Sustaining Central Sands Water Resources

By

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Advisor’s name (print)

____1-8-2014_________________

Date

Forward

This document was written to provide a common framework and language for scientists to communicate within and across disciplines regarding water resource management in the Central Sands region of Wisconsin. The scope of the work was defined by the author and other members of a small, interdisciplinary committee of scientists after a series of meetings to discuss "a path forward" for the "tragedy of the commons" water resources management challenge in the Central Sands region. The committee included four professors from University of Wisconsin-Madison: Ken Potter (professor in the Department of Civil and Environmental Engineering), AJ Bussan (professor in the Department of Horticulture), Ken Bradbury (program leader at the Wisconsin Geological and Natural History Survey and affiliated faculty in the Department of Geology and Geophysics), Jed Colquhoun (professor in the Department of Horticulture) and me (master's candidate in Civil and Environmental Engineering). Throughout the paper, I use the pronoun "we" to honor the committee's input regarding the project scope and framework although the thesis was composed by me. This paper will be used as a strategic planning document for water resources management throughout Central Sands region.

Acknowledgements

I would like to thank my primary advisor, Ken Potter for applying his far-reaching experience in water resources management to this project. I also thank him for his enthusiasm, humor and unwavering project commitment. I would like to thank AJ Bussan for his visionary guidance on this project as well as for his concern for my personal well-being throughout the process. I would like to thank Ken Bradbury for his constructive comments on sections of the paper involving hydrogeologic principles and for sharing his perspective informed by involvement in a myriad of multi-faceted and conflict-ridden groundwater modeling projects. I thank Jed Colquhoun for his knowledge of stakeholder perceptions of how water resources throughout the Central Sands have changed over time and for his responsiveness when asked questions pertaining to this paper. I thank the United States Department of Agriculture - National Resource Conservation Service for financial support.

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Thank you to my family and friends who tolerated my work schedule and made the journey interesting, fun and enjoyable. Finally, I am forever grateful for the love and support from my partner, Joe.

Table of Contents

Abstract

This document aims to provide a common framework for scientists to communicate within and across disciplines regarding future water resource management throughout the Central Sands of Wisconsin. It reviews interdisciplinary research pertinent for 1) understanding how the land cover and land use of the area has changed over time; 2) documenting long-term changes in regional groundwater levels, lake levels, and stream baseflows and related meteorological variables; and 3) investigating water resource management strategies and frameworks that have been implemented in other regions of the United States and throughout the world. Reviewed research shows that the Central Sands aquifer has been a primary resource for agriculture, human water supply, industrial and stock supplies, recreational activities and ecosystems. Land use and land cover has changed from predominantly prairies, savannahs, and forests in the early 1800's to present-day agricultural, forested and urban areas. From 1948 to 2006, average annual temperatures throughout the region have increased 1 to 2°C and growing seasons have lengthened by 15 to 20 days resulting in increased climate-driven average annual evapotranspiration. During this same time period, total annual precipitation has increased between 5 and 15 cm leading to an increase in climate-driven recharge between 5 and 10 cm, despite increases in average annual evapotranspiration. Meanwhile, regional groundwater levels, lake levels and baseflow in streams have declined over the last several decades with the greatest declines in levels and flows close to the groundwater divide and in streams near groundwater pumping wells. The declines in groundwater and associated surface water levels have been understood as a reduction in recharge estimated between 4.5 and 14.2 cm per year depending on the location in the aquifer. The discrepancy between long-term climate trends and surface water and groundwater observations suggests that land use practices, such as irrigated agriculture,

influence groundwater and associated surface water levels. The paucity of long-term groundwater and surface water datasets has limited analyses that can be conducted without the use of more complex groundwater flow models. Drainage ditches throughout the region have potential to lower local groundwater levels and shorten groundwater flow paths. Regional effects of drainage ditches have not been conducted. A shared-vision planning framework and potential strategies for water resources management are summarized to encourage discussions aimed at implementation of solutions with the greatest potential of success toward meeting the goals previously defined during stakeholder conversations. Future groundwater and surface water management will require comprehensive modeling that can test alternative land use and land management strategies. Key knowledge gaps that currently hinder a comprehensive modeling approach include land management influences on rates of evapotranspiration, data collection and ecosystem valuation. Modeling would need to be conducted on a transient time scale to accommodate annual and inter-annual variations in weather and dynamic land management practices. Given the comprehensive review of the state of water resources science in the Central Sands along with a framework and strategies that have been implemented in other areas of the nation and the world, this document aims to support future strategic planning and implementation of water resources management in the Central Sands region of Wisconsin.

Introduction

The Central Sands region of Wisconsin contains an expansive, unconfined aquifer comprised of highly permeable sediments on the order of 30 m (100 ft) thick that lie upon less-permeable sandstone and bedrock (Butler 1978; Devaul and Holt 1971; Summers 1965). Groundwater in this surficial aquifer supports over 300 lakes and 1,290 km (800 mi) of groundwater-fed streams. In the last several decades, declines in groundwater levels, lake levels, and stream baseflows in parts of the region have occurred and stressed local ecosystems (Kraft et al., 2012). Meanwhile, average annual temperatures have increased, growing seasons have lengthened (Kucharik et al., 2010) and irrigation has expanded to support a thriving agricultural industry throughout the region (Butler, 1978; Keene and Mitchell, 2010). Some observers have attributed declining water levels to the cumulative impact of groundwater pumping for agriculture; others have attributed declining water levels to natural climate variability, climate change, and/or land use change. Understanding the hydrogeological system, reviewing reported evidence of regional declines in water levels, and characterizing strategies for managing surface and groundwater resources is critical for maintaining a thriving agricultural and industrial economy, the livelihoods and interests of rural communities and the health of groundwater-dependent surface waters and their associated ecosystems.

I.1 Purpose

This document is meant to provide a common framework and language for interdisciplinary scientists to communicate within and across disciplines regarding water resource management in the Central Sands region of Wisconsin. To do so, this paper reviews interdisciplinary research on the following topics:

1. Geological, ecological, weather, climate and land use factors affecting Central Sands water resources;
2. Reported evidence of declines in Central Sands groundwater levels, lake levels and stream baseflows and of their causes;
3. Water resource management strategies and frameworks that could potentially mitigate the effects of human groundwater use on aquatic ecosystems, while supporting a thriving agricultural economy.

Although groundwater quality and contamination are important concerns for this region, these topics are beyond the scope of this paper.

I.2 Study Area

This paper focuses on the spatial and temporal distribution of groundwater and surface water throughout Wisconsin's Central Sands. The paper uses a delineation of the Central Sands constructed by the Wisconsin Department of Natural Resources (Robert Smail, personal communication) (Figure 1). This delineation is very similar to the delineation of Kraft et al. (2012) and encompasses an area of about 700,000 ha (1.7 million acres). It includes portions of the Fox-Wolf and Central Wisconsin Basins and spans Adams, Marathon, Marquette, Portage, Shawano, Waupaca, Waushara, and Wood Counties.

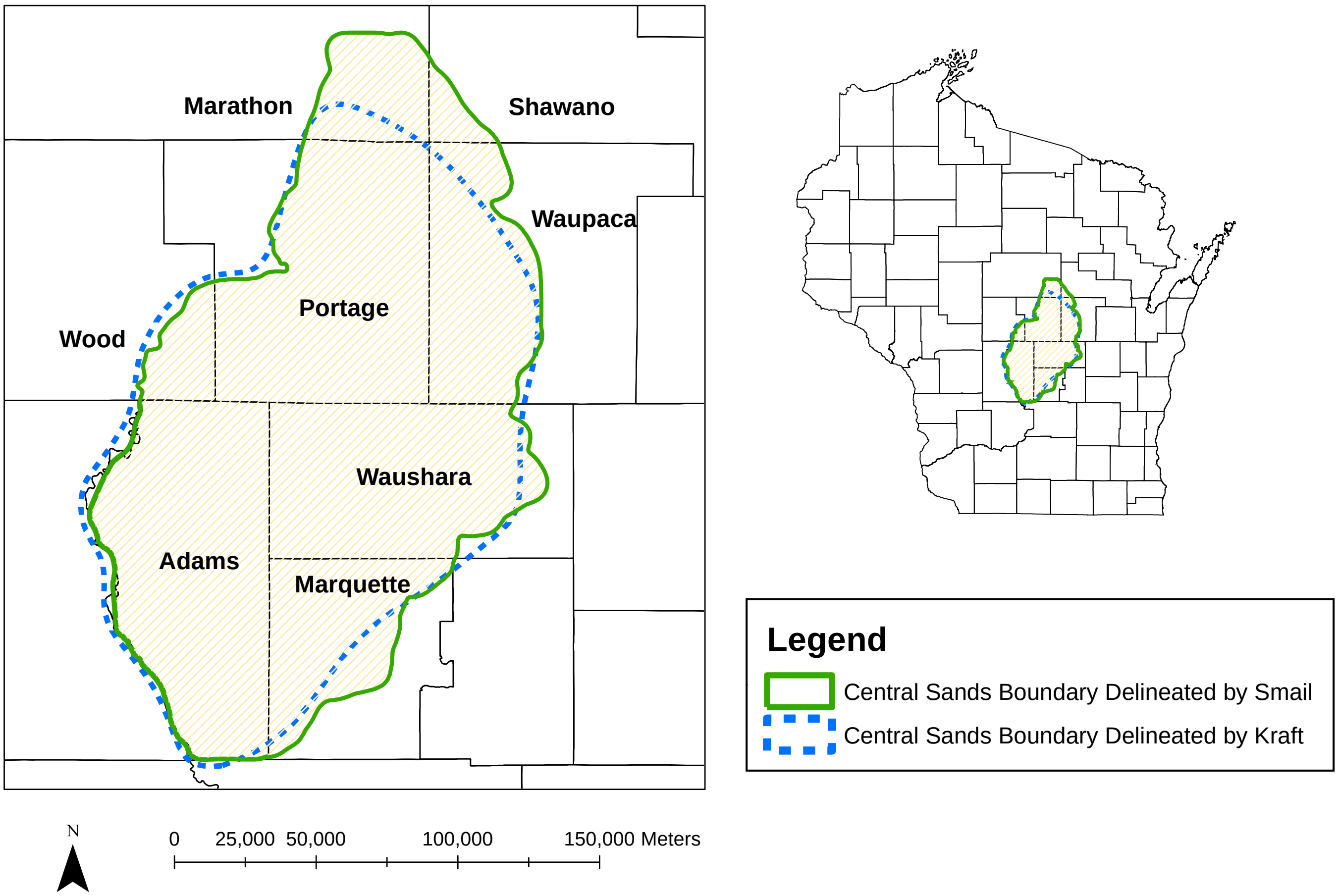


Figure 1. Central Sands region of Wisconsin as delineated by Smail (2012) (solid line) and Kraft et al. (2012) (dotted line) covering approximately 700,000 ha (1.7 million acres). The region includes sections of Adams, Marathon, Marquette, Portage, Shawano, Waupaca, Waushara, and Wood Counties.

Methodology

This document reviews published literature in multiple disciplines including, but not limited to geology, ecology, geography, hydrogeology, hydrology, biophysics, agronomy, horticulture, and water resource management. Since much of the work specific to the Central Sands has been conducted by researchers from local and state agencies and universities, the document reviews a wide range of publication types (Table 1). This document has been reviewed by numerous University-System and public agency personnel involved in Central Sands research.

Table 1. Types of literature reviewed in this paper and referenced databases.

Types of Literature	Databases
Peer-reviewed literature	Web of Science Engineering Village Agricola
U.S. Geologic Survey (USGS) technical papers	USGS online library
Wisconsin Geologic and Natural History Survey technical papers	WGNHS library
Wisconsin Department of Natural Resources (WDNR) technical papers	WDNR library
Master's and PhD theses and dissertations	UW-Madison library system collections

Terrestrial and Aquatic Ecosystems

In the early 1800's, landscapes throughout the Central Sands supported a mixture of prairies, savannahs, and forests (Figure 2) (Mladdenoff and Sickey, 2009). These ecosystem types have been separated into three ecological regions: the Central Sand Plain, Central Sand Hills and the Forest Transition (Figure 3) (WDNR website, 2013). Deciduous and evergreen forests containing *Quercus alba* (white oak), *Quercus rubra* (red oak) and *Quercus velutina* (black oak), *Acer rubrum* (red maple), *Ulmus spp.* (elm), *Pinus strobus* (white pine), *Pinus resinosa* (red pine) and *Pinus banksiana* (jack pine) and other species were spread throughout the region (Figure 4) (De Byle, 1957; Mladdenoff and Sickey, 2009). Aspen, willow and tamarack were generally found in areas of poorly drained soils, which surrounded the marshes (Staatz, 1933). Common species present in forest understories included hazel, sweet fern *Vaccinium parvifolium* (huckleberry), *Vaccinium myrtillus* (blueberry), *Pteridium spp.* (bracken fern), and *Carex pensylvanica* (Pennsylvania sedge) (Staatz 1933). The region was home to large mammals such as *Canus lupus* (timber wolf), *Ursus americanus* (black bear), and *Martes pennanti* (fisher).

Aquatic ecosystems including lakes, streams, wetlands, rare coastal plain marshes, fen and sedge meadows were also located throughout the region. Cool, oxygen-rich springs that connected groundwater flow to lakes, streams, fen-meadows, wetlands and other surface water bodies supported a diversity of species (Macholl, 2007; Swanson, 2013). The spring-fed aquatic ecosystems provided breeding grounds for resident and migratory birds among other aquatic species (Wisconsin Department of Natural Resources, 2006). Calcareous fens, often associated with sedge meadows, created habitats for calciphiles, plants that thrive on lime rich soil (Wisconsin Department of Natural Resources, 2006). Additionally, *Salvelinus fontinalis* (brook trout) depended on cold, groundwater-fed streams throughout the area (Kraft et al., 2010), while kettle lakes supported waterfowl and fish-feeding shorebirds (Wisconsin Department of Natural Resources, 2006).

Today, many of the terrestrial and aquatic ecosystems present throughout the Central Sands landscapes in the early 1800's remain interspersed throughout urban, agricultural, and managed pine plantation landscapes (Figure 5). Maintenance of certain variables (i.e. water temperature, flows, water quality) within these habitats has been identified as critical for supporting the integrity of the ecosystems (Wisconsin Department of Natural Resources, 2002). For example, brook trout depend on the cold temperatures in the baseflow fed streams (Kraft et al., 2010). Additionally, certain rare and endangered species present in the region have been identified as conservation priorities including the Karner Blue Butterfly, Massasuga rattlesnake, Blanding's turtle, rare herptiles and large mammals (Wisconsin Department of Natural Resources, 2002).

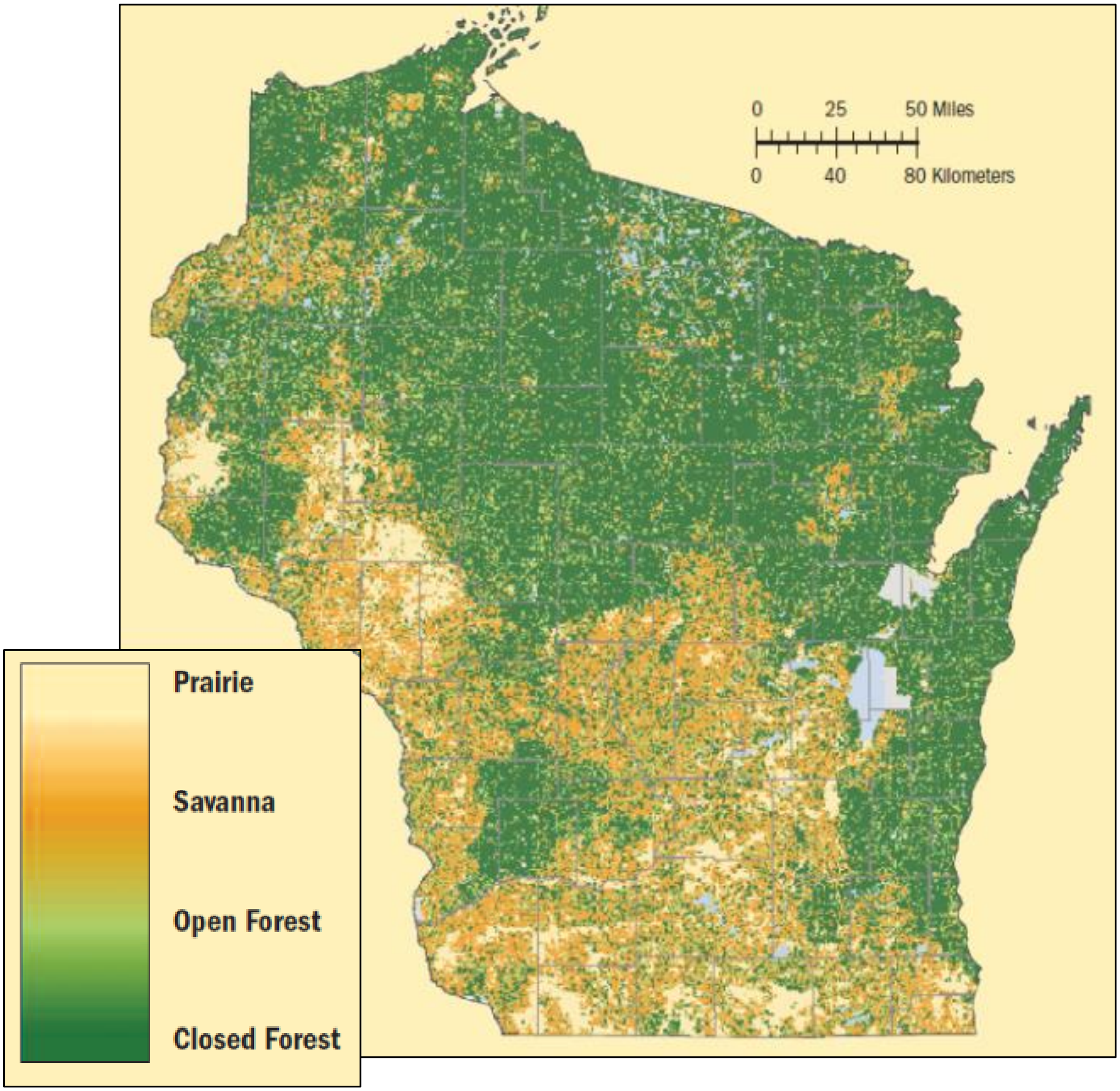


Figure 2. Pre-European settlement forest density map illustrating relative gradients of openness as determined from data collected by U.S. Public Land Survey System contractors between 1832 and 1866 and interpreted by Mladdenoff and Sickey (2009).

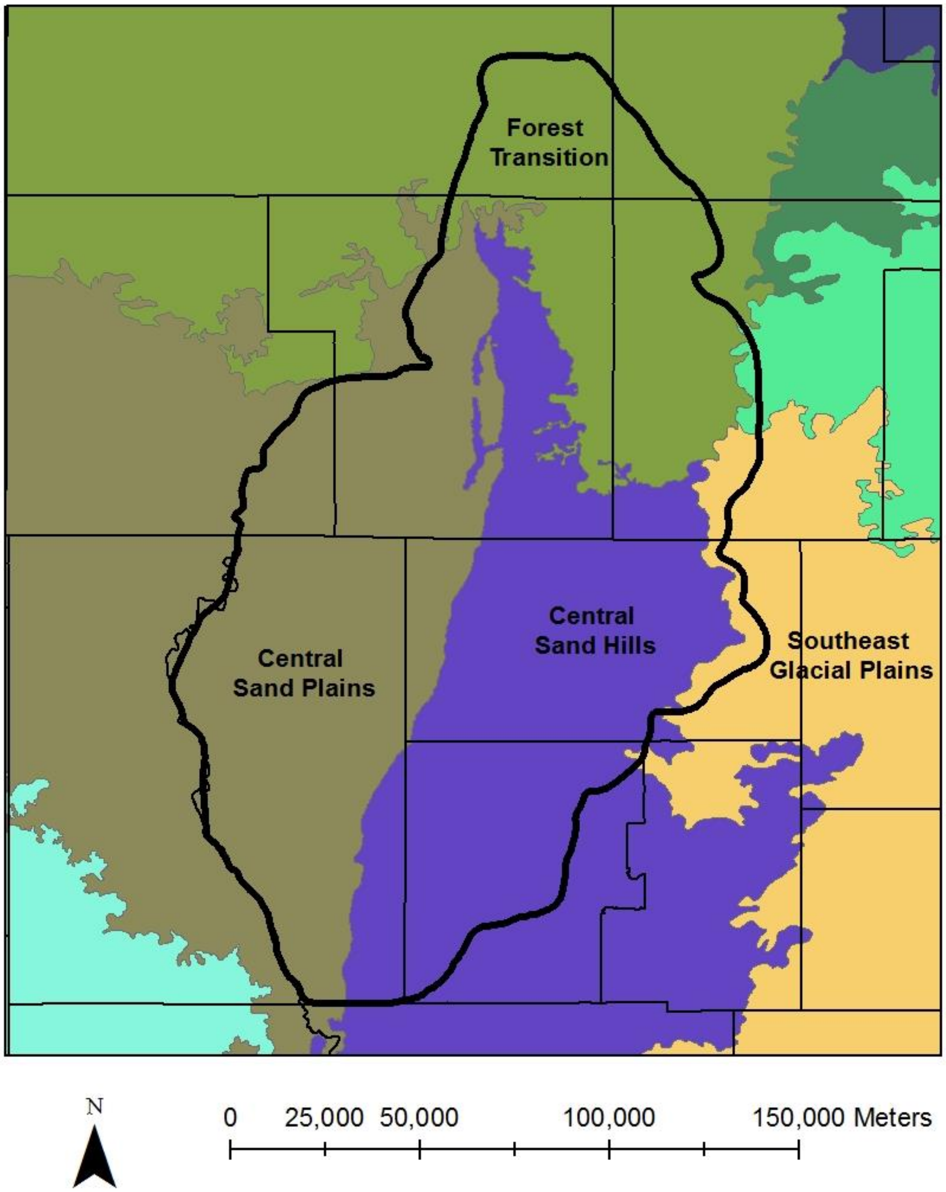


Figure 3. Central Sands ecological landscapes as designated by the Wisconsin Department of Natural Resources (WDNR website, downloaded November 2013).

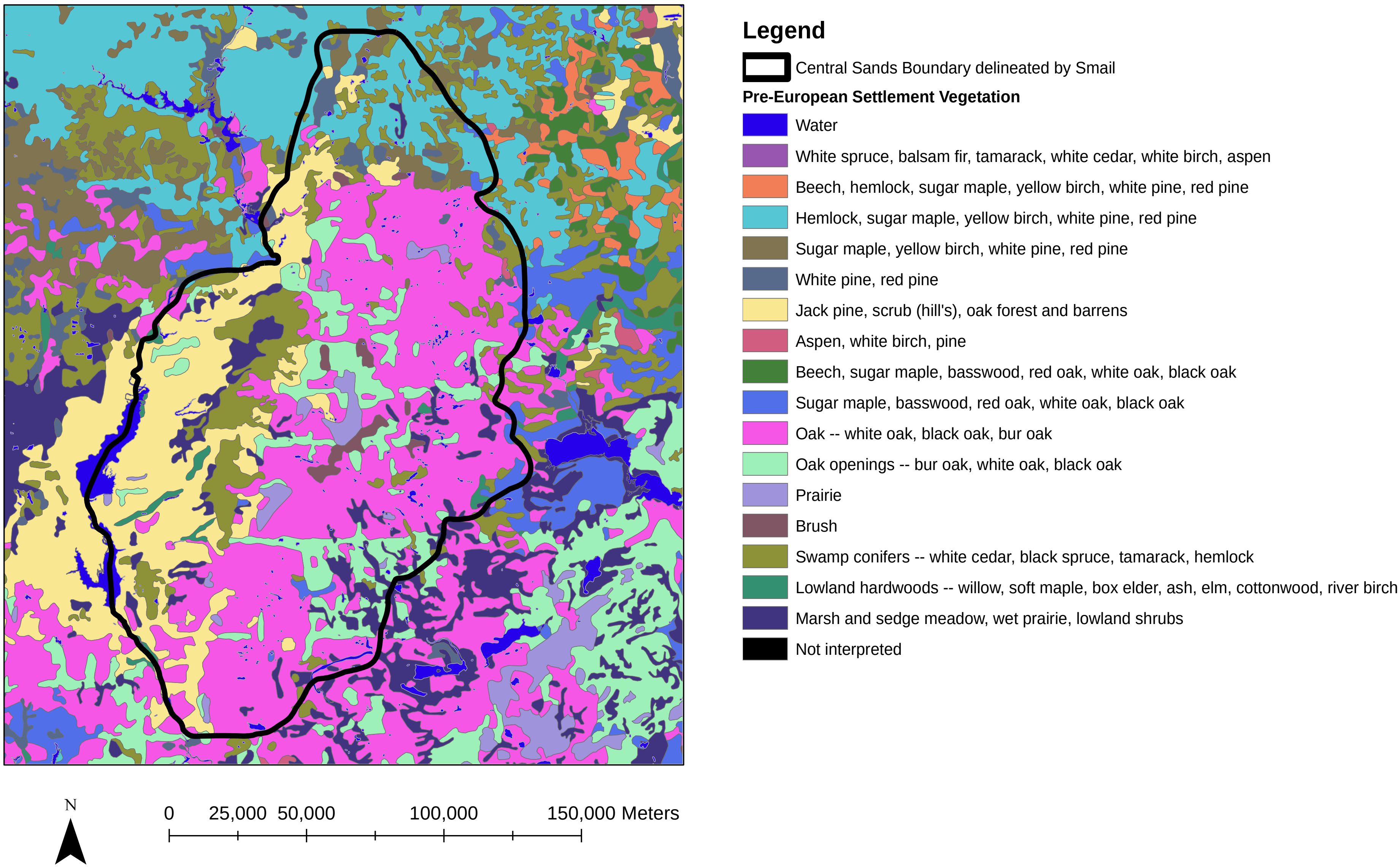


Figure 4. Dominant land cover type present throughout the Central Sands prior to European settlement as surveyed by U.S. Public Land Survey System contractors between 1832 and 1866. Referenced land cover types were interpreted and compiled by Mladdenoff and Sickey (2009).

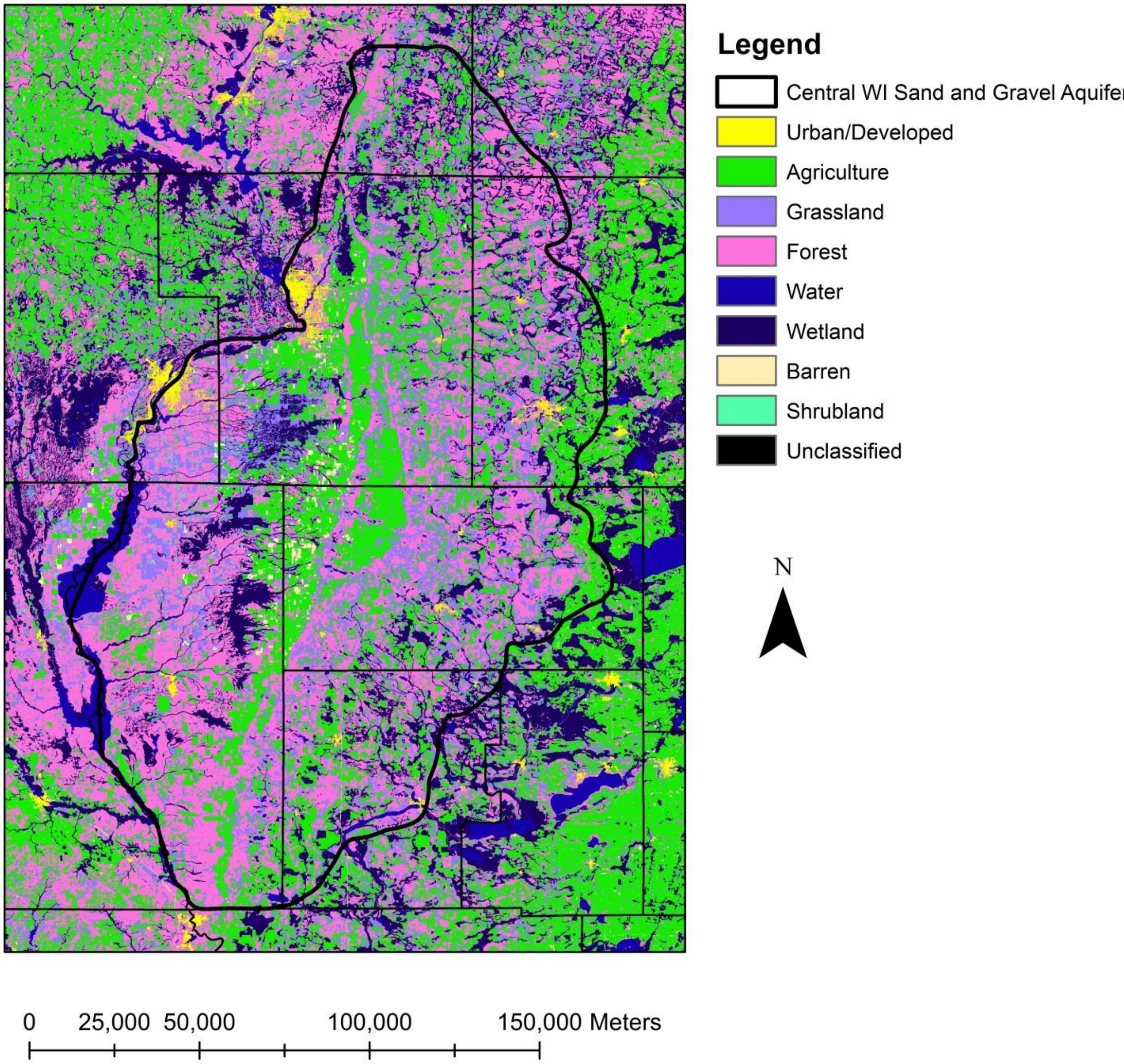


Figure 5. Present-day land cover and land use. Map based on data available through the WDNR website (2013).

Regional Settlement and Land Use Change

Before European settlement, Native Americans inhabited the Central Sands and depended on the water resources and local ecosystems for transportation and sustenance. They traveled by foot and canoe and fished in the streams, lakes and wetlands (Butler, 1978). Native Americans also managed the land with fire, periodically raised small garden plots and hunted large animals (Butler, 1978). These tribes were largely unaffected by European migration until 1787 when a treaty between the US government and the Menominee tribe was signed (Staats, 1933). The treaty claimed Wisconsin as part of the "Northwest Territory" and established US ownership of a 64 x 5 km^2 (40 x 3 mi^2) strip of land along both sides of the Wisconsin River (Staats, 1933).

Major land use change began in the mid- to late-1800s when pine lumbering became the center of the economy throughout the region and to the north (Butler, 1978). Logging provided jobs for a growing population of migrant European settlers in Central Wisconsin and led to the emergence of town centers and railroads (Staats, 1933). Meanwhile, dams were built on the Wisconsin River to create reservoirs that maintained a relatively constant flow of water to carry logs downstream to newly-constructed sawmills (Staats, 1933). Surface water resources supported by spring flow were sources of water for early homesteads and livestock (Swanson, 2013).

The logging period lasted until the 1870s when central Wisconsin pines were harvested and subsequently burnt by drought-induced fire (Butler, 1978; Steinhacker et al., 1973). With the disappearance of the pines, inhabitants shifted to farming, first wheat and then dairy, with varying success. Overgrazed and tilled land with little remaining vegetation led to sand "blowouts" and shifting dunes (Steinhacker et al., 1973). In the late 1800s, only small dairies and paper-making industries were scattered throughout the region (Butler, 1978; Staats, 1933).

In the early 1900s, perennially flowing drainage ditches were constructed in the western portion of the Central Sands to drain wetlands to facilitate settlement and agriculture expansion (Figure 6) (Butler, 1978; Weeks et al., 1965). Many ditches deepened, widened and straightened perennial flowing streams already existing in the area. These generally east-west trending ditches were between 3 to 5 m wide (Chambers and Bahr, 1992; Zheng et al., 1988). Settlers established crops on the drained, organic soils. However, the majority of farms were soon abandoned due to inadequate soil moisture, soil erosion and nutrient depletion (Butler, 1978). Farmers that remained in the area employed techniques to control erosion and increase soil fertility and irrigated with surface water on a small scale (Hindall, 1978).

