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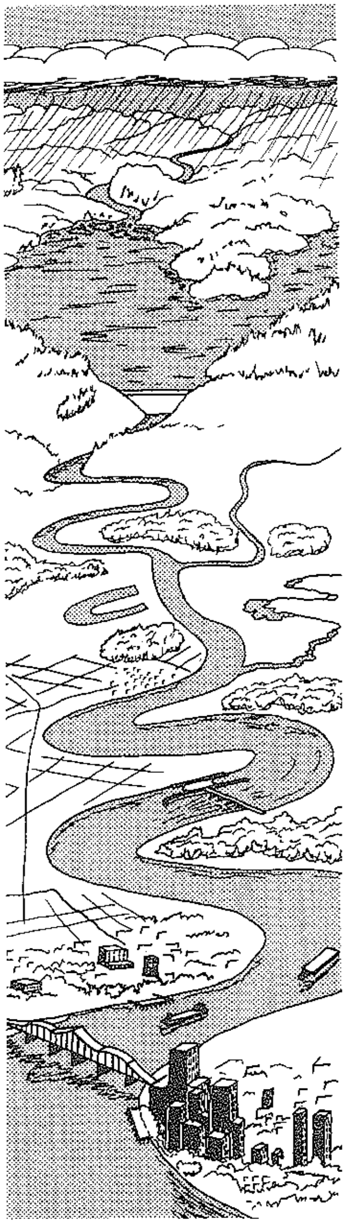

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WATER QUALITY MANAGEMENT FOR RESERVOIRS AND TAILWATERS

Report 1

IN-RESERVOIR WATER QUALITY MANAGEMENT TECHNIQUES

by

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Reservoirs are a valuable national resource and provide flood control, water supply, and hydroelectrical, navigational, recreational, and aesthetic benefits. Human influences threaten the quality of this resource, and appropriate protection and enhancement methods must be developed. Of particular concern are the consequences of eutrophication, the process by which lakes and reservoirs are enriched with sediment, organic matter, and plant nutrients. Elevated nutrient concentrations, rooted plant infestations, reduced transparency, excessive algal growth, reduced dissolved oxygen concentrations in bottom waters, and elevated concentrations of organics in finished drinking water are symptoms of this process.

Available in-reservoir methods for the management of these eutrophication-related problems are reported here. Also described are methods for problem diagnosis and the

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selection of appropriate methods. The manual is intended for water quality managers with a variety of scientific and engineering backgrounds. Detailed reference sections provide sources of additional information for each method.

PREFACE

This report, which is the first in a series of reports describing water quality management technology for reservoirs and tailwaters, was prepared by Dr. G. Dennis Cooke, Department of Biological Sciences, Kent State University, Kent, OH, with Dr. Robert H. Kennedy, Aquatic Processes and Effects Group (APEG), Ecosystem Research and Simulation Division (ERSD), Environmental Laboratory (EL), WES. It provides information concerning various in-reservoir water quality management techniques and was prepared as part of the Water Quality Management for Reservoirs and Tailwaters Demonstration of the Water Operations Technical Support (WOTS) Program. Subsequent reports will describe structural and operational water quality management techniques. The WOTS Program was sponsored by the Headquarters, US Army Corps of Engineers (HQUSACE). Technical Monitors for WOTS were Mr. David Buelow, Dr. John Bushman, and Mr. James L. Gottesman, HQUSACE. Program Managers of WOTS were Dr. J. L. Mahloch and Mr. J. L. Decell, both of EL.

The report was prepared under the general supervision of Dr. Thomas L. Hart, Chief, APEG, and Mr. Donald L. Robey, Chief, ERSD. Dr. John Harrison was Chief, EL. Drs. J. Barko, J. Mahloch, H. Westerdahl, K. Getsinger, R. Engler, T. Wright, and H. Smith, EL; Messrs. E. Theriot and L. Decell, EL; and Mr. S. Wilhelms, HL, provided technical review. The report was edited by Ms. Jessica S. Ruff of the Information Technology Laboratory, WES.

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WATER QUALITY MANAGEMENT FOR RESERVOIRS AND TAILWATERS

IN-RESERVOIR WATER QUALITY MANAGEMENT TECHNIQUES

PART I: INTRODUCTION

The water quality, lifespan, and usefulness of many reservoirs are increasingly threatened by an environmental problem called eutrophication. Eutrophication is the process of the excessive addition of silt, organic matter, and plant nutrients to lakes and reservoirs at rates sufficient to bring about increased production of algae and rooted plants and decreased reservoir volume. Associated with this process are the symptoms of eutrophication: weed-choked shallow areas, "blooms" of algae, low dissolved oxygen in deep waters and discharges, impaired drinking water, accumulation of bottom sediments, and the elimination of a coldwater fishery.

Eutrophication has many consequences. This process will degrade potable water supplies, limit or terminate recreation, alter fishery resources, reduce storage capacity, and harm downstream biological communities. Because there is a high demand for clean water, and because standards for potable and recreational water supplies are increasing, we can expect that eutrophication will produce significant changes in costs to industrial and/or domestic institutions that must clean the water before its use. As well, the costs of reservoir construction and operation could steadily increase to meet standards imposed by regulations and statutes or by user concerns.

The purpose of this manual is to provide the reservoir manager and operations personnel a compendium of experimental and proven methods that can be used in dealing with eutrophication-related conditions. To assist in the evaluation of the various methods, a basic description of limnological processes in reservoirs as they pertain to the eutrophication process and general guidance for diagnosis of eutrophication problems are contained in Part II. The subsequent parts on reservoir water quality management and control methods are divided into methods related to the control of nutrient concentrations and algae and methods of plant biomass control. Each of these parts states the problem addressed and discusses the theory or design of the method, application procedures, effectiveness, costs, and the limitations and concerns. Each part contains a summary and a listing of references that will lead the reader to the primary sources of information.

A manual of this nature can only be an outline or a guide. It will allow reservoir water quality personnel to gain a general understanding of the control and management techniques that are available to address specific eutrophication problems. This manual is not a substitute for the knowledge required of a particular limnological process or a restoration procedure before implementation. Details of effectiveness, applicability, costs, and other factors must be obtained. In every case, however, the reader will be referred to primary sources of information so that a detailed understanding and management plan can be developed.*

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* The reader may also find three US Army Corps of Engineers Information Exchange Bulletins to be of great value: the Water Operations Technical Support (WOTS) Program, the Aquatic Plant Control Research Program (APCRP), and the Natural Resources Research Program (RECNOTES). These periodicals may be obtained by writing to US Army Engineer Waterways Experiment Station, Environmental Laboratory, Vicksburg, MS 39181-0631 (601-634-3657).

PART II: RESERVOIR LIMNOLOGY, PROBLEM DIAGNOSIS, AND SELECTION OF MANAGEMENT METHODS

Reservoir Limnology

Introduction

Impoundments have been constructed for many purposes, often for multiple purposes. Among them are flood control, power generation, water supply, navigation, and recreation. While the same basic physical, chemical, and biological processes occur in reservoirs and natural lakes, reservoirs are obviously much younger geologically, and their morphology (depth, shape), location in the drainage basin, and hydrological characteristics make them unique ecosystems. These characteristics have significant impacts on the development of water quality problems, including the establishment of eutrophic conditions. Thornton et al. (1980), Thornton (1984), and Kennedy, Thornton, and Ford (1985) have developed extensive reviews of the nature and distribution of reservoirs, and of the basic similarities and differences between lakes and reservoirs. The following discussion draws, in part, on these reports.

Reservoir basins are often long, with complex, often dendritic shorelines. Lake basins are more circular. While lakes often receive water from several small streams and ground water, and are usually located nearer the center of their drainage basins, reservoirs are usually supplied by a single large tributary and are located at the bottom of the drainage basin. Water leaves lakes through the ground and/or via an unregulated surface discharge. Reservoir releases are most often discharged through submersed, controlled gates.

Walker (1981) developed a tabular comparison of natural lakes and reservoirs, based upon data from the National Eutrophication Survey of the US Environmental Protection Agency (USEPA). As shown in Table 1, reservoirs on the average have drainage basins more than an order of magnitude greater than lakes, and have much greater surface area, maximum and mean depth, areal water load, ratio of drainage area to reservoir area, and nutrient loading. Their hydraulic residence time, on the average, is less, as well as transparency, biomass of algae (chlorophyll), and the mean concentration of the plant nutrient phosphorus. Reservoirs are located most frequently in the middle latitudes of the United States, an area with different geological,

Table 1

Comparison of Characteristics of Natural Lakes and US Army Corps of Engineers Reservoirs (Modified from Walker 1981)

Variable	Natural Lakes (N = 309)	Reservoirs (N = 107)	Probability That Means Are Equal
Drainage area, km^2	222.0	3228.0	<0.0001
Surface area, km^2	5.6	34.5	<0.0001
Maximum depth, m	10.7	19.8	<0.0001
Mean depth, m	4.5	6.9	<0.0001
Hydraulic residence time, years	0.74	0.37	<0.0001
Drainage area/surface area	33.0	93.0	<0.0001
Phosphorus loading, g P m^{-2} $year^{-1}$	0.87	1.70	<0.0001
Nitrogen loading, g N m^{-2} $year^{-1}$	18.0	28.0	<0.0001
Transparency, m	1.4	1.1	<0.0005
Total phosphorus, mg ℓ^{-1}	0.054	0.039	<0.02
Chlorophyll *a*, μg ℓ^{-1}	14.0	8.9	<0.0001

climatic, and hydrologic properties than the latitudes where natural lakes are concentrated.

According to Kennedy, Thornton, and Ford (1985), reservoirs usually receive water from major tributaries, fed by a network of smaller streams. Materials transported by the river are likely to have been subjected to high degrees of in-stream physical and biological processing, unless the stream has been channelized (Minshall et al. 1985). Also, tributaries to reservoirs often receive discharges from wastewater treatment plants and industries. Reservoirs are thus likely to receive large amounts of fine particulate and dissolved organic matter and inorganic nutrients, which can stimulate algal growth. Organic matter in these forms can also stimulate bacterial metabolism, promote dissolved oxygen depletion, and produce undesirable organic compounds such as chloroform during the chlorination step of a potable water supply system.

Mixing characteristics
and annual thermal history

Hutchinson (1957), Ruttner (1963), and Wetzel (1983) describe annual changes in the thermal structure of lakes. An understanding of these changes is critical to an understanding of the eutrophication process in both reservoirs and lakes. In the spring, water temperature is low and uniform, and wind-driven mixing circulates the entire volume of the basin. Later in spring, surface waters gain heat rapidly and become less dense than the waters in deeper strata. Because of this density difference, winds will not mix these warm, less dense upper waters with the deeper, colder, and denser waters. Throughout the summer this warm upper layer, termed the epilimnion, mixes and exchanges gases with the atmosphere. Just below the epilimnion is a stratum of water with a very sharp thermal gradient. This zone is called the metalimnion. At the reservoir bottom is the hypolimnion, a layer of water that is cold, dense, and stagnant. Figure 1 illustrates the typical summer thermal stratification of a reservoir and the location of these strata.

In the autumn, as heat is lost to the atmosphere, surface waters become cooler and heavier. Eventually the temperature of the upper waters is sufficiently similar to that of the hypolimnion that wind mixing occurs and the reservoir again becomes isothermal and uniformly dense.

It should be noted that reservoirs exhibit varying degrees of thermal stratification. These differences are related to geographic location, operation, and morphometry. Reservoirs located at lower latitudes experience significant heat gain due to warmer climate and duration of the summer period. Temperature profiles for these reservoirs often lack a pronounced temperature discontinuity or thermocline, and temperatures in bottom strata are similar to those in surface strata. Differences in thermal structure are also apparent where surface withdrawal and bottom withdrawal reservoirs are compared (Martin and Arneson 1978). Bottom withdrawal reservoirs discharge cooler hypolimnetic waters and, therefore, store heat. The result is increased temperatures in bottom strata and decreased resistance to wind mixing. Surface withdrawals, on the other hand, lead to pronounced vertical differences in density and greater resistance to mixing. Some reservoirs are so shallow and exposed to wind mixing that they tend to circulate from surface to bottom more or less

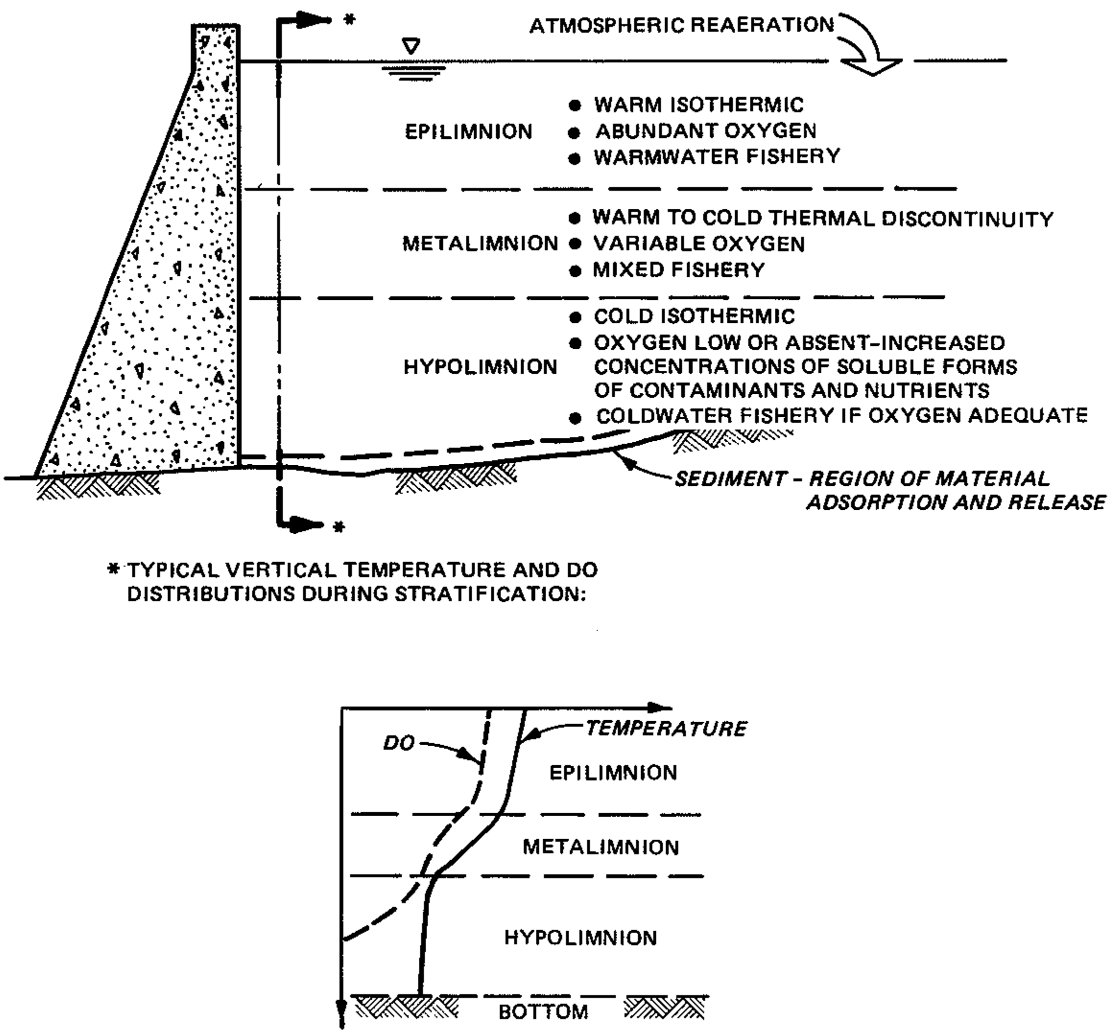

Figure 1. Cross section of a thermally stratified reservoir indicating location and characteristics of the epilimnion, metalimnion, and hypolimnion. Inset depicts typical temperature and dissolved oxygen profiles (from Keeley et al. 1978)

continually. However, even in shallow systems, a hypolimnion may appear near the dam for brief periods during times of calm, hot weather. This is commonly followed by complete circulation once again during windy periods.

Spatial patterns in thermal structure are often observed in reservoirs. In many reservoirs, the upper basin is shallow, and water mixes through the combined forces of tributary flow and wind action. Thermal stratification over as much as half the basin may be absent or occur only during brief periods of low inflow and hot, calm weather. However, the deeper basin toward the dam may exhibit thermal stratification throughout the summer, creating a reservoir with two distinctive habitats, based on their thermal history.

When the density of incoming water is the same as that of the reservoir, the water flows through the upper reaches of the reservoir as a plug flow along the surface, with mixing eventually taking place throughout the water column. When inflows are warmer, and thus lighter, the tributary waters will flow over the reservoir's surface and ultimately mix with surface layers. If this water is nutrient-rich, there may be significant stimulation of algal growth. Colder inflowing water will flow over the reservoir surface until sufficient velocity is lost, at which point (the "plunge point") the inflowing water will plunge beneath the surface, with extensive mixing possible. Inflowing waters with a density intermediate to those for the reservoir's surface and bottom strata intrude in the region of the thermocline as an interflowing density current, while more dense inflows flow along the reservoir bottom as an underflow. Nutrients transported by the interflow or underflow are unavailable to surface water algae at that time. Underflows rich in organic matter will provide a substrate for microbial metabolism and thus lead to a loss of dissolved oxygen in the hypolimnion. Since this layer cannot mix with the atmosphere, oxygen depletion will continue over the summer period.

<u>Longitudinal gradients of water quality</u>

Major gradients of chemical, physical, and biological conditions are pronounced along the length of a reservoir because of reservoir shape and the influence of tributary inflows. At the upper end of reservoirs, the velocity of inflowing waters decreases rapidly and the carrying capacity for suspended solids is reduced. Sedimentation is likely to be heavy, producing shoals or deltas and loss of reservoir volume. Thornton et al. (1980) called this the riverine zone of the reservoir, characterized by mixing, high nutrient concentrations, and possible high sedimentation if water velocity slows sharply. This produces oxygen demand and possible contamination of sediments if the river carries contaminants. In protected areas of clear, shallow water, such as bays, there may be massive development of rooted plants. The next zone down the reservoir, termed the transition zone by Thornton et al. (1980), is the point where colder tributary water plunges. Sedimentation of silt and organic matter is often high. Water clarity may improve sharply, followed by increases in algal growth. Algal blooms may begin here and be transported down the reservoir to the last zone, the lacustrine or lake-like zone at the dam. Figure 2 illustrates the three zones.

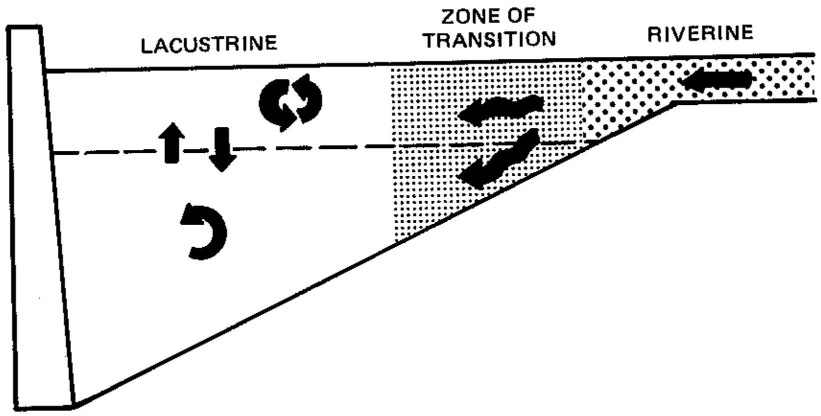

Figure 2. Schematic representation of longitudinal zonation in a typical reservoir. Arrows indicate dominant transport processes. Changes in shading indicate decline in turbidity from the riverine to lacustrine zones (after Thornton et al. 1980)

An important process related to eutrophication occurs in the transition and lacustrine zones. The sedimentation of organic matter transported into the reservoir creates ideal conditions for microbial metabolism and the depletion of dissolved oxygen in the hypolimnion of the transition zone. Under conditions of low or zero dissolved oxygen, reservoir sediments will release phosphorus from iron-hydroxy complexes so that the waters of the hypolimnion become rich in this essential and often growth-limiting element. Also, anoxic waters may have increased concentrations of iron, manganese, hydrogen sulfide, ammonia, and carbon dioxide. Further, the production and death of plants throughout the reservoir, followed by sedimentation of their remains to the hypolimnion, adds significantly to the load of organic matter and subsequent dissolved oxygen losses.

In the lacustrine zone, according to Kennedy, Thornton, and Ford (1985), the magnitude of algal blooms is often limited in extent by nutrient concentrations. The nutrient-rich water of the hypolimnion may be transported or entrained to the epilimnion during the passage of summer cold fronts, which will cool surface waters and provide a mechanism for mixing. Subsequent hot days will bring the nutrient-rich water back to the surface (Stauffer and Lee 1973). Or, seiche activity caused by steady, strong winds that pile up water against one shore, or by reservoir operations, may mix the two temperature zones through the energy of the return underflows of water in a direction opposite to the wind. As described earlier, many reservoirs are too shallow to remain permanently stratified throughout the summer. But, during calm, hot periods the classic three water layers are formed, dissolved oxygen is rapidly depleted in the small volume of the hypolimnion, and nutrients are released from sediments. Windy periods destratify the water column and mix the nutrient-rich water into the entire water column, and may trigger an algal bloom.

Thermal stratification and the associated dissolved oxygen depletion can produce another problem for reservoir operations. If hypolimnetic waters are discharged, their high nutrient concentrations and low or zero dissolved oxygen content may have unacceptable impacts on the biota of receiving waters.

It should be noted that there are reservoirs which receive very low loads of silt, organic matter, and nutrients, which flush rapidly so as to keep concentrations low, or which do not support or contain large amounts of organic matter in the form of rooted plants or rich sediments. These impoundments do not have algal blooms and dissolved oxygen problems.

Biological communities in reservoirs

Reservoirs provide a variety of habitats for aquatic organisms. While the requirements for growth and reproduction for each species of plant or animal vary widely, distinct biological communities or groupings of plants and animals can be broadly defined based on their location in the reservoir. The shallow, nearshore areas, or the littoral zone, exhibit environmental conditions markedly different than the deep, open-water areas, or pelagic zone. The benthic zone or near-bottom area of the reservoir represents a third habitat type. These differences are reflected in the abundance, species composition, and trophic relations of populations and communities present in each habitat type.

The importance of basin shape or morphometry in determining the relative size of each of these zones in two hypothetical reservoirs is depicted in Figure 3. The reservoirs shown are similar in surface area and maximum depth, but different in volume. This difference is due to the fact that the basin of reservoir A slopes more steeply than that of reservoir B. The result is the existence of a more extensive littoral zone in reservoir B.

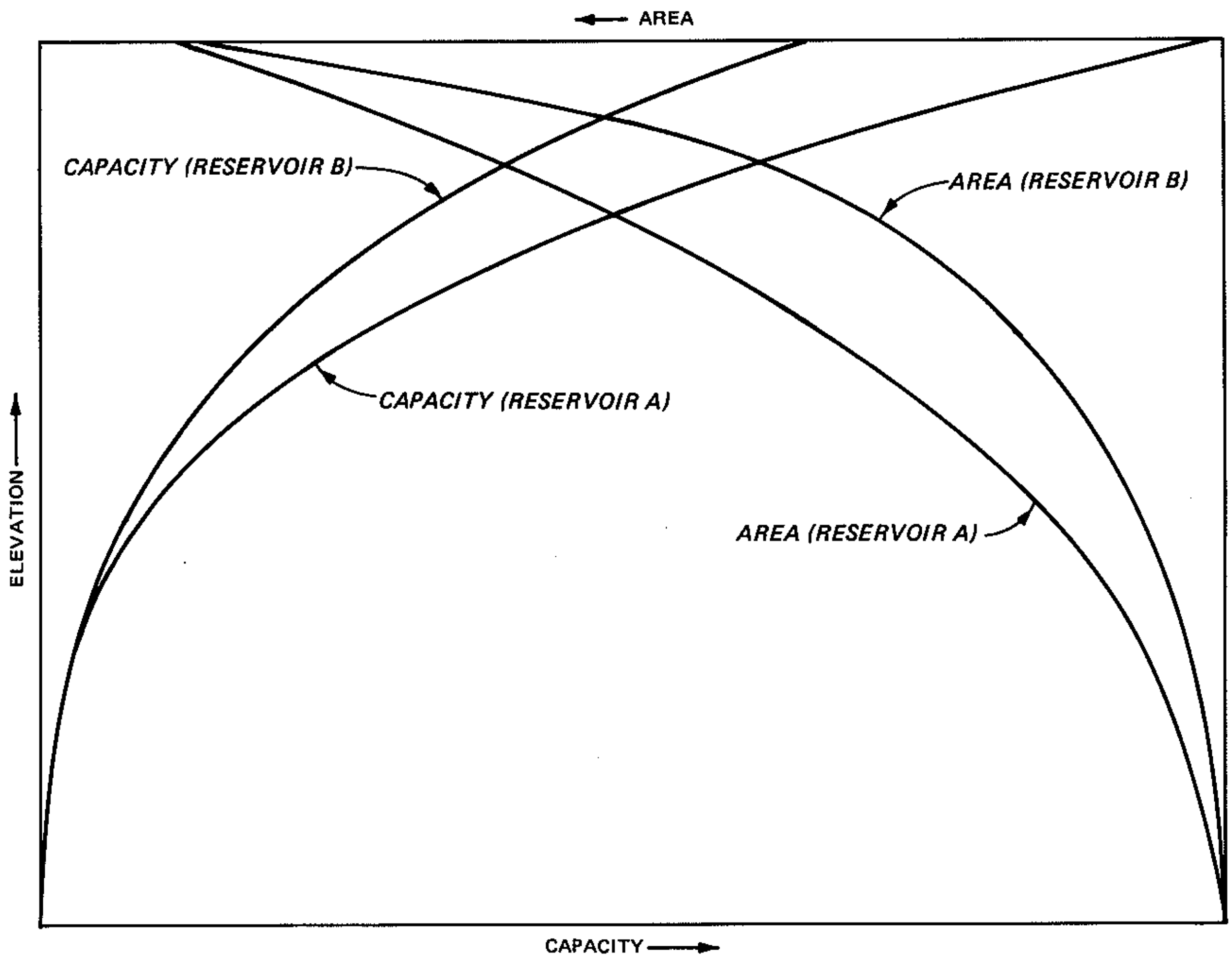

Figure 3. Area-capacity curves for two reservoirs having similar depth and surface area but differing volume and morphometry

This difference can have very significant impacts on reservoir water quality. Reservoir B may have an extensive area of rooted plant growth, assuming that the water is clear and water levels are fairly stable. Plants will grow out to water depths where light or, possibly, hydrostatic pressure becomes limiting. The light limitation depth is often about 2 to 3 m. The littoral zone is rich in animal life, which forms food for fish, and is a site for fish spawning. The plants contribute organic matter to the water column throughout the summer and at their death, which may contribute to dissolved oxygen depletions when this organic matter settles to the bottom. Reservoir A might not have a littoral zone of any significance.

The deep open water of both types of reservoirs is called the pelagic zone. Organisms living here swim (e.g., fish) or float (plankton). Planktonic algae, which may occur as large colonies or filaments, or as single cells, are an important source of energy to the remainder of the food web. Where the water is turbid, the abundance and growth rate of these plants, like the rooted plants of the littoral zone, are limited by the amount of available light. When waters are clearer, as may occur in the lacustrine zone, algal biomass is usually limited by the concentration of a nutrient, most often nitrogen or phosphorus. When waters are nutrient-rich, algal blooms, particularly blue-green algae, may be so extensive that the water becomes "pea-soup" green.

The benthic community is composed of sediment-dwelling animals and microbes. The animals, mainly insects, annelid worms, and molluscs, are a significant part of the fish food web. They also are processors of organic matter in the sediments and change large particles of plant and animal remains into fine particulate organic matter (FPOM). The FPOM is heavily colonized by bacteria and may be quickly solubilized into inorganic substances by microbial metabolism, with a concomitant consumption of dissolved oxygen. A hypolimnion with low or zero dissolved oxygen will have a greatly impoverished benthic fauna. The benthic community, in both deep and shallow waters, is very significant in the storage and regeneration of material for plant growth. An extensive review of the metabolism of this community is found in Wetzel (1983).

The eutrophication process

Eutrophication is the addition of silt, organic matter, and nutrients to a lake or reservoir at rates sufficient to produce an increase in biological production and a decrease in volume or storage capacity. Figure 4 illustrates these incomes and some of the major in-reservoir interactions that promote algal and aquatic plant (macrophyte) growth and loss of volume. This diagram is the basis for an understanding of the management-restoration methods described in this report.

The primary water quality problems of reservoirs are (a) excessive plant biomass (boxes D and F, Figure 4) and the associated impairment of recreation and production of taste and odor in potable water; (b) turbidity, siltation, and loss of storage (box E); (c) low dissolved oxygen, which is associated with problems of sediment release of iron, manganese, hydrogen sulfide, and

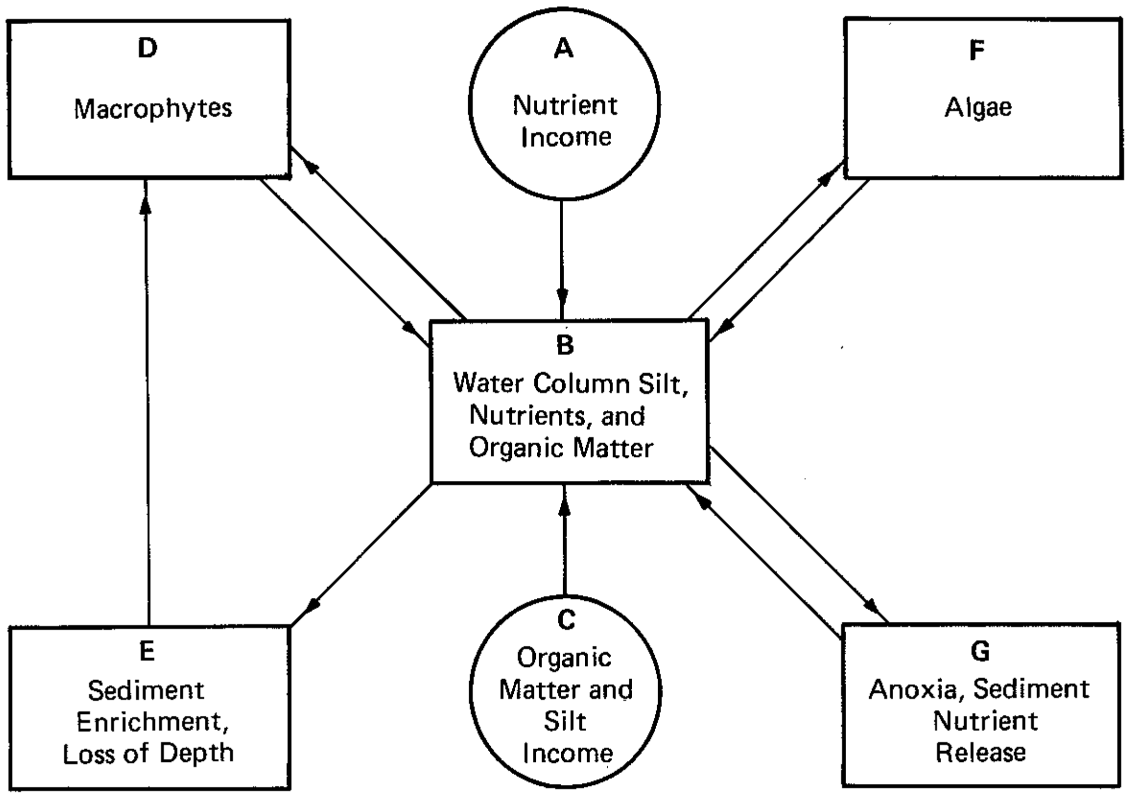

Figure 4. Primary interactions and effects of eutrophication in reservoirs and lakes (after Cooke et al. 1986)

plant nutrients and with poor-quality deep-water discharges (box G); and (d) the appearance of undesirable organic molecules in finished potable water following the chlorination of organic-rich raw water (box B). These problems are ultimately caused by the incomes of silt and organic matter, by the incomes of plant nutrients such as phosphorus and nitrogen (circles A and B, Figure 4), and by important processes in the reservoir, including release of nutrients from sediment storage (box G) and the increase in shallow water suitable for plant growth (box E).

Macrophytes are rooted and free-floating plants that obtain nutrients, such as phosphorus, nitrogen, and potassium, from reservoir sediments and the water column. The abundance and distribution of rooted species are often controlled by types of sediments and by light availability. Aquatic plants are usually insignificant in reservoirs with sandy or extremely organic sediments (Barko and Smart 1986) or in reservoirs that are turbid and/or deep and steep-sided. Significant infilling, such as could occur in any reservoir subject to periodic inflows of silty water, may ultimately provide ideal conditions for plant infestations. These plants, which include waterhyacinth, alligatorweed, milfoil, and coontail, become nuisance weeds when their abundance significantly interferes with reservoir uses. In many cases, their presence is beneficial to fisheries and to waterfowl.

Rooted plants, as shown in Figure 4, contribute to their own success by adding to the buildup of nutrient-rich sediments through the deposition of plant material resistant to decay. Deposition of partially decomposed algal cells may also add to sediment buildup. This positive feedback loop is discussed at length by Carpenter (1980, 1981, 1983a,b) and Wetzel (1983). Once well-established and widespread, rooted plants add to the pool of organic matter (box B, Figure 4) through tissue sloughing and by death at season's end. Some of this organic matter decomposes rapidly, releasing the nutrients taken up by plant roots into the water column, where they may be used by algae. Organic molecules and tissues from macrophytes not only can contribute to sediment accretion, but also are metabolized by microbes and thus stimulate oxygen loss and the chemical changes in anoxic sediments associated with nutrient release to the water column (box G). Finally, organic molecules from macrophytes appear to be precursors of the highly undesirable organics, such as chloroform, which appear in finished potable water following chlorination.

Nuisance algae in reservoirs are found as filamentous mats associated with areas of macrophytes, and as planktonic or free-floating cells and colonies of cells, often seen as "scums." These plants obtain nutrients which have entered the water column from external income, from decomposition of organic molecules (box B), or by direct release to the water from storage in reservoir sediments (box G). When environmental conditions are appropriate for rapid growth, blooms of algae can occur. The abundance of these plants is sometimes directly related to the concentration of the nutrient in least supply, the so-called "limiting nutrient." Often this nutrient is phosphorus. Other factors can also limit algal abundance, including grazing by microscopic animals (zooplankton), light, and loss through water discharge. Large "blooms" or scums of algae contribute heavily to the pool of organic matter in the water column (box B). These organic molecules and large cell fragments behave just as those from macrophytes do, producing sediment accretion, oxygen loss, nutrient release, and the appearance of trihalomethane precursor molecules in raw potable water.

Eutrophication therefore involves not only the excessive addition of materials to the reservoir, but in-reservoir processes that can further enhance the growth of nuisance plants and the appearance of nuisance conditions.

Not all reservoirs can, or need to be, clear, deep, and free of nuisance vegetation. This is so for at least two reasons: (a) the use(s) of the reservoir may be inconsistent with this condition and (b) the richness of area soils, or the amounts of uncontrollable runoff or other regional or local factors, may prohibit any practical reservoir improvement past a certain point. The objectives of reservoir management therefore depend upon reservoir uses, upon the degree of eutrophication acceptable to those uses, and upon reservoir conditions that are attainable for a particular area. For example, the clearest and cleanest water possible for a geographic area might be an objective for a potable water system. In another situation, moderate to high algal productivity might be consistent with good game fishing and, at the same time, not interfere with flood control. In other situations, however, reservoir eutrophication may have proceeded to the point of infestation, or siltation has not only provided ideal conditions for rooted plants but has also significantly reduced water storage capacity. In these and other situations in which the degree of eutrophication interferes with designated reservoir uses, reservoir protection and management may be needed and may be possible.

