Lake Wissota

Diagnostic and Feasibility Analysis

July 16, 1996
sections 1- 1 2 submitted to WI DNR

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Lake Wissota:
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1.0 Lake identification and location

Lake name: Lake Wissota
State: Wisconsin
County: Chippewa
Nearest municipalities: Located within Eau Claire - Chippewa Falls Metropolitan Statistical Area.
Latitude and Longitude of lake center: 44 Degs. 57' N, 9l Degs. 19' 45" W
EPA Region: Region V
EPA Major Basin Name: Mississippi Code: 2
EPA Minor Basin Name: Chippewa River Code:
Major tributaries: Chippewa River
Receiving water body for lake discharge: Chippewa River
Approved State water quality standards for Lake Wissota:
Designated Use: (to be supplied by WI DNR; requested July 1996)

Applicable Criteria: (to be supplied by WI DNR; requested July 1996)

2.0. Geology of the Lake Wissota Watershed

2. 1. Geologic and ecoregion description

Lake Wissota is a 2550 ha impoundment of the Chippewa River. It also receives inflow from the Yellow River, which enters the main basin via an embayment called Moon Bay, and Paint and Stillson Creeks, which flow into Little Lake Wissota. The combined drainage basins of Yellow River and Paint/Stillson Creeks cover roughly 74,000 ha over a large portion of Chippewa County.

Lake Wissota is located in the North Central Hardwood Forests ecoregion to the north of the Driftless Area ecoregion (Omemick and Gallant 1988). Portions of the watershed extend into the Northern Lakes and Forests ecoregion. The vegetation is transitional between the predominantly forested area to the north and the agricultural regions to the south, including various patches with many different types of vegetation. Dairy farming is typical within the ecoregion, covering roughly one third of the area. Agricultural activities occur throughout the ecoregion and occur to a lesser degree in upstream areas blending into the

Northern Lakes and Forests ecoregion. Dairy farming is the most important affect on stream water quality, with associated impacts on stream bank stability and stream water chemistry by manure runoff, erosion of soils and runoff of agricultural chemicals (Omemick et al. 1988).

The original vegetation in Wisconsin has been reconstructed from U.S. General Land Office notes. These records suggest the area immediately surrounding Lake Wissota was dominated by scrub oak forests and barrens. Oak forests were found along the lower lower Yellow River watershed, while white and pines occurred along the Paint Creek and Drywood Creek drainages. The Upper Yellow River watershed and the upper portions of the Drywood Creek drainage were located in a band of sugar maple, yellow birch, white and red pines. Some pockets of more diverse forest cover, which also included beech and hemlock, were also found. Much of the Upper Chippewa drainage basin was originally covered by hemlock, sugar maple, yellow birch, white pine and red pine. The southern limit for pines being a common element in forest communities occurs to the south of the city of Eau Claire, suggesting that pines would have been expected to occur throughout the watershed, especially on mesic sites (Finley 1976).

The Lower Yellow River and Paint Creek basins covered in this study represent 78% of the combined drainage area contributory to Moon Bay and Little Lake Wissota. They are located on a gently rolling till plan underlain by Cambrian sandstone or Precambrian granite and gneiss. Drainage patterns are poorly defined and the permeability of the soils is moderate to poor.

The Stillson Creek basin is formed on an outwash plain underlain by mostly sandy deposits weathered from sandstone. Permeability of these soils is moderate to rapid

Lake Wissota occupies a meander basin of the Chippewa River. The geology of the area is very complex and interesting. In order to appreciate the character of the lake, a more detailed understanding of both the bedrock and the glacial geology of the area is helpful.

Bedrock Geology

The bedrock for most of the area surrounding Lake Wissota is Cambrian sandstone of the Mt. Simon and Eau Claire formations, with the Mt. Simon predominating. This formation is the oldest of the Paleozoic sediments, deposited roughly 550 million years ago. The Eau Claire formation is present only in the higher elevation upland areas at the periphery of the watershed area. These Paleozoic sediments are fairly thin, ranging from a few meters up to around 100 m in thickness beneath the higher hills of the watershed, and are often absent along the major river channel where the rivers have cut down through the Cambrian sandstone to Precambrian bedrock. The Precambrian bedrock is very complex, comprised mostly of metamorphic rocks such as gneisses and amphibolites. These rocks were probably volcanic in origin prior to metamorphism occurring around 1.9 billion years ago. The wide variety of different metamorphic rock types present reflect varying degrees of metamorphism that occurred in the area. Igneous rocks are present mostly as features such as dikes and veins that intruded the host rock during subsequent tectonic events. This period represents a time when the crust was being displaced and the area around Lake Wissota, which is exposed well in the area of the dam, is a failed mid-continent rift that would have made the area a coastal zone.

A brief summary of the structural history would begin with metamorphism of the original rock. Following this metamorphic event were several intrusive events where different igneous rocks such as tonalites and trondhjemites were emplaced into the host rock as dikes and veins. Strike-slip faulting which followed allowed for more veinlets to develop and this was followed by intrusion of granitic pegmatite dikes which were associated with thermal contraction and/or crustal tension. Later, around 1. 1 billion years ago, these rocks were intruded by many gabbro-diabase dikes, which were subsequently deformed by faulting. This deformation is expressed by a high degree of jointing or fracturing in the rock.

Glacial Geology

The bedrock of this most of this area is generally covered by glacial deposits, with a key exception being the area around the dam. Within the extensive lowlands that surround the lake, these glacial sediments consist primarily of glacial outwash which were produced during the last episode of glaciation known as the Wisconsin period, around 15,000 years ago. These sediments of these outwash plains are generally very sandy and therefore have a moderate to high permeability. Within the upland areas that surround these outwash plains, the bedrock is mantled by glacial till rather than outwash. The majority of these glacial sediments originated from a glacial episode much earlier than the Wisconsin period, but was not buried by the glacial outwash due to its higher elevation. The permeability of the glacial fill ranges from moderate to low permeability.

Geomorphic History of the Area

The basin occupied by Lake Wissota has a fascinating origin. The present day impoundment occupies a large meander basin formed by the Chippewa River. This large basin originated when the downcutting Chippewa River encountered hard Precambrian rock at the site of the Wissota Darn. This resistant rock prevented the river from meandering in that area and formed what is known as a constriction in the river valley. However, upstream from the dam, the river had not cut down into the Precambrian rock yet and was free to meander, and with its meandering re-excavated a broad preglacial valley.

The geomorphic history of the Lower Yellow River is somewhat similar to that of the Chippewa River. At the site where the Yellow River enters Lake Wissota, it had cut down to the Precambrian bedrock and formed a narrow dells. Upstream from the dells the river was free to meander in less resistant glacial deposits, and created a wide valley which is occupied by the present day Moon Bay.

The geomorphic orgin of Little Lake Wissota is similar to the lower portions of the Yellow River. Close to Little Lake Wissota, Paint Creek had cut down to the more resistant Precambrian bedrock and created a narrow valley, which prevented the river from shifting. Upstream from this constricted valley was a preglacial valley filled with glacial drift in which the river was able to re-excavate, forming a basin that today is occupied by Little Lake Wissota.

3.0. Public access to the lake

The public has excellent access to the Lake Wissota. The lake has eleven boat launches, including three major public sites. One of those sites is associated with Lake Wissota State Park. All boat access points are shown on the new bathymetric map of Lake Wissota developed for this project (see Figures 12-15 and Appendix B).

Lake Wissota State Park was created in 1961 to provide increased access to the lake. The park is located along the shoreline of the lake and it has one large campground and two group tent areas. One of the hiking trails courses the shoreline of the lake and the public beach area is over 300 feet long. The State Park has approximately 1 mile of lake frontage.

4.0. Lake Wissota user population

4.1 Potential user population

Lake Wissota is contained within the Eau Claire - Chippewa Falls Metropolitan Statistical Area. This area contains approximately 140,000 residents and is ranked by population size as the 237th in the nation (REIS 1994). The population of Chippewa County grew more than 5% each decade from 1940 to 1980, and increased slightly between 1980 and 1990. It is expected to increase at a rate of approximately 5% through 2020 (Chippewa County Population and Economic Profile, 1994). In addition to its location within the Metropolitan Statistical Area, the lake is located only 4 miles east of the city of Chippewa Falls,

and is used extensively by its residents.

4.2 Economic considerations of the Lake Wissota region

Agricultural production is an important characteristic of the region. This production is primarily related to dairy cattle. However, equipment manufacturing accounts for almost 20% of Chippewa County's output, in large part because of Cray Research (Chippewa Valley Center for Economic Research and Development 1995). Manufacturing companies comprise the largest portion of employment in the County, at 30.5% in 1993 (West Central Wisconsin Regional Planning Commission 1994). The total gross regional product for Chippewa and Eau Claire Counties is $2,312 million 1990 dollars.

In 1992 the area had a per capita personal income of $16,600 or 83% of the national average (REIS 1994). Despite the recent closing of a tire manufacturing facility and a supercomputer firm, the economy of the area continues to expand and diversify. Relocations of several firms to the area and new facilities being built is expected to lead to further development.

4.3 Lake Wissota riparian population

The shoreline of Lake Wissota is extensively developed. There are a considerable number of year round residences and permanent structures in the riparian area around Lake Wissota. Most of the structures are located within 200' of the shoreline of the lake. A total of 1,121 structures were found within 1000' of the shoreline of the lake on USGS 1:24, 000 quadrangle maps. Of these, 216 were within 100' of the shoreline and an additional 467 were located between 100 and 200' of the shoreline (Chippewa County Land Conservation Dept., personal communication from Dan Masterpole, July 16, 1996).

5.0 Historical use and development of Lake Wissota and its watershed

Lake Wissota was created by a darn constructed over the period 1915-191 and closed in 1917. The darn is used for power generation and is presently operated by Northern States Power Corporation. Power is generated by peaking practices, which cause fluctuations in water level of up to a foot per week. In addition, the reservoir is drawn down in late winter to accept snowmelt runoff. FERC relicensing is due in 2000.

Land use in the watershed is predominately to support dairy-based agriculture (Chippewa County Land Conservation Dept. 1986). As a consequence, there are approximately 740 barnyards in the drainage area that influences the two embayments of the lake. Associated with dairy herds are row crops and areas of hay and pasture. Forested regions are found in the watershed mainly in the upper reaches of the Lower Yellow River Basin. While the lake is in the Eau Claire - Chippewa Falls Metropolitan Statistical Area, urban land uses are limited to the Village of Boyd, City of Cadott and urbanizing ares of the Towns of Lafayette an Hallie. The lake is, however, within easy commuting distance of all of these areas

The number of farms is declining. For Chippewa County as a whole, there were 1647 farms in 1987, a decline of 119 from 1982. In addition, the total acreage in farms declined by 13,389 acres over the same period (U.S. Dept. Of Commerce 1994). These patterns are expected to continue in the area, including within the Lake Wissota watershed.

5. 1 Growth in residential development

The Metropolitan Statistical Area continues to grow. For the City of Eau Claire, the rate of growth has slowed somewhat from 1.43% from 1970-80 to 0.98 from 1980-90 (Wisconsin Dept. of Administration 1994). Over the period 1980-89, 11.4% of the housing stock was replaced and approximately 550 new homes are added each year (U. S. Dept. Of Commerce 199 1).

While growth rates in Chippewa County overall slowed between 1980 and 1990, higher rates of increase were found in the townships and municipalities near Lake Wissota. In addition, the employment base expanded considerably over the same period, with an increase of 26% in the County (West Central Wisconsin Regional Planning Commission 1994).

5.2 Recreational use

Lake Wissota is one of the largest and most heavily used recreational resources in western Wisconsin. It is used heavily by riparian residents and by non-residents. As one indication of the intensity of its use, visitation at the State Park was 40,000 per year in 197 1, ten years after the Park opened. It grew to 120,000 by 1972 and is now approximately 200,000 visitors per year. This park is one of only three State recreation sites in Chippewa County and is the closest one to the Metropolitan statistical area.

The lake is used extensively for boating and for fishing during the period from spring through fall. Nearly all of the lakeshore residents own boats. The boat ramps at the lake are used intensively and contribute to the heavy usage of the lake. In addition, the lake is a popular place for ice fishing during the winter months.

The median distance traveled to go boating in Chippewa County is estimated at 10 miles, while the median distance in Eau Claire County is 42.5 miles (Panaloza 1992). These estimates reflect the importance of Lake Wissota as a recreational resource in Chippewa County and the further distance traveled by boaters in Eau Claire County to Lake Wissota and lakes to the north on the Chippewa Moraine.

6.0 Population segments adversely affected by lake degradation

Development on Lake Wissota centers on year-round residences. It also is a significant commercial water-based recreational area for the region. in addition to the State Park, there are many water-based businesses, including supper clubs, as well as sporting goods stores and bait shops that depend on business related to the lake. Property values around the lake are higher than in other areas.

Degradation of the lake would impact heaviest on year-round residents but visitors and area businesses would also be affected directly. Property values around the lake would likely decline if water quality deteriorates further.

Given the extensive usage of Lake Wissota, further deterioration of its water quality would have significant economic impact to the surrounding area, particularly in the City of Chippewa Falls.

7.0 Alternative sites for lake users in the region

Lake Wissota is the largest surface water body in a fairly large area of west-central Wisconsin. There are no other similar sized lakes close to or within the metropolitan area. While there are many natural lakes north of Lake Wissota located on the Chippewa Moraine, which begins in northern Chippewa County, these are typically much smaller and they are further from the metropolitan area. In addition, south of the moraine there are very few lakes, especially in Eau Claire County, except for smaller impoundments and oxbow lakes. One of these bodies of surface water is a shallow impoundment of the Eau Claire River, called Lake Altoona, located east of the City of Eau Claire, that experiences rapid sedimentation. None of these sites provides the degree of public access or the range of opportunities for recreation that are found at Lake Wissota. As a consequence, further degradation of water quality in Lake Wissota would mean considerably longer travel distances for similar recreational activities.

8.0 Point source discharge into Lake Wissota

Currently there are no industrial or municipal point source discharges directly into Lake Wissota. The City of Cadott sewage treatment plant discharges into the Lower Yellow River, which flows into Moon Bay. There are multiple upstrewn discharges into the Flambeau and Chippewa Rivers throughout the Chippewa River Basin (WT DNR 1996 - Upper Chippewa River Basin and Lower Chippewa River Basin Water Quality Management Plans). These discharges contribute to the relatively high nutrient loading into the main basin of the lake.

9.0 Lake Wissota Watershed Land Use and Lake Assessment

A comprehensive diagnostic and feasibility analysis has been conducted for Lake Wissota. This study was designed to assess the current conditions in the lake and the possibility that improvements in water quality by improvements in land use within the watershed. The study was a cooperative effort among the Chippewa County Land Conservation Department, the Wisconsin Department of Natural Resources, Winona State University and the University of Wisconsin - Eau Claire.

Lake Wissota covers 6212 acres and it is the largest body of surface water in west-Central Wisconsin. The lake was formed as a hydroelectric impoundment of the Chippewa River, which is one of Wisconsin's major drainage systems. The

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name Wissota was derived from the original developers, the Wisconsin (WIS) and Minnesota (SOTA) Power Company. Currently the hydroelectric plant is operated as a peaking facility by Northern States Power Company under a license that expires in the year 2000. The main lake is roughly 4 miles long and two miles wide, and it has two embayments extending into stream channels. The Wissota project, built beginning in 1915 and closed in 1917, raised water levels 57 feet.

The lake is used heavily for recreation. A large State park is located on the lake, along with four smaller parks, three private campgrounds, five lakeside resorts, a Rod and Gun club with facilities for swimming and boating, several restaurants and a motel. In addition, much of the lakeshore is lined with private residences.

Several issues have emerged related to the operation of the lake and its current condition. First, the lake contains relatively high concentrations of algal nutrients and often develops blooms of blue-green bacteria during the summer. Given the very large watershed of the lake and the extent of agricultural land use, one issue in maintaining or improving water quality is whetlier sources of nutrients within the watershed might be controlled in the two embayments associated with the Yellow River basin and the Paint Creek basin. Second, given the hydrologic setting of Lake Wissota, the reservoir is drawn down in late winter in advance of the major period of snowmelt runoff. A second, major issue is therefore whether the operation of the reservoir impacts fish and other organisms in the lake.

In order to address these issues, this project conducted a number of analyses:

1. categorizing and mapping land use in the watershed and estimating watershed runoff of algal nutrients;
2. developing a hydrologic budget for the lake;
3. developing a new bathymetric map for the lake;
4. evaluating potential strewn runoff of fecal coliform bacteria, especially as associated with storm events;
5. assessing current water quality;
6. determining the impact of reservoir operation on invertebrate populations and fish stranding; and
7. estimating the improvements in water quality that might result from best management practices in the watershed and other controls to reduce nutrient concentrations in runoff.

