A CRITICAL ASSESSMENT OF THE INFLUENCE OF MANAGEMENT PRACTICES ON WATER QUALITY, WATER TREATMENT, AND SPORT FISHING IN MULTIPURPOSE RESERVOIRS IN KANSAS

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The University of Kansas
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AND SPORT FISHING IN MULTIPURPOSE RESERVOIRS IN KANSAS

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ABSTRACT

Many cities in Kansas, especially in the eastern portion of the state, rely upon impounded surface-water supplies for their drinking water. Many of the reservoirs are either multipurpose reservoirs or under pressure to become multipurpose reservoirs, accommodating water-based recreation and, in particular, sport fishing. Traditionally, these reservoirs have been managed primarily to protect the quality of drinking water, with relatively little regard for fish production, but there is now an increasing desire to include enhancement of sport fishing as a major management objective. Unfortunately, protection of drinking water quality and enhancement of sport fishing are not completely compatible objectives, although there are major areas of agreement and areas where some compromise is appropriate. To satisfy both objectives, it is necessary for reservoir managers to have a clear understanding of the potential impacts of their actions on water quality, water treatment, and sport fishing.

This report examines the characteristics of reservoirs and sport fish in Kansas and then critically examines an array of management practices, assessing the potential impacts of each practice on raw and treated water quality, on the cost and difficulty of water treatment, and on sport fishing. The applicability of existing mathematical models to management of Kansas reservoirs is examined; and the report itself represents a conceptual model of the physical, chemical, and ecological functioning of Kansas reservoirs, providing a foundation for development of a regional reservoir management model. Critical research needs are identified, and some suggestions are made as to possible experimental approaches.

KEYWORDS: Fish Management, Kansas, Multipurpose Reservoirs, Water Quality Control
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CHAPTER 1

INTRODUCTION

A. BACKGROUND

Nearly one hundred reservoirs* in Kansas, most of them in the eastern portion of the state, serve as public water supplies; and more reservoirs are likely to be constructed as the demand for water increases. Virtually all of these reservoirs are used for recreational purposes of one sort or another, but many of them, particularly the smaller ones, are managed as single-purpose water supply reservoirs. The growing public demand for sport fishing, and for water-based recreation in general, dictates that planners give serious consideration to construction of multipurpose reservoirs rather than single-purpose water supply reservoirs. Furthermore, there is growing pressure on decision makers to convert existing single-purpose water supply reservoirs into multi-purpose reservoirs accommodating not only water supply but also sport fishing and other forms of recreation, such as boating, skiing, and swimming (Miller, 1985).

In the State of Kansas, state and local water supply reservoirs are designed and managed under guidelines established by the Kansas Department of Health and Environment (KDHE) to maintain the quality of potable water supplies and to minimize treatment costs. Most state and county fishing reservoirs (commonly referred to as lakes) are managed by the Kansas Fish and Game Commission (KFGC) using techniques intended to increase the size and number of sport fish. When a reservoir is to be used for both water supply and fishing, conflict can arise between the two agencies due to concern that fisheries management techniques will degrade water quality or that water quality management practices will interfere with sport fishing. Indeed, since it is generally true that enrichment (eutrophication) causes a deterioration of drinking water quality and an increase in fish yield (see Chapter 2), the management objectives for water supplies and fisheries are, to a certain extent, fundamentally incompatible (Wagner & Oglesby, 1984). Nevertheless, there are areas of agreement (e.g., severe eutrophication is detrimental to both public water use and sport fishing); and there is room for compromise on certain issues.

KDHE and KFGC recently signed a "Memorandum of Understanding," agreeing to collaborate in the design and management of multipurpose reservoirs; but they are in need of mutually agreed upon decision-making criteria. Since maintaining the quality of drinking water and increasing opportunities for sport fishing are both worthy objectives, the management techniques for both purposes need to be critically evaluated to determine which techniques are compatible with both objectives under a given set of circumstances.

*The term "reservoir" is used herein to describe both natural lakes and man-made impoundments larger than the relatively small "farm ponds" that cover only a few acres. There are few true lakes (i.e., natural lakes) in Kansas (the major exception being oxbow lakes), but many of the state's artificial reservoirs are commonly referred to as lakes.
B. RESEARCH OBJECTIVES AND SCOPE

The primary objective of this research effort was to critically evaluate the impacts of reservoir management practices on water quality, water treatment, and sport fishing in Kansas, for the purpose of providing information useful to KDHE and KFGC (and their counterparts in other states) as they work together to develop design criteria and management schemes for multipurpose reservoirs. More specifically, the objectives of the research were:

1. To assess the impact of fisheries management practices (such as habitat modification, stocking, selective poisoning, drawdown, fertilization, and aquatic weed control) on sport fish, on the physical, chemical, and biological quality of raw and treated water supplies, and on water treatment;

2. To assess the impact of water quality management practices (such as clearing of impoundments, algae control, selective withdrawal, elimination of upstream nutrient and suspended solids discharges, destratification, and weed control) on sport fishing, on the physical, chemical, and biological quality of raw and treated drinking water, and on water treatment;

3. To develop a conceptual reservoir management model useful for predicting the results of management practices on typical reservoirs in Kansas; and

4. To identify areas where further research is needed, clearly defining the critical questions and suggesting possible experimental approaches.

This report does not specifically address farm ponds or county fishing reservoirs, since these are seldom used for potable water supply, nor does it attempt to tackle other uses of multipurpose reservoirs, such as water skiing, swimming, irrigation, flood control, and maintenance of in-stream flow needs. The focus of this report is on the scientific and technological aspects of reservoir management, with little or no attention given to related political, legal, and social issues.

C. RESEARCH APPROACH

The research effort consisted of four major sequential tasks: 1) discussions with KDHE and KFGC; 2) a comprehensive literature review; 3) critical evaluation of reservoir management practices; and 4) development of a conceptual reservoir management model.

First, the nature, scope, and objectives of the project were discussed with representatives from KDHE and KFGC and their input was solicited. They were asked to identify the reservoir management practices commonly used in
Kansas as well as those which might be used, and they were also questioned concerning the availability and usefulness of historical data. KDHE has extensive data pertaining to the quality of water withdrawn from water supply reservoirs, data which might be quite useful in assessing the success of design guidelines in controlling water quality, but neither KDHE nor KFGC has much data from controlled studies designed to evaluate specific management practices. Furthermore, there is virtually no historical data base pertaining to the concentration of organic matter in Kansas reservoirs; however, a study is currently being conducted by the USGS (in cooperation with KDHE) to examine the levels of organic carbon and trihalomethane precursors in a number of water supply reservoirs. Under these circumstances, it was considered most appropriate to concentrate the research effort on published literature and not to review the historical data until the utility and purpose of doing so could be more clearly established.

The second task was to conduct a comprehensive (but not exhaustive) review of the literature. Eighteen technical journals considered most likely to contain relevant information were surveyed issue by issue going back ten years. Furthermore, each useful article found was examined for references to other publications. In addition, numerous topical searches were made using the past ten years' issues of Water Resources Abstracts. This report does not cite all of the relevant publications found, since it is not intended to be an exhaustive literature review; rather, selected references are cited at appropriate points in the discussion to aid the reader seeking additional information.

The third task was to critically evaluate each potentially useful management practice to determine its positive and negative impacts upon water quality, water treatment, and sport fishing and to identify areas where future research might prove helpful. In Chapters 4 through 16, various management practices are discussed, and included in each chapter is an assessment of the applicability of the practice to reservoirs in Kansas. Chapters 2 and 3 describe the general characteristics of reservoirs and sport fish in Kansas, and provide other background information helpful in assessing the impacts of management strategies on reservoirs in Kansas.

The fourth task was to develop a conceptual model that could be used to predict, qualitatively, the impacts of management practices on water quality and sport fishing. This report constitutes such a conceptual model, and provides a foundation upon which a computer-based model applicable to Kansas reservoirs can be built. Unfortunately, there are so many complexities and unknown quantities involved in reservoir management that future development of a reliable and useful computer-based model, capable of predicting both the short-term and the long-term effects (physical, chemical, and biological) of any given management practice, appears to be a rather formidable task. Existing mathematical models and their limitations are discussed in Chapter 17.
CHAPTER 2
WATER SUPPLY RESERVOIRS IN KANSAS

A. GENERAL CHARACTERISTICS

Records obtained from KDHE list 248 reservoirs in the State of Kansas, most of which are located in the eastern third of the state. Of these 248 reservoirs, 93 provide water for domestic consumption. Some form of recreational activity is permitted at all of the reservoirs used for water supply and some are also used for other purposes, including flood control and irrigation (Table 1). These reservoirs pose certain management problems as a function of particular characteristics related to their origin, location, use, and size.

**TABLE 1**
SIZE DISTRIBUTION AND USES OF WATER SUPPLY RESERVOIRS IN KANSAS

<table>
<thead>
<tr>
<th>Size</th>
<th>Total Number</th>
<th>No. Used For Flood Control</th>
<th>No. Used For Irrigation</th>
</tr>
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<tbody>
<tr>
<td>Unknown</td>
<td>9</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>5-50</td>
<td>27</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>50-500</td>
<td>30</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>500-1000</td>
<td>3</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>1000-5000</td>
<td>11</td>
<td>10</td>
<td>4</td>
</tr>
<tr>
<td>&gt;5000</td>
<td>13</td>
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Most reservoirs in Kansas are man-made and are therefore of relatively young age compared to natural lakes, which are most often of glacial origin and at least 10,000 years old. They are also significantly different than most natural lakes in regard to their morphology and climatology (O'Brien, 1975; Thornton, 1984); and they are characterized by a single major source of nutrients, rather than the diffuse sources typical of natural lakes (Kennedy, 1984). Man-made reservoirs, due to their elongated and irregular shapes (following old stream channels), often exhibit much greater longitudinal gradients in the concentrations of nutrients, chlorophyll, and suspended solids than do many natural reservoirs (e.g., Kennedy, 1984). There are also significant ecological differences between natural lakes and man-made reservoirs.

O'Brien (1975) conducted an in-depth study of the factors limiting primary productivity in Kansas reservoirs. Reservoir size, orientation, and morphometry were found to be important factors. Large reservoirs (i.e., those greater than 1000 acres in size) were usually unstratified and very
turbid. The turbidity is a consequence of erosion and agricultural activities, as well as resuspension of bottom muds by wind-driven currents. Primary productivity in the larger reservoirs was found to generally be light limited. Smaller reservoirs, on the other hand, were usually strongly stratified during the summer months, allowing much of the turbidity to settle to the bottom (into the hypolimnion). Therefore, primary productivity in smaller reservoirs was often nutrient limited during the summer months, with phosphorus being the primary limiting nutrient.

As a consequence of the young age of man-made reservoirs there has been little time for the biota and the physical-chemical environment to establish the higher level of stability generally more characteristic of lakes. Man-made reservoirs are only in the earliest stages of establishing this stability, particularly in Kansas where most have been constructed since 1930. In addition, man-made reservoirs are created in the midst of a stream or river habitat, with flora and fauna characteristic of a lotic (flowing water) environment, and must undergo the transition to a habitat with lentic (still water) flora and fauna. For example, large predator fish (e.g., northern pike, muskellunge, and walleye) are not native to Kansas streams and rivers, and are thus absent from impounded reservoirs unless artificially added. Such fish, native to most natural lakes, provide natural control of forage and rough fish populations, which are controlled in rivers and streams by natural calamities such as droughts and floods. In man-made reservoirs, such natural controls are lacking, so it is common for forage and rough fish to become overabundant.

Another common manifestation of the relative instability of reservoirs compared to lakes is the excessive growth of certain members of the community, such as algae, aquatic flowering plants, or various insects. In young ecosystems, such instability can easily result from imbalances in predator-prey relationships or between organisms and their nutrient resources. Instability may also be manifested in more fluctuating and more unpredictable water quality as indicated by fluctuating levels of turbidity, dissolved oxygen, and taste and odor. It takes nature hundreds or thousands of years to establish a stable lake ecosystem, so one can hardly expect to create a stable ecosystem in a large body of water in a few short years, especially since scientific knowledge of the complex workings of ecosystems is rather limited.

Man-made reservoirs are not only young and inherently unstable, but they have also been constructed in watersheds where natural processes did not produce a lake or could not perpetuate one to the present time. The watershed may thus impose certain stresses opposing the development of stability, such that long-term management becomes necessary. After the reservoir is first impounded, it tends to be highly eutrophic as the flooded soil and vegetation quickly release nutrients. This can soon cause the classical problems associated with eutrophication, such as excessive plant growth, declining water quality, and shifts from sport fish to herbivorous rough fish. These symptoms account for many of the reservoir management practices initiated in young reservoirs. As a man-made reservoir ages, it may in fact become less eutrophic (opposite to the usual trend for natural lakes), but will still remain in the overall condition of eutrophy. This will often be accompanied by declines in sport fish production, leading some
reservoir managers to consider fertilization. Excessive versus insufficient eutrophication is often a management dilemma in Kansas reservoirs, leading to conflict between the opposing interests of high sport fish production and acceptable water quality.

Man-made reservoirs, as a consequence of their watershed characteristics, are subject to large water level fluctuations, since they are impounded streams or rivers. Indeed, they generally exhibit physical and ecological characteristics somewhat in between those of rivers and natural lakes, and usually contain riverine, transitional, and lacustrine zones (Kimmel & Groeger, 1984). Generally, they have a high rate of water exchange (i.e., a short hydraulic retention time), one consequence of which is a greater influence of the watershed on conditions in the reservoir.

Agricultural activities dominate many reservoir watersheds in Kansas and the impacts of agrichemicals and soil erosion can be additional factors requiring management. For example, soils washing into Kansas reservoirs from either agricultural watersheds, or by natural processes, carry a large component of fine clay particles typical of the soils in this area of the Great Plains. With this comes a prolonged suspension of the finest particles imparting considerable turbidity to the water column. This in turn affects water treatment for domestic consumption and controls the natural ecology of the reservoir in several ways. For example, turbidity reduces visibility for sight-feeding organisms, thus affecting predator-prey relationships. Also, by reducing light penetration, turbidity significantly reduces primary productivity (e.g., Hammer & Hergenrader, 1973; O’Brien, 1975), which may improve drinking water quality. Much of the total particulate matter in the water column eventually settles out, making the reservoir a settling basin in the stream or river system and limiting its life span. A natural lake in such a watershed would have filled in and long ago ceased to exist. Therefore, man-made reservoirs may also be managed for life span and for the consequences of their accelerated aging process.

Another characteristic of many water supply reservoirs in Kansas necessitating management is their multiple use. In addition to domestic water supply, reservoirs may be used for flood control, irrigation, and recreation, including fishing, boating, and swimming. Maintaining a reservoir for such multiple purposes can lead to the use of conflicting management practices. For example, as stated earlier, a eutrophic reservoir may be desirable for fish production but may not provide the best water quality for domestic service. For flood control and irrigation a reservoir is constructed in a watershed particularly prone to releasing large volumes of water resulting in a reservoir with large fluctuations in water level, a high water exchange rate, and perhaps very heavy loadings of sediment. Recreation of any sort in and around the reservoir can reduce water quality in various ways, such as increasing erosion, adding pollutants, and removing specific fish species. Management of multipurpose water supply reservoirs must sometimes address conflicting interests to the point where priorities must be determined in order that certain uses do not suffer irrevocably.

Man-made water supply reservoirs have become integrated into life in Kansas. They are not a natural habitat for the State, but are now heavily relied upon. One cannot expect the natural environment to easily accommodate
or support this unnatural habitat any more than one can expect agricultural habitats to be self-sustaining. This is evidenced by the extensive reservoir management programs used in Kansas and elsewhere.

**B. LARGE RESERVOIRS VS. SMALL RESERVOIRS**

The majority of reservoirs in Kansas are smaller than several hundred acres in size, but 24 reservoirs are over 1,000 acres in size, the largest of these being 16,000-acre Milford Lake (see Table 1). There are some important differences between large reservoirs and small reservoirs, differences that may be quite significant in regard to management practices.

Perhaps the most significant difference between large reservoirs and small reservoirs is the influence of the littoral (shoreline) area on the physical, chemical, and biological processes occurring in the reservoir. For reservoirs of similar shape and bottom slope, the littoral area is proportional to the square root of the surface area. Hence, the littoral zone of a small reservoir will occupy a much greater percentage of the reservoir's surface area and volume than the littoral zone of a larger reservoir of the same shape. Therefore, such things as shoreline vegetation, erosion, periphyton growths, overhanging tree foliage, etc., will have a much greater influence on water quality and fish in a small reservoir than in a large reservoir. The same would be true of a reservoir with a very irregular (long) shoreline compared to one of the same size with a more regular shoreline.

There are other important physical differences between small reservoirs and large reservoirs. Small reservoirs are generally shallower and less turbid than large reservoirs, so they receive more sunlight per unit volume than large reservoirs (and are therefore more productive and warmer), and are more heavily populated by benthic organisms and submerged plants. Large reservoirs have longer fetches, and are more effectively mixed than small reservoirs, making them less likely to be strongly stratified and less likely to have anoxic bottom waters.

Their physical characteristics make small reservoirs much more susceptible than large reservoirs to ecological and water quality problems that require management. These problems are compounded by the fact that managers of small reservoirs also tend to have more limited financial resources and less access to expert advice than the managers of larger reservoirs; and the treatment plants associated with small reservoirs are often designed and operated in such a way as to make it very difficult or expensive to address certain water quality problems arising from eutrophication. However, smaller reservoirs are generally associated with smaller and more manageable watersheds and some management practices (such as draining and cleaning) are generally only feasible for small reservoirs.
The primary management objective for water supply reservoirs in Kansas (and elsewhere) is to control the quality of the raw water supply so that the water can be effectively and economically treated to produce a potable (safe) and palatable (aesthetically pleasing) drinking water. This objective is achieved by managing the watershed to restrict the discharge of materials (toxic chemicals, pathogens, nutrients, sediment, etc.) into the reservoir, by restricting use of the reservoir, and by managing the reservoir to protect and improve the quality of the water it holds. Since most of the water quality problems encountered in reservoirs are directly or indirectly related to eutrophication, i.e., to excessive growths of algae and macrophytes (see Chapter 4), eutrophication is universally viewed as undesirable by the managers of water supply reservoirs; and most of the reservoir management practices they use are primarily intended to retard eutrophication or to counteract its effects.

The literature contains an abundance of information on methods to prevent, retard, or remedy eutrophication, including nutrient removal, destratification, selective withdrawal, aquatic weed control, watershed management, algae control, clearing of impoundments, dredging, draining, and others. In Kansas the most commonly used methods include the use of copper compounds to control algal blooms, control of wastewater discharges and other sources of pollutants within the watershed, weed control, and inclusion of water quality considerations in reservoir design and construction. Most other methods are generally viewed as extreme measures, due to their cost or to the uncertainty of success.

Historically, water quality management practices for water supply reservoirs, particularly small reservoirs, have been implemented with little or no regard to their potential impact on fish. There are several reasons for this. First, even in water supply reservoirs where fishing is permitted, the need to control water quality virtually always takes precedence over fishing, which is generally viewed as a non-essential bonus. Second, few managers of small water supply reservoirs have training in biology, so they are generally unable to predict the impact of their actions on fish. Third, much of the research on management of eutrophic reservoirs has been concerned with water quality rather than fish production; and although many of the impacts of water quality management practices on sport fish can be assessed by someone with proper training, these impacts are not always clear to the reservoir manager who is not a fisheries biologist.

Fish production is, in general, closely related to reservoir fertility, as indicated by correlations of fish yield with total phosphorus (Hanson & Leggett, 1982; Lee & Jones, 1984) and with chlorophyll (Jones & Hoyer, 1982; Wagner & Oglesby, 1984). Therefore, water supply managers may be led to presume that enhancement of sport fishing is necessarily accompanied by degradation of water quality. Indeed, there have been reports that total elimination of fish has led to more oligotrophic conditions (Stenson et al., 1977; Henrikson et al., 1980) or great improvements in water clarity (Erickson, 1981). However, such improvements are attributable to the decimation of benthophagous or plantivorous fish and not to elimination of
sport fish, and similar benefits can be realized by restructuring the fish population to include the large predatory fish preferred by fishermen (Lynch, 1981). Furthermore, excessive enrichment can also be a serious fisheries management problem (e.g., Erickson, 1981), since sport fish are adversely affected by low dissolved oxygen concentrations and by algal blooms and other ecological imbalances associated with eutrophication. In a eutrophic reservoir, it is possible that sport fishing will be improved by a decrease in fertility.

Managers of water supply reservoirs need to recognize that the number and species of fish and other organisms present in a reservoir will greatly influence its sensitivity to nutrient enrichment and will regulate the frequency and intensity of algal blooms (Shapiro, 1979; Lynch, 1981). Only certain types and sizes of sport fish are desirable, and it may be possible to increase the number and size of desirable fish and to simultaneously retard eutrophication or counteract its effects. An historical lack of communication between limnologists and fisheries biologists has hampered research in this area (Rigler, 1980), but enough is known about the management of sport fish to begin to more fully assess their compatibility with water supply reservoirs and management practices. Fishery scientists and biologists should play a much greater role in the development and implementation of reservoir management and restoration projects (Keup, 1979; Shapiro, 1979).

D. WATER TREATMENT PRACTICES

Water treatment has changed very little in the past fifty years. Surface water supplies in Kansas (and elsewhere) are treated by coagulation, sedimentation, filtration, and chlorination; and at some locations hardness is reduced by softening. Plants treating surface waters are typically designed to provide treatment of seasonal taste and odor problems and to remove contaminants from water withdrawn from an anaerobic hypolimnion, if such conditions are likely to occur. Since removal of taste and odor is sometimes expensive, difficult, or incomplete, one of the major goals of water quality management is to reduce or eliminate taste and odor problems prior to treatment by controlling eutrophication and algal blooms. Also, since increased levels of ammonia, sulfide, iron, and manganese can increase the cost and difficulty of treatment and degrade the quality of the finished water, the managers of water supply reservoirs may take steps to prevent anaerobic conditions in the hypolimnion. Thus, most water quality management practices for water supply reservoirs are (and have historically been) primarily intended to alleviate problems that are aesthetic and economic in nature.

In recent years a significantly different problem has confronted the managers of water supply reservoirs: a public health problem. It has been discovered that chlorine, used to disinfect the water, reacts with organic material naturally present in the water to form potentially hazardous by-products (e.g., Rook, 1974; Symons et al., 1981; Bull, 1982; Christman et al., 1983; Fleischacker & Randtke, 1983; Miller & Uden, 1983; Oliver, 1983). Because chloroform, one of the by-products formed, is a known animal carcinogen, the U.S. Environmental Protection Agency has promulgated a primary
drinking water standard of 0.1 mg/L for the class of compounds known as trihalomethanes (THMs), which includes chloroform. Although treatment alternatives are available for reducing the concentrations of these by-products in the finished water (Symons et al., 1981; Randtke, 1984) most of them are expensive and only partially effective. Since water treatment plants in Kansas are designed for prechlorination rather than post-filter chlorination, changes to the chlorination process are generally able to produce only a moderate reduction in THMs at best; and some plants experience considerable difficulty in attaining the standard of 0.1 mg/L. Since the organic precursor materials that react with the chlorine are present in the source water, water supply managers have begun to explore ways to reduce their concentration so that 1) THM formation can be minimized; 2) the cost of THM reduction can be minimized; and 3) expensive treatment plant modifications can be avoided.

Although much research on this problem is still on-going, several conclusions can be drawn from research already completed:

1. In general, the concentrations of THM precursors in surface waters correlate reasonably well with TOC (total organic carbon) concentrations (Symons et al., 1975); and, for a particular water supply, TOC is often a reasonably good surrogate measure of THM precursor concentration.

2. Algae and bacteria both produce precursor materials (Jacquez & Muckerman, 1979; Briley et al., 1980; Hoehn et al., 1980 & 1984; Oliver & Shindler, 1980; Tambo & Kamei, 1981; Oliver, 1983); but the amount formed appears to depend upon species and growth phase. Attempts to correlate THM formation directly with algal blooms and bacterial growths have not yet been very successful, but the data do suggest that such a relationship exists (Hoehn et al., 1984).

3. TOC and chlorophyll concentrations in lakes and reservoirs correlate well with total phosphorus concentrations, but not with nitrogen concentrations (Walker, 1983).

On the basis of this information, it is reasonable to assume that management practices able to control eutrophication will also reduce THM precursor concentrations; but a comprehensive assessment of the impact of water quality and fisheries management practices on TOC and THM concentrations has not been made. Because of the significance of the THM problem to public water supplies, a special effort was made in this research project to search for published information that might be helpful in this regard and to assess the potential impacts of management practices on THM formation.
CHAPTER 3
SPORT FISHING IN KANSAS

Sport fishing in Kansas is considered by the State to be an experience valued by residents and visitors alike (Hartman, 1984; Anon., 1985). At most water supply reservoirs in the state, sport fishing is allowed or even encouraged. Table 2 lists 29 sport fish found in Kansas reservoirs. Sport fish are those fish purposely caught by legal means, being sought after for their food, fighting, or trophy value. Game fish are sport fish whose population sizes are directly manipulated either by stocking or by catch regulations.

Most sport fish in Kansas are predatory in their feeding behavior and visually locate their prey, which are large zooplankton, insects, and smaller fish. Catching sport fish by hook and line, the most common fishing practice, is consistent with such feeding behavior. Sport fish listed in Table 2 that do not feed exclusively in this manner include paddlefish, carp, buffalo, bullhead, and channel catfish. The paddlefish is a filter feeder with little visual requirement for feeding and is caught only by "snagging." The others locate food by smell and taste (as well as visually), which influences the kinds of baits used, but are still readily caught on hook and line. Although clear water is less important to them than to fish that depend only on sight, the production of all sport fish is greatest in clear water (except as production is limited by competition among them). Sport fish production is also strongly influenced by conditions (including water clarity) which regulate the population sizes of their insect and smaller fish prey. Indirectly then, they are affected by most of the traditionally considered water quality parameters, including temperature, turbidity, salinity, pH, alkalinity, oxygen, and plant nutrients, particularly phosphorus and nitrogen.

The habitat requirements for sport fish vary with each species and do not necessarily differ greatly from those of many forage fish (i.e., fish preyed upon by predaceous fish) or rough fish (i.e., fish larger than forage fish considered to be of minimal recreational value). It is possible, however, to identify some habitat conditions in reservoirs which appear to be particularly important to many sport fish. As stated above, most sport fish are obligate visual feeders, so water clarity is very important. As turbidity increases, such fish will expend increasing amounts of energy seeking and catching prey as their field of vision is reduced. It may also be necessary for them to spend more time at shallower depths to compensate for reduced light conditions. This may cause them to be exposed to higher temperatures than they are accustomed or adapted to. For example rainbow trout, which are not native to Kansas reservoirs, might be better able to survive when introduced if they could remain in the deeper colder waters of a stratified reservoir during our typically hot summers. Rainbow trout are accustomed to temperatures below 75°F, which do prevail in the summer hypolimnion of stratified reservoirs in Kansas. However, in this zone (usually at depths below 6 M) light is absent or nearly so and feeding is severely limited. At the same time, oxygen is usually absent due to the absence of plant photosynthesis there and the generally eutrophic condition
### TABLE 2

**RESERVOIR SPORT FISH IN KANSAS, INCLUDING CURRENT STATEWIDE DISTRIBUTION** (east half, west half, or statewide), HISTORY (native or recently introduced, i.e., in the past 100 years), INCIDENCE (rare or common as a catch where found), AND STATUS AS GAME FISH (purposely manipulated by stocking or by imposition of catch limits). Information summarized from Cross and Collins (1975) and F. B. Cross, personal communication (1985).

<table>
<thead>
<tr>
<th>Fish Species</th>
<th>Statewide Distribution</th>
<th>History</th>
<th>Incidence</th>
<th>Game Status</th>
</tr>
</thead>
<tbody>
<tr>
<td>Paddlefish, Polyodon spathula (Walbaum)</td>
<td>east, native, rare</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Bowfin, Amia calva Linnaeus</td>
<td>east, introduced, rare</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Goldeye, Hiodon alosoides (Rafinesque)</td>
<td>east, native, common</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Rainbow trout, Salmo gairdneri</td>
<td>Richardson, statewide, introduced, rare, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Northern pike, Esox lucius Linnaeus</td>
<td>statewide, introduced, rare, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Carp, Cyprinus carpio Linnaeus</td>
<td>statewide, introduced, common</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bigmouth buffalo, Ictiobus cyprinellus (Valenciennes)</td>
<td>east, native, common</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black buffalo, Ictiobus niger (Rafinesque)</td>
<td>east, native, rare</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Smallmouth buffalo, Ictiobus bubalis (Rafinesque)</td>
<td>east, native, common</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black bullhead, Ictalurus melas (Rafinesque)</td>
<td>statewide, native, common</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Yellow bullhead, Ictalurus natalis (Le Sueur)</td>
<td>statewide, native, common</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Channel catfish, Ictalurus punctatus (Rafinesque)</td>
<td>statewide, native, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Blue catfish, Ictalurus furcatus (Le Sueur)</td>
<td>east, native, rare</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Flathead catfish, Pylodictis olivaris (Rafinesque)</td>
<td>statewide, native, common</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Striped bass, Morone saxatilis (Walbaum)</td>
<td>statewide, introduced, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>White bass†, Morone chrysops (Rafinesque)</td>
<td>statewide, introduced, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Smallmouth bass, Micropterus dolomieui Lacepede</td>
<td>east, native, rare, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spotted bass, Micropterus punctulatus (Rafinesque)</td>
<td>east, native, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Largemouth bass, Micropterus salmoides (Lacepede)</td>
<td>statewide, native, common, game</td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Warmouth, Lepomis gulosus (Cuvier)</td>
<td>east, native, common</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Green sunfish, Lepomis cyanellus Rafinesque</td>
<td>statewide, native, common</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Redear, Lepomis microlophus (Gunther)</td>
<td>east, introduced, rare, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bluegill, Lepomis macrochirus Rafinesque</td>
<td>statewide, native, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rock bass‡, Ambloplites rupestris (Rafinesque)</td>
<td>east, introduced, rare, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>White crappie, Pomoxis annularis Rafinesque</td>
<td>statewide, native, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black crappie, Pomoxis nigromaculatus (Le Sueur)</td>
<td>statewide, introduced, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Walleye, Stizostedion vitreum (Mitchill)</td>
<td>statewide, introduced, common, game</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Yellow perch, Perca flavescens (Mitchill)</td>
<td>east, introduced, rare</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Freshwater drum, Aplodinotus grunniens Rafinesque</td>
<td>statewide, native, common</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* Most native species originally occupied stream habitat prior to impoundment.
† Do not commonly reproduce when introduced into state reservoirs.
‡ Possibly native in the last century, but lost and then introduced.
of Kansas reservoirs. The latter results in large quantities of settling organic matter reaching the deeper waters and then decomposing, thereby depleting the oxygen.

Certain structural features of the reservoir are also important to the general sport fish habitat. Maintenance of desirable sport fish populations in a reservoir is in part dependent upon reproduction to supplement or eliminate costly stocking programs. Spawning (i.e., laying of eggs) is often the most sensitive step in the reproduction process, and is closely related to habitat conditions in the reservoir (Noble, 1980). Certain species often require a particular type of habitat to successfully spawn. For example, northern pike seek marshy areas in which to spawn. If a reservoir has little of this habitat at the time of spawning, marshes can be temporarily created by water level manipulation (e.g., raising of the water level, see Chapter 7). Another solution for some sport fish is to provide artificial spawning structures. Some fish require shallow rocky zones, which are seldom available in Kansas reservoirs with their predominantly soil and clay basins. The rock used to line earthen dams to prevent erosion can served as this type of habitat. Purposely adding gravel to shallow zones will also provide spawning habitat.

Even with successful spawning, reproduction may still fail to produce a healthy adult generation. This may begin with the fish embryos and newly hatched fry failing to survive. These stages are particularly vulnerable to poor water quality conditions, such as high turbidity and low dissolved oxygen, and to adverse biological conditions, such as high predation or low food availability. Aquatic macrophytes, often the nemesis of reservoir managers, provide critical shelter for young fish to avoid predation. Vegetation control programs (see Chapter 4) must take this into account. Another means of providing cover is to leave standing timber in the reservoir at the time of construction or to later add brush piles, submerged and anchored (see Chapter 14).

One of the most prevalent problems in sport fish management is the control of forage fish populations. Forage fish include various sunfish, perch, carp, suckers, gizzard shad, red shiners and other minnows, and certain catfish. Such fish, though an important food source for some adult sport fish like largemouth bass and northern pike, often compete with younger sport fish for the same plankton and insect food sources, and may also prey upon the eggs or young of sport fish. This is a problem inherent to many reservoirs, particularly those in Kansas, and is a function of their recent riverine origin. Large predaceous fish like northern pike, muskelunge, walleye, and striped bass, which are not native to Kansas, are top carnivores in the food chains of many natural lakes, particularly those of glacial origin, where they provide natural biological control of forage fish population sizes. In streams and rivers, the native aquatic habitat in Kansas, periodic natural calamities like drought or flood provide a natural check on forage fish populations, reducing the need for larger carnivores. Also, the natural turbidity of streams and rivers discourages the establishment of large visual feeders. With the construction of a reservoir, natural calamities are reduced and, in the absence of predators, many forage fish will proliferate. Subsequent stocking of larger top carnivores like
northern pike, walleye, and white bass represents an attempt to construct a more natural food chain typical of a lake.

Although most water supply reservoirs permit or even encourage fishing, management practices to enhance fishing must be compatible with the maintenance of a potable water source. The remaining chapters of this report focus on the compatibility of reservoir management practices with both fishing and water quality.
CHAPTER 4
CONTROL OF ALGAE AND MACROPHYTES

A. BACKGROUND

Algae and Macrophytes

The typical Kansas reservoir is eutrophic (well nourished) from the moment it is first filled, being supplied with an abundance of nutrients from the soil it is constructed over, from the streamflow feeding it, and from local runoff. These nutrients are used by algae and macrophytes (emergent, floating, or submersed plants) for the primary production of organic plant material that serves as the base of the aquatic food chain. Primary productivity in reservoirs is usually dominated by planktonic algae, though in small reservoirs an increased contribution from periphyton or macrophytes may occur (Mulligan, 1969). When present in quantities that create a nuisance, algae and macrophytes are commonly referred to as weeds.

In Kansas and elsewhere, most of the water quality management practices applicable to water supply reservoirs are primarily intended to prevent or retard eutrophication or to counteract its effects; and most of the water quality problems associated with eutrophic reservoirs are caused, directly or indirectly, by excessive growths of algae and macrophytes (Bernhardt, 1981; Walker, 1983). Algal blooms increase the dosages of chlorine and coagulant required for treatment (Walker, 1983), clog filters (e.g., Bernhardt, 1981; Dice & Beer, 1983), create taste and odor problems (e.g., Lin, 1977; Faust & Aly, 1983), contribute to color, and form potentially hazardous by-products when chlorinated (e.g., Briley et al., 1980; Hoehn et al., 1980; Oliver & Shindler, 1980). Blue-green algae are generally considered to be the most troublesome, since they are frequently associated with taste and odor problems, have been known to cause fish kills, and can fix nitrogen. In turbid reservoirs, such as those typically found in Kansas, blue-green algae have a competitive edge over green algae due their buoyancy.

Certain species of algae produce toxins that can poison fish and birds or adversely affect the health of humans or animals (e.g., Tisdale, 1931; Ingram & Prescott, 1954; Davidson, 1959; Palmer, 1960; Lippy & Erb, 1976; Carmichael, 1980; Billings, 1980; Gorham & Carmichael, 1980; Sykora & Kaleti, 1980; May, 1980; Suess & Dean, 1981). The extent to which water treatment removes these toxins is unknown; however, the severe taste and odor problems associated with large blooms of algae generally result in immediate treatment or a change in the source of supply, such that health effects have rarely been observed. Of course, since algal blooms are temporary, and since the principle health effects (irritations of the skin and the gastrointestinal tract) are sublethal, perhaps the health effects are rarely attributed to their true cause when they do occur.

Macrophytes cycle nutrients from the sediments into the water column (Moore et al., 1984), increase productivity, and increase sediment accretion
rates; and these effects can combine together to create a positive feedback loop (Carpenter, 1983). In addition, they interfere with boating, swimming, and fishing (Engel, 1984); and, although aesthetically pleasing in small quantities, they can be an unsightly nuisance. Macrophytes can also provide breeding habitat for mosquitoes, which are capable of spreading diseases, including malaria and encephalitis.

Both algae and macrophytes eventually die and begin to decompose, releasing nutrients and organic matter into the water column (Landers & Lottes, 1983). The nutrients can stimulate new growths of algae or macrophytes, either immediately or at the onset of the next growing season. The organic matter acts to coagulate turbidity, thereby increasing the depth of the photic zone, further increasing primary productivity, and accelerating sediment accretion. Incorporation of this organic matter into the sediments causes poor compaction (high water content), further aggravating the situation. In addition, the organic matter can support growths of actinomycetes or other bacteria able to cause taste and odor problems (e.g., Palmer, 1960; Silvey & Roach, 1964); and the organic matter or its decomposition products can interfere with water treatment by complexing metal ions used for coagulation, by necessitating treatment for color removal, and by reacting with chlorine. Furthermore, much of the organic matter is not removable by conventional treatment methods, so it enters the distribution system, where it can stimulate the growth of bacteria (perhaps resulting in taste and odor problems or increased corrosion) and make it difficult to maintain a free chlorine residual (e.g., Dorin, 1981; Bernhardt, 1981).

