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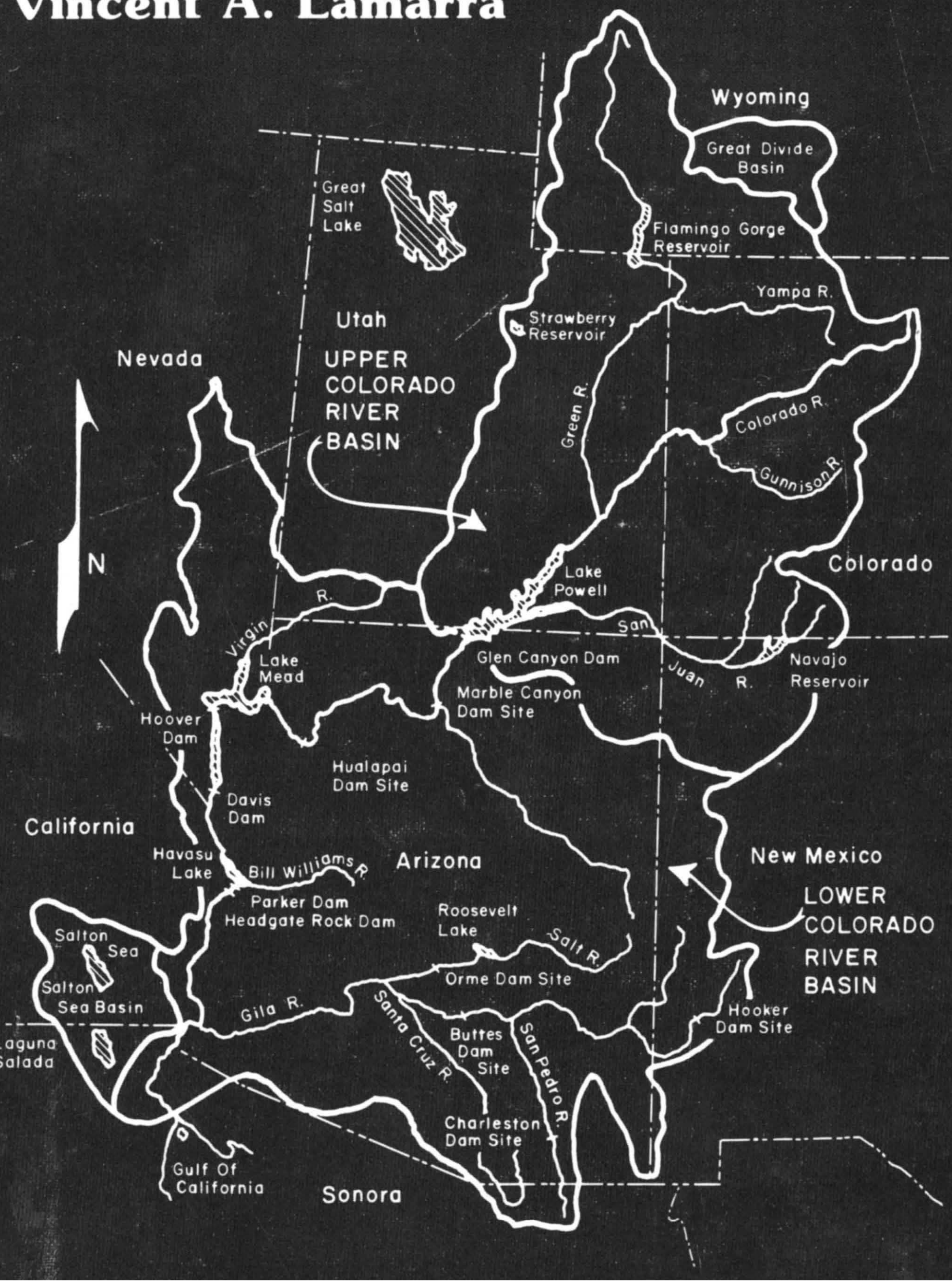

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Edited by

V. Dean Adams
Vincent A. Lamarra

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of the Colorado River Ecosystem,
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PREFACE

The Colorado River system has often been referred to as "the most regulated river system in the world." The Colorado River Basin serves millions of people through agricultural, energy, municipal and industrial uses, fish and wildlife activities, and recreation. The symposium was conceived and organized to allow researchers, private industry, consultants, water users, regulatory agencies, and concerned citizens the opportunity to express needs, desires, and concerns about the vast resources of the Colorado River.

We found that there were a diverse number of problems confronting the individuals who are involved in the management of this important ecosystem. A variety of broad topics have been presented which include: water policy and major diversions; energy impacts; oil shale development--resources and impacts; Lake Mead and the other major reservoirs in the system; the ecology and management of the watershed and the riparian habitat in the system; fisheries; salinity problems; sedimentation; eutrophication; flow depletion; and water augmentation.

This timely symposium brought together many individuals, representing a variety of disciplines, to discuss and transfer information appropriate to the needs of the Colorado River Basin. The results of this symposium, which have been compiled herein, are an attempt to examine current and projected effects of water and land management within the Colorado River Basin and to provide a basis for determining what can be done to better manage the resources within the total context of activities affecting the Colorado River Ecosystem.

V. Dean Adams

ACKNOWLEDGMENTS

A project such as this cannot be accomplished without the outstanding help, cooperation, and assistance of many individuals. We want to thank Ms. Mary Ann Nielsen for the tremendous effort she put forth in organizing the initial phases of the symposium and proceedings. Ms. Kathy Bayn also provided invaluable assistance, contributing to the peer review evaluations, editing, typing and analyzing the proceedings, and overseeing the compilation of all the manuscripts. We owe a tremendous debt of gratitude to both of these women. Ms. Donna Lake also contributed significantly to preparing the final manuscript for publication. We wish to recognize her efforts and also the efforts of the many typists, assistants, and graduate students at the Utah Water Research Laboratory who helped with the technical aspects of presenting the symposium, as well as the proceedings.

The Office of Water Research and Technology and the Utah Water Research Laboratory were sponsors of this project and have our sincere appreciation, for without them, a project such as this would not have been possible. Special recognition is given to Dr. L. D. James, Director of the Utah Water Research Laboratory for his support and contribution to the symposium and proceedings.

Much of the success of the symposium we owe to our session chairpersons whom we thank for their leadership, research efforts, and time spent in preparation for the sessions. These session chairpersons were: Dr. J. G. Carter, Dr. A. B. Davis, Dr. J. G. Dickson, Dr. E. R. Harris, Dr. L. D. James, Dr. A. J. Medine, Mr. J. B. Miller, Dr. L. J. Paulson, and Ms. M. E. Pitts.

We also want to thank all of the peer reviewers for the time and effort they gave in their attempt to evaluate and suggest improvements for the submitted papers. Their efforts have definitely enhanced the quality of this publication.

Finally, we express our gratitude to the invited speakers, the panelists, and the many contributing authors, who, by their research efforts, presentations, and written subject matter have provided us with excellent material regarding the Colorado River System, its resources and management.

V. Dean Adams is Associate Professor and Head of the Division of Environmental Engineering, Utah Water Research Laboratory, Utah State University. He received his BS in Chemistry from Idaho State University and his PhD in Organic Chemistry from Utah State University. Dr. Adams has been involved in numerous research projects, including effects of organic ligand complexation on heavy metals, evaluation of commercially available home water purifiers, and blue-green algae control in the protection of reservoir water quality against toxic organics. He is the author of several publications, technical reports and papers, and co-author of Analytical Procedures for Selected Water Quality Parameters. Dr. Adams is a member of many professional societies, including the American Chemical Society, Water Pollution Control Federation and Utah Water Pollution Control Association.

Vincent A. Lamarra is Adjunct Professor, Department of Civil and Environmental Engineering, Utah State University, and Co-director of the Ecosystems Research Institute. He received his BS in Natural Science from Fresno Pacific and his PhD in Limnology from the University of Minnesota. Dr. Lamarra is currently involved in several research projects, including the use of power plant effluent waters in fish aquaculture, and biological and chemical effects of hatchery effluent on stream ecosystems. He is the author or co-author of several publications and papers. Dr. Lamarra is a member of the American Society of Limnology and Oceanography, American Fisheries Society, International Association of Theoretical and Applied Limnology, and Lake Management Society.

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PART 1

KEYNOTE SPEAKERS

CHAPTER 1

THE COLORADO, A RIVER FOR MANY PEOPLE

Bill Plummer
Bureau of Reclamation
Boulder City, NV

The Colorado River is a life-sustaining water resource that winds more than 1,400 miles through seven stages and two countries. Because this ribbon of water descending from the snowcapped Rockies to Mexico's Gulf of California is the primary source of water for much of the basin it drains, the wellbeing of many communities in the basin is directly related to management of the river.

The Colorado's drainage basin encompasses 242,000 miles^2 in the United States, or one-twelfth of the country's land area, and 2,000 miles^2 in Mexico. Within the basin, the River's waters are used for irrigation, municipal and industrial purposes, hydroelectric power generation, fish and wildlife enhancement, and recreation. In addition, some of its waters are exported outside the basin to densely populated metropolitan areas.

Many water resource people have labeled the Colorado "the World's most regulated river." This need for control is the result of: 1) the scarcity of water in the areas served by the River, and 2) a long history of competition and struggle for this resource.

Use of the River's waters is regulated by various legislative and other legal acts -- known collectively as the "law of the river"-- that have been implemented through the years. The Secretary of the Interior operates the Colorado, in consultation with the seven basin states, according to the mandates of these documents.

The first of these major documents was the Colorado River Compact, dated November 24, 1922. This compact divided use of the River's water between the upper and lower basins at a point about a mile below historic Lee's Ferry, near the Paria River, in northern Arizona. In essence, it apportioned 7.5 million acre-feet of water annually to both the Upper and Lower Basins and paved the way for construction of works to control, regulate, and utilize the stream.

In 1928, the Boulder Canyon Project Act, which authorized Hoover Dam, apportioned the Lower Basin's 7.5 million acre-foot entitlement in a manner which annually provides California with 4.4 maf; Arizona with 2.8 maf, and Nevada with 0.3 maf. This apportionment was reaffirmed by a 1964 United States Supreme Court decision.

In 1944, the United States entered into an international treaty with Mexico which assured that country 1.5 maf of Colorado River water annually. The Upper Colorado River Compact, signed in 1948, permitted Arizona to use 50,000 acre-feet of water annually from the Upper Basin, and apportioned the remaining water among the Upper Basin states. By percentage, that distribution was: Colorado, 51.75 percent; New Mexico, 11.25 percent; Utah, 23 percent; and Wyoming, 14 percent. This compact also provided that the Upper Basin states could divert more than their entitlement if return flows were sufficient to make up the delivery requirement to the Lower Basin states and Mexico.

After all these allocations had been made, the availability of water began to be questioned. The 1922 apportionment between the basins had been based upon river data collected between 1906 and 1921 -- 15 years which now appear to have provided the system with more water than might be expected for the long-term average runoff.

The implications of this revised riverflow information soon became obvious. After delivering the guaranteed 7.5 maf average annual release to the Lower Basin, the Upper Basin may or may not have 7.5 maf available for use.

As it became evident that less water was available than earlier supposed, as agricultural and municipal water supply projects became a reality, and as population and use rates in the southwest soared, prudent management of the Colorado became an absolute necessity. Its waters, after all, are used for many purposes, and serve many diverse interests, throughout the Basin.

For instance, more water is delivered for agriculture in the basin than for any other need. But because of a shorter growing season, the agriculture of the Upper Basin is generally less intensive. In the Lower Basin, agriculture is almost entirely dependent on Colorado River water. The availability of this water, coupled with a year-round growing season, has resulted in some of the world's most productive farmland.

Municipal and industrial water is also a need in both basins. However, the demands for such water in the Lower Basin are currently about 10 times greater than the demands of the Upper Basin. When the Central Arizona Project begins delivering water to Phoenix and Tucson in a few years, and the Southern Nevada Water Project delivers additional water to Las Vegas, that figure may go higher.

Flood control is also a need of both basins. Here again, mainstem flooding of the Colorado has been a far more serious problem on the Lower Basin, particularly at the lower end of the River.

Hydroelectric power generation is another use for the River's waters in both basins, although this is considered an important by-product of the storage and delivery of water for other purposes. During 1980, the Bureau's hydroelectric powerplants on the Colorado River and its tributaries generated 13 billion kilowatt-hours of energy -- enough to supply the needs of 4.3 million people for one year. Much of this power was generated in response to peak demands for electricity. Hydroelectric plants are extremely valuable sources of electricity because of their ability to provide immediate peaking power without costly warmups.

Recreation is an important fringe benefit of our water resource projects in both basins. Two of the most significant recreation areas are Lake Powell, in the Upper Basin, and Lake Mead, in the Lower Basin. Together, these areas attracted approximately seven and a half-million visitors in 1980.

The Colorado River and its adjacent riparian areas continue to provide valuable habitat for fish and wildlife. Trout, largemouth and striped bass, and channel catfish are the dominant gamefish population in the river basin. The Colorado River flyway has long been recognized as a major migration and wintering area for many game and nongame species of birdlife. Working with the Fish and Wildlife Service and state and local agencies, Reclamation has helped improve fish and wildlife habitat along selected sections of the river. Beal Slough, a filled backwater renovated to enhance fish and wildlife values on the Lower Colorado River near Needles, California, is an example of this type of work. Modification of the powerplant intakes at Flaming Gorge Dam in Utah is another example. This work was performed to help restore the blue-ribbon trout fishery on the Green River below the dam.

It is obvious from these very brief user summaries that "managing" the Colorado River means different things to different people. To some, it means a life-sustaining supply of water, to others, flood control for protecting their property, and to still others, it means creation or enhancement of significant recreational resources.

All Colorado River water users would probably agree that management of this river for many people has changed it from a natural menace to a national resource. What some of them overlook is that the benefit they derive from the River is just one of many provided by our multiple-use management programs. The use problems stem from the fact that all of these benefits cannot be fully satisfied without some conflict.

Solving these conflicts is a difficult task, but not an insurmountable one. Reclamation does have defined responsibilities for managing the River. And we perform the task, without owning a drop of the administered water, for the benefit of the people comprising the communities and states of the Basin. In performing this task, we coordinate and consult a great deal with other Federal agencies, state agencies, water users, and other interested parties.

Consider Reclamation's responsibilities and priorities for managing the Colorado River. Current operation of the Colorado River by the Bureau of Reclamation is based largely on forecast of runoff, available storage, and requirements or demand for water -- all according to applicable laws.

As required by the Colorado River Basin Project Act, operation of Reclamation reservoirs in the Basin is coordinated under long-range criteria issued in June, 1970. These criteria state that the objective shall be to maintain a minimum release of 8,230,000 acre-feet of water from Lake Powell annually, and also state that a reservoir operating plan must be developed annually for the Colorado River.

Under these criteria, the Secretary of the Interior determines how much water must be retained in Upper Basin reservoirs each year in order to meet obligations to the Lower Basin without impairing the Upper Basin's consumptive uses. When Upper Basin storage is greater than the amount needed, releases above the minimum are made to maintain, as near as possible, active storage in Lake Mead equal to active storage in Lake Powell.

A third facet of the criteria is that they provide that all reasonable consumptive use requirements of all mainstem users in the Lower Basin will be met without cutback until such time as deliveries commence from the Central Arizona Project.

Releases in excess of downstream water requirements were made in 21 of the 27 years of operation between completion of Hoover Dam and completion of Glen Canyon Dam. With closure of Glen Canyon Dam in March, 1963, the storage capability of the Colorado River reservoir system was essentially doubled. While Lake Powell was filling, essentially all excess water in the Colorado was put into storage -- an annual average of two million acre-feet. However, a combination of three successive years of above average flow, coupled with the June 1980 filling of Lake Powell, resulted in nearly five million acre-feet of water in excess of downstream requirements being released from lower Colorado River dams from May, 1979 to January, 1981.

These excess releases were made in accordance with provisions of the Boulder Canyon Project Act. This legislation tends to alleviate one of the Lower Basin's most pressing management conflicts: when water should be stored for future use, and when water should be released to provide flood storage space in the reservoir.

Basically, the Boulder Canyon Project Act states that flood control will be the number one priority in operating Hoover Dam. Water storage and delivery and hydroelectric power generation have lesser priority.

The criteria for operating Hoover Dam under flood control conditions have been developed jointly by Reclamation and the U.S. Army Corps of Engineers. These criteria are reviewed and modified from time to time as conditions warrant. A public involvement program was conducted in 1979 to obtain updated input from the many people affected by the Dam's operation. A report citing the findings of this program should be published within the next few months.

The report stresses a plan for controlling flood flows to nondamaging levels while simultaneously making optimum use of these flows for hydroelectric generation. It also integrates the Upper Basin reservoirs into the overall flood control capability of Hoover Dam and Lake Mead.

Incidentally, all these excess flows were released through our hydroplants on the lower river. Although we release water only when it has been requested, or when it is dictated by flood control requirements, we <u>do</u> put the water to work as it flows through the system.

Mexico also used these excess flows for leaching, double cropping, and irrigating additional lands. To the extent we were able, we scheduled these excess flows to try to accommodate Mexico's needs and use capabilities.

In January of this year, with a below average runoff forecast, we cut river flows back to the routine condition of water being released only in sufficient amounts to meet downstream requirements. And, although we are temporarily relieved from the threat of high flood control releases, we still foresee a fairly high probability of encountering a similar situation during the next few years.

Encroachment upon the river floodway, particularly in the Lower Basin, has become a serious problem in recent years. Much of this land is in private ownership and not federal control. In the absence of routine flood control releases, development has occurred in and near the floodway that was designated to accommodate such releases. When the Central Arizona Project begins operation in 1985, the additional water used will significantly reduce the likelihood of having to operate the reservoirs under flood control regulations.

Legislation also defines the position of fish and wildlife interests in the operation of the River. The Fish and Wildlife Coordination Act requires that planning for any federally funded water project must include consideration of the project's impact on fish and wildlife. We also operate under direction of the National Environmental Policy Act of 1969 and the Endangered Species Act, and consult regularly with state and federal fish and wildlife authorities.

As an example of our commitment to fish and wildlife interests, consider a study being conducted at Lake Mead. Each spring the water orders from downstream irrigation districts increase. Unfortunately, this coincides with the annual bass spawning period. These increased releases generally lead to a decrease of the lake level, a condition which may affect the bass spawn. Although the reservoir must operate according to the established priorities, a five-year study of the Lake Mead bass population has been initiated in cooperation with the states of Nevada and Arizona which will attempt to identify the role of fluctuating lake levels on the bass population.

For many years we were concerned primarily about the quantity of water available in the River. More recently, we have also become concerned about the quality of this water--specifically, the salinity of Colorado River water. The push for salinity control was given emphasis when Mexico complained in the early 1960s about the increase in the salinity of water being delivered to them under terms of the 1944 treaty. After several years of negotiation between the two countries, and adoption of interim control measures, we entered into an international agreement for a permanent and definitive solution relative to the salinity of Colorado River water delivered to Mexico.

In order to meet the terms of the agreement, the Colorado River Basin Salinity Control Act was signed into law in 1974. The Act had two parts, Title I and Title II. The Title I portion was concerned with salinity control measures upstream of Imperial Dam. Although the act contained no provisions for fish and wildlife mitigation measures, Title I has since been amended to include this provision; to date, no mitigation measures have been included for Title II.

The heart of the Title I measures is the Yuma Desalting Plant, which will remove enough salt from irrigation return flows to make the water acceptable for delivery to Mexico. Preparation of the plant site, four miles west of Yuma, is nearly complete. Contracts have been awarded for the production of the reverse osmosis membrane units, and one of the two manufacturers has been notified to begin production.

Other water salvage operations of the Title I work are also nearing completion. Lining the first 49 miles of the Coachella has now been completed, and our protective and regulatory well field near the U.S.- Sonoran border is partially operative. When completed, Title I features are expected to make over 300,000 acre-feet of additional water available for use in the arid Southwest. Title II measures are designed to reduce salt inflows into the Colorado from particularly saline areas upstream of Imperial Dam.

Four projects -- two in Colorado, and one each in Utah and Nevada -- were originally authorized for construction. Two of these, the Grand Valley and Paradox Valley units in Colorado, are under construction and advance planning is underway on the Las Vegas Wash Unit in Nevada. There is no activity on the Crystal Geyser, Utah, Unit. Planning studies are also underway on twelve additional areas -- four in Colorado, five in Utah, and one each in Wyoming, Nevada and California.

About one-half of the dissolved salts in the River today can be attributed to man's development and utilization of this resource. Increased salinity lessens the quality of the water for both agricultural and municipal use. For every milligram per liter, we can reduce the salinity of water arriving at Imperial Dam, a benefit of about $472,000 may be realized by water users.

What about future management of this highly complex river system? Despite the fact that we have more than the equivalent of three years of average runoff stored in Colorado River reservoirs, and despite our concerns over potentially high flows in the future, there is no overlooking the fact that eventually we must deal with water shortages in the basin. While we cannot absolutely predict when, how long, or how severe these shortages may be, our studies indicate that there is a strong possibility of significant shortage in the Colorado's water supply within the next twenty to twenty-five years.

Because of the importance of the Colorado River for the many millions of people and the wildlife it serves, we must plan for and implement measures that will enable us to minimize the effects of drought periods and make maximum use of the water available during high runoff years.

There are several avenues available to stretch present uses of water to help meet the dry times. Principal among potential water saving methods are better onfarm irrigation efficiency, lining water conveyance facilities, perfecting water transport schedules, recycling return flows, and managing high water-consuming vegetation, to mention only a few. Water supplementing techniques, such as cloud seeding or upper watershed management have also been proposed.

Over the past several years, reclamation has been actively developing "Irrigation Management Services" (IMS). This is a method of providing the farmer with solid recommendations for managing his irrigation practices to assure effective use of the land and water resources. The program basically determines when and how much crops should be irrigated for maximum production and maximum water use. Ultimately, we foresee when an irrigation district's water will be based more precisely on crop need and water holding capability of individual fields rather than on convenience and historic practice.

Future trends in water use have been developing for some time. Present use must frequently be reexamined to ascertain that these trends will preserve water quality and at the same time meet people's needs. Thanks to the existence of reservoirs like Lake Mead and Lake Powell, Colorado River users have both a reliable and a sufficient water supply for some years to come.

But many questions are being asked about future uses of

Colorado River water. To cite a few: Should we use the water to irrigate more lands? Expand cities and industries? Cool thermal electric plants? Develop shale deposits? Improve fish and wildlife habitats? Should we stretch the water supply by encouraging a shift from crops that use a lot of water to those that use less water? The answer to these questions must come from the basin states. They must decide on priorities, within the "Law of the River," and thus direct future water use. We are looking forward to long-term coordination with water, power, wildlife, and land interests to manage the Colorado River to meet the water supply needs of the basin states.

CHAPTER 2

COLORADO RIVER MANAGEMENT TO ENHANCE AQUATIC RESOURCES

Bob Jacobsen
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River management problems as they relate to wildlife and fish will be dealt with first. Problem identification is really quite simple. It is man's uses of water versus fish and wildlife uses of water. The solution to these problems is similar to placing man on Mars: it is going to require a great deal of scientific exploration to achieve a balanced management of water and fish and wildlife resources.

Man's uses of the Colorado River are well documented and they will be further documented in this symposium. Traditional uses, such as dams for irrigation purposes, municipal and industrial purposes are well known, but all too often these projects and uses of waters have continued to result in losses of fish and wildlife. It is an insidious, ever growing loss. Currently, we are beginning to see rapidly expanding losses of fish and wildlife habitat. The expanded, unimpeded coal leasing program, oil and gas leasing, the oil shale program, uranium development, and power production are just a few of those uses by man that we are all too aware of. Losses of riparian habitat are expanding as development occurs. Riparian habitat losses can be lost through transportation and sand and gravel operations which provide for increased populations. The projections for population growth in the Colorado River system point to potentially three and a half-million people living in the upper basin. Salinity control is another problem which may very well result in additional losses of fish and wildlife resources.

Man's needs are readily understood in the Colorado River system. Institutionally, they are well known. There are local organizations and support. There is industrial support for uses of water. There are state water laws, compacts and agreements, all very well understood. However, fish and wildlife needs and uses remain poorly understood to this day.

Traditionally, most everyone in the Colorado River drainage area thought that wildlife was a vast, expendable resource. Tremendous losses have taken place and now listings of species as endangered or threatened are appearing. There has been a poor understanding of the biological requirements of these species. To this day we still have a poor understanding of the numbers of fish and wildlife: deer, for example.

All too often there is never enough time to address a project adequately in terms of what its true impact will be to fish and wildlife. And so the Fish and Wildlife Service and state game and fish agencies are viewed as organizations that are in opposition to the developer when we are merely saying we need to study, study, study. We are forced to try and baffle people with rhetoric. Our posture quite often has been to oppose projects which have less than satisfactory data with less than satisfactory data. There is a lack of true grass roots public support. Thorough public understanding is lacking on fish and wildlife values. For example, the Colorado River Squaw fish, Humpback Chub, Bonytail Chub are considered trash fish by people in Montrose, Colorado. The snail darter has become a symbol to the developer because it stood in the way of progress, and then, when all of a sudden progress took place, we began finding snail darters everywhere. Everybody points to the snail darter. Environmental groups are totally supportive of fish and wildlife values, but quite often for the wrong purpose, using the Endangered Species Act to stop projects when there is no other way to do so. However, I should point out that there is national support for fish and wildlife. Polls recently conducted show that fish and wildlife values are of utmost concern to the United States public. Therefore, I would propose that in working with the Colorado River system, we must look at fish and wildlife as a use of water.

There are a number of laws protecting fish and wildlife resources: state and federal laws. Most of these are poorly understood except for hunting laws. Everybody knows full well that you are not to hunt out of season. However, what acts protect the wildlife when you are not hunting is poorly understood. The Endangered Species Act, Fish and Wildlife Coordination Act, Migratory Bird Treaty Acts, Bald Eagle Act, and the Clean Water Act, Section 404, all protect fish and wildlife. Defining just what species we are actually concerned with in the Colorado River system becomes a problem: trout versus squaw fish, consumptive versus nonconsumptive species, deer versus dicky birds. Consider aquatic resource management. In the biological field we tend to separate aquatic and terrestrial resources, but they are very interdependent. Although this symposium is dealing with aquatics, the same issues apply when dealing with terrestrial resources.

Instream flow is really the bottom line in dealing with aquatic resources. There are legal problems. Few states recognize the value of instream flows for fish. However, Colorado is one state in the Colorado River Basin that actually does recognize instream flow and has the legal mandate to protect instream flow for fishes. Instream flows are poorly understood by most developers and generally the public. When looking at aquatic resources and instream flows, often the developer's standpoint is that if the lowest flow occurred in 1922, that is all the water you need to protect fish. The dynamics of instream flow issues need to be recognized by everyone. These issues include: water quality; watershed inputs in terms of sediments; particulate organic matter and nutrients; flow regime; physical habitat structure, such as channel form; substrate distribution; and riparian vegetation.

Certain specific action has been taken to deal with instream flow issues. In 1980, due in large part to Bill Plummer's efforts, we entered into an agreement which was signed by the Department of the Interior, the Governor of Utah, and the Central Utah Water Conservancy District to recognize instream flows for fish. We are working on a number of Bureau of Reclamation projects, state projects such as the White River Dam, private projects such as Rawley Fisher's Juniper Springs, Cross Mountain Project, and other private projects to determine how the projects can proceed and still provide fish and wildlife habitat. We are looking specifically at the endangered species problem in the Colorado River system.

The Bureau of Reclamation, Bureau of Land Management, National Park Service and Fish and Wildlife Service have funded a long term study (over two years) of the Colorado River fishes in terms of trying to find out what their life history is and what the flow requirements are, not only for those that are listed but also for those that are in danger of becoming extinct. We can no longer afford the time to go out and look and study in the field for the appropriate amount of time precisely what a project is going to do in terms of regional impacts on fish and wildlife. Thus, we are employing the latest in computer technology and considering rapid assessment methodology. We've developed a map indexing system for the states of Colorado and Utah. This system, which is also being used in other basin states, allows the user to look through the computer at what maps might be available for fish and wildlife resources in a given area. We are considering a water-for-energy computer model, which will give the user an opportunity to look at a project and translate the flow all the way down the Colorado River system.

Perhaps more important than these uses of computer technologies is an early input into planning. We are getting involved in a Bureau of Land Management planning for billions of acres in an effort to get fish and wildlife values included before final decisions are made. One question relating to future planning is, "Are we in time to make a difference?" Quite often, we feel that we are losing our resources by bits and pieces when in fact, it is a slow, insidious loss that is hardly recognizable. However, right now we are faced with development in massive proportions and so we need to deal with that. We are looking at regional environmental impact statements for overthrust oil and gas leasing. We are looking at regional environmental impact statements for coal development. We are presently participating in a regional environmental impact statement for synfuels development in the Uintah Basin. We are also looking at pipelines for synfuels delivery. Communication problems between fish and wildlife resource managers and the developers are a vital concern. There are institutional barriers between universities and federal agencies and among federal agencies, and, in some cases, state/federal relations are not the best. However, there are some positive actions which will be completed within the near future.

We will soon be completing our field studies on the Colorado River endemic species. Field work will be completed by January, 1982, and reports issued. We will be preparing a conservation plan for the endangered species of the Colorado River system, which will largely deal with how we can protect and preserve those species that are so near extinction, in some cases, and still allow water development to take place. We are about to complete our data entry into various computer systems which should be useful to many state, federal and private agencies.

Managing fish and wildlife resources in the Colorado River has a long way to go. Many communication problems and problems in getting public recognition of fish and wildlife resource needs still exist and hopefully, this symposium will at least add information which can be useful to all of us.

PART 2

RESERVOIRS

CHAPTER 3

SALINITY AND PHOSPHORUS ROUTING THROUGH THE COLORADO RIVER/RESERVOIR SYSTEM

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INTRODUCTION

The development of storage reservoirs on the Colorado River system, accompanied by the water and land use development during the past 30 to 40 years, has brought about significant changes to the physical, chemical, and biological balances of the Colorado River Basin.

Although sediment transport, temperature, biological productivity, and light penetration are mentioned, we have chosen to focus on salinity and phosphorus relationships in the reservoir sequence of Fontenelle, Flaming Gorge, Lake Powell, and Lake Mead (Figure 1).

A river/reservoir system is never in a state of static equilibrium, but is dynamic in its response to changing hydrological and chemical conditions. Each of the four reservoirs that we are examining has, to varying degrees altered this dynamic equilibrium, requiring the system to establish a new balance.

This paper is an overview of the complex physical and biological interactions which presently define the Colorado River Basin.

In order to develop a perspective of the influence of the four-reservoir sequence, it is necessary to take into consideration the climate, geology, and hydrologic character of the river basin.

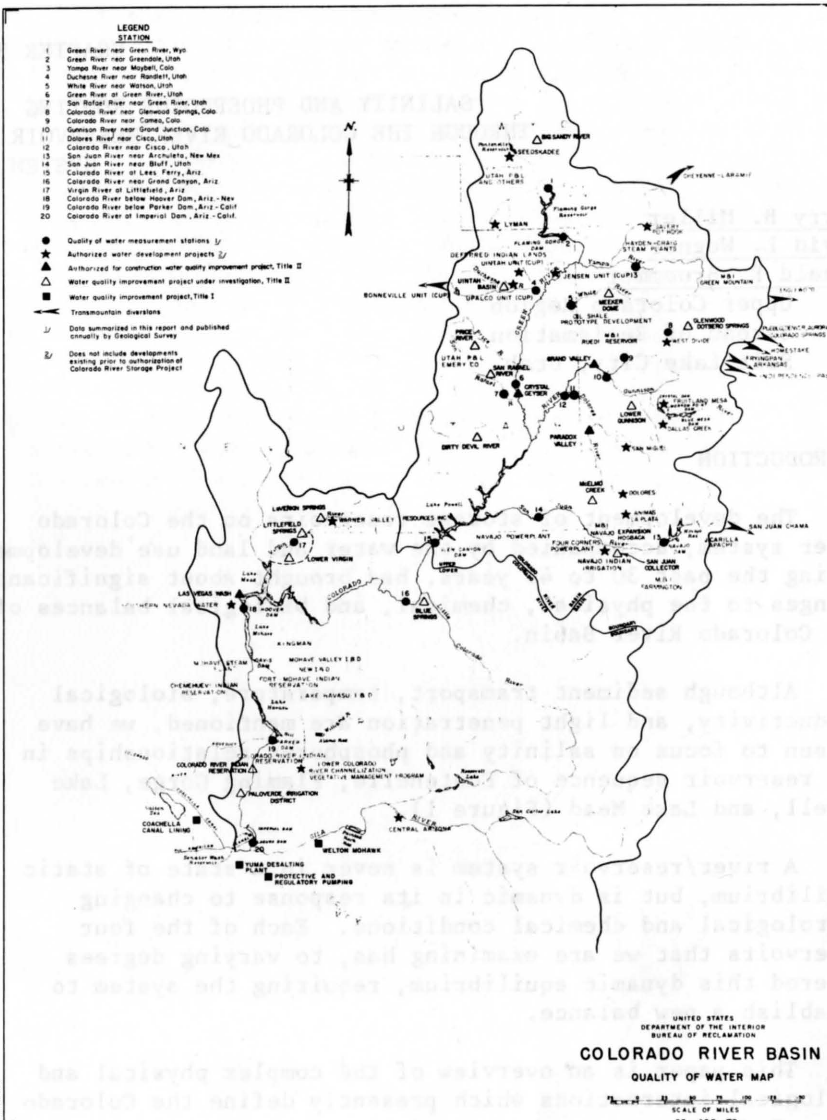


Figure 1. Colorado River Basin: Quality of Water Map.

