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Nutrient and Water Budget Modeling of the Petenwell Flowage Adams, Juneau, and Wood Counties Wisconsin

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Nancy B. Turyk, Senior Research Specialist
Dr. Byron Shaw, Professor of Soil and Water
University of Wisconsin-Stevens Point

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I would like to dedicate this paper to my parents who both passed away during the duration of this project. Through their years they shared with me their passion for the natural world, fostered scientific curiosity, and encouraged me to pursue my dreams. Indeed I was fortunate for the time we spent together.

ABSTRACT

The Petenwell flowage is a highly eutrophic impoundment located on the Wisconsin River in south-central Wisconsin. High nutrient discharge by municipalities and industries up-river results in blue green algae blooms throughout much of the summer, which creates a nuisance and hampers recreational uses of the flowage. The flowage acts as a nutrient sink, with more nutrients entering than are exiting the flowage. This helps control algae problems in down-river impoundments.

The Wisconsin DNR determined that a model to predict the responses of algae blooms and water clarity in the flowage to changes in inflowing total phosphorus could be a useful management tool. The model selected to be used was BATHTUB which was assembled using water quality data collected at five sites in the flowage and three tributary sites along with daily flow measurements. Data were collected monthly between May and October for two years.

The water quality data indicate that internal loading of phosphorus is occurring during summer months in the upper end of the flowage. Detailed information on the internal loading is currently not available, however, during the calibration of the model a loading estimate of 20 $mg/m^2/day$ within this segment best fit the water quality water quality in the reservoir down stream.

The model estimates indicate that the overall reservoir chlorophyll *a* bloom frequency will decrease and Secchi disk depth will increase with a reduction in total phosphorus inputs. A 50% reduction of total phosphorus inputs would reduce algal bloom frequency from 114 to 74 days between May and October. The reduction that would occur in the upper part of the reservoir would be from 171 to 134 days; in the lower part of the reservoir bloom frequency would decrease from 37 to 22 days.

Though this model provides a good general reservoir response tool, it could be further refined by collecting additional water quality data during high flow periods, as well as collecting data with greater frequency. Additional sampling should occur up-river to separate out specific sources of phosphorus entering the impoundment.

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INTRODUCTION

Petenwell Flowage is located on the Wisconsin River in Adams, Juneau and Wood counties. At 23,040 acres, it is the second largest body of water in Wisconsin (Harza, 1993). Throughout the years many studies have been conducted on the Petenwell Flowage; results of many of these studies were addressed in the 10 year management plan which was completed by the Wisconsin Department of Natural Resources (WDNR) in 1996. This plan was comprehensive, and therefore addressed issues pertaining to recreation, aesthetics, environmental quality, habitat, and biota within the reservoir.

Among other problems, Petenwell flowage is a very eutrophic impoundment which experiences excessive blue green algae blooms. These blooms are aesthetically undesirable, a recreational nuisance causing beach closings, and an environmental problem resulting in the reduction of dissolved oxygen, water clarity and occasional fish kills. Nutrient enrichment is the primary cause of the algae blooms (WDNR, 1996, Wisc R Power Co, 1993). Nutrients enter the flowage via the Wisconsin River, its tributaries, and through surface runoff from the Wisconsin River Basin, which extends to the Michigan border. The flowage receives drainage from 15,462 km^2 and 39 sub-watersheds WDNR, 1996 (Figure 1). Nutrient concentrations in the Wisconsin River begin to increase at Tomahawk and continue to increase as you move down stream to the Petenwell Flowage. The flowage is a nutrient sink retaining approximately 48% of the inflow phosphorus (P); an average of 947 kg P enter yet only 490 kg P exit the reservoir daily (WDNR, 1996). Primary point sources of P entering the Wisconsin River are municipal sewage treatment plants, paper industry, and agricultural processing plant effluent discharge (Kreitlow, 1991). These amount to 29 permitted discharge sources on the Wisconsin River above the flowage. Non-point sources of nutrients can be tied to agricultural and residential land use in the Petenwell watershed (Figure 2).

Nutrient sources from land uses include agricultural fertilizer, wind eroded air borne particulates, and lawn fertilizer and septic effluent from intensive housing development around the impounded Fourteen Mile Creek tributary (WDNR, 1992, Burkholder and Cuker, 1991; Daniel, et al, 1990; Sonzogni, et al., 1982, IES Report 105, 1979). Data suggest that adjacent non-point and direct tributary inputs of nutrients are significantly less then nutrients coming into the flowage from the main river.

Figure 1. Map of Wisconsin showing counties, surface hydrology, and the Upper Wisconsin River Basin which is approximately 15,462 km^2 and contains 39 sub-watersheds.

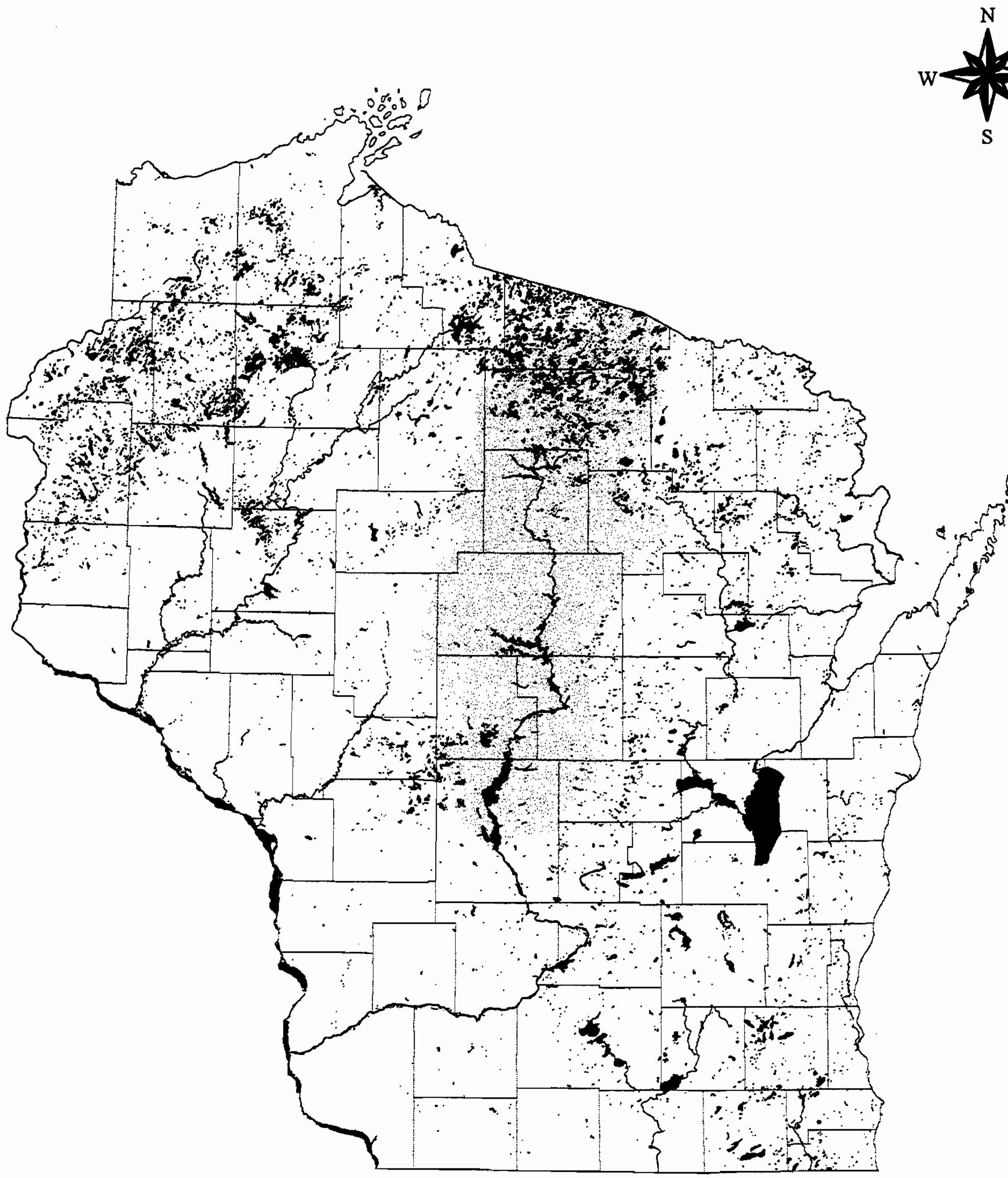
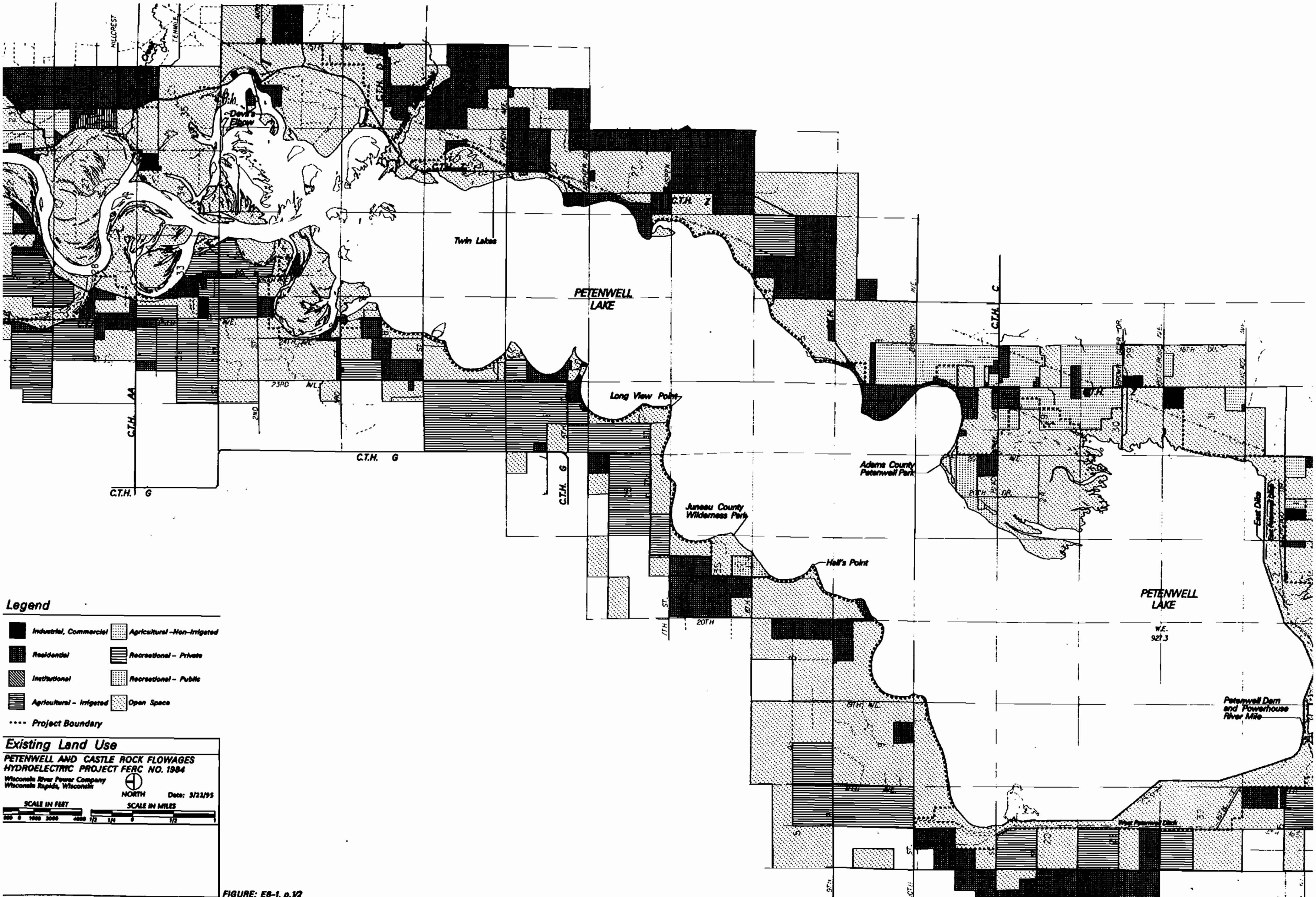