Decaying algae and macrophytes generally settle to the bottom of the reservoir, where their decomposition by bacteria rapidly depletes the dissolved oxygen. When the water in the hypolimnion becomes anaerobic, its quality is reduced by the generation of sulfides, soluble iron and manganese, ammonia, and dissolved organic matter, all of which will increase chlorine demand, cause taste and odor problems (Lin & Evans, 1981), and otherwise interfere with water treatment. Arruda (1985) has shown that the amount of iron and manganese in water samples taken from Kansas reservoirs is proportional to the concentration of chlorophyll-a in the surface water.

For these and other reasons, eutrophication is universally viewed as undesirable by the managers of water supply reservoirs; and it is not uncommon for them to do everything within reason to eradicate algae and macrophytes. However, despite the fact that algae and macrophytes cause many water quality problems, they also serve many important functions (Pandit, 1984). In addition to constituting the base of the aquatic food chain, they provide habitat for fish, wildfowl, insects, and other organisms; they remove pollutants (including turbidity, toxic chemicals, and nutrients) from the water; they can support huge quantities of periphyton, an important source of fish food; and they retard shoreline erosion and resuspension of bottom mud, thereby improving water clarity (and increasing productivity). While it is sometimes desirable to control the growth of macrophytes and algae in a reservoir, it is generally not desirable to completely eliminate them, except in storage reservoirs for treated water.

For fisheries, there is an optimum level of plant growth that will produce the optimum types, numbers, or sizes of fish. If the actual level
of plant growth in the reservoir exceeds the optimum level for fish production, then the objectives of maximum sport fish production and protection of drinking water quality will be compatible in regard to weed control. However, if the actual level of plant growth is less than the optimum level for fish production, any attempt to increase primary productivity to increase fish production will be opposed by the managers of water supply reservoirs on the (justifiable) grounds that increased fertility will cause deterioration of drinking water quality.

In some cases it is necessary for the reservoir manager to decide whether to control algae or macrophytes or both. A sharp reduction in the algal population can result in increased macrophyte growth, due to reduced turbidity and increased nutrient availability. Removal of macrophytes can likewise lead to increased phytoplankton growth (e.g., Canfield et al., 1984), but increased zooplankton grazing may keep the algal population under control (e.g., Newbold, 1976). Also it is possible that removal of macrophytes will reduce internal nutrient loadings, thereby eventually decreasing phytoplankton growth (Carpenter, 1983; Moore et al., 1984). Wile (1978) found that harvesting of weeds in a Canadian lake actually reduced phytoplankton biomass and had no influence on fish populations, but this case was unusual, in that the mass of nutrients exported by harvesting was roughly equal to the external nutrient loading from the watershed.

Kansas reservoirs are generally quite turbid and rather shallow. In large turbid reservoirs that seldom stratify, phytoplankton will generally be light limited and macrophytes are not expected to significantly influence either phytoplankton growth or water quality. In smaller reservoirs that are deep enough to thermally stratify, it may be necessary to control both algae and macrophytes to achieve the desired level of water quality. In most Kansas reservoirs, phytoplankton are the dominant primary producers, and macrophyte control is seldom practiced.

Control Methods

Because algae and macrophytes are the primary cause of many significant problems impacting both water supply and fishing, they are, directly or indirectly, the principal target of most reservoir management practices. The remaining sections of this chapter address those management practices directly aimed at control of algae and macrophytes, including the use of algicides and herbicides, nutrient inactivation, light blockage (to prevent photosynthesis), mechanical harvesting (physical removal of weeds), sediment covering (to control macrophyte growth), and biological control.

Exotic algae control methods, such as electrolysis (Paul et al., 1974), use of naturally produced growth inhibitors (Harris, 1971), and generation of pressure waves (via explosives or ultrasound), are not discussed herein because they are either inadequately developed for use under field conditions (Muchmore, 1978; Raman, 1985) or generally inappropriate for multipurpose reservoirs.

The use of fish species and other organisms to control weeds by grazing is addressed in Section L of this chapter. Control of algae by food chain manipulation, i.e., through stocking of large predator fish, is discussed in
Chapter 5 (Section B: Stocking). Fertilization (Chapter 6) can help to control blue-green algae by causing a shift to green algae, but is most commonly used to increase fish production in farm ponds. Level adjustment (Chapter 7) does control certain macrophytes, but is used in Kansas primarily for the purpose of enhancing fish habitat. Other management practices, such as selective withdrawal, flushing, destratification, hypolimnetic aeration, dredging, draining, and control of nutrients from runoff and waste discharges, can indirectly control weeds by reducing the availability of nutrients. These practices are discussed in subsequent chapters of this report.

B. HERBICIDES—GENERAL REMARKS

Chemical herbicides*, including copper sulfate (the most widely used herbicide), complexed copper, chlorine, permanganate, and an array of organic chemicals, are a particularly attractive means of controlling weeds in water supply reservoirs. They are relatively cheap, they are immediately effective, they generally do not interfere with water treatment, and their use requires little or no understanding of the naturally functioning reservoir ecosystem. Herbicides are frequently used by the managers of water supply reservoirs in much the same way that medical doctors sometimes prescribe drugs for their patients, i.e., to provide quick and effective relief of symptoms, often without complete understanding of the ultimate cause of the problem. They are used because they work, at least temporarily, and because other alternatives, although perhaps cheaper in the long term, are more difficult to evaluate or to implement.

Weed growths can be very detrimental to fish. Excessive amounts of algae can clog gills, some algal species produce toxins powerful enough to kill fish, and decay of dead weeds can consume enough oxygen to suffocate fish. Therefore, fisheries managers occasionally resort to the use of herbicides to control weed growths, particularly blooms of certain blue-green algae. The chemicals and dosages used are not directly harmful to fish, but fish kills in fisheries have been attributed to oxygen depletion brought on by the rapid decay of vegetation killed with herbicides.

Sladekova and Sladecek (1968) suggest that an ideal herbicide should be: 1) selective for the specific nuisance; 2) nontoxic to fish and fish-food organisms; 3) harmless to humans; 4) without any negative influence on water quality; 5) non-accumulative; 6) easily incorporated into the natural elemental cycles; 7) non-corrosive and harmless to equipment; 8) effective under any condition; 9) easy and economical to store and handle; 10) of reasonable cost; 11) available for continuous use; and 12) easily detected

* Herbicides are chemicals able to destroy plants or inhibit their growth; and the term "herbicides", as used herein, includes algicides, which are able to kill algae, a select group of plants.
and quantified in water. No chemical fully satisfies all of these criteria, but a number of chemicals have been found acceptable for limited or even widespread use: copper sulfate, complexed copper, potassium permanganate, and certain organic chemicals (e.g., 2,4-D and endothall). Of these, copper sulfate is used most extensively.

Chemical herbicides are most successful when they are used as a preventive measure rather than as a cure; and they should be applied when they will have their maximum inhibitory effect. This is why it is extremely important for reservoir managers to regularly examine the phytoplankton composition and the growth curves and developmental stages of the organisms occurring in a particular reservoir (Sladekova & Sladecek, 1968). If a mass development (bloom) of algae or macrophytes occurs, it is usually too late for optimal treatment, and severe oxygen depletion may occur, with or without treatment. Furthermore, the decomposition of a large quantity of weeds will release a large quantity of nutrients back into the water, stimulating the growth of another mass of weeds when the herbicide loses its effectiveness. Hence, use of a herbicide to kill large masses of weeds can lead to "chemical dependency," with large dosages of herbicides required at regular intervals.

Heavy growths of macrophytes can be anticipated by visual inspection, but algal blooms are more difficult to anticipate. Monie (1956) suggests that the very first step in the chemical control of algae should be to identify and count the algae present in the water supply using microscopic analysis (e.g., as described in APHA, 1985). Sample collection is an important part of microscopic analysis. Samples should be obtained periodically from different sections of the water body, since the depth of the water, the exposure to sunlight, and other factors often cause a significant difference in the number and types of algae in different sections of the water body. Frequent monitoring should be done early in the seasons in which algae are known to bloom. A well-planned year-round algae sampling program can be a valuable tool in lake management. Reed (1965) reported on the necessity of treating a Boston water supply under ice due to the development of objectionable algae under the ice.

Monie (1956) and others have noted that temperature is a very important factor in algae control. The growth of various types of algae can be anticipated by the temperature of the water, especially the surface temperature; and the critical temperatures at which certain algal species are stimulated to activity are known. It must be realized that there is no set formula to define when it is necessary to treat for algae control. The unique characteristics and history of each water body must be considered in developing an effective, preventive treatment plan.

Closed-loop stripping analysis (CLSA) is a recently developed tool that can help reservoir managers to detect taste and odor problems before they get out of hand, to isolate sources of taste and odor, and to direct the application of chemical algicides (McGuire et al., 1984). The method, now a standard method (APHA, 1985), involves the stripping of semi-volatile odor-causing chemicals from the water and determining their concentration using gas chromatography.
A number of herbicides are discussed in subsequent sections of this chapter, including copper compounds, chlorine, permanganate, and organic chemicals. Sodium arsenite and silver nitrate, once used as algicides (Mulligan, 1969), are not discussed, since they are no longer considered appropriate for general use. Arsenite has a high mammalian toxicity and is a cumulative poison, while silver is more toxic than copper to fish and is very expensive.

C. COPPER SULFATE

Background

Copper sulfate, also known as blue vitrol or bluestone, has been used effectively in the water-supply industry since its introduction as an algicide in 1904 by Moore and Kellerman. Despite the many studies that have been done to investigate alternatives to the use of copper sulfate for algae control, copper sulfate is still generally regarded as the most effective and economical chemical for controlling obnoxious algae (Muchmore, 1978). Numerous reports in the literature testify to the successful control of algae using copper sulfate. Although copper sulfate does not completely satisfy all of the requirements of an ideal algicide, it is generally perceived by many reservoir managers to be: 1) toxic to a majority of nuisance organisms at relatively low concentrations; 2) relatively nontoxic to most fish species; 3) not harmful to the general aquatic environment in the concentrations used; and 4) comparatively inexpensive (Courchene & Chapman, 1975). Unfortunately, many reservoir managers fail to recognize that despite copper sulfate's immediate effectiveness in killing algae, it can kill fish, may create more severe algal problems in the future, may cause harm to the aquatic environment, and at best provides only temporary relief. Hence, careful consideration should be given to the use of copper sulfate, especially in reservoirs used for sport fishing.

Copper sulfate is also effective for control of certain macrophytes, but organic herbicides are more commonly used than copper sulfate for macrophyte control. Where control of both algae and macrophytes is required, it may be more convenient to use copper sulfate for both purposes. Because copper sulfate is the single most important algicide used by the water supply industry, and because it is less commonly used for macrophyte control, this chapter will focus primarily on the use of copper sulfate as an algicide. Nevertheless, much of the discussion is also pertinent to the use of copper sulfate for macrophyte control.

Dosage Requirements

There is no set dosage of copper sulfate which will be effective in all situations, and many factors influence the amount of copper sulfate needed to treat a particular water supply. According to Whipple et al. (1948) the required dosage depends upon the species of algae present, the amount of organic matter present, water hardness, carbonic acid concentration, water temperature, the species of fish present, and the amount of water to be treated. Tables based on laboratory studies and field experience have been
developed to describe the required dosages for treatment of various genera of nuisance organisms (e.g., Hale, 1942; Prescott, 1948); and typical dosages are generally in the range of 0.1-0.5 mg/L. However, use of such tables must be tempered with experience and with knowledge of the characteristics of the particular water supply to be treated (Monie, 1956). Before adding an algicide, short-term bioassays should be conducted, since the minimum effective dose can vary widely between reservoirs and in a single reservoir at different times of the year (Wurtsbaugh & Horne, 1982).

Lueschow (1972) reported that the usual application rate for planktonic blue-green algae is 5.4 pounds copper sulfate per surface acre (1 ppm for the upper 2 feet), that filamentous algae are usually controlled by application of 10 pounds per surface acre repeated at weekly intervals for 3 to 5 weeks, and that Chara control is accomplished with a dosage of 10 pounds per acre applied as close to the bottom as possible to avoid conversion to copper carbonate in the water column.

Various means of fine-tuning copper sulfate dosages have been proposed, including the D.M. test (Monie, 1956), closed-loop stripping analysis (McGuire et al., 1984), and indexing of the dosage to a variable such as methyl orange alkalinity, free carbon dioxide, or temperature. In general practice the dosage is usually adjusted according to the total alkalinity of the water. The usual rate for waters having a total methyl orange alkalinity greater than 40 mg/l is 5.4 pounds of commercial copper sulfate per surface acre. Waters with a total alkalinity of less than 40 mg/l usually receive a dose of 0.9 pounds of copper sulfate per acre-foot of water, calculated for the entire volume of the reservoir (Lueschow, 1972; Courchene & Chapman, 1975; Lin, 1977). Kansas surface waters typically have a carbonate hardness in the range of 140 to 220 mg/l as calcium carbonate, and non-carbonate hardness generally runs between 40 and 50 mg/l as calcium carbonate.

Fair et al. (1971) recommended raising the copper sulfate dosage by as much as 5% for each 10 mg/l of alkalinity. This suggestion stems from the fact that copper sulfate reacts with available carbonate ions in hard waters to precipitate as copper carbonate, which is ineffective as an algicide. Fair et al. (1971) also recommended adding an additional 5% to waters with abnormally low amounts of CO2, adding an additional 2% for each 10 mg/l of organic material present, and adjusting the dose for changes in temperature.

Wageman and Barica (1979) showed that the effectiveness of copper as an algicide depends on the total toxic copper concentration which is equal to the sum of three species: divalent cupric ion and mono- and di-hydroxo copper (II) complexes. The toxicity of the copper hydroxide complexes explains the effectiveness of copper as an algicide in waters with a relatively high pH, while the ineffectiveness of the carbonate complexes explains why higher dosages are needed in waters high in carbonate alkalinity. With the chemical equilibrium models now available, it should be possible to more accurately compute the concentrations of the toxic copper species, facilitating comparison of results and assisting in the determination of dosage requirements.
Copper Sulfate Application

Copper sulfate is available commercially in various particle sizes: large crystals, small crystals, granular (rice), or powdered (snow). The size of the copper sulfate crystals selected for application is determined by the rate of solubility desired. McGuire et al. (1984) found that medium-size chunks of copper sulfate (size C crystal) at a dosage of 200 pounds per acre provided the most effective control of attached growths of blue-green algae. Smaller crystal sizes tended to dissolve before sinking to the bottom. Toth and Riemer (1968) compared the efficiency of granular copper sulfate (1/8-1/4 inch) to that of a solution when applied to shallow ponds. They concluded that granular application resulted in lower copper sulfate concentrations in solution than expected due to the sinking of the granules and absorption by bottom sediments. They found that if copper sulfate is dissolved in water and applied to the surface of a pond, the amount of copper sulfate found in solution in the pond is greater than if the copper sulfate is applied in granular form. The penetration of copper sulfate to any desired depth can be accomplished by appropriate control of the particle size distribution (Goudy, 1936).

There are numerous procedures for applying copper sulfate to a water supply, but they vary widely in their ease and effectiveness. Sladeckova and Sladecek (1968) have summarized the most common techniques: 1) dosing into the flow of the body of water; 2) dispersal of crystals on the water surface using a boat or an airplane; 3) spraying a solution from a boat or an airplane; 4) dissolving copper-sulfate crystals suspended in a burlap bag or a screened hopper behind a boat; and 5) delivering the solution from a pipeline or pipe manifold suspended behind a boat. The main factors determining the technique of choice are the size of the reservoir, the equipment that is available, and financial resources (Courtchene & Chapman, 1975). Other considerations include: the season of the year, the temperature, the species of algae, other biota present in the water, the particular kind of water body, and the purposes for which the water is used.

Historically, the simplest and most common method of copper sulfate distribution has been to suspend burlap bags filled with about 50 pounds of large-crystal copper sulfate in the water behind or along side a motor boat. As the crystals dissolve, the bags are refilled. The propeller action of the motor boat aids in the dissolving and dispersal of the crystals. Screened hoppers mounted on the backs of motor boats and barges have provided a significant improvement over the use of burlap bags (Courtchene & Chapman, 1975).

Domogalla (1935) found that systematic spraying of a copper sulfate solution was much more effective than the burlap bag method, and later studies have also noted the success of copper sulfate spray application. A typical spray apparatus (often barge mounted) has the capacity to deliver about 400 pounds of copper sulfate per hour as a 2-3% solution. The copper sulfate solution is usually applied from the shore toward the open water with the spray passes made parallel to shore.

Blower-type machines mounted on barges may be used to disperse powdered copper sulfate over the surface of the water in much the same way as certain
agricultural chemicals are applied to soil. Several problems have been encountered in the use of blower machines: the machines are generally bulky and heavy; they depend on wind for distribution of the copper sulfate and require constant adjustment for proper feed; and there is a tendency for some of the chemical to be scattered over areas not intended for treatment.

McGuire et al. (1984) used a heavy-duty chemical spreader (1400 lb. capacity) to handle the distribution of large amounts of granular copper sulfate. The spreader was mounted on a barge and calibrated to feed 200 lb/acre at a barge speed of 3 mph, with a spread width of 30 feet. This technique allowed for distribution of 10,000-15,000 pounds of copper sulfate in a 10-hour day. The areas which were to be treated with copper sulfate were marked off with buoys so all attached algae growths could be adequately treated. This shoreline treatment technique appeared to kill attached growths effectively without any adverse effects. Scuba divers were also used in this study to determine the effectiveness of the treatment and locate special problem areas which needed additional copper sulfate application.

Copper sulfate application can also be accomplished by the use of airplanes and helicopters to spread copper sulfate in powder form (Muchmore, 1978). Brouse (1966) reported that a cropdusting aircraft sprayed 20 tons of copper sulfate over a 2,023 acre lake resulting in a copper content of 0.2 ppm in the lake. This was adequate to reduce the algae count significantly.

Reed (1965) reported a novel way to control algae in ice-covered reservoirs. An 18 hp outboard motor can be used to disperse copper sulfate solution under the ice through small holes cut in the ice. The current generated by the motor propeller allows the copper sulfate to be dispersed between 100 and 150 feet.

**Copper Toxicity**

Copper sulfate is often perceived to be less toxic to fish than to algae, but copper is in fact quite toxic to many fish. The dosages of copper shown in Table 3 to be lethal to fish are roughly equivalent to those required to control algae. Fortunately, due to the high concentrations of carbonate alkalinity in Kansas reservoirs, copper is rapidly converted to relatively non-toxic carbonate complexes, protecting fish from its full lethal effects. In addition, copper is strongly bound by certain fractions of organic matter (present in relatively high concentrations in Kansas reservoirs), reducing short-term toxicity, but possibly increasing long-term toxicity by keeping the copper in solution in the water column. Some experts believe that copper produces long-term toxicity when used on a regular basis, but not enough is known to accurately predict when such effects will occur in a particular reservoir. Nevertheless, long-term toxicity is not expected to be a significant problem in Kansas reservoirs treated with the recommended dosages of copper sulfate.

There have been some reports of fish kills attributed to copper sulfate addition, but the actual cause of death has rarely been established. Possible causes of these fish kills include, in addition to copper toxicity,
TABLE 3  
CONCENTRATIONS OF COPPER SULFATE LETHAL TO FISH

<table>
<thead>
<tr>
<th>Fish</th>
<th>Dosage, mg/L</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trout</td>
<td>0.14</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Carp</td>
<td>0.30</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Catfish</td>
<td>0.40</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Pickerel</td>
<td>0.40</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Goldfish</td>
<td>0.50</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Perch</td>
<td>0.75</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Sunfish</td>
<td>1.2</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Black bass</td>
<td>2.1</td>
<td>Moore &amp; Kellerman, 1904</td>
</tr>
<tr>
<td>Stone loach</td>
<td>0.25 (63-day LC-50)</td>
<td>Solbe &amp; Cooper, 1976</td>
</tr>
<tr>
<td>Fathead minnow</td>
<td>0.46 (96-hr. LC-50)</td>
<td>Pickering et al., 1977</td>
</tr>
</tbody>
</table>

Algal blooms (algal toxins), oxygen depletion, and gill clogging. Some fish kills have been attributed to use of copper sulfate dosages much greater than necessary for weed control (e.g., Frost, 1963; Whitaker et al., 1978).

The Maine Department of Environmental Protection (1976) recommended that application of copper compounds for algae control be discontinued in Maine due to their toxicity to fish. However, the natural lakes in Maine and other states in the northeastern U.S. contain water much softer and lower in pH than that in Kansas reservoirs, allowing toxic concentrations of copper to persist for much longer periods of time.

Even though copper is generally considered to be selectively toxic to algae and macrophytes, and relatively harmless to humans, bacteria, and various other organisms, it can have toxic effects on non-target organisms. Effler et al. (1980) found that application of copper sulfate to Cazenovia Lake in New York caused dramatic reductions in bacteria, even though the treatments failed to induce algicidal action and caused no definitive response in submersed macrophytes. They did note short term phytoplankton stress and alteration in the natural seasonal succession of phytoplankton species.

It is interesting to note that copper is an essential micronutrient for algae. Although copper is generally not considered to be a limiting nutrient (Sawyer, 1968), experiments have shown the growth limiting concentration to be in the range of 2-40 micrograms of Cu per liter (Manahan & Smith, 1973; Wurtsbaugh & Horne, 1982), on the lower end of the range of dosages considered toxic. It is, therefore, conceivable that application of copper could actually stimulate algal growth or cause a population shift by altering the availability of nutritional copper.
Assessment

Because Kansas reservoirs are quite turbid, photosynthesis is limited to a rather shallow photic zone. Phytoplankton are generally dominant, except along the shoreline, where filamentous algae and macrophytes may be more abundant. In such reservoirs, copper sulfate is best applied as a spray or by dissolution into the surface water, or by direct spraying onto shoreline vegetation. The water in Kansas reservoirs is also generally high in carbonate alkalinity, so relatively high dosages of copper sulfate are required for effective phytoplankton control.

Although copper sulfate addition is generally effective in achieving immediate control of nuisance algae and macrophytes, has a low mammalian toxicity, is inexpensive, and is in fact the recommended method for water supply reservoirs in Kansas, it can have both short-term and long-term undesirable side effects. Hanson and Stefan (1984) examined a series of lakes treated with copper sulfate for 58 years and identified the following side effects:

1) Dissolved oxygen depletion due to algal decomposition;
2) Accelerated phosphorus recycling;
3) Rapid recovery of the algal population;
4) Occasional fish kills due to oxygen depletion, copper toxicity, or both;
5) Copper accumulation in the sediments;
6) Increasing resistance to copper in nuisance organisms;
7) Increasing dominance of blue-green algae;
8) Selection for rough fish over game fish;
9) Disappearance of macrophytes; and
10) Reductions in benthic macroinvertebrates.

These side effects are neither rare nor surprising; indeed, most of them are entirely possible in Kansas reservoirs in which copper sulfate is relied upon as the primary method of weed control and is regularly applied. Even in the absence of side effects, it may not be the most cost-effective long-term weed control strategy for a given reservoir.

Whether a reservoir is used solely for water supply or for both fishing and water supply, copper sulfate should be added before excessive weed growth occurs, thereby avoiding the rapid decomposition of a large mass of plant growth and the concomitant depletion of dissolved oxygen. Use of copper sulfate to kill a large mass of weeds in a typical Kansas reservoir will almost certainly lead to increased oxygen depletion and will possibly cause fish kills. The increased oxygen depletion will in turn lead to rapid deterioration of hypolimnetic water quality, accelerated phosphorus cycling, rapid growth of additional algal blooms, increasing dominance of blue-green algae, and increasing dominance of rough fish over sport fish. Elimination of macrophytes in shallow areas will reduce sport fish habitat and may increase shoreline erosion.

There have been some reports of accumulation of high concentrations of copper in sediments and potential detrimental effects on benthic organisms, but this is not likely to be a significant problem in Kansas reservoirs in
which the benthos is typically anaerobic and subject to heavy siltation. In Kansas reservoirs, copper is rapidly precipitated as copper carbonate and quickly buried by large amounts of sediment, rendering it unavailable to the biological community. In any event, studies have failed to demonstrate that aquatic organisms are adversely affected by a build up of copper in the sediments (Muchmore, 1978), especially those of turbid hard-water lakes.

However effective copper sulfate may be in providing immediate control of algae, a single application cannot provide long-term control. The copper will be rapidly precipitated, destroying its effectiveness. The dead algae and macrophytes will release nutrients and organic matter into the water, stimulating rapid regrowth of nuisance vegetation and perhaps even clarifying the water, such that even more weeds can grow than before. Therefore, additional treatments will be necessary. Where copper sulfate is relied upon for algae control, it is not uncommon to find that the treatment must be repeated every few weeks throughout the growing season, requiring a sizable expenditure of time and money. Consideration should be given to alternative control methods that might reduce the cost of control while providing better protection of both sport fish and water quality.

Copper sulfate addition should generally be viewed as a "quick fix" remedy to be applied in an emergency, buying time while effective long-term control methods are explored and implemented. In some cases, intermittent use of copper sulfate may in fact be the best long-term control strategy; but this conclusion should be arrived at only after careful study. Elevated nutrient input to the reservoir, perhaps accompanied by ecosystem imbalance, is usually the ultimate cause of excessive weed growth and must be addressed to achieve effective long-term control.

Copper sulfate is expected to have little influence on TOC or THM precursor concentrations, unless it is used frequently enough and in high enough concentrations to substantially reduce primary productivity. Even then it will only reduce TOC if a substantial fraction of the TOC is autochthonous. If copper sulfate is used to kill a massive growth of algae or macrophytes, TOC and THM precursors may sharply increase in concentration due to cell lysis and the release of soluble organic matter into the water.

D. CHELATED COPPER COMPOUNDS

Although copper sulfate is still the chemical compound most widely used for control of algae, the use of chelated copper compounds is increasing. These compounds contain chemicals (chelating agents) that bind (chelate) copper in soluble complexes, such that the copper remains in solution for a longer period of time before being precipitated. Their use is particularly advantageous for waters high in carbonate alkalinity. Calculations presented by Lerman and Childs (1973), for water at pH 8.0 with 1.0 millimoles per liter of carbonate, demonstrate that substantial amounts of copper can be present in solution as a copper-citrate complex, even when the concentration of citrate is quite low.
Assessment

Chelated copper compounds are more expensive than copper sulfate, but they are also more effective and can therefore be used in lower dosages. A decision as to which compound should be used can be based upon cost, taking into consideration the dosage of each compound required for effective control, as determined by laboratory tests.

In general, the impacts of chelated copper compounds on water quality and sport fishing will be approximately the same as those for copper sulfate, with two possible exceptions: 1) the chelated copper, even at reduced dosages, may be relatively more toxic for non-target organisms; and 2) the chelated copper or the chelating agent, may enter the water supply intake in a concentration high enough to cause an unexpected problem (e.g., solubilization of a metal ion at a concentration exceeding the desired level). However, the chelating agents are generally biodegradable (and non-toxic), so that neither they nor their soluble copper complexes are expected to persist for extended periods of time. At the present time, chelated copper compounds are not used in water supply reservoirs in Kansas due to uncertainty concerning possible side effects and due to their cost.

E. CHLORINE AND BROMINE

Chlorine, chlorine dioxide, bromine, and bromine chloride are all known to be algicidal (lethal) or algistatic (growth retarding) at relatively low concentrations. Courchene and Chapman (1975) reported that a free chlorine residual of 0.2-1.0 mg/L is effective in controlling many species of taste- and odor-causing and filter-clogging algae and that chlorine is often effective against organisms that are resistant to copper-sulfate treatment. Some water utilities routinely add chlorine to terminal storage reservoirs* to prevent growths of algae and bacteria in the water awaiting treatment.

Kott and coworkers examined the effects of bromine and chlorine on various species of algae and found that: 1) chlorine and bromine inhibit algal growth at 0.18 to 0.42 ppm, with bromine and bromamine being primarily algicidal and chlorine and chloramine being algistatic (Kott et al., 1966); 2) addition of 10 mg/L of chlorine for 2 hours, 10 mg/L of bromine for 10 hours, and 10 mg/L of copper sulfate for 4 days caused complete kill of the filamentous algae Cladophora (Betzer & Kott, 1969); and 3) chlorine and bromine used together achieved higher algal and bacterial kill than either one separately (Kott, 1969; Kott & Edlis, 1969).

* A terminal storage reservoir is the last reservoir prior to treatment in a multiple reservoir system or the reservoir at the terminal end of a canal or aqueduct.
Assessment

Chlorine is approximately five to ten times cheaper than either copper sulfate or bromine and is readily available at surface water treatment plants, making it an attractive algiocide for single-purpose water supply reservoirs. Unfortunately, chlorine and chloramines are extremely toxic to fish and other aquatic organisms, generally precluding the use of chlorine in reservoirs used for sport fishing. Application of low dosages of chlorine to a small area of a reservoir might be a suitable way to control nuisance vegetation in an emergency, but frequent use of chlorine over large areas of a reservoir would be expected to produce severe adverse effects on the functioning of the ecosystem.

Chlorine is decomposed by sunlight, with chloride as the major end product. Volatile and non-volatile chlorinated byproducts are formed from the reaction of chlorine with organic matter, but these byproducts are not expected to persist for long periods of time in the aquatic environment and in the concentrations found would not be likely to cause harm to organisms. It is reasonable to expect that a slight reduction in the concentration of THM precursors entering the water supply intake will occur if substantial dosages of chlorine are applied to the reservoir, but any advantage might be offset by increased concentrations of chlorinated byproducts in the raw and finished waters. Chlorine dioxide forms much smaller amounts of THMs and chlorinated by-products (Chow & Roberts, 1981); but its toxicity to fish (about the same as chlorine), its cost, and the necessity of on-site generation make it unattractive for use in reservoirs (Aieta, M., pers. comm., 1985).

As with any algiocide, use of chlorine to kill large amounts of vegetation will result in oxygen depletion, release of nutrients back into the water column, and other related adverse effects. Furthermore, its effectiveness will be very temporary, and rapid regrowth of plants or algae is expected to occur. Therefore, use of chlorine should be considered only as an emergency measure or short-term solution, while steps are taken to identify an effective long-term solution to the more basic problem of excess nutrient availability or ecosystem imbalance.

F. POTASSIUM PERMANGANATE

Background

Potassium permanganate is frequently used in water treatment plants to control taste and odor problems, especially those associated with algae and anoxic waters, and to oxidize iron, manganese, and sulfide. It has occasionally been used as an algiocide in water supply reservoirs, cooling towers, and infiltration basins, and as a bactericide in fish ponds.

Several investigators have studied the algiidal properties of permanganate in the laboratory. Kemp et al. (1966) examined the algiidal action of seven concentrations of potassium permanganate on 20 species of algae and
compared the effectiveness of potassium permanganate, copper sulfate, calcium hypochlorite, and sodium arsenite. An 8 mg/L dosage controlled the growth of the majority of species tested; some were controlled with dosages as low as 4 mg/L and all were controlled using a 16 mg/L dosage. Fitzgerald found 1) that a 5-10 mg/L dosage maintained for a 5-6 hour period controlled the growth of a mixed algal population (Fitzgerald, 1964a); 2) that 2 mg/L or less could inhibit the growth of blue-green algae (Fitzgerald, 1964b); and 3) that 1-5 mg/L maintained for 12-72 hours killed 7 of 8 algal species tested (Fitzgerald, 1966). There have also been a number of published reports documenting the bactericidal properties of permanganate (Kemp et al., 1966). There appears to be no information available regarding the effects on permanganate on macrophytes.

There have been several published reports of successful field application of potassium permanganate for algae control. Dice and Beer (1983) found that a 0.5 mg/L dosage (based on total lake volume) of potassium permanganate used to supplement on-going copper sulfate treatment increased filter runs from 2-4 hours to up to 40 hours; but the precise reason for the increase was not determined. Potassium permanganate dosages of 1.0-1.5 mg/L were found to effectively prevent growths of filamentous algae in infiltration basins in West Germany, but permanganate was not an effective cure for existing algal blooms and was ineffective in controlling planktonic algae at dosages up to 4 mg/L (Kötter, 1978). Carr (1976) reported the treatment of a recreational lake with permanganate, alum, and copper sulfate, resulting in sharp reductions in total algal counts and blue-green algae and a substantial increase in visibility; however, the study was uncontrolled, making it difficult to draw any firm conclusions regarding the role and effectiveness of permanganate. Use of permanganate in combination with chlorine or copper sulfate has been reported to be effective in controlling algae in raw water reservoirs (Ficek, 1984).

Although laboratory tests have demonstrated the algicidal properties of permanganate, the dosages reportedly used in field applications are generally inadequate to produce algicidal concentrations. Furthermore, permanganate rapidly reacts with the organic matter and reduced minerals present in lake water to form an inert precipitate of manganese dioxide within a very short period of time (typically 10-30 minutes), allowing insufficient time for algicidal activity and producing contact times much shorter than those used in most laboratory studies. Several explanations have been offered to explain permanganate's apparent success under these circumstances:

1) It may enhance the toxicity of copper to algae by oxidizing organic functional groups that bind copper;
2) It may reduce transmission of important wavelengths of light (Kötter, 1978);
3) It may oxidize organic iron complexes and strip iron and other trace nutrients from the water column (Carr, 1976);
4) It may flocculate colloidal organic matter and particulate phosphorus, removing major nutrients from the water column, especially if alum is used to help settle the manganese dioxide.
Assessment

Despite reports of successful field applications of potassium permanganate for algae control, there is not enough scientific evidence from controlled studies to conclusively demonstrate that potassium permanganate is effective in killing or controlling algae under field conditions. Nevertheless, existing evidence does suggest that permanganate either alone or in combination with other chemicals may be useful in controlling algal blooms; and research should be conducted to examine its effectiveness and mode of action under a variety of field conditions.

There is not a great deal of information available regarding the toxicity of permanganate to fish, but what information is available suggests that reasonable dosages of permanganate are quite harmless to fish if properly applied to the water. Kemp et al. (1966) reported 24–96 hour TL₅₀ values of 4.2 and 3.8 mg/L, respectively, for bluegill and creek minnows in laboratory tests; and Lawrence (1956) found that a variety of fish species could tolerate continuous exposure to 3–5 mg/L of potassium permanganate. Since permanganate rapidly decomposes to manganese dioxide in the field, much higher dosages can be applied. Kemp et al. (1966) reported the use of up to 32 mg/L without fish deaths and cited other studies with similar results. Perhaps the strongest evidence that permanganate is relatively harmless to fish is its use by fish culturists to control external bacterial infections of fish (e.g., Phelps et al., 1977; Tucker & Boyd, 1977).

In general, permanganate treatment should produce mostly positive effects on water quality, including oxidation or adsorption of taste and odor causing chemicals, oxidation and adsorption of reduced iron and manganese, perhaps a slight reduction in oxygen demand, an increase in dissolved oxygen, and perhaps increased water clarity (if the water quality is such that the precipitated manganese dioxide exerts a coagulating effect). Permanganate is not expected to alter the THM formation potential of organic matter in the lake, but it may absorb a fraction of the humic material and remove it from the water column (Colthurst & Singer, 1982). The end product, manganese dioxide, is biologically inert and will settle to the bottom of the lake. If the lake bottom is anaerobic, the manganese dioxide will help to limit sulfide production; however, it may also contribute to increased concentrations of soluble manganese in the hypolimnion.

The biggest obstacles to the use of permanganate are its cost (approximately $1.25 per pound) and uncertainty as to its effectiveness and mode of action. As with any algicide, it should be much more effective in preventing algal blooms than in curing them, and its use to kill heavy growths of algae could result in severe oxygen depletion, fish kills, and related undesirable side effects. Since the effects of permanganate on algae are highly dependent upon the particular species involved and the quality of the water to be treated, short-term bioassays should be conducted in the laboratory, using samples of the water treated, to establish dosage requirements. Potassium permanganate treatment should not be viewed as a long-term solution to an algae control problem, but it does hold promise as an emergency or short-term control measure.
G. ORGANIC HERBICIDES

Background

When herbicides are used to control macrophytes in lakes, reservoirs, and ponds, it is generally the organic herbicides which are used (e.g., Woodford & Evans, 1965; Bennett, 1971; Brooker & Edwards, 1975). Macrophytes commonly targeted for control include the aquatic flowering plants, generally rooted in the substrate but sometimes floating, and the macroscopic algae of the genera Chara and Nitella. Sometimes filamentous algae are grouped with the macrophytes because of their "macroscopic" appearance when they accumulate; but more often they are grouped with the phytoplankton, including the blue-green algae, which are more commonly controlled with copper sulfate treatment (see Section C) than with organic herbicides. Since inorganic herbicides, such as copper sulfate, are more often used to control algae in Kansas water supply reservoirs, the discussion here will emphasize the control of macrophytes with organic herbicides.

As noted in Section A of this chapter, excessive macrophyte growth can produce a variety of undesirable conditions, including reductions in fish production, aesthetic value, and water quality (e.g., Bennett, 1971; Dunst et al., 1974; Bates & Hentges, 1976; Anon., 1982a; Riemer, 1984). Water supply reservoirs in Kansas are rarely treated to control macrophytes (pers. comm., KDHE, Water Quality Section). This is due, in part, to the reluctance of managers to use herbicides or to implement a mechanical harvesting program, and in part to the turbid condition of these reservoirs, which limits the light available to submerged macrophytes and prevents their spread beyond the immediate shoreline. In the future, however, with the continued silting in of these reservoirs, shallower zones will increase in area creating more potential habitat for macrophytes to invade. For this reason the use of organic herbicides is expected to gradually increase.