In the late 1940s and early 1950s, the construction of high-capacity wells and development of the center pivot irrigation system spurred a new agriculture economy (Hindall, 1978). Well construction was enabled by reduced metal prices after World War II as well as diesel and electrical energy development (Butler, 1978; WDNR, 2002). Center pivot irrigation systems were generally constructed on the flat outwash sediments between rocky morainal deposits (Figure 6) (Butler, 1978; Gutenberg, 2009; Last, 1983). The well-drained soils and accessible supply of groundwater for irrigation in the outwash sediments supported the growth of crops.

Throughout the next half-century, irrigation steadily expanded to support agricultural production and with it came job availability and the expansion of cities and towns throughout the region. By 2012, over 2,000 high capacity irrigation wells had been installed within the Central Sands (Figure 7) (WDNR, 2012), enabling the production of potatoes, processing vegetables including snap beans, sweet corn, beets, cucumber, carrots and peas and agronomic crops such as field corn, soybean, wheat and alfalfa. Central Sands growers use central-pivot irrigation systems that pump groundwater at a rate of about 3,785 liter/min (1,000 gal/min) to supplement precipitation and maximize crop yield (WDNR, 2010).[1] Annual pumping volumes vary with precipitation; years with less summer precipitation require more irrigation and vice versa. An economic analysis conducted in 2010 showed that the regional agricultural industry has provided $6 billion of annual economic activity for the state of Wisconsin, $750 million of which has been derived from vegetable and fruit sales (Keene and Mitchell, 2010). The regional agricultural industry also supports Wisconsin's rank as second in the production of processed vegetables in the United States and top supplier of cranberries and ginseng (Keene and Mitchell, 2010).

In addition to irrigated agriculture, the Central Sands water resources (lakes, streams, springs, wetlands and groundwater) support other human uses such as municipal drinking water supplies, industrial supplies, domestic and stock supplies, pine plantations, and recreational activities including fishing, boating, swimming, hiking, camping and bird watching. All types of groundwater use have increased over time. In 2011, Portage, Adams and Waushara Counties were ranked 1, 3 and 4, respectively, for the counties with the greatest groundwater withdrawals in the State of Wisconsin (Figure 8A) (WDNR, 2011). However, in the Central Sands, the amount of groundwater used for municipal, industrial and domestic and stock purposes is small relative to the amount of groundwater used for irrigated agriculture as demonstrated by a water use time series for Portage County (Figure 8B).

[1] High discharging wells fall under the administrative definition of "high capacity" wells, or those that pump 270 liters/min (70 gal/min) or more, and are under state regulation.

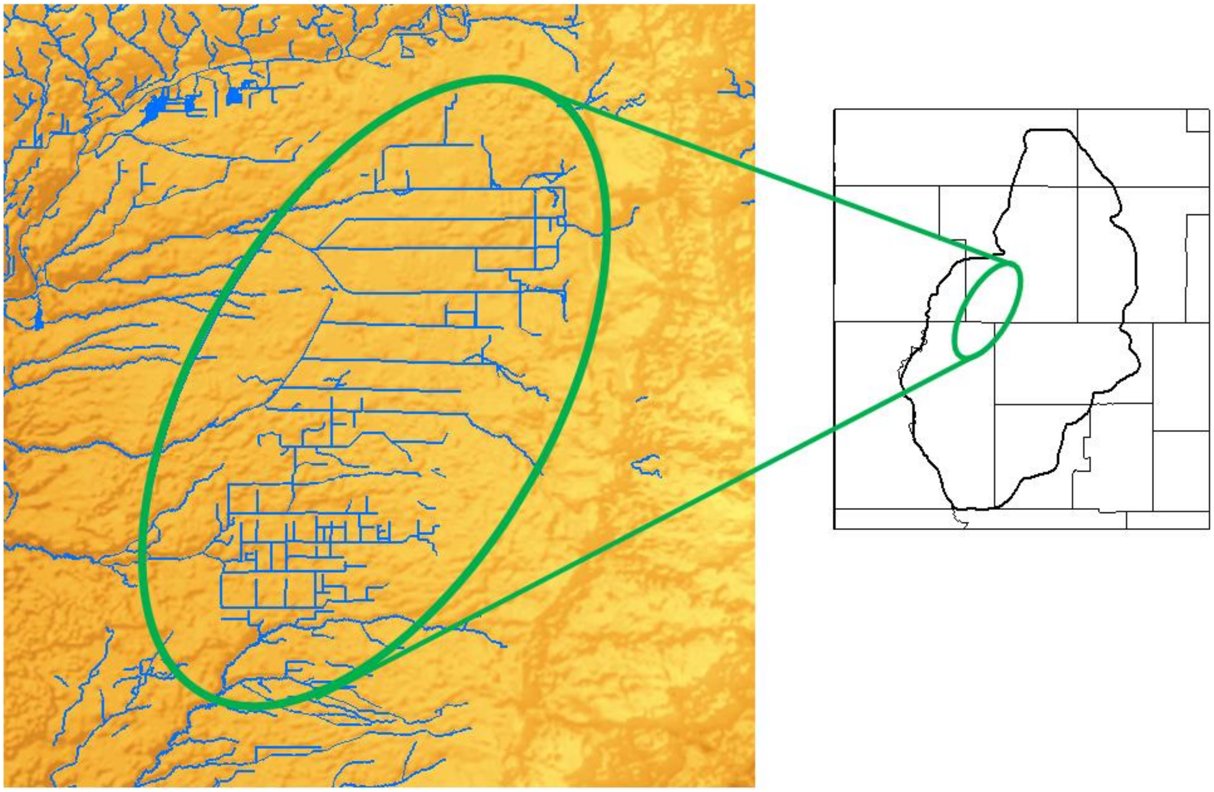


Figure 6. Example of the presence of drainage ditches (blue, straight lines positioned at right angles) in portions of the Central Sands. The figure is based on a section of the Hydro database available through the WDNR website (2013).

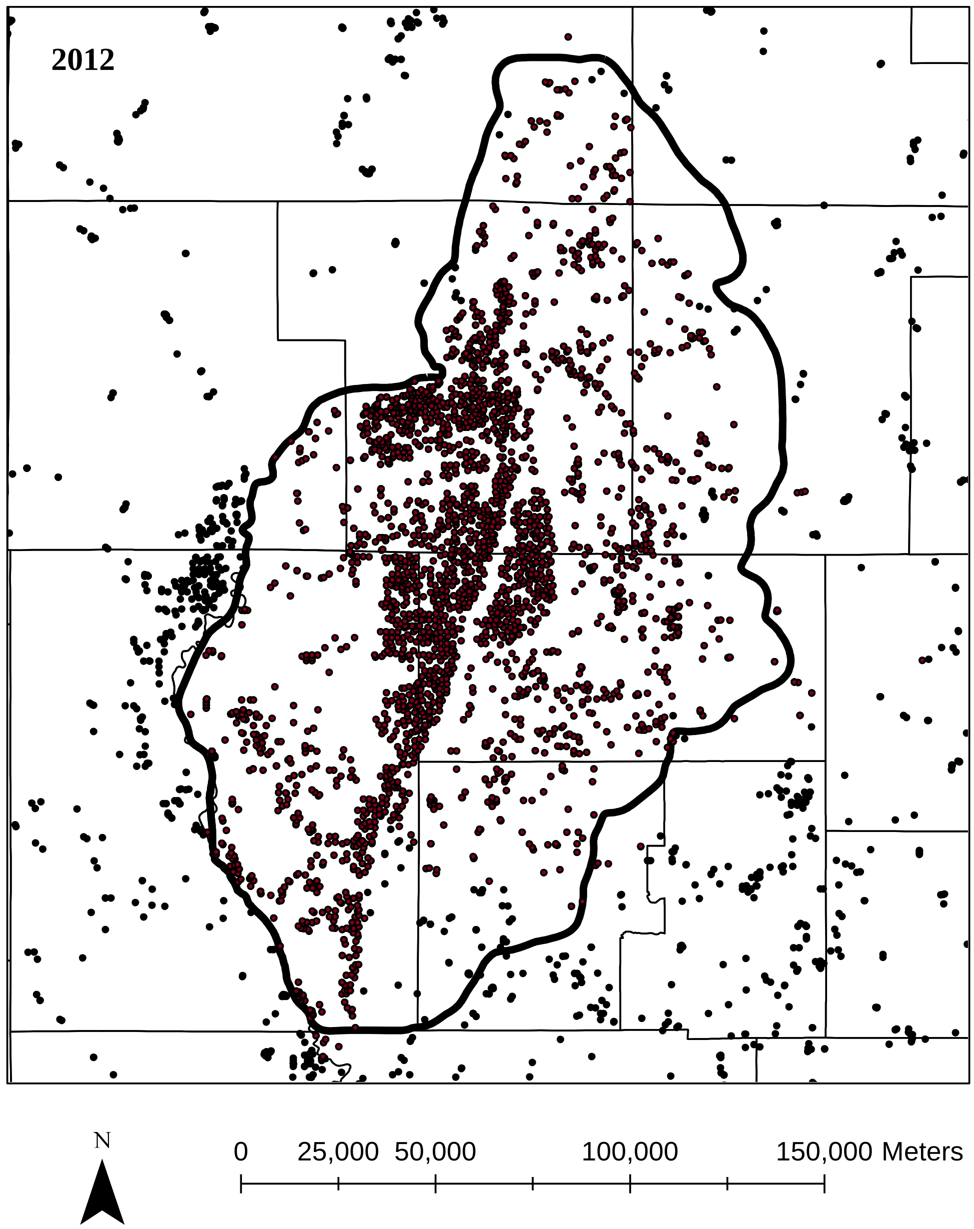
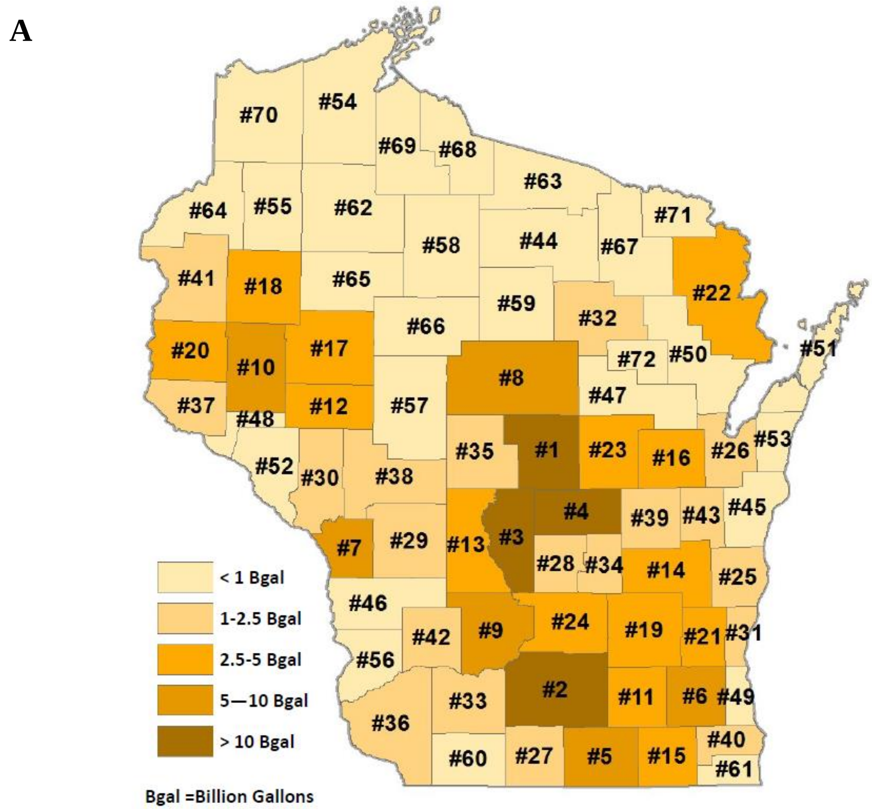


Figure 7. High capacity pumping wells present throughout the Central Sands in 2012 (WDNR, 2012).



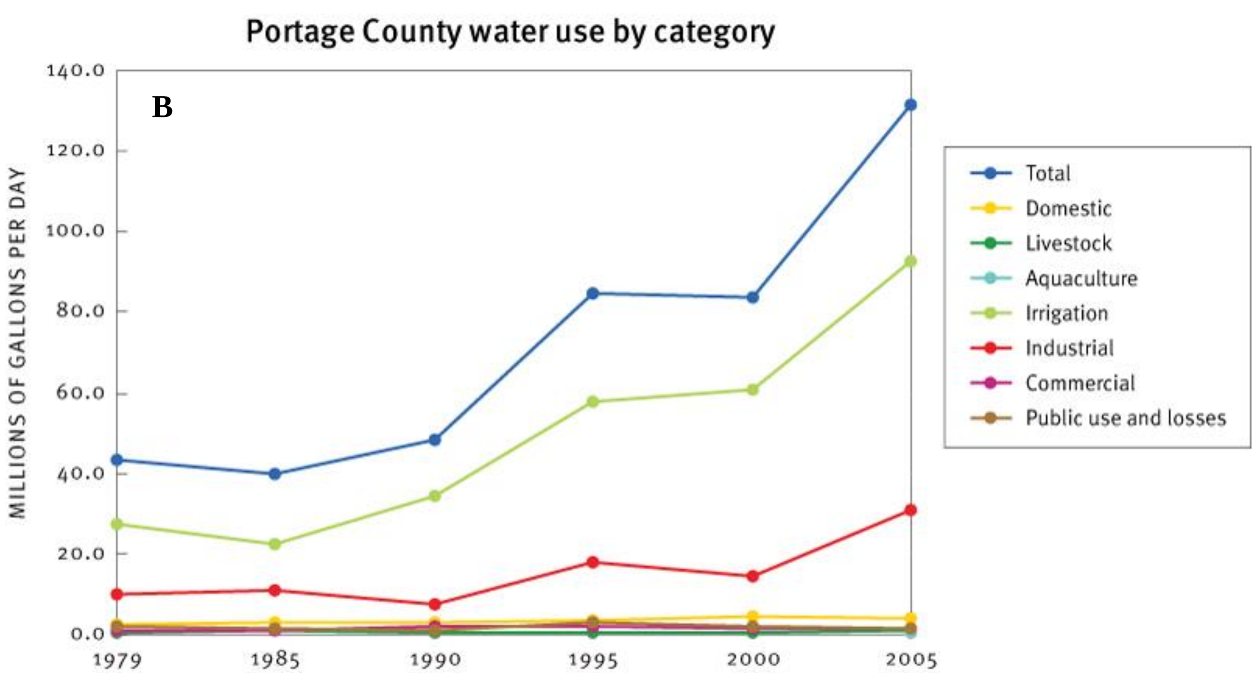


Figure 8. *A:* Ranking of total groundwater withdrawals by county for 2011. *B:* Portage County water use by type. Water use data from U.S. Geological Survey Water Use in Wisconsin reports for calendar years 1979, 1985, 1990, 1995, 2000 and 2005 ("Protection Wisconsin's Groundwater Through Comprehensive Planning" website, 2007).

Central Sands Hydrogeology

During the Pleistocene Epoch, a combination of glacial, glaciolacustrine and glaciofluvial processes deposited unlithified sediments throughout the Central Sands (Attig et al., 1989). Between 10,000 to 14,000 years ago, eolian (wind-related) processes deposited sand dunes and sand sheets on the eastern edge of the Central Sand Plain that are now 2 m to 18 m (7 ft to 60 ft) in relief (Rawling et al., 2008). This section reviews the geologic processes that formed the unconfined aquifer present throughout the Central Sands, and summarizes aquifer characteristics, groundwater flow patterns, and recharge rates throughout the region. This section also discusses the general effects of groundwater pumping wells and drainage ditches on the groundwater system.

I.1 A Brief Geologic History of the Region

A combination of glacial, glaciolacustrine and glaciofluvial processes deposited unlithified sediments throughout the Central Sands approximately 15,000 to 18,000 years ago when the Green Bay Lobe of the Wisconsin Laurentide Ice Sheet reached its maximum extent (Figure 9A) (Attig et al., 1989; Clayton and Moran, 1982). These sediments lie over crystalline bedrock of Precambrian age or, where present, sandstone of Cambrian age (Rawling et al., 2008; Weeks and Stangland, 1971). Cambrian sandstone extends throughout Adams, Waushara, southern Portage County and southern Wood County with erratic surficial exposures. For information regarding these bedrock units, see Weeks et al. (1965), Weeks and Stangland (1971), Clayton (1986), or Clayton (1987).

The Green Bay Lobe of the Wisconsin Laurentide Ice Sheet repeatedly advanced and retreated creating north-south trending terminal moraines comprised of outwash and till (Clayton & Knox, 2008). These moraines impeded the southern flow of the Wisconsin River and formed Glacial

Lake Wisconsin over a portion of the present day Central Sands (Figure 10A) (Rawling et al., 2008). Glacial Lake Wisconsin is estimated to have reached a maximum depth of 306 m (1000 ft) before it drained at the Devils Lake gorge (Rawling et al., 2008).

The former Glacial Lake Wisconsin lakebed is made up of layers of outwash, stratified sands and fine-grained sediment layers. The fine-grained sediments, including the New Rome Member,[2] are comprised of bluish-grey silts and reddish brown clays with an average thickness of 0.5 to 2 cm (0.2 to 0.8 in) (Attig et al., 1989; Brownell, 1986; Faustini, 1985). The presence of these fine-grained sediments suggests that the lake once covered approximately 537,500 ha (1.3 million ac) (Brownell, 1986), or approximately 76% of the Central Sands area delineated by Smail (2013).

Fine silts and clays have also been found in areas of the Central Sands that contained former tunnel channels. Typically 0.5 km wide and generally 10 to 20 km in length,[3] these long, narrow channels ran perpendicular to the margin of the Green Bay Lobe (Figure 10) (Attig *et al.*, 1989). Cutler et al. (2002) used a time dependent, flow line ice sheet model to investigate tunnel channel formation and concluded that tunnel channels were the result of subglacial melt water streams that developed due to the high pressure of the glacier on the permafrost (Figure 9B). After the episodic meltwater flow events, channel walls collapsed and filled with ice fragments that were insulated by debris (Attig et al., 1989). When the glacier and ice fragments melted, the channels became areas of lower elevation. Today, many former tunnel channels contain seepage lakes.

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[2] Brownell (1986) first described the stratigraphic classification for the New Rome Member, the fine sediment layers deposited in glacial Lake Wisconsin when the Green Bay Lobe blocked the Wisconsin River. Later Attig and others (1988) formalized the classification as part of the Big Flats Formation. The classification of the New Rome member indicates that the layers are regionally extensive and mappable.

[3] The approximate length of tunnel channels in the Central Sands region was interpreted from Figure 1 in Attig et al. (1989). Tunnel channels in northern Wisconsin reach up to 40 km in length.

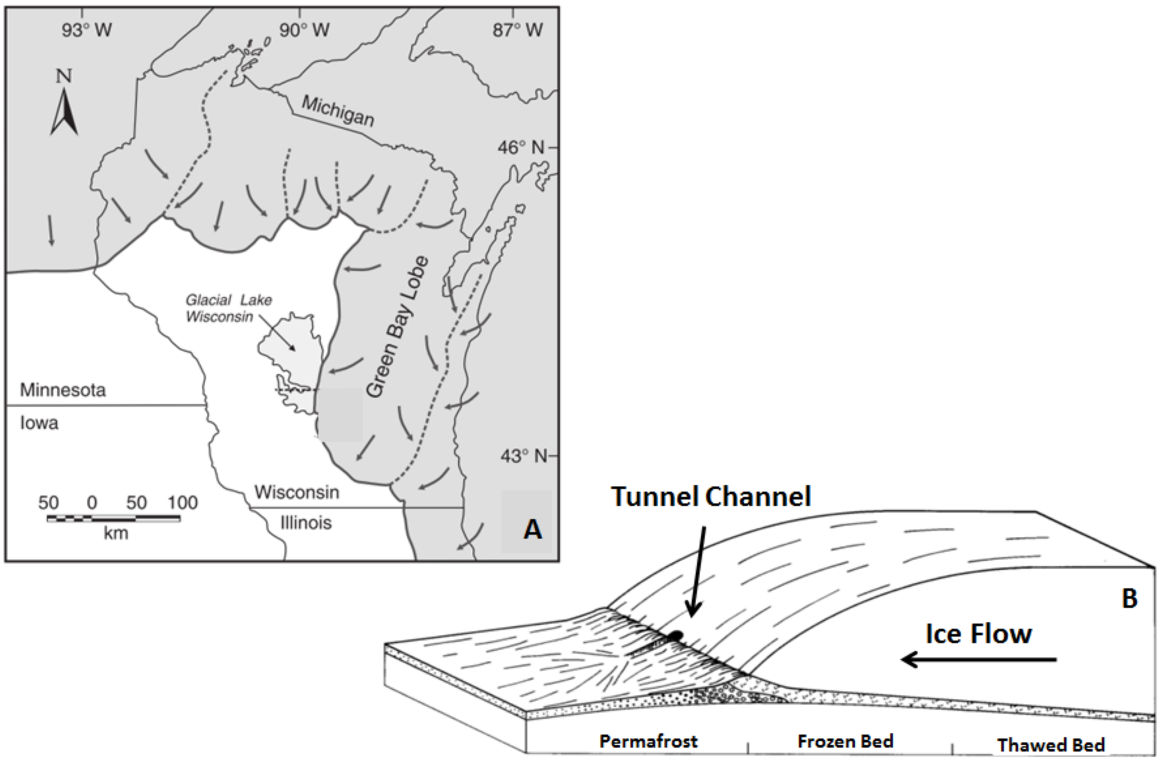

Figure 9. *A:* Furthest extent and flow direction of the Green Bay Lobe of the Wisconsin Laurentide Ice Sheet and location of Glacial Lake Wisconsin (Attig et al., 2011). *B:* Illustration of tunnel channel formation along the ice sheet perimeter (Attig et al., 1989).

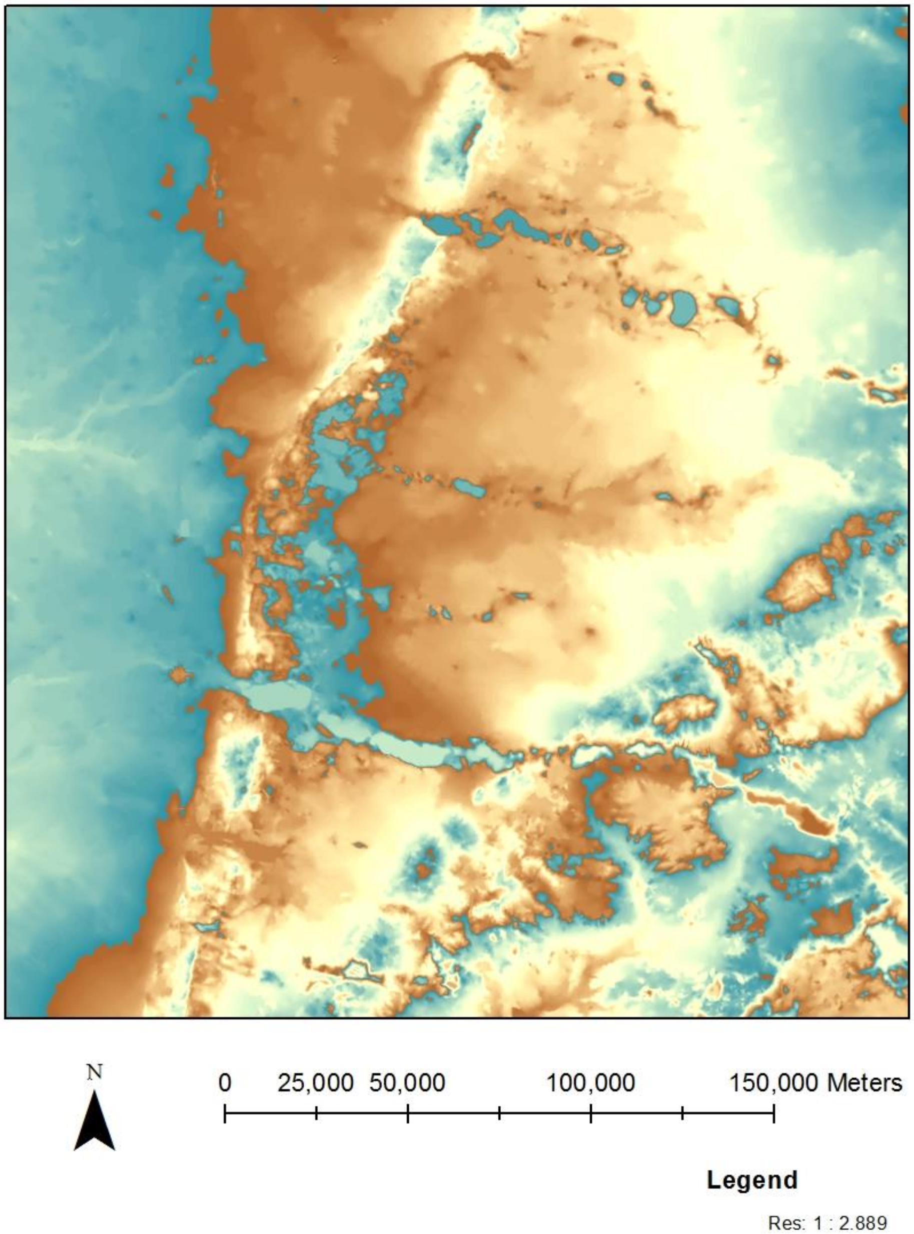

Figure 10. Digital elevation model showing tunnel channels present in the Central Sands. Blue color indicates areas of lower elevation; brown color indicates areas of higher elevation. Present-day Long Lake-Plainfield and Fish Lake are located within the tunnel channels illustrated above. The map is based on a 30 m Digital Elevation Model available through the WDNR website (2013).

Eolian processes also deposited sands throughout the eastern edge of the Central Sand Plain. Research using optically stimulated luminescence dating estimated that sand dune deposition was active between 10,000 to 14,000 years ago (Rawling et al., 2008). Dune formation in central Wisconsin was likely due to the availability of sand from outwash in the Wisconsin River valley or the melting of permafrost (Rawling et al., 2008). Today, eolian sand dunes are typically a few meters thick and range between 2 to 18 m in relief (Rawling et al., 2008) .

To conceptualize the stratigraphy throughout the Central Sands, Brownell (1986) and Clayton (1986, 1987, 1991) created cross sections using data from bore holes, fire protection well records and well-driller logs. As demonstrated in the cross sections in Figures 10 and 11, the region is generally covered by unlithified sediments with thin, intermittent layers of fine sediments and sporadic sandstone exposures. The cross-section topography shows a gradual topographic gradient with highest elevation areas located at the glacial terminal moraines.

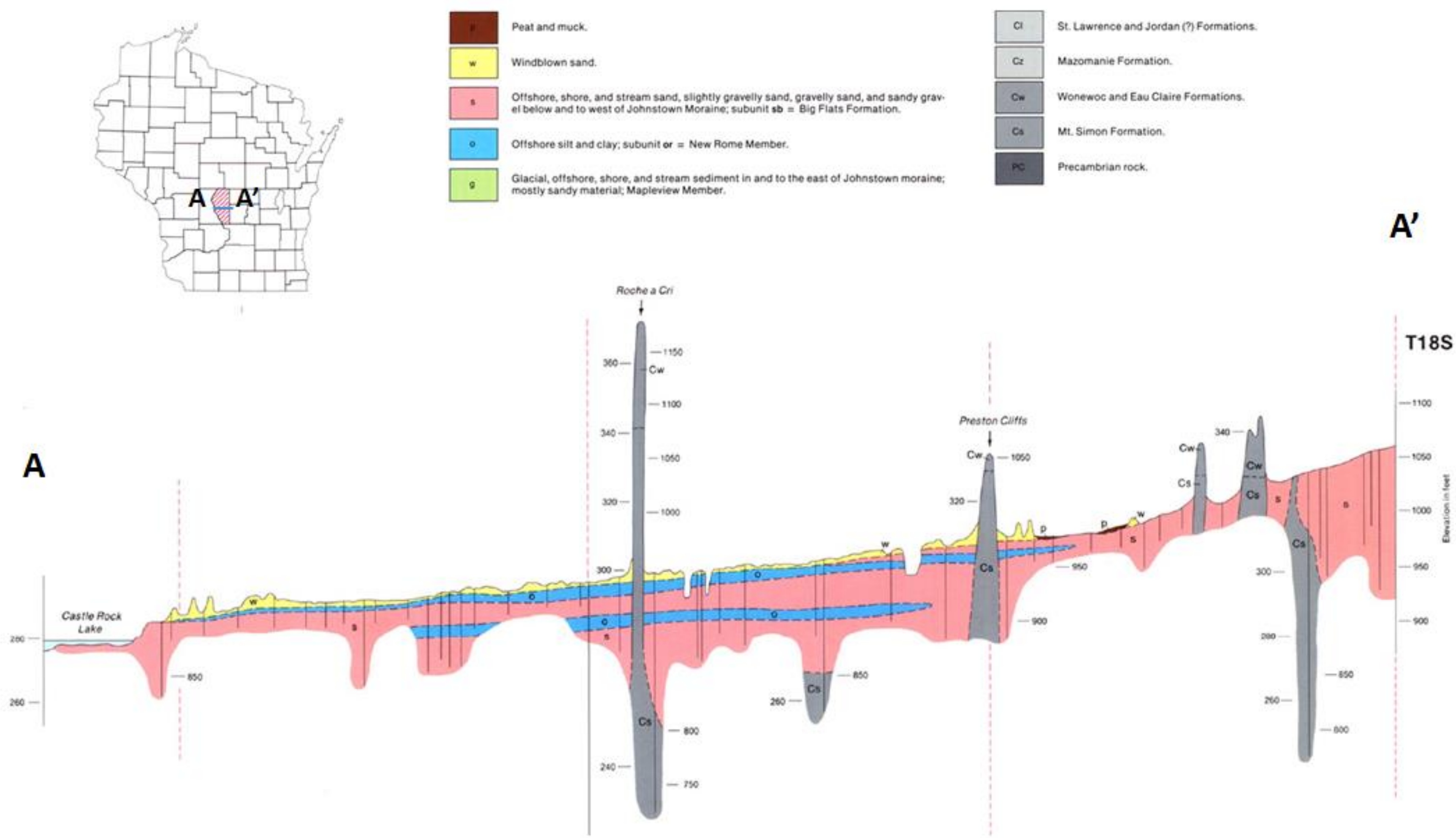

Figure 10. Stratigraphic cross-section (west-east) through Adams County from the Wisconsin River to the eastern county border (Clayton 1987).