The first step in reservoir protection should involve attempts at management of point and nonpoint sources to reduce loading of silt, nutrients, and organic matter (circles A and C, Figure 4). Direct treatment of incoming water to remove nutrients, silt, and organic matter through chemical additions, detention basins, and other procedures could be an alternative or additional step in some situations. The reduction in loading, no matter how it is accomplished, must be sufficient to <u>significantly</u> reduce nutrient concentrations in the reservoir, or the rate of siltation. Reservoir improvement following treatment of incoming water or diversion of loading then involves procedures to control the release of nutrients from sediments (box G) and procedures to control or remove plant biomass (boxes D and F) and the production of organic matter (box B). An understanding of Figure 4 is therefore central to understanding the targets of and expected responses to in-reservoir management methods.

The importance of phosphorus

Many of our reservoir management procedures are based upon the control or limitation of phosphorus income and phosphorus concentration as means to limit problems with nuisance blooms of algae. Because of the widespread importance of problems with algae, and because phosphorus management will be a theme in many sections of this report, an introductory discussion of the relationship between phosphorus and algae and the factors determining phosphorus concentration is presented here.

In many cases, the quantity of algae in the deeper, open water of a reservoir is proportional to open-water total phosphorus concentration. This means that it may be possible to lower the concentration, through land treatments such as wastewater diversion and through in-reservoir management, to a level which limits algal growth. The relationship between open-water phosphorus concentration and chlorophyll (a measure of the amount of algae) for some US Army Corps of Engineer (CE) reservoirs is illustrated as a log-log plot in Figure 5. The scatter in the diagram suggests that, in some cases, factors other than phosphorus (such as zooplankton grazing, light, or another nutrient) may have been limiting at the time of sampling. Nevertheless, the relationship is sufficiently common and powerful that predictions can be made about future levels of algal biomass which could occur following events or management procedures that significantly increase or decrease in-reservoir phosphorus concentration. A general observation is that phosphorus concentrations in the open water of less than 20 $\mu g\ P\ \ell^{-1}$ are usually associated with low algal abundance and that concentrations below 10 $\mu g\ P\ \ell^{-1}$ will be found in reservoirs with few, if any, problems with algal blooms. These phosphorus levels and associated chlorophyll concentrations are therefore management targets in those reservoirs in which low algal abundance is desired.

If nutrient income is significantly reduced, this could produce an improvement in reservoir water quality or could protect the reservoir from further deterioration. But unlike many natural lakes, the sources of nutrients, silt, and organic matter to reservoirs are usually from a very large drainage basin. The basin may contain several political units, each with one or more municipal and industrial wastewater discharges and untreated storm drains, plus many nonpoint sources. Significant reduction in the quantity of such materials to a reservoir is usually not possible due to the large area

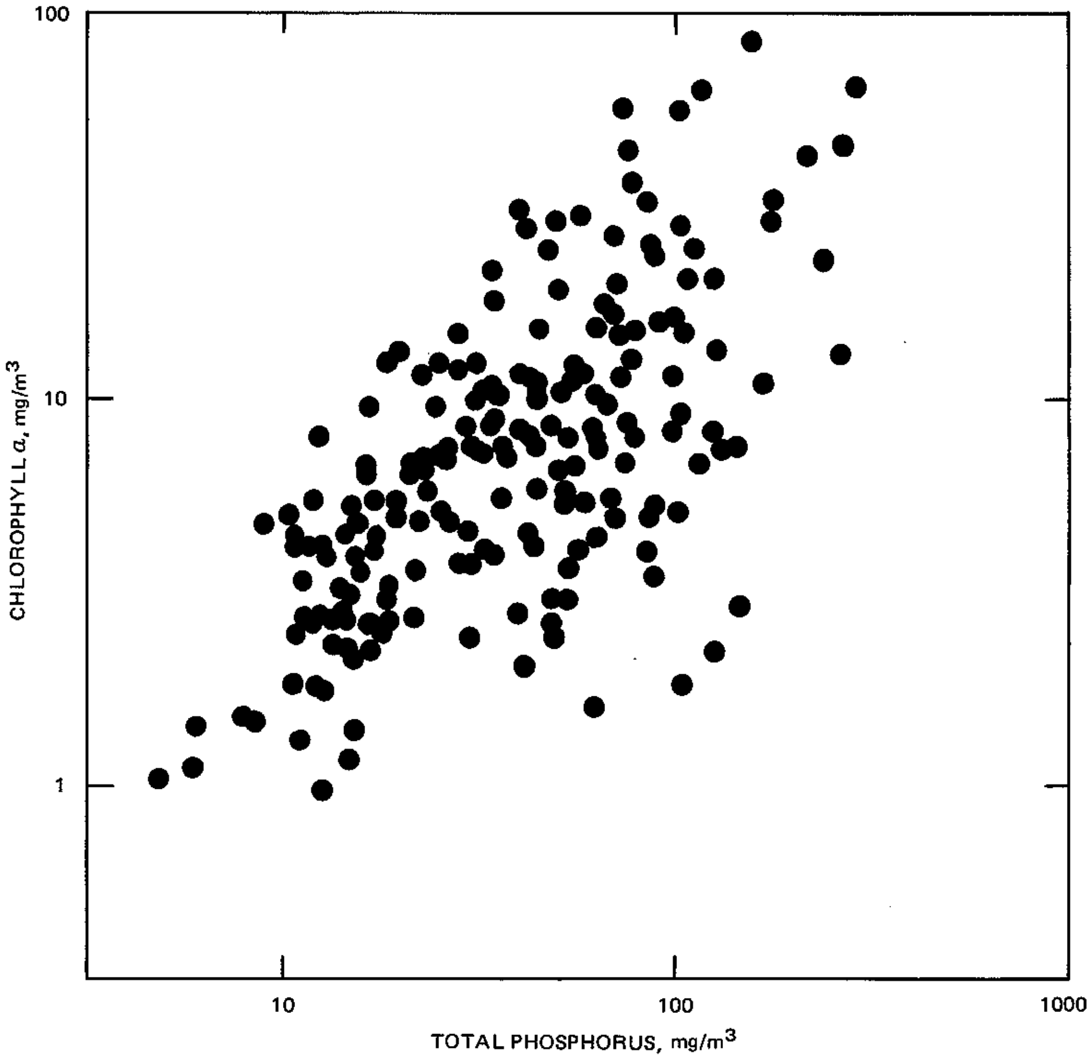

Figure 5. Relationship of average, growing-season chlorophyll $\underline{a}$ concentration and spring total phosphorus concentration for CE reservoirs (after from Walker 1982)

and number of sources and to the difficulties encountered in working with diverse and distinct political units. While the goal of adequate wastewater treatment and proper land use must be continually sought as the best way of achieving long-term improvement in reservoir water quality, steps may have to be taken in the interim within the reservoir. And, as previously pointed out, even with a reduction in the income of silt, nutrients, and organic matter, in-reservoir procedures may be required to effect reservoir improvement. In-reservoir improvement methods are described in subsequent parts of this report.

Phosphorus concentration in the reservoir's water column is determined by its rate of income, deposition, and loss to the outflow (not shown in Figure 4) and by regeneration from sediment storage. Sediment release of phosphorus can be very significant, particularly in shallow reservoirs, and is a process that is part of the feedback loop between organic matter, dissolved oxygen consumption in deep water, and phosphorus release to the water column under both oxic and anoxic conditions. Aquatic systems with a high ratio of sediment surface-to-water column volume can be expected to have significant sediment phosphorus release (Fee 1979).

To assess the amount of phosphorus loading or income to the reservoir that must be eliminated to achieve a significant reduction in algal biomass, the relationship between phosphorus loading and open-water phosphorus concentration must be known. Data from an annual phosphorus and water budget are used in a mass balance model to validate the relationship under current conditions and to estimate expected new phosphorus concentrations following diversion of loading. Walker (1981, 1982, 1985, 1987) has examined this problem for CE reservoirs, and this work is briefly introduced here.

A reservoir phosphorus balance can be expressed in the following terms:

$$\text{Change-in-storage} = \text{Input} - \text{Outflow} - \text{Net sedimentation} \qquad (1)$$

Change-in-storage, which can be either positive or negative, is the difference between the phosphorus content of the reservoir at the end and beginning of an annual cycle. These differences may be due to differences in pool elevation or differences in pool phosphorus concentrations, and are negligible for some reservoirs (Walker 1985). The input term represents the total quantity of phosphorus entering the reservoir during the year from all sources, while the output term is the annual loss of phosphorus due to discharge. The net sedimentation term describes the sum of gains from and losses to the sediments or other sources and sinks within the reservoir.

Since direct measurements of net sedimentation are difficult, methods for estimating this term have been sought. An analysis of CE reservoirs by Walker (1985) indicates that net sedimentation is a second-order decay process and that the rate at which this process occurs is influenced by reservoir flushing rate and the ratio of nonparticulate to particulate phosphorus.

A second-order decay model describing phosphorus sedimentation in reservoirs is as follows:

$$Ps = KP^2 \tag{2}$$

where

Ps = phosphorus sedimentation rate, mg m^{-3} $year^{-1}$
K = effective second-order decay rate, m^3 mg^{-1} $year^{-1}$
P = pool phosphorus concentration, mg m^{-3}

Based on an evaluation of data for CE reservoirs (Walker 1981, 1985), an estimate of the second-order decay rate is 0.1 m^3 mg^{-1} $year^{-1}$. The same evaluation indicated that decay rate tends to be lower for reservoirs with low overflow rates (4 to 10 m $year^{-1}$) and high inorganic-to-total phosphorus ratios for tributary inflows. Surface overflow rate is calculated as annual discharge rate (cubic meters per year) divided by reservoir surface area (square meters). To account for these influences, Walker (1985) suggests the following empirical relationships:

$$K = \frac{0.17Qs}{Qs + 13.3} \quad \text{"low overflow reservoirs"}$$

$$K = \frac{0.056Qs}{Fot(Qs + 13.3)} \quad \text{"high inorganic/total phosphorus ratio"}$$

where

Qs = surface overflow rate, m $year^{-1}$
Fot = tributary inorganic/total phosphorus ratio

If it is assumed that the change-in-storage term is negligible and pool phosphorus concentration is equal to discharge phosphorus concentration (Walker 1985), the phosphorus balance for a reservoir can be quantified by the following expression:

$$QPi = QPo + KVPo^2 \tag{3}$$

where

Q = discharge volume, m^3
Pi = average input phosphorus concentration, mg m^{-3}
Po = average outflow phosphorus concentration, mg m^{-3}
K = second-order phosphorus decay rate, $m^3\ mg^{-1}\ year^{-1}$
V = reservoir volume, m^3

Assuming completely mixed conditions for the reservoir, the solution of the phosphorus balance for pool phosphorus concentration, as approximated by discharge phosphorus concentration, is given by

$$Po = \frac{-1 + (1 + 4KPiT)^{0.5}}{2KT} \tag{4}$$

where T is equal to V/Q (in years).

The final step in using a model of this nature is to link the predicted new phosphorus concentration to a measure of predicted new algal biomass, such as chlorophyll. Reservoirs with a concentration of chlorophyll between 5 and 10 µg Chl a/ℓ can be expected to have a transparency of about 2 to 3 m. Empirical models for such predictions in CE reservoirs are described by Walker (1987) and include features in addition to phosphorus which may control chlorophyll a concentration. A model that incorporates phosphorus and nitrogen concentrations, light availability, flushing rate, and mixed depth is

$$\text{Chl } \underline{a} = \frac{\text{Chl}_{ax}}{(1 + 0.025\text{Chl}_{ax}G)(1 + Ga)} \tag{5}$$

where

$$\text{Chl}_{ax} = (X_{PN})^{1.33}/4.31$$

$$X_{PN} = \left\{P^{-2} + [(N - 150/12)^{-2}]\right\}^{-0.5}$$

P = total phosphorus, mg m^{-3}

N = total nitrogen, mg m^{-3}

$G = Z_{mix}(0.14 + 0.009F_s)$

Z_{mix} = mean depth of mixed layer, m

F_s = summer flushing rate = (inflow - evaporation)/volume, year $^{-1}$

a = nonalgal turbidity (1/m) = (1/Secchi depth) - 0.025 Chl $\underline{a}$

If phosphorus is assumed or demonstrated to be limiting to algal biomass production, a simpler model can be used:

$$\text{Chl } \underline{a} = \frac{\text{Chla}_p}{(1 + 0.025\text{Chla}_p G)(1 + Ga)} \tag{6}$$

where Chla_p is equal to $P^{1.37}/4.88$.

In situations where summer flushing rates are low (F_s < 25 year^{-1}), two other models may be applied. If nonalgal turbidity is low and there is doubt about the nature of the limiting nutrient, chlorophyll $\underline{a}$ can be predicted from

$$\text{Chl } \underline{a} = 0.2X_{PN}^{1.25} \tag{7}$$

Or, if phosphorus is known to be the limiting nutrient, this model can be used:

$$\text{Chl } \underline{a} = 0.28P \tag{8}$$

The reservoir manager uses these equations (see Walker 1987 for available computer software) following the acquisition of appropriate data on loading and reservoir morphology-hydrology to predict the impact of any proposed changes in loading due to restorative efforts or proposed changes in incoming water quality. The next section of this part is concerned with acquiring data that can be used to determine the reservoir's current condition, to recognize changes or deterioration occurring over time, and to evaluate and select in-reservoir management techniques.

Problem Diagnosis

When it is clear that a problem exists, its extent and causes must be evaluated before the feasibility of implementing a management program can be considered. In some cases, the reservoir may be developing problems, but the severity and rate of development are unknown. In both situations, the reservoir manager must implement a diagnostic plan to have a basis for deciding on how quickly a protection and restoration program must begin and what the targets of this program should be. Similarly, proposals for new reservoirs should include a diagnosis of the probability of developing eutrophication problems that will later cause operational difficulties or reduced user benefits.

Reservoir and watershed analyses are the first steps. The purposes of these analyses are to determine the sources and rates of income of substances and the state or condition of the reservoir and to develop predictions regarding future reservoir conditions. Engineer Manual 1110-2-1201 (Reservoir Water Quality Analyses) provides guidance for the assessment of reservoir water quality, including reservoir releases and tailwaters. A reservoir-specific program of data acquisition, compilation, reduction, and modeling has been developed by Walker (1981, 1982, 1985, 1987). Walker's work is briefly described here, but the entire text of at least the 1985 report should be carefully examined prior to the design and implementation of a reservoir diagnostic project. A manual on reservoir sampling design considerations (Gaugush 1987) should also be consulted.

Walker (1987) describes the data requirements for reservoir diagnosis. These include watershed characteristics, nutrient and water loading (income), reservoir morphology, and water quality and hydrology of the pool. Acquisition of these data requires careful examination of extant reports, if any, and a monitoring program designed specifically for the reservoir in question.

The development of both seasonal and annual water and nutrient balances is recommended. Determinations of water income rate and concentrations of total phosphorus, orthophosphorus, total nitrogen, and inorganic nitrogen in the inflow should be made. While not listed by Walker, suspended solids determinations are highly desirable since these materials contribute significantly to the eutrophication process and to the symptoms associated with eutrophication. Minimal and desirable designs for tributary monitoring are outlined. Continuous flow recording combined with periodic grab sampling, especially during high-flow periods, is recommended.

Macrophytes are often a significant reservoir problem, particularly in reservoirs used for recreation and/or (potable) water supply. A survey of the types of plants and the extent of their distribution is an essential step in reservoir evaluation.

A significant amount of reservoir morphology data is required, most of which may be already available. These data include volumes, areas, and lengths of major reservoir segments at typical operating elevations.

Walker (1986) has developed guidelines for monitoring pool water quality. While there may be site-specific variables that must be included, Walker recommends that the following be determined: total phosphorus, orthophosphorus, total nitrogen, inorganic nitrogen, organic nitrogen, chlorophyll _a_, and transparency. Also, the rate of oxygen consumption in the hypolimnion should be calculated. Several stations along the reservoir's length are needed. Methods for sampling design and statistical analysis, prepared specifically for reservoir applications, are found in Gaugush (1986, 1987).

Limnologists describe the degree of eutrophication, or trophic state, of a reservoir through the use of numerical indices that are based on determination of water transparency, total phosphorus concentration, quantity of algae and macrophytes, and amount of dissolved oxygen. The reader is referred to Cooke et al. (1986) for a discussion of the calculation and uses of these valuable indices and to Carlson (1977, 1979) for a description of one of the most widely used indices and a discussion of the theoretical basis for its use. These indices have replaced, for the most part, the use of the term "oligotrophic" to describe a water body with high transparency and low nutrients and "eutrophic" to describe a water body with high biological production.

Walker's (1987) diagnosis requires the use of three computer programs at various stages in the process. FLUX is used to estimate loading to the reservoir from tributary monitoring data, and PROFILE is used to reduce and graphically display water quality data. FLUX and PROFILE are used as data inputs to BATHTUB, a program that is designed to calculate nutrient balance and eutrophication response models such as nutrient-algal biomass relationships described earlier in this report. BATHTUB permits the user to segment the reservoir so that longitudinal gradients in reservoir water quality can be accounted for, as well as relationships between major embayments. BATHTUB can be used in the analytical or reservoir diagnostic portions of a study as well as in a predictive mode. These programs are mathematically and conceptually far less complex than many others, have low data requirements, and are easy to use. For questions related to operational impacts (e.g., changes in withdrawal elevation, modifications of outlet works, etc.) managers should consider more complex, mechanistic models.

Selection of Protection, Restoration, and Management Methods

The diagnostic study is used by the reservoir manager to determine the condition of the reservoir and the cause of its problems or the probability that future eutrophication problems will occur. While it is clear that elimination or reduction in point and nonpoint discharges in the drainage basin should occur, and that every legal and political step should be taken toward this end, it is not realistic to believe that this can usually be accomplished in time to prevent further reservoir deterioration. Steps beyond diagnosis must be taken in the form of active reservoir protection or improvement.

Sanders and Decell (1977) have developed guidelines for the selection of a management plan for reservoirs with macrophyte problems. Their approach could be used in the evaluation of procedures to control other problems. They emphasize that environmental and user constraints must be carefully assessed to determine the impact they will have on the procedure selected. These constraints must be weighed against effectiveness of the procedure. They further emphasize that reservoir responses to the restoration technique that is implemented must be carefully monitored to determine effectiveness and any negative environmental impacts that may arise.

Reservoir restoration or protection techniques are divided into three groups: methods that emphasize prereservoir treatment of incoming water, methods directed at silt removal and in-reservoir control of nutrient concentrations, and methods that emphasize direct plant biomass control or removal.

> Method 1: Prereservoir treatment. Since control of drainage basin sources of silt, organic matter, and nutrients may not be feasible, it is possible to treat incoming waters before they reach the reservoir. Selection of this option will usually require additional land upon which to build a siltation basin, a wetland, or a river treatment plant. Every effort should be made to accomplish prereservoir protection because every in-reservoir treatment can be overwhelmed by continued

high income of substances. Thus, this group of procedures should be given highest priority during a feasibility investigation.

Method 2: Silt removal and control of nutrient concentrations. Loss of storage capacity and interference with recreation indicate a need for sediment removal. Excessive nutrient income, high in-reservoir concentrations, and/or high internal release of nutrients from sediments all indicate a need for nutrient control. All of these methods will be overwhelmed by continued high external silt and nutrient income and thus should be considered as "second steps" following evaluation of the feasibility of reservoir protection. The procedures described in this section emphasize control of the internal release of nutrients from sediments. These methods are phosphorus inactivation, dilution/flushing, hypolimnetic withdrawal, sediment removal, and hypolimnetic aeration. The most understood and most often used procedures among these are phosphorus inactivation and sediment removal, while hypolimnetic aeration is frequently employed to improve problems directly associated with low dissolved oxygen, including potable water intake quality.

Method 3: Control of plant biomass. The excessive growth of algae and rooted plants is ultimately traceable to high nutrient concentrations and to a large area of shallow, nutrient-rich sediments. Some of the procedures described here, especially water-level drawdown, biological controls, and sediment covers, can provide long-term controls beyond symptomatic relief, and several of the procedures may work well in combination. Artificial circulation and algicides are presently the only choices of procedures for control of algal biomass beyond nutrient control. The algicides on the market are essentially confined to the various preparations of copper sulfate, and none of these is more than briefly effective. Therefore, artificial circulation should be given careful consideration. Unfortunately our understanding of how this method could produce a reduction of algal biomass, and the degree of its applicability to the wide variety of situations encountered in reservoirs, is poorly known. This means that in many reservoirs, given our current level of knowledge, long-term control of algal biomass will not occur unless there is nutrient control. Rooted plant biomass, if a nuisance only in small areas (e.g., docks, beaches) can be handled with sediment covers or herbicides. If the reservoir is used for potable water supply, or if a nearby downstream municipality uses the discharged water for drinking purposes, then some herbicides cannot be used. In the case of large-area macrophyte infestations, harvesting, herbicides, or water-level drawdown may be the most feasible and cost effective and should be given highest priority in a feasibility study. Biological controls remain largely in an experimental phase, with some significant exceptions in southern waters. Herbicides can provide effective, though temporary, treatment.

Feasibility of Methods

Each of the descriptions of reservoir water quality protection and enhancement methods in the ensuing parts of this report has been developed with regard to the problem the procedure addresses, the theory behind its use, how to use it and how effective it is, its cost (where known), and its potential negative environmental impacts. To ensure that reservoir managers and operators are aware of all presently known techniques, this report includes both those techniques that have been applied and proven effective and those that are still experimental.

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PART III: PRERESERVOIR TREATMENT

Problem Addressed

Dense populations of algae, particularly blue-green algae, can create nuisance conditions in reservoirs, which can have negative impacts on project uses. These algal "blooms" are caused by a combination of favorable conditions of light, high pH, warm water, and high nutrient concentrations. A second reservoir problem involves the loss of basin volume through the deposition of silt and organic matter from the land. This process creates shallow, well-lighted, nutrient-rich areas for macrophyte growth. It also leads to impaired project use. Both of these problems are related to the transport of material from watershed to reservoir.

Nutrient concentration in the water, as described in an earlier part and in Walker (1987a), is a function of nutrient income, loss to sediments and outflow, dilution by basin volume, and release from sources inside the reservoir. When nutrient income is reduced through advanced waste treatment or land management of urban and agricultural flows, concentration in the reservoir may decline and algal blooms may decrease or be eliminated. The diversion of nutrients from Lake Washington (Edmondson 1970) is an example of this type of response. Similarly, land management to control erosion can curtail silt income and reduce the rate of basin volume loss.

However, nutrient diversion or advanced treatment and land management are often impossible to effectively accomplish in reservoir management. The drainage basin is usually very large, cutting across many political boundaries. This makes action to create lowered nutrient and sediment income very difficult or impossible for the reservoir manager. An alternative or an addition to advanced waste treatment and land management is to pretreat the water from incoming streams through the construction of structures to accelerate nutrient and silt sedimentation, or to add substances to the incoming stream water to precipitate nutrients and particulate matter.

Theory and Design

Siltation basins

A siltation basin is used to detain incoming water long enough to allow significant deposition of nutrients and particulate matter. Water that flows to the main reservoir should have greatly increased quality as a result, while materials deposited in the siltation basin can be periodically dredged. The design of a siltation basin will be site-specific in that management personnel must know the rates of water, nutrient, and silt income and then calculate the size of basin appropriate to detain water long enough to have significant deposition. In some cases, spring and early summer flows may be the only target, since this may be the water that is most stimulatory to algae or that causes, in other ways, the greatest problems in water quality. A basin to intercept summer low flows would be much smaller than a basin to handle runoff from wet seasons. The reports by Jones and Bachmann (1978); Canfield, Jones, and Bachmann (1982), and Walker (1987b) are useful in designing a basin that will allow significant sedimentation of phosphorus.

Prereservoir phosphorus removal

Since upstream treatment of runoff or effluents may be impossible or inadequate, a significant fraction of the incoming phosphorus could be precipitated in the stream or at the head of the reservoir through the addition of iron, calcium, or aluminum salts. This reservoir protection procedure has had very few published applications, and further research is needed.

Iron is added in the ferric (Fe^{+3}) form, usually as ferric chloride. Iron in this state will precipitate as $FePO_4$, or as $Fe(OH)_3$ with inorganic phosphorus sorbed to it (Stumm and Lee 1960, Wetzel 1983) and will be carried in the streamflow and deposited in the stream and upper end of the reservoir. Phosphorus will remain bound in iron complexes as long as the redox potential in the sediments remains high. Unfortunately, the redox potential can be very low in the anoxic hypolimnia of eutrophic reservoirs. A low redox potential will reduce iron, and phosphorus will be released as iron complexes become soluble (Mortimer 1941, 1942). This internal phosphorus release may be high enough to stimulate algal growth. If dissolved oxygen is present at the sediment-water interface, or if dissolved oxygen is introduced through natural or artificial circulation, then iron should remain in an oxidized state and phosphorus will remain sorbed to it. Thus, for an iron addition to be effective, oxidizing conditions must be present continually at the sediment-water interface of the reservoir or stream site where the precipitate is deposited.

In lakes and reservoirs containing substantial amounts of alkaline earths from the solubilization of calcareous deposits in the drainage basin, a buffering action occurs based upon the equilibrium between free CO_2, calcium, bicarbonate, carbonate, and undissociated calcium carbonate. Free CO_2 remains in solution after this equilibrium is reached, the amount dictating the amount of $Ca(HCO_3)_2$ also in solution. If more CO_2 is added to this system the further solution of $CaCO_3$ will occur, producing more $Ca(HCO_3)_2$ and little pH change. If CO_2 is withdrawn, as occurs during extensive photosynthesis by algae and macrophytes, then $CaCO_3$ is precipitated. As Wetzel (1983) has pointed out, this decalcification of hard water can play a major role in regulating the reservoir's metabolism since the precipitation of $CaCO_3$ will also involve the coprecipitation of nutrients such as phosphorus and the sorption of labile organic matter.

This reaction of $CaCO_3$, bicarbonate, and CO_2 could be used to remove phosphorus and organic matter from incoming water, although there appears to be no case history of its use for this purpose. The effectiveness of this procedure could be altered by the quantities and types of organic matter in the stream. According to Wetzel (1983), dissolved organic matter sorbs to $CaCO_3$. In particular, fulvic acids, or low molecular weight humic acids, seem to repress $CaCO_3$ precipitation and allow phosphorus to remain in solution. This could prevent $CaCO_3$ from being effective in some streams. Effectiveness also will be altered by the lower pH-high CO_2 conditions in some reservoirs. These conditions would lead to solubilization of $CaCO_3$ and release of sorbed materials.

The use of aluminum salts to precipitate phosphorus or to prevent its release from reservoir sediments is a well-known in-reservoir water quality management procedure (see Part IV). Aluminum salts could also be added to incoming stream water to precipitate phosphorus before it enters the reservoir. When aluminum sulfate (alum) or sodium aluminate are mixed with water with carbonate alkalinity, a visible floc of aluminum hydroxide is formed to which inorganic phosphorus is strongly sorbed. The formation of the floc may also trap some particulate phosphorus. The $Al(OH)_3$ floc or polymer appears to be inert to redox changes so that sorbed phosphorus will remain out of circulation (Browman, Harris, and Armstrong 1977). A hazard of this procedure is the depression of stream or reservoir pH when using alum, depending upon buffering capacity, and the subsequent appearance of a potentially toxic dissolved aluminum form (Al^{+3} or $Al(OH_2)^-$). If sodium aluminate is used, pH will increase, and at high pH (>8.5), dissolved aluminum again appears (Sung and Rezania 1984). Another hazard involves the smothering of stream invertebrates with the deposited floc. Further details of aluminum chemistry are found in Cooke et al. (1986) and Burrows (1977).

Wetlands

Wetlands, and man-made artificial wetlands or settling basins dominated by rooted plants and their epiphytes, can intercept significant amounts of nutrients and suspended solids under certain conditions. While rooted plants may absorb a comparatively small amount of nutrients, their presence creates barriers to water flow and enhances water detention time and thus contact with the major storage compartments of a wetland, the microflora, detritus, and sediments (Howard-Williams 1985). Deposition of suspended materials will also be enhanced when water flow-through is impeded by the presence of vegetation.

In many cases, the nutrient retention capacity of a wetland is limited on a short-term basis to the growing season, and on a long-term basis to the saturation of storage compartments. High initial nutrient removal rates may be followed, in several years, by large nutrient exports. Harvesting of macrophyte biomass may prolong the wetland as a nutrient sink. Wetlands with predominantly mineral soils having a high aluminum content are far better phosphorus sinks than wetlands with peat soils. Terrestrial ecosystems, however, retain far more nutrients than wetlands (Howard-Williams 1985, Richardson 1985).

Lee, Bentley, and Amundsen (1975) list the following beneficial and adverse effects of wetlands on the quality of water discharged from them:

Beneficial effects:

- Denitrification of nitrate under anaerobic conditions permits methane formation and the degradation of certain organic compounds.
- $CaCO_3$ precipitates, along with other chemicals such as phosphorus.
- Sediments are trapped.
- Nutrients are removed during summer months, especially if flow is diffuse.

- Water storage by the marsh helps reduce fluctuation in water flow.

Adverse effects:

- Nutrients are released during periods of high spring or fall flows, necessitating nutrient interception.
- Nitrogen fixation occurs at a high rate, producing an increase in concentration of organic nitrogen in the marsh discharge.
- Water leaving a marsh may produce taste and odor problems.
- Marsh discharges may be high in organic matter and color, and low in dissolved oxygen.

Diversionary streams

Storm events often exhibit the poorest water quality and can represent a very significant fraction of the total annual loading of nutrients, silt, and organic matter to the reservoir. In some situations, it may be possible to divert some or all of these high flows around the reservoir, especially in reservoirs used solely for recreation or as water supplies, and not for flood control or power generation. The use of this procedure has not been documented with regard to determinations of when or how to divert the water, the effectiveness of the procedure, nor the impact on downstream biotic communities. These data will vary from case to case, and a detailed budget of silt and nutrient loading will form the basis of any design to divert high flows.

Effectiveness, Costs, and Feasibility

Siltation basins

The work of Fiala and Vasata (1982) provides an example of the effectiveness of a siltation basin in removing phosphorus from incoming waters. Jesenice Reservoir, Czechoslovakia, was divided into a small (area = 76 ha; volume = 1.4×10^6 m^3) siltation basin with a detention time of 5 days. It emptied into the main reservoir (area = 670 ha; volume = 51×10^6 m^3), which had a theoretical hydraulic detention time of 180 days. Orthophosphorus fell from over 500 μg P ℓ^{-1} at the inlet of the siltation basin to 30 μg P ℓ^{-1} at its outlet. Orthophosphorus then reached about 10 μg P ℓ^{-1} at the pool behind the main dam. Phytoplankton biomass also declined. The authors note that phosphorus retention by the siltation basin increased with detention time, and they suggest a minimum of 5 days. This could be difficult to achieve on an annual basis in many situations, but could be feasible during summer low flows when symptoms of eutrophication are most extreme. Fiala and Vasata (1982) also note that maintenance of aerobic conditions in the siltation basin is essential to phosphorus removal, presumably because iron and/or calcium complexation of phosphorus may be involved and because anaerobic reservoir sediments may release phosphorus at high rates and thus reduce the efficiency of removal. In deep preimpoundment basins, maintenance of aerobic conditions might require artificial circulation.

Dry dams

In some areas, dry dams have been constructed to aid in flood control. These dams may be particularly effective in silt and nutrient control as well, since they receive and store the "first-flush" runoff, a portion of the hydrograph that can be heavily loaded with pollutants.

Prereservoir phosphorus removal

Wahnbach Reservoir (Federal Republic of Germany) was impounded in 1957. Within 10 years, treatment of the water for drinking purposes became very expensive, and organic compounds excreted into the water by algae were forming precursors for the development of trihalomethanes. Phosphorus was shown to be the limiting element, and nutrient budget studies showed that more than 50 percent of it came from diffuse or nonpoint sources on the watershed, making sufficient diversion nearly impossible. A smaller reservoir (500,000 m^3) to serve as a floodwater retention basin and a phosphorus elimination plant (PEP) were built at the upper end of the main reservoir.

After detention in the smaller reservoir, water is pumped into the PEP and treated with Fe^{+3} to precipitate phosphorus, followed by a cationic polyelectrolyte to form large floc. The water is then filtered through layers of activated carbon, hydroanthracite, and quartz sand. The plant can handle up to 5 m^3 sec^{-1}, and about 95 to 99 percent of phosphorus-containing compounds are eliminated. Output concentration to the main reservoir averaged 4 μg P ℓ^{-1} over 2 years. Also, the PEP has high removal (99 percent) of coliform bacteria, chlorophyll, and turbidity, and lesser removal of chemical oxygen demand (77 percent) and dissolved organic carbon (58 percent). Water discharged to the reservoir approaches drinking water quality, and the trophic state of the reservoir is now nearly oligotrophic. Detailed descriptions of the PEP at Wahnbach Reservoir are provided by Bernhardt (1980, 1981). Costs have not been reported.

Prereservoir phosphorus precipitation has had few reported applications. However, results have been very encouraging, and it will probably be used with increasing frequency. Lathrop (1982) has reviewed this method.

Bannink, van der Meulen, and Peeters (1980) and Hayes et al. (1984) report on the use of iron salts to treat river water entering reservoirs in The Netherlands and England, respectively. Water quality improvements were noted, as well as reduced treatment costs of drinking water. However, Hayes et al. (1984) noted that internal phosphorus release during summer months was responsible for algal blooms, suggesting that iron-bound phosphorus may have been released under anaerobic conditions.

Harper, Wanielista, and Yousef (1983) have suggested the use of the $Al(OH)_3$ sludge, produced during potable water treatment, as a cost-effective compound for treating incoming stream waters. No results appear to be available on their treatment of inflowing storm water to Lake Eola, Florida. Caution should be exercised in the use of the material from a potable water treatment plant since the sludge may have very large amounts of organic matter and phosphorus sorbed to it. Thus, its addition to the upper end of a reservoir could produce a pronounced oxygen demand and little phosphorus removal.

Cooke and Carlson (1986) have found that only a small dose of aluminum sulfate (1 to 5 mg Al ℓ^{-1}) was needed to precipitate all of the soluble reactive phosphorus in the Cuyahoga River just above Rockwell Reservoir, Ohio. The dose to accomplish phosphorus removal was determined with a jar test. To be certain that only insoluble aluminum hydroxide was formed, an attempt was made to keep the dose above the level that would produce a pH of 6.0 or less. Since the experiment was conducted on only a pilot scale in August and September 1985, long after substantial macrophyte and blue-green algae problems had developed in the reservoir, there was no expectation of reservoir improvement, and none occurred. The large volume of aluminum hydroxide floc that deposited in a small area due to the late summer low-flow conditions was deleterious to benthic macroinvertebrates in this area. No changes in macroinvertebrates, compared with upstream controls, were observed at stations nearer the reservoir. This apparently new and simple (compared with Wahnbach) approach to protecting a reservoir is undergoing further evaluation by Cooke and Carlson.

Wetlands

Case histories of the use of wetlands, marshes, or small impoundments with dense vegetation demonstrate that these systems can remove 50 percent or more of the incoming nutrients and suspended solids during the growing season (e.g., Toth 1972; Lee, Bentley, and Amundsen 1975; Spangler, Fetter, and Sloey 1977; Fetter, Sloey, and Spangler 1978; MacCrummon 1980; Sinclair and Forbes 1980; Barten 1983; Herron, LaMarra, and Adams 1984; Weidenbacher and Willenbring 1984; Willenbring 1985).

Barten (1983) describes the deterioration of Clear Lake, Minnesota (area = 257 ha; drainage area = 1,518 ha) due to urban runoff. To protect the lake from further impacts, storm runoff was diverted to a 21-ha marsh, composed of peat underlain with clay loam and having a reed canary grass (*Phalaris arundinacea*) plant community. The marsh was divided into cells controlled by gates. Nutrients and suspended solids were removed by percolation through the peat. The marsh was harvested to remove nutrients and to maintain the absorption potential of the peat. During the winter, storm flows were diverted through the marsh rather than through the cells. Filtration significantly reduced nutrient concentrations, especially phosphorus (90 percent), and suspended solids (70 percent). In 1982, 897×10^6 ℓ was filtered, removing 526 kg of phosphorus. Where possible, a 5-day detention time was used.

