Technical Report
CRWR 215

EVALUATION OF THE EFFECTS OF POINT AND NONPOINT SOURCE PHOSPHORUS LOADS UPON WATER QUALITY IN THE HIGHLAND LAKES

Part I

by
James D. Miertschin
and
Neal E. Armstrong

September 1986

CENTER FOR RESEARCH IN WATER RESOURCES

Bureau of Engineering Research
Department of Civil Engineering
The University of Texas at Austin
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ABSTRACT

The effects of external mass loadings upon phosphorus concentrations in the Highland Lakes, a series of seven reservoirs in Central Texas, were examined. The watersheds of the reservoirs are being subjected to increasing urban development, and concern has been expressed with respect to the potential effects of domestic wastewater discharges and urban stormwater runoff upon water quality. A continuously-stirred tank reactor model was developed for the series of impoundments and applied to the analysis of long-term phosphorus concentrations. The model constructed a phosphorus mass budget for each reservoir that accounted for the annual influx of mass from tributaries, stormwater runoff, precipitation, wastewater treatment facilities, and releases from upstream reservoirs; removal of mass from the water column as a first-order decay process; and transport of mass out of the impoundment through releases from the dam. Point and nonpoint source mass loadings of phosphorus to the reservoirs were defined for historical and projected future conditions within the watersheds. Historical loading estimates were utilized for model calibration, which entailed adjustment of the first-order reaction rate based upon observed concentration data. The model was employed for simulation of future phosphorus concentrations in the Highland Lakes under several sets of alternative conditions for the period 1984-2000. The interrelationships of water quality through the series of reservoirs were examined in order to assess the effects of mass loadings to upstream reservoirs upon phosphorus concentrations in lower reservoirs. Implications of alternative point and nonpoint source control measures were demonstrated by the modeling exercises. The simulations provided a perspective on the relative significance of watershed management or nutrient control measures in either contiguous or upstream watersheds.

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CHAPTER 1.0

INTRODUCTION

The predominant feature of the surface water resources of Central Texas is the chain of reservoirs that impounds the waters of the Colorado, Llano, and Pedernales Rivers--the Highland Lakes. These seven reservoirs, shown in Fig. 1-1, include Lake Buchanan, Lake Inks, Lake Lyndon B. Johnson, Lake Marble Falls, Lake Travis, Lake Austin, and Town Lake. (It is perhaps arguable whether or not Town Lake should be classified as one of the Highland Lakes; nonetheless, it was included in this analysis.) The impoundments provide flood control, water conservation, hydroelectric power, water supply, and recreation. In addition, they contribute to the distinct character of the Central Texas area.

Lake Buchanan is the uppermost impoundment in the series of reservoirs, and its water level fluctuates greatly according to the inflow from the Colorado River and the needs of the downstream reservoirs. Only small communities and developments are located on the shoreline. Lake Inks is maintained at a constant level below Lake Buchanan. This small reservoir is the site of a popular state park. Below Lake Inks lies the constant-level Lake Lyndon B. Johnson (Lake LBJ). Formerly Lake Granite Shoals, the reservoir was renamed in honor of the 36th President of the United States, who frequently cruised its waters accompanied by a flotilla of secret service agents and curious boaters. The unregulated Llano River is a tributary to the reservoir. The small communities of Kingsland and Granite Shoals, and numerous subdivisions and developments, are situated along the shoreline. Lake Marble Falls is next downstream. This constant level reservoir is relatively small and narrow. Residential areas and the community of Marble Falls are fixtures on its shoreline.

Below Lake Marble Falls lies Lake Travis, the largest of the Highland Lakes. Lake Travis receives additional inflow from the Pedernales River. Several large developments and subdivisions are situated on the reservoir. The water level of Lake Travis fluctuates greatly in response to inflow and downstream uses. The constant level Lake Austin lies below Lake Travis. The reservoir is long and narrow, and relatively shallow for most of its length. Along its shoreline are numerous residential dwellings and weekend homes, and several large developments are currently being created within its watershed. The reservoir and its shoreline are incorporated within the City of Austin, and the eastern end of the watershed in particular is becoming heavily urbanized. The lowermost impoundment in the chain of Highland Lakes is Town Lake, situated within the City of Austin. This small reservoir is relatively narrow and shallow and receives a substantial amount of urban runoff.

1.1 NEED FOR STUDY

The Highland Lakes are an important water resource in the central Texas area. Considering the importance of these reservoirs, relatively little effort has been expended in water quality management. This is undoubtedly a consequence of the prevalence of excellent water quality in the Highland Lakes and an historic lack of significant wastewater discharges. Water quality data collection has historically been sporadic and limited in scope. A singular exception was the comprehensive study of Lake Lyndon B. Johnson conducted by the Center for Research in Water Resources at The University of Texas at Austin in the early 1970's. That multifaceted study was conducted to investigate the effects of a new steam-electric generating station, which utilized the reservoir's waters for once-through cooling. With the increasing developmental pressure on the watershed, additional scientific study of the Highland Lakes is urgently needed.

Presently, the Highland Lakes are generally characterized by a high quality of water, but their scenic watersheds are being subjected to continually increasing urban development pressure. Increased urbanization may be accompanied by increased stormwater runoff and pollutant loadings, which could conceivably accelerate the natural eutrophication process in the waterbodies, impact water supply and treatment, and impair recreational activities. Developmental pressure has recently been most intense within the watersheds of Lake Travis, Lake Austin, and Town Lake. A complex of water quality control measures has either been promulgated or proposed. Watershed ordinances have been enacted or proposed that set forth developmental guidelines, with the basic intent of controlling urban runoff in an attempt to maintain acceptable water quality in the reservoirs. For example, in an effort to prevent or mitigate the deterioration of Lake Austin and maintain a high quality water supply, the City of Austin instituted in 1980 a complex of structural and nonstructural stormwater runoff control measures for the Lake Austin watershed. A similar developmental ordinance was recently promulgated for the watershed of Lake Travis. One of the volatile ongoing issues concerns protection of water quality in Barton Creek and Barton Springs (tributary to Town Lake), which entails strict developmental controls in the immediate watershed, adopted in 1981, and proposed controls over an extensive recharge area of the underlying Edwards Aquifer. Debate and opposition have materialized with regard to the proliferation of wastewater discharges which may accompany future developments. In 1983, the Texas Water Development Board enacted a rule prohibiting additional discharges of wastewater to Lake Travis and Lake Austin.

With these immediate water quality concerns, attention has typically focused upon a single reservoir for each issue, when in fact, the Highland Lakes are an interdependent series of reservoirs. That is, when attention has focused upon, for example, watershed ordinances to control runoff water quality into Lake Austin, relatively little analysis was devoted to the long-term water quality characteristics of the releases from Lake Travis or development within its watershed. To the present time, there have been no supporting studies to demonstrate the efficacy of runoff control within a single watershed for maintenance of long-term reservoir water quality.

A need exists to investigate the implications of point and nonpoint source controls upon water quality in the Highland Lakes. A comprehensive analysis of the entire chain of Highland Lakes is required to evaluate the efficacy of water quality management strategies and concepts. The utility of runoff control, the effectiveness of advanced wastewater treatment or nutrient removal, or other concepts should be examined for the series of reservoirs as a long-term water quality management exercise. For this purpose, a mathematical model with predictive capabilities is an indispensable analytical tool.

The thrust of the present investigation revolved around a continuously-stirred tank reactor (CSTR) model and the analysis of long-term phosphorus concentrations. The modeling system was structured for analysis of the Highland Lakes as a series of interconnected reservoirs. The interrelationships of water quality among the Highland Lakes, for example the relationship between pairs of sequential reservoirs, were examined in order to assess the effect of upstream loadings or controls upon concentrations of phosphorus downstream. As a key aspect in the management of water quality in any particular reservoir, the relative significance of watershed management or nutrient control in either contiguous or upstream watersheds was investigated.

Phosphorus is one of the primary nutrients in algal nutrition, and relationships have been demonstrated in numerous waterbodies between the total phosphorus concentration and the standing crop of phytoplankton. Perhaps the key water quality concern within the Highland Lakes is the prospect of algal nuisance conditions, which could potentially become manifest under increased influx of nutrients. The modeling exercises were devised to demonstrate the implications of point and nonpoint source control measures upon phosphorus levels in the study reservoirs. At the outset, it was realized that the development and execution of such a modeling analysis might not provide definitive conclusions, due to limitations of the available data base. However, the analysis was expected to identify management strategies with a likelihood of success, and define the additional data needs, so that a systematic data collection effort and water quality management strategy could be implemented.

Completion of this study, hopefully, will provide impetus for continued interest and research in the Highland Lakes system. As a result of the study, a mathematical model for phosphorus is available and on-line for immediate reference and use. Future data requirements and research needs have been identified, necessary to effect model refinement and enhance predictive capability.

1.2 OBJECTIVES

The overall objective of the study was to develop a water quality model for the series of Highland Lakes that can be used to assess the effects of point and nonpoint source loads upon reservoir phosphorus concentrations. The specific objectives of the study are summarized below:

i) conduct a thorough, systematic compilation and evaluation of phosphorus data for the Highland Lakes, presently available from several sources;

ii) estimate present and projected point and nonpoint source phosphorus loads within the individual watersheds of the study reservoirs;

iii) develop a mathematical model for the Highland Lakes, based upon the concept of a series of continuously-stirred reactors, that can be used to investigate water quality interrelationships through the series of reservoirs, i.e., to evaluate the effects of upstream mass loadings of phosphorus upon water quality downstream;

iv) employ the mathematical model to obtain long-term, time-dependent predictions of phosphorus levels in the reservoirs in order to evaluate the utility of currently implemented or envisioned nonpoint source controls and point source effluent limitations and the implications upon water quality for both the series of Highland Lakes and individual reservoirs;

v) identify future data and research needs for continued study of the Highland Lakes, with an ultimate objective of the development of a more definitive water quality management plan.

1.3 SCOPE OF STUDY

The scope of the present study is described below in order to define the level of effort associated with fulfillment of the primary objectives. The most significant limitation encountered was the lack of a substantial water quality data base. In several respects, data limitations influenced the selection and application of analytical methodologies.

Literature Review

The present study included a review of pertinent literature, which provided exposure to related water quality modeling activities, with emphasis on the various methodologies available for simulation of reservoir systems. As related to the concept of reservoir phosphorus modeling, the literature was surveyed for pertinent information on nutrient behavior, trophic state analysis, and the characterization of phosphorus mass loads. Literature sources included numerous scientific and engineering journals and published reports. Literature pertaining specifically to the Highland Lakes was also reviewed, including several publications by the Center for Research in Water Resources at The University of Texas at Austin.

Data Inventory and Analysis

Phosphorus data for the Highland Lakes, available from several sources, were compiled and evaluated. To complement the phosphorus data, chloride data were compiled for analysis of a conservative constituent. Data sources included the Texas Department of Water Resources (TDWR), U.S. Geological Survey (USGS), The University of Texas at Austin Center for Research in Water Resources (CRWR), and the City of Austin. (Since the time these data were compiled and the present analyses were conducted, the Lower Colorado River Authority has acquired additional sampling data on the reservoirs.) Data were also compiled for major tributaries of the reservoirs.

Data analysis included several activities. Phosphorus and chloride data were examined for spatial and temporal trends. Data sets approximating steady-state hydrologic conditions and defining periods of discernible concentration trends were isolated to support the analysis of phosphorus and chloride budgets on the series of reservoirs.

Hydrological data were also compiled. For major tributaries, flow frequency statistics and daily discharge data were obtained. The operating scheme for the interconnected series of reservoirs was identified, and data for reservoir releases, inflows, and evaporation were obtained from the Lower Colorado River Authority (LCRA).

Point and Nonpoint Source Loads

Data were compiled to describe point and nonpoint source loadings of phosphorus and chloride in the study area. Point source information was obtained from the permit files of the TDWR. Nonpoint loadings were estimated through analysis of available water quality data, primarily stormwater data collection activities conducted in the Austin area by the USGS and routine tributary monitoring activities of the TDWR and USGS.

Model Development and Simulation of Phosphorus Concentrations

A mathematical model for phosphorus was developed and applied to the Highland Lakes, based upon representing the series of reservoirs as a network of continuously-stirred tank reactors (CSTR's). The model was employed for simulation of levels of both a conservative and a nonconservative constituent, viz., chlorides and phosphorus. Chloride levels were simulated for assistance in model calibration. The modeling framework was structured for analysis of the interrelationships of water quality through the series of reservoirs. Simulations emphasized long-term, time-dependent prediction of phosphorus concentrations. The model was employed for simulation of future phosphorus concentrations up to the year 2000 under several sets of alternative conditions. Ten scenarios were formulated to represent a range of future development conditions.

Implications of Controls Upon Water Quality

The model was employed to evaluate the effects of nonpoint source controls and point source effluent limitations upon levels of phosphorus for both the series of Highland Lakes and individual reservoirs. Definition of the interrelationships of water quality among the series of reservoirs enabled assessment of the effects of upstream mass loadings upon constituent concentrations downstream. As a key aspect in the management of water quality in any particular reservoir, the relative significance of water quality management or control in either contiguous watersheds or upstream reaches was evaluated.

Future Research Needs

Based on the modeling exercises, future data requirements and research needs were identified. Future studies should address continued monitoring of water quality and provide for model refinement and enhancement of predictive capability.

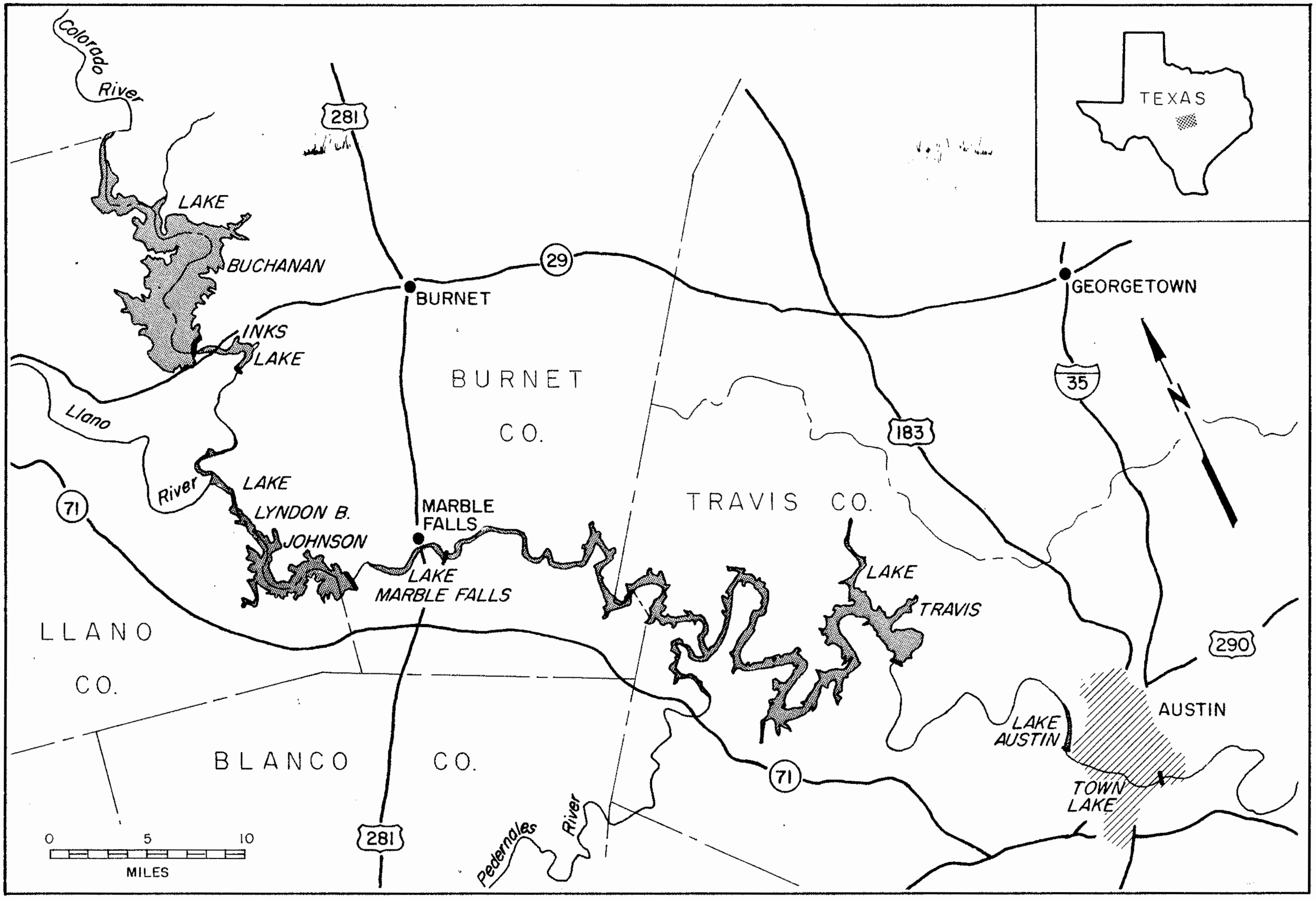


Fig. I-I LOCATION MAP OF HIGHLAND LAKES

CHAPTER 2.0

LITERATURE REVIEW

2.1 RELATED CONCEPTS OF WATER QUALITY IN RESERVOIRS

The principal water quality concern in lakes and reservoirs is typically focused upon eutrophication--the enrichment or fertilization of a waterbody. The terms "eutrophic" and "oligotrophic" were introduced by C. A. Weber in 1907 as synonyms for "rich in nutrients" and "poor in nutrients" in conjunction with a study of German peat bogs (Rodhe, 1969). Eutrophication is a naturally-occurring process, which may be accelerated through man's activities. Natural eutrophication is a consequence of nutrient or organic matter enrichment originating, for example, from the erosion of soil and rock which releases minerals and nutrients to the waterbody. Through the influx of silt and the process of ecological succession, filling of the lake basin may ultimately occur in the aging process. Contributing to accelerated, or cultural, eutrophication are external loadings associated with human influence, such as wastewater discharges, nonpoint source runoff from agricultural land, and stormwater runoff from urban areas.

Edmondson (1974) emphasized separation of the concepts of rate of nutrient supply, nutrient concentration, biological productivity, population density, and ecological succession. In a clarification of the nature of eutrophication and lake productivity, he pointed out that eutrophic means "well fed", not "highly productive." The spectre of eutrophic lakes generates concern because they are likely to produce dense populations of particular kinds of algae. Though a lake that displays a severe algal nuisance is likely to be eutrophic, a eutrophic lake, perhaps with a naturally rich nutrient supply, does not necessarily produce algal nuisances. Differences exist in the response of lakes to nutrient supply, attributable to conditions such as climate, morphometry, water residence time, and the thickness of the epilimnion (Edmondson, 1974). Edmondson's comments may be particularly apropos for an introduction to the study of the Highland Lakes. The vast majority of existing information on eutrophication concepts was developed for waterbodies located in Europe, Canada, and the northern regions of the United States; further, much of the literature is devoted to natural lakes. Impoundments such as the Highland Lakes may display characteristics quite different from natural lakes. For example, the deepest waters of an impoundment are usually located near the dam, rather than in the center of a lake; releases from an impoundment often originate from the hypolimnion, as opposed to surface outflow from a natural lake. With respect to morphometry, Fruh, et al. (1966) discussed the significance of mean depth. A deeper lake will usually have a relatively small euphotic zone, relative to the total volume. Stratification of deep lakes also contributes to the removal of nutrients from the euphotic zone during the warm-weather months by isolation in the hypolimnion. The peculiarities of the Highland Lakes and other reservoirs in the Southwest may affect the relevance of some of the existing literature on eutrophication.

In terms of water quality, reservoirs are often of primary importance among the various hydrologic units of a watershed due to their multiple uses such as water supply and conservation, flood control, and recreation. With the continual sedimentation associated with the often substantial detention features, reservoirs frequently function as the ultimate receiving body for numerous substances. Aquatic environments, and in particular, impoundments, may ultimately receive and concentrate virtually all natural and cultural contaminants produced in a watershed, attributable not only to wastewater discharges but also to increased erosion and runoff (Reid and Wood, 1976). Thus, the significance of pollutant generation and transport throughout a watershed is often magnified within the extant watercourses.

The present modeling analysis focused upon total phosphorus. In a variety of waterbodies, relationships have been demonstrated between the concentration of total phosphorus and the standing crop of phytoplankton or its surrogate, chlorophyll-a (Schindler, 1977; Dillon and Rigler, 1974, Dillon, 1975; Jones and Bachmann, 1976; Canfield and Bachmann, 1981). Phosphorus is one of the primary nutrients in algal nutrition and is frequently assumed to be the algal nutrient of primary concern (Schindler, 1971, 1977; Edmondson, 1974; Lee, et al., 1978). At any particular point in time, phosphorus may or may not be the limiting nutrient in the Highland Lakes. Reported results of algal assays on the series of reservoirs have indicated possible algal growth limitation by phosphorus, nitrogen, or iron (Huang, et al., 1973; Floyd, et al., 1969; Fruh and Davis, 1969b). In many waterbodies, phosphorus is present in short supply compared to nitrogen. Further, certain algae are able to utilize atmospheric nitrogen when other external inputs are deficient. Control of external loadings of phosphorus is technically more easily achievable than control of nitrogen loadings, particularly with point source discharges. Even if phosphorus is not limiting initially, in many instances control of phosphorus input could decrease the available phosphorus concentration in a waterbody sufficiently to render phosphorus the limiting nutrient (Lee, et al., 1978).

This chapter contains a review of literature relevant to the present analysis, with an emphasis upon the modeling of phosphorus and its relation to eutrophication. Additional information from the literature is interspersed throughout the remaining chapters where its application is pertinent.

2.1.1 Limiting Nutrient

The identification of a limiting nutrient involves the examination of the availability and utilization of specific elements for algal nutrition. The concept is embodied by Liebig's "law of the minimum", which asserts that a nutrient present in minimal supply, relative to the requirements of an organism, limits growth. It is frequently assumed that either of the macronutrients nitrogen or phosphorus is limiting to the growth of photosynthetic organisms. Growth limitation may also be attributable to a shortage of one or more micronutrients. Other factors, such as light limitation, should also be considered.

There exist different approaches for identification of a limiting nutrient, which may yield disparate results for specific waterbodies (Verhoff, 1974). The stoichiometric limiting nutrient is determined by the principle of conservation of mass. The stoichiometric composition of algal cells is employed to determine the nutrient that, if totally depleted, will limit the production of biomass. Ratios for the concentration of each available nutrient to the stoichiometric requirement are determined, with the lowest ratio denoting the limiting nutrient. With this approach, only a single nutrient may be limiting at any one time, and it is usually carbon, nitrogen, or phosphorus.

An alternative approach identifies a kinetic limiting nutrient. With this technique, a substance is defined as limiting if an increase in its concentration results in an increased algal growth rate. More than one nutrient can satisfy the criterion and thus be limiting. Limitation may involve both macronutrients and micronutrients (Verhoff, 1974).

In accordance with the stoichiometric composition of algal protoplasm, a nitrogen to phosphorus ratio of approximately 16:1 is generally accepted to be required for algal growth:

$$106\,CO_2 + 16\,NO_3^- + HPO_4^{2-} + 122\,H_2O + 18H^+ + \text{trace elements} + \text{energy}$$

$$p\downarrow\ \uparrow r$$

$$C_{106}\,H_{263}\,O_{110}\,N_{16}\,P_1 + 138O_2$$

which describes a steady state between photosynthetic production (p) and heterotrophic respiration (r) (Stumm and Morgan, 1970). The atomic ratio of 16:1 is equivalent to a mass ratio of 7.2:1. The nitrogen to phosphorus ratio may exhibit variation; for example, Rast and Lee (1978) report a mean atomic ratio of 27:1 for algae from natural waters in the southeastern United States. Theoretically, if the nitrogen to phosphorus ratio in a waterbody is lower than 16:1, nitrogen limitation is indicated, and conversely, phosphorus limitation is implied for ratios in excess of 16:1. The validity of application of the stoichiometric nitrogen to phosphorus ratio to assess nutrient limitation is perhaps questionable, since luxury uptake of either phosphorus or nitrogen by phytoplankton has been demonstrated, and nitrogen fixation may also occur. The forms of the nutrients must be considered for evaluation of nutrient limitation or trophic state. For example, Lee and Jones (1980) indicate that a mass ratio of available nitrogen (ammonia and nitrate nitrogen) to available phosphorus (soluble orthophosphate) in excess of 7.2:1 indicates a phosphorus limited condition. Sawyer's (1947, 1952) widely used criteria suggest that nutrient concentrations in excess of 0.30 mg/l inorganic nitrogen and 0.01 mg/l inorganic phosphorus may produce algal blooms of nuisance density. Sawyer's criteria thus indicate an inorganic nitrogen to phosphorus ratio of 30:1 by weight, or a 66:1 atomic ratio.

2.1.2 Phosphorus Forms and Behavior

Phosphorus is the element most often focused upon in eutrophication studies. Numerous studies have documented the importance of phosphorus in algal nutrition, and it is frequently identified as the limiting nutrient for biological productivity (Schindler, 1971, 1977; Edmondson, 1974; Lee, et al., 1978). However, limitation by nitrogen or trace elements, such as iron, has also been demonstrated (Fruh, 1967; Huang, et al., 1973). Phosphorus is often present in surface waterbodies in relatively small natural concentrations, compared to, for example, ambient levels of nitrogenous compounds. External phosphorus loadings to a waterbody may be associated with both point and nonpoint sources. Point source loadings originating from wastewater discharges, particularly domestic wastes, can contribute substantial nutrient loadings. Nonpoint sources are varied, and include potentially large loadings from agricultural and urban stormwater runoff.

Phosphorus exists in several forms within a waterbody. Historically, the bulk of available data in most situations consists of either total phosphorus or dissolved orthophosphate phosphorus measurements, or both. Hutchinson (1957) distinguished four relevant categories in limnologic studies: soluble phosphate phosphorus, sestonic acid-soluble phosphate phosphorus (mainly ferric and perhaps calcium phosphate), organic soluble (and colloidal) phosphorus, and organic sestonic phosphorus. Aquatic phosphorus forms are differentiated into operationally-defined categories in practice. Soluble and particulate fractions are differentiated by passage of a sample through a 0.45-μm membrane filter. Since this filtration does not assure a true separation of suspended and dissolved fractions, the American Public Health Association (APHA, et al., 1981) employs the contemporary terminology "filtrable" to denote that portion of sample which passes through the filter, and "nonfiltrable" to denote the retained suspended fraction.

Laboratory analyses for phosphorus are described by the American Public Health Association (APHA, et al., 1981). Two general procedures comprise the analysis for phosphorus: conversion of those forms of interest to dissolved orthophosphate, followed by colorimetric determination of dissolved orthophosphate. The total phosphorus, and its filtrable and nonfiltrable components, may be subdivided analytically into three chemical forms of reactive, acid-hydrolyzable, and organic phosphorus. Phosphates measured colorimetrically without preliminary digestive steps are referred to as reactive phosphorus, which includes primarily inorganic orthophosphate, with some condensed phosphates (polyphosphates). Both filtrable and nonfiltrable reactive phosphorus may be detected. Acid-hydrolyzable phosphorus is determined with mild acid hydrolysis at boiling-water temperature. Filtrable and particulate condensed forms are converted by hydrolysis to filtrable orthophosphate. Some organic phosphorus is unavoidably converted to orthophosphate during the hydrolysis procedure. Organic phosphorus forms are converted to orthophosphate by severe oxidative digestion of the organic matter. Both filtrable and nonfiltrable fractions of organic phosphorus may occur.

The interrelationships of the various phosphorus forms in surface waters are depicted in Fig. 2-1 (from Stumm and Morgan, 1970). Dissolved inorganic orthophosphate constitutes the primary source for biological growth in the aquatic environment, though other forms may also be utilized. There is evidence that the phosphorus cycle in a reservoir is extremely active. Suggested phosphorus turnover times include approximately five minutes for the exchange between dissolved inorganic phosphate and phytoplankton. Sediment-water exchanges have been estimated at 15 days for abiotic processes and 3 days for bacterially-mediated processes (Hayes and Phillips, 1958).

Sediment-water interactions are an important element of the phosphorus cycle. In many reservoirs, a net transport of phosphorus to the sediments has been demonstrated, which is effected by the sedimentation of particulate phosphorus. Under anaerobic conditions, sediments tend to release dissolved phosphorus to the overlying water. Thus, in a stratified reservoir, a buildup of phosphorus in the hypolimnion may occur. Transport across the thermocline can deliver a portion of this supply to the euphotic zone, and at seasonal overturn, mixing of epilimnetic and hypolimnetic waters can elevate the mean concentration in the upper water column. However, in an aerobic environment, a portion of the phosphorus is again returned to the sediments by sorption onto particulate iron (Baca, et al., 1976).

The accumulation of soluble phosphate in an anoxic hypolimnion may be attributed to two mechanisms: (1) decomposition of plankton delivered by sedimentation, and (2) reduction of phosphate-containing ferric iron precipitates (Stumm and Morgan, 1970). The exchangeable phosphorus is primarily the dissolved interstitial fraction in the sediment matrix. A sorption-desorption equilibrium exists between the interstitial phosphorus and the phosphorus sorbed onto the sediment particles (Baca, et al., 1976). It is reported by Hutchinson (1957) that an oxidized mud surface prevents diffusion of phosphate and ferrous ions from deeper layers in the sediment, since the ferrous iron is always present in excess, and, when oxidized, precipitates all of the phosphate. It is emphasized that the interaction of phosphate with ferric iron is a chemical reaction dependent upon prevailing concentrations of ferric iron, phosphorus, and pH (Stumm and Morgan, 1970).

As discussed in the introductory section of this chapter, the present modeling analysis focused upon total phosphorus concentrations. Soluble orthophosphate is the form most readily available for algal nutrition. However, examination of only soluble orthophosphate would likely omit a substantial portion of the phosphorus supply. Though a portion of external loadings of total phosphorus is initially lost via sedimentation, subsequent solubilization reactions in the internal phosphorus cycle may reintroduce additional supplies of algal nutrients. Lee, et al. (1978) estimate that of the total external phosphorus load, the biologically available phosphorus load is approximately the soluble orthophosphate plus twenty percent of the difference between the total phosphorus and soluble orthophosphate.

2.2 PHOSPHORUS AND EUTROPHICATION MODELING

As described in the introductory chapter, an integral objective of this study was the development and application of a water quality model for total phosphorus concentrations within the system of Highland Lakes. Water quality modeling for reservoirs covers the gamut of sophistication from simple to complex. For the present study, the development of a model of sufficient sophistication for utilization in water quality management planning was desired. A review of select reservoir water quality modeling activities, with emphasis on phosphorus and eutrophication, is presented below.

There is no single model universally acceptable for all objectives or applicable to all situations. Required formats for, say, modeling long-term chemical constituent behavior or simulation of successional phytoplankton dynamics are markedly different. The most appropriate modeling approach is dictated not only by study objectives but also by the nature and extent of available input data. Certain codes may also require an inordinate amount of computational time--an expense which must often be taken into consideration. Three general approaches to model development were observed by Levins (1966):

(1) sacrificing generality to gain realism and precision;
(2) sacrificing realism to gain generality and precision; and
(3) sacrificing precision to gain realism and generality.

It was suggested that there exists a tendency for development of eutrophication models in accordance with approaches (1) and (3).

2.2.1 Empirical Models for Trophic Analysis

A family of simple empirical models has been developed in recent years for analysis of the general trophic state of a waterbody. These models are basically empirical formulations which relate specific characteristics of a reservoir or external nutrient loadings to a gross trophic delineation.

Vollenweider (1968) formulated a phosphorus loading versus mean depth relationship, thus developing the original trophic model in this category. Vollenweider plotted the annual areal phosphorus load and mean depth of several waterbodies and detected consistent positioning in accordance with trophic state. As indicated in Fig. 2-2, the graph was segmented into eutrophic, mesotrophic, and oligotrophic zones. The permissible loading boundary between the oligotrophic and mesotrophic categories was determined to be

$$L = 0.025\ z^{0.6}$$

while the excessive loading demarcation was

$$L = 0.050\ z^{0.6}$$

where L = areal phosphorus loading, $g/m^2/yr$

z = mean depth, m

The delineations of trophic state recognized that larger waterbodies, as characterized by greater depths, could assimilate more substantial phosphorus loadings before achieving an excessive loading condition.

Vollenweider's original approach was subsequently modified by plotting the areal phosphorus load as a function of the areal hydraulic load (Vollenweider, 1975, 1976, as reported by Rast and Lee, 1978). The ratio of mean depth to hydraulic residence time is equivalent to the hydraulic loading (LT^{-1}), defined as the inflow (L^3T^{-1}) per unit area (L^2). The revised diagram is shown in Fig. 2-3. The hydraulic residence time (volume divided by the outflow) is the reciprocal of the flushing rate of a waterbody. A waterbody with a shorter residence time, or faster flushing rate, could be expected to assimilate larger phosphorus loadings without manifestation of nuisance conditions. Vollenweider's analysis was based upon a simple mass balance model for total phosphorus that accounted for mass inflow and outflow and a first-order decay rate in the water column. According to Dillon and Rigler (1974), Vollenweider assumed that the steady-state phosphorus concentration and loading were related as follows:

$$C = \frac{L}{z(\sigma + \rho)}$$

where
- C = phosphorus concentration, mg/l
- L = phosphorus loading, g/m^2/yr
- z = mean depth, m
- ρ = flushing rate (= outflow/volume), yr^{-1}
- σ = phosphorus sedimentation coefficient, yr^{-1}

With the assumption that the sum of the flushing rate and sedimentation coefficient could be approximated by the reciprocal of the hydraulic residence time, and that the critical phosphorus concentration for departure from oligotrophic conditions was 0.01 mg/l, the boundary line for permissible loading conditions was defined as

$$L = 0.01\ z/\tau$$

where τ = hydraulic residence time (= volume/outflow), T

The excessive phosphorus loading was assumed to be twice the permissible loading, based upon a critical phosphorus concentration of 0.02 mg/l.

In a further modification of the phosphorus loading versus hydraulic loading diagram, shown in Fig. 2-4, Vollenweider (1975, 1976, as reported by Rast and Lee, 1978) incorporated the effects of the phosphorus sedimentation rate. Based upon available data, the phosphorus sedimentation rate was approximated by

$$\sigma = 10/z$$

With this assumption, the boundary line for permissible loadings was defined as

$$L = 0.1 + (0.01\ z/\tau)$$

and the excessive loading delineation was set at

$$L = 0.2 + (0.02\ z/\tau)$$

The modified diagram indicates the existence of a range of values of the ratio of mean depth to hydraulic residence time for which the critical phosphorus loading is relatively constant. In this region of the boundary lines, the capacity for assimilation of phosphorus loadings does not increase with increases in depth or decreases in residence time.

Analyzing the total phosphorus budgets of several lakes in southern Ontario, Dillon (1975) concluded that a number of lakes were not accurately depicted by the Vollenweider phosphorus loading-mean depth relationship, which he attributed to the omission of flushing rate as a model parameter (flushing rate is defined as the ratio of annual discharge to lake volume). Dillon reasoned that the rapid passage of a large volume of water low in nutrients would mathematically yield a high loading rate, but would not necessarily result in eutrophic conditions. Internal processes, particularly sedimentation, also affect the phosphorus dynamics in an impoundment. The fraction of the phosphorus input lost to the sediments was reported to range from 25-50 percent. Data for several lakes indicated that high retention coefficients are characteristic of lakes with a low flushing rate as well as those with low phosphorus loading.

Dillon and Rigler (1974) introduced a retention coefficient into Vollenweider's simple phosphorus budget model as a substitute for the sedimentation rate term. The retention coefficient, R, was expressed as

$$R = 1 - \frac{\rho C}{L/z} = 1 - \frac{\rho}{\rho + \sigma}$$

such that

$$\sigma = \frac{R\rho}{1-R}$$

It was proposed that the retention coefficient be calculated experimentally as

$$R_{exp} = 1 - \frac{Q_o C_o}{\Sigma Q_i C_i}$$

where Q_o = outflow volume, $L^3\ T^{-1}$
Q_i = inflow volume, $L^3\ T^{-1}$
C_o = outflow concentration, ML^{-3}
C_i = inflow concentration, ML^{-3}

Employing the retention coefficient, the steady-state solution to Vollenweider's model formulation was written as

$$C = \frac{L\,(1 - R)}{z\rho}$$

Using this formulation for the steady-state concentration, Dillon (1975) proposed the following parameter to express the effects of flushing rate and retention rate, as well as phosphorus loading, in determining the degree of eutrophy of a lake:

$$\frac{L\,(1 - R)}{\rho}$$

This parameter indicates that increases in both flushing and retention rate reduce the effect of a given loading. A plot of Dillon's model is depicted in Fig. 2-5. The equation for the boundary lines is

$$\frac{L\,(1 - R)}{\rho} = zC$$

where the terms have been previously defined. Classification of lakes is based upon boundary lines that represent steady-state phosphorus concentrations of 0.01 mg/l and 0.02 mg/l for permissible and excessive conditions, respectively.

A graphical eutrophication model was developed by Larsen and Mercier (1976) that examined the mean phosphorus concentration of the incoming runoff versus nutrient retention. The model was based upon the steady-state solution to a simple phosphorus mass balance. Trophic categories of eutrophic, mesotrophic, and oligotrophic were delineated, as shown in Fig. 2-6, employing boundary values of 0.010 mg/l and 0.020 mg/l of phosphorus. The vertical axis represents the potential phosphorus concentration that would be attained if the material acted as a conservative substance (the incoming phosphorus is defined as the annual phosphorus supply divided by the annual water supply). The horizontal axis expresses the aggregation of processes that affect the potential phosphorus concentration through sedimentation. The relative positioning of lakes on the Larsen-Mercier plot and the Dillon (1975) graph is equivalent, because both express the same information.

A graphical method utilizing the critical loading technique was developed by Baca, et al. (1976) for the preliminary assessment of lake trophic state. Differential equations were written to describe the phosphorus balance in the water column and the sediments, and the steady-state solutions were used to develop the critical loading rate graph. The method accounted for the effects of flushing rate, mean depth, and sediment-water interactions on the equilibrium total phosphorus concentration. For application, annual average water outflow, lake surface area, historical annual loading rate, historical phosphorus concentration, and, if under consideration, the estimated new annual phosphorus loading rate are the parameters required. The graphical display of loading rate versus lake parameter (defined as the loading rate divided by the average phosphorus concentration) is shown in Fig. 2-7. The "dangerous" line corresponds to a total phosphorus concentration of 40 μg/l, which was related by the authors to a Secchi depth of 1.5 m and a chlorophyll-a level of 15 μg/l. The line denoted as "admissable" denotes a total phosphorus of 10 μg/l, which corresponds to a Secchi depth of 5.0 m and chlorophyll-a of 3 μg/l, according to Baca, et al. (1976). The method was applied to Lake Washington, Lake Mendota, and Shagawa Lake (located in Washington, Wisconsin, and Minnesota, respectively), with good agreement between predicted results and observed data. For Lake Washington, the method correctly tracked the pre-loading trophic state, the heavy-loading equilibrium state, and the post-nutrient diversion trophic state. The case of Lake Mendota supported the observed lack of a discernible change in trophic state following nutrient diversion. In fact, the results indicated that an order of magnitude decrease in the phosphorus loading would be necessary to affect the trophic state. For Shagawa Lake, improved trophic status was indicated following nutrient diversion.

Imboden (1974) presented a two-compartment lake model, considering an epilimnion and hypolimnion, with phosphorus as the limiting nutrient. Imboden developed a family of curves depicting the tolerable annual phosphorus areal loading rate as a function of mean depth for different hydraulic loading factors (water inflow per unit time per lake surface area) as shown in Fig. 2-8. The criterion employed to distinguish the onset of eutrophy was a hypolimnetic dissolved oxygen decrease of 1 mg/l during a 180-day duration of the midsummer stagnation period.

Simplified empirical models such as those described above have been tested and compared in several studies. The models have been shown to be effective tools for water quality management planning. Bradford and Maiero (1978) utilized both the Dillon and Imboden models to assess the potential for cultural eutrophication in a proposed reservoir in New Jersey. The models indicated that stringent phosphorus control measures would probably be required to prevent eutrophication in the reservoir. Predictions of the Vollenweider, Dillon, and Larsen-Mercier models were compared for 39 lakes sampled in conjunction with the US Environmental Protection Agency's (EPA) National Eutrophication Survey (Hern, 1979). The Dillon and Larsen-Mercier models outperformed the Vollenweider model in an attempt to rank the lakes relative to their ambient phosphorus concentration and trophic state. The Vollenweider phosphorus loading relationship generally correlated well with the trophic state, using data for 38 lakes in the U.S. (Rast and Lee, 1978). Tapp (1978) examined a number of waterbodies in the southeastern United States and concluded that the Vollenweider model, Dillon model, and Larsen-Mercier model were useful for preliminary evaluation of eutrophication problems. It was recommended that further work was needed to relate trophic state and impairment of use to enhance the applicability to southeastern waterbodies, because of innate differences in chemical parameters and turbidity.

Application of simple trophic models, such as Vollenweider's model, to the Twin Lakes Watershed in Ohio was discussed by Cooke, et al. (1974). The Twin Lakes have areas of 26.8 hectares and 34.02 hectares, and volumes of 13.38×10^5 m^3 and 14.8×10^5 m^3. Inflow rates, phosphorus sedimentation rates, and outflows vary widely from month to month in the Twin Lakes. The authors argue that use of averages is inappropriate, and simple trophic models may not be applicable to small watersheds with small lakes of low mean depth. Nutrients generated from relatively minor disturbances in a small watershed may have a significant impact on the nutrient budget of a small lake. Conversely, larger watersheds with larger lakes may respond more slowly to small disturbances in the watershed. Prediction of lake recovery may be more successful with a large lake.

The phosphorus budget models of Vollenweider and Dillon-Rigler were applied to reservoirs in southern Africa by Thornton and Walmsley (1982). Both models successfully predicted steady-state phosphorus concentrations, but further development was suggested before the models could be utilized to predict trophic status. Higgins and Kim (1981) compared predicted steady-state phosphorus concentrations for the Vollenweider and Larsen-Mercier models to data for 18 Tennessee Valley Authority (TVA) reservoirs. Results indicated that phosphorus sedimentation and retention coefficients incorporated within the simple completely mixed mass budget models developed for natural lakes are not directly applicable to the TVA reservoirs. The apparent settling velocity ($v_s = \sigma z$) in the TVA reservoirs was significantly greater than the value assumed by Vollenweider. Vollenweider's relationship for phosphorus removal was more closely approximated in reservoirs with incoming phosphorus concentrations less than 0.025 mg/l. It was suggested that morphometric characteristics of the TVA reservoirs limited the applicability of the simple mass budget models. Canfield and Bachmann (1981) examined data from 704 natural and artificial lakes and concluded that the phosphorus sedimentation coefficient integral to the Vollenweider model should be estimated as

$$\sigma = 0.129\ (L/z)^{0.549}$$

Different formulations were also calculated for natural and artificial lakes. Predicted total phosphorus concentrations with the Vollenweider model were improved with incorporation of this modification.

The empirical phosphorus models described above are relatively simple to apply, and their utility for water quality management planning has been demonstrated. Where the models are employed, it would be prudent to consider the data bases that were used in their formulation. The various data bases included primarily lakes in the northern United States, Canada, and Europe. The empirical models may thus be inappropriate for use in different geographic or climatic regions. Reckhow (1979) recommended that an empirical model should not be applied outside the bounds of the data set used to construct the model without prior verification of the applicability. His evaluation of several models indicated the possibility of predictive biases for certain types of systems, for example, phosphorus-poor or phosphorus-rich lakes.

2.2.2 Mathematical Models

In addition to the simple empirical models described in the preceding section, more complex mathematical models of reservoir water quality have also been developed. These models are generally designed for digital computer application. Advantages of computer simulation include rapid computation time, time-variable solutions, and ability to consider numerous variables.

2.2.2.1 Chemical Models

A mathematical modeling analysis for long-term reservoir water quality was presented by O'Connor and Mueller (1970). The model was applied to the concentration of chlorides in the Great Lakes. Each lake was treated as a completely mixed body of water, and a mass balance was constructed which utilized inflows, outflows, sources, and sinks of material. Good agreement was obtained between the calculated and observed chloride levels. Chloride sources in upstream drainage basins were shown to have a significant effect on concentrations in downstream lakes. The authors concluded that their analysis could provide a framework for development of a coordinated water quality management plan for each individual lake and the entire system.

Chapra (1977) developed a mathematical model for simulation of total phosphorus budgets for each of the Great Lakes. The model employed variables reflecting human development in the basin to derive waste sources in order to estimate the long-term effect of human activities on water quality. The basic framework of the O'Connor-Mueller (1970) model was utilized, though a nonconservative substance, phosphorus, was analyzed as opposed to the conservative material chloride. Average annual values were simulated and each lake was treated as a completely mixed system, with the exception of Lake Erie, which was segregated into three subbasins. Phosphorus input was simulated with a waste source model based on variables such as population and land use. The model incorporated a phosphorus sink to the sediments as a one-way first-order reaction, on the assumption that the net flux of phosphorus on an annual basis is into the sediments.

An historical simulation for the period 1800-1970 was conducted. Results indicated that Lake Ontario was the only lake significantly influenced by an upstream lake, implying that the recovery of Lake Ontario is dependent upon conditions in Lake Erie, which would require a coordinated waste abatement program for both lakes. The model results displayed acceptable correlation coefficients for both loadings and concentration of phosphorus. Generally, phosphorus concentrations were overestimated, implying that in-lake losses to the sediments may be higher than calculated in the analytical framework.

A future simulation for the period 1970-2000 was also conducted in order to predict the magnitude and rate of recovery of the Great Lakes under an idealized phosphorus removal program which assumed a point source effluent limitation of

1 mg/l total phosphorus. Simulation results were interpreted with respect to trophic state under the assumption that mesotrophy is bounded by 10 μg/l and 20 μg/l of total phosphorus (from Dillon, 1975). Results indicated that all lakes would recover to sub-eutrophic levels by 1985, with the exception of one of the Lake Erie subbasins, and further, an oligotrophic state was predicted for Lake Michigan. With this analysis, the usefulness of the model in evaluating the effects of present policies on future water quality was demonstrated.

In order to demonstrate the effects of decreasing or eliminating phosphorus inputs to lakes, Lorenzen, et al. (1976) developed a simple phosphorus budget model and applied it to Lake Washington. The authors point out the general need to apply phosphorus models to a number of lakes where historical data are adequate in order to develop and refine reliable predictive tools. The model was developed to simulate long-term changes resulting from modified loading rates, yet remain as simple as possible. A completely mixed system was assumed, with a phosphorus balance that considered input, outflow, loss to sediments, and release from sediments. An exchangeable or releasable sediment phosphorus concentration was employed to account for phosphorus retained in the sediments. Release of phosphorus from the sediments was represented by first-order kinetics for long-term calculations.

For application to Lake Washington, annual phosphorus loading rates for the period 1931 to 1967 were estimated. Pertinent parameter values used in the simulations included the following: specific rate of phosphorus transfer to the sediments of 36 m/yr; specific rate of phosphorus transfer from the sediments of 0.0012 m/yr; fraction of total phosphorus input to sediment that is unavailable for the exchange process of 0.6; concentration of total exchangeable phosphorus in the sediment at time equal zero of 240 mg/l; initial concentration of total phosphorus in the water column of 0.015 mg/l.

The calculated and observed annual average total phosphorus levels in Lake Washington are displayed graphically in Fig. 2-9. Model calculations indicated that the sediments approach steady state slowly, with implications on lake recovery rates. The importance of a lake's history and initial state was demonstrated. For example, if high phosphorus loading rates are of insufficient duration for sediment equilibration, a quick recovery may be anticipated in response to nutrient diversion. If sediments are equilibrated, longer recovery times may be required to achieve new steady-state phosphorus concentrations. Lorenzen, et al. (1976) point out that their analysis is not applicable to a lake changing from oligotrophic to eutrophic with a periodic anaerobic hypolimnion, due to the effect upon releasable phosphorus fraction. However, it may be possible to employ revised annual average parameter values to represent periods of aerobic and anaerobic conditions. Sensitivity analyses indicated that ±10 percent variation in phosphorus loading rates did not greatly affect model predictions.

The prediction of long-term lake recovery, as related to available phosphorus, was studied by Welch, et al. (1974). Their work was based upon observations from two waterbodies, Lake Washington and Lake Sammamish, which had experienced nutrient budget manipulations. Nutrient diversion from Lake Washington produced measurable indications of recovery:

i) 70 percent decrease in mean winter phosphorus content, from 63 to 20 μg/l;
ii) spring-summer chlorophyll-a decrease from 28 to 8 μg/l;
iii) summer mean Secchi depth increase from 1.0 to 2.5 m.

In contrast, no noticeable response was detected after nutrient diversion from Lake Sammamish. Limnological distinctions between the two lakes were described to account for the differing degree of response. Hypothetical response curves were developed to relate plankton biomass (in terms of chlorophyll-a) to the annual income rate of phosphorus (surface loading rate/mean depth). It was postulated that lake response is not necessarily linearly related to nutrient enrichment. Thus, the response of a lake to a change in its nutrient income would be dependent upon the relative position of the lake under existing conditions on its response curve. For Lake Sammamish, the lack of significant reaction to nutrient diversion could thus be explained by hypothesizing that the reduction in the nutrient income occurred on the asymptotic portion of a response curve that allows for only a small amount of lake response (change in phytoplankton biomass) for a unit increase in nutrient income. The relatively gradual response to nutrient income in Lake Sammamish suggested the influence of an internal mechanism controlling the availability of phosphorus. There is evidence that the internal phosphorus concentration is controlled by iron. Following lake turnover, the phosphorus concentration increases, but it is rapidly complexed by ferric hydroxides and resedimented before it can be utilized by phytoplankton. The hypothetical response curve for Lake Washington is approximately linear up to a certain point where it gradually levels off. Thus, it is assumed that nutrient diversion from Lake Washington occurred on the linear portion of the response curve, producing a marked effect on biomass.

Welch, et al. (1974) attempted to predict the observed phosphorus content in Lake Sammamish. Gross compartment models were employed, which considered phosphorus input, outflow, and exchange with the sediments. Two layers were assumed during the period of stratification; at other times, the water column was completely mixed. Results of two model formulations that differed in their treatment of phosphorus exchange with the sediments were compared. The first model was based on a net loss of phosphorus to the sediments, proportional to the phosphorus concentration in the lake. The second model considered phosphorus sedimentation and release rates separately. Both formulations utilized a phosphorus stratification adjustment factor in place of the effective flushing volume to estimate the actual amount of phosphorus leaving the lake. Annual steady-state values for phosphorus sedimentation and sediment release were estimated from the nutrient budget: sedimentation = 2.7 g/m^2, release = 2.2 g/m^2, retention = 0.5 g/m^2.