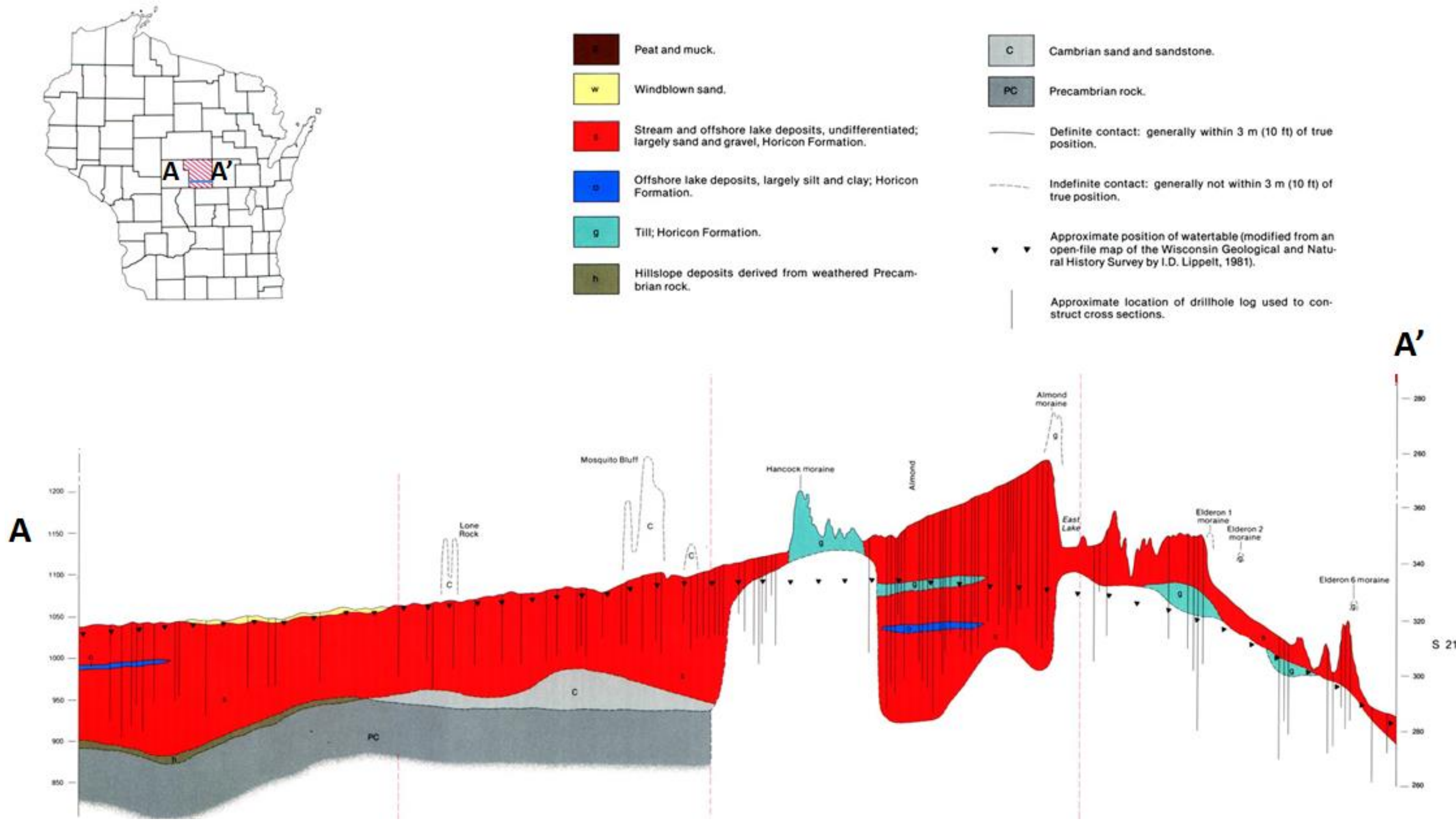

Figure 11. Stratigraphic cross-section (west-east) through southern Portage County (Clayton, 1986).

I.2 Aquifer Characteristics

Where they lie beneath the water table, unlithified sands and gravels throughout the Central Sands comprise a regional unconfined aquifer interspersed with silts and clays. The unlithified sediment thickness throughout the region averages between 30 to 50 m (100 to 160 ft) (Figure 12) (interpreted from Kraft and Mechenich 2010) and attains a thickness over 60 m (200 ft) in bedrock valleys (Weeks and Stangland, 1971). Unlike confined aquifers, for which a low-permeability unit, or aquitard, forms the top of the aquifer, the upper boundary of an unconfined aquifer is the water table. The aquifer is directly connected to local surface water bodies. The Cambrian sandstone present below the unlithified sediments is also an aquifer.

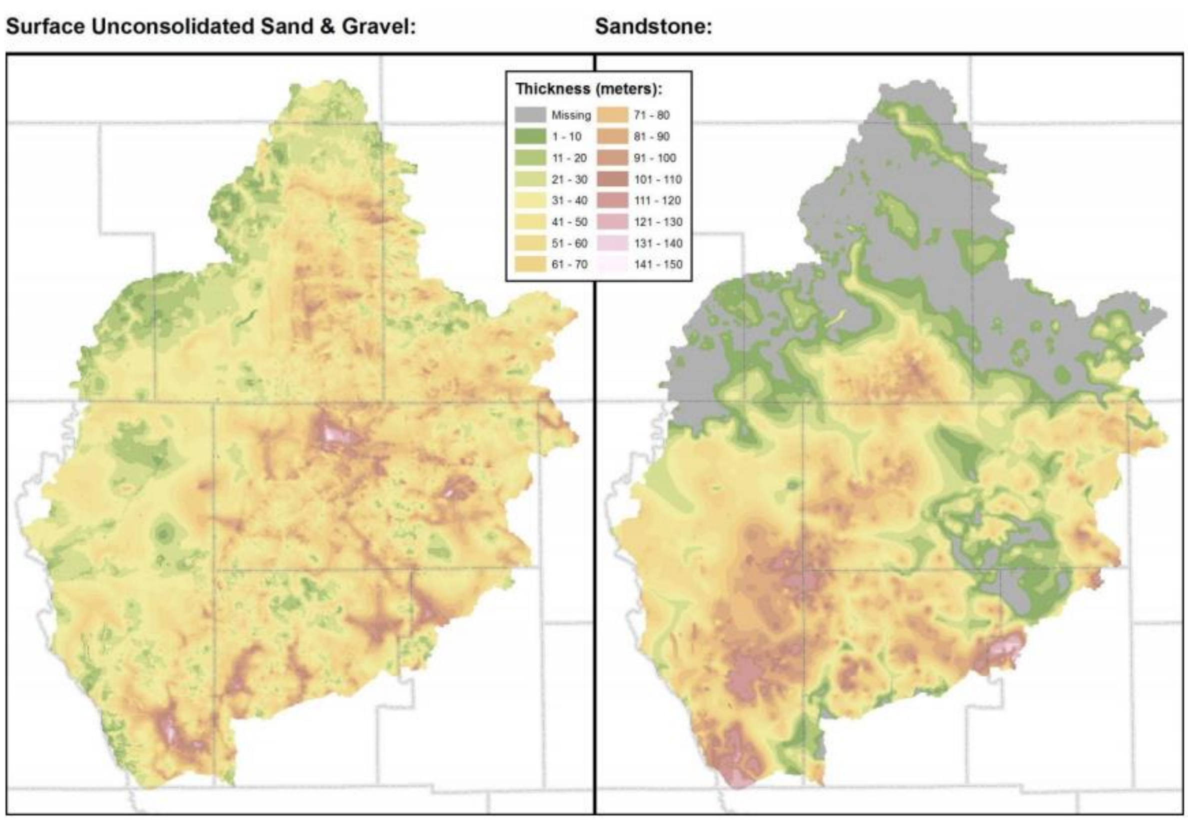

Figure 12. Thickness of the unconsolidated and sandstone layers throughout the Central Sands region (Mechenich et al., 2009).

Over the last century, hydrogeologists have estimated hydrogeologic parameters of the Central Sands surficial aquifer. The three most important parameters related to groundwater pumping and drawdown are hydraulic conductivity (the measure of a material's ability to transmit water), the aquifer thickness, and the storage coefficient, or storativity (the volume of water released from storage per unit decline in head in the aquifer per unit area of the aquifer) (Devaul and Green, 1971; Holt, 1965; Summers, 1965; Weeks et al., 1965; Weeks and Stangland, 1971).The ability of the aquifer to supply water to wells is given by transmissivity, which is simply the hydraulic conductivity multiplied by the aquifer thickness. Likewise, the ability of the aquifer to store or release water is given by the storage coefficient, which is the specific storage multiplied by the aquifer thickness.

The hydraulic conductivity and transmissivity of the Central Sands aquifer are generally very high and have been measured through pumping tests (Holt, 1965; Karnauskas, n.d.; Rothschild et al., 1982; Weeks and Stangland, 1971; Weeks, 1969, 1964), specific capacity tests (Bradbury and Rothschild, 1985), slug tests (Allen et al., 1998; Bradbury et al., 1992), permeameter tests (Bradbury et al., 1992; Stoertz, 1985) and grain-size analyses (Brownell, 1986). Stoertz and Bradbury (1989) determined that the geometric mean of hydraulic conductivity reported from these studies is between 19 and 87 m/day (62 to 285 ft/day) (Table 2). Kraft et al. (2012) found that an average hydraulic conductivity ranging between of 22 to 28 m/day (74 to 91 ft/day) was suited to calibrate a regional groundwater flow model (Figure 13). Studies showed that the ratio of horizontal to vertical hydraulic conductivity ranged between from 1:1 and 20:1 (Weeks and Stangland, 1971) (Table 3). Additionally, Weeks and Stangland (1971) reported transmissivities in the range of 300,500 to 1,700,000 liters/day (80,000 to 450,000 gal/day) based on multi-well pumping tests (Table 2).

Table 2. Central Sands hydraulic characteristics of aquifer, stream and lake sediments along with the location, type of test and publication source and year.

Horizontal Hydraulic Conductivity m/day (ft/day)	**Transmissivity** m^2/day (GPD/ft)	**Location/Type**	**Test Type**	**Author**
75 (247)	NA	Buena Vista basin	Aquifer pumping test (geometric mean)	Stoertz (1989)
63 (207)	NA	Buena Vista basin	Aquifer pumping test (geometric mean)	Bradbury and Rothschild (1985); Holt (1965); Weeks and Stangland (1971); Karnauskas (1977)
87 (285)	NA	Near Hancock	Aquifer pumping test	Manser (1983)
77 (251)	3,353 (270,000)	Sec15, T21N, R9E	Aquifer pumping test	Holt (1965)
90 (294)	4,098 (330,000)	Sec9, T21N, R9E	Aquifer pumping test	Holt (1965)
69 (227)	1,242 (100,000)	Sec34, T24N, R8E	Aquifer pumping test	Holt (1965)
71 (233)	1,739 (140,000)	Sec18, T23N, R9E	Aquifer pumping test	Weeks et al. (1965)
57 (187)	845 (68,000)	Sec35, T22N, R6E	Aquifer pumping test	Karnauskas (1977)
75 (246)	813 (65,500)	Sec26, T22N, R6E	Aquifer pumping test	Karnauskas (1977)
63 (207)	907 (73,000)	Sec34, T22N, R6E	Aquifer pumping test	Karnauskas (1977)
56 (183)	1,043	Sec36, T22N, R6E	Aquifer pumping test	Karnauskas (1977)

	(84,000)			
40 (130)	561 (45,200)	Sec15, T19N, R8E	Aquifer pumping test	Kimball (1982)
87 (285)	1,987 (160,000)	Sec22, T21N, R8E	Aquifer pumping test	Rothschild et al. (1982)
63 (207)	2,037 (164,000)	Sec11, T19N, R8E	Aquifer pumping test	Weeks (1969)
40 (130)	800 (99,400)	Sec22, T21N, R8E	Aquifer pumping test	Weeks (1969)
55 (181)	NA	Buena Vista basin	Specific capacity (geometric mean)	Bradbury and Rothschild (1985) Rothschild (1982)
19 (62)	NA	Buena Vista basin	Piezometer, slug test (geometric mean)	Bradbury et al. (1992)
54 (177)	1,630 (131,260)	Near Hancock	Specific capacity (average of 15 tests)	Rothchild (1982)

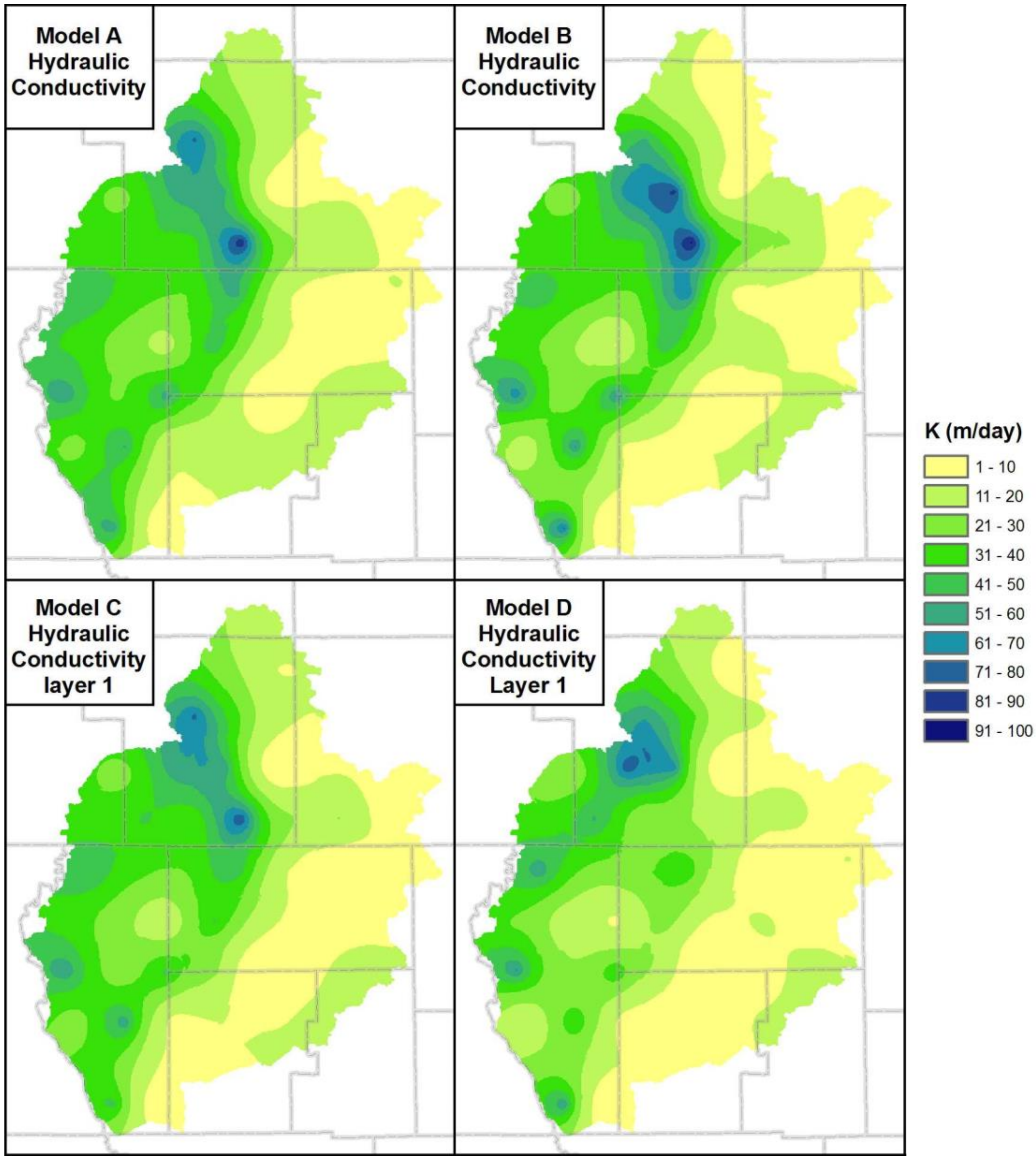

Figure 13. Four model estimates of hydraulic conductivity throughout the Central Sands region as estimated using a steady-state groundwater flow model constructed in MODFLOW (Kraft et al., 2010).

Table 3. Central Sands aquifer characteristics.

	Quantity	Location/Type	Author
Aquifer thickness	30 m (100 ft), average Up to 75 m (250 ft)	Region	Devaul and Green (1971)
Anisotropy	1:1 to 20:1	Region	Weeks (1964, 1968); Weeks and Stangland (1971)
Storage coefficient (storativity)	0.14-0.20	Region	Weeks and Stangland (1971)
Specific yield	0.1 - 0.3	Region	Holt (1965); Weeks (1965); Weeks and Stangland (1971)
Porosity	0.30	Outwash	Summers (1965)
Depth to groundwater	3-20 m (1-60 ft)	Region	Lippelt and Hennings (1981)
Regional horizontal gradient	0.0003-0.005	Regional average	Lippelt and Hennings (1981)
Field capacity	12-14%	Plainfield sand	Haucke (2010)
Linear horizontal groundwater velocity	12-40 cm/day (4.7-16 in/day)	Buena Vista basin	Bradbury et al. (1992)
Residence time	< 28 years	Southern Portage County (Pickerel Lake)	Hennings (1978)

The Central Sands aquifer contains high storage coefficients due to the unconfined nature of the aquifer (Weeks and Stangland, 1971). In unconfined aquifer settings, the storage coefficient is very nearly equivalent to the specific yield ($S = S_y + bS_s$ where S is the storage coefficient, S_y is the specific yield, b is the aquifer thickness and S_s is the specific storage). The specific yield is "the ratio of the volume of water that drains from a saturated rock owing to the attraction of gravity to the total volume of the rock" (Meinzer 1923) and is several orders of magnitude greater than bS_s. The specific yield in the unconfined Central Sands aquifer ranges from 0.1 to 0.3, roughly equal to the storage coefficient for the aquifer (Holt, 1965; Weeks et al., 1965; Weeks and Stangland, 1971). The storage coefficient for the Central Sand unconfined aquifer is orders of magnitude larger than the typical storage coefficient in confined aquifers, which ranges from 0.00001 to 0.001. For this reason, the drawdown after a given period of groundwater pumping in unconfined aquifers is far less than the drawdown for an equivalent period of pumping and discharge at a well in a confined aquifer with similar transmissivity.

The water table in the Central Sands aquifer is the top of the saturated zone and forms the upper boundary. Typically 3 to 20 m (10 to 66 ft) below the land surface, the water table generally slopes eastward and westward from the regional groundwater divide located near the regional topographic divide to the major rivers in the area at a horizontal gradient of about 0.0003-0.005 (Table 3, Figure 14) (Kraft et al., 2010; Lippelt and Hennings, 1981). Throughout the year, the water table naturally fluctuates in response to weather variables. The water table rises as precipitation infiltrates the soil column and reaches the saturated zone. Though precipitation peaks in the summer months with 60% of total precipitation occurring during the frost-free growing season from May 15 to September 15 (Holt, 1965), the water table is highest following spring recharge when precipitation and snowmelt exceed evapotranspiration. In May and June, the water table begins to decline after evapotranspiration exceeds precipitation due to decreased recharge and pumping (Weeks et al., 1965). Groundwater levels continue to decline throughout fall and winter months, with an occasional small recharge event in the fall. Levels often reach an annual minimum in February or March (Holt, 1965).

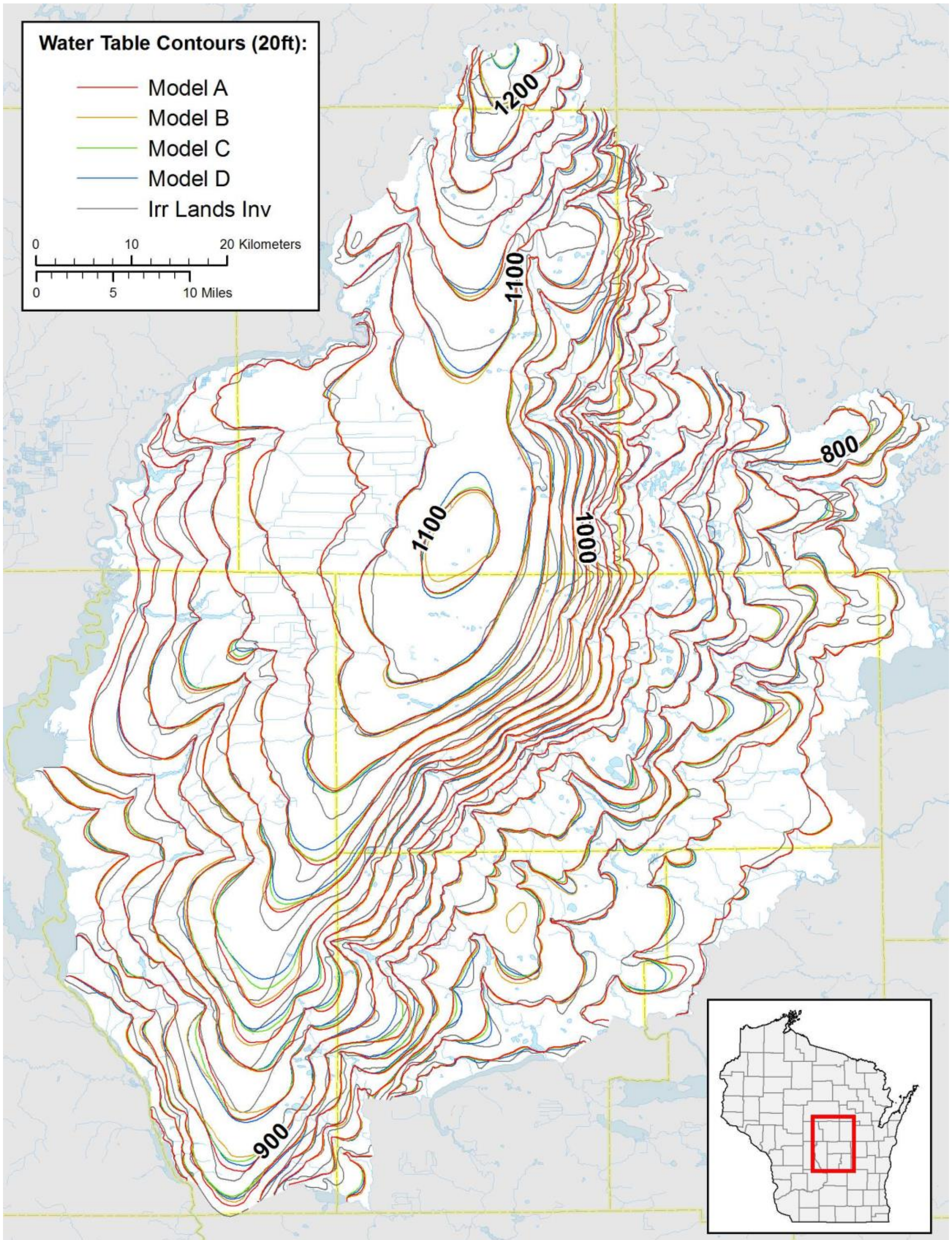

Figure 14. Groundwater table map showing equipotentials or lines of equal hydraulic head, for four groundwater flow model versions along with those described by Lippelt and Hennings (1981) (Kraft et al., 2010).

I.3 Groundwater Flow

Groundwater flows from higher elevation areas, such as moraines, to lower-elevation areas of discharge, such as lakes, rivers, and streams (Bradbury et al., 1992). Groundwater movement is generally perpendicular to equipotential lines, or lines of equal water-table elevation. Based on the regional water-table map of the Central Sands, groundwater moves from the regional groundwater divide generally eastward towards the Fox-Wolf River systems or westward to the Wisconsin River (Figure 14). This regional groundwater movement comprises a groundwater flow system. Locally, regional flow paths are interrupted by discharge to shallow features such as drainage ditches, lakes, wetlands, and pumping wells.

Due to the high hydraulic conductivity soils in the Central Sands, most land surface areas infiltrate water readily and recharge the groundwater if infiltrated water is not evapotranspired. Water that reaches the water table moves along flow paths of various lengths. Shorter flow paths to discharge areas are known as local flow paths, while longer flow paths to discharge areas are called intermediate or regional flow paths (Figure 15) (Winter et al., 1998). Water traveling further distances tends to reach greater depths (Faustini, 1985).

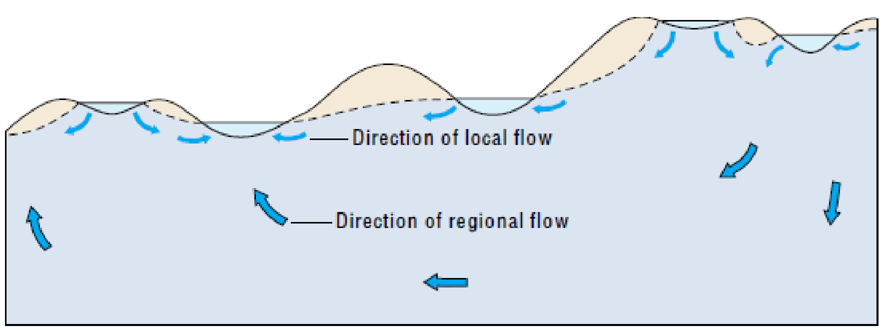

Figure 15. Hypothetical local and regional groundwater flow paths (Winter et al., 1998).

Groundwater flow can be quantified in several ways. Specific discharge is the groundwater flow volume per unit area; it is also called the Darcy flux. The average linear groundwater velocity is the specific discharge divided by the effective porosity.[4] It accounts for the tortuosity of flow paths and hence represents the actual velocity of water traveling through the sediment pore space. Groundwater residence time is the average amount of time that a given particle spends in a groundwater flow system from the point of recharge to the point of discharge. As described by Darcy's law ($q = -k\frac{dh}{dl}$), specific discharge, groundwater velocity, and residence time are dependent on the hydraulic conductivity (k) and the hydraulic gradient ($\frac{dh}{dl}$), the difference in hydraulic head between two points divided by the distance between the two points, of a groundwater system.

Groundwater tracer and other tests in the Buena Vista Basin found the average linear horizontal groundwater velocity to range between 12 to 40 cm/day (4.7 and 16 in/day), depending on local hydraulic gradients (Table 3) (Bradbury et al., 1992). Along with hydraulic conductivity and the hydraulic gradient, residence time depends on the length of a groundwater flow path. Hennings (1978) estimated that the residence time of water discharging into Pickerel Lake in southern Portage County was less than 28 years and flowed at a rate of 30 cm/day (12 in/day) (Table 3).

I.4 Groundwater Recharge

Groundwater recharge is the movement of water into groundwater storage. The course-grained, sandy soils throughout the Central Sands make it possible for recharge to occur almost anywhere. However, rates of recharge vary across the landscape due to the topography, seasonal

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[4] Effective porosity is defined as the volume of interconnected pore space per total sample volume.

weather patterns, vegetation, land use, depth to groundwater below land surface and other factors (Kraft et al., 2010; Stoertz, 1985; Weeks and Stangland, 1971; Weisenberger, 2009). Recharge can result from infiltration of precipitation or overland flow, seepage from surface water bodies, artificial injection, and irrigation (Lin, 2002). Discharge mechanisms include evapotranspiration, pumping, discharge to springs, and seepage to surface water (Lin, 2002; Weeks and Stangland, 1971). For further description of recharge and discharge mechanisms and interactions, see Winter (1999).

Average recharge rates have been estimated at multiple time scales in the Central Sands for a variety of purposes (Table 4). Average annual recharge rates have been estimated for regional basins using water budget modeling (Table 5) (Weeks and Stangland, 1971), water balance modeling (Weeks and Stangland, 1971; Faustini, 1985), stream gage data (Figure 16) (Gebert et al., 2011), groundwater flow modeling (Kraft et al., 2010), and inverse modeling (Stoertz, 1985). Regional average annual recharge values were found to range from approximately 10 to 40 cm/yr (4 to 16 in/yr) (Table 6). A study conducted at five farmland sites in Adams County found that during precipitation events short-term recharge rates were higher beneath prairies compared to irrigated agricultural fields and rarely occurred beneath pine forests (Weisenberger, 2009). Additionally, Weisenberger (2009) found that residue on the soil surface protected soil against extreme freezing temperatures and when present resulted in increased recharge to the aquifer during winter months.

Table 4. Recharge relevance at different temporal and spatial scales (Stoertz 1989). Hydrologic topics are given for each combination of time unit and spatial scale, along with related questions.

Time Unit	**Recharge Relevance at Different Spatial Scales**	
	Local (field or site)	Regional (groundwater basin)
1 hour	Contaminant transport: (Will residence time in unsaturated zone permit attenuation of contaminant?)	Flooding: (What is the basin response to the 100-year storm?)
1 month	Timing of application: (What are seasonal variations in recharge?)	Seasonal groundwater availability: (What are seasonal storage changes?)
1 year	Delineation of recharge and discharge areas: (Is the site in a recharge area?)	Water supply: (Is the aquifer being mined?)

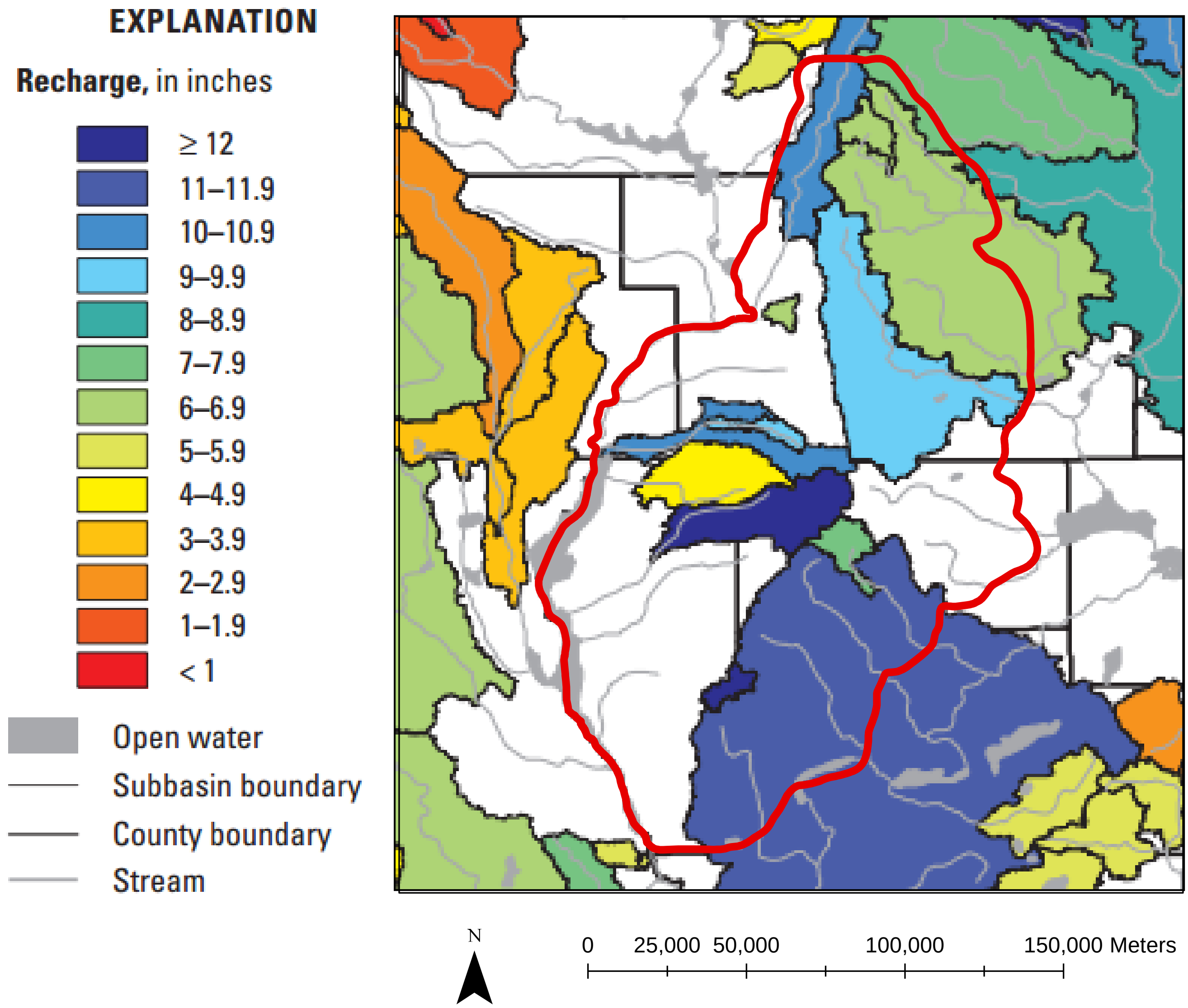

Figure 16. Map of Central Sands annual groundwater recharge (adapted from Gebert 2009-2010).

Table 5. Average annual evapotranspiration and recharge, in inches, for the period 1948-67, as computed for four basins in the Central Sands by the water-balance method (Weeks and Stangland 1971).

