Contract Report 539

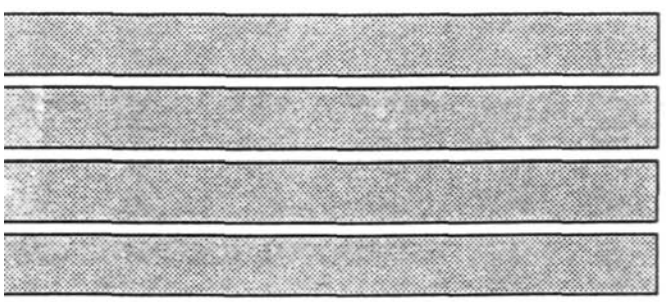
Assessment of the Proposed Discharge of Ground Water to Surface Waters of the American Bottoms Area of Southwestern Illinois

Kenneth R. Rehfeldt
Office of Ground-Water Quality

Raman K. Raman and Shun Dar Lin
Office of Water Quality Management

Robert E. Broms
Goodwin & Broms, Inc.

Prepared for the
U.S. Army Corps of Engineers, St. Louis District



September 1992

Illinois State Water Survey
Hydrology and Chemistry Divisions
Champaign, Illinois

A Division of the Illinois Department of Energy and Natural Resources

ASSESSMENT OF THE PROPOSED DISCHARGE OF GROUND WATER TO SURFACE WATERS OF THE AMERICAN BOTTOMS AREA OF SOUTHWESTERN ILLINOIS

Ground-Water Quality Assessment of the Shallow Alluvial Aquifer

by Kenneth R. Rehfeldt,
Office of Ground Water Quality

Limnological Characteristics of Horseshoe Lake and the Impact of Ground Water Discharges to the Lake

by Raman K. Raman,
Office of Water Quality Management

Removal of Iron and Manganese from Ground Water

by Shun Dar Lin and Raman K. Raman
Office of Water Quality Management

Summary of Water Quality and Effluent Issues

by Robert E. Broms
Goodwin & Broms, Inc.

Prepared for

U.S. Army Corps of Engineers, St. Louis District

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EXECUTIVE SUMMARY

The high ground-water levels encountered during recent years in the alluvial aquifer of the American Bottoms in southwestern Illinois causes flooding of basements and related structural damage, sewer breakage, increased infiltration, and consequent increases in wastewater treatment costs. In response, the U.S. Army Corps of Engineers (USACE), St. Louis District, has developed a plan for reduction of damages caused by the ground-water flooding. To lower ground-water levels, the flood mitigation plan calls for the withdrawal of ground water from 60 wells at 57 locations and discharge of the ground water to surface waters in the area. The study area comprises a portion of the American Bottoms. The Illinois State Water Survey (ISWS), in cooperation with the engineering consulting firm of Goodwin & Broms, Inc. undertook this investigation to assess the possible impacts of the discharges of ground water into surface waters. The primary objective of this study was to determine if treatment of ground water will be required prior to discharge to surface water bodies int he area, with particular emphasis placed on the water quality characteristics of Horseshoe Lake. The ISWS investigation had four main components, titled:

1) Ground-Water Quality Assessment of the Shallow Alluvial Aquifer,

2) Limnological Characteristics of Horseshoe Lake and the Impact of Ground-Water Discharges to the Lake,

3) Removal of Iron and Manganese from Ground Water, and

4) Summary of Water Quality and Effluent Issues.

The report mirrors the main components of the investigation and is divided into sections, with separate authorship for the main sections.

Ground-Water Quality Assessment of the Shallow Alluvial Aquifer

The quality of ground water in the American Bottoms from Cahokia to Granite City was investigated to determine the chemical nature of the water that would be withdrawn under the ground-water pumping plan. Ten deep wells were installed for this investigation and were sampled along with 27 shallow wells from another study to assess the lateral and vertical variation of the ground-water quality. Over a period of nearly two years, five sets of samples were collected from various subsets of the total 27 wells. These samples were chemically analyzed for some combination of primary and secondary inorganic constituents, trace metals, and organic compounds.

The 37 wells were installed at 22 locations with up to three wells nested at different depths at each location. The well locations were selected on the basis of a conceptual model of the ground-water flow system. The conceptual model provides a regional picture of the shallow flow system and indicates likely and unlikely regions of

contaminated ground water. The conceptual ground-water model was constructed from information on land use, ground-water flow direction, soil infiltration rate, potential and known sources of contamination, and locations of observed ground-water contamination.

The chemical analyses revealed only five constituents that exceeded the effluent standard, NH_4-N, PO_4-P, total suspended solids, iron, and manganese. Of those five, only iron and manganese exceed the effluent standard at more than one well. It is expected that treatment to remove iron and manganese will be necessary prior to discharge of the ground water. Detectable levels of trace metals and organic compounds were observed in a number of the wells suggesting low levels of contamination in portions of the aquifer. Significant contamination of aquifer, defined by the concentration of one or more contaminants of ten times the water quality standard, is known to exist at 21 sites in the study area. This significant contamination is generally restricted to plumes of various sizes associated with specific sites. The relatively low levels of contamination detected int he monitoring wells simply indicate that those wells were not located within a specific plume of contamination. However, a well that is pumping will alter the ground-water flow system and may cause a plume of contaminated ground water to be drawn into the well.

Limnological Characteristics of Horseshoe Lake and the Impact of Ground-Water Discharges to the Lake

The high ground-water levels encountered in recent years in the American Bottoms area cause flooding of basements and related structural damage, sewer breakage, increased infiltration, and consequent increases in wastewater treatment costs. The flood mitigation plan proposed by the USACE, St. Louis District, calls for distributed ground-water withdrawals from the unconsolidated shallow aquifer to lower the water-table levels. One of the objectives of the ISWS's investigation is to assess the possible impact of the discharges of ground water (after treatment to remove iron and manganese to meet the Illinois Pollution Control Board's (IPCB) effluent standards) into surface waters, specifically with reference to Horseshoe Lake and its major tributary, Nameoki Ditch in Madison County.

Horseshoe Lake, situated about 2 miles east of the main stem of the Mississippi River, is a meander cutoff of the Mississippi. The lake has a water surface area of about 2150 acres at normal pool elevation, with an average depth of 3 feet except for a 50-foot deep hole near the sand-and-gravel operation. Under normal conditions, the three major sources of flow into the lake are urban runoff through Nameoki Ditch, agricultural runoff through the Elm Slough, and treated waste effluent from the Granite City Steel (GCS) facility. A detailed limnological study was conducted during 1990 to establish baseline lake characteristics data to facilitate projecting the possible impact of ground-water discharges into Nameoki Ditch and Horseshoe Lake.

Based on the criteria proposed in the U.S. Environmental Protection Agency's *Clean Lakes Program Guidance Manual* and on the Carlson's trophic state index

values, detailed limnological investigation of Horseshoe Lake indicates that the lake is eutrophic. Water quality of the test wells in the Granite City area indicates that they meet all IPCB general use and effluent standards, with the exception of iron. Concentrations of iron in the test well samples were high compared with levels found in other parts of the state. Controlled ground-water discharge studies indicate that the DO in the ground-water discharges would quickly recover to acceptable levels. There was no detrimental impact reported for the three Illinois lakes that received ground water to augment water resources. With the treatment of ground water to meet effluent standards for total iron, ground-water quality in Granite City is superior to the lake water quality. If the treated water is diverted into the lake, the lake will benefit from the flushing action on algal growth, increased lake clarity, and probable improved sport fisheries. Alternately, GCS, a division of National Steel Corporation, may be willing to use the treated ground water as its process water, if it is delivered to its premises.

Removal of Iron and Manganese from Ground Water

In order to mitigate basement flooding, distributed ground-water withdrawals has been proposed. However, the high concentrations of iron and manganese in ground-water exceed the IPCB's effluent discharge limits. The purpose of this study was to ascertain an effective and economical treatment scheme for the removal of high levels of iron and manganese from the ground water in the American Bottoms area to meet the effluent standard of 2.0 milligrams/liter (mg/L) of iron and 1.0 mg/L of manganese.

This segment of the report covers a detailed literature review on iron and manganese management, laboratory bench studies of iron and manganese removal from ground water, cost estimates for various removal technologies, and supplemental aeration study for iron and manganese removal.

The "jar test" was employed for the diagnostic feasibility study. Water samples from ten wells were subjected to the jar test using 2-liter beakers in a six-place apparatus (two sets for a total of 12 subsamples were examined for each well sample). Vessels were dosed with zero and incremental amounts of a chosen oxidant. Three oxidants, chlorine (NaOCl), hydrogen peroxide (H_2O_2), and potassium permanganate ($KMnO_4$) were investigated. For each well, at least two samples were examined on two different dates.

The time for the first appearance of floe generally ranged from 30 to 60 seconds. The flocs produced by NaOCl, H_2O_2, and by $KMnO_4$ all settled well in 40 minutes. These left supernatant very clear. However, hours later it turned to yellow with turbidity of 55 to 670 nephelometric turbidity units (NTU). Volumes of residues generated by the chemical oxidation process were estimated. It was observed that the higher the oxidant dose, the greater the amount of solids.

Various oxidant dosages did not cause pH changes. The pH of ground water was between 6.9 and 7.5. The requirement of alkalinity for iron and manganese oxidation was much less than the stoichiometric amounts.

Iron levels of the ten wells studied ranged from 5.2 to 52 mg/L. Nine out of ten well samples met IPCB's manganese limit of 1.0 mg/L. Aeration, even for hours, was not effective in total iron and manganese removal.

Each oxidant required for each well water to meet the IPCB's total iron limit designated as critical dosage is presented in table 4.3. The dosage requirements generally follow the order of NaOCl as Cl_2 > $KMnO_4$ > H_2O_2. The use of $KMnO_4$ increased soluble manganese level, which will result in noncompliance with respect to manganese limit. No single oxidant can treat water from well ME18D to meet both total iron and manganese standards. The amount of an oxidant to oxidize soluble iron was observed to be less than the theoretical value. This is probably due to catalytic reactions of precipitates. Manganese can be oxidized by high concentrations of NaOCl. The possible production of trihalomethanes is a matter of concern.

The quantities of residuals generated by oxidant at the critical dosage are $KMnO_4$ > NaOCl > H_2O_2. Cost for iron and manganese removal from ground waters by H_2O_2 oxidation was the lowest. On the basis of critical oxidant dosage, solids production, and cost, H_2O_2 is recommended as the oxidant of choice.

In a supplemental study, aeration followed by polymer coagulation was found to be effective for iron removal. A 20-40 minute aeration with very small dosage of anionic polymer will treat the ground water to meet the IPCB's iron effluent standard.

Summary of Water Quality and Effluent Issues

- The ground water underlying the American Bottoms area contains iron concentrations in excess of the Illinois Environmental Protection Agency's (IEPA) effluent and water quality standards.

- In order to pump the ground water and discharge it to surface waters, IEPA permits will have to be obtained. In order to obtain the necessary permits, either treatment of the ground water or changes to the effluent and water quality standards are required. If a rule change is sought, there will likely be some public opposition to the degradation of the surface waters, regardless of any public benefit to the lowering of the ground-water table. For any of the 57 locations, it is unlikely that a site-specific rule change to allow discharge without treatment would be successful, given the high iron concentration in the ground water.

- Providing treatment of the ground water pumped nearly doubles the capital cost of the project.

- Petitioning for site-specific rule changes for each discharge point would likely have a total cost of $250,000, and take more than two years to complete with no guarantee that any or all the petitions would necessarily be successful.

- Of the proposed 60 ground-water pumping wells at 57 locations, there is a potential for significant ground-water contamination being drawn into 20 wells at 18 locations. While the costs involved with the potential treatment of the contaminated ground water to achieve effluent and water quality standards are beyond the scope of this report, it would be fair to characterize the treatment for contamination as significantly more expensive, both to construct and to operate, than the iron-removal treatment system. Thus, the potential contamination will likely eliminate those 18 well locations from consideration as part of the overall pumpage system. The cost of long term monitoring of the well discharge for hazardous chemicals is also not included in these cost estimates.

- Excluding the 18 potentially contaminated wells, reduces the amount of ground water pumped from 41.25 million gallons per day (mgd) to 26.0 mgd.

- It is recommended that the Cost/Benefit Analysis, which was completed in 1987, be reanalyzed to determine if the project is still viable if:
    a) The flow rate is reduced to 26 mgd.
    b) Treatment of the ground water is required.
    c) The project is on hold for two or more years in order to obtain the rule change.
    d) The cost of the rule change ($250,000) is factored into the analysis.

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The majority of this investigation was funded by the U.S. Army Corps of Engineers (USACE), St. Louis Division, St. Louis, MO, through the Illinois Department of Transportation (IDOT), Division of Water Resources. The authors gratefully acknowledge the project coordination, guidance, and management provided by Mr. Daniel Marshall (USACE) and the administrative support of Mr. Siavash Mostoufi (IDOT). Several of the Corps staff members provided valuable comments on the draft report. Additional support for the installation and monitoring of the shallow wells was provided by the Illinois Department of Energy and Natural Resources through the Aquifer Assessment Program under the management of Mr. David Baker and Ms. Karen Miller.

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Section 1

INTRODUCTION

The high ground-water levels encountered during recent years in the alluvial aquifer of the American Bottoms in southwestern Illinois causes flooding of basements and related stuctural damage, sewer breakage, increased infiltration, and consequent increases in wastewater treatment costs. In response, the St. Louis District of the U.S. Army Corps of Engineers (USACE) has developed a plan for reduction of damages caused by the ground-water flooding (USACE, 1987). To lower ground-water levels, the flood mitigation plan calls for the withdrawal of ground water from 60 wells at 57 locations and discharge of the ground water to surface waters in the area. The Illinois State Water Survey (ISWS), in cooperation with the engineering consulting firm of Goodwin & Broms, Inc., undertook this investigation to assess the possible impacts of the discharges of ground water into surface waters. The study area (figure 1.1) comprises a portion of the American Bottoms bounded by I-270 on the north, The City of Cahokia on the south, the Mississippi River on the west, and the bluffs on the east. The primary objective of this study was to determine if treatment of ground water will be required prior to discharge to surface water bodies in the area. Particular emphasis was placed on the water quality characteristics of Horseshoe Lake. The ISWS investigation had four main components, titled:

1) Ground-Water Quality Assessment of the Shallow Alluvial Aquifer,

2) Limnological Characteristics of Horseshoe Lake and the Impact of Ground-Water Discharges to the Lake,

3) Removal of Iron and Manganese from Ground Water, and

4) Summary of Water Quality and Effluent Issues.

The first component, an investigation of ground-water quality, characterized the current ground-water quality in the alluvial aquifer of the American Bottoms and also included the sampling of seven surface water sites. The primary objective of this component was to provide the data necessary to determine if treatment of the ground water would be required prior to discharge to surface water. As part of this investigation, ten monitoring wells were installed at depths from 75 to 87 feet to ascertain the quality of ground water at those depths. Herein, these ten wells are referred to as the "deep wells". In addition to the deep wells and surface waters sampled for this study, 27 wells that were installed as part of another investigation into the quality of ground water at depths less than 50 feet (Rehfeldt et al., 1991) were also sampled. Herein, these wells are termed "shallow wells". Where practical, the sampling of the deep and shallow wells was coordinated to provide a larger data set for analysis.

The second component, focused on Horseshoe Lake, was a detailed limnological study designed to establish baseline lake characteristics data. The flood mitigation plan proposed by USACE, calls for discharge of some of the ground water to Horseshoe Lake, a major

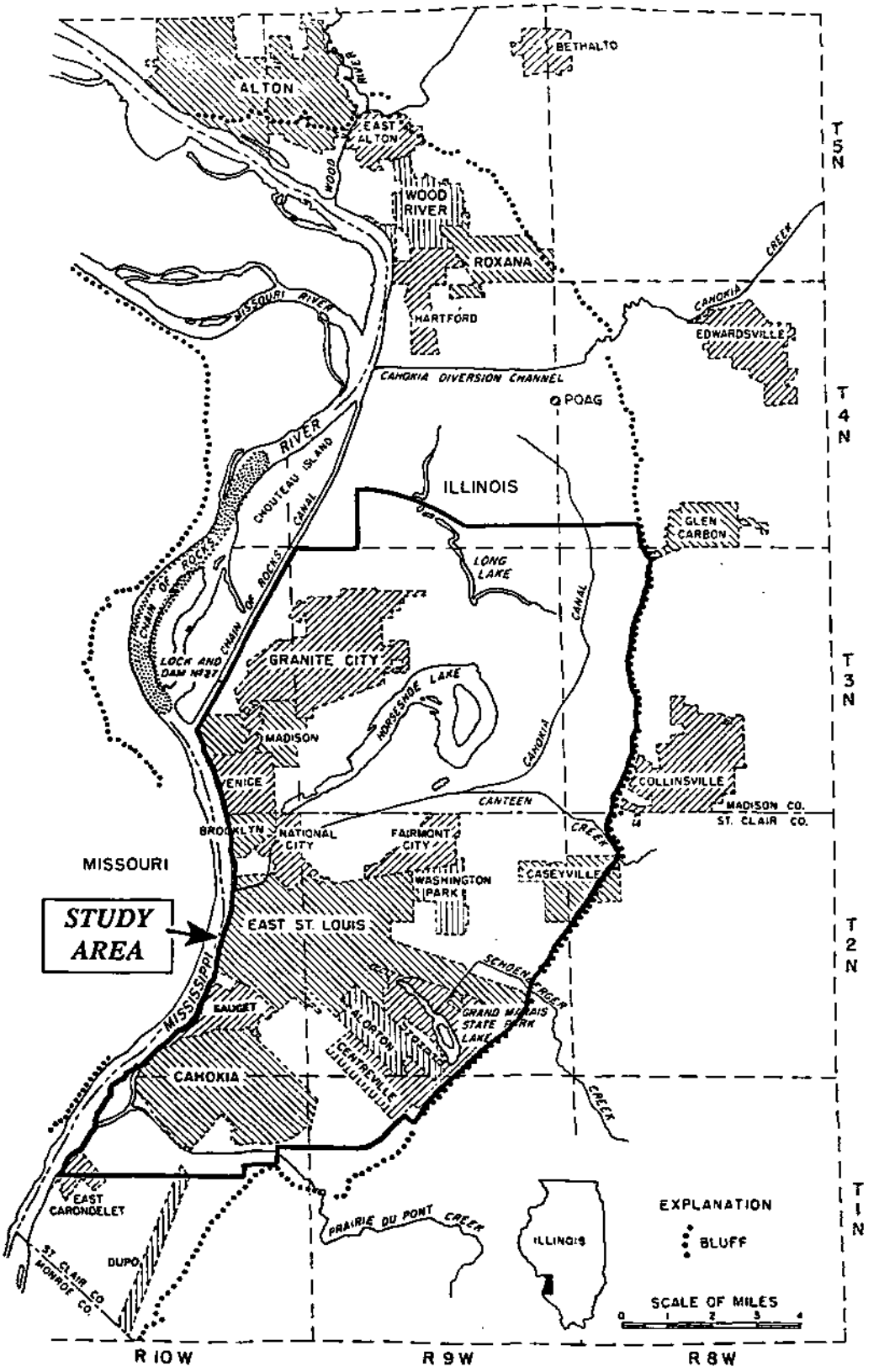


Figure 1.1. American Bottoms study area

recreational resource of the area that could be negatively impacted. The objective of this component of the ISWS investigation was to assess the impact of the discharges of ground water (after treatment to remove iron and manganese to meet the Illinois Pollution Control Board's (IPCB) effluent standards) into surface waters, specially with reference to Horseshoe Lake and its major tributary, Nameoki Ditch in Madison County. The impacts of ground-water discharges on the water quality of Horseshoe Lake were examined in detail using the three-phased approach outlined below:

Phase I:	Assessment of the physical, chemical, and biological ambient water quality conditions of Horseshoe Lake.
Phase II:	Assessment of the changes in water quality characteristics of the Illinois Department of Transportation (IDOT), Division of Water Resources pumpage in underground drainage systems.
Phase D3:	Assessment of the changes in water quality characteristics of controlled well discharges to a surface drain.

The third component was to develop an effective and economical treatment methodology for the removal of high iron and manganese from the ground water. Iron concentrations of ground water from the deep wells ranged from 7 to 44 milligrams per liter (mg/L), well in excess of the 2 mg/L effluent water quality standard (IEPA, 1988). Treatment of the ground water to remove iron and manganese prior to discharge to surface waters will be necessary to meet the effluent standard. This section examines several approaches to iron and manganese removal, determined the most cost-effective method, and estimated the costs of construction of such treatment facilities.

The fourth component summarizes the first three components with respect to the key issues of water quality standards, effluent standards, disharge permits, rule changes, and the costs involved.

The report mirrors the main components of the investigation and is divided into sections, with separate authorship for the main sections. The sections have been combined to make a single document for ease of presentation. The appendices for each section are provided in a companion document titled "Appendices for Assessment of the Proposed Discharge of Ground Water to Surface Waters of the American Bottoms area of Southwestern Illinois."

Section 2

GROUND-WATER QUALITY ASSESSMENT OF THE SHALLOW ALLUVIAL AQUIFER

by Kenneth R. Rehfeldt

APPROACH

Ground-Water Quality

Ground-water quality in the study area was characterized through a program of periodic ground-water sampling, laboratory analysis, and data interpretation. A three-dimensional ground-water quality assessment was accomplished by sampling wells screened over different depths. The sampling well locations were carefully selected to account for the spatial variability of ground-water quality. Water-level measurements were taken to determine the general directions of ground-water movement during the period of study.

The ISWS project team constructed ten dedicated monitoring wells to depths of about 85 feet. The team worked closely with USACE personnel to select these well locations, partly on the basis of a conceptual model of the ground-water flow system. The conceptual model provides a qualitative representation of the shallow ground-water flow system and indicates likely and unlikely regions of contaminated ground water. The conceptual ground-water model was constructed from information on land use, ground-water flow direction, soil infiltration rate, and potential and existing sources of contamination. Ground-water samples were collected from the ten deep monitoring wells and analyzed for a wide range of chemical constituents including heavy metals and organic compounds. In addition, chemical data from ISWS shallow wells installed for another project (Rehfeldt et al., 1991) were also used.

Surface Water Quality

Surface water samples were collected from seven locations, separate from the monitoring well locations, and analyzed for the same compounds as the ground-water samples. These surface water samples will be useful for contrasting current ground-water and surface water quality.

ASSESSMENT OF GROUND-WATER QUALITY

The primary purpose of the ground-water sampling was to assess the quality of the shallow aquifer in the American Bottoms area to determine if the water required treatment before its discharge to surface waters. The large size of the study area makes sampling of ground water difficult with a limited number of wells because of the natural variability in ground-water quality.

Reliance on existing data alone, such as from Environmental Protection Agency (EPA) files, also will produce a very misleading view of regional ground-water quality. A majority of EPA ground- water quality data on organic and metal contaminants comes from sites at which ground-water contamination has already occurred. This information is very useful for documenting existing contamination. However, a compilation of this chemical data from an area such as the American Bottoms would give the false impression that the entire aquifer was contaminated because most samples were of contaminated ground water. Therefore existing data alone cannot be used to give an assessment of regional ground-water quality.

Another approach, the simple random placement of wells, is not likely to yield sufficient data to characterize the ground-water quality in a large region. There is so much variability in the factors that control contamination (such as land use, soil characteristics, and ground-water flow) that a very large number of wells would be required to adequately cover the study area. To better characterize the study area, an approach based on a conceptual model was used in this study.

Conceptual Model of the Ground-Water Flow System

The conceptual model approach is designed to extract the maximum amount of information from each well. The conceptual model is a qualitative description of the flow system that incorporates much of the existing information about potential and active sources of contamination, ground-water flow direction, and ground-water chemistry. The conceptual model is used to identify potential areas with and without contamination. The conceptual model is not quantitative in the sense of an equation or a numerical model. Rather, it summarizes, in a qualitative way, the expected trends or behavior of the ground-water system. A simple example of a conceptual model is the response of a well to pumping. If pumping increases, water levels in the well and the surrounding aquifer decline. This conceptual model of pumping provides the expected behavior of the aquifer (decline in water level), but doesn't yield quantitative results such as the radius of influence or the magnitude of the drawdown. In the same way, the conceptual model developed for the American Bottoms ground-water system cannot, nor is it intended to, provide quantitative results such as expected contaminant concentrations, ground-water velocity, or exact location of plumes. Rather, the conceptual model will provide a qualitative description of where ground-water contamination is expected.

The monitoring wells are placed to confirm the conceptual model. For example, some monitoring wells are placed in regions where contamination is expected and others in regions where contamination is not expected. Once confirmed, the conceptual model can be used to extend the chemistry data from the region immediately around the monitoring well to a much larger area.

Four broad classes of data are brought together to form the conceptual model: ground-water flow direction, soil infiltration rate, potential sources of ground-water contamination, and existing ground-water contamination. Each of these sources of information is important in defining the conceptual model.

The ground-water flow direction helps determine the movement of ground-water contamination. To detect contamination, the monitoring wells must be installed down gradient of contaminant sources. But it is equally important to identify regions unlikely to be contaminated, such as areas up gradient of contaminant sources.

Soil infiltration rate is an indicator of how fast water moves from the land surface to the aquifer. Ground water underlying areas with rapid soil infiltration is more vulnerable to contamination than areas of slow infiltration.

Identifying potential contamination sources to delineate likely contaminated regions of the aquifer is the third key element of the conceptual model. Potential sources include industrial facilities, landfills, and agricultural areas.

Documenting existing contamination is the final element of the conceptual model. Knowledge of existing contamination serves two purposes. The first purpose is a partial confirmation of the conceptual model. In the model, areas of known contamination should fall within regions identified as likely to be contaminated. The second purpose is that monitoring wells do not need to be installed where contamination is known to exist; this avoids duplication of data.

Ground-Water Flow Direction. Ground-water flow direction is determined from a map of the potentiometric surface, the level to which water will rise in a well. To obtain a map of the potentiometric surface, the water level is measured in many wells. The measured elevations are plotted on a map and contours of equal elevation are drawn. Typically, detailed potentiometric surface maps are not available for large areas of Illinois. However, the American Bottoms region is somewhat unique in that the ISWS has had a long-term presence in the region. As a result, the ISWS has periodically obtained potentiometric surface elevations from wells in the region and drawn potentiometric surface maps. Figure 2.1, a map of the potentiometric surface in November 1985, is adapted from data taken from Kohlhase (1987). In November 1990, the ISWS again obtained potentiometric surface elevations from the American Bottoms. A map of a portion of the data corresponding to the study area is presented in figure 2.2. The figure includes locations of the monitoring wells sampled.

To better interpret figures 2.1 and 2.2, the major centers of pumpage in the American Bottoms are shown for 1985 and 1989 (figures 2.3 and 2.4, respectively). These maps are based on pumpage data collected by the ISWS and summarized in reports such as Kirk et al. (1982, 1984, and 1985). The pumpage data are part of the ISWS's continuing effort to collect and maintain information on water usage in Illinois. The 1990 pumpage data were not available, but they are not expected to significantly differ from the 1989 data. As indicated in figures 2.3 and 2.4, the distribution of pumpage in the study area has not changed dramatically between 1985 and 1989. Several features of the potentiometric surfaces are clearly the result of ground-water pumpage.

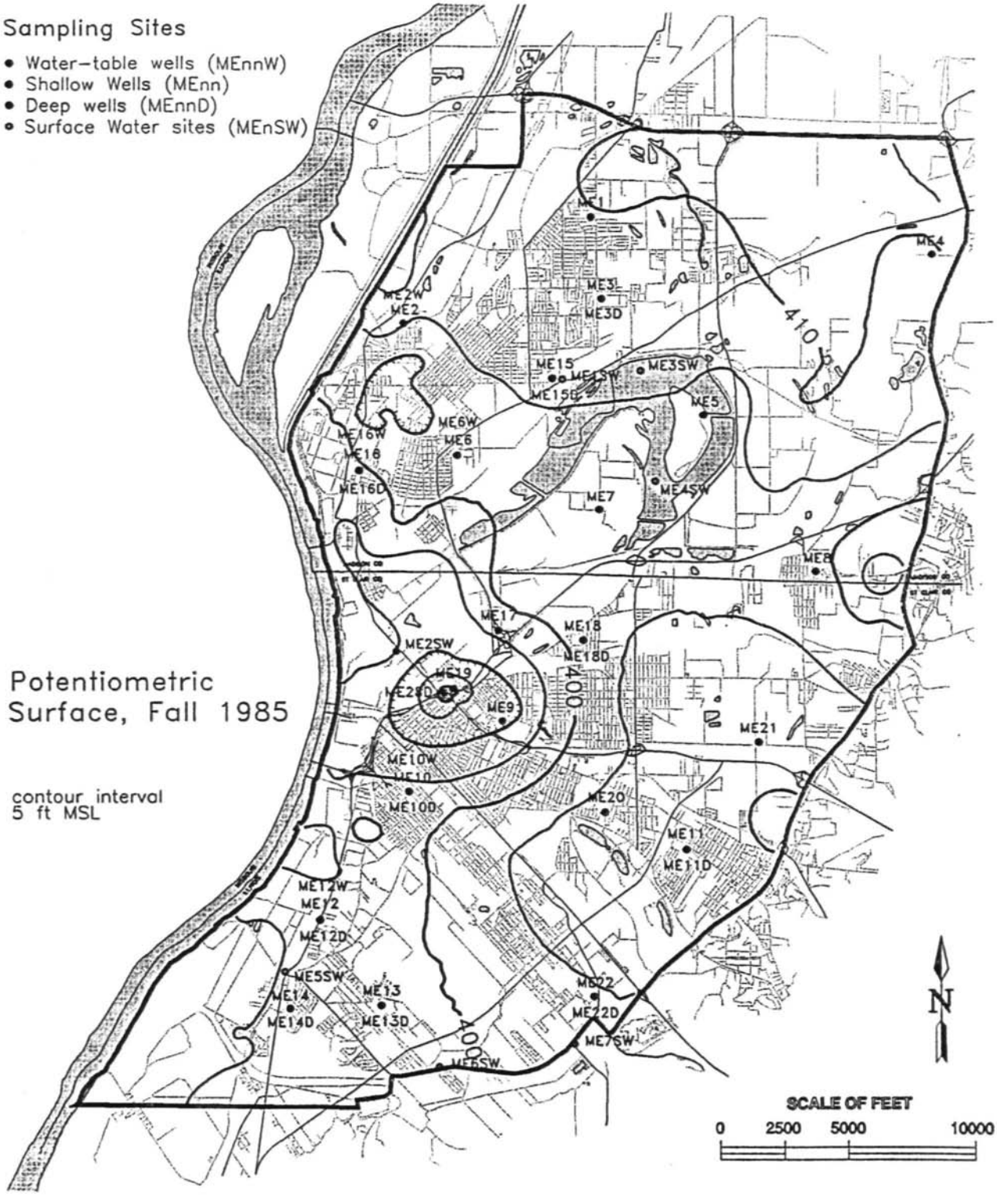


Figure 2.1. Potentiometric surface map, Fall 1985

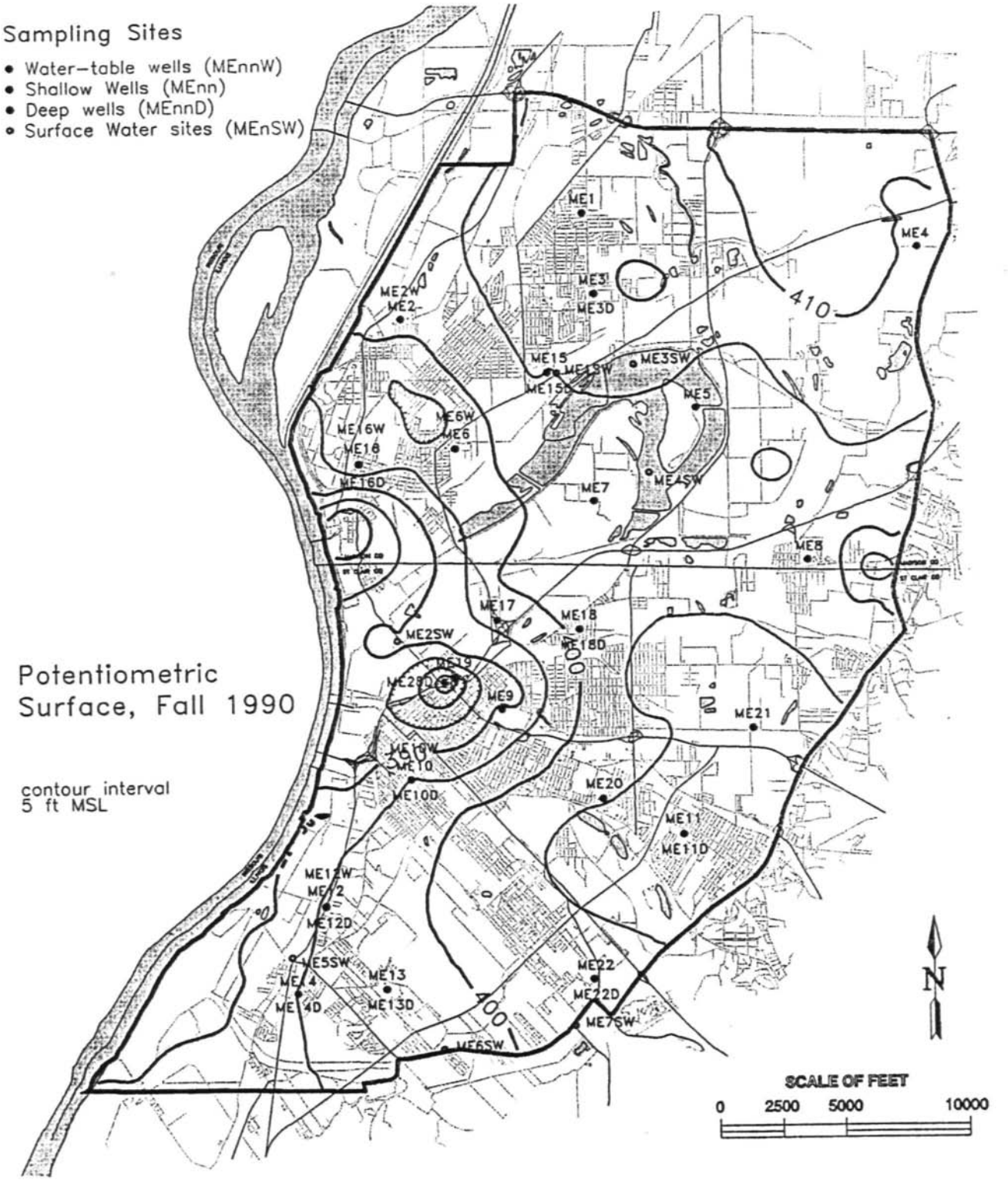


Figure 2.2. Potentiometric surface map, Fall 1990

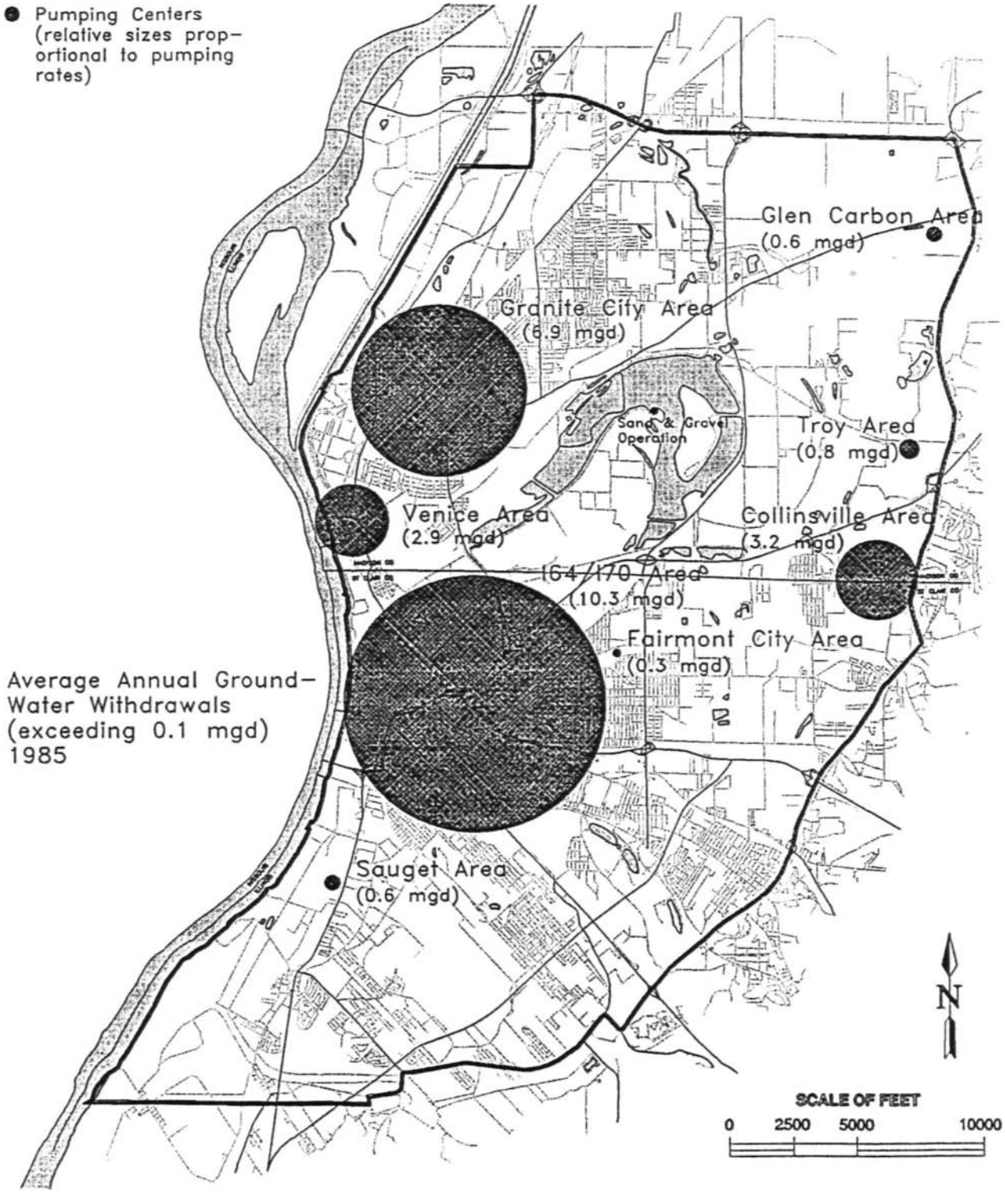


Figure 2.3. Distribution of ground-water pumpage, 1985

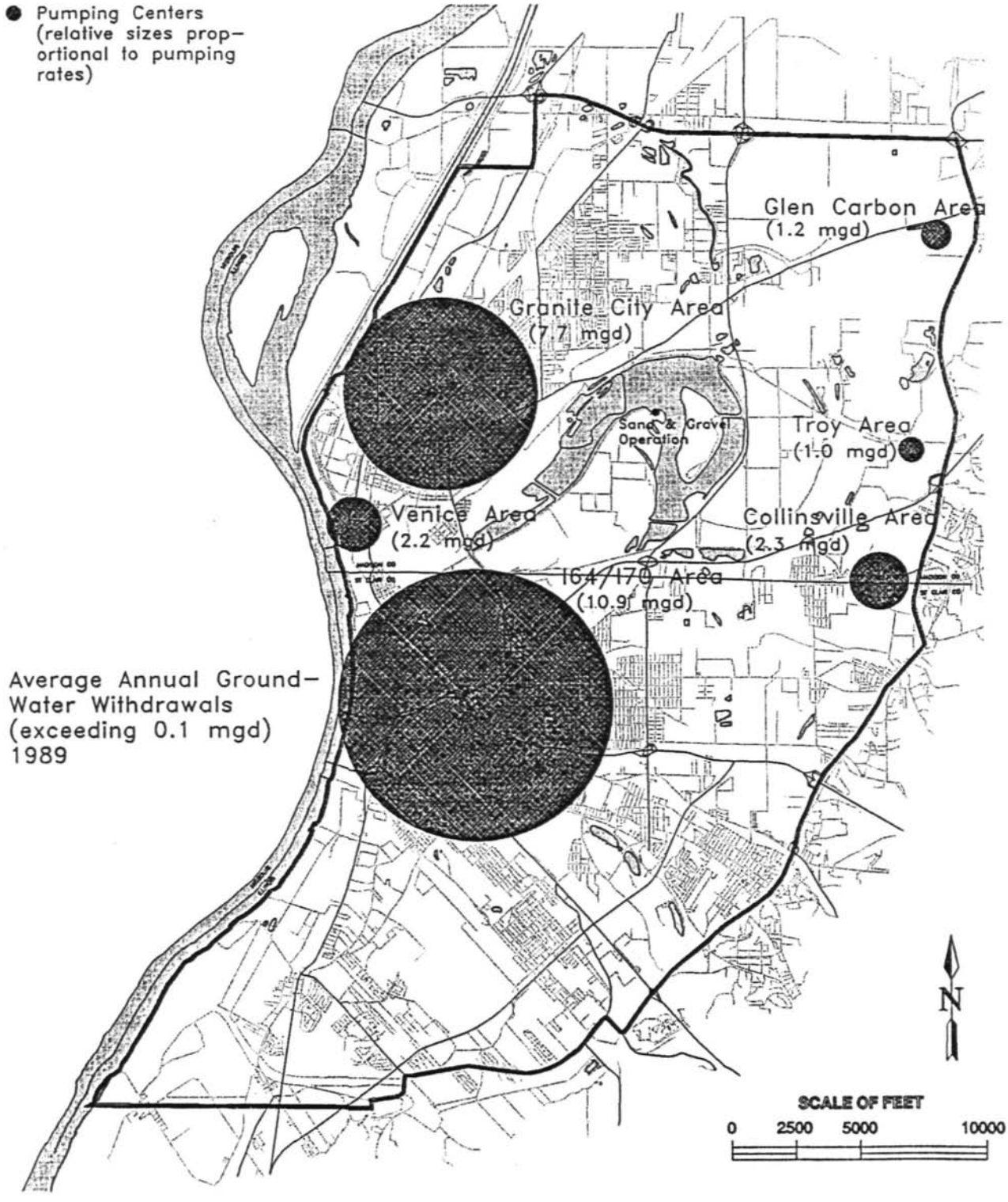


Figure 2.4. Distribution of ground-water pumpage, 1989

First, a large portion of the central study area is dominated by flow to the dewatering wells operated by the IDOT at the I-64/I-70 interchange and by some industrial usage. Several discharge areas are located along the eastern edge of the study area and are associated with the pumpage by the cities of Collinsville, Troy, Glen Carbon, and Maryville. Two other discharge areas are north of the IDOT I-64/I-70 dewatering wells. One pumpage center near Venice is the result of the IDOT dewatering operations at an underpass. The other major area is associated with industrial pumpage in Madison and Granite City. Of note in figures 2.1 and 2.2 are the general directions of flow (toward the Mississippi River or the pumpage centers near the river), which have changed little between 1985 and 1990.

The aquifer is hydraulically connected to Horseshoe Lake by two mechanisms: 1) slow movement through the bottom sediments of the lake and 2) through a hole excavated through the bottom of the lake as the result of a sand-and-gravel mining operation (figures 2.3 and 2.4). Butts and Evans (1979) note that Horseshoe Lake is about 50-feet deep at the sand-and-gravel operation, which provides a direct and highly permeable connection between the aquifer and Horseshoe Lake. The volume of water that could pass through that connection is unknown, but would be a function of the hydraulic head difference between the lake and the aquifer.

Soil Infiltration Rate. The soil infiltration rate will influence the distribution of contaminants in the aquifer. Contamination can be transported very quickly from the land surface to the aquifer in regions with rapid infiltration of water through the soil. Figure 2.5, adapted from the Southwestern Illinois Metropolitan and Regional Planning Commission (SIMRPC) (1975), shows the relative infiltration rate of the soils in the study area. In general, the infiltration rate is faster in areas of urban activity. This is consistent with the observations of SIMRPC that the areas of slow infiltration are generally low lying, poorly drained, and are poor locations for urban development.

Urban areas, are more likely to contain industrial contaminant sources, but agricultural contaminant sources are more often associated with rural areas. In general, the faster soil infiltration rate in the urban areas would be expected to enhance the movement of industrial contaminants into the aquifer compared with agricultural contaminants, which are more often associated with regions of slow infiltration.

The distribution of the surficial materials in the vicinity of Granite City is significantly more complex than is shown in figure 2.5 (Ed Smith, personal communication, 1991). However, for the purposes of this study, the amount detail is adequate. More information regarding the infiltration rates of the soils in the study area can be found in Wallace (1978) and Goddard and Sabata (1986).

Potential Sources of Contamination. Another important component of the conceptual model is the identification of potential sources of contamination. Land use is expected to have a strong influence on the potential for contamination (Cain et al., 1989). Industrialized areas are more likely to be sources of contaminants such as organic solvents and heavy metals than residential areas. Additionally, agricultural areas may be sources of pesticides, herbicides, and nitrates. The land use of the study area was obtained from a database

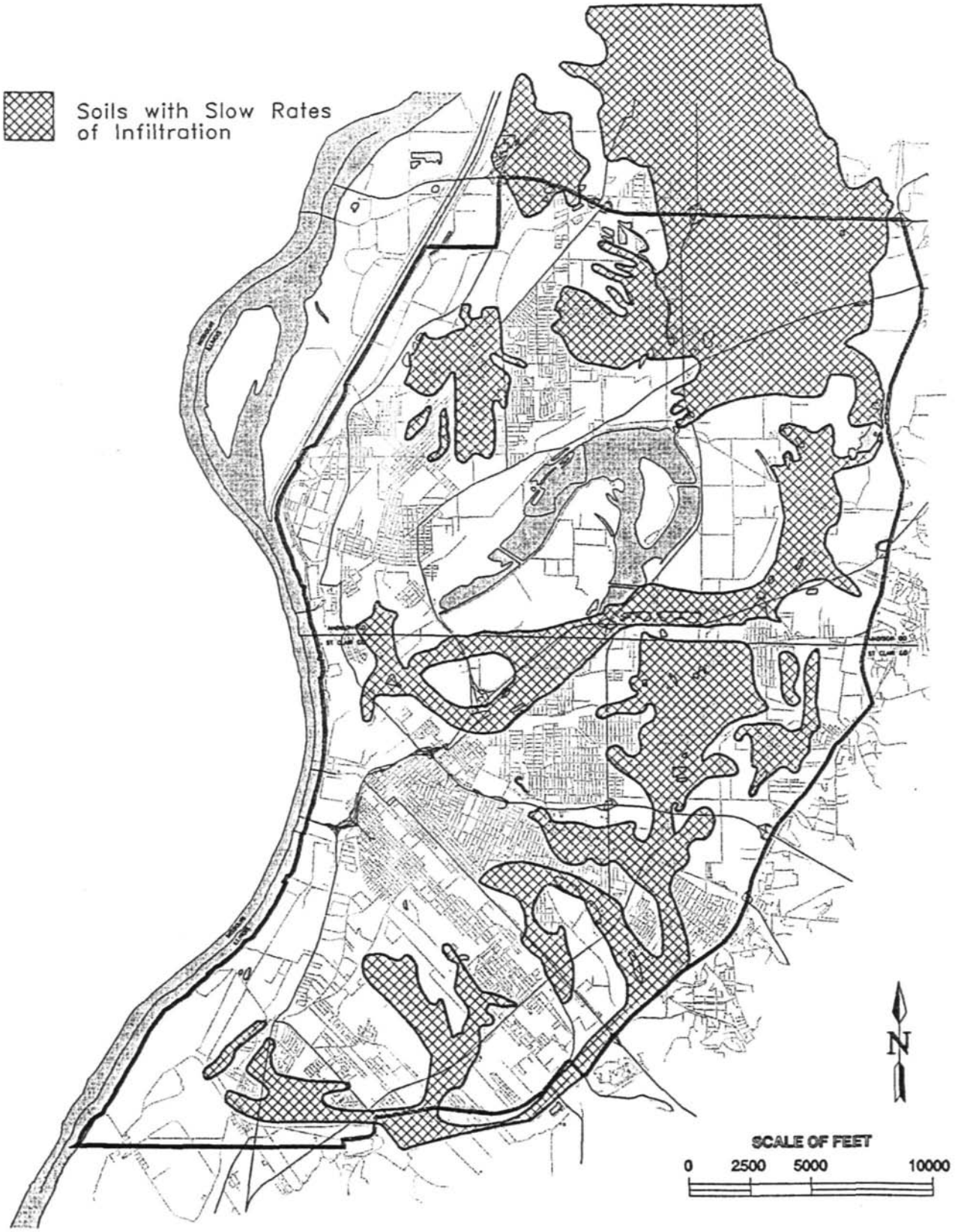


Figure 2.5. Distribution of relative soil infiltration rate

created by the U.S. Geological Survey (USGS) (Fegeas et al., 1983). Two levels of classification are available. The level I classifications are: urban land, agricultural land, forested land, open water, wetlands, and barren land. Figure 2.6 is a map of the level I land use of the study area. Each of the level I classifications can be further subdivided into level II classifications. The area within the level I urban land classification was further subdivided (figure 2.7) into the following level II classifications: residential, commercial and services, industrial, transportation and communications, mixed urban or built-up land, and other urban or built-up land. Of primary interest in terms of potential pollutant sources were the areas of industrial, commercial, and transportation land use.

Other potential sources of contaminants are land-based disposal sites. A compilation of present and former land disposal sites, designated as filled land on Figure 2.8, was obtained from the Hazardous Waste Research and Information Center (HWRIC). That database is described in four recent publications (Dixon et al., 1986; Colten, 1988; Mehnert and Keefer, 1988; Mehnert and Mushrush, 1990). It is important to note that each of these sites does not necessarily contain hazardous material. But as a group, land disposal sites are regarded as potential sources of contamination, because in nearly all cases the content of the fill material is unknown. Exact locations of many of the land disposal sites are not known, so most of the locations in figure 2.8 are approximate and cannot be used to pinpoint specific contaminant sources. However, the locations are very useful for viewing the relative density of land-based disposal sites in different parts of the study area. It is expected that areas with the greatest density of land-based disposal sites would have the highest propensity for ground-water contamination.

Also shown on figure 2.8 are current and former EPA-regulated facilities. A regulated facility is a site for which Illinois EPA has a file under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), Resource Conservation and Recovery Act (RCRA), or Leaking Underground Storage Tank (LUST) programs. The 19 LUST sites in figure 2.8 represent the worst of the approximately 100 incidents of leaks in the study area. The regulated facility data were obtained from files of the Illinois EPA in Collinsville, Illinois, and are summarized in table 2.1. Files for a majority of these CERCLA and RCRA sites date back to the early 1980s, shortly after CERCLA became law in 1980. Each site is identified by name, type, and location to the nearest section. Although nearly all of the 72 sites listed in the table have data on soil contamination, only 29 sites have ground-water contamination data. The last two columns in table 2.1 list the ranking of the sites in terms of ground-water contamination, if ground-water data were collected, or the potential for ground-water contamination if ground-water sampling has not yet begun. The rankings are defined as:

Column 4: Ground Water Contamination

significant: One or more contaminants are present in the ground water at concentrations exceeding an applicable maximum concentration limit by a factor of ten.

observed: Contamination of ground water is observed, but the concentrations do not warrant a classification of significant.