9.1 Watershed land use assessment

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A major conclusion from in-lake sampling of Lake Wissota in 1989 was that water quality conditions in two embayments appeared to be worse than in the main basin of the lake (WI DNR 1993), which was assumed to be related to watershed non-point source loading received to the embayments during a low flow year. Those embayments were known to receive discharge during storm events that contain high concentrations of suspended sediment (see Figure 1). As a result an analysis of watershed land use and nutrient runoff was undertaken for the

Lower Yellow River and Paint Creek watersheds. The locations of Lake Wissota and the watersheds for the two embayments are shown in Figure 2.

Because the primary water quality concern for the Wisconsin Department of Natural Resources (WI DNR) focused on the two embayments, the very large watershed of the Chippewa River is not shown in Figure 2. The entire watershed area was included in the hydrologic budget but it was beyond the scope of the land use assessment conducted as part of this study. This project involved the Lower Yellow River and Paint Creek watersheds, which were divided into sub-watersheds for further analysis (Figure 3).

The analysis of land use in the Lake Wissota watershed was conducted by the Chippewa County Land Conservation Department based on a single seven band Landsat Thematic Mapper (TM) image from June 11, 1992 (see detailed report in Appendix A). Earth Resource Data Analysis (ERDAS) Software was used to classify the raw satellite data into eight land cover classes. The classification recognized:

1) water;
2) wetlands;
3) coniferous forest;
4) deciduous forest;
5) hay/pasture;
6) row crops;
7) heavy urban; and
8) light urban.

Accuracy of the classification was assessed by checking 180 randomly selected points through analysis of USGS Tiger files and interpretation of aerial photographs taken in June and July 1992. This assessment suggested that the accuracy of classification was 85%. Limitations were found in trying to separate hay from permanent grassland and oats from all other classes. The distribution of classified land cover types within the Lower Yellow River and Paint Creek basins is shown in Figure 4.

Both the Lower Yellow River and Paint Creek basins are dominated by agricultural land use, primarily grass, hay and pasture (Figure 5). Rowcrops account for 9% of the area in the Lower Yellow River basin and 11% in the Paint Creek basin. Approximately 40% of the Yellow River watershed is forested and 32% of the Paint Creek watershed is forested. Land cover within the sub-watersheds for each basin is shown in Table 1.

Phosphorus and sediment delivery from upland landuse was estimated by assigning phosphorus delivery and sediment delivery coefficients to the areas of each of the land cover classes. Phosphorus delivery coefficients were selected based on soil types (US EPA, 1990). Sediment delivery coefficients were based on a relationship developed for Duncan Creek, an adjacent watershed (Chippewa County Land Conservation Department, Duncan Creek Watershed Plan, 1991). The phosphorus delivery coefficients used in making these estimates were 1.0 kg/ha-yr for row crops, 0.30 kg/ha-yr for dairy-based grassland/pasture/hay and 0.05 kg/ha-yr for forested areas (values reviewed by Sorge 1995, Panuska 1995, personal communcation to Dan Masterpole).

Estimated rates of phosphorus and sediment delivery associated with upland land use in each of the sub-watershed s are shown in Table 2. The total export of phosphorus was estimated to be __________ kg/yr for the Yellow River basin and 45,000 kg/yr for the Paint Creek basin. The delivery of phosphorus from upland areas in the Yellow River basin produces more phosphorus and the export is higher for some of its subwatersheds in than for the Paint Creek basin (Figure 6). Similar patterns of sediment export were predicted (Figure 7).

The number of barnyards was estimated for each of the watersheds. There are 550 active barnyards in the Lower Yellow River basin and an additional 186 barnyards in the Paint Creek and Stillson Creek basins. Phosphorus delivery associated with the barnyards was estimated from the distance to intermittent or perennial streams and transfer coefficients were based on a comprehensive analysis of barnyards in the nearby Duncan Creek watershed (Chippewa County Land Conservafion, 1992). A total of 8085 kg P/yr was contributed by barnyards in the Lower Yellow River basin and 2734 kg P/yr in the Paint Creek basin (Table 3).

The streambank erosion analysis was based on physical characteristics of streams in the watershed, especially the number of perennial stream segments, the cumulative drainage area, the average length of stream segments, the sinuosity ratio and the estimated rate of streambank erosion. Contributions from each of the sub-basins is given in Table 4 and Figure 9.

High phosphorus and sediment delivery was predicted to occur in several sub-basins and where one was high the other was usually correspondingly high. The areal distributions of phosphorus and sediment delivery from all sources are shown in Figures 10 and 11. These figures indicate a number of priority subbasins where loading is high and management practices should be considered, including the Drywood Creek drainages, Middle Yellow, Middle Paint and Stillson. P loads for all sources are estimated to be quite high in each of these areas.

Estimated upland phosphorus delivery for subbasins within the Lower Yellow River, Paint Creek and Stillson Creek basins were lower than the P export coefficients suggested as most likely by Panuska and Lillie (1995). Estimated P export for a watershed with >50% agricultural activities was 0.56 kg/ha-yr, while most export rates estimated for subbasins were near 0.2 - 0.3 kg/ha-yr, with a maximum of 0.43 kg/ha-yr. These estimates do not include barnyard runoff within the basin. If runoff from barnyards is included, the increase would be 0.14 kg/ha-yr for the Yellow Rivery watershed and 0. 17 kg/ha-yr for the Paint Creek watershed, based on data in Tables 6-8 in Appendix A.

Most of the phosphorus being delivered to strewn networks was estimated to be associated with agricultural activities. Roughly 56% of the phosphorus load came from hay, grass and pasture (Figure 12). The second most important source was associated with barnyards, which co tributed 21-23% of the total. Although rowcrops represented only 9-11% of the area of the watersheds, they contributed 18-21% of the estimated phosphorus being exported. Phosphorus yield from streambank erosion was a very small portion of the total. Similar to the patterns of phosphorus export, most of the sediment delivered to streams in the Yellow River basin and in the Paint Creek basin was from agricultural areas, although streambank erosion accounted for 16-22% of the total (Figure 13).

10.0 Analysis of Limnological Conditions

10.1 Lake mapping

A preexisting lake map was developed in 1940 and was known to contain errors. In order to gain better precision for the hydrologic budget and to provide information for recreational users of the lake, a new bathymetric map was prepared for the lake and its major embayments. The map was prepared based on aerial color-infrared photography flown for Northern States Power in 1992 and intensive measurements made in the field.

The aerial photographs were used to locate the shoreline of the lake. The photos were checked by ground-truthing, which involved the location of 27 GPS (global positioning system - Tremble Pathfmder Plus) latitude/longitude control points at an accuracy of 2-5 m. The shoreline of Party Island was plotted by walking the GPS unit around the perimeter and collecting a string of 68 coordinates.

Depth soundings were made on the lake using a Lowrance depth sounder that plots depth on a paper printout. Locations for the soundings were detemiined by GPS. GPS locational data were collected at the beginning and the end of each transect and intermittently along the transects. Particular features or significant structural change in bottom contours, such as dropoffs or former stream channel trenches, were located by GPS. All GPS data were differentially corrected using a Tremble Community Base Station located on the campus of UW - Eau Claire.

The lake survey consisted of 425 transect profiles collected in 1993 and 1994. The location of the transects and the path of the boat were aided by the use of detailed air photos. Daily logs of lake levels were used to adjust depths such that all depths were calculated for a full pool elevation of 897.1 feet. Transects were routed perpendicular to submerged river channels, and non-perpendicular transects were added when there was increased complexity in the bottom features. In addition, for the three riverine appendages, the sequence of transects proceeded downstream towards the main body of the lake.

Shorelines from each of the fourteen color-infrared photos were merged together and proofed with GPS data collected at the endpoints of transects. The latitudes and longitudes were loaded into Atlas GIS, which was used to calculate lake area.

The positioning of depth contours was accomplished with a combinafion of manual and computerized procedures. Transects were plotted using the GPS locafions for the end and intermediate points, and depth measurements were idenfified for each point. Map positions of the contour depths were located on the transects by comparing the distance along the graph to the actual distance on the map and interpolation between the known GPS points. From the known depths and locations, the positioning of contour lines was accomplished by computer and visual interpolation. Computer interpolation algorithms worked in areas of smooth contours, but it was necessary to employ visual interpolation of contours in areas where bottom features were complex, such as in river chaimels. The depth contour information was digitized and areas we-re calculated for each depth and the depth frustums were summed to calculate the volume in each basin and tributary.

The final lake map (Figures 13-16; not attached but final copies are with WT DNR and also appear as Appendix B) was prepared using Adobe Illustrator software. Depth contours were shaded to improve visual perception of the lake structure and the historic river channels. The locations of aquatic vegetation from earlier surveys and fish cribs were added to the final lake map. Once this map was developed, it also was possible to prepare maps assessing the areal impact of drawdown to various levels (Figures 17-20). The areas and volwnes of the reservoir for each area and at various depths are given in Table 5, which can be used to estimate the approximate decline in lake volmne and surface area during periods of drawdown.

10.3 Methods related to in-lake water quality

Water samples were collected and in-lake measurements were made at four sites on Lake Wissota from May - October 1993. Two stafions were located in the main basin of the lake and were influenced by inflows from the Chippewa River, as well as inputs from the Lower Yellow River via Moon Bay and Paint Creek via Little Lake Wissota (Figure 21). Sites 1 and 2 were located in Little Lake Wissota and Moon Bay, respectively.

Temperature and dissolved oxygen profiles were measured for each sampling date at each lake site using a YSI Model 57 Oxygen meter. The oxygen meter was calibrated to air saturation and checked against a modified Winkler dissolved oxygen titration for two samples on each date. Secchi disc measurements were made at the same time.

Water samples were collected with an opaque, PVC Kemmerer water sampler just below the lake's surace on all dates. An additional sample was collected approximately 0.5 m above the lake bottom from June - August. All samples were collected in accordance with standard protocols used by the WT DNR. Samples were placed in prepared Nalgene bottles, kept on ice and in the dark until return to the laboratory. Samples were preserved according to standard protocols, chilled and shipped on ice to the Wisconsin State Laboratory of Hygiene. All analyses were performed in the State Laboratory using approved analytical procedures and quality control measures (Wisconsin State Lab of Hygiene, 1994). Samples were collected on October 5, 1993 and analyzed for major cations and anions using standard methods. Prior sampling using similar collection and analytical methods was accomplished in 1989 (Wisconsin Dept. Of Natural Resources 1993).

Phytoplankton samples were collected using a 1.5 m integrated sampler made from a PVC pipe. The integrated sample represents the algal cells found in the top 1.5 m of the water column. The sample was placed in a 2 L bottle and transported in the dark to the laboratory. The phytoplankon samples were scanned while the sample was fresh and organisms were live to aid in later identifications, particularly of ciliated or flagellated forms. Then, the sample was preserved with Lugol's solution and an aliquot was transferred to a sedimentation chamber. Following sedimentation, the resulting concentrated sample was counted using an inverted microscope. All idenfifications were made by Dr. Lloyd Ohl an experienced phycologist at the University of Wisconsin - Eau Claire, who also converted the algal counts to biovolumes using procedures outlined in Wetzel and Likens (1991).

10.4 Water chemistry

Water chemistry data are summarized by parameter and by site. The data collected for all lake sites suggest significant loading of nutrients to the lake from the watershed of the Chippewa River and significant loading from the Row River and Paint/Stillson Creeks to the two embayments. As is expected for reservoirs in general, based on an analysis by Lille and (1983) for surface waters in Wisconsin and numerous observations elsewhere, nutrient concentrations were generally higher than in the natural lakes located to the north and on the Chippewa Moraine (Omemick 1991, Brakke 1994). Hydrologic type plays an important role in determining lakewater chemistry in Wisconsin (e.g. Eilers et al. 1988) and with their large watersheds relative to natural lakes, reservoirs generally have higher concentrations of nutrients. This is especially true when runoff is from agricultural areas, which typically have higher runoff coefficients for phosphorus (see summary in Panuska and Lille 1995, Cooke et al. 1993).

Alkalinity and pH.

Alkalinity of Lake Wissota and the two embayments was relatively low and ranged from 20 - 40 mg/l for most. This range was expected based on the areas drained by the Chippewa River and the Yellow River watersheds. Relatively little calcareous material is present in the tills of the two drainage basins. The alkalinity of Lake Wissota is higher than seepage lakes located on the Chippewa Moraine (Stovring 1989, Brakke 1994) but similar to drainage lakes in that area (Brakke 1994, 1995). It is well above dilute seepage lakes in Wisconsin (Eilers et al. 1988), but lower than reservoirs located in the Driftless Area.

Lakewater alkalinity was lower in May and increased over the summer and into autumn. This pattern is undoubtedly related to more dilute snowmelt runoff in the spring compared with summer storm flows that have longer contact times with watershed soils.

Lakewater pH generally ranged from 7.2 to 7.6, but approached 8.0 on days when major blooms of blue-green bacteria were occurring.

Calcium and magnesium.

Calcium concentrations were 12 - 15 mg/l and magnesium concentrations were 4-5 mg/l (Figure 22). These concentrations are similar to values found in some drainage lakes in the Chippewa drainage network, but higher than those found in seepage lakes. As expected, calcium and magnesiumn were the most important cations, accounting for most of the alkalinity in the system. Alkalinity is derived mainly from carbonation reactions associated with watershed soils. General chemical composition in terms of the most important cations and anions is consistent with regional patterns and major controls on surface water chemistry in the Upper Midwest (Gorham et al. 1983).

Chloride.

Chloride concentrations were approximately 4 mg/l. These concentrations are within the range for this part of Wisconsin (Lillie and Mason 1983) and do not indicate any major inputs from road salt or other contamination.

Sulfate.

Sulfate concentrations were 6 mg/l, within a range expected for this area of Wisconsin (Lillie and Mason 1983). Sulfate in this area is primarily derived from weathering reactions and oxidation of reduced sulfur associated with wetlands.

Chlorophyll a.

Most chlorophyll concentrations were <25 ug/l, but peak concentrations during blooms were greater than 100 pig/l, including areas of the main basin. These chlorophyll concentrations are associated with a noticeably green color to the water and the higher concentrations occur-red during dense blooms of blue-green bacteria. Chlorophyll concentrations in Lake Wissota are similar to eutrophic, drainage lakes in the region and reached values typical of hypereutrophic lakes and reservoirs (Lillie and Mason l983, Brakke 1992, 1995, Rast and Holland 1988).

Water clarity (Secchi depth).

Secchi disc depths are valuable indications of lake water transparency and also water quality because of the reciprocal interactions between algal nutrients and water column transparency. In reservoirs, Secchi disc transparencies may also be influenced appreciably by storm events that carry large amounts of suspended sediment into the lake. Secchi disc depths were generally very low and only 1.0-1.5 meters for all dates and sites. Secchi disc measurements on the lake are affected by several variables, including the relatively high water color of the lake (to 100 PCU), and were reduced further by suspended inorganic particulates, particularly during mid to late June associated with high runoff, and by algal populations, especially during blooms of blue-green bacteria. The observed Secchi disc values indicate eutrophic to hypereutrophic conditions in the reservoir (Rast and Holland 1988).

10.5 Little Lake Wissota - Site 1 - Paint Creek Inlet

The water column at Site 1 showed only partial stratification on some dates (Figures 23-25), as would be expected for a relatively shallow water column relative to its surface area (cf. Ragotzkie 1978, Gorham and Boyce 1989). No longer term stratification of the 11 meter water column was observed at this site, although some oxygen depletion occur-red when temporary stratification was present. Such transient stratification was also observed in 1989, especially at higher temperatures.

Surface water pH, alkalinity and conductivity generally increased over the summer (Figures 26-29). Measured values were lower on July 1, in response to an apparent dilution effect related to the major storm events occurring in late June. They increased in August even though rainfall during that month was above average.

Water column transparency is strongly affected by water color and also by turbidity associated with storm events, a characteristic of most reservoirs (see Lind 1986) and algal biomass. Water color was highest in July but even at the lowest values there is a distinct brownish tint to the water (Figure 29). The higher water color indicates an increase in dissolved organic carbon, which is most likely lowest during snowmelt runoff but increased in July in response to the added runoff from the watershed and interaction with peat-bearing areas in the landscape.

Even though the water is stained, increasing absorption of infrared radiation, the cross basin length of the lake is sufficient to allow wind-driven mixing of the water column on most dates. For lakes in this region, cross-basin length and water clarity are important variables regulating mixing depths of the water column in lakes (cf. Fee et al. 1996, Brakke and Cahow in prep.).

Turbidity was generally low, peaking in August (Figure 30), while total solids and suspended solids remained relatively constant during the period (Figure 3 1).

Chlorophyll a concentrations varied considerably over the period of sampling. They peaked at over 50 ug/l in August and were generally lower than might have been expected in June and July, especially when compared to a value of 60 ug/l found in July of 1989 (Figure 32).