Sinclair and Forbes (1980) examined the removal capacity of a swamp, a 16-ha reservoir dominated by waterhyacinth (*Eichhornia crassipes*), and a 0.4-ha reservoir dominated by the submergent plant *Najas* sp. The latter two systems were effective nutrient sinks, but the aerobic system (*Najas*) was most effective. The authors believe that in comparison with the swamp, the waterhyacinth- and naiad-dominated systems have the greatest potential to be nutrient sinks because they can be harvested. The systems could be used in series. Sinclair and Forbes (1980), following the suggestion of Boyd (1970), also recommended cattail systems for removal of nutrients and suspended solids due to their large standing crop, rapid growth rate, high nutrient value to cattle, and ease of harvest.

Limitations and Concerns

Many of the problems that could be encountered with the use of any of these prereservoir treatments will be site- and problem-specific. Therefore only a general listing of the most likely problems is given here.

Siltation basins and wetlands

A critical problem with the use of either of these systems is obtaining the area needed to construct them. If a portion of the upper reservoir is modified to form a smaller reservoir, then significant loss of storage capacity may occur. Both the marsh and the siltation basin may require substantial maintenance in the forms of harvesting of plants and the removal of accumulated silt through dredging.

Wetlands will discharge nutrients during high-flow, low-vegetation periods. More significantly for potable water reservoirs, they also can discharge dissolved organic molecules, which may impart taste and odor, increase the chloride demand, and perhaps contribute to trihalomethane production. Lee, Bentley, and Amundsen (1975) suggest that marsh outflows could be treated with a low dose of aluminum sulfate. Willenbring (1985) notes that channelized flow in a wetland will reduce its removal capacity.

Prereservoir phosphorus removal

An area of significant concern is the potential for adverse effects on stream biota from aluminum salts. The interested reader is referred to Part IV as well as to Burrows (1977), Kennedy and Cooke (1982), and Cooke et al. (1986) for a discussion of the chemistry of aluminum and the environmental conditions under which it can be deleterious to biota. Briefly, when aluminum sulfate is added to natural waters containing bicarbonate-carbonate alkalinity, a visible precipitate of aluminum hydroxide is formed, and pH falls. The floc is sorptive of phosphorus and organic matter, and some materials are trapped with the floc. The forms of aluminum that appear in the water are pH-dependent. Insoluble $Al(OH)_3$ predominates between pH 6 to 8, while soluble species predominate at higher ($(Al(OH)_4^-)$ and lower pH ($Al(OH)_2^+$ and Al^{+3}). $Al(OH)_2^+$ and Al^{+3} are considered to be potentially toxic, and therefore pH must not fall below pH 6.0.

Very few studies of the toxicity of aluminum to aquatic biota have been conducted. Collectively, these studies suggest that concentrations of Al^{+3} below 0.050 mg Al ℓ^{-1} are not toxic to *Daphnia, Tanytarsus dissimilis* (Insecta, Chirononidae) and *Salmo gairdneri* (rainbow trout). Biesinger and Christensen (1972) found that the 48-hr LC_{50} for *Daphnia magna* was 3.90 mg Al ℓ^{-1}, and a 10-percent reproductive impairment occurred at 0.32 mg Al ℓ^{-1}, when animals were reared in Lake Superior water (alkalinity 50 mg $CaCO_3$ ℓ^{-1}, pH 7.74). Lamb and Bailey (1981) report that instars of *T. dissimilis*, reared in laboratory systems at pH 7.8, were unaffected in acute tests at doses from 6.5 to 77.8 mg Al ℓ^{-1}. Dissolved aluminum (Al^{+3}) remained below 0.1 μg Al ℓ^{-1}, and the floc was used by the larvae for tube building. In chronic tests at pH 6.8, Al^{+3} remained at the same low level over the same dose range. However, mortality occurred at every dose, and pupation did not occur over the 55-day study. Narf (1978) found that there were no apparent effects to benthic insects in several lake treatments. Everhart and Freeman (1973) found that a concentration of 0.52 mg Al ℓ^{-1} produced behavioral problems in rainbow trout after several weeks of exposure, whereas a concentration of 0.052 mg Al ℓ^{-1} produced no long- or short-term effects. Buergel and Soltero (1983) found no mortality, physiological stress, gill hyperplasia or necrosis, or retardation of rainbow trout growth after a dose of 12.2 mg Al ℓ^{-1} to hard-water Medical Lake, Washington.

The deposition of floc in the stream may pose some hazard to aquatic organisms and may have an adverse appearance. High flows should displace the floc deposit to the reservoir, and offer the benefit of treating phosphorus-rich reservoir sediments with a substance that may stop internal phosphorus release at the deposition site. The negative features of the use of treatment plant-generated aluminum hydroxide sludges have already been discussed.

Summary

Prereservoir treatments are a partial substitute for watershed management and advanced waste treatment. The object is to detain or remove loads of nutrients, organic matter, and silt by settling basins, marsh filtration, or the addition of nutrient-precipitating chemicals to the stream. With the possible exception of dry dams, these methods have not been widely employed, as yet, but a review of case histories demonstrates their potential effectiveness. These methods can be costly, although reliable cost estimates are uncommon. Also, there are problems with land acquisition to build such basins, with the discharge of nutrients and organic matter from marshes during high flows, the requirement for periodic silt removal, and the potential for creating toxic conditions through the addition of aluminum salts. Table 2 is a summary of this method.

Table 2

Summary of Prereservoir Treatments

Characteristic	Description
Targets	Nutrients, organic matter, and silt income.
Modes of action	Forces deposition in siltation basin. Strips silt and nutrients from water by marsh plants. Precipitates phosphorus in incoming stream.
Effectiveness	Highly effective, depending upon method chosen, season, and maintenance frequency.
Longevity	Months to years.
Negative features	Loss of storage capacity of main reservoir if pre-impoundment basin is constructed. Maintenance requirements can be extensive. Wetlands may lose effectiveness and will discharge unwanted organics. Dissolved aluminum and/or aluminum hydroxide floc may be toxic to reservoir biota.
Costs	Unknown because of site specificity.
Applicability to reservoirs	Not as applicable to high-volume, hydropower reservoirs as to smaller recreational and potable water supply reservoirs.

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PART IV: PHOSPHORUS INACTIVATION

Problem Addressed

Many reservoirs experience intense and prolonged "blooms" of algae during summer months, particularly in the transition and lacustrine zones of the reservoir (see Part II) where the water is clearer. The occurrence and degree of these blooms can often be directly linked to a high income and a high in-reservoir concentration of nutrients, especially phosphorus (Rast and Lee 1978). A substantial reduction in nutrient loading to the reservoir, as would occur in the case of sewage diversion, will usually lead to a predictable decline in concentration in the reservoir. If the decline has been significant, then algal blooms may decrease in frequency and extent, and the degree of eutrophication or trophic state of the reservoir may shift to a far less productive condition. The dramatic improvement of Lake Washington following sewage diversion is one illustration of this type of response (Edmondson 1970).

Phosphorus release to the water column from enriched sediments, especially under conditions of low dissolved oxygen or high pH and temperature, is well known (Bostrom, Jansson, and Forsberg 1982). This "internal loading," as described in Part II, can be a very significant source of nutrients, especially phosphorus, to the water column. Internal nutrient loading may prolong the eutrophic state long after nutrient diversion. Shallow reservoirs, especially those with a prolonged history of nutrient and organic matter loading, are especially susceptible to the impact of internal nutrient release. In these reservoirs the regenerative zone or recycling zone (the sediments) is very close to the lighted, productive zone (the surface waters), and algal blooms may therefore persist after a reduction in nutrient income.

Phosphorus inactivation is a procedure to accelerate the recovery of a reservoir, following a reduction in nutrient income, in those cases where internal phosphorus release is extensive. The target of the treatment is phosphorus in reservoir sediments, and the procedure is to add an aluminum salt to the sediments to bind the phosphorus to aluminum hydroxide. The layer of aluminum hydroxide will persist, even under conditions of anoxia, and has produced a significant decrease in phosphorus concentration in the water column of natural lakes and maintained an improved trophic state in them for many years after treatment. The applicability of this procedure to reservoirs, where adequate treatment of incoming nutrients may not occur, remains open to investigation. This Part describes how the procedure works, how to apply it, and its cost and effectiveness in lakes. More reservoir treatments are needed to better define its effectiveness in this habitat.

Theory and Design

Phosphorus inactivation is carried out through the addition of aluminum sulfate or sodium aluminate (or both) to the lake or reservoir. Aluminum has been the element of choice rather than iron because the complexes and polymers that form after the addition of either of these aluminum compounds are apparently inert to changes in oxidation-reduction potential, such as would occur during the development of hypolimnetic anoxia. Phosphorus will remain bound to these complexes, whereas iron will release phosphorus as the redox potential falls.

Hayden and Rubin (1974), Burrows (1977), and Kennedy and Cooke (1982) have provided reviews of the chemistry of aluminum salts in water. A knowledge of this is essential in determining the correct dose and preventing the development of a high concentration of dissolved aluminum (Al^{+3}), an aluminum species that has been associated with toxicity to aquatic organisms. When aluminum sulfate ($Al_2(SO4)3 \cdot 18H_2O$) or sodium aluminate ($Na_2Al_2O_4 \cdot 14H_2O$) is added to water, the pH dictates the form of hydrolyzed aluminum that will predominate. Settleable, polymerized aluminum hydroxide ($Al(OH)_3$) predominates at pH 6 to 8, aluminate above this range, and dissolved aluminum (Al^{+3}) below it. $Al(OH)_3$, a visible precipitate, is very sorptive of phosphorus, particularly inorganic phosphorus, and is thus the desired form. When aluminum sulfate (alum) is added to water with carbonate alkalinity, the pH and alkalinity of the water will fall at a rate dictated by the water's initial alkalinity. Low initial alkalinity or an excessive dose would allow pH to fall below pH 6.0 and thus decrease the amount of phosphorus-sorbing $Al(OH)_3$ and increase the amount of potentially toxic dissolved aluminum (Al^{+3}).

Phosphorus inactivation is a technique in which as much aluminum sulfate or sodium aluminate as possible, within the bounds dictated by initial alkalinity, pH, and the associated formation of dissolved aluminum (Al^{+3}), is added to the sediments with the purpose of controlling phosphorus release

(Kennedy 1978, Kennedy and Cooke 1982, Cooke et al. 1986). The objective is to control phosphorus release for a period of at least several years. Another procedure, known as phosphorus removal, has been used to add small amounts of an aluminum salt to water for the purpose of removing the phosphorus in the water column rather than giving sediments a maximum dose to control phosphorus release. Phosphorus removal has been used very effectively in special situations such as the interception of nutrients released from decaying vegetation in the fall (Funk et al. 1982), and it has been used as a treatment for incoming streams water (Part III).

Fly ash, the airborne particulate matter (5 to 100 μ) that is trapped in electrostatic precipitators in coal-fired power plants, has been suggested as another type of phosphorus inactivant for lakes and reservoirs (Tenney and Echelberger 1970). Fly ashes have very large sorptive areas and are high in CaO, MgO, Na, and Al. Therefore, they would sorb phosphorus. Fly ash appears to be an attractive option for improving reservoirs because the material is produced in very large quantities and only about 20 percent of it has been used for purposes such as the manufacturing of cement (Adriano et al. 1980). Unfortunately, fly ash treatments have produced serious negative environmental impacts, due primarily to the presence of heavy metal contaminants.

Aluminum dosage to a reservoir for the purpose of removing phosphorus from the water column is determined by jar tests. Aluminum salts, usually aluminum sulfate, are added in increasing amounts to a series of continuously stirred beakers containing reservoir water and reservoir water spiked with known amounts of phosphorus. After settling, phosphorus concentration is measured, and the amount of alum required to obtain the desired phosphorus removal is used to calculate the tonnage of alum needed to treat the water column (Peterson et al. 1973, 1974; Cooke and Kennedy 1981). The amount of alum added is usually so small that large pH shifts and the appearance of dissolved aluminum do not occur. However, pH, alkalinity, and dissolved aluminum must be measured to be certain that potentially deleterious conditions do not occur.

Kennedy (1978) was the first to suggest that the most desirable lake treatment would be to add as much aluminum as possible, consistent with environmental safety, to the phosphorus-rich sediments rather than to the water column, with the purpose of inactivating this sediment store. He developed a procedure for obtaining the maximum dose for a lake by considering the relationships between alkalinity, pH, and aluminum dose. Kennedy's method is reviewed in Cooke and Kennedy (1981), Kennedy and Cooke (1982), and Cooke et al. (1986). The procedure, which is applicable for reservoirs also, is briefly outlined here.

When aluminum sulfate is added to reservoir water, pH and alkalinity fall. At pH 6 to 8, large amounts of the floc aluminum hydroxide are formed, and the dissolved aluminum (Al^{+3}) concentration remains low. This pH range is therefore ideal since large amounts of phosphorus will be sorbed, and toxic conditions will not be present. However, with further additions of alum, pH values below 6.0 occur and the concentration of dissolved aluminum increases rapidly (Figure 6). Therefore, the maximum amount of aluminum sulfate that can be added before the appearance of low pH and high dissolved aluminum (Al^{+3}) is dependent upon the initial alkalinity of the reservoir water. The maximum dose is therefore unique to each reservoir. General guidelines

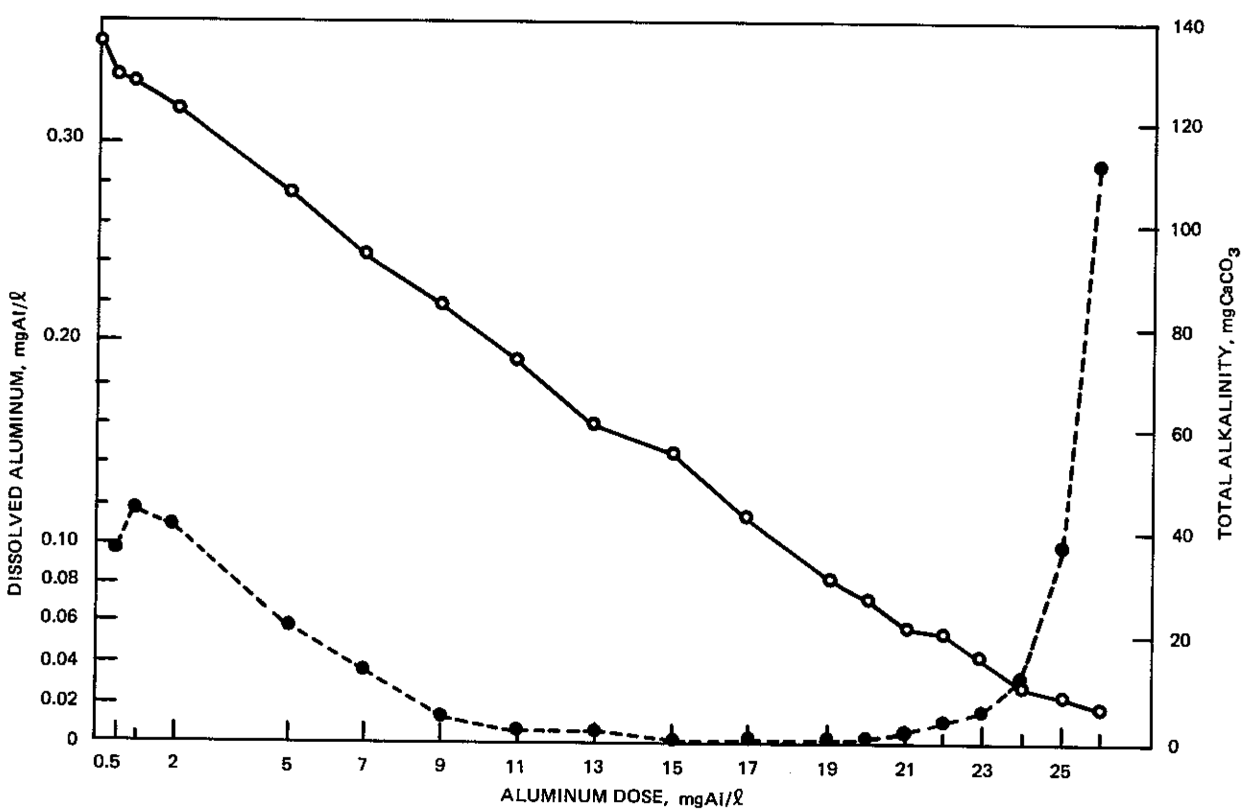

Figure 6. Changes in dissolved aluminum concentration (dashed line) and total alkalinity (solid line) for water from West Twin Lake, Ohio, treated with increasing doses of aluminum sulfate (after Cooke et al. 1978)

for dose determination can also be developed from a knowledge of pH and alkalinity of each stratum and use of the nomograph from Kennedy and Cooke (1982) (see Figure 7).

For dose determination, a vertical series of water samples is obtained and alkalinity is determined. Then, other water samples from the same depths are titrated with stock solutions of alum to pH 6.0. The relation between the aluminum dose and the alkalinity and pH is then used to obtain the maximum dose for any reservoir alkalinity over the range of alkalinity and pH tested. The maximum dose for each depth interval is calculated from the titration and water volume for that depth interval, and these are summed to produce the total dose for the reservoir, or section of the reservoir. Accuracy in treating the reservoir is obtained by dividing the treatment areas, or the reservoir, into zones marked by buoys. The volume and alkalinities in each of the zones are measured, and the amount of alum is then determined. By dividing the reservoir into sections, an overdose to shallow areas or an underdose to deep areas is avoided.

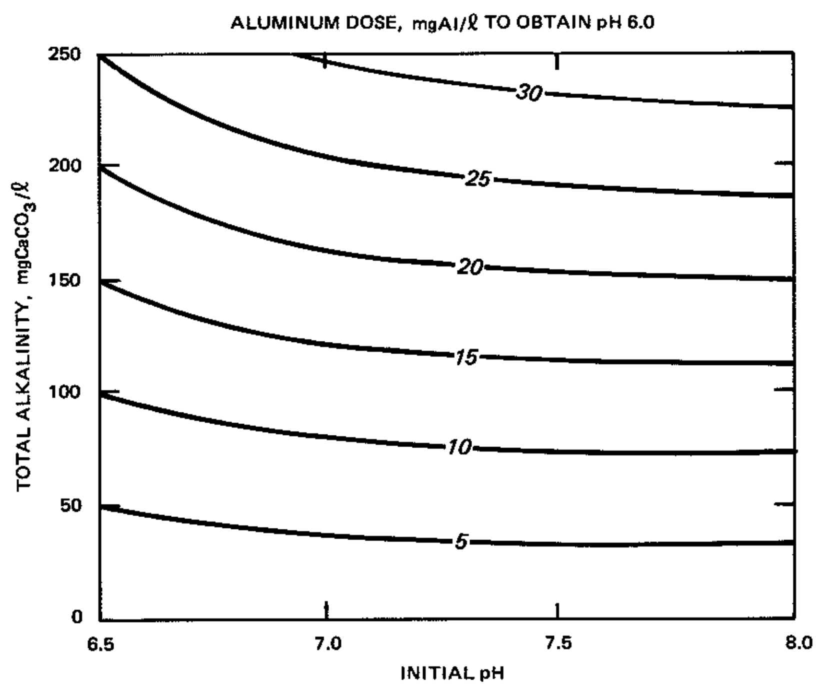

Figure 7. Estimated aluminum sulfate dose (mg Al ℓ^{-1}) required to obtain pH 6 (i.e., "maximum dose") in treated water of varying initial alkalinity and pH (from Kennedy and Cooke 1982)

In soft waters, only small amounts of aluminum sulfate can be added before the pH falls below 6.0. A. R. Gahler and C. F. Powers of the Corvallis Environmental Research Laboratory (USEPA, undated report, Corvallis, OR) suggested that sodium aluminate, which increases the pH of an aqueous solution, could be used with aluminum sulfate to maintain a pH between 6.0 and 8.0. Dominie (1978) was apparently the first to successfully use this dose approach on a large scale when Annabessacook Lake, Maine (alkalinity, 20 mg $CaCO_3$ ℓ^{-1}) was treated with this mixture in an empirically determined ratio of 1:1.6 (alum to sodium aluminate). Another alternative is to add materials such as lime or $CaCO_3$ to buffer the alum. Before attempting an alum treatment, the reader is urged to consult the primary literature, especially Kennedy and Cooke (1982) and Cooke et al. (1986), for a detailed, step-by-step outline of the dosage determination procedure.

Figure 8 illustrates the design of the application equipment used at Dollar and West Twin Lakes, Ohio (Kennedy 1978; Cooke et al. 1978, 1982; Kennedy and Cooke 1982). The delivery system was mounted on barges, and aluminum sulfate, mixed 50-50 with lake water, was pumped to an application manifold that was below the barge at the top of the hypolimnion. This allowed direct injection of the inactivant to the nutrient-rich anoxic hypolimnion and sediments without significant leakage to the littoral zone. As designed, the system added 140 m^3 of liquid aluminum sulfate in 3 days to a hypolimnetic area of 16 ha. Delivery systems similar to this have been used to treat much larger areas, but they are all labor-intensive. The development of a more rapid application system is needed. One option, where a large harvester is available, is to use the front cutter bar to attach the delivery manifold and the weed storage area to hold alum tanks. The harvester's hydraulic system can be used to operate the pumps.*

Ideally, based upon experiences with lakes, the entire area of reservoir sediments should be treated, particularly the area that becomes anoxic. Practically, this may not be possible in reservoirs due to their large size. An alternative is to determine those areas of reservoir sediments with the highest release rates of phosphorus and to treat them. This approach may

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* Personal Communication, 1986, G. N. Smith, Aquatic Control Technology, Inc., Northborough, MA.

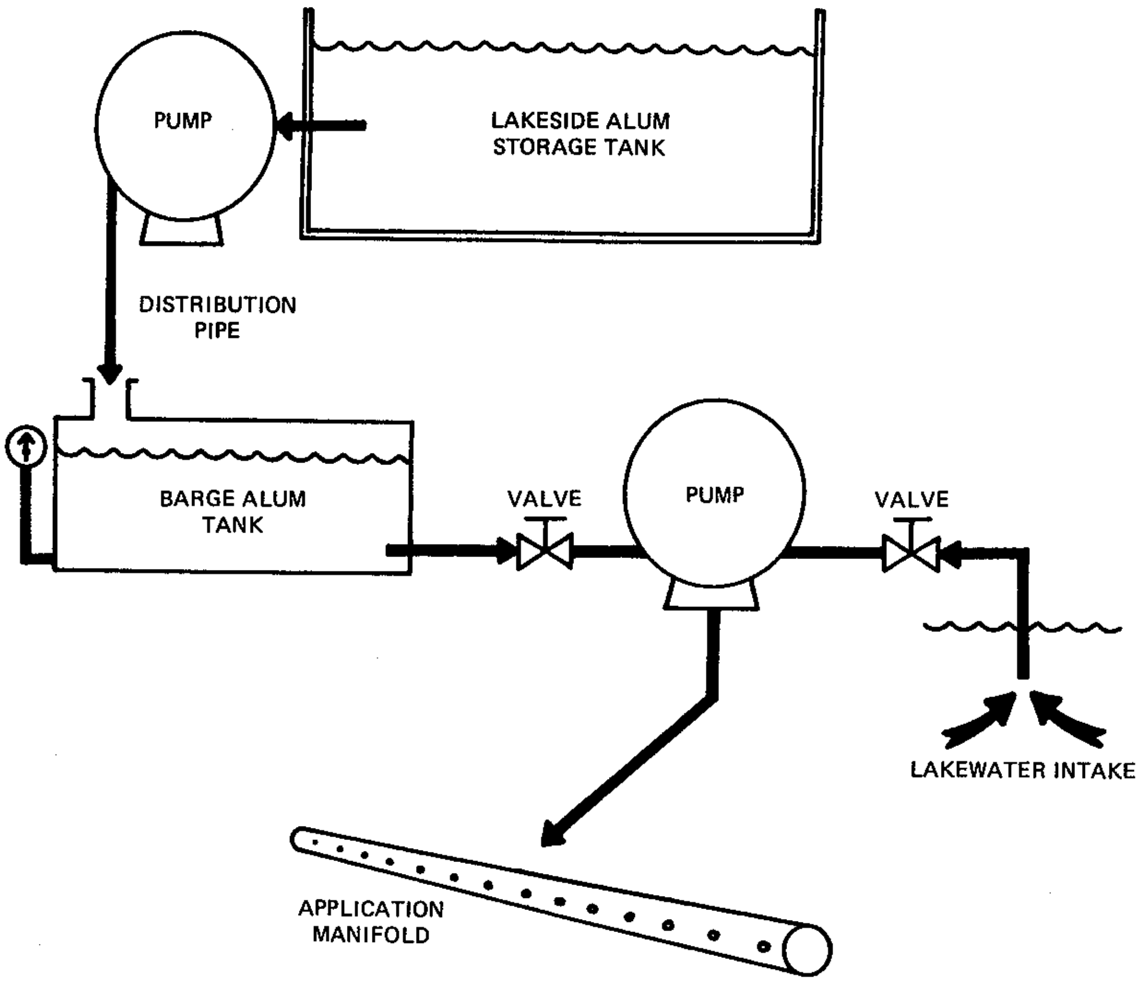

Figure 8. Generalized diagram of an alum application system (from Kennedy and Cooke 1982)

limit the treatment area, but produces nearly the effectiveness of a treatment of the entire reservoir (Cooke and Carlson 1986).

It should be recognized that there may be extensive phosphorus release from aerobic sediments and sediments exposed to high pH, such as may occur in littoral areas during periods of high rates of photosynthesis. Much of the release of nutrients involves microbial metabolism and can be expected to be high when the water is warm. Also, it should be noted that a range of sediment types having various phosphorus sorptive or phosphorus release characteristics may be found in a reservoir. An exceptional review of this topic is provided by Bostrom, Jansson, and Forsberg (1982).

The areas of highest potential phosphorus release can be determined by studying release rates of sediment cores in the laboratory. Samples of sediments are obtained with an Ekman dredge, or a corer, from shallow and deep waters along the length of the reservoir. Sediments from inlets, macrophyte beds, and areas of anoxia should be included. Frevert (1980) and Lennox (1984) provide descriptions of laboratory procedures to evaluate the potential of a sediment sample to release phosphorus under aerobic and anaerobic conditions, high and low pH, and various temperatures. This survey will produce a map of reservoir sediments with regard to their potential to release phosphorus. At least the high-release rate areas could receive an alum application (Cooke and Carlson 1986).

Effectiveness, Costs, and Feasibility

The effectiveness of phosphorus inactivation and phosphorus removal in improving trophic state is best described through case histories that illustrate the range of conditions that have been treated. Four case histories have been chosen to describe the use of the method with aluminum salts, including the first United States treatment, a high alkalinity-high dose case, a soft water-large area application, and treatment of a shallow, nonstratified lake. A case history of fly ash application is also described.

Case histories

Horseshoe and Snake Lakes, Wisconsin. The first US lake to be treated was Horseshoe, in 1970. This small eutrophic lake received a dose of 2.6 g Al m^{-3}. In 1982, 12 years after the application, the concentration of phosphorus in the hypolimnion and the whole lake remains low, compared with pretreatment years. The lake continues to receive nutrient-rich drainage water, but the appearance of the lake has remained better than before (Garrison and Knauer 1984).

Snake Lake, a soft-water lake, was treated in 1972 with a mixture of aluminum sulfate and sodium aluminate. After application, transparency was greatly improved. Phosphorus concentration remained low through 1982 (Garrison and Knauer 1984).

These two case histories are useful to reservoir managers because they clearly demonstrate the longevity of the effect of the aluminum hydroxide floc on phosphorus concentration. Neither lake received a maximum dose, as defined by Kennedy (1978), yet their treatments were long-lasting.

Eau Galle Lake, Wisconsin. This small reservoir (area, 16 ha; mean depth, 3.2 m; drainage area, 16,600 ha) may be the first reservoir in the

United States to be treated with aluminum to control internal phosphorus release. A dose of alum equivalent to five times the average summer internal phosphorus load was added to the hypolimnion in May 1986. Deep-water phosphorus concentration, internal loading, and blue-green algae were reduced relative to previous summers. Algal biomass remained high because a bloom of the dinoflagellate *Ceratium* occurred. External phosphorus loading remained high and may have contributed to the *Ceratium* bloom (Kennedy et al. 1987).

Annabessacook Lake, Maine. Annabessacook Lake is one of the largest (575 ha) water bodies to be treated by this method. The lake supported intense blue-green algal blooms, even following nutrient diversion, due to internal phosphorus loading. Since the water is soft (20 mg $CaCO_3$ ℓ^{-1}), only small amounts of aluminum sulfate could be used before pH 6.0 was reached and dangerous levels of dissolved aluminum (Al^{+3}) appeared. A mixture of aluminum sulfate and sodium aluminate in a 1:1.6 ratio was determined through jar tests to be a dose that would maintain pH in the 6 to 7 range. Over an 18-day period, this dose was applied to the top of the hypolimnion (130 ha) with a barge upon which tank truck trailers had been driven. A concentration of 25 g Al m^{-3} was applied to the 8- to 10-m contour; 35 g Al m^{-3} was applied to the 10-m contour and deeper (Dominie 1980).

A 65-percent reduction in internal phosphorus loading occurred in summer 1979, following the 1978 application. Blue-green blooms were absent in 1979 (Dominie 1980).

Pickerel Lake, Wisconsin, and Long Lake, Washington. Application to shallow, nonstratified lakes was believed to be inappropriate because it was thought that the aluminum hydroxide would be dispersed and relocated during turbulent weather. This problem is important because many shallow, eutrophic reservoirs might also experience this problem.

This concern is supported by the results of the phosphorus removal treatment of shallow, holomictic Pickerel Lake, Wisconsin. A dose of 7.3 g Al m^{-3} was applied in April 1973, and total phosphorus was sharply reduced. After a series of mixing events, total phosphorus returned to pretreatment levels and an analysis of the sediments showed that the aluminum hydroxide floc had been redistributed to the lake's center. This left areas of the sediment free to continue phosphorus release (Knauer and Garrison 1980).

At Long Lake, Washington, however, a maximum dose of aluminum sulfate was applied. Total phosphorus declined, along with phytoplankton biomass and pH. Transparency increased. Internal phosphorus release was curtailed. This effect has lasted 4 years, and the floc was not redistributed during a winter of high winds and high flushing (Welch, Michaud, and Perkins 1982; Jacoby, Welch, and Michaud 1983; Welch et al. 1988). During the fifth summer, phosphorus levels were elevated, along with algae, and transparency declined. The floc layer apparently was dispersed to a deeper layer of sediment and also became covered with new phosphorus-rich materials (Welch, DeGasperi, and Spyridakis 1986).

Lake bottom slope, sediment chemistry, dose, and application procedure are among the factors that could have produced the disparity between the two shallow, holomictic lakes with regard to floc redistribution. It would be logical to be concerned about the problem in shallow, highly mixed reservoirs. Only further testing can provide the answers.

Lake Charles East, Indiana. Theis et al. (1979) describe the treatment of a section of Lake Charles East, Indiana, with fly ash, during summer 1975, for the purpose of sealing the sediments to prevent internal loading. There appears to be no other published report of the full-scale use of this substance for this purpose.

About 1,430 metric tons of fly ash and 275 metric tons of CaO were added to a 8.7-ha area of the lake. Some evidence of a reduction in phosphorus concentration appeared, algal blooms were reduced, transparency increased, and the phytoplankton was no longer dominated by blue-green algae. However, heavy metals, apparently from the fly ash, led to extensive mortality to fish and invertebrates.

This case history, plus the several laboratory experiments with fly ash (reviewed in Cooke 1980 and Cooke et al. 1986), illustrates the danger of using fly ash in lakes and reservoirs. Until further studies are completed, fly ashes should not be used for reservoir restoration (Cooke 1980).

Costs

The principal cost of adding an aluminum salt to a reservoir is labor, and labor costs appear to be dependent upon dose. Cooke and Kennedy (1981) summarized the small amount of published data on costs, and Cooke et al. (1986) provided this equation for estimating man-days of labor from a determination of its maximum dose, based upon reported costs from six lakes:

$$Y = 0.55 + 0.1614X$$
$$r^2 = 0.9411$$

where Y represents the man-days per hectare and X is the dosage in grams of aluminum per cubic meter.

Aluminum sulfate costs vary with the market and, in recent years, a ton of liquid alum has cost about \$160 to \$170. Equipment costs also will vary with the size of the application. Dominie's (1980) technique of using a barge big enough to load tank truck trailers on it represents a way of reducing costs, since lakeshore storage and delivery systems would not be needed. Also, as suggested earlier, a large harvesting machine could be modified for use as an alum applicator. The cost of the equipment, as well as labor, may also vary with the depth of application. Several treatments have been directed toward hypolimnetic sediments only, and a manifold or other injection device was needed that could pump materials to the hypolimnion. A surface treatment could be accomplished with less equipment. Phosphorus inactivation is a procedure that would benefit from new designs for application.

Limitations and Concerns

The potential for serious negative impacts from low pH or the toxic effects of dissolved aluminum clearly exists with the addition of an aluminum salt to a reservoir. Aluminum sulfate, as described earlier, will produce a shift toward a low pH. At pH 5.5, dissolved aluminum Al^{+3} will begin to appear, and its concentration will increase rapidly as pH declines. Toxic conditions could be reached.

Fish mortality has not occurred during alum applications (Funk et al. 1982, Lamb and Bailey 1983). There was little or no appearance or accumulation of aluminum in the tissues of rainbow trout (*Salmo gairdneri*), as reported by Buergel and Soltero (1983), or in tissues of channel catfish (*Ictalurus punctatus*), largemouth bass (*Micropterus salmoides*), and gizzard shad (*Dorosoma cepedianum*), as reported by Berg and Burns (1985), in lakes treated with alum but maintained at a pH of 7.0 or greater. Biesinger and Christensen (1972), Peterson et al. (1973, 1974), and Lamb and Bailey (1981, 1983) have indicated that a dissolved aluminum concentration below 50 μg Al ℓ^{-1} will not bring about harmful effects to *Daphnia magna* (zooplankton), rainbow trout (*S. gairdneri*), and insect (chironomid) larvae (*Tanytarsus dissimilis*). This concentration will not be reached if pH 6.0 or more is maintained (Kennedy and Cooke 1982). Havas and Likens (1985) have found that the zooplankton *Daphnia catawba* and *Holopedium gibberum* and the insects *Chaoborus punctipennis* and *Chironomus anthracinus* were tolerant of aluminum concentrations higher than 300 $\mu g\ Al\ \ell^{-1}$. Narf (1978) reported that there had been no damage to the invertebrate populations of four Wisconsin lakes during several years of monitoring after alum applications. A report by Moffett (1979) suggests that species diversity of planktonic microcrustacea in West Twin Lake, Ohio, was reduced for at least 3 years after an alum treatment in which dissolved aluminum never exceeded 2 $\mu g\ Al\ \ell^{-1}$ and pH and alkalinity returned promptly to normal. Gibbons et al. (1984), however, found no lasting impact to the zooplankton of Liberty Lake, Washington, after an alum application, supporting the conclusion of Moffett that predation may have produced the zooplankton shift.

Much more research, especially field studies, is needed concerning the toxicity of aluminum to aquatic communities. However, it appears, from the laboratory and limited field data, that few risks to biota can be expected if pH 6.0 or above is maintained. It should be noted that soft-water lakes found in regions that receive extensive acid precipitation could be a future hazard after an aluminum treatment, if lake pH falls significantly below pH 6.0 in the years following aluminum treatment.

Bulson et al. (1984) have observed that the aluminum hydroxide floc is very efficient in the removal of fecal coliform and fecal streptococci bacteria during a lake treatment, suggesting that enteric species, including pathogens, might also be accumulated. Bacteria appear to die off in the floc and are not released from it. Bulson et al., however, suggest that there be a posttreatment restriction on recreational use, or a restriction of treatment to the nonpeak recreational season to allow a long die-off of bacteria. They also caution that intake of floc into a potable water treatment plant could pose a health hazard. It is likely, in many cases, that the pretreatment process with alum in potable water treatment plants should remove any floc in the raw water intake.

