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APPLICATION OF A DISCRETE FRACTURE FLOW MODEL FOR WELLHEAD PROTECTION AT STURGEON BAY, WISCONSIN

by

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1998

APPLICATION OF A DISCRETE FRACTURE FLOW MODEL FOR WELLHEAD PROTECTION AT STURGEON BAY, WISCONSIN

Final Report to the Sturgeon Bay Utilities and Wisconsin Department of Natural Resources

by

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ABSTRACT

A wellhead-protection study for the city of Sturgeon Bay, Wisconsin, demonstrates the necessity of combining detailed hydrostratigraphic analysis with groundwater modeling to delineate zones of contribution for municipal wells in a fractured aquifer. We used a three-dimensional numerical model (MODFLOW) combined with a particle tracking codes (MODPATH) to simulate the groundwater system around Sturgeon Bay and to delineate capture zones for five municipal wells.

Detailed hydrostratigraphic characterization involved locating and characterizing vertical and horizontal fractures and high-permeability zones. Correlating stratigraphic interpretations with "hard" hydrogeologic data such as gamma and flowmeter logs, packer tests, and fracture mapping produced a hydrostratigraphic model with 14 gently dipping high-permeability zones related to bedding planes or facies changes. These zones serve as major horizontal conduits for groundwater flow.

The computer model was designed using the hydrostratigraphic model as a conceptual framework. Dipping fracture zones are simulated as thin high-permeability layers. The locations of exposed bedrock and surficial karst features were used to help quantify recharge. The model is transient and replicates seasonal water-level fluctuations in wells near Sturgeon Bay.

Model results show the extreme vulnerability of the Sturgeon Bay wells. Capture zones for the municipal wells extend north and south from the city center and terminate outside the Sturgeon Bay city limits. Travel times from recharge to all wells are generally less than one year even though the capture zones extend for several kilometers. Transient effects are clearly present in the particle paths, confirming that a transient model was required for this analysis.

In fractured carbonate aquifers the groundwater flow velocities can be so high, and resulting travel times so short, that the usual wellhead protection time-of-travel criteria (2 years, 5 years, 10 years, etc.) recommended by most wellhead protection manuals (e.g. USEPA, 1987) have little meaning. In particular, it may be impossible to prioritize areas within the ultimate zone of contribution based on time-of-travel to a well. For example, at Sturgeon Bay, some areas farther from the wells are probably more critical than areas nearer to the wells because of the presence of the horizontal flow zones which can rapidly conduct water to the wells from great distances. Therefore, we recommend that the entire zone of contribution be given equal protection in these types of fractured carbonate aquifers.

Given a well-defined hydrostratigraphic model, the porous-media-based MODFLOW code is adequate for simulating flow in the dolomite aquifer as long as individual horizontal flow zones are recognized and modeled discretely. MODFLOW's lack of capability to fully simulate fracture flow is balanced by its flexibility in simulating the complex boundary conditions found in most real-world problems.

APPLICATION OF A DISCRETE FRACTURE FLOW MODEL FOR WELLHEAD PROTECTION AT STURGEON BAY, WISCONSIN

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INTRODUCTION

Overview and Objectives

Protection of groundwater quality in areas of complex fractured rock presents major challenges for hydrogeologists, water managers, and regulatory officials. This report presents a summary and synthesis of several years of hydrogeologic data collection and groundwater modeling for the purpose of delineating zones of contribution for municipal wells at the city of Sturgeon Bay, Wisconsin. Beginning in 1994, the Wisconsin Geological and Natural History Survey (WGNHS) conducted a research study with funding from the Sturgeon Bay Utilities, the Wisconsin Department of Natural Resources, and the Door County Soil and Water Conservation Department.

Specific objectives of the study are :

-to test and demonstrate a state-of the art methodology for delineation of wellhead protection areas in complex fractured-rock environments; and

-to delineate zones of contribution for municipal wells in the city of Sturgeon Bay.

Background of study

City of Sturgeon Bay

Sturgeon Bay, Wisconsin is a city of about 9,000 residents in Door County, a peninsula bounded by Green Bay on the northwest and Lake Michigan on the southeast (fig. 1). The city is situated on both sides of Sturgeon Bay, a funnel-shaped bay extending southeast from Green Bay, and linked to Lake Michigan by the Sturgeon Bay Ship Canal. The Door Peninsula is composed of fractured dolomite of Silurian age that dips gently to the southeast. The dolomite serves as the principal aquifer for wells in the county (Sherrill, 1975). The area is characterized by thin soils, shallow depth to bedrock, and numerous small karst features. The lack of soil to attenuate contaminants, combined with the rapid groundwater movement through the fractured dolomite aquifer, leads to frequent contamination of groundwater by bacteria, nitrate, and other pollutants (Sherrill, 1975).

The city of Sturgeon Bay relies entirely on groundwater pumped from municipal wells finished in the dolomite aquifer. Since the turn of the century, the city has installed 12 municipal wells within the city limits; nine of these wells have shown signs of bacterial contamination, and seven of the wells have been shut down and abandoned. Currently (1998) the city operates five wells. Water from three of these wells is disinfected using on-site ozonation. Only two wells remain free of bacterial contamination (McMahon Associates, 1991).

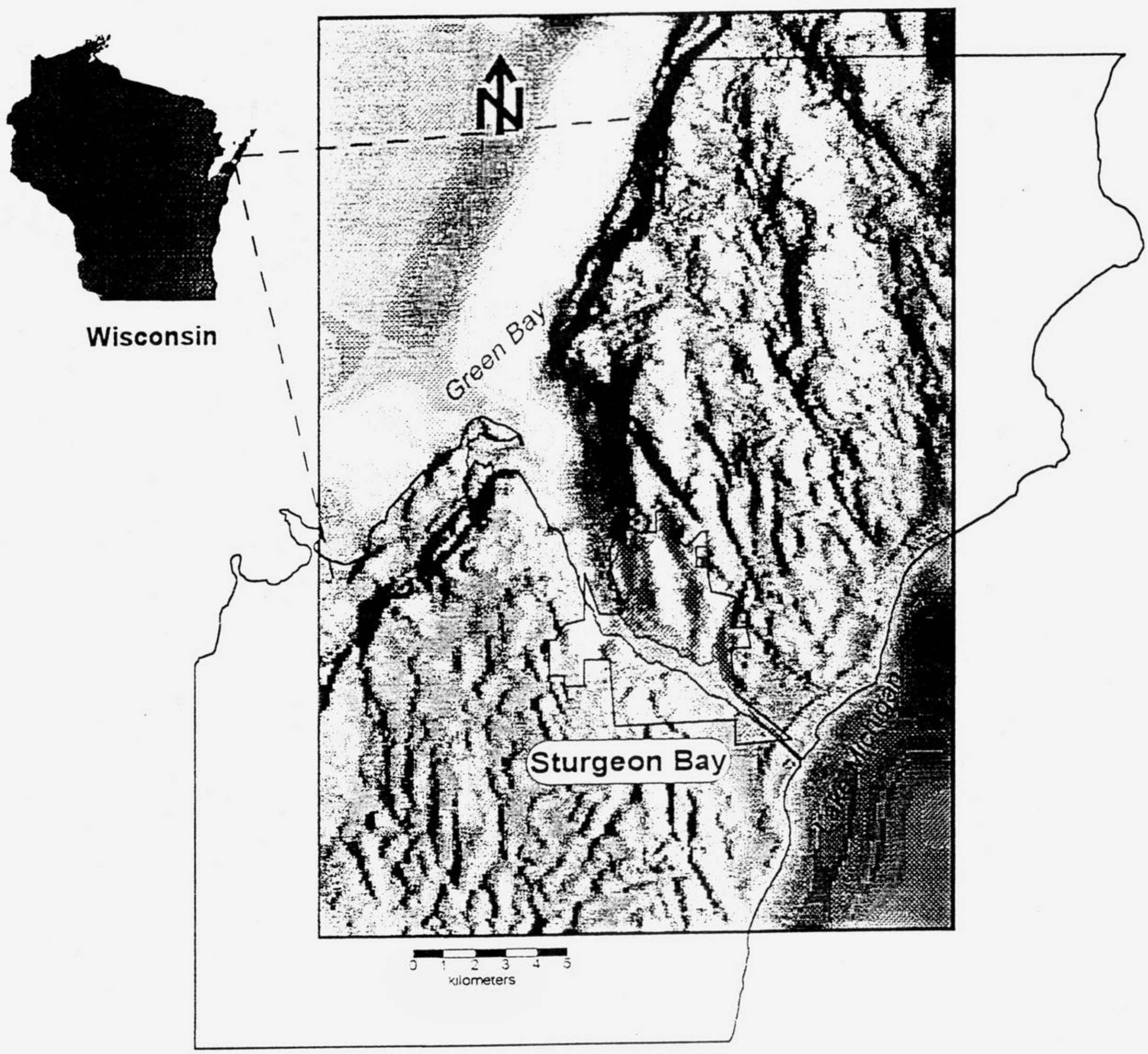


Figure 1. Location of Sturgeon Bay, Wisconsin, showing surface topography based on a digital terrain model.

Wellhead protection in fractured rocks

The goal of the federal wellhead protection program is to protect groundwater used for public water supply by managing potential sources of contamination within the land area that supplies water to a given well or wellfield. The first step in any wellhead protection program is to identify the capture zone, or zone of contribution (ZOC) of a well, defined as the land surface area where recharging precipitation enters a groundwater system and eventually flows to the well (Fig 2). Delineating the zones of contribution for municipal wells is a critical step in establishing wellhead protection areas for the wells. A wellhead protection area (WHPA) is defined by the federal Safe Drinking Water Act as the "surface and subsurface area surrounding a water well or well field, supplying a public water system, through which contaminants are reasonably likely to move toward and reach such water or well field " (USEPA, 1987). In practical terms, the ZOC is a technically-defined area based on groundwater hydraulics, while the WHPA is a legally-defined area including all or part of the ZOC and within which zoning practices or other land-use controls can be implemented to help protect groundwater from contamination.

The Wisconsin Department of Natural Resources (WDNR) has the responsibility and authority to delineate wellhead protection areas for all public water supplies in Wisconsin (WDNR, 1992). In 1992, the WDNR prepared the Wisconsin Wellhead Protection Program Plan (WDNR, 1992), which required the DNR to perform initial ZOC delineations for all existing municipal wells in the State. At the same time, the Wisconsin Administrative Code, Chapter NR811, was revised to require that a wellhead protection program plan be submitted for each new municipal well constructed in Wisconsin after April 1, 1992.

The technical methodologies for ZOC delineation range from simple to complex, and are described in a number of publications such as Born and others (1988), Bradbury and others (1991), Kreitler and Senger (1991), Muldoon and Peyton (1993), and USEPA (1987) among others. Most of these authors suggest simple techniques, such as the fixed-radius methods, as a first approach, but most also recommend the use of numerical groundwater flow models as more sophisticated and reliable methods for ZOC delineation. Unfortunately, due to the cost, data requirements, and level of expertise required, few municipalities in Wisconsin currently have numerical groundwater flow models available for use in ZOC delineation.

Previous work in Door County suggest that vertical fractures as well as horizontal bedding planes and dissolution zones provide the primary pathways for groundwater flow in the Silurian dolomite (Sherrill, 1978; Bradbury and Muldoon, 1992, Gianniny and others, 1996). The groundwater flow models for the Sturgeon Bay area require characterization of vertical fractures, horizontal high-permeability zones, and matrix permeabilities at a regional scale. This report integrates surface and subsurface stratigraphic, geophysical, and hydrogeological data in order to characterize both the horizontal high-permeability zones and matrix permeabilities in the Silurian aquifer. This report integrates and relies on the work of several other authors who have

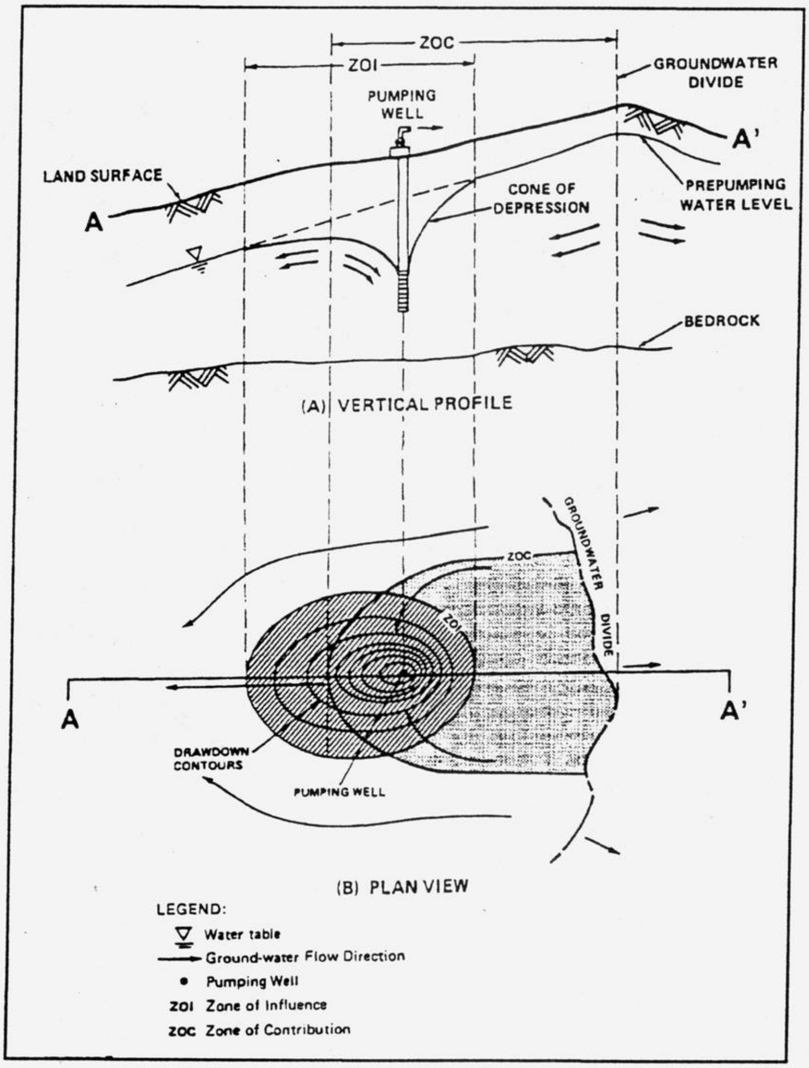


Figure 2. Diagram and terminology for wellhead protection in a simple hypothetical groundwater flow system (from USEPA, 1987).

studied the hydrogeology Silurian dolomite in detail. In particular, the hydrostratigraphic analyses of Ginnniny and others (1996) and the fracture analyses of Roffers (1996) were critical to this study.

Acknowledgments

This project was funded in part by the Wisconsin Department of Natural Resources through the State of Wisconsin joint solicitation of proposals to conduct research and monitoring on groundwater.

Many people and organizations contributed to this study. The Wisconsin Department of Natural Resources and the Sturgeon Bay Utilities provided funding. We are particularly indebted to Bill Schuster, Door County Conservationist, and Scott Adams, Manager of the Sturgeon Bay Utilities, for their vision and continuing support of this project, and for the assistance of their competent employees. WGNHS staff and students assisting with this project include Cristin Harris, Ann Fritz, Tim Eaton, Gary Gianniny, Mike Czechanski, Eric Oelkers, Bill Orr, and Bill Batten. We are also grateful for the assistance of the US Geological Survey in performing packer tests, particularly the work of Jim Raumann. Finally, we thank the residents of the Sturgeon Bay area who provided access to their property and wells.

Physical Setting

Physiography and climate

Sturgeon Bay is located near the center of the Door Peninsula, which separates Lake Michigan from Green Bay (fig 1). Along much of the Green Bay shoreline of Door County, the rocks form a prominent escarpment (the Silurian escarpment). Glaciation has removed much of the soil in the area, and the dolomite is frequently exposed at the surface or covered only by thin soils (Sherrill, 1978). Precipitation averages 31.5 in/yr, and average temperatures range from 17 °F in January to 69 °F in July. The ground is usually frozen from mid-December through mid-March. Dairy farming, forage production, fruit growing, and tourism are the main land uses in central Door County. Ship building and maintenance have long been the major industries in the city of Sturgeon Bay; in recent years the industrial base has shifted toward light industry, service, and tourism.

City well locations

This report refers frequently to specific high-capacity wells, and test wells, within the Sturgeon Bay city limits. Figure 3 shows the locations of these wells, and table 1 summarizes construction details of the wells. Each well has two identifying labels: the city well number (for example, Sturgeon Bay #1) assigned by the Sturgeon Bay Utilities, and a permanent county site identifier (for example, Dr-265) assigned by the WGNHS. The five active city wells are highlighted

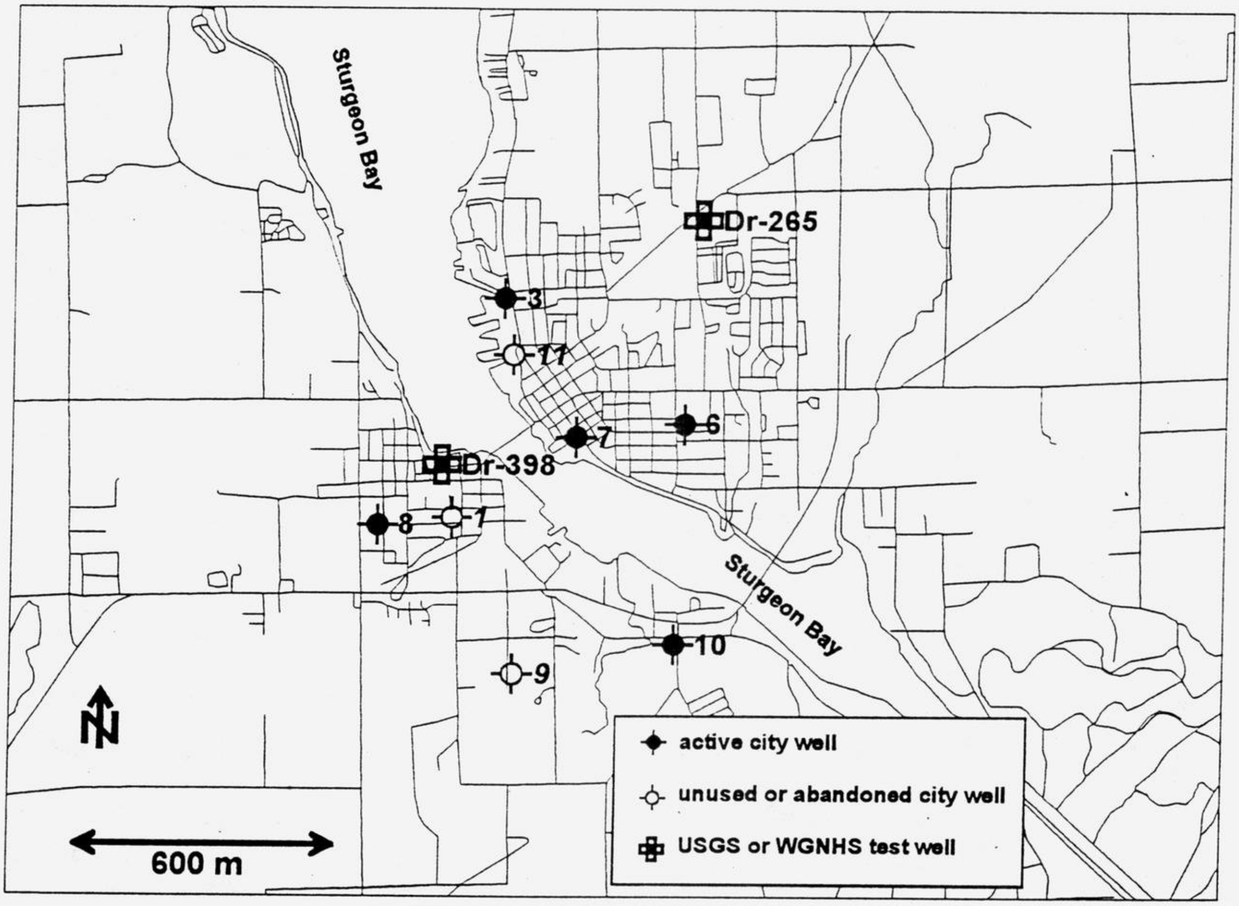


Figure 3. Locations of municipal wells and monitoring wells within the Sturgeon Bay city limits.

by shading in table 1. Two additional test wells were important to this study. Well Dr-265, located near the Door County fairgrounds, was installed as part of Sherrill's (1978) work, and is part of the WGNHS/USGS statewide groundwater-level monitoring network. Long-term water-level measurements are available from this well. Well Dr-398 is a test well installed (and subsequently abandoned) in Ottumba Park in the city specifically for the present study.