Year	Ditch 5 of Tenmile Creek		Big Roche a Cri Creek		Tenmile Creek		Fourteenmile Creek near New Rome	
	ET	Recharge	ET	Recharge	ET	Recharge	ET	Recharge
1948- -	13.3	7.5	14.0	6.8	14.6	6.2	- - -	- - - -
194 - -	17.6	8.0	18.3	7.4	19.0	6.7	- - -	- - - -
1950- -	16.4	11.9	16.9	11.4	17.4	10.9	- - -	- - - -
1951- -	15.8	18.9	16.3	18.4	16.9	17.8	- - -	- - - -
1952- -	15.1	8.7	15.6	8.2	16.3	7.5	- - -	- - - -
1953- -	14.1	11.9	14.8	11.2	15.3	10.7	- - -	- - - -
1954- -	17.3	19.8	17.9	19.2	18.5	18.6	- - -	- - - -
1955- -	16.8	8.8	17.4	8.2	18.1	7.6	- - -	- - - -
1956- -	17.3	11.7	17.8	11.2	18.2	10.8	- - -	- - - -
1957- -	16.1	9.5	16.7	8.9	17.3	8.3	- - -	- - - -
1958- -	14.5	3.4	15.1	2.9	15.6	2.5	- - -	- - - -
1959- -	17.1	22.3	17.7	21.7	18.4	20.9	- - -	- - - -
1960- -	15.5	15.0	17.2	17.0	17.7	16.5	18.2	16.0
1961- -	16.5	13.6	17.4	17.3	18.0	16.7	18.4	16.3
1962- -	18.8	11.2	19.3	8.4	19.8	8.0	20.1	7.7
1963- -	17.1	11.2	18.5	9.7	19.2	9.0	19.6	8.6
1964- -	17.3	6.1	18.0	5.0	18.5	4.5	18.9	4.1
1965- -	18.2	19.7	18.5	19.7	19.0	19.3	19.3	19.0
1966- -	15.4	8.6	16.5	8.2	16.6	8.3	17.4	7.6
1967 1/	12.9	8.7	13.4	8.2	13.7	8.0	14.3	7.2
Average for period of record	16.3	12.0	17.0	11.6	17.6	11.1	18.8	10.6

1/ For first nine months only.

Table 6. Average annual recharge in basins throughout the Central Sands region.

Average Annual Recharge cm/yr (in/yr)	Location	Method	Author	Year Published
17 (7)	Throughout region	Water balance	Stoertz and Bradbury	1985
30.48 (12.0)	Ditch 5 of Tenmile Creek	Water balance	Weeks and Stangland	1971
29.46 (11.6)	Big Roche a Cri Creek	Water balance	Weeks and Stangland	1971
28.19 (11.1)	Tenmile Creek	Water balance	Weeks and Stangland	1971
26.92 (10.6)	Fourteen-mile Creek near New Rome	Water balance	Weeks and Stangland	1971
10.16 to 40.64 (4.1-16.0)	Throughout region	Steady state groundwater flow model calibration	Kraft and Mechenich	2010
10.16 to >30.48 (4 to >12)	Throughout region	Streamflow data measurements	Gebert et al.	2009-2010
13 to >38 (5 to >15)	Buena Vista basin	Computer-aided mapping technique based on water balance	Bradbury et al.	1992

Groundwater – Surface Water Relationships

Groundwater within the first few meters of the land surface supports lakes, streams and wetlands that extend throughout the Central Sands landscape (Weeks et al., 1965). According to the WDNR (2013), the region includes 1300 km (800 mi) of trout streams and over 300 lakes, which are important recreational and ecological resources (Figure 17) (Smail, personal communication). Wetlands extend throughout mid-elevation areas, though they are less widespread today due to drainage ditch construction.

Landscape position influences the type of water body present in a watershed (Figure 18). Most high-elevation water bodies in the Central Sands are small kettle lakes that are fed by precipitation and groundwater inflow (Hennings, 1978). Central Sands water bodies in mid- to low-elevation landscapes are more often groundwater-fed streams or wetlands and act as areas for groundwater discharge (Barlow and Leake, 2012). Each of these water bodies is associated with a surface watershed (Figure 19).

Lakes have three primary types of interactions with the surrounding groundwater system, which can vary seasonally. Lakes can receive groundwater inflow throughout their entire lakebed, they can lose water to the groundwater system throughout their entire lakebed or, more commonly, they can receive groundwater inflow in parts of the lakebed and lose water to seepage in other parts of the lakebed (Figure 20) (Winter et al., 1998). Most Central Sands lakes are seepage (groundwater flow-through) lakes that receive water from and lose water to the groundwater system (Hennings, 1978).

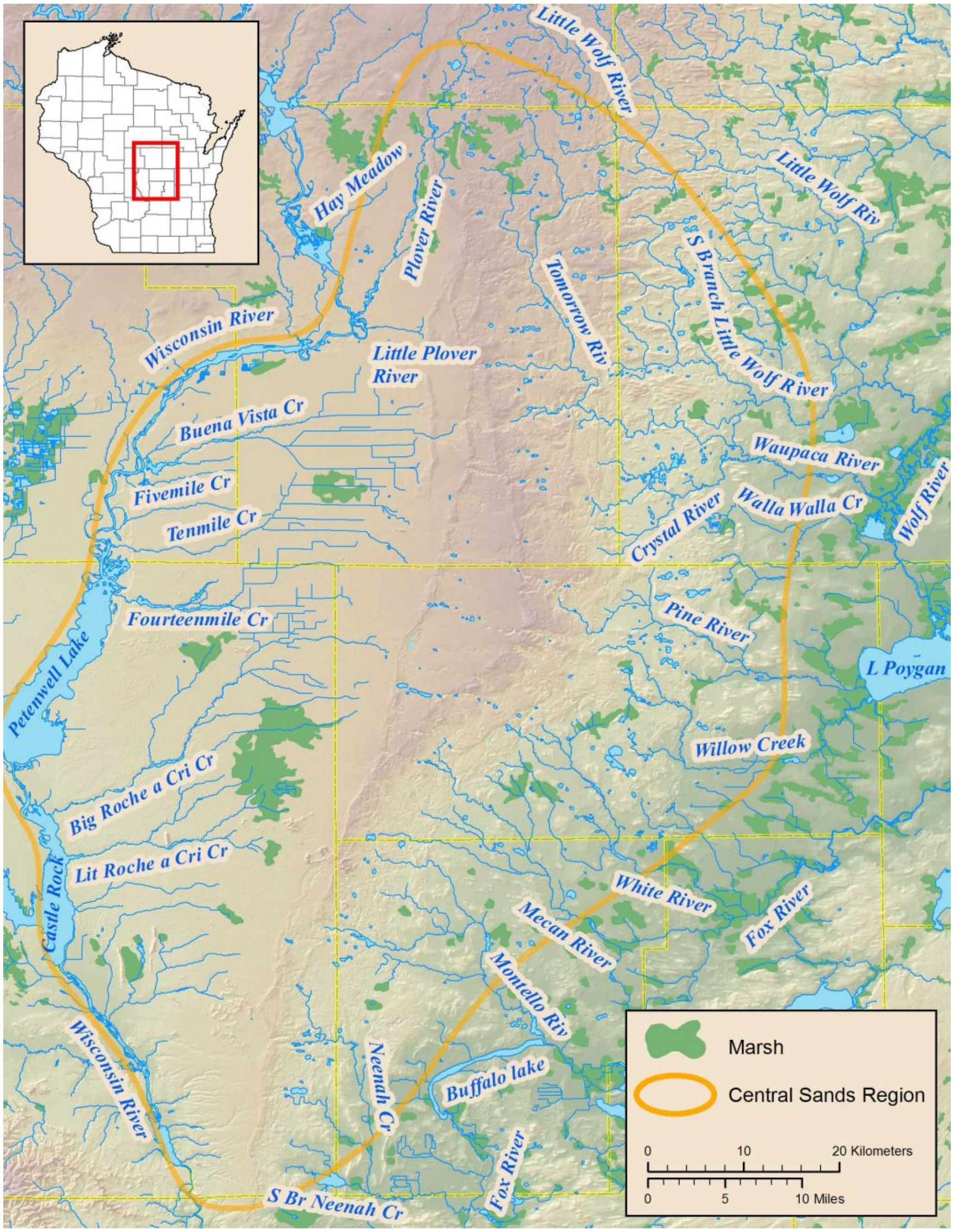

Figure 17. Central Sands water resources and topography (Kraft et al., 2010).

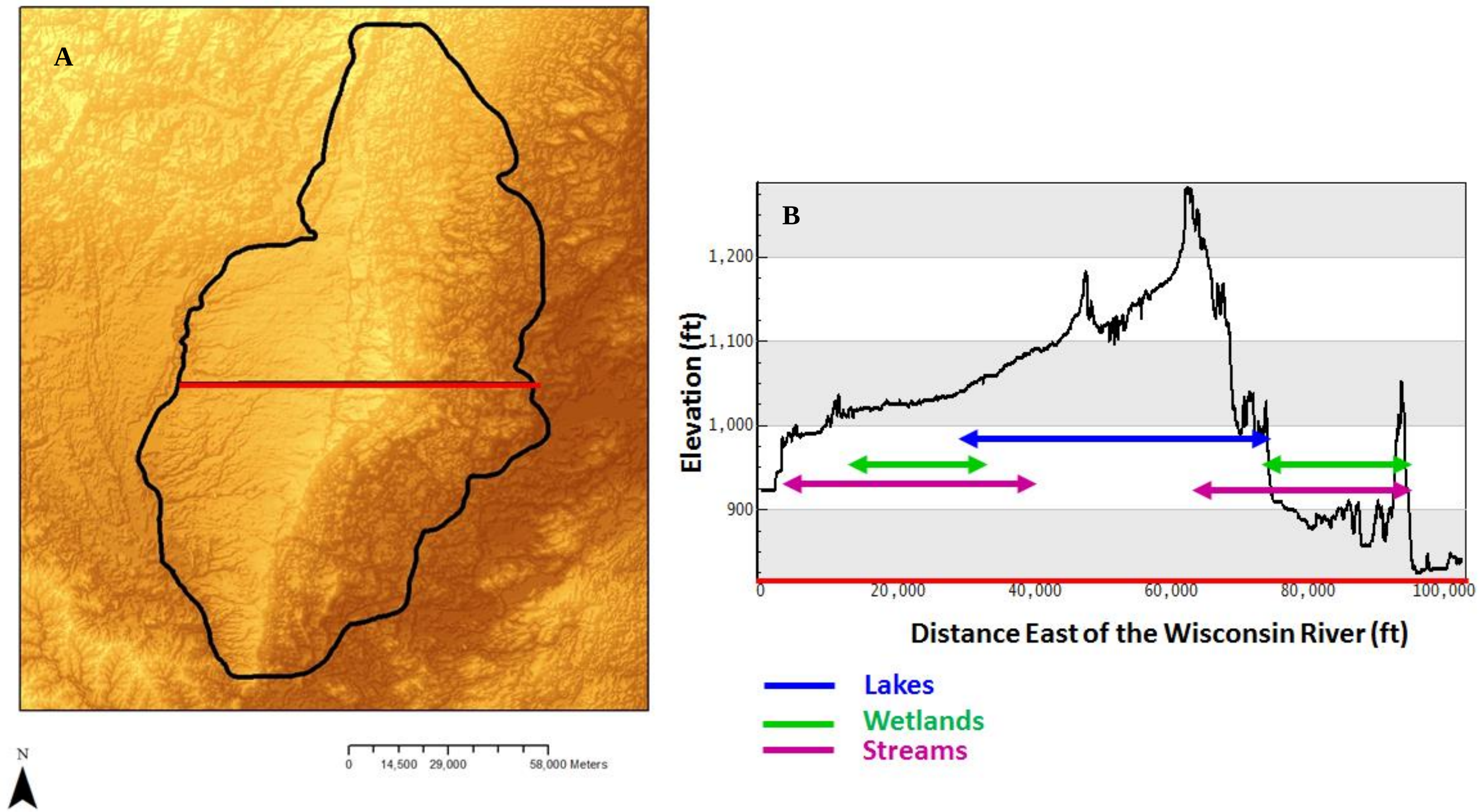

Figure 18. *A:* Digital elevation model illustrating the topography throughout the region. Red line indicates location of topographic profile. *B:* Relation of groundwater-dependent surface water bodies to topographic profile (red line). Lakes tend to be located within tunnel channels in high-elevation areas (blue). Wetlands tend to be located in mid- to low-elevation areas (green) with a low topographic gradient. Streams tend to be located in mid- (headwaters) to low-elevation areas. The map and profile are based on data available through the WDNR website (2013).

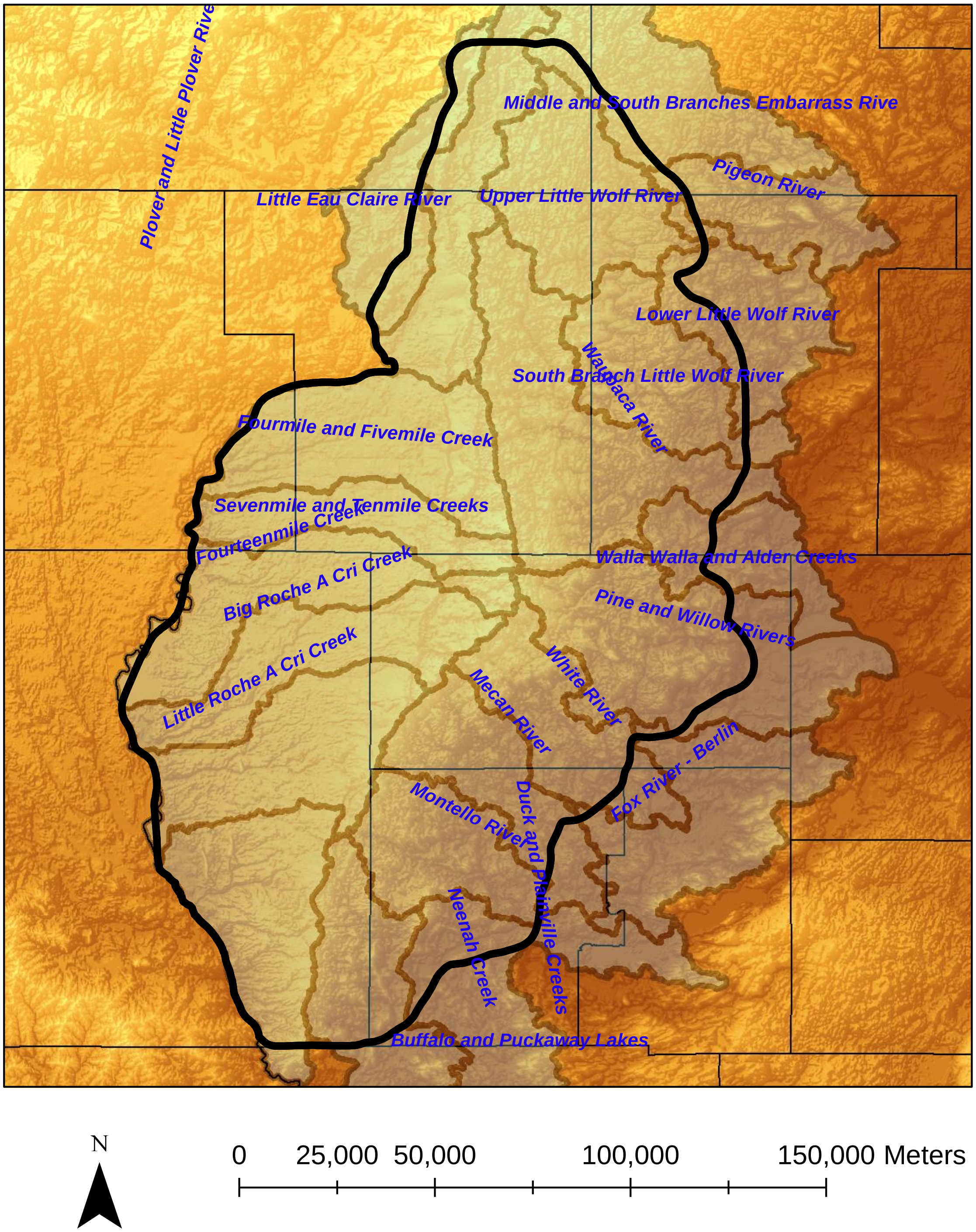

Figure 19. Surface watersheds and topography throughout the Central Sands region. Surface watershed names are in blue. The map is based on data available throughout the WDNR website (2013).

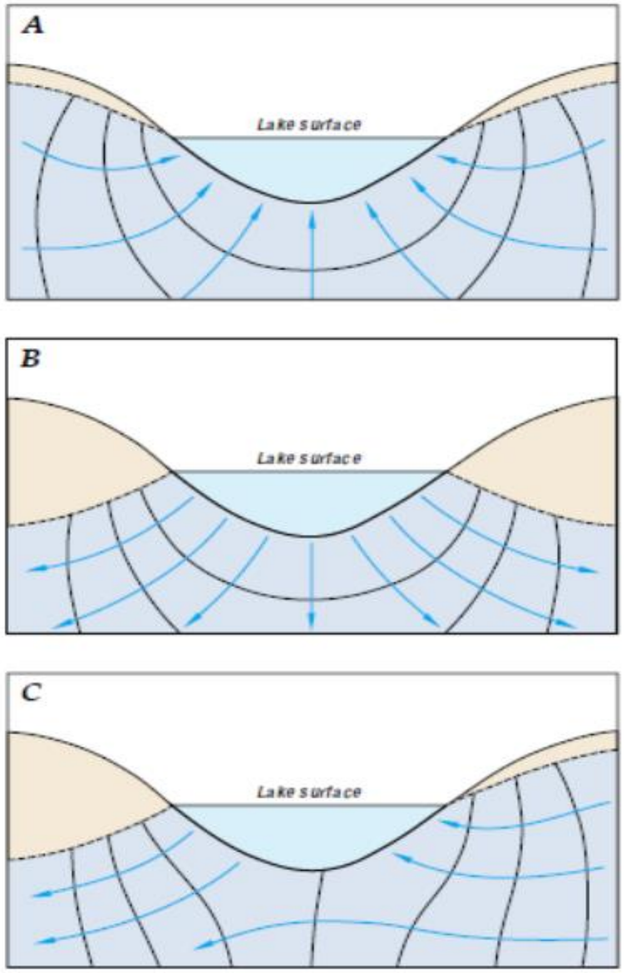

Figure 20. Lakes can receive groundwater inflow (*A*), lose water as seepage (*B*) or both (*C*) (Winter et al., 1998).

Lakes throughout the region are often situated in former tunnel channels (for a definition of tunnel channels, see the section titled “A Brief Geologic History of the Region”) that contain several tens of meters of sand and gravel in the adjacent area, as well as in the channel, suggesting that sands and gravels extend below the bottom of the deepest pits (Attig et al., 1989), Besides sands and gravels, Wisconsin Geological and Natural History Survey well-construction reports have documented intermittent fine sediments within tunnel channels. For example, the Plainfield-Huron tunnel channel breaching the Hancock moraine (Figure 11) contains several seepage lakes (e.g. Plainfield Lake, Long Lake, Lake Huron) with intermittent fine sediment within the bed of the tunnel channel (WGNHS well construction reports). The fine sediments complicate surface water and groundwater connections and can make it more challenging to quantify the relative impacts of weather-driven fluctuations and land use factors, such as irrigated agriculture, on lake levels.

Lake levels fluctuate in response to seasonal, annual and longer timescale weather patterns and the dynamics of water use (Butler, 1978; Winter et al., 1998). The frequency and magnitude of lake level fluctuations depends on the amount and timing of inflows and outflows. Central Sands lakes located in upland areas near the regional groundwater divide that separates the Wisconsin River Basin from the Fox-Wolf River Basin generally lack connections to surface water systems and tend to experience larger magnitude fluctuations than seepage lakes with stream connections (Novitzki and Devaul, 1978).

House (1985) conducted an inventory of fluctuating lake levels throughout the state of Wisconsin. Twelve of the lakes in the study were located in the Central Sands (Table 7). Average annual water level fluctuations ranged between 0.61 and 0.54 m (0.2 and 1.78 ft) for the 12 lakes included in the study. However, the data reported was limited to 1 to 5 years. Quantification of climatic-driven variations in magnitude, frequency and duration of Central Sands lake level fluctuations is important for understanding how land use-land cover change has affected surface water ecosystems.

Similar to lake systems, streams can receive groundwater inflows, lose water through seepage to the aquifer or vary by reach in their relationship to the aquifer depending on the landscape position, change in topographic slope, depth to groundwater and other factors (Winter et al., 1998). The elevation of the groundwater around a stream must be higher than the stream stage for a stream reach to be "gaining" and vice versa for it to be "losing" (Winter et al., 1998). Holt (1965) estimated that for a portion of Portage County south of Stevens Point and west of the Hancock moraine, streamflow is derived entirely from groundwater discharge 95% of the time. Of the cool, groundwater-fed streams in the Central Sands, almost 90% (885 km or 550 mi) are considered Class 1 trout streams that contain sufficient natural reproduction of wild trout to sustain populations at or near carrying capacity (WDNR, 2002).

Numerous springs occur in the Central Sands region, often times in streambeds running along the base of the moraine-formed hills (Figure 21) (Macholl 2007). In general, springs occur where the land surface intersects the water table. At these locations, cool, oxygen-rich groundwater flows to the land surface. Spring flow can be useful water collection sites when investigating biogeochemistry in an aquifer system since they provide a "window" to the aquifer's groundwater resource. Spatial distribution of springs used in association with regional datasets of geochemistry, topography and geology can reveal important controls on groundwater flow and make initial assessments of the vulnerability of spring flow to groundwater withdrawal.

Table 7. Lake names, location, period of record, relationship to groundwater, maximum depth, average annual water-level fluctuations and maximum annual water level fluctuations as reported in House (1985).

Lake Name	County	Near	Period of Record	Hydrologic Class	Max depth (ft)	Average annual water-level fluctuations	Maximum annual water level fluctuations
Bing	Waushara	Coloma	1978-1979	GWF	31	0.76	1.12
Fish	Waushara	Hancock	1971-1972, 1978-1979	GWF	42	0.88	2.2
Fish	Waushara	Wautoma	1966-1977, 1979	SWD	5	0.68	1.14
Long	Waushara	Plainfield	1977-1979	GWF	6	1.26	1.52
Mecan springs	Waushara	Hancock	1978-1979	SWD	21	0.75	1.05
Plainfield	Waushara	Plainfield	1978-1979	GWF	5	1.78	2.63
Huron	Waushara	Plainfield	1978-1979	GWF	37	1.55	2.73
Adams Lake	Portage	Amherst	1978-1979	SWD	51	0.2	0.23
Bear Lake	Portage	Amherst	1965, 1970, 1974, 1978, 1979	GWF	28	1.42	-
Lake Emily	Portage	Amherst Junction	1977-1979	GWF	36	1.08	1.69
Pickerel	Portage	Blaine	1976-1979	GWF	16	0.88	1.5
Ennis Lake	Marquette	Endeavor	1978-1979	SWD	30	0.24	-

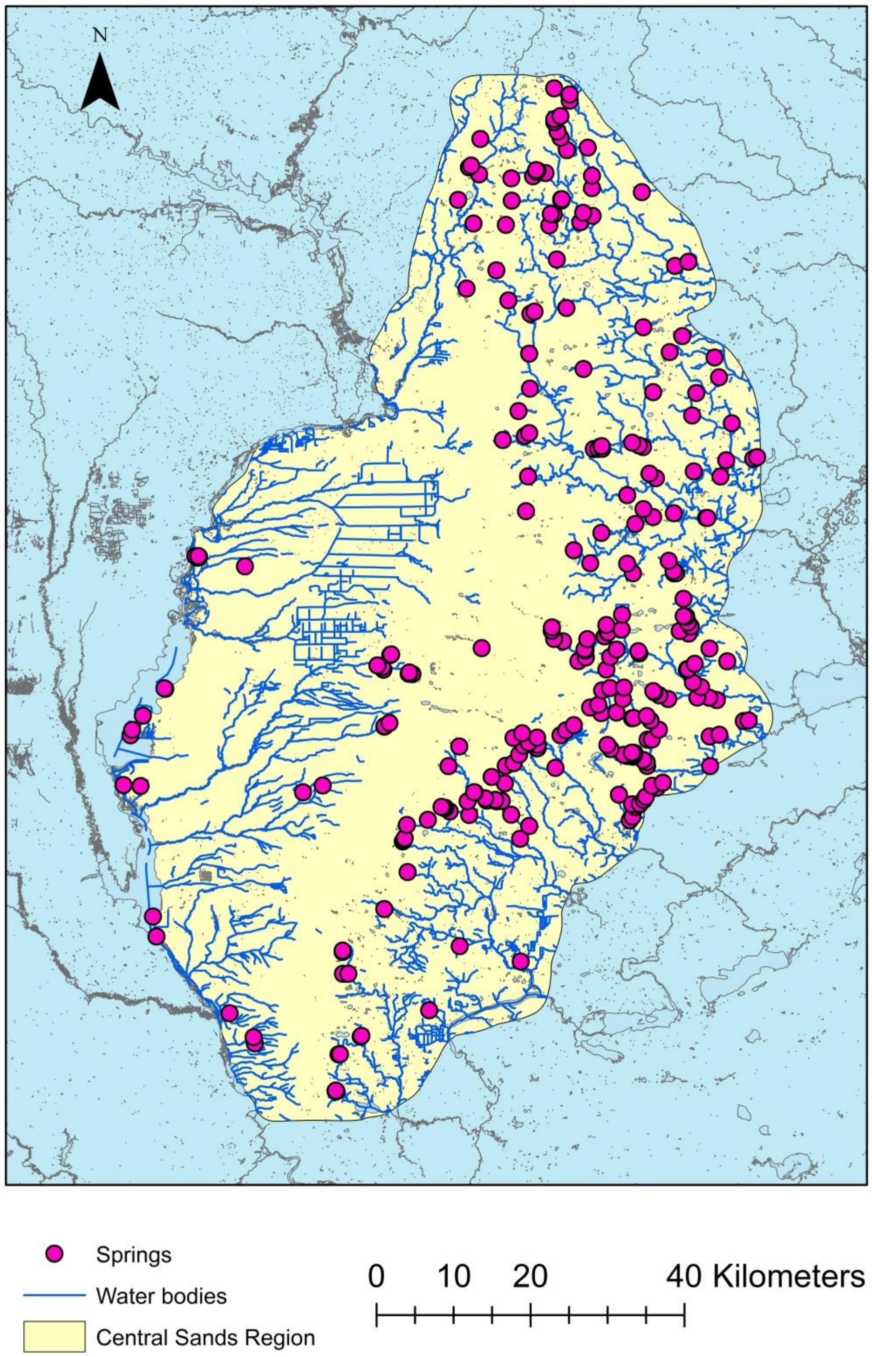

Figure 21. Springs located throughout the Central Sands (Macholl, 2007).

General Effects of Groundwater Pumping

The general effects of groundwater pumping on hydrology, hydrogeology and aquatic ecosystems in glaciated systems are well understood. Hydrogeologists routinely use mathematical equations and computer simulations to predict how withdrawing water from a well will affect nearby water levels and how far these effects will extend. Besides groundwater levels, groundwater pumping affects the hydrologic budget, groundwater basin boundaries and surface water levels and flows by shifting one or more of four sources from which it is derived. This section outlines how groundwater pumping affects hydrogeology and groundwater flow-dependent ecosystems. In particular, the section 1) provides a statement about and an example of methods for assessing impacts of groundwater pumping on groundwater flow systems; 2) describes the sources of pumped groundwater and the short and long-term effects of groundwater pumping on the groundwater system; 3) discusses the dependence of flora and fauna on annual water level fluctuations; and 4) describes how water-level fluctuations affect water temperature, morphometry, sedimentation patterns, light penetration, and invasive species.

I.1 Hydrogeology

Methods for assessing the impacts of high-capacity well pumping on groundwater flow systems in unconfined glacial aquifers are well-established in hydrologic and hydrogeologic literature (Ferris et al., 1962; Zheng, 1986; Winter *et al.*, 1998). The spatial and temporal effects of an individual pumping well on a groundwater flow system can be mathematically predicted. For example, Weeks and Stangland (1971) generated a hypothetical distance-drawdown curve from pumping one well in the Central Sands near the Little Plover River. They found that after 30 days of pumping a well at 3,600 liters per minute (950 gal/min), 0.6 m (2 ft) of drawdown occurred at a distance of approximately 550 m (1,800 ft) from the well. Factors affecting the impact of pumping on unconfined surface and groundwater systems include the pumping rate, proximity of a well to the point of interest, precipitation, number of wells, and hydraulic parameters of the aquifer (Weeks et al., 1965; Weeks and Stangland, 1971; Winter et al., 1998). Methods for assessing the impacts of multiple high-capacity wells on groundwater flows have also been well-established with some general assumptions and limitations (Ferris et al., 1962).

Groundwater pumping alters the hydrologic budget, groundwater levels, groundwater basin boundaries and surface water levels and flows (Weeks et al., 1965). Pumped groundwater is derived from one or more of four sources: 1) water that would otherwise leave the basin as underflow, 2) water that would leave the basin as evapotranspiration, 3) water that would leave the basin as stream discharge or 4) water that would remain as storage within the groundwater reservoir (Weeks et al., 1965). Pumped groundwater for irrigated agriculture is lost to evapotranspiration, is retained in the crops, infiltrates through the soil zone to recharge groundwater or returns to a local surface-water body through the drainage system (Weeks and Stangland, 1971; Winter et al., 1998). Pumping groundwater for irrigation causes a temporary decline in the water levels in the specified area and results in decreased discharge to surface water bodies. After several years, groundwater irrigation pumping reduces the total volume of streamflow by the amount that evapotranspiration is increased by irrigation (Weeks et al., 1965). Determining rates of evapotranspiration for different vegetation types is important for determining longer-term changes in recharge and the resulting effects of groundwater pumping on the groundwater system.

Groundwater pumping can also reverse groundwater flow paths by changing the hydraulic gradient in the system. For example, streams or reaches of streams that typically receive groundwater flow (“gaining” streams) can experience reversed flow and buffer groundwater storage ("losing" stream) during times of pumping (Figure 22). Since irrigation is seasonal, recharge during the spring and fall will generally replace much of the groundwater removed from storage. However, a surface water body may experience low flows during the period of "surface water buffering", particularly during times of drought and/or high rates of groundwater pumping. The ability of surface water bodies to buffer aquifer storage emphasizes the close connection of surface and groundwater resources as well as the importance of understanding groundwater pumping on short timescales. The buffering effect illustrates how groundwater levels do not always demonstrate the spatial and temporal redistribution of water as a result of pumping. Measurements of both groundwater levels and streamflows are necessary to determine how an aquifer system is affected by pumping wells.

I.2 Aquatic Ecosystems

As stated above, groundwater withdrawals can reduce water levels in or flow discharging to a surface water body, induce flow from surface water to groundwater (reverse flow), alter the magnitude and timing of water level fluctuations, and/or cause drying up of a water resource (Leira and Cantonati, 2008; WDNR, 1997). These short-term and long-term changes in water levels and flows can shift distribution and abundance of flora and fauna by affecting their life cycles including habits of feeding, movement, and breeding. For example, water level reductions allow woody plant species to encroach on wet meadows and out-compete the vegetation favored by grassland birds and invertebrates (WDNR, 1997). Simultaneously, the reduction in open water area in lakes can reduce breeding habitat for waterfowl (WDNR, 1997).

Besides water levels and flows, groundwater withdrawals have the capacity to warm water temperatures by reducing the contribution of groundwater to surface water systems since groundwater is generally cooler than surface water (WDNR, 1997). Cold-water dependent fisheries in streams are particularly vulnerable to increased water temperatures. For example, trout have the greatest growth rates in temperatures between 7 to 18 ºC (45 to 65ºF) (White and Brynildson, 1967). Warmer water temperatures may result in a change in stream classification, fish death or susceptibility of groundwater to bacterial contamination from surface water (WDNR, 1997; White and Brynildson, 1967). In lakes, temperature regime shifts due to groundwater withdrawals can cause a deepening of the thermocline and a longer stratification period (Leira and Cantonati, 2008).

Groundwater withdrawals can also alter morphometry, sedimentation patterns, light penetration, and presence of invasive species in aquatic ecosystems (Leira and Cantonati, 2008; WDNR, 1997). These characteristics can then affect ecosystem function and distribution. Groundwater withdrawals can enhance sediment erosion by changing littoral (near shore) sediment and biogeochemical characteristics (Furey et al., 2004). Altered water level/flow regimes can cause contaminated sediments to become resuspended (Rhodes and Wiley, 1993). Additionally, water level changes affect light penetration, which can cause a change in the littoral area available for benthic (bottom dwelling) algae and macrophyte growth. This may result in a shift in the distribution of primary production between pelagic (upper water column) and littoral habitats (Leira and Cantonati, 2008). Finally, decreased water levels can encourage invasive, non-native plant species at the expense of native species (Leira and Cantonati, 2008). Each of these hydrologic and biologic changes are difficult to document due to the time lag between hydrologic change and biological effect, the local nature of the effect and other confounding hydrologic factors, such as precipitation (WDNR, 1997).

