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# Environmental Effects of Hydraulic Engineering Works

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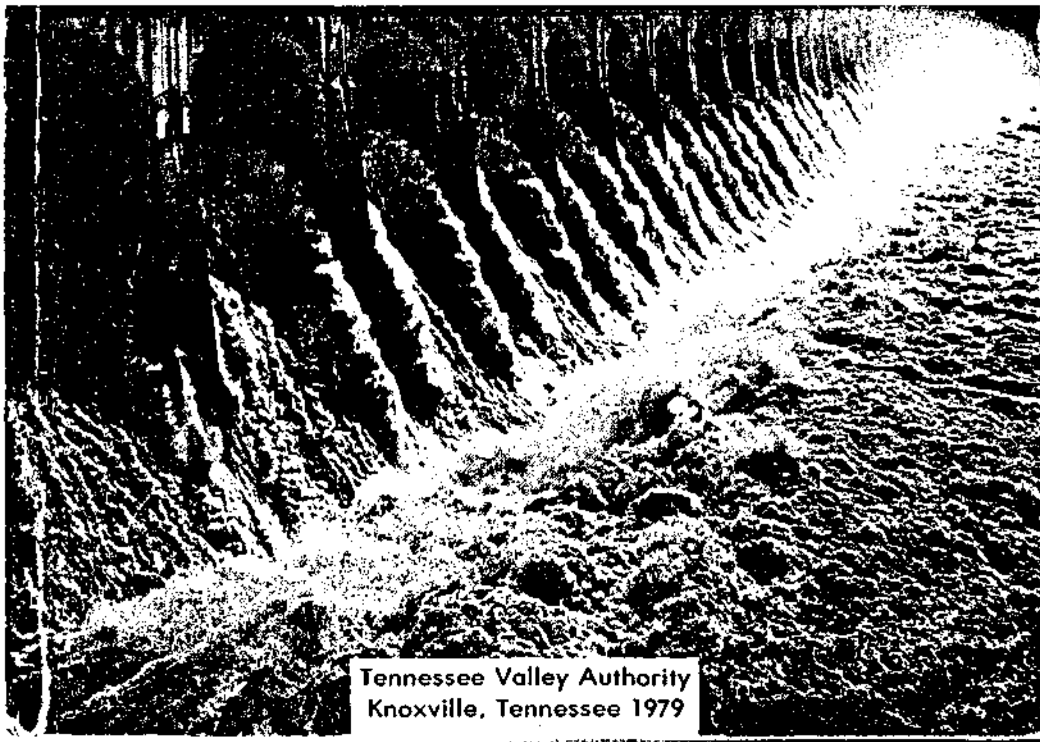

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"An Empirical Analysis of Factors Controlling Eutrophication in Midwestern Impoundments"

by

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ABSTRACT

An analytical strategy is described and implemented to examine empirically the factors controlling algal growth in midwestern impoundments. EPA National Eutrophication Survey data from 23 natural lakes and 27 reservoirs located in Ohio, Indiana, and Illinois provide a basis for contrasting lake and reservoir behavior in this region. Three aspects are studied: (1) phosphorus retention; (2) impoundment phosphorus/outflow phosphorus relationships; (3) algal response as measured by mean summer chlorophyll-a concentration. Results indicate that previously developed empirical models for assessing eutrophication problems based only upon total phosphorus loading and certain morphometric and hydrologic characteristics are not appropriate in this region due to the apparent influences of sediment loadings on phosphorus trapping, light penetration, and nutrient availability. Multiple regression analyses indicate that chlorophyll-a levels are significantly related to phosphorus levels and flushing rates in natural lakes and to phosphorus levels, turbidities, and depths in reservoirs. These results are consistent with the hypothesis that light is more important as a controlling factor in reservoirs than in lakes. Nitrogen is apparently insignificant as a controlling factor in most of the impoundments studied.

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## INTRODUCTION

Methods for assessing eutrophication problems in impoundments range from the relatively simple and empirical to the relatively complex and theoretical. For use in data-limited areas, the simplicity and low data requirements of the former are desirable attributes, provided that they are reliable and flexible enough to address the kinds of questions which might be asked in a water quality management context. Most empirical efforts at eutrophication modeling have been based upon data from natural lakes in northern latitudes (Vollenweider 1975, 1976; Dillon and Rigler, 1974a; Jones and Bachman 1976; Chapra and Tarapchak, 1976; Reckhow, 1977; and Walker, 1977). Because of the empirical nature of these models, they are of questionable validity in impoundments that are artificial and/or located further south. Shannon and Brezonik (1972a, 1972b) and Tapp (1978) have analyzed data from southern latitudes and some reservoirs, but have not systematically tested for differences in model performance due to region or impoundment type.