It is not uncommon for a weed problem in a reservoir to involve a variety of plants at different times or locations in the reservoir, but many managers tend to rely on a single chemical, expecting it to control each type of plant, regardless of the plant species present. When all of the nuisance species are susceptible to the herbicide, which commonly occurs when broad-spectrum herbicides are used, this approach is fine; otherwise, susceptible species may simply be replaced by less susceptible species (e.g., Newbold, 1976). It is also common for reservoir managers to begin treatment only after excessive weed growth has occurred. In reservoirs with a history of weed problems, preventive treatment is highly preferable, since the adverse effects associated with the decay of a massive quantity of weeds are thereby avoided (see Sections A and B of this chapter). Aquatic herbicides are best applied early in the spring when plant biomass is low (Newbold, 1976); and they are most effective in cool (15-18°C), non-turbid, hard water when used to treat young plants prior to seed development (Nichols & Shaw, 1983).
There are a great many herbicides that are used (or could potentially be used) to control macrophytes; but only a few of them have been recommended for use in water supply reservoirs. There is an extensive body of scientific literature devoted to various aspects of herbicide use in aquatic environments, a brief sampling of the literature pertaining to control of aquatic weeds is included in Table 4. Currently, only 2,4-D and endothall are approved by KDHE for use in water supply reservoirs in Kansas (pera. comm., KDHE, Water Quality Section), so the remainder of this discussion will focus on these two herbicides. Brooker and Edwards (1975) have written an excellent review of the direct and indirect effects of eight aquatic herbicides (including 2,4-D) approved for use in Great Britain, and this is a good source of information on some of the aquatic herbicides not discussed in detail here.

The herbicide 2,4-D (2,4-dichlorophenoxy acetic acid) was one of the first organic herbicides developed for terrestrial use. In lower doses it will kill most broad-leaved plants (dicots) and in higher doses it will kill both dicots and monocots (Anon., 1983). It has been used in aquatic habitats to control a variety of macrophytes (e.g., Stennis & Stotts, 1965; Smith et al., 1967; Thomas, 1967; Mulligan, 1969; Steward & Nelson, 1972; Scott et al., 1981; Anon., 1982a & 1982b). Different forms of 2,4-D herbicide have been used (e.g., Johnson & Finley, 1980), and some of their chemical characteristics are described in Anon. (1983). The esters of 2,4-D are more effective in killing target weeds than amides (e.g., Stennis et al., 1962) but are more toxic to animals, as will be discussed below. The esters have often been the preferred formulations for aquatic use at concentrations of a few mg/l because their high affinity for tissue and greater insolubility in water make them less mobile in water supplies (Scott et al., 1981). 2,4-D is not a preferred herbicide for treating filamentous and planktonic algae.

Endothall, 7-oxabicyclo(2,2,1)heptane-2,3-dicarboxylic acid, is available for aquatic use in several forms (e.g., Johnson & Finley, 1980), some of whose chemical characteristics are described in Anon. (1983). Originally used as a terrestrial herbicide able to control a wide variety of weeds, it soon was discovered to also control a wide variety of submersed macrophytes (Woodford & Evans, 1965; Simsman et al., 1976; Riemer, 1984). Endothall has been used to control a variety of submersed macrophytes (Steucke, 1963; Walker, 1963; Houser & Gaylor, 1962; Lopinot, 1965; Yeo, 1970; Bennett, 1971; Serns, 1974; Simsman et al., 1976; Riemer, 1984). The di-sodium and di-potassium endothall formulations are preferred for the control of submersed macrophytes, but not for floating or emergent species. The alkylamine formulations can also control algal macrophytes and filamentous algae, but are not recommended for these or for submersed macrophytes, since they are toxic to animals at the recommended concentrations of 0.3 mg/l to 1 mg/l (Simsman et al., 1976).

Other organic herbicides which have frequently been used to control submersed or emergent macrophytes in the U.S., though they are not currently recommended for use in water supply reservoirs in Kansas, include amitrole, dalapon, dichlobenil, diquat, diuron, silvex, and simazine. Some chemical characteristics of each are presented in Anon. (1983). Of these, simazine, diuron, and silvex (in combination with endothall as aquathol+) have been
<table>
<thead>
<tr>
<th>Investigator(s)</th>
<th>Herbicides Studied</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Brooker &amp; Edwards (1973a &amp; 1973b)</td>
<td>Paraquat</td>
<td>Caused a decrease in photosynthesis and respiration, which later increased due to algal growth.</td>
</tr>
<tr>
<td>Brooker &amp; Edwards (1975)</td>
<td>Chlorthiamid, 2,4-D dalapon, dichlobenil diquat, maleic hydrazide, paraquat, terbutryne</td>
<td>Review of direct and indirect effects on aquatic and terrestrial organisms.</td>
</tr>
<tr>
<td>Butler (1977)</td>
<td>Many compounds</td>
<td>Review of the effects of pesticides (aquatic and terrestrial) on algae.</td>
</tr>
<tr>
<td>DeBussy (1969)</td>
<td>Substituted phenyl-ureas (Diuron)</td>
<td>Submerged plants and filamentous algae controlled with 0.2 to 0.4 mg/L; higher dosages needed for planktonic algae.</td>
</tr>
<tr>
<td>Fitzgerald &amp; Skoog (1954)</td>
<td>2,3-dichloronaphthoquinone</td>
<td>30-55 ppb killed cyanophytes without harming chlorophytes, higher plants, fish, or zooplankton. Repeated applications sometimes required.</td>
</tr>
<tr>
<td>Fitzgerald et al. (1952)</td>
<td>300 compounds</td>
<td>2,3-dichloronaphthoquinone selectively toxic to Microcystis aeruginosa at 2 μg/L.</td>
</tr>
<tr>
<td>Gratteau (1970)</td>
<td>300 compounds</td>
<td>Screened for effectiveness compared to copper sulfate.</td>
</tr>
<tr>
<td>Hawkins (1973)</td>
<td>Diquat</td>
<td>Successful control of blue-green algae in Israel. Lagarosiphon major cleared in New Zealand, but replaced by Nitella.</td>
</tr>
<tr>
<td>Hoffman et al. (1982)</td>
<td>Simazine, diuron</td>
<td>More effective than complexed copper in high pH pond water.</td>
</tr>
<tr>
<td>Hurlbert (1975)</td>
<td>Many compounds</td>
<td>Literature review of the secondary effects of pesticides (aquatic and terrestrial) on aquatic ecosystems.</td>
</tr>
<tr>
<td>Investigator(s)</td>
<td>Herbicides Studied</td>
<td>Comments</td>
</tr>
<tr>
<td>-------------------------</td>
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</tr>
<tr>
<td>Johnson &amp; Finley (1980)</td>
<td>Many compounds</td>
<td>A compilation of results of toxicity tests on fish and aquatic invertebrates.</td>
</tr>
<tr>
<td>Johnson &amp; Julin (1974)</td>
<td>Diuron</td>
<td>Review of use in fisheries; good potential as broad-spectrum herbicide; strongly absorbed; persistent and may bioaccumulate.</td>
</tr>
<tr>
<td>Kerst et al. (1973)</td>
<td>5-(5'-barbiturilidene)-rhodanine</td>
<td>Inhibited algal growths.</td>
</tr>
<tr>
<td>Kirby &amp; Shell (1975)</td>
<td>Karmex (diuron)</td>
<td>Controlled filamentous algae in hatchery ponds, but caused oxygen depletion and algal blooms.</td>
</tr>
<tr>
<td>Lindaberry (1973)</td>
<td>Amine oxide salts of endotheall</td>
<td>Used to control aquatic weeds.</td>
</tr>
<tr>
<td>Mulligan (1969)</td>
<td>2,4-D, silvex, fenac, simazine, endotheall, diquat</td>
<td>Review of use in controlling vascular plants and algae.</td>
</tr>
<tr>
<td>Norton (1973)</td>
<td>Simazine</td>
<td>Discussion of use, safety, application rates, secondary effects.</td>
</tr>
<tr>
<td>O'Neal &amp; Lembi (1983)</td>
<td>Simazine</td>
<td>Photosynthesis by filamentous algae reduced 50% by 1.1-4.7 μM dosages; effectiveness increased with increasing light intensity.</td>
</tr>
<tr>
<td>Otto (1970)</td>
<td>74 Compounds</td>
<td>Nine were more active than copper sulfate; one provided total control of filamentous blue-green algae; use in irrigation canals.</td>
</tr>
<tr>
<td>Otto et al. (1972)</td>
<td>Diquat, endotheall, complexed copper, copper sulfate</td>
<td>Various combinations tested.</td>
</tr>
<tr>
<td>Robson et al. (1976)</td>
<td>12 compounds, incl. terbutryne, triazines, cyanatryn, asulam, cutrine, and diquat</td>
<td>Tested for effectiveness against freshwater algae; success with diquat in the laboratory not matched under field conditions.</td>
</tr>
<tr>
<td>Investigator(s)</td>
<td>Herbicides Studied</td>
<td>Comments</td>
</tr>
<tr>
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<td>--------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Schwartz et al.</td>
<td>Simazine</td>
<td>0.45 mg/L controlled Anabaena circinalis, but not <em>A. flos-aquae</em>, for 2.5 years; more than 3 mg/L needed to control blue-green algae.</td>
</tr>
<tr>
<td>Simonin &amp; Skea</td>
<td>Cutrine, diquat</td>
<td>TLM data for fingerling brown trout; synergism noted.</td>
</tr>
<tr>
<td>Simsiman et al.</td>
<td>Diquat endothall</td>
<td>Literature on their fate in the environment.</td>
</tr>
<tr>
<td>Skaptason (1972)</td>
<td>Many compounds</td>
<td>Contains extensive bibliography on the use of herbicides to control aquatic vegetation.</td>
</tr>
<tr>
<td>Tooby (1976)</td>
<td>Compounds used in Great Britain</td>
<td>Effects on fisheries, including acute and chronic toxicity to fish and invertebrates.</td>
</tr>
<tr>
<td>Tucker &amp; Boyd (1978)</td>
<td>Simazine, copper</td>
<td>Used to control phytoplankton in catfish ponds; drastic reduction in phytoplankton followed by oxygen depletion.</td>
</tr>
<tr>
<td>Walker &amp; Evans (1978)</td>
<td>Quaternary ammonium compounds</td>
<td>Toxicity to <em>Chlorella</em> and duckweed investigated.</td>
</tr>
<tr>
<td>Way et al. (1971)</td>
<td>Paraquat</td>
<td>Ecological effects of addition to a small lake.</td>
</tr>
</tbody>
</table>
used in Kansas reservoirs and ponds (pers. comm., KDHE, Water Quality Section). Most are wide-spectrum herbicides, affecting both algae and macrophytes, and are generally added directly to the water, though some are sometimes sprayed on exposed weed beds following draw down. Silvex, also known as 2,4,5-T and one of the first organic herbicides used for macrophyte control, was registered for aquatic use in the U.S. in the late 1950's and early 1960's along with 2,4-D, diquat, and endothall. Today it is discouraged from use because of its association with dioxin (Esposito et al., 1980), a consistent contaminant in the herbicide. Each of these herbicides has been reported to successfully control various macrophyte weed problems, including some involving algae (e.g., see Table 4).

**Assessment**

There are a number of things to consider in assessing the use of organic herbicides to reduce the growth of macrophyte communities in water supply reservoirs (e.g., Hurlbert, 1975). Of particular importance is the possible disturbance of the natural role of these plants in aquatic habitats and possible effects, either direct or indirect, on other aquatic organisms. A thorough evaluation of the overall effectiveness of any control measure, whether for macrophytes or some other condition in the reservoir, must consider both the immediate desired effects and any side effects which may alter the environment sooner or later.

The macrophyte community of a reservoir can be involved in many natural ecological relationships, as mentioned in Section A of this chapter. Elsewhere in this report the focus may be more on the adverse conditions which excessive macrophyte growth can create. Too often the macrophytes, being more readily observed due to their size and proximity to the shore, are viewed as overly abundant when in actuality they are not. If adverse conditions seem to merit treatment and a macrophyte community is targeted for some degree of reduction, a clear understanding of its natural role in the functioning of that particular ecosystem should be incorporated into the decision making process. Ecological relationships involving aquatic vascular plants have been recently reviewed by Riemer (1984) and will be summarized below in relationship to the potential impact of their loss from the ecosystem. The plant community of most water supply reservoirs in Kansas is not dominated in either productivity or biomass by the vascular plants. This is due primarily to the light blockage effect of the turbidity levels common to these waters, as discussed earlier in this section and elsewhere (i.e., Chapter 2). Therefore, what little macrophyte growth is present may play an even more important ecological role.

According to the review of Riemer (1984), as well as other relevant literature, macrophyte communities, particularly the submersed ones, provide a food source for certain waterfowl (e.g., Martin et al., 1961) and are a link in the food chain of particular invertebrates and fish. In the case of invertebrates, it has been shown that a reduction in submersed macrophytes can lead to biomass and productivity declines (Korinkova, 1967; Smith & Isom, 1967; Dewey, 1986), in some cases as a function of decreased food supply (Lellak, 1965; Wetzel, 1975). Most often, instead of macrophytes being fed upon directly by insects, other invertebrates, and fish, it is the microscopic plants and animals which colonize them that serve as the food
supply. Such a disruption of the food chain could then have effects elsewhere. Submersed macrophytes not only provide habitat for a diversity of aquatic organisms, particularly the microscopic flora and fauna referred to above, but they may also serve as a refuge for certain fish which might otherwise be too heavily preyed upon (Regier, 1963; Walker, 1963; Eipper & Regier, 1965; Werner et al., 1983).

Chemical cycles in aquatic ecosystems are also influenced by the macrophytes, particularly through their effects on pH, oxygen, dissolved organic matter, and movement of chemicals from sediments to the water column, as reviewed by Wetzel (1975). Reduced oxygen conditions (Beaven et al., 1962; Haven, 1963; McCraren et al., 1969; Cope et al., 1970; Brooker & Edwards, 1973) and increased carbon dioxide conditions (Cope et al., 1970; Simpson & Pimentel, 1972; Brooker & Edwards, 1973a & 1975) have followed herbicide treatment. The release of nutrients by decaying vegetation, and/or reduced uptake of nutrients due to live vegetation loss, can make nutrients more available to unaffected organisms. Both plants and animals can be stimulated by such nutrient increases (Tatum & Blackburn, 1962; Crance, 1963; Walker, 1964; Boyd, 1971; Walsh et al., 1971; Brooker & Edwards, 1973a & 1975). Macrophytes are also known to remove pollutants from the water (Boyd, 1970; Steward, 1970; Bingham, 1973; Bates & Hentges, 1976) including nutrients, turbidity, and toxicants.

Considering the various ways in which macrophytes support the natural functioning of a reservoir, the potential effects of a macrophyte removal program on each role of macrophytes must be carefully evaluated. In Kansas reservoirs where such communities may be small to begin with, but not inconspicuous, even greater care is necessary in evaluating the impact of removal. This may be particularly the case with regard to refuge for certain fish for which habitat may be minimal for the entire reservoir, provided only by a localized, perhaps too conspicuous, community of macrophytes.

As emphasized throughout this report, any management practice targeted to control a particular species or type of organism must be considered as possibly affecting non-target organisms as well. Such effects are often overlooked because of the seemingly greater need to control the target organism or because these effects are more separated in time or space from the treatment. There is some evidence from the literature summarized below that the organic herbicides considered here can have effects both on non-target plants and on animals.

The two herbicides (2,4-D and endothall) approved for use in Kansas water supply reservoirs have both been shown to have direct and indirect effects on other members of the aquatic community at concentrations within or close to the range created by the recommended dosages. However, there is no evidence for significant biological magnification of either herbicide within the aquatic food chain. In general, aqueous concentrations of the active ingredient of these two herbicides, and the others mentioned earlier in this section, range from 0.1 mg/l to 2 mg/l when they are applied at the recommended dosages, but actual concentrations in a reservoir can vary and are often considerably higher temporarily in localized zones where higher dosages are applied and allowed to migrate into peripheral areas.
2,4-D is an auxin-type herbicide chemically similar to natural chemicals in flowering plants which stimulate growth as elongation of portions of the plant body. Such elongation ultimately kills the susceptible plant (Moreland, 1980). This mode of action suggests that 2,4-D may have less effect on algae. Boyle (1980), having examined experimental ponds receiving 2,4-D, reported no inhibition of phytoplankton growth over a four month period following addition. Laboratory studies have reported similar conclusions of no inhibition (Butler, 1963; Elder et al., 1970; Poorman, 1973). In a review by Butler (1977), 2,4-D was not toxic to most algae except at concentrations well above those created by recommended dosages. It should be noted that the effects of 2,4-D on aquatic plants are related to the particular formulation of the active ingredient. Among the different types, the esters of 2,4-D are more toxic than the amides to all plants in general, according to the literature cited above.

The effects of 2,4-D on aquatic animals is also related to the formulation, with the esters being more toxic (Lawrence, 1962; Hughes & Davis, 1963; Mullison, 1970; Sanders, 1970; Johnson & Finley, 1980). Active ingredient concentrations of either formulation causing 50% mortality in 96 hours (i.e., 96-hr LC50), or some other measured effect, are generally lower for the invertebrates compared to fish (e.g., Sanders, 1970; Hurlbert, 1975; Jones, 1975; Rehwoeldt et al., 1977; Sigmon, 1979). In these studies the concentrations causing 50% mortality, particularly for fish, were generally, though not always, above those (0.5 mg/l to 2.0 mg/l) typical of treated waters. Concentrations in the range of those created by the recommended dosages have been reported to cause no direct toxic effects in whole-system field studies (Smith & Isom, 1967; Boyle, 1980; Scott et al., 1981). However, indirect effects, such as insect losses with the loss of macrophyte substrate (Smith & Isom, 1967) and stimulation of some non-target plants from nutrient release (Boyle, 1980), are indicated.

The indirect effects of 2,4-D treatment on non-target organisms have been reported to include most of the indirect effects of herbicide treatment discussed in general earlier in this section. Reports of increases in non-target organisms include bacteria (Petruk, 1965), macrophytes (Van Der Weij, 1966; Cope et al., 1970), and invertebrates (Gasaway, 1962; Hurlbert, 1975). Many cases of increases in fish populations have also been reported as the desired result of reducing macrophyte communities by whatever means, as discussed in Chapter 6. Reports of decreases in non-target organisms have concentrated on those of animals, with most studies considering that any declines in non-target plants would also be of benefit. This should not be assumed. Reports of decreases in animals include insects (Smith & Isom, 1967), other invertebrates (Beaven et al., 1962; Haven, 1963), and fish (Beaven et al., 1962). Decreases in dissolved oxygen have been implicated in some of these animal decreases (Beaven et al., 1962; Haven, 1963). Declines in pH, related to free carbon dioxide increases, have also been associated with 2,4-D treatment (Cope et al., 1970), but were not related to any decreases in animals.

Turning now to endothall, this herbicide has been described as affecting several plant processes including respiration (Maestri & Currier, 1958), membrane disruption (Maestri, 1967), inhibition of protein synthesis (Mann et al., 1965), and retardation of lipid metabolism (Mann & Pu, 1968).
Simsiman et al., (1976), in reviewing this subject, concluded that the actual mechanism of action for these effects is still unknown. Seeming to have a broader range of physiologic effects than 2,4-D, endothall might be expected to produce more side effects, although it has been reported to be rapidly inactivated in water (Hiltebran, 1962). Unfortunately there have been fewer investigations of the side effects of endothall than of 2,4-D. In one of the few studies of the direct toxic effects on non-target plants, Walsh (1972) examined a few species of phytoplankton and reported effects only at concentrations well above the 1 mg/l generally encountered with treatment (Simsiman et al., 1976). The alkylamine formulations of endothall have been reported to be toxic to certain filamentous algae at the recommended dosage (Simsiman et al., 1976), and are thus likely to be toxic to other algae as well. The alkylamine formulations, because of their toxicity to animals, are not preferred even though they are more toxic to target plants than the di-sodium or di-potassium endothall formulations.

The effects of endothall on aquatic animals have been given slightly more attention than have the effects on plants, with particular concern focused on the more toxic alkylamine formulations. This form of endothall has been demonstrated to be toxic to invertebrates, amphibians (tadpoles), and fish at concentrations in the range (or slightly higher, at a few mg/l) of those created by recommended dosages, as reviewed by Simsiman et al. (1976) and compiled by Johnson and Finley (1980). The few field studies in treated waters have reported no immediate or longer-term effects on aquatic fauna, including zooplankton (Serns, 1974), other invertebrates (Steucke, 1963), and fish (Steucke, 1963; Yeo, 1970; Serns, 1974).

The indirect effects of endothall on non-target plants and animals at first inspection of the literature appear to be less than those for 2,4-D. However, the indirect effects reported for 2,4-D were the result of the loss and subsequent decomposition of the macrophyte community, which should also result from a successful endothall treatment, or for that matter any other effective herbicide treatment. Therefore, the lack of reports in the literature of indirect effects of this herbicide (or any others) appears to be indicative of too little research rather than the absence of such effects. There is little doubt of the potential effects, either direct or indirect, of the alkylamine formulations; thus, they should not be used. The di-sodium and di-potassium endothall formulations, though less toxic to all organisms (target and non-target alike), have still been found to provide weed control, as reviewed by Simsiman et al. (1976).

The discussion of the general effects of macrophyte removal included earlier in this section pertains not only to 2,4-D and endothall, but to all herbicides. A review of the specific direct and general indirect effects of the other herbicides mentioned earlier will not be presented here, since these herbicides are not currently recommended for use in Kansas water supply reservoirs. Recommendations could change in the future, and some useful initial sources of information concerning the effects of a variety of herbicides on both target and non-target organisms are included in Table 4.

In addition to the direct and indirect effects of herbicides on aquatic organisms, consideration must also be given to their potential to cause acute and long-term effects on terrestrial organisms after the water is
extracted for use (e.g., Brooker & Edwards, 1975). Specifically, herbicide residues and metabolites must not be present in concentrations that will adversely affect human health, irrigated crops, farm animals, lawns, trees, shrubs, domestic animals, pet fish, or the purity of foods (e.g., milk, fish, etc.). In this regard, there are four major considerations:

1) Toxicity to terrestrial organisms.

Fortunately, most herbicides approved for use in reservoirs have a relatively low mammalian toxicity, and therefore do not pose an acute hazard to mammals even when consumed at the maximum recommended concentrations (e.g., Brooker & Edwards, 1975). Endothall has a relatively high mammalian toxicity (Woodford & Evans, 1965), but decomposes rapidly in water (Hiltebran, 1962). Plants, including commercial crops, are naturally somewhat susceptible to herbicide damage, since herbicides are designed to kill plants, and a potential for damages exists if irrigation water is withdrawn from a reservoir shortly after herbicide application (Brooker & Edwards, 1975).

2) Herbicide decomposition rates relative to the interval between herbicide application and water withdrawal.

Herbicide concentrations in reservoirs can be diminished by a variety of mechanisms, including microbial degradation, photolysis, adsorption on colloids or plant material, dilution, dispersion, and volatilization. For example, 2,4-D is photoreactive and can be decomposed by microbial action (Mulligan, 1969). Hence, given sufficient time, the concentration of an aquatic herbicide in water will drop to a negligible level (e.g., Adams, 1983). To reduce the risk of exposure to humans, animals, crops, etc., some regulatory authorities recommend a minimum interval of time between herbicide application and water withdrawal. For the eight herbicides approved for use in Great Britain, this interval is from 10-84 days, including 21 days for 2,4-D applied at a maximum concentration of 5 mg/L (Brooker & Edwards, 1975). In Kansas, KDHE recommends that prevailing wind directions and water currents be considered and that at least 7 days should elapse before water treated with 2,4-D or endothall reaches the intake structure (pers. comm., KDHE, Water Quality Section, 1985).

3) Application method and formulation used

The herbicide concentration that actually occurs in the water is strongly dependent upon the application method, i.e., whether the herbicide is applied directly to the plants or to the water, whether application is limited to shoreline areas or extended to open water, and whether small isolated areas or large areas are treated. Also, some formulations are more water soluble than others, and solutions yield higher aqueous concentrations than pelletized forms.
4) Removal of residues by water treatment

In general, water treatment plants are designed to remove particles rather than soluble chemical species. Residues adsorbed on clay particles, microorganisms, and detritus will be well removed, but residues in solution are not likely to be removed unless activated carbon is used. However, chlorination may chemically alter some herbicides, particularly those with aromatic structures (such as 2,4-D), double bonds, or certain nitrogenous functional groups.

In evaluating any program where organisms are targeted for removal, whether by chemical treatment or other means, the effects which the loss of the organisms may have on the rest of the ecosystem must be considered along with direct effects and any other indirect effects. It should once again be reemphasized that such programs are more often directed toward treating the symptoms of a problem rather than the cause. Considering macrophyte problems, the most likely causes are elevated nutrient conditions or an imbalance in predator-prey relationships resulting in excessive grazing on the competing phytoplankton. With only symptoms treated and causes ignored, such programs will only be temporarily successful at best and may have to be repeated indefinitely, at considerable cost to the public and perhaps to the environment as well.

The use of organic herbicides for macrophyte control can have significant impacts, both short-term and long-term, on water quality. The short-term effects are largely negative and include severe oxygen depletion, nutrient release, stimulation of algal blooms, and release of large amounts of organic matter into the water (Brooker & Edwards, 1975; Hestand & Carter, 1978; Cooke, 1983). Increases in both TOC and THM precursor concentrations can be anticipated soon after herbicide addition to control excessive growths of macrophytes. All of these short-term negative effects can be mitigated by applying the herbicide early in the growing season and by treating only small areas at a time in the recommended manner. The long-term effects of herbicide addition on drinking water quality and on reservoir capacity can be very positive if herbicides are used as a preventive measure and algal blooms are also prevented (e.g., by use of copper sulfate). Such long-term effects include decreased sediment accretion rates, increased dissolved oxygen, decreased TOC and THM precursor concentrations, and fewer taste and odor problems. However, these benefits may be gained at the expense of the sport fish population and serious impairment of the aquatic ecosystem.

It is worth noting here that there is an excellent alternative to herbicide addition for macrophyte control. As discussed in Section J, mechanical harvesting achieves virtually the same benefits as herbicide addition, but with far fewer negative side effects. Properly done, harvesting is both effective and ecologically sound. Furthermore, some have found harvesting to be less expensive than herbicide use. Although not yet fully developed for widespread use in multipurpose reservoirs, biological control methods (see Section L) hold promise for the future as an even better alternative to the use of herbicides. For example, Hestand and Carter (1978) compared application of herbicides (including endothall formulations) to
grass carp in a controlled study. The herbicides rapidly killed the plants, releasing nutrients and stimulating dense phytoplankton blooms, while the grass carp removed the vegetation gradually, causing no increase in either nutrients or phytoplankton.

H. NUTRIENT INACTIVATION

Background

All aquatic plants have certain nutritional requirements that must be met for growth to occur. According to Liebig's "Law of the Minimum," the rate of growth of any organism will be proportional to the concentration of the nutrient that is least abundant relative to nutritional needs. Nutrients can be classified as macronutrients, those needed in large quantities, or micronutrients, those required in trace quantities.

The principal macronutrients for aquatic weeds, and for most other organisms, are carbon, oxygen, hydrogen, nitrogen, and phosphorus. Since weeds can use carbon dioxide as a carbon source, and since water provides an abundant supply of oxygen and hydrogen, either nitrogen or phosphorus will usually be the limiting macronutrient for weed growth. There are many micronutrients, including iron, silica, copper, calcium, etc., but these are generally considered to be abundant relative to nutritional needs (Sawyer, 1968). It is possible for micronutrient availability to limit the growth of certain species, thereby influencing population dynamics; but scientific knowledge of this phenomenon is quite limited. In any event, macronutrient availability should limit primary productivity (the rate of growth of plants and algae) for mixed populations of organisms, such as those occurring in reservoirs.

There is a great body of literature on the subject of lake fertility in relation to the availability of the macronutrients nitrogen and phosphorus. There is general consensus that the primary productivity of a majority of reservoirs is phosphorus limited, but cases of nitrogen limitation have been documented. Dual limitation is also quite possible, since an increase in either nitrogen or phosphorus may stimulate the growth of individual species that are not limited by other nutrients. For example, addition of phosphorus might stimulate the growth of blue-green algae (cyanobacteria), since certain species can fix molecular nitrogen and are therefore not nitrogen limited, while addition of nitrogen might stimulate the growth of green algae, which cannot fix nitrogen. Simple algal bioassays can be used to determine the limiting nutrient in a given reservoir.

Since there are currently no practical ways to remove biologically available nitrogen from reservoirs, and since nitrogen-fixing blue-green algae, which are phosphorus limited, are often associated with taste and odor problems and fish kills, most nutrient inactivation techniques are designed to limit the availability of phosphorus. Also, since rooted aquatic plants can obtain phosphorus directly from sediments, nutrient inactivation is primarily directed at control of planktonic algae.
The selection of a nutrient inactivation technique should be based upon careful consideration of the sources of phosphorus in the reservoir to be treated, i.e., the relative amounts of phosphorus in the water column, in the bottom sediments, and in influent streamflow and runoff. If most of the phosphorus is in the water column and has accumulated there over a long period of time, as is often the case in a reservoir with a very low water exchange rate, then removal of phosphorus from the water column by precipitation and sedimentation should provide effective weed control. If a substantial amount of phosphorus is being released from the sediments, which generally occurs when nutrient rich sediments are exposed to anaerobic conditions, then effective nutrient inactivation requires prevention of phosphorus release from the sediments.

When a large amount of phosphorus enters the reservoir in streamflow and runoff, nutrient inactivation will only be effective if practiced continuously or if combined with substantial reductions in point and/or non-point sources of phosphorus. Continuous nutrient inactivation, accomplished by continuously adding chemicals to the water entering a reservoir, is not likely to be an economical approach in most cases, but has been successfully accomplished at a pumped-storage (off-stream) reservoir in Great Britain (Hayes et al., 1984).

Many different chemicals have been tested to determine their ability to precipitate phosphorus from the water column or to prevent its release from sediments. These chemicals include aluminum sulfate (alum), sodium aluminate, ferric sulfate, lime, gypsum, fly ash, zirconium, lanthanum, ion exchange resins, clay, polyelectrolytes, zeolites, and aerobic lake muds (Peterson et al., 1974). Many of these are generally unacceptable, due to their cost, toxicity, or ineffectiveness. Alum and sodium aluminate are the most widely used, but ferric sulfate, lime, and fly ash have also been successfully employed.

Alum and sodium aluminate are very effective for precipitating phosphorus from the water column and, if higher dosages are applied, for preventing release of phosphorus from sediments; and many successful applications have been reported (Peterson et al., 1973 & 1974; Dunst et al., 1974; EPA, 1980a, 1980b, and 1980c; Cooke & Kennedy, 1981; Garrison & Knauer, 1984). Since alum is an acid and sodium aluminate a base, care must be taken to maintain a pH that will protect fish and plants and retain the phosphorus in an insoluble form (e.g., EPA, 1980b). The optimum pH for phosphorus precipitation is about 6.0, but removal should be effective in the range of 5.5-8.5. Above pH 8.5, both phosphorus and aluminum may be released into the water, causing fertilization and toxicity, respectively. A high rate of photosynthesis can produce a pH value of 10 or higher in certain waters, but values over 8.5 would not be expected to occur in the bottom waters of most Kansas reservoirs, except perhaps in very shallow areas. Kennedy and Cooke (1981) have described procedures for determining the required alum dosage and techniques for applying the alum to a reservoir.

Ferric sulfate can effectively precipitate phosphorus over a broader pH range than aluminum salts. However, under anaerobic conditions iron can be reduced, causing an increase in soluble iron and possibly allowing the
release of previously precipitated phosphorus. The amount of phosphorus released under anaerobic conditions would depend upon the amounts and forms of iron and sulfur present, as well as the depth and composition of the sediments. No evidence was found in the literature review that ferric sulfate has been successfully used to prevent release of phosphorous from sediments.

Higgins et al. (1976) examined the use of fly ash, gypsum, and lime for nutrient inactivation. Fly ash exhibited good sediment sealing properties, but the ability of fly ash to remove phosphorus from the water column varied with the source of the fly ash. Gypsum was ineffective at pH values below 10. Fly ash and lime, used in combination, effectively removed phosphorus and trapped it in the sediments. Since lime is a strong base, it must be carefully applied to avoid damage to fish and other aquatic organisms.

Harper et al. (1983) proposed the use of water treatment plant sludges for nutrient inactivation. Surface application of 200 mg/L of alum sludge to lake water removed significant amounts of chlorophyll, total and ortho phosphorus, TOC, turbidity, and trace metals; and a thin layer of alum sludge prevented release of phosphorus from anaerobic sediments. Calcium carbonate sludge did not prevent release of phosphorus.

When control of nutrient release from the sediments is the primary concern (i.e., when removal of nutrients from the water column is unnecessary), bottom sealing can be an effective nutrient inactivation technique. Various materials potentially useful for bottom sealing have been discussed by Dunst et al. (1974), including plastic sheeting, rubber liners, soil, sand, dredge spoils, fly ash, clay, hydrous metal oxides, and gels. Born et al. (1973a) were able to control both nutrient release and macrophyte growth using plastic sheeting and gravel to cover littoral sediments. The use of bottom covering to control macrophyte growths is discussed in Section K of this chapter.

Assessment

Nutrient inactivation is most likely to succeed in reservoirs with low water exchange rates, nutrient-rich sediments, and low external phosphorus loadings. Such conditions are much more typical of older natural lakes than the man-made reservoirs of Kansas. Most reservoirs in Kansas are rapidly flushed with nutrient rich water and are rapidly filling up with inorganic sediments. Under such conditions, nutrient inactivation is expected to be effective only if used on a frequent or continuous basis or if combined with effective control of external phosphorus loadings, since any phosphorus removed from the water column by precipitation would be quickly replaced with fresh phosphorus from the watershed.

If phosphorus is prevented from entering the reservoir by control of upstream sources, the phosphorus in the water column of a typical Kansas reservoir will be quickly flushed out, leaving sediment release as the major source of phosphorus. Phosphorus release from sediments in a typical Kansas reservoir may be less significant than other phosphorus sources, but most reservoirs do experience anoxic conditions during the warmer months and a certain amount of phosphorus release undoubtedly occurs, although the large
amounts of inorganic sediment entering most reservoirs may retard phosphorus release. If external phosphorus loadings to a reservoir are substantially reduced, sealing of the sediments may help to to keep the phosphorus concentration in the water column low enough to prevent excessive weed growth.

Based upon experiences elsewhere, alum (perhaps in combination with sodium aluminate or fly ash) appears to be the best chemical to use to treat Kansas reservoirs. Other chemicals, although potentially useful, do not have alum's track record of successful application, and would require more extensive testing. Also, alum has been shown to be capable of providing long-term control (Garrison & Knauer, 1984).

Nutrient inactivation, if effective, will almost certainly improve water quality, since it will cause reduced growth of algae, especially blue-green algae, and will reduce the amount of decaying organic matter reaching the hypolimnion. It should, therefore, result in reduced oxygen depletion, lower levels of TOC and THM precursors, and lower levels of dissolved iron, manganese, and sulfide. All of these changes would improve drinking water quality and many of them would also be advantageous for sport fish.

On the other hand, nutrient inactivation can potentially produce certain adverse effects, both direct and indirect, on fish and other aquatic organisms. Direct effects include toxicity of the chemical(s) used to precipitate phosphorus, sudden changes in pH, and smothering of benthic organisms. These effects can be minimized or avoided by selecting chemicals and chemical dosages that are non-toxic to the species of interest and applying them carefully (e.g., EPA, 1980b). Potential indirect effects include reduced fertility (i.e., decreased fish production), succession of undesirable species, increase growth of rooted vegetation due to increased water clarity, and increase sulfide production in the hypolimnion due to reduction of the sulfate component of alum. These effects can be minimized by reducing phosphorus only to the extent needed to control particularly troublesome weeds. Any decrease in fertility may be offset by improvements in habitat.

In summary, nutrient inactivation is unlikely to be an effective means of weed control in most Kansas reservoirs, but if effective and properly done it should have mostly positive effects on both fish and water quality.

I. LIGHT BLOCKAGE

Background

Light blockage is a management technique designed to control the growth of aquatic plants by reducing the amount of light available for photosynthesis. This is most commonly accomplished through the use of dyes (e.g., aniline dyes and various commercial products such as Aquashade) or shades (e.g., floating black plastic sheeting). The use of dyes and shades has been reviewed by Nichols and Shaw (1983).
Assessment

This technique is sometimes appropriate for ponds, especially those that are fertilized to increase fish production, but it is neither feasible nor appropriate for reservoirs, especially multipurpose reservoirs. Dyes provide only short-term control, since they are diluted, degraded, and adsorbed. Also, they cause an increase in water temperature and may be toxic to humans or aquatic organisms. Shades prevent use of the covered area of the reservoir for fishing, boating, and swimming; and they prevent aeration of the water, leading to anoxic conditions.

In Kansas reservoirs, a certain amount of light blockage occurs naturally, due to the high levels of turbidity. This effectively limits the growth of algae and macrophytes in many reservoirs, especially larger ones. It is generally impractical to attempt to artificially increase turbidity by directly adding clay to a reservoir, but it is conceivable that the turbidity of some reservoirs could be indirectly controlled by alterations in design (see Chapter 14), by watershed management practices (see Chapter 15), or by some of the other management techniques described in this report.

