MEASURES FOR THE RESTORATION AND ENHANCEMENT OF QUALITY OF FRESHWATER LAKES

UNITED STATES
ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460
1973

MEASURES FOR THE RESTORATION AND ENHANCEMENT OF QUALITY OF FRESHWATER LAKES

by the

Office of Air and Water Programs
Division of Water Quality and Non-Point Source Control

and the

Office of Research and Development
National Eutrophication Research Program

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
Washington, D.C. 20460
EPA-430/9-73-005

For sale by the Superintendent of Documents, U.S. Government Printing Office, Washington, D.C. 20402 - Price $2.85

FOREWORD

The limited number of publicly owned high quality freshwater lakes in the United States combined with a growing population has resulted in a pressing need for sound management programs designed to protect and enhance the quality of the Nation's lakes.

The Federal Water Pollution Control Act Amendments of 1972 require the Administrator of the Environmental Protection Agency to issue information on methods, procedures and processes as may be appropriate to restore and enhance the quality of the Nation's publicly owned freshwater lakes [Subsection 304(i), PL 92-500]. This report is prepared pursuant to that legislative mandate.

Robert W. Fri
Acting Administrator
Environmental Protection Agency

LIST OF PARTICIPANTS

WORKING GROUP MEMBER CONTRIBUTORS

Office of Air and Water Programs

Division of Water Quality and Non-Point Source Control

Mr. W. L. Kinney, Subcommittee Chairman

Mr. J. I. Lewis, Alternate Subcommittee Chairman

Dr. L. J. Guarraia

Mr. J. P. Cating

Mr. D. K. Boynton, Jr.

Office of Research and Development

Division of Processes and Effects

Dr. F. G. Wilkes

Office of Radiation Programs

Criteria and Standards Division

Mr. R. S. Dyer

NON-WORKING GROUP MEMBER CONTRIBUTORS

National Eutrophication Research Program

Mr. T. E. Maloney

Dr. K. W. Malueg

Mr. D. W. Shults

Dr. C. F. Powers

Dr. S. A. Peterson

Dr. W. D. Sanville

Dr. F. S. Stay, Jr.

WORKING GROUP MEMBER REVIEWERS

Office of Air and Water Programs

Mr. K. M. Mackenthun
Mr. B. W. Everling

Office of Research and Development

Dr. D. Yount
Mr. D. Ehreth

Office of Hazardous Materials Control Programs

Mr. V. Grey
Mr. E. Brooks

Office of Enforcement and General Counsel

Mr. A. W. Eckert

Office of Federal Activities

Mr. P. Smith

Office of Planning and Management

Mr. J. Jacknow

Office of International Affairs

Mr. J. Tarran

NON-WORKING GROUP MEMBER REVIEWERS

Dr. D. Duttweiler
Southeast Environmental Research Laboratory
Athens, Georgia

Dr. L. P. Seyb
Pacific Northwest Environmental Research Laboratory
Corvallis, Oregon

Mr. F. H. Rainwater
National Thermal Research Program
Corvallis, Oregon

Mr. R. M. Brice
Shagawa Lake Research Project
Ely, Minnesota

Mr. H. J. Fisher (ret.)
Region V
Chicago, Illinois

ACKNOWLEDGEMENT

The technical and editorial assistance of Mrs. Marian Musser is gratefully acknowledged.

CONTENTS

List of Tables

APPENDIX

List of Figures

Section I

SUMMARY

The increasing rate of deterioration of the Nation's public waters has resulted in passage of the Federal Water Pollution Control Act Amendments of 1972, PL 92-500. Included within this legislation is the requirement that the Administrator of the United States Environmental Protection Agency issue such information on methods, processes and procedures as may be appropriate to restore and enhance the quality of the Nation's publicly owned lakes [Subsection 304(i)]. This report is prepared pursuant to that legislative mandate. It contains state-of-the-art information only and the methods have not been subjected to cost analyses.

Lakes vary tremendously in their chemical, physical and biological characteristics depending upon their mode of origin, their location, the characteristics of their watersheds and their uses. Consequently, lake problems also vary, and most must be dealt with on a case-by-case basis.

Contaminants may impact upon lake environments in various ways depending upon the nature of the substance. Nutrient rich plant growth stimulators such as domestic sewage and commercial fertilizers cause accelerated eutrophication: sedimentation may add to the eutrophication

problems or create unique problems in the absence of eutrophication: toxic substances may poison water supplies, interfere with normal biological activity or render commercial and sports fish and crustaceous species unfit for consumption. Heated water released to lakes may alter the natural thermal structure and upset the composition of lake communities.

Lake restoration measures are not well developed, with much of the technology still in experimental stages in laboratories or in small pilot lakes. Certain techniques have met with varying degrees of success on individual lakes, but their applicability to other lakes is unknown. At this point in time it is impossible to recommend remedial measures which will prove effective for all lakes or even particular classes of lakes. It is the responsibility of lake managers to define the problems and to implement rehabilitation or enhancement programs which are best fitted to the requirements of particular lakes on a case-by-case basis.

The approach to the rehabilitation of degraded lakes is twofold: (1) restricting the input of undesirable materials and (2) providing in-lake treatment for the removal or inactivation of undesirable materials. Reducing or eliminating the sources of waste loading is the only restorative measure needed to achieve the desired level of improvement in certain lakes in which natural flushing results in substantial improvements in quality. However, in many lakes, particularly those with slow flushing rates, in-lake treatment schemes may also be required before significant improvements will be realized.

Remedial measures which restrict the input of contaminants include advanced wastewater treatment, nutrient diversion and allochthonous sediment control.

Advanced wastewater treatment (AWT) probably represents the best method currently available for curbing nitrogen and phosphorus input to waterways at moderate costs. Phosphorus removal efficiency of 80-95 percent can be achieved by chemical precipitation with alum, lime or ferric salts. Removal of ammonia and other nitrogen species can be accomplished by ion exchange, ammonia stripping, breakpoint chlorination or bacterial denitrification. Although to date there has not been documentation evaluating AWT as a means of restoring a lake, preliminary results both in this country and in Europe have been encouraging.

Nutrient diversion offers a possible restoration technique in situations where the incoming nutrient load is entering from point sources. This technique has been used successfully in Lake Washington and has resulted in some improvement in the Madison Lakes. Preliminary studies on several lakes indicate that the effects of diversion may not be readily apparent in small, shallow, highly eutrophic lakes, due to the remobilization of nutrients from the sediment pool and the continued influx of nutrients from non-point sources.

The useful existence of a lake or reservoir can sometimes be prolonged by implementing control measures to reduce the rate of sedimentation. Prudential land use management practices within the watershed which minimize erosion associated with construction, farming, road building and forestry activities tend to reduce the volume of sediment input to lakes. Filter dams and desilting basins are effective sediment traps under certain conditions. Sediment control measures not only reduce the rate at which a lake basin is filled, but also restrict the input of nutrients adsorbed to sediment particles.

In-lake treatment measures which have been used in lake restoration programs or which are now being investigated include dredging, nutrient inactivation, dilution and displacement, covering of sediments, artificial destratification and hypolimnetic aeration and drawdown.

Lake dredging not only removes sediment buildup, but also serves to remove a potential nutrient source. Little information is available on the chemical and biological effects of dredging, but projects are now under way which will evaluate the total environmental effects. The relatively high costs of dredging make this technique prohibitively expensive on most large lakes, but dredging as a restorative method has been used successfully for years on small lakes and ponds.

Nutrient inactivation in lakes is accomplished by adding some type of material to the water that will bond with, adsorb or otherwise render nutrients unavailable to aquatic plants. Alum, sodium aluminate, fly ash and various other materials have been investigated as nutrient inactivation agents. Although some pilot lake results with this technique have been encouraging, its applicability on a large scale has not been determined.

Under certain conditions the water quality of lakes can be improved by diluting or replacing the existing lake water with water of a higher quality. This technique has been used successfully in Green Lake, Washington and a few others. Its applicability is limited to lakes with ready access to a large supply of high quality water.

Covering of bottom sediments with sheeting materials or particulate matter is being investigated as a means of preventing nutrient exchange and retarding rooted plant growth. Limited experiences with this technique have encountered problems with ballooning of sheeting and rupturing seals of particulate matter when gas is produced within the sediments. Investigations of this technique in pilot lakes are continuing.

It is sometimes possible to replenish the oxygen supply of anaerobic bottom waters of eutrophic lakes by disrupting the thermal stratification or by aerating the hypolimnion directly without disturbing the thermal regimen. Definite improvements in water quality and in the biota have occurred as a result of artificial destratification and hypolimnetic aeration. Although the response of a given lake to these treatment measures is unpredictable, destratification and hypolimnetic aeration are potential mechanisms for improving the water quality of certain lakes.

Lake drawdown has been investigated as a control measure for rooted aquatic vegetation, as a means of retarding nutrient release from the sediments and as a lake deepening mechanism through sediment consolidation. Drawdown has shown promise as a successful remedial method in Florida, but results in Wisconsin are inconclusive. Lake drawdown studies are continuing.

In many lakes in advanced stages of eutrophication attempts have been made to control nuisance organisms through mechanical, biological and chemical means. Mechanical harvesting can be an effective technique for removing excess aquatic plants, but it generally is not economically feasible on a self supporting basis due to the limited market for the product. Biological control agents for algae and macrophytes range from the viruses to the manatee. Although certain organisms have proved to be useful control agents, much work with biological control, particularly with the viruses, needs to be undertaken before it will have universal application. Various chemicals have long been utilized to control or eliminate undesired aquatic flora and fauna. Chemical agents, however, offer only temporary, symptom suppressing relief, and often the treatment must be repeated to achieve the desired results.

Contamination of lakes with various hazardous substances is an ever present threat. In order to avoid major catastrophies resulting from spills, industrial accidents etc., measures for the control and removal of hazardous materials must be implemented.

Decontamination of lakes polluted with toxic substances has been accomplished by filtering the lake water through activated charcoal filters. Several means of removing mercury from waters and sediments have been proposed and used in the laboratory, but few have been demonstrated in field situations.

Several state and local governments have established statutes dealing with various aspects of lake management and rehabilitation as a means of protecting inland lake environments, but explicit statutes authorizing specific state or local programs are often badly fragmented among state agencies and local units of government.

Section II

INTRODUCTION

LEGISLATIVE AUTHORITY

An ever increasing rate of deterioration in the quality of the Nations's waterways combined with increased public need for clean water, has resulted in a public awareness of the Nation's water quality problems and a demand that action be taken to alleviate the problems.

The pressing need for sound water quality management programs has resulted in the enactment of the Federal Water Pollution Control Act Amendments of 1972 designed to restore and maintain the chemical, physical and biological integrity of the Nation's waters. Included within this Act is the requirement that "...The Administrator[of the Environmental Protection Agency] shall, within 270 days after the effective date of this subsection (and from time to time thereafter) issue such information on methods, procedures and processes as may be appropriate to restore and enhance the quality of the Nation's publicly owned fresh water lakes" - Subsection 304(i), PL 92-500.

This report, prepared pursuant to subsection 304(i), PL 92-500, provides background information on lake environments followed by state-of-the-art information on remedial measures for enhancing and restoring the quality of lakes, ponds and reservoirs as required by the legislation. Discussion of major lake problems is included in an appendix. Since most lake restoration techniques are presently in experimental stages, it is impossible to provide a thorough evaluation and complete cost-effectiveness analysis at this time. However, as the experimental programs now underway are evaluated and as new technology becomes available, subsequent reports documenting the latest technological and scientific achievements relating to lake restoration will be forthcoming.

SCOPE OF THE PROBLEM

The limited number of publicly owned fresh water lakes in the United States combined with increasing population and industrial pressures are major factors contributing to their unique and widespread water quality problems. Discharges of organic and inorganic wastes resulting from urbanization, cultural and technological advancement, and new water dependent industries have caused noticeable degradation of lake environments in many areas. The problem, in National perspective, presents a complex interrelationship of urban development, industrial growth, potable water supply demands, recreational needs and maintenance of virgin area resources.

Aesthetic and environmental considerations aside, the demand for clean lakes for private, public, and commercial use is of vital economic concern. Design of a successful water management program depends upon an understanding of the impact of man's activities upon fresh water environments and the means of ameliorating harmful processes.

Effects of waste discharges on the quality of the aquatic environment may be manifested as subtle long term changes in the fauna and flora or dramatic and seemingly immediate as in the sudden appearance of algal blooms, aquatic weeds or dead fish. Along with the alterations of the species composition of the animal or plant life, shifts occur in population densities with the ascendence of large populations of often undesirable species. Sports fish are replaced by "trash" fish, clean water associated benthic organisms are replaced by sludge worms and other pollution tolerant forms, and the normal phytoplankton crops are replaced by large populations of scum forming blue-green algae. In addition, human health becomes threatened due to the establishment of pathogenic microorganisms associated with fecal and other waste discharge.

Reduction of water related activities follows alteration of aquatic life. Boating, swimming, and water skiing activities must be halted as lakes become choked with aquatic weeds and as surface algal scums develop. Economic losses result from a decline of commercially important aquatic species and with the curtailment of water related recreational activities.

Industrial and municipal water supplies are also affected by water quality degradation. Industrial raw water often must be treated to the desired quality. If water is uncontaminated, costs of water processing decrease, possibly affecting final consumer cost. Toxic materials and pathogenic microorganisms in municipal raw water supplies can affect health and increase the costs of processing. The taste, color and odor of water often make people reluctant to draw water from contaminated sources. In effect, this limits water supply and increases the costs to the consumer.

Section III

LAKE ENVIRONMENTS

Lakes are temporary features of the landscape, nearly all of which are very young on the geological, but very old on the human, time scale. With the passage of time, all lakes presumably would cease to exist as a consequence of natural physical and biological processes. Under natural conditions these processes would require several hundreds or thousands of years. With the appearance of man on the scene and as a result of his activities, however, these processes have been accelerated dramatically, and the maturation or aging rates of many lakes have been significantly increased.

In the discussion which follows, the limnological aspects of lake environments including chemical, physical and biological phenomena are briefly explored. A general understanding of the lake as an ecosystem is prerequisite to an appreciation of lake problems.

LAKE TYPES

Often lakes are formed by some geological event such as subsidence, faulting, damming of river valleys or by the eroding and damming action of glaciers. Natural lakes are usually formed in infertile basins with low potential for biological productivity. Thus they are generally poor with respect to dissolved nutrients and biological production in their early history, becoming more fertile with time as nutrients are carried in from the drainage basin. Man-made lakes (reservoirs) are frequently created by the inundation of highly fertile river valleys rich in nutrients necessary for biological production. Such reservoirs which have been created in fertile areas will usually exhibit an immediate high degree of biological activity which, if nutrients are not constantly carried in via tributary streams or other runoff, will decline after a few years as nutrients are accumulated in the bottom sediments or otherwise become biologically unavailable. Many reservoirs, however, are created by the confinement of rivers with very high nutrient concentrations which, through contaminated inflow, maintain the fertility and productivity of the impoundment.

In glaciated North America, nutrient-poor melt waters filled ice-formed basins, creating lakes of various sizes, shapes, and depths. Many of these lakes, particularly large, deep ones, have changed relatively little since their formation and still retain their nutrient-poor characteristics. Such lakes, low in dissolved nutrient content and biological production are of the type classified as "oligotrophic". Oligotrophic lakes are characterized by deep basins with large volumes of deep (hypolimnetic) waters, low organic and nutrient content, high dissolved oxygen concentration at all depths throughout the year, and low biological productivity. Phytoplankton crops are quantitatively restricted, represented by many species of diatoms and green algae. The deep bottom fauna is characteristically sparse and is represented by such forms as fingernail clams, crustaceans, insect larvae and segmented worms. Cold water fishes such as the salmonids and whitefish are typical of oligotrophic lakes.

Many other lakes, usually smaller and shallower, are rich in dissolved nutrients and are highly productive. These are "eutrophic" lakes. In eutrophic lakes organic content of the sediments and the water column is high and nutrients are abundant. Oxygen depletion may occur seasonally in the deeper portions. Diatoms, green, and blue-green algae are the major phytoplankton types, with seasonal shifts in dominance usually apparent. During the summer, blue-green algae blooms may occur regularly, often in nuisance quantities. The benthic organisms of the deeper waters consist of species which are able to survive in the low dissolved oxygen concentrations which occur periodically. Tubificid worms and midge larvae may be very abundant. Fish populations usually consist of warm water species such as perch, pike, bass, panfish, and bullheads. These lakes eventually succeed into ponds, marshes or swamps, and thence to dry land (Fig. 1).

The distinctions between oligotrophic and eutrophic lakes is sometimes not sharply delineated, and the term "mesotrophic" is often used to describe lakes which have characteristics of both. Many of the Nation's better recreational lakes are in a state of mesotrophy, having evolved through their oligotrophic state to the point where they are moderately productive but have not yet developed nuisance conditions.

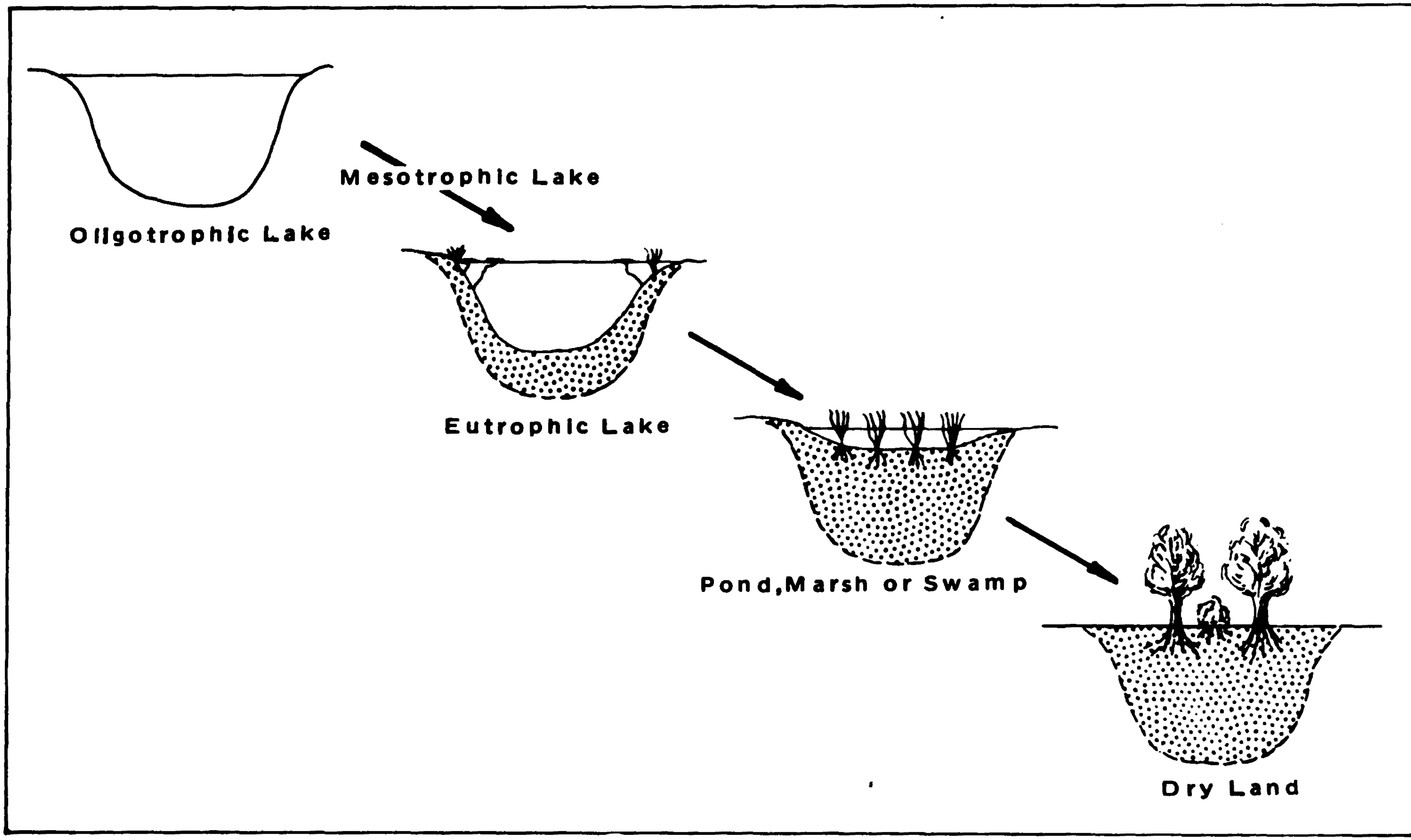


Figure 1 .--Eutrophication - the process of aging by ecological succession.

Source(1)

THERMAL REGIMENS OF LAKES AND RESERVOIRS

The thermal regimens of lakes exert a profound effect upon overall lake ecology, primarily because of the associated phenomenon of thermal stratification.

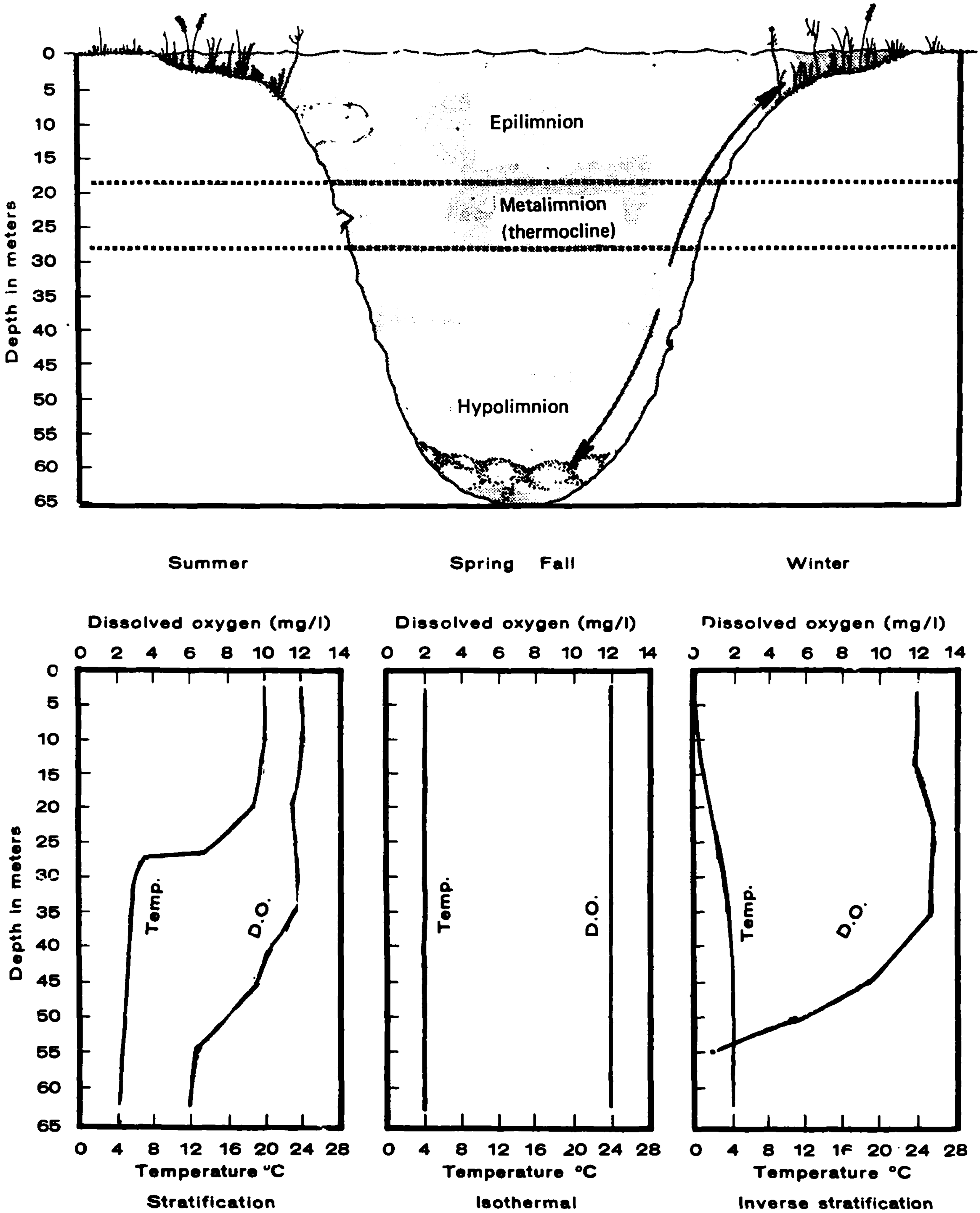
Seasonal changes in air temperature induce changes in water temperature resulting in a cycle of events of mixing and stratification which controls the dispersion of nutrients and dissolved gasses throughout the water column thereby affecting the biological activity in the lake (Fig. 2).

During the winter, surface water under ice cover and frequently open water are very near $0^{o}C$. Since water reaches its maximum density at $4^{o}C$, the warmer, denser waters will occur at the bottom of the lake. This is inverse stratification. With the gradual warming of surface waters in the spring of the year, the lake becomes homothermous throughout at a temperature of $4^{o}C$. Under these conditions, winds generate mixing action which may be complete from top to bottom even in very deep lakes, distributing nutrients, dissolved oxygen and other materials throughout the water. As spring progresses into summer, surface waters continue to warm, and a layer of rapidly decreasing temperature called the "thermocline" or "metalimnion" is formed, acting as a barrier which prevents the warm upper "epilimnetic" waters from mixing with the cool, deeper, heavier "hypolimnetic" waters. The hypolimnetic waters are effectively isolated from the overlying layers and the atmosphere, and if the volume of the hypolimnion is small and the oxygen consumption rate is high, these bottom waters may become depleted of oxygen. This tends to be the case in many eutrophic lakes. This condition will persist until the entire lake once again becomes homothermous in the fall as the surface waters cool. Mixing from top to bottom then occurs, and the bottom waters are reoxygenated. As winter progresses, surface water temperatures again approach $0^{o}C$, and the inverse stratification patterns are again established.

Reservoirs are affected by all of the processes that influence natural lakes, and in addition, are strongly influenced by the hydraulic effects of both the inflow and discharge. Reservoirs with high discharge to volume ratios are often completely mixed during the summer due to the rapid movement of water. Deep reservoirs with a low discharge to volume ratio often exhibit the classical lake stratification cycle. Operation of the reservoir discharge can have a major influence on the thermal structure. The use of multiple outlet structures at various depths can provide pre-selected discharge temperatures when stratification exists, which in turn provides modification of the thermal regimen.

Figure 2

Diagramatic sketch showing thermal characteristics of temperate lakes



NUTRIENT CYCLING

Development of successful water management programs and restoration planning depends upon as complete a knowledge as possible of both the physical and biological processes working within a particular system. The turnover rates and exchange of nutrients with the sediments are in part governed by biological communities.

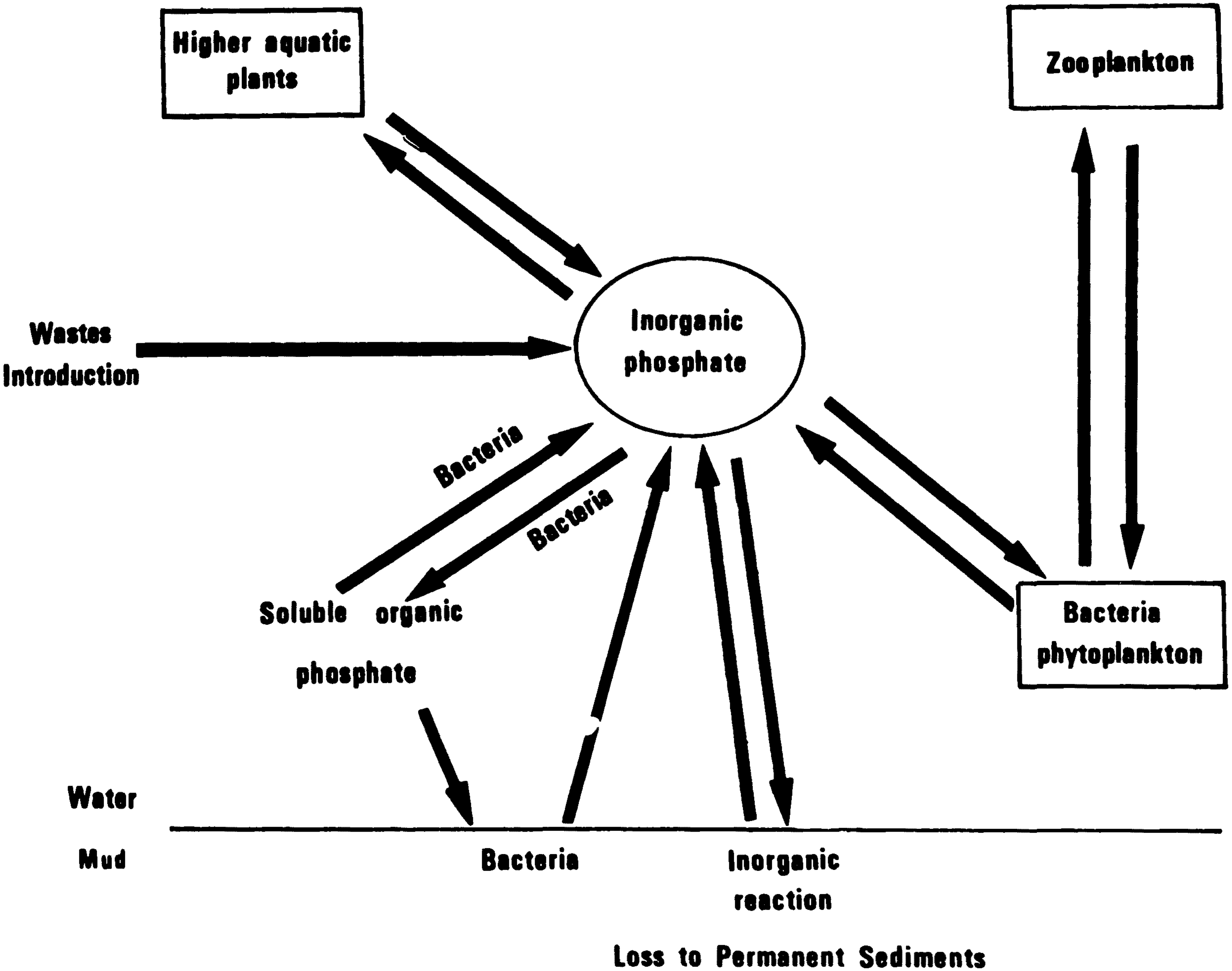
Before proceeding, the term "nutrients" must be defined, because the definition of "nutrient" depends upon the individual involved. "Nutrients" refer to not only organic material, simple and complex, but to trace elements, vitamins, and also the major inorganic elements: phosphorus, sulfur, nitrogen and carbon. For the sake of brevity, only these four major nutrients are discussed.

One nutrient which has received widespread attention is phosphorus. It is known that phosphorus can be limiting to phytoplankton and other organisms. Most of the phosphorus in the aquatic environment is bound in the sediments as an insoluble phosphate salt with availability of insoluble salts being influenced by both the physical-chemical factors (2) and bacterial metabolism (3). As seen in

Fig. 3, loss or precipitation of phosphates to the sediments and solublization of insoluble phosphates from the sediments and exchange among the various biologic communities, is mediated in part by the bacterial community (4 - 10). Three general processes involved in phosphate solubility are the direct metabolic processes involving enzymes, carbon dioxide production leading to a lower pH, and organic acid production (11 - 13). Inorganic phosphate is, in turn, used by higher aquatic plants, zooplankton, and phytoplankton.

