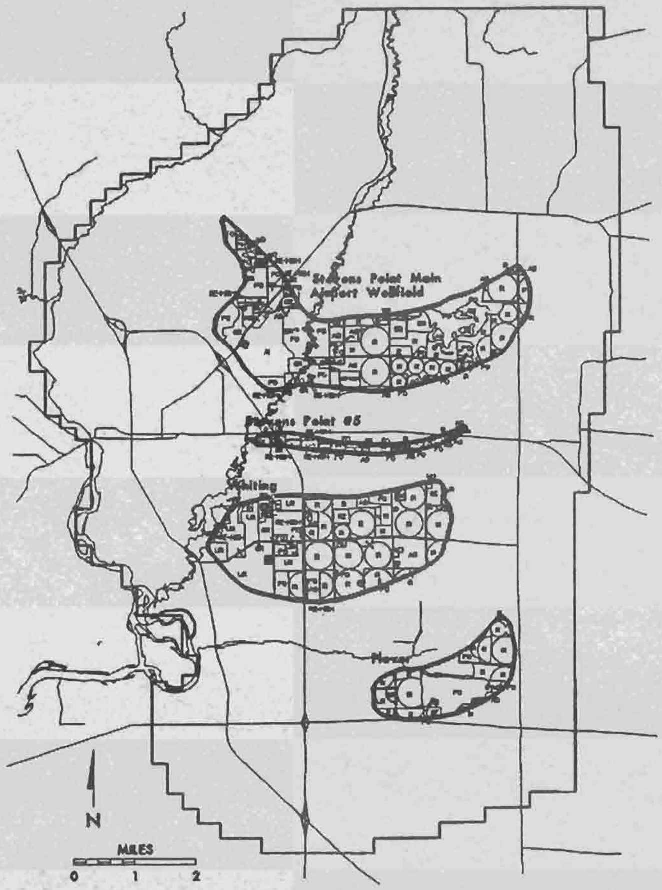
CONTAMINANT SOURCE ASSESSMENT AND MANAGEMENT USING GROUNDWATER FLOW AND CONTAMINANT MODELS IN THE STEVENS POINT - WHITING - PLOVER WELLHEAD PROTECTION AREA



Prepared by the Central Wisconsin Groundwater Center

University of Wisconsin-Stevens Point

1997

CONTAMINANT SOURCE ASSESSMENT AND MANAGEMENT USING GROUNDWATER FLOW AND CONTAMINANT MODELS IN THE STEVENS POINT - WHITING - PLOVER WELLHEAD PROTECTION AREA

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USDA Stevens Point-Whiting-Plover Wellhead Protection Project

for the

Village of Whiting

Village of Plover

City of Stevens Point

Central Wisconsin Groundwater Center

Cooperative Extension Service

College of Natural Resources, University of Wisconsin-Stevens Point

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And finally, we wish to thank the US EPA-Region 5 and Bill Ryan for funding this work and contributing to the informed management of our groundwater resources.

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ARCHIVES

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

1 Database containing well log and construction report data for 694 wells.

2 AutoCAD format land use mapping.

EXECUTIVE SUMMARY

This report documents the development and application of groundwater flow, particle tracking, and nitrate loading models for use as groundwater management tools in the Stevens Point, Whiting, and Plover (SWP) municipal well recharge area. The recharge area supplies groundwater from a glacial outwash aquifer to a population of over 40,000. Groundwater concerns, particularly regarding quality, continue to present management challenges for the community. Nitrate, for instance, exceeds drinking water standards (10 mg/L of nitrate-nitrogen) in 20-25% of area domestic wells, and has forced Whiting and Plover to construct nitrate removal systems at substantial cost. A number of pesticides have also been detected in area groundwater, though these have not been in concentrations near standards in municipal wells.

Growth, changing land use, and changing land management create new demands for groundwater as well as alter groundwater quality and availability. Municipal and county planners and managers need to know what quality of groundwater will evolve from present and anticipated land uses, and what strategies will be effective for maintaining or improving water quality. They also need to know about the adequacy of groundwater supply, and how pumping may affect streams and wetlands. Modeling is a way to evaluate and predict what impacts could result from present and future management decisions. The models documented in this report were used to simulate groundwater flow, delineate zones-of-contribution and times-of-travel for the municipal wells, predict future nitrate conditions, and evaluate the efficacy of nitrate control strategies. The models can be used to answer other groundwater management questions as needs arise.

An assessment of physical (geologic and hydrologic) conditions in the project area was required prior to developing the groundwater flow model. Conditions were assessed from published reports and maps, and an extensive review of well logs. Well log data was entered into a database for future use, and new maps were produced of bedrock and water table elevations. Well log data were also used to estimate aquifer hydraulic conductivity and to map its spatial distribution.

The groundwater flow model was developed using the USGS MODFLOW code. The model utilized the physical conditions assessment for inputs, and was calibrated to historical head data. The calibrated model was then used to evaluate future conditions using anticipated year 2005 pump rates. Zones-of-contribution and times-of-travel were subsequently determined using the USGS MODPATH code.

Flows in the Plover and Little Plover Rivers are being impacted by municipal pumping. Anticipated pumping conditions (year 2005) may remove about 10% of Plover River baseflow, and perhaps greater than 40% of Little Plover River baseflow. Impacts on the Plover River might be subtle, but those on the Little Plover would likely severely impact its ecology and fishery. The flow model was not optimized to examine the Little Plover River, so predictions regarding its fate are approximate and need further examination.

The nitrate mass-balance model was developed to predict steady-state nitrate concentrations at the municipal wells under present land use and land management, and to evaluate the effectiveness of various management options, such as changes in land use or reductions in nitrate loading from agricultural best management practices (BMPs). This model required detailed land-use data and an approximation of nitrate loading from various land uses. Land use data was obtained from the Portage County Planning and Zoning Department geographical information system (GIS). Dryland and irrigated agricultural land-use areas were further characterized by their rotation history. Loading rates were developed for each land use and agricultural management practice using literature values and mass balance calculations.

The predicted steady-state nitrate concentrations for Stevens Point main, Stevens Point #5, Whiting, and Plover wellfields are 4.0, 21.7, 37.8, and 25.9 mg/l respectively under conventional agricultural practices. The predicted concentrations drop to 3.1, 15.5, 25.7, and 18.7 mg/l under complete agricultural BMP participation. Except for Stevens Point main wellfield, the predicted concentrations, even under BMP's, significantly exceed the MCL. The Stevens Point main wellfield prediction is lower because of dilution by induced Plover River recharge and large amounts of no nitrate-loading forest and airport land uses near the wellfield. Current measured nitrate concentrations are around 2, 5, 19, and 15 mg/l respectively, and appear to be increasing over time. The discrepancy between predicted and current concentrations is certainly due to conditions being far from steady-state, and possibly due to error. The major potential source of error may be the assumption of no denitrification in soils and aquifer. However, evidence of this phenomenon has not been observed, nor are conditions conducive to denitrification through most of the project area.

For the combined zones-of-contribution, the ranked nitrate loading sources were agriculture (89%), unsewered residential (7%), surface waters (2%), and urban (1%). The components of agricultural loading are irrigated fields (66%), manure (15%), and dryland fields (8%). Only the Stevens Point #5 well is impacted substantially by another land use (unsewered residential, 31%). The irrigated agriculture land use was the largest potential nitrogen source in the project area. Given the mix of crops and rotations observed on these lands, irrigated agriculture contributes an average of 97 lbs/acre-year under conventional practices, and 69 lbs/acre-year under BMP. Dryland agriculture is slightly lower with 63 lbs/acre-year conventional and 38 lbs/acre-year BMP. (Manure loading not included.) The average loading for the unsewered residential was 43 lbs/acre (average lot size of 1.4 acres). The loading for unsewered residential in the model is likely greater than reality due to assumptions of population by household.

To improve future groundwater quality with respect to nitrate, reducing loading from agricultural lands has to be a major goal. The nitrate mass balance indicates that even universal adoption of agricultural BMPs results in large amounts of nitrate loading. Other strategies need to be explored.

Public process, including information and education efforts, must be components of any plan to address groundwater quality and quantity. Groundwater concerns need to be focused into articulating long-term goals, and developing and implementing management strategies that address the specific problems required to meet groundwater goals.

CHAPTER 1
INTRODUCTION

This report describes research conducted for the municipalities of Stevens Point, Whiting, and Plover to help in wellhead protection efforts. The research consisted of developing groundwater flow, particle tracking, and nitrate loading models that were then integrated with land use and land management information for application as groundwater management tools. These tools allow

* refinement of zone-of-contribution and time-of-travel delineations,

* prediction of nitrate concentrations from non-point loading under current and anticipated land uses and land management,

* evaluation of the efficacy of groundwater protection strategies.

The information and models developed in this project are intended to be used for future groundwater management efforts and to be updated as new information becomes available.

BACKGROUND

The municipalities of Stevens Point, Whiting, and Plover (populations 24,000, 1850, and 8337 respectively) are located in Portage County in the Wisconsin central sand plain (Figure 1.1). The sand plain is an extensive area of glacial sand and gravel outwash. The outwash provides the sole aquifer for the municipalities. While productive, the aquifer is highly susceptible to degradation due to current land uses and low pollution attenuation capacity of the overlying soils.

Land uses within the zones of contribution for the municipal wells are a diverse mix of irrigated and nonirrigated vegetable and dairy agriculture, sewered and unsewered residential and commercial development, and natural areas. While a number of groundwater contaminants are of concern, nitrate has been the most troublesome. Nitrate exceeds drinking water standards in about eighteen percent of private wells in Portage County and perhaps 20-25% of private wells in the study area. Nitrate exceedences forced closure of the Whiting municipal well from 1978 to 1991, during which time the municipality had to procure water from the City of Stevens Point, until completion of a nitrate removal facility constructed at a cost of $670,000. Plover wells began exceeding the nitrate standard in 1993, which forced the Village to install a nitrate removal facility costing $2 million plus

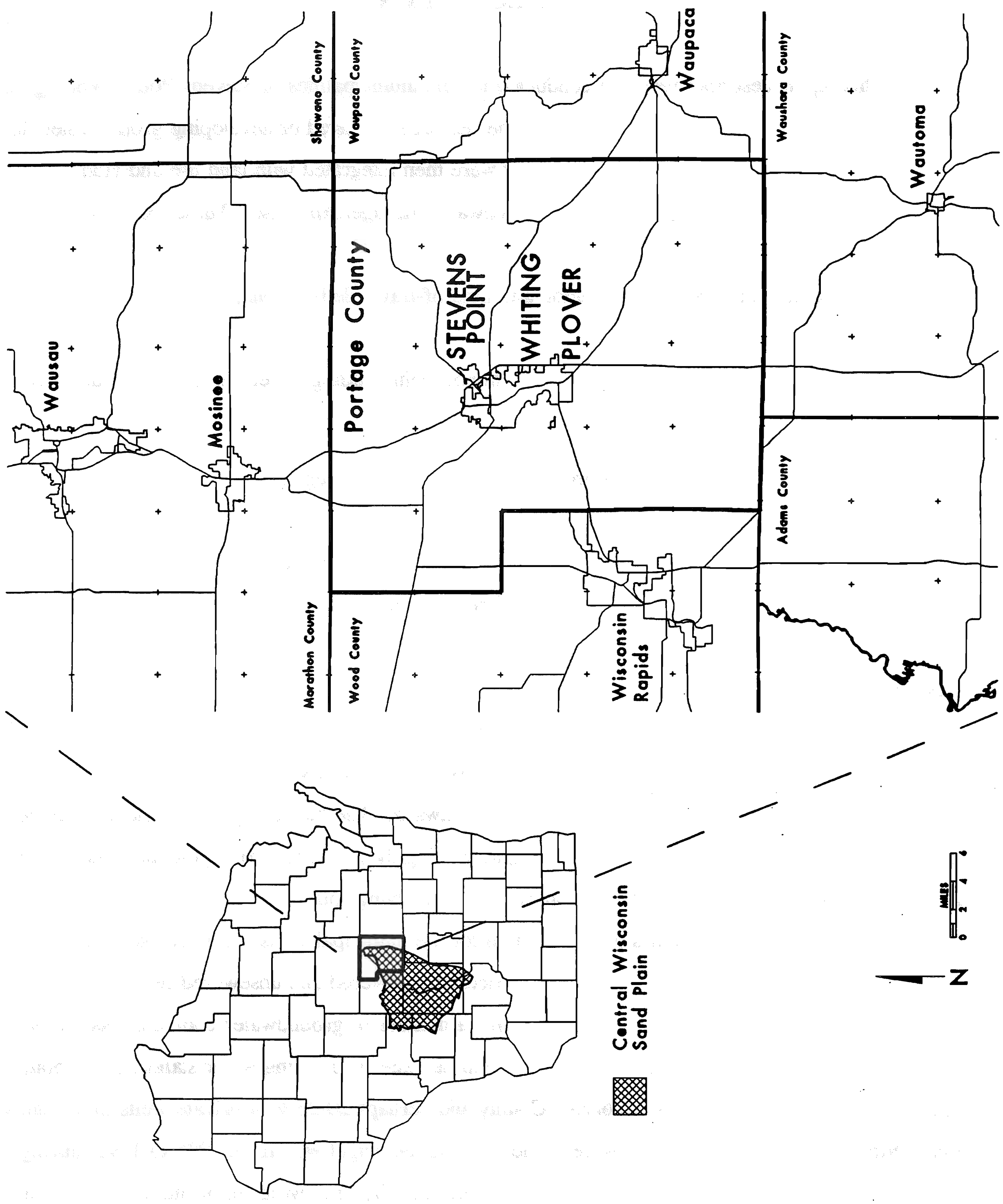

Figure 1.1 Location of study area.

an annual operating cost of $50,000. Trends show that nitrate levels continue to increase (Figures 1.2 and 1.3).

While nitrate is the current threat, pesticides also pose potential problems. A recent Portage County sampling effort indicated that nearly one half of tested wells contain atrazine residues, but scant additional information precludes an accurate characterization of the potential pesticide problem.

An important component of groundwater protection efforts within the study area is the USDA "Stevens Point-Whiting-Plover Wellhead Protection Hydrologic Unit Area Project". The major thrust of the project is to improve or maintain groundwater quality, mainly through reductions in agricultural inputs. This report contains new information which will augment the efforts of the Hydrologic Unit Area Project, and help focus its efforts where most needed.

PROBLEM AND NEEDS

Nitrate from various land uses, and particularly from agriculture, poses the most significant threat to groundwater in the study area. To adopt an effective wellhead protection strategy, guide growth, and make informed decisions affecting their collective future, the communities need to know

* how groundwater quality can be expected to change under present and anticipated land use and land management practices, and

* what strategies will be effective in maintaining or improving groundwater quality.

Tools are required to address these needs. In this case, the tools we developed are models that simulate groundwater flow and predict nitrate concentrations at the wellheads under present and anticipated land use and land management scenarios. While the models are directed at nitrate, the current primary pollutant of concern, they also have utility in predicting concentrations of other contaminants.

The predictive tools were developed as follows. First, existing information was compiled to better define aquifer properties and hydraulic heads throughout the study area. Second, a groundwater flow model was developed and used to refine wellhead protection area (WHPA) delineations and groundwater times-of-travel for the municipal wells. Third, the component land uses within the refined WHPA's were extracted from Portage County's GIS land use database by Portage County personnel. Fourth, nitrate loading rates were assigned to land parcels or land uses by using actual data where available or by using mass-balance loading models. Finally, loading data was combined with

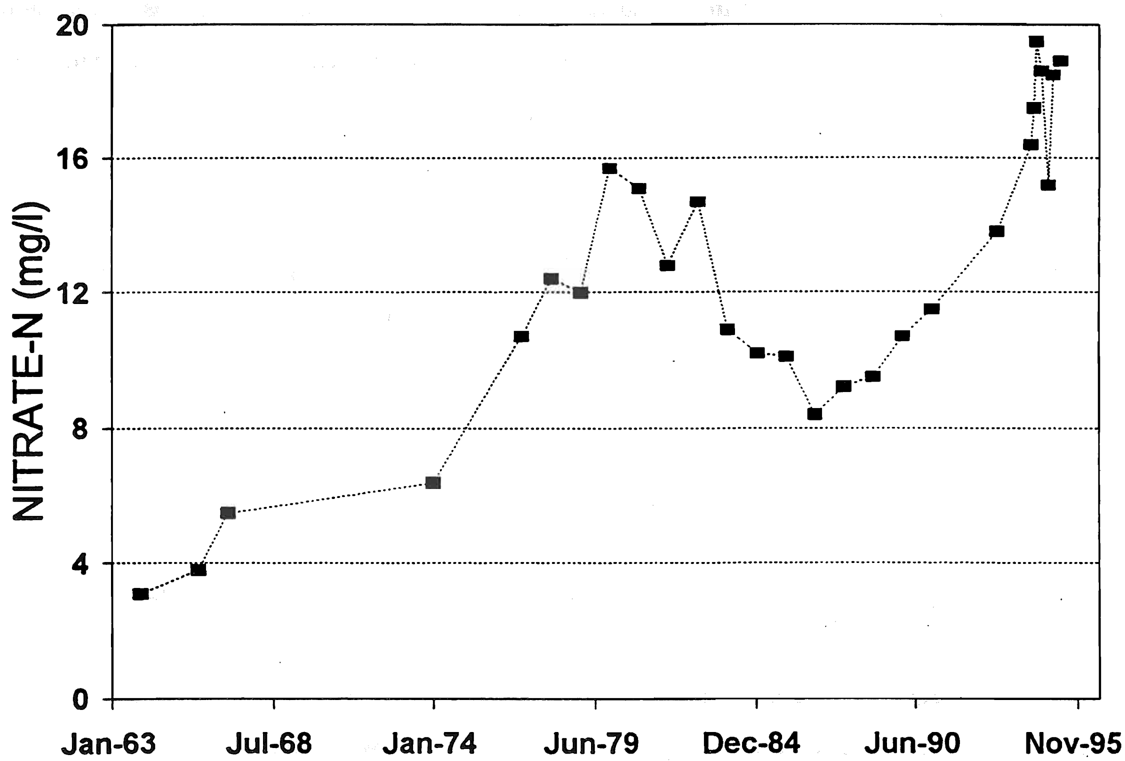

Figure 1.2 Nitrate-N concentrations at Village of Whiting well. (The apparent drop from 1980 to 1987 may be due to the well being largely unused during this period.)

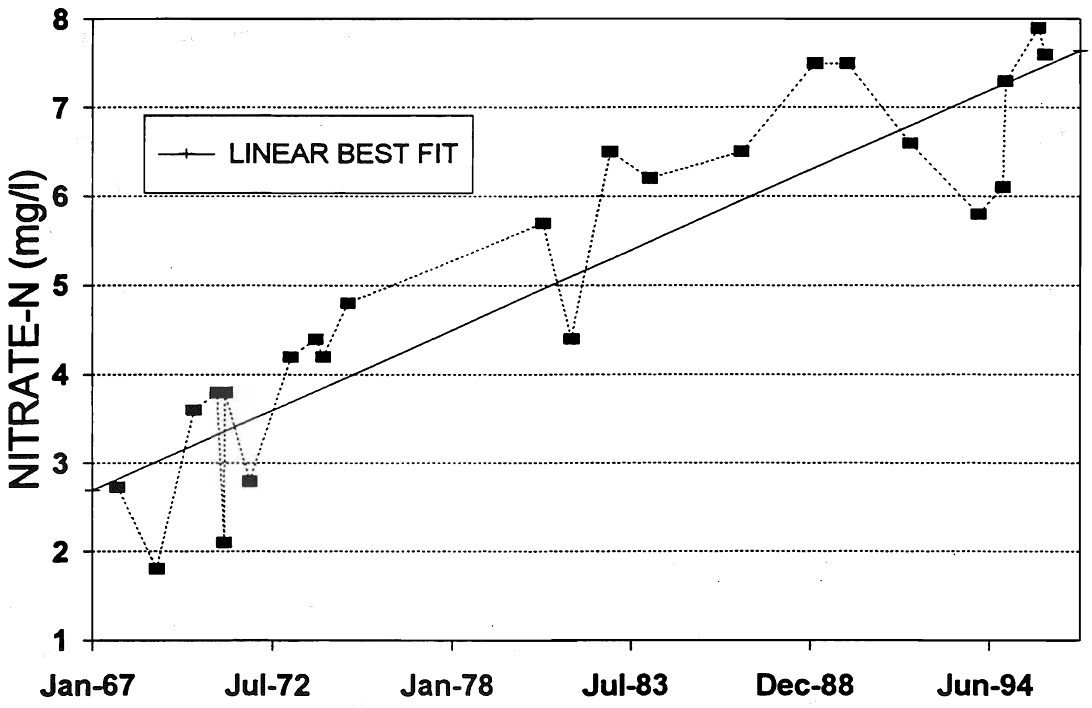

Figure 1.3 Nitrate-N concentrations in Little Plover River baseflow, from sampling at Hoover Road.

groundwater recharge rates to predict steady-state groundwater quality at the wellheads under current and anticipated land uses.

ORGANIZATION OF REPORT

Chapter 1	A brief introduction to the setting, the problem and needs, and the activities undertaken to meet the needs.
Chapter 2	Characterization of the Stevens Point, Whiting, and Plover area. This provides the information needed to develop a conceptual model of groundwater conditions and to satisfy computer model data requirements. Land use information is extracted from the Portage County GIS and specific land management data is developed.
Chapter 3	Describes the development and calibration of a groundwater flow model based on data contained in Chapter 2. The groundwater flow model mimics the water table configuration and the flow dynamics within the model area.
Chapter 4	Describes the development and application of a particle tracking model to refine the Stevens Point, Whiting, and Plover WHPA and time-of-travel delineations. The particle tracking model uses the output of the flow model developed in Chapter 3.
Chapter 5	Describes the process and results of calculating nitrogen loading rates for specific land uses and crops.
Chapter 6	Quantifies the nitrogen loading for the zones-of-contribution delineated in Chapter 4, based on the loading rates developed in Chapter 5. Nitrate concentrations at the wellheads are calculated for existing and anticipated land uses and land management practices. The effect of various management schemes are quantified to help identify urban and agricultural management strategies for protecting the wellheads.
Chapter 7	Reviews the findings of this study and discusses the nitrogen loading/land use relationship for the Stevens Point, Whiting, and Plover area, and the implications for wellhead protection. Recommendations are noted for management strategies and educational outreach.

CHAPTER 2
STUDY AREA CHARACTERIZATION

EXISTING INFORMATION

Considerable hydrogeologic information was available for portions of the Stevens Point, Whiting, and Plover (SWP) area. These data were reviewed in order to develop the conceptual model of groundwater flow in the municipal well zones-of-contribution, develop a database of site information for implementing a numerical groundwater flow model, and identify data deficiencies.

Site Location

The SWP urban area is located along the Wisconsin River in Portage County in central Wisconsin (Figures 1.1 and 2.1). The Wisconsin River is the most prominent hydrologic feature, flowing from the northwest to the southeast and turning westward at Plover. Several dams are sited in the area, taking advantage of considerable drop in the river elevation. The Consolidated dam just south of Highway 10 creates the Stevens Point Flowage at an elevation of 1086 ft MSL. The Consolidated dam in Whiting creates a small pool at elevation 1068 ft MSL, and the Kimberly Clark dam in Whiting creates a small pool at elevation 1046 ft MSL. The river elevation in the Plover area is approximately 1035 ft MSL. Tributaries to the Wisconsin River include Hay Meadow Creek in the northwest, the Plover River passing through Stevens Point and Whiting, and the Little Plover River in the Plover area.

Four distinct wellfields provide the water supply for the SWP municipalities (Figure 2.2). The Stevens Point main wellfield (airport wellfield) consists of 5 wells (#6 - #10) located between the municipal airport and the Plover River. Wells #6 - #9 are 20 inch diameter wells constructed in 1967/1968 (Donohue, 1991). Well #10, constructed in 1994, is a large capacity horizontal collector type well (RUST, 1993). The airport wellfield was sited to take advantage of clean sand and gravel deposits and induced recharge from the Plover River. Stevens Point #5 well, located south of Highway 10 in Iverson Park, is an older 16 inch well constructed in 1966 (Donohue, 1991).

The Whiting wellfield is located along County Highway HH on the east side of the village. This wellfield consists of 6 high capacity wells; #1 serves the Village of Whiting, #2 - #4 serve the Consolidated Paper Company, and #5 and #6 are used by the Kimberly Clark Paper Company. Well #1 is a 16 inch diameter well, constructed in 1964 (WGNHS, 1986).

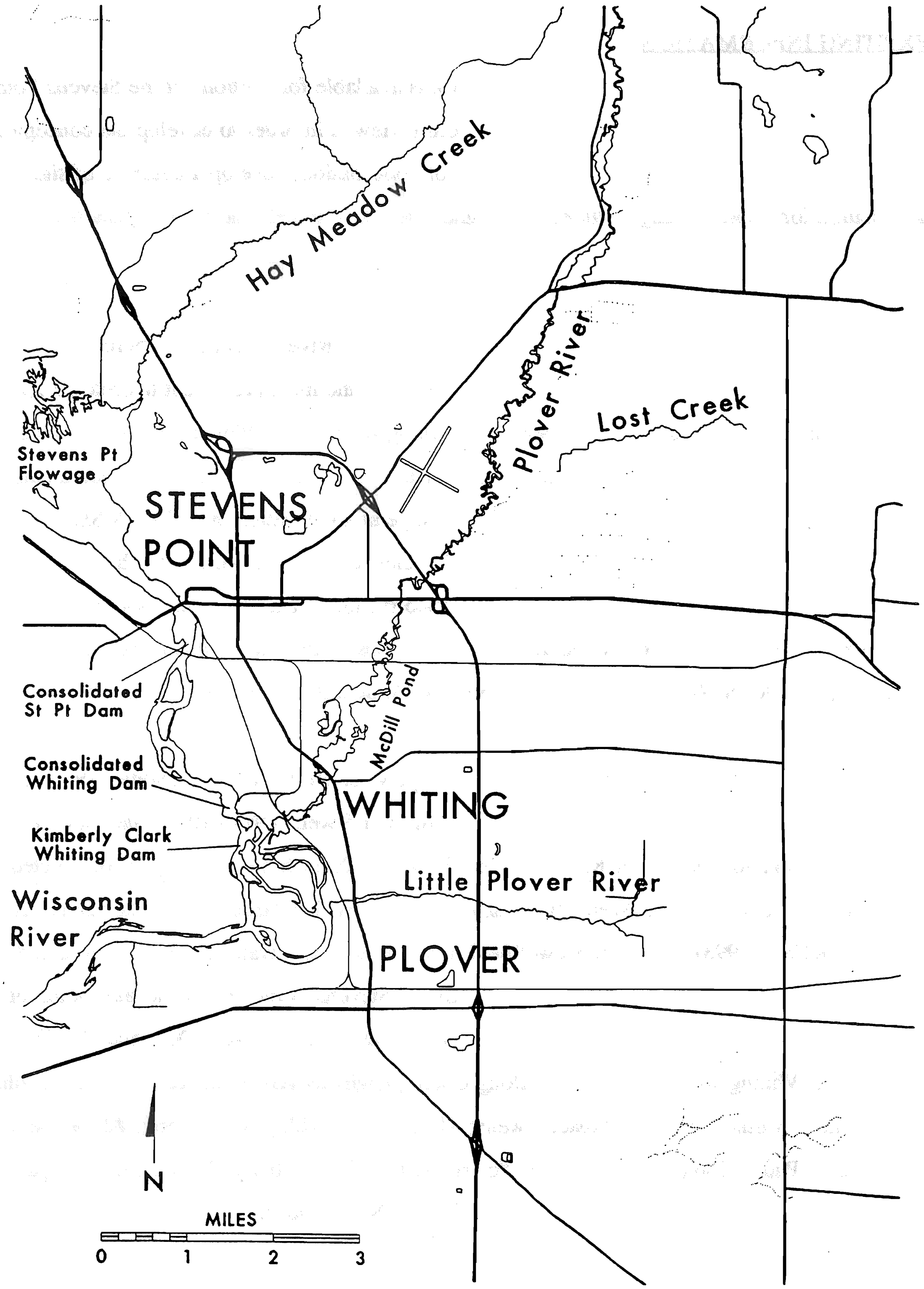

Figure 2.1 The Stevens Point, Whiting, and Plover study area.

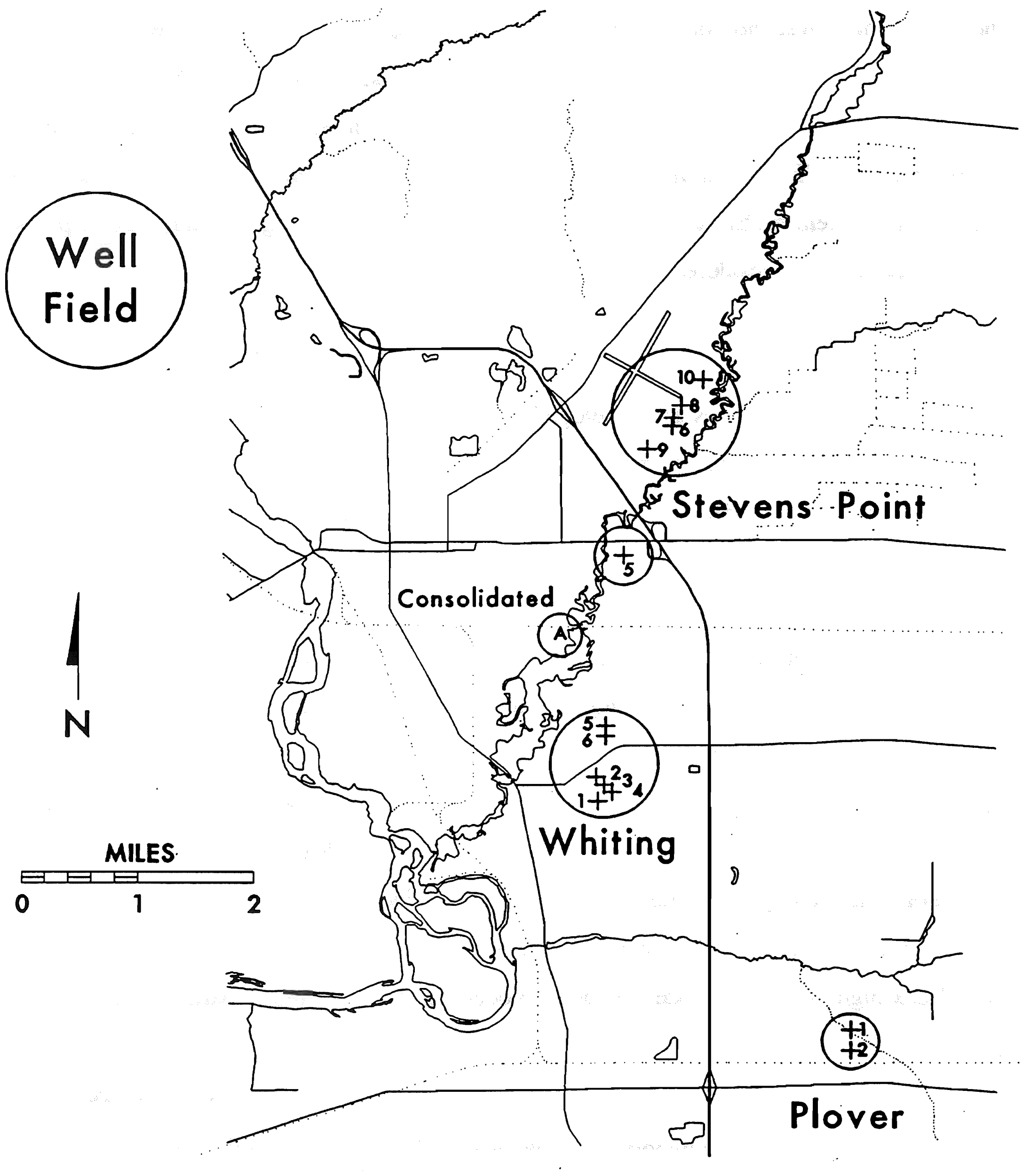

Figure 2.2 The Stevens Point, Whiting, and Plover wellfields.

The Plover wellfield is located on the far eastern edge of the village, just north of County Highway B. This wellfield contains two 20 inch diameter wells, constructed in 1988. Prior to this, the Village of Plover did not have a municipal water supply system (Donohue, 1989).

Consolidated Paper Company also operates an industrial well (A, Figure 2.2) along the Plover River at Patch Street between the Whiting and Stevens Point #5 wellfields. Current total pumpage is approximately 3.75 cfs. At this pump rate, the Consolidated well does not affect the shape of municipal wellfield recharge areas. Should pump rates increase significantly, the impacts of this well would need to be reconsidered.

Aquifer Units and Characterization

The municipal wells are developed in a surficial sand and gravel aquifer of glacial origin. Geologic cross-sections and stratigraphy in the vicinity of Stevens Point and Whiting are typical for the recharge area (Figure 2.3). A "bowl" of unconsolidated glacial meltwater deposits forms a significant unconfined aquifer to the east of the urban area. For the most part, these deposits overlay Precambrian granitic bedrock and a thin layer of bedrock residuum type material (hillslope deposits) that effectively defines the lower extent of the sand and gravel aquifer. Cambrian sandstone is found in the southern part of Portage County, but does not occur in the SWP area except for isolated mounds in the Little Plover River area.

General groundwater flow within this aquifer (Figure 2.4) is towards the Wisconsin River and Plover and Little Plover tributaries (Lippelt and Hennings, 1981). The eastern flow system boundary is a topographic and groundwater high in glacial moraines, approximately 5 miles east of the SWP area, which forms a divide between the Wisconsin River and Tomorrow/Waupaca River watersheds. A bedrock high and groundwater divide also occur in the northwest separating flow between the Wisconsin River/Hay Meadow Creek Tributary and the Plover River.

Holt (1965) described this area as the sand-plain province, an extensive and relatively uniform area of well-sorted Pleistocene outwash sand and some gravel, with little silt or clay. Clayton (1986) describes the Pleistocene material as slightly gravelly sand, gravelly sand, and sandy gravel of the Horicon formation. The material was generally deposited by shallow, braided, melt-water streams extending west from the western extent of the Green Bay lobe of late Wisconsin age. Because of the depositional environment, the outwash tends to be coarser near the source (moraines) and becomes finer moving to the west (Rothschild, 1982; Weeks and Stangland, 1971; Clayton, 1986).

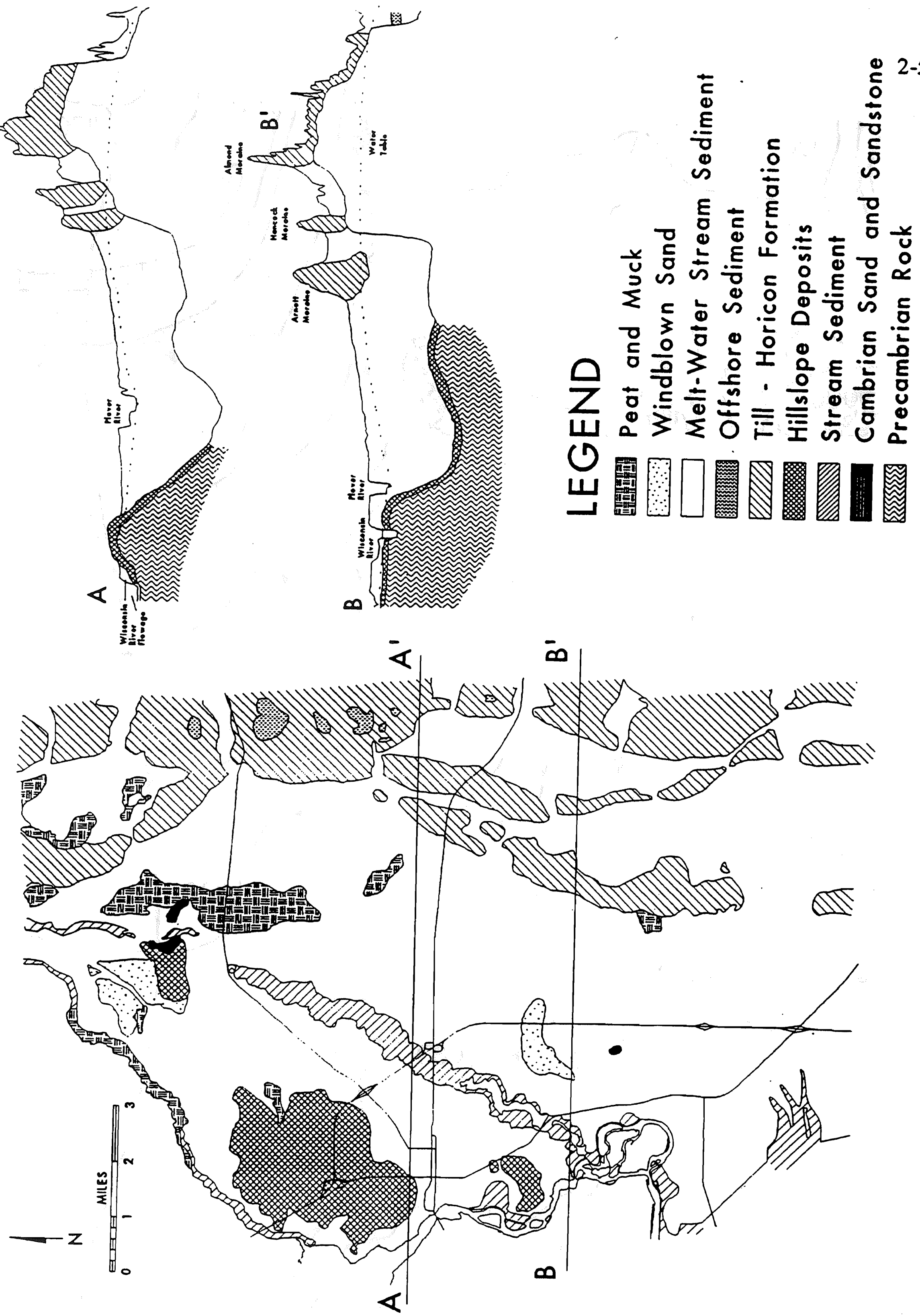

Figure 2.3 Pleistocene geology of the Stevens Point, Whiting, and Plover area (adapted from Clayton, 1986).

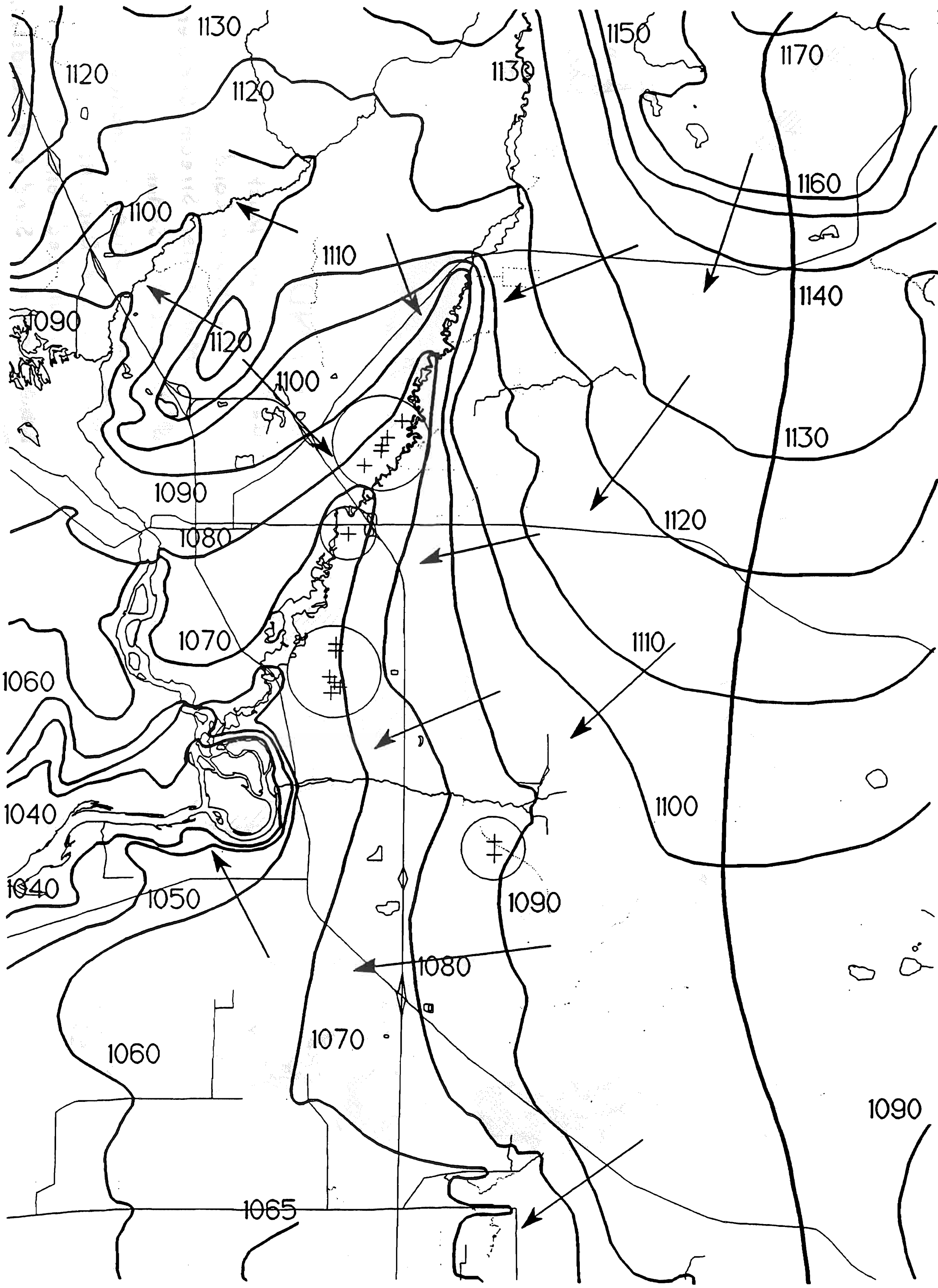

Figure 2.4 Water table elevation in feet MSL (adapted from Lippelt and Hennings, 1981).

No extensive or contiguous silt or clay bedding is noted in the literature for the outwash in the SWP area. The sand and gravel layering noted in well logs for Stevens Point #9, Whiting #2, and Plover #2 is typical for the area (Figure 2.5).

Thickness of the outwash materials in the SWP area averages around 100-115 feet in the Plover wellfield area, and generally thins to the north (Holt, 1965; Clayton, 1986).

The moraines forming the eastern boundary of this province (Figure 2.3) are glacial till and associated deposits, also of the Horicon formation. Typical of tills, these materials are locally variable and contain more silt and clay than the outwash materials. The groundwater divide roughly corresponds with the Hancock or Outer Moraine, westernmost of the moraines associated with the Mapleview member of the Horicon formation. Tills of this member are quite sandy, typically having 5 to 20% gravel, 80 to 90% sand, and 5 to 10% clay or silt (Clayton, 1986). The older Arnott moraine lies just west of the Hancock moraine, and is composed of finer grained and more weathered till of the Keene member of the Horicon formation. Particle size distribution is typically 2 to 20% gravel, 65 to 85% sand, 5 to 25% silt, and 8 to 17% clay (Clayton, 1986). This till overlays older outwash deposits.

Underlying the glacial till and outwash is a relatively thin layer of a more silty or clayey material. Clayton (1986) describes this material as hillslope deposits or mass movement deposits originally derived from locally weathered rock debris, typically 10-12 feet thick. Particle size distribution is typically 0 to 35% gravel, 15 to 65% sand, 15 to 70% silt, and 3 to 35% clay. Because of the finer texture, this layer has significantly lower permeability than the outwash.

The bedrock below the hillslope deposits is generally granitic in the SWP area, often noted to have several meters of rotten or decomposed rock at the contact. While some low yield wells are developed in weathered or fractured granite where significant unconsolidated materials are absent, the granite bedrock is generally considered impermeable and a physical boundary for the overlaying sand and gravel hydrostratigraphic unit (Holt, 1965).

Sandstone of the Dresbach Group of Late Cambrian age overlies the crystalline rock in the southern portion of Portage County, but is found only as isolated mounds in the SWP area. Holt (1965) noted two remnant sandstone mounds just east of Whiting and Plover in sections 2 and 14 respectively of T23N R8E. The mounds generally consist of poorly cemented medium to coarse grained sandstone capped by beds of well cemented sandstone. The mounds in the SWP area are not a significant aquifer because of their very limited extent and the much greater water supply available from the more permeable outwash.