In developing a methodology for evaluating the effects of agricultural practices on water quality, Meta Systems (1978) analyzed data from a collection of lakes and reservoirs located in Ohio, Indiana, and Illinois. Based upon that work, this paper describes a general analytical approach leading to some empirical insights into the factors controlling algal growth in midwestern impoundments. The results provide preliminary indications of the kinds of modifications which might be necessary if empirical approaches to eutrophication assessment are to be applied successfully to lakes and reservoirs in this geographic region.

## DATA BASE

The EPA's National Eutrophication Survey (1975) has provided data on the location, hydrology, morphometry, nutrient budgets, and trophic state indicators of 75 impoundments in the three-state region. These data have been screened to eliminate those impoundments with incomplete information, with unusual characteristics which may have influenced nutrient dynamics and trophic state response (e.g., artificial mixing), or with hydraulic residence times less than three days. A statistical summary of the remaining sample of 23 natural lakes and 27 reservoirs is given in Table 1. Seven of the natural lakes were classified by EPA as mesotrophic. The remaining 16 lakes and 27 reservoirs were classified as eutrophic.

TABLE 1

Statistical Summary of EPA National Eutrophication Survey Data from 50 Indiana, Illinois, and Ohio Impoundments

| | | 23 Lakes | | | 27 Reservoirs | | |
|---|---|---|---|---|---|---|---|
| Variable | Units | Minimum | Median | Maximum | Minimum | Median | Maximum |
| Surface Area | ($km^2$) | .23 | 1.89 | 44.52 | .19 | 5.62 | 105.2 |
| Drainage Area | ($km^2$) | 7.7 | 87.7 | 2418. | 17.7 | 551. | 6937. |
| Mean Depth | (m) | 1.4 | 6.1 | 12.2 | 1.2 | 4.6 | 7.2 |
| Maximum Depth | (m) | 4.0 | 15.1 | 37.5 | 3.7 | 9.4 | 34.7 |
| Hydraulic Residence Time | (yrs) | .03 | .41 | 6.7 | .03 | .21 | 1.25 |
| Overflow Rate | (m/yr) | .56 | 13.4 | 142. | 2.81 | 15.8 | 120. |
| Phosphorus Loading | ($g/m^2$-yr) | .09 | .93 | 29.5 | .28 | 3.14 | 32.9 |
| Retention Coef. | --- | -.03 | .33 | .87 | .17 | .47 | .80 |
| Nitrogen Loading | ($g/m^2$-yr) | 3.30 | 46. | 597. | 6.10 | 75.9 | 469. |
| Retention Coef. | --- | -.21 | .24 | .80 | -.10 | .23 | .55 |
| Chlorophyll-a* | ($mg/m^3$) | 3.8 | 11.5 | 187. | 7.0 | 21.1 | 90.5 |
| Secchi Depth* | (m) | .25 | 1.60 | 3.75 | .36 | .71 | 1.55 |
| Median Total P* | ($mg/m^3$) | 9.0 | 34. | 704. | 25. | 73. | 426. |
| Median Inorganic N* | ($g/m^3$) | .12 | .76 | 3.75 | .15 | 1.18 | 5.73 |

*Summer average; annual average otherwise.

ANALYTICAL STRATEGY

Bioassay studies conducted by the EPA/NES (1975) in these impoundments indicate that nitrogen is present in excess of phosphorus relative to algal growth requirements in most of them. Thus, our initial premise is that algal growth is limited primarily by phosphorus supplies. Possible effects of limitation by light and/or nitrogen in specific impoundments are also assessed. The basic analytical strategy depicted in Figure 1 is similar to that described by Chapra and Tarapchak (1976). Three types of models are used to relate external phosphorus loadings to algal growth responses, as measured by mean summer, photic-zone chlorophyll-a concentrations. First, a phosphorus retention model estimates that fraction of the phosphorus input which is trapped in the impoundment sediments on an average annual basis. A second model relates annual average phosphorus concentration measured in the impoundment outflow to that measured within the impoundment during the summer season and assumed to be available to support algal growth. Finally, an algal growth model relates chlorophyll-a to phosphorus and other controlling factors. For each of these relationships, the lake and reservoir data have been examined in relation to previous models developed for natural lakes in other geographic regions in order to assess model generality across regions and impoundment types.