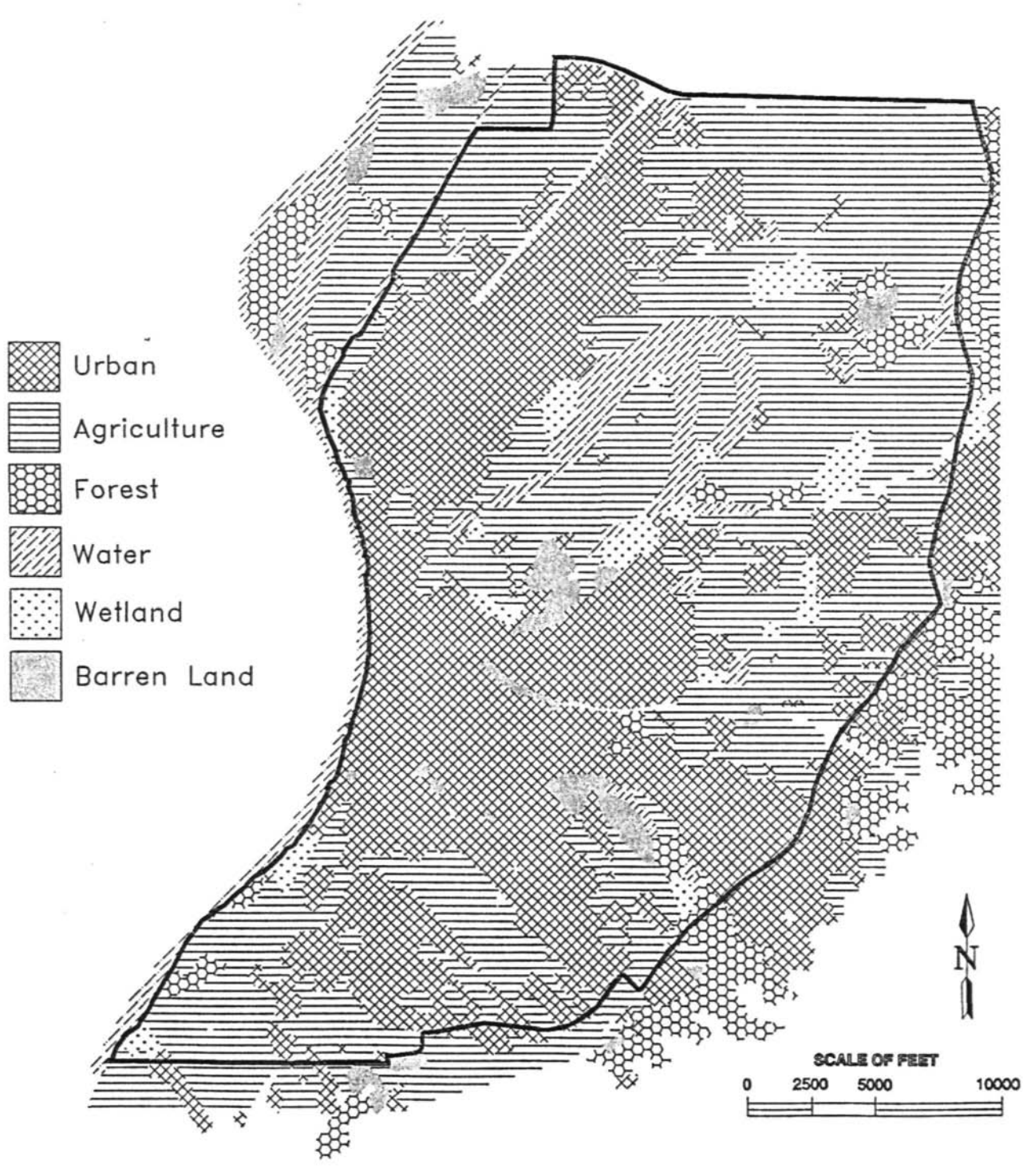


Figure 2.6. Level I land use

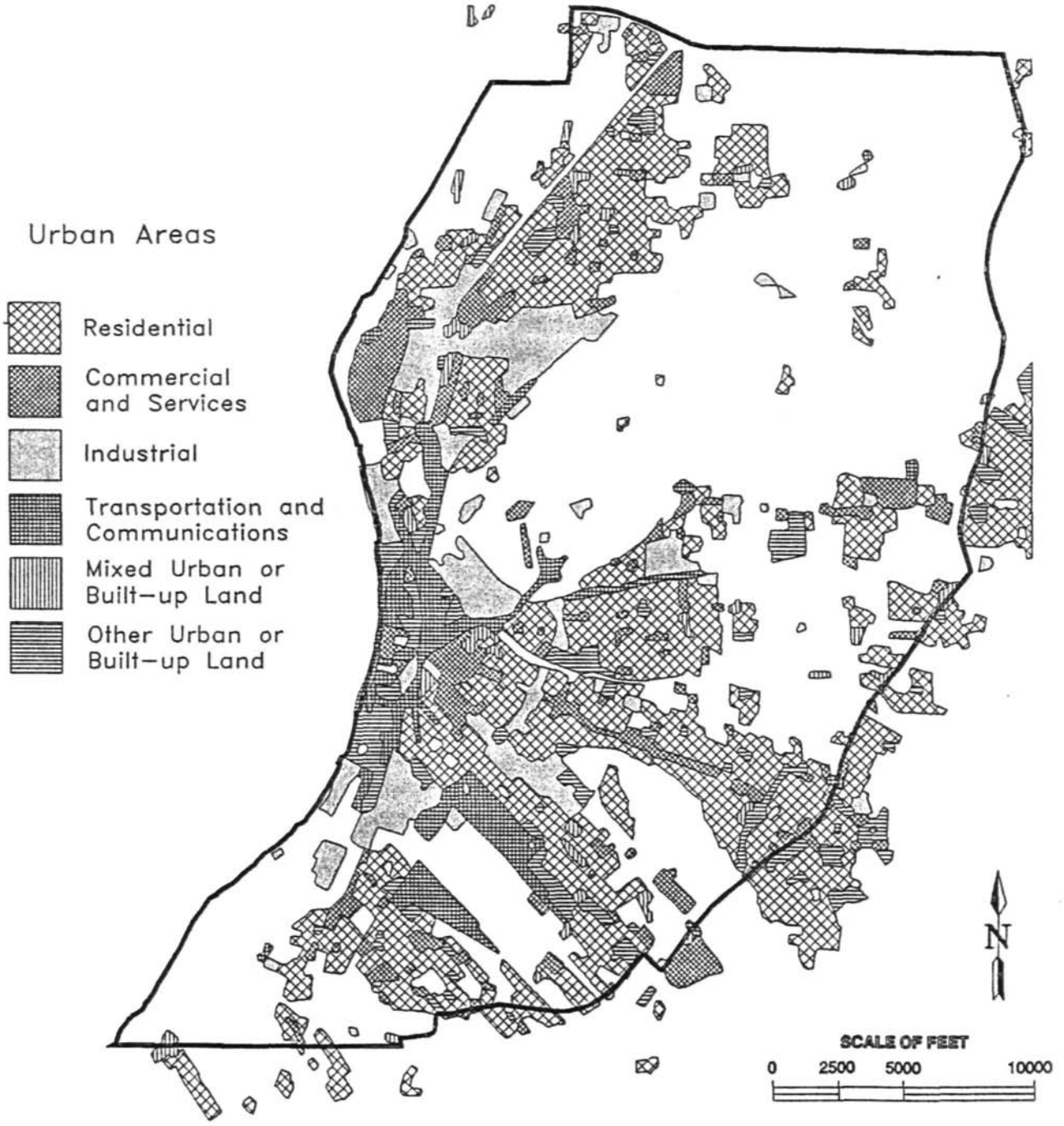


Figure 2.7. Level II land use of the level I urban land use areas

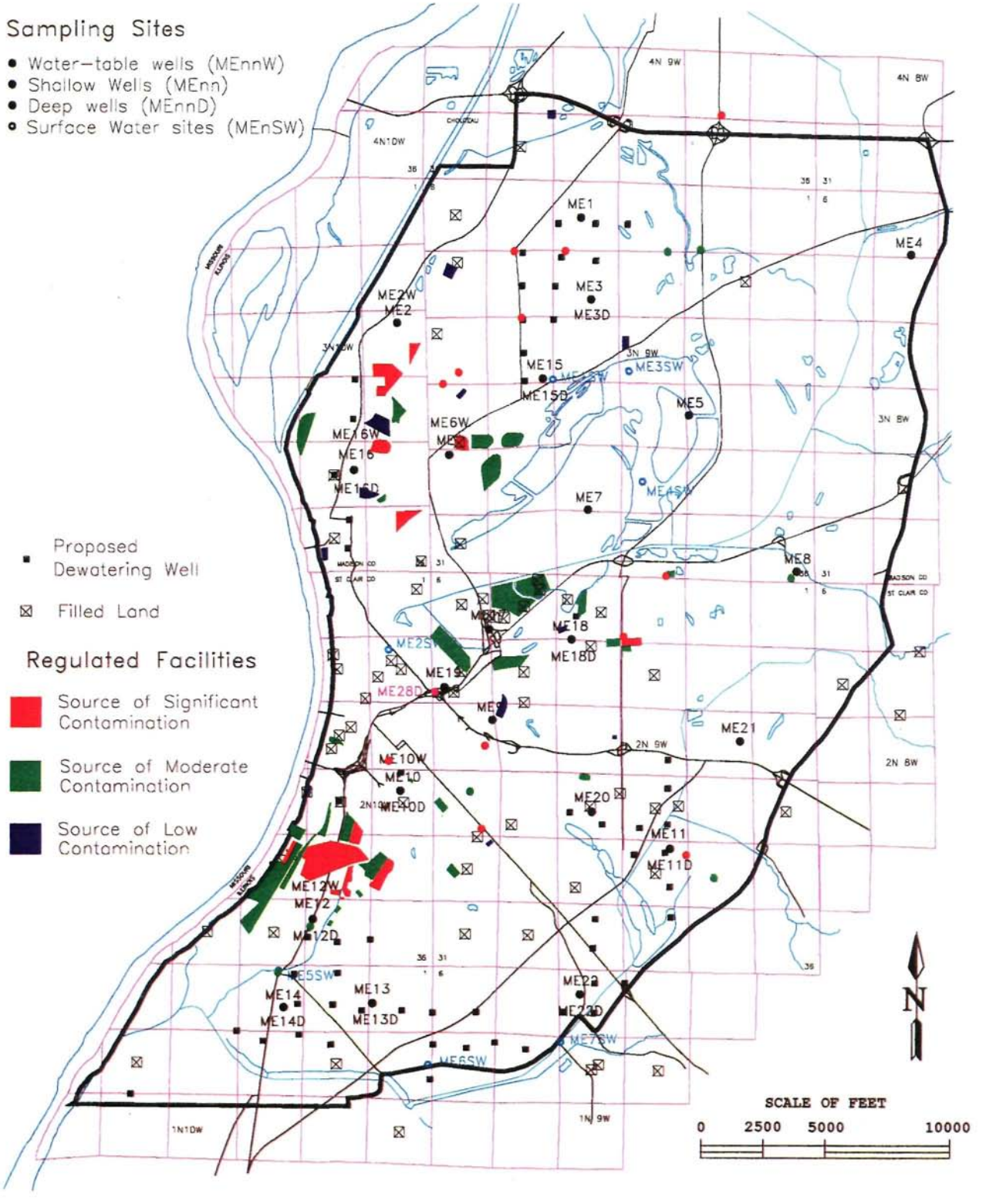


Figure 2.8. Locations of filled land and regulated facilities

Table 2.1. Summary of RCRA, CERCLA, and LUST sites

Site Name	Site Type	Location	Ground-Water Contamination	Potential for Ground-Water Contamination
A.O. Smith	RCRA	MAD T3N R9W Sec 7		low
Aero Aviation	RCRA	MAD T3N R9W Sec 15		low
Affiliated Metals	RCRA	MAD T3N R10W Sec 24		low
Air Products	RCRA	MAD T3N R9W Sec 19		low
Allied Corporation	LUST/ CERCLA	STC T2N R9W Sec 3	significant	
Allwaste Tank Cleaning	RCRA	MAD T4N R9W Sec 32		low
Amax (Big River) Zinc	CERCLA	STC T2N R10W Sec 23		moderate
American Steel Foundaries	CERCLA	MAD T3N R10W Sec 13,24		high
Amoco Oil Co	LUST	STC T2N R10W Sec 35		moderate
Amoco Oil	LUST	MAD T3N R9W Sec 9		high
Amoco Oil Co	LUST	MAD T4N R9W Sec 26		high
Bi-State Development	LUST	STC T2N R10W Sec 13	significant	
Butler	CERCLA	STC T2N R9W Sec 16		low
C and S Auto Plaza	LUST	STC T2N R9W Sec 19	significant	
Cahokia Power Plant	CERCLA	STC T2N R10W Sec 22		moderate
Central States Battery	CERCLA	MAD T3N R9W Sec 34	observed	
Cermak's Auto Repair	LUST	STC T2N R9W Sec 26		moderate

Table 2.1. Continued

Site Name	Site Type	Location	Ground-Water Contamination	Potential for Ground-Water Contamination
City of Granite City	LUST	MAD T3N R9W Sec 18	significant	
Clayton Chemical	CERCLA	STC T2N R10W Sec 27		high
Conrail Train Yard	CERCLA	STC T2N R9W Sec 10		moderate
Dow and Consolidated Alluminum	CERCLA	MAD T3N R10W Sec 25	significant	
East St Louis Housing Auth.	LUST	STC T2N R9W Sec 18	significant	
Ethyl Corporation	CERCLA	STC T2N R10W Sec 23,26		high
GMI Bliss	CERCLA	MAD T3N R10W Sec 25		low
Gateway Petroleum	RCRA	STC T2N R9W Sec 30		low
Granite City Steel	RCRA	MAD T3N R9W Sec 19,29,30		moderate
H.H. Hall Construction	CERCLA	STC T2N R10W Sec 35	observed	
Harcros	RCRA	STC T2N R9W Sec 8,17		low
Jennison Wright	CERCLA	MAD T3N R10W Sec 13	significant	
Jul Fisher Distributors	LUST	STC T2N R9W Sec 19		moderate
Keller Oil - Pontoon Shell	LUST	MAD T3N R9W Sec 2		moderate
Kerr-McGee	CERCLA	MAD T3N R10W Sec 36	significant	
Lanson Chemical	CERCLA	STC T2N R9W Sec 30	observed	

Table 2.1. Continued

Site Name	Site Type	Location	Ground-Water Contamination	Potential for Ground-Water Contamination
Lemear	CERCLA	STC T2N R9W Sec 4		moderate
Lewis Brothers Bakery	LUST	STC T3N R9W Sec 34	significant	
Metro Disposal	CERCLA	STC T2N R9W Sec 8	observed	
Midwest Petroleum	LUST	MAD T3N R9W Sec 36		moderate
Midwestern Drum	RCRA	MAD T3N R10W Sec 25		moderate
Mildred Hediger	LUST	MAD T3N R9W Sec 10		moderate
Mobil Oil	LUST	STC T1N R10W Sec 3		moderate
Mobil Oil	LUST	MAD T3N R9W Sec 17	significant	
Mobil Oil Corp	LUST	MAD T3N R9W Sec 8	significant	
Mobile Oil Terminal	CERCLA	STC T2N R10W Sec 25		moderate
Monsanto Krummrich	CERCLA	STC T2N R10W Sec 26	significant	
Morris Paints	CERCLA	STC T2N R10W Sec 24		moderate
Moss American	CERCLA	STC T2N R10W Sec 25	significant	
Quick Trip	LUST	MAD T3N R9W Sec 18	significant	
Reilly Tar	RCRA/ CERCLA	MAD T3N R9W Sec 19	significant	
Sauget site G	CERCLA	STC T2N R10W Sec 26	significant	
Sauget site H	CERCLA	STC T2N R10W Sec 26	significant	

Table 2.1. Continued

Site Name	Site Type	Location	Ground-Water Contamination	Potential for Ground-Water Contamination
Sauget site I	CERCLA	STC T2N R10W Sec 26	significant	
Sauget site J	CERCLA	STC T2N R10W Sec 26		moderate
Sauget site K	CERCLA	STC T2N R10W Sec 26		moderate
Sauget site N	CERCLA	STC T2N R10W Sec 35		moderate
Sauget site O	CERCLA	STC T2N R10W Sec 26	significant	
Sauget site P	CERCLA	STC T2N R10W Sec 23		moderate
Sauget site Q	CERCLA	STC T2N R10W Sec 27,34	observed	
Sauget site R	CERCLA	STC T2N R10W Sec 27	significant	
SCA Milam Landfill	RCRA/ CERCLA	STC T2N R9W Sec 5	observed	
Shipper Carline	CERCLA	STC T2N R10W Sec 14		moderate
Southern Railroad	CERCLA	STC T2N R10W Sec 22		low
St. Louis Auto Shredding	CERCLA	STC T2N R9W Sec 7	observed	
Star Service	LUST	STC T2N R9W Sec 27		high
Swift Agricultural Chemical	CERCLA	STC T2N R9W Sec 4,9		moderate
Taracorp	CERCLA	MAD T3N R10W Sec 24		moderate
Thomas Garage	CERCLA	STC T2N R9W Sec 4		low
Trade Waste Incinerator	RCRA	STC T2N R10W Sec 26		high

Table 2.1. Concluded

Site Name	Site Type	Location	Ground-Water Contamination	Potential for Ground-Water Contamination
Union Electric	RCRA	MAD T3N R10W Sec 35		low
U. S. Army	CERCLA	MAD T3N R10W Sec 22,23	observed	
U. S. Small Business Administration	RCRA	STC T2N R9W Sec 21		moderate
Waggoner Trucking	CERCLA	STC T2N R10W Sec 26	significant	
Wastex	CERCLA	STC T2N R9W Sec 19		low

Column 5: Potential for Ground-Water Contamination

high: These are sites for which the nature of activity and the hazardous nature of contaminants are such that the likelihood of ground-water contamination by hazardous substances is high. It is expected that if ground-water samples were collected, the concentration of one or more contaminant would exceed applicable standards by a factor of ten.

moderate: Sites for which the nature of the activity is such that contamination is likely, but that the volume of contaminant may be small. When sampled, the ground water is expected to have some contamination.

low: Activity is such that contamination is possible, but would be limited to relatively low concentrations and of a limited volume. Many are RCRA sites where no known major releases of hazardous substances is recorded.

For presentation in figure 2.8, the regulated facilities were grouped into three categories based on their ranking: significant, moderate, or low contamination. The red areas have existing significant contamination or have a high potential for ground-water contamination. The green areas have observed ground-water contamination or have moderate potential for ground-water contamination. The blue areas have a low potential for ground-water contamination.

Observed Contamination. The red or green polygonal regions in figure 2.8 are EPA-regulated facility sites where ground-water contamination was observed, or is expected. The polygons represent the regions of observed contaminant plumes, or the facility boundaries if maps of the plumes were not available. The occurrence of contamination at these locations is considered reliable information because of the strict sampling and testing protocols required by EPA. The specific contaminants are not presented for each site, but at many of the sites the aquifer is contaminated with a variety of organic compounds, including, but not limited to, benzene, xylenes, naphthalene, toluene, trichloroethene, phenols, and arochlors. Some metals, notably lead, arsenic, zinc, copper, chromium, mercury, and nickel, were also detected at some of these sites. Some of these sites are heavily contaminated while others have a few compounds present at concentrations only slightly above the detection limit. These sites are locations where some level of ground-water contamination has been confirmed and there is no need to install additional wells to confirm the EPA data. The proposed locations of the US ACE ground-water withdrawal wells are also shown on figure 2.8 to show the proximity of some of the withdrawal wells to observed or potential ground-water contamination.

Other sources of data about existing contamination include Rehfeldt et al. (1991) and Voelker (1984). Both of these reports present the chemical analyses of ground-water samples taken from existing private wells in the region. As such, there is some concern that the samples may not be representative of the ground water because of variation in well construction and other factors such as the casing and plumbing material and the presence of a home water treatment system that might influence trace levels of contaminants. Nonetheless, these data may be useful in detecting trends in the ground-water quality.

Of the chemical analyses reported in Rehfeldt et al. (1991) and Voelker (1984), only four elements (iron, manganese, mercury, and zinc) were observed in concentrations above the Illinois effluent water quality standard (IEPA, 1988). Figures 2.9-2.12 show the locations and concentrations of samples that exceeded the effluent standard for iron, manganese, mercury, and zinc, respectively.

The concentration of iron (figure 2.9) is in excess of the effluent standard of 2 mg/L over nearly all of the study area. Only five wells (figure 2.10) had manganese concentrations in excess of the effluent standard of 1 mg/L, and of those only two are significantly above the standard.

Six wells (figure 2.11) had mercury concentrations in excess of the effluent standard of 0.0005 mg/L. Five of the six wells from the work of Rehfeldt et al. (1991) are questionable because the concentrations are a factor of 10 larger than the equilibrium solubility of 0.025 mg/L of free-metal mercury - the most stable form (Hem, 1985). It is likely that the five high mercury values are an artifact of the analytical technique used by Rehfeldt et al. (1991) to analyze their private well samples.

Seven wells (figure 2.12) had zinc concentrations in excess of the effluent water quality standard. It is notable that the observed concentrations are only slightly above the standard of 1 mg/L.

Based on figures 2.9-2.12, the larger concentrations were observed most often in the center and southern part of the study area. Because of uncertainties associated with sampling private wells, however, it is not possible to definitively state that these analyses indicate contamination. Therefore, some of the monitoring wells will be installed to confirm the higher concentrations reported by Rehfeldt et al. (1991) and Voelker (1984).

Monitoring Well Locations

Using the data compiled in figures 2.1-2.12, the conceptual model of the expected behavior of the ground-water flow system can be formulated. Ground-water flow is generally east to west toward the Mississippi River. A reversal of this trend, however, caused by ground-water pumpage is evident in the west-central portion of the study area. The vast majority of the EPA regulated facilities are located in the west-central portion of the study areas, which is also the area of relatively faster soil infiltration rates. It is expected that contamination is more likely to enter the aquifer in the west-central, urbanized part of the study area and then move in the direction of ground-water flow either to the Mississippi River or to the major pumping centers near the river. Based on this conceptual model of the expected behavior, a likely contaminated region of the aquifer is defined as the area encompasing the majority of contaminant sources and the portion of the aquifer down gradient from those sources (figure 2.13). Several LUST sites are not included in this likely contaminated region because the contaminant plumes from these sites will be small compared with the land area required to expand the likely contaminated region to those sites. However, these LUST sites will be considered in the evaluation of the data.

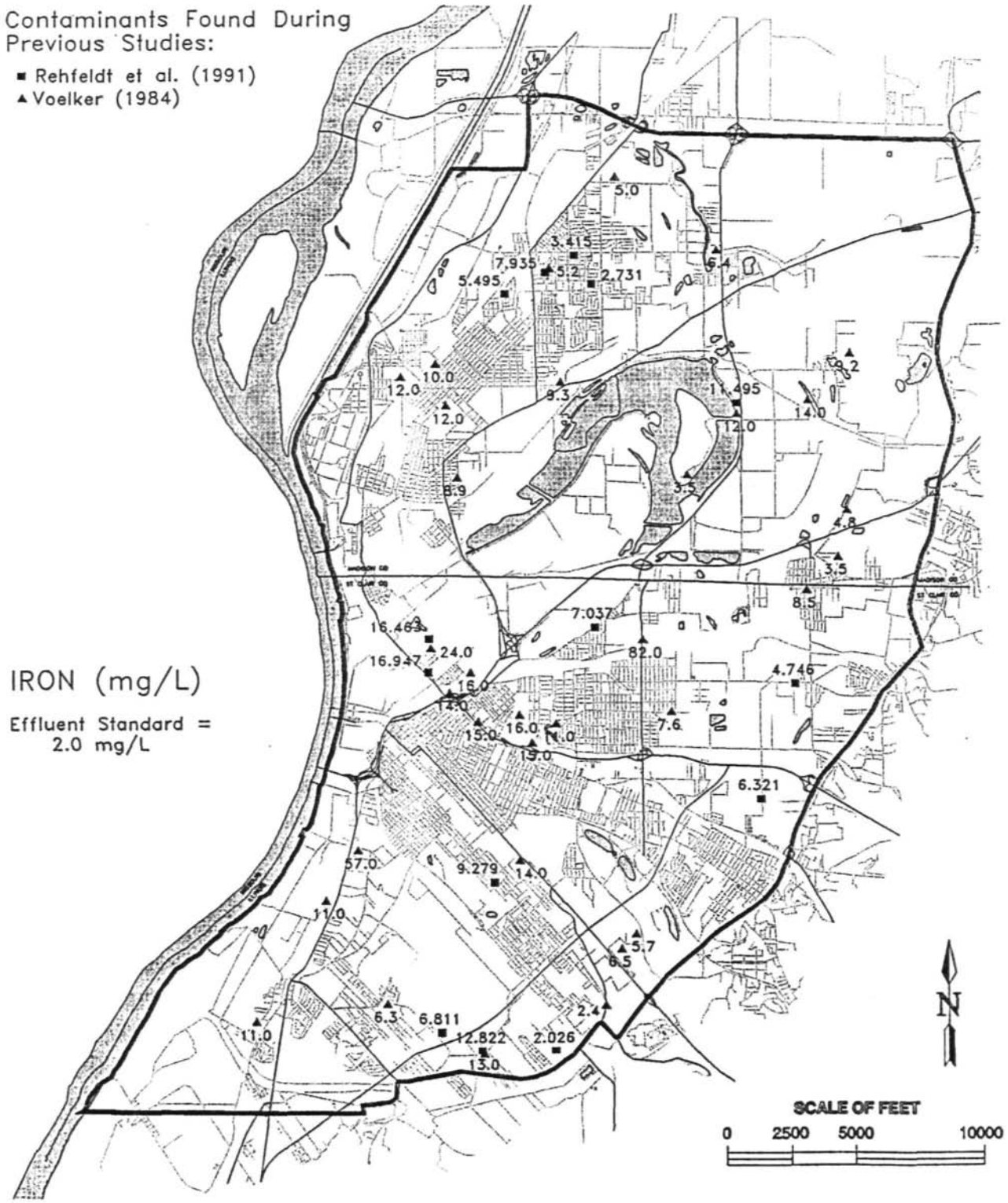


Figure 2.9. Iron concentrations in wells from previous studies

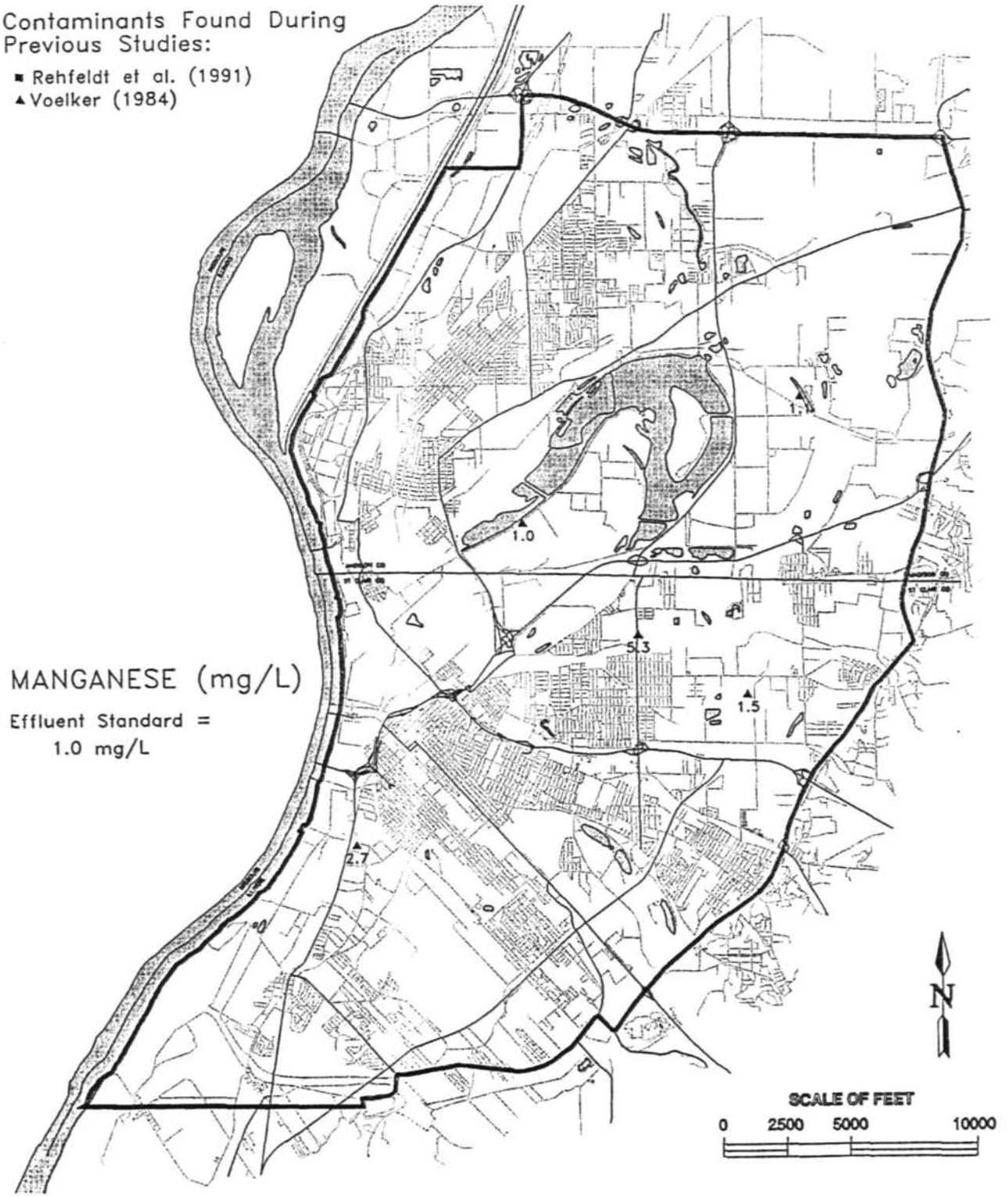


Figure 2.10. Manganese concentrations in wells from previous studies

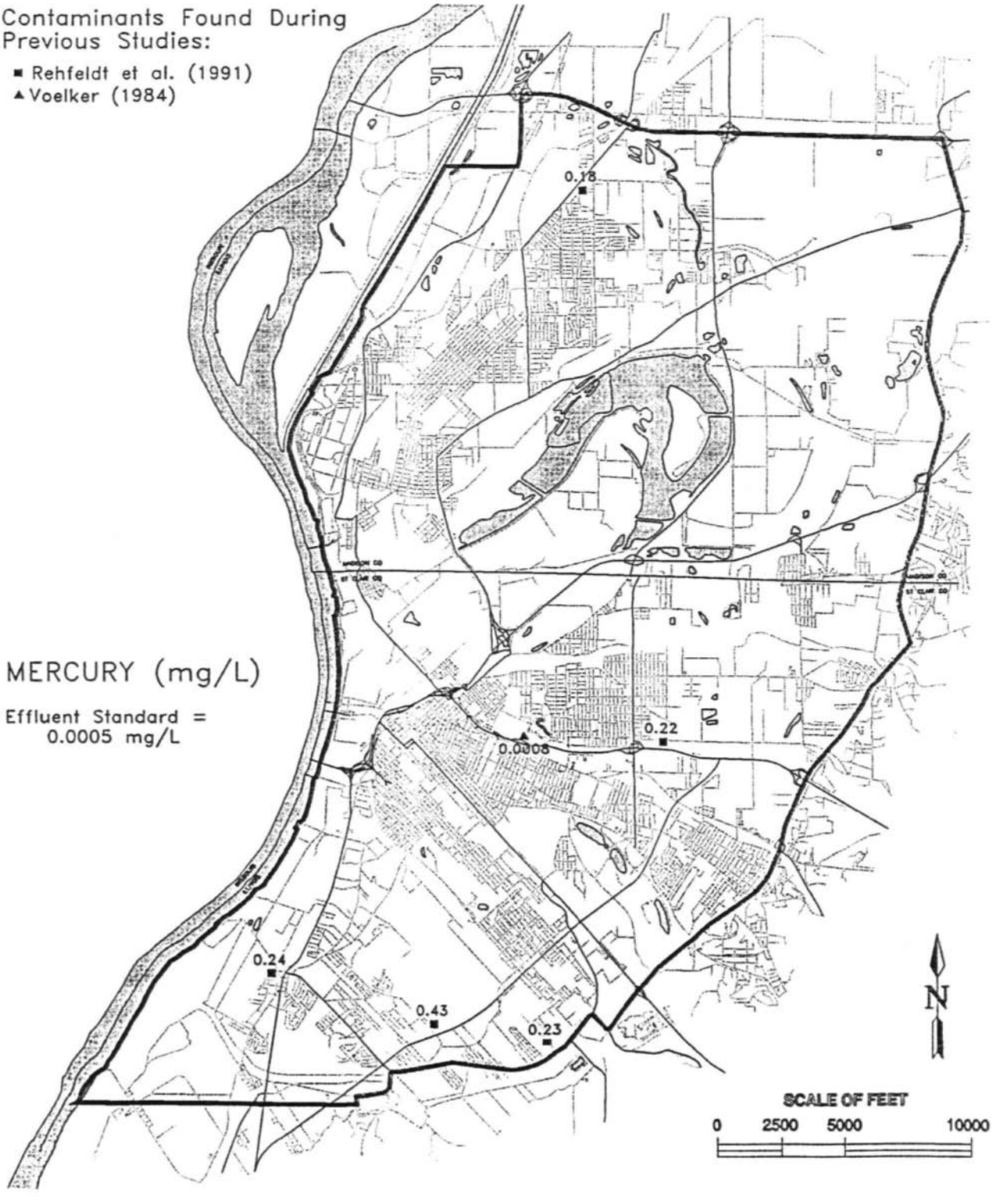


Figure 2.11. Mercury concentrations in wells from previous studies

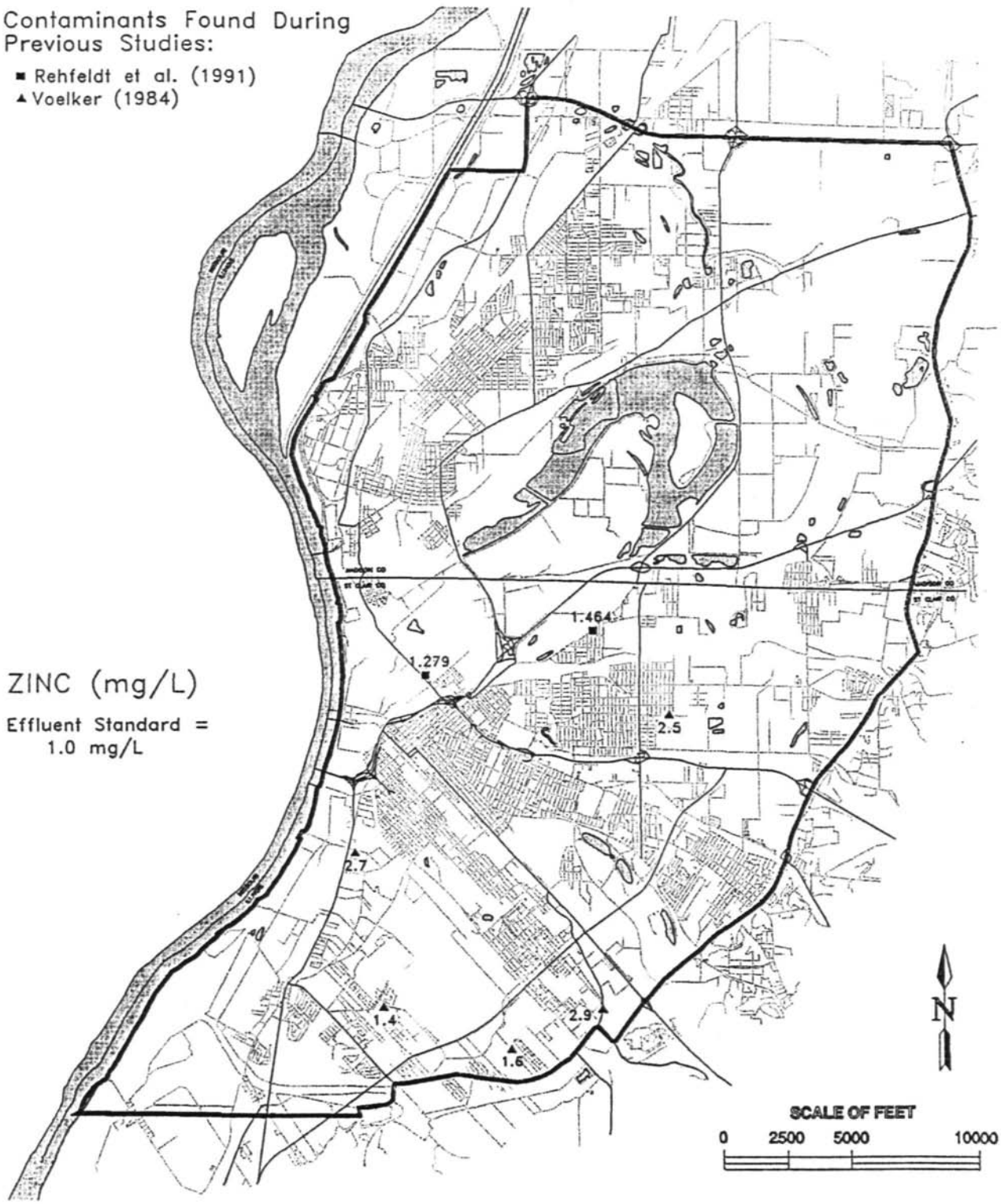


Figure 2.12. Zinc concentrations in wells from previous studies

The locations of the dedicated monitoring wells were chosen to confirm the conceptual model using the following guidelines:

1) Several wells are located down gradient of potential sources of contamination including industrial areas and landfills.

2) Several wells are located up gradient of potential sources. These wells were expected to be in regions of uncontaminated ground water.

3) Several wells are located in or down gradient of agricultural areas.

4) Several wells are located near ground-water contamination identified in previous studies.

5) The deep dedicated monitoring wells installed for this study need to be located near the proposed USACE dewatering wells.

These criteria were used to locate the shallow wells for the shallow ground-water quality assessment of Rehfeldt et al. (1991) and for the deep wells installed for this study. Figure 2.14 is a map of the locations of the monitoring wells installed in the study area and is a useful reference to help locate the wells in figure 2.13. The sampling sites are designated by ME (Metro East) and a location number. Following the number may be a blank, W, D, or SW. The blank and W signify the shallow and water table wells, respectively, of Rehfeldt et al. (1991). The D and SW signify the deep wells and surface water sites, respecitvely, of this study. The wells that satisfy the above criteria are given in table 2.2.

Several wells satisfy more than one criterion. For example, ME3D is up gradient of industrial sources but down gradient of agricultural areas. Therefore, based on the conceptual model, if contamination exists at the ME3D location, it would be expected to be of agricultural origin.

Monitoring Well Installation and Construction. The locations of the dedicated monitoring wells (figure 2.13) were carefully selected according to the conceptual model to best use the resources available for this project. The wells are constructed of corrosion-resistant stainless steel because organic solutes do not readily adsorb to the casing material itself. To avoid the need for drilling mud, the wells were installed with a hollow-stem auger drill rig. Figure 2.15 illustrates a typical well completion. Although no drilling mud was used in drilling the wells, bentonite clay was used to seal the annular space around the wells above the water table. The bentonite clay is not expected to adversely affect the samples that are collected from the dedicated monitoring wells.

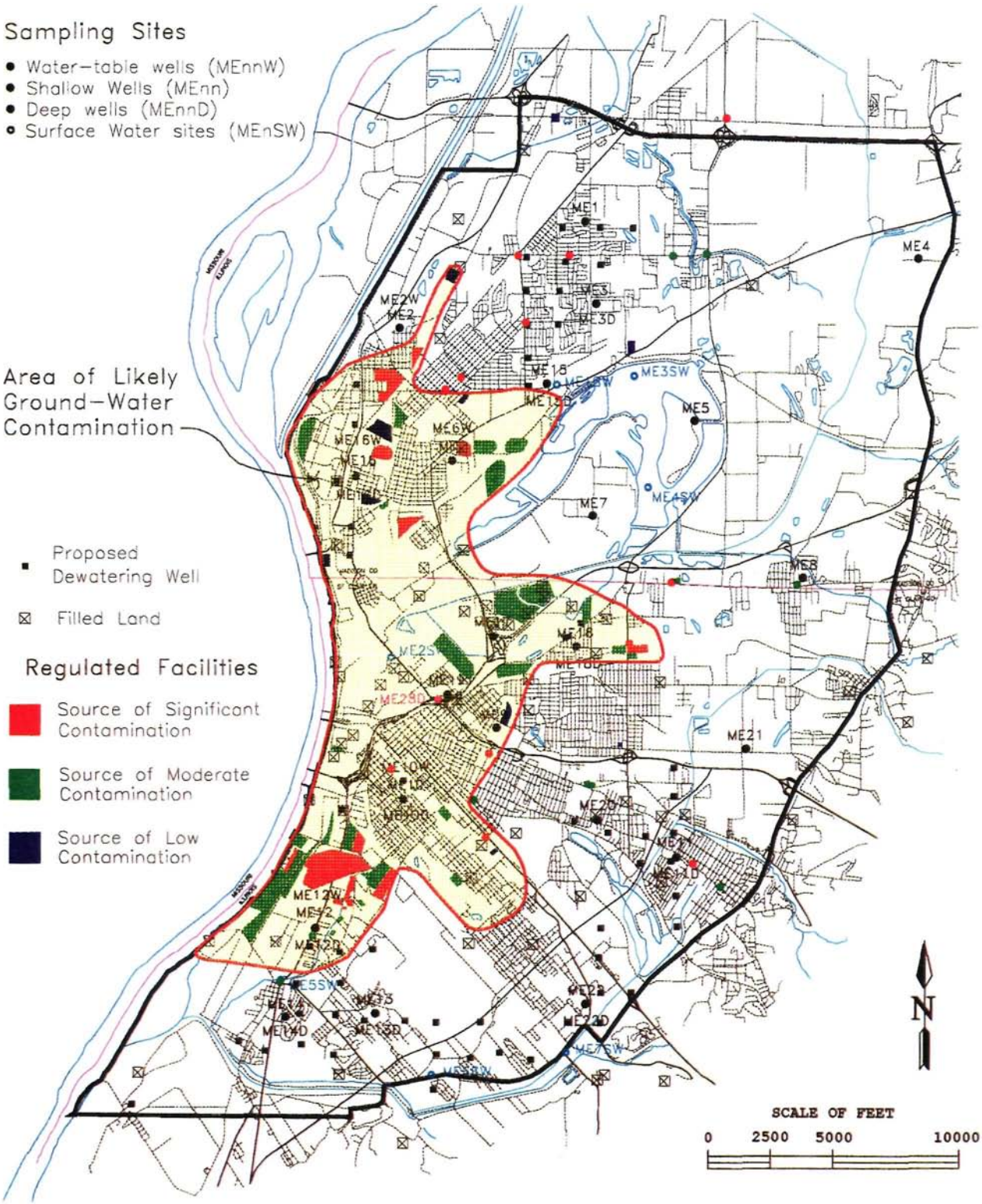


Figure 2.13. Regions of the aquifer where ground-water contamination is likely

Table 2.2

Wells Located with Respect to Conceptual Model Criteria

Criterion	Deep Wells	Shallow Wells
1) down gradient of potential nonagricultural sources	ME10D, ME12D, ME16D, ME18D	ME6, ME6W, ME9, ME10, ME10W, ME12, ME12W, ME16, ME16W, ME17, ME19
2) up gradient of potential sources	ME3D, ME11D, ME13D, ME14D, ME15D, ME22D	ME1, ME2, ME2W, ME3, ME11, ME13, ME14, ME15, ME20, ME22
3) in or down gradient of agricultural areas	ME3D, ME13D, ME14D, ME22D	ME3, ME4, ME5, ME7 ME8, ME13, ME14, ME21, ME22
4) near regions of non-EPA observed contamination	ME13D, ME14D, ME18D, ME22D	ME9, ME13, MEM, ME18, ME22

After the installation of each well, the augers were steam-cleaned on the site to prevent the inadvertent contamination of one site with material from another site. While the augers were being cleaned, compressed air was used to develop the wells to remove suspended solids from the aquifer around the well to produce clean discharge to remove water introduced into the aquifer during drilling.

Each well was installed, constructed, and developed in the same manner. The drill rig was positioned over the well location and the precleaned augers were advanced in 5-foot sections. The material brought to the surface by the augers was described with respect to grain size and color. To install the well, the augers were filled with water and the plug at the end of the lead auger was removed. The water filling the augers was necessary to prevent aquifer material, especially sand, from moving up into the augers after the plug at the bottom of the lead auger was removed. (If sand is not prevented from entering the augers, then installation of the well to the desired depth is impossible.) The well was then lowered into the augers and the bottom of the well was set in the aquifer through the hole at the bottom of the lead auger.

With the well set in the aquifer, the augers were then pulled up, leaving the well in place. Again, water was added to the inside of the augers to prevent sand from entering the bottom of the augers, sand could possibly lock the well inside the augers, which would cause the well to be pulled back out of the ground with the augers. The exact amount of water added to the formation as a result of drilling is not known but is estimated to be no more than 200 gallons based on the total amount of water used in a single day of drilling and the number of wells drilled. Total water used is no more than 300 to 400 gallons per day spread over two

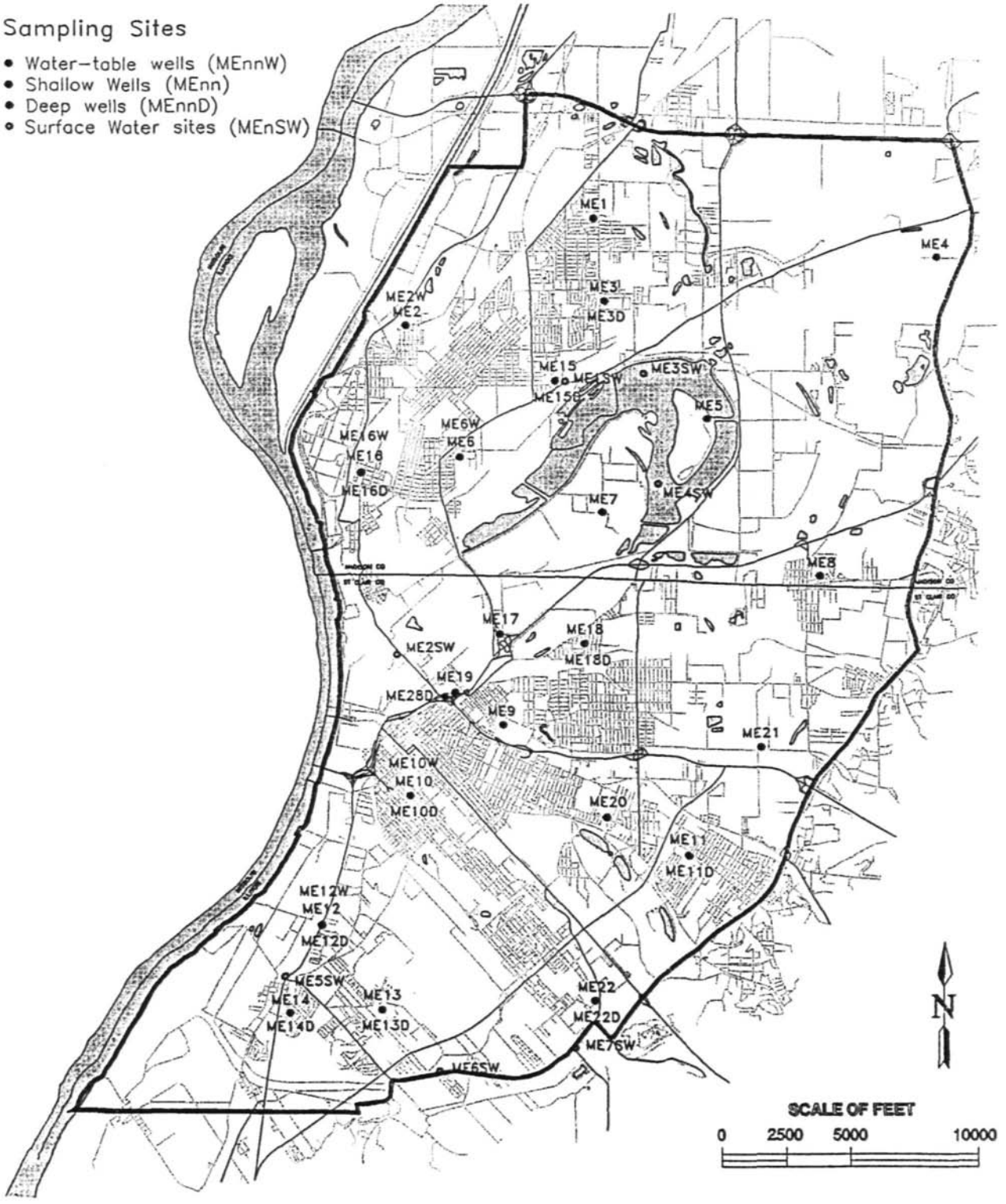


Figure 2.14. Ground-water and surface water sample locations

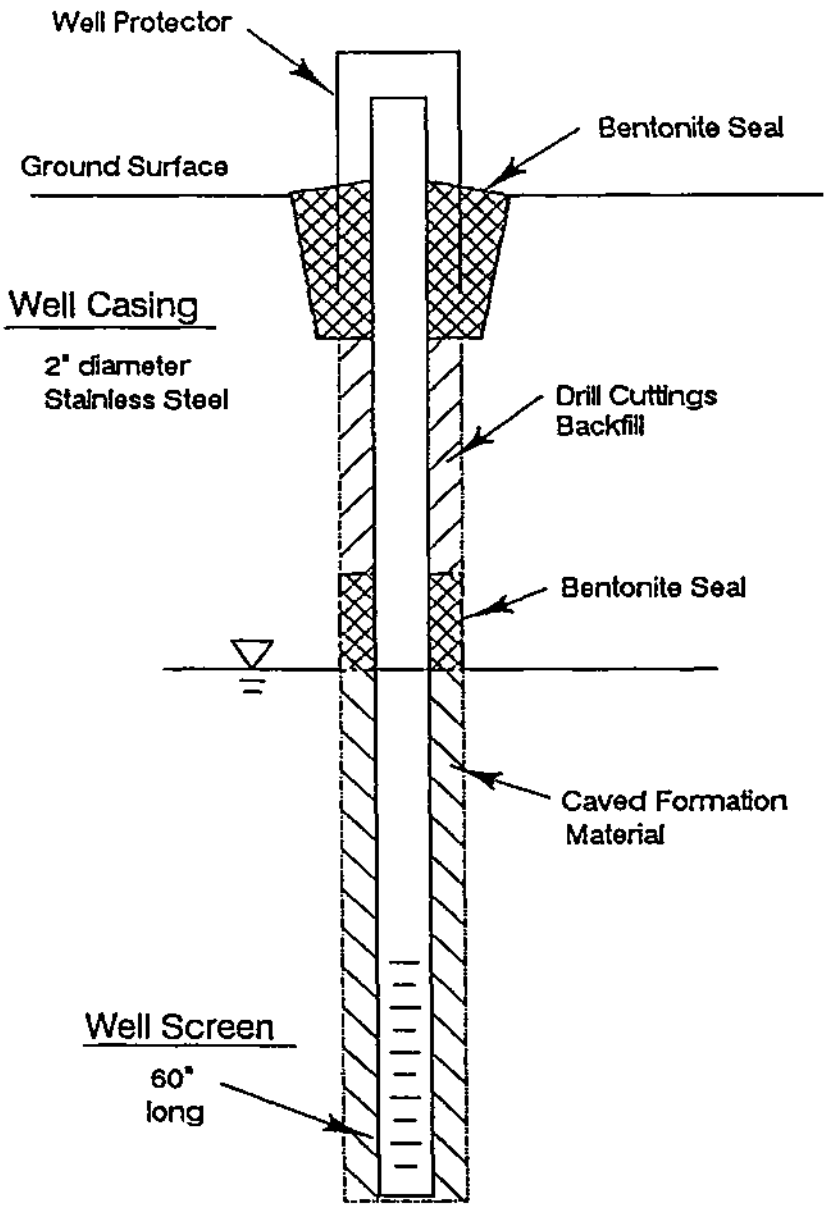


Figure 2.15. Typical well completion

wells. This added water from drilling must be removed prior to obtaining a ground-water sample. Table 2.3 contains the well construction information for each of the shallow and deep wells. Appendix A contains the description of the aquifer material observed during the drilling of each well.

