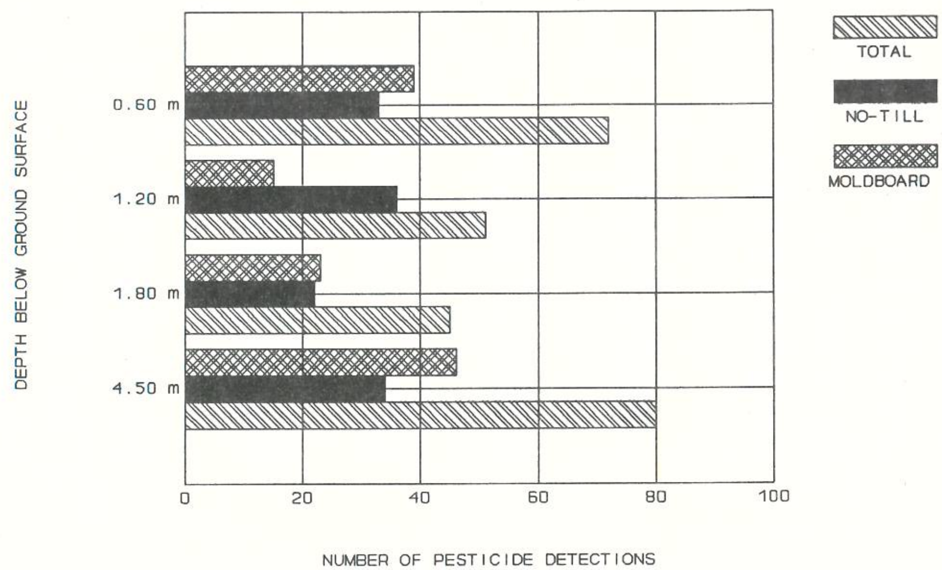


Figure 3-32. Comparison of the number of pesticide detections analyzed by depth for no-till vs. moldboard plow tillage treatments.

3.4 Summary and Conclusions

1) When high amounts of nitrogen fertilizers are broadcast at the soil surface (200 lbs/Ac as N) for corn, higher concentrations of nitrate as N are evident in percolating waters at 4 ft. and 6 ft. below the ground surface for the no-till treatment compared to a moldboard plow treatment.

2) Infiltrating water on a no-tilled field reached 2 ft, 4 ft., and 6 ft. depths below the ground surface from a rainfall-creating-leaching event faster than from a moldboard plowed field for similar soil profile water contents. This is believed to be caused by better macropore development open to the surface on the no-tilled land.

3) Based on data from one growing season, the number of pesticide detections in percolating soil water does not indicate differences or trends between tillage systems consistent with depth.

4) Twenty five percent (25%) of the pesticide detections were from chemicals that had not been applied to the soil for at least three years prior to sample collection. This indicates that residual agricultural chemicals may persist in the soil environment and may be released years later. Verification of these chemicals by gas chromatography/mass spectrophotometer (GC/MS) is forthcoming.

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4.0. NITROGEN-TILLAGE-RESIDUE MANAGEMENT (NTRM) MODEL

Authors: C. G. Carlson
J. H. Bischoff

4.1. Introduction

From the onset of the Oakwood Lakes-Poinsett project, a model was needed to help simulate the effect of the interaction of crop residue, nitrogen fertilizer applications and tillage on potential surface runoff and leaching of chemicals to the ground water for the type of soils in the Central Plains. At that time, NTRM was the most realistic model available that approached this objective.

NTRM, which was developed by ARS researchers in 1979, is a nitrogen, tillage, residue management model developed for tillage, residue, and nitrogen fertilizer managements. It simulates physical, chemical, and biological processes in the soil-water-crop continuum using various integrated submodels.

Denitrification research was done on different crops under varying water contents and temperatures and under different tillage systems at SDSU to provide more data for the model to run. This information was to be used as input into the nitrogen transformation submodels of NTRM to help balance the input and output of nitrogen in the soil. It is now clear that the unsaturated soil water flow submodel does such a poor job of predicting the real world situation, that continuing to apply available data to the model and using it on a watershed basis would only result in an excercise to become more familiar with the model. Subsequently, the denitrification work to relate the effects of crops, tillage, and water content on denitrification rates is good information to have, but it will not be used to input into a large model to simulate the amount of leaching of nitrates is taking place on a watershed basis.

Figure 4-1 is an example of the type of processes simulated by NTRM. Figure 4-2 is a flow diagram of the model and the interactions of various other submodels.

The complex soil-water-nutrient system is largely determined by the saturated and unsaturated flow sub-model. To predict leaching of nutrients, water movement must first be predicted.

Traditional Soil Physics expresses water flow with an energy (h)-volumetric water content (Θ) relationship that is a single-functional relationship (i.e. potential = f(water content)).

Hysteresis of that single-value function due to wetting and drying (Figure 4-3) has been recognized as occurring, but not as significant in explaining water and solute movement in the unsaturated phase.

$$\text{Richard's equation} \quad \frac{\Theta}{t} = -\frac{q}{z} = \frac{-K \quad (h-z)/(z)}{z}$$

is used to explain the change in volumetric water content with time within the unsaturated soil but $h = f(\Theta)$ and $K = f$ (Θ or h) must be known in order to solve this equation. When these relationships are single-valued functions, the process is simple. However, due to the hysteresis involved during wetting and drying, they become multi-valued function relationships between K, h, and θ which are different depending upon whether the soil is drying or wetting and how long since the last infiltration event.

Richard's equation does fairly well in predicting the drainage event when very low potentials are measured, but very little flow occurs at these potentials. The majority of the flow in the soil occurs at high water potentials in the soil. Therefore, there is inflow of water into the soil through macropores and redistribution from the macropores to the micropores. Therefore, the macropores that exist in the soil (larger pores) receive water from a rainfall at a different rate than the micropores. Rather than having plug flow with the water moving in a uniform pattern through the soil with a wetting front, there is a rapid flow of water that fingers through the soil in macropores (fractures, worm holes, and old root channels) and then redistributes to the micropores. Consequently, models based on Richard's equation will never be able to adequately represent the flow of water and solutes until we can formulate mathematical models that more accurately describe the actual processes.

4.2 Methods and Materials

Supporting evidence of the phenomenon discussed above can be shown in figures 4-4 and 4-5. Figure 4-4 shows an infiltration process under a constant intensity of 127 mm/hr for three different plots. All these plots basically behave in identical patterns. The assumption here is that the soil can accept more water at the beginning of the event and after about an hour the soil can take water at a constant maximum rate of around 20-40 mm/hr but no more. Traditionally it was thought that this soil could not receive water any faster than this rate. However, Figure 4-5 shows three plots sprinkled with water with the intensity increased in the middle of the test (after equilibrium conditions were met and the soil "couldn't accept water any faster"). After the rate was increased, the soil accepted water at a faster rate. This suggests that the soil must have two distinct types of pores that can receive water--macropores and micropores. At the low intensity rainfall under the variable rate test, water ran off (overland flow) the plot before it intercepted a macropore to infiltrate, and the infiltration capacity of the macropores was never exceeded; only the micropore infiltration rate was exceeded. When the simulated rain intensity was increased, the macropores accepted some of the water, thus increasing the infiltration rate.

This data is further supported by figures 4-6 and 4-7. Figure 4-6 is a plot of the water potentials in the soil 2' below the ground surface for two tillage systems--no-till and moldboard plow treatments. There are two rainfall events that occur for this time interval--a 16 mm and a 18 mm event. There appears to be a more rapid movement of water through the soil reaching the 2' depth for the NT treatment relative to the MP treatment. The NT treatment shows the water potential as becoming wetter at that depth for the first rainfall event and then a redistribution and/or draining event at different rates compared to the MP treatment. There had been little rainfall prior to the first 16mm rainfall; consequently, the macropores were empty, the soil was drier nearer the soil surface, and the macropores were, consequently, larger due to the shrinkage that occurs during the drying process. Approximately two days later there was a rainfall event of the same amount but at a lower intensity. The differences of the movement of water to the same point are not nearly so evident. A microscopic view is portrayed in figures 4-8 and 4-9. As the rain enters the soil, the macropores fill, but little water has moved into the micro-interstices. After the rainfall event has occurred, the water moves by capillary action from the macro to the micropores. The intensity and duration of the event coupled with the quantity of macropores in the soil profile will determine the depth of penetration of the water to the subsoil.

The rate of water movement to the subsoil can be measured by monitoring the change in matric potential with time during and after a leaching event. Figure 4-7 is a comparison of the rate of movement of water six feet below the ground surface between NT and MP treated plots. From two rainfall events within two days of one another (16mm and 18mm), the draining taking place appears to be at a more rapid rate for the NT treatment, as is indicated by the steeper slope.

When high rates of ammonium nitrate are broadcast on the soil surface, there are also differences in the concentrations of nitrates that leach to different depths based on type of tillage. The fertilizer was dropped on with a drop spreader for both tillage treatments, but was incorporated with a spike-tooth harrow on the MP treatment. It was found that, consistently, in two of three replications, the concentration of nitrates was lower at 2' and higher at 4' and 6' for the NT treatment compared to the MP treatment (Figure 4-10). This was found to be consistent in the latest five leaching events monitored from May through July. This seems to indicate that either 1) the way the fertilizer is incorporated or not incorporated, or 2) the type of tillage used are very important in minimizing the movement of nitrates to the subsoil environment. Which method to use to effectively manage our fertilizer resource is still an unknown.

Consequently, which model to use to explain the flow of water in the unsaturated zone is still under investigation. It is really fascinating to note that the mechanism of water flow in unsaturated soils is really now approaching fuller understanding, all because of the concern for subsurface water quality. In years past, the most improtant understanding was how to get a certain amount of water to a certain soil depth to prevent plant stress and yield declines. Now a fuller understanding of solute movement as the water infiltrates the soil is of prime concern to protect the domestic ground waters from contamination not only from fertilizers, but also from agriculturally applied chemicals.

4.3 Results and Discussion

The best consistent fit of a model using the traditional K vs. Θ and Θ vs. h relationship with realistic and believable numbers is quite aways off, to say the least. Volumetric water contents were measured continuously at the one foot (30 cm) depth using a neutron scattering probe. A sprinkler infiltrometer was used to apply water at a continuous and uniform intensity on a one meter plot. The water was then stopped for a few hours and repeated again (Figure 4-11) and stopped again. This 3000 minute experiment was then modeled using an iterative solution for minimizing the sum of squares for the entire 3000 minutes using the measured values as the true values. The fit of the model compared to the actual event was not very close when traditional K, Θ, and h values were used.

Then, another iterative solution was tried with no maximum values used for K, θ, h, or Ksat (saturated hydraulic conductivity) (Figure 4-12). The fit of the model values compared to the measured values are very close and apparently very accurately represent what is taking place in the soil. On the other hand, the h vs. θ relationship is unrealistic in traditional theory and the hydraulic conductivity value exceeds 8000 feet per minute. What is more disturbing than this is that for the same soil and experiment, by using the same K, θ, and h relationships, the same type of accuracy cannot be predicted. It appears that there are differing K, θ, and h relationships for every set of circumstances when measurements are made.

What can we conclude from all this? First, infiltration and redistribution are different events that use totally different flow paths. These flow paths may change as often as daily. These two flows may have to be modeled as separate events in order to make predictions with any kind of accuracy.

Secondly, in the Central Plains Region (especially native prairie and no-till tillage systems), plug flow is not the dominant flow during infiltration events. Considerable bypass preferential flow occurs which is not accurately described by Richard's equation when using traditionally known moisture release curves and hydraulic conductivities.

Thirdly, the kinetics of the chemical equilibrium between the bypass flow and the soil solution must be further investigated in order to fully understand the management alternatives available to minimize the risk for contaminant transport through the unsaturated zone.

Before these models can predict the water movement with any kind of reliability, the soil-water physics need to be more completely understood. In the meantime, any model (including NTRM) which uses a traditional method of modeling unsaturated soil water flow, will consistently give values that don't truly represent the real world.

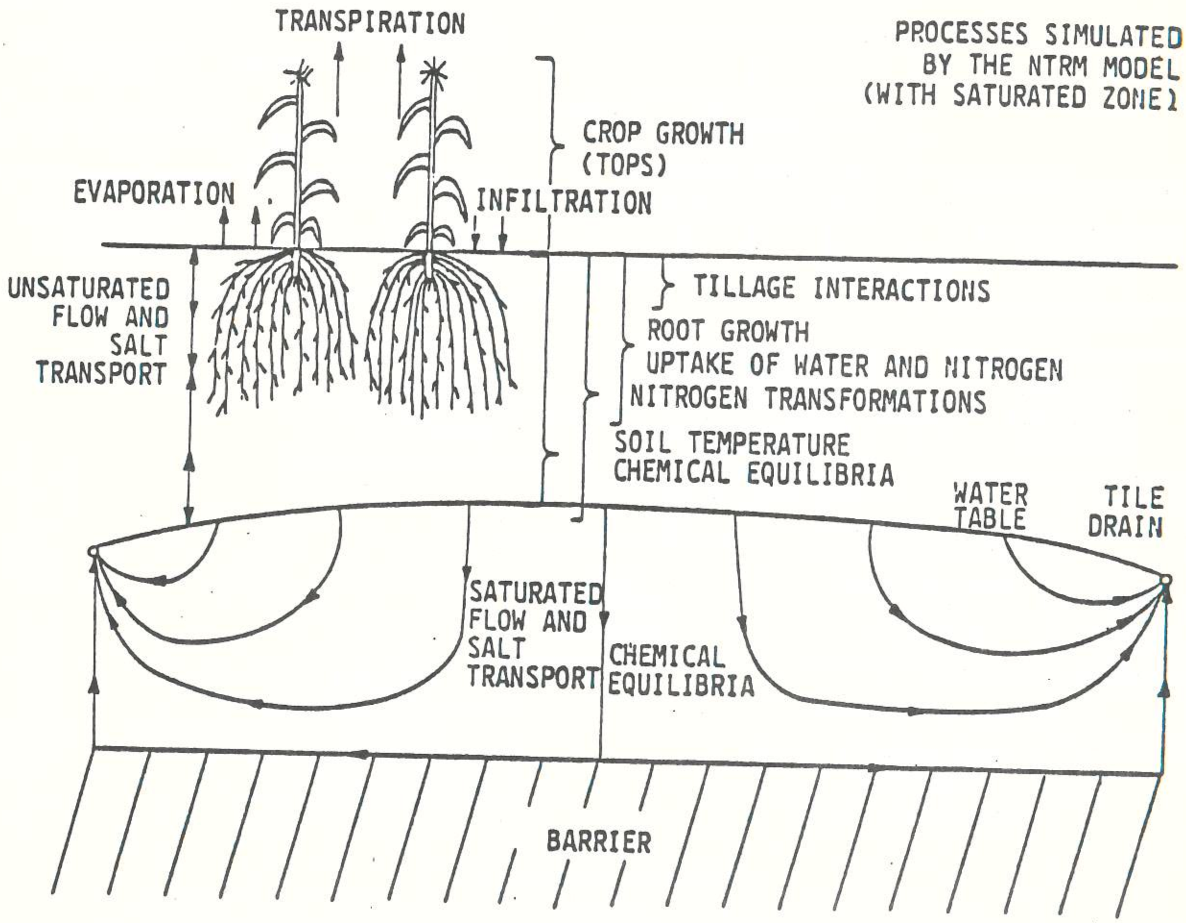


Figure 4-1. Processes simulated by the NTRM model (with saturated zone).

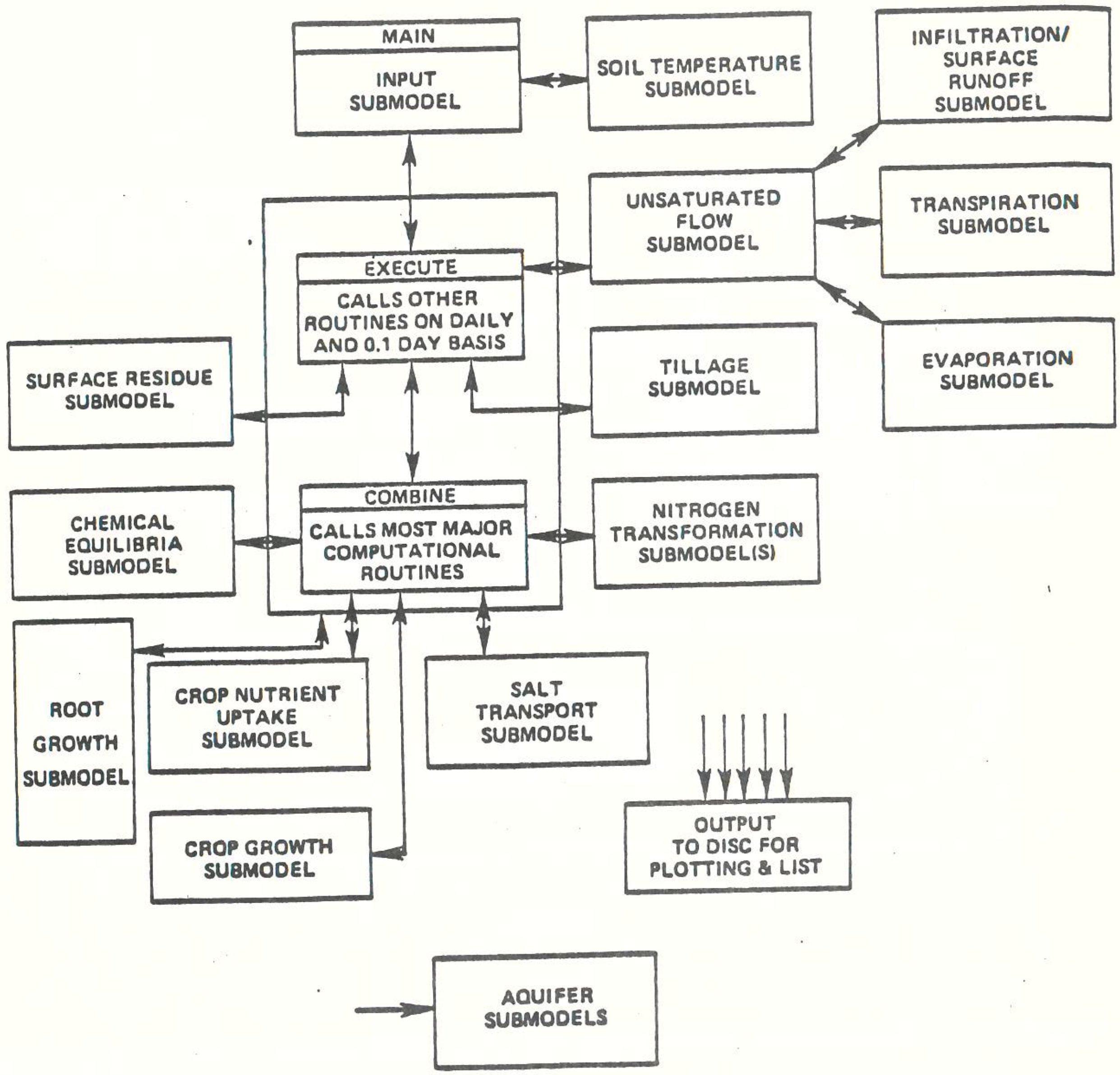


Figure 4-2. Nitrogen-Tillage-Residue-Management (NTRM) Model.

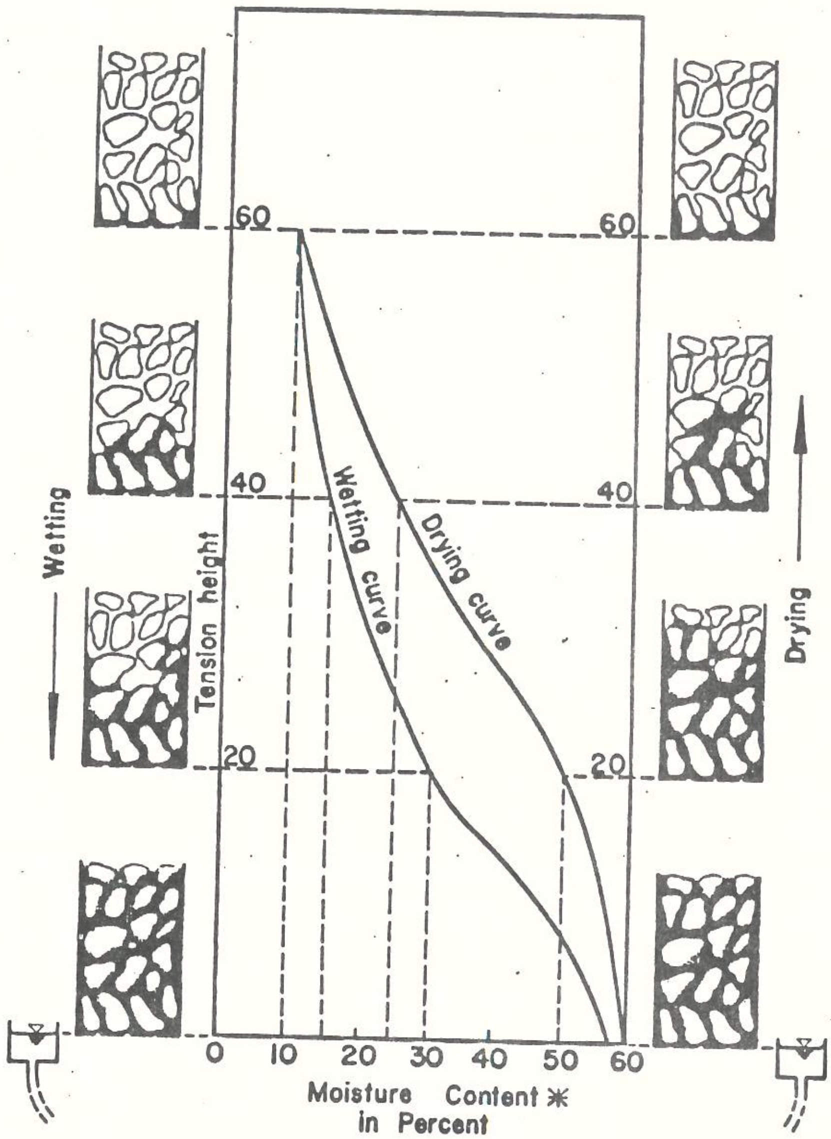


Figure 4-3. Hysteresis effect in Soils subjected to Wetting and Drying
Soil Physics Agronomy 577
M. van Rooyen
Fall, '55

✳ For drawing purposes sp. gr. of soil solids is taken as 2.3.

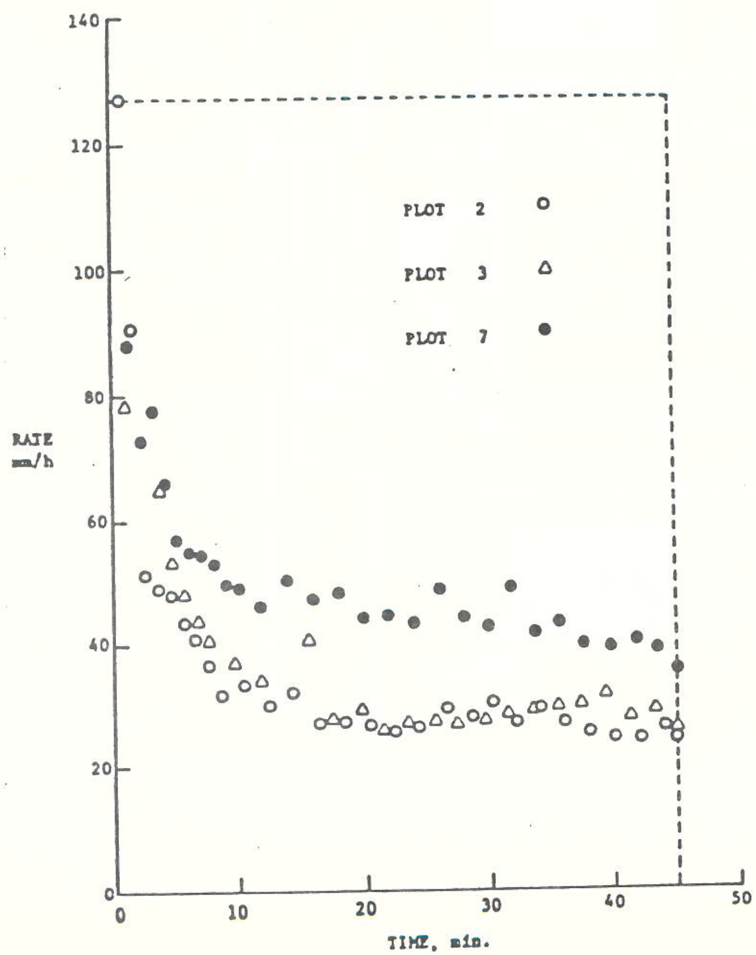


Figure 4-4. INFILTRATION UNDER RAIN INTENSITY OF 127.0 mm/h.

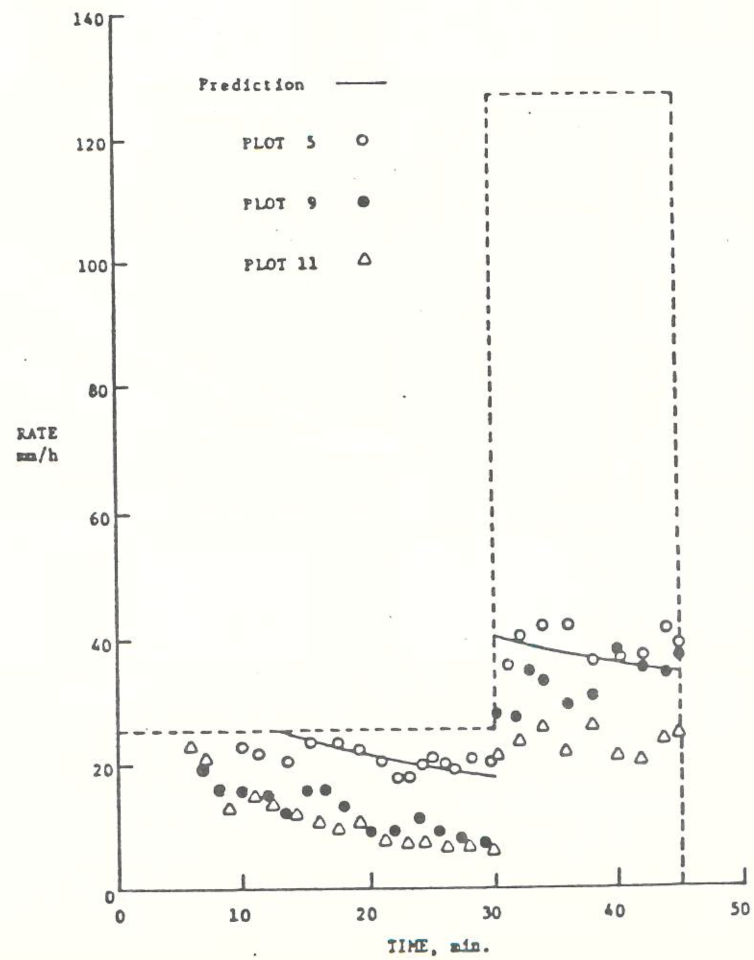


Figure 4-5 INFILTRATION UNDER A VARIABLE RAIN.

WATER PENTRATION BETWEEN TILLAGE SYSTEM

OATS - APRIL 1989 (2 FT.)

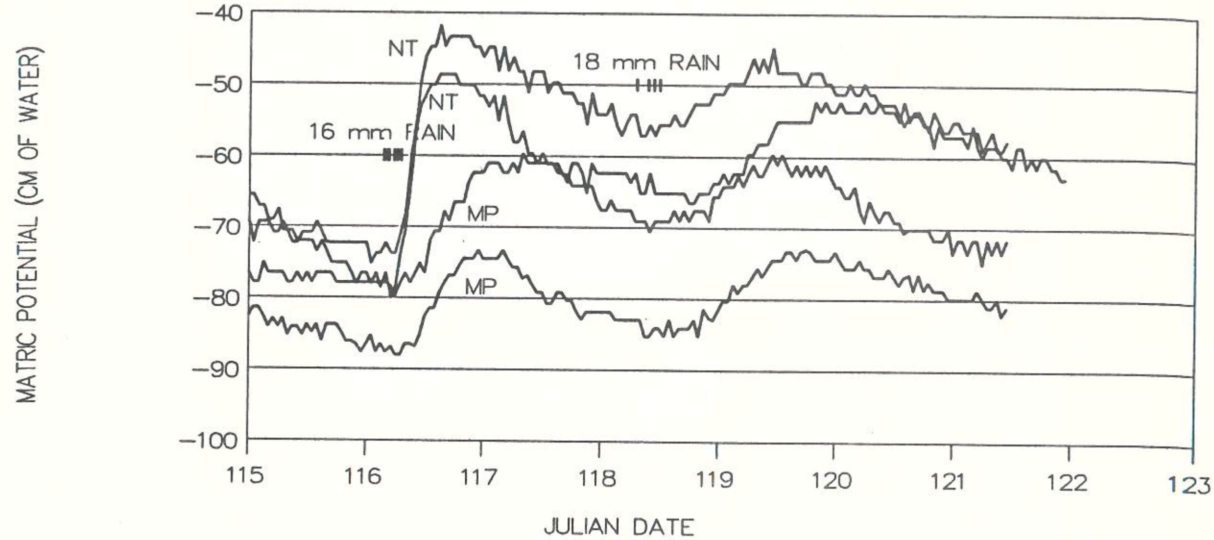


Figure 4-6

MASTER SITE 6 FT TRANSIOMETER DATA

OATS - APRIL 1989

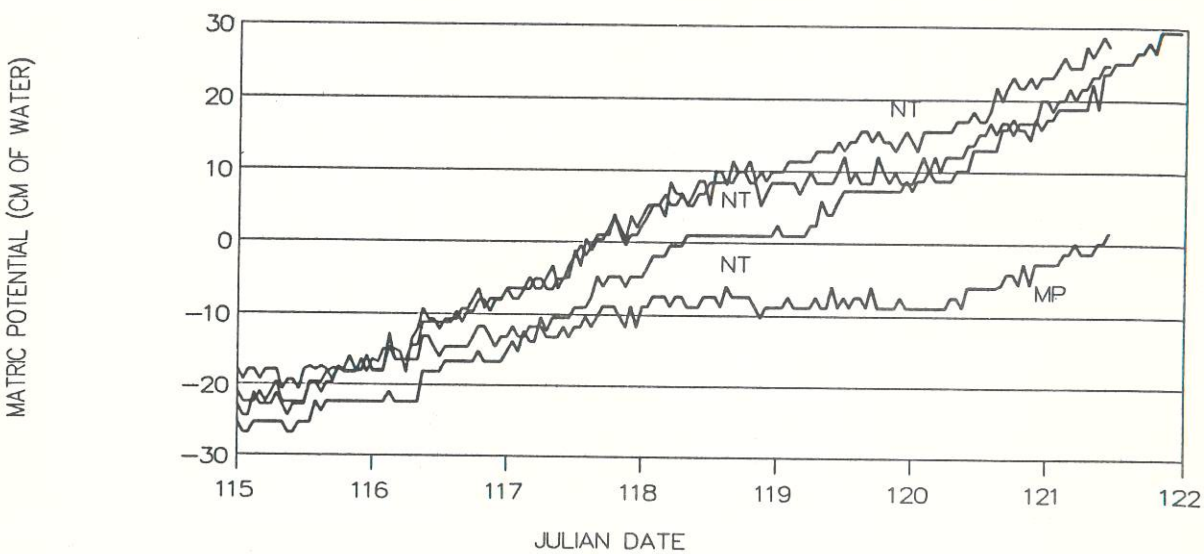
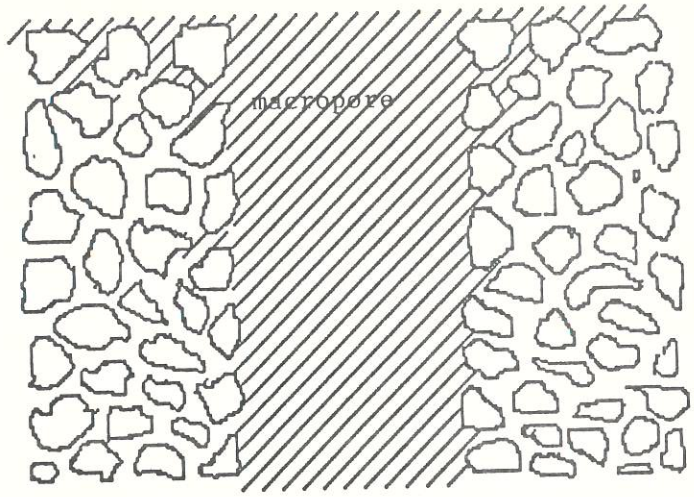
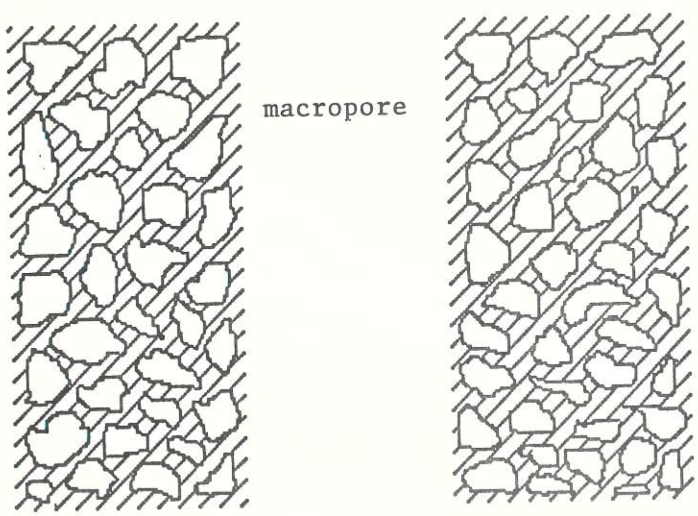