The models were used to predict the response of a 30 percent step reduction in phosphorus income. The first model predicted a sizable reduction in phosphorus concentration, while the second model predicted a rapid, but slight, change, which more closely resembles the observed response on Lake Sammamish. The assumption in the second model of a constant sediment release rate, and the fact that the rate is largely due to the anaerobic nature of the lake, had a significant bearing on the results. For Lake Sammamish, the predictions support the importance of sediment exchange as a factor in phosphorus content, in spite of moderate reductions in phosphorus income (Welch, et al., 1974).

A general trophic analysis presented by Imboden (1974) was described in the previous section on empirical models. The general analysis was developed with a two-compartment mass balance model for phosphorus. The system considered an epliminon and hypoliminion, with compartments for particulate and dissolved phosphorus. The model was based on a system of four equations involving the following parameters: dissolved and particulate phosphorus concentrations, hydraulic loading, phosphorus loading, exchange coefficient between the layers, velocity of sedimentation, rate coefficients of photosynthesis and mineralization, and the exchange of phosphorus at the sediment-water interface. Sets of model coefficients were established for the summer stratification period and for winter circulation conditions.

A predictive model for phosphorus in lakes was developed and verified by Snodgrass and O'Melia (1975). The model simulated summer stratification with compartments for a distinct epilimnion and hypolimnion, and homothermal winter conditions with a single compartment. Each compartment was treated as completely mixed to negate horizontal concentration gradients, and the hypolimnion was assumed aerobic at all times. Data required for predictions included phosphorus loading, hydraulic loading, and mean depth. Phosphorus release from the sediments was not considered. Removal of phosphorus from the water column was represented as a first-order sedimentation process occurring over the sediment-water interface at the lake bottom. Both soluble orthophosphate and particulate phosphorus were included in the modeling analysis.

The authors investigated an observed relationship between predicted phosphorus concentrations and mean depth: a tendency for phosphorus concentrations to increase with increasing lake depth in both the epilimnion and hypolimnion (reported by other researchers as well). Emphasis was placed upon possible mechanisms for transporting particulate phosphorus to the lake sediments. First, a depth-dependent exchange of phosphorus across the thermocline was developed. Empirical studies from several sources were reviewed, which evidenced increased eddy diffusion, or vertical turbulent exchange, with increased mean depth. The stability across the thermocline was lower for deep lakes than shallow ones. In effect, this reaction removes particulate phosphorus from the epilimnion to the hypolimnion at a faster rate in deep lakes than in shallow ones. The second removal mechanism considered was natural aggregation or coagulation. Model results indicated the importance of flocculation in phosphorus removal.

Results of the modeling effort demonstrated the model's utility for predicting the temporal response of a lake to sudden changes in nutrient inputs. A sensitivity analysis indicated that predictions are most sensitive to, in decreasing order, the input loading rate of phosphorus, the settling coefficients, and the decomposition coefficients (particulate phosphorus removed by decomposition to soluble orthophosphate). Sensitivity was also noted to vertical exchange across the thermocline. Priorities for cost-effective limnological studies of a general nature were suggested by the results of Snodgrass and O'Melia (1975). Emphasis should be placed on nutrient inputs, settling of the lake sediments, and decomposition of organic materials when investigating total phosphorus relationships and concentrations.

Lorenzen (1974) reviewed the formulation of three models to demonstrate the effects of different assumptions for the nutrient exchange process upon predictions of lake recovery. The three models all employed idealized assumptions of complete mixing, inflow and outflow of nutrients, and average annual constituent values. The simplest formulation was Model I, in which a net annual loss of nutrients to the sediments was assumed, the loss proportional to the average nutrient concentration in the water column (this is essentially Vollenweider's framework). Model II treated nutrient fluxes into and out of the sediments as independent processes, with the assumption that the concentration in the sediments remained fairly constant over time. In Model III, depletion of nutrients in the sediments was considered, that is, the sediment concentration was variable. A coupled set of differential equations describing the change in nutrient concentrations in the water column and in the sediments was required. Comparison of model predictions under four sets of hypothetical lake conditions yielded some very different results. The results illustrated the effect of initial lake conditions which are not at a steady state with nutrient loading rates. In such a case, it is possible to reduce nutrient loads, and still show an increase in the average nutrient concentration in the lake.

2.2.2.2 Biochemical Models of Lower Trophic Levels

A more complex category of mathematical models for simulation of biological-chemical behavior has been developed in order to relate nutrient concentrations to phytoplankton concentrations. The models generally require more extensive parameter specifications and input data than do the chemical models described in the preceding section.

Schnoor and O'Connor (1980) utilized simplifying assumptions for kinetics and transport equations to develop an alternative nutrient-phytoplankton modeling approach. The model considers completely mixed systems of one or two compartments (i.e., an epilimnion and hypolimnion). Both inorganic and organic nutrient fractions are analyzed, and phytoplankton are simulated on the basis of nutrient concentration. Kinetic coefficients are required for sedimentation, hydrolysis (mineralization of organic nutrient to inorganic nutrient), phytoplankton uptake of inorganic nutrient (autocatalytic growth), and phytoplankton death rate (sum of the endogenous decay and predation rates). The authors concluded that the total phosphorus concentration in a lake was controlled by the sedimentation rate, which controls phosphorus loss to the sediments. This sedimentation rate represents the effective rate of organic nutrient and phytoplankton sinking velocities with respect to the mean depth of the waterbody. Remaining kinetic coefficients for hydrolysis, growth, and death of phytoplankton determine the partitioning of nutrients among the organic, inorganic, and phytoplankton fractions. The model was calibrated and verified with data sets for Lake Lyndon B. Johnson, Texas and simulations were performed for Lake Ontario of the Great Lakes. Schnoor and O'Connor (1980) identified several advantages of their simplified eutrophication model over the genera of lake process models based on nutrient loading typified by the Vollenweider approach: the capability of separating the effects of organic nutrient inputs from inorganic inputs (bioavailability), the calculation of nonsteady-state blooms and periodicities, the direct determination of phytoplankton standing crop, and a better understanding of the nutrient flux rates via phytoplankton growth and death, organic nutrient hydrolysis, and sedimentation.

Schnoor and O'Connor (1980) discussed the application of the steady-state solution of their model using phosphorus data for 81 northeastern and north-central United States lakes collected in conjunction with the USEPA National Eutrophication Survey. Of the total phosphorus present, phytoplankton accounted for 10-40 percent, dissolved phosphorus 35-75 percent, and organic phosphorus 0-40 percent. A general pattern of increasing eutrophic tendency with increasing percentage of organic phosphorus was evidenced. Predicted steady-state concentrations of total phosphorus were converted to chlorophyll-a. Comparison to measured chlorophyll-a concentrations showed considerable scatter. This analysis pointed out the need for a more detailed time-variable solution for water quality management applications (Schnoor and O'Connor, 1980).

Modeling techniques for the simulation of algal growth in eutrophic environments were described by Bierman, et al. (1974). Cell growth was treated as a two-step mechanism involving separate nutrient uptake and cell synthesis processes with an intermediate nutrient storage capacity. An algal growth curve was described from an example calculation for the case of a single algal species with phosphorus as the primary growth regulating nutrient. Model results indicated a lag between the decrease in phosphorus concentration in the water column and a subsequent increase in algal growth, an effect attributable largely to luxury uptake of phosphorus. Under these conditions, cell growth rate was independent of the external phosphorus concentration. A phosphorus limited system with two algal species was simulated to demonstrate the effects of species competition. An application was also made to a system limited by both phosphorus and nitrogen.

Algal growth dynamics in Shagawa Lake, Minnesota were modeled by Larsen, et al. (1974). The available data base for the lake included one year of detailed measurements of available phosphorus (orthophosphate phosphorus), available nitrogen (sum of ammonia, nitrite, and nitrate), and algal biomass (as chlorophyll-a). The model formulation included input variables of light intensity, available phosphorus, available nitrogen, and flowrate. The contents of the lake were treated as completely mixed, and biomass was simulated as chlorophyll-a. Algal dynamics and nutrient concentrations were expressed as three coupled differential equations. Simulation results indicated a spring algal bloom (as observed), with the timing and magnitude of the bloom sensitive to the hypothesized loss coefficient for algal biomass. Simulation of the observed late summer algal bloom was accomplished only with the addition of nutrient supply from the sediments and nitrogen fixation.

Cordeiro, et al. (1974) described a two-layer lake model for simulation of seasonal microbial populations and nutrient concentrations. The model considered not only nutrient transport through the lake, but also nutrient cycling, i.e., the exchange of carbon dioxide, nitrogen, or oxygen with the atmosphere. Each layer included categories for organic and inorganic chemical species. Five mechanisms of nutrient transfer were included: convection, diffusion, sedimentation, chemical reaction, and biological reaction. Growth kinetics for three general algal groups and three groups of bacteria were simulated.

An aquatic ecosystem model for simulation of phytoplankton and nutrients was developed by Lassiter and Kearns (1974). The chemical system included organic and inorganic components of carbon, nitrogen, and phosphorus. Six species of phytoplankton were included, with growth rates a function of inorganic chemical concentration and light intensity. Simulation results for a hypothetical system indicated the importance of rates of secretion and decomposition of organic materials in the timing of algal growth. It was postulated that fast-growing algal species may dominate the early growing season, while slower-growing forms with the capability to use lower nutrient concentrations may dominate late-season growth.

2.2.2.3 Comprehensive Ecosystem Models

Ecosystem models represent a category of complex mathematical models. The increased complexity is generally a consequence of the incorporation of a smaller computational grid (i.e., smaller and more numerous completely mixed cells) or additional variables.

Patten (1974) reported that eutrophication studies typically address only the primary production subsystem of lake ecosystems. Such models consider nutrient loadings and photosynthetic energy inputs as exogenous drives, and incorporate internal processes to deal with algal nutrition, nutrient dynamics, and hydrologic transport, ignoring the remainder of the lacustrine ecosystem. Patten (1974) argued that the appropriate minimal conceptual unit for eutrophication modeling is the whole ecosystem. Since ecosystem components are mutually dependent, points for control of eutrophication exist throughout the ecosystem. Further, control measures directed at sensitive points in the ecosystem may be amplified in propagation to produce significant results.

A complex model for simulation of water quality in a river-reservoir system was developed for the Hydrologic Engineering Center (HEC) of the U.S. Corps of Engineers (HEC, 1975). The model was essentially a combination of previously developed reservoir and stream models. The stream model simulated one-dimensional transport through fully mixed longitudinal segments. The reservoir model was based on an ecological model for deep reservoirs developed for the Office of Water Resources Research (OWWR) of the U.S. Department of Interior (Chen and Orlob, 1972), wherein a stratified reservoir is simulated as a series of one-dimensional, fully mixed horizontal slices. Various criteria are incorporated into the model hydrodynamics to govern the vertical distribution of inflows and outlet releases. Continuity of flow is maintained through vertical advection and diffusion between elements. The following parameters can be simulated: temperature, biochemical oxygen demand, three types of fish, benthos, zooplankton, two types of algae, detritus, organic sediment, orthophosphate phosphorus, ammonia nitrogen, nitrite nitrogen, nitrate nitrogen, dissolved oxygen, coliform bacteria, total alkalinity, total dissolved solids, and pH. A schematic representation of the interdependence of these constituents is displayed in Fig. 2-10. A detailed set of physical, chemical, and biological coefficients are required for implementation of the model. A typical set of coefficient values and ranges is presented in Table 2-1.

A phytoplankton model for Lake Ontario was developed by Thomann, et al. (1975). Two completely mixed layers (epilimnion and hypolimnion) were considered, with ten biological or chemical compartments in each. Included were four trophic levels above the phytoplankton: herbivorous zooplankton, carnivorous zooplankton, and two upper trophic levels. Chemical compartments included organic and inorganic phosphorus; organic, ammonia, and nitrate nitrogen; and chlorophyll-a as a measure of phytoplankton biomass. Model coefficients were calibrated with observed data for 1967-1970. The authors concluded that the model was sufficiently verified for prediction of timing and extent of phytoplankton blooms. The model was used to predict the long-term behavior of peak phytoplankton chlorophyll in Lake Ontario by Thomann, et al. (1977). Four alternate loading schemes were evaluated: present loads, historic loads, and a pair of reduced-load scenarios.

Robertson and Scavia (1979) developed an ecosystem model for the purpose of synthesizing existing knowledge of an ecosystem and providing insight into ecosystem structure and function. The model was a modification of an earlier model for Lake Ontario (Scavia et al., 1976). Two vertical layers were simulated with seasonal cycles of five groups of phytoplankton, five groups of herbivorous zooplankton, one group of carnivorous zooplankton, detritus, available phosphorus, dissolved organic nitrogen, ammonia nitrogen, nitrite plus nitrate nitrogen, soluble reactive silica, detrital silica, dissolved oxygen, and the carbonate system. External forcing functions included solar radiation, winds, allochthonous loads, and hydraulic flushing rates. Sinking of particulate organic matter to the sediment compartment was simulated, and transport of dissolved material across the sediment-water interface was treated as a diffusive process. Decomposers were modeled as first-order decay terms, as were trophic levels higher than secondary consumers (all carnivorous zooplankton). Model output depicted carbon flow between categories or trophic levels, which provided an overall perspective of the Lake Ontario ecosystem. Model results defined operation as a detrital- or producer-based food web, major trophic paths and relations, relative abundance of each trophic level, and other properties.

Huff, et al. (1974) employed a detailed ecosystem model to simulate the pelagic zone of Lake Wingra in Madison, Wisconsin. The model was comprised of nine compartments representing the open water portion of the lake, including phytoplankton, zooplankton, fish, benthos, and chemicals. Simulation of runoff was accomplished with the Hydrologic Transport Model developed earlier by the same author. A detailed water and nutrient budget indicated that 84 percent of the dissolved inorganic phosphorus input entered from storm sewers, while 65 percent of the dissolved inorganic nitrogen orginated from springs and groundwater seepage. The difficulty in obtaining parameter estimates for detailed ecosystem models was discussed. Model results indicated general agreement with trends observed in the real system.

Development of the comprehensive ecosystem model CLEAN (Comprehensive Lake Ecosystem ANalyzer) was described by Bloomfield, et al. (1974). The model was formulated as 28 coupled ordinary differential equations. Major components included producers, consumers, decomposition and organic matter, hydrologic balance, and lake circulation. The producers component included macrophyte and phytoplankton groups, and consumers included three types of zooplankton, three types of fish, and benthic insect larvae. The authors pointed out that the accuracy of prediction had not been demonstrated to be sufficient for use as a management tool, particularly in light of the costs of running the model. Results of the model should, however, provide valuable insight into ecosystem functions.

Modifications to CLEAN resulted in the model CLEANER. Improvements included the addition of several state variables, reformulation of the decomposition submodels, and addition of a subroutine for transformation of biomass values into environmental perception characteristics. The model was further expanded into the version MS. CLEANER. Modifications included additional state variables, enhanced process resolution, and the capacity for simulation of as many as ten vertical and horizontal segments simultaneously, as reported by Park, et al. (1979). The investigators reported that continued model development was limited by the state of knowledge in aquatic ecology, and identified three levels of research needed: (1) process-level research to aid in formulation of environmental responses; (2) detailed measurements for calibration of existing formulations; and (3) comprehensive data collection programs to support verification of models.

Briefly, additional examples of complex ecological models include a food-web model for Lake Michigan developed by Canale et al. (1976). The model was based upon forage-fish predation, and simulated the epilimnion and hypolimnion with compartments for three elements, four types of phytoplankton, seven types of zooplankton, and the alewife. The food-web model was developed in order to evaluate complex ecological problems, such as the effect of eutrophication on the forage fish. Chen et al. (1975) developed a comprehensive water quality-ecological model for Lake Ontario. The model included seven vertical layers with numerous horizontal elements.

Major compartments included nitrogen, phosphorus, silicon, algae, zooplankton, detritus, fish, and organic sediment. Subdivisions for three age groups and four major types were included within the fish compartment. Benndorf and Recknagel (1982) developed the dynamic ecological model SALMO (Simulation by means of an Analytical Lake MOdel). The model was constructed for simplicity with respect to the number of state variables, but maximum complexity regarding ecological control mechanisms was instituted. Only three state variables were considered: phytoplankton, zooplankton, and orthophosphate. Internal control mechanisms included relationships of photosynthesis and phytoplankton respiration to temperature, sedimentation of particulate phosphorus in zooplankton fecal pellets, nutrient remineralization by zooplankton and fish, zooplankton feeding on phytoplankton, fish feeding on zooplankton, and several additional processes.

2.2.3 Other Modeling Techniques

An alternative approach to water quality modeling involves the development of physical models in controlled environments. Falco and Sanders (1974) described an environmental chamber called the Aquatic Ecosystem Simulator. The chamber was designed to simulate biological and chemical responses of flowing streams to environmental perturbations. Information could be obtained on microbial growth kinetics, transport phenomena, and photosynthetic processes. The environmental chamber was 22 m long and 3.66 m wide, housing a channel 19.5 m long, 0.46 m wide, and 0.6 m deep. Rotating paddle wheels regulated mixing, and operational control parameters included air velocity and temperature, relative humidity, influent water temperature, and influent water flowrate. A variety of constituents could be monitored, including organics, nutrients, and dissolved oxygen. A computer provided set points for environmental parameters and served as the data acquisition system.

Hill and Porcella (1974) described the technique of component description and systems analysis as an alternative to compartment models. Effects of potentials or forces and environmental interactions were extracted from the descriptive parameters which constitute a compartment. The authors suggested that inclusion of potentials in models would provide additional realism by (1) differentiating between potential energy storage, kinetic energy storage, and dissipation within a compartment, (2) representing intercompartmental environmental flow resistance, and (3) permitting extrapolation of descriptive parameters to different environments. The method is detailed, and a major disadvantage is the complexity of the mathematics.

2.3 ANCILLARY ASPECTS OF PHOSPHORUS MODELING

Modeling of the eutrophication process requires a fundamental comprehension of several aspects of water quality management. External loads must be quantified within a suitable framework, and internal processes must be examined for assignment of appropriate reaction kinetics. The latter aspect may be of particular importance in model formulation in order to effect certain simplifying assumptions. The success of a reservoir modeling effort is dependent upon representative estimation of external and internal processes. Mass budget analyses are commonly employed for evaluation of loads and reaction kinetics.

A detailed evaluation of sources of chloride loadings to the Great Lakes was reported by O'Connor and Mueller (1970) in conjunction with their mathematical modeling analysis. Major sources included industrial and municipal wastewater, inflow from upstream lakes, and residues from highway de-icing. Utilizing data from the period 1850 to 1900, natural background concentrations in the tributary rivers were calculated to range from 3 to 4 mg/l. Natural lake concentrations of 2 to 3 mg/l chloride were employed for equilibrium calculations. Chloride loadings from municipal sources were estimated at 9.1 kg (20 lb) per capita per year, equivalent to a concentration on the order of 40 to 80 mg/l in sewage. Population growth and the index of the value of chemical products derived from the U.S. Census of Manufacturers were utilized for assessment of industrial growth and thence chloride inputs. Municipal road salt use was estimated as a function of population. A mass balance for chlorides for each reservoir was conducted, with the addition of a category for unidentified sources in order to close the budget. Subsequent modeling of the Great Lakes indicated the relative influences of the components of waste discharges on in situ concentrations. Implications of the imposition of waste load controls were evaluated.

Long-term average nutrient loads were computed for Rend Lake in southern Illinois by Kothandaraman and Evans (1979). Mathematical relationships were developed for correlation of nutrient transport and discharge rate. These relationships were employed to estimate loading rates for total phosphorus and total inorganic nitrogen to the lake surface as 1.29 $g/m^2/yr$ and 3.79 $g/m^2/yr$, respectively. Using guidelines presented by Vollenweider (1968) of 0.13 $g/m^2/yr$ for phosphorus and 2.0 $g/m^2/yr$ for nitrogen as eutrophic levels for a mean depth of 5 m, Rend Lake was determined to be receiving excessive nutrient loads.

Input and export of nutrients to Lake Corpus Christi, Texas, was analyzed by Miertschin and Jensen (1979). Historical data for the Nueces River, the only significant tributary to the reservoir, were compiled, augmented with the results of a high flow data collection program, and load-discharge relationships were determined for selected constituents. Loading equations were integrated with time over the flow-frequency relationship for stations above and below the reservoir to compute long-term average nutrient loadings. Low flow and high flow loadings were also estimated. The calculations indicated that high flow mass transport comprises the major portion of the long-term annual nutrient loadings. Import loadings significantly exceeded export loadings from the reservoir, indicating that Lake Corpus Christi retains a substantial mass of nutrients, both fixed in standing crop and deposited by sedimentation. On an annual basis, inflow loadings of 1.55×10^6 kg/yr of total nitrogen and 2.2×10^5 kg/yr of total phosphorus were computed. The areal phosphorus loading rate to Lake Corpus Christi was estimated to be 2.5 $g/m^2/yr$.

Cooke, et al. (1974) conducted a detailed analysis of phosphorus loading, retention, and flushing on the Twin Lakes Watershed in Ohio. Phosphorus inputs from surface inflow, precipitation, and shallow and deep groundwater flow were monitored and seasonal loads were computed. Surface sources provided the majority of phosphorus loads.

The cost-effectiveness aspects of phosphorus control strategies warrant consideration in the modeling of water quality management alternatives. The development of phosphorus control strategies should be based upon the relationships between the environmental problem, the contributing factors, and the costs (social, economic, technological) associated with the solution, according to Porcella, et al. (1974). A strategy for phosphorus control should be developed to decrease the effects of eutrophication at a minimum cost, in order to avoid implementation of solutions which might be counterproductive in terms of the total economy of a region or nation. The cost-effectiveness analysis commenced with the development of a relationship between phosphorus input and eutrophication. Relationships for admissible or dangerous nutrient levels as a function of nutrient loading and mean depth, based on Vollenweider (1968), were utilized by Porcella, et al. (1974). Index numbers were assigned to levels of relative eutrophication, and the eutrophication estimates were then recast in terms of chlorophyll-a concentrations and annual total phosphorus loading rate to allow definition of lake biota levels corresponding to admissible and dangerous levels of eutrophication. Porcella, et al. (1974) defined their activity analysis as a comprehensive understanding of phosphorus uses, which should be required before development of logical management schemes. A mass flow model for major phosphorus-using activities and natural sources was developed, so that phosphorus output could be calculated according to a variety of different activities. A detailed analysis of various activities was performed, with indentification of major application points for pertinent control tactics. A summary list of possible control tactics was presented (Pocella, et al., 1974), as shown in Table 2-2. Tactics were developed for the supply and demand side of activities, as well as for the technological and treatment side. The application of costs to each control tactic defined the cost-effectiveness relationship. The effectiveness of alternative controls was examined by simulating the application of specific controls, assessing the change in relative eutrophication level, then determining the associated treatment costs. A benefit/cost comparision should test whether the cost is worth the realized benefits, however, benefits from reduction of phosphorus cannot be assessed without the influence of value judgements.

2.4 RELATED STUDIES ON HIGHLAND LAKES

The Highland Lakes have been the subject of a number of studies over the past two decades, frequently under the auspices of the Center for Research in Water Resources of The University of Texas at Austin. No previous modeling activities for phosphorus concentrations in the series of reservoirs have been published.

Limnological and water quality data for the Highland Lakes were collected during 1968 and early 1969 and reported by Fruh and Davis (1969a). Field measurements of temperature, dissolved oxygen, pH, and light intensity were obtained.

Chemical analyses included alkalinity, hardness, conductivity, phosphorus, silica, nitrate plus nitrite, ammonia, iron, total organic carbon, and methylene blue extraction. Enumeration of bacteria and phytoplankton were conducted, and a limited amount of chlorophyll-a data were obtained.

Pittman, et al. (1969) investigated the effect of shoreline development on impoundment water quality. On Lake Lyndon B. Johnson, no significant differences in water quality were found among a developed cove (residential), an undeveloped cove, and the middle of the lake. However, analyses for total phosphorus showed a significant variance between bottom layer sampling stations within the undeveloped cove. Other areas on the Highland Lakes were sampled for an indication of effects of soil conditions, marinas, agriculture, and industrial activity. Shoreline development was occasionally found to impact water quality.

Fruh and Davis (1972) summarized several limnological investigations on the Highland Lakes. Effects of watershed development on water quality, hypolimnetic dissolved oxygen depletion, thermal prediction techniques, and limiting nutrient studies were described. Chemical and physical data for 1969 were presented.

The effect of urbanization on impoundment water quality was investigated by Fruh and Davis (1969b). A significant upward trend in the concentration of coliform bacteria in Town Lake (Colorado River at Austin) since 1949 was detected, as the watershed rapidly urbanized. Taste and odor problems in Town Lake and Lake Austin were also evaluated, and tentatively attributed to rooted aquatic vegetation and attached algae. Nutrient limitation--phosphorus, nitrogen or iron--was observed at various times of the year. Dissolved oxygen resources of Lake Travis, Lake Austin, and Town Lake were also analyzed.

Algal growth potential experiments were reported by Floyd, et al. (1969). Inorganic nitrogen, phosphorus, and iron were used for enrichment of unialgal cultures in the laboratory. Increased growth was observed with the combined addition of nitrogen and phosphorus. The algal growth potential, or fertility, of the Highland Lakes was shown to increase with progression downstream, indicating nutrient additions along the chain of reservoirs.

Huang, et al. (1973) reported on limiting nutrient studies in Lake Travis. Nitrogen, phosphorus, and iron concentrations were generally low, and each nutrient apparently limited phytoplankton growth at different times of the year. The laboratory tests indicated that phosphorus enrichment of Lake Travis would result in increased nitrogen fixation by blue-green algae. Batch laboratory enrichment tests employed both natural mixed populations and unialgal cultures.

A comprehensive study of Lake Lyndon B. Johnson was conducted in order to examine conditions before and after the installation of a power plant (Fruh, et al., 1977). The power plant draws water from a small cove for once-through cooling, then discharges the heated effluent to another cove, modified with an extensive dike to maximize heat dissipation within the cove rather than the main stem of the reservoir. Data were collected monthly from 1971 through May 1974 and presented in several progress reports, theses, and dissertations. Statistical analysis of data from control stations and stations in the discharge core indicated significant differences in temperature, dissolved oxygen, and total phosphorus. Nutrient balance calculations displayed large mass loadings associated with flood events. The calculations indicated that Lake LBJ serves as a sink for nutrients. Scour and resuspension of trapped nutrients could result if floods occur during the period of fall overturn.

The USEPA conducted water quality studies on Lake Travis, Lake LBJ, and Lake Buchanan in 1974 in conjunction with the National Eutrophication Survey. The EPA concluded that Lake Travis was mesotrophic. Algal assay results indicated that Lake Travis was phosphorus limited (USEPA, 1977a). Lake LBJ was classified as eutrophic, based upon phosphorus loading. Reduction of point source loads by 50 percent would reduce the total phosphorus load to a mesotrophic loading, according to the EPA. Phosphorus limitation was indicated by algal assays (USEPA, 1977b). Lake Buchanan was also classified as eutrophic. Essentially all of the annual phosphorus loading was attributed to the Colorado River. Algal assays indicated phosphorus limitation (USEPA, 1977c).

The TDWR conducted an intensive monitoring survey of Lake Travis in February 1975 (Brasier, 1976a). It was concluded that the reservoir was oligotrophic or mesotrophic, though the basis for this assessment was not presented. Algal growth potential studies implicated phosphorus as the limiting nutrient. A similar survey was conducted in Lake Austin in February 1976 by the TDWR (Ottmers, 1976). Both nitrogen and phosphorus were found to be limiting in algal growth potential experiments, with a suggestion that trace elements may also be limiting. An intensive monitoring survey of Lake Buchanan was also conducted by the TDWR in February 1976 (Brasier, 1976b). Total phosphorus levels were observed to be higher in the headwater reach than in the lower pool, attributable to inflow from the Colorado River. Algal assays indicated that addition of nitrogen alone or a combination of nitrogen and phosphorus would stimulate growth.

