Groundwater Pumping Effects on Groundwater Levels, Lake Levels, and Streamflows in the Wisconsin Central Sands

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AUTHORS' NOTE: UNITS OF MEASUREMENT

The hope of the authors is that this report will be used to inform discussions about groundwater management in the Wisconsin Central Sands. The report is aimed at an audience that includes agency staff, policy makers, farmers, conservation groups, as well as professional hydrologists and scientists.

In most scientific writing, SI units of measurement *Système International d'unité* (often termed "the metric system") are preferred or mandated. In the case of this report, we believe accessibility by the audience trumps technical correctness, and that the broader audience (and likely many scientists as well) relates more closely with inches of precipitation, feet of water level decline, and acres of lake size than with millimeters, meters, and hectares of the same. Hence we use English units for all but the parts of this report that will likely have only a scientific audience (the groundwater modeling section in Chapter V as an example).

EXECUTIVE SUMMARY

Background

Prominent hydrologic studies in the 1960s and 1970s warned that the growth in groundwater pumping for agricultural irrigation in the Wisconsin Central Sands could substantially lower regional water levels and streamflows. Irrigation grew in the succeeding decades, and presently encompasses some 2,300 high capacity wells that service 200,000 acres.

Since 2000, Central Sands water levels and stream discharges have been notably depressed, at least in areas that contain large densities of high capacity wells. For instance, the Little Plover River, a formerly high-quality trout stream and a Wisconsin Exceptional Resource Water, was near dry in 2003 and has dried annually in stretches since 2005. (Ironically, the Little Plover was the subject of a 1960s USGS study and film that explored pumping effects on surface waters). Long Lake near Plainfield, which formerly covered 45 acres and had a maximum depth of about 10 feet, has been near dry to dry since 2005. Other lakes in that vicinity have dried, and some that did not (e.g., Pickerel and Wolf Lakes) winter-killed due to depressed water levels.

Questions exist as to whether recent depressed hydrologic conditions are related to drier weather or to groundwater pumping. Or both? Pumping would have its most noticeable impact when wet conditions are unable to mask its effects. Also, irrigation consumption would be expected to be greatest during drier times.

The study summarized here seeks to clarify the impacts of pumping on the Central Sands water resources. The Central Sands region is an extensive, though loosely-defined, region characterized by a thick (often > 100 ft) mantle of coarse-grained sediments overlying low permeability rock, and landforms comprising outwash plains and terminal moraine complexes associated with the Wisconsin Glaciation. Here we address the region between the headwater streams of the Fox-Wolf Basin and those of the Central Wisconsin Basin, which contain some 83 lakes (> 12 acres) and over 600 mi of headwater streams, many of them high-quality coldwater fisheries, in close proximity to large densities of high capacity wells.

Recent indicators of hydrologic conditions

Precipitation, stream discharge, and groundwater level records indicate that climate alone does not explain depressed hydrologic conditions in 2000 to 2009 in parts of the Central Sands with large densities of high capacity wells. (We caution that the Central Sands situation should not be confused with the real and severe long-term drought in the northern part of Wisconsin, nor with flooding in the south, but rather understood in its own context.)

Central Sands annual precipitation in 2000 to 2004 was mostly average to above average while

2005 to 2008 was slightly below to slightly above average. In longer term view, post-1970 precipitation increased by 0.7 to 2.8 in, depending on station, compared with 1940-1970 precipitation. We hypothesize that the post-1960 effects of rapid irrigation expansion in the Central Sands may have been masked by increases in precipitation at about the same time.

Annual discharge records for reference streams (streams not greatly affected by pumping) revealed significant lows over the past 90 years, especially 1931 to 1934, 1948-9, 1957-9, 1964, 1977, and 1988. The 1930s were apparently the driest part of the record with the 1950s coming in second. Reference stream discharges in 2000-2004 were about average for the long term record, and when the Little Plover dried for the first time in 2005 (unprecedented in a 50 year record that included some of the driest years of the last century), reference stream discharges were at 15-26 percentiles. In 2006-7 reference stream discharges were somewhat low, 8-18 percentile, and in 2008 were a more robust 25-50 percentile.

Groundwater and lake levels in areas with few high capacity wells were slightly below to slightly above average in 2000-2005, and in 2006-8 were lower than average, 11 to 16 percentile, but not at rare nor record lows. The available groundwater and lake level record (1950s to present), for areas with few high capacity wells, was generally coincident with stream discharge records, and exhibited 50 year lows in 1958-9.

Water level declines in areas with large densities of high capacity wells

Areas with large densities of high capacity wells experienced record lows in 2000-2008, in sharp contrast to areas with few high capacity wells. In 2000-2008, the Plover monitoring well experienced its seven lowest water level years since 1958, Hancock experienced its lowest two, and Bancroft its lowest four. Coloma NW had its first and third lowest years in a record that started in 1964. "Missing water," water level declines that cannot be accounted for by weather alone, in these wells ranged from about three feet at Plover and Hancock to one foot at Bancroft and Coloma NW. Missing water estimates represent a sort of average for 1999-2008 and not peak amounts. Methodology probably underestimates missing water by about 0.4 to 0.76 feet**.**

Lake levels in areas with few high capacity wells did not show a non-climatic water level decline, such as in the vicinity of Wild Rose and Wautoma. However, lakes in or near areas with many high capacity wells showed substantial and statistically significant missing water amounts in the range of 1.5 and 3.6 feet, depending on the lake's location. These declines represent an average for the 1990s to 2007, do not capture potential peak amounts of missing water, and due to methodology may be underestimated by about 0.4 to 0.76 feet**.**

Groundwater flow modeling

Four groundwater flow model versions for the Central Sands were developed, each representing slightly different conceptual models of the region's hydrogeology. The four models produced similar predictions of water level and streamflow reductions in response to pumping stresses, and appear to reasonably reflect hydrologic reality.

The flow models indicate irrigation pumping may cause large impacts on the region's lakes and streams. In parts of the Central Sands, modeling predicts up to 2.5 feet of water table and lake level decline *per inch* of net recharge reduction on irrigated lands, and greater than 20% flow reduction in headwaters streams *per inch* of net recharge reduction. Given that some estimates of average net recharge reduction on irrigated lands range to 2-3 in, the consequences on lakes and streams are potentially large.

A best fit between statistically estimated and modeled water level declines occurs with a net recharge reduction of 1.9 in on irrigated lands in the flow model. When the flow model is run with this amount of recharge reduction, steady-state water levels declines up to four feet are predicted in areas where in reality lakes are highly water level stressed. Headwater stream discharge reductions are commonly 20-50%. Computed water level and stream discharge declines with a 1.9 in net recharge reduction represent a sort of average and not seasonal nor long term potential maximum declines.

Conclusion

We conclude that climatically driven conditions in 2000-2008 are alone unable to account for the severely depressed water levels and streamflows in areas of the Central Sands that contain high densities of high capacity wells. Declines of around four feet or more in water levels by pumping are possible beyond climatic influences. This is not to say that lake and water levels are unaffected by recent climate or that every lake in the region is so affected. But for broad parts of the Central Sands with large densities of high capacity wells, pumping greatly aggravates or dominates climatic effects.

Impacts on streamflows can only be ascertained through flow modeling as the stream discharge record is too spotty. Modeling indicates headwater streams depletions with the 1.9 in of net recharge reduction are commonly 20-50%. This amount is a sort of average, and does not represent potential annual nor dry period maximum impacts.

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Tech memorandum 1 - Exploring USGS Daily Streamflow Records

Tech memorandum 2 – Exploring Long-Term Streamflow Records

Tech memorandum 3 – The USGS Groundwater Level Record

Tech memorandum 4 – Groundwater Level Trends in the Study Area

Tech memorandum 5.1 – Groundwater Level Trends in the Study Area – Part 2

Tech memorandum 6.1 – Summary – Groundwater Level Trends in the Study Area from Long Term USGS Monitoring Wells

Tech memorandum 7 – Lake Level Records with Possible Applicability for Trend Analysis in the Study Area

Tech memorandum 9 – Precipitation Trends in the Study Area: Available Data Set

Tech memorandum 10 – Precipitation Trends in the Study Area: Hancock Station Record

Tech memorandum 11 - Groundwater Flow Model for the Wisconsin Central Sands

I. INTRODUCTION

Lake levels, groundwater levels, and streamflows in the Wisconsin Central Sands (Figure I-1 and Figure I-2) have been depressed in recent years, greatly so in areas with large densities of high capacity wells (Figure I-3 and Figure I-4) (WLP 2006; Clancy et al., 2009). Accounts of what some consider alarmingly depressed conditions increased beginning in 2005 (Figure I-5). For instance, Long Lake near Plainfield, which in recent times covered 45 acres and had maximum depth of about 10 feet, has been near dry to dry in 2005-2009. Low lake levels have provoked winter fish kills on Pickerel and Wolf Lakes in Portage County. The Little Plover River, which formerly (1959-1987) discharged at a mean of 10 and a minimum of 3.9 cubic feet per second (cfs) (Hoover Road gauge), has mostly flowed at less than the former minimum since 2005 and has dried in stretches every year since 2005. The headwaters of Stoltenberg Creek in Portage County have been dry or nearly dry since 2005. Groundwater levels are at record lows (50-60 year record) in places where high densities of high capacity wells prevail.

We note that some confusion exists among the Central Sands situation, northern Wisconsin severe drought, and southern Wisconsin flooding. As of the end of 2008, the Central Sands was neither in severe drought nor in flooding (NOAA, January 2009; http://www.cpc.noaa.gov/products/predictions/tools/edb/lb-11jan2009.gif). We later show that the Central Sands is about within the normal bounds of climatic dry.

This report describes an investigation into the potential effects of groundwater pumping on groundwater levels and surface water resources in Central Sands region. The Central Sands is an extensive, though loosely-defined, region characterized by a thick (often > 100 ft) mantle of coarse-grained sediments overlying low permeability rock, and landforms comprising outwash plains and terminal moraine complexes associated with the Wisconsin Glaciation. The investigation particularly addressed the region between the headwater streams of the Fox-Wolf Basin and those of the Central Wisconsin Basin, which contain some 83 lakes (> 12 ha) and over 600 mi of headwater streams in close proximity to a great density of high capacity wells (Figure I-2). Specific objectives of the investigation were to:

1. Assemble available lake level, groundwater level, and stream baseflow data.
2. Collect new stream baseflow data.
3. Evaluate assembled stream, lake, and groundwater level data for indications of pumping impacts.
4. Expand and improve upon an existing groundwater flow model for the region.
5. Use the improved flow model to evaluate potential impacts of groundwater pumping on lake levels, groundwater levels, and stream baseflow.

Central Wisconsin contains the state's greatest density of high capacity wells, with about 2300 in the five counties that this study area overlaps (Figure I-1). Here we focus on widely distributed groundwater pumping for irrigation. Other uses (municipal, industrial), while potentially significant locally, are small compared to irrigation (Buchwald, 2009) and have a limited geographic distribution. Some impacts of non-irrigation pumping have been explored by Clancy et al. (2008) and Mechenich and Kraft (1997). Growth in high capacity irrigation well numbers and groundwater pumping has been rapid, minimally controlled, and mainly without regard for impacts on lake and stream resources. This growth mirrors increases in irrigated farmland (USDA NASS, 2008 and others; Figure I-4).

The amount of groundwater pumped for irrigation, applied to fields, and consumptively used (i.e., evapotranspired) is somewhat uncertain. In the vicinity of the Little Plover River, growers estimated from recollection 2006 irrigation amounts averaging 4.4 - 6.1 in depending on crop (Clancy et al., 2008). In 2007, growers were required to report irrigation pumpage under the terms of Wisconsin's new groundwater management law, and thus more robust irrigation estimates should have been achievable. Irrigation pumping for that year (considered a dry one), averaged 12.5 in for 27 fields in the Little Plover vicinity. In 2008, irrigation amounts in the Little Plover area were about 10.9 – 12.4 in (Technical Memorandum 12). The 2007 and 2008 irrigation amounts were about the same as groundwater recharge in an average year (Weeks et al., 1965; Chapter VI of this report).

Early work (e.g., Weeks et al., 1965; Weeks and Stangland, 1971) estimated increased evapotranspiration, and hence reduced groundwater recharge, on irrigated lands amounted to 1-4 in above other land covers. Lowery et al. (2009) are in agreement with these amounts, but note they hope to make additional refinements in their estimates. Current best estimates from plant-soil-atmosphere models are that irrigation results in an average 2 in recharge reduction compared with perennially vegetated lands (W. Bland, pers. comm.).

Potential stream depletions and water table declines were also estimated in early work. Weeks and Stangland (1971) calculated that in the vicinity of Plainfield, a landscape consisting of one-fourth irrigated lands would deplete streamflows by 25-30% and drop the water table about 0.5 ft. During drought years, headwater streamflow depletions might amount to 70-90%, and water table declines might reach 2 to 3 ft near groundwater divides. For half the landscape in irrigated cover during drought periods, headwater streams were predicted to be nearly or completely dry, and water table declines might be 5 ft beyond weather-related declines. The area of predicted maximum water table decline was in an area where lakes, including Long Lake – Oasis near Plainfield, appear to be most impacted. As the area land cover comprises more than half irrigated lands, these projections may underestimate the current impacts of irrigation pumping.

The Little Plover River, a formerly top producing trout stream, has been noticeably impacted by groundwater pumping since the 1970s (Clancy et al., 2009). During 2005-2007 pumping captured 3.2-5.4

cfs of discharge (Hoover Rd. location), provoking dry-ups in the upper reaches of the stream. The missing water was attributed to municipal/industrial pumping (2.1 cfs) and agricultural pumping (average 1.1 to 3.3 cfs with seasonal peaks of 1.7 to 5 cfs).

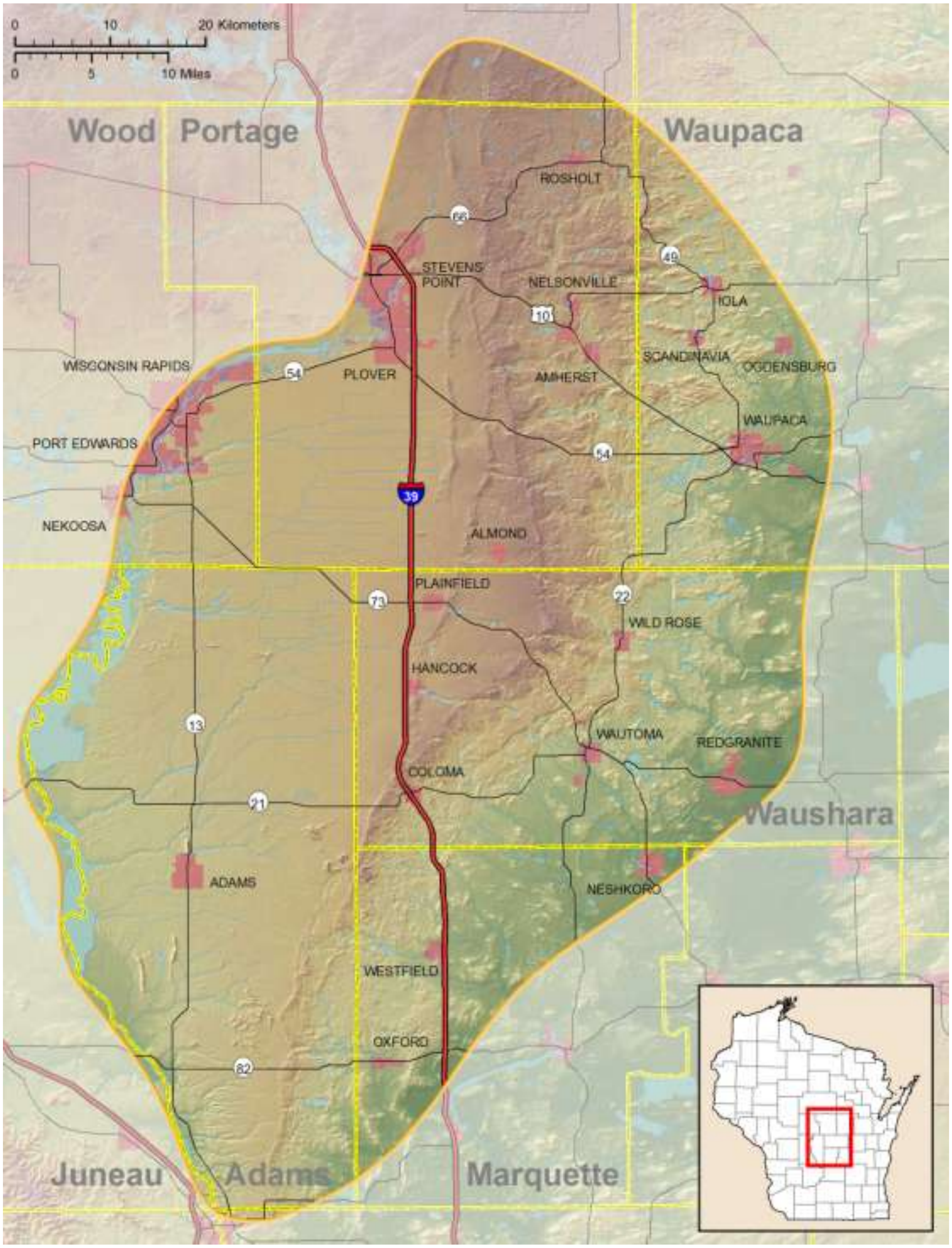


Figure I-1. The Wisconsin Central Sands region with select municipalities and roads shown.

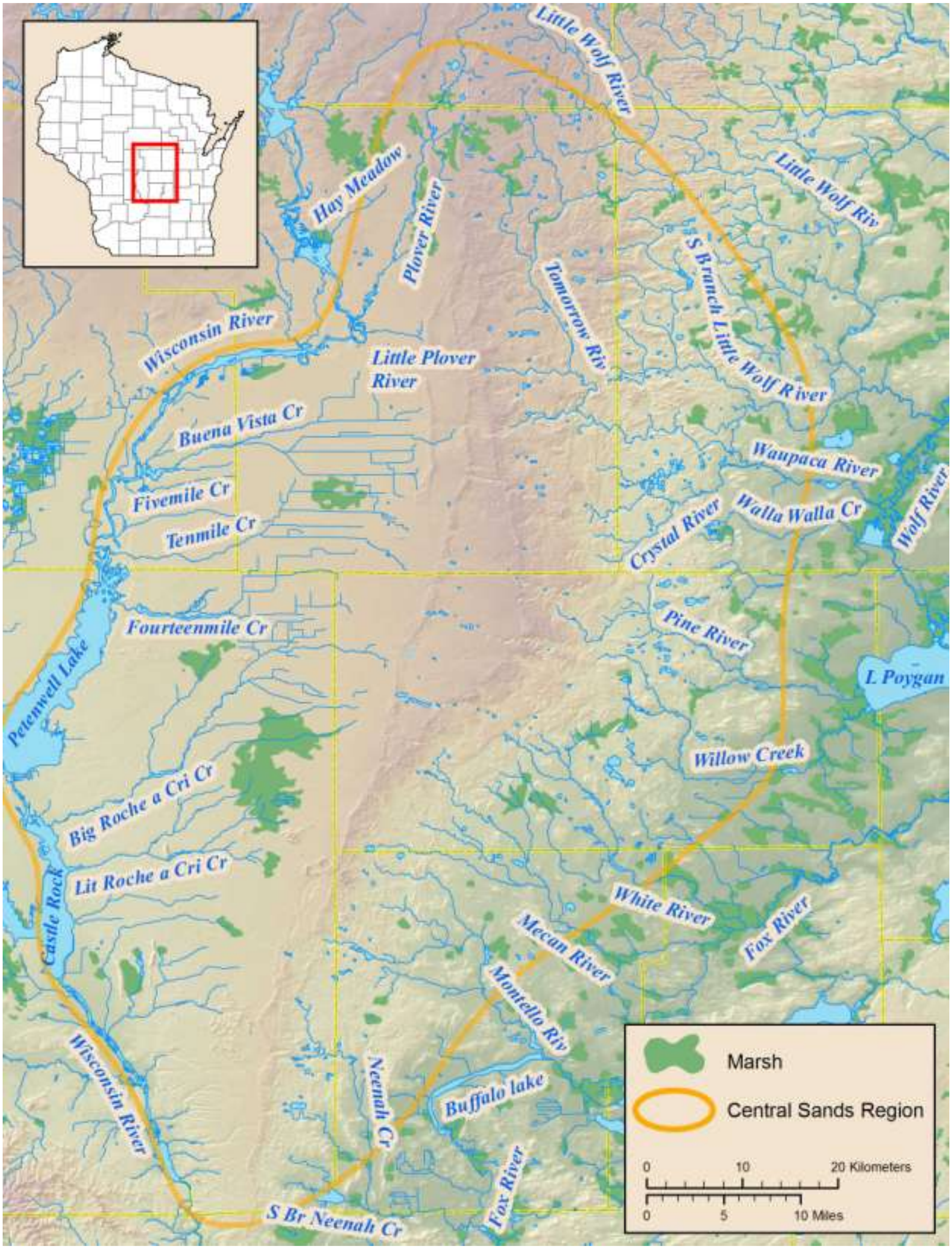


Figure I-2. Hydrography of the Wisconsin Central Sands region.

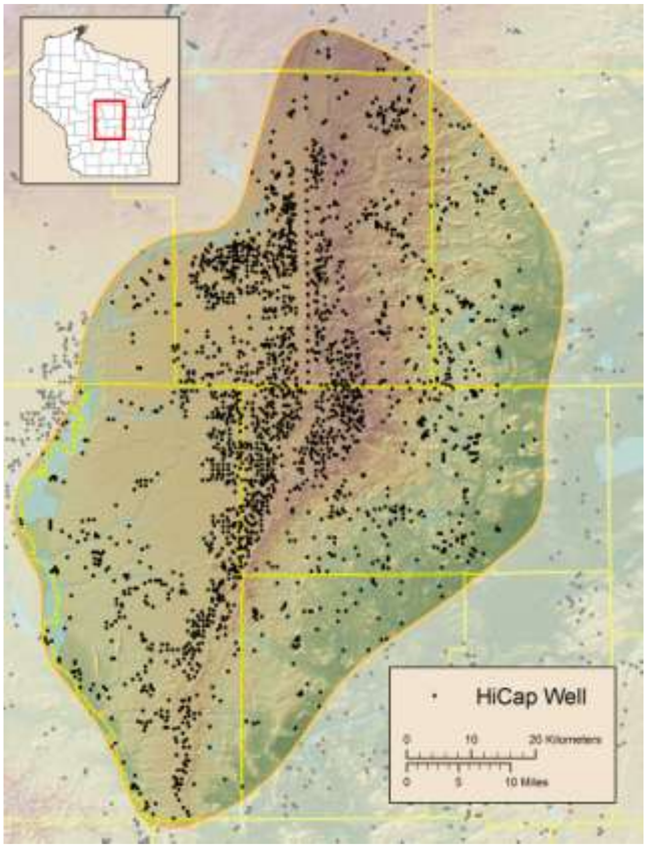


Figure I-3. Locations of high capacity wells in the Wisconsin Central Sands.