Table 1. Details of municipal wells and test wells within the Sturgeon Bay city limits. Shading indicates wells currently in service.

City well no	WGNHS well no	location	year	diam. (in)	total depth (ft)	casing depth (ft)	comments
1	Dr-4	Redwood St	1918	9	1178	139	contaminated; abandoned 1996
3	Dr-3	N 3rd Ave	1935	12	305	138	in service; no contamination
6	Dr-13	N 12 Ave	1951	15	425	212	in service; requires ozone treatment
7	Dr-39	Martin Park	1960	15	425	155	in service; never contaminated
8	Dr-52	Duluth Ave	1966	17	452	150	in service; requires ozone treatment
9	Dr-292	S Neenah Ave	1972	17	507	150	contaminated; never used
10		Tacoma Beach Rd	1977	13	447	170	in service; requires ozone treatment
11	Dr-397	N 1st Ave	1984	12	228	200	contaminated; abandoned 1996
-	Dr-265	County fairgrounds	1972	6	442	170	USGS/WGNHS test well
-	Dr-398	Ottumba Park	1996	6	255	50	WGNHS test well

Geology

The Silurian dolomite aquifer ranges in thickness from 200 to 500 feet in Door County (Sherrill, 1975). The unit is divided into the Niagaran Series and the underlying Alexandrian Series (fig 4). Both series are made up of gray, thinly bedded to massive dolomite with local cherty zones and shale layers. A complete description of dolomite stratigraphy can be found in Gianniny and others (1996).

The Ordovician Maquoketa shale underlies the dolomite, and is considered to be a regional confining unit (Bradbury and others, 1991). Surficial deposits consist of unlithified Pleistocene sediments, which range in thickness from zero to about 10 m in the area surrounding the city. In general, the upland areas have very thin (< 1 m) unconsolidated deposits and the low-lying areas have thicker (> 2 m) deposits.

The dolomite has relatively low primary porosity but it is extensively fractured. Studies by Sherrill, 1978; Bradbury, 1982; Bradbury and others, 1991; Bradbury and Muldoon, 1992; and

SERIES	GROUP	FORMATION	MEMBER
NIAGARAN		Engadine	
		Manistique	Cordell
			Schoolcraft
	Burnt Bluff	Hendricks	
		Byron	
ALEX-ANDRIAN		Mayville	
		Maquoketa	

Figure 4. Simplified stratigraphic column for Door County

Roffers, 1996 show major vertical fracture sets with orientations of about 70° and 155°. Average fracture spacing is about 3 to 6 m (Bradbury and others, 1991). Vertical fractures decrease in aperture (width) and density (number of fractures per unit area) with depth (Sherrill, 1978). They serve as pathways for vertical movement of recharge and contaminants from the land surface. High-permeability horizontal features such as bedding plane joints are evident in quarry walls and outcrops. These features are indicated on borehole geophysical logs as positive spikes on caliper, flowmeter, and resistivity logs. Horizontal flow features are responsible for the majority of lateral groundwater movement.

Hydrostratigraphy

Determining the location and continuity of horizontal "flow zones" in the dolomite aquifer was essential in order to model groundwater flow in the study area. The basis for the conceptual model of the aquifer was the work of Gianniny and others (1996). They combined stratigraphic studies of the dolomite at the outcrop scale with geophysical and hydrogeologic data at different scales to characterize the horizontal bedding plane features from a hydrostratigraphic point-of-view. Geophysical methods used by Gianniny and others (1996) included caliper, natural gamma, single-point resistivity, and spontaneous potential. Hydrogeologic characterization included fluid temperature measurements, heat-pulse flowmeter logging, and discrete-interval permeability tests.

Gianniny and others (1996) identified fourteen horizontal high-permeability features within the Silurian dolomite aquifer in the Sturgeon Bay area. The features are parallel to bedding and are best developed at contacts between lithologies. Five of the zones are continuous in the study area and can be correlated on both sides of Sturgeon Bay. Figure 5 is a cross-section showing the dolomite stratigraphy and horizontal flow features of an area near the City of Sturgeon Bay. The flow zones represent boundaries of contrasting lithologies, layers with high primary porosity (e.g. coquina-like packstones), or lithologies deposited in shallow or supratidal environments that were periodically exposed. In the vicinity of Sturgeon Bay, there are five flow zones that show lateral continuity (zones B, C, D, I, and J of Gianniny and others 1996). These zones were represented by continuous high-permeability layers in the numerical model.

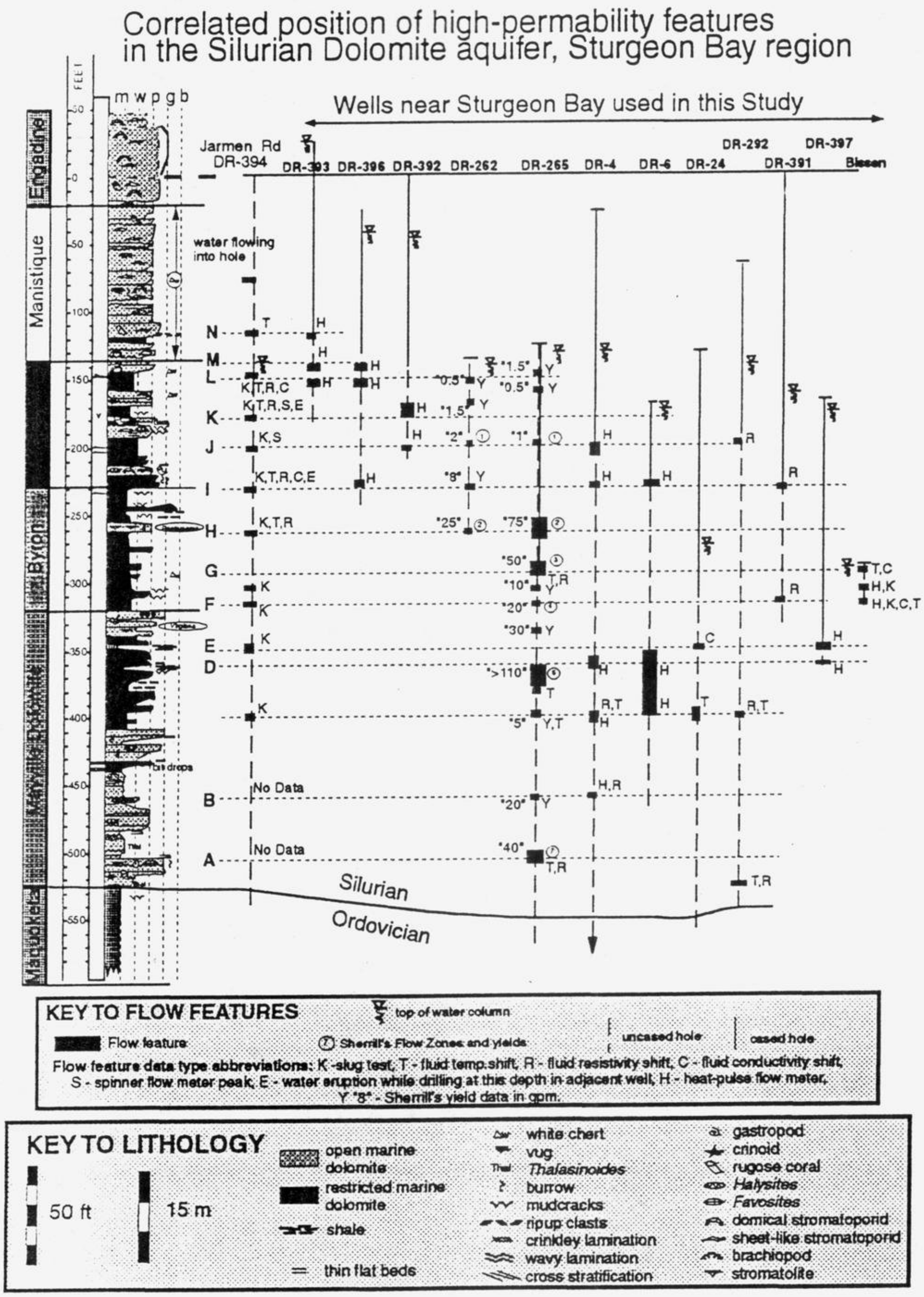


Figure 5. Correlated position of high-permeability features in the Sturgeon Bay area

HYDROGEOLOGIC CHARACTERIZATION

Water-Level Measurement

Water levels in the dolomite aquifer fluctuate seasonally over 30 m due to rapid recharge rates and low effective porosity in the fractured dolomite (Sherrill, 1978). The long-term hydrograph of well DR-265, a USGS observation well located near the Door County fairgrounds, (fig 6) shows the magnitude and rapid nature of water-level fluctuations. Water levels in this well fluctuate annually nearly 10 m. The annual low point occurs in September or October, and the annual high point occurs following rapid recharge in March or April. The dynamic nature of the dolomite aquifer suggests that a single potentiometric map, which represents a "snapshot" of the heads in the aquifer, might be of limited use for delineating zones of contribution to municipal wells.

In order to capture the range of water level fluctuations near Sturgeon Bay, we collected water-level data during two different seasons for use in the construction and calibration of the flow models and to provide insight into groundwater flow directions. During the summer of 1994, we contacted residents in the study area by mail and phone to seek permission to measure water levels in domestic, municipal, industrial, and monitoring wells. If their response was positive, a field crew visited the site and determined the surface elevation of the well top using an altimeter. Coordinates of each well were determined by digitizing the locations from topographic maps.

Field parties from the WGNHS measured water levels at two different times: November, 1994 and April, 1995. Water levels were measured using electrical or chalked water-level tapes after determining that the well was not currently pumping. The resulting hydraulic head information represents composite hydraulic head over the open interval of the borehole, commonly 50 to 100 m. The data were contoured using SURFER (Golden Software, 1994) and hand-edited. The contour lines were then digitized and plotted for use in the construction of the flow models.

The two seasonal potentiometric maps (fig 7) show the configuration of the potentiometric surface and transience of groundwater levels near Sturgeon Bay. Groundwater levels are highest near the crest of the peninsula, and slope toward surrounding surface-crater features: Lake Michigan to the east, Green Bay to the west, and Sturgeon Bay and the ship canal. In Spring, measured potentiometric levels exceed 206 m north of Sturgeon Bay and 210 m south of Sturgeon Bay. In Fall, potentiometric levels are several meters lower.

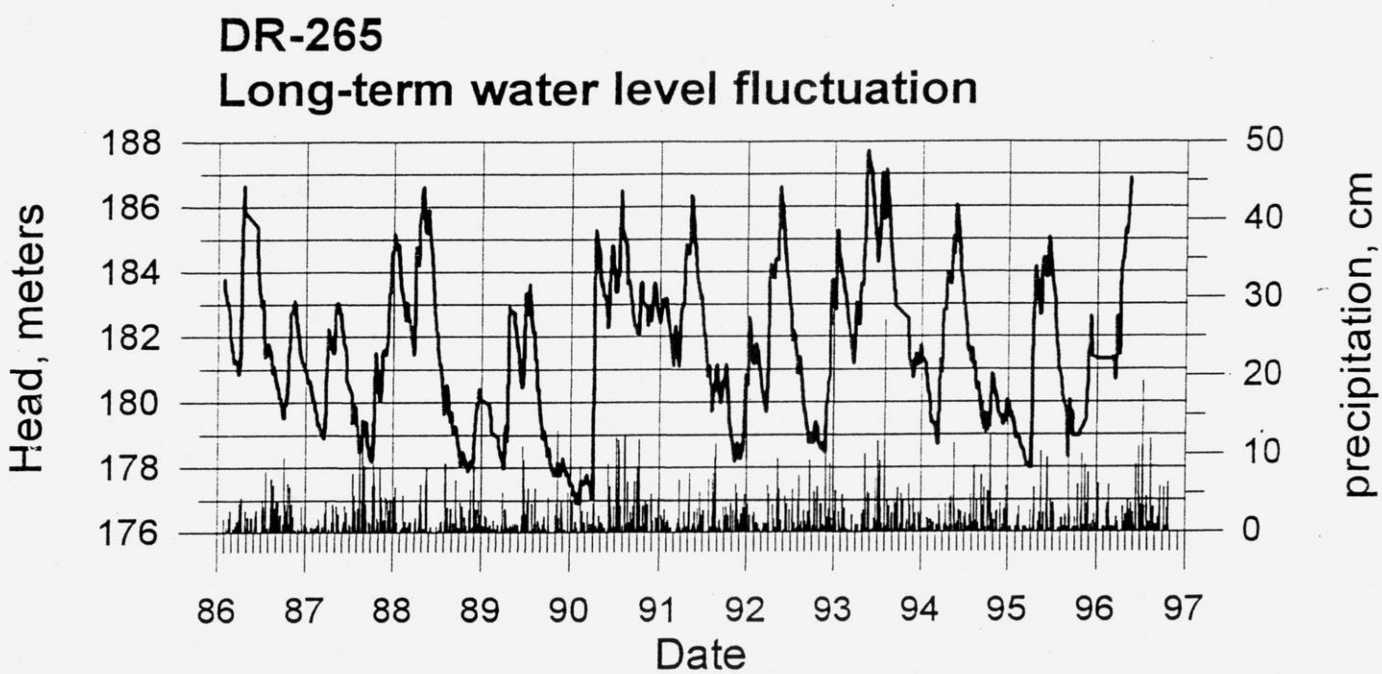
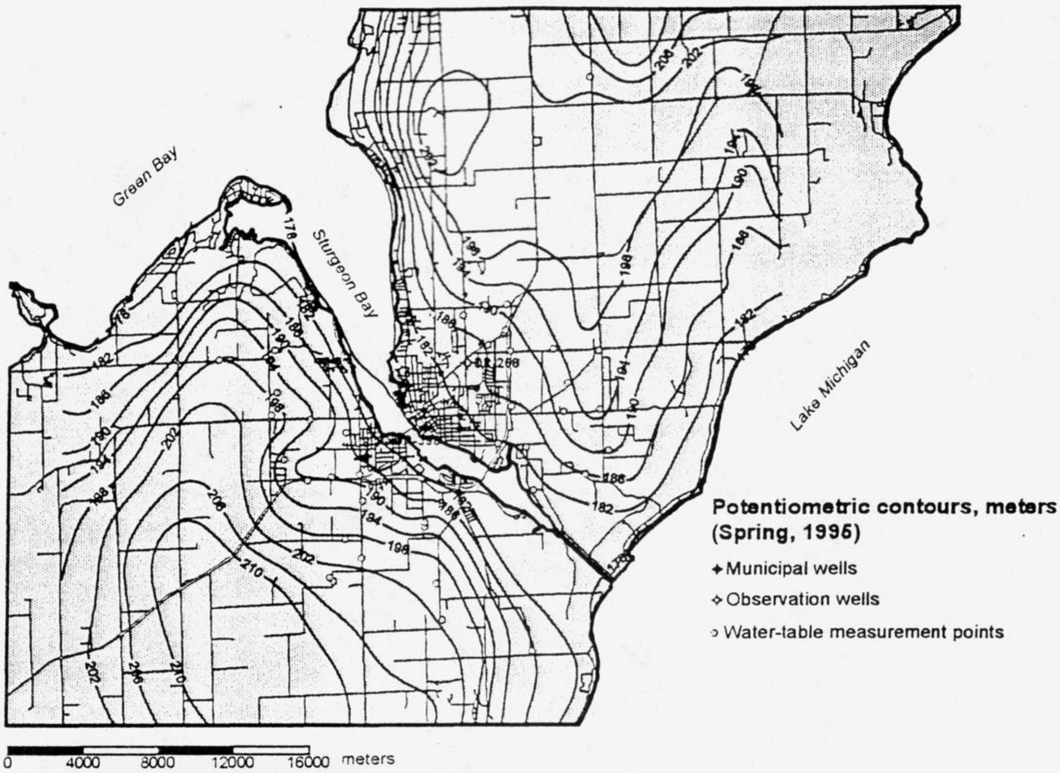


Figure 6. Long-term water-level fluctuations at well DR-265, near the Door County Fairgrounds



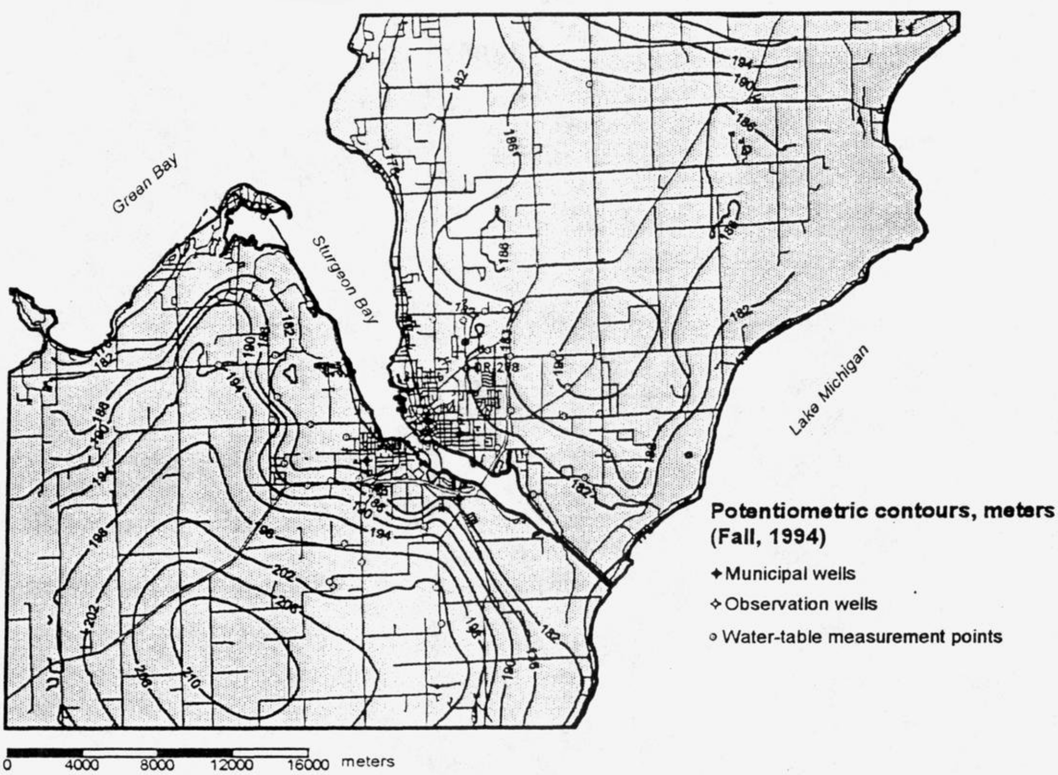


Figure 7. Potentiometric maps for Spring 1995 and Fall 1994 based on field-measured water levels

Estimation of Recharge Rates

Recharge in the Sturgeon Bay area varies in both space and time. Soil characteristics, topography, the presence of karst and fracture features, and the annual precipitation distribution work together to control groundwater recharge rates.

