Wisconsin Lake Planning Grant Final Report

Lake Wapogasset and Bear Trap Lake
Polk County, Wisconsin

Prepared for the
Lake Wapogasset and
Bear Trap Improvement Association

June 1996

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Executive Summary

Introduction

During 1995, the Lake Wapogasset and Bear Trap Improvement Association and their consultant, Barr Engineering Company of Minneapolis, Minnesota conducted a lake water quality study. The study was funded through the Wisconsin Lake Planning Grant Program.

This summary briefly describes the results of the study. A comprehensive final report was prepared for use by WDNR staff and other technical specialists; while a non-technical summary was prepared for distribution to members of the Lake Improvement Association.

Project Goals and Costs

The goals for the Lake Wapogasset and Bear Trap Lake Wisconsin Planning Grant Project were as follow:

- Estimate the amount of phosphorus released from the sediments of both lakes during 1995
- Examine other water quality parameters, such as oxygen concentrations and temperature, to estimate the effects of sediment phosphorus on the water quality of the lake
- Determine the cost and effectiveness of an alum treatment to control sediment phosphorus release

The budget for this study was $13,333. $10,000 was supplied by the State of Wisconsin through the Lake Planning Grant Program; the remainder was supplied by the Lake Improvement Association.

Project Participants and Activities

Lake samples and water quality data were collected each week during May through September, 1995 by volunteers from the Lake Wapogasset and Bear Trap Improvement Association. These volunteers were organized by Don Wegner, and included Frank Fitzpatrick, Jim Uhrhammer, and

Tom Deuring. The samples were immediately packed with ice and shipped to the Wisconsin State Lab of Hygiene in Madison for phosphorus analysis. The volunteers were aided by Craig Roessler from the Wisconsin DNR Northwest District Office in Spooner. Once the lake data were collected, Barr Engineering Company staff performed the necessary analyses and calculations, and wrote the final report.

Project Results

The following briefly summarize the results of the study:

- The trophic state classification of the water quality in both Lake Wapogasset and Bear Trap Lake was mesotrophic immediately after spring overturn; by September, however, the water quality had degraded to a trophic classification of hypereutrophic.

- The hypolimnion in both lakes became anoxic coincident with thermal stratification.

- Thermal stratification commenced in June and continued until September. The stratification in both lakes was weak throughout the summer, and periods of cool and windy weather resulted in intermittent disruption of the stratification, resulting in erosion of the thermocline, and pulse phosphorus loads to the epilimnion.

- The phosphorus concentration in the hypolimnion of both lakes increased dramatically throughout the summer until fall overturn. The epilimnetic phosphorus concentration in both lakes also increased steadily during late-summer due to entrainment of high-phosphorus hypolimnetic water. Periodic increases in the epilimnetic phosphorus concentration during mid-summer were apparently due to intermittent entrainment of hypolimnetic water during cool, windy weather.

- The mass of phosphorus released from the sediments to the hypolimnion of Lake Wapogasset and Bear Trap lake were 2,083 Kg and 505 Kg, respectively.

- The mass of hypolimnetic phosphorus which was entrained to the epilimnion during fall overturn was 1,058 Kg and 242 Kg in Lake Wapogasset and Bear Trap lake, respectively. This internal load comprises 38 percent and 64 percent of the total annual phosphorus load to Lake Wapogasset and Bear Trap Lake, respectively.

- Application of alum to the sediments of the two lakes will result in phosphorus load reduction of 498 Kg in Lake Wapogasset and 133 Kg in Bear Trap Lake. This resulted in a fall-turnover epilmnetic concentration in Lake Wapogasset of 37 µg/L (as compared to the 1995 observed volume-weighted epilimnetic of 63 µg/L); and a fall-turnover concentration of 24 µg/L in Bear Trap Lake (a compared to the 1995 observed concentration of 60µg/L).

- Estimated cost for application of alum to the sediments at depth greater than 15 feet in Lake Wapogasset and Bear Trap Lake is $402,800.00 and $69,600.00, respectively, according to the calculation method of Cooke et. al. (1994); and $278,363.00 and $67,465.00, respectively, according to the calculation method of Eberhardt (1995).

In conclusion, the results of this study have shown that anoxic sediment release of phosphorus and subsequent entrainment of phosphorus to the epilimnion results in severe water quality degradation during late-summer in both Lake Wapogasset and Bear Trap Lake. Water quality modeling simulations indicate that application of alum to the lake sediments will provide a notable improvement in late-summer water quality.

Lakes Wapogasset and Bear Trap Improvement Association Wisconsin Lake Planning Grant Project Final Report

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Introduction

Lake water quality data collected by the Lake Wapogasset and Bear Trap Lake Improvement Association during their 1991-1992 Wisconsin Lake Planning Grant Study indicated that excessive nutrient loads from the lakes' watersheds and anoxic lake sediments may be contributing to the eutrophication of both Lake Wapogasset and Bear Trap Lake. Mitigation and reduction of nutrient loads from the watersheds is currently being carried out by the Wisconsin Department of Natural Resources (WDNR) and the Polk Land Conservation Department (LCD) through the Balsam Branch Priority Watershed Project and the Wisconsin Nonpoint Source Pollutant Abatement Program. Therefore, the Lake Improvement Association applied for a second Wisconsin Lake Planning Grant to assess the effects of the anoxic sediments on the lake water quality; and the grant application was approved in January, 1995. This report, which is intended for use by the Lake Improvement Association, WDNR staff, and other technical specialists, summarizes the results and conclusions of the Lake Planning Grant Study, which was conducted from May 1995 through June 1996 by the Lake Improvement Association and their consultant, Barr Engineering Company of Minneapolis, Minnesota. A brief non-technical summary has also been prepared for distribution to members of the Lake Improvement Association and the general public, and is included in Appendix A.

The project objectives listed in the Planning Grant Study application are as follow:

- Estimate the internal load components of the lakes' annual phosphorus budgets from field data on temporal changes in lake phosphorus concentrations.

- Construct time-depth isopleth diagrams of water temperature, and dissolved oxygen and phosphorus concentrations.

- Determine the likely scope and cost of an alum treatment project designed to mitigate recycle of phosphorus from anoxic sediments into the overlying lake waters.

- Estimate the effectiveness and longevity of a lake alum treatment from a literature search.

The results of each of these objectives are discussed in this report. The following introductory section includes information on lake morphometry, watershed land use, and past studies conducted on the lakes. The "Precipitant and Alum Background Information" section contains a literature review of alum effectiveness and longevity. The "Methodology and Results" section contains a description of the data analysis.

Lakes Description

Lake Wapogasset and Bear Trap Lake are located approximately 5 miles from the town of Amery in Polk County, Wisconsin. Bear Trap Lake is located in Lincoln Township and Lake Wapogasset in both Lincoln and Garfield Townships. The two lakes are connected by a short surface channel (Figure 1). The DNR has historically regarded the two lakes as a single two-basin lake; for the purposes of this report they are treated as two distinct lakes. The lakes are ringed by approximately 610 seasonal and permanent residences, two camps (Minneapolis YMCA Camp Icaghowan and Lake Wapogasset Lutheran Bible Camp) and five resorts. There are two public and two private boat accesses on Lake Wapogasset and one public access on Bear Trap Lake. The main tributary entering the lakes is the Balsam Branch, which flows into the north end of Lake Wapogasset from Balsam Lake. A smaller tributary, Friday Creek, enters Lake Wapogasset along the west shore. Morphometries of the two lakes are listed in Table 1.

Table 1 Lake Morphometries

	Surface Area	Maximum Depth	Mean Depth	Volume
	Acres (hectares)	**Feet (meters)**	**Feet (meters)**	**Acre-feet (cubic meters)**
Lake Wapogasset	1,246.1 (504.3)	30 (9.1)	16.7 (5.1)	20,838 (2.570×10^7)
Bear Trap Lake	263.3 (106.6)	25 (7.6)	16.5 (5.0)	4,340 (5.353×10^6)

Watershed Description

The direct watersheds of both lakes lie within the Balsam Branch watershed; the Balsam Branch flows into Lake Wapogasset (Figure 2). Other area lakes contained within the Balsam Branch watershed include Balsam Lake, Half Moon Lake, Deer Lake, Loveless Lake, Antler Lake, and

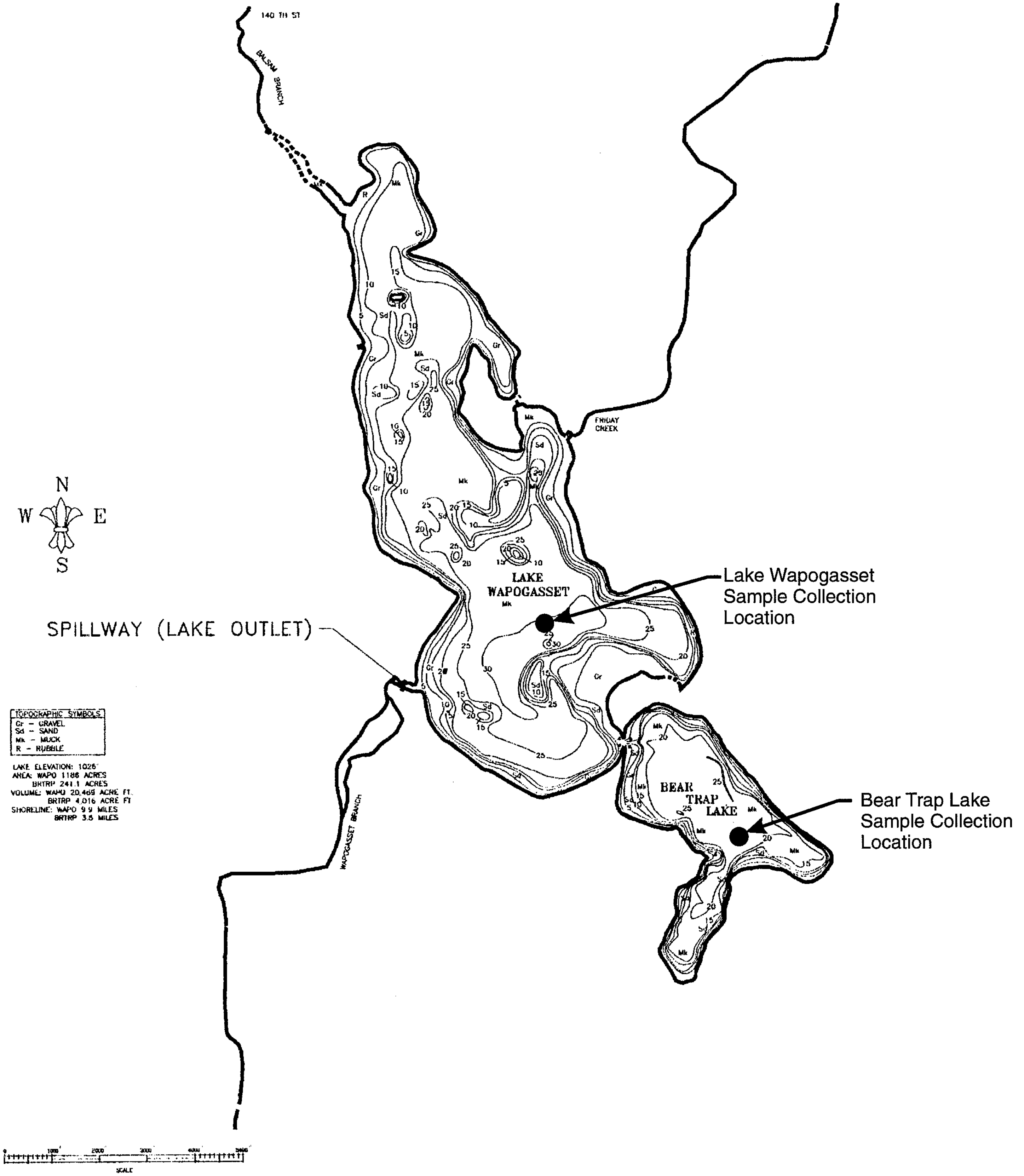
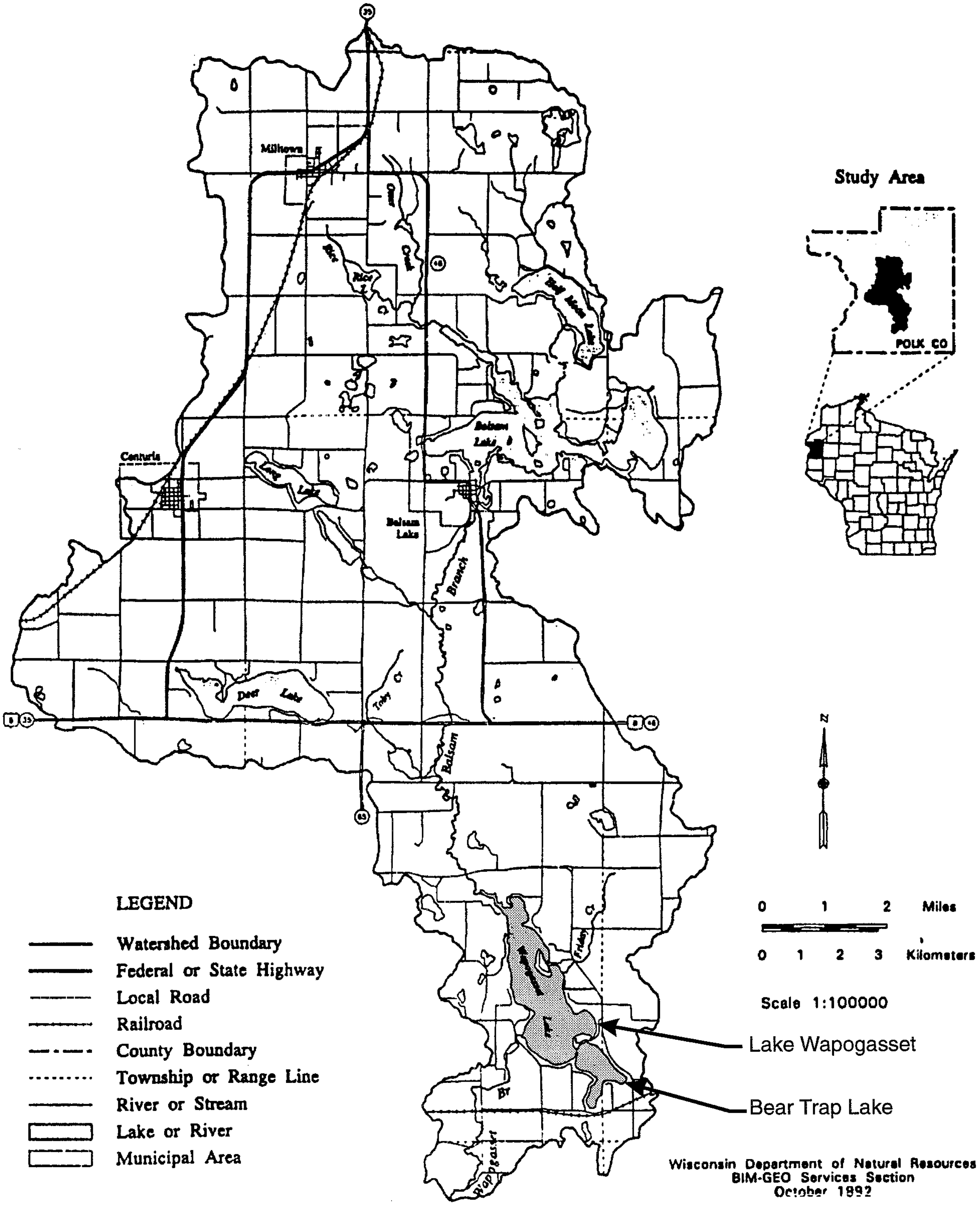


Figure 1

LAKE WAPOGASSET AND
BEAR TRAP LAKE
SAMPLING LOCATIONS



Wisconsin Department of Natural Resources
BIM-GEO Services Section
October 1992

Figure 2

BALSAM BRANCH WATERSHED

Rice Lake. Land use in the direct watersheds of Lake Wapogasset and Bear Trap Lake was identified by the Polk County Land Conservation Department and the WDNR for the purposes of estimating stormwater runoff volume and phosphorus mass during the Balsam Branch Priority Watershed Project (Table 2). Lake Wapogasset also receives the outflows from Balsam, Deer, and Loveless Lakes (which discharge to the Balsam Branch); however, the area and land use in this "extended watershed" is not included in Table 2.

The wastewater treatment ponds serving the Village of Balsam Lake, Wisconsin are located within the Lake Wapogasset watershed. According to the WDNR (Prusak, 1996), the discharge from the ponds seeps to the groundwater, which flows southeast, entering the Balsam Branch approximately 6.5 miles upstream of Lake Wapogasset.

The lake homes surrounding Lake Wapogasset and Bear Trap Lake were connected to a sanitary sewer system in 1978. The wastewater treatment ponds for this system are located approximately 0.3 miles southwest of Bear Trap Lake. The discharge from the treatment ponds seeps to the groundwater; however, there is currently some confusion as to the direction of groundwater flow in the vicinity of the ponds (Prusak, 1996). This issue will be discussed in more detail later in this report.

Table 2 Direct Watershed Land Use

	Total Area	Agricultural	Forested	Wetlands	Residential (Lakeshore)	Residential (Rural)	Residential (Urban)	Wastewater Ponds
	Acres (ha)	**Acres (ha)**	**Acres (ha)**	**Acres (ha)**	**Acres (ha)**	**Acres (ha)**	**Acres (ha)**	**Acres (ha)**
Lake Wapogasset	12,032 (4,869)	4,842 (1,960)	3,865 (1,564)	2,604 (1,054)	428 (173)	256 (104)	36 (15)	n/a
Bear Trap Lake	666 (270)	259 (105)	263 (106)	24 (10)	73 (29)	19 (8)	9 (4)	20 (8)

Past Studies

Several studies have been conducted in the past which have included assessments of the water quality of Lake Wapogasset and Bear Trap Lake. Table 3 provides a list of past studies and report citations, if available. The 1991-1992 Lake Planning Grant Final Report (Lake Wapogasset and Bear Trap Improvement Association, 1993) provides the most comprehensive collection of lake and watershed water quality data, as well as a compilation of the results of a lake user survey. The final report for the Balsam Branch Priority Watershed Project, which has not yet been issued by the WDNR, will contain a comprehensive analysis of potential nonpoint phosphorus sources within the watershed.