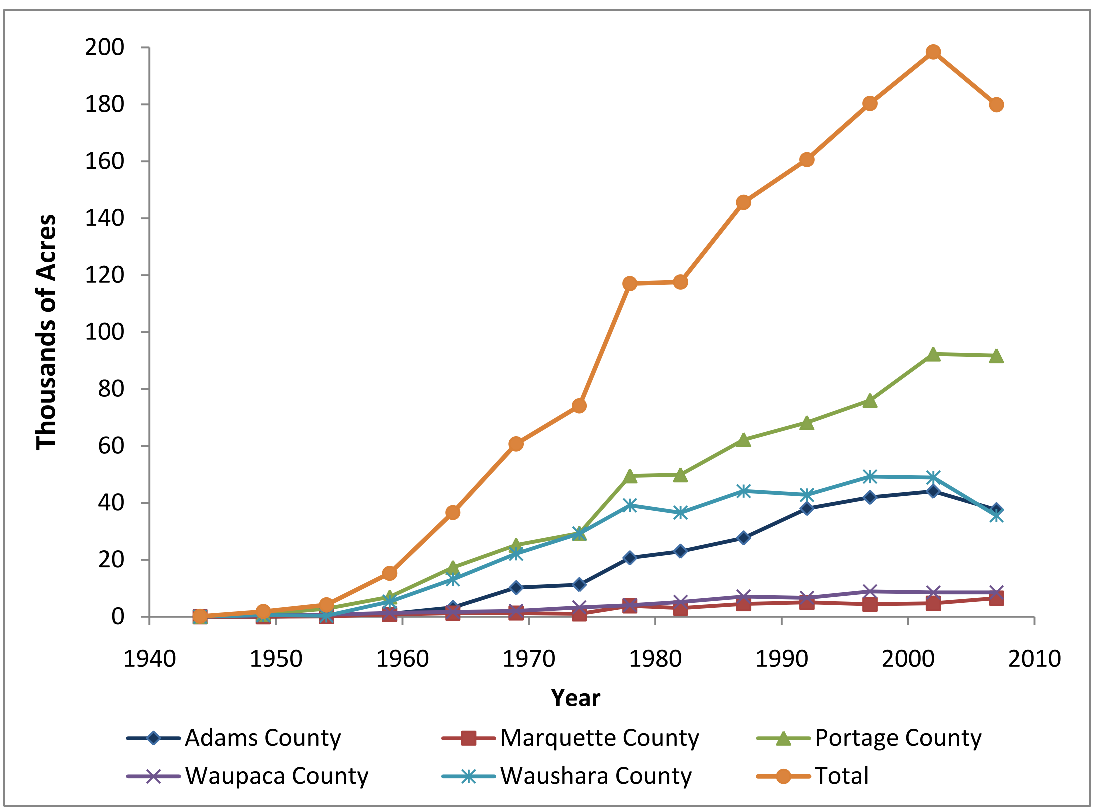


Figure I-4. Increase in irrigated acres in five Central Sands Counties.

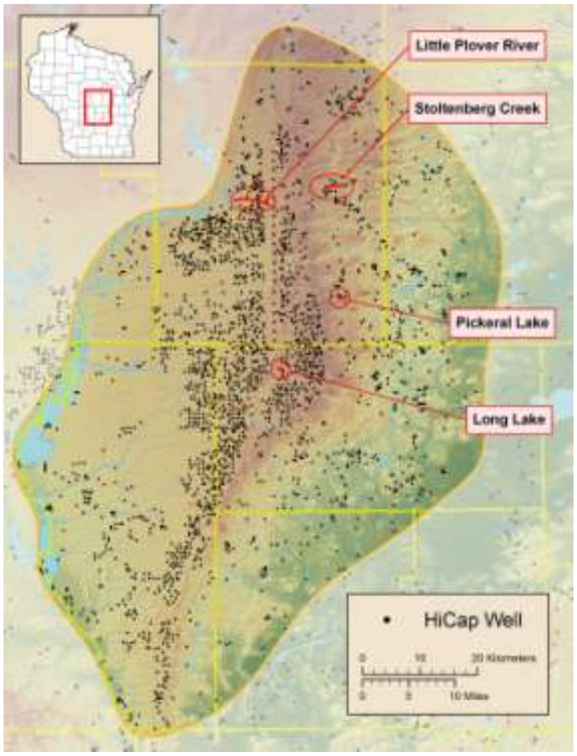


Figure I-5. Central Sands lakes and streams affected by pumping. Clockwise from top left: map of affected lakes and streams, dried up stretch of the Little Plover River, dried up stretch of Stoltenberg Creek, low water levels on Pickerel Lake, and dried up portion of Long Lake.

II. INFERENCES ON RECENT CLIMATE AND HYDROLOGIC CONDITIONS

Summary

Precipitation, stream discharge, and groundwater and lake level information indicate that recent Central Sands (2000-2008) hydrologic conditions have not been abnormally depressed due to some rare climatic conditions.

Annual precipitation in 2000 to 2004 was mostly average to above average for Stevens Point, Hancock, and Wautoma, and in 2005-2008 was slightly below average for Stevens Point and average to slightly above average for Hancock and Wautoma. Long term precipitation trends demonstrate an increase after 1970 by 0.7 to 2.8 in yr^{-1}, depending on gauge location, compared with 1940-1970. We hypothesize that the effects of rapid irrigation expansion in the Central Sands may have been masked by increases in precipitation at the same time.

Discharges in reference streams in areas with few high capacity wells were about average in 2000-2004, somewhat low (8-18 percentile) in 2005-7, and average to slightly below average in 2008 (25-50 percentile). Reference streams provide a history of notable past low flow periods which include 1931 to 1934, 1948-9, 1957-9, 1964, 1977, and 1988. Compared to these, 2000-2008 discharges were more robust. The 1930s contained the lowest flows of record, followed by the late 1950s.

Groundwater levels in areas with few high capacity wells were at 50 year lows in 1958-9, and displayed other lows similar to those in the stream discharge record. Groundwater levels in 2000-2005 were slightly below to slightly above average, and in 2006-8 were lower than average, about the 11-16 percentile, but not at rare nor record lows. A long-term record for a single lake relatively unaffected by pumping agreed well with the groundwater level record.

Precipitation

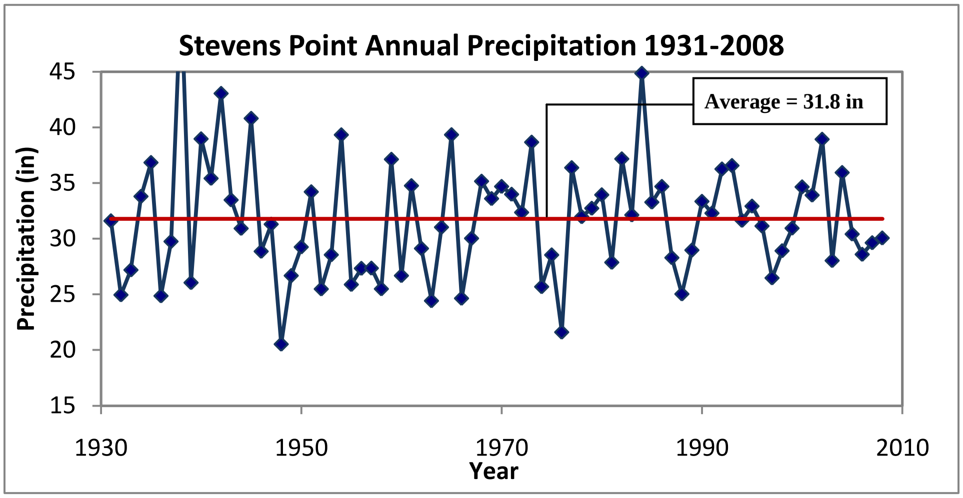
Precipitation records for Stevens Point, Hancock, and Wautoma were evaluated for indicators of long term and recent wetness and dryness. The records for Stevens Point and Hancock were virtually complete, so gaps in the record had to be filled by estimation from other locations on only a few dates (Technical memos 9 and 10, included as electronic media with this report), whereas the entire Wautoma record was interpolated from other locations using the methods of Serbin and Kucharik (in press). Average annual precipitation was 31.8 in at Stevens Point (1931-2008), 31.1 in at Hancock (1903-2008), and 31.1 in at Wautoma (1931-2008) (Figure II-1; only post 1930 data shown). Annual precipitation in 2000 to 2004 was mostly average to above average for all stations. Years 2005-2008 were slightly below average for Stevens Point and average to slightly above average at Hancock and Wautoma.

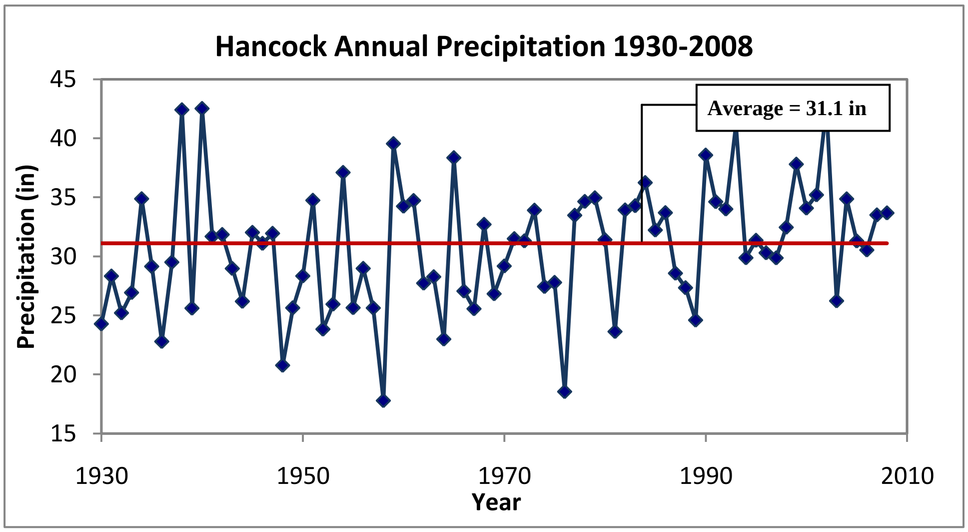
Recent research suggests that wetter conditions have prevailed over much of the eastern US since

1970, including parts of Wisconsin (Juckem et al., 2008). Applying the methodology of Juckem et al., we found conditions have also been wetter in central Wisconsin since 1970 (Figure II-2). Years 1970 through 2008, compared with 1940 through 1970, had an average precipitation increase of 0.7 in yr^{-1} at Stevens Point, 2.2 in yr^{-1} at Hancock, and 2.8 in yr^{-1} at Wautoma (Table II-1). The effects of rapid irrigation expansion in the Central Sands may have been masked by increases in precipitation at the same time.

Table II-1. Comparison of average annual precipitation, 1940-1970 and 1971-2000.

Station	Average Precipitation (in) 1940 - 1970	1971 - 2000
Stevens Point	31.4	32.1
Hancock	29.6	31.8
Wautoma	29.7	32.5





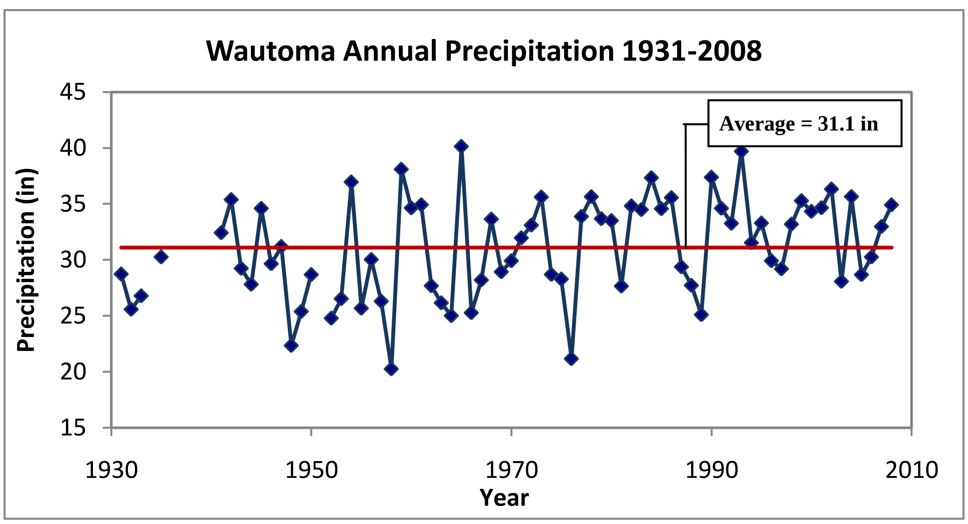
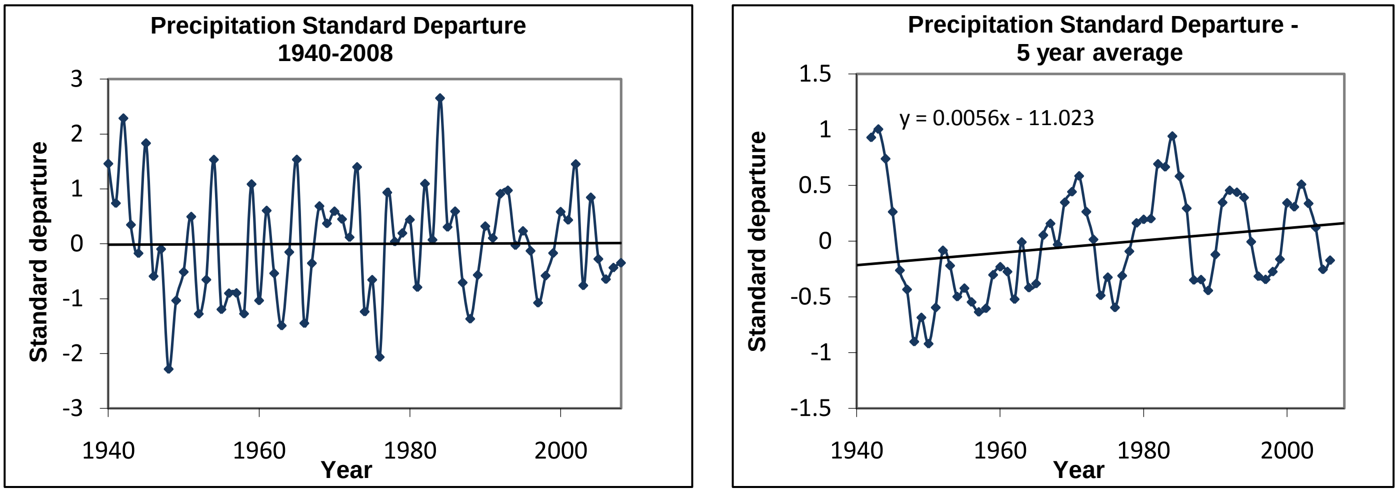
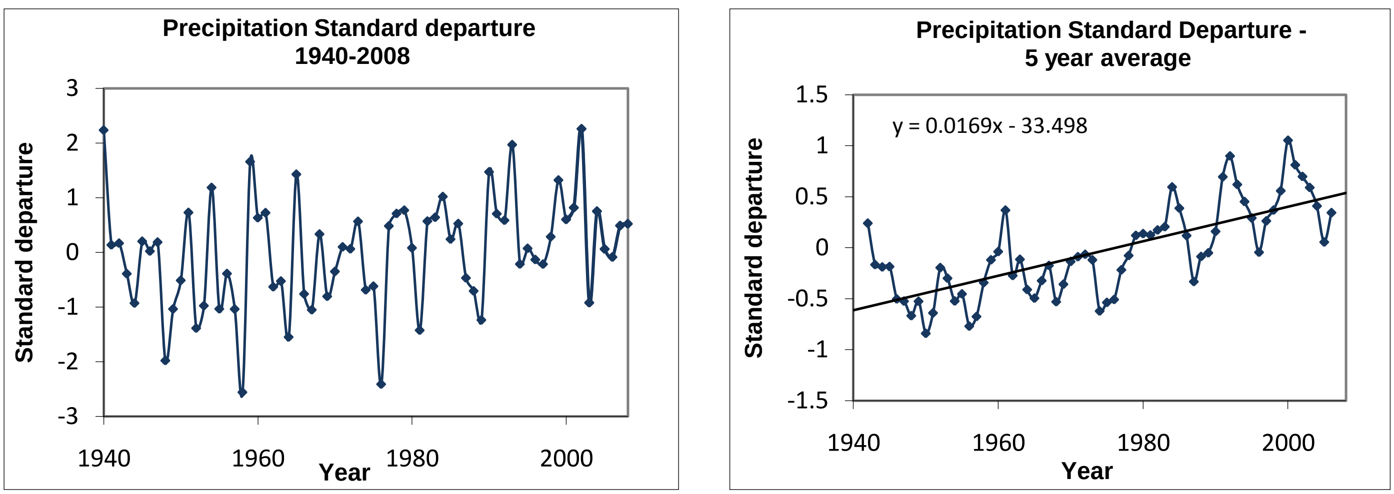


Figure II-1. Precipitation at Stevens Point, Hancock, and Wautoma. Stevens Point and Hancock are from actual record with a few extrapolated values. Wautoma is based on interpolations using methods from Serbin and Kucharik (2009).

Stevens Point



Hancock



Wautoma

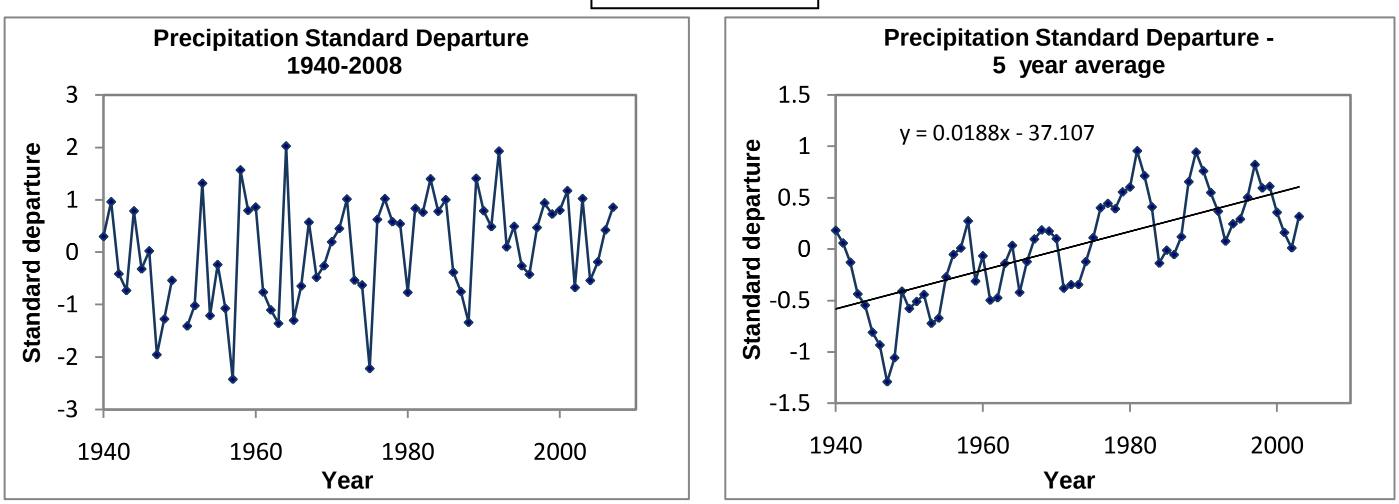


Figure II-2. Standard departure of annual precipitation and five year average of the standard departure for Stevens Point, Hancock, and Wautoma.

Drought Index

The Palmer Drought Index is an indicator of climatic dryness based on precipitation and temperature. Hence, it is an improvement on precipitation alone as an indicator of drought conditions, as it contains an algorithm that uses temperature as a surrogate for evapotranspiration. The Palmer Drought Index indicates that central Wisconsin has been moderately droughty to very moist since about 2000 (Figure II-3). Recent conditions are not particularly dry compared with much of the historical record.

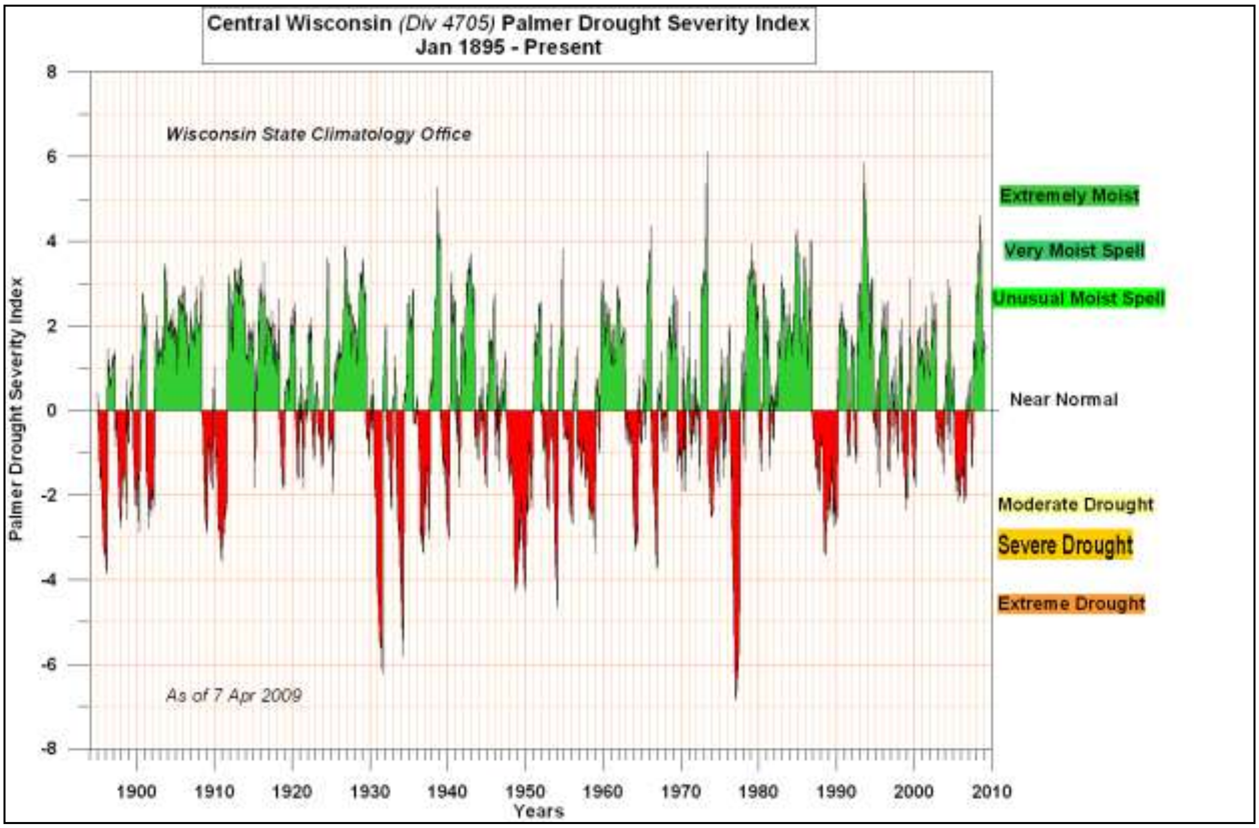


Figure II-3. Palmer drought index graph for central Wisconsin ending spring 2009, produced by the Wisconsin State Climatology Office (2009). Note that the post-2000 period is not substantially droughty compared to the historical record.

Discharges on reference streams

Long term annual discharge records of several area stream gauging stations provide context for how depressed current hydrologic conditions are by comparison. Figure II-4 displays the percentile rank of annual streamflows for six streams: Wolf at New London (1914-2008), Wisconsin at Wisconsin Dells (1935-2008), the Embarrass at Embarrass (1920-2008 with nine missing years), Waupaca at Waupaca (1917-1985 with 18 missing years), and Ten Mile at Nekoosa (1964-2008 with 22 missing years). Significant low flow periods (defined as percentile ranks of 10% or less, which amounts to about a 10 year return frequency) during the past ~ 90 years, include 1931 to 1934, 1948-9, 1957-9, 1964, 1977, and 1988. The 1930s had the smallest discharges of the record. Years 1948 to 1964 mark a long period when low flows were unusually common (6 of 17 years). Years 2000-2004 were about average while 2005-7 discharges were somewhat low. Annual average stream discharges in 2008 were about 25-50 percentile.

Table II-2 lists the lowest 20 year annual discharges in about 90 years for two streams with fairly complete records. Years 1931 and 1934 are indicated as the driest years of almost the last 100, with some years of the 1950s and 1960s making the lowest 10. No years since 2000 are in the driest 10. Year 2005, when the Little Plover first dried and Long Lake – Oasis was at a critical low, was not among the driest 20.