Total phosphorus concentrations ranged from 50 - 90 ug/l during the period of sampling, while dissolved phosphorus concentrations were a high fraction of the total and ranged from 15 - 55 ug/l, indicating that other nutrients were limiting algal growth and much greater biomass of algae could be generated. Concentrations increased in bottom water samples, particularly in August when there appeared to be longer periods of thermal stratification (Figures 33 and 34). Summer total phosphorus concentrations were in a range anticipated for reservoirs and for lakes in this area of Wisconsin (see Omernick et al. 1988, map region 51-11).

Total nitrogen remained relatively constant during the same period, while nitrate-nitrogen declined slightly over the summer and then increased in the fall and ammonia fluctuated considerably. Ammonia was reduced to very low levels in surface waters in August, while it was accumulating in deeper waters (Figures 35-37). During periods of temporary stratification, the increases m ammonia and in phosphorus suggest the possibility of internal sources of phosphorus due to release from lake sediments.

10.5 Moon Bay - Site 2 - Yellow River inlet

The water column in Moon Bay is even shallower than in Little Lake Wissota. As a result, it stratifies for only brief periods of time. Regardless, during periods at higher water temperatures, stratification can occur (a stable water column can exist at elevated temperatures without an apparent thermocline), and oxygen depletion was observed (Figures 36-38). Surface water pH, alkalinity and conductivity were generally higher in Moon Bay than in Little Lake Wissota (Figures 39-41). On the other hand, water color was of similar magnitude and temporal pattern in the two embayments (Figure 42). Turbidity was slightly higher in Moon Bay than in Little Lake Wissota, as were total and suspended solids (Figures 43-44). While total solids increased over the period of sampling, suspended solids declined, except for an increase in August related to an increase in algal biomass (Figure 45). Chlorophyll a remained well under 25 ug/l until a bloom in late August which produced concentrations near 35 ug/l. These levels were considerably lower than those observed in 1989 (45 and 81 ug/l in July and August respectively) or those measured in the main basin of the lake (Figure 46).

Total phosphorus concentrations were consistently higher in Moon Bay than in Little Lake Wissota, except for one date in August. Concentrations ranged from 75-95 ug/l over the summer months in surface waters and increased in bottom water samples during intervals when the water column was stratified in August (Figure 47). As in the case of Little Lake Wissota, a high fraction of the phosphorus was as dissolved P in surface and especially in deeper water samples (Figure 48). Total nitrogen was higher for most dates in Moon Bay and showed little variation over time, while nitrate-nitrogen increased into July and then decreased prior to the highest concentrations during fall circulation (Figures 49-50). Ammonia-nitrogen was much lower in Moon Bay, but also showed higher concentrations in deeper waters (Figure 51).

10.5 Lake Wissota - Sites 3 and 4 - Main basin

As would be expected for a reservoir with short residence times, the sampling stations in the main basin showed little difference for any of the measured physical. and chemical parameters. This can be seen by the nearly identical pH. measurements at sites 3 and 4 from May to October (Figures 52-53). As a consequence of the similarities between the two sites for all dates and parameters, the data collected from site 4 is presented here as being representative of both sites.

The seasonal patten of pH, alkalinity and conductivity in surface waters reflected that occurring in the two embayments, with the values closer to those observed in Little Lake Wissota (Figures 52-54). The decline in alkalinity from June to July and the increase throughout the rest of the summer, indicates a possible dilution effect resulting from the high rainfall experienced in June. Water color also showed the same seasonal pattern in the main lake and the embayments, suggesting that the watershed of the lake, including the area draining into Moon Bay and Little Lake Wissota yields runoff water having similar influences at least in upstream reaches (Figure 55).

Total solids showed little variation over the period of sampling, while turbidity, chlorophyll a and suspended solids increased in August associated with the development of an algal bloom. Chlorophyll a concentrations exceeded 100 ug/l at site 4 and reached nearly 150 ug/l at site 3 (Figure 56). These values are exceptionally high, and much higher than observed in the two embayments during 1993 or in the main basin of the lake in 1989.

10.6 Comparison between sites

Nutrient concentrations and trophic status

Lake Wissota had relatively high concentrations of total phosphorus (>50 and up to nearly 100 ug/l) at all stations (Figure 57). The highest summer concentrations occurred in the main basin, while concentrations in the two embayments peaked in October. In addition, as mentioned previously, the high total P concentrations were associated with a relatively high fraction as dissolved or available phosphorus. These two factors, the high concentration of total phosphorus and the fraction that is readily available for microbial growth, suggest considerable potential to support the growth and development of algal blooms. The fraction of dissolved phosphorus varied and was reduced during the development of algal blooms, suggesting its rapid uptake when other factors were contributory to growth. This pattern is evident related to the bloom of blue-green bacteria found at site 3 on August 11. As the bloom in the main lake declined, total phosphorus was reduced but the fraction of dissolved phosphorus increased (Figures 58 and 59).

Lake Wissota appears to have sufficient phosphorus to support frequent and dense algal blooms, but at least in 1993, total nitrogen remained at levels that would indicate nitrogen limitation. VWhen total nitrogen concentrations were high, as occur-red on August 11 at Sites 3 and 4 in the main basin, bloom conditions resulted leading to major peaks in chlorophyll (Figures 60-62). These results suggest the lake is particularly sensitive to inputs of nitrogen.

Based on results from a large data set representing a wide range of N and P concentrations, Downing and McCauley (1992) found that nitrogen was most frequently the limiting factor for the growth of algal populations if the TN: TP ratio was low (< 14), particularly when phosphorus concentrations were >30 ug/l. TN:TP for Lake Wissota was frequently <14 (Figure 6-3). The ratio increased and exceeded this level in the main basin during bloom conditions, suggesting that external imputs can trigger bloom conditions, perhaps as a result of rainfall events.

In all of the basins and for many periods, based on the ratio of nitrogen:phosphorus, nitrogen appears to be limiting the growth of algae and phosphorus is present in excess. Given the relatively high concentrations of total phosphorus and the high fraction of available P, summer blooms of blue-green bacteria, many of which can fix nitrogen, will be favored during periods when nitrogen:phosphorus ratios approach 14 or below. The decline in ammonia observed during major blooms is further evidence for this relationship, as is the very low N:P ratios found in August in Moon Bay and Little Lake Wissota. Earlier sampling of the lake from September 1972 to August 1973, by the US EPA (1974) as part of the National Eutrophication Survey, also indicated nitrogen limitation of algal growth based on in lake concentrations and on algal assays.

Trophic state index values (Carlson 1977, Lillie et al. 1993) were calculated based on secchi disc measurements, total phosphorus and chlorophyll a concentrations for 1989 and 1993. Most calculated index values were around 60, but maximum values of near or over 75 were found in 1989 and in 1993. In August 1989, Moon Bay had a higher trophic state index value than did Little Lake Wissota or the main basin of the lake (Figures 64-66). By contrast on August 11, 1993, the trophic state index values were higher in Little Lake Wissota than in Moon Bay and the highest values were found at sites 3 and 4 in the main basin of the lake. On August 23, 1993, the highest values were again found in Moon Bay (Figures 67-68). Bloom conditions and high trophic state index values were found at all of the lake sites in the two years of sampling. These results suggest that current nutrient concentrations are adequate throughout the lake to promote the development of frequent algal blooms in mid-summer, which may vary in occurrence depending on concentrations of total nitrogen and its ratio to phosphorus.

10.7 Algal populations

The seasonal cycle of algal populations varied somewhat among the two embayments and the two sites in the main basin. All of the counts and biovolumes calculated for each site and date are found in Appendix C.

Diatoms represented a considerable portion of the algal population during the spring at all of the sites, except at site 3 in the main basin. Cryptophytes were also important at all of the sites (Figures 69-72). Cryptophytes also dominated the early June sampling at three of the sites, but blue-green bacteria represented 59% of the population in Little Lake Wissota. Heavy rains occurring during late June led to high runoff and likely influenced the composition of the plankton community. At three of the sites, diatoms returned in substantial numbers, whereas site 4 in the main basin was dominated by blue-greens (Figures 73-76). This pattern again demonstrates the variability of the phytoplankton populations from site to site and over time.

By mid-July, blue-green bacteria dominated the populations in Moon Bay and Little Lake Wissota but diatoms represented 84-92% of the population in the main lake (Figures 77-80). In mid- to late-August blue-greens represented nearly all of the algal population in the two embayments as they did at site 4 on August 11 and site 3 on August 23 (Figures 81-85). Once again, there was considerable variability from site to site, but a tendency for the development of blue-green algal blooms at each site during the summer months. Those blooms were most commonly caused by Anabaena, Aphanizomenon, and Coelosphaerium. In autumn, diatoms were dominant at three of the sites but blue-greens were most important in Moon Bay (see Appendix C).

10.8 Metals concentrations

A sample was collected at each in-lake site during fall circulation in October 1993. Concentrations of major cations and anions, as well as iron and manganese are shown in Table 6.

10.9 Fecal coliform concentrations in streams and on beaches

Samples were collected on 11 streams within the watersheds of the two embayments for the analysis of fecal coliform concentrations. The samples were collected on 11 dates in 1993, including three dates following storm events. The intent of this sampling was to use fecal coliform concentrations as a surrogate for potential runoff from barnyards and to identify areas where loading of nutrients may be significant. Using this approach we can compare concentrations among sites and before and after storm events.

Fecal coliform concentrations were generally highest in Lotz Creek and the South Fork of Paint Creek (Figures 86-87), although there was considerable variability from site to site and from date to date. Concentrations of fecal coliforms generally increased at all of the sites and at their highest levels following storm events (Figure 88-89). This increase in response to storms may be as much as an order of magnitude or more following storm events at some sites, but the patterns were not predictable and the increases did not occur on all dates or for all sites. These patterns illustrate the highly non-linear patterns in response of runoff and transport of materials in relationship to rainfall. Note also the substantial differences in scale required to plot data for periods following storm events, suggesting substantial transport of particulate material and runoff from barnyards or areas to which manure has been applied during short time frames and at high flow conditions related to storm events.

The results also suggest that there are several streams within the watersheds of the two embayments that could be delivering significant loading of nutrients to the lake

associated with agricultural activities and dairy cows in particular. South Fork Paint Creek and Lotz Creek yielded particularly high NFCC following storms and tended to have high concentrations during other periods (Figure 90).

Fecal coliform samples are routinely collected at three additional areas, including the beaches at Wayside and the Lake Wissota State Park as well as from the

Yellow River at Cadott. Concentrations are normally low in the lake and highest at Cadott, but following the high rainfall and runoff in June 1993, concentrations were quite high at Wayside (Figure 9 1). During the previous summer, concentrations were lower except for a very high level on the Yellow River at Cadott (Figure 92). A smaller peak was also observed at Wayside. These results also suggest significant watershed loading of barnyard wastes during a time period (late June-early July 1993) when strea samples could not be collected due to high water levels.

It is also important to note that stream loading can be substantial during other periods. The stream sampling occurred during a time frame when vegetation was growing. More significant runoff might be expected during early spring when soils are still frozen and more direct runoff is contributed to streams, especially in areas where winter application of manure is commonplace.

1 1.0 Hydrologic budget

A water budget refers to the balance between the inflow and outflow of water to the study area. The inflow of water to Lake Wissota is the sum of precipitation on the lake, of unconcentrated and concentrated surface runoff to the lake, and of groundwater inflow to the lake. The outflow of water from Lake Wissota is the sum of evaporation from the lake surface, of surface water discharge at the Lake Wissota Dam, and of groundwater outflow from the lake.

The data available to complete the water budget of Lake Wissota were existing data of the U.S. Geological Survey (USGS), the Wisconsin Department of Natural Resources (DNR), Northern States Power Company, and USGS 7 1/2 minute topographic maps.

There are several gauging stations located along the Cbippewa River drainage basin (Young and Hindall 1972). Drainage areas in square miles are available for the gauging stations on the Chippewa River at Chippewa Falls (station no. 5-3655), on the Chippewa River near Bruce (station No. 5-3565), on the Flambeau River near Bruce (station no. 5-3605), on the Jump River at Sheldon (station no. 5-3620), and on the Yellow River at Cadott (station no. 5-3640). The drainage area for the drainage basins of Paint Creek, Stillson Creek, and Drywood Creek which drain into Lake Wissota or the Yellow River were also calculated. The total drainage area for Lake Wissota is very large - 5,548 square miles (USGS, 198 1). Table 6 presents the annual mean discharge and the discharge per square mile at the gauging stations for the Chippewa River above the inlet to Lake Wissota and downstream of Bruce and Sheldon, WI; for the Yellow River downstream of Cadott, WT and upstream of Moon Bay; and for the Paint Creek, Stillson Creek, and Duncan Creek drainage areas.

Input data -
Outflows from Lake Wissota

Outflow at Lake Wissota Dam

Discharge data at the Lake Wissota dam are recorded by Northern States Power Company (Lloyd Everhart, Northern States Power Company). Discharge data are recorded in a Northern States Power Company log book, which is available but not easily accessed. The Chippewa River discharge is recorded continuously at Chippewa Falls approximately 1.0 miles downstream from Duncan Creek (NE 1/4, NE 1/4, section 12, T28N, R9W, gauging station #05365500, Holmstrom et al, 1994). The drainage area above the gauging station is 5,650 square miles. The period of record for the gauging station is June 1888 to September 1983 and October 1986 to current year. The USGS states the records for the gauging station are "good". The annual mean discharge for water years 1888-1993 is 5,094 cubic feet per second (cfs) per 5,650 acres of drainage basin (Holmstrom et al, 1994). The annual mean discharge per square mile of drainage area for the Chippewa Falls gauging station is 5,094 cfs divided by 5,650 square miles or 0.90 cfs. The drainage area of Duncan Creek, which is upstream of the Chippewa Falls gauging station and downstream of the Lake Wissota darn is equal to 120 square miles (Ken Schreiber, DNR, Personal Communication January 27, 1995). An annual mean discharge of 108 cfs for Duncan Creek is estimated by multiplying 0.90 cfs per square mile of drainage area times 120 square miles of drainage basin. This calculation assumes that each square mile of the Chippewa River drainage basin above the Chippewa Falls gauging station contributes the same volume of water per interval of time. The estimated annual mean discharge from the outlet at the Lake Wissota Dam is 5,094 cfs minus the Duncan Creek annual mean discharge of 108 cfs or 4,986 cfs. This annual mean discharge neglects baseflow, direct precipitation to and evaporation from the Chippewa River, surface runoff to the Chippewa River between the Lake Wissota darn and the Chippewa Falls gauging station; and the potential loss of Chippewa River water to the cone of depression of the East Well Field of the City of Chippewa Falls.

Evaporation from Lake Wissota

The annual rate of evaporation from Lake Wissota is estimated as 29 inches/year (Ponce, 1989, page 41).

Groundwater Discharge from Lake Wissota

The calibrated water-table map presented by JRT Hydro, Inc. and Ayres Associates (1994) for the Chippewa Falls area shows that the Lake Wissota dam causes groundwater to flow from Lake Wissota around the darn to the Chippewa River. The quantity of groundwater outflow from Lake Wissota has not been quantified for this report.

Inflows to Lake Wissota

Inflow to Lake Wissota from the Chippewa River

The annual mean discharge is 1,476 cfs for the years 1914-1993 for the Chippewa River at Bruce (Holmstrom et al, 1994); 1,817 cfs for the years 1951 to 1993 for the Flambeau River near Bruce (Holmstrom et al, 1994); and 520 cfs for the years 1915 to 1993 for the Jump River at Sheldon (Holmstrom et al, 1994). The annual mean discharge of the Chippewa River above Lake Wissota but downstream of the gauging stations at Bruce, near Bruce, and at Sheldon equals 0.90 cfs per square mile times 951 square miles or 856 cfs. The total annual mean discharge of the Chippewa River into Lake Wissota is 4,669 cfs; which is the sum of 856 cfs, 1,476 cfs, 1,817 cfs, and 520 cfs for each sub basin of the drainage area of the Cbippewa River above Lake Wissota.

hiflow to Lake Wissota Through Moon Bay from Lower Yellow River

The drainage area of the Yellow River above gauging station # 5-3640 at Cadott, Wisconsin is 351 square miles (Young and Hindall, 1972). The annual mean discharge at Cadott for 1942 to 1961 is 273 cfs (Young and Hindall, 1972). The annual mean discharge per square mile for the Yellow River above Cadott is 273 cfs divided by 351 square miles equals 0.78 cfs per square mile of drainage basin.

The drainage area of the Yellow River downstream of Cadott and upstream of Moon Bay of Lake Wissota is 92 square miles. Assuming an annual mean discharge of 0.78 cfs per square mile, the annual mean discharge for the drainage basin between Cadott and Moon Bay of Lake Wissota is 0.78 cfs per square mile times 92 square miles or 72 cfs.

The annual mean discharge into Moon Bay from the Yellow River and from direct runoff to Moon Bay is the sum of the 273 cfs for the drainage area above Cadott and 72 cfs for the drainage area below Cadott. This equals an annual mean discharge of 345 cfs.