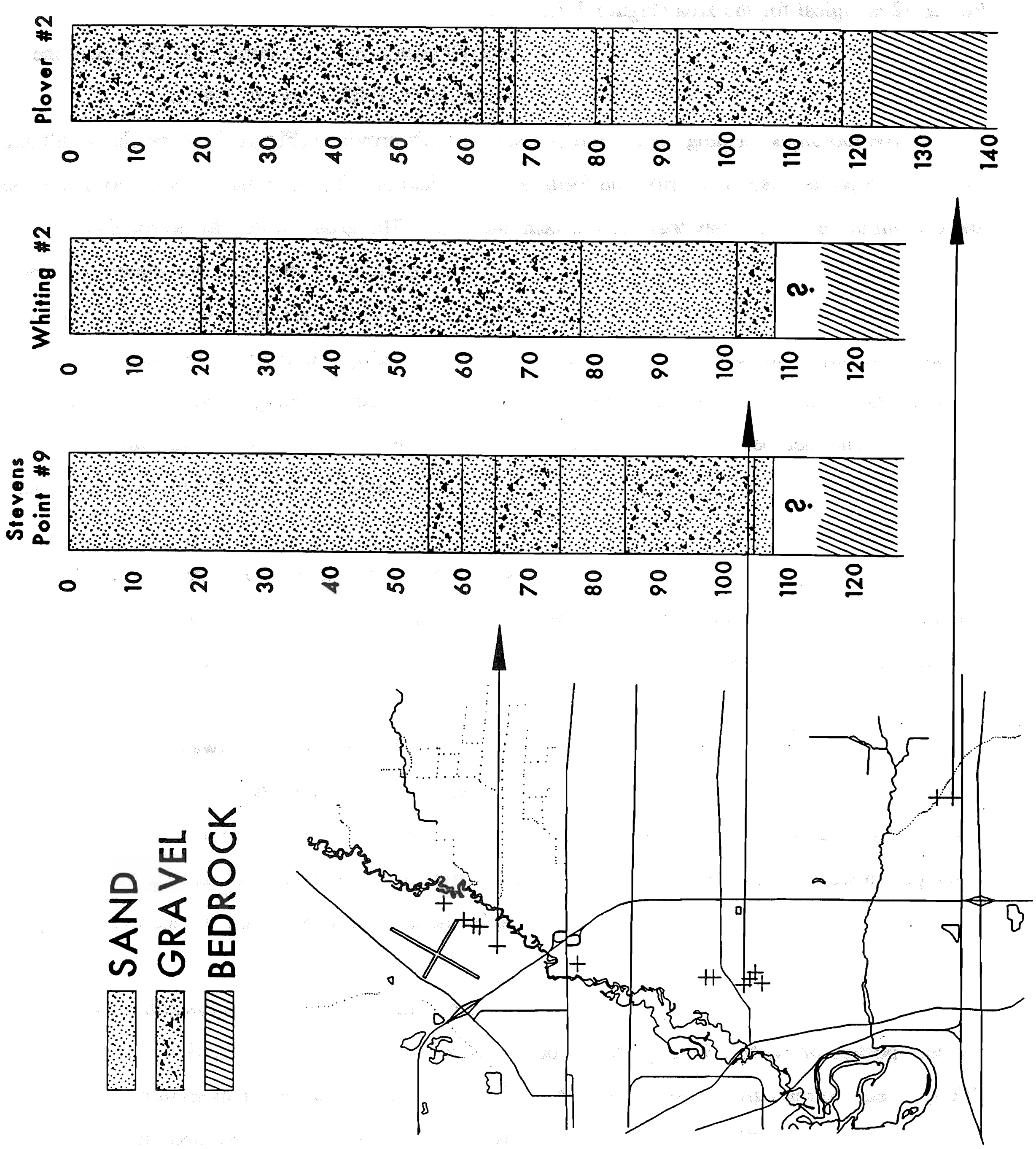

Figure 2.5 Example well logs for the Stevens Point, Whiting, and Plover municipal wellfields.

Hydraulic Properties

The SWP sand and gravel glacial outwash aquifer has a tremendous water supply capacity, estimated to be 500 to over 1000 gal/min (Devaul and Green, 1971).

There are several common hydraulic properties used to describe the storage and transmission of water in an aquifer. Porosity is the ratio of the void volume to the total volume, and represents the maximum water capacity at saturation. The average porosity, as determined by laboratory analysis, for well Pt362 (T23N R9E S18 in the Little Plover River basin) was 0.32 (Holt, 1965). Studies in other central Wisconsin sand plain areas found mean porosities of 0.38 (Stoertz, 1985) and 0.40 (Kraft et al., 1995).

Specific yield is the portion of the porosity from which water drains by gravity; again expressed as a ratio of the volume of drainable water to the total volume. The storage coefficient or storativity is the ratio of the volume of water released or stored per unit surface area per unit change in head. In unconfined aquifers, the storage coefficient is essentially the same as the specific yield. Holt (1965) and Weeks and Stangland (1971) suggest an average specific yield of about 0.2 for the sand and gravel aquifer. The average specific yield for the Stevens Point and Plover wellfields, as determined by pump tests, is approximately 0.21 (Table 2.1). These values of porosity and specific yield are typical for well-sorted sands and gravels and glacial outwash (Fetter, 1988).

Table 2.1. Hydraulic conductivity (K) and specific yield (SY) from pump tests for the SWP area.

Well	Location T/R/S	K ($x10^{-4}$ m/s)	SY	Thickness (m)	Source
Pt279	23/9/18	8.25	0.15	24.4	Holt 1965
Pt57/111	24/8/34	8.02	0.20	18.0	Holt 1965
Pt544	23/7/36	11.3	----	21.6	Weeks 1964
-------	22/8/2	17.9	0.20	32.9	Weeks & Stangland 1971
Pt567	24/8/26	32.0	0.10	27.1	Hickok 1965
StPt#6	24/8/26	44.3	0.25	25.3	Bahr et al. 1990
Plov#1	23/8/24	8.37	0.084	31.1	Donohue 1989
StPt#10	24/8/23	38.5	0.30	19.2	RUST 1993

The ability of the aquifer to transmit water is described by the hydraulic conductivity and transmissivity. Hydraulic conductivity is the rate of flow through a unit area under a unit hydraulic gradient. Hydraulic conductivity times the aquifer thickness yields the transmissivity, defined as the flow through a unit width of the aquifer under a unit hydraulic gradient. Hydraulic conductivity can vary considerably over an area, both spatially and vertically relative to the mix and layering of aquifer materials. Even the relatively homogeneous sands and gravels of the sand plain may have hydraulic conductivities that vary by several orders of magnitude (Table 2.2).

Table 2.2. Typical ranges for hydraulic conductivity for unconsolidated materials.

	Hydraulic Conductivity (m/sec)
Clay	10^{-11} to 10^{-8}
Silt	10^{-8} to 10^{-6}
Silty Sands, Fine Sands	10^{-7} to 10^{-5}
Well-sorted Sands, glacial outwash	10^{-5} to 10^{-3}
Well-sorted Gravel	10^{-4} to 10^{-2}

Source: Fetter, 1988

Hydraulic conductivity can be estimated by laboratory methods (grain size, permeameter), field tests (slug, specific capacity, pump), or by inverse modeling. Generally, the representativeness of the results is directly proportional to the extent of the aquifer being tested (Bradbury and Muldoon, 1990). For example, a laboratory analysis deals with a very small, disturbed sample of the aquifer material while a pump test measures the response of a larger volume of the aquifer delineated by the monitoring wells. Pump tests are generally considered the best method for characterizing the aquifer permeability, although these tests can be costly and time consuming.

Specific capacity, defined as the yield of the well per unit of drawdown, is a common measure of the productivity of a well, usually reported by the well driller on the well log. Specific capacity, while influenced by the well efficiency, is a useful index of the aquifer storage and permeability, and can be used to calculate an estimate of hydraulic conductivity (Rothschild, 1982; Bradbury and Rothschild, 1985). Holt (1965) reported an average specific capacity of 60 gpm/foot of drawdown for the sand plain province near Stevens Point. For comparison, the specific capacity of the area of mixed sandy till and outwash to the east of the sand plain province averaged 24 gpm/ft.

Hydraulic conductivity measurements are sparse for the SWP area. Hydraulic conductivities derived from 8 pump tests ranged from 8.02 $\times 10^{-4}$ to 44.3 $\times 10^{-4}$ m/s (Table 2.1). The higher values in this set appear to be related to localized areas of coarser materials deposited in a preglacial bedrock valley (Rothschild, 1982; Brown et al., 1992). The results of 4 slug tests conducted by Brown et al. (1992) at Stevens Point wellfield monitoring wells mirror the pump tests. Hydraulic conductivity for wells identified as penetrating preglacial bedrock valley sediments ranged from 29 to 60 $\times10^{-4}$ m/s, while a nearby glacial outwash type well ranged from 7.8 to 9.2 $\times10^{-4}$ m/s. Another well that apparently was developed in the bedrock residuum type material overlying the granitic bedrock had a conductivity of only 0.07 to 0.12 $\times10^{-4}$ m/s. A summary of hydraulic conductivity for the Buena Vista basin, a continuation of the sand plain aquifer south and southwest of the SWP area, was compiled by Bradbury et al. (1992). They found a mean hydraulic conductivity of 7.3 $\times10^{-4}$ m/s from 10 pump tests, 6.4 $\times10^{-4}$ m/s from 266 specific capacity calculations, and 9.0 $\times10^{-4}$ m/s from an inverse model solution. It appears that a typical hydraulic conductivity for the outwash sandplain in the SWP area would be around 8 to 9 $\times10^{-4}$ m/s. While the areal distribution of hydraulic conductivity may not vary greatly relative to more heterogeneous areas, spatial patterns related to such considerations as grain size distribution and buried valleys need to be characterized for implementing a numerical model.

Little hard information is available concerning the vertical hydraulic conductivity of the outwash materials. Based on detailed pump tests in the sand plain, Weeks (1969) determined that the ratio of vertical to horizontal hydraulic conductivity ranges from 1:2 to 1:20. Defining the vertical hydraulic conductivity of river and drain beds is important to the accurate representation of these features in a numerical model. This job is especially difficult because of the variable scouring and deposition in these environments. Osborne and Shaw (1988) estimated the hydraulic conductivity of the Plover River bottom sediments in the vicinity of the Stevens Point wellfield by regional watershed analysis, discharge measurements, and seepage meters. They found values ranging from 0.3 to 2.0 $\times10^{-4}$ m/s, with an average of 0.85 $\times10^{-4}$ m/s for the three methods.

<u>Bedrock Surface</u>

The bedrock surface is the bottom physical boundary of the aquifer. Because the outwash aquifer is such a good water source, most wells are finished short of the underlying bedrock and therefore do not provide a good delineation of the bedrock surface. Numerous studies on specific areas of the SWP or adjacent areas suggests significant bedrock topography hidden under the flat sand plain.

Holt (1965) presented a generalized bedrock surface map for Portage County with 100 ft contours. Weeks et al. (1965) described the bedrock geometry with 10 foot contours for the Little Plover area based on project well borings. Weeks and Stangland (1971) mapped the bedrock geology with 20 foot contours for an extensive portion of the sand plain south and west of Plover, primarily south of the SWP area. Hickok (1981) mapped the bedrock surface in some detail in the Stevens Point airport area using test borings and seismic soundings. Osborne (1988) and Brown et al. (1990) compiled information on the bedrock surface for the Stevens Point airport wellfield using well logs as part of a groundwater flow modeling effort. Unfortunately, these coverages are either at too small a scale to be useful for modeling, or are not complete for the SWP area. It is also apparent that these project oriented coverages do not have the continuity of mapping that an areawide interpretation of all data would yield.

The available data suggests a bedrock surface that generally slopes from the north to south, with significant local variation in the form of pre-glacial bedrock valleys. Bedrock outcrops at elevation 1100 ft MSL are visible along the Plover River at Highway 66 northeast of the Stevens Point airport wellfield. In the southern portion of the SWP area in the vicinity of the Plover wellfield, the bedrock is at an elevation of 990 to 1000 ft MSL, and is covered by approximately 100 feet of outwash. Available data suggests significant buried valleys approximately 2 miles east of Whiting, trending to the south and southwest, and just southeast of the Stevens Point airport wellfield, trending northeast to southwest.

Water Table

Considerable information is available describing the general configuration of the water table because surface water (which can be thought of as a groundwater outcrop) elevations are known, and because groundwater elevations can be calculated from numerous production well logs. Unfortunately, these data were collected over many years and therefore are not an instantaneous snapshot of water table conditions. This is a source of error because the water table fluctuates with short term, seasonal, and long term trends. There are also spatial differences in water table fluctuations due to variable recharge/discharge dynamics. As an example, four long-term monitored wells in unconfined sand and gravel aquifers in Wisconsin had an annual amplitude ranging from 0.16 ft to 4.32 ft, with an overall amplitude ranging from 6.08 to 26.72 (Patterson and Zaporozec, 1985). The water-level hydrograph for Well Pt376 in the SWP (Figure 2.6) notes an amplitude of approximately 14 feet for the period 1960 through 1981 (Erickson and Cotter, 1983).

Existing water table maps are available from several sources. Holt (1965) mapped the water table with 20 foot contours based on wells, irrigation pits, and other surface features. A more recent map by Lippelt and Hennings (1981), although limited by the lack of data in certain areas, provides a good approximation of the water table for the SWP area (Figure 2.4) at a regional scale.

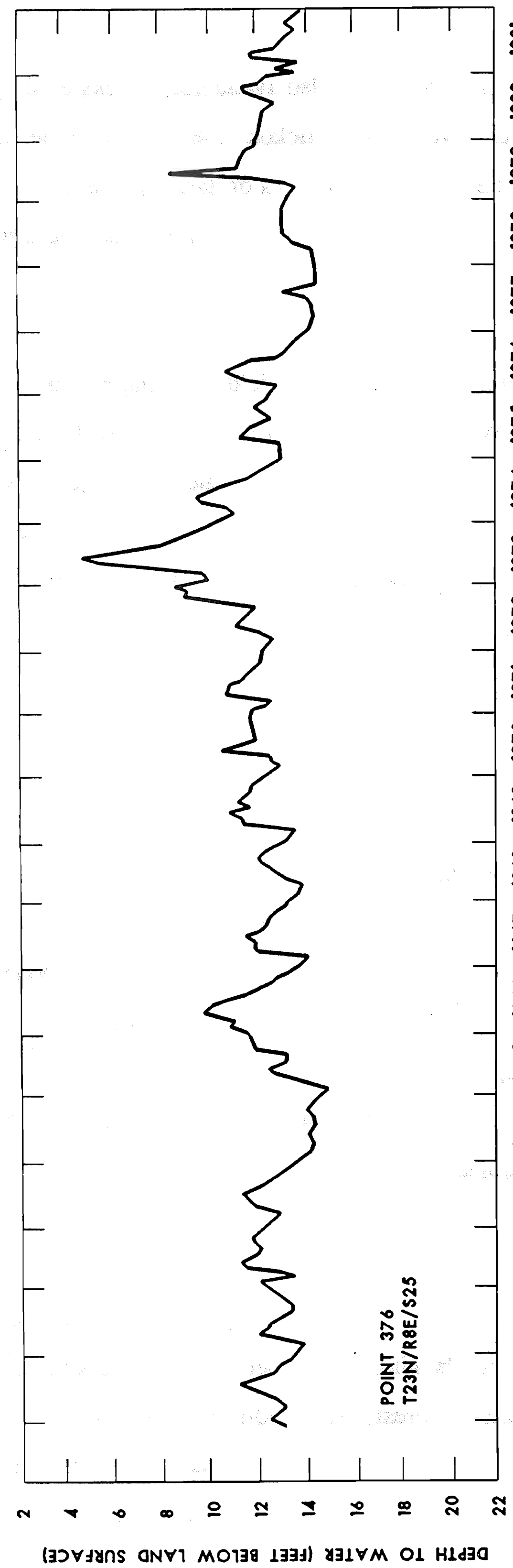

Figure 2.6 Water-level hydrograph for well Pt376 (adapted from Erickson and Cotter, 1983).

Project scale water table maps are also available. Weeks et al. (1965) mapped 5 foot contour intervals for the Little Plover River basin. Hickok (1981) mapped the water table with 2 foot contours for the Stevens Point airport area. These sources of data represent field projects, and the resulting maps provide a detailed description of the water table at that specific time.

Recharge

Holt (1965) estimated the annual water yield of Portage County to be 10.6 inches, defined as the excess of precipitation over estimated evapotranspiration (31.4 inches - 20.8 inches). The water yield includes both surface water runoff and groundwater recharge. Because surface water runoff is minimal for the SWP area because of flat topography and highly permeable soils and outwash, 9 to 10 inches of recharge is a commonly used estimate. For example, 91% of the Little Plover River flow is from groundwater (Weeks et al., 1965).

Stoertz (1985) estimated an average rate of recharge of 8 inches per year for the central sand plain of Wisconsin using an inverse modeling technique. Bradbury et al. (1992) calculated an overall recharge rate of 13 inches in recharge areas of the Buena Vista Basin, the area of the sand plain south and southwest of the SWP area.

The areal pattern of recharge rates can be complex (Stoertz, 1985; Bradbury et al., 1992). A computer derived recharge map of the area surrounding and upgradient of Whiting indicated rates varied from less than 2 inches to more than 12 inches per year (WGNHS, 1986). Land use activities have obvious influences on recharge rates. Irrigated agriculture increases the evapotranspiration component of the water budget, decreasing the net amount of recharge. Irrigated corn, potatoes, and beans were found to increase evapotranspiration by approximately 5.4, 5.9, and 3.4 inches respectively (Weeks and Stangland, 1971).

Land Use

Land use maps are available for all of Portage County, based on the best available information through 1986 (Portage County Planning Department, 1987). General land use categories include agriculture, irrigated agriculture, forest, grassland/brushland/undeveloped, surface water, unsewered residential, sewered urban, mobile home, recreation, industry, utility, airport, and extraction. The land use maps for the SWP area are available as a GIS coverage (Figure 2.7). Three land use categories, irrigated agriculture, forest, and dryland agriculture, account for nearly three fourths of the total SWP area.

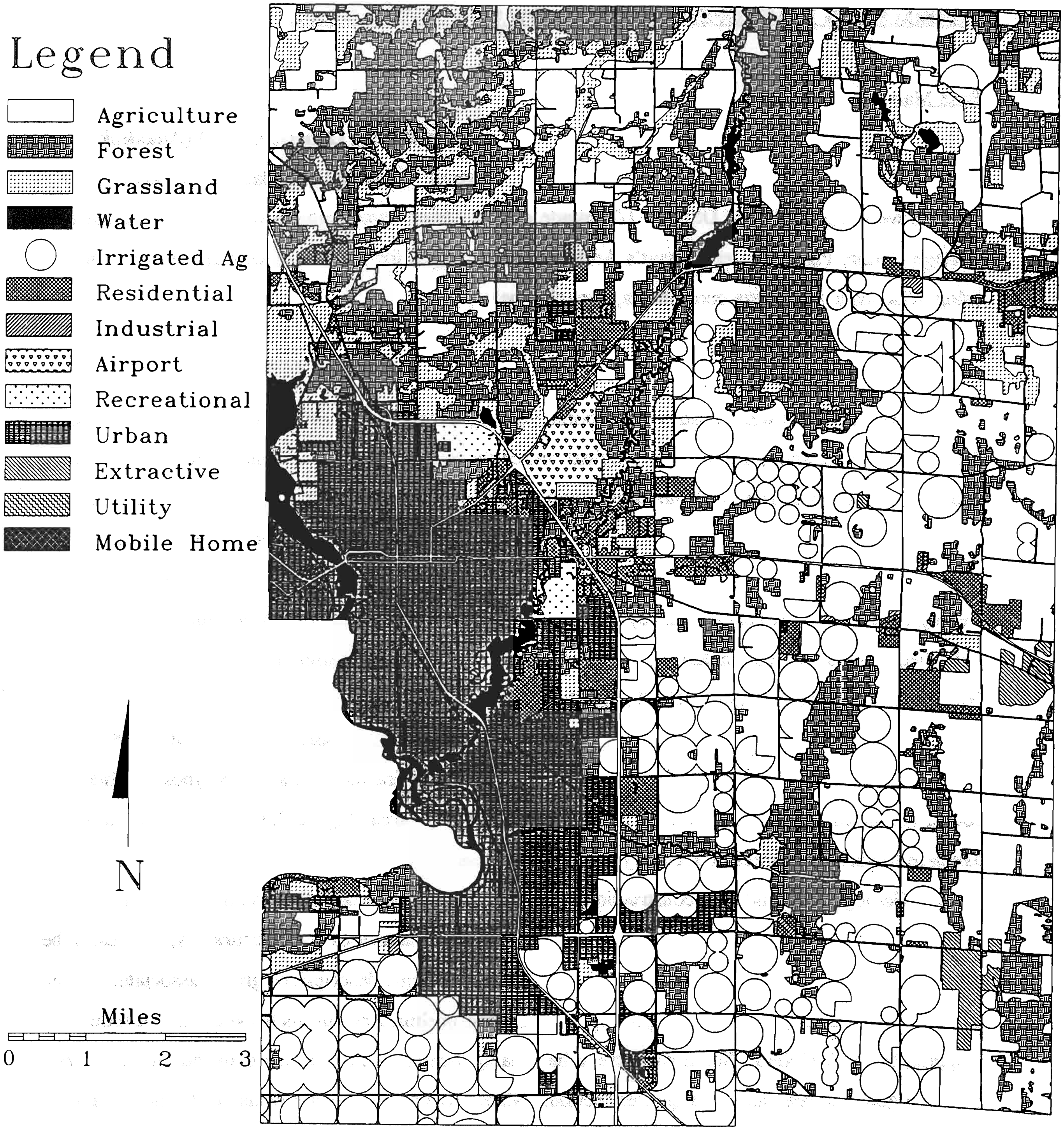

Figure 2.7 Land use in the Stevens Point, Whiting, and Plover area.

INFORMATION DEVELOPED FOR GROUNDWATER FLOW MODEL

Base Map

A base map for the SWP area (Figure 2.1) was created using Generic CADD (Autodesk, 1992). The road network, section corners, railroads, airport, municipal boundaries, and some water features were digitized from USGS 7 1/2 minute quads. Other water features were obtained from the Portage County Planning Department's ArcInfo GIS coverages for the area. All mapping was done using Wisconsin state plane coordinates, central zone.

Well Information Database

Well logs and well construction reports are a significant source of hydrogeologic information for site characterization, and were an original source for much of the background information noted above. In order to update available information as much as possible and to analyze it with consistency and continuity for the entire SWP area, a database of well geologic logs and construction reports through 1988 available from the Wisconsin Department of Natural Resources and the Wisconsin Geological and Natural History Survey was compiled. Some additional well information described in published reports was also included (Weeks et al., 1965; Weeks and Stangland, 1971; Hickok, 1965; Hickok, 1981; Donohue, 1989; RUST, 1993).

The database was implemented using the PC based dBASE IV software (Borland, 1988). Fields were included for well ID, location, construction, pump drawdown test, rock types, confidence codes, and notes (Table 2.3). A total of 694 wells in the SWP area (Figure 2.8) are described in the database to various degrees of accuracy and completeness.

Geologic logs and well construction reports were included when they could be located accurately enough to be useful. Well construction reports are notorious for inaccuracies, and had to be critically reviewed. Location decisions were made based on legal descriptions given, associated owner and plat book information, USGS topo map features, and preliminary but unchecked well location mapping by the WGNHS. Some wells could be reliably located to a point on a map, but many others only to a general area, such as a quarter section. Ninety-five percent of the wells are located to a 40 acre or smaller parcel. The degree of mapping resolution is recorded in the database and can be used to qualify application of the database as needed. The state plane coordinates were determined by digitizing the point or generalized location of the well on a 7 1/2 minute USGS topographic map.

Table 2.3. Table structure for well log database.

Field	Field Name	Type	Width/Dec	Description
1	POINT	Numeric	4	Pt serial number
2	IRR	Numeric	4	Irrigation well serial number
3	PERM	Numeric	5	Permanent well ID number
4	MID	Numeric	4	Miscellaneous well log identifier
5	WUWN	Character	5	Wisconsin Unique Well Number
6	WID	Numeric	4	Database unique well ID
7	TWN	Numeric	2	Township (N)
8	RNG	Numeric	2	Range (E)
9	SEC	Numeric	2	Section
10	GOVLOT	Numeric	2	Government lot number
11	QQQ	Character	2	Quarter/quarter/quarter section
12	QQ	Character	2	Quarter/quarter section
13	Q	Character	2	Quarter section
14	X	Numeric	1	State plane easterly (central zone)
15	Y	Numeric	1	State plane northerly (central zone)
16	QUAD	Character	15	Name of USGS topo map (7 1/2 min)
17	DATE	Character	10	Well construction date
18	LOG	Character	10	Abbreviated geologic log
19	ELEV_SUR	Numeric	7/2	Elevation of land surface (ft MSL)
20	ELEV_WAT	Numeric	7/2	Elevation of water table (ft MSL)
21	ELEV_BED	Numeric	7/2	Elevation of bedrock surface (ft MSL)
22	DEP_WEL	Numeric	6/2	Depth of well (ft)
23	DEP_WAT	Numeric	6/2	Depth to water (ft)
24	DEP_BED	Numeric	6/2	Depth to bedrock (ft)
25	CASE	Numeric	6/2	Casing length (ft)
26	LGTH	Numeric	6/2	Screen length or open interval (ft)
27	DIAM	Numeric	6/2	Diameter of well (inches)
28	SC	Numeric	7/2	Specific capacity (gal/min/ft)
29	T	Numeric	12/6	Transmissivity (m^2/sec)
30	K	Numeric	12/6	Hydraulic conductivity (m/s)
31	GPM	Numeric	7/2	Pump rate (gal/min)
32	LN	Numeric	6/2	Length of pump test (hours)
33	PUMP	Numeric	6/2	Depth to water when pumping (ft)
34	AQTHIC	Numeric	6/2	Aquifer thickness (ft)
35	S	Numeric	6/4	Storage coefficient
36	C	Numeric	6/3	Well loss coefficient
37	STATUS	Character	30	Status of transmissivity calculation
38	BEDWELL	Logical 1		Well developed in bedrock (Y/N)
39	HITBED	Logical 1		Well hit bedrock (Y/N)
40	SOURCE	Character	5	Code for source(s) of well information
41	LOC_CODE	Numeric	2	Code for resolution of well location
42	BED_CODE	Numeric	2	Code for reliability of bedrock data
43	NOTES	Character	7	General notes

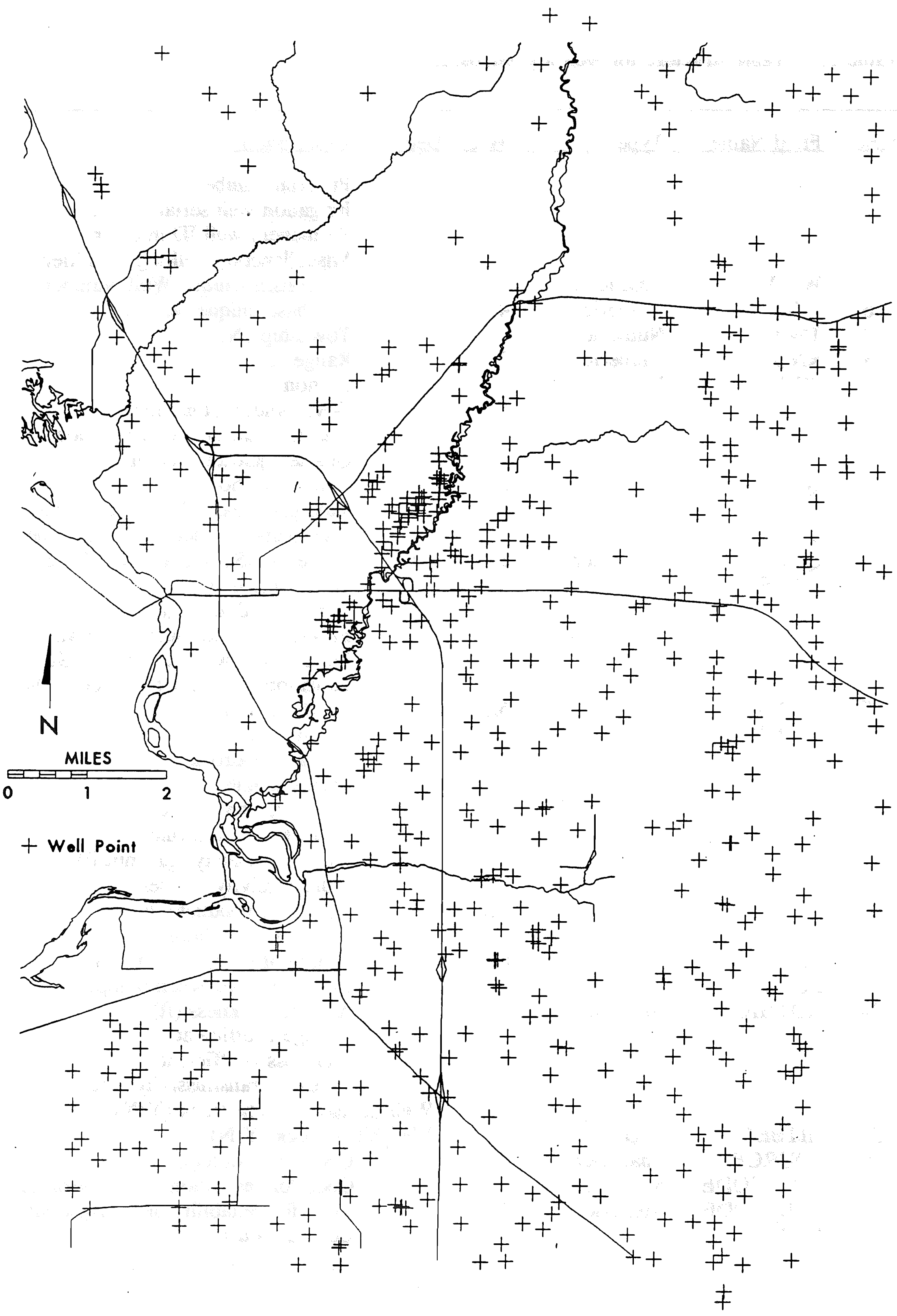

Figure 2.8 Location of wells included in the Stevens Point, Whiting, and Plover well log database.

The surface elevation data contained in the database was usually estimated from USGS 7 1/2 minute topographic maps. Because of the relatively flat surface of the outwash, surface elevation for wells in this area could usually be estimated to plus or minus 2.5 feet. Wells in the moraines with much greater relief could only be estimated when a point location could be determined for the well, and then to plus or minus 10 feet. The surface elevation could not be estimated for 32 wells because of generalized locations and variable relief. Bedrock and water table elevations were calculated relative to the surface elevation. The depth to bedrock or water reported on the well log was subtracted from the estimated surface elevation.

The geologic log information was only captured in an abbreviated form in the database. A review of well logs indicated that the outwash material was relatively uniform and would be treated as a single hydrostratigraphic unit for the regional scale SWP model.

Bedrock Elevation Mapping

A composite map of the bedrock surface for the contiguous SWP area was made using USGS 7 1/2 minute topographic maps for bedrock outcrops and bedrock data from the well information database.

As previously noted, the bedrock is covered by thick unconsolidated glacial deposits in much of the SWP area. While it is known that the bedrock is closer to the surface in the north and northwestern portion of the SWP area, there are few distinct bedrock outcrops. The Plover River intersects a granite outcrop at an elevation of approximately 1100 ft MSL immediately below the Jordan Dam at Highway 66. Holt (1965) and Weeks et al. (1965) map two sandstone mounds just east of Whiting and Plover. These mapped mounds could not be verified from outcrops or well logs. However, a sandstone mound does outcrop along Hoover Road in the SW, NW, S14, T23N, R8E at an elevation of 1090 ft MSL. It is a roughly circular mound approximately 15 feet in height. A bedrock contact or likely contact was also noted for 180 wells in the well information database.

This bedrock surface information was contoured using Surfer (Golden Software, 1990). The resulting preliminary contour map was carefully analyzed for logical inconsistencies or unusual features generated by a lone well point. Eight of the well logs reporting bedrock contacts were subjectively deemed unreliable, leaving 172 well contacts and 2 outcrops.

The other well logs in the database have no indication of bedrock, although many are thought to be finished just above the bedrock. A possible bedrock contact for these was calculated as the surface elevation minus the depth of well, rounded down to the nearest five foot increment. These possible bedrock elevations were posted on the preliminary bedrock contour map. Possible bedrock elevations significantly below the previously mapped bedrock contours indicated a lower bedrock elevation was appropriate at those sites. These were carefully reviewed and only consistent or clustered wells were used. The bedrock contours were then redetermined using these additional wells. This process was automated with a dBASE utility and repeated until all likely near-bedrock wells were identified. An additional 84 wells were identified in this way. While improving the accuracy of the bedrock contours, the use of assumed bedrock elevations only proposes that the bedrock surface is no higher than this, but could, in fact, be lower. Sensitivity of the groundwater flow model to bedrock elevation is noted in Chapter 3 and Appendix C.

The final contour map of the bedrock surface (Figure 2.9) ties many of the features noted in previous reports together for the SWP area. The bedrock surface rises from an elevation of 980 ft MSL in the southern portion of the SWP to 1120 ft in the northwest. Bedrock highs are apparent northwest of the Stevens Point airport, 1 to 2 miles east of Highway 51 along Highway 10, and in the vicinity of the Little Plover River just west of Highway 51. Deep bedrock valleys can be traced past the Stevens Point wellfield, south towards the Whiting wellfield, and southwesterly from the Plover wellfield. While this portrait of the buried bedrock surface is the best available information, it is obvious that 172 known data points and 84 assumed data points can still only provide a generalized description of a 150 square mile area.

Hydraulic Conductivity Mapping

A regional scale interpretation and mapping of hydraulic conductivity was accomplished by estimating the hydraulic conductivity from well construction report specific capacity pump test data using the method of Bradbury and Rothschild (1985). This method uses a computerized iterative technique that corrects the specific capacity data for partial penetration and well loss. The computer code was adopted to the dBASE programming language; input data was read directly from the well information database and calculated results written directly back to the database.

Input variables required include well diameter, screen length or open interval, length of the pump test, pump rate, aquifer thickness, the storage coefficient, the well loss coefficient, and depth to water for pumped and not pumped conditions. This data can generally be found on well construction reports, except for the storage coefficient, well loss coefficient, and aquifer thickness. An average value of 0.20 (Holt, 1965; Weeks and Stangland, 1971) was used for the storage coefficient for all calculations. Well loss was assumed to be negligible (Rothschild, 1982). Aquifer thickness was taken

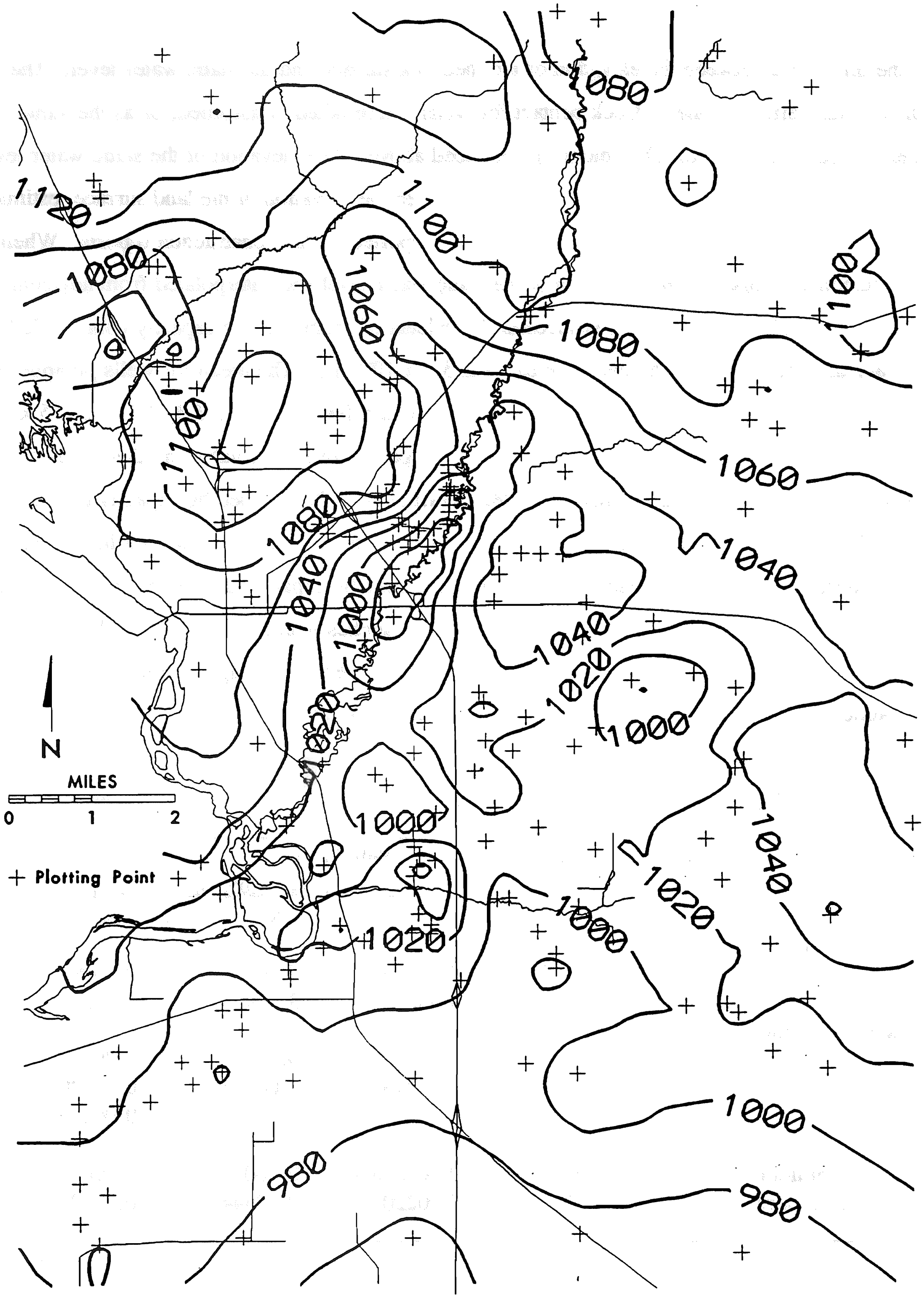

Figure 2.9 Bedrock elevation in the Stevens Point, Whiting, and Plover area (contours are in feet MSL).

as the difference between the elevation of the bedrock surface and the static water level. The bedrock surface was defined as the bedrock contact for wells where noted or assumed, or as the value interpolated from the bedrock contouring described above. The elevation of the static water level could generally be calculated as the difference between the elevation of the land surface, estimated from topographic maps, and the depth to water, as reported on the construction reports. When this information was missing, the elevation of the static water level was interpolated from adjacent wells.

Hydraulic conductivity was successfully calculated from specific capacity data for 327 wells in the database (Table 2.4). The overall mean was 5.1 x 10^{-4} m/s. The mean for wells pumping less than 100 gpm, between 100 and 1000 gpm, and more than 1000 gpm was 3.3, 7.1, and 9.5 x 10^{-4} m/s respectively. These values are comparable to the range of values calculated in other areas of the sand plain (Bradbury et al., 1992; Rothschild, 1982; Kraft et al., 1995). Specific capacity derived hydraulic conductivity compares favorably to that derived from pump tests, for the limited number of wells where both data are available (Table 2.5). The specific capacity derived hydraulic conductivity is lower for 2 of the wells and higher for 4 wells, but matches within a factor of 2. The greatest difference was noted for the Stevens Point #10 test well, where the pump test and specific capacity hydraulic conductivities were 38.5 and 17.4 x10^{-4} m/s respectively.

Table 2.4. Hydraulic conductivity (m/s) calculated from specific capacity data for the SWP area.

	Well Group			
	All	< 100gpm	≥100& < 1000gpm	≥1000gpm
Sample Size:	327	173	70	84
Geometric Mean:				
m/s	0.00051	0.00033	0.00071	0.00095
log m/s	-3.295	-3.486	-3.149	-3.022
Standard Deviation:				
log	0.419	0.416	0.302	0.290
-1 std dev	0.00019	0.00013	0.00035	0.00049
+1 std dev	0.00133	0.00085	0.00142	0.00185
Range:				
minimum	0.000008	0.000008	0.00017	0.00023
maximum	0.00927	0.00220	0.00544	0.00927

Table 2.5. Comparison of hydraulic conductivity determined from pump tests and specific capacity data.

	K (m/s x 10^{-4})		Specific Cap.	Pump Rate	Drawdown
	Pump Test	Calculated	(gpm/ft)	(gpm)	(feet)
Plover #1	8.37	5.77	40	1000	25
Pt279	8.25	8.91	52	1200	23
St Pt #10 (test)	38.5	17.4	100	1340	13.4
Pt567	32.0	48.5	200	1600	8
St Pt #6	44.3	48.5	343	2400	7

The differences related to pump rate probably reflect several factors that are difficult to separate. For many of the wells, the variation is related in turn to spatial differences in aquifer material and permeability. For example, the high capacity irrigation wells tend to be clustered in the more permeable sand and gravel outwash, while many of the smaller wells are residential wells located in the more variable moraines. Within the < 100gpm well category, the mean hydraulic conductivity calculated for 94 wells located in flat outwash was 4.2 x10^{-4} m/s, compared to 2.5 x10^{-4} m/s for 55 wells located in moraine areas.

To lessen the influence of spatial variability due to mapped lithostratigraphic units, the mean hydraulic conductivity was also calculated for the pumpage classes for only wells located in flat outwash ("su" unit of Clayton, 1986). While the means are closer for the three pumpage classes, the same trend with pump rate is apparent (Table 2.6). One factor may be the effects of scale. Bradbury and Muldoon (1990) found that the measured values of hydraulic conductivity tend to increase as the scale of the measurement increases. A well pumping at 1250 gpm samples a larger portion of the aquifer than a well pumping at 50 gpm. The low pumping rate wells also exhibit a higher degree of variability, as indicated by the higher standard deviation of the mean (Table 2.6). This may represent a greater impact of measurement and rounding errors.

A hydraulic conductivity distribution was constructed for the SWP area using the values calculated from specific capacity data (Figures 2.10 A,B). The calculated values were first filtered to remove inconsistent and questionable wells when compared to neighboring wells, or where too few wells were located to reliably characterize an area. Low pump rate wells and small wells (< 100gpm or < 8 inches in diameter) were also not used, except if they were the only data available for an area and appeared reasonably consistent. To smooth the spatial data distribution for a regional perspective, the hydraulic conductivity values were averaged by section, along with the x and y coordinates. A total of 215 wells were averaged to yield 68 plotting points. These points were gridded and contoured using Surfer (Golden, 1990).

Table 2.6. Hydraulic conductivity (m/s) calculated from specific capacity data for wells located in flat sand and gravel outwash in the Stevens Point, Whiting, and Plover area.

	Well Group			
	All	<100gpm	≥100&<1000gpm	≥1000gpm
Sample Size:	225	94	59	72
Geometric Mean:				
m/s	0.00061	0.00042	0.00068	0.00095
log m/s	-3.212	-3.382	-3.170	-3.024
Standard Deviation:				
log	0.379	0.401	0.301	0.301
-1 std dev	0.00026	0.00017	0.00034	0.00047
+1 std dev	0.00147	0.00104	0.00135	0.00189
Range:				
minimum	0.000019	0.000019	0.00017	0.00037
maximum	0.00927	0.00220	0.00544	0.00927

The hydraulic conductivity tends to be lower in the moraines in the eastern portion of the SWP area and in the bedrock high area in the northwest. A significant increase in hydraulic conductivity is apparent for the buried bedrock valley areas in the Stevens Point area. There is no apparent decrease in hydraulic conductivity westward across the outwash away from the source of deposition as might be expected, except possibly in the extreme southwest area. Possibly the bedrock valley environments mask this trend within the limited SWP area.

Water Table Mapping

A composite water table elevation map (Figure 2.11) was constructed from data in the well information database and surface water features noted on USGS 7 1/2 minute topographic maps. From the well information database, the elevation of the water surface was taken as the land surface minus the depth to water. Values that appeared to be in error or were suspect, based on comparisons to neighboring wells, previous water table mapping, or known points of elevation, were not used. A total of 330 well points appeared usable, although they represent a variable water table over many years of record. To help smooth these points, the 330 values were averaged by section to yield 77 plotting points.