Monitoring Well Development. Two rounds of well development were performed to remove any suspended sediment in the aquifer generated by the drilling process and to remove the water that was added during drilling. The first round of development immediately followed the well installation and consisted of placing a compressed air line at the bottom of the well to blow out the sediment and water. The compressor was run for about 30 minutes during which the water and sediment mixture was continually brought to the surface. In nearly all cases, the discharge was free of sediment at the end of the 30-minute period. Unfortunately, the volume of water removed (~300 gallons) could not be precisely determined because it was literally blown out of the well. The shallow wells designated by W could not be developed with compressed air. Due to their shallow depth, these wells also had the least amount of water added during drilling.

A second round of development was initiated about four weeks after the wells were installed. A centrifugal pump was used for the second round of development. Development was aided by cycling the pump on and off in order to move water through the well screen in both directions. The back and forth movement of the water helps to dislodge small particles that are then removed from the well. The volume of water removed was measured during this second round of well development (table 2.4). The volume of water removed during the second round of development, coupled with the water removed during the first round, is larger than the volume of water added to the well during drilling. Following the second round of well development, the wells were available for sampling.

Ground-Water Sample Collection

In support of this study, two sets of ground-water samples were collected from the ten deep monitoring wells. In addition, seven surface water locations (designated by the letters SW) and one IDOT dewatering well (ME28D) located about 1200 feet southwest of ME19 were also sampled (figure 2.14). The first set of samples was taken in June 1990, and the second set was collected between November 1990 and February 1991, using the ISWS Mobile Laboratory. For each set of samples, sampling protocols developed by the ISWS (Barcelona et al., 1985) were followed. To analyze the reliability of the chemical data, field blanks, duplicates, and spiked samples were also analyzed.

In addition to the deep wells and surface waters sampled for this study, the shallow wells that were part of the shallow ground-water quality study (Rehfeldt et al., 1991) were sampled before and during this study. In particular, the June 1990 sampling of the deep wells for this study was coordinated with the sampling of the shallow wells. The wells sampled and the dates of those samples are summarized below.

Table 2.3. Well Construction Data

Well ID	X COORD (ft)[a]	Y COORD (ft)[a]	MP ELEV[b] (est) (ft MSL)	DEPTH[c] <LSD (ft)	RISER HEIGHT (ft)	CASING LENGTH (ft)	SCREEN LENGTH (ft)
ME1	2828261	2082268	422.3	34.6	2.4	32	5
ME2	2813452	2073841	418.4	35.0	2	32	5
ME2W	2813452	2073841	418.04	20.3	1.7	17	5
ME3	2829130	2075730	422.3	35.1	1.9	32	5
ME3D	2829130	2075730	422.75	84.6	2.4	82	5
ME4	2854799	2079313	432.5	49.8	2.2	47	5
ME5	2836958	2066437	412.6	27.8	2.2	25	5
ME6	2817666	2063242	412.1	45.3	1.7	42	5
ME6W	2817666	2063242	412.24	25.1	1.9	22	5
ME7	2828848	2058843	412.7	40.2	1.8	37	5
ME8	2845659	2053899	421.3	34.5	2.5	32	5
ME9	2821139	2041968	419.3	45.1	1.9	42	5
ME10	2813707	2036268	408.5	45.3	1.7	42	5
ME10W	2813707	2036268	408.69	18.1	1.9	15	5
ME10D	2813707	2036268	408.58	85.1	1.9	82	5
ME11	2835468	2031624	413.2	48.4	1.6	45	5
ME11D	2835468	2031624	413.4	75.2	1.8	72	5
ME12	2806712	2025976	414.6	39.9	2.1	37	5
ME12W	2806712	2025976	415.08	24.5	2.5	22	5
ME12D	2806712	2025976	414.8	82.0	2	79	5
ME13	2811492	2019223	414.4	30.0	2	27	5
ME13D	2811492	2019223	414.4	86.9	2.1	84	5
ME14	2804293	2018922	408.7	44.7	2.3	42	5
ME14D	2804293	2018922	408.77	82.6	2.4	80	5
ME15	2825222	2069368	419.7	34.8	2.2	32	5
ME15D	2825222	2069368	420.44	82.2	2.8	80	5
ME16	2809966	2062023	419.9	45.5	1.5	42	5
ME16W	2809966	2062023	419.77	26.0	1	22	5
ME16D	2809966	2062023	420.91	85.1	1.9	82	5
ME17	2820901	2049231	411.5	30.4	-0.4	25	5
ME18	2827513	2048402	424.5	44.9	2.1	42	5
ME18D	2827513	2048402	424.93	84.7	2.3	82	5
ME19	2817314	2044583	420.7	56.3	0.7	52	5
ME20	2829176	2034564	419.1	44.9	2.1	42	5
ME21	2841089	2040262	424	20.5	1.5	17	5
ME22	2828222	2019931	420.6	39.8	2.2	37	5
ME22D	2828222	2019931	419.67	84.0	1	80	5

NOTES:

[a] Coordinates are in Lambert feet
[b] Measurement point elevation
[c] Well depth below land surface datum

Table 2.3. Concluded

Well ID	CASING DIAMETER (in)	CASING TYPE	USE	LSD[e] (est) (ft MSL)	W.PROT[f] HEIGHT (ft)	dH of[g] nest MP (ft)	LEGAL LOCATION
ME1	2	SS[d]	Sample	419.5	2.8	0	MAD3N9W-4.5e2
ME2	2	SS	Sample	416	2.4	0	MAD3N10W-13.4g1
ME2W	2	SS	Sample	415.94	2.1	-0.36	MAD3N10W-13.4g2
ME3	2	SS	Sample	420	2.3	0	MAD3N9W-9.4c2
ME3D	2	SS	Sample	419.95	2.8	0.45	MAD3N9W-9.4c3
ME4	2	SS	Sample	430	2.5	0	MAD3N8W-5.4a4
ME5	2	SS	Sample	410	2.6	0	MAD3N9W-23.8e2
ME6	2	SS	Sample	410	2.1	0	MAD3N9W-30.5h1
ME6W	2	SS	Sample	409.84	2.4	0.14	MAD3N9W-30.5h2
ME7	2	SS	Sample	410.5	2.2	0	MAD3N9W-28.5a
ME8	2	SS	Sample	418.5	2.8	0	MAD3N9W-36.3b
ME9	2	SS	Sample	417	2.3	0	MAD2N9W-18.1g*
ME10	2	SS	Sample	406.5	2	0	STC2N10W-24.4f1*
ME10W	2	SS	Sample	406.39	2.3	0.19	STC2N10W-24.4f2*
ME10D	2	SS	Sample	406.28	2.3	0.08	STC2N10W-24.4f3*
ME11	2	SS	Sample	411	2.2	0	STC2N9W-27.3g2*
ME11D	2	SS	Sample	411.3	2.1	0.2	STC2N9W-27.3g1*
ME12	2	SS	Sample	412	2.6	0	STC2N10W-35.7f1*
ME12W	2	SS	Sample	412.18	2.9	0.48	STC2N10W-35.7f2*
ME12D	2	SS	Sample	412.4	2.4	0.2	STC2N10W-35.7f3*
ME13	2	SS	Sample	412	2.4	0	STC1N10W-1.8d1*
ME13D	2	SS	Sample	412	2.4	0	STC1N10W-1.8d2*
ME14	2	SS	Sample	406	2.7	0	STClN10W-3.3c1
ME14D	2	SS	Sample	405.87	2.9	0.07	STClN10W-3.3c2
ME15	2	SS	Sample	417	2.7	0	MAD3N9W-17.2a1
ME15D	2	SS	Sample	417.14	3.3	0.74	MAD3N9W-17.2a2
ME16	2	SS	Sample	418	1.9	0	MAD3N10W-26.2e1
ME16W	2	SS	Sample	418.37	1.4	-0.13	MAD3N10W-26.2e2
ME16D	2	SS	Sample	418.51	2.4	1.01	MAD3N10W-26.2e3
ME17	2	SS	Sample	411.5	0	0	MAD2N9W-6.1b
ME18	2	SS	Sample	422	2.5	0	STC2N9W-4.7a1
ME18D	2	SS	Sample	422.23	2.7	0.43	STC2N9W-4.7a2
ME19	2	SS	Sample	419.5	1.2	0	STC2N9W-7.6c*
ME20	2	SS	Sample	416.5	2.6	0	STC2N9W-21.4d*
ME21	2	SS	Sample	422	2	0	STC2N9W-14.2e
ME22	2	SS	Sample	418	2.6	0	STC1N9W-4.6f1*
ME22D	2	SS	Sample	418.17	1.5	-0.93	STC1N9W-4.6f2*

NOTES:

[d] SS = Stainless Steel
[e] Land surface datum elevation
[f] Height of the well protector above LSD
[g] Difference of measurement point elevation for nested wells
* Location extrapolated

Table 2.4

Well Development Data

Well	Flow Rate (gpm)	Duration (minutes)	Volume Pumped (gallons)
1	13.0	41	533
2	10.0	41	410
2W	0.3	90	25
3	13.0	38	494
3D	9.3	63	588
4	4.2	42	176
5	13.0	64	832
6	11.0	33	363
6W	7.5	27	203
7	13.5	26	351
8	11.1	44	488
9	4.8	29	139
10	8.7	23	200
10W	0.2	85	17
10D	9.1	53	484
11	15.0	42	630
11D	15.0	42	630
12	7.9	20	158
12W	1.1	66	70
12D	6.4	32	204
13	9.5	28	266
13D	1.8[1]	139	255
14	9.5	50	475
14D	8.8	53	464
15	6.6	30	196
15D	10.0[2]	30	300
16	6.7	37	248
16W	5.6	26	145
16D	5.4	53	285
17	8.4	36	301
18	0.8	60	48
18D	5.9	79	467
19	2.5	77	193
20	5.3	30	159
21	4.3	40	172
22	7.7	37	286
22D	8.1	54	436

[1] Average of several different flow rates between 0.9 and 3.1 gallons per minute (gpm)

[2] Developed with compressed air only, flow rate is approximate

January 1989:

Samples were collected from the end of the pump discharge line following development of the shallow wells which were installed as part of the study of Rehfeldt et al. 1991. One deep well, ME11D, was also installed at this time. The chemical data from these samples were intended to provide a baseline for comparison with later samples to determine if well development had successfully removed the water added during drilling. Shallow wells ME1 -ME10, and ME12 - ME14, and deep well ME11D were sampled.

February 1989:

Samples were collected following the protocols outlined in the next section. Shallow wells ME1 -ME10 and ME12 - ME14, and deep well ME11D were again sampled.

October 1989:

Prior to this sampling, eight additional shallow wells were installed and sampled along with the previous 14 wells. Up to this time, the work was all part of Rehfeldt et al. 1991.

June 1990:

Prior to this sampling, five wells, designated by a W following the well number, were installed to collect samples from the water table, and one shallow well was installed at site 11 as part of Rehfeldt et al. 1991. The remaining nine deep wells for this study were also installed. All shallow wells and all deep wells were sampled this time. In addition, seven surface water sites and the IDOT dewatering well (ME28D) were sampled. The June 1990 sampling was coordinated so that both projects would benefit from the data from the other.

Winter 1990:

The wells were sampled in stages. In November, the deep wells and the surface water sites were sampled. Due to laboratory error, the total organic halogen (TOX) samples were held past the EPA recommended holding time and had to be discarded. The wells and surface waters were resampled in January 1991, and analyzed for total organic carbon (TOC), TOX, nitrates, and ammonia. Based on the TOC and TOX results, deep wells ME6D, ME10D, ME12D, ME15D, ME16D, ME18D, and ME22D were sampled for complete organic analyses. Instead of sampling the three remaining deep well, nine of the shallow wells (ME6, ME10, ME12, ME15, ME16, ME17, ME18, ME19, and ME22), located at or near the seven deep wells list above, were also sampled to provide a more complete analysis of organic compounds at those locations.

Ground-Water Sampling Protocol. All sampling operations were conducted from a sampling van operated by the ISWS. The van is equipped with a generator, an air compressor to supply air to a positive displacement bladder pump, a refrigerator, and adequate bench space for conducting field analytical procedures.

At each sample location, depth to water surface, and date were recorded. Before sampling, each well was purged to remove stagnant water in the casing and to insure a sample representative of the ground water. Well-purging parameters, such as water temperature, specific conductance, pH, and Eh (volts), were monitored with a flow-through-cell system

(Garske and Schock, 1986). Well purging continued at rates of 1 to 1.5 L/minute until the purging indicator parameters (pH, electrical conductivity, temperature, and Eh) changed less than +0.05 pH units, ± 10 micromhos per centimeter (μ mhos/cm), ± 0.1° C, and ± 10 meter volts (mV), respectively over 0.5 to 1 well volume. Once these parameters had stabilized, their values were recorded and ground-water samples were taken. Samples for TOC and volatile organic compound (VOC) determinations were collected after the pump flow rate was decreased to about 100 milliliters/minute (ml/min).

To expedite the sample handling process, all sample bottles were precleaned at the laboratory before their use at the sites. Preprinted labels were supplied and included information specific to the project: ID #, chemical parameter, preservative, and storage and handling precautions. The glass bottles that were used for organic samples were supplied by Daily Analytical Laboratories of Peoria. Acid-cleaned polyethylene bottles were used for trace metal and inorganic species. All samples were refrigerated until they were returned to the laboratory.

Laboratory Determinations. The list of analytes for the January 1989, February 1989, and October 1989, sampling of the shallow wells is given in Table 2.5. Also included is the method of analysis, estimated detection limits and water supply, and general use and effluent water quality standards. The Illinois water quality standards are from the IEPA (1988). All determinations in table 2.5 were performed at ISWS laboratories.

For two of the analytes (cadmium and copper) the detection limit is above one of the water quality standards. For these analytes, a concentration below the detection limit does not indicate that the sample meets the respective water quality standards with respect to those elements. For all elements, the detection limit is below the Illinois effluent water quality standard.

For the June and Winter 1990 samples, the list of analytes was expanded to include more metals and more organic compounds. Table 2.6 lists the analytes for the June 1990 shallow wells and includes the method of analysis, estimated detection limits and standards for Illinois water supply, general use, and effluent water quality. Table 2.7 contains the same information for the June 1990 deep well and surface water samples and the Winter 1990 samples. The method of analysis for some of the constituents differs for the June 1990 shallow and deep well samples because the shallow well samples were part of an Illinois Department of Energy and Natural Resources-sponsored study. For the June and Winter 1990 samples, all analyses except for organic compounds were performed at the ISWS. The organic analyses were performed by Daily Analytical Laboratories, Peoria, Illinois. Table 2.8 is a list of the compound and detection limits for the June and Winter 1990 analyses for organic compounds.

RESULTS AND DISCUSSION

The results of the chemical sampling are presented in a series of tables covering the different sampling episodes and wells. As noted earlier, a concurrent study (Rehfeldt et al., 1991) of shallow ground-water quality in the American Bottoms was ongoing concurrent with this study. Chemical data from that study will also be presented in this report as it impacts on

Table 2.5

Laboratory Determinations
(Sample dates: January 1989, February 1989, and October 1989)

Parameter	Detection Limit (mg/L)	Illinois Water Supply Standard (mg/L)	Illinois General Use Standard (mg/L)	Illinois Effluent Standard (mg/L)	Method
Calcium	0.22				6
Magnesium	0.9				6
Sodium	1.0				6
Sulfate	5.0	250.0	500.0		4
Chloride	0.5	250.0	500.0		5
Potassium	0.1				6
Iron (dissolved)	0.05		1.0[a]	2.0[a]	6
Ammonium	0.012			4.0 as N	1
Nitrate-nitrogen	0.05	10.0	10.0		2
Phosphate	0.01			1.0 as P	3
Cadmium (dissolved)	0.05	0.01[a]	0.05a[a]	0.15	6
Chromium (dissolved)	0.05	0.05[a]		1.0[a]	6
Copper (dissolved)	0.05		0.02[a]	0.5	6
Nickel (dissolved)	0.05		1.0[a]	1.0	6
Lead (dissolved)	0.001	0.05[a]	0.1[a]	0.2	7
Total Organic Carbon	0.111				8
Chloroform	0.00182				9
Benzene	0.00182	0.005[b]			9
Toluene	0.00182				9
Trichloroethylene	0.00182	0.005[b]			9
Tetrachloroethylene	0.00182				9
Base/neutral extractable	1.0				10
Acid extractable	1.0				10

Methods

1	Standard Method 417C
2	Standard Method 418D
3	Standard Method 424F
4	USGS Method 00945
5	Potentiometric Titration
6	Flame atomic adsorption
7	Graphite furnace atomic adsorption
8	Standard Method 505B
9	Standard Method 505B, headspace gas chromatograph
10	USEPA method 625, gas chromatograph/mass spectrometer

Notes:

a	Federal Drinking Water Standard (40 CFR 141)
b	Water Quality Standard is for total concentration

Table 2.6

Laboratory Determinations
(Sample date: June 1990: shallow wells)

Parameter	Detection Limit (mg/L)	Illinois Water Supply Standard (mg/L)	Illinois General Use Standard (mg/L)	Illinois Effluent Standard (mg/L)	Method
Calcium	0.01				6
Magnesium	0.01				6
Sodium	0.03				6
Sulfate	5.0	250.0	500.0		4
Chloride	0.5	250.0	500.0		5
Potassium	1.33				6
Iron (dissolved)	0.015		1.0*	2.0*	6
Ammonium	0.012			4.0 as N	1
Nitrate-nitrogen	0.05	10.0	10.0		2
Phosphate	0.01			1.0 as P	3
Silver (dissolved)	0.01		0.005	0.1	6
Aluminum (dissolved)	0.027				6
Arsenic (dissolved)	0.05	0.05*	1.0*	0.25	6
Boron (dissolved)	0.05		1.0*		6
Barium (dissolved)	0.001	1.0*	5.0*	2.0	6
Beryllium (dissolved)	0.005				6
Cadmium (dissolved)	0.005	0.01*	0.05*	0.15	6
Chromium (dissolved)	0.01	0.05*		1.0*	6
Copper (dissolved)	0.009		0.02*	0.5	6
Manganese (dissolved)	0.005	0.15*	1.0*	1.0	6
Molybdenum (dissolved)	0.014				6
Nickel (dissolved)	0.017		1.0*	1.0	6
Phosphorus (dissolved)	0.17			1.0	6
Lead (dissolved)	0.033	0.05*	0.1*	0.2	6
Sulfur (dissolved)	0.06				6
Antimony (dissolved)	0.16				6
Selenium (dissolved)	0.6	0.01*	1.0*		6
Silicon (dissolved)	0.16				6
Strontium (dissolved)	0.001				6
Thallium (dissolved)	0.15				6
Vanadium (dissolved)	0.013				6
Zinc (dissolved)	0.008		1.0*	1.0	6
Total Organic Carbon	0.5				7
Volatile Organic Compounds		see Table 2.8			8
Base/neutral extractable		see Table 28			9
Acid extractable		see Table 2.8			9
Pesticides/PCBs		see Table 2.8			10

Methods

1	Standard Method 417C	6	Inductively Coupled Argon Plasma Emission Spectrometry
2	Standard Method 418D	7	USEPA Method 415.1
3	Standard Method 424F	8	Federal Register Method 624
4	USGS Method 00945	9	Federal Register Method 625
5	Potentiometric Titration	10	Federal Register Method 608

Notes: * Water Quality Standard is for total concentration

Table 2.7

Laboratory Determinations
(Sample date: June 1990: deep wells; Winter 1990: all wells)

Parameter	Detection Limit (mg/L)	Illinois Water Supply Standard (mg/L)	Illinois General Use Standard (mg/L)	Illinois Effluent Standard (mg/L)	Method
Calcium	0.01				6
Magnesium	0.01				6
Sodium	0.03				6
Sulfate	5.0	250.0	500.0		1
Chloride	0.5	250.0	500.0		1
Dissolved Solids	1.0	500.0	1000.0		2
Potassium	1.33				6
Iron (total)	0.015		1.0	2.0	6
Ammonium	0.012			4.0 as N	4
Nitrate-nitrogen	0.05	10.0	10.0		1
Phosphate	0.01			1.0 as P	1
Suspended Solids	0.2			15.0	3
Silver (total)	0.01		0.005	0.1*	6
Aluminum (total)	0.027				6
Arsenic (total)	0.05	0.05	1.0	0.25*	6
Boron (total)	0.05		1.0		6
Barium (total)	0.001	1.0	5.0	2.0*	6
Beryllium (total)	0.005				6
Cadmium (total)	0.005	0.01	0.05	0.15*	6
Chromium (total)	0.01	0.05		1.0	6
Copper (total)	0.009		0.02	0.5*	6
Manganese (total)	0.005	0.15	1.0	1.0*	6
Mercury (total)	0.00013		0.0005	0.0005*	7
Molybdenum (total)	0.014				6
Nickel (total)	0.017		1.0	1.0*	6
Phosphorus (total)	0.17			1.0*	6
Lead (total)	0.033	0.05	0.1	0.2*	6
Sulfur (total)	0.06				6
Antimony (total)	0.16				6
Selenium (total)	0.6	0.01	1.0		6
Silicon (total)	0.16				6
Strontium (total)	0.001				6
Thallium (total)	0.15				6
Vanadium (total)	0.013				6
Zinc (total)	0.008		1.0	1.0	6
Total Organic Carbon	0.5				5
Volatile Organic Compounds		see Table 2.8			8
Base/neutral extractable		see Table 2.8			9
Acid extractable		see Table 2.8			9
Pesticides/PCBs		see Table 2.8			10

Methods

1	USEPA Method 300.0	6	Inductively Coupled Argon Plasma Emission Spectrometry
2	USEPA Method 160.1	7	Graphite furnace atomic adsorption
3	USEPA Method 160.2	8	Federal Register Method 624
4	USEPA Method 350.1	9	Federal Register Method 625
5	USEPA Method 415.1	10	Federal Register Method 608

Note: * Water Quality Standard is for dissolved concentration

Table 2.8. 1990 Organic Compound Chemical Parameters

Compound	*Detection Limit (mg/L)*	*Illinois Water Supply Standard (mg/L)*	*Illinois General Use Standard (mg/L)*	*Illinois Effluent Standard (mg/L)*
Volatile Organic Analysis				
Chloromethane	0.01			
Bromomethane	0.01			
Vinyl Chloride	0.01	0.002*		
Chloroethane	0.01			
Methylene Chloride	0.005			
Acetone	0.01			
Carbon Disulfide	0.005			
1,1-Dichloroethene	0.005	0.007*		
1,1-Dichloroethane	0.005			
trans-l,2-Dichloroethene	0.005			
cis-l,2-Dichloroethene	0.005			
1,2-Dichloropropane	0.005			
Chloroform	0.005			
1,2-Dichloroethane	0.005	0.005*		
2-Butanone	0.01			
1,1,1-Trichloroethane	0.005	0.2*		
Carbon Tetrachloride	0.005	0.005*		
Vinyl Acetate	0.01			
Bromodichloromethane	0.005			
cis-1,3-Dichloropropene	0.005			
Trichloroethene	0.005	0.005*		
Benzene	0.005	0.005*		
Dibromochloromethane	0.005			
1,1,2-Trichloroethane	0.005			
trans-l,3-Dichloropropene	0.005			
2-Chloroethylvinyl ether	0.005			
Bromoform	0.005			
2-Hexanone	0.01			
4-Methyl-2-Pentanone	0.01			
1,1,2,2-Tetrachloroethane	0.005			
Tetrachloroethene	0.005			
Toluene	0.005			
Chlorobenzene	0.005			
Ethylbenzene	0.005			
Styrene	0.005			
Total Xylenes	0.015			

*Federal Drinking Water Standard (40 CFR 141)

Table 2.8. Continued

Compound	*Detection Limit (mg/L)*	*Illinois Water Supply Standard (mg/L)*	*Illinois General Use Standard (mg/L)*	*Illinois Effluent Standard (mg/L)*
Pesticide/PCB Analysis				
alpha-BHC	0.00005			
beta-BHC	0.00005			
delta-BHC	0.00005			
gamma-BHC (Lindane)	0.00005	0.004	0.004	
Heptachlor	0.00005	0.0001	0.0001	
Aldrin	0.00005	0.001	0.001	
Heptachlor epoxide	0.00005	0.0001	0.0001	
Endosulfan I	0.00005			
4,4'-DDE	0.0001			
Dieldrin	0.0001	0.001	0.001	
Endrin	0.0001	0.0002	0.0002	
4,4'-DDD	0.0001			
Endosulfan II	0.0001			
4,4'-DDT	0.0001	0.05	0.05	
Endrin aldehyde	0.0001			
Endosulfan sulfate	0.0001			
Methoxychlor	0.0005	0.1	0.1	
Chlordane	0.0005	0.003	0.003	
Toxaphene	0.0005	0.005	0.005	
Aroclor-1016	0.001			
Aroclor-1221	0.0005			
Aroclor-1232	0.0005			
Aroclor-1242	0.0005			
Aroclor-1248	0.0005			
Aroclor-1254	0.001			
Aroclor-1260	0.001			
Semivolatile Organic Analysis				
Acid Target Compounds				
Benzoic acid	0.05			
Phenol	0.01			
2-Chlorophenol	0.01			
2-Nitrophenol	0.05			
2-Methylphenol	0.01			

Table 2.8. Continued

Compound	*Detection Limit (mg/L)*	*Illinois Water Supply Standard (mg/L)*	*Illinois General Use Standard (mg/L)*	*Illinois Effluent Standard (mg/L)*
Acid Target Compounds ('Cont'd')				
2,4-Dimethylphenol	0.0			
14-Methylphenol	0.01			
2,4-Dichlorophenol	0.01			
2,4,6-Trichlorophenol	0.01			
2,4,5-Trichlorophenol	0.05			
4-Chloro-3-methylphenol	0.01			
2,4-Dinitrophenol	0.05			
2-Methyl-4,6-dinitrophenol	0.05			
Pentachlorophenol	0.05			
4-Nitrophenol	0.05			
Base/Neutral Target Compounds				
Hexachloroethane	0.01			
bis(2-Chloroethyl)ether	0.01			
Benzyl alcohol	0.01			
bis(2-Chloroisopropyl)ether	0.01			
N-Nitrosodipropylamine	0.01			
Nitrobenzene	0.01			
Hexachlorobutadiene	0.01			
2-Methylnaphthalene	0.01			
1,2,4-Trichlorobenzene	0.01			
Isophorone	0.01			
Naphthalene	0.01			
4-Chloroaniline	0.01			
bis(2-Chloroethoxy)methane	0.01			
Hexachlorocyclopentadiene	0.01			
2-Chloronaphthalene	0.01			
2-Nitroaniline	0.05			
Acenaphthylene	0.01			
3-Nitroaniline	0.05			
Acenaphthene	0.01			
Dibenzofuran	0.01			
Dimethylphthalate	0.01			
2,6-Dinitrotoluene	0.01			
Fluorene	0.01			
4-Nitroaniline	0.05			
4-Chlorophenyl-phenylether	0.01			

Table 2.8. Concluded

Compound	*Detection Limit (mg/L)*	*Illinois Water Supply Standard (mg/L)*	*Illinois General Use Standard (mg/L)*	*Illinois Effluent Standard (mg/L)*
Base/Neutral Target Compounds (Cont'd)				
2,4-Dinitrotoluene	0.01			
Diethylphthalate	0.01			
N-Nitrosodiphenylamine	0.01			
Hexachlorobenzene	0.01			
Phenanthrene	0.01			
4-Bromophenyl-phenylether	0.01			
Anthracene	0.01			
Di-n-butylphthalate	0.01			
Fluoranthene	0.01			
Pyrene	0.01			
Butylbenzylphthalate	0.01			
bis(2-Ethylhexyl)phthalate	0.01			
Chrysene	0.01			
Benzo(a)anthracene	0.01			
3,3'-Dichlorobenzidine	0.02			
Di-n-octylphthalate	0.01			
Benzo(b)fluoranthene	0.01			
Benzo(k)fluoranthene	0.01			
Benzo(a)pyrene	0.01			
Indeno(l,2,3-cd)pyrene	0.01			
Dibenz(a,h)anthracene	0.01			
Benzo(g,h,i)perylene	0.01			
1,2-Dichlorobenzene	0.01			
1,3-Dichlorobenzene	0.01			
1,4-Dichlorobenzene	0.01			

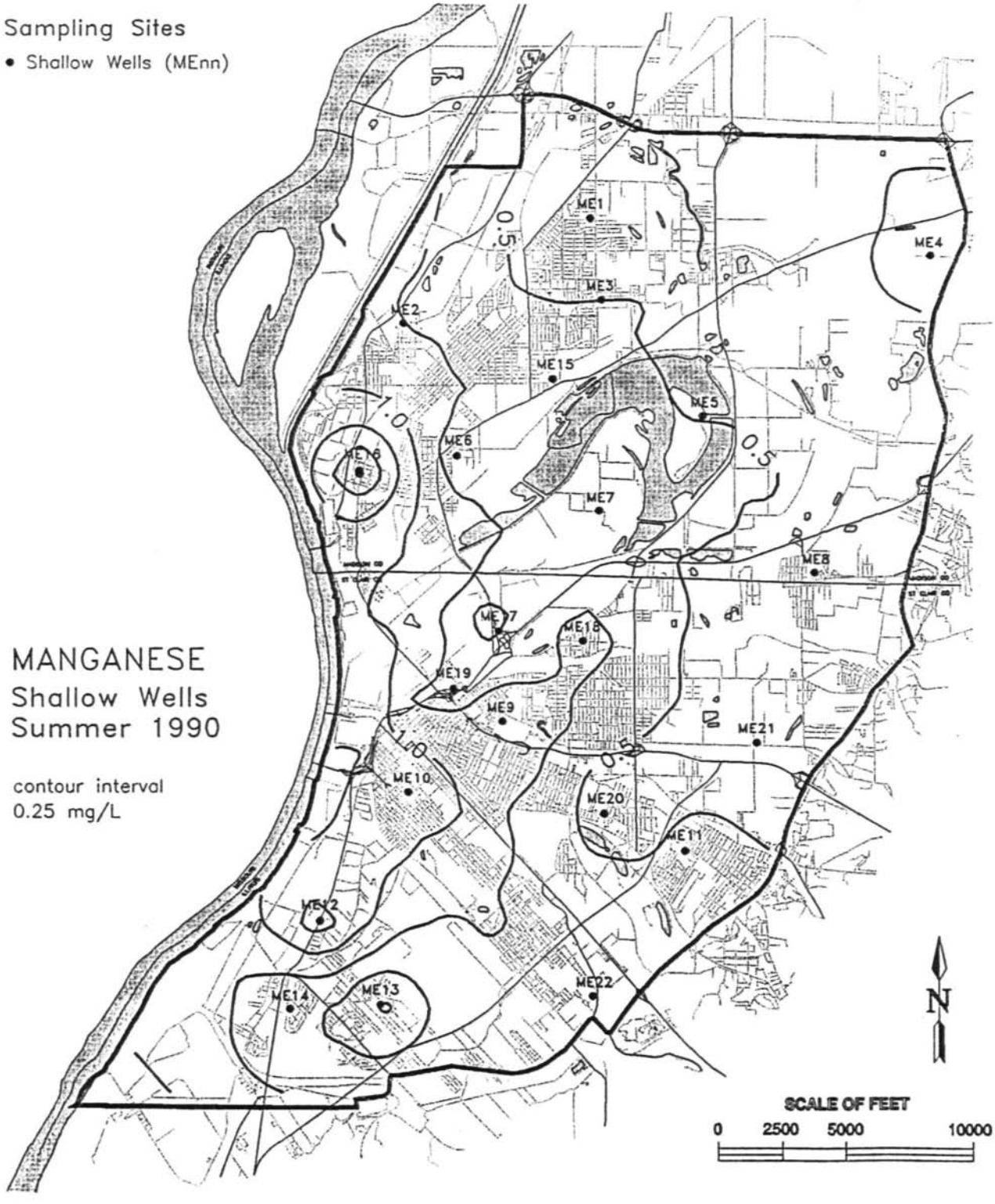


Figure 2.16. Manganese concentration in the shallow portion of the aquifer

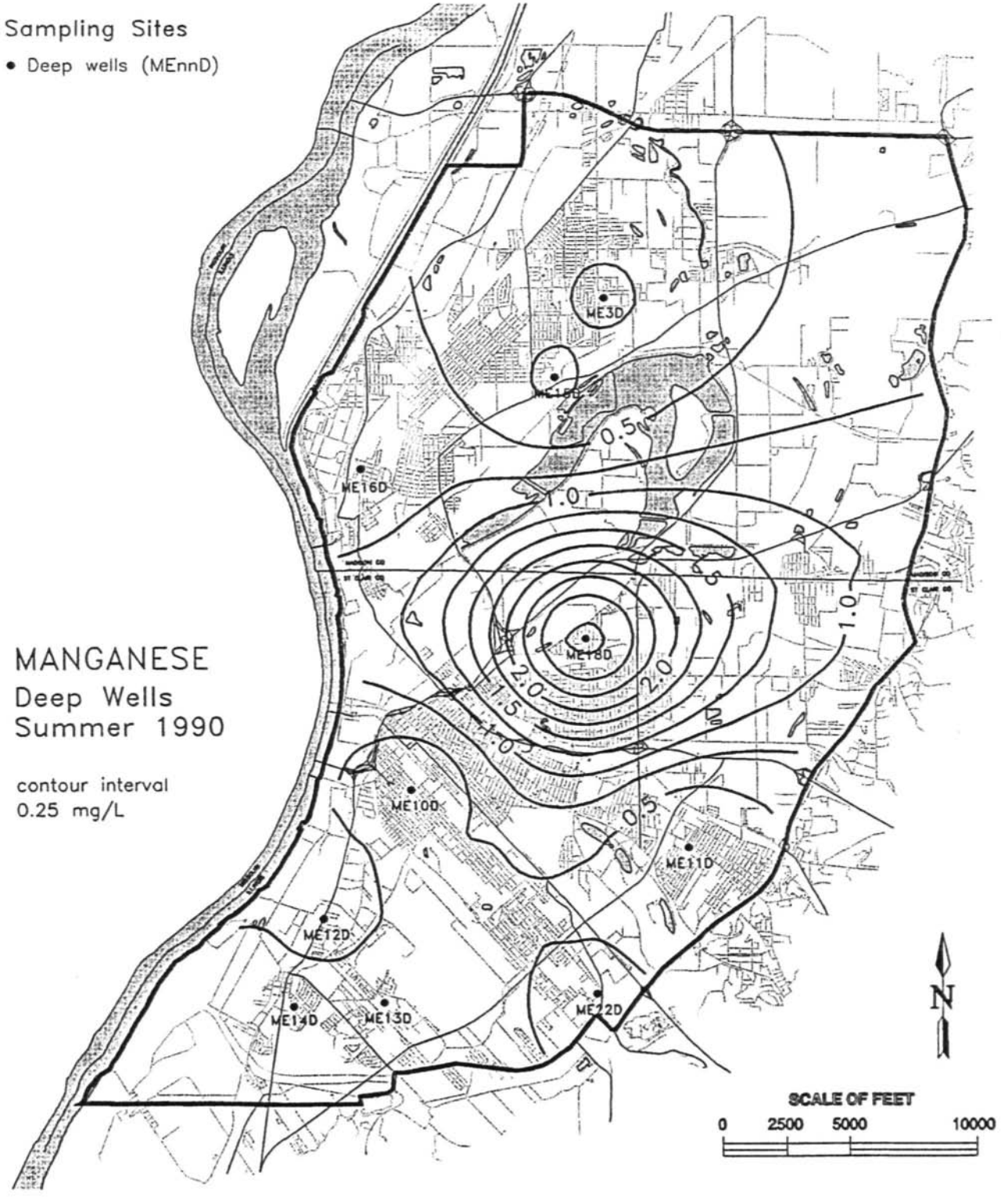


Figure 2.17. Manganese concentration in the deep portion of the aquifer

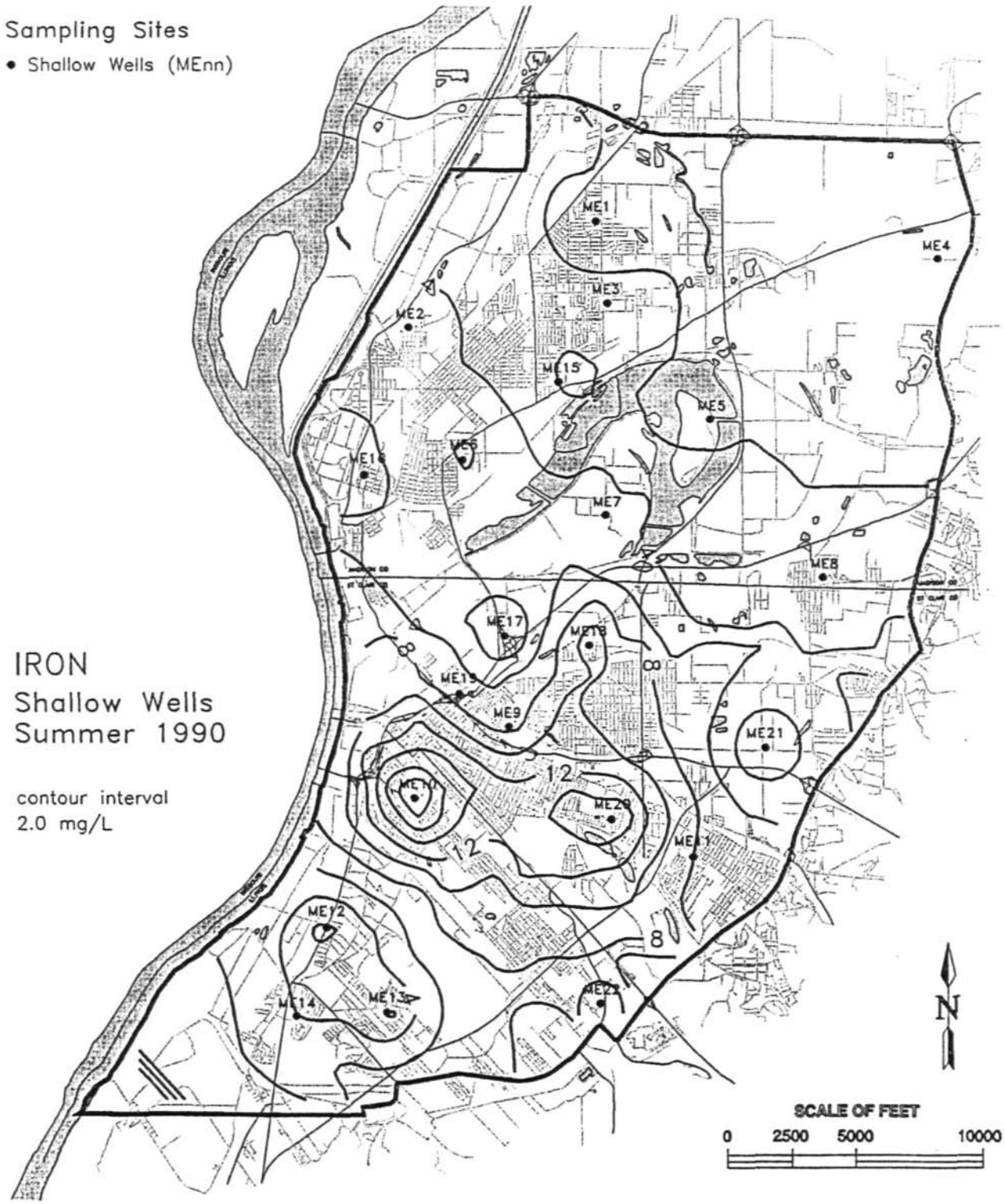


Figure 2.18. Iron concentration in the shallow portion of the aquifer

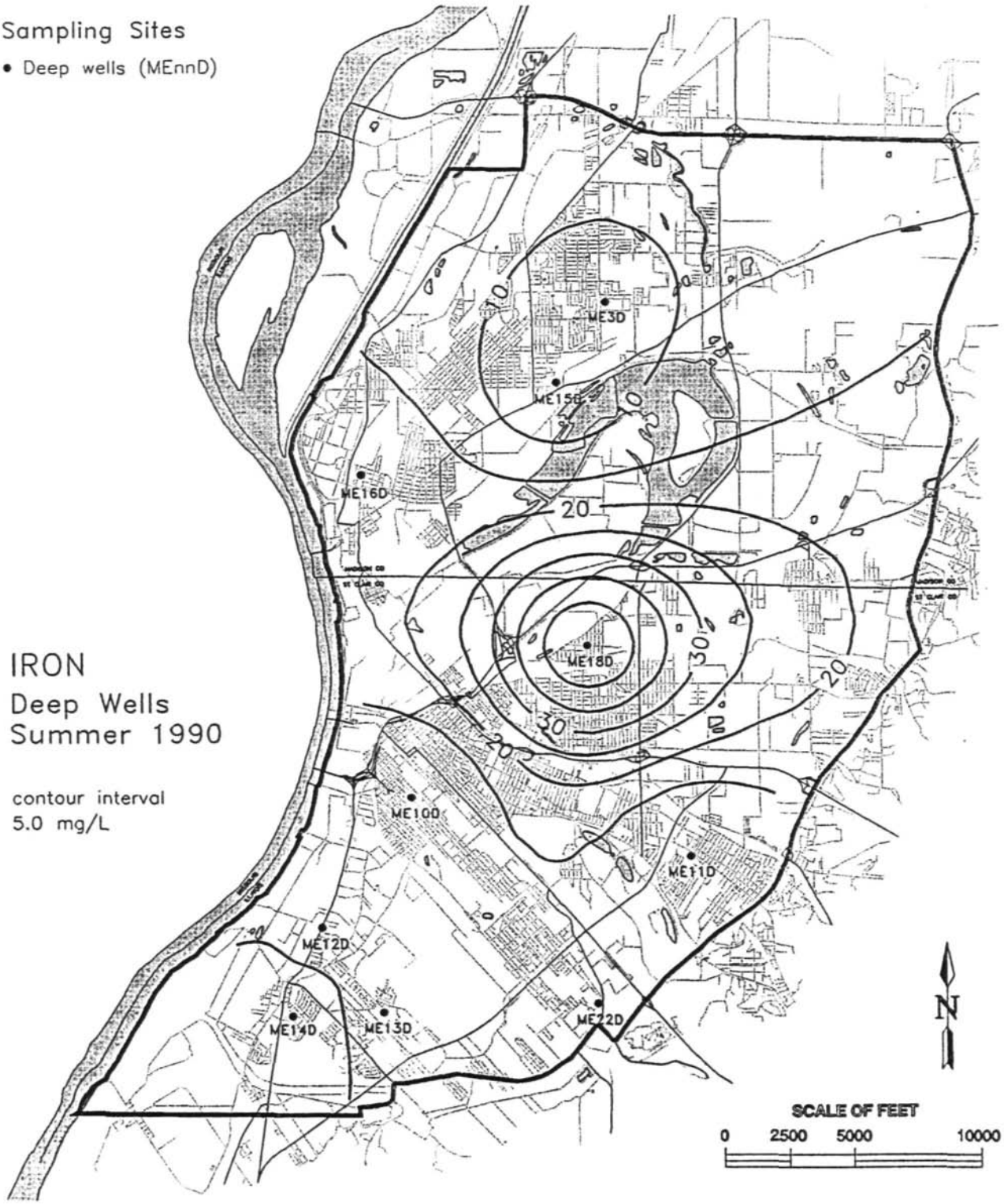


Figure 2.19. Iron concentration in the deep portion of the aquifer

the interpretation of the aquifer water quality. Samples were collected on various dates with different analyses performed as noted above. Appendices B - F summarize the chemical analyses of all the samples for both projects. Each table contains the data from one sampling episode and is organized by source of the water: shallow ground water, deep ground water, or surface water.

Relation to Illinois Effluent Water Quality Standards

A comparison of the chemical results in appendices B - F to the Illinois effluent water quality standards yields only five chemical constituents that were above the standard: ammonium-nitrogen (NH_4-N), iron (Fe), manganese (Mn), total suspended solids (TSS) and phosphate-phosphorus (PO_4-P). NH_4-N, PO_4-P, and TSS were above the effluent standard at only one of the wells, ME18D, ME10, and ME13D, respectively. The high TSS value for ME13D in June 1990 was due in large part to difficulty in development of the well. The Winter 1990 sample from that well did not show large TSS. In general, one would not expect a properly developed well to produce much sediment. The NH_4-N at ME18D and the PO_4-P at ME10 could be easily mitigated by mixing with ground water from other locations that had lower concentrations.

The other two constituents, Fe and Mn were above the effluent water quality standard at several locations. Figures 2.16 and 2.17 are contour plots of the manganese concentration in the shallow wells and the deep wells for the June 1990 sampling. Figure 2.16 shows that the western portion of the region has manganese concentrations in excess of 1 mg/L, with the largest concentration being 2.05 mg/L at ME16. Deeper in the aquifer (figure 2.17), the manganese concentration is below the 1.0 mg/L level everywhere except at ME18D. The contours of manganese concentration in figure 2.17 are dominated by the ME18D value. As will be explored in more detail later, the chemistry at ME18D appears to be anomalous in several ways and organic contamination was detected. With the exception of ME18D, the manganese concentration is no more than twice the effluent standard.

Iron concentrations in the shallow wells (figure 2.18) are above the 2 mg/L effluent standard in all but the extreme northeast and southwest portions of the study area. The largest iron concentrations in the shallow wells are centered in the vicinity of well ME10. All the deep wells (figure 2.19) had iron concentrations in excess of 2 mg/L with the largest value measured at ME18D. Most of the study area has iron concentrations well above the effluent standard.

From this data, it appears that manganese and iron are ubiquitous in the ground water of the American Bottoms. Clearly, iron and possibly manganese will exceed effluent standards.

In terms of other contaminants, no samples had metals concentrations in excess of effluent standards. The organic analyses were performed for a variety of contaminants. Except for phenols and oils, there are no specific effluent standards governing the organic compounds, which will be discussed in the next section.

In general, only iron exceeds the effluent water quality standard consistently over the whole study area. There does not appear to be regional contamination of the shallow alluvial aquifer by either metals or organic substances even though localized contamination is known to exist. Nonetheless, as noted in the next section, a large region of the aquifer may be contaminated and the contamination could be drawn into pumping wells.

Relation to Conceptual Model

Because the monitoring wells sample only a small portion of the aquifer, it is important to evaluate the results of the chemical analyses with respect to the conceptual model presented earlier. This helps verify areas of the aquifer in which contaminants may be present. The conceptual model is a way to extend the monitoring well data to a larger portion of the aquifer that is more appropriate for determining if a pumping well is likely to draw contaminated water. As such, this discussion concentrates on the occurrence of contaminant metals and organic compounds. Although no sample had a concentration of a contaminant metal in excess of the Illinois Effluent Water Quality Standard, several samples had detectable levels of some metals that may be indicative of very low levels of contamination. These metals were chromium, copper, aluminum, nickel, zinc, and mercury.

Samples of the shallow wells taken in February 1989 gave detectable levels of chromium in three wells (ME2, ME4, and ME7) and nickel in one well (ME2). Wells ME4 and ME7 are located in regions where ground-water contamination was not expected. Well ME2 is on the border of the region in which ground-water contamination is more likely to occur. No chromium above the detection limit was observed in subsequent samples of these wells (October 1989 and June 1990). The chromium observed in the February 1989 samples was at concentrations just above the detection limit. That, coupled with the lack of detection in later samples and the location of the samples in regions where contamination is not expected, suggests that the chromium analyses from February 1989 should not be viewed as an indicator of contamination.

In the June 1990 and Winter 1990 samples, concentrations of copper just above the detection limit were observed in wells ME2W, ME9, ME10, ME18, ME18D, and ME28D. Because these wells are within or next to the region identified as having a larger potential for contamination in figure 2.13, the values suggest possible low levels of contamination. Hem (1985) indicates that the use of copper by our industrialized society likely results in greater availability of copper than its low-average natural abundance in rocks would imply.

Aluminum (appendices E and F) was observed in a number of wells (ME2W, ME12D, ME13D, ME22D, and ME28D), most of which are in the regions where industrial contamination would not be expected. Concentrations ranged from 0.037 mg/L to 1.9 mg/L in the wells and 0.123 to 2.05 mg/L in the surface water. According to Hem (1985), aluminum is commonly observed in ground water of neutral pH at concentrations less than 1 mg/L. The concentration of 1.9 mg/L from well ME13D can be explained by the high suspended solids content of that sample in which aluminum-rich mineral particles could have entered the sample even after filtering. Hem (1985) notes that aluminum-rich gibbsite crystals will pass through most filter

media. Therefore the occurrence of aluminum in low concentrations is not considered an indicator of contamination.

Nickel was detected in the June 1990 sample (appendix E) of well ME 17 at a concentration of 0.18 mg/L, well above the detection limit of 0.017 mg/L. Well ME17 is located down gradient of a landfill and next to a major roadway, both of which could be sources of contamination. The landfill is a less likely source of the nickel because of the absence of elevated levels of other constituents, such as sodium, potassium, or chloride, that might be indicative of landfill leachate (Freeze and Cherry, 1979, p. 435)

Zinc was detected in many of the wells scattered over much of the study area (appendices E and F). The concentrations ranged from 0.008 mg/L to 0.43 mg/L). The concentrations of zinc in surface water ranged from 0.02 mg/L to 0.11 mg/L. According to Hem (1985), zinc is a common element in earth materials. Zinc is also widely used in metallurgy and as a white pigment in paint and rubber. Thus, zinc tends to be dispersed widely in the environment as a result of modern industry. The wide availability of zinc from both natural and anthropogenic sources suggests that zinc will not be an important indicator of contamination unless unusually large concentrations are detected. The two largest concentrations were at ME5SW and ME28D, both of which are located in regions considered likely to be contaminated.

Water samples were collected from an IDOT dewatering well (ME28D) located about 1200 feet west-southwest of well ME19 in June and Winter 1990. Concentrations of mercury, chromium, copper, and zinc above the detection limit were observed in this well. With the exception of zinc, the concentrations of these metals were barely above the detection limit, indicating possible low levels of contamination in the aquifer. Well ME28D is significant because it is a pumping well and draws water from a much larger portion of the aquifer than the monitoring wells. The integrating nature of the pumping well leads to a greater likelihood of contaminant detection but an underestimate of the maximum concentration in the aquifer because of mixing of waters in the well itself.

Low concentrations of copper and nickel have been observed in samples from wells ME9, ME10, ME17, ME18, ME18D, ME28D, and ME12D located primarily in the region designated as more likely to be contaminated (appendices E and F). Wells ME2 and ME2W, which had low levels of chromium (appendix C) and copper (appendix E), respectively, are located near the region thought more likely to be contaminated. However, later samples of ME2 did not show detectable chromium. Chromium was also detected in the fall 1989 (appendix C) samples from ME4 and ME7, both of which are located in regions where contamination was not expected. As with ME2, later samples found no detectable chromium, which suggests that the Fall 1990 detections may be spurious. The occurrence of low levels of metal contamination in likely contaminated regions of the study is considered confirmation of the conceptual model.

