

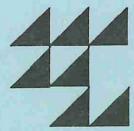
# AN EVALUATION OF THE EFFECTS OF WATERSHED TREATMENT SYSTEMS ON THE SUMMERTIME PHOSPHORUS CONCENTRATION IN METROPOLITAN AREA LAKES

PART TWO OF A REPORT TO THE  
LEGISLATIVE COMMISSION ON MINNESOTA RESOURCES

by  
Richard A. Osgood

METROPOLITAN COUNCIL  
*Mears Park Centre, 230 East Fifth Street, St. Paul, Minnesota 55101 Tel. 612/291-6359*  
*Publication Number 590-89-062b*

June 1989



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Metropolitan Council of the Twin Cities Area  
Mears Park Centre  
230 East Fifth Street  
Saint Paul, Minnesota 55101  
Telephone: 612/291-6359

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## SUMMARY

The Metropolitan Council sampled six lakes (Carver, Crystal, Elmo, Holland, McCarrons and Tanners; Fig. 1) as part of a grant from the LCMR. This report addresses the following tasks in partial fulfillment of the grant:

1. Define sensitive periods in lakes relative to phosphorus inputs.
2. Develop a time-dependent phosphorus model to be used to evaluate the effectiveness of lake management techniques.
3. Evaluate the effectiveness of wetlands and ponds on lake water quality.
4. Evaluate the roles of watershed versus in-lake management techniques for lake management.

A companion report, "The Water Quality Performance of Select Urban Runoff Treatment Systems", summarizes the other tasks required to satisfy the LCMR grant.

The six lakes were sampled from October 1987 through October 1988 at monthly intervals during the winter and every other week during the open-water season. The sampling included measurements of temperature, dissolved oxygen, underwater light, Secchi disk transparency, total phosphorus plus dissolved phosphorus fractions, total Kjeldahl nitrogen plus dissolved nitrogen, chlorophyll *a*, pH, specific conductance, and plankton. These data, except the plankton, are included in the appendices, although this report specifically analyzes the phosphorus dynamics in these lakes.

The dissolved fractions of phosphorus and nitrogen were found in low concentrations at the surface of the lakes and were poorly correlated with algal abundance. The particulate fractions, the difference between the total and dissolved fraction, were highly correlated with algal abundance, indicating that algae contain most of the nutrients available in the lakes.

The six lakes were chosen because they had been managed or changed in some way, making them good candidates for testing the lake phosphorus model that was to be developed. The observed changes in water quality were assessed, although there are generally insufficient data available to detect trends in these lakes. No changes in the water quality of Carver, McCarrons or Tanners Lakes have been observed over the past decade. This does not necessarily indicate that changes have not occurred, rather that the data cannot detect them if they have occurred. Lake Elmo and Holland Lake appear to have improved following the recent diversion of large inflows. Crystal Lake's water quality was worse when the lake was circulated.

Phosphorus is important to lake managers because it controls the general productivity of lakes. Previous Council studies have shown that the summertime total phosphorus concentration is strongly correlated to nuisances associated with over-productive lakes. For example, the frequency and abundance of algae, and the predominance of blue-green algae increase with increases in phosphorus. Thus, phosphorus is a logical management target.

Commonly used annual phosphorus models poorly predict the response of TCMA lakes to changes in phosphorus inputs. It appears that the lakes' response to phosphorus inputs are strongly seasonal, especially during the summer. Since the watershed phosphorus inputs are not normally synchronized with the lakes' most sensitive seasons, the annual phosphorus models are not appropriate. Due to these constraints, the evaluation of lake phosphorus dynamics has been confined to the prediction of summertime surface concentrations. A modified input-output model is used to better predict the short-term (less than 10 years) response of TCMA lakes to changes in phosphorus income.

A summertime phosphorus model was tested using data from the six sampled lakes plus two others. The model was designed to predict the lakes' average summertime phosphorus concentration as a function of watershed inputs of dissolved phosphorus. The predicted and observed average summertime phosphorus concentrations were in close agreement. The model's ability to predict the lakes' response to changing inputs was also assessed. In terms of summertime phosphorus, TCMA lakes are generally insensitive to commonly used watershed management practices; that is, the magnitude of reductions in lake phosphorus is proportionately lower than the magnitude of the watershed load reductions. The model appears to be an adequate tool to evaluate the effectiveness of watershed management scenarios on a cross-section of TCMA lakes.

The model was used to simulate the normal response of the sampled lakes with and without treatments. Watershed treatments, such as those at Carver, McCarrons and Tanners Lakes, reduced the average summertime phosphorus load by up to 17%, while the reduction in the lakes' summertime phosphorus concentration did not exceed 8%. Watershed diversions, such as those at Elmo and Holland Lakes, reduced the summertime phosphorus loads by 78 and 16% (respectively), while the reduction in the lakes' phosphorus was only 29 and 6%. Circulating Crystal Lake led to a 200 - 360% increase in its summertime phosphorus, despite a reduced summertime phosphorus input. The model was further used to generally evaluate the response of TCMA lakes to extremely effective watershed treatments. The results indicate that an average of 3% reduction in lakes' summertime phosphorus is expected with watershed treatments and an average of a 5% reduction in summertime phosphorus is expected with watershed diversions. In other words, these analyses indicate that there will be some small short-term benefits with commonly-used watershed treatments compared to using no treatment. However, since these treatments are most commonly used in association with development projects which simultaneously increase the phosphorus load to lakes, the effects of the treatment will probably be completely negated. Whole-lake circulation does not appear to have any practical applications for reducing the summertime phosphorus in TCMA lakes.

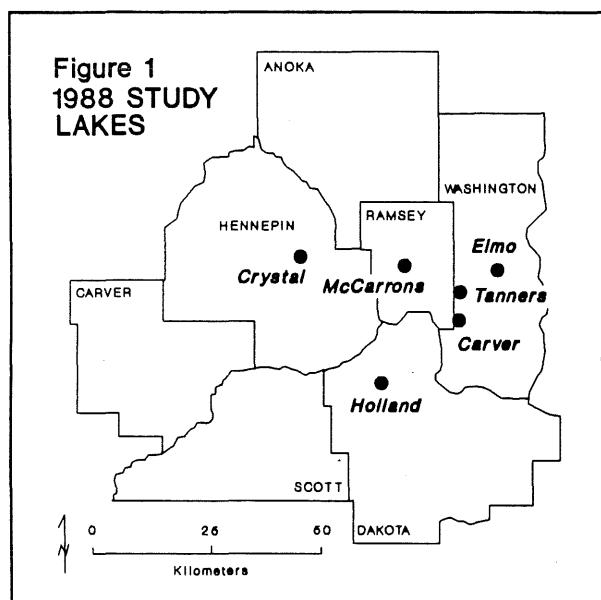
The long-term effects of reductions in watershed loading were also evaluated. Due to the differential seasonal loading relative to the lakes' sensitivities, and also due to the magnitude of internal phosphorus sources, long-term improvements in the lakes' phosphorus following watershed management appear unlikely. In addition, the application of commonly-used watershed treatments (eg. ponding and wetlands), will probably not fully mitigate the impacts of land development because land development generally increases phosphorus inputs to lakes. However, these watershed treatments provide some mitigation as well as many other benefits and the Metropolitan Council is not recommending that watershed treatment practices be

discontinued. Rather, the Metropolitan Council recommends that the impacts of nonpoint source pollution should be mitigated to the fullest possible extent in order to minimize the impacts of land development on lake water quality.

Since it appears likely that the enrichment of TCMA lakes will continue, the use of in-lake management techniques should be considered to treat the symptomatic nuisances associated with eutrophication. Since this approach is less direct than reducing nutrients, it is more costly and probably less effective. The treatment of symptoms without affecting the underlying causes of eutrophication is a long-term cost that appears to be required to manage our lakes. The Metropolitan Council recommends that in-lake management techniques be considered to treat nuisances related to eutrophication only after the most feasible watershed treatments have been included as part of the lake management plan.

## INTRODUCTION

The Metropolitan Council sampled six lakes from October 1987 through October 1988 (Figure 1). These lakes were sampled as part of the Council's cooperative effort with the City of Roseville's restoration of Lake McCarrons (Oberts and Osgood 1988) as well as under a grant from the Legislative Commission on Minnesota Resources (LCMR). This report will address the following tasks to partially satisfy the LCMR's grant:



1. Define sensitive periods in lakes relative to phosphorus inputs.
2. Develop a time-dependent lake phosphorus model that can be used to evaluate the effectiveness of various lake management techniques relative to their effects on summertime phosphorus concentrations.
3. Evaluate the effects of wetlands and ponds on the water quality of lakes.
4. Generally evaluate the roles of watershed versus in-lake management for lake management in the Twin Cities Metropolitan Area (TCMA).

This report summarizes the results of the lakes' aspects of the LCMR study and presents guidelines for directing the efforts of lake managers. A companion report (Oberts et al. 1989) addresses the watershed aspects of the LCMR study and also provides the basis for the watershed loading estimates used in this report.

This report also follows the general format of previous Council lake studies which have examined the water quality of TCMA lakes:

<u>YEAR</u>	<u>NUMBER OF LAKES</u>	<u>REFERENCE</u>
1980	60	Osgood (1981)
1981	30	Osgood (1982a)
1982	7	Osgood (1983)
1983	28	Osgood (1984a)
1984	43	Osgood (1984b)
1985	32	Osgood (1985)
1986/87	10	Osgood (1988a)
1988	6	This study

In addition, the general results of the earlier studies have been summarized in Osgood (1988b).

## METHODS

Six Metropolitan Area lakes were sampled from mid-October 1987 through mid-October 1988, including during the winter. The lakes were sampled at two-week intervals during the open water season and monthly during the period of ice-cover. The lakes were normally visited between 9 a.m. and 1 p.m. on the sampling days.

Samples were collected from one station located over the deepest spot near the center of the lakes (Figure 2). Time of day, surf and weather conditions (or ice and snow conditions in the winter) and station depth were recorded upon anchoring at the site. Temperature and dissolve oxygen were measured at one-meter intervals (half-meter intervals near the thermocline and larger intervals under the ice) using either a Yellow Springs, Inc. (model 50) field oxygen/temperature meter or by Winkler titrations and a mercury thermometer during the winter. Water transparency was measured using a 20cm black-and-white Secchi disk. Underwater irradiation profiles were measured during the open water seasons. Underwater irradiance was measured at half-meter intervals down to a depth where light was < 1% of the surface irradiance using a Li-Cor underwater quantum sensor which measures photosynthetically active radiation (PAR, 400-700 nm).

Water was collected from the lakes' surface (0-2 m) during the open water seasons using a 2 m-long PVC pipe that held two liters of water. Three such samples were mixed in a 8-liter plastic jug. Three 2-liter Van Dorn grabs from 0.5, 1.0, and 1.5-m depths were mixed to composite the surface sample during the winter. The water samples were transported on ice in a dark cooler and processed and preserved within six hours of collection. Water from the surface jug was withdrawn for the following possible chemical analyses: total phosphorus (TP), total dissolved phosphorus (DP), ortho-phosphorus (OP), total Kjeldahl nitrogen (TKN), nitrate-nitrite-nitrogen (N/N), chlorophyll *a* (CLA), pH and specific conductance (COND). Subsurface water samples were also drawn for the following possible analyses: TP, TKN, CLA (McCarbons only), pH and COND. In addition, phytoplankton samples were withdrawn from the surface jug and preserved in the field in a 1% acid Lugol's solution. Zooplankton were collected by vertically towing a 153  $\mu\text{m}$  mesh Wisconsin-type net through the entire water column and preserving the sample in a 4% formaldehyde with sucrose added (Haney and Hall 1973).

The routine chemical analyses were performed at the laboratory of the Metropolitan Waste Control Commission. The TP, DP and OP samples from the surface were analyzed in duplicate and average values are reported. DP samples were filtered through a 0.45  $\mu\text{m}$  membrane filter and analyzed for TP. The two phosphorus samples (TP and DP) were digested with the sulfates of hydrogen, potassium and mercury ( $\text{H}_2\text{SO}_4$ ,  $\text{K}_2\text{SO}_4$  and  $\text{HgSO}_4$ ). Following digestion, phosphorus was analyzed using a modified ascorbic acid reduction method (APHA 1980). Dissolved orthophosphate (filterable reactive orthophosphate) was analyzed using an ascorbic acid method which excludes the oxidative digestion step used for TP and DP.

The TKN samples were chemically reduced the same way as the total phosphorus samples. TKN from the surface were analyzed in duplicate and average values are reported. The TKN samples were color-intensified with sodium nitroprusside and assayed for ammonia colorimetrically. Nitrate-nitrite-nitrogen was measured by reducing nitrate to nitrite, then diazotizing the nitrite and assaying colorimetrically.

Samples for CLA were filtered onto a 0.45  $\mu\text{m}$  glass-fiber-filter, saturated with magnesium carbonate, and stored frozen in the dark until analyzed (within 30 days). Chlorophyll was extracted from the filters by homogenization in 90% aqueous acetone. The optical density of the extract was measured spectrophotometrically at 630, 647, 664 and 750 nm. CLA was calculated from a trichromatic equation that corrects for turbidity (APHA 1980).

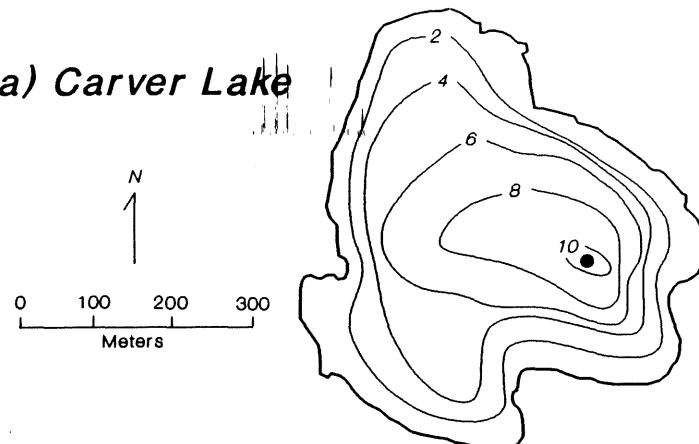
Specific conductance and pH were measured in the laboratory using one of several available lab meters that were calibrated each day.

Preserved phytoplankton samples were settled onto slides and were counted using an inverted microscope. The phytoplankton were identified to genera. A preliminary scan determined those taxa which comprised at least 90% of the community biovolume. These dominant taxa were enumerated in at least five subsamples so that at least 400 individual cells were tallied. In addition, ten random cells of each dominant taxa were measured so their volumes could be computed. Fewer subsamples and fewer measurements were made on the remaining taxa. These data are not discussed in this report, but are available upon request.

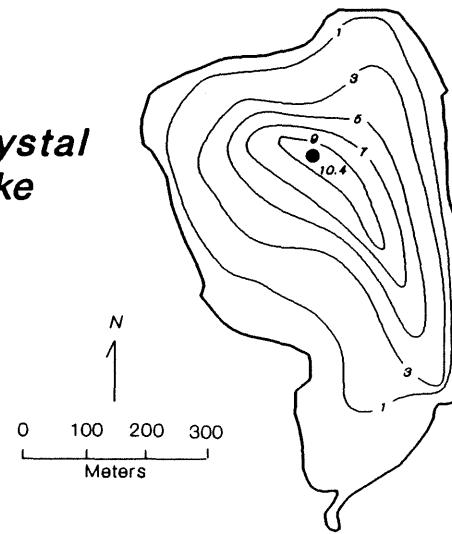
The zooplankton samples were diluted and five subsamples were examined. All adult Cladocera were enumerated in each subsample; Daphnia were identified to species and other cladocerans were identified to genera. These counts included at least 100 of the dominant taxon and at least 400 total adult cladocerans. Copepods were identified to genera, but were not counted. Rather, their abundance relative to the cladoceran community was noted. The relative abundances of the copepods are not reported here. These data are not discussed in this report, but are available upon request.

Figure 2. 1988 STUDY LAKE BATHYMETRIC MAPS

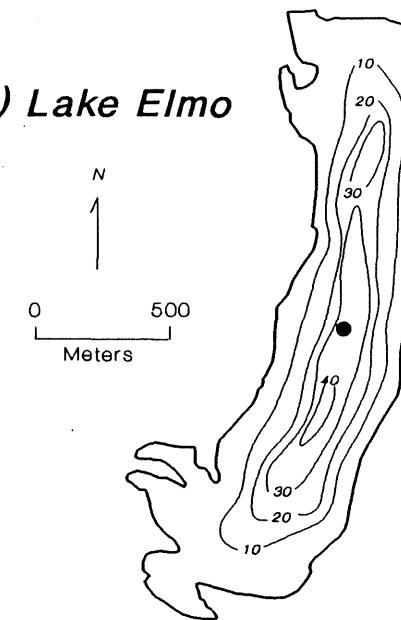
a) Carver Lake



b) Crystal Lake



c) Lake Elmo

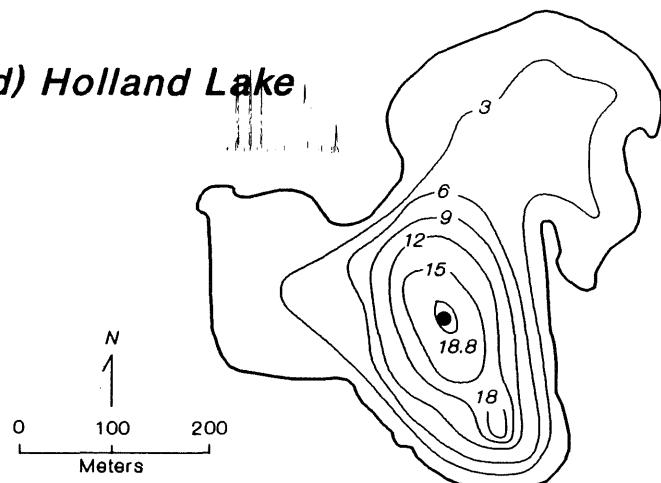


Lake contour lines in meters

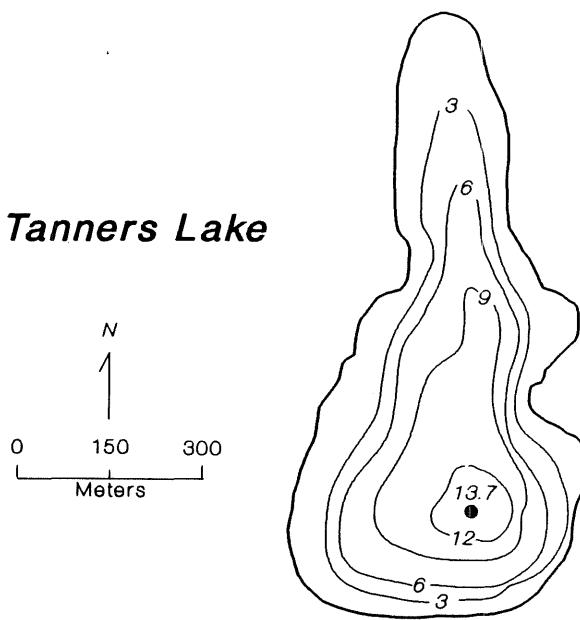
● Sampling station

Figure 2, continued. 1988 STUDY LAKE BATHYMETRIC MAPS

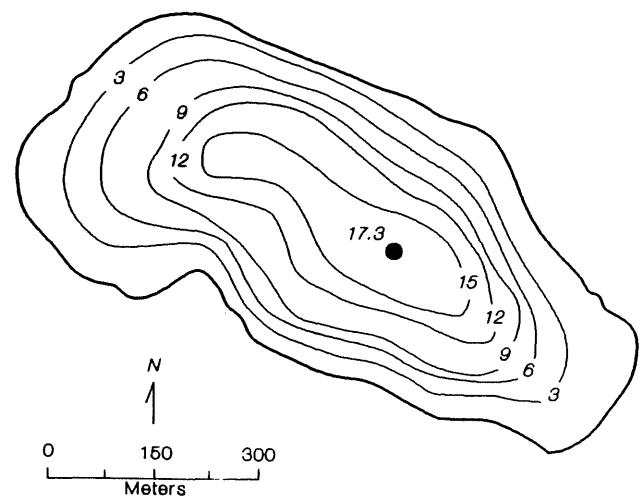
d) Holland Lake



f) Tanners Lake



e) Lake McCarrons



Lake contour lines in meters

● Sampling station

## LAKE WATER QUALITY SUMMARY

The surface nutrient concentrations of the six lakes are summarized in Figures 3a-f. The phosphorus fractions, DP and OP, are generally found in low concentrations in the surface of the lakes during the open water seasons and comprise larger fractions of the total phosphorus during the winter (Figures 3a-f). The particulate phosphorus fractions, as estimated by the difference between TP and either DP or OP, are highly correlated with CLA ( $r > 0.9$ ) during the summer. The same is true for dissolved nitrogen; there is no significant correlation between N/N and CLA, but TKN and CLA are highly correlated ( $r = 0.93$ ) during the summer, which is consistent with the findings in Osgood (1988b). Dissolved nitrogen (N/N) is found in uniformly low concentrations ( $< 0.2 \text{ g} \cdot \text{m}^{-3}$ ). These findings indicate that there is substantial summertime uptake of dissolved nutrients in these lakes.