Figure 2. Map of the Petenwell Flowage in Wisconsin showing adjacent land use. From Wisconsin River Power Company, 1995.



Many upriver point source discharge sites are in the process of adding P removal systems to their effluent processing as a result of Chapter NR 217, Wisc. Admin. Code. It is desirable to determine if these reductions are sufficient to reduce algal blooms in the Petenwell. In 1997 DNR estimated Total P loading to the Wisconsin River above the flowage to be 712,271 lbs/yr (DNR 1996 Table 40). Based on full compliance to the new permits, Total P loading from point source discharges are estimated to be 289,029 lbs/yr (DNR, 1996 Table 41). The majority of facilities are required to reduce Total P discharge to 1.00 mg/l (Table 1). To assist in the determination of the amount of nutrient reduction required to improve the trophic state of the flowage, the Wisconsin DNR recommended that a model of the Petenwell flowage be assembled (WDNR, 1996). This model will be used as a tool to predict changes that may occur in the flowage as a result of changing phosphorus inputs.

OBJECTIVES

The primary objective of this study was to establish a water quality model for the Petenwell Flowage which would serve as a tool for future management decisions by WDNR watershed and fishery managers. Models will summarize morphometric, historic flow and water quality data for the flowage and its tributaries which will assist in the determination of the physical and chemical attributes that are primarily responsible for eutrophication of the impoundment. The use of two models (FLUX and PROFILE) will summarize the data in a format required for the third model (BATHTUB). Development of the Petenwell models will include the following objectives:

1. Identify point and non-point sources of nutrients from the Wisconsin River, tributaries, and surface runoff within the watershed.

2. Quantify inputs and outputs to the Petenwell Flowage by developing a nutrient budget and water budget.

3. Use the model to predict the effects of nutrient reduction in the flowage on ecological variables such as transparency, dissolved oxygen concentrations, and chlorophyll *a*.

4. Use the model to predict amount of reduction of nutrient inputs required to decrease nutrients in the flowage to achieve various reductions in phytoplankton and improvement in water clarity.

Funding for this project is through a cooperative agreement between the Wisconsin Department of Natural Resources, Golden Sands Resource Conservation & Development, and the University of Wisconsin-Stevens Point Environmental Task Force Program.

LITERATURE REVIEW

Models

Modeling of a system is a tool that is popular with many natural resource managers. Frequently the resource manager must make management decisions that may not show measurable results for years, so predictive tools such as numeric models can enhance long term decision making capabilities. This tool can be particularly useful when attempting to predict the response of a water body to hydrological environmental or chemical changes. Generally, changes in the trophic state of large bodies of water occur over long periods of time. Use of a model can assist with land use planning within a basin, estimating permissible effluent discharge levels, or necessary reduction of inputs to an aquatic ecosystem to achieve a targeted water quality condition.

Models designed for use on water bodies can be fairly simple or very complex depending upon the complexity of the system and the required results. Simple models are available to assess one component of a system such as nitrogen or oxygen; more complex models interface many physical and chemical characteristics within the water body and/or the basin. Cost is frequently a consideration as data acquisition is frequently an expensive endeavor and model building can be quite time consuming.

Many lake models are not applicable when modeling a reservoir because of the flow through characteristics and short retention times typical of reservoirs. The Army Corps. of Engineers designed the three component FLUX, PROFILE, and BATHTUB model to be used on these unique bodies of water. This reservoir eutrophication modeling system was applied to the Petenwell flowage. It is an empirical model which utilizes average conditions. The model consists of three components; FLUX, PROFILE, and BATHTUB ultimately resulting in two models, a loading model and an eutrophication response model. The primary purpose of FLUX and PROFILE is for data reduction, which is required for input into BATHTUB.

FLUX is used to estimate nutrient loading from tributaries and the main stem of the

Table 1. Phosphorus loading reductions based on recommendations in Chapter NR 217, Wis. Admin. Code. Table 41 from WDNR, 1996.

Table 41: Phosphorus Loading Reductions as a Result of Chapter NR 217, Wis. Admin. Code

Facility	Effluent mgpd	Total-P mgpl	Total-P kg/day	Total-P lb/yr
Eagle River STP	0.33	1.00	1.25	1,006
Rhinelander STP	1.30	1.00	4.92	3,958
Rhinelander Paper 1	6.12	0.19	4.40	3,540
Rhinelander Paper 2	0.43	1.00	1.62	1,303
Packaging Corp.	4.68	1.00	17.71	14,247
Tomahawk STP	0.44	1.00	1.66	1,335
Ward Paper	0.90	1.00	3.40	2,735
Merrill STP	1.41	1.00	5.34	4,296
Brokaw STP	0.02	2.88	0.23	185
Wausau Paper	8.98	1.00	33.99	27,344
Wausau STP	6.10	1.00	22.75	18,301
Foremost (Rothschild)	0.29	1.00	1.09	877
Weyerhauser	6.41	1.00	24.26	19,516
Rib Mountain STP	1.4	1.00	5.3	4,264
Mosinee STP	0.33	1.00	1.25	1,006
Mosinee Paper	11.51	0.68	29.63	23,836
Stevens Point STP	3.83	1.00	14.5	11,665
Plover STP	.823	1.00	3.11	2,507
Whiting STP	.242	1.00	.91	737
Cons. Paper Water Renewal Center	5.2	.44	8.7	6,968
Neenah Paper	1.98	.47	3.5	2,834
Foremost	.247	1.00	.935	752
ORE-IDA	1.2	1.00	4.5	3,654
Nekoosa Paper	29.43	1.00	111.9	89,635
Wis. Rapids STP	3.23	1.00	12.22	9,747
Port Edwards STP	.456	1.00	1.705	1,372
Vulcan Materials	4.56	--	--	--
Nekoosa STP	.377	1.00	1.43	1,148
Cons. Paper Water Quality Center	27.6	.36	37.6	30,262
TOTAL:	129.7		359.2	289,029

Total Point Source Reduction: ~60% (715,571 lbs/yr -> 289,029 lbs/yr) mgpd = million gallons per day
Point Source Reduction Immediate Source Area: ~41% (393,660 lbs/yr -. 161,280 lbs/yr)

river. Input data required includes sample concentrations of total P, reactive P, total N, inorganic N, and a conservative substance (Cl was used in the Petenwell model). Daily flow is also necessary. Reduced data from FLUX include loading means and coefficients of variation for each input component in each segment. These data are used in the production of a nutrient balance. Based on the loading characteristics of the reservoir and amount of available data, data can be stratified by season, concentration, or flow. This stratification process can provide additional information on relationships that exist within the reservoir and can identify when adjustment of future sampling schemes may be desirable.

PROFILE is used to determine the thermal stratification in a reservoir and for reduction of pool water quality measurements. Reservoir morphometry, location and elevations of monitoring stations, and temperature and oxygen profile data are required to run PROFILE.

The BATHTUB component of the model system is used to produce a nutrient budget and the eutrophication response model. The BATHTUB model assumes steady-state conditions. BATHTUB uses reduced data from the FLUX and PROFILE models. These data include total P, reactive P, total P minus reactive P, total N, organic N, Cl, Secchi disk measurements, chlorophyll *a*, and daily flow values. In the case of the Petenwell flowage, seasonal values (May through October) were used for all components. The nutrient budget and eutrophication model are tools that will be used to predict the levels of reduction of nutrient inputs needed to achieve WDNR target reductions in algal blooms and increased water clarity in the reservoir. Total P, concentration and frequency of chlorophyll *a*, and Secchi disk measurements will be the primary outputs in this study. Preliminary N:P ratios indicated that the system was dominated by P. Therefore, to more accurately model the reservoir nitrogen was held constant.