Climate

The Colorado River Basin ranges in elevation from sea level to over 1450 m (14,000 ft). The component tributaries flow through complex mountain systems, elevated plateaus, and deserts with all but the mountains being primarily arid. While the greatest climate contrast is between the desert province in the south and the mountain province to the north, significant local variances are also present. Temperatures may vary from -40 to 120° F seasonally, with precipitation ranging from 6 to 60 in. per year. In the majority of the river basin's surface area, the evaporation potential far exceeds local precipitation. This high evaporation rate concentrates the dissolved minerals in the remaining water and is a factor in determining the high salinity values in the Colorado River.

Geology

The geology of the Colorado River Basin is as varied as the climate. The igneous and metamorphic rock forming the headwaters region produces cold, crystal clear streams often lacking in sufficient dissolved minerals to support a diverse aquatic community. However, as the river flows downstream it contacts marine deposits containing salts and fine-grain sediments which can over a few miles change the pristine streams into torrents of mud, salts, and nutrients.

Large quantities of salts, sediment, fossil fuels, and evaporite minerals are available, particularly in numerous marine deposits of the geosynclinal basins. The flow regime of the basin has mobilized many of these salt and phosphate deposits. In those areas where the salt has become most mobile, either naturally or due to man's influence, salinity control projects have been designated.

In addition to the readily available salts, deep geosynclinal basins and impermeable aquicludes temporarily isolate highly saline, static, ground water from the hydrologic forces. Control of the major mobilized salt sources in the 14 USBR designated Colorado River salinity control project areas [1] is very important to the future development of the remaining water resources. It is equally important that with the development of the mineral and energy resources within the basin, care be taken not to mobilize presently static saline systems.

Hydrology

Wet and dry cycles have played a significant role in bringing about the development of the Colorado River/Reservoir complex. In the past, the annual flow of the river has varied from less than 6 million acre-feet to over 20 million acre-feet per year [1]. The reservoir system allows storage of sufficient water to maintain the flows of the river to meet downstream needs during dry periods.

The construction and filling of the mainstem reservoirs of the Colorado River Basin have brought about significant changes in the hydrologic cycle. In addition to the major reservoirs, numerous smaller reservoirs are found throughout most of the tributaries. Since major storage began with Lake Mead in 1935, to the conclusion of the initial filling of Lake Powell in 1980, the reservoirs in the Colorado River Basin have developed a storage capacity equal to approximately four times the total average annual flow of the entire Colorado River (Figure 2).

Reservoir Limnology/Downstream Hydrology

During initial reservoir filling, the rising waters inundate soils rich in nutrients, organic matter, and salts. Initially, the rising water will leach out the soils, particularly during the shoreline wave action phase. Quantities of water also go into bank storage, varying with the permeability of the underlying geology.

The typical reservoir of this system is usually highly productive during initial filling. As filling continues, the inflowing sediment is distributed over the reservoir area and temporarily locks in the underlying nutrients and salts. Leaching of those materials below the wind mixed epilimnion is substantially decreased due to the lack of mechanical action. Eventually a chemical balance will develop between the water column, sediments, and bank storage. Again, a fluctuating reservoir is not a static environment, but is in a state of dynamic equilibrium and responds to changes in the hydrologic, climatic, and chemical conditions.

The most readily noted impact from the reservoir is the change in the suspended sediment, water temperature, and flow patterns downstream. Figures 3 and 4 illustrate how the flow and salinity patterns have changed below Flaming Gorge and Glen Canyon Dams.

It is the transition of the inflow conditions through the internal reservoir circulation which determines downstream conditions. The two most significant factors determining

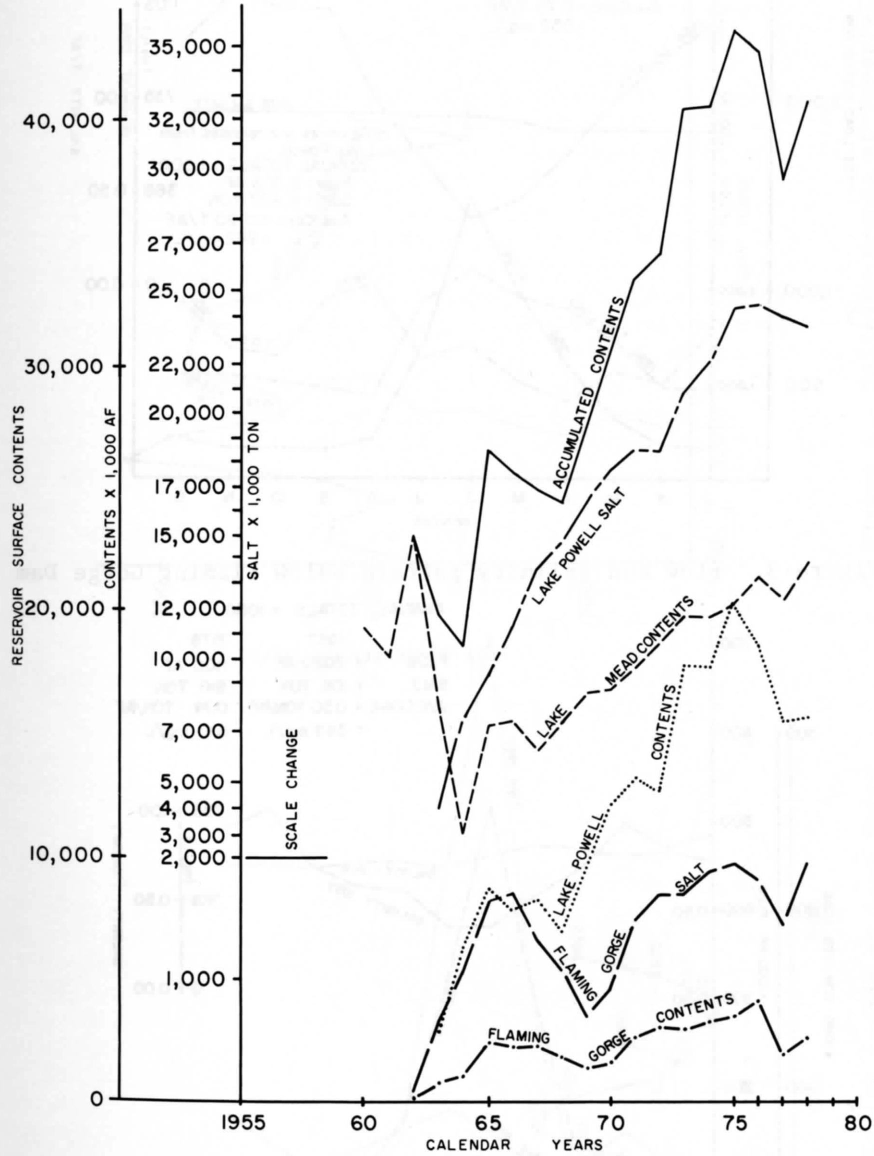


Figure 2. Storage capacity for Flaming Gorge, Lake Powell and Lake Mead

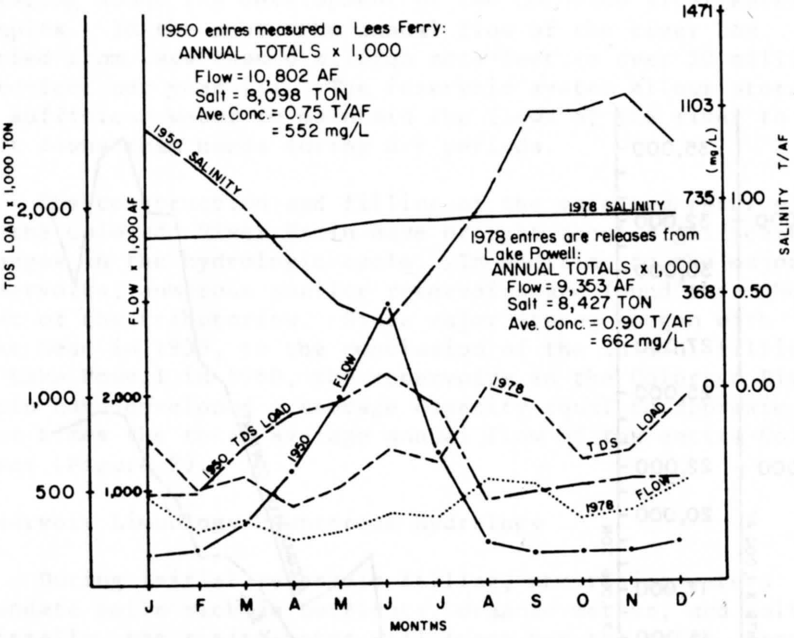


Figure 3. FLow and salinity pattern below Flaming Gorge Dam

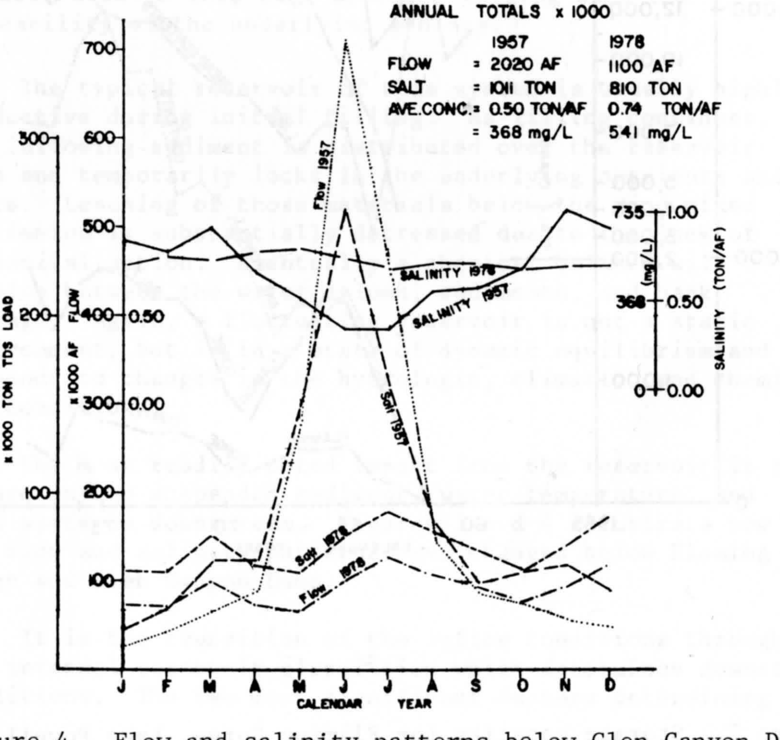


Figure 4. Flow and salinity patterns below Glen Canyon Dam

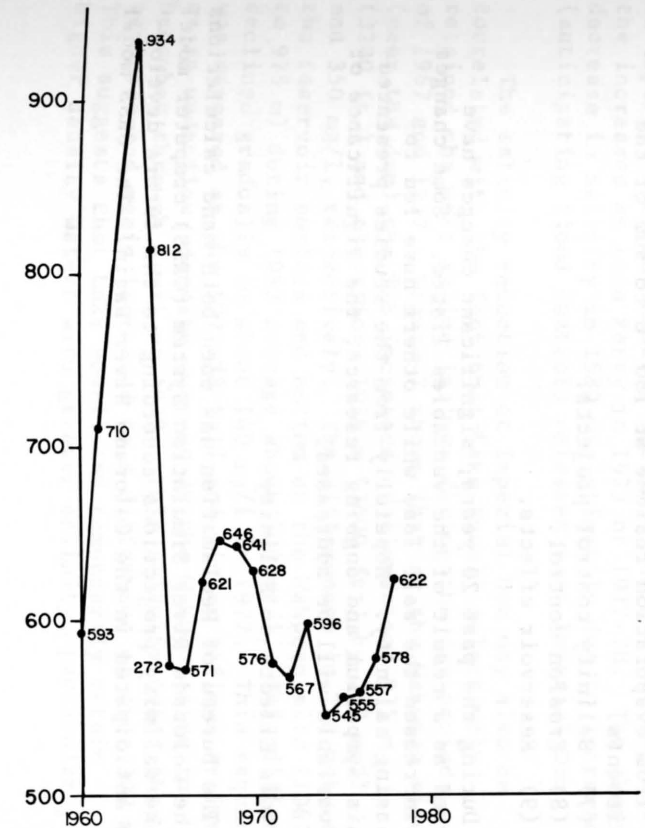


Figure 5. Salinity trends at Lees Ferry

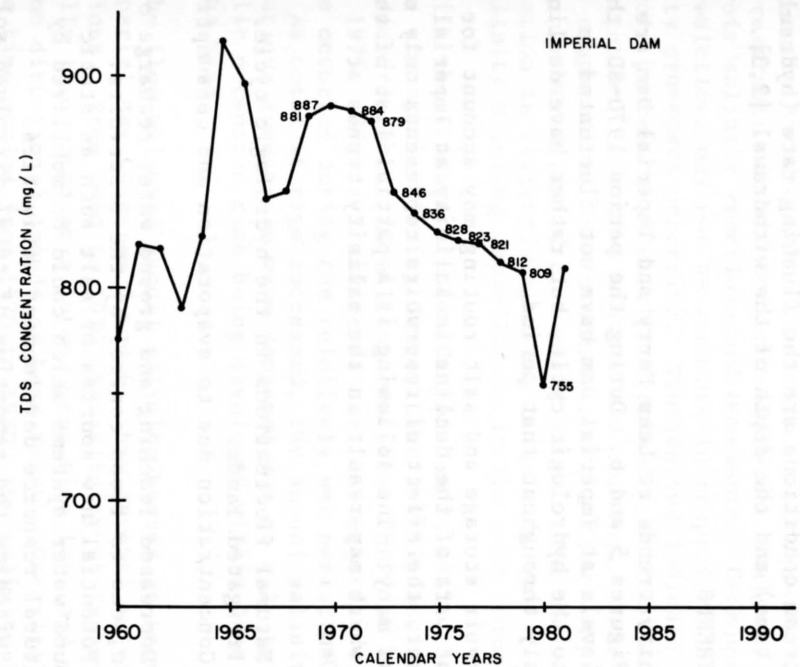


Figure 6. Salinity trends at Imperial Dam

the downstream conditions are the flushing rate (hydraulic detention time) and the depth of the withdrawal [2,3].

SALINITY TRENDS

Salinity trends at Lees Ferry and Imperial Dam are shown in Figures 5 and 6. During the period 1970-80, the salinity levels at Imperial Dam have not fluctuated in relation to the hydrologic cycle, but rather have declined continuously throughout that period.

Reservoir storage and salt routing may account for a significant part of the decline in salinity at Imperial Dam; however, the effect of reservoirs represents only one variable of many. The following is a partial list of the variables which may result in the salinity trends at Imperial Dam:

(1) Natural fluctuations in the hydrologic cycle.
(2) Irrigated lands.
(3) Concentration due to evaporation and consumptive use.
(4) Decreased leaching and ground water recharge of the flood plain, due to flood control by the reservoirs.
(5) Potential new sources of salt such as static saline ground water systems which could be mobilized by various natural resource development activities.
(6) Switching the reporting of total dissolved solids (TDS) from evaporation residue at 180° C to sum of the constituents.
(7) Salinity control projects.
(8) Erosion control.
(9) Reservoir effects.

During the past 20 years, significant changes have occurred as a result of the variables listed. Some changes have increased the salt load while others have led to decreasing salinity. Hopefully from the studies presented at this symposium and ongoing research, the significance of each variable will be addressed.

Reservoir Effects on Salinity

The Bureau of Reclamation has used both hand calculations and the Colorado River Simulation System (CRSS) computer model to make salinity projections according to the future developments anticipated in the Colorado River Basin. The CRSS model is currently based on the assumption of a once-a-month complete reservoir mix and does not allow for analysis of in-reservoir salinity reactions and movements. Consequently, the predictions may not be accurate in respect to the actual salinity processes occurring. Studies now indicate that reservoir processes which affect salinity include leaching, precipitation, selective storage and routing, concentration due to evaporation, bank storage, and the flow weighted averaging over a period of several years. The Bureau of Reclamation is currently studying these problems to improve the salinity modeling capabilities in the near future.

With the closure of Glen Canyon Dam in 1963, the actual surface storage in Flaming Gorge, Lake Powell, and Lake Mead increased from about 20 million acre-feet (MAF) to over 43 MAF by 1975. Figure 2 indicates that the majority of this storage occurred during the relatively wet period of 1968 to 1975. As total storage increased, the annual salinity fluctuations downstream were dampened due to a 2- to 4-year hydraulic detention time being developed within Lake Powell and Lake Mead.

The flow weighted annual salinity trends at Lees Ferry and Imperial Dam (Figures 5 and 6) show several interesting variations. From 1960 to 1970 the salinity at Imperial Dam generally fluctuated at a 1-to-2 year time lag, and with the same directional trends as Lees Ferry. Contrarily, for the time period from 1970 to 1980, the salinity levels at Imperial Dam continuously declined and did not reflect the increases at Lees Ferry in 1973 or 1977-78. The sharp decrease in salinity in 1980 was primarily due to increased (anticipating flood control) releases past Imperial Dam.

The salinity reduction at Imperial Dam may also be correlated with the fact that Lake Powell selectively retained the most saline inflows during the dry periods of 1967 and 1977. Figure 7 shows that at the Wahweap site (near the dam) the TDS increased at elevations 975 m (3200 ft.) and elevations 1036 m (3400 ft.) by 260 mg/l and 350 mg/l, respectively. The differences in TDS between the reservoir surface and bottom at the Wahweap site (1067 m to 975 m) during 1967 average about 300 mg/l. This difference declined gradually to about 140 mg/l in 1977. This sequence was repeated as the 1977-78 fall and winter inflows arrived as an underflow, density current at the Wahweap site in December through April of 1978. This salinity trend then reversed and again increased to a difference of over 300 mg/l. This suggests that Lake Powell can temporarily retain higher density waters with greater salinity, particularly

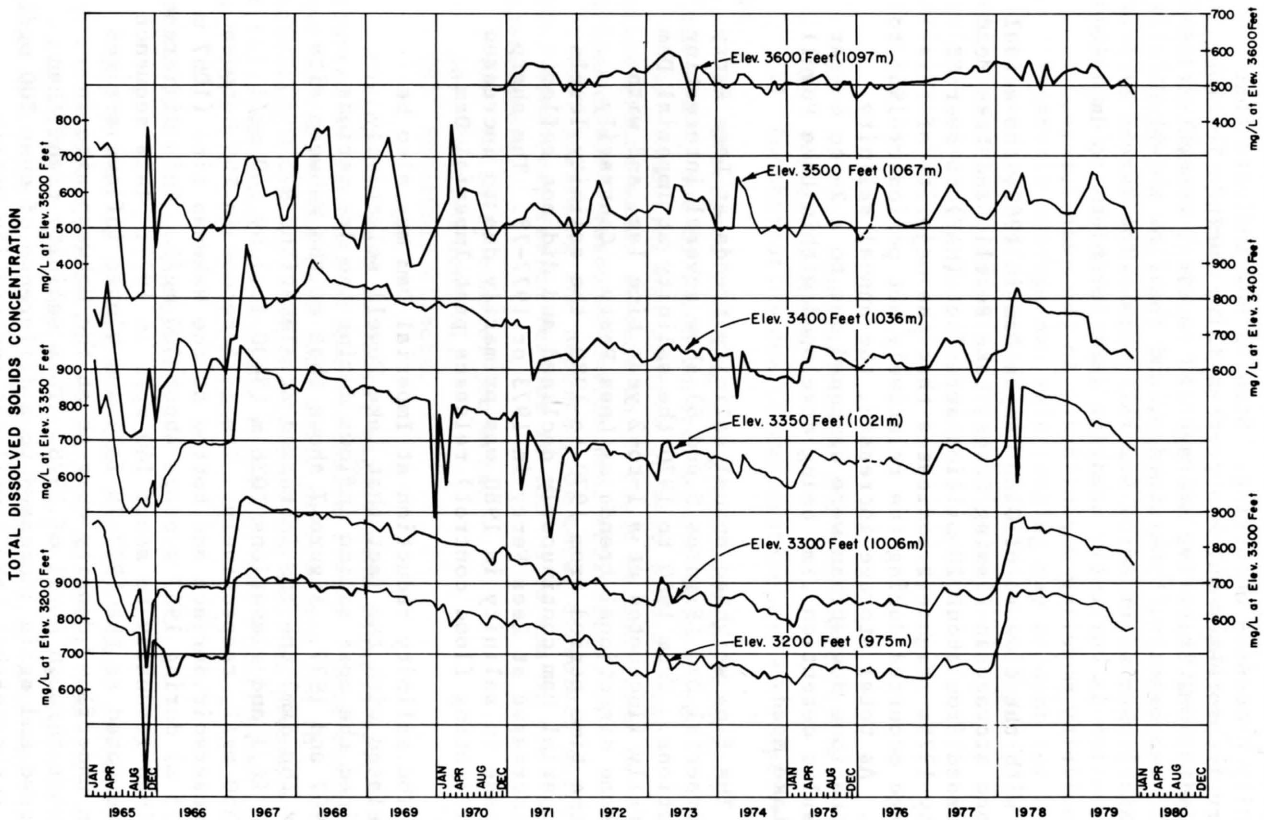


Figure 7. Salinity changes in Lake Powell at the Wahweap Site.

during low inflow periods. The depth of the outlet, flushing rate of the hypolimnion, and density (produced by temperature and salinity) of the inflow all contribute to this process. Once the water is stored, dilution, leaching, precipitation, mixing, and bank storage all affect the saline water retained in the reservoir.

A similar salinity sequence has been observed at Flaming Gorge Reservoir. A pronounced salinity profile and chemocline have been documented by USGS [4,5] and Bureau of Reclamation [6] limnological surveys at Flaming Gorge Reservoir. Figures 8 and 9 show that the surface-to-bottom salinity variance near the dam was 300 micromhos in 1971 and 170 micromhos in 1981. This is very similar to the TDS trends observed at the Wahweap site in Lake Powell.

In 1978 a major operational change to a selective withdrawal system was made at Flaming Gorge Reservoir. The summer releases in 1978 were changed from the hypolimnion to the epilimnion for downstream temperature control. Limnological surveys and analysis are still ongoing, and a final determination on the impact of selective withdrawal on the salt routing, water quality, and aquatic ecology of Flaming Gorge Reservoir has not been made. However, the salinity profile shown in Figure 9 indicates that the chemocline in Flaming Gorge Reservoir is decreasing and the hypolimnion may mix during the fall turnover in 1981 and spring turnover in 1982. The addition of the selective withdrawal and the continuation in the changing salinity levels of the chemocline indicate that Flaming Gorge Reservoir is still undergoing minor chemical adjustments towards a dynamic equilibrium.

Bolke and Waddell [4] reported that Flaming Gorge Reservoir increased the load of sulfate and decreased the load of bicarbonate in the Green River from 1963 to 1972. They predicted that the rate of sulfate leaching would decrease after initial filling in 1972.

Fontenelle, the headwater reservoir in the sequence, has a high flushing rate and a deep hypolimnion outlet. No significant variations in the TDS profiles have been observed with depth in this reservoir.

Under present conditions, Lake Mead does not exhibit significant variation in salinity with depth. The maximum conductivity variations in the lower basin of Lake Mead were

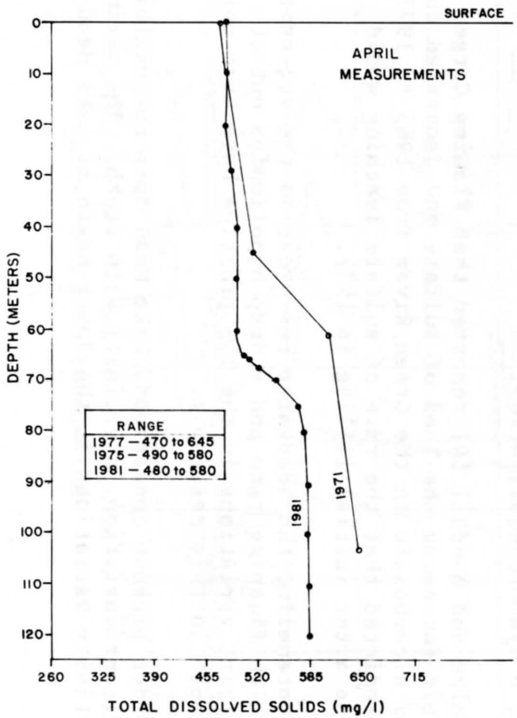


Figure 8. April salinity profiles at Flaming Gorge Reservoir

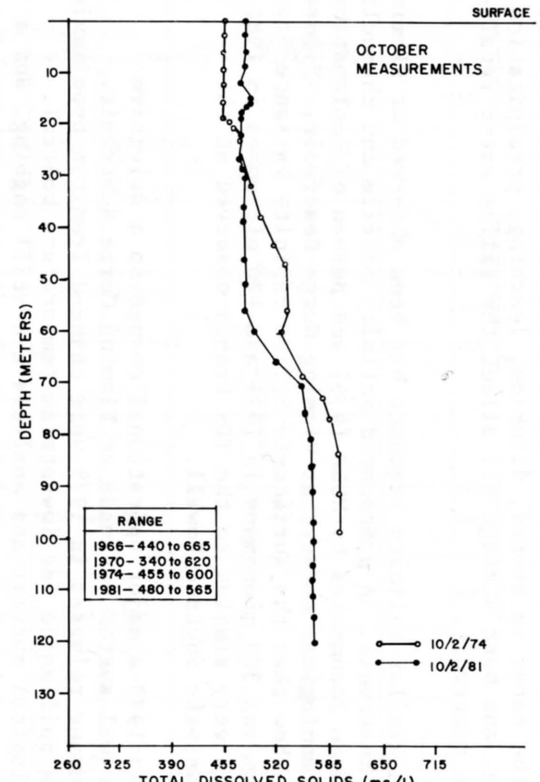


Figure 9. October salinity profiles at Flaming Gorge Dam.

30 to 65 mg/l in 1978 [7]. This is excluding the inflow density currents shown only in the shallow inflow areas. Lake Mead has a deep hypolimnion withdrawal and low seasonal TDS variance in the Colorado River inflow due to the attenuating effects of Lake Powell.

The reservoirs have a significant impact on the seasonal salinity variation downstream, and also have a cumulative effect on the long-term salinity trends at Imperial Dam. There is evidence that the reservoirs trap bicarbonate due to calcium carbonate precipitation, but also leach sulfate (gypsum) [4]. The long term impacts on salinity cannot yet be precisely predicted because the period of record represents the initial filling for over 50 percent of the storage capacity.

Hydrologically, the reservoir pool levels and operation pattern observed since 1975 are probably typical of the future expected conditions. However, Lake Powell did not complete initial filling until 1980, and Flaming Gorge Reservoir is still undergoing minor chemical adjustments. Therefore, minor modifications to predicted trends may be observed.

The data presented also suggest that Lake Powell and Flaming Gorge can selectively trap the most saline inflows and retain these waters for several years. However, there is no assurance that this process will continue as the river/reservoir system approaches a more steady-state condition.

The Bureau of Reclamation has several investigations ongoing to determine the long term effects the reservoirs will have on salinity. These include:

(1) A two-dimensional thermodynamic/salinity reservoir model of Lake Powell and Mead.

(2) An ion constituent study to determine changes in the chemical characteristics of the water, the causes of these changes, and their longevity.

(3) Limnology surveys of Lake Powell, Lake Mead, and Flaming Gorge Reservoirs.

(4) The continued development and improvement of the Colorado River Simulation System.

(5) A study to improve evaporation estimates for the Colorado River Reservoirs.

(6) A study of selective withdrawal from Lake Mead.

As the reservoirs approach a dynamic equilibrium and our understanding of long term effects on salinity improves, the accuracy of the Bureau's salinity predictive tools will also improve.

PHOSPHORUS

Eutrophication refers to the enrichment of nutrients and increases in primary productivity in water [8]. The trophic status, as it is reflected by the types and quantities of algae, largely determines the fishing and recreational potential, dissolved oxygen, aesthetics, and general water quality for potable uses. In fresh water it is most generally considered that phosphorus is the key "limiting nutrient" which regulates primary productivity and determines trophic status. [2].

Eutrophication and its relationship to the available phosphorus supply have been the subject of keen interest in international research. Numerous empirical models have been developed to predict the trophic status of a lake based on the phosphorus budget [9]. It is important to note, however, that the lake must be phosphorus limited.

These empirical phosphorus models are beneficial, but their basic assumptions are conditional to well mixed lakes, not to stratified run-of-the-river reservoirs with deep outlets.

In the Upper Colorado River Basin phosphorus has been mined and exported as fertilizer. In addition, vast formations of oil shale were deposited in ancient eutrophic lakes in Utah, Wyoming, and Colorado. The oil itself is the product of the tremendous algal biomass accumulations in these eutrophic lakes. These eutrophic lake deposits indicate that phosphorus is possibly more abundant in the geochemistry of the Upper Colorado River Basin than is typically found in other river basins.

The seasonal thermal stratifications, deep outlets, and high flushing rates typical of many reservoirs present cbstacles to applying the empirical phosphorus models. In addition, the climatic effects particularly on reservoirs over 4000 ft. in elevation cause seasonal light and temperature variables which become physically limiting factors to primary productivity.

The factors which determine a lake's primary productivity have been classified into three groups [10]: (1) variables related to solar energy input (temperature and light), (2) variables in nutrient supply, and (3) variables in lake morphometry. Flushing rates for hydraulic detention time have also been recognized as being important in determining the residence time and availability of phosphorus [2]. In addition, the depth of the outlet, internal mixing, and density currents are key parameters which influence the residence time and physical/biological availability of phosphorus in reservoirs.

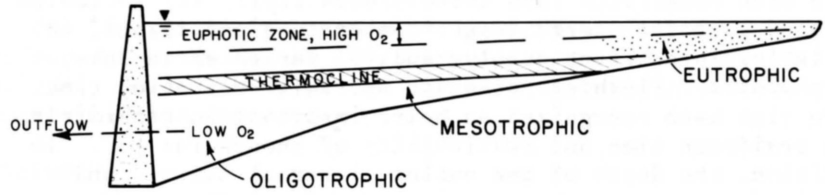
An often overlooked but important factor which physically induces light limitation is the relationship between the euphotic zone (sufficient light for photosynthesis) and the zone of wind-driven turbulent mixing known as the epilimnion. The deeper the mix zone (epilimnion) relative to the euphotic zone, the less time the algae spend in the light, the lower the average amount of light available to the algae, and thus the lower their net rates of photosynthesis and growth (Figure 10)[11]. The algae are physically displaced into the dark portion of the epilimnion.

In reservoirs with deep outlets, the thermocline tends to migrate downward as the cooler hypolimnion water is withdrawn. This deepened thermocline should be considered, as it may induce physical light limitation. The magnitude of this downward thermocline migration is a function of the hypolimnion flushing rate and depth of withdrawal. Deep outlets and high flushing rates may also reduce phosphorus retention.

A summary of these variables which influence a reservoir's primary productivity is illustrated in Figures 10 and 11. It is estimated that on a world wide basis variables related to solar energy input have the greatest influence on primary productivity [12].