Table 3 Past Water Quality Studies

Year	Agency	Citation	Purpose
1972 - 1973	U.S. E.P.A.	n/a	Survey of the Balsam Branch Watershed
1973 - 1976	WDNR	n/a	General Water Quality Survey
1987 - 1988	WDNR	n/a	Water Quality Survey of Lake Wapogasset
1991 - 1992	Lake Wapogasset and Bear Trap Improvement Association	Lake Use and Water Quality Study: Lake Planning Grant Final Report (Lake Wapogasset and Bear Trap Improvement Association, 1993)	Water Quality Survey of Lakes and Tributary Creeks, and Survey of Lake Use
1991- present	WDNR and Polk Land Conservation Department	Balsam Branch Priority Watershed Project : Final Report not yet issued	Land Use and Water Quality Survey of Balsam Branch Watershed
1995 - 1996	Lake Wapogasset and Bear Trap Improvement Association and Barr Engineering Company	Sediment Phosphorus Load and Alum Treatment Assessment: Lake Planning Grant Final Report (this report)	Assessment of Anoxic Sediment Phosphorus Release and Potential Mitigating Effects of an Alum Treatment

Precipitant and Alum Background Information

While this section of the report is not specific to Lake Wapogasset and Bear Trap Lake, it has been included to provide some background information on precipitants in general, and alum in particular.

Introduction to Precipitants

Several chemical compounds have been widely used for decades to reduce phosphorus and/or solids in sewage treatment, lakes, and drinking water. The literature contains many lab and field studies reporting on possible uses and effectiveness. Precipitants discussed here are primarily directed at removing phosphorus from the water column and preventing the resolubilization of sediment phosphorus. Table 4 lists precipitants considered for precipitation, along with their synonyms, chemical formulas, molecular weights, and evaluation dose. The evaluation dose is the concentration generally used in drinking water treatment practices (NSF, 1989).

Table 4 Aluminum, Iron and Calcium Salts Used as Precipitants (NSF, 1989)

Chemical Name	Synonyms	Formula	Approx. Molecular Weight	Evaluation Dose (mg/L)
Aluminum Sulfate	Alum	$Al_2 (SO_4)_3.nH_2O$	594.4 (n=14)	150.0 (25.7 as Al)
Sodium Aluminate	Aluminum Sodium Oxide	$Na_2Al_2O_4$	163.94	40.0 (24.9 as Al)
Ferric Chloride	Iron (III) Chloride	$FeCl_3 . nH_2O$	162.22 (n-0) 270.30 (n=6)	60.0 (20.7 as Fe) 100.0 (20.7 as Fe)
Ferric Sulfate	Iron (III) Sulfate Ferric persulfate	$Fe_2 (SO_4)_3.nH_2O$	399.85 (n=0)	100.0 (28 as Fe)
Calcium Hydroxide	Slaked or Hydrated Lime	$Ca(OH)_2$	74.10	650
Calcium Oxide	Lime, Quicklime	CaO	56.0	500
Calcium Carbonate	Calcite, Limestone, Spent Line	$CaCO_3$	100.09	156g/ 250mL $\simeq$ 624 mg/L

*Evaluation Dose: The concentration of a direct additive used to remove impurities from drinking water. This concentration is based upon use patterns generally found in water treatment practice (NSF 1988).

Chemistry of Precipitation

Precipitants form a floc which is composed of charged particles. Most precipitants are metallic salts. A series of reactions occur upon the addition of metal hydroxide to water. The solubility of metal hydroxides can be written as a dissociation reaction:

$$\mathrm{Me(OH)_n(s)} \rightleftharpoons \mathrm{Me^{n+}} + \mathrm{nOH^-}$$

or as a hydrolysis reaction:

$$\mathrm{Me^{n+}} + \mathrm{nH_2O} \rightleftharpoons \mathrm{Me(OH)_n(s)} + \mathrm{nH^+}$$

The thermodynamic solubility product, K_{sp}, is related to the ion activities, and can be expressed for dissociation as:

$$K_{sp} = \{Me^{n+}\}\{OH^-\}^{n+}$$

or for hydrolysis as:

$$K_{sp} = \frac{\{H^+\}}{\{Me^{n+}\}}$$

If these reactions were limited to a one metal ion and a hydroxides complex, the solubility would be related to pH only. In complex solutions many different species of metal compounds are formed, so that solubility is a sum of the concentration of all soluble species. Whether the reactions are written as a hydrolysis or as a dissociation, pH is important in determining the solubility of a precipitate in water (Skoog and West, 1976). The hydrolysis reaction tends to decrease the pH of aqueous solutions, while the dissociation increases the pH.

The mechanism of coagulation by metallic salts has been elucidated. A combination of mechanisms is responsible for the coagulation or precipitation. A combination of double-layer compression-neutralization of surface charge; adsorption of soluble, polymeric metal-hydroxide species; and precipitation of a metal-hydroxide “sweep floc” are responsible for the actions of alum in water (Dentel and Gossett, 1988).

Aluminum Salts

Aluminum salts have had the widest use and best effectiveness of all the substances reviewed for sediment phosphorus inactivation and water column phosphorus removal. Aluminum sulfate (alum) and sodium aluminate have been used extensively and successfully in lake restoration and water/wastewater treatment for coagulation and removal of solids and dissolved substances. Cooke, *et al.* (1993) reviewed the use of alum for phosphorus inactivation in lakes; Smeltzer (1990), Welch *et al.* (1988), Progressive Architects, Engineers, Planners (1989), Axler, *et al.* (1992) and others have documented the use of alum and results on water quality. Seventeen communities in Minnesota, including Minneapolis and St. Paul, use aluminum in water treatment (see toxicity subsection in Part 2 below).

Alum has been typically used for the removal of color, turbidity, macromolecules and particulates from drinking water. In 1981 more than 153,000 tons (or 300 million pounds) of alum were used in water treatment plants in the United States. In addition, alum has seen increasing use in wastewater treatment applications since the implementation of effluent phosphorus standards. Phosphorus precipitation occurs when the aluminum ion reacts to form insoluble aluminum phosphate. Treatment of wastewater with alum also reduces suspended solids and BOD loads. Aluminum sulfate has also been used for the inactivation and precipitation of phosphorus in fresh water lakes. The alum floc removes phosphorus from the water column and prevents the resolubilization of sediment phosphorus. Treatment of lakes with aluminum sulfate and sodium aluminate is now used worldwide to reduce internal phosphorus loading.

Aluminum Chemistry

Aluminum (atomic element #13) is the third most abundant element in the earth's crust. Aluminosilicates are the most common form of aluminum found in rocks and soils. Aluminum is an important constituent of many inorganic compounds, but is a rare and unimportant component of living organisms. The aluminum content of soils can be quite high, but is present in living organisms in small amounts. Most soil and sediment aluminum is unavailable except under acidic conditions.

Aluminum reactions are slow in approaching true equilibrium. This slow attainment of equilibrium makes measurement of aluminum solutions difficult. In the absence of complexing ions, dissolved aluminum as Al^{3+} hydrates and coordinates with six water molecules to form an octahedral compound. These compounds then form a number of hydrolytic species. This first hydrolysis proceeds as follows:

$$Al(H_2O)_6^{3+} = Al(OH)(H_2O)_5^{2+} + H^+$$

The next step involves the formation of Al $(OH)_2^+$. Formation of monomeric and dimeric hydroxo complexes and polymers to occur as hydrolysis proceeds (Hayden and Rubin, 1974 and Qureshi and Malmberg, 1985).

In natural water systems the aluminum speciation reactions become much more complex. Hem (1970) found that at pH levels below 4, the cation Al^3+ predominates. Acidic waters contain more soluble aluminum than neutral or alkaline waters. Saline waters generally have higher concentrations of aluminum, as will moving waters versus quiescent systems. Dentel and Gossett (1988) examined the speciation of aluminum in water and found that at the pH level of most freshwater systems $Al(OH)_3$ precipitate was prevalent. The chemistry of aluminum in water can thus be viewed as the chemistry of aluminum hydroxide (Burrows, 1977). As the pH values deviated above or below 6, the concentration of the soluble species is related to both pH and aluminum concentration. An examination of the equilibrium constants shows that in natural waters, several competing reactions can occur. The presence of fluoride, silica, sulfate, and hydroxide will have an impact upon the aluminum species formed. Nitrate seems to have little impact upon the precipitation of aluminum. Sulfate affects the pH range of precipitation and stability of the precipitate formed when sulfate concentration is above $2.0 \times 10^{-5}M$. Aluminum-silica reactions exhibit the greatest tendency to proceed to completion ($\log K = 38.7$). The presence of silicic acid inhibits the crystallization of aluminum hydroxide. Fluoride may compete with other anions for available aluminum ions; however, most natural waters with pH 6.5 to 8.5 would have insufficient quantities of fluoride to increase aluminum hydroxide solubility (Burrows, 1977). Equilibrium constants of insoluble aluminum hydroxide for the crystalline phases range from $\log {}^*K_1 = -5.55$ for $AlOH^{2+}$ to $\log {}^*K_4 = -12.35$ for $Al(OH)^-_4$ (Hayden and Rubin, 1974), indicating the stability of the solids.

Investigation of the hydrolysis and precipitation of aluminum (III) by Hayden and Rubin (1974) showed that pH_p (pH of precipitation) and pH_d (pH of dissolution) for aluminum hydroxide were a function of pH and concentration. The pH_p and pH_d values were also affected by temperature and time. The range of pH_p - pH_d represents pH at which $Al(OH)_3$ is insoluble. Hayden and Rubin (1974) indicates that the stability of aluminum hydroxide increases over time, especially at the higher concentrations. The solubility constants for aluminum compounds also suggest that

$Al(OH)_3$ is quite insoluble at a neutral pH ($K_{sp} = 5 \times 10^{-33}$). The removal of phosphorus from water by alum is accomplished by the formation of the insoluble aluminum phosphate precipitate ($K_{sp} = 9.92 \times 10^{-29}$), sorption onto floc, or entrapment of particulate phosphorus by the floc (Cooke *et al.*, 1986).

Aluminum-phosphate complexes are unaffected by redox changes and, therefore, are largely unaffected by the anoxic conditions that are common in the hypolimnia and in sediments. Inorganic phosphorus removal is maximized when Al:P ratios are high.

Iron Salts

Iron salts ($FeCl_3$, $FeSO_4$) are an alternative to aluminum salts in water and wastewater treatment (see case studies below). The results for the iron salts would be similar to the scenario for aluminum salts; that is, the required concentrations of iron salts to precipitate phosphorus from the water would be a small fraction of concentrations used for drinking water treatment. The following equation can be used to estimate iron dose to the lake (Walker 1986):

$$\text{Fe dose (mg/L)} = 0.04\ (P_o - P_t)\ P_t^{\ -0.408}$$

Where P_0 = initial ortho P, P_t = target ortho P
Assuming P_0 = 60 µg/L and P_t = 5 µg/L, the calculated iron dose is 1.14 mg/L, which is approximately 6 percent of the evaluation dose.

In natural waters, phosphorus is removed from oxygenated waters through the precipitation of ferric phosphate and through absorption to ferric hydroxide floc (Cole 1979, Wetzel 1975). Changes in redox potential associated with anoxia lead to the reduction of iron from the ferric (III) to the ferrous (II) state. Ferrous compounds are soluble in water (except FeS), thus allowing the bound phosphate to resolubilize. Therefore, iron salt inactivation is likely to be the most effective precipitant for use in the upper oxic regions of a lake.

Counter Ions

In addition to the aluminum and iron cations (i.e., positively charged ions) that are liberated into the water when aluminum and/or iron salts are used as precipitants, these metal salts each contain a negatively charged counterion called an anion. The anions that are contained in alum and ferric chloride are sulfate (SO_4^{2-}) and chloride (Cl^-), respectively. Both are naturally abundant in surface waters.

Sulfate (SO_4^{2-}) and chloride (Cl^-) are two of the most abundant anions in freshwater, surpassed only by carbonate (Cole 1983). The addition of these anions via alum or iron chloride treatment would not have a significant influence on the background concentrations of these anions. Sulfate is removed from the water by protein synthesis of organisms, reduction to sulfide (which is either bound to sediments or released as a gas, H_2S), or adsorption to mineral particles (which settle out). Chloride is stored in algal cells and combines with all major cations (Na, K, Mg) to form soluble salts. Neither sulfates nor chlorides from precipitant usage would constitute a health or safety risk.

Environmental Effects of Precipitants for Phosphorus Inactivation

The following environmental issues are discussed as they relate to use of these precipitants:

- Groundwater contamination
- Bioaccumulation
- Toxicity
- Sediment accumulation
- Sediment chemistry dynamics
- Long-term efficiency

These environmental issues are common to the three uses discussed above.

Bioaccumulation

The precipitants being considered do not bioaccumulate (Weiss, 1986; Sax and Lewis, 1989). For a chemical to bioaccumulate it must have an affinity for lipids or sulfide functional groups, as do many hydrophobic organic chemicals. These precipitants do not have characteristics of bioaccumulative substances; therefore, they not bioconcentrate or biomagnify in animal tissue.

Toxicity

Human health effects due to exposure to aqueous aluminum have been suggested with regard to Alzheimer's disease (Qureshi and Malmberg, 1985). However, subsequent and more extensive studies by Harriger and Steelhammer (1990) report that the U.S. EPA, FDA and medical authorities concur that aluminum is not a risk factor for Alzheimer's disease. The research linking aluminum to Alzheimer's disease has been seriously undermined by a recent study that demonstrated aluminum in stained brain tissue samples was an artifact-contamination from airborne dust (Landsberg, McDonald, and Watt 1992). Using a new technique that does not rely on the staining procedure, these researchers found no aluminum in the 80 tissue samples from Alzheimer's patients.

Soluble aluminum is the most chemically and biologically available form of aluminum. Soluble aluminum represents an extremely small portion of the total biosphere aluminum mass. The biological cycling of aluminum is quantitatively a small portion of the overall aluminum cycle, as aluminum does not serve as a plant or animal nutrient. Aluminum does not bioaccumulate in living tissue, but at high concentrations is toxic to many organisms (Table 5).

Table 5 Acute Aluminum Toxicity to Bottom Dwelling Aquatic Organisms

Organism	Concentration	Reference
Circumneutral pH		
Midges (*Tanytarsus dissimilis)*	> 79.900 mg/L	U.S. EPA (1988)
Snails (*Physa* sp.)	30.600 mg/L	U.S. EPA (1988)
Fathead minnows (*P. promelas*)	35.000 - > 49.800 mg/L	U.S. EPA (1988)
Catfish (*I. punctatus*)	> 47.900 mg/L	U.S. EPA (1988)
pH = 6.5		
Carp (*C. carpio*)	4.000 mg/L - 30% mortality 48 hrs	U.S. EPA (1988)

A large percentage of aluminum is not toxic at naturally occurring concentrations. Wood (1985) and others have speculated that the use of elements by organisms is dictated by the element's abundance and solubility under anaerobic conditions found 4 billion years ago. Aluminum was abundant, but insoluble, under the conditions that existed as life evolved. Aluminum was unavailable to primitive bacteria and as a result most organisms have not adapted to high aluminum concentrations. Organisms, especially plants, assimilate only limited quantities of aluminum and most organisms experience toxicity from high intracellular accumulations (Driscoll 1989). Solution pH is an important factor when evaluating the toxicity of aluminum compounds, as the lethal limit for many organisms occurs at about pH 4. Most toxicity studies have dealt with three forms of aluminum water: aluminate anion, molecular (suspended) aluminum hydroxide, and aluminum cation (Burrows, 1977). Aluminum toxicity can be either acute or chronic. Numerous investigators have addressed the toxicity of aluminum to organisms under laboratory conditions. This discussion will be limited to the effect of aluminum upon aquatic organisms. Unless otherwise cited the toxicity data are derived from Burrows (1977).

Most aluminum toxicity testing of terrestrial bacteria has been directed at soil organisms. Much of the toxicity data for aquatic bacteria were accumulated through work dealing with the effects of alum dosing in wastewater treatment plants. Hsu and Pipes (1973) noted that 10 mg Al/L did not interfere with nitrification in the activated sludge process. Their research found doses of aluminum hydroxide up to 300 mg/L did not cause changes in the forms of nitrogen found in activated sludge effluent. They suggested that aluminum hydroxide was not toxic to nitrifying bacteria. Morgan, *et al.* (1977) stated that aluminum hydroxide did not have any adverse effects upon biological wastewater treatment activities.

Protozoans were found to be sensitive to aluminum. *Amoeba* underwent an enlargement of contractile vacuoles when exposed to 0.01 mg Al/L. Ciliates were found to have a 28-hour toxicity threshold of 12 mg Al/L at pH 7.5 to 7.8. Many protozoa are sensitive to acidic conditions and most investigators did not distinguish between the effects of aluminum and changes in pH.

The class Chlorophyceae is the only group of algae for which good toxicity data exists. Desmids seem to be very tolerant to aluminum, possibly due to their proliferation in oligotrophic waters where dissolved aluminum concentrations tend to be naturally higher. *Chlorella* was found to tolerate aluminum concentrations of up to 48 mg Al/L before shown a 13 percent inhibition of growth. *Scenedesmus* exhibited a toxicity threshold of 1.5 to 2.0 mg Al/L. *Spirogyra* and *Zygnema* seemed to have the ability to resist plasmolysis in the presence of the aluminum ion.

Daphnia magna has been commonly used for aluminum toxicity testing. Twenty hour exposures of *Daphnia* to 22 mg Al/L resulted in total immobilization, while 64 hour exposure to 1.4 mg Al/L had the same results. Chronic exposure (three weeks) to 0.32 mg Al/L resulted in a 16 percent reproductive impairment. Cooke, *et al.* (1986) suggested that dissolved aluminum concentrations of less than 50 µg/L have no effects of either *Daphnia* or chironomid larvae.

Many common species of aquatic macrophytes are tolerant of aluminum. *Typha* and *Phragmites* were unaffected by exposure to 5 mg soluble Al/L for 15 weeks. *Myriophyllum spicatum* was found to be inhibited by 50 percent when exposed to 2.5 mg Al/L (pH not specified). In contrast to the lack of bioaccumulation in animals, *Lemna minor* has been found to bioaccumulate aluminum, at a concentration factor of 660,000.

A much larger data base is available for aluminum toxicity in fish. Exposure to aluminate causes chronic injury, primarily due to gill hyperplasia (gill-clubbing). Exposure to 0.05 mg aluminate ion/L is acutely toxic to trout. Freshly precipitated aluminum hydroxide does not cause acute toxicity in fish, but it can cause chronic injury. Chronic symptoms are similar to those seen following exposure to aluminates, but develop more slowly (Freeman and Everhart, 1971 and Everhart and Freeman, 1973). Aluminum was not found to be toxic to fish eggs at typical testing doses. Increased receiving water turbidity reduces the toxicity of aluminum to fish. This may be due to absorption of aluminum onto particulates.

The U.S. EPA (1988) states that freshwater organisms would not be effected in a negative manner by aluminum, at a pH between 6.5 and 9.0 if:

> "the four-day average concentration of aluminum does not exceed 0.087 mg/L more than once every three years on the average, and if the one hour average concentration does not exceed 750 µg/L more than once every three years on the average."