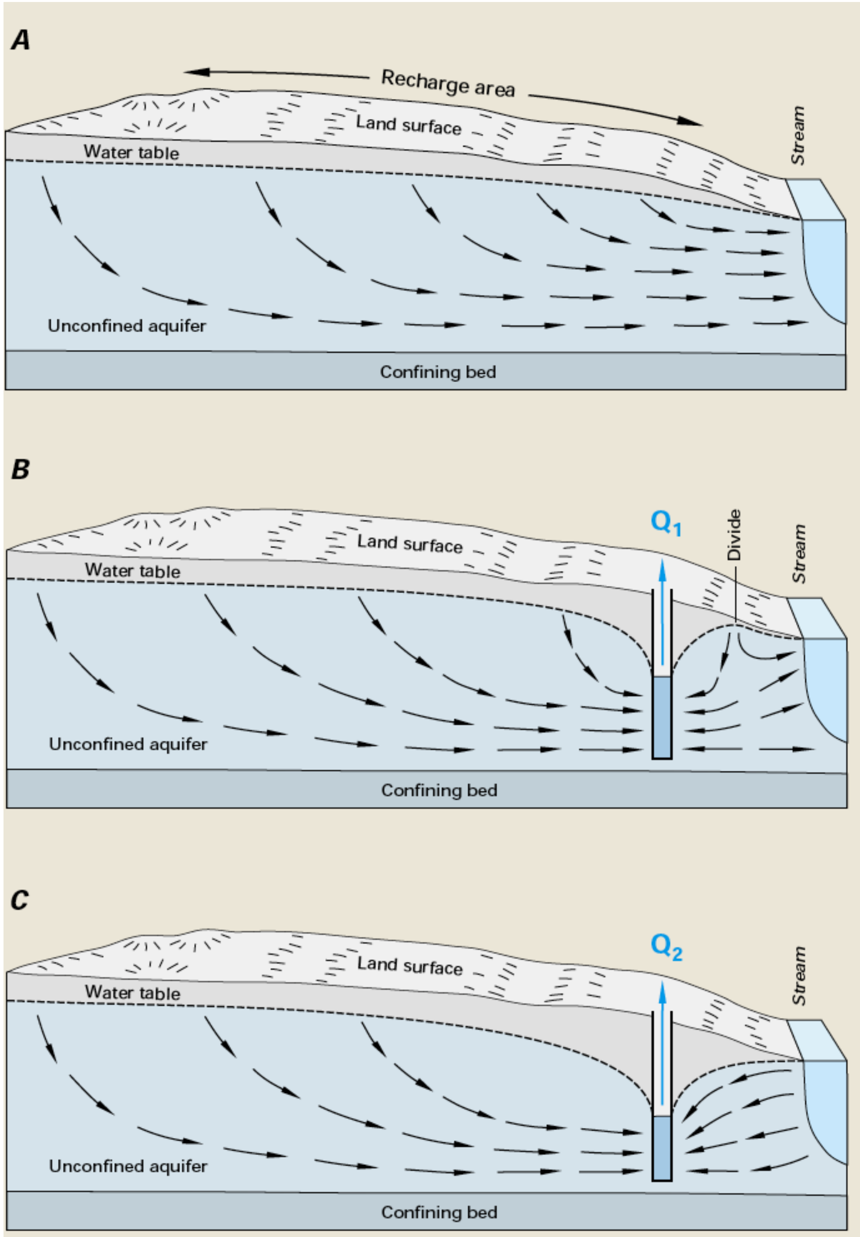

Figure 22. This figure illustrates how groundwater flow is affected by groundwater pumping wells (Winter et al., 1998). *A:* Water recharges the groundwater in higher elevation areas and discharges at lower elevation areas, such as in a stream. *B:* When a pumping well is installed, the well draws water from the surrounding aquifer and shifts groundwater flow patterns, the hydrologic budget, and boundary conditions. *C:* When a pumping well has been pumped for a longer period of time, the well can reverse flows in surface water bodies, such as streams, so that a gaining reach of a stream can become a losing reach.

General Effects of Drainage Ditches

Drainage ditches serve to lower groundwater levels and, thus, affect the hydrogeologic system surrounding the drained area. This section briefly describes how ditches have served to drain wetland areas and can result in shorter groundwater flow paths and altered stream flows. The section also describes how drainage ditches may alter physical and biogeochemical processes in area where drainage ditches are former streams, yet continue to serve unique aquatic ecosystems.

I.3 Hydrogeology

Drainage ditches have served to lower groundwater levels, typically in areas of flat terrain, for the expansion of agricultural and urban development (Butler, 1978). Wetlands, areas where the water table is above the land surface, are common locations for drainage ditch construction that often contain rich, organic soils valuable for agricultural production. The presence of drainage ditches throughout Wisconsin has greatly reduced the total area of land covered by wetlands, while simultaneously expanding agricultural production. Many drainage ditches have been constructed in former streams that are now channelized and deepened. Construction of drainage ditches and/or modification of former stream beds changes the areal distribution of groundwater recharge and discharge (Winter et al., 1998) and often shortens groundwater flow paths. Modification of former stream beds changes the flow rate in the stream/ditch. New drainage ditches installed near streams can reduce baseflow to unmodified streams and can alter the water-holding capacity of topographic depressions (e.g. lakes) by reducing groundwater levels in an aquifer (Winter et al., 1998).

I.4 Aquatic Ecosystems

The influence of drainage ditches on surface water bodies can affect physical, chemical and biological processes (Winter et al., 1998). Drainage ditch construction and channelization affects physical changes in the morphometry, sedimentation patterns, hydraulic characteristics and light penetration and biological changes in aquatic species composition and distribution, soil biological activity and nutrient cycling in new drainage ditches or streams modified into drainage ditches. Still, drainage ditches (new or formerly modified streams) have been shown to provide valuable wet vegetated, non-cropped habitats to aquatic and terrestrial taxa (Herzon and Helenius, 2008; Weeks and Stangland, 1971). Ditches can support common aquatic species, rare aquatic species and aquatic species not found in other farmland habitats due to the supply of food resources that may be lacking in dry, managed cropland (Herzon and Helenius, 2008). Ditches can also perform connectivity functions within a wider landscape (Herzon and Helenius, 2008).

Reported Evidence of Water Resource Impacts

Effects of weather, climate and human-induced changes (e.g. high capacity irrigation) on surface and groundwater levels and flows are difficult to distinguish. Types of models and/or analyses used to investigate the spatial and temporal variation in surface water and groundwater levels and flows include pumping tests, water budget analyses, water balance analyses, statistical models, groundwater flow models, unsaturated flow models and many others. This section begins by reviewing literature that has characterized temporal variability in available Central Sands groundwater level, lake level and streamflow data. The section then reports on changes over time in climate-driven factors that affect water resources, such as temperature and precipitation. The section concludes by summarizing literature that has investigated the impacts of high capacity irrigation well pumping on Central Sands water resources.

I.5 General Observations in Available Water Resource Data

I.5.1 Groundwater Levels

Kraft and Mechenich (2010) determined that there are six active U.S. Geological Survey groundwater monitoring wells in the Central Sands that have sufficient length and completeness to be useful for identifying trends. These wells are Amherst Junction, Wautoma, Hancock, Coloma NW, Bancroft and Plover (Table 8) and have records generally beginning around 1950. (Note that the Plover groundwater monitoring site has had three wells recorded under two different well numbers. For further description about how these datasets were combined, see Kraft et al., 2010.) Groundwater monitoring well records show that groundwater levels experienced lows in 1958 and 1959 (Kraft et al., 2010). Since 1959, average annual groundwater levels generally rebounded and then cyclically fluctuated until the late 1990s when levels again declined. During an average to modest dry period in 2007, wells in low-density irrigated areas at

Amherst Junction and Wautoma declined to their 11 and 14 percentiles, respectively, while other groundwater monitoring wells in high-density irrigated areas (Hancock, Plover, Bancroft, and Coloma) reached record lows (Kraft et al., 2010).

Haucke (2010) presented an analysis of average annual groundwater levels based on the Bivariate Test (Potter, 1981), which tests for a systematic change in mean in a test series compared to a control series. Haucke (2010) compared a series of average annual groundwater levels measured from 1958 through 2008 in wells located in areas having a high-density of irrigation wells (Hancock, Plover, Bancroft, and Coloma) with series of average annual groundwater levels measured over the same period in wells located in areas having a low-density of irrigation wells. Results show a statistically significant decline in average annual groundwater levels in test wells compared to average annual groundwater levels in control wells in the late 1990s, suggesting that the decline is associated with groundwater pumping for irrigation (Figure 23).

Similarly, Kraft et al. (2012) presented estimates of regional groundwater level decline throughout the Central Sands based on comparisons of U.S. Geological Survey observations of groundwater elevations in areas of high and low irrigation well density (Figure 23; Table 9). Results indicated that groundwater monitoring wells in high irrigation well density regions declined more than wells in low irrigation well density areas during the period between 1999 and 2008 (late period) when compared to the period between 1959 and 1968 (early period). The magnitude of the declines in groundwater levels in high irrigation well density areas during the late period relative to wells in the low irrigation well density area ranged between 0 and 0.98 m and started between 1973 and 1990 (Kraft et al., 2012).

Table 8. U.S. Geological Survey groundwater monitoring wells with long-term records (Kraft et al., 2010).

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 Junction	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

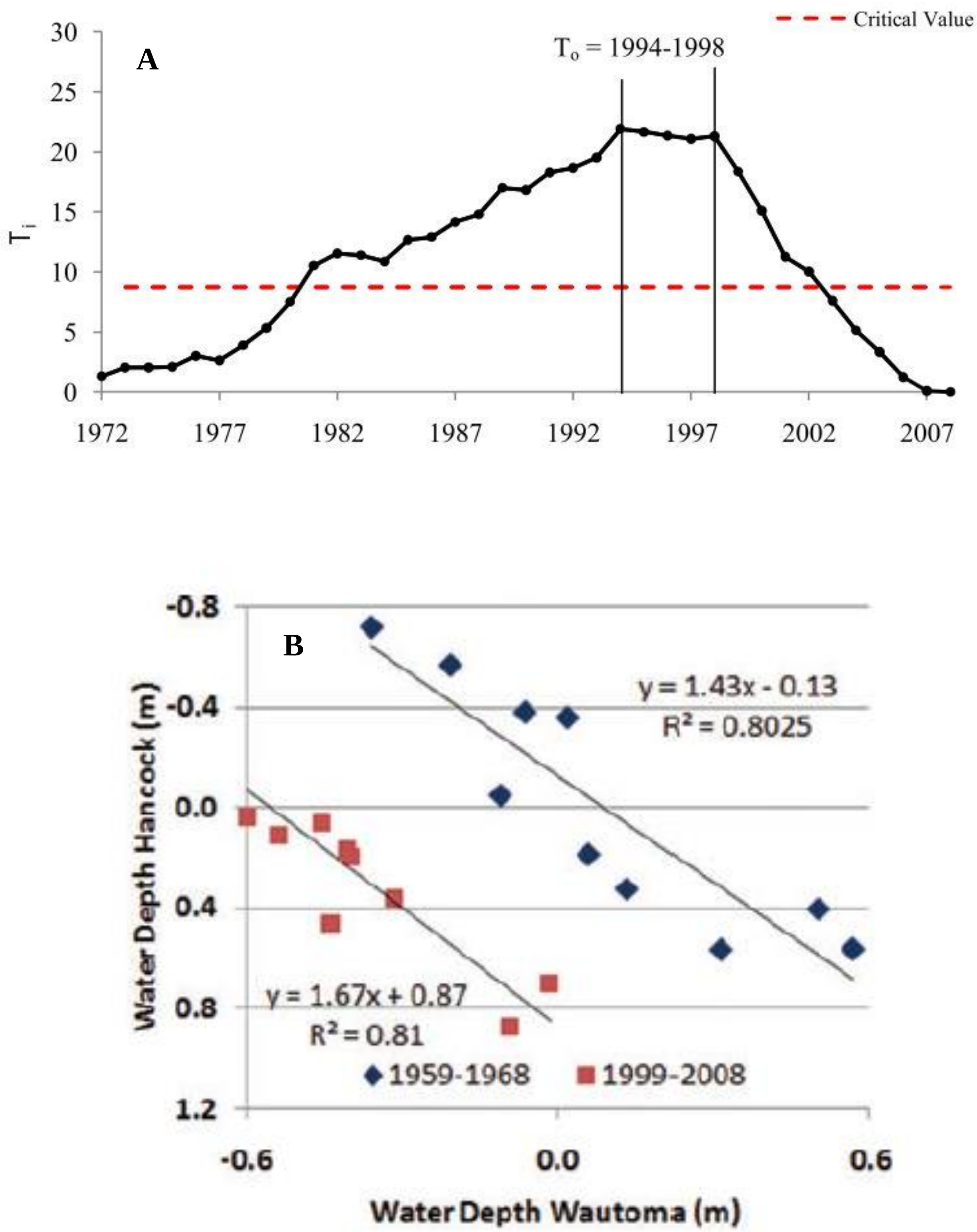

Figure 23. *A:* Results of the bivariate test for a change in mean groundwater levels at Hancock monitoring well when compared to Wautoma monitoring well for the period from 1972 to 2008 (Haucke, 2010). The change in mean occurred in 1999, one year after the last peak in the plateau in 1998. *B:* Regression analysis of average annual groundwater depth in 1959 to 1968 and 1999 to 2008 in Hancock (high irrigation well density area) and Wautoma (low irrigation well density area) (Kraft et al., 2012).

Table 9. Groundwater level decline in high density irrigation well areas during period between 1999 to 2008 as determined from comparing records from U.S. Geological Survey groundwater monitoring wells in high density irrigation well areas to low density irrigation well areas (Kraft et al., 2012). Estimated decline rates and approximate decline start dates are also shown.

Station	**Decline (m)**	**Decline Rate (m/y)**	**Decline Start**
Plover	0.64 (1.04)[1,*]	0.034	1973
Hancock	0.98[*]	0.066	1990
Bancroft[2]	0.25[*]	0.019	1984
Bancroft[3]	0.37[*]	0.019	1984
Coloma NW[2]	0.0	-	-
Coloma NW[3]	0.67[*]	-	1978

[1]Total decline = 1.04 m; irrigation decline = 0.64 m
[2]Comparison against Amherst Junction
[3]Comparison against Wautoma
[*]Decline is significant at the 0.05 level

I.5.1 Lake Levels

County, Wisconsin Department of Natural Resources, and U.S. Geological Survey personnel monitor various lake stages throughout the Central Sands region. Kraft and Mechenich (2010) investigated these data and concluded that 13 of the 39 monitored lakes contained records adequate for trend analyses (Table 9). For most of the monitored lakes, the stage data began around 1978. The average measurement interval was two years. The longest record was collected for 50 years at Long Lake-Saxeville by a local citizen.

Kraft and Mechenich (2010) presented a regression analysis of lake stages in high density irrigation areas (test series) versus groundwater monitoring wells in low density irrigation areas (control series). Comparisons of linear regressions of lake stages between test and control series for an early period of record (circa 1970s to 1980s) and a late period of record (1993 to 2007) showed statistically significant declines between 0.5 and 1.1 m (1.6 to 3.6 ft) for test series lakes (Figure 24; Table 10). Kraft and Mechenich (2010) attributed the declines to the impact of high-capacity irrigation wells.

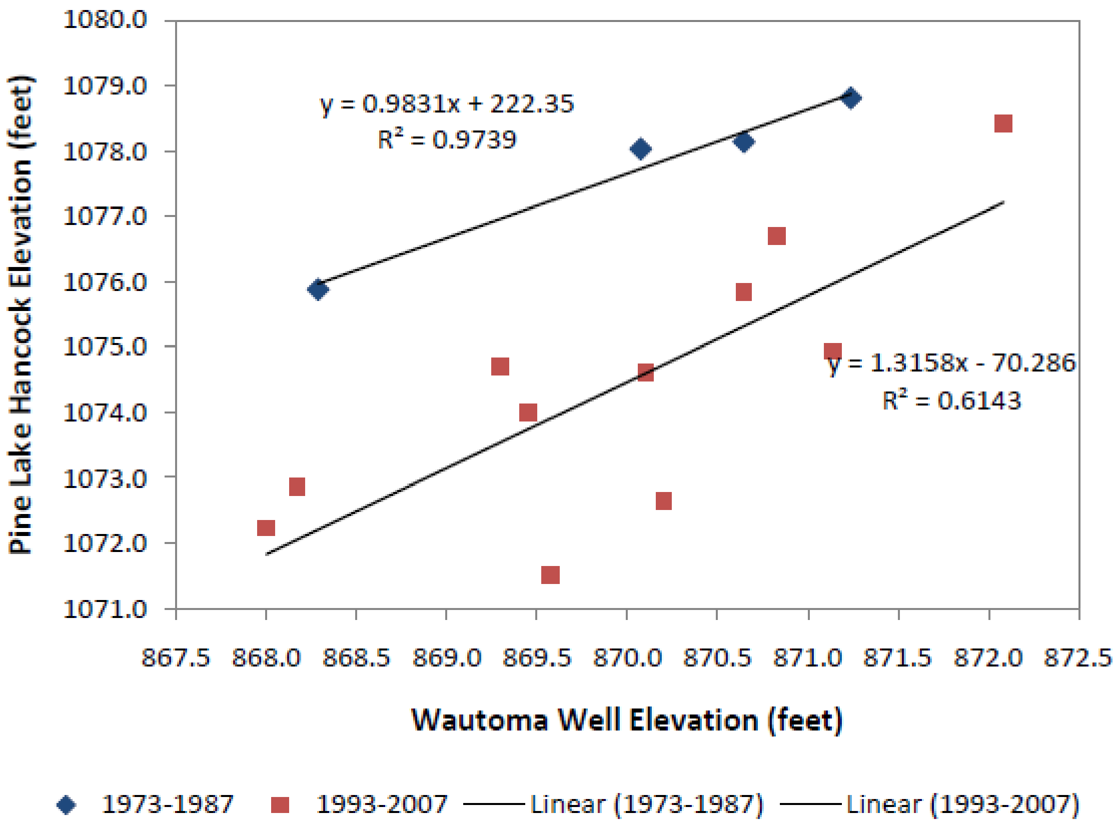

Figure 24. Linear regression of Pine Lake Hancock lake elevation (high density irrigated area) and the Wautoma groundwater elevation (low density irrigated area) in 1973 to 1987 and 1993 to 2007 (Kraft et al., 2010).

Table 10. Estimated decline in lake levels relative to Wautoma groundwater levels. The 95% confidence interval and significance level (p-value) are also shown (Kraft et al., 2010).

		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

I.5.2 Streamflows

Kraft et al. (2010) investigated U.S. Geologic Survey stream discharge records for the Central Sands region and found 71 daily monitored locations. Of the sites, 38 were not from small drainages, special projects (e.g., storm sewer flow monitoring), or those across a major hydrologic boundary. Many of the 38 sites on smaller streams had a limited flow record. Kraft et al. (2010) found that only four small stream sites are presently operational: Tenmile Creek near Nekoosa, 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 to 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. For a more detailed summary of the available streamflow record lengths, see Kraft et al. (2010).

The Little Plover River, a cold, headwater trout stream has been the focus of several past studies (Clancy et al., 2009; Kraft et al., 2012, 2010; Weeks et al., 1965). The Little Plover River experienced low flows and drying between 2003 and 2008, including drying of a 2-km (1.2 mi) reach near Eisenhower Road in 2005. Clancy et al. (2009) conducted a double mass analysis that compared the Little Plover River to the Wolf River at New London. This analysis indicated that the mean flows in the Little Plover River decreased relative to the mean flows in the Wolf River beginning in 1973. The results showed that relative to the Wolf River, the mean flows in the Little Plover River decreased by 0.11 to 0.14 m^3/s (3.9 to 5 ft^3/s) over the period from May to

August of 2005 and by 0.096 m^3/s (3.4 ft^3/s) in 2006. Clancy et al. (2009) attributed this decrease to pumping for municipal, industrial, and agricultural uses.

Kraft et al. (2012) conducted double mass analyses comparing the cumulative discharge of the Little Plover River to the cumulative discharge of the Eau Claire River, Wolf River and Wisconsin River (reference gauges). Results showed a decline in the mean baseflow in the Little Plover River relative to the reference gauges. The decline in the mean baseflow in the Little Plover River relative to the Eau Claire River occurred between 1973 and 1976 and was estimated to be approximately 0.036 m^3/s (1.3 ft^3/s) between 1977 and 1986 and 0.03 to 0.19 m^3/s (1.1 to 6.7 ft^3/s) between 1995 and 2009. Approximately 63% of the mean baseflow diversions from the Little Plover River were attributed to irrigation well pumping.

Kraft et al. (2012) plotted the cumulative baseflow diversions (the difference between cumulative actual and cumulative expected baseflow) from the Little Plover River relative to the discharge from the Eau Claire River, Wolf River and Wisconsin River over time. The plot illustrates that the baseflow diversions increased between 1977 and 1983, and leveled off or even rebounded between 1984 and 1986 (Figure 25). The rebound was attributed to an extraordinarily wet year in 1984 and above average precipitation years in 1985 and 1986. More trends in streamflow are described in the section titled “Reported Evidence for Impacts of Irrigated Agriculture on Water Resources”, subsection “Stream Baseflows”.

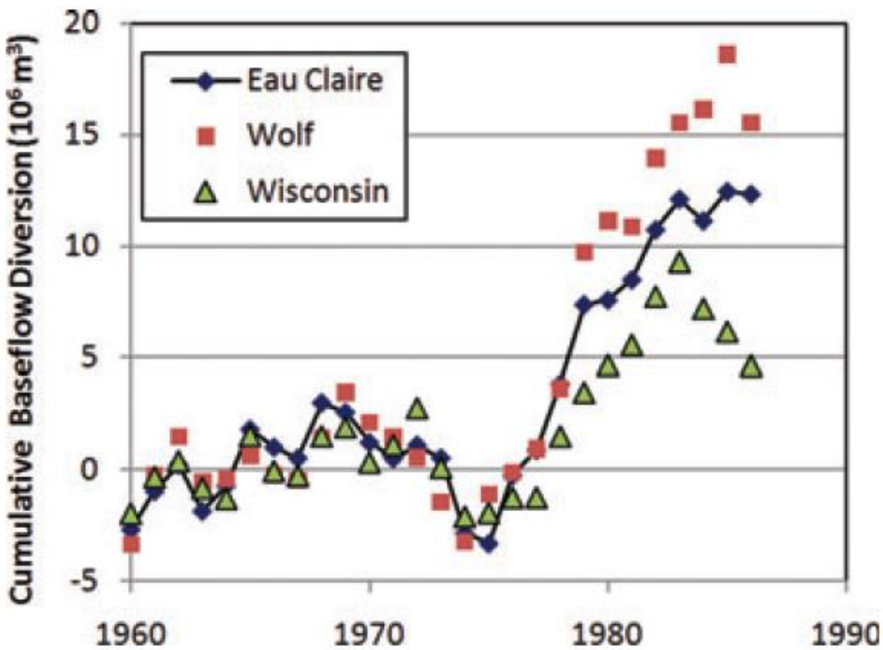

Figure 25. Cumulative baseflow diversions from the Little Plover River at Plover between 1960 and 1986 as determined from double mass analyses that compared the Little Plover River to the Eau Claire River, the Wolf River and Wisconsin River (reference gauges) (Kraft et al. 2012).

I.5.1 *Spring flows*

According to a spring inventory conducted by Macholl (2007), almost 300 springs have been documented throughout the Central Sands. Approximately, approximately 130 springs contain measured spring flow rates, though long-term time series of spring flows are sparse (Macholl, 2007). Mecan Springs is the second largest known spring in Wisconsin with a reported average flow rate of 0.62 m^3/s (22 ft^3/s) (WGHNS website, 2013). The spring system feeds the Mecan River System, which supports Class 1 trout streams and fen vegetation (WDNR website, 2013).

I.6 Reported Evidence for Climate-Driven Impacts on Water Resources

I.6.1 *Temperature*

Air temperature is a key variable driving the hydrologic cycle, particularly evapotranspiration and the resulting groundwater recharge. Studies assessing long-term temperature trends found that average annual temperatures in the Central Sands have increased approximately 1 to 2°C between 1950 and 2006 (Figure 26A) (Kucharik et al., 2010; Motew and Kucharik, 2013; WICCI, 2011). Average annual temperature increases are largely attributed to a greater magnitude of warming in winter and spring (2 to 4ºC) as well as a nighttime low temperature increase at a rate that is faster than daytime highs. Studies found that trends in diurnal temperature ranges have reduced by 0.35°C in spring to as much as 1.2°C in the summer (Kucharik et al., 2010). Additionally, the early arrival of spring, and a trend towards a later occurrence of freezing temperatures in fall has increased the length of the average growing season by as much as 15 to 20 days in Central Wisconsin (Figure 26B) (Kucharik et al., 2010; WICCI, 2011). However, it should be noted that trends toward cooler annual daytime highs have been observed across the region from 1950 to 2006, possibly due to the cooling effects of irrigation (Kucharik et al., 2010).

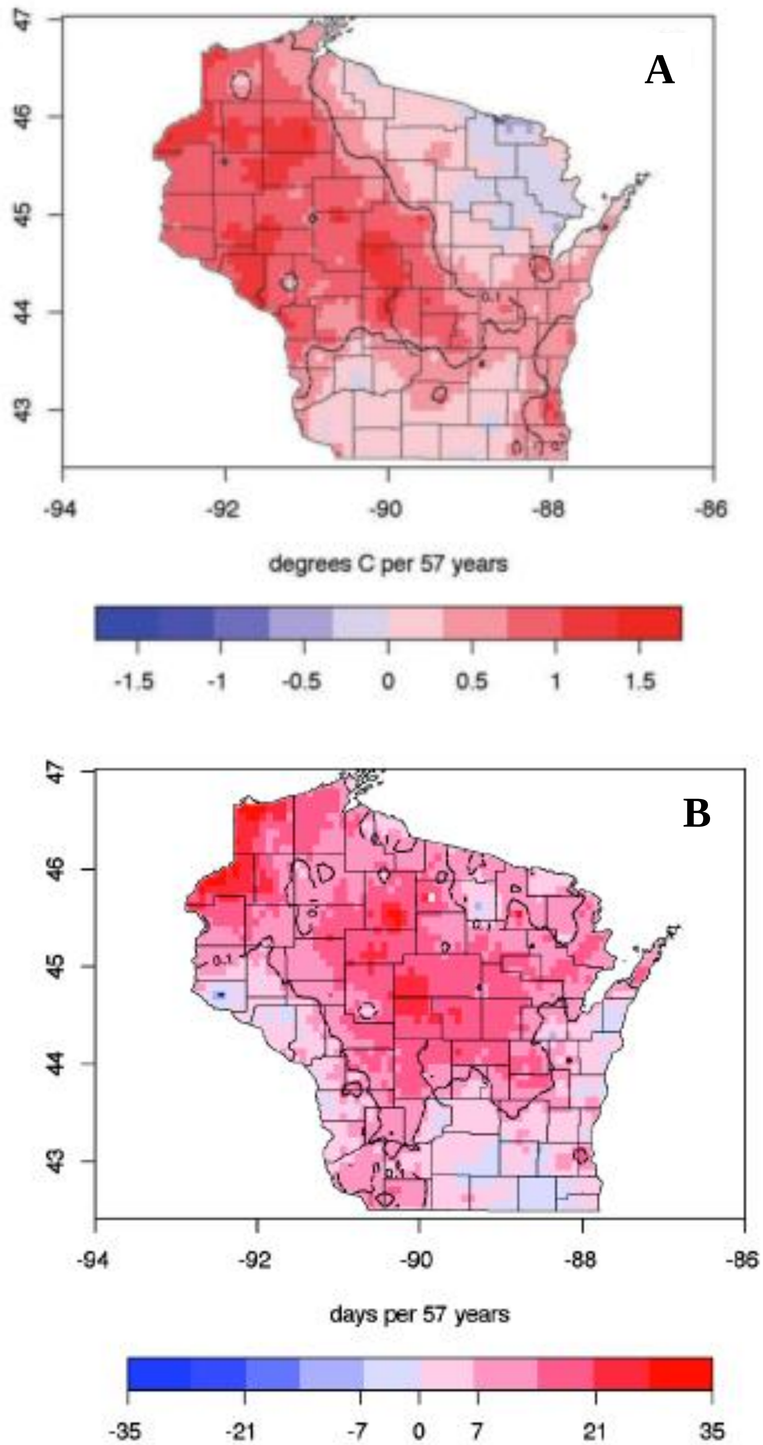

Figure 26. *A.* Average annual temperature increases in degrees Celcius. *B.* Increase in growing season length in days over the last 57 years (Kucharik et al., 2010).

I.6.1 Precipitation

Precipitation is the primary input in the terrestrial hydrologic cycle. Kucharik et al. (2010) investigated precipitation trends throughout Wisconsin between 1950 and 2006 based on gridded daily and monthly precipitation data at an 8-km spatial resolution developed from 176 climate stations (Serbin and Kucharik, 2009). Results showed that total average annual precipitation increased throughout the Central Sands by between 5 and 15 cm from 1950 to 2006, or approximately 0.9 to 2.6 mm per year, and that precipitation increased most during summer months (Figure 27) (Kucharik et al., 2010). Similarly, Motew and Kucharik (2013) evaluated trends in precipitation throughout the upper Midwest based on ZedX Inc. daily gridded weather at a 10-km resolution. Motew and Kucharik (2013) showed that the average annual precipitation throughout the Central Sands increased between 6 and 8 cm from 1948 to 2007, or approximately 1 to 1.3 mm per year.

Haucke (2010) investigated precipitation trends on a smaller scale. Using data from five cooperative observer (COOP) stations throughout the Central Sands, Haucke (2010) conducted a statistical analysis of precipitation data for the period from 1955 (1950 at Montello) to 2008. Station locations included Hancock, Montello, Wautoma, Stevens Point, Waupaca, and Wisconsin Rapids. Haucke (2010) found statistically significant increasing trends in average annual precipitation at Hancock, Montello, and Wautoma, but not at Stevens Point, Waupaca and Wisconsin Rapids (Figure 28). The average annual increase in precipitation across all stations, referred to as the composite increase, was statistically significant and was estimated to be 1.7 mm/yr, or 8.7 cm from 1955 to 2005.