Figure 4-7



$$\Theta_v = 20\%$$

Figure 4-8. **Typical water distribution in the soil during a precipitation event.**



$$\Theta_v = 20\%$$

Figure 4-9. **The distribution of water in the soil profile 3-7 hours after precipitation event.**

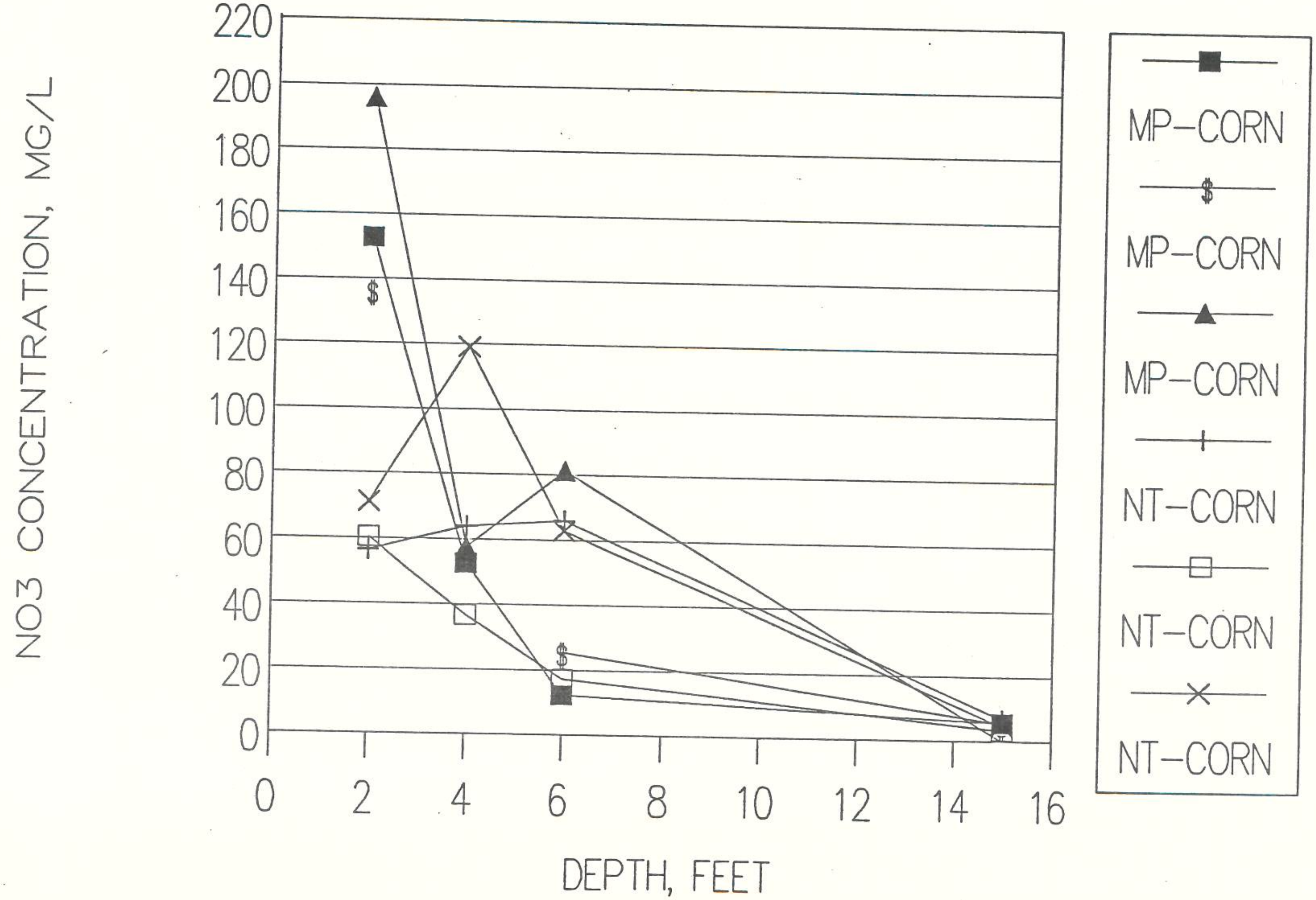


Figure 4-10.Tillage effects on nitrate concentration by depth for a rainfall event on Julv 12. 1989.

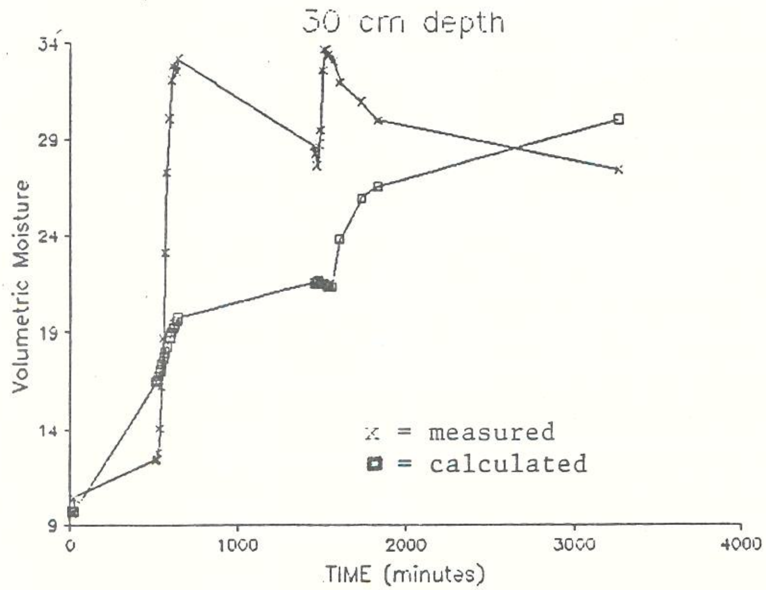


Figure 4-11 Best fit of model using Richard's equation with traditional limited values of K, θ, and h.

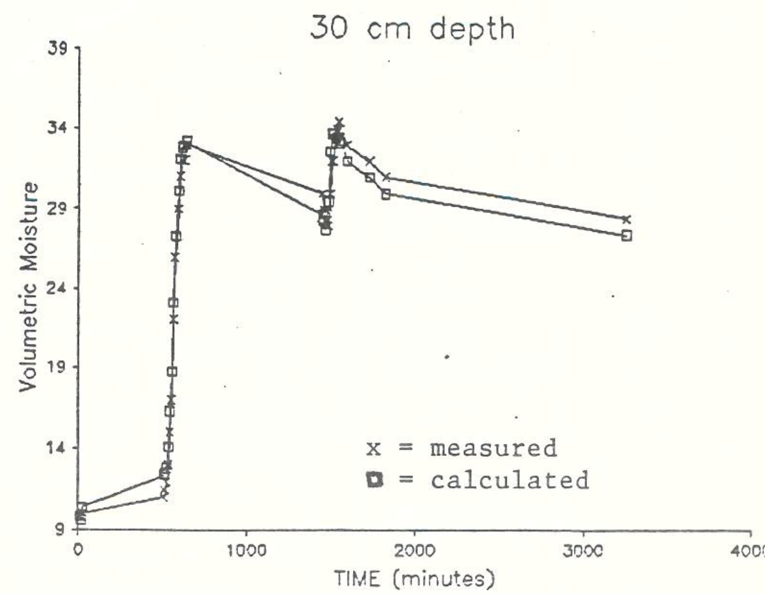


Figure 4-12. Best fit of model allowing K, θ, and h to float with no maximum limits.

5.0 ANALYSIS AND EVALUATION OF THE OAKWOOD LAKES SYSTEM

Authors: David R. German, Mary B. Price and Steve Grosz (surface and ground water quality), Dr. Charles R. Berry Jr. and Rich Brown (fisheries) and Dr. Walter Duffey (zooplankton).

5.1 Introduction

The Oakwood Lakes System Study (OLSS) began in 1987 as an expansion of the Comprehensive Monitoring and Evaluation (CM&E) project. The goal of the OLSS project is to determine if the application of BMPs in agricultural watersheds will affect water quality in shallow, hypereutrophic prairie lakes. To meet this goal it is necessary to collect data on the current trophic state of the lake system, the flux of water and nutrients through the system and various chemical, biological and physical interactions that affect in-lake water quality. Development of annual material budgets for the Oakwood Lakes system and monitoring of in-lake water quality, algae, zooplankton, and fish populations was implemented to meet this goal.

This report represents the current state of data analysis for each OLSS objective. Comprehensive analysis and modeling to describe the system as a whole was scheduled for 1989, but extension of monitoring activities to gain additional runoff data delayed that activity until 1990. A draft report of OLSS data analysis will be prepared by September 30, 1990, with a final report in December 1990.

5.1.1 OLSS Monitoring Strategy

Monitoring before and after BMP implementation to determine water quality effects was inappropriate for OLSS since monitoring was originally scheduled for only two years, and RCWP installation of BMPs preceded the OLSS by several years. A strategy was developed which incorporated water quality and quantity monitoring, biological surveys, land use modeling and, lake modeling to evaluate BMP impacts. The strategy relies upon the development of nutrient, sediment, and hydrologic budgets for the Oakwood Lakes system. These budgets were to be compared with reductions in loadings estimated by watershed models following verification of the Agricultural Non-point Source Pollution (AGNPS) model. Evaluation of factors that predict in-lake water quality and the use of input-output lake models to estimate the potential for BMPs to improve trophic state completes the strategy.

Weather is a large factor in determining both quality and quantity of runoff and determining in-lake water quality. Therefore, weather parameters were monitored to sort out their effect on water quality so effects of BMP implementation could be assessed more accurately.

To estimate the effects of BMP implementation on in-lake water quality, it is necessary to document the current trophic state of the lake and account for external nutrient and sediment loadings from surface water, ground water and atmospheric sources. These material budgets can be used in conjunction with physical characteristics of a lake (surface area and volume), to develop a mass balance equation which can be expressed as:

(accumulation) = (inputs) - (outputs) + (reactions in the lake + or -)

Residence times can also be calculated. These can indicate the potential for changes in the trophic state of the lake, the degree to which the lake is acting as a nutrient sink (or possible source), the potential for improving water quality through BMP implementation, and the time that may be required to pass before changes are measurable.

The primary purpose for collecting tributary water quality data is to calculate the surface water component of nutrient, sediment, and hydraulic budgets for the Oakwood Lakes. The data may also be used to identify water quality problems in watersheds. Tributary water quality is described by evaluating several significant parameters that indicate possible pollution sources in the watershed that impact the receiving water. Variations in the parameters for each tributary are due to watershed characteristics, weather conditions and time of year.

The primary purpose of collecting ground water quality data for the OLSS is calculating ground water contributions to the hydraulic, nutrient, and sediment budgets for the Oakwood Lakes. The data is also useful to determine the extent of quality similarities between ground water surrounding the lake and ground water collected from in-lake wells and seepage meters.

Monitoring the abundance of phytoplankton populations in the Oakwood Lakes system is to their effect on water quality. Since excess phytoplankton are a major cause of water quality degradation in the Oakwood Lakes system, phytoplankton abundance needs to be known to evaluate the success of any management plan. Zooplankton graze on the phytoplankton and lower their abundance, increasing the clarity of the water. Zooplankton abundance must to monitored to estimate what fraction of the changes in phytoplankton numbers can be attributable to zooplankton grazing as opposed to changes in nutrient loading.

The purpose of monitoring the relative abundance and diet of fish in East and West Oakwood Lakes is to indicate the possible relation between the biological community in the Oakwood Lakes and the water quality, i.e. consumer control of lake productivity (Carpenter and Kitchell 1988).

5.1.2 Study Area

The Oakwood Lakes system is a complex of 5 small, interconnected, shallow, hypereutrophic lakes in eastern South Dakota. The lakes have a mean depth of 6 feet, a maximum depth of approximately 10 feet, a combined surface area of 2,082 acres and a total watershed area of 31,600 acres. The lakes are fed by several intermittent streams which drain a predominantly agricultural watershed estimated to be 50% cropland. The lake system has a single outflow from East Oakwood to the Big Sioux River through Mill Creek.

Non-bedrock aquifers in eastern South Dakota consist of glacial outwash deposits resulting from the Pleistocene Epoch glaciation. Within the project area surficial deposits are of the Wisconsin stage of glaciation, specifically the Cary and Iowan substage till, loess, and outwash. The outwash of greatest thickness is the Cary outwash. Morphology of the outwash is that of a valley train outwash which forms along the margin of a glacier as it melts and recedes. This results in long, relatively thin sinuous deposits of sand and gravel of variable thickness and aerial extent. There are three outwash aquifer systems in the area, referred to as the surficial, intermediate and basal aquifers. Connections between these aquifers occur but are not reliably known. The Big Sioux aquifer is the surficial aquifer associated with the Big Sioux River and tributaries and is of greatest importance in the project area. The Oakwood Lakes-Poinsett project is not directly concerned with the other two aquifers of greater depth.

5.2 Methodology

5.2.1 Weather and Atmospheric Deposition

Solar radiation, wind direction and speed were collected at the Master Site weather station and supplemented with data from the Brookings weather station. Rain quantities were measured with instruments at Castlewood, SD. Rain intensities and quantities were also measured at the Master Site and near Oakwood Lakes with two Belfort recording rain gages.

Rainfall data were collected by two Belfort recording rain gauges installed near the lake basins and three volunteer observers at three sites. The Belfort gauges provided rain intensity information as well as rainfall amount. A telephone survey of farmers in the watershed area was made following the two largest rain events in 1989, to map the distribution of rainfall amounts for use in testing the AGNPS model.

Material inputs to the lake budget from atmospheric sources were measured by collecting combined wetfall and dryfall samples following rainfall events. Samples were analyzed for the following parameters:

1. total PO_4-P
2. nitrate
3. nitrite
4. ammonia
5. total Kjeldahl nitrogen
6. pH
7. total dissolved solids

5.2.2 Surface Water

Instrumentation for surface water movement through the Oakwood Lakes system consisted of 13 stage recorders (Leupold and Stevens model F) and stilling wells located at ten sites; six were located at tributaries entering the lake system (T-1 through T-6), three at sites between the lake basins (IL-1 through IL-3) and one (T-0) at the lake outlet (Figure 5-1). Converting the continuous record of stage into discharge volume for each station involved several steps. First, in-field measurements of flow velocity were measured with a portable flow meter (Montedoro Whitney model PVM-2A) and combined with channel (or culvert) area to determine discharge for discreet points on the stage record. The result was an empirical stage-to-discharge relationship. The empirical stage-to-discharge relationship is simply a listing of stage heights and the corresponding discharges for various dates and times. To adequately describe the relationship the listing should represent the full range of stage heights observed for each tributary. These pairs of stage and discharge values are then described by a discharge equation which allows the calculation of discharge for any stage, measured manually or with the stage recorder. OLSS stage recorders produced an analog record of stage which was digitized into two hour segments. The discharge equations were used to convert the digital stage record to a digital record of discharges. In the final step, stream discharge is combined with water quality data to determine material loadings for various periods of time the stream was flowing.

At sites with a stable control (where discharge is only a function of channel shape or culvert size) a logarithmic equation was developed to describe each empirical stage to discharge relationship. The equations that best described the curve as a whole did not always give accurate results near zero flow. They often produced slightly negative discharges. When this occurred, a table of discharge versus stage heights was constructed with points from the low end of the empirical stage-to-discharge plot. This table or the appropriate equation (depending on the stage) was then used within a spreadsheet program to select appropriate discharge values for each stage reading.

Spring snow storms caused disruption in some stage records and shifts in the stage to discharge relationship due to ice for other portions of the record in both 1988 and 1989. Direct measurement of flows and integration between flow points was used to calculate portions of the loadings where stage record was not available.

At three sites between the lake basins (IL-1,2,3) (Figure 5-1) water passes through corrugated steel pipe-arch culverts which were partially submerged by the lakes at either end. This situation requires the development of a complex equation to account for the effects of backwater on the down stream end of the culverts. The Manning formula can be used when the culvert is flowing partially full. The formula relates the average flow velocity V, the channel slope S, the channel roughness n and the hydraulic Rh of the channel cross section (Olson 1967).

$$V = (1.49/n)\ Rh2/3\ S1/2$$

and since discharge = velocity * area or $Q = VA$

$$Q = 1.49/n\ Rh2/3\ S1/2\ A$$

With the exception of the roughness coefficient (n), the information required to calculate discharge with the formula can be determined from stage records. Channel slope S is represented by slope of the water surface in partially

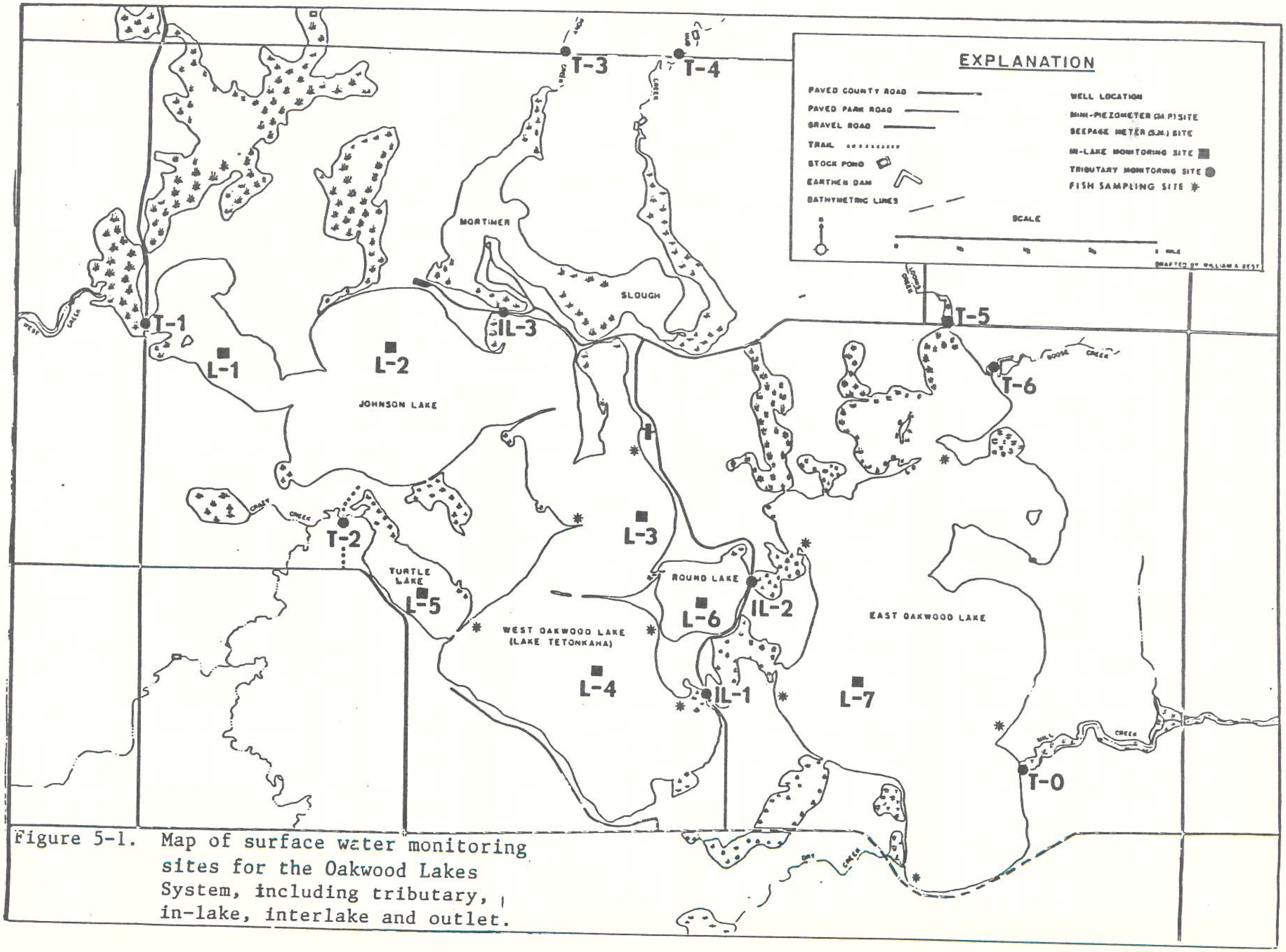


Figure 5-1. Map of surface water monitoring sites for the Oakwood Lakes System, including tributary, in-lake, interlake and outlet.

submerged culverts. Wetted cross sectional area and hydraulic radius were derived from stage height record and a survey of the culvert.

Solving the formula for n, using known values for Q, indicated that n was not a constant but varied depending on depth of water in the culvert. Although the culverts were all constructed of corrugated steel each had mud, sand and gravel or small stones lining the bottom. These factors caused n to vary depending on the amount of wetted perimeter affected. Regression analysis was used to develop an empirical relationship between n and other factors in the Manning formula. Manning's n in most culverts was found to have an inverse relationship with A or Rh2/3. To calculate discharge at the interlake sites, n was predicted for each point on the digital stage record and used in the Manning equation with the other parameters determined from the stage record.

To determine the hydraulic, nutrient and suspended sediment loading for each tributary, the digital discharge record was divided into time intervals determined by the date and time when a sample had been taken. The water quality sample was located at the midpoint of the time interval. This method is patterned after the "trapezoidal rule" proposed by Huber, et al. (1979) for the calculation of event mean concentrations. The material load (kg of each constituent) was calculated by multiplying the volume of water (liters) by the sample concentration (ppm), for each interval and summing over the period of interest (kg/year, kg/month, etc.).

Frequency of sample collection varied and was determined by hydrologic activity. During periods of base flow, collection was weekly for all inlet tributaries and three times a week for the interlake sites and the outlet. During periods of active snowmelt, all sites were sampled once to twice daily. All samples collected during snowmelt and base flow were grab samples. Flow actuated automatic samplers (ISCO) were kept in a state of readiness throughout the ice free seasons in 1988 at all tributaries to collect rain induced runoff.

All samples were collected in polyethylene bottles and transported from the field on ice in a cooler and refrigerated upon arrival at the laboratory. Analysis of most parameters was conducted within 24 hours of collection following EPA approved methods.

Parameters analyzed on tributary and interlake samples included:

1. total PO_4-P
2. ortho PO_4-P (soluble reactive P)
3. nitrate
4. nitrite
5. ammonia
6. total Kjeldahl nitrogen
7. suspended solids

In-lake monitoring was designed to determine current trophic state. Monitoring of zooplankton and fish populations, nutrient parameters and weather produced data for use in a regression analysis to determine factors that best explain variability in dependent variables such as chlorophyll a, total PO_4, algal biomass and Secchi disc transparency. This empirical lake model will be used to determine the role of nutrients and other factors in controlling in-lake water quality.

In-lake water quality samples were collected at seven sites (Figure 5-1). Samples were collected with a project designed, integrated sampler. The sampler collected a column of water from the lake surface to within one foot of the bottom. The sample was poured into a clean plastic bucket to ensure complete mixing of the water column from which subsamples for water quality and biological analysis were taken. For reasons of statistical analysis, a second (independent) cast of the integrated sampler and a second set of subsamples were taken at each location. Secchi disc depths were determined at all sampling locations using a standard black/white 25 cm Secchi disc suspended on a 0.01-foot interval calibrated line. In-situ measurements of dissolved oxygen (DO) and temperature were taken using a dissolved oxygen probe calibrated by the Winkler DO method. Measurements were taken six inches below the water surface and six inches above the lake bottom. If a difference greater than 1 ppm of DO existed between the surface and the bottom a vertical DO/temperature profile was conducted at 3 foot intervals or less.

Samples were placed in 1 liter polyethylene bottles, kept in a cooler until delivery to the laboratory and refrigerated thereafter. Chemical analysis of unstable parameters was conducted within 24 hours of collection. Chlorophyll a analysis began immediately upon arrival in the laboratory. In-lake water quality samples were collected every two weeks from May through October and monthly from November through April. The following parameters were analyzed:

1. total PO_4-P
2. ortho PO_4-P (soluble reactive)
3. nitrate
4. nitrite
5. ammonia
6. total Kjeldahl nitrogen
7. pH
8. chlorophyll "a"
9. algal density (cell counts)
10. suspended solids
11. alkalinity
12. total solids

5.2.3 Ground Water

Instrumentation for ground water inputs includes terrestrial wells (as opposed to in-lake wells to be discussed later), in-lake wells and seepage meters. There were 18 existing monitoring wells (15 on CM&E Field sites) applicable to OLSS projects goals. To complete the coverage of the lakes, 26 additional ground water monitoring wells were installed in November 1987. This brought to 44 the total number of wells available for determining of flow and ground water chemistries. Wells were located in the sand and gravel areas surrounding the lakes to determine horizontal and vertical gradients and chemical stratification (Figure 5-2).

The wells were installed through an 8 inch inside diameter hollow stem auger. Two and a half (2.5) foot split spoon samples at 5 ft intervals were taken and logging was done by a geologist during drilling. Wells were installed approximately 5 to 10 ft below the water table when the water bearing strata was sand and gravel. At many well locations (Figure 5-2) more than one well was installed (referred to as a well nest). The second well, or in some cases the second and third wells, at a well nest was installed at a different depth to allow the calculation of vertical gradients, and sampling of water quality at different depths. The wells were constructed of 2 inch diameter polyvinyl chloride (PVC) pipe coupled (not glued) to 0.018 inch manufactured well screen. Sand and gravel was allowed to collapse around the well screen during withdrawal of the auger to form a natural gravel pack. When collapsing did not occur, pea gravel was installed as a gravel pack. If the borehole penetrated a confining layer, 2 to 3 feet of bentonite was placed in this interval during auger withdrawal to create a seal between the well screen and upper saturated layers. A 3-foot bentonite seal was always placed in the last three feet of hole around the well.

Wells were sampled every two months and water levels were recorded weekly throughout 1988 and 1989. Sampling techniques were the same as those used for the field site ground water investigation (section 2.0). The parameters analyzed on ground water samples were:

1. total dissolved PO_4-P
2. ortho PO_4-P (soluble reactive)
3. nitrate
4. nitrite
5. ammonia
6. total Kjeldahl nitrogen
7. total dissolved solids
8. chloride
9. sulfate

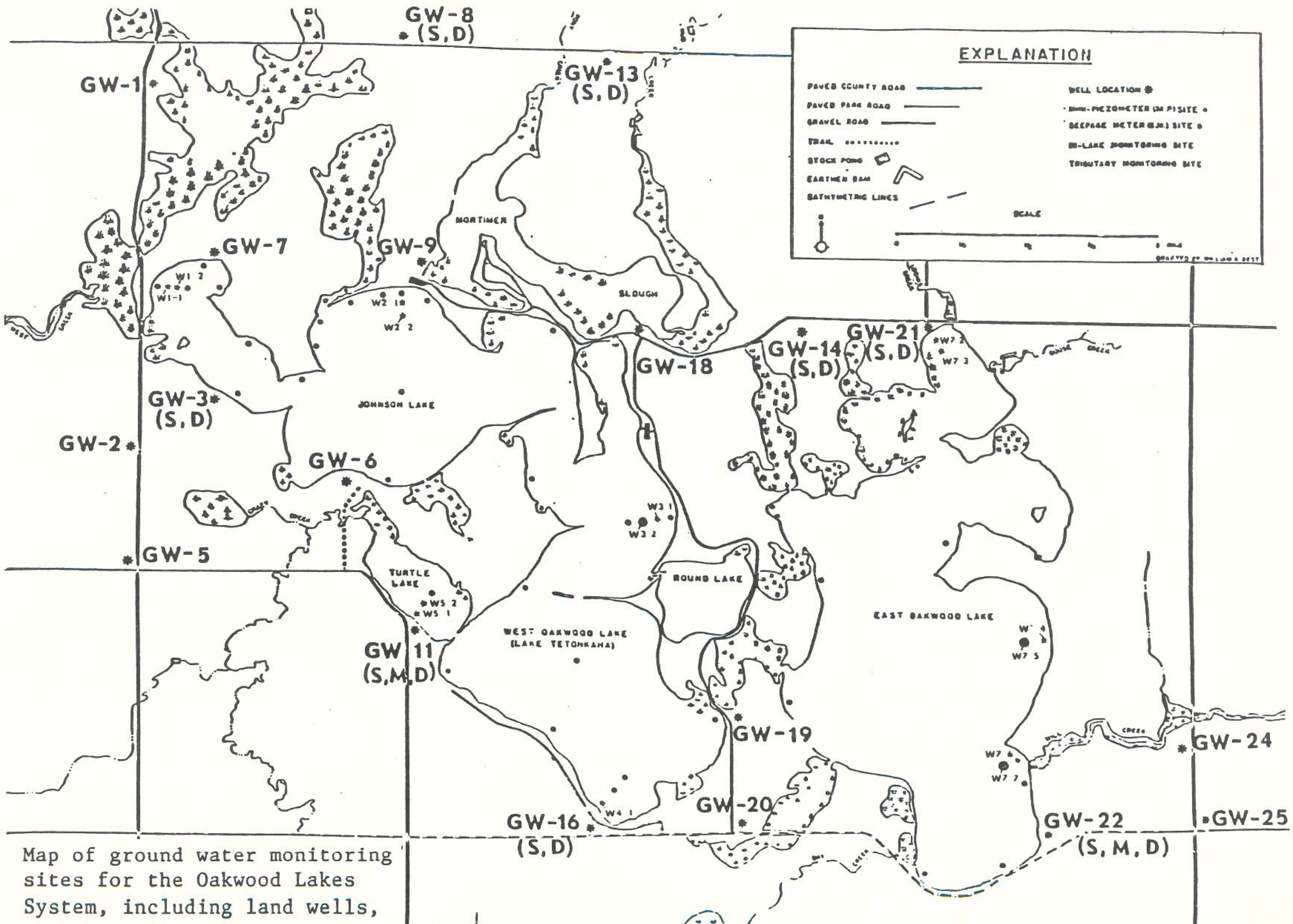


Figure 5-2. Map of ground water monitoring sites for the Oakwood Lakes System, including land wells, in-lake wells, seepage meters and sediment core sites.

In-lake wells were placed into the lake bottom in 1987, to allow ground water level readings beneath the lake. The OLSS in-lake wells were constructed of one-inch diameter polyvinyl chloride pipe with 6 inches of slots at the bottom, wrapped with fiberglass cloth (epoxied in place) for a screen. Installation was accomplished by pounding a steel tube with a drive point on the end, into the lake bottom, to the desired depth. The in-lake well was inserted into the tube and the tube was withdrawn, leaving the in-lake wells and drive point in place. Bentonite and a collar were placed around the base of the well at the lake bottom to seal the well, give the well strength, and help keep the bentonite in place. The collar was weighted to the bottom with large cobble size rocks.

Most of the 21 in-lake wells installed in 1987 were broken or lifted by ice during break up in 1989. The wells were not replaced, which increased the importance of seepage meter readings. In-lake wells were not sampled in 1989.

Seepage meters were constructed by cutting the top or bottom portion off of a 55 gallon drum and attaching a hospital intravenous (IV) bag to it. Seepage meters were placed on the lake bottom, driven into the sediments and sealed around the edge with bentonite. In areas subject to wave action, truck tire tubes were used to protect the bentonite seal. In areas with soft, deep mud, whole barrels were converted into seepage meters to allow contact with firm sediment before the meter was completely covered by mud. Bags partially filled with water were attached to seepage meters. If ground water was moving into the lake, the bag gained water and if ground water was moving out of the lake, the bag lost water. Seepage meters allowed the direct determination of the volume of ground water moving into and/or out of a lake by periodically removing the old bag, measuring the volume of water and attaching a new bag.

Seepage meters were installed adjacent to each in-lake well and at several transects, beginning near shore and extending out into the lake. Seepage meter readings were collected at two or three week intervals depending on weather conditions. Seepage bags were checked 1 to 24 hours of being attached so that pressure due to a full seepage bag did not inhibit seepage. No readings were taken during the winter.