A soil-water mass balance analysis provided initial estimates of groundwater recharge rates. Thornthwaite and Mather's (1957) soil water balance model, as modified by Swanson (1996), was used to estimate potential percolation for each of 68 soil map units in the Sturgeon Bay area. The method is based on a form of the mass balance equation, for a specified time period:

$$S = P - R - E$$

where,

P = precipitation,
R = runoff,
E = evapotranspiration, and
S = increase in soil moisture storage.

Negative values of S represent soil moisture deficit and positive values represent potential percolation (Rushton and Ward, 1979). Each soil has a maximum soil moisture storage capacity, and percolation only takes place when the soil moisture storage capacity is exceeded. Swanson (1996) implemented the soil percolation model for a study of Dane County, Wisconsin, using a computer spreadsheet to calculate monthly and annual estimates of percolation for each soil unit. In the Sturgeon Bay area, we used the same approach using soils data from the Door County soils survey (Link and others, 1978).

The soil percolation model tends to underestimate groundwater recharge because it does not account for water lost to runoff from each soil unit. The model assumes that runoff leaves the system to surface water, when in fact most runoff in Door County becomes infiltration in adjacent soil units or through karst features. Nevertheless, the estimated deep infiltration values provide a starting point for developing the model recharge rates. Table 2 summarizes net annual percolation for each of the 68 soils units in the study area. Net annual percolation ranges from 5.7 to 9.3 in/yr (0.14 to 0.24 m/yr), and averages 7.9 in/yr (0.2 m/yr). For comparison, a previous study in northern Door County near Peninsula State Park determined an annual recharge rate of 12.9 in/yr (0.32 m/yr) (Bradbury, 1982), and Bradbury and Muldoon (1992) determined an annual recharge rate of about 10 in/yr (0.25 m/yr) in central Door County.

The soil infiltration model also indicated that most recharge in the study area should occur only in the spring and the fall, with little or no recharge during other months. In most months of the year the soil model indicates a soil moisture deficit, as evapotranspiration exceeds precipitation. Based on the soil infiltration model, all recharge an average year should occur only during March, April, October, and November, and table 2 shows these estimated monthly rates. During the winter (December through February) the ground is frozen and precipitation accumulates as snowpack. A major recharge event occurs in March and April as the soil thaws, the snowpack melts, and Spring rains occur. From May through September the weather is warm, and evapotranspiration generally exceeds precipitation. During October and November evapotranspiration decreases and Fall rains typically occur. Using both steady-state and transient models to determine seasonal recharge, Bradbury (1982) determined Spring recharge rates of 13.1 to 28.3 in/yr (0.33 to 0.72 m/yr) and Fall recharge rates of 2.8 in/yr (0.07 m/yr).

The presence of shallow bedrock, sinkholes, closed depressions, and other karst and fracture features also have significant effects on groundwater recharge in the study area, but the soil water balance method cannot simulate such features. In order to develop the final recharge distribution for the numerical groundwater model (discussed in the modeling section of this report) we assumed that the presence of karst and fracture features would locally increase groundwater recharge rates. We used maps of solution features at the surface by Stieglitz and Dueppen (1994) and Stieglitz and Johnson (1996), soil maps showing areas of thin or no soil, and fracture density maps at several scales, (Roffers, 1996) to estimate the areal variability of recharge. Areas of exposed dolomite, thin soils, and solution features or intense vertical fracturing were given the highest recharge rates. Areas with no solution features or low fracture density and thick soils were given the lowest rates.

Table 2. Results of soil percolation model

Net Annual Percolation for Soil Types							
Soil Type	Description	Net annual	annual	monthly percolation rates (m/d)			
			(m/d)	March	April	Oct	Nov
AdA	Allendale loamy sand	8.83	6.1E-04	4.9E-03	1.4E-03	0.0E+00	1.3E-03
ApC	Alpena gravelly sandy loam	7.79	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.9E-04
Ax	Angelica loam	7.79	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.9E-04
Be	Beaches	8.72	6.1E-04	4.8E-03	1.0E-03	0.0E+00	1.4E-03
Bn	Bonduel loam	7.79	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.9E-04
Bo	Bonduel Shallow Varient	9.26	6.4E-04	5.0E-03	1.1E-03	0.0E+00	1.6E-03
Bp	Bonduel Wet Varient loam	8.09	5.6E-04	4.7E-03	9.7E-04	0.0E+00	9.9E-04
BrB	Boyer loamy sand	8.20	5.7E-04	4.7E-03	9.7E-04	0.0E+00	1.1E-03
BrC	Boyer loamy sand	7.79	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.9E-04
BrD	Boyer loamy sand	7.16	5.0E-04	4.5E-03	8.6E-04	0.0E+00	5.9E-04
Ca	Carbondale Muck	8.09	5.6E-04	4.7E-03	9.7E-04	0.0E+00	9.9E-04
CcB	Casco sandy loam	8.20	5.7E-04	4.7E-03	9.7E-04	0.0E+00	1.1E-03
CcC2	Casco sandy loam	7.79	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.9E-04
Cm	Cathro Muck	8.09	5.6E-04	4.7E-03	9.7E-04	0.0E+00	9.9E-04
Cp	Chippeny Muck	8.09	5.6E-04	4.7E-03	9.7E-04	0.0E+00	9.9E-04
De	Deford loamy fine sand	9.26	6.4E-04	5.0E-03	1.1E-03	0.0E+00	1.6E-03
DuB	Duel loamy sand	8.42	5.9E-04	4.8E-03	1.0E-03	0.0E+00	1.2E-03
Dv	Duel Variant sandy loam	9.26	6.4E-04	5.0E-03	1.1E-03	0.0E+00	1.6E-03
EmA	Emmet sandy loam	9.26	6.4E-04	5.0E-03	1.1E-03	0.0E+00	1.6E-03
EmB	Emmet sandy loam	8.83	6.1E-04	4.9E-03	1.4E-03	0.0E+00	1.3E-03
EmC2	Emmet sandy loam	7.79	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.9E-04
EmD2	Emmet sandy loam	7.16	5.0E-04	4.5E-03	8.6E-04	0.0E+00	5.9E-04
EmE	Emmet sandy loam	6.75	4.7E-04	4.4E-03	8.2E-04	0.0E+00	4.1E-04
Fa	Fabius silt loam	7.76	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.6E-04
Fu	Fluvaquents	8.34	5.8E-04	4.8E-03	1.0E-03	0.0E+00	1.1E-03
KhA	Kewaunee silt loam	7.75	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.5E-04
KhB	Kewaunee silt loam	6.69	4.7E-04	4.4E-03	8.2E-04	0.0E+00	3.5E-04
KhC2	Kewaunee silt loam	7.50	5.2E-04	4.1E-03	7.0E-04	1.5E-04	1.3E-03
KkD3	Kewaunee soils	5.68	4.0E-04	3.5E-03	4.8E-04	0.0E+00	6.8E-04
YaA	Yahara fine sandy loam	8.83	6.1E-04	4.9E-03	1.4E-03	0.0E+00	1.3E-03
Yv	Yahara Variant silt loam	7.76	5.4E-04	4.6E-03	9.3E-04	0.0E+00	8.6E-04

KmB	Kiva sandy loam	8.20	5.7E-04	4.7E-03	9.7E-	0.0E+0	1.1E-03
KmC	Kiva sandy loam	7.79	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.9E-04
KoA	Kolberg silt loam	7.75	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.5E-04
KoB	Kolberg silt loam	6.69	4.7E-04	4.4E-03	8.2E-	0.0E+0	3.5E-04
KoC2	Kolberg silt loam	7.50	5.2E-04	4.1E-03	7.0E-	1.5E-04	1.3E-03
KvB	Kolberg Variant loam	6.69	4.7E-04	4.4E-03	8.2E-	0.0E+0	3.5E-04
KvC2	Kolberg Variant loam	7.50	5.2E-04	4.1E-03	7.0E-	1.5E-04	1.3E-03
LoA	Longrie loam	7.79	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.9E-04
LoB	Longrie loam	7.16	5.0E-04	4.5E-03	8.6E-	0.0E+0	5.9E-04
LoC	Longrie loam	7.50	5.2E-04	4.1E-03	7.0E-	1.5E-04	1.3E-03
McA	Manawa silt loam	7.76	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.6E-04
MeB	Manistee loamy sand	8.20	5.7E-04	4.7E-03	9.7E-	0.0E+0	1.1E-03
Mk	Markey Muck	8.09	5.6E-04	4.7E-03	9.7E-	0.0E+0	9.9E-04
NaB	Namur loam	6.63	4.6E-04	4.4E-03	8.2E-	0.0E+0	3.0E-04
NaC	Namur loam	7.50	5.2E-04	4.1E-03	7.0E-	1.5E-04	1.3E-03
Nv	Namur Variant loam	9.26	6.4E-04	5.0E-03	1.1E-	0.0E+0	1.6E-03
OmB	Omena sandy loam	8.20	5.7E-04	4.7E-03	9.7E-	0.0E+0	1.1E-03
OmC	Omena sandy loam	7.79	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.9E-04
OmD	Omena sandy loam	7.16	5.0E-04	4.5E-03	8.6E-	0.0E+0	5.9E-04
OvB	Omena Variant sandy loam	8.20	5.7E-04	4.7E-03	9.7E-	0.0E+0	1.1E-03
OzB	Omro silt loam	6.70	4.7E-04	4.5E-03	8.2E-	0.0E+0	3.6E-04
Pn	Pinconning loamy fine sand	9.26	6.4E-04	5.0E-03	1.1E-	0.0E+0	1.6E-03
Po	Poygan silty clay loam	7.76	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.6E-04
Rn	Rondeau Muck	8.09	5.6E-04	4.7E-03	9.7E-	0.0E+0	9.9E-04
RoB	Rousseau fine sand	8.40	5.8E-04	4.8E-03	1.0E-	0.0E+0	1.2E-03
RoC	Rousseau fine sand	7.55	5.3E-04	4.6E-03	9.1E-	0.0E+0	7.6E-04
RpC	Rousseau-Shawano fine sand	7.76	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.6E-04
RpD	Rousseau-Shawano fine sand	6.70	4.7E-04	4.5E-03	8.2E-	0.0E+0	3.6E-04
RrB	Rousseau-Deford fine sand	8.40	5.8E-04	4.8E-03	1.0E-	0.0E+0	1.2E-03
SnA	Sisson fine sandy loam	9.26	6.4E-04	5.0E-03	1.1E-	0.0E+0	1.6E-03
SnB	Sission fine sandy loam	8.00	5.6E-04	4.7E-03	9.5E-	0.0E+0	9.9E-04
SoA	Solona loam	7.06	4.9E-04	4.5E-03	8.6E-	0.0E+0	5.1E-04
Su	Suamico Muck	8.09	5.6E-04	4.7E-03	9.7E-	0.0E+0	9.9E-04
SvA	Summerville loam	7.79	5.4E-04	4.6E-03	9.3E-	0.0E+0	8.9E-04
SvB	Summerville loam	6.70	4.7E-04	4.5E-03	8.2E-	0.0E+0	3.6E-04
SvC	Summerville loam	7.50	5.2E-04	4.1E-03	7.0E-	1.5E-04	1.3E-03
SvD	Summerville loam	6.59	4.6E-04	3.8E-03	5.9E-	0.0E+0	1.1E-03

Determination of Hydraulic Conductivity

City well pumping test

Responses of local wells to municipal pumping show that the groundwater system is laterally well connected in the subsurface. In November, 1995, we monitored water levels in several deep wells in Sturgeon Bay while city well 3 was pumped at 1850 gpm (10,030 m3/d) for 4 hours. Responses to this pumping occurred in city wells 1 and 11 and in Dr-265 (fig 8). Significantly, 1.5 ft (0.45 m) of drawdown occurred at city well 1, which is on the opposite side of the Sturgeon Bay channel from city well 3. The rapid response of well 1 shows that the Sturgeon Bay channel does not act as a significant boundary to lateral groundwater movement beneath it. This test, analyzed using the Theis (1935) method assumes a fully-penetrated confined aquifer, and yields transmissivities ranging from 1750 to 2120 m2/d, and storage coefficients ranging from .00003 to .00006.

In large-scale tests, and as a composite unit, the dolomite aquifer at Sturgeon Bay responds as a classic porous media. Figure 9 shows time-drawdown plot of the pumping test response at the three observation wells. Data are plotted as drawdown (s) versus t/r2, where t is time since pumping started and r is the radial distance from the pumped well to the observation point. Data from two of the wells, 11 and Dr-265, fall almost on top of each other, and data from well 1 approach the other data at late times. All three data sets approximate the classic Theis curve, suggesting that at a regional scale the dolomite aquifer responds as a uniform porous medium. However, this assumption breaks down at the smaller scales needed for wellhead protection and transport studies.

Hydraulic conductivity tests using straddle packers

A series of stratified pumping tests on well Dr-265 (Door County fairgrounds) and Dr-398 (Ottumba Park) using straddle packers demonstrates the importance of near-horizontal fracture zones in controlling groundwater flow in the study area. The tests were designed to measure the hydraulic conductivity of specific fractured and non-fractured zones identified on borehole geophysical logs. The tests also provided reliable measurements of the vertical distribution of hydraulic head at the two sites.

We conducted the packer tests using a packer rig and accessory equipment provided by the U S Geological Survey. The packer string consisted of two gas-inflated 6 in (0.15 m) diameter rubberized packers suspended on drill rod, with a submersible pump mounted between the two packers. The length of the open interval between the packers was 9.6 ft (2.9 m), and was the minimum spacing possible with the available equipment. Submersible pressure transducers mounted just above, between, and just below the two packers monitored hydraulic head during the tests; a digital datalogger recorded the pressure readings. For each test, the packer string was lowered to a pre-selected testing depth based on downhole geophysical data. After positioning the assembly, we inflated the packers while monitoring hydraulic head, then waited for the head readings to stabilize in the packed zone. After stabilization, we pumped the packed intervals for

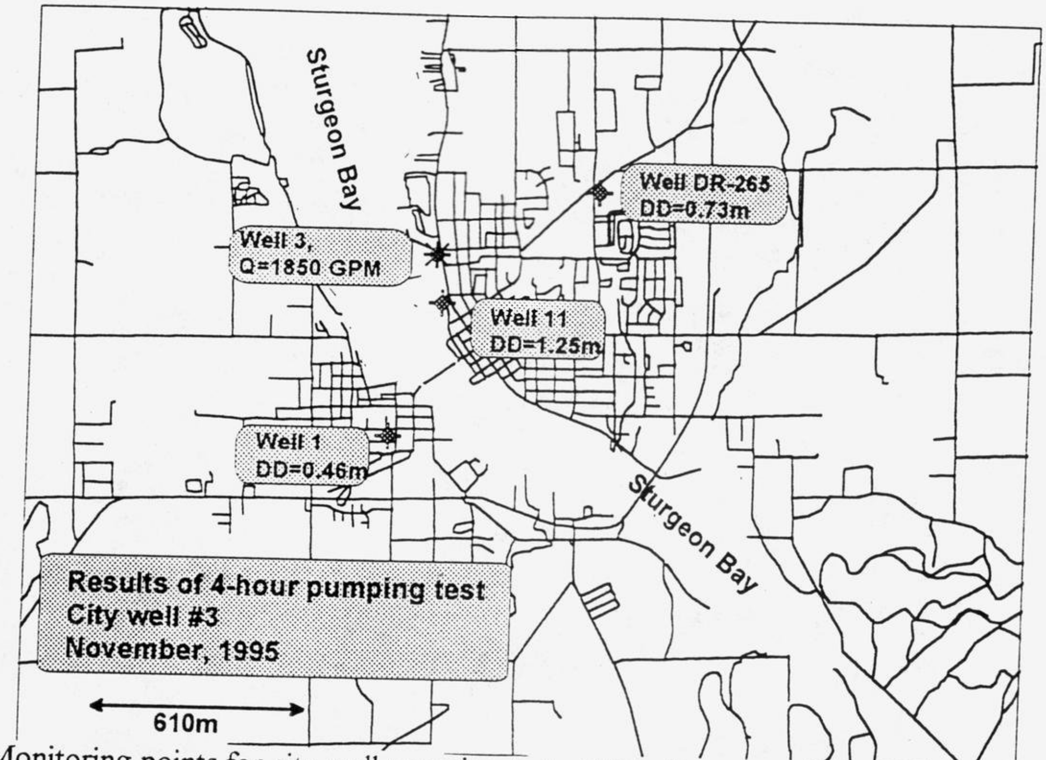


Figure 8. Monitoring points for city well pumping test. DD indicates maximum drawdowns (meters) at the various monitoring points.

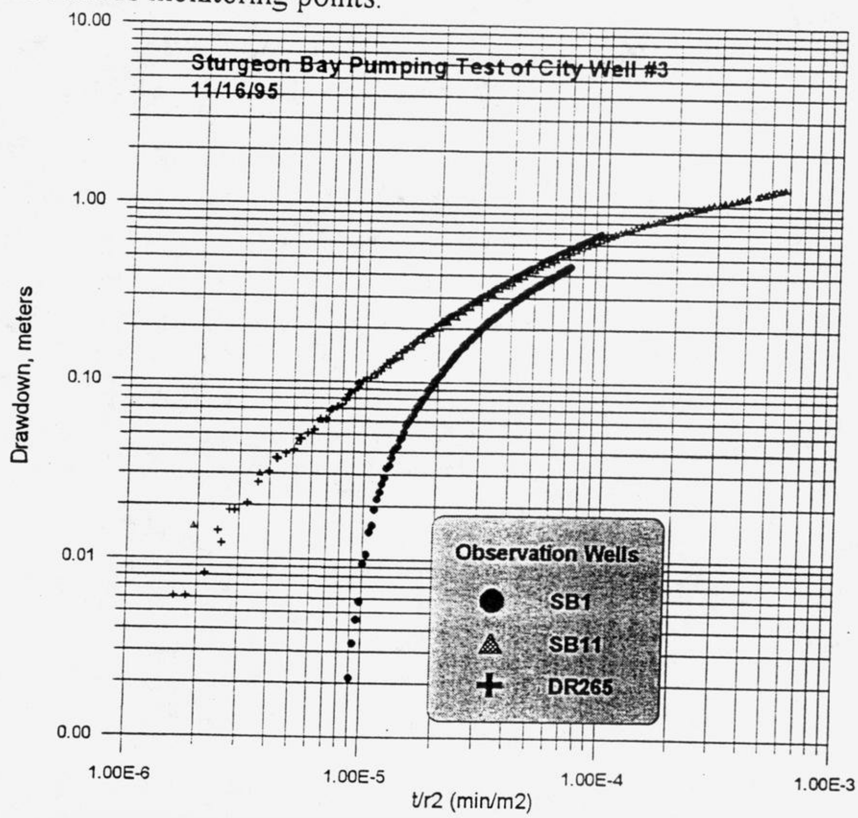


Figure 9. Drawdown versus t/r2 for the city well pumping test

periods ranging from a few minutes to several hours while monitoring drawdown above, between, and below the two packers. Pump discharge was measured using an in-line flowmeter, and discharge rates ranged from about 5 to 50 gallons per minute (27.1 to 271 m^3/day). For each interval we estimated hydraulic conductivity using the Theis nonequilibrium method (Theis, 1935) assuming an aquifer thickness equal to the packer separation. Some nonfractured intervals did not yield enough water to sustain pumping; we analyzed these as slug tests using the Hvorslev method (Hvorslev, 1951).