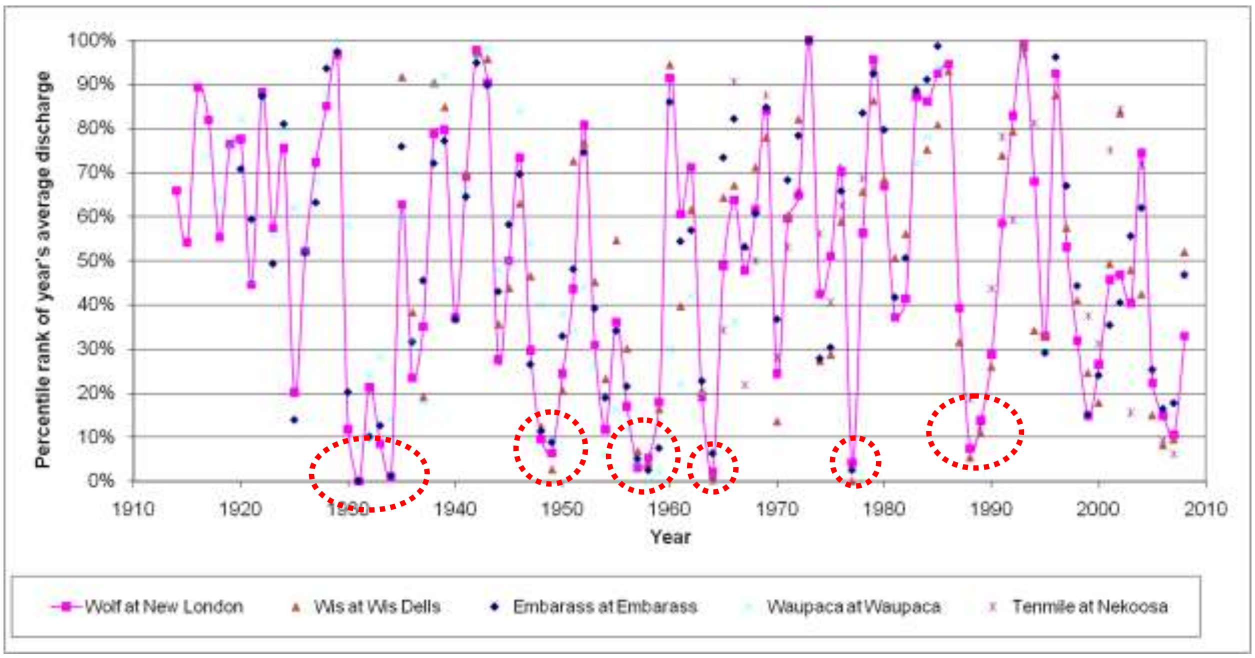


Figure II-4. Percentile rank of streamflows by year, ending 2008. Connecting line is for Wolf at New London only. Significant dry periods (percentile rank <10%) are highlighted by red circles. Note that while streamflows in 2005-7 were dry, the degree of dryness was not uncommon.

Table II-2. Lowest 20 flow years since the early 1900s for two rivers with long record, ranked in order of increasing flow.

Wolf @ New London 1914-2008	Embarrass @ Embarrass 1920-1985, 1994-2008
1931	1931
1934	1934
1964	1958
1957	1977
1977	1957
1958	1964
1949	1959
1988	1949
1933	1932
1948	1948
2007	1933
1930	1925
1954	1999
1989	**2006**
1999	**2007**
2006	1954
1956	1930
1959	1956
1963	1963
1925	**2000**

Groundwater levels in areas with few high capacity wells

The average annual hydrographs of four long-term monitoring wells located in areas with few high capacity wells (Figure II-5 and Figure II-6) are generally consistent with the highs and lows shown in stream discharge (Figure II-4) and precipitation records (Figure II-1). Because of well locations, these hydrographs are only slightly affected by high capacity well pumping (Chapter VII) and are dominantly controlled by climatic conditions. As such, they serve as a reference for how the area's groundwater levels respond in the absence of pumping.

Groundwater levels were at the 50 year low in 1958-9, mostly rose through about 1974, and have cyclically fluctuated since. The 2000-2005 levels at two stations with available records were slightly below average for Amherst Junction and slightly above average for Wautoma. Levels in 2006-8 were lower than average at both locations, but not at rare or record lows. Years 2007 and 2008 were at the sixth and eighth lowest (Table II-2) in 51 years at Amherst Junction (11 and 16 percentile), and 2007 was the seventh driest year at Wautoma (14 percentile).

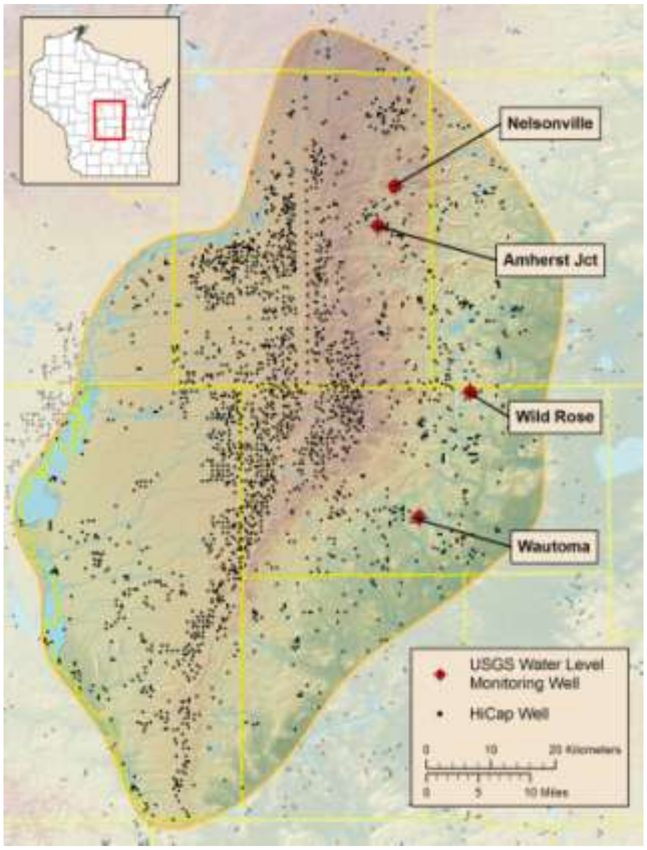


Figure II-5. Locations of four long term monitoring wells in areas with few high capacity wells.

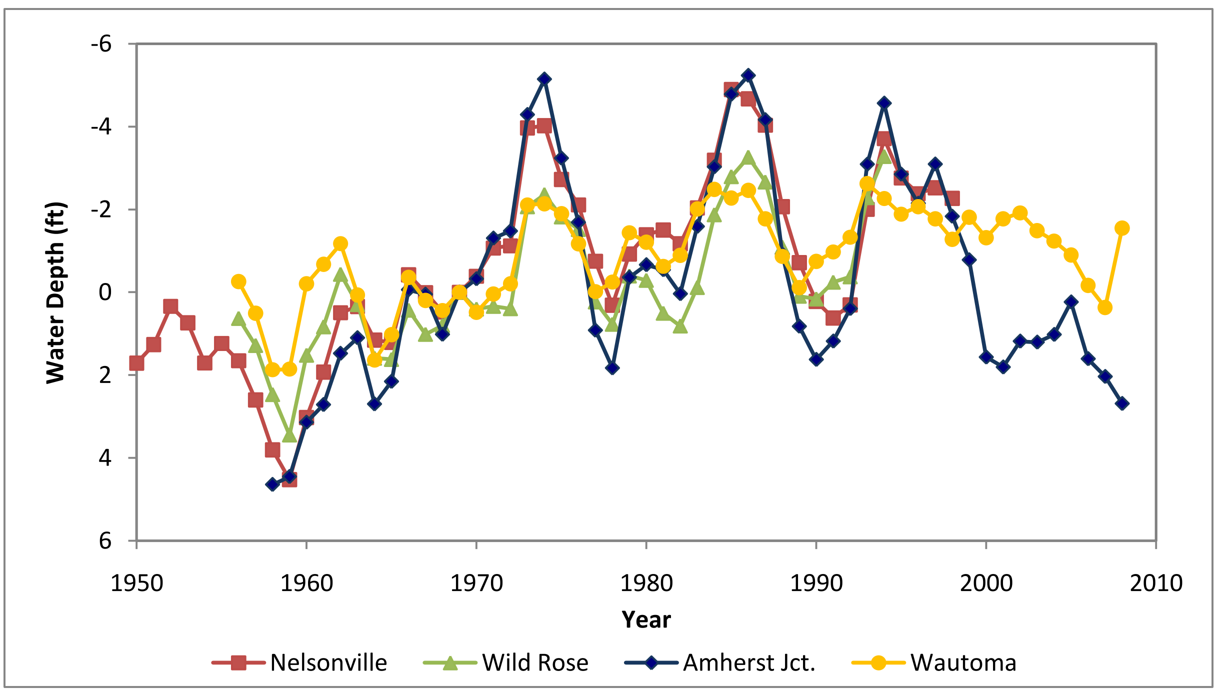


Figure II-6. Annual average depth to water in four long term USGS monitoring wells. For display purposes, water levels were adjusted so that the 1969 value of each is zero.

Table II-3. Lowest ten groundwater levels (1958-2008) for two wells in areas with few high capacity wells, ranked in order of increasing level. No years since 2000 are in the lowest ten percentile.

Amherst Jct.	Wautoma
1958	1958
1959	1959
1960	1964
1961	1965
1964	1970
2008	1968
1965	**2007**
2007	1967
1978	1963
2001	1971

Lake level in an area with few high capacity wells

Few lakes, especially those located in areas with few high capacity wells, have a detailed long term water level record. An exception is Long Lake – Saxeville, where a detailed record from 1958 to present exists, plus two additional measurements from 1947 and 1950 (Figure II-7). (This is not to be confused with Long Lake – Oasis near Plainfield, which has dried in the last few years.)

The Long Lake –Saxeville record correlates closely with that of a long term monitoring well at Wautoma ($r^2 = 0.82$). Lake levels were at a record low in 1959, and rose thereafter before dipping briefly in 1964. Long Lake – Saxeville exhibited a rise from 1964 through about 1974, and has mostly fluctuated cyclically since. In 2000-2006, lake levels remained above their long term average. Levels dropped briefly in 2007 to 1964 levels, and rebounded some in 2008.

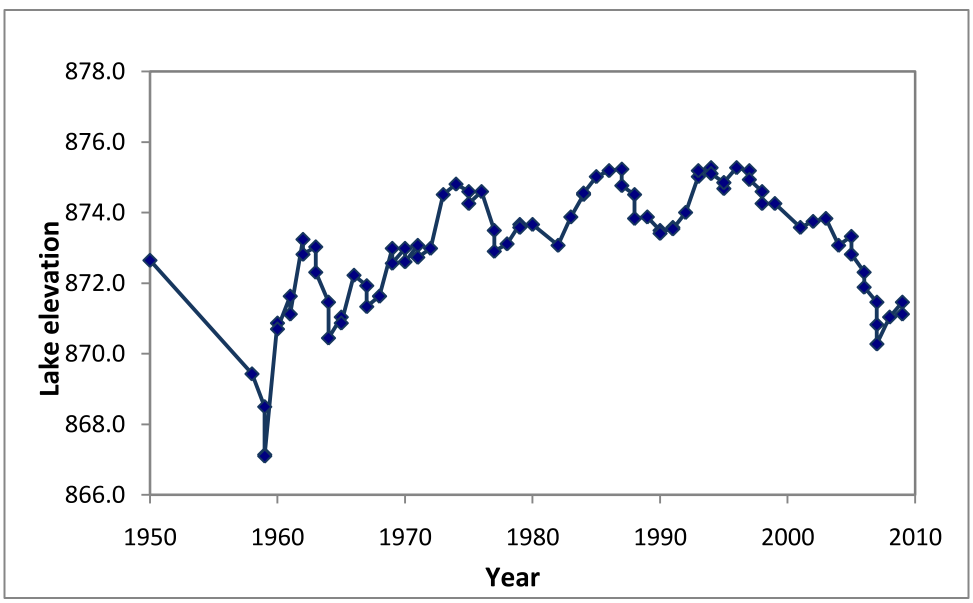


Figure II-7. Hydrograph of Long Lake – Saxeville (not to be confused with Long Lake – Oasis, which dried in 2006.)

III. AVAILABLE RECORD OF GROUNDWATER LEVELS, LAKE LEVELS, AND STREAM DISCHARGES

Summary

The available record of groundwater levels, lake levels, and stream discharges for the study area is both spatially and temporally sparse. The broadest set of groundwater level records is from the USGS, which has archived some 1300 sites. However, most groundwater level sites (1140) have only a single measurement, and only 66 have measurements post-dating 1990. Eight sites have sufficient data for exploring groundwater level trends over the last half-century. Lake level data were available for some 39 lakes, not all of which were sufficient for trend analysis. Stream discharge data for the vicinity are available from USGS daily sites, USGS miscellaneous sites, and a 2006-7 Fox-Wolf basin baseflow study. Stream discharges were measured as part of this study at 41 sites on 30 streams at roughly monthly intervals. Stream discharge data produced for this study were insufficient for evaluating long term discharge trends.

Groundwater level record

The broadest set of available groundwater elevation data is measured and archived by the United States Geological Survey (http://waterdata.usgs.gov/wi/nwis/gw). While the data set spans a large number of central Wisconsin locations, some 1300 sites, and a long time period (1932 to present), most sites (1140) have only a single measurement, and only 66 have measurements post dating 1990. Much of the data set is useful for groundwater elevation mapping and for groundwater flow model calibration (Chapter VI), but not for trend analysis due to the short record, sparse measurements, or influence by localized conditions making them unrepresentative of the broader landscape.

Eight sites had sufficient record for exploring groundwater level trends over the last half-century (Figure III-1, Table III-1). Six of the eight are still being monitored. Two sites, Wisconsin Dells (AD-17/06E/08-0076) and Adams (AD-15/06E/21-0128), were initially promising but were later rejected due to sparseness of record, potential influences of nearby impoundments, uncertainty about the formations being monitored, and confounding by municipal pumping.

A note on the Plover site (PT-23/08E/25-0376): three wells have been located at this site over time with water levels recorded under two different well numbers. Data explored in this study use combined information from these three wells referenced to a common datum. Additional information is presented in Technical Memoranda 4, 5.1, and 6.1, included as electronic media with this report.

Table III-1. Useful USGS water level monitoring wells with long term records.

USGS Station Name	Local or Quadrangle	Well Depth (ft)	First Observation	Last Observation	Number of Observations
PT-24/10E/28-0015	Nelsonville	52	8/24/1950	12/12/1998	1315
PT-23/10E/18-0276	Amherst Jct.	17.4	7/2/1958	2008 +	1687+
PT-23/08E/25-0376	Plover	19	12/1/1959	2008+	1040+
WS-18/10E/01-0105	Wautoma	14	4/18/1956	2008 +	15761+
WS-19/08E/15-0008	Hancock	18	5/1/1951	2008 +	17373+
PT-21/08E/10-0036	Bancroft	12	9/7/1950	2008 +	1550+
PT-21/07E/31-0059	Coloma NW	15.3	8/8/1951	2008 +	665+
WS-20/11E/02-0053	Wild Rose	177	2/6/1956	5/20/1994	442

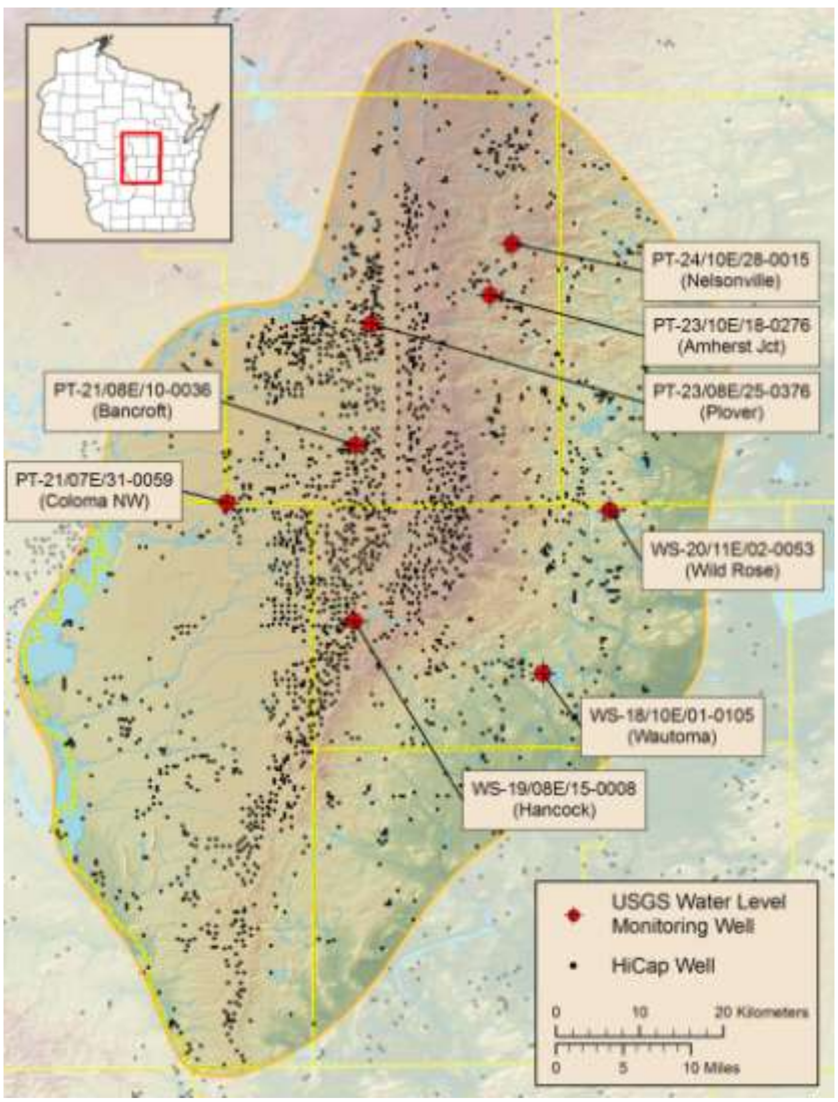


Figure III-1. Location of eight USGS monitoring wells with records sufficient for exploring long term water level trends.

Lake level record

Lake level data were searched for Portage, Waupaca, Waushara, Adams, and Marquette Counties. The search comprised WDNR electronic records, WDNR hard copy records, requests to county Conservation Departments, and requests to lake associations and districts. Level data from 39 lakes (Figure III-2, Table III-2) were collected and compiled in a database. In addition, an excellent record at Long Lake – Saxeville spanning over 50 years was provided by a citizen. The record, containing measurements of distance to water's edge from a benchmark, was correlated with the much shorter water level record measured by Waushara County, resulting in a 50 year lake elevation record.

Similar to groundwater level data, the lake dataset is sparse. On average, 1978 represented the earliest available measurement on a given lake, and the time between measurements taken was about every two years.

Table III-2. Lakes with potentially useful water level information

Lake Name	County	Number of Levels	First Lake Level	Last Lake Level	Avg. Yrs Between Levels
Bean's Lake	Waushara	11	7/10/73	8/3/07	3.10
Big Hills Lake (Hills)	Waushara	10	9/7/95	7/30/07	1.19
Big Silver Lake	Waushara	23	5/14/66	8/1/07	1.79
Big Twin Lake	Waushara	13	6/18/75	7/30/07	2.47
Burghs Lake	Waushara	18	9/7/73	8/1/07	1.88
Crooked Lake	Adams	12	6/14/1973	6/20/1989	1.34
Curtis Lake	Waushara	10	9/12/95	8/3/07	1.19
Deer Lake	Waushara	11	7/28/93	8/1/07	1.27
Fenner Lake	Adams	8	4/25/1974	6/13/1985	1.39
Fish Lake	Waushara	11	7/10/73	8/3/07	3.10
Gilbert Lake	Waushara	28	5/10/62	7/30/07	1.62
Huron Lake	Waushara	13	7/3/73	8/3/07	2.62
Irogami Lake	Waushara	24	1/1/31	8/1/07	3.19
John's Lake	Waushara	11	7/28/93	8/3/07	1.27
Jordan	Adams	20	9/8/1967	9/6/1990	1.15
Kusel Lake	Waushara	26	9/30/63	7/30/07	1.69
Lake Lucerne	Waushara	22	9/30/63	8/1/07	1.99
Lake Napowan	Waushara	14	5/21/85	7/30/07	1.59
Lime	Portage	6	10/2/1940	11/7/1994	9.02
Little Hills Lake	Waushara	7	8/3/01	8/1/07	0.86
Little Silver Lake	Waushara	11	7/20/93	7/30/07	1.28
Little Twin	Waushara	12	5/21/85	8/19/05	1.69
Long Lake	Waushara	23	8/16/61	8/3/07	2.00
Long Lake Saxeville[1]	Waushara	14	11/3/87	7/30/07	1.41
Long Lake Saxeville[2]	Waushara	81	7/1/1947	7/1/2007	1.35
Marl Lake	Waushara	10	4/1/98	8/3/07	0.93
Norwegian	Waushara	12	6/23/75	7/30/07	2.68
Parker	Adams	13	5/26/1983	9/6/1990	0.56
Patrick	Adams	9	5/6/1977	6/16/1986	1.01
Pearl	Waushara	11	6/17/75	8/1/07	2.92
Pine Lake Hancock	Waushara	15	7/10/73	8/3/07	2.27
Pine L (Springwater)	Waushara	27	2/8/61	7/30/07	1.72
Pleasant Lake	Waushara	21	7/9/64	8/3/07	2.05
Porter's Lake	Waushara	6	7/26/02	8/3/07	0.84
Round Lake	Waushara	9	4/1/98	7/30/07	1.04
Sharon	Marquette	72	11/17/84	5/31/1994	0.13
Spring Lake	Waushara	18	10/1/63	8/1/07	2.44
Twin Lakes Westfield	Marquette	11	6/6/02	8/23/2004	0.20
Wilson Lake	Waushara	13	6/18/75	8/3/07	2.47
Witter's Lake	Waushara	20	10/6/63	8/3/07	2.19

[1] Record provided by Waushara County and WDNR

[2] Distance of benchmark to water provided by Long Lake resident.

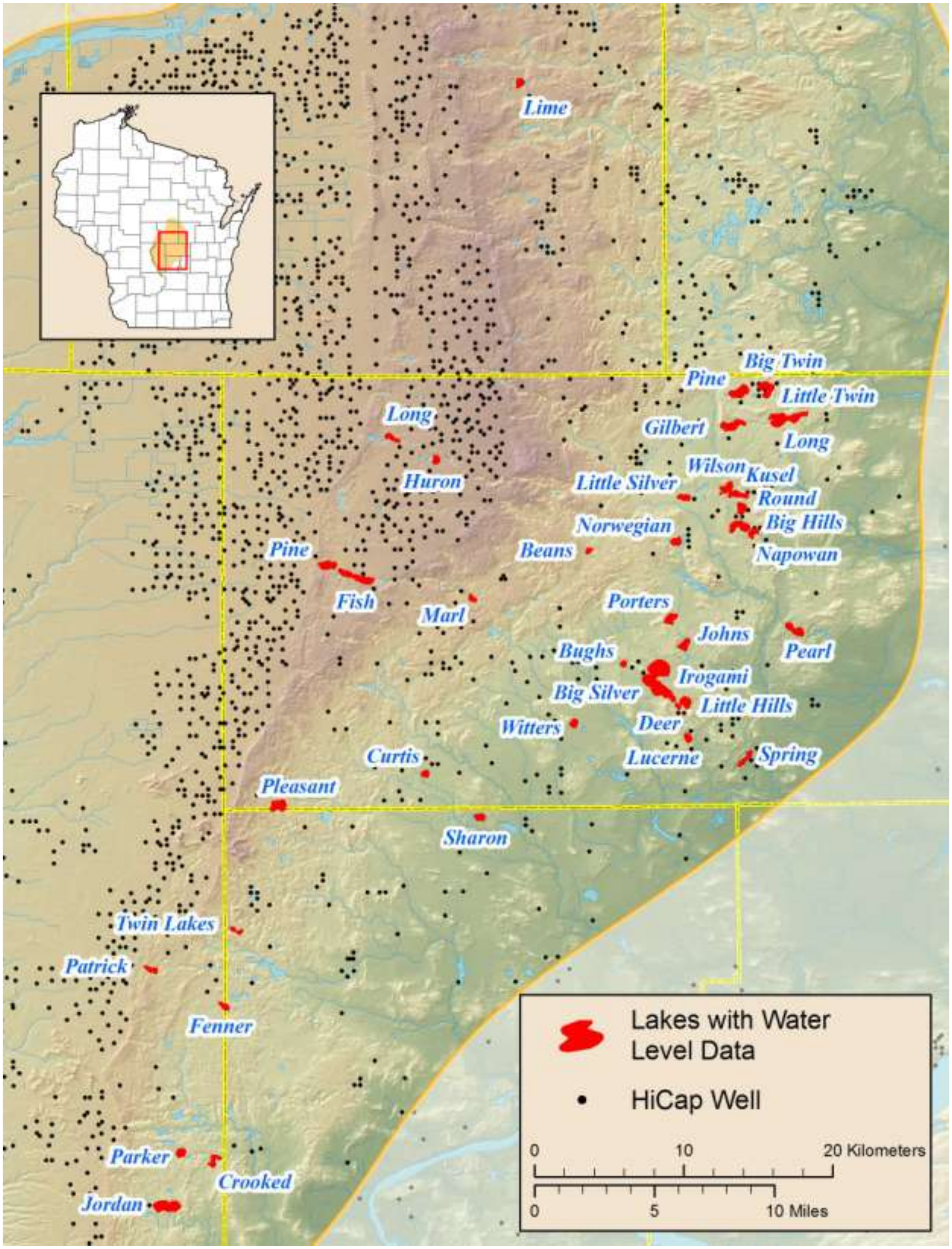


Figure III-2. Location of lakes with water level data in the project database.