Ground Water Data Analysis Design

The goal of analysis and evaluation of ground water data was to determine the quantity and quality of ground water entering and exiting the Oakwood Lakes system. Seepage rates to lake beds can be estimated without using seepage meters. By knowing the hydraulic conductivity and effective porosity of the aquifer and lake bottom, and the gradient between ground water and lake level, a velocity for the ground water can be calculated. The equation for the average linear ground water velocity (Freeze and Cherry 1985) is:

$$A.L.V. = K * I / n$$

A.L.V. = average linear velocity
K = hydraulic conductivity
I = gradient
n = effective porosity

Flow velocity multiplied by the area yields a quantity of water.

$$Q = A * (K * I / n)$$

It is difficult to determine the effective porosity (usually estimated from references based on particle distribution of the material), especially on the lake bottom. Hydraulic conductivity of the aquifer can be determined by pumping tests or single well slug or bail down procedures. In-lake monitoring wells may allow similar tests. The key hydraulic conductivity needed for accurate quantity calculations is the limiting hydraulic conductivity. This is probably located within the strata that comprises the lake bottom sediments. The hydraulic conductivity of the in-lake well may not be located in this limiting layer.

Direct measurements of seepage rates are possible with the use of seepage meters. Seepage rates are combined with chemical concentrations to calculate the total ground water loading to the lake. In its simplest form the following steps are followed:

1) Seepage rates are calculated at every available measuring point.

2) Areas of lake bottom with similar seepage rates for selected time intervals are delineated. The area is multiplied by the seepage rate and the time interval to obtain a volume.

3) Each volume is multiplied by a concentration determined by available water quality data from in-lake or land wells to obtain a loading for each area.

4) The loadings are summed to obtain a total loading for the entire lake bottom for the selected time interval.

Seepage rates can be obtained from direct measurement or by establishing a predictive relationship with wells and hydraulic properties of the aquifer and lake bottom. Establishing a predictive relationship is not only desirable to determine seepage when the seepage meter is no longer present but also because measurements at seepage meters are difficult when the lake is frozen. Once the relationship is derived, seepage can be calculated from the gradient between wells and lake elevation because effective porosity and hydraulic conductivity become a constant and are combined into one term.

5.2.4 Phytoplankton and Zooplankton Study

Replicate phytoplankton and zooplankton samples were collected from all seven stations on 15 dates during 1989. Phytoplankton samples were taken from the same bottle cast as the samples for chemical analysis. Zooplankton were sampled by both vertical net tows (0.3m diameter), and horizontal tows with a larger net (1m diameter). In the laboratory, samples were counted by species and values represented by numbers and biomass (gm) or biovolume (ml) estimates. Water clarity was measured by replicate secchi disc measurements taken at the same time and location as the plankton samples.

5.2.5 Fisheries Study

Relative abundance and size-class distribution of fish in East and West Oakwood Lakes were estimated through assessment netting using four gears:

1) fyke net, 0.9 X 1.8 m frame (1.3 cm mesh) with a 13.7 m lead.

2) fyke net, 0.9 X 1.2 m frame (1.9 cm mesh) with a 25.0 m lead.

3) experimental gill net, 61.0 X 1.8 m multifilament, composed of 5-12.2 m sections of 1.3, 2.5, 3.8, 5.1, and 6.4 mm mesh.

4) bag seine, 13.7 X 1.5 m with 3.2 mm mesh.

Five stations were selected on both lakes to represent similar habitat types (i.e. rock shoreline, point, vegetation) (Figure 5-1). At each station, 20-24 hour sets were made using fyke nets and the gill net. A 30 meter section of shoreline at each station was seined. Sampling was conducted only in May and September, because data were redundant from more frequent sampling that was conducted in the first two years of the study. Total number and weights for each species collected from each gear were recorded.

5.2.6 Lake Sediment Study

5.2.6.1 Sampling procedure

Thirty-six sediment cores were obtained from Oakwood Lakes during March, 1989. Sampling sites were selected based on a nested experimental design of four cores within a site, three sites within a basin, and three basins within the lake system. Each lake basin was sampled along a transect which extended from the shallow near shore area, to the deep water area.

Prior to coring, DO and temperature profiles of the water column were obtained with a Winkler calibrated DO meter (YSI, Model 51B). Water quality samples were collected at 0.30 m intervals with an automatic water sampler (ISCO, Model 2700). Chemical parameters measured were total PO_4-P and ortho PO_4-P.

Intact sediment cores, approximately 0.91 m long, were collected with a project designed sediment corer equipped with removable polycarbonate tubes (4.5 cm o.d. x 1.2 m). Polycarbonate was selected for use because of its resistance to phosphate adsorption (Ryden et al. 1972).

5.2.6.2 Phosphorus extraction

Two cores from each site were chemically analyzed to determine the phosphorus content of the sediment. The cores were extruded from the polycarbonate tubes while held in an upright position to prevent mixing of the sediment layers. Each core was divided into two 7.6 cm (3 in) sections at the top, then into 15.2 cm (6 in) sections thereafter. A brief description noting core length, sediment color, sediment texture and presence of organic matter was logged for each core. The phosphorus content of the sediment was determined by the phosphorus extraction method of Aspila (1976).

5.2.6.3 Incubation experiment

A laboratory experiment was conducted to investigate the extent of phosphorus release from the sediment to the overlying water during aerobic and anaerobic conditions. Two sediment cores from each site were incubated within their original polycarbonate tubes; one under aerobic conditions and one under anaerobic conditions.

The aerobic system was maintained by bubbling compressed air through the overlying water and the anaerobic system was maintained by bubbling nitrogen gas through the overlying water. The DO concentration of the water was checked daily with a DO meter (YSI model 51B) and remained at 7.2 ppm - 9.2 ppm in the aerobic system and less then 1.0 ppm in the anaerobic system.

The overlying water was replaced every two days during the thirty-two day incubation period. The replacement water was analyzed for total PO_4-P and ortho PO_4-P before and after incubation and the quantity of phosphorus released from the sediment was determined from the difference between the before and after phosphorus concentration.

An objective of the incubation experiment was to simulate lake conditions as much as possible except that water low in nutrients was used to exchange the overlying water. This was done to determine the role of the sediments in supplying phosphorus to the Oakwood Lakes if substantial improvements are made in tributary water quality. Groundwater from an aquifer which seeps into Oakwood Lakes was used as the incubation water. Prior to incubation, lake water trapped in the tubes during coring was drawn off and replaced with the groundwater.

All cores were maintained at uniform light and temperature conditions within an incubation chamber. Temperature was between 22° C - 25° C and light was excluded except during water changes, to minimize the impact of algae trapped in the tubes during coring. The incubated cores were analyzed to determine the post-incubation phosphorus content of the sediment after termination of the incubation experiment.

Methods of statistical analysis used to evaluate the results of the sediment study included Analysis of Variance (ANOVA) and Least Significant Difference (LSD). Analysis was conducted using SAS ; SAS procedures used in the analysis were PROC LEAPS and PROC MEANS (SAS 1985).

5.3 Results and Discussion

5.3.1 Weather and Atmospheric Deposition

A summary of winter weather data from the Brookings station for three years of study are presented in Table 5-1 (Lytle, personal communication). The pattern of precipitation, temperature and snow accumulation affects the runoff potential of the spring snowmelt. During 1987 and 1988 snowmelt created the only significant runoff entering the Oakwood Lakes.

The winter of 1986-87 was mild with low precipitation and little snow accumulation; the 1987-88 winter was colder and drier then normal with more snowfall but less accumulation at snowmelt than in 1987. Several rain events occurred in 1987 and 1988 but no significant runoff events occurred after the snowmelt for either year. The below average snowpack and precipitation in 1987 and 1988 probably resulted in tributary water and total PO_4-P inputs lower then what occurs during years of normal precipitation. Although precipitation was again below normal in 1989 normal snow pack just prior to the snowmelt was higher than in 1987 and 1988 due to a 26 inch snowfall in March (Table 5-1). Subsequent warm weather resulted in a rapid snowmelt which was interrupted by a severe cold front. This weather pattern had a negative impact on the measurement of stream discharges.

Significant rain fell on several occasions during 1989. Three runoff events affected all tributaries to varying degrees after the snowmelt runoff had ceased.

5.3.2 Nutrient, Sediment and Hydrologic Budgets

Data produced by the tributary monitoring program permitted calculation of the volume of water and the mass of nutrients and suspended sediment entering the Oakwood Lakes system from several tributaries in 1987, 1988 and 1989. The amounts contributed by each tributary, exchanged between lake basins and the amount lost through the outlet, for each year are presented in Table 5-2. Arrangement within the table is based on the natural sequence of water movement through the system. Estimated ground water flux, direct runoff from unmonitored watershed areas, and the effects of precipitation and evaporation are not included. Data has been collected for these calculations but the analysis has not been completed.

Table 5-1. A summary of winter weather data from the Brookings station for 1986-1989. Prepared by W.F. Lytle, retired State Climatologist.

	Nov. '86	Dec. '86	Jan. '87	Feb. '87	Mar. '87
Mean temperature	25.6	23.4	20.1	30.5	33.0
Normal	30.8	17.1	9.2	15.0	27.2
Departure	- 5.2	+ 6.5	+10.9	+15.5	+ 5.8
Precipitation	0.53	T	0.02	0.28	2.48
Normal	0.66	0.45	0.34	0.50	0.97
Departure	-0.13	-0.45	-0.32	-0.22	+1.51
Snowfall	3.0	T	1.5	4.0	1.0
Normal	2.7	6.0	4.9	6.4	8.2
Departure	+ 0.3	- 6.0	- 3.5	-2.4	- 7.2
Snow on ground Last day	0	0	0	2.0	0
	Nov. '87	**Dec. '87**	**Jan. '88**	**Feb. '88**	**Mar. '88**
Mean temperature	36.1	22.0	4.0	9.8	32.4
Normal	30.8	17.1	9.2	15.5	27.2
Departure	+ 5.3	+ 4.9	- 5.2	- 5.7	+ 5.2
Precipitation	0.77	0.34	0.35	0.14	0.29
Normal	0.66	0.45	0.34	0.50	0.97
Departure	+ 0.11	- 0.11	+ 0.01	- 0.36	- 0.68
Snowfall	1.0	20.2	12.0	2.5	3.0
Normal	2.7	6.0	4.9	6.4	6.2
Departure	- 1.7	+14.2	+ 7.1	- 3.9	- 3.2
Snow on ground Last day	0	20.0	32.0	8.0	0
	Nov. '88	**Dec. '88**	**Jan. '89**	**Feb. '89**	**Mar. '89**
Mean temperature	31.0	19.4	20.1	5.1	25.1
Normal	30.8	17.1	9.2	15.5	27.2
Departure	+ 0.2	+ 2.3	+10.9	-10.4	- 2.1
Precipitation	0.77	0.19	0.01	0.17	0.85
Normal	0.66	0.45	0.34	0.50	0.97
Departure	+0.11	-0.26	-0.33	-0.33	-0.12
Snowfall	2.0	3.0	T	6.0	26.0
Normal	2.7	12.0	4.9	6.4	8.2
Departure	- 0.7	- 9.0	- 4.9	- 0.4	+17.8
Snow on ground Last day	T	3.0	0.0	1.0	0.0

T = Trace of precipitation too small to measure.

Table 5-2. Total Contribution of Water, Suspended Solids, Total Phosphorus and Total Nitrogen for Tributary and Interlake Sites in 1987, 1988 and 1989, with flow-weighted mean concentrations in parts per million (ppm).

			Suspended Solids		Total PO4-P		Total Nitrogen	
		Discharge (acre-ft)	Loading (kg)	Mean (ppm)	Loading (kg)	Mean (ppm)	Loading (kg)	Mean (ppm)
Pony	1987	133.1	950.4	5.8	96.7	0.59	453.9	2.76
Creek	1988	39.4	746.3	15.4	32.6	0.67	110.6	2.28
(T-3)	1989	18.5	234.3	10.3	22.4	0.98	66.9	2.94
Mud	1987	493.2	8,026.0	13.2	263.1	0.43	2,204.8	3.62
Creek	1988	121.1	371.9	2.5	71.1	0.48	283.6	1.90
(T-4)	1989	118.2	1,691.0	11.6	137.6	0.94	823.7	5.65
Mortimer's	1987	512.4	9,684.1	15.3	106.0	0.17	1,197.1	1.89
Crossing	1988	91.4	3,038.4	26.9	39.5	0.35	495.8	4.40
(IL-3)	1989	24.0	-366.1	12.4	20.6	0.70	175.4	5.92
West	1987	2,200.3	43,301.2	16.0	553.5	0.20	6,898.3	2.54
Creek	1988	1,089.9	33,165.4	24.7	403.1	0.30	3,056.6	2.27
(T-1)	1989	1,936.9	149,400.9	62.5	1,625.5	0.68	7,766.5	3.25
Crazy	1987	866.5	41,675.1	39.0	339.4	0.32	3,590.2	3.36
Creek	1988	565.7	10,840.1	15.5	306.4	0.44	1,533.1	2.20
(T-2)	1989	607.3	4,575.9	6.1	532.8	0.71	2,855.1	3.81
Kimball's	1987	3,445.1	66,887.4	15.7	517.1	0.12	6,607.5	1.55
Crossing	1988	874.3	24,994.3	23.2	263.1	0.24	3,231.8	3.00
(IL-1)	1989	603.7	26,985.6	36.2	133.9	0.18	2,037.0	2.74
Round	1987	1,127.5	24,798.0	17.8	172.3	0.12	2,261.6	1.63
Crossing	1988	73.5	2,524.1	27.8	11.6	0.13	230.7	2.54
(IL-2)	1989	4.7	238.5	40.9	0.9	0.15	17.9	3.08
Loomis	1987	104.4	2,105.7	16.3	158.6	1.23	676.8	5.26
Creek	1988	19.4	810.9	33.9	45.0	1.88	174.7	7.30
(T-5)	1989	42.0	6,724.4	129.8	182.7	3.53	529.0	10.21
Goose	1987	53.0	1,509.4	23.1	7.8	0.12	192.4	2.94
Creek	1988	15.8	327.6	16.8	4.6	0.24	66.2	3.40
(T-6)	1989	72.6	2,771.4	31.0	27.0	0.30	359.2	4.01
Mill	1987	4,319.5	55,581.3	10.4	427.9	0.08	7,560.9	1.42
Creek	1988	1,366.8	31,041.3	18.4	200.7	0.12	2,859.0	1.70
(T-0)	1989	0.0	0.0	0.0	0.0	0.00	0.0	0.00

Hydrologic Budget

An accounting of surface water movement through the Oakwood Lakes system in 1987, 1988 and 1989 is presented in Figure 5-3. Net gain or loss for each of the major subbasins is indicated in the boxes. All of the basins gained more surface water than they exported with two exceptions. In 1987 the West Oakwood Lakes including Johnson, Round and Turtle Lakes, had a net loss of 993.4 acre feet of water. In 1988 East Oakwood Lake had a net loss of 384 acre feet of water.

Hydraulic loadings from most tributaries were highest in 1987. Hydraulic loadings were lower in 1988 and 1989. Loadings for 1987 and 1988 were comprised primarily of snowmelt runoff. In 1989 three significant storm events in June and July resulted in significant runoff in several tributaries. West Creek contributed the most water to the system in all three years of study.

There was less transfer of water between basins in each successive year of the study. Lake levels were high at the beginning of 1987 due to several preceding wet years and the large flows between lakes resulted from the lakes draining to normal levels. Dry conditions beginning in 1987 caused lake levels to fall below outlet elevations at the beginning of 1988 and 1989. Part of the tributary contribution was needed to fill the basins before outflow could begin in 1988 and 1989. The culverts at Kimball's Crossing were also partially blocked for most of 1988 and 1989 due to local concern over falling water levels; this caused less water to flow into East Oakwood Lake from West Oakwood Lake. No water left the system through the outlet in 1989 so 100% of the hydraulic, nutrient, and sediment load was retained.

Suspended Solids Budget

Large differences in annual suspended solids loading was observed for most tributaries (Table 5-2). The smallest contributions occurred during 1988 for all tributaries except Pony Creek and Crazy Creek which had lower annual loads in 1989. Highest annual contributions of suspended solids for most tributaries occurred in 1987. The large load of suspended solids carried by West Creek in 1989 was due primarily to a thunderstorm on July 11 when hail and eight inches of rain fell on parts of the West Creek watershed. Other tributaries such as Pony Creek, Mud Creek and Crazy Creek were impacted less by major events in 1989 which resulted in lower suspended solids loadings. The total suspended solids load to the lake system carried by six monitored tributaries was 97,567.8 kg in 1987, 46,262 kg in 1988 and 165,398 kg in 1989.

A flow chart indicating the movement of suspended solids through the Oakwood Lakes system is presented in Figure 5-4. Net gain or loss for each major subbasin is indicated in the boxes. Mortimer's Slough had a net loss of suspended solids in both 1987 and 1988. East Oakwood Lake had a net loss of 2,384 kg of suspended solids in 1988. All of the other basins had a net gain of suspended solids. The lakes were most efficient at trapping suspended solids in 1989.

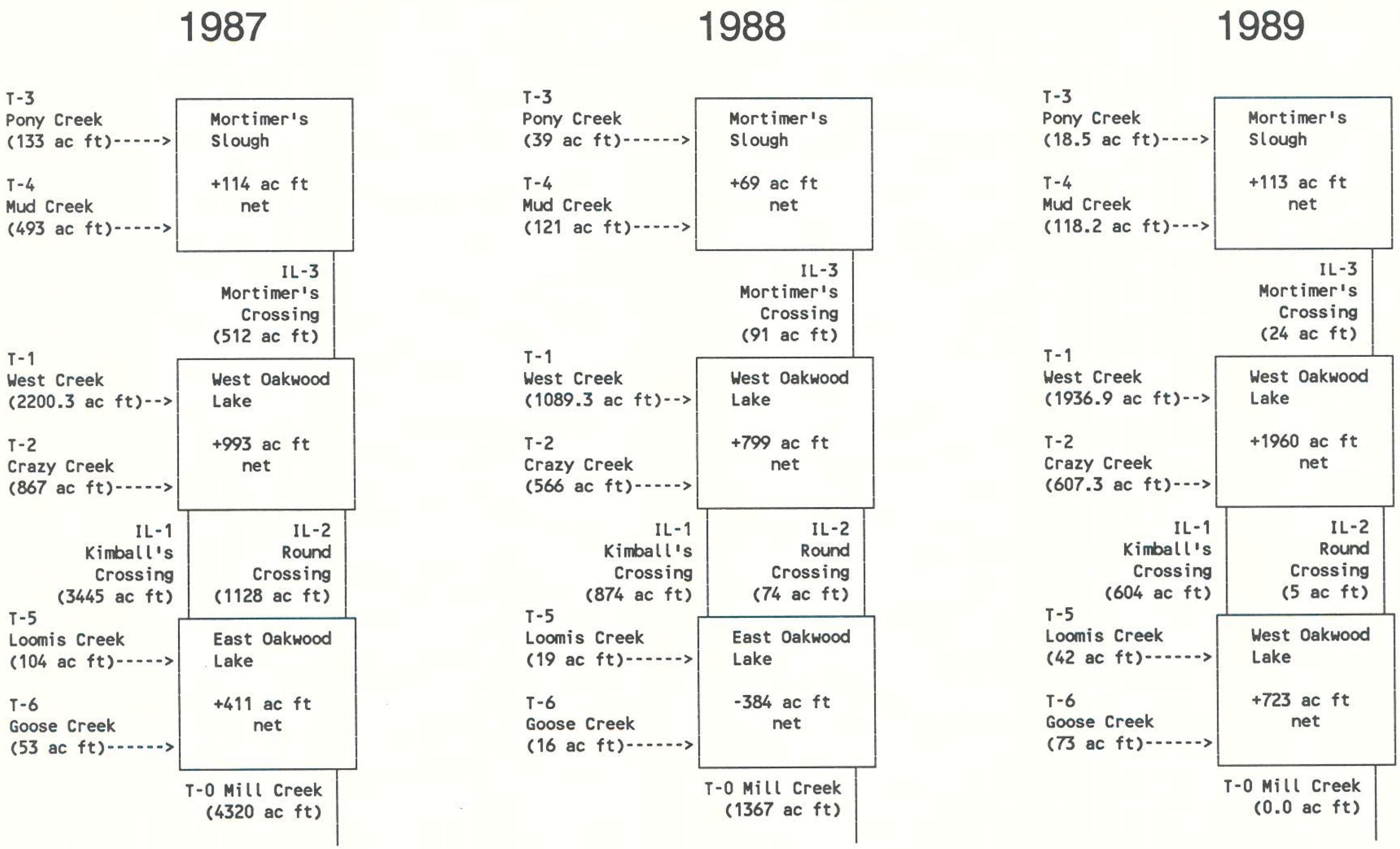


Figure 5-3. An accounting of surface water movement through the Oakwood Lakes system in 1987, 1988 and 1989.

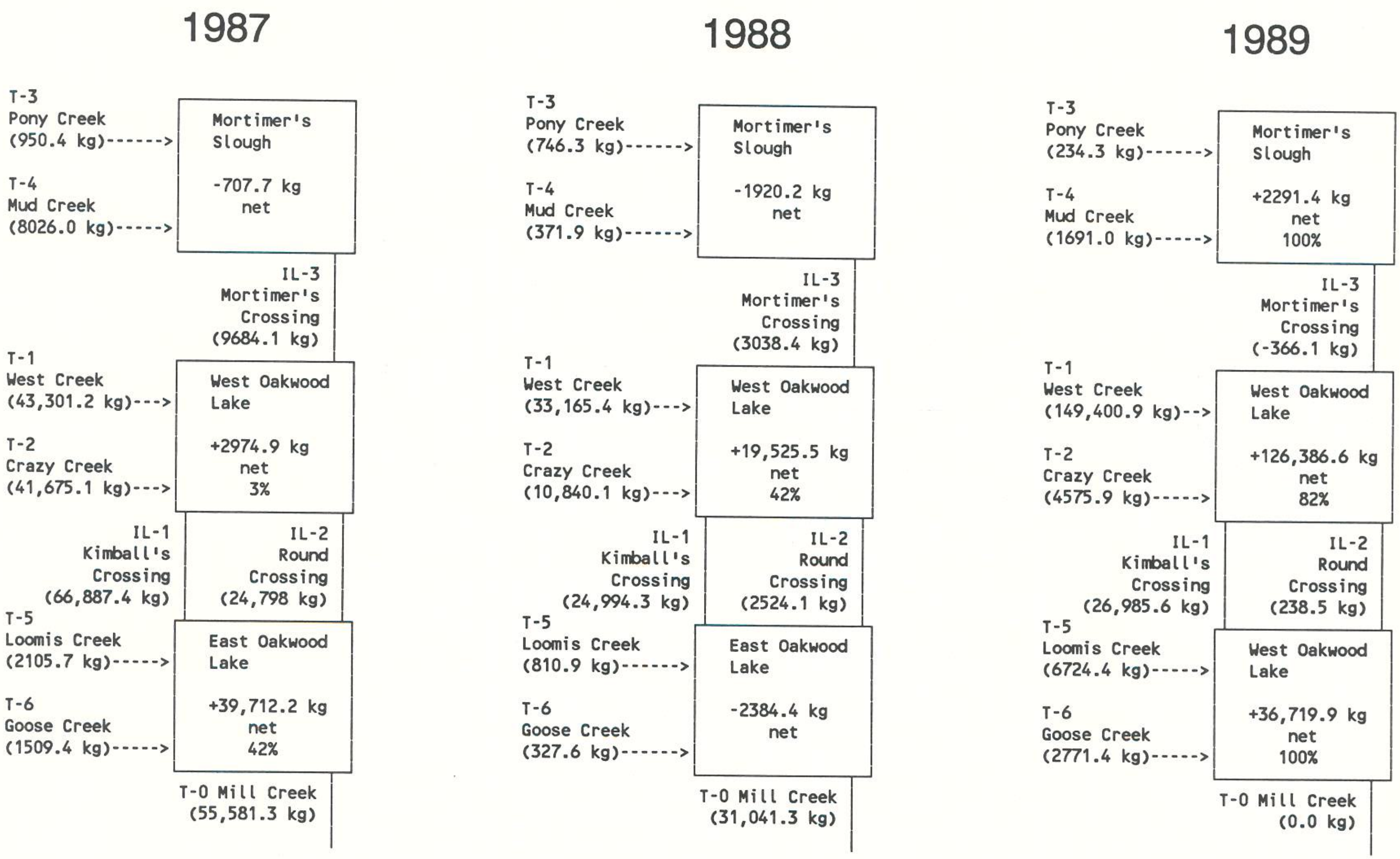


Figure 5-4. An accounting of suspended solids in surface water moving through the Oakwood Lakes system in 1987, 1988 and 1989.

ranging from 82% to 100% efficient (Figure 5-4). The lake system trapped 43%, 33% and 100% of the suspended solids that was contributed by six monitored tributaries in 1987, 1988 and 1989 respectively (Figure 5-4).

Much of the suspended solids that moves between the lakes is in the form of organic material produced within the lake basins. The quantity of suspended solids transferred sites depends on the amount of water flowing between basins and the level of the plankton population. The largest transfer of suspended solids between basins occurred in 1987, due to high lake levels. In 1989, a negative loading of suspended solids occurred at Mortimer's Crossing. More suspended solids entered Mortimer's Slough from West Oakwood Lake when flows reversed during the July 11 runoff event than were transferred into West Oakwood Lake from Mortimer's Slough during the snowmelt runoff.

Phosphorus Budget

An accounting of total phosphorus movement through the Oakwood Lakes system is presented in Figure 5-5. All of the basins had a net gain of total PO_4-P in all three years of study. The basins were most efficient at trapping total PO_4-P in 1989 ranging from 87% to 100% efficient (Figure 5-5). The lake system as a whole trapped 70%, 80% and 100% of the total PO_4-P that was contributed by six monitored tributaries in 1987, 1988 and 1989 respectively (Figure 5-5). The Oakwood Lakes system is operating as a phosphorus sink even though a net loss of water occurred in West Oakwood Lake during 1987 and in East Oakwood Lake during 1988 (Figure 5-3). There will be an accumulation of phosphorus over time and little hope for water quality improvement if these conditions persist.

The total PO_4-P load to the lake system carried by six monitored tributaries was 1,419 kg in 1987, 993 kg in 1988 and 2,528 kg in 1989. This represents an annual tributary total PO_4-P load to the Oakwood Lake system of .16 g/m^2/year in 1987, 0.11 g/m^2/year in 1988 and 0.28 g/m^2/year in 1989. West Oakwood Lake received a tributary load equal to 0.213 g/m^2/year in 1987, 0.159 g/m^2/year in 1988 and 0.46 g/m^2/year in 1989. East Oakwood Lake received a tributary total PO_4-P load equal to 0.230 g/m^2/year in 1987, 0.012 g/m^2/year in 1988 and 0.093 g/m^2/year in 1989. Vollenweider (1968), would consider this a dangerous load (would cause rapid eutrophication) for lakes much deeper than the Oakwood Lakes (permissible load varies with a lakes mean depth). A total PO_4-P loading of 0.13 g/m^2/year for a lake of 5 m mean depth is considered dangerous. In relation to this standard, West Oakwood Lake (max depth = 3.5 m) received excessive tributary total PO_4-P loads in 1987 and 1988, and East Oakwood Lake (max depth = 3.2 m) received an excessive load in 1987.

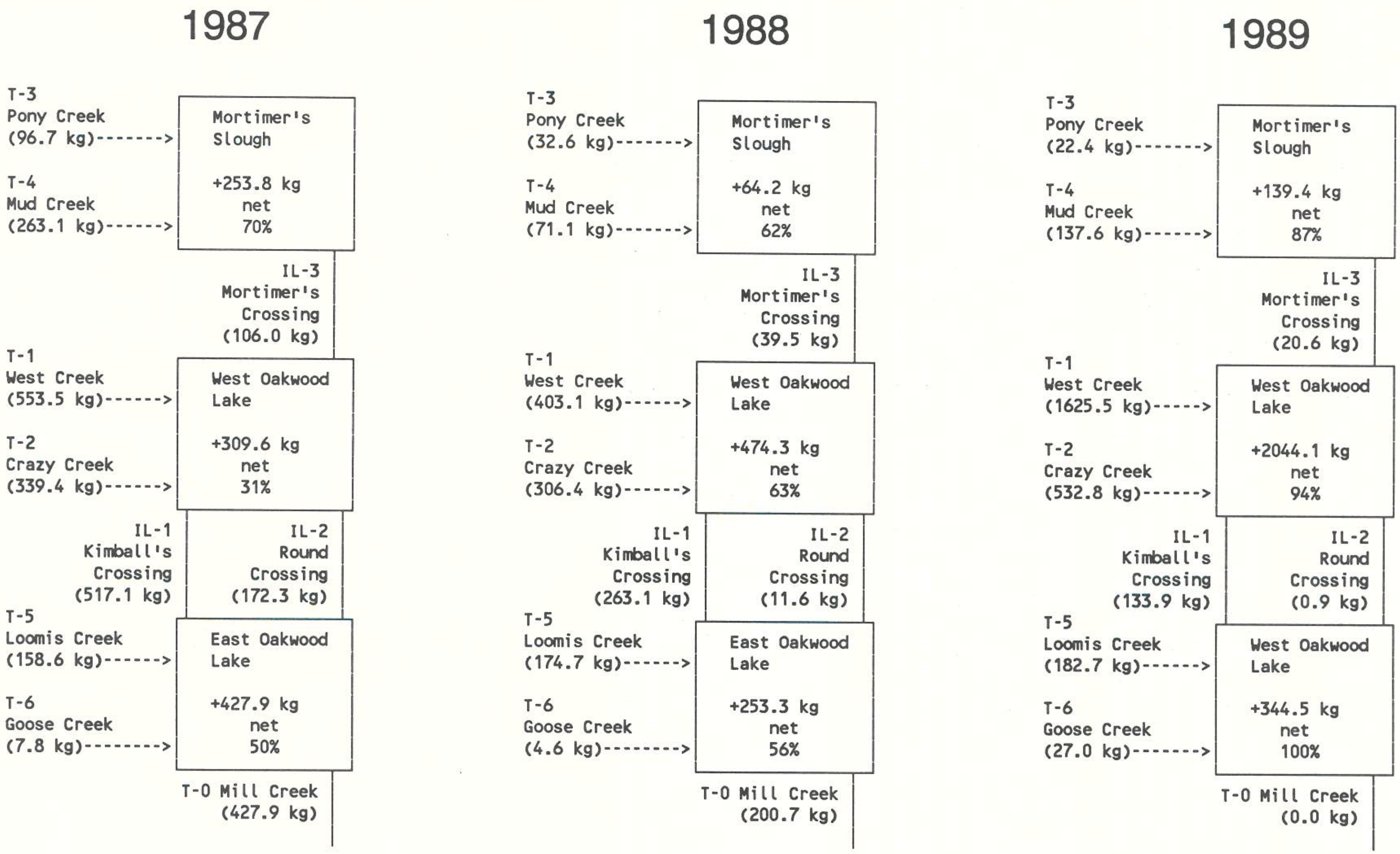


Figure 5-5. An accounting of total phosphorus in surface water moving through the Oakwood Lakes system in 1987, 1988 and 1989.

Nitrogen Budget

Unlike phosphorus, nitrogen may undergo denitrification and be lost to the system and returned to the atmosphere. Nitrogen may also be added to the system through nitrogen fixation by several species of blue green algae. No attempt was made to account for nitrogen lost to denitrification or gained by nitrogen fixation.

Large differences in total nitrogen as N loading between years was observed for most tributaries (Table 5-2). The smallest contributions occurred during 1988 for all tributaries except Pony Creek which had a smaller contribution in 1989. Highest contributions of total nitrogen as N for most tributaries occurred in 1987. The large load of total nitrogen as N carried by West Creek in 1989 was due primarily to the July 11 storm.

An accounting of nitrogen movement through the Oakwood Lakes system is presented in Figure 5-6. All of the basins retained nitrogen except for Mortimer's Slough which had a net loss of 102 kg of total nitrogen as N in 1988. Although the lake system is operating as a nitrogen sink, nitrogen was trapped less efficiently by the lake system than phosphorus. The lake system as a whole trapped 46% of the tributary load in 1987, 44% in 1988 and 100% in 1989. There will be an accumulation of nitrogen reserves over time, if this condition persists.

The largest tributaries (Mud Creek, West Creek and Crazy Creek) contributed the largest amounts of total nitrogen in all three years. The total nitrogen as N load to the lake system carried by six monitored tributaries was 14,016 kg in 1987, 5,095 kg in 1988 and 12,400 kg in 1989. This represents an annual tributary total nitrogen as N load to the Oakwood Lake system of 1.57 $g/m^2/year$ in 1987, 0.57 $g/m^2/year$ in 1988 and 1.39 $g/m^2/year$ in 1989. As with phosphorus, this represents a dangerous load, which would lead to rapid eutrophication, for lakes as shallow as the Oakwood Lakes (Vollenweider 1968).