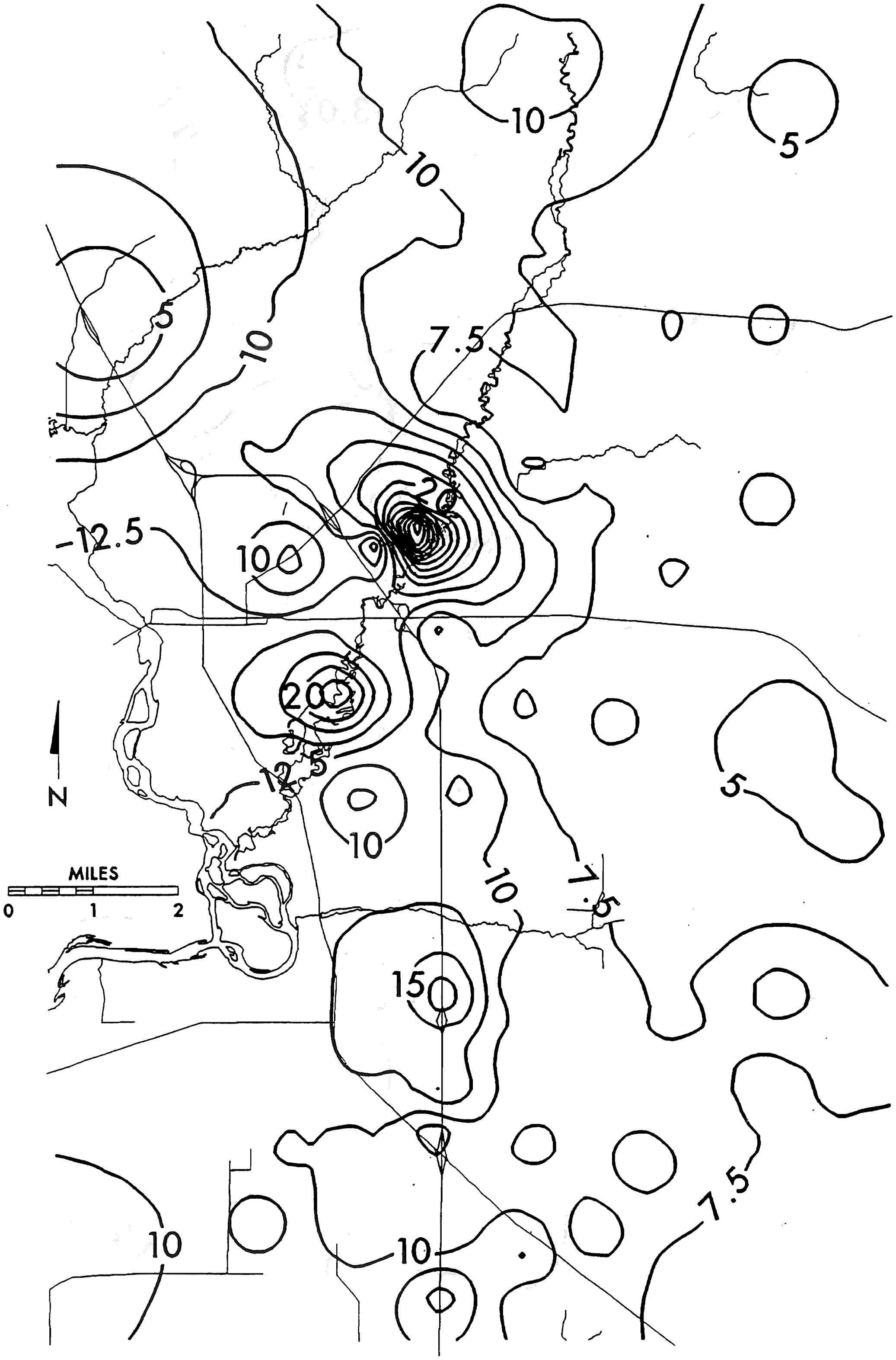

Figure 2.10A Hydraulic conductivity (x 10^{-4} m/s) calculated from specific capacity data for the Stevens Point, Whiting, and Plover area.

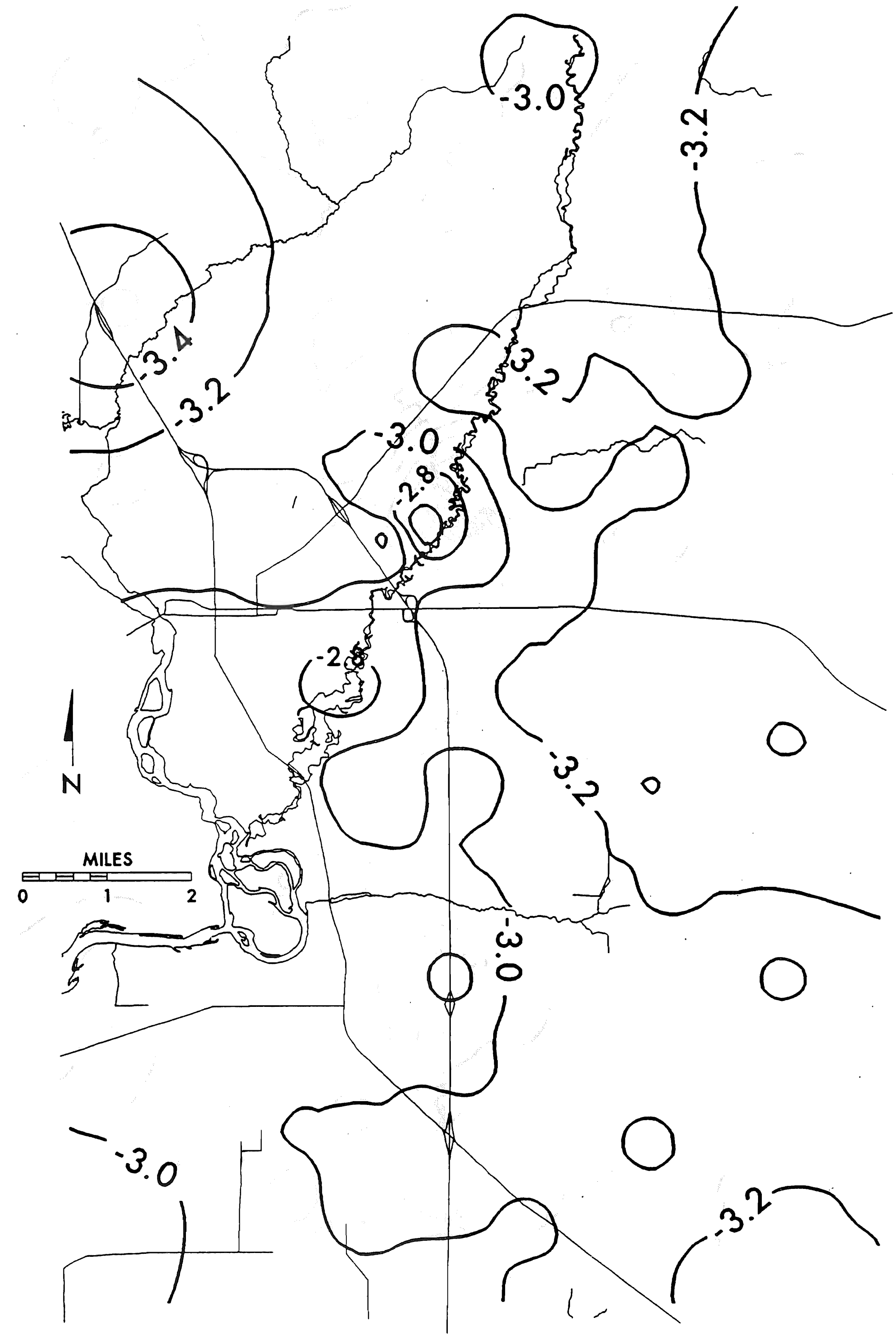

Figure 2.10B **Hydraulic conductivity (log m/s) calculated from specific capacity data for the Stevens Point, Whiting, and Plover area.**

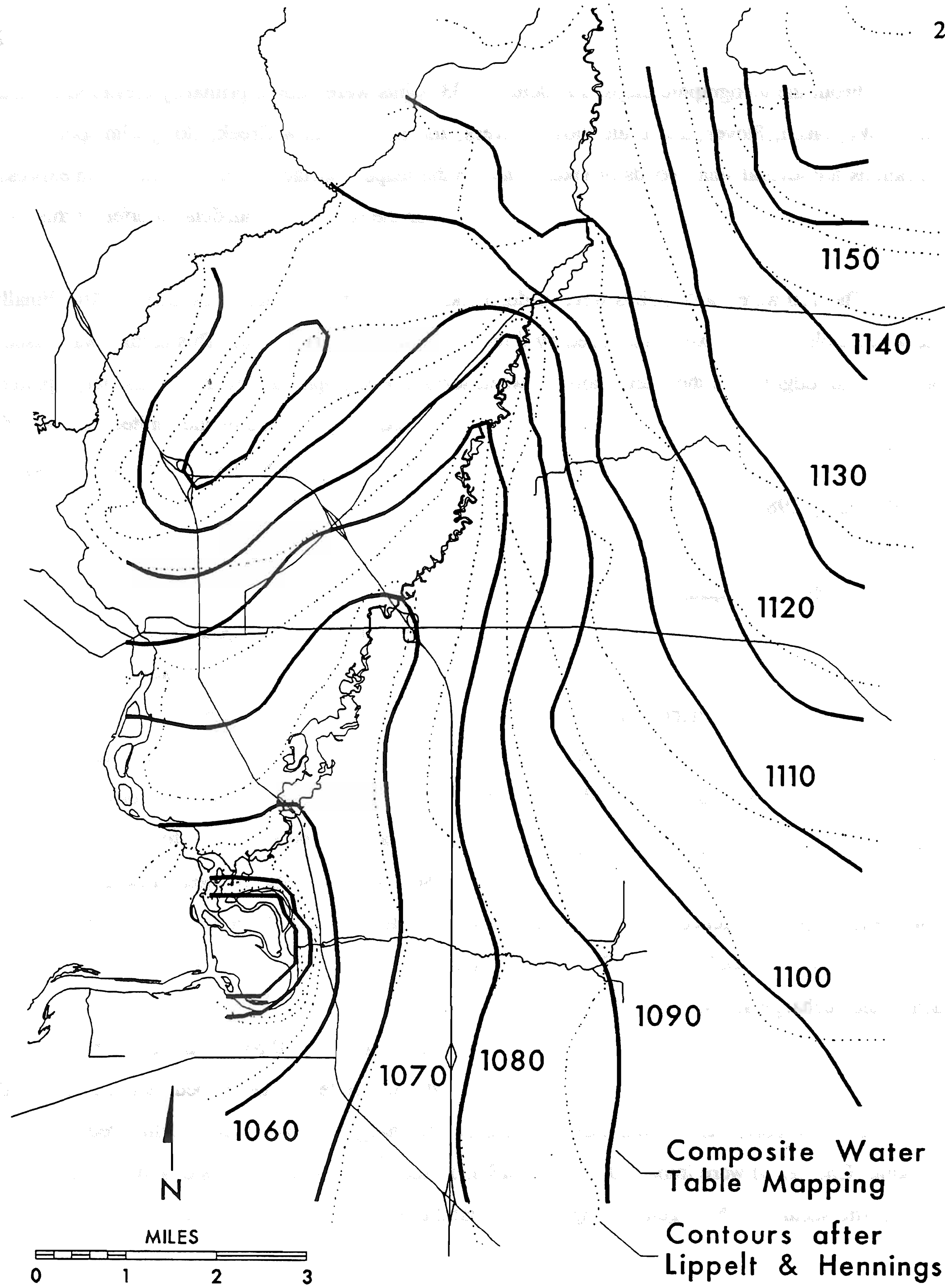

Figure 2.11 Composite water table map for the Stevens Point, Whiting, and Plover area (feet MSL).

From the topographic maps, an additional 33 points were added, primarily elevation contours on the Wisconsin, Plover, and Little Plover Rivers, and Hay Meadow Creek, along with spot elevations for several small ponds or lakes noted on the maps. Surface waters should be an expression of the groundwater table intersecting the land surface for the unconfined surficial aquifer of the SWP area.

The 110 water table points were gridded and contoured with Surfer (Golden, 1990). Finally, the water table contours were fine tuned by directly editing the Surfer grid. This editing was based on professional judgment of the interactions between surface topography and recharge-discharge patterns, especially groundwater-surface water relationships not adequately defined by the limited number of data points. This updated composite portrait of the water table is very similar to the map by Lippelt and Hennings (1981).

Land Management Mapping

Calculation of nitrogen loading rates for the municipal well recharge areas requires detailed information on the crops grown, historical crop rotations, and the nitrogen fertilization rates. To obtain the cropping information, the Portage County generalized land use mapping for the SWP area was further analyzed using road surveys, air photo interpretation, farm management plans, and farm records. This analysis was performed by the USDA SWP Wellhead Protection Hydrologic Unit Area Project staff. Effort was made to identify specific crop rotations where possible for each field, identify land use conversions such as agricultural to residential, and note changes in field boundaries relative to cropping practices. These changes were recorded on a CAD version of the GIS mapping, and then transferred back to the GIS system for land management analysis/tabulation relative to delineated recharge areas.

In addition to the general land use categories, 15 crops with 50 rotations/management programs were identified for the SWP area. A GIS attribute table was developed that characterized 1,894 individual polygons for land use/management. Recharge area boundaries delineated by particle tracking (Chapter 4) were imported into the GIS and used to extract and tabulate land management for the specific subareas. Nitrogen loading for these land uses (Chapter 5) were used to calculate concentrations at the wellheads (Chapter 6).

CHAPTER 3

GROUNDWATER FLOW MODELING

A model is an approximation of a natural phenomenon. Mathematical models represent phenomena with sets of equations, generally solved with the help of computer codes. We developed a flow model describing the groundwater flow system in the SWP area using standard modeling protocol (ASTM, 1993; Anderson and Woessner, 1992): establish the purpose, develop a conceptual model, select a model code, construct the model, perform calibration and sensitivity analysis, and run predictive simulations.

MODEL DESIGN

Purpose

The groundwater flow model was developed for use as a tool for local groundwater management and wellhead protection efforts. The model can be used alone to describe groundwater flow and to predict the effects of pumping wells on other wells and on groundwater quantity or stream discharges. When coupled with a "particle tracking model" (Chapter 4), simulated groundwater flowpaths can be traced from a point back to where they originated, or forward to where they discharge from a flow system.

The major application of models in this project is to help predict groundwater quality at the municipal wells. Citizens and decision makers need to know how to address contaminant sources, and what effects can be realistically anticipated from management strategies. To the present, considerable research and effort has been applied in the SWP area, but a reliable picture of pollution causes-and-effects remains elusive. Given the continuing demands on the groundwater resource, a sophisticated management tool is needed to provide essential information and evaluate options. The groundwater flow model fills part of this need.

Conceptual Model

Our conceptual model of area hydrogeology stipulates that only flow in the glacial outwash and moraine aquifer is significant. The aquifer is unconfined, variably thick, and its base is defined by contact with bedrock or hillslope deposits. Vertical groundwater flow is insignificant at this scale of analysis, and the aquifer can be adequately modeled as a vertically homogeneous hydrostratigraphic unit. Areal heterogeneity, however, needs to be considered.

Groundwater recharge is primarily from precipitation. Discharge is primarily to area streams and wells. Streams may also recharge groundwater if pumping wells are located nearby. East of the Plover and Wisconsin Rivers, groundwater flows generally southwest and west from a regional groundwater divide located in the moraine about 5 miles east of the municipal wells. Flow west of the Plover River is generally southeasterly from the groundwater divide with Hay Meadow Creek. Discharge to irrigation wells is treated as a reduction in net groundwater recharge in areas where irrigation is practiced.

The model boundaries (Figure 3.1) for the flow system are defined with a combination of physical and hydraulic boundaries. Physical boundaries are the Wisconsin River and Hay Meadow Creek which define the western extent. These are modeled as constant head boundaries. Hydraulic boundaries form the remaining limits of the model. The regional groundwater divide forms a no-flow boundary on the east, and flow-line no-flow boundaries form northern and southern boundaries. All external boundaries are located far enough from the area of interest that minor uncertainty does not significantly impact model results.

Model Implementation

The flow model was implemented using the USGS MODFLOW code (McDonald and Harbaugh, 1988), a finite difference numerical model. This code has been rigorously verified and is commonly used for hydrogeologic investigations. The model has the flexibility to handle the boundary conditions, spatial variability, and hydraulic stresses occurring within the SWP area. Based on the site characterization, adequate data are available for implementing the conceptual model. The model design is shown in Figure 3.1. It is one layer, steady-state, and unconfined. ModelCAD386 (Geraghty & Miller, 1993) was used as a preprocessor for the MODFLOW model. Input data files for the final model are reproduced in Appendix A.

Grid

The project area was discretized into a grid of 71 columns by 114 rows (Figure 3.1). For convenience, the grid was oriented colinear with the state plane coordinate system (north/south axes). The grid spacing is a compromise between minimizing the number of nodes relative to computational costs, and maximizing the accuracy of the nodal representation of the water table curvature and the variability in aquifer properties and stresses such as river, drain, and pumping nodes. An irregular grid was therefore used so that smaller cells could be used to define the area of pumping wells and

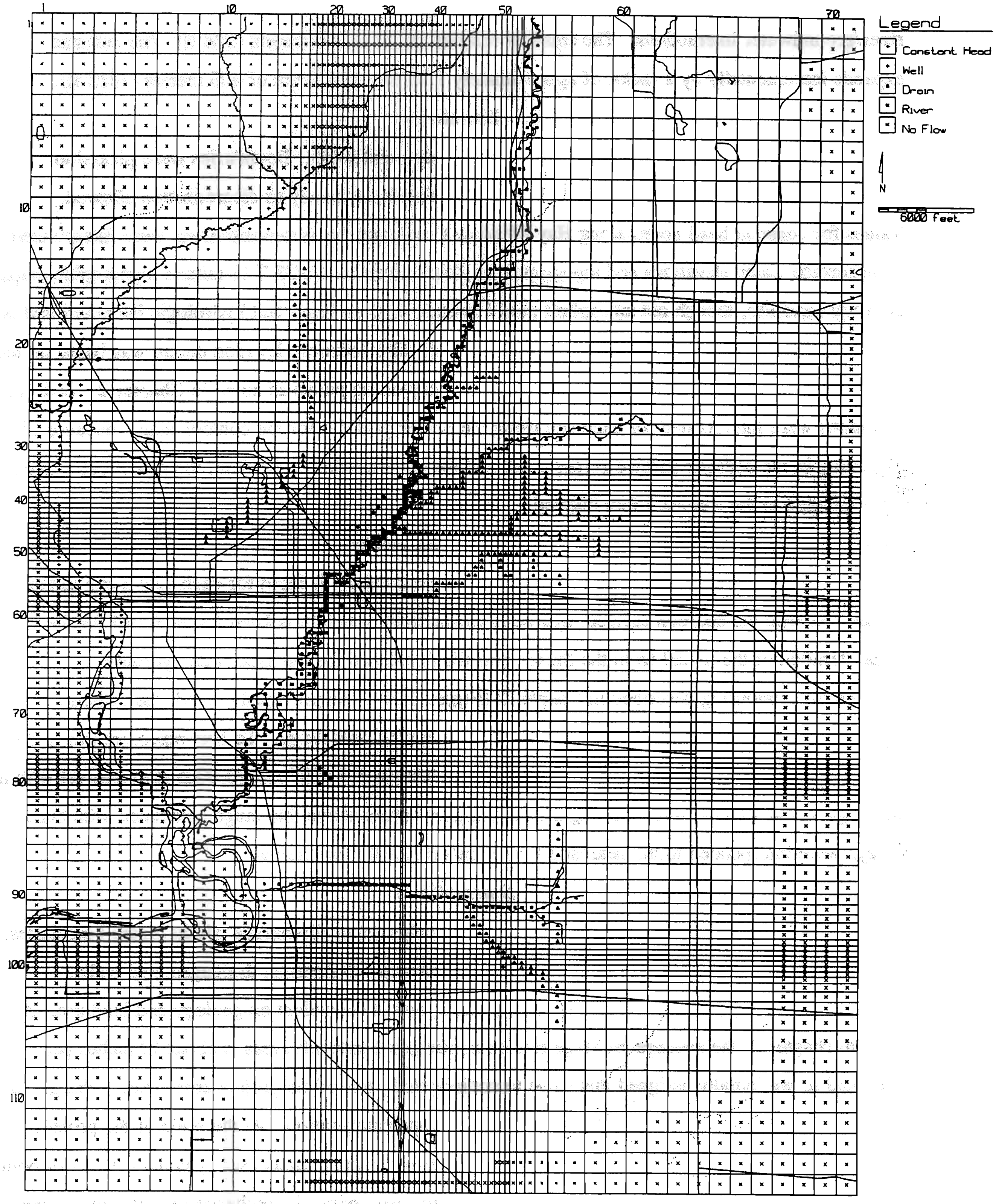

Figure 3.1 Design of the MODFLOW model for the Stevens Point, Whiting, and Plover study area.

river/groundwater interactions. The smallest cell dimension is 100 meters (328 ft). The cell size expands incrementally by a factor of approximately 1.25 to a maximum of 400 meters (1312 ft).

Boundaries

Model boundaries are no-flow on the north, east, and south. Boundaries were picked to represent the curve of the flowlines and groundwater divide as closely as discretization allowed. Values for constant head nodes along Hay Meadow Creek and the Wisconsin River were interpolated from surface water elevations and topographic contours noted on USGS 7 ½ minute topographic maps. Bedrock elevation, though not an explicit model boundary, is an important hydrologic boundary and is important in calculating aquifer thickness in the model. The bedrock elevation datum was based on the bedrock contour map (Figure 2.9) developed from 256 plotting points as noted in Chapter 2. Bedrock elevations were interpolated for each cell using the Surfer grid file for the contour map. Because of variability in the bedrock surface across a cell and the uncertainties of the data, the model cell values were rounded to the nearest 5 foot interval.

Hydraulic Conductivity

Because the aquifer is unconfined, hydraulic conductivity is input for each active cell in the model. As with the bedrock surface, hydraulic conductivity was interpolated for each cell for the initial iteration of the model from the Surfer grid file developed for the area-wide hydraulic conductivity contouring described in Chapter 2 (Figure 2.10). As previously noted, the average hydraulic conductivity for the SWP area is in the 7 to 9 $x10^{-4}$ m/s range. To best represent this in the model, individual cell values were interpolated to the nearest 1.5 $x10^{-4}$ m/s (0.0005 ft/s) interval within the range 1.5 to 15 $x10^{-4}$ m/s. The significantly higher values above this range associated with buried valleys were interpolated to the nearest 3.0 $x10^{-4}$ m/s (0.0010 ft/s) interval.

Sources/Sinks

Model sources and sinks simulate water that recharges or discharges within model boundaries. Some examples include recharge from rainfall and discharges to streams, drainages, and wells. Average net recharge varies across the model domain, and is dependent on geology and land use. A noted in chapter 2, the average recharge rate on nonirrigated outwash areas is about 10 inches per year, and so we initially assigned this value to nonirrigated parts of the project area. Net recharge on irrigated fields is about 6 inches/year, due to higher evapotranspiration. At the scale of the project, few model cells contain all irrigated fields, rather, cells in irrigated areas usually contain a mix of both irrigated and nonirrigated land uses. To reflect this mix, we assigned a recharge rate of 8 inches per year for model cells in irrigated areas.

River fluxes for the Plover River, Little Plover River, and a small portion of Lost Creek were modeled as variable sources/sinks using the MODFLOW river package. This package allows the river cell or node to be a sink if flow is toward the river (gaining stream) or a source if flow is out of the river (losing stream). River fluxes are calculated by the model using a riverbed conductance term and the driving difference in head between the river stage and the calculated groundwater elevation in the cell. However, if the groundwater elevation drops sufficiently below the riverbed, the conductance term becomes limiting and further drops in the calculated groundwater elevation do not influence the flux. This riverbed bottom elevation was taken as one foot below the actual riverbed elevation as interpolated from USGS 7 ½ minute topo maps. The Plover River stage was defined as the reservoir stage or assumed as two feet above the interpolated riverbed. The Little Plover and Lost Creek stage was defined as one foot above the interpolated riverbed elevation. The riverbed conductance term is a function of the riverbed vertical hydraulic conductivity and the riverbed geometry within the cell. A value of 0.42×10^{-4} m/s was considered a reasonable estimate of the riverbed hydraulic conductivity based on the horizontal hydraulic conductivity and the limited data available on vertical conductivity as noted in Chapter 2. A rough estimate of riverbed geometry was made for each cell based on the length of river within the cell.

Additional sinks that need to be considered are the numerous drainage ditches and intermittent streams in the active portion of the model area. These drainages are conduits that can receive significant amounts of groundwater if the water table rises above the drain elevation. These drains are different from river nodes in that they are considered dry and do not contribute or lose water to recharge groundwater when the water table is below the drain elevation. Input for the MODFLOW drain package was determined similar to the river package. Drain elevation was defined as the elevation interpolated from topo maps, and drainbed conductance was defined as for rivers. River and drain cells are indicated on Figure 3.1.

Our calibration excluded large scale municipal and industrial wellfield pumpage. Wellfield pumpage was not high or consistent through much of the 1945 to 1990 time period represented by the composite water table data. A post-modeling analysis indicated such pumpage would have little effect on heads near target data points.

Solver Package

Several solution packages are available in the MODFLOW code. These solvers use iterative techniques which make successive approximations of the head distribution as the solution converges on an error criterion. That is, model convergence is defined as when the specified error criterion is greater than the largest change in calculated head at each active node between successive iterations.

We used the solver Preconditioned Conjugate-Gradient 2 (PCG2; Hill, 1990) with an error criterion of 0.001 ft for maximum change in hydraulic head and 0.001 ft^3 for maximum residual for the iteration. This allows a reasonable interpretation of the calculated heads to the 0.1 ft level. The Strongly Implicit Procedure and Slice-Successive Overrelaxation solver packages by McDonald and Harbaugh (1988) would not converge at an acceptable error criterion.

MODEL CALIBRATION

Calibration is the process of refining the model representation of the hydrogeologic framework, hydraulic properties, and boundary conditions to achieve a desired match between model outputs and field observations (ASTM, 1993). Model calibrations usually attempt to match modeled and field-measured hydraulic heads and stream flows. Model inputs and stresses are then adjusted until the model reproduces the field observations with acceptable error.

Calibration Targets

Ideally, the calibration values for a steady state model would consist of specific points or periods of time in which the system is at equilibrium. Such a data set is almost never available for modeling purposes, and this is true for the SWP area. As noted in Chapter 2, considerable real world transient data is available from well logs and well and streamflow monitoring, but these data represent the dynamic response of the flow system to continually changing stresses.

The composite water table contour map described in Chapter 2 (Figure 2.11) was used as a qualitative calibration target, matching contoured model head values by eye. Two sets of quantitative head targets were also used. The well log database and topographic maps provided 347 usable points of "observed" water table information for active cells. Forty-nine cells had two points that were averaged, yielding 298 cell values (Figure 3.2). A second set was developed by interpolating the Surfer grid file for the water table mapping to each of 6795 active cells. Potential errors in the calibration targets include the effects of measurement/reporting methods, transient effects (climatic, well pumpage, etc) from a long time span of measurements, interpolation errors, nonuniform distribution or representation, target locations not at node, and rounding/smoothing effects.

In addition to hydraulic heads, calibration to streamflow fluxes was also considered. The net loss or gain in streamflow predicted by the model can be compared to measured changes in surface

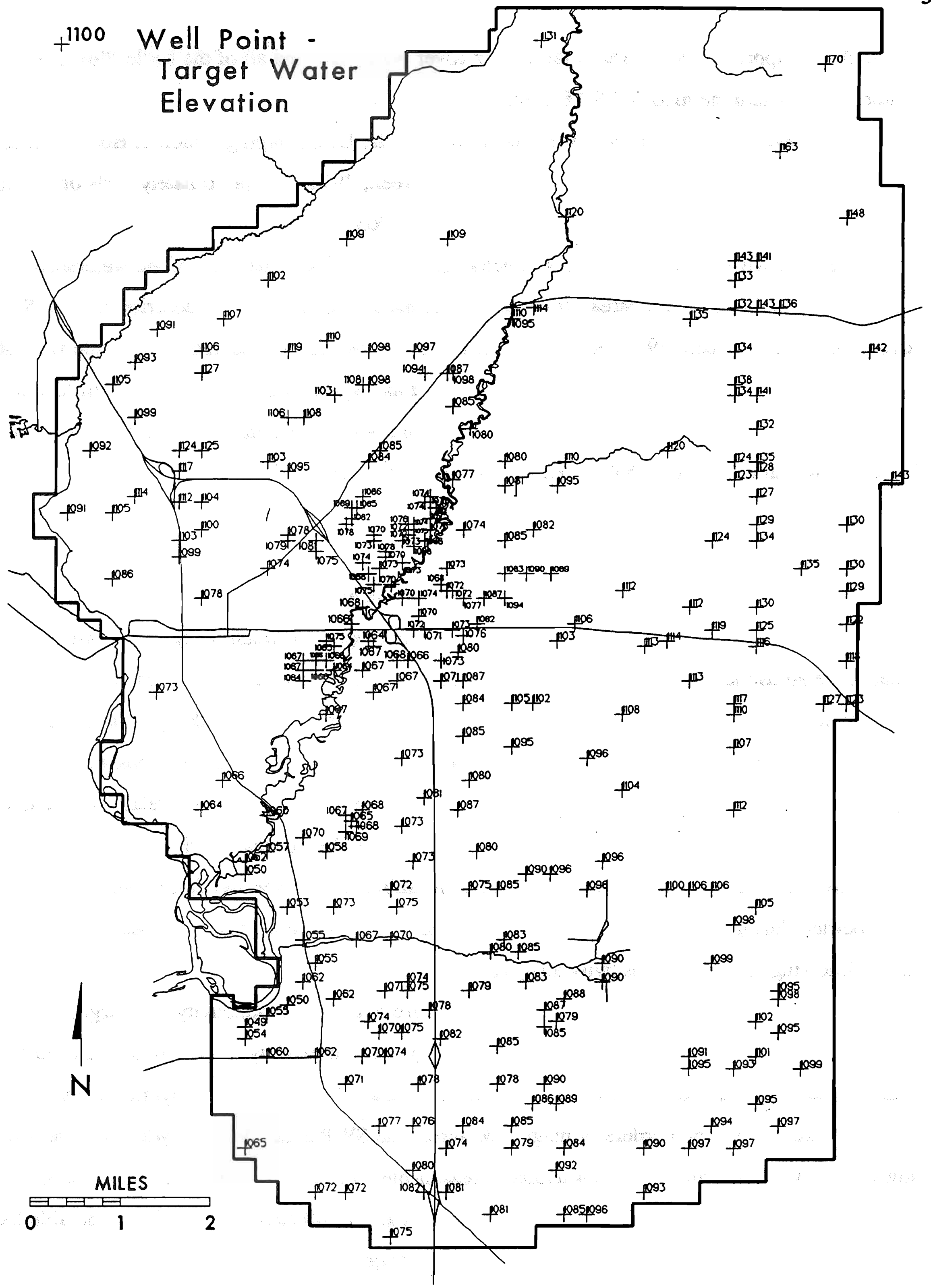

Figure 3.2 Point calibration targets for the Stevens Point, Whiting, and Plover MODFLOW model.

water flow. Approximately 33% of the Plover River watershed and all of the Little Plover River watershed is within the modeled SWP area.

The average flow for the Little Plover River at the USGS gaging station at Hoover Road (1.2 miles upstream from mouth) is 9.6 cfs (Devaul and Green, 1971). Approximately 90% of the flow, or 8.6 cfs, is attributable to groundwater baseflow (Holt, 1965).

Plover River fluxes can be less directly analyzed because a majority of the watershed is outside of the SWP area, and streamflow measurements are not available to describe just the SWP area. Devaul and Green (1971) noted an average flow of 143 cfs for the 136 square mile watershed, or approximately 1.05 cfs/sq mile. Holt (1965) noted that approximately 75% of streamflow is from groundwater, or 0.79 cfs/sq mile. Using this figure, the 44 square miles of watershed above McDill Pond and within the modeled SWP area would contribute 34.8 cfs.

Calibration Process and Results

Calibration strategy

Our calibration strategy was to first adjust model inputs in which we had the least confidence; when these adjustments had to be unrealistic in order to achieve a successful match, we then adjusted the next input in which we had the least confidence. We sought to avoid making adjustments in a patchwork fashion when a larger scale change could be performed. We did not consider modifying boundary conditions as part of the calibration process, since the boundaries are relatively reliable and are located remote from the principal areas of interest. We also considered river and drain node elevations to be reliable and did not modify them in calibration. River and drain conductances were not modified during the calibration because the model is insensitive to these parameters (as discussed in the following section on sensitivity analysis).

Parameters adjusted to achieve calibration were hydraulic conductivity, recharge, and bedrock elevation. Hydraulic conductivity has the highest degree of uncertainty since it was estimated from sparse pump test data and unverified specific capacity data. The data suggest hydraulic conductivity spans a range of up to two orders of magnitude across the SWP area. Where hydraulic conductivity adjustments were insufficient or unwarranted, reasonable adjustments in recharge were made. While inverse modeling studies (WGNHS, 1986; Bradbury et al., 1992) have suggested considerable local variability in recharge is possible, there is little support for drastic changes in the assumed recharge rates within the SWP area for calibration purposes.

Bedrock elevations were judged to be generally more reliable than recharge and hydraulic conductivity, and were therefore adjusted only to correct for interpolation inaccuracies, or to address conditions in the northwest where groundwater flow may occur through bedrock residuum. Interpolation inaccuracies in the original model input occurred in some cells in the Plover River and Hay Meadow Creek. Some cells in these areas initially had bedrock elevations slightly higher than river stage or constant head values. In this instance, bedrock elevations for affected cells were adjusted downward by 5 to 10 feet to eliminate the problem.

Calibration process

When the model was initially run, the overall match with the calibration targets was fair. Problems were encountered in the northwest where cells went dry; in the extreme northeast where output heads were too low; south of the Little Plover River and along the eastern groundwater divide where output heads were too high; and in the Little Plover River, where baseflow was low.

Numerous trial and error model runs were made to calibrate the model. To lower output heads in the east and south, the hydraulic conductivity was increased by 0.001 to 0.002 ft/s, possibly reflecting an initial underestimation of hydraulic conductivity in these permeable outwash areas. To raise output heads in the extreme northeast, an area of complex surficial geology and scattered organic deposits (Figure 2.3), hydraulic conductivity was decreased.

The cell de-watering in the northwest appeared to be related to several causes. The northwest is different than most of the SWP area. Sand and gravel materials are thin and may have low permeability, groundwater flow through the bedrock residuum may be relatively important, and significant surface runoff may occur at the expense of groundwater recharge. In the initial run, the input aquifer base as interpreted from the bottom of the sand and gravel aquifer resulted in numerous cells with a target water elevation below the bottom of the aquifer by up to 21 feet, a situation which is physically impossible. To address these issues, the bedrock in the northwest was uniformly lowered with smooth transitions. This resulted in bedrock elevations being adjusted by an average of -19 feet, with some as much as -40 feet. At the same time, hydraulic conductivity was changed by -0.001 to -0.005 ft/s to reflect less permeable sand and gravel as well as potential flow through residuum. (As noted in Chapter 2, hydraulic conductivity determined by slug tests for this type of material was nearly 2 orders of magnitude less than the average hydraulic conductivity for sand and gravel outwash.) In addition, we changed recharge rates by -2 to -4 inches to reflect the tighter soils and higher surface runoff. We cannot say for certain how well this representation in the northwest reflects reality because of an absence of field data, however, sensitivity analyses indicate effects on zone-of-contribution delineations would be minimal.

In the immediate area of the Little Plover River, we increased recharge rates to bring up the discharge rate. This had a negligible effect on hydraulic heads.

Calibrated model

The final calibrated hydraulic conductivity distribution is given in Figure 3.3. Changes from the initial distribution based on hydraulic conductivity mapping from specific capacity data are shown in Figures 3.4 A,B. These changes were well within the expected range across the SWP area, with the largest reductions apparently related to non-outwash materials. The changes represent a -97.5% to +167% range, with an average change of -4%. For cells adjusted downward, the average change was -44%. For cells adjusted upward, the average change was +43%. The final distribution minimum is $1x10^{-4}$ ft/s and maximum is $1.4x10^{-2}$, or approximately 2 orders of magnitude. A good match with head targets was achieved with these adjustments, except for the northwest area where high bedrock cells continued to go dry during the simulation without bedrock adjustments.

The final recharge distribution for the model is given in Figure 3.5. As previously noted, the average recharge rate for the area is 10 inches/year. Deviations from 10 inches that were incorporated into the model are shown in Figures 3.6 A,B. The initial conditions included a reduction to 8 inches/year for areas dominated by irrigated agriculture. The recharge deviations from 10 inches/year represent a range of -80% to +20% change, with an average change of -30% or -3 inches.

The bedrock adjustments that were incorporated into the model are shown in Figure 3.7. Adjustments were limited to the high bedrock area in the northwest.

Modeled heads compare favorably with the target contour map (Figure 3.8). Residuals (the difference between known heads and the model simulated heads) for the two target groups are given in Table 3.1. The mean residual is very low for both sets, although large negative and positive residuals might be hidden in this statistic. The mean absolute value of the residuals is higher, but small relative to the overall change in heads across the active nodes. The root mean square (RMS) of the residuals is defined as the square root of the average of the squared residuals, and is generally considered the preferred measure of the model fit to calibration data. The RMS for the two calibration target sets is considered adequately small, especially relative to the overall change in heads.

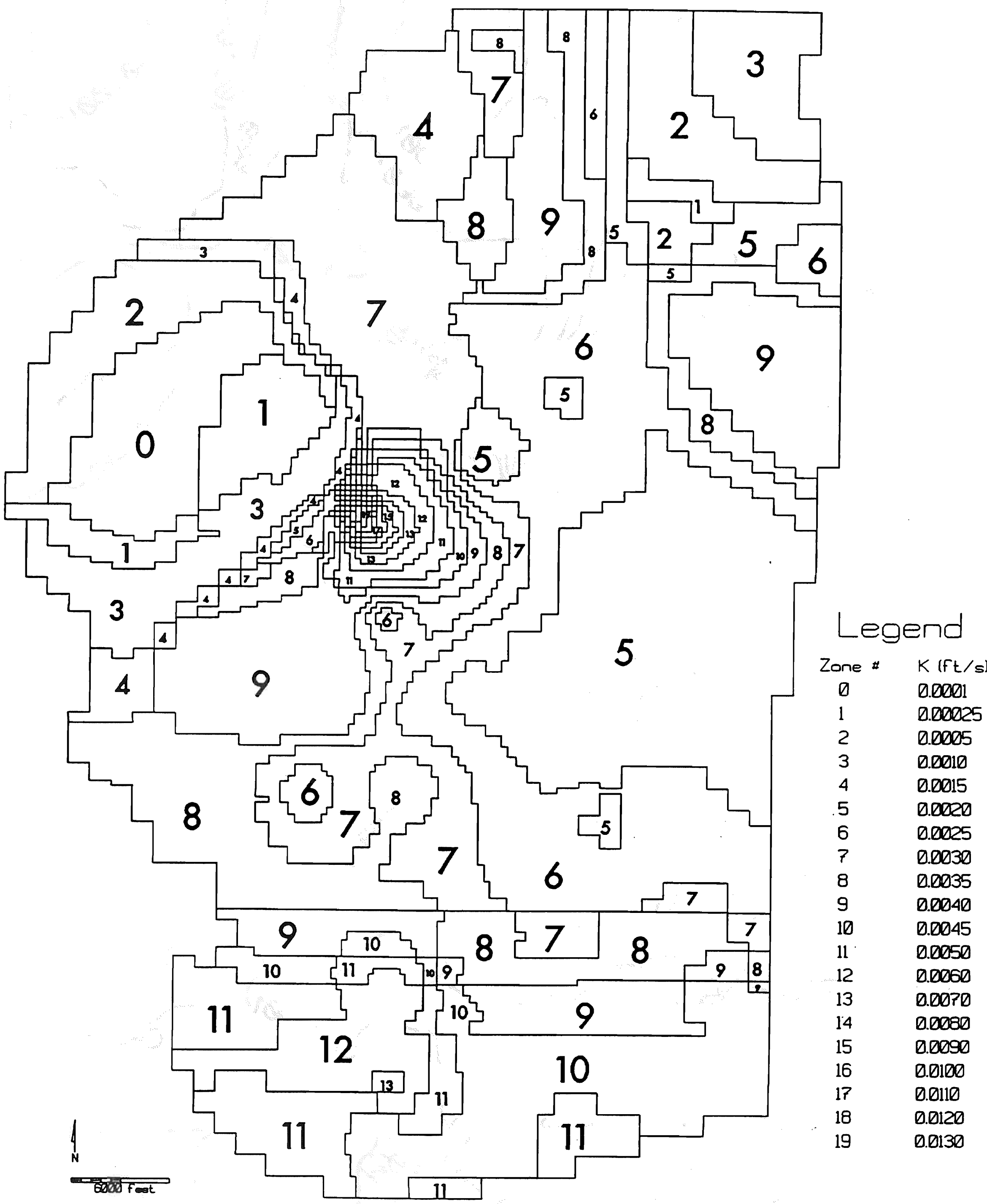

Figure 3.3 Hydraulic conductivity zones for the calibrated Stevens Point, Whiting, and Plover model.

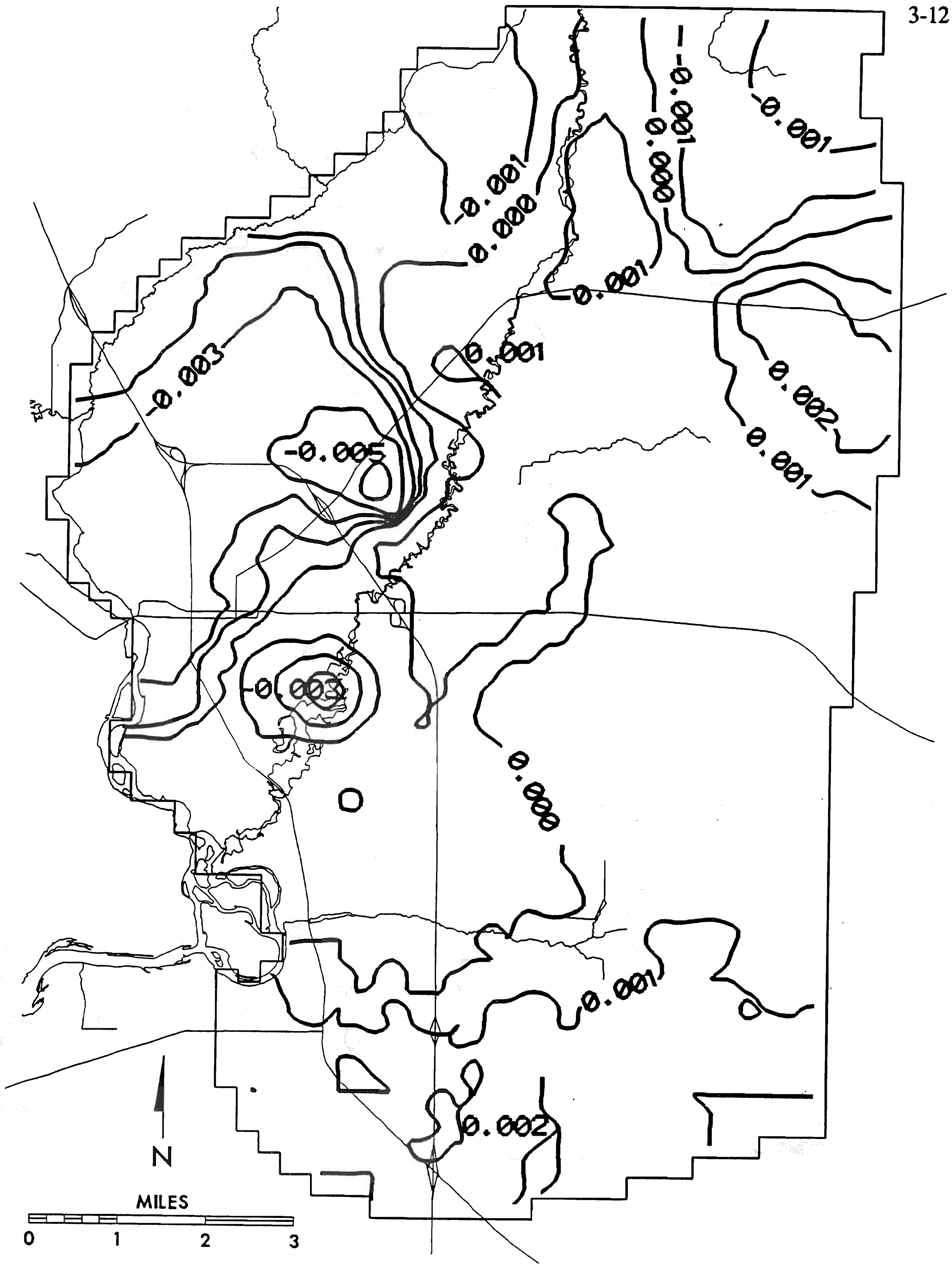

Figure 3.4A Changes in hydraulic conductivity (ft/s) for calibration of the Stevens Point, Whiting, and Plover model.