USGS Stream Daily Record

The stream discharge record inventory for Wisconsin from the USGS website, revealed 71 daily flow sites for basins of geographic proximity (Upper Fox, Wolf, and Castle Rock). After eliminating sites from very small drainages, special projects (e.g., storm sewer flow monitoring), or those across a major hydrologic boundary, about 38 sites of various usability remained. These can be categorized as (Table III-3):

Sites on smaller streams within the project area: 15

Sites on smaller streams adjacent and near the project area: 6

Sites on smaller streams farther from study area: 4

Sites on the Wisconsin, Fox, and Wolf Rivers: 13

Many sites on smaller streams had a limited flow record. Of those within the study area, only one (Tenmile Creek near Nekoosa) is presently operational. Three smaller stream sites adjacent or near the study area (Red, Middle Branch Embarass at Embarass, and Middle Branch Embarass at Wittenberg) are also presently operational. Tenmile has a record spanning greater than 40 years, the Red and Middle Branch Embarass have records spanning 15-20 years, and the Middle Branch Embarass at Embarass has a record of over 80 years.

Other sites with lengthy but non-current discharge records include the Waupaca at Waupaca, Little Wolf at Royalton, Plover near Stevens Point, Little Plover at Arnott, Little Plover at Plover, Fourteen Mile Creek, and Big Roche a Cri near Adams. The variety of sites and length of record on the Wisconsin, Fox, and Wolf provide a useful context for area hydrology.

USGS Stream Miscellaneous Sites

A search of the USGS stream miscellaneous site records revealed some 129 sites within the study area. The record extended back to 1956. On average, 4.2 observations were available per site, with 49 having only a single measurement. The data are useful for groundwater flow model calibration (Chapter VI), but not for trend analysis due to sparseness of measurements.

Fox – Wolf 2006-7 Baseflow Study

Clancy et al. (2008) measured baseflows in headwater streams of the Fox-Wolf Watershed at 304 sites during 2005-6, 139 of which were within the current study area. Measurements were taken during a relative dry period, during which USGS daily discharge gauges averaged 21st and 9th percentiles for 2005 and 2006, respectively. These data were useful for flow model calibration.

Table III-3. List of USGS daily flow sites in or near the study area, with their drainage areas and beginning and end of record. Last updated January 2007.

USGS Site Number	Station Name	Drainage Area (Square Mile)	First Measurement	Last Measurement	Count
UPPER FOX BASIN					
	SMALLER STREAMS, IN STUDY AREA				
4072750	LAWRENCE CREEK NEAR WESTFIELD, WI	13.4	11/1/1967	9/30/1973	2161
4073405	WEST BRANCH WHITE RIVER NEAR WAUTOMA, WI	38.9	10/1/1963	9/30/1965	731
	FOX RIVER LARGE SCALE				
4082400	FOX RIVER AT OSHKOSH, WI	5310	10/1/1991	5/16/2007	5707
4073365	FOX RIVER AT PRINCETON, WI	962	7/1/2001	9/30/2005	1553
4073500	FOX RIVER AT BERLIN, WI	1340		5/16/2007	39947
WOLF BASIN					
	SMALLER STREAMS, IN STUDY AREA				
4080798	TOMORROW RIVER NEAR NELSONVILLE, WI	44	4/9/1993	9/30/1995	905
4080950	EMMONS CREEK NEAR RURAL, WI	25.1	5/15/1968	9/30/1974	2330
4080975	CRYSTAL RIVER NEAR WAUPACA, WI	82	7/16/1971	9/25/1973	679
4081000	WAUPACA RIVER NEAR WAUPACA, WI	265	6/28/1916	9/30/1985	19220
	SMALLER STREAMS, NEAR STUDY AREA				
4077630	RED RIVER AT MORGAN ROAD NEAR MORGAN, WI	114	10/1/1992	5/16/2007	5341
407809265	MIDDLE BRANCH EMBARRASS RIVER NEAR WITTENBERG, WI	76.3	10/1/1989	10/5/2006	6214
4078500	EMBARRASS RIVER NEAR EMBARRASS, WI	384	6/1/1919	5/16/2007	29577
4079602	LITTLE WOLF RIVER NEAR GALLOWAY, WI	22.6	2/9/1973	9/30/1979	2199
4079700	SPAULDING CREEK NEAR BIG FALLS, WI	5.57	6/1/1964	9/30/1966	852
4080000	LITTLE WOLF RIVER AT ROYALTON, WI	507	1/1/1914	9/30/1985	21823
	SMALLER STREAMS, FAR FROM STUDY AREA				
4075200	EVERGREEN CREEK NEAR LANGLADE, WI	8.09	6/1/1964	9/30/1973	3049
4075365	EVERGREEN RIVER BLW EVERGREEN FALLS NR LANGLADE,WI	64.5	12/1/2002	9/30/2006	1400
4076000	WEST BRANCH WOLF RIVER AT NEOPIT, WI	93.2	1/1/1911	2/7/1917	2230
4076500	WEST BRANCH WOLF RIVER NEAR KESHENA, WI	163	3/28/1928	11/12/1931	1325
	WOLR RIVER LARGE SCALE				
4075500	WOLF R ABOVE WEST BR WOLF R NEAR KESHENA, WI	616	10/1/1927	9/30/1962	12784
4077000	WOLF RIVER AT KESHENA FALLS NEAR KESHENA, WI	788	5/10/1907	9/30/1985	27811
4074950	WOLF RIVER AT LANGLADE, WI	463	3/21/1966	5/16/2007	14725
4079000	WOLF RIVER AT NEW LONDON, WI	2260	10/1/1913	5/16/2007	34196
4077400	WOLF RIVER NEAR SHAWANO, WI	816	10/1/1985	6/30/2001	5752
4075000	WOLF RIVER NEAR WHITE LAKE, WI	485	7/1/1935	9/30/1938	1188
CASTLE ROCK BASIN					
	SMALLER STREAMS, IN STUDY AREA				
5400500	PLOVER RIVER NEAR STEVENS POINT, WI	145	1/1/1914	12/31/1951	5113
5400600	LITTLE PLOVER RIVER NEAR ARNOTT, WI	2.24	7/1/1959	7/9/1976	6218
5400650	LITTLE PLOVER RIVER AT PLOVER, WI	19	7/1/1959	9/30/1987	10319
5400853	BUENA VISTA CREEK NEAR KELLNER, WI	53.1	3/1/1964	9/30/1967	1309
5401020	TENMILE CREEK DITCH 5 NEAR BANCROFT, WI	9.73	6/27/1964	9/30/1973	3383
5401050	TENMILE CREEK NEAR NEKOOSA, WI	73.3	10/1/1963	5/16/2007	11827
5401100	FOURTEENMILE CREEK NEAR NEW ROME, WI	91.1	3/1/1964	10/1/1979	5693
5401510	BIG ROCHE A CRI CREEK NEAR HANCOCK, WI	9.61	10/1/1963	9/30/1967	1461
5401535	BIG ROCHE A CRI CREEK NEAR ADAMS, WI	52.8	10/1/1963	10/17/1978	5496

Table III-3. Continued

	WISCONSIN RIVER LARGE SCALE				
5400760	WISCONSIN RIVER AT WISCONSIN RAPIDS, WI	5420	5/21/1914	9/30/2006	30996
5400800	WISCONSIN RIVER-OLD SITE-AT WISCONSIN RAPIDS, WI	5430	10/1/1957	9/30/1981	8766
5401500	WISCONSIN RIVER NEAR NECEDAH, WI	5990	12/1/1902	5/31/1950	6490
5404000	WISCONSIN RIVER NEAR WISCONSIN DELLS, WI	8090	10/1/1934	5/16/2007	26525

Stream Discharges Measured for This Study

Stream discharges were measured during this study at 42 sites on 30 streams at roughly monthly intervals during baseflow (Figure III-3, Table III-4). Discharges were measured to provide new information for locales where little was available, flux targets for groundwater flow model calibration, and modern data for comparison against historical data. We found that comparing modern data against historical was futile because data were too sparse.

Most sites had previous measurement history. Seventeen sites were at or near current and former USGS daily flow sites, and four of the seventeen were gauged as part of the Fox-Wolf project in 2005-6 (Clancy et al., 2008). Eleven sites were USGS miscellaneous measurement sites - those only gauged once to a few times. Four of the eleven sites were also gauged as part of the Fox-Wolf study. Seven more sites had gauging history as part of the Fox-Wolf project, and six sites were new.

Some measurement locations had to be moved from the original USGS or Fox-Wolf locations due to accessibility and practicality issues. Lawrence Creek at Eagle Avenue and the Pine River at Apache Road were moved 0.5 miles downstream, and Witches Gulch at 13 was moved 125 meters downstream. The Buena Vista Creek at 100th Road and Ditch #4 at 100th Road were moved upstream 0.4 and 0.5 miles respectively.

Dams complicated measurements at Little Roche-A-Cri at Friendship Park, Fourteen Mile Creek at Highway 13, Crystal River at County Road K, and the Waupaca River at Harrington Road. Dam influences need to be considered when utilizing the data.

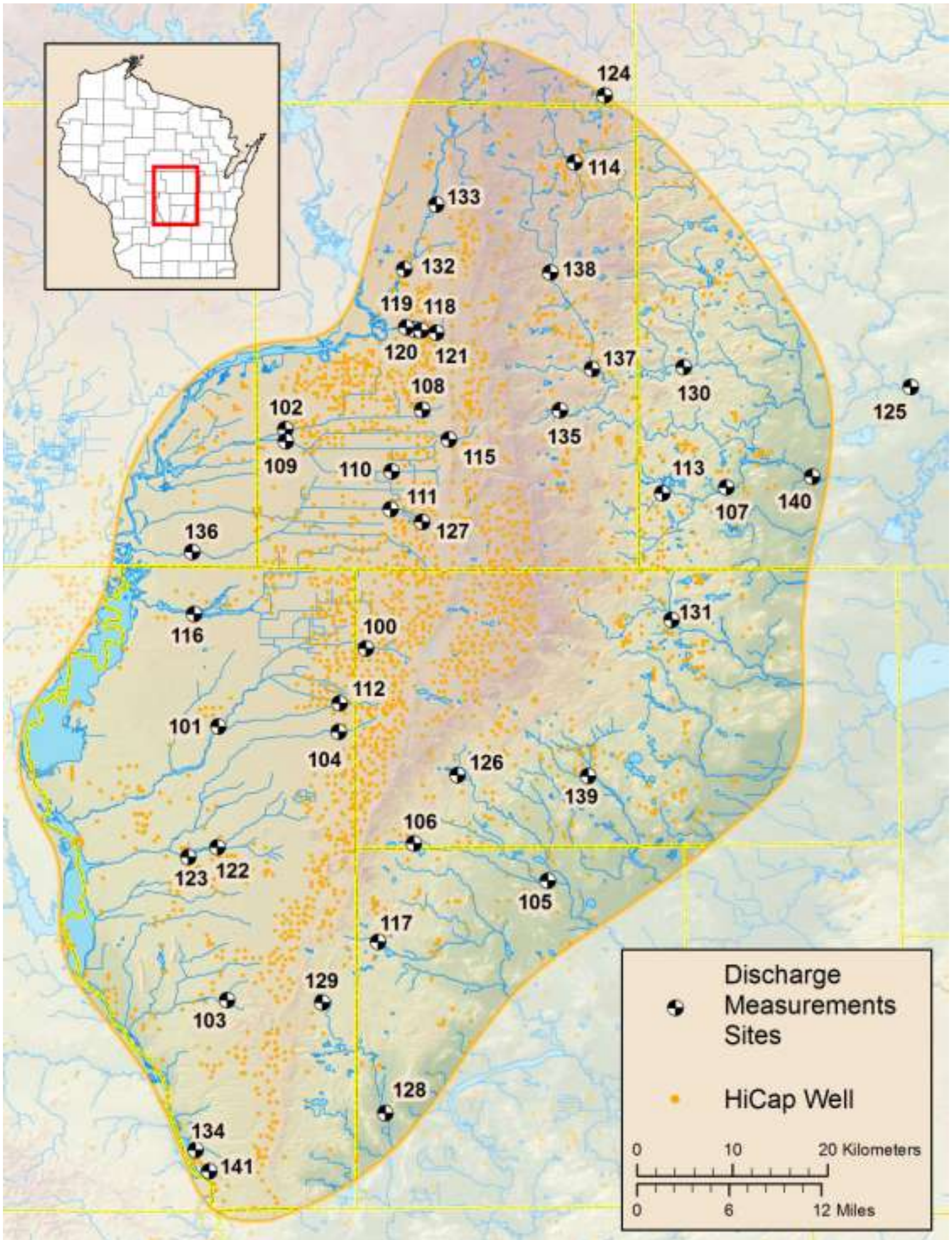


Figure III-3. Discharge measurement sites for this study.

Table III-4. Discharge measurement sites for this study; locations shown in Figure III-3. Also indicated is whether the site had measurements in the USGS Daily or Spot record, or in the Fox-Wolf Project (Clancy et al., 2008).

Map Location	Project Site Name	USGS Site Type1	USGS Years	Fox-Wolf Site	Dam Affected	Comments
100	Big Roche-A-Cri @ 1st Ave	At Daily	1963 - 1967			
101	Big Roche-A-Cri @ Brown Deer Ave	At Daily	1963 - 1978			
102	Buena Vista Creek @ 100th Rd	Near Daily	1964 - 1967			Moved 0.4 Miles Upstream
103	Campbell Creek @ A	At Spot	1971			
104	Carter Creek @ G					
105	Chaffee Creek @ 14th	At Spot	1962 - 1988	Y		
106	Chaffee Creek @ CH			Y		
107	Crystal River @ K			Y	Y	
108	Ditch #2 N Fork @ Isherwood	At Spot	1966			
109	Ditch #4 @ 100th Rd	Near Daily	1964 - 1967			Moved 0.5 Miles Upstream
110	Ditch # 4 @ Taft					
111	Ditch #5 @ Taft	At Daily	1964 -1973			
112	Dry Creek @ G					
113	Emmons Creek @ Rustic Road 23	At Daily	1968 - 1974	Y		
114	Flume Creek in Rosholt @ 66	At Spot	1972 - 1976	Y		
115	Four Mile Creek @ JJ&BB					
116	Fourteen Mile Creek @ 13	At Daily	1964 - 1979		Y	
117	Lawrence Creek @ Eagle	Near Daily	1967 - 1973	Y		Moved 0.5 Miles Downstream
118	Little Plover @ Eisenhower	At Spot	1961 - 1963			
119	Little Plover @ Hoover	At Daily	1959 - 1987			
120	Little Plover @ I-39	At Spot	1961 - 1963			
121	Little Plover @ Kennedy	At Daily	1959 - 1976			
122	Little Roche-A-Cri @ 10th Ave.					
123	Little Roche-A-Cri @ Friendship Park	At Spot	1972 - 1976		Y	
124	Little Wolf @ 49	At Daily	1973 - 1979			
125	Little Wolf @ 54	At Daily	1914 -1985			
126	Mecan @ GG	At Spot	1956 - 1988	Y		
127	NB Ten Mile @ Isherwood/Harding	At Spot	1973			
128	Neenah @ A			Y		
129	Neenah @ G			Y		
130	Peterson Creek @ Q	At Spot	1962 - 1988	Y		
131	Pine River @ Apache			Y		Moved 0.5 Miles Downstream
132	Plover River @ I-39					

Table III-4. Continued

133	Plover River @ Y	At Daily	1914 - 1951			
134	Shadduck Creek @ 13					
135	Spring Creek @ Q			Y		
136	Tenmile Creek @ Nekoosa	At Daily	1963 - 2009			
137	Tomorrow @ A			Y		
138	Tomorrow @ River Rd (Clementson)	At Daily	1995	Y		
139	W Branch White River @ 22	At Daily	1963 - 1965	Y		
140	Waupaca River @ Harrington Rd	At Daily	1916 - 1985		Y	
141	Witches Gulch @ 13	Near Spot	1972 - 1973			Moved 125 Meters Downstream

IV. GROUNDWATER LEVEL TRENDS IN LONG TERM MONITORING WELLS

Summary

In areas of the Central Sands with large densities of high capacity wells, groundwater levels in monitoring wells have declined and since 2000 were at all time lows for their 50 year record. By comparison, groundwater levels in areas with few high capacity wells have been about average to somewhat below average since 2000, but not near record lows and do not show a similar long term decline. Declines beyond those attributable to weather ("missing water") in areas with many high capacity wells ranged from about 3 feet at Plover and Hancock to one foot at Bancroft and Coloma NW. Differences in amounts of missing water are explainable by position in the groundwater flow system and perhaps by amounts of irrigated land cover. Greater amounts of missing water are probably higher in the groundwater flow system. Missing water estimates are likely underestimated by 0.4 to 0.76 ft, as the control locations used in the analysis may be affected by pumping.

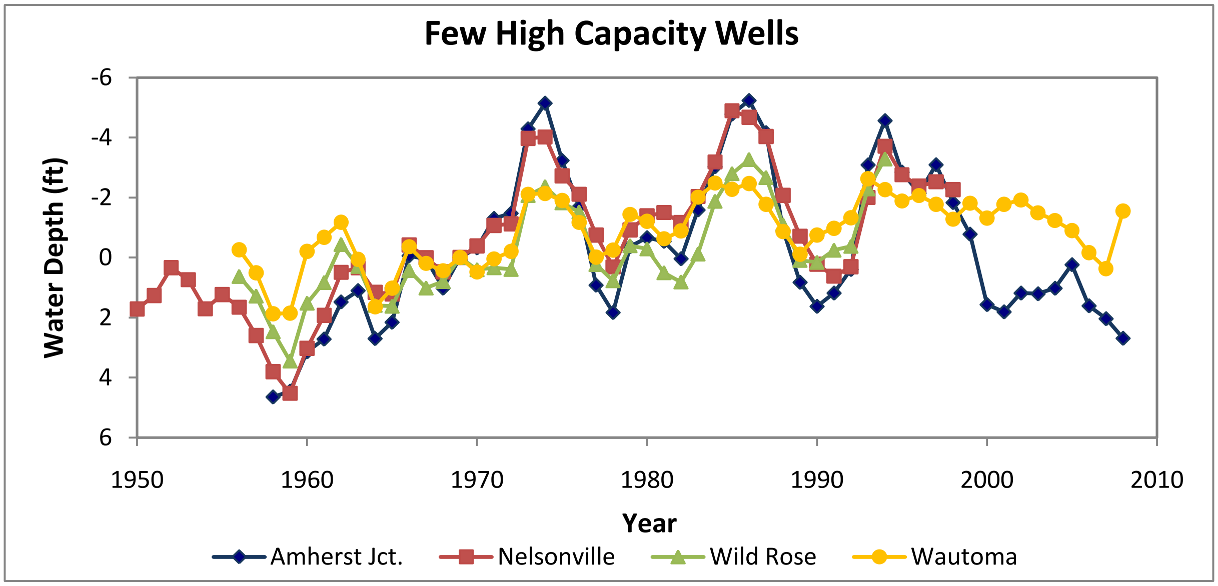
Overview

Eight monitoring well sites have sufficiently detailed and long-term water level records to be useful in examining trends and perhaps teasing out signals of groundwater pumping. Four wells (Amherst Junction, Nelsonville, Wild Rose, and Wautoma) are in areas with relatively few high capacity wells and are designated as "controls," to which the four wells in areas with many high capacity wells (Plover, Hancock, Bancroft, and Coloma NW) can be compared (Figure III-1). Wells in areas with many high capacity wells are designated as "potentially affected."

The water level record suffers several deficiencies. The Wild Rose record terminates in 1994 and that of Nelsonville in 1998; thus they are unable to inform on more current conditions. Some locations have sparse records, particularly Coloma NW. While ideally a long record that pre- and post-dates large-scale pumping development would be available to facilitate before-and-after comparisons, a true pre-development record does not exist, only periods when fewer and greater numbers of high-capacity wells were present. Similarly, control locations would ideally be free of high capacity well influence, while the reality is they are only less affected. (Model-based estimates of pumping impacts at control locations are about 0.4 to 0.76 ft of water level decline; Chapter VI).

Well Hydrographs

Hydrographs of annual average groundwater levels are displayed in Table IV-1, grouped according to control or potentially affected. For display purposes, water levels were zeroed to the measured level of each well in 1969, with positive values indicating a greater depth to water (water level decline) compared to 1969, and negative values a shallower depth (water level rise). Water levels at Coloma NW for years 1983 and 1994 were interpolated from adjacent years because no data was available.



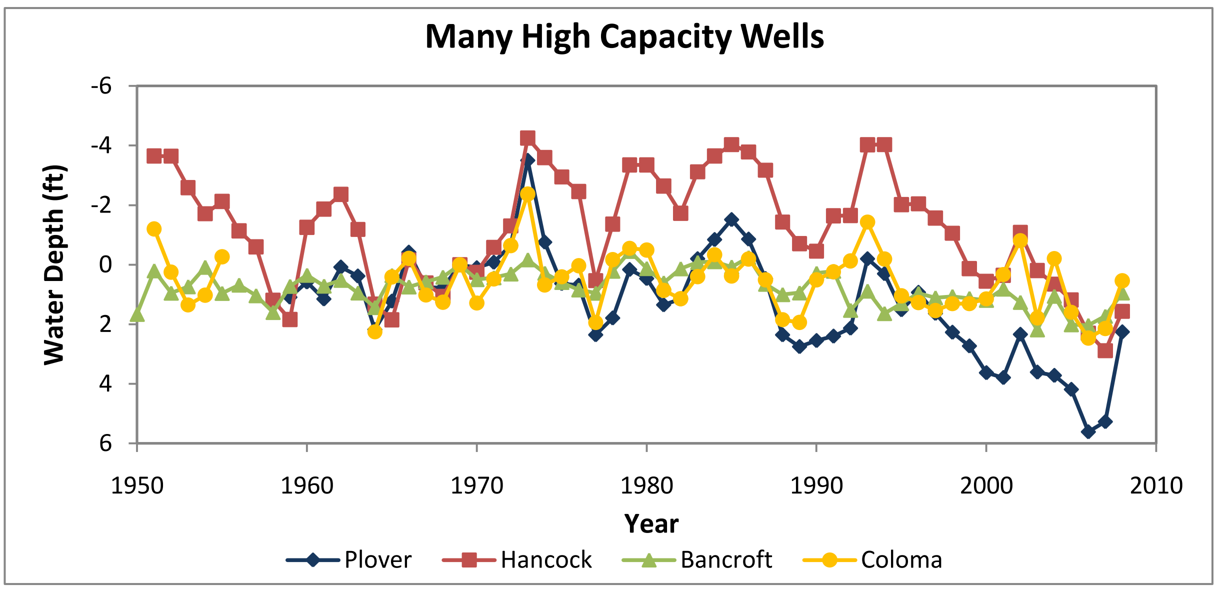


Figure IV-1. Water depths at three monitoring wells in areas with few high capacity wells (top) and four in areas with many high capacity wells. Water depth are adjusted so that 1969 values are zero.

