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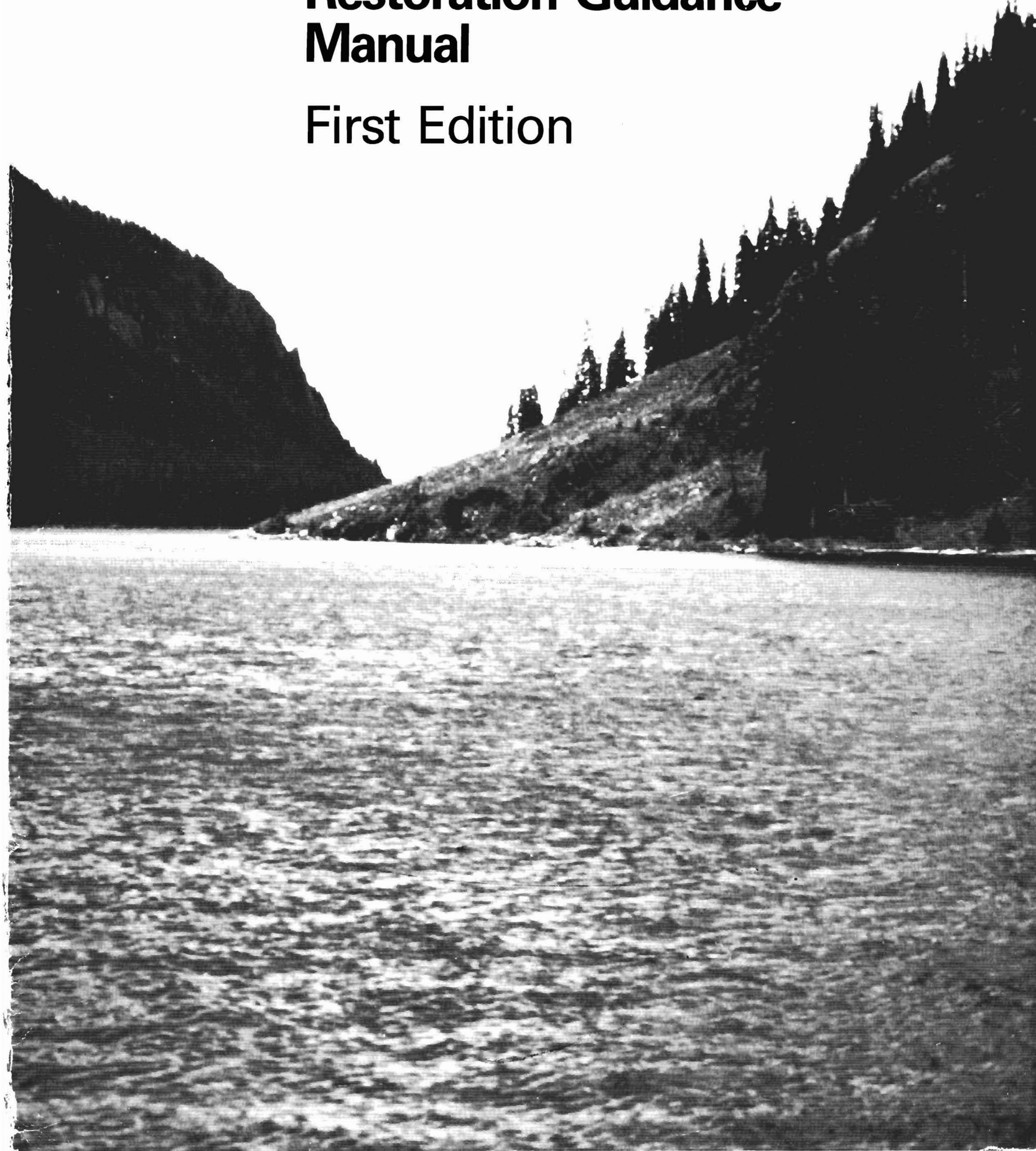
The Lake and Reservoir Restoration Guidance Manual

First Edition



Lake and Reservoir Restoration Guidance Manual

Prepared by the

North American Lake Management Society
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for the

Office of Research and Development
Environmental Research Laboratory
Corvallis, Oregon

and

Office of Water
Criteria & Standards Division
Nonpoint Sources Branch
U.S. Environmental Protection Agency
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Preface

The Lake and Reservoir Restoration Guidance Manual represents a landmark in this nation's commitment to water quality, as it brings to the lake user practical knowledge for restoring and protecting lakes and reservoirs. More than an explanation of restoration techniques, this Manual is a guide to wise management of lakes and reservoirs.

Congress recognized the need for compiling this information and communicating it to the community level, as the logical outgrowth of the Clean Lakes Program established by the Clean Water Act of 1972. For the past 17 years, the Clean Lakes Program has given states matching grants to restore degraded lakes. That process has generated a great deal of information about what techniques to use in restoring these water bodies, how and where they should be used, and how well they work. This Manual is the first step in making that information available in a comprehensive, organized format.

As the Manual was being written, Congress continued its effort to improve this knowledge base by mandating in the Water Quality Act of 1987, that this Manual be updated every two years.

The purpose of the Manual is to provide guidance to the lake manager, lake homeowner, lake association and other informed laypersons on lake and reservoir management, restoration and protection. With this in mind, the reader is invited to send comments and suggestions to the Clean Lakes Program, Nonpoint Sources Branch (WH-585), U.S. Environmental Protection Agency, 401 M Street, S.W., Washington, DC 20460.

Contents

Chapter 3: Problem Identification

Chapter 4: Predicting Lake Water Quality

Chapter 5: Managing the Watershed

Chapter 6: Lake and Reservoir Restoration and Management Techniques

Chapter 7: Hypothetical Case Study

Chapter 8: Implementing the Management Plan

Chapter 9: Lake Protection And Maintenance

Appendices

Acknowledgements

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CHAPTER 4

PREDICTING LAKE WATER QUALITY

Uses of Models

Mathematical models can be useful both in diagnosing lake problems and in evaluating alternative solutions. They represent the cause–effect relationships that control lake water quality in quantitative terms. Model formulas are derived from scientific theories and from observations of the processes and responses in real lakes. There are two basic ways in which models can be employed in lake studies:

1. **DIAGNOSTIC MODE: What is going on in the lake?** Models provide a frame of reference for interpreting lake and watershed monitoring data. They tell the user what to *expect* to find in a lake with a given set of morphometric, hydrologic, and watershed characteristics. These expectations are not always met, however. Differences between measured and predicted conditions contain information on the unique features of the lake under study. They help clarify important cause and effect relationships.

Morphometry: *Relating to a lake's physical structure (e.g. depth, shoreline length).*

2. **PREDICTIVE MODE: What will happen to the lake if we do this, that, or the other thing?** Models can be used to predict how lake water quality conditions will change in response to changes in nutrient inputs or other controlling factors. For practical reasons, it is usually infeasible to predict lake responses based on full-scale experimentation with the lake and its watershed. Instead, mathematical models permit experiments to be performed on paper or on computer.

Examples of questions that might be addressed via lake modeling include

- What did the lake look like before anyone arrived?
- What level of nutrient loading can the lake tolerate before it develops algae problems?

- How will future watershed development plans affect the lake's water quality?
- What are the most important sources of the lake's problems?
- What reduction in nutrient loading is needed to eliminate nuisance algal blooms in the lake?
- Once watershed or point-source controls are in place, how long will it take for lake water quality to improve?
- Given monitoring data collected in the lake and its watershed during a given year, what is the expected range of water quality conditions over several years?
- Given a water quality management goal (such as a target level of lake phosphorus, chlorophyll-*a*, or transparency) and an array of feasible control techniques, what is the probability that restoration efforts will be successful?
- Are proposed lake management goals realistic?

Models are not the only means of addressing these questions, and they do have limitations. For example, modeling is feasible only for evaluating those types of problems that are understood well enough to be expressed in concise, quantitative terms. In some situations, modeling may be infeasible or unnecessary. Why make a lake study more complicated than it has to be?

Models are not monoliths. They are rather frail tools used by lake management consultants in developing their professional opinions and recommendations. The consultant should decide which models (if any) are appropriate, what supporting data should be collected, how the models should be implemented, and how the model's results should be interpreted. Consider the following analogy:

HOME ADDITION	LAKE STUDY
Carpenter	Consultant
Tools	Modeling Techniques
Raw Materials	Monitoring Data

Different carpenters may prefer certain brands of tools to others. The selection of appropriate tools to accomplish a given job is important, but not the only factor determining the success or failure of a project. In home building, the quality of the addition depends less upon which tools are used than upon *how* they are used. The owner hires the carpenter, not his or her tools. The same is true for hiring a lake management consultant. Obviously, the quantity and quality of raw materials are every bit as important as the tools used on the job. The raw materials required for applying a model to a lake are monitoring data and other baseline information developed under diagnostic studies (Chapter 3).

For ease in explaining modeling concepts, English units are used in the examples in this chapter. Lake modeling is far less awkward, however, when metric units are used.