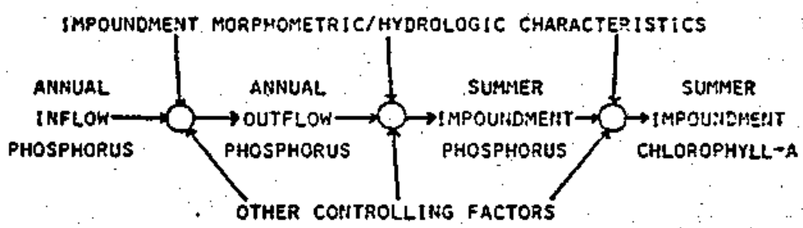



Figure 1: Analytical Strategy

Horizontal variations in phosphorus and chlorophyll-a levels are significant in some lakes and reservoirs, but only average levels are considered here. This is a logical first step, given the types of available data, and does not imply an assumption of complete horizontal mixing. Future development of an expanded data base and modifications of this analytical approach to account for horizontal variations may be justified, particularly for applications involving reservoirs with high length/width ratios.

PHOSPHORUS RETENTION MODEL

The phosphorus retention model estimates the fraction of influent phosphorus which is trapped in bottom sediments due to the net effect of various physical, chemical, and biological processes. From a steady-state mass balance:

$$P_o = P_i \; (1 - R_p) \qquad (1)$$

where,

$P_o$ = annual-average outflow total phosphorus concentration (mg/m$^3$)

$P_i$ = annual-average inflow total phosphorus concentration (mg/m$^3$)

$R_p$ = retention coefficient (dimensionless)

The average inflow and outflow concentrations are flow-weighted and the retention coefficient is measured indirectly for a given impoundment from measurements of flow, inflow concentration, and outflow concentration.

A variety of models have been proposed for estimating $R_p$ as a function of impoundment morphometric and hydrologic characteristics. One version (Vollenweider, 1969, Chapra, 1975) essentially assumes that the removal rate of phosphorus per unit area is proportional to the average phosphorus concentration in the water column. In a completely-mixed system, a steady-state mass balance gives the following formula for the retention coefficient:

$$1 - R_p = \frac{P_o}{P_i} = \frac{Q_s}{Q_s + U_p} \qquad (2)$$

where,

$Q_s$ = surface overflow rate (m/year)

$U_p$ = effective "settling velocity "of phosphorus (m/yr)

Because of the variety of mechanisms involved in phosphorus retention and because the assumption of complete horizontal and vertical mixing is rarely satisfied in lakes and reservoirs, the physical interpretation of the term "settling velocity" is suspect. Equation (3) should be viewed as an empirical rather than theoretical formulation. Retention coefficient and overflow rate data from northern lakes suggest average settling velocities of 10 m/year (Vollenweider, 1969) to 16 m/year (Chapra, 1975). The performance of this model in Corn Belt impoundments is poor. The effective settling velocity is highly variable across impoundments, ranging from -1 to 125 m/yr with a median value of 12.7 m/yr. It is apparent that these systems may be characterized somewhat differently from northern lakes with regard to rates and mechanisms of phosphorus trapping. Similar conclusions are reached when alternative models developed for northern lakes (Kirchner and Dillon, 1975, Larsen and Mercier, 1975) are tested with these data.

Several investigators have suggested that sediment entering waterways as a result of watershed erosion may function as a sink for phosphorus under certain conditions due to absorption-sedimentation reactions (Taylor and Kunishi, 1971; To and Randall, 1975; Olness and Rausch, 1977). These observations, combined with the moderate to high erosion rates characteristic of some watersheds in this geographic region, suggest that phosphorus trapping efficiencies may be partially controlled by impoundment sedimentation rates. To test this hypothesis, sedimentation measurements have been obtained for 15 of these impoundments from the USDA (1973) and the Illinois State Water Survey (1977). Given the scarcity of the coincident phosphorus budget and sedimentation rate data, measurements on Callahan Reservoir, Missouri (Rausch and Schreiber, 1977) have been used to supplement the above. Reported sedimentation rates for these 16 impoundments range from 3 to 277 $kg/m^2$-year and are attributed primarily to external sediment inputs resulting from watershed erosion, as distinguished from internal sources (chemical precipitation and net primary production).