The lakes for this project were selected because they had been managed or changed in some way. Thus, by monitoring these lakes and examining changes that have occurred, we can evaluate the effects of the management practices. For example, Carver, McCarrons and Tanners Lakes all had upstream watershed basins that were monitored; Elmo and Holland Lakes had diversions; and Crystal Lake was circulated. The effects of these management practices will be simulated in the following section. The extent to which that model can simulate the conditions in these lakes, it can be used to generalize the effects of these management practices on other TCMA lakes. The remainder of this section will establish the existing water quality of these lakes and whether or not it may be changing.

The trophic state of the six lakes is summarized in Table 1. Carlson's (1977) trophic state indices show that Elmo and Holland Lakes have 'good' water quality, Carver, McCarrons and Tanners Lakes have 'acceptable but sometimes unpleasant' water quality, and Crystal Lake has 'very poor' water quality (after Osgood 1985). Assuming that a change of ten TSI units indicates a change in trophic state, this analysis indicates that only Crystal Lake has gotten worse. The lakes are all eutrophic, although Elmo and Holland Lakes appear to have improved recently and may be mesotrophic, while Crystal Lake is hypereutrophic.

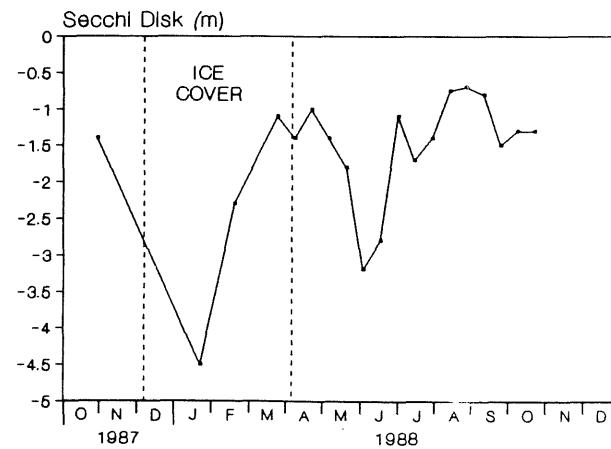
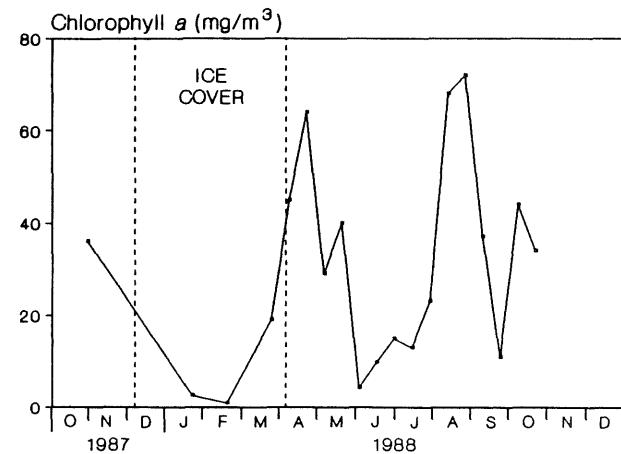
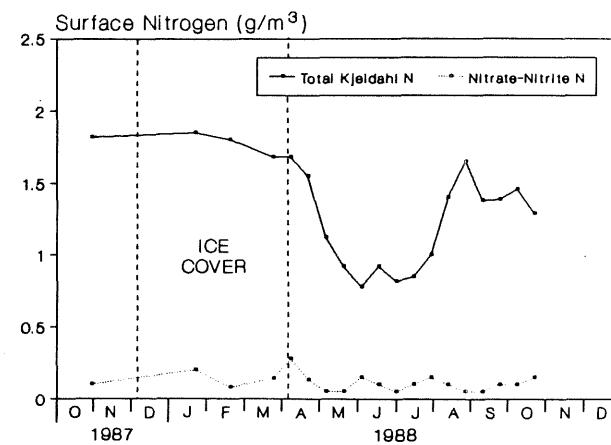
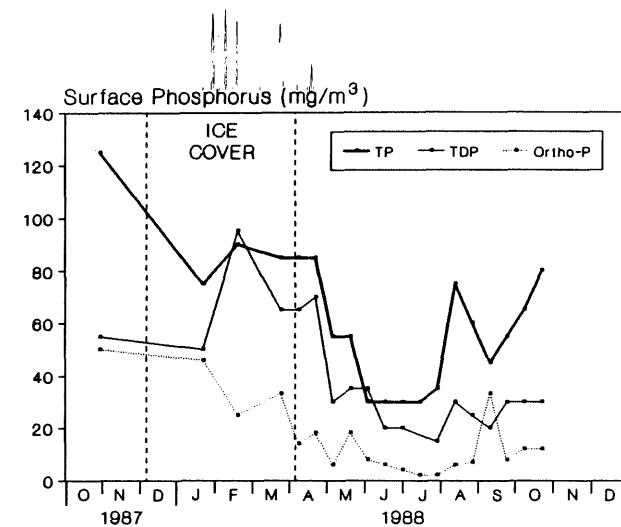
TABLE 1. Reported trophic state indices<sup>a</sup>.

Lake	1980	1981	1982	1983	1984	1985	1986	1987	1988
Carver	--	--	--	--	--	--	--	--	61
Crystal	--	--	--	--	--	--	68	73	88
Elmo	57	52	54	--	--	--	--	--	50
Holland	--	--	--	52-60	56-65	57-66	--	--	53
McCarrons	--	--	--	--	57	57	53	60	63
Tanners	53-62 <sup>“</sup>	--	--	--	--	--	--	--	54-62

<sup>a</sup> After Osgood (1982b).

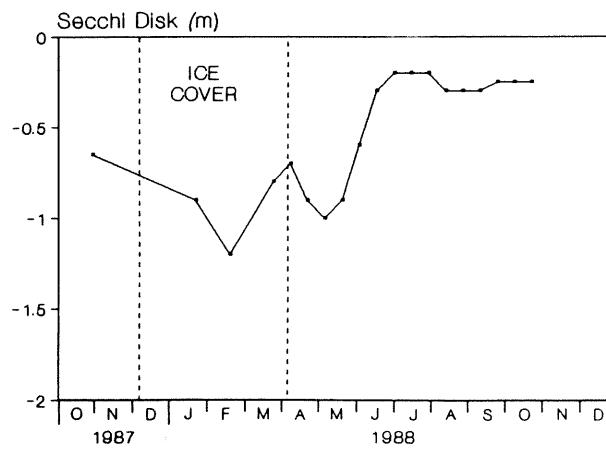
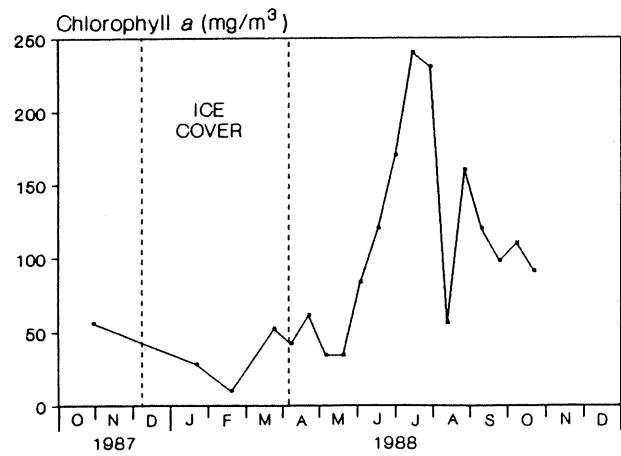
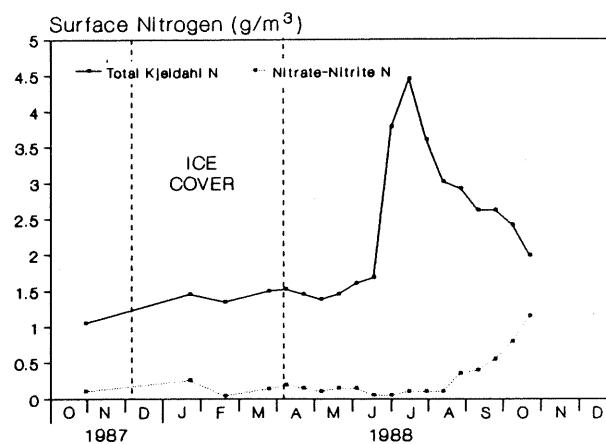
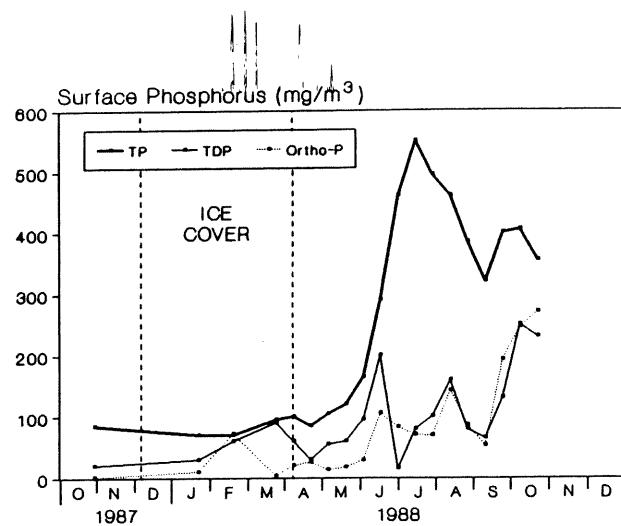
<sup>“</sup> Sampled only twice in 1980.

**Figure 3A. SURFACE PHOSPHORUS, NITROGEN AND CHLOROPHYLL a,  
PLUS SECCHI DISK TRANSPARENCY FOR CARVER LAKE, 1988**



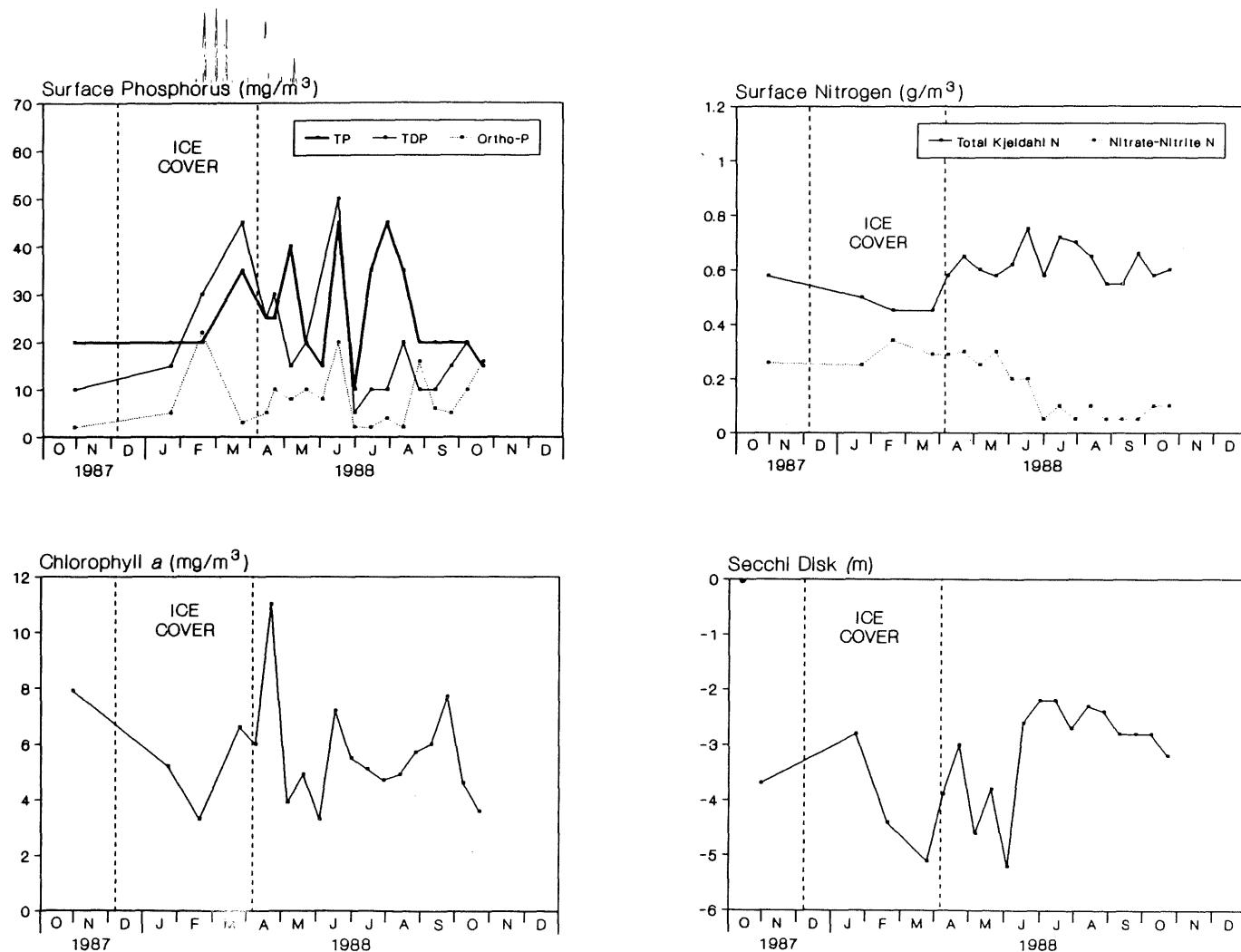
**Figure 3B. SURFACE PHOSPHORUS, NITROGEN AND CHLOROPHYLL  $a$ ,  
PLUS SECCHI DISK TRANSPARENCY FOR CRYSTAL LAKE, 1988**

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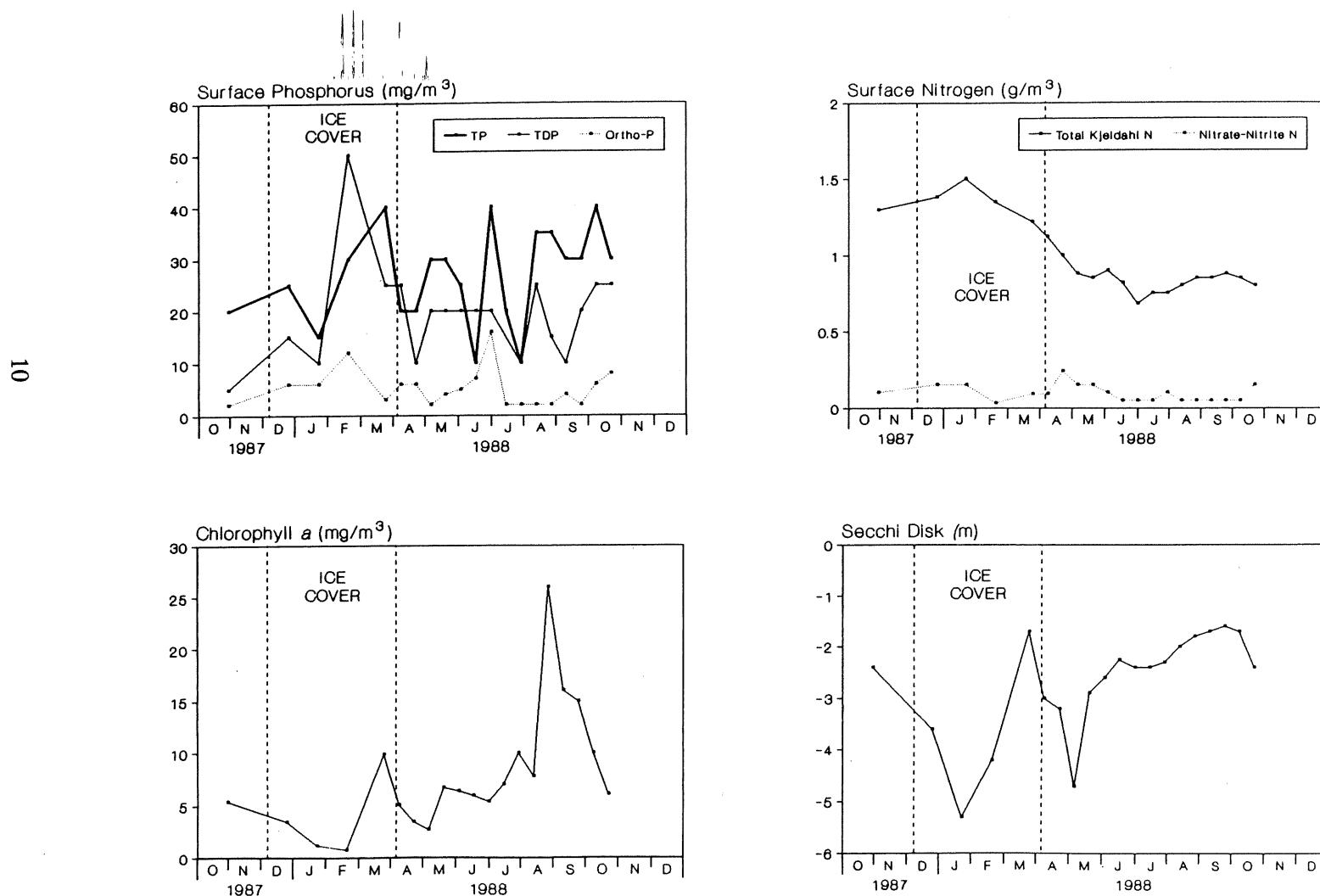