The packer test results clearly demonstrate the vertical heterogeneity of the dolomite aquifer at Sturgeon Bay. Hydraulic conductivities determined from the packer tests (table 3) range over six orders of magnitude, from .0006 to 70 m/day ($2x10^{-8}$ to $3x10^{-3}$ ft/sec). Figure 10 shows the results of the packers tests in well Dr-256; results in Dr-398 are similar. The highest values of hydraulic conductivity, generally above 0.1 m/day, are clearly associated with fracture zones, as shown by test zones A, D, E, F, and J on figure 10. The conductivity of unfractured dolomite, measured at this scale, is about $1.5x10^{-2}$ m/d, as shown by test zones C and G. The base of Dr-265 penetrates shale of the Maquoketa Fm; this material has a hydraulic conductivity of less than 10^{-3} m/day (zone L on figure 10).

It is important to point out that these packer results average the hydraulic conductivity of the rock over the entire thickness of the packed zone (9.6 ft). We hypothesize that the hydraulic conductivity of these intervals is almost entirely due to near-horizontal fractures, while the rock matrix yields little water. Therefore, the packer tests at this scale probably underestimate the hydraulic conductivity of individual fractures. However, these tests are consistent with the discretization scale of the numerical groundwater flow model constructed for the Sturgeon Bay study.

Table 3. Results of straddle packer tests in wells Dr-265 and Dr-398.

Well DR-265				
test zone	midpoint elevation, m	K (ft/sec)	K (m/day)	log k (m/day)
A	135.0	1.4E-04	3.6E+00	0.55
B	125.2	2.4E-06	6.3E-02	-1.20
C	114.9	1.1E-06	2.8E-02	-1.55
D	107.1	1.9E-03	4.9E+01	1.69
E	103.0	2.5E-03	6.6E+01	1.82
F	98.4	9.8E-04	2.6E+01	1.41
G	92.9	6.8E-07	1.8E-02	-1.75
H	87.4	8.5E-06	2.2E-01	-0.65
I	75.9	7.8E-06	2.1E-01	-0.69
J	70.4	2.2E-03	5.8E+01	1.76
K	60.6	7.0E-08	1.9E-03	-2.73
L	57.8	2.3E-08	6.1E-04	-3.22
Well DR-398				
test zone	midpoint elevation, m	K (ft/sec)	K (m/day)	log k (m/day)
A	168.3	8.3E-04	2.2E+01	1.34
B	162.1	6.0E-04	1.6E+01	1.20
C	159.0	2.5E-07	6.6E-03	-2.18
D	149.6	3.3E-04	8.6E+00	0.94
E	141.6	4.8E-07	1.3E-02	-1.90
F	134.3	1.8E-06	4.8E-02	-1.32
G	120.9	2.3E-04	6.0E+00	0.78
H	109.9	4.7E-04	1.2E+01	1.09
I	105.4	5.2E-05	1.4E+00	0.14

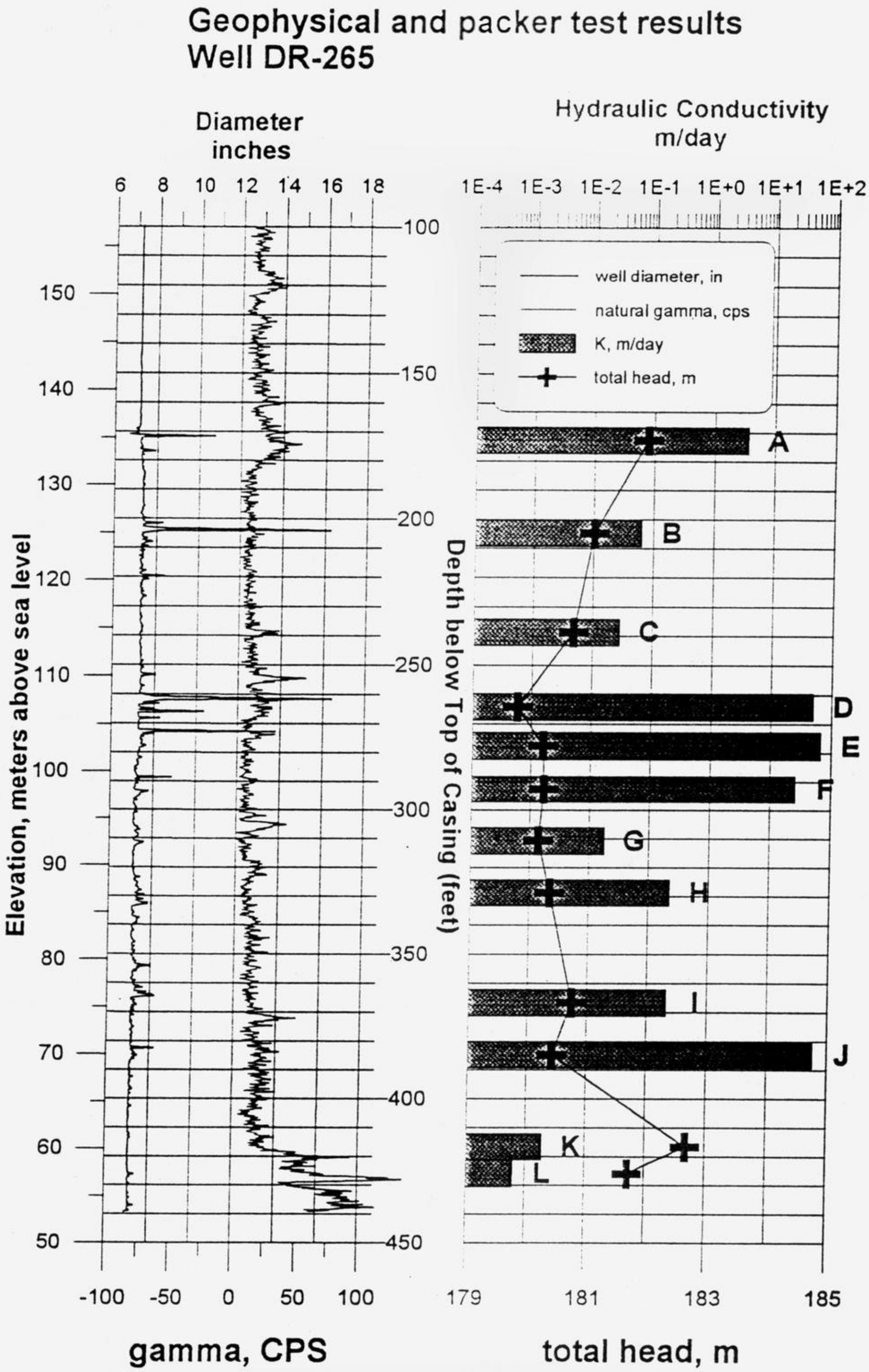


Figure 10. Results of packer tests on well DR-265. Test interval is 9.6 ft (2.9m).

Hydraulic heads measured during the packer tests show that significant vertical hydraulic gradients exist in the dolomite. On figure 10 the total hydraulic head (right-hand curve) in Dr-265 decreases from about 182 m above sea level at test zone A, at 135 m elevation, to under 180 m at test zone D, at 108 m elevation. Below test zone D the head increases steadily to zone K (elevation 58 m) at the top of the Maquoketa Formation. The vertical hydraulic gradient is thus downward (water moves downward) in the upper part of the aquifer, and upward (water moves upward) in the lower part of the aquifer. Significantly, the lowest total head coincides with the position of highest hydraulic conductivity, a large conductive fracture at elevation 108 m in Dr-265.

Specific-Capacity Estimates

Specific-capacity tests in private domestic wells provide additional information on the distribution of hydraulic conductivity in the Sturgeon Bay area. Well contractors commonly measure the specific capacity of private wells, defined as the sustainable pumping rate divided by the drawdown in the well at a quasi steady state, as a guide to predicting well performance. Many investigators have used specific-capacity data to estimate transmissivity and hydraulic conductivity (Bradbury and Rothschild, 1985). We used the TGUESS code of Bradbury and Rothschild (1985) to estimate hydraulic conductivity of 350 wells in the Sturgeon Bay area. These estimates are based on many simplifying assumptions, and are only approximate, but the results can be used to identify regional averages and spatial trends in areas where more rigorous aquifer tests are not abundant.

The specific capacity estimates yield a geometric mean hydraulic conductivity of 0.3 m/day ($1x10^{-5}$ ft/sec) for the Sturgeon Bay area, and represents a composite average over the entire thickness of the dolomite aquifer. Table 4 summarizes the test results.

Table 4. Results of hydraulic conductivity estimates based on specific capacity tests.

Statistic	**ft/sec**	**m/day**
number of tests	350	350
minimum	$4.1x10^{-7}$	.011
maximum	.008	210
mean	$9.5x10^{-5}$	2.5
geometric mean	$1x10^{-5}$	0.27

Although no major spatial trends are apparent in the composite hydraulic conductivity across the study area (fig 11), a number of higher hydraulic conductivities occur just north and south of the Sturgeon Bay city limits. Unfortunately there are few private wells, and thus few data points, within the city limits with which to confirm this trend.

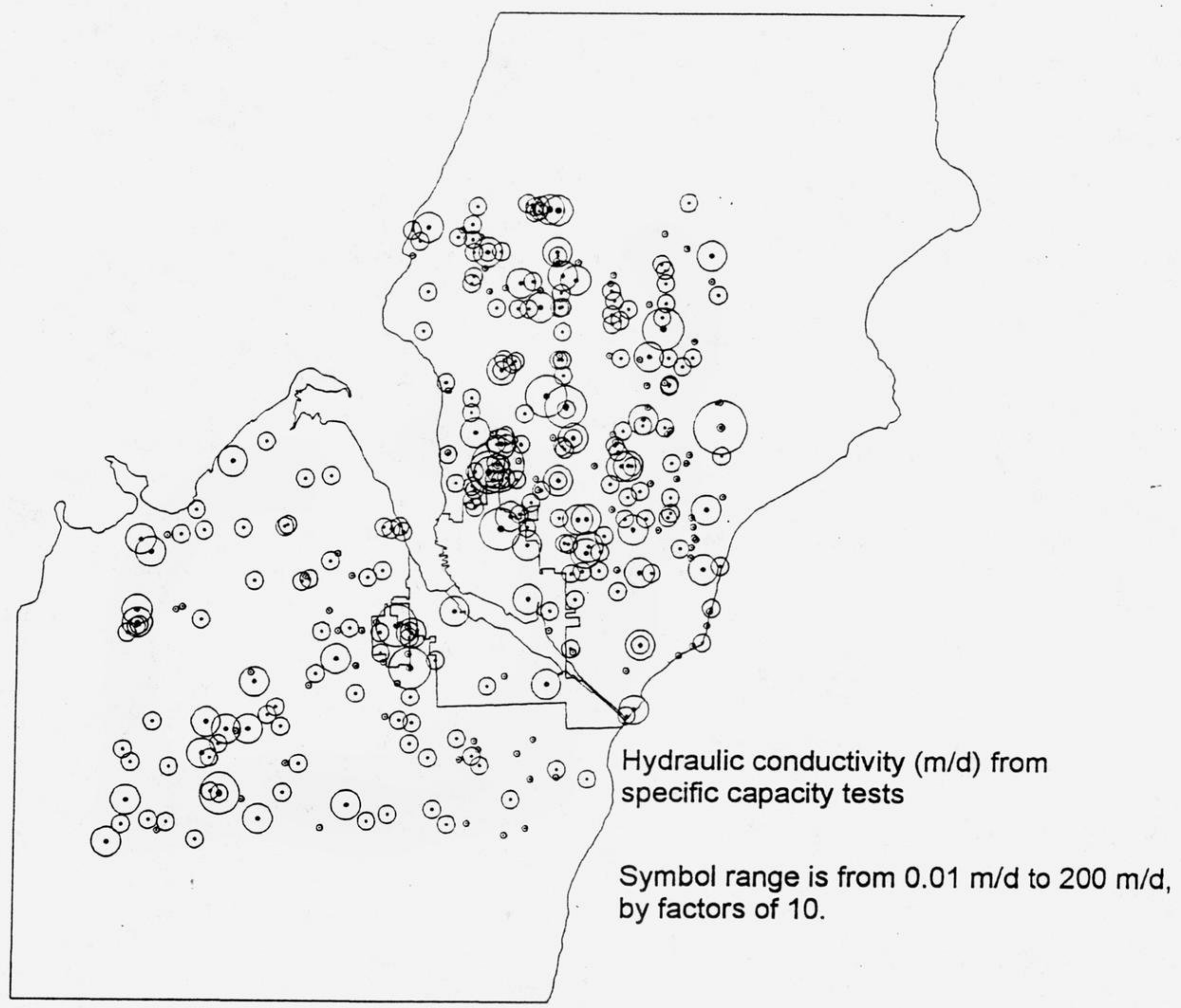


Figure 11. Results of hydraulic conductivity tests based on specific capacity of private wells

Groundwater Geochemistry

The geochemistry and isotopic signature of groundwater can often provide insight to groundwater origin and age (Bradbury and others, 1991). Variations in the geology, flow paths, and distance to recharge areas often influence the geochemistry of water produced by wells. Objectives of major-ion analyses of groundwater in the Sturgeon Bay project were to evaluate the overall water chemistry of groundwater in the Sturgeon Bay area, to observe possible differences in source areas between wells, and to evaluate the possibility of rapid conduit flow to the wells. Objectives of the isotopic analyses were to determine the approximate relative age of water produced by the wells and to look for evidence of surface water being drawn into the wells. Untreated water samples were obtained from the operating Sturgeon Bay municipal wells, and from surface water, during the winter and summer of 1995. Sample analyses were performed by the University of Wisconsin-Extension Soil and Plant Analysis Laboratory (anions and cations) and the University of Waterloo (Ontario) Isotope Laboratory (isotopes). Mineral saturation indices (the saturation state of the water relative to common mineral species) were determined using the PHREEQC speciation code of Parkhurst (1995).

Major ion chemistry

Water produced by the Sturgeon Bay wells has major-ion chemistry typical of carbonate-rock environments (table 5). The dominant dissolved ions are calcium, magnesium, bicarbonate, chloride, and sulfate. The water is slightly alkaline and contains measurable dissolved oxygen. Mineral saturation indices (table 6) are near zero for calcite and dolomite (positive numbers indicate oversaturation and negative numbers indicate undersaturation), suggesting that the water is at or near chemical equilibrium with the rock of the dolomite aquifer.

Overall geochemistry varies slightly between the municipal wells. Figure 12 is a milliequivalent bar graph showing the relative distributions of cations (left bars) and anions (right bars) in the city wells. Well 6 has the highest concentrations of dissolved species, suggesting that groundwater flow paths to well 6 might be longer than for other wells. The cations and anions for well 11 are clearly out of balance, possibly because well 11 was stagnant and out of service when sampled and it was impossible to flush the well prior to sampling. Surface water in Sturgeon Bay clearly has fewer total dissolved constituents than local groundwater.

Isotope chemistry

Isotopic data (table 7) suggest that the water produced by the municipal wells is relatively young and contains little or no surface water. Tritium (^3H) is a general indicator of groundwater age (Bradbury, 1991). Tritium (^3H) is a radioactive isotope of hydrogen which occurs naturally in trace quantities in the atmosphere and water. Atomic weapons testing during the 1960's elevated tritium levels in the atmosphere and in groundwater recharged since that time. Modern precipitation contains between 10 and 20 TU (tritium units), and the tritium content declines with time through radioactive decay. In general, groundwater having less than 2 TU is older than about

35 years, while water with less than 0.2 TU is older than about 50 years. Tritium concentrations in the Sturgeon Bay municipal wells range from about 9 to 16 tritium units and are in the range of current precipitation. These tritium data support the hypothesis that travel times from recharge to the wells are relatively short.

The stable isotopes oxygen-18 (^{18}O) and deuterium (^{2}H) suggest that the Sturgeon Bay wells do not produce significant quantities of local surface water. The ^{18}O:^{2}H ratio for meteoric water and for well-mixed groundwater is fairly constant throughout the world, and deviations from the world-wide meteoric water line usually suggest isotopic fractionation due to physical processes such as evaporation from open water. Figure 13 is a plot of ^{18}O vs ^{2}H concentrations for the Sturgeon Bay data. Groundwater from the municipal wells plots just to the left of the world-wide meteoric water line, while local surface water plots significantly to the right of the line. These results do not indicate any significant groundwater-surface water mixing in the Sturgeon Bay wells.