Cooke, *et al.* (1986) suggests that ingestion of bacteria that could be entrained in the alum floc could be hazardous to swimmers and recommends that lakes treated with alum for phosphorus inactivation be closed for a period of time following application. This is because of the concern about pathogens, not the toxicity of the alum. Normal Al intake in humans from all sources is estimated as 12-14 mg Al per day (Marcus *et al.* 1992). Most dietary aluminum is excreted by the kidney. Healthy individuals have plasma or serum aluminum concentrations of 5 to 15 µg Al/L (Garrot 1986). In their limnological assessment of mine pit lakes for aquaculture use, the Natural Resources Research Institute reviewed literature on potential human health risks of aluminum in drinking water (Axler, et al. 1992). They present an example of aluminum ingestion if people were to drink Twin City-South water without treatment. Assuming 2 liters/day consumption (normal daily adult intake, after 1 year the total amount of aluminum from the water would be 0.3 µg Al, which is <3 percent of the normal daily adult intake.

Free aluminum ion is the only possible toxic product from treatment with these precipitates. The AWWA has set the goal of not more than 0.05 mg free (i.e., dissolved) Al/L in finished drinking water. The AWWA guideline of 0.05 mg Al/L matches the residual dissolved aluminum (RDA) which is the not to exceed level shown to be safe for rainbow trout (Everhart and Freeman, 1971). The nature of these chemicals is that they form very stable precipitants and are insoluble, except under acidic conditions (pH <5.0). In alum-treated waters with alkalinity greater than 20 mg $CaCO_3$/L, there is virtually no dissolved aluminum residual, even at high alum doses, as long as its final pH equals or exceeds 6.0 (Cooke and Kennedy, 1981).

A review of the literature indicates there is no evidence for toxicity of iron salts in solution. Iron is regulated as a secondary drinking water standard because of its potential for staining porcelain tubs, clothing, etc. This is apparent in the widespread use of iron salts for water treatment (see case studies below). For example, the City of Saint Paul water supply reservoir is continually dosed with iron chloride, using over 200,000 pounds per year.

There is no ecological concern evident in the use of iron salts at concentrations proposed for use as a precipitant. Iron is obviously an existing and abundant element in the iron mine pits.

The only human health concern with iron salts is for proper handling of the powdered material to avoid irritation of mucous membranes and skin. The proposed use of iron salts as a precipitant would be to apply it as a liquid into the net pens. Therefore, with proper handling and proper storage (including secondary containment), the proposed use of iron salts as a precipitant would be safe for both environmental and human health.

Sediment Accumulation

The alum floc (aluminum hydroxide) formed on the lake bottom by precipitants will initially not exceed 4-inches in depth. With time, the floc will continue to crystallize becoming more stable and compressed (Cooke *et al.* 1993). Natural sedimentation of erodible soils in the basin and internal production (i.e., dead algae and aquatic plants) will eventually compress and bury the floc layer from the single alum treatment. Thus, the aluminum becomes permanently deposited in the sediments. This sediment deposit of aluminum salts will have no effect on the public's natural resources.

Sediment Dynamics/Chemistry

At the sediment-water interface the chemistry is quite dynamic. Phosphorus and iron (Fe) remain bound in the sediments when the overlying water is aerobic (dissolved oxygen >0). If the water becomes anoxic (due to continued respiration, but no introduction of oxygen due to temperature stratification of the lake), the amorphous ferric-hydroxy-phosphate complex dissociates and releases reduced (i.e., ferrous) iron and phosphate ions into the overlying water. These ions will diffuse upward a distance that is dependent upon the concentration gradient between the water column and the interstitial pore waters. The release of iron as ferrous ions under anoxia will also be regulated by the levels of sulfide generated by anaerobic, sulfate reducing bacteria because ferrous sulfide is insoluble. However, the iron complex will again precipitate to the sediment upon re-oxygenation during the natural process of turnover in the spring and fall season. In contrast, $Al(OH)_3$ from alum treatment continues to sorb phosphorus, and will not dissociate even under the anoxic reducing conditions of mid-summer (Cooke *et al.* 1993).

Long-Term Efficiency

Alum treatments have demonstrated effectiveness in maintaining low P concentrations for at least 12 years following treatment (Cooke *et al.* 1993). The alum is effective because it continues to sorb P long after it has been applied. Successful long-term single treatment generally depends on external loadings of P being reduced or eliminated prior to a one time alum treatment.

Iron salts (ferric chloride, ferric sulfate) are also effective at precipitating P from the epilimnion and will remain insoluble in the sediments, as long as they are oxygenated. If anoxic conditions occur, reduced iron will dissolve into the water column releasing the bound phosphate. This phenomenon is commonly seen in lakes that thermally stratify in summer and become anoxic in the bottom waters. Thus, the long-term efficiency of iron salts is dependent on the dissolved oxygen concentration.

Ore, at this required circumneutral pH, the $Al(OH)_3$ would continue to crystallize and become increasingly stable. Iron (FeIII) remains 100 percent bound to hydrous oxide surfaces above pH 3.0; therefore, iron would remain in a solid form in the oxygenated sludge.

Case Studies

This section presents case studies where alum and iron chloride have been used in:

(1) Lake Restoration
(2) Wastewater Treatment
(3) Water Treatment

Lake Restoration

When the Twin City-South mine pit lake (located in Chisholm, MN) was treated with 29,100 gallons of liquid alum on July 2, 1990, the total mass of phosphorus dropped from 1,856 lbs to 828 lbs (a 55 percent reduction). Aluminum concentrations in the water column returned to baseline within 9 days after treatment and continued to drop below aluminum concentrations prior to treatment. The post-treatment mean value of 23± 13 µg Al/L for epilimnion water was similar to values of 22 and 26 µg Al/L for the City of Duluth, Minnesota's finished water supply for 1989 and 1990.

What follows are other examples of precipitant use to remove and inactivate phosphorus (Table 6 and Table 7). Cooke *et al.* (1993) noted that nearly every aluminum treatment has been successful in reducing phosphorus release from sediments.

Mirror Lake, Wisconsin is a small (5 ha), alkaline (alk.=222 mg/L as $CaCO_3$) urban lake that was treated with alum in 1978 (6.6 mg Al/L) following storm drainage diversion in 1976. The lake is destratified in spring and fall to ensure turnover. Total and dissolved reactive phosphorus was monitored before and after treatment, as well as in 1988, 1989 and 1990. Volume weighted mean P concentrations have remained far below pretreatment levels for 12 years. P levels 10 years following treatment were slightly higher than immediately following treatment. In 1991, the $Al(OH)_3$ layer was 8 to 12 cm below sediment that had accumulated since alum treatment. This new layer of sediments is contributing the higher P levels (Garrison and Ihm 1991).

Table 6 Comparison of Applications to Lakes to Precipitate or Inactivate Phosphorus (From Axler, *et Al.*, 1992), and to Clear Lake, Waseca, Minnesota

Source	Dose	Notes
	Volumetric (mg Al/L):	
Cooke & Kennedy (1981)	Mean: 9.6 + 7.4 Median: 7.7 Range: 0.5 - 26	27 lake applications
Clear Lake, Waseca, MN*	Hypolimnetic vol.: 22	
Twin City-South, Chisholm, MN	Whole-lake: 0.81 Epilimnion: 1.45 (0-20 m)	Calculated from dose assuming uniform mixing
	Areal (g Al/m2):	
Smeltzer (1990)	Mean: 43 + 37 Median: 32 Range 18 - 139	8 hypolimnetic and metalimnetic applications
Clear Lake, Waseca*	38	Calculated from surface dose.
Twin City-South, Chisholm, MN	23	Calculated from surface dose

*Clear lake was a successful MPCA Lake Restoration project.

A case very similar to Mirror Lake is Shadow Lake, Wisconsin. Shadow Lake is larger (17 ha) than Mirror Lake in surface area, but shallower (mean depth for Mirror, 7.8 m; for Shadow, 5.3 m). With a alum dose of 5.7 mg Al/L and alkalinity of 188 mg/L as $CaCO_3$, Shadow Lake had a long-term response to alum treatment that is identical to Mirror Lake. Shadow Lake did not have a destratification system installed. These are two good examples of the long-term benefit of a one-time alum treatment.

A good example of $FeCl_3$ use is in the water supply reservoir for Minnesota's capital city (Walker, et al. 1989). The St. Paul Water Utility (SPWU) has been injecting $FeCl_3$ into its water supply reservoirs since 1985. The water supply system consist of three impoundment reservoirs, fed by diversions from the Mississippi River and local watersheds (Schuler, 1990). $FeCl_3$ has been injected into incoming Mississippi River water since 1985 and into Lambert Creek—the mouth of a highly enriched watershed—since 1990. Previous jar tests indicated the potential for 70 percent removal of soluble P. In the fall of 1986, two hypolimnetic aerators were installed in Vadnais Lake—the terminal water supply reservoir—and $FeCl_3$ injection into the aerators was begun in 1987. The SPWU has injected 80-10 kg/day of $FeCl_3$ into each of the hypolimnetic aerators. Orthophosphate rapidly responds to the $FeCl_3$ injection. Phosphorus has been reduced 100 percent with the combined hypolimnetic aeration and $FeCl_3$ injection. They found that without $FeCl_8$ phosphorus was only reduced by 60 percent. The SPWU has continued to inject $FeCl_3$ at the three locations to keep phosphorus levels low. In 1992, SPWU used 211,000 lbs $FeCl_3$ at these injection points. The SPWU has affectively used Vadnais lake (a natural, popular, metropolitan, multiple-use recreation lake) as an extension of its water treatment plant for processing new water for human consumption. In essence Vadnais lake has been used by the SPWU as a primary clarifier, or the first step in the process of producing finished drinking water.

Table 7 Comparison of Areal Aluminum Dose Rates Used in Nine Metalimnetic or Hypolimnetic Lake Treatments (From Smeltzer 1990)

Lake	Year Treated	Area Treated (HA)	Areal Dose Rate (gAl/M^2)	Reference
Cochnewagon, Maine	1986	91	18	W. Dennis (pers. comm., Cobbossee Watershed Dist., ME)
Annabessacook, Maine	1978	113	18-25	Dominie (1980) and W. Dennis (pers. comm., Cobbossee Watershed Dist., ME)
Three Mile, Maine	1988	259	25	R. Bouchard (pers. comm. ME Dept. Env. Prot.)
Dollar, Ohio	1974	1.4	29	Cooke and Kennedy (1981)
West Twin, Ohio	1975	16	32	Cooke and Kennedy (1981)
Kezar, NH	1984	48	40	Connor and Martin (1989)
Bullhead, WI	1978	5.6	42	Narf (1985)
Morey, VT	1986	133	44	Smeltzer (1990)
Irondequoit Bay, NY	1986	305	139	Monroe County Dept. Health, NY (1986, 1989)

Prior to introduction of $FeCl_3$ treatment to Vadnais Lake, the method had been successful in reducing algae growth in reservoir surface water supply systems in England (Hayes *et al.*, 1984) and the Netherlands (Bannink and VanDer Vlugt, 1978).

The mechanism of phosphorus removal by iron chloride ($FeCl_3$) differs from aluminum salts in that it affects soluble phosphorus and does not floc and trap particulate matter. Phosphate adsorbs to insoluble ferric hydroxides. The adsorption process is influenced by pH, mixing, temperature, and oxidation-reduction potential (Lijklema, 1980). It works best in high alkalinity, circumneutral pH lakes, which is the case for the mine pits. At concentrations similar to those used for alum (approximately 20 mg Fe/L), $FeCl_3$ will produce a sweep floc similar to alum. At the concentrations used by SPWU (3-4 mg Fe/L), however, the iron salt still efficiently removes soluble phosphorus.

Iron chloride has two advantages over alum for continual or intermittent use. First, iron chloride is clearly nontoxic and therefore avoids any controversy about perceived (versus real) human health toxicity. Second, iron chloride is effective at much lower doses than alum in lakes with high alkalinity. In Vadnais Lake, with an alkalinity of more than 125 mg $CaCO_3$/L, SPWU determined that 10 ppm of alum was required to precipitate the phosphorus, whereas only 0.5 ppm of $FeCl_3$ was needed. Given the high alkalinities of the mine pits, the dosage difference between alum and $FeCl_3$ will be at least as large as for Vadnais Lake. Qureshi and Malmberg (1985) compared the use of alum and ferric chloride in drinking water treatments. Ferric chloride proved to be effective at one-half the dose of alum required for effective coagulation. Schuler (1991a) reported that the use of ferric chloride in conjunction with aeration could reduce soluble phosphorus concentrations by 70 percent.

Wastewater Treatment

Alum and iron chloride additions are used as advanced wastewater treatments to clarify and reduce phosphorus to required effluent limits. Examples in Minnesota include Bayport and Stillwater WWTP in the Twin Cities Metropolitan area and Rochester. Bayport uses average alum concentrations of 100 mg Al/L, whereas Stillwater uses an average of 61 mg Al/L. At the Rochester Water Reclamation Plant, both alum and $FeCl_3$ were used at several stages of treatment. Both liquid $FeCl_3$ (38 percent solution) and liquid alum (48 to 49 percent solution) are used at a rate of 800 gal/day for the 13.5 MGD plant. The concentration of $FeCl_3$ averages 11 mg Fe/L and alum only about 2 mg Al/L. Only a low dose of alum is needed because it is used in combination with a polyelectrolyte. These several cases illustrate the wide range of concentrations depending on the specific needs of the facility. As discussed above, the estimated concentration of $FeCl_3$ in the fish manure collection system would be approximately 3 mg Fe/L.

Water Treatment

Alum and $FeCl_3$ are used in potable water treatment for precipitation and coagulation of suspended solids (i.e., turbidity). Removal of turbidity is a prerequisite for effective disinfection.

Water treatment facilities in our nation's capitol offer some good examples of aluminum and iron salt usage. The Washington Aqueduct, run by the U.S. Army Corps of Engineers, treats approximately 200 MGD serving 1 million people. During the summer, the system reservoir is dosed daily with copper sulfate ($CuSO_4$) to control algae. After the water is pumped from the reservoir, alum is added as a coagulant at an average concentration of 20.3 + 4.7 mg/L (based on dosage rates at two plants on July 1, July 15 and July 30, 1993). This is equivalent to 20,500 ± 5,530 lbs/day or 7.5 million lbs/yr. of aluminum sulfate (D. Schell, Washington Aqueduct Division, personal communication).

An example of iron chloride usage in Washington, D.C. is at the WSSC Potomac River Filtration Plants. Like the Washington Aqueduct, the WSSC operates two plants. The smaller plant (40 to 50 MGD) uses alum at a concentration of approximately 8 mg/L alum or approximately 1.1 million lbs/yr of alum. At the larger plant (~165 MGD) iron chloride is used exclusively at an average daily concentration of approximately 12 mg/L. This daily concentration is equivalent to 16,513 lb/day, or approximately 6 million lbs/yr. (B. Fisher, WSSC personal communication). Therefore, in addition to the daily treatments of copper sulfate, the Washington, D.C. water treatment plants use a combined total of approximately 14.6 million pounds of alum, iron chloride, and aluminum sulfate precipitants annually in processing potable drinking water.

Minnesota cities using these coagulants in their water treatment processes are listed in Table 8. This information was collected in 1988 by the Minnesota Department of Health. Since that time, at least three additional communities have begun using alum or iron salts (Ely, Richfield, and Virginia) according to Wm. Anderl, MDH. A total of 32 communities use some type of precipitant as part of their water treatment process. Just over half of these (17) communities use alum as a precipitant. Sixteen of the 18 communities using alum rely on surface water rather than groundwater. Four communities, that include Minneapolis and St. Paul, use iron salts (e.g. $FeCl_3$). Overall, 1.2 million people (27 percent of Minnesota population) are served by utilities using coagulants and, 1.1 million people (25 percent of Minnesota's population) by plants using alum.

For example, the largest utility, Minneapolis, has an average daily production of 64,960,000 gallons of potable water. It uses alum at an average of 8,888 lbs/day (3.2 million pounds of alum per year), with a minimum of 4,267 lbs/day and maximum of 22,143 lbs/day (Paul Koski, Chemist, MWU). This equates to an alum concentration of 12 mg/L to 40 mg/L, or an average of 16 mg/L. In addition, Minneapolis also uses ferric chloride at a rate of 4.5 to 5 mg/L, equating to an average daily usage of approximately 2,500 lbs/day (900,000 lbs/yr). The combined annual precipitant used in processing Minneapolis potable water exceeds 4.1 million pounds of alum and iron chloride. The information in Table 8 is based on a survey of Minnesota water treatment operators by the Minnesota Department of Health. In the multiple choice responses to coagulants/precipitants used, the operator checked "other" if a coagulant/precipitant was not listed.