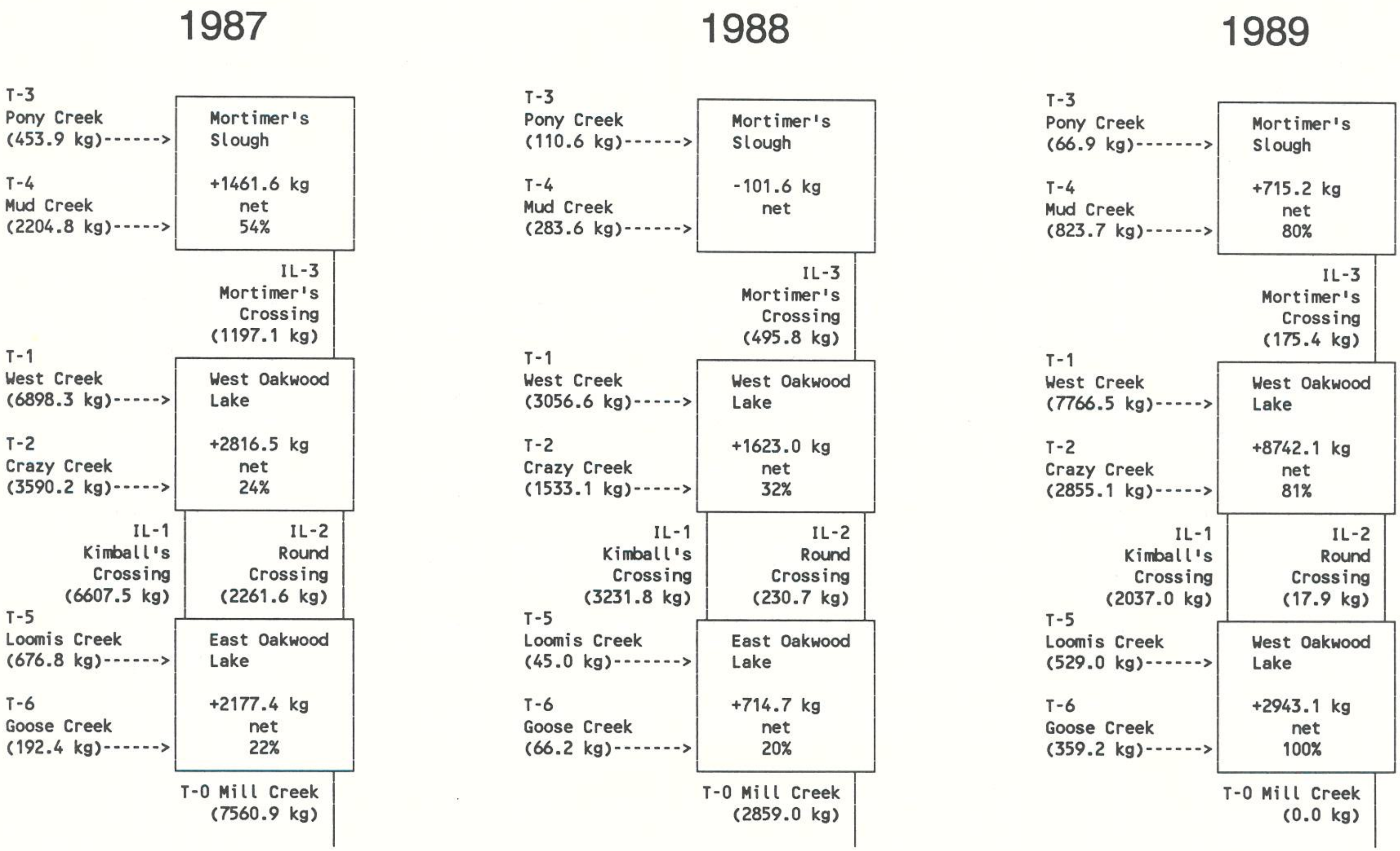


Figure 5-6. An accounting of total nitrogen in surface water moving through the Oakwood Lakes system in 1987, 1988 and 1989.

5.3.3 Tributary Water Quality

Minimum, maximum and mean values for several parameters measured in 1989, are presented for each tributary in Table 5-3. The data provides a basis for comparing tributaries subjected to similar hydrologic conditions and is useful for identifying tributaries with water quality problems. Weighted mean concentrations of nutrients for each tributary (calculated by dividing the nutrient load by the total discharge) is probably a better indication of relative impact on the lake system (Table 5-2). A brief discussion of several parameters follows a description of its use as a water quality indicator. South Dakota water quality standards are cited when appropriate.

Table 5-3. Minimum, Maximum, and Mean Values of Selected Water Quality Parameters for Oakwood Lake System Tributaries for 1989.

		Suspended Solids (ppm)	Dissolved Solids (ppm)	Total Solids (ppm)	Nitrite as N (ppm)	Nitrate as N (ppm)	Ammonia as N (ppm)	Organic Nitrogen as N (ppm)	Total Nitrogen as N (ppm)	Total PO_4-P as P (ppm)	Ortho PO_4-P as P (ppm)	Percent Ortho PO_4-P
West	min	1	200	204	0.000	0.05	0.00	0.94	1.27	0.160	0.000	0.0%
Creek	max	574	1720	1732	0.170	3.40	1.76	3.98	8.23	1.550	0.810	90.0%
(T-1)	mean	43	653	695	0.041	0.77	0.45	1.98	3.13	0.543	0.325	53.0%
Crazy	min	1	130	140	0.000	0.04	0.01	0.92	1.12	0.130	0.070	50.0%
Creek	max	52	1763	1772	0.108	1.74	1.13	5.52	5.91	1.000	0.920	98.8%
(T-2)	mean	11	459	469	0.032	0.52	0.26	1.60	2.36	0.597	0.492	77.4%
Pony	min	2	103	108	0.000	0.10	0.11	1.25	1.71	0.480	0.340	56.7%
Creek	max	71	354	376	0.410	2.24	5.04	9.94	15.91	3.300	1.500	100.0%
(T-3)	mean	10	212	223	0.070	0.67	1.00	2.62	4.40	1.242	0.811	80.1%
Mud	min	4	128	140	0.007	0.16	0.13	1.49	2.00	0.340	0.080	23.5%
Creek	max	23	427	448	1.880	2.68	1.67	3.03	7.40	1.700	1.320	88.0%
(T-4)	mean	12	250	261	0.150	1.32	0.82	2.23	4.53	0.861	0.650	69.2%
Loomis	min	9	188	236	0.000	0.09	0.11	0.97	2.63	1.150	0.310	12.4%
Creek	max	966	1657	1684	1.650	2.84	11.25	16.50	27.60	7.600	5.750	90.6%
(T-5)	mean	113	489	602	0.128	0.72	2.35	5.64	8.34	3.374	2.226	63.9%
Goose	min	9	182	200	0.000	0.05	0.00	1.00	0.87	0.160	0.000	0.0%
Creek	max	79	2115	2128	0.136	3.52	0.75	3.07	7.06	0.980	0.300	100.0%
(T-6)	mean	30	335	365	0.026	0.71	0.19	1.59	2.45	0.281	0.082	29.0%

Suspended Solids

Suspended solids concentrations can be an indication of how much sediment is carried by a stream although it does not include a measure of larger particles that are carried along the stream bed as bed load during higher flows. High concentrations of suspended solids can result from excessive soil erosion or contamination by human or animal wastes. Excessive suspended solids can have detrimental effects on stream and lake fisheries and speed up the aging of lakes by filling in the basin and providing a future source of nutrients for plant growth.

In 1989 suspended solids concentrations ranged from 1 ppm at West and Crazy Creeks to 966 ppm at Loomis Creek (Table 5-3). The state standard for maintaining a warm water marginal fishery is 150 ppm. This level was exceeded at both West Creek and Loomis Creek in 1989. The large concentration of suspended solids carried by both Creeks in 1989 was due to erosion caused by large rain events which occurred in June and July, 1989. West Creek and Loomis Creek also had the largest flow weighted mean concentrations of suspended solids in 1989 (Table 5-2).

In 1987 Crazy Creek had the highest concentrations of suspended solids but in subsequent years the flow weighted means were much lower even though total discharges were similar (Table 5-2). This points out the difficulty of evaluating data from a single season to determine priority watersheds for implementation of non-point source pollution controls.

Ammonia as N

Ammonia nitrogen is an inorganic form of nitrogen derived from the decay of organic material and is present in human and animal wastes and commercial fertilizers. It is directly available to plants as a nutrient for growth. Significant levels of ammonia may indicate organic pollution or fertilizer leaching. High concentrations can be toxic to fish if present as unionized ammonia. In 1989 ammonia concentrations ranged from a maximum of 11.25 ppm at Loomis Creek to below detection limits at West, and Goose Creeks. Loomis Creek also had the highest mean ammonia concentration of 2.35 ppm in 1989 (Table 5-3).

Ammonia concentrations observed Loomis Creek and other tributaries indicate a source of organic pollution in these watersheds. Several small feedlots located on or near these creeks are contributing organic wastes. Several other parameters that indicate contamination by organic wastes are also elevated in Loomis Creek and Pony Creek. Goose Creek had the lowest mean ammonia concentrations in 1989 at 0.19 ppm. The Goose Creek watershed contains significant amounts of pasture land but has no concentrated animal feeding operations.

Nitrate and Nitrite Nitrogen as N

Nitrate and nitrite are readily absorbed sources of inorganic nitrogen for plant growth. High levels in lakes can cause excessive algal growth. Nitrite levels are generally low (0 to 0.01 ppm) in natural surface waters. Nitrate nitrogen concentrations generally range from below detection limits to 10.0 ppm in unpolluted fresh water (Wetzel, 1983).

Oakwood Lakes tributary concentrations of nitrite ranged from below detection limits at most tributaries to a maximum of 1.88 ppm at Mud Creek. Mud Creek also had the highest mean nitrite as N concentration at .15 ppm, and Goose Creek exhibited the lowest mean nitrite as N concentration at 0.026 ppm.

The maximum and minimum nitrate as N concentrations for 1989 both occurred at Goose Creek and ranged from 0.05 ppm to 3.52 ppm. The highest 1989 mean nitrate as N concentration of 1.32 ppm occurred at Mud Creek. Mean concentrations of nitrates for all tributaries in 1989 were higher than concentrations observed for in-lake samples.

Organic Nitrogen as N

Organic nitrogen is a measure of organic material both living and non-living contained in the water. High levels can be an indication of pollution by human or animal wastes. When oxidized, it becomes a source of nitrogen for plant growth. Organic nitrogen was the dominant form of nitrogen in most tributary samples collected in 1989. Concentrations ranged from 0.92 ppm at Crazy Creek to 16.50 ppm at Loomis Creek. Loomis Creeks had the highest mean concentrations of organic nitrogen as N at 5.64 ppm. Other tributaries also exhibit possible sources of organic pollution but to a lesser degree.

Total Nitrogen as N

Total nitrogen as N is the sum of the three inorganic forms plus organic nitrogen. It is a useful parameter for calculating nitrogen loads to a lake. Loomis Creek, Mud Creek and Pony Creek exhibited the largest mean concentrations in 1989 with 8.34 ppm, 4.53 ppm and 4.40 ppm respectively. The lowest mean concentration (2.36 ppm) was observed at Crazy Creek (Table 5-3).

All of the tributaries supply significant amounts of nitrogen to the lake system as indicated by flow weighted means in table 5-2. The highest flow weighted mean concentrations of total nitrogen as N were observed at Loomis Creek in all three years.

Total Phosphorus as P

Phosphorus is an essential nutrient for algal growth and is frequently the limiting nutrient in lakes. However only the orthophosphate form is considered to be immediately available for use by algae. Many of the unavailable forms of phosphorus carried by streams, such as dissolved organics, particulate suspensoids and scoured sediment deposits can be made available by chemical, physical and biological processes after reaching the lake (Wetzel, 1983). This is especially true for a shallow system like the Oakwood Lakes. The concentration of total PO_4-P in tributaries is an important indicator of the potential impact on the receiving water.

Phosphorus concentrations above 0.025 ppm entering a lake would be considered a dangerous load (ie. would lead to rapid eutrophication) according to Vollenweider and Kerekes (1980) (from Wetzel 1983). Lakes with phosphorus concentrations above 0.050 ppm would be considered hypereutrophic according to a classification proposed by Reckhow, et al. (1980). Based on these standards there is little hope that a lake will improve from a hypereutrophic to a eutrophic state as long as it continues to receive substantial quantities of water with total PO_4-P concentrations above 0.050 ppm. Although phosphorus is probably the single most important nutrient involved with degradation of lakes, South Dakota does not currently have a phosphorus standard for waters discharging to lakes.

For discussion, a total PO_4-P concentration level of 0.050 ppm will be used as a reference. All samples taken from tributary, and interlake sites in 1989, had concentrations above .050 ppm total PO_4-P. All monitored tributaries carry an excessive load of total PO_4-P to the Oakwood Lakes system. In 1989 the highest maximum and mean total PO_4-P concentrations of 7.60 ppm and 3.37 ppm, respectively, were found at Loomis Creek (Table 5-3). Loomis Creek also had the highest flow weighted mean concentrations of total PO_4-P in all three years (Table 5-2). These concentrations indicate a severe case of pollution in the Loomis Creek watershed. Pony Creek and Mud Creek also exhibited high flow weighted mean concentrations of total PO_4-P.

Goose Creek is the smallest monitored tributary entering the Oakwood Lakes. It consistently had the lowest flow weighted mean total PO_4-P concentrations when compared to other tributaries although it increased from 0.12 ppm in 1987 to 0.24 ppm in 1988 and 0.30 ppm in 1989. The watershed drains an area dominated by cropland and pasture with no concentrated animal feeding operations. Water from the entire drainage passes through a small dam built by the Soil Conservation Service. These factors probably combine to produce more moderate levels of total PO_4-P in Goose Creek than other tributaries.

Orthophosphorus as P

Orthophosphate (or soluble reactive phosphorus) is a measure of bioavailable PO_4-P in a sample. Concentrations of ortho PO_4-P in a tributary can indicate the short term impact the tributary will have on a water body that is or will become limited by phosphorus. Ortho PO_4-P was measured on all tributary samples where the hold time before analysis was less than 24 hours.

Loomis Creek carried the highest mean levels of orthophosphate in 1989 with 2.226 ppm. Goose Creek had substantially lower levels of orthophosphate (mean of 0.082 ppm) as well as the lowest percentage (29%) of ortho PO_4-P. For all tributaries, ortho PO_4-P represented a high percentage of the total PO_4-P (Table 5-3).

Feedlot Runoff

In 1989 sampling began above and below one feedlot in the Oakwood Lakes watershed to document water quality improvements that may result from the installation of an animal waste management system in 1990, which is under contract through the RCWP project. Results of above and below sampling indicate a substantial water quality impact below the feedlot. Mean concentrations of total PO_4-P above the feedlot was .87 ppm for 15 samples. Mean concentrations of total PO_4-P below the feedlot was 13.52 ppm. Mean concentrations of total nitrogen as N above the feedlot was 4.21 ppm for 15 samples. Mean concentrations of total nitrogen as N below the feedlot was 37.72 ppm.

5.3.4 In-Lake Water Quality

Minimum, maximum and mean values for selected parameters at in-lake, interlake and the outlet sites are presented in Table 5-4. In-lake water quality is described by evaluating several parameters that provide information about the trophic state of the lake, the ability to support a fishery, the nutrients that are available for algal growth and aesthetic factors. Interlake sites and the outlet are also included in Table 5-4 representing lake water flowing between basins or leaving the system. The data are used primarily to calculate mass transfer between basins but provides some insight into in-lake water quality. Mortimer's Crossing at station IL-3 does not have a corresponding in-lake monitoring site and is the only source of water quality data for that basin. IL-3 tends to have high mean values for most parameters when compared with other sites. Station T-0, the outlet at Mill Creek, generally exhibits the lowest mean concentrations for sediment and nutrient parameters which is a reflection of the higher quality water in East Oakwood Lake.

The Oakwood Lakes have been assigned the following beneficial uses:

1. warm water marginal fish propagation
2. immersion recreation
3. limited contact recreation
4. wildlife propagation and stock watering.

A discussion of water quality standards set by South Dakota to meet these beneficial uses will be included where appropriate. Figures are presented in this section indicating trends for several parameters. Two subsamples for each station are plotted with the mean for all stations included for comparison.

Suspended solids

Low suspended solids concentrations are desirable in lakes both for aesthetic reasons and for maintenance of a healthy fishery. Fish populations can be affected by high suspended solids in several ways. Fish can be killed directly or the growth resistance to disease and reproduction success may be reduced. Migrations can also be affected (EPA, 1976). High suspended solids concentrations result in reduced aesthetic value of a lake and often limit recreational use.

The state standard for maintaining a warm water marginal fishery is 150 ppm. The standard was not exceeded at any in-lake sites in 1987, 1988 or 1989. In the Oakwood Lakes, suspended solids is primarily a reflection of the plankton population and does not represent large amounts of suspended sediment. Suspension of sediment probably occurs during high winds in shallow areas but in-lake sampling was not conducted during dangerously high winds.

Table 5-4. Minimum, Maximum, and Mean Values of Selected Water Quality Parameters for In-Lake and Interlake Sites for 1989.

Site		Suspended Solids (ppm)	Dissolved Solids (ppm)	Total Solids (ppm)	Nitrite as N (ppm)	Nitrate as N (ppm)	Ammonia as N (ppm)	Organic Nitrogen as N (ppm)	Total Nitrogen as N (ppm)	Total PO_4 as P (ppm)	Ortho PO_4 as P (ppm)	% Ortho PO_4
Johnson	min	9	595	624	0.001	0.04	0.04	1.58	1.79	0.090	0.000	0.0%
Lake	max	41	1152	1188	0.019	0.12	1.54	2.94	3.97	0.330	0.070	63.6%
(L-1)	mean	27	950	977	0.010	0.08	0.35	2.22	2.66	0.145	0.023	17.6%
Johnson	min	5	860	884	0.000	0.02	0.04	1.61	1.84	0.070	0.000	0.0%
Lake	max	35	1136	1164	0.016	0.10	1.16	2.97	3.43	0.150	0.060	50.0%
(L-2)	mean	22	980	1002	0.008	0.06	0.38	2.14	2.58	0.103	0.016	15.2%
West	min	4	893	940	0.000	0.04	0.00	1.38	2.24	0.060	0.000	0.0%
Oakwood	max	88	1099	1120	0.022	0.10	0.97	3.06	3.37	0.150	0.030	33.3%
(L-3)	mean	23	1006	1029	0.011	0.07	0.47	2.24	2.78	0.089	0.012	12.9%
West	min	6	931	960	0.000	0.03	0.00	1.56	1.94	0.060	0.000	0.0%
Oakwood	max	39	1085	1108	0.021	0.14	0.97	2.89	10.29	0.150	0.030	33.3%
(L-4)	mean	20	1015	1035	0.009	0.07	0.47	2.14	3.11	0.093	0.012	13.7%
Turtle	min	7	927	952	0.000	0.00	0.01	1.61	1.70	0.070	0.000	0.0%
Lake	max	53	1133	1168	0.039	0.14	0.96	2.88	3.11	0.190	0.040	44.4%
(L-5)	mean	24	1018	1042	0.014	0.08	0.36	2.30	2.75	0.122	0.012	10.2%
Round	min	5	946	968	0.000	0.03	0.00	1.43	1.95	0.070	0.000	0.0%
Lake	max	57	1219	1228	0.037	0.10	2.68	4.10	4.36	0.210	0.040	50.0%
(L-6)	mean	26	1074	1099	0.009	0.06	0.75	2.58	3.40	0.102	0.011	12.8%
East	min	4	890	904	0.000	0.03	0.00	1.37	1.63	0.060	0.000	0.0%
Oakwood	max	49	1056	1076	0.031	0.10	1.43	3.77	4.12	0.160	0.030	33.3%
(L-7)	mean	26	966	992	0.006	0.06	0.24	2.18	2.48	0.108	0.013	14.0%
Kimball's	min	1	954	980	0.002	0.02	0.00	1.96	1.86	0.070	0.000	0.0%
Crossing	max	80	1239	1248	0.168	0.12	0.55	3.13	3.75	0.600	0.400	100.0%
(IL-1)	mean	27	1058	1085	0.021	0.07	0.25	2.53	2.67	0.187	0.106	24.3%
Round	min	23	963	992	0.000	0.03	0.06	1.94	1.86	0.070	0.000	0.0%
Crossing	max	117	1112	1172	0.029	0.14	0.55	8.98	9.53	0.660	0.020	20.0%
(IL-2)	mean	45	1014	1059	0.011	0.08	0.23	3.86	3.48	0.193	0.011	5.2%
Mortimer's	min	1	179	180	0.000	0.03	0.07	1.07	1.44	0.110	0.000	0.0%
Crossing	max	64	1053	1068	0.030	0.40	1.34	4.69	5.04	0.780	1.250	81.8%
(IL-3)	mean	19	709	728	0.014	0.15	0.48	2.88	3.29	0.344	0.148	28.0%

Mean concentrations of suspended solids were similar between stations in 1989. Up stream stations L-1 and L-5 had means of 27 ppm and 24 ppm respectively while L-7, the most downstream station, had a mean of 26 ppm (Table 5-4). This differed from suspended solids concentrations observed in 1987 and 1988, which decreased as water moved through the system. Upstream stations, L-1 and L-5 had the highest levels of suspended solids in 1988 with mean concentrations of 49 ppm and 45 ppm, respectively. Both stations L-1 and L-5 receive direct discharge from major tributaries (Figure 5-1) but the dramatic increase in suspended solids loadings from West Creek in 1989 was not translated into higher suspended solids in the lake.

Seasonal trends for suspended solids are presented in Figure 5-7. In-lake concentrations rose sharply at all stations in early June 1989. Stations L-1 and L-5 were above the mean and station L-7 was below the mean for this period. This trend reversed somewhat following the runoff events in late June and early July. Station L-4 was consistently below the mean of all stations during 1989. Mean suspended solids concentrations for interlake sites were similar to in-lake stations in 1989 except for Round Crossing (IL-2) which had a mean concentration of 45 ppm (Table 5-4).

IN-LAKE SUSPENDED SOLIDS

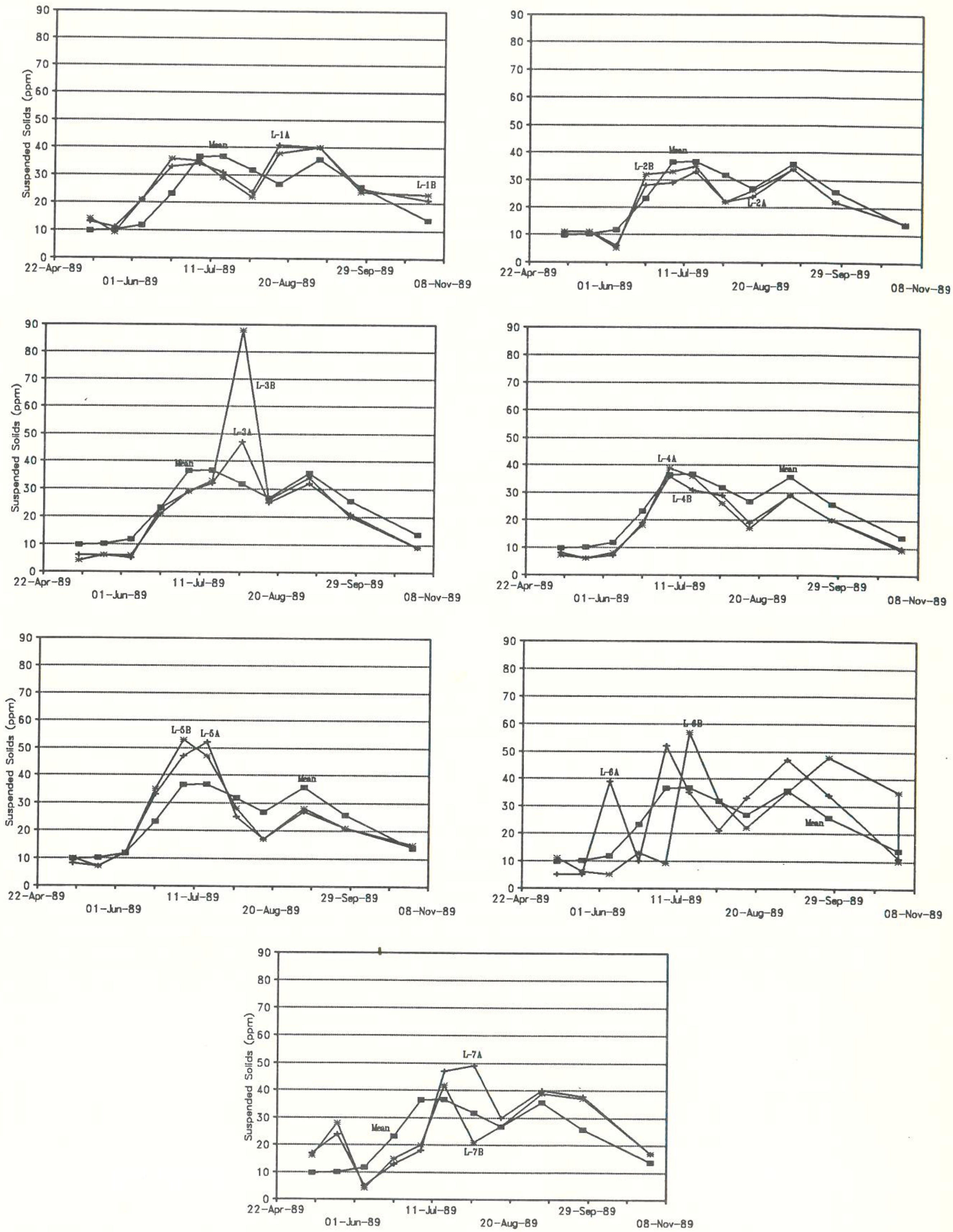


Figure 5-7. Suspended solids concentrations for in-lake stations in 1989.

Inorganic Nitrogen as N

Inorganic nitrogen is a combination of ammonia, nitrite and nitrate. It represents the forms of nitrogen that are immediately available for algal growth. Ammonia was the dominant form of inorganic nitrogen in 1989. A distribution of individual stations about the mean for all stations is presented in Figure 5-8. Inorganic nitrogen was available at all stations from May through mid July 1989. Station L-7 exhibited concentrations below the mean and station L-6 exhibited above the mean through this period (Figure 5-8). The sharp increase in inorganic nitrogen at station L-6 in June was also observed in 1988. From mid July through September concentrations remained low at all stations. Inorganic nitrogen, dominated by ammonia, increased sharply in October 1989 as the summer algal bloom declined. Ammonia nitrogen is an inorganic form of nitrogen in lakes derived from animal excretion products or the decay of organic material. Ammonia levels in the Oakwood Lakes ranged from below detectable limits at several stations, to a maximum of 2.68 ppm at L-6 which also had the highest mean concentration for 1989 at .75 ppm (Table 5-4).

Nitrite concentrations were generally insignificant at all stations in 1989 (Table 5-4). Nitrate concentrations ranged from .01 ppm at several stations to .14 ppm at L-4 and L-5. Mean concentrations ranged from .06 ppm at stations L-2, L-6 and L-7 to .08 ppm at stations L-1 and L-5. Nitrate levels tended to decline as water moved downstream through the system but not as dramatically as other parameters. Nitrate concentrations were similar at interlake sites except IL-3 which had a mean concentration of .15 ppm.

Organic Nitrogen as N

Organic nitrogen ranged from 1.37 ppm at station L-7 to 4.10 ppm at station L-3 in 1989. Means were higher in 1988 than 1987, and ranged from 2.14 ppm at L-2 and L-4 to 2.58 ppm at L-6. Lakes with mean concentrations of organic nitrogen above 1.2 ppm are classified as hypereutrophic (Wetzel, 1983). Organic nitrogen concentrations in 1989 indicate that all of the Oakwood Lakes continue to be hypereutrophic. Round Crossing had a mean concentration of 3.86 ppm organic nitrogen but flows were very low and of short duration. Mortimer's Slough is probably the most eutrophic basin in the system although there is no in-lake monitoring station to document the trophic level. Mean values for organic nitrogen at Mortimer's crossing in 1989 are lower than observed in 1987 and 1988 because much of the flow was from West Oakwood lake <u>into</u> Mortimer's Slough following the large runoff event at West Creek on July 11.

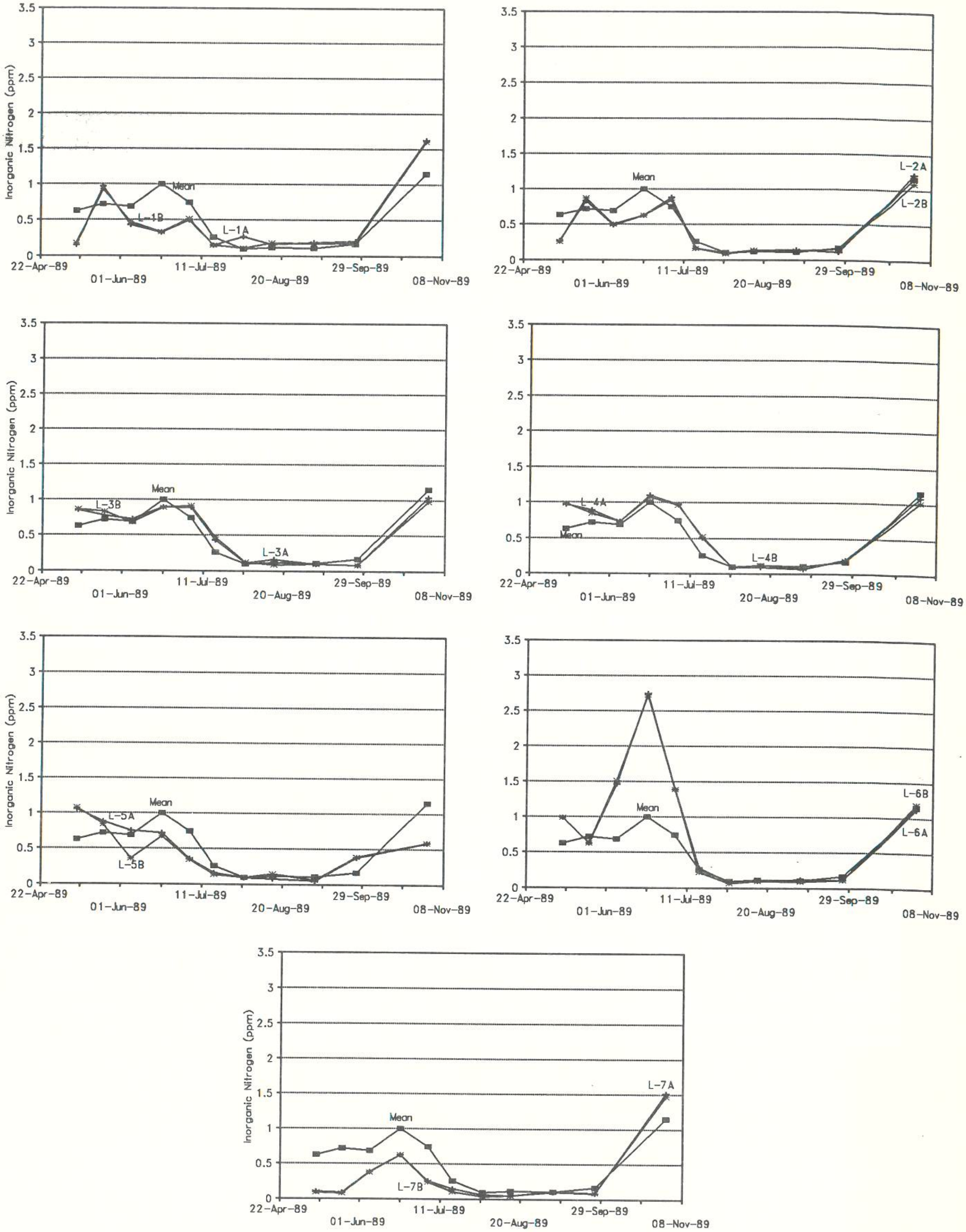


Figure 5-8. Inorganic nitrogen concentrations for in-lake stations in 1989.

Total Phosphorus as P

Phosphorus is frequently a limiting nutrient in freshwater lakes. Numerous investigators have found a strong correlation between total PO_4-P concentrations and the trophic state of the lake (Wetzel, 1983). A trophic state index based on phosphorus put forth by Reckhow, et al (1983), would classify lakes with between .020 and .050 ppm as eutrophic and would classify lakes with total PO_4-P concentrations greater than .050 ppm as hypertrophic. According to this classification, the Oakwood Lakes system has several times more total PO_4-P than would be required to place it in the hypereutrophic classification.

In-lake total PO_4-P ranged from .060 ppm at stations L-3, L-4 and L-7 to .330 ppm at station L-1 in 1989. Mean concentrations were lower in 1989 than in either 1987 or 1988 and ranged from .089 ppm at station L-3 to .145 ppm at station L-1. Total PO_4-P concentrations for interlake sites were higher than in-lake stations above them. Significantly higher total PO_4-P loading to West Oakwood Lake in 1989 did not result in higher in-lake concentrations. A lag time longer than one season may exist between tributary loadings and in-lake concentrations. This factor may complicate modeling of the lake system to estimate the effect of tributary loadings on in-lake water quality.