Aluminum sulfate and sodium aluminate applications will bring about greatly increased water clarity. This benefit of the method could produce a significant increase in the area of the reservoir that is infested with submerged macrophytes, since the outer depth limit of their growth can be light-limited. There is evidence that this has occurred in West Twin Lake, Ohio (Cooke et al. 1978), and in Long Lake, Washington (Jacoby, Welch, and Michaud 1983).

Fly ash presents a serious environmental hazard and should not be used in reservoirs (Cooke 1980). Fly ashes from bituminous coals (eastern United States) are high in sulfur, and aquatic solutions have a low pH. This environment will promote solubility of the heavy metals which they contain. Lignitic coals (western United States) produce a high pH (above pH 12) in solution and also contain heavy metals (Adriano et al. 1980). Theis and DePinto (1976) report the following negative attributes of fly ash: (a) high pH of treated waters, (b) dissolved oxygen depletion, (c) appearance of sulfide, (d) heavy metal release, and (e) physical crushing of biota or clogging of gills. Various laboratory and field studies have demonstrated the toxicity of various fly ashes to fish and invertebrates (e.g., Cairns, Dickson, and Crossman 1972; Guthrie and Cherry 1979).

Summary

Phosphorus inactivation is a technique to control the release of phosphorus from reservoir sediments, a source of "internal loading" that can maintain severe algal blooms even after diversion of nutrient income. Aluminum sulfate or sodium aluminate will produce the formation of aluminum hydroxide in water with carbonate alkalinity. This hydroxide is a visible floc or precipitate that is very sorptive of phosphorus and will not release it under conditions of low dissolved oxygen. A procedure for determining the maximum dose for a reservoir has been outlined. This dose will produce the largest amount of floc possible, consistent with environmental safety.

Case histories of the procedure have been reviewed (Cooke and Kennedy 1981, Cooke et al. 1986). This treatment has been effective for up to 12 years in controlling phosphorus release and in improving the trophic state of lakes. Large (575 ha), deep (18 m), soft-water (20 mg $CaCO_3$ ℓ^{-1}), hard-water (750 mg $CaCO_3$ ℓ^{-1}), and shallow (2 m) lakes have been successfully treated. Application procedures for very large areas, such as many reservoirs, have not been developed.

Aluminum applications pose significant risk to biota and possibly to human consumers of the water if the pH of treated water falls below pH 6.0 and dissolved aluminum (Al^{+3}) appears. The dose determination technique is designed to prevent this occurrence.

Fly ash has also been suggested as a phosphorus inactivant. This material will produce significant adverse environmental impacts and should not be added to reservoirs.

Table 3 is a summary of this method.

Table 3

Summary of Phosphorus Inactivation

Characteristic	Description
Targets	Nuisance algal blooms, low transparency, release of phosphorus from sediments.
Mode of action	Phosphorus release from reservoir sediments is sharply reduced, producing lowered phosphorus concentrations in water column.
Effectiveness	Highly effective, problem eliminated when accompanied by significant diversion of external nutrient loading.
Longevity	Up to 12 years; few long-term evaluations available.
Negative features	Use of aluminum sulfate will lower pH.
	Overdose could produce appearance of toxic dissolved aluminum.
	The floc may contain a high density of bacteria, including pathogens.
	Application is labor intensive.
Costs	Labor and chemical costs will be high but can be determined if dose is known (see text for equations).
Applicability to reservoirs	No published record of use in reservoirs with large areas.
	New methods of application may have to be developed to lower costs.
	Treatment of high phosphorus-release areas should be attempted.

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PART V: DILUTION AND FLUSHING

Problem Addressed

Reservoirs and lakes with high concentrations of nutrients may have severe blooms of nuisance algae. Algal blooms, particularly when blue-green algae are involved, interfere with the recreational use of the water and may dramatically increase potable water treatment costs. Algal excretory and decomposition products, along with other sources of dissolved organic matter, are associated with dissolved oxygen depletions, taste and odor, and with the appearance of trihalomethanes and other organohalides following chlorination of drinking water.

Dilution is a procedure in which water of low nutrient content is added for the purpose of lowering the reservoir's concentration of nutrients to a level at which algal cell growth is limited. Cell washout increases as well. Flushing, on the other hand, emphasizes cell washout through a sharp increase in the water exchange rate. The inflowing water may not necessarily have a lower nutrient concentration. The procedures become equivalent when low nutrient water is added at a rate sufficient to achieve cell washout equal to algal cell growth rate. This normally requires a large volume of scarce low-nutrient water. In practice, the procedures are differentiated because one (flushing) emphasizes what goes out of the reservoir without consideration of nutrient concentration changes and associated changes in cell growth, and the other (dilution) emphasizes a limitation on algal growth through a decrease in nutrient concentration as well as through cell washout (Welch 1981). Dilution can also improve water quality by decreasing the concentration of algal excretory and decomposition products. These procedures can be particularly effective for some reservoirs when treatment of upstream nutrient sources is not feasible. Both techniques, but particularly dilution, are limited in their applicability by the difficulty of finding an additional water source that can be diverted to the reservoir. The reader is referred to Uttormark and Hutchins (1978), Welch (1981), and Cooke et al. (1986) for reviews of these techniques.

Theory and Design

Uttormark and Hutchins (1978) were among the first to clearly describe the effect of adding low-nutrient water to a lake's inflow. According to these researchers, the following changes will occur: (a) the areal and volumetric phosphorus loading will increase with the increased income of water containing phosphorus, (b) the mean phosphorus concentration in inflowing water will decrease, and (c) the flushing rate will be increased and the sedimentation of phosphorus will decrease.

The effect of a change in loading, flushing, and sedimentation on in-lake phosphorus concentration is described by models (Vollenweider 1976, Uttormark and Hutchins 1978, Cooke et al. 1986, and Walker 1987) which assume, on a long-term basis, that lakes can be described as completely mixed reactors in which it is assumed that phosphorus income is constant, that net sedimentation is proportional to the amount of phosphorus in the lake, and that phosphorus is lost through the outlet and by sedimentation. The reader is referred to Walker (1987) for significant additional details and discussions of these models.

At steady state the lake's phosphorus concentration is described as

$$[P] = [P_o] \frac{\rho}{\sigma + \rho} \tag{9}$$

where

$[P]$ = in-lake total P concentration, g m^{-3}

$[P_o]$ = in-flow total P concentration, g m^{-3}

ρ = flushing rate = $\frac{Q}{V}$, $year^{-1}$

Q = annual water flow rate, m^3 $year^{-1}$

V = lake volume, m^3

σ = sedimentation rate, $year^{-1}$

An alternative to Equation 9 is

$$[P] = \frac{L}{\bar{Z}(\rho + \sigma)} \tag{10}$$

where

L = total P income, g m^{-2} year^{-1}

$\bar{Z}$ = mean depth, m

Larsen and Mercier (1976) and Vollenweider (1976) found that the specific phosphorus sedimentation rate, a term that is very difficult to determine empirically, can be estimated as

$$\sigma = \sqrt{\rho}$$

Equations 9 and 10 are thus rewritten as

$$[P] = \left(\frac{L}{\bar{Z}\rho}\right)\left(\frac{\rho}{\sqrt{\rho} + \rho}\right) \qquad (11)$$

or

$$[P] = \left(\frac{L}{\bar{Z}\rho}\right)\left(\frac{1}{1 + \frac{1}{\sqrt{\rho}}}\right)$$

It should be noted, as first pointed out by Uttormark and Hutchins (1978), that phosphorus sedimentation is inversely related to flushing rate so that the amount of incoming phosphorus that is deposited in the reservoir bottom will decrease as the inflow is diluted with additional water. The effect of dilution is that a decrease in the concentration in the inflow may reduce in-lake concentration, but the decrease in sedimentation will increase lake concentration.

Uttormark and Hutchins (1978) derived an expression from Equation 11 which allows comparison of predicted in-lake phosphorus concentrations, following dilution, with that before the addition of dilution water. Thus,

$$\frac{[P]'}{[P]} = \left(1 + \frac{\rho_2[P_0]_2}{\rho_1[P_0]_1}\right)\left(\frac{\rho_1 + \sqrt{\rho}_1}{\rho_1 + \rho_2 + \sqrt{\rho}_1 + \rho_2}\right) \qquad (12)$$

where [P]' is equal to the lake concentration after dilution (subscript 1 refers to conditions before dilution and subscript 2 refers to conditions after dilution).

Figures 9 and 10 illustrate the effects of the addition of dilution water on the in-reservoir concentration. The X-axis gives the flushing rate before dilution, and the lines on the graph show flushing rates due to dilution only, expressed as a constant proportion of undiluted flow. Thus, using Equation 12 and assuming that there is no phosphorus in the dilution water and that the dilution is equal to half the normal flow for a reservoir with a normal flushing rate of 1.0 year^{-1} ($\rho_2 = 0.5\rho_1$), theory predicts a 26-percent reduction in in-reservoir phosphorus concentration. As Uttormark and Hutchins point out, and as Figure 9 illustrates, large quantities of dilution water are needed to produce a significant change in reservoir phosphorus concentration.

Figure 10 illustrates the more realistic case wherein dilution water contains 40 percent of the phosphorus concentration found in the normal undiluted inflow. This graph clearly shows that greater and greater quantities of dilution water do not necessarily result in progressively greater

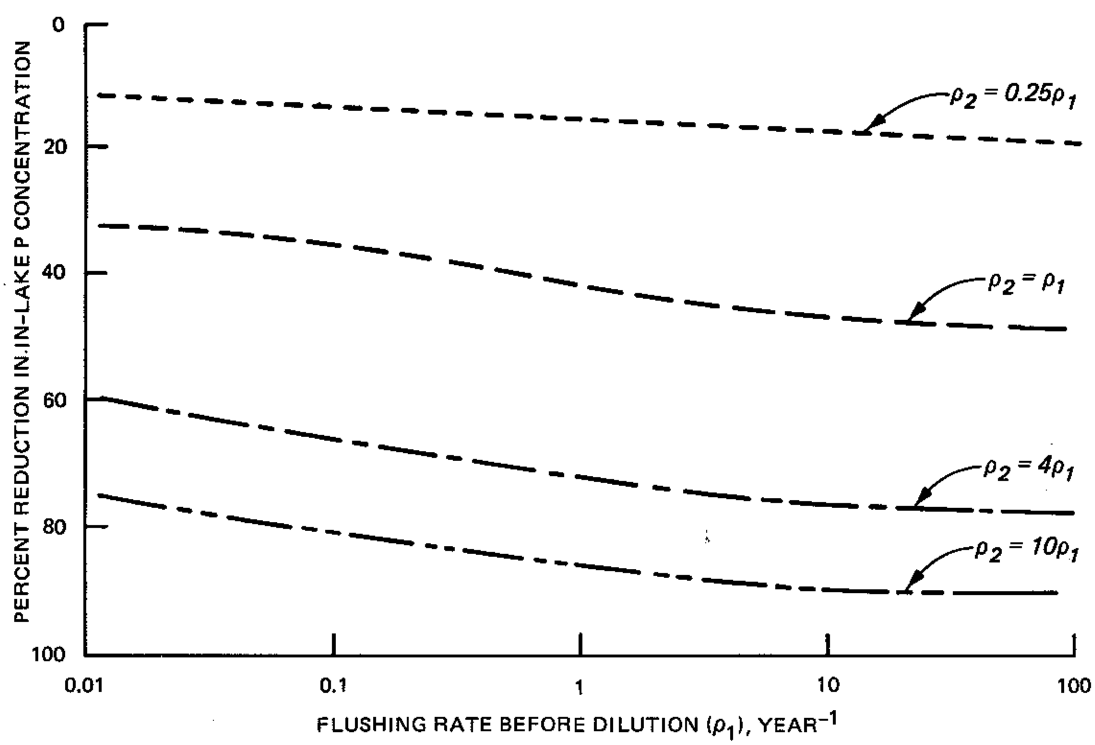

Figure 9. Percent change in in-lake phosphorus concentration following dilution with water containing no phosphorus (after Uttormark and Hutchins 1978). (See text for explanation)

reduction in in-reservoir phosphorus concentration. This graph also illustrates the best possible reduction in phosphorus concentration, given an initial flushing rate before dilution. Thus, if the initial flushing rate is 1.0 $year^{-1}$, a 30-percent reduction in in-reservoir phosphorus concentration approaches the best possible reduction, even with unlimited dilution water. It can also be seen from Figure 10 that in-reservoir concentration can increase by adding dilution water when the counteracting effect of decreased loss to sediments is considered.

Uttormark and Hutchins (1978) conclude that lakes with low flushing rates are poor candidates for improvement through dilution. In these cases, in-reservoir phosphorus concentration could increase (see Figure 10, low flushing rates, <0.1 $year^{-1}$) unless the dilution water is essentially void of phosphorus.

The model used here does not account for internal loading. If summer internal phosphorus loading, a common phenomenon in many lakes and reservoirs, is high, then there may be less reduction in concentration than expected. Substantial empirical studies of dilution are greatly needed.

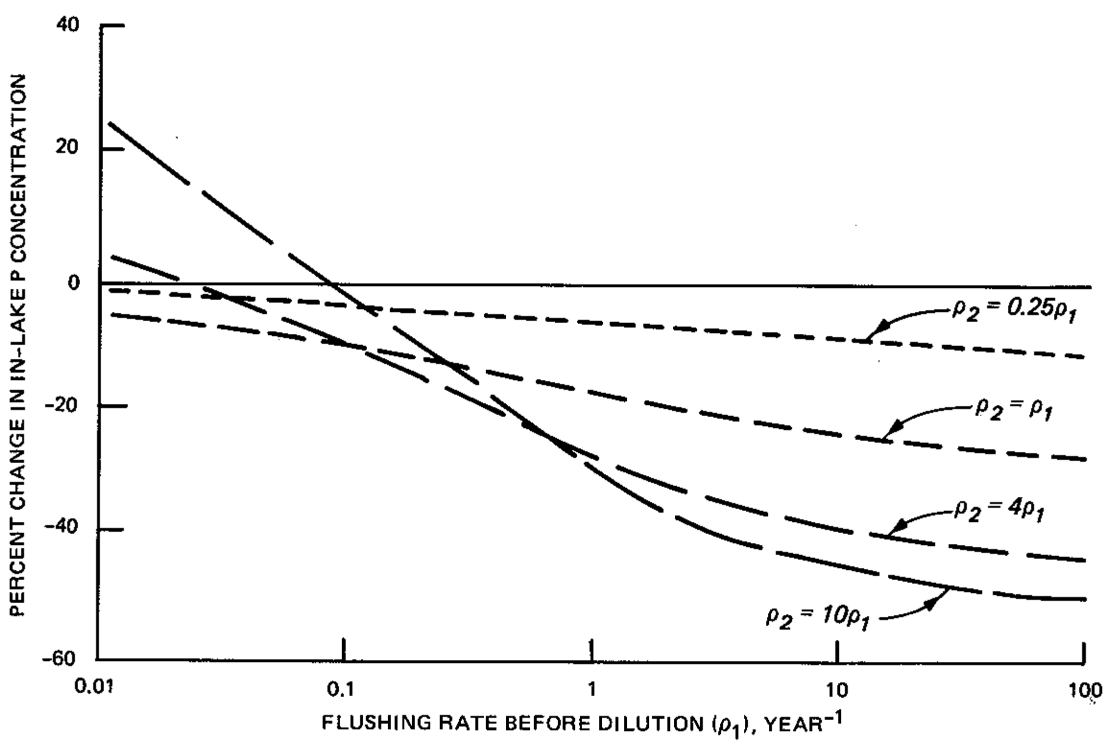

Figure 10. Percent change in in-lake phosphorus concentration following dilution with water having a phosphorus concentration that is 40 percent of the normal, undiluted inflow water (after Uttormark and Hutchins 1978). (See text for explanation)

Flushing, in contrast to dilution, does not require that the nutrient concentration in the inflowing water be less than that of the reservoir. Quantity of algal cells is controlled not by nutrient limitation but by washout. The flushing rate therefore must be close to the algal growth rate to be effective. Flushing rates of 10 to 15 percent per day are believed to be sufficient (Cooke et al. 1986).

Effectiveness, Costs, and Feasibility

There are only a few published case histories of the use of dilution/flushing to improve trophic state, and only two have substantial long-term documentation. These two are Moses Lake, Washington (Welch, Buckley, and Bush 1972; Welch 1979, 1981; Welch and Patmont 1979, 1980; Welch and Tomasek 1980; Welch, Brenner, and Carlson 1984; Cooke et al. 1986) and Green Lake, Washington (Sylvester and Anderson 1964; Oglesby 1968, 1969a,b; Welch 1981; Cooke et al. 1986).

The feasibility of this method for reservoir improvement is very limited since an adequate supply of low-nutrient dilution water or high flows of additional water for flushing are unlikely to be available in most instances. Further, even if there is a potential supply of water, its use for reservoir dilution/flushing may be restricted by prior usage of the water. Since this method of reservoir improvement is likely to have limited use, the results of the Moses and Green Lakes studies will only be briefly reviewed.

Moses Lake, Washington

Crab Creek, the primary water supply to this large (2,753-ha), relatively shallow (mean depth, 5.6 m) lake in eastern Washington, has very high nutrient content. In 1977, dilution water addition to Parker Horn began, using low-nutrient Columbia River water that was diverted through Moses Lake and thence to agricultural areas for irrigation. This produced overall water exchange rates of 0.1 to 0.16 day^{-1} for Parker Horn, and 0.01 to 0.02 day^{-1} for the whole lake. In 1982, dilution water was pumped to previously undiluted Pelican Horn from Parker Horn.

The percent lake water in Parker Horn dropped to less than 30 percent when the dilution rate reached 0.15 day^{-1}. Dramatic improvements in lake quality occurred, not only in Parker Horn but in the entire lake. However, it was obvious that algal blooms and low transparency returned quickly if the amount of dilution water declined. This observation led Welch (1981) to conclude that continual low-rate inputs of dilution water over the entire summer were preferable to very high but irregular rates which are above the amount that can produce a decline in nutrients or a washout of cells. When input of dilution water stopped (August 1982), undiluted high-nutrient water rapidly replaced the diluted lake water.

Another effect of dilution in Moses Lake, in addition to creating nutrient limitation, was the effect of cell washout (Cooke et al. 1986). When water was pumped from Parker Horn to Pelican Horn, a sharp decrease in algal biomass occurred, particularly when the water exchange rates reached 0.09 day^{-1}. Similarly for Parker Horn, cell washout became a significant factor when the mean flushing rate was 10 percent day^{-1}. In the remainder of the lake, where flushing averaged 1.4 percent day^{-1}, cell washout was probably not a significant factor because cell growth rates, at maximum, can exceed 50 percent day^{-1}.

The cost of water for dilution at Moses Lake was zero since water already designated for downstream irrigation was simply routed through the lake. The pump for Pelican Horn cost $324,000 (1983 price), plus overhead for operations. If the water had had a cost similar to that of a typical Washington domestic supply, the 2-month cost of dilution water would have been about $2 million.

Green Lake, Washington

Dilution of Green Lake, in metropolitan Seattle, WA, began in 1962. Domestic water was added at a rate sufficient to increase the water exchange rate from an estimated 0.8 to 2.3 $year^{-1}$. Over the 1965-68 period, the flushing rate, based on dilution water only, ranged from 0.88 to 2.4 $year^{-1}$ (Welch 1981, Cooke et al. 1986). Chlorophyll *a*, phosphorus concentration, and water transparency improved dramatically, and the fraction of algal biomass composed of blue-greens declined substantially. Water quality declined in the 1970s when dilution was reduced, and blooms of algae returned in 1982 when no dilution water was added. High costs were incurred because domestic water is expensive. It was calculated, using the mass balance models described earlier, that 7.6×10^6 m^3 $year^{-1}$ (269×10^6 ft^3 $year^{-1}$) of water would be needed to reduce the mean concentration of phosphorus in this medium-sized lake (area = 104 ha, mean depth, 3.8 m) to 20 μg P ℓ^{-1}.

Limitations and Concerns

These techniques are limited by the availability of low-cost, low-nutrient water. Outlet structures of the reservoir must be capable of handling the added discharge. This may not always be the case, particularly with smaller, older impoundments. Also, the downstream impacts of significantly increased discharge must be considered. Finally, dilution/flushing water must have acceptably low concentrations of contaminants such as heavy metals or pesticides.

Summary

Dilution is a reservoir improvement technique wherein amounts of low-nutrient water are added in quantities sufficient to promote cell washout and to significantly lower in-lake nutrient concentration. The amount of reduction in concentration can be estimated, with assumptions that may not hold true for eutrophic reservoirs, from knowledge of nutrient loading, sedimentation, and flushing rate. Flushing is a procedure to wash out algal cells and does not imply dilution of nutrient concentration in the reservoir unless water with low concentration is used.

Both techniques can produce large improvements in trophic state, as illustrated by two case histories. The primary drawback to their use is the availability of the additional water and possible effects of increased reservoir discharge on downstream areas.

Table 4 summarizes this procedure.

Table 4

Summary of Dilution and Flushing

Characteristic	Description
Target	Blooms of algae.
Modes of action	Dilution water decreases in-reservoir limiting nutrient concentration and increases cell washout.
	Flushing increases cell washout.
Effectiveness	Highly effective.
Longevity	Requires continual water input during growing season.
Limitations and applicability	Dam must be structurally sound.
	Water should be free of toxic substances.
	Downstream impacts of greatly increased discharge could be significant.
	Limited due to shortage of additional and/or appropriate quality water.
Costs	Price could be very high if domestic water supply is used.

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PART VI: SEDIMENT REMOVAL

Problem Addressed

External loading of silt and organic matter, along with the deposition of partially decomposed plant biomass produced in the reservoir, can bring about loss of reservoir volume. As well, nutrient-rich sediments are sources of nutrients to the water column and, in shallow areas, provide ideal conditions for macrophytes. Shoaling may interfere with boating, and loss of storage capacity can have severe impacts in potable water supply and flood control reservoirs. Dense macrophyte infestations interfere with recreation and, along with algae, contribute organic matter to the water column. This promotes loss of dissolved oxygen in deep water, and the organic matter may interact with the chlorination step in potable water treatment to produce organohalides such as trihalomethanes. Some reservoir sediments can contain significant levels of toxic substances (e.g., heavy metals, PCBs) from upstream discharges.

Sediment removal is a highly effective method to deepen reservoirs and to remove shoals, and secondarily to remove nutrient-rich or toxic sediments and to control rooted plants. This procedure has been reviewed by Peterson (1979, 1981, 1982) (see also Cooke et al. 1986).

Theory, Design, and Costs

The object of most sediment removal projects is to regain lost storage capacity, and secondarily to improve water quality by control of internal nutrient release. In the cases of shoaling, loss of volume, or toxic substance contamination, there is little choice except sediment removal. Internal nutrient loading may also be controlled with other methods, and macrophyte control may be better and less expensively handled through harvesting, herbicide treatments, water level drawdown, or biological controls.

Application of the method

There are two means of removing reservoir sediments. First, the reservoir may be drawn down and the sediments allowed to dewater. This is followed by the use of mechanical equipment to remove sediments. Obviously the procedure is limited to those reservoirs where significant and long-term water withdrawal is possible and where sediments will dewater to the degree needed to support heavy equipment. Born et al. (1973) describe the use of this procedure in a small reservoir, and Snow et al. (1979) describe a case history. The more common method of sediment removal is the use of dredges.

The two basic categories of dredges are mechanical and hydraulic, plus some special-purpose designs for the removal of toxic or fine-grained sediments. These are reviewed by Barnard (1978), Peterson (1979), and Cooke et al. (1986). The most common mechanical dredges are the clamshell or grab-bucket designs. Dredges of this type are limited by the requirement that discharge is either in the immediate vicinity of the dredge or into a barge or truck. The bucket types have low productivity rates, can create uneven bottom contours, and produce significant sediment resuspension. They are highly mobile and can work in small areas. The hydraulic cutterhead dredge is the most common type. Cutterhead dredges are faster than the grab-buckets, may produce less turbidity, and are able to dredge over large distances due to their floating pipeline discharge system. However, up to 80 percent of the removed material is water; therefore, confined disposal areas must have adequate volumes to permit the settling of suspended materials. In small reservoirs, there could be some drawdown due to the hydraulic dredging process.

There are also specialized and portable hydraulic dredges. Barnard (1978) has described these, and Clark (1983) has reviewed the operating features of 46 models of portable hydraulic dredges. Also, the Oozer and Clean-up dredges (see review in Cooke et al. 1986) have been developed for removal of contaminated sediments, though it appears that these dredges are unavailable in the United States. Several of these dredges produce very little turbidity and thus little dispersion of toxic materials. Herbich and Brahme (1983) report an average suspended solids concentration of 4.0 and 5.7 mg ℓ^{-1} at 3 and 7 m above the bottom for Clean-up and Pneuma-type dredges, versus 40 to 80 mg ℓ^{-1} for conventional hydraulic cutterhead dredges. However, unless the lake sediments are contaminated, the conventional cutterhead dredges may be used with good results.

Analysis of reservoir
sediments and sediment budget

Sediment removal for the purpose of reducing internal nutrient loading requires a predredging analysis of the sediments to determine those areas of the reservoir with the highest release rates. This analysis should also determine the depth in the sediments to which highly reactive or exchangeable forms of phosphorus and other nutrients extend. The methods of Williams et al. (1971a,b) are recommended for phosphorus. Methods for nitrogen can be found in Chen, Keeney, and Sikora (1979). Release rates should be measured either in situ (Sonzogni et al. 1977) or with sediment cores in the laboratory (Lennox 1984). The study should be conducted in a manner that will produce a map of the reservoir which indicates the areas with high release rates. Appropriate statistically based sampling techniques should be applied to ensure that release rates obtained are representative of the areas examined. For a discussion of considerations required in reservoir sampling and monitoring, see Waide (1986). Sediment removal should be to depths resulting in a significant decrease in nutrient release. There is little value to superficial dredging that leaves nutrient-rich layers exposed.

It is important to know how fast the dredged areas of the reservoir will refill with silt and organic matter. If silt loading is high, it may not be cost-effective to carry out sediment removal. Establishment of appropriate land use management techniques or the construction of prereservoir sedimentation basins might be necessary before dredging. Or, use of dredged materials as top soil could reduce costs (see Stout and Barcelona 1983). Evans and Rigler (1980) and Ritchie and McHenry (1985) describe measurement methods for determining sedimentation rates. Or, direct measurement of the net suspended solids income, particularly during storm runoff events, can be made.

Reservoir sediments in agricultural and industrialized areas may contain PCBs, chlorinated hydrocarbon pesticides, oil and grease, heavy metals, and coliform bacteria. Dredging can release these materials to the water column in association with suspended particulates, and thus the presence of contaminants must be known before initiating operations. Mutagenic substances have been found in reservoir sediments. Allen, Noll, and Nelson (1983) and Lower et al. (1985) describe methods of sediment analysis for mutagenic and toxic materials. Also, an elutriate test (Palermo 1986a,b; 1988) has been devised to evaluate the short-term potential of sediments and disposal area effluents to release hazardous substances into the water column.

Dredge selection

S. A. Peterson's definitive review of sediment removal (in Cooke et al. 1986) provides detailed criteria for the selection of dredge equipment. Additional data are available in Pierce (1970). Since this selection can be highly site specific, the reader is urged to consult these reports.

Containment area design

One of the most common problems with the use of sediment removal is inadequate design of the containment area. Detailed summaries of the procedures for containment area design are found in Palermo, Montgomery, and Poindexter (1978); Montgomery (1978, 1980, 1982, 1984); Averett, Palermo, and Wade (1988); and US Army Corps of Engineers (1987).

The volume of sediments to be removed and the sediment characteristics, such as water content, Atterburg limits, organic content, specific gravity, bulking, grain size, consolidation, and shear strength, must be known. Montgomery (1978) and Averett, Palermo, and Wade (1988) describe the flocculent settling test, which is used to ensure solids retention. These data allow the design of a confined disposal area that will have sufficient volume and area to accommodate continuous hydraulic dredging, and is large and deep enough to allow settling to occur so that the effluent meets suspended solids requirements. Reservoir sediments can be very flocculent, with a low specific gravity (Walsh, Bemben, and Carranza 1984), and the water detention time of the disposal area must be sufficient to allow these materials to settle. If the suspended solids requirement is not met, the project may have to be temporarily stopped or the discharge chemically treated to improve suspended solids removal. In either case, project costs will escalate. Therefore, disposal area design criteria are meant for end-of-project efficiency and not some average or estimated discharge requirements over the entire project period. It is important to note that there is a wide range of settling velocities for sediments so that the use of averages or literature values may produce poorly designed containment areas. The design of a containment area is site specific and should be based on the laboratory settling test (Averett, Palermo, and Wade 1988).

Determination of sediment removal depth

One objective of sediment removal can be the control of internal nutrient release through the removal of nutrient-rich sediments. Reservoir sediments may have a sharp gradient of nutrient concentrations with depth into the sediments, or a horizontal gradient over the reservoir. A map of this vertical and horizontal gradient should be made, as described in earlier paragraphs. Lake Trummen, Sweden, is an example. It was found (Bjork 1972) that 40 cm of silt had accumulated between 1940 and 1965, an interval during which effluents from a flax mill and a wastewater treatment plant discharged to the lake. Sediments below this layer, under both aerobic and anaerobic conditions, had distinctly lower phosphorus release rates. Thus, the depth of sediment removal was judged to be 40 cm.

Stefan and Hanson (1979, 1980) described another method for determining depth of sediment removal to control internal phosphorus release. They observed that in shallow Minnesota lakes, brief periods of summer thermal stratification produced a sharp loss of dissolved oxygen in the hypolimnion, followed by a high rate of phosphorus release from the anoxic sediments. Like shallow, polymictic reservoirs (see Gaugush 1984 for case history), summer wind storms disrupted the thermal stratification, mixed the lakes, and introduced nutrient-rich water to the whole water column. An algal bloom then occurred. Stefan and Hanson calculated the depth that was required for the lake to remain stratified for the entire summer season. This depth became the target depth for sediment removal. This approach, however, requires a massive volume of sediment removal. Cooke et al. (1986) recommend the approach used at Lake Trummen.

There is a direct relationship between transparency and the maximum depth of colonization by submersed macrophytes. While each plant species may have different light requirements and thus different depths to which it can grow, it is possible to estimate the depth to which a reservoir would have to be dredged in order to control nuisance submersed macrophyte growth through light limitation. Canfield et al. (1985) provide the following equations to determine the maximum depth (in meters) of submersed macrophyte colonization (MDC) for Florida and Wisconsin lakes:

State	N	Equation	Coefficient of Determination
Florida	26	log MDC = 0.42 log SD + 0.41	0.71
Wisconsin	55	log MDC = 0.79 log SD + 0.25	0.57

where SD is the Secchi disc depth in meters.

Thus, a Wisconsin lake with a mean Secchi disc depth of 6.6 ft (2.0 m) should have few submersed macrophytes beyond a depth of 9.8 ft (3.0 m), suggesting that sediment removal in shallow, macrophyte-infested areas to this depth might produce significant relief from these plants. In the Florida lakes, a depth of 11.5 ft (3.5 m) might have to be achieved for submersed macrophyte control.

Effectiveness and Costs

Sediment removal is one of the most effective and commonly used methods of improving reservoirs. In most situations where increased depth or storage capacity is desired, or where toxic materials must be removed, sediment removal is the method of choice. In smaller reservoirs, it may be economically and environmentally feasible to dredge the entire reservoir. As the volume of material to be removed increases, so does cost and, more significantly in many cases, so do problems of disposal. Environmental impacts are often short-lived, or can be minimized, assuming that the method is used properly and that adequate containment areas and discharge treatment are available. Negative environmental impacts are most often associated with disposal, and feasibility for any situation may turn on this issue. Case histories of dredging projects are described in Peterson (1981) and Cooke et al. (1986).

Sediment removal has been carefully examined for costs, and detailed reviews are found in Peterson (1982) and Cooke et al. (1986). Cooke et al. (1986) list six factors that influence dredging costs: (a) type of equipment used, (b) volume of material to be removed, (c) availability of a containment site, (d) density of material to be removed, (e) distance to containment area, and (f) ultimate use of removed materials. Saucier et al. (1978) have indicated that costs are also reflected in the price of land for disposal sites, and the value the dredged material may have as a landfill, wildlife site, or future recreation area. Peterson (1981) reports a cost range for 64 US projects of \$0.24 to \$14.00 m^{-3}, with a frequent range for hydraulic dredging projects of \$1.25-\$1.75 m^{-3}. Costs may be reduced through productive or beneficial use of the dredged material (Patin 1981).

Two of the most effective means of controlling internal loading are sediment removal and phosphorus inactivation. Cooke et al. (1986) have compared the cost-effectiveness of these methods, and preliminary evidence suggests that they may be similar when amortized over the effective life of the treatment.

Limitations and Concerns

Sediment removal has high potential for both short- and long-term negative impacts, both at the dredging site and the containment area. Most of these problems are of short duration and can have minimal negative impacts following project completion when containment area design has been proper. Sediments contaminated with toxic materials involve special precautions.

Several possible deleterious actions can occur at the dredging site. These include creation of plumes of turbid water, liberation of nutrients (Churchill, Brashier, and Limmer 1975), destruction of benthic organisms (Carline and Bryneldson 1977), and the release of toxic substances (Murakami and Takeishi 1977). At the disposal site, whether in-reservoir or upland, some of these same problems could occur. In addition, in-reservoir disposal may result in burial of organisms and the creation of new and less desirable substrates. Upland disposal can create nuisance conditions for nearby residents, contaminate ground water, and discharge toxics in the drainage water. Detailed descriptions of these problems are found in Chen et al. (1978); Gambrell, Kincaid, and Patrick (1978); Saucier et al. (1978); and Peterson (1981). In general, these reports indicate that sediment removal and disposal seldom generate significant negative impacts in the short term, except where toxics such as mercury, cadmium, and chlorinated hydrocarbons are involved. Little is known about long-term impacts. A reader contemplating a sediment removal project is urged to consult these reports, especially Gambrell, Kincaid, and Patrick (1978) and Francingues et al. (1985). A brief review of potential environmental problems and some steps to prevent them follows.

Sediment removal itself will create at least a temporary problem with turbidity, nutrient release, and transport of contaminated particles. Normally, particulates settle rapidly. In some situations, turbidity or the transmission of particulate matter to other reservoir areas is undesirable. In these cases, specialized dredges are available (Cooke et al. 1986), or a silt curtain can be installed (Barnard 1978). Montgomery (1984) describes specialized dredging equipment and procedures that can be used to minimize hazards of sediment resuspension while removing contaminated sediments. Disturbance of nutrient-rich sediments may release significant amounts of nutrients, leading to algal blooms. Nutrient levels should return to normal or even lowered concentrations after dredging. Gibbons and Funk (1983) point out errors in hydraulic dredge operation that can produce reservoir problems. In the case of Liberty Lake, Washington, the paths of the cutterhead did not overlap, resulting in mounds and trenches that later merged through slumping. As a result, nutrient-rich sediments still covered the lake bottom, and neither nutrient release nor macrophyte coverage was improved.

Disposal methods and sites are a very important part of the process of minimizing the environmental impacts of sediment removal, and guidelines for their construction for this purpose are available (US Army Corps of Engineers 1987). Upland containment areas are commonly used. Sediment removal for a reservoir improvement project would be defeated by in-reservoir disposal unless the sediments could be placed in very deep water (25 to 30 m) where currents are minimal, or unless the sediments are placed in a containment area used to create an island acceptable to reservoir users. Unconfined disposal in shallow water means that problems may simply be displaced (i.e., creation of new shoals or creation of another site of nutrient release or macrophyte infestation) or that the undesired sediments will be dispersed by currents.

Prior to selection of a disposal method, some preliminary data must be obtained. The short-term pollution potential of nutrients, heavy metals, and organics should be estimated with an elutriate test (Palermo 1986a,b; 1988). While most dredged material poses little risk from release of toxic contaminants, the level of such contamination must be known. In the event that the target sediments are contaminated, environmental risks can be minimized. The reader is referred to the reports referenced above for guidelines and methods to control these factors and risks.