**Figure 3C. SURFACE PHOSPHORUS, NITROGEN AND CHLOROPHYLL a,  
PLUS SECCHI DISK TRANSPARENCY FOR LAKE ELMO, 1988**

6

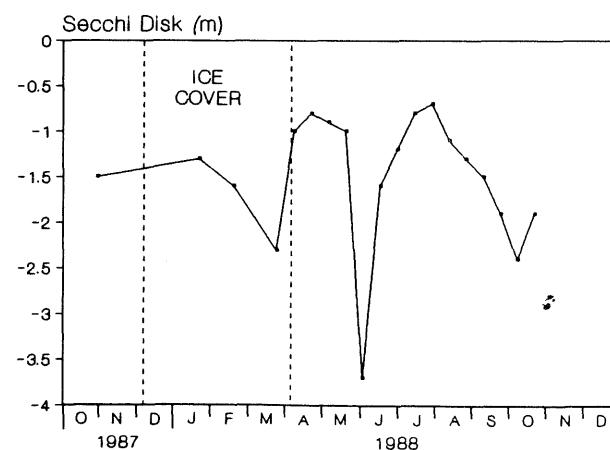
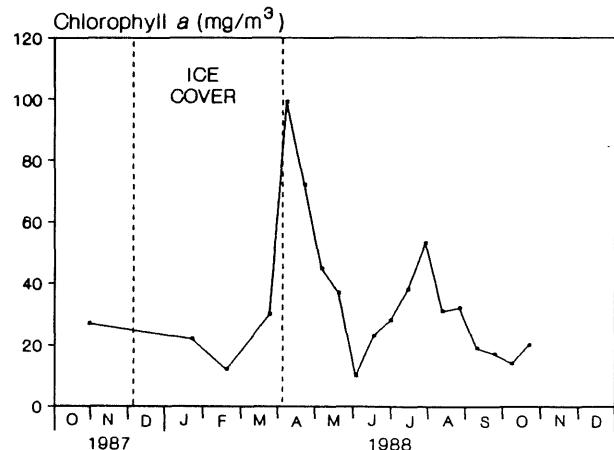
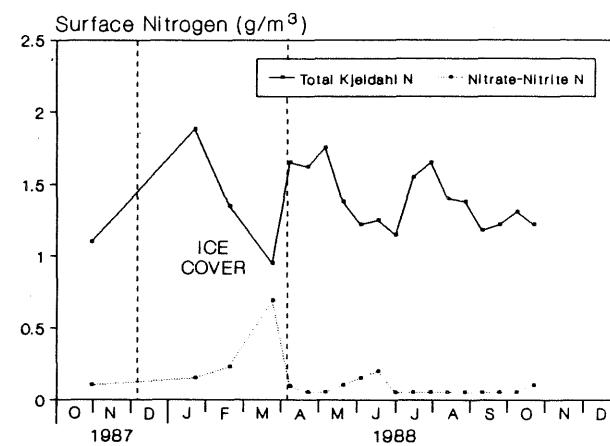
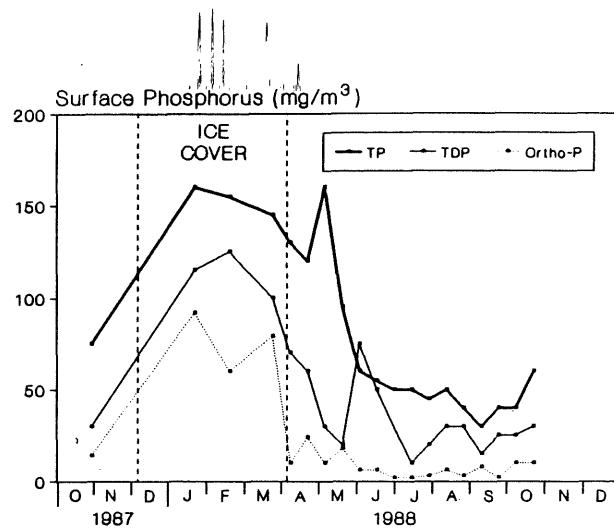


**Figure 3D. SURFACE PHOSPHORUS, NITROGEN AND CHLOROPHYLL a,  
PLUS SECCHI DISK TRANSPARENCY FOR HOLLAND LAKE, 1988**

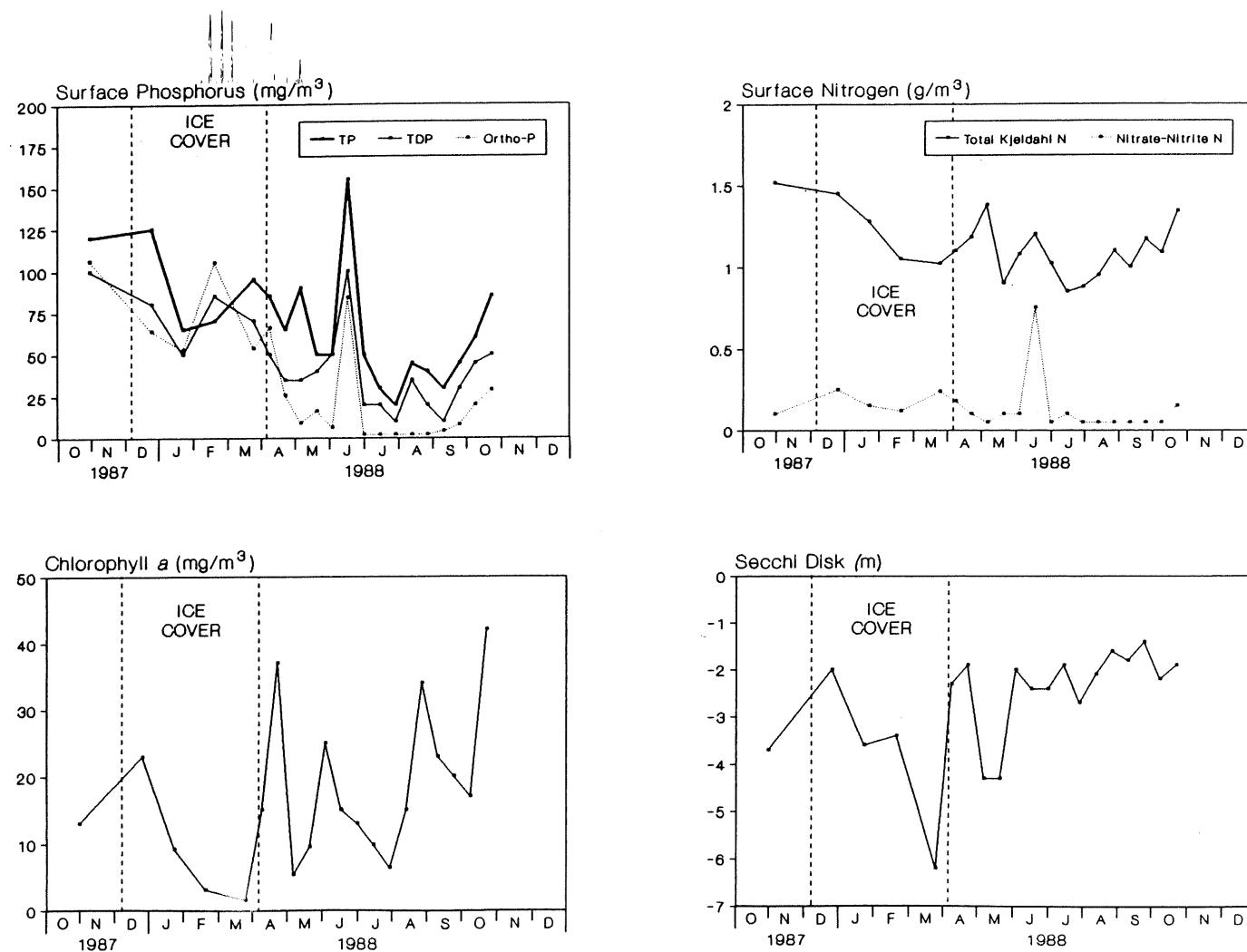


**Figure 3E. SURFACE PHOSPHORUS, NITROGEN AND CHLOROPHYLL a,  
PLUS SECCHI DISK TRANSPARENCY FOR McCARRONS LAKE, 1988**

II



**Figure 3F. SURFACE PHOSPHORUS, NITROGEN AND CHLOROPHYLL *a*,  
PLUS SECCHI DISK TRANSPARENCY FOR TANNERS LAKE, 1988**

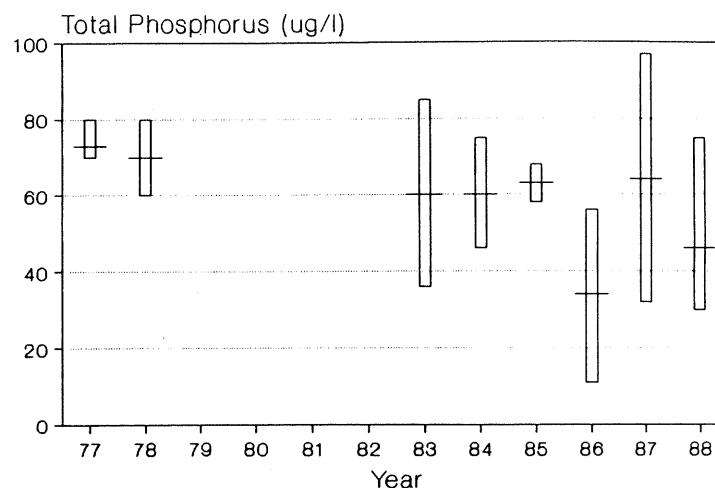
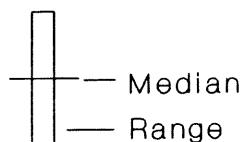


There appear to be inadequate data to evaluate trends in the water quality in these lakes, despite their intentional management. The two basins in Carver Lake's watershed were constructed in 1981 and 1977 (Lake Ridge and Carver Ravine, respectively); Crystal Lake has been circulated since 1973, except during 1986 and 1987; the underwater bypass at Lake Elmo was constructed in May 1987; the diversion at Holland Lake occurred in the autumn of 1986; the wetland management system at Lake McCarrons was functional sometime in 1986; and the wetland system at Tanners Lake was constructed during the spring of 1988. It is fortunate in one respect, that there was a drought in 1988; that is, the inflow of runoff into the lakes was extremely low and the impact of the reduced external phosphorus loading on the lakes should be large.

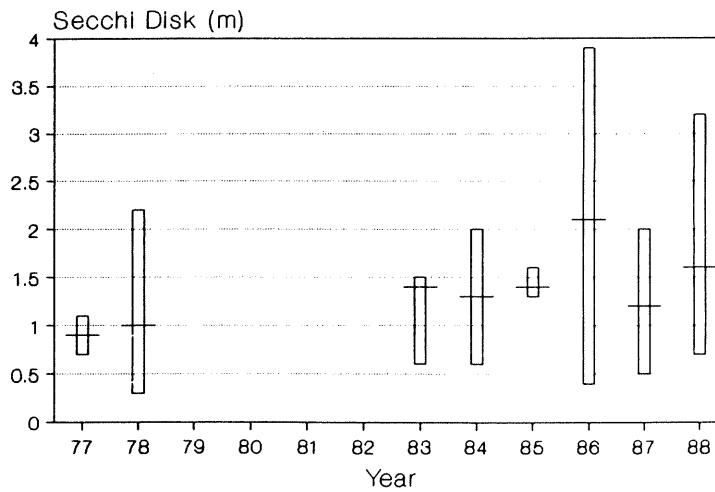
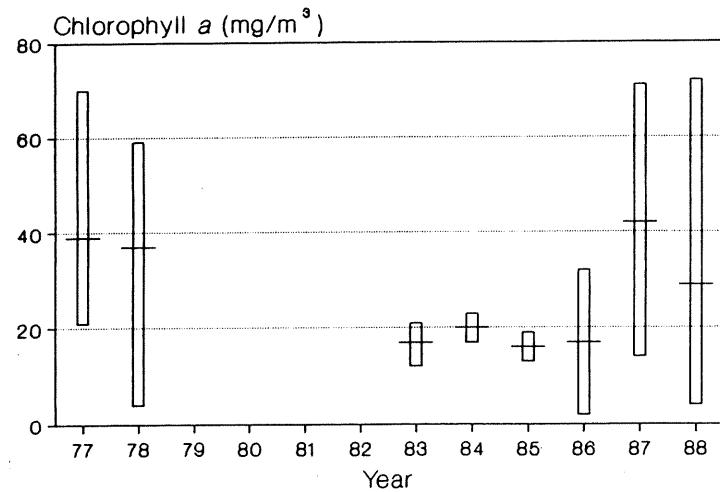
The normal variability in the quality of these lakes is difficult to assess due to the lack of data. However, it is important to understand the differences between trends in water quality that may be affected by intentional management schemes versus apparent trends due to annual or seasonal variability. The following evaluation attempts to summarize all available data from these lakes, then the following section on modelling attempts to explain this variability in order to assess the impacts of lake management practices.

There are no apparent trends in the water quality of Carver Lake (Figure 4a). Changes in water quality following the installation of the two watershed treatment systems in the early-1980s are not apparent from these data. Since these annual averages are from small samples ( $n = 2$  or 3 per year), the annual data were pooled to compare these water quality indicators from the period before the management (1977 and 1978) to the period following the treatments (1983 - 1987). The null hypothesis, that the means do not differ more than by chance, cannot be rejected at a probability of >90%. This means that either the lake's quality, as indicated by total phosphorus, chlorophyll or Secchi disk, has not changed or that there are insufficient data to indicate that a change has occurred.

**Figure 4A**  
**TRENDS IN WATER QUALITY -**  
**CARVER LAKE**  
 (data from Ramsey+Washington Metro  
 Watershed District, except 1988)



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There are no available data from Crystal Lake other than found in Table 2a. These data indicate a definite change in the quality of the lake has occurred with the artificial circulation of the lake. The lake's surface phosphorus and chlorophyll concentrations have doubled or tripled and the lake's clarity has been reduced to half or one third compared to the two earlier years. The few pre-1986 Secchi disk measurements that are available during other years that the lake was circulated (1973 - 1985) similarly indicate that the lake's quality was not improved (eg. no Secchi disk observations > 1 m).

TABLE 2a. Trends in the water quality of Crystal Lake.

Water quality Indicator*	1986	1987	1988
Mean TP ( $\text{mg} \cdot \text{m}^{-3}$ )	83	125	342
Mean TN ( $\text{g} \cdot \text{m}^{-3}$ )	1.40	1.47	2.84
TN/TP Ratio	17	12	8
CLA ( $\text{mg} \cdot \text{m}^{-3}$ )	61	70	113
SD (m)	1.2	0.8	0.4

\* May - September averages from the surface

Lake Elmo's water quality appears to have improved following the diversion of a large portion of its normal inflow in 1987 (Table 2b). The summertime total phosphorus concentration has not changed, but the chlorophyll concentration is substantially reduced and the transparency is greater. It is not clear why there was less algal abundance with no change in phosphorus.

TABLE 2b. Trends in the water quality of Lake Elmo.

Water quality Indicator*	1980	1981	1982	1984	1988
Mean TP ( $\text{mg} \cdot \text{m}^{-3}$ )	35	21	26	20	28
Mean TN ( $\text{g} \cdot \text{m}^{-3}$ )	1.19	0.94	1.13	0.84	0.76
TN/TP Ratio	34	45	43	42	27
CLA ( $\text{mg} \cdot \text{m}^{-3}$ )	14	9.6	12	9.7	5.4
SD (m)	1.9	2.4	2.2	2.5	3.1

\* May - September averages from the surface

The water quality of Holland Lake has changed. Prior to 1980, Holland Lake had the best water quality of lakes in Eagan and probably in all of Dakota County (Ayers et al. 1980). By the mid-1980s, Holland Lake's water quality was degraded (Table 2c). The quality of the lake improved in 1988; however, it is not clear to what extent the improvement is related to the diversion, because the summer's drought eliminated any surface inflows to the lake.

TABLE 2c. Trends in the water quality of Holland Lake.

Water quality Indicator*	pre-1980	1983	1984	1985	1988
Mean TP ( $\text{mg} \cdot \text{m}^{-3}$ )	$\approx 10^{**}$	36	43	38	27
Mean TN ( $\text{g} \cdot \text{m}^{-3}$ )	$\approx 0.80^{**}$	1.04	1.17	1.18	0.90
TN/TP Ratio	--	29	27	31	33
CLA ( $\text{mg} \cdot \text{m}^{-3}$ )	--	20	33	37	9.9
SD (m)	$\approx 3.0^{**}$	2.6	2.2	1.5	2.4

\* May - September averages from the surface

\*\* From Ayers et al. 1980

Lake McCarrons' water quality does not appear to have changed this decade (Table 2d), despite intensive watershed management (Oberts and Osgood 1988). In fact, the lake's phosphorus concentration was actually lowest when external loadings were highest. This unexpected observation has not been satisfactorily explained (Oberts and Osgood 1988).

TABLE 2d. Trends in the water quality of Lake McCarrons.

Water quality Indicator*	1981**	1984	1985	1986	1987	1988
Mean TP ( $\text{mg} \cdot \text{m}^{-3}$ )	52	38	32	28	46	61
Mean TN ( $\text{g} \cdot \text{m}^{-3}$ )	1.31	1.16	1.03	0.93	1.21	1.46
TN/TP Ratio	25	31	32	33	26	24
CLA ( $\text{mg} \cdot \text{m}^{-3}$ )	--	20	19	14	26	30
SD (m)	1.4	2.3	1.9	2.8	1.8	1.4

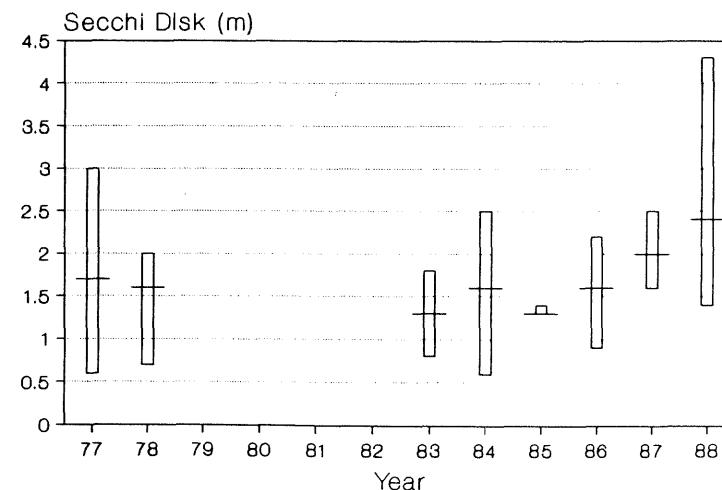
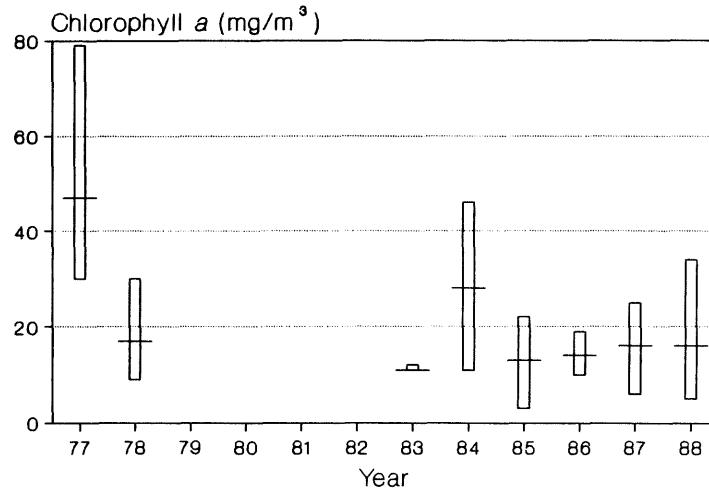
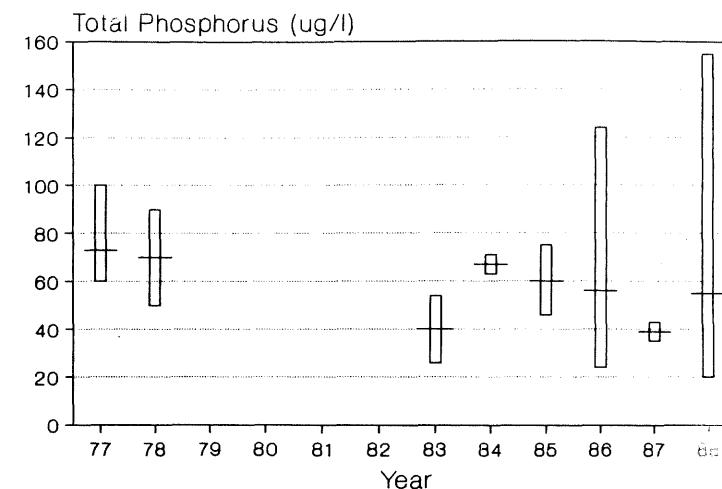
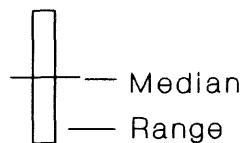
\* May - September averages from the surface

\*\* From Donohue 1983

The water quality of Tanners Lake does not appear to have changed since the installation of the wetland treatment system in the spring of 1988 (Figure 4b); however, there has probably not been enough time to detect any immediate changes. Also, the drought in 1988 reduced the normal amount of runoff to the lake.