Eutrophication Model Framework

Phosphorus loading models are frequently used to evaluate eutrophication problems related to algae. These models link phosphorus loading to the average total phosphorus concentration in the lake water and to other indicators of water quality that are related to algal growth, such as chlorophyll and transparency (Fig. 4-1). Lake responses to phosphorus loading depend upon physical and hydrologic characteristics. Therefore, these models consider lake volume, average depth, flushing rate, and other characteristics in predicting lake responses to a given phosphorus load.

While the terms and equations involved may seem mystical, the underlying concepts are simple:

1. Lake algal growth is limited by the supply of phosphorus.

2. Increasing or decreasing the mass of phosphorus discharged into the lake over an annual or seasonal time scale will increase or decrease the average concentrations of phosphorus and algae in the lake.

3. A lake's capacity to handle phosphorus loadings without experiencing nuisance algal blooms increases with volume, depth, and flushing rate.

In other words, the lake's condition depends upon how much phosphorus it receives from both internal and external sources. A large, deep lake with a high flow will be able to handle a much greater phosphorus load without noticeable deterioration, compared with a small, shallow, or stagnant lake. Models summarize these relationships in mathematical terms, based upon observed water quality responses of large numbers of lakes and reservoirs.

Algal growth in these models is usually expressed in terms of mean, growing-season chlorophyll in the epilimnion concentrations. As discussed in Chapter 3, phosphorus, chlorophyll-*a*, and transparency help to define *trophic state*, a vague concept used to characterize lake condition. Other variables related to algal productivity, such as hypolimnetic oxygen-depletion rate, seasonal maximum chlorophyll-*a*, bloom frequency, or organic carbon, may also be considered in phosphorus loading models.

These methods cannot yet be used to predict aquatic weed densities, which generally depend more upon lake depth, the quantity and quality of lake bottom sediment, and light penetration than upon the loading of nutrients entering the lake from its watershed.

Eutrophication models rely heavily on the lake phosphorus budget, which is simply an itemized accounting of the inputs and outputs of phosphorus to and from the lake water column over a year or growing season. Although budgets can be constructed for other pollutants that cause lake problems (nitrogen, silt, organic matter, bacteria, or toxics, for example) phosphorus budgets are used most frequently. A phosphorus budget provides a means to evaluate and rank phosphorus sources that may contribute to an algal problem. The basic concept and mathematics are relatively simple, although the estimation of individual budget items often requires considerable time, monitoring data, and expertise.

Basic concepts involved in constructing phosphorus budgets and applying eutrophication models are described and illustrated in later sections of this chapter. In some situations, particularly in reservoirs, algal growth may be controlled by factors other than phosphorus, such as nitrogen, light, or flushing rate (Walker, 1985). Models appropriate for these situations are more complex than those discussed below, although the general concepts and approaches are similar.

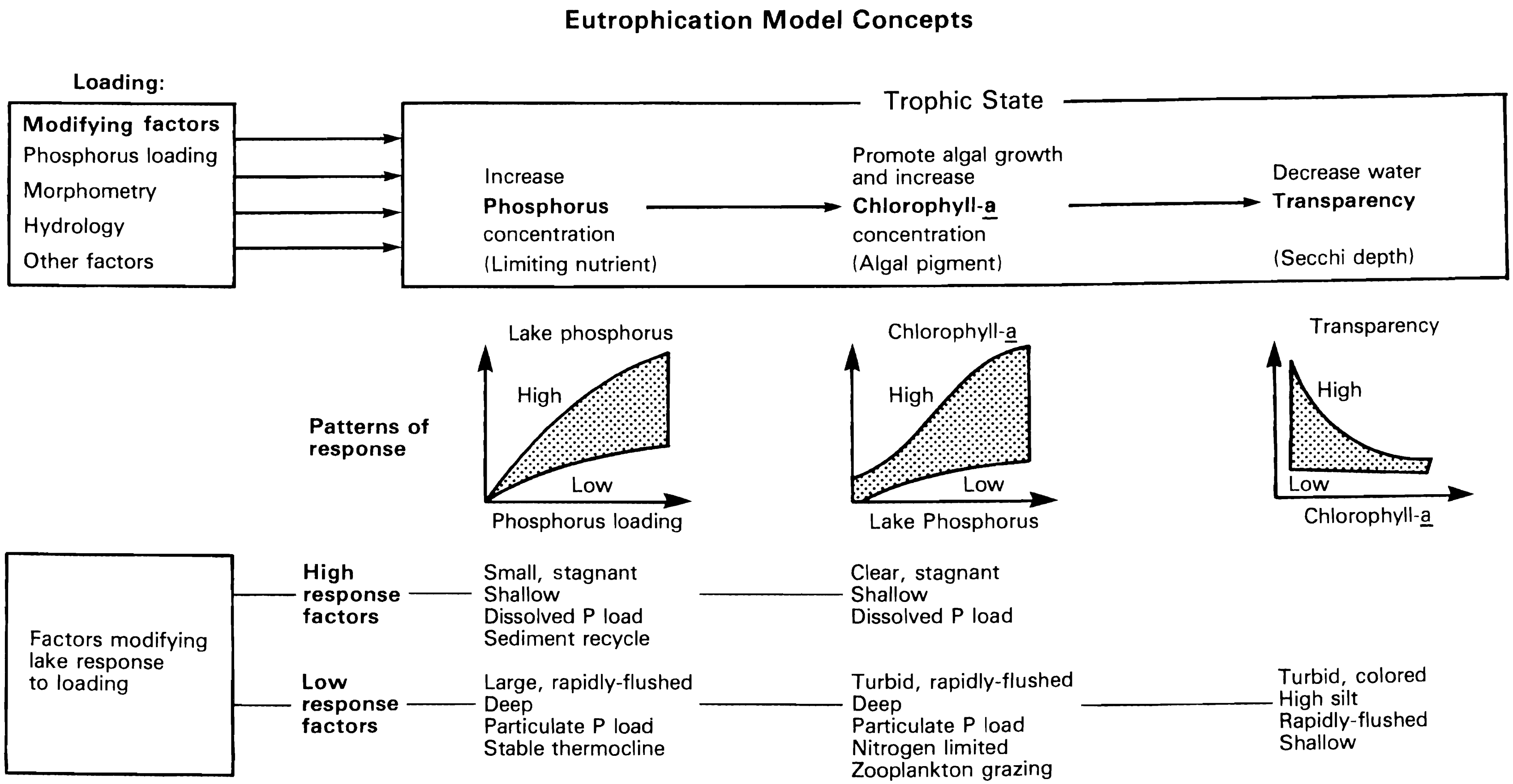


Figure 4-1.—Eutrophication modeling concepts.

Variability

Eutrophication models are geared to predicting *average* water quality conditions over a growing season or year. Unfortunately, this often gives the mistaken impression that water quality is fixed and does not vary in different areas or through time within a given lake. This is not the case. Averaging is typically done over three dimensions:

1. **Over Depth.** Generally within the surface, mixed layer. Vertical variations within the mixed layer are usually small.

2. **Over Sampling Stations.** Sampling station locations might be located in different places of the lake. In a small, round lake, the variations among sampling stations will tend to be insignificant, and one station is usually adequate. In a large lake with several embayments, in a long, narrow reservoir, or in a complex reservoir with several tributary arms, however, water quality may vary significantly from station to station (from oligotrophic to hypereutrophic). In such situations, the "average water quality" may be meaningless, and it may be appropriate to divide the lake or reservoir into segments for modeling purposes (outflow from one segment serves as inflow to the next).

3. **Over Season.** Phosphorus, transparency, and especially chlorophyll-*a* concentrations usually vary significantly at a given station from one sampling date to the next during the growing season. It is not unusual, for example, for the maximum chlorophyll-*a* concentration to exceed two to three times the seasonal average. Because the input data themselves represent values within a range of actual conditions, model outputs also should be considered to represent answers within a range. Thus, model calculations are generally reported as having a certain "percent confidence" to indicate the likelihood that the answer is correct within a given range.

In addition, since chlorophyll-*a*, phosphorus, and transparency vary during the season to begin with, a slight improvement or deterioration in these water quality characteristics is difficult to perceive. A model prediction that conditions would improve slightly, therefore, is not likely to represent a noticeable change in the lake. When the change becomes comparable to normal variations, it is easier to observe an improvement or deterioration.

Because of the above sources of variability, it is more realistic to consider measured or modeled water quality as a "smear" than as a "point." If a consultant says that a lake has a mean chlorophyll-*a* concentration of 10 ppb, for example, the actual mean may be 5 or 20 ppb, depending on monitoring frequency and lake variability. Perhaps more important, even if the seasonal mean is 10 ppb, 90 percent of the samples will be in the 2-to-24 ppb range for a lake with typical seasonal variability.

In a given watershed and lake, year-to-year variations in average water quality may be significant because of fluctuations in climatologic factors, particularly streamflows and factors controlling thermal stratification. Monitoring programs extending for a period of at least 3 years are often recommended to characterize this year-to-year variability and provide an adequate basis for lake diagnosis and modeling.