Summary

Sediment removal is used for deepening, and secondarily to remove nutrient-rich or contaminated sediments and to control macrophyte infestations. Sediment removal projects require careful planning, design, and construction since the costs may be considerable and there is potential for negative environmental impacts. Planning will include dredge selection, sediment analysis, and containment area design. A summary of the method is given in Table 5.

Table 5

Summary of Sediment Removal

Characteristic	Description
Targets	Shoaled areas.
	Nutrient-rich or contaminated sediments.
	Nuisance macrophytes.
Mode of action	Sediments are removed.
Effectiveness	Highly effective; problems eliminated.
Longevity	Years, if dredged deeply and/or sediment income controlled.
Negative features	Temporary turbidity and nutrient release.
	Improper disposal design may lead to release of toxics, or discharge of turbid water with high turbidity, nutrient content, and oxygen demand.
Costs	High (\$0.24 to \$14.00 m^{-3} for uncontaminated sediments).
Applicability to reservoirs	Highly applicable.

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PART VII: HYPOLIMNETIC AERATION

Problem Addressed

Hypolimnetic aeration has been thoroughly reviewed (Fast 1975; Fast and Lorenzen 1976; Pastorok, Ginn, and Lorenzen 1980, 1981; Cooke et al. 1986; McQueen and Lean 1986), although the technique itself has not yet been widely used. Readers contemplating its use are urged to consult these reports directly.

The purpose of hypolimnetic aeration is to increase the dissolved oxygen concentration in the hypolimnion without causing thermal destratification. If successful, several water quality problems related to reservoir eutrophication may be solved. Hypolimnetic dissolved oxygen depletion causes the following reservoir problems: (a) manganese, iron, hydrogen sulfide, methane, ammonia, and total phosphorus increase sharply in anaerobic hypolimnia, producing problems with potable water treatment (increased concentrations of Mn, Fe, and H_2S) or perhaps triggering algal blooms and (b) low dissolved oxygen concentrations will exclude fish, including coldwater species such as salmon and trout, and eliminate the hypolimnion as a refuge from fish predation for algae-grazing zooplankton. This latter problem may mean little grazing control of algae. An additional problem is that some reservoirs discharge oxygen-free hypolimnetic waters, causing environmental impacts downstream.

Theory and Design

Lorenzen and Fast (1977) provide a summary of the basic types of hypolimnetic aeration systems, with an analysis of their performances, where available. They group all designs into three types: mechanical agitation, air injection, and oxygen injection.

The mechanical agitation system, while inefficient, has been used successfully. Water is pumped from the hypolimnion to a splash basin on shore, where it is aerated and then returned to the hypolimnion. The air injection systems are divided into two types, the partial air lift and the full air lift designs. In the partial air lift system, exemplified by Atlas Copco's LIMNOS unit, compressed air is injected at the bottom of the unit, but the air-water mixture does not rise to the surface. Instead, the air and water are separated at depth so that the air is vented to the surface and the aerated water is returned to the hypolimnion. This type of design is described by Bjork (1974) and Fast (1975) and is illustrated in Figure 11. In the full air lift system, compressed air is injected at the bottom of the unit, as in the partial air lift, but the air-water mixture rises to the reservoir surface where the air is released to the atmosphere and the water is returned to the hypolimnion. This design is described by Fast (1971) and Bernhardt (1974) and is illustrated in Figure 12. Fast, Dorr, and Rosen (1975) describe the liquid oxygen system, in which water is withdrawn from the hypolimnion. After passing through the shore-based pump on its way back to the hypolimnion, pure oxygen is injected within the piping system. This design is called "side stream pumping." Another type of oxygen injection involves pumping gaseous oxygen into the hypolimnion through diffusers. The system at Richard B. Russell Reservoir, Georgia, is an example of this type.

Prior to the selection of an aeration system, certain data must be obtained (Fast, Lorenzen, and Glenn 1976). These include (a) determination of hypolimnetic volume to be aerated, (b) determination of the hypolimnetic oxygen depletion rate, (c) determination of the amount of oxygen needed, and (d) selection of the appropriate system. This last step will also involve an estimate of costs. Fast, Lorenzen, and Glenn (1976), Lorenzen and Fast (1977), and Ashley (1985) give detailed guidance in these determinations. A brief summary is presented here.

The hypolimnetic volume is computed from the reservoir's hypograph (capacity curve) and vertical temperature profiles. Fast, Lorenzen, and Glenn (1976) point out that the hypolimnion may enlarge as much as 50 percent during aeration due to mixing and subsequent thermocline erosion. Also, in reservoirs with significant water withdrawal, the volume of the hypolimnion may change.

Cooke et al. (1986) describe the calculation of the hypolimnetic oxygen deficit. (It should be noted that t_s, the duration of stratification, should be omitted from their formula.*) McQueen, Rao, and Lean (1984) caution that aeration may induce oxygen demand, and thus undersizing of the aerator must be avoided. This means that an allowance must be made for this, or the

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* Personal Communication, 1988, R. W. Kortmann, Ecosystem Consulting Services, Inc., Coventry, CT.

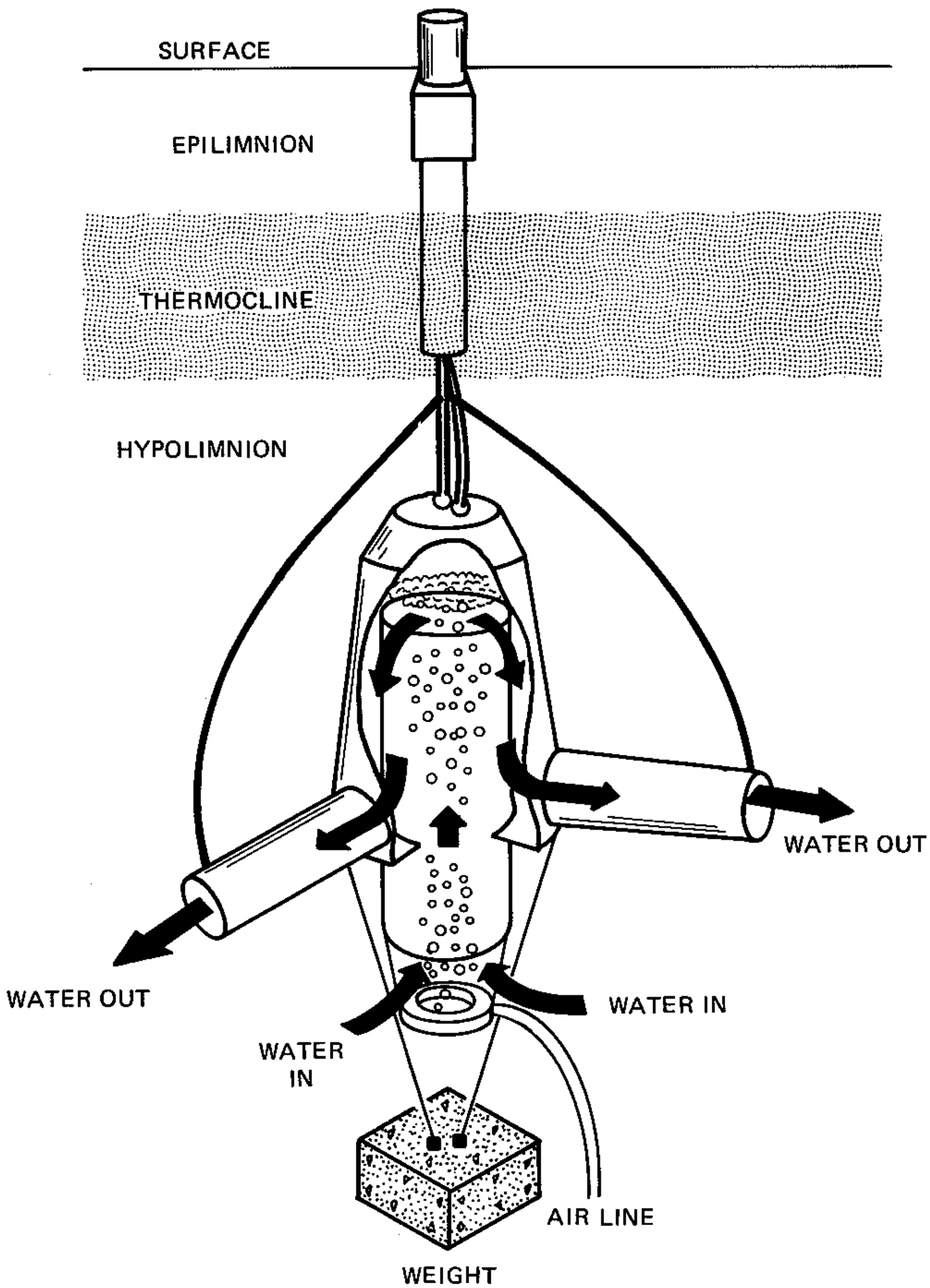


Figure 11. Example of a partial air lift hypolimnetic aeration system (after Fast 1975)

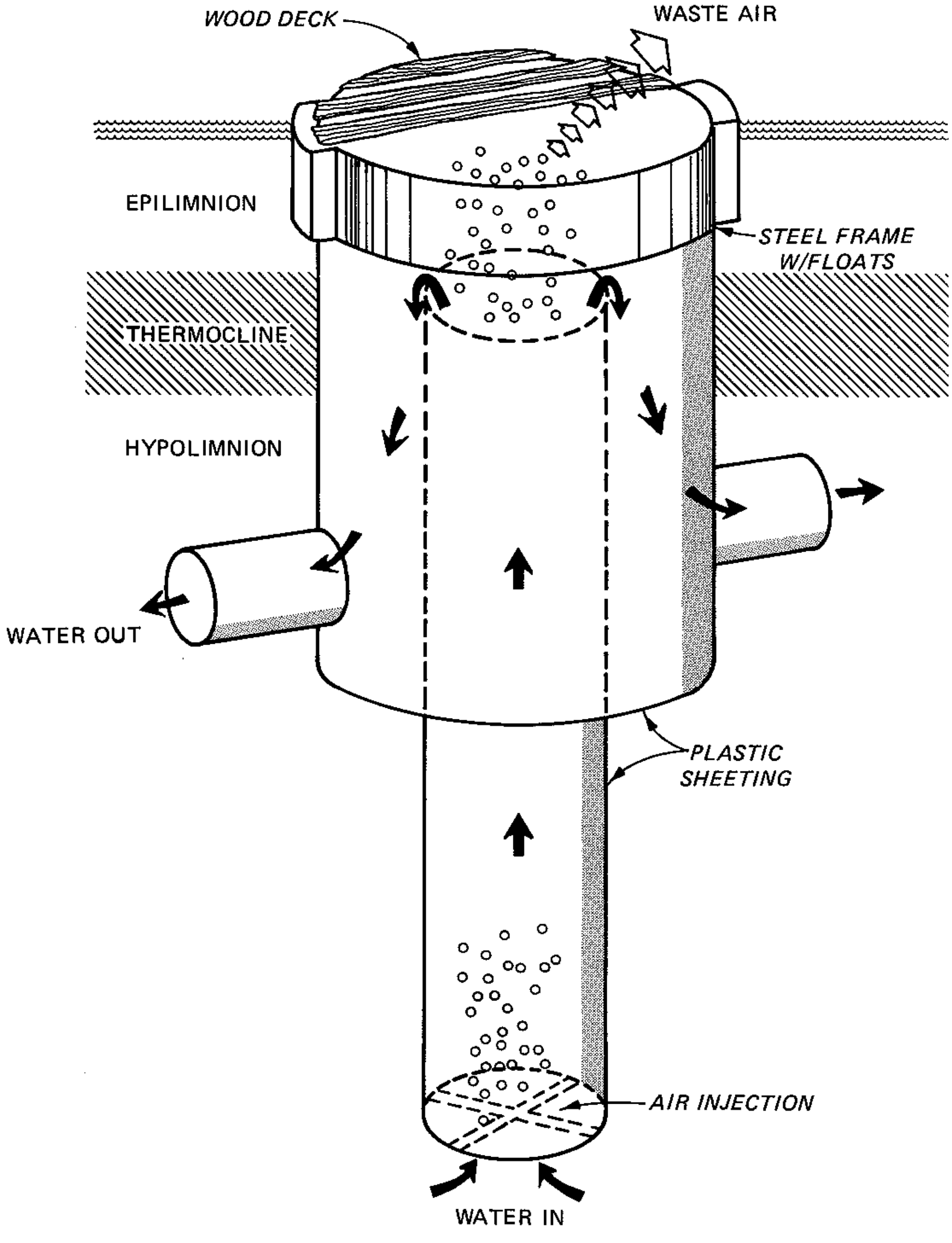


Figure 12. Example of a full air lift hypolimnetic aeration system (after Fast 1971)

system may be inadequate. Cooke et al. (1986) also provide an equation to develop an estimate of the amount of compressed air needed. Ashley (1985) and Ashley, Hay, and Schoeten (1987) present an empirical method for determining the size of a system, a test of the method, and a useful discussion of equipment and performance specifications. The period of time that the aerator must work is based upon knowledge of the thermal history of the reservoir. If there is an oxygen depletion during winter ice cover, aeration may also be required then.

The target dissolved oxygen level to be achieved may vary with the desired uses of the hypolimnetic water. Coldwater fish will require at least 5 mg O_2 ℓ^{-I} (Doudoroff and Shumway 1967), and Steinbert and Arzet 91984) found that high total phosphorus releases occurred at all dissolved oxygen levels below ciency. Bernhardt (1974) reported that up to 50 percent of the injected air was absorbed 5 mg O_2 ℓ^{-1}. This concentration of dissolved oxygen may therefore be a minimum target level.

The most commonly chosen design is the partial air lift, in part because it is commercially available. Lorenzen and Fast (1977) caution that, while this system can be effective, there is evidence of the production of N_2 gas supersaturation in the hypolimnion. This could cause fish death in the reservoir or in downstream receiving water, although there is little documentation at present of this assertion. Lorenzen and Fast (1977) also note that the partial air lift system is less efficient in oxygenating the water because the system aerates less water volume. The full air lift design has a higher efficiency. Bernhardt (1974) reported that up to 50 percent of the injected air was absorbed.

Effectiveness, Costs, and Feasibility

Pastorok, Lorenzen, and Ginn (1982) and McQueen and Lean (1986) have summarized the results of the few published case histories of this technique. All but one recorded that hypolimnetic dissolved oxygen greatly increased; constituents such as hydrogen sulfide, methane, manganese, and ammonia decreased; and hypolimnetic temperature remained low. Despite an increase in dissolved oxygen and a theoretical concomitant change from reducing to oxidizing conditions at the sediment-water interface, the reduction in total phosphorus concentration is often less than expected. In part this appears to have been due to continued external loading. However, Lean, McQueen, and Story (1986), McQueen and Lean (1986), and McQueen, Lean, and Charlton (1986) report evidence that failures to lower phosphorus concentration through hypolimnetic aeration are probably due to low iron availability, perhaps through the production of iron sulfides. They conclude that when available iron is lacking in the hypolimnion, it must be added through the aerator. In experimental enclosures, a ratio of 10:1 (total iron to soluble reactive phosphorus) and a pH <7.5 was needed to produce phosphorus sedimentation.

A metalimnetic dissolved oxygen minimum was produced in several lakes, apparently due to their high content of dissolved and particulate organic matter which settled to the metalimnion. This response is significant because an oxygen depletion in this layer may provide a barrier to the migration of fish and zooplankton, eliminating the hypolimnion as a coldwater fishery and as a refuge for phytoplankton-grazing zooplankton from daytime predation by fish. As Taggart (1984) points out, an increase in aerator capacity may not solve this problem, and it could bring about destratification.

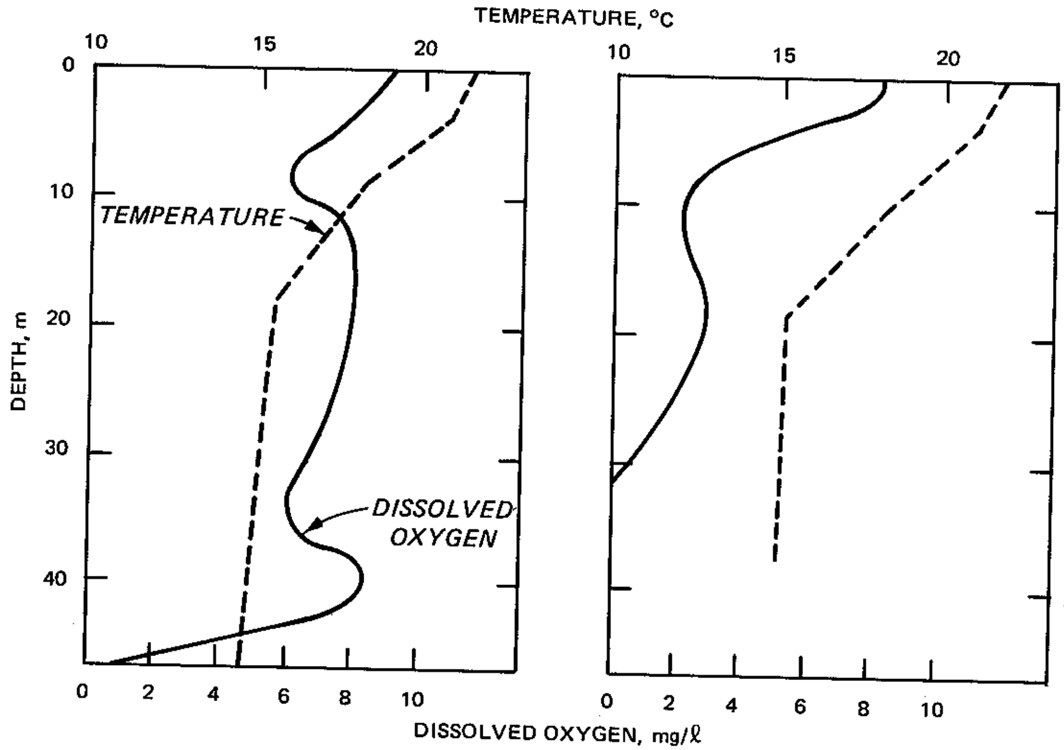
Hypolimnetic aeration appears to have little impact on summer plankton abundance, although the number of documented case histories is very small. It had been predicted that aeration would produce hypolimnetic phosphorus precipitation and thus could limit summer algal abundance in those lakes or reservoirs in which hypolimnetic waters are introduced to the epilimnion through events such as downward-upward thermocline migrations caused by in-reservoir water movements or by storms. As described above, phosphorus concentration in the hypolimnion has not decreased in some cases. An increase in zooplankton, perhaps caused by their escape from fish predation through daytime migration to the hypolimnion, was reported by Fast (1971). Other authors report little or no change in zooplankton density. Further documentation of the responses of plankton is needed.

The response of fish to hypolimnetic aeration is encouraging, although documentation is scarce. Pastorok, Lorenzen, and Ginn (1982) describe three lakes in which trout populations have survived, in part due to the increase in benthic invertebrates.

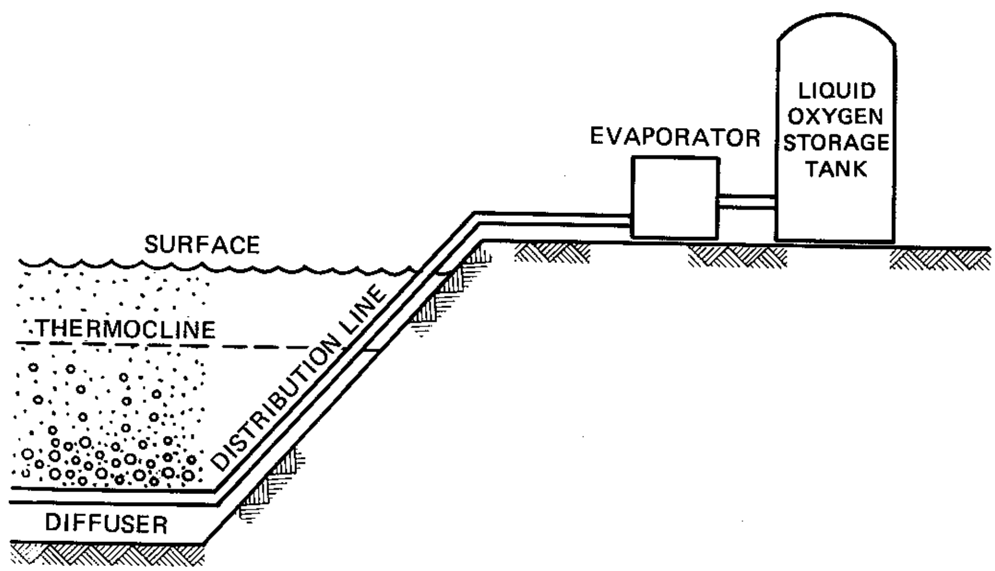
There is an application of hypolimnetic aeration of particular interest to reservoir managers at sites where hypolimnetic water is discharged. If this water is low in dissolved oxygen, significant impact to downstream biota will occur. Pastorok, Lorenzen, and Ginn (1982) discuss the oxygenation of the hypolimnion at Clarks Hill Reservoir, South Carolina. Speece and co-workers have designed an unconfined bubble plume in which the bubbles are released from diffusers at a size that allows for maximum absorption efficiency and minimal risk of destratification (Speece, Rayyan, and Murfee 1973; Speece et al. 1976). The tests of this system at Clarks Hill were successful in increasing the dissolved oxygen of the hypolimnion and the discharge. Location and size of the oxygen diffusers, as well as bubble diameter, appear to be critical factors. The Richard B. Russell Reservoir, Georgia and South Carolina, is another example of this problem of low dissolved oxygen in release waters. It was apparently solved through the use of a deep-water oxygen injection system at a site near the dam (Figure 13). Prior to installation of the system, dissolved oxygen in hypolimnetic waters at the dam fell to zero in May, and a layer of hypolimnetic water with dissolved oxygen less than 2 mg O_2 ℓ^{-1} extended a considerable distance up the reservoir. A pulsed operation and a continuous operation of the oxygen injection system were evaluated. The continuous operation appeared to be more effective because it distributed dissolved oxygen more evenly upstream and thus allowed a much smaller anoxic zone to exist. It was concluded that the oxygen injection system must be operated continuously throughout the stratified period to maintain acceptable dissolved oxygen levels in release waters (James et al. 1985).

Hypolimnetic aeration is a highly feasible procedure for alleviating problems caused by high iron and manganese or low dissolved oxygen in reservoir releases. There is evidence of its usefulness in fish management as a means of developing a "two-story" fishery. There is little evidence, at present, that it is a procedure of value in controlling algal blooms.

Fast, Lorenzen, and Glenn (1976) and Lorenzen and Fast (1977) discuss costs of hypolimnetic aeration. The capital cost items are the air lift devices, the compressor, the air supply lines, and the diffusers. These costs are dependent upon aerator size and thus upon hypolimnetic oxygen demand (McQueen and Lean 1986). Fast, Lorenzen, and Glenn (1976) used San Vicente Reservoir, California, to compare costs of side stream pumping (SSP), partial air lift, and two full air lift designs. This is a large, deep reservoir, containing 111×10^6 m^3, with a maximum depth of 58 m and a maximum oxygen input requirement of 925,344 kg O_2/10 months. Table 6, modified from Fast, Lorenzen, and Glenn (1976), is the cost comparison corrected to 1985 dollars. In their opinion, the full air lift system is the least costly to operate and



a. Example profiles of dissolved oxygen and temperature with (left) and without (right) oxygenation



b. Generalized diagram

Figure 13. Oxygenation system at Richard B. Russell Lake

Table 6

Comparison of Costs for Hypolimnetic Aeration of San Vincente Reservoir, California (1985 dollars) (Modified from Fast, Lorenzen, and Glenn 1976)

Item	SSP	LIMNO	Bernhardt	Fast
Capital costs	\$307,100	\$555,000	\$462,500–\$518,000	\$222,000–\$277,500
Estimated yearly operating costs	\$148,000	\$93,148	\$39,701	\$39,701
Estimated horsepower requirements (total)	150	500	200	200
Estimated efficiency (kg O_2 dissolved kilowatt-hour^{-1})	0.635	0.544	1.089	1.089

the most efficient. If the assertion about problems with N_2 gas supersaturation with the partial air lift system is correct, then the full air lift system seems to be the best choice, based on cost, efficiency, and environmental impacts.

Limitations and Concerns

Lorenzen and Fast (1977), Pastorok, Lorenzen, and Ginn (1982), and McQueen and Lean (1986) describe several problems that could occur with hypolimnetic aeration. One of the most significant is unintentional destratification, particularly in shallow reservoirs, leading to an introduction of nutrient-rich water to the upper waters of the reservoir. If the aerators are undersized or the oxygen demand is greater than anticipated, hypolimnetic dissolved oxygen may not increase at all. This could also occur if there is erosion of the thermocline, making the hypolimnetic volume larger than determined initially with temperature studies. Or, as suggested from the use of aerated and control enclosures by McQueen, Rao, and Lean (1984), aeration may induce oxygen consumption, thus increasing the demands on the system, and may even cause high wintertime dissolved oxygen demand leading to a winter fish kill. Causal factors have not been isolated, but these data suggest that an oversized system should be used. Also, some lakes have developed a metalimnetic oxygen minimum, thus isolating the hypolimnion from fish and zooplankton.

There have been concerns, particularly with the partial air lift design, regarding N_2 gas supersaturation. At present there are few data regarding this problem and any related toxicity with either in-reservoir or tailwater fish. In some cases algal blooms have occurred following the use of hypolimnetic aeration, apparently caused by increased internal phosphorus loading (e.g., Steinberg and Arzet 1984).

Summary

The purpose of hypolimnetic aeration is to increase the dissolved oxygen concentration of the hypolimnion. This procedure may improve tailwater quality, provide a habitat suitable for a "two-story" fishery, reduce internal phosphorus loading, eliminate problems with iron and manganese in potable water treatment plants, and possibly provide some control of phytoplankton biomass through enhanced zooplankton grazing or decreased phosphorus concentration. Hypolimnetic aerators are of these design types: mechanical, oxygen injection, and air injection. The air injection design can be either the partial or the full air lift. Selection of the proper system involves acquisition of basic data, including hypolimnetic volume and oxygen demand. The full air lift system appears to be least costly and most efficient, with fewer negative environmental impacts. Problems with hypolimnetic aerators include undersizing, unintentional destratification, and creation of a metalimnetic oxygen minimum. The procedure is particularly useful in fish management and in improvement of release water quality.

A summary of hypolimnetic aeration is given in Table 7.

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Table 7

Summary of Hypolimnetic Aeration

Characteristic	Description
Target	Anoxic hypolimnion.
Mode of action	Water is aerated through mechanical agitation or through air or oxygen injection.
Effectiveness	Highly effective in increasing dissolved oxygen. Iron addition may be required to precipitate phosphorus.
Longevity	Hypolimnion remains aerobic as long as aerators operate.
Negative features	Unintentional destratification. Supersaturation with N_2 gas. Creation of metalimnetic oxygen minimum.
Costs	Dependent upon size and type selected. Full air lift appears to be most economical.
Applicability to reservoirs	Can be very effective in fish management or in improving tailwater quality.

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PART VIII: ARTIFICIAL CIRCULATION

Problem Addressed

Eutrophic conditions in a reservoir will produce several symptoms that may be alleviated either by preventing thermal stratification from occurring in the spring or by destratifying the reservoir. Thermal stratification isolates the hypolimnion from gas exchange with the atmosphere. These organically rich deep waters and sediments support respiration by microbes, leading to a loss and sometimes complete depletion of dissolved oxygen in the hypolimnion. Under anaerobic conditions, fish cannot survive, and levels of dissolved oxygen below 5 mg O_2 ℓ^{-1} will probably not support coldwater game species. As well, discharges of this cold, low-oxygen water to downstream communities will have significant negative impacts on the biota of the stream. Under anoxic conditions, iron, manganese, and other trace elements can be released from the sediments and accumulate in the hypolimnion, and can produce problems of taste and color in potable water. Anoxic hypolimnetic waters become nutrient-rich and are a potential nutrient subsidy to epilimnetic algae when these waters are introduced to upper, lighted, warmer strata during and following certain partial mixing events.

Thermally stratified reservoirs also present ideal conditions for the development of blue-green algal blooms. In addition to receiving nutrient subsidies via internal loading from the hypolimnion, epilimnetic waters have other factors that favor algal blooms, including high pH and the establishment of calm, hot periods when blue-green algae may form surface scums. Also, the epilimnion may be rich in fish species such as gizzard shad, which can eliminate the zooplankton that graze most heavily on planktonic algae. The usual daytime descent to dark hypolimnetic waters, which can serve as a refuge from sight-feeding fish for the zooplankton, is eliminated by the absence of dissolved oxygen.

Artificial circulation is a procedure to destratify a reservoir to create isothermal and isochemical conditions. Detailed reviews of this method have been provided by Lorenzen and Fast (1977); Pastorok, Ginn, and Lorenzen (1981a,b); Pastorok, Lorenzen, and Ginn (1982), and Cooke et al. (1986).

Theory and Design

Reservoir circulation devices have been described by Lorenzen and Fast (1977), Davis (1980), Pastorok, Lorenzen, and Ginn (1982), Johnson (1984), and Cooke et al. (1986), among others. The devices are classified as air lift pumps, mechanical pumps, and water jet systems. Air lift, or pneumatic diffusers, employ air injection into a pipe laid along the sediment surface. Mechanical destratification systems use pumps or fan blades to move water and are apparently more costly and less efficient than the air lift (Pastorok, Lorenzen, and Ginn 1982). Hydraulic destratification uses a pump-intake-diffuser system to create high-momentum, buoyant jets that induce a great deal of mixing. Water is withdrawn from the epilimnion and pumped through a diffuser in the hypolimnion. Hydraulic destratification is believed to be more efficient than pneumatic destratification, although capital costs may be more (Dortch 1979). Design criteria for hydraulic destratification have been developed by Holland and Dortch (1984). The primary means of artificial circulation at the present time, however, is through the use of the pneumatic diffuser.

The goal of artificial circulation is to develop isothermal conditions. Normally, air lift systems are used to produce a curtain of bubbles with sufficient energy to rapidly move large masses of water. A perforated pipe terminates in the deepest reservoir section, tethered to anchors just above the sediments so that the bubble curtain does not disturb the sediments and increase reservoir turbidity. Rapid mixing will achieve a temperature difference of less than 2° to 3° C from top to bottom. This is critical to success. Approximately 30 standard cubic feet per minute (SCFM) of air per 10^6 ft^2 of reservoir surface area, or 9.2 m^3 min^{-1} km^{-2}, should provide adequate mixing to achieve isothermal conditions and oxygenation of the water column through contact with the atmosphere (Lorenzen and Fast 1977). As discussed later, more than half of the case histories of artificial circulation report a failure to achieve this rate, and many of these projects also failed to obtain expected results. Lorenzen and Fast (1977), Davis (1980), and Cooke et al. (1986) discuss, with examples, the calculation of air flow rate, compressor size, etc., for lakes and reservoirs of various depths and areas. In large reservoirs, destratification may be practical only in limited areas (Pastorok, Ginn and Lorenzen 1981a).

The effect of introducing dissolved oxygen to anaerobic bottom waters will be to increase the oxidation-reduction potential at the sediment-water interface, and this may decrease the release of phosphorus and trace metals such as iron and manganese. Also, mixing will permit the escape of gases such as CO_2 and H_2S and may promote the conversion of ammonia to nitrate.

Improvements in water clarity and algal biomass are expected, although in many cases the expected response does not occur. An understanding of the reasons for the various responses of algae to artificial circulation is not complete, but the expectation of improved conditions is based upon physical, chemical, and biological changes which can occur. These are briefly reviewed below.

Some phytoplankton communities are light-limited. In reservoirs of sufficient depth, where light attenuation is possible, an increase in the mixing depth of algal cells through artificial circulation will subject these cells to increased periods when light is insufficient to support net photosynthesis, leading to a decline in algal biomass. Several models that could be used to evaluate this response to artificial circulation are available. The model of Lorenzen and Mitchell (1975) relates maximum standing crop of algae to mixed depth, using light-limiting conditions only, but nutrients may also exert an effect. Forsberg and Shapiro (1980) and Shapiro et al. (1982) have developed an expanded model to include both light and nutrient limitation. Detailed reviews of these models are given in Lorenzen and Fast (1977), Pastorok, Lorenzen, and Ginn (1982), and Cooke et al. (1986).

Artificial circulation may induce control of algal biomass through a reduction in internal loading. In stratified reservoirs, phosphorus released from anaerobic sediments can be transported to the epilimnion through various vertical entrainment processes and thus subsidize phosphorus-limited cells. Aeration of the sediment-water interface in those reservoirs where iron controls phosphorus solubility should produce sorption of phosphorus by the sediments. Aerobic phosphorus release, however, may be high, and the turbulence induced by artificial circulation can disturb nutrient-rich sediments.

A shift in the dominance of the phytoplankton community from blue-green algae to greens and diatoms is also expected. This is usually considered to be beneficial to water quality. An explanation for this expected shift has been attributed to changes in carbon dioxide concentration and pH, and to cell sinking rates. Blue-green algal cultures, as well as algal communities in in situ enclosures, have been shown to shift to dominance by the less obnoxious green algae in response to decreased pH and increased carbon dioxide (Shapiro 1973, 1984; Shapiro, LaMarra, and Lynch 1975; Shapiro et al. 1982). These chemical changes would be expected following the start-up of a circulator and the introduction of CO_2-rich hypolimnetic waters to the remainder of the reservoir, bringing about a pH decrease. Also, increased contact of the water column with the atmosphere may increase the CO_2 content of the water. Shapiro et al. (1982) have suggested that the pH decrease enhances cyanophage activity. A significant pH decrease may not occur in well-buffered waters. Blue-green dominance in summer months may also be due to their pseudovacuoles, which provide buoyancy and thus allow these cells to dominate the lighted surface layer as "scums." Other algal groups have higher sinking rates and thus cannot compete with scum-forming blue-green algae. Artificial circulation, if powered sufficiently to prevent surface thermal microstratification during hot periods, may eliminate this competitive advantage of blue-green algae over more desirable algae such as diatoms (Pastorok, Lorenzen, and Ginn 1982).

A decline in algal biomass may be expected not only from light limitation or control of phosphorus release, but from increased zooplankton abundance and decreased zooplanktivory by fish. Elimination of the oxygen-free hypolimnion could provide a dark refuge from sight-feeding fish, thus enhancing zooplankton survival. Zooplankton grazing on algae, particularly the more edible green algae which become abundant following the pH shift, may reduce algal biomass. At present there is little evidence to support this scenario of artificial circulation and grazer control of algae.

Effectiveness, Costs, and Feasibility

Table 8, from Pastorok and Grieb (1984), is a summary of the case histories of diffused air systems. For some variables, the number of observations is small, but it is clear that, in most cases, artificial circulation with diffused air has brought about nearly isothermal conditions, decreases in Secchi disc transparency, reduced concentrations of ammonia, iron, and manganese, and an increase in dissolved oxygen. Decreases in pH, algal density,

Table 8

Responses to Artificial Circulation (Diffused Air Systems Only)

(from Pastorok and Grieb 1984)

Parameter	N		Lake Responses* +	−	0	?	Chi-square
ΔT after circulation**	45	No.	15	30	-	-	5.00†
		%	33	67	-	-	
Secchi depth	19	No.	4	10	2	3	6.50†
		%	21	53	10	16	
Dissolved oxygen	41	No.	33	1	2	5	55.2††
		%	80	2	5	12	
Total phosphorus	20	No.	5	5	8	1	0.74
		%	25	30	40	5	
Ammonia	20	No.	3	13	3	1	10.5†
		%	15	65	15	5	
Iron and manganese	22	No.	0	20	2	0	33.1††
		%	0	91	9	0	
Epilimnetic pH	21	No.	1	9	8	3	6.33†
		%	5	43	38	14	
Algal density	33	No.	6	14	8	5	3.71
		%	18	42	24	15	
Blue-green algae	25	No.	5	13	5	2	5.57
		%	20	52	20	8	
Ratio green:blue-green algae	21	No.	11	3	6	1	4.90
		%	52	14	29	5	

* Direction of change (+ = increase, − = decrease, 0 = no significant change, ? = questionable response) in the average value for whole water column.