Organic compounds were detected in several of the wells. Chloroform was observed at 1.21 micrograms/liter (μg/L) in well ME5 from the February 1989 sampling. In October 1989, benzene at 6.6 μg/L and toluene at 6.2 μg/L were observed in wells ME13 and ME22, respectively. Each of these wells is located outside the region expected to be contaminated. This is difficult to reconcile in terms of the conceptual model proposed earlier. In each case,

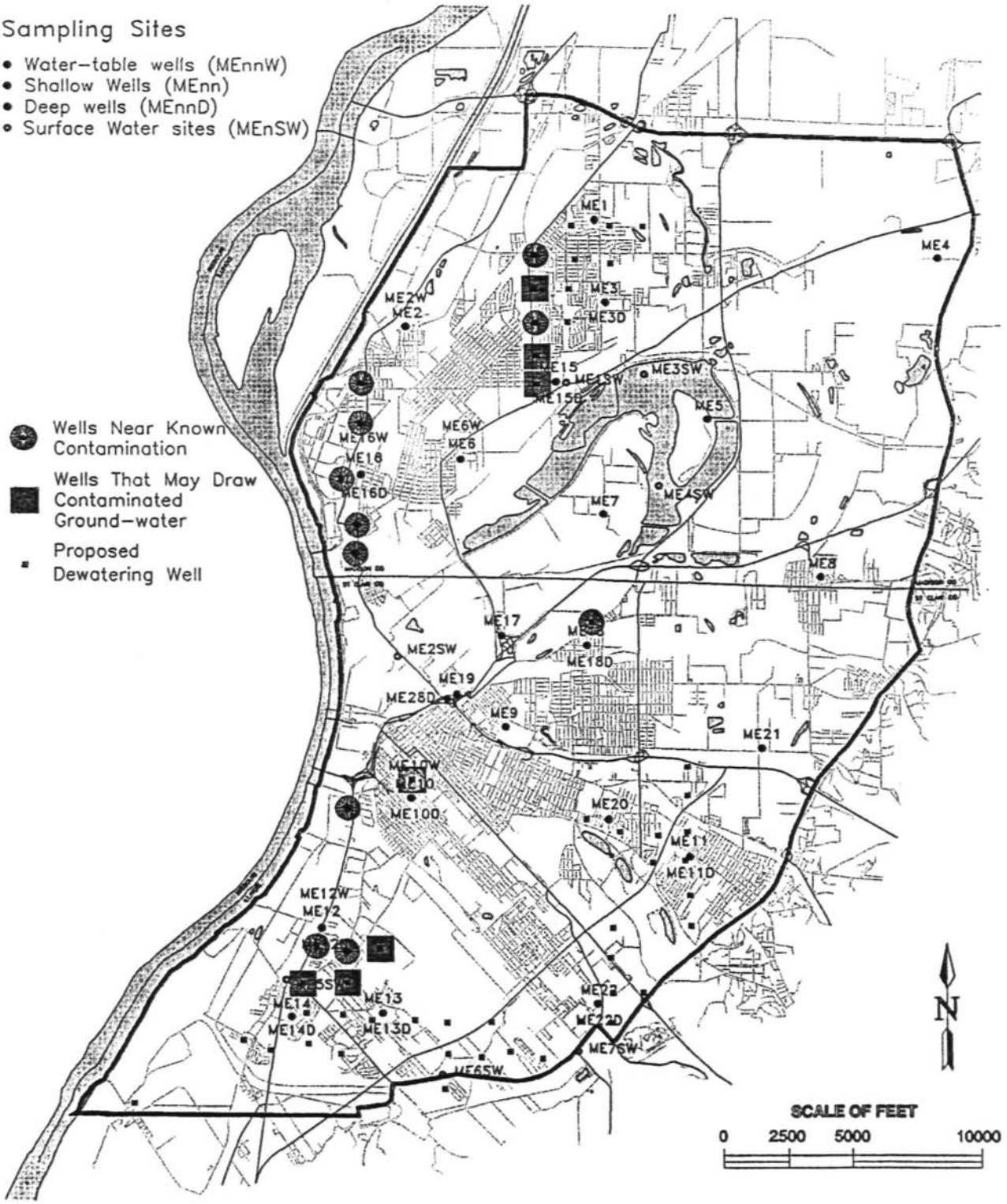


Figure 2.20. Proposed dewatering well locations

the contamination was observed in only one sample of the three (ME5) or four (ME13 and ME22) samples collected from each well. The June and Winter 1990 organic analyses indicated contamination at only three wells, ME15D, ME16D, and ME18D. Wells ME16D and ME18D are within the region that is more likely to be contaminated. Well ME15D is up gradient of the region likely to be contaminated. Methylene chloride and acetone were commonly observed in many of the samples, but often in the method blanks or at concentrations below the detection limit. The presence of either compound in the method blanks indicates a laboratory contamination problem and suggests that low levels of either compound are likely laboratory artifacts.

Nitrate concentrations were all within the Illinois water quality standards. However, several of the wells (ME3, ME4, ME5, ME13, ME14, and ME15) had nitrate values larger than the other wells. Each of these wells is within or down gradient of agricultural activity areas. This is also consistent with the assumptions of the conceptual model: nitrates were more likely to be associated with the agricultural regions.

Comparison of Ground Water and Surface Water Quality

From the data in appendices E and F, it is possible to compare ground-water and surface water quality. In general surface waters tend to be less mineralized than ground water, based on total dissolved solids and the other dominant inorganic elements. Iron concentrations in surface waters are lower than in most ground water, while nitrate-nitrogen tends to be higher. The concentration of metals is comparable in surface water and ground water, but aluminum tends to be higher in surface water. No organic compounds were observed in any of the surface water samples.

CONCLUSIONS AND RECOMMENDATIONS

Figure 2.20 gives the locations of the proposed USACE dewatering wells with respect to the monitoring wells installed in the American Bottoms. The 11 wells designated by circles are located within or very near areas of ground-water contamination. It is highly likely that these 11 wells will yield ground water with detectable levels of contamination. The five wells in the vicinity of ME16 are all in a region considered likely to have contaminated ground water based on figure 2.13. These five wells are surrounded by potential or observed contamination, and the data from ME16D suggest that the aquifer is contaminated by organic compounds at that location. The two northernmost wells are near intersections where gasoline has leaked from underground storage tanks. Both of these sites will have contaminated ground water for some time into the future. The vicinity of ME 18 is likely contaminated based on the anomalous chemistry at ME18D (very high iron, sulfate, etc.), the observed phthalate, and the presence of two up gradient contaminant sources. The single well southwest of ME10 is in a region where contamination is expected and is very close to two regulated facilities with a high potential for contamination. The two wells south of ME12 are near the contaminated Sauget area and would likely draw contaminated ground water from the north.

The seven wells designated by large squares are close enough to areas of observed or likely contamination that they have been identified for special consideration. These wells may or may not draw contaminated ground water. But if these wells are installed, special consideration should be given to monitoring their discharge for the presence of contaminants. The three wells in the north near ME3 and ME 15 could draw contamination from the gasoline spills to the north or from the region of suspected contamination to the south. The well north of well ME10 is near two sources of moderate contamination. The three wells south and east of ME12 are close enough to the contamination in the vicinity of Sauget that possible reversal of the ground-water flow direction due to the pumping (from westerly to southeasterly) could induce contamination to migrate south from Sauget.

The remaining wells are not expected to draw contaminated ground water. For them, only high iron concentration would be a concern with respect to discharge to surface water.

The evaluation of the likelihood of the proposed dewatering wells drawing contaminated water is based, in part, on the conceptual model of the ground-water flow system. This model was only partially confirmed by the ground-water samples. The majority of regulated facility sites with ground-water contamination are in the region designated as the likely contaminated region by the conceptual model. In one sense, this is confirmation of the conceptual model. However, we recognize that contamination was not detected in all samples taken from the likely contaminated region. This means that not all of the ground water in the likely contaminated region is contaminated. It must be recognized that the ground-water samples collected for this study represent the ground-water quality at a specific location in the aquifer at one point in time. Contaminated ground water may be only a few tens of feet away from the monitoring wells, but would not be detected because of the small volume of water removed during sampling. A pumping well, such as the IDOT well (ME28D) sampled during this study, can draw nearby contaminated ground water into it. Such pumping wells often draw contaminated ground water from one portion of the aquifer and uncontaminated ground water from another portion, and then mix them inside the well. Consequently, it is difficult to predict the contaminant concentrations that will be observed in a pumping well.

It is anticipated, that contaminated ground water would be drawn into several of the dewatering wells because of their proximity to nearby contaminated ground water. It is possible that a few of the wells would not have detectable contamination, but given everything we know about the aquifer system, this is considered unlikely.

Section 3

LIMNOLOGICAL CHARACTERISTICS OF HORSESHOE LAKE AND THE IMPACT OF GROUND-WATER DISCHARGES TO THE LAKE

by Raman K. Raman

INTRODUCTION

Horseshoe Lake, situated about 2 miles east of the main stem of the Mississippi River, is a meander cutoff of the Mississippi. Figure 3.1 shows the location and configuration of the lake. Hill et al. (1981a) reported that the lake has a water surface area of about 2150 acres at normal pool elevation, with an average depth of 3 feet except for a 50-foot deep hole near the sand and gravel operation. Under normal conditions, the three major sources of flow into the lake are urban runoff through Nameoki Ditch, agricultural runoff through the Elm Slough, and treated waste effluent from the Granite City Steel (GCS) facility.

The GCS wastewater treatment plant discharges approximately 20 million gallons per day (mgd) into the west leg of Horseshoe Lake. The industrial wastewater treatment system, a partially closed-loop system, includes two large retention basins (lagoons) and several treatment unit processes. The coke-oven process wastes are treated by activated sludge process, and the effluent is returned to the lagoons for recycling. Approximately 20 mgd of the lagoon waters are treated separately prior to their discharge to Horseshoe Lake. The treatment consists of oil skimming, coagulation, sedimentation, sand filtration, chlorination, and sulfonation. The GCS effluent is monitored semiweekly for various conventional pollutants, as well as several metals, cyanide, and phenol. The GCS effluent is presumed to be in compliance with the National Pollution Discharge Elimination System (NPDES) permit stipulations.

During storm events, which are likely to occur about ten times a year (Hill et al., 1981a), storm water flows in the Cahokia Canal are diverted into the lake for flood retention purposes. Lake levels can be drawn down below spillway level through a low-flow weir arrangement operated by the Metro East Sanitary District.

Ground Water

The current hydraulic interchange between the lake water and the ground water is low. During the extended drought of 1953, an estimated 1.5 mgd of lake water leaked into the aquifer. The ground-water contribution to the lake also was about 2.5 mgd from 1973 to 1974 because of higher ground-water elevations. Increased sedimentation of the lake since the data period for which these calculations were made (1953, 1973-1974) has minimized the relative influence of the ground water. Ground-water recharge is not considered a significant factor compared with surface flows (Hill et al., 1981a).

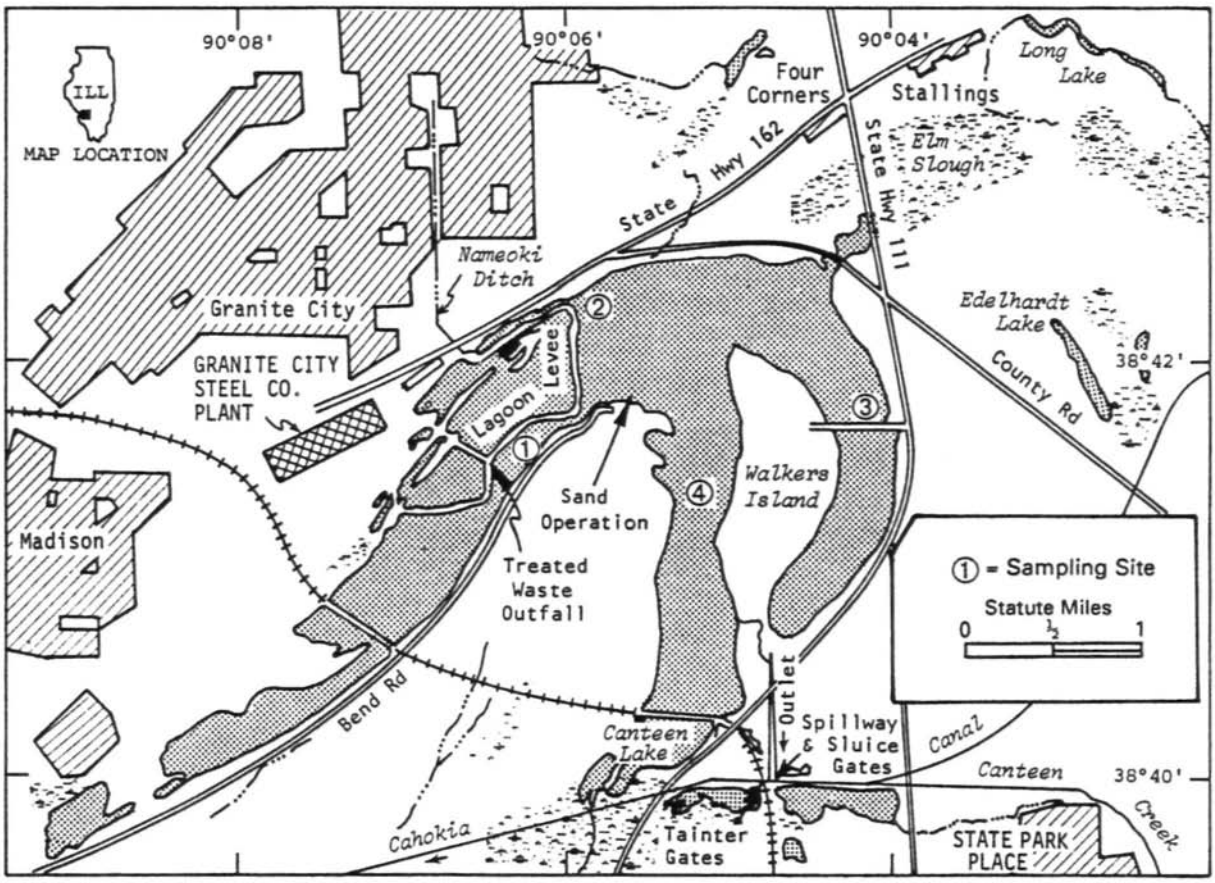


Figure 3.1 Horseshoe Lake and the sampling sites

Horseshoe Lake State Park

The lake and its environs constitute Horseshoe Lake State Park, which is managed by the Illinois Department of Conservation (IDOC) for recreational purposes. Picnicking, boating, fishing, camping, hunting, hiking, and winter sports are permitted at the park. Both commercial fishing and sports fishing are allowed at the lake. Boat fishing is allowed except during waterfowl season. To accommodate waterfowl hunters, numerous public blinds are located in the lake. Hunting is allowed only from these public blinds during waterfowl season. Dove hunting is allowed on a portion of Walkers Island (figure 3.1) during the month of September. There is a 25-horsepower limit for all watercraft. Within the park, 4 miles of hiking trails and during winter months 4 miles of cross-country ski trails are maintained. Other major uses of the lake are flood storage, industrial waste assimilation, and sand and gravel operation.

MATERIALS AND METHODS

General Lake Limnology

To assess the current conditions of the lake, certain physical, chemical, and biological characteristics of the lake were monitored during 1990 at approximately biweekly intervals from mid-April until late October. A total of 13 field visits were made. During each field visit, water samples were also collected from Nameoki Ditch at Route 116 for chemical and biological characterization and to make discharge measurements. Figure 3.1 shows the locations of the tributary and lake monitoring stations.

In situ observations for water temperature, dissolved oxygen (DO), and Secchi disc readings were made at the lake stations. An oxygen meter, Yellow Spring Instrument Company model 58 with a 50-foot lead DO probe and thermister, was used to determine DO and water temperature. The DO probe was calibrated at 100 percent saturation using an air saturation chamber. Water temperature and DO measurements were obtained in the water column at 1-foot intervals commencing from the surface of the lake. Water temperature and DO measurements in the tributary pertain to the surface water.

An 8-inch diameter Secchi disc with black and white quadrant markings attached to a calibrated line was used to measure water transparencies and total water depth. The disc was lowered until it disappeared from view, and the disc's depth of immersion was noted. The disc was lowered farther and then raised slowly until it reappeared. Again, the depth of immersion was noted, and the average of these two observations was recorded as the Secchi disc reading.

Subsurface water samples for chemical and algal identification and enumeration were obtained from all the lake and tributary stations, using appropriate sample bottles.

Water samples for determinations of metals were collected in 1-liter round, glass bottles containing appropriate preservatives. Samples for analyses of phosphates and various forms of nitrogen were preserved with 5 milliliters (mL) of concentrated sulfuric acid. Samples for determining turbidity, total dissolved solids, suspended solids, volatile suspended solids, and

sulfates were collected in plastic bottles, transported to the laboratory in an ice chest, and refrigerated until the analyses were performed.

Determinations for pH and alkalinity were made at the lake site soon after sample collections. *Standard Methods* (APHA et al., 1989) procedures were used except where noted. Table 3.1 gives the methods and procedures for the chemical analyses. Metal analyses for routine lake samples were limited to the determination of total iron, lead, and manganese. Determinations for arsenic and mercury were made once a month.

Water samples in a volume of 380 mL were collected for algal identification and enumeration, preserved with 20 mL of formalin at the time of collection, and stored at room temperature until examined. Each sample was thoroughly mixed and a 1-mL aliquot was pipetted into a Sedgwick-Rafter counting cell. A differential interference contrast microscope equipped with a 10X or 20X eyepiece, 20X or 100X objective, and a Whipple disc was used for identification and enumeration. Five short strips were counted. The algae species were identified and were then classified into five main groups: blue-greens, greens, diatoms, flagellates, and desmids. Individual cells of green algae were counted, but *Actinastrum, Coelastrum,* and *Pediastrum* were recorded by each colony observed. Each cell packet of *Scenedesmus* was counted. Diatoms were counted as one organism regardless of their grouping connections. For flagellates, a colony of *Dinobryon* or a single cell of *Ceratium* was recorded as a unit.

Benthic samples for macroinvertebrate examination were obtained from each lake station on May 3, 1990. Three grab samples for macroinvertebrate analyses were taken at each station with an Ekman dredge (6× 6 inches). The samples were washed in a 30-mesh screen bucket, and the residues were placed in a quart jar and preserved in 95 percent ethyl alcohol. In the laboratory, the samples were washed again, and the organisms were picked from the bottom detritus, identified, counted, and preserved in 70 percent ethyl alcohol.

Diel obervations for temperature, DO, pH, and conductivity were made once at lake stations 1,2, and 4 using Hydrolab Corporation's Data Sonde I monitors at 1/2-hour intervals for the period May 3, 1990, 1300 hours to May 4, 1990, 1300 hours. These Data Sonde units, equipped with data loggers, were supported by suitable floats, anchored in place for the duration of the observations, and then retrieved. The units were calibrated as per manufacturer's instructions prior to deployment in the field.

Nameoki Ditch Discharges

As indicated earlier, flow measurements were made in Nameoki Ditch (whenever there was flow) during each field visit. Discharge was measured using the Marsh-McBirney, Inc. portable flowmeter model 201D and the equal width increment technique of streamflow measurement developed by the United States Geological Survey (USGS). Stream discharge was determined by subdividing the cross section of the channel into partial sections and measuring the flow velocities within each partial section. At each partial section, a single-velocity measurement was made at 0.6 of depth from the surface. Knowing the depths and the widths

Table 3.1. Analytical Procedures

Parameters	*Procedures*
Turbidity	Nephelometric method using HF-DRT 100D ratio turbidimeter
Suspended solids	Dry weight of solids retained on gooch crucible with fiber glass filter
Volatile suspended solids	Loss on ignition of suspended solids at 550° C for 1 hour
Dissolved solids	Residue on evaporation and filtration at 103° to 105° C
pH	Glass electrode method with portable Metrohm Herisau meter, model E588
Alkalinity	Potentiometric method; titration with standard sulfuric acid solution to an end point of 4.5
Conductivity	Metrohm Herisau meter, model E587
Sulfate	Turbidimetric method
Total phosphate-P	Sample digested with sulfuric-nitric acid mixture and determined by ascorbic acid method
Total ammonia-N	Steam distillation, modified phenate method
Dissolved ammonia-N	Modified phenate method after sample filtration through 0.45μ m filter paper.
Dissolved nitrate-N	Chromotropic method after sample filtration through 0.45 μ m filter paper
Total arsenic	Graphite atomic absorption
Total mercury	Cold vapor atomic absorption
Total Fe, Mn, and Pb	Flame atomic absorption

of the partial sections and the mean velocities in these subsections, discharges could be computed. The sum of the individual subsection discharges determined the total stream discharge.

Controlled Ground-Water Discharge Studies

To assess the probable changes in DO, water temperature, and other water quality characteristics when the ground water is discharged into surface streams, a controlled ground-water discharge into a highway surface drain investigation was carried out on August 8, 1990. One of the Illinois Department of Transportation's (IDOT) ground-water pumps at the Trilevel Bridge in the Collinsville, Illinois area was used to discharge water into a concrete open channel drain about 700 feet long. Changes in DO; water temperature; velocity; iron, manganese, and phosphate (all dissolved) were monitored at 100-foot intervals along the open channel flow at three different discharge rates: 600 gallons per minute (gpm), 450 gpm, and 300 gpm. Figure 3.2 shows the views of well discharge, mercury manometer for discharge measurement, covered road drain to lengthen the channel flow and a close-up of the channel flow. Dissolved oxygen and water temperatures were measured using the YSI model 58 probe after calibration, and the velocities were determined at the deepest point using the Marsh-McBirney, Inc. portable flowmeter model 201D. All samples were filtered at the site and preserved for laboratory analyses for iron, manganese, and phosphate.

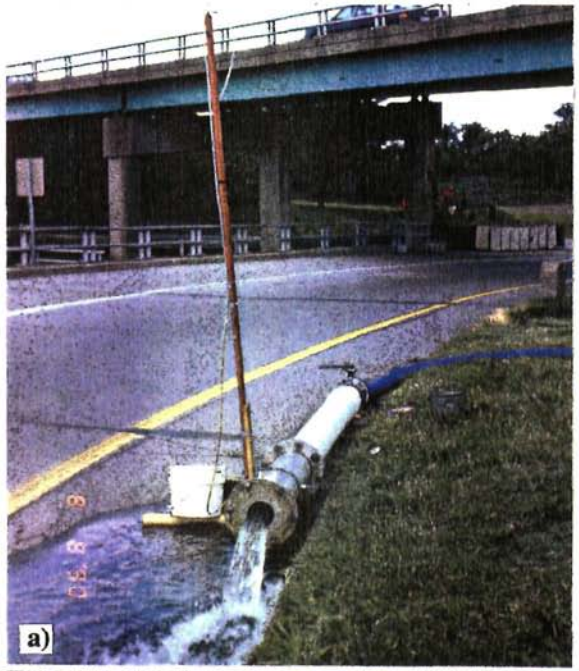
Since the 1985 completion of a major restoration effort of Crystal Lake in Urbana, Illinois, the lake has been receiving ground water continually. Figure 3.3 shows the well head fountain, cobblestone channel flow, and the confluence of the ground water with the lake. Ground water augments the lake water to maintain adequate water level in the lake. The lake, an oxbow cutoff of the meander from the Saline Branch of the Salt Fork River, has a surface area of 7.7 acres, a volume of 66 acre-feet (ac-ft), and a maximum depth of 13 feet. The well pump has a rated capacity of 250 gpm.

In situ observations for DO and water temperature and water sample collections for iron, manganese, and phosphate (all dissolved) were made at two sites, at the fountain and near the confluence of the ground water with the lake, at monthly intervals from May to October 1990. The data collected in Crystal Lake will help assess the impact of ground-water discharges in the American Bottoms area into Horseshoe Lake.

RESULTS AND DISCUSSION

Lake Limnology

Temperature and Dissolved Oxygen. Lakes in the temperate zone generally undergo seasonal variations in temperature through the water column. Deep lakes with depths greater than 15 feet typically exhibit a very pronounced summer stratification and less pronounced inverse winter temperature gradients. These temperature variations are perhaps the most influential controlling factors within the lakes. Dissolved oxygen levels in natural waters depend



a) and b) Views of ground-water discharge and the orifice/manometer

Covered road drain

Open channel flow

Figure 3.2. Controlled ground-water discharge investigation

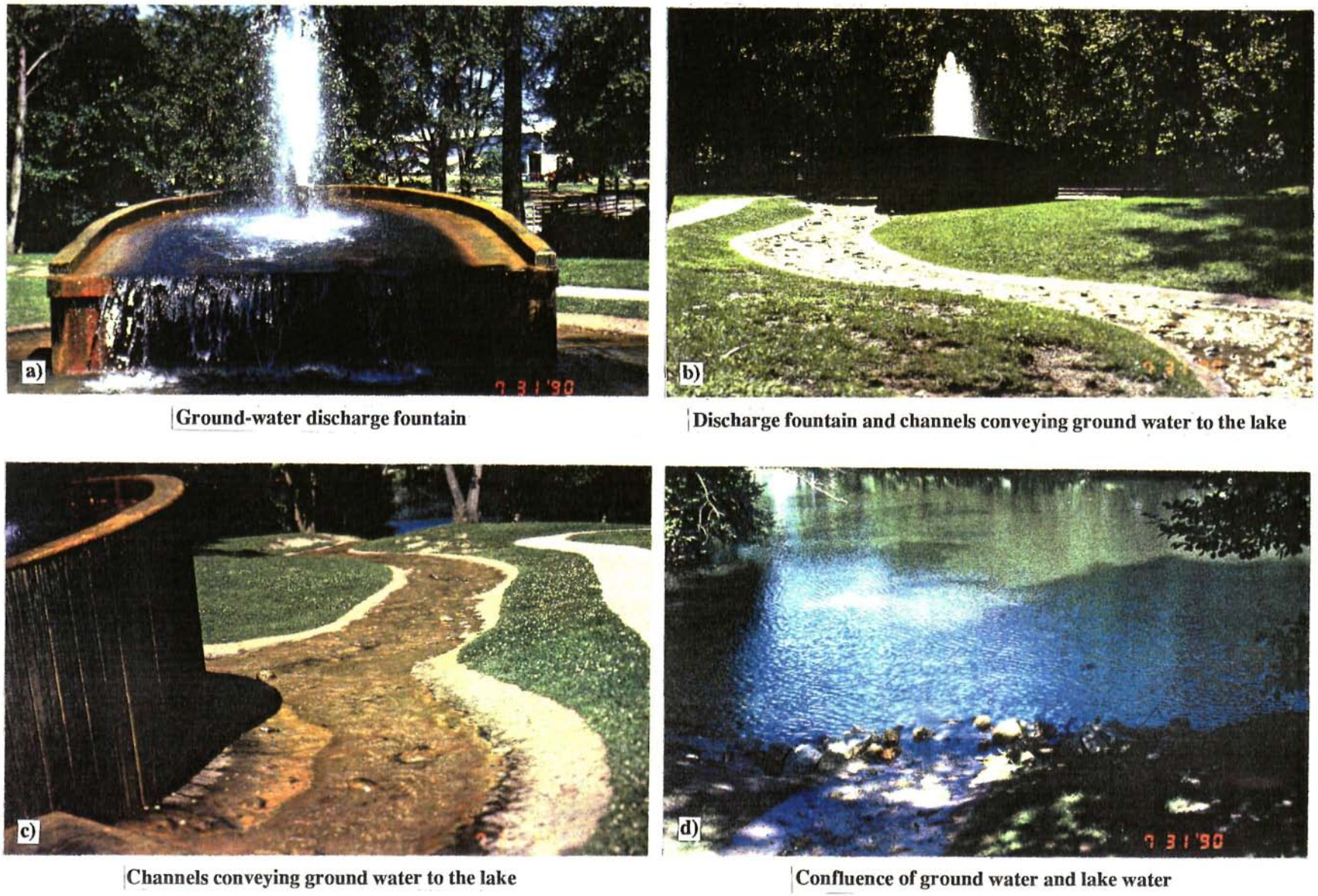


Figure 3.3. Ground-water discharge into Crystal Lake, Urbana, Illinois

on the physical, chemical, and biochemical characteristics of the water body and are considered a key parameter in assessing the overall ecological condition of the system.

Figures 3.4-3.7 show the water temperature and DO profiles for selected dates for lake stations 1-4, respectively. Appendix G gives all DO and water temperature data for these sites. Figures 3.4-3.7 clearly indicate that the DO and water temperature values are mostly uniform throughout the water column at each lake station. The water temperature at station 1 was generally higher than at other sites, and the differences were more pronounced during spring and fall compared with the summer observations. Station 1 is influenced by GCS wastewater effluent discharge and station 2 by the stormwater runoff from Nameoki Ditch. Observations for DO in the lake stations were above the IPCB's minimum standard of 6.0 milligrams per liter (mg/L) except in the near bottom waters and at stations 3 and 4 on June 20, 1990 (figures 3.6 and 3.7 and appendix G, respectively). The observed concentrations of DO in the near bottom waters in the lake stations were always greater than 3.9 mg/L. In contrast, Lake Ellyn in DuPage County, which is a shallow stormwater retention basin with a depth of 5 feet, exhibited total oxygen depletion at depths below 3 feet from the surface during July (Hill et al., 1981b).

Table 3.2 shows the percent DO saturation levels observed in the lake during this investigation. Station 1 showed supersaturated conditions at the surface except on one occasion. Station 2 did not exhibit supersaturated conditions when monitored during or very soon after storm events in the area resulting in stormwater discharges through Nameoki Ditch. The highest saturated value observed was 233 percent (table 3.2).

Secchi Disc Transparencies. Secchi disc visibility is a measure of the water transparency or its ability to allow light transmission. Although not an actual quantitative indication of light transmission, the Secchi disc tranparency serves as an index and a means of comparing similar bodies of water or the same body of water at different sites and times. Transparency is related to water color, algal growth, and suspended sediments consisting of silt, clay, and organic debris.

Table 3.3 gives the mean and range of values observed for Secchi disc readings in the lake stations along with the summary of observations for other physical and chemical water quality parameters. The table also includes this information, except the Secchi disc values, for Nameoki Ditch.

The mean Secchi disc transparency was higher at station 1, as were the minimum and maximum observed values compared with the other three lake stations. This is primarily due to the direct impact of the continuous discharge of treated industrial wastewater effluent from the GCS plant. Values for stations 2-4 were comparable and typical of shallow lakes in Illinois. The temporal variations in Secchi disc readings and other parameters are also shown for lake stations 1-4 in figures 3.8-3.11, respectively.

Turbidity. High turbidity affects the aesthetic quality of the water. Its origins are generally considered to be municipal and industrial wastes; clastic materials derived from the drainage basin; soil erosion resulting from agricultural practices and urban and highway

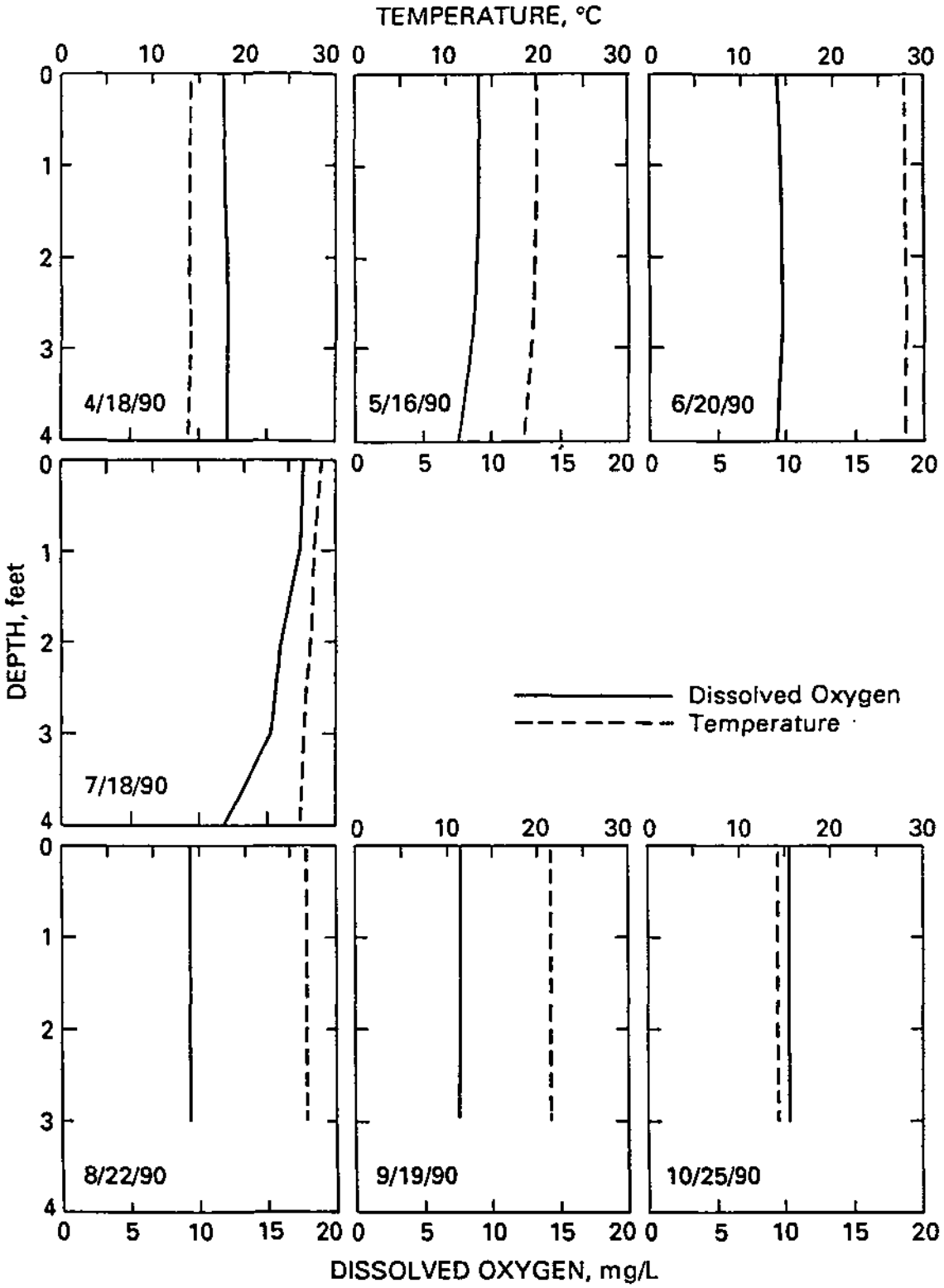


Figure 3.4. Dissolved oxygen and temperature profiles at station 1

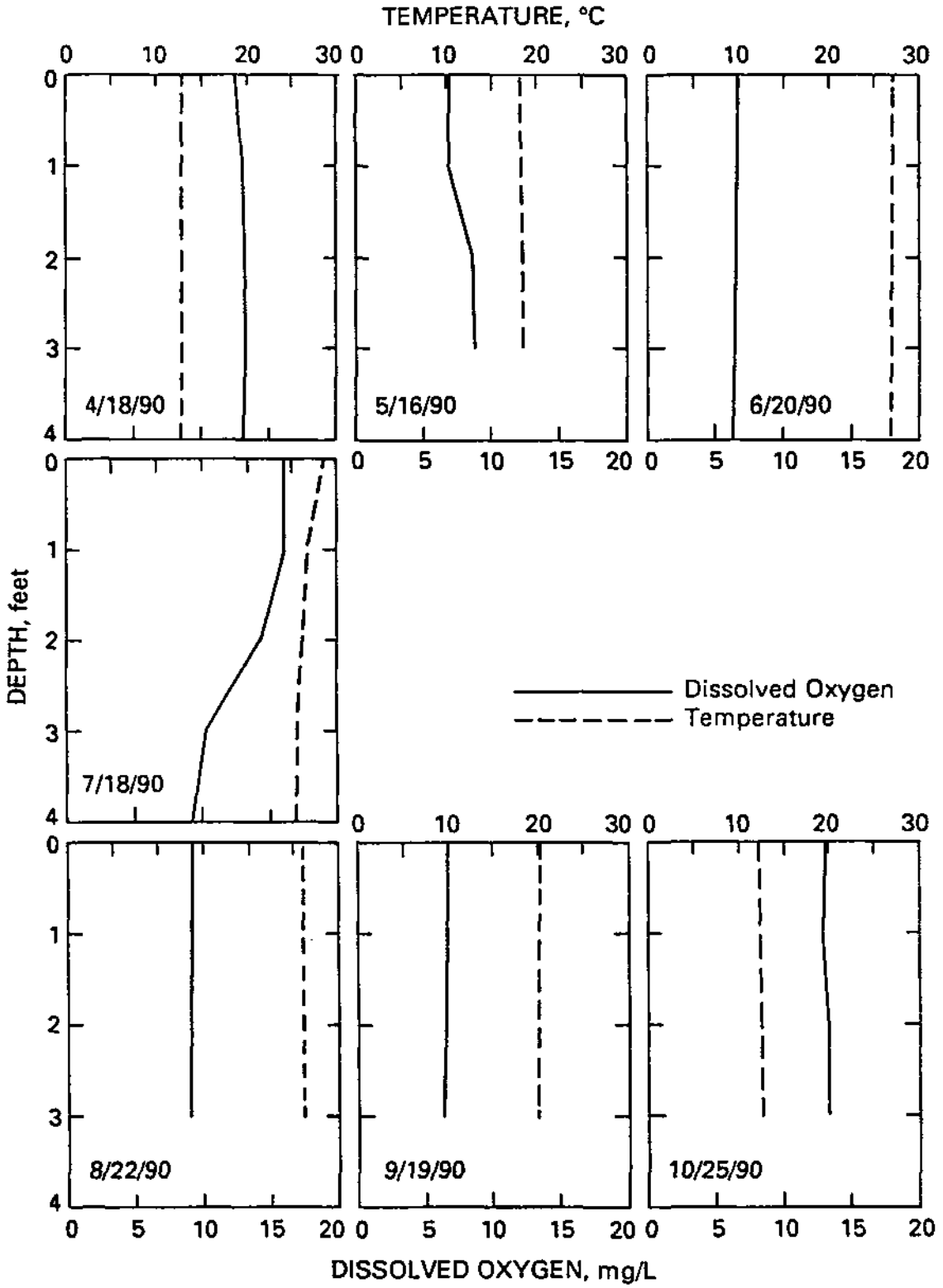


Figure 3.5. Dissolved oxygen and temperature profiles at station 2

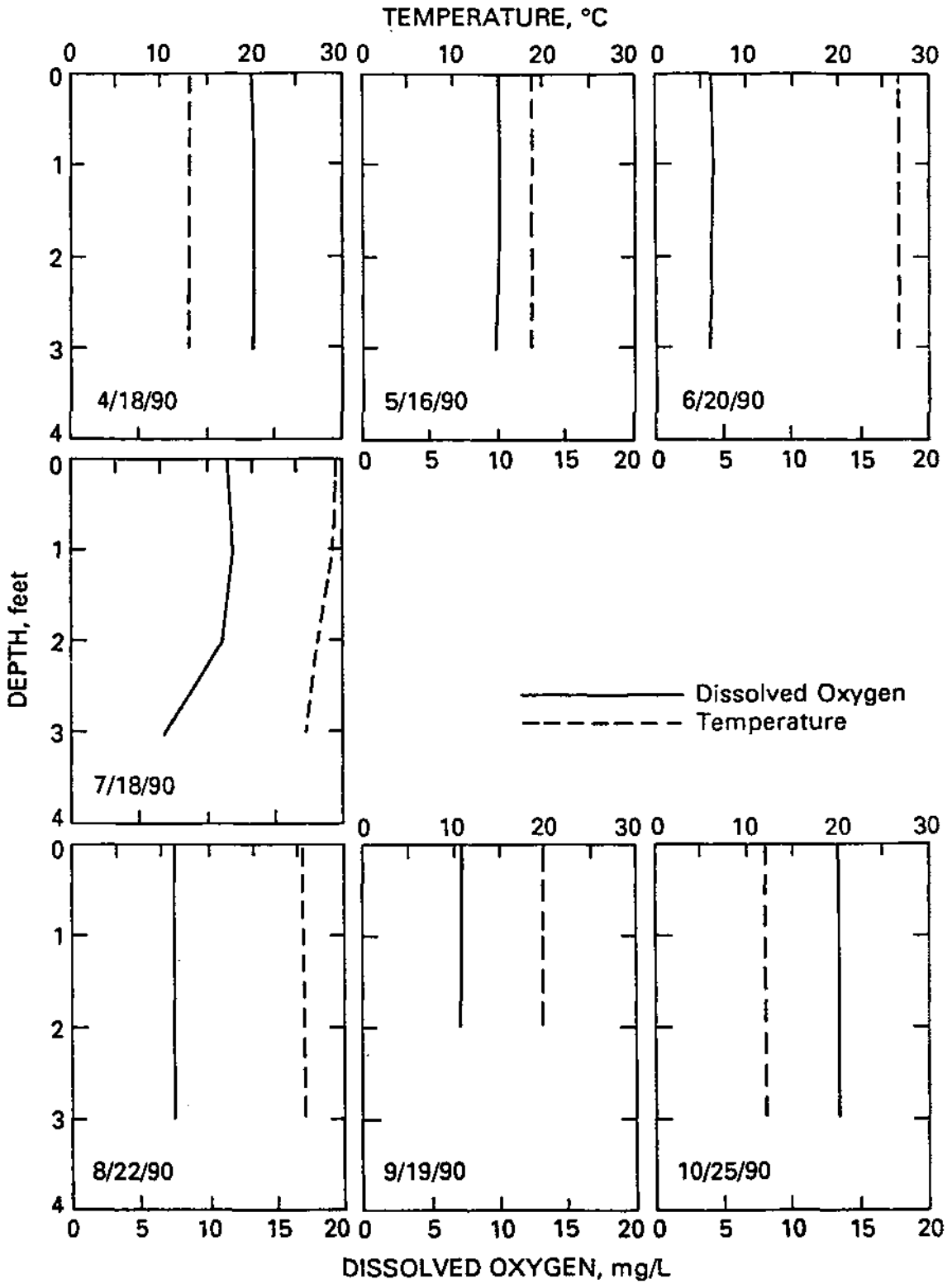


Figure 3.6. Dissolved oxygen and temperature profiles at station 3

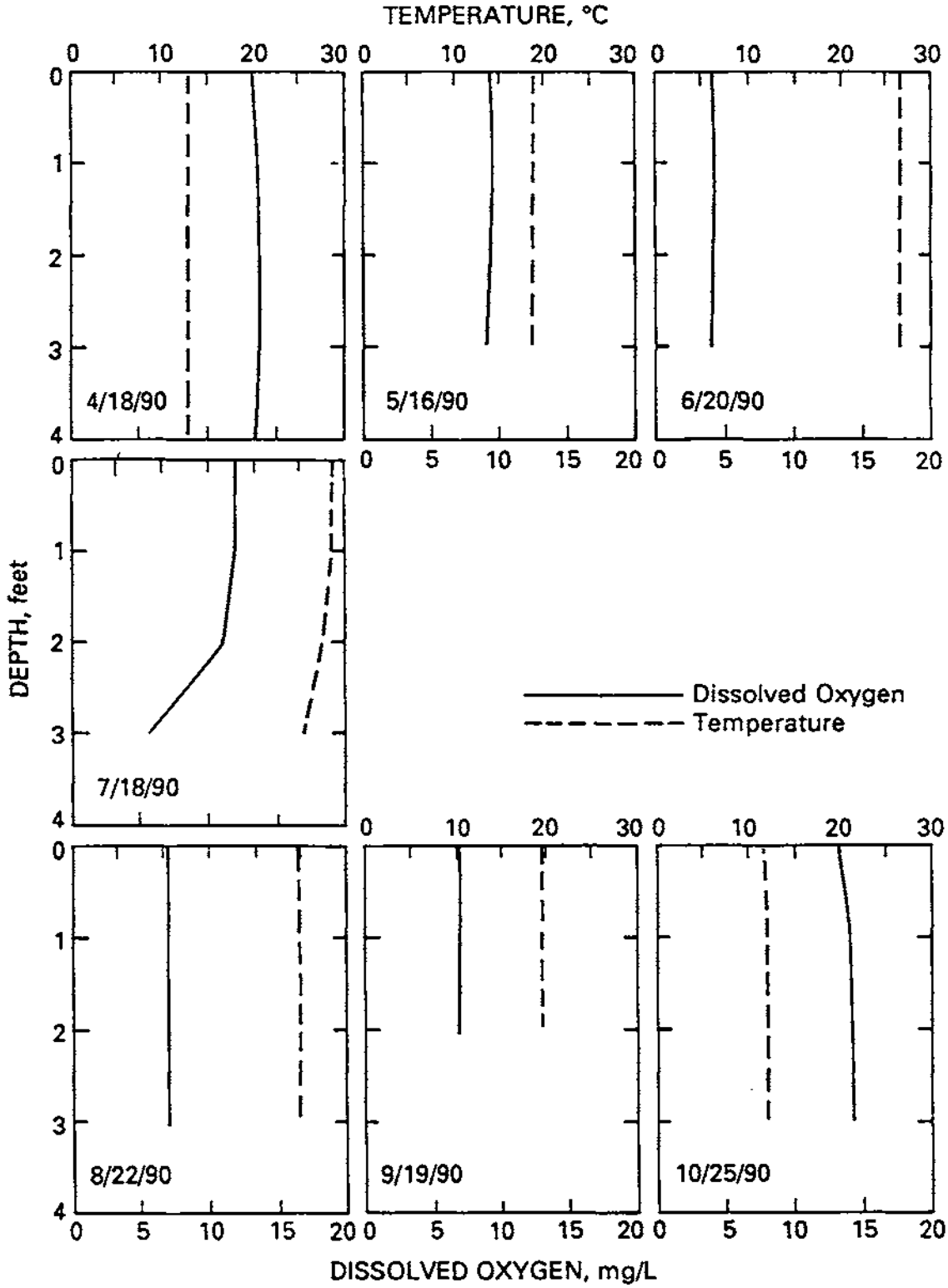


Figure 3.7. Dissolved oxygen and temperature profiles at station 4

Table 3.2. Dissolved Oxygen Saturation
(saturation in percent)

Station	*Depth (feet)*	*Dates* 4/18/90	5/2	5/16	6/20	7/3	7/18	8/9	8/22	9/5	9/19	10/2	10/25
1	0	116	108	101	120	184	233	123	117	171	88	131	101
	1	117	111	101	124	183	225	109	118	175	88	131	101
	2	118	111	99	123	180	206	94	118	138	87	132	100
	3	117	113	92	122	168	194	93	118	123	87	132	100
	4	117	108	80	119	113	149	91					
2	0	117	136	74	83	151	210	138	113	149	74	125	122
	1	123	136	74	84	149	203	139	113	139	74	125	120
	2	124	136	92	81	145	179	129	110	84	72	126	124
	3	124	136	95	81	115	127	106	110	77	71	126	124
	4	123	136		80	108	114						
3	0	126	149	108	52	139	152	93	92	128	79	116	125
	1	127	150	108	53	139	157	93	93	96	78	116	125
	2	128	150	107	51	127	141	66	93	63	78	116	126
	3	128	148	104	49	77	86	54	93				125
4	0	128	134	101	50	125	159	97	85	126	76	120	122
	1	131	135	103	52	125	158	95	85	130	76	120	132
	2	133	135	100	52	121	144	62	85	70	75	120	132
	3	133	134	98	50	61	72	52	85				132
	4	130											
Nameoki Ditch		64	31	72	66	114	111	133	30	75	22	118	77

Table 3.3. Water Quality Characteristics of Horseshoe Lake and Nameoki Ditch

Parameters	*Lake station 1*		*Lake station 2*		*Lake station 3*		*Lake station 4*		*Nameoki Ditch*	
	Mean	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
Secchi disc readings (inches)	15	10-24	12	7-18	12	6-18	11	6-18	.	.
Turbidity (NTU)	26	17-43	38	16-75	37	19-60	40	22-66	37	12-149
Suspended solids	36	16-66	42	14-50	42	19-56	49	25-70	28	6-96
Volatile suspended solids	15	0-28	18	4-29	19	2-30	20	0-33	8	0-21
Dissolved solids	632	528-934	476	240-674	489	434-618	490	390-598	300	72-484
pH (unit)	8.3	7.60-8.78	8.6	7.5-9.0	8.6	7.7-9.2	8.8	7.7-9.9	7.5	6.7-7.9
Alkalinity as $CaCO_3$	133	92-184	100	63-124	101	78-122	99	80-123	171	37-292
Conductivity (μ mho/cm at 25° C)	934	739-1229	679	357-876	712	572-826	718	561-827	451	92-790
Sulfate	137	101-223	106	48-154	109	89-143	110	93-152	55	6-94
Total phosphate-P	0.09	0.04-0.12	0.16	0.10-0.27	0.16	0.05-0.26	0.17	0.09-0.24	0.31	0.08-0.60
Total ammonia-N	0.26	0.08-1.21	0.24	0.08-0.65	0.22	0.10-0.77	0.22	0;09-0.75	0.49	0.15-1.95
Dissolved ammonia-N	0.16	0.02-1.15	0.12	0.01-0.55	0.13	0.03-0.64	0.09	0.00-0.58	0.39	0.03-1.85
Dissolved nitrate-N	1.43	0.52-2.51	0.30	0.08-1.23	0.12	0.03-0.38	0.11	0.04-0.40	0.27	0.04-0.93
Iron	0.49	0.30-0.86	0.90	0.40-2.05	0.87	0.33-1.47	1.01	0.59-1.91	1.46	0.52-4.47
Manganese	0.10	0.05-0.16	0.16	0.06-0.23	0.20	0.05-0.41	0.19	0.06-0.37	0.32	0.06-1.04
Lead	0.02	Tr-0.03	0.02	Tr-0.03	0.02	Tr-0.04	0.02	Tr-0.03	0.02	Tr-0.04