Another source of variability is model error. Statistical analyses of data from large numbers of lakes and reservoirs indicate that phosphorus loading models generally predict average lake responses to within a range of one to two times the average. Differences between observed and predicted water quality, in part, reflect variability in the data (loading estimates and observed lake responses) and inherent model limitations. Differences between observed (directly measured) and predicted (modeled) values may contain useful information for diagnostic purposes, however. Model projections of future conditions resulting from a change in phosphorus loading are more reliable when they are expressed in relative terms (percent change from existing conditions). A good lake and watershed monitoring program can reduce the risk of significant model errors, which may lead to false conclusions and poor management decisions.

Loading Concept

Loadings most accurately express the relative impacts of various watershed sources on lake water quality. For example, a stream with high phosphorus concentration will not necessarily be an important source to the lake, because the stream may have a very low flow and, therefore, contribute a relatively low annual loading.

Because lakes store nutrients in their water columns and bottom sediments, water quality responses are related to the total nutrient loading that occurs over a year or growing season. For this reason, water and phosphorus budgets are generally calculated on an annual or seasonal basis. Water and phosphorus residence times in the water column determine whether seasonal or annual budgets are appropriate for evaluation of a given lake.

Phosphorus loading concepts can be illustrated with the following analogy:

GROCERY BILL	**PHOSPHORUS LOADING**
Item	Source
Quantity	Flow
Unit Cost	Concentration
Cost of Item	Loading From Source
Total Cost of All Items	**Total Loading From All Sources**

The cost of a given item is determined by the quantity purchased and the unit cost. The total cost of all items purchased determines the impact on finances (lake water quality). Funds (lake capacity to handle phosphorus loading without water quality impairment) are limited. Therefore, intelligent shopping (managing the watershed and other phosphorus sources) is required to protect finances (lake water quality).

Loadings change in response to season, storm events, upstream point sources, and land use changes. For example, converting an acre of forest into urban land use typically increases the loading of phosphorus by a factor of 5-20. This results from increases in both water flow (runoff from impervious surfaces) and nutrient concentration (phosphorus deposition and washoff from impervious surfaces). The evaluation of loadings provides a basis for projecting lake responses to changes in land use or other factors.

The grocery bill analogy breaks down in at least one important respect: Shoppers can read the unit costs off the shelves. To estimate phosphorus loading from a given source, both flow and concentration must be quantified over annual and seasonal periods. This is difficult because both flow and concentration vary (much more than supermarket prices) in response to season, storm events, and other random factors. Flow should be monitored continuously in major streams. Concentration is usually sampled periodically (weekly, monthly) and preferably supplemented with samples taken during storms. This is why good lake and watershed studies cost so much. Particularly in small, flashy streams, a very high percentage of the annual loading may occur during short, intense storms. If these events are not sampled, it will be relatively difficult to develop reliable loading estimates.

Because of these factors, loading estimates for each source should be considered with a degree of skepticism. These are not fixed quantities but ranges. Depending upon monitoring intensity and calculation methods, an annual loading estimate for a given stream could be off by a factor of two or more. Where appropriate, monitoring intensity can be increased to provide better data for quantifying loadings, particularly in streams that are thought to be major contributors.

Water Budget

The first step in lake modeling is to establish a water balance. Flows carry pollutants into and out of lakes, and analyses of lake eutrophication and most other water quality problems cannot be conducted without a quantitative understanding of lake hydrology. The basic water balance equation considers the following flow terms, typically in units of acre-feet per year:

INFLOW + PRECIPITATION = OUTFLOW + EVAPORATION + CHANGE IN STORAGE

Water budget concepts are illustrated in Figure 4-2.

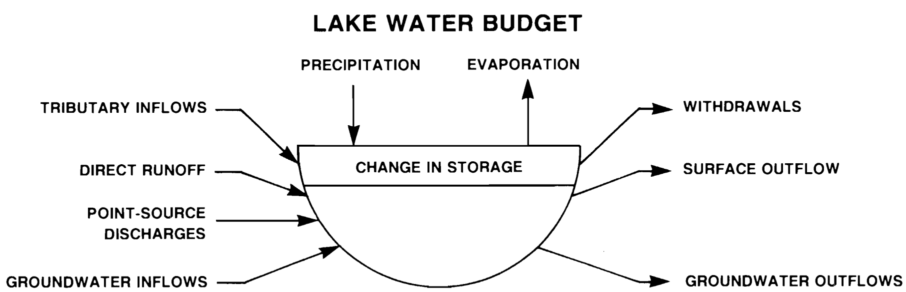


Figure 4-2.—Water budget schematic.

The data for the terms **INFLOW** and **OUTFLOW** should be evaluated over annual or seasonal periods. Inflows may include tributary streams, point-source discharges, runoff from shoreline areas, and groundwater springs. Outflows may include the lake outlet; groundwater discharges; and withdrawals for water supply, irrigation, or other purposes. Major inflow and outflow streams should be gaged directly. Indirect estimation procedures (for example, runoff coefficients) can be used to quantify smaller streams. Precipitation and evaporation can be derived from regional climatologic data. The **CHANGE IN STORAGE** accounts for changes in surface elevation over the study period, which is sometimes significant in reservoirs. This term is positive if lake volume increases over the study period, negative otherwise.

Once the flow terms have been estimated and tabulated, the water balance should be checked by comparing the total inflows (left side of equation) with total outflows (right side). Major discrepancies may indicate an omission or estimation error in an important inflow or outflow term (such as unknown or poorly defined streamflow or groundwater flow). Establishing water balances is relatively difficult in seepage lakes because of the problems and expense of monitoring groundwater flows. In any event, significant errors in the water balance may indicate a need for further study of lake hydrology.

To provide a complete accounting of the watershed, drainage areas should also balance (that is, the sum of the tributary drainage areas plus the lake surface area should equal the drainage area at the lake outlet).

Phosphorus Budget

The lake phosphorus budget (Fig.4-3) provides the cornerstone for evaluating many eutrophication problems. The following terms are evaluated and typically expressed in units of pounds per year:

INFLOW LOADING = OUTFLOW LOADING + NET SEDIMENTATION + CHANGE IN STORAGE

This equation summarizes fundamental cause and effect relationships linking watersheds, lake processes, and water quality responses.

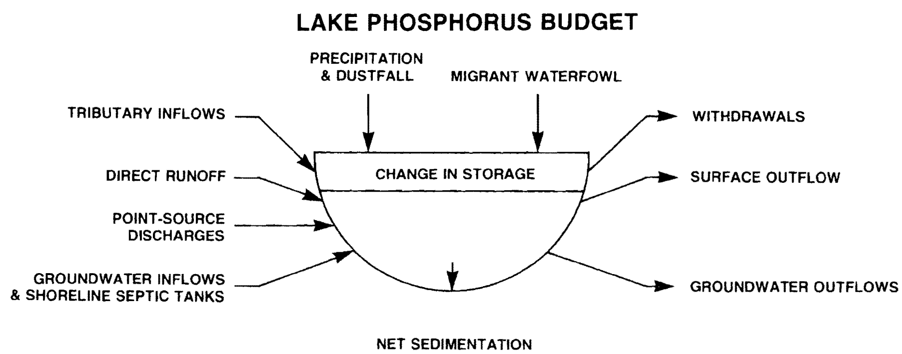


Figure 4-3.—Phosphorus budget schematic.

The **INFLOW LOADING** term is the sum of all external sources of phosphorus to the lake, which may include tributary inflows, point sources discharging directly to the lake, precipitation and dustfall, leachate from shoreline septic tanks, other groundwater inputs, runoff from shoreline areas, and contributions from migrant waterfowl. Estimation of individual loading terms is the most important and generally most expensive step in the modeling process. Investments in intensive monitoring programs to define and quantify major loading sources usually pay off in terms of the quality and reliability of project results. Monitoring of the lake itself is usually conducted during the same period so that loadings can be related to lake responses.

Stream loadings, usually the largest sources, are estimated from streamflow and phosphorus concentrations monitored over at least an annual period. Major tributaries should be sampled just above the lake over a range of seasons and flow regimes (including storm events) to provide adequate data for calculating loadings. In large watersheds, it may be appropriate to sample at several upstream locations so that contributions from individual point and nonpoint sources can be quantified. Special studies may be required to estimate groundwater input terms (for example, groundwater sampling and flow modeling, shoreline septic tank inventories). Loadings in runoff from shoreline areas and from relatively small, unsampled tributaries can be estimated indirectly, as discussed below. Loadings in precipitation and dustfall, usually relatively small, can be estimated from literature values or regional sampling data.

In many cases, indirect estimates of loading from a given stream or area can be derived from information on watershed characteristics. This method is based upon the concept that two watersheds in the same region and with similar land use patterns and geology will tend to contribute the same loading of phosphorus per unit area. This permits extrapolation of data from one or more monitored watersheds to others. "**Export Coefficients**" (lbs phosphorus/acre-yr) have been compiled for various land uses and regions (Chapter 2, see Table 2-1). The applicability of this method depends largely upon the quantity and quality of regional export coefficient data for the land uses and watersheds under study. This approach is much less costly than direct monitoring, but generally less reliable. It is frequently used in preliminary studies (to get a rough handle on the lake nutrient budget before designing and conducting intensive monitoring programs) and for estimating loadings from small watersheds whose contributions to the lake's total phosphorus budget are relatively insignificant.