**Figure 4B**  
**TRENDS IN WATER QUALITY -**  
**TANNERS LAKE**

(data from Ramsey-Washington Metro  
Watershed District, except 1988)



## SUMMERTIME PHOSPHORUS MODEL

### THE MODEL

Steady-state solutions to phosphorus input-output models are commonly used for predicting lake phosphorus concentrations in lake assessments. However, the assumption that lakes are at steady-state, implies that the loading functions, as well as internal settling and reaction processes are constant through time. These assumptions are valid in cases where large-scale changes in phosphorus or water loading have occurred (Edmondson and Lehman 1981; Welch and Patmont 1980). More commonly though, these assumptions are not true. Internal and external phosphorus loading, flushing and the settling and outflow of phosphorus do not occur uniformly within or between years. As a result, changes in lake quality are often not apparent following real changes in nutrient loading (Oberts and Osgood 1988; Ramsey Co. 1988). Perhaps of more immediate importance, is assessing shorter-term changes in the apparent quality of lakes. Since annual steady-state phosphorus models are of little use for this purpose, another analytical approach is needed.

The appearance of lakes during the summer is important to most lake users. Nuisances, such as the frequency and intensity of algal blooms, the predominance of blue-green algae or the clarity of the water, are related to summertime surface phosphorus concentrations (Osgood 1988a, b). It seems reasonable then, to confine the consideration of lake phosphorus concentrations to the summertime epilimnia. Since the inputs and outputs of water and phosphorus to the summertime epilimnia are not equal, the steady-state assumptions cannot be used. Instead, a time-dependent solution to an input-output model may be appropriate.

Existing input-output models assume that the lake is a completely mixed reactor and the pollutants are uniformly distributed (Chapra and Reckhow 1983). This model can be modified by considering inputs, outputs and reactions relative to the summertime epilimnion rather than to the whole lake throughout the year. Since the inputs and outputs are not in balance during the summer, a dynamic equilibrium does not exist. The necessary time and space constraints are as follows: the summertime epilimnion is the surface layer of the lake down to a depth where the temperature gradient exceeds  $1^{\circ}\text{C m}^{-1}$ ; expressed as a summertime average from June through September. Inputs and outputs of water and phosphorus occur across these time and space boundaries. The available data do not allow the definition of loading functions or internal reactions with respect to time during the summer, so they will be assumed to occur at uniform rates during the summer. A time-dependent solution to a step loading function conforms to these conditions.

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The time-dependent solution to a step input has been modified from Chapra and Reckhow (1983) as follows:

$$P_t = W/Q_i + kV [1 - \exp((-Q_o/V) + k)] + P_i[\exp((-Q_o/V) + k)] \quad (1)$$

where  $P_t$  is the epilimnetic phosphorus concentration ( $\text{mg} \cdot \text{m}^{-3}$ ) at the end of the summer,  $P_i$  is the initial summertime phosphorus concentration ( $\text{mg} \cdot \text{m}^{-3}$ ),  $W$  is the rate of available (to the epilimnion) phosphorus inflow into the epilimnion ( $\text{mg} \cdot \text{summer}^{-1}$ ),  $Q_i$  is the inflow of water to the epilimnion ( $\text{m}^3 \cdot \text{summer}^{-1}$ ),  $Q_o$  is the surface outflow of water from the lake ( $\text{m}^3 \cdot \text{summer}^{-1}$ ),

$V$  is the epilimnetic volume ( $m^3$ ), and  $k$  is first-order reaction coefficient (summer $^{-1}$ ). Since it has been assumed that the inputs and reactions occur at constant rates, the time factor has been removed from this solution by setting it equal to the length of the summer (=1). As a result, the form of Eq. 1 is linear, but the actual time-series of summertime epilimnetic phosphorus concentrations in lakes rarely takes this form (see Osgood 1988c). Thus, this equation cannot be used to predict intermediate phosphorus concentrations between  $P_i$  and  $P_t$ , so its only appropriate use is as an estimate of the summertime average epilimnetic phosphorus concentration ( $P$ ):

$$P = (P_i + P_t) / 2 \quad (2)$$

The summertime water loading terms are computed as follows:

$$Q_i = Q_r + Q_p \pm Q_g = Q_e + Q_o \pm Q_g \quad (3)$$

where  $Q_r$  is the inflow from surface runoff,  $Q_p$  is direct precipitation onto the surface of the lake,  $Q_e$  is the water that is lost through evapotranspiration,  $Q_o$  is the lake's outflow and  $Q_g$  is the groundwater flow either to or from the epilimnion. The input of phosphorus to the epilimnion is computed as follows:

$$W = P_r + P_p + P_{int} \quad (4)$$

where  $P_r$  is the mass of phosphorus that enters the epilimnion in surface runoff,  $P_p$  is the mass of phosphorus that enters the epilimnion from precipitation, and  $P_{int}$  is the mass of phosphorus that enters the epilimnion from internal sources. Since field data have shown that the particulate fraction of phosphorus in surface runoff is quickly settled (Oberts and Osgood 1988), inputs of dissolved phosphorus from surface runoff to the epilimnion will be considered here as the available fraction (see Hecky and Kilham 1988).

## TESTING THE MODEL

Reasonably complete data for the inputs to the model have been measured for several lakes in the TCMA. These data plus "good" estimates for other inputs will be used to test the model's performance (Table 3). An alternate approach would be to estimate the input parameters for a large number of lakes and compare the predicted to the observed values in order to empirically determine the values for the constant ( $k$ ) in the model. Since the relative error associated with estimating the input parameters is high (Winter 1981), the uncertainty in the estimate of  $k$  by this method would probably also be high. Since the theoretical basis for the model has been established, testing its performance using "good" input values should confirm its utility.

TABLE 3. Input parameters for testing the time-dependent phosphorus model.

LAKE-YEAR	$Q_i$	$Q_o$	V - ( $10^6 \text{ m}^3$ ) -	$P_p$	$P_r$ --- (kg) ---	$P_{int}$	$P_i$ ( $\text{mg} \cdot \text{m}^{-3}$ )
Carver-1988	.261	0	.736	1.0	39.7	44.3	30
Crystal-1986	.843	0	.669	3.3	170	101.4	55
Crystal-1987	1.01	0	.669	3.9	203	101.4	110
Crystal-1988	.343	0	.951	1.3	69	1260	120
Elmo-1982	.504	.115	5.00	4.8	20.2	0	30
Elmo-1988	.3	0	5.00	5.3	14.7	0	15
Holland-1983	.115	0	.324	1.1	12.5	15.7	15
Holland-1984	.146	0	.324	1.3	15.9	15.7	25
Holland-1985	.107	0	.324	1.0	11.6	15.7	40
Holland-1988	.052	0	.324	0.6	4.3	15.7	17
McCarrons-1984	.527	.187	.853	3.0	59	23	65(20)
McCarrons-1985	.327	.213	.853	2.3	36	23	25
McCarrons-1986	.466	.480	.853	3.3	50	23	20
McCarrons-1987	.290	.198	.853	2.9	37	23	60(30)
McCarrons-1988	.140	0	.853	1.4	17.8	23	60
Spring-1980	2.24	.964	11.6	19	317	6426	120
Spring-1981	2.12	.952	11.6	19	315	6426	101
Spring-1982	1.16	0	11.6	10	165	6426	80
Spring-1984	2.09	.829	11.6	24	403	3672	50
Square-1980	.908	.488	2.89	6.4	14.5	0	20
Square-1981	.905	.485	2.89	6.3	14.5	0	18
Square-1982	.708	.288	2.89	3.3	5.3	0	20
Square-1983	.908	.488	2.89	6.4	14.5	0	10
Square-1984	—	1.01	.593	8.1	18.5	0	20
Square-1985	—	.882	.462	6.0	13.6	0	10
Tanners-1988	.108	0	.858	1.2	12.2	27.2	50

Underlined values are based on direct field observations.

The initial summertime phosphorus concentration in TCMA lakes is consistently observed to be low (Osgood 1988a). Normally, springtime phosphorus concentrations are elevated following ice-off and vernal mixis. The spring clear water phase (Lampert et al. 1986) occurs in mid-May and results in the accelerated loss of phosphorus from the epilimnion. This loss is the result of deposition and loss through the lake's outlet. It appears that the relative loss by these two mechanisms influences the magnitude of  $P_i$ , however very little field data are available to quantify this (see Model Sensitivity below). A summary of TCMA lakes that have been sampled in more than one year or sampled more frequently than monthly indicates that median  $P_i$  is  $25 \text{ mg} \cdot \text{m}^{-3}$  (interquartile range: 20-30, range: 5-180,  $n=72$ ) from sampling dates between 15 May and 15 June.

The hydrologic inputs to the summertime epilimnia include precipitation directly on the lakes surface, surface runoff and sometimes groundwater. Direct precipitation is the total summertime rainfall over the lakes' surface area ( $Q_p$ ). Rainfall data were either gathered near each lake or were taken from the Twin Cities International Airport. Runoff volumes ( $Q_r$ ) at sites adjacent to several TCMA lakes have been measured during several years. These data were used to calculate the runoff to the test lakes (see Oberts 1983; Oberts et al. 1989). In some cases the inputs of groundwater to the epilimnion can be estimated. In these cases, net groundwater inputs are added to  $Q_r$  and  $Q_o$  according to Eq. 3. Finally, in cases where the outflow was known to be  $> 0$ , but had not been measured, it was estimated as the difference between  $Q_i$  and evapotranspiration ( $0.7 \times \text{pan data, ca. } 0.49 \text{ m} \cdot \text{summer}^{-1}$ , see Osgood 1988b).

The phosphorus inputs to the summertime epilimnion include atmospheric sources, phosphorus in the runoff plus internally recirculated phosphorus. Atmospheric phosphorus ( $P_p$ ) was estimated assuming that  $Q_p$  had a phosphorus concentration of  $20 \text{ mg} \cdot \text{m}^{-3}$  (Oberts 1982). The phosphorus loads from surface runoff ( $P_r$ ) were either measured or calculated from the measured inputs as were the runoff volumes (see above). Internal phosphorus inputs ( $P_{int}$ ) could only be measured indirectly or estimated. The magnitude of internal phosphorus inputs into the epilimnion relates to the duration and extent of mixis in TCMA lakes (Osgood 1988c). Dimictic lakes remain stratified throughout the summer and internal phosphorus does not get mixed into the epilimnion. As lakes tend toward polymixis, internally supplied phosphorus comprises a larger fraction of the lakes' phosphorus budget. The frequency and extent of mixis in TCMA lakes are quantitatively related to lake morphometry (Osgood 1988b, c). The internal phosphorus input to lakes from littoral and profundal sources is generally estimated as follows:

Phosphorus input from littoral sources occurs at a rate of  $2.2 \text{ mg} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$  (Jacoby et al. 1982) over the epilimnetic bottom area throughout the summer (120 d).

The epilimnetic bottom area is the area exposed to epilimnetic water (0 down to the epilimnetic depth). The epilimnetic depth can be estimated using the surface area of the lake (see Osgood 1988b).

Phosphorus input from the profundal sediments occurs at a rate of  $12 \text{ mg} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$  (Nürnberg 1984) over the hypolimnetic bottom area during the number of summertime days that the lake is likely to be mixing. This number of days equals the difference between the duration of the summer (120 d) and the duration of stratification. The duration of stratification can be estimated using the relative epilimnetic volume (see Osgood 1988b).

The first-order reaction coefficient describes the loss of phosphorus in the lake. Phosphorus loss is the result of two processes: loss of inflowing phosphorus and loss of in-lake phosphorus. Sedimentation is probably the sum of both processes (Prairie 1988). By considering the external input of total dissolved phosphorus rather than total phosphorus, the loss of the inflowing phosphorus need not be considered. This loss of in-lake phosphorus may be determined empirically (see Canfield et al. 1982), but here,  $k = 1.05 \cdot \text{summer}^{-1}$  (Chapra and Reckhow 1983).

Inflows of water and phosphorus to Carver Lake were measured as part of the present study. Since the lake level was observed to drop throughout the summer, the outflow from the lake was presumed to be zero. Most of Carver Lake's epilimnetic phosphorus came from internal sources (52%) during the dry summer (Table 3). Table 4 shows that the predicted and observed phosphorus concentrations in Carver Lake to agree.

TABLE 4. Predicted and observed average summertime phosphorus concentrations (mg·m<sup>-3</sup>) using eqs. 1 and 2.

LAKE-YEAR	PREDICTED P	OBSERVED P
Carver-1988	47	43
Crystal-1986	95	85
Crystal-1987	133	141
Crystal-1988	403	394
Elmo-1982	22	22
Elmo-1988	11	27
Holland-1983	31	38
Holland-1984	39	47
Holland-1985	48	41
Holland-1988	29	26
McCarrons-1984	63(34) <sup>1</sup>	31
McCarrons-1985	34	32
McCarrons-1986	34	27
McCarrons-1987	57(38) <sup>1</sup>	39
McCarrons-1988	53	48
Spring-1980	238	237
Spring-1981	227	149
Spring-1982	215	149
Spring-1984	130	112
Square-1980	15	24
Square-1981	14	15
Square-1982	14	19
Square-1983	8	12
Square-1984	15	14
Square-1985	8	8
Tanners-1988	47	39 (52) <sup>2</sup>

1. Number in ( ) corresponds to predicted P using P<sub>i</sub> in ( ) from Table 3.

2. Number in ( ) is the summertime average including the observation from June 14, the number outside excludes this observation.

Crystal Lake had been artificially circulated until early-1986, when the Council began studying the lake. The external water and phosphorus loads to the lake were estimated using runoff models developed with data from similar urban areas in the TCMA (Oberts 1983). The circulation system was returned to service during 1988, which affected the internal phosphorus load to the epilimnion by increasing the mixing depth of the lake. Based on observations of the net change in the lake's phosphorus content at a time when external inputs were zero, internal loading during the summer of 1988 occurred at a rate of  $35 \text{ mg} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$  over the epilimnetic bottom area. This rate was used to calculate  $P_{int}$  in 1988, which represented a 12-fold increase compared to the two earlier years. The circulation of the lake also affected the depth of the epilimnion and the initial phosphorus concentration compared to the earlier years. These adjustments were also included in Table 3. The change in the lake's phosphorus concentration is accurately predicted by the model (Table 4).

Inputs of water and phosphorus to Lake Elmo were measured in 1982 (Osgood 1983). Internal phosphorus from littoral sources was found to be zero. Since then, runoff from Eagle Point Lake has been routed into an underwater pipe to bypass Lake Elmo. As a result, no water or phosphorus entered Lake Elmo from Eagle Point Lake during 1988 (Table 3). The model predicts a reduction in phosphorus concentration that was not observed in 1988 (Table 4), although other trophic state indicators showed improvement (Table 2b).

Holland Lake was sampled from 1983 through 1985 and again during this study. Runoff from a large portion of its watershed was diverted in 1986, which led to a substantial reduction in the lake's external phosphorus load (Table 3). Internal phosphorus loading from the littoral area appears to contribute a large fraction of the lake's epilimnetic phosphorus budget. The impacts of the reduced phosphorus load are accurately predicted by the model (Table 4).

Lake McCarrons was sampled from 1984 - 1988 and its watershed was sampled in 1985 and 1987 (Oberts and Osgood 1988). A large fraction of its runoff was treated following the construction of a wetland and detention system in the watershed. This project, completed in 1986, resulted in the overall reduction of the summertime total dissolved phosphorus load by about 25%. The impacts of the reduced phosphorus loading to the lake were confusing (see Oberts and Osgood 1988) because the lake's phosphorus increased (Table 3). It appears that the varying initial phosphorus concentrations observed in the lake had a large effect on the lake's summertime phosphorus concentration. This is further evaluated in the following section.

Inputs of water and phosphorus to Spring Lake were measured in 1982 (Osgood 1983). Internal phosphorus contributed >90% of the lake's summertime phosphorus budget during years when flake-forming Aphanizomenon were dominant (Osgood 1988d). Internal phosphorus loading was observed to occur at rates averaging  $21 \text{ mg} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$  over the entire lake area during the years when flake-forming Aphanizomenon were present (1980-1982) and at rates of  $12 \text{ mg} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$  in 1984 when flakes were not observed. The model predicts the decrease in the lake's phosphorus in 1984 despite the increase in the external load.

The outflow of water and phosphorus from Square Lake were measured in 1982 (Osgood 1983). External inputs were low and groundwater contributed a large portion of flow into the lake. The impacts of the unusually high groundwater flow through the lake are accurately predicted by the model (Table 4).

Inflows of water and phosphorus to Tanners Lake were measured as part of the present study. Since the lake level was observed to drop throughout the summer, the outflow from the lake was presumed to be zero. The model appears to reasonably predict the observed phosphorus in Tanners Lake whether or not the lake's phosphorus concentration from June 14 ( $=155 \text{ mg} \cdot \text{m}^{-3}$ ) is excluded from the summertime average (Table 4).

The predicted average summertime epilimnetic phosphorus concentrations agree very well with observations from the lakes (Table 4). It appears that when "good" input data are used, this model accurately predicts average summertime phosphorus concentration in these lakes. It also appears that the model and the lakes are differentially sensitive to the input variables.

## MODEL AND LAKE SENSITIVITY

The sensitivity to changes in runoff phosphorus and associated water loads (both internal and external), the initial phosphorus concentration and the reaction coefficient,  $k$ , will be evaluated in this section. The model's sensitivity to uncertain input values as well as the lakes' sensitivity to a range of inputs to their epilimnia can be evaluated simultaneously. To the extent that the test lakes represent a cross-section of TCMA lakes, this approach should provide a general assessment of lake behavior relative to common management practices in the TCMA. Once these sensitivities have been defined, the model may be used to evaluate the response of TCMA lakes to management practices which affect their summertime epilimnetic phosphorus dynamics.

The first input to be evaluated is the watershed loading of water and phosphorus. The model (Eqs. 1 and 2) will be used to evaluate the effects of changes in the surface inputs of phosphorus and water while keeping the other parameters constant. The test lakes' parameters from Table 3 will be modified as follows: both the water and phosphorus loads will be decreased and increased by a factor of 2, then only the phosphorus load will be both decreased and increased by a factor of 2. The results of this simulation compared to the unmodified condition are presented in Table 5. Since there were no test lakes with high levels of external loading and low levels of internal loading, Crystal Lake - 1987 input data were modified ( $P_{int} = 0$ ) to mimic this situation.

TABLE 5. Simulated changes in summertime phosphorus concentrations with changes in the watershed inputs of water and/or phosphorus (from Eqs. 1 and 2 and Table 3).

LAKE-YEAR	UNCHANGED P INPUT (mg·m <sup>-3</sup> )	WATER AND P INPUTS x 1/2	x 2	P INPUT x 1/2	P INPUT x 2
Carver-1988	47	43(-9)	57(+21)	35(-26)	54(+15)
Crystal-1987	133	126(-5)	140(+5)	114(-14)	172(+29)
Crystal-1987 (P <sub>int</sub> = 0)	56	43(-23)	87(+52)	37(-34)	95(+70)
Crystal-1988	403	431(+7)	350(-13)	395(-2)	405(+0)
Elmo-1982	22	21(-5)	23(+5)	21(-5)	23(+5)
Holland-1984	39	35(-10)	45(+15)	34(-13)	49(+26)
McCarrons-1985	34	30(-12)	42(+24)	29(-15)	45(+31)
Spring-1982	215	217(+1)	214(-1)	213(-1)	219(+2)
Square-1982	14	14(0)	14(0)	14(0)	14(0)
Tanners-1988	47	45(-4)	50(+6)	45(-4)	51(+9)

Numbers in ( ) are percent change from the unchanged condition.

The results of this exercise indicate that when both the water and phosphorus watershed loads were changed by a factor of 2, the impact on the lake was slight and when only the phosphorus load was changed by a factor of 2, the impact in the lake was relatively greater, but still proportionately small. Further, this analysis indicates that the impact of watershed inputs is less in lakes where internal phosphorus comprises a greater fraction of the summertime budget (Table 5), even to the extreme where reductions in watershed loads can lead to increases in summertime epilimnetic phosphorus concentrations (eg. Crystal and Spring Lakes). This happens when internal phosphorus is a large fraction of the lake's budget, so the decrease in the water load causes the relative inflow phosphorus concentration ( $P_i + P_r + P_{in}/Q_i$ ) to increase. In cases where internal loading is zero, such as dimictic lakes (see Osgood 1988c), the effect of reduced watershed loads is much greater, but still proportionately less than the reduction in the load.