Table 5. Geochemical analyses of water produced by Sturgeon Bay municipal wells

City well no.	Sample	Temp	Cond.	pH	D.O.	Ca	Mg	Na	K	Fe	Mn	NO3-N	Cl	Alkalinity, as CaCO3	SO4
	Date	°C	mS/cm		mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/L	(mg/l)
3	11-4-95	9.9	0.63	7.44	4.4	71.0	33.6	6.7	0.9	0.01	<0.00	2.5	22.5	286	14.7
6	10-5-95	9.2	0.95	7.08	2.6	84.9	36.9	38.8	2.0	0.07	<0.00	2.8	110.4	328	18.8
7	10-4-95	10.5	0.74	7.24	2.5	76.0	34.4	19.8	1.2	<0.01	<0.00	3.1	62.8	300	16.1
8	10-4-95	9.2	0.60	7.31	2.0	67.6	32.9	7.6	1.8	<0.01	0.012	2.7	21.9	279	13.5
10	10-5-95	8.8	0.54	7.29	1.3	63.6	28.0	7.5	1.5	<0.01	0.003	1.0	13.4	457	12.5
11	10-4-95	14.8	0.63	7.41	1.4	67.3	33.4	9.0	1.0	0.34	0.036	1.8	34.0	79	13.9
bay water	10-4-95	16.2	0.30	8.57	-	35.5	12.4	6.6	1.2	0.02	<0.00	<0.5	27.0	114	18.0

Table 6. Mineral saturation indices, calculated using PHREEQC

Well	Sample	Saturation indices			$_{P}CO2$
	Date	calcite	dolomite	gypsum	bars
3	11-4-95	0.04	-0.12	-2.32	-2.09
6	10-5-95	-0.22	-0.69	-2.19	-1.68
7	10-4-95	-0.11	-0.43	-2.28	-1.87
8	10-4-95	-0.12	-0.45	-2.37	-1.97
10	10-5-95	0.03	-0.21	-2.45	-1.74
11	10-4-95	-0.47	-1.03	-2.34	-2.59
Bay	10-4-95	0.60	0.98	-2.46	-3.60

Table 7. Isotopic contents of water from Sturgeon Bay municipal wells and local surface waters

Sample location	sample date	^{18}O	^{2}H	^{3}H	triterr
		permil SMOW		tritium units	
Green Bay @ Big Quarry	11/06/94	-6.31	-47.26	19.6	1.4
Sturgeon Bay @ downtown	11/06/94	-6.51	-50.73	19.4	1.4
Lake Michigan @ Coast Guard Sta.	11/06/94	-6.26	-47.03	21.4	1.6
city well 3	01/03/95	-11.07	-72.30	14.3	1.1
city well 3	07/05/95	-11.00	-72.92	16.2	1.2
city well 6	01/03/95	-10.75	-71.62	14.0	1.0
city well 7	01/03/95	-10.86	-73.37	13.0	1.0
city well 7	07/05/95	-10.66	-72.78	14.2	1.1
city well 8	01/03/95	-10.92	-74.44	10.4	0.8
city well 10	01/03/95	-10.72	-70.83	8.8	0.7
city well 10	07/05/95	-10.60	-72.63	11.2	0.9

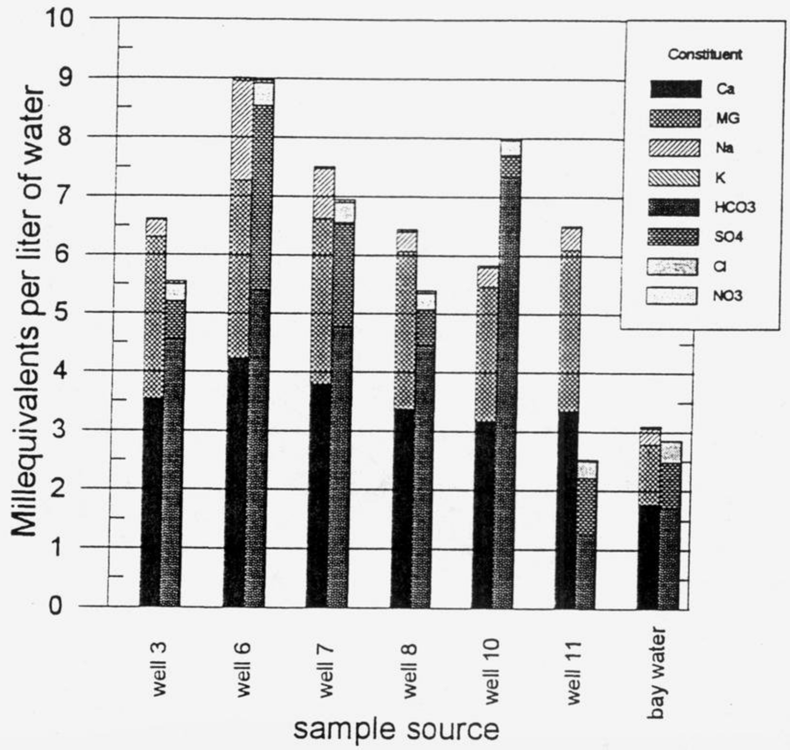


Figure 12. Relative distribution of anions and cations in local groundwater. Left bars indicate cations; right bars indicate anions.

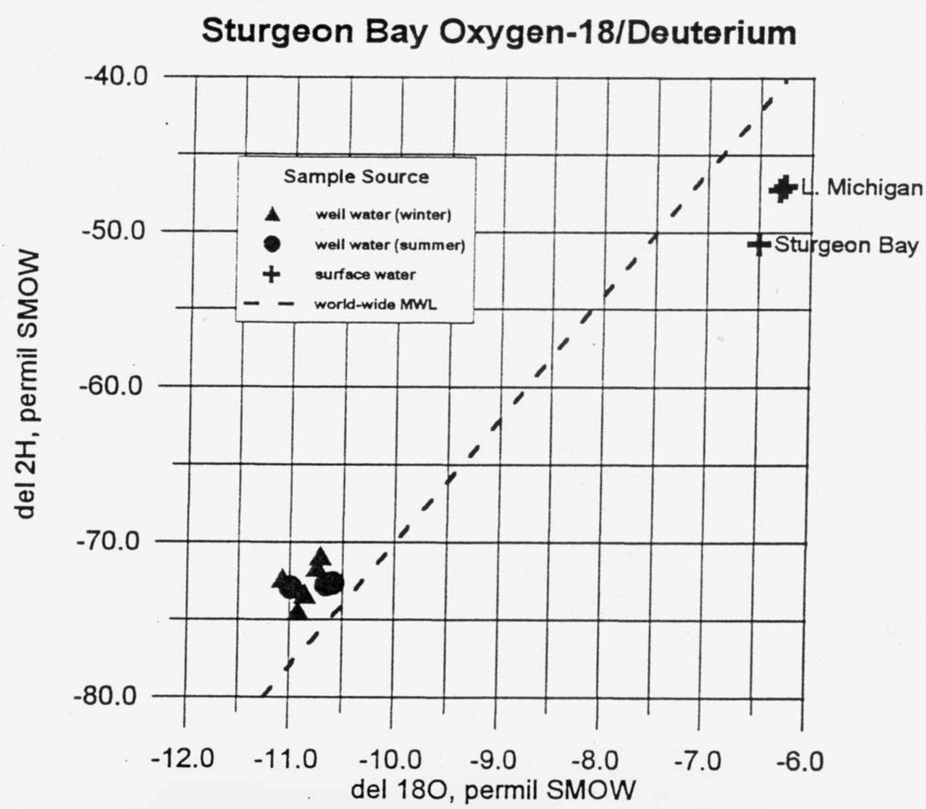


Figure 13. ^{18}O versus ^{2}H for Sturgeon Bay groundwater

Contaminant Source Inventory

Inventories of potential contaminant sources are an important part of any groundwater protection scheme. Comparison of potential sources to delineated well capture zones allows regulators to assess the relative potential for groundwater contamination and to prioritize site remediation efforts. We conducted a search of public records, including files of the Wisconsin Department of Natural Resources, the Sturgeon Bay Utilities, the Sturgeon Bay Fire Department, and the Wisconsin Department of Labor, Industry, and Human Relations (now Commerce). Potential contamination sources were inventoried and plotted using a GIS system.

Many potential contaminant sources exist in the vicinity of Sturgeon Bay (fig 14). Such sites include active and abandoned landfills, waste lagoons, septage spreading sites, leaking underground storage tanks, junk yards, and spill sites. In addition, Department of Commerce records report many registered underground storage tanks in the Sturgeon Bay area. It is important to understand that these sites represent *potential* sources; many may never contaminate groundwater.

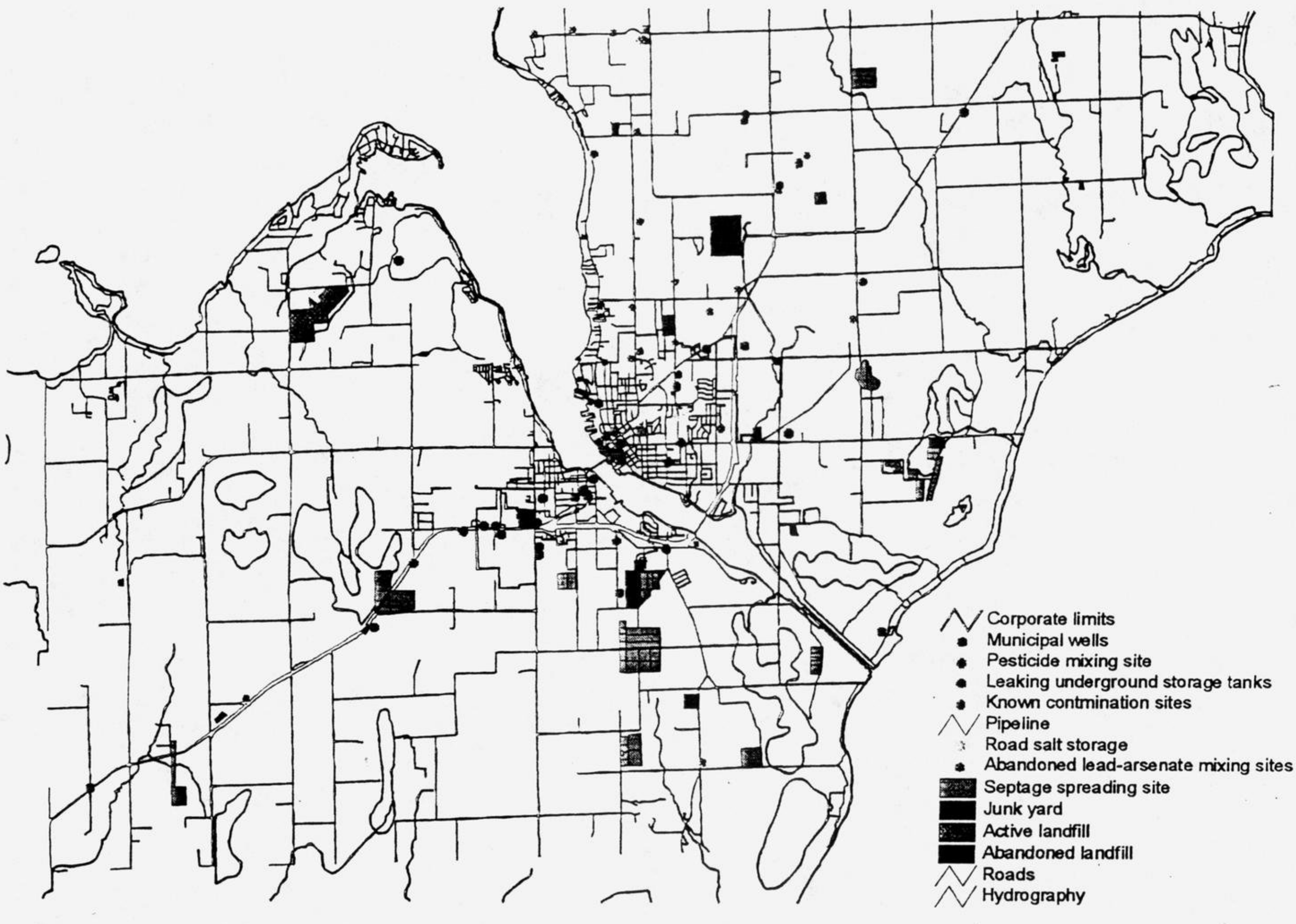


Figure 14. Major potential contamination sources

GROUNDWATER MODELING

Approach

Groundwater modeling is the most precise method for delineating zones of contribution for pumping wells in complex hydrogeologic settings (Bradbury and others, 1991). Groundwater modeling in fractured carbonate rock can be particularly difficult because of the heterogeneity and extreme anisotropy of many fractured-rock systems. Initially, in designing this project, we intended to build a discrete fracture network model of the fracture system using the Fracman/Mafic code of Golder Associates (1995). This code models fractures as discrete planes in space, and builds fracture networks based on field-measured fracture properties. However, during the project we discovered that the FRACMAN/MAFIC code would not easily accommodate the complex boundary conditions and geology of the Sturgeon Bay area. Furthermore, as the project progressed, we identified a conceptual model (described below) that could be solved using standard finite-difference techniques.

Project modeling consisted of three phases. In phase one of the modeling, we used a simple two-dimensional particle-tracking code (the WHPA code of Blandford and Huyakorn, 1993) to approximate zones of contribution for each municipal well based on the configuration of the field-measured potentiometric surface. The purpose of this preliminary modeling was, first, to delineate the approximate area of the capture zones as an aid in designing the more-complex three-dimensional model, and, second, to provide a basis of comparison between results of the simple models often used in wellhead protection studies and resulting ZOC's from the more complex three-dimensional model.

The second, and main, modeling phase used a transient, three-dimensional code (the MODFLOW code of McDonald and Harbaugh, 1988) to more completely simulate the hydrogeologic system and to delineate capture zones with more accuracy than the WHPA code. The purpose of this model was to approximate as nearly as possible the complexity of the groundwater system, with special attention to three-dimensional and transient effects.

A third modeling phase used a simple, two-dimensional fracture network model (using the SDF code of Rouleau, 1988) to study the lateral spreading of water in the fractured unsaturated zone above the water table.

Conceptual Model

Any numerical model must be based on a conceptual model of the aquifer. The conceptual model is created using geological and hydrogeological data that are collected in the field and office. The numerical model is a translation of the conceptual model into a mathematical formulation that can be solved by the computer code.

The basis of the conceptual model for the Sturgeon Bay study includes the stratigraphic study of

Gianniny and others (1996) and previous work by Bradbury and Muldoon (1992) and Sherrill (1978). Our conceptual model of the aquifer serving the Sturgeon Bay municipal wells is as follows: The aquifer is composed of multiple dipping layers of dolomite with an uppermost layer of unlithified material (sand and gravel and lacustrine deposits). The uppermost layer intersects the dolomite unconformably because of erosion of the bedrock by streams and glaciation. Fractures and solution openings in the dolomite are the dominant controls on groundwater movement. A series of planar, dipping, high-hydraulic conductivity fracture zones control groundwater movement and conduct groundwater to the municipal wells. These zones are regionally continuous for up to several kilometers (Gianniny and others, 1996). Groundwater moves rapidly horizontally through these flow zones. Between the flow zones, groundwater movement is mostly vertical, through numerous vertical fractures. Recharge to the groundwater system occurs through these vertical fractures and associated solution features. The distribution of vertical fractures is much denser than the distribution of horizontal flow zones, but the vertical fractures tend to have smaller apertures and are probably poorly integrated and connected at depth.

Hydraulically, the dolomite aquifer near Sturgeon Bay operates as a classic dual-porosity system. The fractures, and in particular the horizontal flow zones, control groundwater movement and are responsible for almost all of the aquifer's transmissivity. The dolomite matrix, between the fractures and flow zones, contributes little transmissivity to the system but is important for groundwater storage. Flow in the dolomite aquifer is primarily in thin, sub-horizontal solution features that in many cases follow bedding planes and formation boundaries. Vertical fractures control the flow between horizontal flow zones but we consider them to be of lesser importance for horizontal groundwater flow.

The recharge rate is variable because of different amounts of runoff versus infiltration of precipitation. Areas of exposed fractured dolomite and areas of karst features receive more recharge than areas where the soil is relatively thick or the rock is relatively unfractured. Boundary conditions include specified head boundaries in Lake Michigan and Green Bay, head-dependent flux boundaries in Sturgeon Bay and streams in the model area, and no-flow boundaries at groundwater divides. Pumping rates for City of Sturgeon Bay municipal wells were averaged over a year to give steady-state withdrawal rates for each well.

WHPA model

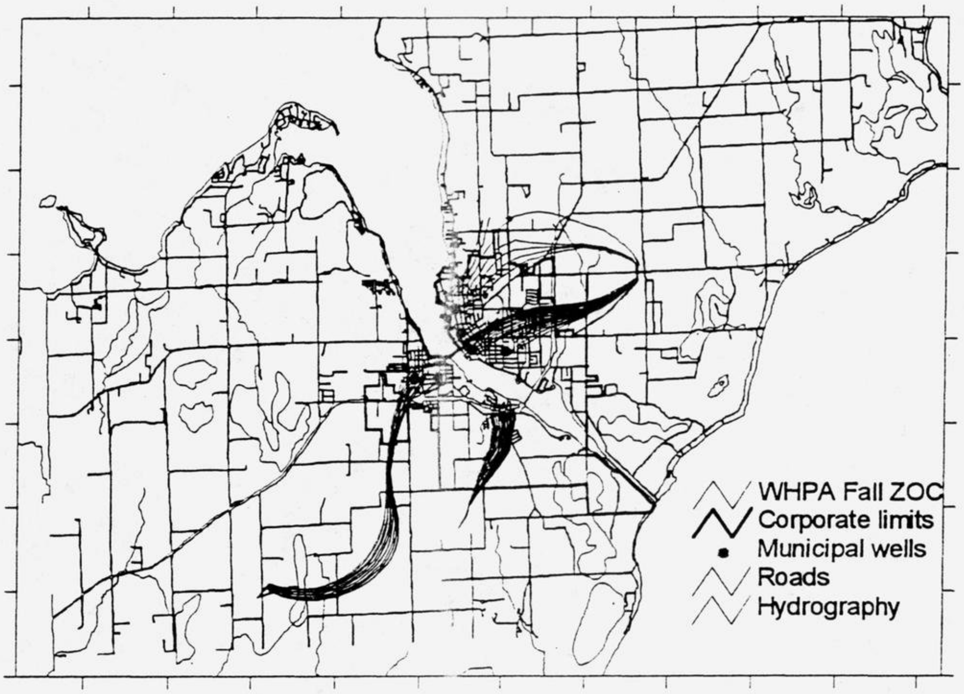
The first phase of modeling, which began after the completion of the potentiometric maps, used the U.S. Environmental Protection Agency Well Head Protection Area code (Blandford and Huyakorn, 1993). The code (WHPA) is a semi-analytical model that calculates steady-state capture zones for wells in two-dimensional, homogeneous aquifers. The numerical option of the General Particle Tracking (GPTRAC) module was used for modeling the aquifer in this study. Essentially, this model simply tracks hypothetical "particles" of water through the field-delineated flow field, represented by the potentiometric surface. The model is not calibrated in the usual sense because hydraulic heads are an input parameter and the model does not maintain mass balance.

The WHPA code assumes the dolomite aquifer is a single layer with uniform porosity, hydraulic conductivity, and recharge. Head values were interpolated by the model from the contours on the field-measured spring and fall potentiometric maps. The hydraulic gradient was calculated by the model based on the head field. The model calculates the velocity field from the input parameters and moves particles through the field. Input to the model is shown in Table 8.

Table 8. Input parameter values for the WHPA model.

Parameter	Value used in WHPA model
Transmissivity	500 m^2/d
Saturated Thickness	100 meters
Porosity	0.01
Length of Particle Tracking	2 years

Figure 15 shows the results of the WHPA modeling using the fall and spring potentiometric surface. Particles were tracked backwards from each of the five Sturgeon Bay wells until they reached a potentiometric divide. The Fall and Spring particle tracks are somewhat different, particularly with respect to the capture zone for city well three, which extends much farther to the south in the Fall than in the Spring. Figure 16 shows a composite WHPA capture zone for all Sturgeon Bay municipal wells. This area encompasses particle paths from both the Spring and Fall potentiometric maps. This area represents a preliminary estimate of the extent of the combined capture zone for the Sturgeon Bay wells.