Inflow to Lake Wissota Through Little Lake Wissota from Paint Creek and Stillson Creek

The drainage areas of Paint Creek and Stillson Creek are 59 square miles and 9 square miles, respectively. Assuming an annual mean discharge of 0.78 cfs per square mile, the annual mean discharge for Paint Creek is 0.78 cfs per square mile times 59 square miles equals 46 cfs. The annual mean discharge for Stillson Creek is 0.78 cfs per square mile times 9 square miles equals 7 cfs. The total annual mean discharge to Little Lake Wissota is 53 cfs.

Precipitation onto Lake Wissota

The annual rate of precipitation on the surface area of Lake Wissota is approximately 31 inches/year for an average year (Young and Hindall, 1972). It is noted that the annual rate of precipitation on Lake Wissota is approximately equal to the annual rate of evaporation (29 inches/year) from Lake Wissota.

Groundwater Recharge to Lake Wissota

The quantity of groundwater inflow to Lake Wissota is not quanfified in this report. Groundwater inflow along the shoreline of Lake Wissota most likely occurs along the northern, eastern, and southern shorelines. As stated in Section 4.13, groundwater outflow occurs along the westem shoreline of Lake Wissota. Brown (1988) maps the bottom of Lake Wissota as Precambrian granite. The Precambrian granite is an aquitard which is a rock of low hydraulic conductivity (permeability). Therefore, groundwater inflow or outflow from the submerged, Precambrian portion of the pre-Lake Wissota, Chippewa River bottom is most likely minimal.

1 1. 1 Calculation of water budget for the main lake and two embayments based on the total period of record up to 1993

The annual mean discharge of surface water to Lake Wissota is 5,067 cfs which equals the sum of the discharges for the Chippewa River drainage area (4,669 cfs), the Yellow River drainage area (345 cfs), the Paint Creek drainage area (46 cfs), and Stillson Creek drainage area (7 cfs). The annual mean. outflow from Lake Wissota is 4,986. The difference of 81 cfs between the inflow and outflow of surface water represents a water balance error of 1.6 percent. Causes for this mass balance error may include: 1) the assumption that each square mile of the Chippewa River drainage basin contributes the same volume of water per time to the Chippewa River at the gauging station below the Lake Wissota dam; 2) errors in the estimates of the size of the drainage areas; and 3) a change in storage of water in Lake Wissota.

The inflow from precipitation (31 inches/year) and the outflow from evaporation (29 inches/year) account for only a 0.95 cfs addition to Lake Wissota (31.0 inches/year minus 29.0 inches/year equals 2.0 inches/year divided by 12 inches/ft equals 0.167 ft times 4,112.8 acres (area of Lake Wissota) times 43,560 feet/acre equals 29,918,646 cubic feet/year divided by 365 days/year divided by 24 hours/day divided by 60 minutes/hour divided by 60 seconds/minute equals 0.95 cfs).

The volume of Lake Wissota is 129,549.3 acre feet (Figure 12). The volume in cubic feet equals 129,549.3 acre feet times 43,560 square feet per acre or 5.643 1675 x 109 cubic feet. This number divided by the annual mean discharge of 5,067 cfs equals 1,113,709.8 seconds divided by 86,400 seconds per day equals 12.9 days. The 12.9 days is the time required to equal a volume of 129,549.3 acre feet at the inflow rate of 5,067 cfs.

The volume of Moon Bay is 4,375.6 acre feet (Figure 13). The volume in cubic feet equals 4,375.6 acre feet times 43,560 square feet per acre or 1.90601 x 108 cubic feet. This number divided by the annual mean discharge of 345 cfs equals 552,467 seconds divided by 86,400 seconds per day equals 6.4 days. The 6.4 days is the time required to equal a volume of 4,375.6 acre feet at the inflow rate of 345 cfs.

The volume of Little Lake Wissota is 6,729.2 acre feet (Figure 14). The volume in cubic feet equals 6,729.2 acre feet times 43,560 square feet per acre or 2.9312395 x 108 cubic feet. This number divided by 53 cfs equals 5,530,640.6 seconds divided by 86,400 seconds per day equals 64.0 days. The 64.0 days is the time required to equal a volume of 6,729.2 acre feet at the input rate of 53 cfs.

Additional information related to the annual water budget is found in Appendix D.

11.2 Calculation of water budget from May 1993 to April 1994

From May 1993 through April 1994, the annual mean discharge of surface water to Lake Wissota is 5,194 cfs which equals the sum of the discharges from the Chippewa River drainage area (4,796 cfs), the Yellow River drainage area (345 cfs), the Paint Creek drainage area (46 cfs and Stillson Creek drainage area (7 cfs ;Figure 19). For May 1993 through April 1994, the mean annual outflow from Lake Wissota is 5,456 cfs which equals 5,564 cfs at the Chippewa Fans gauging station minus 108 cfs for the Duncan Creek drainage area. The difference of 262 cfs between the inflow and outflow of surface water to Lake Wissota represents a water balance error of 4.8 percent. Causes for this mass balance error may include: 1) the assumption that each square mile of the Chippewa River drainage basin contributes the same volume of water per time to the Chippewa River at the gauging station below the Lake Wissota dam; 2) errors in the estimates of the size of the drainage areas; and 3) a change in storage of water in Lake Wissota.

The volume of Lake Wissota is 129,549.3 acre feet. The volume in cubic feet equals 129,549.3 acre feet times 43,560 square feet per acre or 5.643 1675 x 109 cubic feet. This number divided by the mean discharge of 5,194 cfs equals 1.0864781 x 10 Degs. seconds divided by 86,400 seconds per day equals 12.6 days. The 12.6 days is the time required to equal a volume of 129,549.3 acre feet at the inflow rate of 5,194 cfs.

The volume of Moon Bay is 4,375.6 acre feet. The time required to equal a volume of 4,375.6 acre feet at the inflow rate of 345 cfs is 6.4 days.

The volume of Little Lake Wissota is 6,729.2 acre feet. It takes 64.2 days to equal a volume of 6,729.2 acre feet at the input rate of 53 cfs.

11.3 Summer low flows

Using a similar approach, the residence time was calculated for potential summer low flows to Moon Bay and Little Lake Wissota. Based on 1888 to 1994 data the residence time of Moon Bay ranged from 4.1 to 8.4 days (Tinker, 1996). The residence time in Little Lake Wissota was considerably longer and ranged from 40.5 to 84.6 days. Similar results were obtained for 1993. These calculations suggest that while the residence time in the main basin of the lake is relatively short, during summer and particularly in August residence times in the two embayments, especially Little Lake Wissota are longer. This information was based on assuming similar flow relationships between the Yellow River and Paint Creek watersheds and the Chippewa River. Additional information on these calculations based on flow data assumed based on the period of record observations at Chippewa Falls is found in Appendix E.

However, because of the moderate to poor permeability of soils in the Yellow River and Paint Creek watersheds, summer low flows might be considerably lower if runoff varies between the basins. A gauging station was maintained by the U.S. Geological Survey on the Yellow River at Cadott from 1943 to 1961. While no subsequent flow information was available and no gauging station was approved for this project, assuming that the relationship of runoff to precipitation has not changed appreciably in the last 35 years, one can estimate mean flows during the summer for the Yellow River and residence times for Moon Bay. Using data obtained from U.S. Geological Survey records, we can estimate the monthly mean flow, Q 30,5 and Q 3 0, 1 0 (monthly low flows expected every 5 or 10 years) during the summer. On a monthly basis, these results indicate a very high variability in flow from month to month. Mean flow conditions indicate that discharge in June is much higher than July, which exceeds August and September. However, over a ten year period, flow in any one month might be very low, resulting in long estimated retention times in the basin (Table7). These estimates of Q 30,10 have been used in the modeling analysis following consultation with WI DNR staff.

Table 7

Year of Record Flows for Summer - Yellow River entering Moon Bay

Monthly discharge and resulting retenfion time of water in days

1943-1961

	June		July		Aug		Sept
	cfs	RT (dy)					
Q mean	490	9	210	2 1	109	40	150
Q30, 5	159	28	62	71	34	129	18
Q30, 10	28	156	25	175	15	291	10

It is important to note that while flow during an individual month (or over a shorter period) might be very low, variations found from month to month result in the discharge over a growing season to be higher than might be expected based on a single month. High flow months may be followed by periods of low flow. This pattern occurred in 1993, when very high flow conditions were found in June and early July (during the period of major flooding along the Chippewa River and Mississippi River), while the runoff for the entire year was slightly above a long-term average. The long term data for the gauging station at Cadott (Table 8) would indicate that 5 of 19 years had flows <100 cfs as a mean over the growing season. The lowest mean flow observed was 45 cfs, suggesting that approximately every 20 years the residence time in Moon Bay might be as long as 97 days.

Table 8

Mean monthly growing season flows for the Yellow River at Cadott

Year	cfs
1943	895
1944	94
1945	272
1946	322
1947	81
1948	27

1949	158
1950	88
1951	314
1952	127
1953	201
1954	374
1955	231
1956	187
1957	84
1958	326
1959	545
1960	190
1961	45

mean = 236, s = 209

11.4 Operation of Moon Bay and Little Lake Wissota during the summer growing season

During periods of low flow during summer, residence times increase in the two embayments. For Little Lake Wissota, residence times might be relatively long, especially when compared with the main basin of the lake. For Moon Bay, somewhat shorter residence times are likely to occur. For both basins, the distribution of storm events is critical in determining the effective residence time for a "parcel" of water received during a given period. As a consequence, the interaction of a stochastic, climatically-driven process and land runoff will determine the loading of nutrients to each basin and the length of time that algal populations would have to respond. It is important to note that as runoff declines, so does the modeled export of nutrient runoff. Hence, lower concentrations may be predicted for in-lake concentrations during years of lower runoff. The frequency and occurrence of blue-green bacterial blooms in the two embayments is likely determined by a complex interaction of surface loading and the timing and frequency of storm events during the summer. Also, given the nutrient concentrations observed for all periods of record, with any calm, warm period in mid-summer, blue-green blooms might be expected to occur.

12.0 Drawdown effects

A common practice with reservoirs used for hydropower production in temperate areas is to reduce water levels in anticipation of snow melt and spring rains (Hunt and Jones 1972, Kaster and Jacobi 1978). Fluctuation of reservoirs, particularly long-term periods at minimum pool, can profoundly influence community structure and stability of the littoral zone, the area considered to be the most diverse within lake systems (Wetzel 1983).

Each year the level of Lake Wissota is drawn down to accomodate spring snowmelt runoff. The potential effects of drawdown on biological populations were evaluated m two primary ways. The first was a mapping exercise to examine the changes in lake morphometry when the lake was at different levels. The second analysis involved sampling benthic invertebrate populations to examine how their populations change while the lake is being drawn down and how the populations recover as the reservoir is re-filled.

Drawdown typically begins in mid- to late-Febuary and is maintained at lower levels until spring run-off. The lake is drawn down an average of 10 ft (ranging from 5 - 15 ft in some years) at a rate of 0.5 ft(day. The lake surface is still frozen during drawdown, with ice reaching a thickness of 2 ft or more in some years (Delong, personal observation), and the surface ice collapses onto the shoreline as the water recedes. Ice remains over the shoreline throughout the drawdown period until spring thaw. The average date for the onset of spring run-off is 21 March.

Normally drawdown does not reach 10 feet but up until 1984 it was common practice to draw the reservoir down 15 feet each year. Separate maps were prepared to evaluate the impact of drawdown on each of the areas of the lake. These maps were built from the base map described earlier and then 5, 10 and 15 foot drawdowns were plotted and the change in area and volume were calculated (Figures 16-19; Table 4).

Loss of benthic surface area during drawdown varied within the three areas of the lake: Moon Bay, Little Lake Wissota, and the main body of Lake Wissota. Drawdown to 5 ft results in approximately a 10% loss of bottom surface area in Little Lake Wissota, versus a 2.5% and 9.4% loss of total bottom surface area in the main lake and in Moon Bay, respectively. However, the area affected by drawdown, and especially the volume, is enhanced substantially at 10 feet. Total bottom surface area of Moon Bay is reduced by 37% when the lake is drawn down 10 fee% limiting most of the wetted bottom to within the original channel of the Yellow River. The loss of area in Little Lake Wissota increases to 27.7 % when drawn down to 10 ft. By contrast at the same time, this results in only a 6.3% reduction of bottom surface area in the main lake. The main body of Lake Wissota is characterized by steep banks, leaving a narrow littoral zone. Loss of bottom surface area due to drawdown is, therefore, smaller than the proportional losses in Moon Bay and Little Lake Wissota, but it has implications for other communities.

12.1 Fish stranding

The lake maps were evaluated by Joe Kurz, a fisheries biologist with the Wisconsin DNR, to determine whether there were any areas where fish might be stranded during periods of drawdown. These areas were also surveyed during drawdown periods. Because the reservoir filled a stream channel and valley floor as described in the geological description, the pre-existing stream channel represents the deeper troughs in the lake and they also are the best corridors for fish movement as water level declines. Given the morphometry of the basin and the lack of significant constrictions along most of the channels and embayments, there are very few areas where stranding may occur. The maps indicate no large areas where drawdown might strand fish because a corridor in constricted.

There are, however, some minor embayments, such as Pine Harbor, with restricted channels or small areas within Yellow Bay and Little Lake Wissota or the main basin of the lake where some fish stranding might occur (Kurz, personal communication). These areas are generally less than 1 acre in size and are not identified on the lake maps. At a drawdown of 10 fee% there are small areas in the southeastern corner of the lake, around Mermaid Island and possibly in Pine Harbor, where some stranding may occur.

Dissolved oxygen measurements were made in winter and during drawdown when ice conditions were safe for sampling. No significant oxygen depletion was found in Pine Harbor during 1994 or in other areas during drawdown. Oxygen concentrations were low near the sediment.

Other observations suggest that the effect of drawdown on fish populations is not completely the result of barriers to dispersal. At the east end of Moon Bay, fish have been observed to stay in macrophyte areas even while drawdown is occurring (Kurz, personal observation). Mortality of young bluegills results in this case not from stranding but from a lack of responsiveness of the fish to the changes in water level. Given these kinds of behavior, it is very difficult to quantify the overall impact of (drawdown on fish populations. There are no significant barriers to the dispersal of fish, but their populations might be affected nonetheless.

12.2 Aquatic macrophytes

The distribution and density of aquatic macrophytes in Lake Wissota was studied by Borman (1991,Appendix F). Several major factors influence the distribution of aquatic plants in lakes. Among these factors are: water quality and clarity, water level fluctuations, sediment composition, lake morphometry and wave action. As described in the section on limnological conditions, the transparency of the water column in Lake Wissota is affected by water color and by suspended inorganic particulates and algal populations, which in tum are influenced by nutrient concentrations. Based on measurements of Secchi disc transparency, the maximum rooting depth of aquatic macrophytes in Lake Wissota is 1 - 2 m in depth, being lowest in the two embayments and highest m the main body of the lake. Sediment composition also varies between the embayments, which have finer grained sediment with more organic matter, and the main body of the lake, where gravel and sand sediments predominate. Two other factors contribute to differences in macrophyte populations between the main basin and the two embayments. The main basin is steeply sloped and also large enough for substantial wave action, which offers a limited environment for the colonization of aquatic plants (Duarte and Kalff 1986).

A total of 3 1 species of aquatic plants were found in the lake (Bormann 1991). Of these, 12 were emergent species, 5 were floating-leaved and 14 were submersed forms. Aquatic plants were found at 40.5% of the sampling points. While fine-grained sediments occurred at only 8% of the sampling points, those sites had the highest frequency of plant occurrence and also plant density. Aquatic macrophytes were found at 87% of the sites with silt or muck sediment but only 7% of the sites with gravel rubble or boulders. High slope and wave action in the main basin, particularly on the northern shore made these areas difficult ones for plants to colonize and no plants were found along that shore. Because of the true color of the lake and particulates shading the water column, the deepest plants in the survey were growing to depths of only 7.5 feet which is within the drawdown zone. No plants were able to grow at depths below that influenced by drawdown.

AH of the species present in the lake with a frequency of occurrence of 5% or more are classified as "drawdown tolerant" (Nichols 1975, WT DNR 1990). Several species commonly found in other lakes in the area, especially waterlilies, were not found in Lake Wissota, undoubtedly related to the fluctuating water level. In addition, several of the species present with a frequency of occurrence greater than 5% have been found to increase in plant growth in other Wisconsin reservoirs having winter drawdowns. These observations suggest that winter drawdowns play a significant role in shaping the composition of the aquatic plant community and the littoral zone area of Lake Wissota.

12.3 Invertebrate populations

Because drawdown has a considerable affect on area, significant portions of the lake are exposed during periods of the year. The areas are dewatered and the sediment is exposed to freezing temperatures during late winter. Such conditions may subject benthic invertebrates to a wide range of conditions, such as exposure to the open air or remaining in somewhat moist sediments beneath a layer of ice, including freezing and drying.