The rates and magnitudes of internal phosphorus loading are highly variable (Bästrom et al. 1982; Nürnberg 1984). This variability occurs within lakes as well as between lakes, making it difficult to estimate the magnitude of the internal load. It is possible, however, to estimate the potential availability of internal phosphorus supplies during the summer based on the mixing characteristic of lakes (Osgood 1988c). The availability of internal phosphorus to the summertime epilimnia of the test lakes ranges from 0 to 97% of their budgets (Table 3). Clearly, the potential impact of estimation error is large. Since there are no direct field observations and no probability functions available which define this potential error, the sensitivity of the model cannot be fully evaluated. On the other hand, other general analyses of the effects of internal phosphorus loading impacts in TCMA lakes (Osgood 1988c) as well as the analyses of the test lakes presented here (Table 4), indicates that the presumed rates of internal loading are reasonable estimates.

The initial phosphorus concentration appears to have a large effect on the outcome of the model simulations. This is a concern because  $P_i$  appears to be relatively low and constant from lake-to-lake and from year-to-year, and perhaps independently of external events. The extent to which  $P_i$  affects the summertime phosphorus concentration as well as the ability to predict or manage  $P_i$  must be assessed.

The summertime phosphorus concentration is affected by changes in the initial phosphorus concentrations as follows (from Eqs. 1 and 2):

$$P = 0.675 P_i + \text{constant} \quad (5)$$

when the other input variables to Eq. 1 are held constant. The constant in the above equation is the predicted phosphorus concentration from Eqs. 1 and 2 when  $P_i = 0$ . The effect of uncertain or variable  $P_i$  compared to the average summertime phosphorus concentration decreases with  $P$ . For example, assuming that  $P_i$  is normally varies from  $25 \text{ mg} \cdot \text{m}^{-3} \pm 15 \text{ mg} \cdot \text{m}^{-3}$ , then the range of predicted  $P$  is  $20 \text{ mg} \cdot \text{m}^{-3}$ , which is a decreasing percentage of  $P$  as follows:

<u>P at P<sub>i</sub>=25 mg·m<sup>-3</sup></u>	<u>RANGE AS % OF P</u>
10	202
20	101
30	68
40	51
50	41
100	20
200	10
300	7
400	5

indicating a greater sensitivity to changes in  $P_i$  when the summertime phosphorus concentrations are  $< 50 - 100 \text{ mg} \cdot \text{m}^{-3}$ , which is very common in TCMA lakes (see Osgood 1988b). Examples of this effect are clearly seen by examining Tables 3 and 4. Lakes with low summertime phosphorus concentrations (eg. Square and Elmo) have  $P$ 's that vary with  $P_i$  despite the year-to-year changes in the other inputs. In the more enriched lakes (eg. Crystal and Spring), this effect is much less apparent.

The effect of variable  $P_i$  has confounded the interpretation of Lake McCarron's response to reduced loading (Oberts and Osgood 1988). A re-examination of that case indicates that the lake's increased summertime phosphorus concentration following the reduction of external loading may be related to the increased  $P_i$  observed in 1988 (Table 3). Lake McCarron's early-summer phosphorus concentration normally falls to very low levels ( $< 30 \text{ mg} \cdot \text{m}^{-3}$ ); however, it reaches this low level at different times during the summer. For example, the minimum observed surface phosphorus concentrations and their dates are as follows:

<u>YEAR</u>	<u>P ON JUNE 1</u>	<u>MINIMUM P</u>	<u>DATE</u>
1984	65	20	6/27
1985	25	25	5/23
1986	20	20	6/4
1987	60	30	6/25
1988	60	30	9/7

The initial phosphorus concentrations in 1985 and 1986 were low and the model accurately predicts the observed summertime conditions (Table 4). When  $P_i$  is greater on June 1, but still declining, the ~~model~~ poorly simulates the observed summertime conditions in the lake (eg. 1984 and 1987). However, the phosphorus concentration does reach similar low levels during June (see above); when these low levels are used in the model, the predicted summertime concentration is in close agreement with the observed conditions (Tables 3 and 4). During 1988, the phosphorus concentration did not reach these low levels until September and the summertime average phosphorus concentration was high despite a substantial reduction in loading (Oberts and Osgood 1988). It appears that the June 1 phosphorus concentration relates to the loss of phosphorus through the lake's outlet. Fortunately, this has been measured at Lake McCarron and is related to  $P_i$  such that when the springtime outflow phosphorus load is large,  $P_i$  is low (Oberts and Osgood 1988):

<u>YEAR</u>	<u>P<sub>i</sub></u>	<u>APRIL-MAY P OUTFLOW (KG)</u>
1984	65	7
1985	25	9
1986	20	24
1987	50	0
1988	60	0

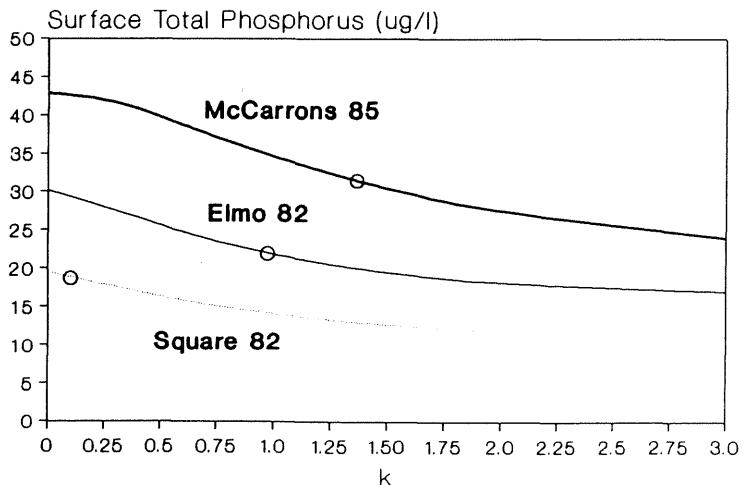
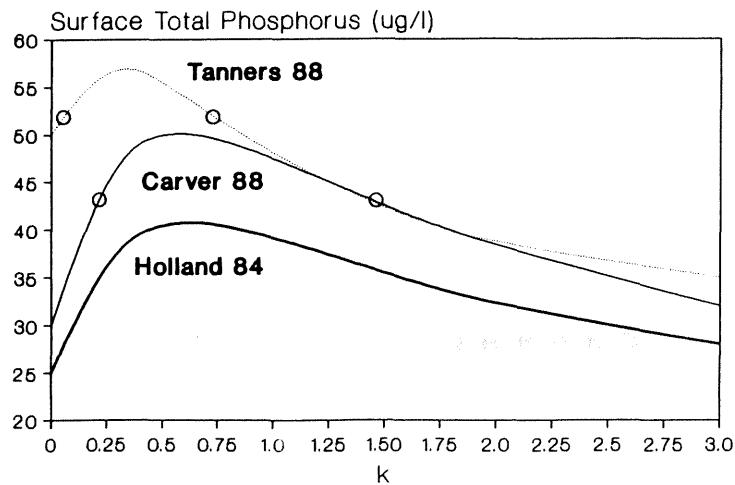
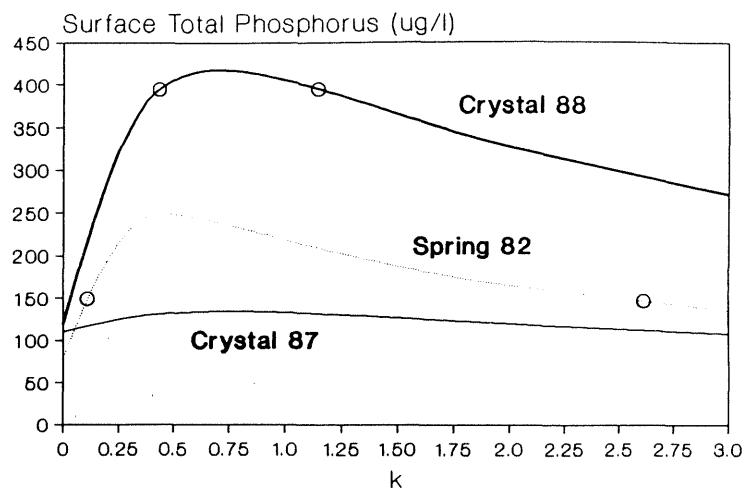
Unfortunately, outflow phosphorus loads have not been measured in other lakes, so the general effect cannot be quantified. Nonetheless, the effect, increased phosphorus loss with increased pre-summer outflow, is probably generally applicable. More significantly, this effect is probably not manageable because its magnitude is inversely related to the intensity of pre-summer runoff events. The best that can be expected when using the model to simulate future conditions is that the normal range of  $P_i$  for lakes can be defined.

Since the first-order reaction coefficient,  $k$ , is a mathematical construct describing the transformation of epilimnetic phosphorus, its physical significance is unclear. It describes the cumulative loss or transformation of phosphorus from the summertime epilimnia as a first-order reaction. The actual loss of phosphorus from the epilimnion has not been measured in this study, but is apparently highly variable in lakes both spatially and seasonally (Canfield et al. 1982). Therefore, it is not possible to further describe the rates or mechanisms of phosphorus loss.

The model's sensitivity to  $k$  can be evaluated by holding all of the input parameters to several of the test lakes constant while varying the value of  $k$ . Values of  $k$  that are less than zero have no physical meaning because that allows  $P_i$  to increase through the summer with no other inputs of phosphorus (Eq. 1). Thus,  $k < 0$  will not be considered. Also,  $k > 3$  indicates that >95% of  $P_i$  will be lost during the summer (Eq. 1). Since greater losses are not likely,  $k = 3$  will be the upper limit to be evaluated. The results of the model's simulations using variable values of  $k$  are shown in Fig. 5. There are two general types of curves that result: lakes with no summertime outflows where  $P = P_i$  at  $k = 0$  and lakes with outflows (Elmo, McCarrons and Square). The first type is a unimodal curve with two potential points of intersection with the observed and simulated  $P$ . The second type of curve descends in value through the range of  $k$  values used here, thus has only one point of potential intersection with the observed  $P$ . These intersection points are indicated in Figure 5. The points occurring on the ascending portions of the curves correspond to values of  $k = 0.2$  (range: 0.1-0.35) while the points occurring on the descending portions of the curves correspond to values of  $k = 1.09$  (range: 0.75-2.35). The descending portions of the unimodal curves are more gently sloping, and values of  $k$  on the second type of curve approximates the descending  $k$  values in the other curves, except for Square Lake. Since  $k = 1.05$  agrees with the descending value of  $k$  in Figure 5 and since using this value in the model adequately simulates the observed phosphorus concentrations (Table 4), it is an appropriate value. Further, this analysis shows that the model is relatively insensitive to changes in  $k$  in this range.

**Figure 5**

**PLOTS OF THE PREDICTED SUMMERTIME PHOSPHORUS CONCENTRATIONS USING INPUT PARAMETERS FROM TABLE 3 IN THE TEXT WITH VARYING VALUES OF K. POINTS ON THE CURVES INDICATE THE OBSERVED SUMMERTIME PHOSPHORUS CONCENTRATIONS IN THE LAKES.**



The form of the model described above (Eqs. 1 and 2) describes the summertime epilimnetic phosphorus dynamics in TCMA lakes and allows the evaluation of various watershed and lake management practices in terms of their impacts on lakes' average summertime phosphorus concentrations. The lakes tested represent a cross-section of TCMA lakes as well as a suitable selection of management practices or measurable changes in environmental conditions. The extent to which internal changes to the lake systems, that is the constants used in the model ( $V$ ,  $P_{int}$  and  $P_i$ ), are effected by the various management scenarios have been presumed to be zero. Further assessment should also compare changes in these internal constants relative to the long-term effectiveness of the management practices and will be presented in a later section. For now, the model appears to an adequate tool to evaluate the short-term effectiveness of a range of management scenarios on a cross-section of TCMA lakes.

## SHORT-TERM LAKE RESPONSES TO THE MANAGEMENT PRACTICES

The model will be used to evaluate the short-term responses of lakes to various management practices. Since 1988 was an abnormally dry year, the effectiveness of the management practices in the studied lakes will also be evaluated under normal hydrologic conditions. Following that analysis, the short-term effects of a suite of watershed and lake treatments will be simulated on a cross-section of TCMA lakes.

The effects of the various management practices on the six lakes as simulated using the time-dependent phosphorus model (Eqs. 1 and 2) are shown in Table 6. Table 6 does not show the change in the external load ( $P_r$ ) as in Table 3, rather it presents the overall summertime load to the epilimnion ( $W$ ). This illustrates: first, that due to internal ( $P_{int}$ ) and atmospheric loading ( $P_p$ ), the overall reductions that occur are relatively smaller than the treatment efficiencies of the monitored systems; and second, that the lakes' summertime P reductions are relatively smaller than the reductions in  $W$ . In particular, the three lakes with watershed treatments had phosphorus reductions ( $W$ ) of 13, 17% and Tanners Lake had a 13% increase, but their summertime P was reduced by only 8, 8% and 4% increase in Tanners for a normal year (see Table 6). The lakes with watershed diversions had larger reductions in phosphorus loading (78% for Lake Elmo and 16% for Holland Lake), but with only 29 and 6% reductions in their summertime P in a normal year. Finally, Crystal Lake's phosphorus load to the epilimnion increased substantially with the circulation. There was a 442% increase in the lake's summertime P with the circulation in 1988, or an estimated 197 - 363% increase in a normal year, depending on the lake's  $P_i$  (Table 6).

The final short-term analysis deals with the effects of extremely effective management practices on the summertime phosphorus concentration in TCMA lakes. The management practices include: 1. a 25% reduction in  $P_r$  with no reduction in  $Q_i$  to simulate watershed treatments; 2. a 50% reduction in  $P_r$  and  $Q_i$  to simulate watershed diversions, and; 3. setting  $V = 90\%$  of the lakes volume with no change in loading to simulate whole-lake circulation (see Table 7). These management scenarios include simulating the maximum phosphorus reductions that can reasonably be expected with general watershed-wide treatments and with whole-lake circulation. The watershed-wide treatments simulate phosphorus reductions through treatment and through diversion. These simulations were run on a cross-section of TCMA lakes (Table 7). The lakes range from dimictic (Holland and McCarrons) to polymictic (Spring and George) and internal loading ranges from 0 (George) to 97% (Spring) of the lakes' phosphorus budgets.

TABLE 6. Input parameters for the assessing the impacts of management practices on the summertime phosphorus concentration (P) in the test lakes using the time-dependent phosphorus model (Eqs 1 and 2 in the text).

	$Q_i$ ----- (10 <sup>6</sup> m <sup>3</sup> ) -----	$Q_o$	V	W (kg)	$P_i$ (mg·m <sup>-3</sup> )	P (mg·m <sup>-3</sup> )
<b>WATERSHED TREATMENTS</b>						
Carver						
1988 with	.261	0	.736	85.0	30	47.0
1988 w/o	.261	0	.736	89.2	30	48.3
Normal with	.366	.272	.736	95.7	30	50.5
Normal w/o	.366	.272	.736	109.5	30	55.1
McCarrons						
1988 with	.140	0	.853	42.2	60	53.7
1988 w/o	.155	0	.853	49.0	60	55.7
Normal with	.227	.660	.853	54.2	25	32.9
Normal w/o	.251	.090	.853	65.3	25	35.9
Tanners						
1988 with	.108	0	.858	40.7	50	46.9
1988 w/o	.108	0	.858	40.6	50	46.8
Normal with	.188	.038	.858	49.7	50	48.6
Normal w/o	.188	.038	.858	43.9	50	46.8
<b>WATERSHED DIVERSIONS</b>						
Elmo						
Normal with	.880	.01	5.00	50.6	25	19.6
Normal w/o	1.76	.89	5.00	226.6	25	27.6
Holland						
1988 with	.052	0	.324	20.6	17	28.5
1988 w/o	.066	0	.324	23.5	17	30.3
Normal with	.084	0	.324	23.8	25	35.1
Normal w/o	.107	0	.324	28.3	25	37.4
<b>CIRCULATION</b>						
Crystal						
1988 with	.343	0	.951	1330	120	403
1988 w/o	.343	0	.669	120	55	74.4
Normal with	.559	0	.951	1375	120	368
Normal w/o	.559	0	.669	165	55(120)	79.6 (124)

TABLE 7. Estimated input parameters for simulating three management scenarios on a cross-section of TCMA lakes using the time-dependent phosphorus model. The management scenarios are as follows: 1. A 25% reduction in  $P_r$  with no reduction in  $Q_i$  to simulate watershed treatments; 2. A 50% reduction in  $P_r$  and  $Q_i$  to simulate watershed diversions; and 3. To simulate whole-lake circulation,  $V = 90\%$  of the lake volume, but no change in  $P_{int}$ .

	$Q_i$ -----( $10^6 \text{ m}^3$ )----	$Q_o$	$V$	$P_p$	$P_r$	$P_{int}$	$P_i$ ( $\text{mg} \cdot \text{m}^{-3}$ )	$P$ ( $\text{mg} \cdot \text{m}^{-3}$ )
<b>George</b>								
1982	.640	0	4.50	7.7	31	0	20	15.8
1.	.640	0	4.50	7.7	23.2	0	20	15.4
2.	.583	0	4.50	7.7	15.5	0	20	14.9
3.	.640	0	4.78	7.7	31	0	20	15.7
<b>Holland</b>								
1988	.052	0	.324	0.6	4.3	15.7	17	28.5
1.	.052	0	.324	0.6	3.2	15.7	17	27.6
2.	.037	0	.324	0.6	2.1	15.7	17	27.3
3.	.052	0	.662	0.6	4.3	15.7	17	20.4
<b>McCarbons</b>								
1985	.327	.213	.853	2.3	36	23	25	34.1
1.	.327	.213	.853	2.3	27	23	25	31.5
2.	.270	.156	.853	2.3	18	23	25	29.3
3.	.327	.213	2.29	2.3	36	23	25	24.1
<b>Medicine</b>								
1983	5.11	3.29	15.0	30	556	1095	20	40.7
1.	5.11	3.29	15.0	30	417	1095	20	39.4
2.	3.31	1.49	15.0	30	278	1095	20	38.3
3.	5.11	3.29	17.2	30	556	1095	20	37.7
<b>Riley</b>								
1982	.487	.021	4.40	4.9	19.5	323	30	44.1
1.	.487	.021	4.40	4.9	14.6	323	30	42.0
2.	.370	0	4.40	4.9	9.7	323	30	42.3
3.	.487	.021	7.23	4.9	19.5	323	30	35.3
<b>Spring</b>								
1982	—	1.16	0	11.6	10	165	6462	80
1.	—	1.16	0	11.6	10	124	6462	80
2.	—	.684	0	11.6	10	82	6462	80
3.	—	1.16	0	(11.2)	10	165	6462	80
								(221)
<b>Stieger</b>								
1984	.510	.190	2.00	6.3	27.6	76.4	25	30.9
1.	.510	.190	2.00	6.3	20.7	76.4	25	30.0
2.	.510	.090	2.00	6.3	13.8	76.4	25	29.5
3.	.510	.190	2.42	6.3	27.6	76.4	25	28.8

The short-term effects of the watershed treatments on the lakes' summertime phosphorus loads are small compared to the reductions observed in the treatment systems. The reduction in W with the 25% reduction in P<sub>i</sub> ranged from 5-20% (median = 6%). The reduction in the summertime phosphorus concentration of these lakes was proportionately even less (range: 3-8%, median = 3%). The overall effect of the watershed diversion was greater, however, it was still small. The reduction in the summertime phosphorus loads (W) following the 50% diversion ranged from 5-40% (median = 13%), with a reduction in the lakes' P ranging from 0-14% (median = 5%). The greatest proportional reductions occurred in lakes with the smallest internal phosphorus budgets (George, McCarrons); however, that is atypical for TCMA lakes (Osgood 1988c).