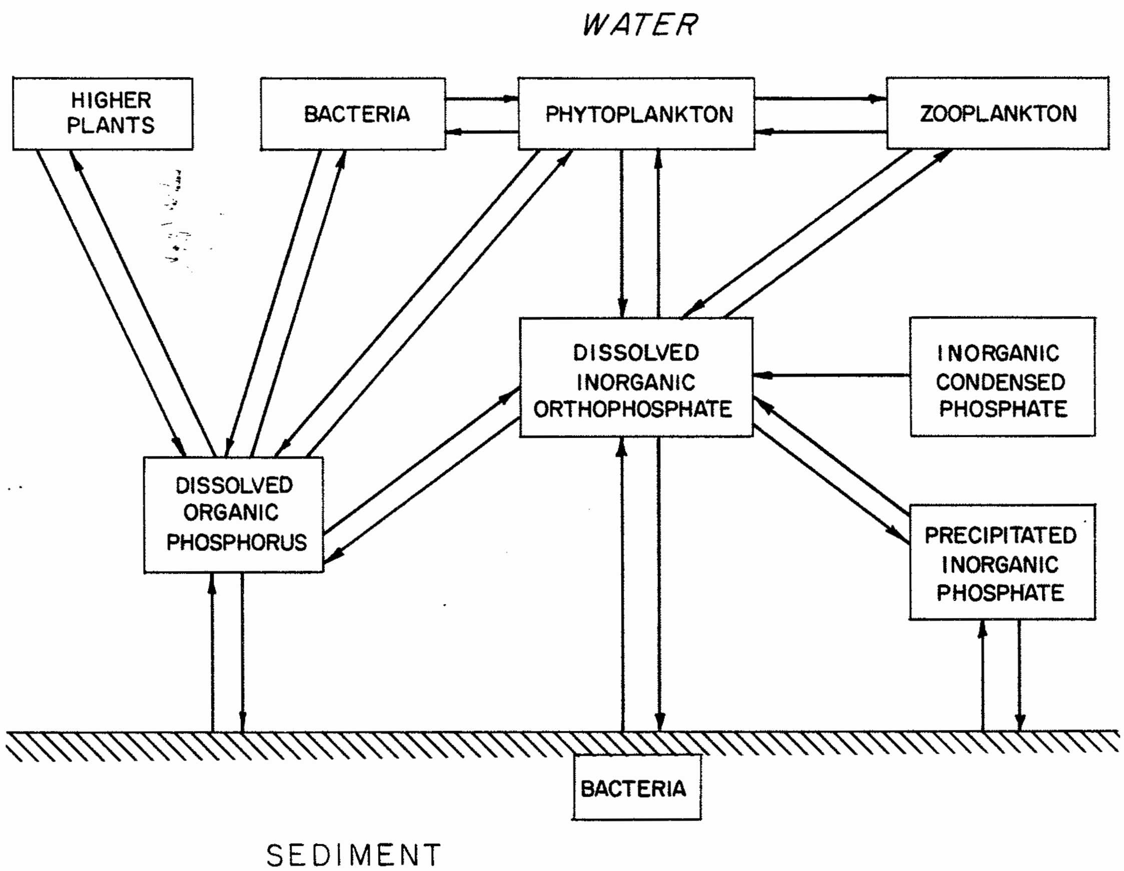


Fig. 2-1 PHOSPHORUS TRANSFORMATIONS IN WATER

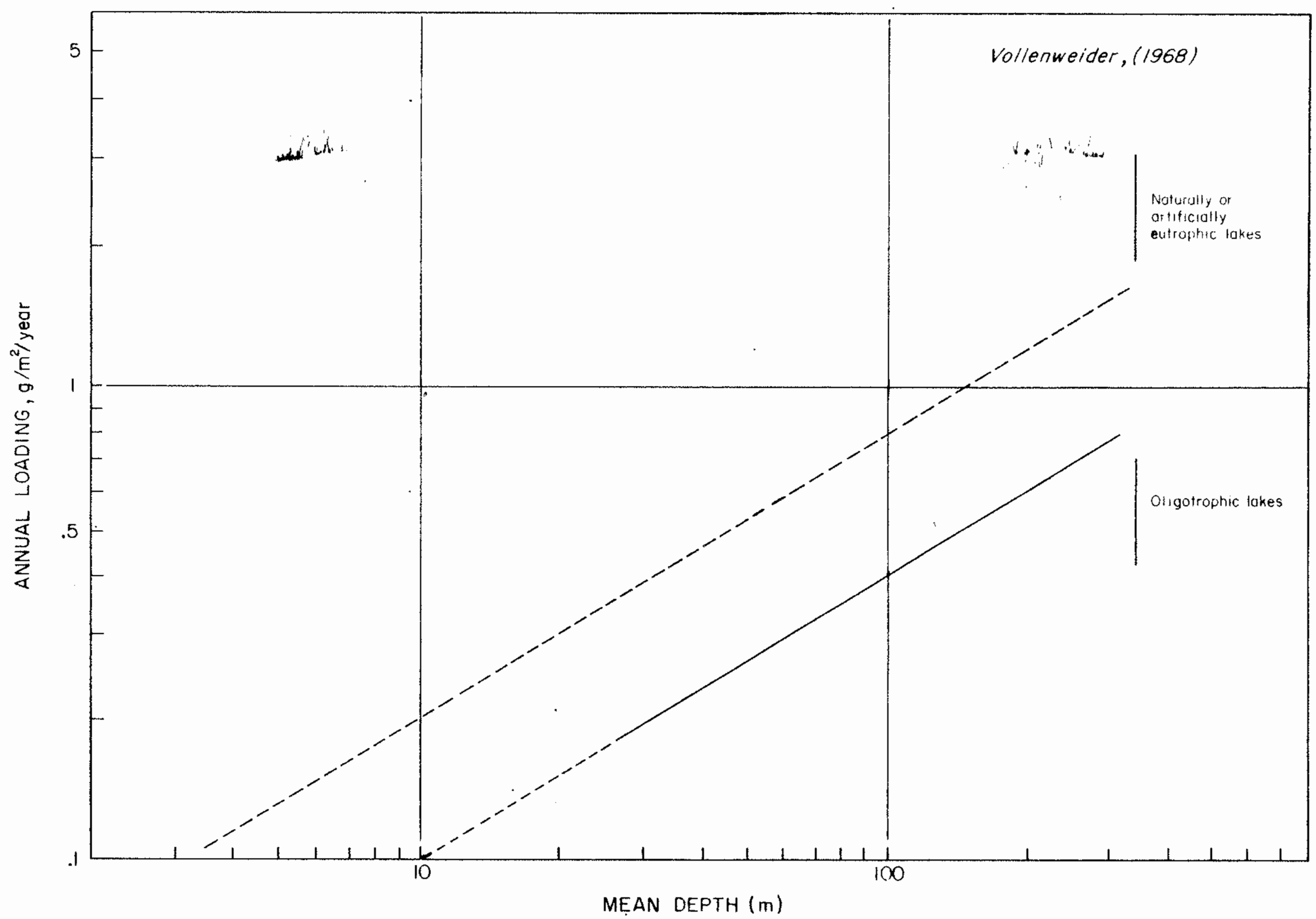


Fig. 2-2 VOLLENWEIDER PHOSPHORUS LOADING - MEAN DEPTH DIAGRAM

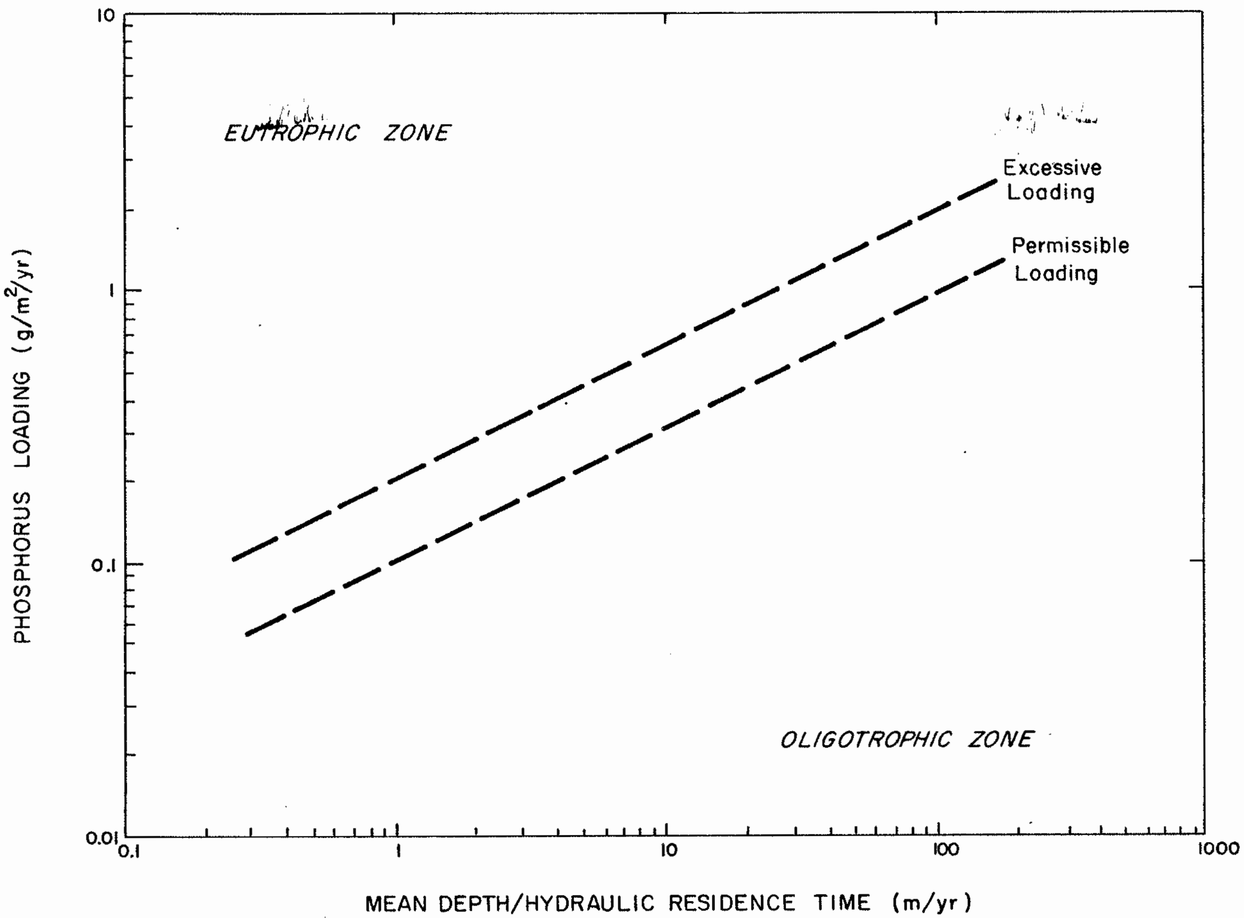


Fig. 2-3 VOLLENWEIDER PHOSPHORUS LOADING- HYDRAULIC LOADING DIAGRAM

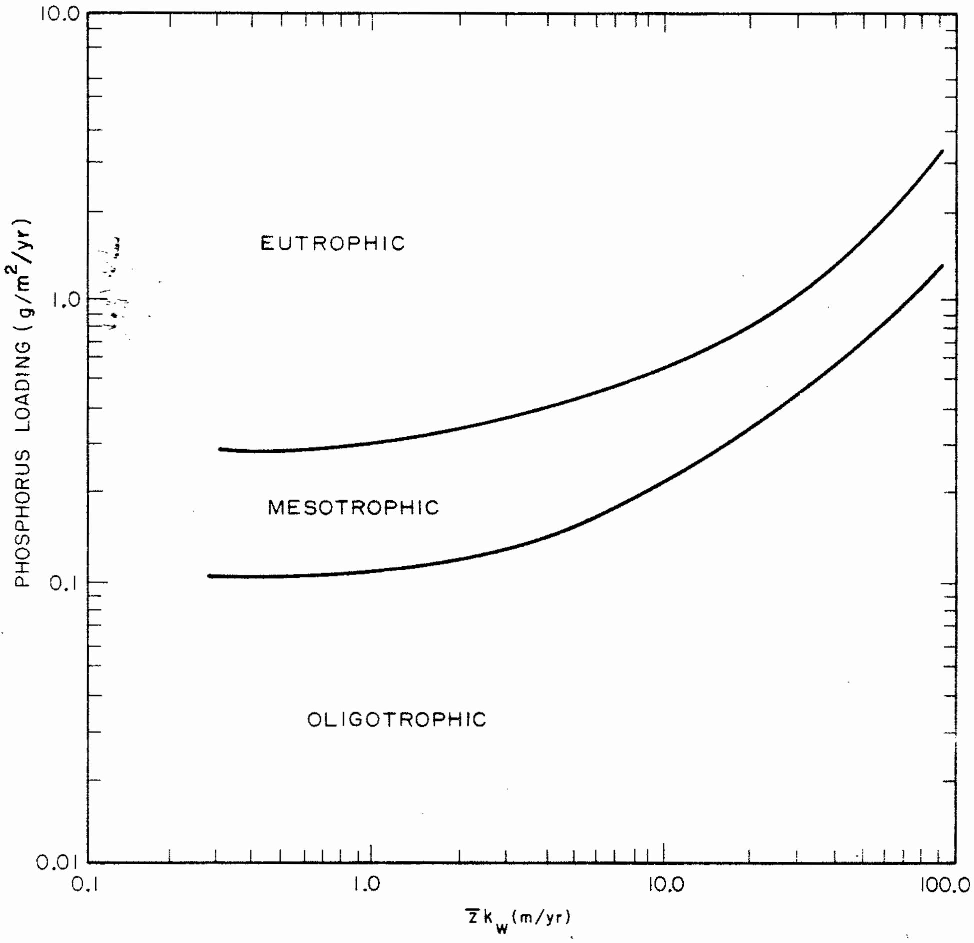


Fig. 2-4 VOLLENWEIDER MODIFIED PHOSPHORUS LOADING - HYDRAULIC LOADING DIAGRAM

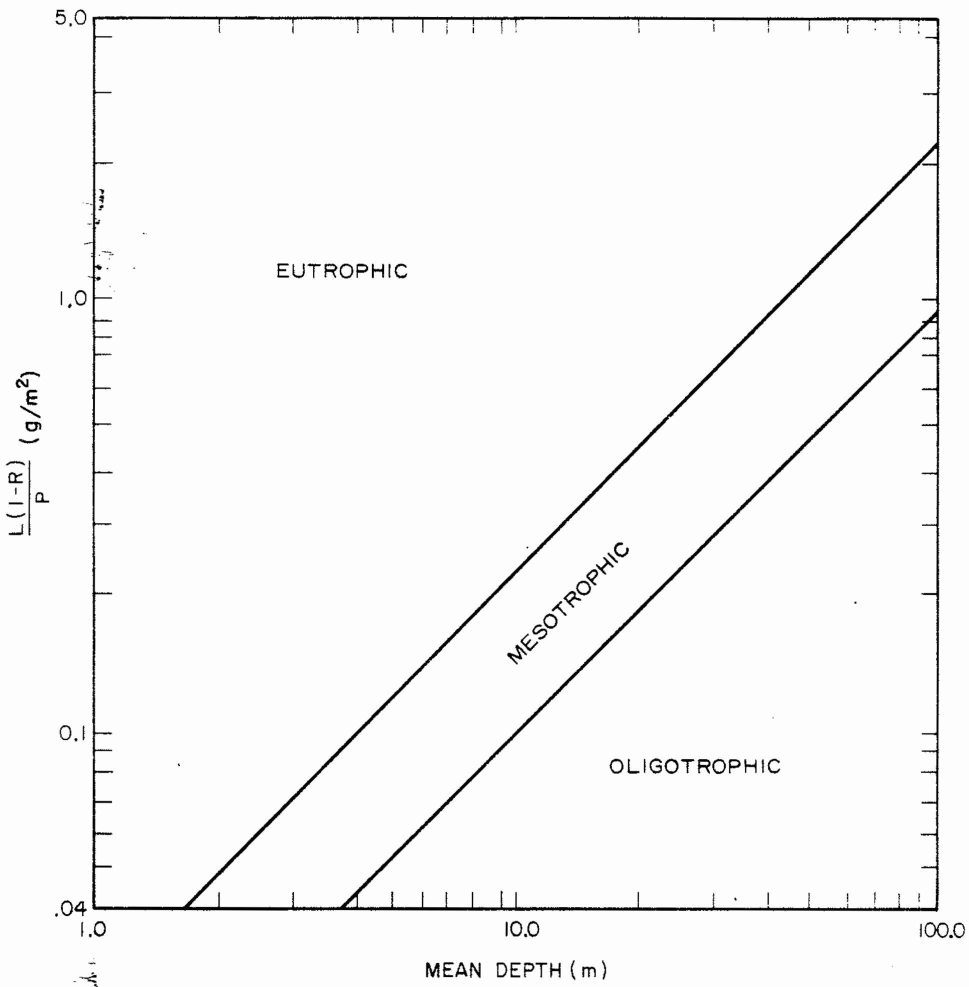


Fig. 2-5 DILLON LOADING PARAMETER - MEAN DEPTH DIAGRAM

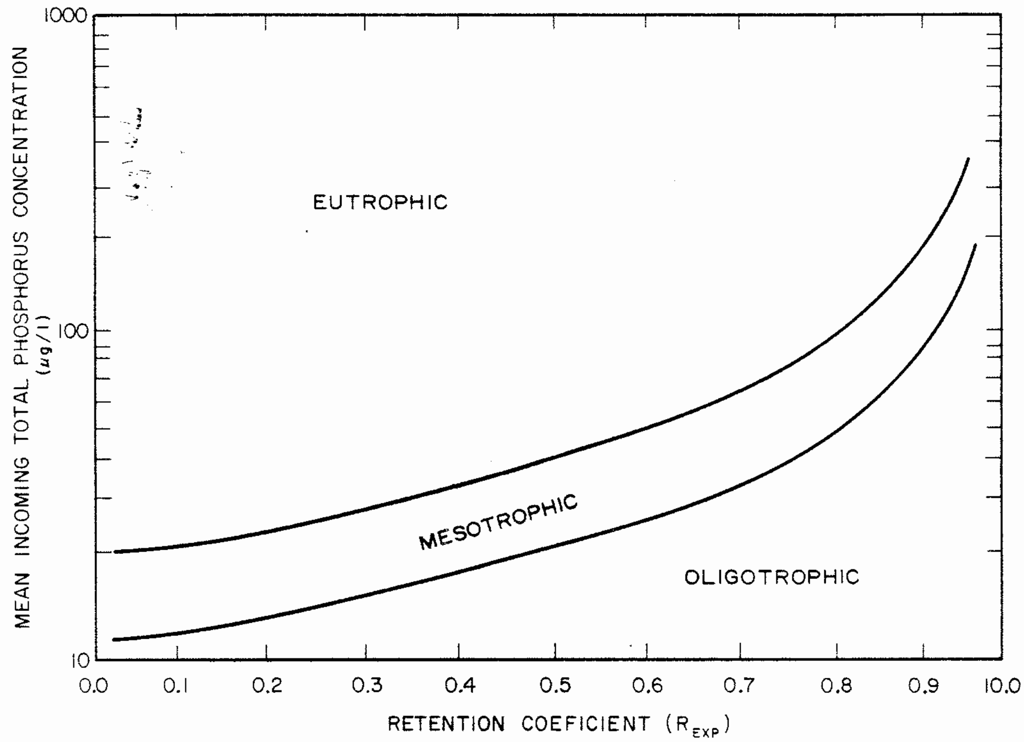


Fig. 2-6 LARSEN AND MERCIER CONCENTRATION RETENTION COEFFICIENT DIAGRAM

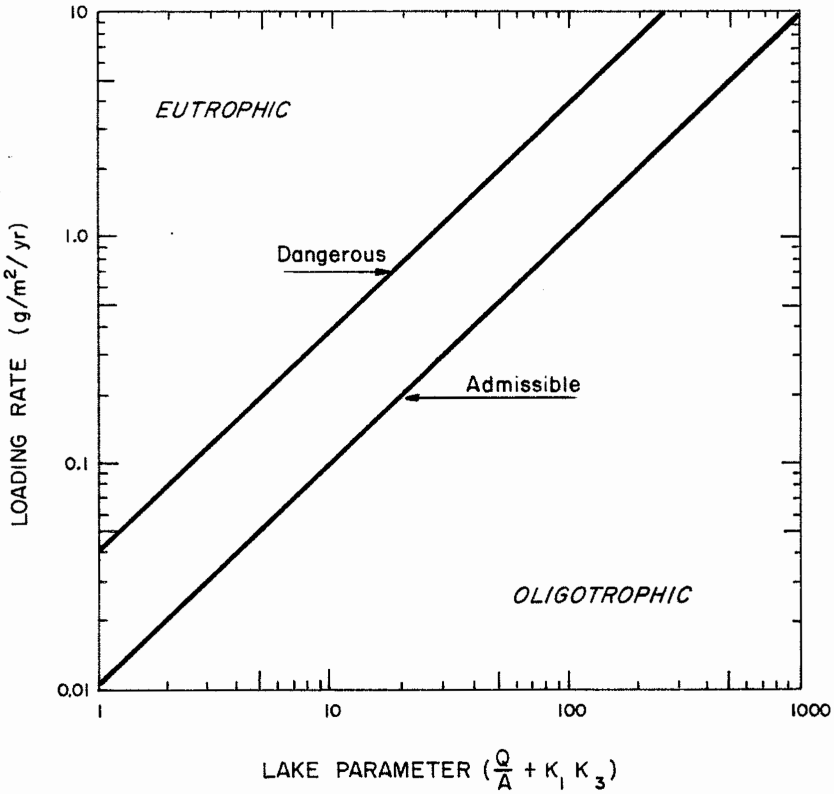


Fig. 2-7 BACA, ET. AL. LOADING RATE - LAKE PARAMETER DIAGRAM

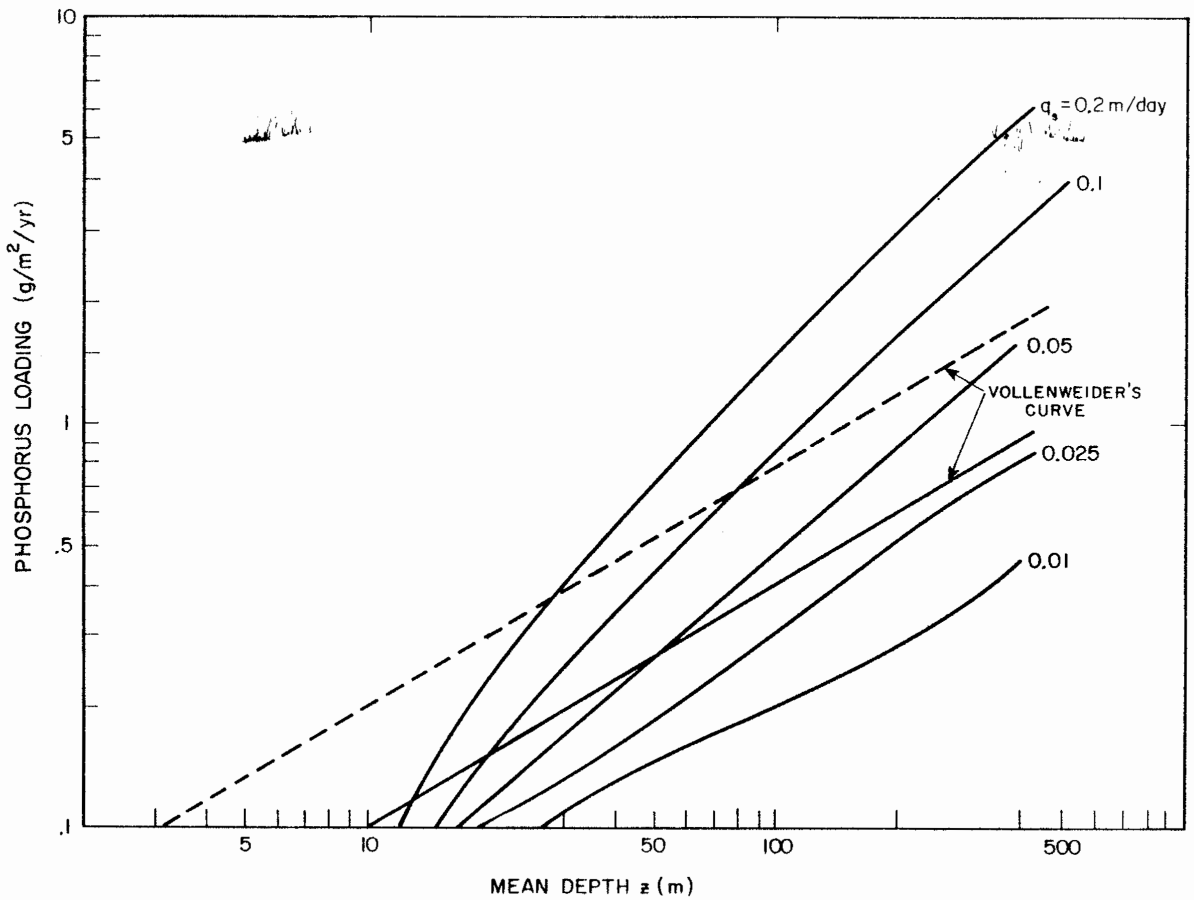


Fig. 2-8 IMBODEN LOADING - MEAN DEPTH - HYDRAULIC LOADING DIAGRAM

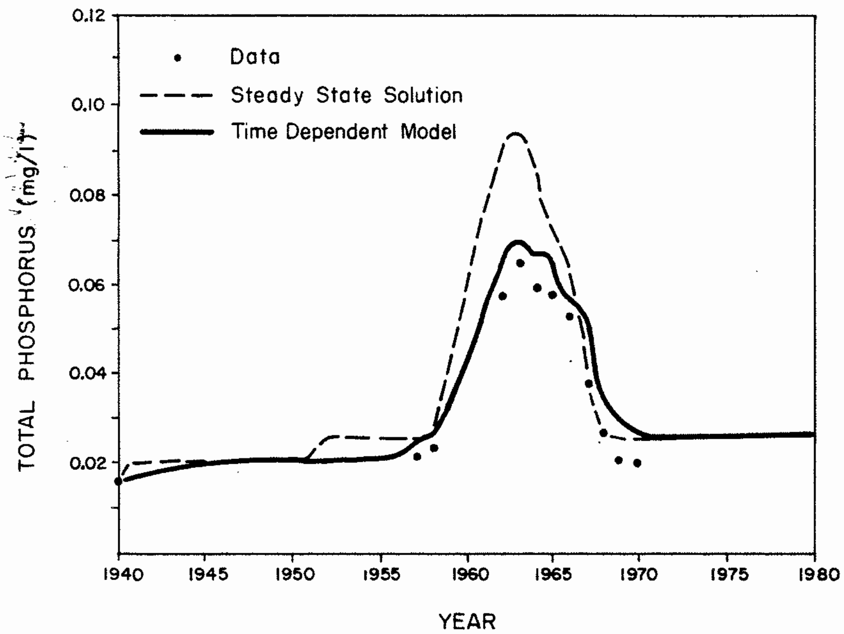
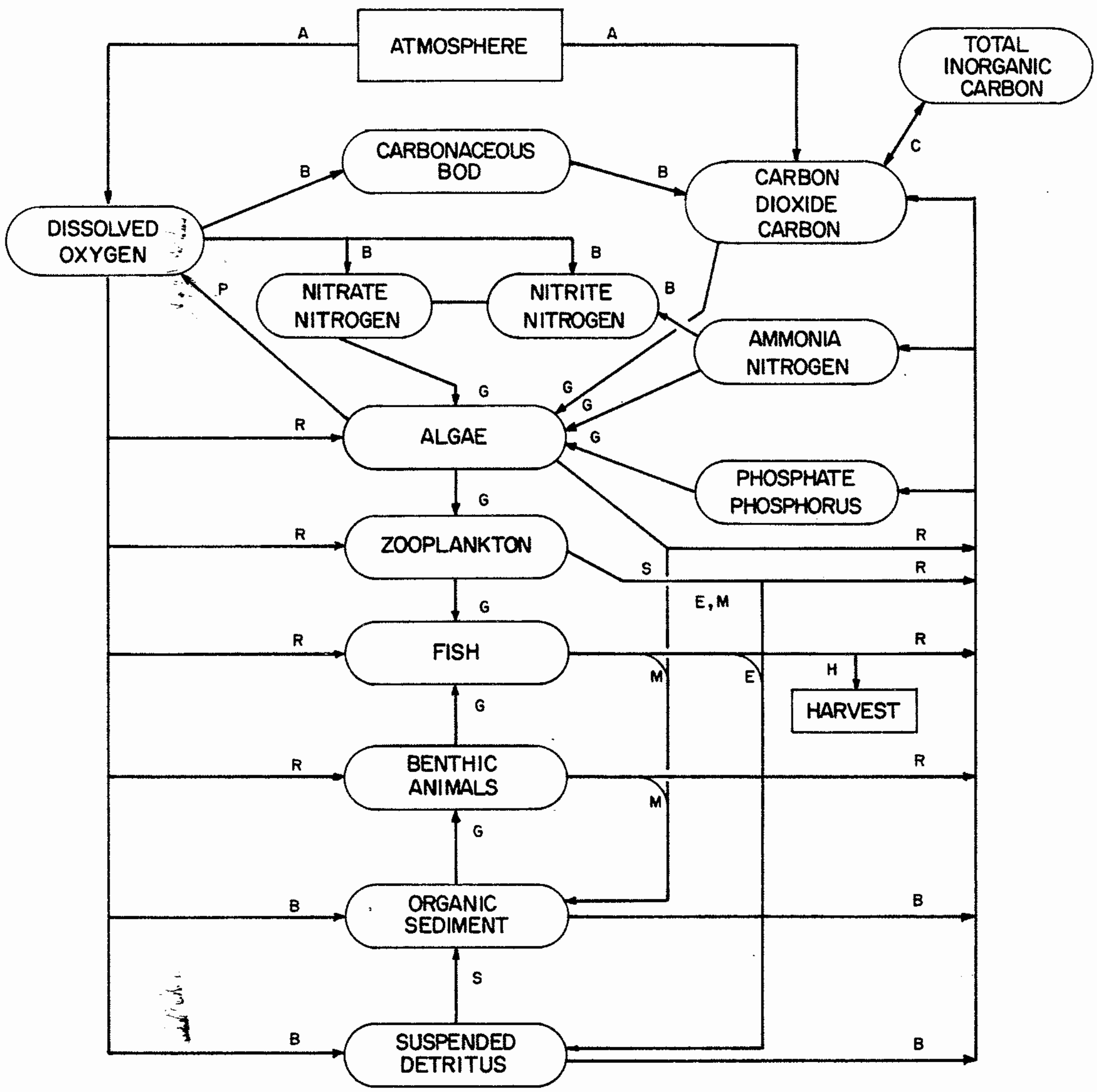


Fig. 2-9 CALCULATED AND OBSERVED ANNUAL AVERAGE TOTAL PHOSPHORUS LEVELS IN LAKE WASHINGTON



A Aeration
B Bacterial Decay
C Chemical Equilibrium
E Excreta
G Growth
M Mortality
P Photosynthesis
R Respiration
S Settling
H Harvest

Fig. 2-10 WATER QUALITY AND ECOLOGIC RELATIONSHIPS

TABLE 2-1

EXAMPLE OF REQUIRED COEFFICIENTS

Temperature coefficients			
BOD DECAY	1.047		
NH_3 DECAY	1.020		
NO_2 DECAY	1.020		
DETRITUS DECAY	1.040		
COLIFORM DIEOFF	1.040		
BIOTA ACTIVITY	1.020		
Chemical Composition of Biota and Detritus			
	C.	N.	P.
ALGAE	.4-.5	.07-.09	.01-.012
ZOOPLANKTON	.4-.5	.07-.09	.01-.012
FISH	.4-.5	.07-.09	.01-.012
BENTHOS	.4-.5	.07-.09	.01-.012
DETRITUS	.2-.5	.05-.09	.005-.0-12
Digestive Efficiency of Biota			
ZOOPLANKTON	.5-.8		
FISH	.4-.6		
BENTHOS	.4-.8		
Stoichiometric Equivalence of Chemical and Biologic Transformation			
O_2/NH_3 DECAY	3.5		
O_2/NO_2 DECAY	1.2		
O_2/DETRITUS DECAY	2.0		
O_2/BIOMASS RESPIRATION	2.0		
CO_2/BOD DECAY	0.2		
O_2/ALGAL GROWTH	1.6		

Temperature Tolerance Limits, °C		
	Lower Bound	Upper Bound
ZOOPLANKTON	5.0	30.0
ALGAE 1	5.0	25.0
ALGAE 2	10.0	30.0
FISH 1	5.0	20.0
FISH 2	10.0	30.0
FISH 3	5.0	30.0
BENTHOS	5.0	25.0
Settling Rate, m/day		
ALGAE 1	0-2.	
ALGAE 2	0-2.	
DETRITUS	0-2.	
Zooplankton Feeding Preference	2/1	
Mortality Rates/Day		
ZOOPLANKTON	.005	
FISH	.001-.005	
BENTHOS	.001	
Repiration Rates/Day	Active	Standard
PHYTOPLANKTON	.05-.2	.001-.01
ZOOPLANKTON	.02-.10	.001-.01
FISH	.001-.01	.0001-.001
BENTHOS	.001-.01	.0001-.001

TABLE 2-1 (Concluded)

Maximum Specific Growth Rates/Day

ALGAE 1	1.-2.
ALGAE 2	1.-3.
ZOOPLANKTON	.15-.3
FISH 1	.02-.03
FISH 2	.02-.03
FISH 3	.02-.03
BENTHOS	.02-.05

Half Saturation Constants of Phytoplankton

	Light	CO_2	N	P
ALGAE 1	.002-.004	.02-.03	.1-.2	.02-.05
ALGAE 2	.004-.006	.02-.03	.05-.1	.04-.08

Half Saturation Constants

ZOOP	graze on ALGAE	.4-.6
FISH 1	graze on ZOOP	.05-.1
FISH 2	graze on ZOOP	.05-.1
FISH 3	graze on BENTHOS	50.0-2000.
BENTHOS	graze on SEDMT	50.0-2000.

Decay Coefficients/Day

BOD	.1-1.5
Ammonia	.05-.2
Nitrite	.2-.5
Detritus	.001-.02
Coliform	.5-2.

TABLE 2-2

SUMMARY LISTING OF CONTROL TACTICS

A. Supply and demand (applies to consumer habits and producer activities)
   1. Subsidies (non-phosphorus products)
   2. Tax breaks and credits
   3. Price controls
   4. Excise taxes or other taxes
   5. Advertising and education
   6. Non-monetary recognition
   7. Content labeling
   8. Moral suasion
   9. Boycotts

B. Resource control, mining and manufacturing
   1. Requirements for recycling
   2. Phosphate mining restrictions (rationing)
   3. Manufacturing/production restrictions
   4. Emission controls

C. Management of phosphorus uses
   1. Resource and product substitution
   2. Technology improvements in processes or uses
   3. Monitor requirements with enforcement of application rates (e.g., fertilizer)
   4. Recycling and reclamation

D. Management of phosphorus discharges
   1. Pollution standards
   2. Land management practices
      a. Reduction of cultivated acreage
      b. Increased fertilizer use
      c. Technical management
      d. Irrigation practices
      e. Green belts and buffer zones
      f. Solid waste recycling
   3. Land use controls
      a. Zoning
      b. Licensing
      c. Leasing
      d. Codes and subdivision regulations
      e. Permits

TABLE 2-2 (Concluded)

4. Solid waste management
    a. Disposal regulation
    b. Fees
5. Effluent charges
6. Bans
7. Fines

E. Judicial controls
1. Judicial review
2. Class action
3. Common law remedies (nuisance trespass, negligence)

F. Wastewater treatment--for phosphorus removal

G. Lake modification

CHAPTER 3.0

ANALYSIS OF AVAILABLE PHOSPHORUS AND HYDROLOGIC DATA

Available data for total phosphorus and chlorides in the Highland Lakes were compiled and reviewed as part of the present analysis. The great majority of available data have been collected by the Texas Department of Water Resources. Additional entities involved in data collection have included the U.S. Geological Survey, the University of Texas Center for Research in Water Resources, and recently, the Lower Colorado River Authority. A substantial amount of data has been collected on Town Lake by the City of Austin. A listing of sampling stations and agencies participating in data collection is displayed in Table 3-1. The present investigation relied primarily upon the TDWR data base. USGS data and CRWR data were limited in availability, but were occasionally used. Much of the TDWR and USGS data were obtained from the files of the Texas Natural Resources Information System (TNRIS, 1983a). At the time these analyses were conducted, data from the LCRA were generally not yet available. Data collected by the City of Austin on Town Lake were occasionally used where appropriate.

The data base for the Highland Lakes was subjected to several types of detailed analyses designed to ascertain spatial or temporal trends in total phosphorus concentrations. Chlorides data were analyzed as a companion exercise to compare the behavior of a conservative constituent. Phosphorus data collected over short time spans throughout the series of reservoirs were analyzed for spatial trends. The data were presented as profiles of concentration from Town Lake through Lake Buchanan. The existence of temporal trends was also investigated. Data were plotted by month for evidence of seasonality, and consecutive sampling surveys were examined for seasonality and effects of external mass loadings and internal reactions. The analyses are described in detail in the following subsections. Also described is a limited analysis of nitrogen data, presented solely for reference purposes.

Additional analyses of phosphorus data are described in subsequent chapters where they are directly used. In Chapter 4, tributary phosphorus data are examined for statistical characteristics and relationships between streamflow rate and mass loading in the development of major tributary and stormwater runoff loadings to the series of reservoirs. Data for chlorides are subjected to a similar analysis in an examination of the chloride budgets of the impoundments in Chapter 5. Also in Chapter 5, historical annual constituent concentrations are described pursuant to model calibration exercises. These analyses are situated within their respective chapters since they are integral components of specific exercises.

3.1 SYSTEM HYDROLOGY

The Highland Lakes are owned and operated by the LCRA, with the exception of Lake Austin, which is owned by the City of Austin and operated by the LCRA. Town

Lake is owned and operated by the City of Austin, and is technically not one of the Highland Lakes, but is included in the chain for practical purposes. The hydraulics of the series of reservoirs incorporate the influences of tributary inflow, releases from upstream reservoirs, and reservoir volumes and detention times. Principal tributaries to the Highland Lakes include the Colorado River, Llano River, Sandy Creek, and the Pedernales River. Pertinent characteristics of the reservoirs are displayed in Table 3-2.

3.1.1 Reservoir Operations

In conjunction with the analysis of water quality data, the fundamental scheme for reservoir operations should be identified. Releases from each of the reservoirs are sporadic. Reservoir releases occur in response to three distinct needs (Birdwell, 1983):

1. Satisfaction of downstream water rights, e.g., water supply, irrigation requirements;
2. Influx of flood waters;
3. Emergency power requirements.

Commonly, the release schedule is dominated by satisfaction of irrigation needs for rice cultivation in the Texas coastal region. The volume of water released is dependent upon the immediate needs of the rice growers. With a typical rice growth cycle, irrigation needs begin to develop in March. Peak irrigation demands usually occur in June, followed by a slight decrease in demand and a subsequent secondary peak usage in August. The demand then declines with the cessation of rice agricultural activity in October. There is typically no rice irrigation demand from November to March of the following year. The volume released varies, but the monthly discharge is frequently in excess of $123.3 \times 10^6 m^3$ (100,000 ac-ft) during peak consumption months. The discharge rate itself fluctuates, but an average daily release rate of 85 m^3/s (3,000 cfs) from Lake Austin is representative. The daily release rate at the various dams may not be constant, rather, the rate is manipulated to peak coincident with peak electrical demand. The peak electrical demand typically occurs in the morning hours in winter months and in the late afternoon and evening hours in summer months.

Releases from Lake Buchanan must be sufficient to maintain constant water levels in several downstream reservoirs, in addition to other demand requirements. Buchanan Dam is outfitted with three hydropower generators for normal releases which can accommodate approximately 102 m^3/s (3600 cfs) at full head. Inks Dam has only one hydropower generator with a normal output of 68 m^3/s (2400 cfs). Thus, the "normal" release schedule from Lake Buchanan is approximately 68 m^3/s (2400 cfs), since such a rate can be passed through the generator at Inks Dam. If releases in excess of 68 m^3/s (2400 cfs) occurred, the water would simply flow over the Lake Inks spillway without the benefit of electrical power generation. Wirtz Dam on Lake LBJ has two hydropower generators with a maximum capacity of roughly 255 m^3/s (9000 cfs). The normal daily release from LBJ is roughly 85 m^3/s (3000 cfs), which is comprised of the 68 m^3/s (2400 cfs) release from upstream reservoirs supplemented by additional tributary inflow to the lake. The two hydropower generators at Max Starcke Dam on Lake Marble Falls normally pass approximately 85 m^3/s (3000 cfs), which represents the normal release from Lake LBJ upstream. Mansfield Dam on Lake Travis has three electrical generators with a maximum throughflow of 142 m^3/s (5000 cfs) at full head and a normal discharge of roughly 85 m^3/s (3000 cfs). The normal discharge through the two hydropower generators at Tom Miller Dam on Lake Austin is thence also 85 m^3/s (3000 cfs).

The preceding description is intended to provide a general indication of the normal release schedule throughout the Highland Lakes. Again, constant releases usually do not occur. The releases generally are executed to coincide with peak electrical demand, with the daily average amount of water released sufficient to meet downstream demands.

3.1.2 Hydraulic Retention Time

The hydraulic retention time varies markedly among the series of reservoirs, due principally to their disparate volumes. Residence times for Lakes Buchanan, Inks, LBJ, Marble Falls, Travis, Austin, and Town are displayed in Fig. 3-1 through 3-7, respectively. For each reservoir, the average annual retention times for the period 1968-1982 are presented. Average annual retention time was computed from the average storage and cumulative outflow for each calendar year, based upon reservoir operation data provided by the LCRA (1983). In addition, each graph depicts retention times based upon moving 3-month average storage and outflow values, using antecedent 3-month periods rather than centered periods. (Only annual retention times are displayed for Town Lake, since detailed records of releases are not available from the LCRA.) The 3-month retention times demonstrate much greater fluctuation and extreme values.

The longest average annual retention times are encountered in Lake Buchanan and Lake Travis, with ranges of 318-3,802 days (0.9-10.4 yrs) and 200-597 days (0.5-1.6 yrs), respectively. Lake LBJ is characterized by an intermediate range of retention time: 29-123 days (0.1-0.3 yr). Shorter annual retention times are displayed in Lakes Inks, Marble Falls, and Austin as a consequence of their relatively small volumes, with computed ranges of 6-60 days (0.02-0.16 yr), 2-7 days (0.005-0.02 yr), and 3-11 days (0.008- 0.03 yr), respectively. Town Lake, the smallest reservoir, exhibits a range of 0.6-2 days (0.002-0.005 yr).

Shown in Fig. 3-8 are cumulative annual residence times for the complete series of reservoirs, i.e., the time theoretically required for water to pass through the chain. Cumulative values were estimated by summation of individual reservoir retention times for each year in the period 1968-1982. Based upon this data set, the average cumulative retention time for the Highland Lakes is 3.7 yrs, with minimum and maximum values of 1.5 yrs and 12.6 yrs, respectively.

3.2 ANALYSIS OF SPATIAL TRENDS IN PHOSPHORUS CONCENTRATION

The available data base was examined for sampling surveys which monitored the series of reservoirs as a continuum over a relatively brief time span. Such data sets present an overall picture of water quality throughout the Highland Lakes at a particular point in time. Analysis of these surveys provides information on the interrelationships of phosphorus and chloride concentrations among the series of reservoirs. The distribution of phosphorus may be affected by thermal stratification within the reservoirs, which signals the importance of temperature and seasonality. Data interpretation should consider the hydrologic regimes attendant to the surveys. For example, surveys conducted under high-flow or low-flow regimes may indicate significant differences in reservoir water quality. In general, if stable inflow conditions are maintained for a sufficient antecedent period, the reservoirs may be assumed to be at a steady-state hydraulic condition. The existence of stable hydraulic conditions does not necessarily ensure that constituent concentrations have achieved a steady state. The length of the antecedent period necessary for establishment of steady-state conditions cannot be precisely defined at the present time due to the absence of sufficiently detailed temporal information. For each survey, antecedent streamflow data were compiled from USGS (1983a) records and presented to indicate the stability of external loadings.

3.2.1 Longitudinal Surveys Under Unstratified Conditions

To avoid potential complications of stratification effects on chemistry in the water column, surveys were selected for detailed spatial analysis to depict periods when the water column was relatively well mixed. The presence or absence of stratification in the series of reservoirs was inferred by direct analysis of the vertical structure of temperature, dissolved oxygen and conductivity in the lower pool of Lake Travis above Mansfield Dam at TDWR monitoring station 1404.01. With the criterion for homogeneity, the data sets subjected to detailed spatial analysis were typically characteristic of periods of cooler temperatures.

3.2.1.1 CRWR Data

Several reports have been published by the CRWR which contain limnological and water quality data for the Highland Lakes. Water quality sampling surveys on the series of reservoirs were routinely conducted during 1968 and 1969 (Fruh and Davis, 1969a). The network of sampling stations included the lower pool of each impoundment, reservoir releases, and major tributaries. Periodic surveys were also conducted on specific individual reservoirs for various research topics. Few data sets are available which indicate conditions throughout the series of reservoirs within a relatively brief time span. Only a single survey was selected for analysis in the present study, the survey of 27 January-1 February 1969, as described below.

CRWR data for total phosphorus were collected at several discrete depths at most sampling stations. Concentrations were typically extremely low throughout the water column. At the low concentrations encountered, the precision and accuracy of the testing procedure were relatively poor (see Fruh and Davis, 1969a). Thus, much of the vertical structure in phosphorus concentration displayed in the CRWR data may in fact be attributable to sampling and analytical error. For the present analysis, surface measurements of phosphorus concentration were employed, and were assumed to be representative of the vertical profile. Where concentrations of zero were reported in the original data set, the values have been replaced by the inferred detection limit of the analytical procedure, namely 0.001 mg/l. (If contemporary detection limits routinely achieved by commercial labortories were substituted instead, non-detectable values would be reported as less than 0.01 mg/l.)

Survey of 27 January-1 February 1969

Antecedent streamflow conditions for the survey of 27 January-1 February 1969 are displayed in Table 3-3. The survey was preceded by a lengthy period of stable inflow from area tributaries. Streamflow records for the Colorado River above Lake Buchanan at USGS Sta. 08147000 near San Saba, Texas indicate maintenance of steady flow conditions. In fact, a relatively stable flow regime was evident for several months prior to the survey. Steady flow conditions are indicated for the Llano River at USGS Sta. 08151500 at Llano, Texas. Sandy Creek near Kingsland displayed a small, steady streamflow rate, as measured at USGS Sta. 08152000. Similarly, a stable flow regime was monitored on the Pedernales River at USGS Sta. 08153500 near Johnson City, Texas. On only two days during the antecedent 30-day period prior to the survey did substantial releases from Mansfield Dam occur. In accordance with these releases, streamflow records of the Colorado River at USGS Sta. 08158000 at Austin, Texas indicate a low, stable flow regime on the order of 1.7-2.3 m^3/s (60-80 cfs).

The survey sampling results are displayed in Fig. 3-9. The observed total phosphorus concentration in the Colorado River upstream from Lake Buchanan was 0.03 mg/l. The concentration dropped below detectable limits within the pool of Lake Buchanan. Total phosphorus was below detectable limits in the lower pools of Inks Lake, Lake LBJ, Lake Marble Falls, and Lake Travis. Detectable levels were found in the release from Lake Buchanan and Lake LBJ. A total phosphorus concentration of 0.01 mg/l was also detected in Lake Travis at the confluence with the Pedernales River. A level of 0.03 mg/l total phosphorus was detected in the lower pool of Lake Austin. Phosphorus was elevated in the lower pool of Town Lake to 0.058 mg/l.

The CRWR data set did not include measurements for chloride concentration. An attempt to obtain a correlation between chloride concentration and conductivity using the TDWR data for Lake Travis was unsuccessful. The lack of correlation is attributable to the narrow range of relatively low chlorides encountered in the Highland Lakes, roughly 50-100 mg/l, and the presence of other ionizable dissolved substances which influence conductivity. For the purposes of this data survey, chloride levels can only be indirectly inferred from analysis of conductivity data.

Observed conductivity levels increased from Lake Buchanan to Inks Lake. Conductivity dropped substantially in Lake LBJ, and then remained fairly steady through Lake Marble Falls. A further decrease in conductivity was detected in the lower pool of Lake Travis. Conductivity levels increased slightly in Lake Austin, and again in Town Lake.

3.2.1.2 TDWR Data

The TDWR collected numerous data sets from the Highland Lakes from 1973 to 1981, the period of record available at the outset of the present analysis (TNRIS, 1983a). The agency generally sampled the entire series of reservoirs over a period of 1-3 days. It was the practice of the TDWR to collect samples for analysis of total phosphorus and chlorides from only the surface layer in the reservoirs. Field parameters, such as temperature, dissolved oxygen and conductivity, were monitored at discrete depth intervals throughout the water column. Currently, the TDWR is scheduled to sample the reservoirs at 3-year intervals, with a quarterly sampling frequency every third year. Several data sets were selected from the TDWR data base for further analysis in the present study, as described below.

The analytical limit of detection for laboratory determination of total phosphorus reported by the TDWR was 0.01 mg/l. (Recall that CRWR data was reported to a level of 0.001 mg/l.) In the early years of the TDWR program, measurements of total phosphorus were reported as phosphate (PO_4), with a detection limit of 0.03 mg/l. The TDWR converted these data to be reported as phosphorus, resulting in a limit of detection of 0.0098 mg/l. For interpretation in the present analysis, 0.0098 mg/l and 0.01 mg/l were taken as equivalent limits. Unfortunately, until recently the TDWR did not consistently encode the "less than or equal to" sign in their computerized data base, making it difficult to surmise if data reported as 0.0098 mg/l were at or below that value. The omission was remedied by examination of the original laboratory data sheets. Much of the TDWR data is reported to four digits after the decimal point, an artifact of the conversion methodology. In the present analysis, data are presented as reported by the TDWR. For chlorides, the convention is to report concentration to the nearest milligram per liter, with an analytical detection limit of 1 mg/l.

Survey of 22-24 January 1974

Area streamflow regimes are displayed in Table 3-4. A stable streamflow is indicated for the Colorado River above Lake Buchanan at USGS Sta. 08147000. The observed flow regime is on the order of 5.7 m^3/s (200 cfs). Similarly, the Llano River at USGS Sta. 08151500 displayed a stable flow regime, again on the order of 5.7-8.5 m^3/s (200-300 cfs). At USGS Sta. 08152000 on Sandy Creek, flows were relatively steady during the antecedent period, typically less than 0.3 m^3/s (10 cfs). The Pedernales River exhibited a relatively stable flow regime near Johnson City at USGS Sta. 08153500 over the approximate range of 1.7-2.5 m^3/s (60-90 cfs). Streamflows in the Colorado River below Town Lake are also displayed in Table 3-4.

Concentrations of total phosphorus and chlorides generally decreased through the series of reservoirs as shown in Fig. 3-10. Concentrations of total phosphorus ranged from 0.0261 mg/l in the headwaters of Lake Buchanan to 0.0131 mg/l in the lower pool of Lake Austin. A corresponding decrease in chlorides level was also observed from 104 mg/l to 55 mg/l over the same reach. A substantial drop in chlorides concentration was indicated in Lakes LBJ and Marble Falls. However, phosphorus concentrations tended to increase in these same two reservoirs. The observed depression in chlorides level and increase in total phosphorus concentration may be an indication of the residual effects of an earlier high-flow period in October 1973, when a substantial influx of water was received from the Llano River, a major tributary of Lake LBJ. Chloride concentration dropped from 99 mg/l in Inks Lake to 28 mg/l in the lower pool of Lake LBJ, and remained relatively constant through Lake Marble Falls. From a concentration of 0.0098 mg/l in Inks Lake, total phosphorus increased through Lake LBJ and Lake Marble Falls up to a maximum value of 0.0261 mg/l in the lower pool of Lake Marble Falls. An observed increase in total phosphorus from 0.0131 mg/l to 0.0654 mg/l in the lower pool of Town Lake may be attributable to a local influx of stormwater from the immediate Austin watershed, as indicated by USGS streamflow data for Waller Creek.

Survey of 1-2 October 1974

The survey of 1-2 October 1974 demonstrates the response of the series of reservoirs to a period of high inflow. In Table 3-5 are displayed area streamflow data for an antecedent period of slightly more than one month. A substantial influx of water from the Colorado River above Lake Buchanan is indicated from the data at USGS Sta. 08147000. Two major storm flow hydrographs were observed, roughly two weeks apart. Lake Buchanan filled to capacity in late September, with subsequent overtopping of the gravity overflow spillway. Data for the Llano River at USGS Sta. 08151500 indicate a similar phenomenon. On the Llano, much higher discharge rates are indicated for the earlier storm hydrograph. The average daily discharge of 1209.3 m^3/s (42,700 cfs) recorded on 29 August 1974 was the highest average daily discharge recorded in Water Year 1974 at this station. Streamflow records for Sandy Creek Sta. 08152000 indicated the occurrence of only one significant storm flow hydrograph, occurring in late August 1974. Two inflow hydrographs were observed on the Pedernales River at USGS Sta. 08153500, with a major influx of stormwater indicated in late August 1974 and a subsequent smaller hydrograph recorded in late September 1974. Releases from Lake Travis, measured below Mansfield Dam and downstream at USGS Sta. 08158000 below Town Lake, were elevated above normal discharge rates, apparently in resonse to the inflow of a substantial volume of water into the upstream Highland Lakes.

As depicted in Fig. 3-11, concentrations of total phosphorus and chloride monitored during the survey of 1-2 October 1974 reflected nonsteady-state streamflow conditions. A total phosphorus level of 0.1928 mg/l was measured in the headwaters of Lake Buchanan. The concentration dropped substantially to 0.0098 mg/l in the lower pool of the reservoir. Phosphorus concentrations increased in Inks Lake, below Lake Buchanan. In Lake LBJ, phosphorus was elevated to 0.1176 mg/l in the headwater, downstream of the Llano River inflow, then decreased to 0.0196 mg/l in the lower pool of the reservoir. The influence of the Pedernales River inflow was evident in the upper reaches of Lake Travis, where a phosphorus concentration of 0.3235 mg/l was measured just upstream of the confluence with that major tributary. An extremely high phosphorus level of 1.0131 mg/l was detected in the upstream reach of Lake Travis. (The TDWR has recently elected to arbitrarily delete this data point from the historical record, with the rationalization that its magnitude suggests analytical or reporting error.) This level may be indicative of localized runoff conditions or may suggest a backwater influence from the Pedernales River confluence downstream. Phosphorus levels decreased with progression through Lake Travis, to a concentration of 0.0131 mg/l in the lower pool area. Concentrations were relatively uniform in Lake Austin, ranging from 0.0196 mg/l total phosphorus in the upper reach to 0.0131 mg/l near the dam.

Chloride concentration demonstrated a marked decrease in upper Lake Buchanan. A chloride level of 48 mg/l was measured in the headwater, while the lower pool area displayed a concentration of 99 mg/l. Chloride levels dropped to 60 mg/l in the headwaters of Lake LBJ, in response to the inflow from the Llano River. Concentrations of approximately 70 mg/l chloride were observed in the lower pool of Lake LBJ and throughout Lake Marble Falls. A substantial drop in chloride levels was detected in the upper reaches of Lake Travis, with a concentration of 25 mg/l encountered above the confluence with the Pedernales River. Concentrations increased with progression toward the lower end of Lake Travis, with a level of 43 mg/l chlorides measured near the dam. Chloride levels in Lake Austin were relatively stable at 44-45 mg/l.

Survey of 6-8 January 1975

Streamflow records for the antecedent period prior to the survey of 6-8 January 1975 are displayed in Table 3-6. Lake Buchanan was at or near capacity throughout much of the antecedent period, and overtopping of the gravity overflow spillway occurred in late December 1974. A relatively stable period of high inflow is indicated for the Colorado River at USGS Sta. 08147000 above Lake Buchanan. Stable high-flow regimes are also indicated for the Llano River at USGS Sta. 08151500, Sandy Creek at USGS Sta. 08152000, and the Pedernales River at USGS Sta. 08153500. Releases from Lake Travis were continual with the exception of a nine-day window of zero discharge in late December 1974. Below Austin at USGS Sta. 08158000, recorded streamflow rates correspond to the releases from Lake Travis with the addition of inflow from area tributaries.

Recorded levels of total phosphorus and chlorides for the survey of 6-8 January 1975 are displayed in Fig. 3-12. Concentrations of total phosphorus were relatively stable throughout the series of reservoirs. A concentration of 0.0131 mg/l total phosphorus was measured in the headwaters of Lake Buchanan and a concentration of 0.0163 mg/l was detected in the lower pool of the reservoir. The lower pools of Inks

Lake and Lake LBJ showed phosphorus concentrations of 0.0163 mg/l. Concentrations of 0.0131 mg/l total phosphorus were detected in the pools of Lake Marble Falls, Lake Travis and Lake Austin. Higher concentrations of phosphorus were encountered in Town Lake, with a level of 0.0359 mg/l in the headwater and 0.0229 mg/l in the lower pool.

Chloride concentration was relatively uniform in Lake Buchanan, dropping from 92 mg/l in the headwaters to 84 mg/l in the lower pool. A substantial decrease in chloride concentration was detected in Lake LBJ, with an observed decline from 85 mg/l in the headwaters to 63 mg/l in the lower pool above the dam. Chloride concentration dropped from 60 mg/l in the upper reaches of Lake Travis to 48 mg/l above Mansfield Dam. A relatively constant level of chlorides was observed throughout Lake Austin and Town Lake, with lower pool concentrations of 49 mg/l and 46 mg/l, respectively.

The data indicate relatively stable concentrations for the series of reservoirs over the survey of 6-8 January 1975. The concentration profile apparently reflects a period of continually elevated tributary streamflow rates.

<u>Survey of 8-9 October 1975</u>

In Table 3-7 are displayed area streamflow records for the period antecedent to the survey of 8-9 October 1975. Above Lake Buchanan, a relatively stable flow regime is evidenced in the Colorado River at USGS Sta. 08147000, on the order of 5.7 m^3/s (200 cfs), with a slight increase in flow observed in mid-September. Flows in the Llano River increased in mid-September 1975, but receded by the end of the antecedent period. Generally, stable flows prevailed in the Llano River, as monitored at USGS Sta. 08151500, averaging 5.7-8.5 m^3/s (200-300 cfs). A relatively constant flow rate was evident at Sandy Creek at USGS Sta. 08152000, with the exception of a small stormwater hydrograph in mid-September 1975. The maximum average daily discharge associated with this storm hydrograph was 10.0 m^3/s (352 cfs). Inflow to Lake Travis from the Pedernales River, as measured at USGS Sta. 08153500, was relatively steady, with a small storm flow hydrograph again evident in mid-September. Releases from Lake Travis exhibited a decrease in mid-September but remained relatively constant through the remainder of the antecedent period. Below Town Lake, flows measured at USGS Sta. 08158000 correlate with the releases from Lake Travis upstream.

In Fig. 3-13 are displayed observed concentrations of total phosphorus and chloride for the survey of 8-9 October 1975. The total phosphorus concentration in the headwaters of Lake Buchanan was 0.0163 mg/l. The phosphorus level decreased through Lake Buchanan to 0.0131 mg/l in the pool above the dam. Phosphorus levels in Lake LBJ and Lake Marble Falls were below the detectable limit of 0.0098 mg/l. The phosphorus level was elevated to 0.0163 mg/l in the upper reach of Lake Travis, but decreased to 0.0098 mg/l in the lower pool. A total phosphorus level of <0.0098 mg/l was detected in the lower pool of Lake Austin, and Town Lake displayed a level of

0.0098 mg/l. Observed levels of chloride were relatively constant through Lake Buchanan and Inks Lake, but dropped substantially in Lake LBJ. From a concentration of 103 mg/l of chloride in the lower pool of Inks Lake, the chloride level decreased to only 60 mg/l at the mid-point of Lake LBJ. Chloride levels showed a slight increase in Lake Marble Falls, but decreased in Lake Travis from 58 mg/l in the headwaters to 52 mg/l in the lower pool above the dam. A chloride level of 51 mg/l was observed throughout Lake Austin and in the headwater reach of Town Lake, with a decrease to 48 mg/l in the pool above Longhorn Dam.

The survey of 8-9 October 1975 represents a period of relatively stable external hydraulic loadings to the series of reservoirs. The observed small inflow hydrograph in mid-September 1975 apparently produced no major impacts upon reservoir water quality, but could account for some of the observed fluctuations in constituent concentrations.

Survey of 13-14 January 1976

Streamflow records for area watercourses during the antecedent period prior to the survey of 13-14 January 1976 are shown in Table 3-8. A relatively steady inflow regime was detected on the Colorado River at USGS Sta. 08147000 above Lake Buchanan, averaging roughly 5.7-7.1 m^3/s (200- 250 cfs). Flows in the Llano River were generally stable, as measured at USGS Sta. 08151500. A small runoff hydrograph in the Llano River was evident during the latter part of December 1975. As indicated at USGS Sta. 08152000, a stable flow regime was encountered on Sandy Creek. A small runoff hydrograph was observed in late December 1975, in conformance with streamflow records on the Llano River. Streamflow records for the Pedernales River at USGS Sta. 08153500 indicate a relatively stable flow regime with, again, a runoff hydrograph displayed in late December 1975. Releases from Lake Travis were discontinued for much of the antecedent period, though a temporary resumption of releases was evident shortly before the survey of 13-14 January 1976. Streamflow in the Colorado River below Town Lake measured at USGS Sta. 08158000 corresponds to the observed releases from Lake Travis, supplemented by inflow from local area tributaries.

Total phosphorus and chloride levels on 13-14 January 1976 are shown in Fig. 3-14. A total phosphorus concentration of 0.0261 mg/l was detected in both the headwaters and lower pool of Lake Buchanan. Phosphorus increased in the lower pool of Inks Lake, up to a concentration of 0.0392 mg/l. Concentrations decreased to 0.0098 mg/l within Lake LBJ, Lake Marble Falls, and Lake Travis. Similarly, a concentration of 0.0098 mg/l phosphorus was detected in the lower pool of Lake Austin, although a sample from the headwaters of that reservoir displayed a concentration of 0.0163 mg/l. There was no concurrent data collection in Town Lake. Chloride levels exhibited a steady decrease from 115 mg/l in the headwaters of Lake Buchanan to 108 mg/l in the lower pool of Inks Lake. Concentrations decreased substantially in Lake LBJ, with a level of 59 mg/l chloride measured in the lower pool. In Lake Travis, chloride levels decreased from 59 mg/l in the upstream portion to

53 mg/l in the lower pool above the dam. A chloride concentration of 51 mg/l was measured in the lower pool of Lake Austin.

Observed data for the survey of 13-14 January 1976 are characteristic of a period of stable external hydraulic loadings to the series of reservoirs.

Survey of 17-18 February 1981

Pertinent streamflow data for area gauging stations are displayed in Table 3-9. Streamflow records at USGS Sta. 08147000 on the Colorado River near San Saba, upstream from Lake Buchanan, indicate a lengthy period of stable streamflow, on the order of 5.7 m^3/s (200 cfs). A stable flow regime of roughly 4.8 m^3/s (170 cfs) is indicated for USGS Sta. 08151500 on the Llano River at Llano, Texas. At USGS Sta. 08152000 on Sandy Creek at Hwy. 71, a relatively stable flow regime of typically 0.3-0.8 m^3/s (10-30 cfs) was observed, although flows were elevated above 2.8 m^3/s (100 cfs) for a two-day period in early February 1981. Flows in the Pedernales River, recorded at USGS Sta. 08153500 near Johnson City, were relatively steady, on the order of 1.4 m^3/s (50 cfs). Releases from Mansfield Dam were variable, as indicated in Table 3-9. A stable low-flow regime was evidenced in Bull Creek at USGS Sta. 08154700 at Loop 360. Flows in Barton Creek at USGS Sta. 08155300 at Loop 360 were relatively stable over the range of roughly 0.3-0.8 m^3/s (10-30 cfs). Streamflow records at USGS Sta. 08158000 on the Colorado River at Austin reflect releases from Town Lake and Lake Austin upstream.

Concentrations of total phosphorus and chloride showed a marked decrease through the series of reservoirs, as displayed in Fig. 3-15. The total phosphorus level ranged from 0.04 mg/l in the headwaters of Lake Buchanan to 0.01 mg/l in the lower pool of Town Lake, while chloride concentration decreased from 101 mg/l to 50 mg/l over the reach. Phosphorus exhibited its most substantial decline in Lake Buchanan, dropping from 0.04 mg/l to 0.02 mg/l, with a subsequent decrease from 0.02 mg/l to 0.01 mg/l in the headwaters of Lake Travis. Chloride levels decreased in Lake Buchanan, from 101 mg/l to 85 mg/l, and most markedly below Inks Lake through Lake LBJ, dropping from 85 mg/l to 55 mg/l. Chloride levels gradually declined to a concentration of 50 mg/l in the lower pool of Town Lake.

The survey of 17-18 February 1981 is representative of a period of stable external hydraulic loadings to the series of reservoirs.

3.2.2 Surveys Under Stratified Conditions

The previous subsection presented a discussion of surveys conducted under unstratified conditions in the Highland Lakes, which typically occur during periods of cooler temperatures. Emphasis on cold-weather surveys is appropriate for the present analysis since the assumption of completely mixed conditions is more closely approximated. Surveys during periods of warmer temperature would typically be characterized by the existence of a marked degree of thermal stratification in the reservoirs.

Under stratified conditions, the available water quality data base may not be readily amenable to quantitative analysis of trends and interrelationships within the series of reservoirs, particularly since the majority of historical data (collected by the TDWR) is comprised of only surface measurements of total phosphorus and chlorides, and therefore information on the vertical structure of phosphorus and chlorides is not available. The only measurements routinely taken throughout the vertical extent of the water column are temperature, dissolved oxygen, and conductivity. The availability of these profiles does enable determination of the presence or absence of stratification. Several available warm-weather water quality surveys were analyzed in order to compare the longitudinal distribution of phosphorus and chlorides with the cold-weather data sets previously described.

Survey of 18-19 July 1974

Streamflow records for a thirty-day antecedent period prior to the survey of 18-19 July 1974 are displayed in Table 3-10. A stable period of very low streamflow was recorded on the Colorado River above Lake Buchanan at USGS Sta. 08147000. The survey of 18-19 July 1974 was in fact preceded by several months of low streamflow. A small runoff hydrograph was observed in mid-May 1974. The Llano River at USGS Sta. 08151500 also experienced a period of low streamflow rates. Only minimal flow was observed in Sandy Creek at USGS Sta. 08152000, with a substantial portion of the antecedent period experiencing zero flow. Low flows were also observed in the Pedernales River at USGS Sta. 0815300. Releases from Mansfield Dam for irrigation needs downstream were relatively steady, as reflected in recorded streamflow rates on the Colorado River at USGS Sta. 08158000 below Town Lake.

Concentrations of total phosphorus and chlorides in the Highland Lakes for the survey of 18-19 July 1974 are displayed in Fig. 3-16. The phosphorus concentration decreased from 0.1013 mg/l in the headwaters of Lake Buchanan to <0.0098 mg/l in the lower pool. In Lake Inks, a concentration of 0.0523 mg/l was detected in the upper reach, while the total phosphorus level in the lower pool was <0.0098 mg/l. A phosphorus level of 0.0229 mg/l was observed in Lake LBJ near the confluence with the Llano River, but concentrations decreased with distance downstream to a level of <0.0098 mg/l in the lower pool. The lower pool of Lake Marble Falls displayed a phosphorus concentration of 0.0163 mg/l, which was also the level observed in the upper reach of Lake Travis near Spicewood. Two downstream locations on Lake Travis showed concentrations of <0.0098 mg/l total phosphorus, while 0.0196 mg/l was detected in the pool above the dam. A level of <0.0098 mg/l total phosphorus was detected in the lower pools of Lake Austin and Town Lake (8 July 1974).

A uniform chloride level of 108 mg/l was determined in both the headwater and the lower pool of Lake Buchanan. The chloride level decreased to 105 mg/l in the lower pool of Inks Lake, then exhibited a substantial decrease within Lake LBJ, to a chloride level of 44 mg/l in the lower pool. Chloride levels were uniform at 42 mg/l in Lake Marble Falls, but steadily increased to 62 mg/l in the lower pool of Lake Travis. A concentration of 55 mg/l of chloride was measured in the lower pool of Lake Austin on 18 July 1974. Samples were not collected from Town Lake coincident with the survey of 18-19 July 1974, however, in early July a level of 54 mg/l was detected in the lower pool.

Data collected on the survey of 18-19 July 1974 are indicative of a period with steady external hydraulic conditions. Generally, a low-flow hydrologic regime was attendant to the period of the sampling survey, with several antecedent months of stable inflow.

Survey of 10-11 July 1975

Streamflow records for area watercourses for a one-month period antecedent to the survey of 10-11 July 1975 are displayed in Table 3-11. The attendant hydrologic regime was characterized by sustained high streamflow rates in a relatively wet year. Streamflow data for the Colorado River above Lake Buchanan at USGS Sta. 08147000 indicate an antecedent period of sustained streamflow fluctuating from roughly 8.5-39.6 m^3/s (300-1400 cfs). A similar flow regime was detected on the Llano River at USGS Sta. 08151500. Flows into Sandy Creek at USGS Sta. 08152000 were also relatively high, ranging from roughly 0.8-10.2 m^3/s (30-360 cfs). Relatively high flows were also detected on the Pedernales River at USGS Sta. 08153500. Releases from Mansfield Dam were typically in excess of 141.6 m^3/s (5,000 cfs) during the antecedent period, with corresponding flow rates observed downstream on the Colorado River at USGS Sta. 08158000.

Total phosphorus and chloride levels for the survey of 10-11 July 1975 are depicted in Fig. 3-17. Total phosphorus concentration decreased from 0.0229 mg/l in the headwaters of Lake Buchanan to <0.0098 mg/l in the lower pool. Levels of total phosphorus were consistently <0.0098 mg/l in the lower pools of Inks Lake, Lake LBJ, Lake Travis, and Lake Austin. A concentration of 0.0163 mg/l total phosphorus was detected in the lower pool of Town Lake on 14 July 1975.

From a concentration of 101 mg/l in the headwaters, the chloride level dropped slightly in Lake Buchanan. A substantial decline in chloride level was detected from the lower pool of Inks Lake, where a level of 100 mg/l was detected, to a concentration of 54 mg/l within the lower pool of Lake LBJ. The chloride level in Lake Travis decreased from 59 mg/l in the headwaters down to 49 mg/l in the lower pool. A concentration of 45 mg/l chloride was observed in the lower pool of Lake Austin, and the lower pool of Town Lake showed a level of 43 mg/l on 14 July 1975.

Survey of 27-28 July 1976

In Table 3-12 are contained the area streamflow data antecedent to the survey of 27-28 July 1976. Within a one-month antecedent period, the first period of relatively heavy inflow was encountered subsequent to at the very least an eight-month period of relatively stable flows. Substantial inflows to Lake Buchanan were indicated on the Colorado River at USGS Sta. 08147000, with average daily flow rates up to 235.1 m^3/s (8300 cfs). High-flow regimes were also observed on the Llano River at USGS Sta. 081515000 and on Sandy Creek at USGS Sta. 08152000, tributaries to Lake LBJ. Substantial flows were observed at USGS Sta. 08153500 on the Pedernales River. Releases from Mansfield Dam ranged from roughly 56.6-141.6 m^3/s (2000-5000 cfs). Observed flow rates on the Colorado River at USGS Sta. 08158000 below Town Lake corresponded to the releases from Mansfield Dam.

Total phosphorus and chloride levels throughout the series of reservoirs are displayed in Fig. 3-18. Total phosphorus decreased from a level of 0.02 mg/l in the headwaters of Lake Buchanan to a concentration of 0.01 mg/l in the lower pool of Lake Buchanan and Inks Lake. Observed concentrations at all remaining reservoir stations downstream were uniformly below the detection limit of 0.01 mg/l total phosphorus.

From a concentration of 113 mg/l in the headwaters of Lake Buchanan, chloride levels decreased to 93 mg/l in the lower pool. A further decline to 84 mg/l chloride was detected in the lower pool of Inks Lake, while concentrations dropped to 66 mg/l in the lower pools of Lake LBJ and Lake Marble Falls. Chloride concentrations in Lake Travis showed an increase from the headwaters to the lower pool, from 44 mg/l to 51 mg/l, respectively. Chloride levels of 53 mg/l were observed in the lower pools of both Lake Austin and Town Lake.

Survey 19 August 1977

Antecedent hydrologic conditions for the survey of 19 August 1977 are displayed in Table 3-13. Relatively stable flow regimes are evident, with the most recent period of heavy inflow occurring roughly four months prior to the survey. Streamflow rates at USGS Sta. 08147000 on the Colorado River above Lake Buchanan were relatively stable, on the order of 4.2-5.7 m^3/s (150-200 cfs). A stable flow regime was observed on the Llano River at USGS Sta. 08151500, with flows on the order of 4.2 m^3/s (150 cfs). Only minimal flows were observed on Sandy Creek at USGS Sta. 08152000, with much of the antecedent period under zero flow conditions. Flows were relatively uniform at USGS Sta. 08153500 on the Pedernales River, ranging from roughly 0.7-1.3 m^3/s (25-45 cfs). Releases from Mansfield Dam were usually below 56.6 m^3/s (2000 cfs), as measured at USGS Sta. 08154510. Flows on the Colorado River at USGS Sta. 08158000 correlated to the releases from Mansfield Dam.