As with phosphorus, sulfur is cycled by the microbial populations in the aquatic environment and has been linked to decreased productivity of fish (see Fig. 4).

FIGURE 3
PHOSPHORUS CYCLE
SOURCE (14)



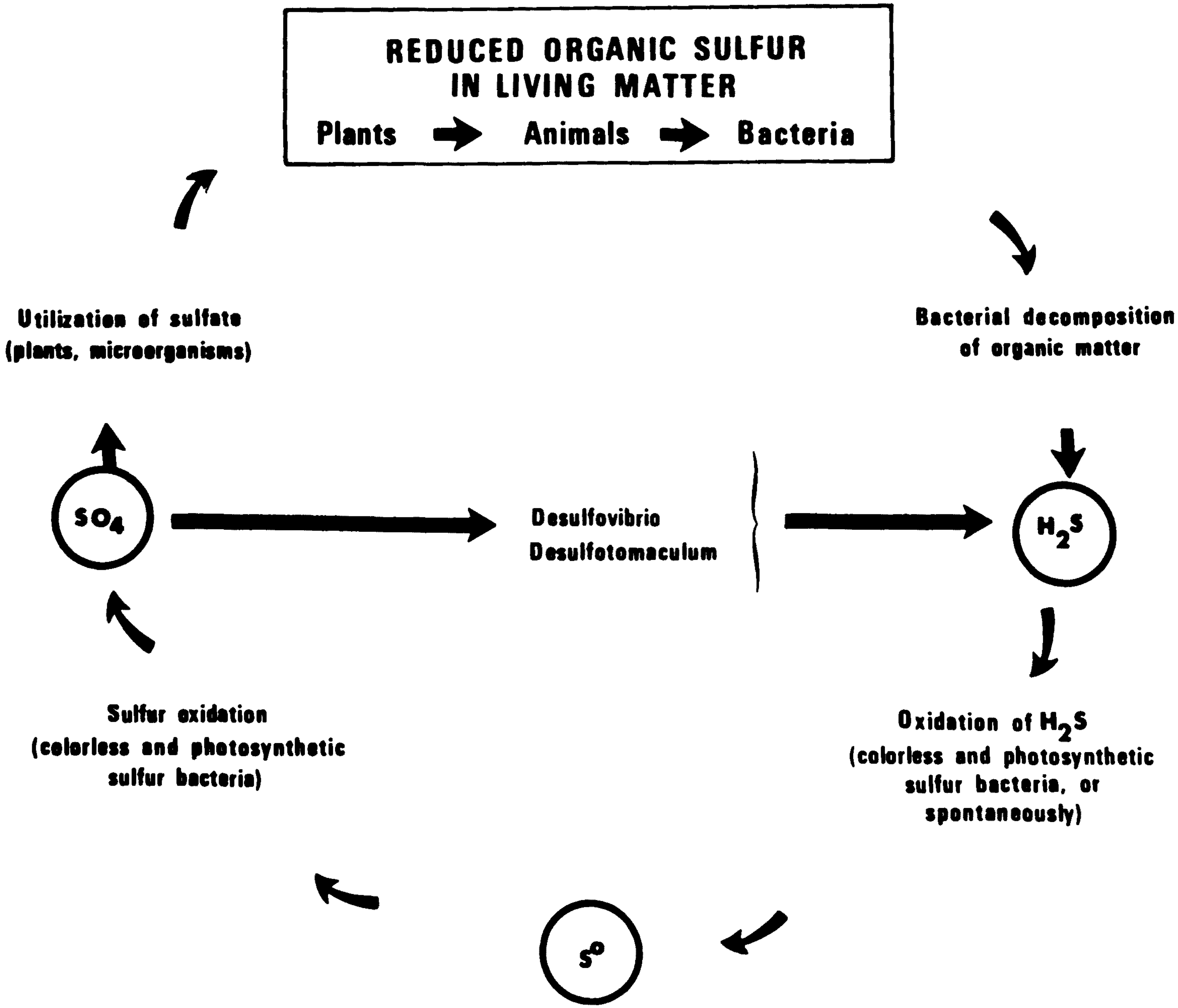
Sulfate can be stoichiometrically reduced to hydrogen sulfide, which in turn can be oxidized chemically, in the presence of oxygen, to elemental sulfur. Elemental sulfur in turn, can be oxidized to sulfate. A specific class of bacteria, the anaerobic dissimilatory sulfate reducers, also leads to the stoichiometric production of hydrogen sulfide and consequent anaerobic environments. On the other side the oxidation of elemental sulfur by Thiobacilli leads to the production of sulfuric acid and their metabolic activity is evident in the acid mine drainage in certain areas of the country.

Biological nitrogen cycling involves, as does the cycling of sulfur and phosphorus, the transition of an elemental nutrient through various chemical states. Fig. 5 is a schematic representation of the cycling of nitrogen. It is convenient to initiate the consideration of the nitrogen cycle at a point where fixation of gaseous nitrogen occurs. Relatively few species of microorganisms populating the earth are capable of metabolizing nitrogen from the air (16 - 19). Once fixed from the atmosphere nitrogen is converted by a relatively few species of bacteria and blue-green algae to organic nitrogenous compounds.

FIGURE 4.

THE SULFUR CYCLE

SOURCE (14)

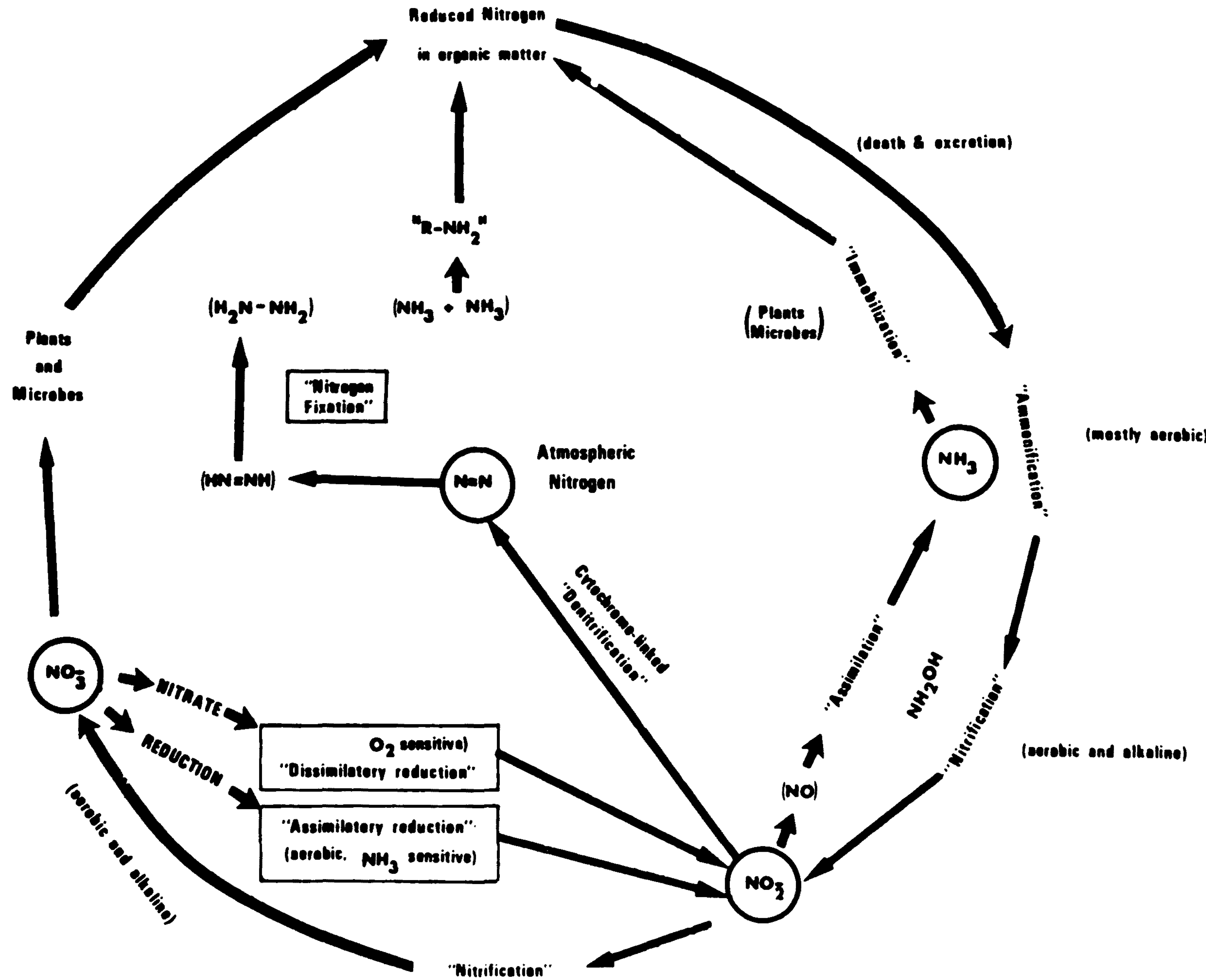


Subsequent to fixation the relative concentrations of the inorganic nitrogen compounds in water, i.e., nitrate, nitrite, and ammonia, depend, in part, on the amount of oxygen available and the oxygen concentrations are dependent upon the organic carbon load and seasonal variations in solubility of oxygen in winter. Attempts to develop a nitrogen balance in lakes and other aquatic environments are hampered by the fact that there are several possible sources for loss of nitrogen. For example, fixed nitrogen can be lost via: (1) lake effluents; (2) loss of volatile nitrogen such as ammonia and nitrogen gas; (3) denitrification by certain microbes; (4) precipitation of nitrogenous compounds into either permanent or semipermanent sediments; and (5) removal of organisms by fishing, weed harvesting or other methods of fauna and flora depletion.

The biochemical mechanisms involved in denitrification have only recently been elucidated in significant detail (20 - 23). These reactions result in the conversion of nitrate to, ultimately, nitrogen gas and are apparently unique to a limited group of microorganisms.

Figure 5

REVIEW OF THE NITROGEN CYCLE
SOURCE (14)



The carbon cycle is composed of an integrated network of physically and biologically mediated pathways encompassing the synthesis, degradation, and transformation of innumerable simple and complex organic molecules (Fig. 6). Superimposed on the carbon cycle are the controls exerted by nutrient availability, and the fixation and evolution of carbon dioxide. Various aspects of the organic carbon cycle in the aquatic environment have been examined with the emergent principle that an overall balance between the production, or synthesis, and decomposition of naturally occurring substances exists in nature (24, 25).

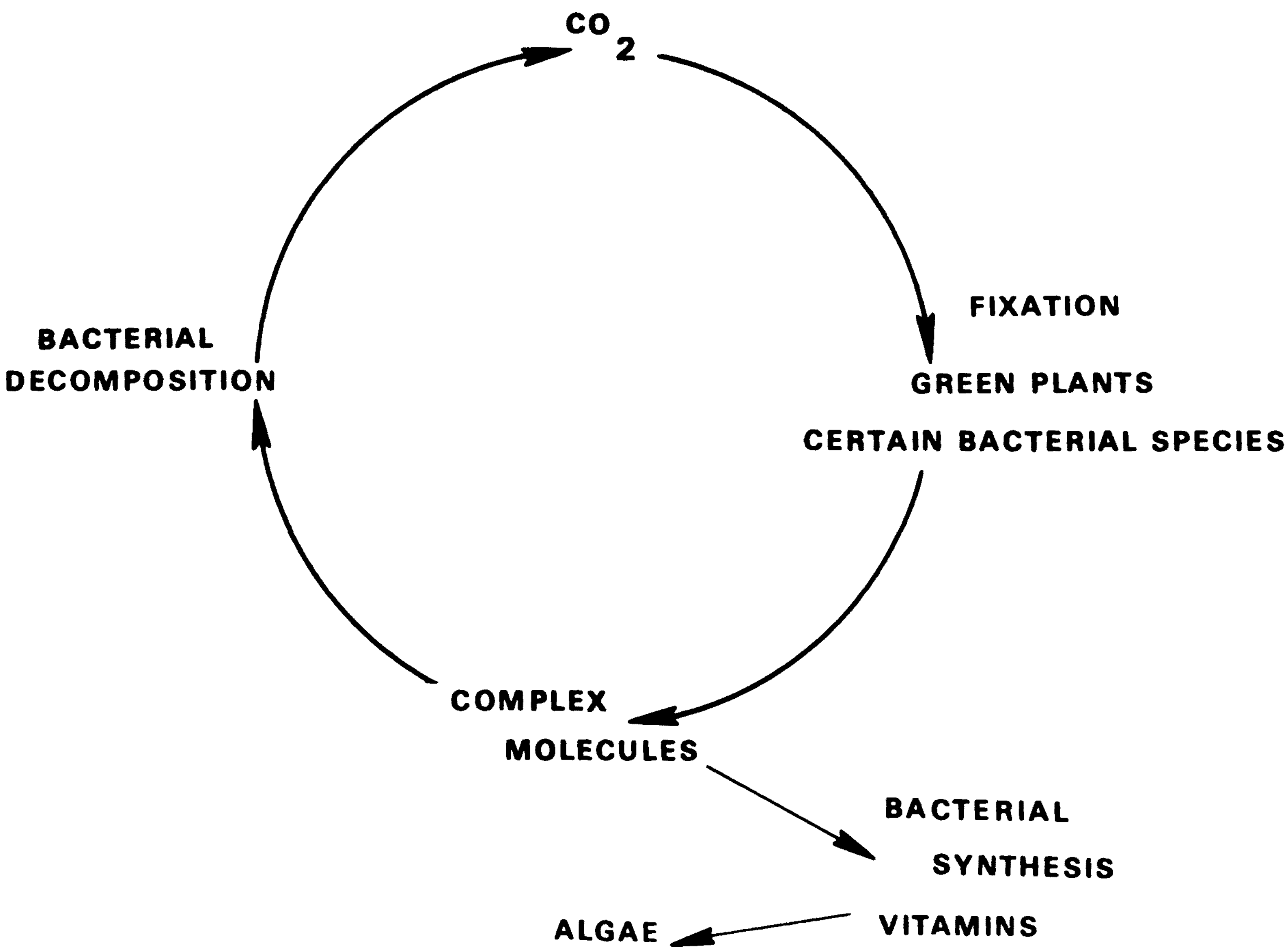
Photosynthetic carbon dioxide fixation by green plants is a major route by which carbon enters the organic carbon cycle. However, fixation by autotrophic bacteria adds to the total carbon budget in the ecosystem (26, 27). Once organic material has been introduced into the aquatic environment the endogenous flora and fauna can either utilize or contribute to, depending upon conditions, an existing reservoir of organic material (28). Some of the ecological questions relating to carbon arise when considering the microbe's direct relationship to carbon cycling are: what effect does microbial synthesis of complex molecules such as vitamins, amino acids, carbohydrates, and lipids have on the aquatic biota; what is the contribution of bacterial biomass, a food source for zooplankton; and what is the significance of microbial degradation of suspended soluble or sedimented organic compounds?

Direct and complex relationships between diverse organisms have evolved based on the needs for various growth factors. Examples of these relationships are seen in the association of various algae and bacteria in the marine and fresh water environments (29 - 32). Also, the degradation of complex, naturally occurring organic compounds such as chitin are affected by the microbial species.

Microbial metabolic activity affects the cycling of the four major inorganic nutrients under consideration. The cycling of each of these nutrients - phosphorus, sulfur, nitrogen, and carbon - is interrelated in that any perturbation in one cycle has far reaching effects in the other cycles. For example, it has been shown that the sulfate reducing bacteria are capable, not only of nitrogen fixation, but of degradation of carbon compounds to carbon dioxide and also of effecting a solubilization of phosphate as a consequence of precipitation of insoluble iron sulfide precipitates. This is but one example. There are many examples of these interrelationships of microbial communities with higher faunal and floral communities and with water quality.

FIGURE 6

MICROBIAL CYCLING OF CARBON



Section IV

POSSIBLE REMEDIAL MEASURES FOR RESTORING AND ENHANCING THE QUALITY OF THE NATION'S PUBLICLY OWNED LAKES

Lake restoration technology is in its infancy. Only a few lake renewal programs have proved successful, and these only on individual lakes. A method of lake rehabilitation which may be highly successful on a given lake, may be totally impractical or unworkable on another. Each lake has its own peculiar characteristics, differing from all others geographically, morphologically, chemically and biologically as well as in the nature of its problems. Consequently, it is impossible at this point in time to recommend remedial measures which will prove to be effective for all lakes or even particular classes of lakes. It is the responsibility of lake managers to define the problems and to implement rehabilitation or enhancement programs which are best fitted to the requirements of particular lakes on a case-by-case basis.

This section presents information on possible remedial measures which have been or are presently being applied in lake rehabilitation programs or in some cases are being evaluated in the laboratory. Many techniques are currently in experimental stages on small lakes, and the results are inconclusive at this time. Other techniques have met with varying degrees of success on individual lakes, but their applicability to other lakes is unknown.

Since eutrophication poses the greatest threat to the Nation's lakes, this report focuses primarily upon those remedial measures which may be applicable to certain lakes displaying symptoms of accelerated or man-induced eutrophication. Possible remedial measures for lakes contaminated with industrial wastes including toxic substances and hazardous materials are only briefly discussed. Subsequent reports will deal with these problems in greater detail. Solutions to problems associated with thermal discharges to lakes are not addressed in this report. Thermal discharge control technology is to be addressed in a forthcoming EPA publication as required by Section 104(t) of the Federal Water Pollution Control Act Amendments of 1972.

The approach to the rehabilitation of degraded lakes is two-fold: (1) by restricting the input of undesirable materials and (2) by providing in-lake treatment for the removal or inactivation of undesirable materials. Obviously, the only means of maintaining the quality of a lake once desired conditions are achieved, is by rigidly restricting the input of undesirable materials. In some lakes reducing or eliminating the primary sources of waste loading is the only restorative measure needed to achieve the desired level of improvement. Once the source of pollution is abated, natural flushing and dilution with uncontaminated water may result in substantial improvements in the quality of the lake. However, in many lakes, particularly in hypereutrophic lakes with slow flushing rates, in-lake treatment schemes may also be required before significant improvements will be realized. In-lake treatment alone without controlling pollutional inflows cannot be termed a restorative measure as only the symptoms or products of eutrophication and pollution are treated and no permanent improvements in quality are achieved. In any lake restoration program, controlling the input of undesirable meterials is the initial step towards permanent lake rehabilitation; all other remedial measures are supplementary to this action.

In the following discussion, measures which may be effective in the restoration and enhancement of the quality of lakes are addressed under four major headings as follows:

1. RESTRICTING NUTRIENT AND SEDIMENT INPUT

   A. Point source nutrient removal and control

   B. Nutrient diversion

   C. Control of allocthonous sediments

2. IN-LAKE TREATMENT AND CONTROL MEASURES

   A. Dredging

   B. Nutrient inactivation

   C. Dilution and dispersion

   D. Covering of sediments

   E. Artificial destratification and hypolimnetic aeration

   F. Drawdown

   G. Harvesting nuisance organisms

   H. Biological control of nuisance organisms

   I. Chemical control of nuisance organisms

3. CONTROL AND REMOVAL OF HAZARDOUS SUBSTANCES

4. POSSIBLE LAKE PROTECTION MANAGEMENT CONSIDERATIONS

RESTRICTING THE NUTRIENT AND SEDIMENT INPUT

Point Source Nutrient Removal and Control

Domestic wastewater represents a significant source of aquatic plant nutrients and therefore is the source that is often considered first for control.

Conventional waste treatment systems using sedimentation and activated sludge or trickling filters remove only suspended and dissolved solids and a portion of the nutrients. Although these systems serve to reduce the BOD load to receiving waters, they generally remove less than 50 percent of the phosphorus and nitrogen (33).

The technology is presently available to remove both phosphorus and nitrogen from wastewater at a moderate cost. Phosphorus removal efficiency of 80 to 95 percent can be achieved by chemical precipitation with alum, lime or ferric salts. Removal of ammonia and other nitrogen species can be accomplished by ion exchange, ammonia stripping at high pH in a gas stripping tower, breakpoint chlorination or bacterial denitrification.

Advanced wastewater treatment (AWT) for nutrient removal probably represents the best method currently available for curbing nitrogen and phosphorus input to waterways. An obvious limitation of advanced waste treatment is its inapplicability to the treatment of most wastes from non-point sources. However, under certain circumstances entire rivers which receive their nutrient loads from diffuse sources may be treated prior to their entry into a lake. In Germany, it has been proposed to treat the entire Wahnbach River using iron to precipitate the phosphorus. The Wahnbach, which forms the Wahnbach Reservoir, receives its wastes primarily from agricultural runoff.

The storage and disposal of waste materials extracted in advanced wastewater treatment plants add to the total treatment costs. The concentrated sludge and liquid must be disposed of in such a manner that the nutrients do not re-enter a waterway. The practice of depositing sludge in marsh areas and along waterways is ecologically unsound.

However, the application of the sludge to cropland to increase production is one beneficial means of disposal.

Information on the cost and efficiency of various advanced waste treatment processes currently in use in the United States is presented in Tables 1 and 2. Table 1 compares total costs and removal efficiency for various nitrogen control processes. Table 2 presents information on average costs of phosphorus removal based upon 1971 data compiled by Cecil (34). From an examination of these data it is apparent that although some processes are more expensive than others, in most instances for comparable levels of nutrient removal efficiency, the cost ranges overlap. The characteristics of the particular situation at hand which influence the cost of the treatment process include: (1) the existing treatment facility, (2) required water quality standards, (3) use and character of the receiving water, and (4) climatic conditions. Since nutrient removal treatment systems are usually built as modifications of existing plants, the most important single factor influencing the selection of treatment processes is the existing treatment facility (35).

TABLE 1

Comparison of Nitrogen Removal Processes a/

Process	Class	Removal Efficiency %	Estimated Cost ¢/3.785 m^3 (¢/1,000 gal.)	Wastes to be Disposed of	Remarks
Ammonia stripping	Physical chemical	64-80	3.8-10	liquid	Efficiency based on ammonia nitrogen only
Ion exchange (Clinoptilolite)	Physical chemical	90-95	5.7-13.6	liquid	Efficiency and costs depend on degree of pretreatment
Breakpoint chlorination	Chemical	99	10.8-20.6	None	Requires strict process control
Nitrification-Denitrification	Biological	90	5.2-17.3	Sludge	Requires some chemical addition and large land disposal area

a/ Data supplied by the Advanced Waste Treatment Laboratory, National Environmental Research Center, Cincinnati, Ohio and the Municipal Technology Branch, Technology Division, Office of Research and Monitoring, Washington, D.C.

TABLE 2

Treatment Plant, Operating and Maintenance Costs for Phosphorus Removal

Treatment Plant Costs a/b/

Plant Size	3,785 m^3/day (1 mgd)	37,854 m^3/day (10 mgd)	378,540 m^3/day (100 mgd)
	Capital Investment Costs in Dollars		
Building and Structures	15,000	40,000	90,000
Process Equipment Installed			
Lime	45,000	150,000	410,000
Aluminum salts	35,000	96,000	300,000
Iron salts	35,000	85,000	250,000
	Operating and Maintenance Costs in Dollars/day		
Labor			
Operating	35	100	200
Maintenance	18	90	225
Amortization			
Lime	13.20	41.80	109.15
Aluminum salts	10.90	29.80	86.50
Iron salts	10.95	29.48	74.80
Chemicals b/ and Sludge c/ Disposal Costs			
80% P Removal			
Lime	36.50	329.45	3,293.20
Aluminum salts	40.45	382.65	3,643.20
Iron salts	43.05	389.15	3,888.75
90% P Removal			
Lime	66.35	584.75	5,303.20
Aluminum salts	53.70	509.30	4,933.20
Iron salts	56.10	508.75	4,998.20
Total Daily Operating and Maintenance Costs			
80% P Removal			
Lime	102.70	561.25	3,827.35
Aluminum salts	104.35	602.45	4,154.70
Iron salts	100.45	608.63	4,338.55
90% P Removal			
Lime	132.55	816.55	4,837.35
Aluminum salts	117.60	729.10	5,444.70
Iron salts	120.05	728.23	5,498.00

a/ Source: (34)
b/ The use of polymers for improved coagulation is included in chemical costs
c/ Land disposal is assumed

In this country there are currently about 1200 wastewater treatment plants, planned or in operation, which incorporate some degreee of AWT technology. However, to date there has not been documentation evaluating AWT as a means of restoring a lake. The EPA program at Shagawa Lake (36) will possibly be the first thorough evaluation documenting restoration of a lake by nutrient removal through AWT of municipal wastewater. Lake Tahoe and the AWT plant there have been studied for a number of years; however, the plant effluent does not enter Lake Tahoe but is diverted to a reservoir outside the watershed.

Several advanced wastewater treatment plants are in operation in Europe but data documenting the effects on lake restoration are incomplete. Preliminary data on the Greifensee in Central Europe indicate that the phosphorus content stopped its upward climb after an AWT plant was built to remove 90 percent of the phosphorus from the Uster municipal wastewater (37).

Other possibilities for removing nutrients from a point-source include spray irrigation, soil infiltration and culturing and harvesting algae or aquatic plants. Spray irrigation of wastewater on land to facilitate the growing of crops results in two methods of nutrient removal. It ties up nutrients, particularly phosphorus, in the soil, and it allows nutrients to be incorporated into a crop that can be harvested and removed from the watershed.

This technique is presently being evaluated as a nutrient removal technique through an EPA grant at Muskegon, Michigan.

Pennsylvania State University has shown that crops that have been irrigated by wastewater effluent can substantially remove nutrients contained in the effluent (38). In the upper 30.5 cm of soil the concentration of nitrate was reduced up to 82 percent and phosphorus up to 99 percent.

Studies in Oklahoma showed that grasses grown in hydroponic culture tanks removed appreciable nitrogen but only slight amounts of phosphorus from secondary wastewater (39). One drawback to the spray irrigation technique is that long term irrigation with water high in sodium or other metals could render a soil unproductive if these materials reach an undesirable concentration.

Soil infiltration, whereby wastewater is allowed to move through the soil, removes or greatly reduces suspended solids, biochemical oxygen demand, microorganisms, phosphorus, fluorides, heavy metals and other substances, including nitrogen if the recharge system is properly managed (40). Peat is particularly good for removing phosphorus. In an EPA study (41) it was shown to remove 95 to 99 percent of the phosphorus from secondary wastewater.

Species of the bulrush, Scirpus, have been used in the biological purification of wastewater (42). Phosphorus and nitrogen are readily taken up by these plants and periodic harvesting of Scirpus will remove the nutrients from the system. The use of Scirpus to facilitate wastewater treatment is being evaluated in Germany.

The culturing and harvesting of algae for nutrient removal have been evaluated. EPA is presently evaluating this technique at Firebaugh, California, to remove nitrogen from agricultural return canals that enter San Francisco Bay. In South Africa (43) culturing and harvesting algae have been studied as a method of producing water suitable for reuse from wastewater.

<u>Nutrient Diversion</u>

Diversion offers a possible restoration technique in situations where the majority of the incoming nutrient load is entering from specific point sources. It has been used as a technique to control nutrient input from municipalities located around the perimeter of lakes.

The major disadvantages include the following:

1) Monetary costs - the expense of installing the necessary collection system for many lakes may be prohibitive.

2) Environmental costs - diversion of untreated sewage from a lake to another waterway may result in the degradation of that waterway and the substitution of one problem for another.

3) Lake morphometry - If the lake basin is shallow, nutrient exchange between sediment and water <u>may</u> recycle nutrients to the extent that no recovery is discernible.

4) Ground water - If the ground water inflow is significant with respect to total hydrologic budget and it is high in nutrients, recovery will be very slow or no recovery may occur.

5) Hydraulic residence time - The rate at which high nutrient water leaves the basin will affect eventual recovery.

Case Studies:

1. Lake Washington - Seattle, Washington, USA (44 - 47)

Lake Washington at Seattle is a former oligotrophic lake which rapidly deteriorated to a state of eutrophy, but which in recent years has shown definite signs of recovery.

The lake lies in an elongate, steep-sided glacial trough with a maximum depth of 65.2 m, mean depth of 32.9 m and a surface area of 8768 hectares (21,650 acres).

Prior to 1963, Lake Washington received heavy nutrient loading from eleven sewage treatment plants discharging directly into the lake. It is estimated that in 1957, 56 percent of the phosphorus and 12 percent of the nitrogen entering the lake was from sewage effluent. Extensive Oscillatoria rubescens blooms were observed in 1955 indicating considerable degradation of water quality. The abundance of algae was approximately 15 times greater in 1962 than in 1950. Secchi disc measurements had been reduced from 3 meters in 1950 to about 1 meter in 1963, 1964, and 1965. Nutrient concentrations increased dramatically. Phosphorus increased from 0.008 mg/l in 1933 to 0.475 mg/l in the 1960's, and nitrate from 0.170 mg/l in 1933 to 0.475 mg/l in the 1960's. Dissolved oxygen concentrations reached zero in the deeper water strata for the first time in 1957.

A series of steps was instituted by Metro (Municipality of Metropolitan Seattle) in the late 1950's to divert the sewage from Lake Washington and to build a series of new treatment facilities which would discharge into Puget Sound. Estimated cost for the project was about $120,000,000. The first phase of the diversion was completed in 1963, at which time approximately 25 percent of the effluent bypassed the lake. In 1965, the effluent volume entering the lake was reduced to approximately 55 percent of the original load, and by 1968, the project was complete with approximately 100 percent of the effluent diverted.

Improvement in water quality has been dramatic since diversion was completed. Phosphorus concentrations in 1969 were 28 percent of the 1963 values and nitrogen concentrations were 80 percent of the 1963 levels. Secchi disc measurements have increased from 1.0 m to 2.8 m. Chlorophyll levels have decreased to approximately 15 percent of the mean winter values for 1963, and noxious blooms of blue-greens have been eliminated.

Lake Washington has shown a significant improvement with the diversion of sewage. A reduction of 50 percent in the phosphorus loading has greatly decreased the algal growth and a significant increase in transparency has occurred. Data indicate that phosphorus is the controlling element with respect to algal growth in Lake Washington and the results of the diversion illustrate this in a dramatic manner.

2. Lake Sammamish, Seattle, Washington, USA (48)

The outlet of Lake Sammamish forms the inlet to the north end of Lake Washington. In 1968 the sewage was diverted from Lake Sammamish, but recovery has not been observed. Approximately 65 percent of the total phosphorus and 22 percent of the nitrate-nitrogen were diverted with the interception system. Surface nutrient concentrations, algal activity, light penetration and hypolimnetic oxygen deficits have not changed.

Although Lake Washington has shown a dramatic recovery, Lake Sammamish has not. Proposed reasons for this include: (1) a greater exposure of epilimnetic waters to sediment in Lake Sammamish (65 percent more than Lake Washington), (2) the lesser state of eutrophication of Lake Sammamish at the time of diversion, (3) the possibility of fungi (actinomycetes) complexing phosphates and removing them from the system, and (4) ground water infiltration from urbanized areas of the lake. No experimental work has been conducted on the first three proposals but the fourth alternative is unlikely because intensive monitoring of the tributaries has failed to detect abnormally high phosphorus concentrations.

3. Madison Lakes, Wisconsin, USA (45, 46, 49, 50)

The city of Madison, Wisconsin is located between Lakes Mendota and Monona, the first and second lakes in a series of four on the Yahara River.

All of the Madison Lakes have a long history of algal problems, but Mendota has been the least troublesome. Lakes in this region are naturally productive, but the problems in the Madison Lakes were attributed to urbanization.