All hydrographs demonstrate common peaks (evident around 1974, 1985, and 1993) and valleys (1959, 1978, 1990, and perhaps 2007), that coincide with indicators of wetter and drier weather (Chapter II). Though water level peaks and valleys coincide, the amplitude and trend differ among wells. Amplitude differences are expected and are predictable by groundwater hydraulics: levels of groundwater near discharge zones are strongly influenced by the water level of the discharge zone, while groundwater levels far from discharge zones are less influenced. Thus, groundwater levels at the Coloma NW and Bancroft locations, which are near groundwater discharge zones, have small amplitudes.

Trend differences conform to whether the hydrograph is for a well in a control location or potentially affected location. Minimum groundwater levels occurred in control locations in 1958-9, consistent with a decade that witnessed some years of the smallest precipitation and stream discharges of the twentieth century (Chapter II). Groundwater levels rose in those locations from 1959 through about 1974, and then until the late 1990s displayed an average cyclical fluctuation. Since 2000, water levels at control locations have generally declined, but not to record levels. Potentially affected locations also demonstrated a low in 1959, but not a low for the record. The Hancock hydrograph generally followed control wells until about 1993 but then dropped precipitously to record lows in 2007 and 2008. Plover has declined since 1973. Both Coloma NW and Bancroft were comparatively flat through their entire record, but have experienced record lows since 2000.

Figure IV-1 shows the lowest 10 annual groundwater levels since 1958 at three wells in areas with many high capacity wells (Plover, Hancock, and Bancroft) compared with two wells in areas with few monitoring wells (Amherst Junction and Wautoma). Since 2000, Plover experienced its lowest seven water levels, Hancock its lowest two, and Bancroft its lowest four. Coloma NW had its first and third lowest years since its record started in 1964. By comparison, control well sites at worst experienced their sixth and seventh lowest years since 2000 in the 1959-2008 record. Thus water levels in potentially affected areas exhibited a decline in water levels over time compared with those in areas with few high capacity wells.

Table IV-1. Comparison of lowest water level years, ranked in order of increasing level, in areas with many and few high capacity wells.

Many high capacity wells			Few high capacity wells	
Plover 1959-2008	Hancock 1958-2008	Bancroft 1958-2008	Amherst Jct 1958-2008	Wautoma 1958-2008
2006	**2007**	**2003**	1958	1958
2007	**2006**	**2006**	1959	1959
2005	1965	**2005**	1960	1964
2001	1959	**2007**	1961	1965
2004	**2008**	1994	1964	1970
2000	1964	1958	**2008**	1968
2003	1958	1992	1965	**2007**
1989	**2005**	1964	**2007**	1967
1999	1968	1995	1978	1963
1990	**2004**	**2002**	**2001**	1971

Estimating the Potential Pumping Influence on Groundwater Levels

The previous discussion shows water levels at potentially affected locations are in long term decline not explainable by climatic variability. Here we estimate the magnitude of this decline through quantitative comparisons of control and potentially affected hydrographs. In essence, control hydrographs are used to subtract out the influence of temporal climate variability on groundwater levels at potentially affected locations. The relation of control and potentially affected well hydrographs in an early period is compared to that relation during late periods. In early periods, pumping influences are hypothesized to be small at both control and potentially affected wells, and hydrograph variability is only climate driven. In late periods, pumping influences are presumed to be more developed. Changes in the relations between early and late periods, if any, are signals of water level change that cannot be accounted for by climatic variability ("missing water"). Linear regressions are used to describe the early and late relations. The method can be written as:

$$\text{Water level decline} = \text{Reg}[h(x_p,t_l) : h(x_c,t_l)] - \text{Reg}[h(x_p,t_b) : h(x_c,t_b)] \qquad \text{[Eqn. IV-1]}$$

where

Reg [Variable 1 : Variable 2] = linear regression function of variable 1 against variable 2

h = groundwater level at location x and time t

p = potentially affected location

c = control location

b = early period time

l = late period time,

Eqn. IV-1 is evaluated at the midpoint of the combined range of $h(x_c,t_l)$ and $h(x_c,t_b)$.

Illustrating the Approach

The approach is illustrated in Figure IV-2, which compares water levels at Hancock (a potentially affected location) with those at Wautoma (control location). Hancock and Wautoma water levels for the entire record (1958-2008; top) exhibit a weak linearity with substantial scatter ($r^2 = 0.50$). The scatter is resolved greatly by regressing at shorter time intervals. For instance, the regression for the early part of the record (1958-1975, Figure IV-2 middle) fits well compared with the record as a whole ($r^2 = 0.89$). Regressions for 1976-1985 and 1986-1995 (Figure IV-2, bottom) also fit the data well and are similar to the 1958-1975 regression. This indicates that Hancock water levels were mostly steady over these times with respect to Wautoma, and non-climatic influence is not indicated. However, the regression of the last time period (1996-2008) deviates greatly from early regressions, and indicates a decline in water levels at Hancock (Figure IV-2, bottom) unlikely due to climatic variability. To estimate the magnitude of the decline, any of the early regressions could serve as a "baseline," which would be subtracted from the

1996-2008 regression. By inspection, the difference is about 3 feet. More formally, the difference should be evaluated at a Wautoma water level that is the midpoint of the combined Wautoma water level values for the baseline and 1996-2008 regressions. The difference at the midpoint is 3.2± 0.9 feet (95% confidence interval), the estimate of groundwater drop beyond climatic variability potentially due to pumping at Hancock.

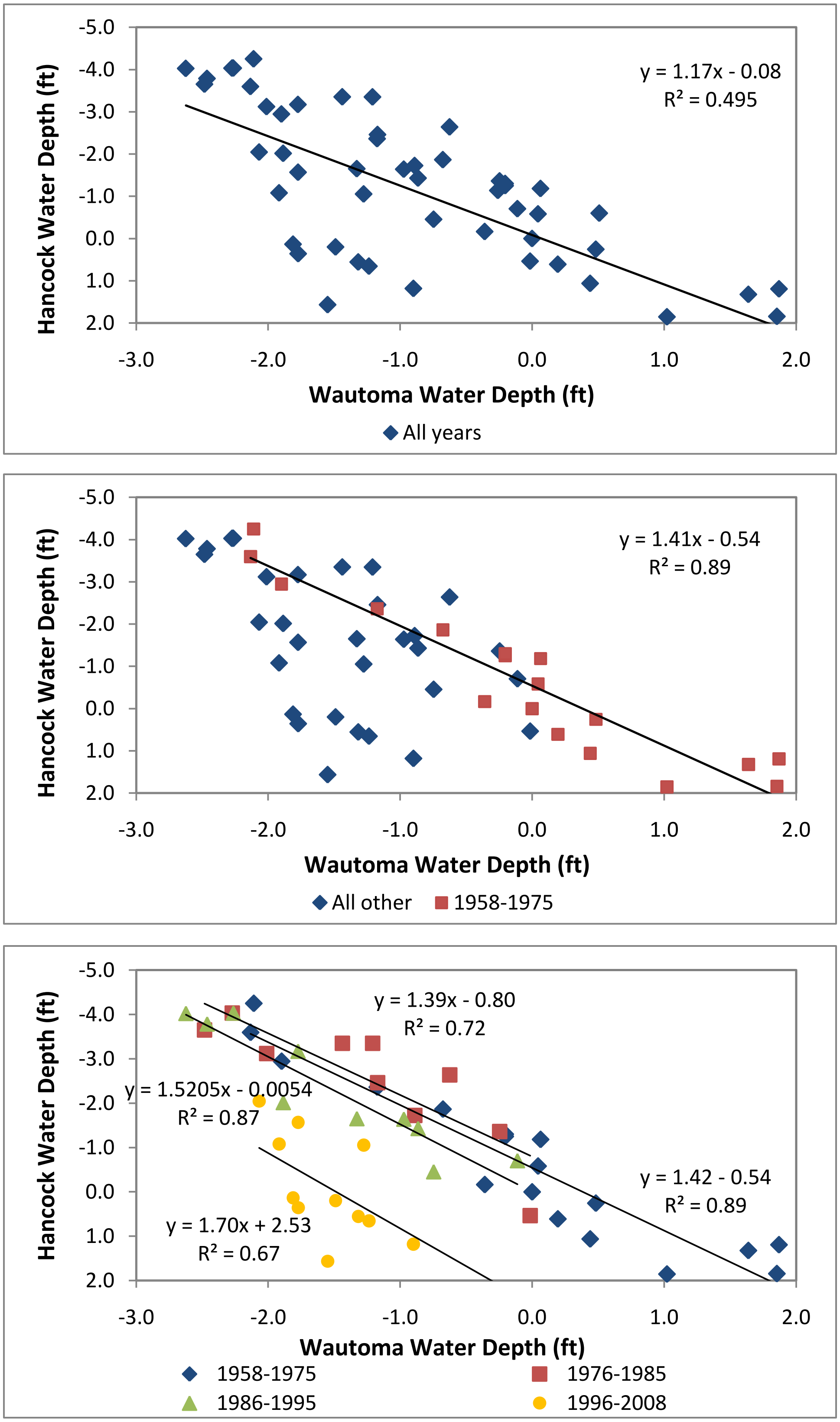


Figure IV-2. Hancock and Wautoma water depths compared. Top: regression of all data. Middle: 1958-1975 regressed separately. Bottom: Later period regressions agree with earlier ones until 1996, at which time Hancock water levels drop compared to Wautoma.

Implementing the method

For each match of a potentially affected location with a control location, two early baseline periods were chosen for comparison against a late (1999-2008) period. One early baseline period was usually 1959-1968, which is the earliest period common among most wells and the period with the least amount of pumping development. A second, longer baseline period was subjectively chosen as a qualitative check that baseline period selection did not unduly influence estimates of missing water. Because Coloma NW records do not become useful until 1964, its early baseline period was chosen as 1964-1973.

In addition to comparing baseline and late periods to estimate missing water, water level trends between potentially affected and reference locations were also calculated, using a modification of the previously described approach. This was done essentially by solving Eqn. IV-1 for an early baseline period and comparing against it successive 10-yr periods for the entire period of record. For instance, the Hancock record (Figure IV-2) was first regressed against Wautoma for the 1959-1968 period to establish a baseline, and then Eqn. IV-1 was evaluated for 1960-1969, 1961-1970, 1962-1971, 1963-1972, etc. The result is illustrated in Figure IV-3, which shows the relative water level relative to 1959-1968. The year of each point in Figure IV-3 is the median of the 10-yr late period interval.

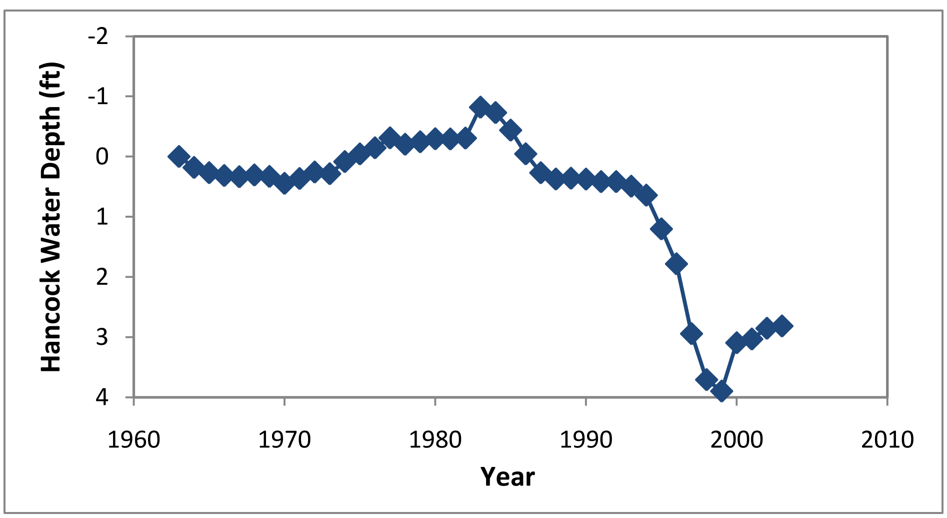


Figure IV-3. Water level deviation at Hancock compared with Wautoma. Median year of the 10-yr interval shown.

Only Amherst Junction and Wautoma were used as control locations. Nelsonville correlated well and consistently against Amherst Junction for their period of overlapping record (1958-1998), so using Nelsonville was redundant. The Wild Rose record ends in 1994, which hampers its use. Since the Wild Rose site behaves midway between Amherst Junction and Wautoma (next section), Amherst Junction and

Wautoma adequately bracket and represent Wild Rose behavior. Comparisons with Wild Rose are presented graphically, however.

Relation among Control Wells

If weather were spatially constant, then control wells would be expected to correlate perfectly over time. As weather indeed is somewhat spatially variable, correlations among wells would be expected to be less than perfect, with more proximate sites relating more closely than more distant ones. Amherst Junction and Nelsonville sites, the northernmost and most proximate locations, correlate well through their entire records. Wautoma, the southernmost of the group and some 30 miles distant, correlates with them more weakly. Wild Rose, which lies between the two (18 miles from the Amherst Junction site, 12 miles from Wautoma), displays a middle response.

In Figure IV-4 Amherst Junction and Wild Rose water levels relative to Wautoma are compared the same way Hancock and Wautoma are in Figure IV-3, thereby providing an indication of the expected effects of weather variability among Central Sands locations (i.e., if pumping is not a factor). Amherst Junction, compared with Wautoma, rose and then fell from about 1963 to 1974 (middle year of 10-yr regression intervals are used for reference date), remaining steady until 1994. Water levels declined and rebounded between 1994 and 2001. For its limited record, Wild Rose mimicked Wautoma more closely than Amherst Junction, exhibiting similar but subdued rises and falls to that of Amherst Junction. This supports a concept that for this region, Amherst Junction and Wautoma act as end members of a range of groundwater responses to weather. Wild Rose, while behaving more similarly to Wautoma, blends some aspects of both.

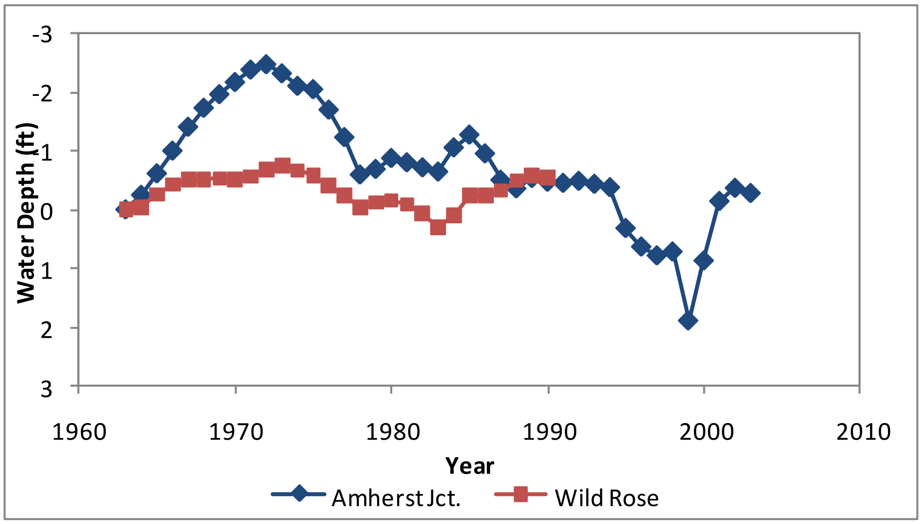


Figure IV-4. Water levels at Amherst Junction and Wild Rose compared with Wautoma. Median year of 10-yr interval shown.

Comparisons of Potentially Affected Wells with Control Wells

Plover

Water levels at Plover demonstrate a consistent and long term decline compared with control locations. During 1964-1972 (years are middle of 10-yr regression intervals) Plover tracked closely with Amherst Junction, but rose relative to Wautoma and Wild Rose, consistent with the comparison among control wells in Figure IV-5. Water levels at Plover dropped precipitously after 1972 compared with all reference stations. Compared to Amherst Junction, the drop averaged 0.09 ft/yr, or 0.9 ft per decade. Using 1959-1968 as a baseline period during which water levels were minimally affected by pumping, water levels in 1999-2008 were depressed 3 ft compared to Amherst Junction and more compared to Wautoma (Table IV-2). A longer baseline period does not substantially affect the estimate of water table decline. Amherst Junction is deemed the best control for comparison to Plover, given its proximity compared with Wautoma (9 miles from Plover to Amherst Junction compared with 30 miles to Wautoma).

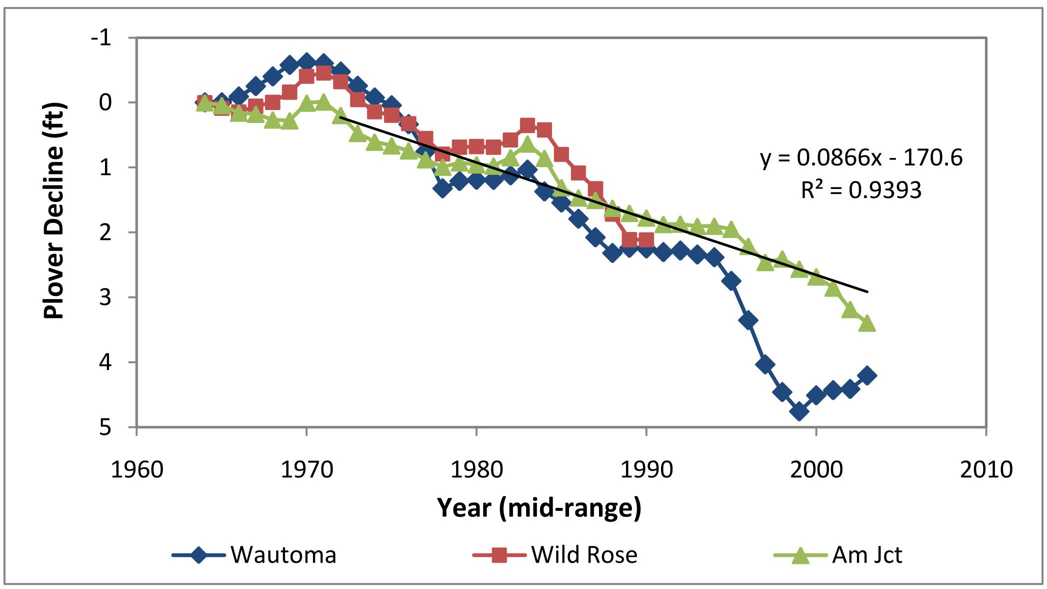


Figure IV-5. Groundwater level decline at Plover compared with control locations. Median year of 10-yr interval shown. Trendline is for Amherst Junction.

Table IV-2. Decline in Plover groundwater levels 1999-2008 (mean +/- 95% confidence interval) compared with two references sites and two baseline periods. BOLD indicates best estimate.

Locations	Baseline years	
	1959-1968	1959-1974
Amherst Junction	**3.0 ± 0.9***	3.1 ± 0.9*
Wautoma	4.5± 0.7*	5.1 ± 0.8*

* Indicates significantly different than zero at p = 0.05.

Hancock

Water levels at Hancock dropped from 1963 to 1970 (middle of 10-yr regression intervals) relative to all control locations, before peaking in about 1983. Water levels then dropped and leveled until the 1990s, and then dropped again, precipitously (Figure IV-6). Water levels at Hancock declined 3.2 ft relative to Wautoma (1959-1968 baseline and 1999-2008 comparison period) and lesser amounts (and not significantly at p = 0.05) compared to Amherst Junction (Figure IV-3). As Wautoma is nearer to Hancock than Amherst Junction (14 miles compared with 27 miles) and tracked closer to Hancock in its early history, we deem the Wautoma comparison as most valid. Assuming a different baseline period did not substantially affect estimates of groundwater decline. [Note: We replicated this analysis using the record for Long Lake – Saxeville for comparison. This confirmed Wautoma results and that Wautoma is the more valid station for comparison.]

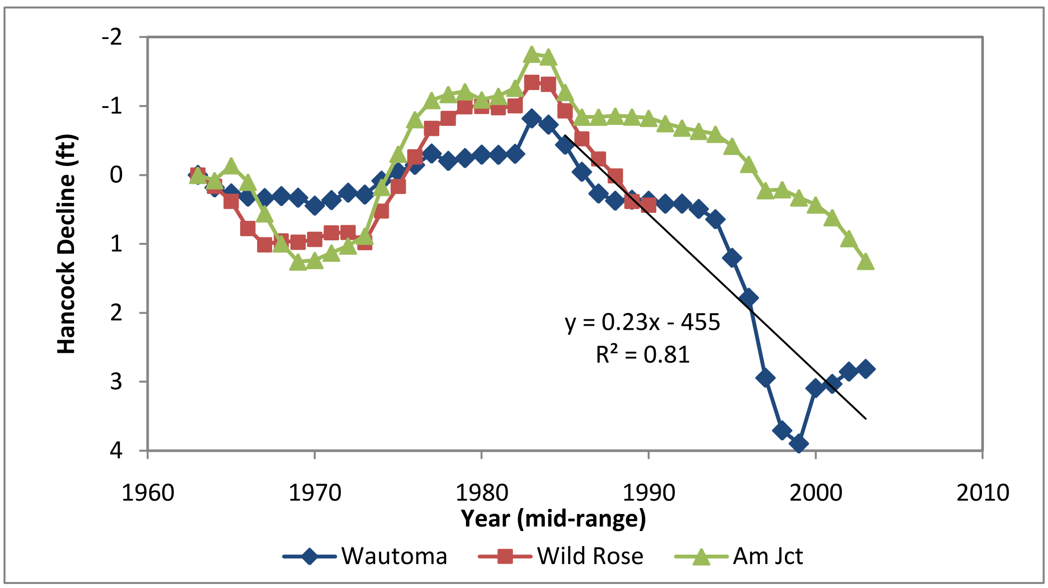


Figure IV-6. Groundwater level decline at Hancock compared with control locations. Median year of 10-yr interval shown. Trendline is for Wautoma.

Table IV-3. Decline in Hancock groundwater levels 1999-2008 (mean +/- 95% confidence interval) compared with two reference sites and two baseline periods. BOLD indicates best estimate.

Locations	Baseline years	
	1959-1968	1959-1988
Amherst Junction	1.1 ± 1.3	1.1 ± 1.5
Wautoma	**3.2± 0.9***	3.3 ± 0.5*

* Indicates significantly different than zero at p = 0.05.

Bancroft

Water levels at Bancroft rose from 1963 to 1983 relative to all control locations, and have declined since, at an annual rate of 0.05 ft/yr (Figure IV-7). Bancroft mimicked Wautoma most closely in the early part of the record. Comparing the 1999-2008 period against the 1959-1968 baseline, water level declines were about one foot (Table IV-4). Assuming a different baseline period did not substantially affect estimates of groundwater decline.