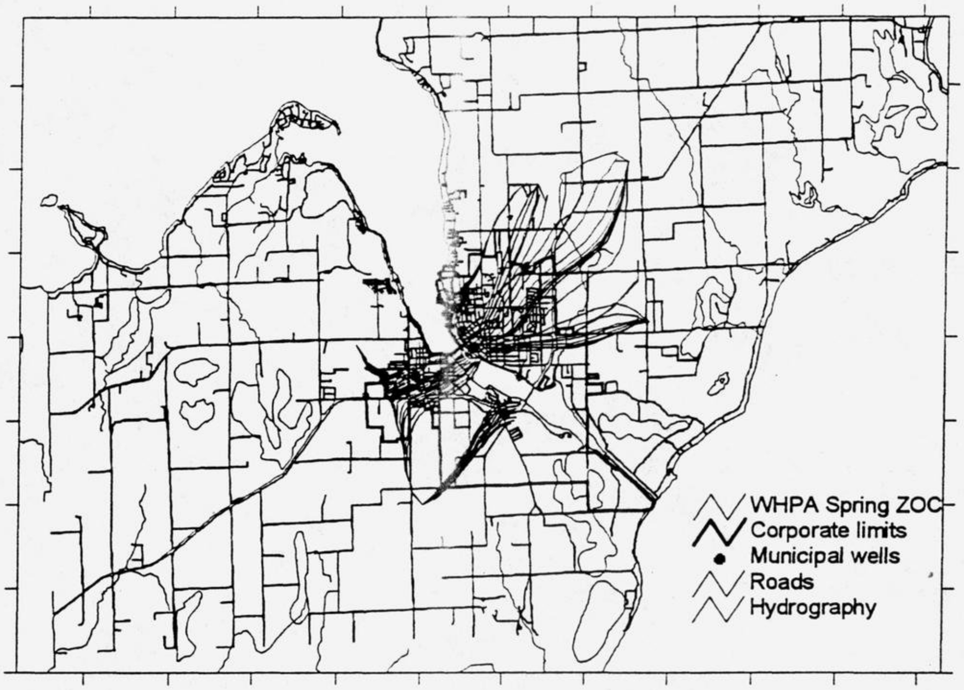


Figure 15. Particle tracks produced by the WHPA model

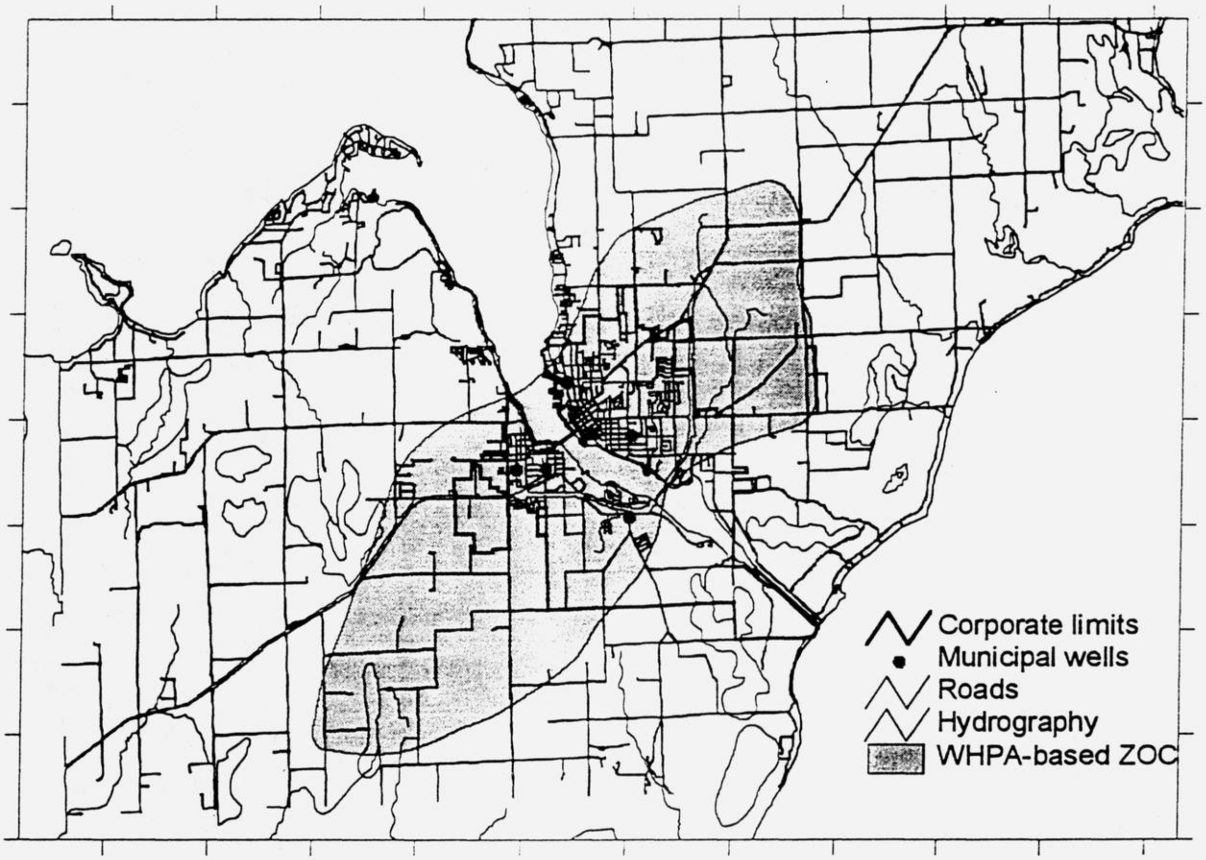


Figure 16. Composite spring and fall zone of contribution determined using the WHPA code

Three-dimensional MODFLOW model

Purpose

The goal of the second stage of modeling was to construct a groundwater model which would simulate the aquifer as accurately as possible. This goal required a three-dimensional transient model with spatial and temporal variability of recharge rates, simulation of horizontal flow zones, and accurate representation of boundary conditions. In this stage we constructed both steady-state and transient models. We believe that a transient model most accurately represents the aquifer because of the extremely dynamic seasonal water-level fluctuations seen in wells in the study area. To construct this model, we used the MODFLOW code (McDonald and Harbaugh, 1988), a modular, three-dimensional, transient, finite-difference model. MODFLOW is probably the most widely-used groundwater model in the United States. We also used Groundwater Vistas (ESI, 1996), a graphical user interface, for model input, output, and data visualization.

Design of the MODFLOW model

The MODFLOW model consists of 99 rows, 97 columns, and 12 layer, for a total of 115,236 finite-difference cells, of which 90,560 are active. Figure 17 shows the model grid, which was aligned 60^0 from north in order to approximate the orientation of the dominant vertical fracture sets. The grid is irregularly spaced, with node spacings ranging from 50 m in the central city to 1000 m at the edges of the model. Grid spacing was also finer near major known karst and fracture features. Figure 18 shows cross sections along rows and columns. The uppermost layer (layer 1) represents unlithified material at the surface. Layers 2-12 represent dipping bedrock, and are arranged to simulate discrete continuous flow zones and related non-flow zones identified by Gianniny and others (1996). The model simulates the flow zones as thin, continuous, highly permeable layers. The intervening non-flow zones were modeled as thicker layers with lower horizontal hydraulic conductivities. The eleven bedrock layers dip approximately 1 degree to the southeast. The top layer, representing surficial unconsolidated material, truncates the bedrock layers unconformably, and has variable thickness. The thickness of each remaining bedrock layer is uniform throughout the model extent, but thickness varies between layers. Layer thicknesses range from less than 1 m to 40 m. Vertical fractures were not explicitely modeled. We assume that vertical fractures are dense enough that their effects can be lumped into the vertical hydraulic conductivity term for each layer.

Boundary conditions

All boundary conditions that were identified in the conceptual model were used in the numerical model. Projected outcrop areas of model layers in Green Bay and Lake Michigan were modeled as constant head boundaries. Head-dependent flux boundaries were used under parts of Green Bay, parts of Lake Michigan, Sturgeon Bay, and in the major streams using the MODFLOW river package. This type of boundary allows the model to compute the flux into or out of the aquifer depending on the head difference between the stream (or bay) and the aquifer. We identified groundwater divides

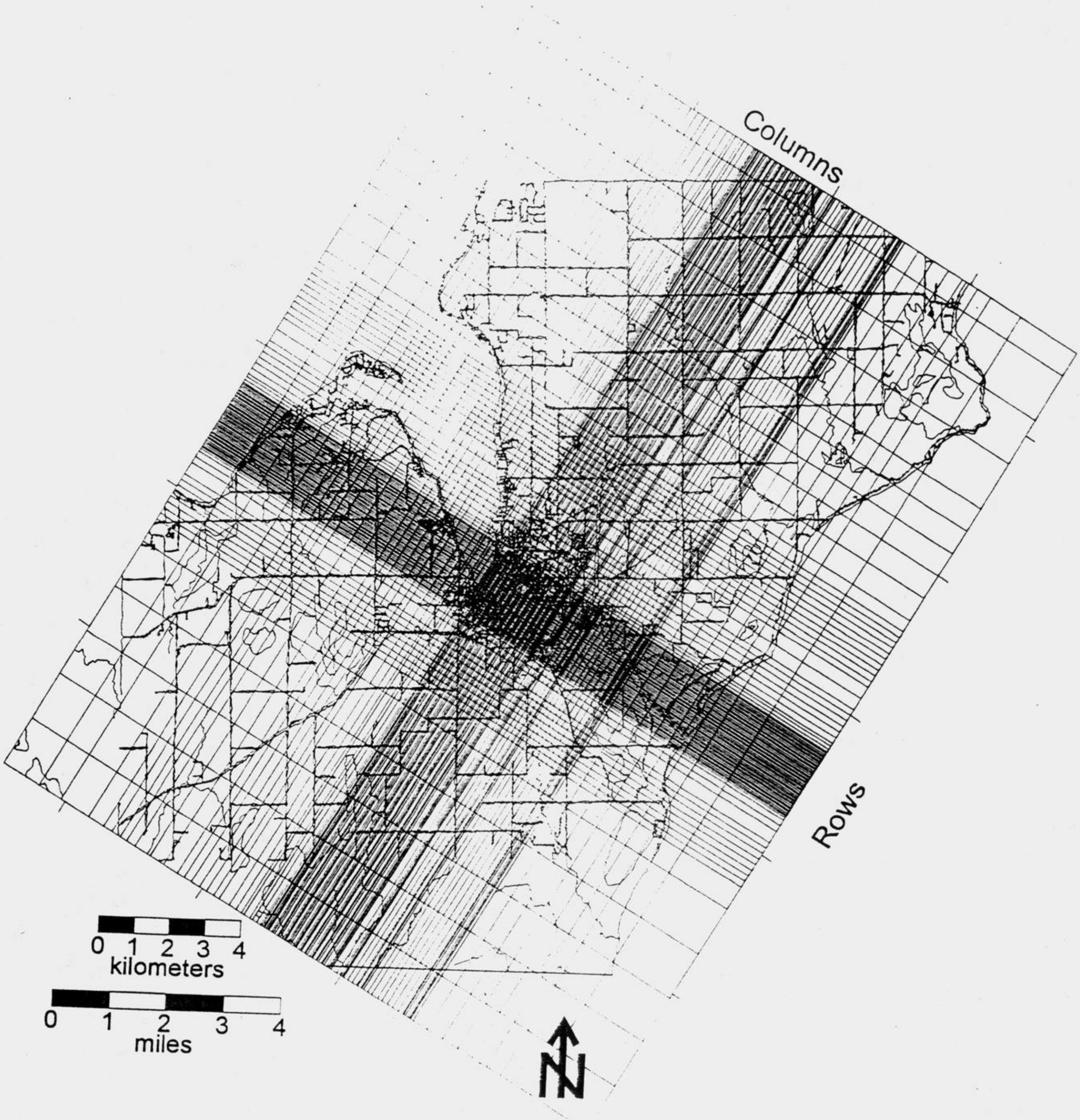


Figure 17. MODFLOW model grid

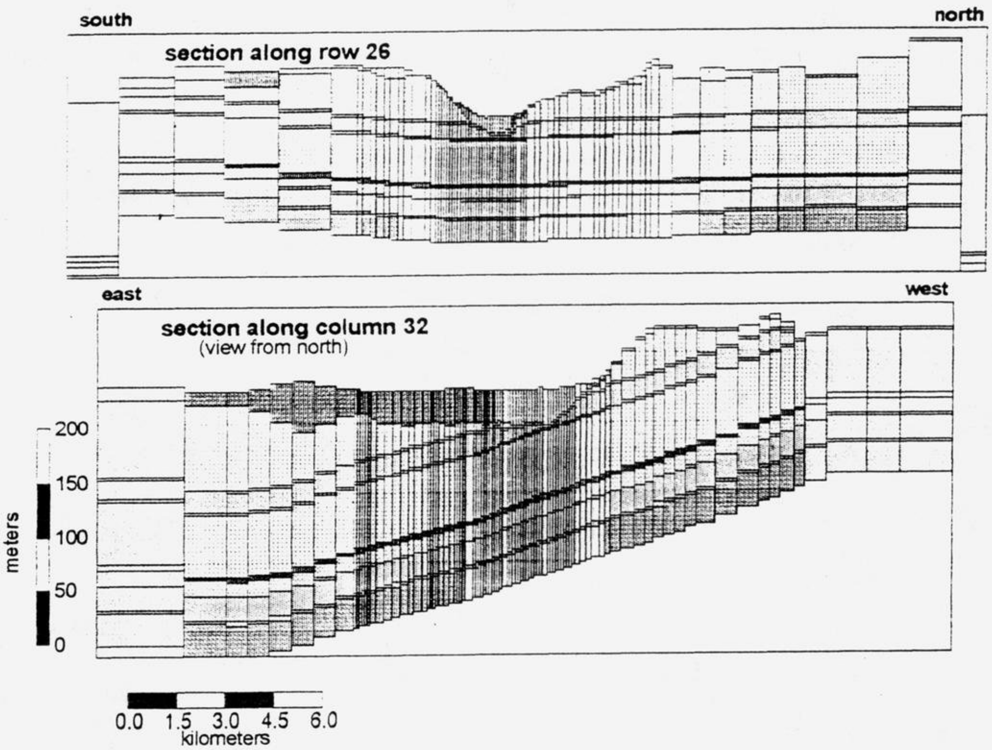


Figure 18. Cross sections along a row and a column of the model grid

from the spring and fall potentiometric maps. These were treated as no-flow boundaries in all layers.

Municipal wells for the City of Sturgeon Bay were modeled by removing water from the model at the location of each well. We calculated daily average pumping rates for each well by summing monthly pumping rates over a one-year period for each well and converting the rate to units of m^3/d. Pumping rates were apportioned to each flow layer in each well by weighting the total pumping rate by the transmissivity of each flow layer using the method of McDonald and Harbaugh, 1988.

Parameter values

Parameters required for the MODFLOW model include hydraulic conductivity, storage coefficient, specific yield, porosity, and recharge. Hydraulic conductivity was determined from pumping tests, packer tests, specific-capacity tests, and slug tests, as described above. Pumping test data from Sherrill (1978) were re-analyzed to verify his analyses. Our analysis of his pumping tests showed little difference from his original interpretations. Vertical hydraulic conductivity of the dolomite was calculated using an estimated anisotropy ratio (Kh/Kv) of 10. The surficial glacial deposits were assumed to be isotropic. Table 9 shows the values of hydraulic conductivity used in the model.

Layers 5 through 10 contain a higher hydraulic conductivity zone in the central area of the model domain (fig. 19). The higher K zone was delineated using the results of transmissivity estimates from specific capacity tests using TGUESS (Bradbury and Rothschild, 1985). The area is a large topographic low area, which suggests more intensive fracturing and hence, more easily eroded rock. Furthermore, this area showed erroneously high head values when the high K zone was not present in early model runs.

Table 9. Values of hydraulic conductivity used in the model. Layers with a * have two conductivity zones (Figure 19) with values shown of K shown for each.

Layer Number	Horizontal K (meters/day)	Vertical K(meters/day)
1	0.1	0.1
2	1	0.1
3*	45	4.5
4	4.5	0.6
5*	45, 350	4.5, 35
6*	1, 55	0.1, 5.5
7*	0.5, 350	0.01, 3.5
8*	2.5, 55	0.3, 5.5
9*	45, 350	4.5
10*	0.95, 55	0.01, 5.5
11	45	4.5
12*	0.2, 55	0.05, 5.5

Storativity was used only in the transient model. Values of storativity determined from pumping tests and packer tests range from 0.0005 to 0.001. We varied the storativity to assist in the calibration; a value of 0.0006 was used in the final version of the model. Storativity in the unconfined layers is approximated by the specific yield. We estimated the value of specific yield to be 0.01, which is typical for fracture
d aquifers.

Effective porosity is not used in the calculation of heads by the model but it is used in the particle tracking model for velocity calculation. We used an estimated effective porosity value of 0.01; this value was estimated by Bradbury and Muldoon (1994) using discrete fracture modeling in central Door County.

The areal distribution of recharge was determined from multiple sources described earlier in this report, with special attention given to the presence of karst and fracture features (closed depressions, exposed fractured rock, known sinkholes). Model nodes falling on such karst features were initially assigned a recharge value of 18 in/yr, or approximately double the rate determined using the soil infiltration model. These values were then varied slightly during model calibration. Figure 20 shows the distribution of recharge zones in the model. Table 10 shows average annual recharge rates associated with the different zones. Recharge rates were varied in the transient model to reflect seasonal variability of precipitation (i.e. very little or zero recharge in mid-winter and summer, high amounts of recharge in November, March, and April). Six recharge periods were used in the final transient model. Table 10 shows the length of each recharge period and the recharge rate in the four recharge zones. Recharge rates in each zone and in each period were varied until the modeled heads matched measured heads in long-term monitoring well Dr-265.

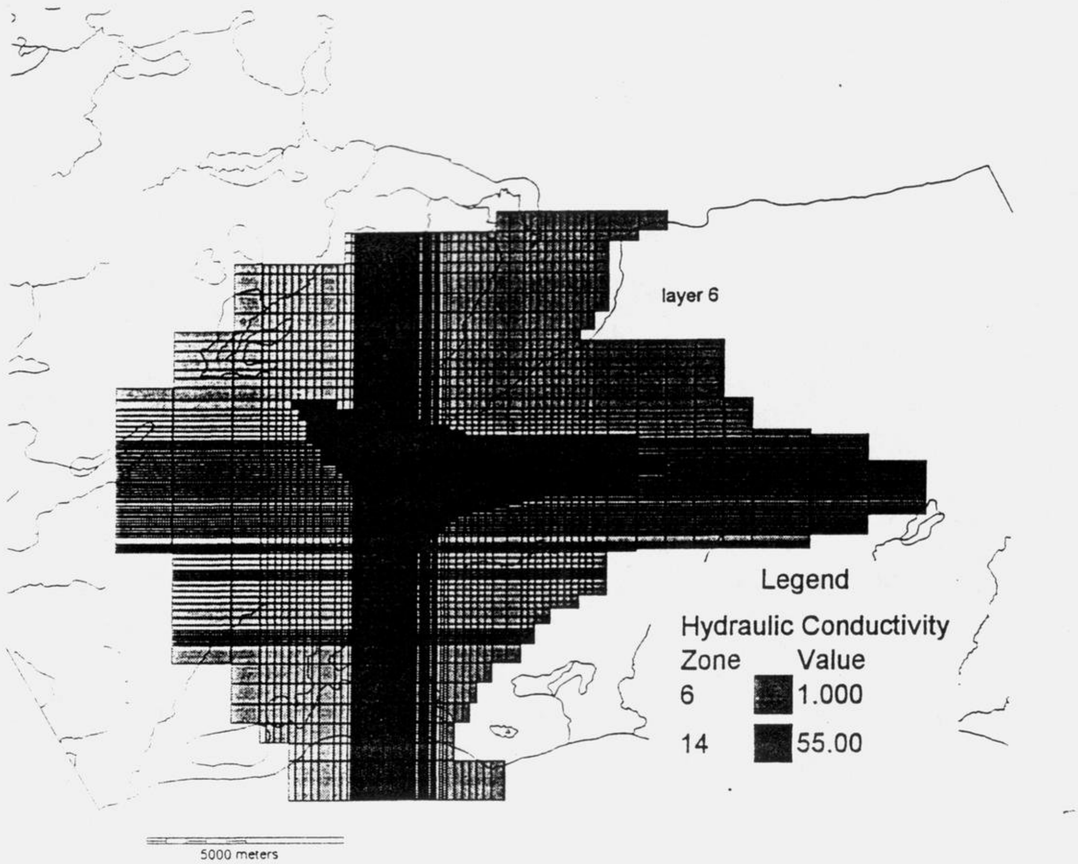


Figure 19. Hydraulic Conductivity distribution in layer 6 of the MODFLOW model, showing the high-hydraulic conductivity zone in the Sturgeon Bay area.