The **OUTFLOW LOADING** term accounts for phosphorus leaving the lake in surface outlet(s); withdrawals for water supply, irrigation, or other purposes; and groundwater seepage. These are usually estimated by direct measurements of flow and concentration, as described above for stream loadings. If the lake outflow is dominated by groundwater seepage, it will be very difficult to determine the outflow loading term directly.

The **NET SEDIMENTATION** term accounts for the accumulation or retention of phosphorus in lake bottom sediments. It reflects the net result of all physical, chemical, and biological processes causing vertical transfer of phosphorus between the water column and lake bottom, as described in Chapter 2. For a given loading, lake water quality will generally improve as the magnitude of the sedimentation term increases because higher sedimentation leaves less phosphorus behind in the water column to stimulate algal growth. Because there are several complex processes involved and these vary spatially and seasonally within a given lake, it is generally infeasible to measure net sedimentation directly. Accordingly, this term is usually calculated by difference from the other terms or estimated using empirical models of the type discussed in the next section.

The **CHANGE IN STORAGE** term accounts for changes in the total mass of phosphorus stored in the lake water column between the beginning and end of the study period. Such changes would reflect changes in lake volume, average phosphorus concentration, or both. This term is positive if the phosphorus mass increases over the study period, negative otherwise.

As formulated above, the water and phosphorus budgets provide important descriptive information on factors influencing lake eutrophication. A useful format for presenting results of budget calculations is illustrated in Table 4-1, based on data from Lake Morey, Vermont. The table provides a complete accounting of drainage areas, flows, and loadings. The relative importance of various sources can be readily derived from the percentage calculations and accompanying pie charts. The mean concentrations (ppb), runoff (ft/yr), and export (lbs/acre-yr) provide means for comparing the unit contributions from various watersheds of different sizes. Often these values are sensitive to land uses, point sources, or geologic factors. For example, the relatively high export value for Pine Brook (.47 versus a range of .04-.21 lbs/acre-year for the other watersheds) reflects erodible soils. High export values for Aloha Camp and Bonnie Oaks Brooks reflect inputs from camp sewage treatment systems.

Comparing the magnitudes of the individual loading terms provides a basis for ranking sources and identifying possible candidates for watershed management or point source control techniques. For example, the Lake Morey phosphorus budget (Table 4-1) clearly indicates that sewering of shoreline areas would not be an effective means of reducing lake eutrophication because septic tanks currently account for less than 1 percent of the total loading.

If the net sedimentation term is unusually low (or negative) for a lake of the type being studied, it may indicate that bottom sediments are releasing significant quantities of phosphorus into the water column and thus that an in-lake

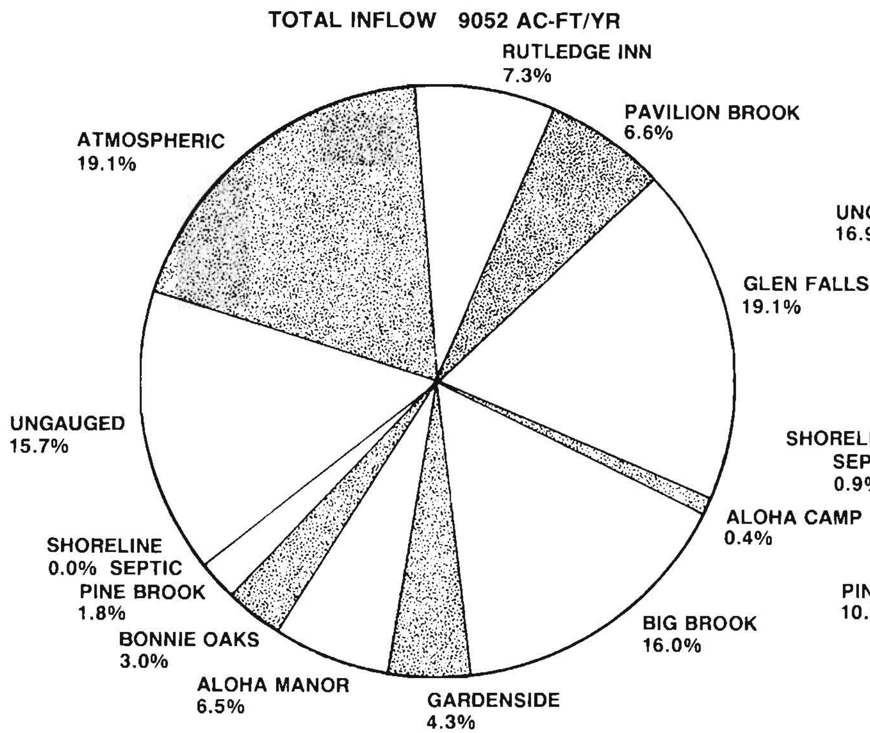
Table 4–1.—Water and total phosphorus (P) budgets for Lake Morey, Vermont

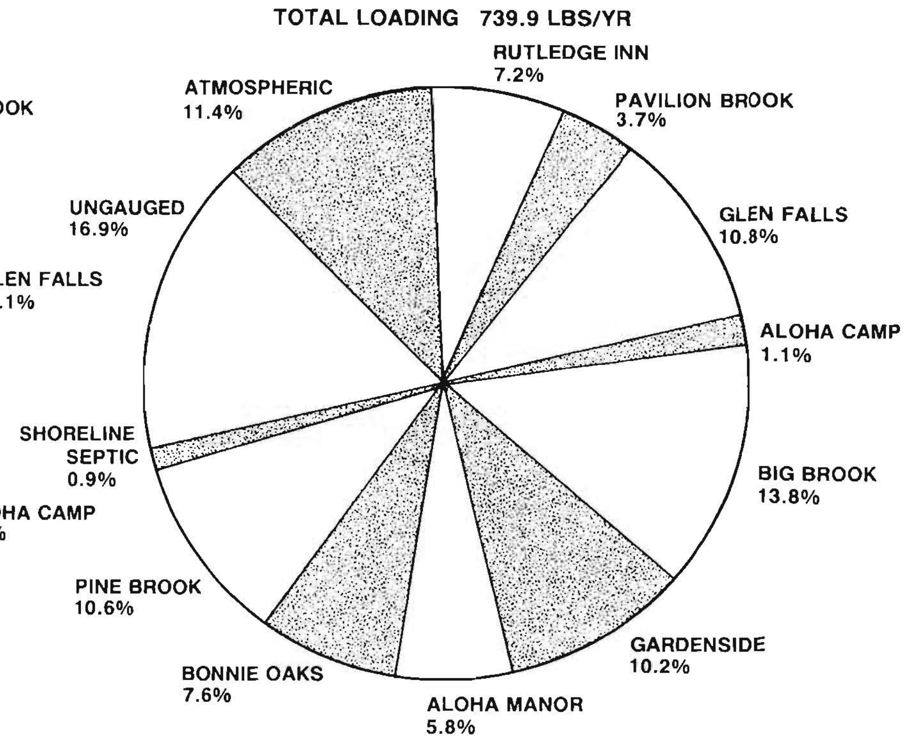
LAKE MOREY, VERMONT
FEB. 1, 1981 THROUGH DEC. 10, 1982

ITEM	DRAINAGE AREA (ACRES)	MEAN FLOW (AC-FT/YR)	WATER INFLOW (%)	TOTAL P LOADING (LBS/YR)	P INFLOW (%)	MEAN CONC. (PPB)	RUNOFF (FT/YR)	TOTAL P EXPORT (LBS/AC-YR)
Rutledge Inn Brook	435	664	7.3	53.5	7.2	30	1.53	0.080
Pavilion Brook	397	598	6.6	27.5	3.7	17	1.51	0.046
Glen Falls Brook	1049	1732	19.1	79.9	10.8	17	1.65	0.046
Aloha Camp Brook	134	41	0.4	8.1	1.1	74	0.30	0.201
Big Brook	908	1452	16.0	102.1	13.8	26	1.60	0.070
Gardenside Brook	237	390	4.3	75.2	10.2	71	1.65	0.193
Aloha Manor Brook	371	587	6.5	43.1	5.8	27	1.58	0.073
Bonnie Oaks Brook	179	272	3.0	56.1	7.6	76	1.52	0.206
Pine Brook	109	166	1.8	78.3	10.6	174	1.53	0.472
Shoreline Septic Systems		(negligible)		6.4	0.9			
Ungauged Direct Runoff	894	1423	15.7	125.0	16.9	32	1.59	0.088
Atmospheric	528	1727	19.1	84.7	11.4	18	3.27	0.049
Total Inflow	5239	9052	100.0	739.9	100.0	30	1.73	0.082
Evaporation		1183	13.1					
Outflow	5239	7769	85.8	639.5	86.4	30	1.48	0.082
Increase in Storage		0	0.0	−217.8	−29.4			
Water Balance Error	0	100	1.1					
Net Sedimentation				318.1	43.0			

WATER BALANCE EQUATION: WATER BALANCE ERROR = TOTAL INFLOW – EVAPORATION – OUTFLOW – INCREASE IN STORAGE

PHOSPHOROUS BALANCE EQUATION: NET SEDIMENTATION = TOTAL INFLOW – OUTFLOW – INCREASE IN STORAGE restoration technique such as sediment phosphorus inactivation (see Chapter 6) may be appropriate for lake restoration.