J. MECHANICAL HARVESTING

Background

Mechanical harvesting includes cutting, uprooting, crushing, and removal of nuisance vegetation by physical means, whether done by hand or with mechanical devices. Hand methods include hand-pulling, hand-raking, and hand-cutting (Riemer, 1984). Mechanical devices include mowers, cutting bars, drag chains, crushers, and high-pressure nozzles (Livermore & Wunderlich, 1969; Burton et al., 1979; Riemer, 1984). Cut, crushed, or dislodged weeds may be left in the water or removed from the water; but most modern harvesters are equipped to remove the weeds from the water. Harvesting may be used to control submersed, emergent, or floating vegetation, and is most frequently used to control submersed rooted vegetation, but no practical methods for harvesting algae have been developed. Therefore, this discussion focuses on macrophytes.

Several other reservoir management methods are closely related to mechanical harvesting. Dredging (see Chapter 12) and draining/cleaning (see Chapter 13) can be viewed as drastic forms of mechanical harvesting where the primary purpose is reservoir deepening. Level adjustment (see Chapter 7) may result in the destruction of macrophytes by dessication or freezing; and exposed weeds may be removed by hand or by mechanical means whether they are dead or alive. Stocking of grass carp or other plant-eating fish (see Chapter 4, Section L) can be considered to result in biological harvesting.

Mechanical harvesting has, in some cases, been found to be an effective means of controlling fertility (i.e., Wile, 1978); but this can only be expected to occur when the mass of nutrients removed along with the weeds (or the mass of nutrients being recycled by the weeds from the sediments to the water column) is large compared to the external nutrient loads entering
the reservoir from the watershed (Burton et al., 1979). Since this is not expected to be possible in Kansas reservoirs, due to high phosphorus loadings and the limitation of macrophyte growth by high turbidity, harvesting in Kansas should be viewed primarily as another means of treating the symptoms of eutrophication rather than as a cure or preventive measure for the underlying problem of eutrophication.

In general, mechanical harvesting is reasonably effective in achieving its immediate objective, the destruction and removal of nuisance vegetation. However, as with other cosmetic approaches to weed control, the benefits may be quite temporary, with fresh stands of vegetation appearing in weeks or months. New growth may occur from the shoots or cut stems left by the harvester or from cut pieces of plants that find their way to shallow water and root. Therefore, harvesting must commonly be repeated at regular intervals to achieve a reasonable degree of control. The interval between harvestings depends on a number of factors, including plant species, cutting depth, harvesting method, season, weather conditions, turbidity, nutrient availability, etc. Nevertheless, it is sometimes found that one or several harvests in one year will significantly reduce plant biomass in the following year (e.g., Dunst et al., 1974; EPA, 1980a).

The advantages and disadvantages of mechanical harvesting have been discussed at length in the literature (e.g., Livermore & Wunderlich, 1969; Carpenter & Adams, 1977; Burton et al., 1979; EPA, 1980a; Carpenter, 1983; Nichols & Shaw, 1983; Cooke, 1983; Mikol, 1984; Engel, 1984; Riemer, 1984) and are outlined only briefly here. Potential advantages include:

1. Opening of boating lanes;
2. Clearing of swimming areas;
3. Improved angling;
4. Reduced oxygen depletion, assuming that the harvested plants are physically removed from the water;
5. Reduced internal cycling of nutrients;
6. Avoidance of the use of toxic chemicals;
7. Reduced likelihood of winterkill;
8. Reduced sediment accretion rates;
9. Removal of nutrients incorporated into plant biomass (though the amounts removed are rarely substantial relative to the external loading);
10. Increased fish production due to a shift in biomass from macrophytes to phytoplankton; and
11. Increased growth of more desirable plants.
Some of the potential disadvantages of mechanical harvesting are:

1. Loss of habitat for fish and for fish-food organisms;

2. Destruction of a fraction of the fry inhabiting the vegetation;

3. Stimulation of algal blooms (a relatively frequent side effect of harvesting);

4. Failure to remove the basal portions of the plants, resulting in rapid regrowth, and failure to remove all of the cuttings, resulting in the spread of the plants by vegetative means;

5. High initial cost due to the need to purchase the harvesting equipment;

6. Problems associated with disposal of the harvested weeds, since most potential uses for the plants (other than its use as compost) are impractical in the U.S.;

7. Short-term increases in turbidity due to resuspension of sediment and detritus;

8. Leaching of nutrients and organic matter from severed stems;

9. Increased erosion in the litteral zone;

10. Increased light penetration (due to removal of the canopy), stimulating growths of algae and benthic organisms;

11. Increased nutrient availability for phytoplankton growth; and

12. Difficulty of operating harvesters along highly irregular or rocky shorelines and around piers, trees, etc.

Assessment

Because the water in most Kansas reservoirs is quite turbid, macrophytes are seldom abundant enough to require treatment. However, macrophytes are a nuisance in some reservoirs and they are expected to become a more serious problem in the future as reservoirs become shallower due to siltation. In the event of a macrophyte problem, a reservoir manager must give serious consideration to harvesting, since the only other short-term control method that is sufficiently developed for general use is herbicide addition.

Setting aside, for the moment, the question of cost, mechanical harvesting appears to be an excellent means of achieving short-term control of macrophytes. All of the potential advantages cited above would or could be realized in Kansas (although the reduction in internal nutrient cycling and the removal of nutrients in the plant biomass would, in all likelihood, be inconsequential); and few of the potential disadvantages cited above would pertain, since Kansas reservoirs are fertile and turbid whether or not
macrophytes are harvested. Over harvesting could certainly have a significant adverse effect on sport fish populations, since a certain amount of macrophyte habitat is highly beneficial to fish and fish-food organisms (see Section 6, Organic Herbicides); but this problem can be avoided by selective harvesting (e.g., Engel, 1984).

The cost of mechanical harvesting is indeed rather high, especially when multiple harvests are required; and the cost is also quite variable from one location to another. Cost surveys have produced estimates of $50 to $140 per hectare (Dunst et al., 1974), $103 to $411 per hectare (Burton et al., 1979), $173 to $250 per hectare (EPA, 1980a), and $148 to $490 per hectare (Cooke, 1983). However, the cost of the major alternative to harvesting, repeated applications of a herbicide, is also quite high and may be substantially higher (e.g., Livermore & Wunderlich, 1969; Cooke, 1983; Conyers & Cooke, 1983). Furthermore, harvesting offers some important advantages over herbicide addition, including 1) greater effectiveness (e.g., Conyers & Cooke, 1983); 2) removal of organic matter and nutrients from the reservoir, reducing oxygen depletion and plant biomass; 3) reduced interference with recreation, since herbicides may require a waiting period before recreational activities are resumed (Cooke, 1983); 4) avoidance of the potential short-term and long-term side effects associated with the introduction of toxic chemicals to the environment (Livermore & Wunderlich, 1969); and 5) the potential for beneficial use of the harvested weeds.

Mechanical harvesting is expected to have primarily positive effects on water quality in Kansas, assuming that algal blooms are kept in check by high levels of turbidity and that cuttings are removed. Resuspension of bottom sediments should cause only a temporary and insignificant increase in turbidity, and any increase in erosion should be more than offset by a decreased siltation rate and greater compaction of the sediments. Provided that adequate habitat is left for fish and fish-food organisms, harvesting is expected to be both ecologically sound and reasonably effective in providing short-term control of macrophytes. By removing vegetation from the water column prior to senescence, harvesting is also expected to reduce TOC and THM precursor concentrations.

**K. SEDIMENT COVERAGE**

Sediment covering is a reservoir management technique intended to control macrophyte growth and, directly or indirectly, to control release of nutrients from the sediments. A layer of impermeable material (e.g., plastic, nylon, or rubber sheeting) is put in place, perhaps weighted down with rocks or gravel. Some macrophytes can be controlled with permeable coverings, such as fiberglass screens (Engel, 1984) or burlap (Nichols & Shaw, 1983). The permeable coverings retard macrophyte growth (and the associated nutrient cycling), but do not prevent release of phosphorus from the sediments. Impermeable coverings need to be vented to prevent gas entrapment and the resulting ballooning and uplift (Perkins, 1984).
Assessment

Sediment covering is generally inappropriate for Kansas reservoirs. High turbidity limits most macrophyte growth to a depth of about three meters, where it can be more easily controlled by other methods, and sediments are generally not a significant nutrient source. Furthermore, the coverings would be rapidly overlaid with a layer of freshly deposited sediment, allowing plants to root on top of the covering.

L. BIOLOGICAL CONTROL

Background

Biological control methods are those management techniques intended to control algae or macrophyte growths by increasing the population of a particular grazer, predator, competitor, or pathogen, or perhaps a group of such organisms. This may be accomplished directly (e.g., by inoculation, introduction, or supplemental stocking) or indirectly (e.g., by fertilization, habitat modification, or manipulation of the food chain).

This section of the report specifically addresses those methods in which populations of organisms able to control the growth of algae or macrophytes are directly increased. Indirect methods, including fertilization (Chapter 6), habitat modification (e.g., Chapter 7), and biomanipulation through fish stocking (Chapter 5), are discussed in other sections or chapters of this report. Of course, virtually every management technique will have some impact upon the various organisms in the reservoir, and some of these impacts can be usefully exploited as discussed in various chapters of this report.

Stocking of grazers (organisms that feed directly upon primary producers) has been one of the most commonly used biological control methods. Reviews by Mulligan (1969), Dunst et al. (1974), Schuytema (1977), Nichols & Shaw (1983), and Cooke (1983) describe the use of a variety of grazers, including protozoans, zooplankton, fish, birds, insects, mites, snails, crayfish, turtles, and mammals. Of these, fish have been the most extensively tried and studied, but there has also been considerable interest in the potential use of herbivorous zooplankton to graze algae.

Nichols and Shaw (1983) identify a number of fish that can potentially be used to control macrophytes, including common carp (Cyprinus carpio), roach (Rutilus rutilus), rudd (Scardinius erythrophthalmus), Tilapia species (including Tilapia zillii and Tilapia mossambica), silver dollar fish (Metynnis roosevelti and Mylossoma argenteum), white amur (Ctenopharyngodon idella, commonly known as grass carp), and hybrids of the white amur. Several of these species cannot withstand cold weather (e.g., Tilapia spp. and silver dollar fish), and can therefore not be used in Kansas for more than a single growing season. Others (e.g., carp) may effectively control macrophyte growths, but can create other nuisances, such as deterioration of water quality, excessive removal of macrophytes causing elimination of
valuable habitat, increased phytoplankton growth, or decline of sport fish populations.

The white amur (or grass carp) has been the most extensively studied herbivorous fish (Nichols & Shaw, 1983) and has been successfully used in a number of locations to control macrophytes. It can survive cold temperatures, and does not reproduce well in many locations (and is therefore not likely to become overabundant). Efforts have been made to develop sterile hybrids for use in locations where there is concern that excessive proliferation of grass carp could be a nuisance (e.g., Sutton et al., 1981; Shelton, 1983). Grass carp can potentially stimulate phytoplankton blooms by releasing plant nutrients into the water column, but there is conflicting evidence in this regard (Hestand & Carter, 1978; Richard et al., 1984). Grass carp can clearly control macrophyte growth, but a number of questions concerning their secondary and long-term effects still need to be answered, including their role in transmission of fish diseases, their effect on sport-fish populations, their impact on nutrient cycling, their feeding selectivity, their ability to reproduce in various environments, and their impact on fish habitat (EPA, 1980a; Cooke, 1981 & 1983; Nichols & Shaw, 1983).

Attempts to control algae with planktivorous fish have not been too successful, in part because some of the more troublesome algal species are not readily digestible (e.g., Dunst et al., 1974). Some fish such as the gizzard shad (Dorosoma cepedianum), though known for their ability to consume phytoplankton (Drenner et al., 1984), have not been thoroughly studied as a control method. There have also been a number of attempts to control algal blooms using herbivorous zooplankton, such as Daphnia pulicaria and Daphnia pulex (Carlson & Schoenbber, 1983; Osgood, 1984; Vanni, 1984); but addition of such organisms to a reservoir will provide only short-term control, if any, unless conditions are altered to encourage their growth and survival. Since herbivorous zooplankton occur naturally, it makes better sense to alter their population through biomanipulation, e.g., stocking of large predator fish that will, by reducing the population of planktivorous fish, indirectly control the population of herbivorous zooplankton, as discussed in Chapter 5.

A number of investigators have sought to control algae and macrophytes with pathogens, including pathogenic fungi (Freeman et al., 1981) cyanophages (e.g., Safferman & Morris, 1964; Desjardins, 1983), actinomycetes (Martin, 1976), and myxobacteria (Burnham & Fraleigh, 1983). Although some of these attempts have met with limited success in laboratory experiments, field experience is very limited, and such methods must be considered experimental at the present time. Use of pathogens holds promise as a short-term control measure, but does not appear useful as a long-term control measure. Even if repeated inoculations are made, a build up of resistant species is likely to occur.

Since plants are often in direct competition for nutrients and light, it should be possible to control growths of undesirable plants by encouraging the growth of more desirable plant species. Examples include control of macrophytes by stimulating phytoplankton growth (Mulligan, 1969), fertilization to cause a shift from blue-green algae to green algae (see Chapter 6),
cultivation of water hyacinths (which cannot withstand freezing) to remove nutrients and shade phytoplankton, and cultivation of spikerush (Eleocharis coloradoensis) to displace offending macrophyte species (Nichols & Shaw, 1984). Carpenter (1983) has suggested cultivation of macrophytes with low nutrient cycling capability as a means of reducing internal nutrient cycling, but the idea has not yet been tested. Fertilization of water supply reservoirs is generally considered inappropriate, and there have been few reports of successful field-scale control of algae or macrophytes through cultivation of more desirable plant species.

**Assessment**

Most biological control methods are still in a state of development and are not ready to be adopted for widespread use. Some methods that are in use in small ponds or single purpose reservoirs are unsuitable for use in multipurpose reservoirs.

One major drawback to biological control methods is that the population of organisms used for control must be sustained if long term control is to be achieved. Attempts to create an artificial shift in the population of a particular organism or group of organisms will generally be neutralized in a short period of time as the ecological balance of the reservoir is restored. Hence, reservoirs inoculated with viruses or zooplankton or myxobacteria would be expected to rapidly revert to their prior condition. Introduction of exotic species can potentially cause permanent or long-term changes in the ecological balance of a reservoir, but the effects of exotic species on other organisms and on water quality are difficult to predict; and until more research is done, such practices must be viewed with great caution.

In Kansas, most multipurpose reservoirs experience much greater problems with algae than with macrophytes. The most promising method for biological control of algae is to increase grazing pressure by restructuring the food chain, i.e., through biomanipulation. This method is discussed in Chapter 5, since it sometimes involves direct manipulation of fish populations. Because this manipulation involves sport fish, there is strong motivation for long-term maintenance of the population of control organisms. A method of this sort might one day be the preferred way to control algal blooms in fertile reservoirs, but additional research is needed before such methods can be recommended with confidence.

**M. Summary**

The preceding sections of this chapter have described in detail a variety of methods for directly controlling nuisance-causing growths of algae and macrophytes. Unfortunately, few of these methods are currently suitable for use in typical Kansas reservoirs. Only three (copper sulfate addition, organic herbicide addition, and mechanical harvesting) can be recommended for general use at the present time, and even these must be employed with caution. Although all three of these methods can provide effective and immediate relief from weed problems, they are expensive, provide only temporary relief, and can have substantial adverse impacts on
fishing and on drinking water quality. Biological control methods hold considerable promise for the future, but are inadequately developed at the moment.

The natural fertility and morphology of Kansas reservoirs, as well as the climate, make weed control a difficult problem to which there appears to be no easy solution. A manager of a multipurpose reservoir should consider the full range of alternatives, including those discussed in subsequent chapters, and should strive to develop a long-term control strategy that not only protects drinking water quality but also maximizes benefits to other users of the reservoir, within the limitations imposed by the need to protect drinking water quality. When enhancement of sport fishing is an important management objective, a weed control strategy should be developed with input from a competent fisheries biologist.
CHAPTER 5
DIRECT MANIPULATION OF FISH

The management of lakes and reservoirs for sport fishing has received considerable attention through the years and often involves direct manipulation of fish populations by either selective removal or addition. Reviews of the various practices involved are common in the literature (e.g., Anon, 1967; Bennett, 1971; Everhart et al., 1975; Gulland, 1978; Lackey & Nielsen, 1980). The primary objective of such practices is to either alter the abundance of selected species or change the species composition of the community. One or another management practice may be initiated when forage fish or rough fish are too abundant. Another circumstance could be a large sport fish population stunted in the size of the individuals and dominated by only one age class due to heavy predation or competition. Here the choice may be to reduce the stunted population or reduce other fish populations. A summary of some of the more common practices, particularly those used in Kansas, is presented below, along with an evaluation of their general success and a consideration of possible side effects. This chapter addresses only the direct manipulation of fish. Other practices, such as level adjustment (Chapter 7), fish attractors (Chapter 14), nutrient addition (Chapter 6), and weed management (Chapter 4) are also used to indirectly manipulate fish populations by altering their habitat.

A. TOTAL OR SELECTIVE REMOVAL

Background

The total removal of fish from a reservoir is often considered to be necessary when forage fish or rough fish have become too abundant. Overabundance occurs when these fish outcompete sport fish for food or consume their young, or when they are so abundant that they alter the habitat (e.g., carp stirring up the bottom sediments and increasing turbidity). To achieve total removal, the reservoir may be drained or a fish toxicant added. Often a combination of these practices is involved. Since it may be impossible to entirely drain the reservoir due to its size or due to the pools and channels inevitably left behind, partial draining followed by poisoning is common. During the draining process a special weir is sometimes constructed so that the valuable sport fish can be retrieved for reintroduction or relocation (Wolf, 1951). The remaining fish in any pools and channels are then poisoned using one of several possible chemicals described below.

The selective removal of certain fish from a reservoir is often considered to be a more feasible task than total removal. The size of the reservoir is one determining factor, since 1) reservoirs greater than a few hundred hectares in area are generally too large to drain and refill again in a reasonable time period; 2) the period that a large reservoir would remain empty might be more objectionable to the general public than the existing poorly-structured fish community; and 3) it might not be feasible
to add enough toxicant to a large reservoir to kill all of the objectionable fish. Furthermore, draining a reservoir (partially or completely), or adding a toxicant, can severely disrupt its ecological functioning, impact non-target organisms, and alter important physical and chemical processes (as will be discussed later). If so, then manipulating the entire body of water may not be advisable.

The selective reduction of target fish populations can be attempted through physical, biological, and chemical means. Physical means such as netting, trapping, and electrofishing are often used when conditions are not considered to be severe enough to warrant removal of a large fraction of the target fish population. These methods are labor intensive and can only capture a small fraction of the fish, particularly in larger reservoirs. In reservoirs where commercial fishing occurs, subsidies may be provided to the fishermen to remove target fish captured in their operation. Biological means, including stocking of fish predators (e.g., turtles or snakes), fish sterilization, and fish hybridization, have been studied (Dunst et al., 1974); but such methods must be considered experimental at the present time.

Toxicants (piscicides) are also used on a selective basis by limiting their application to certain zones of the reservoir, such as bays, shorelines, or specific spawning or nesting sites. Small shallow bays that forage fish tend to frequent may provide the best opportunity for chemical control. Such bays may still be too large to be effectively cleared physically, but are separated enough from circulation with the main body of water that a limited dose of the chemical will be successful. The effectiveness of this approach can be further enhanced by physically separating the bay from the rest of the reservoir using some sort of curtain extended across the zone of confluence.

Specific nesting sites may also be targeted for toxicant addition, reducing the need for introduction of large doses of the chemical. Bluegill are sometimes controlled in this manner, since their young are at first reared in shallow nesting sites. The timing of toxicant addition can also enhance the selectivity of a fish removal operation, for example when attempting a community composition shift within the centrarchids. The small centrarchids, including sunfish and crappie, though of some sport fishing value, can be undesirable when their populations become so large that they prey too heavily on the eggs and fry of a more desirable centrarchid, the largemouth bass. In Kansas, largemouth bass spawn over a relatively narrow time period in the spring, while bluegill spawn from the spring through much of the summer. A toxicant can be added along the shoreline in late summer, when young-of-the-year bass have moved into deeper waters and many bluegill fry are still near their nests.

The choice of a toxicant for total or selective fish removal depends upon a variety of factors, including toxicity to the target population, selectivity for that population, side effects on other fish or other organisms, and cost. Many different chemicals have been used as fish toxicants (e.g., Lennon et al., 1970), but relatively few are widely and routinely used. Even fewer are appropriate for use in water supply reservoirs. Rotenone, under such trade names as Derris and Cube, and antimycin A, under the trade name Fintrol, are two of the most commonly used
fish toxicants in reservoirs, and both have formulations registered for fishery use by the U.S. Environmental Protection Agency according to the Federal Environmental Pesticide Control Act (Schink & Meyer, 1978). The history of the use of such chemicals has been reviewed by Lennon et al. (1970) and Cumming (1975).

Rotenone is a plant extract which kills fish by inhibiting oxygen uptake, thereby causing suffocation. It is generally added to produce a concentration of 1.0 to 2.0 mg/L of the formulated product (which is from 2.5 to 5.0% rotenone); and it causes death within a few minutes to an hour or so. The toxicity of rotenone is a function of the species and size of the fish and such abiotic conditions as temperature, pH, alkalinity, turbidity, and oxygen concentration. Rotenone is most effective at water temperatures above 10°C in waters of lower turbidity, alkalinity, pH, and oxygen concentration. It is slow to penetrate the deeper colder waters of stratified reservoirs and can thus be used to manipulate the epilimnetic community, leaving the deeper dwelling fish (such as trout) less affected (Thompkins & Mullan, 1958). However, in Kansas reservoirs, which are generally eutrophic, hypolimnetic waters are usually low enough in dissolved oxygen that they harbor no resident fish populations of concern. Rotenone remains toxic for several days at temperatures above 10°C; although less toxic at lower temperatures, it may remain active for a month. If the concentration of rotenone is kept low, within the range suggested, it is generally more toxic to small centrachids and shad than to ictalurids and larger centrachids (Davies & Shelton, 1983). Rotenone is not toxic to mammals and birds at the concentrations recommended but will kill some species of animals which obtain their oxygen from the water, including both invertebrates and vertebrates, as will be discussed in greater detail later.

Antimycin A is an antibiotic which kills fish by inhibiting respiration, but it affects a different biochemical site than rotenone. A concentration of 15 ppb of the active ingredient, antimycin A, has been suggested by field studies (Gilderhus et al., 1969), though certain water quality conditions will alter its effectiveness, as is the case with rotenone. The toxicity of antimycin is greatest in lower alkalinity waters of moderate temperature. Under these conditions it generally remains active for up to a week.

Fish species differ more in their sensitivity to antimycin than to rotenone (Muirhead-Thomson, 1971). Walker et al. (1964) group species into those of high, moderate, and low sensitivity, with scaled fish generally more sensitive than the ictalurids (e.g., bullheads and catfish). Most cyprinids (e.g., carp and shiners), catostomids (e.g., suckers), percids (e.g., perch), and centrarchids are killed by 15 ppb of antimycin in waters with the recommended water quality characteristics (Gilderhus et al. 1969). Antimycin at its recommended dosage also appears to have less effect on other aquatic animals and plants than does rotenone at its recommended dosage (Everhart et al., 1975; Davies & Shelton, 1983), as will be further discussed later.

Although the physical (e.g., netting and electrofishing) and chemical (toxicants) methods of removing target fish species are currently the most common methods, some attempts at biological control have been reported
(Everhart et al., 1975). The introduction of predator fish, such as northern pike, has been used to reduce large populations of such fish as sunfish and shad. A further consideration of this method is included under the broader topic of managing fish communities by stocking (see Section B of this chapter).

**Assessment**

There are a variety of things to consider when evaluating the success of fish removal programs in water supply reservoirs. These include both the short-term and long-term effectiveness of the actual population adjustment and the accompanying side effects. The latter may alter water quality directly or so effect the ecosystem such that water quality is altered indirectly. Fish communities in water supply reservoirs are rarely manipulated through total fish removal either by draining or addition of a toxicant. These methods are not compatible with maintenance of a continuous supply of potable water unless alternate supplies are available. Rotenone and antimycin have not been shown to be a direct health hazard to humans, and in addition can be removed from the water by chlorination and potassium permanganate treatment respectively (Everhart et al., 1975). However, public reaction might be a sensitive issue that most managers would rather avoid. The presence of many dead fish, and perhaps other dead aquatic organisms, would likely reduce the water quality for a time as decomposition increased the oxygen demand in the deeper waters of the reservoir. Reduced oxygen levels, particularly in deeper waters and bays with little mixing, could lead to a variety of taste and odor problems. In addition, public reaction to the display of dead organisms in a water supply reservoir would be another sensitive issue.

Partial fish removal, through physical capture, limited toxicant addition, or predator addition, is more acceptable for water supply reservoirs. However, in most cases the control of the undesirable fish populations is only temporary (Noble, 1980). When partial removal is attempted, only a small fraction of the individuals in the population may be reached, even with considerable effort and expense. Even when large fractions of the target populations are removed, the potentially rapid reproductive rates of many such fish often result in rapid reestablishment of the original population sizes within a year or so. These management practices must then become routine and therefore are often too costly for municipalities to support.

Control through stocking of predatory fish is perhaps the most desirable practice in principle, since less disruption of the habitat generally results. Stocking of predators is generally necessary because reservoirs are often unable to sustain a large enough native predator community to keep some forage fish and rough fish populations in check. As discussed in Chapter 3, the original stream fish community that first occupies a newly constructed reservoir does not seem to provide an adequate top predator from the natural contingent present. Continual addition of predator fish, either native or exotic, in large enough numbers to achieve control, can become another costly management practice. The success of stocking programs for this and other purposes is discussed in Section B of this chapter.
The secondary effects of a total or partial removal of fish must also be considered in evaluating the overall success of a manipulation. There is a considerable volume of literature detailing attempts at fish removal, but most reports evaluate success primarily in terms of the fish population and give few details concerning side effects on other components of the ecosystem. As a result, side effects can only be summarized in rather general terms. The limited information on side effects that is available deals primarily with the toxicity of the chemicals to non-target species rather than the effects on the ecosystem of the loss of particular target fish. Both rotenone and antimycin are described throughout the literature to be more toxic to fish than to other aquatic organisms. Terrestrial organisms and other organisms not removing their oxygen from the water are considered to be unaffected. Aquatic plants, including both algae and macrophytes, are also considered to be unaffected at the recommended dosages.

Non-target aquatic animals have been reported as variously affected by rotenone and antimycin addition. Some studies have reported little effect of either rotenone (Brown & Ball, 1942; Walker et al., 1964; Houf & Campbell, 1977; Chandler & Marking, 1982) or antimycin (Walker et al., 1964; Gilderhus et al., 1969; Snow, 1974; Houf & Campbell, 1977). Most studies report varying effects depending on field conditions, major taxonomic group, and species (i.e., some closely related species respond differently) for rotenone (Cushing & Olive, 1955; Almquist, 1959; Binns, 1967; Schottenger et al., 1967; Cook and More, 1969; Burress, 1982) and for antimycin (Schottenger et al., 1967; Lesser, 1972; Beard, 1974; Baumann, 1975; Jacob, 1977; Kotila & Hilsenhoff, 1978). Major taxonomic groups most often studied include mollusks, crustaceans (including plankton), insects, and amphibians (principally frog larvae). Benthic invertebrates, predominantly insects, are generally less affected by both chemicals (Callaham, 1968; Callaham & Huish, 1968, 1969; Houf & Campbell, 1977) than are certain mollusks (Antonioni, 1974; Flowers et al., 1975; Marking & Chandler, 1978; Burress, 1982) and zooplankton (Almquist, 1959; Wollitz, 1962; Kiser et al., 1963; Callaham, 1968; Callaham & Huish, 1968; Rabe & Wissman, 1969). Insect communities affected by either chemical appear to rapidly recuperate (Cook & Moore, 1969 Jacobi & Degan, 1977), while the zooplankton are slower to recuperate (Almquist, 1959; Wollitz, 1962; Kiser et al., 1963; Anderson, 1970). As a result of particular responses of target and non-target organisms, some habitats have experienced nutrient increases (possibly due to animal decay) and phytoplankton increases, possibly due to nutrient increases and/or grazing decreases (Bonn & Holbert, 1961; Jacob & Degan, 1977; Burress, 1982).

In summary, there is enough information from the literature to suggest that total removal of fish by toxicant addition can exert a negative impact on water quality and on some non-target aquatic animals, though for the latter there is a lack of agreement on which organisms are consistently affected and how rapidly they recover. Given these potential side effects, selective removal of fish appears to be preferable to total fish removal. When only a portion of the reservoir receives the toxicant, there is less potential for disruption of the ecosystem as a whole, particularly since untreated areas can provide organisms for rapid recolonization of non-target
species that are negatively impacted. Of course the same rapid recolonization might occur in the target populations.

With regard to the impact of fish removal on water quality, relatively little information is available. Erickson (1981) reported that it is not uncommon for elimination on bottom-feeding fish in Illinois reservoirs to result in a five- to eight-fold increase in water clarity, but this could in turn stimulate excessive growths of algae or macrophytes. Reductions in the numbers of planktivorous fish have reportedly led to decreases in algal biomass (see Section B of this chapter), but such results have not been demonstrated in reservoirs similar to those in Kansas. Until further research is done, it will not be possible to accurately predict the effects of selective fish removal on water quality in Kansas reservoirs.

B. STOCKING

Background

Reservoir stocking programs involving the addition of species not present at the time are termed "introductions". In Kansas (Cross & Collins, 1975), introduced species may be endemic to the area (i.e., the state); these include largemouth bass, bluegill, white crappie, channel catfish, and bullheads. Other introduced species may not naturally occur within the state, but are found within a larger region, such as the Mississippi Valley, where they still come in contact with many native Kansas species. Such introduced species include walleye, yellow perch, white bass, and northern pike. A few introduced species are truly exotics; there are not native to Kansas or to the Mississippi Valley and have had no previous natural contact with native species. Such fish include carp, trout, and striped bass. Resident fish populations in a reservoir can also be artificially increased by stocking, and this practice is termed "supplemental stocking" (Noble, 1980).

Natural self-sustaining fish communities in lakes doubtless take many years to become established, though there has been little opportunity to follow such a natural succession after a lake has come into existence by natural processes. Stocking programs in reservoirs are generally initiated for either the establishment of sport fish populations or for biological control of pest organisms. This chapter will primarily address the former, leaving the latter for consideration in Chapter 4, except for the case where sport-fish stocking is done with the intent (in whole or in part) to control nuisance organisms.

Reservoirs are expected to have sport fish populations attractive to the fisherman within a few years of construction. Stocking programs are therefore initiated early on and must often be repeated, perhaps even indefinitely, if habitat conditions and predator-prey relationships do not become supportive of the sport-fish community. Therefore such management practices as weed removal, water level adjustment, destratification, nutrient manipulation, and others discussed in this report become important components of successful stocking programs. The role which each of these
practices plays in the ability of a reservoir to maintain the desired fish community is discussed within each of the appropriate chapters. The following discussion is thus confined to consideration of the success of various stocking practices in preparation for this longer-term maintenance.

The literature describing the procedures for stocking game fish (i.e., sport fish manipulated by stocking or by catch restrictions) and evaluating their success in terms of the stocked fish is quite extensive. It is not necessary to include a review of this literature here, since the procedures themselves generally do not adversely affect the reservoir environment. The major effects of stocking programs on the reservoir environment are indirect:

1. The introduction of game fish, most of which are predators, can significantly alter predation pressure on lower levels of the food chain, thereby altering the structure of the biological community, which can in turn have other effects (e.g., impacts on nutrient cycling, decreased algal biomass due to a reduction in the number of planktivorous fish feeding on grazing zooplankton, etc.).

2. The particular physical activities of certain types of fish, including some game fish, can produce indirect effects. For example, fish that disturb the sediments can increase turbidity, and bottom feeding fish (e.g., carp and bullheads) can recycle nutrients from the sediments to the water column. These effects can be prevented by total or selective fish removal. Since these effects are rarely associated with stocking, except when carp or catfish are introduced into small reservoirs, they will not be discussed further in this section of the report.

3. The longer-term management practices used to maintain sport-fish populations can have indirect effects on both water quality and fishing, as discussed in the particular chapters dealing with each.

The impact which the addition of game fish can have on a reservoir food chain and on nutrient cycling is being increasingly considered by reservoir managers throughout the nation. This has not been in response to any adverse effects of stocking programs; rather, it is currently of interest in lake reclamation projects. Adding certain fish, particularly top predators, is now being investigated as a means of reconstructing the entire food chain, thereby reducing the growth of certain undesirable plants. Though this literature is still sparse, there is already good evidence that food chains and nutrient cycling can be altered by the addition of certain predaceous fish (e.g., Shapiro et al., 1975; EPA, 1980a; Lynch, 1981; Cooke, 1981; Shapiro & Wright, 1984). Predaceous fish include planktivores which feed on plankton, piscivores which feed on other fish, and benthivores which feed on benthic organisms. The alterations have most often been observed when populations of planktivorous fish have been purposely reduced (e.g., Hrbacek et al., 1961; Hall et al., 1970; Anderson et al., 1978; Stenson et al., 1978; Pott et al., 1979; Lynch & Shapiro, 1980; Shapiro & Wright, 1984) or naturally reduced (e.g., Schindler & Comita, 1972; Brashier et al., 1973; O'Brien et al., 1979). This has been observed to relieve predation pressure on the herbivorous zooplankton with the subsequent increase in their size.
(i.e., larger body-sized individuals, once most heavily preyed upon, now dominate since they are better competitors than smaller individuals) and/or numbers, thereby increasing the grazing pressure on the phytoplankton and reducing their abundance.

Such a purposeful manipulation of the fish population of a natural lake is described by Shapiro and Wright (1984): rotenone was added to eliminate a dominant planktivorous-benthivorous fish community, which was then replaced with one dominated by the piscivorous largemouth bass and walleye. This alteration was being used as a management technique to control nuisance algae accompanying eutrophication. Over the first two years following the restructuring of the fish community, larger-bodied zooplankton increased, with a subsequent increase in grazing pressure and a reduction in phytoplankton abundance. Nutrient conditions were also altered in terms of total nitrogen and phosphorus levels, with observable declines in both.

Assessment

In the absence of enough detailed studies where the entire food chain and the nutrient conditions in the lake or reservoir are followed after restructuring of the fish community, it is not wise to extend the results reported by Shapiro and Wright (1984) to a general prediction of the impact of stocking programs in Kansas. If such results were the general rule, they would be considered an improvement to the reservoir by most managers. However, until it is shown that such results can be consistently produced, particularly in Kansas reservoirs, it must be realized that manipulating the fish community in a Kansas reservoir can have a variety of indirect impacts and the potential for a negative impact therefore exists. Recognizing this, future stocking programs should be evaluated as to their success by observing the responses beyond just those of the manipulated fish populations. Nevertheless, stocking programs to enhance sport fishing in Kansas have no known significant direct impacts on water quality or water treatment; and the indirect effects appear to be either positive or inconsequential. The most significant effects are those associated with long-term maintenance of sport-fish populations by methods described in other chapters of this report.
CHAPTER 6

ARTIFICIAL FERTILIZATION

A. BACKGROUND

Artificial fertilization is the purposeful addition of nutrients in the form of fertilizer to lakes, reservoirs, and ponds in an attempt to increase the production of selected fish through stimulation of plant production or to alter the plant community to discourage the growth of objectionable plant species. The active nutrient ingredients most often incorporated into the various fertilizer products are nitrogen, phosphorus, and potassium. Today, the practice of fertilizing lakes and reservoirs for either purpose is uncommon throughout the U.S., including Kansas, and is particularly discouraged in water supply reservoirs due to its incompatibility with water treatment objectives (Wagner & Oglesby, 1984). Knowledge of the adverse effects of cultural eutrophication (e.g., nutrient enrichment from runoff) on aquatic habitats gained during past two decades (e.g., Hutchinson, 1969 & 1973; Regier & Hartman, 1973; Lee et al., 1978) has played a key role in discouraging this practice. The practice continues to be successful and encouraged for increasing fish production in privately owned ponds (e.g., Boyd, 1981), and is potentially useful in larger bodies of water as a means of selectively discouraging the growth of certain nuisance plants (Barica et al., 1980) or as a means of increasing the overall production of ultraoligotrophic waters (Stockner & Shortreed, 1985). At present, water supply reservoirs are not fertilized in the state of Kansas (pers. comm., KDHE, Water Quality Section), but this practice is herein reviewed to provide a basis for its future consideration.

The practice of fertilizing ponds and lakes to increase fish production has been followed for centuries in Europe and Asia (Bennett, 1971) and was more recently brought to North America (Swingle & Smith, 1942 & 1947). The limitation of fish production by nutrients has been demonstrated in natural lakes, though generally only in the most oligotrophic ones (LeBrasseur et al., 1978; Stockner, 1981). This may result from very low nutrient input from the watershed or a short residence time (producing a rapid flushing condition). Lakes with higher nutrient concentrations may still experience nutrient limitation for particular fish when critical nutrients are incorporated into pathways less available to these fish. As discussed earlier, sport fish are generally carnivores, feeding on zooplankton, insects, and fish. If the plant community is composed of species less available to the particular organisms which the sport fish feed upon, the growth of sport fish may become nutrient limited. This is known to occur when an increased proportion of the phytoplankton community is composed of large colonial or filamentous species (e.g., Lawrence, 1958; Brooks, 1969; Larkin & Northcote, 1969) or when the overall plant community is dominated by aquatic macrophytes (Bennett, 1948 & 1971; Surber, 1961; Boyd, 1971). Such plant communities can also affect fish production in other ways, such as by producing anoxic conditions and releasing toxic chemicals, as discussed in Chapter 4. Since the changes in plant communities discussed above have often been identified to be the result of eutrophication (Lee et al., 1978;
Edmondson & Lehman, 1981), the risks of purposeful fertilizer addition become obvious.