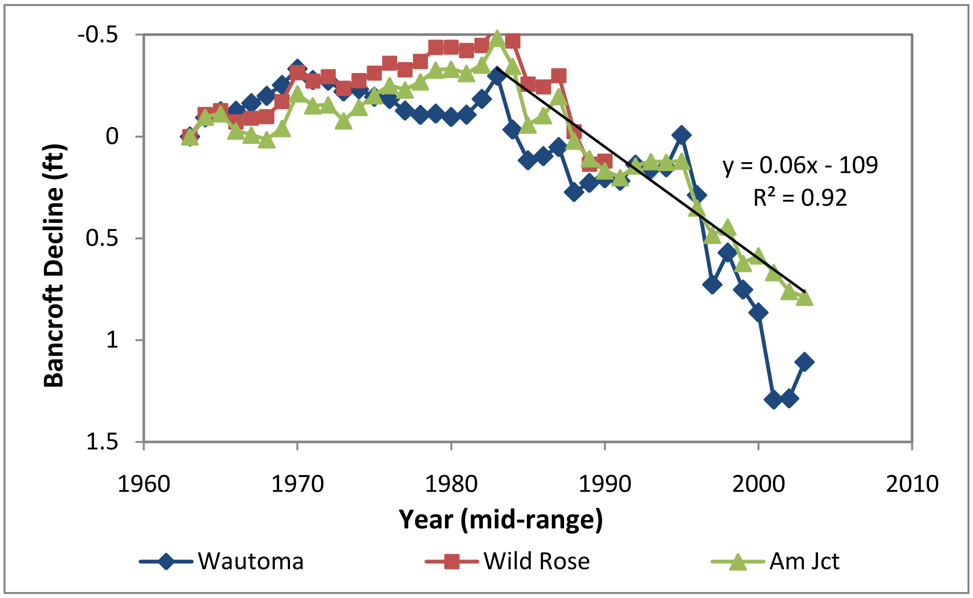


Figure IV-7. Groundwater level decline at Bancroft compared with control locations. Median year of 10-yr interval is shown. Trendline is for Amherst Junction.

Table IV-4. Decline in Bancroft groundwater levels 1999-2008 (mean +/- 95% confidence interval) compared with two reference sites and two baseline periods. Bold indicates best estimate.

Locations	Baseline years	
	1959-1968	1959-1988
Amherst Junction	**0.8 ± 0.4***	0.9 ± 0.5*
Wautoma	**1.2± 0.5***	1.3 ± 0.3*

* Indicates significantly different than zero at p = 0.05.

Coloma NW

This location suffers from an abbreviated early record. Water levels at Coloma NW declined through its entire record relative to all control locations, at an annual rate of 0.15 to 0.07 ft/yr (Figure IV-8). Comparing the 1999-2008 period against the 1964-1973 baseline, water level declines were 2.2 feet relative to Wautoma, and imperceptible against Amherst Junction (Table IV-5). Assuming a different baseline period did not substantially affect estimates of groundwater decline. As Coloma NW is not substantially closer to any control location, estimates from either Amherst Junction or Wautoma are equally valid.

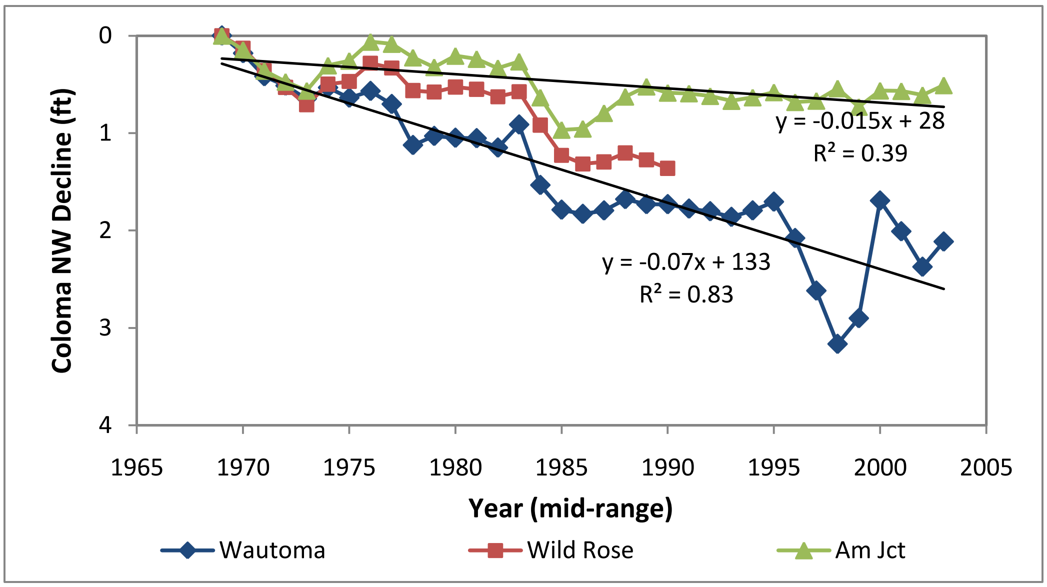


Figure IV-8. Groundwater level decline at Coloma NW compared with control locations.

Table IV-5. Decline in Coloma NW groundwater levels 1999-2008 (mean +/- 95% confidence interval) compared with two reference sites and two baseline periods. BOLD indicates best estimate.

Locations	Baseline years	
	1964-1973	1964-1988
Amherst Junction	**0.1 ± 1.0**	0.2 ± 1.4
Wautoma	**2.2± 0.9***	1.0 ± 0.7*

* Indicates significantly different than zero at p = 0.05.

V. LAKE LEVEL TRENDS

Summary

Records from thirteen lakes were sufficient for providing estimates of level change beyond those resulting from climatic variability alone. Lakes where few high capacity wells are located, in the vicinity of Wild Rose and Wautoma, did not show a non-climatic response. Lakes in or near areas with a large density of high capacity wells, the Hancock – Plainfield vicinity and Pleasant Lake in southwest Waushara County, showed statistically significant declines beyond climatic response, in the range of 1.5 to 3.6 feet. These declines reflect a sort of average "missing" water for the mid 1990s through 2007, and may not adequately reflect current or yearly missing water. Apparent lake level declines in the vicinity of Hancock were similar to the decline in the Hancock monitoring well. Calculated lake level declines corresponded well with those predicted using modeling (Chapter VI).

Overview

Lake level records were evaluated for evidence of change beyond those resulting from climatic variability alone, using an approach similar to that used for evaluating the monitoring well record (Chapter IV). In brief, the approach compared the relationships of water levels in a lake with the water levels at a control location during an early period, when pumping effects were assumed small, to a late period, when pumping was well developed on the Central Sands landscape. Control locations essentially filter out climatic influences on water levels. Differences between early and late period relationships are signals of a water level change that cannot be accounted for by weather variation alone, such as pumping.

Several considerations hampered this effort. Only 14 lakes had sufficient long term water level data suitable for this analysis, and one of these had a record apparently flawed by changing benchmarks [see endnote]. Control locations, which would ideally be free of high capacity well influence, are in reality somewhat affected by pumping. Early records that pre-date large-scale pumping development would be available, but in reality a true pre-development record does not exist, only a time when fewer high-capacity wells were present. Finally, the water level record is much sparser for lakes than the monitoring well record, i.e, observations once every year or so compared with many measurements per year.

The lakes available for comparisons were all located in Waushara County (Table V-1, Figure V-1). These lakes group well into four geographic clusters: Lakes Huron, Pine-Hancock, and Fish are near Plainfield, Hancock, and the Hancock water level monitoring well where pumping is well developed (Chapter IV). Lakes Pine-Springville, Long-Saxeville, Gilbert, and Kusel are near the Wild Rose monitoring well, where pumping is sparser. Lake Burghs, Big Silver, Irogami, Lucern, and Witter's cluster near Wautoma and the monitoring well there, where pumping is also sparse. Lake Pleasant comprises a cluster of one, lying not particularly near any monitoring location, has a few pumping wells in its immediate vicinity, and many wells about two miles to its west.

Only two potential water level control locations were available with records extending through 2007, Wautoma and Amherst Junction (Chapter IV). Wautoma is closer to all lake locations, tracked more closely to the Wild Rose monitoring well (where many lakes are located) through its entire history (Figure IV-4), and tracked more closely to the Hancock monitoring location through its early history (Figure IV-6). Hence, Wautoma is apparently a better baseline location than Amherst Junction for all four clusters, and we use it as the sole control location. (Note: Amherst Junction water levels have dropped about 1-2 ft with respect to Wautoma in the last decades, presumably due to drier climate or more pumping development in the Amherst Junction area. Hence estimates of lake level change using Amherst Junction instead of Wautoma averaged 1.3 feet higher greater than Wautoma estimates.)

Lake levels were matched to the Wautoma monitoring well water level for the same month that the lake level was obtained. Years 1993-2007 were used as the late period. The early period varied among wells due to data availability (Table V-1). Graphs and regressions were produced showing the early and late periods of each lake and the difference between the regressions was determined, similar to the well record, Chapter IV. Example graphs for a lake showing decline compared to Wautoma, and one showing no decline are shown in Table IV-1.

Results

Calculated lake level declines compared with the Wautoma control location differed by cluster. Lakes in the Plainfield - Hancock cluster (Huron, Pine-Hancock, and Fish) showed large and statistically significant declines, 2.7 to 3.6 feet. These declines were similar to those observed at the nearby Hancock monitoring well (3.2 feet, Chapter IV).

The Wild Rose cluster lakes were steady to slightly declining relative to Wautoma. The Wautoma cluster lakes were steady between periods, with Burghs Lake declining 0.9 ft and Lake Lucern rising 1.7 ft. Pleasant Lake declined 1.5 ft.

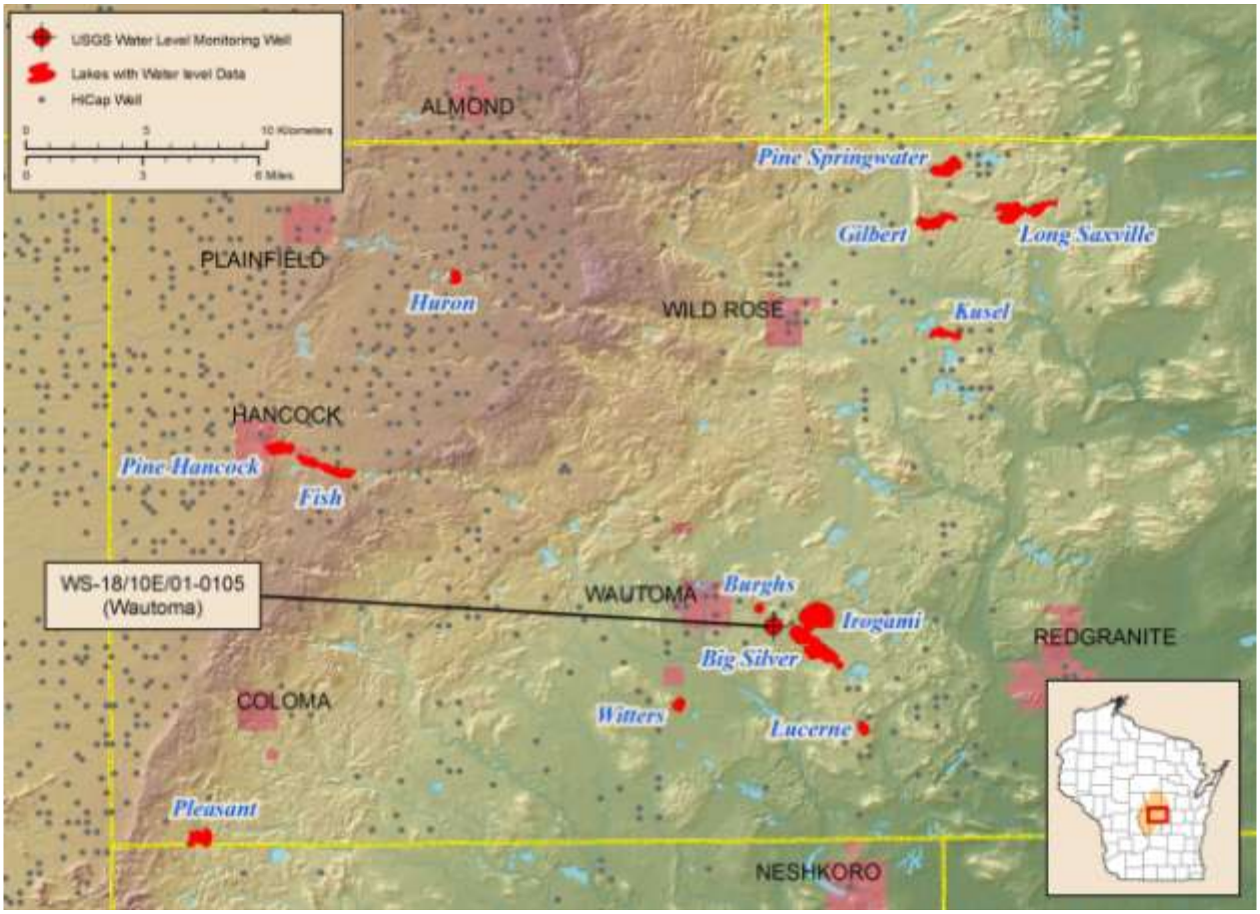
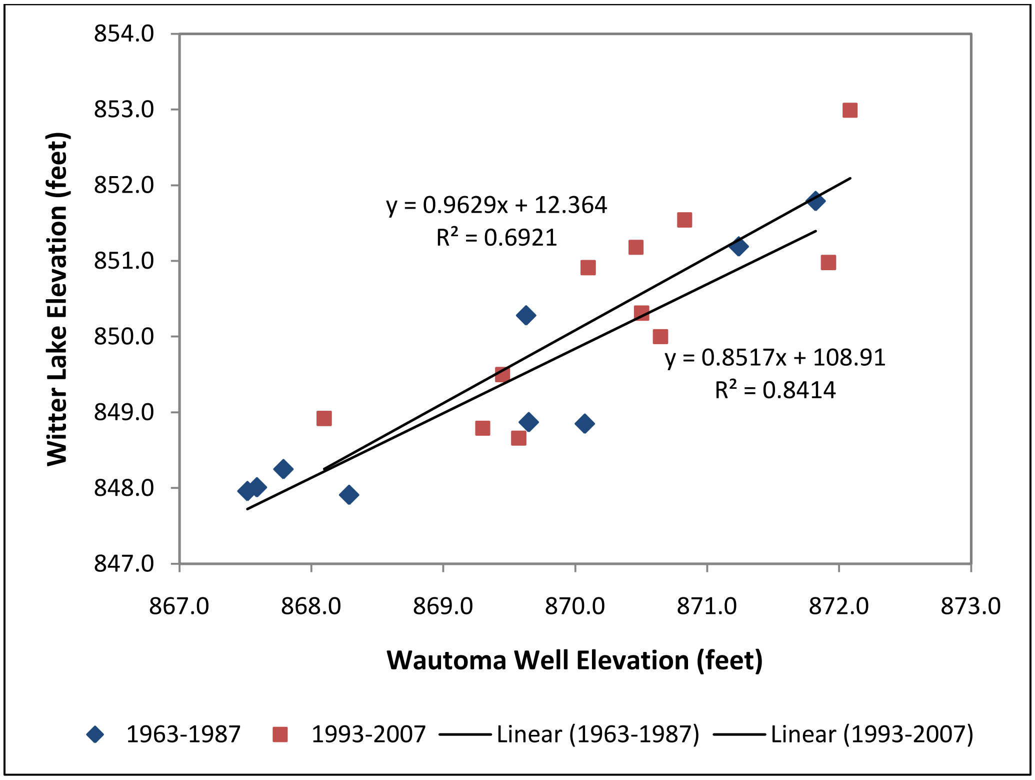


Figure V-1. Locations of lakes in study area along with the Wautoma reference well.



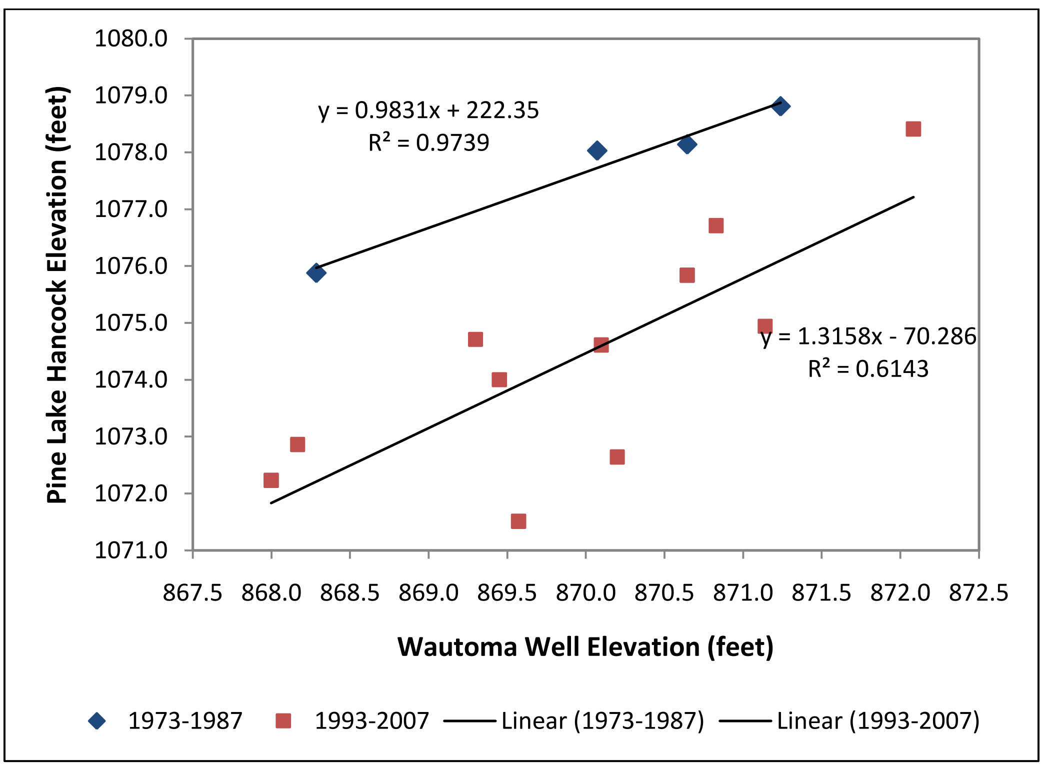


Figure V-2. Top: Correspondence of water levels at Witter's Lake with Wautoma showing no non-climatic changes between early and late period. Bottom: Same for Pine Lake - Hancock showing a decline of 3.2 feet between periods.

Table V-1. Lakes by group; lake name, periods, number of levels.

Lake Name	Cluster #	Early	Early n	Late	Late n
Fish Lake	1	1973-1989	3	1993-2007	8
Huron Lake	1	1973-1987	4	1993-2007	9
Pine Lake Hancock	1	1973-1987	4	1993-2007	11
Gilbert Lake	2	1962-1987	16	1993-2007	12
Kusel Lake	2	1963-1989	15	1993-2007	11
Long Lake Saxeville	2	1959-1974	29	1999-2007	12
Pine Lake Springville	2	1961-1989	15	1993-2007	12
Big Silver Lake	3	1966-1989	13	1993-2007	8
Burghs Lake	3	1973-1987	7	1993-2007	11
Lake Irogami	3	1961-1988	7	1993-2007	10
Lake Lucern	3	1963-1987	11	1993-2007	11
Witter's Lake	3	1963-1987	9	1993-2007	11
Pleasant Lake	4	1964-1989	7	1993-2007	14

Table V-2. Estimated decline in water levels relative to Wautoma; the 95% confidence interval and p different than zero ar ealso shown.

		Early Late Period Comparison		
Lake Name	Cluster #	Decline	P-Value	95% CI
Fish Lake	1	2.7	0.029	± 2.3
Huron Lake	1	3.6	0.009	± 2.5
Pine Lake Hancock	1	3.2	0.001	± 1.6
Gilbert Lake	2	0.3	0.257	± 0.6
Kusel Lake	2	0.5	0.136	± 0.7
Long Lake Saxeville	2	0	0.961	± 0.9
Pine Lake Springwater	2	0.8	0.004	± 0.5
Big Silver Lake	3	-0.6	0.218	± 1.0
Burghs Lake	3	0.9	0.037	± 0.8
Lake Irogami	3	0	0.996	± 0.6
Lake Lucern	3	-1.7	0.004	± 1.1
Witter's Lake	3	-0.4	0.333	± 0.8
Pleasant Lake	4	1.5	0.001	± 0.8

End note – Long Lake Oasis

In the interest of transparency, this end note is included regarding Lake Long-Oasis. During our analysis, we observed that the Long Lake Oasis hydrograph displayed aberrant behavior compared with other lakes in the vicinity, the Hancock monitoring well, and groundwater flow modeling results (Chapter VI).

Closer examination revealed that 1964 reported lake level was lower than the 2006 reported level, when the lake was virtually dry in 2006. This raised the question of how lake levels could be lower in 1964 than when the lake was actually dry. Checks of a dated photographic record supplied by a long time lake resident (M. Williquete, pers. comm.) showed that the lake contained more water during the climatically record dry years of 1958-1959 and 1964 compared to the present, belying the early lake record. How could this be?

Further checking suggests that the most reasonable cause of the conflict is due to benchmarking error. Three benchmarks were established over time on the lake. Conservation Department / Department of Natural Resources staff in the 1960s used a nail in a tree on the lake's north shore at the boat landing as a benchmark (M. Primising; retired DNR biologist, oral communication), referenced at an arbitrary 100 feet. A newer benchmark, referenced to sea level, was established in 1972, on the southern shore of the lake likely across from the boat landing (Rick Ertl, Waushara County Zoning Department, oral communication). The most recent benchmark was established in 1995, back at the boat landing at the north side of the lake. Strong documentation shows that surveyors linked 1972 and 1995 benchmarks, but no record shows that the early 1960s benchmark was ever tied to later benchmarks. The lake level record suggests pre-1972 differences between the benchmark and lake level were simply subtracted from the 1972 benchmark without any correction. Hence pre-1972 lake levels likely are not consistent with those measured afterwards. Insufficient early period post-1972 record exists to make a comparison of early and late periods for Long Lake – Oasis.

VI. ASSESSING PUMPING IMPACTS USING GROUNDWATER FLOW MODELS

Summary

Four groundwater flow model versions for the Central Sands were developed, each representing slightly different conceptual models of the region's hydrogeology. The four models produced similar predictions of water level and streamflow reductions in response to pumping stresses, and appear to reasonably reflect hydrologic reality. The models are available for future regional analyses and for down-scaling to focus on specific locales.

Flow models indicate irrigation pumping may cause large impacts on the region's lakes and streams. In some places, up to 2.5 feet of water table and lake level decline are indicated *per inch* of net recharge reduction on irrigated lands. In headwater streams greater than 20% flow reduction is possible *per inch* of net recharge reduction. Given that some estimates of average net recharge reduction on irrigated lands range to 2-3 in, the consequences on lakes and streams are potentially large.

Analyses done here were performed in the steady-state. Better estimates of spatial and temporal variability of irrigation impacts may be possible upon completion of vadose zone modeling by B. Lowery and W. Bland at the University of Wisconsin-Madison Department of Soil Science.

Introduction

Modeling efforts are described here briefly; details can be found in Technical Memorandum 11, included as electronic media with this report. Groundwater flow modeling efforts involved design, construction, and calibration of four flow model versions, and then applying them to estimate the effects of irrigation pumping on surface water resources. Flow models have been previously constructed for much of the region: a model for the Little Plover River area (Clancy et al., 2009) and one for the Tomorrow/Waupaca watershed (Mechenich, 2000) overlapped in the northern part of the area, and a Source Water Assessment Model (McGinley, 2002) covered much of the total area. The models described here contain several refinements over previous efforts. These include better linkages between groundwater and streams, versions that incorporate the Cambrian sandstone, and improvements in calibration technology. The new models readily lend themselves to adaptation and focusing into smaller parts of the region to answer site-specific questions.

Conceptual models, design and calibration

Each flow model version represents a different realization of a conceptual model of the groundwater flow system. All conceptual models of the study area posit that groundwater originates as a really diffuse precipitation recharge, and is transmitted through the aquifer to the area's streams where it discharges. In two flow model versions, only unconsolidated sediments are considered as an aquifer conveying groundwater. One uses a spatially constant recharge rate and the other a spatially variable recharge rate. In the two other flow model versions, the sandstone underlying unconsolidated sediments was included as a second model layer; one with a constant sandstone hydraulic conductivity and one with a spatially variable sandstone hydraulic conductivity.