Eutrophication

Excessive algal growth in reservoirs can be associated with several conditions. Typically the extent of growth is a function of limiting nutrients. In most freshwater systems P rather than N is the limiting nutrient, however, limitation can change during periods of stratification (Schindler, 1971, Welch, 1992). N:P ratios can be used to determine the limiting nutrient, as well as the trophic status of a lake. Eutrophic lakes tend to have lower N:P ratios

(10:1) than oligotrophic lakes (70:1). These rates also influence type of algal species present. Nutrients in reservoirs are introduced from the main channel flow, tributaries, direct runoff, and groundwater inflow. In addition, the water body can retain and release nutrients from the sediment bed (Singh, 1983). This process is referred to as internal loading. In the Wisconsin River sources of P and N include municipal waste and industrial processing including manufacture of paper and cheese. Non-point sources within the watershed include fertilizers, animal waste, and urban runoff.

Several studies have indicated that a combination of sedimentation and phosphorus enrichment result synergistically, increasing periphyton growth in reservoirs (Kimmel et al, 1981; Burkholder and Cuker, 1991). In fact, Burkholder and Cuker found periphyton to be a large proportion of the total productivity in reservoirs under these synergistic conditions. A study by Schindler (1971) focused on the roles of P, nitrogen (N) and carbon (C) in relation to periphyton growth. P was found to be the primary limiting nutrient. Low concentrations of N and C also limited phytoplankton growth, however, excess concentrations of N and C did not increase phytoplankton growth without the addition of dissolved reactive P.

Phosphorus release from sediment is seasonal, corresponding with temperature variation and oxic versus anoxic conditions. Jensen and Anderson observed that a rise in temperature from 7° to 14° to 21°C were more significant in the release of soluble reactive P then were pH and nitrate-N. Stauffer and Armstrong observed greater release of P occurring in anoxic conditions when pH was less than 8. James, et al (1995) conducted a study on internal phosphorus loading in Lake Pepin and determined that increased P release occurred in anoxic conditions, yet a significant amount of P release also occurred in oxic conditions.

Retention time is another factor related to algal growth. As retention time increases nutrients are more likely to change to dissolved forms which are more readily taken up by plants. In addition, the time available for phytoplankton growth increases, allowing algal populations to become established and bloom before being flushed from the impoundment. The combination of these factors can result in increased phytoplankton productivity (Taylor, 1971).

Other factors that control P release include reservoir draw down, carp activity, and wind. These factors can all create sediment resuspension and P release. Draw down in winter months can also cause roots of aquatic macrophytes to become exposed and freeze. This situation may reduce the amount of rooted plant uptake of available P in summer months leaving additional nutrients for algal uptake.

Phosphorus Loading

Traditionally, P has been the primary nutrient related to increased phytoplankton growth and is becoming the primary focus when addressing eutrophication in fresh water lakes and reservoirs. Phosphorus is found in a variety of forms and availability of P to plants varies with form. Dissolved reactive phosphorus (DRP), $H_2PO_4^-$, and HPO_4^{-2} are directly available to plants for use and P adsorbed on Fe, Al and Ca is partially available to plants when DRP is low. Because of this variation, eutrophication is more closely associated with elevated concentrations of DRP than with total phosphorus (TP), (Sonzogni, et al, 1982; Walker, 1996)

A more thorough characterization of the eutrophication of a water body includes concentrations of chlorophyll *a*, phosphorus (dissolved reactive and total), nitrogen (organic, nitrate, and ammonium), temperature and oxygen gradients, and Secchi disk measurements. Lillie and Mason (1983) developed a water quality index for lakes in Wisconsin which considers chlorophyll *a*, total phosphorus, and Secchi disk measurements (Table 2). Based on this index and using water quality data published by the WDNR in 1993, the Petenwell Flowage ranges from poor to very poor based on P, good to very poor based on chlorophyll *a*, and Secchi disk measurements range from poor to very poor. Later we will see how misleading reservoir averages can be when looking at the response of a flowage as large as Petenwell. The variation in the chlorophyll *a* and Secchi disk measurements is due to seasonality, with the poorest water quality in July and August, corresponding with peak algal blooms, low flow, and poor transparency. The Kruskal Wallis test for seasonality was used to test N:P ratios and at the Nekoosa Bridge (Consolidated data) and Petenwell Dam (Storet data) these data were found to be significant at the 95% CI, indicating that seasonal variation is probably occurring in the flowage (Kreitlow, 1993). This seasonality is likely due to changes in the availability of P throughout the year from variations in flow, temperature, transparency, and color.

Historic water quality, morphometric, and flow data from the Petenwell Flowage are available dating back to the early 1960s. These data were primarily collected at the Nekoosa Bridge and the Petenwell Dam. The water quality data were collected from 2 to 12 times per year and could be useful in examining changes in water quality due to changes in point source loading as well as variation in water quality that may occur in low flow or high flow years. These data were graphed in FLUX to determine if relationships exist between concentration, time concentration and flow. Using basic statistics, no strong relationships exist between either. Many factors within the upper Wisconsin River Basin have changed over time, including land use, land practices, discharge concentrations and discharge volume. A sub-study to document these changes over time would be necessary to further statistically evaluate these relationships. There are not enough historic data sites within the flowage to use in BATHTUB for comparisons of loading over time.

Table 2. Water quality index for Wisconsin lakes and ranges of concentrations of total phosphorus, chlorophyll *a*, and Secchi disk measurements for the Petenwell Flowage (WDNR, 1996). Table base on Lilly and Mason, 1983.

Water Quality Index	Secchi Disk Depth (m)	Chlorophyll *a* (μg/L)	T. Phosphorus (μg/L)
Excellent	>6.0	<1	<1
Very Good	3.0-6.0	1-5	1-10
Good	2.0-3.0	5-10	10-30
Fair	1.5-2.0	10-15	30-50
Poor	1.0-1.5	15-30	50-150
Very Poor	<1.0	>30	>150
Petenwell Flowage	**0.6-1.5**	**8.4-66.6**	**85-171**

STUDY SITE DESCRIPTION

The Petenwell Flowage is located on the Wisconsin River in the counties of Adams, Juneau and Wood; the dam creating it is operated by Wisconsin River Power Company (WRPCO). The main body of the flowage is approximately 15 miles long, covering approximately 30,000 acres. Estimated volume for the flowage is 553,200 acre-ft; depth averages 18 ft, as shown in Table 3, with a maximum depth of 45 ft. Between 1980 and 1997 discharge ranged between 604 and 53,019 cfs, with and average of 5,123 cfs. The water's elevation ranged from 912.5 to 924.9 feet above sea level. The flowage's average retention time is estimated at 47 days (WDNR, 1996).

Table 3. Morphometric data describing the Petenwell Flowage. Information was derived from WRPCO's bathymetric flowage map and planimeter.

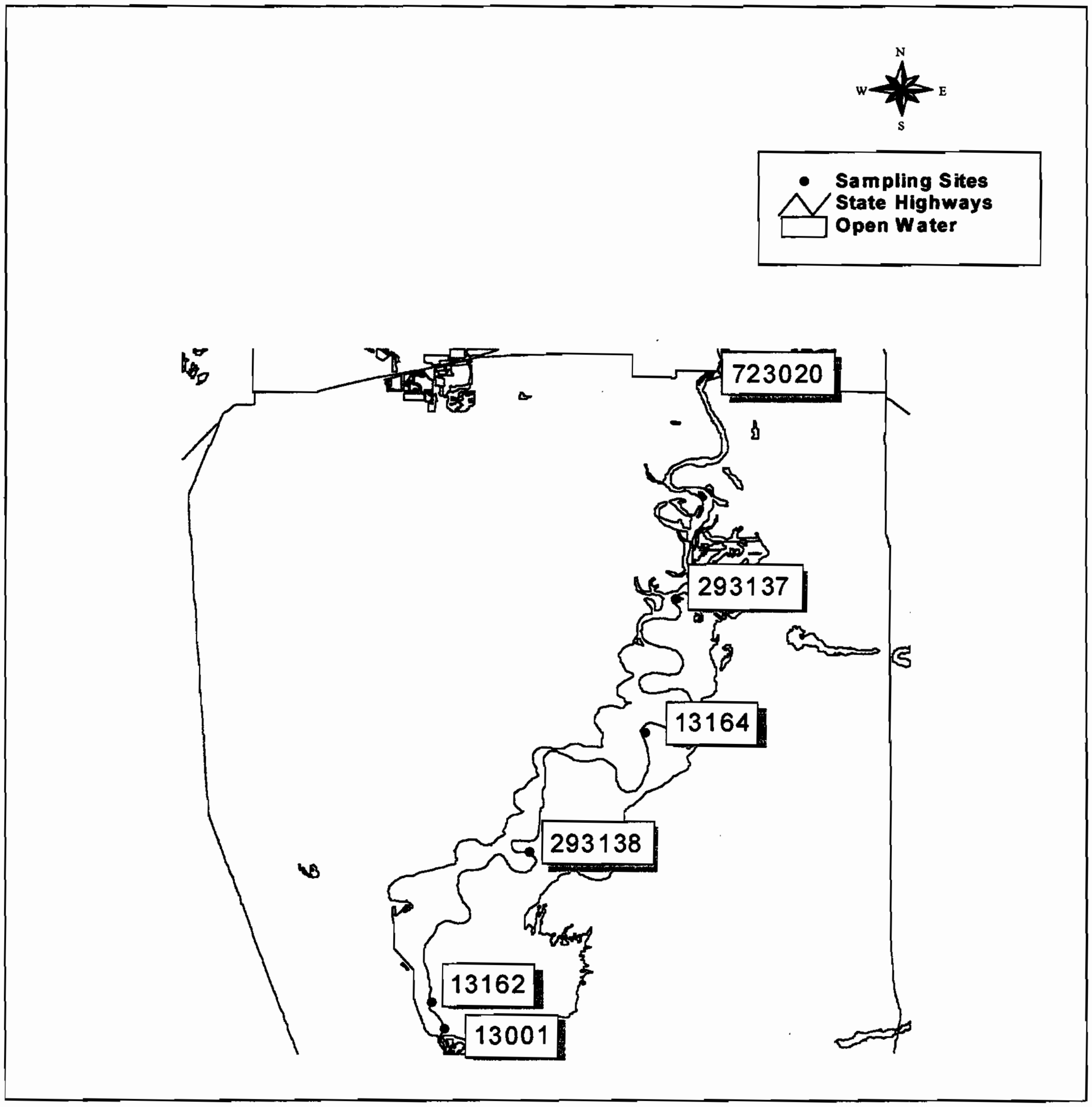
	BATHTUB Model Segment Number						
	1	**2**	**3**	**4**	**5**	**6**	**Total Reservoir**
Area (acre)	2,383	4,408	8,062	5,906	5,682	4,448	30,888
Volume (acre-ft)	17,124	49,296	119,092	132,788	113,180	124,708	553,188
Mean Depth (ft)	7.2	10.5	14.8	22.5	19.9	28	17.9
DNR Sample ID	293137		13164	293138		13162	