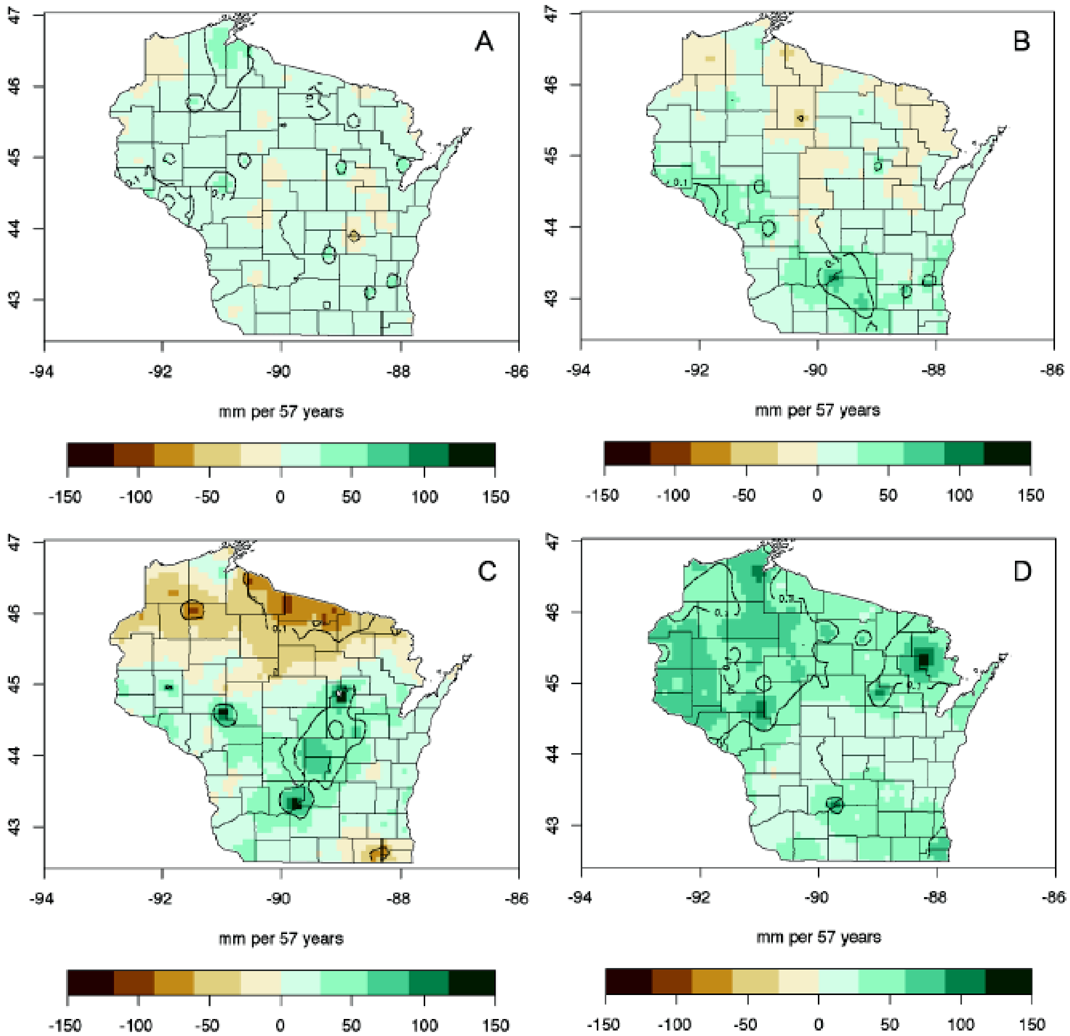

Figure 27. Trends in seasonal precipitation from 1950 to 2006 for winter (A), spring (B), summer (C), and fall (D) (Kucharik et al., 2010).

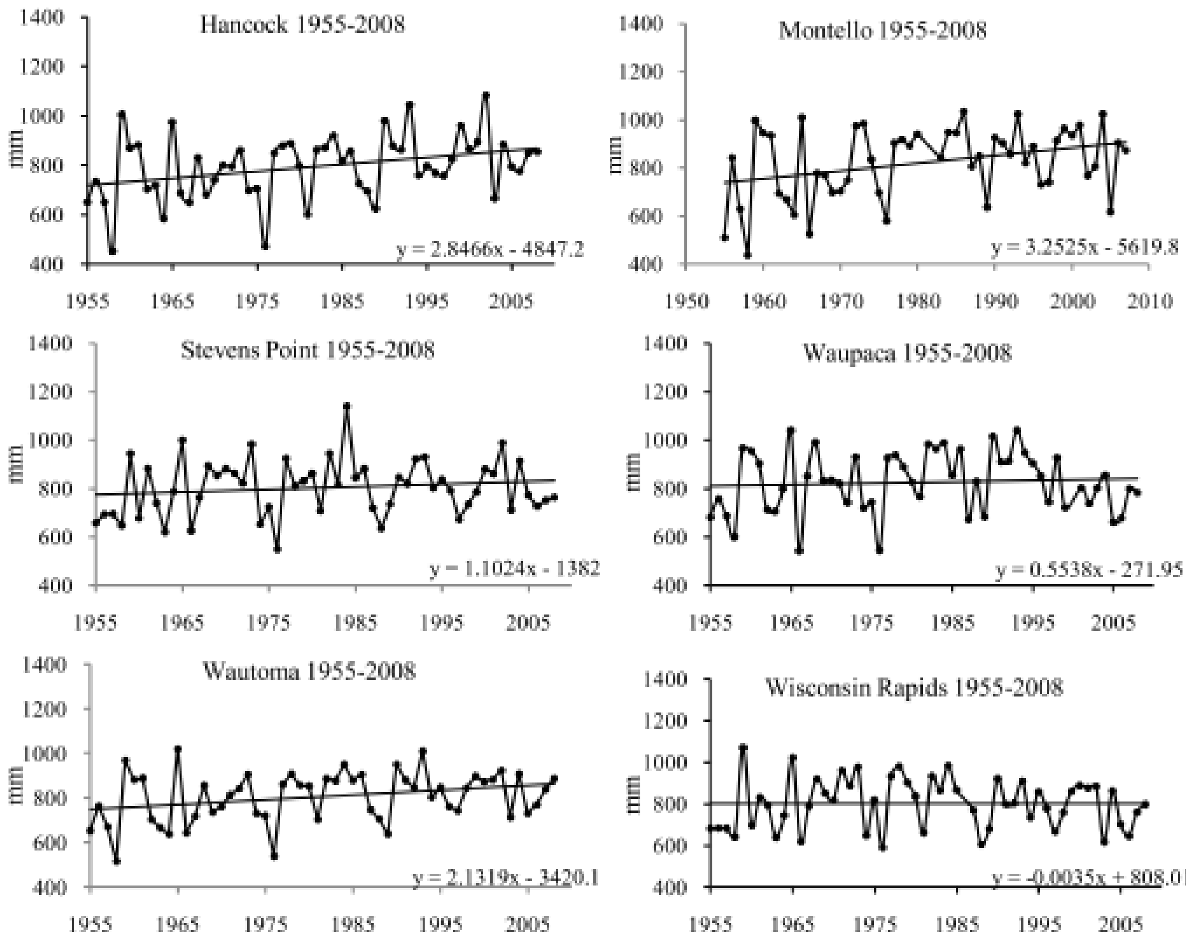

Figure 28. Trends in annual total precipitation from 1955 to 2008 at six precipitation gauges throughout the Central Sands (Haucke, 2010).

I.6.2 Climate-Driven Evapotranspiration

Evapotranspiration reduces recharge. Evapotranspiration rates are determined by a combination of climate factors such as relative humidity, air temperature, wind speed, solar radiation; and vegetation factors, such as basic plant physiognomy (trees, shrubs, grasses), leaf habit, photosynthetic pathway (C3 and C4) and leaf form. Rates of evapotranspiration are susceptible to changes in weather, climate, land cover and land use. Evapotranspiration in humid climates is often limited to the upper 1 to 2 m of soil and water that moves past that depth is likely to become recharge (Stoertz, 1989). In the case of a shallow (<2 m) water table, evaporation can occur from the water table itself (Stoertz, 1989). Additionally, vegetation capable of capturing and using water directly from groundwater are called phreatophytes. The extent of phreatophytes in the Central Sands has been thought to be limited to wooded riparian areas with shallow water tables (Weeks 1971).

To determine the effects of climate change on water cycling in the upper Midwestern United States, Motew and Kucharik (2013) investigated changes in evapotranspiration from 1948 to 2007 using Agro-IBIS, a dynamic ecosystem model. The model was driven by total solar radiation, maximum daily temperature, minimum daily temperature, precipitation, relative humidity and wind speed as gathered from the ZedX Inc. daily gridded weather dataset. The model simulated two biomes, deciduous and evergreen forest (Figure 29A), which included an array of plant functional types - an upper canopy consisting of temperate deciduous and conifer trees, and a lower canopy of shrubs and warm and cool season grasses. For the period 1948 to 2007, Motew and Kucharik (2013) examined the impact of climate drivers, such as increasing temperatures and precipitation and a lengthened growing season, on carbon and water cycling in deciduous and evergreen forest ecosystems that would be classified as natural or potential vegetation of the region. Changes in carbon cycling and hydrology connected to agriculture (e.g., cropping systems and irrigation) and phreatophytes were not simulated in the study. The Agro-IBIS model was validated using evapotranspiration estimated at the Park Falls Fluxnet site in northern WI, and has been previously validated in numerous other studies (Kucharik and Brye, 2003; Kucharik et al., 2006, 2001, 2000; Motew and Kucharik, 2013; Twine and Kucharik, 2008).

Motew and Kucharik (2013) found that total annual evapotranspiration driven by climate variables increased by approximately 4 to 10 cm from 1948 to 2007 (Figure 29B) (Motew and Kucharik, 2013). Concurrently, precipitation minus evapotranspiration (P-ET), an indicator of soil moisture available to plants and groundwater recharge (P-ET-Runoff) increased approximately 2.5 to 15 cm and 5 to 15 cm, respectively (Figure 30A, 30B).[5] Additionally, the ratio of evapotranspiration to precipitation (ET/P) decreased approximately -0.05 to -0.25 during the study period (Motew and Kucharik, 2013). Given that groundwater recharge increased under climate-driven forest biomes, results from Motew and Kucharik (2013) suggest that groundwater recharge throughout the Central Sands would have increased in the absence of agricultural production or other human land uses. Further studies that incorporate crop production and management are necessary for more definitive conclusions. For further discussion on vegetation factors affecting evapotranspiration, see the section titled “Reported Evidence for Impacts of Irrigated Agriculture on Water Resources,” subsection, “Evapotranspiration: Influence of Irrigation and Vegetation Density and Type.”

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[5] Note that the exact quantity provided was interpreted from Figures 29-30 since specific quantities quoted in the text in Motew and Kucharik (2013) averaged data over a larger region that the Central Sands.

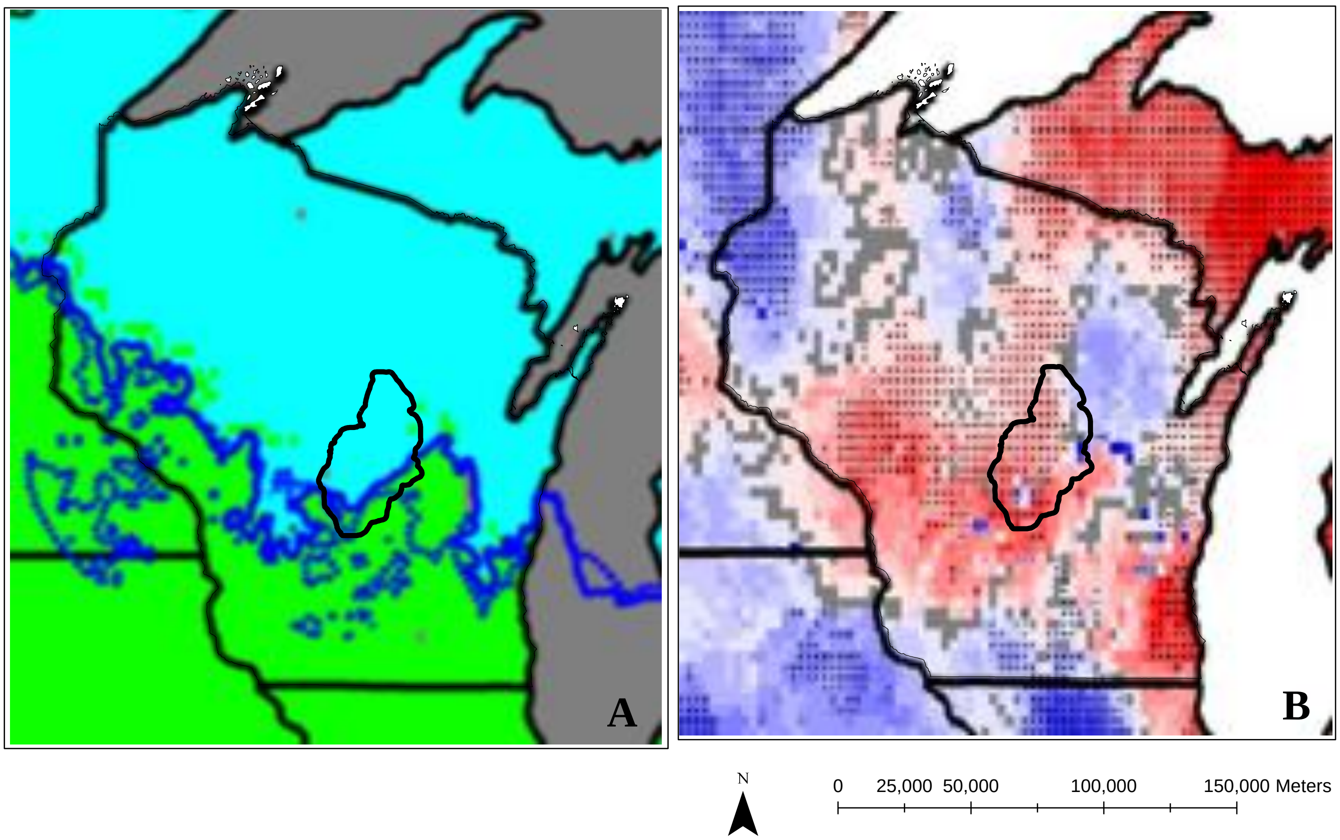

Figure 29. *A:* Simulated Agro-IBIS biomes in Wisconsin: deciduous forest (green) and evergreen forest (blue*)* (Motew and Kucharik 2013). *B:* Trend in actual annual evapotranspiration (mm) from 1948 to 2007 as simulated by Agro-IBIS (Motew and Kucharik 2013).

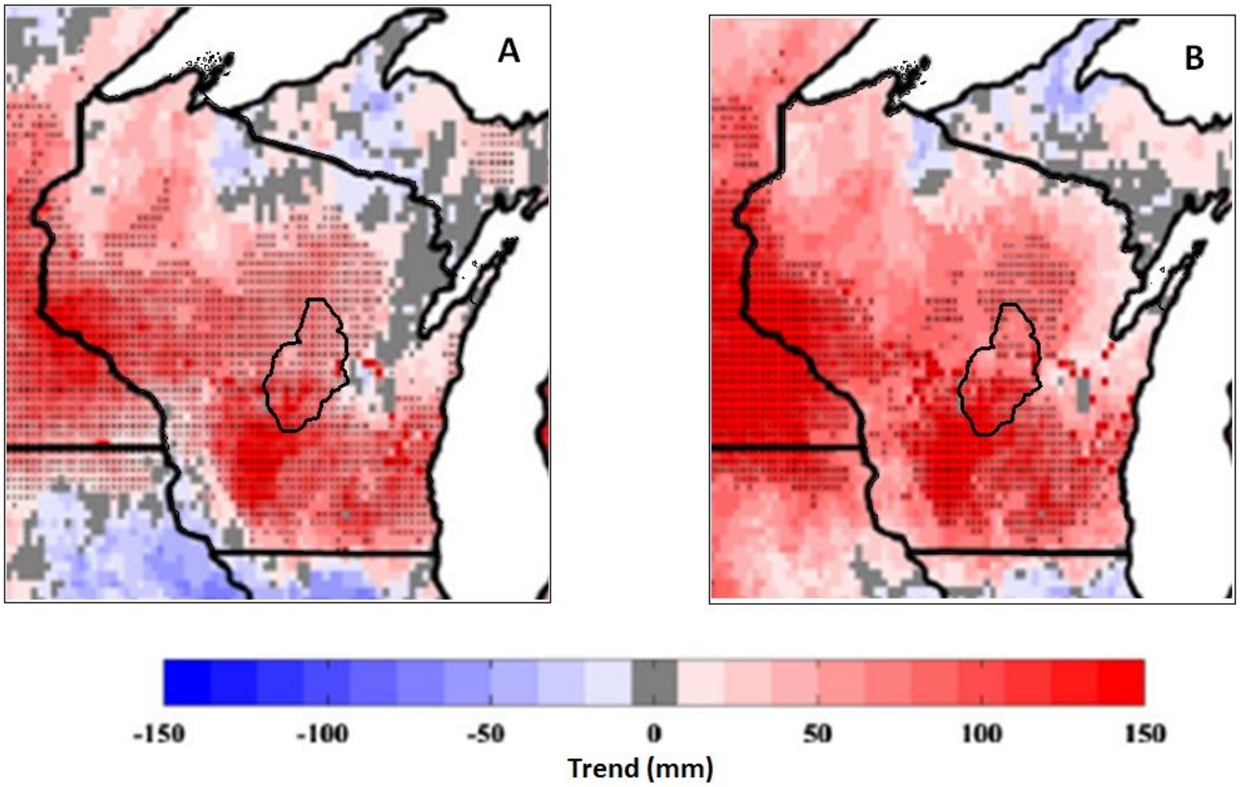

Figure 30. *A:* Agro-IBIS simulated trends in total annual in groundwater recharge (P-ET-Runoff) and (*B)* available soil moisture (P-ET) (mm/yr) from 1948 to 2007 (Motew and Kucharik, 2013).

I.7 Reported Evidence for Impacts of Irrigated Agriculture on Water Resources

I.7.1 Evapotranspiration: Influence of Irrigation and Vegetation Density and Type

Besides climate variables, evapotranspiration is influenced by irrigation volumes and system efficiency as well as properties of the soil, such as infiltration capacity, and vegetation type, such as growing season length, plant rooting depth, leaf area index, stomatal conductance, and plant density. Irrigation satisfies the difference between precipitation and the amount of evapotranspiration that maintains a crop near maximum potential evapotranspiration producing the optimal crop yield. As such, evapotranspiration from irrigated agricultural land is not limited by soil water availability as it the case in non-irrigated lands.

Irrigation in the Central Sands supports shallow-rooted vegetables grown in soil with low water storage capacity, such as Plainfield Sand and Plainfield Loamy Sand, common throughout the area (Figure 31). Since precipitation patterns are unpredictable, evapotranspiration varies appreciably on short timescales (i.e. day to day) and soils contain low water retention capacity, irrigation management can be challenging. Estimates of evapotranspiration are used to help determine adequate irrigation (Curwen and Massie, 2012). During average evapotranspiration conditions and no precipitation, Tanner et al. (1974) found that irrigation delay beyond five days for potatoes in Plainfield Sand can be damaging to yields, while a 3-day period is as short a period between irrigation applications as is feasible for commercial production (Tanner et al., 1974). On the other hand, over-irrigation can result from excessive single applications, frequent small, but excessive applications over a short time period or a lack of irrigation adjustment for rainfall between irrigation events.

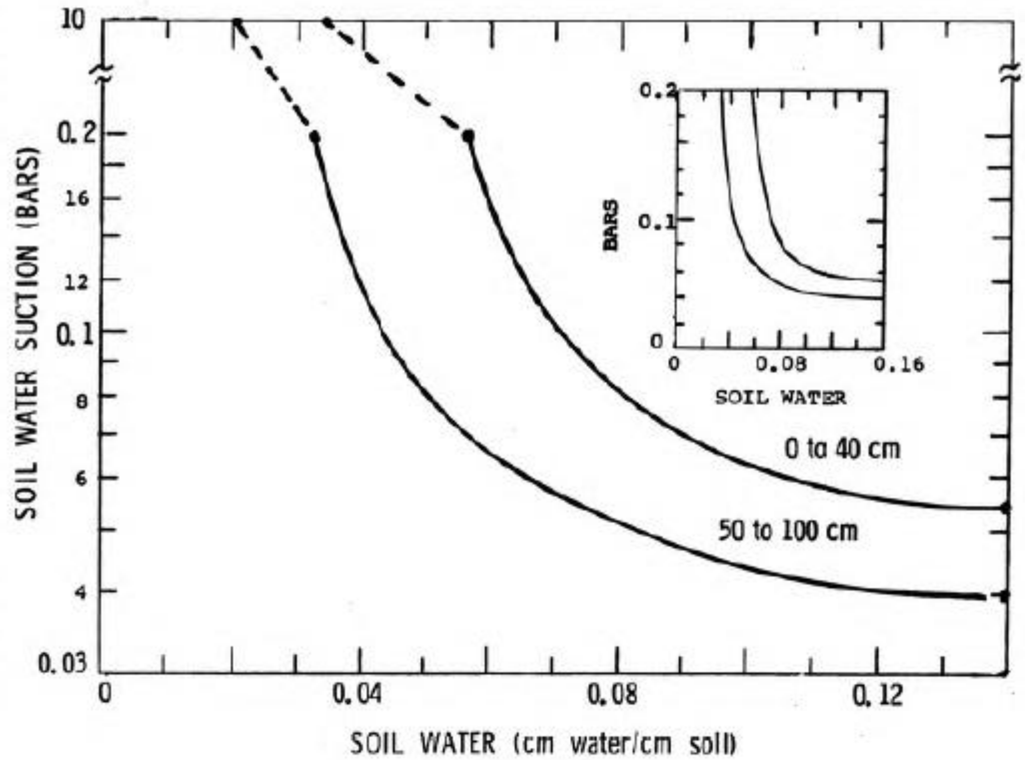

Figure 31. Water retention characteristics of the 0 to 40-cm upper profile and that of the course sand below 40 cm (Tanner et al., 1974).

Values of vegetation-specific evapotranspiration in the Central Sands with and without irrigation have been assessed on daily, monthly, seasonal and annual timescales using a variety of methods (Black et al., 1970; Naber, 2011; Tanner et al., 1974; Weeks and Stangland, 1971). To determine the magnitude of evapotranspiration from different vegetation types with and without irrigation, Weeks and Stangland (1971) conducted water balance computations defined by Thornthwaite and Mather (1955) for nine vegetation types in four surface water basins. The nine vegetation types included five types of non-irrigated vegetation or land use (evergreen forest, deciduous forest, grassland, unirrigated row crops and bare ground), native phreatophytic vegetation (defined as wooded vegetation in areas near the streams or marshes where plants have the ability to tap the groundwater table for transpiration), and three irrigation crops (beans, potatoes and corn). The four surface water basins were Ditch 5 of Tenmile Creek, Big Roche a Cri Creek, Tenmile Creek and Fourteenmile Creek near New Rome. At the time of the study, approximately 30% of total upland areas in all basins had been developed for irrigation (Weeks and Stangland, 1971).

The water balance approach uses measurements of precipitation, estimates of potential evapotranspiration based on climatic data, and estimates of soil moisture available for different vegetation types to conduct successive annual water balances to determine evapotranspiration, soil-moisture storage, and recharge for the basin by accounting for water movement into, storage within, and discharge from the soil zone (Weeks and Stangland, 1971). Non-irrigated evapotranspiration was determined by scaling potential evapotranspiration by the ratio of the actual soil moisture storage within root zone of the vegetation type to the soil moisture content at field capacity (Weeks and Stangland, 1971). Average annual evapotranspiration for phreatophytes was estimated to be equal to potential evapotranspiration. Irrigated crop evapotranspiration was estimated by using monthly water use factors for each crop based on the ratio of the acreage of the crop irrigated in the given month to the total crop acreage. Weeks and Stangland (1971) accounted for incomplete groundcover during the early stages of growth within the water use factor. Each vegetation-specific quantity of estimated evapotranspiration was then multiplied by the area of each vegetation type within the basins. Results for evapotranspiration, soil moisture and recharge were comparable to a separate water budget analysis conducted for the area.

The water balance approach showed that annual average evapotranspiration from 1948 to 1966 was 49.3 cm (19.4 in) for evergreen forest, 48.3 cm (19.0 in) for deciduous forest, 40.6 cm (16.0 in) for grassland, 39.6 cm (15.6 in) for unirrigated row crops and 35.8 cm (14.1 in) for bare ground (Table 11) (Weeks and Stangland, 1971). Average annual evapotranspiration for phreatophytes was 63.0 cm (24.8 in) Estimated average annual evapotranspiration values for irrigated crops were 48.3 cm (19 in) for beans, 54.6 cm (21.5) for potatoes, and 53.3 cm (21.0 in) for corn (Weeks and Stangland, 1971).

Weeks and Stangland (1971) noted that the water balance analysis was unable to account for variability due to different types of vegetation within the specified categories and vegetation density and maturity. Thus, the evapotranspiration values for each category should be used as estimates over large areas. Variability in annual evapotranspiration for unirrigated, nonphreatophytic vegetation types ranged between 25% below to 15% above the 19-year average. The annual variability in evapotranspiration estimates for irrigated crops and phreatophytes was less than the annual variability in unirrigated, nonphreatophytic crops.

Tanner et al. (1974) estimated and measured evapotranspiration for several crops in sandy soils in Hancock, Wisconsin using a weighing lysimeter and long-term weather records (Figure 32). Tanner et al. (1974) then compared the results to observed values of monthly precipitation. Results showed that evapotranspiration substantially exceeds mean precipitation in June, July and August. Measurements showed that the evapotranspiration from irrigated alfalfa is nearly as great as potential evapotranspiration. Evapotranspiration from row crops is less because of the soil exposure and greater stomatal influence. However, when vegetation provides good cover, evapotranspiration from irrigated row crops can be approximately 80% of the alfalfa evapotranspiration rate (Tanner et al., 1974). Additionally, Tanner et al. (1974) found that monthly evapotranspiration from a given irrigated crop “during any month or two-week period of the growing season will rarely vary more than 20% from the long-term average.” Black *et al.* (1970) showed that when row crops cover approximately 60% of the row space, the influence of cover becomes small since the evaporation from the underlying soil is limited by energy supply rather than the soil properties. Weeks et al. (1965) incorporated evapotranspiration estimates from Tanner into a water budget model for the Little Plover River Basin (Table 11).

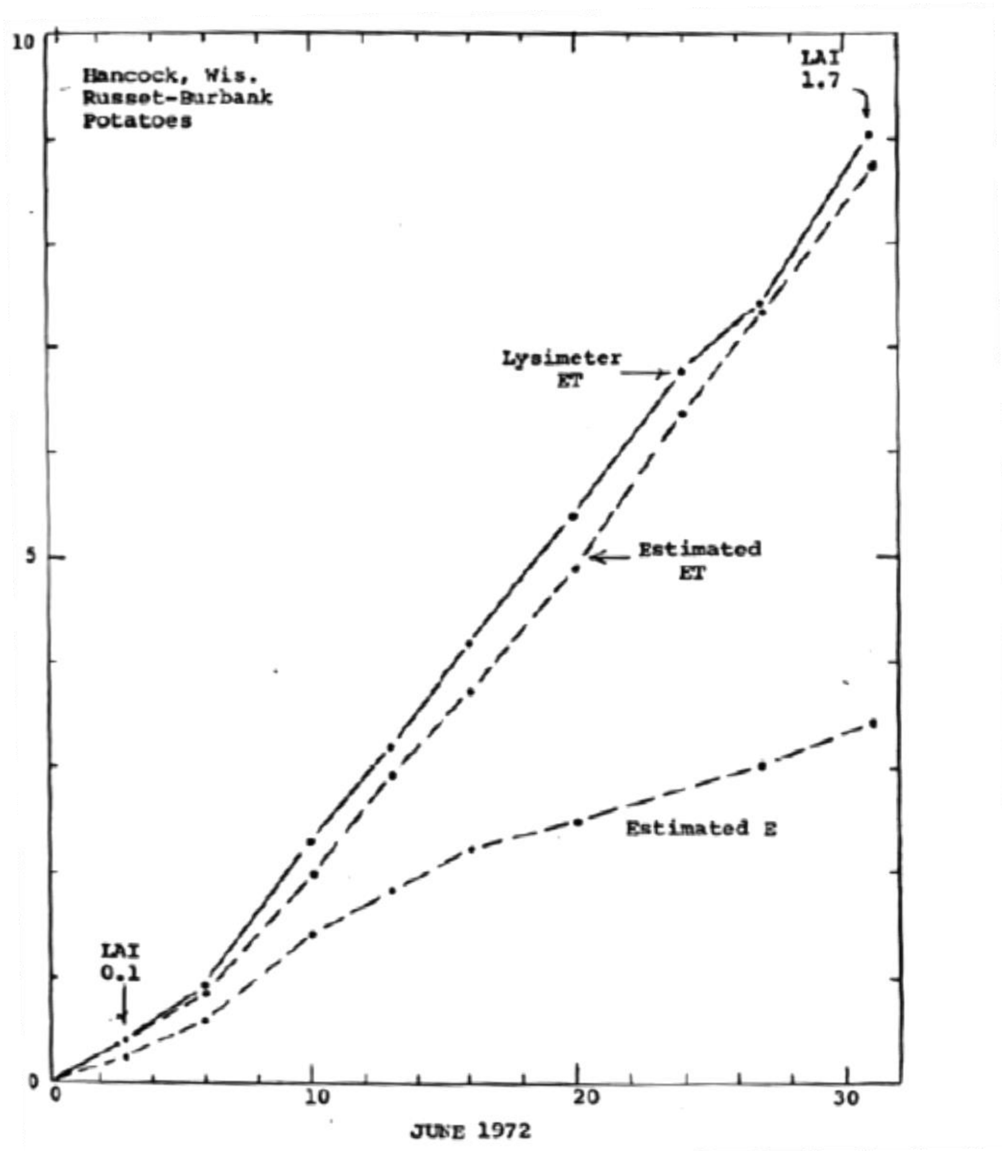

Figure 32. Comparison of estimated and measured evapotranspiration during the partial-cover period of June 1972 for Russet-Burbank Potatoes (Tanner et al., 1974).

Naber (2010) estimated rates of evapotranspiration for Central Sands vegetation using a dynamic ecosystem Integrated Biosphere Simulator (IBIS) model (Kucharik et al., 2000). Model inputs included daily precipitation and temperatures from a gridded dataset based on climate observation stations throughout Wisconsin (Serbin and Kucharik, 2009) and daily average cloud cover, wind speed and specific humidity from the National Center for Environmental Prediction (NCEP) Reanalysis data. Naber (2010) conducted simulations for irrigated field corn, non-irrigated field corn, cool and warm season grasses and shrubs, deciduous (hardwood) forest and coniferous (pine) forest on four sandy soil types from 1950 to 2006.

Results for average annual evapotranspiration were similar to those determined in Weeks and Stangland (1971) and Tanner and Gardner (1974) and showed that irrigated field corn resulted in the highest average annual evapotranspiration (50.6 cm), while prairie vegetation produced the least average annual evapotranspiration (34.6 cm) over the 57-year period (Table 11; Figure 33). Evaporation rates for hardwoods and pines resulted in average annual evapotranspiration rates slightly lower than those for irrigated field corn (48.3 cm and 46.5 cm, respectively) (Naber, 2011). Average annual differences in long-term average recharge beneath irrigated corn compared to pines and irrigated corn compared to prairie were 4.1 cm and 16 cm, respectively. Recharge beneath all vegetation types showed a roughly linear increase in net recharge over time (Figure 33) (Naber, 2011).

Fine-tuning system-specific evapotranspiration response to land management practices such as tillage, crop planting schedule, cover crops, and residue management will continue to be important for evapotranspiration estimation and future water resource management strategies as will be discussed in section titled “Knowledge Gaps.”