Where fertilization has successfully enhanced the production of certain desirable fish, this has generally been accomplished through a balanced and moderate production of both phytoplankton and macrophytes. Species of phytoplankton edible by zooplankton prevail, and the macrophytes provide refugia for young fish and a substrate for an insect community. The macrophytes are not so abundant as to tie up nutrients throughout the growing season, severely limiting algal growth. Too large a macrophyte community can also provide too much refugia for forage fish thus limiting their availability to the larger predators. In order to balance phytoplankton and macrophyte growth, it is critical that nutrients be added prior to the spring onset of macrophyte growth. If added later, much of the nutrient can be taken up by the macrophytes; and with their much slower biomass turnover, nutrients can be held in their biomass until the end of the growing season. In contrast, by stimulating phytoplankton biomass early in the growing season some shading will result (Mulligan, 1969; Nichols & Shaw, 1983), minimizing the tendency for macrophytes (which as a group are less vulnerable to grazing losses) to outcompete phytoplankton in shallow transparent waters (which are sometimes found in the smaller water supply reservoirs in Kansas).

Attempts to alter plant communities to discourage the growth of particular species goes beyond the above situation of selectively manipulating phytoplankton and macrophytes as groups to increase fish production. As noted earlier in this chapter and elsewhere in the report, particular species of algae may be considered objectionable. Certain species of colonial and filamentous blue-green algae, or cyanobacteria as they are sometimes classified, often dominate the plant biomass in eutrophic lakes (Shapiro, 1973; Kalff & Knoechel, 1978). Many of these species of blue-green algae have the capacity to fix atmospheric (diatomic) nitrogen, and thus they have a source of this important nutrient not available to other algae. A steady supply of diatomic nitrogen is maintained in natural waters by diffusion from the atmosphere and from bacterial denitrification in anaerobic or anoxic zones of the water column. The proliferation of such organisms is then supported by the condition, common in eutrophic waters, of phosphorus availability being in excess of demands to the point where sources of nitrogen other than diatomic nitrogen become limiting (Smith, 1979 & 1983).

The addition of enough ammonium or nitrate (preferably nitrate because it has no oxygen demand) to maintain phosphorus as the limiting nutrient can remove the competitive advantage of nitrogen fixation, allowing other algae to become dominant. The nitrogen fixing blue-green algae, being colonial or filamentous and thus large in body size, are not readily eaten by the zooplankton grazers, commonly the next higher level of the aquatic food chain. Thus when their growth is favored, a considerable biomass of blue-green algae may accumulate while comparable growth of other more edible algae may not produce a similar accumulation. Fertilizing to discourage the growth of the blue-green algae might not reduce overall plant growth but could reduce the accumulation of the biomass produced by that growth. The abundance of nitrogen fixing blue-green algae in lakes has been reduced by
such alterations of the relative availability of nitrogen to phosphorus (Findlay, 1978 & 1981; Barcia et al., 1980). Smith (1983), in a survey of 15 natural lakes, found a strong relationship between the ratio of nitrogen to phosphorus and the proportion of blue-green algae, which were rarely found when the ratio exceeded 29 to 1.

B. ASSESSMENT

Under certain conditions fertilizer addition to water supply reservoirs may increase fish production, but it must first be documented that nutrients are limiting fish production. Nutrients may limit fish production by limiting the plant production which is supporting the particular food chain being utilized by the species of fish of concern. Overall plant production may be low in the reservoir, or if actually high it may not be in a form entering the fish food chain. Particularly in Kansas reservoirs, low plant production may not be the result of nutrient limitation but instead the result of light limitation due to excessive turbidity (O'Brien, 1975). In such a case nutrient addition would be costly and would result in no observable benefit to the reservoir.

With high plant production but little support for the fish, nutrient addition early in the growing season may stimulate an early growth of phytoplankton, shading out the later developing macrophytes. This is risky, since if timing is not right and the desired stimulation of phytoplankton growth is either not initiated or not continued long enough, an even larger accumulation of macrophytes may develop, worsening conditions for fish and adversely affecting water quality. Therefore, considering only the benefits to fish production, fertilizing water supply reservoirs in Kansas is not advised because it is too likely to be ineffective and to cause deterioration of water quality to the detriment of the sport-fish population.

Fertilizing may be of some benefit in controlling the abundance of particular species of algae. However, this practice should only be considered when the nitrogen to phosphorus ratio is proven to be low, indicating that nitrogen availability may be a controlling factor in phytoplankton competition and thus community structure. Ideally then, the phosphorus inputs into the reservoir should be reduced, which would also increase the nitrogen to phosphorus ratio and reduce the overall nutrient load. Increasing this ratio by nitrogen addition should only be a last resort, when phosphorus inputs are truly uncontrollable. There is clearly a risk of increasing overall plant production since nutrients are being added rather than removed. Even though bluegreen algae may decline in biomass, the growth and abundance of other algae could have other effects on water quality as discussed in Chapter 4.

A shift from bluegreen algae to green algae is likely to substantially reduce taste and odor problems, but there is not enough information available to allow reliable predictions of the magnitudes of the changes changes that might occur in the concentrations of TOC, THM precursors, turbidity, hypolimnetic dissolved oxygen, etc. However, to the extent that blue-green
algae are less edible than green algae, the changes in water quality resulting from a shift to green algae are expected to be favorable. Nevertheless, even where a shift in algal dominance can be achieved through fertilization, nutrients added to a typical Kansas reservoir would either quickly settle to the bottom (if the reservoir is stratified) or be rapidly flushed out by the inflowing water. Thus, fertilization can be expected to achieve only short-term control at best.
CHAPTER 7
LEVEL ADJUSTMENT

A. BACKGROUND

Newly flooded vegetation, high water fertility, and an abundant food supply make new reservoirs excellent fish habitats. Typical effects of a newly impounded reservoir on fish populations are increased reproductive rates, high survival of offspring, rapid growth and recruitment, and increased fishing success (Keith, 1975).

A major fisheries management problem is the decline in fishing success in the years after the initial impoundment. Removal of large numbers of fish by anglers at a rate that exceeds recruitment during the first year of fishing tends to hasten this decline (Pierce et al., 1963). New reservoir conditions may, however, be prolonged by filling the lake slowly over a period of years (Schroeder, 1978). Each year the reservoir expands with new habitat becoming available with each water level rise.

Unfortunately, in an "old" reservoir, new reservoir conditions may only be accomplished by completely draining the lake, allowing vegetation to grow on the exposed bottom for one or more years, then refilling the lake. Of course, this is not practical for most multipurpose reservoirs because of other water demands.

Fishery biologists have been successful in improving fishing at many older reservoirs by recreating new reservoir conditions on a small scale. Water levels at many Kansas reservoirs can be manipulated a foot or two above or below normal conservation pool level without interfering with most intended reservoir purposes (Schroeder, 1978). Most of the reported successful applications involve a fluctuation of at least two feet.

Typical water-level management plans for Kansas reservoirs include the following:

1) A gradually rising water level in the spring which inundates terrestrial vegetation and rocky areas and enhances spawning and nursery habitat.

2) A relatively stable or slightly rising water level through late spring and early summer.

3) A mid-summer drawdown to expose previously inundated areas for revegetation, make forage fish more available to predator fish, and control rough fish.

4) A slight water level rise in the fall to inundate the lower band of terrestrial vegetation, enhancing waterfowl habitat.
5) A drawdown in the winter to set the stage for favorable fish spawning conditions the next spring (Groen & Schroeder, 1978).

This chapter discusses the effects of water level manipulation on vegetation control and fishery enhancement, and assesses the use of water level adjustment in multipurpose reservoirs.

B. VEGETATION CONTROL

Water level drawdown to manage nuisance aquatic macrophytes is a well established technique. According to Cooke (1980):

"The objective is to retard macrophyte growth by destroying seeds and vegetative reproductive structures through exposure to drying and/or freezing conditions, and by altering their substrate by dewatering and consolidation of sediments. There are several secondary objectives which include turbidity control by sediment consolidation, reduction of nutrient release from sediments, management of fish populations and waterfowl habitats, repair of shoreline structures..., and simultaneous use of other restoration methods such as sediment covering."

The effectiveness of overwinter drawdown depends largely on the susceptibility of the nuisance species to drawdown control. Cooke (1980) surveyed the literature and summarized the responses of 63 macrophyte species to drawdown. Only three macrophytes, Chara vulgaris (muskgrass), Eichhornia crassipes (water hyacinth), and Nuphar spp. (water lily) always seemed to be controlled. Some species exhibited a variable response (e.g., Myriophyllum spp.), while others, including Alternanthera philoxeroides (alligator weed), Lemna minor (duckweed), Leersia oxylepis (cutgrass), Potamogeton spp. (pondweed), and Najas flexilis (naiad) usually increased after drawdown.

Cooke (1980) recommends lake level drawdown for macrophyte control for situations where prolonged (1 month or more) dewatering of lake sediments is possible under rigorous conditions of cold or heat, and where susceptible species are the major nuisances. Fox et al. (1977) point out that lakes with gradual basin slopes are ideal, since small drops in lake level will expose large areas.

Based on their study of three Louisiana lakes, Lantz et al. (1964) found that overwinter drawdown resulted in less turbid water, better spawning and survival of sunfishes, and increased growth rate and survival of the bass population. In two of the lakes, the fish population improved, aquatic vegetation was controlled, and fishermen harvested more fish. Lantz et al. (1964) witnessed over a 90 percent reduction in species of Potamogeton and southern naiad after 2-3 years of drawdowns in these lakes. In the third lake, aquatic vegetation was controlled but fishing was not enhanced.
Nichols (1975) noted good results with an overwinter drawdown on Mondeaux Flowage, Wisconsin. A single overwinter drawdown greatly reduced the amount of littoral zone vegetation. Generally, stem densities were half or less than half of pre-drawdown levels. About 40 percent of the littoral zone was free of aquatic plants after the first year's drawdown, compared to no open area prior to drawdown. *Potamogeton robbinsii* was the dominant pre-drawdown plant in the flowage.

Little additional control was gained by a second year's drawdown except for *Nuphar variegatum*, which may be controlled by mechanical action of ice (Nichols, 1975). Nichols notes that subsequent drawdowns will probably be diminished in effectiveness as drawdown tolerant species replace drawdown susceptible species. To partially alleviate the problem, Nichols recommends water level manipulation no more frequently than every other or third year. Several researchers agree with this recommendation, both for vegetation control and fisheries management (Hulsey, 1959; Ill'ina, 1980; Martin et al., 1981). Nichols (1975) states that alternative management practices might be considered to reduce the frequency of drawdowns.

In Kansas, most water level plans have incorporated a mid-summer drawdown for revegetation of the fluctuation zone and to increase forage availability for piscivorous fishes. In addition to naturally occurring vegetation, shorelines are sometimes seeded with fast-growing vegetation:

"For early [mid-summer] drawdowns, Japanese millet and hybrid sudan-sorghum have produced lush stands valuable for fisheries habitat, turbidity control, and waterfowl food. In Kansas, these species are planted prior to August at rates of 6 to 12 kg/hectare. For late drawdowns, annual ryegrass (*Lolium* sp.), wheat (*Triticum* sp.), or rye (*Secale* spp.) can be seeded during September or early October. Ryegrass is usually planted at 11 kg/hectare, and wheat and rye are planted at 34 to 68 kg/hectare." (Groen & Schroeder, 1978).

C. FISHERY ENHANCEMENT

While overwinter drawdown is used for controlling aquatic vegetation, fisheries are generally improved by high spring water levels. Fortunately, both of these goals can be accomplished with a proper management plan that simulates natural conditions. Water level management plans normally result in an increase in sport fish and a decrease in rough fish numbers.

Lantz et al. (1964) discovered, while using drawdowns for vegetation control, that the fish benefitted from the water fluctuations and the fishermen harvested more fish (the biggest increase being in crappie). The growth rate and survival of the bass population increased and the balance (or percentage of game fish) improved. Lantz et al. (1964) hypothesize that less turbid water resulted in increased predation and therefore a decrease in the shad population. Also, it was learned that if, after an overwinter drawdown, the water level does not return to normal by the early spring, the
fish population will suffer from the increased crowding caused by a smaller lake.

While a high spring water level normally assures spawning success, it seems important that the level remains high and relatively stable throughout the spawning season (Beam, 1983). Drawdown should occur after the production of small forage fish and insects to concentrate the fish and their prey. Heman et al. (1969) implemented a mid-July drawdown on a Missouri impoundment that reduced surface area by 42 percent and volume by approximately 58 percent. Water was drained from the bottom of the lake and the temperature became uniform at 30°C. The mid-summer drawdown resulted in increased feeding activity and accelerated growth of largemouth bass over the previous year for age groups I, II, and III. Largemouth bass harvest declined during August but there were increases in the numbers and average size of fish caught through September and October. The density of fry and intermediate bluegill was reduced after drawdown. Heman et al. (1969) believed that the drawdown was responsible for a portion of this reduction through stranding of small sunfish in weed beds and shallow pools, through increased predation by bass on these small sunfish, and through exposure of nests as the water was lowered.

Miranda et al. (1984) reported a positive relationship between early survival of young-of-year largemouth bass and water level during the spawning period. In the post-spawning period, survival rate and abundance were related directly to water level, but growth was inversely affected. Bennett (1954) also found that abundance of bass was high with elevated spring and summer water levels, but that growth was inversely affected. Following an overwinter drawdown more bass were caught than in any previous season, but the season was considered unusually poor because 80 percent of the bass caught were less than 10 inches in total length. Perhaps summer drawdowns would have led to better fishing seasons by reducing the numbers of bass and consequently increasing their growth rate. During a summer drawdown,

"Small fish that are not stranded...are forced from the protection of rooted vegetation and shallow-water debris into the open water of the lake where they are subject to predation from larger fishes, bullfrogs, and fish-eating reptiles, birds, and mammals. These forces, stranding or trapping and predation, materially reduce the populations of smaller fishes without greatly reducing the numbers of the larger ones. The result is a selective culling action which is more specific for sunfish than for bass, but which may not be extensive enough to be beneficial unless the drawdown: (1) reduces the lake surface area by more than 25 percent and (2) forces the fishes from the protection of beds of aquatic plants." (Bennett, 1971).

Bennett warns that drawdowns in several successive years may result in such a numerical buildup of bass that they will be of smaller average size than under more stable water levels.
Some researchers prefer less frequent manipulations. For example, Keith (1975) suggested that large, programmed drawdowns every 3-5 years were more beneficial to largemouth bass production than smaller annual water level manipulations. Martin et al. (1981) concluded that in Missouri River reservoirs, one high water year out of every three would greatly enhance the fishery resources. According to Beam (1983):

"Although water level management on a 3-5 year cycle has produced quality sport fisheries at some locations (Ploskey, 1982), the uncontrollable conditions of water level fluctuations and flood water releases had statistically significant effects that limited crappie year-class success at Elk City Reservoir, Kansas. These uncontrollable conditions could cause a cyclic water level management plan to result in erratic population trends."

Most successful level adjustment plans in Kansas reservoirs have been annual water-level management plans.

Hulsey (1959) proposed a fisheries management plan dependent upon having a fish management pool and provision for drainage incorporated into the design. The proposal is simply to maintain a cleared and "clean" harvesting area or fish management pool in the very bottom of a reservoir. During periods of drawdown, the fishery manager may recommend, among other things, a harvest of commercial food fishes, a harvest of game or sport fishes, a selective kill of shad or other species, a partial kill of all fishes, or a complete kill. "There are few tools, if any, that might be applied to manipulate a fish population that would not be easier to use in a greatly reduced water area and volume that has been cleared of all standing timber...." (Hulsey, 1959).

As mentioned earlier, Kansas water-level adjustment plans usually include a gradually rising water level in the spring to inundate terrestrial vegetation and rocky areas. This provides valuable fishing and spawning habitat for fishes such as walleye, white bass, and largemouth bass, while providing protection from predation for the newly-hatched fry (Wellborn, 1976). The increased lake volume also encourages abundant production of forage fishes such as gizzard shad (Schroeder, 1978). This abundant food supply is necessary to insure good growth rates of sport fish.

The water level should be held stable at the maximum level through early summer. The flooded terrestrial vegetation provides additional protective cover for certain species. The flooded vegetation also adds nutrients to the reservoir and is partly responsible for the supply of invertebrate foods necessary for the rapid growth of young fish. Decomposing vegetation also helps remove suspended clay particles from the water (Schroeder, 1978). During the first part of July, the reservoir is drawn down, making forage fish available to predator fish and controlling rough fish (Groen & Schroeder, 1978). This mid-summer drawdown exposes the shoreline for revegetation by natural or artificial means.
The next step in a typical water-level manipulation plan is to slightly raise the water level in the fall, inundating the lower band of terrestrial vegetation. The partial flooding of the millet and other grasses provides marshy conditions which attract waterfowl. On several of the reservoirs where conditions have allowed this plan to be implemented, waterfowl numbers have increased (Wellborn, 1976).

In early December, near the end of the waterfowl season, the water is drawn down to its annual low. This winter drawdown sets the stage for favorable fish spawning conditions the following spring. The lowered water level reduces ice and wave damage to the existing shoreline vegetation and provides additional storage capacity for spring runoff. This storage capacity is important in order to avoid drawdowns during the spawning season. Fluctuation zones in Kansas frequently comprise about 20 percent of the reservoir basin. It is important to realize that each reservoir is unique, having its own problems, and that the plan described here represents a general strategy.

Level manipulation seems to be most beneficial to white and largemouth bass, walleye, and white crappie, and detrimental to carp, buffalo, and carpsuckers. In most cases, the increase in sport fish populations is paralleled by a decrease in rough fish numbers. The summer drawdown probably interrupts spawning of rough fishes, forcing young fish out of their protective cover, and limiting their food supply (Schroeder, 1978). Sport fish populations and growth rates are improving in Kansas reservoirs using water-level plans. More forage fish, better water quality, and the summer drawdown are contributing factors (Groen & Schroeder, 1978).

The type of water level management plan described here is not possible on every reservoir. For example, reservoirs in western Kansas may not get enough runoff to refill and are below normal level most of the time. However, Schroeder (1978) remarks that when these reservoirs refill after having been low for one or more years, a fishing boom similar to that found in new reservoirs occurs.

D. ASSESSMENT

A suitable water-level adjustment plan can clearly improve fishing, producing an increase in the size or number of sport fish and fewer rough fish. For this reason, level adjustment is commonly practiced in Kansas reservoirs used for fishing. Other possible advantages of level adjustment are low-cost control of weeds, avoidance of the use of toxic chemicals, and opportunities to make various shoreline improvements.

The effectiveness of level adjustment for vegetation control is species specific and may not provide long-term control (e.g., Jacoby et al., 1983). Some species of weeds are resistant, while others may even be stimulated by this practice. However, other weed control practices (such as herbicide spraying or mechanical removal) are often much easier to implement during drawdown, making level adjustment compatible with macrophyte control if not actually able to effect macrophyte control. In any event, an integrated
approach to macrophyte control is often the best approach (e.g., Nichols & Shaw, 1983).

Potential disadvantages of level adjustment include stimulation of algal blooms, oxygen depletion, reservoir user dissatisfaction during drawdown, failure to refill after drawdown, invasion of undesirable plant species, and lowering of water levels in nearby wells (Nichols, 1975; Cooke, 1980). If level adjustment reduces the growth of shoreline vegetation, a greater percentage of the nutrients in the water column will be available to support phytoplankton growth, and algal blooms could result. However, it should be noted that macrophyte control may substantially reduce internal cycling of nutrients and release of organic matter (e.g., Carpenter, 1983), thereby helping to reduce algal growth; and, since algal growth in Kansas reservoirs is often light limited, a short term increase in nutrient availability may not stimulate algal growth. If level adjustment increases the growth of shoreline vegetation, its subsequent decay can cause increased oxygen depletion and increased release of nutrients and organic matter to the water, possibly causing a shift to blue-green algae (Landers & Lottes, 1983).

Algal blooms are reported to have occurred following reflooding of drawn down reservoirs, but their exact cause is unclear (Cooke, 1980). On the other hand, Fox et al. (1977) simulated drawdown under controlled laboratory conditions and found that the refill water had lower nutrient concentrations, lower turbidity, higher dissolved oxygen, lower temperature, fewer algae, and a more diverse benthic invertebrate population. Since there are conflicting reports, it is reasonable to assume that the impact of drawdown on water quality is site or season specific. Also, level adjustment can cause significant population shifts among organisms, which could account for increases or decreases in algal biomass. For example, if drawdown results in increased predation of planktivorous fish, the population of herbivorous zooplankton could significantly increase, providing natural biological control of algae (see Chapter 4, Section L, and Chapter 5).

Water clarity virtually always increases in years when vegetation in the drawdown zone is flooded (Wellborn, 1976; Groen & Schroeder, 1978; Martin et al., 1981). Rooted plants stabilize the shoreline and reduce turbidity from wave action. Also, decomposing vegetation entering the lake helps to precipitate colloidal clay (Groen & Schroeder, 1978). Tarver (1980) notes that drying and plant dessication produces compaction and stabilization of the sediment; and Hulsey (1957) states that the bottom muds solidify when they dry out and crack open. Hulsey reported that cracks could still be found on the bottom of the lake even after being covered with water for several months. The compaction of the sediment undoubtedly plays a role in reducing the turbidity; but the sediments in Kansas reservoirs are largely inorganic and therefore unlikely to compact much upon drying (e.g., Wedepohl et al., 1983).

Although level adjustment is expected to improve water clarity, its impacts on other measures of water quality (such as dissolved oxygen, organic carbon, and nutrient availability) are difficult to predict. Trihalomethane precursors are released into the water by both growing and decaying vegetation, but the amounts released by various plant species and
the factors influencing their release are almost entirely unknown. Seeding or cultivation of shoreline vegetation, though perhaps beneficial to sport fish, is expected to increase release of nutrients and organic matter to the water, exerting a detrimental effect on drinking water quality. Nutrient availability can be influenced by sediment exposure and desiccation, as discussed by Dunat et al. (1974), but availability may increase or decrease, and the outcome is generally unpredictable.

In reservoirs used as water supplies, it might be worthwhile to monitor water quality as a function of the water level and season, to see whether a cause-and-effect relationship can be established between level adjustment and water quality. In any event, level adjustment exerts its major influence on the shoreline area, so any impacts on water quality should be much more pronounced in smaller reservoirs, which have a greater shoreline to volume ratio.
CHAPTER 8
SELECTIVE WITHDRAWAL

A. BACKGROUND

Selective withdrawal is a management technique in which water discharged from the outlet of the reservoir is selectively taken from either the epilimnion or the hypolimnion, depending upon the objectives to be accomplished. Because selective withdrawal is primarily intended to control downstream water quality, there is a large body of literature examining its effects on downstream water quality and methods for predicting the quality of water released from a given layer of a stratified reservoir. Information on the effects of selective withdrawal on the quality of the water remaining in the reservoir is much more scarce; and theoretical papers presenting the results of model predictions are more numerous than papers describing field-scale experiences.

Since epiimnietic withdrawal is the normal condition for natural lakes, and since little information is available regarding a switch from hypolimnetic to epilimnetic withdrawal, the emphasis of this chapter will be on hypolimnetic withdrawal. Although hypolimnetic withdrawal has not been proven to be an effective restoration technique, its use has been shown to improve water quality. Some common effects of hypolimnetic discharge are increased nutrient export, greater clarity, warming of the hypolimnion, and a reduction in the intensity and duration of hypolimnetic oxygen depletion. Hypolimnetic withdrawal is also generally found to result in a less pronounced temperature gradient, i.e., it has a destratifying tendency.

Dunst et al. (1974) cite three examples where the removal of oxygen depleted water has been beneficial:

(1) Only 1 percent of Fanshawe Lake, Canada, became anaerobic after using bottom discharge, compared to 15 percent before using bottom withdrawal.

(2) In Twin Valley Lake, Wisconsin, severe oxygen depletion continued to occur in the summer but the duration was shortened. In the winter, hypolimnetic oxygen levels increased.

(3) A discharge tube was located 13 m deep in 17 m deep Kortowo Lake, Poland. Before installation, oxygen depletion occurred at 8 m; after installation, oxygen was always present at 13 m.

Dunst et al. note that because of the increase in temperature, increases in the concentration of dissolved oxygen might not be as great as expected. Also, because of the warming, the rates of chemical and biochemical reactions may be greatly accelerated, including the rate of oxygen consumption.

Duever (1980) found that dissolved oxygen (DO) concentrations in Lake Russell, Georgia (surface area = 42 ha, maximum depth = 13 m) were similar in the top 3 m of the water column regardless of the depth of discharge. At intermediate depths, the DO maximum was slightly greater (9-10 mg/L) with
epilimnetic discharge than with hypolimnetic discharge (8-9 mg/L). In deeper water, DO was essentially absent by late summer using epilimnetic discharge. DO concentrations never dropped below 2 mg/L using hypolimnetic discharge. Average net phytoplankton numbers were similar during the summer regardless of discharge depth. During the winter months, net phytoplankton numbers were considerably higher using epilimnetic discharge.

Gaillard (1984) notes the destratifying effect of bottom withdrawal. The difference between top and bottom temperatures in Grangent reservoir, France, may be as high as 20°C when an epilimnetic outlet is used, but does not exceed 10°C when a hypolimnetic outlet is used. The bottom temperature remains below 7.5°C with an epilimnetic outlet, and exceeds 15°C for more than three months of the year using a bottom outlet. Similarly, Gaugush (1984) concludes that decreased thermal stability and increased susceptibility to mixing may be a common feature of bottom withdrawal reservoirs, perhaps exerting considerable influence on nutrient and phytoplankton dynamics.

Gaillard (1984) concludes that although the use of bottom withdrawal helps destratify the reservoir, it is not sufficient to reoxygenate the hypolimnion. The period when the hypolimnion is deoxygenated is reduced, primarily due to an earlier occurrence of fall turnover. However, when thermal stratification is established, the hypolimnion becomes anaerobic. Gaillard (1984) also found that use of a bottom outlet reduces phytoplankton productivity because of the resulting internal mixing, which drives part of the phytoplankton out of the euphotic zone. However, the mixing could just as easily stimulate algal productivity by upwelling nutrients from the hypolimnion following periods of thermal stratification.

Kortman et al. (1983) studied the effects of hypolimnetic withdrawal on Lake Wononscopomuc, Connecticut, a lake with a small (15 m deep) and a large (30 m deep) basin, separated by a submerged ridge immediately below the metalimnion (11 m deep). The lake suffered from metalimnetic blooms of Oscillatoria rubescens, primarily due to internal nutrient loading from the shallow basin. Implementation of hypolimnetic discharge from the shallow basin resulted in increased nutrient export, decreased hypolimnetic oxygen depletion, and reduced internal nutrient loading from the small basin. Combined watershed control efforts and hypolimnetic withdrawal eliminated metalimnetic Oscillatoria rubescens, increased light penetration, and restored a coldwater fishery habitat (Kortman et al., 1983).

Other studies report an increase in nutrient export using hypolimnetic rather than epilimnetic withdrawal (Dunst et al., 1974; Moore, 1976; Rausch, 1980). Cassidy (1980) reported significantly lowered turbidity levels after storms when using hypolimnetic withdrawal.
B. ASSESSMENT

Switching from epilimnetic to hypolimnetic withdrawal is expected to increase nutrient export, increase water temperature, decrease thermal stability, and increase mixing. As a result, there is likely to be a reduction in the duration and intensity of oxygen depletion in the hypolimnion. Increased water clarity and a shift in the abundance and distribution of algal species may also occur. The net impact of these changes on water quality and sport fishing is difficult to assess, both in general and with regard to any particular reservoir.

Increased nutrient export may help control weed growth, but nutrients may not limit photosynthesis in a highly turbid reservoir, and the simultaneous increase in temperature could cause an algal bloom or a shift to a more noxious weed species. Increased mixing may decrease algal productivity (if light limited) or may increase productivity by upwelling nutrients from the hypolimnion. Increased clarity may improve predation for sight feeders, but the increased temperature may cause a decline in feeding activity. A reduction in the duration and intensity of hypolimnetic oxygen depletion may improve the chemical quality of drinking water, but the increase in water temperature will make the water less palatable for drinking purposes and may actually lead to increased complaints regarding taste and odor (due to increased volatility of taste- and odor-causing chemicals). Hence, the impacts of hypolimnetic withdrawal may be positive, negative, or mixed, and the only sure way to determine the impacts is to conduct field-scale testing on the reservoir in question.

Additional complications are introduced where reservoirs serve multiple purposes. If the water is discharged to a stream, especially when the discharge is intended for low-flow augmentation or maintenance of in-stream flow needs, careful consideration must be given to the impact of the discharge on stream water quality. If the water is discharged to a drinking water treatment plant, then the cost of water treatment and the aesthetic quality of the treated water must be considered in the selection of the withdrawal level. If water is withdrawn for agricultural use, hypolimnetic water will generally provide more nutrients and the cooler temperature of the water will help reduce evaporative losses; however, use of the hypolimnetic water may cause problems for other users. Use of epilimnetic water for any purpose may help to control algae by removing phytoplankton and reducing surface water temperature. In any event, the potential impacts of selective withdrawal on all water users should be considered, including the ecological consequences of altering downstream water quality.
CHAPTER 9

DILUTION/FLUSHING

A. BACKGROUND

Dilution/flushing is a reservoir management technique intended to reduce nutrient levels within a reservoir by diluting or replacing high-nutrient water with low-nutrient water, thereby controlling the growth of algae and macrophytes and solving related problems. There are two major approaches (Dunst et al., 1974): (1) pumping (flushing) water out of the reservoir, allowing it to refill with nutrient-poor ground water and precipitation; and (2) adding nutrient-poor surface waters to the reservoir.

Oglesby (1969) postulates three mechanisms for the control of algae using dilution: 1) addition of nutrient-poor water may produce a direct dilution of the principal algal nutrients within the reservoir; 2) a critical trace element, or elements, may be diluted to the extent of becoming limiting for algal growth; 3) as the washout rate of algae approaches growth rate, the standing crop is necessarily reduced. Dilution water may also flush out some of the sediment-bound nutrients (Welch et al., 1972).

The first method, allowing the reservoir to refill by groundwater and precipitation, has been used to a lesser extent than adding dilution water. It has also met with less success. One documented field experiment with this technique was conducted at Snake Lake in northern Wisconsin, a 12.3-acre (5.0 ha) brown-water seepage lake with a maximum depth of 18 feet (Born et al., 1973b). Dunst et al. (1974) summarized the results of the Snake Lake study:

"Approximately 3-2/3 volumes of water were pumped from the lake to a land disposal site. As a result, nuisance blooms of Lemma (duckweed) were eliminated and the nutrient levels were greatly reduced initially. However, phosphorus concentrations were still relatively high and within one year nitrogen had increased to pre-pumping levels. Although leaching from the nutrient-rich sediments limited the effectiveness of this particular experience, the project demonstrated the technical feasibility of the procedure for restoration of small lakes."

Dilutional pumping did not renew Snake Lake. Severe oxygen depletions persisted and prevented the establishment of a sport fishery. Snake Lake had received sewage for many years and its sediments were nutrient-rich. Born et al. (1973b) suggested that this dilution technique might be a more effective means of reducing nutrient levels in a more "normal" lake.

Results of studies using nutrient-poor surface waters for dilution are more encouraging. Generally, adding dilution water from municipal supplies or nearby rivers results in an increase in nutrient export, a reduction in
the prevalence of blue-green algae, and an increase in clarity. Although little documentation on dilution exists, studies at Green and Moses Lakes, Washington, and at Buffalo Pound Lake, Canada, illustrate that this procedure can achieve algal control.

Flushing rates of 3.5 times per year or less significantly decreased nutrient levels in Green Lake and Buffalo Pound Lake. After five years of flushing in Green Lake (surface area = 104 ha, mean depth = 3.8 m), the blue-green algal standing crop was suppressed and there was a shift in dominance, with the elimination of Aphanizomenon (Dunst et al., 1974) and a major increase in macrophytes. Transparency also improved greatly. In this case, water from the city (Seattle) supply entered the lake at the bottom. During the summer this water is generally lower in temperature than the lake. Since water flows out of Green Lake over surface weirs, the phytoplankton may have been displaced from the lake at a rate greater than that predicted from equations applicable to completely mixed systems (Oglesby, 1969).

Dilution water from the Columbia River has been added to Moses Lake (surface area = 2753 ha, mean depth = 5.6 m) since 1977. During 1977 and 1978 there was more than a 50 percent improvement in phosphorus, chlorophyll-a, and Secchi depth values for the period April through July during both years. Although total phosphorus concentrations were not maintained at the predicted low levels, chlorophyll-a was held at surprisingly low average levels during both 1977 and 1978 (Welch, 1979). These low chlorophyll-a concentrations were later determined to be caused by nitrogen limitation, which was temporarily removed by increased nitrate in the inflow after Mount St. Helen's eruption (Welch et al., 1984).

Blue-green algae did not grow in Moses Lake water diluted as little as 25 percent. They were thus washed from the lake at a rate consistent with the water exchange rate (Welch, 1979). While blue-green algae decreased, diatoms increased more than twenty-fold from August 10 to September 29, indicating a preference for diluted lake water in contrast to blue-green algae. Chlorophyll-a remained low during dilution, but biomass remained higher than predicted due to the prevalence of diatoms and growth rates greater than the exchange rate (Welch, 1979).

Studies have shown that the maximum biomass of problem algal species can be reduced in direct proportion to the amount of dilution water added, provided that dilution lowers the concentration of the limiting nutrient (Welch et al., 1972; Dunst et al., 1974). However, there will not be a proportional decrease in total biomass, due to a shift toward species requiring a relatively smaller amount of the limiting nutrient.

Dilution has potential even where a supply of nutrient-poor water is not available. Welch (1979) asserts that the ideal dilution scheme is to attain a long-term reduction of the limiting nutrient through low-rate input of nutrient-poor water; however, if only moderate or high-nutrient water is available, "short-term dilution may work well because of an unknown cause for blue-green inhibition and/or effective washout of growth-limited populations." In general, since algal biomass is a function of both growth rate and cell washout rate, the biomass will decrease as cell washout rate
or dilution rate approaches the growth or doubling rate of algal cells (Welch et al., 1972).

B. ASSESSMENT

Dilution/flushing can improve reservoir water quality by increasing the washout rate of algal cells and reducing nutrient concentrations. Along with dilution/flushing, efforts should be made at reducing nutrient inflows. If a supply of nutrient-poor water is available, the costs involved include the capital cost of facilities to deliver the water, the cost of the water itself, and the costs of maintenance and operation. These costs will be highly variable and may be limiting in some cases.

In Kansas, dilution with clear nutrient-poor water could conceivably exert a positive influence on sport fishing by increasing dissolved oxygen, controlling nuisance algae, and improving water clarity. There is little likelihood that nutrient levels would decrease to the point where fish production would be nutrient limited. Likewise, there is every reason to expect that the impact on drinking water quality would also be favorable, assuming that the dilution water was relatively free of objectionable contaminants.

Unfortunately, many reservoirs in Kansas are already rapidly flushed with nutrient-rich water. Water is scarce in Kansas, making it difficult to find dilution water and undesirable to release stored water unnecessarily. Furthermore, what water does exist is generally nutrient rich and, therefore, unsuitable for dilution/flushing. Practical considerations rule out the use of dilution/flushing for most Kansas reservoirs, especially the larger reservoirs.
Most reservoirs in Kansas thermally stratify into two or three distinct layers (strata) in the summer, a phenomenon that can be detrimental to both water quality and sport fish. The surface stratum, the epilimnion, is warm, wind circulated, and aerobic. The middle stratum, known as the metalimnion or thermocline, is characterized by a sharp temperature drop and is the transitional zone between the epilimnion and the lowest and coolest stratum of the reservoir, the hypolimnion. In unstratified reservoirs, natural wind-driven currents carry dissolved oxygen downward into the deeper regions of the reservoir. When a reservoir stratifies, the hypolimnion can no longer be reoxygenated; and its oxygen supply is soon depleted by a series of chemical and biochemical reactions stemming from the decomposition of organic detritus. When the dissolved oxygen concentration in the hypolimnion declines below the level required by fish (the exact level being species specific), they are thereafter confined to the epilimnion where they may be adversely affected by warm temperatures, overcrowding, or algal blooms.

In an anaerobic hypolimnion, a number of important chemical reactions can take place, including reduction of sulfate to hydrogen sulfide, reduction and solubilization of iron and manganese, and anaerobic biodegradation of organic materials, releasing ammonium, phosphorus, carbon dioxide, and soluble organic matter into the hypolimnetic water. During the fall or spring, as the temperature of the epilimnion approaches that of the hypolimnion, wind currents are able to mix the entire reservoir, an event known as "turnover." When hypolimnetic waters having a high oxygen demand are mixed with epilimnetic waters, the dissolved oxygen level of the mixture can be too low to sustain certain fish species. Also, the release of ammonia, phosphorus, hydrogen sulfide, dissolved iron and manganese, and biodegradable organic matter into the epilimnion can cause rapid oxygen depletion, algal blooms, fish kills, and other water quality problems.