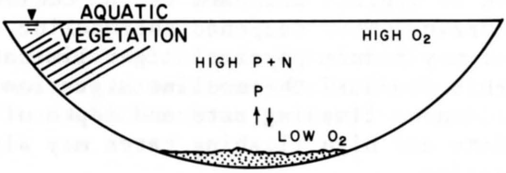
The depth of the outlet and the vertical placement of the inflow based on its density (primarily a function of temperature) may have a major influence on the availability and retention of phosphorus in a reservoir. Not only must the algae remain in the lighted portion of the water column, but the inflowing nutrient supply must also be physically and chemically available in the euphotic zone as well.

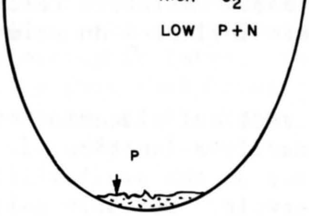
In addition to physical and chemical restrictions to primary productivity, limitations may also be the result of biological actions between groups of algae. Generally, a succession of phytoplankton species occurs throughout the



Summer/Fall conditions: Light limitation is determined by the depth of the epilimnion (wind driven mixing) in relation to the depth of the euphotic zone (sufficient light for photosynthesis). The algae are physically mixed down into the dark; thus, the lower the average amount of light available to the algae, the lower the net rate of photosynthesis and growth. The algae themselves may cause most of the turbidity. A hypolimnion outlet draws the epilimnion deeper.

Figure 10. Reservoir Limnology/hypolimnion outlet





Eutrophic: High water surface to volume ratio, high phosphorus, high rate of phosphorus recycled from sediments due to anaerobic conditions and internal nutrient recycling may exceed inflow nutrient load. The eutrophic lake becomes nutrient self sufficient.

Oligotrophic: Low water surface to volume ratio, low productivity, high phosphorus sedimenting out, HIGH O_2.

Figure 11. Limnology/morphometry variables that influence primary productivity.

year. This sequence of algae succession may trend from desirable species to those which are potentially harmful, such as certain species of blue-green.

Blue-green algae have several distinct characteristics, such as buoyancy and nitrogen fixation, which give them the capability to outcompete more desirable species [13]. Certain species of blue-green algae have the following harmful effect: (1) undesirable as food to grazing zooplankton species; (2) cause reduced light penetration and aesthetics; (3) reduced dissolved oxygen which may mobilize iron and manganese thereby causing additional potable water use problems; (4) the production of organic toxins at death which affect both aquatic and terrestrial life; (5) taste and odor problems in municipal water diversions; (6) production of complex organic compounds which may contribute to the formation of trihalomethanes after chorination; and (7) in excesses they may increase domestic water treatment costs.

The following section is a review of the phosphorus dynamics in the example four-reservoir sequence.

Phosphorus Dynamics

The reservoirs in the Upper Colorado River Basin, which contain natural phosphate deposits in their drainages, are generally eutrophic even though much of the phosphorus is sediment bound and usually biologically unavailable. This could account for the high phosphorus retention rates observed in the reservoirs. Based on the EPA National Eutrophication Surveys [14,15] and U.S. Geological Survey Water Resources data for Water Year 1975 [16], it has been estimated that the four Colorado River Basin Reservoirs can retain 70 to 96 percent of their inflowing phosphorus loads.

Fontenelle Reservoir has natural phosphate deposits in its drainage basin and was calculated to retain approximately 87 percent of the inflowing total phosphorus for Water Year 1975. Reservoirs with bottom releases and high flushing rates tend to retain less phosphorus. Fontenelle's high phosphorus retention rate is apparently due to the mineralized form of the phosphorus which is bound to the sediment and remains predominantly biologically unavailable.

Much of the May/June phosphorus budget is associated with the sediment laden spring runoff and is physically and chemically unavailable in the hypolimnion of Fontenelle Reservoir. Subsequent chemical reductions, organic decompositions, in-reservoir movement, and release of phosphorus from Fontenelle in July through September may contribute to the substantial blue-green algae blooms downstream in the Green River Arm of Flaming Gorge Reservoir.

Flaming Gorge Reservoir is over 90 miles long and has a low surface area to volume ratio. It is a very efficient phosphorus trap and was calculated to have retained 84 percent of the calculated phosphorus load for Water Year 1975. This estimate may be conservative since the single outflow from the dam is probably a better estimate of phosphorus releases than can be made from the multiple inflows. The phosphorus measured at the point sources and from Fontenelle Reservoir exceeded the measured loads in the Green River Arm above Flaming Gorge Reservoir. Flaming Gorge can be classified as eutrophic in both the Green River and Black's Fork inflow areas, mesotrophic through the middle section, and oligotrophic in the downstream canyon portion [17].

Blue-green Algae Relationships

Blue-green algae population levels represent a seasonally significant impact on the Colorado River Basin reservoirs. The occurrence of excessive blooms of blue-green algae in the headwater reservoirs appears to be related to phosphorus dynamics, reservoir dynamics, and other water quality interactions. Blue-green algae exist throughout the basin and the effect of the blue-green algae in the basin reservoirs varies seasonally and annually.

In September 1981, Fontenelle Reservoir experienced fall overturn and mixed the entire water column. The blue-green algae were dispersed throughout the water column with primary productivity being physically limited by light availability. The extent of blue-green population expansion is limited by the elevation, temperature levels, and light intensity.

Blue-green algae blooms are a substantial problem in both the Green River and Black's Fork arm of Flaming Gorge Reservoir. Depending on the magnitude of the blue-green blooms and the climatic conditions of the fall, the cold water fishery may not be continuously maintained in this area. Primarily, this is due to the low dissolved oxygen and high water temperatures. The determining or limiting factor to the blue-green algae blooms in the inflow area of Flaming Gorge Reservoir may be phosphorus controlled, but it is more likely a combination of the length of summer stagnation period, the fall meteorological conditions and the phosphorus supply. In addition, the blue-green algae can often outcompete many of the green algae and diatom species based on their ability to fix nitrogen and control their position in the water column.

A cool wet spring and/or a cool wet fall can greatly reduce the length of summer stagnation due to a reduction in the period that the reservoir is stratified.

The location and stability of the fall thermocline appear to be key factors in determining the timing and strength of the blue-green algae bloom and the amount of reservoir which it affects.

A more thorough investigation of the variables controlling primary productivity in the inflow of Flaming Gorge Reservoir is necessary to determine if phosphate can be reduced sufficiently in the fall to become a limiting factor.

With the shift in reservoir releases from a deep hypolimnion release to a multiple level withdrawal scheme, the availability and concentration of available phosphorus may have been increased. Wright [18] has hypothesized that deep discharge reservoirs may progressively decline in fertility due to withdrawal of nutrient rich hypolimnion water. Conversely, shallow discharge reservoirs may experience an increase in fertiility. This hypothesis and its application to the Colorado River system have been supported by Paulsen [19].

Flaming Gorge Reservoir is also above 6000 ft. in elevation and the fall climate can vary considerably. The reservoir begins to turn over in the inflow areas in early September. There appears to be a relationship between this turnover and the extent of blue-green algae blooms in the fall. Considerable work needs to be done on high elevation reservoirs (such as Fontenelle and Flaming Gorge) to determine the relationship between the fall blue-green algae blooms and available phosphorus supply in the inflow area as a function of the internal phosphorus recycling and meteorological variations.

Lake Powell is the next major downstream reservoir on the Colorado River system. We have estimated that for Water Year 1975, Lake Powell retained approximately 97 percent of the total phosphorus that flowed into it. The reason for this high retention can be directly related to the morphometry of the reservoir basin.

Lake Powell is 170 miles long and is at an elevation of 3650 ft. It was developed within incised sandstone canyons and consequently it is very deep and narrow. This type of morphology and geology is not conducive to high physical or chemical availability of phosphorus, particularly in the horizontal movement down reservoir. The natural phosphorus loads that would have passed Lake Powell have also been reduced by upstream storage in tributary reservoirs, including Fontenelle and Flaming Gorge. However, many unregulated tributaries contribute additional nutrients to the Colorado River above Lake Powell. The high turbidity of the inflow and the short sun day due to the shading effect in the bottom of Cataract Canyon appear to have an influence on primary productivity in the upper end of Lake Powell. No significant blue-green algae blooms have been documented that could impact aquatic or terrestrial life.

Lake Mead, the lowest reservoir in our analysis, has shown significant shifts in its trophic status due to changes in phosphorus availability since 1970. An analysis of the 1975 Water Year data indicates a 69 percent phosphorus retention rate. This lower phosphorus retention rate has been hypothesized to have resulted from several factors, including upstream storage and reduced phosphorus availability due to physical changes in the inflow (primarily temperature) and the deep hypolimnion outlet [7]. The aquatic ecology and nutrient chemistry of Lake Powell and Lake Mead will be further discussed in other symposium papers.

CONCLUSIONS

1. Hydrologically, the reservoir pools and operation did not stabilize until about 1975. Lake Powell completed initial filling in 1980 and the operation of Flaming Gorge Reservoir was changed by the addition of a selective epilimnion withdrawal in 1978. The major chemical and biological adjustments due to reservoir effects are progressing towards an equilibrium. A dynamic equilibrium responsive to hydrologic and climatic conditions is anticipated.

2. The reservoirs have caused major changes in salinity and phosphorus routing in the Colorado River System.

3. The observed salinity trends at Imperial Dam during the 1970 to 1980 period may not be totally understood without an additional period of record.

4. Ongoing studies would provide the needed information to improve predictive capabilities, particularly regarding future salinity and nutrient conditions in the Colorado River System.

5. As water and energy resources are developed in the Upper Colorado River Basin, and the reservoirs become a more important source of water supply.

6. The relationship between blue-green algae population levels, reservoir limnology, and phosphorus dynamics must be defined.

7. Phosphorus retention in the reservoirs above Lake Mead has caused a significant reduction in its nutrient inflow.

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CHAPTER 4

THE EFFECTS OF MAINSTREAM DAMS ON PHYSICOCHEMISTRY OF THE GUNNISON RIVER, COLORADO

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INTRODUCTION

The ways by which dams and diversions impact ecological processes in rivers have received increasing scientific inquiry in recent years [1]. However, almost all knowledge of effects of hydrologic regulation on riverine physicochemistry is based on measurements made at one or a few locations immediately downstream from the point of regulation. While it is obvious that dams alter downstream physicochemical regima profoundly, such impacts have not usually been placed in the context of an entire river system (see [2] for a notable exception).

As a part of a holistic approach to assess the ecology of stream regulation in the Gunnison River, Colorado, we report herein the physicochemical impacts of four mainstream dams on the river system from headwaters to mouth. The changes manifested by this intense regulation greatly influenced patterns and processes within the biotic communities extant in the various river segments [3]. Results reported here are limited to a physicochemical description of this major tributary of the Colorado River before and after regulation.

STUDY AREA

The Gunnison River flows westerly from the Continental Divide in central Colorado to its confluence with the Colorado River near Grand Junction, Colorado. The 20,533 km^2 drainage basin may be divided into two parts, based on basin geology. The upstream portion, above the confluence of the Cimarron River (Figure 1), lies primarily in mountainous terrain and drains granitic soils and relatively insoluble crystalline

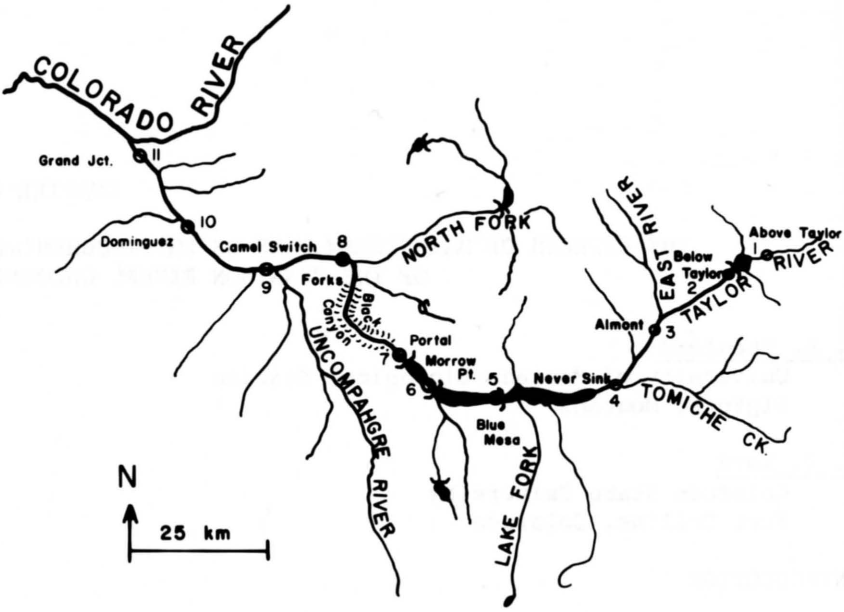

Figure 1. Location of the mainstream dams and eleven sampling sites on the Gunnison River, Colorado.

bedrock. Downstream from the Cimarron, the river drains a variety of mineral-rich sedimentary formations (especially gypsum shales), which characterize the semi-arid, high plateau of western Colorado.

The average monthly extremes in discharge of the Gunnison River at Grand Junction in the last 25 years have varied between a low of <1 m^3/sec to a high of >230 m^3/sec. However, the annual hydrograph is intensely regulated by hydropower and irrigation demands. Four mainstream reservoirs, Taylor Park, Blue Mesa, Morrow Point and Crystal (Figure 1), are impounded behind high dams and severely influence riverine hydrology. All four dams are deep-release (i.e., hypolimnial drain) systems. Taylor Park is an irrigation storage reservoir built in 1936, while the other three comprise the Aspinal Unit of the Colorado River Storage Project. Blue Mesa Dam was finished in 1965; Morrow Point Dam was closed in 1969. Crystal Reservoir began operation in 1975 as a re-regulation dam to dampen the extreme flow fluctuations below Morrow Point Reservoir. Considerable irrigation return flow occurs in the downstream river segment, especially via the lower Uncompahgre River and adjacent areas.

Few data are available concerning the limnology of these mainstream reservoirs. They are impounded within deep, granite walled canyons at ca. 2200 m elevation, where winter temperatures prevail from October - April. Consequently, these impoundments have a low heat budget. All, except Crystal, apparently stratify seasonally; surface temperatures may exceed 20°C for short periods during summer, but the majority of the stored water volume remains below 8°C year around (see Methods).

METHODS

We established eleven sampling sites along the Gunnison River from a headwater location above Taylor Park Reservoir to a point just upstream from the confluence with the Colorado River (Figure 1). Sampling was conducted on eleven occasions during the period September 1979 to October 1980.

Water samples for analyses of ion concentrations were collected in high-density polyethylene bottles, while grab samples for analyses of carbon fractions were collected in acid-washed teflon or glass bottles. All samples were stored on ice and air-freighted to the University of Montana Biological Station for analysis in the Freshwater Research Laboratory. Conductivity (YSI meter) and pH (Corning meter) measurements and alkalinity titrations (as $CaCO_3$) were made in the field. We installed Ryan© thermographs at two locations to augment records provided by Colorado Division of Wildlife.

Ions (Ca^{++}, Mg^{++}, K^{+}, Na^{+}, $NO_3^{=}$ and $SO_4^{=}$) were quantified by raw water injection into a model 16 Dionex© Ion Chromatograph with output integrated and digitized on a Hewlitt-Packard Model 3388 terminal.

Organic carbon present in water samples was separated into two fractions, particulate organic carbon (POC) and dissolved organic carbon (DOC), with glass-fiber filters (Gelman© 0.2 μm pore size). Organic carbon in filtrates was considered to be in the DOC fraction. POC and DOC were converted to CO_2 by hot persulfate digestion in sealed ampules and concentrations subsequently determined by quantification of the liberated CO_2 using an Oceanography International© infrared detector.

Every fifth analysis (ions or carbon) was replicated (i.e., multiple determinations, usually three, of the same parameter on the same sample) and samples were duplicated (i.e., two samples from the same location and time) to permit calculation of analytical precision and natural variation within sample locations. Reagent spikes were utilized (again every fifth analysis) to check accuracy of analytical technique. Standard deviations of replicates and duplicates were consistently less than one percent of the mean (i.e., high precision) and 90-110 percent of the sample spikes were recovered in analyses leading to the data reported herein.

Some chemical data were available in the STORET file of the U.S. Environmental Protection Agency for comparison to those generated during the present study. Discharge data were provided by the U. S. Geological Survey for various river sites and the U. S. Bureau of Reclamation for the dam sites. Available time-series flow data enabled us to compare discharge regima during the study period, with pre- and post-impoundment regimes (i.e., 1900-64 and 1965-present) on the mainstream river. Time-series temperature data were derived from unpublished literature, such as theses and various agency reports. Thermograph records for Sites 8 and 9 were provided by the Colorado Division of Wildlife, while data for Site 11 were provided by the U. S. Bureau of Reclamation. Relationships between discharge and thermal regima were established with the use of polynomial regression analyses and simple plots of annual degree days (a sum of mean daily temperatures over an annual period, [4]) along the river profile.

RESULTS AND DISCUSSION

The pre-regulation discharge regime of the Gunnison River varied from minimum flows during autumn and winter to spring maxima as a result of melting snowpack in the headwaters (Figure 2). The post-regulation flow has been considerably higher in winter and lower during spring (Figure 2), as runoff is stored in the reservoirs and discharged primarily from November to March. Greater than 90 percent of the average annual discharge is derived from precipitation in the headwaters; downstream side flows (i.e., below the North Fork River) in the lowland sedimentary formations contribute significant amounts of water only during short spates in spring and after heavy summer thunderstorms.

Historically, the upstream segment carried substantial sediment and bed loads during spring runoff which were deposited in the lower gradient downstream segment. Thus, for much of the year, the upstream segment flowed low and clear over a cobble and boulder bottom that was annually scoured and re-distributed by the spring freshet. The downstream segment was also fairly clear at base flow, but the bottom was predominantly silt. Occasional rubble riffles occurred in areas where side flows carried large materials into the river channel (Dolan et al. [5] describe this process of riffle or

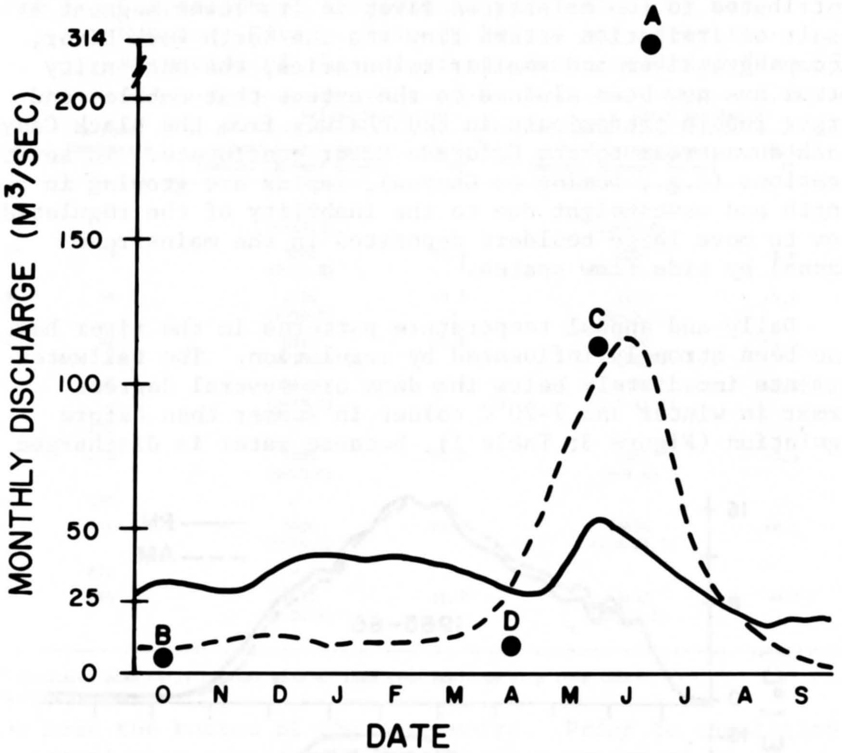

Figure 2. Discharge measured at Site 7, the U.S.G.S. gauging station below Crystal Dam, before regulation (broken line: monthly means 1948-1964) and after construction of mainstream dams (solid line: monthly means 1965-1980). Points A and B identify maximum and minimum pre-regulation discharge (monthly means 1934 and 1957); points C and D represent the maximum (1974) and minimum (1977) monthly flows since regulation (based on U. S. Geological Survey data).

rapids building by side flows on the mainstream Colorado River). Since regulation, silt loads accompanying runoff have been retained in the reservoirs. Thus, discharge below the dams is continually without significant amounts of suspended solids; the river from Taylor Park Reservoir to the East River and from Crystal Dam to the Colorado River is being continually sluiced by clear-water discharges that are of a higher mean volume July to March than prior to impoundment. The result is considerable armoring of the river bottom, the substrata being composed of firmly imbedded large rocks [6]. This situation presently characterizes the Taylor River and Black Canyon segments downstream from the dams to the East River and North Fork River, respectively. Although considerable sediment is contributed to the mainstream river in its lower segment as a result of irrigation return flow via the North Fork River, Uncompahgre River and smaller tributaries, the once silty bottom has now been sluiced to the extent that cobbles and larger rubble predominate in the thalweg from the Black Canyon reach downstream to the Colorado River confluence. In several locations (e.g., Dominguez Canyon), rapids are growing in length and wave height due to the inability of the regulated flow to move large boulders deposited in the mainstream channel by side flow spates.

Daily and annual temperature patterns in the river have also been strongly influenced by regulation. The tailwater segments immediately below the dams are several degrees warmer in winter and 7-20°C colder in summer than before regulation (Figure 3; Table I), because water is discharged

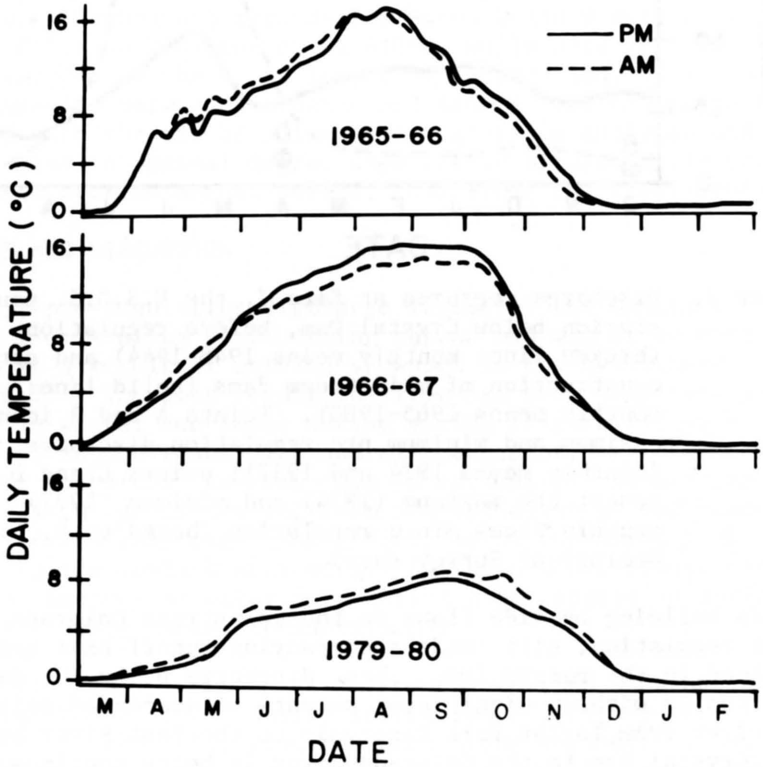

Figure 3. Pre- (1965-66, 1966-67 [from 7]) and post-regulation (1979-80) temperature patterns measured at Site 7, three km downstream from Crystal Dam.

Table I. Comparison of temperature patterns along the Gunnison River continuum before and after construction of the mainstream dams (modified from [3]).

Station No.	Km from Headwaters	Before Regulation Annual Degree Days (Annual Thermal Range)	Before Regulation Daily[a] ΔT	After Regulation Annual Degree Days (Annual Thermal Range)	After Regulation Daily[a] ΔT
1	18	1950 (0-15.0)	-	1950 (0-15.0)	-
2*	24	2000 (0-15.0)	+0.1	1000 (2.5-7.2)	-2.6
3	54	2250 (0-16.5)	+0.7	2150 (0-15.5)	+3.2
4	81	2550 (0-18.8)	+0.8	2250 (0-18.8)	+1.1
5*	115	2650 (0-19.0)	+0.3	2323 (3.3-11.1)	-0.6
6*	130	-	-	-	-
7*	144	2895 (0-20.0)	+0.7	1361 (0-9.4)	-3.8
8	195	-	-	-	-
9	228	3606 (0-24.0)	+2.0	3694 (2.8-21.7)	+6.5
10	271	-	-	-	-
11	290	4132 (0-26.6)	+1.5	3432 (0-23.3)	-0.7

[a]calculated mean daily thermal gain or loss from upstream site (see text).
*tailwater area.

from near the bottom of the reservoirs. Prior to regulation, the annual mean temperature of the river progressively increased downstream (Table I). The daily thermal gain (averaged over 12 months) between the headwater site and the Colorado River was about 6°C. In the Black Canyon National Monument the granite walls and shading greatly influenced the daily thermal regime. Kinnear [7] observed that vernal temperatures in the Black Canyon were actually warmer during the night, than during daytime (Figure 3), due to differential heating and cooling of the canyon walls. Since regulation, this daily cycle has been eliminated by the high-volume, cold discharge from Crystal Reservoir. The post-regulation river thermal regime is summarized in Table I. The major conclusion from these data is that the Taylor River and lower mainstream segments are much colder than before regulation and the thermal gain in Black Canyon is more dramatic (simply because the water is so cold at the head of the canyon during the warmest time of the year). Rhithron conditions [8] now extend well into the lower river segment.

The negative thermal gain of -0.7°C observed between Sites 9 and 11 remains largely inexplicable and may be an artifact of limited time-series data (the post-regulation thermal regima at this site were based only on data for one year, 1978), or a response to groundwater input. Several warm springbrooks (e.g., Tongue and Buttermilk Creeks) flow into the river between Sites 8 and 9. The lower Uncompahgre River is also apparently fed by considerable flow from surface aquifers. These side flows may warm the Gunnison River slightly; a subsequent thermal loss could then eventuate in downstream areas not influenced by groundwaters. Thus, the thermal gain estimate at Site 9 could be slightly high.

A strong correlation (r = .87) between the flow rate from Crystal Dam and river temperatures below the Black Canyon (Sites 8 and 9) was observed. At minimum flows (ca. 16 m^3/sec), which occurred only during spring and summer during the period for which thermograph records exist (1978-81), thermal gain in the canyon was 10-12°C; whereas, high flows (ca. 30 m^3/sec and greater) limited thermal gain to 2-3°C. Thus, a very predictable relationship exists between discharge temperature, discharge volume and temperature of the river at any point downstream, given some knowledge of seasonal trends in air temperature. However, heat storage in the granite walls of the Black Canyon undoubtedly limits variance in this relationship; river channels in more open, low-gradient terrain probably exhibit greater diurnal fluctuations.

The observed significant difference between pre- and post-impoundment temperature minima at Site 9 (0° vs. 2.8°, Table I) may be related to the flow-thermal gain relationship within the Black Canyon. Even though the midwinter thermal gain is generally low, high volume discharge limits heat loss. The canyon walls apparently absorb enough heat to ameliorate heat loss. Prior to regulation, low flows coincided with cold, midwinter air temperatures. Thus, the river froze over for periods of a few days to several weeks until air temperatures moderated to the extent that a thermal gain occurred relative to flow rate.

Concentrations of major ions in solution were highest at the downstream sites, indicating substantial salt loading in the lower river segment. Ion concentration was inversely related to seasonal trends in flow at the least regulated sites during 1979-80 (Figure 4). Dissolved solids in tail-water segments were consistently lower than at upstream sites and concentrations were much less variable (i.e., influenced by flow volume) over all sampling dates (Figures 5 and 6).

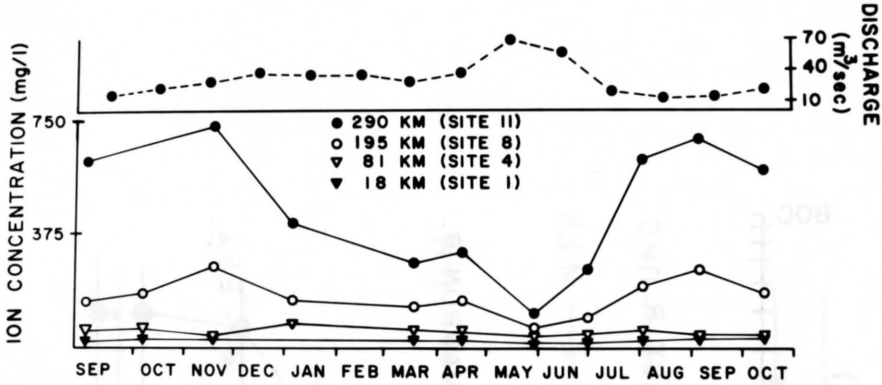

Figure 4. Concentration of the major ions (i.e., sum of Ca^{++}, Mg^{++}, Na^{+} and $SO_4^{=}$ concentrations) in water samples taken in time-series during 1979-80 at four locations along the Gunnison River profile. Flow rates (monthly means) are plotted only for Site 11 located near the confluence with the Colorado River.

Calcium was the dominant ion by percentage composition in the upper river (above the Black Canyon), while sulfate loading from side flows draining gypsum formations characterized the lower river segment (Figure 5). Sulfate-containing salts were observed in high concentrations (e.g., > 3000 mg/l) in the side flows (especially springbrooks and irrigation return flows) between Sites 7 and 9. The propensity of the reservoirs to sediment or precipitate dissolved solids was evinced in our data, but this loss was countered by loading rates nearly two orders of magnitude greater in the lower river segment (Figure 5).

Nitrate concentrations also increased in a downstream direction over the river continuum, but values were consistently elevated in tailwaters in comparison to sites above the reservoirs (Figure 6). The mobilization of nitrate is attributed to mineralization of organic matter (i.e., nitrification) and perhaps nitrogen fixation within the water column of the reservoirs. Nitrates were apparently utilized by autotrophic processes in riverine segments downstream from the dams (Figure 6). This was particularly evident in the Black Canyon, which is the segment least influenced by side flows. Benthic algae, particularly *Cladophora* spp., grow in profusion in all tailwater segments and are a dominant feature of the river bottom from Crystal Dam to Site 8. Tributary effects and turbid irrigation return flows apparently limited excessive growths of filamentous algae below Site 8, even though nutrient loading was apparent (Figure 6). However, thick accumulations of aufwuchs were present at the Dominguez Canyon Site (10) where we measured 3-5 cm accumulations of algae, fine silts, clays and organic detritus firmly attached to cobbles in riffle areas.

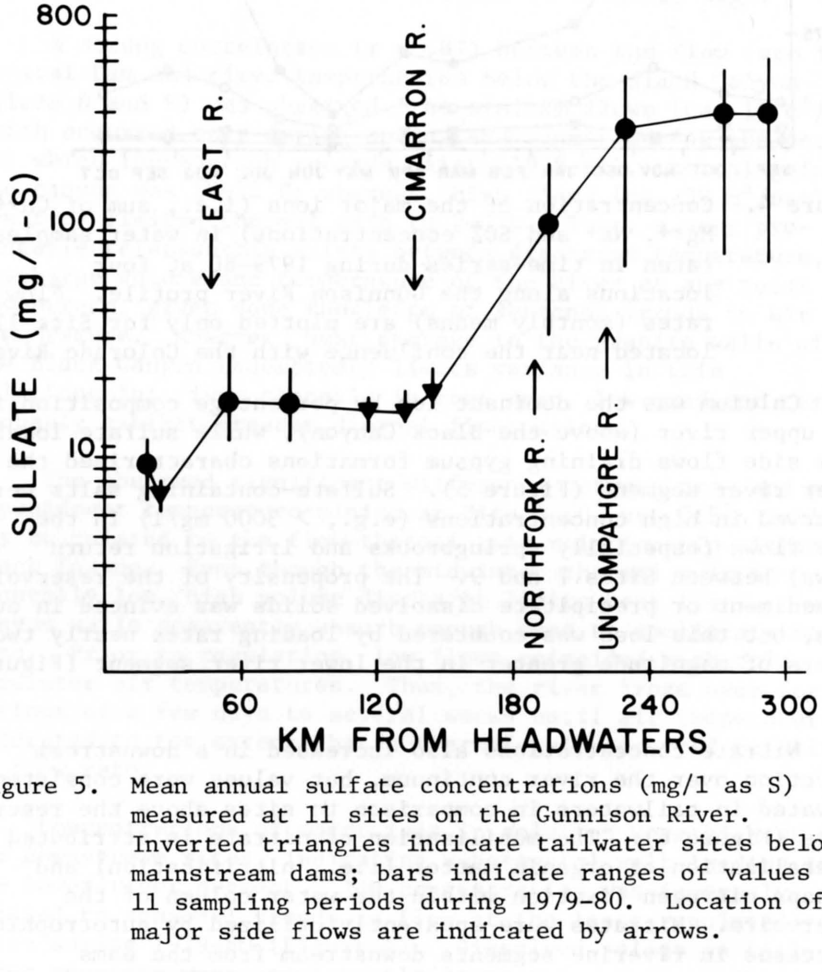

Figure 5. Mean annual sulfate concentrations (mg/l as S) measured at 11 sites on the Gunnison River. Inverted triangles indicate tailwater sites below mainstream dams; bars indicate ranges of values for 11 sampling periods during 1979-80. Location of major side flows are indicated by arrows.