Lake Response Models

Having characterized water and phosphorus budgets under existing conditions, response models can be used to evaluate existing lake conditions and to predict changes in phosphorus, chlorophyll-*a*, and transparency likely to result from changes in phosphorus loading. Several empirical models have been developed for this purpose. These models are based on statistical analysis of monitoring data from collections of lakes and reservoirs.

Models vary with respect to applicability, limitations, and data requirements. The consultant's choice of appropriate models for a given lake or reservoir should be based on regional experience and professional judgment. The consultant should also consider how closely the impoundment characteristics (morphometry, hydrology, lake versus reservoir) reflect the characteristics of the lakes that were used to develop a model. It may be inappropriate, for example, to apply a model developed in a study of Canadian natural lakes to an Alabama reservoir with a very different set of conditions.

Eutrophication models are driven by three fundamental variables that are calculated from impoundment morphometry, water budgets, and phosphorus budgets:

(1) P_I = AVERAGE INFLOW PHOSPHORUS CONCENTRATION (PPB)

$$= \frac{\text{Total Phosphorus Loading (lbs/yr)}}{\text{Mean Outflow (acre-ft/yr)}} \times 368$$

This is the flow-weighted-average concentration of all sources contributing phosphorus to the impoundment. If there were no interactions with bottom sediments, the average inflow, lake, and outflow phosphorus concentrations would be approximately equal. This basic measure of inflow quality is the most important determinant of eutrophication response and is the most frequent focus of long-term management efforts. It is sensitive to watershed point and nonpoint sources.

(2) T = MEAN HYDRAULIC RESIDENCE TIME (YEARS)

$$= \frac{\text{Lake Volume (acre-ft)}}{\text{Mean Outflow (acre-ft/yr)}}$$

This variable approximates the average length of time water spends in a lake or impoundment before being discharged through the outlet. In other terms, it equals the time required for the lake to refill if it were completely drained. As residence time increases, interactions between the water column and bottom sediment have greater influences on water quality. For a given inflow concentration, phosphorus sedimentation usually increases and lake phosphorus concentration decreases with increasing residence time. At very short residence times (less than 1-2 weeks), algae may have inadequate time to respond to the inflowing nutrient supply.

Residence time: *Commonly called the hydraulic residence time—the amount of time required to completely replace the lake's current volume of water with an equal volume of "new" water.*

(3) Z = MEAN DEPTH (FEET)

$$= \frac{\text{Lake Volume (acre-ft)}}{\text{Surface Area (acres)}}$$

Other factors being equal, lakes and impoundments with shallower mean depths are generally more susceptible to eutrophication problems. Shallower lakes have higher depth-averaged light intensities to support photosynthesis and greater sediment/water contact, which can encourage nutrient recycling. Since they both increase with lake volume, mean depth and hydraulic residence time are typically correlated.

Models differ with respect to how these variables are combined in equations to predict lake or reservoir responses for nutrient loading.

One set of equations based on data from northern, natural lakes is presented in Table 4-2 to illustrate modeling concepts. These are not necessarily the "best" models to use in a given application, the choice of which should be left to the lake consultant.

Table 4–2.—Typical phosphorus loading model equations for Northern lakes.

$$\begin{array}{ccccccc} P_I & & P & & \text{Chl.–}\underline{a} & & \\ \text{INFLOW} & \xrightarrow[(1)]{} & \text{PHOSPHORUS} & \xrightarrow[(2)]{} & \text{CHLOROPHYLL–}\underline{a} & \xrightarrow[(3)]{} & \text{SECCHI TRANSPARENCY} \end{array}$$

(1) A model for predicting lake phosphorus concentration was developed by Larsen and Mercier (1976) and Vollenweider (1976):

$$P \text{ (ppb)} = \frac{P_I}{1 + T^{.5}}$$

This equation predicts that average lake phosphorus concentration, P, will increase in proportion to the inflow concentration and will decrease with increasing hydraulic residence time. At low residence times, phosphorus sedimentation is negligible, and the response is controlled primarily by inflow concentration.

(2) The simplest of the chlorophyll–a response models was developed by Carlson (1977):

$$\text{Chl.–}\underline{a} \text{ (ppb)} = .068\ P^{1.46}$$

This equation is similar to others developed from northern lake data by Dillon and Rigler (1974) and by Jones and Bachman (1978).

(3) A similar relationship was also developed by Carlson (1977) to predict Secchi disk transparency:

$$\text{Secchi (meters)} = 7.7\ \text{Chl–}\underline{a}^{-.68}$$

This equation is appropriate for lakes and reservoirs in which transparency is controlled primarily by algae. It will overestimate transparency in impoundments with relatively high concentrations of inorganic suspended solids, silt, or color.

Two of the equations are based on the Trophic State Index (TSI) developed by Carlson (1977). This system, used by many States for classification purposes, is essentially a rescaling of phosphorus, chlorophyll-*a*, and transparency measurements in units that are consistent with northern lake behavior (Fig.4-4). The index provides a common frame of reference for comparing these measurements. The scale is defined so that a decrease of index units corresponds to a doubling of transparency.

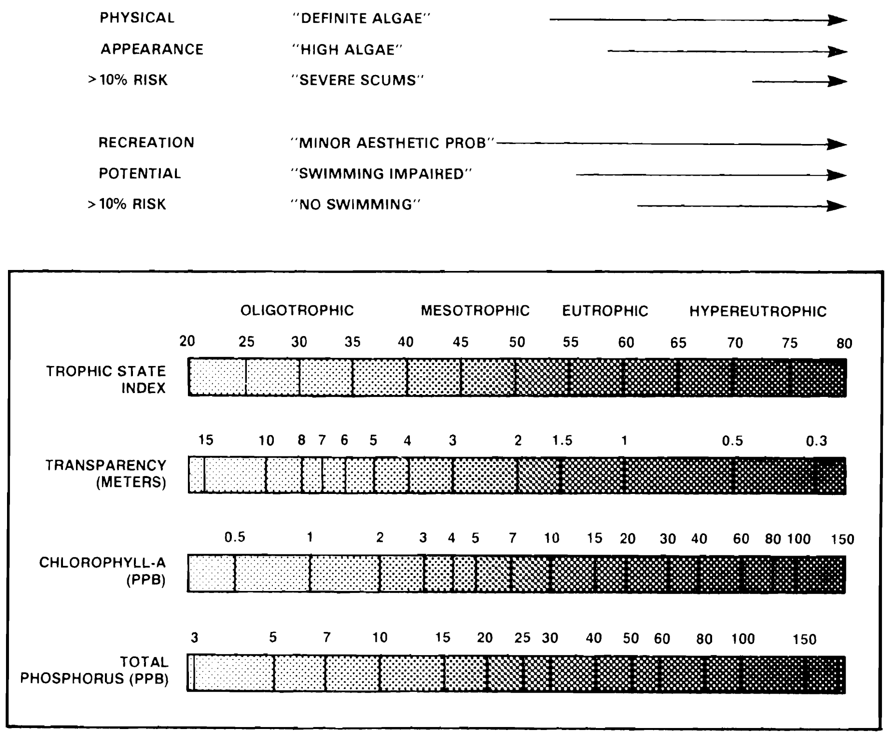


Figure 4-4. – Carlson's Trophic State Index related to perceived nuisance conditions (Heiskary and Walker, 1987). Length of arrows indicate range over which a greater than 10 percent probability exists that users will perceive a problem.

Carlson's index can be used to predict values of one variable from measurements of another. For example, a lake with a measured mean transparency of 6.6 feet (2 meters) would have a TSI of 50. Based on the scales in Figure 4-4, a mean chlorophyll-*a* of 7 ppb and a mean total phosphorus concentration of 23 ppb would also be expected for this lake. These predictions are approximate, however (good roughly to within a factor of two, assuming that the lake under study is typical of other northern lakes).

Various factors influence relationships among phosphorus, chlorophyll-*a*, and transparency (Fig.4-1). Carlson's equations reflect relatively high chlorophyll-*a* and transparency responses found in northern, natural lakes. Turbid, rapidly flushed impoundments tend to have lower responses and less sensitivity to phosphorus loading.

Heiskary and Walker (1987) describe a methodology for relating lake "trophic state," as measured by phosphorus, chlorophyll-*a*, or transparency, to user-perceived impairment in aesthetic qualities and recreation potential. The arrows in Figure 4-4 indicate measurement ranges in which the risk of perceived nuisance conditions (for example, "Swimming Impaired" or "High Algae") exceeds 10 percent, based on surveys of Minnesota Lakes. These ratings may vary regionally.