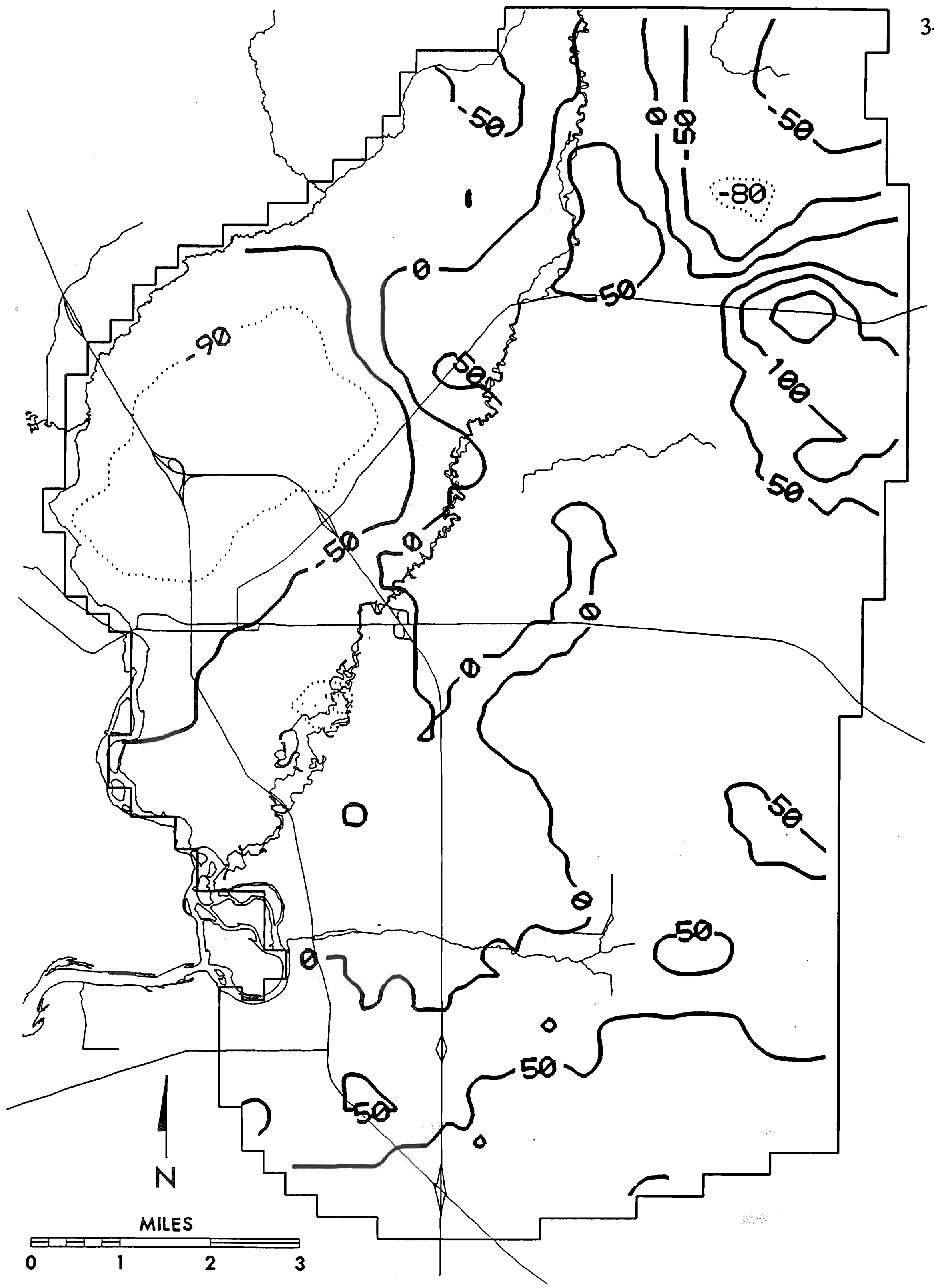

Figure 3.4B Changes in hydraulic conductivity (percent) for calibration of the Stevens Point, Whiting, and Plover model.

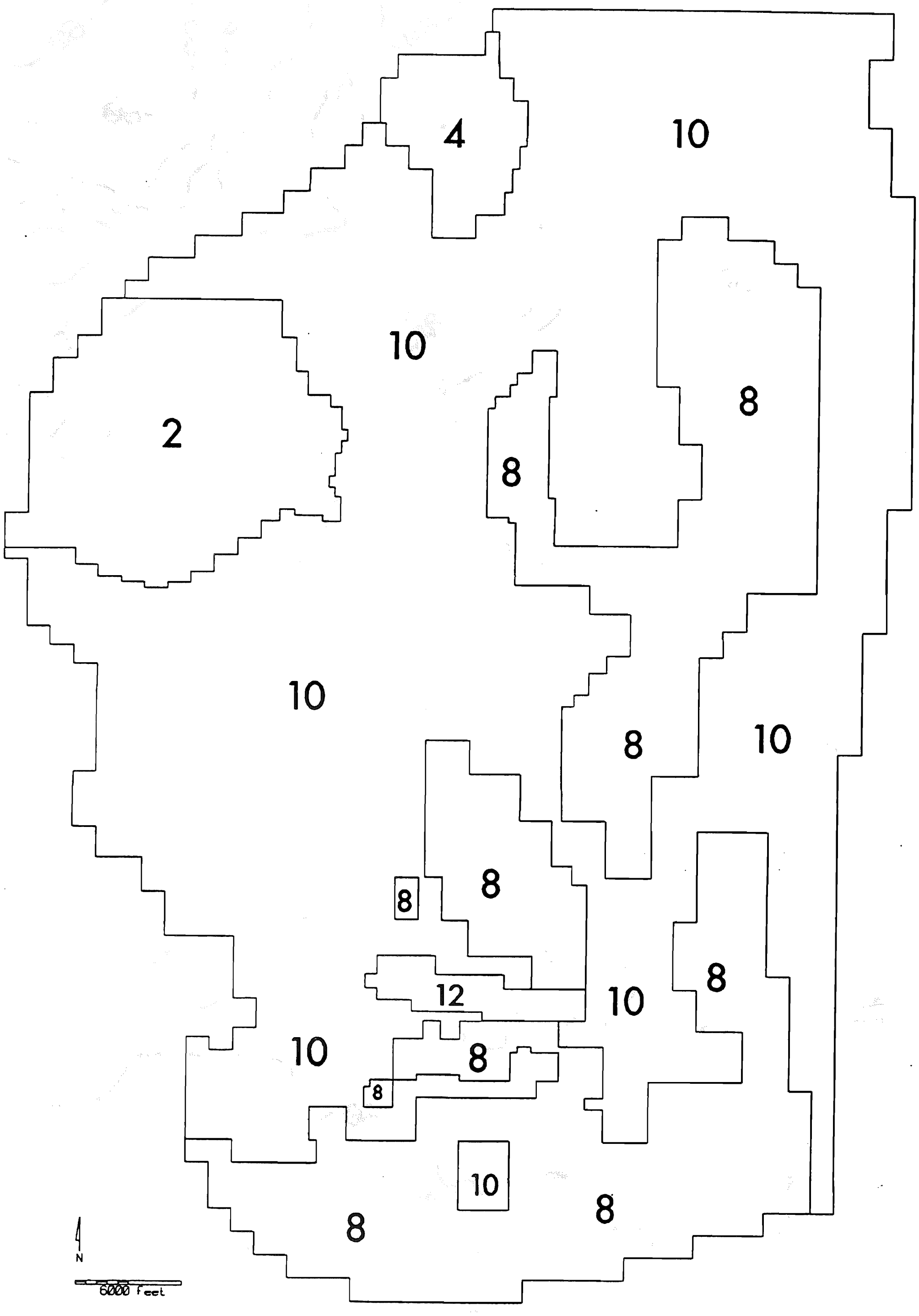

Figure 3.5 Recharge zones for the calibrated Stevens Point, Whiting, and Plover model (inches/year).

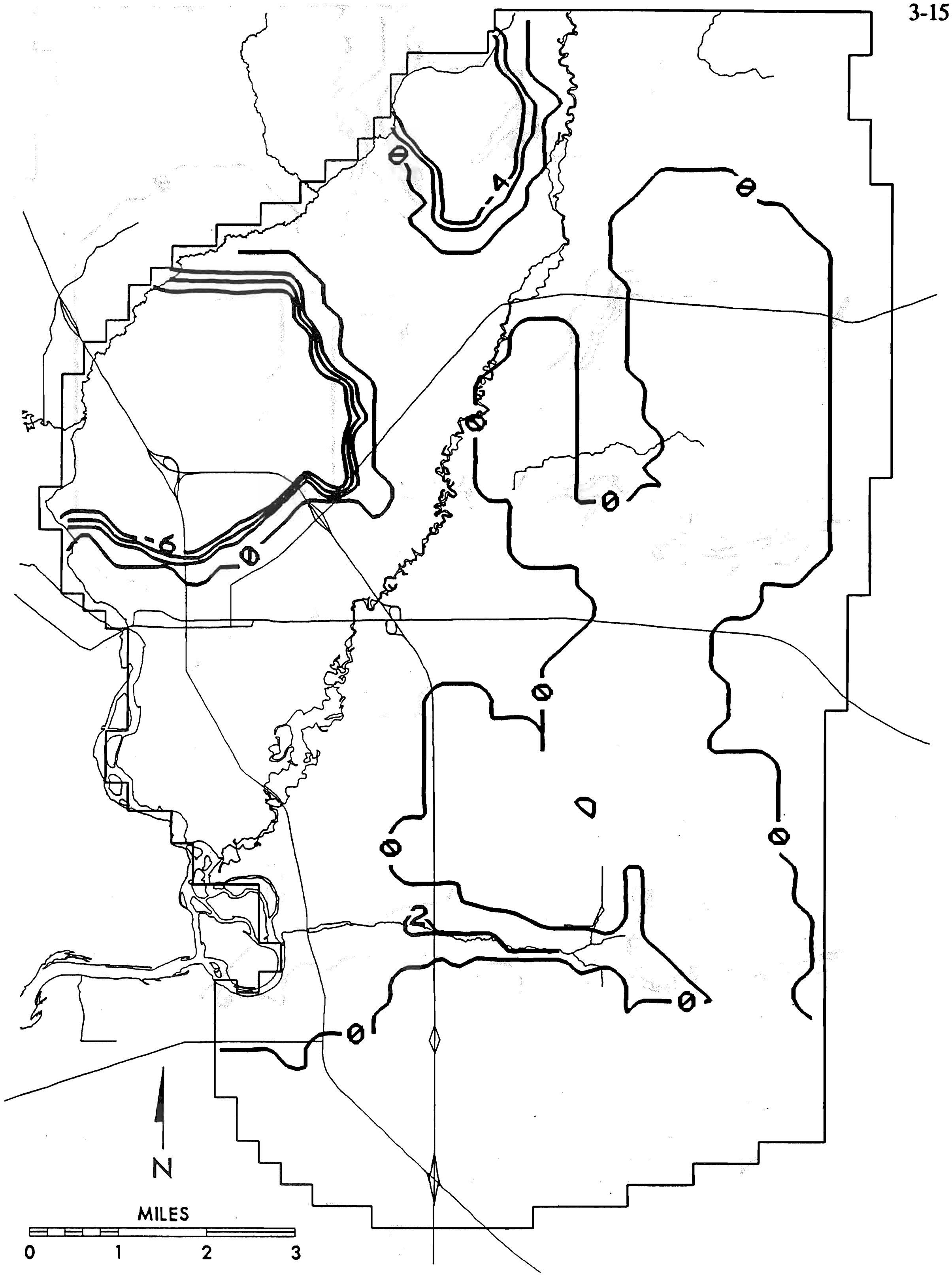

Figure 3.6A Changes in recharge (inches) from 10 inches/year for calibration of the Stevens Point, Whiting, and Plover model.

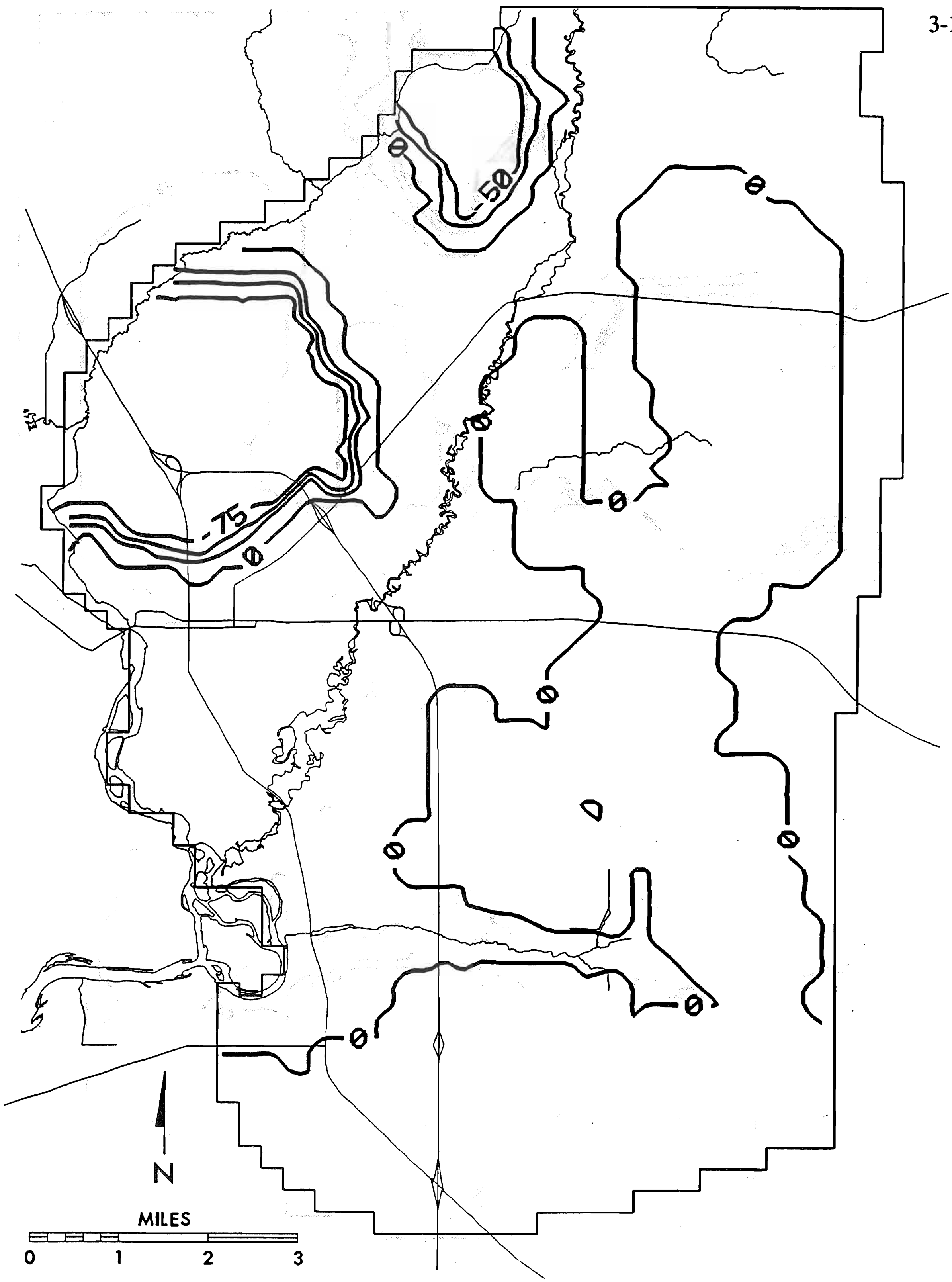

Figure 3.6B Changes in recharge (percent) from 10 inches/year for calibration of the Stevens Point, Whiting, and Plover model.

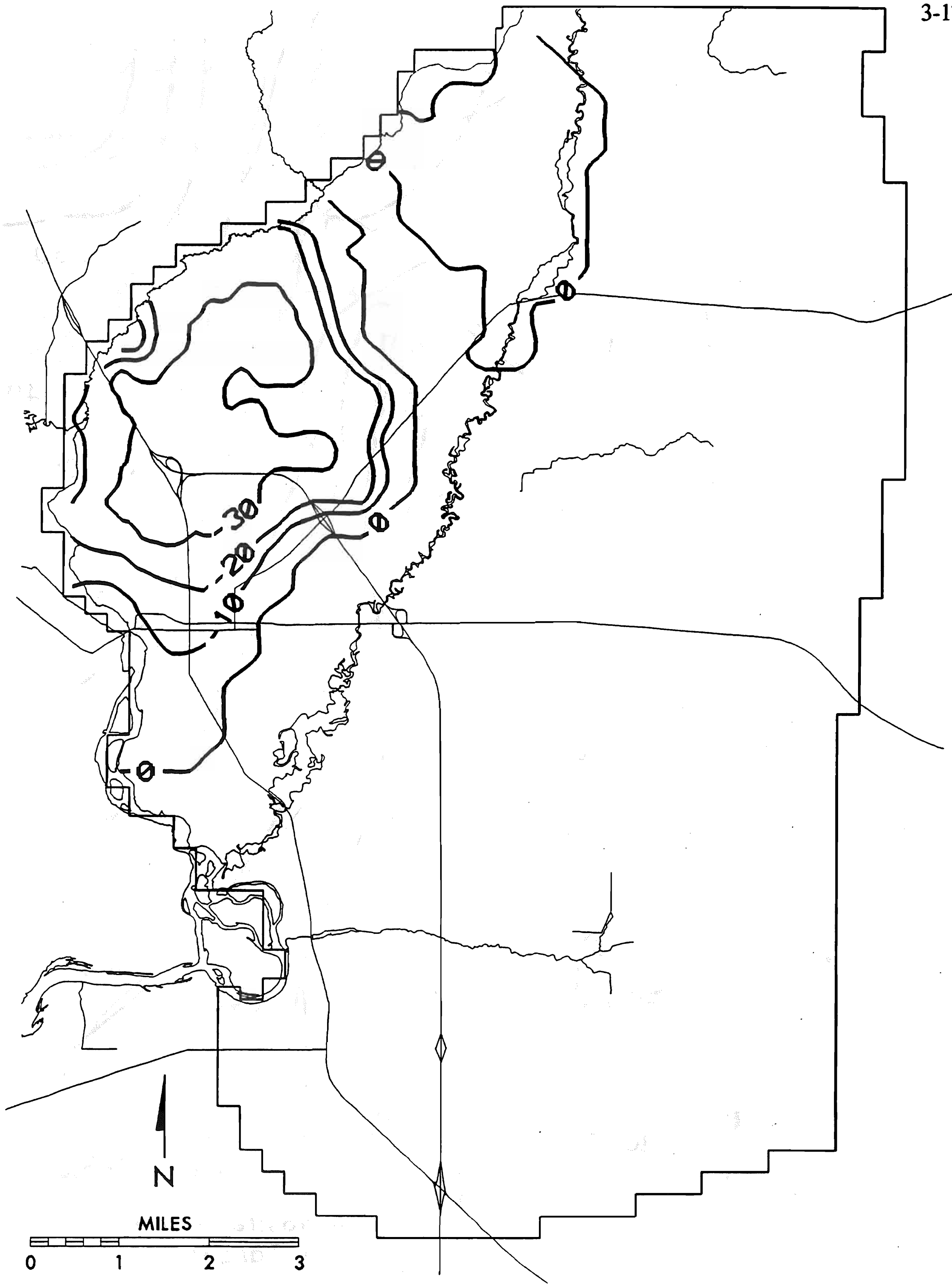

Figure 3.7 Adjustments to bedrock elevation (feet) for calibration of the Stevens Point, Whiting, and Plover model.

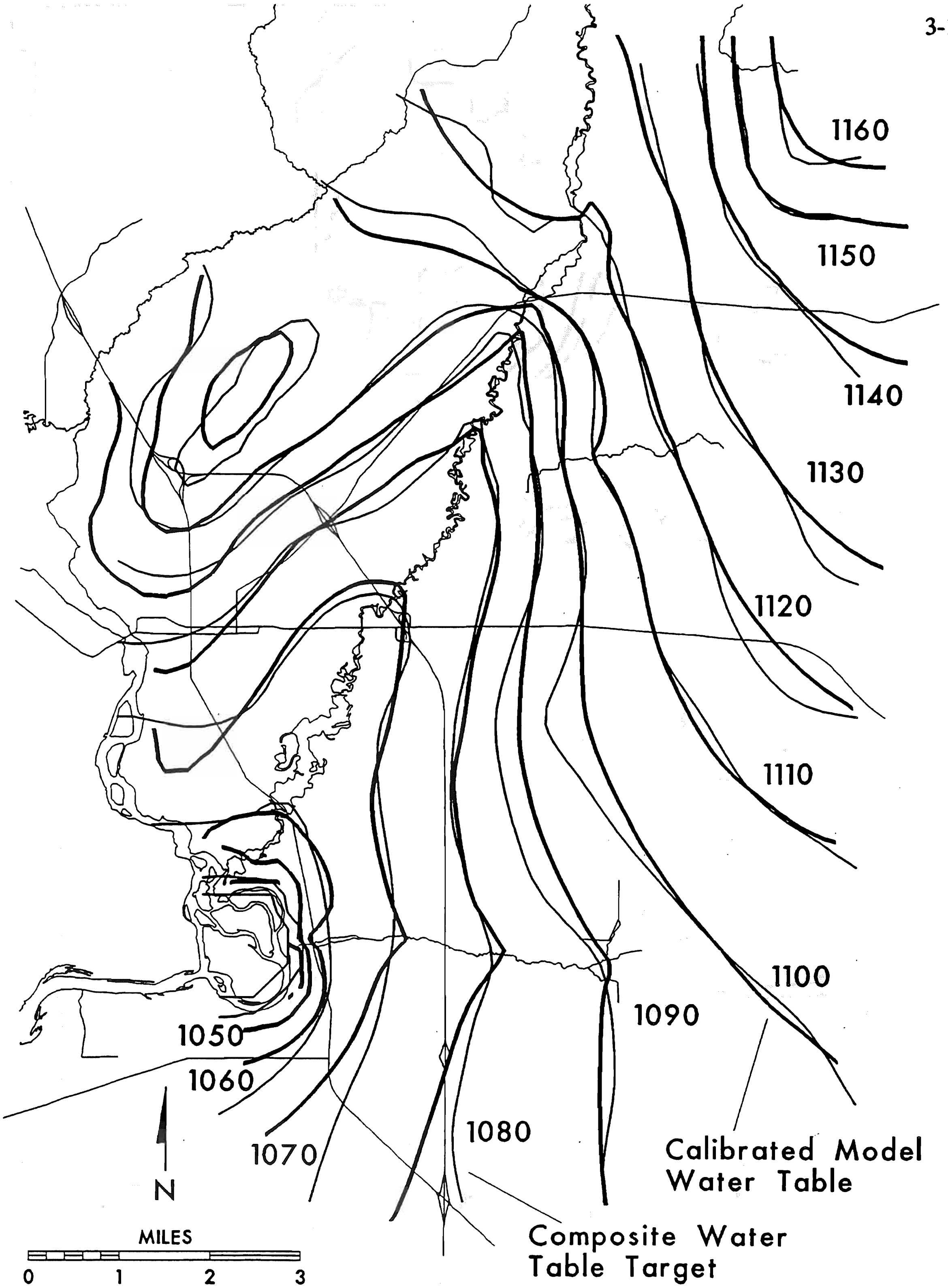

Figure 3.8 Water table contours (feet MSL) for the calibrated Stevens Point, Whiting, and Plover model and the composite target.

Table 3.1. Residuals (feet) for the calibrated SWP model.

	Point Targets	Active Cell Targets
Number of Targets	298	6795
Minimum Residual	-11.75	-10.76
Maximum Residual	18.56	12.02
Residual Std Deviation	4.54	1.97
Mean Residual	-0.02	-0.20
Mean Absolute Residual	3.41 (2.8%)	1.54 (1.3%)
Root Mean Square (RMS)	4.54 (3.7%)	1.98 (1.6%)

(%)=percent of total change in head across model

Higher residuals for the point targets are probably due to error from the target point locations within the model cells not coinciding with the cell node where the model head is calculated, and the transient nature of the point measurements spread over several years. The active cell targets were interpolated for the model node locations and represent transient data that was averaged and smoothed to generalize the measured head distribution over the period of record.

Table 3.2 is a summary of the calibration level achieved for the active cell target set. Ninety-five percent of the active cells were within $\pm$ 4 feet of the measured values. Less than 0.3% were more than $\pm$ 8 feet. Figure 3.9 is a contour map of the residuals, and Figure 3.10 is a plot of the measured heads versus the model derived heads . The largest residuals are in areas where the head changes rapidly, especially relative to cell size. The largest residuals are localized in three areas, in the north central area where the Plover River gradient drops rapidly below Jordan Dam, in the southwest where there is a steep gradient below the Springville Pond and McDill Pond dams to the Wisconsin River, and along the Wisconsin River Flowage. Errors in these areas should not impact the flow analysis for the municipal wellfields. Future generations of the model would benefit from smaller cell sizes in these areas.

Table 3.2. Calibration levels for active cells in the SWP model.

	$\pm$2 ft	$\pm$4 ft	$\pm$6 ft	$\pm$8 ft	$\pm$10 ft	$\pm$12 ft	$\pm$14 ft
Number of Cells	4922	1569	259	28	14	2	1
Percent of Cells	72.4%	23.1%	3.8%	0.4%	0.2%	0.03%	0.01%

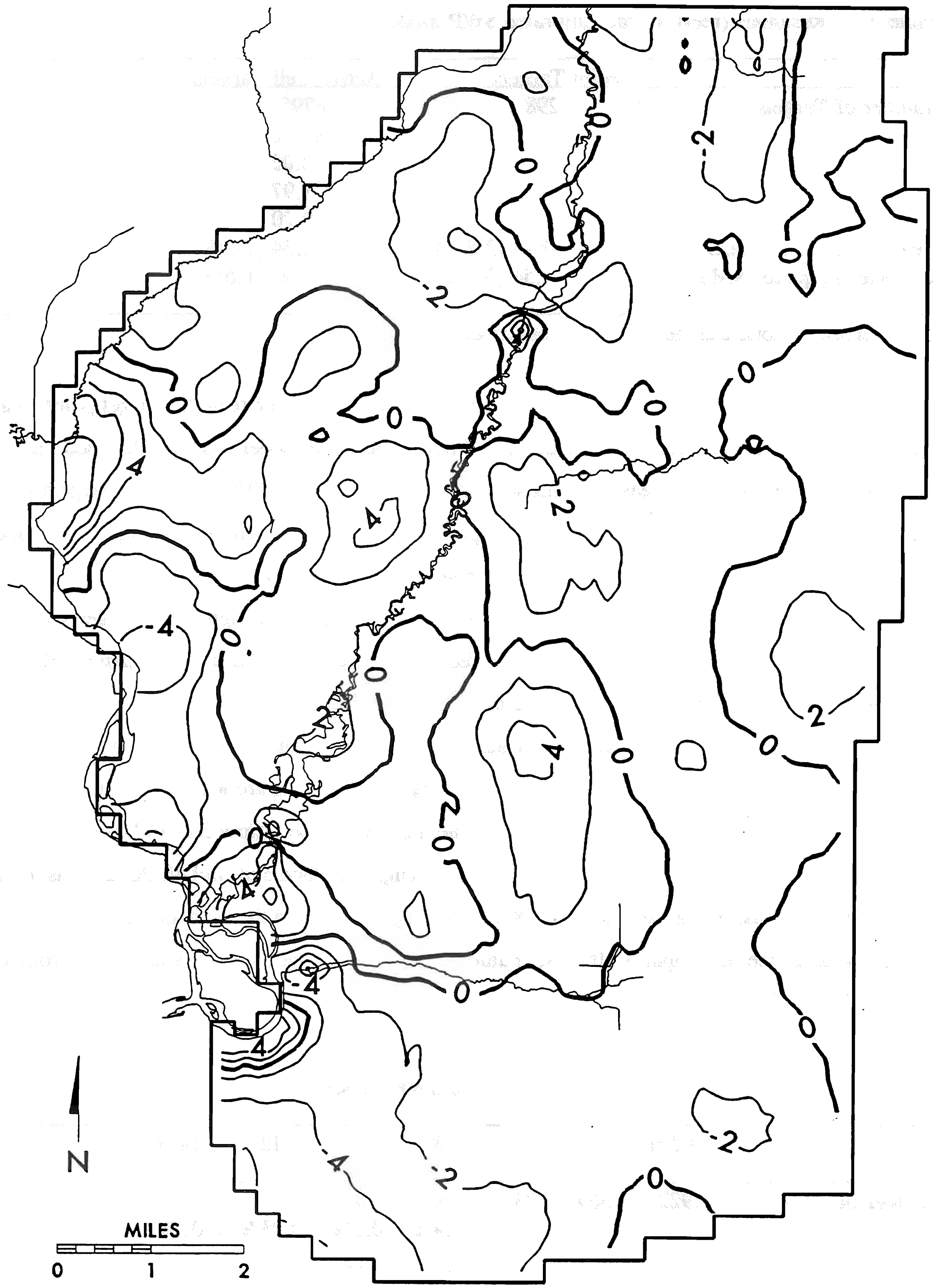

Figure 3.9 Head residuals (measured heads-simulated heads in feet) for the calibrated Stevens Point, Whiting, and Plover model.

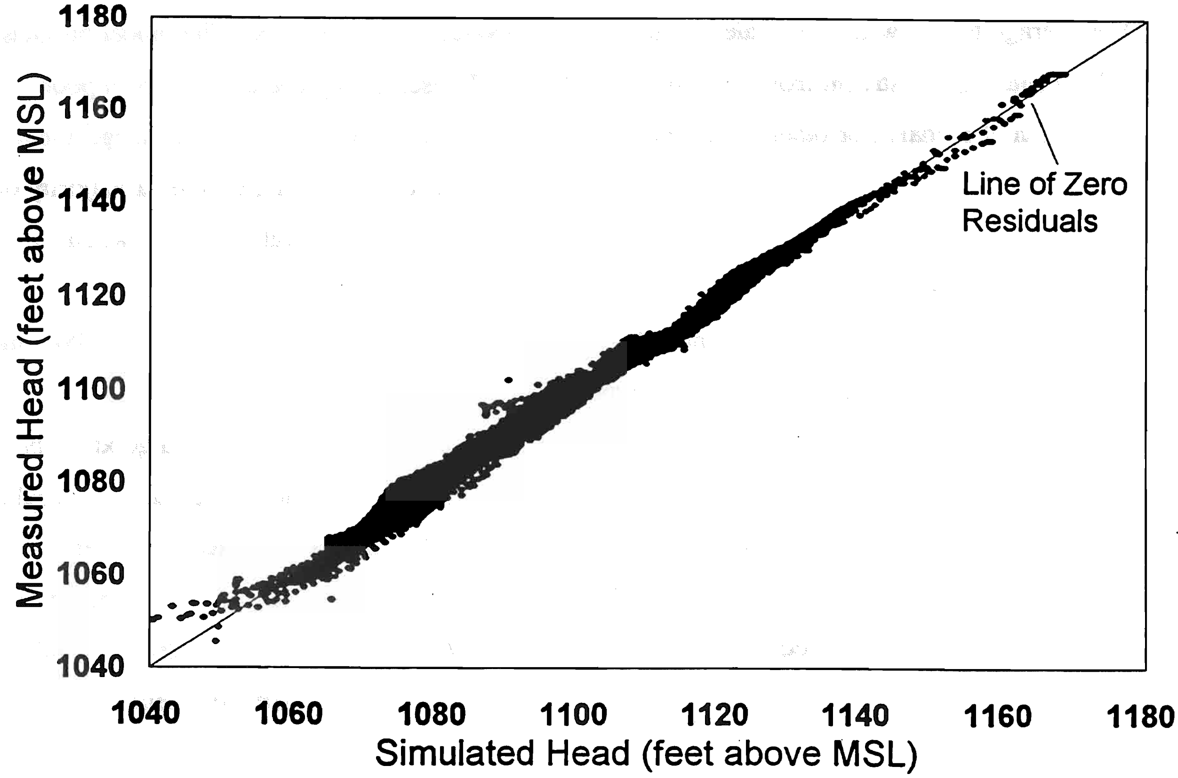

Figure 3.10 Measured (target) heads versus simulated heads for the calibrated Stevens Point, Whiting, and Plover model.

Figures 3.9 and 3.10 also suggests that the residuals are normally distributed. Residuals should not be all positive or all negative, and should not be spatially biased. For example, residual statistics may be excellent, but a model that has all positive residuals grouped in one area and negative in another would not be a good calibration.

As previously noted, model calibration can also be measured in terms of streamflow fluxes. The average baseflow for the Little Plover River at Hoover Road is 8.6 cfs. The model predicts only 6.8 cfs entering the stream from groundwater. This 21% discrepancy could result from poor calibration of recharge or other model parameters in the Little Plover area, or may suggest that the period of gaging records is not consistent with the target water table. For example, a majority of well construction and associated water level measurements may relate to droughty periods when stream flow would be lower.

The Plover River has an estimated net gain of approximately 34.8 cfs within the SWP model area. The model predicts 33.4 cfs entering the stream in this reach.

Although the parameter adjustments and final inputs noted above produce a good calibration and represent our best judgment based on available data, other combinations of parameters could also be used to calibrate the model. Appendix C contains three alternate calibrations and the effect these models would have on zone-of-contribution delineations (Chapter 4). Alternate inputs include an enhancement of the buried bedrock valley as a continuous feature, an alternate hydraulic conductivity distribution based on adjustments to a single average value, and an alternate treatment of the bedrock adjustment in the northwest.

SENSITIVITY ANALYSIS

Sensitivity analysis is used to quantify the impact of input parameter variation on the model results. This analysis helps quantify the uncertainties in the model, identify the parameters that most strongly influence the model, and provides an indication of the level of confidence that can be placed in the model results.

The sensitivity analysis for the SWP model included hydraulic conductivity, recharge, bedrock elevations, and the river/drain conductances. The process involves a systematic variation of the parameter while noting the impact on predicted heads. Parameter variation is expressed as a % change from the calibrated model value, and the impact on heads is expressed as the mean of the absolute value of the differences between the calibrated model heads and the heads calculated with parameter variations. The results of the sensitivity analysis for the SWP model is given in Figure 3.11.

The parameters were varied within the ranges experienced during the calibration process. Within these ranges, the average head changes were generally less than 6 feet, or approximately 5% or less of the total head change across the modeled area. The exception is the large head change relative to decreases of more than 50% in hydraulic conductivity. Because hydraulic conductivity adjustments in this range were related to high bedrock areas and less permeable bedrock residuum, including these low hydraulic conductivities should not significantly reduce confidence in the model results.

Future generations of the model would appear to benefit most from improved hydraulic conductivity data, and better definition of the hydrogeologic conditions in high bedrock areas. While the model is sensitive to the recharge and bedrock parameters, less model uncertainty appears to be associated with these parameters within the modeled ranges. The head calculations appear relatively insensitive to the river and drain conductances, although future modeling efforts targeting river fluxes will require accurate conductances.

MODEL VERIFICATION

Other combinations of parameters are possible that could produce an equally good match to field data. Verification is used to test the model calibration by applying it to different sets of stresses for which head/flow measurements are also available. The model results are then compared to the field measurements. A good comparison suggests that the calibrated model reasonably represents the actual dynamics of the aquifer system, and is reliable for predictive analysis. Unfortunately, there is insufficient data to verify the calibrated SWP model in this traditional sense. But, based on the sensitivity analysis and the strength of the calibration using model parameters within the expected ranges, the model is deemed adequate for regional scale predictive simulations with various point sinks representing municipal wells.

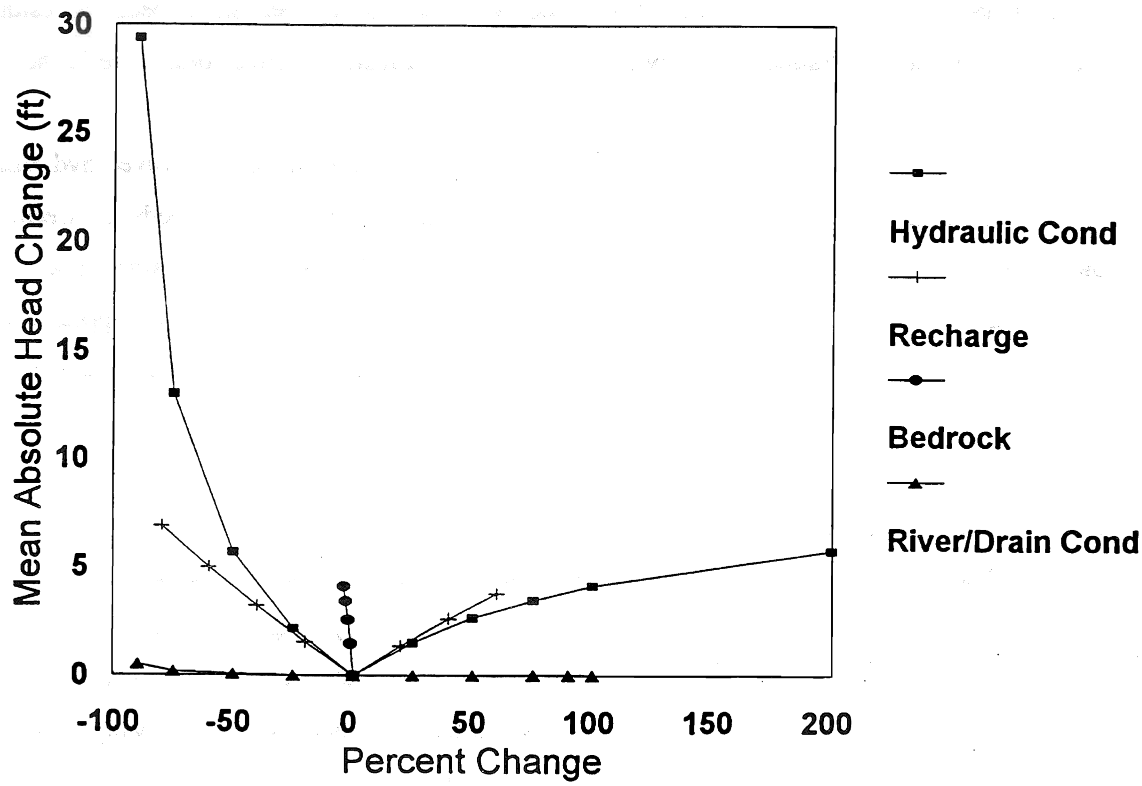

Figure 3.11 Sensitivity analysis for the Stevens Point, Whiting, and Plover model.

PREDICTIVE MODELING

Model design

The calibrated model was used to define the head distribution and internal cell-by-cell flow terms when the system is stressed with the addition of the wellfields operating at projected pumpage levels. Individual wells in wellfields were modeled as point sinks at model nodes using the MODFLOW well package; all other aspects of the calibrated model were the same. The purpose of the simulations was to generate the inputs needed for particle tracking to define zones-of-contribution and times-of-travel (Chapter 4).

For wellhead protection purposes, the question arises as to which pump rates and corresponding zones-of-contribution to use. We chose the estimated year 2005 average daily pump rates (Table 3.3) for delineating zones of influence and times of travel for wellhead protection purposes, as well as predicting future nitrate concentrations. We also simulated year 2005 maximum daily pump rates to examine model sensitivity to pumping rate.

Table 3.3. Municipal wellfield pump rates.

	Model		Pump Rate (cfs)	
Well Name	Row	Col	2005 Avg	2005 Max
Stevens Point #5	58	22	0.422	2.14
Stevens Point #6	42	28	1.238	3.56
Stevens Point #7	41	29	0.984	3.56
Stevens Point #8	39	30	2.100	2.85
Stevens Point #9	45	25	1.062	2.06
Stevens Point #10	35	33	1.336	5.85
Whiting Village #1	80	18	0.380	0.93
Whiting Consolidated #2	77	18	1.672	2.20
Whiting Consolidated #3	78	19	1.672	2.20
Whiting Consolidated #4	79	20	1.672	2.20
Whiting Kimberly Clark Combined #5/#6	72	19	2.298	3.14
Plover #1	97	51	1.350	2.59
Plover #2	100	51	1.350	2.59

The total pumpage and its distribution among wells is based on the best estimate by municipal planners at the time of this writing and may need to be updated as pumpage trends become more defined. The City of Stevens Point anticipates a year 2005 average daily pumpage of 7.142 cfs, compared to the 1990 pumpage of 5.806 cfs (Donohue, 1991). The pumpage for individual wells 5 through 9 were determined by adjusting the 1981-1990 ten year average pumpage for each well (Donohue, 1991) for 1990 pumpage without water sales to the Village of Whiting. Stevens Point Well #10 was brought on-line in 1994, but an anticipated pump schedule was not available. We assumed that to minimize efficiency degradation in wells 5-9, pumpage increases from 1990 to 2005 will be met by the number 10 well. The anticipated maximum day pumpage for 2005 for Stevens Point is 20 cfs, based on 280% of average day demands (Donohue, 1991). The individual well pumpage for maximum day demand for wells 5-9 was based on rated capacity times a long term efficiency factor of 80%, with well #10 assigned any unmet demand. Increased industrial water sales being considered by Stevens Point may significantly increase the anticipated water pumpage.

The Whiting Well #1 serves the village needs. The village does not anticipate any significant change in village water demand, and the average pumpage for 1994 was also used for year 2005 average day pumpage (Schlegel, 1994). The maximum day pumpage was calculated as the 1992-94 average pumpage increased by a factor of 280%, as used for the City of Stevens Point.

The Whiting Consolidated Wells #2-4 average day pumpage for year 2005 was calculated as the average 1994 pumpage (Schlegel, 1994) times an estimated county 10 year growth factor of 10.67% (Census), divided evenly among the wells. The maximum day pumpage was calculated as the 1989-94 average pumpage times a typical maximum day factor (1.37 based on March 1994 daily pumpage) times the projected county growth (10.67%). The Kimberly-Clark Wells #5-6 pumpage were calculated in a similar fashion. A fourth Consolidated well that is being considered may significantly increase pumpage.

The Plover pumpage projections were obtained from Becker-Hoppe (1990), and divided evenly between the two wells.

Results

The effect of the well sinks on the water table configuration is shown in Figure 3.12. Drawdown effects are notable near the wellfields, and to variable extent upgradient.

The model indicates that the Plover wellfield uniformly intercepts westward moving groundwater flow. Even through the wellfield is approximately 1/2 mile south of the Little Plover River, it interacts substantially with the stream. Unlike Stevens Point, the Plover wellfield does not directly induce recharge from the stream, but rather intercepts groundwater that would otherwise discharge to the stream somewhere along its course. The model predicts that pumping the Plover wells at their year 2005 daily average will reduce baseflow by 47% for the stretch above Springville

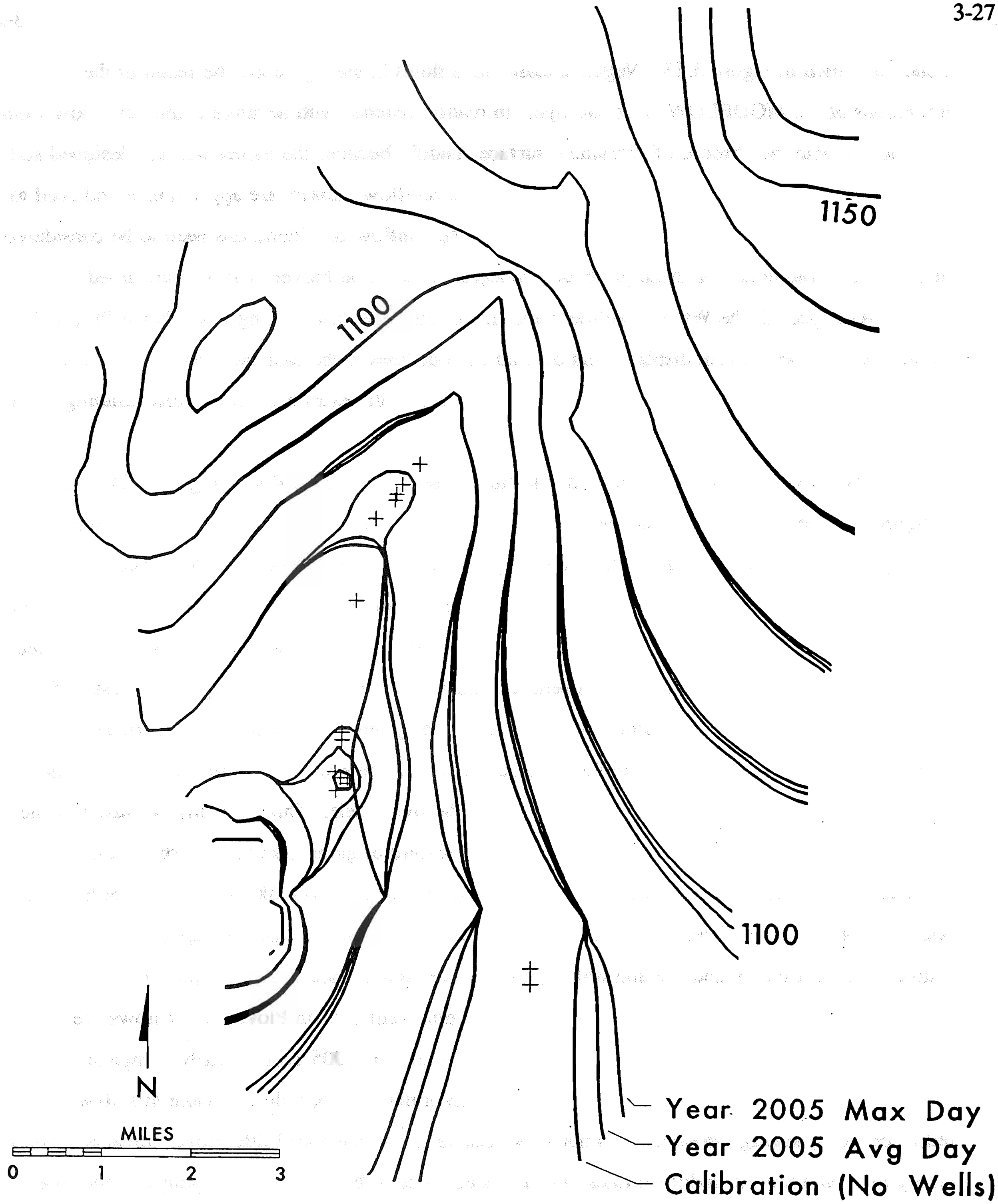

Figure 3.12 Comparison of the water table (feet MSL) calculated for no municipal well pumpage, year 2005 average daily pumpage, and year 2005 maximum daily pumpage.