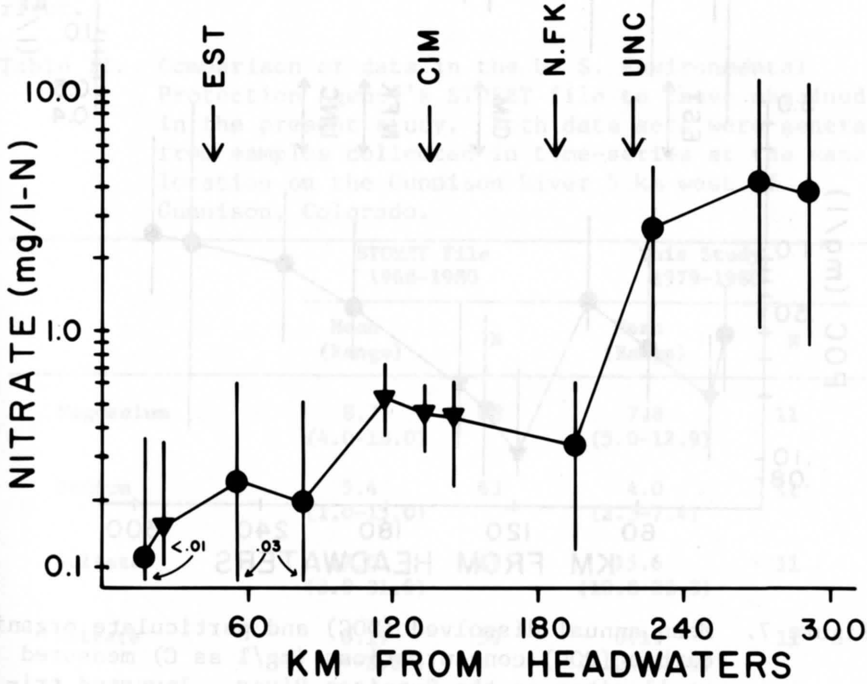

Figure 6. Mean annual nitrate concentrations (mg/l as N) measured at 11 sites on the Gunnison River. Inverted triangles indicate tailwater sites below mainstream dams; bars indicate ranges of values for 11 sampling periods during 1979-80. Location of major side flows are indicated by arrows.

The mineralization effect of the reservoirs was very evident in time-series measurements of particulate and dissolved organic carbon. Despite exports of plankton from the reservoirs, POC levels below the dams were consistently lower than in river segments immediately upstream from the impoundments and vice versa for DOC values. The total organic carbon pool in the river increased from ca. 1.0 to 10.0 mg/l, on the average from headwaters to the mouth (Figure 7). Agglutination processes (i.e., demobilization of dissolved solids by conversion to particulate carbon forms) were responsible for progressively increasing POC values downstream from Taylor Park and Crystal Dams. Much of the seston drift in these segments was due to

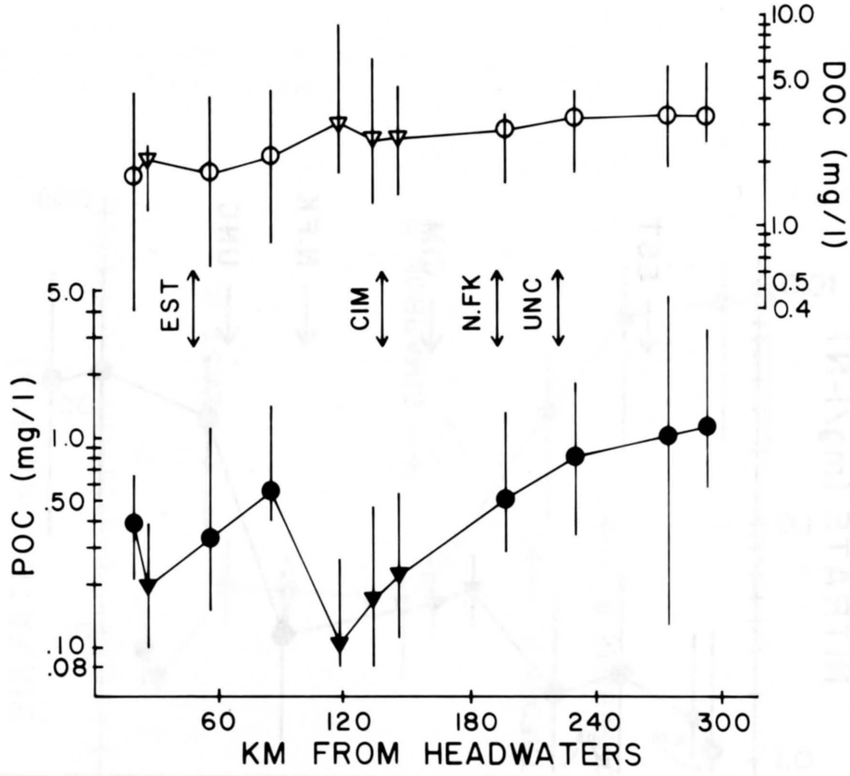

Figure 7. Mean annual dissolved (DOC) and particulate organic carbon (POC) concentrations (mg/l as C) measured at 11 sites on the Gunnison River. Inverted triangles indicate tailwater sites below mainstream dams; bars indicate range of values for 11 sampling periods during 1979-80. Location of major side flows are indicated by arrows.

sloughed filaments of *Cladophora* and other benthic algae. In lower river segments, side flows contributed significant amounts of allochthonous particulates; however, agglutination by autotrophic and micro-heterotrophic activity undoubtedly played a major role in size fractions and POC concentrations in this river segment, except during the spring freshet.

Thus, during 1979-80 the dissolved solids and organic carbon pool increased dramatically in a downstream direction; but, concentrations in the intensely regulated segments were greatly influenced by mineralization and precipitation within the reservoirs and, by agglutination as materials moved downstream in riverine segments. Time-series chemical data for periods previous to our study were limited to Site 4, upstream from Blue Mesa Reservoir. Our data were remarkably similar to these measurements (Table II) indicating that the trends reported here have been the norm since the Gunnison River was regulated. Dissolved and particulate solids loading undoubtedly occurred prior to regulation, but concentrations exported to the Colorado River were likely much lower and more erratic before irrigation return-flows were a significant feature of the lower river.

Table II. Comparison of data in the U. S. Environmental Protection Agency's STORET file to those obtained in the present study. Both data sets were generated from samples collected in time-series at the same location on the Gunnison River 5 km west of Gunnison, Colorado.

	STORET File 1968-1980		This Study 1979-1980	
	Mean (Range)	N	Mean (Range)	N
Magnesium	8.7 (4.0-18.0)	68	7.8 (5.0-12.9)	11
Sodium	5.4 (1.0-15.0)	63	4.0 (2.3-7.4)	11
Sulfate	19.0 (3.0-31.0)	69	15.6 (10.8-22.3)	11
Nitrate	0.19 (*-1.60)	59	0.19 (0.04-0.50)	11

*less than detection limit.

CONCLUSIONS

Hypolimnial-release impoundments on the Gunnison River have altered the physicochemistry of the riverine environment, mainly by reducing seasonal variability. Summer-cold, winter-warm conditions prevail in the river downstream from the dams. Dissolved solids (except $NO_3^=$) and particulate organic matter (POM) are reduced in concentration within reservoir tailwaters in comparison to concentrations in river segments above the reservoirs. Mobilization of $NO_3^=$ and other nutrients in reservoir effluents has stimulated thick growths of periphyton thalweg substrata, which has stabilized (armored) in response to elimination of spring flood flows. Inherent biophysical processes (e.g., communition and agglutination of POM; thermal gain via insolation) and side flows ameliorate or reset the consequences of regulation, as distance downstream from impoundments increases. Although the dissolved solids pool increases down the river profile, conditions 30-40 km downstream from the last dam (i.e., at Site 8) mimic the rhithron environment 115 km upstream (i.e., at Site 4). Physicochemistry of the Gunnison River near its confluence with the Colorado River is similar to pre-regulation, except that annual variance in discharge has decreased and dissolved solids increased.

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CHAPTER 5

THE INFLUENCE OF LAKE POWELL ON THE SUSPENDED SEDIMENT-PHOSPHORUS DYNAMICS OF THE COLORADO RIVER INFLOW TO LAKE MEAD

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INTRODUCTION

The Colorado River has been successively modified by the construction of several reservoirs, beginning in 1935 with the formation of Lake Mead by Hoover Dam. These reservoirs are located in a chain, and each one has an influence on the nutrient dynamics and productivity of the river and downstream reservoir [1]. Lake Mead derives 98% of its annual inflow from the Colorado River [2]. Historically, the Colorado River inflow was unregulated into Lake Mead. Regulation occurred in 1963, when Lake Powell was impounded by the construction of Glen Canyon Dam, approximately 450 km upstream. The formation of Lake Powell drastically altered the physical characteristics of the Colorado River inflow to Lake Mead [1]. Regulated releases from Glen Canyon Dam have eliminated the spring discharge peaks that historically resulted from spring flooding in the Upper Colorado River drainage basin. Temperatures in the Colorado River below Lake Powell have been reduced 5-10°C during the spring and summer, due to cold hypolimnetic releases from Glen Canyon Dam. There were also marked reductions in the suspended sediment loads due to decreases in spring and summer discharge peaks. The turbid overflows that once extended across the Upper Basin of Lake Mead [3] during spring were not evident in 1977-78 [2]. The Upper Basin of Lake Mead is now severely phosphorus deficient, and this appears to have been caused by reductions in suspended sediment loading [1].

Phosphorus has been reported by many investigators as the most common nutrient limiting phytoplankton productivity [4]. Phosphorus loading models are generally based on total phosphorus (total-P), but this fraction may not accurately reflect the amount of phosphorus available for biological uptake in turbid river systems [5]. Total-P loading models greatly overestimate the trophic states in Lake Powell and

Lake Mead [2,6].

Little emphasis has been placed on the interaction between suspended sediments and dissolved inorganic phosphorus in rivers [7,8]. The removal of inorganic phosphorus by suspended sediment, however, does appear to be a sorption rather than a precipitation process [9]. Loosely bound phosphorus on suspended sediments is more readily available than precipitated phosphorus [10]. Wang and Brabec [11], in their work on the Illinois River at Peoria Lake, found that dissolved inorganic phosphorus was actively adsorbed by suspended sediments. Other workers have also observed this process occurring in oxygenated rivers and lakes [12,13]. Mayer and Gloss [14] have shown that phosphorus buffering by suspended sediments in the turbid Colorado River is an important mechanism for sustaining the dissolved inorganic phosphorus pool in Lake Powell. It appears that this same mechanism occurred in Lake Mead when it received turbid inflows from the Colorado River.

The intent of this paper is to discuss the possible effects that the formation of Lake Powell has had on the suspended sediment-phosphorus dynamics of the Colorado River inflow to Lake Mead. This is based on results from recent investigations and on preliminary results of research conducted in the late-summer and early-fall of 1981.

STUDY AREA

This study focuses on a 1000 km stretch of the Colorado River which includes two of the largest reservoirs in the Western Hemisphere, Lake Powell and Lake Mead (Figure 1). Comparative morphometric characteristics for the reservoirs are presented in Table I.

Lake Powell was formed by the construction of Glen Canyon Dam in 1963. The reservoir covers a 300 km stretch of Glen Canyon, and is morphometrically complex with over 3000 km of shoreline. The Colorado and San Juan Rivers provide 96% of the total annual inflow to this reservoir. Approximately 60% occurs in late-spring and early-summer (May-July), as a result of snowmelt [15] from the Upper Colorado River Basin (Figure 2). Lake Mead is the second of four major reservoirs on the main stem Colorado River. It is a large deep-storage reservoir 180 km in length, extending from the mouth of the Grand Canyon at Pierce Ferry to Hoover Dam in Black Canyon (Figure 1). The dominant hydrologic input to this reservoir is from the Colorado River which provides approximately 98% of the total annual inflow. The Virgin and Muddy Rivers discharge approximately 1% into the Overton Arm of Lake Mead. The remainder is derived from Las Vegas Wash, a secondarily-treated sewage and industrial effluent stream from metropolitan Las Vegas, which discharges into Las Vegas Bay [2].

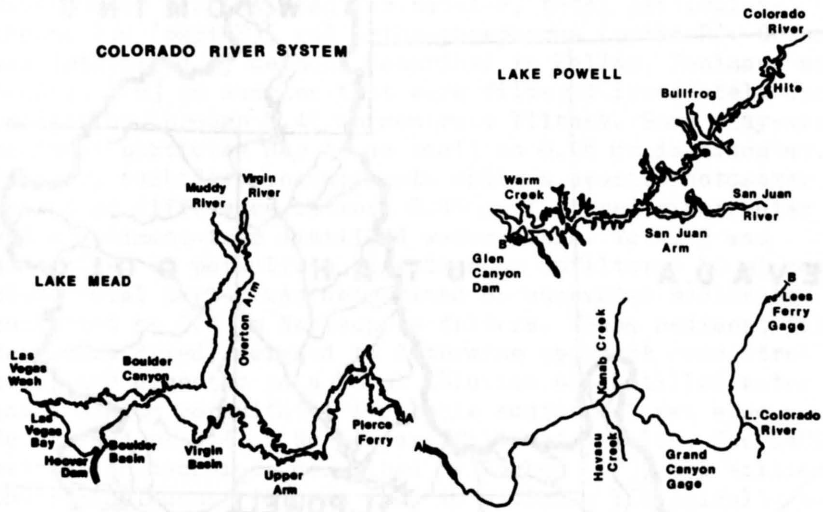

Figure 1. Map of the Colorado River System from Lake Powell to Lake Mead.

Table I. Morphometric Characteristics of Lake Powell and Lake Mead.

Parameter	Lake Powell	Lake Mead
Maximum operating level (m)	1128	374
Maximum depth (m)	171	180
Mean depth (m)	51	55
Surface area (km^2)	653	660
Volume ($m^3 x 10^9$)	33	36
Maximum length (km)	300	183
Maximum width (km)	25	28
Shoreline development	26	10
Discharge depth (m)	70	100
Approximate storage ratio (years)	2	4

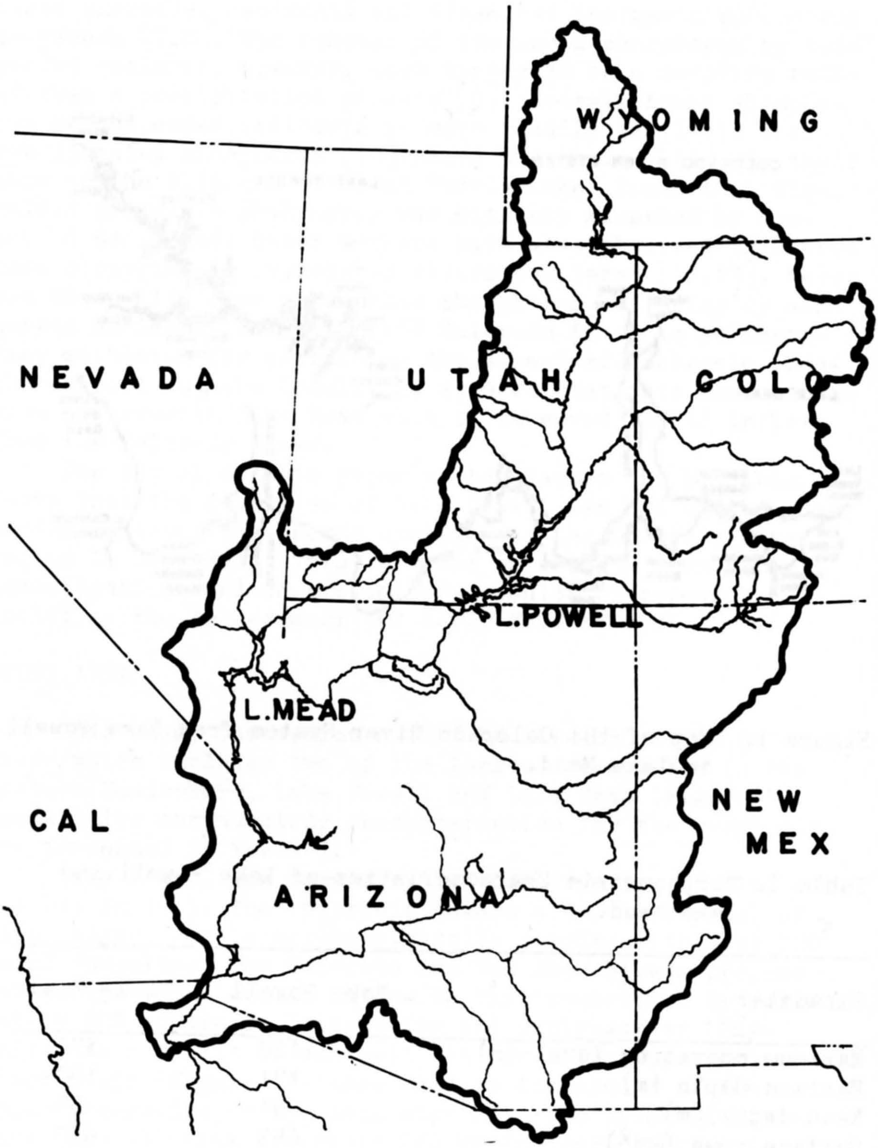

Figure 2. Map of the Colorado River Drainage Basin.

METHODS

The primary inflows to Lake Powell and Lake Mead were sampled monthly from August through October, 1981. A composite sample, consisting of several tows with a 3-liter Van Dorn bottle, was collected from the surface at each station. River water was analyzed for total-P, total particulate phosphorus (part-P), and ortho-phosphorus (ortho-P). Ortho-P was determined by methods described in Kellar, Paulson, and Paulson [16] on samples that were filtered immediately upon collection through 0.45 µm membrane filters. Some clay-sized sediment particles may be as small as 0.06 µm in diameter. However, turbidity measurements using a spectrophotometer showed no difference between 0.45 µm filtered river water and a sediment-free distilled water blank. Total-P was determined by persulfate digestion on unfiltered 50 ml samples. Total part-P was determined on suspended sediments collected on 0.4 µm Nucleopore filters. These sediment-filters were dried, weighed to determine sediment concentration, and digested in a 50 ml solution of distilled water and ammonium persulfate. Available sediment-P was also determined on 0.4 µm Nucleopore filtered samples. The NaOH extraction technique described by Sagher [17], and Williams, Shear and Thomas [18] was used to estimate biologically available sediment-P. NaOH extractable-P gives an approximate estimate of the amount of inorganic phosphorus that is biologically available through sorption reactions with suspended sediments. This fraction includes non-occluded inorganic phosphorus that is loosely bound to iron and aluminum in sediments. Much of the work that has been done on suspended sediment-P dynamics uses only the total-P fraction in high sediment:water ratios. High sediment:water ratios are indicative of soils and sediments rather than suspended riverine sediments [14]. The estimates of total part-P and available sediment-P are based on using natural river water, which has a low sediment:water ratio. The sediment:water ratio appears to be an important factor influencing sorption reactions by suspended riverine sediments.

DATA SOURCES

Suspended sediment data for the Grand Canyon gaging station were derived from "Quality of Surface Waters for the United States," and discharge data were obtained from "Surface Water of the United States," U.S. Geological Survey Water-Supply Papers Part 9. Colorado River Basin (1940-1970). After 1970, these data were taken from "Water Resources Data for Arizona or Nevada" prepared jointly by the U.S. Geological Survey and state agencies.

Suspended Sediment Loads

Suspended sediment loads in the Colorado River at Grand Canyon were extremely high prior to the formation of Lake Powell (Figure 3). In years of high runoff, up to 140 million tons per year of suspended sediments flowed into Lake Mead. The majority of this occurred during the spring runoff periods (Figure 3). Impoundment of Lake Powell in 1963 resulted in a 70-80% reduction in suspended sediment loads in the Grand Canyon (Figure 3). The direct drainage area to Lake Mead was reduced to a few tributary inputs in Grand Canyon (Figure 2). The Little Colorado River, which enters the main stem Colorado River 40 km above the Grand Canyon gaging station, is now the only appreciable source of sediments to the river [19]. Suspended sediment inputs from the Little Colorado River can be quite high when floods occur, but annual loading to Lake Mead is still far below that which occurred in pre-Lake Powell periods (Figure 3).

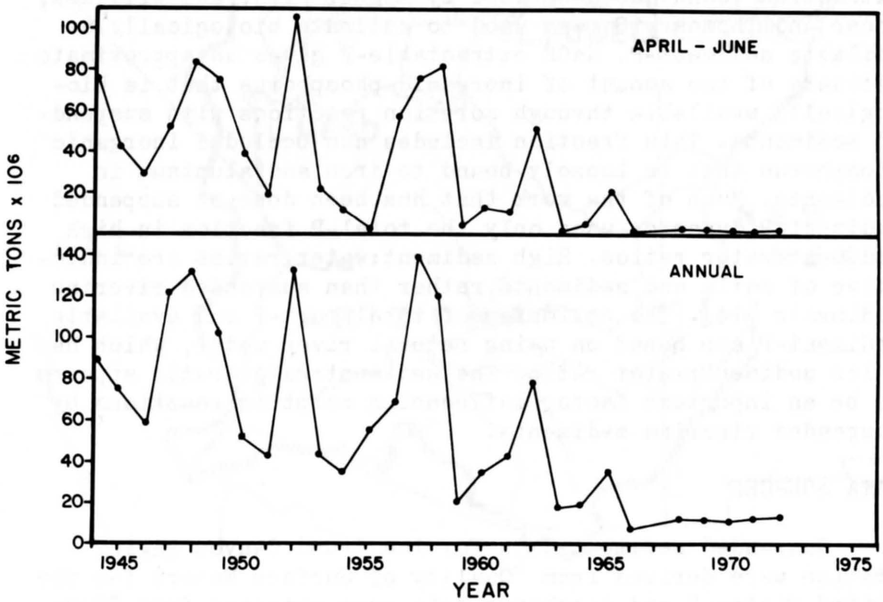

Figure 3. Historical Annual and Spring Suspended Sediment Loads at Grand Canyon Gaging Station. (USGS Data)

Effects on Total-P

Gloss, Mayer, and Kidd [20] demonstrated that total-P concentrations were closely associated with suspended clays in river water. A similar relationship has been observed by other researchers [21]. Preliminary measurements made on the Colorado River above Lake Powell and Lake Mead, and on the San Juan River inflow to Lake Powell, also show a close correlation between total-P and suspended sediment concentrations (Figure 4). The relationship appears to be linear in the range of suspended sediment concentrations that occurred in the Colorado River from August to October, 1981. This research is continuing to determine if the relationship also holds for other seasons.

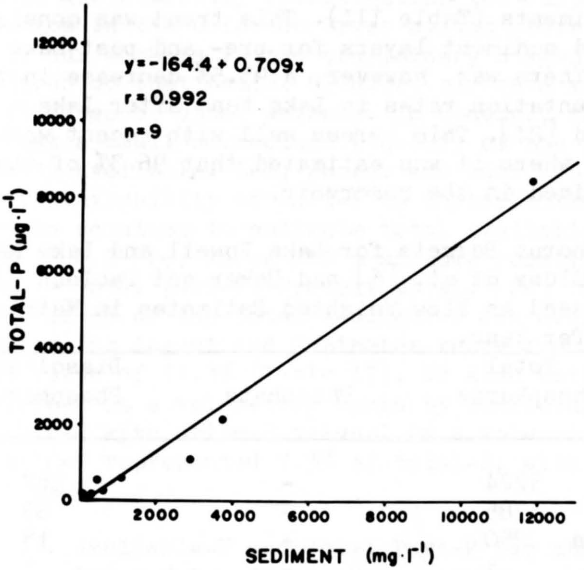

Figure 4. Total-P Concentrations as Related to Suspended Sediment Concentrations in the Colorado River System.

Phosphorus budgets were recently determined for Lake Powell [6] and Lake Mead [22] (Table II). It is readily apparent that Lake Powell is serving not only as a sediment trap, but also as a phosphorus sink. Gloss et al. [6] reported that over 95% of the phosphorus loads entering Lake Powell were in the particulate form. They further concluded that the phosphorus retention coefficients determined for Lake Powell were among the highest reported to date. This probably reflects the strong relationship between phosphorus and suspended sediments in the Colorado River. The phosphorus retention coefficients determined for Lake Mead were not as high as Lake Powell. This was caused by high inputs of ortho-P from the Las Vegas Wash inflow. Las Vegas Wash forms a density current in Lake Mead [22], resulting in a large percentage of the phosphorus input being loaded into the hypolimnion. Hoover Dam is operated from a hypolimnion discharge, which rapidly strips phosphorus from the reservoir [23]. The combination of these two processes greatly reduces retention of ortho-P and total-P in Lake Mead. However, Prentki et al. [24] found that total-P in Lake Mead sediments was high (300-1000 mg/l). Inorganic-P averaged 86% of total-P. Measurements made on various phosphorus fractions in the major river inflows to Lake Powell and Lake Mead also indicate that the majority of total-P is inorganic-P, bound to suspended sediments (Table III). This trend was consistent in Lake Mead sediment layers for pre- and post-Lake Powell periods. There was, however, a 93.5% decrease in the phosphorus sedimentation rates in Lake Mead after Lake Powell was formed [24]. This agrees well with recent work on Lake Powell [6], where it was estimated that 96.3% of the total-P was retained in the reservoir.

Table II. Phosphorus Budgets for Lake Powell and Lake Mead from Gloss et al. [6] and Baker and Paulson [22], Expressed as Flow Weighted Estimates in Metric Tons Per Year.

Location	Total Phosphorus	Phosphate	Dissolved Phosphorus
Lake Powell			
Colorado River	5224	-	267
San Juan River	785	-	83
Other tributaries	250	-	15
Precipitation	1	-	1
Glen Canyon Dam	229	-	100
R =	.963		.727
Lake Mead			
Colorado River	199	56.8	-
Las Vegas Wash	263	136.6	-
Hoover Dam	123	110.6	-
R =	.734	.428	

R = Experimentally determined retention coefficient

Table III. Concentrations of Total-P, Part-P and Part-P Expressed as a Percentage of Total-P for the San Juan and Lower Colorado Rivers for August and September, 1981, in μg/l (±95% CL).

River	Month	Total-P	Part-P	% of Total-P
San Juan	Aug	1149 (±53.6)	1022 (±10.1)	89
	Sep	124 (± 3.8)	100 (±44.2)	81
L. Colorado	Aug	239 (±17.4)	268 (±15.0)	112
	Sep	77 (± 4.2)	70 (± 1.9)	90

Availability of Phosphorus from Suspended Sediment

It has been shown [14] that the suspended sediments in the Colorado River inflow to Lake Powell have the capability of desorbing approximately 20-30 μg/l of dissolved inorganic-P. We are currently investigating the suspended sediment-P dynamics in the Colorado River system above and below Lake Powell. Our work is in the preliminary stages, and must be considered on that basis. However, our data thus far agree with findings of other workers. In general, these data indicate that a small percentage (10-30%) of the total-P is biologically available. Lee, Jones, and Rast [25], in their review of availability of part-P to phytoplankton, have established an equation to estimate total available-P. Available-P = SRP + 0.2 PP_T, where SRP = soluble reactive phosphorus, and PP_T = total part-P. Prentki et al. [24] found that an average of 9% of the total sediment-P was available. Our estimates for August and September range from 7.1-19.2% with a mean value of 11.3% (Table IV). We also estimated total available-P on a volumetric basis by combining sediment available-P with ortho-P values. On a volumetric basis total available-P represented 7.3% of total-P, with a range of 1.7-14.1%.

Table IV. Available-P and Total Part-P for the Upper and Lower Colorado and San Juan Rivers During August and September, 1981, in μg/l (±95% CL).

River	Month	Part-P	Available-P	% of Part-P
U. Colorado	Aug	294 (±13.5)	21.0 (± 5.6)	7.1
	Sep	-	2.4 (± 0.4)	-
San Juan	Aug	72 (± 1.1)	6.9 (± 0.4)	9.5
	Sep	1850 (±15.2)	355 (±43.2)	19.2
L. Colorado	Aug	240 (±18.6)	29 (±28.0)	12.1
	Sep	685 (±99.6)	58.2 (±33.4)	8.5

Effects on Productivity

The formation of Lake Powell in 1963 resulted in marked reductions in suspended sediment loading to Lake Mead. Total-P was reduced accordingly, and the Upper Basin of Lake Mead has since become severely phosphorus deficient [1]. Phytoplankton productivity in the Upper Basin averaged 4612 mg $C/m^2 \cdot day$ during the 1955-62 period [24]. Productivity decreased to an average of 503 mg $C/m^2 \cdot day$ after Lake Powell was formed in 1963. Although only a small percentage of the total-P in the river inflows is biologically available, the historic sediment loads (up to 140 million tons per year) were apparently sufficient to sustain the dissolved inorganic phosphorus pool, and higher productivity.

ACKNOWLEDGEMENTS

We wish to express our appreciation to several people for assistance with this paper. We would like to thank Laurie Vincent and Thom Hardy for typing the manuscript, and Sherrell Paulson and Jim Williams for the drawing and photographing of figures. Also conversations with Richard Prentki, Penelope Naegle, and John Baker were helpful.

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T R I B U T E

T O

T E R R Y D . E V A N S

January 6, 1955 - December 2, 1981

He rode a horse, he drove a boat
He helped us all, he gave us hope
He entered science, he did it well
He had great goals, he did excel
He helped us laugh, he made us cry
He's in our hearts, he'll never die.

- L. J. Paulson

Terry D. Evans was killed in a boating accident in Grand Canyon, a short time after he presented his paper at the symposium from which this book is compiled. Terry and two of his associates were attempting to find a suitable site to set up a water sampler when a large wave swamped the research boat, forcing the occupants to abandon it. The boat driver and the research assistant made it to shore. Terry's body was recovered after a long search by park personnel on December 29.

Terry's thesis research consisted of a study of the suspended sediment-phosphorus dynamics in the principal inflows to Lake Powell and Lake Mead. He was employed as a research associate in the Lake Mead Limnological Research Center and supervised the field sampling programs on Lake Powell, Lake Mead, Lake Mohave, and Lake Havasu. The preliminary results of his research were presented at the symposium.

Terry was awarded a posthumous Master of Science in Biology and the David Bruce Dill Award in Environmental Biology at his memorial service.

Terry would have made an outstanding limnologist. He had already made a significant scientific contribution to our efforts to better manage the Colorado River resources. He was a special individual, devoted to his family, friends and profession. He will never be forgotten by those of us who continue to work on the Colorado River.

CHAPTER 6

MASS BALANCE MODEL ESTIMATION OF PHOSPHORUS CONCENTRATION IN RESERVOIRS

David K. Mueller
Bureau of Reclamation
Denver, Colorado

INTRODUCTION

The significance of phosphorus in reservoirs and lakes stems from its association with the process of eutrophication, or fertilization of the water body. When eutrophication becomes advanced, severe water quality problems can develop. These include blooms of nuisance algae and reduction of dissolved oxygen concentration. Such conditions impact fisheries, domestic water supply, and recreational use, both in the water body and downstream. Phosphorus, in the form of phosphate, is a necessary nutrient and has long been considered a major limiting factor to algal growth. This theory is supported by studies demonstrating that introduction of phosphorus tends to stimulate algal growth [1] and that control of phosphorus loading has the opposite effect [2].