A decline in total PO_4-P concentrations was observed as water moved through the system in all three years. Upstream stations (L-1 and L-5) had consistently higher concentrations than other stations. A distribution of individual stations about the mean for all stations is presented in Figure 5-9. All of the stations followed a similar seasonal trend in total PO_4-P concentrations during 1989. A peak in total PO_4-P was observed at stations L-1 on July 17th following the runoff event at West Creek on July 11. Station L-4 had consistently less total PO_4-P than other stations throughout the year. There was no total PO_4-P peak in August of 1989, as was observed in 1988 (Figure 5-9).

Total Orthophosphorus as P

Total ortho PO_4-P (or soluble reactive phosphorus) is a measure of the phosphorus that is considered immediately available for uptake by phytoplankton. A distribution of individual stations about the mean for all stations in 1989 is presented in Figure 5-10. The seasonal trends were similar between stations with low concentrations in early June, late July and September. Ortho PO_4-P concentrations ranged from below detection limits at all stations to .070 ppm at station L-7. Mean concentrations for each station ranged from .011 ppm at station L-6 to .023 ppm at L-1 (Table 5-4). Although ortho PO_4-P concentrations were low the week following the July 11 storm concentrations increased during both August sampling dates (Figure 5-10).

IN-LAKE TOTAL PHOSPHORUS

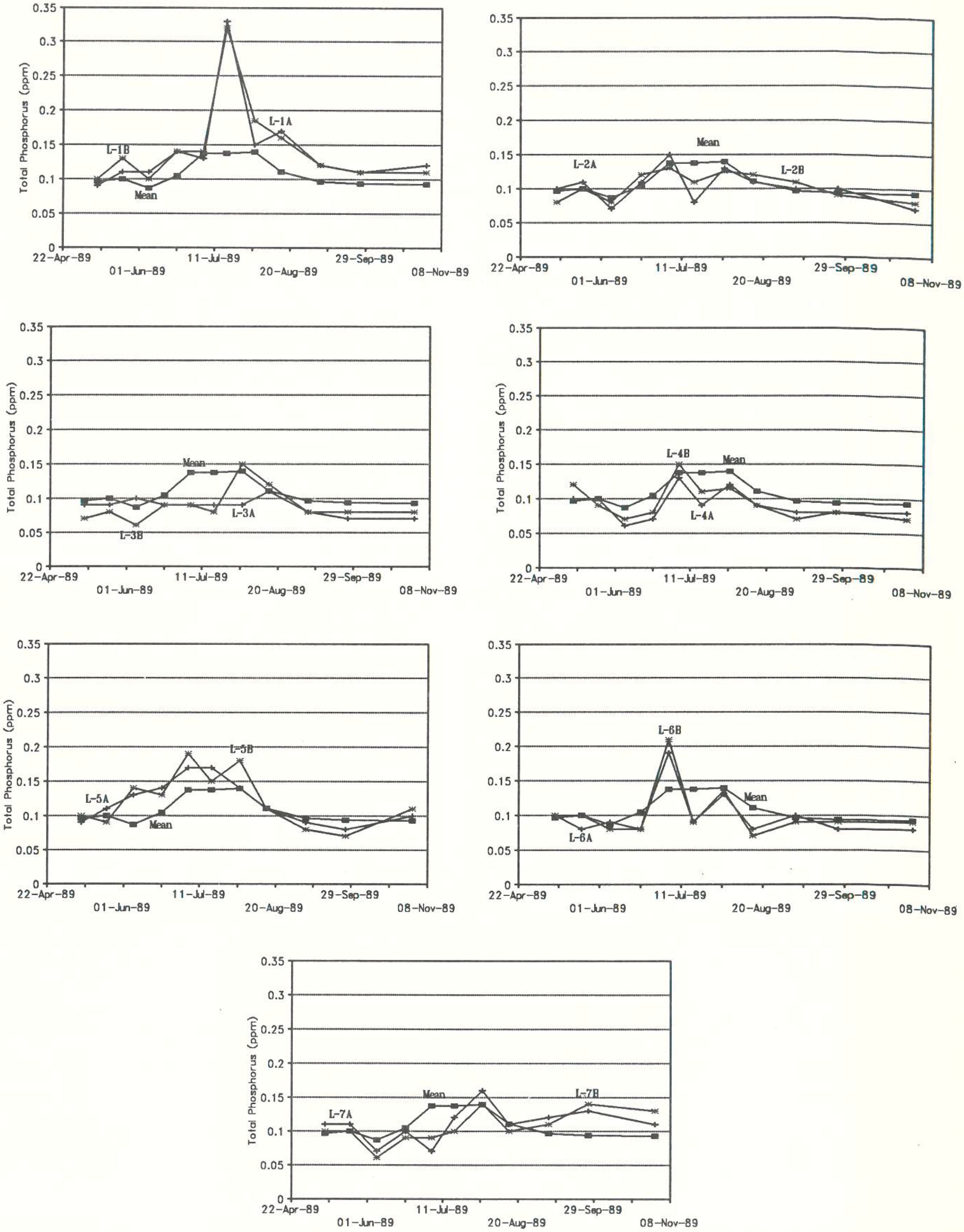


Figure 5-9. Total phosphorus concentrations for in-lake stations in 1989.

IN-LAKE ORTHO PHOSPHORUS

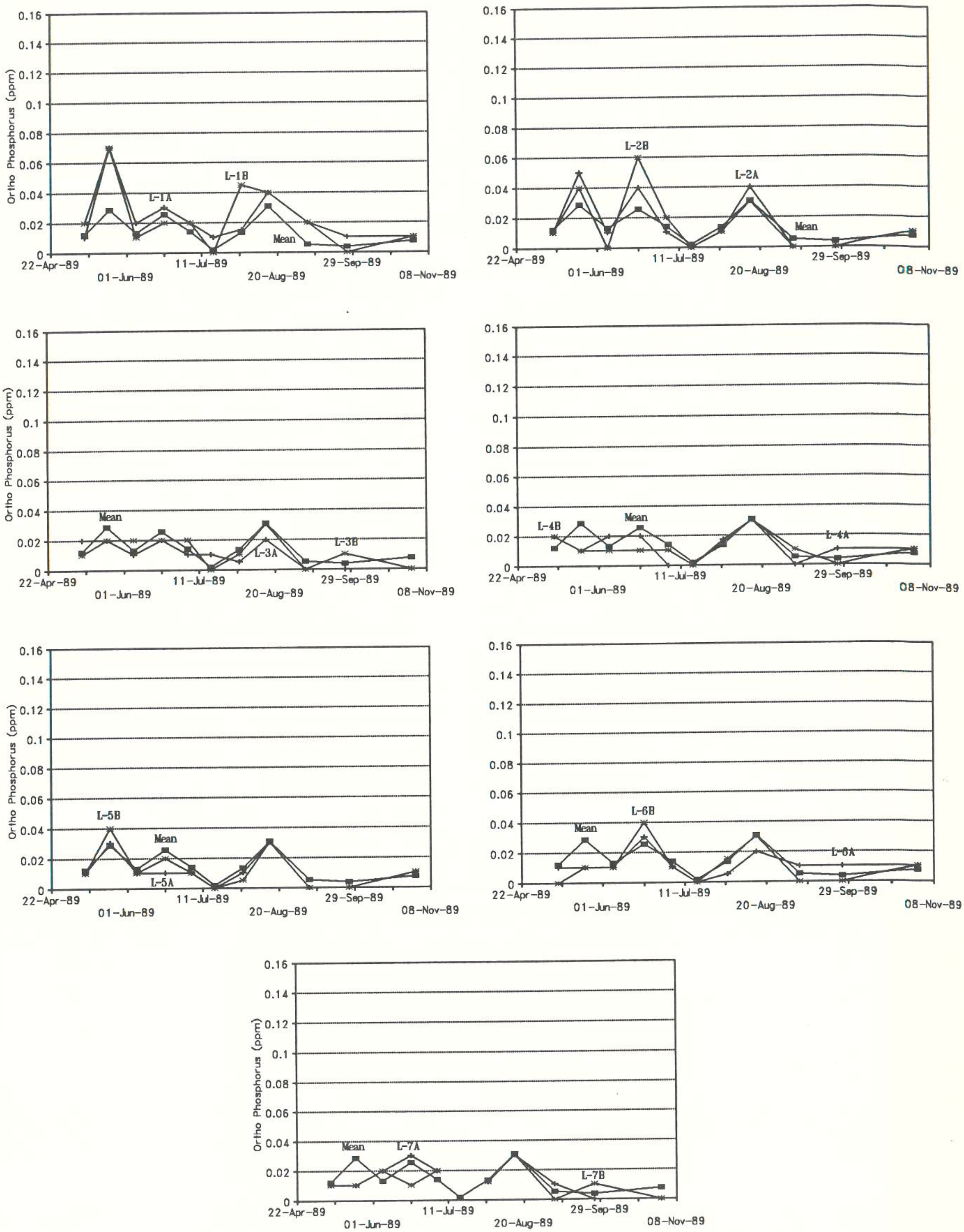


Figure 5-10. Ortho phosphorus concentrations for in-lake stations in 1989.

Dissolved Oxygen

Dissolved oxygen concentrations in the Oakwood Lakes exhibited strong seasonal variations. On most sampling dates oxygen was fairly uniform throughout the water column. Oxygen stratification occurred in the Oakwood lakes in 1989 but was ephemeral. Stratification occurred during periods of no wind, warm weather and high algal biomass. Representative dissolved oxygen profiles for in-lake stations on several dates are presented in Figure 5-11A and 5-11B. Stratification was also observed on June 23, July 16 and August 8 at all stations. The pattern of stratification was similar to those in Figure 5-11A and 5-11B on those dates. Anoxic conditions at the sediment water interface are probably occurring during these events which would facilitate the release of phosphorus from the sediments. Stratification occurred most frequently at the end of July and in August which coincided with the largest increase of in-lake ortho PO_4-P concentrations (Figure 5-10).

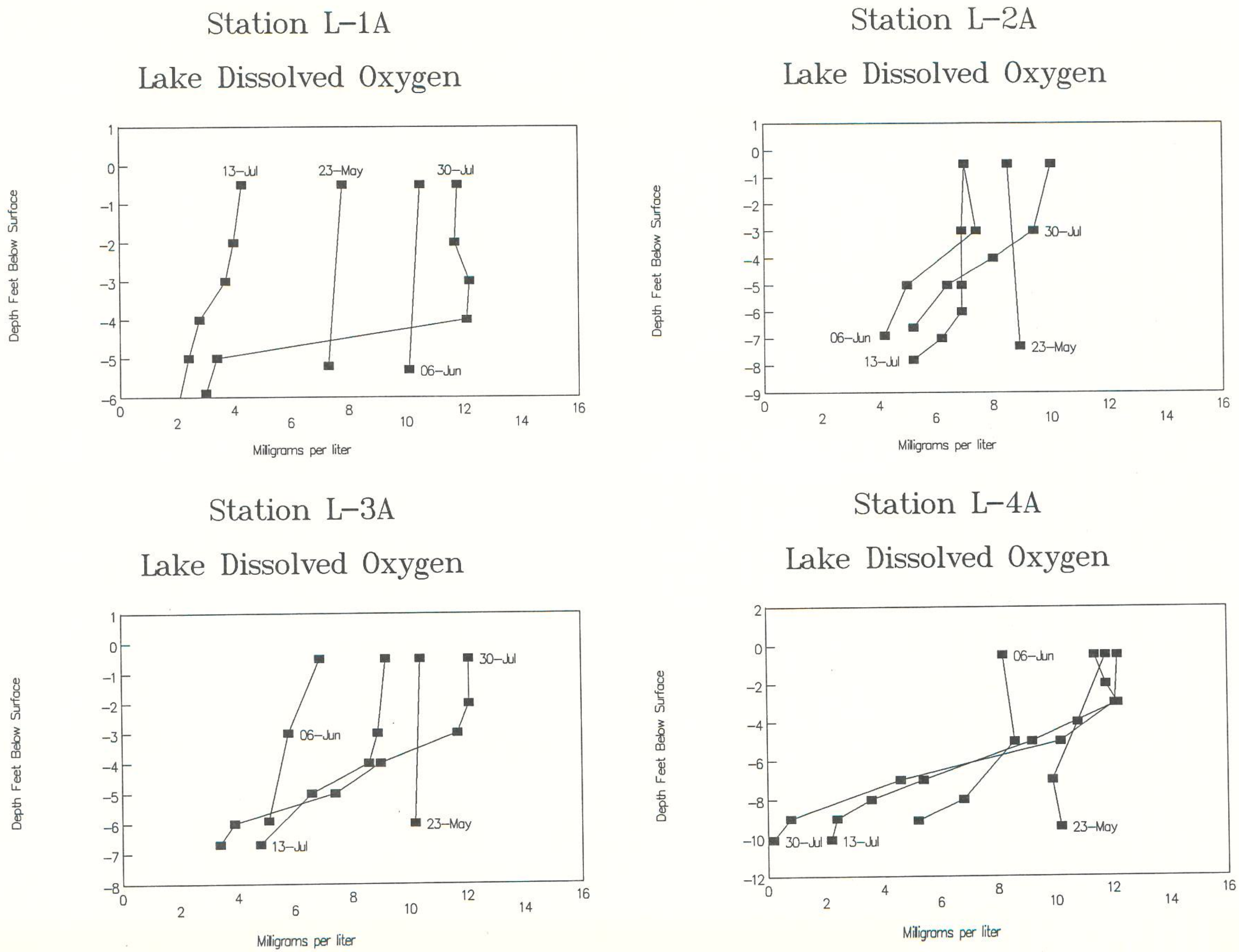


Figure 5-11A. Dissolved oxygen profiles for in-lake stations L-1 through L-4.

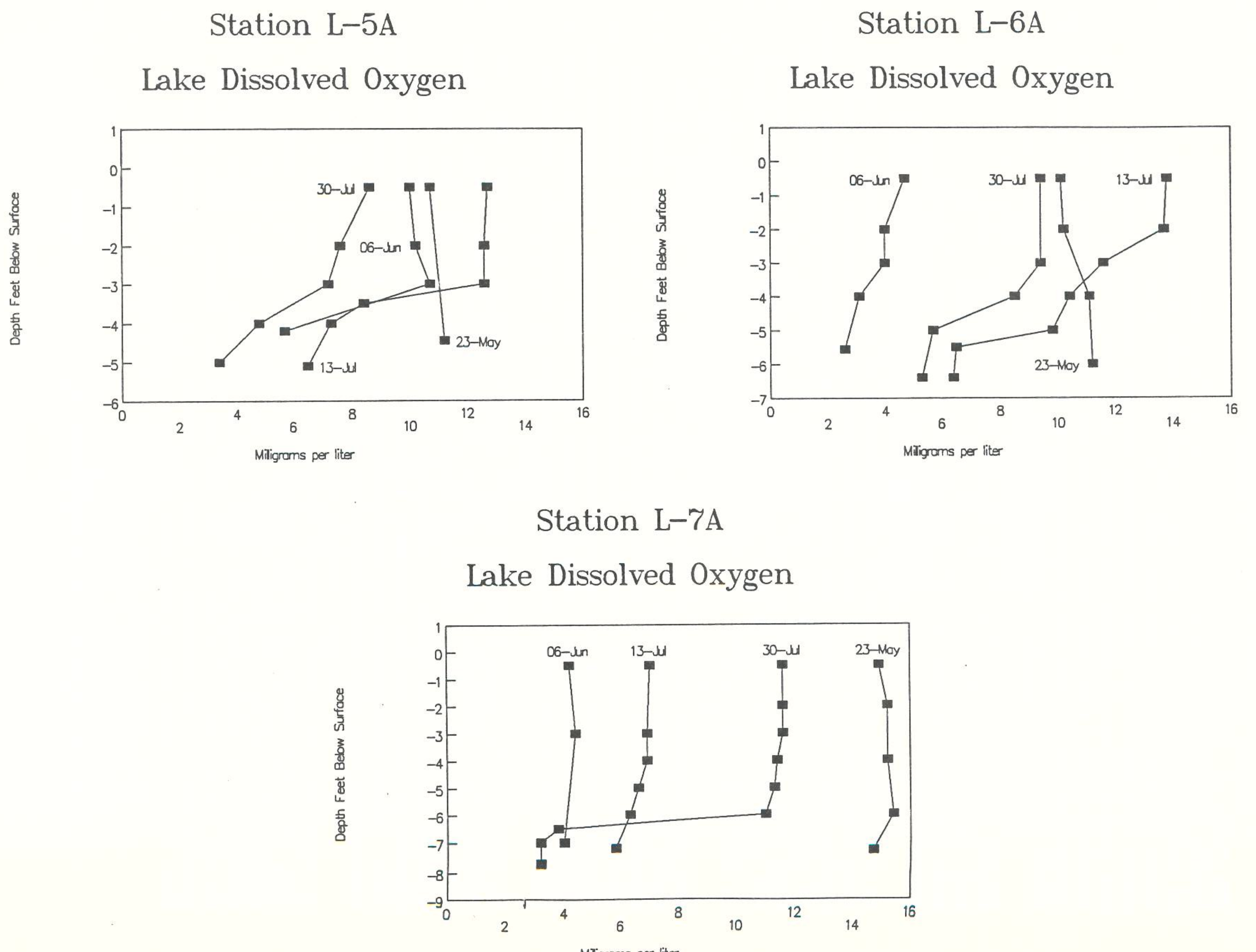


Figure 5-11B. Dissolved oxygen profiles for in-lake stations L-5 through L-7.

Limiting Nutrients

The ratio of nitrogen to phosphorus can determine which nutrient controls or limits the maximum biomass developed during the summer period. The evaluation of limiting nutrients during rapid growth periods is based on the available forms (inorganic nitrogen and ortho PO_4-P). N:P ratios of less than 10 (10 parts nitrogen to one part phosphorus) suggest nitrogen limitation. Ratios of N:P greater than 17 suggest phosphorus limitation. Ratios between 10 and 17 indicate either nutrient could be limiting. To be considered limiting the concentrations of available nutrients have to be close to zero (Forsberg, 1980).

Ratios of available N:P were calculated for all Oakwood Lakes stations for each date during 1987, 1988 and 1989, but were included in Table 5-5 only if mean available phosphorus and nitrogen were less than .012 ppm or .18 ppm, respectively.

In 1987 ratios of N:P below 10 were observed at several stations in April and late June (Table 5-5). Since inorganic nitrogen levels were relatively low during this time nitrogen was potentially limiting. With N:P ratios between 10 and 17 at L-1, L-2 and L-6 in April either nitrogen or phosphorus could have been limiting at those stations. Phosphorus appeared to be limiting at L-4 and L-7 at the end of April. In August 1978 nutrients did not appear to be limiting algal growth.

The most apparent period of nitrogen limitation in 1988 occurred in April and early May at most stations. This may have contributed to the minimum in algal biomass that was observed on May 24th. Round Lake (L-6) was phosphorus limited earlier in the year than other stations (Table 5-5). Phosphorus and nitrogen were both at low concentrations on July 19,1988, and nitrogen was limiting at all stations including Round Lake. This date appears to mark the point at which the lakes switched from nitrogen to phosphorus limitation.

In general, the lakes exhibited more nitrogen limitation in 1987 than 1988 (Table 5-5). Nutrients did not appear to be limiting algal growth in August 1987, during the peak of the algal bloom. In 1988, the lake was phosphorus limited during this period. Conversely, the lakes were nitrogen limited in January of 1987, but had extremely high concentrations of nitrogen in January 1988. These differences may have contributed to the shift in algal species composition that was observed in 1988.

In 1989 nutrients were limiting occasionally at only a few stations until mid July when most stations became phosphorus limited for a short time. All stations had become nitrogen limited by mid August. The pattern of nutrient limitation switching back and forth through the open water season was observed in all three years of study. This feature of the Oakwood Lakes probably contributes to fluctuations in species composition of the algae population that is also observed in the lakes.

Table 5-5. Inorganic Nitrogen to Inorganic Phosphorus Ratios for Oakwood Lake Stations 1988.

	Johnson Lake (L-1)		Johnson Lake (L-2)		West Oakwood Lake (L-3)		West Oakwood Lake (L-4)		Turtle Lake (L-5)		Round Lake (L-6)		East Oakwood Lake (L-7)	
22-Jan-87	6	N	4	N	2	N	3	N	2	N	3	N	2	N
12-Apr-87	57	P	22	P	8	N	10	N or P	6	N	27	P	23	P
28-Apr-87	14	N or P	13	N or P	21	P	5	N	2	N	8	N	37	P
14-May-87	3	N	3	N	2	N	1	N	1	N	2	N	6	N
27-May-87					3	N					7	N	4	N
09-Jun-87	2	N	1	N	3	N	4	N	5	N	5	N	4	N
23-Jun-87	4	N	3	N	14	N or P	21	P	4	N	19	P	5	N
06-Jul-87			56	P	162	P	206	P	84	P	39	P	276	P
20-Jul-87			32	P	21	P	30	P			31	P	18	P
04-Aug-87							4	N						
17-Aug-87														
09-Sep-87	20	P	61	P	147	P	255	P	72	P	27	P	135	P
23-Sep-87	2	N	3	N	2	N	3	N	3	N				
21-Oct-87														
18-Nov-87														
18-Jan-88														
01-Mar-88														
19-Apr-88	no data		no data		no data		7	N	3	N	no data		3	N
11-May-88	1	N	1	N	1	N	1	N	5	N	4	N	7	N
24-May-88					5	N	7	N	3	N	29	P		
08-Jun-88	10	N or P	8	N	14	N or P	3	N	2	N	208	P	3	N
20-Jun-88	13	N or P	10	N or P	13	N or P	12	N or P	10	N or P	44	P	4	N
05-Jul-88	22	P	9	N	8	N	14	N or P	19	P	15	N or P	10	N or P
19-Jul-88	5	N	4	N	8	N	5	N	8	N	3	N	8	N
01-Aug-88	18	P	26	P	16	N or P	166	P	24	P	13	N or P	116	P
15-Aug-88			37	P	206	P	216	P	41	P	26	P	19	P
06-Sep-88	15	N	21	P			19	P	21	P	17	P	84	P
27-Sep-88	48	P	52	P	254	P	373	P	177	P	152	P	387	P
25-Oct-88	188	P	179	P	225	P	217	P	249	P			295	P
10-May-89	11	N or P	26	P					106	P	198	P	10	N or P
23-May-89							87	P			63	P	8	N
06-Jun-89			99	P					56	P	149	P		
21-Jun-89														
05-Jul-89							195	P	35	P	139	P		
17-Jul-89	30	P	32	P	90	P	103	P	30	P	50	P	no data	
01-Aug-89			9	N	15	N or P	6	N	13	N or P	7	N	4	N
15-Aug-89	4	N	4	N	5	N	4	N	4	N	4	N	2	N
05-Sep-89	9	N	27	P	19	P	14	N or P	11	N or P	21	P	21	P
26-Sep-89	38	P	26	P	16	N or P	39	P	77	P	23	P	17	N or P
31-Oct-89	162	P	115	P	203	P	105	P	59	P	116	P	299	P

5.3.5 OLSS Groundwater Quality

OLSS drilling provided more detailed information on the geology of the area which will improve estimates of the contributing ground water areas. A new isopach map was prepared indicating current knowledge of the thickness of sand and gravel areas surrounding the lake (Figure 5-12). The areas of greatest thickness of sand and gravel are located along the southwest sides of Turtle, West Oakwood and East Oakwood Lakes. These same areas produce the largest rates of seepage into the lake during all three years.

Only those wells drilled for OLSS monitoring purposes will be included in the discussion at this time. Minimum maximum and mean values for selected parameters for land and in-lake wells are presented in Table 5-6. Locations of wells can be found on Figure 5-2. Land wells have a "GW" prefix and in-lake wells have a "W" prefix. Wells positioned in well nests are indicated by (S) shallow, (M) mid depth or (D) deepest well.

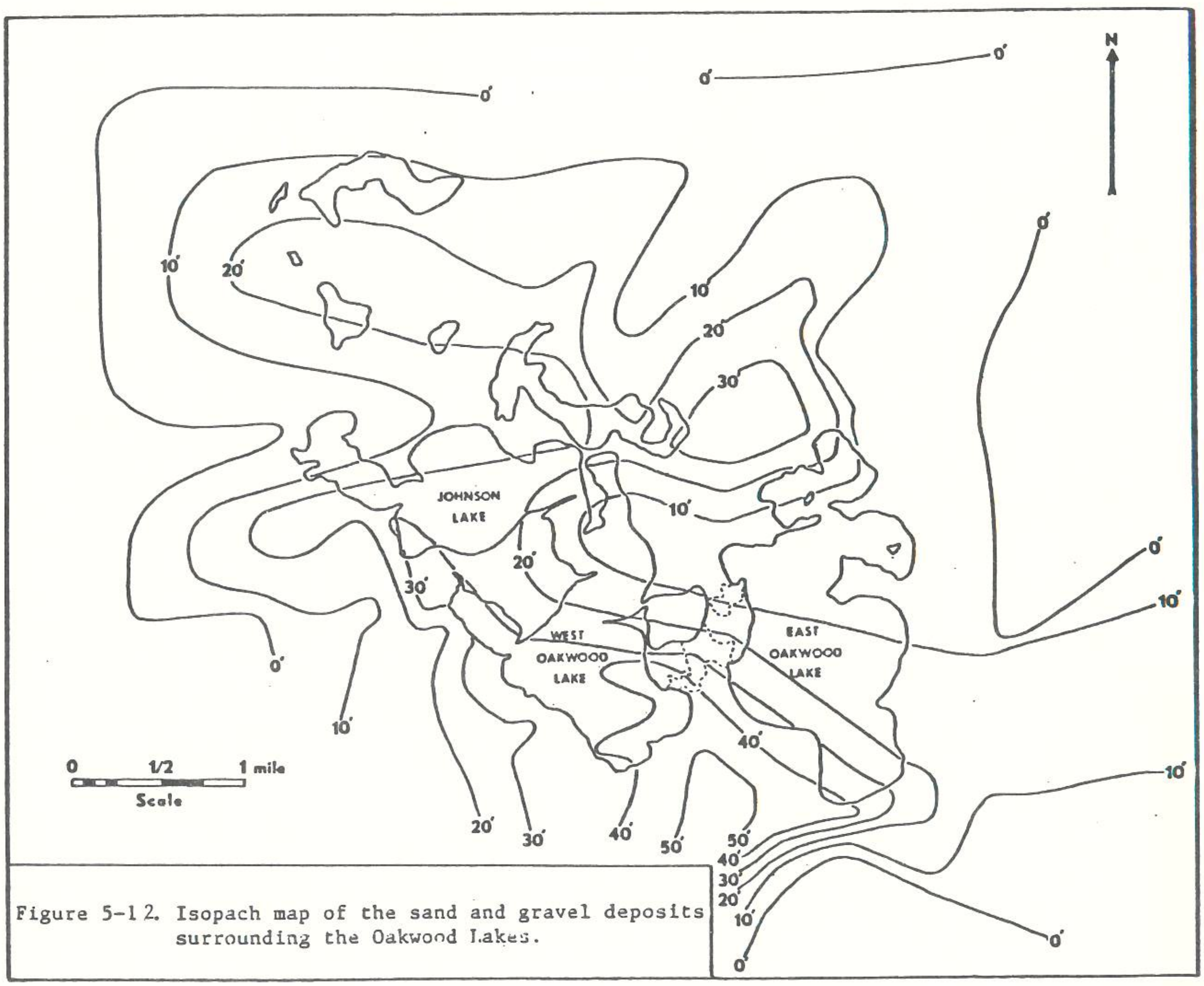


Figure 5-12. Isopach map of the sand and gravel deposits surrounding the Oakwood Lakes.

Table 5-6. Minimum, Maximum, and Mean Values of Selected Water Quality Parameters for In-lake and Land Wells

Well ID		Total Dissolved Solids (ppm)	Chloride (ppm)	Sulfate (ppm)	Nitrite as N (ppm)	Nitrate as N (ppm)	Ammonia as N (ppm)	Organic Nitrogen as N (ppm)	Total Nitrogen as N (ppm)	Dissolved PO_4 as P (ppm)
GW-1	min	604	42	165	0.000	0.04	0.00	0.01	0.113	0.00
	max	1940	128	724	0.018	2.73	0.08	0.25	2.982	0.02
	mean	1380	85	541	0.006	0.71	0.03	0.16	0.907	0.01
GW-2	min	370	1	54	0.001	0.43	0.00	0.00	0.539	0.01
	max	528	3	81	0.029	3.36	0.08	0.19	3.551	0.07
	mean	446	2	72	0.009	1.23	0.03	0.11	1.385	0.03
GW-3 (S)	min	404	3	88	0.000	0.41	0.00	0.04	0.520	0.01
	max	540	8	108	0.013	0.99	0.10	0.17	1.110	0.07
	mean	464	4	97	0.005	0.64	0.03	0.09	0.763	0.02
GW-3 (D)	min	1584	5	680	0.000	0.00	0.72	0.00	0.862	0.00
	max	1780	11	970	0.009	0.23	1.03	0.17	1.160	0.10
	mean	1677	7	870	0.003	0.04	0.90	0.08	1.025	0.02
GW-5	min	1740	1	912	0.000	0.00	0.24	0.00	0.249	0.00
	max	2260	4	1185	0.009	0.04	0.33	0.27	0.561	0.04
	mean	2081	3	1072	0.003	0.02	0.29	0.10	0.408	0.02
GW-6	min	1680	2	506	0.001	0.00	1.67	0.00	1.674	0.02
	max	1888	11	1075	0.077	0.45	2.15	0.14	2.570	0.06
	mean	1766	6	878	0.014	0.08	1.96	0.07	2.114	0.03
GW-7	min	1408	3	750	0.000	0.00	0.93	0.00	0.945	0.00
	max	1892	8	875	0.064	0.04	1.32	0.13	1.497	0.04
	mean	1641	5	825	0.007	0.02	1.14	0.08	1.240	0.01
GW-8 (S)	min	868	1	285	0.000	0.00	0.42	0.00	0.572	0.01
	max	1156	3	425	0.018	0.12	0.62	0.14	0.751	0.03
	mean	1034	2	386	0.004	0.02	0.55	0.08	0.653	0.02
GW-8 (D)	min	1004	1	369	0.000	0.00	0.41	0.00	0.443	0.01
	max	1360	3	469	0.007	0.04	0.66	0.12	0.780	0.04
	mean	1117	2	423	0.002	0.01	0.58	0.06	0.655	0.02
GW-9	min	336	3	44	0.000	4.28	0.00	0.14	4.451	0.08
	max	508	10	86	0.021	8.28	0.06	0.41	8.560	0.18
	mean	418	6	66	0.003	7.04	0.02	0.28	7.345	0.12
GW-11 (S)	min	484	2	63	0.000	12.35	0.00	0.28	12.738	0.00
	max	912	9	80	0.028	20.88	0.08	0.70	21.467	0.08
	mean	650	5	72	0.004	16.70	0.02	0.49	17.221	0.03
GW-11 (M)	min	412	1	49	0.000	3.02	0.00	0.12	3.202	0.00
	max	576	5	64	0.020	6.61	0.08	0.29	6.901	0.08
	mean	476	3	56	0.005	5.02	0.02	0.21	5.258	0.02
GW-11 (D)	min	380	0	141	0.000	0.00	0.00	0.00	0.022	0.00
	max	648	3	188	0.007	0.06	0.08	0.11	0.142	0.06
	mean	540	2	167	0.002	0.02	0.02	0.04	0.080	0.02
GW-13 (S)	min	496	3	64	0.000	4.08	0.00	0.29	4.402	0.00
	max	668	8	111	0.008	8.61	0.16	0.44	9.081	0.03
	mean	617	6	79	0.002	6.17	0.04	0.38	6.588	0.01
GW-13 (D)	min	660	2	224	0.000	0.00	0.07	0.00	0.122	0.00
	max	822	3	281	0.008	0.16	0.44	0.16	0.738	0.02
	mean	737	2	251	0.002	0.05	0.23	0.07	0.344	0.01