** Temperature differential between surface and bottom water (+ means ΔT greater than 3° C, − means ΔT less than or equal to 3° C).

† $P < 0.05$; goodness of fit to uniform frequency distribution for +, −, and 0 responses only.

†† $P < 0.001$.

and blue-green algae and increases in the ratio of green to blue-green algae also occurred in some cases, but these were not statistically significant. Where circulation produced isothermy, there was a decrease in algal blooms in 13 of 23 lakes (57 percent), whereas in situations of incomplete mixing, algal density usually stayed the same or increased. When there was a decrease in pH, a shift in dominance from blue-greens to green algae usually occurred (Pastorok, Ginn, and Lorenzen 1981a; Pastorok, Lorenzen, and Ginn 1982), supporting Shapiro's hypothesis. The desired pH shift is more likely to occur with rapid mixing (Shapiro et al. 1982).

A decrease in Secchi disc transparency occurred under these three circumstances: (a) algae of the photic zone were nutrient-limited and an introduction of hypolimnetic water stimulated an algal bloom, (b) isothermy did not occur, allowing for microstratification at the surface and the formation of blue-green scums, and (c) the intensity of circulation was sufficient to disturb bottom sediments (Pastorok, Lorenzen, and Ginn 1982). Brosnan and Cooke (1987), for example, found that algal blooms and decreased transparency occurred during brief, very warm periods when a low-powered circulation device could not maintain isothermy and therefore allowed surface algal blooms to develop.

The effect of artificial circulation on total phosphorus concentrations has not been impressive (Table 8). In most cases there has either been an increase or no change. In some cases, as pointed out by Cooke et al. (1986), increases in total phosphorus during artificial circulation may be due to aerobic phosphorus release from sediments, which is enhanced by the increase in water temperature and possibly water movement over the sediment surface.

The predicted increase in zooplankton has occurred in a majority of cases. Depth distribution of animals has also increased. Some authors have reported an increase in *Daphnia*, suggesting that predation pressure on these large-bodied phytoplankton grazers may have lessened or their habitat has increased. There is little long-term documentation of species shifts that can be related to circulation or to changes in phytoplankton biomass.

Artificial circulation also brings about an increase in colonizable habitat for benthic invertebrates and fish. As with phytoplankton and zooplankton, long-term evaluations of the effects of artificial destratification on fish or benthic fauna have not been made (Pastorok, Lorenzen, and Ginn 1982).

Artificial circulation is a feasible alternative for reservoir managers, especially for improvement of dissolved oxygen problems associated with reservoir discharge or with potable water supply. As Johnson (1984) has pointed out, in many cases it is more economical to maintain good reservoir water quality through aeration-circulation than to treat water as it is released. In nearly every case history of artificial circulation, dissolved oxygen in the reservoir has improved, and iron and manganese concentrations have declined sharply. This procedure may also be useful in fish management, although it may be detrimental to coldwater game species. Circulation is less likely to produce relief from low transparency and algal blooms, especially in waters that are nutrient-limited and may thus be stimulated by an increase in phosphorus or other nutrient.

The costs of artificial circulation are not comparatively high. Major capital costs will be for air compressors, air supply pipes, and diffusers. Lorenzen and Fast (1977) estimate a cost of $113,000 (1985 dollars) for two complete air compressors, motors, supply lines, and diffusers to deliver 1,200 SCFM. Power costs would be additional.

Limitations and Concerns

Several impacts of concern could occur. If algal productivity is nutrient-limited and circulation increases the supply of a limiting nutrient, then circulation may stimulate an increase in algal biomass (Pastorok, Lorenzen, and Ginn 1982). The oxygen demand by algal blooms, suspended detritus, and bacteria could produce an oxygen depletion, although this should not occur if circulation produces isothermy. Failure to achieve sufficient mixing appears to be one of the chief causes of negative responses to artificial circulation. Undersizing of the system must be avoided. Water temperature will increase following circulation, perhaps as much 15° to 20° C in deep water, and this could have detrimental effects on coldwater fish species in the reservoir or in downstream receiving waters. Finally, for pneumatic destratification, the danger of supersaturation of N_2 gas relative to surface hydrostatic pressure could occur, producing fish mortalities in the reservoir and downstream. This problem is discussed by Fast and Hulquist (1982). It would not be encountered with hydraulic destratification. Pastorok, Lorenzen, and Ginn (1982) have provided a thorough review of the potential negative effects of artificial circulation.

Summary

Artificial circulation is a procedure to destratify or to prevent thermal stratification of reservoirs through the mixing action of compressed air or pumped water. The most common and apparently most cost-effective method for most applications is the diffused air system. This technique has been very successful in increasing dissolved oxygen and decreasing the concentrations of iron and manganese, making it a highly useful procedure for water supply reservoirs and for some situations where there are problems with discharge of water low in dissolved oxygen. In some cases there has been significant improvement in problems with phytoplankton, especially where destratification has been complete and where rapid mixing has introduced enough carbon dioxide to the upper waters to produce a pH decrease. A significant pH change may not occur in well-buffered waters. Deep mixing, in cases where phytoplankton are light-limited, may contribute to algal control by decreasing the length of periods when net photosynthesis is possible. Algal blooms have occurred in cases of incomplete mixing or where nutrient-limited cells are stimulated by circulation-induced nutrient increases. There are several negative effects of artificial circulation, including algal blooms and impacts on fish in the reservoir and downstream communities. Many problems are caused by undersizing the air lift system and a consequent failure to achieve isothermy.

Table 9 is a summary of artificial circulation.

Table 9

Summary of Artificial Circulation

Characteristic	Description
Targets	Low hypolimnetic dissolved oxygen. High hypolimnetic iron and manganese. Algal blooms.
Mode of action	Bubble column or pumped water increases reservoir mixing and allows aeration of water through surface exchange.
Effectiveness	Highly effective in increasing dissolved oxygen and decreasing iron and manganese. Other effects are less predictable.
Longevity	Improvements continue as long as the circulation device is operated. Long-term effects are unknown.
Negative features	Inadequate circulation may produce thermal microstratification, algal blooms, and continued dissolved oxygen problems. Problems for fish from N_2 gas supersaturation have been suggested.
Costs	Comparatively low.
Applicability to reservoirs	Highly applicable for dissolved oxygen, iron, and manganese problems.

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PART IX: WATER LEVEL DRAWDOWN

Problem Addressed

Water level drawdown is a multipurpose reservoir improvement technique. It is used to control some nuisance plants, to provide access to dams, docks, and shorelines for repair and installation purposes, for fish management, for sediment consolidation and removal, and for installation of sediment covers. Reviews of this procedure are found in Cooke (1980), Culver, Triplett, and Waterfield (1980), Triplett, Culver, and Waterfield (1980), Ploskey (1982), and Cooke et al. (1986).

Theory and Design

Water level drawdown for control of nuisance macrophytes has been used successfully against susceptible species in certain climates of the United States. The objective is to expose the plant to freezing-desiccation or heat-desiccation for a period sufficient to destroy the thallus, roots, and rhizomes, and perhaps some reproductive structures.

Water may be withdrawn for several other simultaneous purposes, including access to structures for repairs and installations or for sediment consolidation through drying or sediment removal with dredges or earth-moving equipment. As described in detail in Part XIII, exposed sediments may also be conveniently covered with screens to eliminate rooted plants.

Ploskey (1983) and Ploskey, Aggus, and Nestler (1984) provide detailed reviews of water level changes and their use in fish management. Actions to benefit fisheries can also produce improvement in the trophic state of eutrophic reservoirs, except in instances when the management of the fish community includes stocking of zooplanktivorous fish, such as gizzard shad, for game fish forage. As described in Part XI, elimination of algae-grazing zooplankton may result from this practice, and the reservoir may experience continued algal blooms. Readers interested in the use of water level manipulations for fisheries should consult the above reports.

Prior to the use of a water level drawdown to control nuisance aquatic plants, a survey of the kinds of plants in the reservoir is necessary because this procedure is species specific. Some plants are eliminated, some unaffected, and others may flourish when water levels are restored. The survey will also identify the area of the plant infestation, thus giving the depth to which water must be withdrawn to achieve partial or complete exposure of macrophytes. Procedures for conducting the plant survey are outlined in Forsberg (1959) and Nichols (1982).

As illustrated in the case histories, most drawdowns for the control of macrophytes in northern states have taken place during winter months. Many plants (Table 10; refer to Cooke 1980 and Cooke et al. 1986 for list of responses of 74 plants species) are susceptible to prolonged periods (3 to 4 weeks) of freezing and dewatering. Soils that remain moist or that are not frozen will protect roots and rhizomes, and regrowth will occur.

Water level drawdown will provide access to areas in need of maintenance and repair. Shoreline erosion, a significant source of turbidity in some reservoirs, is a problem that can be treated during water withdrawal. Shoreline erosion may be caused by one or several of these factors: waves, abrupt water level fluctuations, erodible and bare soils, ground-water seepage, bluff slumping, and surface runoff erosion. In one of the few studies of its kind, Wilson (1979) found that 82 percent of the total solids income to a small Ohio reservoir was from shoreline erosion. Shoreline stabilization may be brought about through construction of protective structures, planting of vegetation, development of drainage controls from the land, and by altering bluff slopes. Details of these procedures are available through the Soil Conservation Service (SCS) and in several reports (US Army Corps Engineers 1973; Clemens, undated).

Drawdown also provides access for the installation of sediment covers in areas such as beaches and docks. Normally, these materials are applied during summer months using SCUBA (see Part XII). A less expensive and more effective method is to fasten them to frozen reservoir soils.

With regard to eutrophic reservoirs and their improvement or restoration, water level drawdown is used primarily as a procedure to control rooted macrophytes. Further discussion of this procedure will be confined to this purpose.

Table 10

A Summary of the Response of Some Common Nuisance Macrophytes to Drawdown (Modified from Cooke et al. 1986)

A. Species That Usually Increase
   1. *Alternanthera philoxeroides* (alligatorweed)
   2. *Hydrilla verticillata* (hydrilla)
   3. *Najas flexilis* (bushy pond weed)

B. Species That Usually Decrease
   1. *Ceratophyllum demersum* (coontail)
   2. *Elodea (=Egeria) densa* (Brazilian elodea)
   3. *Myriophyllum* spp. (milfoil)
   4. *Najas guadalupensis* (southern naiad)

C. Species That Are Unaffected or Whose Response is Variable
   1. *Eichhornia crassipes* (waterhyacinth)
   2. *Elodea canadensis* (elodea)

Cooke et al. (1986) describe and review case histories for several climatic regions of the United States and provide a detailed summary of the published responses of 74 species of aquatic plants to summer, winter, and annual exposure to water level drawdown. Table 10 is a summary of the responses of some of the most common plants. Some of these case histories are summarized herein to indicate the effectiveness of this method.

Eurasian watermilfoil (*Myriophyllum spicatum*) has become a nuisance in Tennessee Valley Authority reservoirs, particularly those with small (0.2 to 1.0 m) annual water level fluctuations. The herbicide 2,4-D is effective against this plant in cove areas, but winter water level drawdown has been found to be the most effective means of control. The normal 2-m drawdown at Watts Bar and Chickamauga Reservoirs kills all of the plants along well-drained shorelines. Because these reservoirs are multipurpose, the use of drawdowns is sometimes limited (Smith 1971). At Melton Hill Reservoir, 2,4-D and high-frequency, short-duration winter drawdowns are also used for control of *M. spicatum* (Goldsby, Bates, and Stanley 1978).

Water level manipulation was one of the primary methods of plant control in Louisiana, principally because herbicides were too costly, and harvesting tended to spread the infestation (Richardson 1975). Lantz et al. (1964) and Lantz (1974) have described the use of drawdown for plant control in several Louisiana reservoirs. A mid-summer to mid-October drawdown at Anacoco Reservoir opened it to recreation by eliminating water shield (*Brasenia schreberi*) and by controlling parrot feather (*Myriophyllum brasiliense*) and water lily (*Nuphar odorata*). However, *Chara vulgaris* (muskgrass) increased. An infestation of pondweed (*Potamogeton* sp.) and naiads (*Najas guadalupensis*) was reduced from 285 to 16 ha by a winter drawdown at Bussey Reservoir.

The winter drawdown of Lake Ocklawaha (Rodman Reservoir) in central Florida was probably a failure. Some nuisance plants were controlled, such as coontail (*Ceratophyllum demersum*) and Brazilian elodea (*Elodea densa*), but hydrilla (*Hydrilla verticillata*), waterhyacinth (*Eichhornia crassipes*), and alligatorweed (*Alternanthera philoxeroides*) were not. Hestand and Carter (1975) attribute at least some of this response to a mild winter.

Beard (1973) described the successful use of a winter drawdown of Murphy Flowage, Wisconsin. The pondweeds *Potamogeton robbinsii* and *P. amplifolius*, coontail, and milfoil were controlled, and 80 percent of the reservoir was opened to fishing.

However, Geiger (1983) found that the mild, wet winter of the Pacific Northwest (Oregon) was inappropriate for using drawdown to control milfoil, and a herbicide application was finally required to produce the desired control.

Effectiveness, Costs, and Feasibility

Alligatorweed and hydrilla are serious nuisances in some southern reservoirs, and drawdown apparently does not control them (Table 10), while milfoil, coontail, Brazilian elodea, and southern naiad are controlled. The prospective user of this procedure should be aware that responses to drawdown are species-specific and that successful control of some species may mean that resistant ones will proliferate. This problem may be solved by the use of drawdown followed by 1 to 2 years of no drawdown so that natural competitive interactions between drawdown-sensitive and insensitive species remain and the less sensitive do not become dominant. Winter drawdowns are the most successful for northern reservoirs, interfere least with other reservoir uses, and should refill promptly with spring runoff. In general, the long-term effects of drawdown on aquatic plants are poorly understood.

The feasibility of this method for a particular reservoir is dictated largely by its use. Long-term drawdown could not be used in a hydropower reservoir, for example. Many of these reservoirs, however, have few macrophyte problems due to the absence of shallow areas and to the large water level fluctuations over short periods. The level of mainstream reservoirs is dominated by riverflow and the amount of water level manipulation possible is sometimes limited (Ploskey, Aggus, and Nestler 1984). High-frequency, short-duration withdrawals, as used by Goldsby, Bates, and Stanley (1978) at Melton Hill, could be used for this type of reservoir. Flood control impoundments are good candidates for water level drawdowns, particularly during winter months.

Comparatively very low costs are associated with this procedure, and it is possible that the implementation of other techniques during the drawdown could produce some cost savings in overall reservoir management.

Limitations and Concerns

Ploskey (1983) lists several ways in which drawdown can interfere with reservoir uses, including interference with navigation, access for boaters and swimmers, fishing, and fish management. Most, if not all, of the problems are averted by winter drawdowns and spring refills. Algal blooms after reflooding were reported by Hulsey (1958) and Beard (1973), although the causes of such blooms are poorly understood. High external loading, release of nutrients from sediments, and elimination of competitive effects of higher plants with algae may all be involved. Spring and summer drawdowns can have several negative effects on fishing (Ploskey 1983). These include destruction of littoral food organisms, elimination of cover, and interference with spawning. Also, low dissolved oxygen in the remaining pool can produce a fish kill, or summer drawdowns can eliminate thermal stratification and introduce anoxic waters to the entire reservoir (Geagan 1960, Richardson 1975, Shaw 1983). One very significant problem is the failure to refill due to an unexpected drought or to poor timing of the drawdown relative to expected rainfall. Winter drawdowns for flood control projects have a low probability of refill problems due to the usually high volume of spring runoff. Summer drawdowns can remain low if autumn is dry. An absence of refill problems, minimal negative impact on reservoir users and the fishery, and the best plant control are most likely to be achieved with the use of winter drawdowns, particularly in cold climates.

Summary

Water level drawdown is an effective procedure for the control of certain species of nuisance aquatic macrophytes. Control is achieved through drying and freezing over a period of at least 3 to 4 weeks for projects located in northern areas. Somewhat longer periods may be required for southern projects. Plants that remain in moist soil or in shallow water can be expected to survive.

Drawdown can also be used to implement other procedures, including repair or installation of structures for control of shoreline erosion or for gaining access to dams, docks, and piers. Also, exposed, consolidated sediments can be more easily removed with earth-moving equipment, assuming sufficient consolidation to bear weight, than by removal during normal water level with a hydraulic dredge. Water level manipulation can also be used for fish management and to facilitate the installation of sediment covers.

Negative aspects of this method primarily involve problems of access to water by reservoir users, failure to refill, and possible effects on fisheries. Most of these are avoided at many projects by use of winter drawdowns. This procedure cannot be used at all reservoirs since only some types of operations will permit long-term winter drawdowns. Water level drawdown is summarized in Table 11.

Table 11

Summary of Water Level Drawdown

Characteristic	Description
Targets	Nuisance aquatic macrophytes. Unconsolidated sediments.
Modes of action	Desiccation and freezing of thallus, roots, rhizomes, and other reproductive structures. Sediment consolidation and oxidation.
Effectiveness	Winter drawdown highly effective against some species.
Longevity	Usually effective for at least 1 year.
Negative features	Proliferation of resistant species. Limited access to water during withdrawal. Reduced storage volume.
Costs	Minimal.
Applicability to reservoirs	Highly applicable for reservoirs where operation allows drawdown.

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PART X: HARVESTING

Problem Addressed

Shallow areas of reservoirs often have extensive growths of macrophytes ("weeds"). These plants may inhibit recreation, clog intake structures, release nutrients, and contribute to dissolved oxygen depletions when their tissues, and the dissolved organic matter they may release, are oxidized. Figure 4 illustrates the major interactions of these plants in aquatic systems.

Theory and Design

Harvesting is a procedure by which aquatic plants are cut, collected, and removed from the water. This technique can bring about some control of plant regrowth, open the infested areas to recreation, lower the amount of organic matter in the water column or deposited on sediments, and may contribute to improvement in water quality through the removal of nutrients and organic matter.

Most harvesters in use today are single-stage machines that cut the vegetation with one horizontal and two vertical sickle-blade cutter bars, store the collected plants onboard via a conveyor from the cutterhead to a storage compartment, and unload the plants at shore via additional conveyors aft of the storage compartment. Machine storage capacities vary from about 6 to 23 m^3 of cut vegetation, depending upon model and manufacturer. Cutter depth is usually limited to a maximum of 2 m, since hydraulic drag is considerable if cutting operations are carried out at greater depths. The operator controls the depth of the cutting bar. Some manufacturers also sell shore conveyor units that convey vegetation from the harvester unit onto trucks. Also available are transporter units that are loaded by the harvester at the harvesting site and transport the cut vegetation to shore. Most machines are built on pontoons and driven by diesel-powered paddle wheels.

In practice, the cutter bar should be lowered until it is just in the mud and both root crowns and stems are cut. Most operators attempt to work parallel to shore. Docks, trees, boulders, and other obstacles will hinder operations or damage equipment.

Effectiveness, Costs, and Feasibility

Cooke et al. (1986) have listed the advantages and disadvantages of harvesting for managing nuisance aquatic vegetation. These are:

a. Advantages.

(1) Most harvesting is not regulated by laws, nor is there a waiting period for water use.

(2) Nutrients and organic matter are removed.

(3) Harvesting may facilitate other treatments, such as grass carp or herbivorous insect introductions.

(4) Little impact occurs to nontarget areas.

(5) Costs compare favorably with herbicides in the midwestern United States, but not in southern areas where there are dense infestations of exotic plants such as waterhyacinth (*Eichhornia crassipes*).

b. Disadvantages.

(1) Harvesting is labor intensive.

(2) Relatively small areas can be treated per day.

(3) Fragmentation and spread of nuisance plants may occur.

(4) Harvesting and unloading sites may be separated by great distances.

(5) Operating depths are limited.

(6) Favorable weather is needed.

(7) High initial capital costs occur, and there may be substantial downtime for repairs.

(8) Possible problems of access may occur.

(9) Harvesting is of limited applicability when the growing season is long, regrowth rates high, and infestations very dense.

The effectiveness of harvesting in producing control of vegetation appears to be related to the number of harvests per season, when harvesting occurs, the types of plants present, the amount of vegetation per unit area, and how the machine is used. Most reservoirs are too large to obtain complete control of nuisance vegetation by harvesting. Therefore, selected harvesting has to be planned. Harvesters can be used effectively in more restricted areas such as marinas, swimming areas, docks, and water intakes, if a machine of proper size is used.

Nichols and Cottam (1972) compared the effectiveness of single and multiple harvests in controlling biomass and next-season regrowth of Eurasian watermilfoil in Wisconsin. A single harvest reduced biomass to 10 to 25 percent of the original level; three harvests 1 month apart essentially eliminated all plant material. Reduced growth in the following year was most apparent in plots harvested three times in the preceding summer, and was apparently due to cutting of root crowns. Wile et al. (1977), Wile, Hitchin, and Beggs (1979), and Conyers and Cooke (1983) also found excellent control in northern lakes if the cutter bar was operated in the sediments so as to cut root crowns. Cooke and Carlson (1986) investigated the effects of harvesting frequency and technique on regrowth of a dense infestation of *Myriophyllum spicatum*. They found that season-long control of this plant could be achieved with one harvest if the plant root crowns were also harvested. By way of contrast, Anderson (1984) followed the regrowth of Eurasian watermilfoil, in another area of the same reservoir (LaDue Reservoir, Ohio) as Cooke and Carlson (1986), after the plants had been harvested with the traditional method in which 2- to 5-cm stumps are left and few if any root crowns are removed. Anderson found that the milfoil biomass in the harvested area equaled the original biomass and the biomass of a control area within 21 days. While the technique of cutting and removing root crowns may be more time-consuming and can produce damage to cutter blades, the harvested area may not require a second harvest. Further testing of this approach is needed.

Some macrophyte species are more affected by harvesting than others. Nicholson (1981) has suggested that harvesting promoted milfoil growth in Chautauqua Lake (New York) because this plant can spread and become established from fragments. Other species, such as the pondweed *Potamogeton*, which can be a severe nuisance in reservoirs, are susceptible to harvesting because they emphasize sexual reproduction and regenerate poorly from fragments. Pondweeds therefore might be replaced by milfoil in harvested reservoirs where both species are present.

Efficiency is related to the density of plants, the size of the area to be harvested, the number of harvesters available, the presence of obstacles in the water, and the distance of disposal sites from harvesting areas. For example, some lakes in British Columbia (Newroth,* Cooke et al. 1986) have very narrow bands of dense milfoil beds along their length. Harvesting was

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* Personal Communication, 1986, P. R. Newroth, British Columbia Ministry of Environment, Vancouver, BC.

slow and inefficient there since the machine had to travel very long distances to the shore disposal sites. Obviously, efficiency will be increased if disposal sites are located near the harvesting site, or if a separate transporter unit is used with the harvester.

In southern waters, particularly Florida, harvesting is seldom applicable. Here, plant densities may often be over 62 tons ha^{-1} and as much as 370 tons ha^{-1} in dense waterhyacinth infestations, and regrowth rates can be several hectares per day of new vegetation. In these situations, harvesting rates will be slow, disposal costs will increase, harvests will have to be repeated very frequently, and exotic plant growth rates will exceed the rate at which they can be harvested during some periods. In these cases, harvesting is not an effective management option, and reservoir managers may have to rely on other procedures, such as herbicides, or an introduction of biological controls following a harvest or herbicide application.

Harvesting could have an additional restorative or improvement effect for a reservoir when large amounts of nutrients are removed as plant biomass. Removal of nutrients would have to be sufficient to significantly lower the net external nutrient loading or to significantly interfere with the nutrient release that occurs during the autumnal dieback of plants. Nutrient removal to this extent is unlikely in many large reservoirs. Burton, King, and Ervin (1979) have listed the conditions that must be met to accomplish sufficient nutrient removal: (a) macrophyte densities must be high, (b) phosphorus loading to the reservoir must be less than 1.0 g m^{-2} $year^{-1}$, (c) most of the reservoir surface must be covered with plants, and (d) macrophytes must regrow every year. They provide a useful nomograph to estimate the macrophyte harvest required to equal net phosphorus income as a function of percent coverage by plants. While many reservoirs have nutrient loading in excess of the above value, few have complete macrophyte coverage, and the harvest season may be short. As well, complete macrophyte removal could be detrimental to a sports fishery, could contribute to increased turbidity through erosion of littoral sediments on windswept shores, or may stimulate an algal bloom. Harvesting is therefore unlikely to be a factor in improving reservoir trophic state through nutrient removal alone.

Harvesting may lead to improvement in dissolved oxygen conditions through the removal of particulate and dissolved organic matter, which is continually produced by sloughing of plant tissue and by plant decay at summer's end. Evidence regarding the significance of this removal to reservoir water quality is insufficient. Experiments with enclosures (Landers and Lottes 1983) and calculations from shallow Lake Wingra, Wisconsin (Carpenter 1980), suggest that macrophyte decay is a significant factor to trophic state.

Carpenter (1983) also points out that macrophytes can close a positive feedback loop (see Figure 4) that enhances sediment accumulation and thus macrophyte growth. This occurs through stimulation of algal growth by release of nutrients and organic matter. Sedimentation of dead algal cells and macrophyte tissues adds to sediment accumulation. Since these plants are limited by light penetration, any factor that promotes a decrease in depth, as increased sedimentation would do, will ultimately promote an increase in the area of coverage of macrophytes in the reservoir. Harvesting of plants may be one factor in disrupting this positive feedback loop.

The costs of harvesting are related not only to high purchase price and problems with efficiency, but also with machine breakdowns and the number of reharvests per year. Downtime may increase sharply when an undersized machine is employed or where the equipment is heavily stressed or not operated properly. The British Columbia lakes were an example of high stress on the equipment. A typical operating year there consisted of 2,764 hr of work, of which 44 percent was downtime (Cooke et al. 1986). It is strongly recommended by manufacturers that the machine purchased be of a size appropriate to the area to be harvested, as well as to plant density. Or, the reservoir manager may wish to employ one of the several contract harvesting companies and, in this way, test harvesting as a solution of that particular reservoir's weed problems.

Costs for harvesting vary regionally and reflect differences in the density of plant infestations, their regrowth rate, and other factors affecting operations. Table 12 is a comparison of harvesting and herbicide costs for the midwest and Florida. Literature cost values have been converted to 1987 dollars by using changes in the consumer price index.* In the Midwest, expenditures for harvesting and herbicides are clearly comparable, but in Florida, harvesting is significantly more expensive (and less effective). Cost comparisons are difficult to make due to wide variances in reporting of

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* Personal Communication, 1988, Dr. Thomas Lough, Kent State University, Kent, OH.

Table 12

Comparison Between Midwest and Florida Cost Ranges (1987 Dollars) for Harvesting and Herbicide Treatments of Lakes and Reservoirs

Procedure	Cost Range (per hectare)
Harvesting	
Midwest	\$333-918
Florida	\$734-35,531*
	\$734-2,900**
Herbicides	
Midwest	\$467-905
Florida	\$434-863

Note: Based on data from Koegel, Livermore, and Bruhn 1974, 1977; Culpepper and Decell 1978; Dunst and Nichols 1979; McGehee 1979; Smith 1979; Cannellos 1981; Sassic 1982; Shireman 1982; Shireman et al. 1982; Conyers and Cooke 1983; Sabol and Hutto 1984; Cooke et al. 1986; and Thayer and Ramey 1986.

* Larger number refers to cost for dense waterhyacinth population.

** Larger number refers to cost for dense hydrilla population.

factors such as overhead, labor, transportation, and disposal, and most investigators do not report costs at all so that data are scarce. Further, costs vary with the type of plant infestation. For example, Thayer and Ramey (1986) report a harvesting cost range for *Hydrilla* to be \$1,230 to \$2,900 ha^{-1}, but for waterhyacinth, the cost range is \$10,960 to \$35,531 ha^{-1}.

Good estimates of effort (manpower), time, and cost of a proposed mechanical harvesting operation can be obtained using the US Army Corps of Engineers' computer model HARVEST (Sabol and Hutto 1984). Input requirements of the model are generally straightforward and easily measured or estimated. Naturally, the more precise the input variables provided by the user, the more precise the results will be. Output includes estimated time and costs, given specific machinery configurations, and density of vegetation growth. Compared with actual operations performed, these time/cost estimates are accurate enough to give a "best-case" estimate of the proposed effort to the planner. The HARVEST program, which runs on a personal computer, can be obtained by contacting the Program Manager, Aquatic Plant Control Research Program, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, MS 39181-0631.

Limitations and Concerns

Carpenter and Adams (1977) have reviewed the environmental impacts of harvesting. Macrophyte removal constitutes habitat removal for organisms such as snails, insects, and young fish, and the abundances of these animals can be sharply reduced. The evidence of a negative impact on fish is conflicting. Haller, Shireman, and DuRant (1980) found that harvesting removed about 85 kg of fish per hectare (76 lb per acre), primarily sunfish. Whereas Wile (1978), Storch and Winter (1983), and Mikol (1984) found that fish populations of lakes remained stable during harvesting. At Saratoga Lake (Mikol 1984) and Chautauqua Lake (Storch and Winter 1983), New York, small sunfish, perch, and bullheads were the dominant fish removed by the harvester. This suggests another possible benefit of harvesting. Removal of sunfish and perch also means removal of the organisms that can have a significant role in size-selective predation on the zooplankton species that graze algae. Ultimately, a significant removal of these small fish may mean lake improvement with regard to algae as well, since herbivorous zooplankton may have increased population density when fish predation on them is lowered.

Some investigators have been concerned about the initiation of algal blooms following extensive plant removal. This has been found in some lakes and reservoirs (Neel, Peterson, and Smith 1973; Nichols 1973; Anderson 1984; Cooke and Carlson 1986), but not in others (Wile, Hitchin, and Beggs 1979). The causes of this phenomenon are very poorly understood. Cooke and Carlson (1986) found much higher phosphorus concentrations in a bay after harvesting, although there is no evidence that this caused the algal bloom. There have been many suggestions of an antagonistic effect between algae and macrophytes (e.g., Shireman et al. 1983) so that removal of macrophytes "releases" algae from these restraints, and blooms occur. The inhibition postulated by these investigators could include shading, release of an inhibitory substance, or removal of cover for algae-grazing zooplankton species. Normally this problem will not occur on large reservoirs where complete vegetation removal is neither desirable nor feasible, and could therefore be an environmental problem only in some coves or bays. Another more significant impact of macrophyte harvesting (or any other procedure that produces plant elimination) may be the increased erosion of the littoral zone (Burton, King, and Ervin 1979). Rooted plants are important in preventing wave-generated erosion and turbidity. Windswept shores and coves should probably not have 100 percent plant removal.

Summary

Harvesting of aquatic plants is a procedure that can quickly improve portions of a reservoir for recreation (such as marinas and swimming areas), and at the same time may interfere with the release of nutrients and organic matter from these plants to the open water of the reservoir. Harvesting in which the cutter bar also cuts root crowns appears to be the best technique in retarding regrowth. Costs can be estimated with the US Army Corps of Engineers' model HARVEST (Sabol and Hutto 1984). The environmental impacts of harvesting are normally minor.

Table 13 is a summary of this technique.

Table 13

Summary of Harvesting

Characteristic	Description
Targets	Aquatic macrophytes.
Mode of action	Cut, collect, and remove plants.
Effectiveness	Moderately effective; symptoms may remain; nutrients and organic matter removed.
Longevity	Weeks to months.
Negative features	Habitat removal. Turbidity. Occasional algal blooms.
Costs	High initial equipment costs. Operator, storage, and maintenance costs may be high. Costs can be estimated through computer simulation (HARVEST).
Applicability to reservoirs	Could not manage infestation over large area unless several machines were available. Poor applicability in some southern waters with dense, rapidly growing infestations of exotic plants. Excellent for coves, marinas, and beach areas.

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PART XI: BIOLOGICAL CONTROLS

Problem Addressed

The biological control of nuisance macrophytes and algae is potentially the most effective method in terms of costs and long-term control. However, our knowledge of how to use biological controls is not well developed.

Biological controls are employed to obtain an acceptable level of plant biomass through the introduction of species that graze or parasitize specific plants, or by the manipulation or elimination of endemic animal species that directly or indirectly control plant growth. The use of biological controls can produce a slow, gradual response that may be long-lasting, in contrast to other methods.

This part describes biological control techniques that are currently available or are undergoing testing and provides general guidelines for their use. Emphasis is placed on phytophagous fish and insects and upon the control of nuisance algae through manipulation of fish populations. Portions of these topics have been reviewed by Schuytema (1977) and Cooke et al. (1986).

Phytophagous Fish

The grass carp (white amur, *Ctenopharyngodon idella* Val.) was introduced to the United States in 1963. Earlier, it was exported to Europe, Japan, and Mexico, primarily as a food fish and later as a controller of macrophytes. The fish's native habitat is in eastern China. The animal has some characteristics which make it desirable as a biological control agent for aquatic plants, including tolerance to low dissolved oxygen and to a wide range of temperatures (Opuszynski 1972). In other ways, this fish is far from ideal, since its tolerances are so high that it cannot be restricted in range and it is not easily controlled or eliminated if it escapes to a nontarget lake or reservoir.

The following paragraphs discuss the biology of grass carp. Information on their reproduction (including the development of sterile triploids), feeding preferences, stocking rates, effectiveness, costs, and associated problems is included. Because grass carp have the potential to drastically alter the biota of a reservoir, and in some states they cannot be used at all, managers are urged to consult with the appropriate state agency before stocking this fish.

Reproduction

A significant question about the use of grass carp in reservoirs, where escape to downstream ecosystems could easily occur, is whether they will infest or reproduce in sites where vegetation is desirable. A marsh or lake that is important for nesting or migrating waterfowl would be an example of this kind of habitat. It was originally believed by early importers of grass carp that the stringent conditions for reproduction would not be found in the United States. Spawning occurs in the deep channels of large rivers following a sharp rise in water level, temperature above 17° C, and a current velocity above 0.6 to 0.8 m sec^{-1}. Survival of fry depends upon a downstream quiescent area where plankton are abundant and predation low (Smith and Shireman 1983, Pauley and Thomas 1987). However, direct evidence of reproduction in Arkansas and Louisiana has been reported (Connor, Gallagher, and Chatry 1980), leading investigators to search for a means of obtaining sterile fish which are as effective in consuming vegetation as the fertile, diploid (both members of each pair of chromosomes in each cell) fish originally introduced to the Nation's waters.

Early attempts to eliminate the possibility of reproduction involved the use of monosex populations and surgically sterilized animals. Shortcomings of this approach included the possibility of an unwanted introduction of animals of the opposite sex and the regeneration of gonads. A second approach involved the intergeneric cross of female grass carp and male bighead carp (*Hypophthalmicthys mobilis*) to produce sterile, hybrid offspring. Two major problems with the hybridization approach were the production of some subvital but fertile diploids and a comparatively low feeding efficiency (Allen and Wattendorf 1987; Bonar, Thomas, and Pauley 1987).

A solution to the problems of the sterile hybrid involved the production of pure (unhybridized) triploid (three members of each chromosome in cells) grass carp. Hydrostatic pressure or high temperature techniques are used to produce nearly 100-percent triploids (Cassani and Caton 1986). Since no known procedure can produce 100-percent triploidy consistently, and because the diploids and triploids cannot be accurately separated by looking at them, fish producers needed a technique to verify that every fish sold is triploid.

Currently, the best technique is to use a Coulter Counter to examine a drop of blood taken from each fish. Triploid red blood cells are larger than diploids, and the Counter easily verifies cell size. Three workers can examine 2,000 to 3,000 fish per day. Triploids are apparently functionally sterile, and there is an extremely low probability that triploids could be a source of a large population of reproducing diploids (Allen and Wattendorf 1987; Allen, Thiery, and Hagstrum 1986).

The production and verification of 100-percent sterile fish has prompted several more states to allow their introduction. As of September 1987, 18 states prohibit grass carp, 12 have no constraints on the use of fertile diploids, 16 allow only triploids, and 4 are studying the triploid prior to release (Allen and Wattendorf 1987).