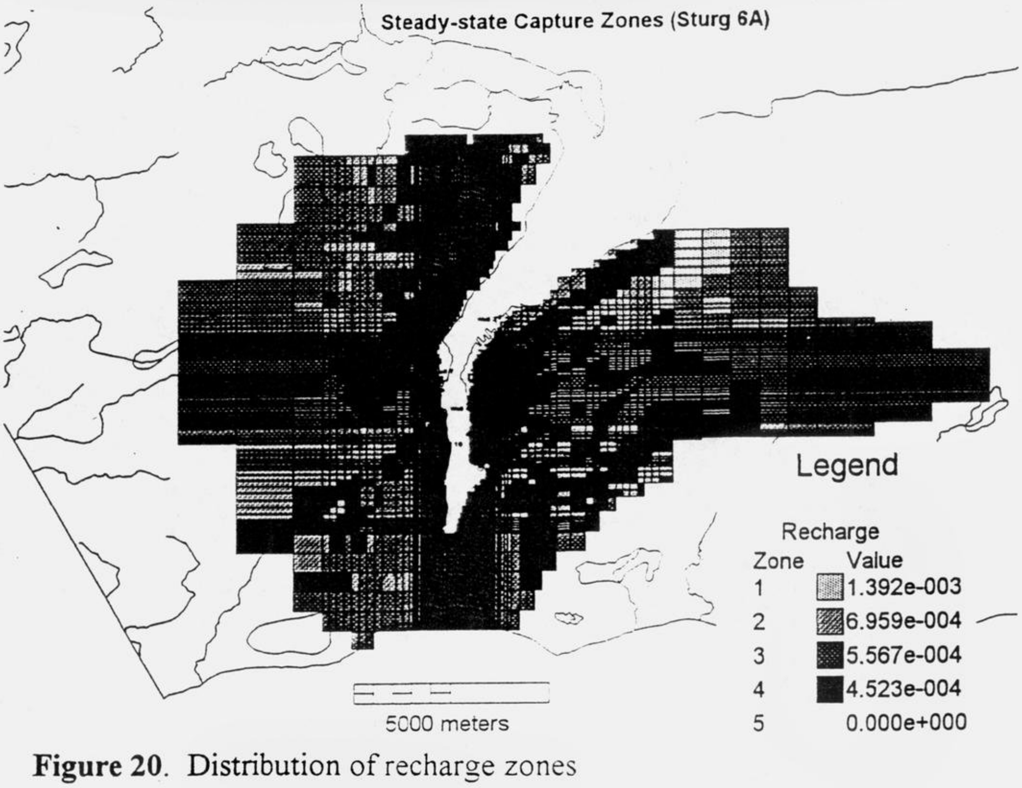


Figure 20. Distribution of recharge zones

Table 10. Recharge period lengths and recharge rates (m/d) in the four recharge zones for steady-state and transient model runs.

Recharge period	Length of period (days)	Recharge zone	Recharge rate (meters/day)
Steady state	--	1	1.2E-03
	--	2	1.1E-03
	--	3	6.0E-04
	--	4	5.0E-04
1	31	1	4.000E-05
	October	2	3.000E-05
		3	8.000E-06
		4	5.000E-06
2	30	1	4.900E-03
	November	2	5.100E-03
		3	3.200E-03
		4	2.200E-03
3	91	1	0.00
	December-February	2	0.00
		3	0.00
		4	0.00
4	31	1	6.100E-03
	March	2	6.300E-03
		3	5.000E-03
		4	4.200E-03
5	30	1	1.500E-03
	April	2	1.200E-03
		3	9.200E-04
		4	8.500E-04
6	153	1	2.500E-04
	May-September	2	1.500E-04
		3	1.200E-04
		4	3.000E-05

Model calibration

Most groundwater modeling protocols (e.g, Anderson and Woessner, 1991) recommend constructing and calibrating a steady-state model prior to constructing a transient model. However, the groundwater system near Sturgeon Bay, and probably in many fractured-rock environments, is rarely, if ever, at steady state. Due to the low effective porosity and thin soil cover, the system responds very rapidly to precipitation, and groundwater levels are rarely steady (see figure 6, for example). Therefore, although we achieved a steady-state calibration to both Spring and Fall water levels, we believe the transient calibration is more representative of the actual flow system. The steady-state calibration procedure involved adjusting values of hydraulic conductivity and recharge within a reasonable range (i.e. within measured values) until the best match between measured heads (from the potentiometric maps) and modeled heads was attained. We also adjusted the hydraulic conductivity and thickness of stream sediments to calibrate to measured baseflow values. Ranges of hydraulic conductivity values for stream and lake sediments were taken from slug tests performed by Bradbury (1982). Stream sediment thicknesses were measured and estimated from field work. Stream flows were measured in August 1995.

Model calibration consisted of repeated model runs attempting to match hydraulic heads in 75 calibration targets distributed among 9 of the 12 model layers while adjusting hydraulic conductivity and recharge rates. The calibration targets were water levels in individual wells measured during 1994 and 1995 plus the vertical distribution of hydraulic head measured during the packer experiments on wells DR-265 and Dr-398. In addition, the long-term daily average water levels measured in well Dr-265 were used to constrain the transient calibration. Calibration of the transient model was accomplished by adjusting values and timing of transient recharge.

The transient model was constructed to simulate the groundwater system through one idealized water year, from October through September. The MODFLOW code discretizes time into stress periods, between which hydrologic parameters can change, composed of time steps of varying length. The Sturgeon Bay model uses six stress periods, ranging in length from 30 to 150 days, and each having different recharge rates (table 10). These stress periods are each subdivided into either three or four time steps, with a time step multiplier of 1.2. Initial head conditions for the transient runs were taken from the previously-calibrated steady-state model.

Both the transient and steady-state models were initially unstable, and would not converge to a solution. The model instabilities are apparently related to the presence of thin layers near the water table. These layers, some less than 1 m thick, occasionally become unsaturated during model iterations, and, unless allowed to rewet, cause incorrect transmissivity variations in the model. We were able to achieve model convergence using the PCG2 solver combined with the wet-dry option (McDonald and Harbaugh, 1988), using closure criteria of 0.01 m.

The transient model showed an additional instability, in which the initial starting heads, thought to represent fall conditions, were not replicated after one model cycle (six stress periods). The model did not achieve transient stability (the ability to replicate one "model" year with the next model year) until at least two cycles of the model were complete. The final model, then, consisted of 12 stress periods, representing two identical years.

Values of hydraulic conductivity and river node parameters from the steady-state model were used in the transient model. The final calibrated model gives an acceptable reproduction of water levels at the calibration targets. Figure 21 shows the distribution of hydraulic head in model layer 7 at transient time steps representing Spring and Fall conditions (compare with fig 7). Figure 22 is a plot of measured versus modeled heads for the transient model during the Fall time period. Model residuals range from -34.8 m to +7.5 m, with a residual mean of -0.19 m. Most of the data fall near the ideal 45° line of perfect calibration, but some outliers also occur. These outliers could be due to measurement error as well as model error, or to problems of well construction in the target wells. Figure 23 is a hydrograph from Dr-265 and the computed values of head during the six stress periods. The transient model is able to acceptably reproduce the average daily water levels in this well. The computed heads are within one standard deviation of the 9-year mean water level for most of the simulation period. Notice how the model responds to the varying recharge rates.

Particle tracking

Particle tracking simulations were done with steady-state and transient versions of the model using the MODPATH III code (Pollock, 1994), chosen for its capability to do reverse particle tracking in transient groundwater flow. MODPATH operates as a post-processor to the MODFLOW model, and uses the MODFLOW cell-by-cell mass balance files to calculate flow velocities throughout the aquifer. MODPATH then tracks particles through this flow field. Particles were initialized in rings of ten particle placed around the five municipal wells in flow zones open to each well. MODPATH then tracked these particle backwards through the transient flow field produced by the transient model until the particles reached the water table. The outline of the resulting paths delineates the zone of contribution for the well.

To test the results of reverse particle tracking in a transient model, forward simulations were performed with the particles starting at locations determined by the ending locations from the reverse runs. The results were different from the reverse runs, implying that reverse particle-tracking may not be appropriate in transient models. This is probably due to the temporal changes in the configuration of the water table and potentiometric surfaces in a transient model run. To resolve this problem with reverse particle tracking in a transient model, we adapted an approach in which particles are started from wells at each time step in the transient model. We started particles at the beginning of each of the 12 stress periods in the two-year model run. The resulting collection of particle paths outlines the transient capture zone for each well.

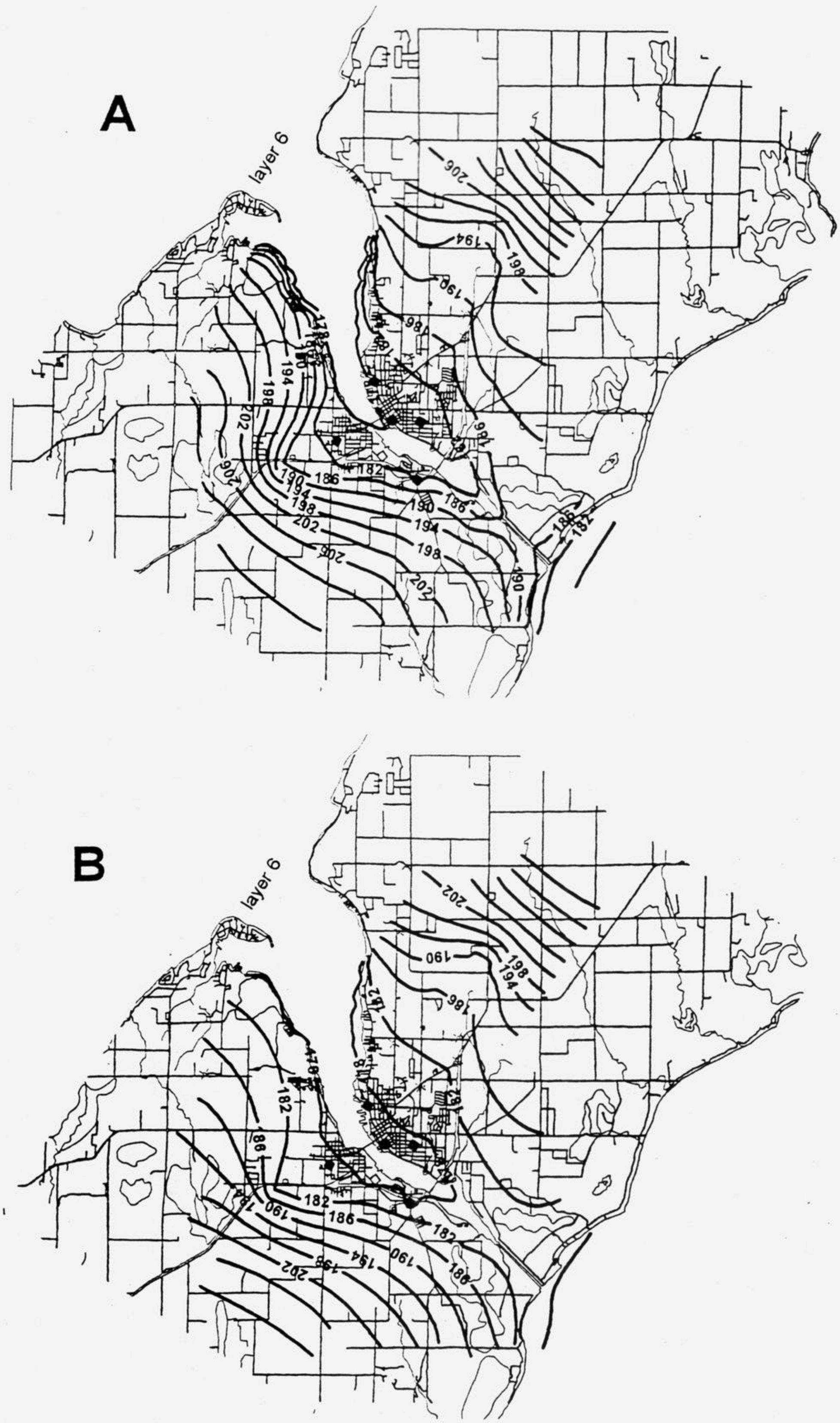


Figure 21. Model-simulated hydraulic head for spring (A) and fall (B) periods in the transient model.

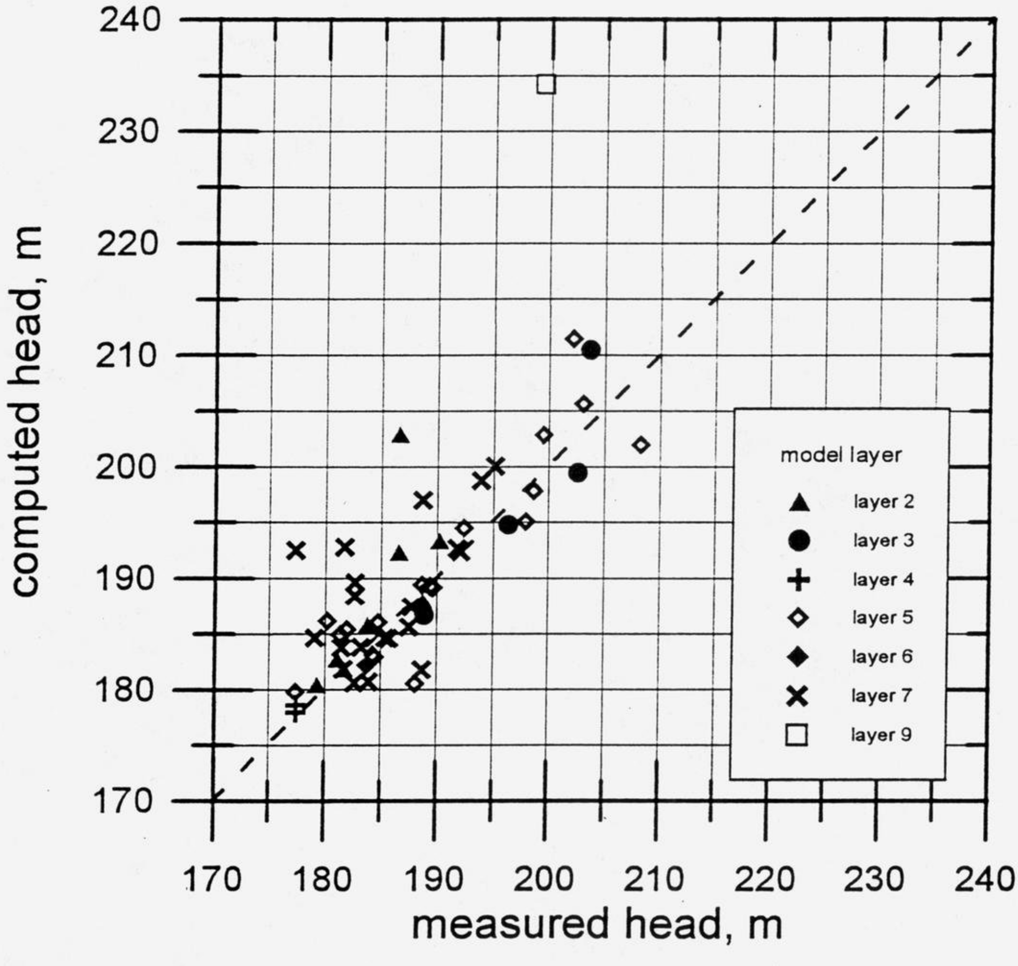


Figure 22. Measured versus computed hydraulic head at calibration targets

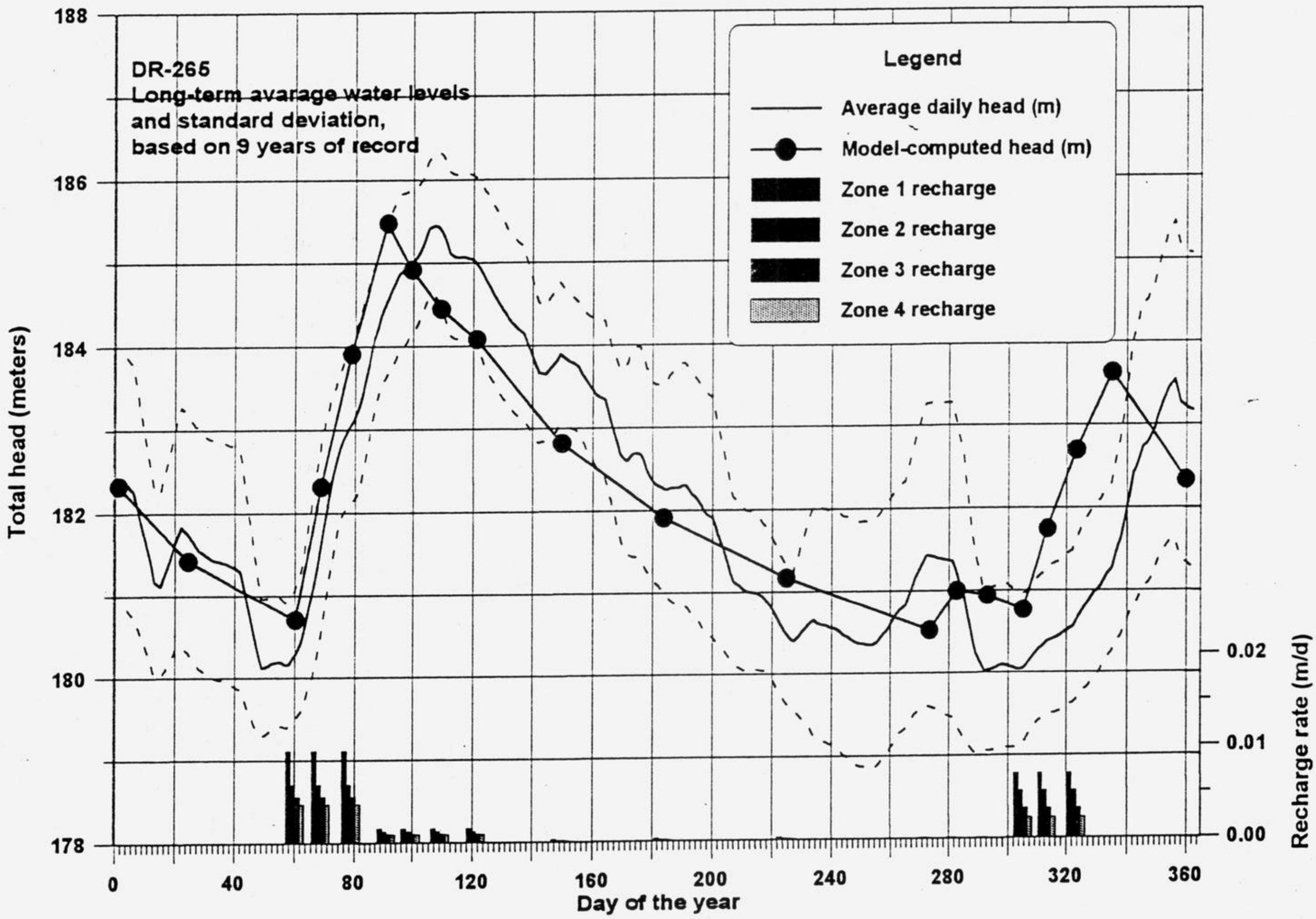


Figure 23. Computed versus measured long-term hydrograph of well DR-265. Bar graph along bottom axis indicates recharge periods used in the model.