Table 8 Minnesota Communities Using Precipitants in Their Water Treatment (Source: Minnesota Department of Health, 1988)

Community	Coagulant/ Precipitant	Pop. Served	Avg. Daily Prod. (1,000 gal.)	Source
Beaver Bay	alum/polymer	265	34	L. Superior
Biwabik	alum/polymer	1,432	160	Canton Pit
Breckenridge	alum	3,900	550	Groundwater/
Duluth	alum/polymer	87,000	22,000	L. Superior
Eden Prairie	iron salt/	20,000	3,100	Groundwater
Fairmont	alum/iron salt	11,506	2,300	Groundwater/Budd L.
Fergus Falls	iron salt	12,600	1,570	L. Wright
International Falls	alum/"other"	6,700	900	Rainy R.
Little Falls	alum	7,388	1,307	Groundwater
Minneapolis	alum/iron salt	480,526	64,960	Mississippi R.
Oslo	alum/polymer	400	60	Red R.
St. Cloud	alum	48,255	4,833	Mississippi R./
St. Paul	alum/iron salt	385,000	50,600	Mississippi R./
Silver Bay	alum/polymer	2,200	225	L. Superior
Staples	alum	2,800	312	Groundwater
Stephen	alum	958	90	Tamarac R.
Thief River Falls	alum/polymer	9,950	1,200	Red Lake R.
Two Harbors	alum/polymer	4,400	850	L. Superior
White Bear Lake	alum	24,000	2,400	Groundwater
Bellingham	polymer	290	20	Groundwater
Carlton	lime/polymer	862	165	Groundwater
Chisholm	"other"	5,150	500	Fraser Pit
Currie	polymer	359	36	Groundwater
Eagle Bend	polymer	623	45	Groundwater
E. Grand Forks	soda ash/"other"	9,000	1,250	Red Lake R.
Granite Falls	"other"	3,200	275	Minnesota R.
Mankato	polymer	28,651	4,000	Groundwater
Moorhead	"other"	30,200	4,400	Groundwater/Red R.
Nicollet	polymer	709	55	Groundwater
Red Rock	polymer	1,234	224	Groundwater
Round Lake	polymer	480	47	Groundwater
Tonka Bay	polymer	1,453	200	Groundwater

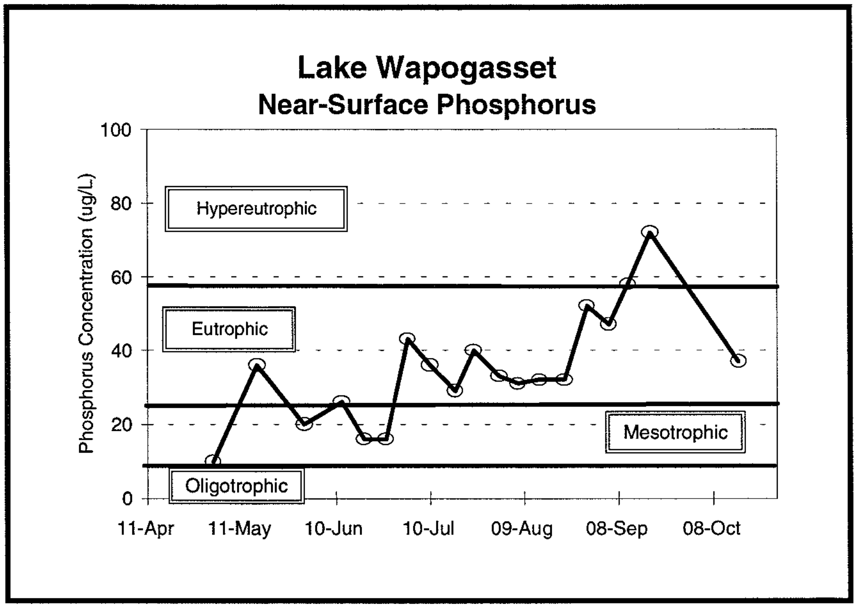
Data Collection and Water Quality Assessment

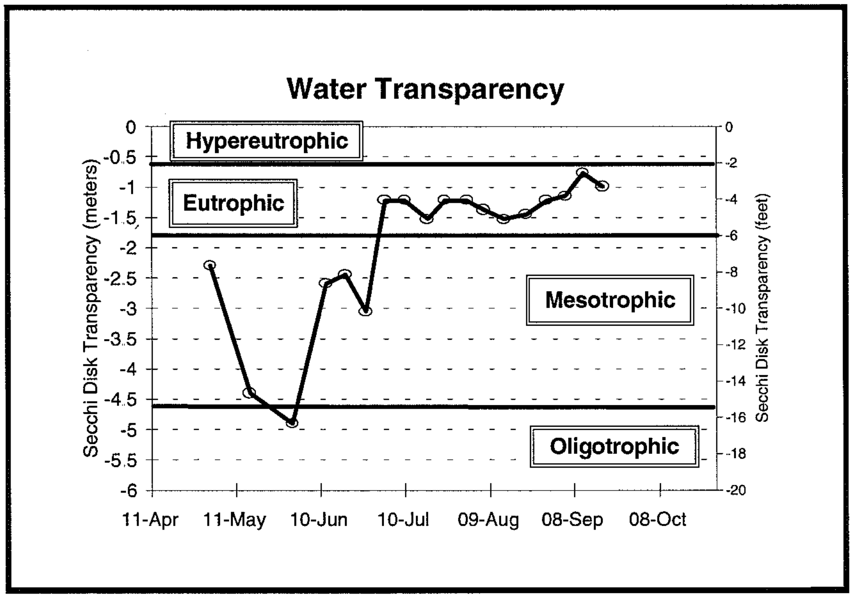
Field data was collected from the deep hole in each lake (Figure 1) by Lake Wapogasset and Bear Trap Improvement Association volunteers, with some assistance from WDNR personnel. Samples were collected biweekly from early May to mid June, 1995 and thereafter collected weekly until September 18, 1995. Temperature and dissolved oxygen data were collected at 1 meter intervals in each lake using an YSI Model 51B dissolved oxygen meter. Total phosphorus samples were collected using a peristaltic pump from 1, 3, 5, 6.5, 7.5, 8.5, and 9.5 meter depths in Lake Wapogasset, and from 1, 3, 4.5, 5.5, and 6.5 meter depths in Bear Trap Lake. Total phosphorus samples were preserved with sulfuric acid, and forwarded to the Wisconsin State Lab of Hygiene in Madison for analysis. Field and chemistry data are included in Appendix B. In addition, Barr Engineering analyzed some additional samples for total dissolved phosphorus. These results were used to corroborate the internal phosphorus load results derived from the State Lab of Hygiene concentrations. The results of Barr's laboratory analysis are also included in the water quality table (Table B-2).

Surface water phosphorus and chlorophyll *a* concentrations and Secchi disc transparencies for Lakes Wapogasset and Bear Trap were plotted and classified according to the Carlson Trophic State Index (Carlson, 1977) (Figures 3a and 3b). Both lakes exhibit a similar pattern in seasonal water quality. During May, the surface phosphorus and chlorophyll *a* concentrations were quite low, while the water transparency was relatively high (resulting in trophic state classifications of oligotrophic/mesotrophic). By mid-September, the phosphorus and chlorophyll *a* concentrations had increased dramatically, and as a result the water transparency diminished considerably; the trophic state classification of both lakes was eutrophic/hypereutrophic. Both lakes also exhibited a brief elevation in surface water phosphorus concentration during early-July. As will be discussed later, this elevation in concentration appears to be associated with a temporary degradation of the thermal stratification, most likely associated with a period of cool and windy weather.

Time-depth diagrams of temperature, dissolved oxygen, and total phosphorus isopleths were also prepared for the two lakes (Figures 4a and 4b). Both lakes began to thermally stratify in early June. Stratification in both Lake Wapogasset and Bear Trap Lake was relatively weak throughout the summer. Attempts were made to calculate and identify the thermocline depth for each

Figure 3a: Lake Wapogasset Trophic State Indices





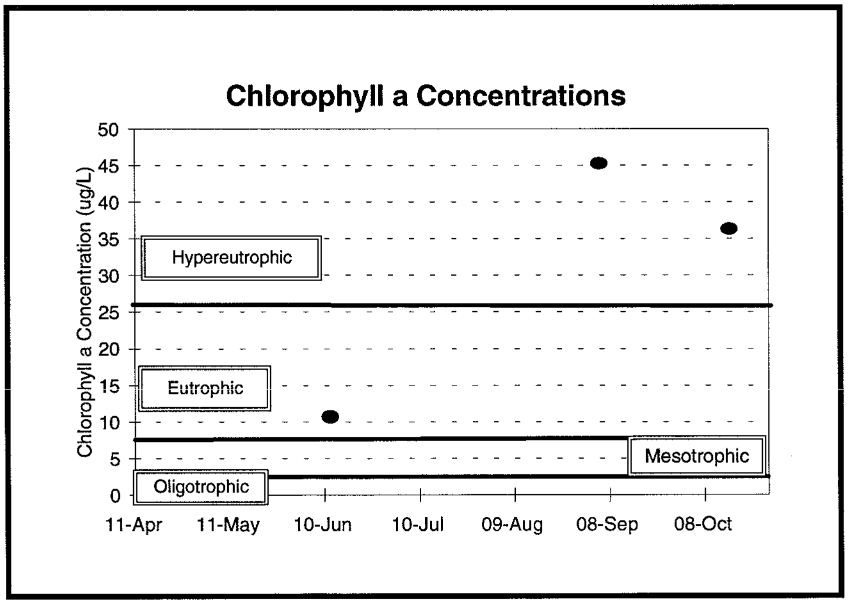
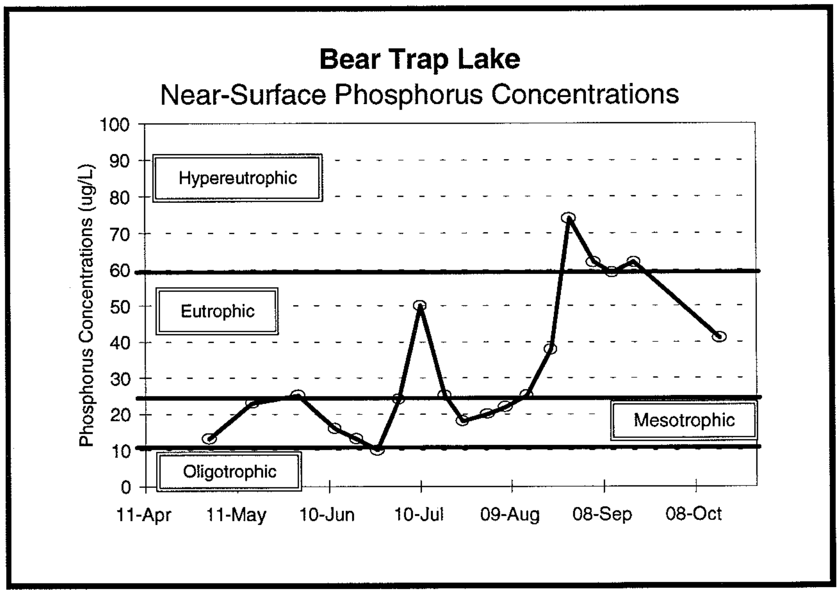
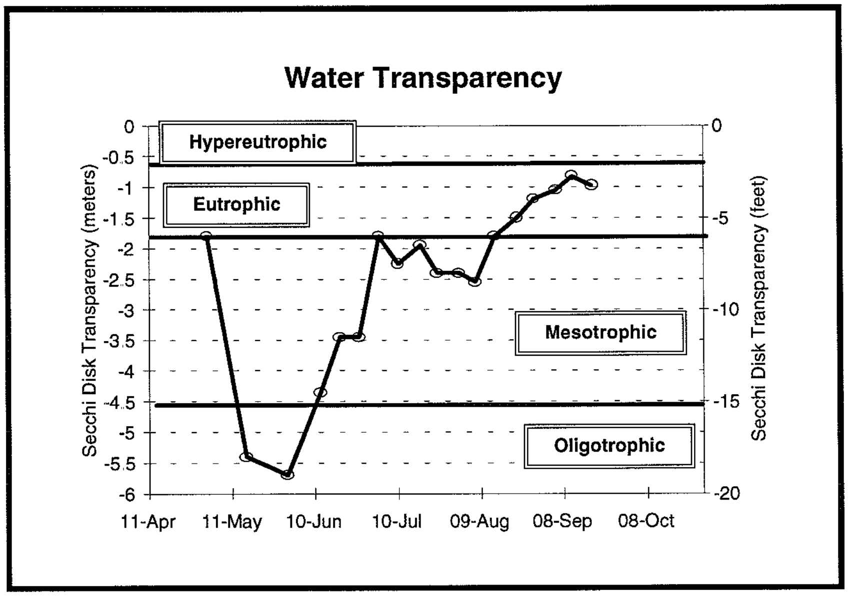
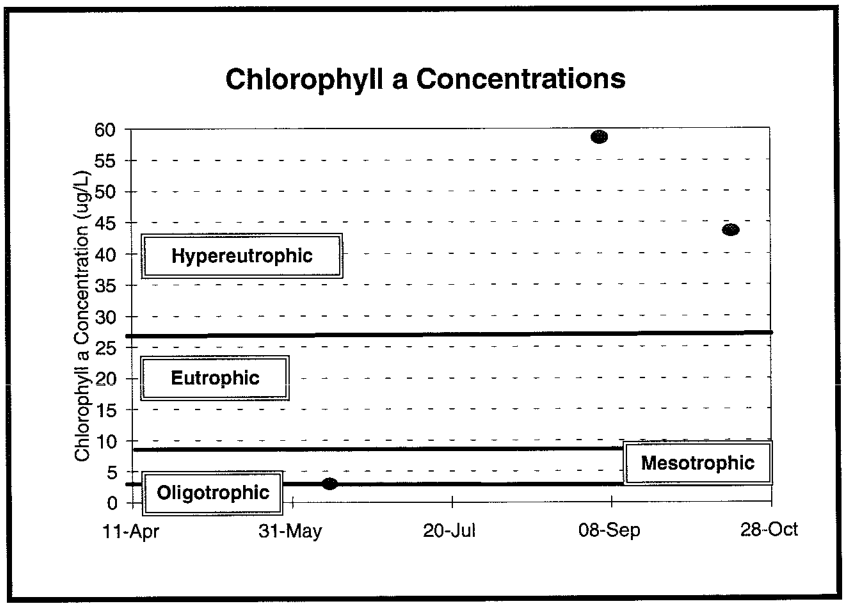


Figure 3b: Bear Trap Lake Trophic Status

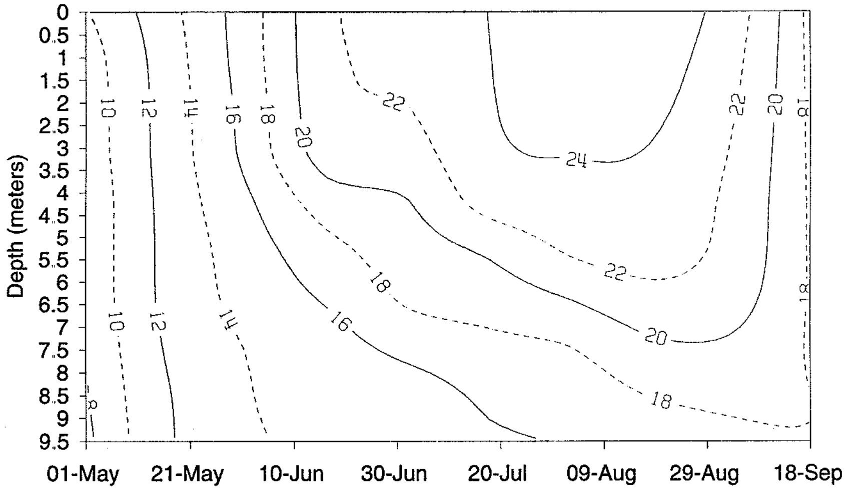




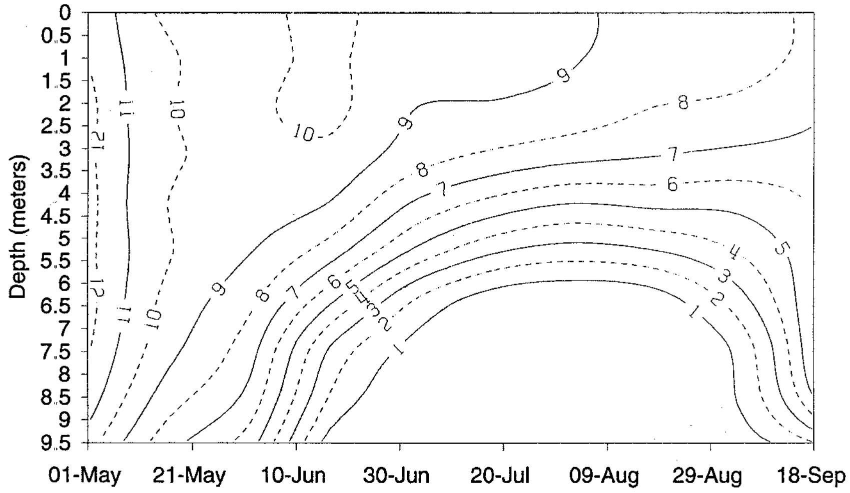


Lake Wapogasset -- 1995 Data

Time - Depth Diagram of Temperature (C) Isopleths



Time - Depth Diagram of Dissolved Oxygen (mg/L) Isopleths



Time - Depth Diagram of Total Phosphorus (ug/L) Isopleths

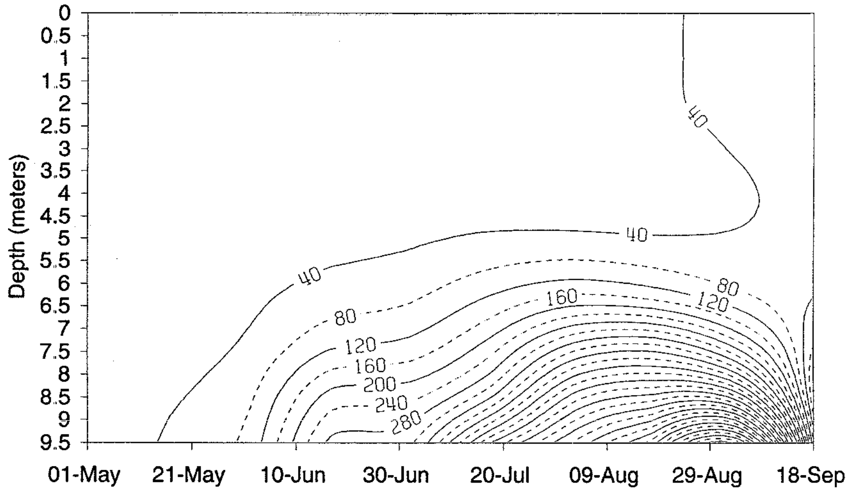
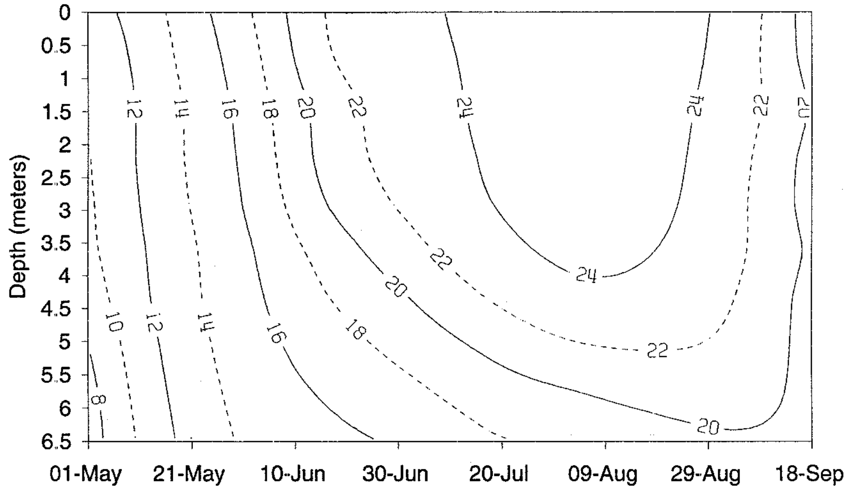