In the early history of the city, Lake Monona received the sewage from the city of Madison. As a consequence in 1912 algal growths had become so prolific that copper sulfate was used to kill the algae, and in 1925 a regular program of treatment with copper sulfate was established. The condition of the lake deteriorated steadily. In 1928 the Nine-Springs plant was placed in operation and the effluent from this operation was carried via Nine-Springs Creek to the Yahara River downstream from Lake Monona.

Algal productivity in Lake Monona was not measured directly, but the quantities of copper sulfate used to control algal growths may be indicative of the intensity of algal crops. Since relocation of the plant, the amount of copper sulfate needed to prevent obnoxious blooms has decreased dramatically. A total of 1,579 kg (3,481 pounds) was used from 1955 to 1963 as compared to 45,587 kg (100,500 pounds) used in 1934 only. A change in species composition has also occurred. The algae presently inhabiting the lake do not cause surface scums, thus the need for copper sulfate has diminished.

The relocation of the sewage treatment plant did not end the Madison Lakes' problems. Shortly after the effluent was moved downstream from Lake Monona, the symptoms of overenrichment in Lakes Waubesa and Kegonsa began to intensify and copper sulfate treatment in large doses was required.

The community eventually adopted a plan by which the effluent was diverted from the Madison Lakes via the Badfish River to the Yahara River downstream from the lakes. The diversion project was completed in 1958. Since diversion, the condition of the lakes appears to have improved, but the Badfish River has deteriorated considerably.

It is difficult to relate the Madison Lake diversion project to the Lake Washiqton case, because the Madison area is much richer in dissolved minerals than is the Lake Washington area, and consequently the Madison Lakes are naturally more productive. In addition the Madison Lakes are much smaller and much shallower than Lake Washington (see Table 3).

TABLE 3

PHYSICAL CHARACTERISTICS OF THE MADISON LAKES

Lake	Length km	Length (miles)	Width km	Width (miles)	Area, km^2	Area, $(mi)^2$	Max depth, m	Mean depth, m
Mendota	9.5	(5.9)	7.4	(4.6)	39.4	(15.2)	25.62	12.1
Monona	6.7	(4.2)	3.9	(2.4)	14.1	(5.4)	22.57	8.4
Waubesa	6.8	(4.2)	2.3	(1.4)	8.2	(3.2)	11.16	4.9
Kegonsa	4.8	(3.0)	3.6	(2.3)	12.7	(4.9)	9.58	4.7

4. Red Lake (Rotsee), Lucerne, Switzerland (45)

Sewage was diverted from Red Lake (Rotsee) in 1933, but the lake continued to produce nuisance quantities of algae. The reasons for the lack of improvement in Red Lake following diversion are attributed to the lake's small size and the considerable drainage it receives from fertilized and cultivated cropland.

5. Lyngby-So, Copenhagen, Denmark (45)

The sewage was diverted from Lyngby-So in 1959 and productivity, as measured by the rate of photosynthesis, decreased markedly for the next four years. The submerged rooted aquatic vegetation disappeared from the lake after 1956 presumably from shading by algae, but the aquatic macrophytes are now becoming reestablished. It appears in this case that recovery began immediately.

6. Stone Lake, Michigan, USA (51)

Stone Lake which has a surface area of 56.6 hectares (140 acres) began to receive secondary sewage in 1939. The treatment plant was replaced in 1965 and the treated wastes since then have been disposed of outside the drainage basin. The only remaining sources of pollution are a few household septic tanks located on the periphery of the lake. The lake has shown little response to the cessation of nutrient influx from the treatment plant.

Several reasons are suggested for the failure of the remedial technique. Although over 95 percent of the phosphorus and 50-75 percent of the nitrogen were removed, some pollution is still entering the lake (organic materials). Because of the relatively shallow morphology (mean depth 6.1 m.) sediment-water nutrient interchange may be responsible for recycling previously deposited materials. Further, hydraulic retention time is of such a magnitude (11 years) that insufficient time has elapsed to observe significant improvement in water quality.

7. Lake Annecy, France, (52)

Lake Annecy was classified in 1937 as "becoming eutrophic". Conditions became more pronounced with time because of human waste disposal to the lake. Low dissolved oxygen became the norm and species composition of the phytoplankton indicated an advanced eutrophic state during the 1960's. In 1961, a diversion system was begun. By 1971 approximately 44 percent of the population around the periphery of the lake was using the system.

Changes in algal composition indicate that the water quality of the lake is improving, but the studies have not been carried on long enough to determine the long term effect.

8. Diamond Lake, Oregon, USA (53)

A sewage interceptor system has been installed around one-half the periphery of 1,944 hectare (4,800 acre) Diamond Lake. The lake is primarily used for trout fishing, and extensive camping, lodge and summer home facilities are located around the circumference. The U. S. Forest Service installed a waste collection system to replace existing septic tank systems in 1971, but it is not as yet fully operational. Studies have been made to obtain background data on the chemical, physical and biological characteristics of the lake. These will be monitored in the future to determine changes in water quality.

9. Lakes Tegernsee and Schliersee (54)

Sewage diversion (finished 1964/65) from two Bavarian lakes (Tegernsee and Schliersee) resulted in a reduction of the phosphate load to the lakes to about 10-20 percent of the former amount while nitrogen income was diminished to about 25-40 percent. In 1967 improvement was observed, especially with better hypolimnetic oxygen conditions at summer stratification. Subsequent years, however, showed a relapse in the highly eutrophicated Schliersee to oxygen-free hypolimnion again, while improvement at the Tegernsee was more or less maintained. Intensive remobilization of nitrogen and phosphorus from lake deposits permanently increased nutrient levels in the Schliersee up to 1970. A partially meromictic (permanent stratification) situation seems to be mainly responsible for this process. Different circumstances which may promote or restrain improvement after sewage diversion include hydrological and climatic conditions, progress of eutrophication at the moment of sanitation, and intensity of nutrient-turnover. Notwithstanding eutrophication parameters, sewage diversion has removed primary pollution of the lakes and their tributaries which is of great importance for their recreational function.

Control of Allochthonous Sediments

Sedimentation of lakes and reservoirs is a major factor restricting the available acreage of the Nation's recreational waters. In terms of volume, sediments are the greatest pollutant.

Sedimentation rates in lakes and reservoirs can frequently be retarded by prudential land use management practices within the watershed. Construction and logging activities should shun the steepest slopes, and projects which denude the landscape should be timed to avoid seasonal rainy periods. Agricultural practices such as strip cropping, contour plowing, and proper grazing practices prevent rural erosion and consequent sedimentation in streams and lakes. Terraced hillsides and banks of watercourses stabilized by riprap or gabions are also effective erosion preventive measures.

Sediment Traps

Sediments may sometimes be trapped before they enter lakes and reservoirs by filter dams and desilting basins installed downstream from all large cleared areas and other sources of silt. Detailed descriptions of sediment traps and their use as well as other effective sediment controls may be found in the publications by Thronson (55) and the National Association of Counties Research Foundation (56).

Analysis of Sediment Transport

The mechanics of sediment transport have been extensively studied and hydraulic and mathematical model studies of bedload and suspended load transport are described by Bogardi (57). Three stages of statistical analysis can be recognized in sedimentology (58). The first stage is descriptive statistics in which the sample is the object of interest. The second stage is analytical statistics in which the population is the object of interest. The third stage is the application of stochastic process models in which the objective is to discern the probabilistic elements in sedimentary processes. Krumbein's paper, as abstracted in Selected Water Resources Abstracts by the U.S. Department of the Interior, states (58), "Stochastic process models thus provide one way of examining sedimentary processes through time or over an area. In conjunction with deterministic models they provide a framework for exploring the underlying physical, chemical, and biological controls on sedimentary processes and deposits...." Using turbulent diffusion theory (59) a non-steady-state model was developed for sediment transport.

Cost Effectiveness Models

The economic benefits to be gained by controlling erosion and sedimentation are compared to the control costs in a cost effectiveness study. Such a study on the Seneca Creek watershed, near Washington, D.C. (60) compared cost to effectiveness and damage values for many sediment control methods. Present control practice includes sediment basins, diversion basins, level spreaders, grade stabilization structures, sodded ditches, seeding, and straw mulch tacked with asphalt or disked. The average conventional system is estimated to cost $2780/hectare and to control 91% of the potential erosion. Control systems incorporating large sediment basins can boost control to 96% at less total cost. Economic aspects of sedimentation are also discussed by Maddock (61).

IN-LAKE TREATMENT AND CONTRCL MEASURES

Dredging

Many lakes have suffered the consequences of filling and nutrient enrichment as the result of allochthonous materials entering from the watershed. Highly eutrophic lakes also receive large amounts of autochthonous materials resulting from massive algal populations. Much of the organic material entering the system will not be decomposed because of the low oxygen conditions in bottom waters associated with increased productivity. The eventual result is an accelerating rate of sedimentation and a filling of the lake basin.

Dredging has thus been proposed as a possible remedial technique. This would serve to remove the sediment build-up, thus increasing the depth of the lake, and removing a potential nutrient source. A large number of lakes have been dredged but no information is available on the chemical or biological effects.

A number of disadvantages are obvious with respect to this technique:

1) The relatively high costs of dredging operations may make this technique prohibitively expensive on large lakes.

2) The dredging operation may release nutrients from the sediments, making them available for reinvolvement in the food web. The nutrient content of many sediments may remain high at considerable depths, making it impossible to reach a low nutrient level in the sediment.

3) The elimination of shallow zones which maintain large macrophyte beds, may result in a considerable increase in the algal populations. The nutrients formerly tied up in macrophyte biomass could become available for algal growth. The result may be the substitution of one problem for a second.

4) Turbidity resulting from the dredging process may persist for a considerable time during and following dredging.

5) Disposal of the dredged spoils economically is often impossible. Sediments may prove unsuitable for agricultural purposes and in such a case, could be used for land fill only.

6) Interstitial waters contained in sediments are frequently high in nutrients, consequently, disposal of the sediments must be in such a manner that leeching of nutrient rich waters back to the lake is prevented.

Case Studies:

1. Lake Trummen, Vajko, Sweden (62)

Lake Trummen is a shallow (1.1m mean depth) small (1.0 km^2) lake located in central south Sweden. The lake is one of a series of oligotrophic lakes, indicating that Lake Trummen was also once oligotrophic. Waste water has entered the lake since the turn of the century resulting in a highly eutrophic condition for many years. Studies indicate that a 20 cm layer of black, highly organic gyttja has been deposited since human habitation began around the lake.

Plans were instituted in 1966-67 to develop some type of restoration program. The final decision has been to dredge to a depth sufficient to remove the recent gyttja deposits, and to dispose of these materials in bay areas of the lake. An estimated 600,000 m^2 of sediment will be removed. The water released from the sediments upon their deposition on land will be treated with aluminum sulfate to remove phosphorus before it is allowed to reenter the lake.

The Lake Trummen study is extremely comprehensive, involving individuals from at least sixteen disciplines relating to water quality management. Data have been collected for two years prior to the study, and will be collected during the 2-year dredging operation. The lake will be monitored for 8 years after completion of the dredging. Dredging operations were to begin in 1970, but no information is yet available.

2. Lake Herman, South Dakota, USA (63)

A suction type dredge was used to remove silt from eutrophic Lake Herman. Analytical results indicated that water from the dredged material when it returned to the lake was lower in pH and total phosphates, and almost as clear as the lake water. Total orthophosphate-phosphorus increased dramatically in the lake water (approximately doubled) during dredging, but the concentrations of the other nutrients remained at approximately the same level.

The following synopsis is taken from Technical Bulletin Number 46, Inland Lake Dredging Evaluation, Department of Natural Resources, Madison, Wisconsin (64). Very general data are presented and none of the lakes appear to have been investigated to determine water quality changes associated with the dredging operations.

3. Wazeecha Lake, Wood County, Wisconsin USA

The upstream end of Lake Wazeecha, a 60 hectare (148 acre) impoundment of Buena Vista Creek, was dredged by the County over a four to five-year period. The dredged area was deepened by 1.2 to 1.5 m by the removal of 133,530 m^3 of sediment. The total cost of the operation, including the purchase price of a second-hand 19.6 cm hydraulic cutterhead dredge was $66,859, at a unit price of $0.50/$m^3$ ($0.38/$yd^3$). The dredged spoil was pumped onto the shoreline, improving the conditions of the shoreline. One area was diked off and filled, and a new park was created on the fill.

4. North Twin Lake, Calhoun County, Iowa USA

North Twin Lake, a 207 hectare (510 acre) lake in the predominately agricultural plains country of west central Iowa, had undergone rapid sedimentation resulting from severe bank erosion and sheet erosion from the surrounding farmland. The lake had been filled with as much as four meters of sediment, reducing the lake depth to 0.6 to 0.9 m. Dredging first began in 1940 when 55 hectares (135 acres) were dredged to a depth of 4.3 to 5.5 m. Dredging was then discontinued until 1960 when five dredging contracts were let to private contractors. During the 1960's the entire lake beyond 45 m from the high water line was dredged to a depth of 3.7 to 4.3 m, removing 1,523,498 m^3 of sediment. The project was completed in 1969 at a cost of $934,931. The dredging was done by two contractors, one using a 30.5 cm and the other a 35.6 cm cutterhead. The contractor using the larger cutterhead excavated approximately 0.6 as much material in a period of 4 months as the other contractor did in six years. His total unit costs, including dike work, were estimated at $0.52/$m^3$ ($0.40/$yd^3$) of excavated material, whereas combined unit costs for both contractors averaged $0.61/$m^3$ ($0.47/$yd^3$). The unit costs do not include administrative and engineering expenses.

Benefits to the lake, other than increased water depths, have not been defined as yet.

5. Lake George and Lake Sisseton, Fairmont, Minnesota USA

Lake George and Lake Sisseton are in a chain of five lakes located within Fairmont, Minnesota. The city draws its municipal water from these lakes, and their eutrophic conditon was contributory to high water treatment plant operating costs and a warm municipal supply. In 1966 the city of Fairmont purchased a 30.5 cm portable hydraulic cutterhead dredge and appurtenant equipment at a cost of about $175,000 for the purpose of dredging the entire chain of lakes. To date only Lakes George and Sisseton have been dredged.

Prior to dredging, water depths in the lakes averaged 1.8 to 2.0 m. The lakes were dredged in all areas beyond 46 m from the high water line, sloping down to a maximum depth of 7.6 m or until a hard substrata was reached.

Dredged materials from Lake Sisseton were deposited on an adjacent city-owned 69 hectare (170 acre) farm. The disposal site is sufficiently large to permit adequate settling so that the dredged water which returns to the lake has a very low suspended solids content. The material dredged from Lake George was pumped to a different disposal site which is presently being developed into a park.

It is estimated that 382,328 m^3 of sediment are dredged yearly at a cost of $35,000 to $50,000. Unit costs of dredged materials including engineering and administrative costs, but excluding disposal site costs are estimated at about $0.13 to $0.16/$m^3$ ($.10 to $.12/$yd^3$).

Dredging is part of an overall lake improvement program being undertaken in these lakes. A complete sanitary sewer system was also installed, and the combined effects have reportedly been a marked improvement in water quality although no quantitative data are available to support this contention. The benefits derived from the total project include: greater water depths and volume, lower water temperatures, habitat improvement for fish and desirable aquatic organisms, a general increase in recreational value and reduced water treatment costs.

Additional information is given for several other lakes which have been dredged, but the above examples are representative of hydraulic cutting head dredging experiences in the Great Lakes Region. A survey of 49 inland lakes and ponds indicated that contract unit costs will usually vary between \$0.59 and \$0.98/m^3 (\$0.45 and \$0.75/yd^3) when all costs are considered. The major factors determining costs are : 1) the project size, 2) the type of material to be excavated, 3) distance to disposal sites, and 4) the availability of properly equipped dredging contractors. Such factors as obstructions in the lakes such as tree stumps and boulders, purchase cost of disposal site (if necessary) and experience of the contractor can also influence total costs.

There is no information available on the total ecological effects of dredging upon lake environments or on the water quality. Complete biological, physical and chemical assessments of pre and post dredging conditions need to be made on several lakes with varying characteristics before the benefits derived from dredging can be thoroughly evaluated.

Nutrient Inactivation

It has become apparent after some nutrient diversion studies that nutrients may remain in the water for several years and noticeable improvement in water quality may be delayed. This seems to be particularly true of lakes which have practically no water flow- through to replace that which is high in nutrients. A possible alternative to simply allowing the lake to remain in a highly eutrophic state, is to attempt some method of nutrient inactivation. This process can be defined as the adding of some type of material that will bond with, adsorb, or otherwise immobilize necessary algal nutrients, thus preventing them from being utilized by these organisms for their growth.

Present studies have been directed toward the most common growth limiting nutrients, phosphorus and nitrogen. Phosphorus removal has been used in field studies on three occasions, and some work has been done on a laboratory scale using ammonia ion exchange resins.

Many problems remain to be answered before this technique can be considered operational. A few of the more obvious potential problems are listed below:

1) The relatively high expense of treating the body of water may be prohibitive. Materials may not in themselves be overly expensive, but manpower necessary for application and transportation costs may be considerable.

2) Possible toxic effects by the introduction of an excess of a metal used as a precipitant may have toxic effects on the biota.

3) Adverse biological effects may result from the formation of a floc. The material used may be non-toxic, but the floc could conceivably suffocate aquatic organisms by interfering with their respiratory mechanisms. It is also possible that the floc material resting on the sediments could interfere with the benthic ecology of the system.

4) In order to obtain maximum effectiveness, it may be necessary to either raise or lower the pH of the system, which could have serious biological consequences.

5) The addition of certain salts, such as sulfates and chlorides, may increase the conductivity of the water to an unacceptable level. In the case of sulfate, if the hypolimnetic waters should become anaerobic after treatment, reduction of the sulfate would lead to the release of hydrogen sulfide.

6) Little information is available on the effective duration of the treatment. Wind action, continued inflow of nutrients, bacteriological and benthic organism activity are a few of the phenomena which could possible influence the longevity of treatment effects.

7) The time of application of the inactivent may be critical; it may be necessary to apply the material when the maximum nutrient content is present in the water.

Case Studies:

1. Lake Langsjon, Stockholm, Sweden (65, 66)

Lake Langsjon is a shallow (max. 3 m depth), 35 hectare (86.5 acre) lake which has received municipal waste. Near the end of April 1968, 30 metric tons of aluminum sulfate were added in an attempt to inactivate phosphorus. The final aluminum concentration was about 50 mg/l of lake water. Immediate results included an increase in secchi disc measurements from 50-60 to 250 cm, a reduction in total phosphorus by approximately 50 percent and a reduction in phosphate phosphorus by a factor of 12 (60 to less than 5 ug/l). Total phosphorus increased during 1968, but a concomitant increase in "thero-stalle" coliform bacteria indicated that municipal sewage was entering the lake. During 1969-1970, there was an increase in phosphorus levels during winter stagnation.

In May 1970, the lake was again treated with 32 metric tons of aluminum sulfate. Total phosphorus values were reduced by approximately two-thirds (170 ug/l to 50 ug/l). During the summer of 1970, the phosphate phosphorus remained about 30 ug/l, slightly above the levels encountered the previous summer.

The investigators concluded that the aluminum sulfate was effective with respect to total phosphorus reduction. There was also a slight improvement in dissolved oxygen conditions during winter stagnation: The period between the formation of the ice cover and the development of anaerobic conditions was extended. They concluded, however, that the effects of aluminum sulfate are not long lasting. A substantial increase in both phosphate phosphorus and total phosphorus concentrations occurred during the period of winter anaerobic conditions.

Following treatment, phytoplankton remained about the same with respect to total number of organisms but there was a change in species composition, with a general reduction in the proportion of blue-greens. No adverse effect was noted on the other biota, although little quantitative work was done to verify this.

2. Horseshoe Lake, Manitowoc Country, Wisconsin USA (67)

Horseshoe Lake, an 8.9 hectare (22 acre), 16.7 m deep, eutrophic lake in east-central Wisconsin was selected for treatment and evaluation. The Lake was treated in May 1970, by distributing 10 metric tons of slurried alum in the top 60 cm of water. Alum concentrations in the treated volume were about 200 mg/l (18 mg Al/l), which, based on laboratory testing, resulted in maximum phosphorus removal with minimal ecologic risk. The results of the treatment include: (1) a decrease in total phosphorus in the lake, during the summer following treatment, (2) no large increase in total phosphorus in the hypolimnion during the following two summer stratifications, (3) some increase in the transparency of the water during the summer following treatment, (4) a short-term decrease in color, (5) an absence of the nuisance planktonic algal blooms that had been common in previous years, (6) marked improvement in dissolved oxygen conditions, especially during the following two winters, and (7) no observations of adverse ecological consequences. Manpower, equipment and cost information are summarized in Table 4.

TABLE 4

Summary of Manpower, Basic Equipment and Costs for Alum Treatment of Horseshoe Lake, Wisconsin. 1/

	Item	Costs 2/
Sampling	8 man hours per trip @ $5.00/hour	$ 40.00
	270 miles round trip @ $3.00 day + $0.06/mi.	$ 19.20
Analyses	12 samples per trip @ $30/sample	$ 360.00
Staff	1 professional	$13,000.00
	overhead	$ 7,300.00
Chemicals	10.88 Metric Ton Alum @ $66.18/metric ton (12 ton @$60/ton)	$ 720.00
	Delivery to site	$ 180.00
Labor for Treatment	12 man days @$40/day	$ 480.00
	+ expenses	$ 100.00
Equipment List	2-18 ft. workboats 2-10 ft.x20 ft. barges 4-outboard motors, 18-25 hp 1-amphibious truck, 21/2 ton, DUKW-353 4-gasoline driven pumps 1-4,000 watt generator 2-electric pumps 3-electric mixers 4-55 gallon slurry tanks 2-200 gallon slurry tanks Piping, valves, hose, plastic tubing, marker flags, gasoline, plastic tarp, rope, dust masks	essentially all equipment was on loan

1/ Source: (67)

2/ Many of the costs associated with this treatment are entirely dependent on local salary levels, distances to site, sampling plan, magnitude of treatment, and local availability of equipment. In essence, treatment costs must be estimated for a specified situation.

3. Clines Point--Oregon USA (68)

A 0.4 hectare (1 acre) farm pond with an average depth of 2.4 m was treated with a neutralized solution of sodium aluminate. A concentration of 10 mg Al/l (3 mg/l $NaAlO_2$) was achieved by the adoption of 227 kg of sodium aluminate. The aluminum compound was neutralized with hydrochloric acid prior to its application to form an aluminum hydroxide floc.

The first year's results were encouraging. Total phosphate, ammonia, total kjeldahl nitrogen, iron and manganese remained lower than in previous years, and the algal standing crop was reduced. A shift in dominance from blue-green to green algae was noted. Dissolved oxygen, transparency and pH also indicated a significant improvement in water quality.

Costs excluding labor were $100 for 227 kg of sodium aluminate and $60.00 for the hydrochloric acid. It required five people one full day to treat the pond.

4. Twin Lakes, Ohio; Stone Lake, Michigan USA

There are presently two demonstration grants funded by EPA which anticipate the use of nutrient inactivation as a lake restoration tool. The one project at Twin Lakes, Ohio will combine nutrient diversion and nutrient inactivation. Nutrient diversion is presently being undertaken and plans are to treat the lakes with alum (aluminum sulfate). The other project at Stone Lake, in southern Michigan, will probably utilize fly ash and lime, in an attempt to precipitate the phosphorus as well as seal the bottom (5 cm layer of fly ash in the deeper areas). Laboratory studies have been encouraging using this technique, but the possible hazards must be weighed against anticipated benefits when considering the application of fly ash to lakes.

Many metals have been suggested as possible nutrient inactivation materials. Lanthanium and zirconium have been investigated in the laboratory by the National Eutrophication Research Program, EPA, with varying degrees of success. Other suggested metals include iron, calcium, activated aluminum, bauxite and several of the rare earths. Clays which would serve as adsorption sites for the phosphate and bottom sediments have also been suggested.

These would include such substances as bentonite, montmorillonite and kaolinite. Polyelectrolytes are also feasible but their cost is so great that it may be prohibitive. Materials such as straw and sawdust have also been used but the eventual decomposition of these materials would be expected to create severe problems. Their use would be highly questionable. Another possibility is the resuspension of low nutrient bottom sediment which would absorb phosphorus as it resettled through the water column.

Nutrient inactivation would have to be evaluated on a lake-to-lake basis. No universally acceptable substance has been discovered which could be acceptable under all environmental conditions. Because the technique represents the addition of a foreign material to the water it should be used very carefully. Long term effects on the biota or water chemistry have not been determined for any of the substances listed above.

Dilution and Displacement

The water quality of some lakes can be improved by diluting or replacing the existing water with water of a higher quality. In using this method the replacement water must be readily available and there must be a convenient and acceptable means of discharging lower quality water. Water replacement can be done in one of two ways: (1) by introducing high quality water directly into the lake, thus displacing an equal volume of lower quality water or (2) by removing a given volume of the existing water and replacing it with water of higher quality.

Case Studies

1. Green Lake, Seattle, Washington USA (69)

The displacement technique has been successfully employed in Green Lake, located in Seattle, Washington.

Green Lake is a 104 hectare (256 acre), naturally eutrophic lake, with an average water depth of 3.8 m. Sedimentation has been rapid in Green Lake, with an estimated two-thirds of the volume of the basin filled with sediment by the early 1900's. The present rate of sedimentation is about 0.9 cm per year, practically all of which is autochthonous organic matter. During the summer, mixing of the entire lake is complete except for a very small portion within the confines of the 6 m contours, at which depth thermal stratification persists.

It is estimated that Green Lake has been eutrophic for 7,000 years. The blue-green algae production is very high, and rooted aquatic plants are abundant throughout the littoral zone which comprises much of the total area. Herbicides are applied periodically to control rooted vegetation.

In 1962 water was diverted to the lake from the city's municipal supply for the first time. Between 1962 and 1968, the equivalent of approximately eight lake volumes of water have been flushed through the lake.

A comparison of pre and post flushing data indicates that substantial changes in water quality have occurred following the addition of dilution water. Phosphate levels have dropped considerably, particularly during August and September when blue-green algae growths reach their peaks.

Decreases in nitrate nitrogen were even more pronounced, with the maximum discrepancies occurring during midsummer.

Since dilution definite changes in the species composition of the phytoplankton have been observed. The blue-green algae which, in 1959, were dominant during all months of the year, were the most prominant form during only 5 or 6 months during 1965 and 1966. There has also been a shift amongst the blue-green algae to those species which are able to fix gaseous nitrogen. For example, *Aphanizomenon flos-aquae* which was the major nuisance alga during 1951 has all but disappeared from the lake, and *Anabaena cirinalis* and *Gleotrichia echinulata* have increased in abundance.

Further investigation of the Green Lake situation is continuing, and attempts are being made to develop a kinetic model that can be used in developing similar programs elsewhere.

2. Snake Lake, Wisonsin USA (70)

The water removal/refill technique has been carried out at Snake Lake, Wisconsin. In the summer of 1970 about three lake volumes of water were pumped and deposited on land above the lake. Nutrient-poor water from contiguous ground water aquifers and precipitation were allowed to replace the removed water. In the fall phosphorus concentration was half that of the previous years. The lake nutrients and other effects continue to be monitored.

Additional dilution and dispersion experiments have been proposed at Moses Lake (71) and Vancouver Lake (72), Washington, and at Lake Bled, Yugoslavia (73).

Covering of Sediments

Covering of bottom sediments with sheeting material (plastic, rubber, etc.) or particulate material (clay, fly ash, etc.) can theoretically perform two functions in restoring eutrophic lakes. First it can prevent the exchange of nutrients from the sediments to the overlying water, and second, it can prevent or retard the establishment of rooted aquatic plants.

One problem encountered when covering sediments is the ballooning of sheeting, or rupturing the seal of particulate material, when gas is produced in the underlying sediments.

For particulate material, the small sizes which have relatively low effective specific gravity (i.e. clays, fly ash) appear to be best suited for sediment covering. Materials of larger size (sand and silts) tend to sink below flocculant sediments. Sands and silts, however, can be effective in areas where the sediments are more consolidated. Materials such as Kaolinite and fly ash, which have a high water soluble lime content, have the added advantage that they will remove phosphate from the water and carry it to the bottom in a relatively insoluble form.

Covering of sediment to improve lake conditions has been done at Marion Millpond, Wisconsin (70). About 12.1 hectares (29.9 acres) of this 44.5 hectare (69.9 acres) lake were physically treated by (a) sand blanketing, (b) scraping of overburden to a sand substrate, and (c) covering the sediments with black plastic sheeting anchored with sand and gravel.

The University of Notre Dame is evaluating fly ash to cover and prevent sediment nutrients from entering the overlying waters (74). This appears to have promise not only as a barrier between sediment and water but also as a material to remove phosphate from the lake during application.

The possible consequences resulting from the application of fly ash to lakes as a sediment covering agent should be thoroughly evaluated prior to application. Fly ash frequently contains numerous impurities including several heavy metals, phosphorus, boron, radioactive wastes and many others. The damage resulting from "treatment" with fly ash could conceivably offset any benefits.

Artificial Destratification and Hypolimnetic Aeration

Possible techniques for altering the water quality in eutrophic lakes include artificial destratification and hypolimnetic aeration. These methods are of particular value in improving the water quality of lakes in which the hypolimnion is void of dissolved oxygen and thus uninhabitable by aerobic organisms. The effective actions of both of these processes are to increase oxygen concentrations in the water, promote the oxidation of reduced organic and inorganic substances and enhance biotic distribution. It must be emphasized, however, that these techniques are palliative in nature, i.e., they will not, in themselves, restore a lake. Aeration techniques generally treat the symptoms of over fertilization rather than the source. Permanent restoration will be accomplished only be removing or significantly reducing the primary nutrient inputs to a lake. Following such a reduction aeration methods may be effective in increasing the rate of recovery.

Artificial destratification of a thermally stratified lake is most often accomplished by injecting air into the water at the deepest point. As the bubbles rise to the surface vertical water currents are generated. The colder and denser bottom water mixes with the warmer surface water, sinks to a level of equal density and spreads out horizontally. Oxygen is added to the water directly from the compressed air as well as by contact with the atmosphere and by photosynthesis of aquatic plants. As the mixing process continues, complete circulation is achieved and the lake approaches an isothermal condition in which the water temperature and dissolved oxygen level are approximately equal from top to bottom. Likewise, with elimination of distinct epilimnion, metalimnion and hypolimnion zones, the whole watermass becomes inhabitable by the biota. The time required to reach this condition depends on the time of year, size of the lake, degree of stratification and method of air injection.

Artificial destratification may also be accomplished by utilizing a mechanical pump to move the bottom water to the surface. Although this technique does effect complete mixing, it does not afford the advantage of oxygenation directly from the air bubbles produced by air diffusion systems.

In contrast to artificial destratification, the process of hypolimnetic aeration does not disrupt the thermal stratification of a lake. The aerator consists of a large diameter pipe which extends from the lake bottom to above the water surface. Water inlet ports are located near the bottom of the pipe and outlet ports are located below the metalimnion. The top of the pipe is open to the atmosphere.

Air is released through a diffusor near the bottom of the pipe. As air rises in the pipe, water is drawn in through the bottom ports. Oxygen diffusion occurs as the water rises to the surface with the air bubbles. At the top of the pipe the air escapes into the atmosphere. The water sinks to the outlet port where it flows back into the hypolimnion. After the establishment of a hydraulic head in the pipe, water flows directly from the inlet to the outlet ports without rising to the surface. Hypolimnetic water, therefore, is aerated but not significantly heated or mixed with epilimnion or metalimnion water. Thus, the dissolved oxygen level of the bottom waters is increased, but the integrity of the thermal strata is maintained, with the warm water of the epilimnion overlying the cold water of the hypolimnion.