Consequently, the need arose for predictive techniques for the evaluation of phosphorus reduction as a means to control eutrophication. Since 1969, a variety of models has been proposed using the input-output or mass balance approach. Though most model verification has been conducted using data from natural lakes in Europe and eastern North America, there has been a tendency to extrapolate validity to lakes and reservoirs throughout the northern temperate zone. The purpose of the present work is to test that assumption using the extensive data base developed by the U.S. Environmental Protection Agency's NES (National Eutrophication Survey). Specifically, several model formulations are compared as to their accuracy in predicting phosphorus concentrations in reservoirs in the western United States.

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Model and Data Selection

Several models have been developed from the steady-state solution to a phosphorus mass balance equation proposed by Vollenweider [3]. Five of these, commonly used in lake quality assessment, were chosen for evaluation and comparison.

As listed in Table I, these are:

1. The Vollenweider-1975 model [4], which assumes a constant settling velocity;
2. The Jones-Bachmann model [5], which assumes a constant sedimentation coefficient;
3. The Dillon-Rigler model [6], which uses phosphorus retention calculated from observed data;
4. The Dillon-Kirchner model [7], which estimates phosphorus retention as a function of hydraulic load;
5. The Vollenweider-1976 model [8], which estimates phosphorus flushing from the inverse of hydraulic detention.

Variables used in these models are defined as follows:

P = phosphorus concentration (mg/L)
L = areal phosphorus loading rate ($g/m^2/yr$)
z = mean lake depth (m)
τ = hydraulic detention time (yr)
q_s = areal hydraulic loading rate, z/τ (m/yr)
R = fraction phosphorus retention
R_p = empirical estimate of R

Data were compiled from NES [9]. Selection was based on the following criteria:

1. The water body was a manmade reservoir located in the western continental United States;
2. All data were available for solution of the phosphorus loading models listed in Table I;
3. The phosphorus retention calculated from inflow and outflow data was greater than zero.

The resultant data set included 68 reservoirs, distributed by state as shown in Figure 1. A statistical summary is given in Table II. Using criteria in Table III, 5 reservoirs were classified oligotrophic, 16 mesotrophic, and 47 eutrophic. This data set then represents wide geographic, hydrologic, morphologic, and trophic ranges.

Table I. Forms of Phosphorus Mass Balance Models

1.	Vollenweider-1975	$P = \frac{L}{10 + z/\tau}$
2.	Jones-Bachmann	$P = \frac{0.84\ L}{z(0.65 + 1/\tau)}$
3.	Dillon-Rigler	$P = \frac{L\tau}{z}\ (1-R)$
4.	Dillon-Kirchner	$R_p = 0.426 \exp\ (-0.271\ q_s) + 0.574 \exp\ (-0.00949\ q_s)$
		$P = \frac{L\tau}{z}\ (1-R_p)$
5.	Vollenweider-1976	$P = \frac{L/q_s}{(1 + \sqrt{\tau})}$

Figure 1. Distribution by state of reservoirs included in the data set.

Table II. Data Statistics

	P (mg/L)	L (g/m^2yr)	z (m)	τ (yr)	R -
Arithmetic Mean	0.055	7.44	16.2	1.37	0.51
Geometric Mean	0.037	1.82	11.8	0.67	0.39
Standard Deviation	0.061	25.78	12.7	2.14	0.30
Maximum	0.371	183.51	59.2	15.20	0.99
Minimum	0.007	0.08	0.6	0.003	0.03

Table III. Trophic State Classification Criteria

Comparison	Classification		
	Oligotrophic	Mesotrophic	Eutrophic
Total phosphorus (mg/L)	<0.010	0.010-0.020	>0.020
Chlorophyll _a_ (μg/L)	<4	4-10	>10
Secchi depth (m)	>3.7	2.0-3.7	<2.0
Hypolimnetic D O (% sat.)	>80	10-80	<10

(after Allum _et. al_.[12])

Each model form was fit to the data set using a Gaussian nonlinear fitting algorithm available on the level 8.0 version of the Statistical Package for the Social Sciences [10, 11]. Fitted forms and resulting coefficient values are listed in Table IV. An interesting comparison can be made between the fitted versions of the Vollenweider-1976 and the Jones-Bachmann models. The latter is:

$$P = \frac{0.882\ L}{z(1.61 + 1/\tau)} \tag{1}$$

Solving to eliminate the coefficient yields:

$$P = \frac{L}{z(1.83 + 1.13/\tau)} \tag{2}$$

which may be approximated:

$$P = \frac{L}{z(2 + 1/\tau)} \tag{3}$$

The Vollenweider-1976 best fit version is:

$$P = \frac{L/q_s}{(1 + 2.09\tau^{0.832})} \tag{4}$$

Multiplication by τ/τ leaves, on rearrangement:

$$P = \frac{L}{z(2.09\tau^{-0.168} + 1/\tau)} \tag{5}$$

for which equation 3 is, again, a reasonable approximation. For this reason, equation 3 was named the Combination best fit model and was included in the analysis.

ANALYSIS

Two criteria were used to judge model accuracy. The first of these was the root-mean-square error of logarithmically transformed estimations, for which the computational form is:

$$S_m = \left\{ \frac{\sum_{i=1}^{68} \left[\log_{10}(P_o)_i - \log_{10}(P_e)_i\right]^2}{d_f} \right\}^{1/2} \tag{6}$$

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Table IV. Best Fit Models

Type	Form	Coefficients
Vollenweider-1975	$P = \dfrac{L}{\alpha + z/\tau}$	$\alpha = 16.4$
Jones-Bachmann	$P = \dfrac{\alpha L}{z(\beta + 1/\tau)}$	$\alpha = 0.882$ $\beta = 1.61$
Dillon-Kirchner 1/	$R_p = \alpha \exp(\beta q_s) + (1-\alpha) \exp(\gamma q_s)$ $P = \dfrac{L\tau}{z}(1-R_p)$	$\alpha = 0.290$ $\beta = -0.556$ $\gamma = -0.00483$
Vollenweider-1976	$P = \dfrac{L/q_s}{1 + \alpha\tau^{\beta}}$	$\alpha = 2.09$ $\beta = 0.832$

1/ Due to numerical problems in the computational algorithm only 64 reservoirs were included in fitting this equation.

where S_m = standard error of estimation for the model
P_o = observed phosphorus concentration (mg/L)
P_e = estimated phosphorus concentration (mg/L)
d_f = model degrees of freedom

The variable d_f was calculated for each model as the difference between sample size and the number of fitted parameters in the model.

Confidence intervals for the estimation at the meso-eutrophic boundary phosphorus concentration can then be calculated from the model error values:

$$CL = 0.020 * 10^{\left[\pm t_{d_f}^{\alpha/2} \, (S_m)\right]} \tag{7}$$

where CL is the upper or lower confidence limit in mg/L.

The second accuracy criterion was the correlation between observed and estimated phosphorus concentrations. This was calculated as the Pearson product moment coefficient. A comparison of standard errors, 90 percent confidence intervals, and correlation coefficients is given in Table V. The Dillon-Rigler model is the only one which achieves a tolerable fit with the observed data (R = 0.86). Graphical results of estimated vs. observed phosphorus concentration for the Dillon-Rigler and Combination models are shown in Figures 2 and 3.

Using the two judgment criteria, the models were grouped by relative accuracy as shown in Table VI. Significance of differences between groups was then tested by comparing mean standard errors and correlation coefficients. Squared error values were compared with an F test of variance. Correlation coefficients were tested using the Fisher transformation method, as described by Bryant [13, p. 140]. Test statistic values and significance levels are given in Table VII. These results leave little doubt that the Dillon-Rigler model is indeed more accurate than any other. The differences among the remaining groups are also significant, though to a lesser degree between groups 3 and 4 and between groups 4 and 5.

RESULTS

The Dillon-Rigler and Combination models were used to develop standard Dillon graphs (Figures 4 and 5) of computed areal phosphorus load vs. reservoir depth. These graphs show the effect of model uncertainty on the prediction of trophic state. The dashed line indicating a phosphorus

Table V. Model Statistics for 68 Reservoirs

	Standard 1/ error	Correlation 1/ coefficient	90% Confidence 2/ interval
Original Formulations:			
Vollenweider-1975	0.417	0.52	0.004-0.097
Jones-Bachmann	0.367	0.65	0.005-0.080
Dillon-Rigler	0.200	0.86	0.009-0.043
Dillon-Kirchner	0.387	0.56	0.005-0.086
Vollenweider-1976	0.387	0.64	0.005-0.087
Best Fit Formulations:			
Vollenweider-1975	0.407	0.48	0.004-0.097
Jones-Bachmann	0.327	0.67	0.006-0.069
Dillon-Kirchner	0.371	0.60	0.005-0.081
Vollenweider-1976	0.324	0.68	0.006-0.068
Combination	0.325	0.68	0.006-0.068

1/ Based on $\log_{10}$ transformed values.
2/ For P_e = 0.020 mg/L.

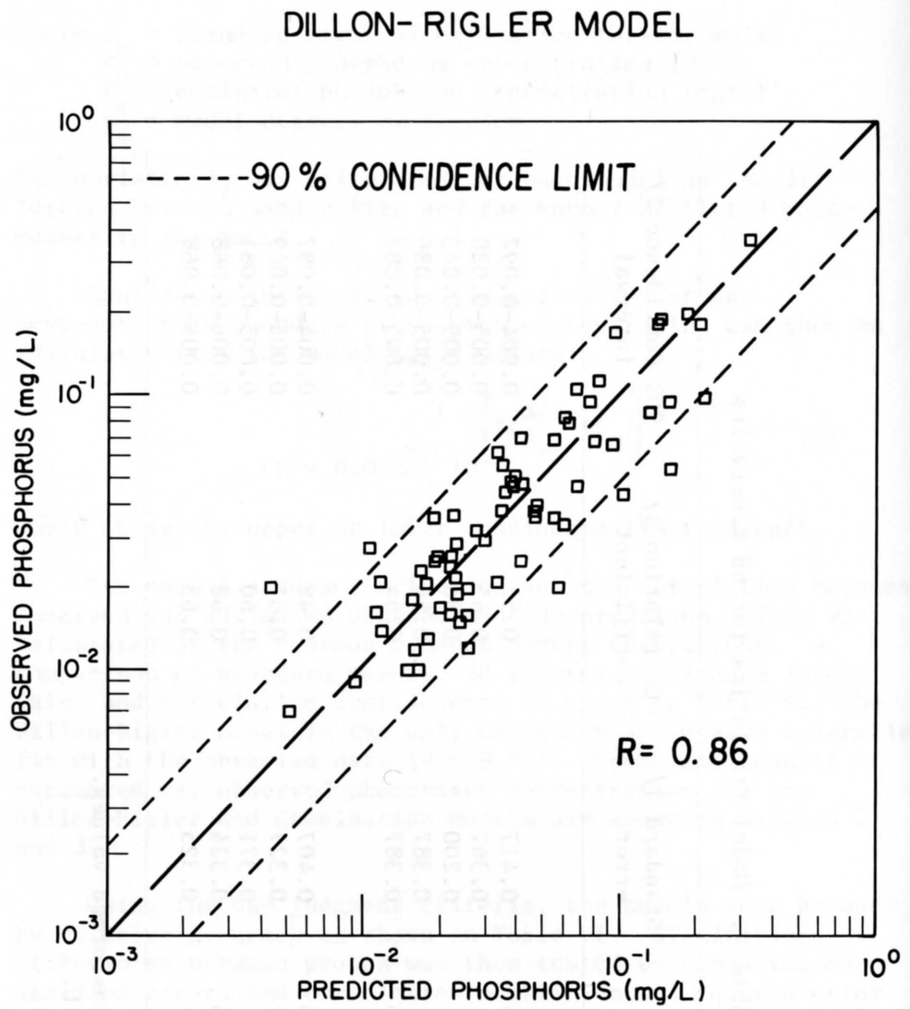

Figure 2. Comparison of observed phosphorus concentrations and Dillon-Rigler estimations showing the 90 percent confidence interval.

COMBINATION MODEL

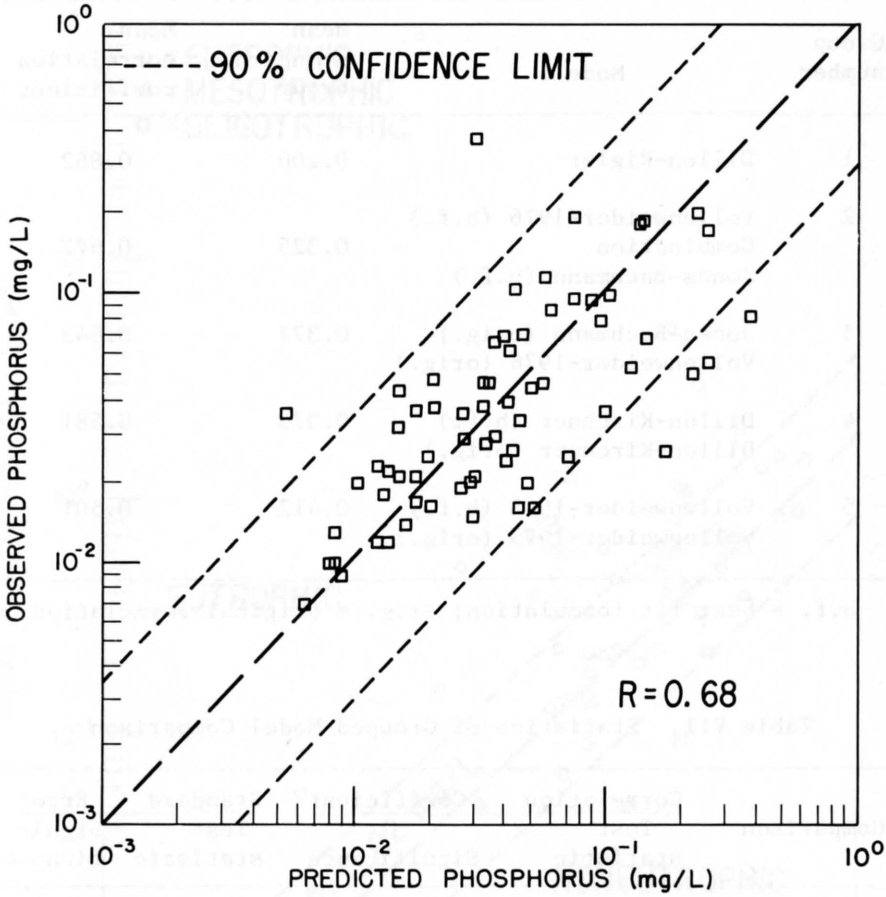

Figure 3. Comparison of observed phosphorus concentrations and Combination model estimations showing the 90 percent confidence interval.

Table VI. Models Grouped by Relative Accuracy

Group number	Models	Mean standard error	Mean correlation coefficient
1	Dillon-Rigler	0.200	0.862
2	Vollenweider-1976 (b.f.) Combination Jones-Bachmann (b.f.)	0.325	0.677
3	Jones-Bachmann (orig.) Vollenweider-1976 (orig.)	0.377	0.643
4	Dillon-Kirchner (b.f.) Dillon-Kirchner (orig.)	0.379	0.581
5	Vollenweider-1975 (b.f.) Vollenweider-1975 (orig.)	0.412	0.501

b.f. = best fit formulation; orig. = original formulation

Table VII. Statistics of Grouped Model Comparison

Comparison	Correlation Coefficient		Standard Error	
	Test statistic	Significance	Test statistic	Significance
1 vs. 2	2.76	<0.01	2.64	<0.01
2 vs. 3	0.35	0.36	1.35	0.11
2 vs. 4	0.92	0.18	1.36	0.11
2 vs. 5	1.57	0.06	1.61	0.03
3 vs. 4	0.58	0.28	1.01	0.48
3 vs. 5	1.24	0.11	1.19	0.24
4 vs. 5	0.66	0.26	1.18	0.24

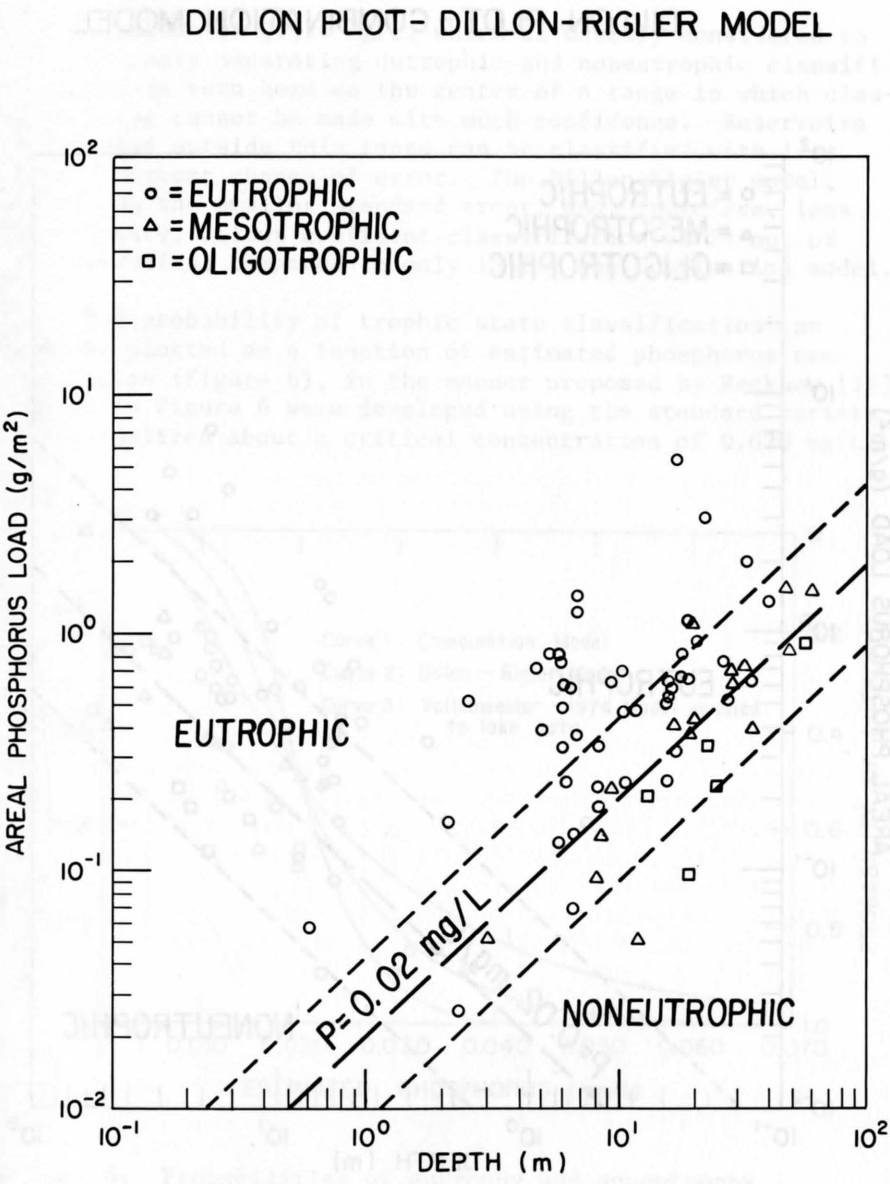

Figure 4. Dillon graph of Dillon-Rigler model results showing the eutrophic-noneutrophic boundary phosphorus concentration with a 90 percent confidence interval.

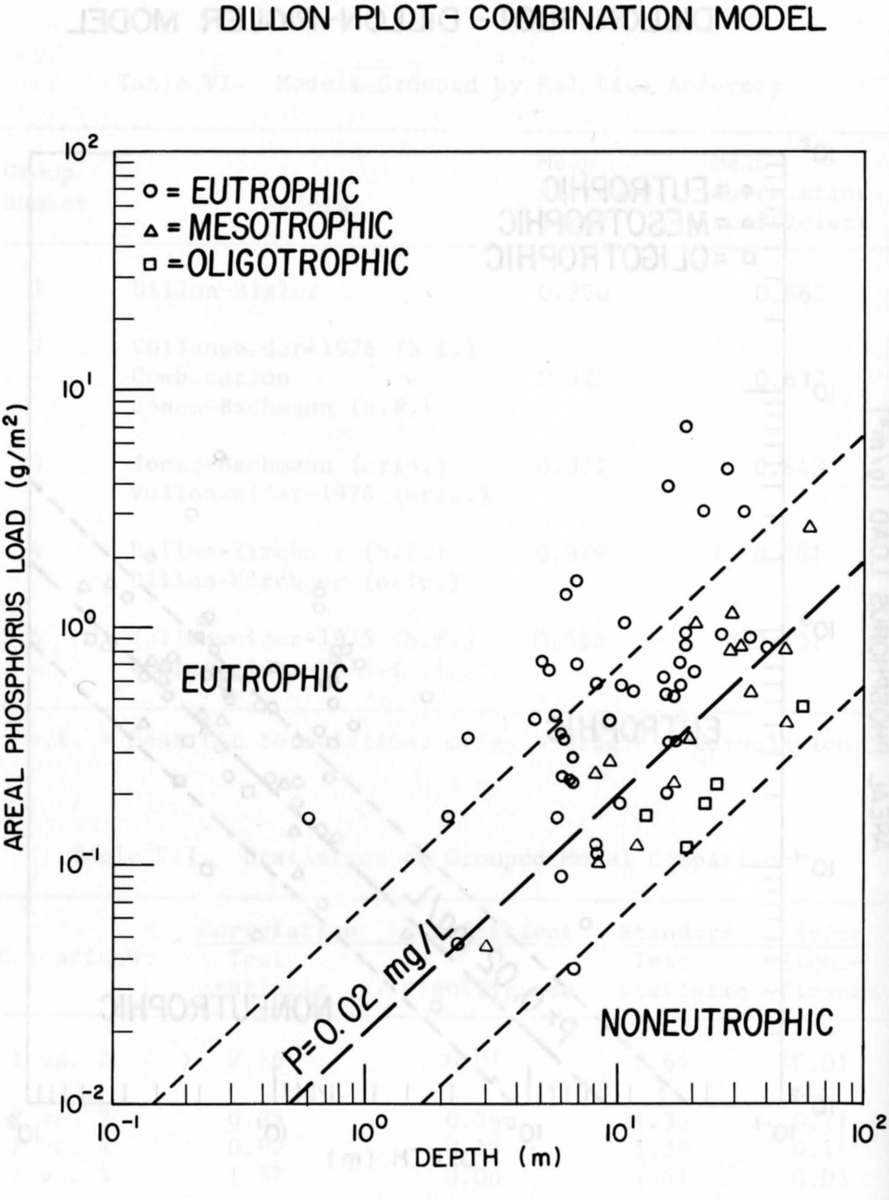

Figure 5. Dillon graph of Combination model results showing the eutrophic-noneutrophic boundary phosphorus concentration with a 90 percent confidence interval.

concentration of 0.020 mg/L, which is usually considered to be a boundary separating eutrophic and noneutrophic classifications, is seen here as the center of a range in which classification cannot be made with much confidence. Reservoirs which plot outside this range can be classified with less than 5 percent chance of error. The Dillon-Rigler model, which has the smaller standard error, and, therefore, less uncertainty, allows confident classification of 28 out of 68 reservoirs, compared to only 18 for the Combination model.

The probability of trophic state classification can also be plotted as a function of estimated phosphorus concentration (Figure 6), in the manner proposed by Reckhow [14]. Curves in Figure 6 were developed using the standard variate (z) normalized about a critical concentration of 0.020 mg/L:

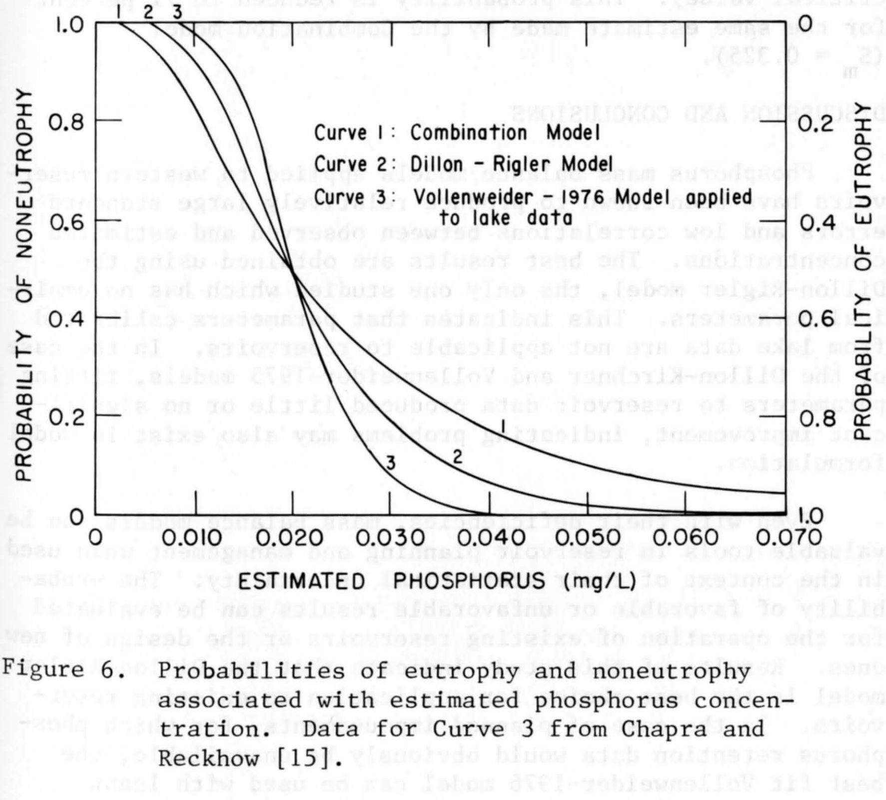

Figure 6. Probabilities of eutrophy and noneutrophy associated with estimated phosphorus concentration. Data for Curve 3 from Chapra and Reckhow [15].

$$z = \frac{\log_{10}(0.020) - \log_{10}(P_e)}{S_m} \quad (8)$$

The three curves represent: (1) the Combination model, (2) the Dillon-Rigler model, and (3) the Vollenweider-1976 model applied to 117 North American natural lakes with a standard error (S_m) of approximately 0.17 [15]. Again, the advantage of a smaller standard error can be seen. As S_m increases, the uncertainty also increases at all values of estimated phosphorus except the critical concentration. For example, a reservoir with a phosphorus concentration of 0.030 mg/L estimated by the Dillon-Rigler model (S_m = 0.200), has an 81 percent probability of being accurately classified eutrophic (having a phosphorus concentration greater than the critical value). This probability is reduced to 71 percent for the same estimate made by the Combination model (S_m = 0.325).

DISCUSSION AND CONCLUSIONS

Phosphorus mass balance models applied to western reservoirs have been shown to produce relatively large standard errors and low correlations between observed and estimated concentrations. The best results are obtained using the Dillon-Rigler model, the only one studied which has no empirical parameters. This indicates that parameters calibrated from lake data are not applicable to reservoirs. In the case of the Dillon-Kirchner and Vollenweider-1975 models, fitting parameters to reservoir data produced little or no significant improvement, indicating problems may also exist in model formulation.

Even with their deficiencies, mass balance models can be valuable tools in reservoir planning and management when used in the context of their statistical uncertainty. The probability of favorable or unfavorable results can be evaluated for the operation of existing reservoirs or the design of new ones. Results of this study indicate that the Dillon-Rigler model is the best choice for application to existing reservoirs. In the case of planned impoundments, for which phosphorus retention data would obviously be unavailable, the best fit Vollenweider-1976 model can be used with least uncertainty. However, the best fit Jones-Bachmann and Combination models would provide statistically similar results.

While it is clear that these methods are not ideally suited for application to western reservoirs, they provide a basis for judgment of improved techniques. New or modified models should be accepted on the basis of their ability to reduce the uncertainty inherent in the currently available ones.

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CHAPTER 7

ANALYSIS OF POTENTIAL SEDIMENT TRANSPORT IMPACTS BELOW THE WINDY GAP RESERVOIR, COLORADO RIVER

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INTRODUCTION

Background

Development of water resources in the upper Colorado River Basin is a difficult task due to internal and external demands of users. A significant use of water is by transmountain diversion to the agricultural lands and population centers of the Colorado Front Range. In order to meet demands of Front Range water users, increased diversions have become necessary. These diversions will be met in part by construction of a small forebay reservoir and a pumping plant capable of up to 16.3 cms withdrawal (at this time) from the Colorado River. This reservoir, with normal maximum volume of 0.0005 km^3, will be located near Granby, Colorado (Figure 1) and is referred to by the name of a nearby geologic feature, Windy Gap. Water pumped from the Windy Gap Reservoir will be piped back into Lake Granby and then conveyed through the Colorado-Big Thompson Project to Eastern Colorado.

Owners of the reservoir and pumping plant, the Northern Colorado Water Conservancy District, saw a need to study the effects of withdrawing water from the river system. An aquatic ecologist, Dr. Robert Erickson, was hired and subsequently recommended an indepth investigation of the hydrology and sediment transport of the river. The investigation focused on post-pumping effects of potential aggradation in the stream channel below the reservoir and downstream

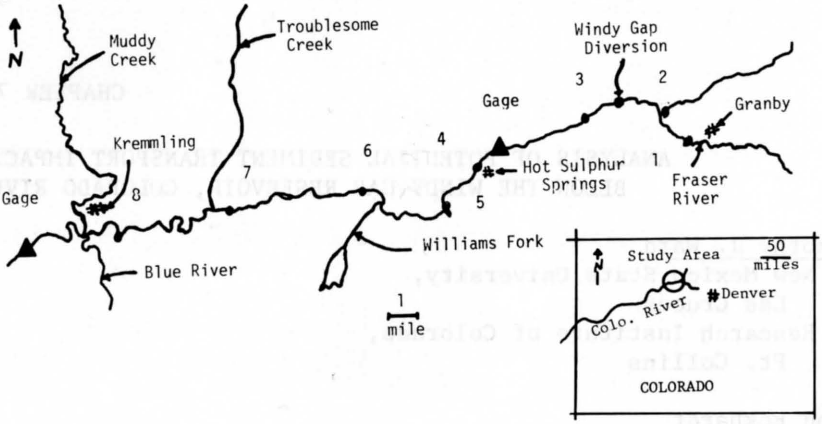

Figure 1. Sketch map of study area showing key locations and sampling sites (one mile equals 1.61 km).

for approximately 48 km (Figure 1). Excessive aggradation created by reductions of the sediment transporting capacity of the river could create situations where habitat conditions of food, protection, and spawning beds would no longer support the current trout population. Excessive aggradation would have a significant impact because this segment of the Colorado River supports several private and public fishing reaches.

What was required were estimates of potential aggradation (or degradation) for several intermediate river reaches using 20 years as a base. Water years (WY) 1958 through 1977 were selected to determine pre- and post-pumping flow conditions. These flow conditions along with collected field data were then used to calculate hydraulic and sediment transport characteristics at selected sites. A mass balance approach for distributing discharges between points was employed. For aggradation computations, another mass balance between end sections of the selected reach was used. Results using this approach indicated that flow conditions were sufficient to prevent excessive aggradation at the selected sites for the assumed conditions. Details of the study are presented by Ward [1].

HYDROLOGY AND CHANNEL MORPHOLOGY

The Colorado River between its headwaters on the Continental Divide and the confluence with the Blue River near

Kremmling is controlled by Lake Granby, Grand Lake, and Shadow Mountain Reservoir, all above Windy Gap (WG) Reservoir. Tributaries above WG Reservoir include the Fraser River (710 km^2), and Willow Creek (347 km^2) which is controlled by Willow Creek Reservoir. Contributing area to the WG Reservoir is 2023 km^2, of which 837 are controlled by Lake Granby. The major controlled tributary is the Fraser River which is estimated to produce about 60% of the inflow.

Between the WG Reservoir and the confluence with the Blue River, the river is influenced by two major tributaries, 15 minor tributaries, and numerous diversions for agricultural and domestic needs. The two major tributaries are the Williams Fork River (598 km^2) which is controlled by the Williams Fork Reservoir and Troublesome Creek (440 km^2) which is not regulated to any extent. The minor tributaries include approximately 388 km^2. The major and minor tributaries account for over 90% of the contributing area between the reservoir site and the Blue River confluence. However, inflows from the minor tributaries are relatively insignificant in comparison to the major one. Over 20 diversions have been identified with the largest water right being 1.8 cms (one cms equals 35.3 cfs) [2].