Data for total phosphorus and chlorides collected on the survey of 19 August 1977 are shown in Fig. 3-19. Continuity of the data set is absent within the Lake Austin and Town Lake stations, as samples were collected at later dates. Total phosphorus concentrations in Lake Buchanan ranged from 0.05 mg/l in the headwaters to 0.02 mg/l in the lower pool. A level of 0.03 mg/l total phosphorus was observed in the lower pools of both Inks Lake and Lake LBJ. The lower pool of Lake Marble Falls displayed a total phosphorus concentration of 0.02 mg/l, while a level of 0.01 mg/l was detected in the lower pool of Lake Travis. Phosphorus levels were below the detection limit of 0.01 mg/l in the lower pool of Lake Austin (31 August 1977) and the lower pool of Town Lake (23 August 1977).

Observed chloride levels in Lake Buchanan decreased from 104 mg/l to 96 mg/l from the headwaters to the lower pool. A concentration of 104 mg/l chloride was observed in the lower pool of Inks Lake, with a subsequent decrease to 77 mg/l in the lower pool of Lake LBJ. Chloride decreased from 79 mg/l in the lower pool of Lake Marble Falls to 72 mg/l in the upper reach of Lake Travis near Spicewood. A level of 57 mg/l chloride was detected in the lower pool of Lake Travis and throughout Lake Austin. On 23 August 1977, the chloride level in the lower pool of Town Lake was 54 mg/l.

Data collected on the survey of 19 August 1977 were characteristic of a period of stable external hydraulic conditions. Constituent concentrations may have been affected by a period of heavy inflow approximately four months prior to the survey.

Survey of 27-28 July 1981

Antecedent flow conditions for area streams prior to the survey of 27-28 July 1981 are displayed in Table 3-14. The table includes streamflow rates from mid-June in order to display the most recent period of high inflows. Generally, the records indicate an antecedent period of continually decreasing streamflow rates, with a period of heavy inflow in the previous month. Streamflow rates on the Colorado River at USGS Sta. 08147000 decreased steadily from roughly 56.6 m^3/s (2000 cfs) in mid-June to 0.7 m^3/s (26 cfs) on 28 July 1981. A declining streamflow regime is also evident on the Llano River at USGS Sta. 08151500, with the flow rate decreasing to roughly 5.7 m^3/s (200 cfs). A decrease from substantial to minimal flow was also detected on Sandy Creek at USGS Sta. 08152000. A similar decline was observed at USGS Sta. 08153500 on the Pedernales River. Releases from Mansfield Dam were relatively large in mid-June, then decreased to more typical levels for the remainder of the antecedent period. Streamflow records for Bull Creek, tributary to Lake Austin, and Barton Creek, tributary to Town Lake, displayed relatively high flow rates for their respective watersheds. Observed flows below Town Lake on the Colorado River at USGS Sta. 08158000 correlated with the releases from Mansfield Dam upstream.

A total phosphorus level of 0.04 mg/l was detected in the headwaters of Lake Buchanan, with a subsequent decrease to 0.02 mg/l in the lower pool, as displayed in Fig. 3-20. The lower pool of Inks Lake displayed a total phosphorus concentration of 0.03 mg/l, while levels of 0.02 mg/l total phosphorus were detected in the lower pools of Lake LBJ and Lake Marble Falls. The lower pools of Lake Travis, Lake Austin, and Town Lake uniformly displayed a total phosphorus concentration of 0.01 mg/l.

A chloride concentration of 95 mg/l was detected in the lower pool of Lake Buchanan, a slight increase from the headwater concentration. Chloride decreased from 93 mg/l in the lower pool of Inks Lake to 45 mg/l in Lake LBJ below the confluence with the Llano River. The chloride level increased slightly in the lower pool of Lake LBJ, but then declined to 45 mg/l in the lower pool of Lake Marble Falls. The upper reach of Lake Travis near Spicewood showed a chloride concentration of 35 mg/l, with an increase to 46 mg/l in the lower pool of the reservoir. Chloride levels of 44 mg/l and 43 mg/l were detected in the lower pools of Lake Austin and Town Lake, respectively.

3.2.3 Vertical Profiles

Data for phosphorus and chloride concentration through the vertical extent of the water column are extremely limited on the Highland Lakes. Vertical profiles for the series of reservoirs are available from the CRWR data set collected in 1968 and 1969. Other sources provide data on specific reservoirs. For example, the USGS conducted occasional sampling surveys on Lake Austin and Town lake. The most voluminous source of data on the vertical structure of phosphorus in a reservoir was assembled by the CRWR on Lake LBJ during the period 1971-1975. Vertical profiles of total phosphorus and other constituents were measured at typically monthly intervals in the lower pool and at several other sampling locations throughout the impoundment.

Examples of the concentration profiles for each reservoir from the 1968 CRWR data set (Fruh and Davis, 1969a) are shown in Fig. 3-21. Profiles were selected to depict unstratified and stratified conditions in the lower pool of each impoundment. Profiles for either February or March 1968 depict essentially isothermal conditions in the reservoirs in the absence of stratification. The reservoirs were thermally stratified with distinct epilimnion and hypolimnion layers at the time the July 1968 profiles were collected, with the exceptions of Lake Austin and Town Lake. Chloride data were unavailable, so conductivity was substituted as an indicator of dissolved solids levels in the data plots. The data set includes numerous phosphorus measurements at very low levels, as discussed in Sec. 3.2.1.1. There is little evidence of systematic trends in the phosphorus profiles. A trend to higher hypolimnetic concentrations of phosphorus is indicated, but the data are highly variable. The observed variability may be attributable in part to sampling and analytical errors (see Fruh and Davis, 1969a).

Data from the intensive study of Lake LBJ (Fruh, et al., 1975a) were also examined. In Fig. 3-22 are displayed selected vertical profiles of phosphorus and chloride measured in the lower pool of the reservoir during 1973. Based upon the profiles of temperature and dissolved oxygen collected in 1973, isothermal conditions in the water column persisted until late April, which marked the onset of thermal stratification. By early May, the stratification was more pronounced and dissolved oxygen levels began to decrease in the lower layers of the water column. Anoxic conditions in the hypolimnion were encountered in early June. Through the summer, the depth of the thermocline was roughly 4.6-9.2 m (15-30 feet). Stratification was still evident in early October, but temperatures in the water column were becoming more uniform. Under the influence of a flood event, isothermal conditions in the water column were exhibited by late October. The profiles in Fig. 3-22 indicate that phosphorus and chloride concentrations are roughly homogeneous under cold-weather conditions in the absence of stratification. Under stratified conditions, phosphorus concentrations typically display an increase in the hypolimnion. However, substantial variability in the vertical profiles is indicated. The effects of a large flood in the fall of 1973 are exhibited in the December profile: phosphorus levels are elevated, while chloride concentrations are depressed. The CRWR measurements of total phosphorus in Lake LBJ display generally higher concentrations than the TDWR data set, but the source of the discrepancy cannot be discerned. Fruh, et al. (1977) reported coefficients of variance of 50 percent for the laboratory tests and 70 percent for the field sampling procedures. The CRWR data for Lake LBJ are examined further in Sec. 5.5.3.

3.2.4 Discussion of Spatial Trends

Profiles of phosphorus concentration through the Highland Lakes were presented for several sampling surveys. The profiles display the concentration trends typically encountered under both unstratified and stratified conditions, based upon surface measurements. Data for chlorides were also presented, in order to illustrate the corresponding behavior of a conservative constituent.

Phosphorus concentrations are relatively low throughout the Highland Lakes. The highest concentrations are usually encountered in Lake Buchanan, at a typical range of 0.01-0.05 mg/l, with higher values occasionally detected in the headwater reach. In the reservoirs below Lake Buchanan, the majority of observations range from 0.01-0.02 mg/l total phosphorus. Values below the detection limit (0.01 mg/l) are frequently encountered, particularly in the lower reaches of Lake Travis. Sporadic elevations in phosphorus concentrations observed within the series of reservoirs appear to be localized, probably in response to tributary inputs.

The profiles generally indicate that surface layer phosphorus concentrations decrease from the upper to the lower reservoirs. Concentrations are usually greatest within Lake Buchanan, which receives inflow from the Colorado River. Phosphorus levels typically decrease from the headwaters of Lake Buchanan to the lower pool. Concentrations in Inks Lake reflect the composition of the releases from Lake Buchanan. Decreasing concentrations are often exhibited in Lake LBJ, where headwater phosphorus levels can be elevated in apparent response to inflow from the Llano River. Phosphorus levels in Lake Marble Falls are generally similar to the lower pool of Lake LBJ upstream. Concentrations typically decrease in Lake Travis, with localized elevations apparently due to inflow from the Pedernales River. In Lake Austin, phosphorus levels are usually stable or show a slight decrease. Concentrations are sometimes elevated within Town Lake. These general trends are characteristic of both unstratified and stratified conditions.

Profiles of chloride concentration are similar to phosphorus trends. However, chloride levels consistently display a marked decrease within Lake LBJ. This is apparently a result of the lower chloride levels in inflow from the Llano River and

Sandy Creek. Where phosphorus levels show localized elevations, chloride concentrations often decrease substantially in response to tributary inflow.

The sampling surveys described above dated from 1969-1981. Hydraulic retention times for the series of reservoirs were described in Sec. 3.1.2. for the period 1968-1982. The retention times can affect the interpretation of the concentration profiles. Phosphorus concentrations are affected by internal reactions, as well as hydraulic retention time. Retention times in Lake Buchanan ranged from 0.9-10.4 yrs. This is in effect the time required for inflow loads from the Colorado River at the head of the reservoir to demonstrate effects upon concentrations in the lower pool, based upon hydraulics only. Thus, phosphorus concentrations observed in the headwater are reflective of recent incoming mass loads, but the effects require longer periods of time to become manifest in the lower pool, particularly in the larger reservoirs. Lower pool concentrations in the larger impoundments would be expected to be influenced primarily by internal reactions rather than incoming mass loads. Retention times in Inks Lake are short, 0.02-0.16 yr. The reservoir is dominated by throughflow from Lake Buchanan upstream. Concentrations observed in Inks Lake are thus closely related to the lower pool conditions in Lake Buchanan. Lake LBJ displays retention times of 0.1-0.3 yr. Concentrations can therefore be strongly influenced by tributary inflows throughout the length of the impoundment. Lake Marble Falls is a throughflow-dominated impoundment, with retention times of 0.005-0.02 yr. Phosphorus concentrations are thus strongly dependent upon conditions in Lake LBJ upstream. Since Lake LBJ can also have a relatively short retention time, recent incoming mass loads to the upstream reservoir can theoretically have a substantial effect upon conditions in Lake Marble Falls. Longer retention times, 0.5-1.6 yrs, are exhibited in Lake Travis. With the longer retention times, phosphorus concentrations in the lower pool are probably influenced primarily by internal kinetics rather than responses to recent mass inflows. Both Lake Austin and Town Lake are throughflow dominated, with very short retention times (0.008-0.03 yr and 0.002-0.005 yr, respectively). Thus, phosphorus concentrations should reflect closely the characteristics of releases from Lake Travis upstream or localized mass inflow.

The interpretation of constituent profiles may also be influenced by the extant vertical structure. The present analysis emphasized for the most part surface measurements of phosphorus, since these comprise the majority of the available data base. However, examination of surface concentrations may result in misinterpretation of trends under certain conditions. Vertical profiles of phosphorus concentration at lower pool stations were described in the preceding section. Under conditions of thermal stratification, phosphorus concentrations may be elevated in the hypolimnion. Homogeneity is evidenced under unstratified conditions. The vertical structure may be further influenced by mixing conditions within the reservoirs. Layering of inflow due to differences in density can occur under certain conditions. Density differences have been encountered in the Highland Lakes in association with releases from upstream impoundments and tributary inflow (Fruh and Davis, 1969a).

3.3 ANALYSIS OF TEMPORAL TRENDS IN PHOSPHORUS CONCENTRATION

The data base for phosphorus and chloride in the Highland Lakes was subjected to a detailed analysis of temporal trends. For the temporal analyses, the data at specific sampling stations were examined for variability with time. The objective of the analysis was to obtain a characterization of the constituent fluctuations within a reservoir and possibly determine the primary forcing mechanism for observed fluctuations (for example, external loadings or internal kinetics).

Interpretation of phosphorus data for temporal trends may be handicapped for a variety of reasons. The hydraulic retention times of the reservoirs may significantly affect observed concentrations. With the smaller reservoirs in particular, short retention times may relegate primary importance to external loadings rather than internal kinetics. Responses to variations in inflow could thus obliterate temporal trends. The present evaluation of surface phosphorus measurements may obscure seasonal trends in the water column as a whole, due to the effects of thermal stratification and mixing conditions within the reservoirs. In addition, the extent of the historical data base may simply be insufficient for a demonstration of temporal trends, both in terms of an inadequate number of samples at any particular location and an inadequate sampling frequency.

3.3.1 Seasonality

Data for total phosphorus were plotted versus month for selected sampling stations in order to determine whether any seasonal trends were evident. Generally, only the routine monitoring data collected by the TDWR were plotted, which consist entirely of surface measurements. Theoretically, surface measurements of total phosphorus could be expected to demonstrate a marked degree of seasonality. Surface concentrations of total phosphorus would be expected to be lowest during warm weather months when the reservoir is under stratified conditions. Under stratified conditions, phosphorus would be expected to decline in the epilimnion due to settling of particulate phosphorus and phytoplankton uptake and subsequent sedimentation of the biomass to the hypolimnion. Under anoxic conditions, elevated phosphorus levels are anticipated in the hypolimnion, due to sedimentation from the epilimnion and release from the sediment layer. With the onset of colder weather, stratified conditions are eliminated and the reservoir contents become mixed. The comingling of hypolimnetic and epilimnetic waters could then produce a higher level of total phosphorus in the surface zone. Generally, seasonality was not evident in the phosphorus data set, as discussed below. Chloride data were subjected to the same analysis to provide a comparison with a conservative material. Marked seasonality was not anticipated with the chlorides data. However, mineral levels are commonly influenced by the streamflow rate, such that trends could potentially develop in response to seasonal variations in the flow regime.

Town Lake

Total phosphorus data for Town Lake Sta. 1402.08 are shown in Fig. 3-23(a). Seasonal trends are not evident in the data set. Concentrations at or below the phosphorus detection limit occur throughout the year. Phosphorus concentrations typically range from 0.010-0.020 mg/l, though substantially higher values are occasionally recorded. Chlorides data, also presented in Fig. 3-23(a), display no apparent seasonality. Town Lake is a throughflow-dominated reservoir, with short hydraulic retention times, particularly under the influence of releases from Lake Travis for downstream demands. With the substantial throughflow, constituent concentrations in Town Lake would be expected to be influenced primarily by external loadings rather than internal kinetics.

Lake Austin

Total phosphorus data for Lake Austin near the dam at Sta. 1403.01 are displayed in Fig. 3-23(b). There is no pronounced trend of seasonality in the data set. Concentrations at or below the detection limit occur throughout the year. (The existence of a large number of data points reported as less than the detection limit may in fact diminish the utility of the present analysis.) The only noteworthy feature of the Lake Austin data set is the absence of any phosphorus values in excess of the detection limit for the period of June-September. Substantial releases from Mansfield Dam upstream typically occur from March through November for satisfaction of irrigation and hydropower generation demands. The phosphorus dynamics in Lake Austin for much of the year are probably influenced primarily by external loads rather than internal kinetics, due to the throughflow-dominated nature of the impoundment. Data for chlorides (including data for USGS Sta. 08154900) are also shown in Fig. 3.23(b) and indicate no seasonal trends.

Lake Travis

Total phosphorus data for Lake Travis have been plotted for three stations: Sta. 1404.01 near the dam (Fig. 3-23(c)), Sta. 1404.03 near Lakeway (Fig. 3-23(d)), and Sta. 1404.06 near Spicewood (Fig. 3-23(e)). Seasonality in observed phosphorus concentrations is not exhibited at Sta. 1404.01. The majority of reported values are at or below the detection limit. At the station near Lakeway, no clear trends are evident for phosphorus behavior, with the majority of values lying at or below the detection limit. Total phosphorus data for Lake Travis near Spicewood show more variability than the data for the lower reservoir stations. Seasonal trends, however, are not apparent. An extremely high total phosphorus concentration of 1.0131 mg/l was recorded in October 1974. This particular sampling survey was conducted under a period of high inflow to the reservoir, which could provide an explanation for the elevated level. On the same date, a phosphorus concentration of 0.0523 mg/l was detected downstream at the station near Lakeway. Chlorides data for the three stations do not display seasonal trends.

The hydraulic residence times in Lake Travis are relatively large. Thus, the effects of throughflow upon constituent concentrations could be expected to be diminished, which could amplify the effects of internal kinetics. The fact that the data plots do not display any systematic trends may imply either that strong trends do not exist or that the temporal density of the data is insufficient for their detection. In fact, the extent of the data set is limited, and its utility is diminished by the preponderance of values below the detection limit.

Lake Marble Falls

Data for Sta. 1405.01, on Lake Marble Falls near the dam, are displayed in Fig. 3-23(f). Generally, the data fluctuate to a greater degree than those discussed previously for Lake Travis. Seasonality is not evident; however, there is an absence of values in excess of 0.02 mg/l total phosphorus during the period extending from May through October. Data for chlorides do not display seasonal trends. Lake Marble Falls is a throughflow-dominated impoundment, as attested by the short hydraulic residence times. Therefore, fluctuations in constituent concentrations are probably influenced predominantly by hydraulics and external loadings rather than internal reactions.

Lake LBJ

Total phosphorus concentrations near the dam on Lake LBJ, collected at Sta. 1406.01, are depicted in Fig. 3-23(g). The data do not indicate a seasonal trend. In every season, concentrations at or below the detection limit as well as concentrations of at least 0.02 mg/l total phosphorus were encountered. Seasonal trends are not evident in the chloride data. Detention times can be relatively short in Lake LBJ, amplifying the effects of throughflow originating from the Llano River and releases from upstream reservoirs.

Inks Lake

As shown in Fig. 3-23(h), total phosphorus levels in Inks Lake, collected at Sta. 1407.01 near the dam, fluctuate through all seasons of the year. Seasonal trends are absent. Similarly, trends are not evident in the chlorides data. With its short retention times, constituent concentrations in Inks Lake can be anticipated to be dominated by releases from the upstream reservoir.

Lake Buchanan

Total phosphorus data for Lake Buchanan at Sta. 1408.01 near the dam are shown in Fig. 3-23(i). Seasonality is not apparent, as phosphorus concentrations fluctuate throughout the year. All values from the period of approximately November through March were well above the detection limit, a phenomenon which did not materialize in the other reservoirs. In Lake Buchanan, the only occurrences of values at or below the detection limit were encountered from the months of May through October. Seasonality is not evident in the chlorides data. As with Lake Travis, hydraulic retention times in Lake Buchanan are relatively large, which could enhance observation of the effects of internal reactions upon constituent concentrations.

Colorado River at Bend, Texas

Data for the Colorado River at Sta. 1409.01, located at Bend, Texas, were plotted in a similar fashion in order to illustrate phosphorus behavior in the major tributary above Lake Buchanan. The data, displayed in Fig. 3-23(j), were usually collected at quarterly intervals throughout any specific sampling year. Generally, highest inflow concentrations of total phosphorus were monitored in the summer and fall. The lowest values in the record were measured during the month of December, although the concentrations in this month were not consistently low. Chlorides data do not display strong seasonal trends, and concentrations fluctuate over a wide range. Instream concentrations can potentially correlate with streamflow rate (see Chapter 4).

The preceding analysis indicates a general absence of seasonality in observed phosphorus concentrations in the Highland Lakes. Pronounced seasonal trends are not evident, and only in a few instances were any uniform tendencies evident. Throughout the data set, both high and low total phosphorus levels occur at varying times of the year. Similarly, marked trends are absent in the chlorides data set. Several of the reservoirs are throughflow-dominated, and thus, constituent concentrations may be influenced primarily by the extant hydraulics and inflow mass loadings. Internal kinetics would be expected to demonstrate a stronger influence upon observed concentrations in impoundments with longer residence times.

3.3.2 Consecutive Sampling Surveys

Measurements of total phosphorus from consecutive sampling surveys were examined for temporal trends in concentration. The analyses focused upon data collected by the TDWR in their routine monitoring surveys at a frequency of three or four samples per year. Data surveys examined under this criterion generally encompassed the periods of October 1973 to July 1976 and December 1980 to July 1981.

As noted previously, routine TDWR data sets consist of surface samples only. Anticipated seasonal trends could include reduced levels of total phosphorus within the surface layer in warm weather months under stratified conditions, followed by increased total phosphorus concentrations in colder months under completely mixed conditions. Reduced levels of total phosphorus during warmer periods would be attributable to sedimentation of particulate phosphorus compounds and uptake by phytoplankton with subsequent settling of the biomass out of the epilimnion. During cold weather periods, the increased total phosphorus levels in the hypolimnion would be mixed with the epilimnetic waters, producing a potential elevation in surface concentrations. Alternatively, reservoir phosphorus levels may fluctuate in response to tributary inflow. Examination of total phosphorus levels from consecutive sampling surveys may provide an indication of the relative response to internal cycling versus response to external loadings. Data for chlorides were examined to provide a comparison with the behavior of a conservative constituent.

Response of surface concentrations to internal cycling and external loadings may not be straightforward, due to mixing conditions within the reservoirs. For example, releases from an upstream impoundment often differ in temperature from the receiving waterbody, which can result in significant density currents. Density differences can also be encountered with tributary inflow.

The hydraulic retention time can have a substantial effect upon the observed concentration profiles, as noted in the preceding section. The effects of retention time have implications upon the temporal density of the data set. In the throughflow-dominated reservoirs with short retention times, the frequency of sampling must be commensurate with the retention time in order to ascertain the possible trends associated with the hydraulic characteristics. With longer residence times, less temporal density may be required to demonstrate the effects of external loadings upon internal concentrations. The limited sampling frequency of the historical data set probably diminishes the utility of the present examination of seasonality.

Lake Buchanan

Total phosphorus and chlorides data for Lake Buchanan are displayed in Fig. 3-24. Trends at two stations are depicted: Sta. 1408.03 in the upper reach near Bend and Sta. 1408.01 in the lower pool near the dam. Also displayed in Fig. 3-24 is a record of total monthly inflow into the reservoir, based upon LCRA (1983) data, with the Colorado River the principal tributary.

In the upper reach of the reservoir, levels of total phosphorus and chloride demonstrate substantial fluctuation through time. There are indications of correlation of constituent levels with inflow, though the association is inconsistent. In October 1973, a substantial inflow peak was recorded, which was accompanied by an elevated level of total phosphorus (0.0588 mg/l) and a depression of chloride concentration. During a subsequent period of reduced inflow, phosphorus levels decreased while chlorides exhibited a steady increase. However, phosphorus showed an increase in April 1974 in the absence of a substantial increase in inflow. Under a high inflow regime with a peak in September 1974, a substantial increase in phosphorus level was recorded with a corresponding decrease in the level of chloride. A peak in chloride concentration was recorded in April 1976 following a lengthy period of stable, low inflow, which was accompanied by an observed increase in total phosphorus level. Surveys conducted in 1981 indicate substantial fluctuation in chloride level, but uniform phosphorus measurements of 0.04 mg/l.

At Sta. 1408.01 in the lower pool near the dam, total phosphorus and chloride fluctuate over narrower ranges than those encountered in the headwater reach. A degree of correspondence of constituent concentrations with inflow is indicated, however, a persistent trend is not evident. There does appear to be some evidence of warm weather/cold weather cycling within the reservoir, as demonstrated by reduced levels of total phosphorus in the summers of 1974, 1975, 1976, and 1981. Cold weather surveys generally indicated increased levels of total phosphorus. Evidence of response to inflow loads is also indicated, for example, in the data collected in January 1975, which displayed an increase in total phosphorus level accompanied by a substantial decrease in chloride level, both of which appear to be manifest in response to an antecedent period of substantial tributary inflow over the preceding four to five months.

In summary, the data for Lake Buchanan may indicate a degree of response to internal cycling and external loading. Constituent magnitudes and trends display substantial variability between the upper and lower stations on Lake Buchanan. The upper station appears to be controlled primarily by inflow loading. With the relatively large retention times characteristic of Lake Buchanan, the effect of the inflow loads is dampened during transport to the lower end of the reservoir, where effects of internal cycling would be expected to dominate. Observed chloride levels appear to be more closely correlated to external inflow. The possible effects of density currents originating from the Colorado River inflow cannot be quantified with the existing data base.

Inks Lake

In Fig. 3-25 are displayed data for total phosphorus, chlorides, and total monthly inflow for Inks Lake. The inflow record depicts accumulation of releases from Lake Buchanan upstream and immediate tributary runoff. Two stations within Inks Lake were examined: Sta. 1407.03 in the upper reach and Sta. 1407.01 in the lower pool near the dam.

The limited data at Sta. 1407.03 in the upper reach of the resevoir demonstrate response of chloride levels to inflow, with levels of the conservative mineral increasing during periods of low, stable inflow and decreasing in response to increased inflow. However, only with the initial data point recorded in October 1973 do the phosphorus levels demonstrate an elevated response subsequent to a period of substantial inflow. Observed phosphorus levels do not display consistent seasonality.

In the lower pool of Inks Lake, chloride levels again demonstrate response to external inflow. Phosphorus levels show only a slight indication of correlation to inflow (most notably in October 1973). There does appear to be a degree of association of lower phosphorus levels with warm weather surveys.

As a throughflow-dominated reservoir, constituent concentrations in Inks Lake would be expected to demonstrate correspondence to external mass loadings. Additionally, observation of the effects of internal kinetics would be diminished.

Lake LBJ

Total phosphorus and chloride data for Lake LBJ at Sta. 1406.03 near Kingsland, Sta. 1406.02 near Sherwood Shores, and Sta. 1406.01 in the lower pool near the dam are displayed in Fig. 3-26, accompanied by a record of total monthly inflow to the reservoir. Lake LBJ receives releases from Inks Lake as well as substantial inflow from the Llano River and Sandy Creek watersheds. Comparison of inflow records with those for Lake Buchanan indicates substantially greater total inflows into Lake LBJ in conjunction with high-flow regimes.

Data for Sta. 1406.03, which is located immediately downstream of the confluence of the Llano River and the Colorado River arms, generally vary without correspondence to inflow. With the initial data set collected in November 1973, an increase in total phosphorus and decrease in chlorides were observed, in concert with a peak inflow period. Chloride levels subsequently exhibited a steady increase with little correspondence to recorded inflow. Phosphorus levels fluctuated during the same period, but do not appear to be related to inflow. The lowest phosphorus levels were recorded during cold weather surveys while the highest level occurred in the summer of 1974. Data for the period 1980 through 1981 indicate greater association with inflow. Chloride levels exhibited a marked depression in response to a period of high inflow centered in mid-1981. Conversely, phosphorus levels increased substantially during the same high inflow period. Data for total phosphorus indicate lower concentrations in the cold weather months of 1980, with a subsequent elevation in concentration during the warm weather months of 1981.

Chloride data for Sta. 1406.02 near Sherwood Shores indicate a degree of correspondence with inflow. However, only with the initial data point recorded in November 1973 does phosphorus display a marked increase in response to a high inflow period. From early 1975 to mid-1976, phosphorus levels remained at or below the detection limit, in contrast to the observed fluctuation in chloride levels.

As with the upstream stations, an elevated phosphorus level and decreased chloride level were observed at Sta. 1406.01 in November 1973 after a period of high inflow. Chloride levels subsequently exhibit a degree of correspondence with inflow in a manner quite similar to the trends observed at the upstream Sta. 1406.02. Trends in phosphorus concentration are also very similar to those observed at the upstream station. Data for the period 1980-1981 indicated a decline in chloride level, which can be attributed to increased inflow, accompanied by a uniform level of total phosphorus.

In summary, the data for the two lowermost stations on Lake LBJ demonstrate a great deal of similarity in constituent behavior. The data set for the station below the confluence of the Llano and Colorado Rivers is not as extensive as those of the lower reservoir, which hampered the analysis of trends. Lake LBJ represents a reservoir of intermediate volume in the Highland Lakes. It is not throughflow-dominated to the extent of some of the other impoundments in the chain, but hydraulic retention times are relatively short, typically less than 0.3 yr. Observation of effects of both external loadings and internal kinetics would be anticipated. The data for Lake LBJ generally appear to be more closely correlated with external inflows to the reservoir as opposed to internal cycling mechanisms.

Lake Marble Falls

Total monthly inflow, phosphorus, and chloride data for Lake Marble Falls are shown in Fig. 3-27. Data for Sta. 1405.02 at Highway 281 and Sta. 1405.01 in the lower pool near the dam are included. The inflow to Lake Marble Falls is primarily derived from releases from Lake LBJ. Further, the inflow record resembles closely the inflow record into Lake LBJ, which is a logical consequence of the maintenance of Lake LBJ at a constant water level.

Only limited data are available for Sta. 1405.02. The data do indicate some correspondence to inflow, with depressed chloride levels observed in November 1973 and January 1975 in response to high inflow regimes. Phosphorus levels demonstrate an increase in apparent response to those same high inflow regimes. In 1974, phosphorus levels exhibited a substantial decrease from spring to summer months, with a minimum value recorded in October, which may be indicative of a seasonal trend.

A more extensive data set is available for Sta. 1405.01 in the lower pool near the dam. Chloride levels exhibit fluctuations in response to high inflow periods, but also demonstrate fluctuations which cannot be explained hydrologically. In most cases, observed chloride levels in Lake Marble Falls track closely with those observed in the lower pool of Lake LBJ. The phosphorus levels also appear to be associated with those detected in the lower pool of Lake LBJ. Warm weather/cold weather cycling is not evident in the data set.

Constituent concentrations in Lake Marble Falls generally appear to be controlled by external inflow. The reservoir is dominated by throughflow from releases from Lake LBJ upstream, and hydraulic retention times are relatively short.

Lake Travis

In Fig. 3-28 are displayed data for total phosphorus and chloride in Lake Travis at Sta. 1404.06 near Spicewood, Sta. 1404.03 near Lakeway, and Sta. 1404.01 in the lower pool near the dam. Total monthly inflow records are also presented. Inflow to Lake Travis is derived from releases from Lake Marble Falls upstream, supplemented by additional tributary drainage, principally from the Pedernales River watershed.

At Sta. 1404.06 near Spicewood, data for chloride exhibit fluctuations in response to inflow, with depressed levels observed in October 1974 and July 1976. However, certain high inflow periods did not appear to depress chloride levels. Phosphorus showed a marked increase in apparent response to peak inflows in October 1974 and July 1981.

Data for Sta. 1404.03 near Lakeway exhibit some similarities with the data recorded at the upstream station. Responses to high inflow manifest by elevated phosphorus levels and depressed chloride levels were evident in October 1974 and July 1981. An increased frequency of phosphorus levels at or below the detection limit was generally observed at this station.

Fluctuations in constituent concentrations are dampened considerably in the lower pool at Sta. 1404.01. There appears to be little correlation of phosphorus levels with inflow, though there are still some indications of association of chloride levels (which may be attributable to the differences in behavior of nonconservative and conservative constituents). Phosphorus levels have remained at or below the detection limit from mid-1975 to mid-1976 and late 1980 to mid-1981, with no evidence of seasonality.

Lake Travis is characterized by relatively long hydraulic retention times. As a consequence, it would be reasonable to anticipate that phosphorus concentrations would be effected primarily by internal kinetics rather than external loadings. In fact, data for the lower pool display little apparent association with inflow. However, the longitudinal extent of the impoundment is substantial, and constituent concentrations in the upper reach may demonstrate greater correlation with inflow. The magnitude of potential effects from inflow layering or density currents in the upper reservoir is unknown.

Lake Austin

Phosphorus and chloride levels for Lake Austin, accompanied by a record of total monthly inflow, are displayed in Fig. 3-29. Data are presented for Sta. 1403.03 near Lakeland Park and Sta. 1403.01 in the lower pool near the dam.

Data for Sta. 1403.03 in the upper reach of the reservoir appear to fluctuate without correspondence to fluctuations in inflow. Chloride levels are closely tied to those observed in the lower pool of Lake Travis, with less correspondence in phosphorus levels.

Constituent fluctuations are reduced at Sta. 1403.01 in the lower pool. Chloride levels exhibit substantial similarity between the upper and lower stations on Lake Austin. Phosphorus levels differ in that higher concentrations were occasionally observed in the upstream reach. There is some evidence of a seasonal fluctuation in phosphorus levels. Phosphorus concentrations decreased from early 1974 through the warm weather months, then demonstrated increased levels through the subsequent cold weather period, followed by a decrease in the warm weather months of 1975.

Hydraulic retention times in Lake Austin are generally short--on the order of several days during periods of release from Lake Travis upstream. Constituent concentrations would be expected to be effected primarily by external loadings. With the limited temporal density of sampling, assessment of trends may be futile. Observed constituent levels may correspond poorly to monthly inflow due to the short detention times. In addition, density currents from upstream releases have been encountered under certain conditions, which could affect the analysis of trends.

Town Lake

Phosphorus and chloride levels for Town Lake are displayed in Fig. 3-30. (The inflow record for Lake Austin was utilized in Fig. 3-30, since monthly data for Town Lake are not available from the LCRA).

Data for Sta. 1402.08 in the lower pool of the impoundment do not appear to fluctuate in accordance with fluctuations in inflow. Phosphorus concentrations are generally similar to concentrations in the lower pool of Lake Austin, though elevated levels are occasionally observed. Chloride levels appear closely correlated to conditions in Lake Austin.

Town Lake is a throughflow-dominated reservoir with short hydraulic retention times during periods of upstream releases. As expected, concentrations appear to be effected primarily by external loadings from upstream releases.

3.4 NITROGEN DATA

Within the following section is presented a brief examination of available data for nitrogen species within the Highland Lakes. The available nitrogen data set is primarily derived from the routine surveys of the TDWR (TNRIS, 1983a). Generally, the TDWR collects samples for analysis of nitrate nitrogen and ammonia nitrogen as a companion exercise to the collection of phosphorus data. The emphasis of the present study is upon the analysis of phosphorus data; the following information is provided solely to identify typical levels of inorganic nitrogen in the Highland Lakes.

Reported TDWR detection limits for nitrogen data varied from 0.020-0.100 mg/l for ammonia nitrogen and 0.010-0.030 mg/l for nitrate nitrogen. For plotting purposes, the inorganic nitrogen level was set at the lowest reported limit when both ammonia nitrogen and nitrate nitrogen were below detection limits. Ammonia nitrogen data were frequently below detectable levels throughout the series of reservoirs.

3.4.1 Analysis of Spatial Trends

Several of the data sets examined for spatial trends of phosphorus and chloride distribution, described previously in Sec. 3.2, were analyzed for trends in inorganic nitrogen levels. For the present analysis, which utilizes primarily data from the TDWR, inorganic nitrogen is defined as the sum of nitrate nitrogen and ammonia nitrogen. It is assumed that the magnitude of nitrite nitrogen concentration is negligible (though data are generally not available).

3.4.1.1 Surveys Under Unstratified Conditions

Several surveys were selected for spatial analysis under the conditions of a relatively well-mixed water column.

Survey of 1-2 October 1974

Area streamflow data antecedent to the survey of 1-2 October 1974 were presented in Sec. 3.2. The data indicated a substantial influx of water from area tributaries. Marked fluctuation in inorganic nitrogen levels was observed throughout the chain of Highland Lakes, as displayed in Fig. 3-31. The inorganic nitrogen level decreased from 0.29 mg/l in the headwaters of Lake Buchanan to a level below the detection limit in the lower pool. A substantial increase in concentration was observed in Lake LBJ, with a level of 0.29 mg/l inorganic nitrogen detected below the confluence with the Llano River. The inorganic nitrogen level in the lower pool of Lake LBJ was 0.19 mg/l. A slight decrease in concentration was observed in Lake Marble Falls. In Lake Travis, inorganic nitrogen levels increased to 0.35 mg/l above the confluence with the Pedernales River, but concentrations declined to 0.06 mg/l in the lower pool of the reservoir.

Survey of 13-14 January 1976

Streamflow records for area watercourses during the antecedent period prior to the survey of 13-14 January 1976 were discussed in Sec. 3.2. A relatively stable flow regime was indicated for area tributaries. The inorganic nitrogen concentration decreased within Lake Buchanan and Inks Lake, as shown in Fig. 3-32. The level rose to 0.15 mg/l inorganic nitrogen in Lake LBJ, decreased slightly in Lake Marble Falls, and decreased in the upper region of Lake Travis. The inorganic nitrogen level increased downstream in Lake Travis, with a concentration of 0.16 mg/l observed in the lower pool. A decrease in inorganic nitrogen concentration was observed through Lake Austin.

Survey of 17-18 February 1981

The attendant streamflow data for the survey of 17-18 February 1981 were described in Sec. 3.2. The data indicated a period of relatively stable streamflow on all major area tributaries. Concentrations of inorganic nitrogen are displayed in Fig. 3-33. Inorganic nitrogen concentrations fluctuate to a greater extent than either total phosphorus or chloride levels. The inorganic nitrogen level demonstrates a decrease from the headwaters to the lower pool of Lake Buchanan. Levels subsequently increased to 0.36 mg/l inorganic nitrogen within Lakes LBJ and Marble Falls. The concentration decreased in the upper reach of Lake Travis, then steadily increased to a level of 0.31 mg/l inorganic nitrogen in the lower pool. A decrease in concentration was observed in Lake Austin, with a subsequent increase in Town Lake.

3.4.1.2 Surveys Under Stratified Conditions

Two warm weather surveys were analyzed in order to compare the longitudinal distribution of inorganic nitrogen with the cold weather data sets described previously.

Survey of 18-19 July 1974

Streamflow records for a thirty-day antecedent period prior to the survey of 18-19 July 1974 were displayed in Sec. 3.2. A period of stable, low streamflow was recorded on all area tributaries. As displayed in Fig. 3-34, all measurements of inorganic nitrogen in the Highland Lakes from the headwaters of Lake Buchanan to the lower pool of Lake Travis indicated concentrations at or below analytical detection limits (0.03 mg/l for nitrate nitrogen, 0.1 mg/l for ammonia nitrogen). Inorganic nitrogen levels slightly above the detection limit were observed in the upper reaches of Lake Austin, though the maximum level detected was only 0.05 mg/l.

Survey of 27-28 July 1981

Antecedent flow conditions for area steams prior to the survey of 27-28 July 1981 were described in Sec. 3.2. An antecedent period of generally decreasing streamflow rates was observed. As shown in Fig. 3-35, inorganic nitrogen levels in the Highland Lakes were at or slightly above detectable limits from the headwaters of Lake Buchanan to the lower pool of Lake Travis (0.01 mg/l for nitrate nitrogen, 0.02 mg/l for ammonia nitrogen). A slight increase in inorganic nitrogen levels was observed in Lake Austin, with a concentration of 0.1 mg/l detected in the lower pool. The inorganic nitrogen concentration increased to 0.24 mg/l in the lower pool of Town Lake.

3.4.2 Analysis of Temporal Trends

The data base for inorganic nitrogen in the Highland Lakes was examined for evidence of temporal trends, in a manner similar to the phosphorus analysis described in Sec. 3.3. TDWR data for ammonia nitrogen and nitrate nitrogen collected in the lower pool of each reservoir were plotted versus month of the year for evidence of seasonality. The data are displayed in Fig. 3-36. Seasonal trends are generally not evident, although at the majority of the stations inorganic nitrogen concentrations are depressed during summer months. The vast majority of ammonia nitrogen measurements found concentrations below the detectable limit of 0.1 mg/l. (Data from more recent surveys feature a lower limit of detection, i.e., 0.02 mg/l.) Within Lake Austin and Lake Travis, inorganic nitrogen levels are typically below 0.3 mg/l. Within Lake Marble Falls, the majority of inorganic nitrogen measurements fall below 0.3 mg/l, but there is an increase of observations in the range of 0.3-0.4 mg/l. The distribution of observations in Lake LBJ is similar to that encountered in Lake Marble Falls. Concentrations in excess of 0.2 mg/l occur only infrequently within Lake Inks and Lake Buchanan. Data for the Colorado River above Lake Buchanan at Bend, Texas are also displayed in Fig. 3-36. Riverine levels of nitrate nitrogen are frequently substantially higher than those observed within the Highland Lakes, ranging up to 3.03 mg/l.