Advantages

The benefits of artificial destratification and hypolimnetic aeration are most pronounced on eutrophic lakes which undergo oxygen depletion in the hypolimnion, in contrast to oligotrophic lakes which never become oxygen deficient. The changes in water quality which are induced by these techniques include the following:

1. Due to the increased oxygen levels in the hypolimnion, there is a reduction in the anaerobic release of nutrients from bottom sediments (75). This results in a general decrease in productivity of the body of water.

Also due to higher hypolimnetic oxygen levels, oxidation of reduced organic and inorganic materials occurs in the water (77). This is particularly important when the lake serves as a raw water supply. In such cases, the need for specialized water treatment processes to remove taste and odor carrying materials such as iron and manganese is obviated.

2. The range of benthic populations is extended into areas which were once anaerobic (75). An increase in the number of fish and a shift to more favorable species could result due to the greater availability of food organisms (75, 78).

3. Favorable changes in algal populations occur with a decrease in undesirable blue-green species and an increase

in green algae species (79). This is a result of the continued movement of the algae from the aphotic to the euphotic zones (76), the lowering of water temperature of the epilimnion, and the modificaiton of the nutrient availability. The decrease in blue-green algae could result in a reduction in raw water taste and odor problems. There also appears to be a reduction in actinomycete population which could improve water taste (80).

4. Artificial destratification increases the heat budget of a lake by inducing complete circulation (75). An increased rate of productivity results. This is of particular importance in oligotrophic bodies of water.

5. Artificial destratification reduces evaporation rates by slightly reducing surface temperatures during the summer (81). In areas such as the southwest United States where water is in short supply and is expensive, significant savings can be achieved by reducing the rate of evaporation.

6. Artificial destratification often results in increased water clarity (75). This appears to be associated with reduced algal populations.

7. Winter fish kills may be prevented by artificial destratification due to the maintenance of high oxygen levels under ice (82).

Disadvantages

Problem areas associated with these two methods may include the following phenomena:

1. The increased heat budget produced by artificial destratification may be deleterious to cold water fishes, particularly in shallow lakes in which the temperature is increased excessively at all depths (81). Also, warmer lake waters may reduce a lake's usefulness as a source of cooling water for industry and, if the lake is a public water supply, the attractiveness of drinking water derived from the destratified lake (77).

2. Both artificial destratification and hypolimnetic aeration may increase water turbidity due to the

resuspension of bottom sediments (80). This is often a temporary problem, however, and may be resolved by continued mixing or a change in the location of aerators.

3. In most investigations these methods have produced a reduction in blue-green algae populations with a subsequent increase in green algae such that total productivity remains about the same (76, 83). In other instances there has been no observable effect on blue-green algae populations with the result that problems associated with these organisms have remained (84).

4. If oxygenation is insufficient to increase the hypolimnetic oxygen concentration rapidly enough during destratification, fish kills may occur (85).

5. The artificial destratification procedure may induce foaming, an aesthetically undesirable phenomenon (76).

6. The oxygen demand of resuspended anaerobic mud may result in a decrease in oxygen concentrations to the extent that fish kills occur (77). This is particularly true of small, very eutrophic lakes.

Costs

The costs of applying artificial destratification techniques depend on such considerations as systems design, length of operation, power cost, degree of stratification and oxygen deficit in the lake to be destratified. The findings of a survey of water-utility managers who have applied these techniques in an effort to improve or maintain the water quality of impounded water supplies reveal that although costs vary, certain generalities may be made (86). It was found, for example, that both the initial cost per unit volume and the operating cost per unit volume declined as the volume of the reservoir increased. No clear trend emerged with regard to the costs associated with the type of equipment (homemade or commercial) and the operating schedule used (continuous, continuous all summer, or intermittant). It should be noted that 89 percent of the respondents utilized aeration devices and only 4 percent used mechanical pumps. Other less widely used techniques were employed by the remaining 7 percent The majority of the operators (83%) used electrical power. One third of the respondents employed continuous operation, one third continuous during the summer and one third intermittant operation. The concensus costs of all the survey respondents are presented in Tables 5 and 6. These data combine all types of equipment and operation.

Table 5

Initial Costs Per Unit Volume
(Purchase and Installation)

Maximum	Mean	Minimum
$16.00/1000 m^3 ($60.50/mil gal)	$3.54/1000 m^3 ($13.40/mil gal)	$0.04/1000 m^3 ($0.15/mil gal)
1.6¢/m^3 ($0.051/1000 gal)	$0.003/m^3 ($.013/1000 gal)	$0.0054/m^3
$159.70/ha-m ($19.70/acre-ft)	$35.59/ha-m ($4.39/acre-ft)	$0.41/ha-m ($0.05/acre-ft)

Table 6

Operating Costs Per Unit Volume and Time
(Energy and Maintenance)

Maximum	Mean	Minimum
$3.67/1000 m^3/yr ($13.90/mil gal/yr)	$0.77/1000 m^3/yr ($2.90/mil gal/yr)	-- ($0.01/mil gal/yr)
$0.37/m^3/yr ($0.014/1000 gal/yr)	-- ($0.003/1000 gal/yr)	-- --
$37.46/ha-m/yr ($4.62/acre-ft/yr)	$7.62/ha-m/yr ($0.94/acre-ft/yr)	$0.02/ha-m/yr ($0.003/acre-ft/yr)

These costs represent the actual costs incurred in past applications of aeration devices. It should be noted, however, that cost estimates for future applications must be generated on a case-by-case basis. It will be difficult to determine precise cost estimates, however, as there remains a lack of information on the exact amount of mixing needed to improve water quality in a given circumstance and on how mixing can be maximized with a given power input, thereby minimizing cost so the the highest benefit/cost ratio can be obtained (86).

Case Studies

1. El Capitan Reservoir, California, USA (78, 81, 87)

The El Capitan Reservoir is an impoundment on the San Diego River. This body of water typically experiences one annual period of thermal stratification usually lasting from March or April to November or December. The reservoir was continuously aerated by diffuse air injection during the summers of 1966 and 1967. The chemistry and biology of the lake were investigated during these periods as well as during the summers of 1964 and 1967 when normal stratification was allowed to occur. During the course of the study the depth of the lake rose from 24.8 m in 1964 to 33.3 m in 1967 and the total volume increased from 1,136 ha-m (9,200 acre-ft) in 1964 to 2,698 ha-m (21,845 acre-ft) in 1967.

The total cost of equipment, materials and labor to install the system was approximately $6,010. At 6 percent interest, the 10-year amortization cost will be $825 annually. With continuous operation on a 6-month basis each year, total power consumption was approximately $1,674. Monthly electrical service charges totaled an additional $177. Including the amortization and power costs, plus an estimated $250 per year for maintenance and repair, the estimated annual cost of operating the destratification system on El Capitan Reservoir was $2,926.

Changes in the chemistry and biology of the lake were quite evident following artificial destratification. The lake became isothermal from top to bottom. The heat budget increased. For example, the maximum heat content of the lake in 1966 during destratification was 25,116.0 cal/m^3 above 0.0^{o}C as compared with the maximum of 22,548.4 cal/m^3

above 0.0^{0}C observed during the thermal stratified condition of 1967.

During destratification dissolved oxygen was distributed to all depths and was essentially uniform from top to bottom. It was observed, however, that the surface oxygen concentration of about 5 mg/l found during 1965 was significantly lower than the 8 mg/l which occurred in 1964 under stratified conditions. This indicates that an accelerated oxidation rate may have occurred during forced circulation.

Phosphorus concentrations in the hypolimnion decreased from as high as 1.4 mg/l during stratified conditons to 0.1 - 0.2 mg/l during destratification. During destratification the phosphorus level was uniform from top to bottom.

Prior to destratification the combined concentrations of iron and manganese were 0.65 and 1.46 mg/l at 7 and 17 meters respectively. These values exceed the level of 0.3 mg/l recommended for potable water by the U.S. Public Health Service. Following destratification, the combined concentrations of iron and manganese were below 0.3 mg/l at all depths.

Benthic organisms such as midge larvae and pupae, oligochaete worms, nematode worms and freshwater clams were absent from the hypolimnion prior to destratification. During destratification these organisms invaded the deepest part of the lake and increased in total numbers. Zooplankton populations were also affected by destratification. For example, over 85 percent of the zooplankton were found below 10 meters on June 17, 1965, under destratified conditions, whereas less than 10 percent were observed below this depth the previous year under stratified conditions. Destratification, therefore, allowed a greater depth distribution of these organisms.

Although no data have been reported on the effect of destratification on algal populations in El Capitan Reservoir, the results presented indicate that artificial destratification did produce a significant improvement in water quality in this body of water.

2. Wahnbach Reservoir, Germany (77)

Wahnbach Reservoir is used as a water supply and as a source of industrial cooling water. It contains 4,168 ha-m

(33,740 acre-ft), has a maximum depth of 42.9 m and an average depth of 19.2 m. The lake is rapidly becoming more eutrophic due to the introduction of domestic sewage and agricultural runoff. During periods of thermal stratification, a complete lack of dissolved oxygen exists in the hypolimnion.

The reservoir was aerated by diffused air injection during the summer of 1964. Oxygen was maintained throughout the lake. Unlike previous years, the oxygen content did not decrease to below 30 percent saturation at the mud-water interface at any time during 1964. Compared to previous years without aeration when manganese concentrations of up to 20 mg/l were observed, aeration generally reduced the concentration of manganese throughout the lake to less than 1.0 mg/l. Some increase in dissolved phosphorus levels in the surface water was evident during aeration although this had occurred previously when there was no aeration. No increase in production occurred during the destratified period. A decrease in the population of the blue-green algae <u>Oscillatoria</u> sp. was observed, however.

Although improvements in many aspects of the water quality of Wahnbach Reservoir were produced by artificial destratification, a detrimental effect occurred. Increases in water temperatures rendered the water unsuitable for drinking and for industrial cooling water purposes. To overcome this disadvantage, a system of hypolimnetic aeration was employed to raise oxygen levels without increasing water temperature.

Hypolimnetic aeration of the reservoir was employed from July to November, 1966. Thermal stratification was maintained and the lake became aerobic throughout. Manganese concentrations fell to below 0.1 mg/l. Phosphorus concentrations declined from 80 ug/l prior to aeration to 20 ug/l after aeration. Hypolimnetic aeration, therefore, produced water quality changes similar to artificial destratification without adversely affecting the temperature regime of the lake.

The installation cost for the diffused air injection apparatus was $3,750. This includes $2,500 for the purchase of a 36.5-kw compressor and $1,250 for the air distribution pipe. The operational costs, primarily electrical power costs, for approximately 5 months of continual aeration were $2,250 in 1964. The annual operating cost was $0.15/1000 m^3

($.57/mil gal) of drinking water withdrawn from the reservoir - 14.8 million m^3/yr (3,900/mil gal/yr).

The hypolimnetic aeration equipment required a higher capital investment - $8,250. In addition, $4,500 was required for a raft with an overhead crane used for assembling the apparatus. The compressors and plastic pipe cost $3,750. Total installation cost, therefore, equaled $16,500. The operational costs for hypolimnetic aeration were only slightly less than for artificial destratification.

3. Hemlock Lake, Cheboygan County, Michigan, USA (76)

Hemlock Lake is a 1.8 hectare marl lake having a maximum depth of 18.6 meters. Hypolimnetic aeration was applied to the lake continuously from June 14 to September 7, 1970.

Aeration increased hypolimnetic oxygen levels from zero to over 11 mg/l. The temperature of the hypolimnetic water increased more than 12^{o}C above its normal level. This was due in part to heat conduction through the aeration tower and can be minimized by using insulation.

After an initial increase in phytoplankton cell numbers immediately after initiating aeration (attributed to the leakage of nutrient rich hypolimnetic water through the tower into the epilimnion), the standing crop decreased from over 30,000 cells/ml to less than 500 cells/ml. Concomitantly, Secchi disk measurements increased to over 9 meters, the deepest ever recorded for Hemlock Lake. Aeration did not appear to affect the periphyton standing crop. Following aeration, zooplankton inhabited the lower lake waters and their numbers increased until predation stress by fish caused zooplankton numbers to decline. The total number of zoobenthos was increased by aeration although the biomass remained the same. The zoobenthos were able to inhabit the deep water during aeration as were rainbow trout.

The results of this study indicate that hypolimnetic aeration may be an effective method of alleviating the eutrophic condition of a body of water.

4. Boltz and Falmouth Lakes, Kentucky, USA (79, 88, 89)

Boltz Lake and Falmouth Lake were artificially destratified by diffused air injection during the summer of 1966. Bullock Pen Lake, also in Kentucky, was not destratified and acted as a control. The morphological characteristics of the lakes are given in Table 7.

Table 7

Morphological Characteristics of Bullock Pen, Boltz and Falmouth Lakes

	Volume m^3	Maximum Depth m	Average Depth m	Surface Area hectares
Bullock Pen	3.95×10^6	14.6	7.0	56.8
Boltz	3.58×10^6	18.9	9.1	38.4
Falmouth	5.69×10^6	12.8	6.1	90.0

Intermittant aeration was used. Boltz Lake was destratified four times during the period June to September and Falmouth Lake five times.

In both lakes, the temperature of the bottom water increased during the aeration periods and decreased or leveled off between aeration periods. The net effect of intermittant aeration was to increase the bottom temperatures by over 20°C during the course of the summer. Likewise, hypolimnetic oxygen concentrations increased during the mixing process and decreased between periods of artificial destratification. The net effect through the summer was to increase the oxygen levels of the deeper water and decrease the concentrations of reduced materials such as iron and manganese.

The sum of the ammonia and nitrate nitrogen concentrations at the 1.5 m depth in the unmixed lake remained at about 0.2 - 0.3 mg/l throughout 1966. Boltz Lake exhibited concentrations in about the same range at the 1.5 m depth except for the last mixing period. During this period the concentration of NH_3-N plus NO_3-N increased to between 1.0 and 1.5 mg/l after which it leveled off and subsequently declined to former levels. Smaller increases in the concentration of ammonia and nitrate nitrogen occurred during two of the mixing periods in Falmouth Lake. In each case, the concentration declined after aeration stopped.

The soluble phosphorus concentration at the 1.5 m depth in the unmixed lake varied between 20 and 50 ug/l in 1966. Concentrations at the 1.5 m depth in Boltz Lake, however, exhibited an increase from 5 - 10 ug/l in May to approximately 100 ug/l in September. Falmouth Lake exhibited a net decrease in phosphorus concentration at the 1.5 m level from May to September although the concentrations increased during most of the mixing periods.

The surface plankton counts in the unmixed lake were between 1,000 and 3,000 per ml from June to mid-September. Boltz Lake exhibited declines in plankton counts during three of the four mixing periods and Falmouth Lake during two of the five. When these declines took place they occurred at all depths. In most cases, an increase in plankton counts took place after mixing stopped. These increases were not of "bloom" proportions, however, and despite periodic increases in nitrogen and phosphorus caused by mixing, excessive algal growth never occurred during any of the artificial destratification experiments. Whereas the unmixed lake exhibited the predominance of blue-green algae characteristic of the geographical area, a shift to green algae occurred during several of the mixing periods in both test lakes.

It may be concluded from this study that artificial destratification eliminates thermal stratification, adds dissolved oxygen to the water, causes oxidation of reduced substances and can produce a shift in algal predominance from blue-green to green species. The results also indicate that artificial destratification should be initiated in the spring or early summer and should be continued, at least periodically, throughout the summer for best improvement of water quality.

5. Parvin Lake, Colorado, USA (90)

Parvin Lake was artificially destratified from November, 1969, to December, 1970, in an effort to improve the normal winter hypolimnetic oxygen deficit and summer blue-green algae blooms. Continuous air diffusion was employed. Parvin Lake has a surface area of 19 ha, maximum depth of 10 m and a mean depth of 4.4 m. It is located at an elevation of 2,500 m in the Rocky Mountains of Colorado.

Total phytoplankton abundance decreased in Parvin Lake during artificial destratification, but the decrease was not uniform among all phyla. Green algae declined during destratification. This may have been due to the colder than normal winter water temperatures and warmer than normal summer temperatures. Planktonic diatoms decreased in population size during the winter when they normally dominate. Several blue-green algal species increased in number during the summer over previous, untreated years, namely *Anabaena flos-aquae*, *Aphanizomenon flos-aquae* and *Gomphosphaeria lacustris*.

These results indicate that complete understanding of the response of eutrophic lakes to artificial destratification is lacking. Whereas other investigators observed decreases in blue-green algal populations during mixing periods, this study found that several of these species increased in number. Because of this difference in observed response, it is evident that the potential effects of artificial destratification should be evaluated on a case-by-case basis.

Drawdown

Water level manipulation exists as a potential mechanism for enhancing the quality of certain lakes and reservoirs. Lake drawdown has been investigated as a control measure for submersed rooted aquatic vegetation, as a means to retard nutrient release from the sediment nutrient pool, and as a mechanism for lake deepening through sediment consolidation.

Observations from natural drawdown and subsequent exposure of the bottom sediments have indicated marked improvement in the water quality of two Florida lakes. Before drawdown the lakes produced heavy algal crops. After drawdown and sediment drying, rooted aquatic plants replaced the algal community making the lakes more amenable to game fish.

Experiments with sediments from Lake Apopka, Florida, in 1967-68 showed that when the sediments were dried and reflooded a balance of aquatic weed and shoreline (emergent) vegetation grew (91). Further, the sediments oxidized and would not resuspend upon flooding. It was concluded from these studies that drawdown for 6 to 8 weeks during the dry season should result in a suitable aquatic weed crop.

Drawdown has been carried out in three Wisconsin lakes: Marion Millpond, Snake Lake (70) and Jyme Lake (92). At Marion Millpond many manipulations besides sediment exposure were made: bottom stumps and logs were removed; some sediment was removed and sand and plastic were placed in some of the littoral areas. Therefore, the effects of sediment drying were masked by these other rehabilitation techniques.

At Snake Lake the primary objective was to restore the lake by pumping nutrient-rich water from the lake and allowing it to be replaced with nutrient-poor groundwater. In lowering the water level by 3.35 meters the sediments were exposed to air which resulted in extensive compaction and likely chemical alterations by oxidation. The phosphorus concentration decreased by half after the lake refilled but this likely was mainly attributable to the dilution water.

Jyme Lake is a 0.45 hectare, 3.7 m deep acid-bog seepage lake in Oneida County, Wisconsin. Beginning in October 1971 water was pumped intermittantly for a 10-day period to a nearby low-lying cattail marsh in an effort to drain the lake to allow investigation of sediment consolidation as a lake deepening technique. Attempts to completely drain the lake were unsuccessful due to the flow of low-solids mud and peat on the lake bottom and from beneath the vegetative mat of the bog, and a subsequent subsidence of the level of the bog. The wood fragments in the mud clogged the pump impeller forcing termination of the project prior to winter freeze-up. The Jyme Lake experience indicates that although lake drainage and sediment consolidation is a potential physical deepening technique, to be effective the lake must be completely drained and the water table must be maintained below the surface of the lake sediment surface. Because of possible pumping problems and slumping difficulties encountered during the draining of bog lakes, this technique may be more applicable to lakes with a greater percentage of inorganic sedimentary fill.

In the Tennessee Valley Authority lakes it was observed that lowering the water level 1.83 meters for a period of 21 to 25 days during the winter provided a 90 percent reduction in the acreage infected with *Myriophyllum spicatum* (93).

Studies on drained marsh areas have shown that water removed during the drainage period would carry with it much of its total burden of nutrients (94). It was concluded that frequent drainage could heavily deplete the fertility of marsh environments.

Harvesting Nuisance Organisms

Algae

Algal harvest is particularly difficult because the algae are normally in dilute suspensions and of small physical size. For these reasons most attempts at algal harvest have been conducted on lagoon waste water effluents which have a relatively high concentration of algal cells. Even at these concentrations Oswald and Golueke (95) indicate that in order to obtain a usable, economically feasible end product, the following three steps are necessary: 1) initial concentration of the algal suspension, 2) dewatering and concentrating the resulting slurry, and 3) drying the dewatered algae for storage and handling.

Algal harvest may be accomplished by centrifuging, filtering, coagulation, microstraining, sonic vibration, flotation, and changing of ionic characteristics of the algae with ion exchange resins (96). Coagulation of algae with aluminum sulfate, lime, and alum have been used in combination with the above methods (97).

A high grade end product is most cheaply obtained by centrifugation, whereas a lower grade product is most cheaply obtained by combining centrifugation with the coagulation or flocculation process (96). According to 1967 estimates one metric ton of dry algae (low grade) could be produced at a cost of $66 to $88 ($60 to $80/ton). No firm market value for the finished product could be established in 1968, but it was estimated to be worth about $95 per metric ton ($86/ton) with an additional $10.00 per 3785 m^3 (mil. gal.) of high quality process water as a fringe benefit. Products of the process would then yield an income of nearly $105 per metric ton of dry algae at a production cost of $66 to $88 for a net profit of $17 to $39 per metric ton of dry algae. According to Levin and Barnes (98) a similar quality low grade product may be produced by the froth flotation process for approximately $52 per metric ton ($47 per ton). Assuming the same market value for the end product this method might realize a net profit of nearly $55 per metric ton ($50 per ton). If these figures represent real numbers a municipality or industry might be able to economize significantly on wastewater treatment by harvest of algae.

Algal harvesting studies which utilized the effluent from wastewater lagoons have demonstrated that nearly 90 percent of the nitrogen can be removed in the form of algal protein (99, 100). One field study demonstrated 50 to 70 percent inorganic nitrogen removal and 19 to 68 percent phosphorus removal from wastewater with algal harvest (101). Soluble phosphate has been reduced by 90 percent using high rate algal culture techniques (102).

Under highly favorable climatic conditions up to 70 percent of the nitrogen and 50 percent of the phosphates have been removed from wastewater by algal action alone (103). In laboratory cultures under controlled conditions 50 percent of the total inorganic nitrogen was removed by algae in one day and 95 percent in four days (104).

It has been estimated (95) that .6 million hectares (1.5 million acres) of land devoted to algal culture would satisfy the oxygen demand of all liquid-borne wastes in 1967. The need by the year 1990 is projected to be about 2.42 million hectares (6 million acres). The algae recovered would meet approximately one-quarter of the protein needs of the nation's livestock industry, and since the U. S. has about 121.4 million hectares (300 million acres) devoted to protein production, the savings in water resources normally used to produce protein could amount to $2.5 \times 10^{11} m^3$ (200 million acre-feet) each year (96). This technique should be evaluated with regard to efficient land use practices.

Data regarding algal harvest from lakes are severely lacking, probably owing to the relatively sparse algal populations found in lakes. Levin and Barnes (98) noted that the efficiency of harvest was inversely proportional to culture density. Another probable reason for lack of lake data is that algal bloom populations usually consist of blue-green algae for which there is a limited market. Green algae usually associated with the nutrient-rich wastewater lagoons, on the other hand, are a potentially valuable source of protein.

Despite the apparent success of some methods of algal harvest as a measure of nutrient removal, many problem areas still remain. It appears now that there is little hope of developing an *in situ* lake-oriented method of harvesting algae that would be economically feasible.

Macrophytes and Higher Organisms

Excessive macrophyte growths due to nutrient imbalances in eutrophic situations pose difficult problems in control from practical, economic, and aesthetic standpoints. Most authors (106 - 113) encourage management via periodic mechanical harvesting, rather than eradication, as an ecologically sound control method for macrophytes. Individual species require specially developed methods or harvest because of basic physical differences in growth form. For instance, water hyacinth (Eichornia crassipes), an unrooted, floating plant, allows free flow of water beneath even very large beds of the plant, so that control may be accomplished by subsurface mowing. The harvested parts could be processed for food supplements, such as protein concentrates and pelletized roughage (107, 111). Furthermore, existing industrial and agricultural equipment such as alfalfa pelletizers and citrus pulp processors may be used directly or modified to process the crop (106, 108, 110, 113).

Rooted species, such as water milfoil (Myriophyllum) and pondweed (Potamogeton), severely impede the flow of water and trap large amounts of sediment. Devices have been designed to physically uproot and destroy these plants (107), but both types of plants have economic value if controlled; pondweed as the major duck food plant in the U. S. (114), and milfoil as a feed supplement (115, 116).

Development of specialized cutting machines has progressed to the point where relatively efficient cutting can be accomplished, but the major expense comes in collecting the cut debris and removing it from the water. Various devices for reduction of weight and volume of the crop have been designed, such as screw presses (117, 118, 110), high pressure crusher-rollers (111, 117), brush chippers and crushers (106), as well as assorted efficiency improving pretreatments (108, 117, 110, 111).

From a health related standpoint, especially with reference to food production, Abou-El-Fadl <u>et al</u> (119) observed no infectious stages of helminths (schistosomes) in harvested water hyacinth (this is a severe problem in temperate, tropical, and subtropical countries), but noted that the crop must be composted before use as an organic manure.

Various estimates of nutrient removal efficiency have been made, but there is widespread disagreement. Lee (120) is of the opinion that harvesting, in general, does not make significant inroads in the nutrient balance of the lake, although it does remove certain amounts of nitrogen and phosphorus. Rogers (121), however, points out that 1 hectare of water hyacinth could absorb in 6 months the annual nitrogen and phosphorus wastes of about 550 people. Livermore and Wunderlich (106), cite work (106, 122) that indicates that the harvest of six species of plants in Lake Mendota, and milfoil harvest in Caddo Lake, could yield up to 202 kg/hectare/year (180 lb/acre/yr) of nitrogen and 31.8 kg/hectare/year (28 lb/acre/yr) of phosphorus, which would represent substantial nutrient removal in many lakes. Yount and Grossman (107) indicate that primary production is reduced by harvesting, but only if the intact plants are removed from the area but, "if too much vegetation is removed, the availability of these pollutants to other organisms (such as algae) is increased....the problem is resolved by managing a population on a sustained yield basis." See also (123).

Steward (124) indicates that emergent macrophytes are substantially more productive (in terms of dry weight) than submerged rooted species, and cites the work of the Orange County Pollution Control Department, Orlando, Florida, where the nutrient balance of a small eutrophic lake has been successfully restored by growing water hyacinths in a fenced area in the center of the lake. After one year the lake was clear and supporting fish.

In studies of two full scale treatment plants used in processing citrus pulp waste (113), the performance of water hyacinth in the removal of nutrients in aerated lagoons and oxidation ponds has been evaluated. It was determined that a minimum of 5 days retention time was required to attain substantial nutrient removal, and the hyacinths were most efficient at D.O. concentrations below 0.5 mg/l, and further that the microbiota attached to the roots of the hyacinth were responsible for substantial reduction of BOD (70 percent) and COD (47 percent). A considerable amount of nutrients (contained in the presswater) were released during squeezing in a drying process, i.e., .4 hectare (1 acre) of hyacinths at 336 metric ton/hectare (150 ton/acre), would yield 128.7 m^3 (34,000 gal) of pressed liquor containing 63 mg/l PO_4-P and 335 mg/l total N). Analysis showed an animal feed value of the processed hyacinth comparable to alfalfa hay.

Steward (124) further calculates that water hyacinth has the highest nutrient reduction potential of eight species compared. Taylor, Bates, and Robbins (125) assayed the protein content of water hyacinth finding that, "although the quantity of the protein extracted was low, it appeared to be of good nutritional quality as evidenced by the proportions of essential amino acids." The crude protein concentrate (33.6% recovery by alkali) ranged from a summer low of 4.7 percent (dry weight) to 5.8 percent in winter to 9.2 percent in the spring.

There has been considerable work done in Germany over the past 10 years with the bulrush, _Scirpus lacustris_ L., as a biological filter for use in pond reclamation and sewage treatment (126 - 128). This rush, which has worldwide natural distribution, can grow in an astonishing variety of situations, including saline water and highly contaminated freshwater. _Scirpus_ has been shown to have the ability to penetrate and break chemically precipitated hardpans in holding ponds, allowing percolation to the ground water. Before the introduction of the rush, the water stagnated. _Scirpus_, by virtue of a root exudate termed a "phytondice", is able to lyse (kill) common sewage bacteria (_E. coli_, _Salmonellae_, etc.) completely, rectify the pH of the entering sewage effluent to 7 ± 0.5, and is capable of removing large amounts of organic and inorganic nutrients, storing these nutrients in its "phyllosphere" or leaf blade which is harvested periodically and utilized in a number of ways, e.g. fuel, cattle feed roughage, paperboard fiber. In pilot plant operations, flow-through channels of Scirpus have shown the ability to reduce BOD by 96 percent, often to less than 5 mg/l, phosphate by 50+ percent, and ammonia by more than 99 percent (22 mg/l to 0.1 mg/l). The process improves the activated sludge process, is capable of 80 percent reduction of total nitrogen, and its metabolism is reduced by only 40 percent under ice cover. The designs are suitable for small cities in the 20,000 to 40,000 population range. There are an increasing number of installations in European countries, treating both domestic and industrial effluents. Steward (124) reports that an industrial installation in Germany treats 5 million cubic meters of effluent per day by passage through 20 basins, 400 meters long by 50 meters wide, planted with Scirpus.

The other area of possible interest in nutrient removal is that of vegetation consumers, such as fish and shellfish. Some research seems promising. Greer and Ziebell (129) tested various fish and shellfish, and found the oriental clam, Corbicula fluminea was most effective; at concentrations of 5.0, 10.0, and 15.0 mg/l PO_4, this system can remove the PO_4 ion to below 0.30 mg/l in 16 days or less, yielding a clear effluent. This process occurs partly by sedimentation of psuedo-feces (mucous bound undigested pellets of algae) which are not resuspended. X-ray diffraction of sediments showed that PO_4 had been precipitated in the form of hydroxyl-apatite. They concluded from studies with Tilapia and channel catfish, that where algal blooms could be controlled, removal of nutrients via sport fishery could be feasible (algal blooms generally result in massive fish kills).

Corey et al (129) estimated that fish harvest, on a sustained yield basis, would result in catches of 337 kg per hectare (300 lb/acre) of water surface annually (sport fishing about 1/3 of the total), which would represent removal of about 7.8 kg of nitrogen and 0.67 kg of phosphorus per hectare (7 lb N/acre and 0.6 lb P/acre).

There has been recent work in this country concerning the use of Ctenopharyngodon idella Val., the white amur or grass carp, in controlling aquatic plants. Claims have been made that experiments in Arkansas have proven the white amur to be one of the best control agents for aquatic vegetation (130).

The state of Arkansas has released the white amur into a number of waterways including some which will provide the fish access to the Mississippi River Basin tributaries. The neighboring states of Texas and Missouri, however, have banned the importation of the grass carp.

Results from studies in Europe and Asia on the use of this fish for weed control purposes are less encouraging than those from the Arkansas studies. Opuszynski (131) reported that grass carp fry eat only animal food such as zooplankton and Chironomidae larvae until they reach a weight of 1.8 g and a length of 36-43 mm, and that the use of macrophytes in their diets increases with increases in size. It was also reported that animal protein apparently is a necessary addition to the diet for normal growth and development of these fish (131). When given a choice the grass carp seemed to prefer macrophytes to algae. According to the Sport Fishing Institute Bulletin (132) a recent release from the Missouri Department of Conservation provides preliminary evidence that the white amur preferred amphipods over weeds when given a choice, and ate weeds only in the tank deprived of amphipods.