Table 5-6 continued

Well ID		Total Dissolved Solids (ppm)	Chloride (ppm)	Sulfate (ppm)	Nitrite as N (ppm)	Nitrate as N (ppm)	Ammonia as N (ppm)	Organic Nitrogen as N (ppm)	Total Nitrogen as N (ppm)	Dissolved PO_4 as P (ppm)
GW-14	min	444	1	136	0.000	0.00	0.00	0.01	0.050	0.00
(S)	max	576	4	181	0.009	0.04	0.10	0.12	0.184	0.03
	mean	535	3	153	0.002	0.01	0.04	0.06	0.108	0.02
GW-14	min	564	2	192	0.000	0.00	0.00	0.03	0.073	0.00
(D)	max	772	6	236	0.006	0.04	0.14	0.14	0.276	0.06
	mean	640	3	214	0.002	0.01	0.10	0.07	0.186	0.02
GW-16	min	488	2	168	0.000	0.00	0.04	0.01	0.125	0.00
(S)	max	936	7	248	0.020	0.38	0.14	0.33	0.650	0.08
	mean	769	4	201	0.003	0.05	0.08	0.12	0.251	0.03
GW-16	min	776	13	300	0.000	0.00	0.02	0.00	0.093	0.00
(D)	max	1080	24	395	0.033	0.04	0.11	0.36	0.433	0.07
	mean	942	17	357	0.004	0.01	0.05	0.12	0.192	0.02
GW-18	min	832	1	337	0.000	0.00	0.14	0.02	0.264	0.00
	max	1048	4	380	0.013	0.16	0.31	0.32	0.693	0.10
	mean	933	2	361	0.003	0.04	0.26	0.13	0.425	0.03
GW-19	min	584	1	200	0.000	0.00	0.10	0.00	0.103	0.00
	max	816	6	278	0.008	0.09	0.41	0.34	0.676	0.06
	mean	724	3	239	0.003	0.05	0.20	0.16	0.405	0.03
GW-20	min	436	3	135	0.000	0.00	0.09	0.09	0.212	0.00
	max	644	8	214	0.007	0.04	0.30	0.24	0.515	0.06
	mean	528	5	171	0.002	0.02	0.18	0.15	0.347	0.03
GW-21	min	500	1	162	0.000	0.00	0.12	0.01	0.203	0.00
(S)	max	684	4	192	0.006	0.05	0.29	0.13	0.366	0.06
	mean	612	2	179	0.002	0.02	0.23	0.05	0.296	0.02
GW-21	min	464	1	175	0.000	0.00	0.14	0.02	0.183	0.00
(D)	max	680	4	212	0.006	0.04	0.29	0.20	0.416	0.04
	mean	628	2	187	0.001	0.01	0.23	0.05	0.298	0.02
GW-22	min	480	4	76	0.000	0.84	0.02	0.09	1.020	0.00
(S)	max	748	10	230	0.016	4.28	0.16	0.45	4.830	0.10
	mean	582	6	137	0.004	2.86	0.07	0.24	3.172	0.03
GW-22	min	540	1	192	0.001	0.00	0.72	0.11	1.022	0.04
(M)	max	756	3	222	0.007	0.34	1.03	0.30	1.316	0.15
	mean	653	2	207	0.004	0.09	0.88	0.16	1.134	0.08
GW-22	min	572	1	210	0.000	0.00	0.74	0.06	0.984	0.00
(D)	max	780	6	314	0.010	0.24	1.04	0.29	1.231	0.12
	mean	713	3	248	0.004	0.05	0.91	0.14	1.101	0.05
GW-24	min	1052	4	412	0.000	0.02	0.00	0.17	0.241	0.00
	max	1316	17	551	0.008	1.82	0.11	0.35	2.094	0.06
	mean	1166	9	497	0.003	0.59	0.04	0.23	0.863	0.02
GW-25	min	456	2	272	0.001	0.00	0.38	0.10	0.572	0.00
	max	924	6	360	0.012	0.04	0.62	0.39	0.882	0.05
	mean	804	4	311	0.003	0.02	0.49	0.17	0.690	0.02
W1-1	min	3940	4	2300	0.003	0.06	0.69	0.98	1.876	0.02
(S)	max	5060	8	2856	0.017	0.35	1.56	1.87	3.493	0.02
	mean	4372	6	2612	0.007	0.17	1.11	1.32	2.605	0.02
W1-2	min	3936	5	2240	0.005	0.04	1.69	1.01	3.964	0.00
(S)	max	4744	6	2335	0.024	0.18	3.17	2.36	4.477	0.02
	mean	4276	5	2295	0.010	0.10	2.64	1.40	4.153	0.01
W2-1	min	520	2	355	0.001	0.00	0.05	0.13	0.191	0.01
(S)	max	1164	5	372	0.005	0.00	0.17	0.35	0.505	0.02
	mean	864	4	362	0.003	0.00	0.11	0.19	0.308	0.01

Table 5-6 continued

Well ID		Total Dissolved Solids (ppm)	Chloride (ppm)	Sulfate (ppm)	Nitrite as N (ppm)	Nitrate as N (ppm)	Ammonia as N (ppm)	Organic Nitrogen as N (ppm)	Total Nitrogen as N (ppm)	Dissolved PO_4 as P (ppm)
W2-2	min	952	1	385	0.002	0.00	0.53	0.09	0.706	0.01
(D)	max	1156	3	400	0.008	0.04	0.66	0.34	1.028	0.02
	mean	1009	2	393	0.005	0.03	0.58	0.24	0.850	0.01
W2-2	min	888	1	360	0.003	0.03	0.52	0.15	0.733	0.02
(S)	max	1148	3	365	0.022	0.07	0.61	0.25	0.862	0.03
	mean	983	2	363	0.009	0.04	0.55	0.19	0.796	0.02
W3-1	min	3288	4	1560	0.009	0.18	4.57	0.82	5.597	0.01
(S)	max	3296	6	1775	0.027	0.29	6.50	0.85	7.649	0.03
	mean	3292	5	1668	0.018	0.24	5.54	0.84	6.623	0.02
W3-2	min	4282	2	2130	0.005	0.10	16.80	0.60	17.955	0.20
(S)	max	4812	15	2150	0.014	0.16	17.25	2.26	19.234	0.21
	mean	4547	8	2140	0.010	0.13	17.03	1.43	18.595	0.21
W4-1	min	908	14	371	0.001	0.03	0.00	0.09	0.145	0.01
(D)	max	1120	16	385	0.005	0.15	0.56	0.29	1.005	0.36
	mean	1027	15	378	0.004	0.07	0.23	0.16	0.464	0.13
W4-1	min	892	11	385	0.001	0.03	0.08	0.14	0.262	0.01
(S)	max	1004	11	385	0.012	0.04	0.21	0.69	0.941	0.54
	mean	948	11	385	0.006	0.04	0.15	0.42	0.602	0.28
W5-1	min	852	1	420	0.001	0.00	0.12	0.05	0.261	0.01
(D)	max	1012	2	450	0.016	0.04	0.38	0.14	0.486	0.03
	mean	942	2	437	0.006	0.01	0.28	0.09	0.388	0.02
W5-1	min	924	1	466	0.001	0.00	0.36	0.13	0.491	0.01
(S)	max	1116	45	500	0.078	0.04	0.71	0.29	0.923	0.05
	mean	1037	16	485	0.022	0.02	0.52	0.19	0.747	0.04
W5-2	min	2472	1	1128	0.002	0.02	3.04	0.69	3.774	0.01
(S)	max	2980	2	1172	0.010	0.06	6.40	1.24	7.332	0.04
	mean	2768	1	1144	0.006	0.04	3.94	0.98	4.961	0.02
W7-2	min	2668	8	1528	0.001	0.00	0.76	0.67	1.536	0.00
(D)	max	2988	11	1625	0.026	0.18	1.04	0.80	2.036	0.02
	mean	2794	10	1571	0.013	0.07	0.90	0.74	1.730	0.01
W7-2	min	2044	10	1049	0.000	0.00	0.13	0.52	0.006	0.00
(S)	max	4224	408	1100	0.006	0.08	0.38	0.75	1.193	0.07
	mean	2867	143	1075	0.003	0.05	0.25	0.64	0.633	0.02
W7-3	min	4348	5	2320	0.001	0.04	0.66	1.40	2.551	0.00
(D)	max	4956	6	2325	0.005	0.06	1.32	1.85	2.862	0.03
	mean	4611	6	2323	0.003	0.05	1.05	1.63	2.733	0.01
W7-3	min	3384	3	1688	0.007	0.10	4.78	1.93	7.726	0.00
(S)	max	4368	4	1975	0.044	0.22	13.60	2.83	15.766	0.04
	mean	3999	3	1809	0.021	0.17	10.62	2.32	13.133	0.02
W7-4	min	924	2	328	0.001	0.00	1.39	0.36	1.961	0.00
(S)	max	3384	52	1688	0.007	0.16	11.50	2.09	13.757	0.12
	mean	1494	15	677	0.003	0.05	3.60	0.81	4.463	0.05
W7-5	min	1672	3	610	0.002	0.03	11.30	0.73	12.136	0.00
(S)	max	2716	113	720	0.006	0.10	14.80	1.83	16.462	1.14
	mean	2012	40	673	0.005	0.07	13.43	1.10	14.598	0.46
W7-6	min	1250	6	520	0.002	0.00	2.30	0.75	3.092	0.00
(D)	max	1256	8	569	0.007	0.22	3.07	0.93	4.024	0.17
	mean	1253	7	545	0.004	0.09	2.76	0.86	3.708	0.06
W7-6	min	916	6	296	0.001	0.03	1.72	0.68	2.513	0.00
(S)	max	1268	8	415	0.019	0.10	2.88	0.91	3.896	0.23
	mean	1108	7	356	0.007	0.07	2.47	0.79	3.335	0.09
W7-7	min	1184	3	410	0.002	0.03	3.53	0.67	4.262	0.00
(S)	max	1848	4	426	0.011	0.06	5.17	3.42	8.622	0.47
	mean	1409	4	418	0.005	0.04	4.16	1.64	5.848	0.28

Total Nitrogen as N

Total nitrogen as N concentrations observed in wells surrounding the Oakwood Lakes will be used to calculate the ground water contributions to the nitrogen budget for the Oakwood Lakes. Land wells that exhibited the highest total nitrogen as N concentrations are plotted at a larger scale in Figure 5-13A. Wells with less than 3 ppm total nitrogen as N are presented in Figure 5-13B. Wells north and south of Johnson Lake and south of Turtle Lake appear to have the greatest potential for contributing significant quantities of nitrogen to the lake system.

Mean total nitrogen as N concentrations found in land wells ranged from .080 ppm at GW-11D to 17.2 ppm at GW-11S. Concentrations at in-lake wells were generally higher and ranged from .388 ppm at W5-1D to 18.59.3 ppm at W3-2S (Table 5-6). Note that two land wells with high nitrogen, GW-11S (17.2 ppm) and GW-11M (4.84 ppm), are only 75 feet from in-lake wells W5-1S and W5-1D (Figure 5-2) with .388 ppm and .747 ppm, respectively. They are all four screened in the same layer of sand and gravel, and GW-11M and W5-1D are screened at the same elevation. W5-1S is screened at an elevation between GW-11S and GW-11M. Lower nitrogen observed at W5-1D may indicate that the origin of water at W5-1D is deeper in the profile than where GW-11M is screened or that nitrogen is being lost as it moves toward the lake.

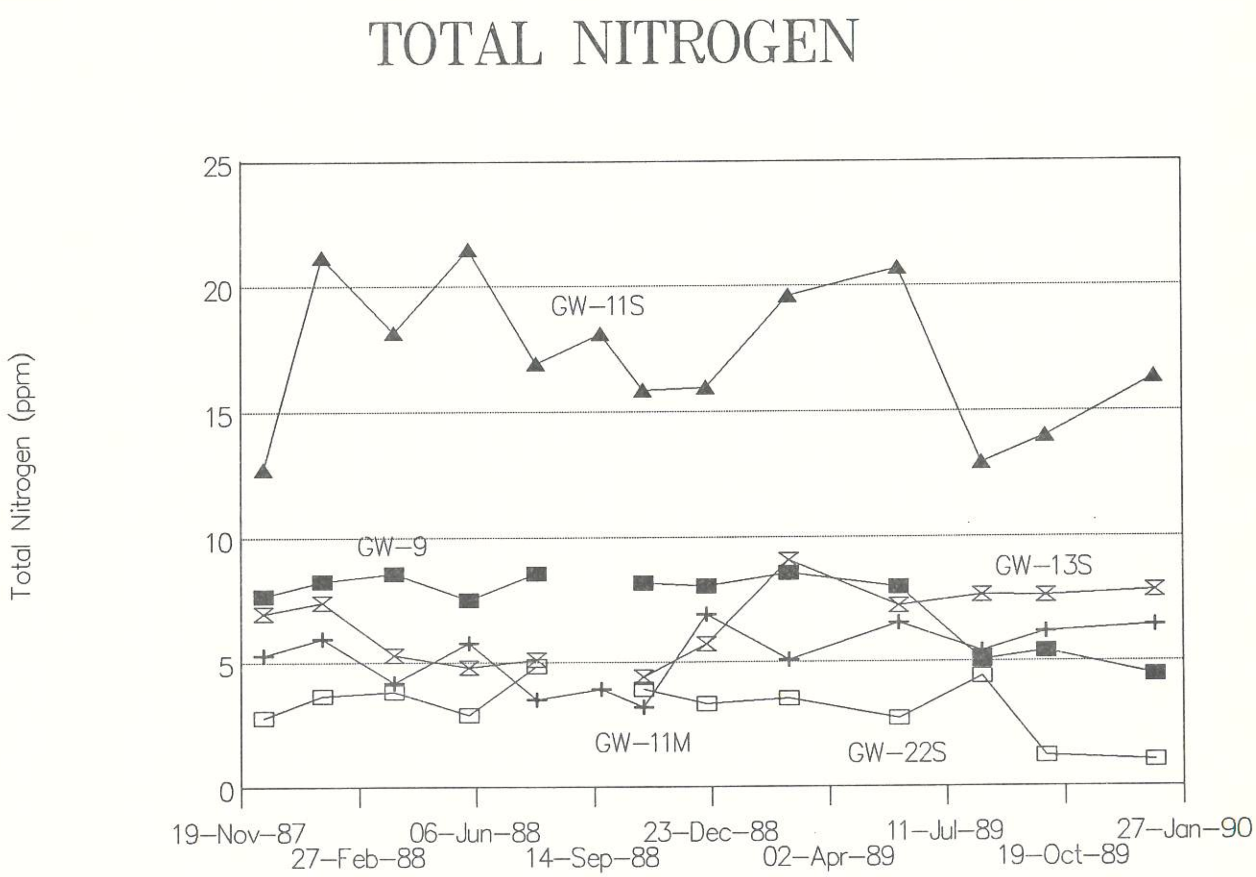


Figure 5-13A. Total nitrogen as N concentrations in Oakwood Lake land wells.

TOTAL NITROGEN in OLSS WELLS

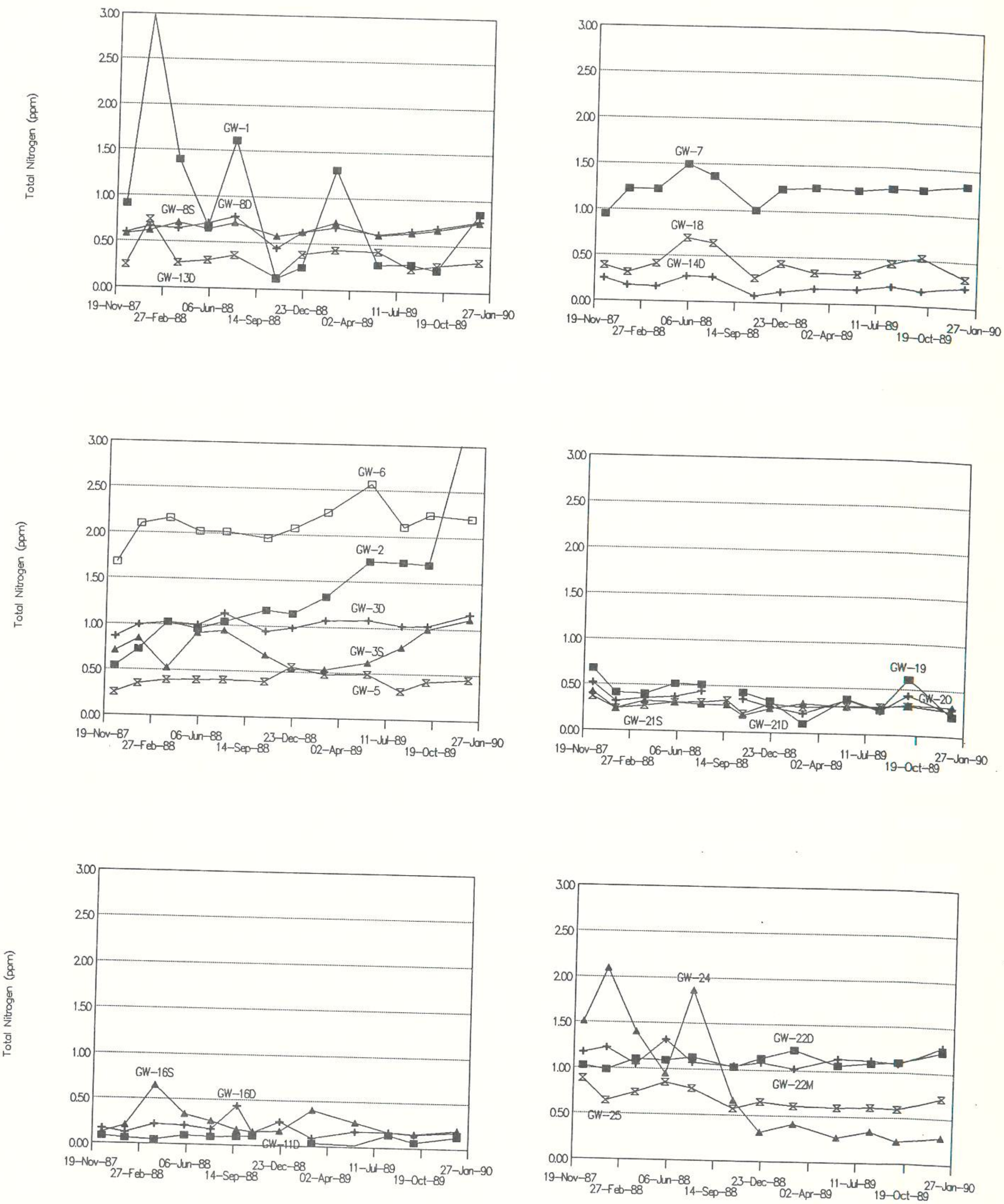


Figure 5-13B. Total nitrogen as N concentrations in Oakwood Lake land wells.

Concentrations of total nitrogen as N was relatively stable over the two years they were sampled (Figure 13A and 13B). Exceptions to this included GW-1 and GW-24 which exhibited a significant decline in total nitrogen as N. GW-2 exhibited a significant increase over the same period.

Ammonia

Ammonia was the dominant form of nitrogen in some OLSS land wells. Wells GW-6 and GW-7 with 1.96 ppm and 1.14 ppm, respectively, had the highest mean concentrations of ammonia among land wells. Several in-lake wells, (W3-2S, W7-3S and W7-5S) had very high ammonia concentrations (Table 5-6). These wells are all shallow (3 feet) and screened in mud, although other shallow wells screened in the similar materials do not exhibit high ammonia levels. In-lake wells W2-1S and W4-1S had the lowest concentrations of ammonia of in-lake wells with .11 ppm and .15 ppm, respectively. Both are screened in gravel and are adjacent to land wells that are also low in ammonia.

Nitrate

Nitrate was the other dominant form of nitrogen in OLSS land wells. Mean nitrate concentrations were above 1 ppm in only five wells. GW-2, GW-9, GW-11S, GW-11M and GW-13S had mean nitrate concentrations of 1.23 ppm, 7.04 ppm, 16.70 ppm, 5.02 ppm and 6.17 ppm, respectively. GW-2 is located on the edge of an alfalfa field and the others are located in areas with pasture or trees as the land use directly above them. All are all screened in sand and gravel. GW-9 is located approximately 75 feet from a pit privy near the swimming beach at the Oakwood State Park. This may account for some of the nitrates even though the privy is between the lake and the well and is probably down gradient. Well GW-13 and the GW-11 wells both have cropland approximately 100 feet up gradient and across the road from their location. They may be influenced by this adjacent land use. Several other wells that are located in pasture or trees that are located further from cropland have much lower nitrates. The in-lake wells were all quite low in nitrate. W3-1S had the highest mean nitrate concentration among in-lake wells with 0.24 ppm. This is probably due to the reducing environment common to lake sediments.

Total Dissolved Phosphorus

Total dissolved PO_4-P concentrations observed in land wells surrounding the Oakwood Lakes will be used to calculate the ground water contributions to the phosphorus budget for the Oakwood Lakes. Concentrations of total PO_4-P for all OLSS land wells are presented in Figure 5-14. Most wells fall in the .02 to .06 ppm range which is significant for ground water. Other wells are much higher. Ground water is probably a significant contributor of phosphorus to the system, although it has not yet been quantified.

Mean total PO_4-P concentrations found in land wells ranged from .12 ppm at GW-9 to .01 ppm at several wells (Table 5-6). GW-9 is the well located in the Oakwood State Park near the pit privy and may be affected by leachate. GW-22M is another well with a high mean (0.08 ppm). It is located at the southeast corner of East Oakwood Lake.

TOTAL DISSOLVED PHOSPHORUS

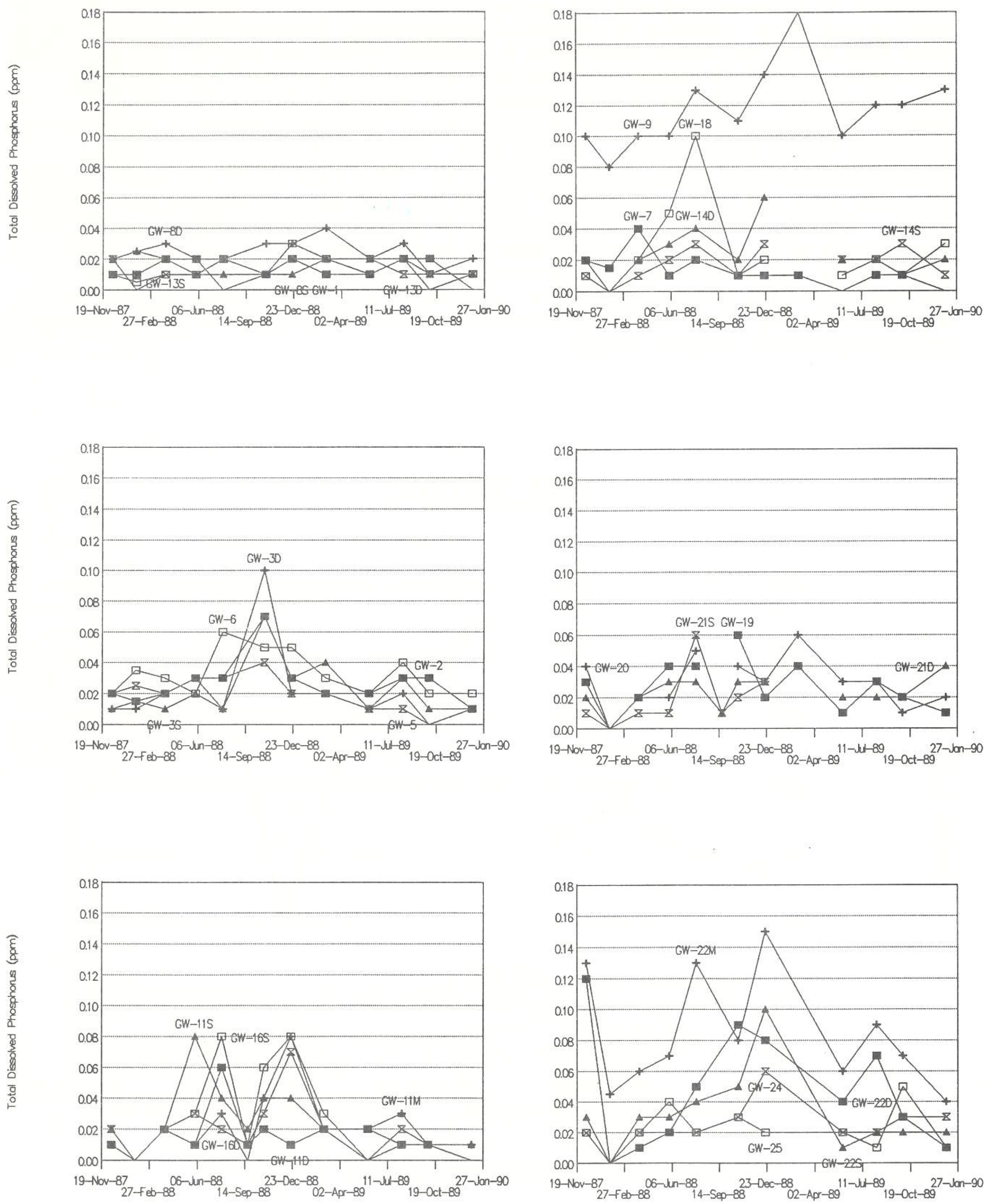


Figure 5-14. Total dissolved PO_4-P in Oakwood Lake land wells.

Chloride

Chloride is conservative and was included in the set of parameters analyzed in OLSS ground water samples to determine similarities between ground water taken from land wells and water collected in in-lake wells and seepage meters. This information may help confirm connections that are suspected from drilling information and analysis of flow gradients. For this information to be useful, differences in chloride concentrations between individual wells in the same area would be desirable. Wells GW-1, GW-16D and GW-24 had higher concentrations than other wells (Table 5-6). GW-1 and GW-24 are some distance from the lake and will not be very useful for drawing connections with in-lake wells. GW-16D with a mean concentration of 17 ppm is significantly different than GW-16S with 4 ppm and may be useful in drawing comparisons. In-lake wells W4-1S and W4-1D that are located near the GW-16 land wells are intermediate with 11 ppm and 15 ppm mean chloride respectively (Table 5-6). Wells GW-16D and GW-24 exhibited a decline in chloride while other wells were relatively stable (Figure 5-15B).

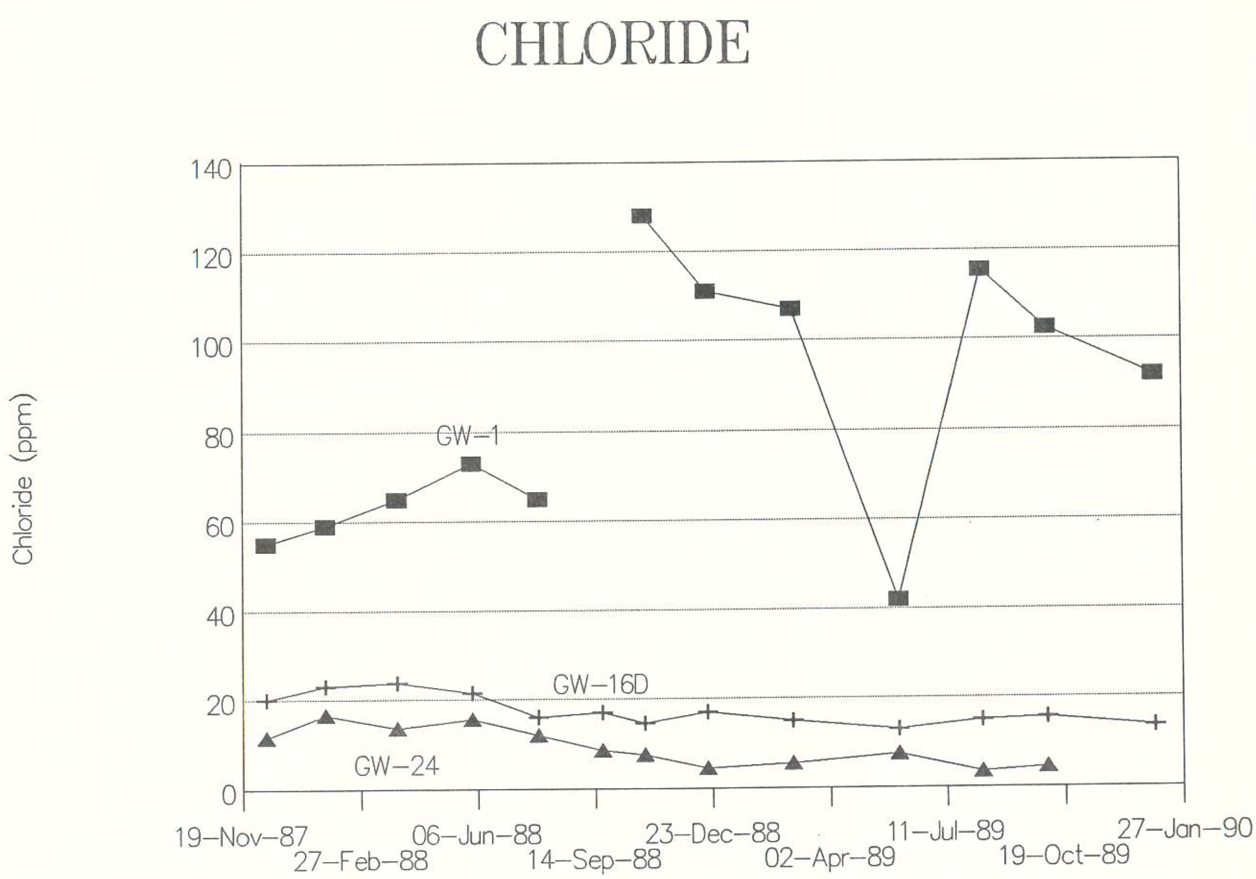


Figure 5-15A. Chloride Concentrations in OLSS land wells.

CHLORIDE in OLSS WELLS

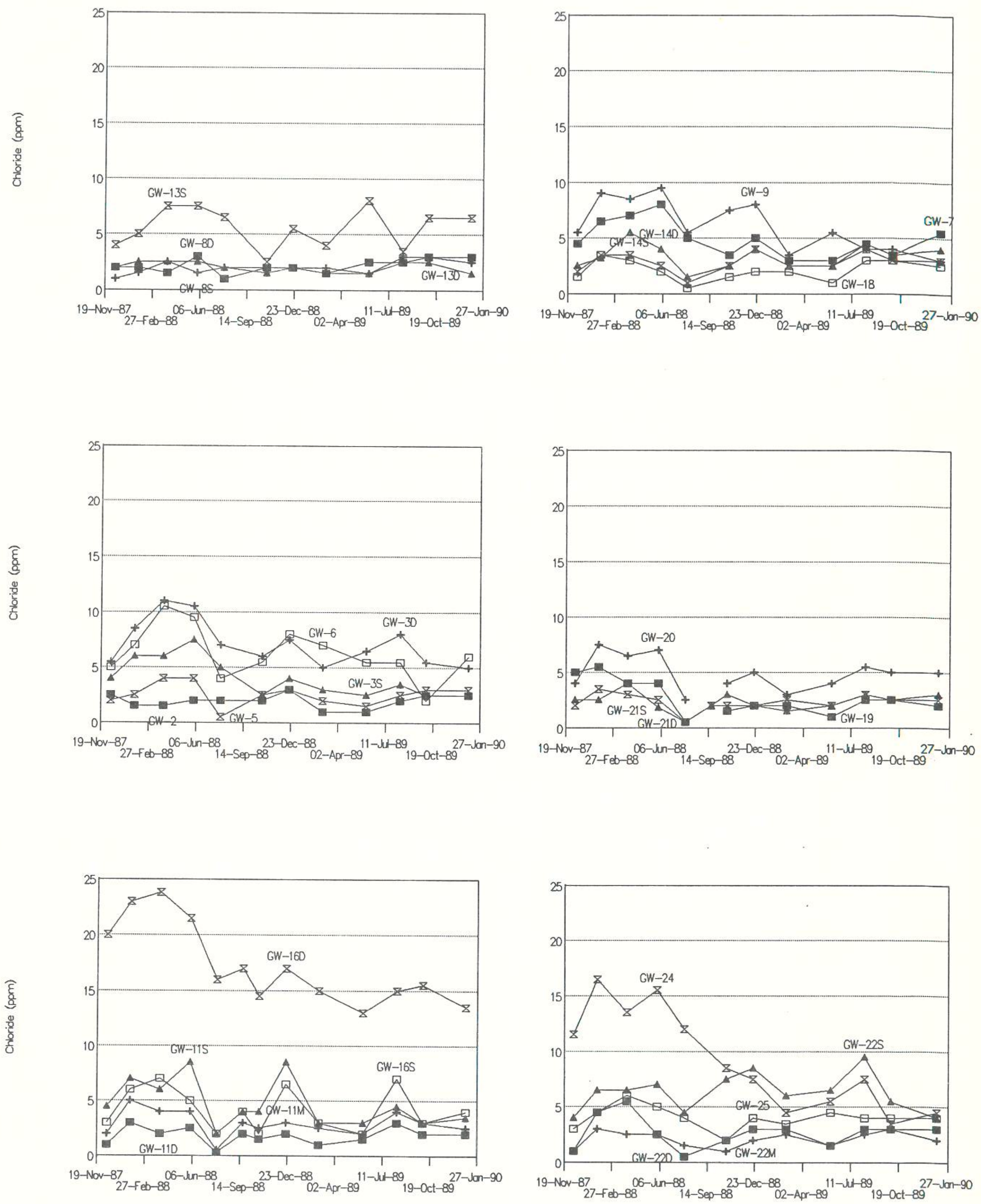


Figure 5-15B. Chloride Concentrations in OLSS land wells.