Many benthic invertebrates are of tolerating freezing temperatures, with mortality approaching 100% under severe circumstances (Andrews and Rigler 1985, Danks 199 1, Oswood et al. 1991). Many taxa, however, have the ability to survive such conditions. Some members of Oligochaeta and Chironomidae form cocoons in the winter, whereas other invertebrates, such as Chironomus plumosus and some Ephemeroptera, can burrow deeper into the sediments to avoid freezing condition at the surface (Danell 1981, Olsson 1981, Andrews and Rigler 1985). Highly mobile invertebrates, such as many Ephemeroptera and Crustacea (e.g., Amphipoda and Isopoda), may be able to leave dewatered areas, providing the rate of drawdown is not exceedingly rapid. Less mobile invertebrates, such as the Sphaeriidae and many Trichoptera, typically suffer extensive mortality in areas prone to freezing (Danks 1991, Palomaki and Koskenniemi 1993).

This study examined the effects of late-winter drawdown of a reservoir in central Wisconsin on the benthic community of the littoral zone. The objectives of this study were: to compare benthic invertebrate community structure before and after drawdown to assess the initial impact of drawdown; and to examine the recolonization of the zone impacted by late-winter drawdown.

Drawdown began on 14 February 1994 when the surface lake elevation was 897.49 ft. Drawdown continued until the lake reached a surface elevation of 888.0 ft on 1 1 March 1994. The lake fluctuated at this level until it began to refill around 22 March 1994. Lake Wissota returned to its previous surface elevation on 10 April 1994, which coincided with the melting of the remaining surface ice. Sampling to assess drawdown effects and to monitor recolonization of the impact zone (the shoreline area dewatered during drawdown) began on 15 April 1994. AH samples taken following this date are referred to as postdrawdown samples.

Sample Procedures

Benthic invertebrate samples were collected beginning 12 - 13 November 1993. These samples were intended to represent the community prior to drawdown. Samples were collected immediately following ice out on 16 - 17 April 1994 to determine postdrawdown community structure. On both of these dates, 22 transects were sampled throughout the lake. Selected transects (denoted by ** in Figure 93) were sampled to follow the recolonization of the zone affected by drawdown. Samples to monitor recolonization were collected on 23 - 24 April, 30 April - 1 May, 14 - 15 May, 9 - 10 June, and 23 - 24 June 1994.

Samples were collected at four depths along each transect: 2, 5, 10, and 15 ft. Samples from 2 and 5 ft were within the zone affected by drawdown, whereas samples from 10 ft were at the transition between dewatered and continuously inundated substrata and 15 ft samples were in areas where substrata remained inundated throughout the study. All methods and analyses are detailed in Delong and Mundahl (1996, Appendix G).

Benthic invertebrate samples were collected in most instances with a Petersen dredge. A 0.09-m2 ring with 200-um mesh netting was used in shallow cobble areas where the dredge was ineffective. AU samples were washed through a 200-PM sieve in the field and preserved with 70% ethanol and rose bengal stain. In the laboratory, samples were sorted using the kerosene-alcohol technique of Barmuta (1988). Large organisms (> 1 nun) were sorted under a 1 OX sorting lamp and smaller organisms were sorted at 2OX with a dissecting microscope. AH taxa, except Chironomidae and Oligochaeta, were idenfified to the lowest possible taxonomic level.

Data were analysed using the three areas of the lake, Moon Bay, Little Lake Wissota, and the main body of Lake Wissota, separately; and for the lake as a whole. The latter was achieved by pooling data from the three lake areas. In all instances, major taxonomic groups were used to determine if their mean densities differed significantly with depth or sample date. Significant differences in mean total number of individuals and mean number of taxa were also tested. Data did not satisfy the assumptions of normality until they were transformed using $\log10(x + 1)$. Data were analyzed for both the separate sample area tests and pooled-data test using two-way analysis of variance in the PROC GLM procedure of SAS, inc. (1989). Factors used in the statistical tests were depth (levels = 4) and date (levels = 7). A depth*date interaction term was also included in the ANOVA model. A non-significant interaction term would mean that the four sample depths follow the same pattern on all dates, that is they exhibit the same statistical differences or similarities, making it unnecessary to examine each depth separately. In contrast a significant interaction term indicates that patterns for the sample depths differs among sample dates, so that each depth must be examined separately to determine where significant differences occurred among dates for each depth. When significant differences were detected, a least-squares mean comparison test was performed to determine where differences existed among depths or dates.

Results

A total of 105 taxa was collected from Little Lake Wissota, Moon Bay, and Main Lake Wissota (Delong and Mundahl 1996, Appendix G). Major taxonomic groups included: Ephemeroptera (mayflies), Trichoptera (caddisflies), Diptera (true flies), Annelida (worms), Crustacea, and Mollusca. For purposes of analysis, Diptera was divided into Chironomidae and non-chironomid Diptera (referred to as Diptera through the remainder of the report). All taxa not contained within the major taxonomic groups listed above were combined and are referred to as "other" invertebrates. Division of taxa by total number of individuals, not including Chironomidae, identified 18 taxa that represented >1% of the entire Lake Wissota invertebrate community throughout the course of the study. The same examination of the three regions sampled on Lake Wissota revealed that 14 taxa comprised >1% of the non-Chironomidae community in Moon Bay, whereas 17 and 19 taxa were the most abundant representatives of the communities found in Little Lake Wissota and main Lake Wissota, respectively.

Comparisons among different areas of Lake Wissota

Moon Bay: Analysis of variance detected significant differences among depths for the total number of individuals, number of taxa, and all major taxonomic groups except Ephemeroptera, Diptera, and Mollusca. The total number of individuals and number of taxa also differed significantly among dates, as did numbers of Trichoptera and Diptera. The interaction term, depth*date, was significant for Trichoptera and total number of species present.

The total number of individuals was greatest for samples taken at 2 ft (Figure 94) and was lower at 5 and 10 feet, where almost no difference existed between these two depths. Total number of individuals was lowest at 15 ft, thus accounting for the significant difference for total number of individuals at 15 ft from the other sample depths. The largest decrease in density seen from 2 ft down to the other depths occurred among the Chironomidae. The "other" invertebrate group followed the same trend. Trichoptera were only abundant at samples from 5 ft, whereas Crustacea exhibited similar densities at all depths, except at 15 ft where numbers were lowest. Ephemeroptera, represented primarily by Hexagenia limbata, were most abundant at 10 ft.

The total number of individuals in November 1993 was substantially greater than totals collected on all postdrawdown sample dates (Figure 95). Examination of mean comparisons revealed that this trend of high numbers in November 1993 and low densities in postdrawdown samples was followed by all major taxa, whether significant differences existed or not with the exception of the Ephemeroptera and

Annelida. Invertebrate densities fluctuated but remained low, relative to densities of November 1993, throughout the postdrawdown sampling period, never approaching more than 55% of the predrawdown densities (doing so on 14 May). None of the major taxonomic groups, with the exception of the Ephemeroptera and Annelida, had postdrawdown densities approaching predrawdown densities.

Examination of total number of individuals across sample dates for each depth revealed that total number of individuals decreased at all depths from November 1993 to April 16, 1994 (Figures 95 and 96). In other words, there was no increase in total number of individuals at depths below the impact zone following drawdown, which would have suggested movement of organisms from shallower to deeper water. Nearly all major taxonomic groups declined m numbers from pre- to postdrawdown sampling, with only the Annehda appearing to be unaffected.

Densities at 2 ft began to rise after 30 April 1994, reaching total densities statistically comparable to those of November 1993 on 23 June 1994. The same was evident at 5 ft. Mean values for most invertebrate groups at 2 ft changed little throughout the 1994 sample period, although Crustacea and Chironomidae densities were higher in samples from 14 May - 23 June. Total number of individuals fluctuated widely at 10 ft, with mean densities exceeding predrawdown densities on 23 April and 14 May before dropping to their lowest point on 23 June. Densities of Ephemeroptera, Annehda, and Chironomidae were responsible for the widest fluctuations observed in postdrawdown samples taken at 10 ft. Community structure at 10 ft in Moon Bay actually appeared to be the least affected by drawdown. The community was dominated by Crustacea-in November 1993. While crustacean densities remained low following drawdown, all other major taxonomic groups appeared to have returned to or exceeded predrawdown densities.

Little Lake Wissota: Very few significant differences were observed for samples from Little Lake Wissota. Only numbers of Mollusca differed significantly among depths, while Ephemeroptera, Diptera, Annehda, Chironomidae, and total number of individuals differed significantly among dates (see Appendix G).

The total number of individuals was nearly identical at 2 ft and 5 ft, but decreased at 10 ft and 15 ft (Figure 97). Nearly all major taxa represented the same or nearly the same relative proportion of the community at all depths from which samples were collected. Diptera increased in numbers with increasing depth, whereas the Mollusca were abundant only in samples from 2 ft. These deviations from the community pattern observed were apparently sufficient to produce the significant differences noted for these two taxonomic groups.

Total invertebrate densities were highest in November 1993 at 2 and 5 ft, whereas total density was highest on 16 April 1994 at 10 and 15 ft (Figure 98). Densities at 2 and 5 ft declined substantially from November 1993 to 16 April 1994, with much of the reduction at 2 ft a result of lower densities of Chironomidae and Mollusca. Chironomidae were also responsible for the decline from November 1993 to 16 April 1993 at 5 ft. Densities of Ephemeroptera, Trichoptera, and Annelida also dropped substantially from before to following drawdown. The increase in density at 10 ft from November 1993 to 16 April 1994 was almost entirely attributable to greater numbers of Chironomidae. While higher numbers of Chironomidae were primarily responsible for the total density observed at 15 ft on 16 April densities of Crustacea and Annehda were also high. Diptera, excluding Chironomidae, were the only major taxonomic group to decrease substantially from November 1993 to 16 April in samples from 15 ft.

Densities at 10 and 15 ft increased from November 1993 to 16 April 1994, and densities at 2 and 5 ft decreased over the same interval, suggesting the possibility of movement to deeper depths. However, the increases in community densities in deeper samples on 16 April were not of taxa abundant at shallower depths in November 1993. There appears, therefore, to be little linkage between densities in shallow areas to densities of deeper areas in Little Lake Wissota.

Main Lake Wissota: Ephemeroptera, Diptera, Crustacea, Mollusca, and Chironomidae differed significantly among the different depths from which samples were taken (see Appendix G). Significant differences among dates of sample collection were noted for all taxonomic groups except Ephemeroptera, Diptera, other invertebrates, and number of taxa. The interaction term was significant only for the Crustacea. No clear pattern was revealed by mean comparisons among depths (see Appendix G). Densities of Trichoptera prior to drawdown were both higher than and significantly different from densities on all dates for which samples were taken following drawdown. Densities of Chironomidae and total number of individuals were highest in November 1993 and were significantly different from densities from all 1994 samples except 23 June 1994. While densities of Mollusca were highest prior to drawdown, densities in November 1993 were not significantly different from densities observed on 23 and 30 April 1994. Predrawdown densities were, however, significantly different from all of the remaining postdrawdown samples. Significant differences for Crustacea at 2 ft demonstrated no clear trend, whereas densities at 5 ft for November 1993 were significantly different only from densities on 16 April 1994. There were no significant differences among sample dates at 10 and 15 ft for Crustacea.

The total number of individuals was highest at 5 ft, followed closely by 10 ft (Figure 94). Total numbers were lowest and 2 and 15 ft. This trend was reflected

directly by the Chironomidae. Crustacea also contributed substantially to the high densities found at 10 ft. Diptera and Ephemeroptera deviated from the pattern seen for total number of individuals, with densities increasing with increasing depth. Densities of Ephemeroptera were a result of high numbers of Hexagenia sp. whereas Diptera were dominated by Chaoborus bicinctus at 15 ft.

The total number of individuals decreased substantially from November 1993 to 16 April 1994 (Figure 95). Total densities remained low, although densities appeared to increase slightly by 14 May. Densities declined slightly on 9 June but increased beyond 14 May densities by 23 June 1994. The most prominent decrease from before to after drawdown was within the Chironomidae; however, densities of non-chironomid invertebrates also exhibited a marked decrease with densities declining by approximately 50%. This was particularly evident in the decrease in numbers of Crustacea, Trichoptera, and Mollusca. Among these three groups, only densities of Crustacea increased in later samples.

The number of taxa did not differ for depth or date. Mean number of taxa in Main Lake Wissota was the lowest of the three lake areas sampled and never exceeding 10 taxa.

Densities at 2, 5, and 10 ft decreased dramatically from November 1993 to 16 April 1994, dropping, on average, 7,000 - 9, 000 individuals/m^2. The total number of individuals also declined at 15 ft, although not as markedly as at the shallower depths. As was the case for Moon Bay, densities at 2 and 5 ft increased slightly from 16 to 30 April 1994, with Crustacea contributing to most of the increase at 2 ft and with Chironomidae doing the same at 5 ft. Densities at 2 ft continued to increase through 23 June, with Chironomidae contributing to the higher density on this date. Mean total density at 2 ft on 9 June was roughly 50% that of mean total densities prior to drawdown. Total density at 5 ft remained about the same on 14 May but dropped significantly on 9 June as a result of lower densities of Chironomidae. Total density increased to 8,100 individuals/m^2 by 23 June 1994. Numbers declined through April at 10 and 15 ft, whereas densities remained almost unchanged through April 1994 at 15 ft. Total number of individuals declined further after 16 April and remained low throughout the sample period until increasing again on 23 June 1994.

As was the case for Moon Bay, it was apparent that densities were substantially lower from 2 - 10 ft in the post-drawdown period when compared to invertebrate densities from November 1993. Densities observed in April again indicate that benthic invertebrates did not move to deeper areas in response to the declining water levels.

12.4 Drawdown Effects: Moon Bay and Main Lake Wissota

Benthic invertebrate densities observed in Moon Bay, Main Lake Wissota, and the pooled-lake data all demonstrated a substantial decrease from November 1993 to 16 April 1994, the first date on which samples were taken following drawdown. The decline resulted in the total number of individuals and densities of several major taxonomic groups in November 1993 being significantly higher than all April 1994 samples.

The decrease in invertebrate densities in Moon Bay and the main basin of Lake Wissota were highest at 2 feet and decreased with depth. There was an 85% decline in mean total number of individuals/m2 at 2 ft in Moon Bay and a 790/o decrease from November 1993 to 16 April 1994 at 5 ft. Densities decreased by 54% and 6 1% at 10 and 15 ft, respectively. Invertebrate losses from November 1993 to 16 April 1994 were 840/o, 750/o, 700/o, and 470/o going from 2 to 15 ft in the main body of Lake Wissota.

Total invertebrate densities in Moon Bay and the main body of Lake Wissota were almost always higher within the impact zone than in the permanently inundated areas. This pattern was seen in samples collected both before and after drawdown. An original hypothesis of this study was that benthic invertebrates within the impact zone would migrate to permanently inundated areas during drawdown. The fact that invertebrate densities at 10 and 15 ft on 16 April 1994 were lower than those observed in November 1993 in Moon Bay and the main body of Lake Wissota do not support this hypothesis. Instead, there was an overall decrease in invertebrate densities for nearly all of the major taxa following drawdown. Even mobile taxa such as the amphipod Hyalella azteca and the isopod Asellus racovitzae, were present in lower numbers following drawdown.

Invertebrate densities declined substantially from November 1993 to 16 April 1994, particularly at the 2 and 5 ft sample depths. Although there is abundant evidence that invertebrates, especially Chironomidae, can survive freezing conditions, it should be assumed that at least part of the decline in density at 2 ft can be attributed to shoreline freezing (e.g., Oswood et al. 1991). In addition, there was a substantial decline of invertebrates at 5 ft in Moon Bay and the main body of the lake, where the decrease in total density and densities of several major taxonomic groups exceeded the decline in densities witnessed at 15 ft, which remained inundated during the drawdown period.

Low dissolved oxygen concentrations were observed in Moon Bay and the main lake when samples were taken in November 1993. Additional data on Lake Wissota dissolved oxygen concentrations indicate that low dissolved oxygen concentrations are not generally a problem during the winter except near the sediment surface (Brakke, personal observation). It should also be noted that invertebrate losses were still higher at 5 ft than they were at 10 and 15 ft, where dissolved oxygen concentrations were lowest on 12 November 1993. There is no strong evidence, therefore, to suggest that invertebrate losses were caused by low oxygen concentrations during the winter.

12.5 Recolonization of the Impact Zone: Moon Bay and Main Lake Wissota

Densities of some major taxonomic groups were statistically similar to predrawdown densities by 14 May 1994; however, most taxa were still present in numbers lower than those observed in November 1993. This trend continued for samples taken on 23 June 1994, despite a decline in total densities on 9 June. Although mean total densities on 23 June 1994 remained less than those observed prior to drawdown, the statistical similarity in mean densities of most major taxa would suggest that the lake was approaching, if not necessarily reaching, recovery from possible drawdown impacts within 11 weeks of returning to full pool.