Figure 4a

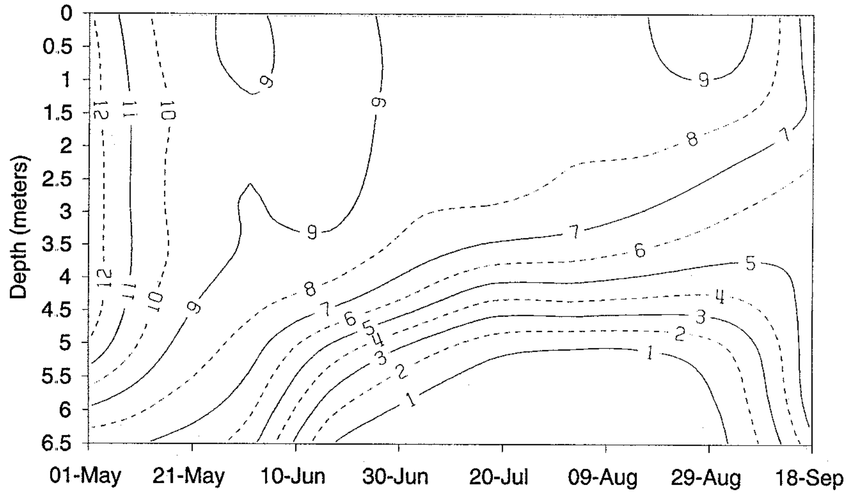
LAKE WAPOGASSET 1995
TIME-DEPTH ISOPLETH DIAGRAMS

Bear Trap Lake -- 1995 Data

Time - Depth Diagram of Temperature (C) Isopleths



Time - Depth Diagram of Dissolved Oxygen (mg/L) Isopleths



Time - Depth Diagram of Total Phosphorus (ug/L) Isopleths

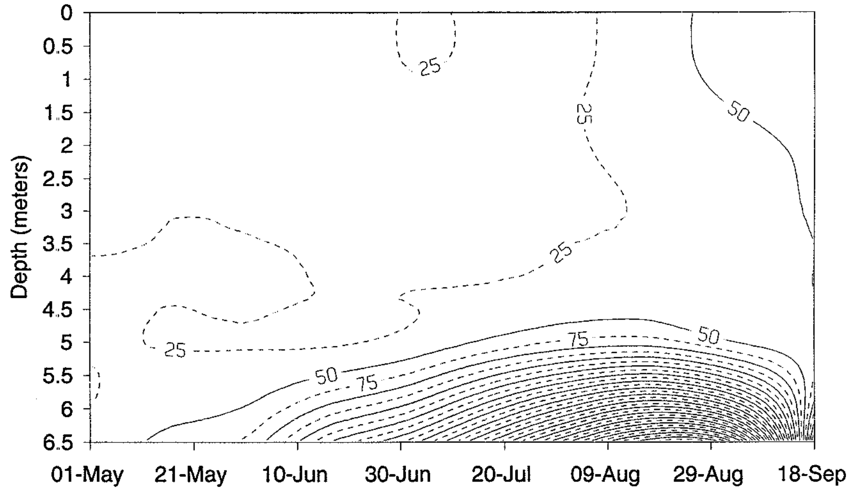


Figure 4b

BEAR TRAP LAKE 1995
TIME-DEPTH ISOPLETH DIAGRAMS

sampling date according to the first and second-derivative method of Stauffer and Lee (1973), however stratification was too weak to produce definitive results. Temperature profiles for each sampling date were then plotted, and the thermocline depth was identified visually. From examination of the thermocline depths, it appears stratification was strongest in mid-June. By early July, each lake exhibited severe thermocline erosion, which most likely resulted in entrainment of hypolimnetic waters to the epilimnion in a "pulse" load (Figures 5a and 5b). 1995 climatic data from Chetek, Wisconsin (Agricultural Weather Observation Network) and Amery (National Weather Service Station) indicates that the pulse phosphorus load observed in early July was preceded by a period of unusually cool and windy weather (Figure 6). It appears that both lakes then restratified during mid-July. Both lakes were completely mixed by the September 18 sampling date.

Dissolved oxygen concentrations correspond closely to thermal stratification patterns. The near-sediment waters became anoxic at the onset of thermal stratification. By late-July, Bear Trap Lake was anoxic (i.e., dissolved oxygen concentration less than 2 mg/L) below 5 meters, while Lake Wapogasset was anoxic below 6 meters. Near-sediment dissolved oxygen was replenished in both lakes during fall turnover.

Total phosphorus concentrations corresponded to the dissolved oxygen concentrations in both lakes. Anoxic sediment release of phosphorus obviously coincided with the onset of thermal stratification and hypolimnetic anoxia. Observed near-sediment total phosphorus concentrations reached a maximum on August 7 in Bear Trap Lake (789 µg/L) and on September 5 in Lake Wapogasset (1840 µg/L). The near-sediment concentrations decreased dramatically during fall turnover, reaching 84 µg/L in Lake Wapogasset and 85 µg/L in Bear Trap Lake on September 18.

Watershed Phosphorus Load Calculations

In order to quantify the mass of phosphorus transported from the sediments to the epilimnion of Lakes Wapogasset and Bear Trap, it was necessary to estimate the phosphorus mass delivered to the lakes from external sources. Watershed mass loads were estimated using the export coefficients listed in Table 9 and the land use delineation listed in Table 2. Watershed stormwater runoff volumes were calculated using a runoff coefficient of 0.58 (WDNR, 1996; Roessler, 1995). The mass of phosphorus delivered to each lake due to atmospheric deposition was estimated from an export coefficient of 0.56 kg/hectare/year (Tetra Tech, 1992). The mass of phosphorus and volume of discharge contributed to Lake Wapogasset from the outflows of Balsam, Deer, and Loveless Lakes was estimated by the WDNR (WDNR, 1996; Roessler, 1995). Since Lake

Figure 5a

Lake Wapogasset

Midsummer, 1995 Temperature Profiles

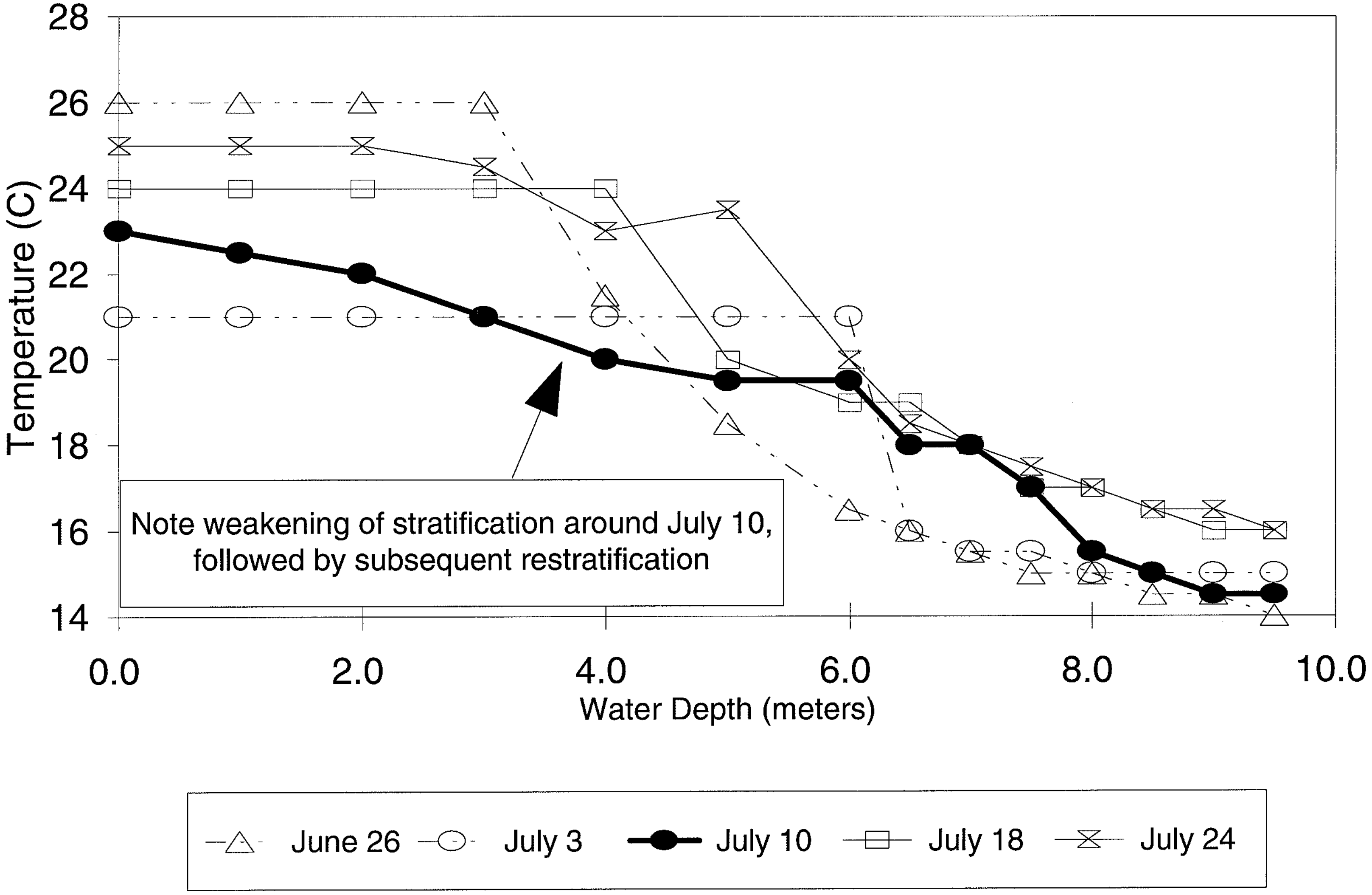
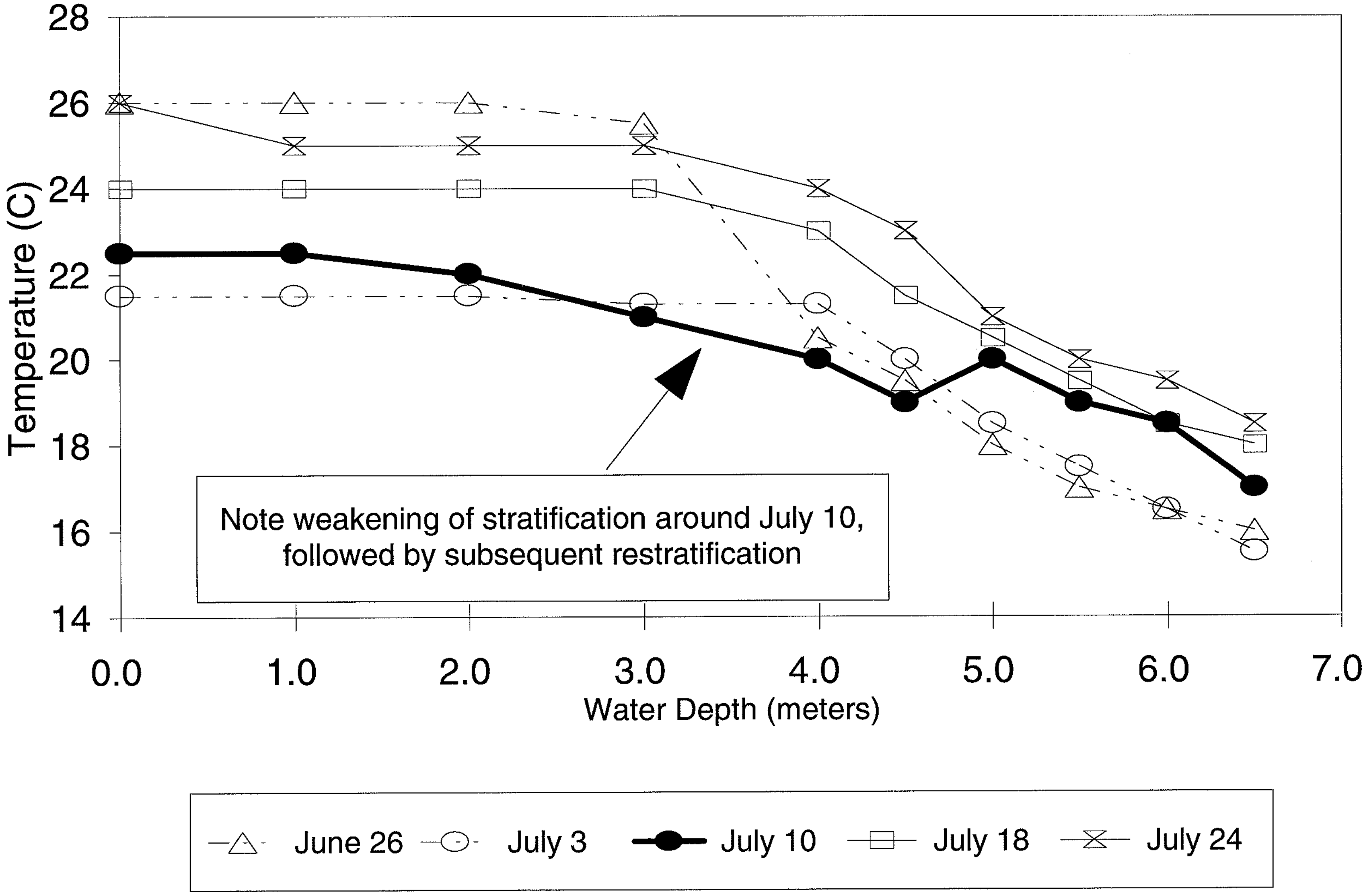

Note weakening of stratification around July 10, followed by subsequent restratification

Figure 5b

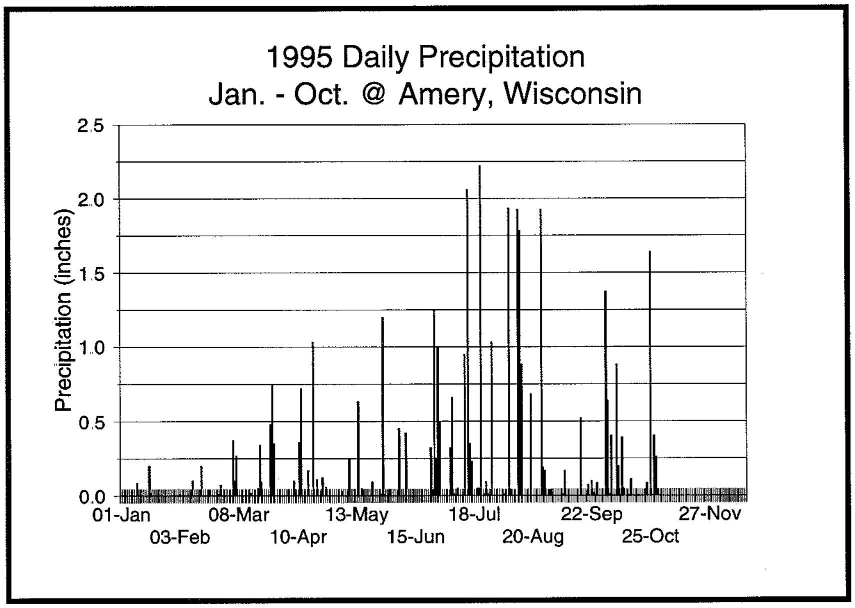
Bear Trap Lake

Midsummer, 1995 Temperature Profiles

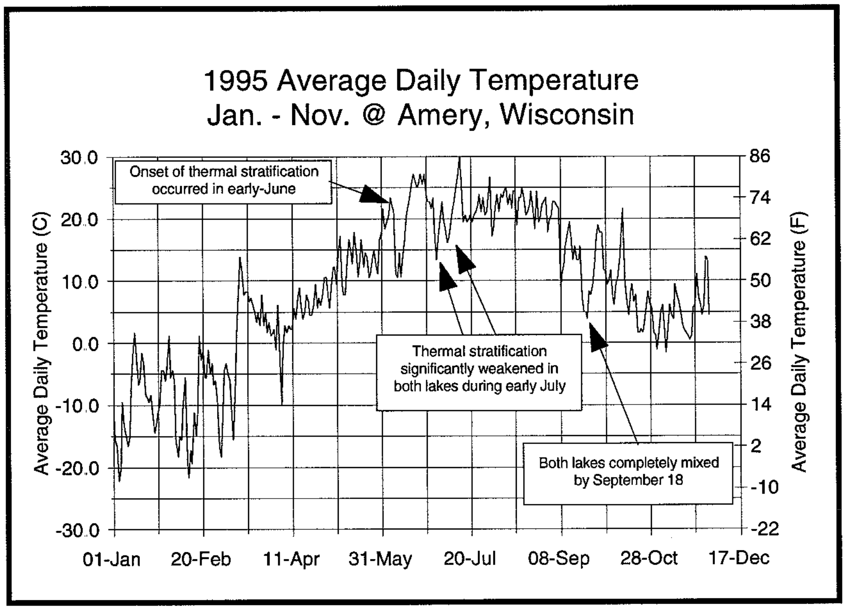


Note weakening of stratification around July 10, followed by subsequent restratification

Figure 6: 1995 Meterological Conditions Near Lake Wapogasset and Bear Trap Lake



1995 Daily Precipitation
Jan. - Oct. @ Amery, Wisconsin

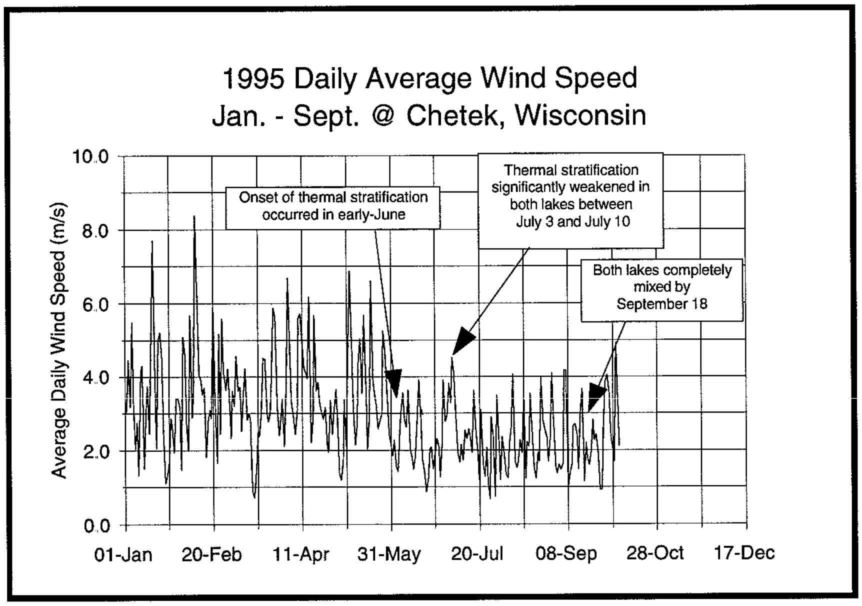


1995 Average Daily Temperature
Jan. - Nov. @ Amery, Wisconsin

Onset of thermal stratification occurred in early-June

Thermal stratification significantly weakened in both lakes during early July

Both lakes completely mixed by September 18



1995 Daily Average Wind Speed
Jan. - Sept. @ Chetek, Wisconsin

Onset of thermal stratification occurred in early-June

Thermal stratification significantly weakened in both lakes between July 3 and July 10

Both lakes completely mixed by September 18

Wapogasset and Bear Trap Lake are connected by a surface channel, it is likely that some water flow occurs between the two lakes. The calculation of the mass of phosphorus transported between the two lakes is believed to be minor and the net phosphorus mass transfer was assumed to be zero, therefore.