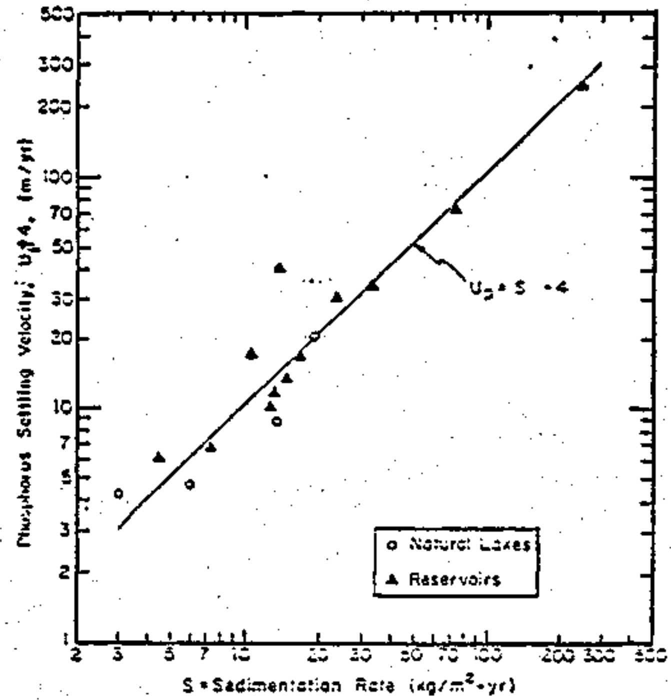
Equation (2) can be solved for $U_p$ to estimate an effective settling velocity for each impoundment based upon reported $Q_s$ and $R_p$ values. The relationship between $U_p$ and sedimentation rate for 16 impoundments is depicted in Figure 2 and summarized by an equation of the following form:

$$U_p = -4 + S \qquad (3)$$

where,

S = sedimentation rate ($kg/m^2$-year)

Allowing the estimated settling velocity to assume a minimum value of zero, equations (2) and (3) explain 73 percent of the variance in the reported retention coefficient data with a standard error of 0.12.




A variety of alternative empirical formulations for predicting retention coefficients in these impoundments have been tested (Meta Systems, 1978). In addition to sedimentation rate, depth, overflow rate, and inflow phosphorus concentration have been included as independent variables. The results of the analysis indicate that sedimentation rate has by far the strongest predictive ability. The limited amount of data available (16 impoundments), however, does not permit clear distinction among alternative forms for the model. Equation (3) is the simplest formulation tested and should be considered a preliminary result. At this time, insufficient data are available to sort out mechanisms or to detect differences between natural lakes and reservoirs, once sedimentation rates are taken into account. The generally higher sedimentation rates characteristic of reservoirs, however, would indicate generally greater phosphorus trapping efficiencies.

Figure 2: Relationship between Sedimentation Rate and Phosphorus Settling Velocity for 16 Impoundments

A potential exists for recycling of bottom sediment phosphorus into the water column due to wind-induced resuspension and/or to changes in phosphorus adsorption chemistry under the anaerobic conditions which are characteristic of bottom sediments and some impoundment hypolimnia (Syers et al., 1973). Anaerobic conditions were detected by the EPA/NES in the bottom waters of seven of the fifteen impoundments studied here, all of which have mean depths less than five meters. The results may not hold true in deeper lakes or reservoirs with more extensive stratification and greater potential for phosphorus recycling through anaerobic bottom waters. It is possible that the negative intercept in equation (3) reflects the importance of this recycling in impoundments with relatively low sedimentation rates. Additional data are needed to test the generality of the above results in deeper impoundments and in other geographic areas.

OUTFLOW PHOSPHORUS/IMPOUNDMENT PHOSPHORUS RELATIONSHIPS

In order to link the phosphorus retention and algal growth models, it is necessary to relate annual-average outflow phosphorus concentration to that measured during the summer within the impoundment. In general, a direct equivalence of these two quantities cannot be assumed because of the possible effects of seasonal variations, lack of complete horizontal mixing (e.g., plug flow behavior) and/or vertical stratification. Regression analyses have been performed to relate $P_o$ and $P_s$ values in the sample of 23 natural lakes and 27 reservoirs. As tabulated by the EPA/NES (1975), $P_s$ is defined as the spatial and temporal median total phosphorus concentration measured during the summer within the impoundment. Regression analyses performed for lakes and reservoirs separately do not indicate significant differences between the two data groups. The relationship can be summarized by:

$$P_s = 0.78\ P_o \quad (4)$$

An identical relationship has been derived using EPA/NES data from northcentral and northeastern impoundments (Walker, 1978). As shown in Figure 3, equation (4) explains 88 percent of the variance in the base-10 logarithm of $P_s$ with a standard error of 0.14. Residuals do not exhibit any patterns when plotted against a number of morphometric and hydrologic factors.