Sulfate

Sulfate is another fairly conservative parameter that was included in the set of parameters analyzed in OLSS ground water samples to determine similarities between ground water taken from land wells and water collected in in-lake wells and seepage meters. For this information to be useful, differences in sulfate concentrations between individual wells in the same area would be desirable. Mean concentrations of sulfate for land wells ranged from 56 ppm at GW-11M to 1072 ppm at GW-5 (Table 5-6). Concentrations of sulfate were relatively stable during the sampling period (Figures 5-16A and B). Wells GW-1, GW-3D, GW-5, GW-6 and GW-7 had higher concentrations than other wells (Figure 5-16A). These wells are located south of Johnson Lake except for GW-1 (Figure 5-2). Comparisons with in-lake wells may not be valid due to high concentrations of sulfate in some in-lake wells that may be influenced by sediment chemistry (Table 5-6). For example, the GW-11 well nest had mean sulfate from 72 to 167 ppm but adjacent W5-1 in-lake wells had 437 to 485 ppm sulfate (Table 5-6). Mean concentrations of sulfate for in-lake wells ranged from 362 ppm at W2-1S to 2612 ppm at W1-1S.

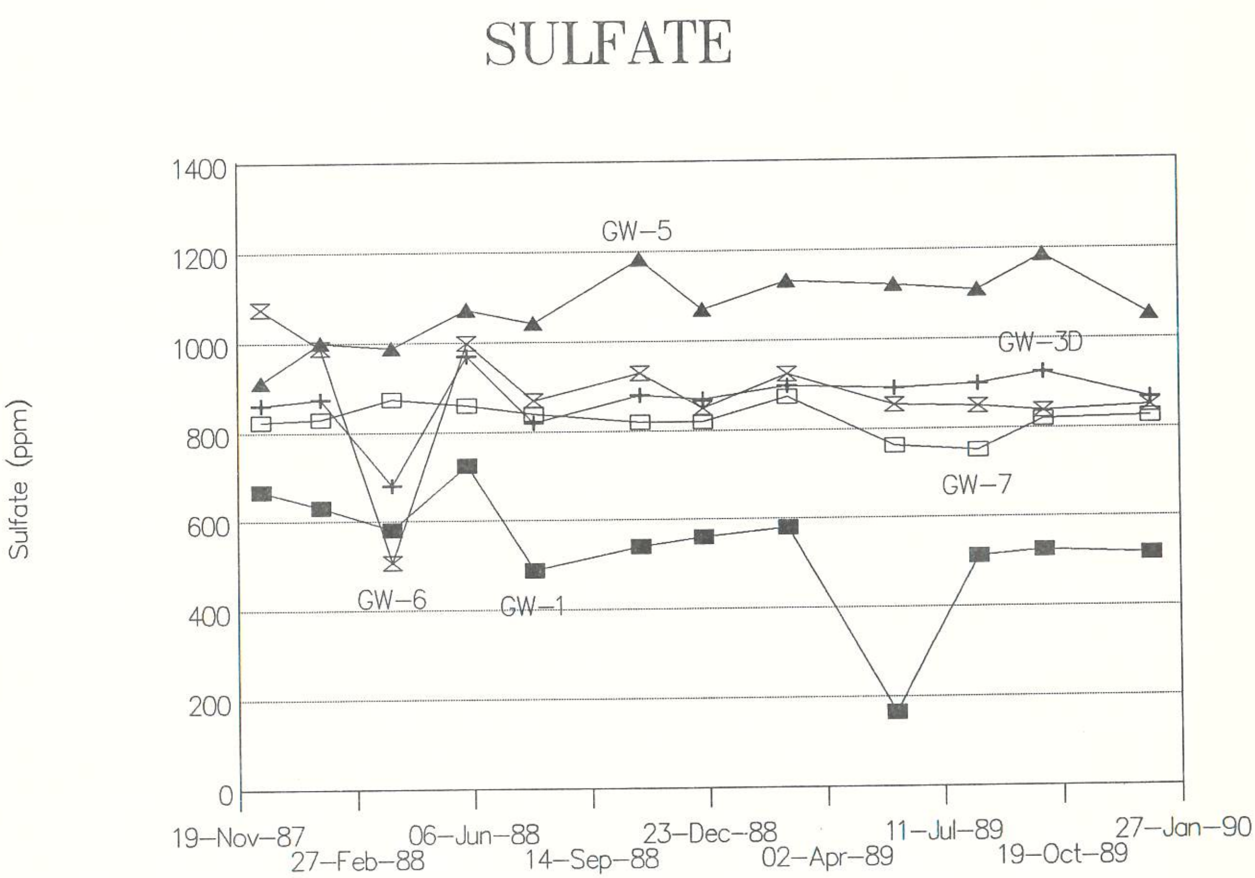


Figure 5-16A. Sulfate Concentrations in OLSS land wells.

SULFATE in OLSS WELLS

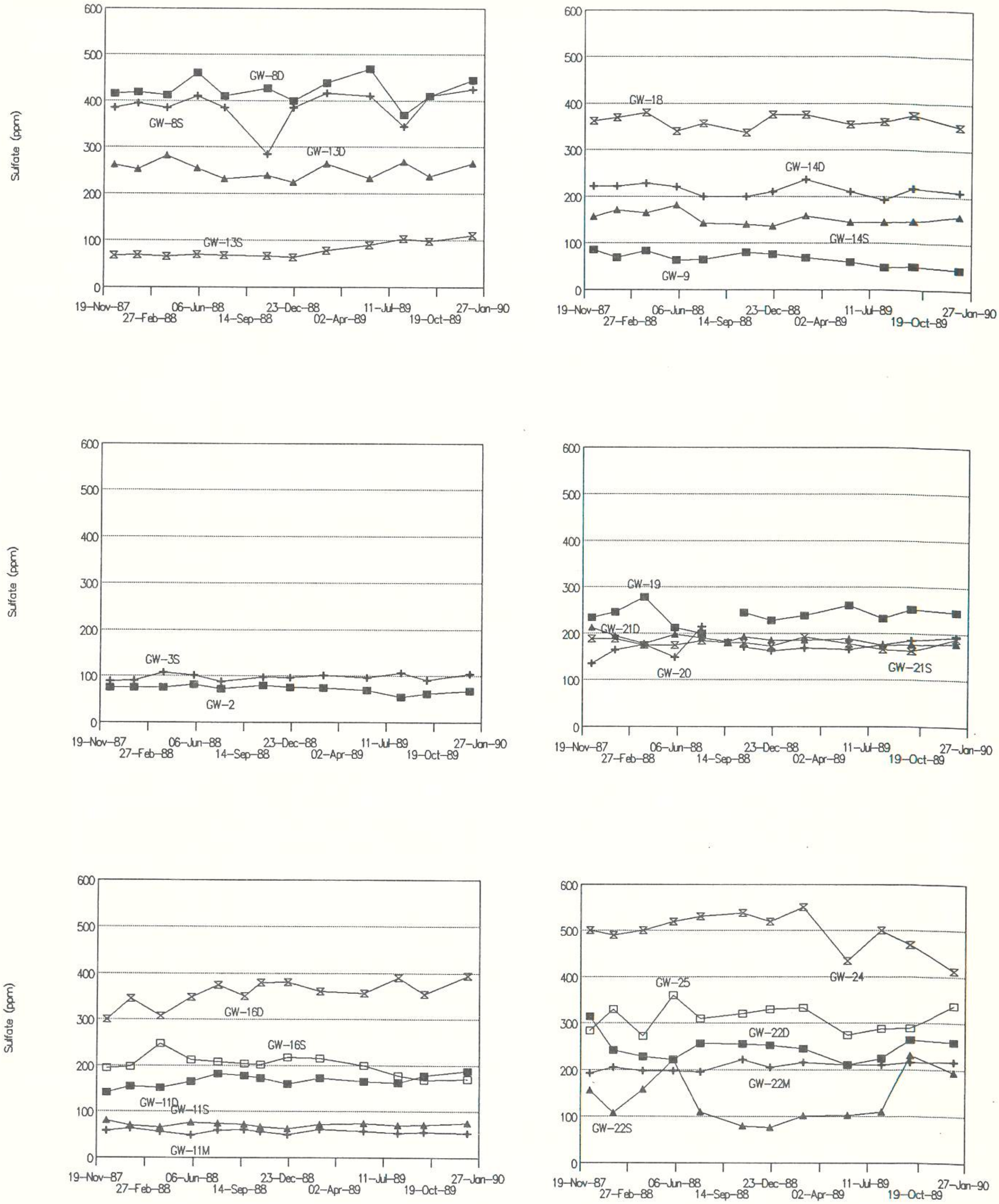


Figure 5-16B. Sulfate Concentrations in OLSS land wells.

Total Dissolved Solids

Total dissolved solids (TDS) is another parameter that was included in the set of parameters analyzed in OLSS ground water samples to determine similarities between ground water taken from land wells and water collected at in-lake wells and seepage meters. Wells GW-1, GW-3D, GW-5, GW-6 and GW-7 had higher concentrations of TDS than other wells and are plotted at a larger scale in Figure 5-17A. These wells are located south of Johnson Lake except for GW-1 (Figure 5-2). For this information to be useful, differences in TDS concentrations between individual wells in the same area would be desirable. For some nests of wells, this is true; for example, GW-3S, which has a mean of 464 ppm TDS, and GW-3D has a mean of 1677 ppm (Table 5-6). In other well nests, however, concentrations of TDS are similar when comparisons between wells are made. Mean concentrations of TDS for land wells ranged from 418 ppm at GW-3S to 2081 ppm at GW-5 (Table 5-6). Concentrations were relatively stable over the two year sampling period (Figure 5-17B).

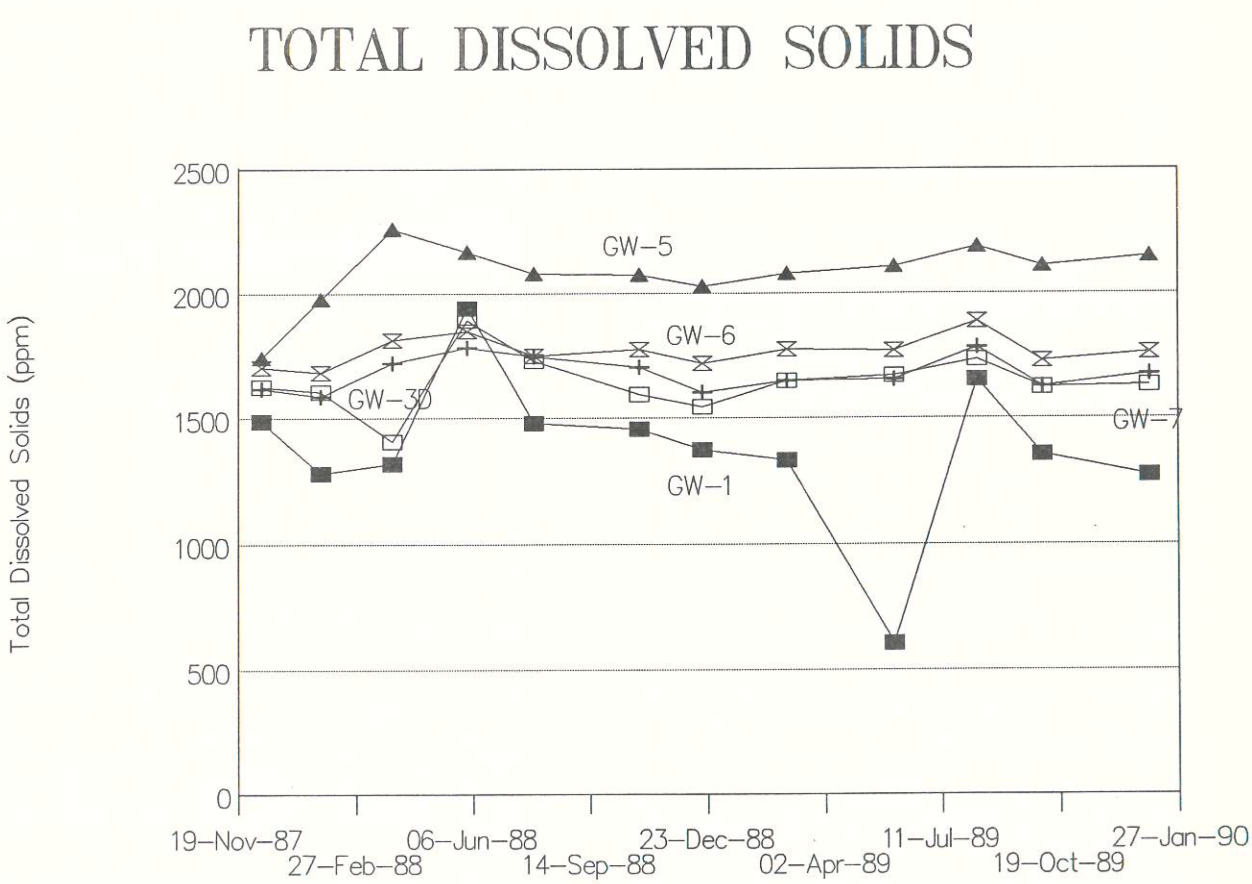


Figure 5-17A. Total Dissolved Solids Concentrations in OLSS land wells.

TOTAL DISSOLVED SOLIDS in OLSS WELLS

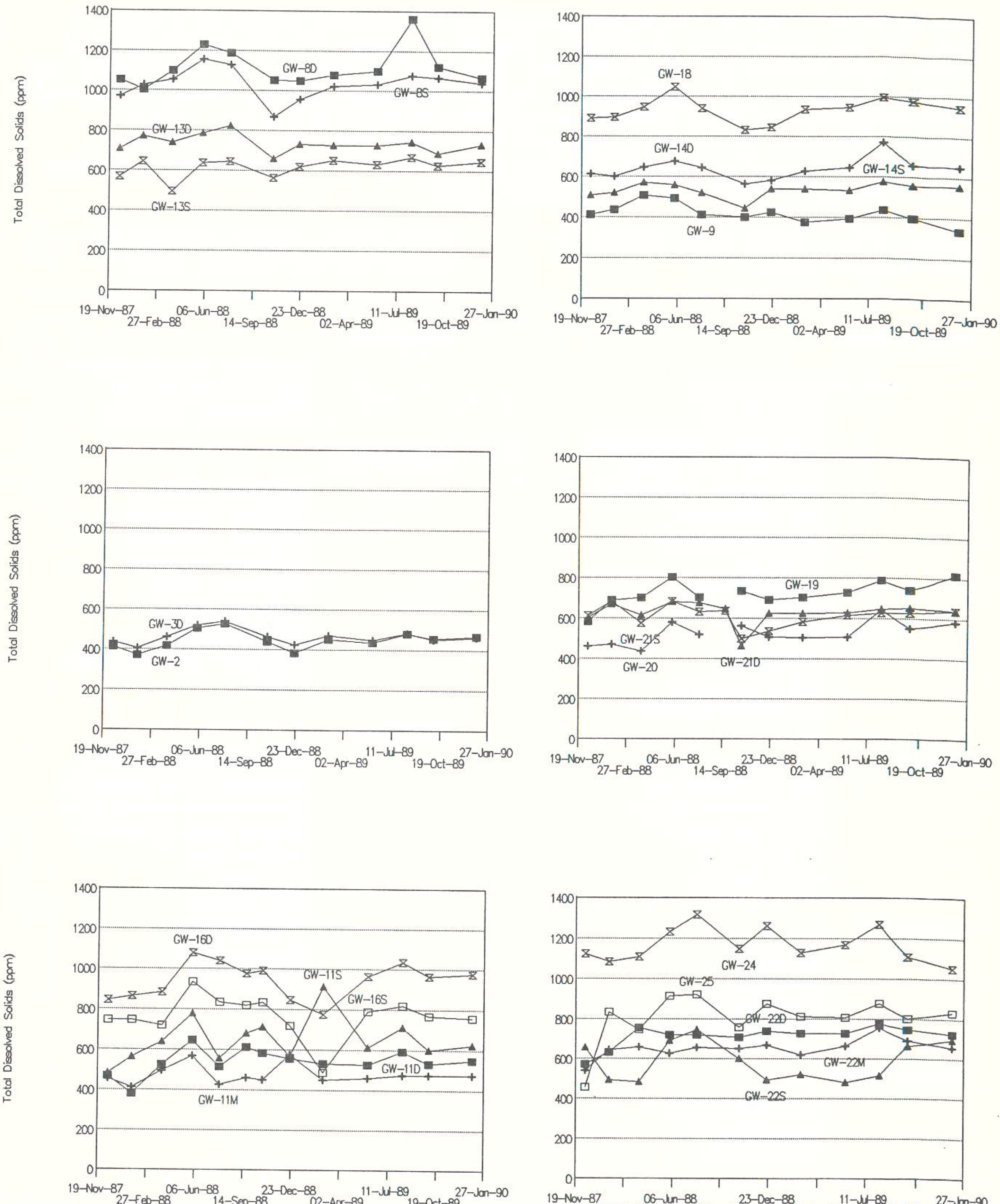


Figure 5-17B. Total Dissolved Solids Concentrations in OLSS land wells.

5.3.6 Phytoplankton and Zooplankton Dynamics

The plankton work for the OLSS was done by a different graduate student in each of the three years of study. Species of algae present in the Oakwood lakes were similar in 1987, 1988 and 1989 seasons (Table 5-7) but numbers and total biomass varied. Differences in the conversion factors used to calculate biomass prevent a valid comparison between years at this time. These conversions will be standardized and the biomass data will be used as dependant variable in multiple regression analysis to identify major factors affecting water quality in the Oakwood lakes.

Table 5-7A. Major algal genera in the Oakwood Lakes.

Bluegreens:

Non heterocystous:
- Agmenellum
- Anacystis aeruginosa
- Anacystis cyanea
- Anacystis. incerta
- Coccochloris
- Gomphosphaeria
- Lyngbya
- Merismopedia
- Oscillatoria

Heterocystous:
- Anabaena
- Aphanizomenon
- Cylindrospernum

Greens:
- Ankistrodesmus
- Dictyosphaerium
- Closteriopsis
- Pediastrum
- Schroederia

Diatoms:
- Melosira
- Synedra
- Pinnularia
- Stephanodiscus

Flagellates:
- Chlamydomonas
- Cryptomonas
- Chroomonas

Table 5-7B. Major zooplankton genera in the Oakwood Lakes.

Copepoda:
- Cyclops bicuspidatus
- Cyclops vernalis
- Diaptomus sicilis

Cladocera:
- Daphnia pulex
- Daphnia galeata
- Daphnia parvula
- Daphnia catawba
- Daphnia rosea
- Ceriodaphnia
- Diaphanosoma
- Bosmina longirostris
- Alona

Rotifera:
- Keratella quadrata
- Keratella cochlearis
- Brachionus angularis
- Brachionus cuadatus
- Polyarthra
- Filinia
- Asplanchna
- Trichocerca
- Kellicottia
- Syncheata

5.3.7 Fisheries Study

Fish Populations

We collected 38,180 fish representing 10 species (Table 5-8). Bullheads (mostly Ictalurus melas and to a lesser degree I. natalis) dominated the catch in total number (86%). Northern pike (Esox lucius), yellow perch (Perca flavescens), and walleye (Stizostedion vitreum) accounted for 2.5% of the total number. Carp (Cyprinus carpio), bigmouth buffalo (Ictiobus cyprinellus), and white sucker (Catostomus commersoni) accounted for 2.9% of the total number. Fathead minnows (Pimephales promelas) accounted for 3.5% of the total number of fish collected. Darters (Etheostoma sp.), green sunfish (Lepomis cyanellus), and shiners (Notropis sp.) were collected in previous years but only one green sunfish was collected in 1989. Catches were similar among stations (Table 5-8).

Table 5-8. Number of fish collected using four gear types at five sampling sites in East and West Oakwood Lakes, 1989.

Species	1		2		3		4		5	
	East	West	East	West	East	West	East	West	East	West
BBH	3294	1920	6231	2181	3197	1658	7828	732	1031	4680
YBH	115	65	219	1233	115	31	388	42	189	21
YP	60	21	46	64	28	60	40	8	126	20
WAE	4	20	1	62	0	18	1	48	0	171
CARP	32	562	6	40	19	86	19	9	9	6
WHS	8	21	33	3	2	5	19	75	6	7
BMB	5	17	1	2	2	42	30	7	2	14
FHM	43	924	17	97	50	112	12	0	0	73
NOP	22	3	34	5	25	2	32	3	20	4
GSF	0	0	0	1	0	0	0	0	0	0
OSS	0	0	0	0	0	0	0	0	0	0

The four gear types used to collect fish in East and West Oakwood Lakes were selective. The seine was the least productive but captured smaller fish which inhabited the littoral zone (Table 5-9). Black bullheads, carp, yellow perch, walleye, and white suckers collected with the seine were mostly young-of-the-year fish. Fyke nets were effective on all species except yellow perch, which were captured more frequently in the gill net. Overall, the variety of passive and active collection techniques in both shallow and deep water probably provided a good index of the species richness and relative abundance in the Oakwood fish community.

Table 5-9. Number of fish captured using four gear types in East and West Oakwood Lakes, 1989.

Species	3/4" TN		1/2" TN		Gill Net		Seine		Totals
	East	West	East	West	East	West	East	West	
BBH	9071	6710	12246	4249	250	179	4	33	32,742
YBH	434	235	342	1128	38	8	212	0	2,397
YP	31	57	49	27	130	85	90	4	473
WAE	2	81	1	4	3	227	0	0	318
CARP	26	591	4	77	10	3	45	32	788
WHS	38	22	25	87	5	2	0	0	179
BMB	30	26	10	50	0	6	0	0	122
FHM	0	0	2	0	0	0	120	1206	1,328
NOP	46	7	39	4	48	6	0	0	150
GSF	0	1	0	0	0	0	0	0	1
									38,180

We calculated the relative weight of three game fish as an index of the well being of these species (Anderson and Gutreuter 1983). Relative weight is given by the equation

$$W_r = \frac{W}{W_s} \times 100$$

where W is the weight of an individual and W_s is a length-specific standard weight. Mean W_r values of 100 ±5 are expected in healthy population where food and feeding relationships are adequate. Four W_r indices were below expectations in West Oakwood (Table 5-10). Perhaps the greater frequency of winterkill in East Oakwood keeps the fish population below carrying capacity, whereas, in West Oakwood, there is greater competition for prey.

Table 5-10. Relative weight values for three species of fish in East and West Oakwood Lakes, 1987-1989.

Species	Year	East Oakwood			West Oakwood		
		n	W_r	Std. err.	n	W_r	Std. err.
Northern Pike	1987	304	95.7	1.2	84	98.5	1.6
	1988	51	95.4	3.3	72	71.4	3.4
	1989	55	93.4	3.2	10	95.8	4.7
Yellow Perch	1987	152	102.3	1.7	31	88.9	4.3
	1988	84	100.6	1.7	6	100.2	8.7
	1989	105	104.5	2.1	21	92.1	9.9
Walleye	1987	27	96.5	3.9	38	96.5	2.7
	1988	14	101.4	3.6	78	90.5	1.7
	1989	3	87.8	47.2	6	100.7	13.0

The black bullhead has dominated the weight (percent biomass) of collected fish over all years of the study (Figures 5-18 and 5-19). In 1987, black and yellow bullheads made up from 27% (East) to 32% (West) of the weight of all captured fish. The percentage increased about 50% in 1988 and remained at about 1988 levels in 1989. Northern pike declined from 39% (mean of both lakes) in 1988 to about 8% of the biomass the following two years.

Percent Biomass for East Oakwood

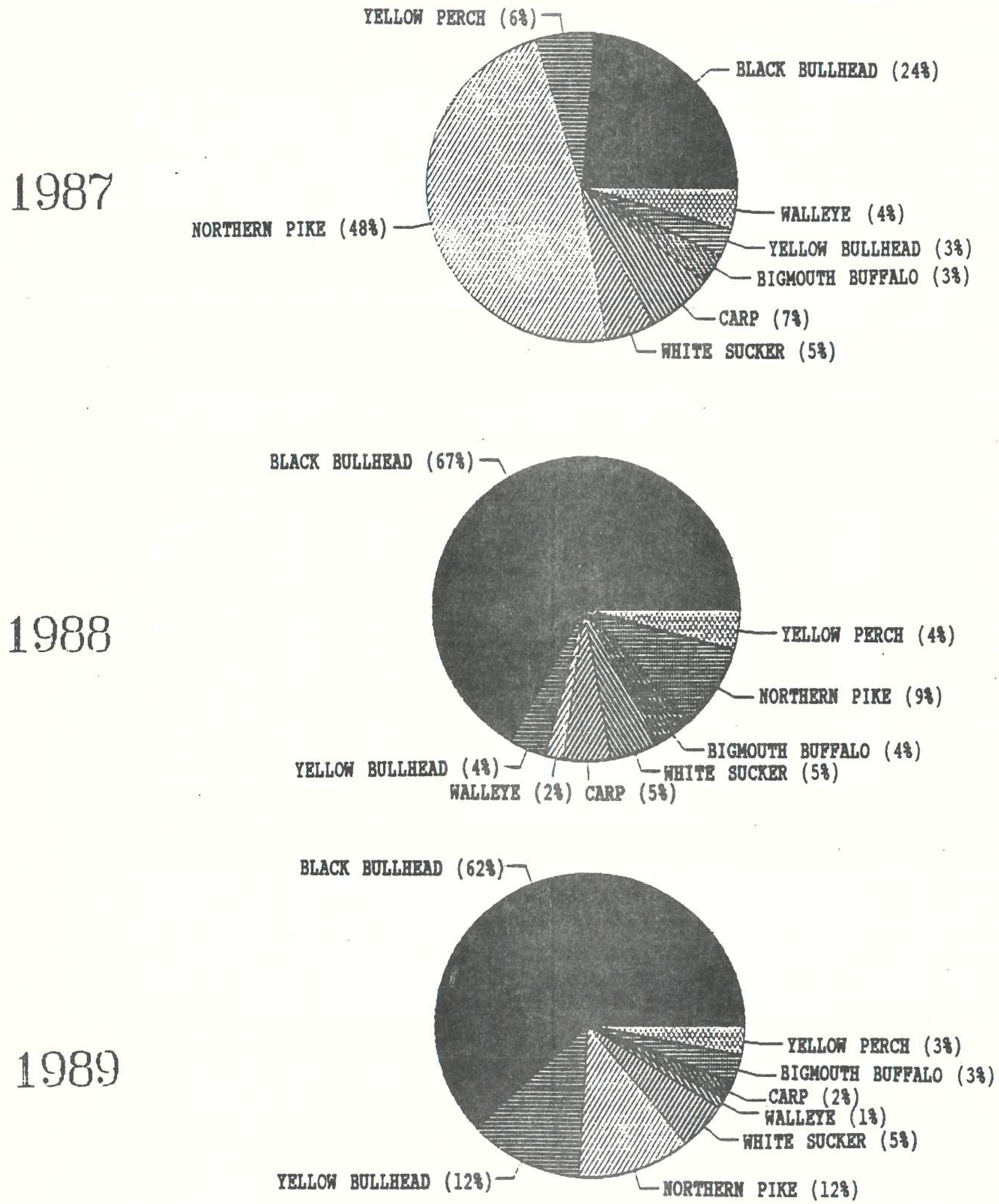


Figure 5-18. Composition of the fish population in East Oakwood Lake

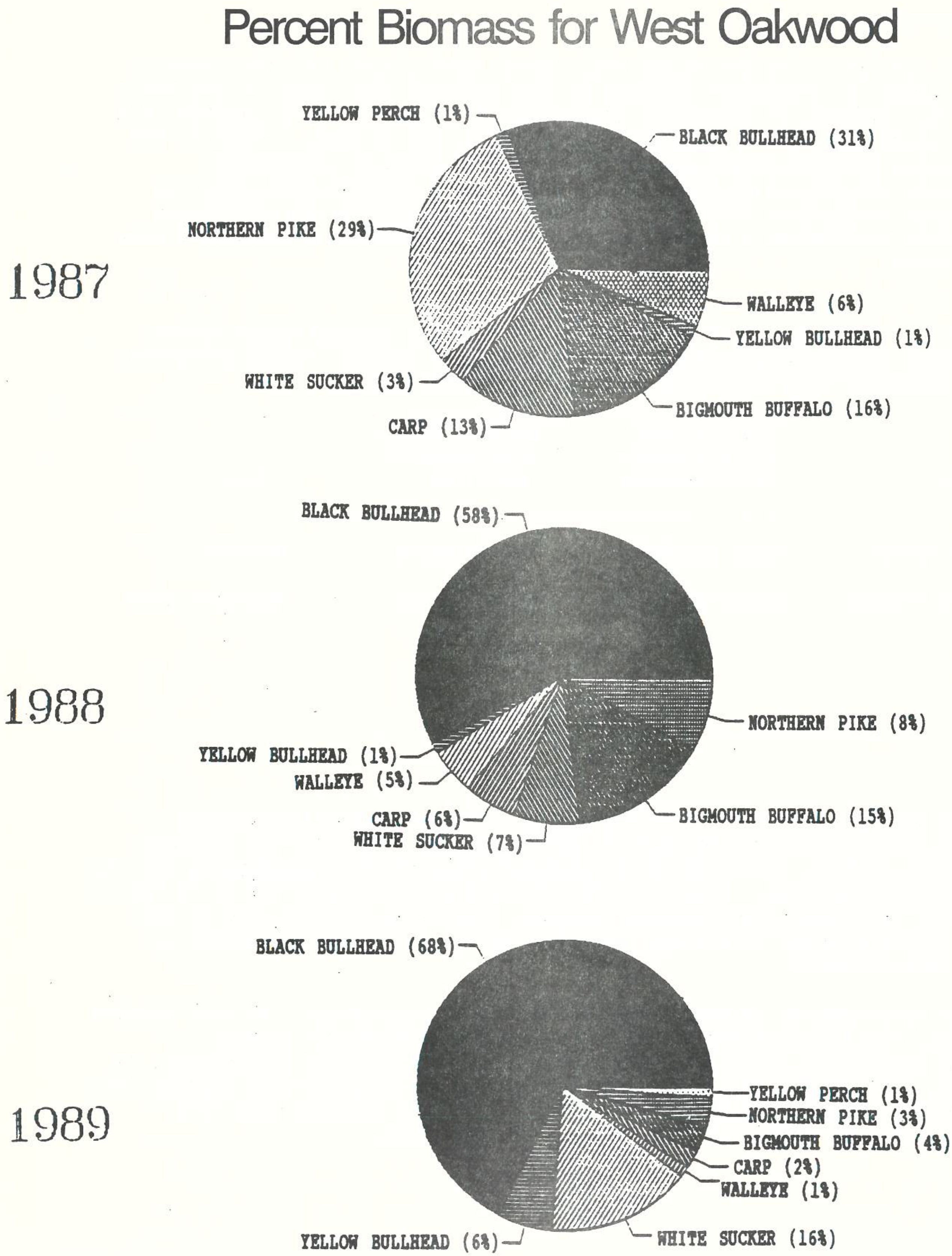


Figure 5-19. Composition of the fish population in West Oakwood Lake.

Fish Food Habits

The fish species collected during this survey were placed into four general groups, based on literature accounts of the dominant prey items in the adult diet. The dominant or codominant items include 1) plankton, 2) plankton and benthic invertebrates, 3) benthic invertebrates, and 4) fish. Based on the adult diets, fathead minnow and bigmouth buffalo would be considered specialists feeding on plankton, while walleye and northern pike would be specialists feeding on fish. The remainder of the species are considered generalists feeding on a variety of planktonic and benthic invertebrate prey (Table 5-11).

Table 5-11. Feeding guilds of adult fishes from the Oakwood Lakes system. Preliminary guild assignment based on a literature review. Guild assignment may be adjusted later depending on results of diet analysis.

Plankton	Plankton and Benthic Macro-invertebrates	Benthic Macro-invertebrates and Fish	Fish
Fathead minnow	Black bullheads	Yellow perch	Walleye
Bigmouth buffalo	Yellow bullheads	White sucker	Northern pike
Darter		Carp	
Shiner			

Analysis of fish food habits is continuing. We have made a preliminary analysis (Table 5-12) of the diet of the bullheads, which are the dominant fish in the system. Most bullheads consumed zooplankton; Diaphanosoma was the most preferred species. From 9 to 35% of the black bullheads and 36 to 50% of the yellow bullheads also consumed insects. Fish were a minor part of the diet. Our data agree with those of Repsys et al. (1976) for bullheads in nearby Lake Poinsett.

Our diet data must be interpreted with caution at this point. We have grouped all dates, all size classes of fish, and all zooplankton and insects. Seasonal succession among zooplankton species , particularly cladocerans, is expected. Therefore, if a zooplanktivorous fish was sampled when Daphnia were at a maximum, the diet might be expected to have more Daphnia than other zooplankters. We expect diet to change with fish size.

Table 5-12. Percent frequency of occurrence of zooplankton, insects, and fish in the diet of black and yellow bullheads collected from Oakwood Lakes in 1988.

Diet item and number of samples	Black bullhead East	Black bullhead West	Yellow bullhead East	Yellow bullhead West
No. stomachs examined	63	38	24	32
No. stomachs without food	16	7	0	7
Zooplankton	91	94	75	60
Insects*	9	35	50	36
Fish	2	29	17	12
Other	2	3	42	8

* Also includes worms, mollusks, and leeches.