In summary, it appears that although the technical methods for nutrient removal via harvesting are becoming increasingly diverse and sophisticated, none is economically feasible on self-supporting basis, although the costs appear to be within reason for some situations. Increased research indicates that markets will develop for products created by these harvesting procedures. There is substantial agreement among authors that complete eradication of any species of plant is undesirable, and management, especially by biological means, is the ultimate goal.

Biological Control of Nuisance Organisms

Algae

Hasler and Jones (133) reported that dense growths of aquatic macrophytes were inhibitory to the growth of phytoplankton, both by direct competition for nutrients and by shading. On the other hand, Mulligan (114) reported that the technique (used primarily by fish culturists) of massive fertilization stimulates blooms of green algae, which shade submerged macrophytes and prevent their development. Neither technique really solves anything, exchanging one problem for another.

Porter (134) reported on the effect of grazing by *Daphnia* and related zooplankton on natural phytoplankton populations in a mesotrophic kettle lake experimental *in situ* set up. Selective reduction and significant suppression of numbers of phytoplankton are accomplished by *Daphnia*.

Mattox, Stewart, and Floyd (135) reported the presence of virus particles in four genera of Ulotrichalean (green) algae (viruses were previously unknown in eukaryotic algae). These findings greatly increase the chances of developing viral control procedures for green algae. Safferman and Morris (136) reported the first isolation of blue-green algal virus, which was found to be highly specific for several closely related blue-green algae, *Lyngbya*, *Plectonema*, and *Phormidium*, all of which are now classified by Drouet as *Schizothrix calcicola*.

A significant amount of work has been done to develop the virus as a control measure (137, 138) in order to take advantage of the observed natural phenomenon of abrupt massive die-offs of blue-green algal blooms caused by viruses.

Broad-spectrum control of blue-green algae is also exhibited by a bacterium, *Myxobacter* sp., as reported by Shilo (138). Bacteria and fungi apparently hold some promise as control agents, but much work, as with viruses, remains to be done.

Macrophytes

Cappelman (139) has developed a culture technique for detached water hyacinth leaves that has allowed demonstration of the pathogenicity of *Alternaria* sp., an aquatic fungus, to the hyacinth. The system holds considerable promise for the demonstration of pathogenicity of other organisms to the hyacinth, including viruses and insects. Sculthorpe (140) reports African work on the sedge, *Cyperus rotundus*, noting that the planting of *Eucalyptus* trees nearby reduces the growth of the sedges.

Also, work has been done in Italy on the control of the grass *Echinochloa crus-galli* by the smut fungus, *Sorosporium bullatum*.

The water hyacinth weevil, *Neochetina bruchi* Hustache, showing promise as a control agent, has been introduced to numerous waterways in Florida by the Army Corps of Engineers (141) after extensive research in South America. Coulson (142) reporting on the original research and future plans for arthropod control agents, believes that significant control of the hyacinth will result with widespread distribution. One species of mite, *Orthogalumna terebrantis* Wallwork, apparently introduced with the hyacinth, also shows considerable control activity. Work is being conducted in Uruguay on the crambine moth *Acigona intusella* (Walkor), whose larvae are stem borers and although they may feed on sugar cane and rice, are only able to complete their development on water hyacinth (*Eichornia*) or *Pontederia* (a closely related genus).

The nymph of an acridid grasshopper, *Cornops aquaticum* Bruner, has also shown considerable ability to defoliate water hyacinth.

The developmental work on these arthropods is being done at the ARS Laboratory in Albany, California, and world-wide under PL480 funds. Thrips, previously introduced, have not successfully controlled the hyacinth.

During the period 1962 to 1967 work was concentrated on the flea beetle, *Agasicles*, which is most specific for the alligator weed, *Alternanthera philoxeroides*. Its introduction into problem areas has resulted in somewhat successful control (114, 142, 143).

Baloch, Khan, and Ghani (144) reported on the isolation and study of four insects which feed on water milfoil (*Myriophyllum* spp.) in Pakistan, and their possible use as control agents. Frequent fluctuations in water levels prevent the buildup of large enough populations of these curculinoids to significantly affect the milfoil, but if such fluctuations were prevented, the introduction of these insects, which damage both seed-bearing capacity and submerged parts of the plant, might result in effective control.

Attempts have been made (114, 140, 145) to take advantage of the accidental introduction of the snail, Marisa cornuarietis L., for the biological control of aquatic weeds. Although they are voracious consumers of aquatic weeds, they tend to disperse rather than build up dense populations, and have not been numerous enough to significantly affect the standing crop. One approach suggested is to confine them to small lakes, rather than the canals through which they have spread.

Potamogeton sp. is the primary food plant of numerous wild fowl, such as ducks, but these birds apparently do not have significant impact on aquatic weed populations.

Only one mammal, the manatee, Trichechus manatus latirostris, has been experimentally considered for control purposes (140, 146), but despite the fact that it consumes tremendous quantities of aquatic weeds, it is a rare animal, difficult to locate, catch, and transport, has not bred in captivity or in freshwater, and simply does not exist in large enough numbers to have a significant impact in terms of overall programs. In individual experiments, although expensive to conduct, the manatee has proven to be a very efficient weed control agent.

Chemical Control of Nuisance Organisms

Algae

Fitzgerald (147) has compiled an excellent review of algicides, especially as they apply to lake management. Although the most desired method of alleviating eutrophication is to restrict nutrient input, many situations have deteriorated to the point where direct measures must be taken to control algal growth, and one of these is the use of algicidal chemicals.

Copper sulfate is probably the most widely used chemical against taste and odor causing algae, floating blue-green algae and filter clogging algae. Over 11,000 metric tons of copper sulfate are used for this purpose per year (147) at concentrations ranging from less than 0.5 mg/l to more than 10 mg/l, according to the density of algae and relative water quality. Application methods vary from spraying from a boat, or dragging a sack of crystals behind a skiff, to aerial systems including helicopters.

Test tube experiments with potassium permanganate have shown it to be more toxic to certain algal species than copper sulfate (148). Because potassium permanganate not only kills algae, but also eliminates tastes and odors and removes iron and manganese sulfates, it may find usage in the treatment of raw water reservoirs (148). Although organic mercurial algicides are potent and very effective, they are more hazardous in the long term to higher organisms in the food chain, including man, and must be used with extreme care. Other algicides of some use are the resin amines, triazine derivatives (such as simazine), a mixture of copper sulfate and silver nitrate, and ammonium compounds. Since the resin amines and copper are toxic to fish, they must be used with caution. Simazine, which has a relatively low mammalian toxicity (114), controls planktonic and filamentous algae through inhibition of the Hill reaction. Although it does not appear toxic to zooplankton and fish at recommended levels, it is taken up and concentrated in fish tissues. Mulligan (114) reports that a 30:5 weight ratio of copper sulfate and silver nitrate has been effective in Czechoslovakia.

Macrophytes

Timmons (149) and Mulligan (114) have reviewed the means of chemical control thoroughly. The following herbicides are the most widely utilized presently: 2, 4-D and other phenoxy compounds, dalapon (2, 2 dichloroproprionic acid), diquat (6, 7-dihydrodipyrido (1, 2-a:2;1'-c) pyrozinedium salts), paraquat (1, 1'-dimethyl-4, 4'- bipyridinium salts), acrolein, xylene, dichlobenil (2, 6-dichlorobenzonitrile), and diuron (3-(3,4-dichlorophenyl)-1, 1-dimethylurea).

Diquat, paraquat, and dalapon are the most widely used throughout Europe; but diuron is used almost exclusively for control of aquatic and bank weeds in the Netherlands, 3-amino-s-triazole (amitrole) and 3-amino-s-triazole + ammonium thiocyanate (amitrole-T) are the most widely used herbicides for aquatic and bank weeds in Australia. These latter compounds are restricted or banned in the U. S. (149). Diquat, paraquat and dalapon are safe for fish; dalapon and diuron are safe for humans and livestock.

Mulligan (114) indicates that esters of 2, 4-D are much more effective in killing aquatic plants than amides of 2, 4-D, although there is much controversy over 2, 4-D residues accumulating in food organisms such as shellfish. 2, 4-D is reported to be photo-oxidizable and can be broken down by soil microorganisms in 4-6 weeks to humic acids. The butoxyethanol ester of 2, 4-D was used to control Myriophyllum in the TVA Lakes in 1967 and Trapa natans (water chestnut) in the Hudson and Mohawk Rivers. Application of 2, 4-D usually results in temporary increases in heterotrophic populations in the waters (114).

Silvex is a non-selective, slow acting herbicide which remains in the water up to 5 weeks. Different formulations have differing toxicities to food chain organisms, the least toxic of which is the potassium salt (114)

Fenac (2, 3, 6-trichlcrophenylacetic acid) is a persistent non-selective agent, reportedly of low food chain toxicity (114).

Endothal (3, 6 Endoxohexahydropthalic acid) is used to control submergent plants, sometimes in combination with silvex. There is substantial concern over its unknown mode of action and unpredictable toxicity to fish and other food chain organisms.

Diquat-bromide (1, 1-ethylene-2, 3-dipyridylium dibromide) kills submerged plants on contact, has relatively low toxicity, and can be removed from the water by adsorption onto clay particles and subsequent sedimentation.

In summary, the following observations should be made: inadequate information exists concerning algicide and herbicide residues, breakdown rates, and long-term effects on other organisms; when plants are killed chemically, oxygen levels quickly decline, often to levels toxic to other organisms; plant-bound nutrients are released into the water; chemical agents offer only temporary, symptom suppressing relief; treatments must be repeated frequently, often semiannually or more; chemical kills of macrophytes are frequently followed by massive algal blooms; and some herbicides, such as endothal and silvex, may damage crops if the water is subsequently used for irrigation.

CONTROL AND REMOVAL OF HAZARDOUS MATERIALS

Contamination of lakes with various hazardous substances is an everpresent threat. Industrial accidents, spills occurring during transport, intentional dumping or plain carelessness may result in the release of a variety of toxic or noxious substances to the environment, with subsequent transport to lakes and reservoirs.

The initial effort in combating the problems of lake contamination with hazardous substances must be the establishment of sound preventive measures through the cooperative efforts of the public, industry and government. Prevention, in order to be effective, must be a requirement of law, with appropriate controls and guidance imposed by the various levels of government. Secondly, industry must meet its moral and legal commitments to society and the environment by implementing appropriate precautionary measures including proper inhouse plant design, adequate safeguard mechanisms and procedures, and conscientious management policies and operational practices.

Precautionary Measures

Even with the best preventive methods in effect, accidental and deliberate contamination of lakes with hazardous substances will occur. A line of defense geared for an immediate response to spilled substances is essential if major catastrophies are to be averted. The essential components of the response mechanism include capabilities for the containment or confinement of the spilled substances and removal or inplace treatment (inactivation) while the material is concentrated in a localized area.

Containment of spilled materials will not always be possible, even though an efficient spill response system is in operation. A percentage of the spills will not be detected or reported until after the spilled material has dispersed throughout the lake. Also, continuous or intermittent discharges of toxic or other hazardous materials over a period of time may cause lake-wide contamination which precludes the use of containment devices.

Decontamination

In lakes where widespread contamination has occurred and ecological damage has resulted, restoration programs will have to be initiated once the source of contamination has been curtailed. If ecological damage has been severe and the contaminating material is present in the lake in sufficient quantities to impede the natural recolonization of disturbed ecosystems or to hinder artificial propagation efforts, a procedure for treating or removing the contaminant must be implemented.

Few *in situ* techniques for removing or treating hazardous materials in lakes have proved effective. Since many toxic substances such as heavy metals and pesticides are readily sorbed onto particulate matter, incorporation of the material into the sediments and biota occurs very rapidly. Consequently, such schemes as flushing, dilution and filtering of lake water do not necessarily remove the contaminant from the system, as the materials are still available for recycling from the sediment and biological reservoirs.

Lambou (150) summarized existing experiences and approaches which have been considered for dealing with mercury contamination in aquatic systems. Since mercury is one of the most hazardous of the heavy metals in the environment, due to its tendency to be biologically methylated, procedures which are effective in restoring mercury contaminated lakes may possibly be applied to lakes contaminated by other toxic substances. The following is taken from Lambou [150]:

> "The continuing supply of mercury from bottom sediments to the water and the slow rates of excretion of mercury by fish give little hope for quick improvement in levels of mercury residue in fish. The Swedish experience confirms this. In Sweden mercury in pike in most lakes has dropped little if at all since mercury bans became effective in early 1966. These lakes where the fish residues have not dropped tend to be biologically poor and acid. Only about three lakes apparently have had mercury levels in pike drop to a demonstrable extent. Rivers have a better chance due to continual flushing action.
>
> "Jernelov (1969) [151] calculated that it would take from 10 to 100 years for the methylation process to remove the mercury from the bottom of lakes. These calculations were based on the yield over a period lasting from 1 week to 2 months of mono and dimethylmercury from bottom sediments taken from contaminated lakes and rivers and kept under natural conditions. In Minamata Bay, Japan, once the cause of the pollution was determined and eliminated, mercury levels in shellfish dropped from 35 ppm to 10 ppm over a two year period and remained constant for at least a five year period (Trukayama, 1966) [152]. Rivers should have a better chance of being decontaminated because of the flushing action of currents moving sediments downstream. Mercury levels of salmon placed in cages below former sources of mercury in some Swedish rivers showed considerable improvement within 3 years (Study Group on Mercury Hazards, 1970) [153].
>
> "Swedish workers have considered the following approaches to the decontamination of

mercury contaminated waterways: (1) introduce oxygen-consuming materials to create continuous anaerobic conditions in the sediments, thereby reducing methylation, (2) increase the pH of the sediments to favor dimethylation and increased volatilization, (3) cover the sediments with fresh finely divided materials with high adsorptive affinity (e.g., quartz and silicates), (4) cover the sediments with inorganic inert materials of any type, i.e., bury them, and (5) remove mercury-bearing sediments by dredging or pumping (Study Group on Mercury Hazards, 1970) [153]. The first two approaches appear to be impractical, however Sweden is evaluating the other approaches (Study Group on Mercury Hazards, 1970) [153].

"Experiments have been conducted in Sweden to evaluate covering sediments by layers of inorganic sediment of varying thicknesses (0-20 cm), with and without Tubificidae (oligochaete worms) and Anodonta (a bivalve) (Study Group on Mercury Hazards, 1970) [153]. These studies have revealed that: (1) in the absence of Tubificidae, methylmercury accumulated in fish only when the sediments were uncovered, (2) in the presence of large populations of these worms, fish accumulated methylmercury when the covering layer was less than 2 cm, and (3) in the presence of Anodonta, which stirs the sediments, leakage of methylmercury occurred if the covering layer was less than 9 cm.

"Swedish workers have conducted tests to evaluate the effectiveness of ground silicate, on the uptake of mercury by fish from sediments contaminated with metallic mercury, ionic mercury, and phenylmercury (Study Group on Mercury Hazards, 1970) [153]. These tests have revealed that there was no reduction in uptake when the pollutant was phenylmercury; however, a decrease in uptake by a factor of two occurred when inorganic mercury was the pollutant.

> "The removal of mercury contaminated sediments by dredging appears to have some serious shortcomings. For one thing, the cost to dredge any extensive area may be excessive. The dredging of a Finnish port increased the soluble mercury concentration in the water from a level of 0.5 to approximately 10 ug/l (Stephan, 1971) [154]. This increase took 'some weeks' to reach a peak; however, it returned to background in a 'few more weeks' (Stephan, 1971) [154]. Swedish workers were of the opinion that by dredging there was a considerable risk of increasing the rate of methylation of mercury in the sediments (Stephan, 1971) [154]. Measurements taken on sludges dredged from mercury sludge banks in Sweden indicated that while some 95 percent of the suspended solids can be retained in the sludge, only 50-60 percent of the mercury will remain in the sludge, the remaining 40-45 percent being discharged with the supernatant (Stephan, 1971 [154]."

Additional information on the effects of sand and gravel overlays on the release rates of mercury from mercury enriched sediments is summarized from Bonger and Khattak [155] as follows:

It was found in laboratory studies that overburden layers of sand or gravel 6 cm thick prevented the release of mercury from the underlying enriched sediments. Layers less than 6.0 cm thick were less effective in preventing mercury loss from the sediments. Little differences were observed in the rate of release from organic or inorganic sediments. It was noted that Tubificidae worms when present in the sediments in large numbers apparently were responsible for the vertical transfer of mercury. This suggests that additional coverage of mercury enriched sediments may be required in areas where sludgeworm activity is high.

Although field tests were not conducted the approximate cost of applying this abatement procedure in a representative field situation were calculated. The area selected for economic analysis was the Trenton Channel of the Detroit River near Wyandotte. Cost estimates for treating .8, 10.1 and 20.2 hectares of mercury contaminated sediment with 7.6 cm of sand overlay are listed in Tables 8 and 9. This cost evaluation is preliminary, and such site-dependent factors as local transportation, sediment characteristics, topography of the area, water currents and depth, weather conditions and the availability of labor, materials and hardware would affect the actual costs.

Table 8

Estimated fixed/variable costs of distributing sand in an area south of Wyandotte. 1/

Fixed Costs ($):	
Spreading Equipment System (i.e. swivel piler, conveyor, clam shell, fixtures, hopper, etc.)	20,000.00
Variable Costs:	
Sand, dockside, per cubic meter	2.94
Tug boat and crew, per 12-hour day	1,900.00
Deck scow, 612 to 765 m^3 (500 - 1000 yd) capacity per day	100.00
Equipment barge, per day	30.00
Labor, per day (2)	80.00
Equipment maintenance, per day	10.00

1/ Source: Bonger and Khattak (155)

Table 9

Estimate of the cost involved in the application of 7.6 cm of sand to .8, 10.1 and 20.2 hectares of sediment contaminated with mercury 1/.

Hectares (Acres)	0.8 (2)	10.1 (25)	20.2 (50)
Fixed Costs ($)	20,000	20,000	20,000
Variable Cost ($):			
Sand	1,670	20,800	41,600
Tug Rental	1,900	24,700	47,500
Scow Rental	100	1,300	2,500
Barge	30	390	750
Labor	80	1,040	2,000
Maintenance	10	130	250
S/Total	3,790	48,360	94,600
Number of Days	1	13	25
m^3 of Sand	566	7,037	14,145
(Yards of Sand)	(740)	(9,250)	(18,500)

1/ From: Bonger and Khattak (155)

Suggs, Petersen and Middlebrook (156) conducted laboratory investigations of the effectiveness of several agents in removing mercury from the water column and the underlying sediments. It was found that both elemental sulfur and thio-organic compounds dispersed in recoverable materials were capable of removing mercury. However, elemental sulfur coated on a cotton meshwork was found to be most effective, particularly in anaerobic sediments. It was also found that the rate of removal of metallic mercury with elemental sulfur was proportional to the surface area of the "mercury getter".

Other mercury getters investigated were polyvinyl alcohol gel systems, paraffin, sulfur dispersed in paraffin, sulfur tablets, cotton and paper, plastics, paraffin-thiourea, polyvinyl alcohol-cystene and iron oxides. Of these only the polyvinyl alcohol gel systems containing sulfur or phenyl thiourea were found to be effective in removing mercury from contaminated water and sediments, but were not considered applicable where sediment contamination levels were below 25 to 50 mg/l. Cost for actual application to field situations were not provided, but a research-demonstration test plan has been proposed.

When a spill of hazardous substances occurs, the contingency plan of the appropriate agencies must be implemented immediately, frequently under adverse conditions. Such an event occurred in Pond Lick Reservoir, Ohio, in 1971. The following summary of that experience is condensed from reports prepared by Ryckman, Edgerly, Tomlinson and Associates, Inc. (157) and by Nye (158) of the Ohio Department of Natural Resources.

The Pond Lick Lake Incident - A Case Study

On June 2, 1971, Pond Lick Reservoir (known locally as Shawnee Lake) near Portsmouth, Ohio was maliciously poisoned with about 4.54 liters of an endrin solution mixed with strychnine treated corn. Pond Lick Lake is about 300 meters long and approximately 75 meters wide at its widest point, with maximum and average depths of 12 and 4.5 meters, respectively. Fortunately at the time of the poisoning, the lake was thermally stratified, thus restricting the toxic substances primarily to the epilimnion.

The effects of the poison were immediately apparent. The entire fish population was destroyed, and the only aquatic vertebrates surviving were tadpoles which were apparently unaffected by the pesticide.

Pond Lick Lake discharges to the Ohio River via Pond Lick Creek and Turkey Creek. The total distance separating the lake from the Ohio River is less than 16 kilometers. Cincinnati, on the Ohio River about 160 kilometers below the confluence point, was vitally concerned about its water supply.

As a means of containing the pesticide within the lake, the spillway was sandbagged and an earthen dam was built upstream on the inflowing Pond Lick Creek. The creek was then diverted around the lake via a 25.4 cm aluminum pipe and two 13,620 liters per minute pumps. Bags of activated carbon were added to the spillway to remove the pesticide seeping through, and the seepage was pumped back into the lake.

At the time the spill was discovered endrin concentrations of 9 mg/l were present in the epilimnion waters with lower concentrations below the thermocline. Strychnine was not detectable in the lake.

Since endrin is extremely toxic, even in concentrations as low as 0.2 mg/l, is highly stable and can be concentrated biologically by factors of 10,000, it was imperative that essentially all the endrin be removed as rapidly as possible. A heavy rain would overload the by-pass system releasing the contaminated lake water to the receiving stream.

Suggestions considered for resolving the problem were as follows:

1. Dilution
2. Spray irrigation
3. Adsorption - bentonite, fly ash
4. Biological removal
    a. Sewage
    b. Fish
5. Chemical treatment
    a. Cracking
    b. Oxidation with ozone
6. Adsorption and filtration through activated carbon.
7. Physical removal by use of tank trucks.
8. Filter through alfalfa hay.

Most of the suggestions were discarded as impractical or ineffective.

An initial attempt was made to reduce the concentration of endrin by broadcasting approximately 3,178 kg of 40 mesh activated granular charcoal over the lake. This proved to be ineffective as there was insufficient contact time before the charcoal settled out.

A pilot plant was next constructed of a 45.7 cm diameter pipe, 2.4 meters high and filled with activated charcoal. Water from the lake was run into the bottom and out the top of the cylinder. This system proved to be very effective as the endrin concentration of lake water which passed through the column was reduced to near zero.

A large treatment plant was then designed based upon the success of the pilot plant. A channel filter was constructed consisting of a 122x244x549 cm wooden box containing gravel and a 1.8-meter deep charcoal bed. Water was pumped through the bottom. The filter was effective but slow, with a flowthrough rate of about 304 liters per minute. It was discovered that underground springs were feeding the lake faster than it could be filtered, so an additional charcoal filter was constructed in the spillway outlet, with the filtered water diverted into a stilling flume and retained until analysis indicated that endrin concentrations were below 0.1 mg/l.

Analysis of the lake sediments indicated that endrin was being adsorbed by sedimented organics. Since the lake level was not being decreased as rapidly as desired, another filtering device, constructed to hold 166 bales of hay, was designed based upon another pilot plant study. This system could handle approximately 5.7 m^3 per minute with no evidence of endrin detectable in the discharge. When the lake was eventually drained, endrin concentrations in the sediments along the bank were approximately 100 mg/kg. Concentrations in the lake bottom sediments were much lower.

The bottom and sides of the lake were cleaned and scraped, and the spoils disposed of in a prepared area outside the watershed. Approximately 4,940 cubic meters of sediment were distributed over a .8 hectare spoil area to a depth of 45.7 cm and mixed with clay. Three months after the poisoning event, the lake was fertilized, the banks reseeded, the lake refilled and fish restocked. Total estimated costs were $100,000.

The Pond Lick Lake incident serves to demonstrate that in the event of a hazardous substance spill, no matter how hopeless the case appears to be, a possible solution may exist. In the Pond Lick Lake case the cooperative effort of Federal, State and county governments and various local agencies, private consultants and industries provided a solution to the problem.

The control of hazardous substances in the aquatic environment has been the target of efforts by industry, universities and governments. The various aspects of the problems are discussed in the Proceedings of the 1972 National Conference on Control of Hazardous Materials Spills (159).

POSSIBLE LAKE PROTECTION MANAGEMENT CONSIDERATIONS

Several state and local governments have established statutes dealing with various aspects of lake management and rehabilitation as a means of protecting inland lake environments. Kusler (160) has summarized the state and local statutes which establish preventive or remedial programs, lists applicable statutes and sets out examples of representative statutes. Kusler (160) points out that explicit statutes authorizing specific state or local programs for lake protection, management and rehabilitation are rare, and that protection and management estimates are often badly fragmented among several state agencies and local units of government. This fragmentation of efforts coupled with high costs and lack of technical expertise have discouraged comprehensive lake protection, management and rehabilitation efforts (160).

Problems relating to lake shore development regulations, shoreland management including economic impacts of artificial lake development and legal problems of property owners associations are addressed in Various Inland Lake Renewal and Shoreland Management Demonstration Project Reports (161 - 165).

Section V

REFERENCES

1. Greeson, P. E. 1969. Lake Eutrophication - A natural process. Water Res. Bull. 5:16-30.

2. Pomeroy, L. R., E. E. Smith and C. M. Grant. 1965. The exchange of phosphate between estuarine water and sediments. Limnol. and Oceanogr. 10:167-173.

3. Hayes, F. R. and J. E. Phillips. 1958. Lake Water and Sediment. IV. Radiophosphorus equilibrium with mud, plants, and bacteria under oxidized and reduced conditions. Limnol. and Oceanogr. 3: 459-475.

4. Sackett, W. G., A. J. Patten and C. W. Brown. 1908. The solvent action of soil bacteria upon the insoluble phosphates of raw bone meal and natural rock phosphate. Zentralblatt f. Backteriol. 2:688-703.

5. Sperber, J. I. 1958. Solution of apatite by soil microorganisms producing organic acid. Aust. J. Agr. Res. 9:778-781.

6. Johnston, H. W. 1959. The solubilization of "insoluble" phosphates. V. The action of some organic acids on iron and aluminum phosphates. New Zealand. J. Of. Sci. 2:215-218.

7. Wang, T. S., S. Y. Cheng and H. Tung. 1967. Extraction and analysis of soil organic acids. Soil. Sci. 103:360-366.

8. Sperber, J. A. 1958. Release of phosphate from soil minerals by hydrogen sulfide. Nature 181:934.

9. Hayes, F. R. and C. C. Coffin. 1951. Radioactive phosphorous exchange of lake nutrients. Endeavour 10:78.

10. Mackenthun, K. M. and W. M. Ingram. 1967. Biological associated problems in fresh water environments: their identification, investigation, and control. U.S. Dept. Interior, FWPCA, U.S. Government Printing Office 0203402.

11. Postgate, J. R. and L. L. Campbell. 1966. classification of desulfovibrio species, the non-sporulating sulfate-reducing bacteria. Bacteriol. Rev. 30:732-738.

12. Postgate, J. R. 1970. Nitrogen fixation by sporulating sulfate reducing bacteria including rumen strains. J. Gen. Microbiol 63:137-139.

13. LeGall, J., S. C. Senez et F. Pinchinoty. 1959. Fixation de l'azote par les bacteries sulfato-reductrices et caracterisation de souches actives. Ann. Inst. Pasteur (Paris) 96:223-230.

14. Guarraia, L. J. and R. K. Ballentine. 1972. Influences of microbial populations on aquatic nutrient cycles and some engineering aspects. U.S. EPA T.S. 00-72-06.

15. Sisler, F. D. and C. E. Zobell. 1951. Nitrogen fixation by sulfate-reducing bacteria indicated by nitrogen-argon ratios. Science 113:511-512.

16. Burris, R. H. 1966. Biological nitrogen fixation. Annu. Rev. Plant Physiol. 17:155-184.

17. Goering, J. J. and J. C. Nees. 1964. Nitrogen fixation in two Wisconsin lakes. Limnol. and Oceanogr. 9:530-539.

18. Vollenweider, R. A. 1963. Studi sulla situazone atterole del regime chimico e biolgico del Lago d'Orta. Inst. Ital. Hydrolbiol. 16:121-126.

19. Ohle, W. 1962. Der Stoffhauhalt der seen als Grundlage einer allgmeiner Stoffwechseldynamik der Gewasser. Kieler Merresferschg. 18:107-120.

20. Organic matter in Natural Waters. 1970. D. W. Hood, ed. Institute of Marine Science Ossasional Publ. No. 1. University of Alaska.

21. Marine Food Chains. 1970. J. H. Steele, ed. University of California Press.

22. Riley, G. A. 1963. Organic aggregates in sea water and the dynamics of their formation and utilization. Limnol. and Oceanogr. 8:372-381.

23. Kuznetsov, S. E. 1968. Recent studies on the role of microorganisms in the cycling of substance in lakes. Limnol. and Oceanogr. 13:211-224.

24. Seki, H. 1968. Relation between production and mineralization of organic matter in Aburatsubo Inlet, Japan. J. Fish. Res. B. Can. 25:625-637.

25. Hugh, R. 1970. A practical approach to the identification of certain non-fermentative Gram-negative rods encountered in clinical specimens. J. Confr. Publ. Health. Lab. Dir. 28:168-187.

26. Seki, H., J. Skelding and T. R. Parsons. 1968. Observations on the decomposition of a marine sediment. Limnol. and Oceanogr. 13:440-447.

27. Hendricks, C. W. 1971. Enteric bacterial metabolism of stream sediment eluates. Can. J. Microbiol. 17:551-556.

28. Guillard, R. R. L. and J. H. Ryther. 1962. Studies on marine planktonic diatoms. I. Cyclotella nana (Hustedt) and Datonula confervacea (Cleve) Gran. Can. J. Microbiol. 8:229-239.

29. Burckholder, P. R. and L. M. Burckholder. 1956. Vitamin B12 in suspended solids and marsh muds collected along the coast of Georgia. Limnol. and Oceanogr. 1:202-208.

30. Burckholder, P. R. and L. M. Burckholder. 1958. Studies on B vitamins in relation to productivity of Bahia fosforescenti, Puerto Rico. Bull. Mar. Sci. Gulf and Carribbean 8:201-113.

31. Kurata, A. and M. Kimata. 1968. Studies on marine bacteria producing vitamin B12. I. On the distribution of marine bacteria producing Vitamin B12 and the vitamin production of them. Res. Inst. Sci. Kyoto Univ. 31:26-34.

32. Seki, H. 1964. Studies on microbial participation of food cycle in the sea. Inter. J. Oceanog. Soc. (Japan) 20:122-134.

33. Convery, J. J. 1970. Treatment Techniques for removing phosphorus from municipal wastewaters. Water Pollut. Contr. Res. Ser. No. 17010-01/70. U.S. Government Printing Office, No. 1970-389-930 Bulletin.

34. Cecil, L. K. 1971. Evaluation of processes available for removal of phosphorus from wastewater. EPA Project Report. WO. 17010. DRF.

35. Rohlich, G. A. and P. D. Uttormark. 1972. Wastewater treatment and eutrophication. Nutrients and eutrophication. Special Symposia, Vol. I. pp. 231-243.

36. Malueg, K. W., R. M. Brice, D. W. Schults, and D. P. Larsen. In Press. Interim Report No. 1, The Shagawa Lake Project; lake restoration by nutrient removal from wastewater effluent. EPA Internal Report.

37. Thomas, E. A. 1969. The process of eutrophication in Central European lakes. In: Eutrophication: causes, consequences, correctives. Proceedings of a Symposium, National Academy of Sciences, Washington, D. C. p. 37.

38. Pennypacker, S. P. 1967. Renovation of wastewater effluent by irrigation of forest land. J. Water Pollution Control Federation. 39:285.