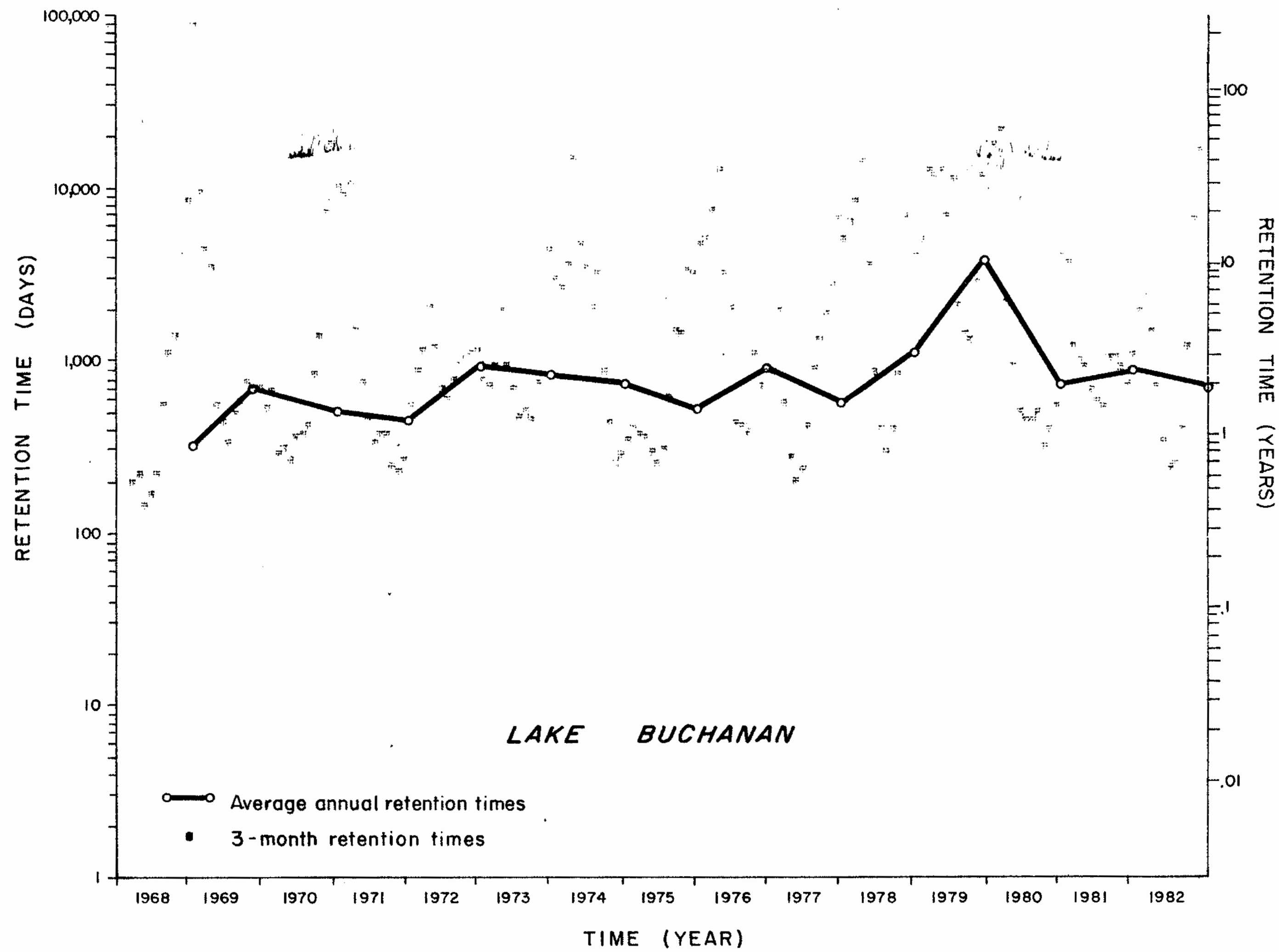


Fig. 3-1 HYDRAULIC RETENTION TIME, LAKE BUCHANAN, 1968-1982

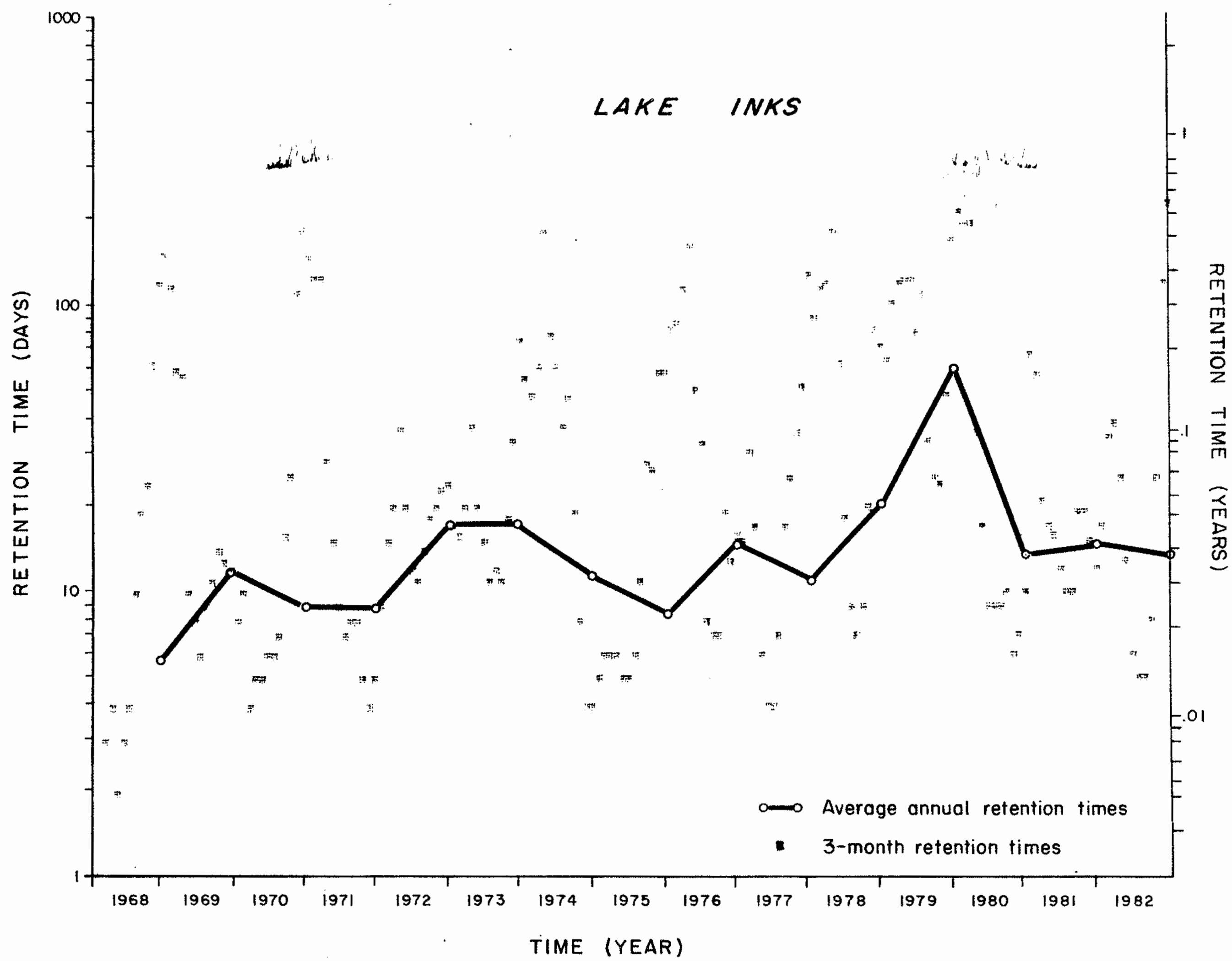


Fig. 3-2 HYDRAULIC RETENTION TIME, LAKE INKS, 1968-1982

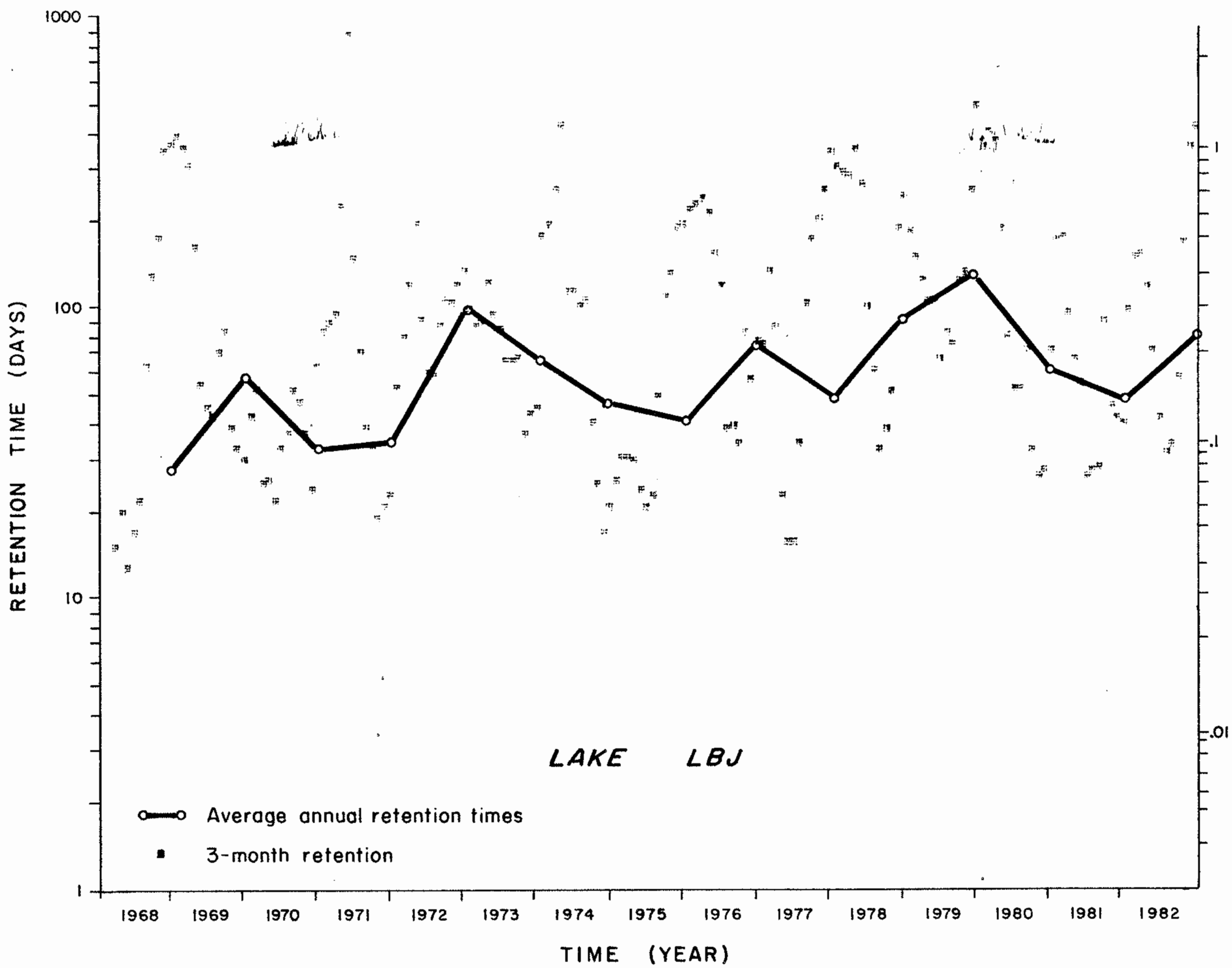


Fig. 3-3 HYDRAULIC RETENTION TIME, LAKE LBJ, 1968-1982

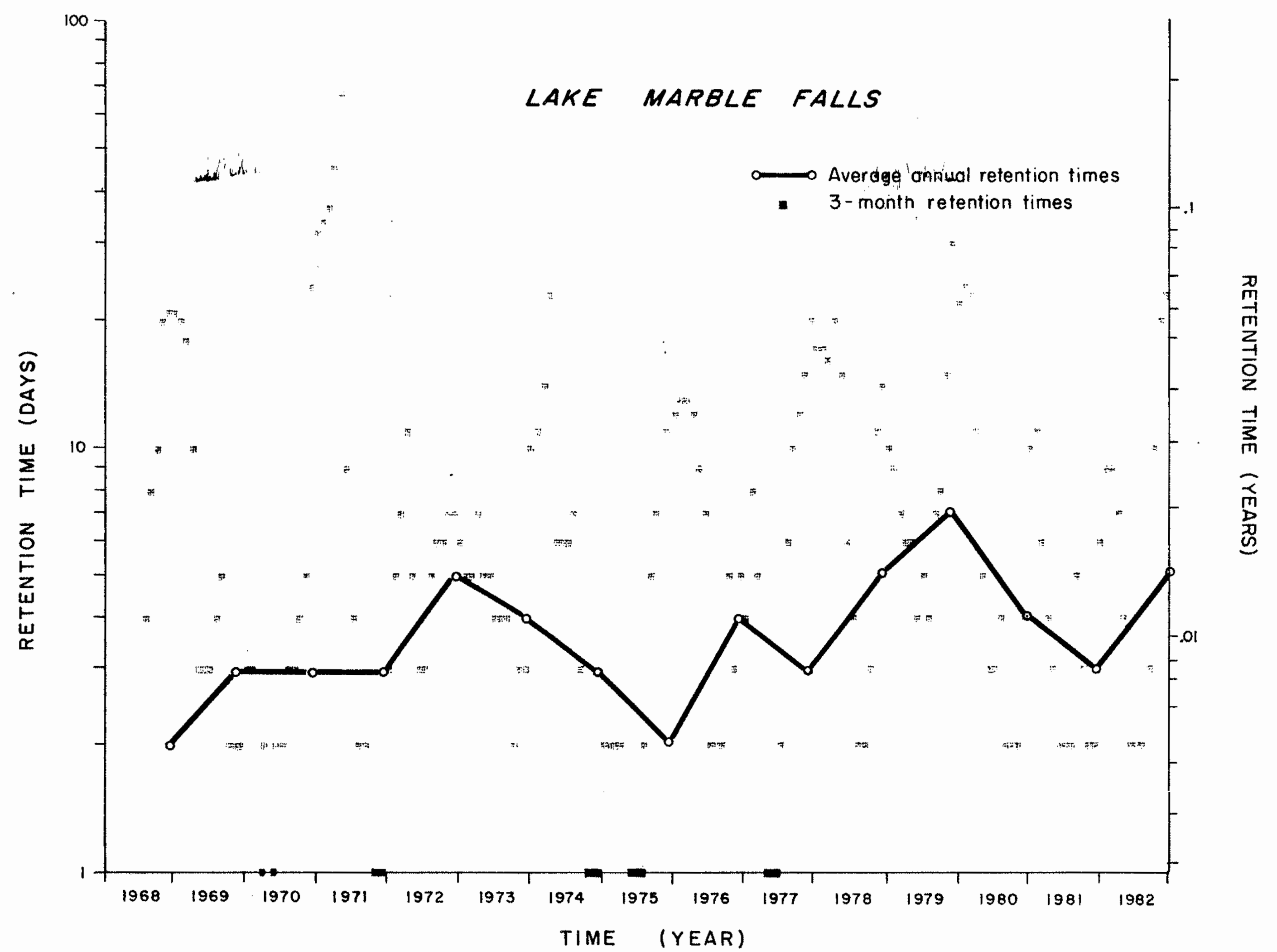


Fig. 3-4 HYDRAULIC RETENTION TIME, LAKE MARBLE FALLS, 1968–1982

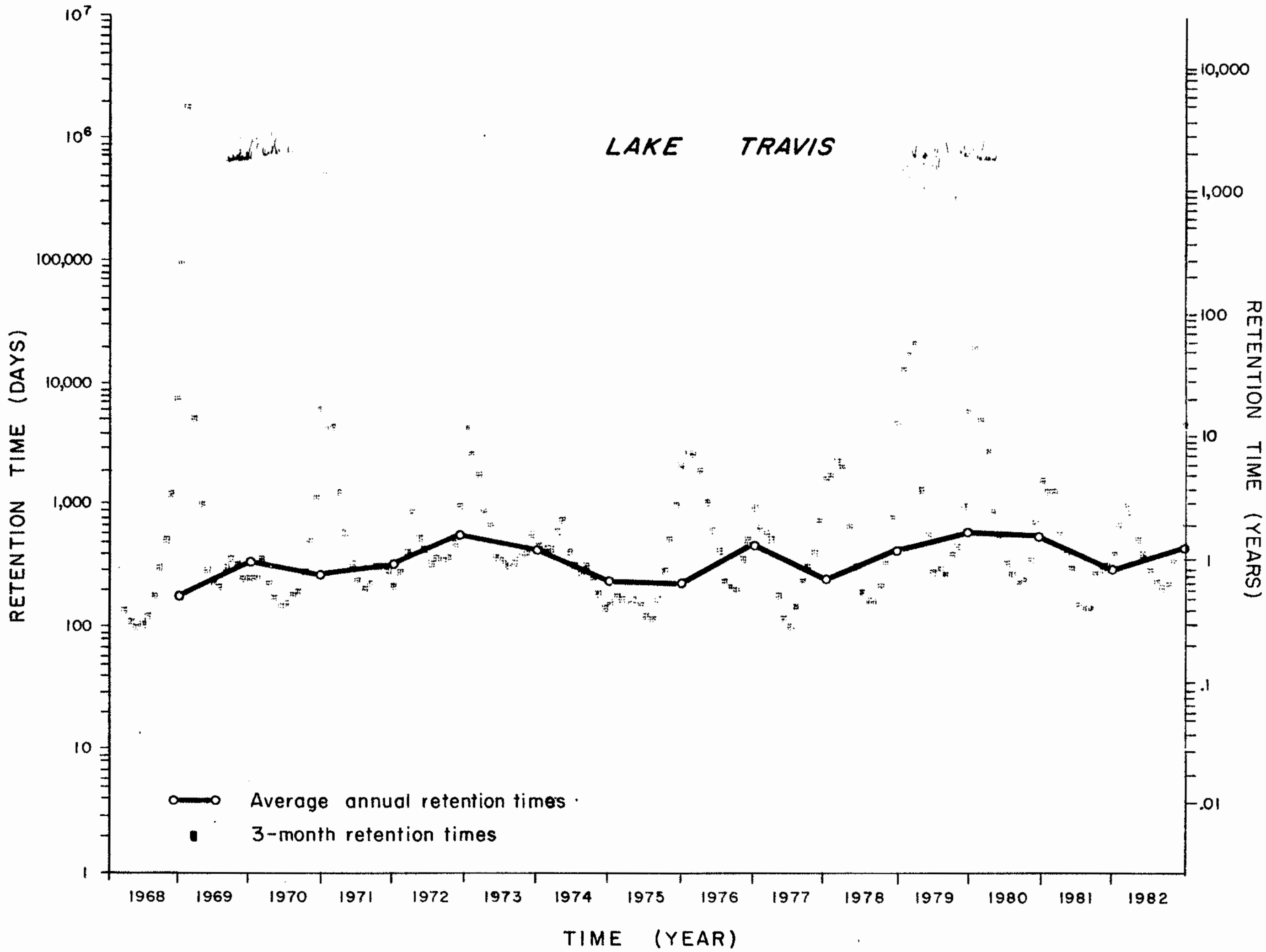


Fig. 3-5 HYDRAULIC RETENTION TIME, LAKE TRAVIS, 1968–1982

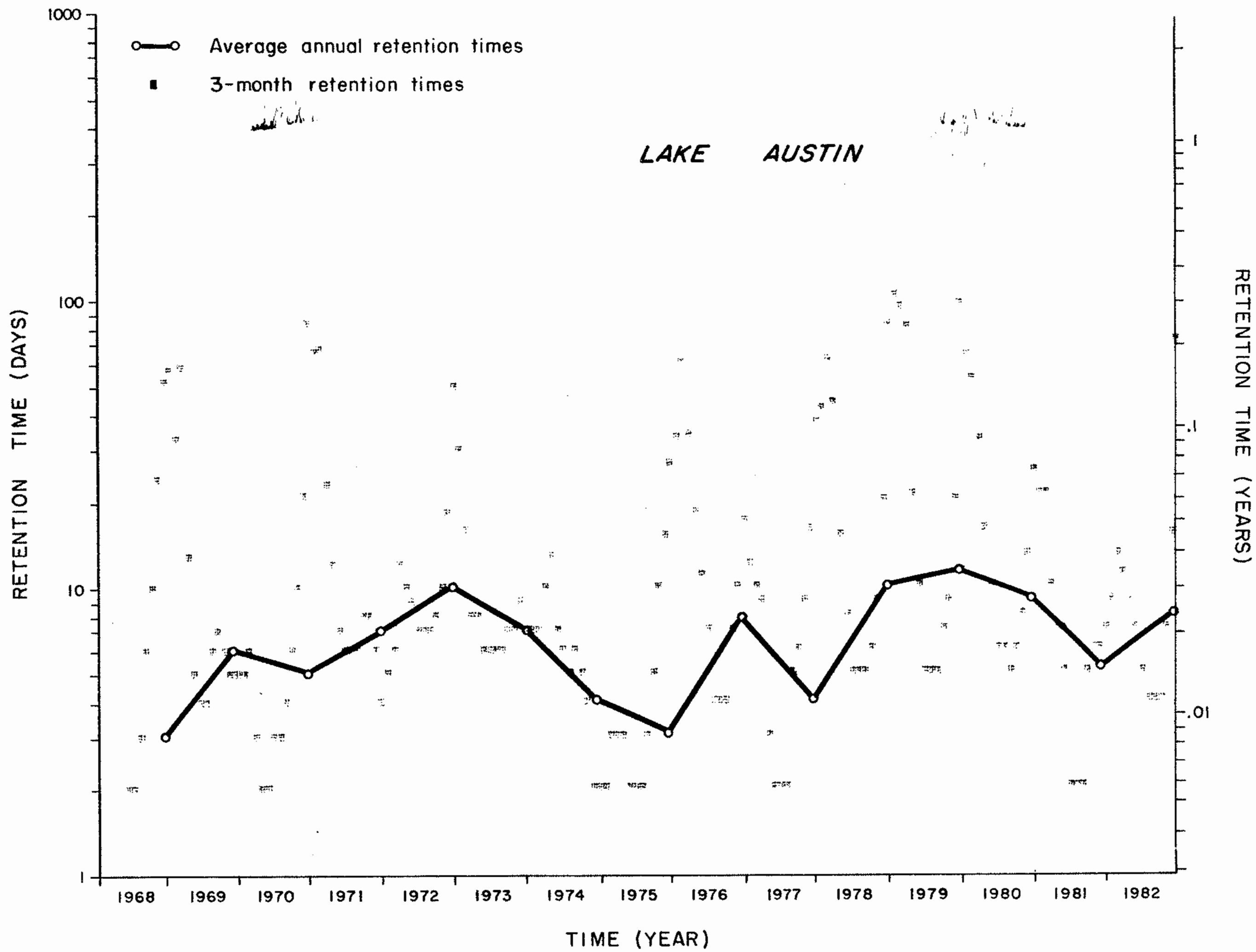


Fig. 3-6 HYDRAULIC RETENTION TIME, LAKE AUSTIN, 1968–1982

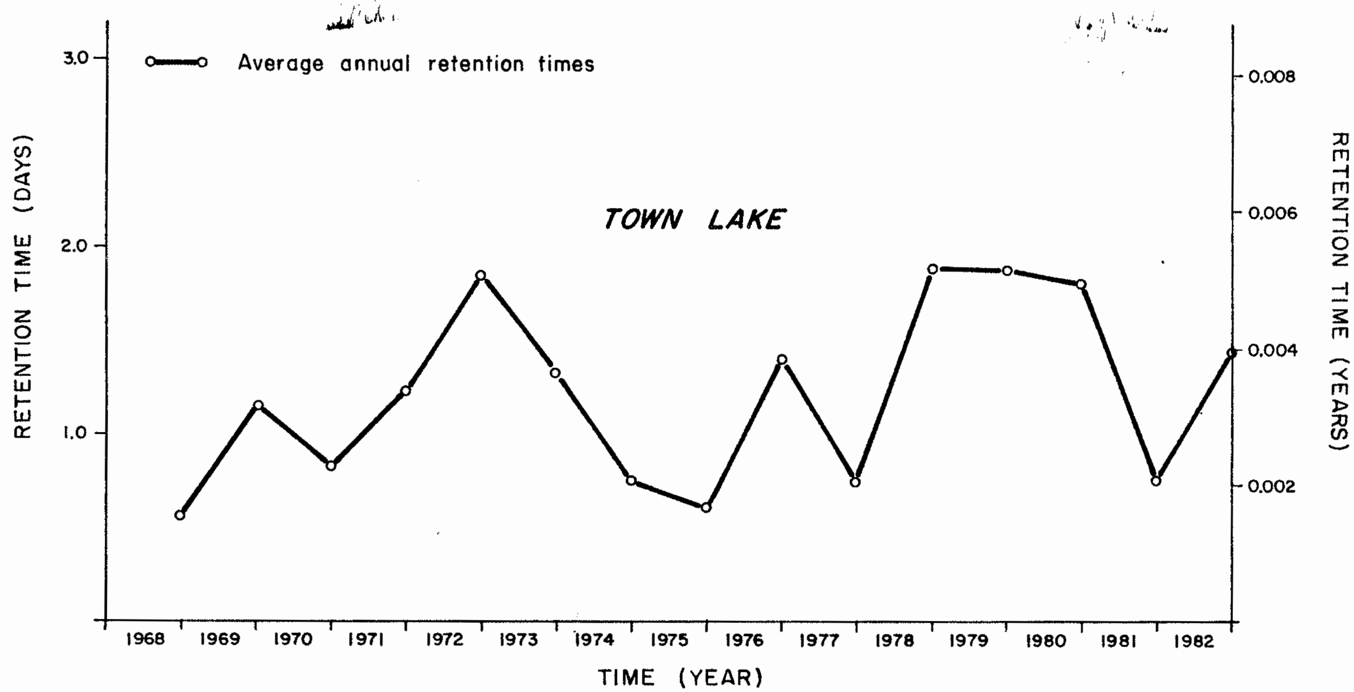


Fig. 3-7 HYDRAULIC RETENTION TIME, TOWN LAKE, 1968-1982

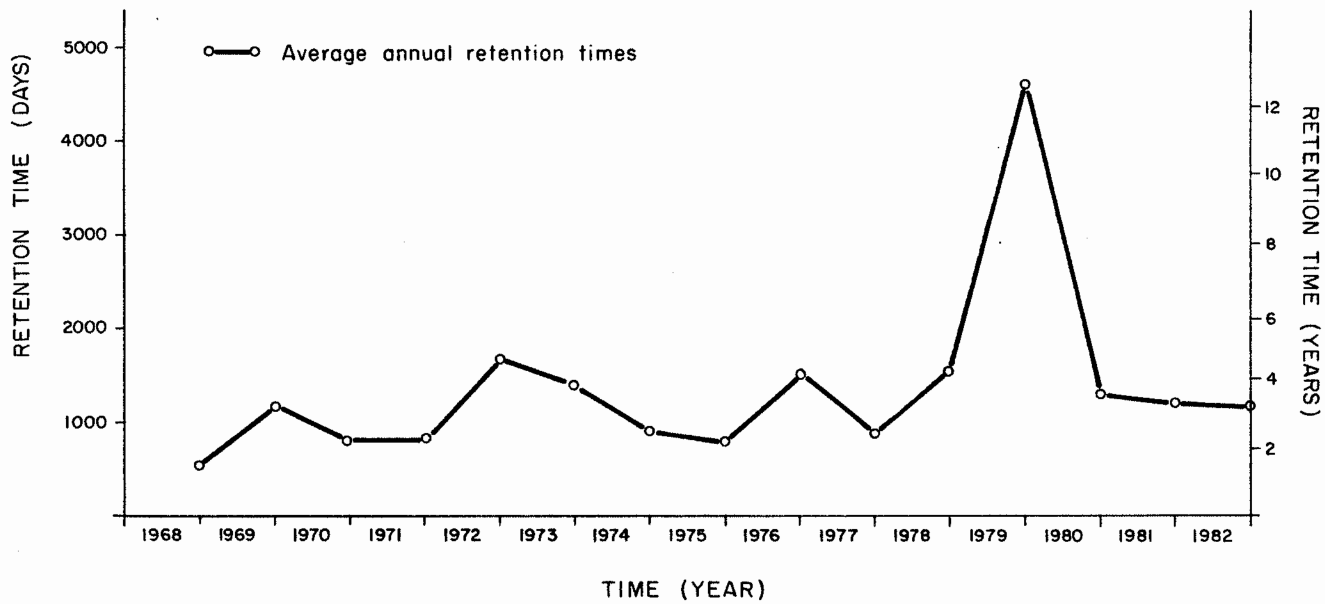


Fig. 3-8 CUMULATIVE HYDRAULIC RETENTION TIME, HIGHLAND LAKES, 1968-1982

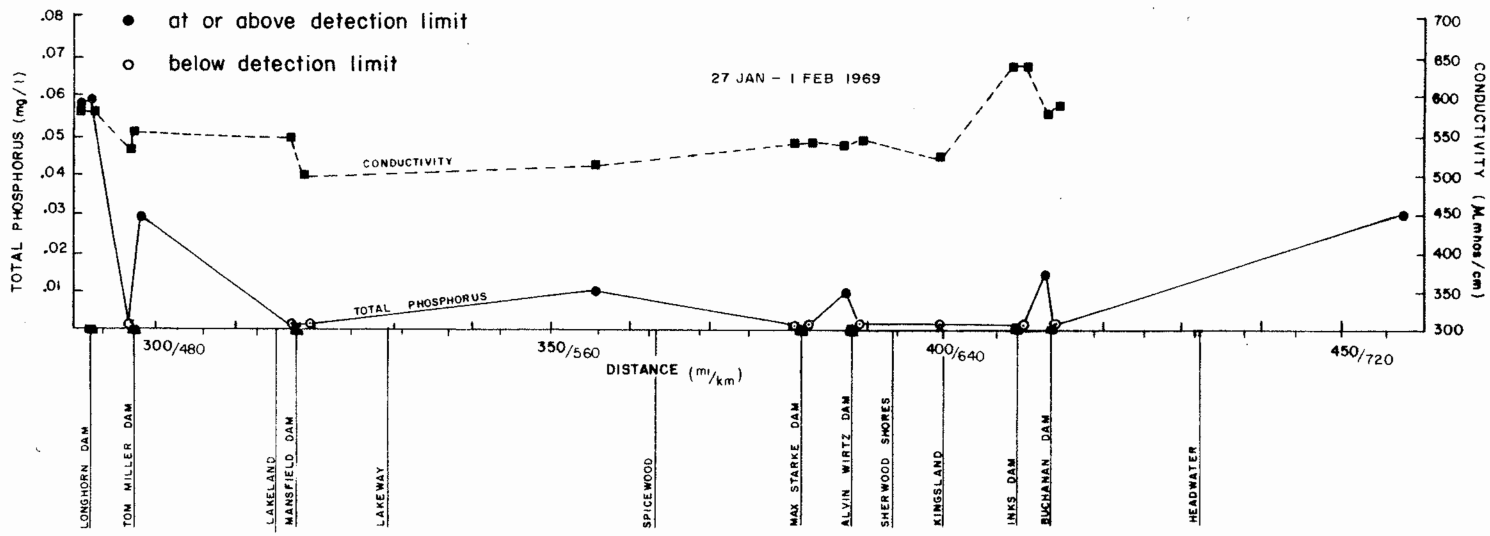


Fig. 3-9 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CONDUCTIVITY, 27 JANUARY– 1 FEBRUARY 1969

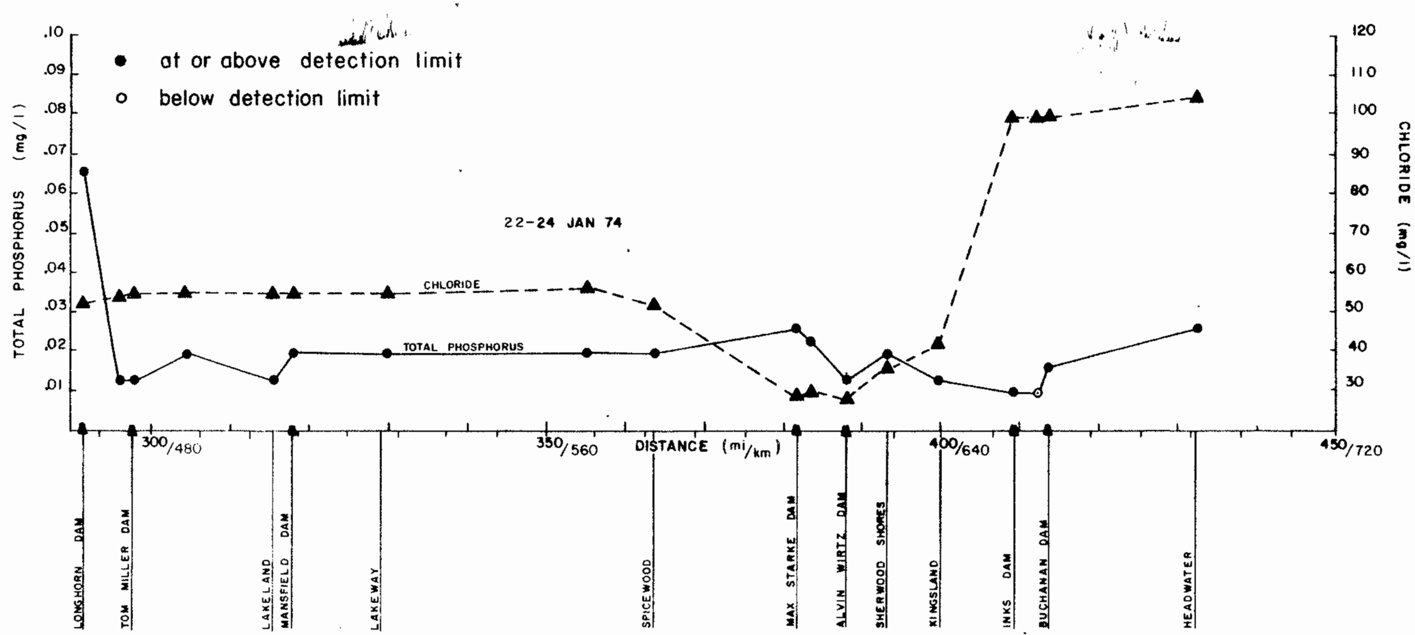


Fig. 3-10 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 22-24 JANUARY 1974

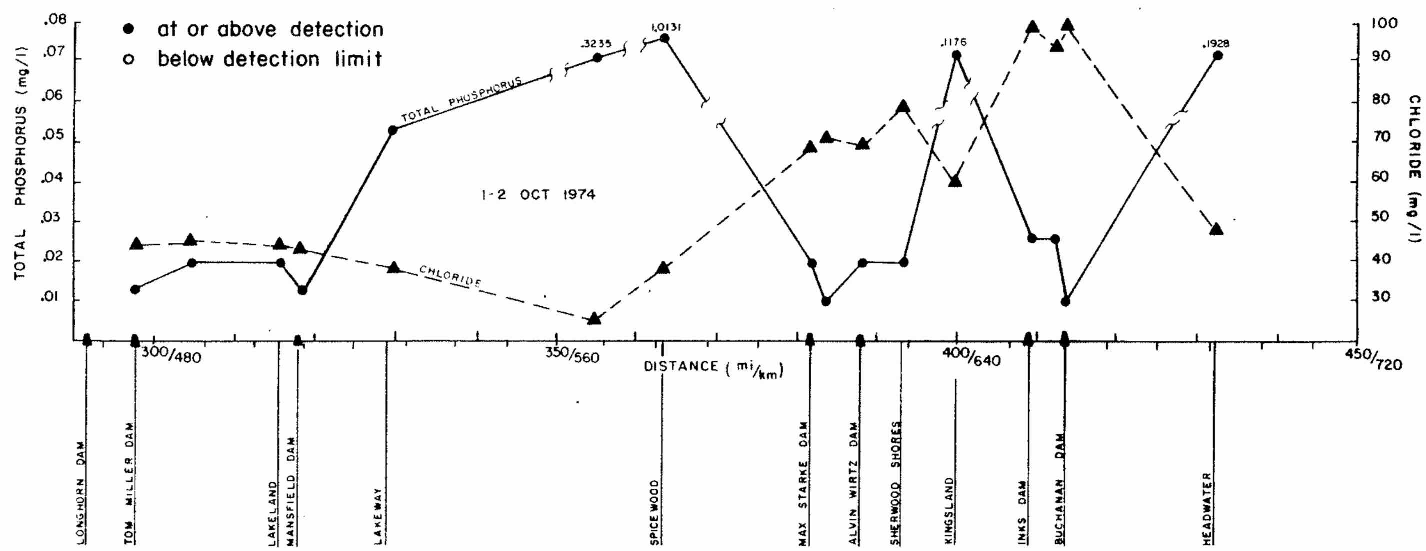


Fig. 3-11 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 1-2 OCTOBER 1974

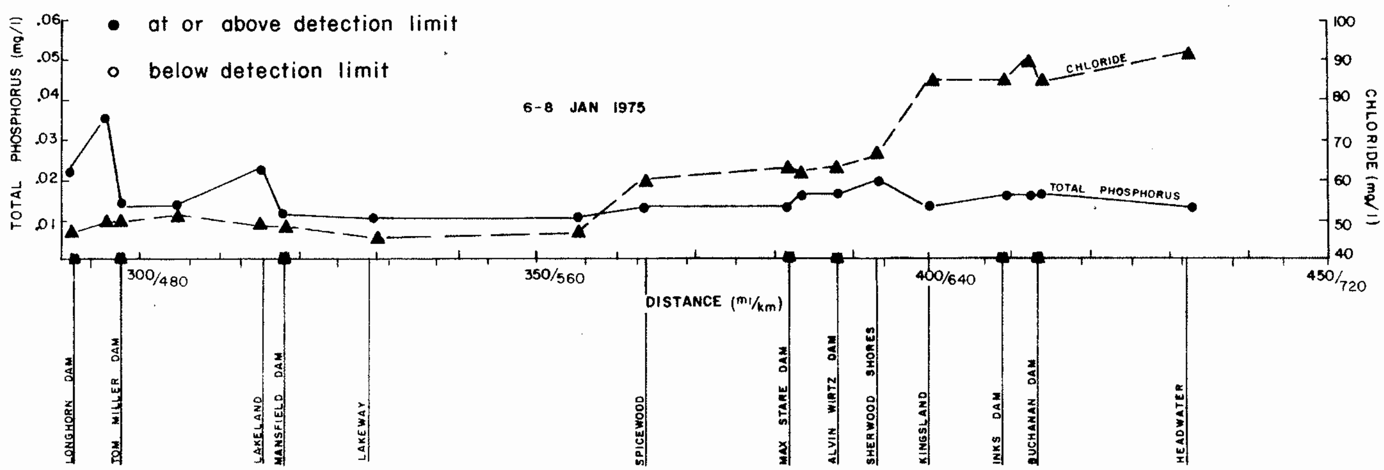


Fig. 3-12 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 6-8 JANUARY 1975

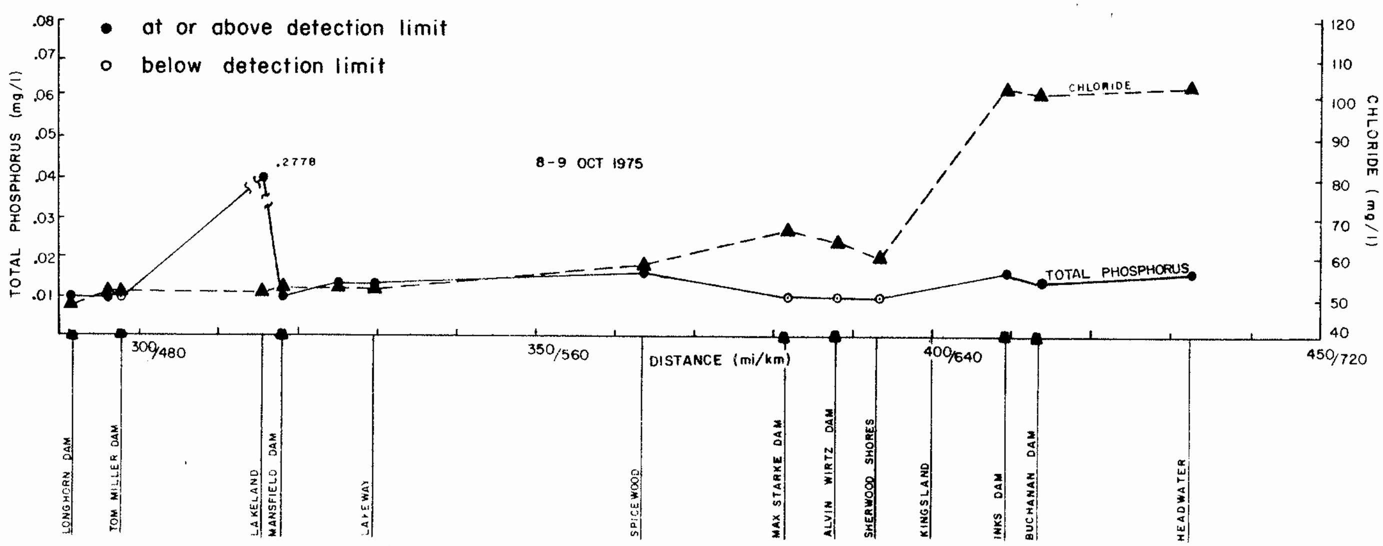


Fig. 3-13 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 8-9 OCTOBER 1975

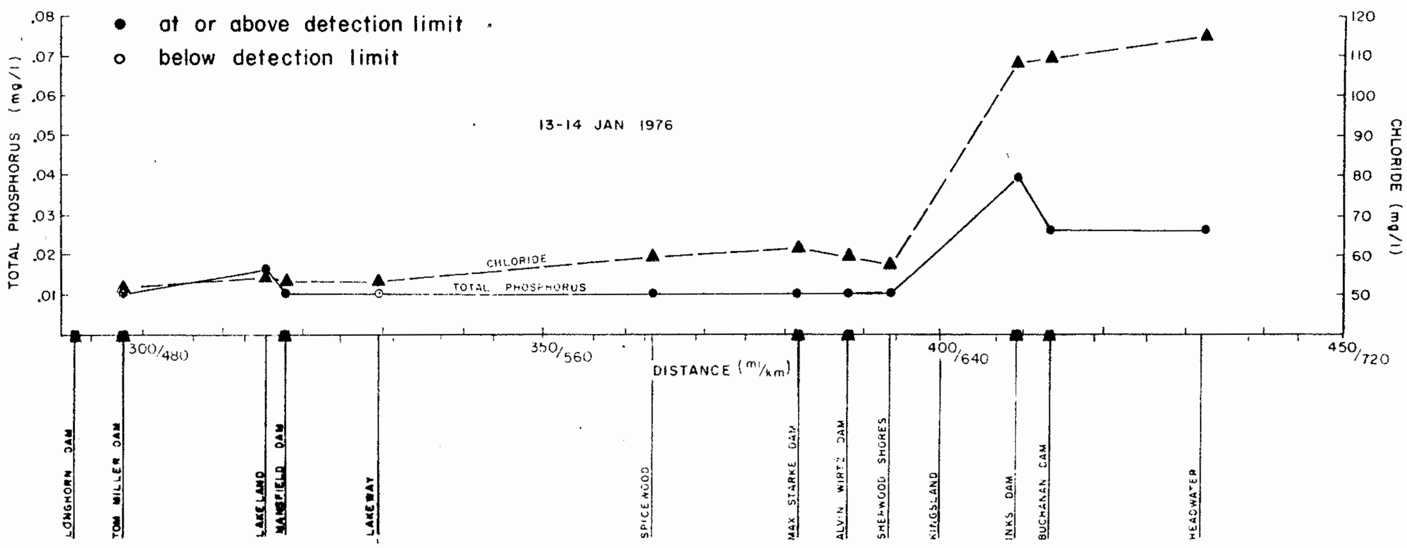


Fig. 3-14 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 13-14 JANUARY 1976

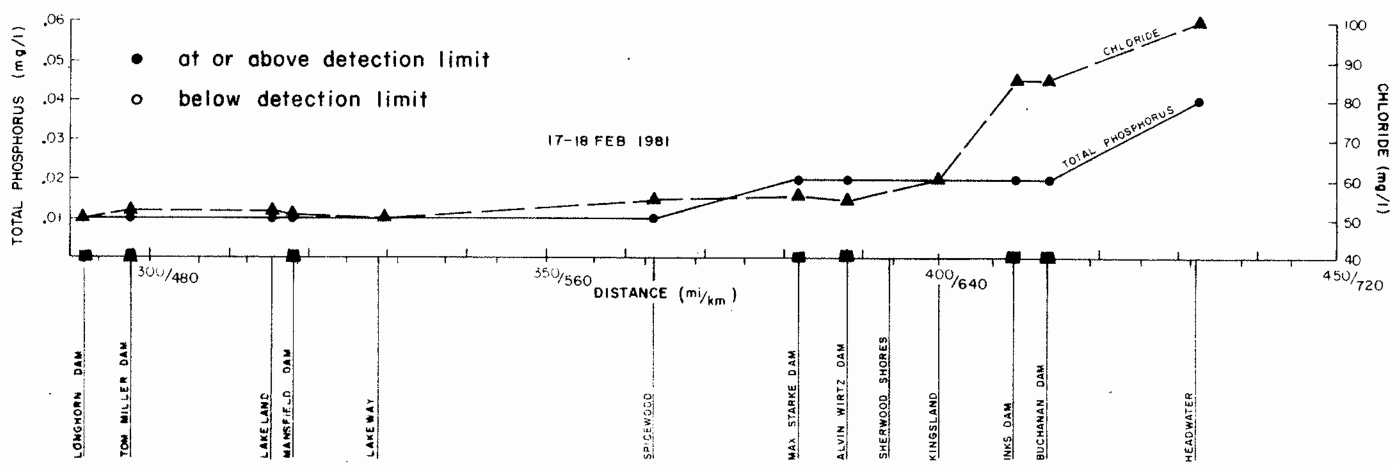


Fig. 3-15 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 17-18 FEBRUARY 1981