Although complete, long-term records for existing and abandoned U.S. Geological Survey (USGS) gaging sites are not available, enough record is available for simulation of daily flows at key locations or tributaries. Because of the long-term records, the Hot Sulphur Springs flows were selected as a point of discussion for the entire segment.

The 20-year base from WY 1958 through WY 1977 represents the flow conditions during operation of Lake Granby. Flow statistics for this period are shown in Table I for Hot Sulphur Springs. Even with the upstream controls there was significant variation in the flow.

Table I. Flow Statistics for Hot Sulphur Springs Gage, Water Years 1958 through 1977.

Statistic	Arithmetic Average	Range
Peak, cms	40	81-9.8
Minimum daily, cms	1.5	2.0-1.2
Yield, km^3	0.20	0.43-0.10

Although the flow has been quite variable and effects of upstream regulation appear in the long-term record, significant historic changes in river form were not detectable. Aerial photographs from 1938, 1950, 1967, and 1974 were obtained and analyzed. During the period only two noticeable changes occurred, both the Fraser River and Troublesome Creek straightened naturally or were straightened near their confluences with the mainstem Colorado.

In general, the river flow during the 20-year period was quite variable. Unfortunately, corresponding discharge measurements were not taken on all the major or minor tributaries during that period, necessitating a mass balance approach for distributing river flows from measured and simulated data. Even with the variability, physical conditions of the river bed were such that few significant changes occurred.

SEDIMENT TRANSPORT

Sediment available for transport in this segment of the Colorado River is derived from upstream inflows, tributary inflows, or the channel bed and banks. In order to ascertain the type and magnitude of sediments in transport and available for transport, an intensive sampling and measurement program was conducted during the spring, summer and fall of 1980. Eight sites were selected for sampling of suspended material and bed load. Measurements for five different flows at each site were collected for a total of 40 site-samples. Samples on the rising and falling limbs of the 1980 runoff hydrograph were fortuitously chosen (Figure 2). In addition

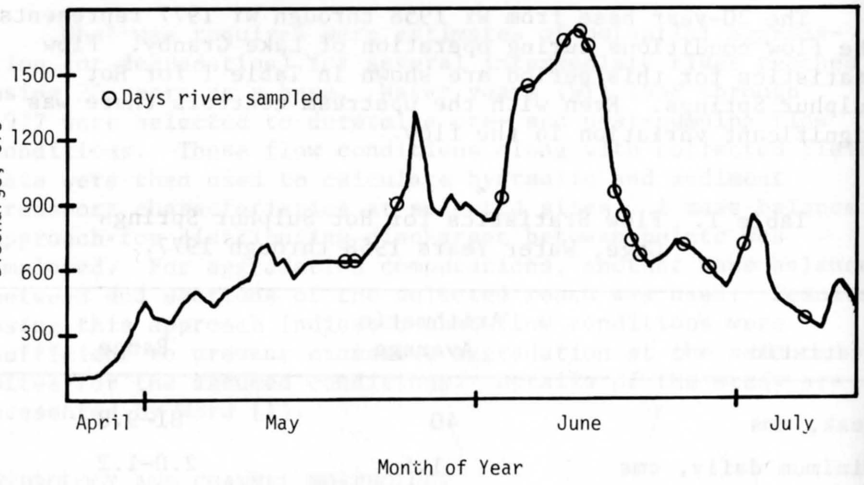

Figure 2. Daily discharges at Hot Sulphur Spring Gage for WY 1980 showing days river sampled at various sites. (One cfs equals 0.028 cms).

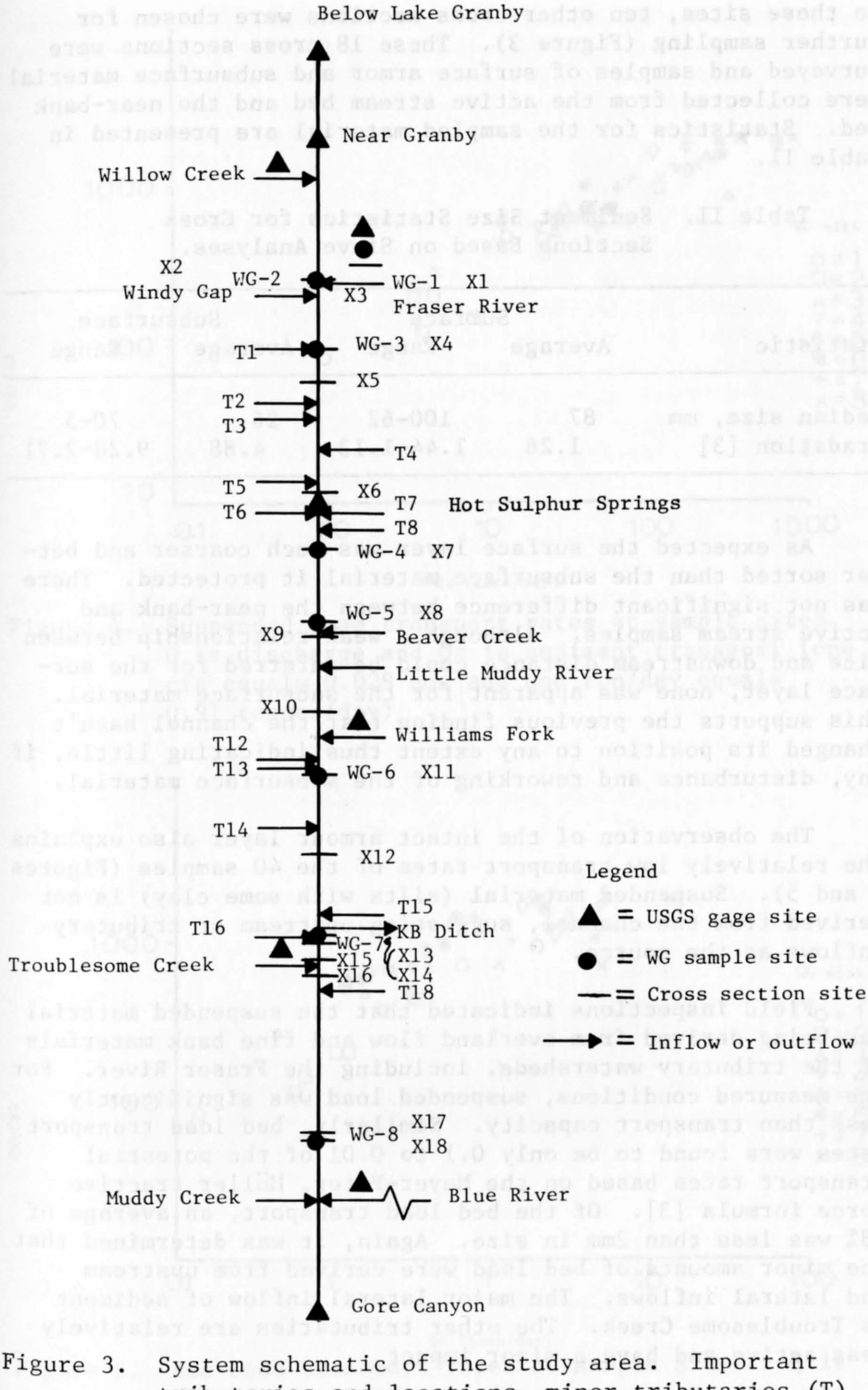

Figure 3. System schematic of the study area. Important tributaries and locations, minor tributaries (T), sampling sites (WG-1) and cross sections (X) are shown. Flow is from top of page. Drawn to scale.

to these sites, ten other cross sections were chosen for further sampling (Figure 3). These 18 cross sections were surveyed and samples of surface armor and subsurface material were collected from the active stream bed and the near-bank bed. Statistics for the sampled material are presented in Table II.

Table II. Sediment Size Statistics for Cross Sections Based on Sieve Analyses.

Statistic	Surface Average	Surface Range	Subsurface Average	Subsurface Range
Median size, mm	87	100-62	26	70-5
Gradation [3]	1.26	1.44-1.13	4.88	9.28-2.71

As expected the surface layer was much coarser and better sorted than the subsurface material it protected. There was not significant difference between the near-bank and active stream samples. Although a weak relationship between size and downstream distance could be inferred for the surface layer, none was apparent for the subsurface material. This supports the previous finding that the channel hasn't changed its position to any extent thus indicating little, if any, disturbance and reworking of the subsurface material.

The observation of the intact armour layer also explains the relatively low transport rates of the 40 samples (Figures 4 and 5). Suspended material (silts with some clay) is not derived from the channel, suggesting upstream or tributary inflows as the source.

Field inspections indicated that the suspended material was being derived from overland flow and fine bank materials of the tributary watersheds, including the Fraser River. For the measured conditions, suspended load was significantly less than transport capacity. Similarly, bed load transport rates were found to be only 0.1 to 0.01 of the potential transport rates based on the Meyer-Peter, Müller tractive force formula [3]. Of the bed load transport, an average of 78% was less than 2mm in size. Again, it was determined that the minor amounts of bed load were derived from upstream and lateral inflows. The major lateral inflow of sediment is Troublesome Creek. The other tributaries are relatively less active and have a minor impact.

Preliminary transport computations indicated two things. First, suspended load transport capacity of post-pumping

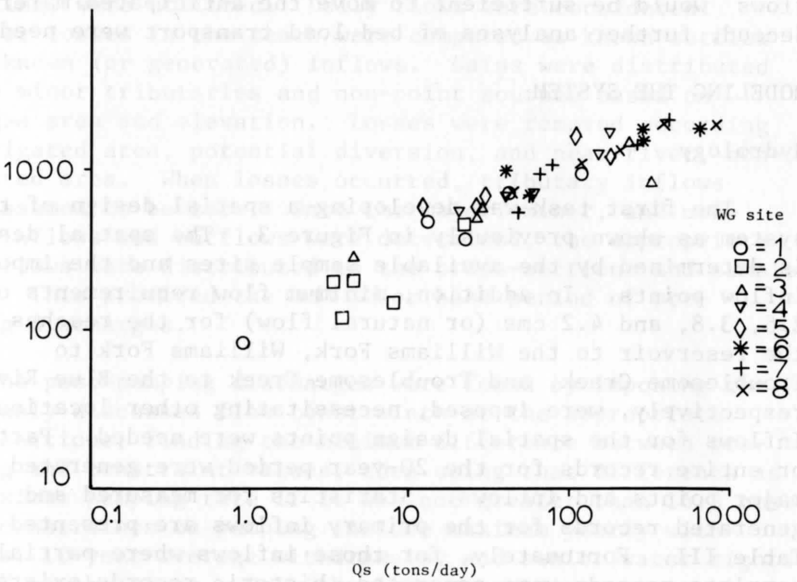

Figure 4. Suspended load transport rates at sample sites. Q is discharge and Qs is sediment transport (one cfs equals 0.028 cms and one ton/day equals 0.91 tonnes/day)

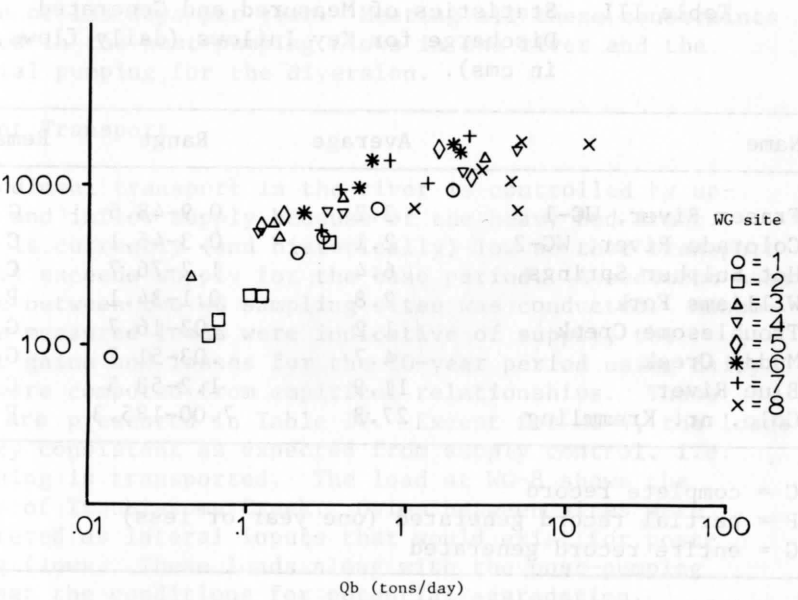

Figure 5. Bed Load transport rates at sample sites. Q is discharge and Qb is sediment transport (one cfs equals 0.028 cms and one ton/day equals 0.91 tonnes/day).

flows would be sufficient to move the anticipated materials. Second, further analyses of bed load transport were needed.

MODELING THE SYSTEM

Hydrology

The first task was developing a spatial design of the system as shown previously in Figure 3. The spatial design is determined by the available sample sites and the important inflow points. In addition, minimum flow requirements of 2.5, 3.8, and 4.2 cms (or natural flow) for the reaches from the reservoir to the Williams Fork, Williams Fork to Troublesome Creek, and Troublesome Creek to the Blue River, respectively, were imposed, necessitating other locations. Inflows for the spatial design points were needed. Partial or entire records for the 20-year period were generated for major points and inflows. Statistics for measured and generated records for the primary inflows are presented in Table III. Fortunately, for those inflows where partial or complete records were generated, historic records existed which were related to other long-term measurements in the same watershed or at nearby stations. Flows were then generated from information at the nearby stations.

Table III. Statistics of Measured and Generated Discharge for Key Inflows (daily flows in cms).

Name	Average	Range	Remarks
Fraser River, WG-1	3.2	0.9-48.5	G
Colorado River, WG-2	2.1	0.3-46.1	C
Hot Sulphur Springs	6.4	1.2-76.7	C
Williams Fork	2.8	0.1-34.1	P
Troublesome Creek	1.2	.03-16.7	G
Muddy Creek	4.7	.03-51.7	G
Blue River	11.9	1.2-53.5	C
Colo. nr. Kremmling	27.3	7.00-185.3	P

C = complete record
P = partial record generated (one year or less)
G = entire record generated

Flow distribution was conducted on a mass balance approach. Two long reaches, the reservoir to Hot Sulphur Springs and Hot Sulphur Springs to the Gore Canyon gage near

Kremmling, were utilized. Daily flows were considered. Gains or losses in the reach were computed as known outflow minus known (or generated) inflows. Gains were distributed to the minor tributaries and non-point sources based on draingae area and elevation. Losses were removed according to irrigated area, potential diversion, and near-river, non-irrigated area. When losses occurred, tributary inflows were assumed to be zero. Once the major, minor, and non-point inflows and outflows were determined, the appropriate river flows were distributed to the cross-sections and sample sites. This provided the initial or base period of pre-pumping discharges.

The post-pumping discharges were found by imposing the previously discussed flow constraints at the appropriate cross-sections, finding the minimum difference between pre-pumping and constraint values, then using that difference as the maximum pumping rate if it was not greater than 16.8 cms. Other constraints on pumping include maximum yearly withdrawals, 10-year average withdrawals, and senior water rights "calls" on the river. Only the last constraint was considered in addition to the 16.8 cms pumping right and minimum flows because the other two would permit increased flow and higher sediment transport, a beneficial result. Generally "calls" run for about eight months and pumping would be permitted in the period between about April and August, an average of 135 days per year. Meeting all these constraints resulted in the post-pumping flows in the river and the potential pumping for the diversion.

Sediment Transport

Sediment transport in the river is controlled by upstream and inflow supply because of the heavy bed armor. Supply is currently (and historically) low so that transport capacity exceeds supply for the base period. A sediment mass balance between the WG sampling sites was conducted. Assuming the measured loads were indicative of supply, the current gains and losses for the 20-year period using daily flows were computed from empirical relationships. These loads are presented in Table IV. Except for WG-4, the loads are very consistent as expected from supply control, i.e. everything is transported. The load at WG-8 shows the effects of Troublesome Creek. Gains between sites were interpreted as lateral inputs that would exist for post-pumping flows. These loads along with the post-pumping flows set the conditions for potential aggradation.

Potential sediment transport was computed using a form of the Meyer-Peter, Müller equation modified to account for

Table IV. Computed Bed Load Passing WG Sampling Sites for the Base Period.

Site	Load Passing Tonnes	Gain or Loss in Reach Tonnes
WG-1 + WG-2	784	--
WG-3	901	117
WG-4	1992	1091
WG-5	916	-1076
WG-6	913	-3
WG-7	1214	301
WG-8	4935	3721

shear stress against the grain created by the flow velocity. This theoretical transport was compared with measured values to confirm that grain movement did occur for the various sediment sizes collected. Transport rates for individual grain-size fractions were computed and comparison with measured data indicated that potential transport was 10 to 100 times greater. This is reasonable as steep channel (average bed slope was 0.006 in the mainstem) experiments indicate that bed loads can easily be greater than 500 ppm, a level never approached in any sample. All of these findings and computations led to the following results.

RESULTS AND CONCLUSIONS

Pumping Rates

Application of three of the five constraints indicated that pumping could occur 2232 days out of a possible 2692 over the 20-year period. The average rate would be 6.2 cms or .06 km^3 per year. Fifty-four percent of the time pump rates of 4.2 cms or less would be permitted. Seventeen percent of the time the maximum rate of 16.8 cms could be attained.

River Flows

The effect on the river flows varied from year to year. As comparisons, a "wet", high runoff year (WY 1962) and a "dry", low runoff year (WY 1977) are shown in Figures 6 and 7. Note the scale differences. The effects on transport capacity at the WG sites for the two years are presented in Table V.

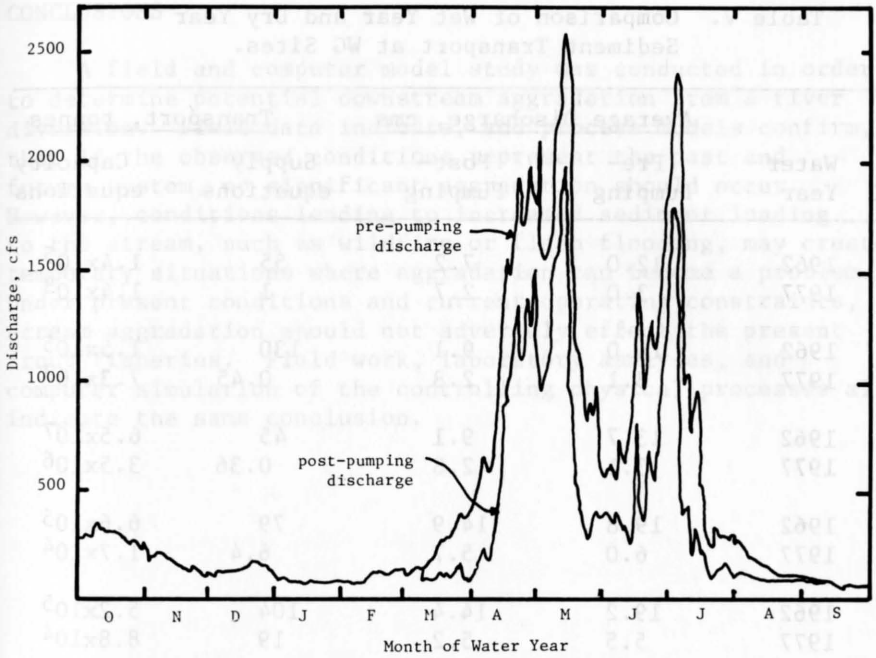

Figure 6. Pre- and post-pumping flows at Hot Sulphur Springs Gage WY 1962 ("Wet" Year) (one cfs equals 0.028 cms).

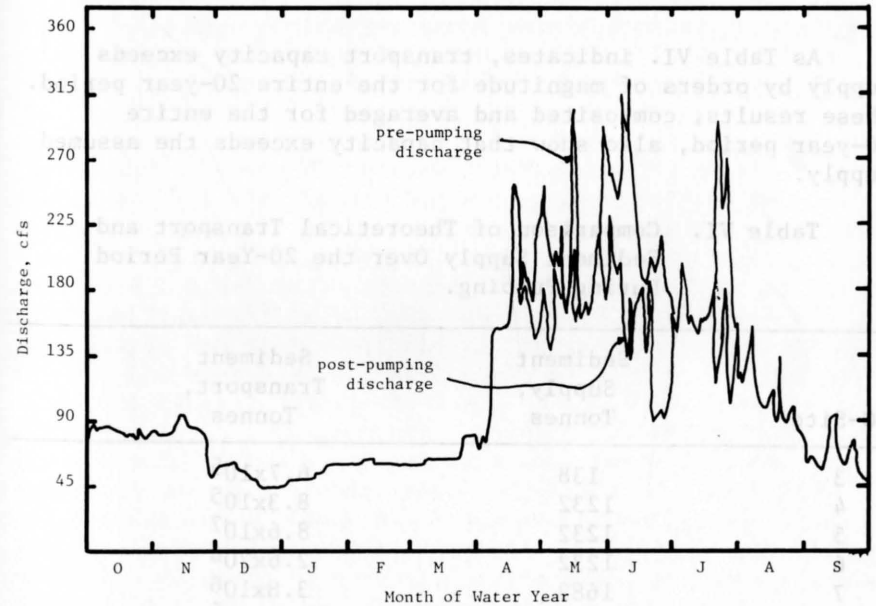

Figure 7. Pre- and post-pumping flows at Hot Sulphur Springs Gage WY 1977 ("Dry" Year) (one cfs equals 0.028 cms)

Table V. Comparison of Wet Year and Dry Year Sediment Transport at WG Sites.

		Average Discharge, cms		Transport, tonnes	
WG	Water Year	Pre-Pumping	Post-Pumping	Supply equations	Capacity equations
3	1962	12.0	7.2	55	$1.4x10^5$
	1977	3.0	2.7	1	$1.4x10^3$
4	1962	14.0	9.1	130	$1.5x10^5$
	1977	3.1	2.8	0.45	$7.3x10^3$
5	1962	13.7	9.1	45	$6.5x10^7$
	1977	3.1	2.8	0.36	$3.5x10^6$
6	1962	19.8	14.9	79	$6.6x10^5$
	1977	6.0	5.7	6.4	$1.7x10^4$
7	1962	19.2	14.4	104	$5.2x10^5$
	1977	5.5	5.2	19	$8.8x10^4$
8	1962	21.0	16.1	458	$1.2x10^5$
	1977	6.0	5.7	43	$1.2x10^4$

As Table VI. indicates, transport capacity exceeds supply by orders of magnitude for the entire 20-year period. These results, composited and averaged for the entire 20-year period, also show that capacity exceeds the assumed supply.

Table VI. Comparison of Theoretical Transport and Sediment Supply Over the 20-Year Period During Pumping.

WG-Site	Sediment Supply, Tonnes	Sediment Transport, Tonnes
3	138	$6.7x10^5$
4	1232	$8.3x10^5$
5	1232	$8.6x10^7$
6	1232	$2.6x20^6$
7	1683	$3.8x10^6$
8	5388	$7.5.10^5$

CONCLUSIONS

A field and computer model study was conducted in order to determine potential downstream aggradation from a river diversion. Field data indicate, and process models confirm, that if the observed conditions represent the past and future system, no significant aggradation should occur. However, conditions leading to increased sediment loading to the stream, such as wildfire or flash flooding, may create temporary situations where aggradation can become a problem. Under present conditions and current operating constraints, stream aggradation should not adversely effect the present trout fisheries. Field work, laboratory analyses, and computer simulation of the controlling physical processes all indicate the same conclusion.

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CHAPTER 8

HISTORICAL PATTERNS OF PHYTOPLANKTON PRODUCTIVITY IN LAKE MEAD

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INTRODUCTION

Lake Mead was impounded in 1935 by the construction of Hoover Dam. The Colorado River was unregulated prior to then and therefore was subjected to extreme variations in flows and suspended sediment loads. Hoover Dam stabilized flows and reduced suspended sediment loads downstream [1], but Lake Mead still received silt-laden inflows from the upper Colorado River Basin. The Colorado River contributed 97% of the suspended sediment inputs to Lake Mead, and up to 140 x 10^6 metric tons (t) entered the reservoir in years of high runoff [2]. Most of the sediments were deposited in the river channel and formed an extensive delta in upper Lake Mead [3,4]. However, sediments were also transported into the Virgin Basin and Overton Arm by the overflow that occurred during spring runoff [5]. The limnology of Lake Mead is thought to have been strongly influenced by this turbid overflow until Glen Canyon Dam was constructed 450 km upstream in 1963.

The construction of Glen Canyon Dam and formation of Lake Powell drastically altered the characteristics of the Colorado River inflow to Lake Mead [2]. The operation of Glen Canyon Dam stabilized flows, reduced river temperatures and cut the suspended sediment loads by 70-80% [2]. Nitrate loads decreased initially during 1963 and 1964, then increased through 1970, but have since decreased again to a lower steady state [6]. Phosphorus loads were decreased due to reductions in suspended sediment inputs [2]. Lake Powell now retains 70% of the dissolved phosphorus [1] and 96% of the total phosphorus [7] inputs that once flowed into Lake Mead. The Colorado River still provides 85% of the inorganic nitrogen to Lake Mead, but Las Vegas Wash now contributes 60% of the phosphorus inputs [2].

Wastewater discharges from Las Vegas Wash into Las Vegas Bay increased steadily during the post-Lake Powell period. The morphometry and hydrodynamics of Lake Mead are such that the Las Vegas Wash inflow is confined to the Lower Basin where historically it has elevated phytoplankton productivity. However, high phosphorus loading and productivity have resulted in decreases in nitrate concentrations, and the Las Vegas Bay and parts of Boulder Basin have become nitrogen limited since 1972 [6]. A unique situation has therefore developed in Lake Mead in that the Upper Basin has become more phosphorus limited and the Lower Basin more nitrogen limited since the formation of Lake Powell. Paulson and Baker [2] theorized that these changes in nutrient loading and limitation must also have been accompanied by decreases in reservoir-wide productivity.

There is some evidence for this hypothesis in apparent improvements in water quality of Las Vegas Bay since 1968 [6]. Chlorophyll-a concentrations in the inner Las Vegas Bay have decreased considerably since the first measurements were made in 1968 [8] and during the period of the Lake Mead Monitoring Program [9-12]. Improvements in water quality of the bay have confounded efforts to establish water quality standards on effluent discharges and are contrary to predictions made in the early 1970s that water quality would continue to degrade with increased phosphorus loading [13]. The decline in the largemouth bass fishery documented by the Nevada Department of Wildlife [14] could also be a symptom of lower productivity in Lake Mead.

In this paper, the hypothesis that algal productivity has declined in Lake Mead as a result of impoundment of Lake Powell is evaluated. The chemical status of six stations in the Upper and Lower Basins of Lake Mead is analyzed and current and past rates of organic carbon and phosphorus sedimentation are calculated. The relationship between algal productivity and accretion of organic carbon in sediment is determined, and this is used to construct a historical record of algal productivity for Lake Mead.

METHODS

Sampling Locations

The productivity and siltation patterns in Lake Mead are extremely heterogeneous due to the irregular reservoir morphometry and variable influence of nutrient loading from Las Vegas Wash and the Colorado River [15]. In order to insure that this heterogeneity was adequately represented in the survey, multiple sediment cores were collected from several locations in the reservoir. The location of drilling sites are shown in Figure 1, and site characteristics are listed in Table I. Station locations were surveyed with an echo-sounder and the final sites were selected to provide a reasonably flat, undisturbed sediment surface. The stations were purposely placed outside the old river channel to avoid possible sediment disturbances from the Colorado River density current. Station 1 was a shallow-water site in a small embayment of the inner Las Vegas Bay, near the point of the sewage inflow from Las Vegas Wash. Stations 2 and 3 were placed in the Lower Basin; one of these in Boulder Basin (Station 3). Two stations were also placed off the old river channel in the Upper Basin: the Virgin Basin (4) and Bonelli Bay (5) stations. The sixth station was located in the Overton Arm, near Echo Bay.

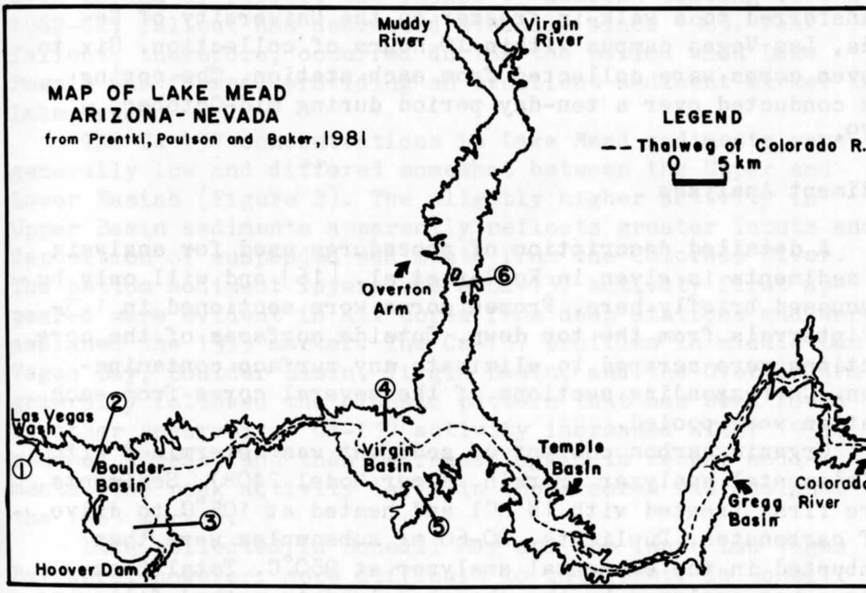

Figure 1. Map of Lake Mead Sediment Coring Stations.

Table I. Physical Characteristics at Sediment Coring Stations in Lake Mead.

Station	Water Depth (meters)	Number of cores	Date of submersion (month-year)	Relict material
1*	14	8	6-38	gravel
2	60	11	7-35	gravel
3	90	6	7-35	gravel
4	80-95	10	7-35	soil
5	102	11	7-35	sand
6**	75	8	7-35	sand

* This station was dry in low water years

** Fine sediment was not deposited above sand until 3-40

Sediment Coring

The sediment coring was conducted by an oceanographic drilling company (Ocean/Seismic/Survey Inc., Norwood, NJ). A hydraulically-operated vibra-corer was used to obtain undisturbed sediment cores of 8.6 cm effective diameter. Coring rates were monitored with a penetration recorder. Coring was terminated when coring rates indicated that contact had been made with the old reservoir floor. The corer was retrieved and the core was immediately inspected through the Lexan liner for signs of marbling or other disturbance. Undisturbed cores were capped and stored upright. They were transferred to a walk-in freezer on the University of Nevada, Las Vegas campus within 10 hours of collection. Six to eleven cores were collected from each station. The coring was conducted over a ten-day period during mid-October, 1979.

Sediment Analyses

A detailed description of procedures used for analysis of sediments is given in Kellar et al. [16] and will only be discussed briefly here. Frozen cores were sectioned in 1.3-cm intervals from the top down. Outside surfaces of the core sections were scraped to eliminate any surface contamination. Corresponding sections of the several cores from each station were pooled.

Organic carbon content of sediment was determined with an elemental analyzer (Perkin Elmer Model 240B). Sediments were first treated with 1N HCl and heated at 105°C to drive off carbonates. Duplicate, 20-60 mg subsamples were then combusted in the elemental analyzer at 950°C. Total phosphorus was analyzed by the phosphomolybdate method following ignition of 0.5 g samples at 550°C and subsequent extraction of phosphorus from the residue into 1N H_2SO_4.

Sediment bulk density and calcium carbonate content measurements were necessary in order to calculate the organic carbon sedimentation rates but are not reported here. These data and description of their analytical methodology are described by Prentki et al. [17].

The Cesium-137 counting of 500-1000 g samples was performed by Controls for Environmental Pollution Inc. (CEP), a commercial laboratory in Santa Fe, NM. The required sample size necessitated pooling two to three adjoining 1.3-cm sediment sections. A few samples were also counted by the U.S. Environmental Protection Agency, Office of Radiation Programs, Las Vegas, NV, and by the Southern Plains Watershed and Water Quality Laboratory, Durant, OK, for quality assurance purposes.