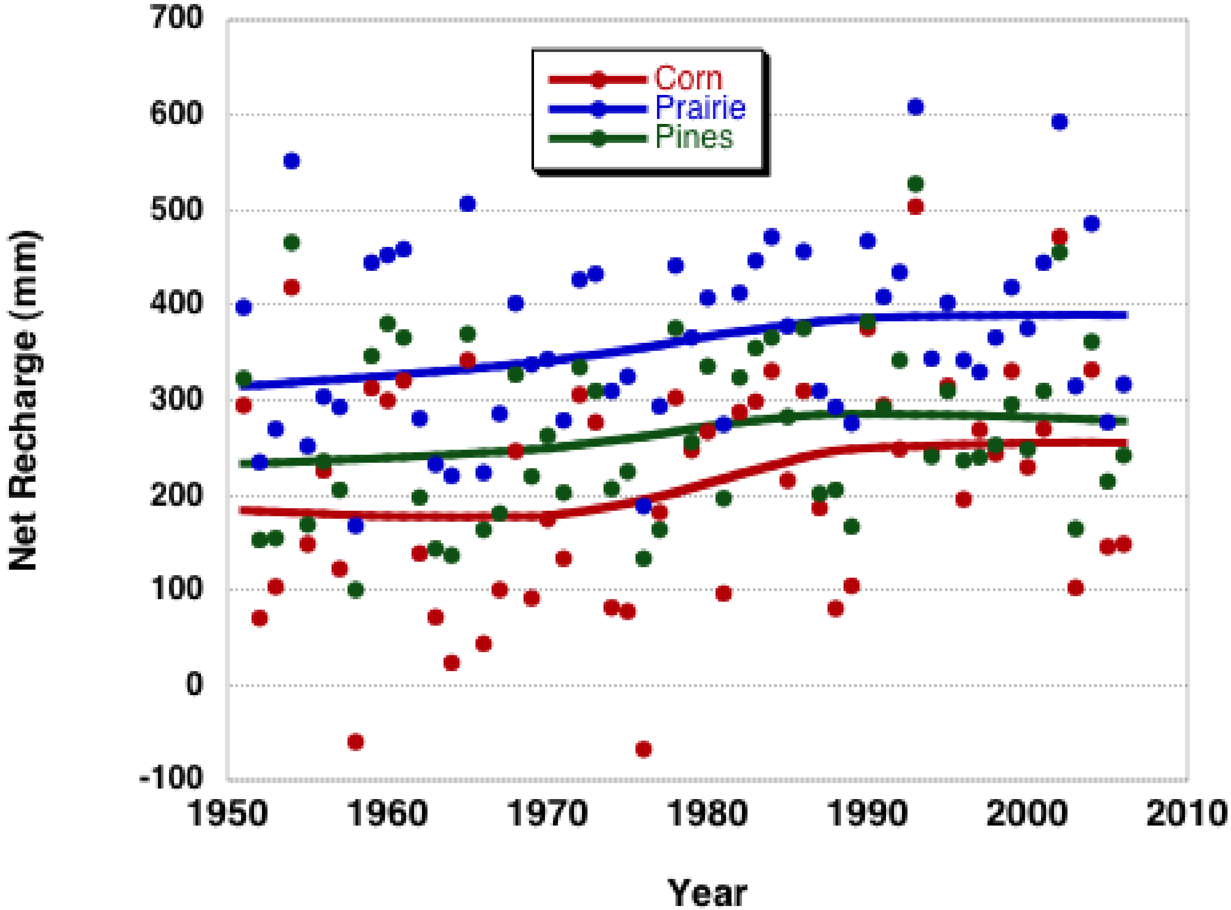

Figure 33. Modeled annual net recharge from 1950 to 2006 for corn, prairie and pine as determined from daily precipitation and estimated daily evapotranspiration using dynamic ecosystem IBIS model (adapted from Naber 2011). Smoothed lines for each vegetation type illustrate long-term trends.

Table 11. Average annual evapotranspiration for vegetation types in the Central Sands.

Crop	Average Annual ET (cm)	Method of Estimation	Author (year published)
Irrigated alfalfa-brome	61.0	From CB Tanner, written communication	Weeks (1965)
Irrigated beans	48.3	Water Balance	Weeks and Stangland (1971)
Irrigated corn	50.8	From CB Tanner, written communication	Weeks (1965)
Irrigated corn	53.3	Water Balance	Weeks and Stangland (1971)
Irrigated corn	50.6	IBIS	Naber (2011)
Irrigated potatoes	54.6	Water Balance	Weeks and Stangland (1971)
Irrigated grain	50.8	From CB Tanner, written communication	Weeks (1965)
Unirrigated crops	39.6	Water Balance	Weeks and Stangland (1971)
Forest	69.9	From CB Tanner, written communication	Weeks (1965)
Coniferous forest	46.5	IBIS	Naber (2011)
Evergreen forest	49.3	Water Balance	Weeks and Stangland (1971)
Forest (phreatophytes)	63.0	Water Balance	Weeks and Stangland (1971)
Deciduous forest	48.3	IBIS	Naber (2011)
Deciduous forest	48.3	Water Balance	Weeks and Stangland (1971)
Grasslands	40.6	Water Balance	Weeks and Stangland (1971)
Scrub	38.4	IBIS	Naber (2011)
Prairie	34.6	IBIS	Naber (2011)
Bare ground	35.8	Water Balance	Weeks and Stangland (1971)

I.7.2 Net Regional Recharge

Groundwater flow modeling is the current state of practice for understanding and evaluating groundwater flow dynamics. Groundwater flow models use computers to solve equations that describe the physics of groundwater flow. MODFLOW, a groundwater flow model code developed by the U.S. Geological Survey, is considered an international standard for groundwater flow modeling and is one of the most widely-used codes (McDonald and Harbaugh, 1996, 1988).

In contrast to directly measuring or estimating average evapotranspiration, long-term average evapotranspiration can be determined with a groundwater flow model by estimating long-term average net changes in recharge entering the groundwater flow system. This approach has been applied to the Central Sands using two MODFLOW groundwater flow models: one conducted by Clancy et al. (2009) and one conducted by Kraft and Mechenich (2010). Results for each of these models are summarized in Kraft et al. (2012) and will be described in parallel with one another.

Clancy et al. (2009) conducted a three-dimensional, transient model of the area surrounding the Little Plover River and incorporated groundwater level, stream level and streamflow data. Four calibration scenarios were constructed with varying the distributions of hydraulic conductivity and recharge. Simulations included ditches, streams, and pumping wells. Similarly, Kraft and Mechenich (2010) constructed a regional, steady-state, three-dimensional groundwater flow model constructed in MODFLOW. The model contained 200-meter discretized cells and used approximately 10,000 well logs to map the aquifer and sandstone bases (Hartman, 2007; Kraft et al., 2012; WDNR, 2010; WGNHS, 2008). Net groundwater recharge reductions were determined by decreasing recharge rates on irrigated areas in the groundwater flow models until a match between observed and modeled conditions was achieved (calibration).

Kraft (2012) concluded that water level changes in headwater areas between 1999 and 2008 were consistent with a net recharge reduction (and evapotranspiration increase) of 4.5 cm. The net decrease in groundwater recharge resulted in a groundwater table reduction of up to 1.2 m in heavily irrigated upland areas containing seepage lakes with minimal reductions near low-elevation discharge areas (Figure 34) (Kraft et al., 2012). Kraft (2012) also found that a 14.2 cm change in net recharge was consistent with flow reductions in the Little Plover River. Model results showed that the decline in Little Plover River baseflow varied according to reach, proximity to municipal, industrial and agricultural pumping wells and seasonal pumping regimes (Clancy et al., 2009). The differences between net recharge estimates in headwater areas versus areas near the Little Plover River may represent differences between regional precipitation amounts, cropping systems, pre-existing land cover and/or data or model limitations (Kraft et al., 2012).

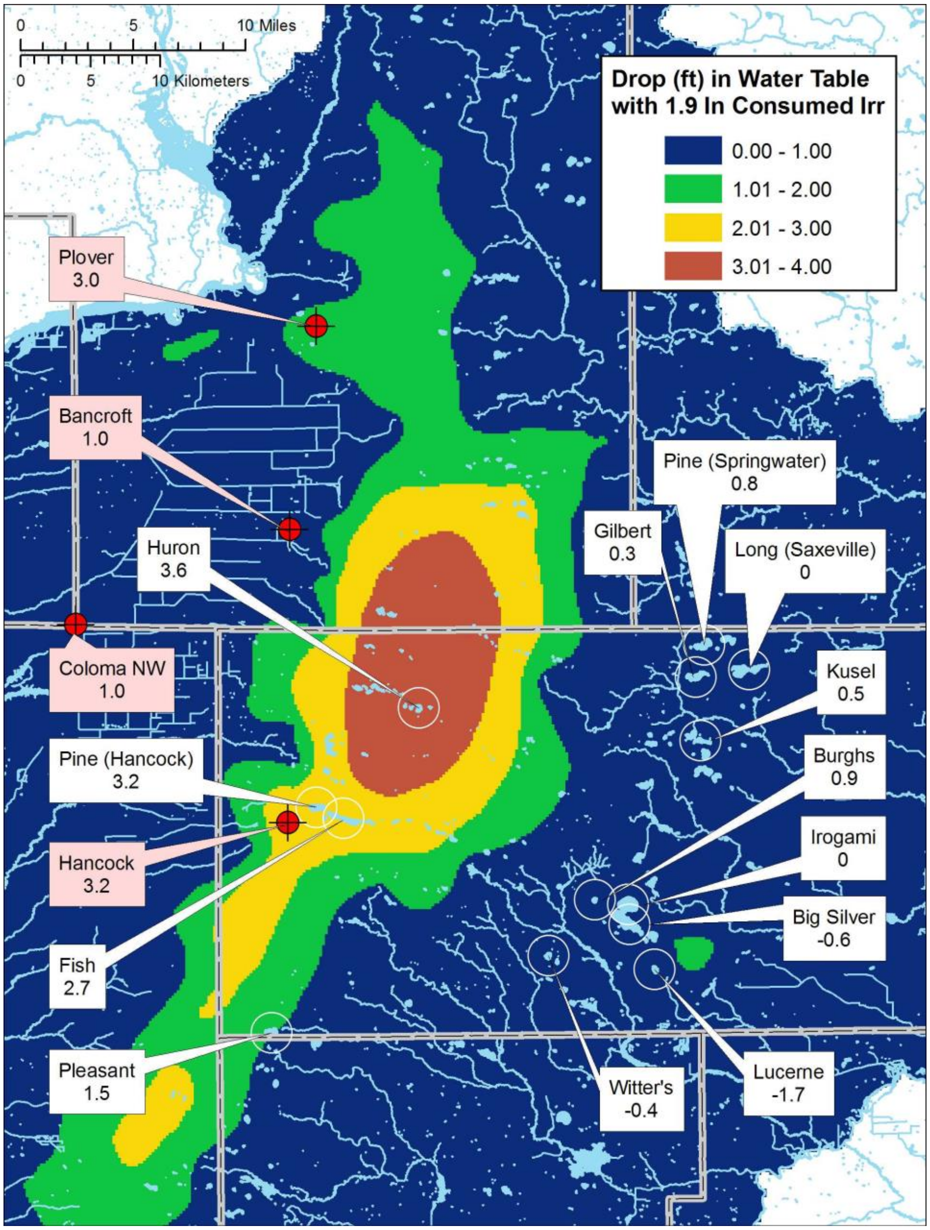

Figure 34. Groundwater level declines simulated using a regional, steady state groundwater flow model with a reduction in recharge on irrigated land equal to 4.5 cm per year. Monitoring well (pink boxes) and lake level (white boxes) declines determined from hydrographs are also shown (Kraft et al., 2012).

I.7.3 Groundwater and Lake Levels

By conducting a water budget analysis, Weeks and Stangland (1971) found that irrigation decreased groundwater elevations in 1966 and 1967 by approximately 15 cm per year in flat, upland areas in addition to the 60 to 90 cm seasonal decline that occurred from natural groundwater drainage to streams. Using this magnitude of decline, they back-casted the irrigation impact to the period from 1948 to 1967 and determined that irrigating 30% of landscape would have led to a long-term decline of 60 to 90 cm in groundwater levels (Figure 35). Expanding the irrigated area to 50% of the landscape area would have resulted in summer declines in groundwater levels along the divide by as much as 1.2 to 1.5 m beyond natural fluctuations and may have resulted in streamflow depletion or drying (Weeks and Stangland, 1971). [For comparison, note that during a drought year in 1958 the departure from average near the groundwater divide was a decline over 2 m].

According to Weeks and Stangland's calculations, converting grassland in the Big Roche a Cri Creek near Hancock to forest or tree plantations would also increase evapotranspiration, decrease soil moisture storage, and decrease net recharge similar to the effects of irrigation. Using a stream-aquifer model, they conducted a hypothetical scenario of converting 10% of the headwater area in Big Roche A Cri Creek basin from grassland to forest or tree plantations. Results showed that late summer stream discharge (August and September) would have decreased by approximately 4% for the period from 1952 to 1967 due to increased evapotranspiration and the reduction of summer and fall recharge, though results may be less significant during a drought (Weeks and Stangland, 1971).

I.7.1 Stream Baseflows

Weeks et al. (1965) estimated the effect of irrigation pumping on the Little Plover River Basin by accounting for differential rates of evapotranspiration for different vegetation types to determine the reduction in stream discharge. Evapotranspiration rates for different vegetation types were provide by Tanner (personal communication, 1967). Starting in 1965, Weeks et al. (1965) found that the maximum monthly rate of depletion in the Little Plover River due to irrigating 200 ha (500 ac) the first year and an additional 20 ha (50 ac) each year for 10 years was approximately $1.1x10^{-2}$ m^3/s (0.4 ft^3/s) the first year and approximately $1.4x^{-2}$ m^3/s (0.5 ft^3/s) after 10 years. Weeks et al. (1965) concluded that groundwater pumping and long-term increased rates of evapotranspiration leads to declines in stream discharge, particularly during the highly irrigated summer months.

To explore the connectivity of the Little Plover River to the surrounding groundwater, Weeks et al. (1965) conducted an aquifer pumping test. With a well located approximately 100 m (328 ft) from the river, Weeks et al. (1965) pumped at a rate of 4,200 liters/min (1,100 gal/min) for 74 hours and continuously measured groundwater levels and streamflows. During pumping, groundwater storage was reduced and caused a cone of depression around the well (Weeks et al., 1965). Results showed that after 56 hours, approximately 30% of water pumped was derived from streamflow and 70% was derived from groundwater storage. According to Weeks et al. (1965) even short periods of pumping in proximity to streams reduces streamflow.

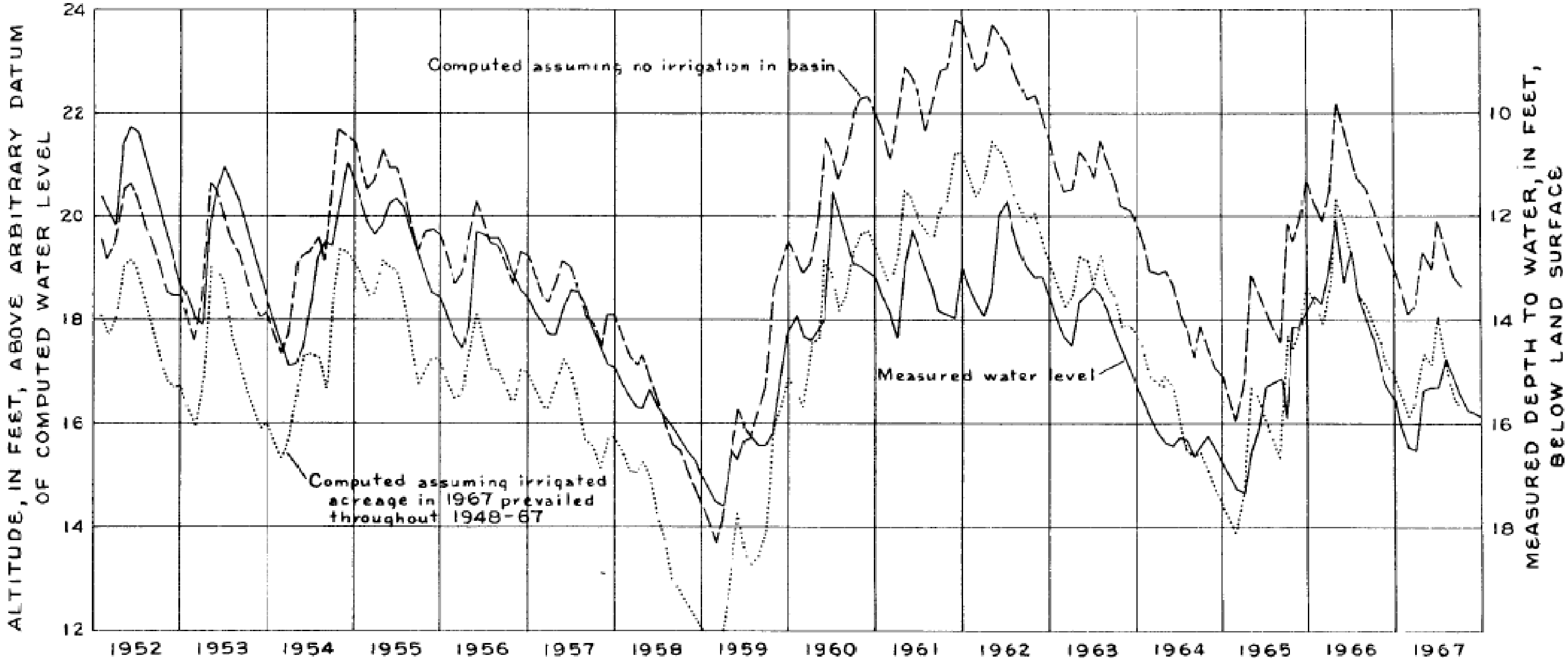

Figure 35. Measured (solid line) and hypothetical groundwater levels in well WS-20/8/14-7 from 1952 to 1967. Hypothetical groundwater levels calculated by back-casting the impact of irrigation on 0% (gapped line), 30%, (dotted line) and 50% (not shown) of the landscape from 1948 to 1967 are also shown.

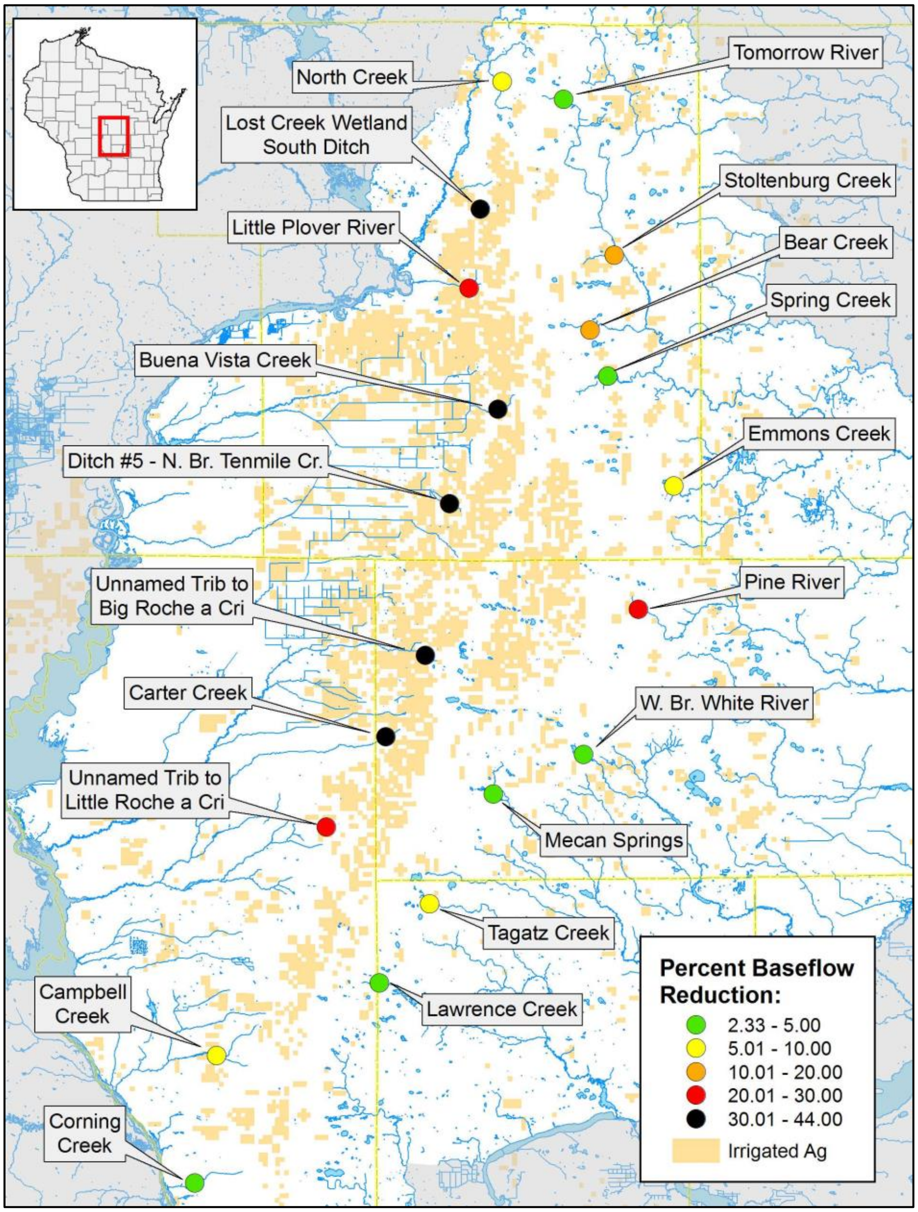

Figure 36. Declines in stream baseflows throughout the Central Sands as determined using a regional, steady-state groundwater flow model. Declines are represented as a percent reduction from expected flows (Kraft and Mechenich 2010).

According to the regional steady-state groundwater flow model, Kraft and Mechenich (2010) found that stream discharge throughout the Central Sands had declined. Declines varied according to the proximity of a stream to high-density pumping zones. To understand baseflow declines, Kraft (2012) used a base flow reduction index [(stream base state – stream altered state)/stream base state]. Using this method, Kraft et al. (2012) found stream headwaters near large densities of irrigated land to have the greatest reductions in baseflow. With a 4.8 cm (1.9 in) decline in net recharge on irrigated lands, headwater stream discharges decreased between 20 to 50% (Kraft et al., 2010). The Buena Vista Creek, Ditch #5 in the northern branch of Tenmile Creek, a tributary of Big Roche a Cri Creek, Carter Creek and Lost Creek Wetland South Ditch experienced baseflow reductions between 30-44%, while the Little Plover River, an additional tributary of the little Roche a Cri Creek, and the Pine River experienced a reduction in baseflow between 20-30% (Figure 36) (Kraft et al., 2010). In contrast, most lakes and streams in the eastern portion of the Central Sands with fewer high capacity wells only experienced baseflow reduction between a 2 and 5%.

I.7.2 Biological Impacts in Aquatic Ecosystems

The authors of this review found little documentation of the biological impacts of irrigated agriculture impacts on aquatic ecosystems specific to the Central Sands. Weeks and Stangland (1971) analyzed water temperatures in headwater and downstream forested areas. Results showed that maximum stream temperatures were dependent on the rate of baseflow and the extent of shading from bank vegetation, with lower temperatures in areas of higher flow rates and more shading. Greatest maximum July temperatures occurred at low-flow, headwater sites. Temperatures in downstream, forested areas were fairly uniform. Some downstream, unshaded streams and ditches in marsh areas did not support trout.

Weeks et al. (1965) conducted hypothetical scenarios of impact of irrigation on streamflows and found that streamflow depletion resulting from irrigation pumping could be detrimental to trout in the Little Plover River. Weeks et al. (1965) noted that lower stream stages may lead to a general reduction of living space, reduce the productive stream depth and hiding places, reduce the availability of food, and raise the summertime temperature of the stream. Weeks et al. (1965) reported that trout in Big Roche a Cri Creek had a slower growth and a higher natural mortality during years when the stream stage was relatively low.

I.8 Reported Evidence for Impacts of Drainage Ditches

Drainage ditches are predominant features in 4 of the 21 surface watersheds, approximately 30% of the total area in the Central Sands area. Drainage ditches serve to lower groundwater levels. Many drainage ditches throughout the Central Sands are former streams that have been dredged and channelized. Dredging deeper and wider channels for ditches enables a more rapid flux of water out of the aquifer system at earlier times in the season (Faustini, 1985; Zheng, 1986; Zheng et al., 1988). Where additional drainage ditches are present, groundwater flow paths are shortened, such as in the Buena Vista Groundwater Basin[6] where flow paths are seldom more than 5 to 6 km (3 to 4 mi) long (Figure 37) (Faustini, 1985; Zheng, 1986; Zheng et al., 1988).

Drainage ditches that were former streams and continue to provide ecosystem habitat experience exacerbated lows during summer months, particularly during years with dry summers. Similarly, drainage ditches near existing streams can exacerbate stream low flows, particularly during years with dry summers. Weeks and Stangland (1971) determined that installing one additional ditch in

[6] Buena Vista groundwater basin is a 44,000 ha (109,000 ac) area located in the northwest portion of the Central Sands.

the Tenmile Creek marsh area increased discharge out of the groundwater system an average of 185,000 m^3 to 200,000 m^3 (49 to 52 million gallons) during the month of August for the period between 1952 and 1967.

Drainage ditches also inhibit regional groundwater flow. During summer months, groundwater discharge to the ditches can occur throughout the depth of the aquifer such that the basin does not contribute to the regional groundwater flow system (Faustini, 1985; Stoertz, 1985). Underflow (groundwater recharge in the basin that does not discharge in the basin) may occur during other seasons depending on ditch bottom conditions such as depth and vegetation growth (Faustini, 1985; Zheng, 1986).

To date, no research has shown that ditches influence water levels in upland areas of the Central Sands. Additionally, the authors did not encounter literature reporting impacts of drainage ditches on local aquatic ecosystems. However, abundance and biomass of invertebrates, a major food source for juvenile and adult trout, was found to be higher in macroinvertebrate beds than in sand/gravel patches in the West Branch of the White River, Waushara County (Shupryt and Stelzer, 2009). Practices that increase sand/gravel patches in riparian systems, such as ditch channelization, may decrease invertebrate abundance and biomass.

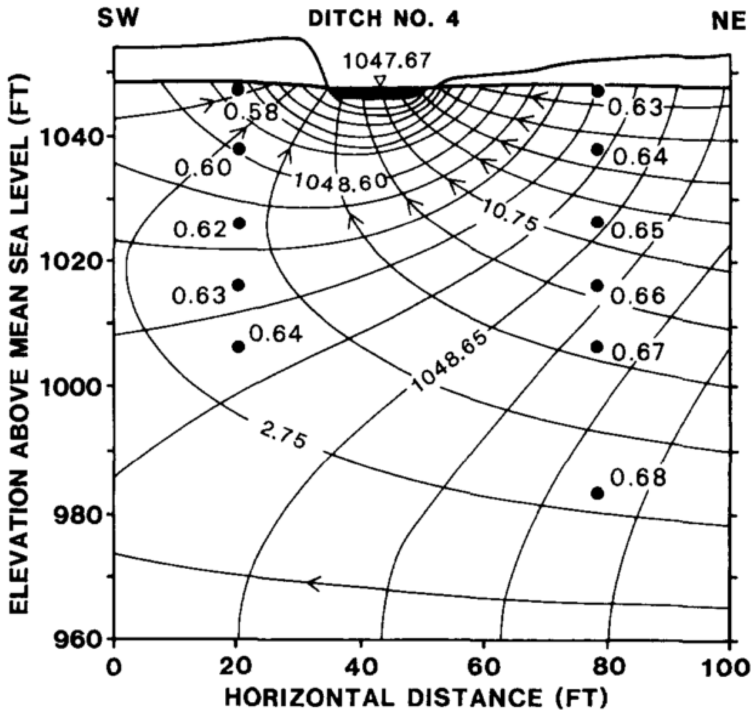

Figure 37. Impact of a drainage ditch. A drainage ditch is located in the top center portion of the figure. Flow lines with arrows headed in the direction of groundwater flow point toward the drainage ditch. Equipotentials, lines of equal head, are perpendicular to the flow lines (Zheng et al., 1988).

Future Climate Projections

Mean annual air temperatures across Wisconsin are expected to increase by 2.6 to 3.6°C (4.7 to 6.5ºF) by the mid-21st century and by 5.1°C (9.2ºF) by the late 21st century. This temperature increase will support longer growing seasons and an increase in growing degree day accumulations (Kucharik et al., 2010; Motew and Kucharik, 2013; WICCI, 2011). The greatest warming is projected to occur at nights and during the winter season (WICCI, 2011).

Increases in mean annual air temperature and the length of the growing season may increase average annual potential evapotranspiration by 10 to 20 cm across Wisconsin (WICCI, 2011). However, additional mechanisms such as increased cloud cover, increased atmospheric water vapor, and decreased plant stomatal conductance due to increasing atmospheric carbon dioxide may counteract and limit the increase of evapotranspiration (Joachim, 2012; WICCI, 2011).

Future precipitation is expected to remain at or near current levels in the Central Sands (WICCI, 2011). However, the time of year and the amount of extreme events may change such that wetter springs may occur and the top year-round precipitation events may be 20% heavier by the mid-21st century (WICCI, 2011). The Central Sands may also experience large inter-annual fluctuations in precipitation patterns, which may create challenges for agricultural land management (WICCI, 2011).

Future temperature and precipitation analyses indicate that climate change could mask or exacerbate impacts of groundwater pumping for irrigation. For example, a drier and warmer climate would increase demand for irrigation and result in further stress to aquatic ecosystems (WICCI, 2011). In July 2012, the Central Sands region experienced an extreme severe drought (“NCDC Drought Data,” 2012). Due to the reliability of high-capacity irrigation systems, the

Central Sands was able to provide animal feed to non-irrigated farmers across the Midwest. A high percentage of irrigation water was derived from groundwater storage during the 2012 drought.

Knowledge Gaps

Future groundwater and surface water management will require comprehensive modeling that can test alternative land use and land management strategies. Comprehensive modeling would require a groundwater flow model, an economic model, and a method for ecosystem valuation that is relevant to stakeholder goals and objectives for an identified area of interest. Modeling would need to be conducted on a transient time scale to accommodate annual and inter-annual variations in weather and dynamic land management practices. This section outlines key knowledge gaps that currently hinder a comprehensive modeling approach. Filling these gaps may allow for comprehensive modeling that can test alternative land use/land management scenarios. Note that the investigation of all knowledge gaps may not be necessary for assessment and implementation of all management strategies.

I.9 Comprehensive and Dynamic Modeling

Most modeling to date has been conducted under steady-state conditions and used to inform investigators about fundamental processes in the surface water and groundwater flow systems. To allow for regional land use and land management planning, future models need to have the capacity to test land use and land management scenarios in a way that accounts for dynamic variations in weather and management practices on a seasonal or shorter-term basis and incorporate both economic and ecosystem measures along with surface and groundwater flow processes. Coupling economic, crop-based or ecosystem-focused models to transient groundwater flow models would allow for testing of alternative management scenarios.

I.10 Land Management Influences on Rates of Evapotranspiration

Further research investigating system-specific evapotranspiration response to land management in the Central Sands is critical for integrating water management strategies with agricultural land use practices and assessing alternative land use and land management scenarios. Land management practices that influence system-specific rates of evapotranspiration include tillage, crop planting schedule, cover crops, and residue management. One method for estimating system-level rates of evapotranspiration and drainage is vadose zone lysimetry. The Kucharik Agroecology Research Group at UW-Madison is currently collaborating with Isherwood Farms (Plover, WI) to apply this method at 26 different sites in the northern portion of the Central Sands.

Another method for investigating system-level rates of evapotranspiration is through the use of eddy flux towers installed over different land use types. Eddy flux towers can be used to measure exchanges of carbon dioxide, water vapor and energy between terrestrial ecosystems and the atmosphere to provide estimates of evapotranspiration ("Fluxnet," 2013). The eddy flux tower closest to the Central Sands is located in Park Falls in northern Wisconsin and is not sensitive to the temperature and precipitation regime in the Central Sands. Installing eddy flux towers over different vegetative complexes within the Central Sands would be helpful for dynamically measuring system-scale variability in rates of evapotranspiration. Additionally, eddy flux towers incorporate irrigation system inefficiencies into the collected observations and would provide additional information about crop needs versus applied water.

I.11 Data Collection

Historic, present and future data monitoring drives our understanding of how the surface water and groundwater system has changed over time. Historic missing data and gaps in data limit the capacity to analyze trends in the surface and groundwater flow system as has been illustrated through incomplete stream gage and lake level records. For this region, present and future data collection of lake levels, stream discharge and groundwater levels are important for system and trend analyses and the construction of groundwater flow models. Other types of data that will be useful to collect simultaneously include land use and land cover types, land management (e.g. pumping rates, crop types, etc.), and ecosystem assessments that assess dynamics of organisms including fish, aquatic vegetation, birds and/or mammals.

I.12 Ecosystem Valuation

Given that water resources in the Central Sands support many land uses, there are inevitable tradeoffs. Some types of land uses, such as agricultural productivity, are more readily quantified. Whereas, other land uses, such as ecosystem conservation, are more difficult to quantify. The value of ecosystem conservation has been defined using the term ecosystem services, or benefits that people obtain from ecosystems (Lamarque et al. 2011). Ecosystem services are often grouped into four categories: 1) provisional services, such as food, clean water or raw material; 2) regulating services, such as regulation of floods, drought; 3) cultural services, such as recreational, spiritual, and other nonmaterial benefits; and 4) supporting services, such as soil formation and nutrient cycling (Reid et al., 2005). Ecosystem valuation is a relatively new field in science that seeks to understand the context of a given water resource management problem and its community in order to determine the most appropriate mode of valuation (Bingham et al.

1995). Finding a relevant mode of ecosystem valuation for the Central Sands community remains an important component of any future water resource management strategy or set of strategies.