39. Law, J. P., Jr. 1969. Nutrient removal from enriched waste effluent by the hydroponic culture of cool season grasses. Water Pollution Control Research Series 16080. U.S. Dept of Interior.

40. Bouwer, H. Water quality aspects of intermittent systems using secondary sewage effluent, U.S. water Conservation Laboratory, Phoenix, Arizona. Paper #8.

41. Tilstra, J. R., K. W. Malueg, W. C. Larson. 1972. Removal of phosphorus and nitrogen from wastewater effluent by induced soil percolation. J. Water Pollution Control Federation. 44:796-805.

42. Seidel, K. 1967. Wasserpflanzen reinigen abwasser. Umschau in Wissenschaft und Technik. 67:565.

43. Hemens, J. and G. J. Stander. 1969. Nutrient removal from sewage effluents by algal activity. In: Advances in Water Pollution Research, Prague 1969. Proceedings of the 4th International Conference. pp. 701-715. Pergaman Press.

44. Anderson, G. C. 1961. Recent changes in the trophus nature of Lake Washington - A review. In: Algae and Metropolitan Wastes, Trans. 1960 Seminar. U.S. Department of Health, Education and Welfare. pp. 27-33.

45. Edmondson, W. T. 1968. Water quality management and lake eutrophication: The Lake Washington case. In: Water Resources Management and Public Policy, edited by T. H. Campbell and R. O. Sylvester. University of Washington Press. pp. 139-178.

46. Edmondson, W. T. 1969. Eutrophication in North America. In: Eutrophication: causes, consequences, and correctives. National Academy of Sciences, Washington, D. C. pp. 124-149.

47. Edmondson, W. T. 1970. Phosphorus, nitrogen, and algae in Lake Washington after diversion of sewage. Science 169:690-691.

48. Emery, R. M., C. E. Moon, and E. B. Welch. 1971. Delayed recovery in a mesotrophic lake following nutrient diversion. Presented at the 38th Annual Meeting of the Pacific NW Pollution Control Assoc, Spokane, Washington.

49. Sarles, W. B. 1961. Madison's lakes, must urbanization destroy their beauty and productivity. In: Algae and Metropolitan Wastes, Trans. 1960 Seminar. U.S. Dept. of Health, Education, and Welfare. pp. 10-18.

50. Mackenthun, K. M., L. L. Lueschow and C. D. McNab. 1960. A study of the effects of diverting the effluent from sewage treatment upon the receiving stream. Wisc. Acad. Sciences, Arts and Letters. 49:51-72.

51. Tenney, M. W., W. F. Echelberger, Jr., and T. C. Griffing. 1970. Effects of domestic pollution abatement on a eutrophic lake. Partial Report on FWQA Demonstration Grant No. WPD-126.

52. Laurent, P. J., J. Garaucher and P. Vivier. 1970. The condition of lakes and ponds in relation to the carrying out of treatment measures. 5th International Water Pollution Research Conference. Pergamon Press Ltd. In Press.

53. Sanville, W. D., and C. F. Powers. 1972. Progress Report No. 1, Diamond Lake Studies-1971. NERP, Pacific NW Environmental Research Laboratory, Corvallis, Oregon.

54. Hamm, M. A. 1971. Limnologische Untersuchungen an Tegernasee and Schliersee Nach der Abwasserfernhaltung, Wasser und Abwasser-Forschung. Nr. 5.

55. Thronson, R. E. 1971. Control of erosion and sediment deposition from construction of highways and land development. U.S. Environmental Protection Agency.

56. National Association of Counties Research Foundation. 1970. Urban soil erosion and sediment control. Dept. of the Int., Federal Water Quality Administration. 15030DTL05/70.

57. Bogardi, J. L. Sediment transportation in alluvial streams. 2nd Int. postgrad course on Hydrol Methods for Developing Water Resources Management, Budapest, Hungary. Jan-July, 1968. Manual No. 13.

58. Krumbein, W. C. 1968. Statistical models in sedimentology. In: Sedimentology 10:7-23.

59. Barfield, B. J. 1969. Prediction of sediment profiles in open channel flow by turbulent diffusion theory. Water Resources Res. 5:291-299.

60. Brandt, G. H., E. S. Conyers, F. J. Lowes, J. W. Mighton, and J. W. Pollack. 1972. An economic analysis of erosion and sediment control for watersheds undergoing urbanization. NTIS PB 209212.

61. Maddock, T., Jr. 1969. Sedimentation engineering Chapter VI: Economic aspects of sedimentation. Proc. Amer. Soc, Civ. Eng. S. Hydraul Div. 95:191-207.

62. The Lake Restoration Researchers Team. 1971. The Lake Trummen Restoration Project. A Presentation, University of Lund, Sweden.

63. Johnson, C. S. 1971. Silt removal from a lake bottom. Interim Progress Report, EPA Project 16010-ELF.

64. Wisconsin Department of Natural Resources. 1970. Inland lake dredging evaluation. Technical Bulletin No. 46, Madison, Wisconsin.

65. Landner, L. 1970. Lake Restoration. Trials with direct precipitation of Phosphorus in polluted lakes; the binding of mercury in lakes; trials with grass carp. Swedish Water and Air Pollution Research Laboratory, Stockholm, Sweden.

66. Jernelov, A. 1970. Phosphate reduction in lakes by precipitation with aluminum sulfate. Fifth International Conference on Water, San Francisco, USA.

67. Peterson, J. O., J. P. Will, T. L. Wirth, and S. M. Born. 1973. Eutrophication Control: Nutrient inactivation by chemical precipitation at Horseshoe Lake, Wisconsin. Tech. Bull. No. 62. Wisconsin Dept. of Nat. Res., Madison.

68. Gahler, A. R., W. D. Sanville, J. A. Searcy, C. F. Powers and W. E. Miller. Studies on lake restoration by phosphorus inactivation. NERP, Pacific NW Environmental Research Laboratory, Corvallis, Oregon. (In manuscript).

69. Oglesby, R. T. 1969. Effects of controlled nutrient dilution on the eutrophication of a lake. In: Eutrophication: causes, consequences, and correctives, National Academy of Sciences, Washington, D. C. pp. 483-493.

70. Born, S. M. 1970. Final Report (May 1, 1968-January 31, 1971). The Inland Lake Renewal and Management Demonstration Project, submitted to Upper Great Lakes Regional Commission. pp. 5-10.

71. Welch, E. B., J. A. Buckley, R. M. Bush. 1972. Dilution as a control for nuisance algal blooms. J. Water Pollution Control Federation. 44:2245-2265.

72. Bhagat, S. K., J. F. Orsborn. 1971. Water quantity and quality studies of Vancouver Lake, Washington. Washington State University, Pullman, Washington.

73. Sketelj, J. and M. Rejie. 1966. Pollutional phases of Lake Bled. In: Advances in Water Pollution Research, Vol. I, Proceedings of 2nd International Conference, Tokyo. pp. 345-362.

74. Tenney, M. F., W. F. Echelberger, Jr. 1970. Fly ash utilization in the treatment of polluted waters, Bureau of Mines Information Circular 8488, Ash Utilization Proceedings. pp. 237-265.

75. Fast, A. W. 1971. The effects of artificial aeration on lake ecology. U. S. Govt. Printing Office, Washington, D. C.

76. Wirth, T. L. and R. C. Dunst. 1967. Limnological changes resulting from artificial destratification and aeration of an impoundment. Progress Report. Wisconsin Conservation Department, Research and Planning, Madison, Wisconsin.

77. Barnhardt, H. 1967. "Aeration of Wahnbach Reservoir without Changing the Temperature Profile." J. Amer. Water Works Assoc. 59:943-964.

78. Inland Fisheries Branch. 1970. Effects of artificial destratification on the distribution of bottom organisms in El Capitan Reservoir. State of California, Department of Fish and Game, Fish Bull. No. 148.

79. Symons, J. M., J. K. Carswell and G. G. Robeck. 1970. "Mixing of Water Supply Reservoirs for Quality Control." J. Amer. Water Works Assoc. 62:322-334.

80. Symons, J. M. 1971. Quality control in reservoirs for municipal water supplies. Quality Control in Reservoirs Committee, Water Quality Division, Amer. Water Works Assoc., A Committee Report, "Pre-publication."

81. Fast, A. W. 1968. Artificial destratification of El Capitan Reservoir by aeration. Part I - Effects on chemical and physical parameters. Calif. Dept. of Fish and Game, Fish Bull. 141.

82. Lackey, R. T. 1972. Evaluation of two methods of aeration to prevent winter kill. The Progressive Fish Culturist. 34(e):175-17b8.

83. Malueg, K. W., J. R. Tilstra, D. W. Schults and C. C. Powers. In Press. The effects of induced aeration upon stratification and eutrophication processes in an Oregon farm pond. International Symposium on Man-made Lakes. Knoxville, Tennessee.

84. Bernhardt, H., P. Cooley, J. A. Steel, J. E. Ridley and H. Ambuhl. 1967. Impoundment destratification by mechanical pumping. J. San. Div. Amer. Soc. Civ. Eng. 93(SA-4):136-143.

85. Leach, L. 1968. Eufaula Reservoir aeration research. Proc. Okla. Acad. Sci. 49:174-181.

86. Symons, J. M. 1971. Artificial destratification in reservoirs. Report of AMWA Committee on Quality Control in Reservoirs. J. Amer. Water Works Assoc. 63:597-604.

87. Fast, A. W. 1971. Effects of articifial destratification on zooplankton depth distribution. Trans. Amer. Fish Soc. 2:355-358.

88. Symons, J. M., W. H. Irwin, E. L. Robinson and G. G. Pobeck. 1967. Impoundment destratification for raw water quality control using either mechanical or diffused-air pumping. J. Amer. Water Works Assoc. 59:1268-1291.

89. Robinson, E. L., W. H. Irwin and J. M. Symons. 1969. Influence of artificial destratification on plankton populations in impoundments. Trans. Kentucky Acad. Sci. 30, Nos. 1 and 2.

90. Lackey, R. T. 1973. Artificial reservoir destratification effects on phytoplankton. J. Water Poll. Control Fed. 45:668-673.

91. Sheffield, C. W., R. T. Kaleel. 1970. Lake Apopka and aquatic weeds. Hyacinth Control Journal 8:45-47.

92. Smith, S. A., J. O. Peterson, S. A. Nichols and S. M. Born. 1972. Lake deepening by sediment consolidation Jyme Lake. Inland Lake Demonstration Project Report University of Wisconsin and the Wisconsin Department of Natural Resources.

93. Smith, G. E., T. F. Hall, R. A. Stanley. 1967. Eurasian water milfoil in the Tennessee Valley. Weeds 15:95-98.

94. Cook, A. H., C. F. Powers. 1958. Early biochemical changes in the soils and waters of artificially created marshes in New York, New York Fish and Game Journal 5:9-65.

95. Oswald, W. J. and C. G. Golueke. 1968. Harvesting and processing of waste-grown microalgae. In: D. F. Jackson (ed.). Algae, Man and the Environment. Syracuse University Press. p. 371-389.

96. Golueke, C. G., W. J. Oswald and H. K. Cree. 1964. Harvesting and processing sewage-grown planktonic algae. Serl. Repl. No. 64-8. Sanitary Eng. Res. Lab. U. of Calif, Berkeley, Calif. Mimeo.

97. McGauhey, P. H., R. E. Eliasson, G. Rohlick, A. G. Ludwig, and E. A. Pearson. 1963. Comprehensive study on protection of water resources of Lake Tahoe Basin through controlled waste disposal. Lake Tahoe Area Council, Al Tahoe, Calif.

98. Levin, G. V. and J. M. Barnes. 1964. Harvesting of algae by froth flotation. Final Report, Public Health Service Contract No. PH86-63-75.

99. Anon. 1957. Sewage stabilization ponds in the Dakotas. Vol. I and II. Joint Report of the North and South Dakota Depts. of Health, and the U. S. Depts of Health, Education and Welfare, Washington, D. C.

100. Neel, J. K., J. H. McDermott, and C. A. Monday, Jr. 1961. Experimental lagooning of raw sewage at Fayette, Missouri. Jour. Water Pollution Cont. Fed. 33(6):603-641.

101. Bush, A. F., J. D. Isherwood and S. Rodgi. 1961. Dissolved solids removal from waste water by algae. Jour. San. Eng. Div., Am. Soc. Civil Eng. 89, SA3:39.

102. Bogan, R. H. 1961. The use of algae in removing nutrients from domestic sewage. In: Algae and metroplitan wastes. TR W61-3, U. S. Public Health Service, Cinn., Ohio.

103. Oswald, E. J., C. G. Golueke, R. C. cooper, H. K. Cree, and J. C. Brenson. 1962. Water reclamation, algal production and methane fermentation in waste ponds. Manuscript No. 25, Int. Conf. Water Pollution Res., London.

104. Gates, W. E. and J. A. Borchardt. 1964. Nitrogen and phosphorus extraction from domestic waste water treatment plant effluents by controlled algal culture. Jour. Water Pollution Control Fed. 36:443.

105. Livermore, D. 1954. Harvesting underwater weeds. Water Works Eng. Feb. pp. 118-120.

106. Livermore, D. F. and W. E. Wunderlich. 1969. Mechanical removal of organic production from waterways. In: Eutrophication: causes, consequences, correctives. Printing and Publ. Office, N.A.S., Washington, D. C.

107. Yount, J. L. and R. A. Grossman, Dr. 1970. Eutrophication control by plant harvesting. JWPCF 42(5):R173-R183.

108. Bruhn, H. D., D. F. Livermore, and F. O. Aboaba. 1970. Physical properties and processing characteristics of macrophytes as related to mechanical harvesting. Paper #70-582. Am. Soc. Agr. Engineers, St. Joseph, Michigan (also avail. from NTIS as PB 198-129).

109. Nichols, S. A. 1971. The distribution and control of macrophyte biomass in Lake Wingra, Wisconsin Water Resources Center, Madison. Final Completion Report OWRR B-019-WIS(4). (Also avail. from NTIS as PB-205-152.)

110. Bagnall, L. O., T. W. Casselman, J. W. Kesterson, J. F. Basley, and H. E. Hellwig. 1971. Aquatic forage processing in Florida. Paper #71-536. Am. Soc. Agr. Eng. OWRR A-017-FLA(2).

111. Koegel, R. G., H. D. Bruhn, and D. F. Livermore. 1972. Improving surface water conditions through control and disposal of aquatic vegetation, Phase I: Processing aquatic vegetation for improved handling and disposal or utilization. Wisc. Water Resources Center, Madison, Technical Completion Report. OWR B-018-WIS(4).

112. Boyd, C. E. 1971. The Limnological role of aquatic macrophytes and their relationship to reservoir management. In: Reservoir fisheries and limnology. Special publ. #8 Am. Fisheries Soc. Washington, D. C. pp. 153-166.

113. Goodson, J. B. and J. J. Smith. 1970. Treatment of citrus processing wastes. Water Pollution Control Research Series #12060-10/70. EPA, Water Quality Office, Contract WPRD 38-01-67.

114. Mulligan, H. F. 1969. Management of aquatic vascular plants and algae. In: Eutrophication: causes, consequences, correctives. Printing and Publ. Office, NAS, Washington, D. C.

115. Lange, S. R. 1965. Commercial possibilities of dehydrated aquatic plants. Southern Weed Conference, Proceedings 18:543-551.

116. Bailey, T. A. 1965. Commercial possibilities of dehydrated aquatic plants. Southern Weed Conference, Proceedings 18:543-551.

117. Aboaba, F. O. 1971. Physical processing characteristics of some aquatic macrophytes. PhD. Thesis, Univ. Wisconsin (Avail. as OWRR B-018-WIS(3)).

118. Cifuentes, J. 1971. Screw press design parameters for dewatering water hyacinth (<u>Eichornia crassipes</u>). M. S. Thesis in Engineering, Univ. Florida, Gainesville, Dept. Agr. Eng. (WRSIC accession #W 72-03232.)

119. Abou-El-Fadl, M., S. G. Rizk, A. F. Abdel Ghani, M. K. El-Mofty, and M. F. A. Khadr. 1970. Utilization of water hyacinth as an organic manure with special reference to water-borne helminths. J. Microbiol. U.A.R. 3:27-34.

120. Lee, G. F. 1970. Eutrophication. Univ. Wisconsin Water Resources Center, Madison, Occasional Paper #2, Eutrophication Information Program.

121. Rogers, H. H., Jr. 1971. Nutrient removal by water hyacinth. M. S. Thesis, Auborn Univ., Alabama. WRSIC accession #W72-04776.

122. Gerloff, G. C. and P. H. Krombholz. 1966. Tissue analysis as a measure of nutrient availability for the growth of angiosperm aquatic plants. Limnol. and Oceanogr. 11:529-537.

123. Cottam, G. 1969. Changes in water environment resulting from aquatic plant control. Proceedings of meeting, joint industry/Government Task Force on Eutrophication. Univ. Wisconsin, Madison, Nov. 24-25, 1969.

124. Steward, K. K. 1970. Nutrient removal potential of various aquatic plants. Hyacinth Control Journal 8:34-35.

125. Taylor, R. G., R. P. Bates, and R. C. Robbings. 1971. Extraction of protein from water hyacinth. Hyacinth Control Journal 9:20-22.

126. Seidel, K. 1968. Elimination von Schmutz-und Ballaststoffen aus belasteten Gewassern durch hohere Pflanzen (Elimination of substances of mud and ballast from waters by higher plants). Zeitschrift Bitalstoffe-zivilisations Krankheiten Nr. 4.

127. Kiefer, W. 1968. Pflanzen biologische Reinigung von Abwasser (Biological wastewater treatment with plants). Umschau:210.

128. Greer, D. E., and C. D. Ziebell. 1972. Biological removal of phosphates from water. JWPCF 44:2342-2348.

129. Corey, R. B., A. D. Hasler, F. H. Schraufnagel, and T. L. Wirth. 1967. Excessive water fertilization: Report to water subcommittee, Natural Resources Committee of State Agencies. Madison, Wisconsin (Unpublished).

130. Bailey, W. M., and R. L. Boyd. 1972. Some observations on the white amur in Arkansas. Hyacinth Control Journal 10:20-22.

131. Opuszynski, K. 1972. Use of phytophagous fish to control aquatic plants. Aquaculture 1:61-74.

132. Sport Fishing Institute Bulletin. 1972. Grass carp problem. No. 240. Washington, D. C.

133. Hasler, A. D., and E. Jones. 1949. Demonstration of the antagonistic action of large aquatic plants on algae and rotifers. Ecology 30:359-364.

134. Porter, K. B. 1972. Control of natural phytoplankton populations by grazing zooplankton. (Abstract of paper presented at AAAS-Ecological Society) Bull. Ecol. Soc. Am. 53:9.

135. Mattox, K. R., D. Stewart, and G. L. Floyd. 1972. Probable virus infections in four genera of green algae. Can. Jour. Bot. 18:1620-1621.

136. Safferman, R. S., and M. E. Morris. 1963. Algal virus: isolation. Science 140-679-680.

137. Padan, E. and M. Shilo. 1969. Distribution of Cyanophages in natural habitats. Verh. International Verein. Limnol. 17:747-751.

138. Shilo, M. 1971. Biological agents which cause lysis of blue-green algae. Mitt. International Verein. Limnol. 19:206-213.

139. Cappleman, L. E. 1972. Detached leaf culture of Eichornia crassipes and application to the culture of its pathogens. M. S. Thesis, Florida Atlantic University, Boca Raton (unpublished).

140. Sculthorpe, C. D. 1967. The biology of aquatic vascular plants. Edward Arnold (Publishers) Ltd., London.

141. Gillies, P. (Ed.) 1972. The Water Hyacinth. Water News Letter 14:2. Water Information Center, Inc. Publ. Port Washington, L.I., N.Y.

142. Coulson, J. R. 1971. Prognosis for control of water hyacinth by arthropods. Hyacinth Control Journal 9:31-34.

143. Hawkes, R. B. 1965. Domestic phases of program designed to use insects to suppress alligator weed. Proc. S. Weed Conf. 18:584-585.

144. Baloch, G. M., A. G. Khan, and M. A. Ghani. 1972. Phenology, biology and host-sepcificitty of some Stenophagous insects attacking Myriophyllum spp. in Pakista. Hyacinth Control Journal 10:12-16.

145. Blackburn, R. D., and L. W. Weldon. 1965. A fresh water snail as a weed control agent. (Abstract) Proc. S. Weed Conf. 18:589.

146. Sguros, P. L., T. Monkus, and C. Phillips. 1965. Observations and techniques in the study of the Florida manatee - reticent but superb weed control agent. Abstract. Proc. S. Weed Cond. 18:588.

147. Fitzgerald, G. P. 1971. Algicides. Literature Review No. 2, Eutrophication Information Program, the University of Wisconsin Water Resources Center.

148. Fitzgerald, G. P. 1966. Use of potassium permanganate for the control of problem algae. Jour. Amer. Water Works Assoc. 58:609-614.

149. Timmons, F. L. 1970. UNESCO Meeting on Ecology and Control of Aquatic Vegetation, December 16-18, 1968. Paris. Hyacinth Control Journal 8(2):23-26.

150. Lambou, V. W. 1972. Problems of mercury emissions into the environment of the United States. Report to the Working Party on Mercury, Sector Group on Unintended Occurrence of Chemicals in the Environment, OECD, Environmental Protection Agency.

151. Jernelov, H. 1969. Conversion of mercury compounds. In: Chemical fallout. Charles C. Thomas, Springfield, Illinois.

152. Trukayama, K. 1966. The pollution of Minamata Bay and Minnemata Disease. Adv. Disease. Adv. Water Pollution Research, Proc. Int. Conf. 3:153-180.

153. Study group on mercury. hazards. 1970. Hazards of mercury. Special report to the secretary's Pesticides Advisory Committee, Dept. of Health, Education and Welfare and Environmental Protection Agency.

154. Stephan, D. G. 1971. Trip report: Finland and Sweden, Feb. 21-25, 1971. Assistant Commissioner, Research and Development, Water Quality Office, Environmental Protection Agency.

155. Bongers. L. H. and M. N. Khattak. 1972. Sand and gravel overlay for control of mercury in sediments. Research Institute for Advanced Studies, prepared for the Environmental Protection Agency, Washington, D. C.

156. Suggs, J. D., D. H. Petersen and J. B. Middlebrook, Jr. 1972. Mercury pollution control in stream and lake sediments. Advanced Technology Center, Inc. Prepared for the Environmental Protection Agency, Office of Research and Monitoring, Washington, D. C.

157. Ryckman, Edgerley. Tomlinson and Associates, Inc. Pesticide poisoning of Pond Lick Lake, Ohio investigation and resolution, June 2 - July 5, 1971. Final report prepared for the Environmental Protection Agency, Division of Oil and Hazardous Materials, OWP, Washington, D. C.

158. Nye, W. B. 1972. The hazardous material spill experience in Shawnee Lake, Ohio - a case history. In: Control of hazardous materials spills. Proc. 1972 Nat. Conf. on the Control of Hazardous Materials Spills, March 21-23, 1972, Houston, Texas.

159. Control of hazardous materials spills. 1972. Proc. Nat. Conf. on the Control of Hazardous Materials Spills, March 21-23, 1972, Houston, Texas.

160. Kusler, J. A. 1972. Survey: lake protection and rehabilitation legislation in the United States. Inland Lake Renewal and Shoreland Management Demonstration Project Report. Univ. of Wisc. Madison.

161. Yanggen, D. A. 1971. Preserving lakes by protecting their shorelands. In: Proceedings Workshop Conference on Reclamation of Maine's Dying Lakes, Univ. of Maine, Bangor, March 24 and 25, 1971. Conf. Report No. 2, Water Resources Center, Univ. of Maine, Orono.

162. Klessig, L. L. and D. A. Yanggen. 1972. Wisconsin Lake Shore Property Owners Associations: Identification, description and perception of lake problems. Inland Lake Renewal and Shoreland Management Demonstration Project Report, Univ. of Wisc., Madison.

163. Kusler, J. A. 1971. Artificial lakes and land subdivisions. Report from Wisconsin Law Review, Vol. 1971, No. 2, Univ. of Wisc., Madison.

164. Lejeune, H. 1972. Economic impacts of artificial lake development: Lakes Sherwood and Camelot - a case history. Inland Lake Renewal and Shoreland Management Demonstration Project Report. Univ. of Wisc., Madison.

165. Yanggen, D. A. and Z. L. Zigurds. 1972. Legal problems of property owners' associations for large water-oriented recreational housing complexes. Inland Lake Renewal and Shoreland Management Demonstration Project Report. Univ. of Wisc., Madison.

APPENDIX

Section VI

LAKE PROBLEMS

SOURCES OF WATER QUALITY PROBLEMS IN LAKES

Water quality problems have resulted as increased amounts of wastes have been introduced to aquatic receiving systems. Molecules of diverse chemical structures have been synthesized resulting in compounds which are refractory to degradation. The ability of microorganisms to metabolize pollutants to carbon dioxide and water and thus to remove them from the aquatic environment is the primary biological method for "self purification" of waters. As organisms advance evolutionally, the inherent ability to assimilate and degrade new and diverse products is rapidly diminished. Evidence of this is seen in the alarming levels of certain chlorinated hydrocarbons. Although contaminants may originate from a variety of sources, they can usually be broadly classified as industrial, municipal or agricultural wastes.

Industrial Wastes

Industrial wastes often create unique problems in the aquatic environment. They are frequently in the form of liquid containing substances which are difficult if not impossible to remove from drinking water. The magnitude of the problem is brought to light by the fact that there are approximately 240,000 water using establishments in the United States which consume 75,700 m^3 (20,000,000 gallons) or more water (1). Industrial waste water effluent has three to four times more oxygen-demanding wastes than the total sewered population in America (2). As industries expand and diversify the attendant problems of industrial effluents increase at a proportional rate. Atmospheric rain-out resulting from industrial stack and automobile emissions also contribute to the contamination of waterways.

No detailed inventory of industrial wastes is available; however, as seen in Table I, the amount of water used and waste generated is enormous. Water and airborne wastes contain organic and inorganic solids, suspended material, toxic substances, and biological growth stimulants.

The magnitude of industrial waste loading can be illustrated by using thermal pollution as an example of the total problem. The electric power industry, the single largest producer of waste heat, and a contributor of other pollutants, is increasing at a rate of 7.2 per cent annually, almost doubling every ten years (4). As seen in Table II, this trend is expected to continue. Other industries also require water for cooling purposes (Table III). The metal, chemical, petroleum and coal, paper, food, and various manufacturing industries are among those requiring large quantities of cooling water.

It has been estimated that by 1980 electric power cooling operations alone will require the equivalent of one-fifth the total fresh water runoff to the United States (4). However, the thermal loading associated with power generation is only one example of water quality degradation caused by industry. Other industries have effluents which can be more difficult to deal with.

TABLE I

Estimated Volume of Industrial Wastes
Before Treatment, 1964 1/

	Waste Water Volume (billion m^3)	Waste Water Volume (billion gallons)	Process Water Intake (billion m^3)	Process Water Intake (billion gallons)	BOD (million kg)	BOD (million pounds)	Suspended Solids (million kg)	Suspended Solids (million pounds)
Food and kindred products	2.61	(690)	0.98	(260)	1952.2	(4,300)	2996.4	(6,600)
Meat Products	0.37	(99)	0.20	(52)	290.6	(640)	290.6	(640)
Dairy Products	0.22	(58)	0.05	(13)	18.2	(400)	104.4	(230)
Canned & frozen food	0.33	(87)	0.20	(51)	544.8	(1,200)	27.2	(600)
Sugar refining	0.83	(220)	0.42	(110)	685.6	(1,400)	2270.0	(5,000)
All other	0.83	(220)	0.16	(43)	304.2	(670)	49.9	(110)
Textile mill products	0.53	(140)	0.42	(110)	404.1	(890)	--	--
Paper & allied products	7.19	(1,900)	4.92	(1,300)	2678.6	(5,900)	1362.0	(3,000)
Chemical & allied products	7.19	(3,700)	2.12	(560)	4403.8	(9,700)	862.6	(1,900)
Petroleum & coal	4.92	(1,300)	0.33	(88)	227.0	(500)	208.8	(460)
Rubber & plastics	0.61	(160)	0.07	(19)	18.2	(40)	22.7	(50)
Primary metals	16.28	(4,300)	3.79	(1,000)	204.3	(450)	2137.8	(4,700)
Blast furnaces & Steel mills	13.63	(3,600)	3.29	(870)	76.6	(160)	1952.2	(4,300)
All other	2.80	(740)	0.49	(130)	145.3	(320)	195.2	(430)
Machinery	0.57	(150)	0.09	(23)	27.2	(60)	22.7	(50)
Electrical cmachinery	0.34	(91)	0.11	(28)	31.8	(70)	9.1	(20)
Transportation equipment	0.91	(240)	0.22	(58)	54.5	(120)	--	--
All other manufacturing	1.70	(450)	0.72	(190)	177.1	(390)	422.2	(930)
All manufacturing	49.58	(13,100)	14.00	(3,700)	9988.0	(22,000)	8172	(18,000)
For comparison: Sewered population of the U.S.	20.06	(5,300) 2/	-	-	3314.2	(7,300) 3/	3995	(8,800) 4/

1/ Columns may not add, due to rounding.
2/ 120,000,000 persons times 0.452 m (120 gallons) times 365 days.
3/ 120,000,000 persons times 0.0757 kg (1/6 pound) times 365 days.
4/ 120,000,000 persons times 0.0808 kg (0.2 pound) times 365 days.

Source: (3)

Table II

U.S. Electric Power - Past Use, Future Estimates

Year	In billion Kilowatt-hours
1912	12
1960	753
1965	1,060
1970	1,503
1975	2,022
1980	2,754
1985	3,639

Source: (5)

TABLE III

Use of Cooling Water by U.S. Industry

Industry	Cooling Water Intake m^3 (billions)	(billions of gallons)	Percent of Total
Electric power	154.0	(40,680)	81.3
Primary metals	12.8	(3,387)	6.8
Chemical and allied products	11.8	(3,120)	6.2
Petroleum and coal products	4.6	(1,212)	2.4
Paper and allied products	2.3	(607)	1.2
Food and kindred products	1.5	(392)	0.8
Machinery	0.6	(164)	.3
Rubber and plastics	0.5	(128)	.3
Transportation equipment	0.4	(102)	.2
All others	1.0	(273)	.5
Total	189.5	(50,065)	100.0

Source: (5)

Municipal Wastes

Municipal waste treatment accounts for the disposal of a heterogeneous variety of liquid and solid material which comes from domestic (55%) and industrial (45%) facilities (4). Added to this constant waste load is the periodic storm sewer runoff, which in certain areas of the country (Northeast, Midwest and Far West) may contain deicing chemicals and organic and inorganic pollutants. Domestic waste treatment sewers service approximately two-thirds of the total population (4). Of this sewered population, approximately 60 per cent have adequate treatment facilities (4).

A major contribution of phosphates and nitrates to lakes and reservoirs comes from municipal plants (4). In addition to the inorganic nutrients are various organic compounds, such as detergents, which can act as a substrate for a variety of microorganisms. The organically and chemically rich effluents serve as an ideal millieu for the growth of the endogenous bacteria in the receiving waters. It is the growth of these normal inhabitants which lowers the dissolved oxygen and is reflected as biochemical oxygen demand (BOD).

Agricultural wastes

Agricultural wastes in waters originate basically from either fertilizers and pesticides supplied to growing crops or as wastes from livestock. Fertilizers contain predominately nitrogen and phosphorus, which when applied to the land, can wash into the aquatic environment. These two nutrients stimulate the growth of algae, bacteria and aquatic weeds leading to a shift in the normal aquatic life.