Note: Values in mg/L unless otherwise indicated

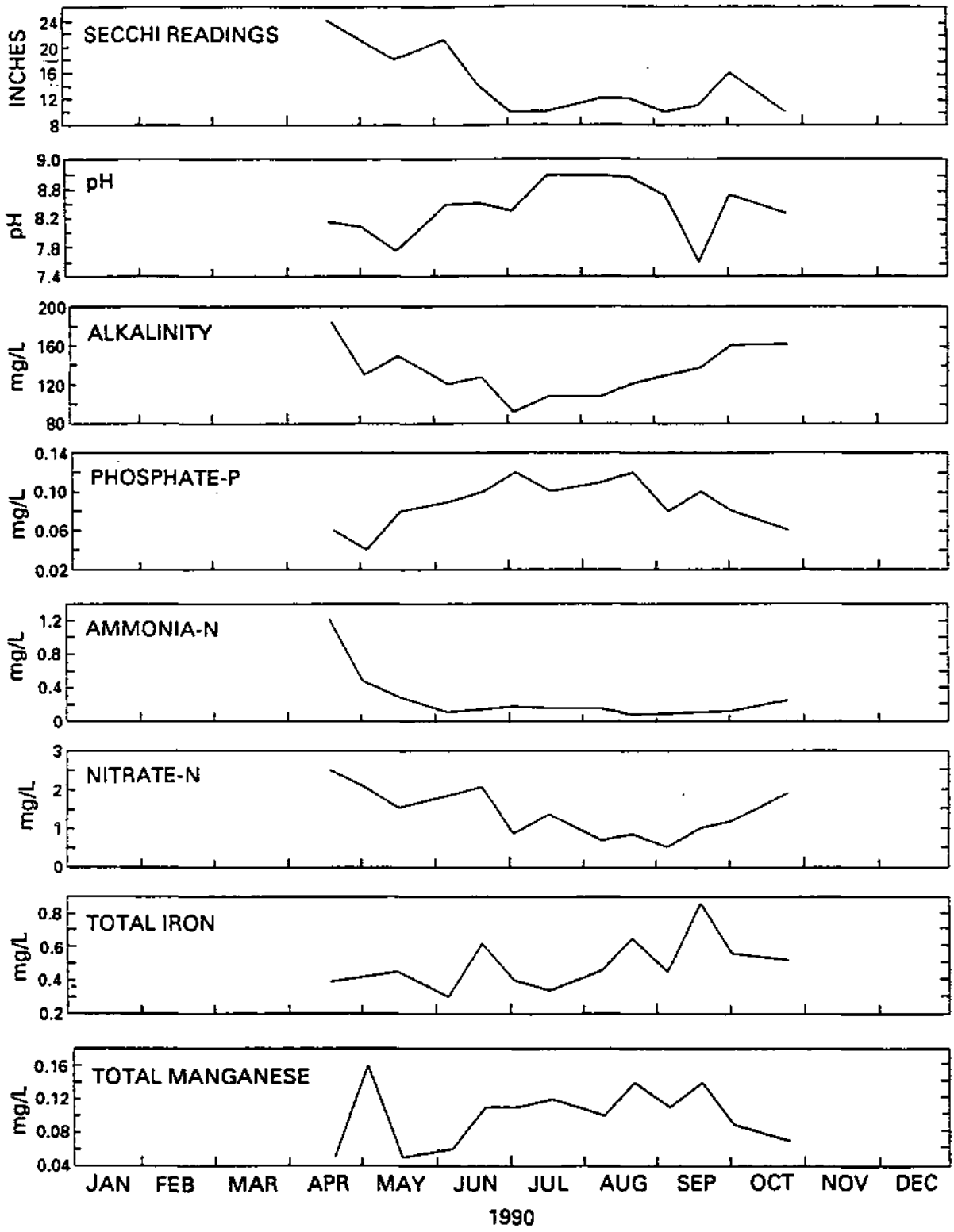


Figure 3.8. Temporal variations in water quality characteristics at station 1

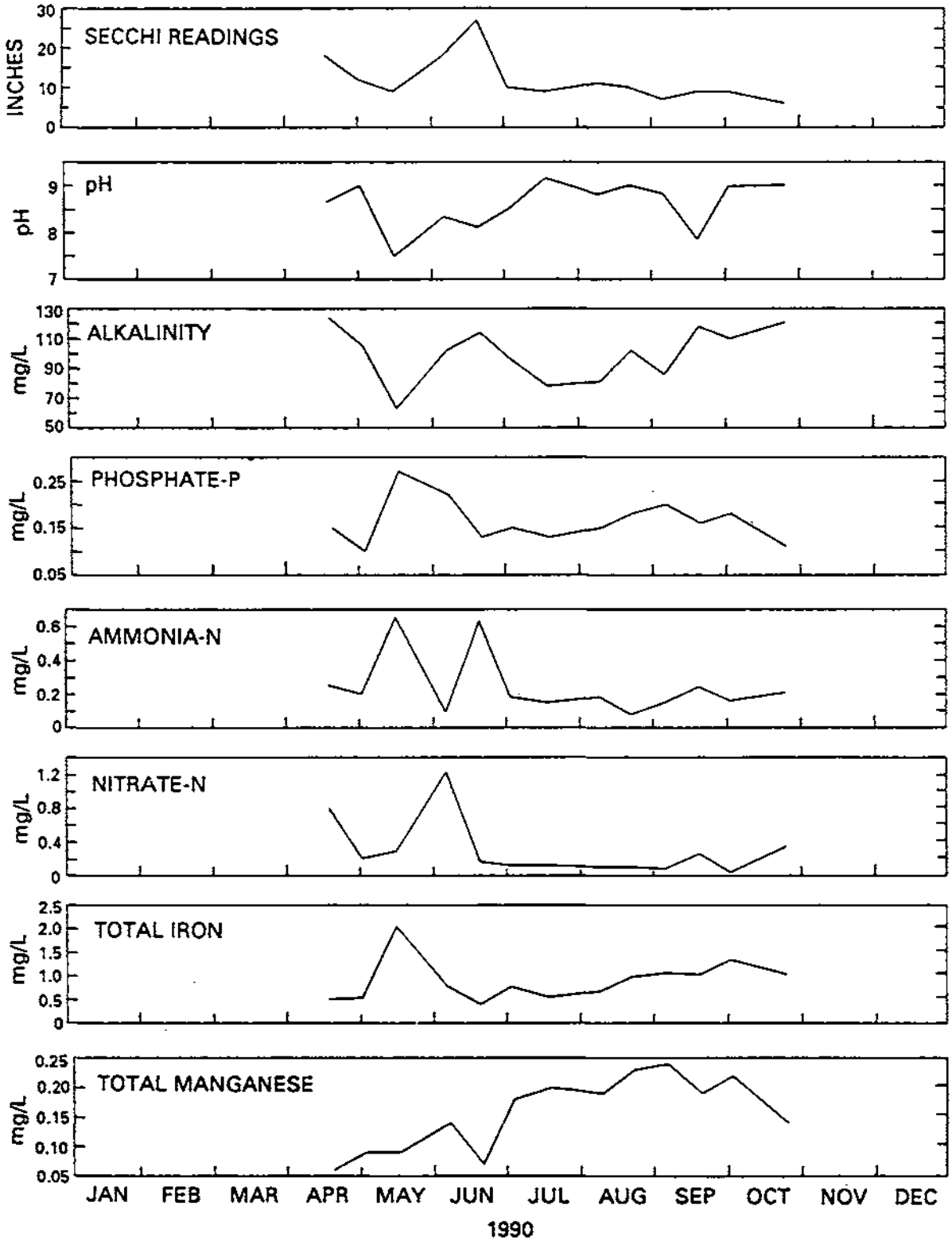


Figure 3.9. Temporal variations in water quality characteristics at station 2

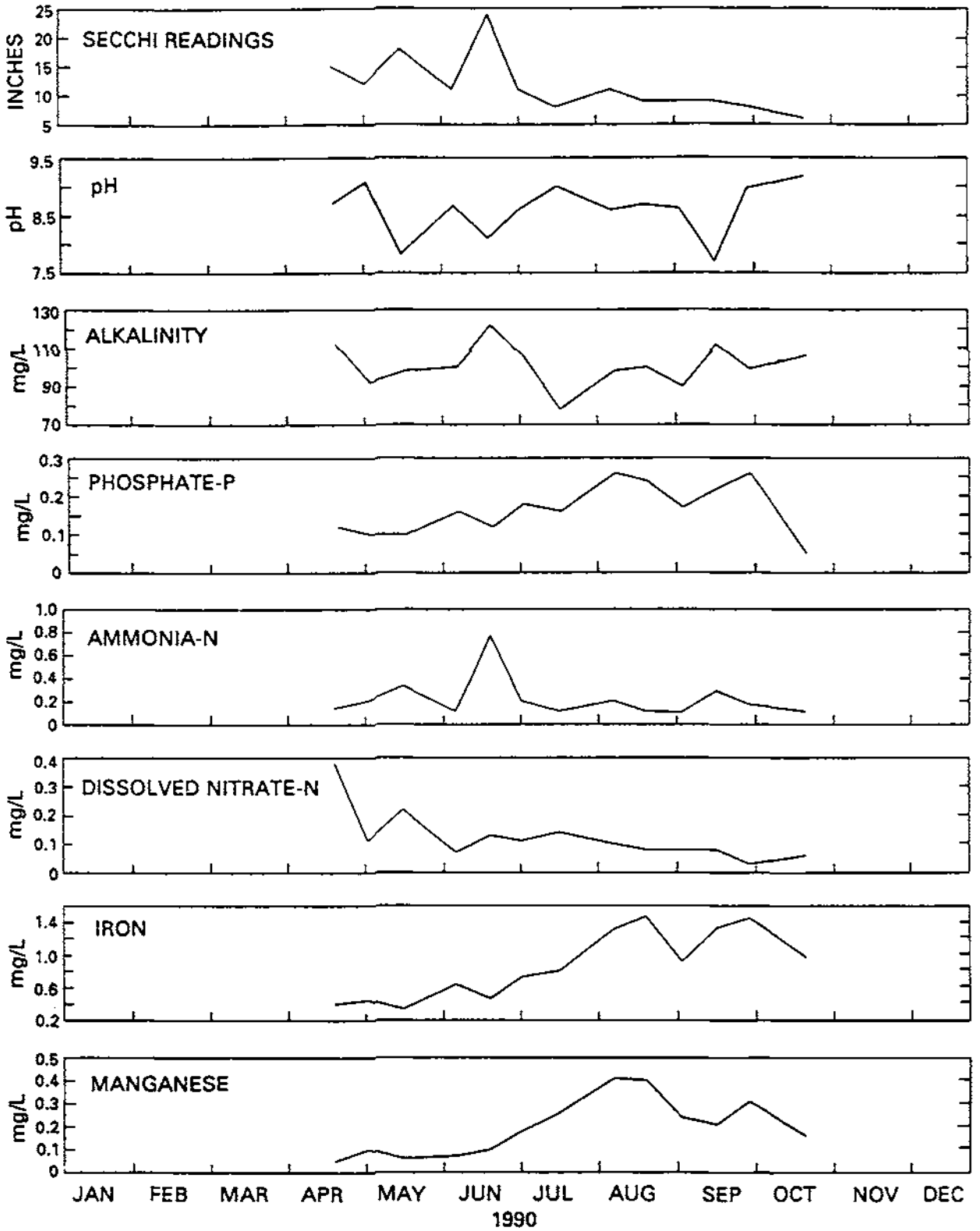


Figure 3.10. Temporal variations in water quality characteristics at station 3

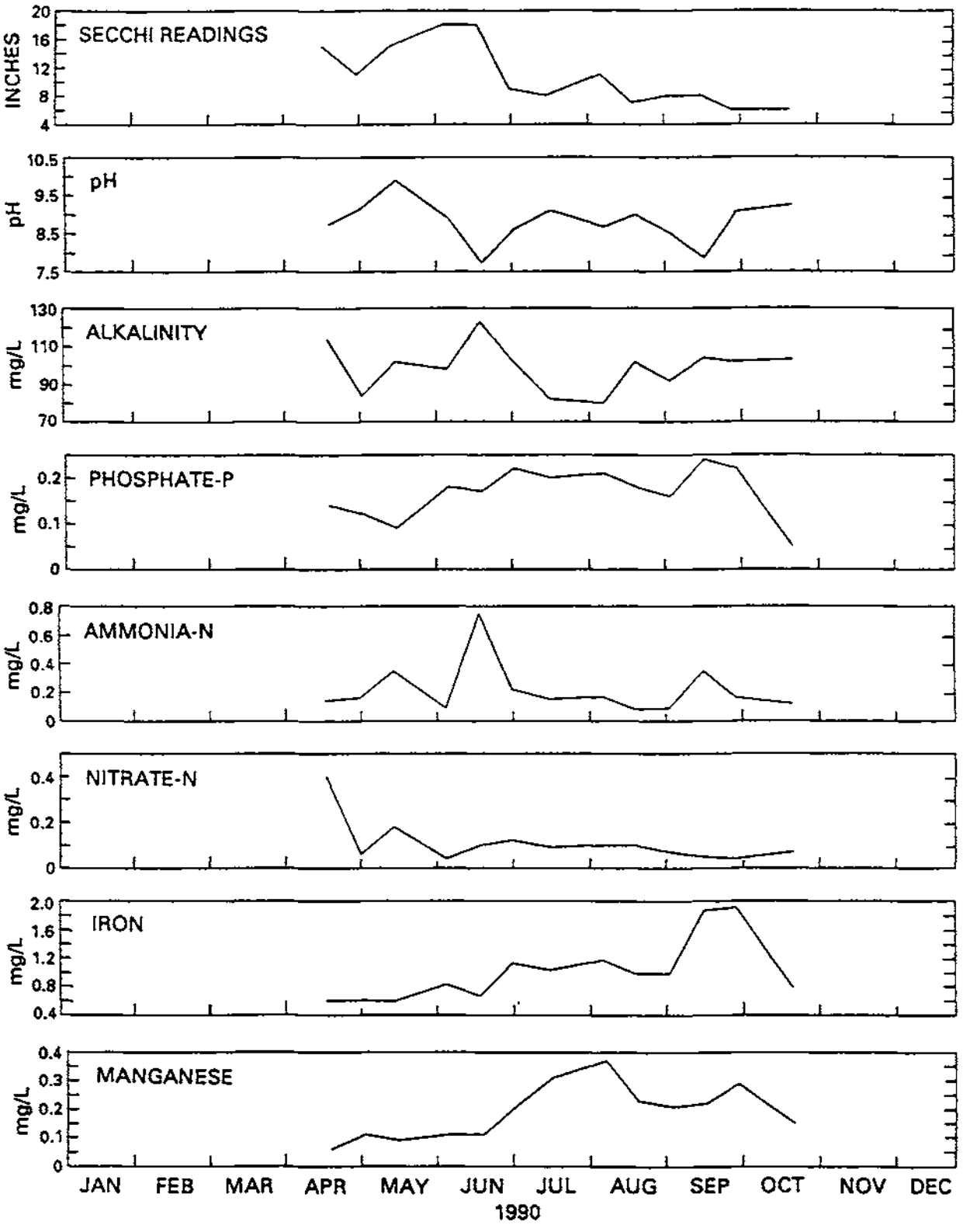


Figure 3.11. Temporal variations in water quality characteristics at station 4

developments; lake sediments stirred by wind, waves, and boating activities in shallow lakes; and detrital remains of algae and aquatic and terrestrial plants and animals.

Turbidity was lowest at station 1 compared with all other lake stations and Nameoki Ditch, which had comparable mean values (table 3.3). The highest turbidity value, 149 nephelometric turbidity units (NTU), was observed in the Nameoki Ditch sample collected on May 15, 1990, during an intense rainstorm resulting in 3.2 inches of precipitation.

Dissolved. Suspended, and Volatile Solids. In natural waters, the dissolved solids consist mainly of carbonates, sulfates, chlorides, phosphates, and nitrates of calcium, magnesium, sodium, and potassium with traces of iron, manganese, and other substances. The constituent composition of these minerals is dependent on the geochemistry of the area and municipal and industrial waste discharges.

Greeson (1971) observed that high dissolved solids content of Oneida Lake (New York) in 1967 and 1969 accompanied the high production of algae. In 1968, low dissolved solids content accompanied lesser algal production. Greeson concluded that these relationships indicate that the dissolved solids content is an important index of potential productivity conditions, because no element, ion, or compound is likely to be a limiting factor in algal production when the dissolved solids content is high.

The amount of suspended solids found in impounded waters is small compared with the amount found in streams because solids tend to settle to the bottom of lakes. However, in shallow lakes, this aspect is greatly modified by wind and wave action and by the type and intensity of use to which these lakes are subjected.

Suspended solids were generally lower at station 1 than at the other three lake stations. However, even though the dissolved solids levels at station 1 were higher because of the impact of industrial discharge, they were still much below the limits stipulated for general use standards by IPCB. The mean of suspended sediments values for Nameoki Ditch is low, because stagnant water was found on several occasions at the sampling site and there was no flow. However, suspended sediments were high on May 16, and June 6, 1990, when samples were collected during or soon after precipitation events. Volatile fractions of the suspended sediments in stations 1-4 and Nameoki Ditch were 42, 43, 45, 41, and 29 percent, respectively.

pH and Alkalinity. pH values above 8.0 in natural waters are produced by a photosynthetic rate that demands more carbon dioxide (CO_2) than the quantities furnished by respiration and decomposition. Photosynthesis by aquatic plants uses CO_2, removing it from bicarbonate when no free CO_2 exists in the water column. Bicarbonate, carbonate, and hydroxide components generally give water its alkalinity or capacity to accept protons. Active photosynthesis results in a decrease in bicarbonate alkalinity and an increase in the pH, whereas decomposition and respiration tend to reduce pH and increase bicarbonates.

Figures 3.8-3.11 clearly bring out the inverse relationship between pH and alkalinity values at the lake stations. During summer months, pH values were higher than 8.0, indicating algal photosynthesis and thus corroborating the DO supersaturation values observed in the lake.

pH values in Nameoki Ditch did not exceed 8.0. Mean alkalinity values in the lake were 137 mg/L at station 1 and differed within a very narrow range from 106 to 110 mg/L at the other stations.

Conductivity. Specific conductance provides a measure of a water's capacity to convey electric current and is used to estimate dissolved mineral quality of the water. This property is related to the total concentration of ionized substances in water and the water temperature at which the measurement was made.

Conductivity was highest at station 1 and differed negligibly among the other stations, which is also true with respect to the dissolved solids. The conductivity value for Nameoki Ditch was nearly half that for station 1, with a similar trend for dissolved solids.

Sulfates. Sulfates occur naturally in water as a result of teachings from gypsum and other common minerals. Sulfates may be discharged in numerous industrial wastes such as those from tanneries, pulp mills, and other plants that use sulfates or sulfuric acid. Lead, barium, and strontium sulfates are relatively insoluble, whereas sodium, potassium, and aluminum sulfates are highly soluble.

The mean sulfate values in the lake stations ranged from 133 mg/L to 99 mg/L, with the station 1 value being the highest. The mean sulfate value for Nameoki Ditch was 55 mg/L. The sulfate values closely followed trends for dissolved solids and conductivity values for all the monitoring stations. All observed sulfate values were much below the IPCB's general use standard of 500 mg/L.

Phosphorus. Because phosphorus is essential to the plant growth process, it has become the focus of attention in the entire eutrophication issue. With phosphorus the most limiting nutrient and the one most easily controlled by removal techniques, various facets of phosphorus chemistry and biology have been extensively studied in the natural environment. To prevent biological nuisance, the IPCB (1990) stipulates "Phosphorus as P shall not exceed 0.05 mg/L in any reservoir or lake with a surface area of 8.1 hectares (20 acres) or more or in any stream at the point where it enters any such reservoir or lake."

From his experience with Wisconsin lakes, Sawyer (1952) concluded that aquatic blooms are likely to develop in lakes during summer months when concentrations of inorganic nitrogen and inorganic phosphorus exceed 0.3 and 0.01 mg/L, respectively. These critical levels for nitrogen and phosphorus concentrations have been accepted and widely quoted in scientific literature.

Table 3.3 summarizes the observations for total phosphate-P for the four lake stations and Nameoki Ditch. Figures 3.8-3.10 depict temporal variations in phosphorus content in the lake. The mean total phosphorus levels in the lake varied from 0.09 to 0.17 mg/L; in the tributary, 0.31 mg/L. These levels are much higher than the state's general use water standards.

Nitrogen. Nitrogen in natural waters is generally found in the form of nitrate, organic nitrogen, and ammonia nitrogen. Nitrates are the end product of the aerobic stabilization of

organic nitrogen; as such, they occur in polluted waters that have undergone self-purification or aerobic treatment processes. Ammonia nitrogen, a constitutent of the complex nitrogen cycle, results from the decomposition of nitrogenous organic matter. Ammonia nitrogen can also result from municipal and industrial discharges to streams and rivers. Inorganic nitrogen concentrations in excess of 0.3 mg/L are considered sufficient to stimulate nuisance algal blooms (Sawyer, 1952). The EPCB stipulates that ammonia-N shall in no case exceed 15 mg/L, and if it is less than 15 mg/L and greater than 1.5 mg/L, then the unionized ammonia-N shall not exceed 0.04 mg/L. Unionized ammonia is a factor dependent on pH and temperature of the waters.

Table 3.3 includes the mean and range of values for total and dissolved ammonia-N and dissolved nitrate-N for the lake stations and Nameoki Ditch. Figures 3.8-3.11 show the temporal variations in these parameters for the lake stations. Dissolved ammonia-N and dissolved nitrate-N are the readily available forms of nitrogen for bioassimilation. Ammonia-N levels in the lake stations and Nameoki Ditch were within the stipulated standards. However, the readily available forms of nitrogen were much higher than the criteria with respect to algal blooms suggested by Sawyer (1952) in lake stations 1 and 2 and in Nameoki Ditch.

Metals. Table 3.3 shows the mean concentrations and range of values for iron, manganese, and lead for the lake stations and Nameoki Ditch. Because the analyses for arsenic and mercury for all the monitoring stations were below detection limits, they are not included in the table. Figures 3.8-3.11 show the temporal variations in iron and manganese for the lake stations.

All observed lead and manganese concentrations were within the IPCB standards except on August 22, 1990, for manganese in Nameoki Ditch when it was 1.04 mg/L as opposed to the 1.0 mg/L general use standard. For iron, 4 of the 13 observations exceeded the 1.0 mg/L standard at lake stations 2 and 3; 5 of the 13 observations for station 4; and 9 of the 13 observations for Nameoki Ditch. Surprisingly, observed iron concentrations were all within the limits for station 1, which is opposite the GCS wastewater treatment plant effluent discharge.

Diel Observations. Table 3.4 summarizes the diel observations made for water temperature, DO, pH, and conductivity at lake stations 1,2, and 4 using Data Sonde I units. As indicated earlier, the observations were recorded at half-hour intervals during the period May 3, 1990, 1300 hours to May 4, 1990, 1300 hours. Even though the diel observations were made during the spring when the temperatures are moderate, DO and pH values observed in the lake stations clearly indicate high algal activity with DO supersaturated conditions. The algal respiration process did not tend to depress the DO levels significantly during nighttime. The lowest DO level observed in the lake was 9.3 mg/L during this period. Conductivity values were much higher at station 1 compared with the other two lake stations.

Algae. Tables 3.5-3.9 give the total algal counts and the species distribution of algae found in the lake and in Nameoki Ditch. Algal densities in the lake were always of bloom proportion except for one observation at station 3 on May 16, 1990. Algal densities greater than 500 counts/mL are considered growths of bloom proportion (Fruh, 1967). Algal densities in Nameoki Ditch were not high even during periods of stagnation.

Table 3.4. Diel Observations in Horseshoe Lake
(May 3, 1990, 1300 hours to May 4, 1990, 1300 hours)

	Water temperature		*Dissolved oxygen*		*pH*		*Conductivity*	
Station	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
1	17.0	16.5-18.1	12.0	11.0-13.2	7.8	7.7-8.1	1296	1199-1341
2	16.0	15.8-17.3	10.9	9.3-12.4	8.4	8.1-8.6	938	873-1013
4	15.2	14.2-18.3	12.2	9.8-14.4	8.8	8.5-9.0	866	842-888

Note: Water temperature = ° C, Dissolved oxygen = mg/L, pH = units, and conductivity = μ mho/cm

Table 3.5. Algal Types and Densities at Station 1 in Horseshoe Lake, 1990

	Algal species	Dates 4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/17	10/2	10/25	Number of times found
BG:	*Agmenellum sp.*								42				95		2
	Anabaena spiroides					1134	74	21			116				4
	Anacystis thermalis						21					116			2
	Aphanizomenon flos-aquae	1376	620			924	116	32	347	231	84	462	231	105	11
	Lynebyra contorta														
	L. digneti	1250	2951	8726		2793									4
	Oscillatoria angustissima												168		1
	O. chlorina		1166					74					200		3
	O. sp.	641		147	76		74			74		137			6
	Spirulina major														
	S.sp.													347	1
G:	*Actinastrum hantzschii*	2951													1
	Coelastrum microporum					126			21						2
	Crucigenia rectangularis	84	158	168		74	168	84	294	662	347	210		168	11
	Micactum pusillum														
	Oocystis borgei	137				63		21	32				95	32	6
	Pediastrum duplex		116	21			53	11							4
	P. simplex														
	P. tetras														
	Scenedesmus avadricausla														
	S. carinatus					1155									1
	S. dimorphus	767	903	63	46		32		63			95	147	74	9
	Tetraedron sp.														
	Ulotrix variabilis														
D:	*Asterionella formosa*				254	137									2
	Caloneis amphisbaena														
	C. bacillum														
	Coconeis placentula														
	Cyclotella michiganiana	1376				977			431					147	4
	C. meneghiniana		1197												1
	C ocellata														
	Cymatopelura solea					21									1
	Cytnbella prostata														
	Diatoma														
	D. vulgara					74									1
	Diploneis smithii														
	Gomphomema olivaceum														
	Gyrosigma macrum												11		1
	Mastoglosia braunii									63	84			21	3

Table 3.5. Concluded

	Algal species	Dales 4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/17	10/2	10/25	Number of times found
D:	*Melosira ambigua*						53								1
	M. binderana														
	M. granulata		189			1386	95	32	200						5
	Navicula cryptocephala			200											1
	N. cuspidata														
	N. giacilei														
	Neidium productum														
	Nitzschia sigmoidea														
	Pediastrum duplex														
	P. tetras														
	Stauroneis phoenicentron														
	Stephanodiscus niagarae														
	Surviella ovata														
	Synedra acus	113													1
	S. delicatissima	287	3318		403	1764	284,812	259,875	19,803	9356	10,175	8810	12,212	4305	12
	S. ulna	61													1
F:	*Dinobryon sertularia*	63		42											2
	Euglena acus				2								21	32	3
	E. gracilis							11		147				42	3
	E. oxyurus	53					11								2
	E. viridis	231	105	53	8										4
	Glenadinium sp.					11									1
	Phacus pleuronectes		32		8	63		11						32	5
	Tracheolomonas crebea	137	74	32		21	21		53					74	7
	T. herrida														
De:	*Staurastrum comutum*														
Total		13,902	10,826	9450	798	10,721	285,527	260,180	21,284	10,532	10,805	9828	13,178	5376	
Types		15	12	9	7	16	12	10	10	6	5	6	9	12	

Note: Density in counts per milliliter: BG = blue-greens, G = greens, D = diatoms, F = flagellates, and De = desmids

Table 3.6. Algal Types and Densities at Station 2 in Horseshoe Lake, 1990

	Algal species	Dates												Number of times found
		4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/7	10/2	10/25	
BG:	*Agmenellum sp.*												305		1
	Anabaena spiroides	294						32					116	42	4
	Anacystis thermalis			116					95		305		189		4
	Aphanizomenon flos-aquae	2268	851	147	46	1376	84	63			179				8
	Lyngbyra contorta											1187	1428		2
	L. aimed	3287	5208	4232											3
	Oscillatoria angustissima														
	O. chlorina		630			147									2
	O. sp.	809		221			32	11	21			200		116	7
	Spinilina major														
	S. sp.													305	1
G:	*Actinastrum hantzschii*	3297	189	63					11					116	5
	Coelastrum microporum			32					21	84	32				4
	Crucigenia rectangularis		63	84	63	95	189	95	347	294		200	200	641	11
	Micaclum pusillum					32									1
	Oocystis borgei	147	63		13	63	32	21		74	95				8
	Pediastrum duplex	105						11						146	3
	P. simplex					74							21	53	3
	P. tetras														
	Scenedesmus avadricausla														
	S. carinatus										105				1
	S. dimorphus	672	326	95	11	1229	42	11	95	168		105			10
	Tetraedron sp.									21					1
	Ulotrix variabilis					200									1
D:	*Asterionella formosa*														
	Caloneis amphisbaena														
	C. bacillum														
	Coconeis placentula														
	Cyclotella michiganiana							462	315						2
	C. meneghiniana	1061											158		2
	C. ocellata							95							1
	Cymatopelura solea					21							11		2
	Cymbella prostata			21											1
	Diatoma														
	D. vulgara														
	Diploneis smithii														
	Comphomema olivaceum														
	Gyrosigma macrum														
	Mastoglosia braunii			21			53	11	11				21		5

Table 3.6. Concluded

Algal species	Dates 4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/17	10/2	10/25	Number of times found
D: *Melosira ambigua*														
M. binderana														
M. granulata	1061	452	273	46	1533	137	200	378	294	168		157	210	12
Navicula cryptocephala									662	315			326	3
N. cuspidata						11								1
N. giacilei														
Neidium productum														
Nitzschia sigmoidea													42	1
Pediastrum duplex					189									1
P. tetras					11									1
Stauroneis phoenicentron														
Siephanodiscus niagarae														
Surviella ovata												11		1°
Synedra acus			137	2										2
S. delicatissima	3287	4862		521		6395	23,625	17,210	11,876	6258	8495		1691	10
S. ulna														
F: *Dinobryon sertularia*														
Euglena acus			11									11	11	3
E. gracilis			63	11	105					63	158	21	63	7
E. oxyurus		21												1
E. viridis	420													1
Glenadinium sp.														
Phacus pleuronectes							11					11		2
Tracheolomonas crebea			32		53	21	21			189		21	42	7
T. herrida														
De: *Staurastrum cornutum*				2									. 11	2
Total	16,496	12,663	5544	714	5124	6993	24,665	18,501	13,472	7707	10,343	3707	3812	
Types		12	10	15	9	14	10	14	10	8		6	15	

Note: Density in counts per milliliter: BG = blue-greens, G = greens, D = diatoms, F = flagellates, and De = desmids

Table 3.7. Algal Types and Densities at Station 3 in Horseshoe Lake, 1990

	Algal species	Dates 4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	0/5	9/17	10/2	10/25	Number of times found
BG:	*Agmenellum sp.*							305	21						2
	Anabaena spiroides								42			305	1439		3
	Anacystis thermalis			74					116		137				3
	Aphanizomenon flos-aquae	431	1082		61	1082	38	116	147						7
	Lyngbyra contorta												2027		1
	L. aimed	4715	4158	3213		2331				221					5
	Oscillatoria angustissima														
	O. chlorina														
	O. sp.	420	452	74		630		42				126			6
	Spirulina major														
	S. sp.											399		305	2
G:	*Actinastrum hantzschii*	357	326	63				11						116	5
	Coelastrum microporum	53			4	32			32						4
	Crucigenia rectangularis			53		95	50	168	294	389	273	1061	147	672	10
	Micactum pusillum														
	Oocystis borgei		21			231		95	105	126	147	179		74	8
	Pediastrum duplex	116	126		8	168		21						21	6
	P. simplex														
	P. tetras														
	Scenedesmus avadricausla													21	1
	S. carinaius														
	S. dimorphus	525	588	42	11	462	23		137	210	147		116	63	11
	Tetraedron sp.														
	Ulolrix variabilis	1239													1
D:	*Asterionella formosa*				50										1
	Caloneis amphisbaena														
	C. bacillum													11	1
	Coconeis placentula														
	Cyclotella michiganiana					294		34	389						3
	C. meneghiniana														
	C. ocellata														
	Cymatopelura solea	11					2					11			3
	Cymbella prostata														
	Diatoma														
	D. vulgara														
	Diploneis smithii														
	Gomphomema olivaceum														
	Gyrosigma macrum														
	Mastoglosia braunii			21	4			21			32	95		32	6

Table 3.7. Concluded

Algal species	Dales 4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/7	10/2	10/25	Number of times found
D: *Melosira ambigua*									105					1
M. binderana														
M. granulata		189		17	567	105	147	231	200	168	116	168	294	11
Navicula cryptocephala														
N. cuspidata		53							74					2
N. giacilei														
Neidium productum														
Nitzschia sigmoidea														
Pediastrum duplex														
P. tetras														
Stauroneis phoenicentron														
Stephanodiscus niagarae	200													1
Surviella ovata											21			1
Synedra acus	389	221		4										3
S. delicatissima	38	1862			3161	1069	24,812	11,876	10,091	8100	6437	6332	1691	11
S. ulna														
F: *Dinobryon sertularia*														
Euglena acus			32					11	11		11			4
E gracilis		189	74	13	74	15			42		126	11	147	9
E. oxyurus		11				8								2
E viridis	189	42					32			21				4
Glenadinium sp.														
Phacus pleuronectes	21							11	11		32	21	11	6
Tracheolomonas crebea	42	63		4		6				42		21	63	7
T. herrida														
De: *Staurastrum comutum*														
Total	11,918	12,380	3644	176	9125	1317	26,114	13,409	11,477	9062	8915	10,280	3518	
Types	15	15	9	10	12	9	12	13	11	9	13	9	14	

Note: Density in counts per milliliter: BG = blue-greens, G = greens, D = diatoms, F = flagellates, and De = desmids

Table 3.8. Algal Types and Densities at Station 4 in Horseshoe Lake, 1990

	Algal Species	Dates												Number of times found
		4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/17	10/2	10/25	
BG:	*Agmenellum sp.*							483	63						2
	Anabaena spiroides										231		1292	326	3
	Anacystis thermalis											116			1
	Aphanizomenon flos-aquae	620	1019		86	1201	132	116							6
	Lyngbyra contorta											242	693		2
	L. aimed	3255	3350	4074	1103	3791									5
	Oscillatoria angustissima												420		1
	O. chlorina														
	O. sp.	557		735		1061	57	105	74		137	326			8
	Spirulina major											378			1
	S. sp.														
G:	*Actinastrum hantzschii*	693	609	116									74	95	5
	Coelastrum microporum		42		4				32						3
	Crucigenia rectangularis	158	95			189		168	231	147	431		200	977	9
	Micactum pusillum														
	Oocystis borgei	74				326			189	32	63		32	147	7
	Pediastrum duplex		116					42	11						3
	P. simplex			42											1
	P. tetras								11						1
	Scenedesmus avadricausla														
	S. carinatus											126			1
	S. dimorphus	389	746	63	101	630	8			221		210	95		9
	Tetraedron sp.														
	Ulotrix variabilis	1376													1
D:	*Asterionella formosa*														
	Caloneis amphisbaena						4	11							2
	C. bacillum														
	Coconeis placentula														
	Cyclotella michiganiana						36	410			147			273	4
	C. meneghiniana														
	C. ocellata														
	Cymatopelura solea						2								1
	Cymbella prostata														
	Diatoma														
	D. vulgara								21						1
	Diploneis smithii														
	Gomphomema olivaceum														
	Gyrosigma macrum														
	Mastoglosia braunii				8		4	147	11	11		116	116	63	8

Table 3.8. Concluded

Algal species	Dates 4/18	5/2	5/3	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/17	10/2	10/25	Number of times found
D: *Melosira ambigua*														
M. binderana									168					1
M. granulata	179	273			1166	103							326	5
Navicula cryptocephala												242		1
N. cuspidata	21				21	4								3
N. giacilei														
Neidium productum											11			1
Nitzschia sigmoidea														
Pediastrum duplex														
P. tetras														
Stauroneis phoenicentron														
Stephanodiscus niagarae														
Surviella ovata														
Synedra oats								21						1
S. delicatissima	2426	3728			4526	1329	9188	11,960	8411	5355			2216	9
S. ulna														
F: *Dinobryon sertularia*														
Euglena acus			21			2	21				32	32		5
E. gracilis			168		116		63	21		74	32	84	.	7
E. oxyurus		32				4								2
E. viridis	179	74		15					126					4
Glenadinium sp.														
Phacus pleuronectes			11											1
Tracheolomonas crebea	63					13		11	21	32	84		53	7
T. herrida														
De: *Staurastrum cornutum*														
Total	9986	10,080	5229	1317	13,094	1699	10,752	12,653	8505	6468	1670	3276	4473	
Types	13	11	8	6	10	13	11	13	8	8	11	11	9	

Note: Density in counts per milliliter: BG = blue-greens, G = greens, D = diatoms, F = flagellates, and De = desmids

Table 3.9. Algal Types and Densities in Nameoki Ditch, 1990

	Dates												Number of
Algal species	*4/18*	*5/2*	*5/16*	*6/6*	*6/20*	*7/3*	*7/18*	*8/22*	*9/5*	*9/17*	*10/2*	*10/25*	*times found*
BG: *Agmenellum sp.*													
Anabaena spiroides												32	1
Anacystis thermalis													
Aphanizomenon flos-aquae		42		46	19		48						4
Lynebyra contorta													
L. dignet$_i$									61				1
Oscillatoria angustissima													
O. chlorina													
O. sp.				21									1
Spirulina major													
S. sp.											57	44	2
G: *Actinastrum hantzschii*												11	1
Coelastrum microporum									13				1
Crucigenia rectangularis					59				63			109	3
Micactum pusillum													
Oocystis borgei								15					1
Pediastrum duplex													
P. simplex													
P. tetras													
Scenedesmus avadricausla													
S. cannulas													
S. dimorphus	23	42				8			15			13	5
Tetraedron sp.													
Ulotrix variabilis													
D: *Asterionella formosa*													
Caloneis amphisbaena													
C. bacillum													
Coconeis placentula						4							1
Cyclotella michiganiana													
C. meneghiniana													
C. ocellata													
Cymatopelura solea		2											1
Cymbella prostata													
Diatoma													
D. vulgara		6											1
Diploneis smithii								4					1
Gomphomema olivaceum	8	4											2
Gyrosigma macrum													
Mastoglosia braunii					4	8						8	3

Table 3.9. Concluded

	Algal species	Dales 4/18	5/2	5/16	6/6	6/20	7/3	7/18	8/22	9/5	9/17	10/2	10/25	Number of times found
D:	*Melosira ambigua*													
	M. binderana													
	M. granulata					48	92							2
	Navicula cryptocephala	15			74				99	92				4
	N. cuspidata													
	N. giacilei				4									1
	Neidium productum													
	Nitzschia sigmoidea													
	Pediastrum duplex													
	P. tetras													
	Stauroneis phoenicentron						4							1
	Stephanodiscus niagarae													
	Surviella ovata		15											1
	Syncdra acus						4							1
	S. delicatissima							239						1
	S. ulna													
F:	*Dinobryon sertularia*													
	Euglena acus											4		1
	E. gracilis		13			4								2
	E. oxyurus													
	E. viridis	36					11	13			11	8		5
	Glenadinium sp.													
	Phacus pleuronectes						6	4			4	6		4
	Tracheolomonas crebea				6		8			4	25	4	13	6
	T. herrida											4		1
De:	*Staurastrum cornutum*													
Total		82	86		158	134	147	305	118	248	44	79	230	
Types		4	7	0	5	5	9	4	3	6	4	5	7	

Note: Density in counts per milliliter: BG = blue-greens, G = greens, D = diatoms, F = flagellates, and De = desmids

Aphanizomenon flos-aquae (**BG**), *Crucigenia rectangularis* (**G**), and *Synedra delicatissima* (**D**) were the most frequently found algal species at stations 1 and 2. At station 2 opposite Nameoki Ditch, *Scenedesmus dimorphus* (**G**) and *Melosira granulata* (**D**) were also found more frequently. *Synedra delicatissima,* a species usually found with other planktons, was the most ubiquitous in the lake, often at high densities. This diatom measures about 110 microns *(μ)* with a length-to-breadth ratio of 30:1.

The number of algal species found in the lake at any time varied from 4 to 15 and is more diversified than at other eutrophic lakes monitored by the Water Survey. No one group of algae (i.e., blue-greens, greens, diatoms, or flagellates) was always dominant in the lake. Densities of blue-green algae were relatively too low to cause unsightly surface algal scums during summer months as is the case in most deep eutrophic lakes in Illinois.

Benthic Organisms. Table 3.10 gives the types and densities of benthic macroinvertebrate communities in the lake sediments. Macroinvertebrate densities found in stations 1,3, and 4 were comparable. Surprisingly, the density at station 2 near the confluence of Nameoki Ditch was very low compared with the other stations. Since this sampling site is quite far from the mouth of Nameoki Ditch, low density could not be due to the scouring effect of the stormflows in the tributary. Generally, the macroinvertebrate types and densities in the lake are comparable with those found in other Illinois lakes with mud bottoms.

Fisheries. According to Hill et al. (1981a), the sport fishery in Horseshoe Lake is marginal. The lake sustains a commercial fishery consisting mainly of carp, buffalo, channel catfish, and freshwater drum. They reported that annual yields varied from 12,000 to 88,700 pounds between 1957 and 1973 and 16,000 pounds between 1974 and 1977. In the past, largemouth bass and bluegill were significant in the sport fishery, but a population census conducted in 1977 by IDOC indicated that the reproduction and growth rates of these two species were so poor that they are no longer considered a fishery.

The fish survey conducted by IDOC on June 13, 1989 (Marbut, 1990) indicated that good populations of bigmouth buffalo and freshwater drum were present in the lake, contributing to the limited commercial fishery. Due to poor reproduction, the number of largemouth bass collected, 24 catch per unit effort, was much below the desired number, 68. No fish (bass) less than 8 inches was collected. Because of the low recruitment into the fishery, this population would not support even moderate angling pressure. Bluegill fishery had the most to offer an angler: 16 percent of the 352 bluegill collected in the survey were between 6.0 and 6.8 inches and in good condition. The numbers of channel catfish and white crappie collected were small, indicating poor reproduction. Gizzard shad was the best population in the lake for desirable numbers and length frequency. About 65 percent of the total 1,192 fish collected were smaller than 2.8 inches, constituting a forage base for mid-sized predators. No mid-sized shad between 3.0 and 6.0 inches were collected, but good adult breeding population existed, of which 25 percent were larger than 7 inches.

Trophic State. Eutrophication is a natural process that affects every body of water from the time of its formation. As a lake ages, the degree of enrichment by nutrient materials increases. In general, the lake traps a portion of the nutrients originating in the surrounding

Table 3.10. Benthic Macroinvertebrates in Horseshoe Lake
(Collected on May 3, 1990)

Organisms	*Station* 1	2	3	4
Chironomidae (nonchironomus)	100	301	847	574
Chaoborus (phantom midge)	29		57	
Ceratopogonidae (biting midge)			29	
Oligocnaeta (aquatic worm)	1048		459	804
Total	1177	301.	1392	1378

Note: Data in numbers per square meter

drainage basin. In addition, precipitation, dry fallout, and ground-water inflow are the other contributing sources.

A wide variety of indices of trophic conditions has been suggested in the literature. Indices have been based on Secchi disc transparency, hypolimnetic oxygen depletion, nutrient concentrations, and biological parameters including species abundance and diversity. The U.S. Environmental Protection Agency (USEPA) (1980) suggests in its *Clean Lakes Program Guidance Manual* (table 3.10-4) tile use of four parameters as trophic indicators: Secchi disc transparency, and concentrations of carbon, phosphorus, and chlorophyll-a. The criteria suggested therein are reproduced here (table 3.11).

In addition, the lake trophic state index (TSI) system developed by Carlson (1977) on the basis of Secchi disc transparencies, chlorophyll-a, and total phosphate-P values can be used to evaluate a lake's trophic state. The index number can be calculated from Secchi disc transparency (SD) in meters (m), chlorophyll-a (CHL) in micrograms per liter (μ g/L), and total phosphorus (TP) in μ g/L as follows:

$$TSI\ (SD) = 60 - 14.41\ \ln\ (SD)$$

$$TSI\ (CHL) = 9.81\ \ln\ (CHL) + 30.6$$

$$TSI\ (TP) = 14.42\ \ln\ (TP) + 4.15$$

The numerical index ranges from 0 to 100. Lakes with TSI values less than 40 are classified as oligotrophic (nutrient-poor or relatively unproductive, biologically speaking), and those with TSI values greater than 50 are classified as eutrophic (highly productive). Each major division (10, 20, 30, etc.) represents a probability of a doubling biomass.

Table 3.12 gives the mean and range of values of TSI for the lake stations, using the observed SD and TP values. Based on the USEPA criteria and the TSI values for these water quality parameters, Horseshoe Lake is highly eutrophic. It is also noteworthy that lake station 1 opposite the industrial waste discharge exhibited the least signs of eutrophy (Secchi disc, total phosphate, algal density, etc.) among all four lake stations.

Environmental Science & Engineering, Inc. (1990) reported that the toxicity testing data indicated that no short-term acute mortality responses or long-term sublethal effects resulted from exposure to the GCS effluent. The biological community assessment of Horseshoe Lake indicated that the areas near GCS outfall supported community composition and densities of organisms similar to areas far from the discharge.

Nameoki Ditch. This tributary to Horseshoe Lake is a storm drain for Granite City's urban area. There was practically no flow during most of the field monitoring except during or soon after heavy rainstorms with suitable antecedent conditions. Figure 3.12 shows the precipitation in Belleville, Illinois (a nearby raingaging station) and the flow characteristics in Nameoki Ditch. The lowest and highest observed DO values were 2.0 and 12.1 mg/L (appendix H). The highest turbidity and suspended solids were noted during the most intense storm, which

Table 3.11. Quantitative Definitions of Lake Trophic State
(USEPA, 1980)

Characteristics	*Oligotrophy*	*Eutrophy*
Total phosphorus (winter), μg/l	≤10-15	≥ 20-30
Chlorophyll-a (summer), μg/l	<2-4	>6-10
Secchi disc depth (summer), m	≥ 3-5	≤1.5-2
Primary productivity		
Carbon ($mg/m^2/yr$)	30-100	300-3000
Carbon ($\mu g/m^2/day$)	7-25	75-700

Table 3.12. Trophic State Index Values for Lake Stations

	Station 1		*Station 2*		*Station 3*		*Station 4*	
Parameter	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
Secchi disc	75.1	67.1-79.7	78.4	65.0-87.1	78.6	67.1-87.1	79.9	71.3-87.1
Total phosphate-P	67.8	57.3-73.2	77.2	71.9-84.9	76.5	60.6-84.3	77.0	60.6-83.2

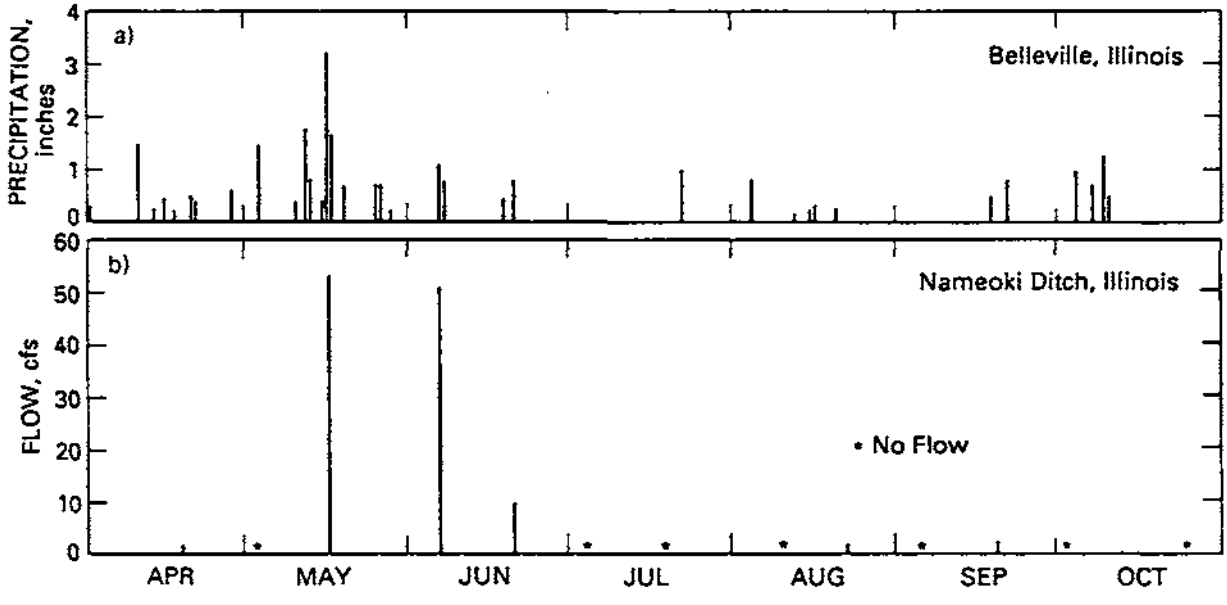


Figure 3.12. Precipitation for Belleville, Illinois, and flow pattern for Nameoki Ditch

occurred during this investigation on May 16, 1990. All the observed phosphorus values were in violation of the IPCB's general use standard. Iron values were also in violation 9 out of 13 times. Manganese excursion was noted only once. All other parameters were within stipulated state standards. The algal data (table 3.9) indicate that algal growth is not a water quality problem in this water course, even during periods of stagnation.

Impact Assessment

Controlled Discharge. To assess the changes in the DO and temperature of the ground water when discharged to a surface drain, a controlled discharge investigation was carried out on August 8, 1990, as indicated earlier. In addition to monitoring DO and water temperature, changes in pH and alkalinity were also determined at the site. Table 3.13 presents the results. Figure 3.13 depicts the spatial variations in velocity, DO, and water temperature for the three different discharge rates.

The DO and water temperature at station 1 near the pump discharge varied from 2.2 to 2.9 mg/L and 15.8° to 16.1° C, respectively, during the period from 9:30 a.m. to 3:30 p.m. of this field study when the ambient temperature varied from 21.0° to 25.8° C. The data reveal that the DO recovered from a low of 2.2 to 5.0 mg/L in a 5-minute period, flowing about 700 feet. The recovery of DO at the three discharge rates appears to reach an asymptotic value irrespective of the rate of discharge. However, it should be noted that the velocity of this channel flow is relatively high compared with stream channels.

The increase in water temperature was 0.6° C, but the initial water temperatures observed 20 feet from the pump discharge were different, reflecting the changes in ambient temperatures during the investigation. When the dewatering plan is implemented, the discharge through natural water courses will result in much longer travel time for the pumped discharge to reach Horseshoe Lake, other lakes, or impoundments, providing ample time for the DO and water temperatures to recover to levels dictated by the environmental conditions. The ground-water temperature will be stable around 16° C or 60.8° F. The initial oxygen deficits and the channel flow characteristics will be the driving forces in the recovery of oxygen conditions of the discharged waters. These prospects are greatly enhanced if the discharges are required to be controlled to meet the iron and manganese effluent standards.

Table 3.14 shows the striking changes in dissolved iron and dissolved total phosphate-P in the controlled discharge study. Values for pH, alkalinity, and dissolved manganese are also given. All these analyses were performed in the laboratory with the samples filtered at the site using 0.45 μm filters. The samples show a progressive decrease in the total dissolved iron with distance, presumably due to the oxidation of ferrous iron into insoluble ferric form by atmospheric oxygen. There was a striking change in total dissolved phosphate-P level, being reduced to zero from about 0.32 mg/L, probably due to the coprecipitation of phosphate with ferric iron. This change is an important phenomenon in view of the fact that Horseshoe Lake is highly eutrophic. Other parameters appear to be little affected.