The simulation of the whole-lake circulation shows reductions in the lakes' P (Table 7), contrary to the observations from Crystal Lake (Table 2a). The largest reductions occur in the most strongly stratified lakes and no reductions occur in the polymictic lakes, which normally circulate during the summer. Intermediate mixis lakes, such as Medicine and Stieger, show 7% reductions in P. Since whole-lake circulation is generally not used to reduce lakes' phosphorus content (Cooke et al. 1986), this exercise may be superfluous. The large reductions in the dimictic lakes are due to dilution, however, the energy requirements to circulate strongly stratified lakes probably renders this technique infeasible. Also, this simulation assumed that internal phosphorus loading to the mixing zone and P<sub>i</sub> would not increase following the circulation. This is not what occurred in Crystal Lake (Table 2a), but there is inadequate information available to generalize the impacts of circulation-induced increases in internal phosphorus loading. It appears that whole-lake circulation is not a practical technique to reduce lakes' summertime phosphorus concentration in TCMA lakes.

#### LONG-TERM LAKE RESPONSES TO PHOSPHORUS REDUCTIONS

The long-term affects of watershed management are more difficult to evaluate. The consideration of annual nutrient and water loading using the steady-state models yields results that conflict with field observations. However, it is instructive to use a steady-state model to simulate changes in lake phosphorus concentration following watershed treatments and compare the results with the seasonal analyses that have been developed here.

A steady-state solution to the input-output model for lakes is as follows (after Chapra and Reckhow 1983):

$$P = W / (Q + kV) \quad (6)$$

where P is the steady-state phosphorus concentration ( $\text{mg}\cdot\text{m}^{-3}$ ), W is the annual dissolved phosphorus loading ( $\text{mg}\cdot\text{year}^{-1}$ ), Q is the annual water input ( $\text{m}^3\cdot\text{year}^{-1}$ ), V is the lake volume ( $\text{m}^3$ ) and k = 1.05. Notice that the parameters are the same as in Eq. 1, except that here they are annual values.

When Eq. 6 is used to simulate the effects of watershed changes similar to those in Table 7, it appears that the lakes are more responsive to these changes on an annual basis (Table 8). The results of this simulation on the same cross-section of lakes indicates that a 25% annual reduction of dissolved phosphorus loading yields an overall reduction in loading to

the lake of 12% (median; range: 3 - 19%) with a similar reduction in the lake's phosphorus concentration (median = 12%; range: 3 - 19%). The results of the simulated diversion (50% reduction in water and phosphorus loading) indicates an overall reduction in loading to the lake of 26% (median; range: 7 - 38%) with a lower reduction in the lake's phosphorus concentration (median = 19%; range: 0 - 36%). As with the seasonal simulations, internally loaded phosphorus accounts for the lowered overall reductions (Table 8). A time-dependent model analogous to Eq. 1, except with annual loading rates and the whole-lake volumes substituted, indicates that the lakes in Table 8 would achieve 90% of their lowered steady-state P within two years of the reductions in loading. However, neither result of the simulation using annual data, the magnitude and speed of P reduction, has been observed in TCMA lakes (eg. Lake McCarrons, Oberts and Osgood 1988).

TABLE 8. Estimated annual input parameters for simulating the impacts of three management scenarios on the annual P in a cross-section of TCMA lakes using the steady-state phosphorus model (Eq. 6 in the text). The management scenarios are as follows: 1. A 25% reduction in  $P_r$  with no reduction in  $Q_i$  to simulate watershed treatments; and 2. A 50% reduction in  $P_r$  and  $Q_i$  to simulate watershed diversions.

	$Q_i$ ----	$Q_o$ $(10^6 \text{ m}^3)$ ----	V	$P_p$ ----	$P_r$ $(\text{kg})$ ----	$P_{int}$	P $(\text{mg} \cdot \text{m}^{-3})$
George							
1982	1.73	0	5.31	26	84	0	15.1
1.	1.73	0	5.31	26	63	0	12.2
2.	1.52	0	5.31	26	42	0	9.6
Holland							
1983	.23	0	.736	3.0	16.4	15.7	35.0
1.	.23	0	.736	3.0	12.3	15.7	30.9
2.	.17	0	.736	3.0	8.2	15.7	28.5
McCarrons							
1985	.69	.66	2.55	5	82	23	32.7
1.	.69	.66	2.55	5	62	23	26.7
2.	.48	.45	2.55	5	41	23	21.9
Medicine							
1983	8.79	6.93	19.1	53	1228	1095	82
1.	8.79	6.93	19.1	53	921	1095	72
2.	5.72	3.86	19.1	53	614	1095	68
Riley							
1982	2.44	1.25	8.03	40	360	323	66.5
1.	2.44	1.25	8.03	40	270	323	58.2
2.	1.65	0.46	8.03	40	180	323	53.9
Spring							
1982	7.7	6.5	12.4	38	1064	6462	365
1.	7.7	6.5	12.4	38	799	6462	352
2.	7.7	6.5	12.4	38	532	6462	395
Stieger							
1984	.74	.42	2.69	9.2	42.0	76.4	35.8
1.	.74	.42	2.69	9.2	31.5	76.4	32.9
2.	.60	.28	2.69	9.2	21.0	76.4	31.1

The explanation for the lack of observable lake responses following real changes in external loading probably deals with the magnitude of internal loading in TCMA lakes (Osgood 1988c) which reduces the effect of external load reductions. In addition, the differential effect of seasonal loading relative to lakes' responsiveness, which has been illustrated with the seasonal time-dependent phosphorus model (Eqs. 1 and 2), reduces the apparent effects of the annual load reductions. The above analysis using annual loading estimates and a steady-state model highlights the lack of validity of the steady-state assumptions relative to the behavior of TCMA lakes. It has been shown earlier in this report, that lakes are more sensitive to changes in external phosphorus that occur during the summertime. Those analyses indicate that substantial, long-term reductions in the summertime P of TCMA lakes will require reductions in the initial summertime phosphorus concentration ( $P_i$ ) and in the internal phosphorus loading ( $P_{int}$ ) as well as reductions in the external loading of dissolved phosphorus during the summer.

It appears that the year-to-year variation of  $P_i$  in TCMA lakes is unrelated to large changes in watershed loading (see Oberts and Osgood 1988). Therefore, long-term reductions in external phosphorus loading does not appear to affect  $P_i$ . This is further supported by the observation that  $P_i$  in most TCMA lakes occurs within a narrow range. There is no evidence to suggest that watershed management will lead to long-term changes in the initial summertime phosphorus concentration of TCMA lakes.

Long-term reductions in the summertime phosphorus in TCMA lakes must result from the combination of sustained reductions in the external phosphorus load to the lakes and the reduction in the rate of internally supplied phosphorus. Unfortunately, we have data on neither the long-term effectiveness of watershed management systems nor the decay of internal phosphorus supplies following reduced external loads. In order to expect any long-term improvements in lake quality to occur, the length of effectiveness of watershed treatments must exceed the time necessary for the internal phosphorus sources to become substantially depleted. The depletion of phosphorus available in the lakes' sediments will occur only after the annual accumulation falls below some critical level, above which net accumulation still occurs. Since lakes in the TCMA retain  $\approx 90\%$  of the annual external total phosphorus loads (Osgood 1981), very large reductions in external loads are probably required in order to deplete the pool of available phosphorus in the lakes' sediments. Based upon our observations of the Lake McCarrons Wetland Treatment System (Oberts and Osgood 1988), it appears that without intensive maintenance of watershed treatment systems, that the depletion of lakes' internal phosphorus supplies will occur on time scales much longer than the normal effectiveness of the watershed systems. Thus, the time-dependent phosphorus model presented here adequately predicts the response of TCMA lakes to nutrient load modifications that apply for the duration of reasonable planning time scales.

## SUMMARY OF PHOSPHORUS MODELLING

The modelling presented above shows TCMA lakes to be generally insensitive to changes in external phosphorus inputs, especially during the summer. It appears that substantial changes in the lakes' summertime phosphorus concentrations are unlikely to result following commonly-used watershed management techniques. Furthermore, due to the predominance of internal phosphorus loading to the lakes' summertime phosphorus budgets, it appears that the lakes are unlikely to improve over the long-term following annual reductions in phosphorus

loading. The time-dependent phosphorus model, which has been shown to be appropriate for simulating the average summertime phosphorus concentrations in TCMA lakes in response to short-term changes in watershed loadings, probably also adequately simulates long-term impacts of watershed changes. Substantial, long-term reductions in the lakes' phosphorus concentration requires sustained reductions in the external phosphorus load to the lakes as well as the attenuation of the internal phosphorus supply.

The modelling presented here has been shown to simulate the response of TCMA lakes to changes in phosphorus income and is useful for understanding the general behavior of the lakes. The model can be used for individual lake analysis, provided equally "good" input parameters are available. Preliminary tests of the model using gross estimates of inputs provided poor results. It appears that several years' data from the lake and its watershed are necessary to provide "good" inputs to the model.

## RECOMMENDATIONS FOR LAKE AND WATERSHED MANAGEMENT

The TCMA lake managers should plan to mitigate the impacts of nonpoint source pollution to the fullest possible extent, consider the use of in-lake management techniques to treat symptomatic nuisances in lakes and consider the long-term economic and environmental costs associated with both practices when planning development in the TCMA.

It has been shown that watershed treatments are generally ineffective with respect to reducing TCMA lakes' summertime phosphorus concentrations. Since land development also affects the phosphorus loading to lakes, the effectiveness of watershed treatments to mitigate the impacts of development is compromised. Thus, the common practice of requiring watershed treatments for new developments will probably lead to a net degradation of the lakes. The lakes will become more eutrophic; however, based on the analyses included here and observations over the past decade, this process is occurring slowly. It appears that the only watershed treatments that have the potential for reducing lake phosphorus are runoff diversions, which are seldom feasible.

The discontinuation of the watershed treatments and wetland preservation is not being recommended here. Rather, it is recommended that the impacts of nonpoint source pollution should be mitigated to the fullest possible extent in order to minimize the impacts of land development on lake water quality. In fact, watershed treatments and wetland preservation should occur more commonly in order to best address the long-term impacts on lakes. These systems provide plenty of other benefits which justify their continued use. In addition, they provide enough treatment of phosphorus, with proper maintenance, to at least partially mitigate the impacts of development (Oberts and Osgood 1988; Oberts et al. 1989).

Presuming that wholesale changes in the lakes' summertime phosphorus are not likely to occur, what approach is left for the management of eutrophication-related aquatic nuisances? The in-lake treatment of symptoms, rather than the underlying causes, is perhaps philosophically unpleasant, but may be the best remaining alternative available to TCMA lake managers. It is recommended that in-lake management techniques be considered to treat nuisances related to eutrophication only after the most feasible watershed treatments have been included as part of the lake management plan. In-lake treatments are inherently energy-demanding. They may lead to observable changes in lakes; however, these changes, while they may mimic improvements associated with lowered nutrient concentrations, are fundamentally different (see Lehman 1988). The discussion of specific in-lake management techniques will not be included here.

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## APPENDIX A

### Physical/Chemical Data

<u>Parameter Code</u>	<u>Parameter</u>	<u>Units</u>
DM	Depth	m
TC	Temperature	°C
DO	Dissolved Oxygen	g·m <sup>-3</sup>
LIGHT	Underwater Light	µE·m <sup>-2</sup> ·s <sup>-1</sup>
TP	Total Phosphorus	mg·m <sup>-3</sup>
TDP	Total Dissolved Phosphorus	mg·m <sup>-3</sup>
OP	Ortho-phosphorus	mg·m <sup>-3</sup>
TKN	Total Kjeldahl Nitrogen	g·m <sup>-3</sup>
N23	Nitrate-Nitrite-Nitrogen	g·m <sup>-3</sup>
CLA	Chlorophyll a	mg·m <sup>-3</sup>
SDM	Secchi Disk	m
COND	Specific conductance	µmho·m <sup>-2</sup> ·s <sup>-1</sup>
PHL	pH	standard units

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9:17 FRIDAY, FEBRUARY 24, 1989

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9:17 FRIDAY, FEBRUARY 24, 1989

4

OBS	LAKE	LAKENAME	LAKEID	DATE	TIME	DM	TC	DO	LIGHT	TP	TDP	OP	TKN	N23	CLA	SDM	COND	PHL
221	CARV	CARVER	82-0166	880906	935	6.5	9.9	.	.	.	.	.	.	.	.	.	.	.
222	CARV	CARVER	82-0166	880906	935	7.0	8.6	1.00	.	.	.	.	.	.	.	.	.	.
223	CARV	CARVER	82-0166	880906	935	7.5	7.6	.	.	.	.	.	.	.	.	.	.	.
224	CARV	CARVER	82-0166	880906	935	8.0	7.1	1.00	.	.	.	.	.	.	.	.	.	.
225	CARV	CARVER	82-0166	880906	935	8.5	6.8	.	.	.	.	.	.	.	.	.	.	.
226	CARV	CARVER	82-0166	880906	935	9.0	6.6	0.90	.	640	.	.	.	.	.	.	615	.
227	CARV	CARVER	82-0166	880921	950	0.0	16.6	5.60	156.00	55	30	8	1.390	0.100	11.0	1.50	350	.
228	CARV	CARVER	82-0166	880921	950	0.5	.	.	77.00	.	.	.	.	.	.	.	.	.
229	CARV	CARVER	82-0166	880921	950	1.0	16.6	5.60	34.50	.	.	.	.	.	.	.	.	.
230	CARV	CARVER	82-0166	880921	950	1.5	.	.	18.40	.	.	.	.	.	.	.	.	.
231	CARV	CARVER	82-0166	880921	950	2.0	16.6	5.60	9.40	.	.	.	.	.	.	.	.	.
232	CARV	CARVER	82-0166	880921	950	2.5	.	.	5.40	.	.	.	.	.	.	.	.	.
233	CARV	CARVER	82-0166	880921	950	3.0	16.6	5.60	3.10	70	.	.	.	.	.	.	350	.
234	CARV	CARVER	82-0166	880921	950	3.5	.	.	1.65	.	.	.	.	.	.	.	.	.
235	CARV	CARVER	82-0166	880921	950	4.0	16.6	5.60	0.99	.	.	.	.	.	.	.	.	.
236	CARV	CARVER	82-0166	880921	950	5.0	16.6	5.30	.	.	.	.	.	.	.	.	.	.
237	CARV	CARVER	82-0166	880921	950	6.0	16.0	2.70	.	70	.	.	.	.	.	.	350	.
238	CARV	CARVER	82-0166	880921	950	6.5	12.2	.	.	.	.	.	.	.	.	.	.	.
239	CARV	CARVER	82-0166	880921	950	7.0	9.7	0.70	.	.	.	.	.	.	.	.	.	.
240	CARV	CARVER	82-0166	880921	950	7.5	8.3	.	.	.	.	.	.	.	.	.	.	.
241	CARV	CARVER	82-0166	880921	950	8.0	7.6	1.00	.	.	.	.	.	.	.	.	.	.
242	CARV	CARVER	82-0166	880921	950	9.0	7.1	1.10	.	670	.	.	.	.	.	.	585	.
243	CARV	CARVER	82-0166	881004	940	0.0	13.8	8.80	850.00	65	30	12	1.460	0.100	44.0	1.30	360	8.26
244	CARV	CARVER	82-0166	881004	940	0.5	.	.	265.00	.	.	.	.	.	.	.	.	.
245	CARV	CARVER	82-0166	881004	940	1.0	13.9	8.70	83.00	.	.	.	.	.	.	.	.	.
246	CARV	CARVER	82-0166	881004	940	1.5	.	.	33.00	.	.	.	.	.	.	.	.	.
247	CARV	CARVER	82-0166	881004	940	2.0	13.9	8.60	15.00	.	.	.	.	.	.	.	.	.
248	CARV	CARVER	82-0166	881004	940	2.5	.	.	7.10	.	.	.	.	.	.	.	.	.
249	CARV	CARVER	82-0166	881004	940	3.0	13.9	8.60	3.40	70	.	.	.	.	.	.	355	8.28
250	CARV	CARVER	82-0166	881004	940	3.5	.	.	1.60	.	.	.	.	.	.	.	.	.
251	CARV	CARVER	82-0166	881004	940	4.0	13.9	8.60	.	.	.	.	.	.	.	.	.	.
252	CARV	CARVER	82-0166	881004	940	5.0	13.8	8.50	.	.	.	.	.	.	.	.	.	.
253	CARV	CARVER	82-0166	881004	940	6.0	13.7	8.40	.	60	.	.	.	.	.	.	360	8.28
254	CARV	CARVER	82-0166	881004	940	6.5	13.5	.	.	.	.	.	.	.	.	.	.	.
255	CARV	CARVER	82-0166	881004	940	7.0	12.8	3.20	.	.	.	.	.	.	.	.	.	.
256	CARV	CARVER	82-0166	881004	940	7.5	8.9	.	.	.	.	.	.	.	.	.	.	.
257	CARV	CARVER	82-0166	881004	940	8.0	7.6	1.10	.	.	.	.	.	.	.	.	.	.
258	CARV	CARVER	82-0166	881004	940	8.5	7.4	.	.	.	.	.	.	.	.	.	.	.
259	CARV	CARVER	82-0166	881004	940	9.0	7.1	1.00	.	740	.	.	.	.	.	.	565	7.10
260	CARV	CARVER	82-0166	881018	930	0.0	11.6	8.90	218.00	80	30	12	1.290	0.150	34.0	1.30	360	8.66
261	CARV	CARVER	82-0166	881018	930	0.5	.	.	72.00	.	.	.	.	.	.	.	.	.
262	CARV	CARVER	82-0166	881018	930	1.0	11.6	8.90	25.40	.	.	.	.	.	.	.	.	.
263	CARV	CARVER	82-0166	881018	930	1.5	.	.	12.00	.	.	.	.	.	.	.	.	.
264	CARV	CARVER	82-0166	881018	930	2.0	11.6	8.80	5.50	.	.	.	.	.	.	.	.	.
265	CARV	CARVER	82-0166	881018	930	2.5	.	.	2.80	.	.	.	.	.	.	.	.	.
266	CARV	CARVER	82-0166	881018	930	3.0	11.6	8.70	1.36	40	.	.	.	.	.	.	360	8.66
267	CARV	CARVER	82-0166	881018	930	4.0	11.6	8.90	.	.	.	.	.	.	.	.	.	.
268	CARV	CARVER	82-0166	881018	930	5.0	11.6	8.60	.	.	.	.	.	.	.	.	.	.
269	CARV	CARVER	82-0166	881018	930	6.0	11.5	8.70	.	.	.	.	.	.	.	355	8.67	.
270	CARV	CARVER	82-0166	881018	930	7.0	11.2	7.90	.	50	.	.	.	.	.	.	.	.
271	CARV	CARVER	82-0166	881018	930	7.5	9.6	.	.	.	.	.	.	.	.	.	.	.
272	CARV	CARVER	82-0166	881018	930	8.0	8.4	0.80	.	.	.	.	.	.	.	.	.	.
273	CARV	CARVER	82-0166	881018	930	8.5	7.2	.	.	.	.	.	.	.	.	.	.	.
274	CARV	CARVER	82-0166	881018	930	9.0	6.8	0.80	.	1180	.	2	1.050	0.100	56.0	0.65	595	7.02
275	CRSR	CRYSTAL(ROBINSDALE)	27-0034	871028	840	0.0	6.5	10.00	.	85	20	2	1.050	0.100	56.0	0.65	310	8.22

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9:17 FRIDAY, FEBRUARY 24, 1989