Pond, as shown in Figure 3.13. Negative cumulative flows in the figure are the result of the limitations of the MODFLOW river package. In reality, reaches with negative cumulative flow would become dry with the absence of substantial surface runoff. Because the model was not designed and calibrated for a Little Plover area scale project, the streamflow impacts are approximate and need to be studied and modeled in greater detail. However, streamflow considerations need to be considered in addition to traditional wellhead protection concerns if the Little Plover is to be maintained.

As expected, the Whiting wellfield appears to intercept flow moving towards the Plover River, as indicated by the uniform displacement of head contour lines to the east and northeast. Some induced recharge from the Plover River is also likely due to the extra head conditions resulting from McDill Pond.

The Stevens Point main wellfield is located close to the Plover River (Figure 2.2), and was designed to benefit from direct induced recharge from the river. The model indicates that wellfield pumpage has little impact on the water table (Figure 3.12), which is largely due to considerable induced recharge. Induced flow particularly limits the amount of groundwater from east of the river that would otherwise find its way to the wellfield. Geologic conditions west of the river also influence groundwater flow. Aside from a buried bedrock valley (Figure 2.9) the recharge area west of the river has low transmissivity resulting from the aquifer being thin (high bedrock), and contributes little water to the wellfield. It is not possible to exactly apportion the source of groundwater being pumped by the Stevens Point wellfield to east, west, or from the river itself. This difficulty is caused by the river cells in the wellfield area exhibiting a complex mixture of gaining and losing stretches. Considering that the model calculates only a net for each cell, it is even likely that some cells contain both gaining and losing faces. Some assumptions, based on a water budget, for apportioning the source areas relative to land use and water quality concerns are discussed in Chapter 6.

The total impact from Stevens Point and Whiting wellfields on Plover River flows are significant, as would be expected (Figure 3.14). The total year 2005 average daily pumpage of these wellfields of 14.8 cfs are a direct or indirect reduction of the river baseflow. While this 10% reduction in the average streamflow is not as noticeable as the potential Little Plover impacts, effects on aquatic communities, sediment rates, and aesthetics need to be considered as pumpage increases.

CONCLUSIONS

The MODFLOW model is a reasonable representation of the unconfined aquifer utilized by the municipal wellfields, and suitable for supporting a particle tracking analysis of the

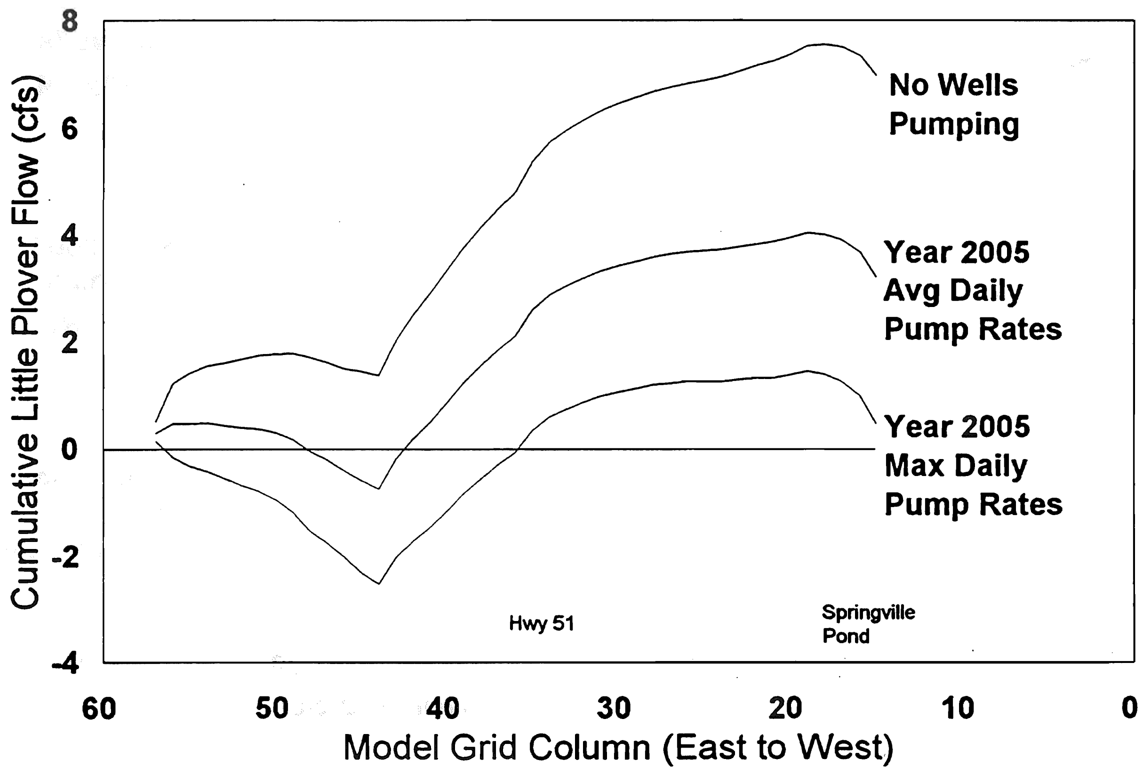

Figure 3.13 Cumulative baseflow in the Little Plover River under three pumping scenarios.

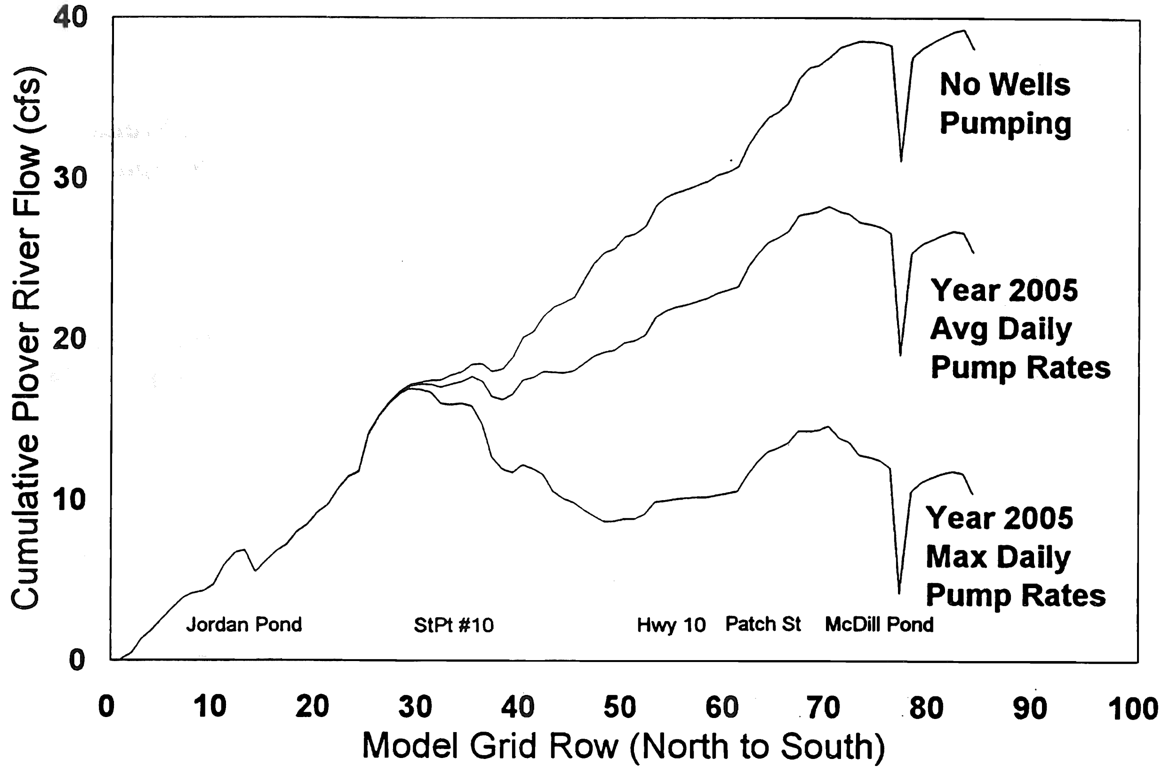

Figure 3.14 Cumulative additions to the Plover River baseflow under three pumping scenarios.

wellfields. As additional data become available, the model will be adjusted to maintain its usefulness as a management tool. Groundwater-stream interactions should be explored in more detail, especially in the Stevens Point main wellfield area. Model design and input data fit and support a regional analysis; the model should be interpreted with caution for localized or site specific study.

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CHAPTER 4

ZONE-OF-CONTRIBUTION DELINEATION

INTRODUCTION

A well's "zone-of-contribution" is the land surface that contributes groundwater recharge to the well. Frequently, the zone-of-contribution or the area within a certain time of travel within the zone-of-contribution is adopted as a wellhead protection area. A number of methods are available to delineate a well or wellfield's zone-of-contribution. We used particle tracking model techniques to accomplish the delineation for the municipal wells in this study. Particle tracking models use the distribution of hydraulic heads (water table elevation) and aquifer parameters from a groundwater flow model to calculate flow paths for imaginary fluid particles moving through a groundwater flow field. If the coordinates for imaginary particles are placed around pumping wells and transported backward in time and space, the flow paths they trace delineate the zones-of-contribution. Particle tracking models can also calculate cumulative times-of-travel at intermediate positions for each particle, allowing times-of-travel (TOT) to be delineated. While in this study the particle tracking model was implemented with particular interest in the municipal wells, the model can be used to trace groundwater flow up- or down-gradient from any other point of interest in the domain.

PARTICLE TRACKING MODEL

Model Code and Inputs

The particle tracking code used for this delineation is the USGS MODPATH model (Pollock, 1989). MODPATH requires the MODFLOW input and output files plus porosity information and an indicator for each component of the stress packages that specifies how the flow is handled relative to the cell geometry. A uniform porosity of 0.32 was used (see Chapter 2). Recharge, river, and drain flow were assigned to the top cell face. Wells were treated as internal sinks.

Limitations/Assumptions

While particle tracking models are widely used for wellhead protection area delineation, certain limitations should be considered. The quality of the groundwater flow model used as input to the particle tracking model is a potential limitation because uncertainties in the flow model are inherited by the particle tracking model. Grid discretization can also become a limitation. Too coarse of a grid can result in a weak sink condition, which occurs when a sink within a cell (e.g., well or river) discharges less water than that entering the cell. In such a case, the model has no way to determine whether a particle entering one face of such a cell should discharge to the sink or pass through another cell face. Another possible limitation is a two-dimensional simulation of a three-dimensional problem. Two-dimensional flow models can introduce particle tracking errors even though the flow model appears to adequately describe the head distribution (Barlow, 1994). Vertical averaging in 2-dimensional models cannot describe some features that can significantly impact particle flowpaths, such as discrete vertical zones of low hydraulic conductivity, large horizontal to vertical hydraulic conductivity ratios, partially penetrating wells, shallow streams (weak sinks), and low pump rates.

Most of the potential limitations listed above are not problematical for the current application. As noted in Chapter 3, the MODFLOW flow model appears to adequately represent the SWP area. Model confidence will increase as additional parameter and verification data become available. While cell size remains a tradeoff between minimizing cell size and computational logistics, a concerted effort was made to minimize cell size for well and important river cells. The choice of a 2-dimensional model is generally adequate for the outwash aquifer, and unrepresented vertical heterogeneity and anisotropy should not be a significant problem.

Weak sink conditions do pose a problem for particle tracking relative to Plover River cells in the vicinity of the Stevens Point main, and to a lesser degree, the Whiting wellfield. In the real world, the river was fully penetrating before the wellfields were developed; the river captured all groundwater flow from the east and west. With wellfield development, the river ceased to be fully penetrating in the vicinity of the wellfields. In the model, when the simulated wellfields are activated, cells representing the river are no longer fully discharging to the river. Some water passes through the river cells and to the wells, and some discharges to the river. MODPATH allows the user to specify whether particles in this situation should stop at a weak sink, pass through the cell, or pass through only if a specified fraction of water passes through. For this study, the model was constructed so that particles pass through the weak sink cells. This then delineates a potential zone-of-contribution; water recharged in the area delineated by these particle traces may discharge either to the wellfield or to the Plover River. Impacts on Stevens Point main and Whiting wellfields are discussed later in this chapter. A more exact description of the Plover River flow dynamics could be made using a localized 3-dimensional flow model with smaller grid size.

The particle tracking model introduces one more parameter to the problem, that is, aquifer porosity. Error in the use of a uniform porosity of 0.32 would be manifested in errors in times-of-travel. To illustrate, consider the 10 year time-of-travel. If the porosity value is 20% low, the line delineating the 10 year time-of-travel would actually be the 8.3 year time-of-travel; if the value is 20% high, it would be the 12.5 year time-of-travel. The consequences of a nonuniform spatial distribution of porosity might have similar types of consequences. In addition, it might make the time-of-travel lines appear somewhat more ragged, but would not likely change delineations substantially.

Particles

The model can track a particle forward or backward in time and space. For wellhead protection, the particles are typically started at the well and tracked backward to their source. The location of the particles at specific times allows delineation of recharge subareas for management based on a time-of-travel criteria.

For the SWP model, 30 particle starting locations were equally spaced on a circle with a radius of 300 feet around each well (Figure 4.1). An adjustment was made for time-of-travel between the offset and the well. The adjustment required an estimation of average groundwater velocity in the 300 foot zone, which was calculated from hydraulic gradients between the pumping well cells and adjacent cells and local hydraulic conductivities. The mean average velocity for all the wells in the municipal wellfields was 7.2 ft/day for the 300 foot offset. Therefore, the particle starting locations represent an average time-of travel off-set of about 42 days. These calculations rely on a number of assumptions, however, even gross errors make little difference to the delineated times-of-travel. Even an error of 100% in time-of-travel offset creates only a 10% error at the 1 year time-of-travel.

As noted in the previous section, the mixed flow patterns in the Plover River area adjacent to the Stevens Point main wellfield complicated particle tracking analysis. Barlow (1994) noted that forward tracked particles are often a better choice for delineating zones-of-contribution in complex flow systems. To better define the potential contributing area, particles were also started at various locations in the potential northeast zone-of-contribution for forward tracking analysis.

ZONE-OF-CONTRIBUTION MAPPING

MODPATH Zones-of-Contribution

Using the flowpaths delineated by MODPATH, the recharge zones and times-of-travel were mapped for the municipal wellfields using year 2005 average daily pump rates (Figure 4.2). The areas included within the plotted flowpaths represent the total zone-of-contribution, but the density of

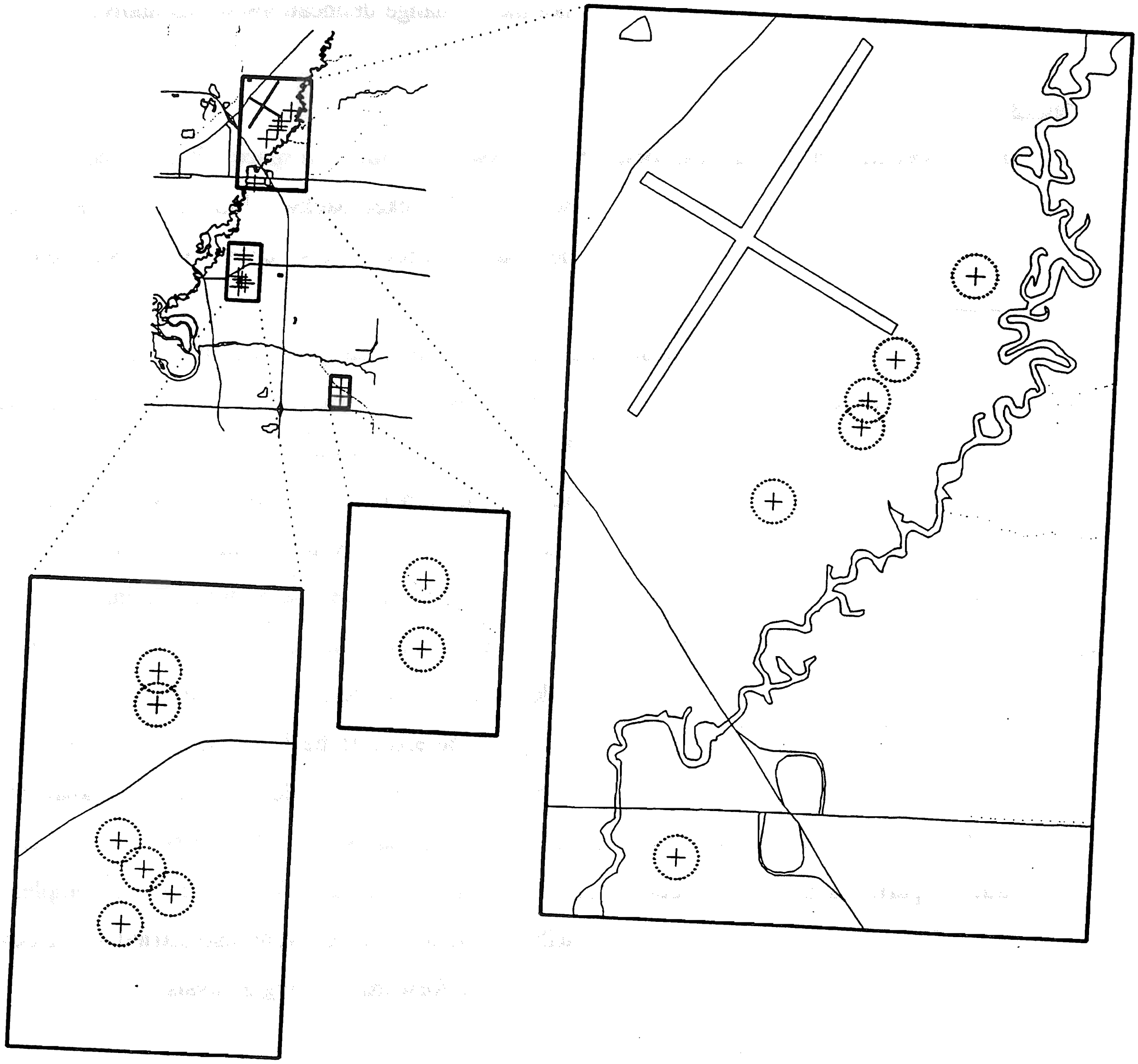

Figure 4.1 Starting locations for particles used to delineate zones-of-contribution.

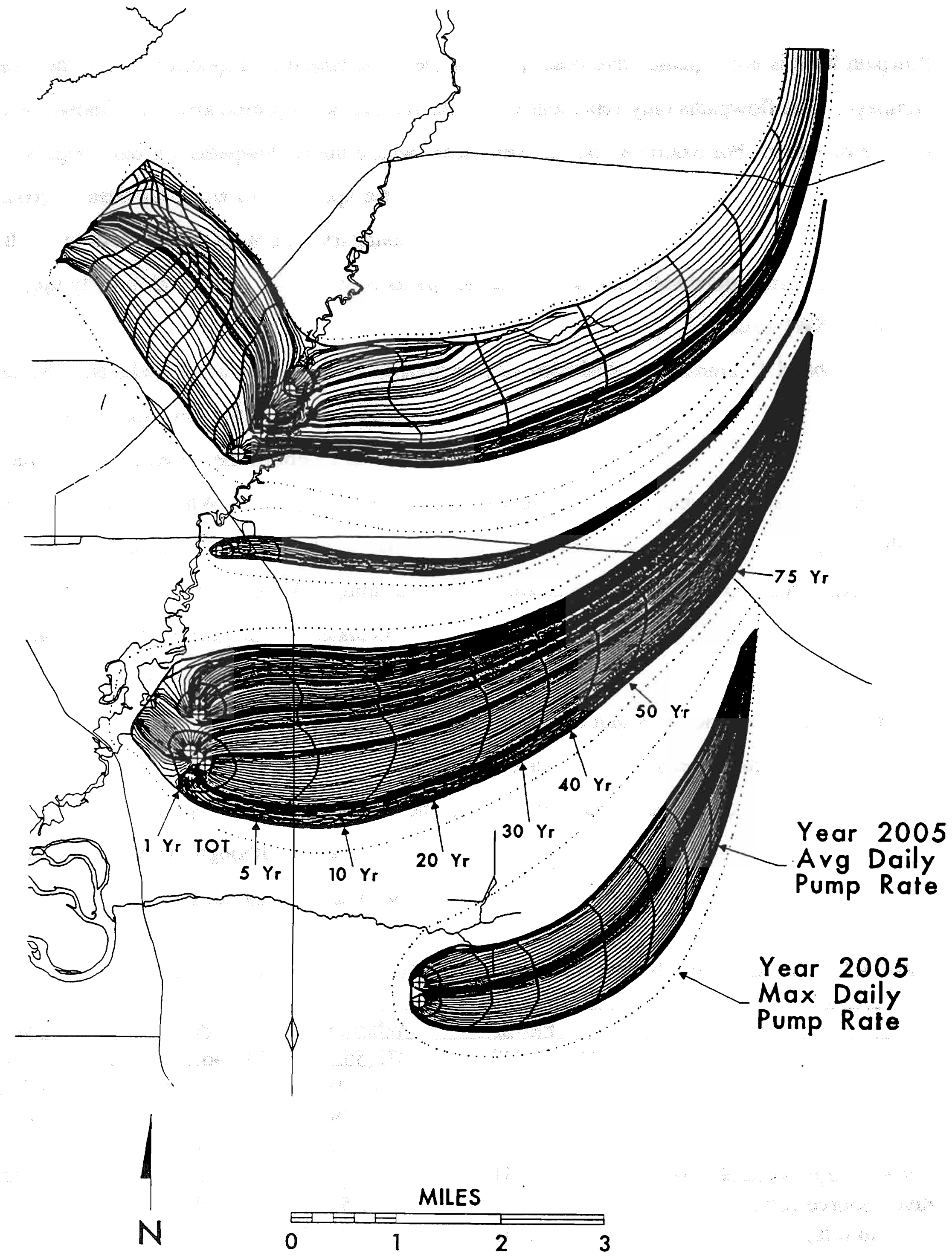

Figure 4.2 Zones-of-contribution and times-of-travel for the Stevens Point, Whiting, and Plover municipal wellfields using MODPATH.

flowpath lines is not a quantitative description of the contribution of a specific area to the total well pumpage. The flowpaths only represent a particle and are not representative of a known or consistent volume of water. For example, the "darker" areas where many flowpaths are close together do not necessarily contribute more water to the wells. Also, the upper limits along the eastern groundwater divide are truncated slightly to provide a reasonable boundary. Because of the north to south gradient along the eastern groundwater divide, all the flowpaths converge and track indefinitely upgradient within the SWP area.

Table 4.1 summarizes total zone-of-contribution sizes and net water budgets. The calculated recharge volumes for the delineated areas balance reasonably well with pump and river flow rates. The indefinite delineation and truncation of the easternmost extent of the zones-of-contribution may contribute to discrepancies. Water budgets for Stevens Point main and Whiting are also complicated by the complex interactions with the Plover River as discussed above for weak sink situations. The net water budget for Stevens Point main is somewhat misleading. Although the Plover River has a net gain of approximately 2 cfs, the flow model actually calculates a total gain of 6.3 cfs along with a total loss of 4.3 cfs for the wellfield reach. Given the flow system, it is reasonable to assume that the 4.3 cfs lost from the Plover is induced recharge to the wells. This has significant implications for the loading calculations presented in Chapter 6, where a source partitioning of water pumped at the wellfield is presented. In the vicinity of Whiting, the Plover River has a net loss of 1.5 cfs. The Whiting wellfield captures some of this flux as induced recharge, although the exact fate of this flow is unknown. Implications for loading calculations are also noted in Chapter 6.

Table 4.1. Size and water budget for municipal wellfield zones-of-contribution.

	Plover	Whiting	St Pt #5	St Pt Main
Area: sq feet	94,410,838	256,072,552	23,146,882	278,368,650
acres	2,167	5,879	531	6,390
sq miles	3.39	9.19	0.83	9.99
Avg Recharge Rate (in/year)	9.24	9.21	9.37	8.23
Tot Recharge Volume (cfs)	2.31	6.23	0.57	6.05
River Source (cfs)	0	1.5?? (McDill)	0	2.42 (Lost Cr)
Total In (cfs)	2.31	6.23 (7.73??)	0.57	8.47
Pumpage (cfs)	2.70	7.69	0.44	6.72
River Sink (cfs)	0	0	0	1.96
Total Out (cfs)	2.70	7.69	0.44	8.68

The recharge area for the Stevens Point main wellfield can be separated into distinct areas, one west of the Plover River, which generally trends northwest of the wellfield, and one east of the Plover River, trending to the northeast (Figure 4.2). Much of the west recharge area has high bedrock and contains low permeability aquifer materials, and this is reflected in closely spaced time of travel lines which indicate slower groundwater movement in this area. Travel times are less certain in the shallow bedrock area due to a lack of geologic data and uncertainty regarding hydraulic conductivity and recharge rates. However, the time of travel delineations are sufficiently accurate for wellhead protection management decisions. In the northern portion of the west recharge area, the effect of thicker outwash materials associated with the buried bedrock valley is evident in the bending and greater spacing of the time of travel lines. The western recharge area extends to the groundwater divide with the Hay Meadow Creek basin.

In the eastern zone-of-contribution of the Stevens Point main wellfield, the complexity of groundwater flow patterns relative to Plover River weak sinks is apparent. As previously noted, the east zone-of-contribution must be considered a maximum potential delineation because of the unknown fate of individual flowpaths at river cells. Lost Creek further complicates the delineation because it is a variably gaining/losing stream. Ultimate management of pump rates also complicate the delineation. From a wellhead protection perspective, these unknowns would not obviate the need for contaminant management. The maximum extent of the eastern zone-of-contribution trails off to the northeast along the groundwater divide with the Tomorrow/Waupaca River basin.

The Stevens Point #5 well zone-of-contribution is a narrow finger extending to the east and northeast. The model indicates drawdown is insufficient to cause an interaction with the Plover River to the west. The precise location of this narrow zone-of-contribution is more susceptible to model uncertainties than the other zones-of-contribution. A slight change in the water table configuration could significantly shift the location of the zone-of-contribution at greater distances from the well.

The Whiting wellfield has a large zone-of-contribution extending to the east and northeast to the groundwater divide. Particle tracking also indicates that flowpaths extend from the wellfield west to McDill Pond, creating the weak situation previously noted. A qualitative assessment of the particle traces suggests some induced recharge from the Plover River, but very little underflow from the west. Given the uncertainties and the very small potential amounts of underflow, no zone-of-contribution was included for Whiting west of McDill Pond.

The Plover zone-of-contribution is shaped similar to the others, with upper limits trailing upgradient to the north along the groundwater divide. There is no direct interaction with the Little Plover River.

Zones-of-contribution for the year 2005 maximum daily pump rates are also mapped on Figure 4.2. These exhibit varying sensitivity to the pump rate. For example, the Plover boundary expands disproportionately to the north. The northern boundary of the Stevens Point main wellfield changes very little, probably related to the efficiency of the #10 collector well to induce recharge from the Plover River. Wellhead protection managers need to anticipate long term water needs and manage the corresponding recharge zones.

<u>Comparisons with Previous Delineations</u>

Zones-of-contribution and times-of-travel for the municipal wells were previously delineated individually by a variety of methods. While the basic representation of the flow system should be consistent in all cases, boundary and time-of-travel details will necessarily vary because of the delineation method used, differences in pump rates, and available hydrogeologic data.

The Plover zone-of-contribution was previously delineated by Becker-Hoppe (1990) using hydrogeologic mapping. This method is analogous to the MODPATH method presented in this report, except that flowpaths were drawn subjectively based on water table maps rather than being calculated mathematically. A comparison of the two delineations (Figure 4.3) shows that they have similar shapes, but the previously mapped zone-of-contribution is larger, dips substantially towards the south, and misses an area to the northeast. These differences probably reflect the subjective interpretation of flowpaths. The hydrogeologic mapped area is also not directly based on any specific pump rate as the MODPATH delineation was.

Plover times-of-travel were also previously drawn by Becker-Hoppe (1990) using Darcy's Law, which relates groundwater velocity to the hydraulic conductivity, groundwater gradient, and effective porosity of the aquifer material. From the velocity, travel distance was computed for chosen elapsed times, and used to draw arcs of uniform distance from a well. The MODPATH 5 and 10 year time-of-travel limits extend further upgradient than the average velocity based distances. The MODPATH times-of-travel should be more accurate as they incorporate the effects of the spatial distribution of hydrogeologic parameters and cell-by-cell flow rates, and reflect the travel time of individual particles traversing different portions of the zone-of-contribution.

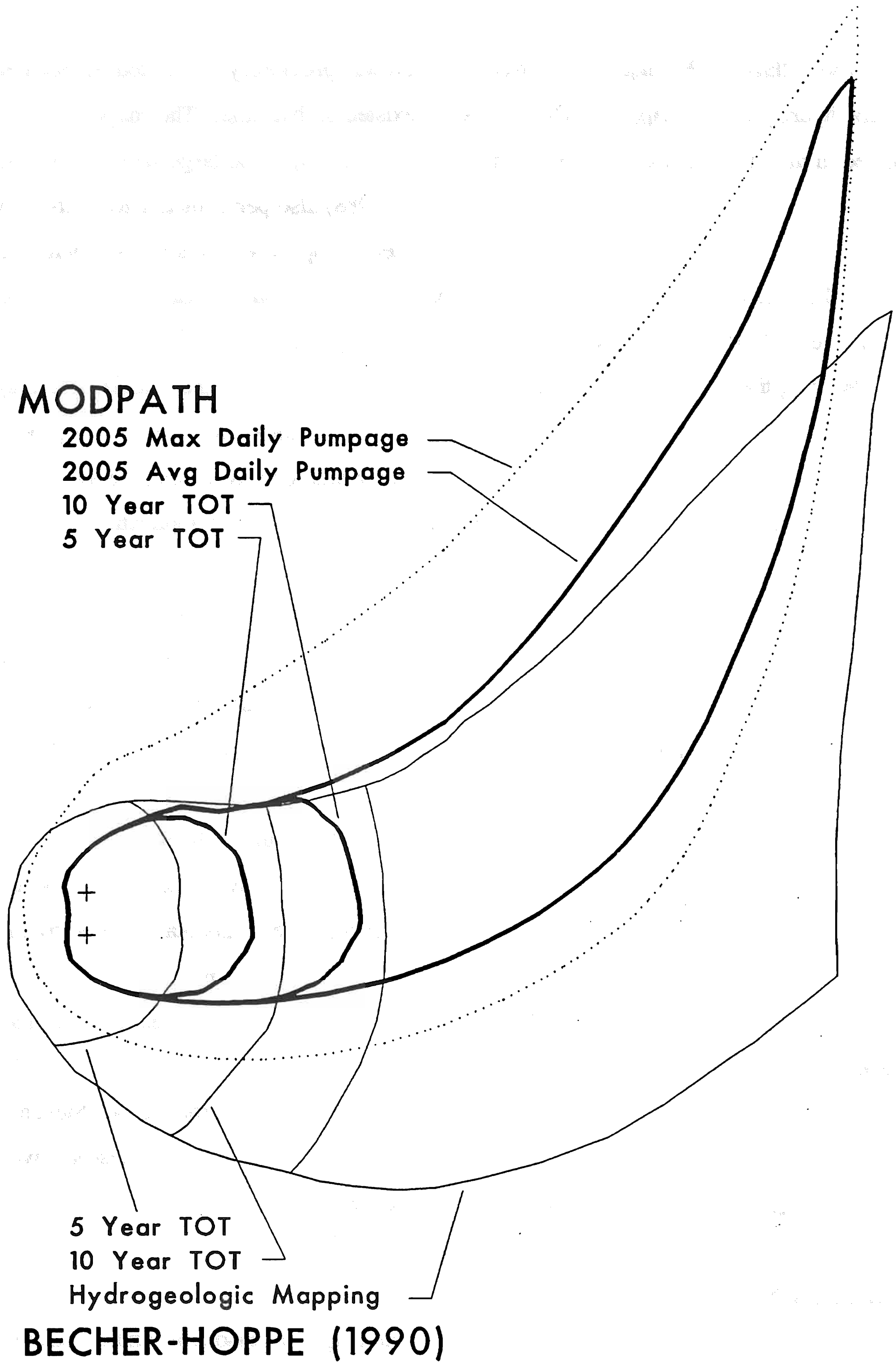

Figure 4.3 Comparison of zone-of-contribution delineations for the Plover municipal wellfield.

The Village of Whiting's zone-of-contribution was previously delineated by Born et al. (1988), also using hydrogeologic mapping. Only wells 1-4 existed at that time. The shape is similar to that delineated in this study, but the size is much larger (Figure 4.4). The large size is apparently a conservative interpretation of flowpaths. Born et al. (1988) also performed a zone-of-contribution delineation based on the minimum recharge area needed to supply 4.25 cfs for wellfield pumpage with a uniform 8 inches of recharge per year. This zone-of-contribution delineation is smaller and more similar to the MODPATH delineation which uses a pumpage of 7.69 cfs.

Whiting times-of-travel were also previously determined using the calculated average linear velocity method described for Plover. The 1, 5, and 10 year time-of-travel lines are generally similar to MODPATH determinations (Figure 4.4). The 20 year time-of-travel using the average velocity calculation was considerably further upgradient, suggesting limits to the application of averaged values away from the well.

A previous recharge delineation for Stevens Point wells 5-9 was done by Brown et al. (1992), and for well #10 by RUST (1994), using the FLOWPATH combined flow and particle tracking model. Substantial differences exist between the two models, due to different inputs (Figure 4.5). The FLOWPATH model pump rates for wells 5-9 were approximately equal to the year 2005 maximum daily rate, while the #10 well pump rate of 11cfs was nearly twice that rate. The FLOWPATH model produces a pronounced difference in the orientation of the flowpaths to the southeast before swinging to the northeast compared to the MODPATH easterly orientation. Available regional water table mapping does not support the southeasterly flows, which could be an anomaly caused by input data or modeled heads not adequately representative of the regional flow system. Because simulated pumping for well #10 was at a much higher rate for FLOWPATH, the recharge delineation in that area is not comparable.

The times-of-travel delineations are also considerably different for the two Stevens Point models. The FLOWPATH simulation consistently locates the 5 and 10 year times-of-travel considerably upgradient compared to MODFLOW/MODPATH.

CONCLUSIONS

The MODPATH delineated zones-of-contribution appear reasonable and consistent. The MODFLOW/MODPATH modeling uses updated input data to provide a regional perspective of the individual zones-of-contribution, and resolves conflicting delineations between previous efforts. The

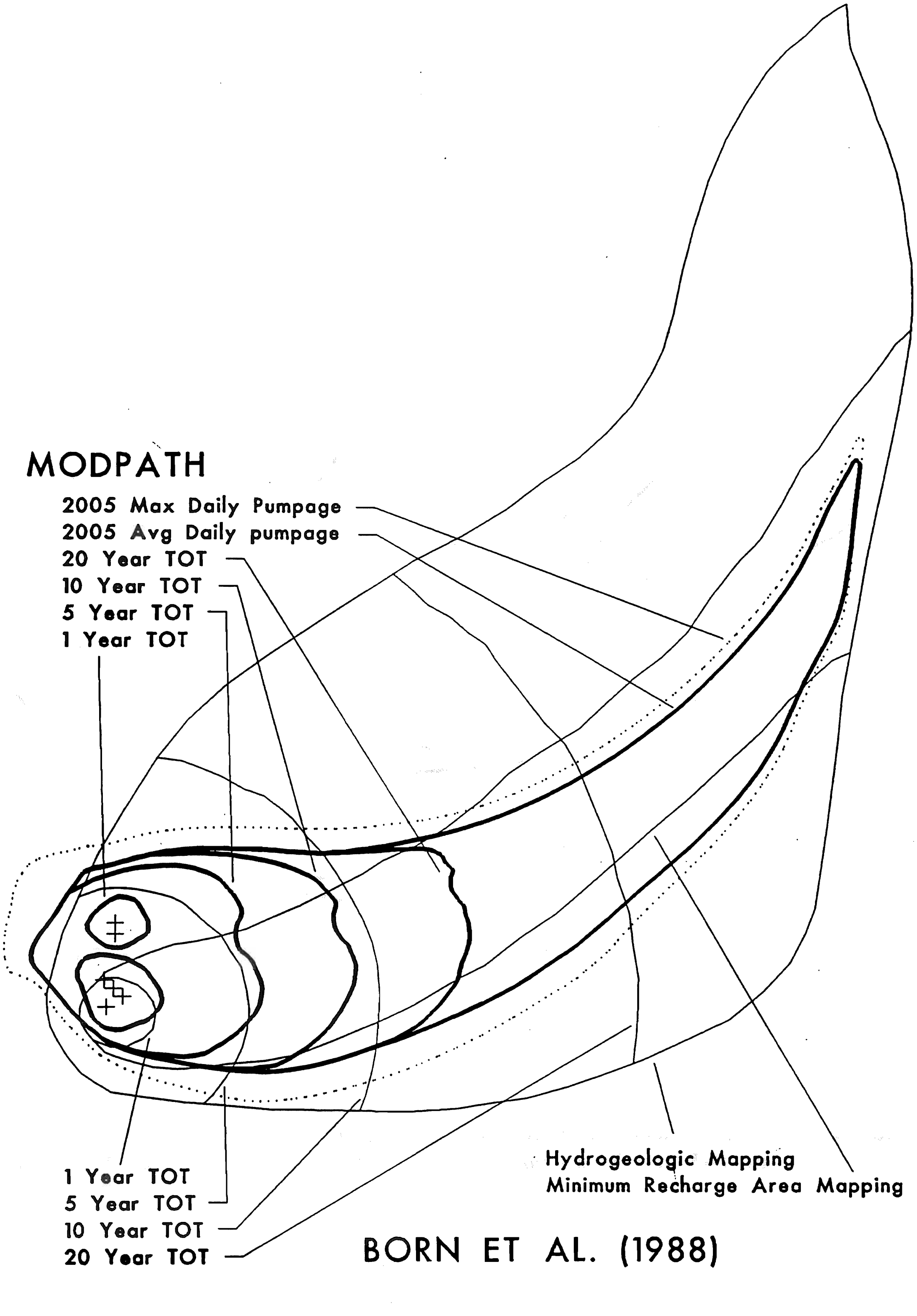

Figure 4.4 Comparison of zone-of-contribution delineations for the Whiting municipal wellfield.

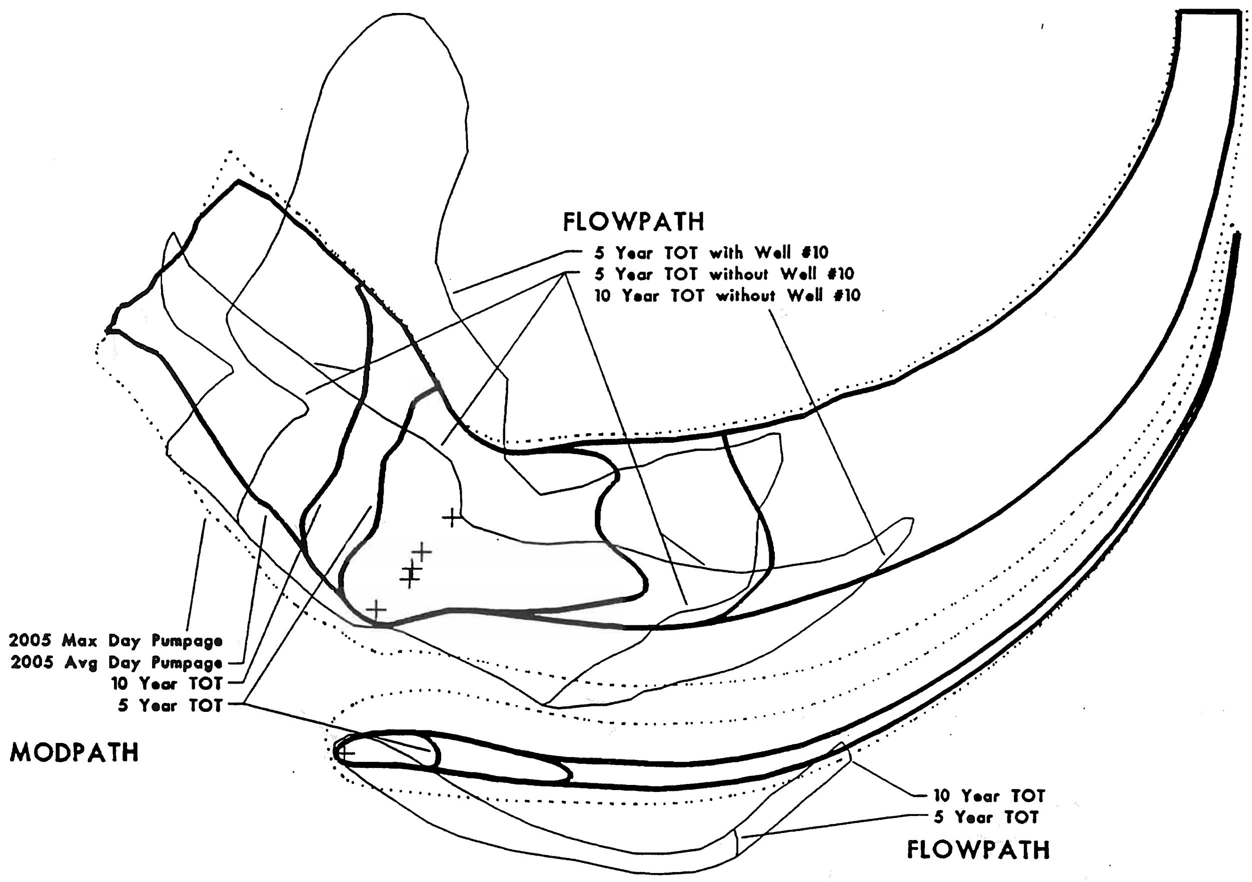

Figure 4.5 Comparison of zone-of-contribution delineations for the Stevens Point wellfields (FLOWPATH delineations adapted from Brown et al., 1992, and RUST, 1994).

complex flow patterns around the Plover River in the vicinity of the Stevens Point main and Whiting wellfields provide the most uncertainty, although a workable analysis for the wellfields is provided.

Future work beyond the scope of this project could include development of a three-dimensional small cell size model for the Plover River in the vicinity of the Stevens Point main and Whiting wellfields in order to better define river sinks, sources, and underflow. Quantitative forward tracking flowpath analysis could also be used effectively with a 3-dimensional Stevens Point model to better define zones-of-contribution east of the Plover River. Wellfield specific models might also be extracted for the other wellfields to reduce cell size and improve the flowpath definition.

The Stevens Point main wellfield induces considerable recharge from the Plover River, which minimizes the contribution from the potential zone-of-contribution to the east. At assumed pump rates, water quality will be primarily influenced by Plover River quality and recharge from the northwest. Significant increases in anticipated pump rates will require further investigation to redefine contributing areas.

Whiting and Plover wellfields primarily capture groundwater flow from the east. Water quality will be defined in terms of the land use in the areas to the east.

CHAPTER 5

NITRATE LOADING TO GROUNDWATER

This chapter develops nitrate loading estimates for the nitrate sources occurring within the study area. The succeeding chapter uses these data to calculate an average, steady-state nitrate concentration for the municipal wells and their recharge areas.

Groundwater pumped by a municipal wellfield can be partitioned as that originating from precipitation recharge and that originating from surface water recharge. With this in mind, a predicted steady-state nitrate concentration at the municipal wells can be calculated as

$$[NO_3\text{-}N]_{SS} = \frac{Q_P \times [NO_3\text{-}N]_{P,SS} + Q_S \times [NO_3\text{-}N]_S}{Q}$$

where

$[NO_3\text{-}N]_{ss}$	= steady-state NO_3-N concentration for the wellfield
Q_P	= wellfield pumpage originating from precipitation-recharged groundwater
$[NO_3\text{-}N]_{P,SS}$	= average steady-state NO_3-N concentration in precipitation-recharged groundwater
Q_S	= wellfield pumpage originating from surface water recharged groundwater
$[NO_3\text{-}N]_S$	= average NO_3-N concentration in surface water recharged groundwater
Q	= total wellfield pumpage

The steady-state nitrate concentration in precipitation-recharged groundwater is the main focus of this chapter. It can be expressed as

$$[NO_3\text{-}N]_{P,SS} = \frac{N_T}{R_P}$$

where

N_T	= Mass of NO_3-N loaded to groundwater annually from the surface of the recharge area
R_P	= Volume of annual precipitation-recharged groundwater

A simplifying assumption is that no nitrate sinks (e.g., denitrification) are operative in the aquifer. This assumption appears valid based on extensive monitoring within the basin.