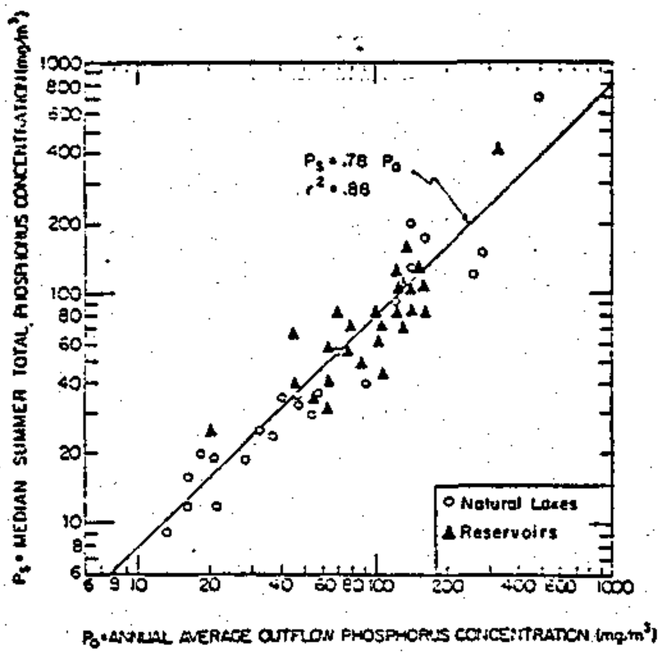



Figure 3: Relationship Between Median, Summer, Impoundment Phosphorus and Annual Outflow Phosphorus Concentrations

These results suggest that, on the average, seasonal effects on phosphorus concentration may dominate over those of plug flow, since plug flow behavior would tend to result in $P_s/P_o$ ratios greater than one. In future work, it would be interesting to examine $P_s/P_o$ ratios as a function of length/width ratio or other morphometric variables which may govern the extent of horizontal mixing. Summer phosphorus levels may tend to be lower than annual average levels because of longer residence times and higher rates of biological activity. Another effect which may be important and requires further testing is that of seasonal variations in external phosphorus loadings, which would tend to be lower during the summer in impoundments dominated by non-point sources but relatively constant with season in impoundments dominated by point sources. Internal phosphorus loading resulting from bottom sediment releases may also influence seasonal variations in phosphorus levels. Specific data on outflow withdrawal levels in reservoirs are needed to test for the possible effects of low-level outlets, which tend to release relatively enriched waters and thereby influence both phosphorus trapping efficiencies and outflow phosphorus/impoundment phosphorus relationships.

CHLOROPHYLL-a MODEL

The third part of the analytical framework depicted in Figure 1 is the relationship between impoundment phosphorus and chlorophyll-a levels. Based upon data from northern, phosphorus-limited lakes, Dillon and Rigler (1974b) obtained the following result:

$$\log_{10} B = -1.14 + 1.45 \log_{10} P_v \qquad (r^2 = .92,\ S.E.E. = .22) \quad (5)$$

where,

$B$ = mean summer chlorophyll-a (mg/m³)

$P_v$ = mean spring total phosphorus (mg/m³)

Nearly identical slopes (1.45) have been derived elsewhere using data from different sets of northern lakes and based upon summer rather than spring total phosphorus concentrations (Jones and Bachman, 1976, Walker, 1978). The slope of the B versus $P_s$ relationship in these impoundments is 0.87 ± .10, significantly different from 1.45. Even when fourteen, possibly nitrogen-limited impoundments ($N/P < 12$) are removed (as done by Dillon and Rigler), equation (5) under-predicts chlorophyll-a levels slightly at low phosphorus levels and over-predicts substantially at high phosphorus levels.