OBS	LAKE	LAKENAME	LAKEID	DATE	TIME	DM	TC	DO	LIGHT	TP	TDP	OP	TKN	N23	CLA	SDM	COND	PHL
496	ELMO	ELMO	82-0106	871028	955	6.0	8.0	8.40	.	.	.	.	.	.	.	.	315	8.11
497	ELMO	ELMO	82-0106	871028	955	8.0	8.0	8.40	.	20	.	.	.	.	.	.	.	.
498	ELMO	ELMO	82-0106	871028	955	10.0	8.0	8.30	.	.	.	.	.	.	.	.	.	.
499	ELMO	ELMO	82-0106	871028	955	12.0	8.0	8.30	.	.	.	.	.	.	.	.	.	.
500	ELMO	ELMO	82-0106	871028	955	14.0	8.0	8.10	.	.	.	.	.	.	.	.	.	.
501	ELMO	ELMO	82-0106	871028	955	16.0	7.5	7.30	.	20	.	.	.	.	.	315	8.07	.
502	ELMO	ELMO	82-0106	871028	955	17.0	7.0	4.60	.	.	.	.	.	.	.	.	.	.
503	ELMO	ELMO	82-0106	871028	955	18.0	6.5	1.40	.	.	.	.	.	.	.	.	.	.
504	ELMO	ELMO	82-0106	871028	955	19.0	5.5	1.10	.	.	.	.	.	.	.	.	.	.
505	ELMO	ELMO	82-0106	871028	955	20.0	5.5	0.90	.	.	.	.	.	.	.	.	.	.
506	ELMO	ELMO	82-0106	871028	955	22.0	5.5	0.80	.	.	.	.	.	.	.	.	.	.
507	ELMO	ELMO	82-0106	871028	955	24.0	5.5	0.70	.	60	.	.	.	.	345	7.52	.	
508	ELMO	ELMO	82-0106	871028	955	26.0	5.5	0.60	.	.	.	.	.	.	.	.	.	.
509	ELMO	ELMO	82-0106	871028	955	28.0	5.5	0.50	.	.	.	.	.	.	.	.	.	.
510	ELMO	ELMO	82-0106	871028	955	30.0	5.5	0.50	.	.	.	.	.	.	.	.	.	.
511	ELMO	ELMO	82-0106	871028	955	32.0	.	.	.	70	.	.	.	.	345	7.50	.	
512	ELMO	ELMO	82-0106	880121	1000	0.0	.	.	.	20	15	5	0.500	0.250	5.2	2.80	.	.
513	ELMO	ELMO	82-0106	880121	1000	1.0	2.0	10.20	.	.	.	.	.	.	.	.	.	.
514	ELMO	ELMO	82-0106	880121	1000	8.0	2.5	9.70	.	20	.	.	.	.	.	.	.	.
515	ELMO	ELMO	82-0106	880121	1000	16.0	2.5	9.45	.	20	.	.	.	.	.	.	.	.
516	ELMO	ELMO	82-0106	880121	1000	24.0	2.5	9.15	.	20	.	.	.	.	.	.	.	.
517	ELMO	ELMO	82-0106	880121	1000	32.0	3.0	8.85	.	20	.	.	.	.	.	.	.	.
518	ELMO	ELMO	82-0106	880218	945	0.0	.	.	.	20	30	22	0.450	0.340	3.3	4.40	.	.
519	ELMO	ELMO	82-0106	880218	945	1.0	2.0	9.20	.	.	.	.	.	.	.	.	.	.
520	ELMO	ELMO	82-0106	880218	945	8.0	2.5	8.25	.	20	.	.	.	.	.	.	.	.
521	ELMO	ELMO	82-0106	880218	945	16.0	2.5	8.25	.	10	.	.	.	.	.	.	.	.
522	ELMO	ELMO	82-0106	880218	945	24.0	2.5	8.00	.	10	.	.	.	.	.	.	.	.
523	ELMO	ELMO	82-0106	880218	945	32.0	2.5	6.35	.	20	.	.	.	.	.	.	.	.
524	ELMO	ELMO	82-0106	880324	930	0.0	.	.	.	35	45	3	0.450	0.290	6.6	5.10	.	.
525	ELMO	ELMO	82-0106	880324	930	1.0	3.5	10.25	.	.	.	.	.	.	.	.	.	.
526	ELMO	ELMO	82-0106	880324	930	8.0	3.5	9.30	.	20	.	.	.	.	.	.	.	.
527	ELMO	ELMO	82-0106	880324	930	16.0	3.5	9.20	.	10	.	.	.	.	.	.	.	.
528	ELMO	ELMO	82-0106	880324	930	24.0	3.0	8.45	.	20	.	.	.	.	.	.	.	.
529	ELMO	ELMO	82-0106	880324	930	32.0	3.0	5.25	.	20	.	.	.	.	.	.	.	.
530	ELMO	ELMO	82-0106	880411	1000	0.0	6.3	10.70	1160.00	25	25	5	0.580	0.290	6.0	3.90	350	7.91
531	ELMO	ELMO	82-0106	880411	1000	0.5	.	.	830.00	.	.	.	.	.	.	.	.	.
532	ELMO	ELMO	82-0106	880411	1000	1.0	.	.	565.00	.	.	.	.	.	.	.	.	.
533	ELMO	ELMO	82-0106	880411	1000	1.5	.	.	420.00	.	.	.	.	.	.	.	.	.
534	ELMO	ELMO	82-0106	880411	1000	2.0	6.0	10.70	330.00	.	.	.	.	.	.	.	.	.
535	ELMO	ELMO	82-0106	880411	1000	2.5	.	.	243.00	.	.	.	.	.	.	.	.	.
536	ELMO	ELMO	82-0106	880411	1000	3.0	.	.	178.00	.	.	.	.	.	.	.	.	.
537	ELMO	ELMO	82-0106	880411	1000	3.5	.	.	130.00	.	.	.	.	.	.	.	.	.
538	ELMO	ELMO	82-0106	880411	1000	4.0	5.8	10.50	104.00	.	.	.	.	.	.	.	.	.
539	ELMO	ELMO	82-0106	880411	1000	4.5	.	.	76.00	.	.	.	.	.	.	.	.	.
540	ELMO	ELMO	82-0106	880411	1000	5.0	.	.	59.00	.	.	.	.	.	.	.	.	.
541	ELMO	ELMO	82-0106	880411	1000	5.5	.	.	45.00	.	.	.	.	.	.	.	.	.
542	ELMO	ELMO	82-0106	880411	1000	6.0	5.7	10.60	35.00	.	.	.	.	.	.	.	.	.
543	ELMO	ELMO	82-0106	880411	1000	6.5	.	.	26.80	.	.	.	.	.	.	.	.	.
544	ELMO	ELMO	82-0106	880411	1000	7.0	.	.	20.00	.	.	.	.	.	.	.	.	.
545	ELMO	ELMO	82-0106	880411	1000	7.5	.	.	17.50	.	.	.	.	.	.	.	.	.
546	ELMO	ELMO	82-0106	880411	1000	8.0	5.6	10.50	13.70	20	.	.	.	.	.	.	.	.
547	ELMO	ELMO	82-0106	880411	1000	10.0	5.4	10.00	.	.	.	.	.	.	.	.	.	.
548	ELMO	ELMO	82-0106	880411	1000	12.0	5.2	10.10	.	.	.	.	.	.	.	.	.	.
549	ELMO	ELMO	82-0106	880411	1000	14.0	5.1	10.10	.	.	.	.	.	.	.	.	.	.
550	ELMO	ELMO	82-0106	880411	1000	16.0	5.0	10.10	.	40	.	.	.	.	350	7.81	.	

OBS	LAKE	LAKENAME	LAKEID	DATE	TIME	DM	TC	DO	LIGHT	TP	TDP	OP	TKN	N23	CLA	SDM	COND	PHL
551	ELMO	ELMO	82-0106	880411	1000	18.0	5.0	9.80	-	-	-	-	-	-	-	-	-	-
552	ELMO	ELMO	82-0106	880411	1000	20.0	4.9	9.20	-	-	-	-	-	-	-	-	-	-
553	ELMO	ELMO	82-0106	880411	1000	22.0	4.8	9.10	-	-	-	-	-	-	-	-	-	-
554	ELMO	ELMO	82-0106	880411	1000	24.0	4.8	9.10	-	50	-	-	-	-	-	-	350	7.77
555	ELMO	ELMO	82-0106	880411	1000	26.0	4.7	8.80	-	-	-	-	-	-	-	-	-	-
556	ELMO	ELMO	82-0106	880411	1000	28.0	4.6	8.70	-	-	-	-	-	-	-	-	-	-
557	ELMO	ELMO	82-0106	880411	1000	30.0	4.6	8.70	-	-	-	-	-	-	-	-	-	-
558	ELMO	ELMO	82-0106	880411	1000	32.0	-	-	-	30	-	-	-	-	-	-	350	7.69
559	ELMO	ELMO	82-0106	880421	1005	0.0	7.0	11.80	665.00	25	30	10	0.650	0.300	11.0	3.00	335	8.21
560	ELMO	ELMO	82-0106	880421	1005	0.5	-	-	390.00	-	-	-	-	-	-	-	-	-
561	ELMO	ELMO	82-0106	880421	1005	1.0	-	-	224.00	-	-	-	-	-	-	-	-	-
562	ELMO	ELMO	82-0106	880421	1005	1.5	-	-	170.00	-	-	-	-	-	-	-	-	-
563	ELMO	ELMO	82-0106	880421	1005	2.0	7.0	-	132.00	-	-	-	-	-	-	-	-	-
564	ELMO	ELMO	82-0106	880421	1005	2.5	-	-	110.00	-	-	-	-	-	-	-	-	-
565	ELMO	ELMO	82-0106	880421	1005	3.0	-	-	81.50	-	-	-	-	-	-	-	-	-
566	ELMO	ELMO	82-0106	880421	1005	3.5	-	-	66.00	-	-	-	-	-	-	-	-	-
567	ELMO	ELMO	82-0106	880421	1005	4.0	6.9	11.70	51.00	-	-	-	-	-	-	-	-	-
568	ELMO	ELMO	82-0106	880421	1005	4.5	-	-	38.50	-	-	-	-	-	-	-	-	-
569	ELMO	ELMO	82-0106	880421	1005	5.0	-	-	30.50	-	-	-	-	-	-	-	-	-
570	ELMO	ELMO	82-0106	880421	1005	5.5	-	-	23.20	-	-	-	-	-	-	-	-	-
571	ELMO	ELMO	82-0106	880421	1005	6.0	6.9	-	17.50	-	-	-	-	-	-	-	-	-
572	ELMO	ELMO	82-0106	880421	1005	6.5	-	-	13.80	-	-	-	-	-	-	-	-	-
573	ELMO	ELMO	82-0106	880421	1005	7.0	-	-	10.90	-	-	-	-	-	-	-	-	-
574	ELMO	ELMO	82-0106	880421	1005	7.5	-	-	9.30	-	-	-	-	-	-	-	-	-
575	ELMO	ELMO	82-0106	880421	1005	8.0	6.8	11.20	7.00	50	-	-	-	-	-	-	345	8.14
576	ELMO	ELMO	82-0106	880421	1005	8.5	-	-	5.50	-	-	-	-	-	-	-	-	-
577	ELMO	ELMO	82-0106	880421	1005	10.0	6.8	-	-	-	-	-	-	-	-	-	-	-
578	ELMO	ELMO	82-0106	880421	1005	12.0	6.7	11.10	-	-	-	-	-	-	-	-	-	-
579	ELMO	ELMO	82-0106	880421	1005	14.0	6.7	-	-	-	-	-	-	-	-	-	-	-
580	ELMO	ELMO	82-0106	880421	1005	16.0	6.6	10.80	-	50	-	-	-	-	-	-	345	8.08
581	ELMO	ELMO	82-0106	880421	1005	18.0	6.5	-	-	-	-	-	-	-	-	-	-	-
582	ELMO	ELMO	82-0106	880421	1005	20.0	6.1	10.50	-	-	-	-	-	-	-	-	-	-
583	ELMO	ELMO	82-0106	880421	1005	22.0	5.7	9.30	-	-	-	-	-	-	-	-	-	-
584	ELMO	ELMO	82-0106	880421	1005	24.0	5.4	8.70	-	10	-	-	-	-	-	-	345	7.92
585	ELMO	ELMO	82-0106	880421	1005	26.0	5.3	-	-	-	-	-	-	-	-	-	-	-
586	ELMO	ELMO	82-0106	880421	1005	28.0	5.2	8.40	-	-	-	-	-	-	-	-	-	-
587	ELMO	ELMO	82-0106	880421	1005	30.0	5.1	8.30	-	-	-	-	-	-	-	-	-	-
588	ELMO	ELMO	82-0106	880421	1005	32.0	-	-	-	20	-	-	-	-	-	-	355	7.78
589	ELMO	ELMO	82-0106	880504	1040	0.0	13.7	11.10	1330.00	40	15	8	0.600	0.250	3.9	4.60	340	8.34
590	ELMO	ELMO	82-0106	880504	1040	0.5	-	-	920.00	-	-	-	-	-	-	-	-	-
591	ELMO	ELMO	82-0106	880504	1040	1.0	13.5	-	690.00	-	-	-	-	-	-	-	-	-
592	ELMO	ELMO	82-0106	880504	1040	1.5	-	-	490.00	-	-	-	-	-	-	-	-	-
593	ELMO	ELMO	82-0106	880504	1040	2.0	13.1	11.30	410.00	-	-	-	-	-	-	-	-	-
594	ELMO	ELMO	82-0106	880504	1040	2.5	12.8	-	315.00	-	-	-	-	-	-	-	-	-
595	ELMO	ELMO	82-0106	880504	1040	3.0	11.7	12.10	255.00	-	-	-	-	-	-	-	-	-
596	ELMO	ELMO	82-0106	880504	1040	3.5	-	-	200.00	-	-	-	-	-	-	-	-	-
597	ELMO	ELMO	82-0106	880504	1040	4.0	11.0	12.10	157.00	-	-	-	-	-	-	-	-	-
598	ELMO	ELMO	82-0106	880504	1040	4.5	-	-	125.00	-	-	-	-	-	-	-	-	-
599	ELMO	ELMO	82-0106	880504	1040	5.0	10.4	-	102.00	-	-	-	-	-	-	-	-	-
600	ELMO	ELMO	82-0106	880504	1040	5.5	-	-	81.50	-	-	-	-	-	-	-	-	-
601	ELMO	ELMO	82-0106	880504	1040	6.0	9.9	11.80	64.50	-	-	-	-	-	-	-	-	-
602	ELMO	ELMO	82-0106	880504	1040	6.5	-	-	52.50	-	-	-	-	-	-	-	-	-
603	ELMO	ELMO	82-0106	880504	1040	7.0	9.4	-	41.50	-	-	-	-	-	-	-	-	-
604	ELMO	ELMO	82-0106	880504	1040	7.5	-	-	34.00	-	-	-	-	-	-	-	-	-
605	ELMO	ELMO	82-0106	880504	1040	8.0	8.8	11.50	28.00	10	-	-	-	-	-	-	340	8.28





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OBS	LAKE	LAKENAME	LAKEID	DATE	TIME	DM	TC	DO	LIGHT	TP	TDP	OP	TKN	N23	CLA	SDM	COND	PHL
991	ELMO	ELMO	82-0106	881019	950	4.0	11.8	8.60	38.50	.	.	.	.	.	.	.	.	.
992	ELMO	ELMO	82-0106	881019	950	4.5	.	.	30.50	.	.	.	.	.	.	.	.	.
993	ELMO	ELMO	82-0106	881019	950	5.0	11.8	8.60	23.30	.	.	.	.	.	.	.	.	.
994	ELMO	ELMO	82-0106	881019	950	5.5	.	.	17.90	.	.	.	.	.	.	.	.	.
995	ELMO	ELMO	82-0106	881019	950	6.0	11.8	8.50	13.50	.	.	.	.	.	.	.	.	.
996	ELMO	ELMO	82-0106	881019	950	6.5	.	.	10.90	.	.	.	.	.	.	.	.	.
997	ELMO	ELMO	82-0106	881019	950	7.0	11.7	8.60	8.50	.	.	.	.	.	.	.	.	.
998	ELMO	ELMO	82-0106	881019	950	7.5	.	.	6.40	.	.	.	.	.	.	.	.	.
999	ELMO	ELMO	82-0106	881019	950	8.0	11.7	8.70	5.15	10	.	.	.	.	.	.	320	8.38
1000	ELMO	ELMO	82-0106	881019	950	8.5	.	.	4.25	.	.	.	.	.	.	.	.	.
1001	ELMO	ELMO	82-0106	881019	950	9.0	11.7	8.60	.	.	.	.	.	.	.	.	.	.
1002	ELMO	ELMO	82-0106	881019	950	10.0	11.6	8.10	.	.	.	.	.	.	.	.	.	.
1003	ELMO	ELMO	82-0106	881019	950	11.0	11.6	8.30	.	.	.	.	.	.	.	.	.	.
1004	ELMO	ELMO	82-0106	881019	950	12.0	11.5	8.20	.	.	.	.	.	.	.	.	.	.
1005	ELMO	ELMO	82-0106	881019	950	13.0	10.9	7.20	.	.	.	.	.	.	.	.	.	.
1006	ELMO	ELMO	82-0106	881019	950	14.0	9.6	3.30	.	.	.	.	.	.	.	.	.	.
1007	ELMO	ELMO	82-0106	881019	950	15.0	7.9	1.10	.	.	.	.	.	.	.	.	.	.
1008	ELMO	ELMO	82-0106	881019	950	16.0	7.1	0.80	.	30	.	.	.	.	.	.	350	7.56
1009	ELMO	ELMO	82-0106	881019	950	17.0	6.5	0.80	.	.	.	.	.	.	.	.	.	.
1010	ELMO	ELMO	82-0106	881019	950	18.0	6.3	0.70	.	.	.	.	.	.	.	.	.	.
1011	ELMO	ELMO	82-0106	881019	950	19.0	6.2	0.70	.	.	.	.	.	.	.	.	.	.
1012	ELMO	ELMO	82-0106	881019	950	20.0	6.1	0.70	.	.	.	.	.	.	.	.	.	.
1013	ELMO	ELMO	82-0106	881019	950	22.0	6.0	0.70	.	.	.	.	.	.	.	.	.	.
1014	ELMO	ELMO	82-0106	881019	950	24.0	6.0	0.60	.	90	.	.	.	.	.	.	365	7.47
1015	ELMO	ELMO	82-0106	881019	950	26.0	6.0	0.60	.	.	.	.	.	.	.	.	.	.
1016	ELMO	ELMO	82-0106	881019	950	28.0	6.0	0.60	.	.	.	.	.	.	.	.	.	.
1017	ELMO	ELMO	82-0106	881019	950	30.0	6.0	0.60	.	.	.	.	.	.	.	.	360	7.47
1018	ELMO	ELMO	82-0106	881019	950	32.0	.	.	.	90	.	.	.	.	.	.	.	.
1019	HOLL	HOLLAND	19-0065	871029	825	0.0	6.5	7.00	.	20	5	2	1.300	0.100	5.4	2.40	240	7.85
1020	HOLL	HOLLAND	19-0065	871029	825	1.0	6.5	6.80	.	.	.	.	.	.	.	.	.	.
1021	HOLL	HOLLAND	19-0065	871029	825	2.0	6.5	6.60	.	.	.	.	.	.	.	.	.	.
1022	HOLL	HOLLAND	19-0065	871029	825	3.0	6.5	6.60	.	.	.	.	.	.	.	.	.	.
1023	HOLL	HOLLAND	19-0065	871029	825	4.0	6.5	6.40	.	.	.	.	.	.	.	.	235	7.84
1024	HOLL	HOLLAND	19-0065	871029	825	5.0	6.5	6.40	.	10	.	.	.	.	.	.	.	.
1025	HOLL	HOLLAND	19-0065	871029	825	6.0	6.5	6.30	.	.	.	.	.	.	.	.	.	.
1026	HOLL	HOLLAND	19-0065	871029	825	7.0	6.5	6.30	.	.	.	.	.	.	.	.	.	.
1027	HOLL	HOLLAND	19-0065	871029	825	8.0	6.5	6.20	.	.	.	.	.	.	.	.	235	7.83
1028	HOLL	HOLLAND	19-0065	871029	825	9.0	6.5	6.20	.	10	.	.	.	.	.	.	.	.
1029	HOLL	HOLLAND	19-0065	871029	825	10.0	6.5	6.10	.	.	.	.	.	.	.	.	.	.
1030	HOLL	HOLLAND	19-0065	871029	825	11.0	6.0	1.90	.	.	.	.	.	.	.	.	.	.
1031	HOLL	HOLLAND	19-0065	871029	825	12.0	5.5	1.30	.	.	.	.	.	.	.	.	.	.
1032	HOLL	HOLLAND	19-0065	871029	825	13.0	5.0	1.00	.	.	.	.	.	.	.	.	275	7.37
1033	HOLL	HOLLAND	19-0065	871029	825	14.0	5.0	0.80	.	160	.	.	.	.	.	.	.	.
1034	HOLL	HOLLAND	19-0065	871029	825	15.0	5.0	0.80	.	.	.	.	.	.	.	.	.	.
1035	HOLL	HOLLAND	19-0065	871029	825	16.0	5.0	0.70	.	.	.	.	.	.	.	.	.	.
1036	HOLL	HOLLAND	19-0065	871029	825	17.0	5.0	0.60	.	.	.	.	.	.	.	.	300	7.27
1037	HOLL	HOLLAND	19-0065	871029	825	18.0	5.0	0.60	.	390	.	.	.	.	.	.	.	.
1038	HOLL	HOLLAND	19-0065	871029	825	19.0	5.0	0.60	.	.	.	.	.	.	.	.	.	.
1039	HOLL	HOLLAND	19-0065	871228	845	0.0	.	.	.	25	15	6	1.380	0.150	3.5	3.60	.	.
1040	HOLL	HOLLAND	19-0065	871228	845	1.0	3.0	7.40	.	.	.	.	.	.	.	.	.	.
1041	HOLL	HOLLAND	19-0065	871228	845	4.0	3.0	.	.	30	.	.	.	.	.	.	.	.
1042	HOLL	HOLLAND	19-0065	871228	845	8.0	3.5	6.35	.	20	.	.	.	.	.	.	.	.
1043	HOLL	HOLLAND	19-0065	871228	845	12.0	4.0	5.30	.	30	.	.	.	.	.	.	.	.
1044	HOLL	HOLLAND	19-0065	871228	845	16.0	4.0	0.85	.	40	.	.	.	.	.	.	.	.
1045	HOLL	HOLLAND	19-0065	880119	820	0.0	.	.	.	15	10	6	1.500	0.150	1.2	5.30	.	.