Food preferences

Grass carp (diploid and triploid) are voracious consumers of aquatic plants but exhibit distinct feeding preferences that seem to vary from region to region in the United States. Table 14 is a summary of these preferences. The data for Florida are extensive, reflecting the longer history of the species' introduction and the number of investigators.

The regional differences in grass carp food preferences could have important management implications. *Elodea densa* is a preferred species in Florida but is not preferred or may not be eaten when plants grown in Oregon-Washington are offered to fish. *Ceratophyllum demersum* is a preferred plant in Florida, variably eaten in Oregon-Washington trials, but not eaten by Illinois grass carp. Feeding trials in Illinois were with diploid fish. The question remains whether palatability of plants varies from region to region, whether there is some genetic basis to carp feeding behavior or whether further studies will demonstrate that these apparent geographical differences are produced by the design of experiments. It appears that when preferred plants such as *Hydrilla*, many species of *Potamogeton*, *Chara*, or native *Elodea* are present, and stocking rates are moderate to low, nonpreferred or noneaten nuisance species that are also present could predominate after several years of grass carp feeding.

It should be noted that major nuisance exotic species, including water-hyacinth, alligatorweed, and Eurasian watermilfoil, are not eaten or may be nonpreferred plants. Much additional research is needed with regard to grass carp feeding preferences and their management implications.

Table 14

Feeding Preference List, in Approximate Order of Preference, for Triploid Grass Carp in Florida, Illinois, and Oregon-Washington Studies

Florida	Illinois*	Oregon-Washington
	Preferred Plants	
Hydrilla verticillata (hydrilla)	*Najas flexilis* (brittle naiad)	*Potamogeton crispus* (curly-leafed pondweed)
Potamogeton illinoiensis (Illinois pondweed)	*Najas minor* (naiad)	*Potamogeton pectinatus* (sage pondweed)
Potamogeton spp. (pondweeds)	*Chara* (muskgrass)	*Potamogeton zosteriformis* (flat-stemmed pondweed)
Najas guadalupensis (southern naiad)	*Potamogeton foliosus* (pondweed)	*Elodea canadensis* (elodea)
Elodea densa (Brazilian elodea)	*Elodea canadensis* (elodea)	*Vallisneria* sp. (tapegrass)
Elodea canadensis (elodea)	*Potamogeton pectinatus* (sago pondweed)	
Chara spp. (muskgrass)		
Lemna spp. (duckweed)		
Nitella spp. (stonewort)		
Ceratophyllum demersum (coontail)		
Eleocharis acicularis (needle rush)		
Pontederia lanceolata (pickerelweed)		
Wolffiella spp. (bog mat)		

(Continued)

Note: Data based on Hestand and Carter 1978; Osborne 1978; Nall and Schardt 1980; Van Dyke, Leslie, and Nall 1984; Sutton and Van Diver 1986; Bowers, Pauley, and Thomas 1987; and Leslie et al. 1987.

* Diploid carp.

(Sheet 1 of 3)

Table 14 (Continued)

Florida	Illinois	Oregon-Washington
	Preferred Plants (Continued)	
Wolffia spp. (watermeal)		
Azolla spp. (azolla)		
Spirodela (duckweed)		
	Variable Preference - May Eat	
Myriophyllum spicatum (Eurasian watermilfoil)	*Potamogeton crispus* (curly-leafed pondweed)	*Myriophyllum spicatum* (Eurasian watermilfoil)
Bacopa spp. (bacopa)		*Ceratophyllum demersum* (coontail)
Polygonum spp. (smartweed)		*Utricularia vulgaris* (bladderwort)
Utricularia spp. (bladderwort)		*Polygonum amphibium* (amphibious smartweed)
Cabomba spp. (fanwort)		
Fuirena spp. (umbrellagrass)		
Nymphaea spp. (waterlillies)		
Brasenia schreberi (watershield)		
Hydrocotyl spp. (pennywort)		
Panicum spp. (torpedograss)		
Stratiotes aloides (water aloe)		

(Continued)

(Sheet 2 of 3)

Table 14 (Concluded)

Florida	Illinois	Oregon-Washington
	Nonpreferred - Does Not Eat	
Nuphar luteum (spatterdock)	*Ceratophyllum demersum* (coontail)	*Potamogeton natans* (floating leaf pondweed)
Vallisneria americana (tapegrass)	*Myriophyllum* spp.	*Brasenia schreberi* (watershield)
Myriophyllum brasiliense (parrotfeather)		*Elodea densa* (Brazilian elodea)
Eichhornia crassipes (waterhyacinth)		
Alternanthera philoxeroides (alligatorweed)		
Nymphoides spp. (floating heart)		
Pistia stratiodes (waterlettuce)		
Phragmites spp. (reed)		
Carex spp. (sedge)		
Scirpus spp. (bulrush)		
Ludwigia octovalis (water primrose)		
Colocasia esculentum (elephant-ear)		

(Sheet 3 of 3)

The existence of these feeding preferences suggests the possibility that grass carp could allow nonpreferred plants to become abundant, particularly when understocking or fish escape occurs and only the most palatable species are then consumed. Fowler and Robson (1978) and Fowler (1985) report a shift in dominance from *Potamogeton* to *M. spicatum* in an understocked lake. In Deer Point Lake, Florida (Van Dyke, Leslie, and Nall 1984; Leslie et al. 1987; Van Dyke*), a large reservoir stocked in 1976, *M. spicatum* returned as a problem. Carp have either escaped or have been removed by predation from this impoundment, and the remaining low density of animals has been unable to control the milfoil. This lake has been restocked with grass carp.

Grass carp exhibit a metabolic strategy unlike most fish (Wiley and Wike 1986). Their aerobic metabolism rate is about half that of other fish, but their average consumption rate (at 21° C or higher) is about 50 to 60 percent of body weight per day, which is 2 to 3 times that of carnivorous fish. These two factors offset their very low assimilation efficiency, about one third that of carnivorous fish. About 50 percent of ingestion, on the average, is egested. Triploids have a growth rate of about 9 g day^{-1} (for a 1-kg carp fed 420 g of *Potamogeton crispus* per day). An energy budget for triploid carp is (Wiley and Wike 1986):

$$100I = 13M + 74E + 13G \tag{13}$$

where

I = ingestion

M = metabolism

E = egestion

G = growth

Average daily growth rates from Florida lakes range from 10.0 to 10.4 g day^{-1}. The largest grass carp (a diploid) recorded (Lake Wales, Florida) in the United States weighed 32.7 kg (72 lb).**

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* Personal Communication, 1987, J. M. Van Dyke, Florida Department of Natural Resources, Tallahassee, FL.

** Personal Communication, 1987, Harold Revels, Florida Fish and Game Commission, Tallahassee, FL.

Stocking rates

An appropriate stocking rate is critical to successful aquatic plant management with grass carp. Some managers may choose to stock at a rate that will produce plant eradication. This strategy is achievable with grass carp but is also associated with water quality changes and is unlikely to be compatible with sport fishing. Stocking rates are directly affected by the length of the growing season, water temperature, palatability of the plants, size of fish stocked, present and desired plant coverage, and preapplication plant management procedures. Fewer fish are needed when the feeding season is long, the nuisance plants are palatable, the water body is only partially plant-covered (perhaps due to harvesting or chemical treatment), and the management plan calls for some vegetation to remain. Also, serial stocking, where fish are added at intervals, usually requires fewer fish than batch stocking and also allows better management of the rate of plant reduction.

Aquatic plant infestations vary in coverage, biomass of plants per unit area, and species. Grass carp stocking rates such as "rule-of-thumb" formulas (e.g., 10 fish per acre of reservoir), which do not take these factors into account, can produce a density of grass carp that is too low when there is a dense, widespread infestation of a nonpreferred or unpalatable plant (e.g., *Myriophyllum spicatum*) or too high when opposite conditions are found. In one case, plant control may not be achieved. In the other case, eradication can occur.

Stocking rate guidelines, based upon reservoir-specific variables, have been developed (Bonar, Thomas, and Pauley 1987; Leslie et al. 1987; Wiley, Tazik, and Sobaski 1987), and these should be consulted. The most accurate stocking rates are developed from a program of data acquisition and the use of one of the computer models. Models in use now include (a) the US Army Engineer Waterways Experiment Station model (Miller and Decell 1984), (b) the Illinois Herbivorous Fish Stocking Simulation System (Wiley and Gorden 1985), and (c) the Colorado model (Swanson and Bergersen 1986). A model for the Pacific Northwest is under development (Pauley and Thomas 1987). Reservoir managers planning to introduce grass carp are strongly urged to consult a model appropriate to their area or to examine reports by Leslie et al. (1987) and Wiley, Tazik, and Sobaski (1987).

The effectiveness of grass carp is also related to sources of carp mortality. Fish less than 450 mm total length are susceptible to largemouth bass predation in Florida (Shireman, Colle, and Rothman 1978). Also, grass carp may escape in large numbers from some reservoirs, and barriers may have to be constructed.

Case histories

Detailed case histories of grass carp additions to large impoundments are not abundant, and most are from southern states because problems are most severe there and carp have been under study for years. Additional reports will become available for midwestern and far western lakes and reservoirs in the near future.

Lake Conroe, an 8,100-ha impoundment near Houston, TX, was filled in 1973 and is used for recreation and potable water supply. By 1981, *Hydrilla* had infested about 44 percent of its area, along with Eurasian watermilfoil and coontail. In 1981, triploid grass carp were stocked at a rate of 75 per vegetated hectare (about 23 kg ha^{-1}), a rate considered to be "overstocked." Two years later, nuisance plants had been eliminated. There was also a 40-percent reduction in water clarity due to a phytoplankton bloom in the year following plant eradication (Martyn et al. 1986; Noble, Bettoli, and Betsill 1986). The treatment was a success with regard to protecting lakeshore property values and enhancing recreation. Impacts on drinking water quality have not been reported.

Lake Conway, a 729-ha multipool impoundment in Orlando, FL, was stocked with diploid grass carp at a rate of about 15 fish per vegetated hectare (about 6 kg per vegetated hectare). In 2 years, *Hydrilla* and *Nitella* were eliminated in all pools, although *Vallisneria* was unaffected. The treatment is considered to be a success. There was minimal negative impact, although blue-green algae increased and waterfowl population decreased. Fishing for largemouth bass improved dramatically (Miller and King 1984).

There are several useful case histories of grass carp introduction to small impoundments (e.g., Mitzner 1978) and urban recreational lakes (e.g., Van Dyke, Leslie, and Nall 1984; Leslie et al. 1987).

Costs

Grass carp may be the least expensive means of achieving long-term control/eradication of some nuisance aquatic plant species (see Table 14 for lists of preferred and nonpreferred plants). In southern waters, in particular, plant harvesting is often either ineffective or too expensive, leaving herbicides and biological controls as primary alternatives. Shireman (1982) and Shireman et al. (1985) provide the following case history which demonstrates the comparative low cost-high effectiveness of grass carp.

At Lake Baldwin, an 80-ha lake in Orlando, FL, about 80 percent of the area was infested with *Hydrilla*. About \$100,000 was spent over 3 years to control the plants with HYDOUT (an endothall salt), at an average annual cost of \$33,333 for 64 ha (\$520 ha^{-1}). Treatments provided control but had to be repeated annually. In 1975 and 1979, grass carp were added at a total stocking rate of 35 fish per vegetated hectare and a cost of \$8,499 (\$106 ha^{-1}). There was no appreciable growth of aquatic plants between 1980 and 1985. When costs are amortized over the expected duration of effectiveness (10 years or more*), they fall to \$11 ha^{-1} (1975 dollars) or \$22 ha^{-1} in 1987 dollars. Over that same 10-year interval, had herbicide treatments continued with no inflation of chemical and labor costs, about \$333,333 would have been spent.

Environmental Impacts

The positive aspects of the use of grass carp to control aquatic plants are easily identified. These include elimination of the target plants (and often all plants); very low initial and long-term costs; low maintenance costs (unless fish removal is required); greatly enhanced boating, water skiing, and sometimes fishing; and long-term effectiveness. These animals can approach the "ideal" biocontrol agents when the system to be treated is large (making other management procedures very costly) and when the nuisance plant is a preferred species such as *Hydrilla*.

The negative effects are less easily identified, in part because so little is known about the long-term consequences of the elimination of all macrophytes from a lake or reservoir. Eradication of submersed and emergent plants is not uncommon when stocking densities are high; it has so far proven difficult to stock just enough carp to allow a certain desired percentage of plant cover to remain, but stocking models (e.g., Wiley, Tazik, and Sobaski 1987) may correct this problem. With techniques such as harvesting or herbicides, far more precise control of plant densities is possible. In many cases, distinct water quality changes accompany macrophyte control or eradication by grass carp. These changes almost always include increased

* Personal Communication, 1987, J. M. Van Dyke, Florida Department of Natural Resources, Tallahassee, FL.

turbidity and shoreline erosion, perhaps because macrophytes previously damped wave action. Erosion in some carp lakes has been extensive enough to uproot trees along the shore. In some cases, there have been increases in algae, particularly nuisance blue-green algae, and a replacement of phytophilic invertebrate species with bottom dwellers. A major concern had been the potential impacts of the high rate of egestion of organic and inorganic matter. There appears to have been no report of a "pulse" of nutrients following carp introduction, in contrast to procedures such as herbicide treatments where very large amounts of plant biomass may be left to decay. Instead, there appears to be a continuous transformation of plant matter to fish tissue and to egested materials so that some nutrients in plant biomass are lost to a "sink" (fish tissue) and the rest to decomposing organic matter.

There have been concerns about negative impacts to other fish species. Some investigators have reported enhanced fishing success following elimination of vegetation (e.g. Bailey 1978, Miller and King 1984), while others have found interferences with spawning and growth (e.g., Ware et al. 1975, Forester and Lawrence 1978). Further studies in this area are required. There is some speculation that a slow release of nutrients from decomposing carp fecal material can be a subsidy to the planktonic food web and in this way contribute to enhanced energy flow to game fish.

Overstocking of grass carp can present other problems in addition to the direct effects of plant eradication. For example, waterfowl are dependent on aquatic plants for food. Lakes and reservoirs on significant flyways may be important resting/feeding sites and could be avoided if plants were eradicated. Further, if significant numbers of carp escape to downstream sloughs and marshes, further damage to desirable communities could occur.

Once stocked, management of fish density has proven to be very difficult in Florida waters.* Carp apparently avoid nets and traps and can escape electroshockers. It appears that once a lake or reservoir is stocked, users of the reservoir are committed to their decision until the fish die (10 years or so).

In summary, triploid grass carp are clearly an effective, low-cost, long-term agent for the eradication of some nuisance aquatic plants. Their use in the management of macrophytes to achieve a desired coverage has proven

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* Personal Communication, J. M. Van Dyke, Florida Department of Natural Resources, Tallahassee, FL.

to be more difficult, and significant environmental changes can occur after their stocking. A nationwide research program is now under way to establish regional food preferences and stocking rates and to assess the effectiveness of carp introductions.

Tilapia zillii, another phytophagous fish, has been used successfully to control rooted plants and filamentous algae, but there are few literature reports concerning this fish. A distinct characteristic of this species is its intolerance to prolonged exposure to temperatures below 10° C. This limits the use of this species for northern lakes or reservoirs. Childers and Bennett (1967) report that a *Tilapia* density of 1,000 $acre^{-1}$ (2,500 ha^{-1}) was sufficient to control *Chara*, *Potamogeton foliosus*, and filamentous algae in an Illinois pond. This fish was used to control *Utricularia vulgaris* (bladderwort) in a North Carolina cooling water impoundment. A stocking density of 50 $acre^{-1}$ (124 ha^{-1}) eliminated the problem with bladderwort, but blue-green algae and rooted plants then invaded the lake (Schiller 1984). *Tilapia* has also been used successfully to control *Elodea densa* in Hyco Reservoir, North Carolina (Schiller 1984).

Insects and Plant Pathogens

The history of the deployment of insects and plant pathogens as biocontrol agents of nuisance exotic vegetation is in striking contrast to that of the grass carp. Alligatorweed (*Alternanthera philoxeroides*) and waterhyacinth (*Eichhornia crassipes*) are exotic plants that have become nuisances of great economic significance in southern waters of the United States. Several cooperative investigations, involving scientists from several universities, the US Department of Agriculture, and the US Army Corps of Engineers (e.g., Coulson 1977, Cofrancesco 1984, Balciunas 1986), revealed species of insects in the native habitats and ranges of these plants that might be imported to the United States and released as biocontrol agents. These insects were kept under strict quarantine while studies were conducted to assess their specificity to the target plant and the possibility that their introduction might also introduce parasites and diseases.

Six species of insects have been imported and released for plant control. *Neochetina eichhorniae*, *Neochetina bruchi* (Coleoptera:Curculionidae), and *Sameodes albiguttalis* (Lepidoptera:Pyralidae) were imported from

Argentina, a native habitat of waterhyacinth. *Neochetina eichhorniae* is obligatorily monophagous on hyacinth, and *N. bruchi* is found on plants of only two genera. Its distribution does not exceed that of waterhyacinth, and its life cycle can be completed only on the hyacinth (Perkins and Maddox 1976, Center 1981). The first insect to be released in the United States following importation, quarantine, and testing was the alligatorweed flea beetle *Agasicles hygrophila* (Coleoptera:Chrysomelidae). Two other species have been introduced for alligatorweed control, *Vogtia malloi* (Lepidoptera:Pyralidae) and *Amynothrips andersoni* (Thysanoptera:Phlaeothripeae). These species are also confined in their feeding or life cycle to the target plant (Coulson 1977).

Insects have been successful in exerting control of alligatorweed and waterhyacinth in several states. The primary control of alligatorweed in Florida, Louisiana, Mississippi, and south Alabama is by the alligatorweed flea beetle *A. hygrophila* and the alligatorweed stem borer *V. malloi*. Both have contributed to control in Georgia, but their impact is limited in North Carolina because the colder weather inhibits or completely prevents over-wintering of the insects (Cofrancesco 1984). The alligatorweed thrips (*Amynothrips andersoni*) overwinter better and thus stress the plants earlier in the season. They also attack the terrestrial form of this plant, while the flea beetle and stem borer require the aquatic form (Cofrancesco 1984, McGehee 1984). *Neochetina eichhorniae* and *N. bruchi* have been released in Florida, Georgia, Louisiana, Mississippi, Texas, and California for control of water-hyacinth, and effective control has been reported from Florida, Louisiana, and Texas (Center and Durden 1984). *Sameodes albiguttalis*, the waterhyacinth moth, has been found to be most effective against small, luxuriantly growing plants, such as might occur following a chemical or mechanical treatment (Center and Durden 1984).

The impact of the two weevils on waterhyacinth in Louisiana serves as an example of insect effectiveness (Perfetti 1983, Goyer and Stark 1984, Sanders 1984). The distribution of waterhyacinth reached a statewide peak of 690,000 ha in 1975. By 1980, the beetles had reached swarming levels, and dry weather had concentrated the plants. Plant distribution declined to 122,000 ha in 1980; the insects, along with the drought and increased efficiency of herbicide spraying, are believed to have been the major factors.

Goyer and Stark (1984) reported that beetle densities as low as one individual per plant had an adverse effect on plant height and number of reproductives.

Plant harvesting or herbicide application can sharply reduce the effectiveness of an established population of *Neochetina* on waterhyacinth. After such treatments, insects disperse and the plants regrow in log phase without control. It may then take several years for insect densities to reach effective levels (Center and Durden 1984, 1986). Haag (1985) has found that diquat, 2,4-D, and glyphosate were nontoxic to the weevils at low doses and that adult insects will migrate to nearby untreated plants. Integrated methods of aquatic plant management may ultimately prove to be among our most successful approaches. Haag (1986) describes an experiment in which insects and a herbicide were used together to control a waterhyacinth infestation in a pond. About 75 percent of the weed mat was sprayed with 2,4-D, in increments of 20 percent, and both species of waterhyacinth weevils were "herded" into an unsprayed area where they overwintered, increased sharply in density, and exerted complete control of the weeds in the next spring. Additional experiments with this approach are in progress in larger lakes and reservoirs.

Some native insect species are known to damage aquatic plants. Work is now in progress to identify these species and to determine their usefulness in plant control (Buckingham, Haag, and Habeck 1986; Haag, Habeck and Buckingham 1986; Habeck, Haag, and Buckingham 1986).

Plant pathogens should be ideal for control of rooted aquatic plants because they are numerous and diverse, usually are host specific, easily disseminated and self-maintaining, exert a limiting influence on target plants without eradication, and normally are not dangerous to man and domestic animals (Zetter and Freeman 1972; Freeman, Charudattan, and Conway 1975; Freeman 1977). Only one plant pathogen has been significant in plant control, the fungus *Cercospora rodmanii*, which was isolated from waterhyacinth in Rodman Reservoir, Florida. Pilot field tests were held in Louisiana in 1977-80, and a commercial formulation has been developed by Abbott Laboratories (Freeman et al. 1981; Perfetti 1983). The effectiveness of the fungus in Louisiana cannot be separated from the impacts of the beetles, but it is believed that its effect on waterhyacinth has been less than expected (Sanders 1984). Another species, *Cercospora piaropi*, was generally believed to produce only moderate plant damage. Martyn (1985), however, has documented a situation at Lake Conroe, Texas, where this fungus has apparently caused widespread damage to waterhyacinth. There are other fungal pathogens, including *Fusarium roseum* for control of *H. verticillata* (Freeman, Charudattan, and Cullen 1980) and *Fusariium sporotrichoides* for control of milfoil, *M. spicatum* (Andrews and Hecht 1981). Neither is operational.

The application of insects and/or plant pathogens may produce a reduction of plant biomass slowly, and control may be long-lasting. Few negative side effects are expected. Research is continuing with this type of biological control.

Biomanipulation

Biomanipulation is a term coined by Shapiro, La Marra, and Lynch (1975) and Shapiro (1978), although one of the first observations that algal blooms were absent in ponds with a certain type of fish community was made by Caird (1945). Biomanipulation includes some potentially effective but currently experimental procedures to control algal biomass. Among them is the manipulation or management of food webs to control fish species that recycle nutrients during browsing and feeding, or that promote algal growth through their predatory activities on microscopic animals (zooplankton) which graze on algae. These procedures may be difficult to implement in large reservoirs without continual management due to the frequent introduction of undesirable fish species, and because fish management is difficult in such habitats. However, it appears that food web manipulation can improve eutrophic systems. When biomanipulation is combined with control of nutrient loading, major declines in algal abundance can be expected (Benndorf 1987). Fish stocking programs should be initiated with regard for problems of algal biomass, as well as sports fishing. This means that the addition of gizzard or threadfin shad as forage for game fish, for example, should be undertaken with great caution. These fish preferentially consume the species of zooplankton which are the major grazers of some species of algae. Reservoir manipulations such as an occasional extreme drawdown can provide an ideal opportunity to restructure the fish community.

Figure 14 illustrates the open-water food chain or web (Shapiro et al. 1982). Briefly, zooplanktivorous fish (e.g., gizzard shad, alewives, perch, and small sunfish) graze on the largest species of zooplankton. These zooplankton are the most efficient grazers of algae, and their absence eliminates

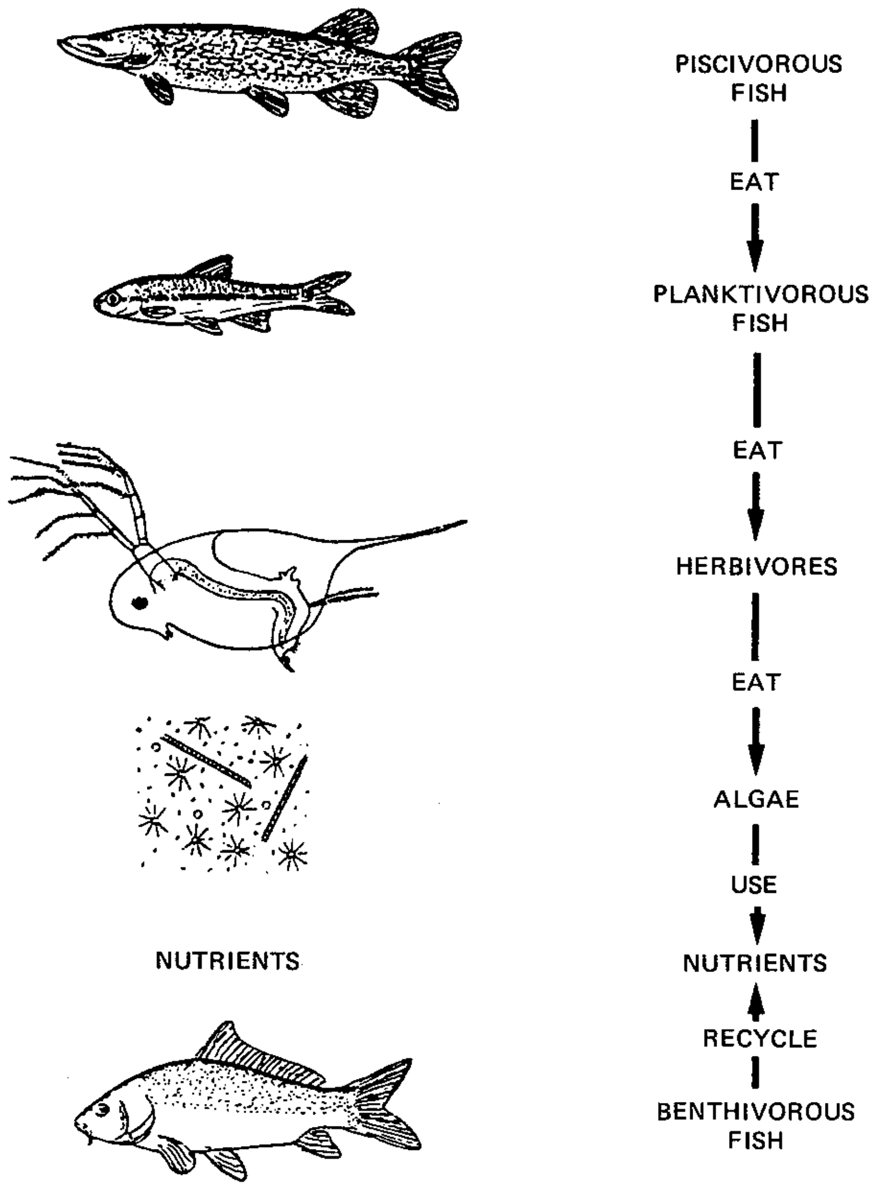

Figure 14. The aquatic food chain, indicating interactions between the components of the biomanipulation model (after Shapiro et al. 1982). See text for an explanation of the biomanipulation model

a significant source of mortality to algae. In a number of experiments with enclosures or small lakes, fish have been eliminated either by winter fish kill or by rotenone application (e.g., Lynch and Shapiro 1981, 1982; Shapiro and Wright 1984; Stenson et al. 1978). In these experimental systems, blooms of some algal species have been eliminated through zooplankton grazing alone, even under conditions of nutrient enrichment. In controls, where fish predation has eliminated the most effective algal grazers, leaving a system dominated by rotifers and small Cladocera, algal blooms and low water transparency continue.

Similar effects have occurred in macrophyte-dominated lakes to which grass carp have been introduced. When macrophytes are eliminated, the zooplankton community may shift from one dominated by large-bodied Cladocera, or Copepoda-Cladocera, to one dominated by Rotifera and the small-bodied Cladocera, *Bosmina* and *Ceriodaphinia*. The algal blooms that sometimes accompany the elimination of macrophytes (whether through grass carp or through mechanical or chemical methods) are often attributed to increased nutrient availability. They may in fact be due to the elimination of cover for zooplankton, followed by intense predation by fish and the elimination of effective algal grazers. Or, algal blooms may in part be explained not by the low grazing rate of small-bodied zooplankton, but by the high rates of nutrient cycling associated with small-bodied zooplankton (Henry 1985).

A reduction in zooplankton grazing on algae may also occur when dissolved oxygen in deep water is low or absent. Zooplankton migrate to the cooler, darker waters of the metalimnion and hypolimnion during the day to escape predation by sight-feeding fish, and then return to surface waters to graze at night when sight-feeding by fish is less (Vuorinen, Rajasilta, and Solo 1983). Low dissolved oxygen will restrict this daily migration and permit effective fish feeding on zooplankton during daylight. As discussed in another section, one of the benefits of artificial circulation and hypolimnetic aeration may be to provide an oxygenated refuge for zooplankton from the sight-feeding of fish.

Any factor that produces significant and prolonged zooplankton mortality may bring about persistent algal blooms, low transparency, and associated problems with dissolved oxygen and quality of potable water. Shapiro (1979) points out that agricultural runoff can have significant concentrations of pesticides that are lethal to zooplankton in trace amounts. He lists 10 commonly used insecticides, including malathion, diazinon, and Baytex, for which the 48-hr LC_{50} concentration to zooplankton is less than 1.0 $\mu g\ \ell^{-1}$. Copper sulfate, the most commonly used algicide, is very toxic to zooplankton (DeMayo, Taylor, and Taylor 1982). Winner and Farrell (1976) and Winner et al. (1977) found significant mortality to species of *Daphnia*, an effective algal grazer, at a copper concentration of about 40 μg Cu ℓ^{-1}, a concentration well below that normally used for algal control. Chelated copper algicides are somewhat less toxic to *Daphnia* (Biesinger, Andrew, and Arthur 1974). The often observed "rebound" of algal biomass shortly after a copper treatment may be due to mortality of herbivorous zooplankton.

Fish have been shown to be very significant in the regeneration of nutrients from sediments to the water column. The common carp (*Cyprinus carpio*) can add phosphorus to the water column of lakes at rates similar to the external loading (La Marra 1975). Similar findings have been made about the brown bullhead (*Ictalurus nebulosus*) by Keen and Gagliardi (1981). Control or removal of these rough fish can improve water clarity by elimination of their bottom-browsing activities and by lowering the rate at which phosphorus is recycled from sediments to water and then to algae.

Several fish management techniques can produce an improvement in the trophic state of the reservoir. The first step is to evaluate the reservoir, as outlined in an earlier section, to establish the types of fish and zooplankton and thus the likelihood that improvement through fish manipulation can occur. The addition of piscivorous fish to control planktivorous fish may meet with limited success, if any (Bennett 1970), but this idea has had little study as far as trophic state improvement is concerned. Rough fish removal by seining has also met with little success, primarily because the technique is labor intensive and very inefficient.

The use of rotenone, a fish poison, to eliminate all fish may be the only feasible procedure to correct fish problems, as illustrated by the work of Shapiro and Wright (1984). Round Lake, Minnesota, a small, shallow, natural lake, was treated with rotenone to eliminate a fish community dominated by planktivorous bluegill (*Lepomis macrochirus*), black crappie (*Pomoxis nigromaculatus*), and the benthivorous black bullhead (*Ictalurus melas*). Following treatment, the piscivorous largemouth bass (*Micropterus salmoides*) and walleye (*Stizostedium vitreum*) were added, along with channel catfish (*Ictalurus pumetalus*) to control the reestablishment of the black bullhead, a fish that can increase internal nutrient loading. A marked improvement in transparency occurred. Prior to rotenone application, the mean summer Secchi disc transparency was 2.1 m; in the two subsequent summers, it averaged 4.8 m and 4.7 m, respectively. *Daphnia pulex*, a large-bodied herbivorous zooplankton species that was rare before elimination of the planktivores, became abundant.

The use of rotenone to eliminate fish in a reservoir may not be feasible due to the large area to be treated, the possibility of damage to downstream communities, the possibility of reinvasion from upstream, and the use of water for drinking. According to Bennett (1970), a dose of 1.0 mg ℓ^{-1} will produce a complete fish kill if applied when the weather is warm (water temperature 20° C or warmer). No toxicity to fish should remain after 7 days, although this must be checked by placing caged fish in the reservoir. Rotenone may not be used in potable waters.

Water level drawdown can be very effective in restructuring fish communities, as well as in eliminating certain species of macrophytes (see Part IX). Pierce, Frey, and Yaun (1963) and Lantz (1974) describe the enhancement of piscivorous fish populations through drawdown. Small fish are trapped in vegetation and killed. Forage-size bluegills, which are among the planktivores, decline conspicuously, apparently due to bass predation in the remaining pool (Herman, Campbell, and Redmond 1969; Bennett 1970). Lantz et al. (1964) report that winter drawdown can remove gizzard shad and sunfish and that summer drawdown can prevent their spawning. Drawdown when spring water temperature reaches 10° to 15° C has been used effectively to kill common carp eggs (Shields 1958). Hulsey (1958) recommended that new reservoirs have a deep channel or harvest basin that will support fish during drawdown. Then, seining or use of rotenone could take place without endangering downstream fish communities.

Restocking of a reservoir must be done with the objectives of the reservoir users in mind. Highly productive water bodies are usually ideal for fishing, but often have low transparency, high algal biomass, and problems associated with dense algae. Fish stocking in a fishery reservoir should be directed toward that activity. However, in those cases in which other types of recreation are important, a drinking water supply is involved, or an oxygen-free hypolimnion poses a problem, fish stocking to enhance zooplankton grazing and stocking to reduce or eliminate nutrient recycling by fish such as common carp or bullheads would be an important consideration. The use of planktivores such as shad as forage for game fish would be inconsistent with an objective of high water transparency or potable water free of taste and odor.

Summary

Biological control of nuisance algae and macrophytes is a rapidly developing area of lake and reservoir management. A number of insects and plant pathogens are operational and have been proven to be successful in control of waterhyacinth and alligatorweed. Restructuring of fish communities offers great promise for algal control but may be impractical for large reservoirs.

The introduction of phytophagous fish (primarily sterile, triploid grass carp) for control, or more likely elimination, of macrophytes has proven to be successful and inexpensive. Significant problems can be associated with their use. Grass carp are operational in several states.

Table 15 is a summary of biological control techniques.

Table 15

Summary of Biological Controls

Characteristic	Description
Targets	Macrophytes and algae.
Modes of action	Grazing by fish, insects, and zooplankton. Infection with pathogens.
Effectiveness	Variable, dependent on biological control agent and target plant.
Longevity	Short- to long-term.
Limitations	Possible replacement of one noxious plant species by another. Few proven biological agents are currently available. Environmental conditions can control distribution and effectiveness of control agents.
Costs	Potentially less expensive than typical mechanical or chemical controls.
Applicability to reservoirs	Questionable due to open nature of some reservoirs.

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PART XII: SEDIMENT COVERS

Problem Addressed

Sediment covers are used to control or eliminate nuisance rooted plant growth, particularly in limited areas of reservoirs such as coves, marinas, swim beaches, and boat docks. The procedure can provide reservoir users with immediate and long-term access to areas previously choked with plants. This method of plant control has been reviewed by Cooke (1980).

Theory and Design

Sediment covers control the growth of rooted aquatic plants in several ways. Some covering materials are actually screenlike in nature so that gas bubbles do not accumulate under them. The screens appear to operate by physical limitation of the plants, since light can penetrate the screen (Perkins, Boston, and Curren 1980). Other materials are completely opaque and may exert control by light limitation also.

Engel (1982) lists these advantages of the use of sediment covers for rooted plant control:

- Use is confined to specific areas.
- Covers are usually out of sight.
- Covers can be installed where harvesters or sprayers cannot gain access.
- No toxic substances are used.
- Permits or licenses are usually not required.
- They are easily installed over small areas.

Engel (1982) cites these disadvantages to the use of sediment covers:

- They do not address the cause of the problem of excessive rooted plant growth.
- They are expensive.
- They are difficult to apply over large areas or over areas with obstructions.
- They may slip on steep grades or float to the surface if gases accumulate beneath them.
- They can be difficult to remove or relocate.

- They may rip during application.
- Some materials are degraded by sunlight.

The successful use of sediment covers to control rooted aquatic vegetation is directly related to the technique of application, as well as to the selection of materials. Materials that have been used as sediment covers in reservoirs and lakes include polyethylene, polyvinyl chloride, polypropylene, nylon, fiberglass, synthetic rubber, and burlap. Some of these materials are highly unsuitable; some are very effective.

The cover should be applied directly on the sediment surface without "ballooning" or bare spots. In many cases, especially with soft, muck sediments, this will be very difficult since stakes may not hold, and bricks or other weights will anchor only the site at which they are placed. When covers are installed over heavy vegetation, as is often the case, there will be great difficulty in getting the material flush with the sediment surface.