Kaster and Jacobi (1978) proposed that both the Chironomidae, represented primarily by Chironomus plumosus and adult Oligochaeta, consisting mostly of Limnodrilus spp., burrowed into sediments and remained there until the impact zone was reinundated. Chironomidae were capable of moving back up to the surface sediments following reinundation and Limnodrilus spp. rapidly recolonized the impact zone by reproduction. As a result, recolonization occurred within 3 months in Big Eau Pleine Reservoir. Palomaki and Koskenniemi (1993) also noted a relatively brief recolonization period. Kaster and Jacobi (1978) further noted that low numbers of larval Chironomidae at two stations were probably of little importance in recolonization because large numbers of larvae were surviving below the drawdown limit.

Total invertebrate densities observed during the period following drawdown indicate that invertebrate declined following drawdown, at least in Moon Bay and the main body of Lake Wissota. If benthic invertebrates were burrowing into the bottom sediments at even moderate numbers, the disparity between pre- and postdrawdown should be low and densities should have been approaching predrawdown levels by the end of the postdrawdown sample period (i.e., Kaster and Jacobi 1978). While total densities as well as densities of Annelida Mollusca, and Chironomidae were statistically similar to densities observed in November 1993, the actual mean densities were still lower than those observed prior to drawdown. Furthermore, all 1994 densities of Trichoptera remained significantly different from predrawdown densities.

Another possible source of recolonization could be areas with large amounts of detritus. Palomaki and Koskenniemi (1993) noted that survival in ice covered areas was higher where detritus was abundant than in sandy regions. Detritus was not abundant in the littoral zone of either Moon Bay or Lake Wissota. Both areas are subjected to wind-induced wave action, which probably moves detritus from nearshore areas to deeper areas of the lake. Further-more, Moon Bay and the main lake both have a rapid water flushing rate (4 and 12 days, respectively). As a result detrital refugia are rare in Moon Bay and the main body of Lake Wissota, further limiting the ability of benthic invertebrates to rapidly recolonize the impact zone of these areas. During a preliminary survey of Moon Bay, invertebrates such as Asellus racovitzae and Pseudocloeon sp. were found on the bottom substrata when detrital mats were located through fissures in the ice.

Little Lake Wissota

Data from Little Lake Wissota followed a trend much different from that observed in Moon Bay and the main body of the lake. There were few significant differences among depths and sample dates for the Little Lake. The total number of individuals dropped by 78 and 55% at 2 and 5 feet respectively, from November 1993 to 16 April 1994. There was an increase of 119 and 81% in invertebrate density at 10 and 15 ft, respectively, over this time period. Of note, however, is the fact that invertebrate densities at all four sample depths continued to decline or remain at densities lower than those observed on 16 April 1994. There was no increase in numbers at 2 and 5 ft in response to decreases in total numbers at 10 and 15 ft on these later dates. Much of the decline at 2 and 5 ft in Little Lake Wissota was a result of lower numbers of Chironomidae; however, it should be noted that both Ephemeroptera and Diptera other than Chironomidae also showed declines from November 1993 to 16 April 1994. The increased densities at 10 and 15 ft following drawdown did appear to be a result of increased densities of Chironomidae. Numbers of Crustacea were also higher on 16 April than they were at 15 ft prior to drawdown, but there was no apparent increase in Ephemeroptera. Comparison of benthic invertebrate community structure in Little Lake Wissota before and after drawdown do suggest there may have been migration of at least members of the Chironomidae and Crustacea, but community structure at all depths on succeeding dates suggests these individuals may have remained in deeper water rather migrating back to the drawdown zone.

As previously stated, most major taxonomic groups and total number of individuals did not differ significantly among depths in Little Lake Wissota. A possible reason for this, and the differences in community structure witnessed in Little Lake Wissota when compared to Moon Bay and the main lake, is the presence of detritus at all depths of Little Lake Wissota. Although usually not of a sufficient quantity to be characterized as a primary or secondary substratum (the substrata classification system used requires at least 50% of the substrata be organic matter in order to be characterized as organic matter substrate), leaf litter was consistently most abundant in samples from Little Lake Wissota. The abundance of detritus and its capacity to serve as refugium (e.g., Palomaki and Koskenniemi 1993) may account for the patterns of invertebrate community structure found in Little Lake Wissota.

Despite the greater abundance of benthic invertebrates immediately following drawdown, invertebrate densities in Little Lake Wissota declined over the course of the sample period, dropping to levels comparable to those observed in other parts of the lake. Both Moon Bay and the main body of Lake Wissota, on the other, had a significant decline in invertebrate density from November 1993 to 16 April 1993. Furthermore, densities remained low for many major taxonomic groups for the remainder of the sample period, never approaching densities observed prior to drawdown. The decrease in invertebrate abundance in Little Lake Wissota and the continuing low densities of Moon Bay and the main lake appear to be a result of insect emergence. Many of the benthic insects obtained in our samples over-wintered as mid- to late-instar larvae. As water temperatures increased, surviving members of the community emerged, as evidenced by the increased abundance of pupae in samples and high number of pupal exuviae observed both in our samples and on the lake surface during collection of samples (Delong, personal observation). The timing of sample collection corresponds with the emergence of many species of Chironomidae from this area (Hilsenhoff 1966). The compounding effect of invertebrate loss, primarily through drawdown-related mortality and insect emergence, appeared to have delayed recolonization of the drawdown zone until the offspring of surviving individuals hatch and begin to use this area. It is our hypothesis that the invertebrate community of Lake Wissota does not retum to numbers comparable to those observed in November 1993 until at least mid-summer and possibly into late late-summer (August) after eggs deposited by adults emerging in the spring and early summer have hatched and the hatching insects have distributed themselves throughout the littoral zone.

The same response most likely occurs in other invertebrates, such the isopod, Asellus racovitzae. Juveniles were observed on females of A. racovitzae on 14 May 1994 and were still observed on females on the last sample date. Previous studies of drawdown impacts (e.g., Grimas 1965; Kaster and Jacobi 1978) have indicated that recovery following reinundation of drawdown areas can occur within 3 months. The sample period for Lake Wissota fell just short of three months; however, the densities of many of the taxonomic groups did not show any indication of approaching predrawdown densities 11 weeks after the lake had returned to full pool. A primary reason for this protracted period of recolonization is that the low numbers and diversity of all invertebrates below the drawdown limit (> 10 ft) provides a very reduced pool of individuals and taxa for the recolonization of the dewatered area.

Benthic invertebrate densities in Moon Bay and the main body of Lake Wissota, Wisconsin, were lower following mid-winter drawdown than densities observed prior to drawdown. Decreases in densities were observed at all depths sampled, with numbers showing the greatest decline at 2 and 5 ft. Differences in densities were lowest at 15 ft, where bottom substrata remained inundated throughout the draw down period. Assuming the reduction in density at 15 ft is representative of natural mortality, lower invertebrate densities within the zone of drawdown unpact in excess of observations at 15 ft are probably a result of other phenomena.

While bottom substrata within 2 ft of the shoreline may be subjected to ice scouring, the benthos from 2 - 10 ft are not subjected to scouring under normal conditions. It appears most likely, therefore, that the difference in total number of individuals and densities of major taxa from November 1993 to April 1994, particularly at 5 and 10 ft, are associated with the drawdown of Lake Wissota. The absence of an increase in invertebrate densities below the drawdown zone of Moon Bay and the main body of Lake Wissota did not support the hypothesis that invertebrates would migrate to deeper areas in response to dewatering. As a result, the pool available for the recolonization of the drawdown area consisted of individuals surviving within the drawdown zone and the oviposition and hatching of the next generation of invertebrates. Densities of some major taxa on 23 June 1993, 11 wk after the lake returned to full pool, did not differ significantly from densities in November 1993; however, some taxa (e.g., Trichoptera) remained lower than November densities. While densities of some taxa on 23 June 1994 did not differ statistically from November 1993, the overall low mean total number of individuals on 23 June 1994, relative to November 1993, suggests that recolonization of the drawdown area is protracted to at least mid-summer, if not later.

The invertebrate community of Little Lake Wissota responded differently from that of the main lake. Invertebrate densities did increase at 10 and 15 ft following drawdown in conjunction with decreased densities at 2 and 5 ft. Invertebrate densities did, however, decline during the period following drawdown. These lower densities were most likely a response to insect emergence, especially among the Chironomidae. Comparisons of the physical and chemical structure of Little Lake Wissota suggests that this area shares characteristics common to either Moon Bay or Little Lake Wissota. Two marked differences between Little Lake Wissota and the other two areas are: 1) water remains in residence in the Little Lake for a much longer period (64 days versus 4 and 12 days for Moon Bay and the main lake, respectively); 2) there is a greater abundance of detritus associated with bottom sediments. These two factors may interact to increase the stability within Little Lake Wissota, even during drawdown, thus reducing any adverse affects on the benthic invertebrate community.

12.6 Potential interaction of aquatic macrophytes, benthic invertebrates and fish

Annual drawdown of the lake results in an aquatic macrophyte community that can tolerate exposure and freezing. In some circumstances, reservoir drawdown is done to control aquatic plants (see Cooke et al. 1993), which certainly indicates its impact on some aquatic plants. It is therefore an expected result that drawdown would impact aquatic plant populations, especially if it occurred during periods when freezing conditions occurred or shortly after growth began.

Most of the main basin of the lake contains coarse-grained substrates and aquatic plant densities are generally low as is species diversity. There are higher densities and greater diversity in the two embayments, but these areas also have a community that tolerates late-winter drawdown. If there were no periods of drawdown, it is expected that the two embayments in particular would be colonized by additional species and the macrophyte beds would likely be more extensive. In addition, the embayments and shallow areas of the lake, if stable in water level, represent filters for incoming nutrients. The possible interaction of nutrient dynamics with aquatic populations is unknown, but these inshore areas are important contributors to ecosystem metabolism (Wetzel 1990).

By restricting the area that can be colonized by benthic invertebrates, drawdown impacts their populations. Even in cases where there are increases at depth in densities, those increases cannot compensate for the areal loss in invertebrate populations. The decline in population density represents the total mortality over the period influenced by drawdown. During this time there will be some natural mortality and the effect of the drawdown is superimposed. The mortality observed below the depth affected by drawdown was less than that found in the drawdown zone indicating that a considerable amount of mortality observed is related to drawdown.

The drawdown of the lake creates a significant periodic disturbance of the community and it appears that the existing community takes several months to recover from the disturbance. Recovery is in reference to the existing benthic invertebrate community. It is not known what community might be present if there was no major fluctuation in water level that was a periodic disturbance. In addition, if aquatic plant communities changed under a hydrologic regime without drawdown, then benthic invertebrate communities would be expected to respond to changes in habitat.

Wintertime drawdown would be expected to affect fish populations as well. Fish typically spawn in shallow areas and aquatic plant beds are important areas for juvenile fish. It is likely that drawdown affects fish in several ways by restricting the available cover associated with aquatic plants and by reducing the biomass of prey items during the recovery period, which includes the critical time period associated with spawning and the development of young-of-the-year fish. It is not known whether there is a change in the quality of the food resource available for fish that might also be associated with the drawdown.

Community structure in lakes, reservoirs and ponds is determined by physical factors, such as anoxia or dessication, and biological effects resulting from ecological interactions (Wellbom et al. 1996). While there are several uncertainties associated with the impact of the drawdown on biological populations and it is not possible to quantify the unpact. drawdown has a periodic impact on benthic invertebrates and it has created an aquatic macrophyte community that is tolerant of drawdowns. Clearly the strong physical factor of periodic drawdown is shaping the community. It is likely that the aquatic macrophyte and benthic invertebrate communities would include additional species and greater densities particularly during drawdown and post-drawdown periods. How these changes would be translated into fish biomass is unknown. In addition, because of the tolerant macrophyte and benthic invertebrate communities that result, and the influence of drawdown on physical and biological structures that represent habitat for organisms, as well as ecological interactions, it is expected that the structure of the aquatic community would change if drawdown periods were reduced in magnitude or eliminated.

13.0 Feasibility analysis

The two main issues related to improving conditions in Lake Wissota involve the operation of the reservoir for power production, including drawing down the lake during late winter and its associated impacts on lake biota (described in the section on unpacts of drawdown), and nutrient loading to the reservoir from non-point source runoff and a wastewater treatment plant at Cadott. The study did not address watershed land use and other point sources along the Upper Chippewa River basin, (cf. WI DNR PUBL-WR-345-96-REV) or in the Lower Chippewa River basin, e.g. McCann Creek and Fisher River Watershed, including the Cornell Wastewater Treatment Plant, (cf. WI DNR PUBL-WR-216-96-REV) that affect the main basin of Lake Wissota. It also did not examine and the Gilman Wastewater Treatment Plant in the Upper Yellow River watershed.

Based on water quality observations within the lake during 1993 and other records of nutrient concentrations, the Chippewa River basin runoff results in substantial loading of nutrients to the main basin of the lake. Given the flow of water entering the reservoir from the Chippewa River and water quality results from 1993, it appears that the Upper Chippewa River Basin has a dominant influence on water quality conditions in the main basin of the lake. In fact, US EPA (1974), based on monthly sampling of watershed inflows, estimated that 78% of the annual total phosphorus to the lake was via the Chippewa River.

Based on conclusions from data collected in 1989 on Lake Wissota, where the observation was made that water quality tended to be worse in the two embayments than in the main basin of the lake, the WI DNR decided to address water quality issues related to Little Lake Wissota and Moon Bay and the land use associated with their watersheds. Therefore, the design of the study and the analysis of watershed land use was restricted to the Yellow River and Paint Creek watersheds entering the two embayments of the lake. These areas are largely within Chippewa County.

The Upper Yellow River Watershed is primarily forested, with the lower portion having significant agricultural activity as the river drains to the west towards Chippewa County. No information is available on the Yellow River below Chequamegon Waters Flowage, although it is suggested in the Lower Chippewa River Basin Plan (WI DNR 1996) that an evaluation of the effects of polluted runoff from Babit, Elder and Hay Creeks be done. In the Lower Yellow River Watershed, agriculture is the predominant form of land use. Whereas 94% of the original vegetation was estimated to be fores% that cover has been reduced substantially so that nearly 60% of the watershed is agricultural (Lower Chippewa River Basin Plan). Although the number of farms in the Chippewa County is declining (section 5. 0), agricultural runoff remains a key issue for surface water quality.

The lower section of the watershed also includes a point source associated with the Cadott Wastewater Treatment Plant. -Because of the analysis of nutrient export related to land use (Appendix A), water quality conditions including dense blooms of blue-green bacteria observed in the lake 1989 and in 1993, the primary questions involve what reductions in nutrient runoff from the watershed and from the single point source in the Lower Yellow River watershed can be accomplished do would result in improved conditions in Moon Bay and Little Lake Wissota. What is the feasibility of reducing runoff from the watershed and from the treatment plant at Cadott and what would the impacts be on water quality m the lake?

13.1 Scenario 0 - Do nothing

Untreated waterbodies experiencing eutrophication are not expected to improve and often conditions worsen resulting in greater expenditures later (Rast and Holland 1988). In the case of the Lake Wissota watershed, some changes in agricultural practices and a reduction in the number of farms may result in some small improvements in reservoir water quality. However, given the levels of phosphorus and nitrogen observed during the study, in 1989 and in 1972-73, the lake is expected to remain in an "eutrophic" condition and experience frequent algal blooms in summer.

13.2 Scenario 1 - Identification of nutrient sources from bacterial sampling of strewns

The sampling of streams for coliform bacteria, including several samplings following storm events, resulted in the identification of drainage subbasins that appear to produce significant runoff of nutrients. Paint Creek and Lotz Creek were two areas that showed evidence of very high levels of coliform bacteria. The results of the bacterial sampling also demonstrate the major role of storm events in determining the runoff of nutrients. As would be expected from numerous studies (see e.g. Dunne and Leopold 1978, Rekolainen 1993, Likens and Bormann 1995), storm event runoff dominates the transport of particulate material loading to streams, but it is highly non-linear. The majority of loading occurs in a small fraction of time and in a small percentage of total annual runoff. The impact of storm runoff on Moon Bay and Little Lake Wissota is demonstrated visually in Figure la.

Based on the identification of drainage subbasins that might be considered "hotspots" of animal waste runoff, the Chippewa County Land Conservation Department has initiated a survey of barnyards and other practices in a number of drainage areas. Once the sources of the runoff have been identified, the feasibility of management practices to control the runoff can be considered. As indicated in Schultz et al. (1996), "it is likely that a small number of high discharge operations located in

close proximity to the strewn network may contribute a high percentage of the total barnyard phosphorus load in select basins." Moreover, 25% of the 530 active barnyards in the Lower Yellow River basin are located within 150 m of perennial streams. These two facts suggest do significant reductions in barnyard P export might be realized by focusing on high discharge operations close to perennial stream channels, especially along streams where bacterial concentrations soared during storm events (e.g. Lotz Creek).