METHODS

Experimental Design

Historic surface water quality data for the Petenwell flowage is available, however, the calibration of the model system requires more annual water quality data than was available. Therefore, additional water quality samples/year were collected for this project over two years (1996/7). The PROFILE model is capable of segmenting a reservoir and determining the optimum number and location of water quality samples sites, however, historic data from five sample sites within the Petenwell flowage could be used with the BATHTUB model after calibration, so the same sample sites were used in this study for surface water collection. Walker (1996) reviewed data available for the Petenwell flowage and confirmed that it would be desirable to continue using these same historic sample sites. Surface water sampling by the W DNR occurred at five sites in the Petenwell Flowage; at the headwaters near Nekoosa, in the primary river channel near New Rome, below Barnum Bay, near the Bighorn Access, and just above the Petenwell Dam (Figure 3). In addition to these sites, samples were taken

Figure 3. Map of the Petenwell Flowage showing historic water sampling sites.



from Fourteen Mile, Ten Mile, and Chester Creeks. For this study, six water quality samples sets were collected from the five flowage and three tributary sites between May and October in both 1996 and 1997.

Preliminary calibration of the model indicated seasonal internal loading in the upper part of the reservoir. To determine the boundaries of the internal loading, samples were taken every mile along the river channel (Figure 4). This sampling occurred twice in June and once in September 1998. For uniformity, surface water samples for all sample sites were collected on the same day. These data indicated the need for re-segmentation of the reservoir. To insure the appropriate representation of sample data six segments were used instead of the four originally proposed (Walker, 1999).

Field Sampling Methods

The five surface water sample sites on the flowage were located using a global positioning system (GPS) mobile receiver. At each location pH and Secchi disc readings were obtained and temperature/oxygen profiles were measured with an YSI electrode and digital meter. Following these measurements, surface water samples were acquired for water chemistry analysis. Water samples were retrieved using a VanDorn bottle at the surface, mid-depth and 3 ft. above the sediment; then combined to create a composite sample. Composite samples were transferred into three bottles, two un-acidified and one acidified with H_2SO_4 for nutrient analysis; all were kept on ice until delivery to the labs. The State Lab of Hygiene, located in Madison, Wisconsin analyzed the samples from within the flowage for Chlorophyll *a*, reactive phosphorus, total phosphorus (TP), NO_2+NO_3-N, NH_4, TKN, chloride, and turbidity. UWSP's Environmental Task Force (ETF) Lab analyzed the tributary samples for reactive phosphorus, TP, NO_2+NO_3-N, NH_4, TKN, and Cl. Both labs used state certified techniques and methods of analysis.

Data Acquisition/Manipulation

Water quality and flow data were obtained from a variety of sources. Daily flow and water elevation data for the Wisconsin Rapids and Petenwell dams were supplied by the Wisconsin Valley Improvement Company (WVIC, 1996). Flow information was needed for the Nekoosa Dam (located just above the Petenwell Flowage) however flow measurements were not available, therefore, Wisconsin Rapids flow data were adjusted to reflect the 3.5%

increase in flow at Nekoosa (Morgan, S., 1997).

Gaging stations currently do not exist on Fourteen or Ten Mile Creeks, so estimates were used. Flow data were available for both the Wisconsin River (at Petenwell) and Ten Mile Creek. These data were regressed, with an $R^2 = 0.92$. The ratio of flow in the Wisconsin River versus the Ten Mile Creek was 1:0.0132. Missing flow data for the years used in this model (1996/7) were estimated using this relationship. Flow was also needed for Fourteen Mile Creek. By use of regressions, Oberhoffer (1993) developed equations to predict the flows at Fourteen Mile Creek from the Ten Miles flows. These equations were used to predict missing flows for this model (Eq.1 and 2). Flows and inputs from the Chester Creek are insignificant when compared with the Wisconsin River, so the Chester Creek watershed was combined with Fourteen Mile's watershed in the BATHTUB model.

Equation 1. Ten Mile Creek Flow > 50 cfs:

14 Mile Creek (cfs) = (x-4.41)/ 0.10

Equation 2. Ten Mile Creek Flow < 50 cfs:

Fourteen Mile Creek (cfs) = (x-29.1)/0.83

Water quality data for five sites within the Petenwell Flowage were obtained from the STORET system at the Wisconsin DNR. Tributary sample chemistries were obtained from the ETF lab. Chemistry and physical measurements required for the models included: temperature, dissolved oxygen, Secchi disk, Chlorophyll *a*, reactive P, TP, NO_2+NO_3-N, NH_4, TKN, chloride, color. Units of measure are shown in Table 4. The BATHTUB model requires other data which were derived from the measured chemistry/physical data; total N, organic N, TP minus reactive P, and non-algal turbidity.

The reservoir was segmented based on the location of sample sites taking into account the distribution of chemical concentration within the reservoir which resulted in the use of six segments (Figure 5). Morphometry for each segment was determined using WRPCO's bathymetric flowage map (1993) and a planimeter; area, volume, and mean depth were then calculated for each segment (Table 3).

Table 4. Reported units and levels of detection for laboratory water quality analyses.

Analysis	Reported Units	Level of Detection
Reactive P	mg/l	0.002
Total P	mg/l	0.002
NO_2+NO_3-N	mg/l	0.2
NH_4-N	mg/l	0.006
TKN	mg/l	0.08
Chlorophyll *a*	µg/l	1
Chloride	mg/l	1
Turbidity	NTU	1

Figure 4. Map of Petenwell Flowage showing approximate location of 16 water sampling sites. Samples were obtained in June and September 1998.