Current Water Policies in Wisconsin

According to Wisconsin law, all surface water and groundwater resources within Wisconsin including the Central Sands are “waters of the state.” As stated in Wisconsin Statute 281, “portions of Lake Michigan and Lake Superior within the boundaries of this state [Wisconsin], and all lakes, bays, rivers, streams, springs, ponds, wells, impounding reservoirs, marshes, watercourses, drainage systems and other surface water or groundwater, natural or artificial, public or private, within this state or its jurisdiction [are]…waters of the state.”

Groundwater management areas are defined as “a multi-jurisdictional area including towns, cities, villages and counties within which the level of the groundwater potentiometric surface in any of its underlying aquifers has been reduced by 150 ft (46 m) or more from the level at which the potentiometric surface would be if no groundwater withdrawals had occurred (State Statute 281-34).” No areas in the Central Sands constitute a groundwater management area. Surface water resources in the Central Sands depend upon the first few meters of the groundwater water table to maintain water levels and flow. Fluctuations in the unconfined aquifer throughout the Central Sands vary significantly less than confined aquifers throughout the state due to their differing hydraulic characteristics. As such, the current definition of groundwater management areas in State Statute 281-34 is not hydrogeologically relevant to the Central Sands unconfined aquifer.

High capacity wells for irrigation are the largest water users in the Central Sands (Smail, 2013; Wisconsin Department of Natural Resources, 2012). According to Wisconsin law, the Wisconsin Department of Natural Resources (WDNR) is the primary governing body that determines placement of high-capacity wells for the State of Wisconsin. As stated on the WDNR’s website,

> "*the DNR reviews each application for a new high capacity well to determine whether the well, along with other wells on the same property, would result in significant adverse environmental impacts to waters of the state – which includes all streams, lakes, wetlands and public and private wells. Following completion of a technical review, if the DNR determines the well could directly result in significant impacts, the DNR would either deny the well application or impose conditions on the construction and operation of the well to prevent such impacts. The DNR bases the need to impose conditions or deny an application on the projected impacts to the affected water resource, e.g., estimated reductions in streamflow or lake level, and the resultant impacts to water temperature, the fishery and other ecological aspects of the stream or lake. In conducting these assessments, DNR considers site-specific hydrogeology, separation distance between the well(s) and the water resource, the hydrology and characteristics of potentially-affected surface waters, construction details of nearby wells, characteristics of the proposed wells such as construction, pump capacity, and the water use and pumping schedule for the proposed well and any other existing wells on the property*."

Thus, if the WDNR determines that a proposed well could have a significant adverse effect on waters of the state, it will deny the application or condition any approval to avoid the impact, and this may include moving the well, reducing the well's maximum capacity or altering the well construction.

In 2009, the Village of East Troy, the Lake Beulah Management District, the Lake Beulah Protective and Improvement Association and the WDNR were involved in a dispute of over the WDNR's position in permitting the installation of a high-capacity municipal well. This dispute was settled in Wisconsin State Supreme Court, which ruled that the WDNR has the authority and a general duty to consider whether a proposed well may harm waters of the state. To comply with this general duty, the WDNR must consider the environmental impact of a proposed high capacity well when presented with specific, concrete scientific evidence of potential harm to waters of the state (*Lake Beulah Management District v. Dep't of Natural Res.*, 2011 WI 54 (July 6, 2011). Following this ruling, the WDNR reviews high capacity well permits for impacts on all waters of the state in addition to its specific review of proposed wells in groundwater protection areas or near springs with a normal flow of 1 cubic foot per second. In addition, according to

State Statute 281.34(5m), “no person may challenge an approval, or an application for approval, of a high capacity well based on the lack of consideration of the cumulative environmental impacts of that high capacity well together with existing wells.”

A Path Forward

Declining water levels and the expansion of irrigated agriculture throughout the Central Sands have raised stakeholder concerns about regional water resources management. At this time, there is no ongoing legal or voluntary decision-making strategy for organizing stakeholders, experts and decision-makers to formulate and conduct regional water resources management in the Central Sands. However, components of a voluntary strategic process have been implemented.

According to the co-author's assessment, shared vision planning, as summarized by Palmer et al. (2013), offers a strategy that appears to be suitable for the region. Shared vision planning employs disciplined planning, structured stakeholder participation and computer modeling to develop and implement management strategies. Palmer et al. (2013) organizes the shared vision planning process into the following steps: 1) build a team and identify problems, 2) develop objectives and metrics for evaluation, 3) describe the status quo, 4) formulate alternatives to the status quo, 5) evaluate alternatives, 6) select and implement an alternative, and 7) exercise, update and use the plan. This section describes previous collaborative efforts in the region in the context of these shared vision planning steps; presents a portfolio of management strategies that have been researched and/or implemented in other locations; and makes recommendations regarding future activities that would lead to regional water resources management.

I.13 Previous Collaborative Efforts in the Central Sands

Over the last decade, stakeholders including growers, canning factory workers, individuals representing multinational corporations, residents who fish and kayak in surface waters and others who affect or are affected by water resources in the Central Sands have gathered to discuss water resource management practices. These gatherings have been organized by a local leaders, experts and decision makers from a variety of entities. In 2002, the Wisconsin Department of Natural Resources organized the Central Wisconsin River Basin Partnership Team in effort to increase awareness about natural resource issues throughout the basin (Wisconsin Department of Natural Resources, 2002). In 2009, the Wisconsin Potato and Vegetation Grower Association (WPVGA) formed the Water Task Force, a group of community leaders, producers, food processors, and researchers, to advocate for agricultural productivity, while protecting groundwater and associated streams, lakes and wetlands (WPVGA website, 2013). Additionally, the Friends of the Central Sands (FCS) formed to promote healthy landscapes through stewardship, community involvement, scientific knowledge and advocacy (FCS website, 2013). Similarly, the Central Sands Water Action Coalition (CSWAC) has worked to advance sustainable groundwater policies within the Central Sands region (CSWAC website, 2013).

In 2011 and 2012, the College of Agriculture and Life Sciences (CALS) and Wisconsin Institute for Sustainable Agriculture (WISA) at UW-Madison organized a series of stakeholder engagement meetings, called the Central Wisconsin Groundwater Initiative. The Central Wisconsin Groundwater Initiative involved expert presentations and stakeholder-facilitated conversations during which the identified regional challenge was *"*to create a way of using this bountiful resource [water] that will allow the economy and agriculture to prosper while protecting the water resources that we all share (Nowak, 2013)*."*

The Central Wisconsin Groundwater Initiative provided an opportunity for individuals to express their perspectives on water management. In general, these perspectives were polarized. Some residents felt that high-quantity water users have the responsibility to prove that their water consumption will not have adverse effects on the water rights of others and the environment prior to consumption. In contrast, other residents asserted that high-quantity water users have the right to consume water to maintain their livelihood and the State's economic prosperity until evidence supports that this is not in the best interest of the majority. Determining who bears and manages the burden of proof is a philosophical disagreement that has yet to be clarified and directly affects current and future water resource management in the region.

In spite of these differing perspectives, there was agreement on the following goals:

1. Maintain healthy waters and ecological resources in the Central Sands region during future water development;

2. Restore healthy waterways in the Central Sands region that have been compromised;

3. Promote and maintain a vibrant agriculture industry (Nowak, 2011).

From these three goals, specific objectives and metrics for evaluation could be established, in accordance with the shared vision planning process articulated by Palmer et al. (2013). Objectives and metrics can be developed via back-casting, the process of working backward from a desirable future to determine a timeline for necessary actions (Alley and Leake, 2004; Brandes and Brooks, 2006). Metrics are often defined according to direct human benefits, indirect human benefits, direct benefits to environmental systems, or indirect benefits to environmental systems. Examples of metrics include gross domestic product, agricultural productivity or yield, biodiversity or target species indicators, and environmental flows. For a review of approaches for establishing environmental flow metrics, see King (1999). For additional descriptions of how the goals can fit within the wider scope of the shared vision planning process, see Palmer et al. (2013).

I.14 Portfolio of Strategies

The formulation of alternative water resource management strategies in shared vision planning is often an iterative process involving input from stakeholders, particularly since “solutions…are often not within the power of any one entity to implement (Palmer et al., 2013).” To provide a starting point for stakeholder strategy-focused discussions, this subsection explores strategies that have been researched and/or implemented in other locations. These strategies vary with respect to the magnitude and uncertainty of their potential impact and their difficulty of implementation. The strategies are divided into three categories: technological adaptation, landscape scale management and water transfer and are summarized in Table 12.

I.14.1 Technological Adaptation

Water use efficiency (WUE) is defined as total crop biomass, dry matter growth or crop grain per unit of water expressed as transpiration, evapotranspiration or total water input to the system per unit time (Sinclair, 1984; Vazifedoust et al., 2008). This section describes three types of strategies for bringing farmers closer to reaching the WUE benchmark function: precision agriculture, deficit or deferred irrigation, and drip irrigation (Figure 37).

Table 12. Type, description, challenges, and examples of potential strategies for Central Sands water resource management.

Technological Adaptation	**1) Precision Agriculture** Employ mobile computing and other technologies to conduct dynamic system management in response to spatial and temporal field variability to reduce rates of evapotranspiration	Challenges: Lack of research showing significant quantitative impact Example: Nebraska
Technological Adaptation	**2) Deferred or Deficit Irrigation** Strategically account for rainfall within irrigation applications or apply less water than needed without significantly affecting yields	Challenges: Reducing applied irrigated water without significantly affecting yields Examples: Delaware, Denmark
Technological Adaptation	**3) Drip Irrigation** Install drip irrigation technology	Challenges: Costly Example: Maine
Landscape Level Management	**4) Drainage Ditch Management** Seasonally manage drainage ditches	Challenges: Installation costs and potential flooding of fields Example: Ohio
Landscape Level Management	**5) Low ET Landscapes** Alter planted species and/or cultivars, plant GMO crops, use low-density planting	Challenges: Socio-economic factors affecting crop choice and density, opposition to GMO crops Example: None
Landscape Level Management	**6) Comprehensive Irrigation Management Plan** Design diverse, sub-basin scale solutions such as groundwater conservation zones, agricultural priority zones, and/or water deficit sharing	Challenges: Decentralized decision-making, mediation of conflicting stakeholder values Examples: Washington, High Plains Aquifer, Germany
Water Transfer	**7) Regional Water Transfer** Divert river water (i.e. Wisconsin River) via large canal or pipeline from river to area of interest	Challenges: Socio-economic pushback, costly Examples: Colorado River Aqueduct, California State Water Project, Texas Water Project, North China Plain South-North Water Transfer Project
Water Transfer	**8) Local Flow and Water Level Augmentation** Augment water-stressed streams or lakes via pumping groundwater or surface water from less stressed areas directly into or near the water-stressed water body	Challenges: Groundwater levels surrounding water-stressed body may be too low for water augmentation to be helpful; stratigraphy may be unsuitable for maintaining water levels Examples: Cedar Lake, Wisconsin

Precision Agriculture

Precision agriculture employs satellite positioning systems, in-field and remote sensing, mobile computing and data acquisition, advanced information processing and telecommunications to conduct dynamic system modeling and management in response to spatial and temporal field variability throughout the production process (Greenwood *et al.*, 2009; Zhang *et al.*, 2002). In a study conducted on the Tri-Basin Water Resources District in Nebraska, grain yield was maximized with 900 mm of applied water, yet over 55% of the total fields in the district received excess water over the 900 mm requirement and resulted in yields that were 20% below the mean WUE benchmark function. Zhang et al. (2002) and Sadras and Angus (2006) discussed how accounting for spatial and temporal variability in soil, topography, precipitation, crop factors, and field management practices can reduce the total amount of applied water for irrigation (Figure 37).

Deferred or Deficit Irrigation

Deferred irrigation is the process of applying irrigation in a way that decreases the amount of applied irrigated water by accounting for rainfall. Thus, irrigation pumping is reduced, though crop evapotranspiration is maintained. Deficit irrigation is the process of applying irrigation in a way that decreases the amount of applied irrigated water by accounting for rainfall *and* applying irrigated water below potential evapotranspiration, which results in water stress on plants (Greenwood et al., 2009). Researchers in Central Wisconsin are currently collecting data while testing deferred and deficit irrigation practices in efforts to quantify the effects of each of these practices on crops in the Central Sands region.

Studies have found deficit irrigation to be effective at reducing water use while maintaining crop yield if targeted during stages of crop production when the crops are tolerant of stress. Others have shown that lowering crop evapotranspiration causes reductions in yield. In Delaware, watermelon yields were not significantly affected by reductions in irrigated water indicating their adaptability to low water conditions and adaptability to variable rainfall conditions (McCann et al., 2007). Deficit irrigation increased water productivity in potatoes by 5% in coarse sand and 36% in sandy loam relative to full irrigation practices in Denmark (Ahmadi et al., 2010). However, these methods decreased water productivity in loamy sand by 15% and 13%, respectively. Results indicate that loamy sand produces the highest yield under non-limited water resources conditions, while sandy loam and course sand produce higher yields under water-saving irrigation regimes (Ahmadi et al., 2010).

Drip Irrigation

Drip irrigation is another technological adaptation that can increase WUE through localized application and flexible scheduling (Patel and Rajput, 2007). Drip irrigation provides improved water application efficiency by applying water to a specific location and depth without wetting foliage (Mmolawa and Or, 2000). In potato fields in Maine, drip irrigation was more efficient in applying water to the center of a potato ridge and resulted in less runoff flowing into furrows (Starr et al., 2008). Unlike transpiration, which facilitates photosynthesis, plant biomass accumulation, and crop productivity, evaporation of water from the soil surface has little benefit to crop productivity. Mechanisms or practices that reduce evaporation have potential to reduce evapotranspiration on agricultural landscapes with little to no influence on crop productivity.

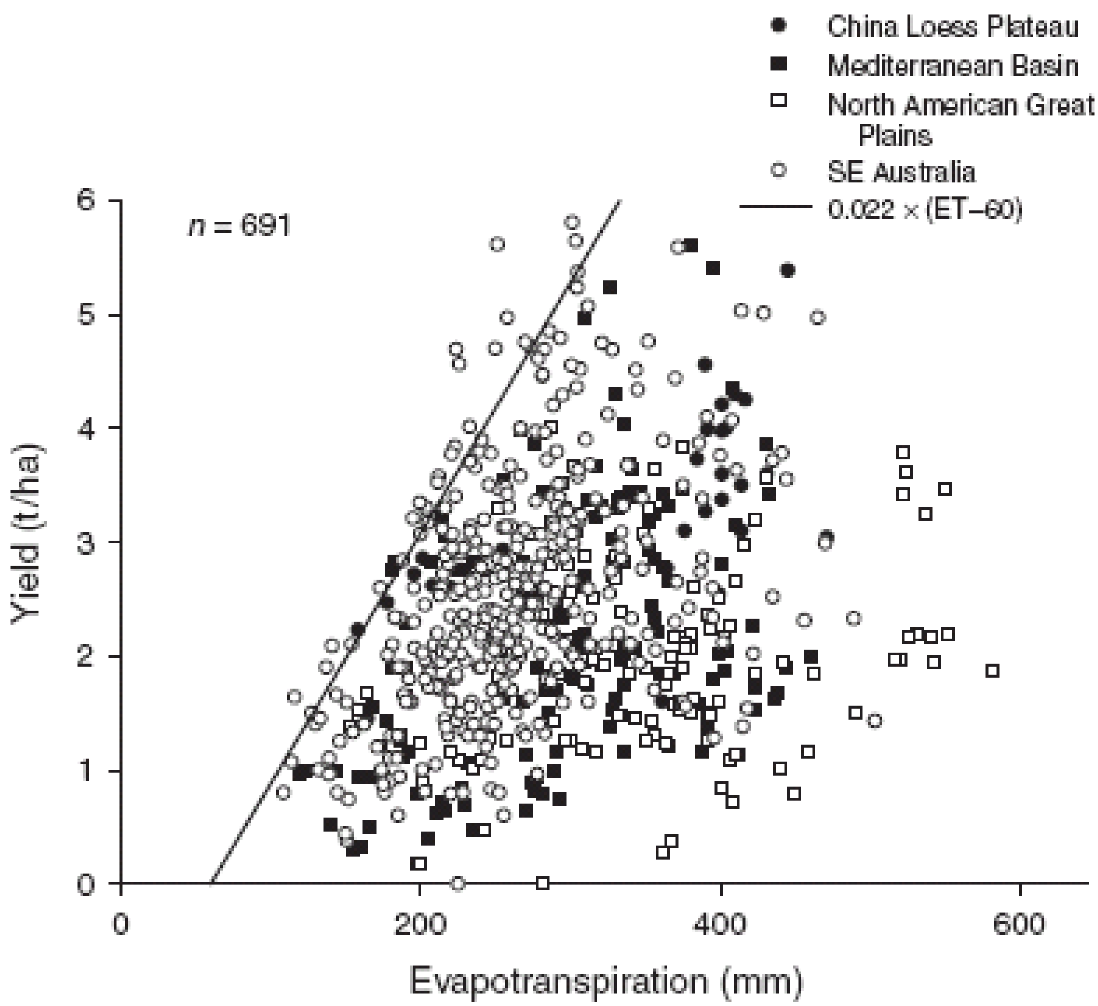

Figure 37. Scatter plot of grain yield and seasonal evapotranspiration in four dry environments. The line marks the water use efficiency (WUE) "benchmark function" (Sadras and Angus 2006).

I.14.2 Landscape-Level Water Resource Management

Certain strategies may be most effect if implemented on a landscape-level scale. This section describes the three following landscape-level water resource management strategies: drainage ditch management, low-evapotranspiration crop systems and comprehensive irrigation management.

Ditch Management

Recent drainage ditch management strategies offer a way to increase WUE and decrease groundwater extraction by seasonally managing groundwater that otherwise would flow out of the system as a source of water for crops (Allred et al., 2003; Banedjschafie et al., 2008). By managing ditch flow such that water table depths are sufficient for farming, yet within a level that crops can draw from it as a form of irrigation, drainage ditch management aims to reduce groundwater extraction (Allred et al., 2003; Gavin, 2003).

Agricultural drainage ditch management is employed in the Maumee River Basin in northwestern Ohio (Allred et al., 2003). Drainage ditches are composed of underground drainage pipes connected to a constructed wetland and water storage reservoir (Allred et al., 2003). The system captures runoff and subsurface drainage in a constructed wetland, which naturally filters the water and routes it to a storage reservoir where it can drain or add water to the crop root zone depending on the desired groundwater management plan. Maumee River Basin drainage management has increased crop yields for unirrigated corn and soybeans by 34.5 and 38.1% during dry growing seasons and 14.4 and 9.7% during near average to wetter growing seasons, respectively (Allred et al., 2003). Studies in other areas have shown that ditch management has increased yields compared to conventional drainage for navy beans, sugar beets and corn yields and promoted higher water levels for ecological purposes (Cooper et al., 1999, 1991; Fisher et al., 1999; LeCureux et al., 1995; Parsons et al., 1990).

Low-Evapotranspiration Crop Systems

With an understanding of system-specific rates of evapotranspiration, vegetation systems can be strategically planted to increase regional rates of recharge. A growing opportunity for improving WUE is maintenance of yield with crop varieties and hybrids that contain shorter growing seasons. Current studies in the Central Sands are investigating the growing season length of variety of potatoes, snap beans, and sweet corn to select for varieties and hybrids with fewer growing degree day requirements to reduce water use (e.g. Russet Norkotah and Russet Burbank). Additionally, researchers are investigating the feasibility of engineering low-evapotranspiration cropping systems (Hu et al., 2006). Targeted mechanisms for decreasing crop evapotranspiration include reducing the number of field days until harvest, decreasing the number of plant stomata and increasing stomatal efficiency (Farooq et al., 2009). Species of canola (Wan et al., 2009), rice (Hu et al., 2006; Xiao et al., 2007, 2009) and maize (Castiglioni, 2008; Nelson, 2007; Wang et al., 2008) have been successfully genetically engineered.

Comprehensive Irrigation Management

The inherent trade-offs between different land uses can be managed by setting priority areas for agriculture and ecosystem conservation. In Walla Walla, Washington, a watershed council worked in collaboration with stakeholders and state and federal agencies to develop a coordinated, performance-based irrigation management plan for designing solutions at a watershed or sub-basin scale (Ruckelshaus Center, 2007). This voluntary, business planning approach implemented on-farm irrigation efficiency projects, piping and lining irrigation ditches, water metering, riparian buffer enhancements for fish, in stream habitat improvements, and upland conservation protection by converting conventionally tilled cropland to native vegetation m (Martin and Siemann, 2007). As of 2007, at least 17 planning and assessment efforts had been conducted in the basin aiming to improve ecosystem and agricultural health and efficiency (Martin and Siemann, 2007).

Similarly, in the Donauried, Germany, a non-governmental committee initiated a voluntary compromise program that prioritized agriculture in 50% of the area, drinking water in 34% of the area and ecosystems in 16% of the area (Haakh, 2002). Farmers determined how priority agricultural land could be distributed around key aquatic ecosystems to support the water resources necessary for their survival (Haakh, 2002). Surface water systems are sacrificed in the agricultural priority areas and agricultural production is limited in the drinking water and ecosystem priority areas.

I.14.3 Water Transfer

Water transfer is the process of mechanically supplementing water from one location to another to support groundwater storage, irrigation capacities or water bodies in high-valued areas. Water transfer strategies are separated into regional and local water transfer efforts.

Regional Water Transfer

Water can be supplemented from less-valued areas within or outside the Central Sands to support high-valued agricultural, industrial, recreational and/or ecological activity within the Central Sands that are supported by groundwater storage, irrigation capacities or water bodies. This can be achieved by diverting the course of a river via construction of a large canal or pipeline that can transport available water (Yan et al., 2012). Water transfer projects have occurred throughout the United States, particularly in the southwest. Projects include the Colorado River Aqueduct, which uses Colorado River to supply water to the city of Los Angeles; the California State Water Project, which transfers water from northern to southern California; and the Texas Water Project, which transfers water from the humid eastern region of Texas to the arid and semi-arid western regions. Similarly, in effort to supply water to the North China Plain, an area that contains high-density irrigated agriculture representing two-fifth of the country's farmland and municipal pumping supporting one third of the country's population, the South-North Water Transfer Project is currently being designed to supply water from the Yangtze River in the wet southern region to the dry northern region (Gleick et al., 2009; Yan et al., 2012).

Local Flow and Water Level Augmentation

Similarly, surface water and groundwater can be used to locally supplement water levels in lakes and streams of particular value to communities. Water level augmentation was applied to increase water levels in Cedar Lake, Wisconsin (McLeod, 1980). Groundwater was pumped at a rate of 2 m^3/min from the local surficial aquifer. The volume of water pumped was sufficient to raise the stage 119 cm, or 41% of the total lake volume at a normal lake stage. Lake stage measurements made by U.S. Geological Survey showed that 90% of water pumped was lost from the lake as seepage or recycled to the well (McLeod, 1980). Due to the high permeability of the stratigraphy of the glaciated area, the method failed to be a solution for water resources management in this lake (McLeod, 1980).

I.15 Strategy Evaluation and Selection

Many methods to evaluate alternative water resource management strategies exist. Methods involved constructing models to test the potential impact, uncertainty and feasibility of management strategies prior to implementation. Such methods often couple groundwater flow models with vadose zone, econometric crop-water productivity management, ecosystem or optimization models to test alternative strategies and assess risks (Aeschbach-Hertig and Gleeson, 2012). Other studies incorporate game theory (Ganji et al., 2008; Shirangi et al., 2008) to determine the most acceptable water allocations scenarios.

The collaborative groundwater modeling effort between the Wisconsin Geological and Natural History Survey and the U.S. Geological Survey to construct a groundwater model for area around the Little Plover River will be an example of groundwater flow model coupled to an optimization model that will allow for the testing of alternative management solutions. This model aims to incorporate stakeholder-identified value optimization and facilitate a search for feasible and effective solutions. By focusing on a specific geographic area within the Central Sands and using the physical science to inform the process, this collaborative computer modeling project may be an opportunity to determine whether or not a disciplined planning, stakeholder collaborative, computer modeling effort is a feasible approach for developing and implementing water resource management in the Central Sands.

I.16 Strategy Implementation

The strategy implementation process can be economically-driven, voluntary/cooperative, regulated by the government or include a combination of implementation methods. Implementation that is economically-based is often controlled via incentives, pricing or taxes. Business reporting can also be used to promote water-efficiency amongst high-user groups (Gleick *et al.*, 2009; "The World's Water"). Additionally, regional or local voluntary self-governance, or “bottom-up”, approaches are becoming more common. In this case, management goals are often based on a common stakeholder understanding about the limitation of the common-pool resource at stake. Voluntary governance strategies have been implemented in Texas and India where farmers are managing water resources and leading the process of data collection as part of a voluntary and collaborative project (Gleeson et al., 2012). Another implementation strategy is to establish locally-governed groundwater conservation districts. For more information on groundwater conservation district development and implementation, see Johnson et al. 2011. Many government-regulated water resource management strategies exist. However, a discussion of these strategies is not within the scope of this paper.

Due to the diffuse nature of water, solution implementation is often not within the power of any one entity (Palmer et al., 2013). The most successful water resource management projects often incorporate diverse water management strategies and diverse strategy implementation methods (Martin and Siemann, 2007).

I.17 Exercise, Update and Use the Plan

A groundwater management plan developed through a shared vision planning process is not a static document. Rather, it is an opportunity for the exercise, adaption and use of a dynamic and integrated plan. The plan will help document historic, ongoing and future water resource management metrics, assessments, implementation successes, failures and lessons learned. This process requires adaptive management. For further information about adaptive management and environmental assessments, see Holling *et al.* (1978) and Stankey *et al.* (2005).

Conclusion

This document aimed to provide a common framework and language for current and future dialogue regarding the state of water resource science and future management for the Central Sands of Wisconsin by reviewing existing interdisciplinary published literature related to hydro-, eco-, and agricultural systems affecting water resources. A review of literature showed that the sandy soils deposited by glacial and eolian processes created an unconfined, unlithified regional aquifer that is in close connection with surface bodies. This groundwater-surface water interaction accentuates the connectivity of socio-, hydro-, eco-, and agricultural systems. The region's land and water resources that supported the hunting and gathering way of life of the Native people on the prairie and oak savannah landscapes in the Central Sands did not provide a reliable and economically stable livelihood for European settlers until the technological advancement of irrigated agriculture.

Reviewed research showed that surface water resources and groundwater levels in parts of the Central Sands have declined. However, due to the paucity of long-term datasets, these conclusions were difficult to quantitatively assess without the use of groundwater flow models. Regional long-term trends temperatures have increased throughout the season. Regional long-term trends in precipitation increased in the southern portion in the Centrals Sands in all studies. The long-term precipitation trends in northern portion of the Central Sands increased in one study and showed no long-term trend in another study. Regional long-term trends in evapotranspiration increased, though not as great as trends in precipitation indicating that climate variables may have driven long-term, regional net increases in groundwater recharge. No studies have shown that climate-driven variables have caused long-term declines in recharge throughout the region.

To date, studies show that the discrepancy between observed surface and groundwater declines and the increase in recharge due to climate-driven variables can be explained as the result of the influence of irrigated agriculture. Irrigated agriculture was found to contain, on average, higher rates of evapotranspiration than the original vegetation types that existed in the majority of the region's area (e.g. prairies, wetlands and open forest systems). Phreatophytes found along wooded, riparian areas with shallow water tables were estimated to have higher rates of evapotranspiration than irrigation agriculture. However, phreatophytes are present over a relatively small portion of the landscape. The increase in evapotranspiration due to the expansion of irrigated agriculture was shown to decrease the total water entering the system as recharge. Additionally, irrigation was found to have a greater impact on surface water bodies during dry periods, such as during summer months and/or droughts.

Drainage ditches are also present throughout approximately 30% of the landscape. Drainage ditches serve to lower water tables on a field-scale. Many drainage ditches are former streams that have been deepened and channelized. The total impact of drainage ditches on the regional water budget has not been documented in literature.

Given current policies in Wisconsin and past stakeholder engagement efforts, shared-vision planning could provide a path forward for the Central Sands community. Shared-vision planning involves creating a strategic framework, engaging stakeholders and conducting collaborative modeling efforts. A comprehensive modeling effort useful for water resource management in the Central Sands would couple groundwater flow, unsaturated zone, economic and ecosystem valuation models. Ongoing collection of water resource and land management data, incorporation of transience into models, and establishment of ecosystem valuation processes relevant to local stakeholders would benefit the shared-vision planning and implementation process.

The shared-vision planning process presents a philosophical challenge and opportunity, namely, to integrate research questions and methodologies of multiple disciplines. The review of interdisciplinary research highlighted the different focus of agronomic and hydrogeological research questions. Agronomy research questions focus on a given plant and/or plant-atmospheric interaction. The control volume for agronomic research questions is the atmospheric hydrological budget and leads agronomists to ask questions about how Central Sands evapotranspiration rates vary between crop types over time. Hydrogeological research questions focus on a given aquifer system. The control volume for hydrogeological research questions is the terrestrial hydrological budget and leads hydrogeologists to ask questions about how Central Sands recharge rates change over time. These differing approaches to research allow each discipline to address different aspects of the hydrological system. Both are critical aspects in the study of water resources management. Future research questions will require integration of atmospheric and terrestrial hydrological budgets to investigate water resource management.

The portfolio of strategies provided in this document could be used for future conversations on water resources management. Note that the strategies reviewed in this document are not comprehensive, nor mutually exclusive, but aimed at encouraging ongoing discussions on managing Central Sands water resources. We recommend focusing on actions that ultimately lead to implementation of solutions on farms and within communities across the landscape. Prioritization could be given to strategies with the greatest potential for success, largest impact on water body protection, and greatest likelihood of implementation. We also recommend focusing and targeting future research on strategies with the greatest priority.

Water resource management success will ultimately be defined by the region's ability to meet the three stakeholder-identified goals for water resource management listed below:

- Maintain healthy waters and ecological resources in the Central Sands region during future water development;
- Restore healthy waterways in the Central Sands region that have been compromised;
- Promote and maintain a vibrant agriculture industry (Nowak, 2011).

To achieve these goals, the process of establishing a mechanism for implementation of priority strategies that demonstrate the greatest potential for success will be important. Next steps may include:

- Identification of short- and long-term funding that allows for research on potential strategies including modeling and ground-truth efforts
- Priority ranking of management issues and specific areas to be addressed
- Implementation of strategies (and secured funding for implementation of strategies) designated as high priority on a small scale
- Broad implementation of strategies that have been successful on a small scale in other regions and are relevant to the characteristics of the Central Sands system.

We hope that this document will be useful in providing a context for future strategy development and implementation of water resources management in the Central Sands region of Wisconsin.

Appendix I: Definitions

Eolian: Wind-blown.

Effective Porosity: The volume of interconnected pore space per total sample volume. See Bradbury et al. (1992) for further information.

Environmental Flows: Water that is left in a river system, or released into it, for the specific purpose of managing the condition of that ecosystem.

Evapotranspiration: The combination of transpiration from vegetation and evaporation from vegetative and soil surfaces linking land, vegetation, and atmospheric interactions.

Headwater Streams: The uppermost streams in a stream network furthest from the stream's endpoint of confluence with another stream.

Hydraulic Conductivity: Rate of groundwater flow per unit area under a unit hydraulic gradient.

Littoral: Near-shore.

Specific yield: Dimensionless ratio of volume of water drainage by gravity from saturated aquifer material to the total volume of that material.

Steady state groundwater flow model: [adapted from the U.S. Geological Survey website] The magnitude and direction of groundwater flow is constant with time and throughout the entire domain. This does not mean that no movement of groundwater occurs, just that the amount of water within the domain remains the same, and amount of water flows into the system is the same as the amount of water that flows out. Time is not an independent variable and the governing equation does not contain a storage term.

Transient groundwater flow model: [adapted from the U.S. Geological Survey website] The magnitude and direction of groundwater flow changes with time. Time is an independent variable and the governing equation contains a storage term which adds more complexity to the model compared to a steady-state model.

Storage coefficient (storativity): The volume of water released from storage per unit decline in head in the aquifer per unit area of the aquifer.

Transmissivity: Rate of groundwater flow per unit width under a unit hydraulic gradient.

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