13.3 Scenario 2 - Mass balance estimates of reducing watershed runoff of nutrients

Nonpoint source pollution is a major contributor of nutrients to lakes and reservoirs (Rast and Lee 1983). Most of the phosphorus load to Lake Wissota is estimated to be associated with upland land use, and with runoff from barnyards (see Appendix A). Some improvement would be expected simply as the number of farms in the watershed decline, as menfioned as a trend throughout Chippewa County earlier in this report. However, that is not expected to result in a substantial reduction in nutrient loading to the stream because the percentage reduction in farms to date has been <10%.

Runoff from row crops can be reduced by best management practices, whereas runoff control for grass, hay and pasture is not cost-effective (Dan Masterpole, personal communication). Barnyard runoff also can be controlled. The current total phosphorus load in the Lower Yellow River basin is estimated 26,311 kg/yr and 5686 kg/yr for the Paint Creek basin. These figures are not measured or validated by in-stream measurements, but are estimates are based on phosphorus export coefficients applied to different land use classes mapped using satellite imagery. The phosphorus delivery coefficients used in making these estimates were 1.0 kg/ha-yr for row crops, 0.30 kg/ha-yr for dairy-based grassland/pasture/hay and 0.05 kg/ha-yr for forested areas. The barnyard runoff estimates are based on transfer of results from a more detailed study of nearby Duncan Creek (Chippewa County Land Conservation Department 1991) and applying it to the Yellow River Watershed. Both are reasonable estimates of P runoff that can be used in modeling, but do not include nitrogen loading that might be required as an input vaniable in modeling.

Soil erosion influences the long-term productivity of agricultural soils, but the runoff of nutrients can have significant impact on downstream watercourses. Several studies have indicated the feasibility of using different agricultural practices to reduce nutrient runoff. For example, Sojka et al. (1993) demonstrated that changes in irrigation practices and tillage can result in reduced sediment loss and transport of nitrogen and phosphorus. Sharpley et al. (1993) found similar results. In Illinois, altemative com and soybean tillage and rotation schemes have been evaluated using the EPIC model (Phillips et al. 1993). Those results indicated that no-till significantly reduced soil erosion rates and losses of nitrogen and phosphorus. However, under fertilized soil conditions, P losses to surface water were higher under no fill conditions, presumably related to a lack of incorporation of fertilizer, but losses associated with sediment were lower.

Using Best Management Practices (BMPs), it is thought that runoff of phosphorus from barnyards can be reduced by 80% within the Yellow River and Paint Creek basins and that runoff from rowcrops might be reduced by an average of 50% (Dan Masterpole, Chippewa County Land Conservation Department, Buzz Sorge, WI DNR, personal communication). Given the areal extent of land use classes in the two basins and the distribution of barnyards, and the estimates of land use and P runoff, implementing BMP's to the extent it can be estimated that such practices could result in a reduction of total phosphorus export from each subbasin by approximately 28%. This would lower loading to 19,087 kg/yr for the Lower Yellow River basin and 4, 101 kg/yr for the Paint Creek basin.

One difficulty in making a mass balance estimate of the change in concentrations in the lake is that the background phosphorus concentration in the Yellow River is unknown, as is a problem in modeling any potential pollutant having some background concentration (cf. Brakke et al. 1989, Brakke and Henriksen 1989). The DNR currently lacks water quality information for upstream areas such as the Chequamegon Waters Flowage or the Yellow River prior to any influence from agricultural areas to the west that are within the area of study. Without such information, the background concentrations of nutrients, which are required to set a baseline beyond which reductions in-lake concentrations could not occur, can only be estimated from other sources or from export coefficients developed for forested catchments (see several references in Panuska and Lillie 1995). Export rates from forested watersheds averaged 0.09 kg P/ha-yr for the watersheds cited in Panuska and Lillie (1995), which might represent the lower limit of phosphorus concentrations being approximately 10-20% of present values if the entire watershed were forested. However, with forested areas currently amounting to only 30-37% of the two subbasins, current phosphorus concentrations even assuming no substantial agricultural activity would be expected to be considerably higher. This level would represent that lowest possible concentration if no agricultural activity occurred in the area.

Other data suggest that lakes and reservoirs in the region might be expected to have relatively high "background" concentrations of total phosphorus. Omernick et al. (1988) indicated that the phosphorus "ecoregion" containing Lake Wissota has typically very high P concentrations, most being > 50 ug/l, as opposed to adjacent areas with different surficial geologic characteristics. Part of the reason for the higher phosphorus was related to the extensive agricultural practices in the area.

Mass balance models for phosphorus are based on assuming the receiving water column is a continuously mixed reactor (see Reckhow and Chapra 1983). Volume is held constant and mass balance is determined by inflow and outflow rates, as well as in-lake processes. These relationships are the basis for estimates of equilibnium phosphorus developed by Vollenwieder (1969) and modified by Dillon and Rigler (1974). Steady-state mass balance models have been verified for a large number of lakes. They are also the basis of empirical observations by Walker (1986, 1987).

Depending on the change in transport rates to streams and in stream uptake of nutrients, it might be expected that control of barnyard and runoff from row crops could result in a reduction of 'in-reservoir concentrations by up to 28% (see Table 8). A reduction in watershed runoff would translate into lower concentrations of phosphorus in the subbasins during the summer growth period for algal populations, with the magnitude of the decline dependent on the seasonal distribution of phosphorus export from-the land surface. However, it is important to note that regardless of the modeling approach used, this uncertainty remains: tying to determine the load during the summer growing season, especially related to barnyard runoff, and its affect on 'in-reservoir phosphorus concentrations during the time of mid- and late-summer algal blooms.

The use of BMP's is feasible and should result in lowered concentrations within the lake that reduce P significantly and decrease the frequency of blue-green algal blooms. However, it is expected that given the total concentrations of nutrients observed in the two embayments and in the main basin in both years, mass balance estimates suggest that nutrient concentrations will still remain high enough to produce "eutrophic" conditions in the lake (e.g. total phosphorus may still fall in a range from 60-70 ug/l, which would be expected to generate algal blooms, although at reduced frequency).

13.4 Scenanio 3 - mass balance estimate of reductions in effluent from Cadott Wastewater Treatment Plant point source

The Cadott Wastewater Treatment Plant represents a point source of pollution to the Yellow River. Because of compliance monitoring, its influence on concentrations in the lake can be estimated quite simply. For example, the volume of Moon Bay is 4376 ac-ft (1,670,000,000 l).

At a discharge of 0.17 Mgd, as observed in August 1996,

vol Moon Bay = 2599 days
vol disch @ 0. 17 Mgd

Assuming that all P discharged from Cadott is loaded to Moon Bay in one month, and that it adds directly to the concentration in the bay, an estimate of the influence of the Cadott Wastewater Treatment Plant is as follows.

At a loading of 267 lbs P/mo 121 kg/mo = + 7.25 ug/l in Moon Bay

At a loading of 193 lbs P/mo 88 kg/mo = + 5.3 ug/l in Moon Bay

By requiring 75% reductions in P concentrations in effluent from the treatment plant at Cadott, which are feasible using appropriate technology, this could reduce concentrations in Moon Bay by 4-7 ug/l. This range represents a relatively small fraction of the total concentration observed in Moon Bay, but it is a fraction that can be controlled. John Panuska (WI DNR, personal communication) agreed with this approach in estimating the impact of the Cadott treatment plant and its significance. At low flow conditions during periods of the summer, the treatment plant represents a larger fraction of the total load, suggesting that resulting improvement in P concentrations in Moon Bay would be greater during low flow periods with further controls placed on the Cadott treatment plant.

It is interesting to note that the National Eutrophication Survey (US EPA 1974) indicated that loading to Lake Wissota was three times that proposed by Vollenweider as "dangerous," but estimated that only around 2% of the load could be attributed to point sources at Cornell and Cadott. US EPA (1974) indicated that "phosphorus removal at the two point sources would have little effect on the trophic condition at Lake Wissota."

13.5 Other scenarios examined from model results using BATHTUB

A series of model simulations were run using an annual time scale to estimate the reductions in in-lake nutrient concentrations that might be expected from controls on nonpoint source runoff. Those analyses produced estimates for mean annual conditions and indicated that improvements in water quality would result from the application of BMP's in the watershed. However, WI DNR Western District was interested in trying to estimate summer or growing season concentrations (Art Bearnhart memo to Buzz Sorge, August 26, 1996).

John Panuska of WI DNR (Madison, WI) adjusted the model input parameters to a growing season and used recent estimates of phosphorus export coefficients for agricultural areas (Panuska and Lillie 1995) and developed the input files to be used for the BATHTUB model runs. WI DNR requested modeling runs that would evaluate reductions in row crop export, effluent controls at the Cadott wastewater treatment plan and the influence of low flow seasons on total phosphorus and chlorophyll a concentrations in Moon Bay (Buzz Sorge, personal communication).

Scenario 4. Reductions in row crop export of total phosphorus

Data from 1993 were considered to represent an average flow year (note: although runoff was very high in June, annual rainfall was 3" above normal). BATHTUB model simulations suggest a stepwise decline in Moon Bay phosphorus concentrations as row crop export of phosphorus declines (Figure 99). A 50% reduction in row crop export is believed to be possible within the watershed (Dan Masterpole, personal communication). With this level of reduction, total P is modeled to be reduced from 85 ug/l to 75 ug/l, indicating that there are appreciable additional sources of P in the watershed, but also demonstrating that conditions in the lake would be expected to improve. The frequency of chlorophyll a > 20 ug/l is reduced from 40% to slightly more than 30% if row crop export is reduced by half Chlorophyll a concentrations > 30 ug/l are projected to decline from 18 to 12%.

Scenario 5. Reduction in point source from Cadott wastewater treatment plant

The influence of the Cadott treatment plant was also estimated using BATHTUB for conditions in 1993. The model predicts that total P would be reduced by approximately 3 ug/l if the treatment plant effluent was reduced by 75% (note the mass balance estimates given earlier of 4-7 ug/1) (Figure 100). Some slight improvement in chlorophyll concentrations was also projected.

Scenario 6. Point source reductions and row crop export reductions

When coupled with a 50% reduction in row crop export, a 75% reduction in treatment plant effluent was projected to result in a only slightly more reduction in total P concentrations than estimated for controls on row crop export (Figure 100).

Scenario 7. Reductions in row crop export and point source effluent during a low flow year

Low flow conditions were estimated based on the observed frequency of occurrence and magnitude of discharge for the period of record at the gauging station at Cadott. Using the BATHTUB model, total P concentrations for a low runoff year (represented as monthly flow experienced each five years) were higher than predicted for 1993 (91 vs. 85 ug/1) (Figure 101, Table 9). As a consequence, the percentage of time represented with chlorophyll a concentrations > 20 and 30 ug/l were higher. Reductions in effluent from the Cadott wastewater treatment plant of 75% during a low water year was predicted to result in a 2 ug/l decrease in total P concentrations in Moon Bay and a change in the frequency of chlorophyll a > 20 ug/l from 47 to 44%.

The model results suggest that even with very significant changes in runoff from agricultural areas (a 50% decline in row crop export) and reductions of 75% in the effluent concentrations of P from the Cadott wastewater treatment plant, that total P concentrations might be reduced by only 10 ug/l. Some improvement was also predicted for chlorophyll a concentrations. However, the model also suggests that during lower runoff years, water quality conditions would likely mask the effect of runoff controls and higher P concentrations would be seen even though runoff from the land surface and the Cadott treatment plant had been reduced considerably (Figure 101, Table 9). Annual, mass-balance estimates suggest somewhat greater reductions, particularly when barnyard runoff of P was included along with control of row crop export using BMP'S.

Watershed BMP's are feasible as is reduction of effluent concentrations of P from the Cadott wastewater treatment plant. Both are expected to result in improvements in water quality in Lake Wissota. Given the loading from the larger Chippewa River basin and background loadings from the Paint Creek and Yellow River watersheds, it is expected that total phosphorus concentrations will still range from 60 - 80 ug/l depending on the year and the flow conditions. These concentrations will continue to generate blooms of blue-green bacteria.

13.6 Uncertainties in BATHTUB model estimates of the effect of reducing export from land runoff and the Cadott Wastewater Treatment Plant

Given the complexity of non-linear relationships between flow and nutrient load, the diffuse nature of non-point source runoff, the general lack of detailed information on input variables or loading estimates and their distribution in time and space and the lack of quantitative information on the impact of changing agricultural practices on external loads, it is a challenge to model the expected changes in surface water quality that might result from reducing the export of nutrients from the landscape and from point sources. Moreover, as has been found in detailed modeling exercises elsewhere, improved agricultural practice can be swamped by changes in climate and differences in flow conditions.

Modeling is useful to illustrate the potential changes that might occur in response to a specific management activity. Two major concerns in any modeling are the propagation of errors, especially with many linked and estimated terms, and the eventual interpretations of results from modeling. Sensifivity analysis, such as might be accomplished by Monte Carlo simulation of input variables and multiple model projections, was beyond the scope of this study. Interpretation of model outputs must recognize the uncertainties associated with input variables. For example, the land classification (Appendix A) had an accuracy of approximately 85%. Land use classes were given P export coefficients chosen from other studies but not validated for. the Yellow River and Paint Creek watersheds in order to estimate the runoff of nutrients from each subbasin and watershed. Flow and these loading values were adjusted to estimated equilibrium P concentrations in the lake and then model simulations were run. Each of the steps involves uncertainty, which should be considered in evaluating model outputs.

The BATHTUB program was selected by the WI DNR for application in estimating the potential effect of changes in land use on water quality conditions in Lake Wissota. The model was developed over a number of years by Walker (I 987) based on analyses of U.S. Army Corps of Engineers reservoirs, often in the southeastern U.S. It is based on a series of empirical relationships developed between phosphorus loading and phosphorus sedimentation, as well as phosphorus concentrations and predicted chlorophyll a concentrations, which included the influence of non-algal turbidity, so common in reservoirs. Walker (1987) developed three programs: FLUX, to calculate nutrient loadings; PROFILE, to display constituents in the reservoir; and BATHTUB, which calculates the nutrient balances in the system, predicting the response of a reservoir (see Cooke et al. 1993). BATHTUB runs in a DOS-shell and not under Windows, requiring considerable memory for its application. Applications of BATHTUB have not been cited frequently in the literature.

There are several possible subroutines or models within the BATHTUB program to select as options to describe various relationships depending on whether a reservoir might be phosphorus- or nitrogen-limited or phosphorus-chlorophyll relationship works best. The BATHTUB program is set up to run on an annual basis and also for application to situations where the land surface is not frozen or snow covered, but exposed throughout the year.

Initially, the BATHTUB program was to be run on an annual basis for Lake Wissota, however paying attention to the two embayments and not the main basin of the lake. The program was fun using specific sub-models suggested by Panuska (1996, personal communication) based on experience with lakes in Wisconsin. Model subroutines selected were #1 for phosphorus, #5 for chlorophyll due to the presence of blue-green bacteria in high concentrations and #1 for Secchi disc.

Several model simulations of changes in water quality related to changes in land use were done for Moon Bay and the Lower Yellow River Watershed and for Little Lake Wissota and the Paint Creek watershed using an annual time step. These results were presented to WI DNR in July 1996. However, WI DNR believed based on 1989 data that it was more important to focus on the summer growing season and to run the model for that period only. The rationale for this revised approach was that with longer residence times in the two embayments, water quality conditions would worsen.

Two BATHTUB files were created by John Panuska of WI DNR. The first used 1993 water quality data along with estimated flows for the year and loading. Tributaries were established to represent: 1) direct loading to Moon Bay; 2) non-point source loading between Miller Dam and Moon Bay; and 3) the Cadott WWTP and discharge from Miller Dam. The 1993 flow to Moon Bay was estimated from gauged flows at Cadott adjusted to Moon Bay's watershed area based on representative flow conditions in 1946, a year when annual mean flow and June-September rainfalls were as close to 1993 as possible. The concentration of P in the Miller Dam outflow was set by trial and error to match the observed seasonal mean total P in Moon Bay. After fitting the total P value, the chlorophyll model was calibrated to observed conditions (Panuska 1996).

The second file was used to represent low flow conditions. The flow in the Yellow River at Moon Bay was assumed to be 45 cfs from June-September with a total P of 30 ug/l (the estimated background total P). Non-point source runoff volume was 0.01 m, which is slightly higher than the runoff from the lowest flow year in the 1943-1961 record at Cadott. The Cadott WWTP was assumed to be discharging at 1996 flow and at a concentration of 6.3 mg/l. A total P value of 20 ug/l was assumed for Miller Dam. The predicted in-lake concentration at low flow was 61 ug/l, reflecting the lower runoff contribution from the land surface (Panuska 1996).