It has long been recognized that many of the aforementioned problems could be avoided if stratification could be prevented or broken up, and if an adequate concentration of dissolved oxygen could be maintained in the hypolimnion. Many attempts have been made to accomplish this through artificial destratification and hypolimnetic aeration. The usual methods are: 1) releasing compressed air near the bottom, which transfers oxygen to the water and subsequently rises to produce a mixing action; 2) pumping hypolimnetic water to the surface or surface water to the hypolimnion; and 3) aerating the hypolimnion without destroying the thermal stratification. This chapter discusses the effects of artificial destratification of reservoirs on water quality, algae, and fish. Chapter 11 discusses hypolimnetic aeration, a closely related management practice.
Both mechanical mixing and diffused air mixing have been used with success. The advantages and disadvantages of various devices are summarized by Johnson (1984), who points out that each device is best suited for particular applications and that the designer must select the most appropriate device for a particular application. Holland (1984) gives guidance for the design of hydraulic destratification systems, and an article by Lorenzen (1977) summarizes the power requirements and air flowrates used to treat a variety of reservoirs. Additional design information is provided by Lorenzen and Fast (1977), Pastorok et al. (1981), and Henderson-Sellers (1984).

**Temperature and Dissolved Oxygen.** Temperature and dissolved oxygen are very important factors with respect to both water quality and fish, are easily measured, and are quite different in the epilimnion and hypolimnion of a stratified lake. Therefore, they are frequently used as measures of the effectiveness of destratification. The following paragraphs discuss the influence of destratification on temperature and dissolved oxygen.

The time required to destratify a lake can vary from less than a day to about two weeks, depending on the size of the lake and the equipment used. Garton (1976) reported that after four days of pumping Ham's Lake (surface area = 40 ha, maximum depth = 9.5 m), a 10.1°C temperature difference was reduced to 1.2°C. The entire body of water continued warming after pumping began (a common effect of artificial destratification) and remained between 27° and 29°C until the fall turnover in early September. In the same four days the hypolimnetic dissolved oxygen concentration increased from zero to 2.9 mg/L, while the dissolved oxygen in the surface water decreased from 8.0 to 7.2 mg/L. Subsequently, the dissolved oxygen remained above 2.0 mg/L in the hypolimnion and above 5.0 mg/L at the surface. In this study surface water was pumped downward from a floating platform using a 746 watt electric motor, a 100:1 gear reducer and a 1.83 m propeller, producing a flowrate of 1.58 m³/sec. According to Garton (1976) and Strecker et al. (1977), this pump destratified the lake more quickly and maintained a slightly lower temperature difference than a pump with a flowrate of 0.65 m³/s used in earlier studies.

In Summerfelt and Cross's (1983) study, McFarland Lake (surface area = 2.8 ha, maximum depth = 5.3 m) was thermally and chemically destratified within 24 hours. During this period the dissolved oxygen in the hypolimnion increased from 4.8 to 11 mg/L, and then remained uniform at about 11 mg/L, much more than necessary to sustain fish and other aerobic organisms. This study, as well as those of Garton (1976) and Strecker et al. (1977) employed continuous mixing; but other studies have been successful using periodic mixing (e.g., Symons, 1970).

In general, properly implemented artificial destratification techniques are successful in nearly eliminating the thermal gradient and replenishing hypolimnetic oxygen. Steichen et al. (1979) found that a longer period of time was needed to achieve uniform dissolved oxygen concentrations than to achieve thermal destratification, and this is normally the case.

Undersizing of the pump has been one of the key technical problems associated with artificial destratification of reservoirs (Pastorok et al.,
1981). At times, in larger reservoirs, the thermocline may be lowered by axial flow pumping, but the reservoir may not destratify and the bottom will therefore remain anoxic (Toetz, 1977). Dissolved oxygen profiles are a useful indicator of the effective mixing depth (e.g., Toetz, 1981).

**Water Quality.**---Destratification, by introducing oxygen into the deeper regions of a reservoir, will substantially alter the quality of bottom waters. In the presence of even a low concentration of oxygen, sulfide production, iron and manganese reduction and solubilization, and anaerobic biological activity in the water column will be prevented, greatly improving water quality. Also, internal cycling of nutrients from the sediments will be retarded, since anaerobic biological activity in the uppermost layer of sediment will be greatly reduced. Epilimnetic water quality may improve or deteriorate, depending on changes in temperature, nutrient availability, algal biomass, mixing depth, etc.

Pastorok and Grieb (1984) summarized the responses of a large number of reservoirs to artificial circulation by diffused-air systems. Only 67% of the reservoirs had a post-circulation top-to-bottom temperature gradient of less than 3°C, so the results appear to be from a combination of destratification and hypolimnetic aeration systems. A majority (>50%) of the reservoirs for which data were available showed decreased Secchi depth (increased turbidity), increased dissolved oxygen, decreased ammonium, decreased iron and manganese, decreased blue-green algae, and an increased ratio of green to blue-green algae. Phosphate, total phosphorus, nitrate, pH, algal density, and chlorophyll decreased more often than they increased. Green algae increased in 39%, decreased in 22%, and showed no change in 39% of the reservoirs.

Kothandaraman and Evans (1983) reported the use of a low energy (1.5 HP) mechanical, reversible draft destratifier to restore Lake Eureka (36 acres, 18 feet maximum depth) to a condition suitable for use as a public water supply source. The lake was completely destratified and aerobic throughout. Iron and manganese were reduced by 97% and chlorine demand by more than half. Blue-green algae declined to insignificant levels, and the taste and odor problems that had previously caused abandonment of the supply no longer occurred.

**Algae.**---Artificial destratification can sometimes be very useful as an algae control technique, preventing extensive algal blooms that can lead to massive fish kills, nuisances for recreational users, increased taste and odor problems, and increased water treatment costs (see Chapters 2 and 4). Fast (1975) postulated several possible mechanisms by which artificial destratification might reduce algal populations: 1) reduction of internal nutrient loadings by prevention of anaerobic conditions; 2) reduction of photosynthesis by increasing the mixing depth of the algae; and 3) the effects of turbulence and rapid changes in hydrostatic pressure on the algae as they are swept through large vertical distances. Blue-green algae are naturally buoyant, and prefer to reside near the surface, perhaps for better access to atmospheric carbon dioxide (Klemer, 1983). This may also provide them with a competitive advantage in turbid water, as discussed earlier, since they can remain within the euphotic zone.
The effects of artificial destratification on algae vary, with some investigators reporting increased algal biomass and others reporting decreased algal biomass (Knoppert et al., 1970; Fast, 1979; Toetz, 1981; Pastrok & Grieb, 1984; Henderson-Sellers, 1984). However, the majority of reports indicate that destratification has a beneficial effect on algal dominance, causing a shift from blue-green algae to the more desirable green algae. Frequently, blue-green algae are almost completely eliminated. A shift in algal dominance from blue-green algae to green algae is significant because green algae are more beneficial to fish and cause fewer taste and odor problems (Steichen et al., 1979).

The shift from blue-green to green algae and changes in algal biomass can be attributed to changes in pH (Lorenzen, 1977; Shapiro, 1979), nutrient availability, mixing characteristics, zooplankton abundance, and habitat availability (Fast, 1979; EPA, 1980a; Forsberg & Shapiro, 1981; Pastrok & Grieb, 1984). In the past the effects of destratification on algal abundance and dominance have been considered unpredictable. However, recent research indicates that such changes can be predicted based on mixing depth and expected changes in nutrient availability (Lorenzen, 1977; Forsberg & Shapiro, 1981; Pastrok & Grieb, 1984). If algal productivity is considered to be either light limited or nutrient limited as a function of the mixing depth, then as shown by Lorenzen (1972 and 1977), algal biomass will increase upon mixing if the algae are nutrient limited and decrease upon mixing if the algae are light limited.

Reservoirs that are continuously aerated may experience algal blooms if the mixing device is shut down. Streeker et al. (1977) and Garton (1976) report that because of blown fuses, their pump was shut down from 8:00 p.m. on July 6 to 9:00 a.m. on July 7 and from 5:00 p.m. on July 8 to 9:00 a.m. on July 9. An algal bloom was observed the morning of July 9.

**Fish.**—Although data on the effects of artificial destratification on fish are scarce, the available data suggest mostly beneficial effects. Laverty and Nielson (1970) state that during the period of aeration of Lafayette Reservoir (surface area = 130 acres, average depth = 30 ft), there was an increased food supply (plankton), a greater volume of water containing this supply (due to the homogeneity), frequent spawning, and exceptional growth rates. Bass are the only fish mentioned in the report.

Gebhardt and Clady (1977) noted good results in Arbuckle Lake. Seasonal growth rates of gizzard shad and channel catfish, bottom feeding fish, were greater during the destratified period when more of the bottom area was available for feeding. White crappie and white bass (pelagic feeders) were not significantly affected. They concluded that:

"Mechanical mixing increased the amount of AFH (available fish habitat) in late summer which probably benefitted growth, but the best growth occurred when AFH was expanded in early summer. In order to maximize growth, mixing should begin in early May with the objective of preventing vertical stratification at any time during the summer." (Gebhart & Clady, 1977).
Gebhart and Clady (1977) also called attention to a study by Johnson in 1966 that measured growth of fish before and after destratification of a lake used to rear coho salmon. The average growth of the fish decreased slightly after destratification, probably due to an increase in survival rate from 12.9% before to 42.1% after destratification, but this improved survival rate increased total production by over 300%.

In addition to its effects on the growth of fishes, artificial destratification can also be used to prevent fish kills. Summerfelt and Cross (1983) report that winter aeration can effectively prevent winterkill, thereby eliminating the need for costly restocking of fish and allowing for individual fish to gain additional years of growth. They also found that summer aeration was successful in eliminating winterkills:

"Artificial aeration in the summer reduced WBOD (water column BOD) concentrations in McFarland Lake, a chronic winterkill lake, such that depletion rates of DO (dissolved oxygen) under ice cover were reduced and sufficient DO was maintained throughout the winter for survival. Summer aeration proved to be as desirable an alternative to the traditional winter aeration strategy for prevention of winterkill of a lake's fish population."

There has been at least one account of a fish-kill associated with artificial destratification, as reported by Nicholls et al. (1980). Artificial destratification of Heart Lake, Ontario, in 1976 precluded the development of blue-green algae, but led to an increase in the density of herbivorous zooplankton which controlled the development of smaller planktonic algae. According to Nicholls et al., Ceratium hirundinella flourished because there was little competition for nutrients from other algae and because the Ceratium cells were too large to be grazed by the zooplankton; the sudden collapse and subsequent decomposition of the Ceratium population depleted dissolved oxygen and resulted in a fish-kill.

In 1976, the Kansas Fish and Game Commission began an aeration program. The aeration system Fish and Game uses is known as a helixor unit. The unit is comprised of three plastic tubes through which air is pumped. The tubes are connected to a fifteen horsepower blower and placed in the hypolimnion. At both Pottawatomie and Neosho State Fishing Lakes, the helixor's performance exceeded expectations. Before the unit was set up, fish were confined to the upper six feet of Pottawatomie. After running the blower, biologists found fish at depths as great as fifteen feet. The unit produced similar results at Neosho. Fish, confined to the upper fifteen feet of the lake prior to aeration, began to appear down to the twenty-five foot depth after installation of the helixor. At Pottawatomie, the channel catfish catch better than tripled in 1976, the first year of aeration, and the 1977 catch outnumbered 1975 figures by over 500 percent (Mosher, 1979).

KFGC is also using the helixor to lengthen the growing season for fish. Reservoir managers operate the unit during the warmest part of the day in early spring and fall. During the summer, managers run the helixor during the coolest part of the day to prevent overheating of the water. Four hours
a day of running time has been adequate to prevent the reservoir from stratifying (Mosher, 1979).

**B. ASSESSMENT**

Artificial destratification of reservoirs in Kansas should generally improve both water quality and sport fishing, reduce the cost of water treatment, and improve the taste and odor of the finished water. Destratification will result in increased hypolimnetic dissolved oxygen, a shift from ammonia to nitrate, and decreased concentrations of iron, manganese, sulfide, and perhaps ammonium and phosphate in the hypolimnion. In general, a decline in populations of blue-green algae can be expected; total algal biomass may increase or decrease, but will probably decrease as a result of increased turbidity. Following startup of destratification in a typical nutrient-rich Kansas reservoir, an increase in algal biomass may occur as nutrients in the hypolimnion are upwelled into the photic zone. However, the combination of high turbidity and mixing is expected to limit photosynthesis, ultimately resulting in decreased algal biomass and a higher ratio of green to blue-green algae.

An American Water Works Association committee (1971) surveyed 29 reservoirs that used artificial destratification techniques (55% used diffused air). The study revealed a success rate or 86%, with success defined as some combination of improved water treatment, improved raw-water quality, improved finished-water quality, and improved aesthetics in the reservoirs. Furthermore, 90% of the water-supply managers stated that they planned to continue the mixing operation.

In general, artificial destratification should have a positive effect on the warm-water sport fish residing in Kansas reservoirs (Fast, 1979), due to the improvement in water quality. Oxygenation of the hypolimnion will increase available fish habitat, but this improvement is likely to be only marginal for most sport fish in a typical Kansas reservoir, since high turbidity will exclude light from the hypolimnion and prevent visual feeding. In less turbid reservoirs, where the euphotic zone extends into the hypolimnion, destratification should be extremely helpful to both sport fish and water quality, since such reservoirs are likely to experience frequent algal blooms and severe oxygen depletion. In all reservoirs, destratification should help prevent both winterkill and summerkill. The major adverse effects of destratification on fish are increased water temperature (which will increase respiration rate and lower dissolved oxygen concentrations in the epilimnion) and possibly increased turbidity (if bottom sediments are resuspended or if increased hypolimnetic dissolved oxygen allows bottom-feeding fish to thrive).

Artificial destratification is only expected to be economically feasible for small reservoirs, generally those less than about 100 acres in size (Toetz, D.W., pers. comm., 1985). Large reservoirs, such as the large federal reservoirs in Kansas, would require an excessive amount of power and equipment to destratify; and because of their large surface area (and hence their long fetches) they are generally rather well mixed by wind action
anyway. Smaller reservoirs are more likely to be stratified and, when stratified, are much more likely to have anoxic hypolimnia.

The effects of destratification on TOC and on THM precursors are largely unknown. Summerfelt and Cross (1983) reported that destratification resulted in decreased water column BOD and decreased sediment BOD. Similar results have been noted by others and are to be expected on the basis of theoretical considerations. However, much of the soluble organic matter in a reservoir is expected to be relatively non-biodegradable, and there is probably not a close relationship between BOD and TOC or between BOD and THM precursors. It is entirely reasonable to expect that destratification will reduce concentrations of TOC and of THM precursors in the hypolimnion, but the magnitude of the reduction is difficult to predict. Epilimnetic TOC may increase or decrease, depending upon the effects of destratification on algal biomass and nutrient export.
CHAPTER 11
HYPOLIMNETIC AERATION

A. BACKGROUND

Hypolimnetic aeration, as distinguished from artificial mixing or destratification (see Chapter 10), retains thermal stratification of the water column while adding oxygen to the reservoir's deeper water. Equipment that can be used for this purpose has been described by EPA (1980a), Johnson (1984), and others; and practical design considerations have been discussed in detail by Ashley (1985). A good review of the practice is given by Taggart and McQueen (1981). Management applications of hypolimnetic aeration include reduction of internal nutrient loading for eutrophication control, improvement of water quality for domestic use, and prevention of fish winterkill (Ashley, 1983).

McQueen and Lean (1983) have given a general summary of the effects of hypolimnetic aeration on water chemistry:

"1) the thermocline remains intact and hypolimnetic temperatures are not appreciably increased;
2) hypolimnetic O$_2$ levels increase;
3) hypolimnetic PO$_4$ generally decreases;
4) hypolimnetic total phosphorus decreases;
5) NO$_3$ increases or remains unaltered;
6) NH$_3$-N increases or decreases;
7) total Fe and Mn decrease;
8) pH generally remains unaltered;
9) H$_2$S and CH$_4$ should be decreased; and
10) N$_2$ may be increased."

Ashley (1983) also notes that increased hypolimnetic turbidity may be a common feature of hypolimnetic aeration. However, he adds that this should not decrease epilimnetic transparency, since circulation currents are confined to the hypolimnion by thermal stratification.

McQueen and Lean (1983) also summarized the general biological effects of hypolimnetic aeration:

"1) chlorophyll levels are generally unaltered;
2) zooplankton populations usually exhibit little or no response although in one case abundance decreased by 45%.

3) benthic macro-invertebrates generally increase in terms of numbers, diversity and distribution;

4) cold water fish stocking programs are usually successful and endemic fish populations increase their depth distributions."

As for oxygenation of the hypolimnion, Fast et al. (1973) reported that after nine days of continuous aeration of a Michigan lake, hypolimnetic oxygen levels jumped from zero to 8 mg/L. Shortly thereafter, they reached and remained above 10 mg/L for most of the summer; and they often exceeded the surface values due to low hypolimnetic water temperatures and greater hydrostatic pressures. Ashley (1983) reported that aeration produced hypolimnetic oxygen values of about 2.7 mg/L compared to 0.2 mg/L in a control lake. He gives the following account:

"As the aerator supplied more oxygen, the sediments and water column consumed more oxygen. As the aerator was physically unable to exceed the additional oxygen demand, the system oscillated about a narrow range (1.6–3.7 mg/L) all summer."

McQueen and Lean (1983) found that, on the average, approximately 15 minutes of aeration per day were required to maintain oxygen levels greater than 4 mg/L. The aerator was only turned on every 3 or 4 days and the oxygen levels were rapidly increased over a period of 1–2 hours. However, during the winter when no air was added, the experimental enclosure experienced severely depleted hypolimnetic and metalimnetic oxygen levels. This was probably because low phytoplankton concentrations in the experimental basin resulted in little photosynthesis taking place. Also, winter bacterial counts in the experimental enclosure were higher than in the control, possibly resulting in higher oxygen demand.

McQueen and Lean (1983) and others have noted that high diatomic nitrogen concentrations are associated with hypolimnetic aeration, and that this could cause gas bubble disease in fish. However, there appear to be no reports of adverse effects on fish as a direct consequence of hypolimnetic aeration.

Bernhardt (1967), from his experiments on Wahnbach Reservoir, concluded that complete destratification and hypolimnetic aeration both had a favorable effect on water quality, but aeration of the hypolimnion had the additional advantage of preserving a quantity of cold water for use as drinking water. Fast et al. (1973) noted that hypolimnetic aeration is sometimes more desirable than destratification in fisheries management and in the provision of domestic and industrial waters, since complete mixing may promote increased algal growth. Ashley (1983) also recognized that hypolimnetic aeration can prevent winterkill in temperate lakes and can create "two-story" fisheries in warmer climates.
Hypolimnetic aeration normally reduces internal nutrient loading by interfering with release of nutrients from the sediments under anaerobic conditions. Garrell et al. (1977) state that although hypolimnetic aeration may be sufficient to decrease the phosphorus concentration and increase the rate of conversion of ammonia nitrogen to nitrate nitrogen for internally cycled nutrients, considerable loading from external nonpoint sources can easily overwhelm the system. For watersheds in which external loading dominates, other management techniques should be considered where nutrient reduction is desired.

B. ASSESSMENT

Hypolimnetic aeration of thermally stratified reservoirs in Kansas is expected to have mostly positive effects on both water quality and fishing. Elimination of anoxic conditions in the hypolimnion will reduce concentrations of sulfide, soluble iron and manganese, ammonia, oxygen demanding substances, and taste and odor causing substances, all of which are undesirable from a water treatment standpoint. Compared to destratification, the hypolimnetic water temperature will be cooler, improving the aesthetic quality of drinking water and perhaps having ecological consequences. Internal cycling of nutrients will be reduced, including movement to the surface waters, which could stimulate algal growth. However, such nutrient movement will have little impact on reservoirs receiving heavy external nutrient loadings, as is typical in Kansas.

There is very little information available on the impacts of hypolimnetic aeration on algae, zooplankton, or fish. Nevertheless, it is reasonable to expect that sport fish will be favorably affected by the increase in hypolimnetic dissolved oxygen, which will increase their available habitat. However, if turbidity is high, as is typical in Kansas, visual feeding fish will be unable to feed efficiently in the hypolimnion. Nevertheless, the increased habitat for benthic organisms and forage fish should still enhance sport fishing, if only marginally. The incidence of both summerkill and winterkill should be reduced by hypolimnetic aeration.

Since hypolimnetic aeration does not destratify a reservoir, it should have minimal effect on epilimnetic water quality, including nutrient enrichment, except for whatever effects may be brought about by reduced internal nutrient cycling. Hypolimnetic TOC and THM precursor concentrations may be reduced, but current knowledge of this subject is too limited to predict how significant the change might be.
CHAPTER 12
DREDGING

A. BACKGROUND

One technique that may be used to restore sediment filled or highly eutrophied reservoirs to their former condition is dredging. The methods available for removing sediments from the bottom of a reservoir may be classified as either mechanical or hydraulic (Pierce, 1970). Mechanical dredges are similar to conventional land-based excavation equipment, and can be operated from either the shore or from a barge. Hydraulic dredges usually operate from a work boat or barge and pump up a mixture of sediment and water that is transported to the disposal site through a pipeline. The hydraulic cutterhead dredge is the most commonly used piece of equipment for reservoir dredging. The design and operation of the dredging equipment can certainly influence the short-term impact of dredging on a reservoir, but the long-term benefits associated with dredging are virtually the same for mechanical and hydraulic dredges.

There are both positive and negative effects of dredging on the water quality and ecology of freshwater lakes and reservoirs. Peterson (1977, 1979, & 1981) has discussed the "environmental concerns associated with dredging" and has compiled summaries of past experiences in lake restoration using hydraulic dredging. He has identified the following effects of dredging:

1. Deepening of the reservoir;
2. Removal of potentially recyclable nutrients;
3. Removal of toxic substances;
4. Removal of rooted aquatic vegetation;
5. Improvement of fish habitat;
6. Increasing turbidity during the dredging operation, resulting in release of toxic substances and nutrients into the water, oxygen depletion, temperature alteration, and reduced primary production.

The main objective of dredging is usually to deepen the reservoir, thereby enhancing certain recreational and water supply uses. Dredging "improves boating, water skiing, fishing, and other uses impaired by the effects of shoaling" (Peterson, 1981). It was once thought that dredging would adversely affect fish populations, but there has been little documentation of adverse effects and it now appears that the effects are largely beneficial (Peterson, 1979). Fishing may especially benefit from reservoir deepening in the colder parts of the United States, where reservoirs must be at least 4.5 m deep to avoid winter fish kills (Toubier & Westmacott, 1976). Fishing, boating, and swimming may also be improved by the removal of rooted aquatic plants that often interfere with these activities.
Knauer (1984) reported an excellent example of the recreational benefits that can be obtained through a lake deepening operation. Creve Coeur Lake, an oxbow lake located in the Missouri Bottoms area of St. Louis County, MO, has been used since the early 1900's for recreational purposes. During the early 1900's, the lake had a surface area of approximately 400 acres and an average depth of 10 feet. By 1971 the lake, which had been subject to severe sedimentation from erosion of the surrounding watershed, had a surface area of less than 180 acres and an average depth of less than 2 feet. Between 1974 and 1981 approximately 4,800,000 cubic yards of sediment were removed, restoring the lake to 320 acres with an average depth of 10 feet. Knauer (1984) reports that Creve Coeur Lake now "provides excellent boating and fishing areas" as well as having additional usable park space on the graded and seeded dredged material deposits.

The most obvious benefit gained by water supply reservoirs from dredging projects is increased storage capacity. The storage capacity of a water supply reservoir must be great enough to supply the needed water plus the additional losses due to seepage and evaporation. In 1971 the U.S. Department of Agriculture recommended that small reservoirs in various parts of the country have depths ranging from 1.5 m to 4.5 m just to compensate for water loss through evaporation and seepage. In most states the supply capacity of reservoirs is decreasing due to sedimentation, and in many cases the best way to recover the lost capacity is through dredging. In Illinois this problem has been addressed by Stout et al. (1982, 1983) who report Illinois Environmental Protection Agency figures indicating that "approximately 30 percent of Illinois's 84 municipal water supply reservoirs are experiencing severe sedimentation problems, with losses of 1 percent or more each year." In response to this situation, a pilot study was completed on Lake Paradise, a small water supply reservoir near Matton, IL, that had lost 31 percent of its storage capacity since 1907. Stout et al. (1982, 1983) demonstrated that dredging is a feasible method to recover lost storage in Lake Paradise; and Lembke et al. (1983) showed the additional benefit of using Lake Paradise's sediment as an agricultural soil conditioner. It should be noted that although dredging may restore lost capacity to a reservoir subjected to severe sedimentation, the benefits will be short-lived if a watershed soil conservation plan is not put into effect to slow erosion (EPA, 1980a).

Nuisance growths of algae and macrophytes resulting from high nutrient levels (most often phosphorus) are a major factor affecting the water quality in many reservoirs (see Chapters 2 and 4). Dredging to remove these nutrients, which can be recycled through the bottom sediments, has promise in the restoration of eutrophied reservoirs, especially if combined with other restoration techniques. Peterson (1981) cites several examples in which the internal phosphorus load ranged between 25 and 66 percent of the annual load. From these values it is evident that internal loading can constitute a major portion of the total load and that removal of the sediment may reduce the nutrient problem. Breithaupt and Lamb (1983) reported that dredging sediments from Liberty Lake, Washington, "accomplished the goal of removing a substantial portion of the nutrient reservoir residing in the sediments of Liberty Lake" and that "this should help reduce the internal nutrient recycling found to occur in the lake." Of course, nutrient levels can be quite high, even at considerable depth, making it difficult to
reach sediments low in nutrients. Sediment sampling and analysis can reveal the depth of the nutrient-rich sediment layer. Also, to minimize macrophyte growth in turbid reservoirs, the dredging depth should exceed the depth of the photic zone. Stefan and Hanson (1981) have developed a technique that estimates the dredging depth required to minimize internal nutrient cycling by assessing the likelihood of thermal stratification followed by wind-induced circulation.

Sediment containing toxic materials poses a problem to some reservoir restoration projects. As with nutrients, one way of dealing with this problem is to remove the contaminated material. A major concern during sediment removal is the reintroduction of the toxicant into the water column (Peterson, 1981), which occurs when particles are stirred up by the dredge. In order to minimize turbidity and contamination problems, a number of specialized dredges have been developed (Montgomery, 1984), including the Mud Cat, the Oozer, several types of specialized suction-head systems described by Peterson (1979), and a pneumatic dredge that has been used to remove mercury-contaminated sediments (Spencer, 1984). Specialized dredges have been successfully used to dredge PCB-contaminated sediments from Japanese waterways (Peterson & Randolph, 1977). Suda (1979) compared the levels of turbidity produced by specialized dredges and conventional cutter-head dredges and found that there was approximately a "10-fold reduction in suspended sediments around the specialized Japanese dredges."

Rooted aquatic plants in the littoral zone pose three types of problems for reservoir managers: 1) they interfere directly with recreational uses such as fishing, swimming, and boating; 2) they may be responsible for a large portion of in-reservoir nutrient cycling (Peterson, 1979); and 3) they may greatly accelerate sediment accretion rates (Carpenter, 1983). Aquatic plants are usually more economically removed by harvesting or by chemical treatments than by dredging, but dredging has been shown to be a more permanent solution than other options due to the complete removal of the plants and the partial removal of their habitat. This effect was noted by Carlile and Brynildson (1977) while investigating the effects of dredging on Wisconsin spring ponds. They reported that 60% of one pond was covered with Chara beds before dredging, but five years after dredging only 28% of the pond area had been revegetated. One problem that arises when removing macrophytes is the possibility of clogging the dredge. Breithaupt and Lamb (1983) report that when dredging thick beds of Elodea in Liberty Lake, Washington, "the Elodea had a tendency to wrap around the auger and plug the suction intake of the dredge."

In clear lakes, removal of macrophytes by dredging could result in more frequent or more severe algal blooms due to decreased competition for available nutrients (EPA, 1973), unless the macrophytes had been releasing substantial amounts of nutrients and organic matter. However, algal growth in Kansas reservoirs is often light limited rather than nutrient limited, and removal of macrophytes is expected to cause a slight increase in turbidity due to increased erosion and a slight decrease in the concentration of soluble organic matter (which has a coagulating effect on turbidity). For this reason, and because nutrient release from the sediments may be retarded, removal of macrophytes by dredging may help to control algal
blooms in Kansas. Also, the decrease in organic matter will have a favorable impact on trihalomethane formation and sediment accretion rates.

Improvement of fish habitat, resulting in increased fish biomass and an increase in average fish size, is one of the benefits observed during fish surveys in reservoirs that have been dredged. Spitler (1973) reported that four years after a dredging project to improve fish habitat in Long Lake, Michigan, the average bass length had increased nearly 4.8 cm over predredging length. Other fish species either remained the same size or became slightly smaller. Carline and Brynildson (1977) also noted several beneficial effects of dredging on sport fishing in Wisconsin spring ponds. These include increased trout biomass, increased utilization of ponds by fishermen, and increased yield. It should be noted that dredging will temporarily destroy the benthic community of fish food organisms. Carline and Brynildson (1977) found that fish populations will be stunted until the benthic community becomes reestablished. One way of reducing this effect is to leave small sections of the bottom undredged.

Although dredging has many long term benefits, as discussed above, there are several short term effects that are detrimental to fish production and water quality. As previously mentioned, the main concern during dredging is resuspension of sediments. Colloidal particles of both organic and inorganic composition may have toxic substances or nutrients adsorbed to their surfaces. When sediments are resuspended some of the chemicals "may be liberated in soluble form and taken up by plankton," thus entering the food chain (Peterson, 1979).

There are generally two types of toxic chemicals that may be of concern when dredging: 1) synthetic organic chemicals, such as pesticides, herbicides, and industrial solvents; and 2) toxic metals. To illustrate some of the reasons for concern about toxic substances during dredging projects, Peterson (1979) cites a report by Dames and Moore on a pilot scale dredging project on Lake Vancouver, Washington. Before dredging, the aldrin concentration was approximately 0.012 mg/L, but the amount found during dredging was three times higher at one site and ten times higher at another site. Also, the return water from the settling ponds was found to have aldrin concentrations as high as 0.336 mg/L, more than 100 times the criterion level of 0.003 mg/L. This report indicated that the aldrin was probably not in soluble form, but adsorbed to suspended particulate material.

The release of toxic metals during dredging poses a similar problem to that of synthetic toxicants, although it is much less widespread, usually occurring in lakes near urban areas. In a study by Gambrell et al. (1983) of City Park Lake (a shallow, highly eutrophic municipal lake in Baton Rouge, LA), numerous sediment and water quality characteristics were measured. It was found that lead, zinc, and possibly cadmium may cause adverse environmental effects during dredging and disposal of the sediments. Lead levels in the recent sediments were found to be four to six times those in the original bottom soils, but there was only twice as much easily solubilized lead.
Like toxic chemicals, nutrients adsorbed to the sediments and dissolved in interstitial waters may be released during dredging. These nutrients have the potential for increasing primary production, especially if the suspended solids resettle quickly, allowing sunlight to penetrate the water. On the other hand, if the water remains turbid, primary production may be reduced due to decreased light penetration.

Oxygen depletion caused by rapid bacterial decomposition of organic suspended solids may also reduce primary production by killing phytoplankton. Fish may be killed by this same mechanism if the reservoir being dredged is small and large turbidity levels are produced. Increased turbidity levels may also reduce the oxygen concentration by allowing the water to heat to higher temperatures, lowering the solubility of oxygen. It has been noted by Peterson (1979) that "rapid decomposition may also result in a downward shift of pH if the suspended sediments are of a noncalcareous composition and provide little buffering capability."

A number of water quality and sport fishing concerns associated with dredging have been discussed. Most of the long term effects of dredging tend to be positive while the short-term effects are negative. Results from numerous dredging projects indicate that the long-term positive benefits far outweigh the short-term negative effects. In the future, as the need for good quality recreational and water supply waters increases and available reservoir space decreases, the use of dredging is expected to increase. Further research and improvements in the areas of cost reduction and control of sediment resuspension will help make dredging a more economically and environmentally sound form of reservoir restoration.

B. ASSESSMENT

Most reservoirs in Kansas are quite shallow, subject to heavy loads of stream-borne sediment, and designed with a life expectancy of perhaps 50-100 years. For many of these reservoirs, dredging is likely to become an attractive alternative at some point in time. The biggest obstacle to dredging is its high cost; but if two or more objectives can be accomplished at the same time, the cost of dredging can be reasonable compared with other alternatives. For example, it would not be reasonable to use dredging solely for weed control; but if increased storage capacity or depth were also needed, and if the cost of dredging could be shared among various users (e.g., fisherman, boat owners, swimmers, and water utility customers), then dredging might be the most cost-effective long-term solution to the combined problems of a particular reservoir.

Dredging is expected to produce significant and positive long-term benefits for both water quality and fishing, provided that adequate consideration is given to fish habitat needs and perhaps to reestablishment of the benthic community. The short-term negative impacts of dredging (increased turbidity, resuspension of nutrients and metals, increased oxygen depletion, etc.) can be minimized by careful planning and operation and are not expected to be significant relative to long-term benefits.
Dredging should be combined with implementation of reasonable measures to control sediment inputs from the watershed, and the potential benefits of nutrient inactivation should be examined where phosphorus inputs from streamflow and runoff are low. Careful consideration should be given to proper disposal of dredged materials and the return waters associated with hydraulic dredging, so that the nutrients or other contaminants in the sediment are not returned to the reservoir (e.g., Churchill et al., 1975; Montgomery, 1984; Raymond & Cooper, 1984). Also, an effort should be made to identify beneficial uses of the dredged material (e.g., Patin, 1981; Stout & Barcelona, 1983), giving proper consideration to the physical and chemical properties of the dredged material (e.g., Barcelona, 1981; Gunkel et al., 1984; Kelly et al., 1984), which may vary spatially in reservoirs. Unfortunately, dredge spoil is generally composed of very fine particles and does not usually make good topsoil (Riener, 1984); but the coarser material deposited in the riverine zone of a reservoir may be very suitable for application to agricultural land.
CHAPTER 13
DRAINING AND CLEANING

A. BACKGROUND

Draining and cleaning is a drastic measure that can be taken to restore a highly polluted reservoir. The reservoir is drained, partially or completely; and when the sediments are sufficiently dry, conventional earth moving equipment is used to remove sediment, decaying vegetation, and debris from the bottom of the reservoir. Thus, this method represents a combination of drawdown, weed harvesting, and dredging, in which removal of weeds and sediment is greatly simplified as a result of the dry conditions produced by drawdown.

Draining and cleaning is generally regarded as a last resort, and is likely to be implemented only when: 1) conditions have deteriorated to the point where most water users are willing to forego use of the reservoir for a period of time, generally one year; 2) an alternate source of water for domestic consumption exists, if complete draining is done; 3) the reservoir is relatively small, such that the reservoir can be drained and refilled in a reasonable amount of time and the cost of the lost water is relatively small; and 4) the cost of draining and cleaning is competitive with the cost of conventional dredging and weed harvesting methods or other reservoir restoration techniques. Because these conditions are seldom met, draining and cleaning is rarely used; but it can be a very effective restoration technique.

B. ASSESSMENT

There are many positive long-term benefits of draining and cleaning, including reservoir deepening, removal of nutrients and decaying vegetation, weed control, and consolidation of the sediments. In other words, the long-term benefits are the same as those associated with dredging, drawdown, and weed harvesting (see Chapters 4, 7, and 12). In addition, it is a relatively easy matter to cover the bottom of the reservoir after cleaning (using plastic and gravel, fly ash, sand, or other materials) to prevent or retard the release of nutrients into the water upon refilling. Also, rip-rapping of the shoreline can be done where shoreline erosion is a concern.

The impacts of draining and cleaning on water quality are expected to be largely beneficial: 1) decreased chlorophyll, organic carbon, and nutrient concentrations, due to removal of nutrients and organic matter and to deepening of the reservoir; 2) decreased temperature, as a result of deepening; and 3) increased hypolimnetic oxygen concentrations, resulting in lower concentrations of ammonia, hydrogen sulfide, and soluble iron and manganese.
It is difficult to predict how turbidity will be influenced. Turbidity could decrease slightly due to consolidation of the sediments, but the water used to refill a typical Kansas reservoir would probably be quite turbid to start with. Also, removal of large amounts of shoreline vegetation and organic debris would be expected to substantially reduce the concentration of organic matter in the reservoir. Since organic matter can have a coagulating effect on turbidity, the turbidity in a cleaned reservoir might very well increase.

In clear reservoirs, removal of shoreline vegetation can result in algal blooms because of reduced competition for nutrients. However, this is unlikely to occur in Kansas because algae are generally light limited in turbid reservoirs. In fact, it is more likely that turbidity will increase slightly, thereby helping to control algal blooms as well as the taste and odor problems associated with algae.

Draining and cleaning can have both positive and negative impacts on fishing. Obviously, if the reservoir is completely drained, the existing fish population will be eradicated and restocking will be necessary. After refilling and restocking, fishing should be significantly improved due to reservoir deepening, removal of dense growths of vegetation (which interfere with angling), and elimination of algal blooms. However, fish habitat can be severely impaired by draining and cleaning. Some shallow areas and shoreline vegetation, preferably near the reservoir inlet, should be left uncleated to provide habitat, and fish attractors can be constructed before the reservoir is refilled (see Chapter 14). Cleaning can eliminate much of the benthic community, but heavy siltation greatly impairs growth of a healthy benthic community in most reservoirs in Kansas, whether they are cleaned or not.
CHAPTER 14
DESIGN GUIDELINES

A. GENERAL CONSIDERATIONS

When planning and designing a multipurpose reservoir, the designer must carefully consider the many factors that may influence water quality, aesthetics, fishing, and other recreational activities. Important factors to be considered are site selection, morphometry, orientation, watershed activities, inlet and outlet capabilities, design aesthetics, extent of clearing, and fish habitat requirements.