For this analysis, nitrate loading is divided into four compartments; residential, croplands, legume forage, and manure. This can be expressed as

$$N_T = N_R + N_C + N_L + N_M$$

where

N_R = NO_3-N loaded from residences

N_C = NO_3-N loaded from cropland

N_L = NO_3-N loaded from legume forage

N_M = NO_3-N loaded from manure

All the above are expressed in units of lb/yr. We eliminated nitrate loading from other land-uses and management from consideration because they are minor at the scale of this analysis.

NO_3 LOADING FROM RURAL RESIDENTIAL LAND USE

The primary sources of NO_3-N loading in unsewered residential areas are on-site septic systems and lawn fertilization. Septic system loading is dependent on the number of systems per acre and the number of persons using each system. We adopted a loading value of 10 pounds of nitrogen per person per year based on previous studies (Shaw et al., 1993). We further assumed an average of 4 persons per system, yielding a septic system loading of 40 lbs N per system per year. (Current census data indicate an average of 3 people per system, so this is likely an overestimate of current septic system loading.)

Lawn fertilization is a secondary component of the nitrogen loading in residential and urban areas. Loading variables include the percentage of residential lot owners who apply lawn fertilizers, the number of applications per year, the amount of nitrogen per application, lot size, the percentage of the lot in turf, and the nitrogen leaching rate. A study of two subdivisions in the SWP area (Shaw et al., 1993) determined that an average of 1.6 fertilizer applications are made per year, including homeowners who apply none, to a minimum of approximately 40% of the total lot area, with a leaching rate of 25%. Assuming 40 lbs fertilizer per 10,000 square feet at 28% N, approximately 8 lbs N/acre/year are leached to groundwater in residential areas.

We identified four types of residential land uses within the study area, higher density unsewered (~1 acre lots), lower density unsewered (~4 acre lots), urban sewered, and mobile home park unsewered (0.25 acre lots).

The total N loading for the higher density unsewered residential land use is 48 lbs N/acre/year, 40 lbs from the septic system and 8 lbs from lawn fertilization. Low density unsewered residential loading is 12 lbs N/acre/year. This is due to 40 lbs of loading from septic systems divided over 4 acres (10 lbs/acre), plus loading from lawn fertilizer. The amount of turf in low density appears to be about the same as for the higher density unsewered residential. When spread over the 4 acres of the land use, the 8 lbs N from lawn fertilization results in a loading of 2 lbs N/acre/year.

Nine acres of rural mobile home park land use were identified in the study area. For nitrogen loading calculation purposes, 0.25 acre lots were assumed, yielding a loading rate of 160 lbs N/acre/year from septic systems. When loading from lawn fertilizer is added, unsewered mobile home parks are assigned a loading of 168 lbs N/acre/year.

In summary, nitrate loading from residential areas is due to septic systems (in unsewered areas) and lawn fertilization. We used loading estimates of 10 lbs/person-year of nitrogen for septic systems, and 8 lbs/acre-year for lawn fertilization. These yield loading estimates from the four identified residential land uses of: higher density unsewered - 48 lbs/acre-yr; lower density unsewered - 12 lbs/acre-year; sewered - 8 lbs/acre-year; and mobile home park - 168 lbs/acre-year. These are likely overestimates for two reasons. First, the numbers of persons per household used in the septic system estimate are higher than what is current. Second, lawn fertilization estimates likely assume more rural residents fertilize than what is actual.

NO_3 LOADING FROM CROPLANDS

NO_3-N loading from croplands within a recharge area can be estimated by summing the NO_3-N loading (lb/acre-yr) from each crop multiplied by the area occupied by that crop. This can be written as

$$N_C = \sum_{i=1}^{i} (N_{c,i} \times A_{c,i})$$

where

$N_{c,i}$ = NO_3-N leached under crop i (lb/acre-yr)

$A_{c,i}$ = Area in the WHPA occupied by crop i (acres)

For instance, if the only crops in a WHPA are field corn, potato, and sweet corn, the equation becomes

$$N_C = \textit{(lb/acre-yr } NO_3\textit{-N from sweet corn} \times \textit{acres of sweet corn)} + \textit{(lb/acre-yr } NO_3\textit{-N from potato} \times \textit{acres of potato)} + \textit{(lb/acre-yr } NO_3\textit{-N from snap bean} \times \textit{acres of snap bean)}$$

$N_{c,i}$ estimates for the crops grown in the basin were developed using a budget approach and considered both conventional and best management practice scenarios. Forage-legume and manure-N are neglected for the present, and accounted for in their own compartments. This budget approach is a modification of Meisinger and Randall (1991). It is well suited for this application, because the study area groundwater/soil system is highly responsive and because it is particularly applicable for predicting steady-state scenarios. $N_{c,i}$ in this report is equivalent to the Meisinger and Randall "LPLN", Long Term Potentially Leachable Nitrogen. Recent work (Kraft et al., 1995) in the Wisconsin Central Sand Plain has shown that this approach agrees well with loading estimates based on groundwater monitoring.

The N budget for a given crop between the plant canopy and bottom of root zone can be written as

$$0 = N_{input} - N_{output} - \Delta N_{st} - N_{c,i}$$

where

N_{input}	= N that enters the field
N_{output}	= N that leaves the field, excluding leached nitrogen
ΔN_{st}	= change in stored N

A steady state ($\Delta N_{st}=0$) is assumed with respect to inorganic N, that is, the inorganic N content of the soil doesn't increase or decrease consistently over time. In addition, to help simplify the accounting procedure a bit, we treat the mineralization of soil organic matter as an input rather than as a change in storage. Considering these and rearranging, the N budget becomes:

$$N_{c,i} = N_{input} - N_{output}$$

The inputs considered in development of this budget were fertilizer, nitrogen fixation, precipitation, dry deposition, mineralization, and crop seed. Irrigation water nitrogen is neglected because it represents nitrate already leached to groundwater; including it would cause a double counting of this nitrogen. Outputs considered were harvested crop, ammonia and denitrification losses, soil erosion and runoff, and miscellaneous gaseous losses.

Nitrogen Inputs

Fertilizer

Fertilizer-N inputs for various crops for both conventional agricultural practice (CON) and best management practice (BMP) scenarios were supplied from surveys collected by the USDA Stevens Point - Whiting - Plover Wellhead Protection Project (Table 5.1; Ebert, written comm., 1995)

Table 5.1. Fertilizer-N applied to crops in the study area under best management practices (BMP) and conventional practices (CON) (Ebert, written comm., 1995).

	Fertilizer-N (lb/acre)	
Crop	BMP	CON
Field corn, irrigated	150	161
Field corn, nonirrigated	120	120
Oat	60	70
Pasture	0	0
Pea	40	60
Pickle	100	100
Potato*	175	200
Rye	50	70
Snap bean	60	100
Sod	40	45
Sorghum-sudan forage**	120	150
Soybean	0	60
Strawberry	45	55
Sweet corn	150	179

* Average of late and early potato.

** Sorghum-sudan grown for fumigant receives 20 lb/acre (BMP) or 40 lb/acre (CON). We are neglecting fumigant sorghum-sudan in this study.

Microbial Fixation

N fixation is a significant input for soybean and snap bean. We used a value of 68 lb/acre for soybean based on the average of Midwestern conditions (Meisinger and Randall, 1991; p. 132). N fixed by snap bean was estimated at 29 lb/acre based on the following assumptions:

- harvested portion = 4.5 tons/acre
- pod N content is 7.8 lb/ton
- above ground plant-N content = 1.2 x pod N content
- 70% of above-ground plant-N is fixed by plant (Note: this is the low end of the 70-95% range provided in the literature)

Additional detail is provided in Meisinger and Randall (1992, p. 100).

Atmospheric Deposition

Atmospheric deposition has both wet (precipitation) and dry components. Annual average Wisconsin precipitation-N was reported at 13 lb/acre by Andraski and Bundy (1990) and about 18 lb/acre by Hoeft et al. (1972). These estimates are supported by work in the upper Midwest by MacDonald et al. (1992). We adopted 13 lb/acre as an estimate of precipitation-N based on the more recent and Wisconsin specific work of Andraski and Bundy. Dry deposition-N can be approximated as about the same as precipitation-N (Meisinger and Randall, 1991; Schepers and Mosier, 1991), bringing total atmospheric deposition N to 26 lb/acre.

Mineralization

Oberle and Keeney (1990) found a net N release of 40 lb/acre-yr from soil organic matter mineralization on Plainfield loamy sand during the growing season. However, neither wet nor dry deposition were taken into account in the study. For this study, we estimated soil mineralization-N at 14 lb/acre, which is the Oberle and Keeney 40 lb/acre minus 26 lb/acre of wet + dry deposition N. This estimate is low compared to the rule of thumb, which estimates that 2% of soil organic matter mineralizes each year and soil organic matter contains 5% nitrogen. Estimates based on this rule for a soil of 1.5% organic matter amount to about 30 lb/yr compared to our estimate of 14.

Crop Seed

Based on average seeding rates, corn seed adds less than 0.25 lb/acre; potato seed, about 6-10 lb/acre (Meisinger and Randall, 1991). We included potato seed-N at 8 lb/acre and viewed seed-N from other crops as negligible.

<u>Nitrogen Outputs</u>

Harvested Crop

Harvested N (Table 5.2) was estimated from crop yield (Ebert, pers. comm., 1995) and literature-reported crop-N content (Table 5.4 in Meisinger and Randall, 1991). The sweet corn harvest-N excludes stalks harvested for fodder. This was neglected because only 15% or so of sweet corn fodder is removed from fields (Ebert, pers. comm., 1995).

Table 5.2. Crop yield and harvested-N per acre for crops grown in study area. Harvested-N per unit obtained from Meisinger and Randall (1991). Harvest per acre provided by Ebert, pers. comm. (1995).

Crop	Harvested-N/unit (range)	Harvested-N/unit (average)	Units	Units/acre	Harvested-N (lb/acre)
Field corn, irrigated	0.64-0.83	0.73	lb/bu	150	109
Field corn, nonirrigated	0.64-0.83	0.73	lb/bu	90	66
Oat*	0.83-1.06	0.94	lb/bu	80	75
Pasture					
Pea	14-19	17	lb/ton	3	51
Pickle	2.0-2.9	2.4	lb/ton	10	24
Potato***	0.3-0.5**	0.4	lb/100 lbs	400	160
Rye	0.95-1.2	1.05	lb/bu	60	63
Snap bean	6.5-9.0	7.8	lb/ton	5	39
Sod					
Sorghum-sudan	6.8-12	9.5	lb/ton	7	66
Soybean	3.1-3.5	3.3	lb/bu	30	99
Strawberry	2.0-2.9	2.4	lb/ton	1.5	4
Sweet corn	7.6-9.7	8.6	lb/ton	9	77

* Includes harvest of straw.

** This range excludes some important Wisconsin work by Saffigna et al. (1977) that reports a potato-N content of 0.28 lb/100 lb.

*** Average of early and late potato.

Fertilizer Volatilization

When ammonium or ammonium-yielding fertilizers are not incorporated, N can be lost to ammonia volatilization (Blackmer, 1987), however, evidence indicates that ammonia volatilization is not an important process in the study area. Saffigna et al. (1977) claim that ammonia volatilization will not occur on Plainfield soils due to the soil's acidity. Also, Oberle and Bundy (1987) found that 2.5 mm of rainfall within 4 days after urea application is sufficient to limit ammonia losses. For these reasons, we will assume ammonia volatilization from fertilizer is negligible.

Denitrification

Denitrification occurs when soils are waterlogged during periods when soil temperature exceeds 50°F. Most soils in the study area are unsaturated when the temperature is this warm. Saffigna et al. (1977) suggest that denitrification is generally nonexistent in these soils. In addition, Kraft et al., (1995) observed no evidence of denitrification when vegetable fields were flooded during a portion of the growing season due to heavy rains in 1993. For purposes of this N budget, denitrification losses will be assumed negligible, though it may be a factor in a few fields where soil drainage is poor.

Soil Erosion Losses and Surface Runoff

Nitrogen may be lost by erosion as soil particles are removed from fields by wind or water. We assume these losses are not significant to the total budget.

Miscellaneous Gaseous Losses

These losses include N_2O evolution during nitrification, decomposition of nitrous acid, and reactions of nitrous acid with soil minerals and organic constituents. We used the suggestion by Meisinger and Randall (1991) that these losses be approximated as 1% of the total N inputs.

Gaseous Losses from Crop Senescence and Crop Residues

Gaseous losses (ammonia and volatile amines) during plant senescence and from crop residues are poorly understood. Meisinger and Randall (1991) suggest estimates of 2-8% of total aboveground plant N for senescence. The following is an example of approximating senescence losses in sweet corn and potato. The following are assumed:

- losses are 5% of above-ground plant-N;
- above-ground plant-N for potato is 70 lb/acre (Olson and Kurtz, 1982); and
- above-ground plant-N for sweet corn, not including the harvested portion, is 92 lb/acre (Olson and Kurtz, 1982).

Calculated N losses associated with senescence are then 3.5 lb/acre for potato and 4.6 for sweet corn. Since these losses are a small part of the N budget, they will be approximated as 4 lb/acre for all crops.

Little information is available regarding possible gaseous losses of N from crop residue in this setting, but evidence indicates it is probably small. For instance, Wagger et al. (1985) recovered 100% of residue ^{15}N from incorporated sorghum and wheat residues after one growing season. Parker (1962), measuring N content of field corn residue as it decomposed on the soil surface, found that while the total residue mass decreased 50% over 140 days, the mass of N recovered in the residue actually increased slightly.

Summary of N Outputs

Table 5.3 is a summary of the total nitrogen output accounted for through harvest and various gaseous loses.

Table 5.3. Summary of N outputs (lb/acre).

Crop	Harvest-N	Gaseous loss*	Total outputs
Field corn, irrigated	109	5.5	115
Field corn, nonirrigated	66	5.2	71
Oat	75	4.6	80
Pasture			
Pea	51	4.5	55
Pickle	24	5.0	29
Potato**	160	5.9	166
Rye	63	4.6	68
Snap bean	39	5.1	44
Sod			
Sorghum	66	5.3	72
Soybean	99	4.9	104
Strawberry	4	4.5	8
Sweet corn	77	5.6	83

* Gaseous losses include N released in plant senecense and misc. gases. See text.
** Average of early and late potato.

N Loading to Groundwater

Estimates of NO_3-N leaching to groundwater from crops ($N_{c,i}$) range from 4 to 137 lb/acre-yr, depending on crop and BMP vs. CON (Figure 5.1 and Table 5.4). Of the commonly grown crops, $N_{c,i}$ was lowest for BMP soybean and highest for CON sweet corn, snap bean, and field corn. This analysis indicates that BMPs can reduce N leaching by 10-30% for the more common crops grown in the Wisconsin central sand plain. The most striking improvement is for soybean, where use of BMPs is predicted to reduce N loading from 64 to only 4 lb/acre - a reduction of 94%. Also striking is the large $N_{c,i}$ associated with sweet corn. Apparently, this is mainly due to the large amount of immature crop residue left by sweet corn (Kraft et al., 1995). When the residue decays, it releases N that is available for leaching.

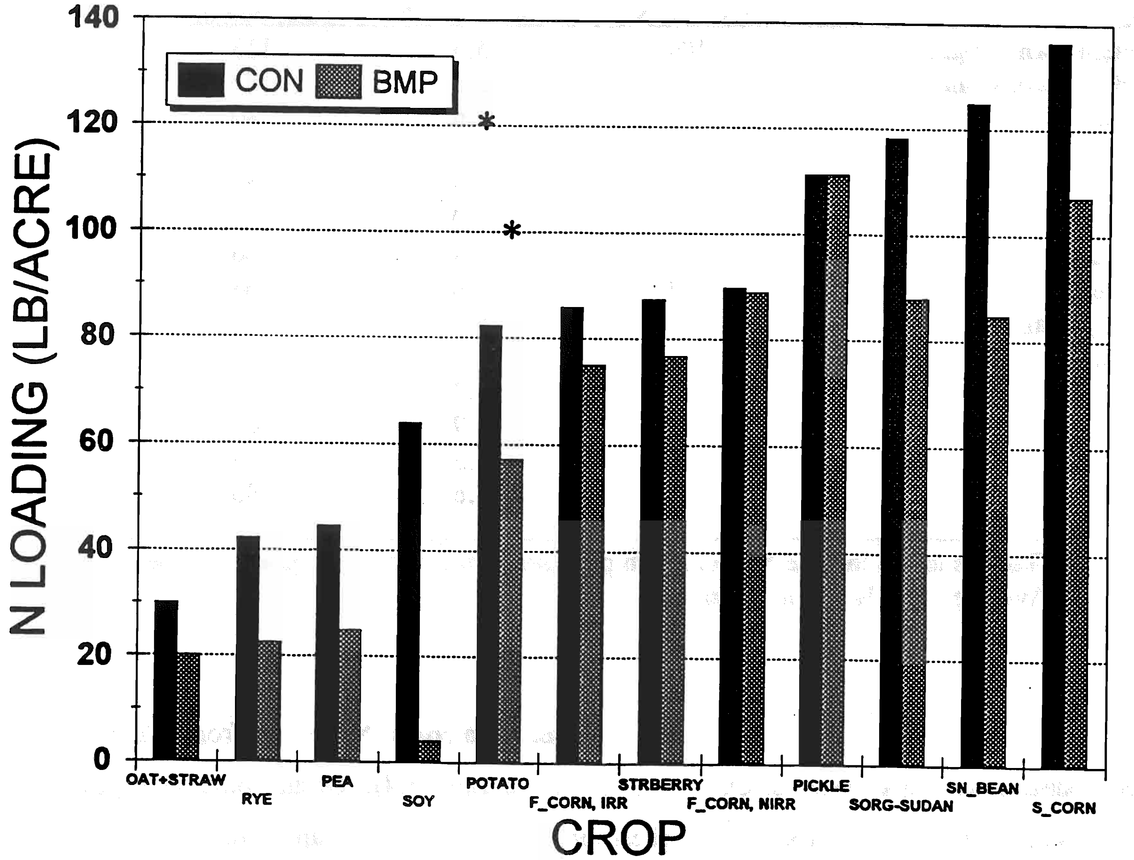

Figure 5.1 Estimated nitrogen loading for crops in the Stevens Point, Whiting, and Plover area under conventional and BMP agriculture. Asterisks denote predicted loading by potato given Wisconsin-specific data (see text).

Table 5.4. Calculated N loading to groundwater from each crop ($N_{c,i}$). Units are lb/acre.

Crop	Total inputs BMP	Total inputs CON	Total outputs	$N_{c,i}$ BMP	$N_{c,i}$ CON
Field corn, irrigated	190	201	115	75	86
Field corn, nonirrigated	160	161	71	89	90
Oat	100	110	80	20	30
Pasture**					
Pea	80	100	55	25	45
Pickle	140	140	29	111	111
Potato*	223	248	166	57	82
Rye	90	110	68	23	42
Snap bean	129	169	44	85	125
Sod***	80	85		20	21
Sorghum	160	190	72	88	118
Soybean	108	168	104	4	64
Strawberry	85	95	8	77	87
Sweet corn	190	219	83	107	137

* Average of early and late potato.

** Insufficient data available.

*** $N_{c,i}$ calculated at 25% leaching rate of total inputs (Shaw et al., 1993)

The $N_{c,i}$ calculation involved a number of components. Many components are minor, and would have little effect on $N_{c,i}$ even if the values provided are highly inaccurate. However, errors in the major components of this budget, fertilizer-N and harvest-N, could cause large errors in $N_{c,i}$. The fertilizer-N component used in this analysis is probably low for the long term average, because in wet years, growers may reapply fertilizer to replace that which they perceive leaches out of the soil. When this occurs, $N_{c,i}$ is substantially higher than the estimates provided here.

Harvest-N was calculated from the product of crop yield and N content of the harvested crop. In general, the range of N content provided by Meisinger and Randall (1991) is fairly narrow, resulting in a small error in harvest-N (Figures 5.2 and 5.3). However, potato has a large range in the estimate of N content, 0.3-0.5 lb/100 lb. Some evidence indicates that the 0.4 lb/100 lb average of this range, which was used in this study, is too high. At the Hancock experimental station, Saffigna et al. (1977) found that potato contained 0.28 lb N/100 lb, and this is consistent with the results of Kunkel et al. (1973). The use of 0.4 lb/100 lb of harvested N provided a low estimate of Nc,i compared to field measurements (Kraft et al., 1995). As a result, the $N_{c,i}$ estimate for potato (BMP - 57 lb/acre-yr; CON 82 lb/acre-yr) is probably low. More reasonable $N_{c,i}$ estimates for potato, using a harvest-N of 0.3 lb/100 lb, may be in the range of 100 lb/acre (BMP) or 120 lb/acre (CON).

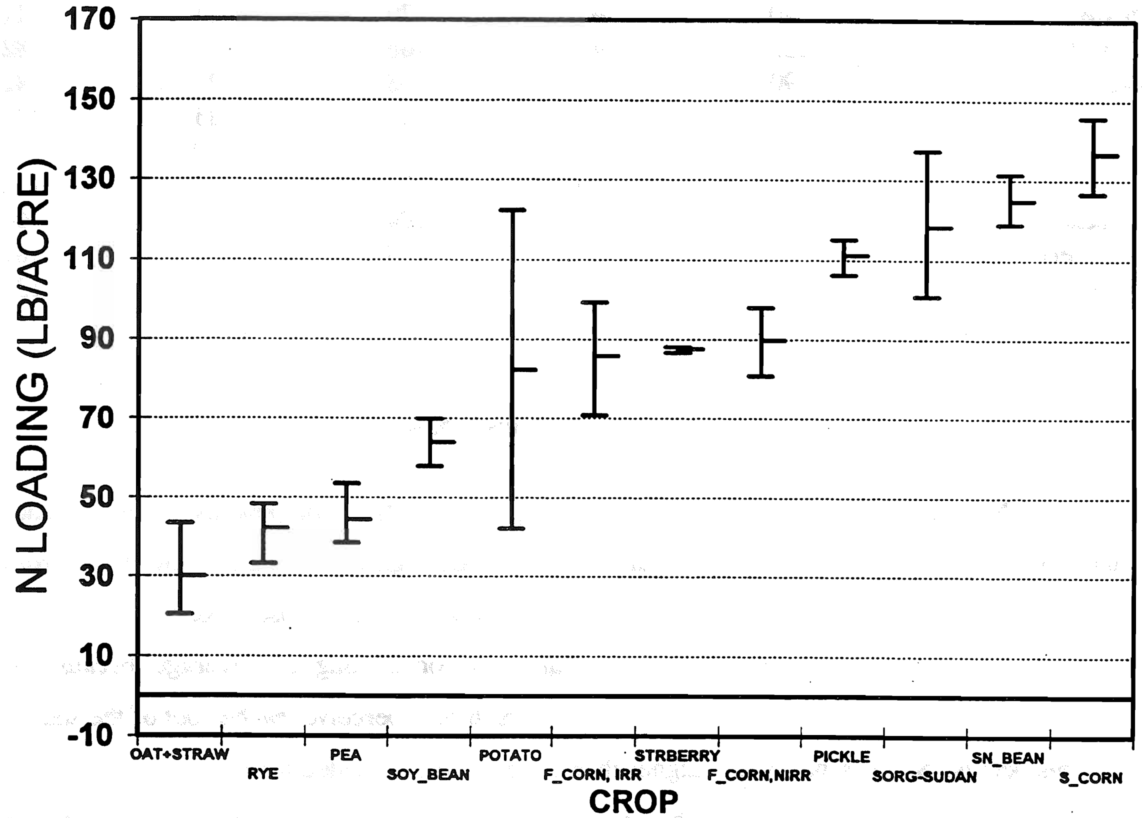

Figure 5.2 Estimated nitrogen loading average and range for crops in the Stevens Point, Whiting, and Plover area under conventional agricultural management.

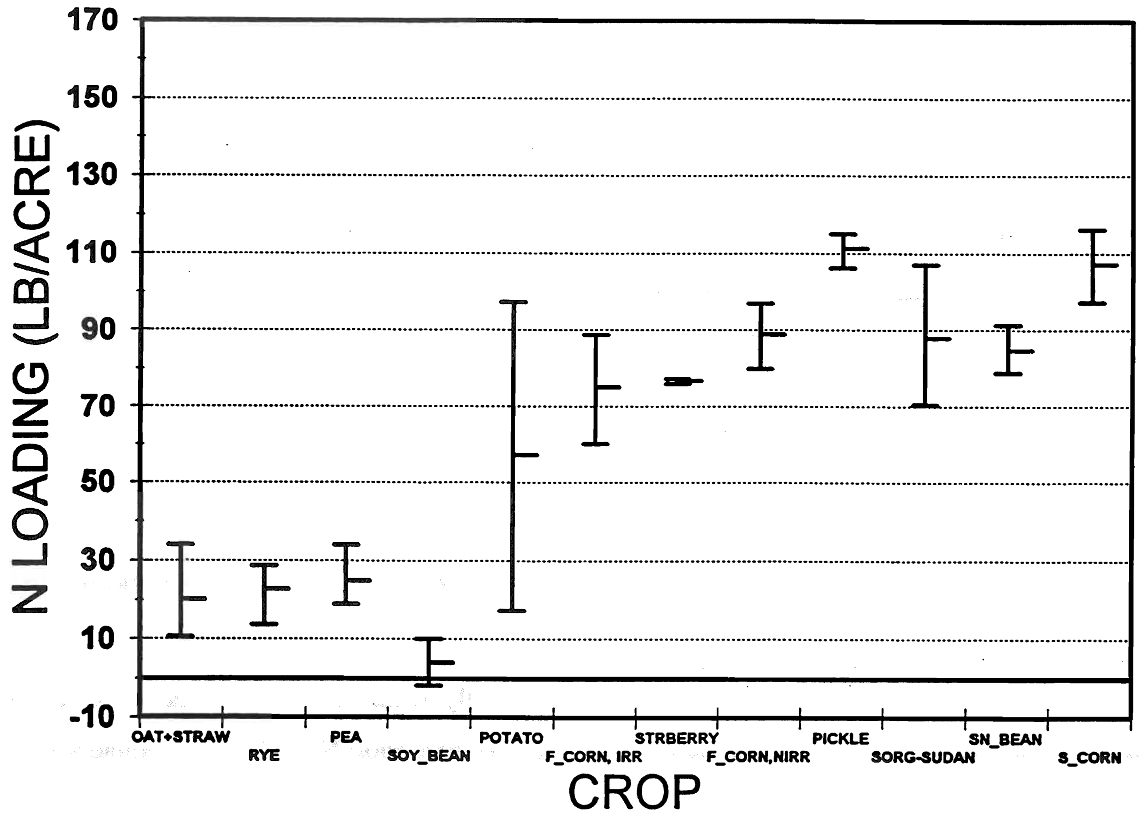

Figure 5.3 Estimated nitrogen loading average and range for crops in the Stevens Point, Whiting, and Plover area under BMP agriculture.

N LOADING FROM LEGUME FORAGES

Legume forages such as alfalfa have the ability to incorporate substantial atmospheric N into plant tissues. These forages are very nitrogen efficient, and they leach virtually no NO_3 until plowdown (Shaw and Trapp, 1993). After plowdown, organic-N in plant tissue mineralizes and becomes available for plant uptake, leaching, or transfer to other parts of the N cycle. The N loading due to legume forage plowdown (N_L; lb/yr) can be expressed as

$$N_L = N^*_{L,P} \times A_{L,P}$$

where

$N^*_{L,P}$	= N loaded by forage legume at plowdown (lb/acre)
$A_{L,P}$	= Area of legume forage plowdown per year (acres/yr)

$N^*_{L,P}$ can be estimated as

$$N^*_{L,P} = \textit{(Total N in legume forage)} - \textit{(N sinks)} - \textit{(N used by subsequent crop)}$$

Neither total N in legume forage nor N sink data are readily available for local conditions, so we approximate the difference between these as equal to UW-Extension N crediting recommendations:

$$\textit{(Creditable N)} \approx \textit{(Total N in legume forage)} - \textit{(N sinks)}$$

So that

$$N^*_{L,P} \approx \textit{(Creditable N)} - \textit{(N used by subsequent crop)}$$

Creditable N is the amount of N that can be used to replace chemical fertilizer in the subsequent crop. The UW-Extension recommended creditable N is dependent on several factors, including soil texture and legume stand condition at plowdown. For current purposes, we will use a creditable N of 110 lb/acre at plowdown.

Managers may or may not credit the full amount of creditable legume N to the subsequent crop. To allow for this, the amount of forage legume N that leaches to groundwater is

$$N^*_{L,P} = (Creditable\ N) - (Creditable\ N \times Fraction\ credited)$$

Under BMP agriculture, 100% of creditable N is credited. However, conventionally, few managers take all available legume N credits. Exo (1993) found that in the study area, only 24% of farmers took some N credit for legume forages, averaging 63 lb/acre credit. As a result, the fraction credited is only 14% that available. Using the above equation, and substituting 110 lb/acre for creditable N, 1.0 for the fraction of N credited under BMP agriculture or 0.14 for the fraction credited under conventional agriculture, the N loading for legume plowdown ($N^*_{L,P}$) is :

BMP	0 lb/acre
Conventional	95 lb/acre

Because of the uncertainty in creditable N (Total N in legume - Other N sinks), these estimates are likely low by 40-80 lb/acre of leachable N at plowdown.

N LOADING FROM MANURE

NO_3 leaching from manure occurs when organic and inorganic forms of N oxidize to the more mobile NO_3 form. We estimate N loading from manure (N_M; lb/yr) as:

$$N_M = (N^*_M) \times (tons\ of\ manure\ spread\ /\ yr)$$

where

N^*_M = (lb manure N loading / ton manure)

N^*_M can be estimated as

$$N^*_M = (Total\ manure\text{-}N) - (Other\ N\ sinks) - (Manure\text{-}N\ used\ by\ subsequent\ crops)$$

Dairy cattle produce about 12 tons of manure per animal-year, which contains about 10 lb of nitrogen per ton. Depending on storage and handling methods, 15-80% of manure N may be lost to the atmosphere. For the types of manure handling practices common in the study area, about 40% of manure-N may be lost to the atmosphere and 60% is available for plant uptake or leaching to groundwater (Meisinger and Randall, 1991). The above equation then becomes

$$N_M^* = (10 \text{ lb/ton total manure-N}) - (4 \text{ lb/ton manure-N lost to atmosphere}) - (\text{Manure N used by subsequent crops})$$

so that

$$N_M^* = (6 \text{ lb/ton}) - (\text{Manure N used by subsequent crops})$$

where

$$(\text{Manure-N used by subsequent crops}) = (\text{Creditable N}) \times (\text{Fraction of farmers crediting}) \times (\text{Fraction of manure-N credited})$$

According to UW-Extension recommendations, 3.5 lbs/ton of manure-N is "creditable" or available for crop uptake. Under BMP agriculture, 100% of manure credits are taken by 100% of farmers, so that

$$\begin{aligned} N_M^* \text{ for BMP} &= 6 \text{ lb/ton} - 3.5 \text{ lb/ton} \\ &= 2.5 \text{ lb/ton} \end{aligned}$$

However, not all farmers credit manure-N and those who do frequently don't take all available credits. In the study area, 43% of farmers who handle manure in the study area take some manure-N credit (Exo, 1993), however, no study area specific data on the fraction credited is available. Shepard

(1991) found that in Wisconsin, farmers who took manure credits took only 51% of available credit. Combining these data indicates only 22% of credits are taken. Therefore,

$$\textit{(Manure-N used by subsequent crops)} = 3.5 \text{ lb/ton} \times 0.43 \times 0.51$$

and

$$\begin{aligned} N_M^* (CON) &= (6 \text{ lb/ton}) - (3.5 \text{ lb/ton} \times 0.43 \times 0.51) \\ &= 5.2 \text{ lb/ton} \end{aligned}$$

CHAPTER 6

PREDICTED NITRATE CONCENTRATIONS AT THE MUNICIPAL WELLFIELDS

Predicted steady-state nitrate concentrations for the municipal wellfields are calculated in this chapter using the methodology and loading information detailed in Chapter 5 along with land use, crop census, and water budget data.

We used the 30 year time-of-travel (TOT) zone as the basis for calculating nitrate loading to precipitation-recharged groundwater pumped by the municipal wellfields (Figures 4.2 and 6.1). We chose to work with the 30 year TOT instead of the entire zone-of-contribution for three reasons. First, the 30 year TOT is more workable in size, and 30 years is an appropriate planning horizon. Second, zone-of-contribution delineations become less accurate with distance from wells (Born et al., 1988). In the present case, the 30 year TOT is mapped with good accuracy, but less accuracy is likely associated with, say, the 75 year TOT. Third, land uses within the 30 year TOT are representative and therefore scalable to the entire zone-of-contribution. Analysis of the area/TOT relationship indicates that the 30 year TOTs represent 62-71% of the total zones-of-contribution, and generally reflect a breakpoint of diminishing additional area relative to time-of-travel (Figure 6.2).

LAND USES AND MANAGEMENT IN 30 YEAR TOTs

As discussed in Chapter 2, land uses and management were identified for the SWP area and mapped into the Portage County geographic information system (GIS). The 30 year TOT designations (Chapter 4) were exported to the GIS, where an overlay allowed precise tabulation of land uses and management for the selected zones-of-contribution (Figure 6.3).

Land uses for the 30 year TOT are summarized in Table 6.1. The Stevens Point main 30 year TOT contains 31% irrigated agriculture, 33% forest, and the remainder spread over various other uses. Land uses within the Stevens Point #5 30 year TOT are dominantly residential(33%) and dryland agriculture (27%). Whiting and Plover 30 year TOTs contain approximately 50% irrigated agriculture. Whiting is secondarily influenced by residential/urban uses (28%) while Plover has 36% in forest.

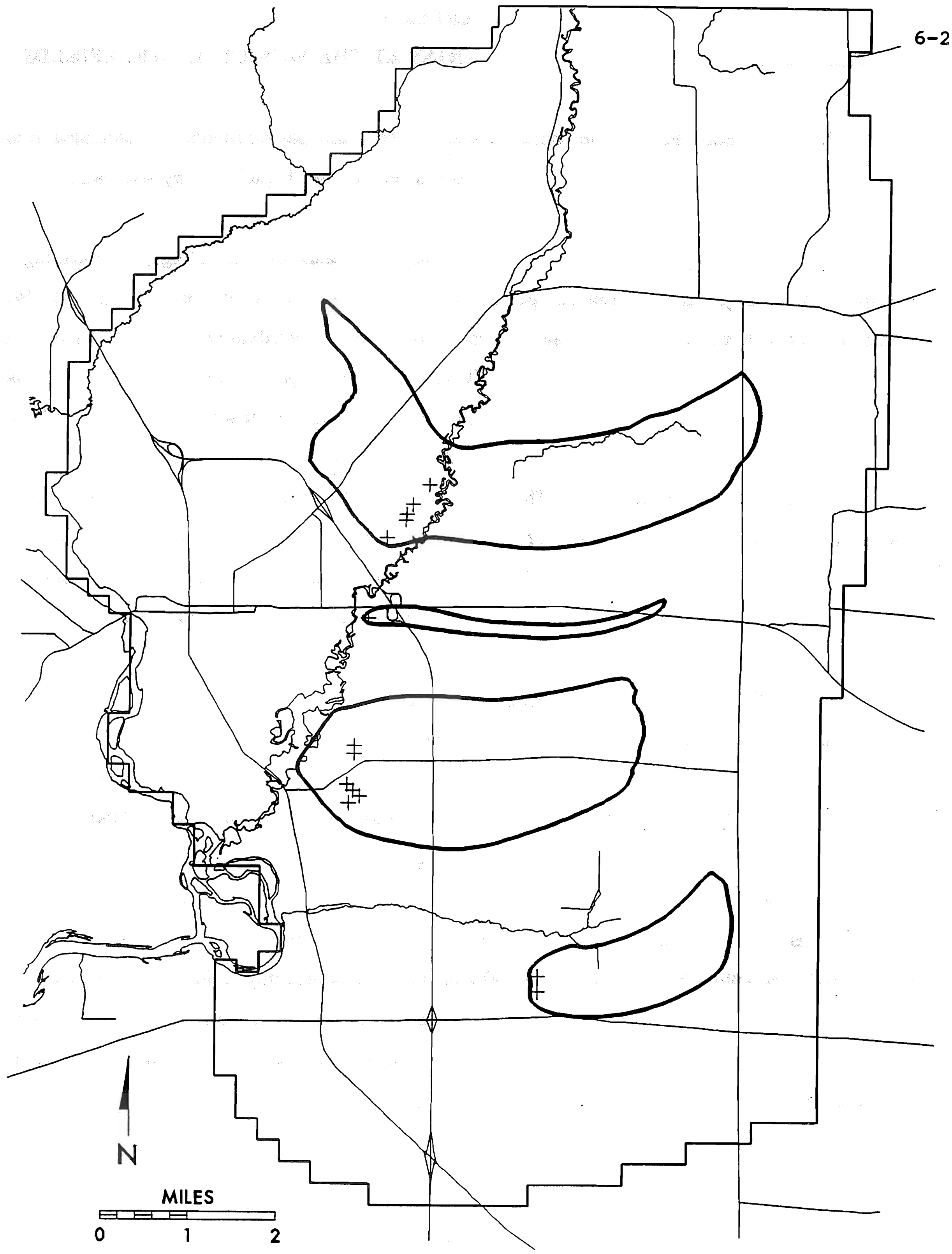

Figure 6.1 Thirty year zones-of-contribution for the Stevens Point, Whiting, and Plover municipal wells.

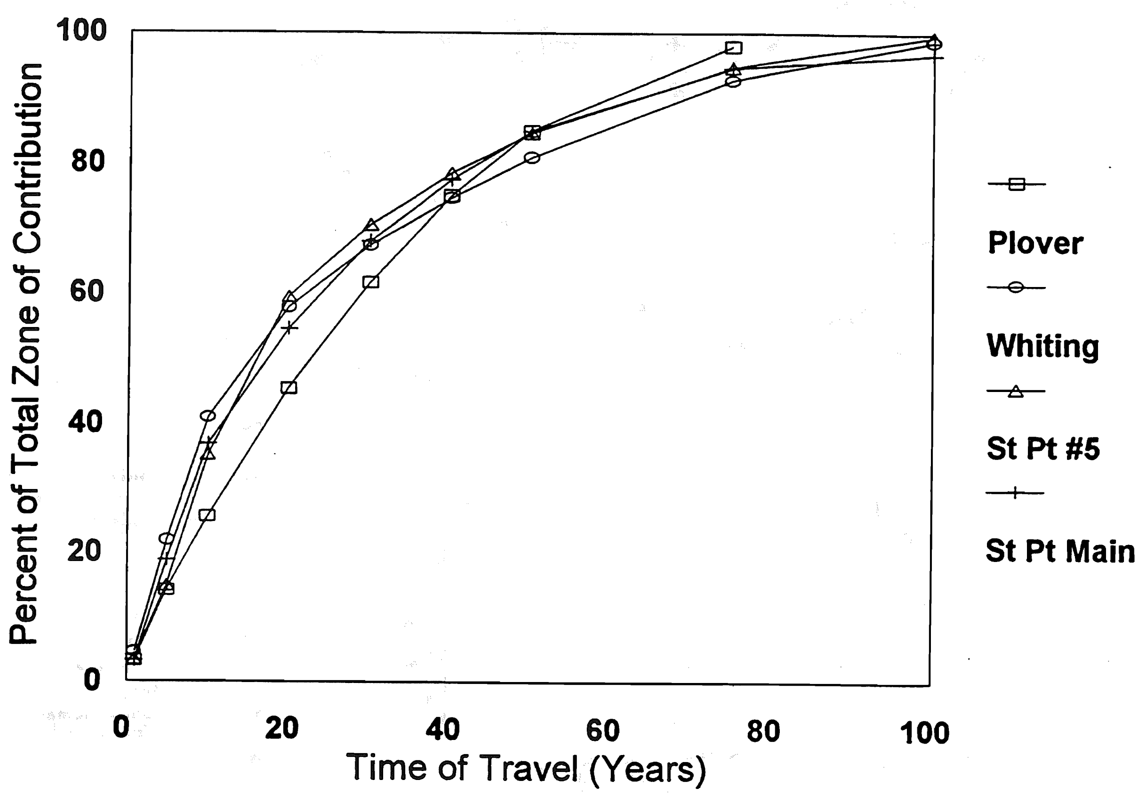

Figure 6.2 Relationship between zone-of-contribution and time-of-travel for the Stevens Point, Whiting and Plover municipal wells.

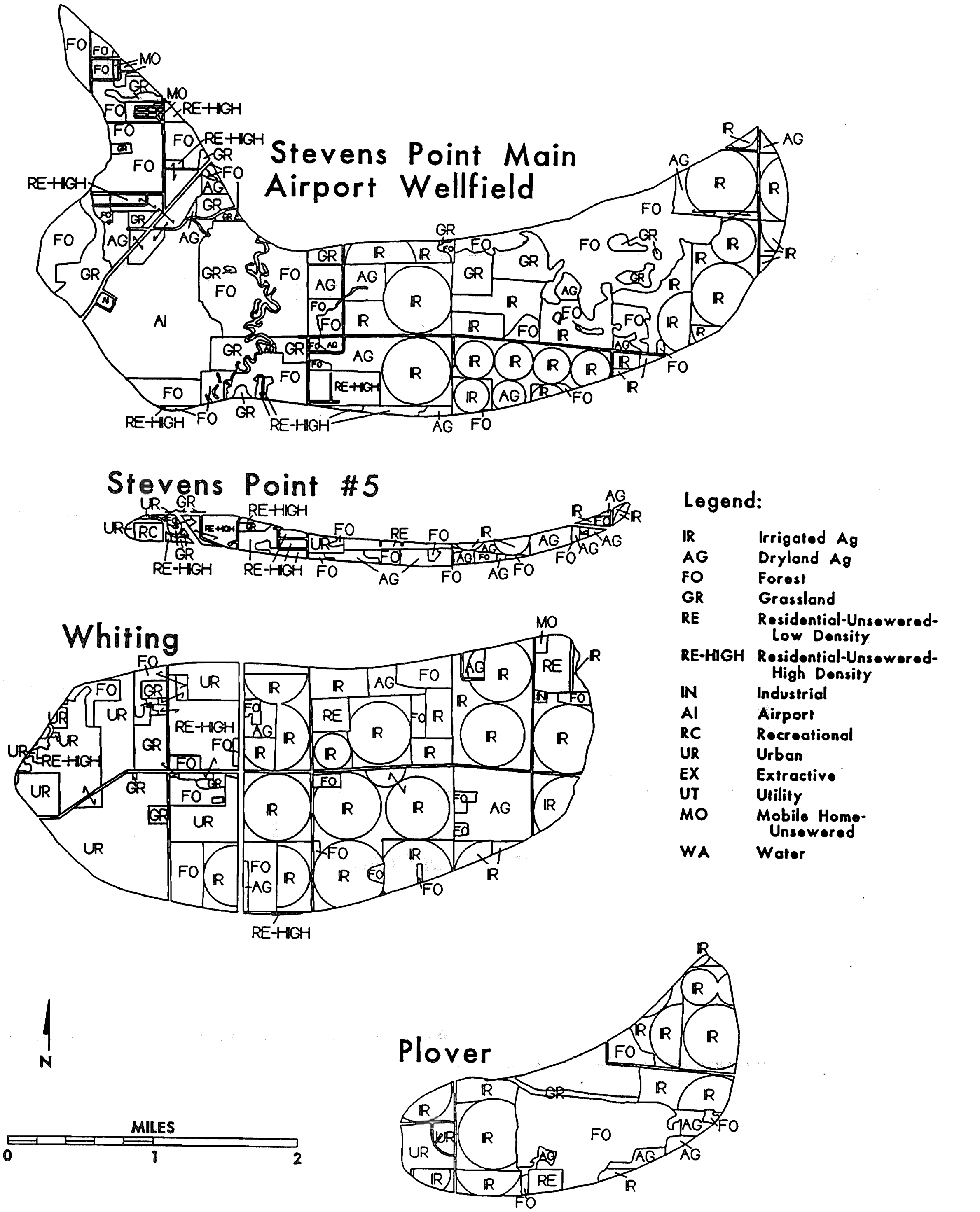

Figure 6.3 Land use in the 30 year zones-of-contribution for the Stevens Point, Whiting, and Plover municipal wells.