Figure 4-5 provides additional perspectives on the relationship between impoundment phosphorus concentrations and eutrophication responses, as measured by mean chlorophyll-*a* and transparency. The figure is based on cross-tabulations of median total phosphorus, mean chlorophyll-*a*, and mean

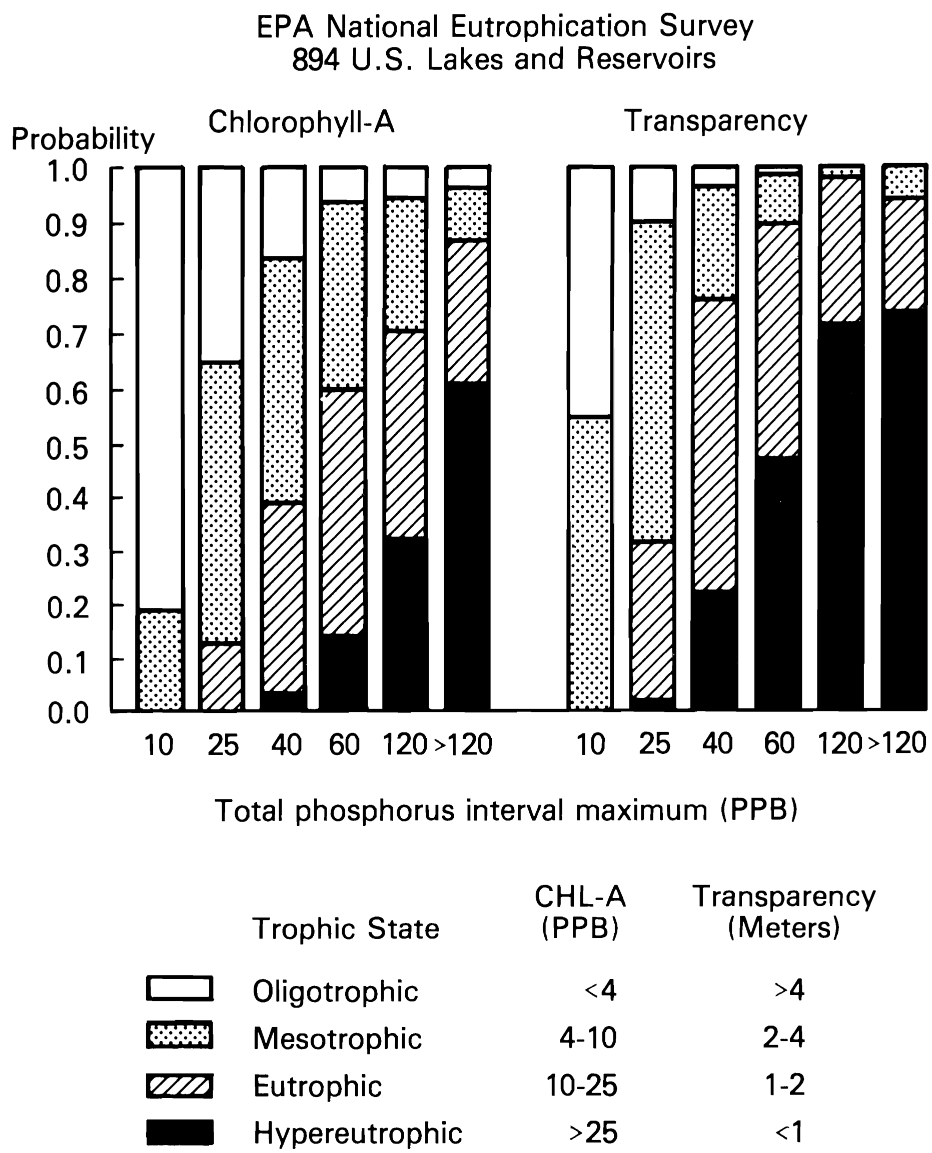


Figure 4-5.—Responses of mean chlorophyll-a and transparency to phosphorus.

transparency values from 894 U.S. lakes and reservoirs (U.S. Environ. Prot. Agency, 1978). Phosphorus values are classified into six intervals (0-10, 10-25, 25-40, 40-60, 60-120, 120 ppb), and the probabilities of encountering mean chlorophyll-*a* and transparency levels in oligotrophic, mesotrophic, eutrophic, and hypereutrophic ranges have been calculated for each phosphorus interval. For example, if phosphorus is in the 25-40 ppb range, the probability of encountering a mean chlorophyll-*a* in the eutrophic range (10 ppb) is about .4, or 40 percent, and the probability of encountering a mean transparency less than 6.6 feet (2 meters) is about .75, or 75 percent. Variations in the response factors such as depth, flushing rate, or turbidity (see Fig. 4-1) contribute to the distribution of chlorophyll-*a* and transparency that can be expected for a given phosphorus load.

Flushing rate: *The rate at which water enters and leaves a lake relative to lake volume, usually expressed as time needed to replace the lake volume with inflowing water.*

Tracking Restoration Efforts

Figure 4-6 illustrates a type of "phosphorus loading diagram" often used to depict modeling results (Vollenweider,1976). This diagram is developed by solving the equation for phosphorus concentrations from the Secchi depth of inflowing waters and the hydraulic residence time (Equation 1 in Table 4-2.) The dotted lines (representing phosphorus concentrations of 10, 25, and 60 ppb) are not sharp boundaries of lake condition, but roughly delineate trophic state categories based on average phosphorus concentrations. Corresponding chlorophyll-*a* and transparency probabilities can be derived from Figure 4-5. The object of the game is to move the lake away from the "HYPEREUTROPHIC" (northeast) corner and toward the "OLIGOTROPHIC" (southeast) corner in Figure 4-6, usually by reducing watershed point or nonpoint sources and decreasing the average inflow phosphorus concentration (y-axis).

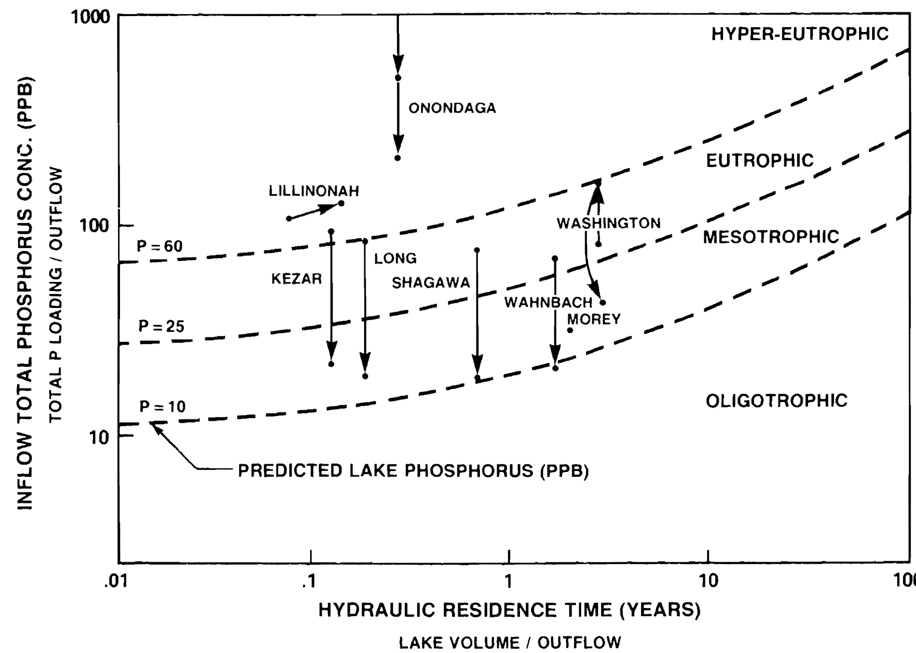


Figure 4-6.—Restoration efforts tracked on Vollenweider's (1976) phosphorus loading diagram.

The paths of eight documented restoration efforts are also plotted in Figure 4-6, based upon data summarized in Table 4-3. These case studies provide a context for illustrating important modeling concepts. Figure 4-7 plots measured mean phosphorus, chlorophyll-*a*, and transparency for each lake and time period. These are compared with predicted values derived from the models in Table 4-2. The predictions are driven by the inflow concentrations and hydraulic residence times listed in Table 4-3. These comparisons illustrate model capabilities to predict lake conditions before and after each restoration activity.

Figure 4-8 summarizes measured phosphorus budget information (inflow, inflow-lake, and lake concentrations) for each case and time period. The difference between the inflow and lake concentrations approximately reflects the net influence of bottom sediments as a phosphorus sink (positive) or source (negative) during each time period.

Each of the following sections discusses a particular case study.