A series of multiple regression analyses has been done to test for possible effects of limiting factors other than phosphorus. The ratio of average outflow nitrogen concentration to average outflow phosphorus concentration has been included as an independent variable to test for nitrogen limitation. The effect... is apparently insignificant in all but two impoundments, both of which have N/P ratios less than 9. This is consistent with EPA/NES bioassay results. Because two data points do not provide an adequate basis for estimating model parameters describing effects of nitrogen limitation, these two impoundments (one lake and one reservoir) have been excluded from the analysis.

Secchi depth measurements have been used to provide an estimate of light extinction due to turbidity and/or color in each impoundment. Secchi depths are influenced partially by chlorophyll-a concentration because of light extinction by algal biomass. The following model has been derived from Secchi depth, light extinction, and chlorophyll-a concentration data from midwestern impoundments (Meta Systems, 1978):

$$\varepsilon = \frac{1.66}{Z_s} = \alpha + .03\ B \qquad (6)$$

where,

$\varepsilon$ = visible light extinction coefficient ($m^{-1}$)

$Z_s$ = Secchi depth (m)

$\alpha$ = light extinction due to water, dissolved color, and non-algal suspended solids ($m^{-1}$)

This assumes that the Secchi depth is inversely proportional to the light extinction coefficient, which, in turn, is a linear function of chlorophyll-a. Solving equation (6) for $\alpha$ and substituting measured $Z_s$ and B values provides and approximate estimate of non-algal turbidity and color in each impoundment. This variable has been included as a test for possible algal growth limitation by light and/or the effects of particulate and colloidal materials, which may render a portion of the total phosphorus measured in each impoundment unavailable to support algal growth.

Mean depth and surface overflow rate have also been included as independent variables in the regression analyses. The former may reflect an influence of light limitation since the depth-averaged light intensity in a totally-absorbing water column is inversely proportional to depth. Overflow rate has been included as a test for possible importance of flushing as a controlling factor.

The results of regression analyses performed using data from 22 lakes, 26 reservoirs, and both groups combined are summarized in Table 2. Because the data tend to be lognormally distributed, logarithmic transformations have been used for the variables B, $P_s$, and Z. Based upon examinations of partial residuals plots for alternative variable transformations, linear forms of $\alpha$ and $Q_s$ appear to be the most appropriate. The standard regression coefficients (Dixon and Brown, 1977) may be taken as approximate measures of the relative explanatory power of each independent variable. When both data groups are combined, each regression coefficient

TABLE 2

Statistics Derived from Chlorophyll-a Regression Analyses[a]

| Variable | Natural Lakes | Reservoirs | Both |
|---|---|---|---|
| | Regression Coefficients | | |
| Intercept | -.256 | .490 | .070 |
| $\log_{10}P_s$ | 1.103 ± .121* | .875 ± .189* | 1.032 ± .104* |
| $\alpha$ | -.055 ± .053 | -.231 ± .039* | -.164 ± .031* |
| $\log_{10}Z$ | -.219 ± .154 | -.586 ± .193* | -.435 ± .131* |
| $Q_s$ | -.0049 ± .0010* | -.0007 ± .0013 | -.0036 ± .0011* |
| | Standard Regression Coefficients | | |
| $\log_{10}P_s$ | 1.015* | .660* | .938* |
| $\alpha$ | -.094 | -.801* | -.419* |
| $\log_{10}Z$ | -.149 | -.426* | -.301* |
| $Q_s$ | -.408* | -.065 | -.298* |
| | Regression Statistics | | |
| N | 22 | 26 | 48 |
| $R^2$ | .925 | .742 | .836 |
| S.E.E.[b] | .133 | .148 | .156 |

*Coefficient Significant ($p<.01$)
[a]Predicted Variable = base-10 logarithm of mean summer chlorophyll-a concentration ($mg/m^3$)
[b]S.E.E. = Standard Error of Estimate (base 10 logarithm)

is statistically significant ($p < 0.01$). When the data set is stratified according to impoundment type, the coefficients reveal a contrast in lake versus reservoir behavior. Regression coefficients vary significantly across data groups ($p=0.029$). The quality of the fit in lakes is somewhat better than in reservoirs, but both are superior to the Dillon and Rigler model, as gauged by the standard error of estimate. Observations are plotted against predictions in Figure 4, using separate sets of coefficients for lakes and reservoirs.