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OBS	LAKE	LAKENAME	LAKEID	DATE	TIME	DM	TC	DO	LIGHT	TP	TDP	OP	TKN	N23	CLA	SDM	COND	PHL
1266	HOLL	HOLLAND	19-0065	880727	815	1.5			139.00									
1267	HOLL	HOLLAND	19-0065	880727	815	2.0	25.9	7.90	83.00									
1268	HOLL	HOLLAND	19-0065	880727	815	2.5			52.50									
1269	HOLL	HOLLAND	19-0065	880727	815	3.0	25.8	7.30	31.00									
1270	HOLL	HOLLAND	19-0065	880727	815	3.5	24.6		20.00									
1271	HOLL	HOLLAND	19-0065	880727	815	4.0	22.2	5.40	12.70									
1272	HOLL	HOLLAND	19-0065	880727	815	4.5	19.5		6.60									
1273	HOLL	HOLLAND	19-0065	880727	815	5.0	15.3	1.20	4.50	20						220	7.87	
1274	HOLL	HOLLAND	19-0065	880727	815	5.5	12.4		3.05									
1275	HOLL	HOLLAND	19-0065	880727	815	6.0	10.5	0.30	1.88									
1276	HOLL	HOLLAND	19-0065	880727	815	6.5	8.8											
1277	HOLL	HOLLAND	19-0065	880727	815	7.0	7.8	0.20										
1278	HOLL	HOLLAND	19-0065	880727	815	7.5	6.6											
1279	HOLL	HOLLAND	19-0065	880727	815	8.0	6.0	0.20										
1280	HOLL	HOLLAND	19-0065	880727	815	8.5	5.3											
1281	HOLL	HOLLAND	19-0065	880727	815	9.0	5.1	0.20		20						250	7.41	
1282	HOLL	HOLLAND	19-0065	880727	815	9.5	4.9											
1283	HOLL	HOLLAND	19-0065	880727	815	10.0	4.8	0.20										
1284	HOLL	HOLLAND	19-0065	880727	815	11.0	4.7	0.20										
1285	HOLL	HOLLAND	19-0065	880727	815	12.0	4.6	0.20										
1286	HOLL	HOLLAND	19-0065	880727	815	13.0	4.6	0.20										
1287	HOLL	HOLLAND	19-0065	880727	815	14.0	4.6	0.20		120						280	7.05	
1288	HOLL	HOLLAND	19-0065	880727	815	15.0	4.6	0.20										
1289	HOLL	HOLLAND	19-0065	880727	815	16.0	4.6	0.20										
1290	HOLL	HOLLAND	19-0065	880727	815	17.0	4.6	0.20										
1291	HOLL	HOLLAND	19-0065	880727	815	18.0				580						310	7.01	
1292	HOLL	HOLLAND	19-0065	880809	840	0.0	26.6	7.00	880.00	35	25	2	0.800	0.050	7.8	2.00	215	8.89
1293	HOLL	HOLLAND	19-0065	880809	840	0.5	26.7	7.10	395.00									
1294	HOLL	HOLLAND	19-0065	880809	840	1.0	26.7	7.00	248.00									
1295	HOLL	HOLLAND	19-0065	880809	840	1.5			158.00									
1296	HOLL	HOLLAND	19-0065	880809	840	2.0	26.6	6.80	92.00									
1297	HOLL	HOLLAND	19-0065	880809	840	2.5			65.00									
1298	HOLL	HOLLAND	19-0065	880809	840	3.0	26.6	6.50	43.00									
1299	HOLL	HOLLAND	19-0065	880809	840	3.5	26.2		24.60									
1300	HOLL	HOLLAND	19-0065	880809	840	4.0	23.6	3.70	13.20	50						230		
1301	HOLL	HOLLAND	19-0065	880809	840	4.5	20.1		6.75									
1302	HOLL	HOLLAND	19-0065	880809	840	5.0	15.8	0.20	4.00									
1303	HOLL	HOLLAND	19-0065	880809	840	5.5	12.9											
1304	HOLL	HOLLAND	19-0065	880809	840	6.0	11.3	0.20										
1305	HOLL	HOLLAND	19-0065	880809	840	6.5	10.0											
1306	HOLL	HOLLAND	19-0065	880809	840	7.0	8.2	0.20										
1307	HOLL	HOLLAND	19-0065	880809	840	7.5	7.2											
1308	HOLL	HOLLAND	19-0065	880809	840	8.0	6.3	0.20										
1309	HOLL	HOLLAND	19-0065	880809	840	8.5	5.6											
1310	HOLL	HOLLAND	19-0065	880809	840	9.0	5.3	0.20		30						260		
1311	HOLL	HOLLAND	19-0065	880809	840	9.5	5.0											
1312	HOLL	HOLLAND	19-0065	880809	840	10.0	5.0	0.20										
1313	HOLL	HOLLAND	19-0065	880809	840	11.0	4.7	0.20										
1314	HOLL	HOLLAND	19-0065	880809	840	12.0	4.7	0.20										
1315	HOLL	HOLLAND	19-0065	880809	840	13.0	4.6	0.20										
1316	HOLL	HOLLAND	19-0065	880809	840	14.0	4.6	0.20		270						290		
1317	HOLL	HOLLAND	19-0065	880809	840	15.0	4.6	0.20										
1318	HOLL	HOLLAND	19-0065	880809	840	16.0	4.6	0.20										
1319	HOLL	HOLLAND	19-0065	880809	840	17.0	4.6	0.20										
1320	HOLL	HOLLAND	19-0065	880809	840	18.0	4.6	0.20		590						325		

OBS	LAKE	LAKENAME	LAKEID	DATE	TIME	DM	TC	DO	LIGHT	TP	TDP	OP	TKN	N23	CLA	SDM	COND	PHL
1321	HOLL	HOLLAND	19-0065	880823	845	0.0	24.1	6.50	745.00	35	15	2	0.850	0.050	26.0	1.80	225	8.60
1322	HOLL	HOLLAND	19-0065	880823	845	0.5			330.00									
1323	HOLL	HOLLAND	19-0065	880823	845	1.0	24.2	6.40	155.00									
1324	HOLL	HOLLAND	19-0065	880823	845	1.5			92.00									
1325	HOLL	HOLLAND	19-0065	880823	845	2.0	24.2	6.30	54.00									
1326	HOLL	HOLLAND	19-0065	880823	845	2.5			32.00									
1327	HOLL	HOLLAND	19-0065	880823	845	3.0	24.1	6.10	18.90									
1328	HOLL	HOLLAND	19-0065	880823	845	3.5			11.50									
1329	HOLL	HOLLAND	19-0065	880823	845	4.0	23.9	4.50	6.60									
1330	HOLL	HOLLAND	19-0065	880823	845	4.5	22.3		4.20									
1331	HOLL	HOLLAND	19-0065	880823	845	5.0	16.8	0.90	2.97	30						230	7.75	
1332	HOLL	HOLLAND	19-0065	880823	845	5.5	13.6											
1333	HOLL	HOLLAND	19-0065	880823	845	6.0	10.7	1.00										
1334	HOLL	HOLLAND	19-0065	880823	845	6.5	9.4											
1335	HOLL	HOLLAND	19-0065	880823	845	7.0	8.1	1.10										
1336	HOLL	HOLLAND	19-0065	880823	845	7.5	6.9											
1337	HOLL	HOLLAND	19-0065	880823	845	8.0	6.3	1.20										
1338	HOLL	HOLLAND	19-0065	880823	845	8.5	5.6											
1339	HOLL	HOLLAND	19-0065	880823	845	9.0	5.3	1.20		20						250	7.20	
1340	HOLL	HOLLAND	19-0065	880823	845	10.0	4.9	1.20										
1341	HOLL	HOLLAND	19-0065	880823	845	11.0	4.8	1.20										
1342	HOLL	HOLLAND	19-0065	880823	845	12.0	4.7	1.20										
1343	HOLL	HOLLAND	19-0065	880823	845	13.0	4.6	1.20										
1344	HOLL	HOLLAND	19-0065	880823	845	14.0	4.6	1.20		230						280	7.00	
1345	HOLL	HOLLAND	19-0065	880823	845	15.0	4.6	1.20										
1346	HOLL	HOLLAND	19-0065	880823	845	16.0	4.6	1.20										
1347	HOLL	HOLLAND	19-0065	880823	845	17.0	4.6	1.30		520						300	6.94	
1348	HOLL	HOLLAND	19-0065	880906	830	0.0	19.7	8.40	520.00	30	10	4	0.850	0.050	16.0	1.70	235	8.88
1349	HOLL	HOLLAND	19-0065	880906	830	0.5			274.00									
1350	HOLL	HOLLAND	19-0065	880906	830	1.0	19.7	8.40	130.00									
1351	HOLL	HOLLAND	19-0065	880906	830	1.5			74.00									
1352	HOLL	HOLLAND	19-0065	880906	830	2.0	19.7	8.40	42.50									
1353	HOLL	HOLLAND	19-0065	880906	830	2.5			26.60									
1354	HOLL	HOLLAND	19-0065	880906	830	3.0	19.7	8.30	16.30									
1355	HOLL	HOLLAND	19-0065	880906	830	3.5			10.20									
1356	HOLL	HOLLAND	19-0065	880906	830	4.0	19.6	8.20	6.55									
1357	HOLL	HOLLAND	19-0065	880906	830	4.5	19.4		4.10									
1358	HOLL	HOLLAND	19-0065	880906	830	5.0	19.1	5.90	2.73	30						220	8.83	
1359	HOLL	HOLLAND	19-0065	880906	830	5.5	16.2											
1360	HOLL	HOLLAND	19-0065	880906	830	6.0	11.8	1.40										
1361	HOLL	HOLLAND	19-0065	880906	830	6.5	9.7											
1362	HOLL	HOLLAND	19-0065	880906	830	7.0	8.0	1.30										
1363	HOLL	HOLLAND	19-0065	880906	830	7.5	7.2											
1364	HOLL	HOLLAND	19-0065	880906	830	8.0	6.3	1.30										
1365	HOLL	HOLLAND	19-0065	880906	830	8.5	5.8											
1366	HOLL	HOLLAND	19-0065	880906	830	9.0	5.3	1.20		20						265		
1367	HOLL	HOLLAND	19-0065	880906	830	9.5	5.0											
1368	HOLL	HOLLAND	19-0065	880906	830	10.0	4.9	1.20										
1369	HOLL	HOLLAND	19-0065	880906	830	11.0	4.7	1.10										
1370	HOLL	HOLLAND	19-0065	880906	830	12.0	4.7	1.00										
1371	HOLL	HOLLAND	19-0065	880906	830	13.0	4.6	1.00		140						295		
1372	HOLL	HOLLAND	19-0065	880906	830	14.0	4.6	1.00										
1373	HOLL	HOLLAND	19-0065	880906	830	15.0	4.6	0.90										
1374	HOLL	HOLLAND	19-0065	880906	830	16.0	4.6	0.90										
1375	HOLL	HOLLAND	19-0065	880906	830	17.0	4.6	0.90		470						325		

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OBS	LAKE	LAKENAME	LAKEID	DATE	TIME	DM	TC	DO	LIGHT	TP	TDP	OP	TKN	N23	CLA	SDM	COND	PHL
2146	TANN	TANNERS	82-0015	880921	1050	4.0	16.8	7.60	1.50	.	.	.	.	.	.	.	.	.
2147	TANN	TANNERS	82-0015	880921	1050	5.0	16.8	7.60	.	.	.	.	.	.	.	.	.	
2148	TANN	TANNERS	82-0015	880921	1050	6.0	16.8	7.40	.	50	.	.	.	.	.	385	.	
2149	TANN	TANNERS	82-0015	880921	1050	6.5	16.7	.	.	.	.	.	.	.	.	.	.	
2150	TANN	TANNERS	82-0015	880921	1050	7.0	14.3	3.50	.	.	.	.	.	.	.	.	.	
2151	TANN	TANNERS	82-0015	880921	1050	7.5	11.7	.	.	.	.	.	.	.	.	.	.	
2152	TANN	TANNERS	82-0015	880921	1050	8.0	9.9	.	.	.	.	.	.	.	.	.	.	
2153	TANN	TANNERS	82-0015	880921	1050	8.5	8.9	.	.	.	.	.	.	.	.	.	.	
2154	TANN	TANNERS	82-0015	880921	1050	9.0	8.5	0.80	.	620	.	.	.	.	.	475	.	
2155	TANN	TANNERS	82-0015	880921	1050	10.0	8.1	.	.	.	.	.	.	.	.	.	.	
2156	TANN	TANNERS	82-0015	880921	1050	11.0	7.8	1.00	.	1100	.	.	.	.	.	510	.	
2157	TANN	TANNERS	82-0015	881004	1030	0.0	13.9	8.00	1020.00	60	45	20	1.090	0.050	17.0	2.20	375	8.28
2158	TANN	TANNERS	82-0015	881004	1030	0.5	.	.	530.00	.	.	.	.	.	.	.	.	
2159	TANN	TANNERS	82-0015	881004	1030	1.0	13.9	7.90	260.00	.	.	.	.	.	.	.	.	
2160	TANN	TANNERS	82-0015	881004	1030	1.5	.	.	135.00	.	.	.	.	.	.	.	.	
2161	TANN	TANNERS	82-0015	881004	1030	2.0	13.9	7.90	83.00	.	.	.	.	.	.	.	.	
2162	TANN	TANNERS	82-0015	881004	1030	2.5	.	.	47.00	.	.	.	.	.	.	.	.	
2163	TANN	TANNERS	82-0015	881004	1030	3.0	13.9	7.80	26.50	60	.	.	.	.	.	380	8.31	
2164	TANN	TANNERS	82-0015	881004	1030	3.5	.	.	15.80	.	.	.	.	.	.	.	.	
2165	TANN	TANNERS	82-0015	881004	1030	4.0	13.9	7.90	10.10	.	.	.	.	.	.	.	.	
2166	TANN	TANNERS	82-0015	881004	1030	4.5	.	.	6.50	.	.	.	.	.	.	.	.	
2167	TANN	TANNERS	82-0015	881004	1030	5.0	13.9	7.90	4.00	.	.	.	.	.	.	.	.	
2168	TANN	TANNERS	82-0015	881004	1030	6.0	13.9	7.80	.	50	.	.	.	.	.	380	8.30	
2169	TANN	TANNERS	82-0015	881004	1030	7.0	13.9	7.80	.	.	.	.	.	.	.	.	.	
2170	TANN	TANNERS	82-0015	881004	1030	8.0	13.9	7.80	.	.	.	.	.	.	.	.	.	
2171	TANN	TANNERS	82-0015	881004	1030	8.5	10.5	.	.	.	.	.	.	.	.	.	.	
2172	TANN	TANNERS	82-0015	881004	1030	9.0	8.9	1.00	.	300	.	.	.	.	.	415	7.66	
2173	TANN	TANNERS	82-0015	881004	1030	9.5	8.5	.	.	.	.	.	.	.	.	.	.	
2174	TANN	TANNERS	82-0015	881004	1030	10.0	8.0	0.90	.	.	.	.	.	.	.	.	.	
2175	TANN	TANNERS	82-0015	881004	1030	10.5	7.8	.	.	.	.	.	.	.	.	.	.	
2176	TANN	TANNERS	82-0015	881004	1030	11.0	7.6	0.80	.	.	.	.	.	.	.	.	.	
2177	TANN	TANNERS	82-0015	881004	1030	12.0	7.5	0.80	.	1410	.	.	.	.	.	520	7.14	
2178	TANN	TANNERS	82-0015	881004	1030	13.0	7.3	0.80	.	.	.	.	.	.	.	.	.	
2179	TANN	TANNERS	82-0015	881018	1020	0.0	11.8	8.70	355.00	85	50	29	1.340	0.150	42.0	1.90	375	8.50
2180	TANN	TANNERS	82-0015	881018	1020	0.5	.	.	107.00	.	.	.	.	.	.	.	.	
2181	TANN	TANNERS	82-0015	881018	1020	1.0	11.8	8.90	59.00	.	.	.	.	.	.	.	.	
2182	TANN	TANNERS	82-0015	881018	1020	1.5	.	.	35.50	.	.	.	.	.	.	.	.	
2183	TANN	TANNERS	82-0015	881018	1020	2.0	11.8	8.80	22.00	.	.	.	.	.	.	.	.	
2184	TANN	TANNERS	82-0015	881018	1020	2.5	.	.	12.20	.	.	.	.	.	.	.	.	
2185	TANN	TANNERS	82-0015	881018	1020	3.0	11.8	8.80	7.35	100	.	.	.	.	.	385	8.51	
2186	TANN	TANNERS	82-0015	881018	1020	3.5	.	.	4.65	.	.	.	.	.	.	.	.	
2187	TANN	TANNERS	82-0015	881018	1020	4.0	11.8	8.60	2.94	.	.	.	.	.	.	.	.	
2188	TANN	TANNERS	82-0015	881018	1020	5.0	11.8	8.40	.	.	.	.	.	.	.	.	.	
2189	TANN	TANNERS	82-0015	881018	1020	6.0	11.8	7.60	.	80	.	.	.	.	.	380	8.38	
2190	TANN	TANNERS	82-0015	881018	1020	7.0	11.7	4.70	.	.	.	.	.	.	.	.	.	
2191	TANN	TANNERS	82-0015	881018	1020	8.0	11.4	3.20	.	.	.	.	.	.	.	.	.	
2192	TANN	TANNERS	82-0015	881018	1020	8.5	11.1	.	.	.	.	.	.	.	.	.	.	
2193	TANN	TANNERS	82-0015	881018	1020	9.0	10.6	0.80	.	220	.	.	.	.	.	405	7.60	
2194	TANN	TANNERS	82-0015	881018	1020	9.5	9.3	.	.	.	.	.	.	.	.	.	.	
2195	TANN	TANNERS	82-0015	881018	1020	10.0	8.3	0.70	.	.	.	.	.	.	.	.	.	
2196	TANN	TANNERS	82-0015	881018	1020	10.5	8.0	.	.	.	.	.	.	.	.	.	.	
2197	TANN	TANNERS	82-0015	881018	1020	11.0	7.8	0.70	.	.	.	.	.	.	.	.	.	
2198	TANN	TANNERS	82-0015	881018	1020	12.0	7.4	0.70	.	1460	.	.	.	.	.	505	7.12	