Table 4–3.—Data for restoration cases discussed in Chapter 4. These data were used to plot the progress of restoration efforts on the Vollenweider curve shown in Figure 4-6

RESTORATION CASE HISTORIES

LAKE OR RESERVOIR LOCATION IMPOUNDMENT TYPE	DURATION AND TYPE OF RESTORATION ACTIVITY	YEARS	OBSERVED WATER QUALITY RESPONSES INFLOW P (PPB)	LAKE P (PPB)	CHL-A (PPB)	SECCHI (METERS)	HYDRAULIC RES. TIME (YEARS)	MEAN DEPTH (FEET)	SURFACE AREA (ACRES)
Lake Washington		1957	94	26	13	2.2	2.84	108.0	21634
Washington[1]	1963-68 Point Source Diversion	1963	155	62	35	1.0			
Natural Lake		1978	48	19	3	6.4			
Onondaga Lake	1971 Detergent Ban & Sewer Repairs	1970	3667	2310		2.1	0.28	39.4	2889
New York[2]	1979-81 Point Source Treatment	1974	509	382	45	1.2			
Natural Lake		1985	224	143	43	0.9			
Long Lake	1978 Point Source Treatment	1972-77	85	71	15		0.19	47.9	5136
Washington[3]		1978-82	22	18	8				
Reservoir									
Shagawa Lake	1973 Point Source Treatment	1971-72	79	55	28	1.9	0.70	18.7	2272
Minnesota[4]		1974-78	20	35	26	2.4			
Natural Lake									
Kezar Lake	1970 Point Source Treatment	1970	95	70	32	1.0	0.13	9.2	180
New Hampshire[5]	1981 Point Source Diversion	1981	24	30	18	1.4			
Natural Lake	1984 Hypolimnetic Alum Treatment	1985	24	16	5	3.0			
Lake Morey	1986 Hypolimnetic Alum Treatment	1981	35	30	10	4.4	1.93	27.6	543
Vermont[6]		1986	35	12	3	6.0			
Natural Lake									
Wahnbach Reservoir	1977 River Inflow Treatment	1969-70	73	25	18	3.0	1.70	59.0	558
West Germany[7]		1978-79	21	8	5	6.0			
Reservoir									
Lake Lillinonah	1977 Point Source Treatment	1976	119	65	35	1.1	0.08	39.0	1899
Connecticut[8]		1977	136	68	33	1.6	0.14		
Reservoir									

Sources: [1] Edmonson and Lehman (1981)
[2] Devan and Effler (1983); Walker (1977)
[3] Soltero and Nichols (1984)
[4] Larsen and Malueg (1980)
[5] Connor and Smith (1983, 1986); Connor and Martin (1986)
[6] Walker (1983); Smeltzer and Swain (1985); Smeltzer (1987)
[7] Bernhardt (1980)
[8] Jones and Lee (1981)

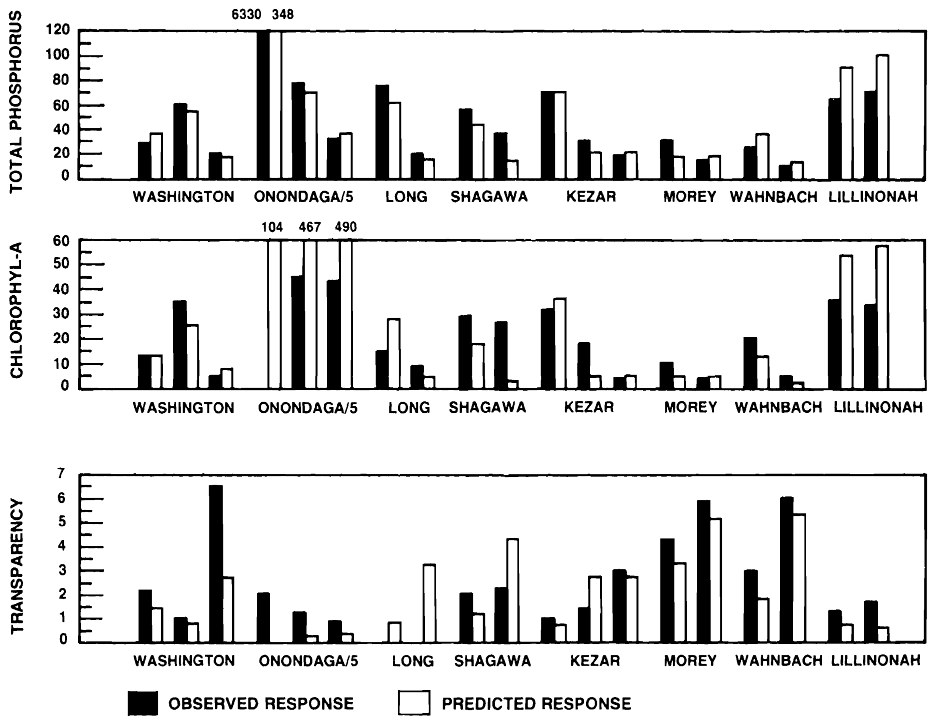


Figure 4-7.—Observed and predicted responses to restoration efforts.

Lake Washington, Washington: "You Should Be So Lucky!"

Between 1957 and 1963, eutrophication progressed with increasing sewage loadings from Metropolitan Seattle. Between 1963 and 1968, sewage discharges were diverted out of the lake basin, reducing the total phosphorus loading to the lake by 69 percent, relative to 1963. Observed and predicted conditions in 1978 reflect dramatic improvements in water quality that followed within a year or two after the sewage diversion. Observed phosphorus concentrations agree well with model predictions for each time period. Decreases in chlorophyll and increases in transparency were somewhat more dramatic than predicted by the models. Lake Washington is perhaps the most successful and fully documented lake restoration project to date.

Onondaga Lake, New York: "Far Out. 93 Percent Is Not Enough."

Onondaga received primary treated sewage from Syracuse for many years. Between 1970 and 1985, phosphorus loadings were reduced by over 93 percent as a result of a phosphorus detergent ban, combined sewer repairs, and tertiary treatment for phosphorus removal. Lake phosphorus levels responded in proportion to loading reductions and in agreement with model predictions (Fig. 4-7). No significant improvements in chlorophyll-*a* or transparency were achieved, however.

The lack of algal response reflects the fact that pre- and postrestoration phosphorus levels were extremely high (exceeding 100 ppb; note the scale factor of 5 for this lake in Figs.4-7 and 4-8). Phosphorus usually does not limit algal growth in this concentration range, particularly in deeper lakes. The chlorophyll model (Equation 2 in Table 4-2) does not apply and substantially overpredicts algal concentrations. Despite the substantial loading reductions as of 1985, Onondaga remained well within the hypereutrophic region of Figure 4-6 and on the flat portion of the chlorophyll response curve shown in Figure 4-1.

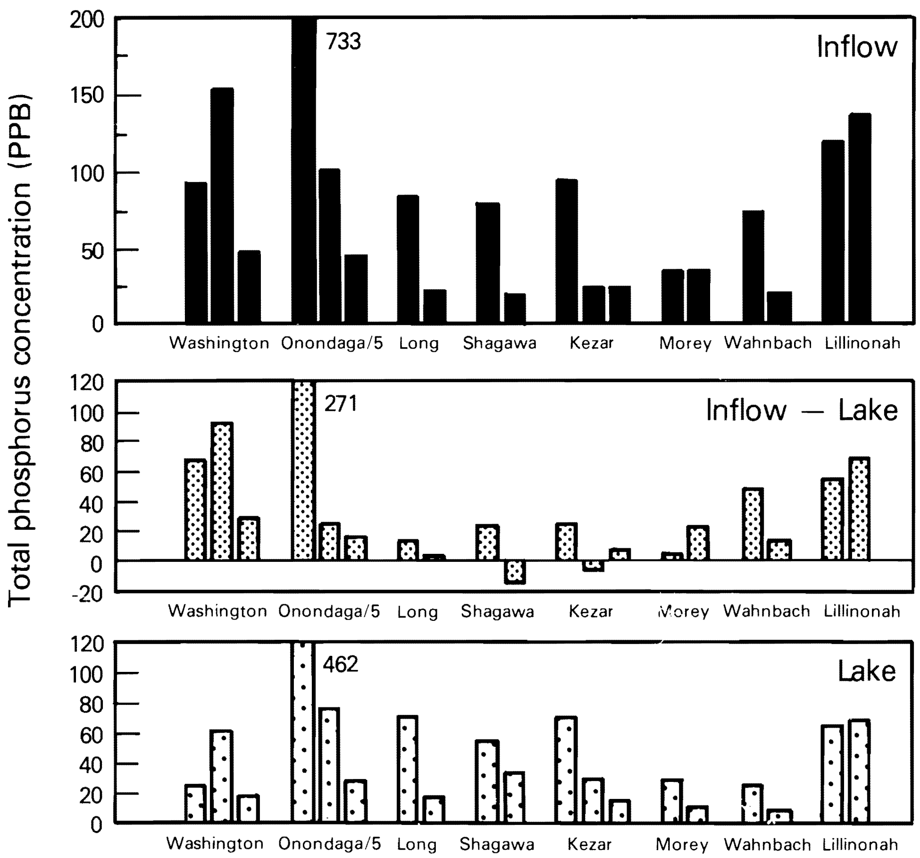


Figure 4-8.—Observed responses of phosphorus budget components to restoration efforts.

Onondaga illustrates the fact that some lakes subject to point-source phosphorus discharges may be susceptible to nuisance algal growths, even with tertiary treatment to remove phosphorus. Although chlorophyll and transparency did not respond, the disappearance of severe blue-green algal blooms following the loading reductions was a significant water quality improvement.

Why didn't Onondaga Lake respond like Lake Washington? It started off in much worse shape (Fig. 4-6). Onondaga has much shorter hydraulic residence time (.28 versus 2.8 years) and, therefore, less opportunity for phosphorus sedimentation. The loading plot (Fig. 4-6) essentially captures the relative responses of these two lakes to restoration efforts.

Long Lake, Washington: "What's This? Reservoir Restoration?"