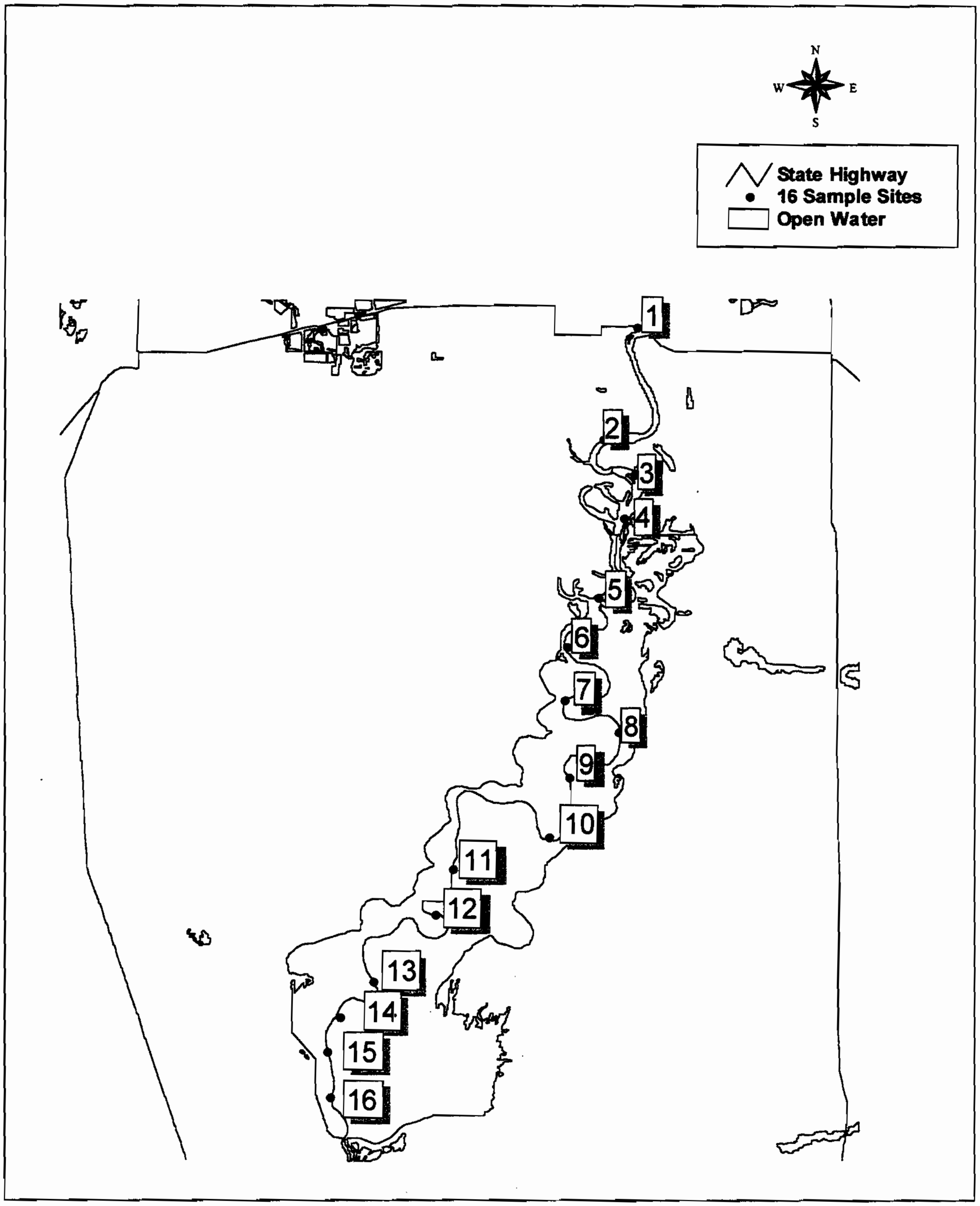
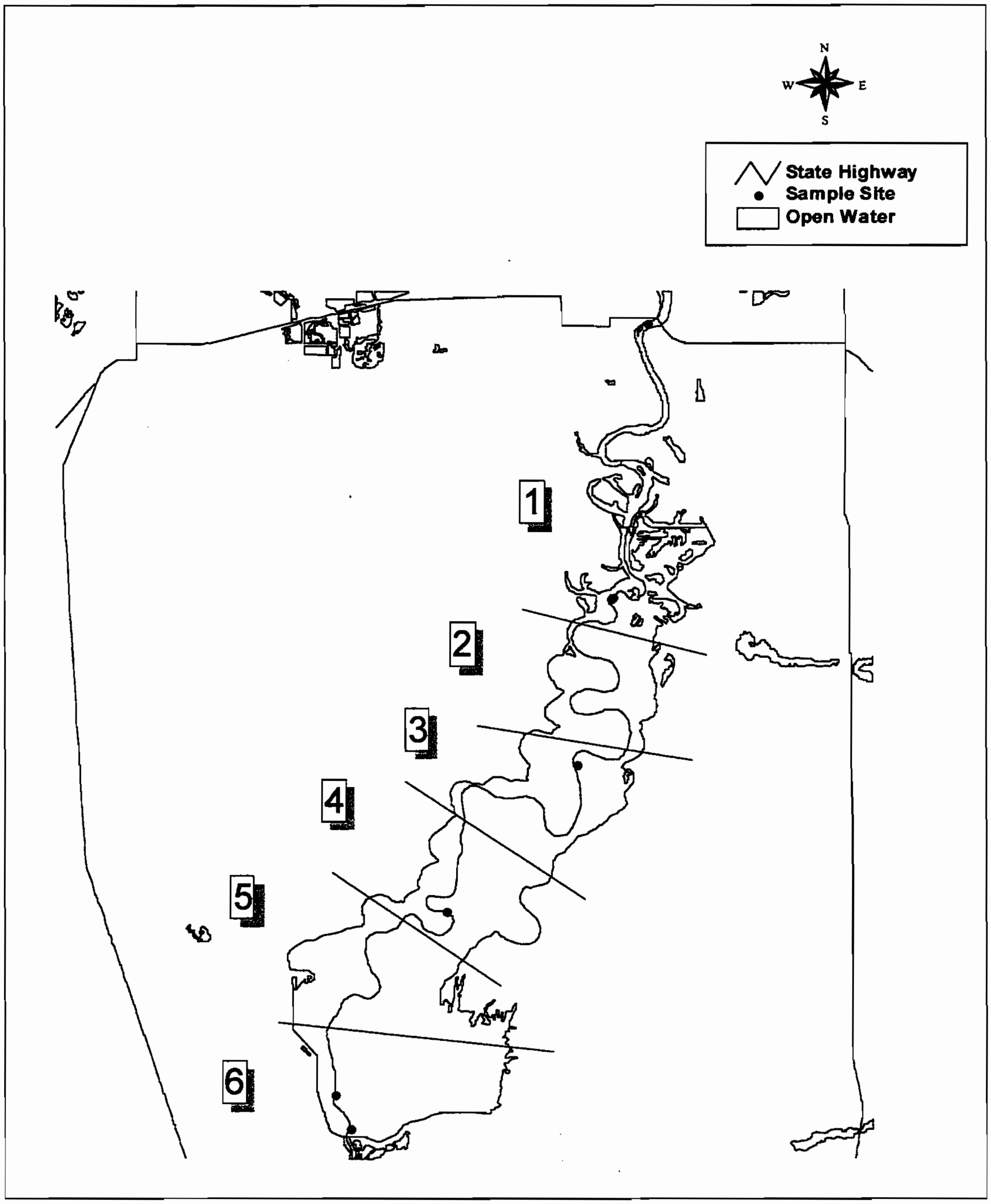


Figure 5. Map of the Petenwell Flowage showing sample sites and approximate location of segment boundaries used in the BATHTUB model.



BATHUB Model

A non-networked model was used for the Petenwell Flowage. The flowage was divided into six segments (Figure 3). Four of these segments correspond with sample sites that have historically been used by the Wisconsin DNR or Consolidated Papers. The other two segments were added to provide a more representative model of the flowage. This model was designed to predict changes in Chlorophyl *a* with the reduction of input P loading. In addition, three tributaries were sampled, including Fourteen Mile, Ten Mile, and Chester Creeks. Increase in P occurred consistently in flowage within Segment 2, presumably due to internal loading. This was accounted for using an internal loading term of 20 mg $TP/m^2/day$. The Wisconsin River and Ten Mile Creek were input tributaries to Segment 1. Fourteen Mile Creek and internal loading were an added inputs to Segment 2.

The reservoir eutrophication modeling system which was applied to the Petenwell flowage is an empirical model which utilizes average conditions. The model contains three components; FLUX, PROFILE, and BATHTUB which ultimately result in two models, a nutrient balance and a eutrophication response model.

FLUX is used to estimate nutrient loading in tributaries and the main stem of the river. Input data required includes, total P, reactive P, total N, inorganic N, and a conservative substance (Cl was used in the Petenwell model). Daily flow is also necessary. Loading means and coefficients of variation result for each input component in each segment and are used in the production of a nutrient balance. Based on the loading characteristics of the reservoir and amount of available data, data can be stratified by season, concentration, or flow. This stratification process can provide additional information on relationships that exist within the reservoir and can identify when adjustment of future sampling schemes may be desirable.

PROFILE is used to determine the thermal stratification in a reservoir. Morphometric, temperature and oxygen profile data are required to run PROFILE. Temperature and oxygen data obtained from the Petenwell indicate the flowage is mixed year round.

The BATHTUB component of the model system is used to produce a nutrient budget and the eutrophication response model. The BATHTUB model assumes steady-state conditions. BATHTUB uses reduced data from the FLUX and PROFILE models. These data include Total P, reactive P, Total P - reactive P, Total N, organic N, Cl, Secchi disk measurements, chlorophyll *a*, and daily flow values. In the case of the Petenwell flowage seasonal values (May through October) were used for all components except flow. The eutrophication model was calibrated for each year of thorough flow and water quality data and validated by Dr. William Walker. Once calibrated, scenarios with varying nutrient inputs were run to predict the level of water quality and algal reduction that is feasible for this reservoir. The nutrient budget and eutrophication model are tools that will be used to predict the levels of reduction of nutrient inputs needed to achieve WDNR target reduction in algal blooms in the reservoir.

Model Calibration/Validation

The BATHTUB model was assembled using 1996 water quality and flow data and was sent to Dr. Walker for review and calibration. The water chemistry in the upper end of the reservoir indicated the presence of internal loading, but with the minimal number of sample sites it was difficult to discern the spatial extent of the internal loading. Dr. Walker recommended supplemental sampling with additional sample sites within the reservoir. Water quality samples were collected every mile within the reservoir three times in 1998 . These data helped to indicate the boundaries of the internal loading in the upper end of the reservoir. The reservoir was re-segmented and morphometry of each segment was re-calculated. The model was sent to Dr. Walker who tested it with 1997 data and re-calibrated the model based on these results.

Default model calibration factors were used with the exception of the following changes that were made by Dr. Walker to the BATHTUB model during the calibration process to better fit the model to the observed Petenwell flowage data and response.

- The P calibration factor for TP includes a decay rate calibration factor which was changed from 1 to 0.4. Also, an internal loading rate of 20 mg/m^2-day was specified in the second segment. These adjustments were necessary to simulate the increase in P concentration between the inflow and upper pool to represent the observed P gradient in the middle and lower portion of the reservoir. Internal loading with these factors/rates is about 17% of the total loading.
- Dispersion rate is the estimation of exchange flows between adjacent segments. This was changed to Model 0 (not calculated). This was necessary to match the

longitudinal gradients in P and other water quality parameters. With the default value (1), longitudinal mixing rates were substantially over-predicted, resulting in a flat profile. This adjustment most likely reflects the tortuous shape of the main river channel within the reservoir (thalweg). Sharp curves in the flow path impede longitudinal mixing. If spatial resolution within the reservoir is important for evaluating management scenarios, a more detailed segmentation scheme (using shorter segment lengths) should be developed.

- Chlorophyll a/Secchi Slope = 0.015 m^2/mg (default = 0.025 m^2/mg). This adjustment is typically required in reservoirs with blue-green algae having relatively large cell sizes and low light attenuation per unit of chlorophyll a. This adjustment influences predicted chlorophyll a and transparency values.
- Chlorophyll a Coefficient = 1.3 (default = 1.0). This was necessary to match the area-weighted mean chlorophyll a concentration in the reservoir. It reflects differences in algal types, analytical procedures, and growth response relative to the population of reservoirs used in developing the model.