Several steps, if included, can help ensure a successful installation. Use SCUBA to survey the reservoir sediments. Locate and mark obstructions such as stumps or boulders. Test the sediments for their ability to hold a stake. If the sediments are highly fluid or muck-like, then either very long stakes will be needed or anchors such as bricks will have to be used. Stakes cannot be used in hard sediments such as gravel. If the sediments are unsatisfactory for steel stakes, examine the possibility of sewing link chain into the edge of the fabric. Obviously, some sediment covering materials cannot be used for this type of holdfast.

The most commonly used stake has been made of 1/8- or 1/4-in.-diam steel wire. Bend the stake so that one end is L-shaped. Sharpen the other end to ease the penetration of the fabric and the mud. The length of the stake needed will vary directly with the degree of softness of the mud.

In nearly every situation, SCUBA will be needed for an effective application. A successful procedure in many applications has been to build a reel in the stern of a boat. Most of the materials are sold in rolls of 7- to 8-ft width and 100-ft length. Two applicators, one on each side of the boat, unroll the screen from the reel. The screen is pressed and smoothed to the sediment surface and fastened. Stakes, bricks, or other fasteners will be needed every 1 to 2 m, depending upon the density of the vegetation. In deeper waters, divers will be needed to apply the covers. Be certain that a backup, fully suited diver is prepared to assist the applicators.

The screen is laid down out to the edge of the zone of rooted plants, if needed. Be certain to overlap the rolls so that there are no bare spots. The application should be examined in about 2 to 3 weeks to reposition stakes and smooth the screen to the sediment surface. A poor application will have missed areas, and extensive "ballooning" or even lift-off of the cover could occur. This will be a nuisance to boaters and swimmers, and any screen with a specific gravity less than 1.0 poses a hazard to reservoir operation if substantial amounts of it break free and float toward intake structures.

Application is best accomplished during water level drawdown. Plants are absent, obstructions can be avoided or removed, and the applicators can see their work. Otherwise, early summer before dense vegetation develops is optimum. As Perkins, Boston, and Curren (1980) and Engel (1982) have shown, screens can be removed after 2 months and placed elsewhere. The area treated first (May-June) should remain essentially free of weeds during the remainder of the summer, and the screen can be used for July-August control elsewhere. Harvesting prior to application should also make the job easier and the coverage more complete.

Effectiveness, Costs, and Feasibility

Three materials, Aquascreen (fiberglass), Dartek (nylon), and Typar (polypropylene), have received extensive testing and have been shown to be effective in controlling rooted aquatic plants. The use of these materials is best described by presenting brief case histories of their use. The other materials will also be briefly described. The reader should consult Cooke et al. (1986) for other case histories and descriptions of techniques.

Aquascreen

Aquascreen is a flexible, heavy (specific gravity = 2.54) fiberglass screen coated with polyvinyl chloride. It resembles window screen. It is sold in rolls of 7 by 100 ft, with a mesh size of 62 apertures cm^{-2} at \$0.20 ft^{-2} or \$21,000 ha^{-1} (1984 prices; Menardi-Southern Division of US Filter Corporation, Augusta, GA), plus installation charges.

Aquascreen has been shown, in many investigations, to be completely effective in controlling rooted aquatic plants, at least for the season of application. Mayer (1978) first described the use of this product following extensive experiments in Lake Chatauqua, New York. He reported that

Myriophyllum spicatum (Eurasian watermilfoil) and *Potamogeton crispus* (curly-leafed pondweed) infestations decayed in 2 to 3 weeks following application. The deposition of sediment on the screen was significant after 2 to 3 years of placement, and this permitted regrowth of the plants. Mayer found that autumnal removal of the screens and repositioning in the spring maintained 95-percent control of nuisance weeds.

Perkins, Boston, and Curren (1980) examined the relationship between coverage time by Aquascreen and control of Eurasian watermilfoil. Panels that measured 30 by 80 ft (9 by 24 m) were set out in shallow (0.5 to 2.0 m) and deep (2 to 3 m) areas of Lake Washington, and then removed at 1-, 2-, and 3-month intervals. Compared with control areas, panels in place for 1 month produced 25- and 35-percent decreases of biomass in shallow and deep plots, and plant regrowth was small. Two months of coverage produced 78- and 56-percent decreases in standing crop of plants and little regrowth. Three months of coverage essentially eliminated the plants. Both Perkins, Boston, and Curren (1980) and Engel (1982, 1984) found that Aquascreen was most effective when applied tightly to the sediment surface. Growth will occur under loosely applied screen. Engel emphasizes that screens left in position for a second season will have plant growth on them. The screens should be removed, cleaned, and repositioned. Similar conclusions were made about Aquascreen by Lewis, Wile, and Painter (1983), following experiments in an Ontario lake. Newroth and Trvelson (1984) reported an average period of plant control in British Columbia lakes to be 2 years. The screen was found to allow some plants to grow through it, while fragments of some plants were able to root through it.

Newroth and Trvelson (1984) have found that vinyl-coated window screen is a satisfactory and less expensive substitute for Aquascreen.

Dartek

Dartek is a black-pigmented, impermeable nylon sheeting material (DuPont, Canada) with a specific gravity greater than 1.0. It is sold in 2.5- by 30.5-m rolls (about \$8,000 ha^{-1}) (1983 prices). Perkins (1984) evaluated the effectiveness of Dartek in Green Lake, Washington, where one roll was placed over soft unconsolidated organic muck that supported a population of *Elodea canadensis*. Four panels of Dartek were placed in Lake Washington over sand-gravel sediments. Sheeting without gas-venting slits and those with diagonal slits lifted off the sediments. Panels that had 12 cross-hatch slits per square meter lifted less than 20 cm. After 35 days, plant decomposition beneath the panels was nearly complete. Perkins concluded that Dartek was "highly effective." Newroth and Trvelson (1984) report that Dartek, on the average, was effective for 2 years.

Polypropylene

Polypropylene is a spun-bonded, woven, semipermeable sheeting with positive buoyancy. It is sold under various trade names, such as Typar (DuPont) and Terratrack (Terratrack Ltd., Rexdale, Ontario). Engel (1982) reports that Typar (1984 price) costs about $8,000 ha^{-1}, plus installation. Terratrack (Lewis, Wile, and Painter 1983) costs about $6,254 ha^{-1}.

Cooke and Gorman (1980) found that Typar, anchored with cement blocks, was completely effective in eliminating rooted plant biomass in the treatment area for one season. No evaluation in subsequent seasons was made. Filamentous algae grew profusely on the surface of the Typar in shallow water. Engel (1984) also found that Typar completely controlled vegetation during the first year. Removal and repositioning were difficult, and plants regrew on Typar after sediments accumulated on the panels. In contrast, Terratrack, anchored with concrete blocks, did not allow Eurasian watermilfoil to reroot on it over three summers, even though fragments were found (Lewis, Boston, and Curren 1983). Newroth and Trvelson (1984) report that the use of sand to anchor Typar provided a substrate for rerooting of plants.

Other materials

The experiences with polyethylene sheeting have not been satisfactory, largely because polyethylene is impermeable and buoyant (Armour, Brown, and Marsden 1979; Nichols 1974; Petersen, Born, and Dunst 1974; Engel 1982; Engel and Nichols 1984). Since polyethylene floats, it is very difficult to apply, and the anchoring must be very secure so that the sheets do not move under the influence of waves. Sand has been used as an anchor, but plants grow on it. Also, large slits must be cut to allow the escape of gases, the material deteriorates in sunlight, and it will slip down steep inclines (Armour, Brown, and Marsden 1979). It is not very expensive, however. Armour, Brown, and Marsden (1979) report a price of about $4,600 ha^{-1} (1984 prices), plus installation.

Hypalon, a synthetic rubber, is effective due to its strength and weight but is impermeable. It is also very costly. In 1984, the inflation-corrected price was $59,270 ha^{-1} (Armour, Brown, and Marsden 1979). Polyvinyl was also found to be effective, in part because it is negatively buoyant. It is impermeable and thus requires venting, and it is expensive (inflation-corrected to 1984, the price was \$22,194 ha^{-1}). Polyvinyl also tended to crumple and was easily dislodged by waves (Armour, Brown, and Marsden 1979).

Burlap (10 oz yd^{-1}) was found by Jones and Cooke (1984) to be effective for one season in an Ohio reservoir, but the material rotted even when treated with a preservative. Despite the permeability of burlap, it ballooned, suggesting that the pores may have been clogged with organic matter and the associated microbial community. Newroth and Trvelson (1984) report control of Eurasian watermilfoil with burlap for two or three growing seasons in British Columbia lakes. Despite its negative buoyancy, burlap must be securely anchored. Cost of burlap according to Jones and Cooke (1984) is about \$3,400 ha^{-1}.

Sediment covers, while very effective in eliminating problems with nuisance rooted aquatic plants, cannot usually be used over large areas due to costs. A 30- to 40-ha treatment with Dartek would cost over \$400,000 plus installation. However, smaller areas, such as docks and marinas, could be treated at a much lower cost. In most cases, this would be an appropriate use of sediment covers.

Reservoirs often receive substantial silt loads, and waves may further increase the quantity of suspended solids that could settle on the covers. This problem could limit the longevity of control by sediment covers since plant fragments may root in the deposited silt. Newroth and Trvelson (1984) report that a fabric called Texel, a negatively buoyant, needle-punched, polyester fabric, may be particularly applicable to this situation because fragments of plants appear to have difficulty in attaching to the upper surface and in penetrating the fabric. In many cases, the reservoir manager will have to be aware that sediment screens will require periodic cleaning and repositioning to prevent new plant growth.

Several candidate materials are available, as reviewed earlier. Feasibility in reservoirs depends not only costs and the possibility of plant regrowth on deposited sediments, but also the effectiveness and ease of application. An onsite evaluation of this should be conducted prior to full-scale application to a particular area. Ease of application, longevity of materials, and duration of control should be evaluated for several types of sediment covers.

Limitations and Concerns

While little study has been made of their impacts, a few negative impacts of the use of sediment covers have been reported to date. Engel (1982, 1984) found that macroinvertebrates were eliminated by Aquascreen in Cox Hollow Lake, Wisconsin, apparently due to low dissolved oxygen. Boston and Perkins (1982), however, reported that plant death following Aquascreen application was slow enough that they believed deoxygenation would be a minor problem. Additional studies are needed.

Applications of sediment covers to eliminate rooted plants at swimming beaches, while completely successful in meeting this objective, have produced complaints by swimmers about walking on the screens.

Summary

Table 16 is a summary of the properties, costs, and effectiveness of seven materials that have been used as sediment covers. Table 17 is a summary of this reservoir management technique.

Table 16

Summary of Features of Sediment Covering Materials

Material	Specific Gravity	Weatherability	Cost $/ha	Application Difficulty	Permeability to Gases	Comments
Black polyethylene	0.95	Poor	4,600	High	Impermeable	Easily dislodged, vents permit plant growth, "balloons"
Hypalon	>1.0	Excellent	59,270	Moderate	Impermeable	Strong, effective, "balloons"
Polyvinyl	1.2-1.5	Good	22,194	Moderate	Impermeable	Easily dislodged, crumples, effective
Polypropyl (Typar)	0.90	Good	8,000	Low	Permeable	Effective
Aquascreen	2.54	Good	21,500	Low	Permeable	Highly effective, easily removed and repositioned
Burlap	>1.0	Poor	3,400	Low	Permeable	Effective, "balloons," rots when placed on highly organic muds
Dartek	>1.0	Fair	8,000	Moderate	Impermeable	Effective, "balloons"

Table 17

A Summary of Sediment Covers

Characteristic	Description
Target	Nuisance rooted plants.
Mode of action	Prevents plant growth by a physical barrier over reservoir sediments.
Effectiveness	Effective; problem often eliminated.
Longevity	Months to years.
Negative features	Elimination of habitat. Installation may be difficult and costly. May float to surface and clog intake structures. May annoy waders at swim beaches.
Costs	Ranges from \$3,400 to \$60,000 ha^{-1}, plus installation, depending upon material selected.
Applicability to reservoirs	Suitable for eliminating plants in selected areas such as marinas, swimming areas, and docks.

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PART XIII: HERBICIDES AND ALGICIDES

Problems Addressed

Excessive biomass of rooted aquatic plants can interfere with recreational uses of the water and may produce organic substances in the water, which lead to dissolved oxygen depletions or to problems with taste, odor, and production of trihalomethanes in potable water plants. Nuisance "blooms" of algae, often blue-green algae, can also produce the same problems.

Herbicide and/or algicide applications are used to kill the plants, and together represent one of the most commonly used means of managing the symptoms of eutrophication. There are few modern scientific reviews of their use. However, the reader will find the following useful: Brooker and Edwards (1975); Kearney and Kaufman (1975); Robson and Barrett (1977); Brown (1978); Janik, Taylor, and Barko (1980); Hanson and Stefan (1984); Westerdahl and Getsinger (1988).

Theory and Design

Herbicides and algicides are applied either as liquids or granules to nuisance plant populations at concentrations known to be sufficient to produce elimination or control of biomass, usually through interference with plant cell metabolism. In some instances, chemicals are used together (e.g., diquat and copper) for controlling macrophyte and algal problems simultaneously, or chemicals may be used in conjunction with other procedures such as harvesting, grass carp, or insects.

Until the last 15 years, the use of chemicals to alleviate the symptoms of eutrophication was essentially the only effective method available to improve or open reservoirs and waterways. Today, other methods that are restorative or protective in nature, or with less environmental impact, have often been substituted for them, especially where longer term control is desired. These methods have been described in earlier parts of the report and in Cooke et al. (1986). In many instances, however, particularly in densely infested southern waters, herbicides remain as one of the more cost-effective management options. When the causes of excessive algae or macrophyte growth cannot be remedied, or while nutrient control or other restorative procedures are being implemented, there may be no other option than the use of algicidal or herbicidal chemicals.

The chemical and physical properties of herbicides are described by Beste (1983), including a list of manufacturers who will provide data on application, dose, target species, and user precautions. A handbook entitled "Aquatic Weeds: Their Identification and Methods of Control" has been published by the Illinois Department of Conservation, Springfield, IL. This and similar reports prepared by other states can be useful in calculating doses and in determining the identity of nuisance plants. Reservoir managers should be aware that a commercial applicator's license, along with liability insurance, is required when the work is contracted. Prospective users of these chemicals should consult the appropriate state regulatory agency to obtain a list of approved chemicals for that state. Assistance in determining which chemicals are approved for use cannot be obtained from the Office of Pesticide Programs of the US Environmental Protection Agency at this time.

Effectiveness and Feasibility

The following paragraphs briefly review the most commonly used chemicals with regard to effectiveness, persistence, toxicity to nontarget organisms, and other factors.

Copper-containing compounds

Copper has been used as an algicide throughout this century to produce short-term relief from nuisance algal blooms. More recently, chelated copper compounds have been used to keep the copper concentration in the water column high for longer periods. Usually a dose of 1 to 2 ppm (0.8 mg Cu ℓ^{-1}) provides significant algal control, apparently through inhibition of photosynthesis and nitrogen fixation, for a period ranging up to 10 days.

Unless nutrient input and water column nutrient concentration of a eutrophic reservoir are significantly reduced, or zooplankton grazing on algae sharply enhanced, problems with nuisance algal blooms will likely continue. The use of copper or other algicides such as simazine may be the only feasible choice for control of an algal bloom if nutrient reduction does not occur.

The use of copper results in several significant environmental impacts. A massive algal kill, such as could occur if too great an area is treated per day, may produce a large decline in epilimnetic dissolved oxygen as the cells decay. Hypolimnetic dissolved oxygen may fall sharply. Copper is a highly toxic metal to animal groups, especially in soft water of low organic content (Hodson, Borgmann, and Shear 1979). Some fish, particularly salmonids and walleye, are very sensitive to copper, and effects will appear at concentrations between 10 and 20 μg Cu ℓ^{-1}. Bluegills (*Lepomis macrochirus*) will exhibit sublethal effects at 40 to 160 μg Cu ℓ^{-1}, and perch (*Perca flavescens*) at 40 μg Cu ℓ^{-1}. Significant tissue accumulation of copper occurs in bluegills exposed to between 40 and 160 μg Cu ℓ^{-1} in soft water (45 mg ℓ^{-1} as $CaCO_3$). Persistent use of copper could produce extinctions of fish species in a particular reservoir due to effects on reproduction and larval stages. Some fish food organisms, including *Daphnia* (Cladocera), *Gammarus* (Amphipoda), *Physa* (Mollusca), and some insects, are also extremely sensitive to copper (Benoit 1975; Birge and Black 1979; Hodson, Borgmann, and Shear 1979; Ingersoll and Winner 1982; Harrison, Knezovich, and Rice 1984; Collvin 1985; Blaylock, Frank, and McCarthy 1985). The elimination or reduction in population density of algae-grazing zooplankton, such as *Daphnia pulex*, may account for the rebound in phytoplankton density that can occur after a copper application. It should be noted that the use of chelated copper in combination with the herbicide diquat, a treatment technique to attempt to control both rooted plants and algae at once, greatly enhances the chemicals' toxicity to trout (Simonin and Skea 1977).

Ingestion of copper in potable water apparently will not produce copper toxicosis in humans (Scheinberg 1979).

Diquat

Diquat, a bipyridilium salt, is known to be an effective herbicide, but is quickly deactivated in turbid water through sorption to inorganic particles. Even a rainstorm that introduces turbid water shortly after application may reduce effectiveness. Diquat persistence of up to 160 days in the mud has been noted by Frank and Comes (1967), and Birmingham and Colman (1983) have found that chronic applications could lead to sufficient desorption of "loosely bound" diquat from sediments to produce phytotoxic conditions in the water. The duration of its effectiveness ranges from weeks to an entire summer (Blackburn and Weldon 1964, Schenk and Jarolimek 1966). Apparently its mode of action is to inhibit photosynthesis and stimulate respiration (Funderburk and Lawrence 1964). The recommended dose ranges from 0.5 ppm for

Potamogeton to 1.0 ppm for *Myriophyllum spicatum, Ceratophyllum demersum*, and *Najas flexilis*.

As noted earlier, the combined use of diquat and copper is particularly toxic to brown trout (*Salmo trutta*) and also to some invertebrate animals of the fish food web (May, Hestand, and Van Dyke 1973; Simonin and Skea 1977). Crustacea, a group of animals of great significance to fish diets, are extremely sensitive to diquat at levels well below those needed for plant treatment (Wilson and Bond 1969; Nicholson and Clerman 1974; Storch and Winter 1983; Williams, Mather, and Carter 1984). Other invertebrate animals appear to survive doses well in excess of those that would be encountered during reservoir treatment (Naqvi, Leung, and Naqvi 1980; Marshall 1984). Bluegills tolerate high levels of diquat (Surber and Pickering 1962), but perch and rainbow trout (*Salmo gairdneri*) exhibit sublethal effects that would interfere with successful spawning (Bimber, Boening, and Sharma 1976; Dodson and Mayfield 1979). The latter study has shown that rainbow trout accumulate significant amounts of diquat in their tissues, although the study did not specify whether these were edible or nonedible tissues.

Endothall

Endothall is produced in several formulations, including the liquid (Aquathol K) and granular (Aquathol) dipotassium salt and the di(N,N-dimethylalkylamine) salt (Hydrothol 191) in liquid and granular forms. Endothall acts on plant tissues to produce abnormal permeability, loss of water, and wilting (Keckemet 1969). Susceptible plants may be controlled for weeks to months. For example, Rodgers, Reinert, and Henman (1984) found that it took 2 months before *M. spicatum* began to reinvade after treatment. Doses of endothall to target macrophytes range from 1 to 3 ppm, depending on target species. The use of endothall has been reviewed by Armstrong (1974).

Both the mono and di(N,N-dimethylalkylamine) salts have been found to be very toxic to some fish at concentrations below those needed for plant control (Walker 1963, 1964; Finlayson 1980). Armstrong (1974) reports that rainbow trout, chinook salmon (*Oncorhynchus tshauytscha*), mud minnows (*Pimephales promelas*), largemouth bass (*Micropterus salmoides*), and bluegills will tolerate disodium endothall concentrations from 10 to 100 times the concentration recommended for plant treatment. The disodium and potassium salts apparently persist in water for periods ranging from 2 to 46 days (Hiltibran 1962, Yeo 1970, Simsiman and Chesters 1975, Holmberg and Lee 1976, Serns 1977).

2,4-D

The herbicide 2,4-D is a phenoxyacetic acid. It is available as the acid or in salt or ester form. The herbicide must be absorbed by plant tissue to be effective. The sodium and potassium salts penetrate poorly, the ammonia and amine salts somewhat better, and the hydrolyzed ester (e.g., butoxyethanol ester, BEE) is readily absorbed and translocated. The action of 2,4-D in the plant is unclear except that it behaves like an auxin (Loos 1975, Westerdahl and Hall 1983).

The herbicide 2,4-D is particularly effective against *Myriophyllum spicatum*. Gangstad (1982) considers 2,4-D to be the most effective and economical treatment of this plant. Doses of 20 to 40 kg acid equivalent/ha, normally of 2,4-D DMA (dimethylamine salt) or 2,4-D BEE, are usual. Goldsby, Bates, and Stanley (1978) have reported that 2,4-D is effective against milfoil, particularly when combined with water level drawdown. Adams (1983) cautions that root contact is essential for long-term milfoil control and that the use of granular formulations can be ineffective if the pellets become trapped and suspended in the foliage. If an early season application is not possible, then the use of a harvester prior to application will allow the herbicide good contact with the roots. Adams (1983) also recommends that the normal dose of 22 kg active ingredient/ha of the granular formulation of 2,4-D BEE (Aqua-Kleen) be doubled for dense infestations in deep water (>2.4 m) where there is a high water turnover rate. Otherwise, dilution will be too great to produce control.

Depending on the dose and degree of infestation prior to treatment, 2,4-D remains effective against milfoil for at least the season of application, and often longer (Smith 1971; Aiken, Newroth, and Wile 1979; Getsinger, Davis, and Brinson 1982; Adams 1983). Other plants may not be adequately controlled. Controlled-release formulations may provide even longer control of milfoil (Van, Steward, and Jones 1986). Pierce (1960) found that species of *Potamogeton* returned in 1 month and grew heavily, while *Utricularia* was unaffected at doses up to 6 ppm. Adams (1983) also reported an invasion of *Potamogeton* after elimination of milfoil.

Although 2,4-D appears to have a short persistence in the water column, it can be detected in mud samples for months (Faust and Aly 1964; Smith and

Isom 1967; Cope, Wood, and Wallen 1970; Adams 1983; Birmingham and Colman 1985). Degradation is far slower in anaerobic sediments than in aerobic (DeLaune and Salinas 1985).

At the concentrations achieved with the usual dose of 2,4-D, there is little evidence of toxicity to fish and invertebrates, with some significant exceptions (Smith and Isom 1967, Vardia and Durve 1981, Couch and Nelson 1982, Adams 1983). Low doses of 2,4-D BEE are toxic to developmental and juvenile stages of sockeye salmon (*Oncorynchus nerka*), chinook salmon, and rainbow trout, according to McBride, Dye, and Donaldson (1981) and Finlayson and Verrue (1985). Cope, Wood, and Wallen (1970) found that the propylene glycol butyl ether ester produced lesions and liver, blood, and central nervous system abnormalies in bluegills. It apparently does not bioaccumulate to significant levels in tissues of bluegills, channel catfish, or largemouth bass (Schultz 1973). Areas within 0.5 mile (0.8 km) of a potable water intake cannot be treated with 2,4-D. Moreover, it can be used only for Eurasian watermilfoil control in Tennessee Valley Authority reservoirs unless specifically approved by EPA under Section 18 or 24C of FIFRA.

Mullison (1981) concluded that there is a wide margin of safety for humans with respect to the use of 2,4-D. However, a case study (Colton 1986, Hoar et al. 1986) provides evidence for increased risk of non-Hodgkin's lymphoma among men exposed to 2,4-D for more than 20 days per year. This finding suggests that 2,4-D should not be used in potable water and that applicators should use every precaution to avoid exposure.

Fluridone

Fluridone is registered under the trade name SONAR and is sold as an aquatic suspension or as pellets. A review of its mode of action, effectiveness, dose, and environmental impacts has been provided by Schmitz (1986). Fluridone is a slow-acting, rapidly degradable herbicide that is very effective against a broad spectrum of submersed and emergent aquatic plants (Table 18). A more complete list of plants controlled by fluridone can be found in Schmitz (1986) and Westerdahl and Getsinger (1988). Its action is to inhibit synthesis of plant pigments which protect chlorophyll from photodegradation. The normal dose for reservoirs with a mean depth greater than 4 m is 2.2 to 4.5 kg (active ingredient)/ha, and treatments are most effective when applied prior to rapid plant growth. Treatments when plants are visible

Table 18

Common Aquatic Weed Species and Their Responses to Herbicides*

Plant	Diquat	Endothall	2,4-D	Glyphosate	Fluridone
Emergent species					
Alternanthera philoxeroides (alligatorweed)			YES	YES	YES
Dianthera americana (water willow)			YES		
Glyceria borealis (mannagrass)	YES	NO	NO		
Phragmites spp. (reed)				YES	
Ranunculus spp. (buttercup)			YES	YES	
Sagittaria sp. (arrowhead)	NO	NO	YES		YES
Scirpus spp. (bulrush)	NO	NO	YES	YES	YES
Typha spp. (cattail)	YES	NO	YES	YES	YES
Submersed species					
Ceratophyllum demersum (coontail)	YES	YES	YES	NO	YES
Chara spp. (stonewort)	NO(1)	NO(1)	NO(1)	NO(1)	NO(1)
Elodea spp. (elodea)	YES	?	NO	NO	YES
Hydrilla verticillata (hydrilla)	YES(2)	YES	NO	NO	YES
Myriophyllum spicatum (milfoil)	YES	YES	YES	NO	YES
Najas flexilis (naiad)	YES	YES	NO	NO	YES
Najas guadalupensis (southern naiad)	YES	YES	?	NO	YES
Potamogeton amplifolius (large-leafed pondweed)	?	YES	NO	NO	
P. crispus (curly-leafed pondweed)	YES	YES	NO	NO	
P. diversifolius (waterthread)	?	YES	NO	NO	
P. natans (floating leaf pondweed)	YES	YES	YES	NO	YES
P. pectinatus (sago pondweed)	YES	YES	NO	NO	YES

(Continued)

Note: Data based on Arnold 1979, McCowen et al. 1979, Pennwalt Corporation 1984, Monsanto Company 1985, Nichols 1986, Schmitz 1986, Westerdahl and Getsinger 1988.

* Abbreviations are defined as follows: YES = controlled, NO = not controlled, (1) = controlled by copper sulfate, (2) = plus chelated copper sulfate, and ? = possible control. No entry indicates data unavailable.

Table 18 (Concluded)

Plant	Diquat	Endothall	2,4-D	Glyphosate	Fluridone
Submersed species (Continued)					
P. illinoiensis (Illinois pondweed)				NO	YES
Floating species					
Brasenia schreberi (watershield)	NO	?	YES	NO	NO
Eichhorniae crassipes (waterhyacinth)	YES(2)		YES	YES	NO
Lemna minor (duckweed)	YES	NO	YES	NO	YES
Nelumbo lutea (American lotus)	NO	?	YES	YES	NO
Nuphar spp. (cowlily)	NO	?	YES	YES	?
Nymphaea spp. (waterlily)	NO	?	YES	YES	?

in spring-summer are also effective. Fluridone acts slowly, and 30 to 90 days may be required to establish plant control under optimum conditions.

Fluridone appears to have a very low toxicity to fish and invertebrates and does not accumulate in animal tissues. Because it is slow acting, dramatic changes in physicochemical variables, such as dissolved oxygen, are unlikely. Fluridone cannot be applied within 0.25 mile (0.4 km) of a potable water intake. There is no waiting period following application.

<u>Glyphosate</u>

Glyphosate, registered under the trade name RODEO, is used for treatment of emergent vegetation. It is ineffective against submersed plants. Table 18 lists some of the plants known to be controlled. Glyphosate is formulated as a liquid combined with a surfactant, and appears to affect amino acid metabolism in treated plants. Glyphosate is new, and limited data are available in the scientific literature regarding toxicity, bioaccumulation, and persistence. Studies performed by Monsanto Company (1985) indicate that RODEO biodegrades, does not bioaccumulate, and has very low invertebrate and mammalian toxicity. A study by Servisi, Gordon, and Martens (1987) concludes that glyphosate antagonizes the toxicity of its surfactant MONO 818 and that the surfactant is more toxic than the herbicide. Glyphosate cannot be applied within 0.5 mile (0.8 km) of potable water intakes. There is no waiting period for water use after application.

Table 18 summarizes the effectiveness of commonly used herbicides against some aquatic plants. Westerdahl and Getsinger (1988) have developed a thorough guide to the types of aquatic plants that may appear in reservoirs and have listed the herbicides that are effective against them. Readers contemplating a herbicide application should examine this report.

Costs

Costs of herbicide treatments range widely. Plant density, area to be treated, types of plants, and other factors will influence cost greatly. Table 12 (Part X) is a comparison of cost ranges for harvesting and herbicides. In the Midwest, cost ranges are clearly comparable (about \$350-\$900 ha^{-1}) for these two techniques and will be affected by some of the local factors listed above. In Florida (and other areas with dense, rapidly growing populations of exotic plants such as waterhyacinth), herbicide treatments are usually less costly than harvesting. In some southern waters, harvesting cannot keep up with plant growth and becomes a continuous operation, whereas herbicide applications may be sufficient to manage the problem in one or a few applications.

Limitations and Concerns

Brooker and Edwards (1985) point out that most discussions of the negative effects of algicides and herbicides deal only with direct toxic effects to selected species. Some of these effects have been briefly outlined in a previous section. While these reports provide some useful information, they are not very informative about the effects of the addition of toxic materials to reservoirs. Reservoirs are complex units of interacting biological, chemical, and physical components called ecosystems. The ecosystem is the actual level of biological organization to which herbicides and algicides are applied, not the species level. There are very few studies about the effects of these chemicals on reservoir or lake ecosystems. Some of these studies are briefly outlined here, based upon the list of concerns in Conyers and Cooke (1982, 1983) and Cooke (1983).

Hanson and Stefan (1984), in one of the few long-range ecosystem-level studies of the impact of chemical treatments on an aquatic system, found both short- and long-term impacts. Copper sulfate applications over a period of 58 years were effective in providing temporary, short-term (days) control of algae, but produced oxygen depletions and increased phosphorus cycling and occasional fish kills from copper toxicity and low dissolved oxygen. The longer term effects over the years included copper accumulation in sediments, increased tolerance by some algae to copper sulfate, a shift in species composition from green to noxious blue-green algae and from game fish to rough fish, disappearance of macrophytes, and reductions in benthic macroinvertebrates. Hanson and Stefan (1984) conclude that the short-term gain of expedient and brief algal control is essentially traded for long-term degradation of the lake. However, the use of chelated copper compounds, in place of copper sulfate, alone and in combination with other herbicides is EPA-approved. Moreover, previous problems with copper toxicity attributable to copper sulfate are not apparent with the chelated copper compounds.

When plants are killed with chemicals, their biomass, including the plant nutrients contained therein, is left in place to decompose. Many authors, among them Walker (1964), Nichols and Keeney (1973), Carter and Hestand (1977), Carpenter and Adams (1978), Hestand and Carter (1978), Myers (1979), Peverly and Johnson (1979), Morris and Jarman (1981), Wingfield and Johnson (1981), Getsinger, Davis, and Brinson (1982), James (1984), and Goldsborough and Robinson (1985), have observed a pulse of nutrients, and in some cases a loss of dissolved oxygen and a phytoplankton bloom, following a herbicide treatment. Thus, one problem, excessive macrophytes, is replaced by one or more other severe problems (oxygen depletion, algal bloom). However, these side effects can often be lessened or avoided through careful planning and proper use of chemicals. Ways to prevent these effects include chemical application in spring before biomass develops, staged applications so that limited areas are treated each day until the entire target area receives an application, use of a harvester for plant biomass removal prior to application, or installation of aeration/circulation devices.

A common occurrence in herbicide use is the replacement of the nuisance target plant with another species that is unaffected by the chemical. The alga *Chara* and the rooted plant *Potamogeton* are often replacement species. Reiser (1976), Conyers and Cooke (1982, 1983), Conyers (1983), Cooke (1983), and Richard, Small, and Osborne (1984) describe case histories of this response to herbicides. Nichols (1986) provides a useful list of common aquatic nuisance plants and their responses to endothall, diquat, 2,4-D, and water level drawdown.

Another potential negative effect of herbicide use is the waiting period (3 to 10 days, depending upon chemical) between application and water use. Diquat, endothall, and 2,4-D all have waiting periods.

Summary

Herbicide and algicide applications for the control of nuisance macrophyte and algal problems are widely used methods of reservoir management. These chemicals provide an expedient and often highly effective means of producing at least short-term control of problem species. However, when one nuisance species is controlled, another species may take its place.

It is important to note that the use of registered herbicides may be the only feasible solution in many instances.

Table 19 is a summary of the use of herbicides and algicides.

Table 19

Summary of Algicides and Herbicides

Characteristic	Description
Target	Excessive algal and/or macrophyte biomass.
Mode of action	Toxic to plants.
Effectiveness	Highly effective against susceptible species.
Longevity	Algicides are effective for days.
	Some herbicides provide at least seasonal control.
Negative features	Decay of plants may produce a temporary dissolved oxygen depletion, and release of nutrients may stimulate an algal bloom.
	Some chemicals are toxic to fish and fish food organisms.
	Long-term effects on ecosystems remain unclear.
Costs	Costs range from about \$430 to \$900 ha^{-1} (\$175 to \$370 $acre^{-1}$).
Applicability to reservoirs	Their use may be the only feasible option; however, use in potable waters is restricted, except for copper, or restricted to use at some specified distance from potable water intakes. For some herbicides, a waiting period is designated following treatment.

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PART XIV: SUMMARY

Since the water quality of lakes and reservoirs is determined to a great extent by the quality and quantity of influent materials, the logical first step in water quality management is the control of these material inputs. However, unlike most natural lakes, many reservoirs are large, complex, and receive water and material loads from relatively large tributaries draining extensive watersheds. Since these watersheds are so large and often cross administrative and/or regulatory boundaries, measures to control material inputs are difficult to implement.

When the inputs of materials, particularly sediments and nutrients, become excessive, problem conditions occur. These include loss of storage, excessive algal and macrophyte production, poor water clarity, reduced dissolved oxygen concentrations in bottom waters, and an impaired fishery. The occurrence of problem conditions in the reservoir often leads to problems in the tailwater. The release of water low in dissolved oxygen and high in concentrations of reduced materials threatens fisheries and impacts downstream waterbodies. Such problems diminish project benefits.

Water quality management techniques aimed at control of in-reservoir water quality processes provide potentially valuable tools to the reservoir manager. Targeted by these techniques are such processes as the release or recycling of nutrients stored in sediments or biomass, the growth and reproduction of aquatic plants, the decomposition of organic material and the resultant utilization of oxygen, and the accumulation of sediment. In general, many of the techniques discussed here attempt to reduce the severity of undesirable water quality responses rather than to eliminate or modify the cause of the response. For instance, hypolimnetic aeration provides a means to replenish hypolimnetic dissolved oxygen stores reduced by the decomposition of excessive autochthonous or allochthonous inputs to bottom waters.

Given the difficulty in addressing the direct causes of reservoir water quality problems, successful reservoir water quality management will require careful evaluation of the importance of various chemical, physical, and biological processes. Management goals and alternatives will have to be established and evaluated based on a sound understanding of these processes. The methods chosen must be realistic from a scientific and engineering standpoint, cost-effective, and offer the desired degree of control. Often, this will require the use of more than one technique. Consideration should also be given to localized treatment of problem conditions.

This report provides water quality management personnel information concerning a wide variety of management techniques that have been used successfully in lakes or reservoirs. The information presented will serve as a general guide, and the reader is urged to seek additional, more detailed information. Sources of this information are referenced for each technique.