RESULTS AND DISCUSSION

Sediment Core Dating

Cesium-137 radioactivity from atmospheric bomb fallout has been widely used to date reservoir sediments [18]. Cesium-137 is strongly adsorbed by fine soil particles and, if eroded from the watershed, will be deposited in reservoir sediments. The first occurrence of Cs-137 activity in the bottom of a sediment profile indicates that the layer was deposited after the first testing in 1954. The most intensive period of fallout was caused by Russian testing during 1962-64; fallout has decreased steadily since 1963. Peak fallout, therefore, occurred during the period when Lake Powell was formed, providing an excellent sediment marker in Lake Mead.

The Cs-137 concentrations in Lake Mead sediments were generally low and differed somewhat between the Upper and Lower Basins (Figure 2). The slightly higher activity in Upper Basin sediments apparently reflects greater inputs and deposition of suspended sediments from the Colorado River. The bottom sediment layers where Cs-137 activity first appeared were evident in all cores from deep stations and were assigned the 1955 marker. The Cs-137 profiles in middle Las Vegas Bay, Boulder Basin, Virgin Basin, and the Overton Arm generally followed the classic pattern that has been found in other reservoirs. Cs-137 activity increased after 1955, reached a peak, and then decreased again in recent sediments. The peak activity layer in these cores was assigned the 1963 marker.

Data collected in Bonelli Bay and the inner Las Vegas Bay were, however, more difficult to interpret. In Bonelli Bay, peak Cs-137 activity occurred at 17-19 cm sediment depth, far below that found at the other Upper Basin Stations. In Virgin Basin, the peak activity occurred at 8-9 cm, and in the Overton Arm, it occurred at 3-4 cm sediment depth. In order to resolve the obvious discrepancies with other Upper Basin cores, we assigned the 1963 marker to the secondary Cs-137 maximum that occurred 3-4 cm from the sediment surface in Bonelli Bay. This is consistent with changes in other chemical parameters of this layer [17] and reasonable in terms of known reductions in suspended sediment loading and siltation in the Upper Basin after 1963.

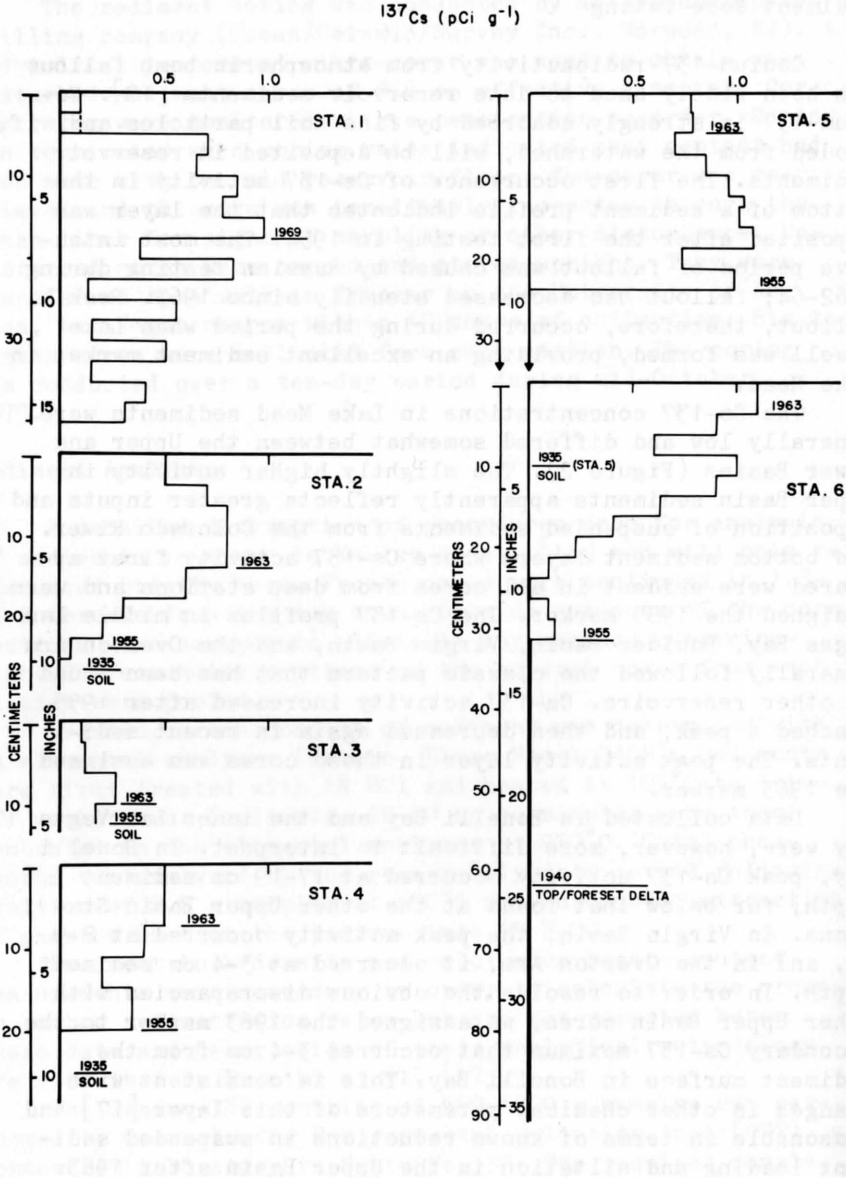

Figure 2. Cesium-137 Profiles of Lake Mead Sediments and Dates of Various Sediment Layers.

The Cs-137 profile in the inner Las Vegas Bay was also difficult to interpret because activity was found in gravel layers deep in the core. This station was shallow and in the past has been subject to water level fluctuations and periodic desiccation. This area was dry until 1938, very shallow (1-2 m) during 1947 and from 1951-57, and then dry again from 1963-69. Because of possible reworking of sediment during dry or low water years, we were unable to use the disappearance of Cs-137 activity to indicate the 1955 marker. Moreover, the peak in Cs-137 activity must reflect 1969 rather than 1963, because this area was dry over the period from 1963-69.

Apart from some difficulties in interpreting Cs-137 profiles in Bonelli Bay and the inner Las Vegas Bay, the Cs-137 data provide reliable markers of the 1955 and 1963 sediment layers. It is also possible to establish a third marker, the old reservoir floor of 1935, by obvious discontinuities between pre-reservoir soils and reservoir sediments. Sediments were underlain by gravel in the middle Las Vegas Bay, gravel and soft rock in Boulder Basin, unconsolidated desert soils in Virgin Basin, and sand in Bonelli Bay. A similar discontinuity existed in Overton Arm, but the sediment depth here was also influenced by delta deposits from the Virgin River as the reservoir was filling. Gould [19] reported that in 1935 and 1936 the mouth of the Virgin River was located at Bitter Wash, a few kilometers upstream from our station. He was, therefore, unable to distinguish between sand deposited by the river and that in the pre-reservoir deposits. Clay sediments were deposited once lake levels increased and caused the point of river inflow to recede up the Overton Arm. This occurred in 1940. Layers below that represent siltation from the Virgin River inflows during 1935-40.

Sediment Chemical Structure

Organic carbon in Lake Mead sediments was very low. Values ranged from 0.3% of sediment dry weight in early sediments to 1.7% in recent sediments (Figure 3).

Total phosphorus concentrations of Lake Mead sediments were appreciable and ranged from 300 ppm of dry weight in old reservoir sediments to 1000 ppm in recent sediments (Figure 4). In the inner and middle Las Vegas Bay, phosphorus increased steadily in sediments deposited after 1963, but elsewhere phosphorus concentrations decreased or remained stable. The organic carbon:phosphorus ratios were very low, ranging from 10 to 20. These ratios are tenfold lower than found in plankton and considerably lower than those reported in other lake sediments [17]. The low C/P ratios were caused by the presence of large amounts of biologically unavailable particulate phosphorus which entered Lake Mead from the Colorado River [20].

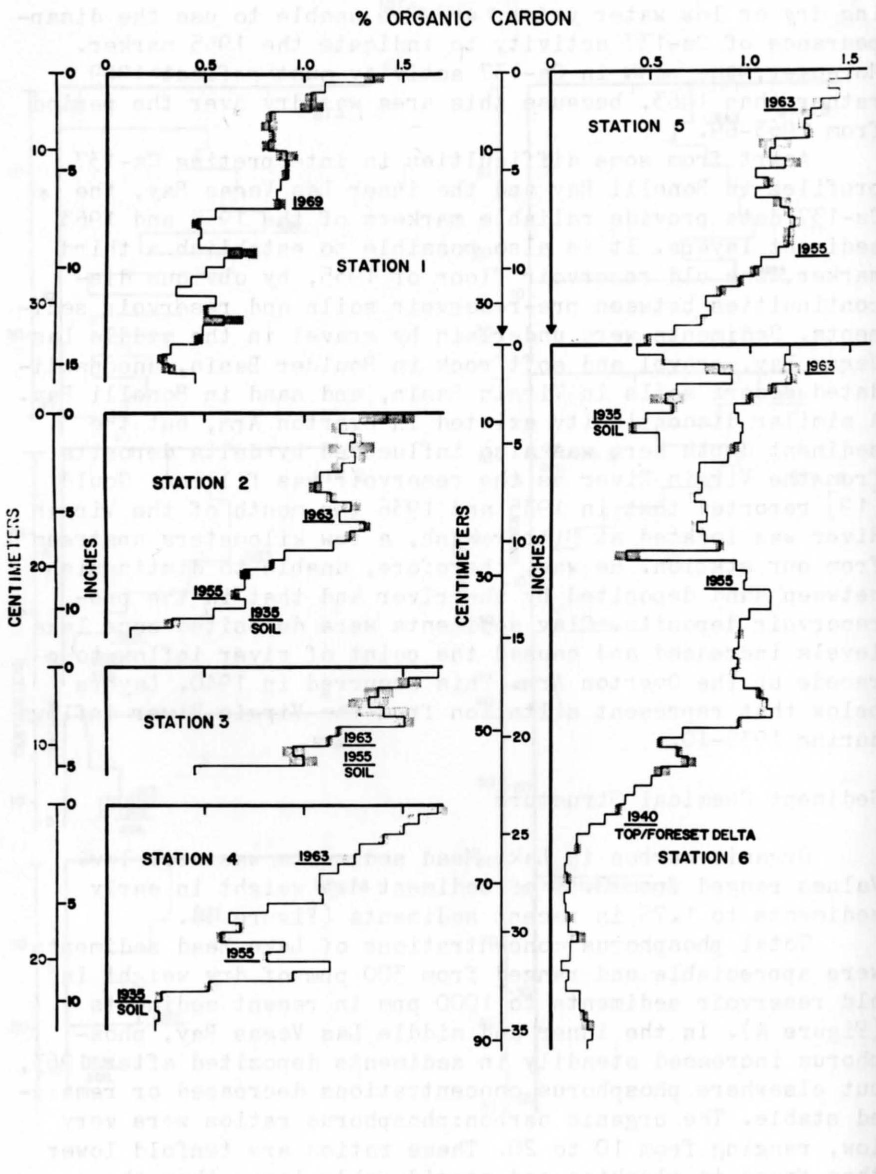

Figure 3. Organic Carbon Content of Lake Mead Sediments (Range in Replicate Analyses Shown by Shading).

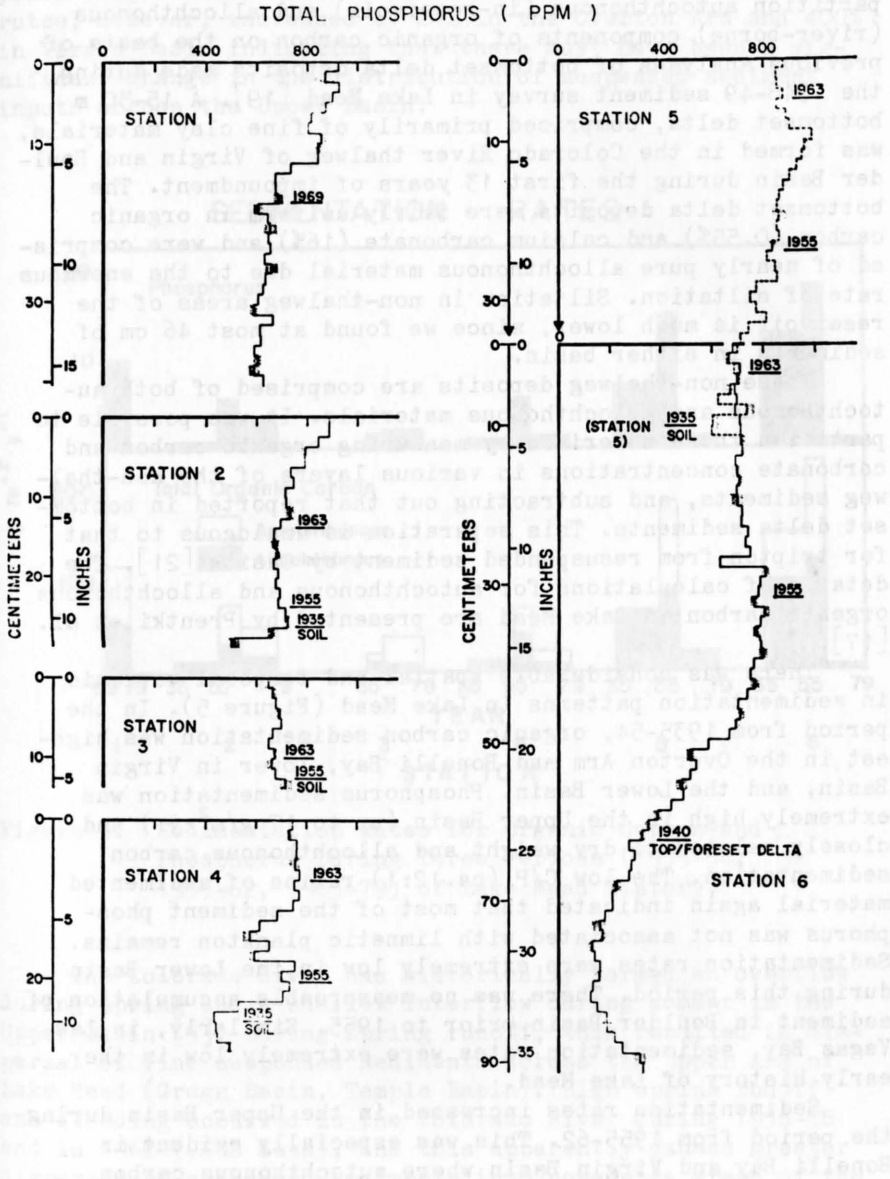

Figure 4. Total Phosphorus Content of Lake Mead Sediments (Range in Replicate Analyses Shown by Shading).

Spatial and Temporal Patterns in Sedimentation

The Cs-137 data and chemical analyses enabled us to estimate annual sedimentation rates for organic carbon and

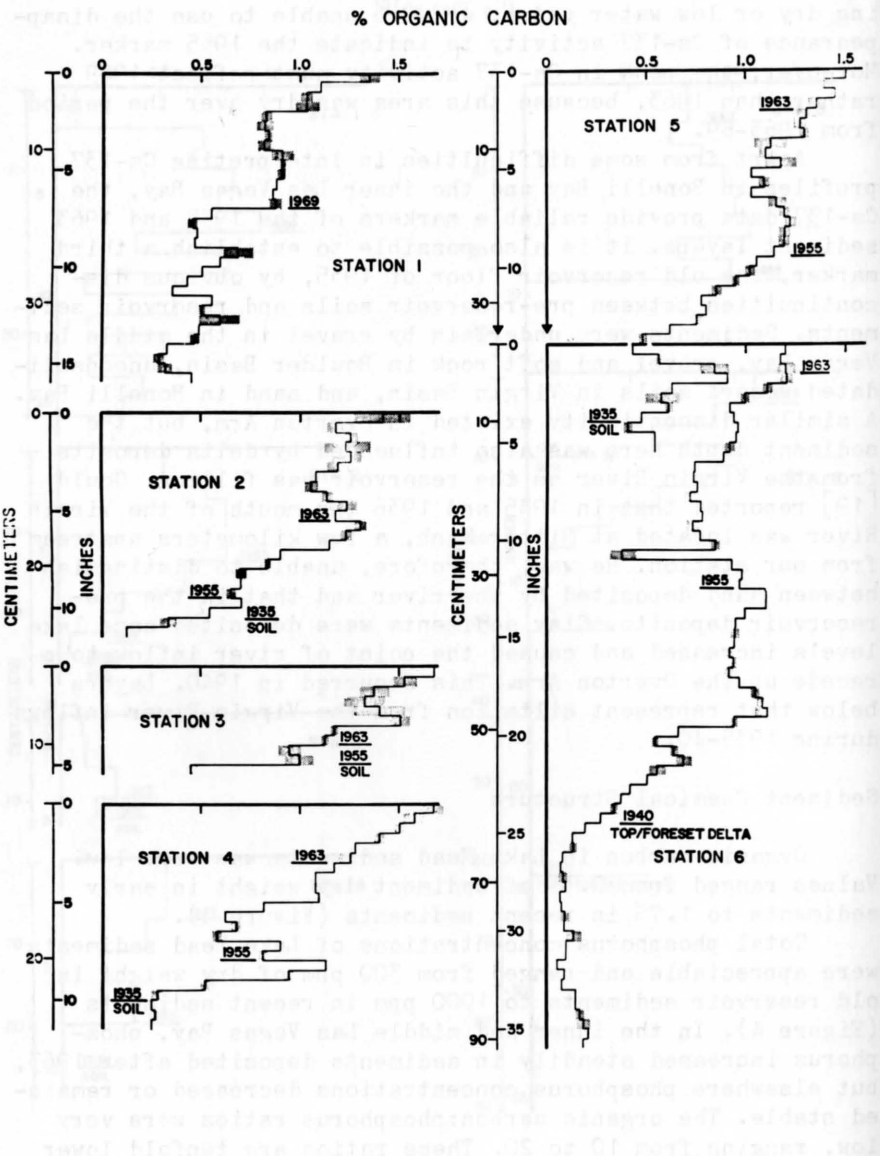

Figure 3. Organic Carbon Content of Lake Mead Sediments (Range in Replicate Analyses Shown by Shading).

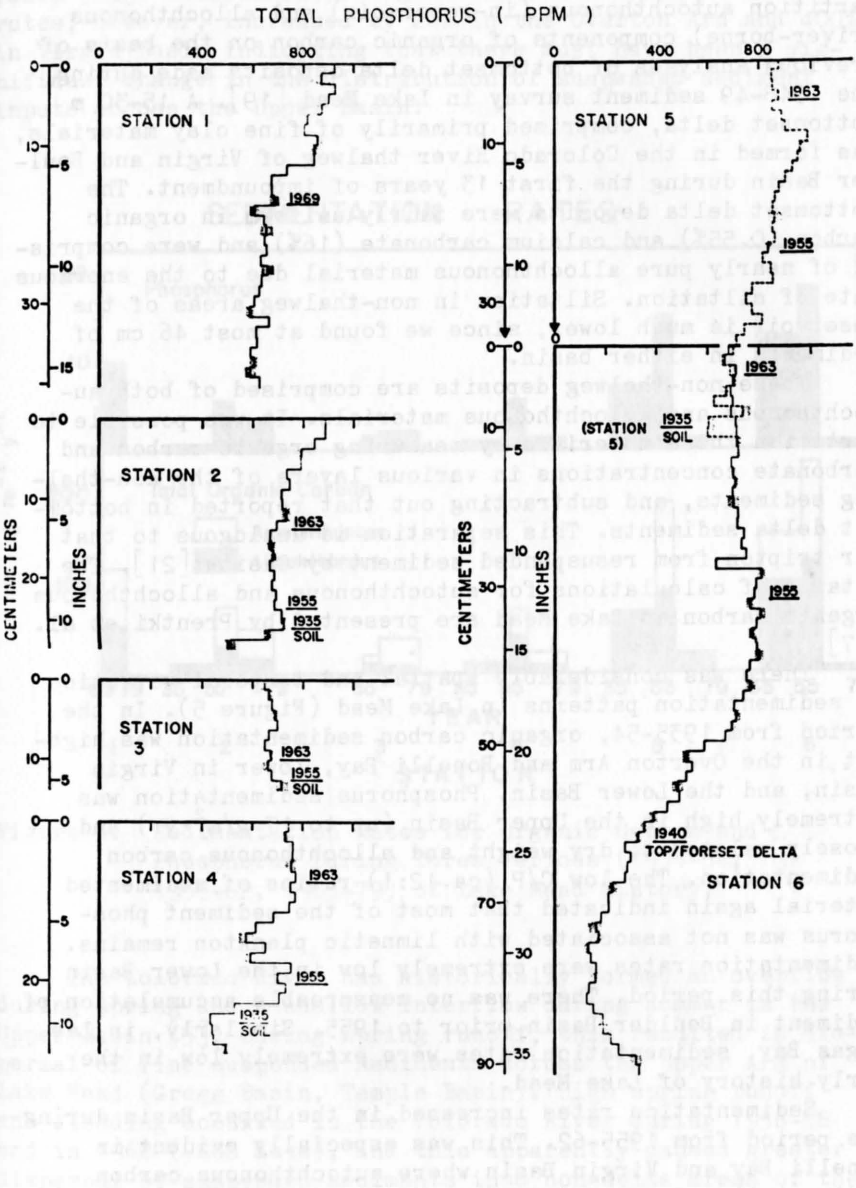

Figure 4. Total Phosphorus Content of Lake Mead Sediments (Range in Replicate Analyses Shown by Shading).

Spatial and Temporal Patterns in Sedimentation

The Cs-137 data and chemical analyses enabled us to estimate annual sedimentation rates for organic carbon and phosphorus during three periods of reservoir history (1935-54, 1955-62 and 1963-79). In addition, it was possible to partition autochthonous (in-reservoir) and allochthonous (river-borne) components of organic carbon on the basis of previous analyses of bottomset delta deposits made during the 1948-49 sediment survey in Lake Mead [19]. A 15-30 m bottomset delta, comprised primarily of fine clay materials, was formed in the Colorado River thalweg of Virgin and Boulder Basin during the first 13 years of impoundment. The bottomset delta deposits were fairly uniform in organic carbon (0.55%) and calcium carbonate (16%) and were comprised of nearly pure allochthonous material due to the enormous rate of siltation. Siltation in non-thalweg areas of the reservoir is much lower, since we found at most 46 cm of sediments in either basin.

These non-thalweg deposits are comprised of both autochthonous and allochthonous materials. It was possible to partition these materials by measuring organic carbon and carbonate concentrations in various layers of the non-thalweg sediments, and subtracting out that reported in bottomset delta sediments. This separation is analogous to that for tripton from resuspended sediment by Gasith [21]. The details of calculations for autochthonous and allochthonous organic carbon in Lake Mead are presented by Prentki et al. [17].

There was considerable spatial and temporal variation in sedimentation patterns in Lake Mead (Figure 5). In the period from 1935-54, organic carbon sedimentation was highest in the Overton Arm and Bonelli Bay, lower in Virgin Basin, and the Lower Basin. Phosphorus sedimentation was extremely high in the Upper Basin (up to 17 $g/m^2 \cdot yr$) and closely related to dry weight and allochthonous carbon sedimentation. The low C/P (ca.12:1) ratios of sedimented material again indicated that most of the sediment phosphorus was not associated with limnetic plankton remains. Sedimentation rates were extremely low in the Lower Basin during this period. There was no measureable accumulation of sediment in Boulder Basin prior to 1955. Similarly, in Las Vegas Bay, sedimentation rates were extremely low in the early history of Lake Mead.

Sedimentation rates increased in the Upper Basin during the period from 1955-62. This was especially evident in Bonelli Bay and Virgin Basin where autochthonous carbon sedimentation increased twofold over the preceding period. Phosphorus sedimentation also increased in the Upper Basin but not as drastically as what was observed for carbon. It is somewhat surprising that these sedimentation rates increased during this period because average suspended sediment loading decreased by 34%. The suspended load in the Colorado River averaged 110 x 10^6 t/yr prior to 1955 but then decreased to 73 x 10^6 t/yr during the 1955-62 water years [22]. Allochthonous organic carbon sedimentation rates, however, increased by 20% in the Overton Arm and 400% in Virgin Basin indicating that there must have been a significant change in the distribution of suspended sediment inputs across the Upper Basin.

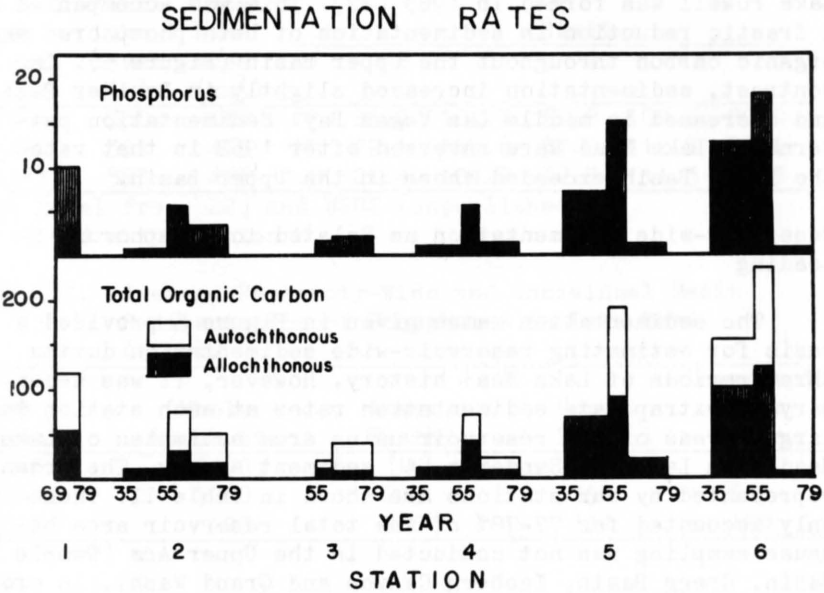

Figure 5. Sedimentation Rates for Organic Carbon and Phosphorus During Three Periods (1935-54, 1955-62, 1963-79) of Lake Mead History.

The Colorado River has historically formed an overflow during spring and a shallow interflow during summer in the Upper Basin [5]. During spring runoff, this resulted in dispersal of fine suspended sediments across the Upper Arm of Lake Mead (Gregg Basin, Temple Basin). High spring runoff and flooding occurred in the Colorado River during 1956-58 and in 1962 (USGS data), and this apparently caused greater dispersal of suspended sediments into non-delta areas of the Virgin Basin, Bonelli Bay and the Overton Arm. The magnitude of spring runoff and seasonal frequency of flooding appear to be more important factors than is average, annual suspended sediment loading in determining sedimentation in non-delta areas of the reservoir. However, even during years of extreme spring runoff, it does not appear that much Colorado River suspended sediment is transported into the Lower

Basin. There was only a slight increase in sedimentation rates of allochthonous organic carbon in Boulder Basin during the period 1955-62 (Figure 5). There was a greater increase in sedimentation in the middle Las Vegas Bay, but this was probably due to increased discharge of sewage effluents into the Lower Basin.

Suspended sediment loading in the Colorado River decreased to an average of 16 x 10^6 t/yr in the period after Lake Powell was formed in 1963 [22]. This was accompanied by a drastic reduction in sedimentation of both phosphorus and organic carbon throughout the Upper Basin (Figure 5). In contrast, sedimentation increased slightly in Boulder Basin and decreased in middle Las Vegas Bay. Sedimentation patterns in Lake Mead were reversed after 1962 in that rates in the Lower Basin exceeded those in the Upper Basin.

Reservoir-wide Sedimentation as Related to Phosphorus Loading

The sedimentation rates given in Figure 5 provided a basis for estimating reservoir-wide sedimentation during three periods of Lake Mead history. However, it was necessary to extrapolate sedimentation rates at each station to larger areas of the reservoir using area estimates of Lake Mead from Lara and Sander's [4] sediment survey. The areas represented by our stations are shown in Table II. These only accounted for 77-78% of the total reservoir area because sampling was not conducted in the Upper Arm (Temple Basin, Gregg Basin, Iceberg Canyon and Grand Wash). In order to obtain an estimate of reservoir-wide sedimentation, we used data from station 5 to characterize the Upper Arm of Lake Mead.

The formation of Lake Powell markedly reduced phosphorus sedimentation in the Upper Basin of Lake Mead. Phosphorus sedimentation in the Upper Basin was extremely high during the early history of Lake Mead but decreased by 93.5% after formation of Lake Powell (Table III). Phosphorus sedimentation in the Lower Basin decreased by only 2% in the post-Lake Powell period. Reservoir-wide phosphorus sedimentation, however, decreased from an average of 5200 t/yr during 1955-62 to 623 t/yr after 1962.

There are no long-term loading data for phosphorus, but it must have been high, particularly during 1955-62, to account for the high rates of phosphorus sedimentation during the pre-Lake Powell years. Phosphorus loading was probably on the order of that recently measured for Lake Powell by Gloss et al. [7]. They estimated that the Colorado River currently provides 5224 t/yr of total phosphorus to Lake Powell. However, only 229 t/yr of phosphorus is currently discharged from Glen Canyon Dam [7], and about the same amount, 198 t/yr enters Lake Mead from the Colorado River [23]. These numbers represent a 96% reduction in total phosphorus loading into Lake Mead which accounts for the abrupt decrease in phosphorus sedimentation in the Upper Basin.

Table II. Reservoir Mean Surface Areas (km^2) Characterized by Sediment Coring Stations.

Time Interval	Mean* Lake Level (m)	Total Lake Area (km^2)	Station 1	2	3	4	5	6
≤ 1954	350	447	**	21.7	101.7	35.8	108.7	80.0
1955-62	352	465	**	22.1	103.6	37.0	112.5	85.1
≥ 1963	353	475	0.8	21.4	104.2	37.7	114.3	87.3

*Lake level from [22] and USGS (unpublished).
**Combined with station 2

Table III. Average Reservoir-Wide and Individual Basin Sedimentation of Phosphorus in Lake Mead (t/yr).

Time Interval	Whole Reservoir	Lower Basin	Upper Basin	Lower and Upper Basin
≤ 1954	2470	15	1780	1795
1955-62	5200	273	3390	3663
≥ 1963	623	268	220	488

Sewage effluent discharges and nutrient loading from Las Vegas Wash, however, rose steadily in the post-Lake Powell period. Las Vegas Wash now contributes 60% of the phosphorus input to Lake Mead [23]. The morphometry and hydrodynamics of Lake Mead [15] are such that the phosphorus-rich Las Vegas Wash inflow is confined to the Lower Basin. Phosphorus sedimentation in the Lower Basin has been maintained, therefore, at levels equal to that in the 1955-62 period (Table III).

The historical patterns of phosphorus sedimentation in each basin of Lake Mead generally agree with historical changes in loading. However, there is a considerable difference in sedimentation estimated from nutrient budgets (apparent sedimentation) [23] and absolute sedimentation measured in this study.

Phosphorus loading to Lake Mead was 198 t/yr from the Colorado River and 263 t/yr from Las Vegas Wash in 1977-78 [23]. Total phosphorus loading to Lake Mead was about 460 t/yr because the Virgin and Muddy Rivers contribute minimal phosphorus to the reservoir [24]. Phosphorus loss from Hoover Dam was 123 t/yr in 1977-78 [23]. The fish harvest also resulted in an annual loss of 25 t of phosphorus from the reservoir [25]. The combined phosphorus losses from Lake Mead would therefore be 148 t/yr. Apparent phosphorus sedimentation would be 312 t/yr. Absolute phosphorus sedimentation, as measured in this study, was 268 t/yr in the Lower Basin, 220 t/yr in the Upper Basin and 623 t/yr in the whole reservoir during the post-Lake Powell period (Table III). Absolute sedimentation thus exceeded 1977-78 apparent sedimentation by 311 t/yr. It is unknown whether loading for 1977-78 reflects average annual loading in recent years. However, the discrepancy between the two retention numbers is most likely caused by a higher nutrient output from Lake Powell during the first years of impoundment than is now occurring [2,20].

Organic Carbon Sedimentation and Phytoplankton Productivity

The historical changes in nutrient loading to Lake Mead have also been accompanied by marked changes in organic carbon sedimentation and, as will be shown, phytoplankton productivity. Reservoir-wide autochthonous carbon sedimentation was low prior to 1955 but increased sharply during the period from 1955-62, followed by an abrupt decrease in the post-Lake Powell period (Table IV). The same trends were also evident for allochthonous organic carbon sedimentation. Organic carbon sedimentation was consistently higher in the Upper Basin during the pre-Lake Powell period and accounted for over 90% of reservoir-wide organic carbon sedimentation. This pattern was reversed after 1962, and the Lower Basin now contributes over 50% of organic carbon sedimentation in Lake Mead. However, reservoir-wide sedimentation has still been reduced by 76.8% of that which occurred in the 1955-62 period.