As part of the planning of a reservoir intended to serve as a potable water supply source in Kansas, state policy (KDHE, 1983) requires a sanitary survey of the area to collect and evaluate information on various pollutants. This survey should include an evaluation of watershed, stream, and storage characteristics in terms of their natural and developed states. Most important are the activities on the watershed, significant sources of pollution and nutrients, water quality variations with time and distance, and pollution control facilities.

In selecting a site for a reservoir in Kansas, state policy (KDHE, 1983) requires the designer to consider topography, catchment area, potential pollution and nutrient sources, storage capacity versus dam and spillway required, geology, safety, and water rights. The consulting engineer must evaluate the watershed area for potential pollutant and nutrient contributions and project the effect that they may have on the eutrophication rate and water quality. If the effect can be interpreted as contributing to taste and odor, violations of drinking water standards, trihalomethane formation, or turbidity problems, special attention to additional management of the watershed or to design of the treatment processes will be required. For additional planning and design requirements, the reader should consult KDHE (1983).

Factors such as depth, shape, volume of water impounded, exchange rate, and inlets and outlets will affect nutrient accumulation and should be carefully considered in siting and design. Ideally, water supply reservoirs should be large and deep (but not deep enough to thermally stratify), with a watershed size and water exchange rate that will maintain turbidity at high enough levels to limit primary productivity. Primary productivity, especially in large reservoirs, is closely related to morphometry (e.g., Rawson, 1955), and there is an inverse relationship between the mean depth of a reservoir and its productivity (Fee, 1979), i.e., shallow reservoirs are more productive than deep ones of the same size. Similarly, smaller reservoirs and those with highly irregular shorelines tend to have more extensive littoral zones; thus, they tend to be more productive (see Chapter 2).

Small reservoirs in Kansas are likely to thermally stratify, resulting in decreased turbidity and increased primary productivity (O'Brien, 1975).
For a recreational lake, a basin with a minimum area of shallow water is best, since shallows frequently become choked with aquatic vegetation. These areas, when filled with dense rooted aquatic plants, are useless for fishing, boating, and swimming, and may become a breeding location for mosquitoes (Bennett, 1971). Also, the decomposing vegetation may seriously impair water quality and increase the concentration of THM precursors.

Reservoirs designed for maximum fish production, on the other hand, should be of medium size, deep enough to thermally stratify, and of highly irregular shape, having extensive littoral zones (O'Brien, 1975). Such reservoirs would be less turbid and more productive than the ideal water supply reservoir. Thus, the designer is sometimes forced to choose between maximum drinking water quality and maximum fish production. However, the natural topography of the area where a reservoir is to be constructed can greatly limit the range of choices left to the designer.

Reservoir (dam) orientation and topography can have a significant impact on water quality, due to their influence on solar insolation and on wind-driven currents. Reservoirs in basins open in the direction of prevailing winds will be better mixed and more highly oxygenated, but may require protection against shoreline erosion along certain sections of the shoreline. Reservoirs having their major axis in a north-south orientation will receive less sunlight, especially if bordered by hills or tall trees, and this will reduce productivity and improve drinking water quality.

Activities in the watershed can greatly influence both water quality and fishing, and reasonable measures should be taken to control the entrance of pollutants into the reservoir (see Chapter 15, Watershed Management). The American Water Works Association (1985) makes the following recommendations for siting and design of small reservoirs, some of which are also good suggestions for larger reservoirs:

1) the location of the watershed and reservoir should be selected to reduce the possibility of contamination;

2) the watershed should be clean (preferably grassed), free from barns, septic tanks, privies, and soil absorption fields, protected against erosion and drainage from livestock areas, and fenced to exclude livestock;

3) the reservoir should be at least 8 feet deep and should have the maximum possible percentage of water storage in areas over 3 feet deep.

During construction, care should be taken to ensure that the quantity of pollution which enters the stream and reservoir is kept to a minimum. Erosion control should be implemented at the beginning of the job. The Bureau of Reclamation (1977) suggests that roads, cut slopes, and borrow areas should be provided with terraces, berms, or other check structures if excessive erosion is probable. Holding ponds or contour ditches could also be used to reduce pollution inputs. Recreational developments such as boat ramps and boat docking facilities should be constructed at the same time as the dam to prevent contamination of the reservoir by erosion after filling.
Also, KDHE (1983) recommends that site preparation include protection from floods during construction.

Careful attention should be given to design of inlet and outlet structures, particularly the latter. In some locations it is possible to control the quality of the water in the reservoir by allowing only good quality water to enter. This can be made possible by constructing an off-stream reservoir or by diverting polluted water around the reservoir (Chapter 15), but such alternatives are rarely feasible in Kansas. Another option is to design a treatment facility to remove nutrients from the inflowing water, but this is generally possible only for an offstream reservoir (e.g., Hayes et al., 1984), where the flowrate can be controlled, or for treating one heavily polluted stream of several feeding into a reservoir (e.g., Bernhardt & Schell, 1982). Water supplies for fish hatcheries have been biologically treated to remove ammonium (Caufield, 1985), as have public water supplies in Europe (Rittmann & Snoeyink, 1984), but such treatment applied to an influent stream of a water supply reservoir would be costly, probably ineffective (where phosphorus is the limiting nutrient), and perhaps detrimental (nitrogen removal could cause a shift to blue-green algae).

There are two methods of controlling influent water quality that are potentially effective and may be economically feasible: 1) construction of upstream sedimentation basins (e.g., Bureau of Reclamation, 1977; EPA, 1980a; Roberts, 1981), check weirs (e.g., Singh, 1981), or sediment traps (e.g., Siegel, 1979); and 2) passage of the inflowing water through wetlands (e.g., Herron et al., 1984; Weidenbacher & Willenbring, 1984). Large basins, perhaps naturally present or easily constructed, can remove much of the sediment and nutrients that would otherwise reach the main body of the reservoir. During times of drought or low flow the sediment can be removed and used for agricultural purposes. Wetlands can capture large amounts of sediment and nutrients during periods of high flow; and although some of the nutrients are later leached back into the water, there can be a net loss of nutrients (e.g., Herron et al., 1984) and a great reduction in the amount of sediment reaching the reservoir. Potential disadvantages of these alternatives include increased algal productivity (as a result of decreased turbidity) and increased organic matter due to increased productivity and to decomposition of algae and plants growing in upstream basins or wetlands areas.

The design of the outlet structure(s) is important in any reservoir. In very small reservoirs used for water supply there is generally an overflow structure or spillway accompanied by a single primary outlet (the water supply intake), which if placed too close to the bottom will draw in excessive amounts of turbidity and decaying organic material. It is best to design the outlet so that water can be withdrawn from at least three different levels in the reservoir, giving the water treatment plant operator an opportunity to select the best available water. Such an outlet also permits manipulation of downstream water temperatures and dissolved oxygen levels to create favorable conditions for spawning. Other considerations include capability to adjust the water level to control vegetation and enhance fish habitat (see Chapter 7), provision for draining and cleaning (see Chapter 13), adequate flow capacity, winter operation (ice problems), and avoiding entrainment of fish and debris.
Since most reservoirs are used for at least some form of recreation and will be visible to the public, proper attention should be given to the aesthetic qualities of the design. Design guidelines of the Bureau of Reclamation (1977) identify three goals in regard to environmental design: 1) keeping the natural beauty of the surrounding area intact; 2) creating esthetically satisfying structures and landscapes; and 3) causing minimal disturbance to the area ecology." Examples mentioned include a curved dam axis in lieu of a straight axis (to make the reservoir more nearly resemble a natural lake), providing a scenic overlook for viewing the dam and reservoir, designing necessary buildings with low profiles to blend with the surroundings, minimization of access roads, and erosion control. The embankment design should use materials found in and/or near the reservoir area, and a temporary viewing site, having signs which show the completed project and explain its purpose, should be constructed to promote good community relations. The Bureau of Reclamation (1977) also gives many suggestions for protection and enhancement of fish and wildlife.

When a reservoir has been filled with sediment to the point that an expensive dredging operation is under consideration, or when additional storage capacity is needed, it is sometimes possible to redesign the reservoir and raise the level of the dam, thereby increasing the size and depth of the reservoir. This requires modification of the dam, and perhaps the spillway and stilling basin, and careful attention must be given to dam safety and structural requirements (Singh, 1981). Raising the dam could be much cheaper than dredging and may provide most of the same benefits.

**B. FISH HABITAT ENHANCEMENT**

Many different things can be done during design and construction of a reservoir that will increase available fish habitat, promote spawning, improve the survival and reproductive success of sport fish, or facilitate commercial or recreational fish harvesting:

1) leaving standing timber and brush in a large percentage of the area to be inundated;

2) leaving shallow vegetated areas to provide shelter and food for young sport fish and suitable spawning areas for certain species of sport fish;

3) constructing fish attractors (e.g., brush or tire piles) to concentrate fish where they can be caught;

4) removal (or clearing) of degradable organic material from the basin prior to filling;

5) making adequate provisions for water level adjustment (see Chapter 7);
6) leaving a cleared fish-management pool in the bottom of
the reservoir to facilitate fish harvesting and manipula-
tion (Hulsey, 1959);

7) reduction of turbidity (to improve sight feeding) by
construction of a sedimentation basin, design of struc-
tures to control shoreline erosion, and implementation of
an erosion control program in the watershed;

8) adding gravel or rock to create areas suitable for spawn-
ing of certain fish species;

9) providing adequate depth to prevent winterkill; and

10) planning (in consultation with a fisheries expert) a
program to stock appropriate types of fish; and

11) considering fish habitat requirements during the design
stage.

Items 1 and 4 are obviously contradictory, reflecting the uncertainty
that exists as to the benefits of clearing or not clearing brush prior to
impoundment. There is general agreement that the other measures listed
above do in fact enhance sport fishing. Unfortunately, there is concern
that certain of these measures (namely, items 1, 2, and 3) will have an
adverse impact on water quality, especially the quality of drinking water,
and on the cost of drinking water treatment. There are four major areas of
controversy: 1) standing timber; 2) brush clearing; 3) construction of
brush piles or other fish attractors; and 4) shallow areas. Each of these
is discussed in the paragraphs below.

Standing Timber.--It is generally agreed that leaving standing timber
in reservoirs is beneficial to sport-fish populations. Reported benefits
include increases in reproductive success, fish-food organisms, sport-fish
standing crop, and sport-fish harvest (Layher, 1984). Although reproductive
benefits are difficult to measure and therefore somewhat speculative
(Layher, 1984), it is entirely reasonable to expect such benefits in reser-
voirs where structure would otherwise be rather deficient, as is typical in
Kansas. Such structure also provides habitat for many organisms that serve
as food for fish, particularly for young fish. While standing crops of many
sport fish are higher in timbered areas than in cleared areas, this does not
indicate an increase in overall productivity, but is presumably due to the
well known ability of structure to attract fish, a very positive benefit to
fishermen.

Hulsey (1959) has summarized the benefits of leaving large uncleared
areas in reservoirs to be used for fishing:

1) "... a substantial saving in the total cost of a reser-
voir, as compared to complete clearing;

2) the timbered areas will tend to keep wave action down and
prevent the waves from eroding the dam and shoreline;
3) the dead timber and litter will retard erosion when the shoal areas are exposed during drawdowns;

4) the organic material will produce carbon dioxide (from decomposition) which will help flocculate the colloidal clay turbidity;

5) the standing timber, litter and debris will tremendously increase the surface area exposed to the water for attachment of periphyton and other organisms, thereby increasing the productivity of the reservoir; and

6) the timbered areas will give a different type of fish habitat than the open water areas."

Water supply managers, on the other hand, generally take a much different view of standing timber and are reluctant to leave standing timber in a reservoir due to concern that it will:

1) clog the intake;
2) pose a hazard to boating;
3) release nutrients into the water;
4) release trihalomethane precursors into the water;
5) accelerate eutrophication and lead to a series of related problems;
6) increase the siltation rate; and
7) interfere with control of algae and macrophytes and with future dredging or cleaning operations.

Trees left standing in a reservoir decay at a very slow rate, and the tree trunks are likely to remain relatively unchanged and immobile for decades. Of course, if the water level drops significantly for an extended period of time, the tree trunks may dry out, begin to rot, and become buoyant enough to float away when the water level subsequently rises. Smaller branches will fall off or be broken off over a period of years, and these can indeed clog intakes and interfere with boating. Of course, standing timber can also screen out branches and debris carried downstream, so its net impact on clogging may be a positive one, depending upon its location.

Trees and their branches do pose a hazard to boaters, but this can be minimized by leaving them standing well above the waterline, which will make them clearly visible to boaters and provide perching habitat for birds (Layher, 1984). Also, areas where timber is left to attract fish can be marked with buoys and posted with speed limits.

Trees do contain nutrients (nitrogen and phosphorus) and THM precursors, but they decay so slowly that the amounts released are expected to be rather negligible. The indirect effects of trees on eutrophication and THM formation are a bit more difficult to predict, and are related to their ability to support growths of periphyton, their impact on siltation, and their interference with control of algae and macrophytes.
The submerged surfaces of trees provide a suitable substrate for attached algae (or periphyton). Such algae will remove nutrients from the water, leading to a short-term reduction in planktonic algae if the planktonic algae are in fact nutrient limited. If the planktonic algae are light limited, no reduction will occur. However, since periphyton will, on the average, remain in the reservoir for a much longer period of time than the planktonic algae, they will increase the retention of nutrients in the reservoir, leading to nutrient enrichment (eutrophication) and increased levels of organic matter. Such organic matter can coagulate colloidal solids (and the periphyton themselves can adsorb colloidal solids), increasing light penetration and hence increasing primary productivity (and phytoplankton growth). Therefore, substantial growths of periphyton would be expected to be ultimately detrimental to water quality and to lead to acceleration of the eutrophication process. Fortunately, most reservoirs in Kansas are quite turbid, and much less periphyton growth occurs on submerged surfaces than would occur in clear-water reservoirs.

Standing timber has not been conclusively demonstrated to increase or decrease siltation of reservoirs, so some difference of opinion exists as to its impact on siltation. Indeed, Layher (1984) believes that standing timber helps prevent siltation and shoreline erosion, and Hulsey (1959) states that timbered shoals help to prevent shoreline erosion. There is no question that large amounts of settleable solids flow into reservoirs in Kansas, eventually decreasing depth and causing significant deterioration of water quality. This occurs regardless of whether trees and brush are present. However, there are also large quantities of colloidal solids entering many reservoirs in Kansas, and capture of such solids can increase both the sediment accretion rate and primary productivity. The question then is whether or not standing timber results in the direct or indirect capture of a substantial quantity of colloidal solids.

Since colloidal particles can attach to surfaces, any structure in a reservoir will increase capture of such particles, thereby increasing siltation. The extent of capture will be related to the surface area. Since trees have a relatively small surface area compared to dense growths of aquatic plants and shoreline vegetation, they are therefore less important with respect to solids capture by an adsorptive mechanism. Nevertheless, they do increase siltation by directly capturing solids, even if the increase is rather slight; and this in turn can increase primary productivity when light is limiting.

As noted previously, organic matter plays a critical role in capturing colloidal solids. To the extent that timber decomposes (even slightly), supports growths of periphyton, and captures biodegradable debris, it will contribute to an increase in organic matter and hence an increase in siltation and primary productivity (due to the reduction in turbidity). The increased primary productivity will in turn result in an additional increase in organic matter, thereby creating a positive feedback loop similar to that described by Carpenter (1983). Hence, even a slight increase in organic matter can be greatly magnified over time, leading to substantial deterioration of water quality. As organic matter builds up in the sediment, it will cause poor sediment compaction, shortening the life span of the reservoir. Nevertheless, if a small quantity of standing timber is left near the inlet
area and in a few isolated coves, and if the timber covers only a small percentage of the total surface area of the reservoir, its impact on coagulation of colloids may be rather minimal.

Other important aspects of siltation are shoreline erosion and the resuspension of bottom sediments by wind-generated currents and wave action. Trees will reduce scouring of bottom deposits, thereby preventing their resuspension and perhaps their eventual exit from the reservoir. In this regard, trees clearly increase siltation, and indeed they serve as an anchor for shoals.

Shoreline erosion is undesirable, since a portion of the eroded material will be deposited in the reservoir, decreasing its depth. Shoreline erosion is caused primarily by wave action, something trees do not by themselves usually prevent or significantly retard. Unless trees near the shoreline are dense enough to capture a large quantity of debris, they will not prevent shoreline erosion. However, they will capture silt and eroded material, preventing its transport to deeper areas of the reservoir. In this way, they maintain (or even propagate) shallow areas, which do retard shoreline erosion. Hence, timber can slow down infilling of the deeper areas of the reservoir, thereby exerting a positive influence on the reservoir. Also, as noted by Hulse (1959), timber and debris help prevent erosion of shallow areas exposed during drawdown. Nevertheless, the decomposition of a large amount of debris near the shoreline can potentially cause a substantial increase in the concentrations of TOC and THM precursors.

Chemical treatment of small reservoirs to control algae and macrophytes is generally done by boat or barge, and standing timber can obviously impede such treatment. Also, the timber can interfere with mixing of the chemical with the water and can shelter organisms from the full effects of the chemical, leading to their rapid regrowth. Furthermore, small reservoirs in Kansas are likely to require dredging in the not too distant future to restore their capacity and improve the quality of the water. Therefore, it makes sense to restrict the amount and location of standing timber in a reservoir where dredging or chemical treatment for control of algae or macrophytes may be desirable at some point in time.

**Brush Clearing.**—Years ago, agencies such as the Bureau of Reclamation and the U. S. Army Corps of Engineers recommended total clearing of reservoirs during construction. The rationale was that removal of biodegradable materials and nutrient sources would slow eutrophication and improve water quality. Also, there was (and is) concern over hazards to boaters (Bronoski, 1979). However, in more recent times reservoirs, particularly large federal reservoirs, have been built with little clearing at all, with only those areas near water supply intakes, docks, beaches, etc., being cleared. This change in practice is a result of two considerations: 1) the high cost of clearing the entire impoundment of brush; and 2) its limited effectiveness in improving water quality, especially in large fertile reservoirs.

Depending upon the types and amounts of brush present, the brush may be quite beneficial to sport fish during the first few years of reservoir
operation. In fact, it has often been noted that fish production is better as the reservoir is filled and for several years thereafter, but drops significantly after several years of operation. The brush provides both habitat and nutrients, increasing fish production. Once the brush decays, this source of habitat and nutrients is no longer available. Certain benefits of brush to fish can be realized over extended periods of time by constructing brush piles of larger branches, as discussed in the next section of this chapter.

Since brush does provide a source of nutrients, it can contribute to poor water quality and to eutrophication. Indeed some reservoirs have been observed to be noticeably less eutrophic several years after filling than when first filled, giving water supply managers an incentive to design and construct reservoirs in such a way as to reduce this initial fertilization. Hence it is not uncommon for agencies to require or recommend complete clearing of small reservoirs. For single-purpose water supply reservoirs, the State of Kansas (KDHE, 1983) requires removal of trees and brush to the conservation pool elevation without major disturbance of the original surface area. For multipurpose reservoirs, some trees and brush may be left standing, the locations and amounts to be approved on a case-by-case basis.

Unfortunately, the clearing process can disturb the topsoil to such an extent that increased release of soil nutrients more than offsets any advantage of brush removal (e.g., Armstrong et al., 1979). This can be rectified by avoiding disturbance of the soil, by removing the topsoil, or by placing a layer of material (such as flyash) over the soil to prevent release of phosphorus during filling (e.g., Dunst et al., 1974). Gunnison et al. (1983) examined the relationship of soils and sediments to water quality and found that newly flooded soils had a capacity to exert high rates of oxygen consumption, carbon dioxide production, and methanogenesis.

In any event, many of the impacts of brush on both fishing and water quality are expected to be temporary, since the brush quickly decays. In rapidly flushed reservoirs, such as those in Kansas, the nutrients and organic matter released from the brush will be quickly transported downstream and the nutrient balance will soon be dominated by other sources. However, large brush that fails to decay rapidly, particularly if present in shallow areas, can be expected to increase primary productivity and sediment accretion rates.

**Fish Attractors.**—Fish attractors, constructed of brush, used tires, discarded Christmas trees, concrete pipe, or other materials are sometimes placed in certain locations within a reservoir to concentrate fish for harvesting by fishermen. Their ability to concentrate sport fish (most likely by concentrating their prey) has been thoroughly documented (e.g., Layher, 1984; Thomas & Wilson, 1984) and is not in question. It should be noted that fish attractors are not added to fertilize the reservoir, but only to attract fish, and that they are generally small structures placed in coves or near shoreline areas conducive to fishing. Unfortunately, water supply managers have a number of concerns regarding their use, concerns that are virtually identical to those for standing timber.
With regard to clogging of intakes, boating safety, and interference with chemical treatment or dredging of the reservoir, it is important 1) that brush piles be securely staked down or tied down to prevent them from being washed away (e.g., Harrison, 1974; Bureau of Reclamation, 1977); and 2) that all fish attractors be clearly marked with signs or buoys (e.g., Timmons & Shelton, 1982), not only to reduce hazards to boaters but so that fishermen will be able to determine their location. If a brush pile washes away, it will not only interfere with boating and possibly clog the water supply intake, but it will not serve its intended purpose. Properly constructed fish attractors should last many years (e.g., Bronoski, 1979).

The other concerns water supply managers have in regard to fish attractors are the release of nutrients and THM precursors by decaying brush and the potential for accelerated eutrophication and siltation posed by structures made of any type of material. It is obvious that brush piles will release a certain amount of nitrogen, phosphorus, and organic matter into the water. The amounts released will depend upon a number of factors, including: 1) the size and number of brush piles relative to the size of the reservoir; 2) the diameter of the branches used; 3) whether or not leaves or needles are stripped off or left on the branches; and 4) the species of the branches or brush used. It is difficult to predict just how much material might be released and how significant this might be, but proper construction of a few small brush piles in a eutrophic reservoir is unlikely to result in the release of substantial amounts of nutrients and organic matter. A reasonable assessment of the potential impact could be made by estimating the weight of material used, its chemical composition, and its rate of decay and comparing the nutrient loadings attributable to other sources, such as runoff and rainfall (e.g., EPA, 1980a).

With regard to siltation and eutrophication, fish attractors are expected to accelerate both. Their surfaces will provide habitat for attachment of periphyton and thereby increase nutrient retention. The structures themselves will increase capture of colloidal solids and material eroded from the shoreline and will impede resuspension of bottom sediments by currents. Indeed, by reducing turbidity and providing surfaces for periphyton growth, fish attractors function in a manner similar to macrophytes; thus, they can complete a positive feedback loop similar to that described by Carpenter (1983), thereby accelerating the rate of sediment accretion. Nevertheless, such effects might indeed be negligible if a small number of fish attractors of reasonable size are constructed in appropriate locations within the reservoir. In fact, it makes little sense to construct a large number of fish attractors, since that would defeat their purpose of concentrating fish in selected locations.

Shallow Areas.—Sport fish may require shallow vegetated areas for spawning and may also benefit from higher concentrations of food organisms present in such areas (see Chapter 7: Level Adjustment). This is especially true for young sport fish, who also benefit from the shelter from predators. Hence, a certain amount of shallow water is highly beneficial to sport fish. However, such areas are generally detrimental to water quality, due to increased productivity, increased warming, and growths of aquatic weeds. Therefore, such areas must be limited to protect drinking water quality.
If, during the design of a reservoir, it is decided to leave shallow areas for fish enhancement, such areas should be of limited size and located near the inlet, at a substantial distance from the outlet. Swamps, loose fill, and shallow areas near the outlet should be eliminated during construction. As noted by Arruda (1985):

"When lakes are constructed with large areas of shallow water, habitat is created for submerged aquatic plants (weeds) that can interfere with recreation and may increase the pool of dissolved organic compounds when the plants senesce and die at the end of the growing season. Importantly, these shallow areas could, if deeper, contribute to the total volume of the lake and further decrease the effects of nutrients inputs."

**C. ASSESSMENT**

Reservoir planning and design can have very substantial impacts on both fishing and water quality; and certain opportunities that exist during the design stage are lost once construction is underway. In general, water supply reservoirs should be relatively large and deep, but not deep enough to strongly stratify during the summer months. Shallow areas are beneficial for sport fish, but should be minimized to protect water quality. For most design decisions, the best choice is independent of whether fishing or water quality is the primary concern. However, conflicts can and do arise, particularly in regard to fish habitat enhancement, i.e., clearing of timber and brush, construction of fish attractors, and establishment of shallow vegetated areas.

Standing timber and fish attractors constructed of brush are unlikely to make a significant direct contribution to nutrient loadings or THM precursor concentrations, even in relatively small reservoirs. However, their indirect contributions to increases in primary productivity and sediment accretion rates are potentially quite significant. Nevertheless, in reasonable quantities and locations, they should have minimal impact on water quality in large reservoirs.

Small reservoirs are, by their very nature, susceptible to water quality problems associated with eutrophication; and the operating personnel may, due to lack of training, time, equipment, or financial resources, be hard pressed to deal with such problems. Standing timber and fish attractors can potentially create or aggravate water quality problems in small reservoirs, so caution and sound judgement should be exercised when incorporating them into the design of a small reservoir. In general, small areas of standing timber near the inlet may not pose a significant threat to water quality, and may in fact be somewhat beneficial. Fish attractors of reasonable size located near inlets and in a few small coves should cause relatively little harm to water quality. In small reservoirs, littoral vegetation is likely to exert a much more significant impact on drinking water quality than small amounts of standing timber and a few brush piles,
but all of these should be adequately controlled to protect the quality of the drinking water.

Total clearing of timber and brush from large reservoirs appears to be unnecessary and may represent a significant waste of time and money, although some clearing is usually necessary and desirable, especially in shallow areas and near beaches, docks, water intakes, etc. In small reservoirs, complete clearing may significantly improve the quality of the water during the first few years of operation, provided that care is taken to prevent increased release of soil nutrients.

Shallow vegetated areas incorporated into the design of a reservoir can be very beneficial to sport fish. In large reservoirs a reasonable amount of shallow water, properly located and managed, should have little or no impact on drinking water quality and treatment. In small reservoirs, areas of shallow water are much more likely to adversely affect water quality; they may necessitate an increase in the time and money spent on weed management and may increase the cost and difficulty of water treatment.

Standing timber, brush, fish attractors, and shallow areas located near the inlet and away from the outlet may actually help to improve water quality by removing sediment and nutrients before the water reaches the main body of the reservoir. In this way, the inlet area is converted into an artificial wetlands, serving as a sink for nutrients and sediment, providing habitat for fish and wildlife, and slowing eutrophication of the reservoir. The major problem with such an arrangement is that during storm events the material deposited near the inlet can be flushed into the reservoir, creating serious water quality problems and perhaps even fish kills. One possible solution is to build an upstream dam, a submerged dam, or a sedimentation basin, creating what would essentially be two reservoirs in sequence. The first reservoir would remove most of the sediment, would be more eutrophic (productive), and could be managed for maximum fish production. The second reservoir would have better water for drinking purposes (assuming that fine silt, passing through the first basin, maintained a high enough level of turbidity to limit algal growth); and it could be managed to maintain maximum drinking water quality.
CHAPTER 15
WATERSHED MANAGEMENT

A. BACKGROUND

Reservoirs may receive very substantial loads of nutrients, sediment, and organic carbon from their watersheds, leading to algal blooms, oxygen depletion, rapid infilling, and other consequences that can significantly impair water quality, fishing, and recreation. Such pollutant loads can sometimes be substantially reduced through implementation of appropriate watershed management practices, resulting in enhancement of both water quality and fishing. This chapter briefly reviews various sources of pollution originating in reservoir watersheds, identifies management options for reducing pollution from these sources, and assesses the potential impact of watershed management practices on water quality, water treatment, and sport fishing in Kansas reservoirs.

Potential sources of pollution originating in a watershed include agricultural runoff, municipal and industrial wastewater discharges, groundwater and septic tank seepage, various human activities in the immediate vicinity of the reservoir, atmospheric deposition, and runoff from urban areas, industrial sites, mining and drilling operations, construction sites, forests, natural grasslands, and roads. Methods for estimating the contributions of some of these sources to lake nutrient loadings are discussed by Uttermark et al. (1974) and EPA (1980a). Most reservoirs in Kansas are significantly impacted by human activities in the watershed, most notably by agricultural activities.

In most Kansas watersheds, the majority of stream-borne pollutants in excess of those that would occur naturally are attributable to agricultural runoff, which carries nutrients, sediment, organic matter, and pesticide residues downstream to reservoirs. Nutrient loads in agricultural runoff can be reduced by applying fertilizers at the proper time of the season, by storing manure during the winter rather than spreading it on frozen ground (EPA, 1980b), and by avoiding application of fertilizers to waterways, roads, and other areas not utilized for crop production. Since large fractions of the nitrogen, phosphorus, and organic matter in agricultural runoff are particulate, erosion control can greatly reduce the concentrations of nutrients and organic substances in agricultural runoff.

Sediment transport (erosion) is a problem for several reasons: 1) it represents a potentially significant loss of soil, which could eventually cause the land to become less productive; 2) nutrients associated with the sediment are displaced, causing a loss of soil fertility and enrichment of downstream reservoirs; 3) the sediment is deposited in reservoirs, greatly reducing their lifespan; and 4) pesticides adsorbed on sediment are transported downstream and may adversely affect aquatic food chains. There are many steps that can be taken to reduce sediment loss, including reduced tillage, changes in tilling equipment and timing, terracing, construction of sediment retention basins, planting of grass in waterways, and establishment
of untilled areas to encourage growth of riparian vegetation along stream banks and drainage ditches. Sundmacker (1981) has presented a good brief discussion of soil conservation practices; and extensive information on this subject is available from the Soil Conservation Service of the U.S. Department of Agriculture.

Riparian vegetation is very important in agricultural areas for a number of reasons (e.g., Schlosser & Karr, 1981; Lawrence et al., 1984). It removes sediment, nutrients, and organic matter from the runoff entering the stream, significantly improving water quality. It provides habitat for numerous terrestrial and aquatic organisms, including game. It helps to stabilize stream banks against erosion; and it provides shade, lowering stream temperature, reducing growth of filamentous algae, and improving habitat for many types of aquatic organisms. The benefits of riparian vegetation are substantial, not only in agricultural areas but also in urban settings.

Pesticides are widely used to increase crop yields by controlling insects and weeds. Many pesticides are hydrophobic and absorb on soil, accumulating most heavily in the smallest and lightest (most organic) particles, those most likely to be entrained in the runoff. Other pesticides are somewhat soluble and can dissolve into the water during a runoff event. Some pesticides are relatively persistent (non-biodegradable) and can accumulate in the sediments of streams and reservoirs, gradually working their way into aquatic food chains. Certain widely used herbicides (e.g., atrazine) inhibit weed growth by blocking photosynthesis, and, if transported into reservoirs via runoff, can potentially inhibit phytoplankton photosynthesis, thereby altering ecological conditions in the reservoir (deNoyelles & Kettle, 1980). Control of sediment loss can greatly reduce pesticide concentrations in runoff, and can also decrease the need for pesticides. The amounts of pesticides reaching a reservoir can also be reduced by selecting non-persistent pesticides, careful application (with attention to timing, formulation, dosage, and placement), crop rotation, utilization of pest-resistant plant species, and exploitation of alternatives to pesticide use (e.g., pheromone traps).

Feedlots can be a very significant source of agricultural pollution, since a large number of manure producing animals are concentrated in a small area to facilitate feeding. Only a limited amount of the feed is converted into animal tissue (perhaps 10-50%), and much of the remainder is excreted as manure and urine, generating large quantities of nutrients, biodegradable organic matter, suspended solids, and perhaps synthetic chemicals and pathogens. When runoff travels across a feedlot, carrying these waste materials into a stream, the pollution problems that result are as serious as those created by untreated domestic sewage. The solution is to prevent runoff and to collect the wastes and treat them. Both aerobic and anaerobic treatment processes have been successfully employed, and anaerobic processes can potentially be used to generate energy.

Runoff from forests and grasslands also carries sediment, nutrients, and organic debris, but this occurs naturally and transports relatively small quantities of pollutants per acre compared to those in agricultural runoff. Nevertheless, human activity in natural forests and grasslands can
greatly increase the pollutant loads carried by runoff. Construction of roads, bridges, and utility lines (for communications, gas, and electricity) can scar the landscape, leading to great increases in erosion, especially in hilly or mountainous areas. Use of appropriate construction methods and careful attention to drainage problems can control this source of pollution.

Mining and oil drilling operations are another potential source of pollution. Depending upon the type of operation, pollutants can include sediment (from the operation itself or from increased erosion due to disturbance of the land surface), nutrients, metal ions, dissolved solids, acids (e.g., from oxidation of pyrite), and oxygen demanding substances. Preventive measures include construction of tailings ponds and sediment traps, control of runoff, and treatment of waste discharges.

Urban runoff can carry substantial amounts of sediment, nutrients, biodegradable organic matter, metals, and oils into nearby streams and reservoirs. In certain locations and under certain hydrologic conditions, urban runoff can be as heavily polluted as raw or treated sewage; and it is often more polluted than agricultural runoff (e.g., Glandon et al., 1981). Many steps can be taken to reduce the pollutant loads in urban runoff; but each of them is potentially cost-effective only under certain circumstances, and careful planning is required for effective and economical control of urban runoff. Field (1985) presents a very thorough discussion of the management options for controlling urban runoff, and Lee (1972) has identified numerous ways in which residents of a watershed can help to improve lake water quality.

Upstream discharges of municipal and industrial wastewater can significantly increase the amounts of nutrients, sediment, organic matter, and toxic chemicals entering a reservoir. Federal regulations now require treatment of virtually all wastewater discharges, but some pollutants may remain even after treatment. For example, municipal wastewaters must receive secondary treatment to remove oxygen demanding substances and suspended solids, but substantial amounts of nitrogen and phosphorus remain in the effluent.

There are a number of treatment processes (commonly designated as "advanced") that can be employed to remove nutrients and other pollutants from wastewater effluents, but the processes are generally quite expensive and should not be implemented without good evidence that the wastewater is a significant contributor to the entire pollutant load entering the reservoir. In the Occoquan watershed, near Washington, D.C., advanced wastewater treatment was installed to remove most of the nutrients originating from point sources, but large quantities of nutrients in stormwater runoff caused continued eutrophication and deterioration of water quality in Occoquan Reservoir (Randall et al., 1978). Bachman and Jones (1976), after studying 143 lakes and reservoirs in Iowa, concluded that removal of phosphates from detergents and sewage-treatment-plant effluents would not significantly lower algal productivity due to the high loadings of phosphate present in runoff. Also, phosphate removal will have little impact where primary productivity is light limited.
One option for dealing with upstream discharges is to intercept them before they reach the reservoir and route them around the reservoir to a point downstream. This technique is known as nutrient diversion, and it has been successfully employed, most notably at Lake Washington, in Seattle (Dunst et al., 1974). After diversion of treated wastewater effluents around the Lower Madison Lakes, Sonzogni et al. (1975) found improvements in winter phosphorus concentration, greater algal species diversity, fewer blooms of blue-green algae, and improved water quality; but the lakes continued to be highly eutrophic. Similar results were reported at Lake Sammamish, where diversion of one-third of the nutrient loading reduced blue-green phytoplankton by 50%, but had no impact on biomass or visibility (Welch et al., 1980).

In some cases a substantial fraction of the allochthonous pollutant load originates in the immediate vicinity of the reservoir, e.g., from shoreline erosion, leaching of water from nearby septic tanks, erosion along access roads, and fertilization of crops or lawns adjacent to the shoreline. Such sources of pollution can be minimized by establishing "sanitation zones" around reservoirs, in which strict sanitation codes and land use regulations are enforced (e.g., Beuscher, 1969; Ferguson, 1981). Shoreline erosion can be reduced by structural methods (e.g., Niccum, 1981) or by maintenance of a suitable cover of vegetation. Where septic tanks are a potential problem, e.g., where the water table is shallow (Sawney & Starr, 1977), careful attention to construction requirements and intermittent dosing with alum (Brandes, 1977) can reduce pollutant loads entering the reservoir. Centralized management of septic systems may be necessary to insure adequate maintenance (e.g., Otis, 1979).

B. ASSESSMENT

Many watershed management practices have already been implemented in Kansas, producing substantial benefits to water users. The need for additional controls in a particular watershed is best assessed by examining all the sources of the pollutant of interest within the watershed (e.g., EPA, 1980a), and within the reservoir itself, and then deciding which sources are significant, what benefits would be produced by their control, and what control methods are workable and affordable. It is worth noting that many watershed management practices produce benefits not only for downstream water users but also for those who implement the controls. For example, soil conservation not only reduces sedimentation of reservoirs but also improves soil fertility; riparian vegetation reduces soil loss; proper application of pesticides and fertilizers saves money; and precipitation of phosphorus from municipal wastewater increases the nutritional value of the sludge. All such benefits should be considered in a watershed management program.