RESULTS

Scenarios

The DNR proposed several scenarios to be run with the BATHTUB model. The model should provide reservoir response to changes of TP inputs and/or flow regimes. The scenarios included the mean TP for 109 impoundments in Wisconsin, average TP at Lake DuBay, average TP at Nekoosa (prior to NR 217), loading if point source facilities between Lake DuBay and Petenwell were all in compliance with NR 217, loading if discharge from point source facilities between Lake DuBay and Petenwell was 0.5 mg/l, 1.5 times the model concentration of 128 ug/l, and 2 times the model concentration (Table 5). Inflow concentrations were calculated for each scenario. The percent difference between these inflow values and the calibrated model inflow concentration (128 ug/l) were calculated; this same percent difference was used to calculate internal loading concentrations from the model's 20 mg/m^2/day internal TP loading rate. This determination was based on the assumption that inflow TP and internal loading TP are directly correlated to one another and therefore, if inflow TP is reduced, internal loading should also be reduced. Without a more thorough understanding of the internal loading processes these types of assumptions must be applied.

The above scenarios and calculated TP inflow concentrations are shown in Table 5. The resulting inflow concentrations ranged between 64 and 256 ug/l (50 to 200% of the model standard of 128 ug/l). Because many of the scenarios resulted in similar concentrations, only five of them were selected for discussion in this paper. These concentrations included 64, 112, 128, 155, and 256 ug/l (shown shaded on Table 4). Internal loading TP concentrations were 10, 17, 20, 24, and 40 ug/l, respectively. The percent of the model standard were 50, 87, 100, 120 and 200, respectively.

Results indicate that TP concentrations are not static as they move from the upper to the lower end of the flowage. Regardless of input TP concentrations, concentrations in Segment 2 always increase due to internal loading, then TP decreases in each subsequent segment. As the TP moves through the system, reduced concentrations in the water results from plant (predominantly algae) uptake and/or adsorption/settling out.

Table 5. Inflow/internal loading TP concentrations calculated for various scenarios used to determine loading with the BATHTUB model of the Petenwell Flowage. Shaded loading pairs are discussed in this paper.

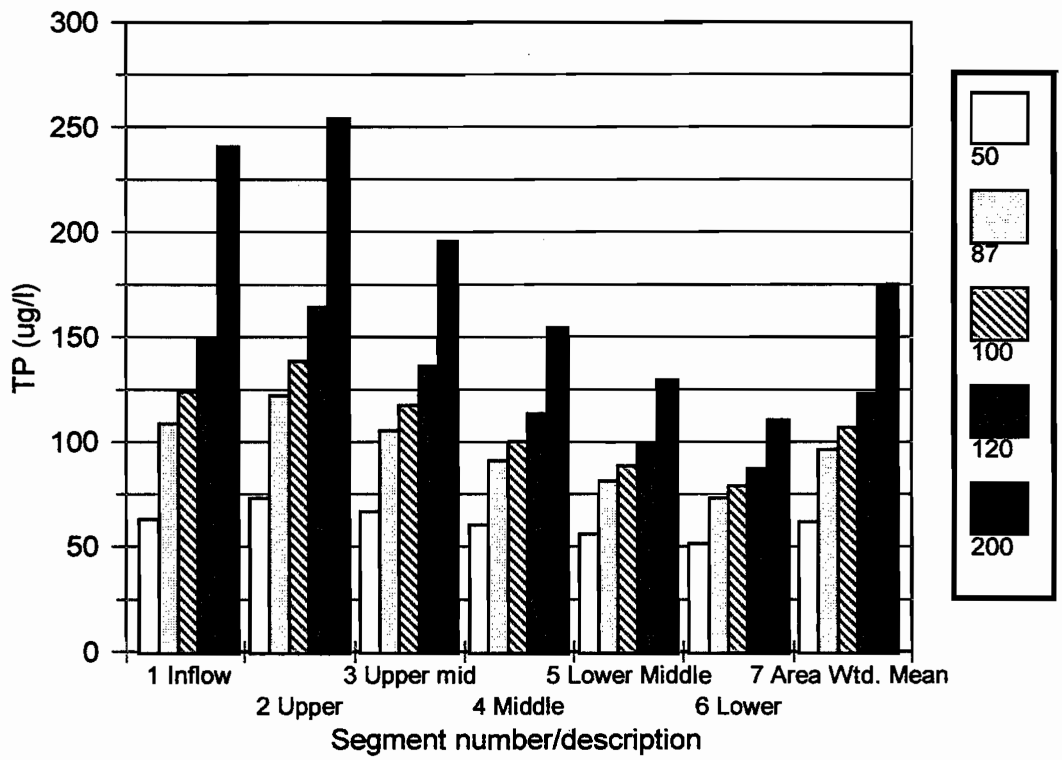
TP Inflow/TP Internal Load (mg/l)	Percent of 128 mg/l	SCENARIO
64 /10	50	Mean TP for 109 Wisc. Impoundments
85/13	66	Upper range of Wisc. 109 Impoundments
100/16	78	Background Ave. TP at DuBay
101/15	79	Loading based on 0.5 ug/l TP [1]
112/17	87	Loading based on 1 ug/l TP [1]
128/20	100	1996 Average loading at Nekoosa (Model Loading)
155/24	120	Average TP LOADING at Nekoosa
192/30	150	1.5 x Model Loading
256/40	200	2 x Model Loading

[1] Calculated for point sources between Lake DuBay and Nekoosa.

DNR's target for TP in the reservoir was 64 ug/l (the mean TP for 109 impoundments in Wisconsin). The 50% reduction (64/10 scenario) was the lowest concentration run through the model and therefore, was the only scenario able to achieve this goal throughout the reservoir. If discharge from up-river point sources between Lake DuBay and Nekoosa were in compliance with recommendations in NR217 (Scenario 112/17), 87% of the model's standard inputs would exist and input in the lower half of the reservoir would come close to meeting the DNR goal with the area weighted mean TP for the total reservoir at 96 ug/l, however, the upper reservoir (Segments 1 to 4) would all exceed the target. Responses for all other scenarios would exceed the TP target value (Figure 6).

TP concentrations were plotted against chlorophyll *a* data for the four primary sample sites. Sample site 293137 (located on the upper end of the flowage) showed a poor relationship between TP and chlorophyll *a* concentrations (R^2 -0.27). This site is located above the flowage and is predominantly riverine with some adjacent wetlands. All other sample sites indicated a reduction in TP which correlated with a reduction in chlorophyll *a*.

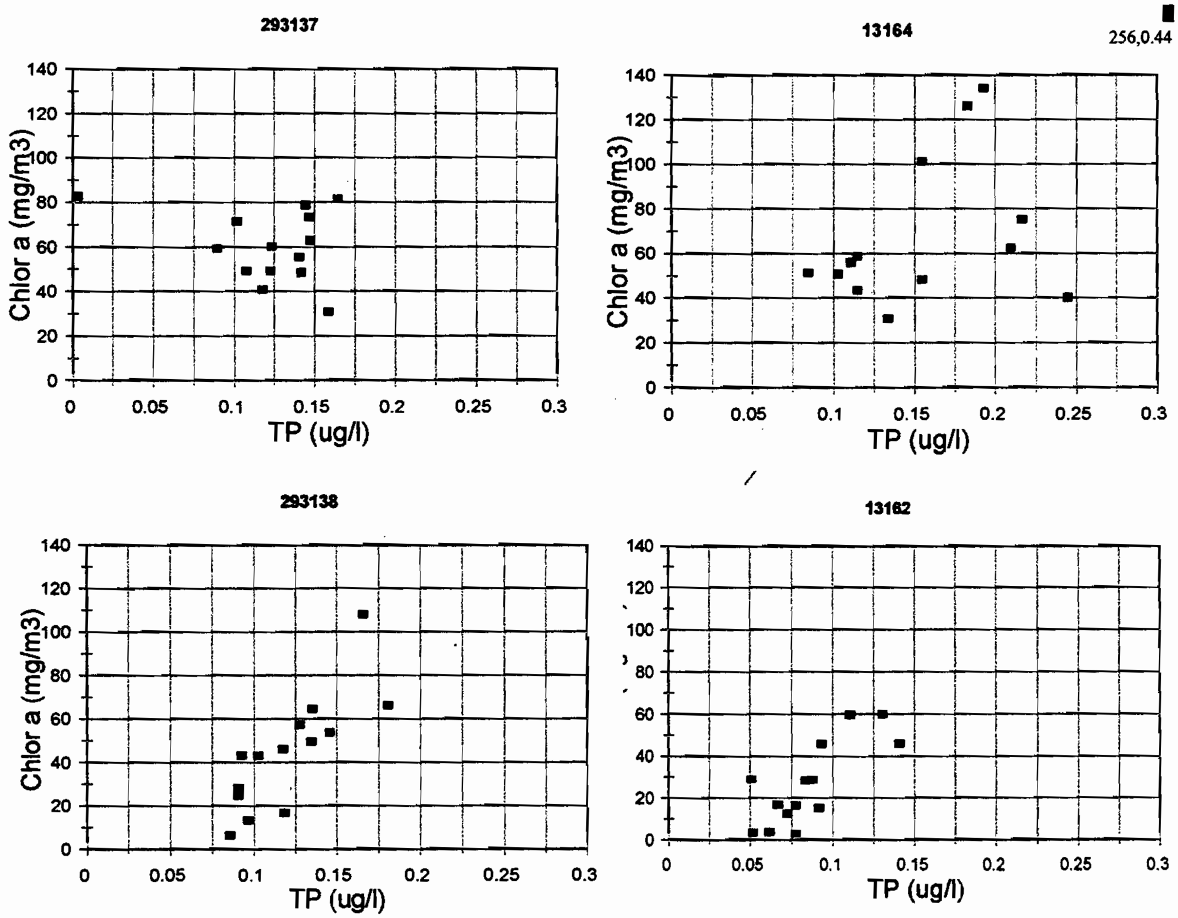
Figure 6. BATHTUB estimates of TP concentrations for each segment using five scenarios of inflow concentrations and internal TP loading. Values shown in the legend are the percent of inflow and internal TP loading based on the model standard of 128ug/l and 20 mg/m²/day, respectively.