Table 3.13. Changes in Water Quality Characteristics from Ground-Water Discharges into a Concrete Open Channel

Discharge (gpm)	*Station*	*Distance from discharge point (ft)*	*Velocity (fps)*	*Dissolved oxygen (mg/L)*	*Temperature (° C)*	*pH (units)*	*Alkalinity (mg/L as $CaCO_3$)*
600	1	20	-	2.9	15.8	7.2	489
	2	120	2.5	3.6	16.0		
	3	220	3.4	3.8	16.1		
	4	320	4.1	4.1	16.3		
	5	420	3.5	4.3	16.4		
	6	520	3.9	4.6	16.5		
	7	686	-	5.1	16.5	7.5	489
450	1	20	2.5	2.7	15.9	7.4	473
	2	120	2.2	3.0	16.1		
	3	220	3.0	3.3	16.2		
	4	320	3.9	3.8	16.3		
	5	420	3.4	4.0	16.4		
	6	520	3.7	4.4	16.4		
	7	686	3.9	5.0	16.5	7.5	501
300	1	20	0.3	2.2	16.1		
	2	120	1.9	2.6	16.1		
	3	220	2.5	3.0	16.1		
	4	320	3.6	3.5	16.3		
	5	420	3.1	3.9	16.4		
	6	520	3.5	4.4	16.5		
	7	686	3.4	5.0	16.7		

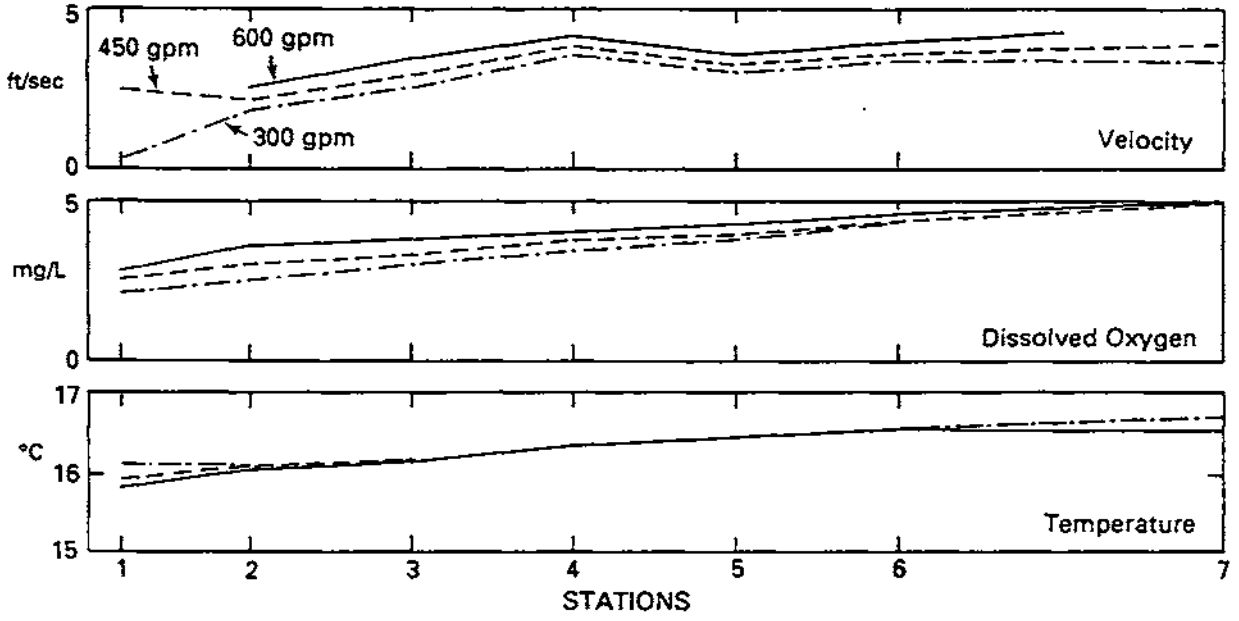


Figure 3.13. Spatial changes in velocity, dissolved oxygen, and temperature during a controlled discharge study

Table 3.14. Changes in Soluble Iron, Manganese, and Phosphate-P and Other Parameters in Controlled Discharge Study

Discharge (gpm)	*Station*	*Distance from discharge point (ft)*	*pH (units)*	*Alkalinity (mg/L as $CaCO_3$)*	*Dissolved iron (mg/L)*	*Dissolved manganese (mg/L)*	*Dissolved phosphate (mg/L)*
600	1	20	7.6	445	3.49	0.21	0.35
	2	120	7.5	443	3.44	0.26	0.23
	3	220	7.6	444	3.30	0.23	0.08
	4	320	7.5	447	3.06	0.23	0.02
	5	420	7.4	446	2.94	0.25	0.01
	6	520	7.5	442	2.76	0.23	Tr
	7	686	7.6	445	2.59	0.24	0.00
450	1	20	7.4	443	3.44	0.23	0.30
	2	120	7.6	446	3.35	0.23	0.18
	3	220	7.7	450	3.27	0.24	0.14
	4	320	7.6	447	3.12	0.23	0.02
	5	420	7.8	443	2.94	0.22	0.01
	6	520	7.8	446	2.67	0.22	0.01
	7	686	7.7	444	2.18	0.26	Tr
300	1	20	7.7	444	3.67	0.24	0.32
	2	120	7.6	446	3.50	0.25	0.22
	3	220	7.6	441	3.34	0.23	0.12
	4	320	7.8	446	3.13	0.22	0.03
	5	420	7.5	446	2.84	0.23	0.01
	6	520	7.6	442	2.67	0.21	0.01
	7	686	7.7	448	1.98	0.23	0.01

Crystal Lake. Urbana. Illinois. Table 3.15 gives the results of monitoring of the ground-water discharge into Crystal Lake, Urbana, Illinois. The length of the cobblestone-studded discharge channel is 125 feet. As seen from the table, the DO at station 1 was high initially because the discharge jet in the fountain shot high and splashed on the slanting fountain slab (figure 3.3). The water sample was collected at the lip of the fountain slab. In spite of the high initial DO, the DO also increased at station 2 before the ground water flowed into the lake. The temperature of the ground water did not exceed 15.3° C during summer months when the lake surface water temperatures would have hovered around 28° to 30° C. The dissolved iron concentrations decreased significantly between stations 1 and 2. Similarly, dissolved phosphate-P also showed a significant decrease. Total iron concentrations in pre-construction ground-water samples collected on May 24, 1984, and March 8, 1985, were 3.07 and 1.47 mg/L, respectively. Iron concentrations found in the American Bottoms area were much higher than these values. However, there was no change in the manganese levels. The fountain sloping slab, fountain periphery wall and well, and the discharge channel were discolored with rust stain (figure 3.3).

The lake volume is estimated as 21.5 million gallons (mil gal). The pumping rate is 150 gallons per minute (gpm) or 216,000 gallons per day (gpd). The volume displacement rate of the ground-water pumpage is approximately 100 days. During the planning period for the lake restoration, the primary concern of the regulatory agency (IEPA) for the discharge of ground water into the lake was the probable toxic effect of ammonia in ground water on the lake fish population (Wauthier, 1990), not the DO and water temperature aspects. During the six months of observations of the lake near the confluence of discharge stream and the lake, there were no visible signs of fish in distress. There were no fish kills observed or reported, and no adverse impact was noticed.

Lake Eureka. Eureka. Illinois. Lake Eureka, with a surface area of 36 acres and a volume of 227 acre-feet (ac-ft), serves as a water supply source to the village of Eureka, which has a population of 4300. The lake's watershed area is only 1700 acres. To augment the water resources of the water supply system, ground water is pumped from Sankoty aquifer, piped underground for about 3 miles, and discharged directly into the shallow head-end of the lake. The ground-water temperature in this region is about 13° C (55° F). The pumping capacity is 800 gpm. The pump is operated only during the nighttime to make use of the off-peak power rates.

The annual pumpage into the lake during the period from 1982 to 1989 varied from 17.39 to 187.5 mil gal, the highest rate occurring during the 1989 drought. The Water Survey has monitored the lake intensively from 1981 to 1983 and has maintained close contact with the village since then for proper water quality management in the lake. At the highest pumpage rate recorded to date, the volume displacement time due to ground-water addition alone is about 145 days. There were no adverse environmental impacts reported due to water temperature, DO, ammonia-N, or phosphate-P levels of the ground water. The water temperatures reported for the lake station near the ground-water discharge during 1982 were very similar to those for the deep station (2500 feet downstream). The maximum observed surface temperature near the ground-water discharge was 2° C cooler during July and August. There were practically no differences in the DO levels at these sites (Raman and Evans, 1984).

Table 3.15. Water Quality Changes in Crystal Lake, Urbana, Illinois, 1990

		Temperature			*Iron*		*Manganese*		*Phosphate-P*	
Date	*Station*	*Air (° C)*	*Water (° C)*	*DO mg/L)*	*Total (mg/L)*	*Dissolved (mg/L)*	*Total (mg/L)*	*Dissolved (mg/L)*	*Total (mg/L)*	*Dissolved (mg/L)*
5/23		19.7								
	1		13.1	8.8						
	2		12.9	9.6						
6/22		21.5								
	1		13.9	7.9		0.02		0.03		0.02
	2		14.3	9.4		0.02		0.03		0.03
7/31		24.0								
	1		13.3	7.5		0.02		0.06		0.04
	2		14.2	9.8		0.02		0.06		0.02
8/28		27.5								
	1		14.0	7.7		0.05		0.04		0.02
	2		15.3	10.2		0.03		0.04		0.03
9/27		22.0								
	1		13.3	8.7	1.96	0.11	0.04	0.04	0.17	0.02
	2		13.6	9.7	1.97	0.06	0.04	0.04	0.16	0.03
10/19		8.0								
	1		12.4	9.1	1.93	0.02	0.04	0.03	0.14	0.01
	2		12.1	10.5	1.69	0.04	0.03	0.03	0.11	0.02

Lake of the Woods. Mahomet. Illinois. Lake of the Woods serves primarily as a recreational resource and has a surface area of 22 acres and a capacity of 225 ac-ft. The diagnostic feasibility study of the lake and the subsequent implementation of the recommendations from 1983 to 1985 were sponsored and partially funded by USEPA under the 314 Clean Lakes Program. One of the recommendations for lake management was to install a deep well pump to flush and maintain lake water levels.

The ground-water input system, completed in summer 1985, consisted of a 300-foot deep well with a 600 gpm capacity, an aeration basin, two retention basins (which serve as water hazards for the surrounding golf course), and a tile drain to the lake. The primary concern of the regulatory agency (IEPA) was the nutrients (ammonia and phosphate-P) and the DO levels of the ground-water discharges to the lake. The aeration and retention basins of the ground-water treatment system were reported to be effective in increasing oxygen and decreasing concentrations of nutrients and metals in the well water (Sefton and Mitzelfelt, 1987). Prior consideration was not given to the impact of the ground-water temperatures on the receiving lake. However, no adverse impact due to the ground-water discharge into the lake was ever reported.

Horseshoe Lake. Madison County. Illinois. The ground-water quality of the test wells (ME03D and ME15D), given in the ground-water quality section of this report, are representative of the quality of 12 mgd of ground water proposed to be pumped under the P5AFREQ plan (U.S. Army Corps of Engineers, 1987). Ground-water qualities reported here conform to all the water quality standards for general use and effluent qualities of IPCB (1990) except for total iron and total phosphate-P concentrations. Concentrations of several metals of concern, such as arsenic, cadmium, chromium, copper, lead, and mercury, were either below detection levels or well below the requirements for general use or both. Restrictions on such constituents as arsenic, barium, cadmium, down to zinc for effluent discharges (IPCB, 1990) are more than adequately met, with the exception of iron. Concentrations of iron in these test wells were relatively high, nearly three to five times those observed in Crystal Lake and Lake Mahomet wells.

Water quality characteristics of Nameoki Ditch, a storm drain for Granite City and a tributary to Horseshoe Lake (appendix H) indicate that the water course is in total violation with respect to phosphorus as P, exceeding the limit of 0.05 mg/L at the point where it enters Horseshoe Lake. The general use standard for iron was exceeded 9 out of 13 times monitored. In spite of the above normal precipitation in the region during 1990, there was no measurable flow in 7 out of 14 site visits. The discharge twice exceeded 50 cubic feet per second (cfs). The discharge was once less than 2 cfs and twice in the range of 2 to 10 cfs. Steady discharge of 12 mgd (18.6 cfs) is well within the carrying capacity of the water course. It is also beneficial in establishing a stable population of aquatic organisms as opposed to the frequent, rapid changes of wet and dry conditions except at deep, stagnant pools.

The discharge of ground water directly to the surface drain will result in discoloration (due to iron precipitation) of the vegetation, streambanks, streambeds, and other man-made structures that come in contact with the ground-water discharge. The highway concrete open channel used in the controlled discharge study was completely brown after about 6 hours of

ground-water flow through it. The conveyance channels used in Lake of the Woods and Crystal Lake were also discolored (figure 3.3). However, in all these cases, iron concentrations were much less compared with those found in the American Bottoms area. As the predominantly dissolved form of iron in the ground water is oxidized to ferric form when exposed to the atmosphere, the oxidized iron will be mostly in colloidal suspended form, which will tend to increase the turbidity of the streamflow. Also, a steady source of iron in the water course could lead to the establishment and proliferation of iron bacteria, creating aesthetically objectionable conditions.

If the ground water is treated for iron removal prior to discharge to the surface drain, the discharge is likely to be nutrient poor as phosphate-P and iron are easily removed as coprecipitates. Based on the jar test studies, the supernatant after a short gravity settling period was found to be clear. Thus, the addition of 12 mgd clear, nutrient-poor waters to eutrophic Horseshoe Lake will have beneficial impact, enhancing the overall chemical and biological characteristics.

Dilution/flushing is one of the accepted in-lake management techniques (Wisconsin Department of Natural Resources, 1974; USEPA, 1973). This has been attempted primarily to alleviate excessive algal growths and associated problems by reducing nutrient levels within a lake. Nutrient-rich lake waters are replaced with nutrient-poor waters and the phytoplankton are washed out. Lake restoration projects have attempted nutrient dilution by two procedures: (1) pumping water out of the lake, thus permitting the increased inflow of nutrient-poor ground water and (2) routing additional quantities of nutrient-poor surface waters into the lake.

The addition of clear, nutrient-poor waters to Horseshoe Lake will result in the reduction of nutrient levels of lake waters, improvement in lake clarity, and a reduction in the frequency and magnitude of algal blooms currently experienced. Increased clarity and reduced lake turbidity will be conducive for sport fisheries that are primarily sight feeders. As mentioned earlier, sport fisheries in Horseshoe Lake have declined in recent years.

Nameoki Ditch, downstream of State Highway 162, winds its way through a network of marshes adjoining (on the north) the GCS wastewater treatment lagoons. It is likely that the ground-water discharges would have traversed more than a mile and would have had ample time to reach near-ambient temperature conditions prior to reaching the lake. Also, the ground-water temperature in the American Bottoms area is about 60° F. Based on actual observations of ground-water discharges in three Illinois lakes, it can be predicted that temperature perturbances would have no deleterious impact on Horseshoe Lake.

Likewise, the ground-water discharge will have ample opportunity to recover from low DO conditions to adequate levels. These prospects are enhanced significantly if the iron removal aspects are incorporated in the project plan.

The estimated lake volume is 6450 ac-ft. The addition of ground water at the rate of 12 mgd, considered alone, will result in a volume-displacement rate of 178 days. This water renewal rate (in days) is longer than that for Crystal Lake or Lake Eureka. Consequently, it should have a lesser impact, if any, on Horseshoe Lake compared with those lakes. It is also

important to recall that Horseshoe Lake's water quality near the industrial waste discharge was better than at other locations with respect to clarity, nutrients, and phytoplankton densities.

SUMMARY

Based on the criteria proposed in the USEPA's *Clean Lakes Program Guidance Manual* and on the Carlson's trophic state index values, detailed limnological investigation of Horseshoe Lake indicates that the lake is eutrophic. Water quality of the test wells in the Granite City area indicates that they meet all IPCB general use and effluent standards, with the exception of iron. Concentrations of iron in the test well samples were high compared with levels found in other parts of the state. Controlled ground-water discharge studies indicate that the DO in the ground-water discharges would quickly recover to acceptable levels. There was no detrimental impact reported for the three Illinois lakes that received ground water to augment water resources. With the treatment of ground water to meet effluent standards for total iron, ground-water quality in Granite City is superior to the lake water quality. If the treated water is diverted into the lake, the lake will benefit from the flushing action on algal growth, increased lake clarity, and probable improved sport fisheries. Alternately, GCS,a division of National Steel Corporation, may be willing to use the treated ground water as its process water, if it is delivered to its premises.

Section 4

REMOVAL OF IRON AND MANGANESE FROM GROUND WATERS

by Shun Dar Lin and Raman K. Raman

INTRODUCTION

Iron (Fe) and manganese (Mn) are abundant elements in the earth's crust. They are mostly in the oxidized state (ferric, Fe^{+3}, and Mn^{+4}) and are insoluble in natural waters. However, under reducing conditions (i.e., where dissolved oxygen is lacking and carbon dioxide content is high), appreciable amounts of iron and manganese may occur in ground water and in water from the anaerobic hypolimnion of stratified lakes and reservoirs. The reduced forms are soluble divalent ferrous (Fe^{+2}) and manganous (Mn^{+2}) ions or are chemically bound with organic matter. Iron and manganese get into natural water from dissolution of rocks and soils, from acid mine drainage, and from corrosion of metals. Typical iron concentrations in ground water are 1.0 to 10 mg/L, and typical concentrations in oxygenated surface waters are 0.05 to 0.2 mg/L. Manganese exists less frequently than iron and in smaller amounts. Typical manganese values in natural waters range from 0.1 to 1.0 mg/L (James M. Montgomery, Consulting Engineers, 1985). Voelker (1984) reported that iron and manganese levels in ground waters in the American Bottoms area of southwestern Illinois ranged from <0.01 to 82.0 mg/L and <0.01 to 4.70 mg/L, respectively, with mean concentrations of 8.4 mg/L and 0.56 mg/L, respectively.

Generally, iron and manganese in water are not a health risk. However, in public water supplies they may discolor water, stain plumbing fixtures and laundry, and cause tastes and odors. Iron and manganese may also cause problems in water distribution systems because metal depositions may result in pipe encrustation and may promote the growth of iron bacteria which may in turn cause tastes and odors. Iron and manganese may also cause difficulty in household ion exchange units by clogging and coating the exchange medium.

To eliminate the problems caused by iron and manganese, the U.S. Environmental Protection Agency (1987) has established secondary drinking water standards for iron at 0.3 mg/L and for manganese at 0.05 mg/L. The Illinois Pollution Control Board (IPCB, 1986) has set effluent standards of 2.0 mg/L for total iron and 1.0 mg/L for total manganese.

The techniques for removing iron and manganese from water are based on the oxidation of relatively soluble Fe(II) and Mn(H) to insoluble Fe(HJ) and Mn(III,IV) and the oxidation of any organic-complex compounds. This is followed by filtration to remove the Fe(UJ) and Mn(III,IV) precipitates, sometimes preceded by a sedimentation process. The detailed treatment processes for iron and manganese removal are discussed later.

IRON AND MANGANESE MANAGEMENT

Iron and manganese in water can be managed by treatment techniques for removal, sequestration, and *in situ* source control. Many methods exist for iron and manganese removal. The four major techniques are: 1) oxidation-precipitation-filtration, 2) manganese zeolite process, 3) lime softening-settling-filtration, and 4) ion exchange. Aeration-filtration, chlorination-filtration, and manganese zeolite process are the processes most commonly used in the public water supply industry. Only these three processes and the use of strong oxidants will be discussed.

Oxidation. Precipitation, and Filtration

The most common treatment method is oxidation of soluble (reduced) iron and manganese by oxidants such as dissolved oxygen (air aeration), chlorine, or potassium permanganate ($KMnO_4$), followed by removal of the insoluble precipitates by sedimentation and filtration processes. For economical reasons, aeration-filtration and chlorine-filtration processes are very popular. For waters with high organic content, a strong oxidant such as $KMnO_4$, chlorine dioxide (ClO_2), ozone (O_3), or hydrogen peroxide (H_2O_2) is required (Knocke et al., 1987, 1988; Lorenz et al., 1988).

Aeration. In the water supply industry, aeration for iron and manganese removal usually can be achieved by cascades, coke trays, sprays, or other mechanical aeration devices followed by sedimentation and filtration. Aeration is kinetically slow and very dependent on pH and alkalinity. Oxygen may be introduced by aeration, which will strip carbon dioxide from the water and result in a raise in the pH. Aeration has been successful in oxidizing ferrous iron at a pH of greater than 6.5 (Lorenz et al., 1988). Iron removal by aeration is rather slow when the pH is lower than 6.5.

The oxygen reaction can be expressed as follows (Jobin and Ghosh, 1972):

$$Fe^{++} + 1/4\ O_2 + 2OH^- + 1/2\ H_2O \rightarrow Fe(OH)_3 \qquad (1)$$

Alkalinity has an effect on the oxygenation rate of ferrous iron.

Oxygenation of manganous manganese can be expressed as follows (Dean, 1979):

$$M^{++} + O_2 \rightarrow MnO_2 + 2e \qquad (2)$$

In theory, 0.14 mg/L of oxygen is needed to oxidize 1 mg/L of iron and 0.29 mg/L of oxygen for each mg/L of manganese (Benefield et al., 1982).

It has been reported that high concentrations of silica can catalyze ferrous iron oxidation and retard hydrolysis of ferric iron, thus hindering sedimentation and filtration (Schenk and Weber, 1968; Robinson, Jr., 1974, 1975). A high concentration of silica gives a false (higher)

bicarbonate alkalinity reading than the actual. Therefore lime or soda ash should be added to negate the interference of silica to promote iron oxidation.

Manganese is more difficult to remove than iron. Manganese oxidation may be poor at a pH below 8.6 and is considered acceptable at a pH of 9.5 (Graveland and Heertjes, 1975). A detention time of several hours may be necessary after aeration. The major disadvantages of the aeration-filtration process are that the initial costs are high and that if the soluble manganese concentration is greater than 1 mg/L, additional retention time and supplemental chemical treatment may be required (Wong, 1984). Aeration is slower than other chemical methods and is ineffective when iron and manganese exist as complexes or chelates (Humphrey and Eikleberry, 1962; Ficek, 1978).

Chlorination. Many water treatment plants practice prechlorination with chlorine gas or hypochlorites for iron and manganese removal. The process is more rapid than aeration. Chlorine oxidizes iron at a pH of 7 to 10 and manganese at a pH greater than 8.5; therefore pH adjustment is usually necessary (Benefield et al., 1982). Sometimes a small retention tank and a pH adjustment system to feed soda ash, caustic soda, or lime are used. To achieve manganese removal to a level less than 0.05 mg/L using chlorine, a pH of 8 or greater is required. Oxidation of manganese by chlorine essentially ceases at water temperatures of less than 40° F (4.4° C) (Lorenz et al., 1988). Chlorination-filtration is often slow and incomplete. At pH 8.4 and above, the chlorination-filtration process effectively removes iron and manganese. The costs for oxidation in chlorination-filtration are less than those for potassium permanganate - manganese greens and filtration (Wong, 1984). Again, chlorination-filtration does not efficiently oxidize manganese or organically bond iron and manganese.

Chlorine, in either free or combined available form, will oxidize ferrous iron and manganous manganese to insoluble forms as follows (White, 1972):

$$2Fe(HCO_3) + Cl_2 + Ca(HCO_3)_2 \rightarrow 2Fe(OH)_3 + CaCl_2 + 6CO_2 \quad (3)$$

$$MnSO_4 + Cl_2 + 4NaOH \rightarrow MnO_2 + 2NaCl + Na_2SO_4 + 2H_2O \quad (4)$$

The stoichiometric amounts of chlorine required to oxidize each mg/L of iron and manganese are 0.62 mg/L and 1.3 mg/L, respectively. Oxidation by chlorine is slower than by other stronger oxidants and is often incomplete. In most cases, more than theoretical quantities are needed. Chlorine gas and hypochlorites must first react with water to form hypochlorous acid, the active oxidizing agent. The application of chlorine requires special equipment.

Care must be taken to avoid thrihalomethane (THM) formation when excess chlorine is added to water containing organic material. Many water treatment plants no longer prechlorinate their waters to minimize THM formation; instead, chlorine is added just prior to filtration. Unfortunately, this modification drastically reduces the contact time available for iron and manganese oxidation. Furthermore, many plants are replacing chlorine with other oxidants such as chlorine dioxide (ClO_2) or ozone. The iron and manganese removal efficiency of these alternative oxidants has not been well established.

Chlorine Dioxide. Chlorine dioxide is a strong oxidant that will rapidly oxidize soluble iron and manganese. Because of its high cost, it has seldom been used for oxidizing iron, although it is used to some extent for the oxidation of the last traces of soluble manganese (Welch, 1963).

In theory, 2.45 mg/L of chlorine dioxide is required for oxidation of each mg/L of soluble manganese, assuming that the reduction by-product is chlorite. Knocke et al. (1987) reported on laboratory studies showing that dosages of at least twice the stoichiometric quantity were needed to lower the soluble manganese level to <0.05 mg/L. They also claimed that chlorine dioxide was effective for manganese oxidation. Dosages of 1 to 1.5 mg ClO_2/L effectively oxidized a high level of soluble manganese (0.25 mg/L) from river water over a wide pH range, with total organic carbon (TOC) ≤2.5 mg/L. More than 3 mg ClO_2/L was required to reduce the soluble manganese level from 0.25 mg/L to 0.05 mg/L for water from a different river (TOC of 8-10 mg/L).

Ozonation. Ozone is one of the strongest oxidants used in the water industry for disinfection purposes. The oxidation potential of common oxidants relative to chlorine is as follows (Peroxidation Systems, 1990).

Fluoride	2.32
Hydroxyl radical	2.06
Ozone	1.52
Hydrogen peroxide	1.31
Potassium permanganate	1.24
Chlorine	1.00

Ozone can be very effective for iron and manganese removal. Because of its relatively high capital costs and operation and maintenance costs, the ozonation process is rarely employed for the primary purpose of oxidizing iron and manganese. Since ozone is effective in oxidizing trace toxic organic matter in water, pre-ozonation instead of pre-chlorination is becoming popular. In addition, many facilities are using ozone for disinfection purposes. Ozonation can be used for two purposes: disinfection and metal removal.

Laboratory studies by Knocke et al. (1988) showed that stronger oxidants such as ozone, chlorine dioxide, and potassium permanganate resulted in soluble manganese oxidation immediately upon addition to prefiltered water. Typically, 75 to 90 percent of the manganese applied to the filters in studies of these oxidants was in the form of colloidal manganese oxides.

Hydrogen Peroxide. Hydrogen peroxide (H_2O_2) is one of the strongest oxidants. It is a clear, colorless liquid that is waterlike in appearance and nonflammable. It has a characteristic pungent odor and is miscible with water in all proportions. Commercially available solutions are 35, 50, and 70 percent hydrogen peroxide. The 50 percent solution is the one most commonly used.

Solutions of 50 percent and less are classified as non-hazardous substances. They are considered safer to handle and store than chlorine gas. Hydrogen peroxide is stable when

properly stored; decomposition or rate of loss is minimal. In large containers, such as tanks, the loss rate is less than 1 percent per year at normal ambient temperatures; in small containers such as drums, the decomposition rate is less than 2 percent per year (FMC Corp., 1975). The hydrogen peroxide process produces no hazardous waste streams, gaseous emissions, or other secondary wastes.

Hydrogen peroxide is not used for iron and manganese removal in water supply, possibly because of its higher cost. Very limited information on this process is available in the literature.

Hydrogen peroxide oxidizes soluble iron to insoluble forms by either of two mechanisms as follows (Kreuz, 1962):

1. Direct oxidation:

$$2Fe^{++} + H_2O_2 + 2H^+ \rightarrow 2Fe^{+++} + 2H_2O \quad (5)$$

2. Decomposition to oxygen:

$$H_2O_2 \rightarrow H_2O + 1/2\ O_2$$

followed by oxidation:

$$2Fe^{++} + 1/2\ O_2 + H_2O \rightarrow 2Fe^{+++} + 2OH^- \quad (6)$$

Although direct oxidation is very fast, the brine components may rapidly decompose hydrogen peroxide to oxygen, and it is probable that both of the above reaction mechanisms are in operation during iron removal with hydrogen peroxide (Kreuz, 1962). In either case, two moles of ferrous iron are oxidized per mole of hydrogen peroxide, or 0.61 mg/L of 50 percent H_2O_2 oxidizes 1 mg/L of ferrous iron.

For oxidation of manganese by hydrogen peroxide, the following reaction applies (H.M. Castrantas, FMC Corporation, Princeton, NJ, personal communication, 1990):

$$Mn^{++} + H_2O_2 + 2H^+ + 2e \rightarrow Mn^{+++} + e + H_2O \quad (7)$$

To oxidize 1 mg/L of soluble manganese, 1.24 mg/L of 50 percent H_2O_2 is required.

Potassium Permanganate. Chlorination is not usually used as the only means for manganese oxidation because of slow kinetics. Instead, it is often used in conjunction with potassium permanganate ($KMnO_4$) addition for manganese oxidation to reduce chemical costs (Adams, 1960; Cleasby, 1975; Wong, 1984). If the point of pre-chlorination is moved to just before filtration to avoid THM formation, other oxidants such as $KMnO_4$ are usually used.

Potassium permanganate has long been used very effectively by the water industry for iron and manganese removal. Its chemical reaction is rapid and complete. $KMnO_4$ reactions can be shown as follows (Ficek, 1978):

$$3Fe^{2+} + MnO_4^- + 4H^+ \rightarrow MnO_2 + 3Fe^{3+} + 2H_2O \quad (8)$$

$$3Mn^{2+} + 2MnO_4^- + 2H_2O \rightarrow 5MnO_2 + 4H^+ \quad (9)$$

In a stoichiometric reaction, 0.94 mg/L of $KMnO_4$ is needed for each mg/L of soluble iron and 1.92 mg/L for each mg/L of soluble manganese.

In practice, a 1 to 4 percent solution of $KMnO_4$ is fed at the low-lift pump station or at the rapid-mix point. It should be totally consumed prior to filtration. However, the required $KMnO_4$ dosage is generally less than the theoretical values (Humphrey and Eikleberry, 1962; Wong, 1984). Secondary oxidation reactions occur as follows:

$$Fe^{2+} + MnO_2 \rightarrow Fe^{3+} + Mn_2O_3 \quad (10)$$

$$Mn^{2+} + MnO_2 \cdot 2H_2O \rightarrow Mn_2O_3 \cdot X(H_2O) \quad (11)$$

and $$2Mn^{2+} + MnO_2 \rightarrow Mn_3O_4 \cdot X(H_2O) \quad (12)$$

or $$MnO \cdot Mn_2O_3 \cdot X(H_2O) \quad (13)$$

Additional catalytic oxidation occurs at the surface of MnO_2 and Mn_2O_3. Jar tests are required to determine the optimum $KMnO_4$ dosage by the empirical method for each water source (Humphrey and Eikleberry, 1962).

Recently, Knocke et al. (1987) reported that for waters low in TOC, $KMnO_4$ efficiently oxidized manganese over the pH range of 5.5-10.0. The oxidation kinetic was rapid. Most manganese was oxidized within the first five minutes of contact. When TOC was <3 mg/L, the required $KMnO_4$ dosage was typically less than 10 to 20 percent of the theoretical value. For waters with high TOC levels, the required $KMnO_4$ dosage for manganese oxidation was well above the stoichiometric value. The range of pH required is 6 to 9, although Benefield et al. (1982) suggested that the pH should be above 10 to oxidize manganese by $KMnO_4$.

In addition, $KMnO_4$ readily reacts with hydrogen sulfide, cyanides, phenols, and other taste- and odor-producing compounds. There is no potential of THM formation with $KMnO_4$. However, $KMnO_4$ is more expensive than chlorine. Excess $KMnO_4$ can increase the soluble manganese concentrations and import a pink color to the water if not properly controlled.

On the basis of bench studies, O'Connor (1971) reported that iron removal by $KMnO_4$ could be improved by mixing. This implies that the aerator and reaction basin might be capable of oxidizing most of the iron in raw water. By feeding $KMnO_4$ after the iron had been oxidized, the $KMnO_4$ demand could be reduced. In a full-scale study, Curry and Reynolds (1983) showed

that the by-products of iron removal could effectively remove manganese from a ground-water source in Roxana, Illinois. The process included $KMnO_4$ feed, ferric hydroxide from the reaction basin as a primary coagulant, and a cationic polymer as a coagulant aid.

Filtration. Following oxidation, the oxidized iron and manganese can be removed by the settling-filtration or filtration-only processes. Filtration preceded by sedimentation can prevent excess headloss buildup across the filters. Direct filtration usually has a suspended solids limitation. For example, the upper limit of suspended solids is 5 mg/L at the Village of Castle Rock, Colorado (Lorenz et al., 1988). Filtration can be used with sand or dual media filters or with the manganese greensand (manganese zeolite) process.

Manganese Zeolite Process

In the manganese zeolite process, iron and manganese are oxidized to the insoluble form and filtered out, all in one unit, by a combination of sorption and oxidation. The filter medium can be manganese greensand, which is a purple-black granular material processed from glauconitic greens, and/or a synthetic formulated product. Both of these compositions are sodium compounds treated with a manganous solution to exchange manganese for sodium and then oxidized by $KMnO_4$ to an active manganese dioxide. The greensand grains in the filter become coated with the oxidation products. The oxidized form of greensand then adsorbs soluble iron and manganese, which are subsequently oxidized with $KMnO_4$. One advantage is that the greensand will adsorb the excess $KMnO_4$ and any discoloration of the water.

Regenerative-Batch Process. The regenerative-batch process uses manganese treated greensand as both the oxidant source and the filter medium. The manganese zeolite is made from $KMnO_4$-treated greensand zeolite. The chemical reactions can be expressed as follows (Humphrey and Eikleberry, 1962; Wilmarth, 1968):
Exchange:

$$NaZ + Mn^{2+} \rightarrow MnZ + 2Na^{+} \qquad (14)$$

Generation:

$$MnZ + KMnO_4 \rightarrow Z \cdot MnO_2 + K^{+} \qquad (15)$$

Degeneration:

$$Z \cdot MnO_2 + \begin{matrix} Fe^{2+} \\ Mn^{2+} \end{matrix} \rightarrow Z \cdot MnO_3 + \begin{matrix} Fe^{3+} \\ Mn^{3+} \\ Mn^{4+} \end{matrix} \qquad (16)$$

Regeneration:

$$Z \cdot MnO_3 + KMnO_4 \rightarrow Z \cdot MnO_2 \quad (17)$$

where NaZ is greensand zeolite and Z • MnO_2 is manganese zeolite. As the water passes through the mineral bed, the soluble iron and manganese are oxidized (degeneration). Regeneration is required after the manganese zeolite is exhausted.

One of the most serious problems with the regenerative-batch process is the possibility of soluble manganese leakage. In addition, excess amounts of $KMnO_4$ are wasted, and the process is not economical for water high in metal content. Manganese zeolite has an exchange capacity of 0.09 lb of iron or manganese per cubic foot of material, and the flow rate to the exchanger is usually 3.0 gpm/sq ft. Regeneration needs approximately 0.18 lb $KMnO_4$ per cubic foot of zeolite.

Continuous Process. For the continuous process, 1 ~ 4% $KMnO_4$ solution is continuously fed ahead of a filter containing anthracite bed (6-9 in. thick), manganese-treated greensand (24-30 in.), and gravel. The system takes full advantage of the higher oxidation potential of $KMnO_4$ as compared to manganese dioxide. In addition, the greensand can act as a buffer. The $KMnO_4$ oxidizes iron, manganese, and hydrogen sulfide to the insoluble state before the water reaches the manganese zeolite bed.

Greensand grain has a smaller effective size than silica sand used in filters and can result in comparatively higher head loss. Therefore a layer of anthracite is placed above the greensand to prolong filter runs by filtering out the precipitate. The upper layer of anthracite operates basically as a filter medium. When iron and manganese deposits build up, the system is backwashed like an ordinary sand filter. The manganese zeolite not only serves as a filter medium but also as a buffer to oxidize any residual soluble iron and manganese and to remove any excess unreacted $KMnO_4$. Thus a $KMnO_4$ demand test should be performed. The continuous system is recommended for waters where iron predominates, and the intermittent regeneration system is recommended for ground water where manganese predominates (Inversand Co., 1987).

Ion Exchange

Ion exchange resins are effective in removing small quantities of soluble iron and manganese. Precautions must be taken to prevent the water from being aerated prior to its reaching the ion exchange unit. Otherwise the soluble iron and manganese may be converted to the insoluble forms and be deposited on the resin, causing fouling. This will reduce the useful life of the resin. Also the resins are impractical to use in treating high concentrations of iron and manganese, because they are easily fouled by the oxide hydrate. It is difficult to remove the adsorbed iron and manganese, and the capital costs for ion exchange are relatively high.

For ferrous iron levels of 5 mg/L or less, ion exchange units are adequate for iron removal. Customarily used in areas with hardness problems, such units operate by exchanging sodium for iron to convert ferrous bicarbonate to sodium bicarbonate (Gass, 1977).

Lime Softening

The lime or lime-soda softening process removes iron and manganese along with other hardness-causing metals. Adjusting pH to 9.5 or above is an economically feasible and effective means of iron and manganese removal if lime or lime-soda softening is practiced. The softening process becomes more effective when preceded by aeration.

Sequestration

Sequestration is an attempt to sequester soluble iron and manganese by sequestering agents instead of by using a removal process. Polyphosphates or hexametaphosphate are sometimes used to keep iron and manganese in solution, preventing them from being oxidized and deposited in the treatment plant or in the distribution lines, and from coloring tap water. Generally, 1 to 5 mg/L of polyphosphate is required when the soluble iron concentration is below 2 to 3 mg/L (U.S. Department of the Army, 1952). The results of using sequestration are mixed. Some water plants are satisfied with polyphosphates, whereas others have discontinued their use.

The use of polyphosphates has some drawbacks. They revert to ineffective orthophosphates in hot water at temperatures above 150° F. In these cases, when soluble iron and manganese are oxidized to the insoluble forms in the distribution lines, the polyphosphates will not sequester those forms, and discoloration will occur. In addition, phosphates support microbiological growth in distribution systems.

Few research results on sequestration are available in the literature. Robinson et al. (1987) studied the effects of several ions for sequestering iron by sodium silicate and hypochlorite under laboratory conditions. They found that calcium greatly increased the silicate dosage required to maintain low turbidity and high iron filterability. Iron was stabilized for more than 10 days as SiO_2 at a 12 mg/L silicate dosage. Without calcium, however, iron precipitated after three days in the presence of 10 mg/L of $CaCO_3$. With 100 mg/L of $CaCO_3$, iron precipitated in less than a day.

Source Control

Source control uses an in-ground treatment system, such as the Vyredox method (Hatva et al., 1973; AWWA Conference, 1984; Zienkiewicz, 1985; Maogong, 1988). This method works by improving the conditions of the soil around the well and by removing iron and manganese from ground water at the source, in the ground, before the water is drawn to a well.

The Vyredox method is a chemical and biological contact oxidation process. After undesirable gases are removed, the oxygenated wastes are forced into an iron-rich aquifer through special diffusion wells that form a treatment radius of 15 to 30 m (50 to 100 ft). The oxygen-rich water maintains oxidizing conditions, thus preventing the reduction of ferric iron or the oxidization of soluble iron *in situ.* The oxygenated water causes precipitation of iron and

manganese. The ferric hydroxide film which has been deposited on the surface of the sand of the aquifer absorbs oxygen and iron from the water, and the catalytic reaction between oxygen and iron takes place on the surface of the sand grains, assisted by iron bacteria. This recharge is conducted every two or three weeks after the production well is shut down for one day. After recharge, water with low levels of iron and manganese can be pumped until five to ten times more water has been pumped than was recharged.

The Vyredox system has been used in Europe and other parts of the world since 1955 and has been marketed in the United States since the late 1970s (Zienkiewicz, 1985). Limited discussions of the process have been published. In Seekonk, Massachusetts, the Vyredox process was used to treat raw well water containing 6.2 mg/L of manganese to meet the standard of 0.05 mg/L (AWWA Conference, 1984). It reduced iron from 2.55 to 0.30 mg/L.

METHODS AND PROCEDURES

Samples

Ten sampling wells in the American Bottoms region were studied. Their locations are shown in figure 4.1. The wells were dug 80 to 100 feet deep, and well water samples were drawn from 60 to 70 feet deep by a pump with a generator. Each well was allowed to discharge for 10 minutes to flush stagnant water from the well, and water samples were collected after at least 10 minutes of pumping. Water samples were pumped directly into plastic bags placed in 20-gallon ice chests. Care was taken to minimize turbulence that might lead to premature aeration. The plastic bags were filled with well water and tied shut with very little or no air space. For each well, at least two water samples were collected on different dates.

Well water temperature was recorded at the site. Raw water samples for metal analyses were collected at the well head in two 50-mL glass vials (filtered and nonfiltered) and in 125-mL plastic bottles for turbidity, alkalinity, and hardness determinations. The water samples analyzed for metals were acidified with 2 drops of concentrated nitric acid. The samples for soluble metals were filtered through 0.45-μm cellulose membrane filters.

Experiments

Laboratory studies were carried out at the materials testing laboratory of the Illinois Department of Transportation in Collinsville immediately after sample collection. Water samples from each well were subjected to a jar test using 2-L rectangular reactors in a six-place jar-test apparatus. One reaction vessel was untreated and kept as a control, while the other reactors were dosed with various amounts of a chosen oxidant. At least two jar test runs were performed for each sample. The first run was used to assess the approximate range of the required dosage required to meet the total iron effluent standard of 2 mg/L. The second run was used to better define the required oxidant dosage. Except for control samples, all the jars were dosed with incremental doses of the chosen oxidant. Attempts were made to keep the sample temperature

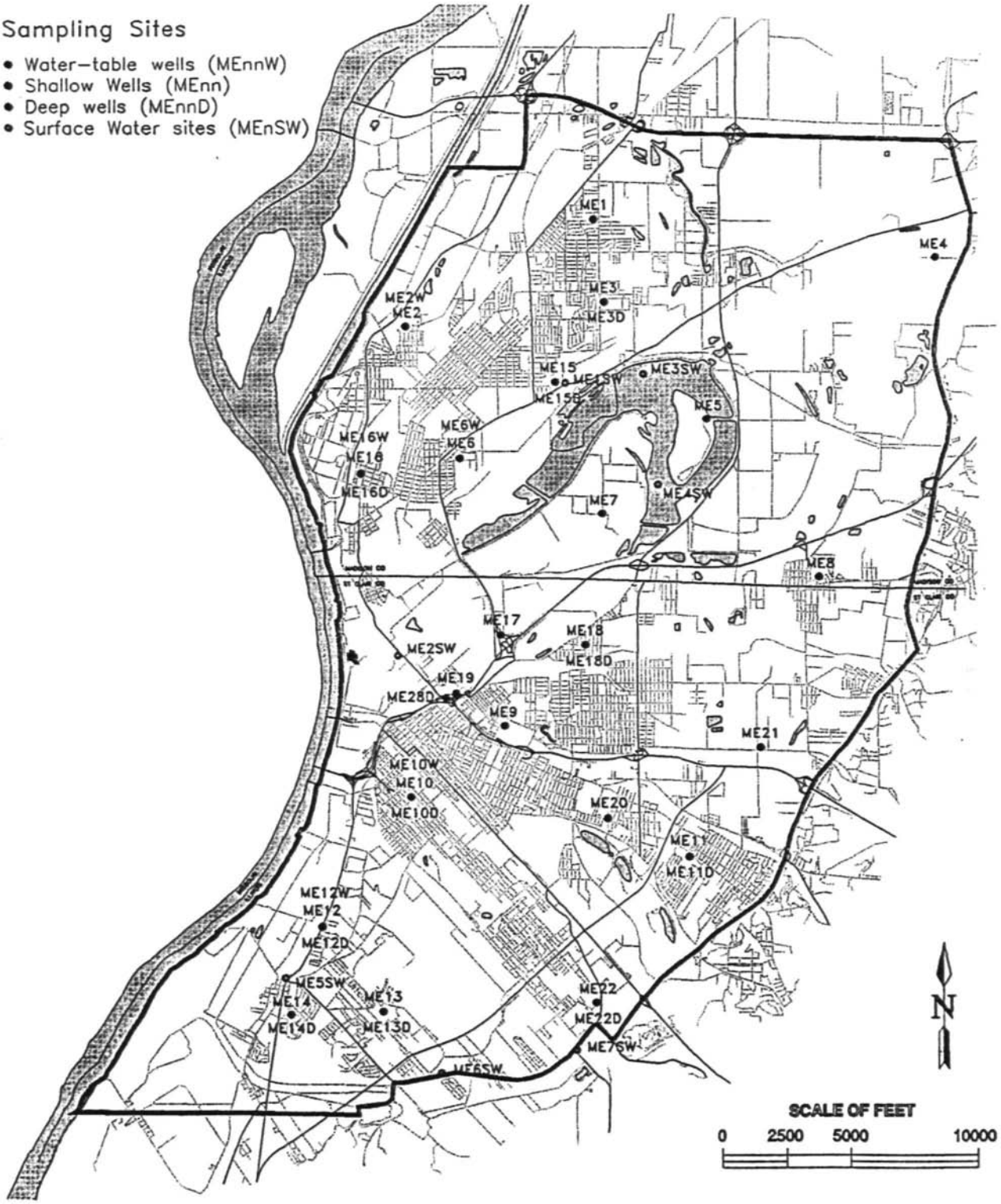


Figure 4.1. Locations of the sampling wells in the American Bottoms area

during the jar test as close as possible to the initial well water temperature. However, this was difficult, and there was generally a 2 to 3° C increase in the water temperature.

The oxidants used in this study were potassium permanganate, hydrogen peroxide, and chlorine (sodium hypochlorite, NaOCl). These chemicals are commonly used in water treatment facilities as oxidants. All oxidants were freshly prepared at the beginning of each jar test. The 0.2 percent working oxidant solutions were prepared with solid $KMnO_4$, a 30 percent H_2O_2 solution, and household bleach. Chlorine concentration was verified by amperometric titration. Appropriate chemical oxidant dosages were added to the samples before the rapid mixing at 100 revolutions per minute (rpm) for 2 minutes. The samples were then flocculated at 20 rpm for 20 minutes and allowed to settle for 60 minutes. Water temperature was recorded during the jar test period. For each of the reaction vessels, observations were recorded on time of appearance of first floc, color of solution after mixing, floc size, settling characteristics, color of supernatant, estimation of residue production, color of liquid in the Imhoff cone, and pH before and after settling. Supernatant was drawn from each vessel at the end of settling and/or at the desired time intervals for chemical analyses. Three subsamples of supernatant were collected each time for various laboratory analyses. The procedure used was similar to the procedure previously indicated for collecting raw water at the well head.

At the end of the jar tests, the supernatant was decanted off from each jar and the precipitates were transferred to Imhoff cones. The residues were allowed to settle quiescently for one hour. Then the volume of precipitates was noted to estimate the amount of solids generated in the iron/manganese removal process.

Analyses

Chemical analyses were performed for total and soluble iron, total and soluble manganese, turbidity, alkalinity, and hardness. The procedures followed were in accordance with *Standard Methods* (APHA et al., 1989). The iron and manganese concentrations were measured by the atomic absorption technique. Turbidity was determined in a 90° light-scattering nephelometer. The pH was measured on-site or in the laboratory with a portable pH meter.

RESULTS AND DISCUSSION

General Results

At least two water samples were obtained from each of the ten sampling wells for the jar tests. The results of the jar tests are listed in appendices I-1 through R-1 and I-2 through R-2. Each of the letters I through R represents one of the ten wells, and the numbers 1 and 2 represent the sample numbers. A third sample was collected and an extra experiment was performed for wells ME3D, ME15D, ME18D, and ME12D. These results are given in appendices I-3, J-3, L-3, and O-3, respectively. Table 4.1 lists the raw water quality characteristics of the ten wells studied.

Table 4.1. Well Water Quality Characteristics

Well no. ME D	Date, 1990	Temperature, ° C	pH	Alkalinity.* mglL as $CaCO_3$	Hardness.* mg/L as $CaCO_3$	Iron. mg/L Total	Iron. mg/L Soluble	Maneanese. mg/L Total	Maneanese. mg/L Soluble	Turbidity, NTU*
3	4/10	14.8	7.09	(363) 402	-	9.66	7.76	.25	.27	161
	6/28	15.6	7.10	(357) 388	433	9.66	8.45	.26	.24	153
	8/16	14.5	7.05	-	-	10.04	10.43	.31	.30	155
	10/16	15.5	-	-	-	9.65	9.35	.26	.24	126
15	4/9	14.7	7.20	(274) 281	322	5.19	4.93	.21	.22	84
	6/27	15.6	7.31	(275) 302	333	6.58	6.07	.27	.24	87
	8/16	14.5	7.02	335	-	6.63	6.85	.27	.26	108
16	4/17	14.5	7.03	528	-	18.12	-	.64	-	-
	7/11	15.5	7.19	(473) 525	747	18.96	19.40	.75	.77	59
18	4/16	15.6	6.82	(315) 392	1686	44.36	-	2.94	-	641
	7/2	16.2	6.70	(328) 414	1705	45.00	48.28	2.92	3.02	580
	8/15	15.5	6.90	-	-	52.48	48.88	3.12	2.86	430
	9/12	16.0	-	-	-	47.68	-	2.78	-	-
	9/13	15.5	-	-	-	48.56	-	3.16	-	-
	10/18	14.9	-	-	-	46.00	42.08	2.34	2.60	547
10	4/25	15.7	7.00	311	-	9.02	-	-	-	-
	7/18	16.0	7.07	(375) 392	477	10.16	10.44	.37	.31	158
11	5/14	15.4	7.02	258		10.72	9.86	.43	.39	-
	7/17	15.5	7.28	(240) 270	484	9.87	9.87	.27	.29	170
12	5/7	15.4	7.12	258		13.28	13.16	.91	.76	-
	7/3	15.8	7.12	(413) 441	539	14.68	13.20	.82	.99	215
	8/16	14.5	-	-	-	12.80	12.20	.90	.78	199
13	5/15	14.9	7.11	302	-	9.70	9.42	.37	.39	-
	8/15	15.0	7.15	(293) 345	370	10.93	10.35	.36	.46	134
	10/17	14.9	-	-	-	8.76	8.38	.29	.30	130
14	5/8	15.6	7.10	337	-	7.50	7.47	.45	.32	-
	7/10	15.0	7.32	(291) 315	398	7.46	7.43	.34	.33	109
22	4/24	15.7	6.95	351	-	13.52	-	.17	.17	-
	8/14	15.5	7.15	(328) 345	392	14.92	14.36	.27	.28	182
	10/17	14.9	-	-	-	12.92	12.68	.19	.18	191

* Measured a few days after collection and not immediately after withdrawal; values in parentheses were measured 1 hour after withdrawal.

The appendices mentioned above include data on water temperatures during the jar tests, various oxidant dosages, flocs, residual settlement in jars and in Imhoff cones, and water quality parameters such as pH before and after settling, alkalinity, hardness, turbidity, and total and soluble iron and manganese.

Temperature. An attempt was made to perform the jar tests at water temperatures as close as possible to ground-water temperature. However, since no constant-temperature water bath was used, all jar tests were conducted at water temperatures 1° to 3° C above ground-water temperatures.

Flocs and Colors. The first appearance of floc generally can be observed during rapid mixing (the first 2 minutes). The first flocs appeared within 10 to 20 seconds, and some appeared only after 3 to 4 minutes. The most common case was for flocs to appear within one-half to one minute of rapid mixing. These values are influenced by individual observer, color of liquid, floc size, and concentration of oxidant dosage. Each well sample was usually collected in the morning, and generally six jar test runs were carried out. The late runs were performed 7 or 8 hours after collection. During that time the water turned slightly yellow or yellowish-brown. For these cases, the first appearance of floc was extremely difficult to determine. Occasionally, it was necessary to slow down the mixing speed for a few seconds to detect flocs.

All ground waters were clear and colorless just after pumping. They were still clear at the beginning of the first jar test run or even at the beginning of the second jar test run. The color of water in the control jar changed from clear before mixing to milky after mixing. The water in the oxidant-dosed jars was yellow, amber, or brown. The higher the dosage, the darker the color. The color recorded was also dependent on the individual observer. After dosage with potassium permanganate, the waters were brown or dark brown in color and were pink if overdosed (in excess of $KMnO_4$ demand).

Actual sizes of flocs were not measured. However, the floc size in each reaction vessel was rated by visual estimation as small, medium, large, or extra large. Floc size grew bigger through the rapid mixing, flocculation, and settling process. In general, flocs generated by $KMnO_4$ were large in size and stronger in floc characteristics. Floc formed by H_2O_2 had the weakest floc strength. This could be observed during transfer from a 2-L square vessel to an Imhoff cone.