While the model was developed to run on an annual basis, it can be "tricked" to run over a shorter time period. However, several questions arise in adapting the model for use in a specific season during the year, such as the period from June to September. Specifically, how are the annual exports of phosphorus actually apportioned on a seasonal or monthly basis? The pattern win be influenced by a large number of factors, including the distribution of storm events, especially as they are related to the exposure of the soil surface to erosion and as affected by changes in cover with growth of crops over time, seasonal differences in runoff, and changes in agricultural practices during the year, especially those related to the spreading of manure on fields related to the surface cover at the time of application. Most of the phosphorus transported from agricultural land is bound to soil particles, and is highly unavailable to biological organisms. However, the fraction of available phosphorus can increase during snowmelt runoff (Rekolainen 1989). As a consequence, application of manure to frozen fields can result in high runoff of available phosphorus to the reservoir system. Manure management is a key issue within the watershed, and given the increases in transport observed during spring runoff peaks, manure storage during winter and until ground cover is established should be considered as prudent practice.

The results observed by WI DNR in 1989 were assumed to represent low flow conditions. However, while the annual rainfall during 1989 was 3.92 in below normal, the rainfall during the period from June to September was only 1.00 in below normal. In fact, the departure from normal summer flow conditions occurred in September, only after the period when water quality conditions were observed to be poor during July and August (rainfall records sent from John Panuska to Buzz Sorge, October 2, 1996).

While 1993 annual rainfall was somewhat above average (+ 3.64 in), precipitation during the summer growing season suggested for modeling the two embayments was +3.05 in during that period. Therefore, 1989 conditions do not represent low flow during the summer and monitoring data were inadequate for low flow analysis (John Panuska, personal communication to Buzz Sorge and David Brakke October 2, 1996). Further, mass balance was suggested as the best approach for low flow conditions, especially for analysis of the influence of the Cadott WWTP.

One can modify annual estimates of export rates of phosphorus from land use classes or from barnyards and apportion the loads to a fraction of the year, but this may not adequately represent the actual seasonality of export or reflect the relationship between runoff, land use change, reductions in runoff from barnyards and in-lake nutrient concentrations. In particular, the proportional apportioning of the application of manure to agricultural fields if lower in summer could result in significant over-prediction of phosphorus reduction. Also, for low runoff years, one would expect less export from the landscape. At issue in the case of Lake Wissota is the relationship between storm events and water residence times, as well as the seasonal distribution of agricultural practices that affect runoff of animal wastes. How long after storms does the water remain in the two subbasins, which might allow for the development of bloom conditions of blue-green bacteria?

It is important to note that during 1993, water quality conditions in the main basin of the lake were as poor or worse than observed in the two embayments. Loading from the very large watershed of the Chippewa River upstream of Lake Wissota results in concentrations of nutrients in the main basin that are close to those found in the two subbasins, and chlorophyll a concentrations in the main basin exceeded those in the embayments during 1993. It is not known the influence of the high runoff during June 1993 and also August 1993 on this pattern, however, water quality in the main basin of the lake appears to be regulated in large part by input from the Chippewa River and not dominated by runoff entering via Little Lake Wissota and Moon Bay.

The BATHTUB program has limitations where runoff from agricultural areas is being predicted unless that export is quantified or estimated with some precision and is then used as inputs to the model. According to Rekolainen (1993 and personal communication in Eau Claire and Helsinki, Finland with Seppo Rekolainen and Lea Kauppi), who has published extensively on phosphorus and nitrogen loads from agricultural areas, as well as estimating storm loading of nutrients, the CREAMS and AGNPS models are designed to estimate agricultural losses to surface waters and best equipped to conduct modeling of their impacts. As expected, Rekolainen (1993 b) indicated that estimating non-point source losses to surface waters requires a great deal of resources because of the very high spatial and temporal variations in non-point sources. With sufficient data, outputs from models such as CREAMS and AGNPS could be coupled to the Universal Soil Loss Equation (or modification) and then to BATHTUB or other models to predict in-reservoir nutrient concentrations, but he did not think the input data collected on streams in the watershed were adequate to run those models. Without running something equivalent the input data relating watershed conditions to lakewater concentrations are useful in an exploratory sense, but there are many uncertainties that must be considered in making careful interpretation of model outputs or accepting the model projections as precise estimates.

In addition, due to the diffuse origin of nutrients loaded from agricultural areas, estimation of their fluxes is exceedingly difficult. In addition, the transport of nutrients is highly dependent on climatic variables and on the highly non-linear

relationships of runoff and load. The nutrient load from land areas is highly basin-specific. Moreover, sampling at an intensity of up to 12 times per year is in no way sufficient to explain reliable estimates of mean concentrations of inflow P. For example, Walling and Webb (1982) found that the sampling interval required for estimating the mean annual concentration to within plus or minus 20% would be 2 hours for suspended sediment and 8 days for nitrate. Because a large fraction of the phosphorus load to the lake is associated with suspended sediment, this suggests that a very high sampling frequency would be require to provide estimates of mean annual phosphorus concentrations. In fact, because most of the particulates are transported during high runoff events, sampling strategies have been developed to increase frequency during high flow periods (cf. Young et al. 1988)

Equilibrium and outflow P concentrations in reservoirs are most sensitive to the inflow P concentrations because of relatively short residence times in reservoirs, which minimizes the sedimentation term for P (see Walker 1987, Cooke et al. 1993). As residence times decline below 0.2 yr, as is the case for Lake Wissota, the inflow term becomes most important. No information is available from sampling to set the inflow P concentration with confidence as an input to the BATHTUB model.

Additional uncertainties

There are several additional uncertainties in estimating the influence of changes in land use and controls on barnyards using the BATHTUB program that should be identified. These uncertainties include and are not restricted to the following, which in some cases represent missing input data that were not collected as part of the initial study design and sampling proposed and or as modified and eventually funded by the US EPA:

1. no stream gauging information was available for the period covered by the study on Paint Creek or the Yellow River as inflows to the two embayments;
2. no in-stream nutrient concentrations were available for the streams along their course, which might indicate areas where nutrient additions occurred, or as they entered the embayments of the lake as P inflow terms;
3. no 'm-stream data were available for the Yellow River prior to influence by agricultural areas in the Lower Yellow River Watershed;
4. there are indications from the water quality data that the lake is nitrogen-limited, at least in part, but there are no estimates available for export rates of nitrogen that could be used as inputs to a model or what might be adequate to run the phosphorus or nitrogen limited subroutine of the model;

5. if nitrogen limitation occur's, the relationship between phosphorus and chlorophyll a concentrations is affected for the in-lake calculations and requires a different subroutine;
6. if nitrogen limitation is a frequent or dominant phase, as suggested by the data, including the N:P ratios and the high proportion of dissolved P: total P, the improvement that might be expected in water quality conditions with decreased runoff of phosphorus will only result after phosphorus limitafion occurs, therefore making the ratio of N:P reduction in land runoff a critical factor;
7. although N limitation may be a key to reservoir functioning, there is no information on watershed sources of nitrogen or estimates of N loading to the embayments;
8. reservoirs respond quickly to storm events, with the magnitude and frequency of storm events in the watershed being important factors, because watershed export of particulates (with most phosphorus expected to move associated with particulates) and stream runoff is highly dependent on storm episodes, as indicated also by the bacterial sampling done for the study; and
9. the relation between the incoming streams and the two embayments could not be calibrated due to the lack of available data resulting in the need to try to simulate the in-lake concentrations (in addition, many simultaneous changes could produce similar responses);
10. the models used to generate pollutant load estimates have not been validated for the study area and contain uncertainties associated with the estimates both for land use exports and barnyard estimates (see Appendix A);
11. the cur-rent land use study did not document the extent of the area where BMP's are being used, which makes it difficult to determine exactly the potenfial for further reducing nonpoint source runoff (see Appendix A); and
12. little information is available for a quantitative assessment of the effectiveness of BMP's in reducing nutrient export.

Because of these uncertainties, modeling results should be examined with caution. Without more monitoring data it is very difficult to do much more with modeling (John Panuska, WI DNR - Madison, personal communication to Buzz Sorge, September 26, 1996).

11.8 Conclusions from modeling and other assessments related to land use and phosphorus export

In spite of the many uncertainties identified above related to developing precise estimates of water quality conditions or predictions, taken together, a number of pieces of information and different approaches suggest that agricultural practices within the Yellow Creek and Paint Creek watersheds lead to higher nutrient concentrations in Lake Wissota and its embayments than would be the case without appreciable agricultural activity. Land use evaluations and estimates based on export coefficients suggest that upland land use and barnyards are the major sources of phosphorus load downstream. Stream sampling for coliform bacteria suggest high influences of agricultural activities in some areas. Priority areas to evaluate based on land use runoff estimates include the subbasins: Drywood Creek drainages, Middle Yellow, Middle Paint and Stillson, where P loads for all sources are estimated to be high (> 0.3 kg/ha-yr). In-lake monitoning and modeling efforts also suggest that nutrient concentrations exceed levels that would be expected in the absence of heavy agricultural use within the drainage basin for rowcrops and dairy farming. Common sense suggests that surface water loading should be reduced by all practical means to result in the best possible water quality in Lake Wissota.

The point source at Cadott can be controlled, reducing its contributions by up to 75%. However, nonpoint sources along the watershed certainly dominate the flux of nutrients to streams and the reservoir. The Chippewa County Land Conservation Department should work cooperatively with the WI DNR and with landowners to identify major sources of non-point source pollution and engage in best management practices for agriculture and other activifies within the Yellow River and Paint Creek watersheds. In addition, the WI DNR should work with other counties and municipalities to reduce the non-point source and point source loading in the Upper Yellow River, McCann Creek and Fisher River and mainstem Chippewa River watersheds.

14.0 Selection of lake restoration alternatives

Rast and Holland (1988) have developed useful framework for considering and selecting eutrophication control measures. Ryding and Rast (1988) developed suggestions for data required for evaluating lake and reservoir trophic condition. Both provide a context for the evaluation of possible mitigation measures for the Yellow River and Paint Creek watersheds.

External vs. In-lake controls

Restoration techniques for lakes and reservoirs fall into two main classes: external controls on nutrient sources and in-lake methods for reducing nutrient concentrations (see Cairns et al. 1992, Cooke et al. 1993). However, given the geologic setting of Lake Wissota and the limnological conditions observed in the two embayments and the main basin of the lake, in-lake methods do not appear to be feasible in reducing nutrient concentrations. Moreover, managing "symptoms" occurring in the lake is ineffective and prohibitively expensive in systems with short residence times and continual loading of nutrients from watershed areas.

Some recent experience suggests possibilities for altering the composition of aquatic communities without changing nutrient income. One of the major challenges of aquatic ecology is to isolate the influence of resources, e.g. algal nutrients as a "bottom-up" effect, versus consumers, e.g. fish populations and algal grazers as a "top-down" effect, on algal populations. Trophic interactions among aquatic communities may result in changes in the composition of algal and other communities and modification of the communities can bring about changes in algal populations ("biomanipulation" as coined by Joseph Shapiro, see Carpenter and Kitchell 1992, 1993). While these various interactions have been demonstrated in mesocosms and small lakes (see Vannie and Layne 1997, Vanni et al. 1997), applications in altering phytoplankton communities in complex reservoir systems have not been accomplished. Moreover, while enhancing grazing by zooplankton might control algal biomass over a range of P concentrations and input rates, it may not result in the suppression of cyanobacterial blooms (Carpenter et al. 1995). In addition, shorter residence times in reservoirs might limit the influence of grazers. Therefore, any improvement in reservoir water quality should best focus on realizing appreciable reductions in lakewater concentrations of nutrients that would come from regulating external sources of nutrients.

Control methods for external loading of nutrients

Control methods for external nutrient loading to lakes or reservoirs involve 5 main categories or options (Cairns et al. 1992):

1. stream or wastewater diversion;
2. municipal wastewater treatment;
3. product modification (e.g. ban on P in detergents);
4. treatment of inflow strewns; and
5. land use practices.

Of these, the first and third are not practical or would not result be expected to result in sufficient reduction in nutrient income to alter water quality due to the existing sources within the watershed.

The second option can be considered but only for the Cadott wastewater treatment plant and other plants located along streams or rivers contributory to the reservoir system. The estimated reduction in P concentrations with enhanced treatment at the plant were summarized in section 11. Costs for requiring this modification of the Cadott Wastewater Treatment Plant are being evaluated by the WI DNR (Buzz Sorge, personal communication). The cost effectiveness and the influence on in-lake concentrations of nutrients for requiring additional treatment at other wastewater plants is not known.

Treatment of inflow strewns

Treatment of inflow streams could be considered but the costs would be relatively high due to the large volume of inflowing water, especially via the Chippewa River, but also the Yellow River and Paint Creek basins during snowmelt or storm runoff periods. This would involve the design of pre-lake interceptor systems, including detention and retention basins, artificial or natural wetlands and in-stream phosphorus precipitation. Detention basins have been evaluated primarily for urban areas (e.g. Walker 1987b) and pre-reservoir retention basins have been used in only a few areas. Some experience has been gained with artificial and natural wetlands, but without any comprehensive evaluation of the benefit. In-stream phosphorus precipitation has been accomplished for some lakes, but is usually not a cost-effective approach, especially when the volumeof inflowing water is high, as in the case of reservoirs (Cairns et al. 1992).

Agricultural-related non-point source runoff

Nutrient runoff from agricultural watersheds is diffuse and complex across a landscape. In addition, it is highly dependent on a number of environmental factors associated with soils, slope and topography and climatic variables creating pulses of loading. Managing nutrient reductions must be done in the context of the watershed and recognizing the institutional constraints and framework.

Mitigation measures for reducing nutrient concentrations include what are known as Best Management Practices (BMP's). These practices have been developed in part to improve agricultural production while minimizing the impacts on downstream areas by decreasing losses of soil, nutrients or potentially other contaminants from croplands and pastures. The effectiveness of the various practices appears to vary considerably and many are still in a stage where they are being developed and modified, with little or no quantitative information available on their ability to reduce nutrient or soil loss. As such, making precise predictions of their influence is not possible at this time.

As pointed out by Cairns et al. (1992), "although BMP's seldom provide the complete solution in restoring degraded lakes, they are key elements in an evolving strategy that recognizes that lakes can be managed and protected effectively only in the context of the watershed in which they exist." This statement strongly suggests the management of Lake Wissota in the context of its watershed and with the application of techniques appropriate to the situation.

Common sense suggests that prudent agricultural practice should minimize the impact on runoff waters and that as a result some improvement in reservoir water quality can be accomplished.

The treatment of agricultural runoff with BMP's involves a number of possibilities that might be applicable to the Lake Wissota watershed (Cairns et al. 1992):

1. Runoff controls to change peak flow and volume
   a. no or minimum tillage
   b. winter cover crop
   c. contour plowing and strip cropping
   d. terraces
   e. grassed outlets; vegetated borders
   f. detention ponds
2. Nutrient loss controls
   a. timing and frequency of fertilizer application
   b. amount and type of fertilizer used
   c. control of fertilizer transformation to soluble forms
   d. crop rotation with legumes
   e. storage of manure during winter

Many of these practices are feasible and can be implemented within the Yellow River and Paint Creek watersheds. Some practices are already underway but the impact of them to date is unknown (Dan Masterpole, personal communication). Of those potential pracfices listed above, no or minimum tillage and strip cropping or contour plowing would result in lower transport of sediment and particulates to stream channels, reducing loading of particulate phosphorus to the system.

Animal waste

Animal waste appears to be a significant problem within the Yellow River and Paint Creek drainages (see Appendix A). A variety of options should be considered to minimize the runoff of nutrients associated with animal waste. The application of manure to fields in the watershed should be carefully examined to determine whether holding requirements and seasonal applications should be considered. In addition, barnyard runoff should be examined and provisions made

14.1 Developing an effective management strategy for Lake Wissota

The current status of Lake Wissota has been defined. The reservoir and its two embayments have elevated phosphorus and nitrogen concentrations resulting in algal densities that reduce transparency to low levels. Blooms of blue-green bacteria are frequent.

The watershed is the most reasonable scale to focus on restoration efforts. At the suggestion of the WI DNR, this study addressed only that portion of the Lake Wissota watershed entering the two embayments, including only the lower drainage portion of the Yellow River system. A number of alternative strategies have been identified and several have been evaluated through different modeling approaches. Uncertainties associated with the modeling approaches have been identified.

In order to analyze the costs and benefits associated with meeting certain water quality goals, those goals need to be defined. Some constraints have been identified in limiting the overall improvements in water quality, however no water quality goals have been set for the Wissota system by the WI DNR in the ater Quality Management Plan for the Lower Chippewa River Basin (WT DNR 1996) or developed for this project. Public participation is required to develop these goals.

Improvements in water quality can result from a combination of BMP's and point source controls, but it is not known to what goal or desired future condition for the reservoir these estimated improvements are being compared to. In addition, it is not possible to develop sensible cost-benefit analyses in the absence of clear goals for water quality.

The development of an effective eutrophication management strategy requires the specification of water quality goals (Rast and Holland 1988, Rast et al. 1989, Cooke et al. 1993). These goals can then be communicated to the public along with information on the costs of meeting the goals through various practices within the watershed. At the same time, evidence collected as part of this study suggests that work be initiated to control the major, idenfifiable sources of nutrient runoff, particularly from barnyards, and begin to control those sources. Further, the institutional frameworks should be identified to minimize future problems related to eutrophication.

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