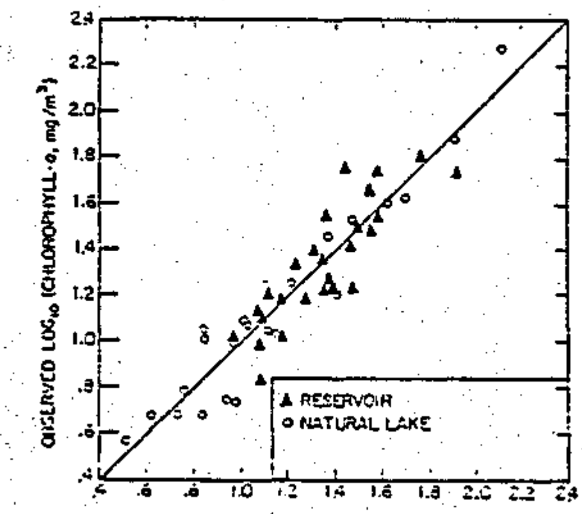



Figure 4: Observed versus Estimated Chlorophyll-a

Results suggest an importance of phosphorus and flushing in controlling chlorophyll-a levels in lakes, while phosphorus, non-algal turbidity and color ($\alpha$), and depth appear to be most important in reservoirs. These results are consistent with a hypothesis that light is more important as a controlling factor in reservoirs than in lakes. Average $\alpha$ values in these data groups are 1.68 $m^{-1}$ and 0.84 $m^{-1}$, and, as discussed above, may represent influences of turbidity on light penetration and/or phosphorus availability, although we cannot separate these effects without additional information on the distribution of phosphorus among various fractions (e.g., dissolved and particulate). The apparent significance of overflow rate as a limiting factor in lakes but not in reservoirs may be due to summertime withdrawal of primarily epilimnetic waters in lakes versus primarily hypolimnetic waters in some reservoirs. Flushing would be a more important algal removal mechanism in the former since growth occurs chiefly in surface waters. Specific data on reservoir outlet levels are needed to further study this relationship.

While a theoretical basis for including each of the above terms in the regression analyses has been presented, the results of the analyses do not necessarily prove or disprove the hypothesized causal relationships. Because of correlations among these factors and because the simple linear or log-linear models assumed are not mechanistically realistic, it is tenuous to accept the results of these analyses as proof of mechanisms. (Blalock, 1961). Results should be considered as suggestive rather than conclusive regarding causal relationships. In future work, it would be useful to test some more theoretically based formulations of the above model. Some initial attempts along these lines are discussed by Meta Systems (1978). The results presented here serve primarily to point out the possible importance of factors other than total phosphorus in controlling chlorophyll-a levels in lakes and reservoirs in this region.

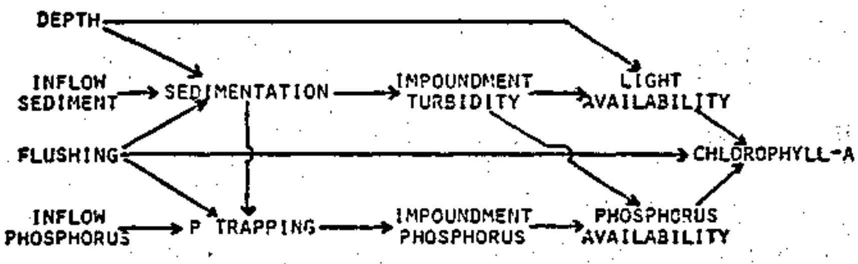



Figure 5: Control Pathways For Chlorophyll-a Concentrations.

## CONCLUSIONS

Based upon the results of the above analysis, Figure 5 depicts what appear to be important pathways controlling algal growth in lakes and reservoirs in this region. Previous empirical efforts at eutrophication modeling have been based primarily on data from northern, natural lakes. These models appear to be inappropiate for Corn Belt impoundments, possibly because of the influences of sediment loading on phosphorus trapping, phosphorus availability, and light penetration. The generality of these results needs to be examined in other geographical regions. Differences between lake and reservoir behavior within this region may be attributed partially to differences in sedimentation rates. Possible influences of hydrodynamic factors have not been examined in detail here but should be in the future in order to provide a better basis for applying these types of models to reservoirs. Some important implications with regard to the potential effects of erosion control practices on impoundment eutrophication are discussed by Meta Systems (1978). These empirical results indicate that theoretical approaches to modeling the eutrophication of these impoundments involving elaborate specification of phytoplankton dynamics will not be successful unless sediment dynamics and influences are properly understood and represented.

## ACKNOWLEDGEMENT

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