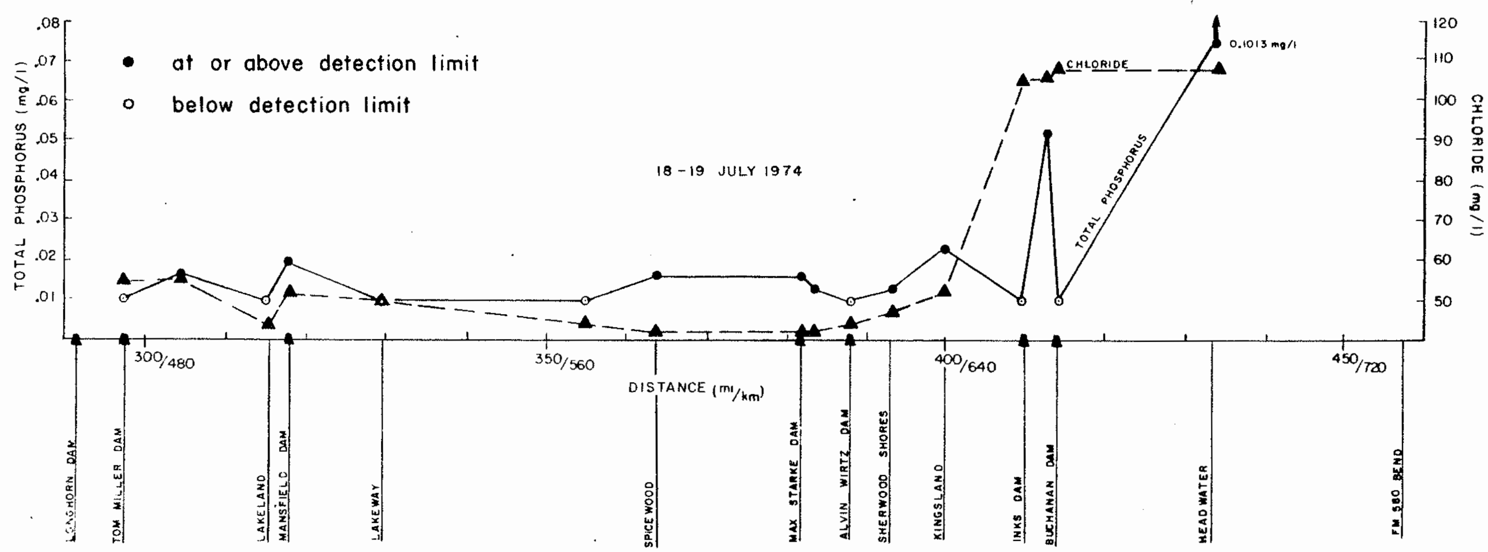


Fig. 3-16 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 18-19 JULY 1974

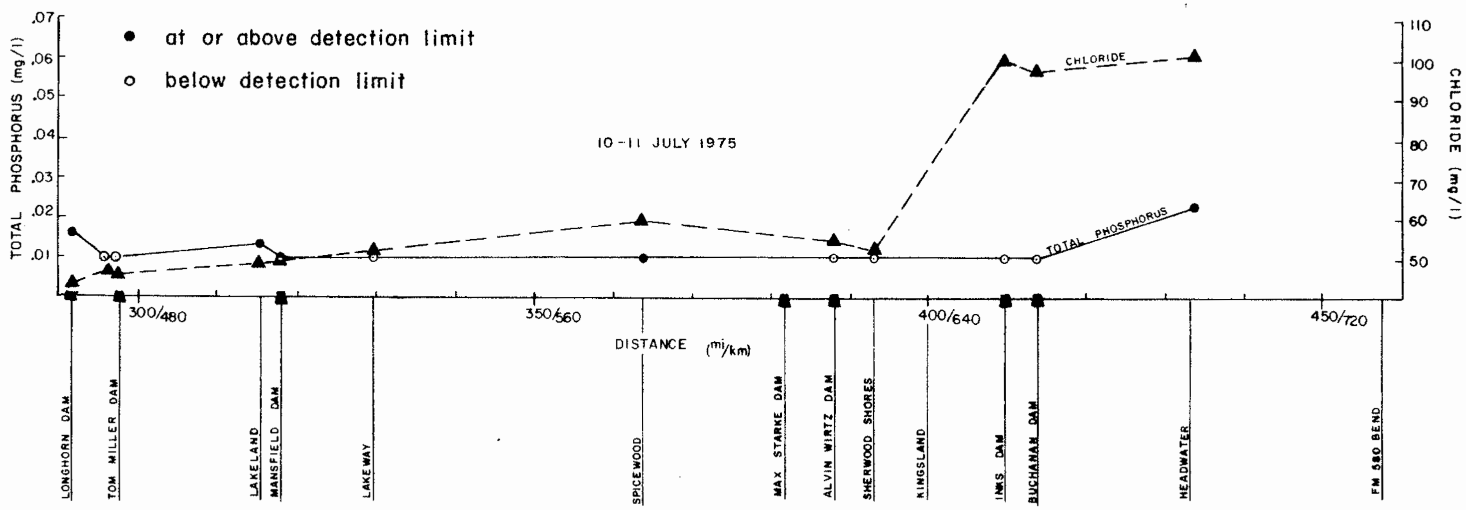


Fig. 3-17 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 10-11 JULY 1975

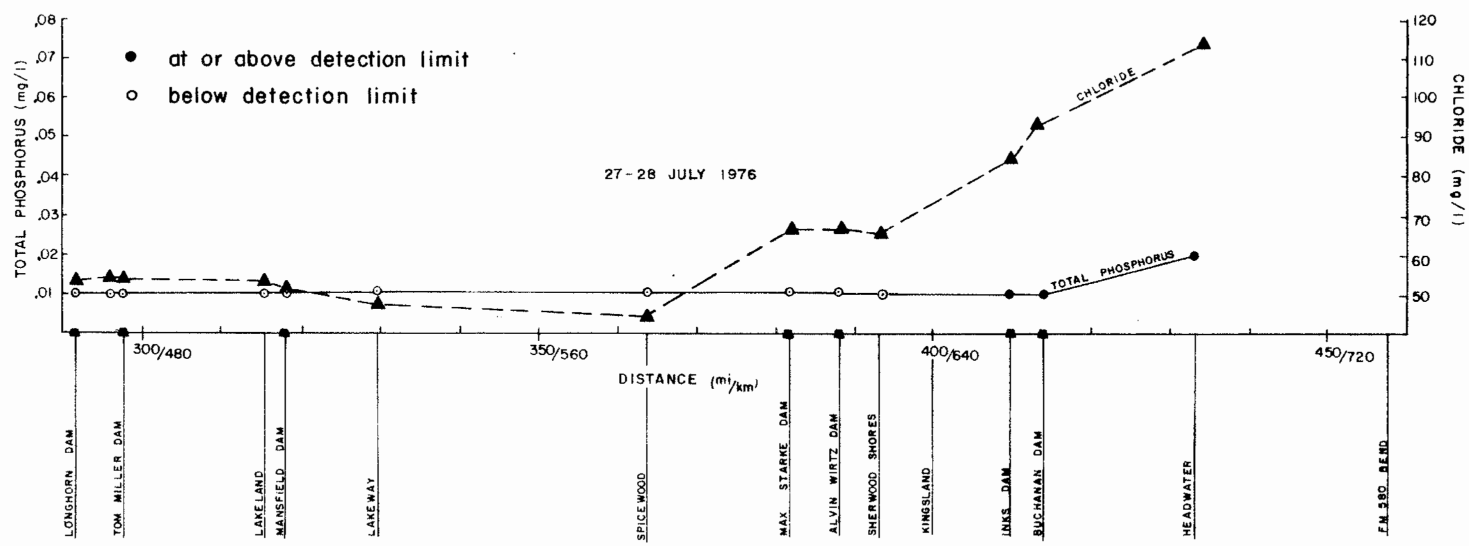


Fig. 3-18 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 27-28 JULY 1976

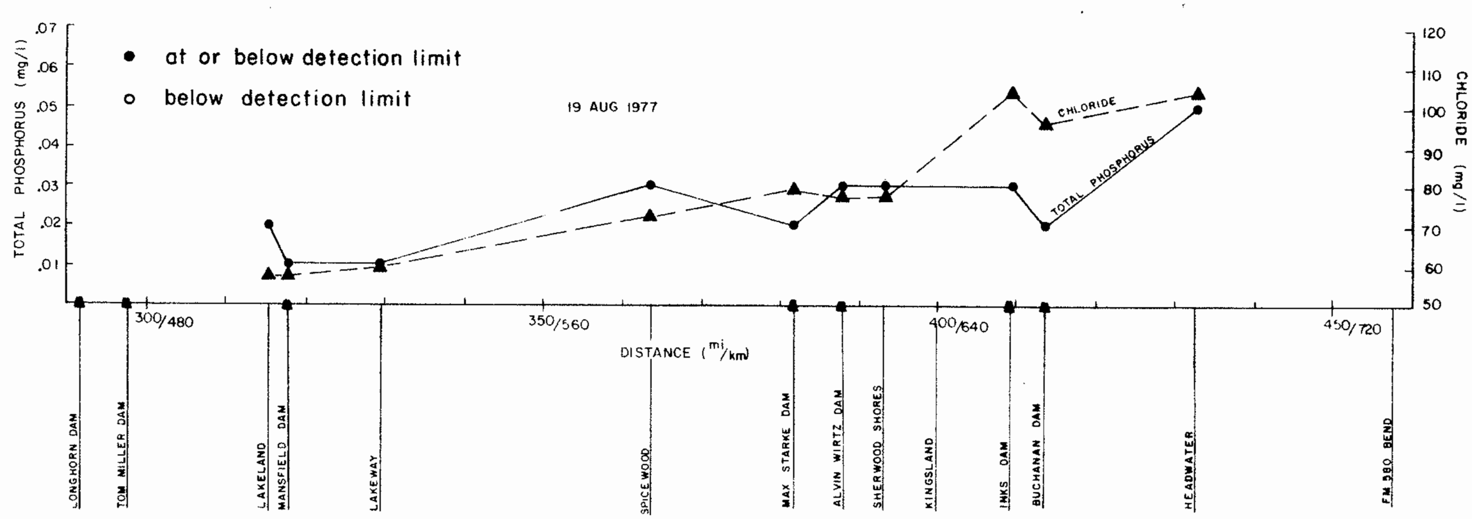


Fig. 3-19 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 19 AUGUST 1977

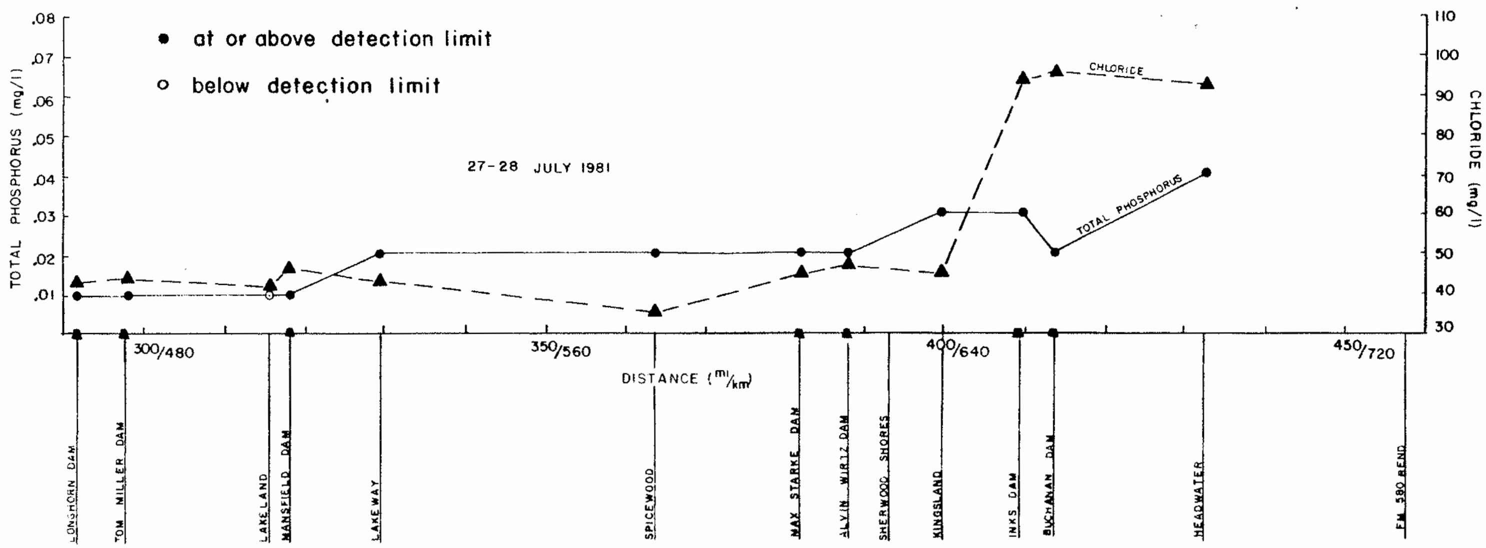


Fig. 3-20 LONGITUDINAL TRENDS OF TOTAL PHOSPHORUS AND CHLORIDE, 27-28 JULY 1981

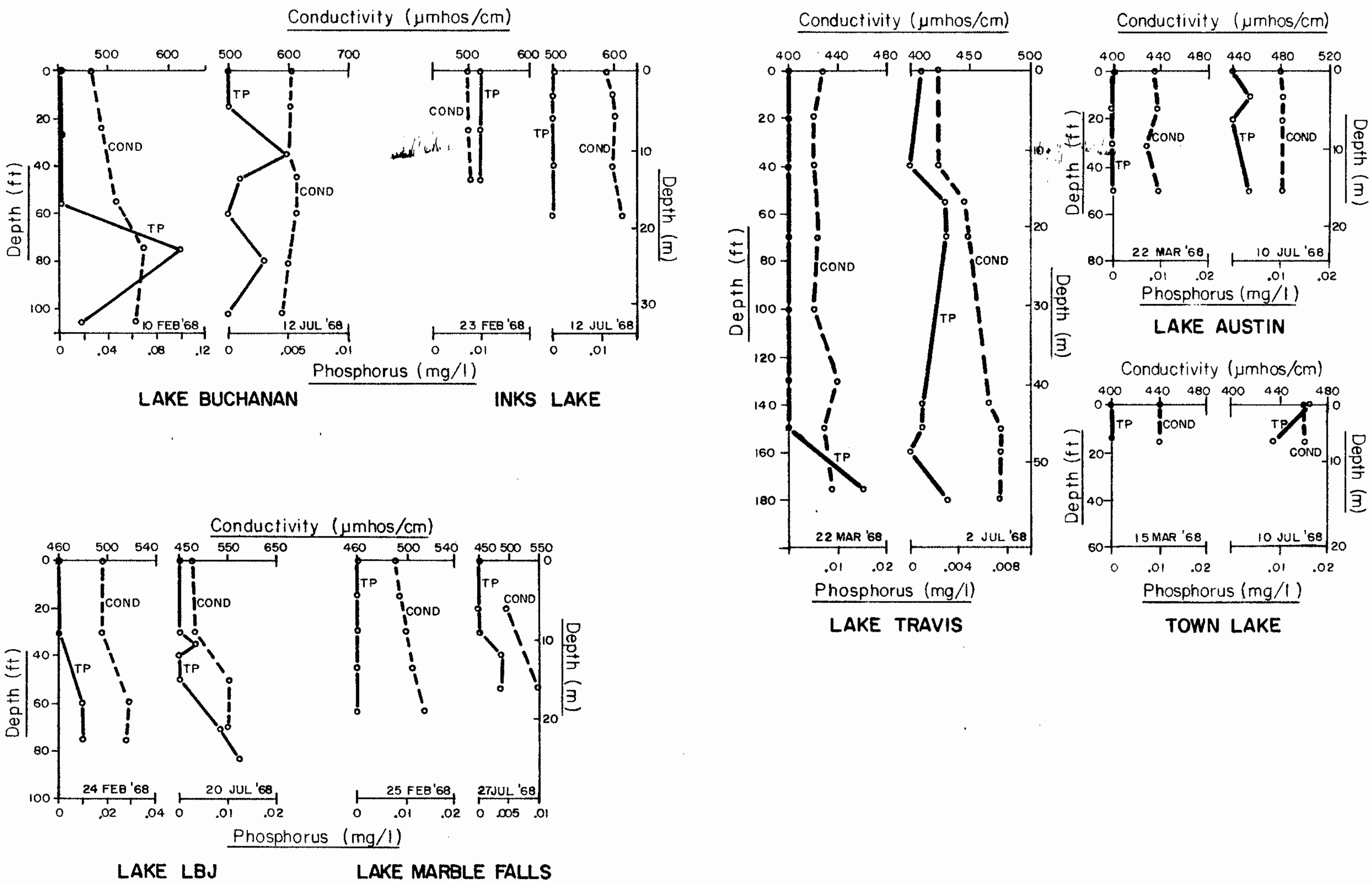


Fig. 3-21 VERTICAL PROFILES OF PHOSPHORUS AND CONDUCTIVITY, HIGHLAND LAKES, 1968

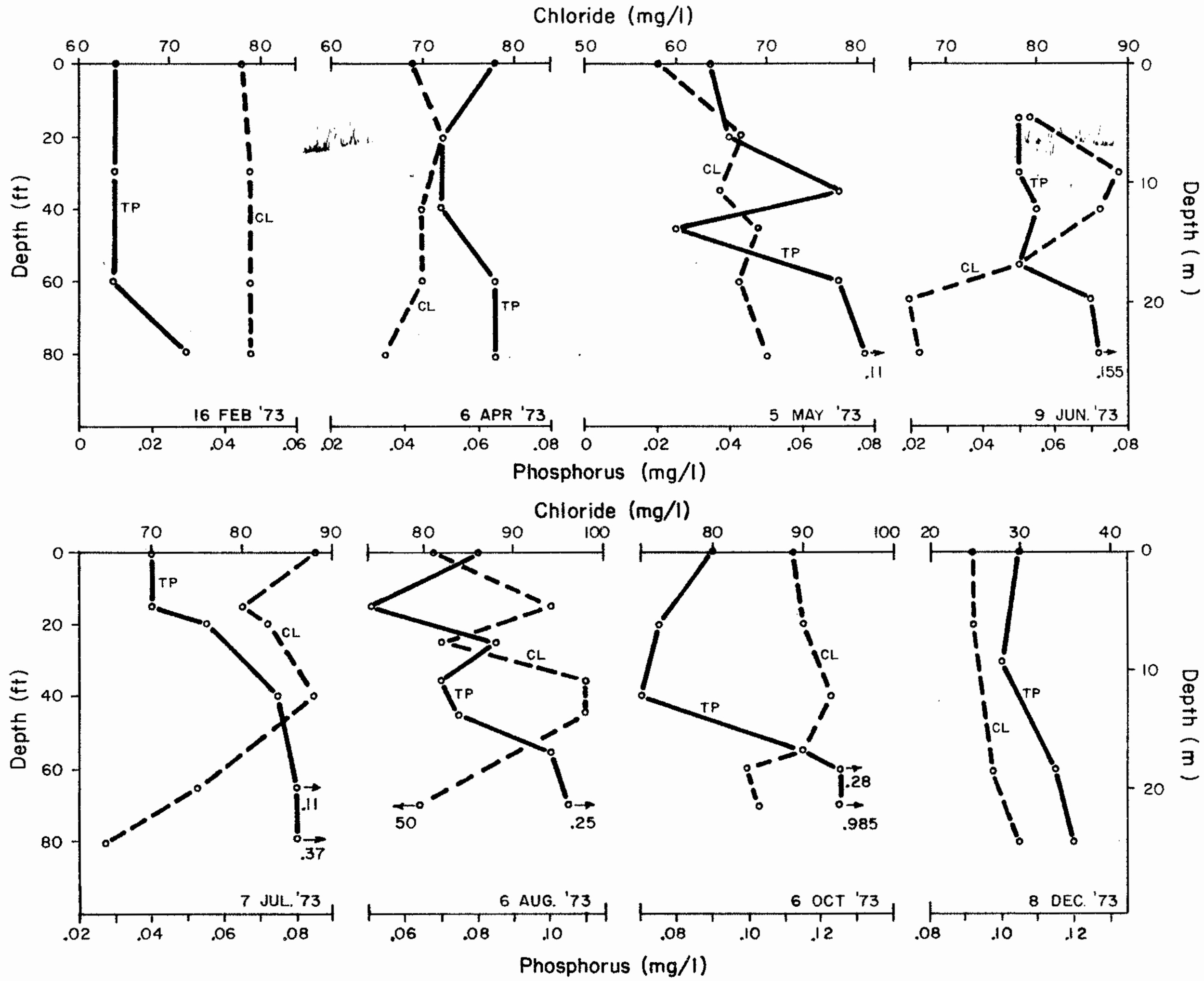


Fig. 3-22 VERTICAL PROFILES OF PHOSPHORUS AND CHLORIDE, LAKE LBJ POOL, 1973

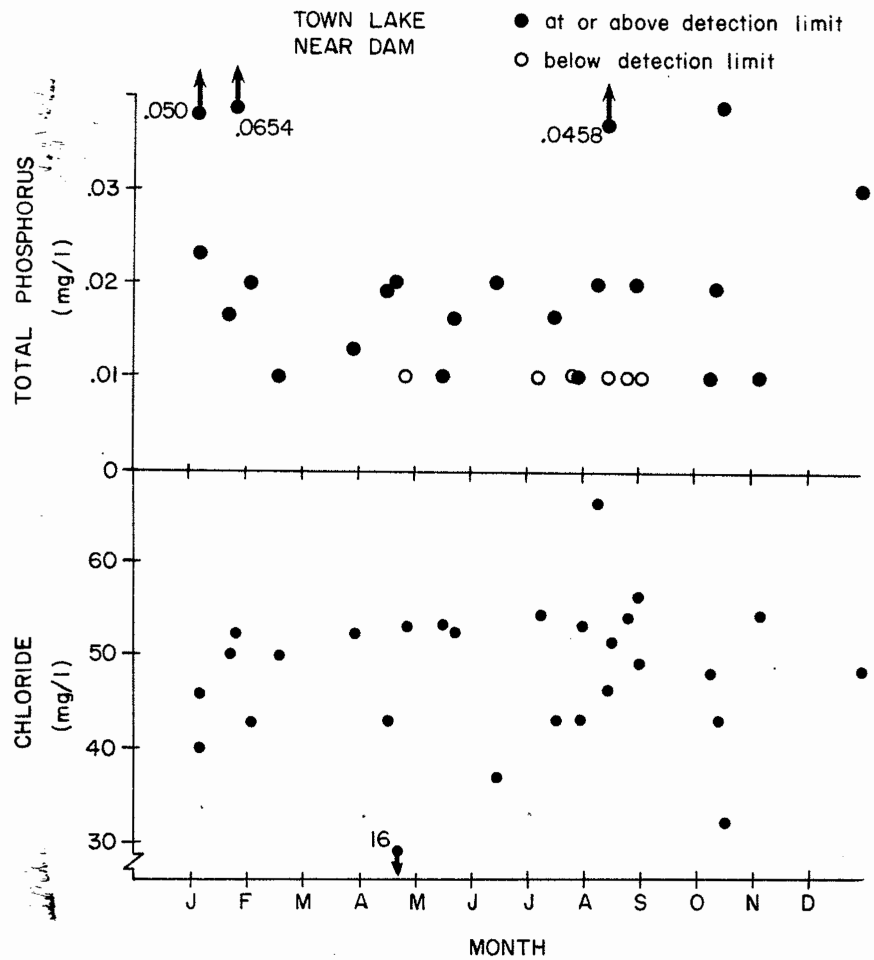


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(a) TOWN LAKE, ST. 1402.08

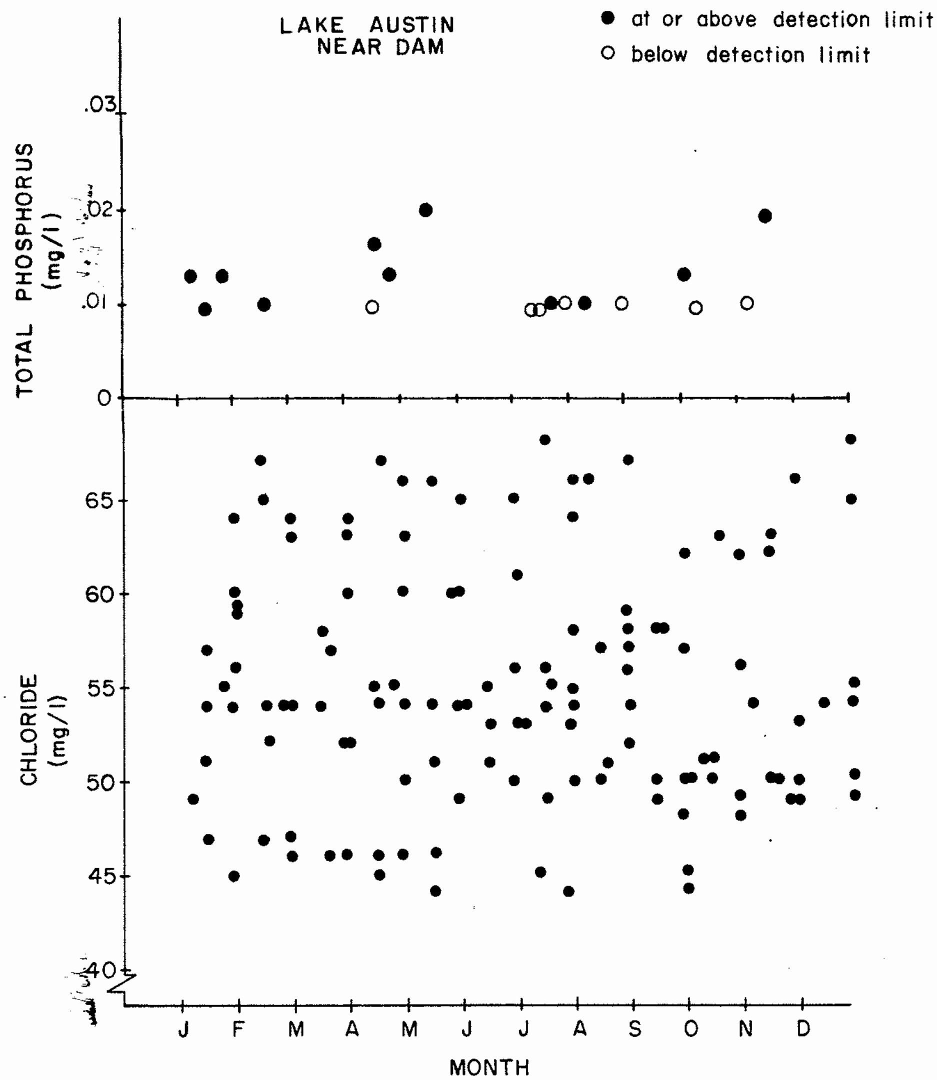


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION: (b) LAKE AUSTIN, ST. 1403.01

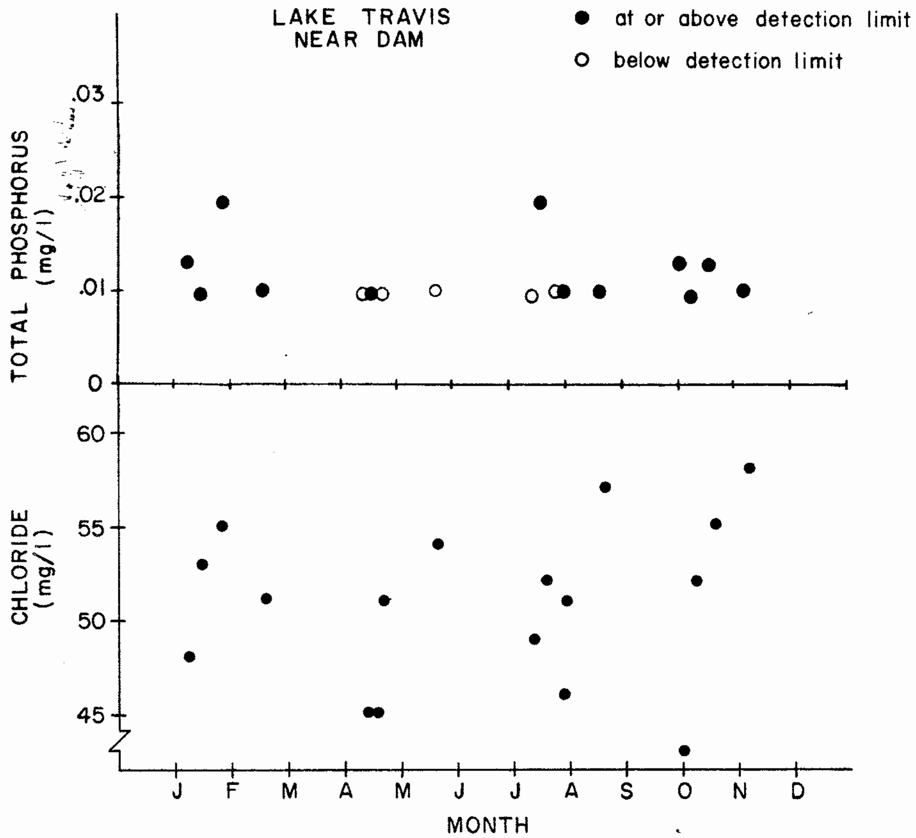


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(c) LAKE TRAVIS, ST. 1404.01

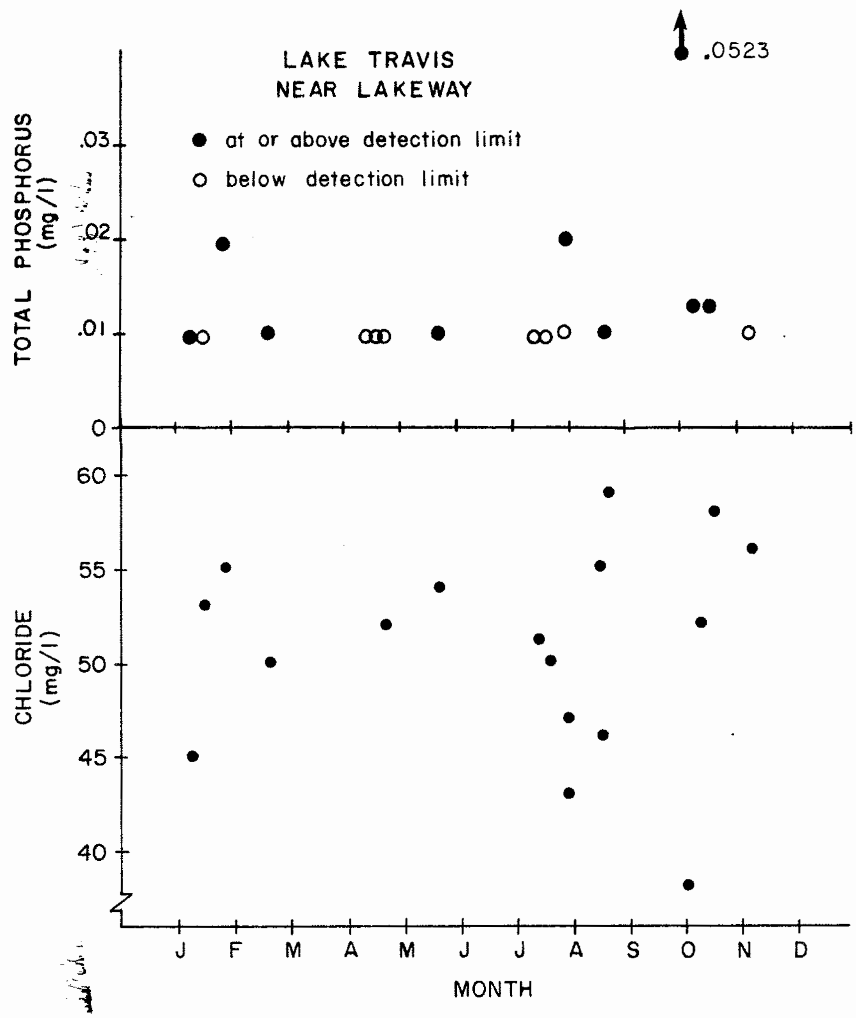


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(d) LAKE TRAVIS, ST. 1404.03

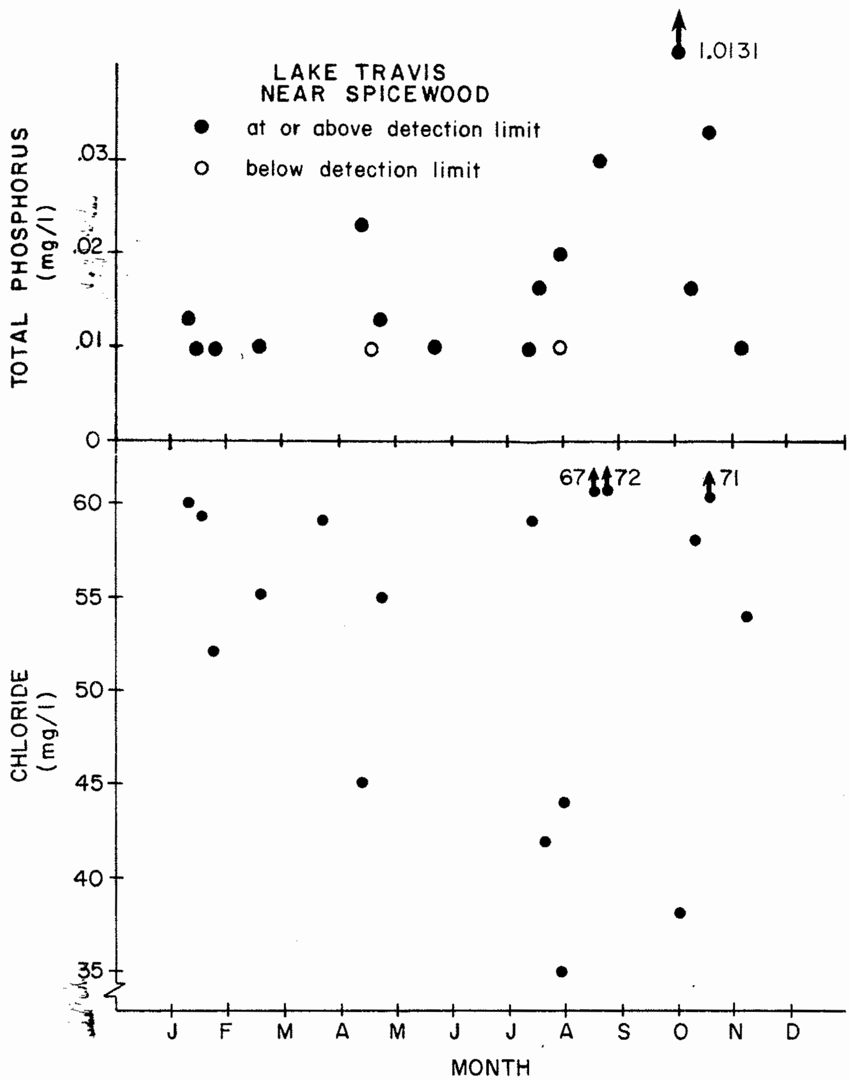


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(e) LAKE TRAVIS, ST. 1404.06

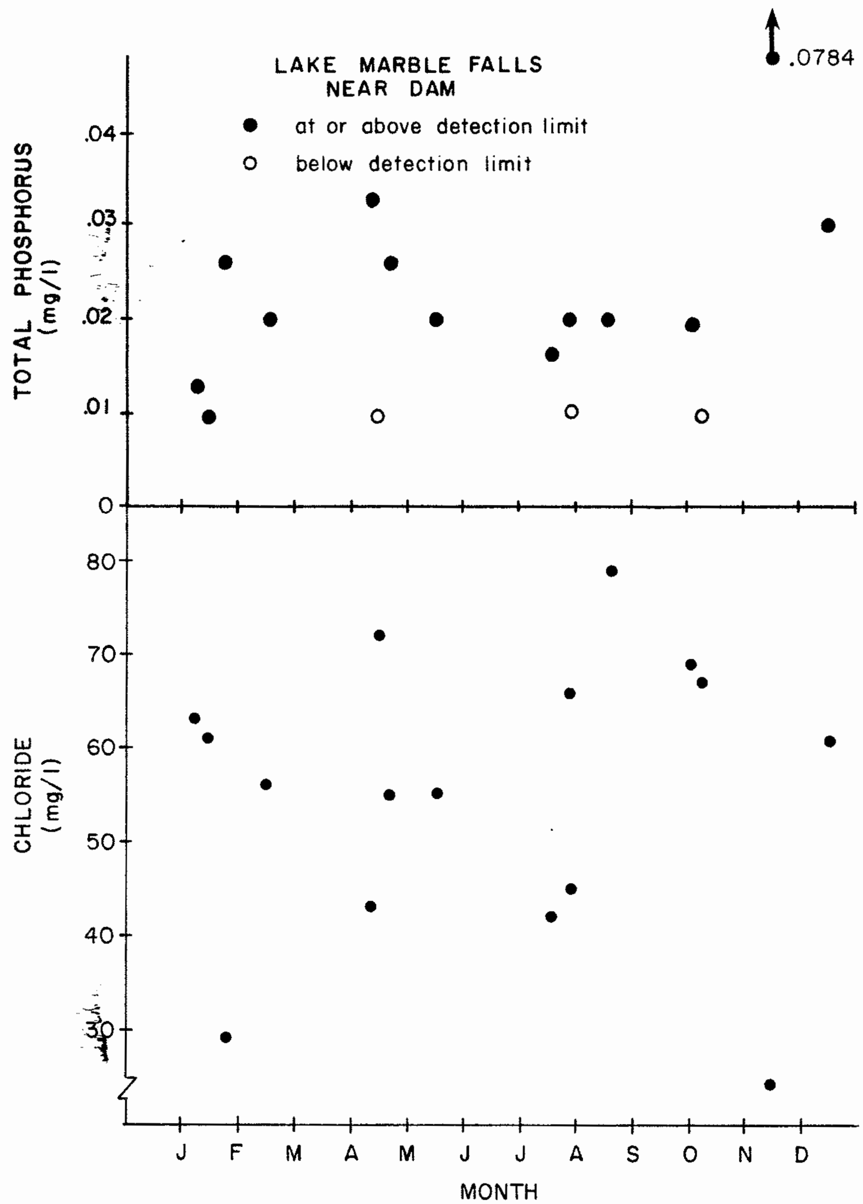


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(f) LAKE MARBLE FALLS, ST. 1405.01

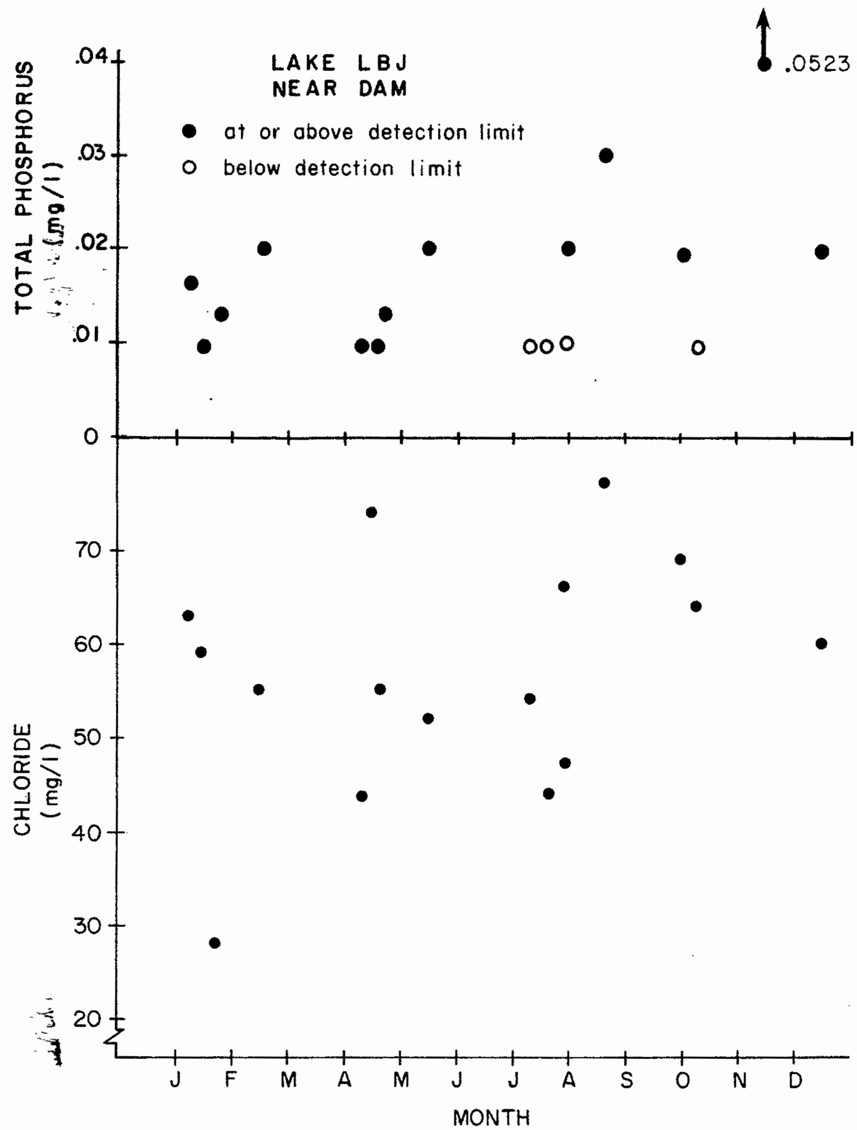


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(g) LAKE LBJ, ST. 1406.01

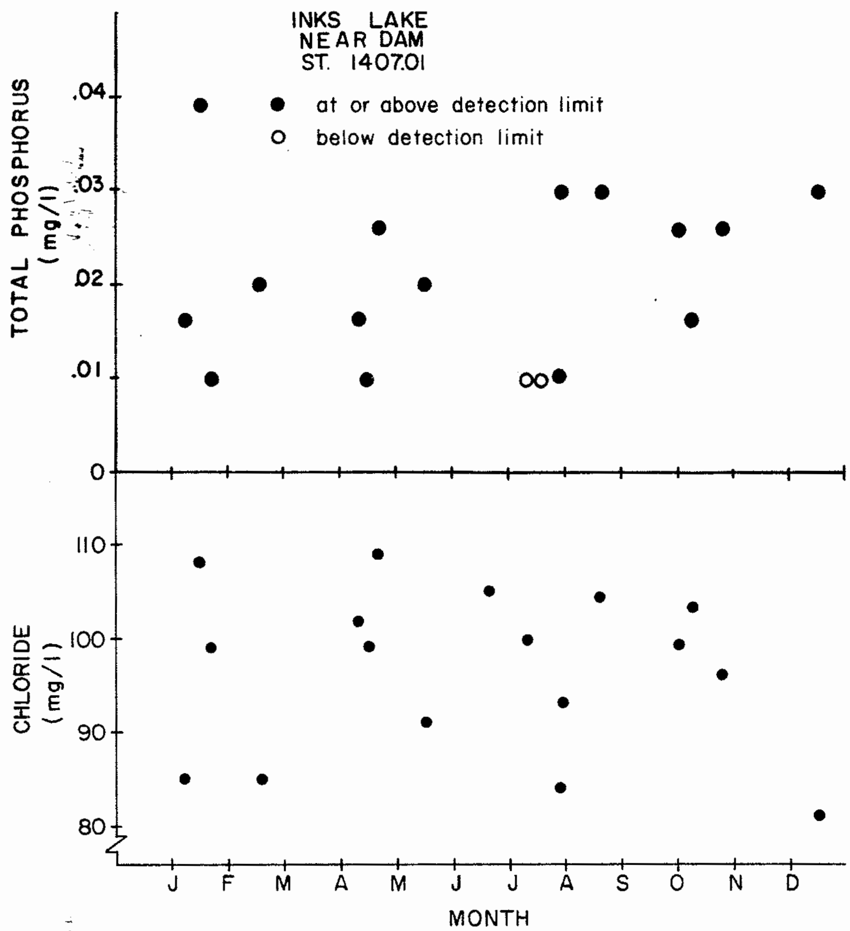


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(h) INKS LAKE, ST. 1407.01

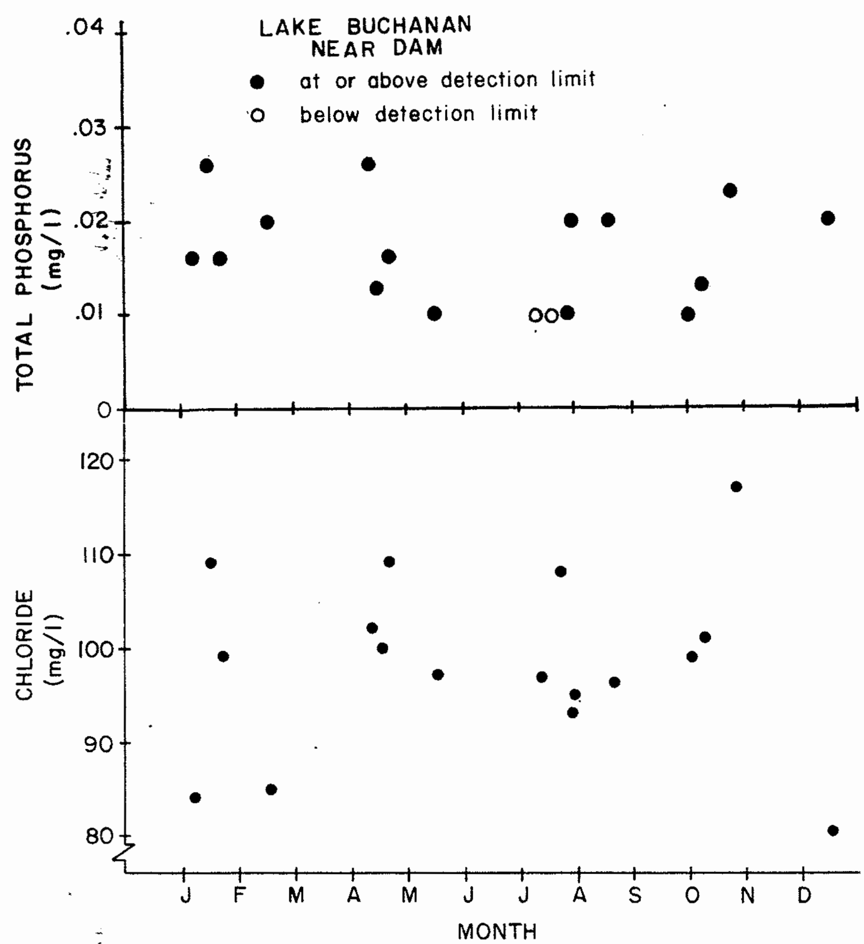


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(i) LAKE BUCHANAN, ST. 1408.01