Table IV. Reservoir-Wide and Individual Basin Sedimentation of Autochthonous and Allochthonous Organic Carbon in Lake Mead (t C/yr).

Interval	Whole Reservoir	Lower Basin	Upper Basin	Lower and Upper Basin
		Autochthonous		
≤ 1954	7710	48	6150	6198
1955-62	33400	2290	20300	22590
≥ 1963	7720	3830	2450	6280
		Allochthonous		
≤ 1954	18900	85	13800	13885
1955-62	32500	1710	21700	23410
≥ 1963	3300	1200	1320	2520

For the post-Lake Powell period, autochthonous organic carbon sedimentation in various locations of Lake Mead (Figure 6) was closely related to recent phytoplankton productivity measurements made at these locations by Paulson et al. [15]. There was a good correlation (r=0.979, N=6) between annual autochthonous organic carbon sedimentation and annual phytoplankton productivity (1977-78) at the six sediment sampling stations (Figure 6). Linear regression of organic carbon sedimentation against phytoplankton productivity (Equation 1) provided a means of predicting historical productivity in the reservoir.

$$PPR = -7 + 19.7\ (AOC) \qquad (1)$$

where PPR = rate of phytoplankton productivity ($g\ C/m^2 \cdot yr$)

AOC = autochthonous organic carbon sedimentation ($g\ C/m^2 \cdot yr$)

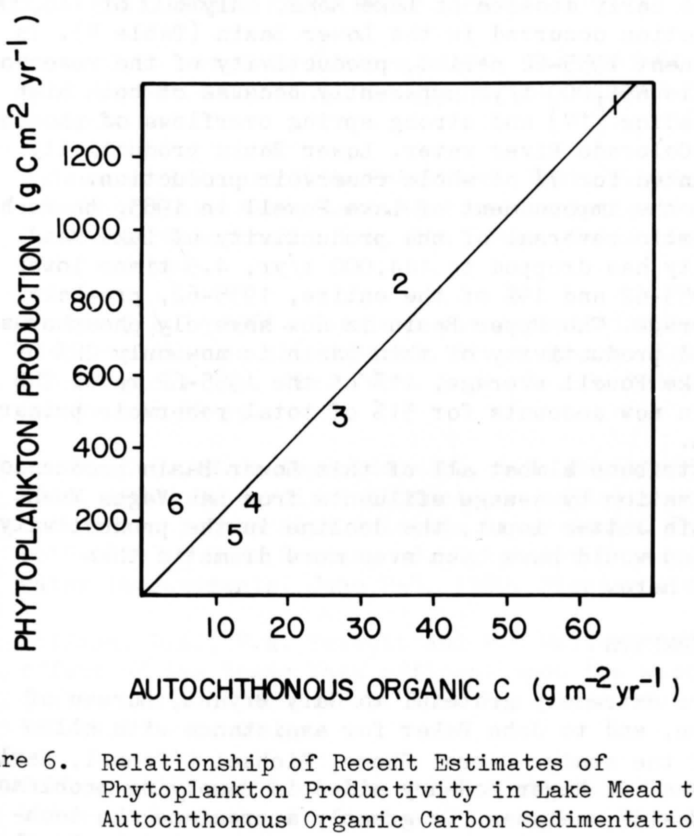

Figure 6. Relationship of Recent Estimates of Phytoplankton Productivity in Lake Mead to Autochthonous Organic Carbon Sedimentation in the Post-Lake Powell Period.

Rates of phytoplankton productivity estimated for each station with Equation 1 were extrapolated over larger areas of the reservoir to estimate reservoir-wide and individual basin total annual production (Table V). The spatial and historical trends in total production (Table V) necessarily follow those for autochthonous organic carbon sedimentation (Table IV) and thus do not provide different information. However, historical rates in units of productivity enable us to better reconstruct the trophic history of Lake Mead.

Table V. Reservoir-Wide and Individual Basin Estimates of Historical Rates of Phytoplankton Production ($t\ C/yr \times 10^3$).

Interval	Whole Lake	Lower Basin	Upper Basin	Lower and Upper Basin
≤ 1954	146	0.6	117	118
1955-62	651	43	395	438
≥ 1963	144	73	44	117

In the early decades of Lake Mead, only 600 of 146,000 t/yr production occurred in the Lower Basin (Table V). In the subsequent 1955-62 period, productivity of the reservoir increased to 651,000 t/yr apparently because of both high nitrate loading [17] and strong spring overflows of phosphorus-rich, Colorado River water. Lower Basin productivity then accounted for 7% of whole reservoir production.

Since the impoundment of Lake Powell in 1963, there has been a drastic reversal of the productivity of Lake Mead. Productivity has dropped to 144,000 t/yr, 4.5 times lower than in 1955-62 and 49% of the entire, 1935-62, pre-Lake Powell average. The Upper Basin is now severely phosphorus limited and productivity of this basin is now only 22% of the pre-Lake Powell average, 11% of the 1955-62 rate. The Lower Basin now accounts for 51% of total reservoir primary production.

We attribute almost all of this Lower Basin production to fertilization by sewage effluents from Las Vegas Wash. Without this latter input, the decline in the productivity of Lake Mead would have been even more dramatic than documented here.

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CHAPTER 9

WATER QUALITY TRENDS IN THE LAS VEGAS WASH WETLANDS

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INTRODUCTION

The Las Vegas Wash is a wetlands ecosystem that acts to buffer the effects of wastewater discharges on the receiving waters of Lake Mead. The wash is the terminus for the 4,144 km^2 Las Vegas Valley drainage basin, emptying into Las Vegas Bay of Lake Mead (Colorado River). It is in the northern Mojave desert, which receives an average of only 10 cm of rainfall annually. The Las Vegas Wash is technically an artificial wetland supported almost entirely by the perennial flows from sewage treatment plants. These flows contribute an average of 3.7 t of nutrients (nitrogen and phosphorus) and 4 t of oxygen consuming organic material (BOD_5) to Lake Mead per day. High nitrate and total dissolved solid loads (2.7 and 603 t/day respectively) are derived primarily from groundwater inputs in the lower wash [1,2,3]. The contaminated groundwater originates from large underground salt mounds that were formed from discharges of industrial effluents into unlined evaporation ponds until 1978.

Conflicting interests among municipal, recreational, and down-river users make the Las Vegas Wash a focal point in current legal disputes regarding the need for advanced wastewater treatment (AWT). In light of rapidly escalating costs, especially for energy and chemicals needed for AWT, many municipalities nation-wide are investigating alternative treatment techniques. Public Law 92-500, Section 210 (parts d and f) specifically encourages the reclamation and recycling of wastewaters. Operation of treatment facilities to produce revenue through the production of agriculture, silviculture, or aquaculture products is encouraged. Combinations of open space and recreational uses with waste treatment management techniques are also emphasized in PL 92-500.

The Las Vegas Wash ecosystem has been identified as a potential wastewater treatment system. Previous investigations [4,5] indicate that the ecosystem could be removing substantial amounts of nutrients from wastewaters. Goldman and Deacon [5] recommended "that a specifically designed nutrient removal management program be developed and implemented with the flow distribution and erosion control program necessary to maintain wetland wildlife habitat."

The purpose of this paper is to describe historical and current water quality and to quantify the degree of nutrient removal presently occurring in the Las Vegas Wash.

DESCRIPTION OF THE STUDY AREA

Las Vegas Wash is located in Clark County, Nevada, between the City of Las Vegas and Lake Mead (Figure 1). The boundaries of the Wash are defined by a large drainage system that was once part of the pluvial Las Vegas River [6]. Our research was focused in the 18 km stretch downstream of the City of Las Vegas sewage treatment plant (STP) to Las Vegas Bay of Lake Mead.

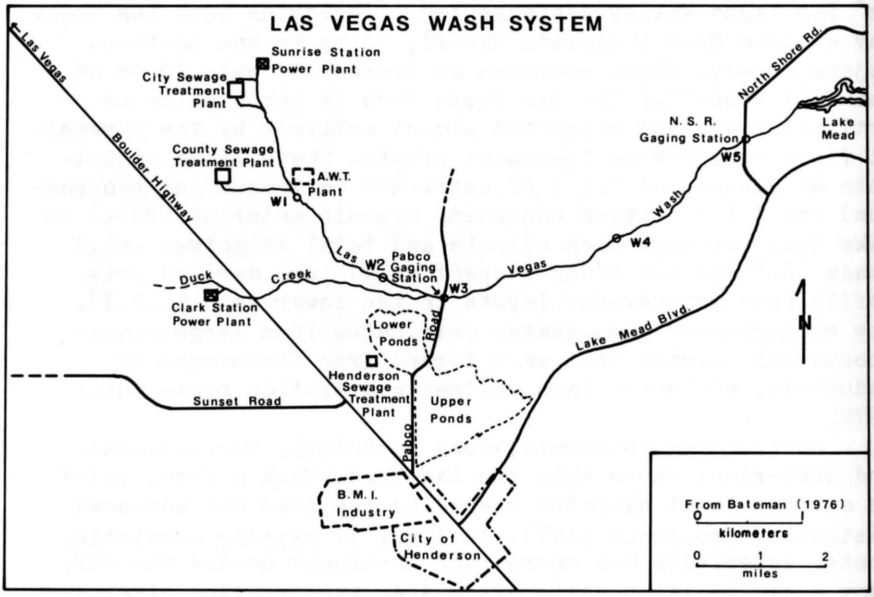

Figure 1. Sampling Site Location Map.

The water in Las Vegas Wash is comprised of 90% secondarily-treated wastewater from the City of Las Vegas and Clark County STPs. Vegas Creek, the last natural creek in the Las Vegas Valley, dried up in the late 1940's [7]. Flows in the present riparian and marsh wetland have increased with the population of Las Vegas Valley. The Valley is home to nearly one-half million permanent residents and host to approximately nine million tourists annually. Las Vegas has become one of the fastest growing urban areas in the United States. Concomitant with this growth has been an increase in wastewater discharges to Lake Mead. Total discharges currently average 2.8 m^3/sec (100 cfs). This amount is twice the flow rate measured at Northshore Road in 1970 (Figure 2).

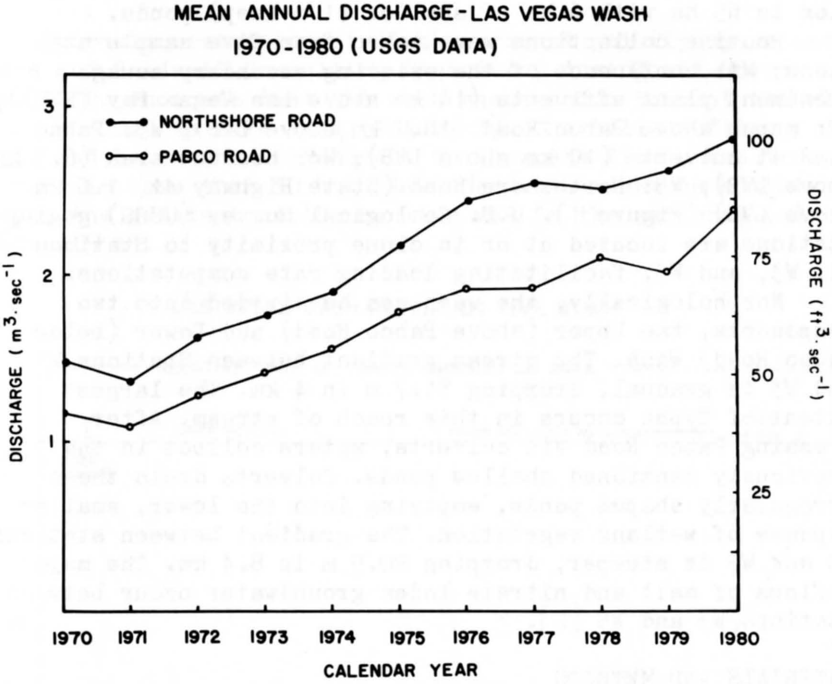

Figure 2. Historical Discharges From Las Vegas Wash.

Increasing volumes of perennial surface water as well as stormwater discharges have transformed sparse desert shrub and mesquite woodland habitats into dense growths of hydrophytic wetland vegetation dominated by Typha domengensis (cattail) and Phragmites communis (common reed). Extensive growths of the introduced phreatophyte Tamarix petandra (salt cedar) border the wetland and riparian zones.

In 1975, a channelization program was initiated in the upper reach of the wash from the City and County STPs to 1.6 km downstream. This man-made channelization has steadily decreased the extent of wetland from the 1969 to 1975 maximum of approximately 730 ha to 120 ha in 1979. Increased flow velocities and unstable soils in the lower portion of the wetland have also facilitated increased erosion rates at areas known as "headcut" regions for the past 5 yr. Sediment transport in 1979 and 1980 was particularly large. Erosion and headcutting in the lower reach of the Las Vegas Wash is especially prominent during flash flooding and accelerated erosion occurred during this study. The principal headcut region advanced approximately 1.5 km upstream during a single storm event of February 1980. Upstream progression of erosion has resulted in the draining of another 50 ha of wetland creating a riparian habitat with channel depths often exceeding 6 m. Present areal extent of wetland vegetation is 65 ha with 6 ha of shallow (1 m deep) ponds.

Routine collections were taken from five sample stations; W1: confluence of the existing secondary sewage treatment plant effluents (14 km above Las Vegas Bay (LVB)); W2: marsh above Pabco Road (10.7 km above LVB); W3: Pabco Road at culverts (10 km above LVB); W4: headcut area (6.3 km above LVB); W5: Northshore Road (State Highway 41, 1.6 km above LVB) (Figure 1). U.S. Geological Survey (USGS) gaging stations are located at or in close proximity to Stations W1, W3, and W5, facilitating loading rate computations.

Morphologically, the wash can be divided into two components, the Upper (above Pabco Road) and Lower (below Pabco Road) Wash. The stream gradient between Stations W1 and W3 is gradual, dropping 31.7 m in 4 km. The largest extent of Typha occurs in this reach of stream. After crossing Pabco Road via culverts, waters collect in the previously mentioned shallow ponds. Culverts drain these irregularly shaped ponds, emptying into the lower, smaller expanse of wetland vegetation. The gradient between stations W3 and W5 is steeper, dropping 80.5 m in 8.4 km. The major inflows of salt and nitrate laden groundwater occur between Stations W3 and W5 [2].

MATERIALS AND METHODS

These five sampling stations were monitored biweekly from July 1979 to December 1980. Special studies were also conducted to determine diurnal variations in nutrients and flow regimes. Over 40 sampling rounds were conducted during the 18 month study.

Field measurements and sample collections were performed under contractual agreement between the Lake Mead Limnological Research Center, University of Nevada, Las Vegas, and Brown and Caldwell Consulting Engineers of Sacramento, California. Water samples were collected with a large plastic bucket, and subsamples were collected in plastic bottles and preserved on ice. Samples for soluble nutrient analyses were filtered through GF/C filters upon return to the laboratory. The samples were iced and shipped to the Brown and Caldwell laboratory in Emeryville, California for analysis. All analyses were performed as prescribed by U.S. EPA [8].

In addition to physical and chemical measurements, rhodamine WT dye was introduced into segments of the wash during a special study to determine hydraulic retention time. The dye was tracked using a Turner Designs Model 10 fluorometer, and Instrumentation Specialty Corporation (ISCO) automatic water samplers.

RESULTS AND DISCUSSION

Hydraulic retention studies performed jointly by us and Brown and Caldwell Consulting Engineers, during mid-November, 1980, indicate that the wash has a short time of travel from the STPs to Lake Mead (Figure 3). Channelized flows above and below the wetland act to increase flows, and travel time is less than 20 h to Lake Mead. As might be anticipated, the greatest residence time was in the wetlands and ponds (15 h).

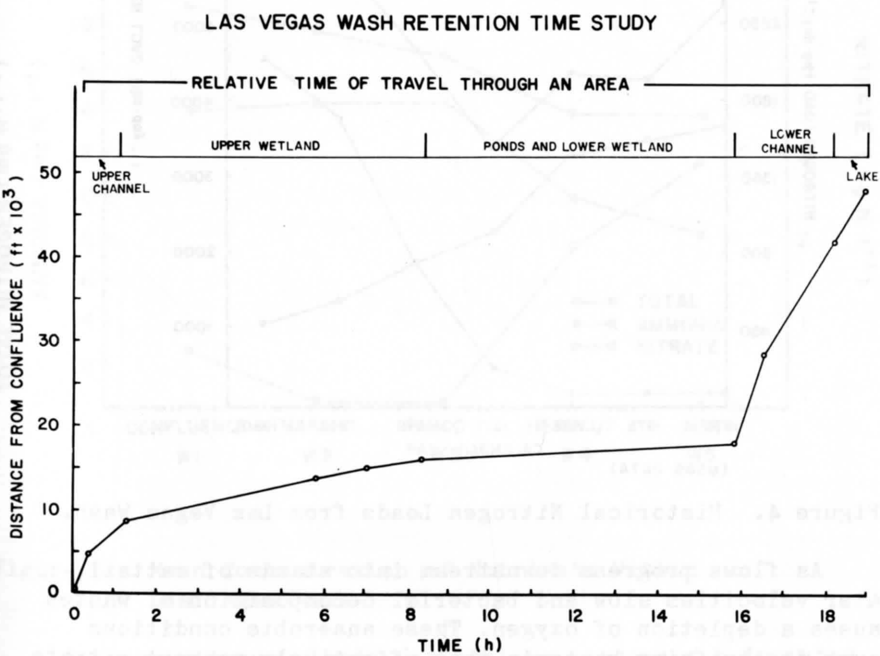

Figure 3. Hydraulic Retention Time Within Las Vegas Wash.

Relatively complete historical nutrient data are available in the USGS records for Las Vegas Wash. Summaries of past data indicate that some dramatic changes have occurred during the last 6 yr of monitoring. Nitrogen loads at Northshore Road have steadily increased since 1977, a drought year (Figure 4). Discharges of industrial wastes into unlined ponds, constructed in the 1940's, was discontinued in the mid-1970's. This has led to a gradual decline of nitrate loads contributed by shallow groundwater aquifers. Ammonia, however, has steadily increased. Ammonia loads at Northshore Road were less than 10% of the total nitrogen load prior to 1977, while current levels exceed 60%.

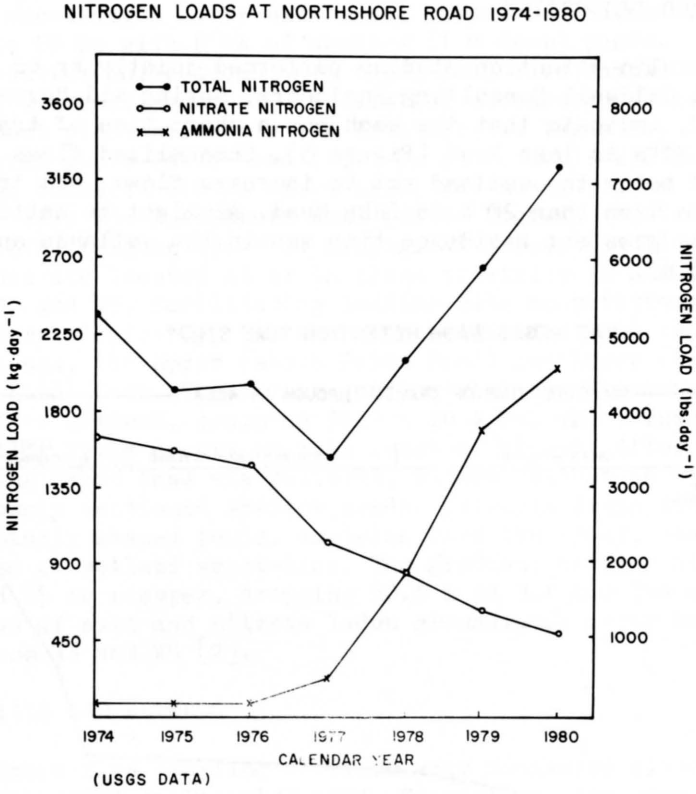

Figure 4. Historical Nitrogen Loads from Las Vegas Wash.

As flows progress downstream into stands of cattail, water velocities slow and bacterial decomposition of wastes causes a depletion of oxygen. These anaerobic conditions favor denitrifying bacteria that effectively convert nitrate to nitrogen gas. No rate measurements are currently available on this, but it appears to result in decreased nitrate concentrations in the upper marsh throughout the year (Figure 5). Denitrification has been generally cited as the major reason that wetlands are nitrogen traps or sinks. Ammonia concentrations increased slightly in this area, apparently due to bacterial decomposition of organic nitrogen compounds.

Overall, total nitrogen concentrations (and loads) decreased within the wash system. Total nitrogen was reduced by 27% between Stations W1 and W5 during the summer of 1980. This was as high as 47% removal on some occasions and averaged 15.4% during the entire study. Nitrate concentrations at Northshore Road increased to 12 mg/l on three occasions. One event on March 4, 1980 was traced to a leaking pipe which transports industrial wastes to lined evaporation ponds in upland areas of the wash. This resulted in six to seven-fold increases over normal nitrate loads to Lake Mead. Because of these perturbations, the mean removals of nitrogen are conservative.

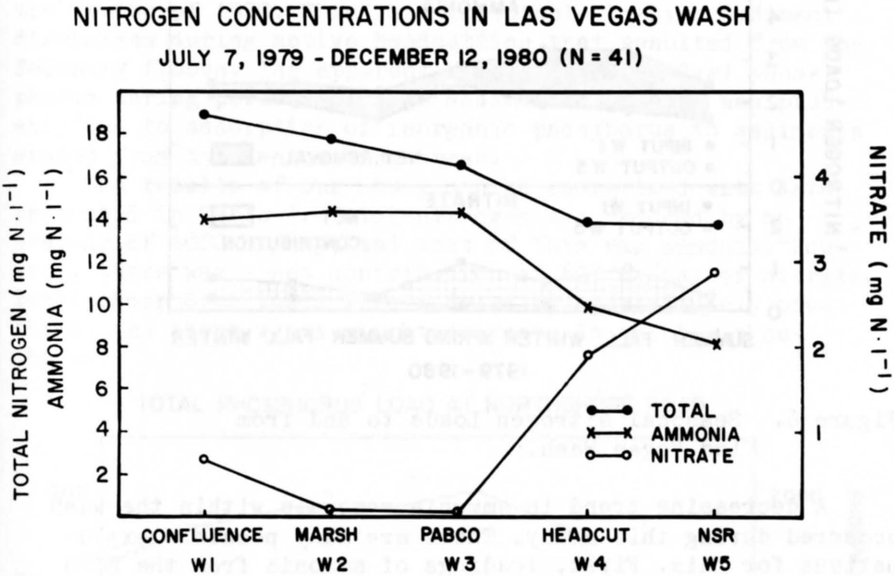

Figure 5. Mean Concentration of Nutrients Within Las Vegas Wash.

Average nitrogen loads for various seasons are depicted in Figure 6. Loads were calculated from average flows recorded by USGS for the day water samples were taken. There were net removals of total nitrogen and ammonia in the wash. However, there was a net contribution of nitrate, primarily as a result of groundwater inputs in the lower wash. The effects of perturbations discussed earlier can be seen in peaks of nitrate loads during fall of 1979 and spring of 1980.

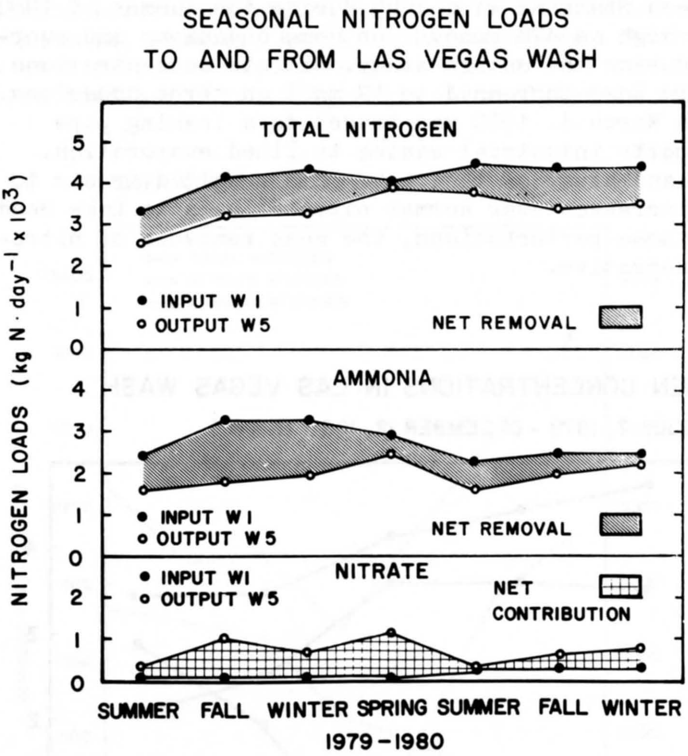

Figure 6. Seasonal Nitrogen Loads to and from Las Vegas Wash.

A decreasing trend in ammonia removals within the wash occurred during this study. There are many possible explanations for this. First, loadings of ammonia from the STPs were lower during later portions of the study. Second, storm events of winter 1979-1980 caused a major upstream advancement of erosion. Decreased removal efficiencies during lower loadings were also observed by Morris et al. [9] in related wetland studies in the Lake Tahoe Basin. They found the most dramatic nutrient and sediment removals occurred when loads were greatest. Another relevant conclusion was that one of the most important factors in determining the effectiveness of wetland treatment was the degree of sheet flows across the wetland. In the case of the Las Vegas Wash, channelization limits spatial and temporal contact of waters with wetland vegetation and, therefore, limits nutrient reducing capabilities.

Goldman and Deacon [5] suggested that one mechanism that may be responsible for ammonia removals within the Las Vegas Wash is the adsorption to clay particles. These investigators indicated 90% reductions in ammonia loading as measured at Northshore Road in comparison to lower (31%) reductions seen during this study. It is possible that active headcutting zones may be eroding strata of less clay content than historical headcut zones. Based on elevational differences between past and present headcut zones, it seems that this may be true. However, a more detailed analysis of the system is required to give a definitive answer as to what mechanism plays a dominant role.

Historical phosphorus data measured at Northshore Road (Figure 7) indicate that recent upgrading of sewage treatment facilities is reducing total phosphorus loads in comparison to previous years. Seasonal analysis of phosphorus (Figure 8) shows a net contribution of total phosphorus in the spring of 1980. This is attributable to high sediment discharges during active headcutting that resulted from the February floods. The apparent removal of dissolved phosphorus during periods of high sediment discharge was probably due to adsorption of inorganic phosphorus to sediments eroded from the headcutting areas.

The results of our study can be summarized with data presented in Table I. Total nitrogen was reduced by an average of 1000 kg/day, and most of this was ammonia. However, there was a net contribution of 697 kg/day of nitrate. Total phosphorus loads were reduced by approximately one-third, and there was a slight decrease in soluble phosphorus.

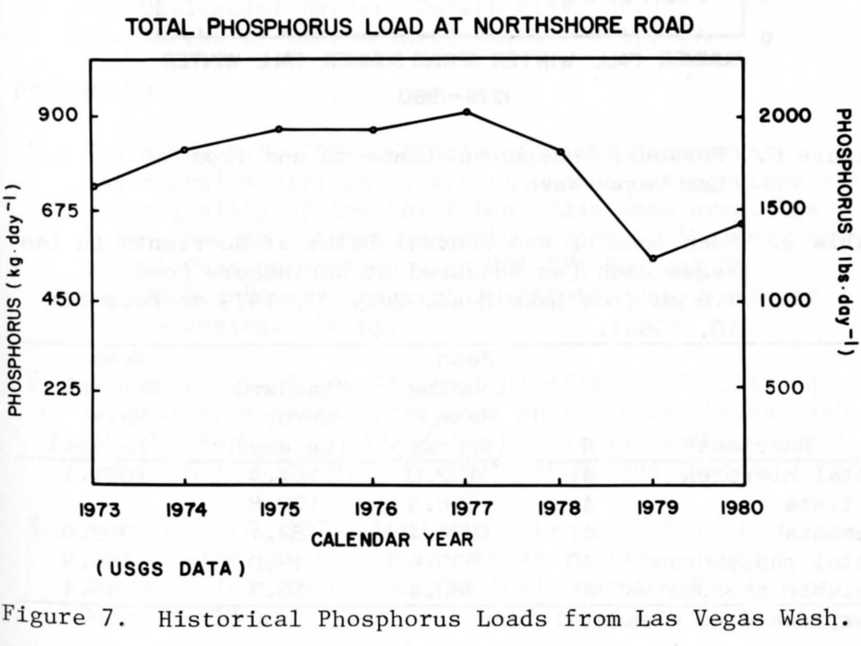

Figure 7. Historical Phosphorus Loads from Las Vegas Wash.

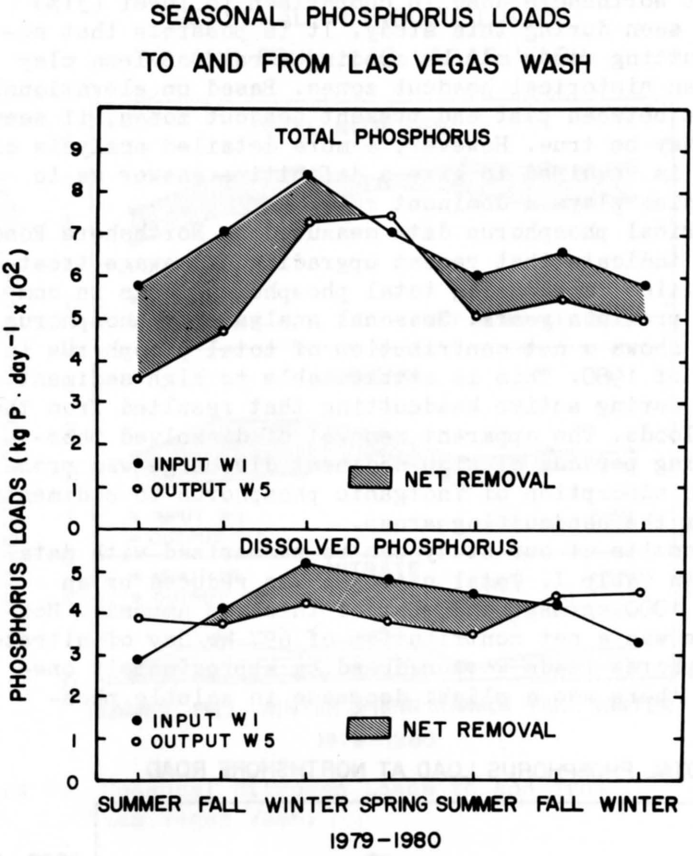

Figure 8. Seasonal Phosphorus Loads to and from Las Vegas Wash.

Table I. Mean Loading and Removal Rates of Nutrients in Las Vegas Wash (as Measured at Northshore Road, 1.6 km from Lake Mead, July 23, 1979 to December 18, 1980).

Nutrient*	N	Mean Loading Rate (kg/day)	Standard Error (kg/day)	Mean Removal Rate (kg/day)
Total nitrogen	41	3172.0	164.4	1001.1
Nitrate	41	696.9	107.8	-
Ammonia	41	1874.4	82.1	992.0
Total phosphorus	40	539.6	45.0	156.9
Soluble phosphorus	41	380.4	16.7	49.4

*Expressed as elemental form

Although the wastewaters are retained within the Las Vegas Wash wetlands for a relatively short period of time, this ecosystem is behaving seasonally as a nitrogen and phosphorus trap. This results in an improvement of the quality of water discharged to Lake Mead. Efficiency of nitrogen and phosphorus removal is a function of the loads entering the system and the degree of contact of waters with the wetland. Increasing velocities and volumes of flows have decreased retention time resulting in less contact time with the wetland. Rates of nutrient removal in the Las Vegas Wash as described by URS and Clark County Department of Comprehensive Planning in 1978 [10] appear to be declining as a result of changes in flow regimes. Improving the efficiency of nutrient removal by proper management of this wetland appears to be feasible. Further studies should be conducted to elucidate specific mechanisms of nutrient removals.

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