The groundwater flow system was modeled using the USGS MODFLOW code (Harbaugh et al., 2000) (Figure VI-1). Model development was accomplished in GMS (Aquarco, 2005) and then ported to Groundwater Vistas (ESI, 2007). The model area was discretized into 200 meter square cells, 625 rows by 500 columns (WTM origin at 517000, 351000), with 171,406 active cells per layer within the boundaries. External boundaries were usually constant head. These usually coincided with major streams, and where major streams were not present, usually minor streams. Internal streams and drainage ditches were modeled as MODFLOW rivers and drains. Drains were used especially in headwaters areas where uncertainty would exist as to whether the feature was wet or dry. Constant head, river, and drain elevations were input from digital USGS 7.5 minute topographic maps (DRGs) or in a few cases, field data where available. The conductance of the river and drain cells reflect the approximate length and width of the stream in a cell, ranging from 10 m/d per unit length for small headwaters to 40 for larger rivers/lakes based on previous models.

Sandstone and granite surfaces were mapped using the WDNR HiCap and Well Construction databases (WDNR, 2008), WiscLITH (WGNHS, 2008), and a Portage County well database (Hartman, 2007) as data sources. These data were managed and interpreted using ArcGIS software (ESRI, 2008). The surfaces were contoured using recorded rock contacts and other well points that might represent the surface (well bottom below the surface calculated only from recorded contacts). The land surface and base for calculating elevations from well depths was a 10 m National Elevation Dataset DEM with elevation in decimal meters for the entire model area.

The water table in the study area is usually contained in the unconsolidated aquifer. Exceptions occur where relatively rare bedrock mounds (both sandstone and granite) pierce unconsolidated sediment. The most noticeable examples are distinctive sandstone mounds such as those at Roche a Cri State Park in Adams County. In order to make the model numerically stable and prevent cell drying, some bedrock highs were included in the upper layer.

Calibration was performed by adjusting recharge and hydraulic conductivity (K) parameters. The simplest model version (Model A) included a single layer, primarily representing the unconsolidated sediments. This model was calibrated to a single recharge rate and a K that was allowed to vary continuously over the model domain. Model B was also single layer, but the recharge rate was allowed to vary across the domain during calibration. Model C included a lower sandstone layer, with recharge and sandstone K constrained to a single value, while allowing the upper layer K to vary continuously. Model

D was similar to C, but allowed sandstone K to also vary continuously. PEST parameter estimation software (Doherty, 2007) was used for initial model calibration followed by minor manual adjustment employing the “pilot point” method to vary parameters continuously over the model domain. A combination of 500 head and 84 flux calibration targets were used to guide PEST and calculate calibration statistics for comparison. Targets were weighted on their apparent reliability and representativeness relative to a base value of one. The modeled contours were also qualitatively compared to the Irrigable Lands Inventory water table elevation maps (Lippelt and Hennings, 1981) for reasonableness.

All four model (Figure VI-2) versions produced similar water table configurations that compared favorably with the Irrigable Lands Inventory water table maps (Figure VI-3). Allowing recharge rates to vary (Model B), did not improve the calibration, but may be a better representation of reality. Estimated recharge as a single parameter or as an average of pilot points was very similar, 8.78 to 8.85 inches. Two layer model calibrations appear somewhat better in terms of smaller absolute residual mean, but the difference does not seem important in comparison to the added complexity. The K distribution in the surficial aquifer was similar for all four models, averaging 22.4 to 27.6 m/day. In two layer models, the lower layer was 2.1 m/d in Model C (single K value calibration) and averaged 6.6 m/day in Model D.

Assessing Effects of Irrigation Pumping on Lakes and Streams

The impacts of irrigation pumping on lakes and streams were evaluated in the models as a reduction in net groundwater recharge on irrigated lands. This evaluation was done in the steady state, using a single value of net recharge reduction in each model run. Better estimates of spatial and temporal variability of irrigation impacts may be possible upon completion of vadose zone hydrology modeling by B. Lowery and W. Bland at the University of Wisconsin - Madison.

Irrigated land coverages were not universally available for the counties in the Central Sands region. Thus, we generated an artificial irrigated land coverage through the use of high capacity well locations and county-based irrigated land acreage in the Census of Agriculture (USDA NASS, 2008). The artificial coverage was generated by expanding quarter-quarter sections around high capacity irrigation wells until the total acreage equaled the reported acreage. The resulting pattern closely mimicked irrigated land coverages where they were available.

For each model version, eight runs were made simulating 1 to 8 inches of reduction in recharge on irrigated lands. Declines in the water table elevation and stream discharges were approximately linear with reduction in recharge, and can therefore be mapped as a rate of decline per inch of recharge reduction.

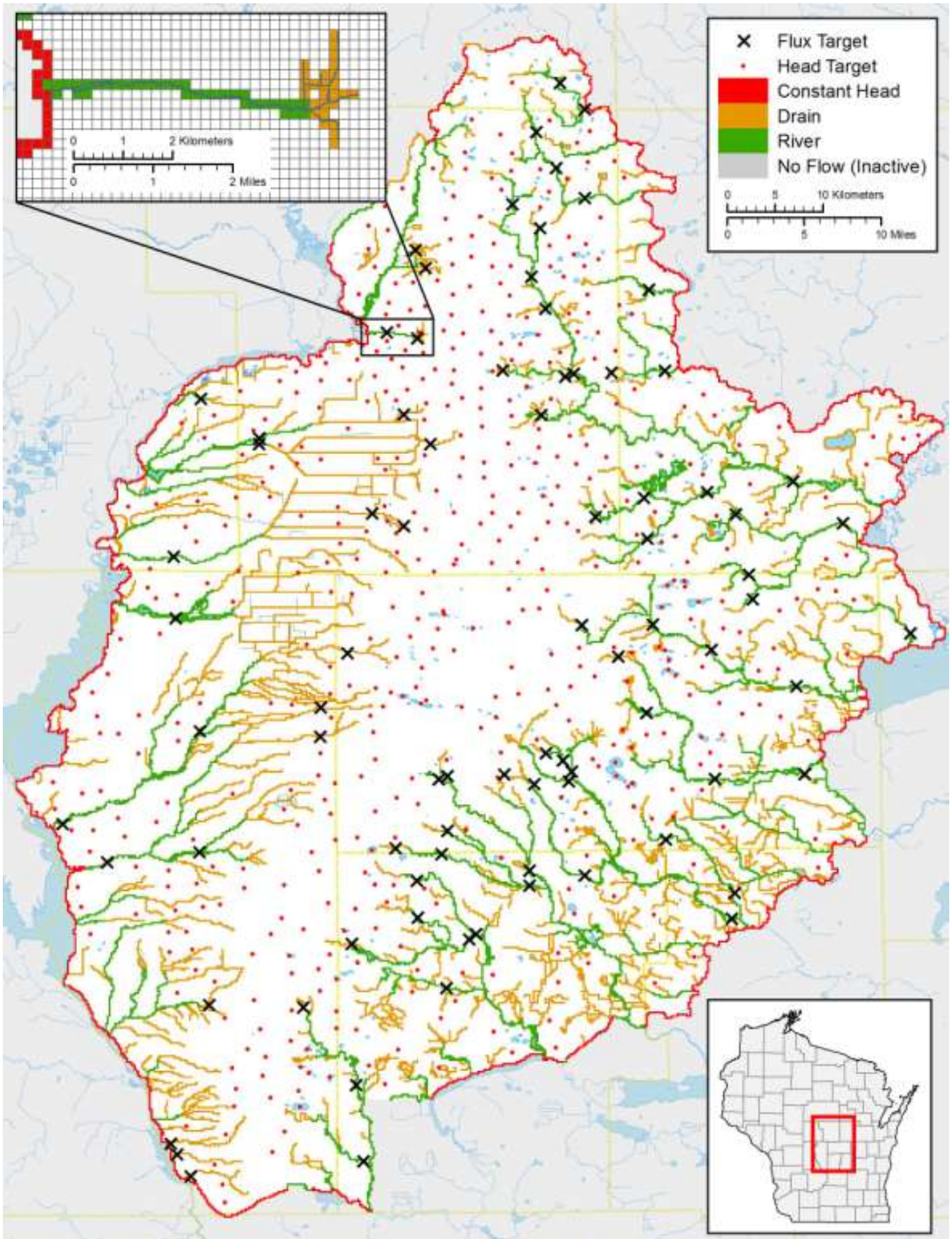


Figure VI-1. Model features including discretization and boundaries.

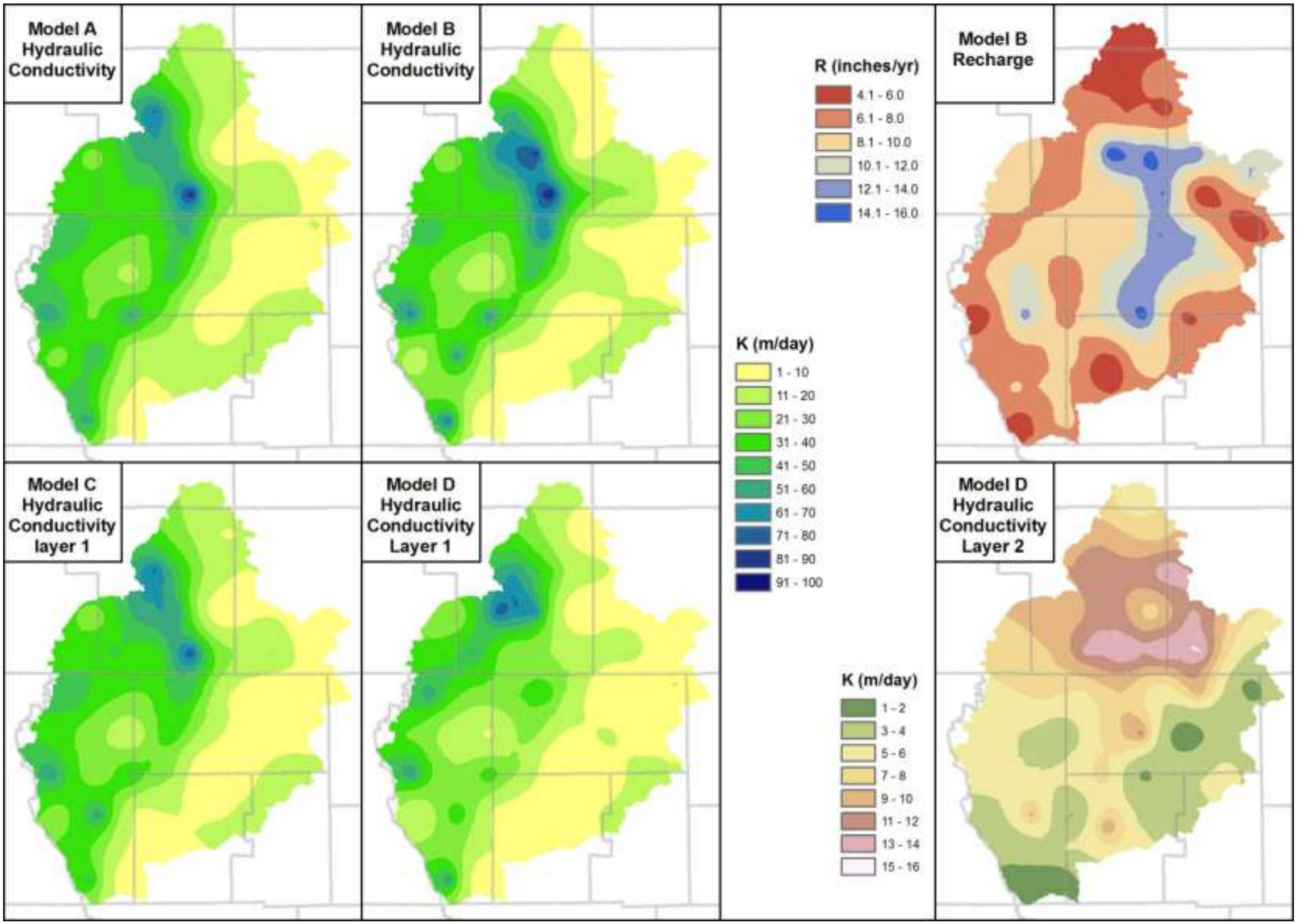


Figure VI-2. Hydraulic conductivity and recharge distributions for the calibrated models.

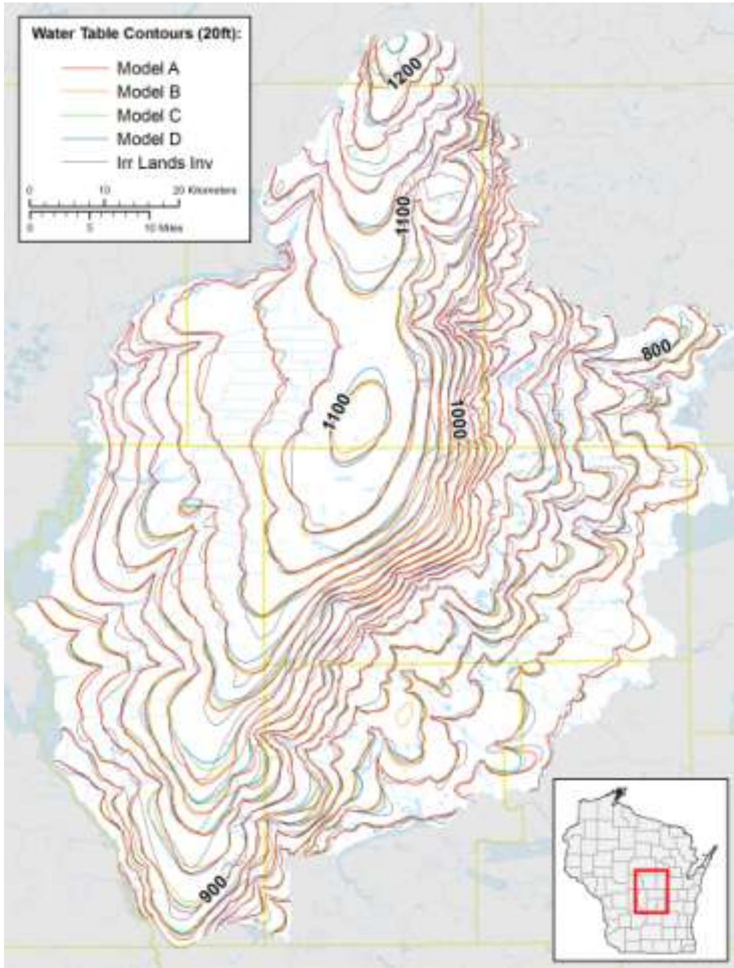


Figure VI-3. Match of heads in four model versions to those in Lippelt and Hennings (1981).

Irrigation Effects on Groundwater and Lake Levels

Modeled declines in groundwater levels (and the declines in lake levels as well because they are strongly groundwater connected) ranged up to 2.6 feet per inch reduction in recharge on irrigated lands (Figure VI-4). The most sensitive portion of the landscape is the region with greatest densities of high capacity wells distant from groundwater recharge areas in southeastern Portage County and northwestern Waushara County. Water level declines at the Wautoma and Amherst Junction reference sites were computed to be 0.20 and 0.37 ft per inch recharge reduction, respectively.

Declines in stream discharges with reduction in recharge on irrigated lands were evaluated in two ways. First, the discharge decline was evaluated at select locations that have actual discharge measurements. As locations with a discharge measurement history are sparse and generally distant from stream headwaters, these estimates are biased toward places where streamflows are relatively large and will under-reflect potential harms in headwaters areas. The modeled discharge declines (cfs reduction per inch recharge reduction), expressed as a percentage of measured discharge (Figure VI-5), are smaller far from high densities of high capacity wells and for larger streams. Percentage streamflow decline is greater near stream headwaters, particularly for tributary streams to the Wisconsin River where large densities of high capacity wells are close to streams. Streams with larger declines (> 10%) include the headwaters of the Buena Vista Creek, 10-Mile Creek, 14-Mile Creek and Little Roche a Cri systems, Spring Creek, the Pine River, the Little Plover River, a tributary to the Mecan, and the Montello.

The second way that percentage discharge decline with reduction in recharge was evaluated used modeled discharges, with and without recharge reductions. This calculation was done at 20 locations on headwater streams at an arbitrary distance of one mile from where modeled discharge to the stream begins. Results (Figure VI-6) were consistent with those using locations with discharge measurements, but indicated more serious discharge losses in headwaters areas, where declines were commonly 15-25% per inch recharge reduction.

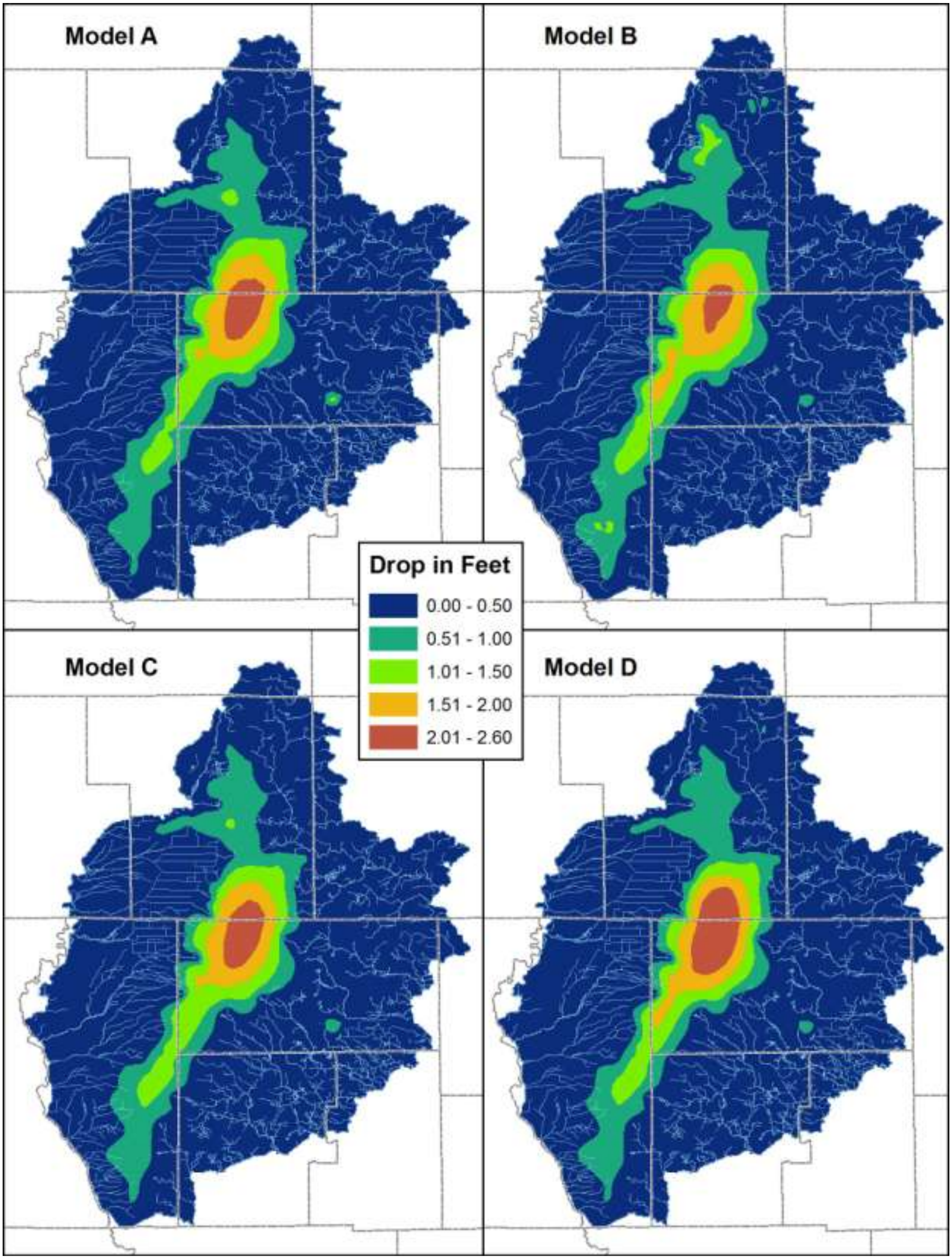


Figure VI-4. Drop in water table per inch reduction in net recharge on irrigated lands by four model versions.

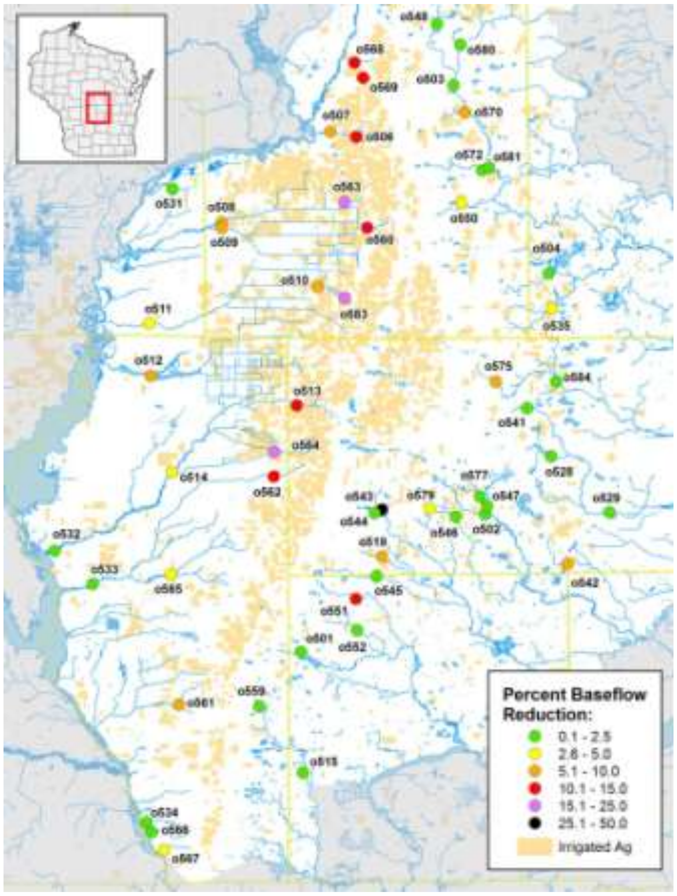
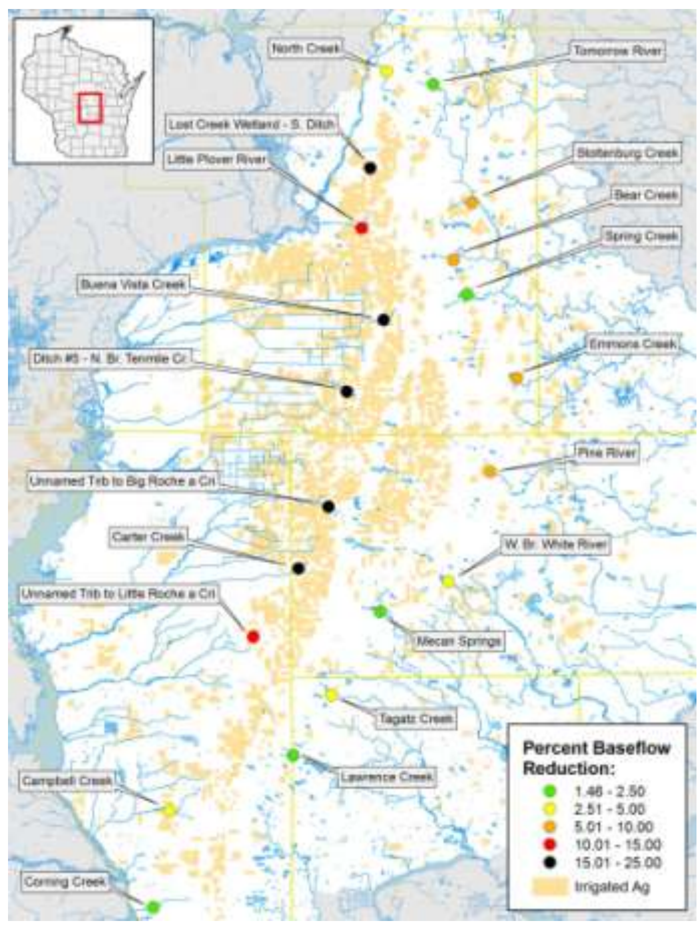


Figure VI-5. Modeled percent stream baseflow reductions compared with measured discharges mostly at non-headwaters locations.