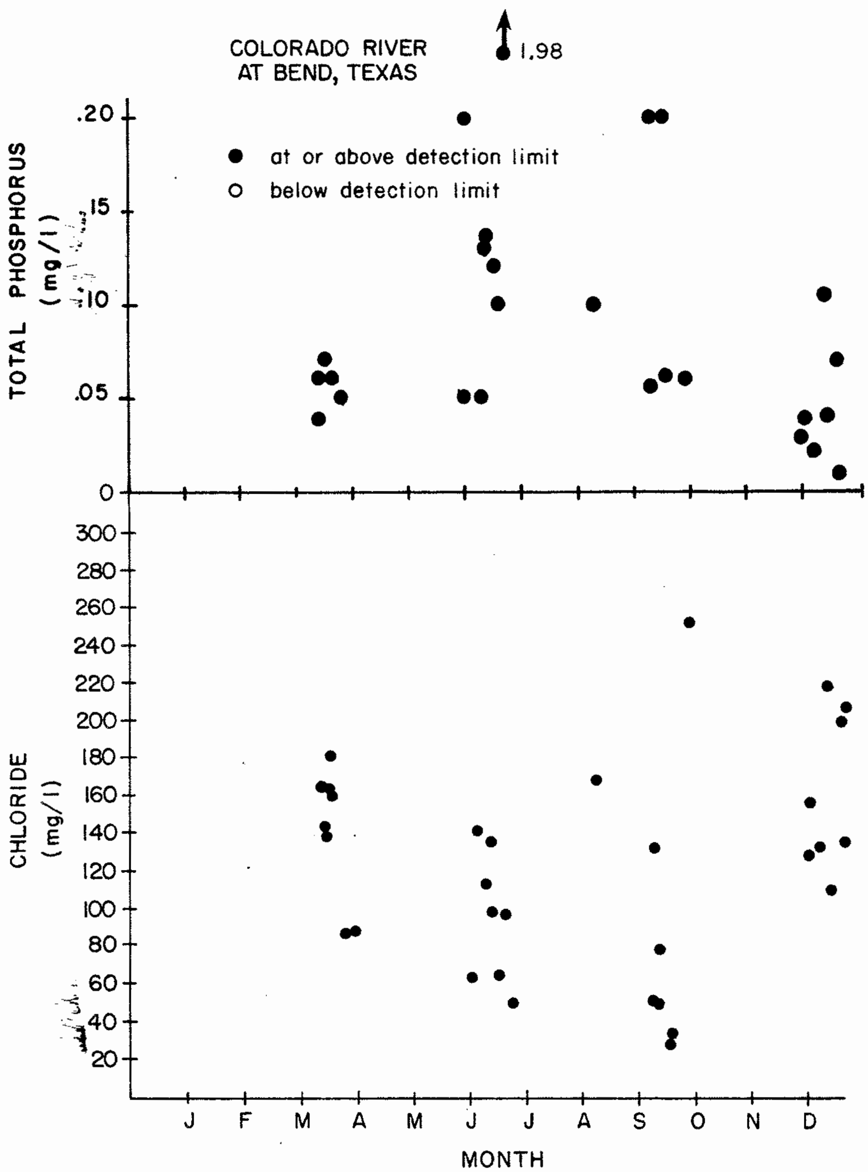


Fig. 3-23 TOTAL PHOSPHORUS AND CHLORIDE BY DATE OF COLLECTION:
(j) COLORADO RIVER, ST. 1409.01

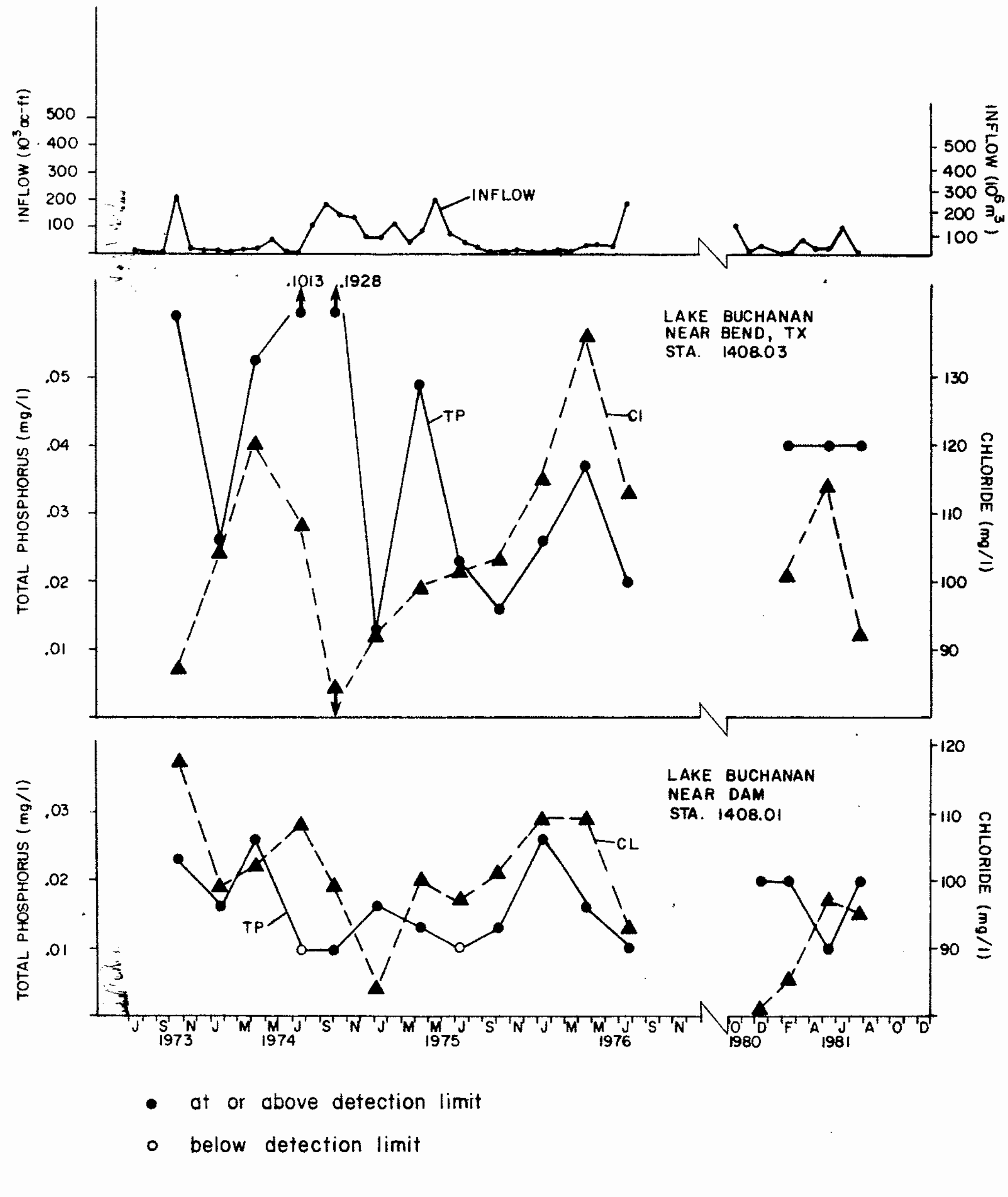


Fig. 3-24 TEMPORAL TOTAL PHOSPHORUS AND CHLORIDE TRENDS, LAKE BUCHANAN

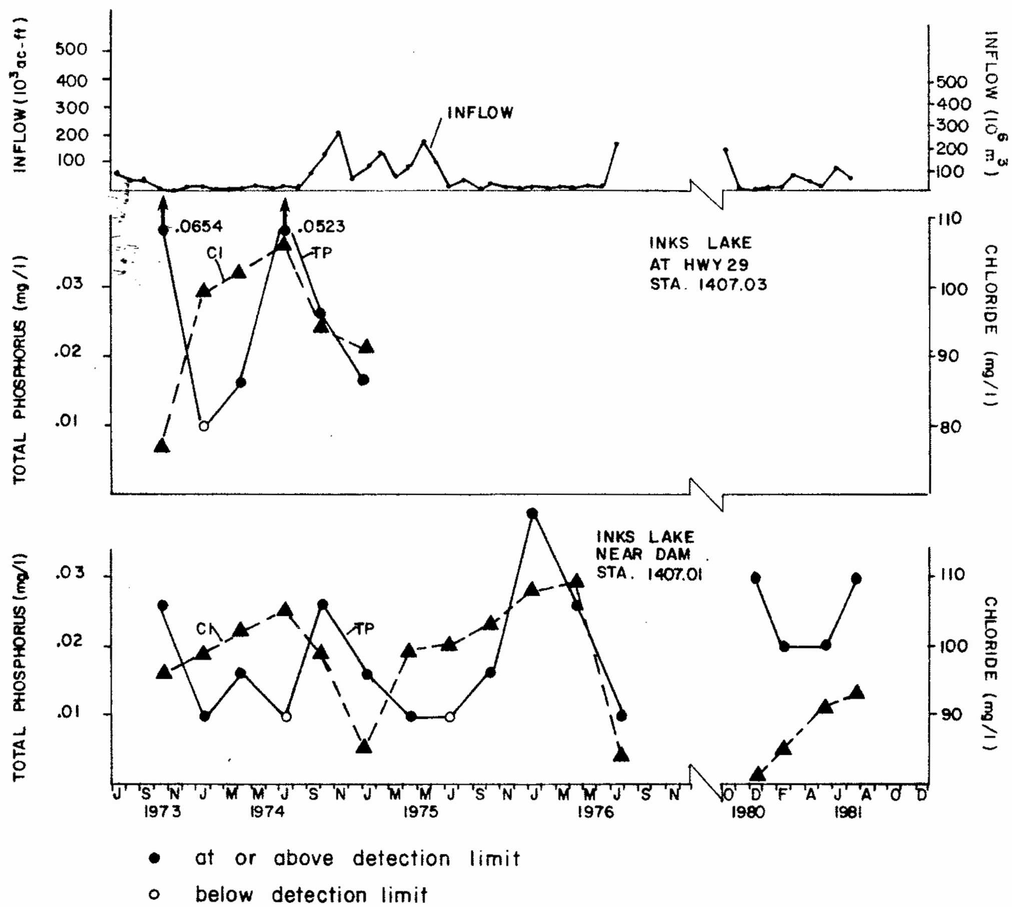


Fig. 3-25 TEMPORAL TOTAL PHOSPHORUS AND CHLORIDE TRENDS, INK LAKE

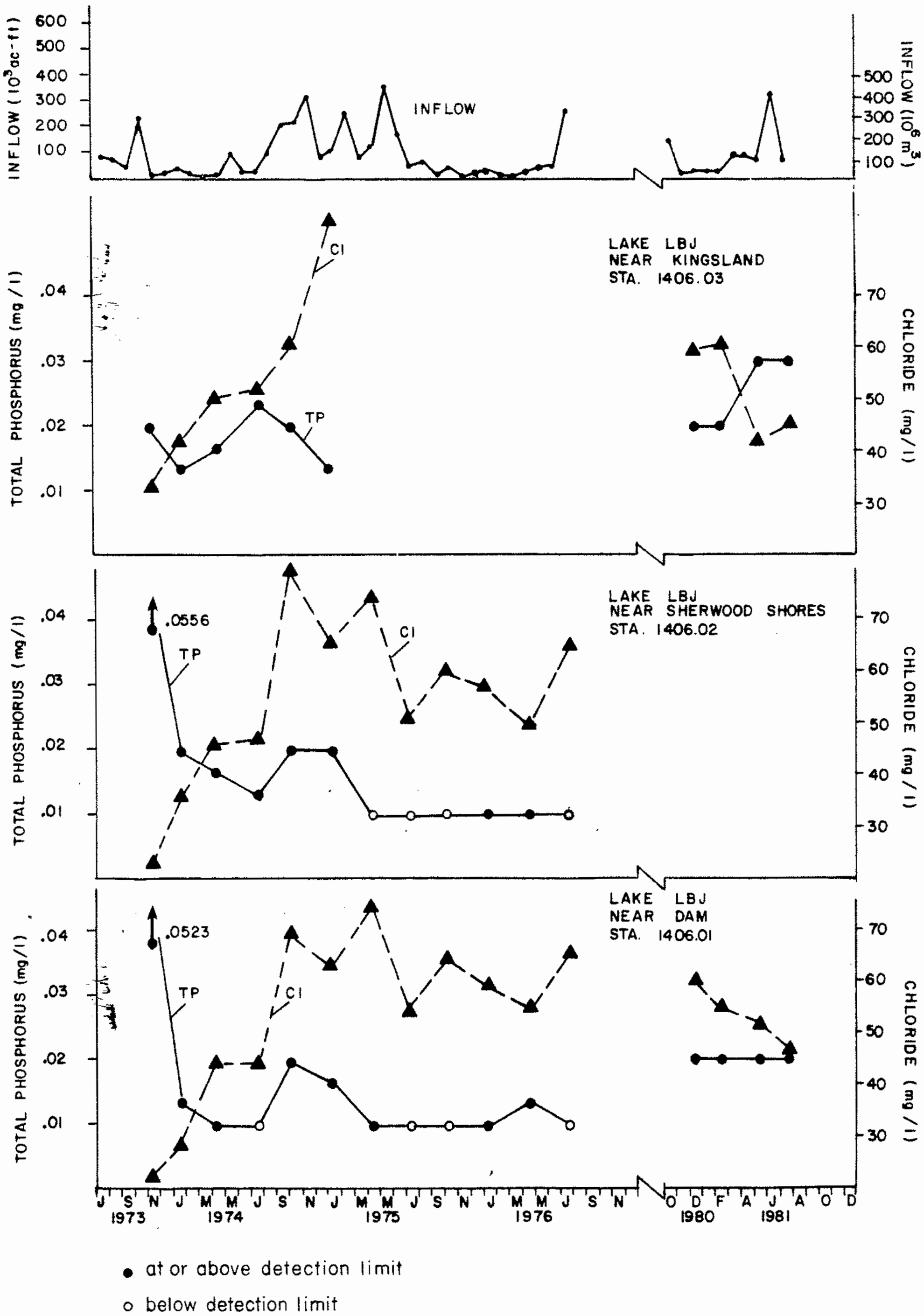


Fig. 3-26 TEMPORAL TOTAL PHOSPHORUS AND CHLORIDE TRENDS, LAKE LBJ

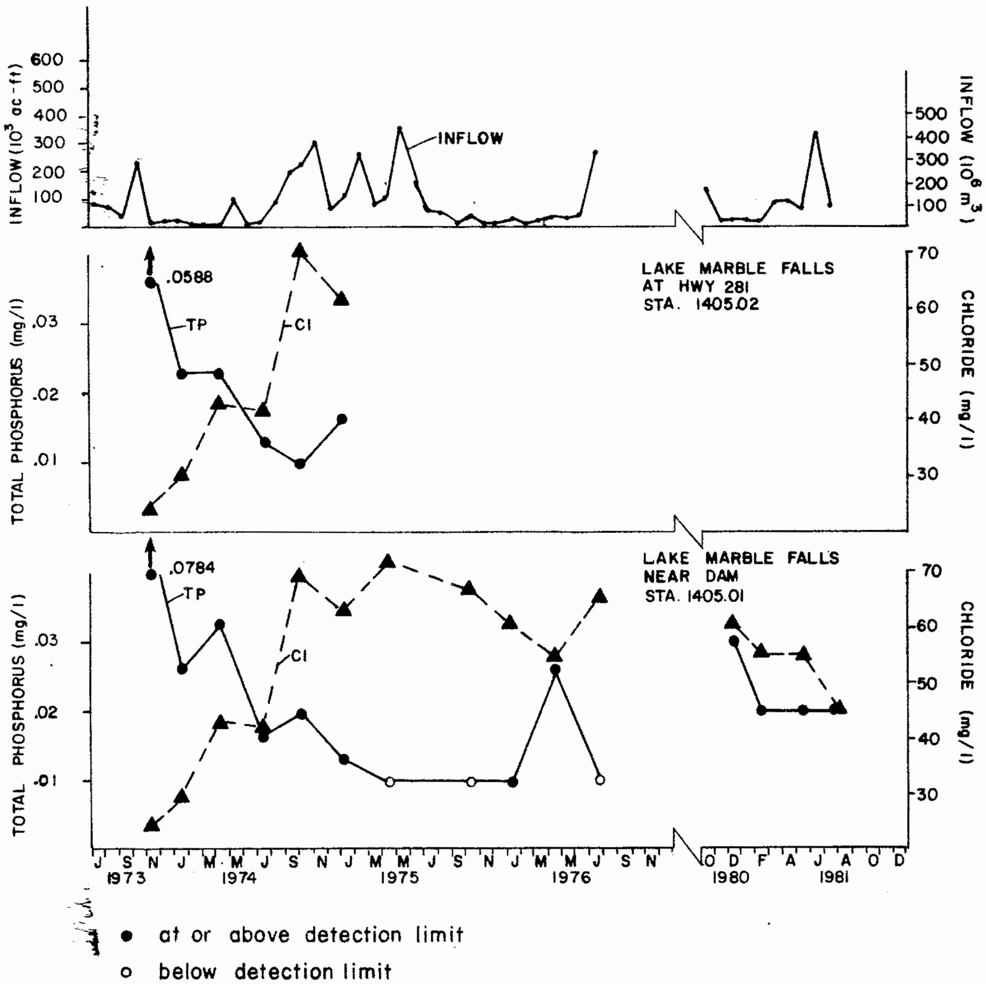


Fig. 3-27 TEMPORAL TOTAL PHOSPHORUS AND CHLORIDE TRENDS, LAKE MARBLE FALLS

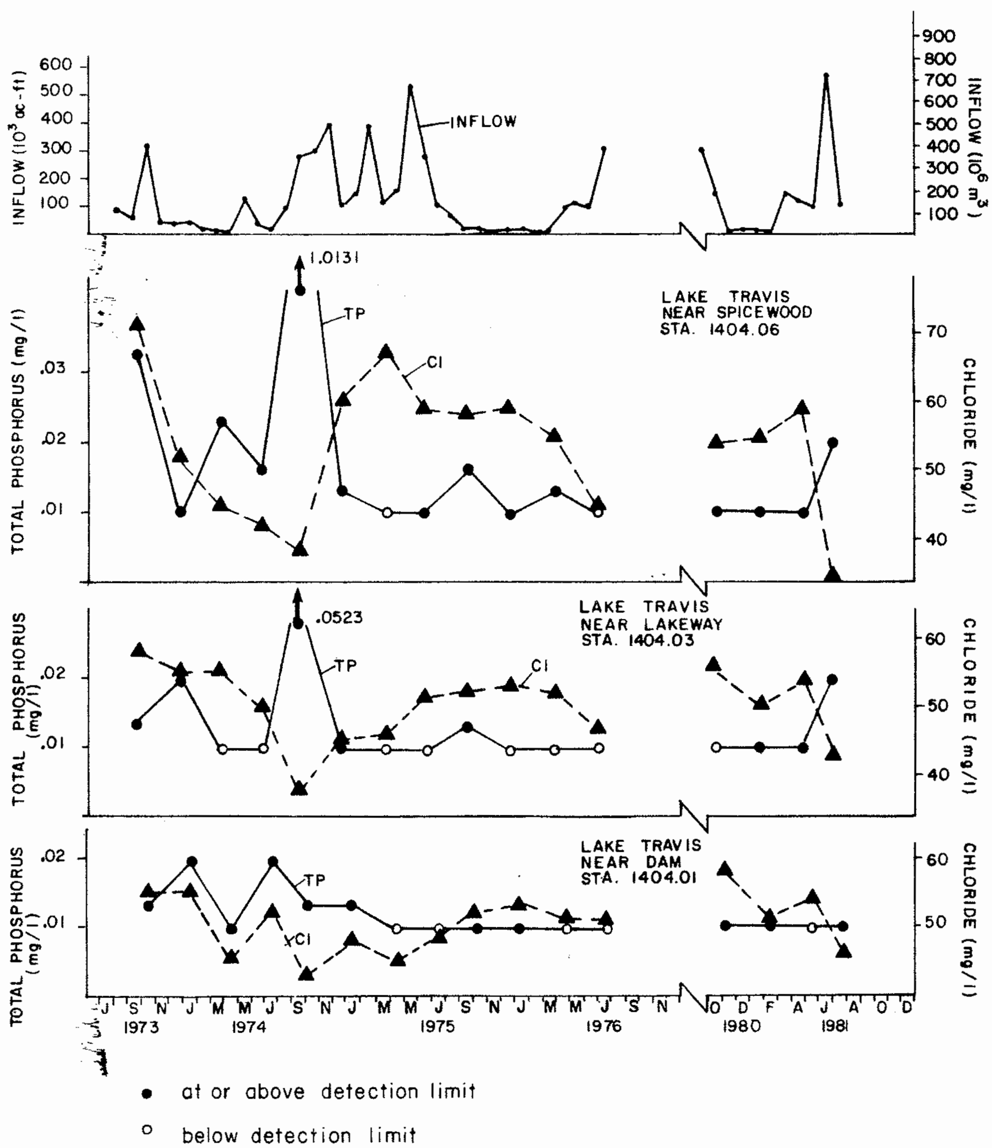


Fig. 3-28 TEMPORAL TOTAL PHOSPHORUS AND CHLORIDE TRENDS, LAKE TRAVIS

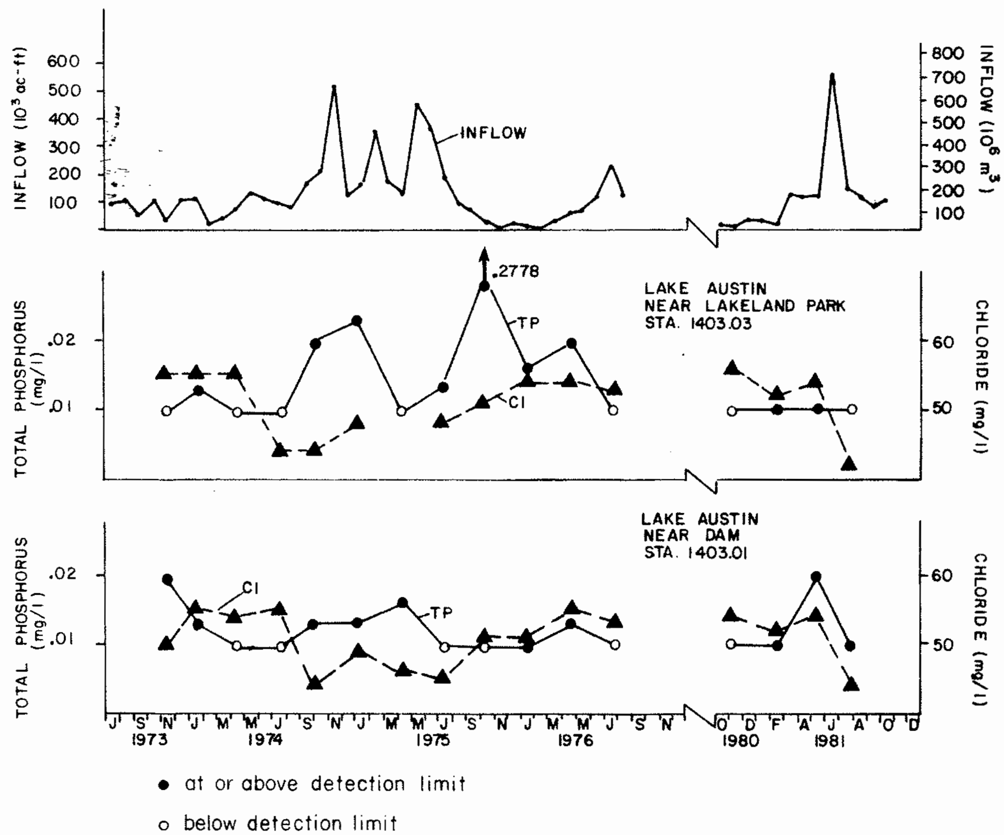


Fig. 3-29 TEMPORAL TOTAL PHOSPHORUS AND CHLORIDE TRENDS, LAKE AUSTIN

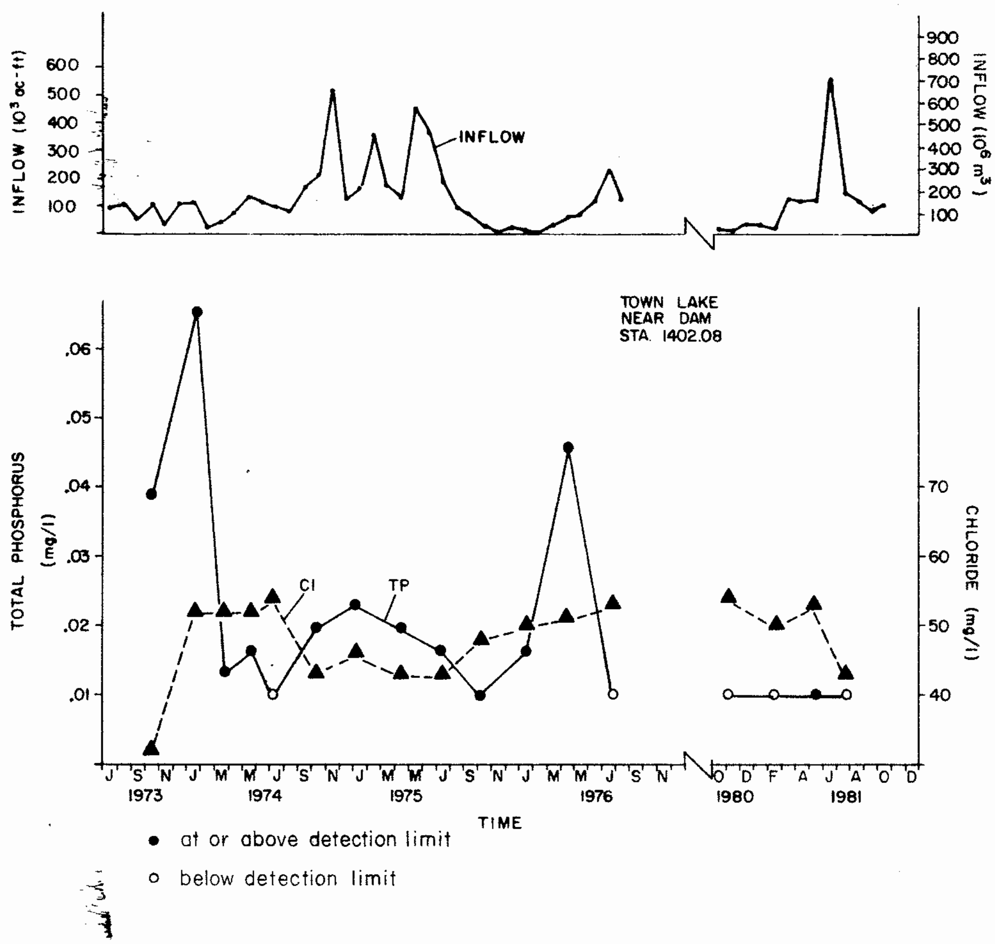


Fig. 3-30 TEMPORAL TOTAL PHOSPHORUS AND CHLORIDE TRENDS, TOWN LAKE

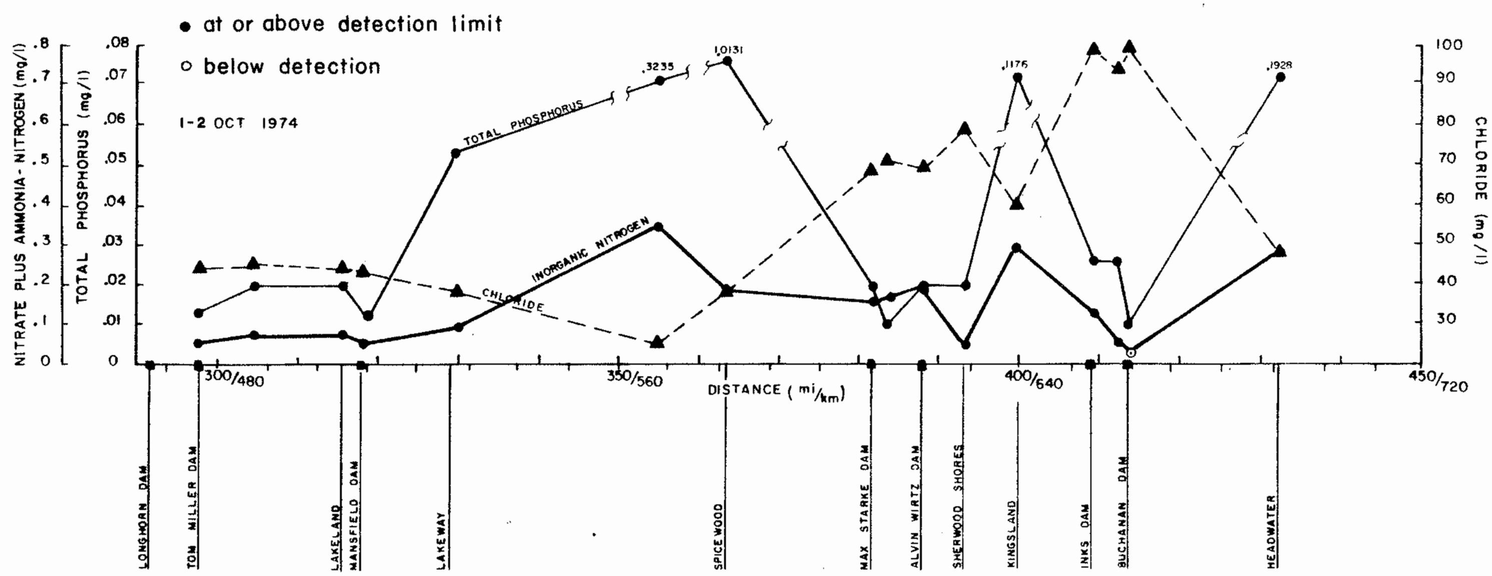


Fig. 3-31 LONGITUDINAL TRENDS OF INORGANIC NITROGEN, 1-2 OCTOBER 1974

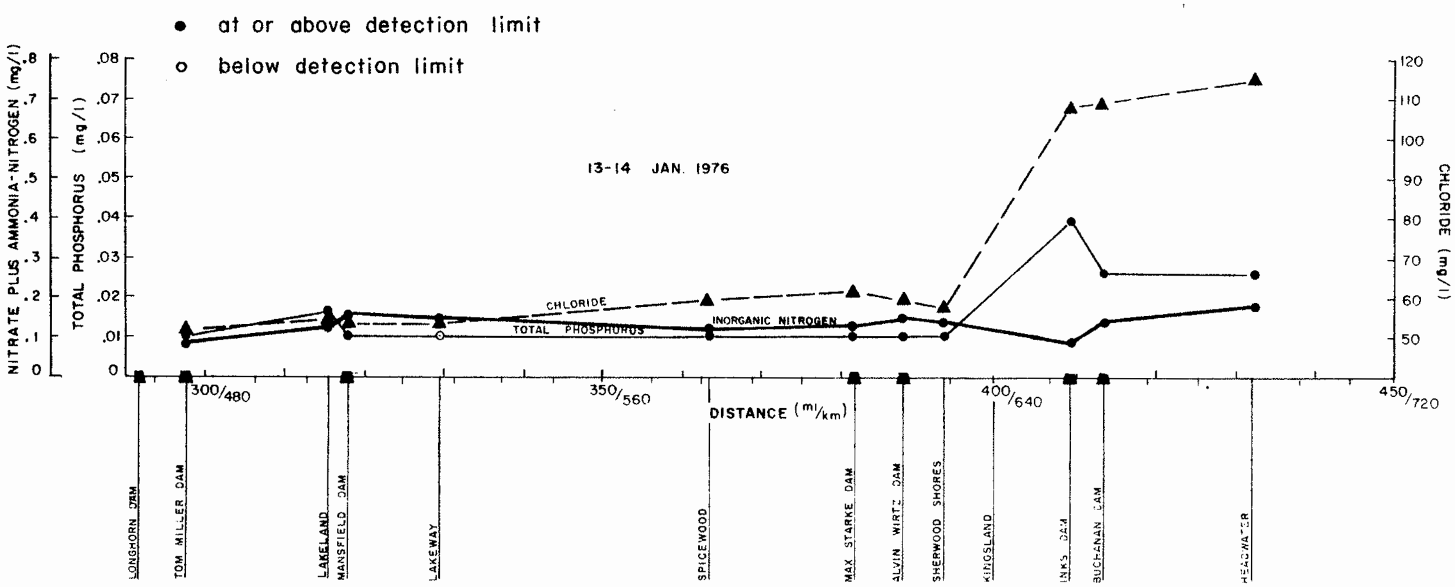


Fig. 3-32 LONGITUDINAL TRENDS OF INORGANIC NITROGEN, 13-14 JANUARY 1976

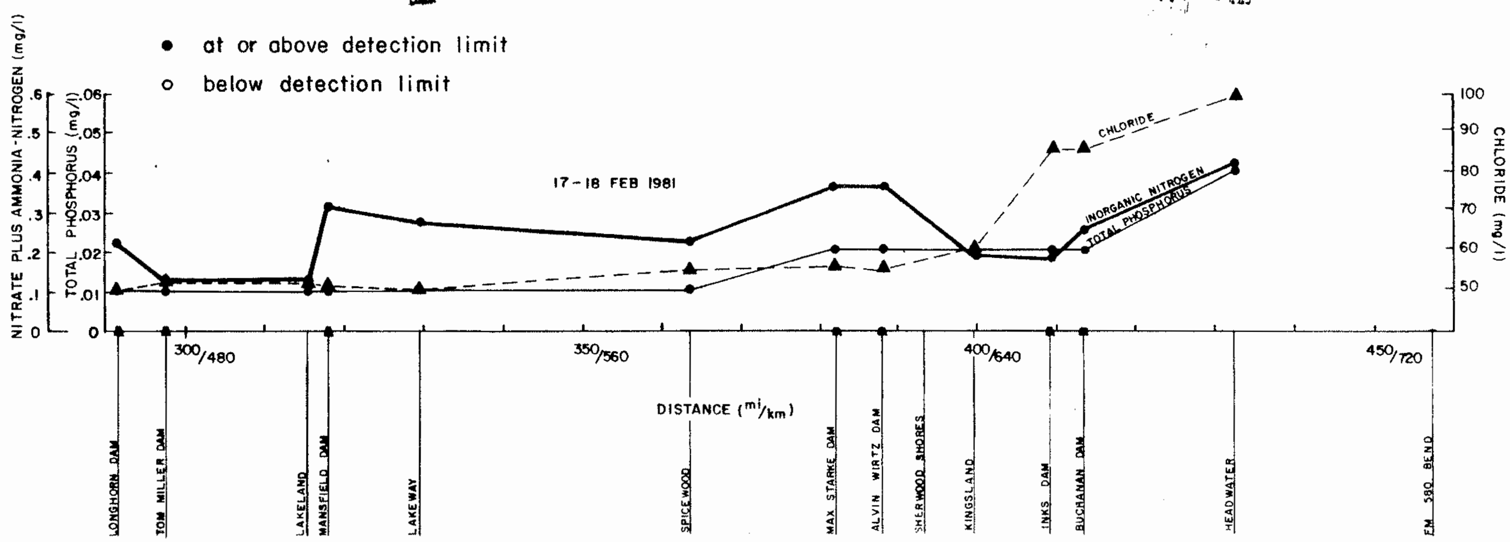


Fig. 3-33 LONGITUDINAL TRENDS OF INORGANIC NITROGEN, 17-18 FEBRUARY 1981

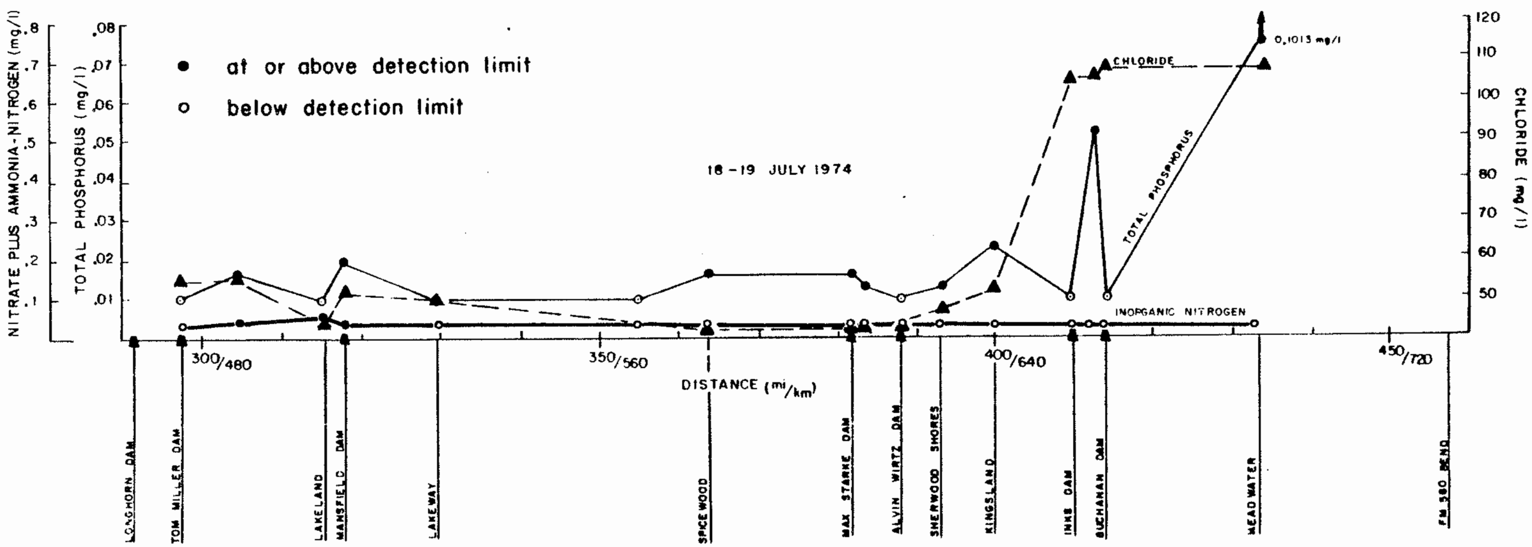


Fig. 3-34 LONGITUDINAL TRENDS OF INORGANIC NITROGEN, 18-19 JULY 1974

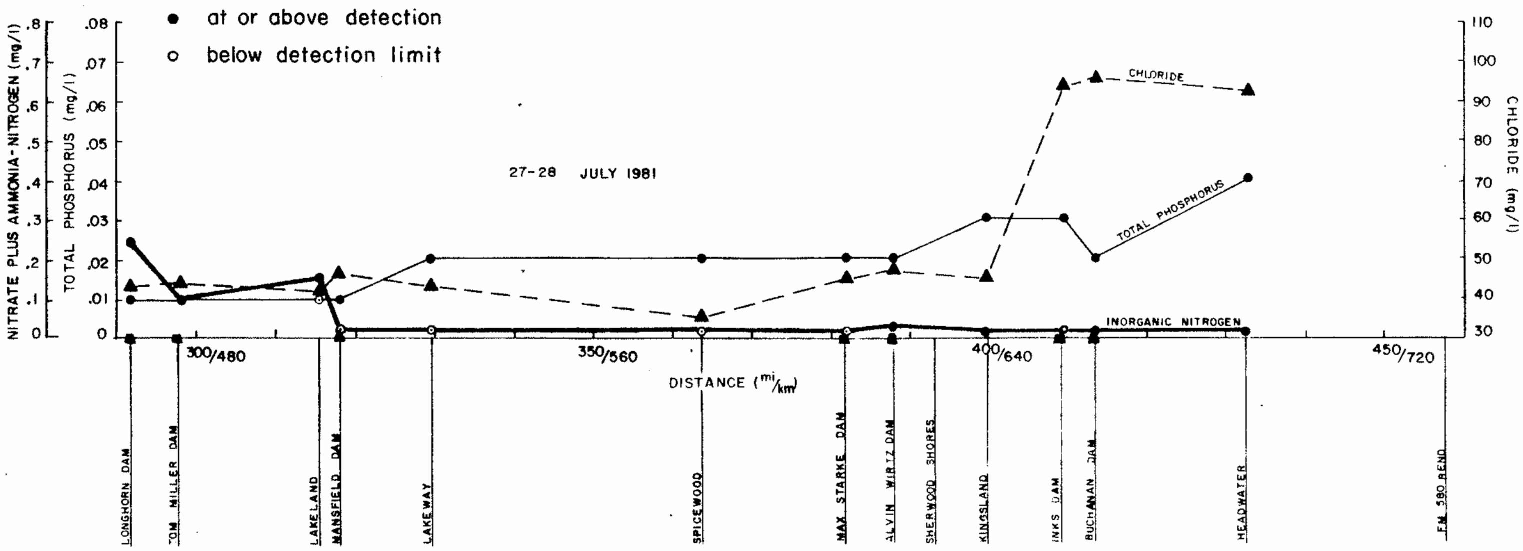


Fig. 3-35 LONGITUDINAL TRENDS OF INORGANIC NITROGEN, 27-28 JULY 1981

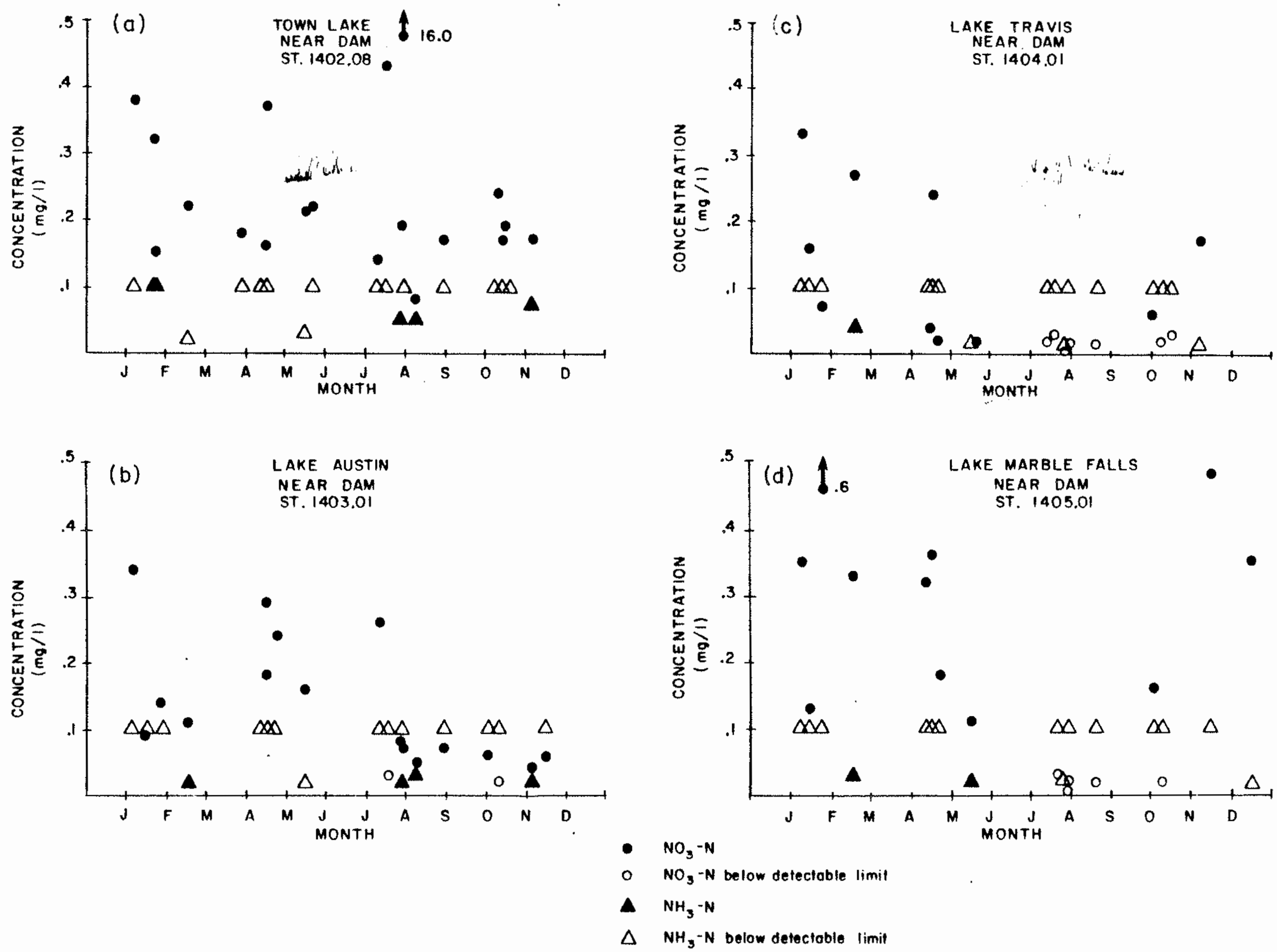


Fig. 3-36 DISPLAY OF INORGANIC NITROGEN BY DATE OF COLLECTION: (a) TOWN LAKE, ST. 1402.08, (b) LAKE AUSTIN, ST. 1403.01, (c) LAKE TRAVIS, ST. 1404.01, (d) LAKE MARBLE FALLS, ST. 1405.01

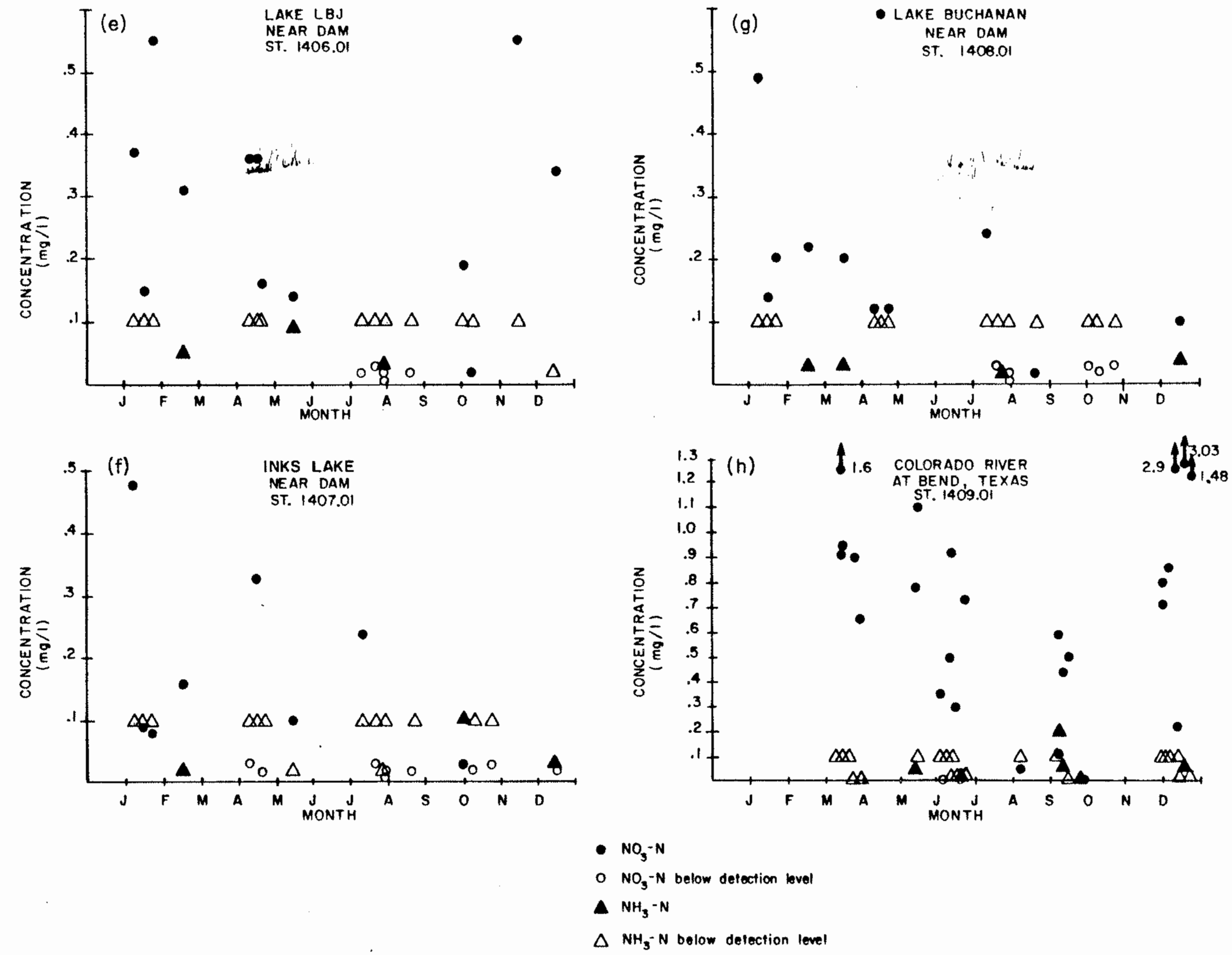


Fig. 3-36 (cont'd): (e) LAKE LBJ, ST. 1406.01, (f) INKS LAKE, ST. 1407.01, (g) LAKE BUCHANAN, ST. 1408.01, (h) COLORADO RIVER, ST. 1409.01

TABLE 3-1

AVAILABILITY OF ROUTINE HISTORICAL PHOSPHORUS AND CHLORIDES DATA FOR THE HIGHLAND LAKES AND MAJOR TRIBUTARIES

Agency	Station No.	Station Location	Approximate Period of Record	Current Sampling Frequency	Comments
TDWR	1402.075	Colorado River in Austin, Hwy 183	69-71	inactive	no TP data
	1402.08	Town Lake at Longhorn Dam	73-81	quarterly/3 yrs	sporadic frequency
	1042.09	Town Lake at Loop 1	73-78	inactive	sporadic frequency
	1403.01	Lake Austin near Tom Miller Dam	73-81	quarterly/3 yrs	frequency varies
	1403.02	Lake Austin near Metropolitan Park	73-75	inactive	few data points
	1403.03	Lake Austin near Lakeland Park	73-81	quarterly/3 yrs	frequency varies
	1404.01	Lake Travis near Mansfield Dam	73-81	quarterly/3 yrs	frequency varies
	1404.02	Lake Travis at Big Sandy Creek Park	73-77	inactive	quarterly
	1404.03	Lake Travis near Lakeway	73-81	quarterly/3 yrs	frequency varies
	1404.04	Lake Travis in Pedernales River arm	73-81	quarterly/3 yrs	frequency varies
	1404.05	Lake Travis above confluence with Pedernales River	73-75	inactive	few data points
	1404.06	Lake Travis near Spicewood	73-81	quarterly/3 yrs	frequency varies
	1045.01	Lake Marble Falls near Starcke Dam	73-81	quarterly/3 yrs	frequency varies
	1405.02	Lake Marble Falls near Hwy 281	73-75	inactive	few data points
	1406.01	Lake LBJ near Wirtz Dam	73-81	quarterly/3 yrs	frequency varies
	1406.02	Lake LBJ near Sherwood Shores	73-77	inactive	quarterly
	1406.03	Lake LBJ near Kingsland	73-81	quarterly/3 yrs	frequency varies
	1407.01	Inks Lake near dam	73-81	quarterly/3 yrs	frequency varies
	1407.02	Inks Lkae in Clear Creek arm	73-77	inactive	quarterly
	1407.03	Inks Lake at Hwy 29	73-75	inactive	few data points
	1408.01	Lake Buchanan near dam	73-81	quarterly/3 yrs	frequency varies
	1408.02	Lake Buchanan in Morgan Creek arm	73-75	inactive	few data points

TABLE 3-1 (Concluded)

Agency	Station No.	Station Location	Approximate Period of Record	Current Sampling Frequency	Comments
TDWR	1408.03	Lake Buchanan near headwater	73-81	quarterly/3 yrs	frequency varies
	1409.01	Colorado River at Bend, FM 580	74-82	quarterly	--
	1414.01	Pedernales River at Johnson City, Hwy 281	68-81	annually	frequency varies
	1415.01	Llano River above Kingsland	73-82	quarterly	
USGS	08158000	Colorado River in Austin, Hwy 183 (TDWR 1402.075)	68-79	monthly	TP available since 1974
	08157900	Town Lake at Longhorn Dam (TDWR 1402.08)	75-77	inactive	recent special surveys
	08154900	Lake Austin near Davis plant intake (TDWR 1403.012)	68-78	inactive	Cl data only
	08147000	Colorado River at Hwy 190 East of San Saba (TDWR 1409.015)	68-79	monthly	some TDWR data also
	08153500	Pedernales River at Johnson City, Hwy 281 (TDWR 1414.01)	69-81	monthly	Cl data only
	08151500	Llano River at Llano, Hwy 16 (TDWR 1415.02)	79-81	monthly	TDWR data available
Austin-Travis Co. Health Dept.	1402.08	Town Lake at Longhorn Dam	77-82	weekly	diss. orthophosphate
	1402.082	Town Lake at Blunn Creek	77-82	weekly	diss. orthophosphate
	1402.0825	Town Lake at Waller Creek	77-82	weekly	diss. orthophosphate
	1402.085	Town Lake at Shoal Creek	77-82	weekly	diss. orthophosphate
	1402.0875	Town Lake at Barton Creek	77-82	weekly	diss. orthophosphate
	1402.09	Town Lake at Loop 1	77-82	weekly	diss. orthophosphate
City of Austin Water Dept.	--	Various stations in Lake Travis, Lake Austin, and Town Lake	75-82	weekly	TP only, also daily intake data
City of Austin Wastewater Dept.	--	Various stations in Lake Travis, Lake Austin, and Town Lake	75-82	weekly	TP only

Sources: Texas Natural Resources Information System (1983a)
USGS (1983a)
Remaley (1983)

TABLE 3-2

PHYSICAL CHARACTERISTICS OF THE LAKES

	Lake Buchanan	Lake Inks	Lake LBJ	Lake Marble Falls	Lake Travis	Lake Austin	Town Lake
Date deliberate impoundment of water began	1937	1938	1951	1951	1940	1939	1960
Spillway elevation (ft. above msl)	1,020.5	888.5	825.0	738.0	714.1	492.8	428.2
Area at spillway elev. ($10^6 m^2$)	93.3	3.2	25.8	3.2	117.4	7.4	1.7
Capacity at spillway elev. ($10^6 m^3$)	1223.6	21.6	170.8	10.8	2410.2	25.9	4.3
Mean depth (m)	13.1	6.7	6.6	3.4	20.5	2.1	2.6
Approximate maximum depth (m)	32.0	18.3	25.9	16.8	61.0	15.2	6.1
Length by river channel (km)	49.3	6.8	34.0	9.2	102.6	32.6	9.6
Maximum width (km)	8.0	0.9	3.3	0.3	3.5	0.4	0.3
Total length of dam (m)	3349.0	471.7	1673.8	262.0	2163.5	484.6	—
Purposes: S - sediment control; C - conservation; FC - flood control	S,C	S,C	C	S,C	S,C,FC	S,C	C
Electrical generation capacity (KW)	38,000	12,000	50,000	30,000	85,000	15,000	—

Sources: Fruh and Davis (1969a); TDWR (1971); TDWR (1978).

TABLE 3-3

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 27 JANUARY - 1 FEBRUARY 1969

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam (LCRA release)	Colorado River at Austin 08158000
30 Dec 68	155	113	9.4	48	—	3,640
31	156	110	7.9	46	—	2,090
01 Jan 69	153	112	8.9	45	—	68
02	153	123	11	42	—	99
03	154	125	11	40	—	84
04	150	122	9.4	38	—	84
05	150	119	9.4	37	—	66
06	150	119	9.4	36	—	64
07	150	120	9.4	35	—	70
08	146	119	9.4	35	—	79
09	141	114	7.9	29	—	74
10	135	110	7.4	28	—	62
11	140	109	7.9	29	—	66
12	141	109	7.9	30	—	70
13	141	111	8.4	29	—	77
14	141	113	9.4	30	—	77
15	146	117	10	32	—	66
16	150	123	11	33	—	50
17	150	124	11	34	—	60
18	155	123	10	31	—	62
19	160	121	10	30	—	62

TABLE 3-3 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam (LCRA release)	Colorado River at Austin 08158000
20	156	120	9.4	31	—	64
21	151	156	8.9	32	—	68
22	144	159	7.9	29	—	72
23	141	137	7.9	29	—	77
24	140	122	7.4	26	—	77
25	137	114	7.4	25	—	70
26	137	112	7.9	28	—	68
27	139	114	8.4	28	—	256
28	133	117	9.4	32	—	72
29	132	120	10	35	—	70
30	133	119	11	33	—	89
31	130	121	13	36	—	62
01 Feb 69	133	125	13	44	—	64

Note: Data not available for Colorado River below Mansfield Dam.
Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-4

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 22-24 JANUARY 1974

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam (LCRA release)	Colorado River at Austin 08158000
26 Dec 73	205	302	8.9	71	—	2,480
27	199	302	8.3	68	—	2,720
28	202	302	8.8	68	—	2,860
29	196	302	9.1	64	—	2,040
30	197	298	8.4	66	—	2,530
31	206	298	7.5	64	—	2,660
01 Jan 74	205	294	7.8	62	—	2,760
02	213	294	8.0	62	—	2,400
03	220	287	9.8	56	—	3,880
04	224	287	9.8	64	—	3,640
05	222	283	10	70	—	3,030
06	221	280	11	71	—	2,270
07	221	276	11	69	—	2,370
08	223	270	11	71	—	2,430
09	222	270	11	68	—	2,690
10	222	266	11	65	—	2,700
11	216	263	11	64	—	2,320
12	215	259	9.3	63	—	2,670
13	221	253	8.8	60	—	3,560
14	221	249	8.8	62	—	1,300
15	220	246	8.8	60	—	2,480

TABLE 3-4 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam (LCRA release)	Colorado River at Austin 08158000
16	212	242	8.5	60	—	2,180
17	211	239	7.3	60	—	2,520
18	209	232	6.9	84	—	2,770
19	253	229	7.2	93	—	2,560
20	325	229	6.8	65	—	2,290
21	230	225	6.5	61	—	2,620
22	209	219	6.1	58	—	2,640
23	197	213	12	62	—	2,780
24	198	207	16	90	—	2,810

Note: Data not available for Colorado River below Mansfield Dam.
Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-5

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 01-02 OCTOBER 1974

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam (LCRA release)	Colorado River at Austin 08158000
26 Aug 74	121	147	1.0	6.0	—	2,060
27	123	168	1.0	5.0	—	1,990
28	530	2,740	4,780	11,100	—	2,510
29	2,680	42,700	2,540	4,430	—	1,180
30	13,500	31,800	2,730	7,430	—	514
31	2,580	19,900	793	1,740	—	2,930
01 Sep 74	2,180	3,990	385	513	—	3,200
02	1,060	2,190	211	1,680	—	3,510
03	777	1,560	203	2,260	—	3,480
04	590	1,570	148	424	—	3,420
05	467	1,410	95	239	—	3,340
06	385	1,020	71	180	—	3,610
07	332	864	69	149	—	3,750
08	291	780	48	128	—	3,660
09	277	721	33	117	—	3,540
10	263	680	43	187	—	3,510
11	245	639	30	121	—	3,080
12	231	601	25	112	—	3,670
13	231	619	28	274	—	4,160
14	527	600	28	165	—	2,360
15	333	550	23	121	—	138

TABLE 3-5 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam (LCRA release)	Colorado River at Austin 08158000
16	280	522	26	112	—	2,140
17	13,300	908	25	139	—	2,260
18	29,300	8,500	27	112	—	3,580
19	28,400	3,070	36	107	—	4,960
20	8,090	1,530	28	98	—	8,000
21	13,700	1,840	168	1,490	—	12,300
22	14,400	5,250	96	1,350	—	9,600
23	12,500	4,050	53	328	—	5,320
24	5,980	2,040	39	215	—	5,990
25	3,370	1,350	35	177	—	6,380
26	2,730	1,110	34	160	—	5,680
27	2,590	967	35	149	—	6,020
28	2,860	878	28	132	—	6,040
29	2,450	804	27	122	—	5,860
30	2,070	754	25	112	—	5,610
01 Oct 74	1,800	702	23	107	5,410	5,990
02	1,580	658	19	99	5,420	5,890

Note: Data not available for Colorado River below Mansfield Dam until October 1974.
Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-6

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 06-08 JANUARY 1975

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
10 Dec 74	1,930	556	45	212	3,090	3,570
11	1,690	611	70	303	3,070	3,440
12	1,120	608	63	265	3,060	3,430
13	947	573	47	213	3,080	3,450
14	867	546	42	196	3,060	3,490
15	821	534	41	186	3,060	3,500
16	792	525	42	178	3,820	3,480
17	826	521	41	172	3,090	3,340
18	1,800	504	40	165	3,140	3,400
19	1,130	501	40	163	3,000	3,420
20	868	490	39	162	3,060	3,060
21	760	484	37	154	960	1,720
22	711	468	36	155	0	241
23	687	477	36	157	0	231
24	667	467	36	156	0	236
25	1,000	471	42	187	0	249
26	1,090	553	73	436	0	255
27	916	606	78	409	0	230
28	946	576	67	311	0	229
29	998	552	66	277	0	224
30	1,010	541	64	266	0	268

TABLE 3-6 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
31	1,080	582	70	285	1,680	2,010
01 Jan 75	1,760	581	77	297	1,850	2,230
02	1,330	549	75	290	1,930	2,270
02	1,120	551	82	304	2,200	2,310
04	1,070	516	75	270	3,030	3,450
05	1,030	489	68	245	3,030	3,110
06	975	463	67	236	4,280	3,970
07	1,390	455	66	233	5,000	5,320
08	1,670	438	64	222	3,440	4,250

Note: Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-7

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 08-09 OCTOBER 1975

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
09 Sep 75	198	249	16	80	1,540	1,880
10	196	233	13	81	1,530	1,870
11	189	244	11	85	1,610	1,840
12	188	262	8.6	94	1,630	1,910
13	188	273	7.1	87	1,340	1,580
14	184	290	8.8	80	1,550	1,550
15	186	295	15	87	1,340	1,760
16	408	329	352	116	1,420	1,610
17	226	313	147	295	954	965
18	213	281	39	144	1,140	1,130
19	198	257	22	97	853	1,040
20	195	308	17	93	709	1,400
21	231	288	16	116	720	953
22	238	313	15	98	0	198
23	253	326	13	84	327	447
24	260	313	11	81	736	902
25	244	296	9.6	75	695	841
26	288	270	8.8	72	756	897
27	218	259	8.6	64	832	916
28	213	256	8.3	64	790	906
29	211	8.2	64	812	898	

TABLE 3-7 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
30	215	241	7.4	64	776	896
01 Oct 75	212	227	7.2	58	744	895
02	200	217	6.4	57	776	898
03	190	210	6.1	57	801	904
04	184	210	6.1	57	756	897
05	182	212	6.1	56	856	908
06	179	218	6.1	57	790	968
07	179	218	6.1	56	827	915
08	175	216	6.1	53	810	913
09	172	220	6.0	54	1,230	1,160

Note: Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-8

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 13 JANUARY 1976

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
15 Dec 75	229	228	11	59	292	295
16	220	224	15	59	216	311
17	217	212	13	59	288	295
18	207	203	11	59	1,460	1,400
19	207	201	12	61	386	361
20	207	205	12	57	0	137
21	207	205	11	57	0	143
22	208	206	11	57	0	134
23	214	216	11	57	0	119
24	221	303	34	84	53	361
25	246	308	58	149	0	165
26	243	302	43	128	0	143
27	243	296	29	97	0	143
28	254	279	21	87	0	150
29	251	262	17	78	0	137
30	244	251	15	69	0	134
31	246	251	15	67	0	140
01 Jan 76	256	246	13	65	0	146
02	256	241	13	63	25	137
03	243	236	11	62	0	128
04	233	230	11	60	0	113

TABLE 3-8 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
05	229	230	11	59	0	113
06	231	236	11	58	0	113
07	230	230	11	58	2,080	1,320
08	222	220	11	57	2,800	3,450
09	219	220	11	57	1,280	1,200
10	227	225	11	60	0	131
11	227	220	11	57	0	138
12	227	220	11	59	0	111
13	232	220	11	60	0	111

Note: Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-9

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 18 FEBRUARY 1981

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Bull Creek near Austin 08154700	Barton Creek at Austin 08155300	Colorado River at Austin 08158000
20 Jan 80	266	226	33	64	684	6.9	17	554
21	265	232	26	60	387	4.9	25	498
22	262	216	21	55	570	4.5	24	556
23	253	194	16	51	1,020	4.4	22	1,010
24	253	180	14	52	487	4.4	22	676
25	251	171	14	50	27	4.4	21	174
26	247	165	14	45	160	4.4	19	267
27	240	161	13	45	419	4.2	18	495
28	236	158	12	45	488	4.1	17	600
29	232	158	11	46	120	4.1	15	212
30	225	152	11	46	232	3.9	15	324
31	220	147	10	45	430	3.7	14	541
01 Feb 81	220	163	12	45	451	4.9	13	511
02	215	153	10	42	0	4.0	12	123
03	212	159	8.8	42	652	3.7	11	771
04	215	166	14	48	1,110	4.5	9.4	1,180
05	224	189	30	57	663	6.9	9.3	862
06	233	193	28	59	269	5.9	13	391
07	235	203	24	62	0	5.6	17	97
08	231	194	20	57	0	5.1	18	94

TABLE 3-9 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Bull Creek near Austin 08154700	Barton Creek at Austin 08155300	Colorado River at Austin 08158000
09	229	182	18	54	66	5.0	17	149
10	218	237	106	56	822	9.5	24	1,160
11	218	220	130	64	1,280	5.7	27	1,630
12	215	194	65	65	753	5.2	27	447
13	215	169	36	59	0	5.0	24	87
14	215	169	26	60	0	5.0	25	86
15	210	169	24	57	14	5.0	26	80
16	210	169	21	54	44	5.0	27	124
17	208	169	20	54	0	5.0	27	74
18	205	163	19	53	223	5.0	24	267

Note: Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-10

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 18-19 JULY 1974

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 0815300	Colorado River below Mansfield Dam (LCRA Release)	Colorado River at Austin 08158000
19 Jun 74	52	93	3.0	34	—	2210
20 Jun 74	46	89	2.0	32	—	2220
21 Jun 74	48	85	1.6	26	—	2270
22 Jun 74	41	79	1.5	21	—	2110
23 Jun 74	40	72	1.1	21	—	2400
24 Jun 74	41	65	1.1	20	—	2350
25 Jun 74	43	63	0.83	22	—	1600
26 Jun 74	41	71	0.18	21	—	2310
27 Jun 74	41	77	0.01	22	—	2320
28 Jun 74	39	77	0	22	—	2430
29 Jun 74	36	75	0	18	—	2190
30 Jun 74	27	67	0	16	—	2200
1 Jul 74	23	57	0	16	—	2210
2 Jul 74	26	56	0	17	—	2110
3 Jul 74	31	54	0	17	—	1900
4 Jul 74	33	55	0	17	—	1940
5 Jul 74	37	57	0	15	—	2020
6 Jul 74	46	55	0	13	—	1940
7 Jul 74	48	51	0	13	—	1990
8 Jul 74	44	52	0	12	—	1910
9 Jul 74	40	46	0	11	—	1910

TABLE 3-10 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 0815300	Colorado River below Mansfield Dam (LCRA Release)	Colorado River at Austin 08158000
10 Jul 74	37	38	0	10	—	1880
11 Jul 74	32	35	0	9.5	—	1920
12 Jul 74	30	40	0	12	—	1640
13 Jul 74	27	42	0.03	16	—	1610
14 Jul 74	23	41	0.32	14	—	1550
15 Jul 74	22	40	0.46	11	—	1540
16 Jul 74	23	43	0.41	10	—	1580
17 Jul 74	23	42	0.26	8.8	—	1560
18 Jul 74	21	48	0.23	8.4	—	1560
19 Jul 74	17	55	2.0	9.1	—	1660

Note: Data not available for Colorado River below Mansfield Dam.
Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-11

STREAMFLOW DATA
ANTECEDENT TO SURVEY OF 10-11 JULY 1975
(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
11 Jun 75	1370	1170	361	1400	5440	6490
12 Jun 75	1030	2070	222	917	5440	6070
13 Jun 75	922	2230	171	719	5440	5940
14 Jun 75	847	1250	142	621	5440	5890
15 Jun 75	731	895	119	558	5440	6060
16 Jun 75	613	754	108	524	5440	6080
17 Jun 75	528	676	103	487	5320	5780
18 Jun 75	470	599	95	445	5440	5330
19 Jun 75	412	554	84	407	5440	5680
20 Jun 75	378	528	78	379	5480	5580
21 Jun 75	375	514	78	390	5530	5270
22 Jun 75	345	493	78	458	5530	5650
23 Jun 75	325	481	77	411	5530	5640
24 Jun 75	368	466	68	346	5530	5610
25 Jun 75	535	441	80	427	5530	6170
26 Jun 75	454	435	135	476	5530	6030
27 Jun 75	670	608	305	1130	5530	5840
28 Jun 75	725	658	146	1270	5530	5920
29 Jun 75	497	576	107	863	5520	6060
30 Jun 75	627	948	86	448	5530	5880

TABLE 3-11 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
1 Jul 75	878	720	68	382	5530	6310
2 Jul 75	923	590	63	423	5530	5900
3 Jul 75	915	623	162	792	5530	5370
4 Jul 75	926	591	100	503	5530	5680
5 Jul 75	1170	517	59	376	5530	5670
6 Jul 75	835	466	48	327	5530	5870
7 Jul 75	737	415	41	298	4020	4750
8 Jul 75	606	382	39	274	3260	3690
9 Jul 75	480	356	38	258	3000	3030
10 Jul 75	445	338	34	243	3010	3180
11 Jul 75	409	350	47	242	2820	3560

Note: Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-12

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 27-28 JULY 1976

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
28 Jun 76	173	365	156	303	2040	2490
29 Jun 76	145	313	92	191	2050	2370
30 Jun 76	133	241	62	152	2090	2380
1 Jul 76	124	202	48	131	2430	2310
2 Jul 76	111	177	39	115	2150	2420
3 Jul 76	127	166	32	104	2170	2380
4 Jul 76	2610	9120	1490	2460	1870	2640
5 Jul 76	4500	793	861	1690	3030	3580
6 Jul 76	1360	439	327	666	3030	3530
7 Jul 76	1460	428	248	479	2750	3310
8 Jul 76	988	380	194	295	2900	2380
9 Jul 76	670	319	161	285	3030	3530
10 Jul 76	832	356	211	970	3030	3650
11 Jul 76	1770	3230	1280	3230	3100	3600
12 Jul 76	3000	1210	634	1280	3950	3990
13 Jul 76	1600	709	338	720	5080	5330
14 Jul 76	1930	548	310	631	5060	5660
15 Jul 76	1340	587	284	808	4870	5690
16 Jul 76	1780	1870	244	641	4860	5580
17 Jul 76	8310	2400	194	788	5090	5540
18 Jul 76	4290	2020	165	471	5060	5180

TABLE 3-12 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
19 Jul 76	2420	979	118	369	5040	5180
20 Jul 76	1510	580	101	315	4990	5480
21 Jul 76	1130	429	92	279	4990	5380
22 Jul 76	986	369	83	267	5050	5410
23 Jul 76	866	332	74	251	4990	5380
24 Jul 76	681	310	67	234	5010	5350
25 Jul 76	561	293	81	320	4990	5500
26 Jul 76	494	280	92	333	3840	4650
27 Jul 76	683	263	64	234	3120	3660
28 Jul 76	1220	245	52	203	3200	3690

Note: Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

TABLE 3-13

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 19 AUGUST 1977

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Colorado River at Austin 08158000
21 Jul 77	198	181	1.2	45	1140	1260
22 Jul 77	201	179	.71	44	863	1130
23 Jul 77	187	178	.36	42	813	1030
24 Jul 77	179	176	.11	40	862	1020
25 Jul 77	175	170	0	40	1100	1030
26 Jul 77	175	164	0	38	1080	1290
27 Jul 77	173	161	0	35	1220	1400
28 Jul 77	290	165	0	35	1230	1340
29 Jul 77	349	170	0	35	1540	1620
30 Jul 77	402	176	.12	35	1390	1830
31 Jul 77	272	172	0	35	1590	1680
1 Aug 77	244	165	0	33	1450	1680
2 Aug 77	232	165	0	33	1520	1690
3 Aug 77	222	159	0	32	1430	1710
4 Aug 77	204	155	0	40	2050	2000
5 Aug 77	189	148	0	43	1790	2080
6 Aug 77	176	146	0	34	1780	2060
7 Aug 77	176	144	0	27	1900	2060
8 Aug 77	172	144	0	27	2030	2130
9 Aug 77	164	142	0	27	1920	2090
10 Aug 77	144	135	0	26	1690	2020

TABLE 3-14

STREAMFLOW DATA

ANTECEDENT TO SURVEY OF 27-28 JULY 1981

(CFS)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Bull Creek at Loop 360 08154700	Barton Creek at Loop 360 08155300	Colorado River at Austin 08158000
16 Jun 81	2140	5320	7080	14000	14200	504	2720	19100
17 Jun 81	1140	16200	981	2930	20300	172	1300	22100
18 Jun 81	987	15400	439	1680	22700	97	747	21300
19 Jun 81	803	3110	261	1230	20300	71	513	19600
20 Jun 81	506	1700	177	983	15300	59	366	16700
21 Jun 81	378	1240	117	802	15300	49	269	16700
22 Jun 81	373	1010	10	661	13300	42	216	15300
23 Jun 81	349	864	91	560	8360	35	172	9340
24 Jun 81	303	758	78	524	5530	31	140	6130
25 Jun 81	253	689	93	603	5530	30	130	6260
26 Jun 81	214	638	72	606	5530	28	169	6160
27 Jun 81	198	593	54	469	5530	24	114	6070
28 Jun 81	191	582	47	412	5530	22	138	5990
29 Jun 81	179	566	48	383	5530	20	207	5980
30 Jun 81	165	535	46	356	5530	19	194	5860
1 Jul 81	153	504	42	324	4180	17	174	5090
2 Jul 81	144	483	39	294	3570	15	152	3810
3 Jul 81	133	461	35	274	3540	14	142	3780
4 Jul 81	128	449	32	253	3170	13	122	3800
5 Jul 81	127	433	31	265	3130	4	286	4740
6 Jul 81	131	426	45	702	3340	21	269	3980

TABLE 3-14 (Concluded)

Date	Colorado River near San Saba 08147000	Llano River at Llano 08151500	Sandy Creek near Kingsland 08152000	Pedernales River near Johnson City 08153500	Colorado River below Mansfield Dam 08154510	Bull Creek at Loop 360 08154700	Barton Creek at Loop 360 08155300	Colorado River at Austin 08158000
7 Jul 81	337	456	34	348	3230	19	174	3900
8 Jul 81	313	418	32	265	3230	26	169	3810
9 Jul 81	203	399	32	241	3230	21	145	3780
10 Jul 81	220	392	30	221	3230	17	120	3780
11 Jul 81	268	387	28	208	3230	15	98	3750
12 Jul 81	217	371	26	199	3420	13	73	3590
13 Jul 81	174	362	24	182	3240	11	63	3780
14 Jul 81	155	353	24	169	3240	11	49	3800
15 Jul 81	145	347	22	162	3440	9.4	41	3770
16 Jul 81	126	332	17	151	3090	8.0	38	3730
17 Jul 81	115	320	16	142	1670	7.7	29	2640
18 Jul 81	102	316	17	138	1310	7.2	27	1840
19 Jul 81	87	307	16	132	1970	6.7	23	1730
20 Jul 81	80	301	15	125	1320	6.1	19	1700
21 Jul 81	72	294	15	120	1660	5.7	16	2000
22 Jul 81	55	280	13	115	1780	5.1	12	1970
23 Jul 81	48	269	14	106	1500	5.0	7.1	1980
24 Jul 81	37	260	13	101	1930	4.7	6.0	2270
25 Jul 81	28	254	13	96	1990	4.7	4.9	2340
26 Jul 81	25	244	13	96	1820	3.8	3.8	2380
27 Jul 81	26	245	13	116	1800	4.7	2.7	2350
28 Jul 81	26	239	14	117	1720	4.6	1.6	2440

Note: Data reported by USGS as cfs (35.31 cfs = 1 m^3/s).