In order to generate a conservative capture zone analysis, we increased the simulated pumping rates of the city wells to the maximum pump capacity for each well before running the final transient particle simulations. For most of the wells this pumping rate is about double the actual rate of use. Using the higher pumping rates produces slightly larger capture zones, and is considered appropriate for future planning scenarios in which water use in Sturgeon Bay might increase or one or more wells have to be taken off line for repairs.

Figure 24 shows the combined particle paths for all five city wells. The wells on the northern side of the city have capture zones extending nearly 10 kilometers to the northeast, while capture zones for the wells on the south side of the city extend nearly 7 km to the southwest. The longest particle paths originate from city well 3, on the northwest side of the city. Groundwater travel times from the water table to the municipal wells vary with depth in the well but in all cases are quite short. The average travel time from the water table to the wells was 152 days. The minimum and maximum travel times were, respectively, 14 days and 729 days. The majority of travel times were less than one year. Several of the particles had paths originating in Sturgeon Bay, indicating that some municipal wells may be inducing surface water in the Bay to flow into the aquifer.

The near-horizontal flow zones clearly control groundwater movement to the municipal wells. Figure 25 is a cross section along model row 27 showing the vertical sense of particle movement to city wells 8 and 3. Particle movement is largely vertical from the water table to the first major flow zone. Upon entering a flow zone, particle movement is mostly horizontal.

Figure 24. MODPATH-generated particle paths for the five city wells

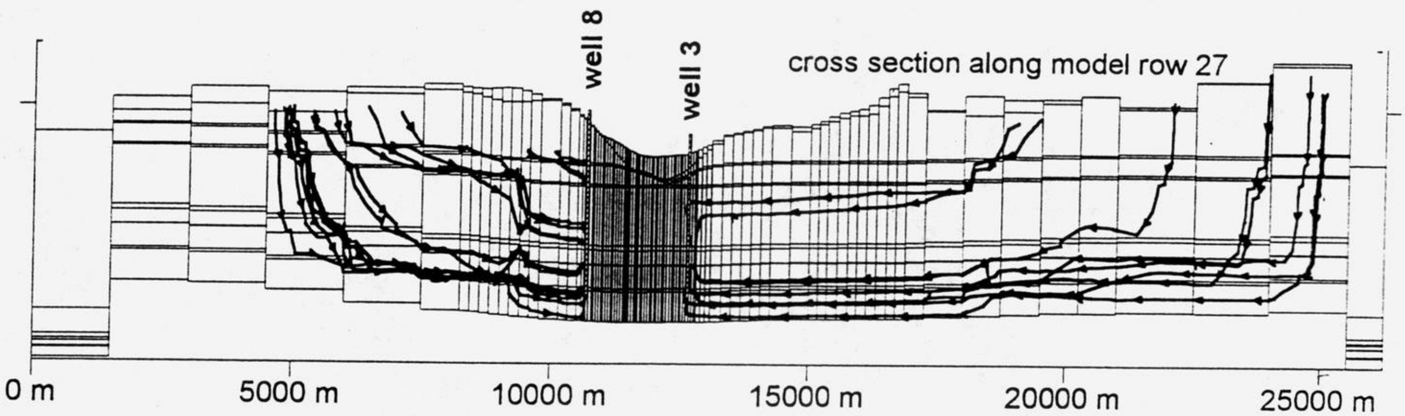


Figure 25. Cross section showing vertical particle movement

Two-Dimensional Fracture Network Model

The saturated zone particle tracking using the MODFLOW and MODPATH models delineates well capture zones relative to the saturated zone only. This procedure has the potential to underestimate the lateral extent of the capture zones and thus does not account for lateral spreading in the unsaturated zone above the water table. The unsaturated zone in the Sturgeon Bay area, particularly in areas north and south of the city limits, can be up to 30 m thick. Recharging water moves through the shallow fracture network above the water table, and the complex fracture pathways might lead to significant lateral movement of water.

We used a simple two-dimensional discrete fracture flow model, SDF (Rouleau, 1988), to estimate the amount of lateral spreading that might occur in the unsaturated zone between the land surface and the water table. Bradbury and Muldoon (1994) modified the original SDF code to include particle tracking and applied it to simulation of groundwater movement near the Town of Sevastopol, in central Door County. The SDF model assumes saturated flow through discrete fracture pathways based on statistical properties of fractures measured in the field. The model creates stochastic synthetic fracture networks, which are statistically similar to field measurements, and tracks particles of water through the networks. In this application the saturated fracture-flow model was used to estimate processes in the unsaturated zone. The apparent contradiction in using a saturated flow model for an unsaturated situation is not a problem because during recharge events the fracture network probably does become saturated for short times, while the rock matrix remains unsaturated. This model does not consider flow in the rock matrix.

The objective of the spreading analysis was to determine the probable lateral distance a contaminant might move while moving from the land surface to the water table. The spreading analysis uses a hypothetical problem domain of a cross section 100 m long and 20 m thick. The top and bottom boundaries are constant head, with a 5-m head loss across the section from top to bottom. The side boundaries are no-flow. Particles are injected along a 2-m wide strip centered at the top of the section (from 49 to 51 m in the X direction). Particles were initiated at 0.1 m increments across this distance, for 21 particles in all. The model uses three sets of fractures, with statistical properties based on measurements of Bradbury and Muldoon (1994) and Roffers (1996). Set 1 represents near-horizontal bedding-plane fractures. Sets 2 and 3 represent near-vertical fractures. Figure 26A shows the problem domain.

Based on 10 stochastic realizations of the fracture network, the lateral spreading at 20 m below the land surface is centered 5.5 m down dip, with a standard deviation of 15.2 m (fig 26A). Assuming this trend is linear with depth, and that the average depth to groundwater beneath the capture zone boundaries is 40 m, the model analysis gives a horizontal spreading distance of about 26 m. However, this analysis contains many uncertainties, including local variations in fracture characteristics, water table depth, and recharge rates. For the propose of wellhead protection it is wise to err on the conservative side of the analysis. Therefore, we chose a lateral spreading zone of 100 m beyond the capture zones delineated in the three-dimensional model. The final capture zones were extended horizontally 100 m in every direction to account for this potential spreading.

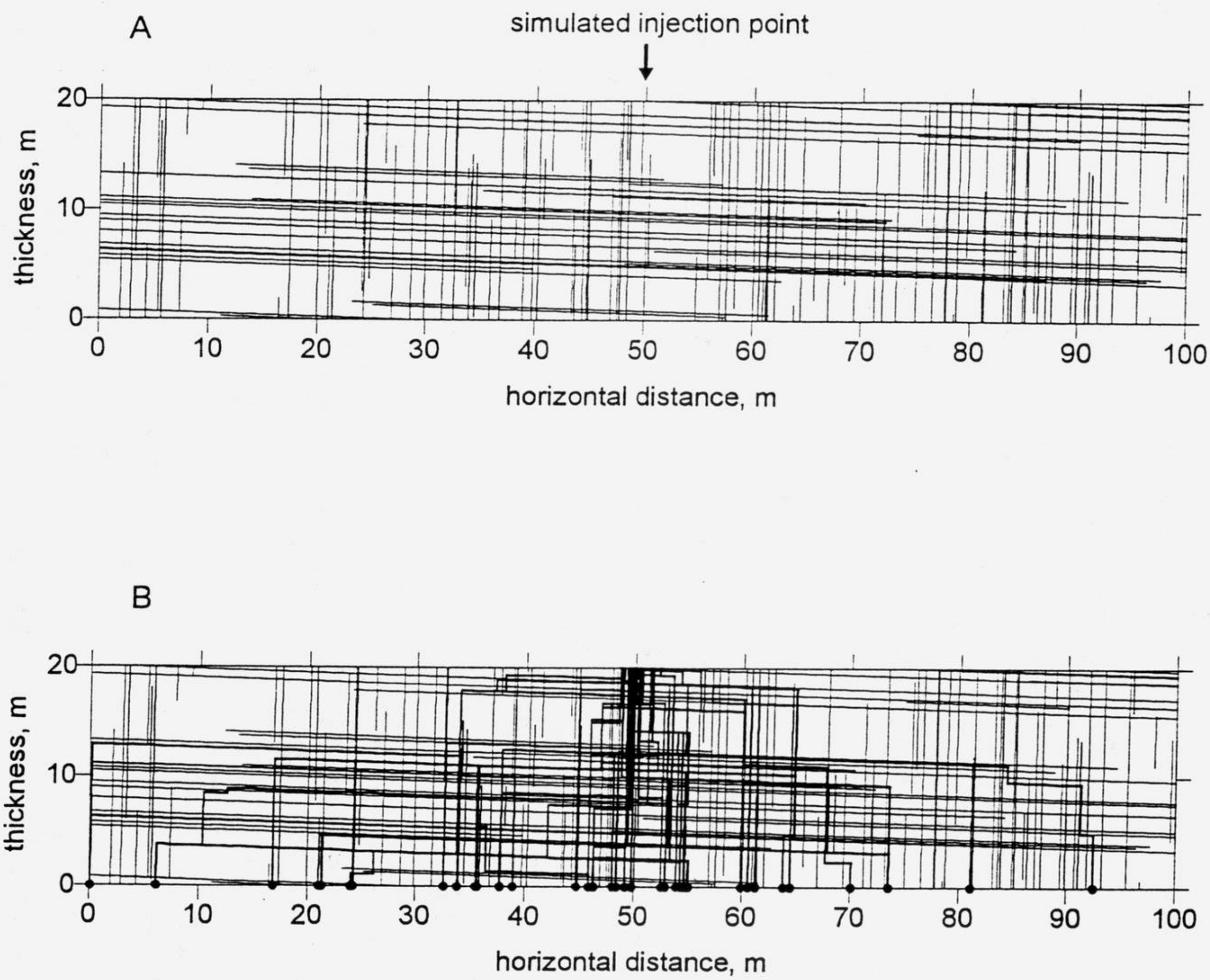


Figure 26. Problem domain and results for the cross-sectional fracture network model. A. Problem domain, showing fracture network. B. Particle paths for one realization of 10 particles.

RESULTS AND IMPLICATIONS

Results

Areal extent of zones of contribution

The final zones of contribution for the Sturgeon Bay wells extend up to 10 km to the north and up to 7 km to the south of the city (fig 27). We delineated these final zones by digitizing the boundaries of the collection of transient particle paths for each well (fig 24) and then extending the zones outward by 100 m to account for spreading in the unsaturated zone. Under this procedure, the capture zones for wells 6 and 7 overlap, and are shown as one zone of fig 27. At least two wells (6 and 7) have the potential to draw small amounts of surface water from Sturgeon Bay.

It is important to understand that, while water originating anywhere in the capture zones has the *potential* to be captured by a city well, very little of the water originating in the delineated capture zones is *actually* withdrawn by the city wells. Most of the water originating in the capture zones bypasses the city wells and eventually discharges to surface water. This bypass is possible for several reasons. First, the city wells are not fully penetrating, but are cased to significant depths below the water table. Shallow groundwater cannot enter the wells. Second, the wells do not operate continuously, and groundwater can flow past the wells while they are not pumping. Third, groundwater falling in the capture zones can also be intercepted by domestic wells and by surface water features upgradient of the city wells.

Travel times to the Sturgeon Bay wells

Travel times from the land surface to the Sturgeon Bay wells are can be less than one year from anywhere in the zones of contribution. Longer travel times are also possible, given the complexity of the fractured groundwater system. However, it is unlikely that travel times ever exceed two or three years.

These travel times are far more rapid than travel times to most municipal wells in other parts of Wisconsin, and are directly related to the hydrogeology of the fractured dolomite in the area. For comparison, travel times to municipal wells in Dane County, Wisconsin, range from hundreds to thousands of years (Bradbury and others, 1996).

Reliability of models and results

Questions always arise about the reliability of, and validity of, zones of contribution determined by numerical models, and these concerns are clearly warranted given the complex hydrogeology of the Sturgeon Bay area. While errors and alternative interpretations of the data are always possible, it is important to point out that the analyses presented in this report are internally

Figure 27. Final zones of contribution for the Sturgeon Bay wells

consistent and are also consistent with other work in Door County. In particular we emphasize the following points:

- The geochemical and isotopic analyses of Sturgeon Bay groundwater are consistent with our model of rapid groundwater flow. All water samples obtained from the city wells were isotopically "young", and have been isolated from the atmosphere for less than about 35 years. Ion concentrations and saturation indices, while not definitive of groundwater age, are similar to results reported by Bradbury and Muldoon (1992) from samples of shallow, young (a few days or weeks) groundwater in central Door County.

- The regional hydraulic gradient, based on potentiometric maps, is the major control on the extent of the capture zones, and the regional hydraulic gradient is well understood. The fall and spring potentiometric maps are the major data sets used in the simple WHPA model. The capture zones generated by the WHPA model (figs 15 and 16) are similar in shape and orientation to the results of the MODFLOW model, even though WHPA uses major simplifying assumptions.

- Although this study did not include tracer experiments, the rapid groundwater flow velocities predicted by the model are consistent with the few existing groundwater tracer experiments that others have conducted in central Door County. In particular, Bradbury and Muldoon (1992) reported groundwater velocities of 17 m/day and 115 m/d from planned and unplanned tracer experiments in the Town of Sevastopol in central Door County. Muldoon and Bradbury (1998) observed velocities of up to 90 m/d in carefully controlled tracer experiments at the Bissen Quarry south of Sturgeon Bay.

- The intensive hydrostratigraphic analyses of Gianniny and others (1996) delineated the connected flow zones in the dolomite and provided a firm foundation for building the numerical model.

Implications and Recommendations

Implications and recommendations for the city of Sturgeon Bay

The zones of contribution to the municipal wells determined by the transient model are large and extend well outside of the city limits. Land use in the parts of the zones of contribution that are outside the city limits is not currently under the jurisdiction of the city. City officials will have to work with Door County and State of Wisconsin officials to regulate land use in these areas if protection of the recharge areas of the city's wells is desired.

The unfortunate combination of a single, hydraulically well-connected fractured aquifer, the lack of attenuating surficial sediments, the rapid travel times from recharge area to wellhead, the size of the zones of contribution of the municipal wells, and the types of land use practices in the zones of contribution make the Sturgeon Bay wells extremely vulnerable to contamination. Furthermore, numerous contamination sources occur within the mapped zones of contribution. Recommendations for using this information in a wellhead protection program for Sturgeon Bay include the following:

- Prioritized remediation, monitoring, and removal of potential contamination sources within the mapped zones of contribution. Often, financial resources are not available for addressing all potential groundwater contaminants in an area; the mapped zones of contribution provide a method for prioritization of groundwater cleanup efforts.

- Prioritized installation of agricultural best management practices within the mapped zones of contribution, as recommended in the Sturgeon Bay/Red River Priority Watershed Project (DC SWC, 1995).

- Proper abandonment of all unused wells in the zones of contribution. The presence of unused wells is a particular problem because the open well bores and potentially failing well casings provide pathways for rapid vertical movement of groundwater between the major horizontal flow zones supplying the city wells. The Sturgeon Bay Utilities has already abandoned several disused city wells as part of their contribution to this project.

- Continued use of the three-dimensional groundwater flow model as a planning and management tool for the Sturgeon Bay Utilities. In addition to its particle-tracing capabilities, the model is useful in siting new wells and developing optimum pumping strategies for existing wells.

- The mapped zones of contribution should be used as a public education tool to help local citizens understand the sources of local water and the links between land surface activities and groundwater quality.

Implications and recommendations for other wellhead protection studies

This project has significant implications for other wellhead protection studies in fractured carbonate aquifers. A sufficiently detailed hydrostratigraphic model was critical to understanding the hydrogeology of the Sturgeon Bay area, and should be carried out for any serious hydrogeologic analyses in carbonate aquifers. The work of Gianniny and others (1996) was the basis for the conceptual model of the aquifer and allowed us to use a porous media model to simulate groundwater flow and delineate zones of contribution in the aquifer. The porous-medium approach to modeling can be used with success in fractured carbonate rock if detailed hydrostratigraphy is understood. Specific recommendations are as follows:

- In fractured carbonate aquifers the groundwater flow velocities can be so high, and resulting travel times so short, that the usual wellhead protection time-of-travel criteria (2 years, 5 years, 10 years, etc.) recommended by most wellhead protection manuals (e.g.

USEPA, 1987) have little meaning. In particular, it may be impossible to prioritize areas within the ultimate zone of contribution based on time-of-travel to a well. For example, at Sturgeon Bay, some areas farther from the wells are probably more critical than areas nearer to the wells because of the presence of the horizontal flow zones which can rapidly conduct water to the wells from great distances. Therefore, we recommend that the entire zone of contribution be given equal protection in these types of fractured carbonate aquifers.

- Simplified, two-dimensional flow models, such as the WHPA code, have significant value in wellhead protection studies even in an area as complex as Sturgeon Bay. The areal zones of contribution generated by the WHPA code were quite reasonable when compared to the zones generated by the MODFLOW model.

- Given a well-defined hydrostratigraphic model, the porous-media-based MODFLOW code is adequate for simulating flow in the dolomite aquifer as long as individual horizontal flow zones are recognized and modeled discretely. MODFLOW's lack of capability to fully simulate fracture flow is balanced by its flexibility in simulating the complex boundary conditions found in most real-world problems.

- Construction of groundwater flow models requires adequate and available water-level measurements for model calibration. In particular, long-term records of water-level fluctuations and trends are essential for the construction of transient groundwater models. In Wisconsin, such records are provided by the statewide groundwater level network, operated cooperatively by the USGS and WGNHS. In recent years, budget cuts have threatened this program, but continued collection of long-term water-level data is essential for models such as the Sturgeon Bay model.

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