Table 9 Phosphorus Export Coefficients

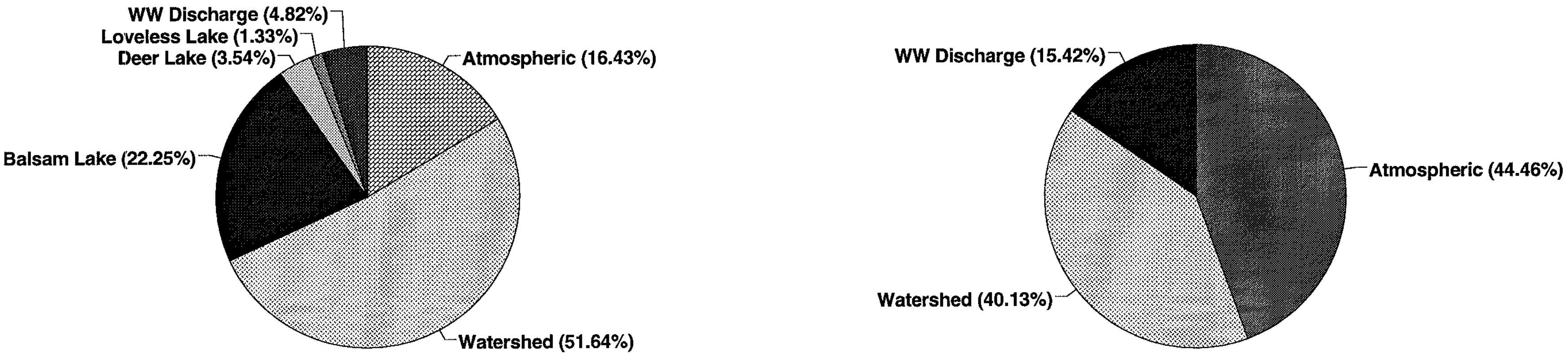
Land Use	Phosphorus Export Coefficient (kg/acre-year)	References
Agricultural	0.3	Uttormark et. al. (1974)
Forested	0.18	Reckow et. al. (1980)
Residential-Lake Shore	0.04	U.S. EPA (1983) and Pitt and McLean (1986) as compiled by Pitt and Voorhees (1995)
Residential-Rural	0.04	U.S. EPA (1983) and Pitt and McLean (1986) as compiled by Pitt and Voorhees (1995)
Residential-Urban	0.5	U.S. EPA (1983) and Pitt and McLean (1986) as compiled by Pitt and Voorhees (1995)

The calculation of the mass of phosphorus reaching Lake Wapogasset from the Village of Balsam Lake wastewater treatment ponds was complicated for several reasons: phosphorus concentrations have not been measured on either samples collected from the wastewater treatment system seepage pond or from local monitoring wells; the removal of phosphorus by soil adsorption during groundwater transport is not known; and the removal of phosphorus from the Balsam Branch during detention in the creek's many wetland areas has not been quantified. The average flow into the wastewater treatment system is 80,000 gallons/day; estimated phosphorus concentration in the seepage pond is 3 mg/L (Prusak, 1996). A soil retention factor (SRF) of 0.75 (Reckhow et. al., 1980) was used to estimate removal of phosphorus from groundwater flow via soil adsorption, (i.e., 75 percent of the phosphorus mass present in groundwater flow was removed by adsorption onto soil particles). Therefore, the annual phosphorus mass entering Lake Wapogasset from the Village of Balsam Lake wastewater ponds was estimated to be 182 lbs (82.8 kg); the annual discharge volume was calculated to be 89.4 acre-feet.

The calculation of the mass phosphorus reaching Bear Trap Lake (or Lake Wapogasset) from the Wapogasset/Bear Trap sanitary system was also complicated for several reasons: the flow of groundwater in the vicinity of the treatment system is not understood; phosphorus concentrations have not been measured on either samples collected from the wastewater treatment system seepage pond or from local monitoring wells; the removal of phosphorus by soil adsorption during groundwater transport is not known. Four monitoring wells are located in the vicinity of the wastewater treatment system. Each well is monitored quarterly for dissolved nitrate/nitrite, chloride, total dissolved solids, dissolved ammonium, dissolved organic nitrogen, alkalinity, and total hardness. Groundwater elevations (to 0.01 feet) are collected each quarter, as well (Prusak, 1996). Due to possible groundwater contamination by nitrates from an adjoining farm, using the chemical data to track groundwater flow provides inconclusive results. In addition, the elevations of the monitoring well casings have not been verified by the WDNR, and may be questionable; the groundwater elevations collected from the wells provide inconclusive results of groundwater flow direction. During the course of this study, a Barr Engineering Company groundwater hydrogeologist examined both the WDNR monitoring well data and the U.S.G.S. 7.5 minute topographic quadrangle map of the area ("Wapogasset Lake, Wisconsin"). He concluded that localized groundwater flow caused by seepage from the treatment ponds may skew the groundwater levels measured in the monitoring wells. Also, the surface topography in the vicinity of the treatment ponds definitely slopes towards Bear Trap Lake. Since shallow groundwater flow almost always mimics surface water flow, he concluded that groundwater flow most likely proceeds from the ponds towards Bear Trap Lake. Therefore, for the purposes of this report it was assumed that the seepage from the Wapogasset/Bear Trap sanitary system treatment ponds does flow towards Bear Trap Lake. During 1995, the inflow to the ponds ranged from 14,000 gallons/day to 32,000 gallons/day; the daily average was 20,000 gallons/day (Prusak, 1996). The estimated phosphorus concentration in the seepage cell discharge is 3 mg/L (Prusak, 1996). A soil retention factor (SRF) of 0.75 (Reckhow et. al., 1980) was used to estimate removal of phosphorus from groundwater flow via soil adsorption, (i.e. 75 percent of the phosphorus mass present in groundwater flow was removed by adsorption onto soil particles). Therefore, the annual phosphorus mass entering Bear Trap Lake from the wastewater ponds was estimated to be 45.5 lbs (20.7 kg); the annual discharge volume was calculated to be 22 acre-feet.

The overall external phosphorus and water budgets for Lakes Wapogasset and Bear Trap are summarized in Table 10, and illustrated on Figure 7.

Figure 7: Lake Wapogasset and Bear Trap Lake Annual External Phosphorus and Water Budgets


Lake Wapogasset
Annual External Phosphorus Budget

Bear Trap Lake
Annual External Phosphorus Budget

Annual Water Budget

Annual Water Budget

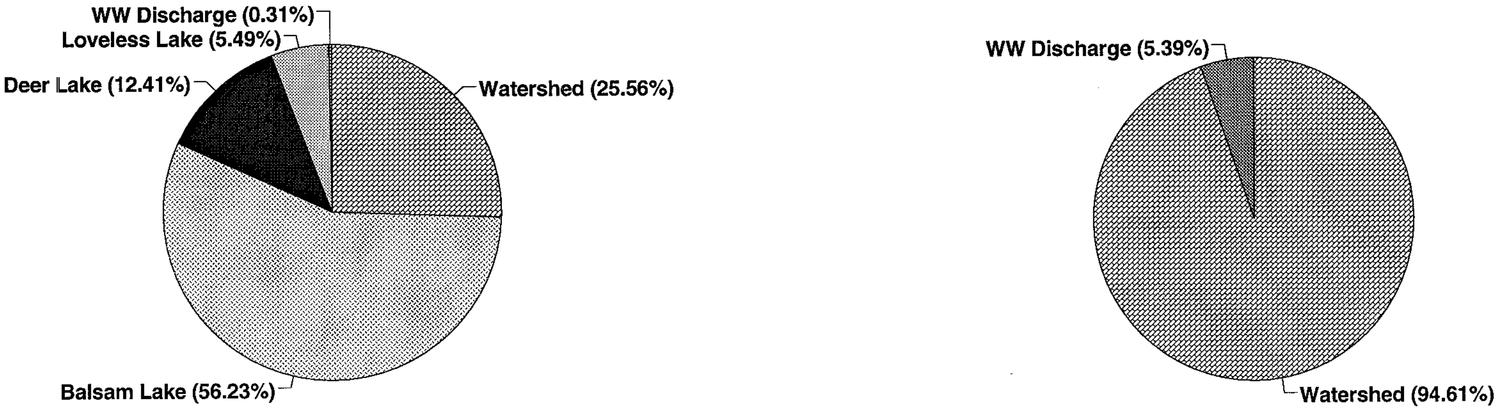


Table 10 Lake Wapogasset and Bear Trap Lake Annual External Phosphorus and Water Budgets

	Lake Wapogasset			Bear Trap Lake		
	Inflow Volume (acre-feet)	Phosphorus Mass lbs (kg)	Phosphorus Concentration (µg/L)	Inflow Volume (acre-feet)	Phosphorus Mass lb (kg)	Phosphorus Concentration (µg/L)
Watershed	6,978	1,953 (888)	103	386	119 (54)	113
Atmospheric	–	620 (282)	–	–	132 (60)	–
Balsam Lake	15,350	841 (382)	20	–	–	–
Deer Lake	3,389	134 (61)	15	–	–	–
Loveless Lake	1,499	50 (23)	12	–	–	–
Village of Balsam Lake Wastewater Treatment System	84	182 (83)	751	–	–	–
Wapogasset/Bear Trap Sanitary System	–	–	–	22	46 (21)	763
Total	27,300	3,780 (1,718)	–	408	297 (135)	–

Internal Phosphorus Load Calculations

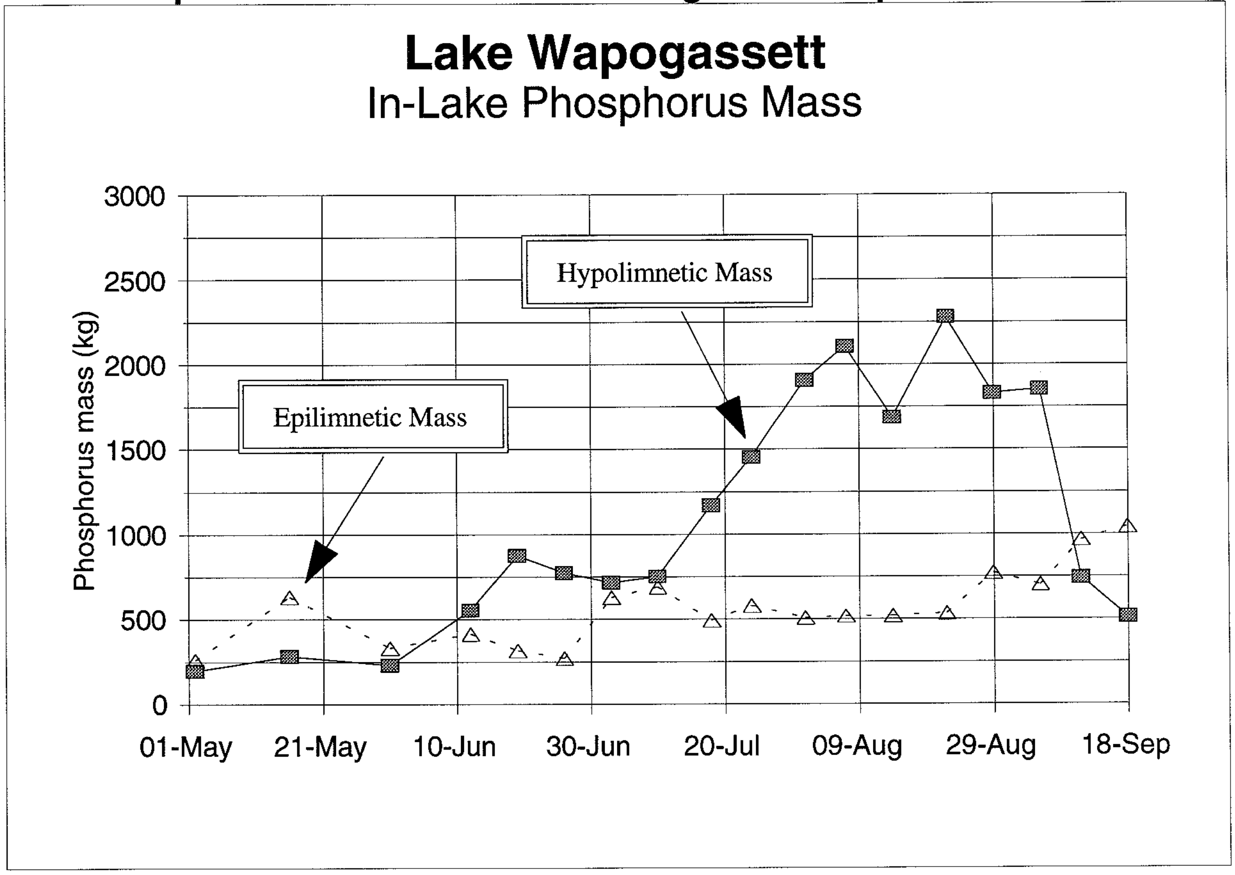
Nurnberg (1987, 1984, and 1985) suggests several methods for the calculation of phosphorus mass released from the sediments and accumulated in the hypolimnion during anoxia. An areal phosphorus release rate (expressed in units of mg P/m^2-day) can be determined from laboratory incubations of sediment cores. The mass of phosphorus released from the sediments to the hypolimnion is then calculated from this rate and the areal extent and duration of anoxia. Similarly, the areal phosphorus release rate can be calculated from in-situ hypolimnetic phosphorus concentrations, the sediment area of the hypolimnion, and the duration of anoxia. Thirdly, when sufficient phosphorus concentration data has been collected at various depths throughout the water column, the mass of phosphorus present in the hypolimnion and epilimnion can be calculated from measured phosphorus concentration at depth and the associated water volume. Since samples were collected frequently at various depths in both Lake Wapogasset and Bear Trap during May through September, 1995, the third method appeared most appropriate. The water area at various depths was determined by planimetry of Wisconsin Department of Conservation bathymetric survey maps of both lakes. The water volume associated with each depth interval was calculated from the mean water area within the depth interval multiplied by the depth of the interval (Table 11).

Table 11 Lake Wapogasset and Bear Trap Lake Hypsographic Data

Depth (feet)	Lake Wapogasset		Bear Trap Lake	
	Area (acres)	Cumulative Volume (acre-feet)	Area (acres)	Cumulative Volume (acre-feet)
0	1,246.1	20,838	263.4	4,340
3	–	–	246.9	3,574
5	1,063.2	15,064	226.8	3,101
10	915.2	10,118	205.6	2,020
15	715.9	6,041	184.8	1,044
20	528.8	2,929	116.3	291
25	290.4	881	0.2	–
30	61.9	–	–	–

Volume segments were defined in each lake such that each sampling depth was located in the center of a segment. For example, samples were collected from 1, 3, 5, 6.5, 7.5, 8.5, and 9.5 meters. Therefore, the volume segments used in calculating the phosphorus mass present in the lake were 0-2 meters (encompassing the 1-meter sample), 2-4 meters (encompassing the 3-meter sample, 4-5.75 meters (encompassing the 5-meter sample), etc. Once the phosphorus mass present in each volume segment on each sampling date was calculated, an average epilimnetic depth of 4 meters was identified for each lake. This depth is rather arbitrary, and used primarily to identify phosphorus mass fluctuations between the near-surface and near-bottom waters. Using this depth, an epilimnetic and hypolimnetic phosphorus mass for each sampling date was readily calculated; the epilimnetic and hypolimnetic phosphorus masses were also divided by the associated water volume to yield the volume-weighted phosphorus concentration for the two layers. Both the calculated epilimnetic and hypolimnetic phosphorus masses and volume-weighted concentrations for Lake Wapogasset and Bear Trap Lake are plotted on Figures 8a and 8b. Each plot exhibits the characteristics observed in lakes with anoxic phosphorus release. The hypolimnetic phosphorus mass in both lakes began to increase mid-June, coincident with stratification and hypolimnetic anoxia, and to decrease mid-August as thermal stratification began to breakdown. The epilimnetic mass fluctuated slightly throughout May and early June, but increased dramatically coincident with the decrease in hypolimnetic phosphorus mass. The volume-weighted phosphorus concentrations in the epilimnion and the hypolimnion were similar

Figure 8a: Lake Wapogasset
1995 Phosphorus Mass and Volume-Weighted Phosphorus Concentration