In Kansas, most reservoirs receive substantial loads of sediment, nutrients, and organic matter from cultivated land. Any measures taken to substantially reduce soil loss in a watershed should result in substantial benefits to the reservoir, including increased life span, greater depth than
would be present in the absence of controls, higher dissolved oxygen concentrations, better water quality, and better fishing. Conceivably, nutrient concentrations might also be reduced below the levels needed to sustain growths of nuisance vegetation. However, most reservoirs in Kansas have very large ratios of watershed area to reservoir area, making upstream control of non-point sources of nutrients extremely difficult or impractical. Primary production in many Kansas reservoirs is often light limited, even if soil loss is kept to a minimum. Nutrient control is most likely to be effective for small thermally stratified reservoirs where primary productivity is generally not light limited (O'Brien, 1975).

While it is certainly undesirable to place any additional economic burdens on Kansas farmers at the present time, some steps can be taken to reduce pollution from agricultural sources, including the strengthening of educational programs and the adoption of policies that will encourage soil conservation practices without placing undue hardships on farmers (e.g., Radosevich & Skogerboe, 1979; Kramer et al., 1984). An excellent example of the latter is the very commendable proposal, included in the preliminary draft of the 1986 State Water Plan, to exempt uncultivated riparian land from property taxes (Kansas Water Office, 1985).

There is very little information available to serve as a basis for predicting the effects of watershed management on the concentrations of TOC and THM precursors in reservoirs; but in reservoirs such as those in Kansas, the majority of the organic matter is likely to be allochthonous. Furthermore, there are clearly substantial amounts of THM precursor materials present in agricultural runoff (e.g., Morris & Johnson, 1976). Therefore, practices that reduce downstream transport of soil and leaves should reduce TOC and THM precursor levels in reservoirs. However, it is important to note that large percentages of TOC and THM precursors are usually dissolved in the water, rather than in particulate form; and the impacts of watershed management practices on dissolved organic matter are largely unknown. Since it is the dissolved materials that most influence finished water quality, additional research is needed to determine whether these materials are autochthonous or allochthonous and to assess the feasibility of source control of dissolved THM precursors.
CHAPTER 16
RERESTCTION OF RECREATIONAL USE

A. BACKGROUND

Recreational use of a reservoir can result in a significant deterioration of water quality. Access roads and various human activities in the watershed can increase loadings of nutrients, suspended solids, and litter. Boat motors increase the concentrations of lead, oil, and fuel in the water, and can resuspend sediment from the reservoir bottom. Swimming increases turbidity and nutrient concentrations, while all body-contact sports increase the opportunity for spread of water-borne diseases. Because the managers of water supply reservoirs have historically been concerned primarily with the quality of the finished water, they have by-and-large done everything in their power to keep recreational use to a minimum (Miller, 1985). In water-rich regions, it is not uncommon for water supply reservoirs to be off limits to the public. Where water resources are more limited, water supply reservoirs must often accommodate some level of recreational use.

The single most important objective of water treatment is to provide potable water free of pathogenic organisms; hence, body-contact sports (swimming, skiing, etc.) are the most likely forms of recreation to be restricted. Since pathogen concentrations are the highest near their sources (e.g., Hendry & Toth, 1982), and since pathogens tend not to survive in warm eutrophic water for extended periods of time, body-contact sports can generally be accommodated in large water-supply reservoirs, especially if beach areas are located away from the water treatment plant intake. Body-contact sports are generally prohibited in smaller reservoirs due to concern over the spread of disease and because smaller reservoirs are generally unable to assimilate the increased loads of pollutants associated with such use.

Fishing is, in general, quite compatible with water supply, since the increased pollution attributable to fishing is expected to be largely offset by the removal of nutrients from the reservoir in the form of harvested fish. Of course, this assumes a reasonable number of fishermen and proper control of fish populations through stocking and other manipulations and through size and creel limits. In addition, access roads should be constructed in a manner that minimizes erosion and runoff; and trash bins and toilet facilities should be provided to encourage proper disposal of waste materials. If necessary, motor horsepower and boat speed can be limited to control lead, oil, and fuel inputs and to minimize resuspension of sediments.
B. ASSESSMENT

Increasing demand for water-based recreation, coupled with limited water resources, exerts pressure on the managers of water supply reservoirs in Kansas to accommodate various forms of recreation, all of which can have deleterious effects on water quality. In large reservoirs, properly managed body-contact recreation will have only marginal effects, if any, on water quality, water treatment, and sport fishing. However, public health and water quality considerations dictate that body-contact recreation be prohibited in most small water supply reservoirs (i.e., those less than several hundred acres in size). Fishing, if properly managed, is compatible with water supply in any size of reservoir; but smaller reservoirs are more vulnerable to fishing-related pollution and should be managed accordingly. Recreation is not expected to have significant effects on concentrations of THM precursors and TOC other than those associated with increased productivity due to increased nutrient availability.

In the State of Kansas, swimming is not recommended in any water supply impoundment except for designated federal reservoirs (KDHE, 1983). This position is justified not only on the basis of water quality protection but also for reasons of safety. Kansas reservoirs are typically very turbid; and poor visibility is known to contribute to drownings by interfering with supervision of swimmers (especially young children) and by impeding rescue efforts (EPA, 1980d). At least one state has set a standard on visibility for recreational waters used for swimming (Carr, 1976).
CHAPTER 17
RESERVOIR MODELING

A. BACKGROUND

Models, often mathematical computer-based models, are now frequently used by planners, designers, and managers of reservoirs to help them evaluate the impacts of a range of alternatives on such things as water quality, trophic state, nutrient loadings, fish production, thermal stratification, and downstream dissolved oxygen concentrations. This chapter briefly summarizes the types of models available and assesses their applicability to the problems facing managers of multipurpose reservoirs in Kansas used for water supply and sport fishing. Since the total number of models applicable to lakes and reservoirs is exceedingly large, no attempt is made to discuss each one individually. Rather, particular types (or classes) of models are discussed.

As noted by Reckhow and Chapra (1983), there are several different schemes that can be used to classify models. On a very basic level, models can be classified as mathematical, graphical, or conceptual. The emphasis of this chapter is on mathematical models, but the other chapters of this report in essence constitute a conceptual model of the functioning of reservoirs in Kansas.

Reservoir models can also be classified on the basis of their inclusion of physical, chemical, or biological phenomena. Physical models are used to predict or describe such things as temperature profiles, density currents, and circulation patterns. Chemical models, generally known as water quality models, are used to examine such things as dissolved oxygen concentrations and nutrient loadings. Biological or ecosystem models are used to examine aquatic food chains, fish production, and population dynamics. More complex models (e.g., Chen & Orlob, 1972; DiToro, 1976) incorporate physical, chemical, and biological phenomena in an attempt to approximate the complexity of real reservoirs.

Reservoir models differ greatly in their assumptions regarding the basic physical description of the reservoir. One-box models assume that a reservoir is a single completely mixed body of water. Two-box models view a reservoir as a two-story entity, having a completely mixed upper story (the epilimnion) and a completely mixed lower story (the hypolimnion). Other models are one-, two-, or three-dimensional. One dimensional models (e.g., Corps of Engineers, 1982) consider variations in one direction only, such as a variation in temperature with depth or a variation in nitrogen along the length of a channel. Two- and three-dimensional models accommodate variations in more than one direction, and are therefore more complex and perhaps better able to accurately describe certain phenomena.

Reckhow and Chapra (1983) have identified a number of additional criteria that can be used to classify mathematical models:
1. Empirical (based primarily on data fitting and correlations) or theoretical (based primarily on fundamental principles);
2. Simulating (descriptive) or optimizing (able to identify the best alternative);
3. Static (time independent) or dynamic (time dependent);
4. Lumped parameter (zero dimensional) or distributed parameter (one, two, or three dimensional in space);
5. Deterministic (excluding probability) or stochastic (probabilistic, i.e., attempting to deal with the uncertainty of the real world situation); and
6. Cross-sectional (describing the behavior of a group of reservoirs) or longitudinal (describing the behavior of a single reservoir over time).

Extensive research has been done for many years on eutrophication, which is generally at the heart of most problems involving water quality or poor fishing in reservoirs. For this reason many models, especially those that are widely used, are primarily designed to examine some parameter or group of parameters associated with eutrophication, including such things as dissolved oxygen, chlorophyll-a, total phosphorus, total nitrogen, Secchi depth, phytoplankton density, and primary productivity. The models that are used most extensively are those that describe phosphorus concentrations, since: 1) algal growth in a majority of reservoirs is phosphorus limited; 2) empirical cross-sectional models show a strong correlation between chlorophyll-a and total phosphorus; 3) algae are associated with most water quality problems; and 4) phosphorus inputs are often significantly influenced by human activity and can therefore be managed.

Mathematical reservoir models are frequently an integral part of research investigations and management programs (Scavia & Robertson, 1979), and are used for a variety of purposes. Examples include examination of the effects of management on ecosystems (e.g., Hutchins, 1982), prediction of water quality in new or altered impoundments (e.g., Bradford & Maiero, 1978; Yahnke, 1981; Antosch, 1984; Lee & Jones, 1984), and assessment of the degree of phosphorus control needed to achieve acceptable water quality (e.g., Lorenzen, 1974; Bradford & Maiero, 1978; Bingham et al., 1985). Nevertheless there are some severe problems associated with many uses of reservoir models, including poor predictive ability, improper application, and the expense of acquiring good input data. These problems are discussed further in subsequent paragraphs, with emphasis on modeling of phosphorus and chlorophyll concentrations.

Many models are empirical or semi-empirical, and hence are based upon historical data from a number of reservoirs (cross-sectional models) or from a single reservoir over a period of time (longitudinal models). The ability of a cross-sectional model to describe the functioning of a reservoir depends upon the degree of similarity between the reservoir of interest and the reservoirs upon which the data base was built, while the ability of a longitudinal model to describe a reservoir depends upon the adequacy of the data base and the ability of the model to accurately describe all of the important relationships influencing the variable of interest. Thus, poor predictive ability can result from inappropriate application, from use of
models that incompletely describe the reality of the reservoir of interest, or from failure to acquire an adequate data base.

There are numerous examples of poor predictive ability in the literature, only a few examples of which are given here. Smith and Shapiro (1981) analyzed time-series data for 16 lakes and showed that reductions in phosphorus produced consistent declines in chlorophyll, contradicting a cross-sectional model developed from the National Eutrophication Survey that predicted that phosphorus removal would not influence the chlorophyll concentration until a threshold level was reached. Moore et al. (1981) found that a computer simulation model calibrated with actual data from Wahiawa Reservoir in Hawaii afforded low predictive accuracy. Mueller (1982) examined a number of lake eutrophication models to determine their applicability to reservoirs in the Western United States and found only one that gave reasonably accurate results.

Misapplication of models is a major problem, stemming largely from a desire to save time and money by using an existing model (perhaps one that oversimplifies the problem) and by collecting as little data as possible. As noted by Shapiro (1979), determining the phosphorus budget of a reservoir can be both time consuming and expensive. Because various data bases from large numbers of lakes in the U.S. (e.g., Dillon & Rigler, 1974; Jones & Bachmann, 1976; Hern et al., 1981; Jones & Lee, 1982) and around the world (e.g., Sakamoto, 1966; Vollenweider, 1976; OECD, 1982) show a strong correlation between phosphorus and chlorophyll-a, there is a strong temptation to use an empirical trophic state model based on such a correlation when modeling a reservoir, especially a small reservoir. However, there are several problems with this:

1. Such correlations are often based on the logarithms of the concentrations (in order to linearize the data), and despite the seemingly good correlation coefficients there is actually a substantial amount of scatter in the data (e.g., Knowlton et al., 1984). Hence, the difference between the chlorophyll concentration predicted from the line of best fit and the actual concentration in any given reservoir can be very great—so great as to render the correlation entirely inappropriate for a particular reservoir (Shapiro, 1979).

2. As noted by Reckhow and Chapra (1983), a model that provides a good description of the cross-sectional behavior of a number of reservoirs may not be applicable to a single reservoir for time-series purposes (e.g., phosphorous may not even be the limiting nutrient under all conditions). Empirical trophic state models typically represent a time-varying process (phosphorus input and algal growth) with a single annual average value. A mass-balance technique that considers the time varying nature of water quality can greatly improve the analysis and, if kept simple, is appropriate for small reservoirs (Bingham et al., 1985).
3. As noted by Reckhow and Chapra (1983), almost all phosphorus loading research to date has focused on temperate natural lakes, whereas much of the water in arid climates is stored in reservoirs where factors such as turbidity and morphometry can significantly influence water quality. Pearse (1984) examined 40 reservoirs in South Carolina and found that chlorophyll-a levels were considerably lower than predicted by eight different models based on data from temperate northern lakes. Trophic state indices based on natural temperate lakes have been found inappropriate for TVA reservoirs (Cox, 1984) and Florida lakes (Edmiston & Myers, 1984). Higgins & Kim (1981) found that phosphorus sedimentation coefficients developed for natural lakes could not be directly applied to TVA reservoirs. Phosphorus-chlorophyll relationships are weaker in reservoirs than in natural lakes (Canfield & Bachmann, 1981), as expected due to differences in outlet depth (Wright, 1967) and turbidity, and to differences in sedimentation coefficients (Bachmann & Jones, 1976). (See Chapter 2 for a discussion of other differences between natural lakes and artificial reservoirs.)

4. There are considerable differences among the various phosphorus-chlorophyll relationships proposed in the literature (see Figure 6 in Shapiro, 1979 or Figure 8-22 in Reckhow & Chapra, 1983); therefore, data collection is necessary to select the most appropriate relationship for the region of interest. However, once the data are collected there is no longer a need to use correlations from the literature, although models similar to those found in the literature can perhaps be used.

5. Most correlations fail to consider macrophyte growth and attached algae (periphyton), which, though not dominant in Kansas reservoirs, can be extremely significant in some reservoirs.

A major dilemma faced by model users is the choice between a simple model, one that may be less accurate due to oversimplification of important processes, and a more complex model that will require a greater amount of input data but may provide more accurate and more comprehensive results. For example, a simple phosphorus loading model may be able to predict, with reasonable accuracy and with relatively little data, the reduction in phosphorus concentration that can be achieved by a particular management strategy. It may even be able to give a good estimate of the resulting average annual chlorophyll concentration. However, the manager may really want to know the likelihood that algae blooms will be eliminated, a question which the model is incapable of answering quantitatively. A more complex model that incorporates physical and biological phenomena (e.g., DiToro, 1976; Lum et al., 1981) could perhaps provide such an answer, but would require more extensive data gathering to adequately define all the constants and coefficients needed by model. In general, the more complex the model,
the more time and money will be required to obtain the data necessary to properly use it. Therefore, decision makers should use the simplest model that will give reliable answers to the necessary and specific questions being asked.

A related problem is that of model calibration or verification. Unless a model can be shown to accurately describe the reservoir in question, or one very much the same, the decision maker cannot (and should not) place much confidence in the model predictions. Such confidence is gained by 1) calibrating the model with data from the reservoir in question and testing its ability to simulate historical data trends; or 2) verifying the model by making predictions and collecting data to test their accuracy. In either case, some data collection is necessary unless adequate calibration or verification has already been done. "Inadequate model confirmation increases the risks associated with the application of the model" (Chapra & Reckhow, 1983).

Due to the many difficulties that can arise in using mathematical models, some decision makers choose to ignore them, believing that models cannot adequately describe the complex processes occurring in a reservoir or watershed. Instead, they prefer to rely on their own knowledge of the situation and perhaps their intuition. What such individuals may fail to realize, as Reckhow and Chapra (1983) point out, is that they are in fact using conceptual "black-box" models tucked away in their minds. Mathematical models have an advantage in that the assumptions made are generally explicit and open to scrutiny and modification.

B. ASSESSMENT

The majority of water supply reservoirs in Kansas are small and turbid, and are not typical of those that have been more extensively studied. Also, limited resources are available for data gathering, research, and implementation of corrective measures. Under these circumstances complex models are, in their present state of development, inappropriate, as are cross-sectional models developed with data from natural lakes or from reservoirs located elsewhere. The most appropriate mathematical models for management of small reservoirs (and even large reservoirs) in Kansas are the simpler ones, such as phosphorus input-output models, that enable a decision maker to assess, with a minimum expenditure of time and money, whether a given strategy is likely to produce a significant impact (e.g., whether reducing a particular source of phosphorus will significantly reduce the total phosphorus loading to a reservoir).

Any model, whether simple or complex, should be used with caution (Smith & Shapiro, 1981), an adequate data base, verification or calibration, and sound judgement. An empirical cross-sectional trophic-state model, calibrated for Kansas reservoirs, could be a useful tool for managing the state's reservoirs, especially the smaller ones, since development and calibration of a complex model for each reservoir is not presently feasible. Perhaps the investigation currently underway by KDHE and USGS will provide a foundation for development of an appropriate cross-sectional model, which
will ultimately need to include a description of the relationship between turbidity, phosphorus, and primary productivity (e.g., Brown, 1984). In any event, it must be recognized that knowledge of chemical processes, ecosystem functioning, meteorological phenomena, etc., is very limited, and that many model predictions, even those of mathematically sophisticated models, actually represent little more than an educated guess.
CHAPTER 18

SUMMARY AND CONCLUSIONS

Reservoirs in Kansas, as a consequence of their geological origin, morphometry, climate, and watershed characteristics, are generally quite fertile; and their fertility enables them to support substantial fish populations, which can be managed to improve sport fishing. Unfortunately, their fertility also encourages the growth of algae and macrophytes, which directly and indirectly cause deterioration of drinking water quality, increase the cost and difficulty of water treatment, and, if excessive, adversely affect sport fishing. Furthermore, organic matter derived from algae and macrophytes tends to clarify the water, increasing primary productivity and accelerating sediment accretion, thereby creating a positive feedback loop that can cause rapid deterioration of water quality and sport fishing. Furthermore, an increase in the organic carbon content of the sediments can result in poor sediment compaction, further accelerating the cycle and reducing the life span of the reservoir. Therefore, water supply reservoirs must be managed to limit their fertility and to minimize the effects of fertility on water quality and treatment, on sport fishing, and on reservoir life span. Although protection of drinking water quality and improvement of sport fishing are often conflicting management objectives with regard to reservoir fertility, they are compatible in many other respects; and managers of multipurpose reservoirs should be encouraged to adopt management strategies that will achieve both objectives.

There are numerous techniques that can be employed in managing a reservoir, but for any given reservoir in Kansas, only a few of these are likely to be appropriate and cost effective. Also, caution and sound judgement must be exercised in assessing the potential effects of management techniques on multipurpose reservoirs in Kansas, since they are significantly different than the more extensively studied lakes and reservoirs in other regions of the country, especially with respect to turbidity, morphometry, and ecological stability. A variety of management practices are listed in Table 5, along with a brief summary of their anticipated effects on water quality, water treatment, and sport fishing in a typical Kansas reservoir. Also included are some comments regarding such considerations as effectiveness, cost, frequency of use, and state of development, all with particular regard to Kansas.

There are a number of conclusions that can be drawn from this investigation:

1. The fundamental area of conflict in regard to drinking water quality and sport fishing is fertility. Water supply managers and fisheries managers have fundamentally different objectives in regard to the desired level of fertility and in regard to certain aspects of management of the consequences of fertility. Maximum sport-fish production is associated with high fertility (but not too high) and low turbidity (to improve sight feeding), while the best drinking water quality is associated with low fertility and, in the case of a fertile reservoir,
<table>
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<tr>
<th>PRACTICE</th>
<th>EFFECTS</th>
<th>COMMENTS</th>
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<tr>
<td>Addition of Copper Sulfate or Chelated Copper Compounds</td>
<td>Direct</td>
<td>Widely used in Kansas</td>
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<td></td>
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<td>Only temporarily effective, but good for emergencies</td>
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<td>Indirect</td>
<td>Best used as a preventive measure</td>
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<td>Other techniques may be more effective or more economical for long-term control</td>
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<td>Can harm fish if improperly applied</td>
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<td>More appropriate for smaller reservoirs</td>
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<td>Accelerated internal cycling of phosphorus</td>
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<td></td>
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<td>Possible damage to non-target organisms</td>
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<td>Addition of Chlorine and Bromine Compounds</td>
<td>Direct</td>
<td>Rarely used, but handy in an emergency</td>
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<td>Highly toxic to fish</td>
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<td>Produces halogenated by-products</td>
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<td>Relatively inexpensive</td>
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<td>Provides only short-term control</td>
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<td></td>
<td>Indirect</td>
<td>Oxidation of iron, manganese, sulfide, and other substances</td>
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<td>May directly kill algae, but unlikely at dosages typically used</td>
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<td>May remove essential trace nutrients</td>
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<td>May indirectly control algae</td>
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<td>May reduce oxygen depletion</td>
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<td>May improve water quality and clarity</td>
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<td>Inadequate evidence of effectiveness</td>
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<td>Additional research needed</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Appears harmless to fish</td>
</tr>
<tr>
<td>PRACTICE</td>
<td>EFFECTS</td>
<td>COMMENTS</td>
</tr>
<tr>
<td>-------------------------------</td>
<td>--------------------------------------------------</td>
<td>--------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Use of Organic Herbicides</td>
<td>Direct</td>
<td>Effective, but only temporarily</td>
</tr>
<tr>
<td></td>
<td>Destruction of algae and macrophytes</td>
<td>Commonly used for macrophyte control, but harvesting may be a better choice</td>
</tr>
<tr>
<td></td>
<td>Possible damage to non-target organisms</td>
<td>Provides only short-term control</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td>Only 2,4-D and endothall are currently recommended for use in Kansas</td>
</tr>
<tr>
<td></td>
<td>Oxygen depletion and nutrient release</td>
<td>Seldom used in water supply reservoirs in Kansas at the present time</td>
</tr>
<tr>
<td></td>
<td>Stimulation of algal blooms</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Food chain disruption</td>
<td></td>
</tr>
<tr>
<td></td>
<td>May harm non-target organisms</td>
<td></td>
</tr>
<tr>
<td>Nutrient Inactivation</td>
<td>Direct</td>
<td>Not expected to be effective for most Kansas reservoirs</td>
</tr>
<tr>
<td></td>
<td>Removes phosphorus from the water column</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Retards phosphorus release from sediments</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Chemicals used can harm aquatic organisms</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Reduced growth of algae and macrophytes, assuming that the depth of the photic zone is not substantially increased</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Reduced fish production, if fish production is nutrient limited</td>
<td></td>
</tr>
<tr>
<td>Light Blockage</td>
<td>Direct</td>
<td>Occurs naturally in most Kansas reservoirs</td>
</tr>
<tr>
<td></td>
<td>Reduced photosynthesis</td>
<td>Still experimental; more research needed</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td>Inappropriate for multipurpose reservoirs</td>
</tr>
<tr>
<td></td>
<td>Reduced amounts of algae and macrophytes</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Interference with sight feeding</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Interference with other water uses</td>
<td></td>
</tr>
<tr>
<td>Mechanical Harvesting</td>
<td>Direct</td>
<td>An excellent alternative to herbicide use</td>
</tr>
<tr>
<td></td>
<td>Removal of macrophytes and associated nutrients and organic matter</td>
<td>Labor intensive and expensive</td>
</tr>
<tr>
<td></td>
<td>Clearing of navigation channels, etc.</td>
<td>More appropriate for smaller reservoirs</td>
</tr>
<tr>
<td></td>
<td>Reduction of available fish habitat</td>
<td>Provides only short-term control</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Seldom done in water supply reservoirs in Kansas at the present time</td>
</tr>
<tr>
<td>PRACTICE</td>
<td>EFFECTS</td>
<td>COMMENTS</td>
</tr>
<tr>
<td>--------------------------</td>
<td>-------------------------------------------------------------------------</td>
<td>-----------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Mechanical Harvesting</td>
<td>Indirect</td>
<td>Generally inappropriate for multipurpose reservoirs</td>
</tr>
<tr>
<td>(cont'd)</td>
<td>Improved drinking water quality</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Increased turbidity</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Increased or decreased fish production</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Invasion of other plant species</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Reduced internal nutrient cycling</td>
<td></td>
</tr>
<tr>
<td>Sediment Covering</td>
<td>Direct</td>
<td>Not likely to be effective in Kansas, due to rapid siltation</td>
</tr>
<tr>
<td></td>
<td>Reduced macrophyte growth</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Retardation of internal nutrient cycling</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Reduction of available fish habitat</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Likely to improve water quality</td>
<td></td>
</tr>
<tr>
<td></td>
<td>May decrease fish production</td>
<td></td>
</tr>
<tr>
<td></td>
<td>May increase turbidity</td>
<td></td>
</tr>
<tr>
<td>Biological Control</td>
<td>Direct</td>
<td>Still in developmental stages</td>
</tr>
<tr>
<td></td>
<td>Artificially increased populations of certain organisms</td>
<td>More research needed</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td>May provide only short-term control</td>
</tr>
<tr>
<td></td>
<td>Population shifts to more desirable species</td>
<td>Promising means of weed control for the future</td>
</tr>
<tr>
<td></td>
<td>May improve drinking water quality by controlling algal blooms</td>
<td></td>
</tr>
<tr>
<td></td>
<td>May influence fish production</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Undesirable species may invade</td>
<td></td>
</tr>
<tr>
<td>Fish Manipulation</td>
<td>Direct</td>
<td>Stocking of sport fish very common in Kansas</td>
</tr>
<tr>
<td></td>
<td>Increased size or number of sport fish</td>
<td>Not generally used to improve water quality, but potential exists</td>
</tr>
<tr>
<td></td>
<td>Fewer rough fish</td>
<td>Requires long-term maintenance due to the ecological instability of Kansas reservoirs</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td></td>
</tr>
<tr>
<td></td>
<td>May improve water quality through indirect control of algal blooms</td>
<td></td>
</tr>
<tr>
<td>PRACTICE</td>
<td>EFFECTS</td>
<td>COMMENTS</td>
</tr>
<tr>
<td>-----------------------</td>
<td>-------------------------------------------------</td>
<td>--------------------------------------------------------------------------</td>
</tr>
</tbody>
</table>
| Artificial Fertilization | Direct  
  Increased fertility  
  Indirect  
  Increased fish production, if nutrients are a limiting factor  
  Deterioration of drinking water quality | Not advisable or economical in most Kansas reservoirs  
 Undesirable in water supply reservoirs  
 A possible solution to the problem of blue-green algae, but risky; field demonstrations lacking |
| Level Adjustment      | Direct  
  Control of certain species of nuisance vegetation  
  Improved sport fish habitat and predation  
  Indirect  
  Decreased turbidity after reflooding  
  May influence the concentrations of nutrients, TOC, dissolved oxygen, etc., but changes difficult to predict | Commonly used, especially in large reservoirs  
 Beneficial effects on fish well established  
 Low cost, if accommodated during design  
 Facilitates shoreline improvements  
 Can improve waterfowl habitat  
 Reservoir could fail to refill during drought |
| Hypolimnetic Withdrawal | Direct  
  Export of nutrients and oxygen demand  
  Decreased thermal stability and increased mixing  
  Increased water temperature  
  Indirect  
  Reduced hypolimnetic oxygen depletion  
  May increase water clarity  
  Changes in phytoplankton species and productivity | Impacts on sport fish and water quality mixed and difficult to predict  
 Other constraints may preclude or require hypolimnetic withdrawal |
| Dilution or Flushing  | Direct  
  Decreased nutrient availability  
  Indirect  
  Reduced primary productivity  
  Increased clarity and dissolved oxygen | Generally infeasible in Kansas  
 Coat depends on cost and availability of nutrient-poor water  
 Conceivably beneficial for both sport fishing and water treatment  
 Primarily applicable to small reservoirs |
<table>
<thead>
<tr>
<th>PRACTICE</th>
<th>EFFECTS</th>
<th>COMMENTS</th>
</tr>
</thead>
</table>
| Destratification         | Direct
Increased mixing
Oxygenation of the hypolimnion
Increased water temperature
Indirect
Greatly improved hypolimnentic water quality
Expansion of available fish habitat
Changes in algal productivity and dominance that are generally beneficial and may be predictable
Reduced summerkill and winterkill
May increase turbidity   | Often effective and expected to improve both sport fishing and water quality
Successfully used by KFGC
Seldom used in water supply reservoirs, but perhaps should be used more often
Primarily applicable to small reservoirs
Somewhat expensive, but perhaps worthwhile
More research needed to examine usefulness of this practice in Kansas |
| Hypolimnetic Aeration    | Direct
Increased hypolimnetic dissolved oxygen
Indirect
Greatly improved hypolimnetic water quality
Expansion of available fish habitat
Reduced summerkill and winterkill
Reduced internal nutrient cycling | Seldom used in Kansas, but has potential
Somewhat expensive, but perhaps worthwhile
Primarily applicable to small reservoirs
Ecological effects largely unknown
More research needed to examine applicability to Kansas |
| Dredging; Draining & Cleaning | Direct
Removal of nutrients and organic matter
Reservoir deepening, increased capacity
Opening of channels for boaters
Indirect
Reduced primary productivity
Improved water quality, sport fishing, and recreation | Very expensive and often disruptive
Waste disposal can be a problem
Rarely used now, but use should increase in the future as reservoirs become silted in
Provides long-term benefits
Adverse effects are minimal and short-term |
<table>
<thead>
<tr>
<th>PRACTICE</th>
<th>EFFECTS</th>
<th>COMMENTS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Good Design and Planning</td>
<td>Direct</td>
<td>May increase initial cost</td>
</tr>
<tr>
<td></td>
<td>Maximum depth without stratification</td>
<td>Will reduce O&amp;M costs, perhaps greatly</td>
</tr>
<tr>
<td></td>
<td>Optimal size and orientation</td>
<td>Required by KDHE for new reservoirs</td>
</tr>
<tr>
<td></td>
<td>Control of shoreline erosion</td>
<td>Especially important for small reservoirs</td>
</tr>
<tr>
<td></td>
<td>Protection of fish, wildlife, water quality, and aesthetics</td>
<td>May govern water quality in large reservoirs</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Improved water quality and sport fishing</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Extended reservoir life</td>
<td></td>
</tr>
<tr>
<td>Leaving Brush, Timber, and Shallow Areas;</td>
<td>Direct</td>
<td>May reduce construction costs</td>
</tr>
<tr>
<td>Constructing Fish Attractors</td>
<td>Improved fish habitat</td>
<td>May increase O&amp;M costs and water treatment costs</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td>Commonly practiced by KPCC</td>
</tr>
<tr>
<td>Watershed Management</td>
<td>Increased fertility</td>
<td>Should be done with caution in small water supply reservoirs</td>
</tr>
<tr>
<td></td>
<td>Increased fish production</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Increased fishing success</td>
<td></td>
</tr>
<tr>
<td></td>
<td>May impair drinking water quality</td>
<td>May interfere with boating, dredging, or chemical treatment</td>
</tr>
<tr>
<td></td>
<td>May reduce reservoir lifespan</td>
<td></td>
</tr>
<tr>
<td>Use Restrictions</td>
<td>Direct</td>
<td>Control of non-point nutrient sources is generally impractical as a mean of</td>
</tr>
<tr>
<td></td>
<td>Reduced loadings of sediment, nutrients, and organic matter</td>
<td>limiting reservoir fertility in Kansas</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td>Many watershed management practices have already been implemented in Kansas</td>
</tr>
<tr>
<td></td>
<td>Increased reservoir life span</td>
<td>Benefits to watershed residents can be substantial</td>
</tr>
<tr>
<td></td>
<td>Improved water quality</td>
<td>Decreased turbidity may increase primary productivity</td>
</tr>
<tr>
<td></td>
<td>May reduce trihalomethane formation</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Can influence turbidity</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Improved fishing where weed growth is excessive and nutrient limited</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Direct</td>
<td>Must be balanced against user needs</td>
</tr>
<tr>
<td></td>
<td>Reduced loadings of nutrients, sediment, and other pollutants</td>
<td>More important for small reservoirs</td>
</tr>
<tr>
<td></td>
<td>Indirect</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Improved drinking water quality</td>
<td></td>
</tr>
<tr>
<td></td>
<td>May improve sport fishing</td>
<td></td>
</tr>
</tbody>
</table>
2. For given levels of fertility and turbidity (which may be largely governed by factors outside the control of a reservoir manager), there are many things that can be done by managers to improve both drinking water quality and sport fishing. Potentially useful techniques include control of excessive plant growth, sport fish stocking, reduction or elimination of certain rough fish, level adjustment, selective withdrawal, destratification, hypolimnetic aeration, dredging, encouragement of good design practices, and implementation of appropriate watershed management strategies.

3. Techniques that are generally ineffective, uneconomical, or insufficiently developed for multipurpose reservoirs in Kansas include biological control of algae and macrophytes, fertilization, flushing/dilution, nutrient inactivation, draining and cleaning, and control of non-point sources of nutrients.

4. Kansas reservoirs can not be expected to be filled with clear blue water and cold-water trout. They must be managed in the context of what is reasonably achievable under local conditions.

5. Small reservoirs are much more likely to experience problems with drinking water quality and fish kills than are large reservoirs. Therefore, very careful attention should be paid to the design and management of small reservoirs.

6. There is a tendency for reservoir managers to depend rather heavily on the use of herbicides to control algae and macrophytes. Herbicides can provide only short-term control at best and can have deleterious side effects on both water quality and sport fishing. Mechanical harvesting should be given serious consideration as an alternative to the use of herbicides for macrophyte control, and reservoir managers should seek effective and economical long-term solutions to the problem of nuisance vegetation.

7. Most reservoir management problems are better addressed by "preventive medicine" and ecologically sound long-term programs than by "quick-fix" solutions implemented during a crisis.

8. There is a general lack of good information regarding the secondary (indirect) and long-term effects of many reservoir management practices. However, to the extent that such information is available, it should be carefully considered by reservoir managers seeking long-term solutions.

9. Very little information is available regarding the influence of reservoir management practices on TOC or on THM precursors. However, in turbid rapidly flushed reservoirs, such as those found in Kansas, much of the organic matter is likely to be allochthonous. Hence, concentrations of TOC and THM precursors are likely to be more strongly influenced by conditions in the watershed than by in-reservoir management practices. Nevertheless, excessive growths of algae and weeds can
be expected to substantially increase the concentrations of TOC and THM precursors in a reservoir, particularly a small reservoir.

10. Existing trophic-state models and correlations between phosphorus and chlorophyll-a found in the literature are expected to be inappropriate for Kansas reservoirs. An empirical cross-sectional model, calibrated for Kansas and describing the relationships between turbidity, phosphorus, and chlorophyll-a could be a useful management tool, especially for smaller reservoirs.
CHAPTER 19
RECOMMENDATIONS FOR FUTURE RESEARCH

There are clearly a great number of unanswered questions regarding the impacts of reservoir management practices on drinking water quality and sport fishing. A general lack of knowledge concerning the secondary and long-term effects of various management practices makes it difficult for managers to develop optimal long-term strategies and encourages them to operate in a "crisis-mode". Described below are some specific areas where well conceived, carefully controlled, well documented, and properly executed research investigations might provide information of considerable value to managers of Kansas reservoirs.

1. Control of Algae by Food Chain Manipulation

There is good evidence that algal populations can be held in check by manipulating the food chain in such a way as to increase the number of large herbivorous zooplankton. This might be accomplished by selective removal of small planktivorous fish or by stocking of large predator (piscivorous) fish (see Chapter 5). It is not known whether this approach will work in a typical Kansas reservoir or what its short-term and long-term side effects might be. A controlled field study should be conducted in a Kansas reservoir and the biological and chemical effects of such a manipulation should be carefully followed for an extended period of time.

2. Mechanical Harvesting

An inventory should be made of the species of macrophytes that are (or are likely to become) a nuisance in Kansas reservoirs, and the ability of mechanical harvesting to control these species should then be assessed. Using a small to medium sized weed-infested reservoir to accommodate a field study, the long-term costs of mechanical harvesting should be compared to the long-term costs of herbicide addition. Macrophytes are currently not so great a problem in water-supply reservoirs in Kansas that they are frequently treated or harvested, but they are likely to become a more serious problem in the future as reservoirs fill in. Also, they might be more frequently controlled if there were more reliable and effective control methods available. Research on this topic should be conducted before the problem becomes more severe, so that the research results will be available when needed.

3. Artificial Destratification and Hypolimnetic Aeration

Both of these techniques could potentially result in a significant improvement in drinking water quality in small stratified reservoirs. A demonstration project should be conducted at a suitable multipurpose Kansas reservoir and the effects on fish and water quality should be thoroughly documented.
4. Design Guidelines

The quality of water in existing reservoirs should be carefully evaluated with respect to the design criteria used for construction and with respect to the current condition of the reservoirs. An effort should be made to determine what improvements could be made in state design criteria and which aspects of reservoir design have the greatest influence on water quality.

5. TOC and THM Precursors

There is a need for better information regarding the sources of TOC and THM precursors in Kansas reservoirs, including the factors influencing their concentrations in water supply reservoirs and the effects that various management practices might have on their concentrations.

6. Reservoir Siltation

Samples of sediment from a cross section of Kansas reservoirs should be analyzed for organic carbon and moisture content to determine the extent to which organic matter might be contributing to increased sediment accretion rates and decreased reservoir life.

7. Modeling

A data base suitable for development of a cross-sectional trophic-state model applicable to Kansas would be quite helpful, particularly in managing small reservoirs. (On-going research by KDHE and USGS may provide such a data base.)

An ecosystem model similar to that of Hutchins (1982) should be developed and tested for Kansas reservoirs. Such a model, if successfully developed and demonstrated to be reliable, would greatly facilitate evaluation of the potential impacts of management practices on sport fishing in Kansas.
Readers seeking additional information may find some of the following sources helpful:


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