Pesticide runoff is another problem associated with, but not exclusive to, agricultural activities. Productivity reportedly has increased with the increased use of insecticides and the consequent reduction of plant pests. However, in some areas, the cost ecologically has been manifested in either the elimination of or decrease in numbers and diversity of certain aquatic organisms. As the population increases with attendant demands for more food a continued, if not increased, pesticide use will be required.

Feedlot wastes are a potential contributor to the pollution of waters in various areas of the country. Modern methods for raising beef cattle, poultry and swine, along with dairy farm operations produce concentrated waste sources of potential water pollution. The animal wastes produced today are estimated to be the equivalent of the waste produced by 2 billion people (4). This figure does not necessarily mean that a proportional amount of animal waste ends up in water, since much does not reach the aquatic ecosystem. However, it is a measure of the pollution potential.

Miscellaneous Sources

Mine drainage

Acid drainage comes from mines where the water and air mix allowing the growth of sulfur oxidizing bacteria. As a consequence of this growth sulfuric acid is produced resulting in a pH, in extreme cases, of less than one. It has been estimated that in the Appalachia region, where 75 per cent of coal mine pollution occurs, about 168,000 kilometers of streams are polluted (4). Other mining operations for phosphate, iron, copper, gold and aluminum also are responsible for acid mine discharge.

Oil and Hazardous Materials

Pollution of the aquatic environment due to oil and hazardous materials spills has grown steadily in the past years. As seen in Table IV, the number of spills over 15,900 l (100 barrels) increased dramatically in a period of one year. The number of spills is expected to increase as the flow of oil to refineries increases to meet rising fuel demands. Disposal of spent motor oils and lubricants also presents a problem. It has been estimated that 1,330,000 kiloliters of used oil per year have to be disposed of by gas service stations (4).

TABLE IV - Number of Reported Oil Spills in U.S. Waters over 15,900 l (100 Barrels)

	1968	1969
Vessels	347	532
Shore facilities	295	331
Unidentified	72	144
Total	714	1,007

Source: (6)

Watercraft Wastes

Pollution resulting from sewage discharged from watercraft is primarily of health significance rather than organic or oxygen depleting significance. It has been suggested that the total potential sewage from vessels is equal to a town of 500,000 people (4). However, sewage waste disposal from vessels can present a problem in confined areas such as boat harbors and marinas.

IMPACT OF CONTAMINANTS ON LAKE ENVIRONMENTS

Impairment of lakes can result from an isolated instance of the introduction of a contaminant, such as occurs during an accidental spill, through continuous or intermittent industrial or municipal point source discharges, or through surface runoff and contributions from tributary streams and ground waters.

The nature of contaminants and their effects on lake environments vary widely. In general, the various contaminants can be grouped into categories based upon the manner in which they affect a lake ecosystem. The major groups are the organic wastes, inorganic nutrients, silts and sediments, toxic substances, and heated waters. Other contaminants include radicactive wastes, various non-toxic salts, and many others which produce a wide range of effects on lake environments. The impact of each of the major groups of contaminants is discussed below under the respective headings of eutrophication, sedimentation, thermal problems and selected toxic substances. Radioactive wastes and non-toxic salts are briefly discussed under the heading of miscellaneous problems.

Eutrophication

Eutrophication may be broadly defined as nutrient or organic matter enrichment, or both, that results in high biological productivity and a decreased volume within a lake ecosystem. Eutrophication is, therefore, a process by which a lake gradually evolves from a condition of low productivity (oligotrophic) to a highly productive condition (eutrophic). Organic matter and nutrients are carried into the lake by runoff and leaching from the drainage basin, stimulating increased biological productivity of all kinds. Products of erosion carried to the lake, and excessive quantities of organic matter, both plant and animal, produced within the lake, lead to a gradual filling-in, and the lake becomes shallower and smaller. The waters consequently become generally warmer. Rooted aquatic plants take over increasingly more space, their dead remains accelerating the filling of the basin. Eventually the lake becomes a marsh, upon which terrestrial vegetation progressively encroaches until the lake ceases to exist, being replaced by a dry-land environment. The lake then, not only evolves from oligotrophy to eutrophy, but, if the aging process is permitted to proceed to completion, eventually is subjected to total extinction.

Natural and Accelerated (Cultural) Eutrophication

The gradual enrichment and aging of lakes is a natural process which takes place under completely natural conditions, in the absence of man, provided that a sufficient nutrient supply is available from the drainage basin. For lakes situated within a relatively sterile drainage area, the aging process may span geologic time. Other lakes, subject to heavy nutrient loading from naturally fertile drainage basins apparently were highly eutrophic prior to their exposure to civilization.

The role of man in the eutrophication process may completely override natural forces. Many lakes have been observed to become enriched and to age very rapidly from the effects of domestic or industrial waste disposal, or from drainage basin disruptions or alterations resulting from man's activity. Nutrient flux to lakes can be increased manyfold by, for example, the input of nutrient-containing wastes, agricultural fertilization, clearing of forest lands, and roadbuilding and other construction. Many lakes exposed to increased nutrient input are currently exhibiting symptoms of rapidly increased rates of eutrophication; this condition is referred to as accelerated or cultural eutrophication, and it is an ever-growing problem in the United States and other countries.

Consequences of Eutrophication

The progressive eutrophication of a lake results in distinct physical, chemical, and biological changes, generally in the direction of impairment of the lake's utility to man. Oligotrophic lakes have the highest quality water (although perhaps not the best fishing), and the water is well suited to a variety of uses. Oligotrophic lakes are good multi-purpose lakes.

Very definite changes in the quantity and quality of the biota occur as eutrophy proceeds. With the increased productivity associated with accelerated rates of eutrophication comes the filling of the basins with decaying organic materials and sediments resulting in an increased oxygen demand on the overlying waters. The increased oxygen demand may result in total depletion of oxygen in the cooler bottom waters during the summer, accompanied by an increase in the products of respiration and decomposition, namely carbon dioxide, methane, and hydrogen sulfide. These developing anaerobic conditions result in replacement of existing benthic organisms with less desirable types, and cold-water species of fish, such as trout and salmon, are no longer able to exist; they are replaced by forms tolerant of higher temperatures. During the winter, under heavy ice and snow cover, shallow eutrophic lakes may be subjected to complete oxygen depletion. As a result entire fish populations may be eliminated, as frequently happens in the northern states.

In addition to restricting fish populations, highly eutrophied lakes are undesirable aesthetically and with respect to water use. Algal blooms produce taste and odor problems, and create unsightly surface scums which discourage water contact recreational activities. Dense growths of rooted aquatic plants may accompany, or occur in place of, the nuisance algal blooms. Such intense plant production greatly inhibits use of the water for swimming, fishing, or boating. Accumulation of algal mats and dense weed growths are most pronounced near shore, where man's contact with the water is greatest. The accumulated algal masses begin to decay in a short period of time, resulting in extremely foul-smelling conditions. Excessive plant production, then, can render a lake virtually unfit for recreational purposes or shoreline development.

In addition to their deleterious effects on aesthetic and recreational aspects of lakes, the excessive growth of aquatic plants can seriously affect water quality. Large quantities of planktonic algae frequently, and to a serious extent, increase the rate of clogging of sand filters at water treatment plants.

Probably even more serious is the increased frequency of taste and odor problems resulting from algae in eutrophic lakes. These can originate from either living or dead algae, or from the fungi which grow on algae remains. Tastes and odors may be produced by members of all the major algal groups: the blue-greens, greens, diatoms, and flagellates. No one group is responsible.

Still other water quality problems resulting from eutrophication are increased color in the water, resulting from plant growth, and concentrations of iron, manganese, and sulfide which may occur as the result of oxygen depletion.

Certain blue-green algae have been shown to have toxic effects on animals. Domestic animals, such as cattle and sheep, as well as fish and aquatic invertebrates, may be susceptible to toxic substances excreted by algae of this group. Water in which certain blue-green algae have bloomed may produce death in mammals and fish even when the algal cells themselves are excluded. There is also evidence that allergic reactions and gastrointestinal disturbances may result in humans from contact and ingestion of lake water in which algae exist in bloom proportions.

Sedimentation

Sediments are an integral part of lake ecosystems, providing habitats for benthic organisms and serving as a pool for nutrients necessary for aquatic plant growth. Ion exchanges and nutrient transport between the mud and water significantly affect the lake's productivity. Accelerated erosion and subsequent deposition of sediments in lakes can result in a degradation of these natural ecosystems. In terms of volume, sediment is today's greatest water pollutant. It reduces the storage capacity of reservoirs, fills lakes and ponds, clogs stream channels, buries aquatic habitats and increases turbidity.

Effects of Sediment

Sediments influence the physical, chemical and biological processes occurring in lakes. Perhaps one of the most obvious is the filling of lakes and impoundments by sedimentation thus restricting the useful life of the water body. A detailed survey of 148 artificial lakes (7) revealed the average annual loss of water retaining volume which as shown in Table V, varies between 0.5 to 2 percent annually.

Table V

ANNUAL LOSS OF RETAINING VOLUME FOR 148 LAKES

No. of Lakes	% Annual Volume Loss
34	0.5
39	0.5 to 1.0
39	1.0 to 2.0
36	2.0

Source: (7)

Suspended sediments increase turbidity and reduce the depth to which light penetrates below the water surface thus restricting the growth of photosynthetic flora and reducing the lake's productivity. Increased turbidity also affects aquatic food chains by impairing the sight and food gathering efficiency of predators. The European Inland Fisheries Advisory Commission (8) reports the effect of inert suspended solids on freshwater fish as shown in Table VI.

Table VI

EFFECT OF INERT SUSPENDED SOLIDS ON FRESHWATER FISH

Concentration, mg/l	Effect
25	No evidence of harmful effects
25-80	Good to moderate fisheries
80-400	Good fisheries unlikely
400	Poor fisheries

Source: (8)

High concentrations of suspended materials may also be deleterious to aquatic vertebrates by reducing their resistance to disease, preventing the successful development of eggs and larvae, modifying natural migrations and reducing the abundance of food. Buck (9) removed the fish from 39 farm ponds having a wide range of turbidities, and restocked the ponds with largemouth black bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*) and red-ear sunfish (*Lepomis microlophus*). After two growing seasons, the fish crop was harvested and the effects of various turbidity levels on reproduction were compared as seen in Table VII.

TABLE VII

EFFECT OF TURBIDITY ON FISH REPRODUCTION

Yield, kg/ha	(lb/acre)	Turbidity, mg/l
181.2	(161.5)	25
105.5	(94.0)	15-100
32.9	(29.3)	100

Source: (9)

Buck (9) also reported that largemouth black bass, crappies (Pomoxis) and channel catfish (Ictalurus punctatus) grew more slowly in a reservoir where the water had an average turbidity of 130 mg/l than in another reservoir where the water was always clear.

Lake sediments provide habitats for benthic organisms including bacteria, fungi, algae, flagellates, ciliates, sponges, mussels, worms, insects and snails. Some of these organisms have commercial value, and others are essential links in food chains which sustain fish, water fowl and other wildlife. When accelerated erosion resulting from farming, timber harvest and other activities causes heavy sediment inputs to a lake, the benthic flora and fauna may be blanketed with layers of silt. Feeding grounds and spawning sites as well as entire populations may be destroyed, causing radical changes in the lake ecosystems.

By the ion exchange process at the mud-water interface, nutrients are either released to the bottom water or are removed from the water by the sediments. These ion exchanges are caused by oxidation-reduction (redox) reactions. The oxidation potential of a solution is determined by the type and proportion of oxidized and reduced ions in the solution.

When oxygen is available to the lake bottom, the top strata of its sediments are oxidized. This layer acts as a barrier against diffusion from the mud to the water and holds nutrients in the sediments. However, when a lake's benthos becomes anaerobic this layer becomes thinner and may disappear entirely. As the oxidized layer of sediments is destroyed, nutrients in reduced form (i.e., Fe^{++}, Mn^{++}, NH_3 and P) are released from the sediments into the water and are available for assimilation by the biota.

Suspended solids entering a lake may adsorb both nutrients and toxic materials removing them from possible involvement in the food web as deposition of suspended particles occurs. Gumerman's (10) study of sterile sediments from Lake Erie and Lake Superior demonstrated that the maximum phosphate adsorbing capacity of the sediments is in the top 3.5 mm, and is reduced to zero below 14 mm. Gumerman (10) also found that the release of adsorbed phosphorus from sediments will maintain sufficient concentrations of phosphates to sustain algal growths for some time after phosphate input has ceased. Another study on phosphate equilibrium between reduced sediments and water (11) revealed that sediments in a reduced state will adsorb less phosphate than the same sediment in an oxidized state. Consequently, under low oxygen tensions at the mud water interface, phosphates are released into the water by chemical reduction reactions and by a physical tendency of the sediment particles to adsorb fewer molecules and ions.

Sources of Sediments

Lake sediments fall into two general categories, depending upon their origin. Autochthonous sediments are generated within the lake itself, and are often composed primarily of decomposed aquatic plants. A highly productive eutrophic lake will have a larger proportion of autochthonous sediments than an oligotrophic lake. Allochthonous sediments are transported into the lake from an outside source. Under natural conditions these sediments are generally the result of three geologic processes - erosion, transportation, and deposition. Human activities associated with forestry, agriculture, mining, urban development, highway construction, and channelization often tend to accelerate the natural geologic processes thereby increasing several fold the natural sedimentation rates of lakes.

Timber harvesting operations may be responsible for increased sedimentation. On a steep forested slope in Oregon clear-cutting with no roads increased sedimentation three times more than that of a control slope (12). Erosion on patch-cut areas with forest roads has reportedly increased sedimentation more than 100 fold.

Runoff from cultivated land carries a heavier silt load than that from either forest or grassland. However, soil conservation practices, including contour plowing and strip cropping, have greatly reduced agricultural land erosion.

Strip mine runoff and erosion of mine tailings are a major source of sediment in some areas. The annual sediment yield from unmined areas of Cane Branch, Kentucky, averaged about 8.8 metric tons per square kilometer (13). Erosion of mine spoil banks in this same drainage basin resulted in an average annual yield of 9,455 metric tons per square kilometer, and erosion of abandoned coal haul roads at steep grades was also severe.

Urban land development resulting in exposure of bare soil at construction sites is also a cause of accelerated erosion. Yorke and Davis (14, 15) indicate that a direct relation exists between the sediment yield of a basin and the area of land under construction, the season of the year, slope of the land, and proximity of construction sites to stream channels. Streamflow and sediment data were collected at gauging stations on Bel Pre Creek in Montgomery County, Maryland, between 1963 and 1967. Pasture and woodland dominated the landscape prior to March 1965, however, between March 1965 and August 1967, 15 percent of the watershed was developed into garden apartment and townhouse complexes. Suspended sediment discharged increased 14 times as a result of this construction (14, 15). A study on the effect of urbanization on sediment yield in New Jersey (16) also suggested that yields are proportional to the degree of urbanization. The low population density pine barrens yielded 4 - 14 metric tons of sediment per square kilometer per year, while the urbanized Delaware River area yielded 9 - 35 metric tons per square kilometer per year, and in the Philadelphia area, the yield was up to 175 metric tons per square kilometer per year. This corresponds to the 70 - 175 metric tons per square kilometer per year sediment yield reported (17) for the Washington and Baltimore urban and suburban areas.

Sediment transported by storm runoff was measured (18) for 25 storm events on a 23.5 hectare watershed in Kensington, Maryland. Between July 1952 and January 1962, 89 single family homes were built and 171 metric tons of sediment per acre were lost from this watershed. It is apparent that sediment yield is controlled by the combined effect of runoff and vegetation cover, both of which are affected by human use of the land.

The extent of erosion and transportation of soil exposed by highway construction was studied (19) in a 11.8 square kilometer watershed in Fairfax County, Virginia. Sediment yield was measured at gauging stations and revealed that, with average precipitation, erosion was 10 times that normally expected for cultivated land and 200 times that expected of grassland and 2,000 times that expected from forest land.

Eolian sediments are composed of material that was borne, deposited, produced, or eroded by the wind. Lakes in evergreen forests are at times so covered with pine pollen that their surface takes on a golden hue. This material is eventually deposited as organic sediment. Lakes nearby industrial plants or construction sites also receive fallout which may contain lead, mercury, and a host of other contaminants.

Thermal Pollution

With the settling of North America vast stands of forest canopy and tall prairie grass were removed, exposing the soil beneath to direct solar radiation. An obvious result was a general warming of the continent's streams and lakes. Today an urbanized society and an industrial economy, with continually rising demands for power plants and factories, many of which discharge thermal energy, contribute to the warming of our waterways.

Effects of Thermal Pollution

An increase in ambient water temperature caused by thermal effluents entering a lake may increase the metabolic rate of aquatic organisms and cause a corresponding increase in the food required for maintenance of body weight with no growth. Members of the freshwater family of fishes, Centrarchidae, reportedly ate three times as much food at 20°C as at 10°C (20), and brown trout, Salmo trutta, showed a constantly increasing feeding rate from 10°C to 19°C, above which the rate declined abruptly. When water temperatures rise, the swimming speeds of fish may also be affected. Acclimated goldfish increased their swimming speeds as temperatures were increased from 5°C to 20°C (21). Cruising speeds remained fairly constant until temperatures reached 30°C and then dropped off rapidly with further temperature increases.

The optimum temperature for maximum growth depends on available food. Young sockeye salmon raised in tanks with surplus food grew best at temperatures near 15°C (22), and at higher or lower temperatures their growth rates declined sharply. However, when given a small daily food ration these fish grew best at near 5°C and did not grow at all at 15°C. Increasing the temperature of a relatively barren water body, resulting in increased food requirements of the fish populations, could conceivable lower the fish supporting capacity of the lake or impoundment.

Increased water temperature reduces the solubility of oxygen thus reducing the dissolved oxygen available to aquatic fauna. This harmful effect is intensified because the oxygen consumption of aquatic vertebrates is approximately doubled for every ten degrees' C rise in temperature (23).

Fishes will adapt to higher temperatures, but the success of this process depends on the absolute temperature, the length of exposure to high temperature and the rate of temperature change. Gradually exposing fishes to higher and higher temperatures acclimates them to these elevated temperatures, but it lessens their ability to survive at low temperatures (24). It follows that the thermal shock caused by a large reduction in thermal effluent, during a power generating station's shutdown, could be more damaging to aquatic biota than the original water temperature increase. Meyer (25) points out that subtropical fishes are living much closer to their thermal limit than are polar species. Thus thermal pollution may be more critical in southern states than in northern states.

Elevated water temperatures may stimulate the activity of parasites and disease. Hedgpeth and Gonar (26) noted that maintaining bivalves in warm waters had the disadvantage of increasing the predatory gastropod activity, since oyster and mussel pests such as Urosalpinx and Thais thrive at warmer temperatures.

Many biological cycles are initiated by a temperature stimulus. Such an impulse induces sexual activity in marine animals (27). Salmon do not spawn if the water temperature is too high. The ability of a species to adapt to an incremental temperature rise may be different at various ontogenic stages. For example, fish eggs and larvae may have more sensitive temperature requirements than the adults. Trout eggs do not hatch if they are incubated in water that is too warm. and some fish species require a winter chill period for successful reproduction. In the vicinity of a thermal outfall fish might hatch too early in the spring before their natural food has become plentiful. Insect nymphs in an artificially warmed water body might emerge too early for mating flight and be immobilized by the cold air.

Sublethal temperature effects are also important. For example, the embryos of brown trout reared at high temperatures (13°C) yielded significantly smaller embryos than those hatched at 2.8°C (28, 29). A larger proportion of the yolk is required for metabolism of embryonic tissues at the higher temperature.

Temperature increases within the ranges tolerated by the existing species tend to increase productivity, provided that light and nutrients are not limiting. In northern lakes added heat might make the water more attractive for swimmers, but if this also resulted in extensive growth of filamentous algae or other types of noxious vegetation the advantage may be offset. Increased algal productivity may also reduce the ability of predatory fish to see their prey. When temperature ranges of existing populations are exceeded, the species composition will change. Below 30°C diatoms are often represented by the largest members of species (30) with green algae becoming more abundant at temperatures from 30°C to 35°C. Above 35°C blue-green algae frequently dominate the flora.

If a thermal discharge flows out over the surface of the lake, it will reinforce any tendency of the lake to stratify into density layers. Such stratification inhibits mixing between the surface waters, which are generally rich in dissolved oxygen, and the hypolimnetic waters, which may become oxygen depleted if not replenished.

Artificially induced temperature changes may trigger the spawning migration at the wrong time of year. Migrating fishes must be able to avoid zones of unfavorable temperature, as such zones may block the migration, and spawning may be thwarted.

Sources of Thermal Pollution

Power generating plants are the prime source of thermal pollution. This trend may continue since, in the United States, power generation has doubled every ten years since 1945, and indications are that future requirements will demand an even higher rate of increase. Other sources of thermal pollution are industrial effluents, sewage effluents, and exothermic reaction associated with oxidation of organic matter.

Selected Toxic Substances

Historically, natural weathering of mineral rich rock formations was the primary mechanism for release of toxic substances to the aquatic environment. During the past few decades the man-induced release of naturally occurring toxic materials combined with the discharge of synthetic toxic compounds has far exceeded the input through natural weathering. As a consequence of the increased rate of input, low level residues of toxic substances are found throughout the total biosphere.

Pesticides

The most widely dispersed of all man-made toxic materials in the environment are the pesticides. Included in this rather heterogeneous group of compounds are agents designed to eliminate or control a variety of nuisance organisms. Many of the compounds are toxic or potentially toxic to most life forms while others are specific in their killing. Both inorganic and organic compounds are used.

Increased and frequently indiscriminate use of pesticides during the past 20 to 30 years has resulted in an ubiquitous low level residue of certain classes of these compounds in the total biosphere. Release of these agents to the environment comes about not only as a consequence of agricultural activity but also from manufacturing processes, accidental spills, and disposal of containers and unused or outdated agents.

In the United States approximately 900 chemicals are formulated into over 60,000 pesticidal preparations which include the insecticides, fungicides, herbicides and plant growth regulators (31). The majority of the pesticides in use today are synthetic organic compounds, however, inorganic pesticides and plant extracts are still used. The inorganic pesticides include such compounds as lead arsenate, calcium arsenate, copper sulfate, mercuric chloride and Paris Green. The advent of the more effective organic pesticides has caused a decline in the use of the inorganic pesticides.

Certain botanicals or plant extracts such as pyrethrum and rotenone are still in demand, as they are relatively safe to handle, are quite specific in their killing, and do not persist very long in the environment. These pesticides are widely used around livestock as they are relatively non-toxic to mammals (31).

The synthetic organic pesticides include the familiar chlorinated hydrocarbons or organochlorines such as DDT, dieldrin, chlordane and toxaphene. Also included are the organic phosphates (malathion, parathion, etc.), and the carbamate insecticides such as carbaryl (Sevin) and several fungicides, herbicides and defoliants.

In 1967 the United States production of all pesticides totaled 476.3 x 10^6 kg (31). Between the years 1964 and 1968 total pesticide production increased at the rate of 9 percent per year. However, recent data indicate that this trend has reversed, as total sales of synthetic organic pesticides were down 6.9 percent in 1971 from the 1969 total (32). Present trends suggest that the pesticide industry may be on a three-year plateau, after which sales are expected to increase at an unknown rate (32). The domestic use of DDT and other persistent pesticides has been declining in recent years, reflecting a shift to the use of the less persistent chlorinated hydrocarbons and organic phosphates. Between the years 1956 to 1970 domestic supplies of the chlorinated hydrocarbons dropped from nearly 110.8 million kg (244 million lbs) to about 14 million kg (31 million lbs). Conversely, during the same period, production of the organophosphates increased from 3.2 million kg (7 million lbs) to 25.9 million kg (57,000,000 lbs) (33). Recently the Administrator of the United States Environmental Protection Agency issued an order restricting the use of DDT primarily to Public Health Officials and physicians for the control of disease vectors, lice and for health quarantine purposes (34). This order, which became effective on January 1, 1973, may result in substantially increased use of other insecticides for insect control.

The major pathways of pesticides into the fresh water environment are through direct application on surface waters and from surface runoff (31). Industrial and domestic sewage, and fallout from atmospheric drift and precipitation also contribute to the contamination of waterways by pesticides.

Upon reaching a stream, downstream transport of pesticides occurs through movement of the solubilized fraction and residues sorbed onto suspended or saltated particles. As a result of downstream transport, pesticide concentrations in upstream reaches tend to diminish rapidly, while levels in the downstream reaches and in receiving lakes and reservoirs may be increased substantially.

Sediments of lakes and reservoirs, particularly those in eutrophic water bodies rich in organics, have a high affinity for pesticides, and act as sinks or pools for the residues. Consequently, pesticides may be removed from the water and incorporated into the bottom sediments fairly rapidly. If siltation rates are high, pesticides in lake sediments may be effectively isolated from the overlying waters and removed from involvement in the food web. On the other hand in lakes with lower siltation rates, sedimented pesticides may be taken up by the benthic biota, which is in turn consumed by fish and other predators and thus the pesticides are reintroduced into the food web. Pesticide entrapment in lake sediments may be only temporary and persist only during the period in which the lake is thermally stratified. Once turnover occurs, if mixing is complete, the pesticides may be released from the sediments and redistributed throughout the water.

The recovery rates of lakes treated with pesticides were studied in Oregon, where two mountain lakes were treated with the organochlorine, Toxaphene (35). One lake was deep and biologically unproductive and the other shallow and rich in aquatic life. The shallow lake recovered rapidly and trout were restocked within one year. Restocking of trout in the deep lake, however, was delayed for 6 years due to toxic levels of Toxaphene in the water. The reasons given for the slower recovery of the deep lake were thermal stratification, slower flow through time and reduced biological activity (35).

All organic pesticides are subject to degradation. With most pesticides, depending upon environmental conditions, degradation may be complete in a few days to a few months.

The organophosphates, for example, are readily hydrolized in alkaline water at high temperatures, however, at reduced pH and temperatures they persist for several months (36). The non-persistent pesticides, as with the organophosphates, although acutely toxic, do not pose long term hazards to aquatic life and apparently are not accumulated through the food chain. The organochlorine compounds, however, are highly resistent to degradation, or the degradation products may be persistent. These compounds may be accumulated by the biota directly from the water (37) or through the food chain, resulting in concentrations in the tissues of higher trophic level animals that may be several thousand times that found in the ambient waters.

That persistent pesticides are rapidly removed from the water and concentrated in the sediments and biota was demonstrated by Bridges et al (38) who described the dispersion and persistence of DDT in a farm pond. Sufficient quantities of DDT were applied to a pond to yield a 0.02 mg/l concentration in the pond water. The distribution of DDT in the water, sediments and biota was observed for about 18 months. DDT had disappeared from the water after 3 weeks. Maximum concentrations in the sediments of 8.30 mg/kg were recorded one day after treatment, but had declined nearly to pre-treatment levels after 8 weeks. Vegetation samples revealed maximum concentrations of 30.7 mg/kg one-half hour after treatment, and after eight weeks contained 5.1 mg/kg. DDT concentrations in the new vegetation crop, one year after application corresponded to post treatment levels. Accumulation in fish of DDT and its metabolites reached 3 to 4 mg/kg within 1 month after treatment. Concentrations in excess of 2 mg/kg were still present in fish when the study was terminated.

High level pesticide residues in lakes have posed problems in recent years by interfering with the reproductive patterns of fish or rendering them unfit for consumption due to excessive contamination. Concentrations of DDT exceeding 4.75 mg/kg in the eggs of lake trout resulted in up to 100 percent mortality in developing fry in New York lakes whose watersheds had been treated with DDT for gypsy moth control (39).

In Lake Michigan similar mortalities of coho salmon fry were attributed to DDT, dieldrin and PBC concentrations in the eggs (31). Reinert (41) found DDT and dieldrin in fishes from all the Great Lakes. Concentrations in Lake

Michigan fishes were found to be 2 to 7 times as high as those in fish from the other Great Lakes. Samples from canned coho salmon had DDT and dieldrin concentrations of 7.10 and 0.09 mg/kg respectively. Concentrations in adult salmon caught just prior to spawning exceeded 12 mg/kg DDT and 0.14 mg/kg dieldrin. Levels in excess of those established by the FDA have resulted in Lake Michigan coho, and several other species, being removed from the interstate market.

The behavior of pesticides in lake sediments and their availability for recycling back into the biota are not fully understood. Studies on the rates of interchange across mud-water interfaces and between the water and the biota are needed before the magnitude of the problem of pesticide pollution in lakes can be thoroughly assessed.

<u>Mercury</u>

The problems of mercury contamination in United States waterways were drawn to public attention in April 1970, when Canadian investigators reported mercury pollution in Lake

St. Clair and other boundary waters (42). Subsequent investigations by the United States Federal Water Quality Administration (now the Environmental Protection Agency) revealed that the mercury pollution problem was not limited to the Great Lakes area, but was of national scope (42).

Mercury is a particularly hazardous contaminant in aquatic systems, owing to its tendency to be transformed from a relatively immobile inorganic metal to a highly toxic organic form by the biological process of methylation. The methylation process is accomplished by certain aquatic bacteria living in the bottom muds (43), and all inorganic mercury introduced for aquatic systems is potentially subject to bacterial methylation, and subsequent uptake by the biota. Aquatic organisms are able to concentrate methylmercury directly from the water or through the food chain (42 - 47). In general mercury in fish food organisms increases at each trophic level of the food chain (48). A concentration factor of 5,000 or more from water to pike has been reported (49) and methylmercury magnification in brook trout has been shown to exceed 10,000 after long term exposure (50). Such factors as the metabolic rate, food selection and the epithelial surface area of the individual fish have been implicated as parameters which affect the rate at which mercury is concentrated by fish (44, 51).

The toxicity of mercury compounds to aquatic organisms has been summarized by various investigators with widely differing results. It is established, however, that the toxic level of mercury is affected by several aspects of water quality including termperature, pH, organic pollution loading, hardness, alkalinity, heavy metal loadings and dissolved oxygen (50).

In respect to toxicity in natural waters, it is methylmercury which is of primary concern. Experiments at the National Water Quality Laboratory indicate that 0.2 mg/l methylmercury will kill fathead minnows within 6 to 8 weeks (50). Toxicity data from the same laboratory on invertebrates, Gammarus and Daphnia, a top minnow and a brook trout is said to indicate than none are more sensitive that the fathead minnow (50).

Plankton is particularly sensitive to mercury poisoning. Exposure of phytoplankton to concentrations of 0.1 ug/l of methylmercury compounds caused a significant reduction in photosynthesis, and at levels of 0.50 ug/l photosynthesis was stopped (52).

Sources of mercury release to the environment include natural weathering, burning of fossil fuels, mining, farming, industrial operations, hospitals, laboratories and a host of others. Sources of mercury input to the environment, both man made and natural, are summarized by Lambou (42). The natural weathering process is said to release a maximum of 230 metric tons of mercury to the environment yearly, whereas the amount released by burning coal is on the order of 3,000 tons annually, and anotner 3,000 tons are emitted as industrial wastes (53).

Mercury pollution in the nation's lakes and rivers poses a serious public health threat and has restricted sports fishing and commercial fisheries operations in many areas. Table VIII summarizes data compiled by the United States Geological Survey on concentrations of total mercury found in many U. S. lakes and rivers. Concentrations of total mercury above the minimum detection limit of 0.5 mg/l were found in 140 of the 719 samples analyzed (42).