The settleability of flocs produced by $KMnO_4$, H_2O_2, or NaOCl was generally very good. Most of the flocs settled in one-half hour. In general, within 40 minutes a complete settling occurred and the liquid in the reactor vessel was clear.

Residues. The amount of residue produced by the iron and manganese removal processes was estimated. For each reactor vessel of the jar tests, the supernatant samples for chemical analyses were withdrawn from the drainports of the vessels after a 1-hour settling period. After the samples had been collected, the supernatant was drained out of the vessel by gravity until the flow stopped. Then the remainder (470 mL) in the vessel was carefully transferred to a 500-mL Imhoff cone. The quantity of residue was recorded after 1 hour of settling in the cone.

During the transfer, some flocs were broken up, and hindered settling was observed in the cones. Settling in the Imhoff cones was never complete. Approximately 5 to 20 percent of the precipitates was found either resuspended or stuck to the wall of the cone. The amounts reported in the appendices are thus underestimates. For design purposes, it is recommended that solids production be revised upwards to allow for this. The higher the oxidant dosage, the greater the amount of residue generated.

pH. For all jar test runs, the pH of the liquid ranged from 6.9 to 7.5. No pH adjustment was made during the normal jar tests. There was no significant change in pH values between the various oxidants dosed and the control, and no significant change in pH from before to after the sedimentation process.

Alkalinity and Hardness. In the Midwest, waters typically have high alkalinity and hardness. The ten wells studied at the American Bottoms had total alkalinity ranging from 240 to 528 mg/L as calcium carbonate ($CaCO_3$) and contained hardness from 322 to 1705 mg/L as $CaC0_3$ (table 4.1). Well ME16D had the highest alkalinity, and the highest hardness occurred at well ME18D. The alkalinity levels presented in table 4.1 were determined in the laboratory soon after sample collection. Alkalinity concentrations for raw well waters reported in the appendices were measured a few days after collection and found to be lower than the values reported in table 4.1. However, those values are provided for comparison purposes.

Equations 6 and 7 suggest that 0.9 mg/L of alkalinity as $CaCO_3$ will be required to oxidize 1 mg/L of iron; and for each mg/L of manganese oxidized, 3.4 mg/L of alkalinity as $CaCO_3$ is required (White, 1972). Not all samples were analyzed for alkalinity and hardness because these parameters were not found to be affected by the oxidants during the early jar test runs. Evaluation of the results indicated that alkalinity consumption due to iron and manganese oxidation was much less than the stoichiometric values.

Turbidity. The turbidity of the ground water was not determined immediately after pumping. Turbidity for the well ME13D sample collected on October 17, 1990, was 5.9 NTU, determined at the East St. Louis water treatment plant 20 minutes after collection. The turbidity of the same sample measured at the Water Survey laboratory in Peoria a few days after collection was 130 NTU.

The values of turbidity reported in the appendices and in table 4.1 were all obtained a few days after the jar tests. They are not precise values and are used for purposes of comparison only. The turbidity values for the raw waters and the control samples (with no oxidant added) can be taken as a measure of the turbidity of the ground waters after pumping if they are discharged without treatment. The turbidities for the control samples ranged from a low of 55 NTU at well ME16D (appendix M-1) to a high of 670 NTU at well ME18D (appendix L-2). The turbidities for the control jar test samples for the three oxidants tested were usually comparable except for the sample collected at well ME10D on April 25, 1990. The turbidities for the zero concentration jar tests with respect to $KMnO_4$, H_2O_2, and NaOCl were respectively 55, 80, and 122 NTU (appendix M-1). The reason for this was unknown.

The results of this study showed that good turbidity reductions were achieved with the use of any of the three oxidants. Most of the effluent turbidities ranged from 3 to 10 NTU.

Iron and Manganese. Samples from the ten deep wells studied were found to have high total iron concentrations ranging from 5.19 to 52.5 mg/L (table 4.1). Most iron was in the soluble form. The highest iron levels occurred at well ME18D in Fairmont City. In previous studies in the American Bottoms, Bruin and Smith (1953) found that iron content in 86 ground-water samples ranged from 0 to 120.0 mg/L. The highest iron concentrations were 120 mg/L at well 78 in Cahokia (73 feet deep), 94.8 mg/L in East St. Louis (38.5 feet deep), and 75.0 mg/L in National City (110 feet deep). All these wells are located 1 to 2 miles east of the Mississippi River. Wells ME10D, ME12D, and ME13D of this study are located in Cahokia (Well ME12D is close to well 78 of the study by Bruin and Smith), but all have total iron concentrations less than 11 mg/L.

As previously mentioned, Voelker (1984) found iron concentrations in the American Bottoms ranging from <0.01 to 82.0 mg/L. Recently, Shafer (1987) observed that iron concentrations for 39 wells in the American Bottoms were less than 17 mg/L.

As can be seen from table 4.1, the total manganese concentrations in water samples from nine of the ten wells were below the IPCB's 1.0 mg/L limit. Only well ME18D exceeded the effluent standards for manganese. Like iron, most manganese in the well samples was found to be in the soluble form. In a few cases, soluble manganese was determined to be higher than total manganese. The reason for this is unclear. It might have been due to analytical error or to the release of manganese and iron from the filter paper.

Removal of iron and manganese from the ground water by three oxidants, and estimations of residue production for each test, will be discussed separately.

Aeration

For ground waters collected from test wells ME16D, ME22D, and ME10D, 2 L of sample was aerated vigorously in a graduated cylinder for up to 3 hours. The aerated samples were analyzed for residual iron and manganese. Table 4.2 presents the results of aeration as well as the data for raw water and for the control samples in chemical oxidation jar tests. It can be seen from table 4.2 that aeration oxidized soluble iron and manganese. Manganese content was generally low in these ground waters. However, no visible flocs formed in the reactor vessel. From the laboratory results, it is surmised that the oxidized iron was primarily in the colloidal form. Although turbidity was not measured for the aerated samples, the turbidity of these samples was at the same levels as that of the controls. The aerated samples were also similar in color to the control samples. On the basis of color, turbidity, and total iron level after settling, aeration is considered to be ineffective for removing iron from the ground waters in the American Bottoms region.

According to the American Water Works Association (AWWA, 1969), when iron is present alone (with a minimum amount of manganese), simple aeration followed directly by

filtration may be effective in the production of an acceptable effluent. Under certain conditions, a 5-minute reaction period or less may be sufficient. However, when high concentrations of manganese are present, or when iron and manganese exist as organic complexes or chelates, air oxidation is generally ineffective (Ficek, 1978). In such cases, strong oxidants, such as chlorine, chlorine dioxide, or potassium permanganate, are employed in conjunction with aeration (AWWA, 1971).

Iron and Manganese Removal by Chemical Oxidants

The effects of oxidant addition on both total and soluble iron and manganese are presented in the appendices. In figures 4.2-4.11, only data for the removal of total iron are depicted. From these figures, the oxidant concentrations required for reducing total iron concentration to the IPCB's standard of 2.0 mg/L were determined. They are shown in table 4.3. The residues generated from the oxidation/clarification processes at these critical oxidant concentrations are estimated from the jar test data and are also listed in table 4.3. The highest values (those in parentheses in table 4.3), rather than averages, were selected for design purposes.

Table 4.4 presents data on the quality of the raw well waters, collected as part of a comprehensive sampling program in 1990. These data are provided for reference purposes.

During the course of this study, oxidant addition was seldom greater than oxidant demand. It was easy to notice when $KMnO_4$ dosage exceeded the $KMnO_4$ demand. In general, there was no residual oxidant after rapid mixing. Results of removal of iron and manganese for each test well are discussed individually.

Well ME3D. The total and soluble iron concentrations in the four ground-water samples taken from well ME3D ranged from 9.65 to 10.04 mg/L and 7.76 to 10.43 mg/L, respectively (table 4.1). Approximately 80 to 100 percent of the iron was in the soluble form. Total manganese levels were low: between 0.25 and 0.31 mg/L. Manganese was primarily in the soluble form.

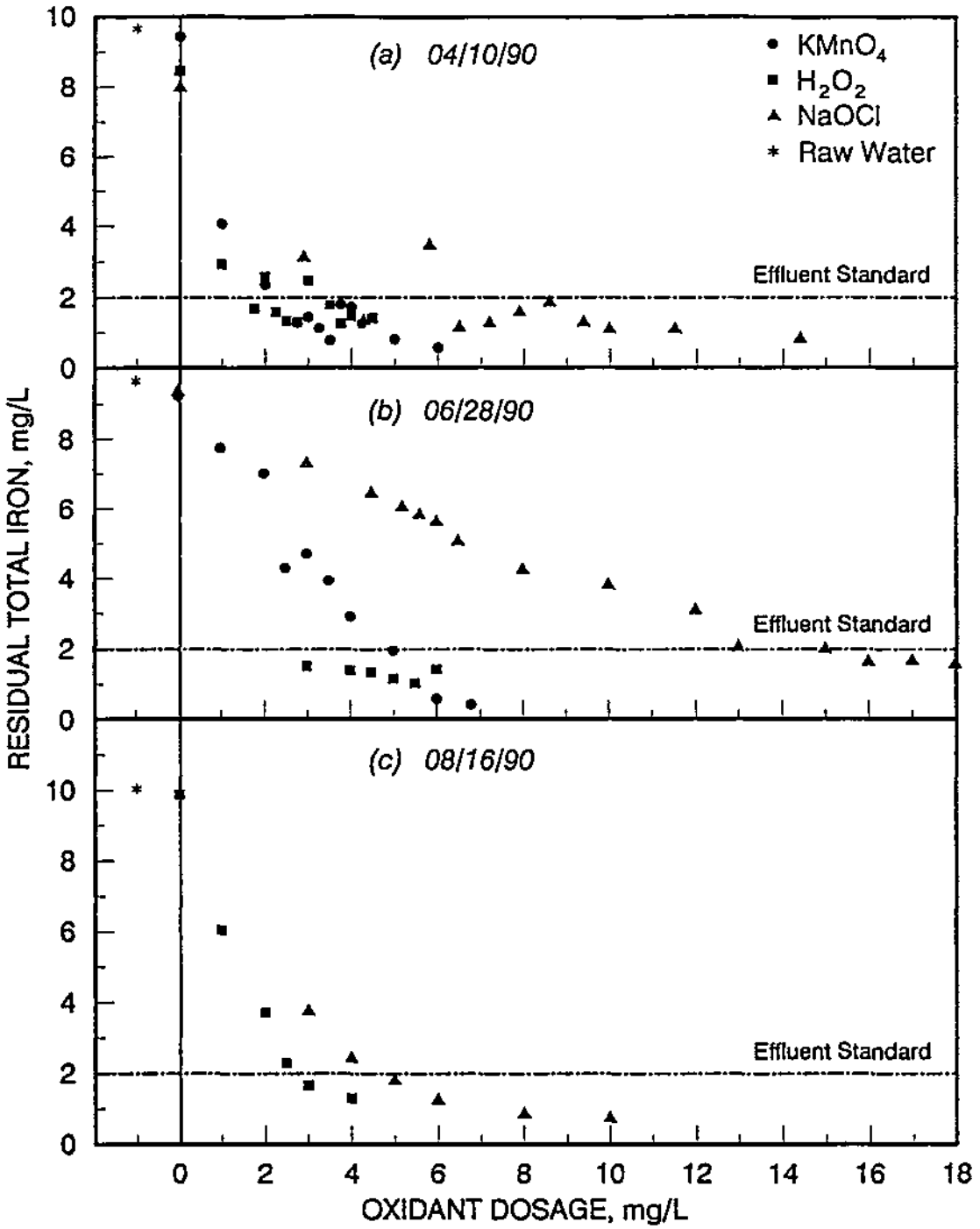
The dosage required on June 28, 1990, to lower the total iron concentration to 2.0 mg/L with $KMnO_4$ was double that required on April 10, 1990 (table 4.3, figures 4.2a and 4.2b). This was the case despite the fact that the quality of the raw ground water from well ME3D was not significantly different on these two dates. Removal of total iron by H_2O_2 was found to be more replicable. Figure 4.2 and table 4.3 indicate that the H_2O_2 dosages required to lower the total iron to 2.0 mg/L (referred to here as the critical dosages) were 2.1, 2.8, and 2.8 mg/L for the three test runs. The critical NaOCl dosages were found to be 5.0, 13.0, and 5.0 mg as Cl_2 per liter for the three runs. Because of the identical value for the April 10 and August 16 samples, 13 mg/L of Cl_2 was not used for design purposes.

For design purposes, the dosages required to meet the total iron effluent standard were set at 5.0, 2.8, and 5.0 mg/L for $KMnO_4$, H_2O_2, and NaOCl (as Cl_2), respectively. The

Table 4.2. Results of Aeration of Selected Ground-Water Samples

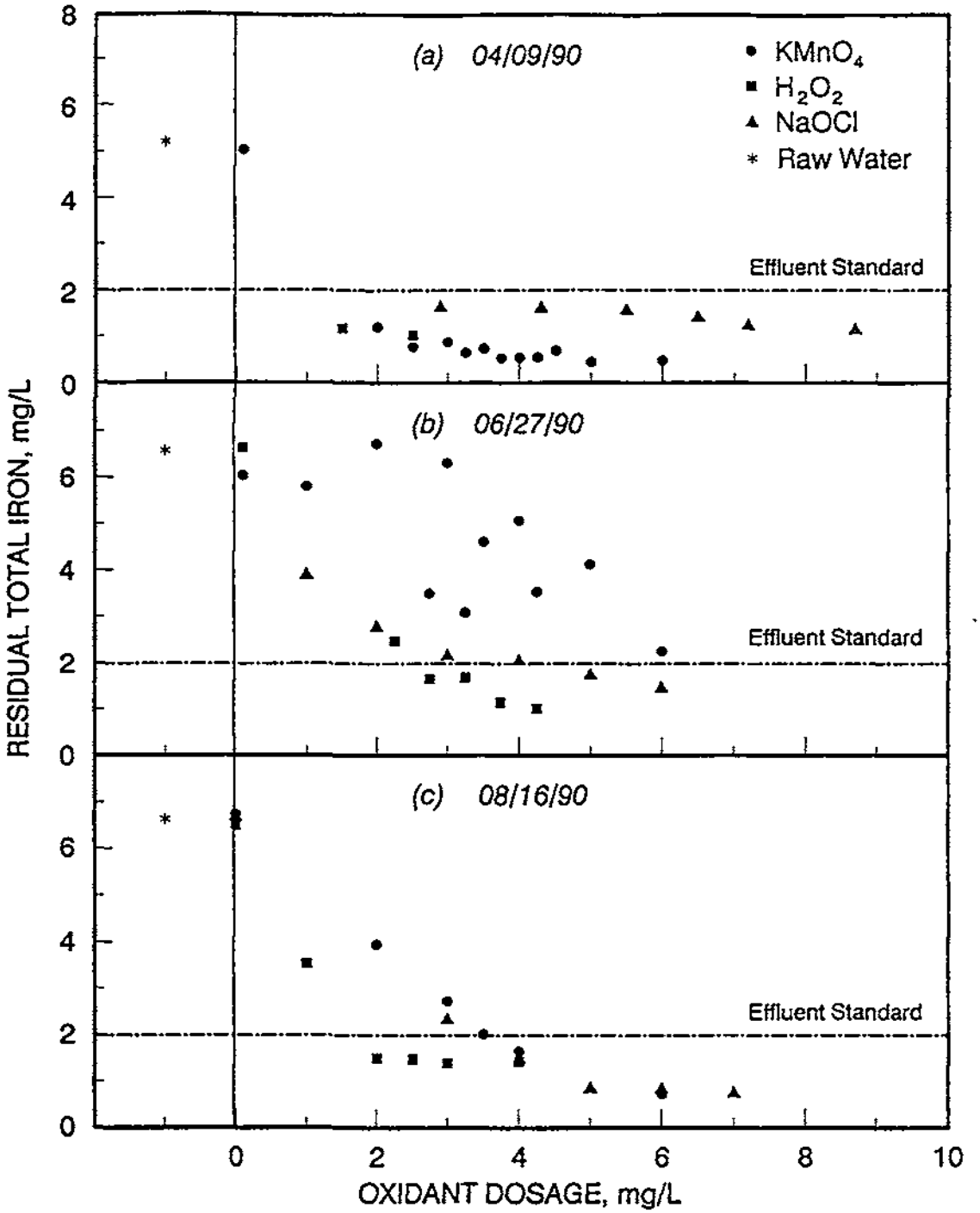
Date	*Well, MED*	*Sample or minutes aerated*	*Iron, mg/L*		*Manganese, mg/L*		*Turbidity NTU***
			Total	*Soluble*	*Total*	*Soluble*	
4/17/90	16	Raw	18.12	-	0.64	-	-
		Control -K*	15.96	1.55	0.62	0.58	250
		-H	17.76	3.02	0.64	0.57	280
		-C	12.18	1.14	0.64	0.58	207
		60min	16.10	0.68	0.32	0.09	-
4/24/90	22	Raw	13.52	-	0.17	-	-
		Control -K	12.96	6.85	0.17	0.18	169
		-H	11.36	3.51	0.17	0.17	150
		-C	11.88	3.66	0.17	0.13	133
		0 min	12.80	8.67	0.26	0.18	-
		60 min	12.04	0.09	0.20	0.21	-
		120 min	11.96	0.07	0.17	0.05	-
		180 min	11.36	0.07	0.11	0.04	-
4/25/90	10	Raw	9.02	-	-	-	-
		Control -K	9.56	2.14	0.30	0.51	55
		-H	9.68	2.82	0.31	0.37	80
		-C	9.68	3.31	0.31	0.49	122
		0 min	-	8.33	-	0.51	-
		60 min	9.51	0.03	0.31	0.33	-
		120 min	9.25	0.14	0.13	0.05	-

Notes: * Control for K ($KMnO_4$), H (H_2O_2), and C (NaOCl) test runs, with 2-min rapid mix, 20-min flocculation, and 60-min settling.
** Measured a few days after collection.



Well ME3D

Figure 4.2. Oxidant dosages required to lower total iron concentrations at well ME3D to the effluent standard of 2.0 mg/L



Well ME15D

Figure 4.3. Oxidant dosages required to lower total iron concentrations at well ME15D to the effluent standard of 2.0 mg/L

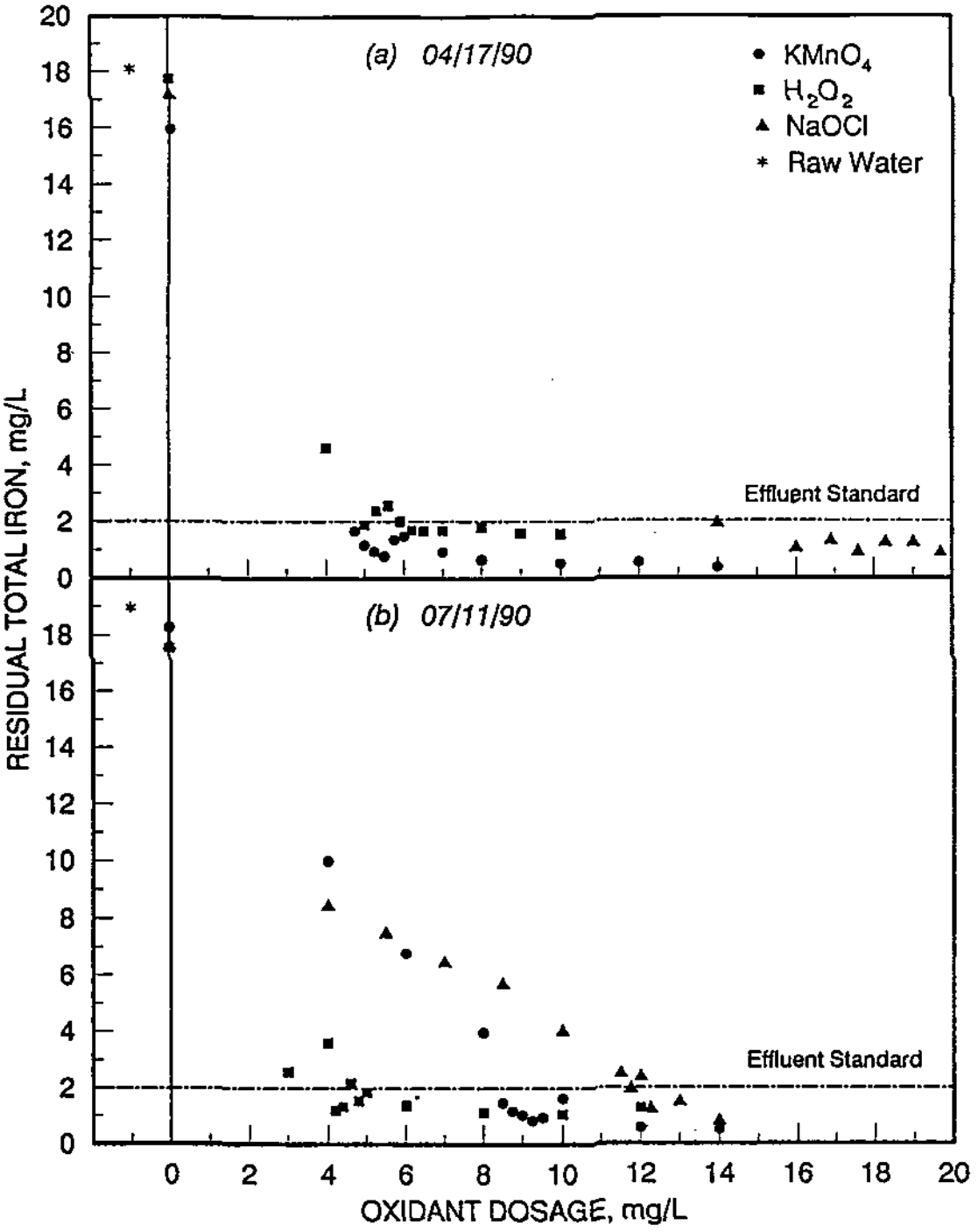
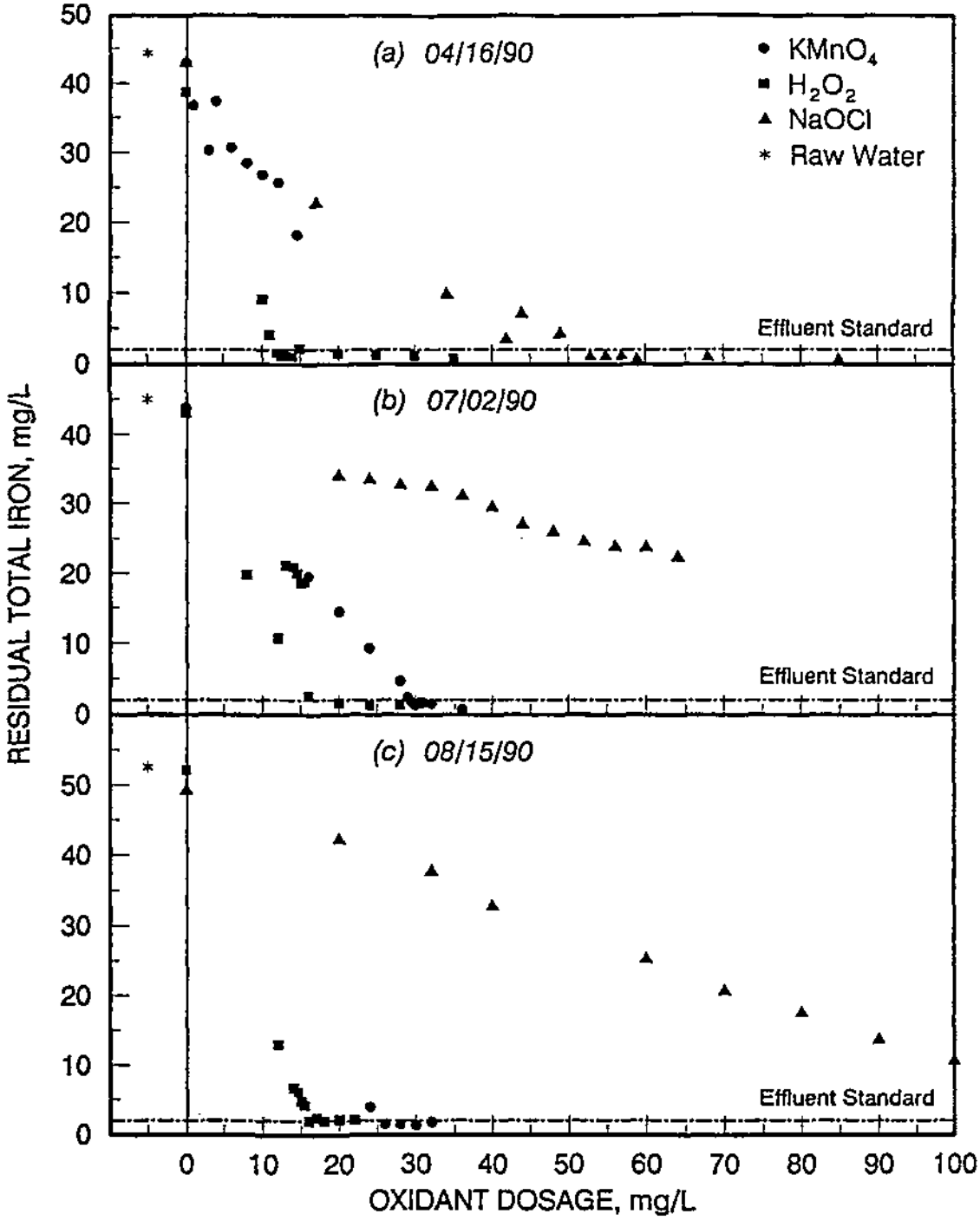
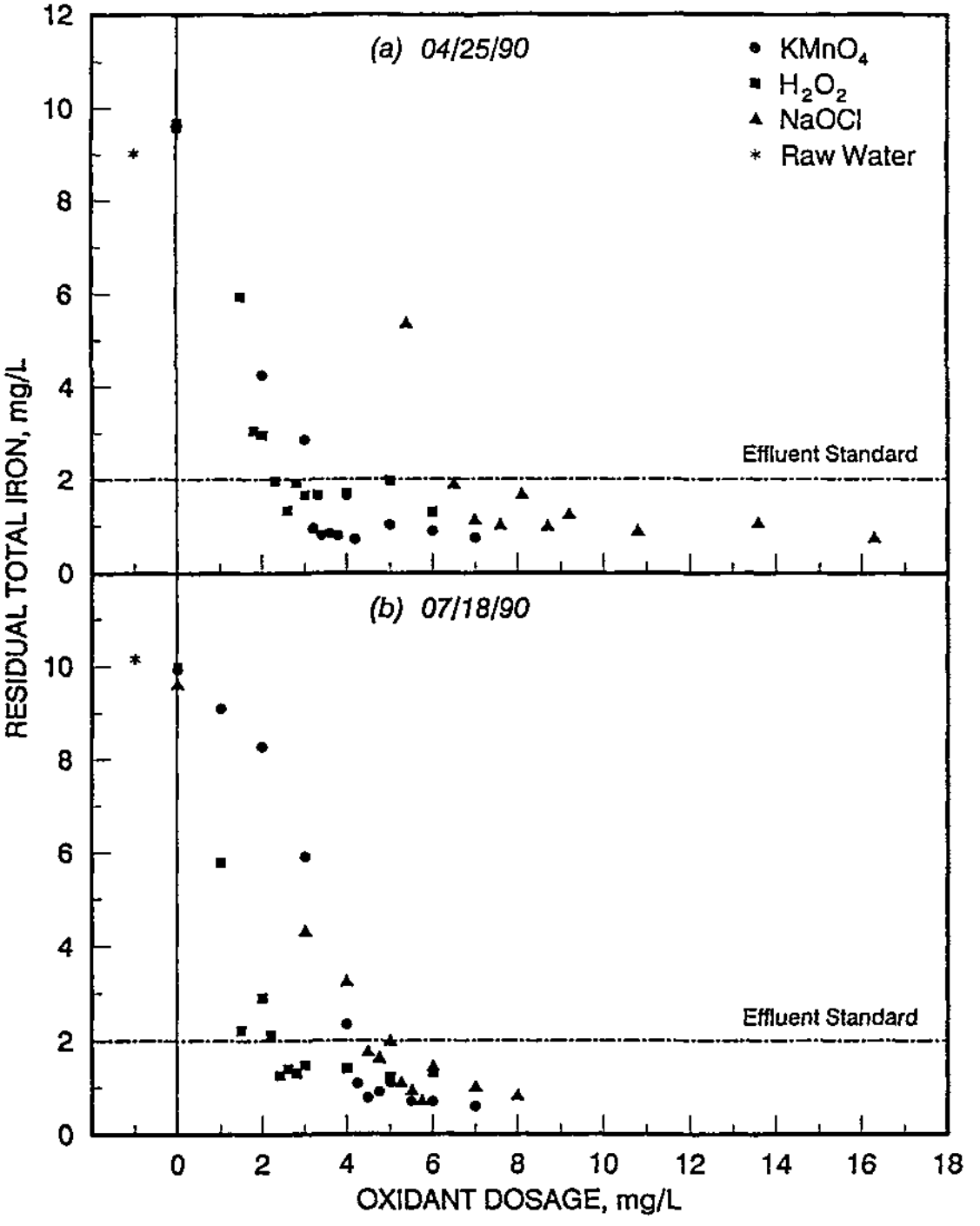


Figure 4.4. Oxidant dosages required to lower total iron concentrations at well ME16D to the effluent standard of 2.0 mg/L



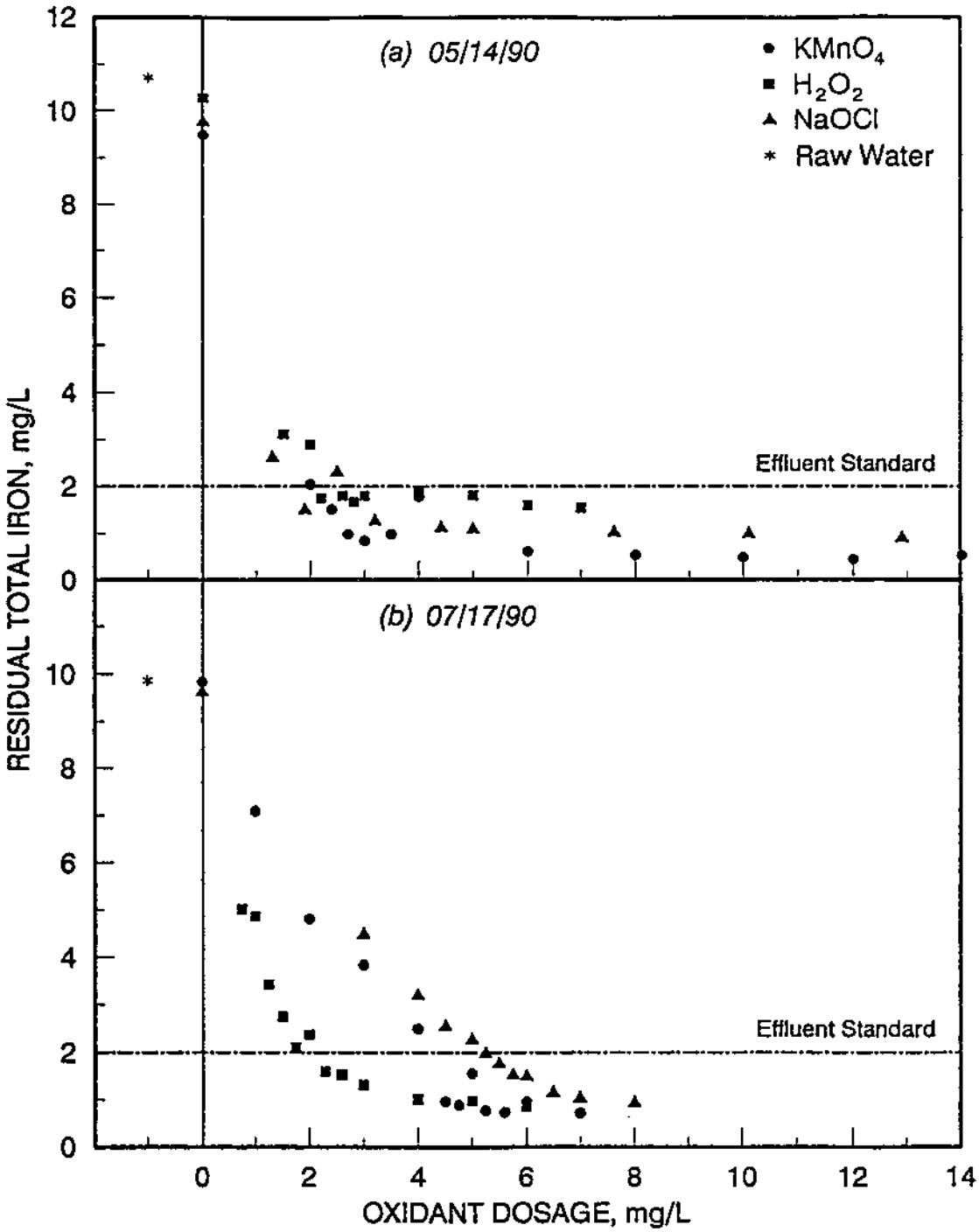
Well ME18D

Figure 4.5. Oxidant dosages required to lower total iron concentrations at well ME18D to the effluent standard of 2.0 mg/L



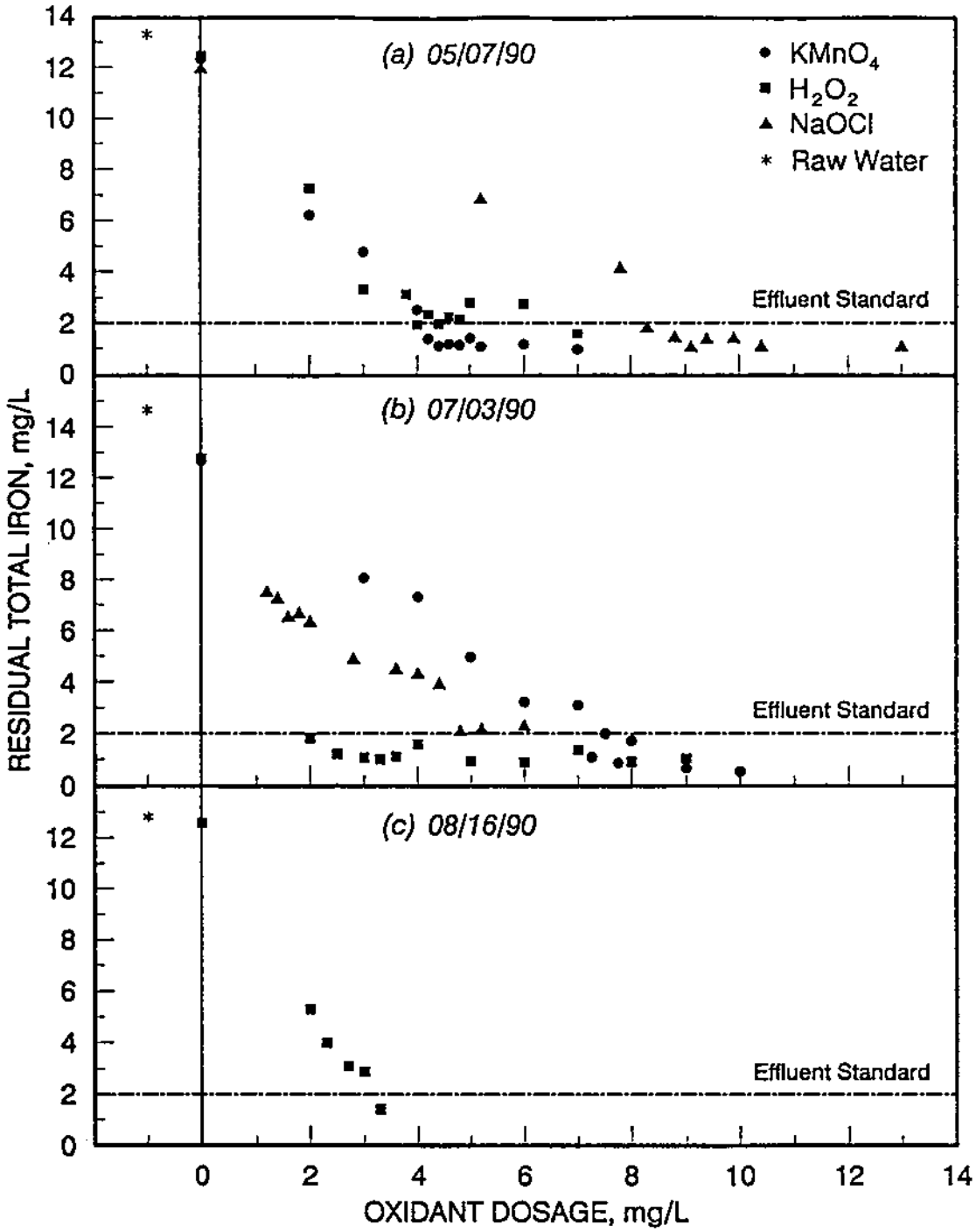
Well ME10D

Figure 4.6. Oxidant dosages required to lower total iron concentrations at well ME10D to the effluent standard of 2.0 mg/L



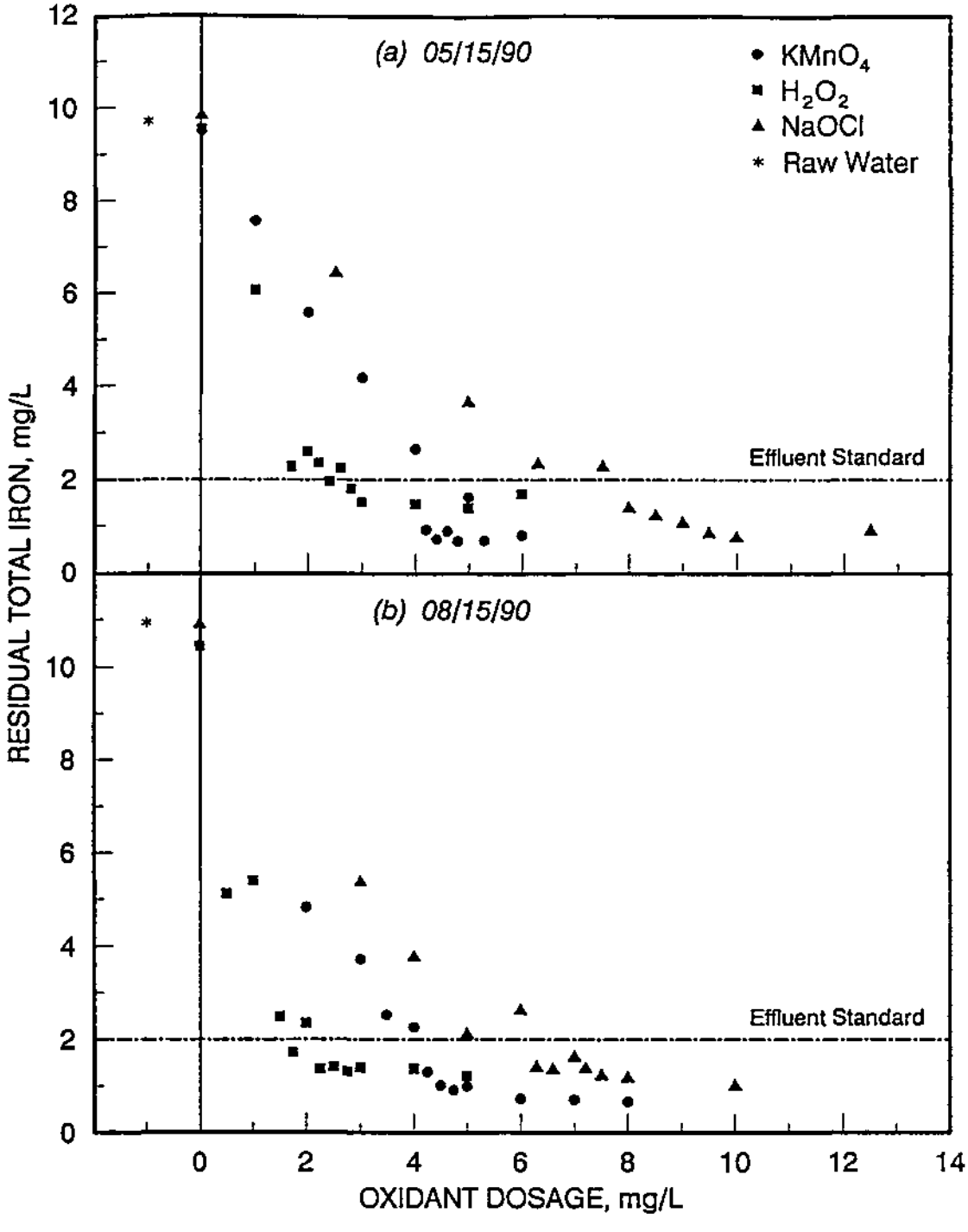
Well ME11D

Figure 4.7. Oxidant dosages required to lower total iron concentrations at well ME11D to the effluent standard of 2.0 mg/L



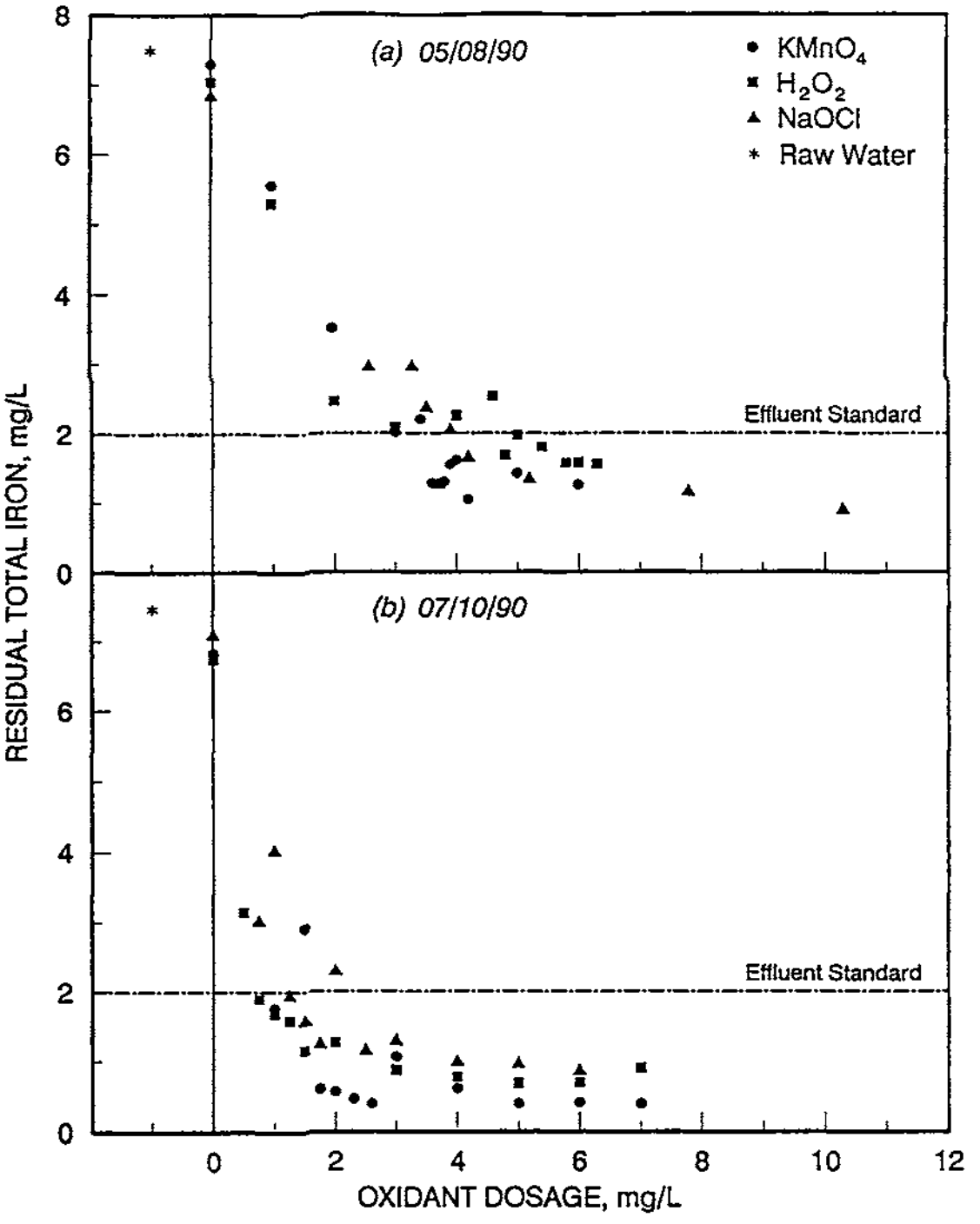
Well ME12D

Figure 4.8. Oxidant dosages required to lower total iron concentrations at well ME12D to the effluent standard of 2.0 mg/L



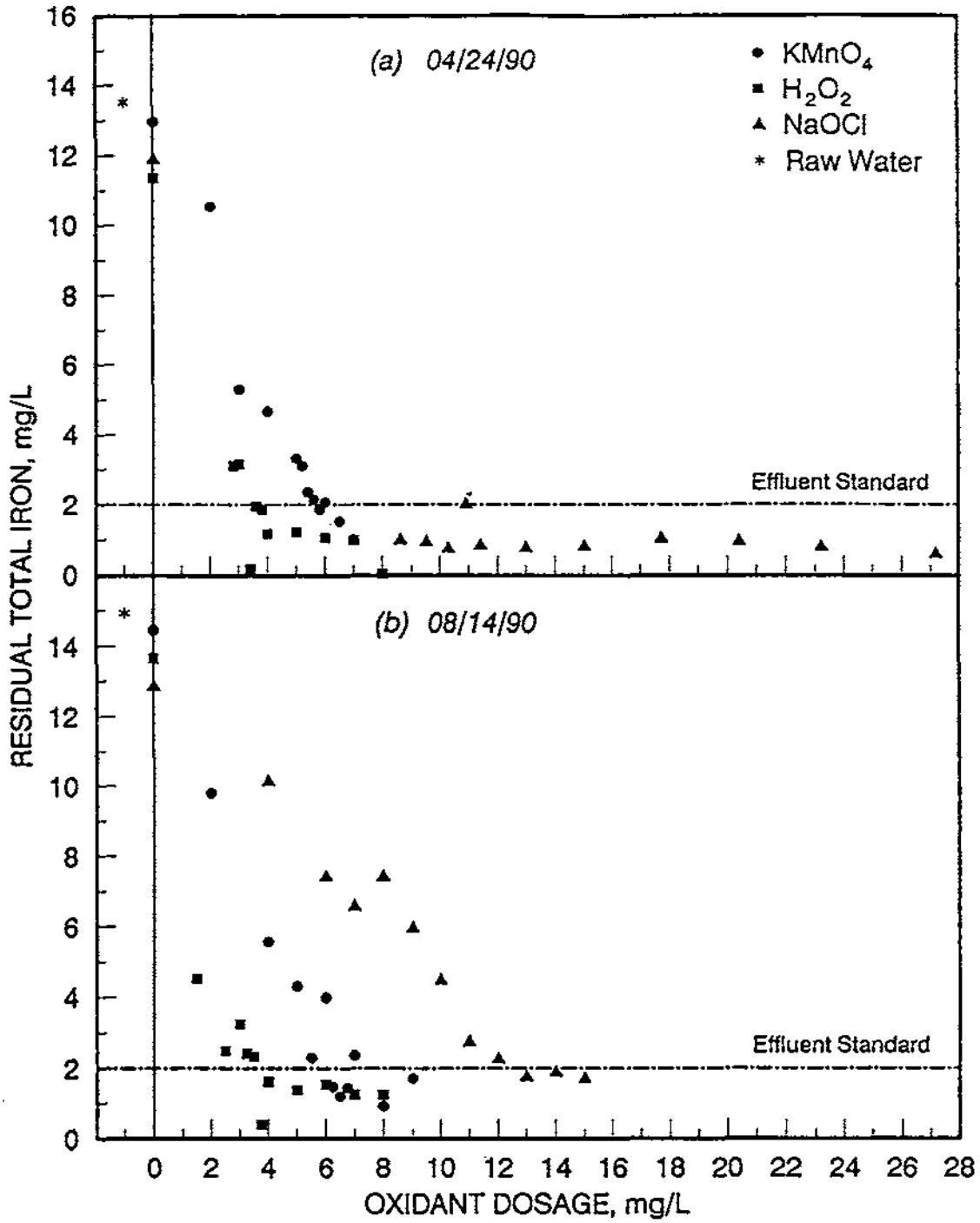
Well ME13D

Figure 4.9. Oxidant dosages required to lower total iron concentrations at well ME13D to the effluent standard of 2.0 mg/L



Well ME14D

Figure 4.10. Oxidant dosages required to lower total iron concentrations at well ME14D to the effluent standard of 2.0 mg/L



Well ME22D

Figure 4.11. Oxidant dosages required to lower total iron concentrations at well ME22 to the effluent standard of 2.0 mg/L

Table 4.3. Oxidant Required and Residues Produced to Meet the IPCB's Total Iron Effluent Limit of 2 mg/L

Well, MED	*Date, 1990*	*Oxidant concentration required to meet the iron effluent limit, mg/L*			*Residues produced at the required oxidant concentration. mL/L (o/oo)*		
		$KMnO_4$	*H_2O_2*	*NaOCl as Cl_2*	*$KMnO_4$*	*H_2O_2*	*NaOCl*
3	4/10	2.5	2.1	5.0	4.6	2.5	2.9
	6/28	5.0	2.8	13.0	4.2	4.5	3.0
	8/16	–	2.8	5.0	–	6.0	5.5
	Design:	(5.0)	(2.8)	(5.0)	(476)	(6.0)	(5.5)
15	4/9	1.3	1.0	2.1	-	0.5	0.8
	6/27	-	2.6	4.0	-	2.5	2.0
	8/16	3.5	1.9	3.4	5.2	2.5	4.2
		(3.5)	(2.6)	(4.0)	(5.2)	(2.5)	(4.2)
16	4/17	4.4	5.2	12.0	6.5	5.0	5.5
	7/11	8.2	4.0	12.0	15.5	12.0	14.0
		(8.2)	(5.2)	(12.0)	(15.5)	(12.0)	(14.0)
18	4/16	-	12	52	-	21	36
	7/2	29	16	-	40	30	-
	8/15	26	16	-	49	30	-
		(29)	(16)	(52)	(49)	(30)	(36)
10	4/25	3.0	2.3	6.6	4.8	2.7	6.0
	7/18	4.1	2.3	4.5	9.0	4.1	4.2
		(4.1)	(2.3)	(6.6)	(9.0)	(4.1)	(6.0)
11	5/14	2.0	1.9	2.7	2.7	2.5	2.5
	7/17	4.0	2.2	5.3	5.5	3.1	4.5
		(4.0)	(2.2)	(5.3)	(5.5)	(3.1)	(4.5)
12	5/7	4.3	4.1	8.3	7.7	5.1	7.0
	7/3	7.0	2.0	6.5	13.0	5.5	6.0
	8/16	–	3.2	–	–	11.5	–
		(770)	(4.1)	(8.3)	(13.0)	(5.5)	(770)
13	5/15	4.1	2.5	7.4	7.0	3.6	6.0
	8/15	4.2	2.2	6.2	7.0	4.0	5.5
		(4.2)	(2.5)	(7.4)	(7.0)	(4.0)	(6.0)
14	5/8	3.5	3.2	4.1	4.5	1.5	4.0
	7/10	1.1	0.9	1.8	2.3	0.5	1.0
		(3.5)	(3.2)	(4.1)	(4.5)	(1.5)	(4.0)
22	4/24	6.0	3.6	7.0	10.5	6.7	5.0
	8/14	6.0	3.8	13.0	11.0	6.0	6.8
		(6.0)	(3.8)	(13.0)	(11.0)	(6.0)	(6.8)

Table 4.4. Quality of Raw Well Waters

Well, ME D	*1990 sample*	*Calcium, mg/L*	*Iron, mg/L*	*Manganese, mg/L*	*Sulfur, mg/L*	*Chloride, mg/L*	*Sulfate, mg/L*	*TDS, mg/L*	*TOC,* mg/L*
3	Spring	117	9.19	.22	20	19	63	543	1.9
	Fall	123	9.62	.25	23	20	73	580	3.1
15	Spring	90	6.33	.22	21	18	62	448	36.0
	Fall	117	7.61	.28	23	119	72	711	3.9
16	Spring	200	17.2	.64	85	47	269	1072	3.2
	Fall	233	20.8	.83	80	160	248	1304	2.9
18	Spring	429	44.3	2.83	459	198	1400	2938	1.8
	Fall	411	41.8	2.72	430	180	1440	2851	3.1
10	Spring	129	10.3	.33	86	30	252		2.8
	Fall	137	10.7	.35	88	33	261	910	3.1
11	Spring	116	10.4	.26	87	18	245	704	2.6
	Fall	111	8.27	.25	79	18	238	709	1.6
12	Spring	122	11.5	.66	30	20	95	464	2.2
	Fall	128	13.2	.81	31	24	92	629	1.6
13	Spring	93	11.7	.36	25	7.0	73	453	2.7
	Fall	88	8.71	.28	23	8.2	74	453	3.0
14	Spring	104	6.97	.32	32	28	96	560	1.7
	Fall	102	6.53	.31	27	31	86	539	2.5
22	Spring	100	14.2	.17	28	19	79	513	2.3
	Fall	96	13.3	.16	25	21	79	540	3.7

* TOC samples were collected in the spring and winter of 1990.

volumes of solids generated were estimated as 0.46, 0.60, and 0.55 percent, respectively (table 4.3).