The R^2 values for this relationship at sites 13164, 293138, and 13162 ranged from 0.77 to 0.81 (Figure 7).

Target concentrations for chlorophyll *a* was 22 mg/m³ (average for 109 impoundments in Wisconsin). BATHTUB estimates this can only be achieved in Segment 6 of the reservoir, however, within this segment targets could be achieved with most of the TP input concentrations (Figure 8). Since it appears to be difficult to universally achieve this goal within the reservoir, we chose to look at reduction in frequency of chlorophyll *a* blooms >30 mg/m³ between May and October. The upper end of the reservoir clearly has the most significant problem with blooms. The model standard scenario (128/20) showed algae blooms

Figure 7. TP concentrations (ug/l) versus Chlorophyll *a* (mg/m^3) at four sample sites within the Petenwell flowage. May through October 1996 and 1997.

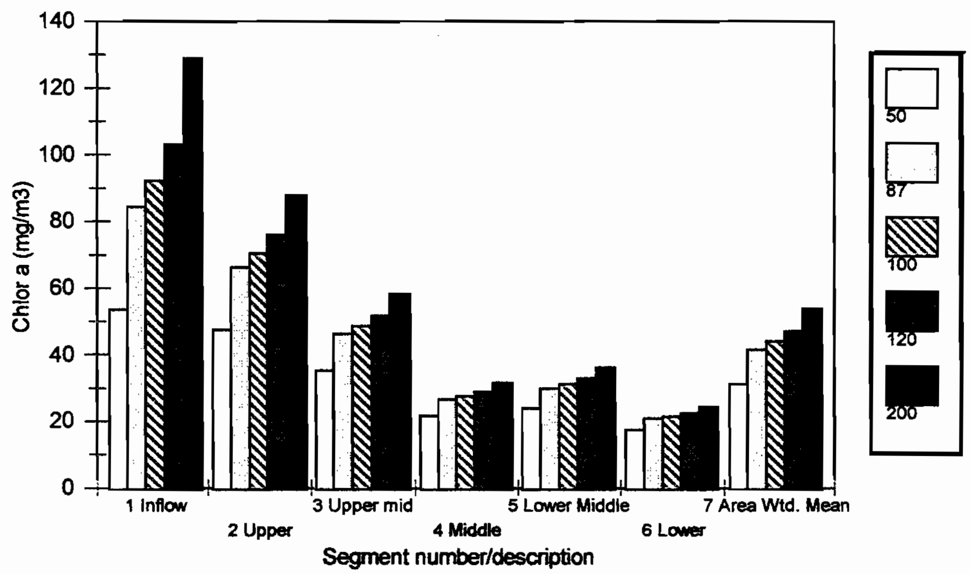


occurring in the first segment 93% of the May to October time period. The frequency can be reduced to 73% of this period if inflow and internal loading TP is reduced by 50% to 64/10, or increased to 97% if TP is increased to 200% of the model standard (256/40). In contrast, Segment 6 had algae blooms 20% of the same time period in 1996. This segment's bloom frequency can be reduced to 12% with the 50% scenario, or increased to 26% with the 200% scenario. Overall, the reservoir experienced blooms 62% of the May through October time period in 1996. Overall reservoir bloom frequency could be reduced to 40% in the same time period with the 50% scenario, or increased to 74% with the 200% scenario. These data are shown in Table 6.

Table 6. Frequency (days) of algae blooms > 30 ug/l occurring between May and October for five TP inflow/internal loading scenarios. Response of the Petenwell Flowage is shown by segment and reservoir mean.

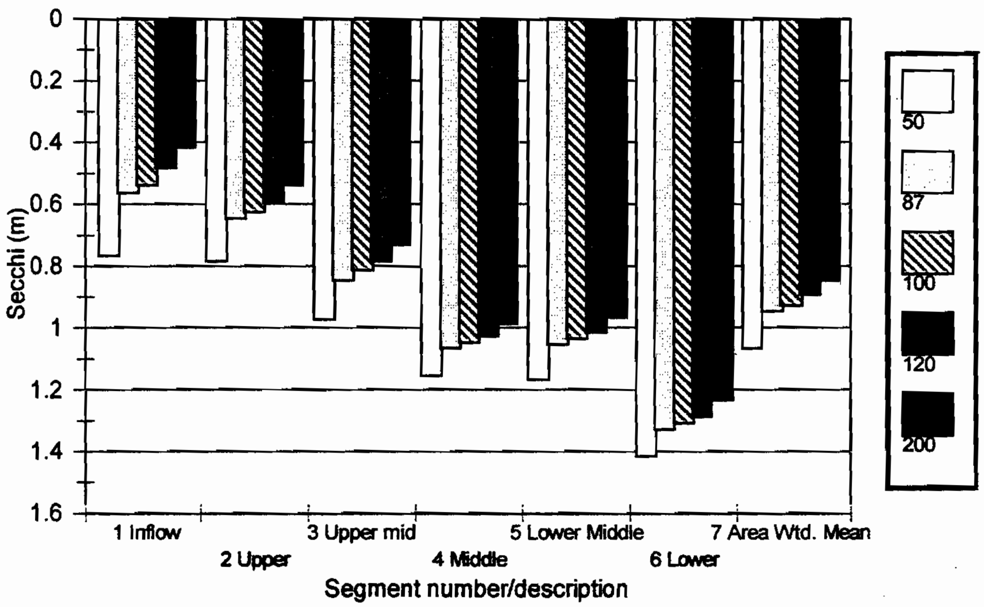
Percent Change	Segment Number						Area wtd. mean
	1	2	3	4	5	6	
50	134	122	88	37	46	22	74
87	167	152	119	57	69	34	107
100	171	157	122	61	74	37	114
120	175	162	131	66	80	40	121
200	179	169	142	76	91	48	135

Figure 8. BATHTUB estimates of chlorophyll *a* values for each segment using five scenarios of inflow concentrations and internal TP loading. Values shown in the legend are the percent of inflow and internal TP loading TP based on the model standard of 128 ug/l and 20 mg/m²/day, respectively.



Secchi disk measurements were estimated by the model in an attempt to determine the potential clarity of the reservoir. The goal for this measurement was 1.3 m which is the average for 109 impoundments in Wisconsin. Regardless of scenario, the upper five segments will not be able to achieve this goal (Figure 7). It is possible to achieve this goal in Segment 6 if all up-river point sources between Lake DuBay and Nekoosa were compliant with NR217.

Figure 9. BATHTUB estimates of Secchi disk measurements for each segment using five scenarios of inflow concentrations and internal TP loading. Values shown in the legend are the percent of inflow and internal TP loading based on the model standard of 128 ug/l and 20 mg/m²/day, respectively.

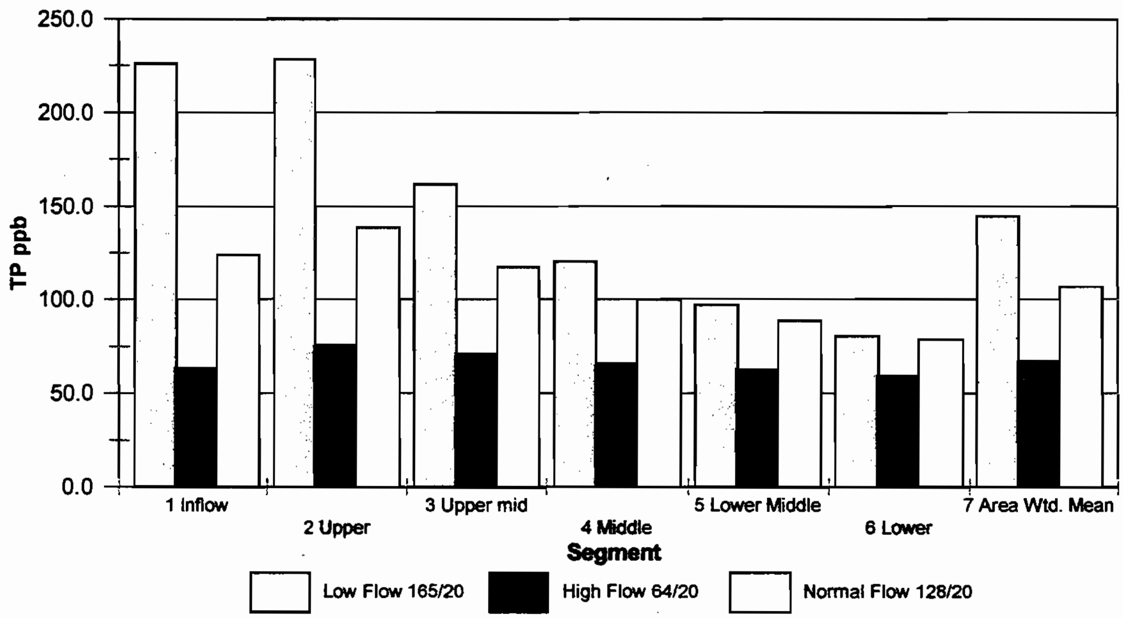


High and Low Flow Scenarios

High and low flow scenarios were also run in BATHTUB. It should be noted that few high flow events were captured with the data collected to calibrate the model, so the accuracy of the output for high flow events is questionable. Historic USGS flow data were used to determine annual mean high and low flow rates. These data were for the Wisconsin River at Wisconsin Rapids, so flows were adjusted by multiplying the flow rate times 1.035 to estimate flows at Nekoosa. This resulted in an annual mean high flow of 8,754 cfs and an annual mean low flow of 2,107 cfs. To adjust TP concentrations relative to flow conditions, we assumed that point sources are responsible for the majority of TP inputs. This assumption was based on historic data for a variety of flow conditions which generally shows an inverse relationship between TP concentrations and flow. Therefore, in the low flow scenario TP was increased to 165 ug/l and in the high flow scenario TP was decreased to 64 ug/l. Internal loading was held at 20 mg/m3/day for both scenarios. Results of these scenarios are shown in Figures 10, 11, and 12.