Table 6.1. Summary of land use in the 30 year time-of-travel zone-of-contribution (acres and %total).

	St Pt - Main	St Pt - #5	Whiting	Plover	Total
Irrigated Ag	1327 31%	29 9%	1799 51%	658 50%	3813 40%
Forest	1395 33%	46 14%	389 11%	476 36%	2307 24%
Dryland Ag	397 9%	90 27%	246 7%	64 5%	798 8%
Residential-Unsewered	220 5%	112 33%	394 11%	24 2%	751 8%
Urban-Sewered	28<1%	24 7%	619 17%	70 5%	740 8%
Grassland	391 9%	17 5%	89 2%	22 2%	520 6%
Airport	430 10%	0	0	0	430 5%
Surface Water	38<1%	0	12<1%	0	50<1%
Recreational	0	16 5%	0	0	16<1%
Mobile Home-Unsewered	0	3<1%	6<1%	0	9<1%
Industrial	0	0	5<1%	0	5<1%
Utilities	0	0	2<1%	0	2<1%
Total	4226	337	3562	1315	9441

Analysis of the agricultural land uses by USDA SWP Wellhead Protection Project staff provided an average crop census and historical data on crop rotations. Potato, snap bean, field corn, sweet corn, and hay are the 5 principal crops grown in the recharge areas (Table 6.2). Crops are rotated in 2 to 10 year rotations, with 2 to 5 year rotations most common. The most common crop rotations are potato/sweet corn, field corn/snap bean, and potato/field corn/snap bean. Thirty specific dryland and irrigated crop management classes were identified within the 30 year TOTs, and a tabulation of areas for each made using the 30 year TOT overlay in the Portage County GIS.

Table 6.2. Average crop census for the 30 year TOT areas (acres).

	StPt-Main	StPt-#5	Whiting	Plover	Total
Potato	432	3	438	136	1009
Snap Bean	235	3	466	192	896
Field Corn-Irrigated	183	6	452	132	774
Sweet Corn	258	6	245	118	627
Hay	253	36	252	40	580
Field Corn-Dryland	124	35	92	20	271
Oat+Straw	134	29	39	8	210
Sod	92	0	0	0	92
Pea	14	1	31	22	68
Rye	0	0	0	55	55
Sorghum	0	0	29	0	29
Total	1724	119	2045	723	4611

NITRATE LOADING TO PRECIPITATION-RECHARGED GROUNDWATER

To review Chapter 5, the predicted average steady-state nitrate-N concentrations for precipitation-recharged groundwater in 30 year TOTs can be expressed as

$$[NO_3\text{-}N]_{P,SS} = \frac{N_T}{R_P}$$

where N_T is the mass of NO_3-N loaded to groundwater annually from the surface of the recharge area and R_P = volume of annual precipitation-recharged groundwater. N_T has components of loading from crops, legume forage, residential, and manure. The crop and legume components were calculated as follows. The average annual crop and legume forage census (acres of crop) for the irrigated and dryland agriculture land uses within the 30 year TOTs was multiplied by the crop nitrate-N loading rate (Table 5.4) or forage legume loading rate for both conventional and best management practices. The residential component was determined by multiplying the loading rate for the various residence types times their area within the 30 year TOTs. Manure loading was calculated slightly differently, from the mass of manure spread within the individual 30 year TOTs (tons) times the nitrogen loading per mass (lb/ton) for both conventional and best management practices. Manure tonnages were 1350, 375, 11310, and 65 tons respectively for Stevens Point main wellfield, Stevens Point #5, Whiting, and Plover (Ebert, 1995).

Loading results for the four major nitrate loading land uses are presented in Table 6.3 (note: manure loading is neglected for this table). Loading rates varied substantially among the land uses, from 8 lb/acre-year (urban - sewered) to about 100 lb/acre (irrigated agriculture). The loading for an individual land use frequently differed among the various TOTs depending on the mix of crops, rotations, or unsewered residential mix prevailing within the TOT. Agricultural land use loading also depended if practices were conventional (CON) or best management practices (BMP).

Table 6.3. Average annual nitrate-N loading for various land uses (lb/acre) for the 30 year times of travel.

	StPt-Main	StPt-#5	Whiting	Plover	Overall
Irrigated Ag (IR)*	91 (65)	84 (48)	100 (71)	100 (74)	97 (69)
Dryland Ag (AG)*	64 (39)	71 (43)	61 (36)	61 (30)	63 (38)
Residential-Unsew (RE)	48	43	41	12	43
Urban-Sewered (UR)	8	8	8	8	8

( ) Average with BMPs. * Manure loading not included.

WATER BUDGET FOR MUNICIPAL WELLFIELDS

Precipitation is the major source of recharge for groundwater pumped by most of the municipal wellfields. Precipitation recharge for the Stevens Point #5, Whiting, and Plover wellfields was determined using the recharge rates specified in the MODFLOW flow model (Figure 6.4, Table 6.4).

In addition to precipitation recharge, surface water recharge also needs to be considered for the Stevens Point main wellfield, because a considerable amount of pumped water there originates as induced recharge from the Plover River. Groundwater flow to this wellfield has to be apportioned as to whether it originates from east or west of the river, or directly from the river. This is not straightforward, since the current flow model does not allow a direct quantitative apportionment. However, the model provides sufficient information to allow a reasonably accurate apportionment, as follows. The total modeled wellfield pumpage is 6.72 cfs. It may be safely assumed that all water originating from the entire zone-of-contribution west of the river (1.80 cfs) is captured by the wellfield. The model also shows 13 losing cells for the Plover River in the vicinity of the wellfield. We further assume that this water (4.33 cfs) is captured by the wellfield. The remaining pumpage (0.59 cfs) is assumed to originate from the east zone-of-contribution. We are unable to say where on the east side of the river this 0.59 cfs originates. In reality, the patterns of groundwater recharge and discharge from the river are complex. Complicating factors such as seasonal scouring and deposition as well as pumping schedules confound the task of apportioning groundwater origin. While this apportionment estimate (27% from the west, 64% from the river, and 9% from the east) is approximate, the low concentrations of nitrate in wellfield water are supportive.

The Whiting wellfield may also receive induced recharge. Particle track modeling indicated flow paths from Whiting's wellfield extend to McDill Pond. The volume of induced recharge is minor compared to Stevens Point main. The maximum induced recharge component is less than 20%, and may be considerably less, depending on permeability of McDill Pond sediments. As a result, we neglected induced recharge for the Whiting wellfield for nitrate loading and concentration calculations, and revisit the issue when discussing uncertainties and sensitivities.

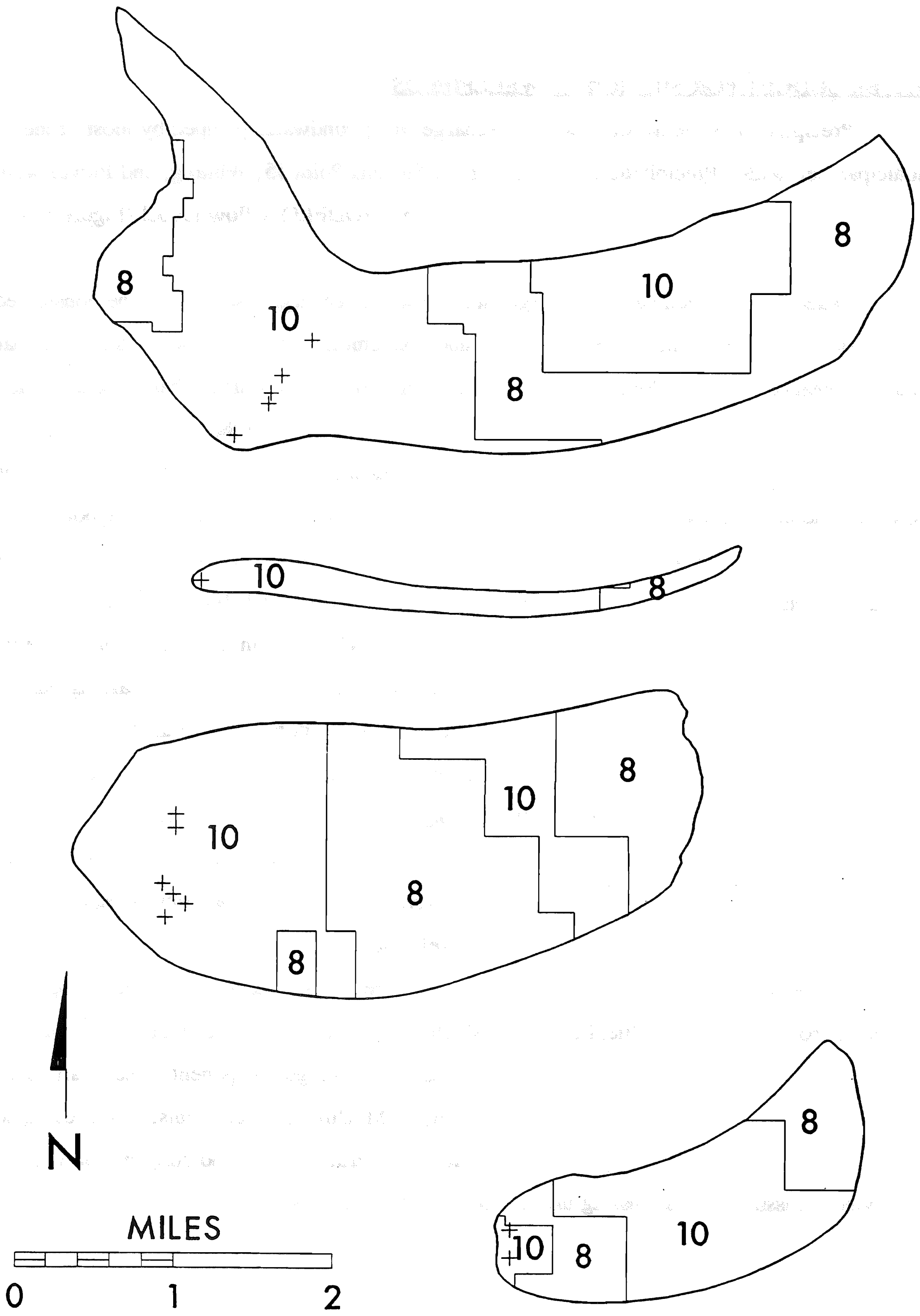

Figure 6.4 Groundwater recharge rate (inches/year) in the 30 year zones-of-contribution for the Stevens Point, Whiting, and Plover municipal wells.

Table 6.4. Water budget for 30 year zones-of-contribution for concentration calculations.

	StPt Main West	StPt Main River	StPt Main East	StPt #5	Whiting	Plover
Area (acres)	1526		2701	337	3562	1315
Recharge (in/yr)	8.8		9.1	9.6	9.0	9.2
Vol Rech (R_T) (cfs)	1.55	4.33	2.81	0.37	3.69	1.40
Apportionment*	27%	64%	9%			

* Apportionment factors for calculating nitrate contribution to Stevens Point-main wells (see text).

NITRATE CONCENTRATIONS AT MUNICIPAL WELLS

As developed in Chapter 5, the predicted steady-state nitrate-N concentration at the municipal wells can be calculated as

$$[NO_3\text{-}N]_{SS} = \frac{Q_P \times [NO_3\text{-}N]_{P,SS} + Q_S \times [NO_3\text{-}N]_S}{Q}$$

where

Q_P = wellfield pumpage originating from precipitation-recharged groundwater

$[NO_3\text{-}N]_{P,SS}$ = average steady-state NO_3-N concentration in precipitation-recharged groundwater

Q_S = wellfield pumpage originating from surface water recharged groundwater

$[NO_3\text{-}N]_S$ = average NO_3-N concentration in surface water recharged groundwater

Q = total wellfield pumpage ($Q_P + Q_S$)

For all but the Stevens Point main wellfield, the surface water components are zero or are treated as neglectable. (Whiting will be considered later in a sensitivity analysis.) In these cases, predicted steady-state nitrate-N concentrations at the municipal wells are simply equal to the average nitrate-N concentration for precipitation-recharged groundwater in the 30 year TOT ($[NO_3\text{-}N]_{P,SS}$). The nitrate concentration for the Stevens Point main wellfield was calculated as the weighted average (based on source apportionment of pumpage) of the Plover River nitrate concentration and the separately calculated steady-state concentrations for the east and west zones-of-contribution. Plover River nitrate concentrations have historically been low because much of the land use in the watershed in northern Portage County and in Marathon County to the north is wood and wetlands. Renaud (1987) measured Plover River nitrate concentrations in the Stevens Point area on 6 dates in 1984-1985. The values ranged from 0.4 to 2.0 and averaged 1.2 mg/l N. This average concentration will be used for loading calculations.

A mass balance model was developed for these data, incorporating the formulas presented in Chapter 5 into a spreadsheet format. The mass balance model may be used by county and municipal groundwater managers in the future to facilitate "what if" analyses. The loading formulas included variables for base loading rates for general land uses and individual crops, adding fallow years to rotations, manure crediting, and global loading multiplication factors for irrigated ag, dryland ag, unsewered residential, and urban categories.

Predicted nitrate concentrations under current conditions

Predicted steady-state nitrate-N concentrations range from 4.0 (Stevens Point main) to 37.8 mg/L (Whiting), for current conditions and conventional agricultural practices (Table 6.5 and Figure 6.5), with all wellfields except Stevens Point main exceeding the federal MCL of 10 mg/L. The low nitrate concentration in the Stevens Point main wellfield is due to the large amount of river water induced by pumpage. This wellfield would also exceed the MCL if it was largely dependent only on precipitation-recharged groundwater. Assuming 100% of agricultural lands are managed using Best Management Practices (BMPs), the nitrate-N range is reduced to 3.1 to 25.7 mg/L, with all wellfields except Stevens Point main still in excess of the MCL. Achieving even the modest groundwater improvements possible with BMPs will be difficult, since watershed projects are rarely successful in sustaining even 50% participation from farmers (Shepard, pers. comm., 1995).

Table 6.5. Predicted steady-state nitrate-N concentrations (mg/l) under existing land uses.

	Conventional Ag	BMP Ag
StPt-Main Net	4.0	3.1
(West	2.8	2.4)
(River	1.2	1.2)
(East	28.1	19.7)
StPt-#5	21.7	15.5
Whiting	37.8	25.7
Plover	25.9	18.7

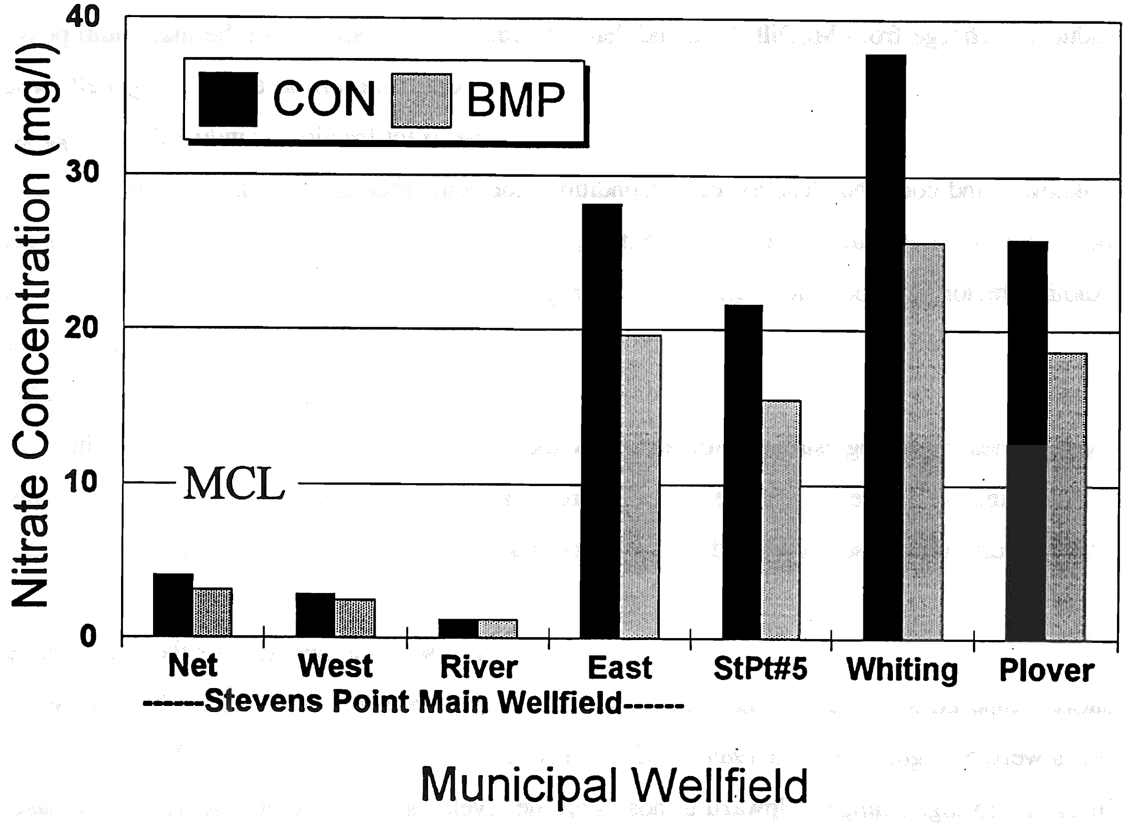

Figure 6.5 Predicted steady-state nitrate-N concentrations under existing land use.

COMPARISONS TO CURRENT CONDITIONS

All predicted nitrate concentrations exceed current concentrations at the municipal wellfields. This suggests that either the model contains some inaccuracies, or that nitrate concentrations will increase over time as groundwater quality comes into equilibrium with prevailing land uses.

For the Whiting wellfield, nitrate concentrations have clearly increased over time (Figure 1.2), from about 4 mg/L nitrate-N in the early 1960s to about 19 mg/L at present. Recent data continue to suggest an upward trend. Factors which would decrease predicted concentrations include induced recharge from McDill Pond and denitrification in the aquifer. At the maximum possible induced recharge fraction of 20%, predicted nitrate-N concentrations at the Whiting wells would lower to about 30.5 mg/L (CON) and 20.8 mg/L (BMP). The exact fraction of induced recharge is unknown, and could be close to zero. Conditions for denitrification in soils and groundwater appear to be absent in the Whiting zone-of-contribution, but could exist, though this possibility seems remote. Another factor that could account for the discrepancy between predicted and current concentrations is that the predicted concentration is for the entire wellfield, containing 5 industrial wells and one municipal well, whereas the measured concentrations are only for the municipal well. Finally, inaccuracies in loading estimates for the crops and land uses may affect the predicted nitrate concentrations. However, these estimates were generally on the low side, further refinements would likely result in increased predicted nitrate concentrations. In sum, the model predictions appear valid at this time.

The trend in the Village of Plover wellfield has also been upward, but the period of record is short compared to Whiting. The Village wells were in the 8.5-9.5 mg/L nitrate-N range when the wells were brought on-line in 1989. They exceeded the 10 mg/L standard in 1993, and are currently in the 11-15 mg/L range. Upward trends in nitrate levels have also been observed in the baseflow of the Little Plover River, located near the Whiting and Plover wells (Figure 1.3). The potential for unaccounted dentrification or inaccuracies in nitrogen loading predictions to affect predicted nitrate concentrations appear to be the same as for Whiting. Again, in sum, model predictions appear realistic at the present time.

The Stevens Point #5 well is currently around 5 mg/L nitrate-N, considerably less than the predicted 15.5 to 21.7 mg/L. While the nitrate concentration appears to be slowly increasing, up from 3.5 in the late 1960's, the predicted concentrations are still quite high in comparison. Several factors may explain this discrepancy. First, well #5 is considered standby capacity, especially with well #10 on-line, and is not pumped consistently at the modeled rate so that a scenario approximating a steady-state may never develop. Second, an inaccuracy in the model may neglect actual induced recharge from the river which would tend to dilute the nitrate. For example, if the average daily flow used in the model was actually intermittent pumpage at much higher rates, induced recharge from the river is more likely. Finally, the nitrate loading estimate for residential land uses is likely high compared to reality. Since the Stevens Point #5 well has a fair amount of this land use in its recharge area, a more realistic estimate might decrease the predicted concentrations by a few mg/L. From a qualitative standpoint, induced recharge seems like it might be responsible for a larger part of the discrepancy, followed by higher than actual estimates for the unsewered residential land use. Model predictions appear to be high given the current use being made of the Stevens Point #5 well.

For the Stevens Point main wellfield, measured values are congruous with modeled nitrate concentrations. Current nitrate-N values are generally in the 1.5 to 2.5 mg/L range, with no apparent trends. Well #7 has fluctuated up to about 5 mg/L. Nitrate-N levels in the Stevens Point distribution system have fluctuated from 1 mg/L in 1975 to 2.2 in 1990, but the origin of the nitrate is unknown. Predicted nitrate-N concentrations for the area east of the river (28.1 and 19.7 for conventional and BMP management, respectively) are high, but are they realistic? The limited number of monitoring wells located along the eastern flank of the Plover River near the wellfield do not support these high predicted values, however, these wells are too sparse to be conclusive. Extensive private well monitoring data for this area is inconclusive in establishing the predicted values as being realistic. The data do indicate elevated nitrate levels are common, with over one third of the analyses exceeding the MCL of 10 mg/L N. The area east of the river of all the zones of contribution is most likely to have a possibility for denitrification in soils and groundwater, due to some large areas of organic soils.

SOURCES OF NITRATE TO THE MUNICIPAL WELLS

The relative contribution of the various land uses to nitrate loading varies by wellfield (Figure 6.6). For Plover 93-94% (CON-BMP) is from irrigated agriculture, which accounts for 50% of the land use. Even if all nitrate sources other than irrigated agriculture are completely eliminated, predicted steady-state nitrate-N concentrations would only be reduced by about 1.1 mg/L (BMP scenario). Nitrate reduction strategies for Plover must therefore target irrigated agriculture.

Whiting is strongly impacted by agriculture, which is responsible for about 92-89% (CON-BMP) of nitrate loading. Irrigated cropland, comprising 51% of the total land use, contributes 65-69% of the loading, dryland agriculture contributes 5%, and manure contributes 22-15%. Unsewered residential use also contributes 6-9% of the loading. Eliminating all the non-agricultural loading only

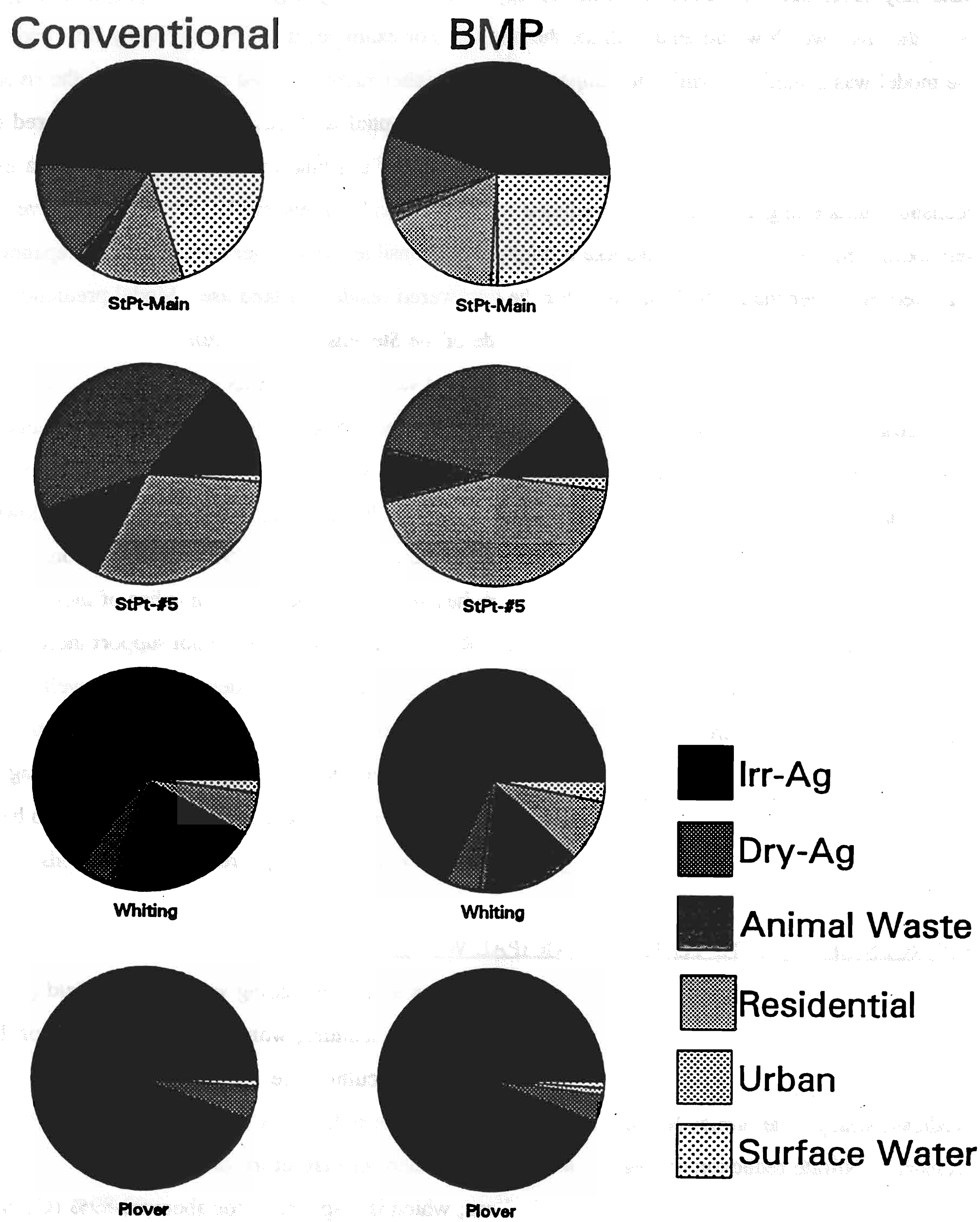

Figure 6.6 Nitrate loading sources for the Stevens Point, Whiting, and Plover municipal wells under conventional and BMP agriculture.

reduces nitrate concentrations by 2.9 mg/L (BMP scenario). Management strategies must principally target irrigated agriculture and manure handling, and to a much lesser extent unsewered residential and dryland agriculture.

The major loading for Stevens Point #5 is from both agriculture (67-54%) and unsewered residential (31-44%). The agricultural components are dryland agriculture (40-34%), irrigated agriculture (15-12%), and manure (12-8%). The predicted steady-state nitrate concentrations are lower than for Whiting and Plover because the dryland agriculture and residential uses tend to have a lower per acre loading than the irrigated crops more common to Whiting and Plover. Management needs to first focus on dryland agriculture and unsewered residential uses, and secondarily on irrigated agriculture and manure.

Water quality in the Stevens Point main wellfield is largely influenced by the Plover River. The models indicate the river contributes 64% of wellfield water but only 20-25% of the nitrate loading. An important management goal for Stevens Point is to ensure nitrate concentrations in the Plover River remain low. Should the Plover River concentration rise to 12 mg/l, models predict that wellfield pumpage would reach the MCL of 10 mg/l N.

Also, the City must consider that increased pumping rates may shift the proportion of water coming to the wells from the east side of the river, which has much higher predicted nitrate-N concentrations (28.1-19.7 mg/L). If the assumed percentage of pumpage from the Plover River drops from 64% to 27%, and the percentage from the eastern recharge zone rises from 9% to 46%, the overall wellfield concentration rises to the MCL. Nitrate loading east of the river is dominated by agricultural land uses, irrigated agriculture contributes 78-80%, dryland agriculture contributes 15-13%, and manure contributes 5-3%. As for Plover and Whiting, management strategies would need to target these land uses should wellfield pumpage capture additional flow from this area.

The Stevens point main zone-of-contribution west of the river has a predicted steady-state nitrate-N concentration of 2.8-2.4 mg/L due to the high proportion (87%) of land in low nitrate loading land uses (forest, airport, and grassland). Principal nitrate loading sources are unsewered residential (65-78%) and dryland agriculture (32-19%).

POTENTIAL MANAGEMENT SCHEMES FOR REDUCING NITRATE CONCENTRATIONS

Depending on public policy and groundwater quality goals, the modeling process indicates that a nitrate loading reduction may be needed in four areas; the zones-of-contribution for Stevens Point main-east, Stevens Point well #5, Whiting, and Plover. Assuming a hypothetical goal of reducing average nitrate-N concentrations to the MCL of 10 mg/L, what strategies might attain this goal?

Universal use of agricultural BMPs could theoretically reduce nitrate concentrations by about 30% (Table 6.6). However, universal adoption would be difficult to achieve; BMP adoption rates are rarely higher than 50% in watershed projects (Shepard, pers. comm., 1995). Even assuming universal adoption, predicted nitrate-N concentrations still lie in the 15.5-25.7 range (Table 6.6) in the four areas.

Table 6.6. Some comparisons for evaluating potential management schemes.

Area		Predicted Nitrate-N Concentrations (mg/l) CON	CON-remove non-ag N	BMP	BMP-remove non-ag N
Stevens Point	Main	4.0	3.4	3.1	2.5
	West	2.8	0.9	2.4	0.5
	East	28.1	27.2	19.7	18.8
Stevens Point #5		21.7	14.7	15.5	8.5
Whiting		37.8	34.8	25.7	22.8
Plover		25.9	25.6	18.7	18.4

Eliminating all nonagricultural nitrate loading could be accomplished by banning nonagricultural fertilizer use, extending public sewers, and requiring nitrate removing septic systems. This approach has potential only in the Stevens Point #5 zone-of-contribution, which would decrease the predicted nitrate-N concentration from 15.5 to 8.5 mg/L (BMP scenario). In the other three areas, nonagricultural loading is only about 10%of the total, so eliminating this source drops nitrate concentrations only 0.3-3 mg/L (Table 6.6).

Reductions in agricultural loading

Since agriculture is the dominant nitrate loading land use, contributing 89-85% (CON-BMP) of the loading to the municipal wells, any attempt to meet the hypothetical goal must consider reducing the nitrate loading beyond the BMP. Agricultural loading would have to be further reduced by 53% in the Stevens Point main - east, 66% in Stevens Point #5, 69% in Whiting, and 47% in Plover.

Given current cropping practices and land uses, loading can be reduced by either decreasing nitrogen inputs or increasing nitrogen outputs. Inputs might be reduced if recommended N credits for manure and forage legumes were increased (Minnesota recommends greater credits than does

Wisconsin). However, even a 50% increase in manure credits would reduce overall agricultural loading by only about 7%. A closer look at existing data by University research might reveal that nitrogen recommendations could be reduced somewhat beyond BMPs without affecting yields. Perhaps nitrogen outputs could be increased, by utilizing more of the crop residue that is presently left in the field. Utilization of sweet corn residue as silage, for instance, would significantly increase N outputs.

Technical advances in the future may help if research is guided in that direction. Nutrient scavenging crops grown after the main crop could be used to hold nutrients for the subsequent crop. This is currently being investigated in the Hancock Agricultural Research Station, but practical application is many years away. New research may indicate that nitrogen use could be trimmed back somewhat. Also, deploying irrigation wells in a way to recycle nitrogen in irrigation water may prove useful. Finally, better nutrient utilizing crops may have potential in the long term.

Changes in land uses could be used to reduce nitrogen loading sufficiently to meet the hypothetical goal. For instance, in the Plover zone-of-contribution, 49% of irrigated land could be placed in a low nitrogen loading land use, such as forest or grassland. Alternatively, 59% of irrigated land could be placed in 4 acre lot unsewered residential use.

Changes in crops is another mechanism. Increasing acreages of soybean, pea, and hay at the expense of higher leaching crops may have utility. For example, under BMP management, adding 4 soybean years to the main irrigated crop rotations in the Plover zone-of-contribution (potato/corn/snap bean, sweet corn/snap bean/corn/rye, potato/pea/sweet corn/snap bean, potato/sweet corn/snap bean, 2potato/sweet corn/snap bean, and sweet corn/snap bean) meets the MCL goal for Plover.

CONCLUSIONS

The combination of detailed land use and management data, nitrate loading estimates for various land uses, and zone-of-contribution delineations allowed the development and implementation of tools to predict future steady-state nitrate concentrations at the municipal wellfields. The tools are available and can be used to evaluate the efficacy of nitrate reduction strategies in the future.

Predicted concentrations for the municipal wells are partly dependent on whether conventional (CON) or best management practices (BMP) are assumed for the agricultural land use. Predicted steady-state nitrate-N concentrations at the municipal wellfields are mostly higher than the nitrate MCL. Predicted concentrations are: Plover - 25.9 mg/L (CON) and 18.7 mg/L (BMP); Whiting - 37.8 and 25.7; Stevens Point # 5 - 21.7 and 15.5; and Stevens Point main - 4.0 and 3.1.

Predicted concentrations are substantially higher than current for all but the Stevens Point main wellfield. This is due to the groundwater system not being at steady-state, and possibly to model error. Current water quality data suggest that the predictions for the Plover, Whiting, and Stevens Point main wellfields are plausible, and that the predictions for Stevens Point #5 may be high. One potential model error common to all wellfields is the assumption that denitrification is not occurring in soils or groundwater. We know of no evidence which indicates denitrification is likely, while substantial evidence suggests that denitrification is generally absent, with the possible exception of the Stevens Point main zone-of-contribution. No obvious sources of model error are available to lower predicted concentrations at the Plover well. For the Whiting well, the major source of potential error is neglecting potential induced recharge, which might result in overestimating nitrate-N concentrations by a few mg/L. Of all wellfields, Stevens Point #5 has the largest discrepancy and appears least likely to attain predicted concentrations. This may be due to the way the well is used currently compared to the model, and interactions with the Plover River. Nitrate loading estimate (Chapter 5) errors would be more likely to increase predicted concentrations than to lower them.

The importance of specific sources of nitrate loading varied among wellfields. For the combined zones-of-contribution, the nitrogen loading sources (CON scenario) were 66% irrigated agriculture, 15% manure, 8% dryland agriculture, 7% unsewered residential, 2% surface water, and 1% sewered urban. The combined agricultural loading accounts for 89% of the total loading.

The mass balance model was useful for "what if" analysis of potential wellhead protection management efforts. To make significant progress towards reaching the MCL, dramatic changes are needed (in addition to reaching full BMP participation), possibly including: reducing agricultural inputs below current BMP levels, removing agricultural land from production, adding groundwater friendly crops to rotations, and sewering high density unsewered residential areas.

CHAPTER 7

CONCLUSIONS

SITE CHARACTERIZATION

The Stevens Point - Whiting - Plover municipal well recharge area lies east of the urban area and to the northwest of the Stevens Point airport. The municipal wells tap an extensive unconfined glacial outwash aquifer. As part of this study, we characterized the area's hydrogeology through an extensive review and analysis of previous reports and well logs. Well logs and other data relating to geologic and groundwater conditions were entered into databases for future use.

The outwash is a relatively uniform sand and gravel varying in thickness from 0 to 180ft, with about 100-115ft being typical. Underlying the outwash are Precambrian granite and isolated outliers of Cambrian sandstone. The bedrock surface has considerable topography hidden under the flat outwash plain. Deeply incised buried bedrock valleys are important conduits for groundwater flow and wellfield water supply. We developed bedrock elevation maps from well logs and mapped outcrops for current and future uses.

A limited number of pump tests on the outwash aquifer estimate hydraulic conductivity at 8 to 44 $x10^{-4}$ m/s. (Pump tests are better than single-well tests for characterizing large scale hydraulic conductivity.) We derived substantially more hydraulic conductivity data from specific capacity data contained in 327 well construction reports. Specific capacity derived hydraulic conductivity ranged from 3.3 to $9.5x10^{-4}$ m/s, depending on the well pumpage class used. A hydraulic conductivity map for the outwash aquifer was developed from the specific capacity data and is available for future use. Previous reports estimate an average porosity of 0.32 and a specific yield of 0.2.

We developed a composite water table map from topographic maps and well construction reports. These indicate the principal discharge areas controlling the groundwater flow system are the Wisconsin, Plover, and Little Plover Rivers, and Hay Meadow Creek. A major groundwater divide bounds the eastern extent of the flow system, approximately 5 miles east of the SWP urban area.

Irrigated agriculture is the dominant land use, followed by forest area. Land use in the 30 year TOT zones-of-contribution are typical for the SWP area, and consist of 40% irrigated agriculture, 24% forest, 8% dryland agriculture, 8% unsewered residential, 8% sewered urban, 6% grassland/brushland, and 6% various other. The main crops are potato, snap bean, field corn, sweet corn, and hay. The most common crop rotations are potato/sweet corn, irrigated corn/snap bean, and potato/irrigated field corn/snap bean.

FLOW AND PARTICLE TRACKING MODELS

A numerical computer model was developed to simulate area groundwater flow. After calibration, predictive runs were made with the addition of municipal well sinks based on projected year 2005 average and maximum daily pump rates. A particle tracking model was added to delineate zones-of-contribution for the municipal wellfields. The zones-of-contribution ranged from 10 sq miles for the Stevens Point main wellfield, 9.2 sq miles for Whiting, 3.4 sq miles for Plover, and 0.8 sq miles for the Stevens Point #5 well.

The groundwater flow model indicated substantial interaction between the Plover River and Stevens Point main wellfield. Flow patterns in this vicinity were difficult to analyze, and created difficulties in delineating the extent of this recharge area east of the river. The zone-of-contribution delineation for the Stevens Point main wellfield east of the river, therefore is the maximum possible contributing area. A water budget used to partition flow to the wellfield shows that approximately 64% is from the Plover River, 27% from west of the river, and only 9% from east of the river.

The flow model also indicated that pumping in the Plover wellfield at projected 2005 levels will have a substantial impact on the Little Plover River, decreasing its flow by over 40%. The model was not implemented to specifically examine Little Plover interactions, so the prediction is approximate. However, ample evidence exists to support the general conclusion that flow in the Little Plover will substantially decrease, and certain stretches may quit flowing during parts of the year.

NITRATE LOADING

Nitrate loading rates for agricultural crops were estimated using a mass balance approach. N loading for row crops was estimated at 4 to 137 lbs/acre-year, depending on crop, and whether conventional (CON) or best management practices (BMPs) were used. The CON loading rates for the 4 most common crops are potato - 82 lb/acre, snap bean - 125, irrigated field corn - 86, and sweet corn - 137. BMP's reduce loading by 10 to 30%. Crop loading estimates are probably low, especially for potato and legume forages. Nitrogen loading rates for land-spread manure were at 2.5 lb/ton (BMP) and 5.2 lb/ton (CON).

Residential loading was estimated from septic system plus lawn fertilization loading. The septic system loading used in the study was 40 lbs N/system-year. Lawn fertilization loading was based on an average 25% leaching rate of applied fertilizers. The rate for residential is likely higher than actual by 10-25%.

PREDICTED STEADY-STATE NITRATE-N CONCENTRATIONS

Model predicted concentrations for the municipal wells are partly dependent on whether conventional (CON) or 100% farmer best management practice (BMP) implementation are assumed for the agricultural land use. Predicted steady-state nitrate-N concentrations at the municipal wellfields are mostly higher than the nitrate MCL. Predicted concentrations under current land uses are:

Plover - 25.9 mg/L (CON) and 18.7 mg/L (BMP)

Whiting - 37.8 and 25.7

Stevens Point # 5 - 21.7 and 15.5

Stevens Point main - 4.0 and 3.1.

These predicted concentrations are substantially higher than current conditions. An analysis of errors and assumptions indicates that only the Stevens Point #5 well may be substantially in error, due to differences between modeled pumping schedule and volumes compared to current well deployment. Trends in nitrate concentrations in the Whiting wellfield and Little Plover River support the notion that nitrate concentrations will rise over time.

The Stevens Point main wellfield prediction is lower than the others because of dilution by induced Plover River recharge and large amounts of no N-load forest and airport areas around and to the west and north of the wellfield. However, the predicted nitrate concentrations for the portion of the Stevens Point main wellfield recharge area east of the Plover River are 28.1 (CON) and 19.7 (BMP) mg/l N. With the pump rates studied, the models suggest only a small amount of recharge from this area actually contributes directly to wellfield pumpage.

NITRATE SOURCES

The major nitrate loading sources to the municipal wells are agriculture (89%), unsewered residential (7%), surface waters (2%), and urban (1%). Agricultural loading components are irrigated fields (66%), manure (15%), and dryland fields (8%). The principal nitrate loading source varies somewhat by wellfield. Plover and Whiting loading are dominated by agriculture (98% and 92% respectively). Stevens Point #5 is impacted by both agriculture (67%) and unsewered residential (31%). The primary loading source for Stevens Point main is agriculture (67%), originating primarily in the small portion from the eastern zone-of-contribution that passes under the Plover River. Other sources include Plover River surface water (20%), and unsewered residential (13%). For Stevens Point main east of the Plover River, 97% of the loading is from agriculture.

The irrigated agriculture land use has the highest loading rate. Estimated irrigated agriculture loading averages 97 lbs/acre-year CON and 69 lbs/acre-year BMP, while dryland agriculture averages 63 lbs/acre CON and 38 lbs/acre BMP, manure loading excluded. For residential land uses, loading rates were: urban sewered - 8 lb/acre-yr, low density unsewered - 12 lb/acre-yr, high density residential - 48 lb/acre-yr, mobile home park - 168 lb/acre-year. The agricultural loading estimates are likely somewhat low, and the residential somewhat high, compared to reality.

Managing Agricultural Nitrate Loading

Maintaining or improving existing water quality requires reducing N loading from agricultural sources. Considerable emphasis is currently placed on using best management practices to reduce agricultural nitrate loading. For BMPs to protect groundwater, sufficient numbers of farmers must voluntarily adopt the practices, and the practices must effectively reduce the specific causes of groundwater pollution to acceptable levels. Universal BMP adoption achieves about a 30% reduction in nitrate loading, though traditionally, it has been difficult to sustain a 50% BMP participation rate in most water quality projects. Even with universal adoption, predicted nitrate-N concentrations remain in the 15.5-25.7 mg/L range for all but the Stevens Point main wellfield. BMPs are obviously a valuable tool, but they are clearly not the total solution.

To reach the nitrate MCL, agricultural loading would have to be further reduced by 53% in the Stevens Point main east recharge area, 66% in Stevens Point #5, 69% in Whiting, and 47% in Plover. Nitrate loading might be decreased by increasing the N credit for legumes and manure, reducing fertilizer inputs below current BMP recommendations, increased crop residue utilization, changes in land use (such as conversion of high N load per acre cropland to forest or low density residential use), and changes in crops grown (such as adding soybean or hay years to rotations).

Managing Residential/Urban Loading

Eliminating all nonagricultural loading achieves the MCL goal only in the Stevens Point #5 zone-of-contribution when coupled with universal adoption of agricultural BMP's. Predicted nitrate concentrations are reduced from 15.5 to 8.5 in this scenario. Eliminating nonagricultural loading in the other areas has minimal impact.

On-site septic systems are the largest N source in unsewered residential areas, contributing an estimated 40 lbs/system/year. This loading is not because some systems have failed or are substandard; working on-site systems pollute groundwater. The best "BMP" for reducing residential

N loading from existing development is to provide community sewer services. Replacement of existing systems with improved designs for nitrogen removal might be an acceptable alternative where sewering is not possible.

RECOMMENDATIONS

1. Develop water quality goals to guide management of SWP zones-of-contribution. Goals are crucial in developing strategies for improving water quality.

2. Education efforts are needed to provide current information on groundwater status to leaders, citizens, and farmers to enable them to make informed decisions. Educational topics should include:

* Groundwater recharge and flow in the SWP area
* Sources of groundwater contamination in the SWP area; cause-effect relationships between land use activities and groundwater quality
* Continued degradation of groundwater quality with current land use
* Potential impacts or threats from the contaminants; community water quality goals
* Measures to effectively achieve water quality goals; limitations of BMP's

3. BMP's should be vigorously pursued, even though they are only partially helpful in improving groundwater quality.

4. Viable methods to change land uses to lower N loading uses should be investigated, including conversion to sewered urban uses in the urban fringe that are compatible with a wellhead protection district, economic incentives, and community purchase.

5. Encourage adding or replacing existing crops with groundwater friendly low N-load crops, such as soybeans under BMP practices. Investigate use of "scavenger" crops that would take up and recycle nutrients after the main crop is harvested.

6. Explore ways to introduce crops requiring lower inputs into rotations.

7. Encourage investment of agricultural research resources to the development of crop varieties that have lower N needs and/or better uptake efficiency, additives or fertilizers that reduce leaching potential, and methods to recover or recycle nutrients leached to groundwater back to crops.

8. Encourage community efforts to extend municipal sewer to high density unsewered residential areas along the urban fringe, such as intergovernmental agreements that address cost and tax roadblocks. Promote development and use of alternate nitrogen removal on-site treatment systems.

9. Maintain and upgrade the models developed by this project as dynamic educational and management tools for the SWP communities. Incorporate new hydrogeologic data as it becomes available. Encourage transfer of the technologies to other communities.