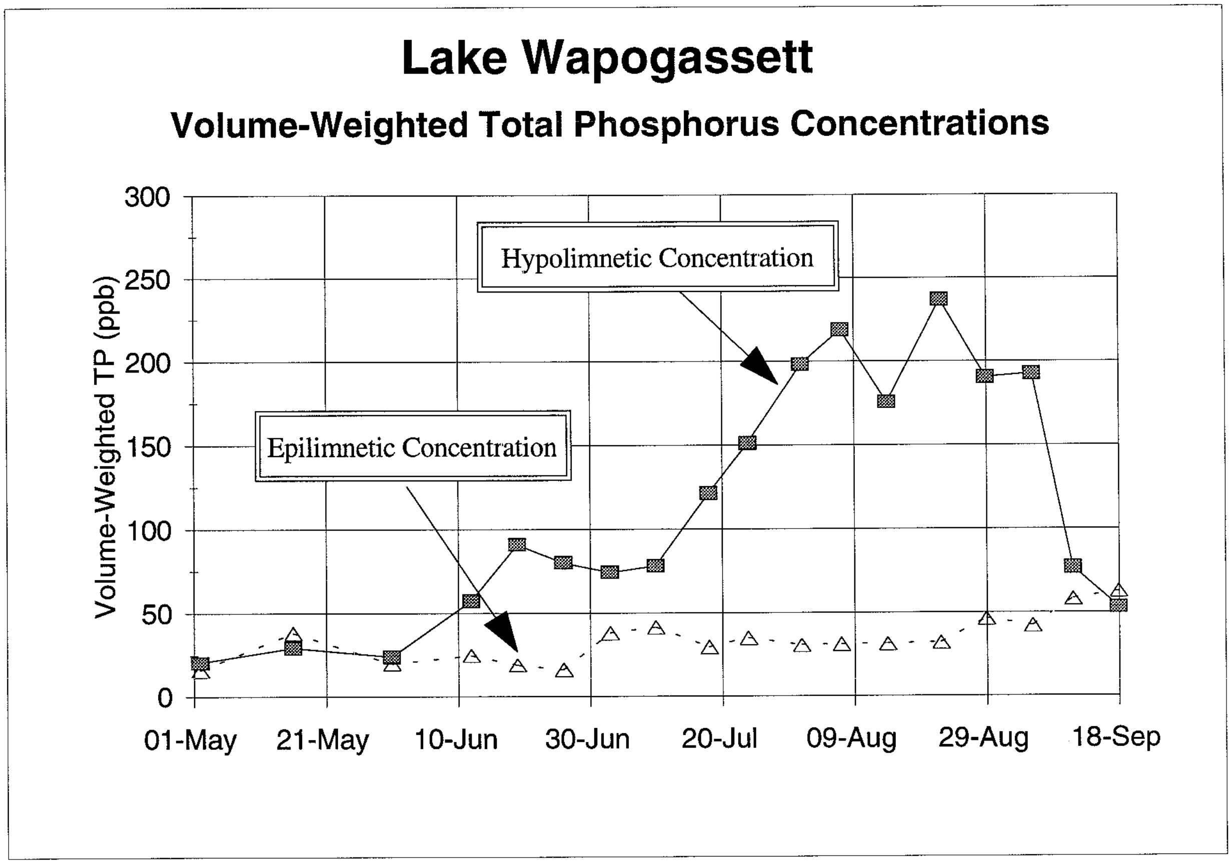
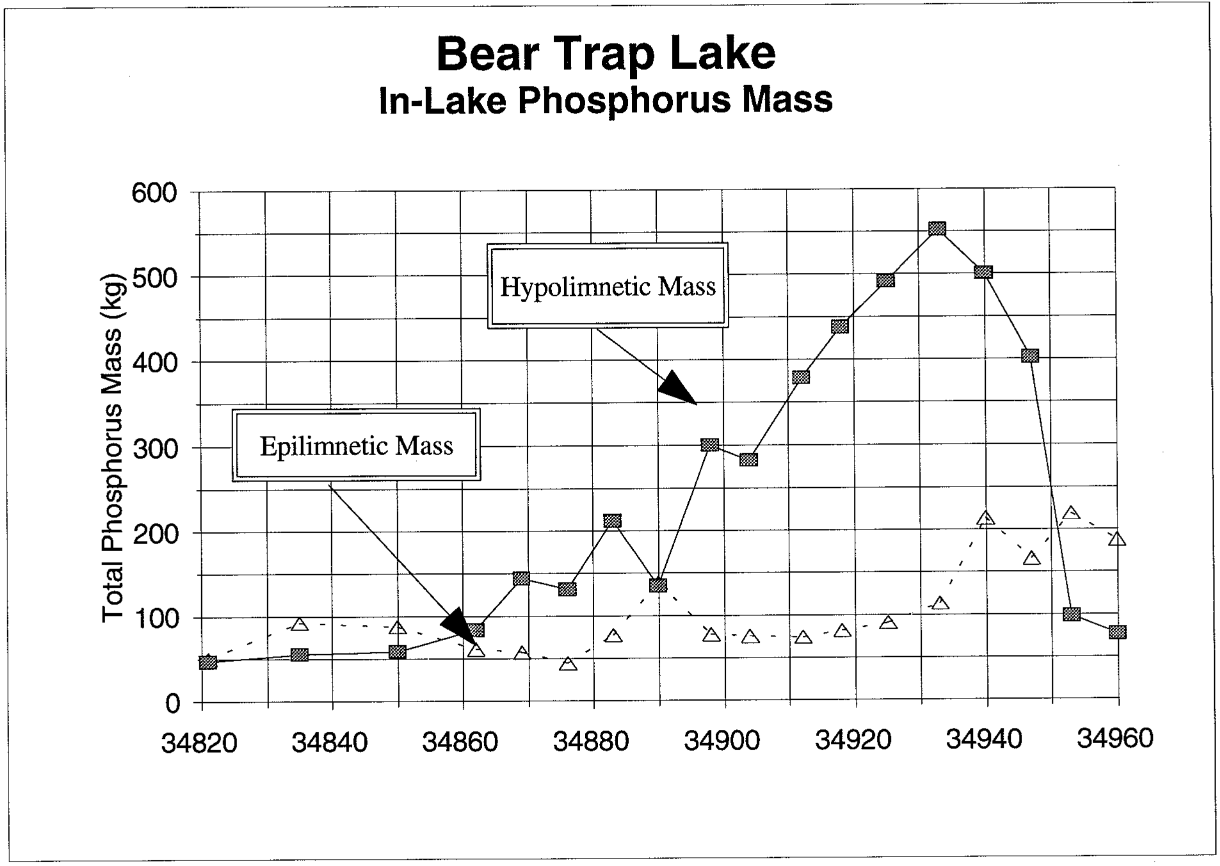
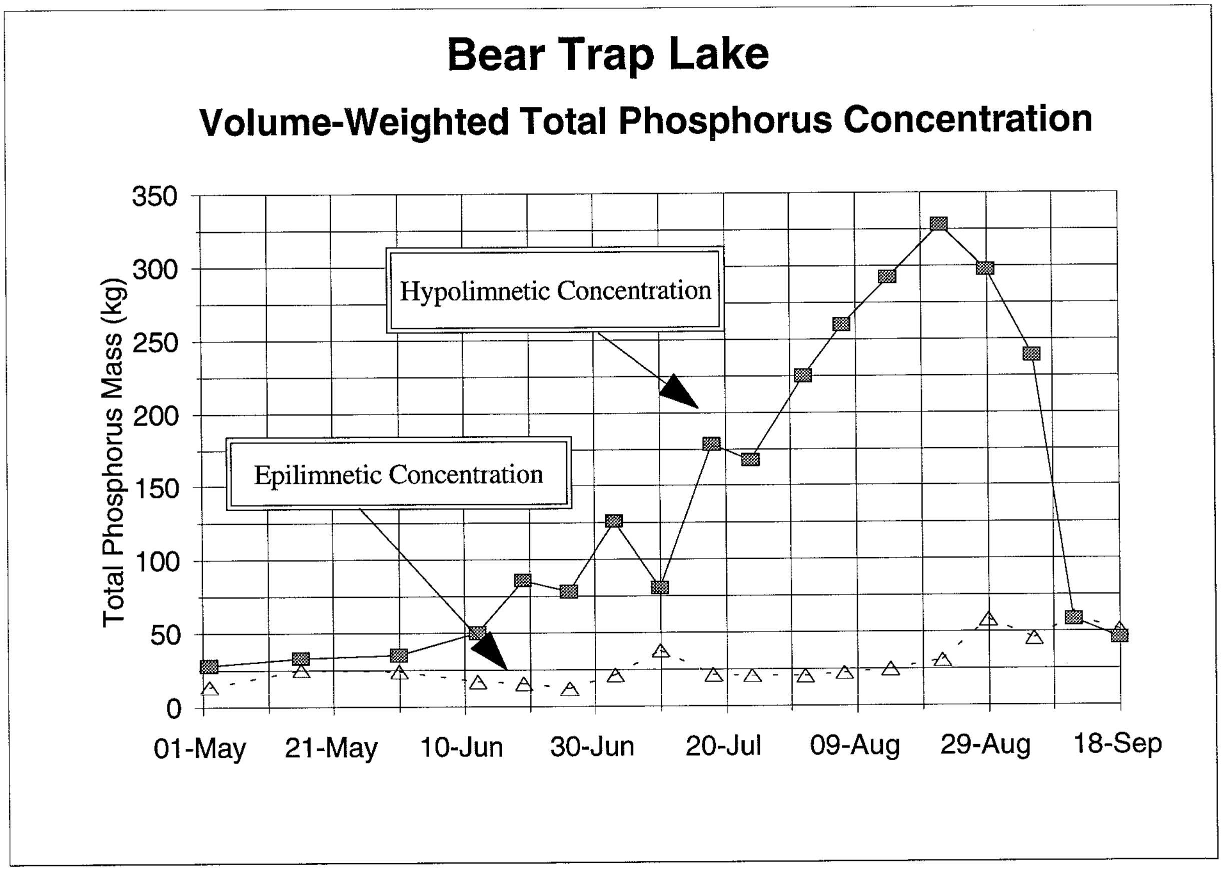


Figure 8b: Bear Trap Lake
1995 Phosphorus Mass and Volume-Weighted Phosphorus Concentration





prior to the onset of thermal stratification. Once thermal stratification commenced, the hypolimnetic phosphorus concentration increased dramatically, while the epilimnetic concentration remained fairly stable. At the onset of stratification breakdown, the hypolimnetic concentration declined drastically, while the epilimnetic concentration increased (presumably due to entrainment of hypolimnetic phosphorus). By fall turnover, the epilimnetic and hypolimnetic volume-weighted phosphorus concentrations were equivalent.

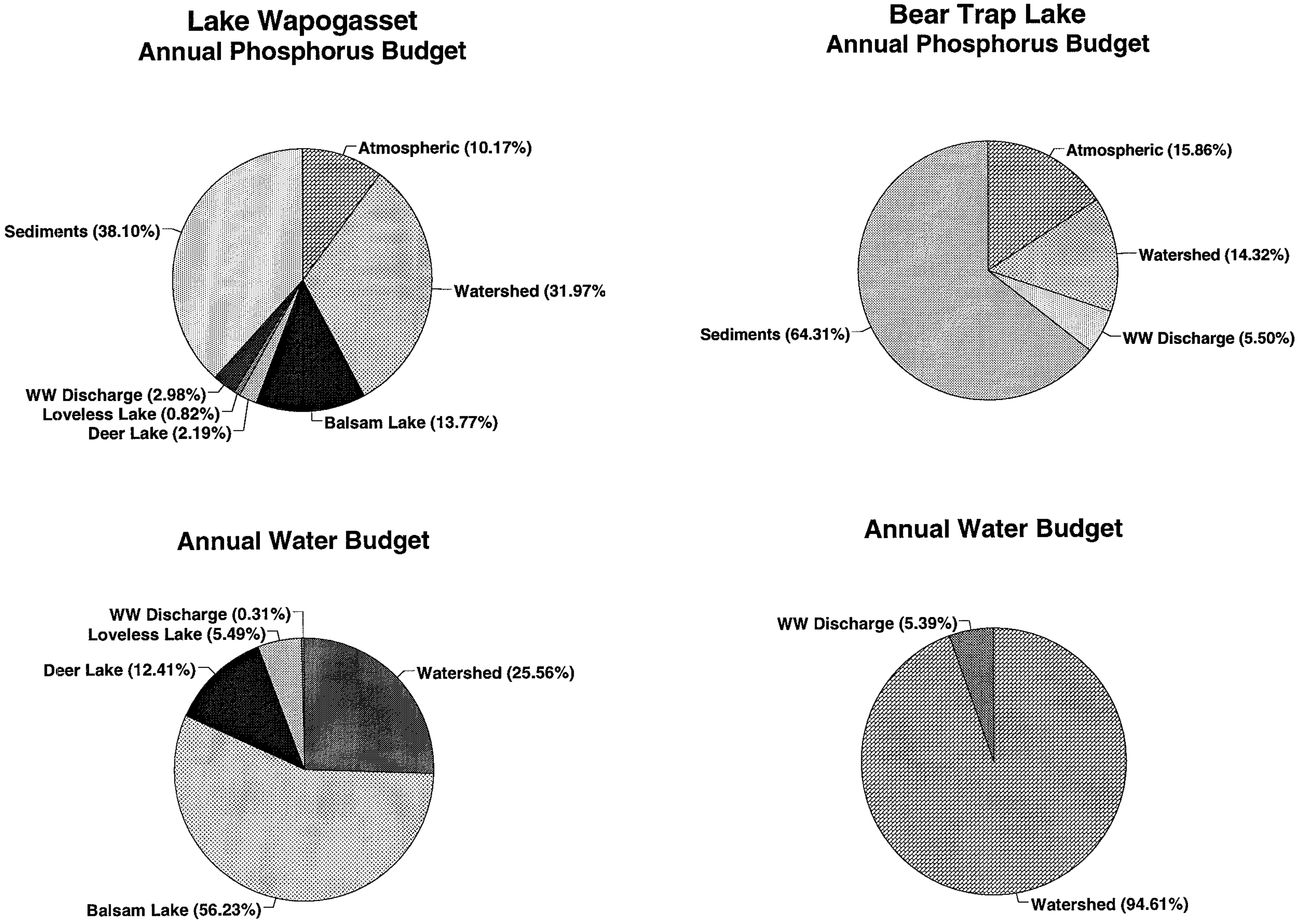
The total mass of phosphorus released from the anoxic sediments to the hypolimnion of each lake was calculated from the difference of the mass of phosphorus present in the hypolimnion just prior to stratification and the maximum mass of phosphorus present just prior to the late-summer breakdown of thermal stratification. For Bear Trap Lake, the total mass of phosphorus released from the sediments to the hypolimnia was calculated to be 505 kg (i.e. 552 kg maximum hypolimnetic phosphorus mass - 47 kg hypolimnetic phosphorus mass at onset of stratification = 505 kg); for Lake Wapogasset, the total mass was calculated to be 2,083 kg (i.e. 2,278 kg - 195 kg = 2,083 kg).

Only a portion of the phosphorus mass released from the sediments to the hypolimnion and entrained during mid-summer wind mixing and fall turnover will actually reach the epilimnion due to adsorption of some phosphorus to ferric hydroxide ($Fe(OH)_3$) precipitate. The total mass of phosphorus entering the epilimnion during the summer months was calculated from the difference of the epilimnetic phosphorus mass just prior to the onset of thermal stratification breakdown (June 26, 1995) and the epilimnetic mass at fall turnover (September 18, 1995). The total mass of phosphorus entering the epilimnion of Lake Wapogasset during July through September, 1995 was 772 kg [i.e. 1,045 kg (epilmnetic phosphorus mass on September 18, 1995) - 272 kg (mass on June 26, 1995) = 772 kg]. Similarly, the mass entering the epilimnion of Bear Trap Lake during that same period was 174 kg (i.e. 219 kg - 45 kg = 174 kg). However, this increase in epilimnetic phosphorus mass is the net balance of watershed phosphorus inflows, lake phosphorus outflows, and hypolimnetic phosphorus entrainment. Therefore, the mass of phosphorus entrained from the hypolimnion to the epilimnion during June through September was calculated from the total increase in epilimnetic phosphorus mass minus watershed inflows plus phosphorus discharged from the lake outlet plus any pulse loads of phosphorus entrained to the epilimnion during mid-summer windy periods. Based on precipitation data collected at Amery during 1995, it was assumed that 61 percent of the annual watershed water and phosphorus load occurred during June through September. The mass of phosphorus discharged through the lake outlet during June through September was calculated from the volume of runoff entering the lake and the average epilimnetic phosphorus concentration during that period. The mass of phosphorus entering the epilimnion during pulse-loads associated with mid-summer windy periods was calculated from the mass prior to the windy period minus the peak phosphorus mass observed immediately thereafter; such a pulse was observed in both lakes around July 10, 1995. Therefore, the actual mass of phosphorus entrained to the epilimnion from the hypolimnion during June through September in Lake Wapogasset was 1,058 kg [772 kg (from above) - 735 kg (watershed inflows) + 516 kg (lake outlet discharge) + 505 kg (pulse phosphorus load) = 1,058 kg]. Similarly for Bear Trap Lake the actual mass entrained from the hypolimnion was 242 kg [174 kg - 36 kg + 10 kg + 94 kg = 242 kg]. Therefore, the fraction of the phosphorus released from the sediments that was entrained to the epilimnion was 51 percent (1,058 kg / 2,083 kg) in Lake Wapogasset and 48 percent (242 kg / 505 kg) in Bear Trap Lake. These estimates seem reasonable in comparison to the range of 35 percent to 60 percent reported by Nurnberg (1984). The overall phosphorus and water budgets for Lake Wapogasset and Bear Trap Lake are illustrated on Figure 9.

In-Lake Water Quality Modeling

The in-lake water quality at spring and fall turnover in Lake Wapogasset and Bear Trap Lake was predicted using the empirical relationship between external phosphorus load, annual hydraulic flushing rate, lake morphometry, and phosphorus retention developed by Dillon and Rigler (1974) and modified by Nurnberg (1984). Prediction of the in-lake water quality is portentous for several reasons: accurate prediction of the spring overturn concentration provides assurance the hydrologic and phosphorus budgets are reasonable, accurate prediction of the fall overturn concentration provides assurance that the internal phosphorus load mass is reasonable, and evaluation of watershed and sediment phosphorus load reductions using the calibrated equations provides qualitative assessment of the resulting in-lake water quality benefits. The observed average spring overturn concentration in each lake was used to assess the plausibility of the model; using historical data collected by the Lake Improvement Association and the WDNR, spring overturn concentrations of 20 µg/L and 13 µg/L for Lake Wapogasset and Bear Trap Lake, respectively, seem reasonable. The parameters used in the in-lake modeling are summarized in Table 12. The phosphorus retention in each lake was calculated using the equations developed by Nurnberg (1984), Chapra (1975), Kirchner and Dillon (1975), and Larsen and Mercier (1976). Each retention coefficient was then inserted in Dillon and Rigler's equation, and the resulting spring turnover concentration was compared to the average observed concentration; the phosphorus retention coefficient which generated the calculated concentration closest to the observed concentration was selected as most representative of the lake system. The phosphorus retention coefficients predicted for Lake Wapogasset by each of the equations were similar, however the coefficient predicted by Nurnberg's equation resulted in a spring overturn concentration of 20 µg/L (exactly the average

Figure 9: Lake Wapogasset and Bear Trap Lake Annual Phosphorus and Water Budgets



observed spring overturn concentration). The phosphorus retention coefficients predicted for Bear Trap Lake by each of the equations were quite disparate, which is mostly likely due to the extremely small overflow rate for Bear Trap Lake (0.47 m/year). The retention coefficient predicted by Larson and Mercier's equation resulted in a spring overturn concentration of 13 µg/L (exactly the average observed spring overturn concentration).

The epilimnetic phosphorus concentration at fall turnover was estimated using the spring turnover concentration predicted above, along with the increases in epilimnetic phosphorus concentration from June through September estimated for watershed inflows and internal load. For Lake Wapogasset, the spring overturn concentration was calculated to be 20 µg/L, as discussed above. The watershed phosphorus load during June through September (219 kg, discussed previously) is equivalent to a volume-weighted phosphorus concentration of 13 µg/L. The internal phosphorus load entrained to the epilimnion was calculated to be 553 kg (discussed previously), equivalent to a volume-weighted phosphorus concentration of 34 µg/L. The phosphorus mass added from mid-summer pulse loads was not included, since the resulting elevated epilimnetic phosphorus concentration was temporary, and soon decreased to approximately the concentration observed before the pulse load.