During the low flow scenario TP and chlorophyll *a* in the upper half of the flowage remained high and Secchi disk measurements low. In the middle Segment (4) TP, chlorophyll *a*, and Secchi disk measurements all drop nearer to the 1996 scenario results and in the lower part of the reservoir (Segments 5 and 6) TP, chlorophyll *a,* and Secchi measurements are very similar to the 1996 scenario results. This suggests that increased concentrations have the greatest effect in the upper part of the reservoir.

Figure 10. Estimates of TP concentrations from BATHTUB using annual mean high flow rates, annual mean low flow rates, and 1996 model estimates. Values are shown in legend are Inflow TP (ug/l)/Internal Loading TP (mg/m^2/day).



The high flow scenario results in little variation in TP concentrations as it moves through the reservoir with ranges from 59 to 76 ug/l, however, chlorophyll *a* changes from 51 to 17 mg/m^3 and Secchi disk measurements increase from 0.8 to 1.4 m moving from the upper to the lower reservoir, respectively. The faster flow through and dilution of point sources suggests more improvement will occur during wet years compared to dry years.

In the high flow scenario the TP loading entering the reservoir was estimated to be 1,245,890 lbs/year compared to the 1996 loading of 1,352,275 lbs/year (Table 7). Ouflow from the reservoir was estimated at 1,164,992 lbs/year during high flow and 848,357 lbs/year for the 1996 model. In an average year (1996) 423,020 more lbs/year TP were retained in the flowage than in a high flow year. In the low flow scenario inflow TP loading was estimated at 622,464 lbs/year and outflow estimates were 338,155 lbs/year. This low flow scenario resulted in retention of 219,608 fewer pounds of TP per year than in the average year.

Figure 11. Estimates of chlorophyll *a* concentrations from BATHTUB using annual mean high flow rates, annual mean low flow rates, and 1996 model estimates. Values are shown in legend are Inflow TP (ug/l)/Internal Loading TP (mg/m^2/day).

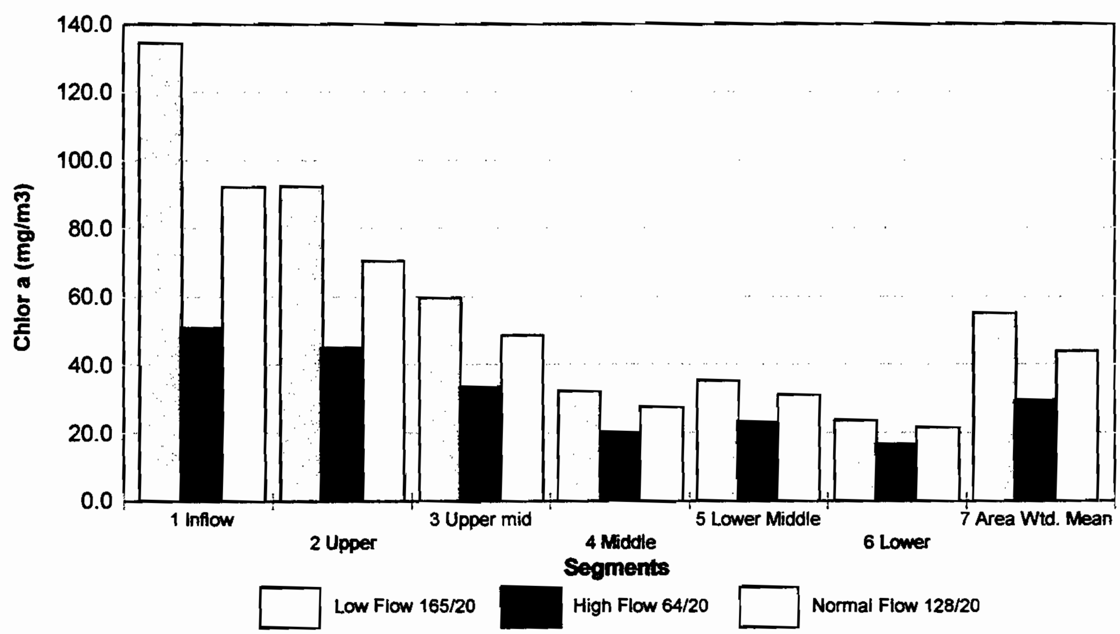


Figure 12. Estimates of Secchi disk measurements from BATHTUB using annual mean high flow rates, annual mean low flow rates, and 1996 model estimates. Values are shown in legend are Inflow TP (ug/l)/Internal Loading TP (mg/m²/day).

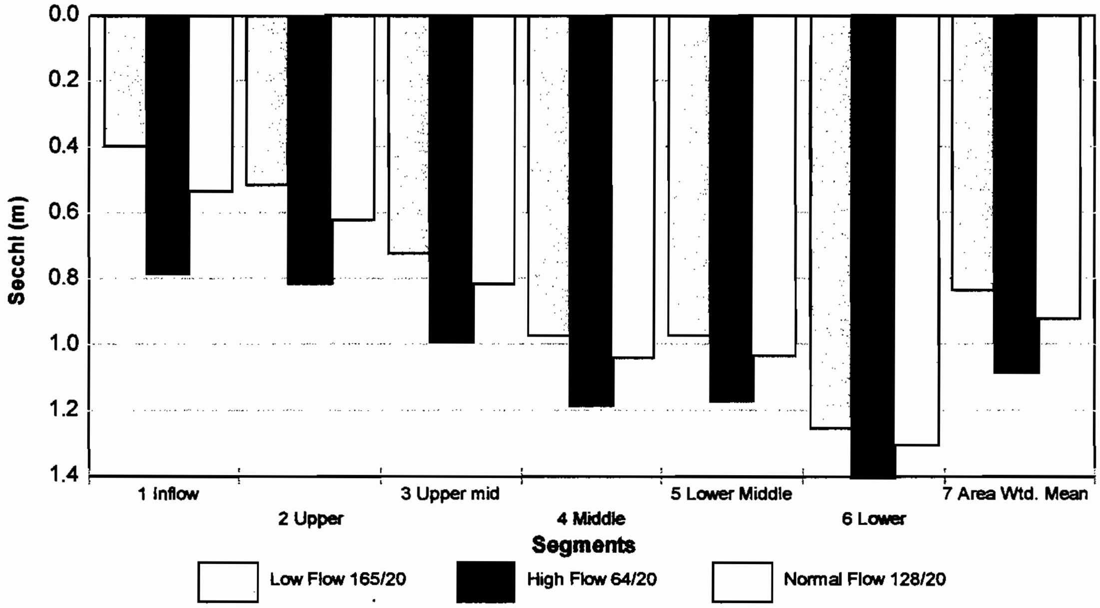


Table 7. Total phosphorus loading and retention and water budget estimates for seven scenarios using the BATHTUB model of the Petenwell Flowage.

Scenario	TP Load (lbs/year)		Retention (lbs/year)	Water Budget (mgd)	
Inflow/Internal Loading	Inflow	Outflow		Inflow	Outflow
64/10	681,512	556,100	125,412	3,490	3,529
112/17	1,184,584	784,900	399,685	3,490	3,529
128/20	1,352,275	848,357	503,918	3,490	3,529
155/24	1,635,253	938,155	697,098	3,490	3,529
256/40	2,693,801	1,187,609	1,506,192	3,490	3,529
High Flow 64/20	1,245,890	1,164,992	80,898	6,385	6,423
Low Flow 165/20	622,464	338,155	284,310	1,621	1,659

Water Budget

The reservoir water budget is shown in Table 7. In 1996 the Wisconsin River was the primary source of water to the Petenwell Flowage at 97.3% of the water budget. The 10 Mile Creek supplied approximately 1.3%, and the Chester Creek and 14 Mile Creek supplied approximately 0.7% of the inflow. Precipitation in the reservoir accounted for the remaining inflow. Approximately 99.8% of the water left the flowage as outflow. The remaining 2% was lost though evaporation.

CONCLUSIONS

1. The Wisconsin River is the major contributor of phosphorus inputs to the Petenwell Flowage. Loading from this source is significantly greater than from tributaries.

2. Phosphorus inputs to the Petenwell Flowage would need to be reduced by 50% to achieve DNR chlorophyll *a* goals of 22 mg/m^3 in at least half of the flowage.

3. Based on BATHTUB model estimates, if all point source discharge between Lake Dubay and Nekoosa were in compliance with NR217 algae bloom frequency (chlorophyll *a* >30 mg/m^3) in the flowage would be reduced from 114 to 107 days in the flowage between May and October. This frequency could be further reduced with greater reduction in phosphorus inputs to the flowage.

4. Internal loading in the Petenwell flowage should be studied to determine a realistic time period for reduction or exhaustion of loading from this source if inflow phosphorus concentrations were decreased.

5. Current inflow data is not adequate to accurately separate impacts from point and non-point sources. More frequent sampling of more sites from Wausau to Nekoosa are needed to better define the movement of phosphorus through this stretch of river.

6. The Petenwell flowage should be sampled throughout the year (including high flows) and with greater frequency to better define flowage characteristics and responses and to further refine the BATHTUB model.

7. A more dynamic model using data obtained above (#5) may be desirable before requiring more restrictive point source discharge levels.

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