10. Portage County should continue to coordinate municipal wellhead protection management. Work groups should focus community groundwater quality concerns into specific long-term water quality goals and suggest measures, such as noted in this report, for the community to consider. Work groups should include representation from local governmental units, concerned citizens, farmers, other landowners, university and extension staff, related state and federal agency staff, and agri-business professionals. The groundwater management recommendations contained in the county's groundwater management plan (Portage County Planning Department, 1988) should be reviewed for effectiveness in reducing non-point contaminant loading.

11. Work groups should form and begin discussions for measures that might protect surface water resources in light of potential for decreased flows and pollution from groundwater.

QUALIFICATIONS/ASSUMPTIONS

1. The models, and resulting zone-of-contribution delineations and water budgets, were based on and calibrated with the best available information. The models appear reasonable and robust, and can be considered the best available description of the SWP area. However, use of the models must not be extended beyond their data limitations.

2. The 2-dimensional models provide limited detail for groundwater interactions with the Plover River in the vicinity of the Stevens Point main wellfield. Errors in the amount of induced river recharge to the wells assumed from the models could significantly impact concentration calculations.

3. The contaminant mass-balance model is a steady-state description of likely concentrations resulting from uniform mixing of loading under sustained land management practices. Although the SWP area is too dynamic to ever truly reach a steady-state equilibrium, the predicted concentrations are useful for evaluating the potential impacts of different land uses and management practices. The guiding caveat should be, "This is the quality of groundwater that will result if land-uses and management remain as there are now." The predicted concentrations are good approximations of long-term concentrations that the communities can expect to have to deal with. The predicted concentrations are only valid for the wellfields and not any other points in the defined flow systems.

4. The mass-balance model assumes no internal sources and sinks for nitrate. If denitrification is occurring, the predicted concentrations are too high.

5. The municipal wells are considered a consumptive use, with no return flow to the zone-of-contribution. While minor return flow might occur in sewered urban areas, such as by lawn watering,

this amount would be extremely minor compared to the recharge water budget used for the calculations.

6. The 30 year time-of-travel area delineated for the year 2005 average daily pump rates is assumed to be representative of the recharge and loading to the wellfield. Predictions could be higher or lower if certain land uses were concentrated in areas not included or disproportionately included within the delineated area for land use loading calculations.

7. The loading rates used in this study are assumed to be a reasonable estimate of the average long-term loading that can be expected. Many factors, including climatic and crop growth variations, could raise or lower the actual leaching rate to groundwater in a given year.

8. This study targeted the principal non-point nitrogen loading sources associated with identified land uses in the SWP area. Other loading sources, such as intermittent land disposal of sludge materials or septage waste, were considered minor on a regional scale relative to the sustained loading from the basic long-term land uses identified. While one-time point sources, such as fertilizer spills, could have severe localized impacts, they would not be part of the steady-state long-term predictions. Inaccuracies in the land use/management mapping would be a source of error in the nitrate concentration calculations.

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APPENDIX A

MODFLOW INPUT FILES

Appendix A has been published as a separate document. Please contact the Central Wisconsin Groundwater Center for availability. The MODFLOW input files are also available as ASCII computer files.

APPENDIX B

MODPATH INPUT FILES

Appendix B has been published as a separate document. Please contact the Central Wisconsin Groundwater Center for availability. The MODPATH input files are also available as ASCII computer files.

APPENDIX C
ALTERNATE FLOW MODEL CALIBRATIONS

ALTERNATE BEDROCK CONFIGURATION

The flow model developed in Chapter 3 defines the bottom of the aquifer as the bedrock surface (Figure 2.9) delineated from well logs and bedrock outcrops. A major feature of the bedrock topography is a bedrock valley trending north to south. The highly permeable sand and gravel filling the bedrock valley is an important component of the aquifer. The bending of the time-of-travel lines in the northwest portion of the Stevens Point main zone-of-contribution (Figure 4.2) is an example of the impact the buried valley has on the flow dynamics. The available data gives us a fairly good picture of this feature, but does not completely define its course as a contiguous feature.

We wished to examine the sensitivity of zone-of-contribution delineations to alternate interpretations of the bedrock valley. To do this, an alternate model was developed with the bedrock surface modified as shown in Figure C.1. While the hypothetical bedrock surface depicted in Figure C.1 is by no means the only interpretation possible, we believe it is a reasonable extrapolation of the available data used to construct Figure 2.9, and is adequate to test the sensitivity of zone-of-contribution delineations.

This alternate model was developed by taking the model described in Chapter 3, revising the aquifer bottom elevations as per Figure C.1, and recalibrating to the same targets with adjustment of the hydraulic conductivity. In general, the enhanced bedrock valley initially lowered heads, most noticeably south of the Little Plover River, through the central area between Whiting and Stevens Point #5, and north of the Stevens Point main wellfield. A calibration comparable to the Chapter 3 calibration was obtained with reasonable adjustments in hydraulic conductivity (Figure C.2 and Table C.1). The revised bedrock valley facilitated calibration south of the Little Plover River.

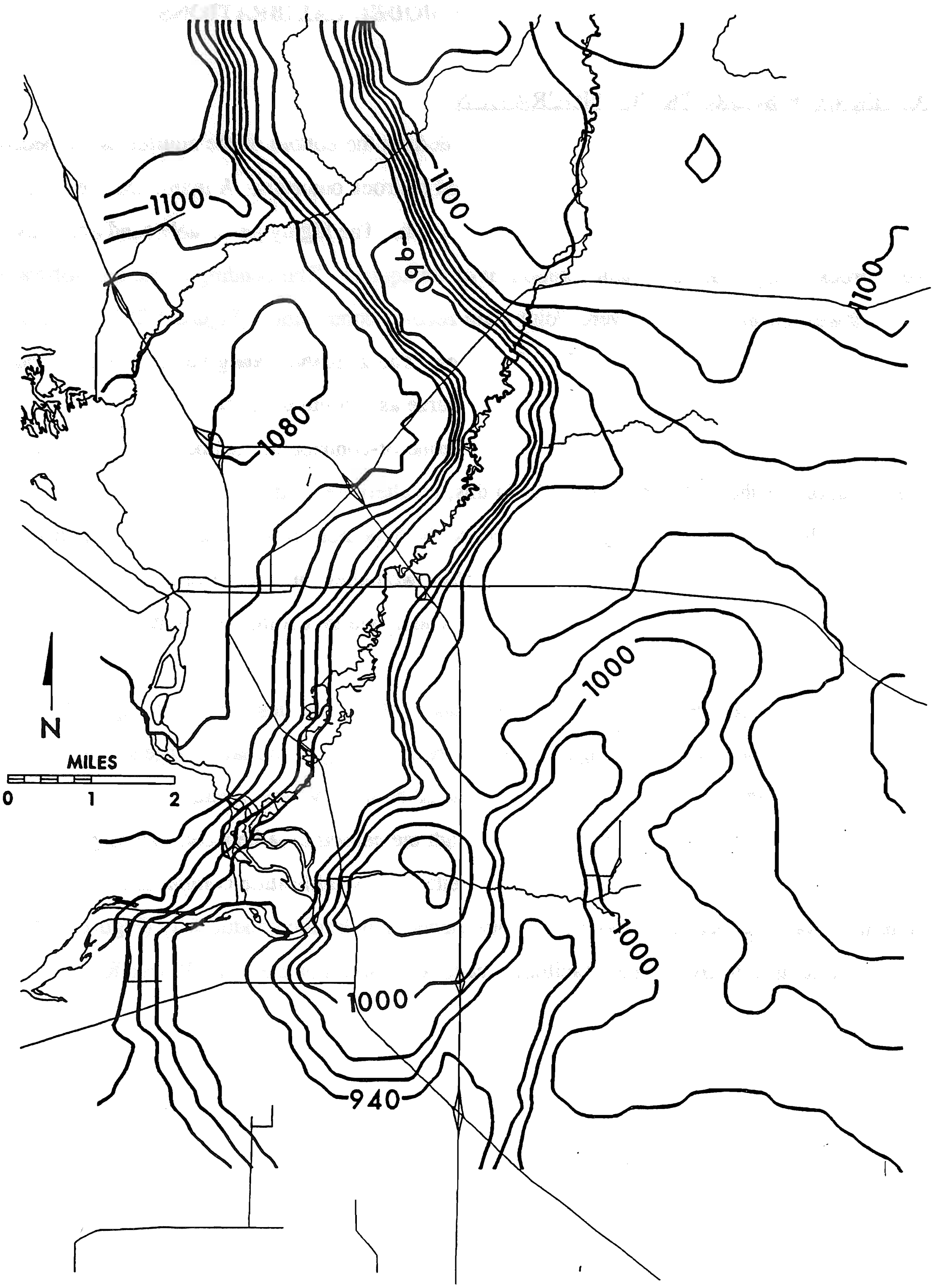

Figure C.1 Revised bedrock topography for the alternate bedrock model (contours are in feet MSL).

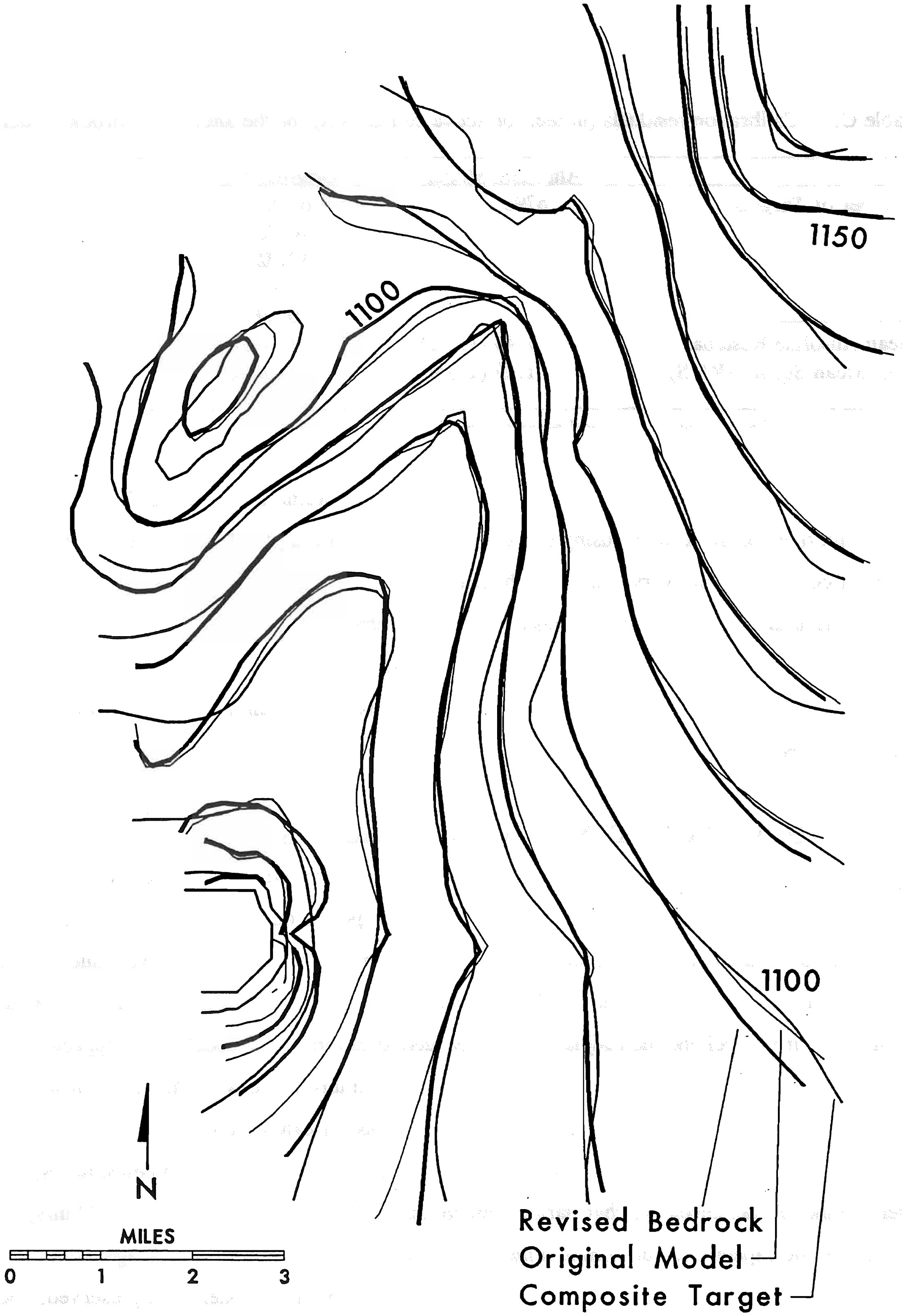

Figure C.2 Water table contours (feet MSL) for the original calibrated model, the revised bedrock model, and the composite calibration target.

Table C.1. Calibration residuals (in feet for active cell targets) for the alternate bedrock model.

	Alternate Model	Original Model
Number of Targets	6795	6795
Minimum Residual	-10.36	-10.76
Maximum Residual	14.49	12.02
Residual Std Deviation	1.86	1.97
Mean Residual	0.59	-0.20
Mean Absolute Residual	1.44 (1.2%)	1.54 (1.3%)
Root Mean Square (RMS)	1.95 (1.6%)	1.98 (1.6%)

(%)=percent of total change in head across model

The 30 year zones-of-contribution for the municipal wellfields (Figure C.3) were delineated for the alternate bedrock model using procedures described in Chapter 4. The size of the alternate zones-of-contribution are very similar, with small changes in alignment. The most pronounced change was a shift in the Stevens Point main eastern zone-of-contribution slightly to the north. Our conclusion is that potential errors due to bedrock valley uncertainties are not likely to significantly impact zone-of-contribution or time-of-travel delineations, nor the analysis of nitrogen loading and wellhead nitrate concentrations.

ALTERNATE HYDRAULIC CONDUCTIVITY CALIBRATION

The flow model described in Chapter 3 utilized an initial hydraulic conductivity (K) distribution based on calculations from specific capacity reported on well logs and construction reports (Figure 2.10). The final K distribution (Figure 3.3) is complex because it preserves much of the spatial diversity inherited from the well data. An alternate approach would be to start with an average K for the entire model and then adjust the K as needed to calibrate the model. This approach would downplay the significance of the spatial K patterns based on unverified well data, and allow an independent K distribution based on reasonable adjustments for calibration.

To test the impacts of a different K distribution on zone-of-contribution delineations, an alternate model was developed that started with an uniform K value of 0.003 ft/s ($9x10^{-4}$ m/s). Based on the specific capacity calculations and literature values, this is a reasonable average value for the SWP area as a whole. All other conditions of the original calibrated model were preserved, and only K needed to be adjusted for calibration.

The final K distribution for this model is given in Figure C.4. While there are similarities to the final K distribution presented in Chapter 3 (Figure 3.3), the alternate K distribution is much

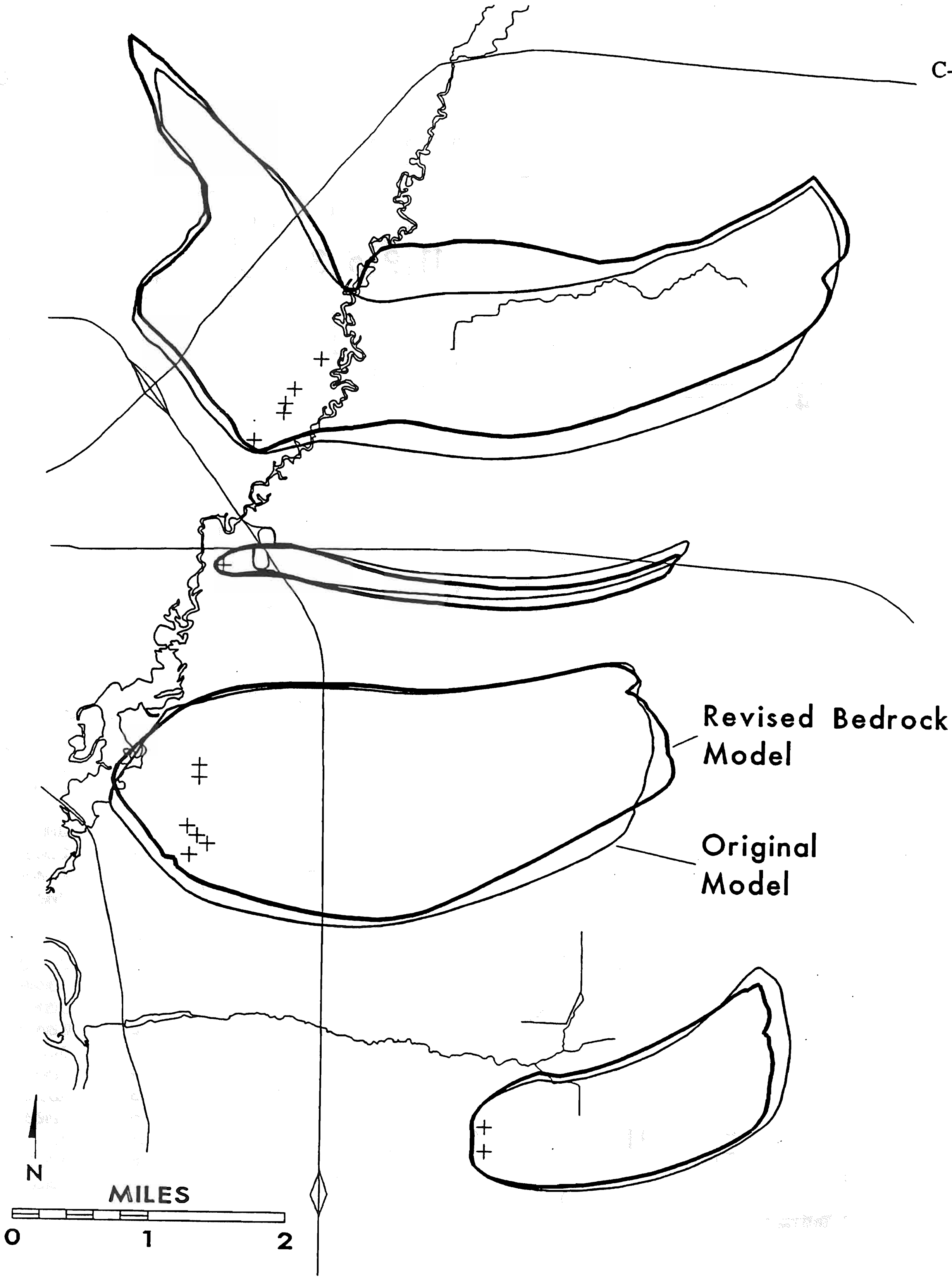

Figure C.3 Comparison of 30 year zones-of-contribution for revised bedrock model and original model.

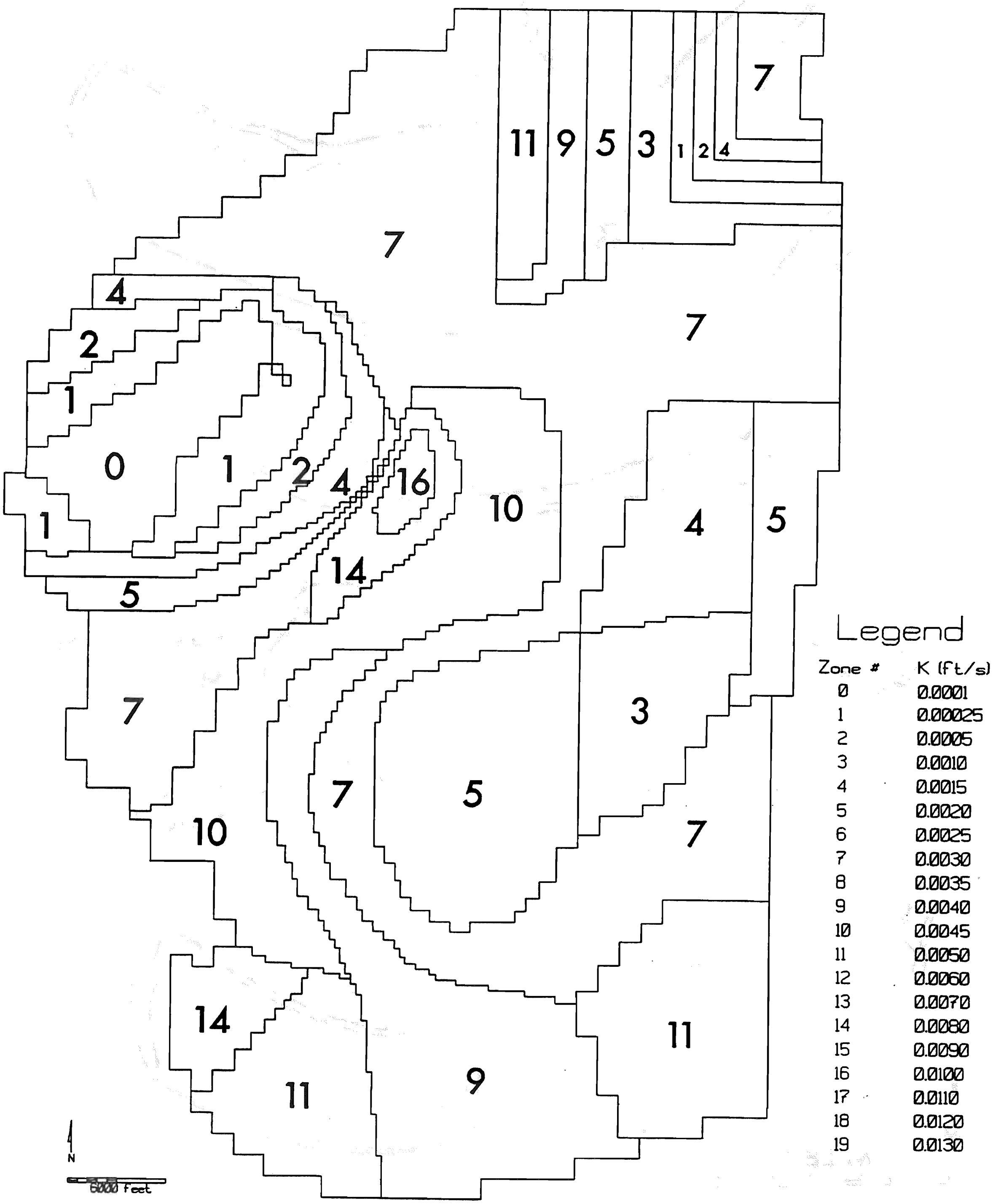

Figure C.4 Final hydraulic conductivity zones for the alternate hydraulic conductivity model.

simpler. The calibrated heads and residuals for this model compare closely with the original model described in Chapter 3 (Figures C.5 and Table C.2).

Table C.2. Calibration residuals (in feet for active cell targets) for the alternate hydraulic conductivity model.

	Alternate Model	Original Model
Number of Targets	6795	6795
Minimum Residual	-10.66	-10.76
Maximum Residual	12.02	12.02
Residual Std Deviation	1.69	1.97
Mean Residual	0.24	-0.20
Mean Absolute Residual	1.31 (1.1%)	1.54 (1.3%)
Root Mean Square (RMS)	1.71 (1.4%)	1.98 (1.6%)

(%)=percent of total change in head across model

The 30 year zones-of-contribution delineated with the alternate K model are very similar to the original delineations (Figure C.6), and we would not expect significant impacts on nitrogen loading and concentration calculations. The most notable differences occur at Whiting and Stevens Point main. Whiting's 30 year zone-of-contribution does not extend as far upgradient; Stevens Point's eastern zone extends further south and the northwest extension is much less pronounced.

ALTERNATE TREATMENT OF BEDROCK HIGH AREAS

As noted in Chapter 3, the area northwest of the Stevens Point main wellfield is different because of high bedrock and possible flow through less permeable bedrock residuum. If the aquifer bottom in this area is defined as the bottom of the sand and gravel as extrapolated from sparse well logs, numerous cells have a target water elevation below the bottom of the aquifer by up to 21 feet. This problem is due to sparse data, interpolation/averaging errors, and possible flow through bedrock residuum. The model was therefore calibrated by lowering bedrock and hydraulic conductivity. Bedrock was adjusted an average of -19 feet, with some cells by as much as -40 feet (Figure 3.7).

Another approach would be to adjust bedrock as needed to keep target heads above the bottom of the aquifer, but to not assume any significant flow below this level of adjustment deeper in the bedrock residuum. That is, limit bedrock adjustments to no more than approximately -20 feet. An alternate model using this approach was developed using the calibrated model from Chapter 3.

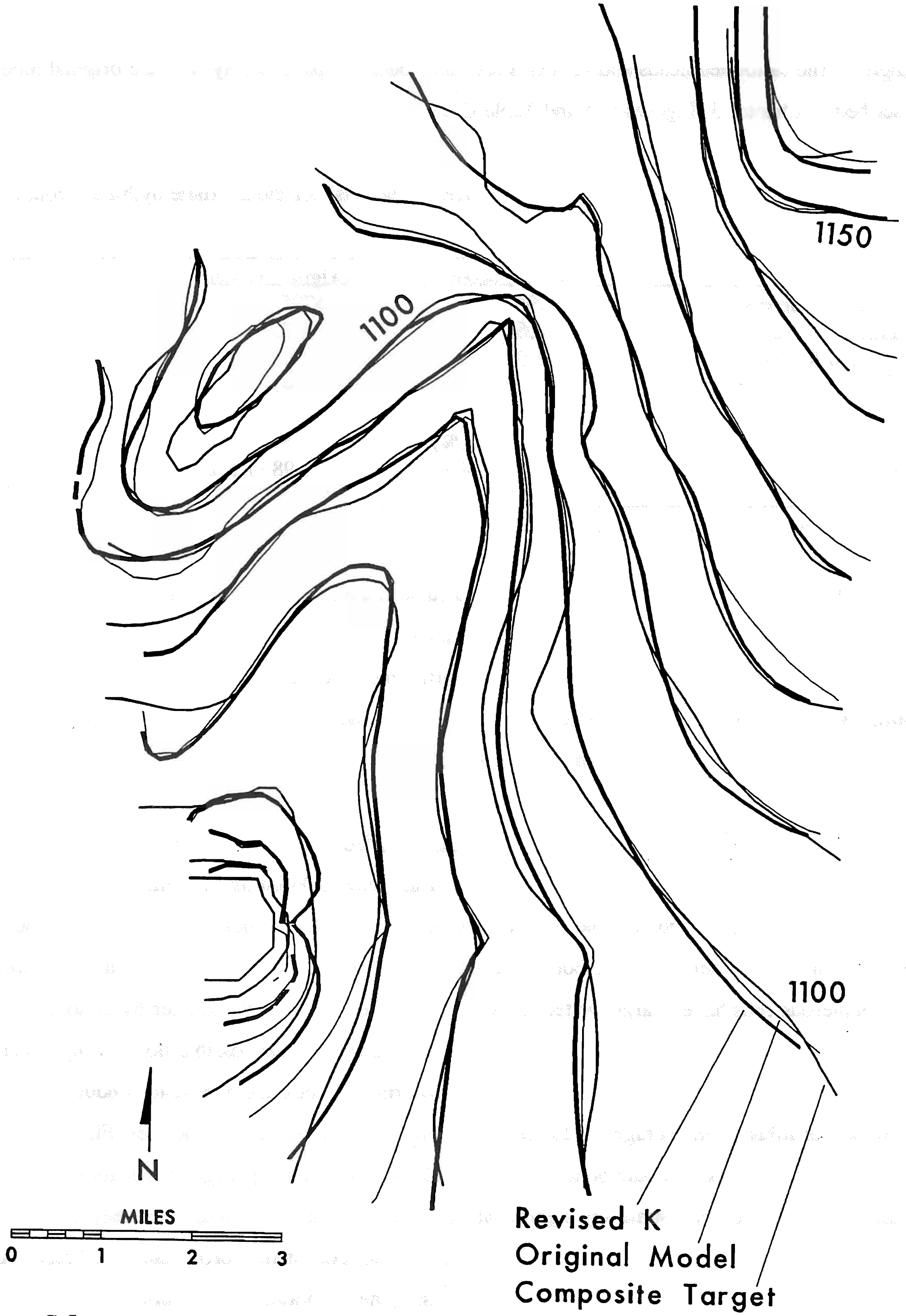

Figure C.5 Water table contours (feet MSL) for the original calibrated model, the alternate hydraulic conductivity model, and the composite calibration target.

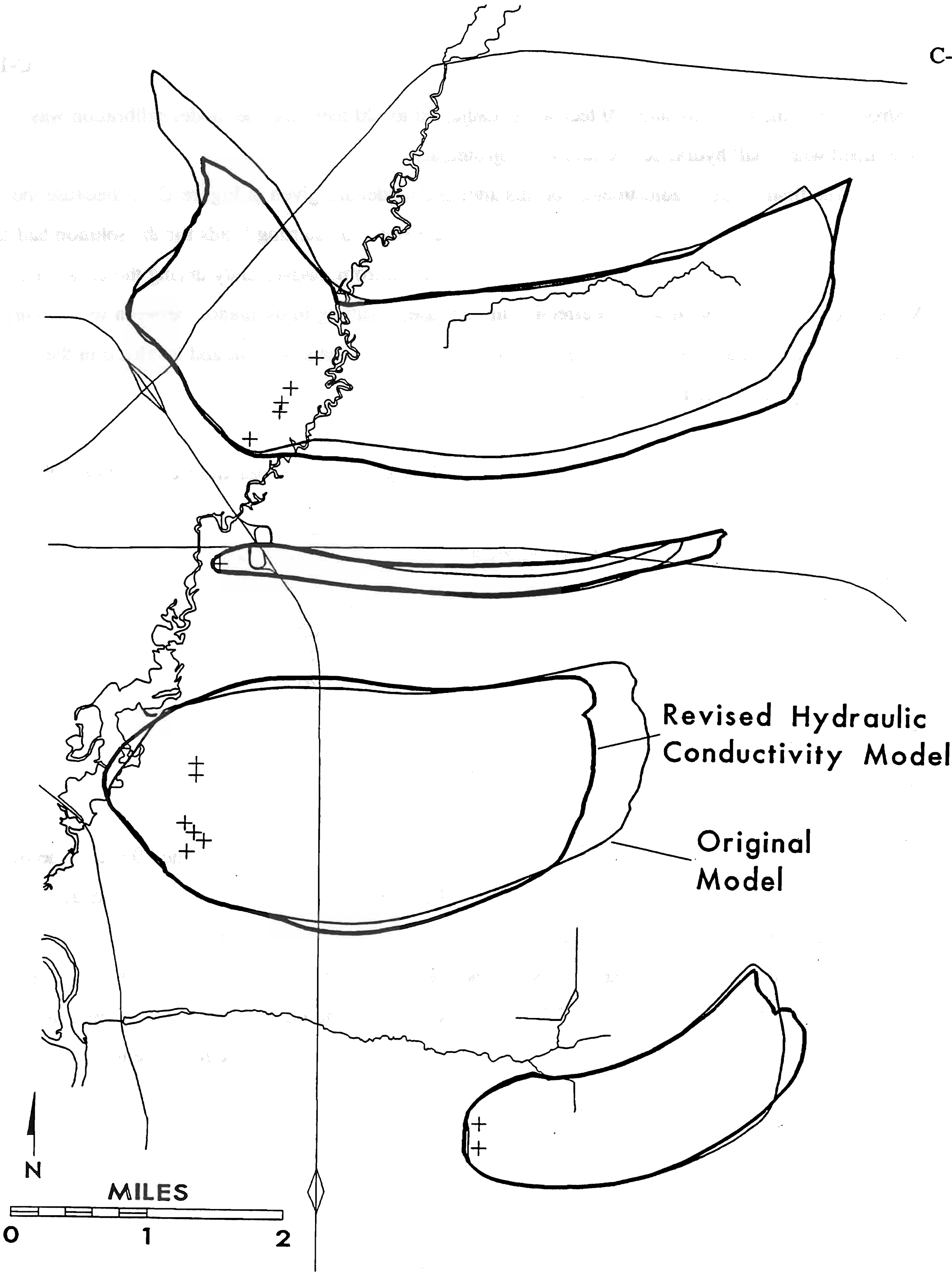

Figure C.6 Comparison of 30 year zones-of-contribution for revised hydraulic conductivity model and original model.

Bedrock adjustments more than -20 feet were readjusted to -20 feet, and the model calibration was fine-tuned with small hydraulic conductivity adjustments.

The final bedrock adjustments for this alternate model are given in Figure C.7. Because the saturated thickness in this scenario is very thin for some cells, the starting heads for the solution had to be very close to the final heads to prevent model oscillation from inadvertently drying the cells. The MODFLOW wetting option was not effective in this case, resulting in oscillation between wet and dry. The impact of this alternate model on heads and calibration residuals is slight and localized in the northwest (Figure C.8 and Table C.3).

Table C.3. Calibration residuals (in feet for active cell targets) for the alternate treatment of high bedrock areas.

	Alternate Model	Original Model
Number of Targets	6795	6795
Minimum Residual	-10.76	-10.76
Maximum Residual	12.02	12.02
Residual Std Deviation	1.97	1.97
Mean Residual	-0.27	-0.20
Mean Absolute Residual	1.55 (1.3%)	1.54 (1.3%)
Root Mean Square (RMS)	1.99 (1.6%)	1.98 (1.6%)

(%)=percent of total change in head across model

This alternate treatment of the high bedrock area had minimal impact on the 30 year zone-of-contribution delineation for Stevens Point (Figure C.9). We would not anticipate any significant changes to nitrogen loading calculations. As noted in Chapter 3, the area of high bedrock is difficult to model with the available information. We cannot be sure which treatment better describes the area, although the results of this alternate model suggest zone-of-contribution delineations and nitrogen loading calculations are relatively insensitive to a reasonable range of bedrock adjustments.

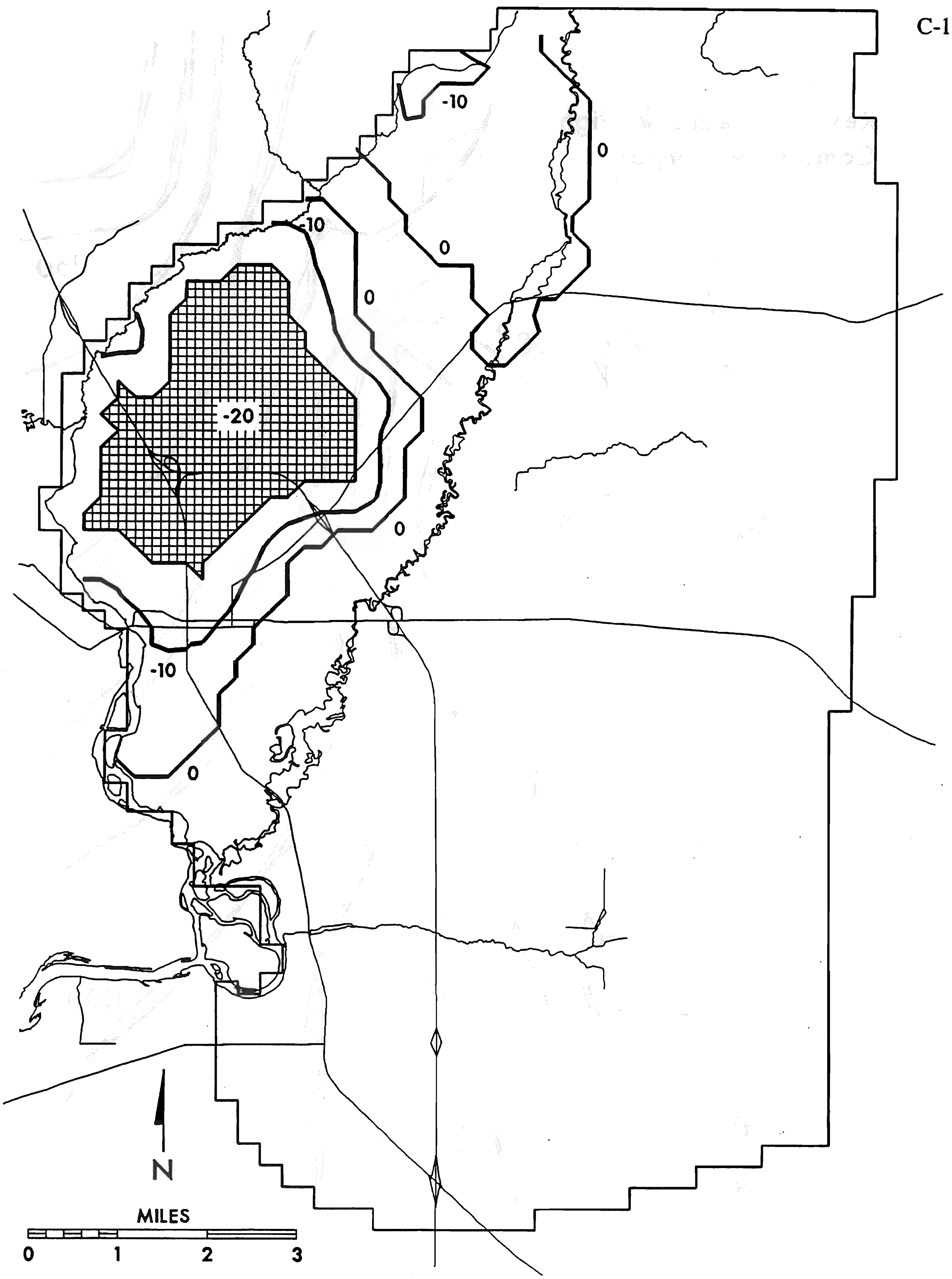

Figure C.7 Adjustments to bedrock elevation (feet) for the alternate treatment of bedrock highs model.

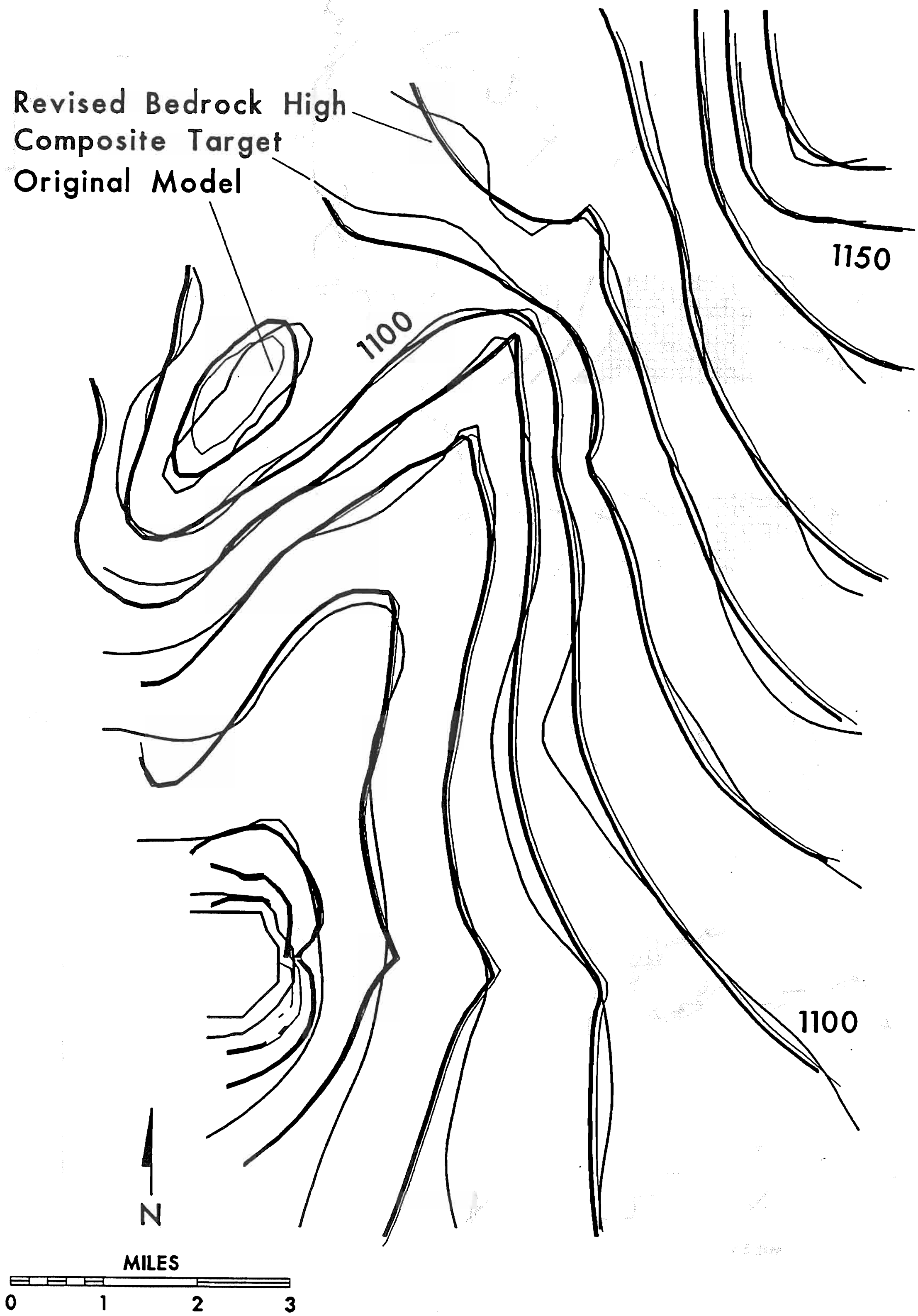

Figure C.8 Water table contours (feet MSL) for the original calibrated model, the alternate bedrock high model, and the composite calibration target.

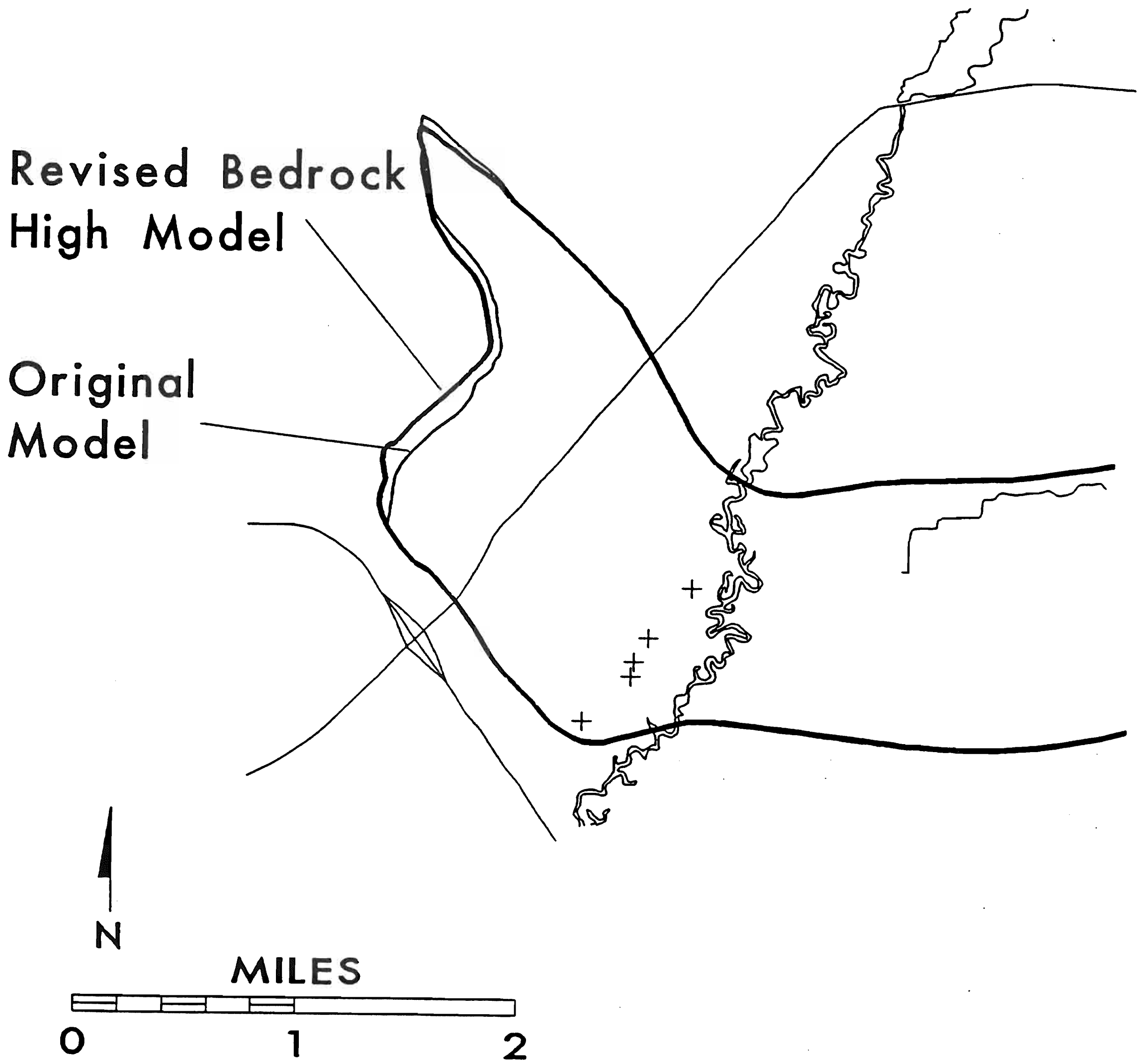

Figure C.9 Comparison of 30 year zones-of-contribution for revised bedrock high model and original model.