Table 5-13. Size range (mm) of zooplankton eaten by seven species of fish in Oakwood lakes. Data are from July and August, 1988, and June, August, and October, 1989.

Zooplankton genera		Fish Species						
		BBH	WS	BMB	FHM	YBH	YP	CARP
Daphnia		0.7	---	0.4	0.9	---	---	1.2
	n	5	---	3	1	---	---	1
Diaphanosoma		0.7-1.1	0.7-1.0	0.7-0.9	---	1.0	0.7-1.1	0.6-1.1
	n	162	17	16	---	8	15	10
Bosmina		0.3	0.4	0.3-0.4	0.3-0.4	---	---	0.3-0.5
	n	101	11	106	43	---	---	59
Cyclops		0.6-1.3	0.7-0.9	0.6-0.7	1.0	0.8	0.5-0.8	0.5-1.0
	n	170	37	142	1	5	14	40
Diaptomus		0.8-0.9	0.5-1.0	0.8-0.9	0.2-0.8	---	0.8	1.0-1.1
	n	27	8	18	3	---	11	18

The size of zooplankters in the diet of seven species of fish was measured when an intact zooplankter was found. Results are presented in Table 5-13. Fish size had little effect on size of zooplankton that was eaten. For example, all black bullheads collected in October 1989, (100 to 300 mm in total length) ate Diaphanosoma that were 0.9 mm long, and Cyclops that were 0.7 mm long. These data will be most useful when compared with the size and kinds of zooplankters available in the water column. If the size that was eaten is larger than the size available, then the fish are selecting the larger individuals, an observation that has been commonly made in other studies (Wetzel 1983, p. 463).

About 90% of the fish in the Oakwood lakes can be classed as zooplanktivorous. The cascading trophic interaction theory suggests that because there is a negative relation between the densities of zooplanktivorous fish and zooplankton, then the zooplankton community might be too small to control the phytoplankton community. While the relation between fish and zooplankton has been shown in a number of studies, the relation at the next lower trophic level is usually much weaker (Post and McQueen 1987). Reasons for this include nutrient input from sediments (Ahlgren 1977), type of phytoplankton bloom (Wetzel 1983, p. 425), nutrient regeneration by zooplankton (Bergquist and Carpenter 1986), and strong "bottom up" effects at the TP--chlorophyll a link.

Success of biomanipulation of fish as a technique to reduce phytoplankton abundance may be limited. Drastic alterations in the fish community would be needed to produce only marginal effects on phytoplankton abundance and water clarity (Post and McQueen 1987). For example, conventional stocking rates of 120 walleye and 12 pike fingerlings per hectare in Lake Mendota did not depress standing stocks of planktivorous fish. Higher piscivore stocking rates would probably have to be maintained for 3-7 years to reduce planktivore biomass in that lake to levels at which planktivory would be reduced 50%. In Oakwood, stocking piscivores would probably not depress the bullhead population to levels that would effect water clarity.

Bullheads may not have as large an effect on zooplankton as our data suggest. For example, in Lake Christina (Minnesota) researchers learned that bigmouth buffalo, not bullheads, were the most important zooplanktivorous fish. Buffalo, but not bullheads, were removed and water clarity improved (M. Butler, NDSU, pers. comm.). Butler's results and those of Kolterman (1990) also indicate that bullheads may not be as important in resuspension of bottom sediments as has been suggested (Hope and Peterson 1962). However, they may assist in phosphorous recycling from bottom sediments (Keen and Gagliardi 1981).

5.3.8 Lake Sediment Study

Sediment physical and chemical characteristics

At the time of sediment sampling the DO concentration in all lakes decreased with water depth to near zero at the sediment surface. The low DO concentrations near the lake bottom may have created a flux of phosphorus from the sediment to the water column prior to collection of the sediment cores. If so, the phosphorus content of the sediment, and subsequently the quantity available for release to the overlying water may be greater than this study determined.

The sediment is a silty clay, highly organic muck, black to black-grey in color. The muck is a mixture of allocthonous inorganic silt and clay and autochthonous organic matter primarily detritus of the lake biota. Most cores contained shells and shell fragments often in high concentrations and sometimes in distinct layers. Partially decomposed coarse plant material was found in East Oakwood Lake within the 45-60 cm section. The hydrogen sulfide (H_2S) odor exuded during coring indicates the sediment exists in an extremely reduced state.

Deposition of the muck has occurred wherever water depth is sufficient to prevent scouring of the lake bottom by wave action. Sediment thickness is at least 2 m (maximum length of the coring device) and is probably much thicker. The muck type sediment is estimated to cover 85% of the lake bottom. For the remaining discussion the use of the term "sediment" will mean the soft muck described above and not the sandy deposits that also occur in the lake system.

The top 15 cm of all cores was extremely flocculent and is the site of greatest sediment-water interaction. Sediment density increased in the deeper sections as quantified by the increase in dry bulk weight with sediment depth.

The phosphorus concentration profile shows total phosphorus and organic phosphorus concentrations were greatest at 0-7 cm then decreased with sediment depth (Figure 5-20). Conversely, the concentration of inorganic phosphorus increased with sediment depth and reached a maximum concentration at 45-60 cm. The ratio of organic phosphorus to total phosphorus was 42% within the 0-7 cm layer and decreased to 23% within the 45-60 cm layer. The ratio of inorganic phosphorus to total phosphorus was 58% at the 0-7 cm layer and increased to 77% within the 45-60 cm layer.

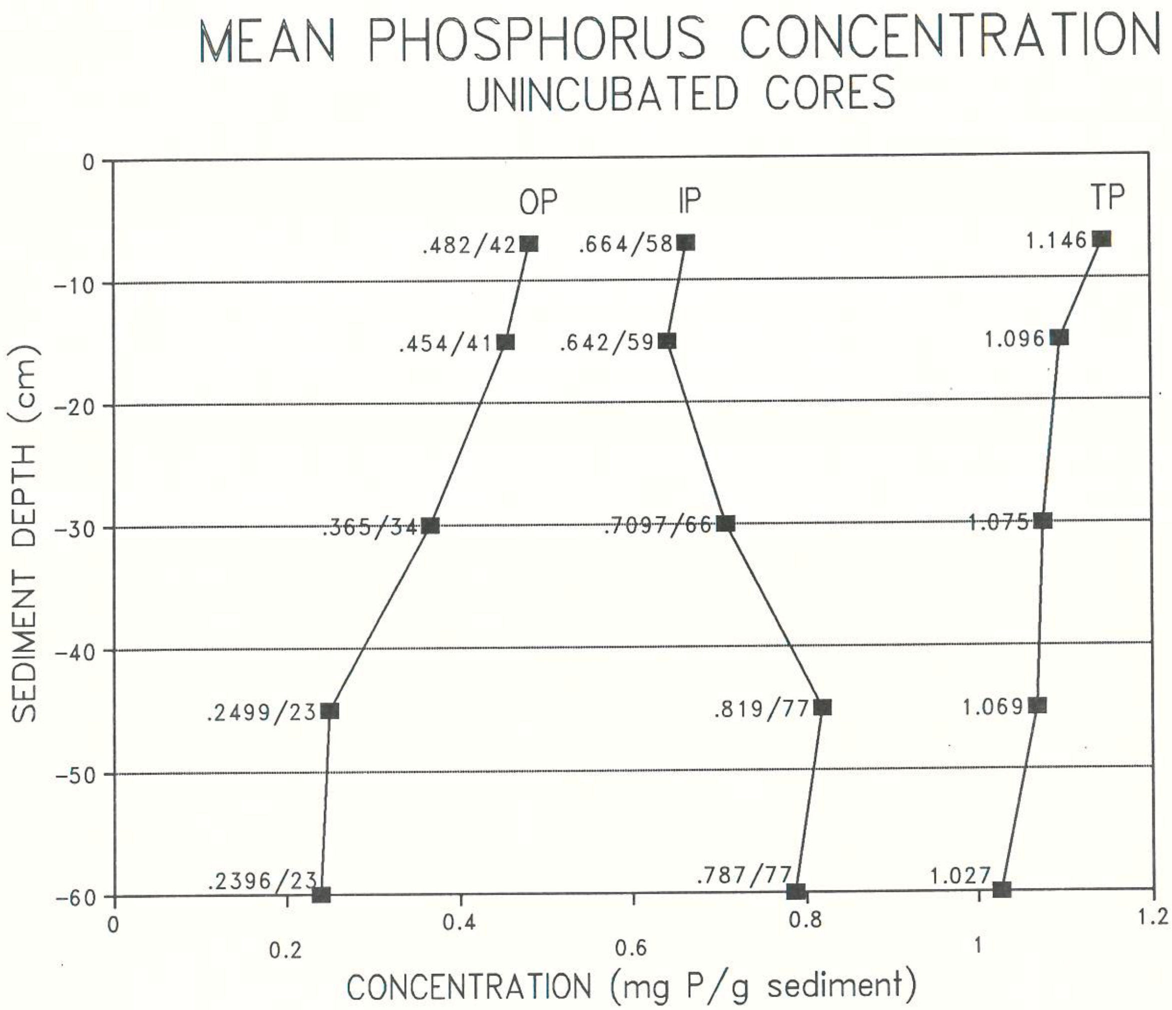


Figure 5-20. Mean phosphorus concentrations of unincubated sediment cores.

The highest sediment concentrations of total, inorganic and organic phosphorus were observed in West Oakwood Lake followed by Johnson Lake and East Oakwood Lake (Figure 5-21). The ratio of inorganic phosphorus to total phosphorus was between 65-70% and the ratio of organic phosphorus to total phosphorus ranged between 30-35%. These results agree with other studies reporting total inorganic phosphorus averaging 69% and total organic phosphorus averaging 33% (Theis and McCabe 1978; Hosomi et al., 1982; Kamp-Neilsen 1974).

ANOVA indicated the sediment concentration of total, inorganic and organic phosphorus varied significantly with depth at the .01 significance level but depth by lake interaction was not detected. Between lakes differences exist at the .05 level of significance for total, inorganic and organic phosphorus but sites within lakes were not significantly different. The analysis indicates the level of sediment phosphorus is lake specific, however, the pattern of distribution, both vertically and horizontally, is similar in all the study lakes.

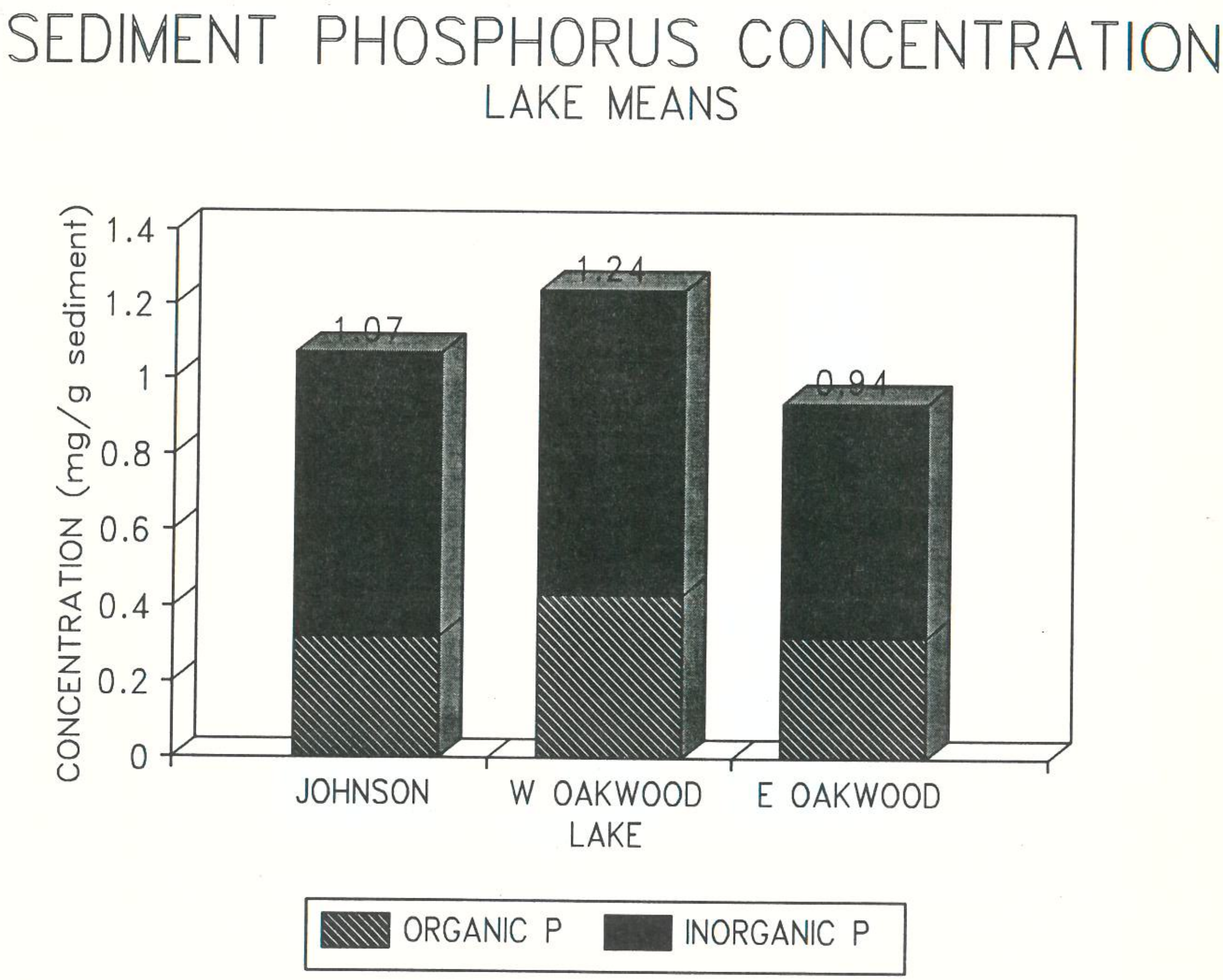


Figure 5-21. Mean concentration of inorganic and organic phosphorus by lake.

Incubation experiment

The overlying water of all anaerobically incubated cores experienced a net gain of Total PO_4-P indicating phosphorus was released from the sediment (Figure 5-22). A smaller net gain of Total PO_4-P occurred in the water of the aerobically incubated cores. Two cores from West Oakwood Lake experienced a net loss of Total PO_4-P in the overlying water, indicating the sediment had removed phosphorus from the water column of these cores, resulting in the negative aerobic treatment mean.

The LSD analysis of the treatment means shows the anaerobically incubated cores released significantly more Total PO_4-P (0.305 g/m^2) than the aerobically incubated cores (0.080 g/m^2). Based on a sediment surface area of 7.12 x 10^6 m^2, the potential sediment phosphorus load to the Oakwood Lakes system is 2,041.0 kg Total PO_4-P during anaerobic conditions and 467.76 kg Total PO_4-P during aerobic conditions. ANOVA indicated the release of Total PO_4-P varied significantly between lakes for both aerobic and anaerobic incubation at the .03 significance level. ANOVA of the sediment phosphorus concentration of the unincubated cores also detected significant between lakes variation.

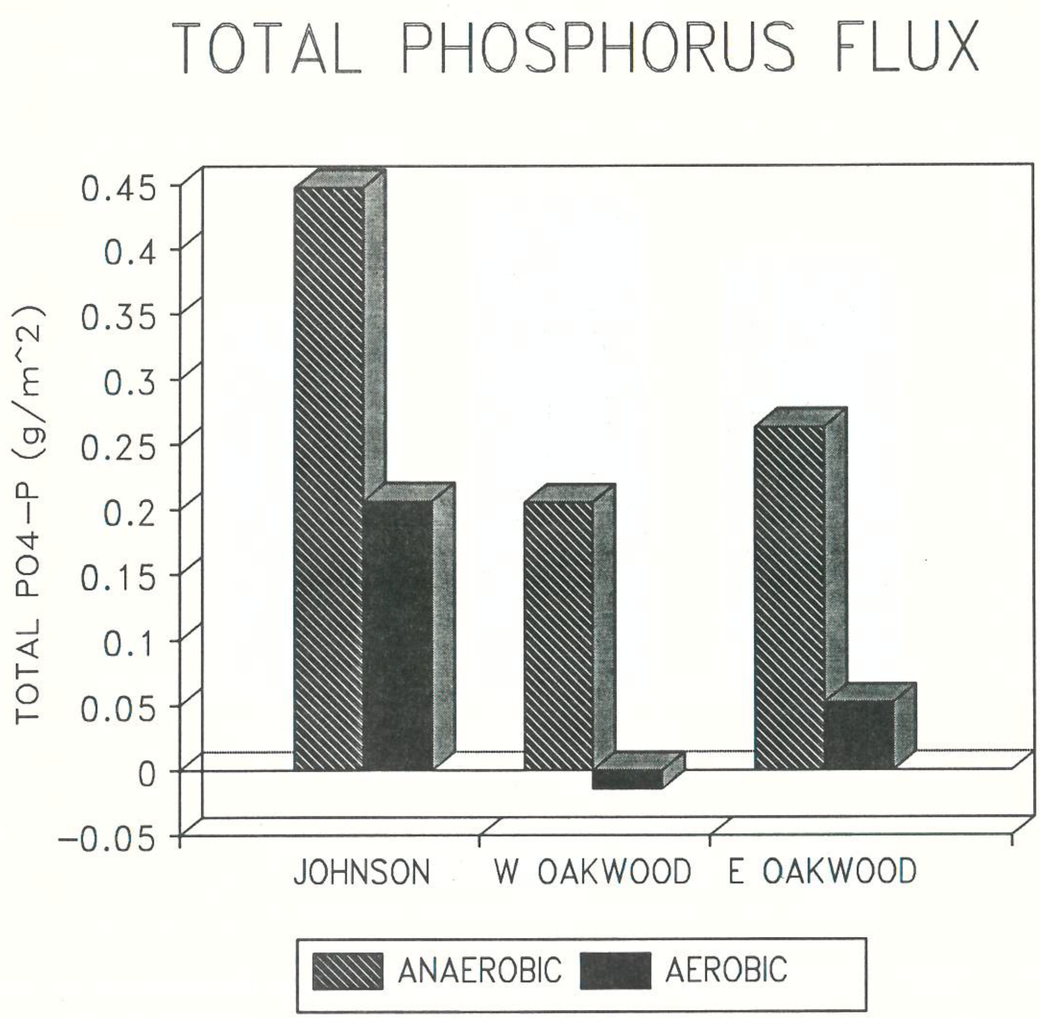


Figure 5-22. Flux of phosphorus between sediment cores and overlying water.

A comparison of the observed tributary phosphorus load and the calculated sediment phosphorus load to West Oakwood Lake and East Oakwood Lake is presented in Figure 5-23. The sediment load during anaerobic conditions is equal to or greater than the tributary loadings of phosphorus during 1987 and 1988. During aerobic conditions the sediment load of phosphorus was substantially less then the tributary loads for both years. The results suggest the internal loading of sediment phosphorus may be sustaining the observed in-lake levels of phosphorus during periods of no tributary inputs.

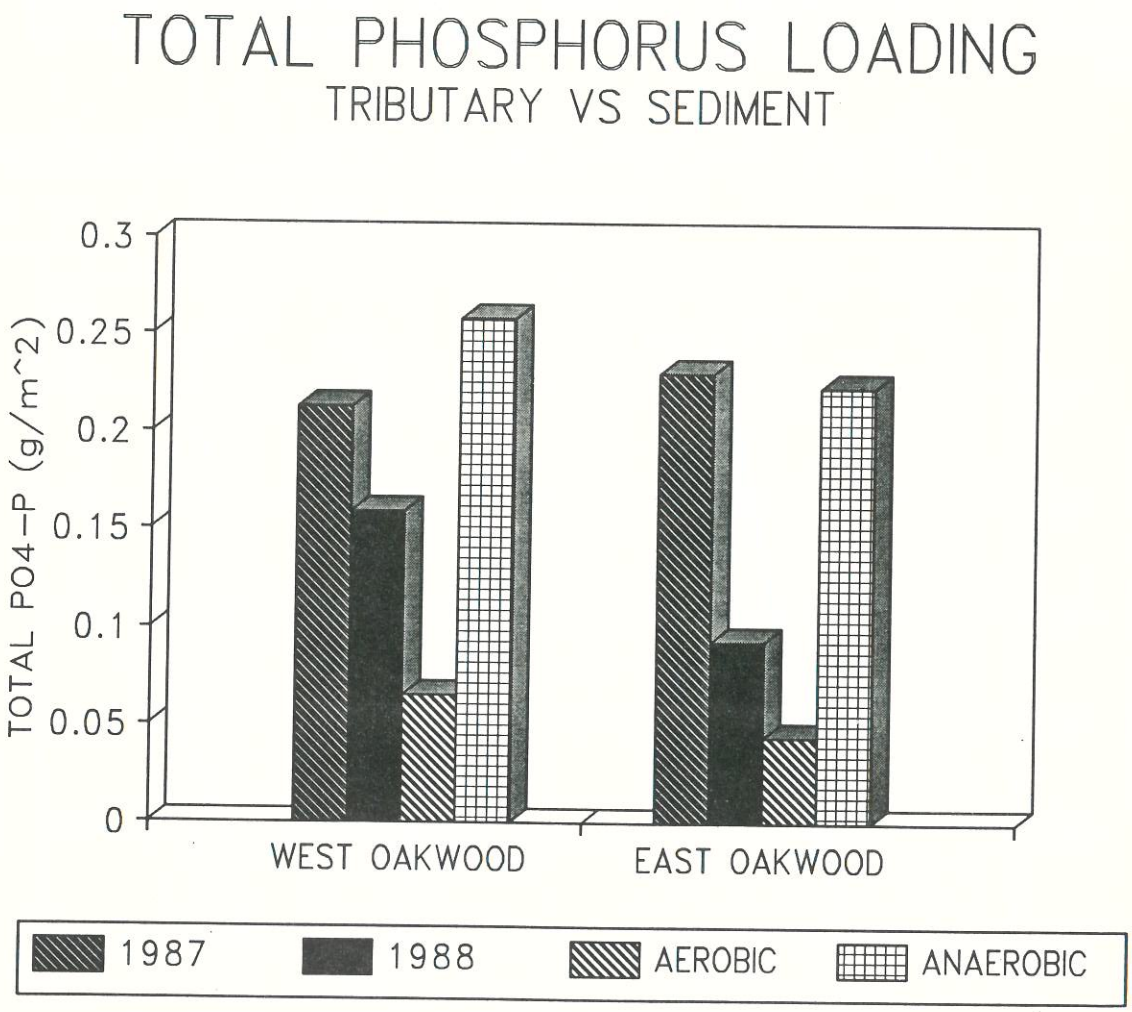


Figure 5-23. Comparison of tributary loadings and potential sediment loadings.

The phosphorus concentration profiles of the incubated sediment cores indicate anaerobic conditions enabled the release of phosphorus from a sediment depth of 45 cm (Figure 5-24). The release of total phosphorus from the anaerobic cores is consistent with the release of inorganic phosphorus whereas organic phosphorus appears to be unaffected. However, organic phosphorus concentration in the anaerobic cores within the 0 cm - 7 cm layer is slightly less than in the aerobic cores suggesting that some mineralization of organic compounds may have occurred in the surface layer. The results agree with other studies reporting the release of inorganic phosphorus during anaerobic conditions (Hosomi et al. 1982).

The ANOVA of the inorganic phosphorus content of the incubated cores detected a significant difference between treatments at the .05 significance level. ANOVA was not able to detect a treatment by lake interaction, indicating incubation had an equal effect on the sediment phosphorus content in all lakes.

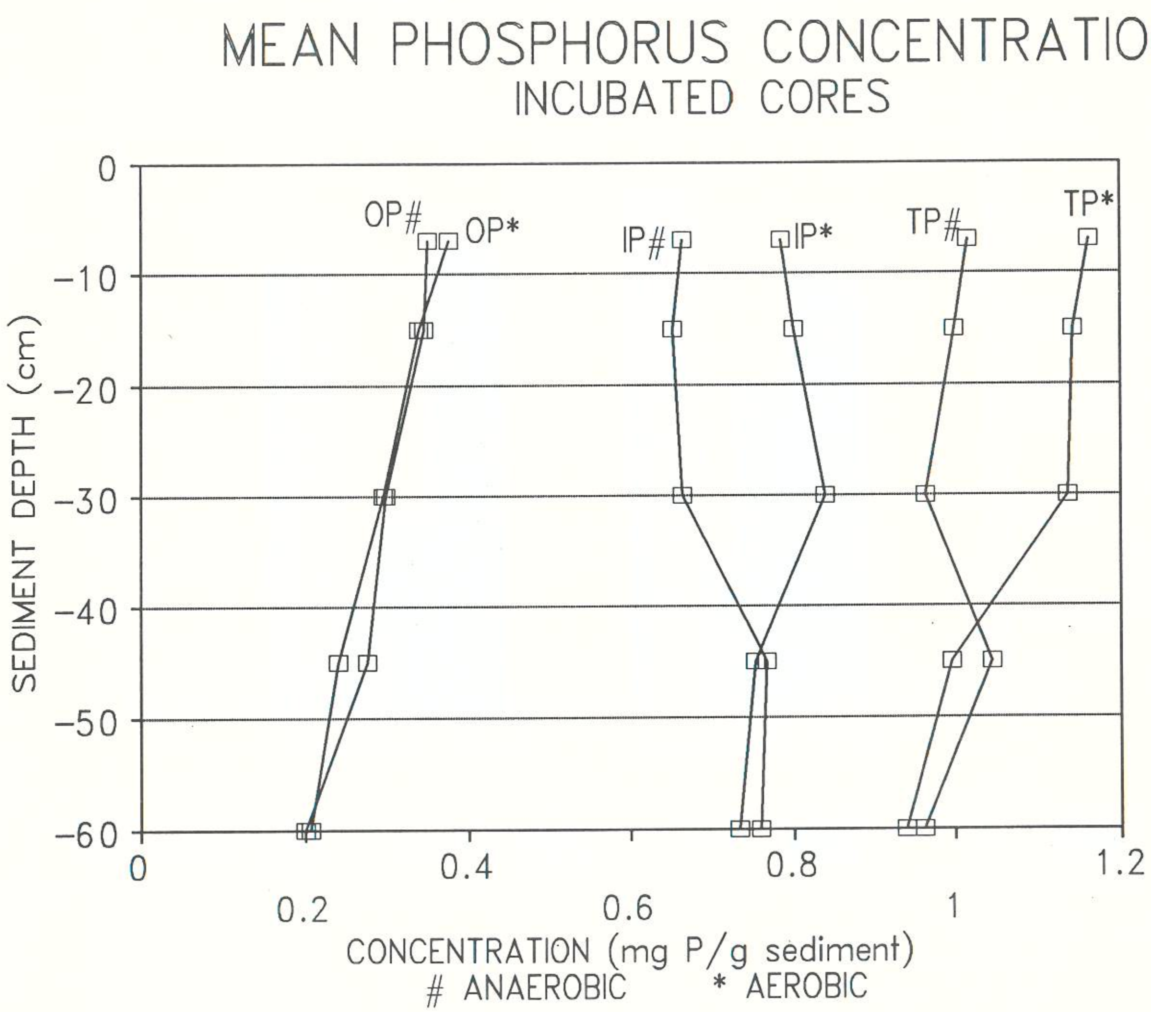


Figure 5-24. Mean phosphorus concentration of incubated sediment cores.

5.6.0 Literature Cited

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6.0 REPORT MANAGEMENT

6.1 Preparation of the FY 91 Comprehensive Report

It is anticipated all monitoring activities will be completed by December 1990, and all comprehensive report writing activities will be accomplished during 1991. The final comprehensive report is scheduled to be completed for distribution by the State Coordinating Committee by November 15, 1991.

A detailed plan of work for the completion of the Oakwood Lakes-Poinsett RCWP comprehensive report has been prepared by the report management (RM) team. This team consists of representatives of SCS, ASCS, CES, DWNR, and WRI. The District Conservationists from all three counties are also involved.

Table 6-1 describes in tabular form the detailed work plan for final report preparation. The agency responsible for each section of the report has been identified. Information for the individual sections will be required from several different agencies, but one individual will be responsible for organizing the information into a readable format. Dates have been established for the completion of the draft and final copies of each section. Review groups have also been identified to review and comment on individual sections.

The RM team has discussed the advantages of having a project leader(s) responsible for ensuring the report preparation is on schedule, reviews are completed, information is collected and available, and the report is compiled into one working document within the established time frames. This will be done through the RCWP Coordinator and CM&E Project Leader.

PLAN OF WORK FOR PUBLICATION AND TECHNOLOGY TRANSFER
OAKWOOD-POINSETT RCWP, SOUTH DAKOTA

Publication	Responsible Organization	Input From	Responsible Person	Review Group	Date to Complete Draft	Date to Complete Final	Remarks
Executive Summary	RM Team - SCS	WRI, DWNR, SCS, ES, ASCS, ERS	M. Kuck	SCC	6/91	11/15/91	
Project Findings & Recommendations	WRI	WRI, CES, DWNR ASCS, SCS, ERS	A. Bender	RM Team SCC	3/91	9/91	
Introduction & Project Background	DWNR	WRI, SCS, ES, ERS, ASCS, LCC, SCC	J. Goodman	Report Team	05/90	6/91	
RCWP Implementation	ASCS	ASCS, SCS, DWNR, WRI	P. Nettinga	SCC RM Team	9/90	4/91	
Project Information & Educational Activities	ES	ES, ASCS, SCS, DWNR, WRI, LCC, SCC	C. Ullery	SCC RM Team	11/90	10/91	
Evaluation of Institutional Relationships & Economics	ERS	WRI, DWNR, ERS, SCS, ASCS	D. Magleby A. Bender L. Holtsclaw	SCC	3/91	9/91	
Impact of BMP's on Surface Water Quality	WRI	WRI, CES, SCS, DWNR, LCC	D. German	RM Team LCC	9/90	12/90	
Impact of BMP's on Ground Water Quality	DWNR	DWNR, WRI	J. Davis	RM Team	3/91	9/91	
Tracking through Vadose Zone	WRI, ES	ES, (Lemme) SDSU	J. Bischoff	RM Team	12/90	4/91	
Synopsis	ES-SDSU	ERS, ASCS, SCS, DWNR, WRI, EPA, CD, LCC, ES	C. Ullery	SCC, Vermont RCWP	8/91	11/15/91	

(The last annual technical report will be May 1990)
(The last annual progress report will be Nov. 1990)

APPENDIX A

Personnel Directory

AGRICULTURAL CHEMICAL LEACHING STUDY

Principal Investigator John H. Bischoff, Research Associate, Water Resources Institute, SDSU
Dave Bjorneberg, Graduate Research Assistant, Water Resources Institute, SDSU

FIELD SITES

Principal Investigator John R. Davis, Hydrologist, Department of Water and Natural Resources
Principal Investigator John H. Bischoff, Research Associate, Water Resources Institute, SDSU
William Best, Research Assistant, Water Resources Institute, SDSU
Kevin Benck, Natural Resources Technician, Water Resources Institute, SDSU

OAKWOOD LAKES SYSTEM STUDY (OLSS)

Objective 1--Principal Investigator Alan Bender, Coordinator of SWRIP, Water Resources Institute, SDSU

Objective 2--Principal Investigator C. Gregg Carlson, Associate Professor, Plant Science Department, SDSU

Objective 3--Principal Investigator Dave German, Research Associate, Water Resources Institute, SDSU
Mary Price, Graduate Research Assistant, Water Resources Institute, SDSU
Steve Grosz, Graduate Research Assistant, Water Resources Institute, SDSU
Objective 4--Principal Investigator Charles Berry, Professor, Wildlife and Fisheries Department, SDSU
Principal Investigator Walt Duffy, Associate Professor, Wildlife and Fisheries Department, SDSU
Richard Brown, Resource Specialist, Wildlife and Fisheries Department, SDSU
Scott Buskerud, Graduate Research Assistant, Biology Department, SDSU
Diane Aston, Research Assistant, Wildlife and Fisheries Department, SDSU
Margaret Chapman, Research Assistant, Wildlife and Fisheries, Department, SDSU

LAND SURFACE NUTRIENT BUDGET

Principal Investigator Teresa Lemme, Microbiology Department, SDSU

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CM&E Project Director Jeanne Goodman, Department of Water and Natural Resources
CM&E Project Coordinator Alan R. Bender, SWRIP Coordinator, Water Resources Institute, SDSU
Cheryl Beste, Senior Secretary, Water Resources Institute, SDSU

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Project Coordinator Michael D. Kuck, Area Engineer, Soil Conservation Service
LeRoy Holtsclaw, Water Quality Specialist, Soil Conservation Service
Paul Nettinga, Conservation Specialist, Agricultural Stabilization and Conservation Service
Diane Clayton, Program Assistant, Agricultural Stabilization and Conservation Service

WATER QUALITY LABORATORY

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Cheri Rath, Research Assistant, Water Resources Institute, SDSU

PESTICIDE LABORATORY

James Rice, Assistant Professor, Chemistry Dept., SDSU
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APPENDIX B

Project Documents: Oakwood Lakes-Poinsett, South Dakota

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