Therefore, the volume-weighted epilimnetic phosphorus concentration at fall turnover was estimated to be 67 µg/L (20 µg/L + 13 µg/L + 34 µg/L). This corresponds closely to the volume-weighted epilimnetic concentration observed on September 18, 1995 of 63 µg/L. Similarly for Bear Trap Lake, the spring overturn concentration was calculated to be 13 µg/L, the watershed inflow between June and September was estimated to be 26 kg, equivalent to a volume-weighted epilimnetic concentration of 7 µg/L, and the internal load mass of 148 kg (not including pulse-loads) was equivalent to a concentration of 41 µg/L. Therefore, the volume-weighted epilimnetic phosphorus concentration at fall turnover was calculated to be 61 µg/L (13 µg/L + 7 µg/L + 41 µg/L), which corresponds closely to the volume-weighted epilimnetic concentration observed on September 18, of 60 µg/L.

Table 12 Lake Wapogasset and Bear Trap Lake Modeling Parameters

Modeling Parameter	Lake Wapogasset	Bear Trap Lake
External Areal Phosphorus Load: L_{ext} (mg/m^2-year)	340.9	125.9
External Water Load, Q_s: (m3/year)	33,669,125	503,187
Water Surface Area, S: (m2)	5,042,970	1,065,980
Lake Water Volume, V: (m3)	25,700,249	5,354,527
Mean Depth, z:[1] (m)	5.1	5.0
Hydraulic Detention Time, τ:[2] (years)	0.763	10.6
Hydraulic Flushing Rate, ρ:[3] (1/years)	1.31	0.094
Overflow Rate, q_s:[4] (m/year)	6.68	0.472
Retention Coefficient, R_p:	0.61 (61%)[5]	0.77 (77%)[6]
Modeled Spring Overturn TP Concentrations (µg/L)	20	13
Observed Spring Overturn TP Concentrations (µg/L)[7]	20	13
Modeled Fall Overturn TP Concentrations (µg/L)	67	61
Observed Full Overturn TP Concentrations (µg/L)[8]	63	60

[1] $z = V/S$

[2] $\tau = V/Q_s$

[3] $\rho = 1/t$

[4] $q_s = Q_s/S$

[5] Calculated from the equation developed by Nurnberg (1984): $R_p = 15/(18 + q_s)$

[6] Calculated from the equation developed by Larson and Mercier (1976): $R_p = 1/(1+\rho^{1/2})$

[7] Averaged from April Concentrations observed during past studies.

[8] Observed during this study (1995)

Estimated Water Quality Benefits from Alum Treatment

The reduction in the fall overturn epilimnetic phosphorus concentration resulting from application of alum to the lake sediments to control internal phosphorus load was predicted for Lake Wapogasset and Bear Trap Lake. It was assumed that an alum treatment would reduce the internal phosphorus load in each lake by 90 percent. This results in an internal load reduction in Lake Wapogasset of 498 kg and an internal load transported to the epilimnion during fall turnover of 55 kg (equivalent to a volume-weighted epilimnetic concentration of 3.3 µg/L). Therefore, the epilimnetic concentration predicted at fall turnover after application of alum is 37 µg/L (13 µg/L + 20 µg/L + 3 µg/L, as in the previous section). Similarly, the internal load reduction in Bear Trap Lake from an alum treatment was estimated as 133 kg, and the actual internal load transported to the epilimnion during fall turnover as 15 kg (equivalent to a volume-weighted epilimnetic concentration of 4 µg/L). Therefore the epilimnetic concentration predicted at fall turnover after application of alum is 24 µg/L (13 µg/L + 7 µg/L + 4 µg/L, as in the previous section).

The overall percent reduction in epilimnetic phosphorus concentration due to application of alum to the sediments is estimated as 45 percent in Lake Wapogasset, and 60 percent in Bear Trap Lake.

Preliminary Alum Dose Rate Determinations and Cost Estimates

Preliminary alum dose rate calculations were completed using two methods to provide a range of treatment doses and costs: the alkalinity-based dosing method described by Cooke et. al. (1993) and the hypolimnetic phosphorus mass method outlined by Eberhardt (1995). Cooke's method calculates the maximum dose of alum that can be added to a lake without lowering the pH below 6.0; at pH values below 6.0, the fish, plant, and animal communities within the lake may be harmed. Eberhardt's method calculates alum dose based on the mass of phosphorus released from the sediments, and should result in a lower alum dose than Cooke's method. While the deep, anoxic sediments release the bulk of the internal phosphorus load, shallow littoral zone (near-shore) sediments may also release some phosphorus to the water column during brief periods of anoxia within the sediment interstitial waters. However it is extremely difficult to quantify this phosphorus mass. Some alum applicators recommend treating the entire sediment area in order to reduce both deep anoxic and littoral zone sediment phosphorus release. Alum applied to the littoral sediments is susceptible to disturbance by wind-mixing, boat motor and jet-ski turbulence, and ice movement. Therefore, the cost estimates were calculated based on a proposed treatment depth of 15 feet and greater in order to cover the deep, anoxic sediments which are less susceptible to disturbance (Figure 10).

Since formation of the alum floc tends to lower pH, the amount of alum that can be added to a lake without causing detrimental effects to the ecosystem is dependent on the alkalinity of the lake water, which may vary slightly with season and depth. Cooke et. al. (1993) have developed a nomograph to provide an estimate of the maximum alum dose that can be added to a lake based on initial alkalinity and pH. The average alkalinity and pH measured in Lake Wapogasset and Bear Trap Lake during the 1991 WDNR Lake Planning Grant study were 96 mg/L as $CaCO_3$ and pH 8.8 (Lake Wapogasset) and 105 mg/L as $CaCO_3$ and pH 8.7 (Bear Trap Lake). Using Cooke's alum dose nomograph, it appears that an alum dose of 12 mg Al/L is appropriate for both lakes. This dose rate was applied to the volume of each lake to calculate the total mass of alum required; the calculations are summarized in Table 13.

Table 13 Alum Dose Rate Calculations Based on Cooke et. al. (1994) for Application to Lake Wapogasset and Bear Trap Lake at Depths Greater than 15 feet

	Lake Wapogasset	Bear Trap Lake
Alum dose rate according to nomograph developed by Cooke et. al. (1994)	12 mg Al/L	12 mg Al/L
Total mass of dry alum to be dosed[1]	985,000 kg 2,167,010 lb	170,230 kg 374,500 lb
Total volume of alum solution to be dosed[2]	402,800 gallons	69,600 gallons
Areal Dose Rate[3]	3,030 lb dry alum per acre 563 gallons alum solution per acre	2,030 lb dry alum per acre 377 gallons alum solution per acre
Total Cost[4]	$402,800.00	$69,600.00

[1] Mass of dry alum to be dosed calculated using a mass ratio of 11.0156 mg Alum/mg Al^{+3}(assuming a molecular weight of 594.4 g/mole for commercial alum, $Al_2(SO_4)_3{*}14H_2O$) and water volumes at depth greater than 15 feet of 6,041 acre-ft in Lake Wapogasset and 1,044 acre-ft in Bear Trap Lake.

[2] Volume of alum solution was calculated from an Al^{+3} content of 4.4% and a solution density of 11.1 lb/gallon.

[3] Areal dose rate calculated using that portion of the lake surface area which overlies depths of 15 feet or greater (Lake Wapogasset = 715.9 acres; Bear Trap Lake = 184.8 acres)

[4] Cost for alum and application as estimated by Tom Eberhardt of Sweetwater Technology is $1.00/gallon (1996 dollars). This price includes mobilization/demobilization, and does not include the cost of any special dosing titrations or contract/bid document preparation that may be required by the state.

Rather than calculating alum dose rate based on alkalinity, Eberhardt (1995) calculates alum dose based on the annual mass of phosphorus released from the sediments. A copy of his alum dose rate calculation procedure is included in Appendix C. Table 14 summarizes the parameters used in Eberhardt's calculations, as well as the results of the alum dose calculations for Lake Wapogasset and Bear Trap Lake.

Table 14 Alum Dose Rate Calculations Based on Eberhardt (1995) for Application to Lake Wapogasset and Bear Trap Lake at Depths Greater than 15 feet

	Lake Wapogasset	Bear Trap Lake
Depth at which alum treatment begins, H:	15 feet	15 feet
Total area of the lake deeper than H, A_h:	716 acres	184 acres
Total area of the lake deeper than H with nutrient bearing sediments, A_s:	716 acres	184 acres
Total mass of phosphorus recycled from the sediments each year, P_i:	4,583 lbs	1,111 lbs
Total mass of phosphorus recycled from sediments deeper than H, P_h:	3,437 lbs	833 lbs
Areal phosphorus release rate, P_s:[1]	4.8 lbs P/acre-year	4.5 lbs P/acre-year
Expected duration of alum treatment effectiveness, T:	10 years	10 years
Distribution efficiency of the application equipment, K_d:	0.9 (90% efficiency)	0.9 (90% efficiency)
Ratio of maximum sediment phosphorus release rate to average sediment release rate, K_r:	4	4
Ratio of molecular weight of aluminum to the molecular weight of phosphorus, K_a:	0.89	0.89
Areal Al^{+3} dose to be applied to sediments deeper than H, D_a:[2]	190 lbs Al^{+3}/acre	179 lbs Al^{+3}/acre
Total Al^{+3} mass required to treat sediments deeper than H, D_t:	135,952 lbs	32,950 lbs
Total alum mass required to treat sediments deeper than H, A_t:[2]	3,089,828 lbs	748,859 lbs
Total alum solution volume required to treat sediments deeper than H, A_v:[3]	278,363 gallons	67,465 gallons
Areal Dose Rate[3]:	4,315 lb dry alum per acre 389 gallons alum solution per acre	4,070 lb dry alum per acre 367 gallons alum solution per acre
Estimated Cost[4]:	$278,363.00	$67,465.00

[1] $P_s = P_h/A_s$

[2] $D_a = T*((K_r*K_a)/K_d)*P_s$

[2] Volume of alum solution was calculated from an Al^{+3} content of 4.4% and a solution density of 11.1 lb/gallon.

[3] Areal dose rate calculated using that portion of the lake surface area which overlies depths of 15 feet or greater (Lake Wapogasset = 715.9 acres; Bear Trap Lake = 184.8 acres)

[4] Cost for alum and application as estimated by Tom Eberhardt of Sweetwater Technology is $1.00/gallon (1996 dollars). This price includes mobilization/demobilization, and does not include the cost of any special dosing titrations or contract/bid document preparation that may be required by the state.

Conclusions/Recommendations

The following briefly reviews the results of the calculations outlined above:

- The trophic state classification of the water quality in both Lake Wapogasset and Bear Trap Lake was mesotrophic immediately after spring overturn; by September, however, the water quality had degraded to a trophic classification of hypereutrophic.

- The hypolimnion in both lakes became anoxic coincident with thermal stratification.

- Thermal stratification commenced in June and continued until September. The stratification in both lakes was weak throughout the summer, and periods of cool and windy weather resulted in intermittent disruption of the stratification, resulting in erosion of the thermocline, and pulse phosphorus loads to the epilimnion.

- The phosphorus concentration in the hypolimnion of both lakes increased dramatically throughout the summer until fall overturn. The epilimnetic phosphorus concentration in both lakes also increased steadily during late-summer due to entrainment of high-phosphorus hypolimnetic water. Periodic increases in the epilimnetic phosphorus concentration during mid-summer were apparently due to intermittent entrainment of hypolimnetic water during cool, windy weather.

- The mass of phosphorus released from the sediments to the hypolimnion of Lake Wapogasset and Bear Trap lake were 2,083 Kg and 505 Kg, respectively.

- The mass of hypolimnetic phosphorus which was entrained to the epilimnion during fall overturn was 1,058 Kg and 242 Kg in Lake Wapogasset and Bear Trap lake, respectively. This internal load comprises 38 percent and 64 percent of the total annual phosphorus load to Lake Wapogasset and Bear Trap Lake, respectively.

- Application of alum to the sediments of the two lakes will result in phosphorus load reduction of 498 Kg in Lake Wapogasset and 133 Kg in Bear Trap Lake. This resulted in a fall-turnover epilmnetic concentration in Lake Wapogasset of 37 µg/L (as compared to the 1995 observed volume-weighted epilimnetic of 63 µg/L); and a fall-turnover concentration of 24 µg/L in Bear Trap Lake (a compared to the 1995 observed concentration of 60µg/L).

- Estimated cost for application of alum to the sediments at depth greater than 15 feet in Lake Wapogasset and Bear Trap Lake is $402,800.00 and $69,600.00, respectively, according to the calculation method of Cooke et. al. (1994); and $278,363.00 and $67,465.00, respectively, according to the calculation method of Eberhardt (1995).

In conclusion, the results of this study have shown that anoxic sediment release of phosphorus and subsequent entrainment of phosphorus to the epilimnion results in severe water quality degradation during late-summer in both Lake Wapogasset and Bear Trap Lake. Water quality modeling simulations indicate that application of alum to the lake sediments will provide a notable improvement in late-summer water quality.

Based on the results of this study, the following actions are recommended:

- Encourage the Wapogasset/Bear Trap Sanitary District to monitor for phosphorus during all routine water quality assessments of the treatment ponds and the surrounding groundwater monitoring wells, to install additional monitoring wells closer to Bear Trap Lake, and to accurately survey the elevations of the existing well casings. While this recommendation has little to do with the original project scope, there appears to be some confusion as to the direction and extent of groundwater flow from treatment pond seepage. A cursory examination of monitoring data, monitoring well water levels, and the area topography by a Barr Engineering Company hydrogeologist suggests that the direction of groundwater flow in the area of the treatment ponds is towards Bear Trap Lake. Accurately surveying the elevations of the existing well casings will ensure that water level data collected is interpreted correctly. Since seepage from the treatment ponds may cause localized groundwater mounding (thereby skewing the water levels in the existing monitoring wells), it is recommended that one or two additional monitoring wells be installed in the apparent groundwater drainage pathway between the ponds and Bear Trap Lake. Data from these wells will provide more conclusive information as to the actual direction of groundwater flow, and will allow the monitoring of phosphorus concentrations in the groundwater entering Bear Trap Lake. If it appears that unusually high phosphorus concentrations are reaching the lake via this groundwater flow, remedial action should be taken. Pete Prusak of the WDNR can provide assistance in issues pertaining to the wastewater treatment system and monitoring (Table 15).

Table 15 Agency and Technical Contacts

Agency	Name	Responsibility	Telephone	E-mail	Address
Wisconsin DNR	Dan Ryan Craig Roessler	Lake Planning Grant/Lake Protection Grant Application Assistance	715-635-2101	ryand@dnr.state.wi.us roesc@dnr.state.wi.us	WDNR Northwest District Headquarters Box 309 Spooner, WI 54801
Wisconsin DNR	Jane Malischke	Balsam Branch Priority Watershed Project Pollution Abatement Funding Assistance	715-635-2101	malij@dnr.state.wi.us	WDNR Northwest District Headquarters Box 309 Spooner, WI 54801
Polk County Land Conservation Department	Cheryl Bursik Jeff Timmons	Balsam Branch Priority Watershed Pollution Abatement Program Implementation	715-485-3725	n/a	Polk County LCD PO Box 460 Balsam Lake, WI 54810
Wisconsin DNR	Paul Garrison	Alum Treatment Technical Assistance	608-221-6365	garrp@dnr.state.wi.us	WDNR
Wisconsin DNR	James Pardee	Environmental Assessment Coordinator	608-266-0426	pardej@dnr.state.wi.us	WDNR Box 7921 101 South Webster Street Madison, WI 53707
Sweetwater Technology	Tom Eberhardt	Commercial Alum Applicator	218-927-2200 800-428-9900	n/a	Sweetwater Technology 317 Minnesota Ave. N. Aitkin, MN 56431
Wisconsin DNR	Pete Prusak	Polk County Wastewater Treatment Plant Discharge Monitoring	715-822-3590	prusp@dnr.state.wi.us	WDNR Box 397 Cumberland, WI 54801
Barr Engineering Company	Hal Runke Karen Jensen	Consultant for Lake Improvement Association	612-832-2600	hrunke@barr.com kjensen@barr.com	Barr Engineering Company 8300 Norman Center Drive Minneapolis, MN 55437

- Apply alum to the sediments at depths greater than 15 feet in Lake Wapogasset and Bear Trap Lake. The results of this study have shown that alum treatment is an appropriate management practice to remediate the effects of anoxic sediment phosphorus release, and will more than likely be quite successful. Two caveats most be added to this recommendation. First, Paul Garrison of the WDNR has had good success decreasing internal phosphorus load from anoxic sediments in Cedar Lake (a Wisconsin lake with similar morphometry to Lake Wapogasset) using a hypolimnetic aerator to prevent the lake from stratifying. Costs for the original aeration system are not available at this time, but operation of the system costs approximately $7,000 per summer. Before the Lake Wapogasset/Bear Trap Improvement Association commits to an alum treatment, it may be worthwhile to speak to Mr. Garrison about aeration instead. Most of the information compiled in this report will be useful in determining if aeration is a viable possibility for Lake Wapogasset and Bear Trap Lake. Mr. Garrison is in the process of writing a final report for this project; the Lake Improvement Association may want to request a copy. (Paul Garrison's address and phone number are included in Table 15.)

  Second, the WDNR has stated that funding for an alum treatment of Lake Wapogasset and Bear Trap Lake will not be approved until watershed nonpoint phosphorus sources are reduced as much as is feasible. This is to ensure that the alum applied to the sediments is not rapidly covered by suspended solids originating from watershed inflows.

  It is recommended that the dose rate as calculated by the method of Eberhardt (1995) be used to treat the sediments. This dose rate should be efficient for a treatment longevity of about 10 years (if watershed phosphorus loads are sufficient reduced). The estimated cost for treatment of Lake Wapogasset is $278,400 and Bear Trap Lake is $67,500.

  Two sources of funding are available for the proposed alum treatment. The Wisconsin Nonpoint Pollution Abatement Program is administered by the WDNR; this program is currently funding the Balsam Branch Priority Watershed Project. Since Lake Wapogasset and Bear Trap Lake are within this watershed, projects to improve the lake water quality are eligible for funding under this program. The state and local share would probably be 70 and 30 percent, respectively. The local share may be funded by other grants or governmental monies that the Lake Improvement Association can obtain. The contact person for the Pollution Abatement Program is Jane Malischke of the WDNR; her address and phone number are included in Table 15. The Lake Protection Grant Program (also administered by the WDNR) is another source of funding. This program provides up to $266,66.67 (75 percent state share, 25 percent local share); Dan Ryan of the Northwest

District Office is the contact person for this grant program (Table 15). A copy of the grant description and application is included in Appendix D.

In addition, the State of Wisconsin may require completion of an Environmental Assessment Worksheet prior to the alum application. A copy of this document is included in Appendix E. Most of the technical information required for completion of this document should be included within this report. James Pardee of the WDNR can answer any questions about the preparation of this document.

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