With only a 2-minute rapid mix, 20-minute flocculation, and 1-hour settling period, without oxidant addition, the percentage of soluble iron removed in control vessels was 79 to 98 percent (appendix I-1), 52 to 81 percent (appendix I-2), and 58 percent (appendix I-3) for the three samples tested. These removals were due to atmospheric air oxidation. For the sample taken on April 10, 1990, addition of oxidants was very effective for oxidizing soluble iron. Even as low a dosage as 1 mg/L of either $KMnO_4$ or H_2O_2, or 2.9 mg/L of Cl_2, reduced soluble iron to less than 1 mg/L (appendix I-1). The dosages were well below the stoichiometric amounts. This may be due to catalytic reactions of precipitates.

Inspection of appendix I-2 indicates that the residual soluble iron in the two control vessels for $KMnO_4$ tests was significantly different: 4.05 and 1.68 mg/L. Residual soluble iron with $KMnO_4$ doses of up to 3.5 mg/L was higher than 2.0 mg/L (which was higher than for the control test of run 2). A similar situation was observed in runs 1 and 2 for NaOCl oxidation.

For the third sample collected at well ME3D, oxidation of soluble iron by H_2O_2 and NaOCl was effective (appendix I-3).

As expected, total manganese concentrations increased with increased dosages of $KMnO_4$ (appendices I-1 and I-2). Most of the total manganese was in the soluble form. When $KMnO_4$ dosages were greater than 2 mg/L, both total and soluble manganese content exceeded 1.0 mg/L. Therefore the use of $KMnO_4$ (above 2.5 mg/L) appears to result in meeting the total iron limit but not the total manganese standard.

As can be seen from appendices I-1 through I-3, no significant manganese oxidation was noticed with H_2O_2 dosages up to 6 mg/L. This concentration is well above the theoretical requirement. Similar results were reported by Knocke et al. (1987). They reported that H_2O_2 had no effect on the oxidative removal of soluble manganese from river water over the pH range 5.0 - 8.0. Nevertheless, the total manganese concentrations at well ME3D were low and meet the effluent limit.

Some reduction of total and soluble manganese occurred with NaOCl oxidation (appendices I-1 through I-3). Especially with the highest NaOCl dosages, soluble manganese could be reduced from 0.23-0.30 mg/L to 0.02-0.07 mg/L.

Well ME15D. Samples from well ME15D had total iron concentrations of 5.19-6.63 mg/L and total manganese concentrations of 0.21-0.27 mg/L (table 4.1). Both of these metals were in the soluble form.

For the first sample, collected on April 9, 1990, all three oxidants were effective at low dosages in reducing total iron to lower than 2 mg/L (figure 4.3a, table 4.3). However, the dosages exceeding 2.0 mg/L seemed not to be much more effective in total iron removal. Although total iron was not high, total organic carbon (TOC) was high in this sample (table 4.4). Organic compounds might complex with iron, interfering with the iron removal process.

Total iron removal with $KMnO_4$ was poor for the June 27, 1990, sample (appendix J-2, figure 4.3b). Total iron removal with H_2O_2 was also poor during run 1. These results were due primarily to small floc formation and poor settling. Only the H_2O_2 jar test results for run 2 were used for the critical dosage evaluation. The critical oxidant dosage for the third sample (collected on August 16, 1990) was significantly higher than that for the April 9, 1990, sample (figures 4.3a and 4.3c, table 4.3).

As shown in table 4.3, the oxidant concentrations required to meet the effluent total iron limit are suggested to be 3.5, 2.6, and 4.0 mg/L for KMnQ,, H_2O_2, and NaOCl (as Cl_2), respectively. The highest amounts of residues generated would be 0.52, 0.25, and 0.42 percent, respectively. The estimated solids produced by the oxidant additions varied greatly from sample to sample.

As can be seen from appendices J-1, J-2, and J-3, most of the soluble iron can be oxidized for removal to meet the IPCB standard with 2-3 mg/L of the three oxidants used. Both total and soluble manganese concentrations increased significantly with $KMnO_4$ oxidation, as anticipated. With the critical $KMnO_4$ dosage of 2.6 mg/L, there is a possibility of exceeding the total manganese effluent limit. Hydrogen peroxide dosages also showed no effect on manganese removal. However, higher NaOCl dosages reduced manganese concentrations.

Well ME16D. The total iron levels for the two samples collected from well ME16D were 18.12 and 18.96 mg/L; the total manganese levels for the same samples were 0.64 and 0.75 mg/L (table 4.1). All were in the soluble form. These metal concentrations were found to be the second highest among the ten wells studied.

The dosages of H_2O_2 and NaOCl needed to remove the iron to acceptable levels were able to be duplicated for the second tests, which was not the case with $KMnO_4$ (table 4.3, figure 4.4). The dosages of $KMnO_4$, H_2O_2, and NaOCl required to lower the total iron levels to 2.0 mg/L were found to be 8.2, 5.2, and 12.0 mg/L, respectively.

A peculiar phenomenon was noticed for the tests carried out on April 17, 1990 (appendix K-1). The quantity of residues generated in test run 2 was less than the quantity generated in run 1 for comparable dosage rates in the case of all three oxidants. The reasons are unknown (appendix K-1). This made it difficult to estimate the solids generation. The quantities of residues generated at the critical dosages for the July 11 sample were almost 2.6 times those for the April 17 sample (table 4.3). For design purposes, the estimated amounts of residues generated by $KMnO_4$, H_2O_2, and NaOCl were taken as 1.55, 1.20, and 1.40 percent, respectively.

Appendices K-1 and K-2 indicate that the three oxidants were effective for oxidizing soluble iron, especially for the sample collected on April 17, 1990. On that date, residual soluble iron levels generally reached 0.10 mg/L or less.

Although $KMnO_4$ oxidation can lower iron concentrations to below the effluent total iron limit, an unfortunate result was that both total and soluble manganese levels increased

significantly with $KMnO_4$ to unacceptable concentrations (appendices K-1 and K-2). $KMnO_4$ alone can not be used for iron and manganese removal for well ME16D.

An examination of the data in appendices K-1 and K-2 shows that hydrogen peroxide at the dosages applied does not result in decreases in either total or soluble manganese. Reductions of both total and soluble manganese occurs with NaOCl dosages.

Well ME18D. The highest number (six) of samples were collected and studied at this well. Among the ten wells studied, the ground water at well ME18D contained the highest levels of iron, manganese, hardness, and turbidity (table 4.1), and the highest levels of the parameters shown in table 4.4. However, its pH was the lowest and was less than 7.0. The measured total iron concentrations ranged from 44.36 to 52.48 mg/L, and total manganese levels were between 2.34 and 3.16 mg/L.

Figure 4.5a and appendix L-1 indicate that the $KMnO_4$ applied was not enough to bring the total iron content down to 2.0 mg/L in the April 16 samples. The $KMnO_4$ requirements for meeting the total iron standard were found to be 29 and 26 mg/L for the samples collected on July 2 and August 15, respectively (figures 4.5b and 4.5c). From figures 4.5a, 4.5b, and 4.5c, critical dosages of H_2O_2 for the three samples were estimated as 12, 16, and 16 mg/L. A high NaOCl dosage (52 mg/L as Cl_2) was required for the April 16 sample (figure 4.5a). The data shown in figures 4.5b and 4.5c suggest that high NaOCl dosages of up to 64 and 100 mg/L cannot achieve the desired total iron effluent limit. With H_2O_2 oxidation for iron removal, 3.0 percent of residues will be produced (table 4.3).

Inspection of the data in appendices L-1, L-2, and L-3 reveals that all three oxidants oxidize soluble iron very well. However, significant increases of both total and soluble manganese were observed with $KMnO_4$ oxidation. Both total and soluble manganese levels were unaffected by H_2O_2 and NaOCl oxidation. It is concluded that the three oxidants studied cannot reduce the manganese levels to meet the effluent standards. Ground water at site ME18D appears to be the most difficult to treat to meet the required standards.

Well ME10D. The two samples of ground water from well ME10D were found to have total iron concentrations of 9.02 and 10.16 mg/L and total manganese concentration of 0.37 mg/L (table 4.1).

Figures 4.6a and 4.6b indicate that all three oxidants were effective for total iron removal. The dosages of $KMnO_4$, H_2O_2, and NaOCl required for reducing total iron concentrations to 2 mg/L were estimated as 4.1. 2.3, and 6.6 mg/L, respectively (table 4.3). The estimated residues generated would be respectively 0.90, 0.41, and 0.60 percent.

As with ground waters from other wells, the three oxidants were found to be effective for oxidizing soluble iron (appendices M-1 and M-2). Both total and soluble manganese increased above 1 mg/L when $KMnO_4$ was used. The data in appendices M-1 and M-2 also show that H_2O_2 was not effective in removal of total and soluble manganese. Only high dosages of NaOCl were effective in manganese removal. However, concentrations of naturally occurring manganese in the well water were less than the stipulated maximum level.

Well ME11D. Total iron concentrations in the two samples collected from well ME11D were 10.72 and 9.87 mg/L, and total manganese levels were 0.43 and 0.27 mg/L (table 4.1). The higher values were found in the first-round sample (May 14, 1990). As with other well samples, the majority of these two metals were in the soluble form.

From the data depicted in figures 4.7a and 4.7b, the critical dosage requirements for $KMnO_4$ and NaOCl for the second-round sample were found to be double those for the first-round sample (table 4.3). The H_2O_2 dosages required for the two samples were almost the same. The suggested dosages for meeting merely the total iron limit with $KMnO_4$, H_2O_2, and NaOCl are taken as 4.0, 2.2, and 5.3 mg/L, respectively. All three oxidant requirements are found to be less than the stoichiometric values.

Appendices N-1 and N-2 also indicate that total and soluble manganese concentrations will exceed 1.0 mg/L when 2 or 3 mg/L of $KMnO_4$ is used. Total and soluble manganese levels are unaffected by the dosages of hydrogen peroxide applied. Removal of manganese will occur when NaOCl dosage exceeds 8 mg/L.

Run 1 for both $KMnO_4$ and H_2O_2 generated greater amounts of residues than run 2 (appendices N-1 and N-2). The reason is unknown. The estimated solids productions with $KMnO_4$, H_2O_2, and NaOCl are 0.55, 0.31, and 0.45 percent, respectively (table 4.3).

Well ME12D. Table 4.1 indicates that the total iron content for well ME12D ranged from 12.80 to 14.68 mg/L, and the total manganese ranged from 0.82 to 0.91 mg/L. More than 90 and 83 percent of the total iron and manganese, respectively, were in the soluble form. Except for well ME18D, ME12D has the highest manganese levels among the ten wells studied.

The critical oxidant dosages for the ground water from well ME12D were estimated from the data displayed in figures 4.8a, 4.8b, and 4.8c. The dosage requirements for $KMnO_4$, H_2O_2, and NaOCl were found to be 7.0, 4.1, and 8.3 mg/L (table 4.3). Soluble iron was oxidized to less than 0.2 and 0.1 mg/L with $KMnO_4$ and H_2O_2, respectively, for the first (May 7) and the second (July 3) samples when $KMnO_4$ or H_2O_2 dosages exceeded the critical dosages listed in table 4.3. For the third-round (August 16) sample, the residual soluble iron concentration was 0.15 mg/L when the H_2O_2 dosage exceeded the critical value of 4.1 mg/L. On the basis of the results of soluble iron oxidation, the observed oxidant requirements are less than the theoretical amounts.

Precipitate values for the August 16 samples were much higher than normal. This might have been due to an error in observation; hence these values were not considered as design values. The expected residuals generated to meet the total iron effluent limit with $KMnO_4$, H_2O_2, and NaOCl were 1.30, 0.55, and 0.70 percent, respectively (table 4.3).

The data in appendices O-1 and O-2 show that the total manganese content increased to over 1.0 mg/L when $KMnO_4$ dosages were as low as 2 to 3 mg/L. Because of this, use of $KMnO_4$ as an oxidant for iron and manganese removal is not acceptable.

The results presented in appendices O-1, O-2, and O-3 suggest again that H_2O_2 does not affect total and soluble manganese removal. Also, the NaOCl dosage levels in water from well ME12D are not effective for manganese removal.

Well ME13D. The total iron concentrations in three samples from well ME13D ranged from 8.76 to 10.93 mg/L. These samples had total manganese levels between 0.29 and 0.37 mg/L (table 4.1). Soluble iron content was more than 95 percent of the total iron in the samples. Soluble manganese concentration was determined to be higher than total manganese. As stated previously, the reason for this is unclear.

From the data depicted in figures 4.9a and 4.9b, it is seen that the removal of total iron from the two samples is found to be almost identical for each of the oxidants. The dosages required with $KMnO_4$, H_2O_2, and NaOCl (as Cl_2) for meeting the total iron effluent limit were 4.2, 2.5, and 7.4 mg/L, respectively.

The quantities of residues generated from the two samples with each oxidant addition were also very close. The estimated residues produced would be 0.70, 0.40, and 0.60 percent, respectively, at the critical dosages of $KMnO_4$, H_2O_2, and NaOCl.

It is seen from appendices P-1 and P-2 that approximately 99 and 97 percent of the soluble iron in the first and second samples, respectively, could be oxidized by each oxidant. The dosages consumed were found to be less than the theoretical amounts.

It can be seen further from appendices P-1 and P-2 that when $KMnO_4$ dosages were over 3 mg/L, total and soluble manganese exceeded 1.0 mg/L, which is the recommended effluent limit. At the dosages of H_2O_2 and NaOCl applied to the two samples from well ME13D, no effect on total or soluble manganese removal was noticed.

Well ME14D. The total iron concentrations in the two raw water samples from well ME14D were 7.50 and 7.46 mg/L, almost all of which was soluble iron (table 4.1). The concentrations of manganese for those samples were low: 0.45 and 0.34 mg/L.

Comparisons of the data depicted in figures 4.10a and 4.10b reveal that the three oxidant dosages required for compliance with the total iron effluent limit stipulated by the IPCB were significantly higher for the May 8 samples than for the July 10 sample. The higher dosages are used for design purposes. The critical dosages for $KMnO_4$, H_2O_2, and NaOCl (as Cl_2) are taken as 3.5, 3.2, and 4.1 mg/L, respectively.

Inspection of appendices Q-1 and Q-2 shows that the amounts of precipitate generated in run 1 for each oxidant jar test were higher than those in run 2. The precipitate results for H_2O_2 oxidation for the May 8 sample did not provide good information for the ranges near the critical dosage of 3.2 mg/L. The amount of precipitate produced is thus estimated on the basis of engineering judgment. The expected solids generated from the critical dosages of $KMnO_4$, H_2O_2, and $NaOCl_2$ are 0.45, 0.15, and 0.40 percent, respectively.

The data listed in appendices Q-1 and Q-2 suggest that residual soluble iron was less than 0.15 mg/L in most cases when oxidant dosages were at or more than the critical dosage required to meet the ICPB's total iron limit of 2 mg/L. The amounts of oxidants needed for oxidizing soluble iron were generally found to be less than the theoretical stoichiometric values except for the May 8 sample with H_2O_2 oxidation. For this sample, the stoichiometric requirement of H_2O_2 was 2.24 mg/L. Figure 4.10a indicates a requirement of 3.2 mg/L of H_2O_2. However, for the July 10 sample only 0.9 mg/L of H_2O_2 was found to be needed (figure 4.10b, table 4.3).

Appendices Q-1 and Q-2 also indicate that the use of $KMnO_4$ cannot bring samples into compliance with the IPCB's total manganese limit of 1.0 mg/L. Hydrogen peroxide does not affect the concentrations of total or soluble manganese. High dosages of NaOCl (> 10 mg/L) will remove manganese.

Well ME22D. The total iron concentrations in three samples from well ME22D ranged from 12.92 to 14.92 mg/L. These samples had the lowest total manganese (0.17 - 0.27 mg/L) among all the wells examined for this investigation. Almost all the iron and manganese was found to be in the soluble form (table 4.1).

A comparison of the data presented in figures 4.1 la and 4.1 lb indicates that removal of total iron with $KMnO_4$ and H_2O_2 in the two samples collected on different dates (April 24 and August 14) was nearly identical. However, the NaOCl dosage required to achieve the recommended total iron level of 2 mg/L for the August 14 sample was almost twice that required for the April 24 sample (table 4.3, figures 4.1 la and 4.1 lb). The critical dosages of $KMnO_4$, H_2O_2, and NaOCl were 6.0, 3.8, and 13.0 mg/L, respectively. The residues generated were respectively 1.10, 0.60, and 0.68 percent (table 4.3). For the other well samples tested, although the use of $KMnO_4$ can achieve the effluent total iron limit, $KMnO_4$ dosages increased the total and soluble manganese levels, thus violating the recommended total manganese standard.

On the basis of the data for soluble iron and oxidant dosages (appendices R-1 and R-2), it was found that the three oxidants required to oxidize soluble iron in water from well ME22D are less than the theoretical amounts. Hydrogen peroxide again showed no effect on the removal of total and soluble manganese from this well water. Appendices R-1 and R-2 suggest that both total and soluble manganese can be removed when NaOCl dosages are higher than 15 mg/L.

Cost Estimates for Iron and Manganese Removal Units

To develop preliminary cost estimates for construction of iron and manganese removal units and for their operation and maintenance, information was obtained from a consulting engineering firm and chemical manufacturers. The cost estimates were used in comparing and evaluating process alternatives. Combined treatment of ground water from a number of wells is expected. Therefore preliminary cost estimates were made for plant capacities of 1, 2, 3, 4, 5, and 10 million gallons per day (MGD), on the assumption that several pumping units would be clustered together for treatment to provide a higher scale of economy.

Further research is needed to identify costs of sludge management, including handling, storage, treatment, and disposal of sludge. Sludge would most likely be lagooned and disposed of in a landfill. No information on iron and manganese sludge disposal costs have been compiled by the USEPA or any other agency.

The following estimates cover only preliminary piping construction, and treatments using chlorination, potassium permanganate, and hydrogen peroxide.

Cost of Piping. Piping costs were not estimated for clustered treatment. However, the unit costs for plastic pipe and ductile iron pipe are presented in table 4.5. Table 4.5 includes costs for various pipe sizes, excavation and backfill, and finished grading with soil or concrete. For 6- and 8-inch pipes, the excavation would be 4 feet deep and 2 feet wide, and the cost of excavation and refill would be $2 per foot. Wider trenches would be needed for 10- and 12-inch pipes, which would increase the cost by 50 percent. Costs for grading, fertilizer, and seeding are $2 per foot. Finishing with concrete or asphalt would be $10 to $12 per foot.

Cost of Chlorination. Cost estimates for iron and manganese removal from ground water by chlorination were prepared by Environmental Science & Engineering, Inc., Peoria, Illinois. Appendix S shows the breakdowns for each component of the treatment facility. The cost estimates for annual operation and maintenance (O & M) with chlorination are presented in appendix T. Data from appendices S and T were used to calculate cost estimates for iron and manganese removal from ground waters by chlorination. These estimates are summarized in table 4.6. The total costs for new construction (excluding sludge management) range from $264,000 to $1,188,000 for 1 MGD to 10 MGD plants, respectively. The total cost includes the chlorinator, flocculation and settling units, and housing. The annual operation and maintenance costs are estimated to range from $36,000 to $335,000 for 1 MGD to 10 MGD plants.

Cost of $KMnO_4$ Oxidation. Although the use of $KMnO_4$ will not enable waters to meet the manganese effluent standards, the cost estimates for iron removal by $KMnO_4$ are presented for purposes of comparison. A $KMnO_4$ injector (BIF 2507) is much cheaper than a chlorinator. The injector ($5,000) also can be leased at $500 per month. One unit can be used for plants up to 5 MGD. The estimated total construction costs and annual O & M costs for various plant sizes are presented in table 4.7. Flocculation, sedimentation, and housing costs listed in table 4.7 are adapted from table 4.6.

Chemical costs were estimated on the basis of the following data: The design $KMnO_4$ dosage is 5 mg/L (≈ 40 lb/d for 1 MGD). The cost of $KMnO_4$ is $1.50 per pound. The annual maintenance cost is estimated as 5 percent of the total construction cost.

Cost of H_2O_2 Oxidation. The results of this study indicate that a dosage of 3 mg/L of H_2O_2 is generally required to meet the iron and manganese effluent standards for the ground waters in the American Bottoms. At this H_2O_2 dosage rate, for a 1 MGD plant, the required amount of 50 percent H_2O_2 is approximately 50 lb/d. A 50-gallon drum contains 500 pounds of H_2O_2, which costs $1.00 per pound. A drum can be used for a 10-day period in a 1 MGD plant, whereas a 10 MGD plant needs 1 drum (500 pounds) per day. A larger plant may need

Table 4.5. Costs for Piping and Finishing

Material	*Size, diameter in inches*	*Cost, $per foot*	*Cost of excavation and fill, $*	*Finished eroding*		*Total cost, $/ft*	
				Fertilizer & seed, $	*Concrete, $*	*Seed*	*Concrete*
Plastic	6	6.35	2	2	10	10.4	18.4
	8	8.45	2	2	10	12.5	20.5
	10	10.05	3	3	12	16	25
	12	18.20	3	3	12	24	33
Ductile iron	6	11.3	2	2	10	15.3	23.3
	8	15.7	2	2	10	20	28
	10	20	3	3	12	26	35
	12	25	3	3	12	31	40
	14	32	4	4	14	40	50
	16	36	4	4	14	44	54

Table 4.6. Cost Estimates for Iron and Manganese Removal from Ground Water by Chlorination

Plant capacity, MGD	*Chlorinator, etc., $*	*Rapid mix, $*	*Flocculator, $*	*Settling tank, $*	*Building, shops, garage, $*	*Total construction cost, $*	*Annual operation & maintenance, $*
1	53,000	40,000	61,000	72,000	38,000	264,000	36,000
2	56,000	70,000	91,000	92,000	38,000	347,000	67,000
3	58,000	124,000	129,000	125,000	38,000	474,000	97,000
4	61,000	172,000	174,000	157,000	38,000	602,000	129,000
5	76,000	177,000	197,000	171,000	54,000	675,000	160,000
10	135,000	309,000	232,000	404,000	108,000	1,188,000	335,000

Note:

1. The construction costs are based on the water being pumped to the plant site.
2. Equipment costs were provided by General Filter Company and Wallace and Tiernan.
3. Chlorine costs were provided by Valley Chemical and are included in the O & M costs.
4. O & M costs were based on EPA publication 625/4-89/023, *Technologies for Upgrading Existing or Designing New Drinking Water Treatment Facilities.*
5. The cost estimates are based on a chlorine dosage of 10 mg/L and no filtration.

Table 4.7. Cost Estimates for Iron Removal by Potassium Permanganate ($KMnO_4$)

Plant capacity, MGD	*$KMnO_4$ injector, BIF 2507*		*Flocculation, sedimentation, & housing, $*	*Total construction cost, $*	*Annual KMnO4 cost, $*	*Annual maintenance cost, $*	*Annual O & M cost, $*
	No. of unit	*Cost, $*					
1	1	5,000	211,000	216,000	22,000	11,000	33,000
2	1	5,000	291,000	296,000	44,000	15,000	59,000
3	1	5,000	416,000	421,000	66,000	21,000	87,000
4	1	5,000	541,000	546,000	88,000	27,000	115,000
5	1	5,000	599,000	604,000	110,000	32,0000	142,000
10	2	10,000	1,053,000	1,153,000	220,000	57,000	277,000

Table 4.8. Cost Estimates for Iron and Manganese Removal by Hydrogen Peroxide (H_2O_2)

Plant capacity, MGD	*50% H_2O_2 usage rate, lb/d*	*Equipment cost, $*	*Flocculation, sedimentation, & housing, $*	*Total construction cost, $*	*Annual H202 cost,$*	*Annual maintenance cost, $*	*Annual O&M cost, $*
1	50	6,000	211,000.	217,000	18,000	11,000	29,000
2	100	6,000	291,000	297,000	36,000	15,000	51,000
3	150	6,000	416,000	422,000	55,000	21,000	76,000
4	200	6,000	541,000	547,000	73,000	27,000	100,000
5	250	6,000	599,000	605,000	91,000	32,000	123,000
10	500	6,000	1,053,000	1,154,000	183,000	57,000	240,000

a 5,000-gallon storage tank instead of a 50-gallon drum. However, the equipment cost for tank feed will be significantly higher, 5 to 7 times that of drum feed.

The estimated equipment costs and annual O & M costs are presented in table 4.8. One drum feed system will be enough for a plant of up to 10 MGD. The equipment costs include an aluminum drum, aluminum pipings and valves, aluminum pump, and other accessories. Construction work on H_2O_2 equipment and the design and installation of the chemical feeding system will be done by the chemical manufacturer. The costs of flocculation and sedimentation units are the same as for chlorination (table 4.6).

From the data presented in tables 4.6 through 4.8, it is seen that the costs of iron removal by chlorination are the highest, and iron removal costs by H_2O_2 oxidation are the lowest. Therefore, H_2O_2 is the oxidant of choice based on its effectiveness and economy.

SUMMARY AND CONCLUSIONS

On the basis of results obtained from the jar tests, the following conclusions can be drawn:

- The time for the first appearance of floc ranged from 10 seconds to 4 minutes. However, it was between 30 to 60 seconds in most cases.
- The color of the water after mixing was either yellow, amber, or brown.
- The size of flocs increased during the processes of rapid mixing, flocculation, and sedimentation. The flocs produced by $KMnO_4$, H_2O_2, and NaOCl all settled well in 40 minutes. These left the supernatant very clear.
- The estimates of residue produced by the iron and manganese removal processes are conservative. For design purposes, solids volume should be taken as upward of 15 to 20 percent.
- The quantity of solids produced by oxidation of iron and manganese and measured with an Imhoff cone was found to be underestimated (by 5 to 20 percent). Generally, the higher the oxidant dose, the greater the amount of solids.
- All jar tests were performed at pH of 6.9 to 7.5 without special pH adjustment. Various oxidant dosages and sedimentation did not cause any significant pH changes.
- The alkalinity required for iron and manganese oxidation was found to be much less than the theoretical amounts.
- If ground waters were withdrawn from the ten wells studied and discharged directly to the surface without iron and manganese removal, the turbidities of these discharged

waters would be expected to be in the range of 55 to 670 NTU. The turbidities for most of the treated effluents ranged from 3 to 10 NTU.

- Total iron concentrations for the ten wells studied ranged from 5.19 mg/L at well ME15D to 52.5 mg/L at well ME18D. Total manganese levels were between 0.17 and 3.12 mg/L. Most of the iron and manganese was found to be in the soluble form. For nine of the ten wells, the total manganese concentrations met the IPCB's 1.0 mg/L limit.

- Aeration and sedimentation alone was found not to be effective in total iron and manganese removal. The turbidity and color of aerated water also were not acceptable with just aeration and sedimentation.

- The oxidant dosages required to lower the total iron concentration to the recommended limit (2 mg/L) and the estimated solids generated at those dosages are listed in table 4.3. The values in parentheses are suggested for design purposes.

- Oxidant dosages required to achieve the IPCB's effluent total iron limit of 2 mg/L, as well as the estimated amount of residues produced at these critical dosages, are presented in table 4.3. Although the numerical values are different for each well, nine of the ten wells studied (all except well ME18D) indicate values within a narrow range.

- For well ME18D, no single oxidant can be used to remove iron and manganese to meet the IPCB's effluent standards of 2 mg/L for total iron and 1 mg/L for total manganese.

- For nine of the wells studied (all except well ME18D), $KMnO_4$ can be used to achieve the total iron limit but not the total manganese limit, because $KMnO_4$ increased total and soluble manganese levels.

- Oxidant dosages required to meet the recommended effluent total iron standard generally follow the order of NaOCl as $Cl_2 > KMnO_4 > H_2O_2$.

- The amount of residues generated by oxidants at the critical dosages follow the order of $KMnO_4 > NaOCl > H_2O_2$. Generally $KMnO_4$ produced the largest, strongest, and most dense flocs.

- Except in a very few cases, the amount of an oxidant required to oxidize soluble iron was found to be less than the theoretical stoichiometric value. This was probably due to catalytic reactions of precipitates.

- Total manganese removal and soluble manganese oxidation occurred when high dosages of NaOCl were applied. It is probable that trihalomethanes were produced in well water treated with NaOCl. Trihalomethanes were not measured during this study.

- Hydrogen peroxide was found to be effective for the removal of total iron and for the oxidation of soluble iron, but it had no effect on the total or soluble manganese in ground

waters. The concentrations of total manganese in waters from nine wells (all except well ME18D) were below the recommended limit.

- Cost estimates for construction, operation, and maintenance for oxidation with chlorine, $KMnO_4$, and H_2O_2 are presented in tables 4.6—4.8.

- Costs for iron and manganese removal from ground waters by H_2O_2 oxidation are found to be the lowest. On the basis of both effectiveness and economy, H_2O_2 is the oxidant of choice for iron removal in the American Bottoms.

Section 5

SUMMARY OF WATER QUALITY AND EFFLUENT ISSUES

by Robert Broms
Goodwin & Broms, Inc.

INTRODUCTION

The St. Louis District of the U.S. Army Corps of Engineers (USACE) has developed a plan for the reduction of damages caused by high ground-water levels in the American Bottoms area of southwestern Illinois (centered around the city of East St. Louis, Illinois). The prime operational aspect of this project is the pumping and discharging of ground water into surface streams of the area (USACE, 1987). The reader should note that the the tables for Section 5 are contained in Appendix X of the companion document titled "Appendices for Assessment of the Proposed Discharge of Ground Water to Surface Waters of the American Bottoms area of Southwestern Illinois."

Ground-water and surface water studies of the area completed by the Illinois State Water Survey (Rehfeldt, Section 2, this document) have shown that high levels of iron exist in the ground water throughout the study area. The ISWS study has also identified potential pumping areas that have the potential to draw contaminated ground water.

The purpose of this section report is to summarize the cost and water quality findings as they impact the project. The key issues of concern are water quality standards, effluent standards, discharge permits, and costs involved.

DISCHARGE RULES AND REGULATIONS

Federal Law as required under the Clean Water Act and Amendments, requires National Pollutant Discharge Elimination System (NPDES) permits for all point source discharges into U.S. waters. In the state of Illinois, the Illinois Environmental Protection Agency (IEPA) was delegated authority by the USEPA (in 1977) to implement the NPDES permit system.

When issued for new sources, NPDES permits will require that the source achieve the effluent and water quality standards required by the Illinois Pollution Control Board (IPCB 35, Subtitle C: Chapter I). These standards require that iron levels in the effluent from any discharge not exceed 2.0 mg/L (§ 304.124) and that the receiving stream after an appropriate mixing zone (§ 302.102) not exceed 1.0 mg/L (§ 302.208).

The key contaminant of concern is the dissolved iron concentration in the ground water, which is in excess of both water quality and effluent requirements in essentially the entire project area (figures 2.18 and 2.19, Rehfeldt, Section 2, this document).

In order to maintain compliance with state and federal water pollution control requirements, either the limitations required by state and federal law must be relaxed, or treatment of the pumpage to the levels required must be provided. It should be noted that the treatment, if provided for iron, will also provide treatment for any excess manganese levels.

TREATMENT COSTS

The original plan (USACE, 1987) was to have 60 different pumping units discharging a total of 41.25 mgd into different water courses in the area. The aerial distribution is as shown in figure 5.1, and well designation location and the approximate length of discharge pipe are shown in table 5.1 (appendix X). This layout as originally proposed was designed to provide maximum benefit for the least cost and was not intended to be used if treatment was required.

Since it is possible that treatment may be required, two options of treatment works were developed. The first option was to provide treatment of the pumpage at each well and discharge directly into a nearby water course. There is enough flexibility in the spotting of the locations of the pumps that this would be relatively easy to complete.

In order to determine if there was any economy of scale, a second option was developed in which the 60 wells were clustered into 21 treatment works. Based on a cursory review of the locations, table 5.2 (appendix X) and figure 5.2 were developed showing 21 clusters of discharges that could be grouped for treatment. These use the basic original locations outline by the USACE.

Based on the locations in tables 5.1 and 5.2 and the costs developed by Lin and Raman (Section 4, this document), the values in table 5.3 (appendix X) were developed and are summarized as follows:

Option	Description	Project cost	Annual cost
1	No treatment, rule change required	$9,031,000	$1,312,000
2	Treatment of discharge, individual pumps	$18,277,000	$2,605,000
3	Treatment of discharges, with clusters	$16,779,000	$2,394,000

It should be noted that the costs for the treatment of pumps at clusters (Option 3) do not include the costs of any easements, major utility relocation, or disruption of traffic for the interconnecting piping. All of these items will have a cost impact. Thus, it is likely that any cost savings due to economy of scale at the treatment units will be offset by the cost of

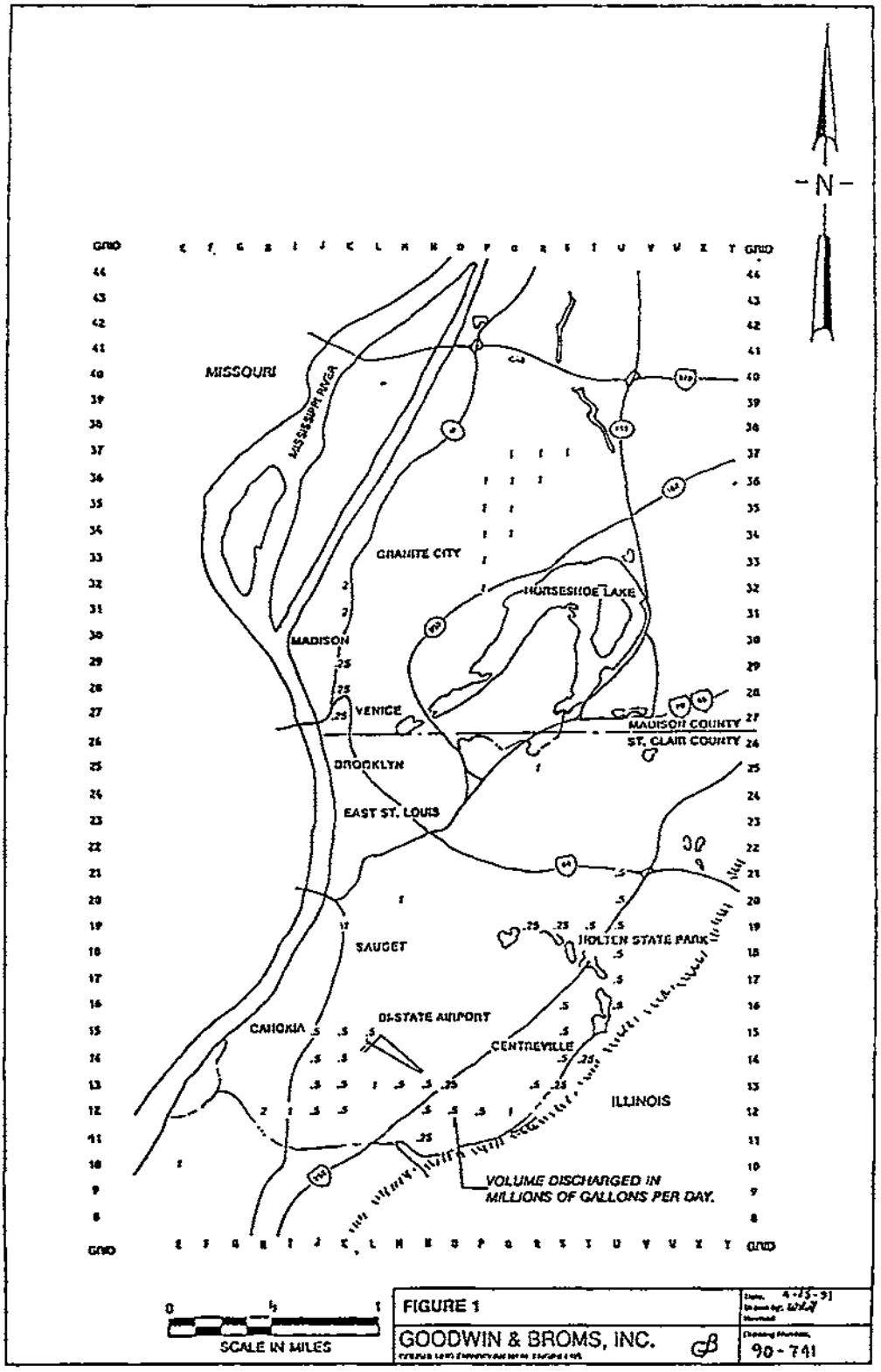


Figure 5.1. Volume of pumping at proposed pump location

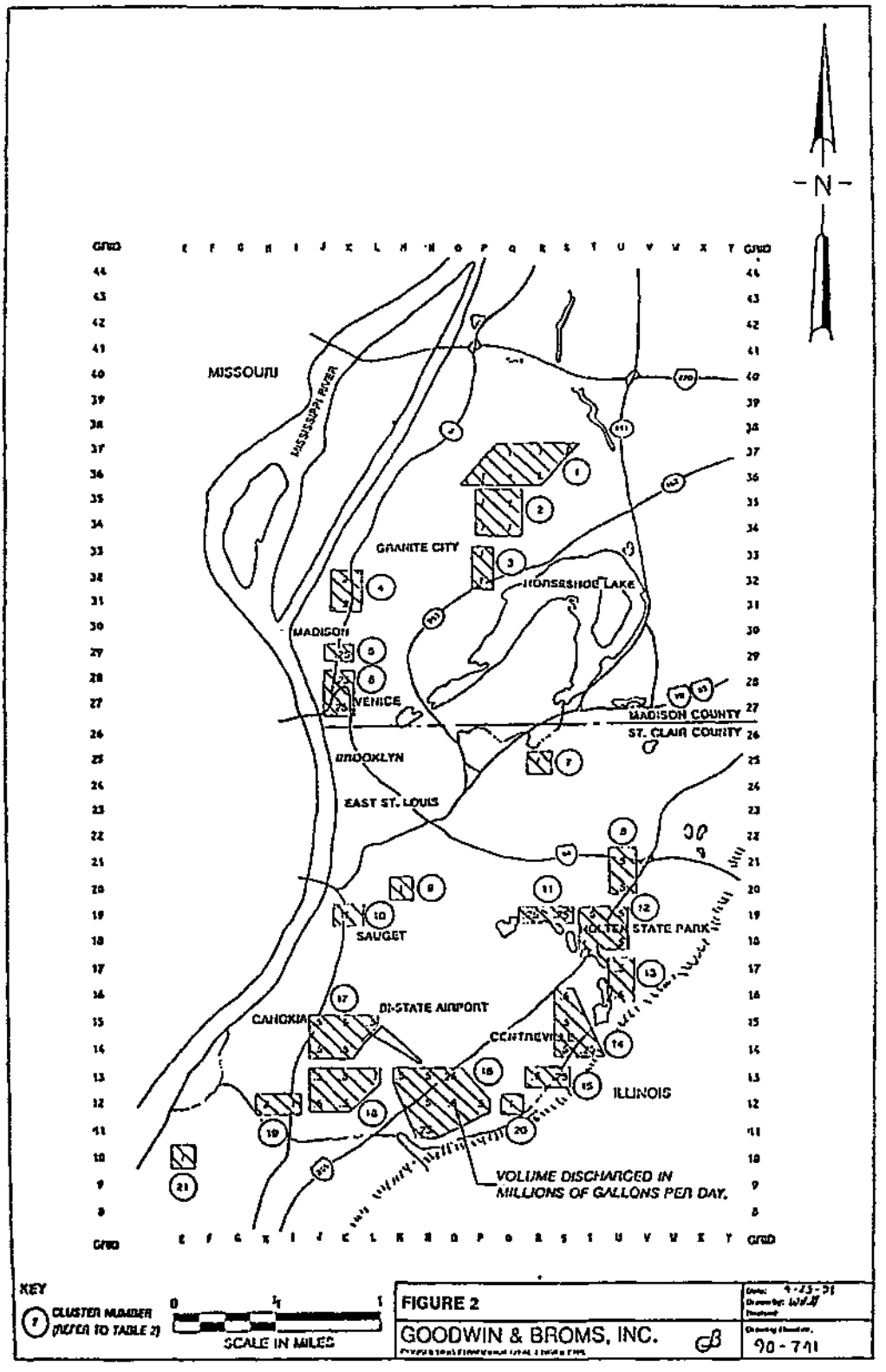


Figure 5.2. Proposed clusters

the interconnecting piping. Based on these considerations, it is unlikely that there will be any major savings by grouping the discharges for treatment.

CONTAMINATED GROUND WATER

According to Rehfeldt (Section 2, this document), 18 of the proposed 57 monitoring well locations have the potential to draw contaminated ground water (see figure 5.3 and table 5.4 in appendix X). Because of the potential hazards, either appropriate treatment needs to be provided for these withdrawals or they should not be included as part of the ground-water withdrawal system.

While the costs involved with the potential treatment of the contaminated ground water to achieve effluent and water quality standards are beyond the scope of this report, it would be fair to characterize it as significantly more expensive, both to construct and to operate, than the iron-removal treatment system. Thus, the potential contamination will likely eliminate those 18 well locations from consideration as part of the overall pumpage system. The cost estimates also do not include the expense of long term monitoring of well water quality that may be necessary for wells located near regions of contamination.

This then reduces the withdrawal rates and changes the costs (see table 5.5 in appendix X and appendix D) for treatment as follows (the costs of Options 1-3 have been repeated for reference):

Option	Locations	Flow rate (mgd)	Description	Project cost	Annual cost
1	57	41.25	No treatment, rule change required	$9,031,000	$1,312,000
2	57	41.25	Treatment of discharge at each pump	$18,277,000	$2,605,000
3	57	41.25	Treatment of discharge at clusters	$16,779.000	$2,239,000
4	39	26.0	No treatment, rule change required	$6,730,000	$911,000
5	39	26.0	Treatment of discharge at each pump	$12,966,000	$1,780,000
6	39	26.0	Treatment of discharge at clusters	$11,611,000	$1,594,000

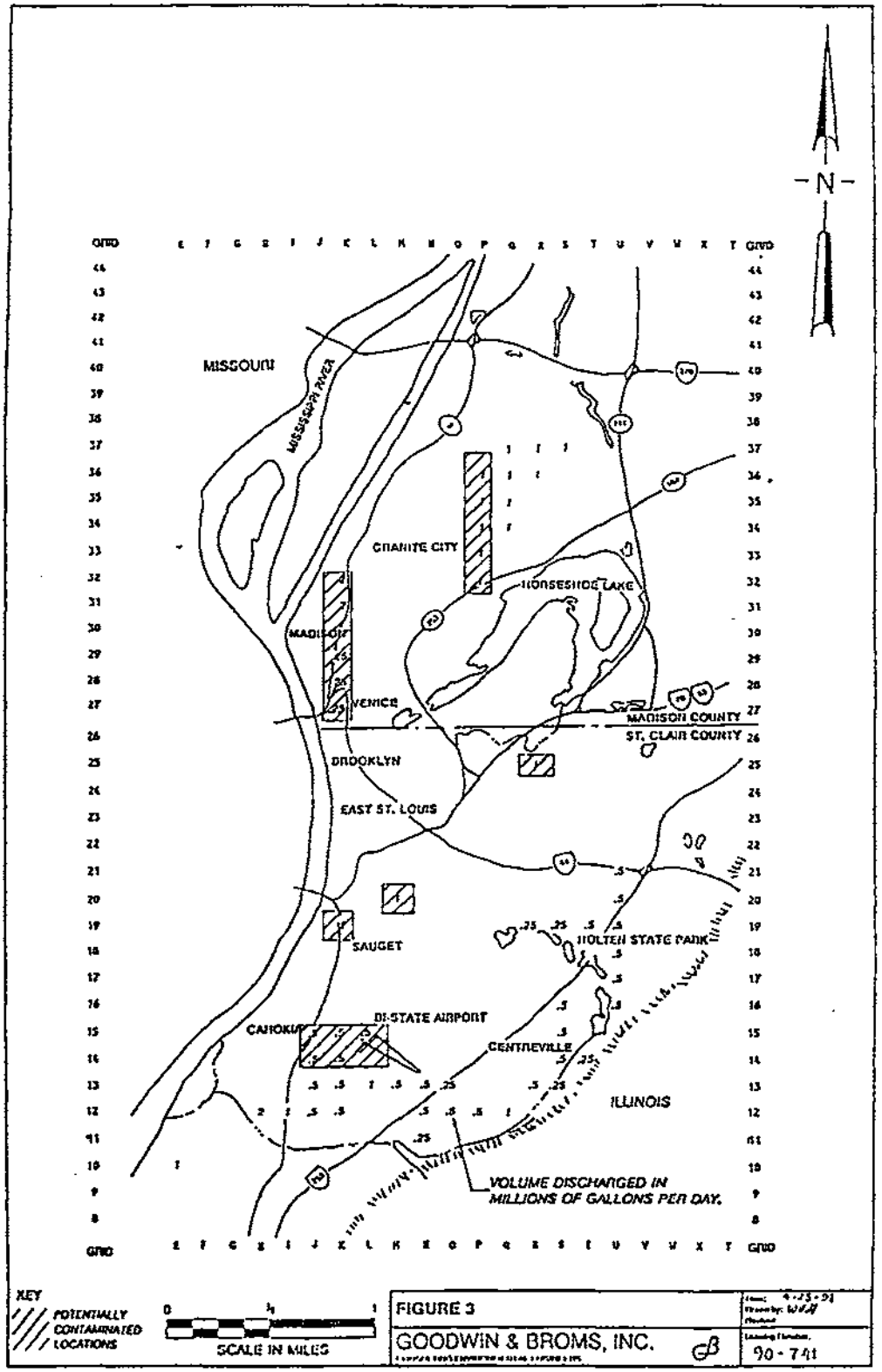


Figure 5.3. Potentially contaminated pumping locations

REVISION TO EFFLUENT AND WATER QUALITY STANDARDS

The IPCB has mechanisms in place for the adjustment of the effluent and water quality standards through the process referred to as "a site-specific rule change." This mechanism includes requirements for a demonstration of the impact on the specific water body involved for each site, allowance for public hearings, and an economic impact analysis.

The procedures for the modification of water quality standards are reproduced in appendix U. These require:

1) A proposal
2) A statement of reasons supporting the proposal
3) A petition with 200 signatures
4) A public hearing including discovery and subpoenas
5) Examination of any witnesses
6) A series of economic impact hearings

The process can take up to two years or more to complete.

The difficulty with the site-specific rule change for the American Bottoms Project is that there are a number of different water courses and lakes involved, and the impact on each would have to be addressed individually. In order to prepare an effective proposal, some site-specific work, sampling, and analysis would need to be completed particularly in regard to the potential color problem, sludge deposits, and impact on aquatic life.

If the IPCB completes approval of the new Water Quality Standards, they would still require approval by the USEPA since all state water quality standards must be federally approved.

It is projected that the costs to complete the appropriate studies to initially make the submission of the rule change would cost about $150,000, with hearing and submission costs through both the IPCB and the USEPA being about $100,000, thus bringing the total cost of the rule change to $250,000. This cost has been factored into Options 1 and 4 as shown in tables 5.3 and 5.5 (appendix X).

Normally to obtain approval for site-specific rule changes, there need to be compelling public benefits to justify the downgrading of the standards in a receiving stream. It is possible that the petitions for rule changes could be rejected in whole or in part.

Any impact analysis provided would need to show that the hardship of not allowing the relaxed standards is severe enough to justify the anticipated negative impact on the receiving stream segments. This hardship goes beyond a standard cost/benefit analysis and needs to show the arbitrary and unreasonable hardships that maintaining the current standards would impose.

KEY ISSUES

There are some extremely critical issues that have to be resolved if this project is to continue.

1. Does the USACE wish to pursue the 57 site-specific rule changes and associated costs involved? In addition to the costs, the negative impact on the receiving streams has to be justified (color, sludge, and impact on aquatic life). There will likely be some public opposition to the degradation of the surface waters, regardless of any public benefit to the lowering of the ground- water table. For any of the 57 locations, it is unlikely that a site-specific rule change to allow discharge without treatment would be successful, given the high iron concentration in the ground water.

2. If the site-specific rule changes are not pursued, then is the project justified economically if treatment is required? Will the local municipalities agree to the long-term operation and maintenance of the treatment facilities, and the long-term monthly sampling and reporting to IEPA through the NPDES permits?

3. Because of the potential of 18 of the 57 well locations being contaminated by various chemicals, what is the impact of the cost/benefit analysis if the flow is reduced from 41.25 to 26.0 mgd? What is the cost/benefit impact if treatment has to be provided to the 26.0 mgd plan?

SUMMARY

- The ground water in the American Bottoms area contains iron concentrations in excess of the IEPA effluent and water quality standards.

- In order to pump the ground water and discharge it to surface waters, IEPA permits will have to be obtained. In order to obtain the necessary permits, either treatment of the ground water or changes to the effluent and water quality standards are required.

- Providing treatment of the ground water to current standards nearly doubles the capital cost of the project.

- The cost of obtaining permits is not affected by the level of treatment, since the permit can only be issued to meet applicable standards regardless of what they are.

- Petitioning for site-specific rule changes for each discharge point would likely cost a total of $250,000 (see Table 5.6 in appendix X), and take more than two years to complete with no guarantee that the petition would be successful.

- Of the proposed 60 ground-water pumping wells at 57 locations, there is potential significant ground-water contamination being drawn into 18 of the locations. Excluding the 18 potentially contaminated sites, reduces the amount of ground water pumped from 41.25 mgd to 26.0 mgd.

- It is recommended that the Cost/Benefit Analysis, which was completed in 1987, be reanalyzed to determine if the project is still viable:

  a) If the flow rate is reduced to 26 mgd.
  b) If treatment of the ground water is required.
  c) If the project will be on hold for two or more years in order to obtain the rule change.
  d) If the cost of the rule change ($250,000) is factored into the analysis.

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