The problem of mercury pollution in lakes, particularly the Great Lakes, is of such a magnitude that many states imposed fishing restrictions or warnings of some type because of high levels of mercury in fish taken from contaminated lakes. Table IX summarizes State restrictions which were in effect as of September 1, 1970. Mercury levels in fish from selected areas of the Great Lakes are summarized in Table X. These data, based upon composite homogenized samples collected by the U. S. Fish and Wildlife Service (55) reveal relatively low total mercury residue levels in the upper Great Lakes fishes, with increasing concentrations in fishes taken in the lower Great Lakes. Average residue levels in the Lake Ontario fishes exceeded the 0.5 mg/kg level for edible portions established by the Food and Drug Administration.

TABLE VIII

Summary of total mercury measured in water samples from U.S. rivers and lakes obtained during October and November, 1970. 1/

State	Number of samples with ug/l .5 2/	.5-.9	1.0-1.9	2.0-2.9	3.0-3.9	4.0-4.9	5.0-5.9	6.0-6.9
Alabama	18	-	-	-	-	-	-	-
Alaska	8	-	1	-	-	-	-	-
Arizona	10	-	-	1	-	-	-	-
Arkansas	10	3	-	-	-	-	-	-
California	6	11	10	3	-	1	-	1
Colorado	17	2	-	-	-	-	-	-
Connecticut	24	1	-	-	-	-	-	-
Delaware	3	-	-	-	-	-	-	-
District of Columbia	1	-	-	-	-	-	-	-
Florida	8	4	3	1	-	-	-	-
Georgia	17	-	-	-	-	-	-	-
Hawaii	4	2	2	-	-	-	-	1
Idaho	5	1	1	-	-	-	-	-
Illinois	13	2	1	-	-	-	-	-
Indiana	19	2	-	-	-	-	-	-
Iowa	8	1	2	-	-	-	-	-
Kansas	-	4	4	2	1	1	-	-
Kentucky	3	3	1	-	-	-	-	-
Louisiana	11	1	1	-	-	-	-	-
Maine	6	1	-	-	-	-	-	-
Maryland	13	-	-	-	-	-	-	-
Massachusetts	8	6	-	-	-	-	-	-
Michigan	15	4	-	-	-	-	-	-
Minnesota	12	1	-	-	-	-	-	-
Mississippi	8	-	-	1	1	-	-	-
Missouri	9	-	2	1	1	-	-	-
Montana	8	-	-	-	-	-	-	-
Nebraska	7	2	1	-	-	-	-	-
Nevada	3	4	1	-	-	-	-	-
New Hampshire	3	1	-	-	-	-	-	-
New Jersey	18	-	-	-	-	-	-	-
New Mexico	15	-	-	-	-	-	-	-
New York	27	6	-	-	-	-	-	-
North Carolina	21	-	-	-	-	-	-	-
North Dakota	5	1	-	1	-	-	-	-
Ohio	9	8	7	-	-	-	-	-
Oklahoma	9	2	1	-	-	-	-	-
Oregon	13	-	-	-	-	-	-	-
Pennsylvania	42	1	-	-	-	-	-	-
Rhode Island	4	-	-	-	-	-	-	-
South Carolina	16	-	-	-	-	-	-	-
South Dakota	5	1	-	-	-	-	-	-
Tennessee	10	2	-	-	-	-	-	-
Texas	27	-	3	-	-	-	-	-
Utah	8	2	1	-	-	-	-	-
Vermont	3	-	-	-	-	-	-	-
Virginia	11	-	-	-	-	-	-	-
Washington	13	-	1	-	-	-	-	-
West Virginia	12	-	-	-	-	-	-	-
Wisconsin	15	-	1	-	-	-	-	-
Wyoming	9	-	-	-	-	-	-	-
Puerto Rico	10	-	-	-	-	-	-	-
Total	579	79	44	10	3	2	0	2

1/ Summarized from Durum et al, (1970).

2/ Below detection limit.

Source: (41)

TABLE IX

State fishing restrictions because of mercury -- September 1,1970

State	Closure of sport fishery	Closure of commercial fishery	Warning or catch and release for sport fishery	Embargo or warning to commercial fishery
Michigan	So. L. Huron, West L. Erie take no walleye, drum, white bass	Detroit R., L. St. Clair, St. Clair R. closed. So. L. Huron, West L. Erie closed to walleye, drum, white bass	Detroit R., L. St. Clair, St. Clair R. catch and release only	Embargo on species other than walleye, drum, white bass
Wisconsin			Wisconsin R., catch and release recommended; no more than 1 meal per week	
Ohio		L. Erie closed to walleye	Lake Erie - warning released via news	Embargo on white bass
New York	L. Onondago		L. Champlain, Erie, Ontario, Oswego R., Niagara R., St. Lawrence R. danger warnings	
Vermont			L. Champlain, L. Memphremagog, danger warning	L. Champlain, L. Memphremagog, embargo on sales
Pennsylvania			L. Erie, danger warning for walleye, drum, smallmouth bass, white bass	
Alabama		Tombigbee R. closed Mobile R., Tensaw R., Mobile-Tensaw system, Tennessee R. and impoundments, closed	Tombigbee R. up to Jackson Dam, warning Mobile R., Tensaw R., Mobile-Tensaw system Tennessee R. and impoundments, warning	
Mississippi		Pickwick L. closed	Pickwick L., warning	
North Carolina			Danger warning (general)	
Tennessee		Pickwick L. closed	Pickwick L., warning, catch and release	

Source: (41)

TABLE X

Mercury residues in fish, 1969 and 1970

Station Location	Species	1970 No. of Fish	1970 Average Size Length (cm)	1970 Average Size Weight (kg)	Total Mercury (mg/kg)	1970 No. of Fish	1970 Average Size Length (cm)	1970 Average Size Weight (kg)	Total Mercury (mg/kg)
				GREAT LAKES DRAINAGE					
Genessee River Scottsville, N.Y.	White sucker	4	35.6	0.54	.15	5	38.4	0.68	.13
	Redhorse sucker (R)	4	33.5	0.41	.19	5	18.3	0.14	.22
	Rock bass	4	20.6	0.23	.39	2	43.7	0.73	.25
	Walleye					2	43.7	0.73	.25
	Northern pike	4	35.0	0.32	.17				
					Avg. .24				Avg. .20
St. Lawrence River Massena, N.Y.	White sucker	3	43.7	0.68	.22				
	Yellow perch	5	17.8	0.09	.20				
	Yellow perch (R)	5	21.1	0.14	.18				
	Northern pike	4	52.3	0.95	.39				
					Avg. .27				
Lake Ontario Port Ontario N.Y.	Yellow perch	5	21.6	0.18	.86	4	26.4	0.27	.48
	Yellow perch (R)	5	21.3	0.18	1.00				
	White perch	5	21.1	0.18	1.30	5	24.1	0.23	.43
	Rock bass	5	16.5	0.09	.30	3	21.8	0.27	.65
					Avg. .84				Avg. .52
Lake Erie Erie, Pa.	White sucker	5	45.7	1.20	.31	3	37.6	0.68	.10
	Freshwater drum	5	35.8	0.59	.43	5	34.3	0.50	.15
	Yellow perch	5	24.6	0.18	.23	5	23.9	0.18	.13
	Yellow perch (R)	5	22.6	0.14	.15				
					Avg. .31				Avg. .13
Lake Huron Bay Port, Mich.	Carp	5	49.8	1.82	.07	5	41.1	0.95	.05
	Channel catfish	5	41.1	0.64	.07	5	40.4	0.68	.13
	Yellow perch	5	23.1	0.14	.08	5	25.1	0.23	.09
	Yellow perch (R)	5	9.1	0.3	.05				
					Avg. .07				Avg. .07
Lake Michigan Sheboygan,	Bloater	5	28.4	0.27	.09	5	30.5	0.36	.09
	Bloater (R)	5	23.9	0.86	.10				
	Yellow perch	5	27.9	0.27	.07	5	26.2	0.27	.27
					Avg. .09				Avg. .18
Lake Superior Bayfield, Wis.	Bloater	5	26.1	0.14	.15	5	28.5	0.18	.16
	Lake whitefish	5	44.7	0.77	.08	5	40.9	0.54	.05
	Lake whitefish (R)	5	46.0	0.86	.06				
	Lake trout	5	59.4	1.82	.29	4	55.9	1.36	.14
					Avg. .17				Avg. .10

Source: (41)

Polychlorinated Biphenyls (PCB's)

Recently, evidence has been compiled which indicates that the PCB's are widely distributed throughout the environment and that they can have adverse ecological and toxicological effects (54).

An Interagency Governmental Task Force (54) investigating the effects of PCB's in the environment concluded that PCB's present a potential, but not an imminent, health hazard, except for accidents which result in high level exposure. They have, however, been found in fish and wildlife at levels which may adversely affect aquatic organisms.

PCB's have been manufactured commercially since 1929. Historically PCB's in the United States were used in a variety of applications including plasticizers, hydraulic fluids and lubricants, surface coatings, inks, adhesives, pesticide extenders, and microencapsulation of dyes for carbonless duplicating paper. Beginning in 1971, however, the Monsanto Company reportedly reduced its production volume, limiting its distribution to industries concerned with the manufacture of electrical apparatus (54).

The water environment is thought to be the principal sink and transport mechanism for PCB's, but there are few data on the removal, disappearance and sequestering of the substance in soils or bottom sediments of rivers, lakes, estuaries or the ocean (54). Concentrations in fresh water away from any immediate source of waste discharges contain less than one ug/l; sediment samples contain up to several hundred mg/kg near some industrial outfalls.

PCB's are fat soluble and tend to be concentrated at succeedingly higher levels as they pass through the various steps of the food chain. They have been shown to accumulate in fish and aquatic invertebrates to levels of 75,000 times the ambient water concentration, and to be accumulated from concentrations as low as 0.05 ug/l (54).

PCB's are lethally toxic to fish and aquatic invertebrates in concentrations of a few ug/l. Metabolism and excretion of PCB's by these organisms is very slow (54). PCB's are only moderately toxic to birds and mammals and have not resulted in sufficient mortalities to affect populations, although they are thought to have contributed to direct mortalities of some birds in the field. The sublethal physiological effects on wild animals appear to be of greater significance than the lethal toxicity.

Phthalate Esters

Phthalate ester residues have been discovered in various segments of the aquatic environment in North America, occurring principally in water, sediment and aquatic organisms in industrial and populated areas (55). Phthalate esters are widely used as plasticizers particularly in polyvinyl chloride (PVC) plastics (50). They have also been used as insect repellents and in pesticide formulation to retard volatilization.

The acute toxicity of phthalate esters appears relatively significant. However, these compounds may be detrimental to aquatic organisms at low chronic concentrations. *Daphnia magna*, exposed to 10 mg/l of ^{14}C di-n-butyl phthalate showed a magnification of 6000 fold. Upon transfer of the organisms to uncontaminated water, however, approximately 50 percent of the material was excreted within three days (49).

Arsenic

Arsenic compounds in the lake environments pose potential hazards to aquatic life and wildlife and even to man. Arsenic enters waterways through various routes including industrial and municipal waste discharges, mine drainage, pesticides, lead shot, coal burning and smelting of ores (55). Many detergents and laundry products contain arsenic and their discharge in waste effluents contributes substantially to arsenic contamination of waterways as most sewage treatment plants do not remove arsenic (56).

Arsenic was frequently applied to lakes and ponds for the control of submerged aquatic vegetation. In the period from 1950 through 1962, over 4.54×10^5 kg (1 million pounds) of arsenic trioxide were applied to Wisconsin lakes for weed control (57). In Minnesota nearly 4.31×10^4 kg (95,000 pounds) of arsenic trioxide were applied for submerged aquatic plant control in 1958 (57). Michigan and other states also reported using arsenic trioxide as a weed control agent, but in unknown quantities.

It is known that arsenic can be biologically concentrated and magnified in the food web (58) as well as accumulated in lake bottom muds (59). Some concentration factors for certain marine organisms were given by Lowman (58) as follows: Benthic algae, 2000; mollusc muscle, 650; crustacean muscle, 400; and fish muscle, 700. Concentration in bottom samples taken in a treated lake ranged from 10 to 82 mg/kg (60). Dupree (59) studied the arsenic content of the water, soil and biota of lakes which had been treated with soil arsenite and subsequently drained and refilled 2 to 3 times. The following year after treatment the sodium arsenate content of the water ranged up to 0.3 mg/l, in plankton up to 7.4 mg/kg, and in bottom soil up to 0.38 mg/kg. These data suggested that arsenic could be released from bottom muds providing a source to the water and biota for a considerable period after application (59).

The literature on the toxicity of arsenic is rather confusing. Arsenic is toxic to all animals with a central nervous system and to most higher plants, but may not be toxic to lower organisms (56). The toxicity of arsenicals is influenced by the form in which it is accumulated. The organic compounds which may reside in bottom sediments are less toxic to man than the inorganic compounds, and the pentavalent compounds (arsenates) are generally much less toxic than the trivalent arsenicals (arsenites).

Arsenic trioxide, a common aquatic weed control agent, has been found to be harmful to fish food organisms in concentrations as low as 2.0 mg/l over an unspecified length of time (56). Conversely, concentrations as high as 17.1 mg/l have been tolerated by minnows for one hour with no harmful effects, and 10.0-20.0 mg/l were tolerated by insect larvae for an unspecified period of time without apparent damage (56).

Sodium arsenite applied to experimental ponds in concentrations of 4 mg/l substantially reduced the numbers of bottom organisms and reduced bluegill production. A 4 mg/l application also killed microcrustacea and greatly reduced the rotifer population (56).

Because the relatively insoluble arsenicals are present in many waterways, potential hazards to those forms which accumulate arsenic, exist. Arsenic builds up slowly in the body and, according to some medical sources, long term arsenosis may not be detectable for two to six years or more (56).

Ammonia and Sulfides

Both ammonia and sulfides are potentially toxic substances which are discharged from a wide variety of industrial processes as well as municipal sewers.

In unpolluted lakes ammonia and sulfides are usually present in low concentrations. However, in lakes receiving decaying organic waste loads or with high natural organic sediment content, the biological production of ammonia and hydrogen sulfide in unusually high concentrations may pose potential toxicity problems.

During the summer stagnation periods the concentration of free ammonia and hydrogen sulfide in lakes generally increases with depth. The bottom ooze may contain many times the concentrations found in the overlying waters. The development of isothermal conditions and subsequent mixing tends to distribute the dissolved gases throughout the water column. Consequently ammonia and hydrogen sulfide concentrations in the bottom waters are usually lowest during the periods of spring and fall overturn.

The toxicity of both ammonia and sulfide is determined to a large extent by the pH of the water. Gaseous ammonia is readily soluble in water forming ammonium hydroxide which dissociates into ammonium and hydroxide ions in a pH dependent reaction. The toxic component of ammonia solution is non-ionized ammonia. Since the percentage of non-ionized ammonia increases with increased pH, the toxicity of the solution does also (50). Sulfides derive their toxicity from hydrogen sulfide which is formed by reaction with the hydrogen ion when added to water. Hydrogen sulfide dissociates in solution yielding the HS and H ions, and the higher the pH the more complete the dissociation reaction, therefore at higher pH values toxicity is reduced. Numerous other factors such as temperature, dissolved oxygen tensions and free carbon dioxide concentration also influence the rate of the reactions involving these substances, hence influencing the toxicity.

Toxicity problems arising from excessive concentrations of ammonia and hydrogen sulfide are more common in streams, particularly those with a heavy industrial or municipal water loading, than in lakes. The potential for toxic problems exists in lakes, however, particularly in those with high organic content in the sediments. In shallow northern lakes toxic levels of ammonia may develop under heavy ice cover, and in combination with low oxygen tensions contribute to stress conditions for aquatic life and in some cases result in heavy fish mortalities.

Miscellaneous Problems

Non-Toxic Salts

In the northern United States the practice of applying salts to streets and roads to control ice accumulations has become increasingly common. During the past few decades the amount of salt (mostly sodium chloride) used for deicing purposes has increased exponentially, nearly doubling every five years (61). During the winter of 1969-70 an estimated 7,700,000 metric tons of salt were used for deicing purposes (61, 62).

Much of the salt used for deicing purposes is carried off in melt waters and transported to lakes via storm sewers, ground and surface waters. As a consequence of the salt influx, the physical and chemical characteristics of the lakes may be changed substantially resulting in significant ecological alterations and impairing the lake's utility as a resource. Such is the case in Irondequoit Bay, near Rochester, New York.

The 435 km^2 Irondequoit Bay drainage basin, with a 1970 population of 206,000 receives approximately 1 percent (77,000 metric tons) of the deicing salt used in the United States (61, 62). Irondequoit Bay is connected to Lake Ontario by a shallow channel, but little exchange of the deeper bay water with the lake occurs. The surface area of the Bay is 6.7 km^2 and maximum depth is 23 m (61).

During the winter of 1969-70, approximately 10 metric tons of salt were stored in the Bay, while 11,000 metric tons went out the outlet. Approximately one half of the 77,000 metric tons applied to the roads were stored in soil and ground water, part of which will eventually reach the Bay (61).

The winter influx of salt resulted in the development of a vertical density gradient sufficient to prevent the Bay from mixing during the 1970 and 1971 spring seasons. It also prolonged the period of summer stratification by about one month in the fall seasons of 1969 and 1970 (as compared to the fall of 1939) (61, 62).

The full ecological consequences of the artificial disruption of the circulation patterns due to salt influx are not known. One effect is to prolong the anaerobic conditions of the bottom waters. In a normal dimectic lake anoxic bottom waters are replenished with oxygen during both the spring and fall turnover. Due to the lack of a complete spring mixing period, the hypolimnetic water of Irondequoit Bay remain anaerobic for about 9 months of each year.

It is not presently known how many of the Nation's northern lakes are similarly affected by salt runoff, as the problem has received little attention until recent years. Present trends in uses of deicing salts suggest that the potential for serious problems may be developing.

Radioactive Wastes

The development of the nuclear power generating plant, with its dependence upon large volumes of cooling water, has introduced yet another form of contaminant to the lake environment - radioactive material. As the number of nuclear generating stations increases, the number of nuclear fuel reprocessing plants will also increase, some impacting on lakes. The parallel development of these facilities will increase the potential for radionuclide contamination of freshwater lakes.

Radioactive wastes create a unique environmental problem in the form of ionizing radiations of varying energies, but the primary consideration is the potential for human exposure to these radiations. In this regard, radionuclides of concern in the aqueous environment include cesium, cobalt, iodine, strontium, tritium, and plutonium.

Consequences of Release of Radioactive Wastes

While many radioactive wastes are of very short half-life and low energy, others present problems because of their persistence in the aquatic environment (e.g., ^{129}I, ^{137}Cs), reconcentration potential in aquatic food chains leading to man, and subsequent toxicity to man. Bioconcentration of radioiodine (^{131}I) is of special concern in this respect since it is readily metabolized and concentrated in the thyroid, and may become a significant hazard via the cow-milk-child pathway. In addition to presenting a potential threat to the biota itself, bioconcentrated radionuclides could render food sources such as fish unsafe for human consumption. Significant quantities of soluble radioactive materials would also endanger lakes used as municipal water supplies. Discussions concerning bioconcentration of radionuclides, and their transfer through aquatic food chains are contained in respective publications of the Lawrence Livermore Laboratory (63) and the National Academy of Sciences (64).

The virtual non-removability of radioactive materials in the aqueous environment coupled with the problem of radionuclide reconcentration in the biota necessitates careful control of nuclear facilities which release radioactive wastes in the vicinity of freshwater lakes.

REFERENCES

1. Anon. 1971. Water use in manufacturing. 1967 census of manufacturing. U. S. Dept. of Commerce.

2. Powers, T. J. 1967. National Industrial Waste Assessment. U.S. Department of Commerce.

3. Parker, F. L. and P. A. Krenkel. 1969. Thermal pollution: Status of the Art. Vanderbilt University Nashville, Tenn. Report No. 3.

4. Environmental quality. 1970. First Annual Report of the Council on Environmental Quality, pp. 30-39.

5. Anon. 1968. Industrial waste guide on thermal pollution. Federal Water Pollution Control Federation, U.S. Department of Interior.

6. Anon. 1970. Clean water for the 1970's. Federal Water Quality Administration, U.S. Department of the Interior.

7. Happ, S. C. 1941. Sedimentation in artificial lakes. In: A symposium on hydrobiology. Wisconsin Univ. Press, Madison, Wis.

8. European Inland Fisheries Advisory Commission. 1965. Working part on water quality criteria for European freshwater fish. Report on finely divided solids and inland fisheries. Internat. J. Air Water Pollution 9:151-168.

9. Buck, H. D. 1956. Effects of turbidity on fish and fishing. Trans. N. Amer. Wildlife Conf. 21:429-61.

10. Gumerman, R. C. 1970. Aqueous phosphate and lake sediment interaction. Proceedings 13th Conference on Great Lakes Research, part 2. Great Lakes Research Center, Detroit, Mich.

11. Olsen, S. 1964. Phosphate equilibrium between reduced sediments and water. Verh. Int. Ver. Limnol., Copenhagen Univ., Denmark 15:333-341.

12. Fredricksen, R. L. 1970. Erosion and sedimentation following road construction and timber harvest on unstable soils in three small western Oregon watersheds, Forest Service Research Paper PNW-104.

13. Collier, C. R. 1970. In: Influences of strip mining on the hydrologic environment of parts of Beaver Creek Basin, Kentucky, 1955-1966, Geological Survey Professional Paper 427-C, P C31-C46.

14. Yorke, T. H. and W. J. Davis. 1971. Effects of urbanization on sediment transport in Bel Pre Creek Basin, Maryland. In: Geological Survey Research 1971, Chapter B, Professional Paper 750-B, P B218-B223.

15. Yorke, T. H. and W. J. Davis. 1972. Sediment yields of urban construction sources, Montgomery County, Maryland, Geological Survey Open-File Report.

16. Anderson, P. W. and J. E. McCall. 1968. Urbanization's effect on sediment yield in New Jersey, J. Soil and Water Conserv. 23:142-144.

17. Wolman, M. G. and A. P. Schick. 1967. Effects of Construction on fluvial sediment, urban and suburban areas of Maryland. Water Resources Res. 3:451-464.

18. Guy, H. P. and G. E. Ferguson. 1970. Stream sediment: an environmental problem. J. Soil and Water Conservation 25:7-22.

19. Vice, R. B., H. P. Guy, and G. E. Ferguson. 1969. Sediment movement in an area of suburban highway construction, Scott Run Basin, Fairfax County, Virginia, 1961-64. Geol. Surv. Water-Supply Paper 1591-E, P. E1-E41.

20. Hatheway, E. S. 1927. The relation of temperature to the quality of food consumed by fishes. Ecology 8:428-434.

21. Fry, F. E. J. and J. S. Hart. 1948. Cruising speed of goldfish in relation to water temperature. J. Fish. Res. Bd. Can. 7:169-175.

22. Brett, J. R., J. E. Shelbourn and C. T. Shoop. 1967. The relation of temperature and food ration to the growth rate of young sockeye salmon. In: Abstracts of Paper at the American Fisheries Society 97th Annual Meeting. Toronto, Ontario, Canada.

23. Black, D. S. 1969. Keynote address. In: P. A. Krenkel, and F. L. Parker (ed.). Biological aspects of thermal pollution. Vanderbilt Univ. Press, Nashville, Tenn.

24. Wurtz, C. B. 1969. The effects of heated discharges on freshwater benthos. In: P. A. Krenkel, and F. L. Parker (ed.). Biological aspects of thermal pollution. Vanderbilt Univ. Press, Nashville, Tenn.

25. Mayer, A. G. 1914. The effects of temperature upon tropical marine animals. Pap. Tortogas Lab. 6:1-24.

26. Hedgpeth, J. W. and J. J. Gonor. 1969. Aspects of the potential affect of thermal alteration on marine and estuarine benthos. In: P. A. Krenkel, and F. L. Parker (ed.). Biological Aspects of Thermal Pollution. Vanderbilt Univ. Press, Nashville, Tenn.

27. Orton, J. H. 1920. Sea-temperature, breeding and distribution in marine animals. J. Mar. Biol. Ass. U. K. 12:339-336.

28. Gray, J. 1928a. The growth of fish. II. The growth rate of the embryo of <u>Salmo fario</u>. J. Exp. Biol. 6:110-124.

29. Gray, J. 1928b. The growth of fish. III. The effect of temperature on the development of the eggs of <u>Salmo fario</u>. J. Exp. Biol. 6:125-130.

30. Patrick, R. 1969. Some effects of temperature freshwater algae. In: P. A. Krenkel, and F. L. Parker (ed.). Biological aspects of thermal pollution. Vanderbilt Univ. Press, Nashville, Tenn.

31. Mrak, E. M. (Chairman). 1969. Report of the secretary's Commission on pesticides and their relationship to environmental health. Parts I and II. U. S. Department of Health, Education and Welfare. December 1969.

32. Johnson, O. 1972. Pesticides '72. Chemical Week 110:34.

33. Anon. 1972. Environmental indicators for pesticides. Stanford Research Institute.

34. Ruckelshaus, W. D. 1972. Consolidated DDT hearings: Opinion and order of the Administrator, Environmental Protection Agency, concerning the registrations of products containing the insecticide DDT. Federal Register 37:13369-14476.

35. Terrier, L. C., U. Kiigemagi, A. R. Gerlack and R. L. Borovilka. 1966. The persistence of toxaphene in lake water and its uptake by aquatic plants and animals. J. Agr. Food Chem. 14:66-69.

36. Gakstatter, J. L. and C. M. Weiss. 1965. The decay of anticholinesterase activity of organic phosphorus insecticides on storage in waters of different pH. Proceedings Second International Water Pollution Research Conference, Tokyo. 1964. pp. 83-95.

37. Wilkes, F. G. and C. M. Weiss. 1971. The accumulation of DDT by the dragonfly nymph, Tetragoneuria. Trans. Amer. Fish. Soc. 100:222-236.

38. Bridges, W. R., B. J. Kallman and A. K. Andrews. 1963. Persistence of DDT and its metabolities in a farm pond. Trans. Amer. Fish. Soc. 92: 421-427.

39. Burdick, G. E., E. J. Harris, H. J. Dean, T. M. Walker, J. Skea, and D. Colby. 1964. The accumulation of DDT in lake trout and the effect on reproduction. Trans. Am. Fish. Soc. 93: 127-136.

40. Johnson, H. E. and C. Pecor. 1969. Coho salmon mortality and DDT in Lake Michigan. Trans. 34th N. A. Wildlife and Nat. Res. Conf. pp. 157-166.

41. Reinert, R. E. 1970. Pesticide concentrations in Great Lakes fish. Pest. Mont. J. 3:233-240.

42. Lambou, V. W. 1972. Problems of mercury emissions into the environment of the United States. Report to the Working Party on Mercury, Sector Group on Unintended Occurrence of Chemicals in the Environment, OECD, Environmental Protection Agency.

43. Erl, T. B. 1970. Methyl mercury poisoning in fish and human beings. Modern Medicine 38:135-141.

43. Johnels, A., T. Westermark, W. Berg, P. I. Perssonant, B. Sjostrand. 1967. Pike (Esox lucius L.) and some aquatic organisms in Sweden as indicators of mercury contamination in the environment. Oikos 18:323-333.

45. Hannerz, L. 1968. Experimental investigations on the accumulation of mercury in water organisms. Fisheries Board of Sweden, Institute of Fresh Water Research, Drottningholm, Report No. 48. pp. 120-126.

46. Hasselrot, T. B. 1968. Report in current field investigations concerning the mercury content in fish, bottom sediment, and water. Fisheries Board of Sweden, Institute of Fresh Water Research, Drottningholm, Report No. 48. pp. 102-111.

47. Miettinen, V., E. Blankenstein, K. Rissanen, M. Tillander, J. K. Miettinen and M. Valtouen. 1970. Preliminary study on the distribution and effects of two chemical forms of methyl-mercury in pike and rainbow trout. FAD Technical Conference on Marine Pollution and its Effect on Living Resources and Fishing. Rome, Italy. December 9-18.

48. Hamilton, A. 1971. Mercury levels in Canadian fish. Paper by E. B. Bligh In: Mercury in man's environment. Royal Society of Canada; 395 Wellington St., Ottawa, Ontario. KIAON 4.

49. Johnels, A. G. 1971. Mercury: observed levels and their dynamics in the environment. Results from Sweden. In: Mercury in man's environment, Proc. Symposium February, 1971. Royal Society of Canada, Ottawa. pp. 62-72.

50. Unpublished Report, Environmental Protection Agency by the National Academy of Sciences. Water quality criteria, Section III, Freshwater Aquatic Life and Wildlife, Final Draft. July 22, 1972.

51. Wobeser, G., N. O. Nielsen and R. H. Dunlap. 1970. Mercury concentrations in tissues of fish from the Saskatchewan River. J. Fish. Res. Bd. Canada 27:830-834.

52. Harris, R. C., D. B. White and R. E. Macfarlane. 1970. Mercury compounds reduce photosynthesis by plankton. Science 170:736-737.

53. Joensuu, O. I. 1971. Fossil fuels as a source of mercury pollution. Science 172:1027-1028.

54. Interdepartmental Task Force on PCB's. 1972. Polychlorinated biphenyls and the environment. National Technical Information Service. Com-72-10419. 181 pp.

55. Stalling, D. L. 1971. Proc. 2nd Annual International Congress of Pesticide Chemistry, Tel Aviv, Feb. 22-26.

56. Lambou, V. and B. Lim. 1970. Hazards of arsenic in the environment, with particular reference to the aquatic environment. Fed Water Qual. Adm., U. S. Dept. of Interior.

57. Mackenthun, K. M., W. M. Ingram and R. Porges. 1964. Limnological aspects of recreational lakes. U.S. Dept. of Health, Education and Welfare, Public chemical elements in edible aquatic organisms. Lawrence Livermore Laboratory. UCRL-50564 Rev. 1 (Also avail. from AEC as TID-4500, US-48).

64. National Academy of Sciences Report. 1973. Radionuclides in foods. Pub. No. ISBN 0-309-02113-8.

Health Service, Cincinnati.

58. Lowman, F. G., et al. 1970. Accumulation and redistribution of radionuclides by marine organisms. Bureau of Comm. Fish., USDI. Unpublished.

59. Dupree, H. K. Arsenic content of water, plankton, soil and fish from ponds treated with sodium arsenate for weed control. Proc. Southeastern Assoc. of Game and Fish Comm.

60. Mackenthun, K. M. 1961. Status of arsenic studies. Memorandum to the Chairman of Subcommittee on Aquatic Nuisance Control, Wisconsin Committee on Water Pollution.

61. Bubeck, R. C., W. H. Diment, B. L. Deck, A. L. Baldwin and S. D. Lipton. 1971. Runoff of deicing salt: Effect on Irondequoit Bay, Rochester, New York. Science 172:1128-1132.

62. Bubeck, R. C., W. H. Diment and A. L. Deck. 1971. Some limnological changes due to deicing salt runnoff in Irondequoit Bay, Rochester, N.Y. Abstr. Geological Society of American Animal Meeting, November 1971, Washington, D. C.

63. Thompson, S. E., C. A. Burton, D. J. Quinn and C. N. Yook. 1972. Concentration factors of chemical elements in edible aquatic organisms. Lawrence Livermore Laboratory. UCRL-50564 Rev. 1 (Also avail. from AEC as TID-4500, US-48).

64. National Academy of Sciences Report 1973. Radionuclides in foods. Pub. No. ISBN 0-300-02113-8.