Beginning in 1978, tertiary treatment of sewage from Spokane reduced the average seasonal phosphorus loading to this 22-mile-long reservoir on the Spokane River by 74 percent. This impoundment has a relatively short hydraulic residence time (.19 years, or 70 days). Accordingly, the inflow and reservoir phosphorus concentrations are similar, and the sedimentation term is relatively small (Fig. 4-8). Reservoir phosphorus levels responded roughly in proportion to the loading. Mean chlorophyll-*a* concentrations were reduced by 45 percent and were apparently less sensitive to the phosphorus loading reductions than predicted by Equation 2 in Table 4-2. Northern lake models (such as Equation 2) tend to overestimate chlorophyll-*a* sensitivity to phosphorus in some reservoirs because of effects of algal growth limitation by flushing and light (Walker, 1982,1985).

Shagawa Lake, Minnesota: "The Little Lake That Couldn't."

During 1973, external phosphorus loadings to this northern Minnesota lake were reduced by 75 percent via point source treatment. Although average lake phosphorus levels during ice-free seasons were reduced by 35 percent, mean chlorophyll-*a* and transparency did not respond according to model predictions (Fig. 4-7). The lack of response has been attributed to phosphorus releases from bottom sediments. These releases reflect historical loadings and the high susceptibility of this relatively shallow lake to hypolimnetic oxygen depletion and wind mixing. The fact that lake phosphorus exceeded the inflow concentration during the postrestoration period (Fig. 4-8) is indicative of sediment phosphorus release.

Despite the fact that the phosphorus loading diagram (Fig. 4-6) places Shagawa Lake at the oligo-mesotrophic boundary following load reductions, mean chlorophyll-*a* concentrations remained in the hypereutrophic range during the first few years following loading reductions. Over time, the rate of phosphorus release from bottom sediments may eventually decrease and permit the lake to respond to the change in loading. This case points out the fact that loading models of the type demonstrated here do not account for unusually high sediment phosphorus release rates, which may defer lake responses to changes in external loading.

Kezar Lake, New Hampshire: "The Little Lake That Could (With a Little Help)," Or "Shagawa Revisited . . ."

This shallow, rapidly flushed lake was subject to a municipal sewage discharge and in hypereutrophic condition for many years. Following installation of phosphorus removal facilities in 1970 and, eventually, complete elimination of the discharge in early 1981, the external loading was reduced by about 75 percent. Like Shagawa, the lake phosphorus concentration exceeded average inflow concentration during the initial period following loading reduction (Fig. 4-8). Kezar

Lake (maximum depth = 27 feet) was thermally stratified in 1981. Significant accumulations of phosphorus released from thick, phosphorus-rich bottom sediments accompanied depletion of oxygen from the hypolimnion. Surface algal blooms (chlorophyll-*a* = 60 ppb) were experienced during August 1981 and were apparently triggered by escape of hypolimnetic phosphorus into the mixed layer.

Because of sediment phosphorus releases, responses of lake phosphorus, chlorophyll-*a*, and transparency to the 1981 sewage diversion were less dramatic than predicted by the models (Fig. 4-7). In 1984, a hypolimnetic alum treatment was conducted to address the sediment nutrient release problem. Monitoring data from 1985 indicate that phosphorus, chlorophyll-*a*, and transparency levels responded in agreement with model predictions following the alum treatment. This case illustrates use of both watershed (point source control) and in-lake restoration (alum treatment) techniques to deal with a lake problem. Decreases in transparency following 1985 indicate that the book is not yet closed on Kezar Lake, however.

Lake Morey, Vermont: "Strange Mud. . ."

Morey is a resort lake sheltered in the mountains of eastern Vermont. Aside from the shoreline, the watershed is largely undeveloped. From the late 1970's to 1985, severe algal blooms and user complaints were experienced at increasing frequency. Summer mean chlorophyll-*a* concentrations ranged from 8 to 30 ppb, transparencies ranged from 2 to 5 meters, and spring phosphorus concentrations ranged from 17 to 48 ppb. These variations in water quality could not be explained by changes in land use, other watershed factors, or climate. Peak algal concentrations were usually found in the metalimnion and were supplied by phosphorus released from bottom sediments during periods of summer and winter anoxia. The hypolimnion was relatively thin (mean depth = 7 feet) and covered approximately 59 percent of the lake surface area. Bottom waters lost their dissolved oxygen early in June and remained anaerobic through fall overturn.

A 2-year intensive study indicated that large quantities of phosphorus were stored in the lake water column and sediments. At peak stratification in August 1981, for example, the total mass of phosphorus in the water column was about five times the annual phosphorus loading from the watershed. Phosphorus balance calculations (see Table 4-1) indicated that the lake inflow and outflow concentrations were approximately equal, despite the relatively long hydraulic residence time of 1.93 years. Equation 1 (Table 4-2) predicts that a lake with this residence time should trap 58 percent of the influent phosphorus. Study results indicated that Lake Morey was particularly susceptible to phosphorus recycling from bottom sediments because of its shape (broad, thin hypolimnion susceptible to rapid oxygen depletion) and iron-poor sediments (Stauffer,1981).

Model predictions for the Lake Morey prerestoration period were substantially below observed values of phosphorus and chlorophyll-*a* (Fig. 4-7). This reflects the fact that phosphorus retention capacity was unusually low. Observed transparency was higher than predicted, however, because of the tendency for algae to concentrate in the metalimnion, below the mixed layer where transparencies were measured.

Because the phosphorus budget indicated that the Morey's problems were primarily related to internal recycling and not to watershed loadings, a hypolimnetic alum treatment was conducted during early summer of 1986. The treatment reduced average phosphorus and chlorophyll-*a* concentrations during the period following treatment down to levels that were consistent with model predictions. Despite no significant changes in external loadings, the alum treatment apparently restored Lake Morey to a mesotrophic status, consistent with its position on the phosphorus loading diagram (Fig. 4-6).

The longevity of the treatment remains to be evaluated through future monitoring. This is an example of how phosphorus budgets can be used to diagnose lake problems, regardless of whether or not the solutions involve reductions in external loading. Sewering of shoreline areas (a restoration activity previously proposed and on the drawing boards for Lake Morey) would have had little impact.

Wahnbach Reservoir, Germany: "When All Else Fails . . ."

Wahnbach Reservoir, a water supply for Bonn, Germany, was subject to high phosphorus loadings from agricultural runoff and municipal point sources during the period prior to 1977. The resulting severe blooms of blue-green algae that developed in the reservoir caused major problems for the water supply. For various reasons, the loadings from the watershed were largely uncontrollable. In response to this problem, a detention basin and treatment plant were constructed at the major inflow to the reservoir in 1977. The treatment plant was designed to remove more than 95 percent of the phosphorus inflow via sedimentation, precipitation, flocculation with iron chloride, and direct filtration. Operation of this plant reduced the average inflow phosphorus concentration to the entire reservoir by about 71 percent.

As illustrated in Figures 4-6 and 4-7, the inflow treatment restored Wahnbach Reservoir from eutrophic to oligotrophic status during 1978-1979. Observed and predicted lake phosphorus concentration dropped below 10 ppb. Chlorophyll-*a* concentrations are consistently overestimated by the model, although the relative reduction in chlorophyll-*a* is correctly predicted. This relatively extreme and costly restoration measure was justified in relation to the severe impacts of eutrophication on drinking water quality and water treatment economics.

Lake Lillinonah, Connecticut: "You Can't Fool Mother Nature . . ."

Data from this 10-mile impoundment on the Housatonic River in Connecticut illustrate the sensitivity of some reservoirs to hydrologic fluctuations. During 1977, phosphorus removal was initiated at a municipal point source above the reservoir. This program reduced phosphorus loading from the point source by 51 percent and reduced total loading to the reservoir by 8 percent during 1977.

Compared to the case studies discussed above, this loading reduction was relatively small, and a major change in reservoir water quality would not be anticipated. In fact, observed and predicted phosphorus and chlorophyll-*a* concentrations were slightly higher during 1977 (Fig. 4-7). The concentrations increased primarily because the flow through the reservoir decreased by about 43 percent during 1977. As indicated by Equation 1 (see Table 4-2), the average inflow concentration is the most important variable determining phosphorus predictions, particularly in reservoirs with low hydraulic residence times. Inflow concentration is determined from the ratio of loading to outflow. The inflow concentration increased by 14 percent in 1977 because the small decrease in loading was more than offset by the decrease in flow.

For both time periods, the models overestimate reservoir phosphorus and chlorophyll-*a* concentrations and underestimate transparency. Apparently, phosphorus sedimentation in the Lillinonah was somewhat greater than predicted by Equation 1. This is not unusual for long and narrow reservoirs with high inflow phosphorus concentrations (Walker, 1982,1985). The loading plot (Fig. 4-6) correctly predicts a hypereutrophic status for the Lillinonah during both monitoring years.

Monitoring over a longer time period that includes years with flows similar to those experienced during 1976 would be required to track the response of the reservoir to the phosphorus loading reduction. Because the loading reduction is relatively small, impacts may be difficult to detect in the context of year-to-year variations. More substantial reductions in upstream point or nonpoint loadings, or both, would be required to restore the reservoir to a eutrophic or mesotrophic level.