Figure VI-6. Modeled percent stream baseflow reductions in headwater locations, defined as 1 mile from modeled wetup locations.



VII. COMBINING STATISTICALLY ESTIMATED AND MODELED WATER LEVEL DECLINE RESULTS

Summary

Statistically estimated water level declines in monitoring wells and lakes were overlaid on flow-modeled declines and demonstrated a good agreement: lakes and wells for which a large decline was calculated lie in areas that the model computes should have a large decline, and the same applies for lakes and wells where a small impact has been observed. A best fit between estimated and modeled results occurs with a net recharge reduction of 1.9 in on irrigated lands in the flow model. When the flow model is run with this amount of recharge reduction, modeled water levels decline by 2-4 feet in areas where highly stressed lakes have been observed. Modeled headwater streams discharge reductions with this amount of net recharge reduction are commonly 20-50%. Computed water level and stream discharge declines with a 1.9 in net recharge reduction represent a sort of average and not annual nor dry period maximums.

Statistically estimated water level decline estimates at monitoring wells and lakes (Chapters IV and V) assumed that the water level estimates at Wautoma and Amherst Junction reference sites were unaffected by pumping. Extrapolating the model based water level declines to reference wells with 1.9 in net recharge reduction indicates that pumping may have depressed water levels at the reference sites by 0.4 to 0.76 feet. Hence estimated water level declines for lakes and monitoring wells could be greater by these amounts.

Overlaying estimated and modeled water level declines

Statistically estimated water level declines in monitoring wells (Chapter IV) and lakes (Chapters V) are overlaid on flow-modeled declines (Figure VII-1). Groundwater Model C results were used as the underlay; other model results are similar (Figure VI-5).

To be clear, flow modeled declines and statistically estimated declines represent slightly different things. Estimated declines were calculated from actual water level measurements and represent a sort of average amount of missing water not explainable by climate alone during 1999-2008 for monitoring wells and the mid 1990s-2007 for lakes compared to an earlier reference period. Modeled declines are not absolute, but are the amount of decline per inch of net recharge reduction on irrigated lands. Due to the uncertainties, an actual net recharge reduction cannot be assigned a priori (Chapter VI). Modeled declines compare differences between a steady-state at a base recharge rate and a steady-state with the one inch recharge reduction. Neither statistically estimated nor modeled water level declines describe peak amounts of missing water, for instance what might have happened during dry years.

Caveats aside, estimated and modeled approaches still get at something similar and the overlay of estimated on modeled results shows good agreement: lakes and wells for which a large decline was calculated lie in areas that the model computes should have a large decline, and the same applies for lakes and wells where a small impact has been observed. Plotting estimated against modeled declines per inch recharge reduction (Figure VII-2) exhibits a good linearity ($r^2 = 0.71$). The slope of the regression line, [(ft of decline / (ft of decline / in recharge reduction) = in recharge reduction] provides an estimate of actual steady-state recharge reduction on irrigated lands, amounting to 1.9 in ($\pm$ 0.7 in, 95% confidence interval). A 1.9 in net recharge reduction on irrigated lands is within the range of average produced using simple plant-soil-atmosphere models (W. Bland, pers. comm.). As the groundwater system may not be at long term equilibrium because irrigation has been increasing on the central sand landscape, the 1.9 in recharge reduction may represent an underestimate. The amount is doubtlessly greater seasonally and during dry years.

Modeled water level and streamflow declines with 1.9 in of net recharge reduction are presented in Figure VII-3 and Figure VII-4. Many water level stressed lakes lie in zones where modeled water levels are 2-4 feet lower on average with this recharge reduction amount. Headwater streams discharge reductions with this amount of net recharge reduction are commonly 20-50%. These water level and stream discharge declines represent a sort of average, and not annual nor dry period low.

We present the Baseflow Reduction Index for Central Sands streams in Figure VII-5. This index, attributed to Prof. Doug Cherkauer at the University of Wisconsin – Milwaukee, computes the percent reduction in baseflow as expressed as

$$\frac{\text{(Groundwater discharge with pumping – Groundwater discharge without pumping)}}{\text{(Groundwater discharge without pumping)}} \times 100\%$$

The index has been suggested as a way to designate areas that need to be managed broadly to mitigate pumping impacts from many pumpers.

Effects of Water Level Declines at Wautoma and Amherst Junction

Water level decline estimates at monitoring wells and lakes (Chapters IV and V) assumed that the water level estimates at Wautoma and Amherst Junction were unaffected by pumping. Extrapolating the model based water level declines to reference wells with 1.9 in net recharge reduction indicates that pumping may have depressed these water levels by 0.4 to 0.76 feet. Hence estimated water level declines computed for lakes and monitoring wells could be greater by these amounts.

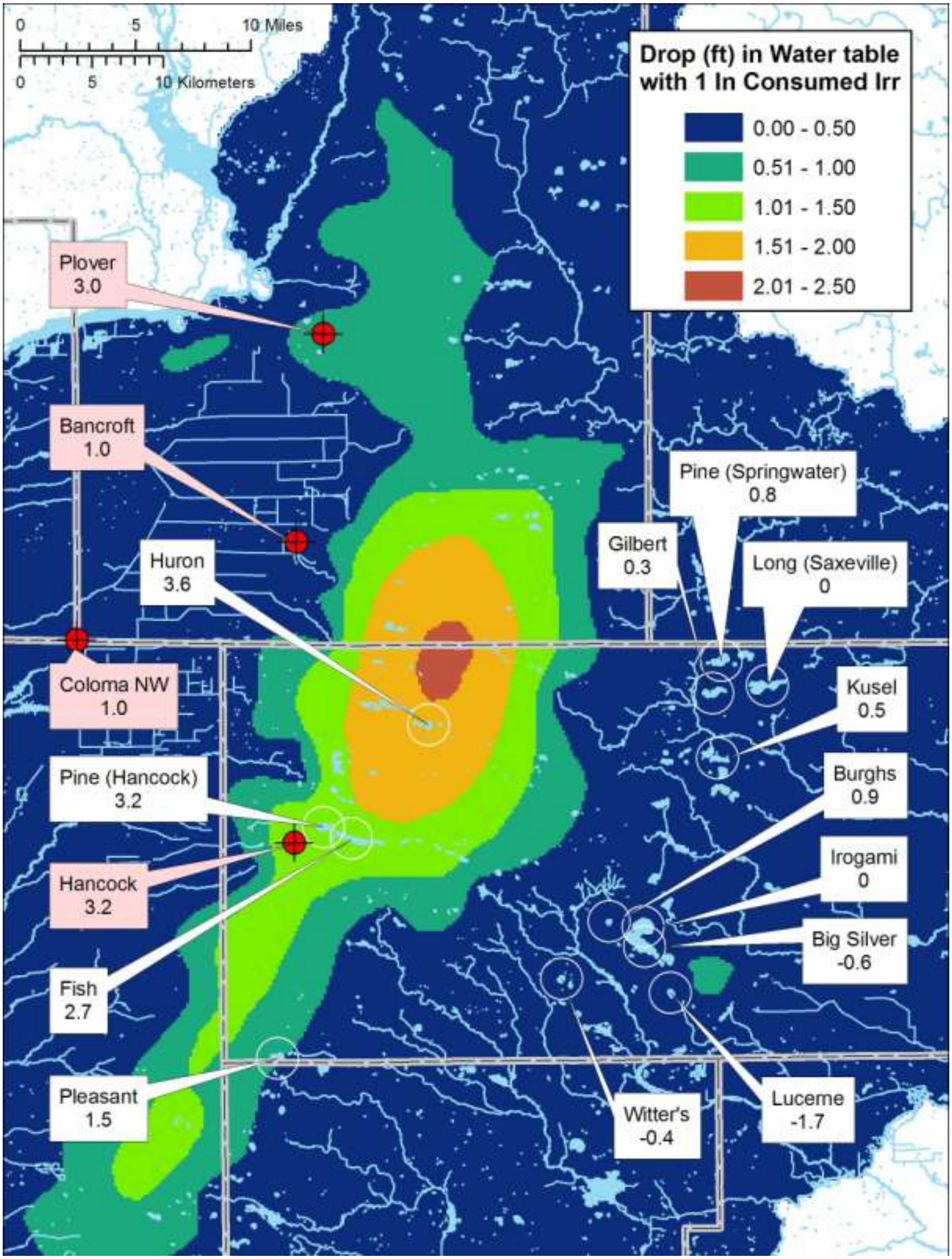


Figure VII-1. Statistically estimated amounts of "missing water" in monitoring wells (pink boxes) and lakes (white boxes) overlaid on modeled declines with a one inch reduction in recharge on irrigated lands. "Missing is that beyond that explainable by dry weather." Missing water is a sort of average for the 1999-2008 period.

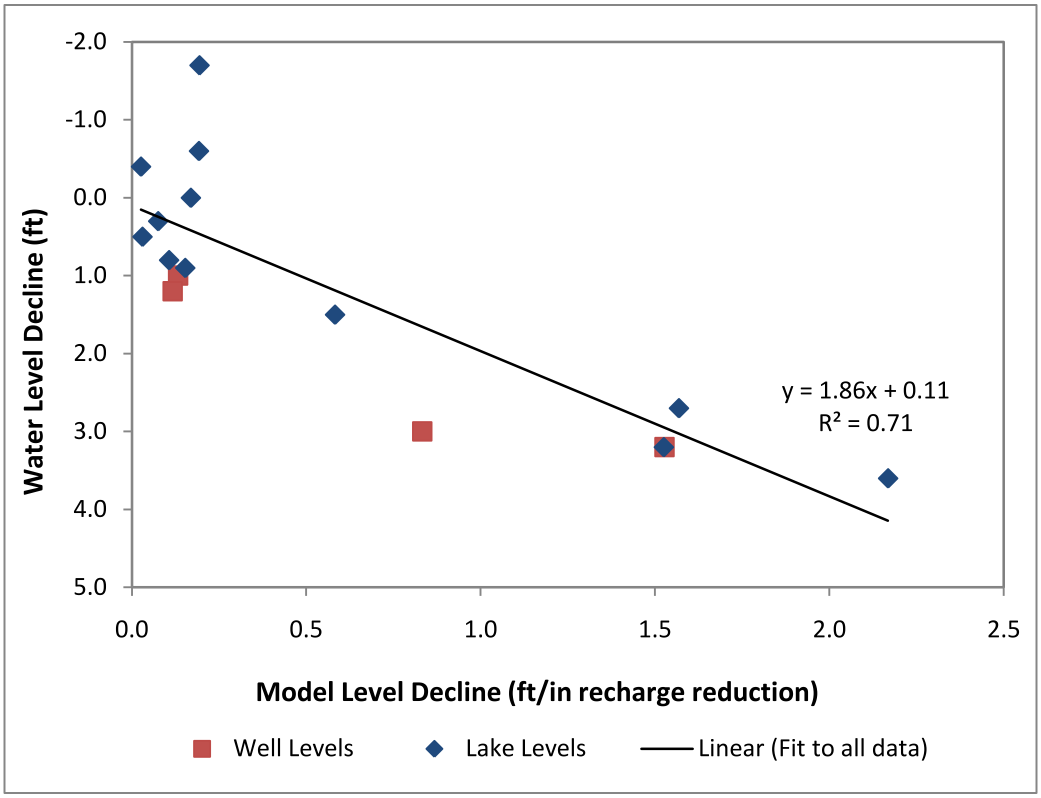


Figure VII-2. Comparison of calculated water level decline beyond weather influences and model level decline per inch recharge reduction.

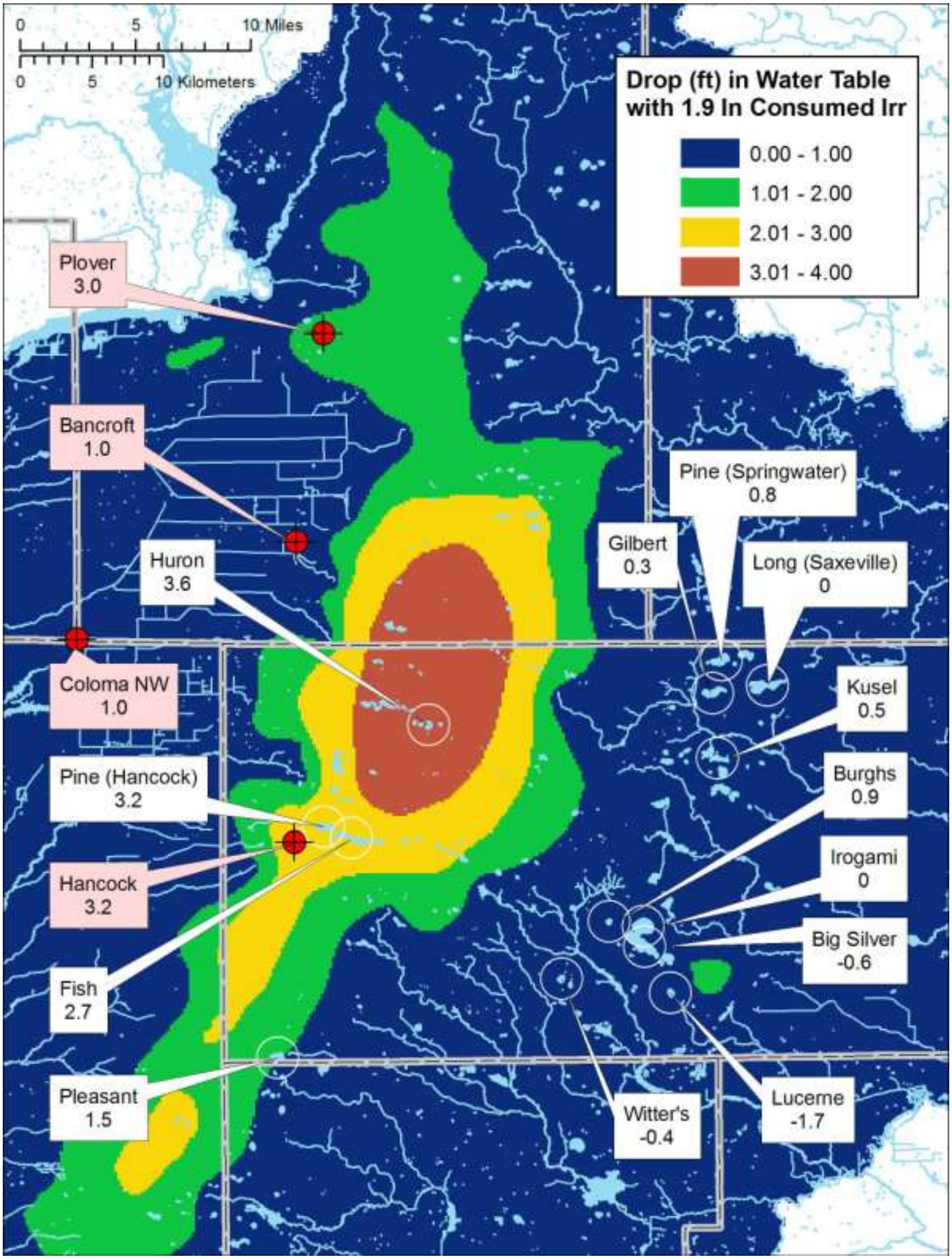


Figure VII-3. Statistically estimated water level declines beyond weather influences at monitoring wells (pink) and lakes (white) compared with modeled declines for 1.9 in reduction in recharge on irrigated lands.

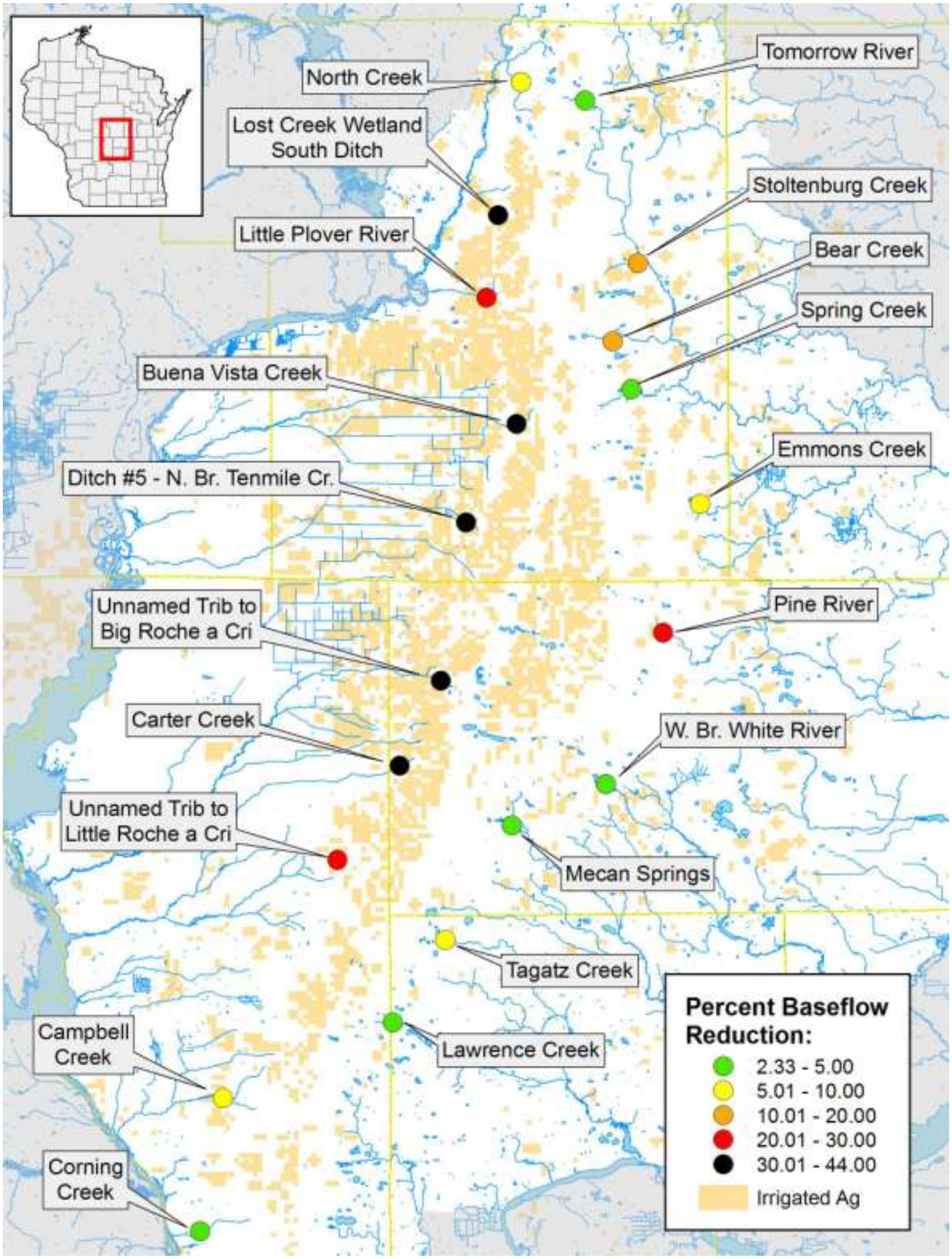


Figure VII-4. Modeled percent steady state ("average" of sorts) flow decline in headwater streams, 1 mile below the source, for 1.9 in recharge reduction on irrigated lands. Note that seasonal and dry year declines would be larger.

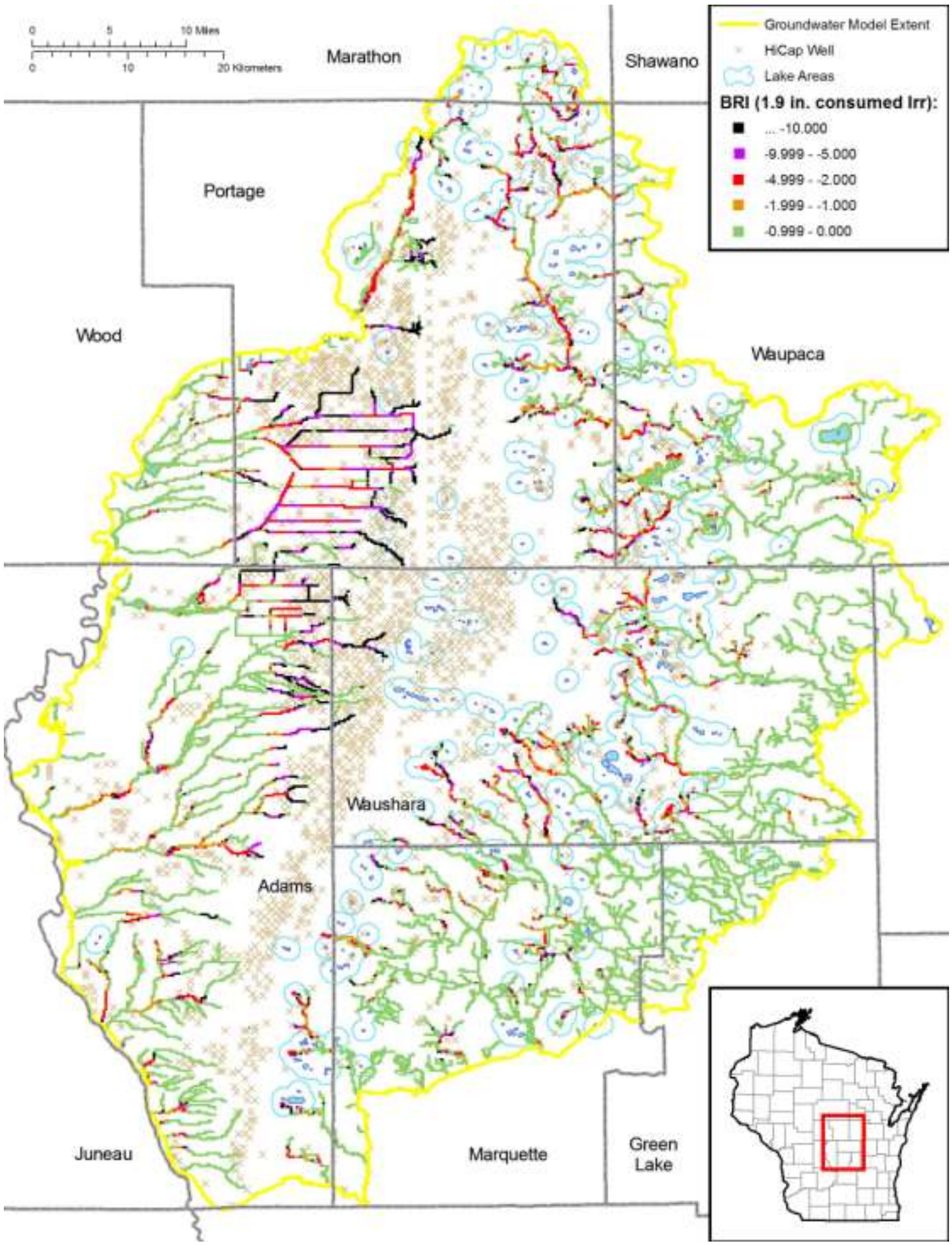


Figure VII-5. Baseflow reduction index for streams in central Wisconsin based on 1.9 in recharge reduction on irrigated lands.

VIII. CONCLUSION

Low water levels are evident in groundwater, lakes, and streamflows in the Wisconsin Central Sands. Some frequently cited examples and consequences are the drying of the Little Plover and Stoltenberg Creek; low water levels in Huron, Pickerel, and Wolf Lakes, likely triggering winterkills in Pickerel and Wolf; and the complete drying of Long Lake – Oasis, Pumpkinseed, and other lakes.

Some low water level phenomena are certainly due to somewhat below average moisture conditions, and some water bodies are only affected by these below average conditions. However, moisture conditions have not been close (as of the end of 2008) to long term record lows and cannot alone explain extremely depressed water levels in some locations. Missing water is evident in monitoring wells, lakes, and streams where substantial groundwater pumping is occurring.

The amount of missing water only explainable by pumping amounts to several feet in some lakes high in the groundwater flow system where high capacity wells are prevalent. Far from high densities of high capacity wells and lower in the groundwater flow system the impacts are muted. Impacts on streams may reach half of their average baseflow in headwater locations.

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