CHAPTER 4.0

MASS LOADINGS

Quantification of mass loadings of phosphorus to the Highland Lakes requires identification of both point source and nonpoint source loads. Point source loads are defined as those which emanate from a definable point or origin, the most common example being an effluent discharge from a wastewater treatment plan. Such wastewater discharges require permits from an appropriate regulatory authority. In Texas, discharge permits must be obtained from the Texas Department of Water Resources and from the U.S. Environmental Protection Agency. The discharge permit specifies allowable limitations on effluent loads or constituent levels and flowrates.

Nonpoint source loadings are those which arise from sources other than identifiable outfalls. They are often characterized as originating from diffuse sources. Typically, the most substantial nonpoint source loads are associated with mass transport in stormwater runoff events. In an urban setting, nonpoint source loads include, for example, washoff of street surface contaminants, erosion from denuded areas in various stages of construction, and runoff from chemically-treated residential lawns. Rural nonpoint source loads include runoff from fertilized cropland, cleared pasture land and forested areas. Seepage of leachate from individual home septic tank and drainfield systems constitutes an often controversial type of nonpoint source loading. In many situations, particularly within watersheds that contain few point sources, nonpoint sources may account for the predominant portion of various constituent mass loads.

4.1 POINT SOURCE LOADINGS

There are relatively few significant point source dischargers within the watershed of the Highland Lakes. A listing of all municipal and industrial permits (TDWR, 1983a) is displayed in Table 4-1.

Though the number of permit holders may appear large, the vast majority have very small discharges, and are remotely located from the reservoirs. Municipal wastewater treatment plants within the watershed of the Highland Lakes are typically small, serving specific residential developments. The majority of municipal plants dispose of treated effluent via irrigation systems, often on golf courses. The few industrial permit holders are associated with wood products or minerals processing. Several animal feedlot operations are also permitted in the watershed.

Permit limitations for significant dischargers are presented in Table 4-2. In the majority of permits, phosphorus is not specifically regulated. Thus, Table 4-2 includes data for permitted flow, biochemical oxygen demand (BOD_5), and total suspended solids (TSS) for comparison of effluent characteristics. All permit holders identified in Table 4-2 are domestic wastewater treatment plants; there are no significant industrial discharges in the watershed of the Highland Lakes.

For treatment plants where phosphorus levels are regulated by permit, the level is typically 1 or 2 mg/l as a daily average. Total phosphorus levels were estimated for the remainder of the plants which are not subject to permitted limitations. Reported phosphorus content in untreated domestic wastewater exhibits an approximate range of 4-20 mg/l (Metcalf and Eddy, 1979; Steel and McGhee, 1979). Approximately 20-40 percent of the phosphorus is removed in aerobic biological wastewater treatment processes (Steel and McGhee, 1979; Stumm and Morgan, 1970). Assuming a 30 percent removal, treated effluent phosphorus levels of 3-14 mg/l would be anticipated. Armstrong, et al. (1985) determined an average total phosphorus concentration of 10 mg/l for domestic wastewater treatment plant effluent in the Lake Travis area. For the present analysis, total phosphorus levels for domestic discharges not subject to specific permit limitations were estimated at 10 mg/l.

D'-charge permit holders are required to report effluent characteristics to the TDWR on a routine basis. Self-reporting data (TDWR, 1983b) for area dischargers are summarized in Tables 4-3 and 4-4. Data for the five existing point sources which discharge directly to the reservoirs are displayed in Table 4-3. The Kingsland Municipal Utility District (MUD) and City of Marble Falls are the largest dischargers. In Table 4-4 are presented self-reporting data for the four domestic treatment plants adjacent to the Highland Lakes which dispose of treated effluent via irrigation.

4.1.1 Historical Loadings

Historical point source loads to the Highland Lakes were estimated using the information compiled in the preceding section. Relatively few point source discharges have been in existence in the watershed of the Highland Lakes. For determination of loads, only domestic treatment plant discharges were of concern--there have existed no significant industrial sources in the area. Estimated loads are presented in Table 4-5. Loads were based upon reported data where available. Where phosphorus data were unavailable, the effluent phosphorus level was assumed to be 10 mg/l. Only those point sources which discharge directly to the reservoirs were included in the loading calculation. A total of five point source permit holders have maintained direct discharges of wastewater into the reservoirs. Silver Creek Lodge discharges into Lake Buchanan. Lake LBJ receives discharges from Kingsland MUD and Commanche Point. The City of Marble Falls discharges into Lake Marble Falls, and Lake Travis receives discharges from the Travis Vista plant. The most substantial point source of phosphorus loads has been the Marble Falls plant, since it has not incorporated a phosphorus removal system. The total point source loading to the series of reservoirs for 1983 was estimated to be 6,540 kg/yr. Of this total, the Marble Falls discharge accounts for the major portion, 4,896 kg/yr, a result of its volumetric loading and absence of a permit phosphorus limitation.

4.1.2 Projected Loadings

In addition to the identification of historical loads, point source phosphorus loadings for future developmental conditions were projected. Population projections for the counties surrounding the Highland Lakes are displayed in Table 4-6, as developed by the TDWR (1983c). Anticipated population increments were applied (as percent increases) to the estimated phosphorus loadings under present conditions to obtain projected loadings for Marble Falls, Commanche Point, Kingsland MUD, and Silver Creek Lodge. For the Commander's Point plant, it was assumed that the plant would be constructed and the load would reach the allowable permit level by 1990 and remain at that level. Travis Vista is currently discharging in excess of its permit load. It was assumed that the load for this small development would remain at the existing level. It was assumed that the Villa on Travis plant would commence discharging and attain the allowable permit load by 1990, and remain at that level to the year 2000. Projected phosphorus loadings up to the year 2000 are presented in Table 4-7.

4.2 NONPOINT SOURCE LOADINGS

Several methods are available for determination of nonpoint source loads within a watershed. Comprehensive stormwater sampling surveys can be conducted in order to obtain detailed site-specific information on runoff flow and water quality. An alternate method involves a detailed review of historical water quality and hydrologic data within a watershed, which presupposes that a substantial localized data base is available. Nonpoint source loads can also be estimated from appropriate information developed across the nation as reported in the literature.

Nonpoint source loads are conventionally identified as a function of various land uses. Typical land use categories include urban, forest, agricultural, and pasture. In addition to these broad categories, more specific delineations are sometimes considered, such as residential, industrial, row crops, nonrow crops, etc.

Nonpoint source loading estimates are sometimes expressed in terms of mass generation per unit area of watershed per unit of time. Numerous publications have reported study results from individual watersheds as well as more general compilations of reported data (see, for example, Reckhow, et al., 1980; Rast and Lee, 1983; Loehr, 1974; Shannon and Brezonik, 1972). Constituent export rates are applied to delineations of land use acreages in order to obtain estimates of annual constituent loadings. In order to demonstrate the variability of reported export coefficients, loading rates for total phosphorus and total nitrogen from several studies are displayed in Table 4-8.

Another common technique for determination of nonpoint loads, which was employed in the present analysis, involves the specification of representative or average constituent concentrations for runoff flows. Representative concentrations may be estimated from reported values from studies conducted across the nation, or may be be based upon site-specific information.

Constituent concentrations are multiplied by the runoff volume, on either a single storm or an annual basis, in order to obtain estimates of runoff loads. Runoff volumes can be computed with a variety of hydrologic techniques. For the watershed of the Highland Lakes, nonpoint source phosphorus loads from major tributaries were determined from analysis of gauged streamflows and historical phosphorus concentrations. Estimates of runoff volumes from ungauged drainage areas adjacent to the series of reservoirs were obtained with the Soil Conservation Service (SCS) rainfall-runoff relationship (Mockus, 1972). The methodology incorporates an empirical variable called a curve number to transform rainfall into runoff volumes. The curve number is sensitive to several factors, including land use and soil types. Loads from the adjacent watersheds were defined based upon the SCS analysis of runoff volumes and an evaluation of phosphorus data from area tributaries. The following sections present a discussion of the historical data base for phosphorus concentrations in the study area tributaries, and a detailed description of the determination of phosphorus loads to the Highland Lakes from major tributaries and adjacent watersheds.

Definition of phosphorus loadings to the Highland Lakes requires that both incoming flows and phosphorus concentrations be specified. Stormwater phosphorus content has been reported in numerous studies across the country. Literature values are often employed in areas where site-specific data are not available. Several studies have attempted to correlate constituent concentrations or loads with a variety of watershed variables. Correlations have been attempted with land use type, impervious cover, slope, soil type, and a myriad of other variables with varying degrees of success. In general, data available in the literature display wide variation in most constituents, including phosphorus. Examples of reported constituent concentrations are shown in Table 4-9 (from Loehr, 1974). For many water quality management applications, the use of typical constituent concentrations reported in the literature is an acceptable procedure. But in general, it is preferable to employ site-specific data where available. Use of site-specific data may not be a panacea, since the utility of a particular data set is affected by its representativeness, temporal or spatial coverage, and numerous other factors. In many respects, the utility of a data set depends upon the intended application. For determination of long-term loads, a data set should include measurements over a variety of hydrologic or seasonal regimes.

Phosphorus data for major and minor tributaries in the Highland Lakes watershed were compiled and reviewed (TNRIS, 1983a; USGS, 1983a). Available data are derived primarily from the ongoing monitoring efforts of the TDWR and the USGS. The total phosphorus levels in the major tributaries were examined for direct input into the modeling analysis as incoming mass loadings. Minor tributaries were examined for characterization of runoff phosphorus levels from watersheds adjacent to the reservoirs.

4.2.1 Major Tributary Mass Loads

Tributary mass loads are a function of stream discharge rate and constituent concentration. Typically, tributary data bases are very limited. USGS streamflow monitoring stations are often available at many locations, providing a complete record of stream discharge rate in the form of mean daily flows. Comprehensive streamflow records from USGS gauging stations are available for the Colorado, Llano, and Pedernales Rivers in the study area. However, water quality sampling data are typically available on a more limited basis. In the watershed of the Highland Lakes, phosphorus concentration data are available from the USGS and the TDWR, but at best, data are available no more frequently than monthly. Thus, there exists a common disparity between the amount of flow data and chemical concentration data available at any one location.

In order to accurately characterize water quality for constituent loading calculations, collected data should be representative of extant conditions. There is no a priori test for representativeness. It can generally be assumed that more frequent collection of data provides a more accurate representation of ambient conditions. In some situations, routine sampling data may exhibit a bias towards low streamflow rates (Dolan, et al., 1981). In that event, use of the sampling data for estimation of an annual constituent loading rate could lead to either over- or under-estimation of the true loading. If low-flow constituent concentrations are lower than companion high-flow values, under-estimation of the annual load may result. Similarly, over-estimation of the annual load could be obtained in those cases where ambient constituent concentrations are higher under low-flow conditions. This potential error has its origins in situations where strong correlation is exhibited between streamflow rates and constituent concentration or loading.

4.2.1.1 Overview of Available Phosphorus Data

Historical phosphorus data are available for the three major tributaries in the Highland Lakes chain: the Colorado, Llano, and Pedernales Rivers. Monitoring stations examined in the present study are identified in Table 4-10 (station locations are depicted in Fig. 4-1). Two stations were analyzed on each of the tributaries. On the Colorado River above Lake Buchanan, a monitoring station (TDWR Sta. 1409.01) is located at Bend, Texas, at river mile 458.4 approximately 40 km (25 mi) above the reservoir. Water quality samples are collected at Bend by the TDWR. Farther upstream at river mile 474.3, the USGS maintains a streamflow and quality monitoring station near San Saba, Texas (USGS Sta. 08147000, TDWR Sta. 1409.015).

Monitoring stations on the Llano River are situated at Kingsland (river mile 6.5) and Llano, Texas (river mile 29.3). The TDWR collects water quality samples at the station near Kingsland (TDWR Sta. 1415.01), located approximately 3 km (2 mi) above Lake LBJ. The USGS monitors streamflow and water quality at the Llano station (USGS Sta. 08151500, TDWR Sta. 1415.02), and the TDWR collected water quality samples in the past.

The USGS maintains a streamflow gauge (USGS Sta. 08153500, TDWR Sta. 1414.01) on the Pedernales River near Johnson City, Texas, approximately 64 km (40 mi) above Lake Travis at river mile 48.0. Water quality is monitored at the station by both the USGS and the TDWR. A USGS streamflow gauging station is also located upstream near Fredericksburg at river mile 88.7 (USGS Sta. 08152900, TDWR

Sta. 1414.02). The TDWR collects water quality samples at the station near Fredericksburg.

Summary statistics for total phosphorus at the major tributary monitoring stations are displayed in Table 4-11. The table includes identification of observed ranges, means, and standard deviations for the data sets.

Data for the two stations on the Colorado River exhibit very similar mean values of total phosphorus: 0.155 mg/l and 0.118 mg/l. The stations are situated only 26 km (16 mi) apart, and the lower station drains an additional 137 km^2, less than one percent of the total watershed area at the upstream site. Therefore, data from the two stations were combined to represent instream conditions on the Colorado. With the combined data sets, a mean total phosphorus of 0.125 mg/l was obtained. A flow-weighted average total phosphorus of 0.557 mg/l was also calculated for the combined data sets. The flow weighted average is substantially greater than the arithmetic mean. The discrepancy is a consequence of a few data points (out of a combined total of 136 data points) collected at exceptionally high streamflow rates.

Llano River stations at Kingsland and Llano exhibit similar mean phosphorus levels of 0.026 mg/l and 0.040 mg/l, respectively. The difference in drainage area between the two stations is 542 km^2, which is only five percent of the total watershed area at the upstream station. The combined data sets were used to represent phosphorus concentrations in the Llano River. If the data sets for the two stations are combined, a mean of 0.035 mg/l total phosphorus is obtained. For the combined data set, a flow-weighted total phosphorus of 0.044 mg/l was calculated. Flows for the Kingsland station were estimated from observed discharge rates at the gauging station upstream at Llano.

Mean values of total phosphorus for the two stations on the Pedernales River exhibit a substantial difference--0.048 mg/l versus 0.137 mg/l--with the downstream station showing the lower concentration. The drainage area more than doubles between the two stations, which are located 66 km (41 mi) apart. The decrease in mean phosphorus level may be attributable to sedimentation in pool areas through the reach. A flow-weighted average total phosphorus of 0.057 mg/l was calculated for the data collected at the station near Johnson City, which is slightly higher than the arithmetic mean of 0.048 mg/l. The Johnson City data set was used to represent phosphorus concentrations in the Pedernales River.

4.2.1.2 Determination of Tributary Loads

There exists an assortment of methodologies for estimation of tributary mass loadings. The methodologies vary widely in complexity and sophistication. Selection of an appropriate methodology should be based upon the nature and extent of the available data sets, computational requirements, and the intended application. There is debate of course concerning the perceived precision and accuracy of alternative techniques. Frequently, the fundamental limitation affecting any load computation technique is the unavailability of a comprehensive data base. Where data are limited, complex statistical analyses may create an illusion of precision and accuracy, but no matter how elegant the analysis, the output is only as strong as the input data set.

Consideration and selection of an appropriate loading methodology is illustrated within the present analysis of the Highland Lakes. To initiate the selection process, the anticipated end result of the analysis must be considered. For the present analysis, the objective was to develop representative annual mass loading values for major tributaries. Annual phosphorus loads were needed for two potential applications. First, phosphorus loads representing historical inflows were needed. Optimally, historical streamflow values would be available on a year to year basis. Annual mass loads would be sensitive to historical meteorological conditions through their relationship to annual streamflows. Projection of future loading conditions represents the second potential application of annual phosphorus loads.

Several alternative methods may be employed for calculation of constituent loadings. Most commonly, available concentration data are averaged over some discrete period of time and multiplied by the corresponding observed discharge rate in order to obtain the load estimate. This method is variable in the duration of the time period for averaging. For example, concentrations could be averaged, and loads computed, over a weekly, monthly or annual term. As discussed previously, errors may potentially result if the sampling data collected in the averaging period are not representative of actual conditions, or more precisely, are not representative of the dynamic concentration fluctuations over the time frame of interest. Another approach involves application of a flow-concentration or flow-loading relationship to observed daily discharge rates. The method is attractive in that it makes use of an extensive streamflow data set where available on a watercourse. However, the success of the method is still dependent upon the availability of sufficient concentration sampling data to accurately define the relationship with streamflow. The use of a ratio estimator has been suggested to remove bias which may exist in the loading estimate (Tin, 1965; Dolan, et al., 1981). Application of the ratio estimator is based upon the premise that available concentration data are biased towards low-flow values. The ratio estimate is derived from the ratio of the mean of measured loads to the mean of flows when samples were collected. An estimate of the annual load is obtained by application of the ratio to the observed annual average flow. A bias adjustment term is included in the ratio estimator, derived from the sample set variance and covariance. The formula for the ratio estimator is as follows:

$$\mu_y = \mu_x \frac{m_y}{m_x} \left(\frac{1 + \frac{1}{n} \frac{S_{xy}}{m_x m_y}}{1 + \frac{1}{n} \frac{S_x^2}{m_x^2}} \right)$$

where μ_y = estimated load

μ_x = mean daily flow for the year

m_y = mean daily loading for the days on which concentrations were determined

m_x = mean daily flow for the days on which concentrations were determined

n = number of days on which concentrations were determined

$$S_{xy} = \frac{1}{(n-1)} \sum_{i=1}^{n} x_i y_i - n\, m_x m_y$$

$$S_x^2 = \frac{1}{(n-1)} \sum_{i=1}^{n} x_i^2 - n\, m_x^2$$

x_i = individual measured flows

y_i = daily loading for each day on which concentration was determined

The performances of several load calculation methodologies were investigated in the present analysis, as described below.

(i) Mean annual flow x historical mean concentration

This method involves the product of the mean annual streamflow rate for the tributary and the mean constituent concentration from the entire historical data base for the tributary to obtain an estimate of the annual mass load.

(ii) Mean annual flow x historical flow-weighted mean concentration

A flow-weighted mean concentration is calculated for the historical data base and multiplied by the mean annual streamflow rate to obtain an estimate of the annua mass load.

(iii) Mean annual flow x annual flow-weighted mean concentration

Annual loads are calculated as the product of the mean annual flow and the flow-weighted average concentration for each year's sampling days. Upon rearrangement, the method is equivalent to multiplication of the average annual sampling day load times the ratio of the mean annual flowrate to the mean annual sampling day flowrate. The latter formulation is known as the simple ratio estimator (Cochran, 1977).

(iv) Sampling day flow x sampling day concentration

With this method, the flowrate and concentration for each sampling day are converted to loads. An annual load is obtained from the calculated average sampling day load over a specific time interval.

(v) Unbiased ratio estimator

The unbiased ratio estimator described above was employed to calculate annual loads. The unbiased ratio estimator is equivalent to the simple ratio estimator multiplied by a bias adjustment term derived from the sampling variance and covariance.

(vi) Load-flow relationship

Relationships between streamflow rate and mass loading rate were investigated for tributary data sets. The best-fit relationship was applied to the available records of mean daily streamflow to calculate daily loads, which were then summed over a one-year period for an estimate of annual load.

4.2.1.3 Analysis of Colorado River Phosphorus Loads

A detailed analysis of Colorado River phosphorus loads was conducted in order to select an appropriate load calculation methodology. The performances of alternative methodologies were reviewed for the Colorado River because it has available the most substantial data base among the major tributaries. The method selected for calculation of phosphorus loads on the Colorado was then employed for the other major tributaries.

4.2.1.3.1 Flow and Concentration Data

The determination of mass loadings from the Colorado River to Lake Buchanan illustrates the methodology selection process. Optimally, records of gauged inflow and phosphorus concentrations would exist throughout the history of the reservoir. If sufficient data are available, mass loadings could be characterized on, say, a daily basis by continual integration of the streamflow and constituent concentration data. The time series of loadings so obtained would provide delineation of phosphorus loads delivered by the watercourse over any particular time frame of interest. However, such extensive data sets are usually not available. For the Colorado River, the frequency of the data is insufficient for determination of a detailed loading analysis on a daily or even monthly interval. Further, data do not extend far enough back in time to conduct a complete analysis of historic loadings. Though gauged inflow data are available as mean daily discharge rates for the Colorado River above Lake Buchanan since closure in 1932, a total of only 136 instantaneous measurements of total phosphorus concentration have been collected, over the period 1968 to 1982.

The data set for the Colorado ver is comprised of total phosphorus measurements from two stations, TDWR Sta. 1409.01 and 1409.015, supplemented with mean daily discharge rates at 1409.015 (USGS Sta. 08147000). The Colorado River flow and concentration data sets were examined prior to estimation of loads. The frequency distribution of discharge rates was investigated for both a complete historical record and the subset of sampling day measurements. The flow frequency distributions for the two data sets, displayed in Fig. 4-2, exhibit close agreement. Median discharge rates are similar: 6.8 m^3/s with the complete historical record and 6.6 m^3/s with the sampling day subset. The curves diverge slightly in the high-flow frequency region.

The similarity of the frequency distribution curves demonstrates that the sampling data flow rates are indeed representative of the historical flow regime.

The distribution of total phosphorus concentration data for the Colorado River is illustrated in a histogram in Fig. 4-3, and a cumulative frequency diagram is presented in Fig. 4-4. Total phosphorus measurements range from less than 0.01 mg/l to 1.98 mg/l, with an arithmetic mean va ue of 0.125 mg/l. A median phosphorus concentration of 0.066 mg/l is displayed in the cumulative frequency diagram. The flow-weighted mean concentration of the phosphorus data set is 0.557 mg/l, which is substantially greater than the arithmetic mean. As indicated above, the discrepancy is a consequence of a few data points (out of a combined total of 136 data points) collected at exceptionally high flowrates. Deletion of the two highest and two lowest extreme flow rate data points results in a flow-weighted average concentration of 0.248 mg/l. Because the exceptionally high-flow data points occur very infrequently, the latter flow-weighted average value should provide a more representative characterization of the year-to-year annual average phosphorus level. Exclusion of extremely high-flow data points is justified in order to prevent an unrealistic weighting of the long-term average phosphorus concentration.

The Colorado River data set was examined for relationships between flow and total phosphorus concentration. Regressions were run for the following formulations:

linear function:	$y = a + bx$
logarithmic function:	$y = a + b \log x$
exponential function:	$y = ae^{bx}$
power function:	$y = ax^{b}$

In these equations, the dependent variable y is constituent concentration (mg/l), the independent variable x is streamflow rate (m^3/s), and a and b are constants. Results of the regression analysis are displayed in Table 4-12. Three of the relationships have similar correlation coefficients (r) and standard errors, with the linear fit exhibiting the highest r of 0.57 and the lowest standard error.

Similarly, relationships between flow and loading rate were investigated. Daily loading rate (g/d) was calculated as the product of flowrate and constituent concentration, with appropriate conversion factors. Loading relationships are shown in Table 4-13. Correlation coefficients are higher than in the concentration analysis, due to the normalizing effect of regressing flow on a term that also contains flow. The power function exhibited the highest correlation coefficient, 0.91, and the linear fit displayed an r of 0.80. The standard error was lowest for the linear relation, with the power function next lowest. Normally, the equation with the lowest standard error would be selected as the best fit to the observed data. The linear relation, however, displayed a large negative intercept on the ordinate. Because of this negative intercept, the equation predicts negative phosphorus mass loadings for flowrates less than approximately 7.0 m^3/s, which on the Colorado, occur historically roughly 50 percent of the time. The linear relationship is thus unacceptable for prediction of mass loadings on the Colorado River, and the power function was selected. The power function is superimposed over the data set in Fig. 4-5.

The selection of a loading relationship has a significant effect upon predicted loads. In Table 4-14 are shown predicted loads for the alternative relationships at a series of flowrates. The variability of predicted loads among the alternative equations is substantial. While a best-fit relationship for the limited historical data set may be selected on the basis of standard error or correlation coefficient, the available data are simply insufficient to impart a large degree of confidence in the accuracy of predicted loads via any of the relationships.

4.2.1.3.2 Comparison of Loads from Alternative Methods

Annual phosphorus loads for the Colorado River above Lake Buchanan were estimated for the years with available concentration sampling data, 1968 to 1982, using the methods described in Sec. 4.2.1.2. In Table 4-15 and Fig. 4-6 are displayed the series of calculated annual phosphorus loads. Substantial variability is evident among alternative methods. There is no indication of a systematic trend among the alternatives. The methods based upon the product of annual flow and concentration provide loads which correspond to the fluctuations in recorded flows. Loads derived from the ratio estimator display less correspondence to flows, and the year-to-year variability of predicted loads appears to be greater. Predictions derived from the load-flow relationship are generally similar to the flow-concentration methods.

For the 15-year historical period of record, the mean load and standard deviation were calculated for each alternative method. A t-test was applied to estimate the confidence bounds upon the mean at a 0.05 level of significance. Results of the analysis are shown in Table 4-16. The predictions obtained with the methods utilizing long-term average concentrations, as well as the load-flow relationship, display lower standard deviations and narrower confidence ranges than results from the ratio estimator methods, which employ an annual accounting of concentration data. The means and confidence bounds of the long-term average methods are encompassed within the confidence ranges of the ratio techniques. There is no evidence to support the accuracy of any of the alternative methods, since comprehensive site-specific test data do not exist. However, it is apparent that use of the ratio estimator technique imparts a much wider confidence range upon the mean.

In the absence of test data, an unequivocal selection of the most appropriate method for calculation of loads cannot be resolved. As a framework for the selection process, it would be prudent to disregard those methods that predict extremely large or small loads, and favor instead methods that appear moderate in performance. In addition, the chosen method should display correspondence to the annual tributary flow. While the ratio estimator has been demonstrated to perform admirably in other areas, the concentration data set available for the watershed of the Highland Lakes is simply too sparse to impart confidence in the technique. The key shortcoming in the data set is that the frequency of concentration sampling is not adequate to reflect the actual phosphorus dynamics within the tributary. For the present situation, the method of multiplying average annual flow times historical flow-weighted mean concentration is probably adequate for use and is certainly the simplest technique.

With this method, deficiencies in the annual phosphorus data set are recognized and replaced with the compromise of the long-term flow-weighted average concentration.

The load-flow relationship developed from the historical sampling data set was selected for use in the present analysis. The relationship is based upon the complete set of observed phosphorus loadings, though admittedly the set is limited. Further, the method maximizes the use of the historic streamflow data, which are available as records of mean daily discharge. The load-flow relationship method thus makes the best use of all available historical data for the Colorado River. The method will be employed to determine the long-term historical phosphorus loads and predict future loads. The accuracy of predicted loads derived from the load-flow relationship cannot be assessed in the absence of adequate test data (see comparison of alternative loads in Table 4-14 and Table 4-15). However, the selected method is anticipated to perform as well as possible given the fundamental limitations.

4.2.1.3.3 Historical Loadings

Determination of historical phosphorus loads in the present analysis covered the period 1952-1983. Annual loads were estimated with the load-flow relationship described previously, using recorded daily streamflow rates for the Colorado River at San Saba, Texas. Phosphorus loads for the Colorado were adjusted to represent the station downstream at Bend, Texas by application of a drainage area correction factor of 1.007 to the streamflow data. Estimated historical loads are shown in Table 4-17.

4.2.1.3.4 Projected Loadings

Prior to the estimation of future loadings, the streamflow records for the Colorado River were examined for an evaluation of the effects of hydrologic modifications in the watershed. Completion of Robert Lee Dam on E. V. Spence Reservoir in the upper watershed in 1968 appeared to effect a reduction in average streamflow rates, as described in Appendix A.

For the present analysis, future loadings were projected with the load-flow relationship described previously, which requires specification of future daily flows. To forecast future flows, it was assumed that historical flow regimes would be repeated. Historical flow records for 1967-1983 were used to simulate post-1983 flow conditions to reach the year 2000. Prior to 1969, recorded flows, and thus loads, were reduced according to the ratio of the synthesized annual mean discharge (see Appendix A) to the reported annual mean discharge in order to account for the hydrologic modification in the watershed. Since the flow records are for the San Saba gauge, a drainage area correction factor of 1.007 was applied to adjust flows to the location downstream at Bend, Texas. Projected phosphorus loadings are shown in Table 4-18.

4.2.1.4 Analysis of Llano River Phosphorus Loads

4.2.1.4.1 Flow and Concentration Data

The data set for the Llano River is comprised of total phosphorus measurements from stations at Kingsland (TDWR Sta. 1415.01) and Llano, Texas (TDWR Sta. 1415.02). A total of 76 phosphorus measurements are available in the combined data set, collected over the period 1972-1982. Streamflow records are available at the Llano station (USGS Sta. 08151500).

The frequency distribution of observed streamflows was investigated for both a complete historical record and the subset of sampling day measurements. Flow frequency distributions for the two data sets are displayed in Fig. 4-7. The median discharge for the historical data set is 4.0 m^3/s, compared to 6.5 m^3/s for the sampling day flows. Generally, the sampling day flows represent higher than normal discharge rates, based upon the complete historical record.

The frequency distribution of total phosphorus data for the Llano River is shown in Fig. 4-8. The distribution is strongly skewed toward small concentrations. The cumulative frequency distribution for observed phosphorus data is displayed in Fig. 4-9, which indicates a median phosphorus concentration of 0.019 mg/l. The arithmetic mean of the phosphorus data set is 0.035 mg/l, and the flow-weighted mean concentration is 0.044 mg/l.

The Llano River phosphorus data set was analyzed for relationships between flow and concentration, as described in the preceding section on the Colorado River analyses. Regression results are displayed in Table 4-19. The standard error is similar for the four relationships investigated, with the linear and logarithmic fits displaying the lowest value. Correlation coefficients are low for all relationships. Regression with the exponential relationship provided the highest r, 0.24.

Relationships between flow and phosphorus loading rate were similarly investigated. Loading relationships are shown in Table 4-20. As with the Colorado River analysis, correlation coefficients are higher with the regression on oadings. The linear fit provides the lowest standard error and the highest correlation coefficient. However, a large negative intercept on the ordinate precludes use of the linear relationship. The power relationship displayed the next lowest standard error and next highest correlation coefficient, and was therefore selected for use in the analysis of Llano River phosphorus loadings. In Fig. 4-10, the power relationship is shown along with the phosphorus data set.

4.2.1.4.2 Historical Loadings

Phosphorus loadings were calculated for the period 1952-1983. The flow-load relationship was used to estimate annual phosphorus loadings, based upon recorded daily streamflow rates for the Llano River at Llano, Texas. A drainage area correction factor of 1.05 was applied to streamflow data in order to reference the selected loading point downstream at Kingsland. Estimated historical phosphorus loads are presented in Table 4-17.

4.2.1.4.3 Projected Loadings

As with the Colorado River, future phosphorus loadings for the Llano River were projected based upon repetition of historical flow regimes. Streamflow records for Sta. 08151500 were available for the period 1939-1983. Discharge rates can be directly employed for the calculation of loads due to the absence of significant hydrologic modifications in the watershed. Historical flow records for 1967-1983 were used to simulate post-1983 flow conditions up to the year 2000, as shown in Table 4-18.

4.2.1.5 Analysis of Pedernales River Phosphorus Loads

4.2.1.5.1 Flow and Concentration Data

Pedernales River discharge rates and water quality are monitored at Johnson City, Texas (TDWR Sta. 1414.01, USGS Sta. 08153500). A total of 24 phosphorus measurements were available for the period 1969-1981.

The frequency distribution of observed sampling day streamflows is compared to the complete historical flow record in Fig. 4-11. The historical median flowrate is 1.3 m^3/s, while the median flow for the sampling days is 2.8 m^3/s. The frequency curve for the sampling day flows generally lies above the historical trend.

Figure 4-12 shows the phosphorus frequency distribution for the Pedernales River data set. A skew toward small concentrations is exhibited. The cumulative frequency distribution for observed phosphorus data is displayed in Fig. 4-13, which indicates a median concentration of 0.042 mg/l. For comparison, the data set has an arithmetic mean of 0.048 mg/l and a flow-weighted mean of 0.057 mg/l total phosphorus.

A regression analysis between flow and concentration was conducted for the Pedernales River data set, as shown in Table 4-21. Standard errors were similar for the four relationships investigated. The highest correlation coefficient, 0.26, was obtained with the exponential fit.

Relationships between flow and loading rate were examined for use in the estimation of phosphorus loadings. Regression parameters are shown in Table 4-22. Though the linear fit exhibited the lowest standard deviation and the highest correlation coefficient, it displayed a large negative intercept on the ordinate. The power relationship was selected for estimation of loads, since it had the next smallest standard error and next highest correlation coefficient. The power fit is superimposed upon the phosphorus data set in Fig. 4-14.

4.2.1.5.2 Historical Loadings

The flow-load relationship developed in the preceding section was employed to calculate historical phosphorus loadings for the period 1952-1983, as shown in Table 4-17. Recorded daily streamflow rates at the Johnson City gauge provided the basis for load calculations. A drainage area correction factor of 1.37 was applied to streamflow data in order to adjust phosphorus loads to a downstream location below the confluence with Cypress Creek, near the FM 962 crossing.

4.2.1.5.3 Projected Loadings

Phosphorus loadings for the Pedernales River were projected under the assumption that historical flow regimes would be repeated. Streamflow records for USGS gauge 08153500 were available for the period 1939-1983. In the absence of significant hydrologic modifications in the watershed, historical discharge rates can be directly employed. Projected loads are shown in Table 4-18, based upon use of historical flow records for the period 1967-1983 to simulate post-1983 conditions.

4.2.1.6 Analysis of Sandy Creek Phosphorus Loads

4.2.1.6.1 Flow and Concentration Data

The historical data base for Sandy Creek is limited. Phosphorus data are generally not available, but streamflow records are available for the period 1966-1983 for a gauging station (08152000) at the Hwy. 71 crossing near Kingsland, Texas. The historical flow frequency distribution for Sandy Creek is shown in Fig. 4-15.

For the present analysis, it was assumed that Sandy Creek characteristics would be similar to the Llano River, in terms of flow frequencies and phosphorus concentrations, due to the proximity of the two watersheds. Thus, the flow-load relationship developed for the Llano was employed for estimation of Sandy Creek phosphorus loadings.

4.2.1.6.2 Historical Loadings

Phosphorus loadings were calculated for the period 1952-1983, using the flow-load relationship developed for the Llano River. Historical flow records for the Sandy Creek gauge near Kingsland were used for the years 1967-1983. Prior to 1967, flow records for the Llano River gauge at Llano, Texas were employed, with application of a drainage area correction factor of 0.0825. Historical loads are shown in Table 4-17.

4.2.1.6.3 Projected Loadings

Historical flow records for Sandy Creek for the period 1967-1983 were used to simulate the projected post-1983 flow regimes. Projected phosphorus loadings to the year 2000 are shown in Table 4-18.

4.2.1.7 Releases from Upstream Reservoirs

The Colorado River drains directly into Lake Buchanan at the head of the Highland Lakes. The Colorado continues to provide input to all of the reservoirs, albeit in the form of releases from upstream impoundments. These reservoir releases may provide substantial phosphorus loadings. Loadings associated with releases from upstream impoundments will be evaluated with the mathematical modeling exercises. Historical data for characterization of the phosphorus content of the releases is extremely limited. In the present study, the phosphorus concentration will be estimated from the concentration within the reservoir pool calculated through the mass budget analysis.

4.2.2 Loadings from Adjacent Areas to the Highland Lakes

Loadings from major tributaries to the Highland Lakes were defined in a straightforward manner using streamflow and water quality data from the various monitoring stations situated on the watercourses. However, the watershed areas contiguous to the shoreline of the reservoirs are generally ungauged. These adjacent watershed areas represent a source of potentially significant loadings to the reservoirs. Loadings from the adjacent watersheds are predominantly associated with stormwater runoff which drains into the reservoirs. The potential significance of runoff loads is greatly enhanced by the occurrence of increasing urban development within the adjacent watershed areas. Urban development can produce an increase in phosphorus loading via the associated increase in runoff volumes due to increased impervious cover and by the introduction of additional sources of phosphorus in an urban environment.

Runoff volumes from adjacent areas were determined through a hydrologic analysis based upon Soil Conservation Service (SCS) methods. The SCS method is based upon an empirical relationship between runoff volume, precipitation, and retention storage (Mockus, 1972). Evaluations of land use, soil classifications, and rainfall statistics are integral components of the methodology, as described in the following sections.

4.2.2.1 Overview of Available Phosphorus Data

4.2.2.1.1 Minor Tributaries in the Study Area

A comprehensive water quality monitoring program on several streams in the Austin urban area was initiated in 1975 by the USGS, in cooperation with the City of Austin. An objective of the program was to determine the variations in water quality during different seasons and flow conditions in representative watersheds under various types of urban development (USGS, 1983b). Major streams in the study area include Onion Creek, Barton Creek, Walnut Creek, Bull Creek, Boggy Creek, Shoal Creek, Williamson Creek, Slaughter Creek, Bear Creek, and Waller Creek. All of the streams are within the watershed of the Colorado River, though several drain into the reach below Town Lake. By design, the data base includes several samples collected under storm flow regimes.

Locations of Austin-area water quality monitoring stations are displayed in Fig. 4-16. Sampling stations examined in the present analysis are described in Table 4-23. Stations were selected to be generally representative of rural, urban, and mixed rural-urban areas. Summary statistics for phosphorus measurements at the tributary stations are shown in Table 4-24.

Data for tributaries in the Austin urban area exhibit a wide range of mean total phosphorus concentrations, from a low of 0.025 mg/l at Onion Creek near Driftwood, up to 0.478 mg/l at Shoal Creek. The data display some correlation with the degree of urbanization in the various watersheds. Those drainage areas located within highly urbanized areas of Austin generally feature the highest mean phosphorus levels. Stations which reflect relatively high urbanization include Shoal Creek at 12th St., Boggy Creek at Hwy 183, Walnut Creek at Webberville Rd., and Williamson Creek at Oak Hill and at Jimmy Clay Rd. Stations within predominantly rural areas are characterized by lower levels of mean phosphorus. Rural stations include Onion Creek near Driftwood, Barton Creek at Hwy 71, and Bear Creek below FM 1826. Intermediate levels of phosphorus are indicated for watersheds with a mixture of urban and rural characteristics, such as Bull Creek at Loop 360, Barton Creek at Loop 360, Onion Creek at Buda, Onion Creek at Hwy 183, Slaughter Creek at FM 2304, and Little Bear Creek at FM 1626. The preceding categorization of the sampling stations is qualitative-- substantial overlap beteen rural and urban areas is inevitable. However, the generalization holds best at the extremes of the ordering, namely, for either predominantly rural or highly urban watersheds.

Mean phosphorus concentrations for each station were grouped according to the preceding generalization. A mean value (of individual station means) was then calculated for the undeveloped, intermediate development, and highly urbanized categories. (The individual phosphorus measurements for the stations were not grouped and analyzed because this would weight the calculation toward stations with the greatest number of data points.) The confidence range for the mean value for each category was determined with a t test at a 0.05 level of significance (presented as lower bound $\leq$ sample mean $\leq$ upper bound):

largely undeveloped areas	$0.013 \leq 0.028 \leq 0.043$ mg/l
intermediate development	$0.063 \leq 0.122 \leq 0.181$ mg/l
highly urbanized areas	$0.113 \leq 0.341 \leq 0.569$ mg/l

Smoothing the boundaries among these groupings, general categories of mean phosphorus levels are indicated as follows:

largely undeveloped areas	≃ 0.02-0.04 mg/l
intermediate development	≃ 0.04-0.20 mg/l
highly urbanized areas	≃ 0.20-0.60 mg/l

These observations are based upon analysis of reported data at all flowrates.

The data from the Austin-area tributaries were also screened for isolation of high-flow data. To differentiate beteen low and high-flow data, a streamflow rate of 0.28 m^3/s (10 cfs) was established as an arbitrary minimum requisite for inclusion in the storm runoff data base. For the Austin-area tributaries, this flowrate should be sufficient for classification as storm-induced runoff. Screened high-flow data are summarized in Table 4-25. In general, tributary storm flows exhibit mean phosphorus levels in excess of average values characteristic of complete (low- and high-flow) data sets.

Mean high-flow phosphorus concentrations at each station were grouped into undeveloped, intermediate development, and highly urbanized categories as described above. A mean value for each category was calculated, and the confidence range for the mean was determined with a t test at a 0.05 level of significance (presented as lower bound ≤ sample mean ≤ upper bound):

largely undeveloped areas	$0.021 \leq 0.032 \leq 0.043$ mg/l
intermediate development	$0.070 \leq 0.146 \leq 0.222$ mg/l
highly urbanized areas	$0.107 \leq 0.513 \leq 0.919$ mg/l

If the boundaries among the groupings are generalized, the high-flow data indicate the following typical classifications:

largely undeveloped areas	≃ 0.02-0.05 mg/l
intermediate development	≃ 0.05-0.20 mg/l
highly urbanized areas	≃ 0.20-1.00 mg/l

Though these generalizations are subject to limitations, the delineations provide adequate guidelines for estimation of runoff phosphorus concentrations.

4.2.2.1.2 Supplemental Studies

Runoff water quality data for two study areas in the Colorado River basin were collected in conjunction with programs developed to satisfy the requirements of Section 208 of the Federal Water Pollution Control Act Amendments of 1972. In addition, a sampling program was conducted in the Austin area as part of the Nationwide Urban Runoff Program. These supplemental sources of data are described below.

<u>Austin Intensive Planning Area 208 Study</u>

Stormwater data were collected in the Austin Intensive Planning Area during 1977 (Altman, 1978a). Sampling stations were established in complex land use areas, areas having predominantly one land use, and the Colorado River at Austin. Data were collected during two storm events. Estimated rainfall depths in the study watersheds ranged from 1.80-2.92 cm (0.71-1.15 in) for the first storm and 1.65-2.74 cm (0.65-1.08 in) for the second. Stormwater total phosphorus levels for six stations are summarized in Table 4-26. The highest average phosphorus concentration, 0.52 mg/l, was observed in the West Bouldin Creek watershed, which features a land use breakdown of 46% low density residential, 5% high density residential, 15% commercial/public, and 34% undeveloped area. The Ferguson Branch watershed, with 91% undeveloped and 9% low density residential area, was characterized by the lowest observed average total phosphorus concentration, 0.12 mg/l.

San Angelo Intensive Planning Area 208 Study

Stormwater data were collected in the San Angelo Intensive Planning Area to characterize runoff entering the Concho River, a tributary of the Colorado River (Altman, 1978b). Data were collected during three storm events in 1977 and 1978, with rainfall amounts of 1.55-3.18 cm (0.61-1.25 in), 1.52-2.39 cm (0.60-0.94 in), and 1.32-5.61 cm (0.52-2.21 in). Phosphorus data for seven sampling stations are summarized in Table 4-27. A total phosphorus concentration of 0.61 mg/l in Red Arroyo was the highest average level encountered. The Red Arroyo watershed is predominantly undeveloped (77%), with a portion (15%) of single-family residential area. The lowest average total phosphorus concentration, 0.23 mg/l, was observed in the South Concho River which drains a nonurban watershed above the study area.

Austin National Urban Runoff Program

The Austin Nationwide Urban Runoff Program (NURP) sampling study was conducted in conjunction with a nationwide series of urban runoff studies funded by the U.S. Environmental Protection Agency (Engineering-Science, Inc., 1983). The Austin program had two primary aspects: stormwater monitoring and receiving water monitoring. Sampling surveys were conducted in 1981. Watersheds were selected for study on the basis of differences in the level of impervious cover or availability of structural control devices. Three watersheds were selected for study: Northwest Hills, Rollingwood, and Turkey Creek.

The Northwest Hills site was characterized by moderate density residential land use served by central sewer service. With a residential density of 3-4 lots per acre, the watershed had a 39 percent impervious cover level. Two stations were sampled in the Northwest Hills watershed, one of which monitored releases from the Woodhollow Dam stormwater drainage detention facility. The Rollingwood site featured low density residential land use with onsite wastewater disposal in septic tank/drainfield systems. Average residential density in the watershed is 1-1.3 lots per acre, and the impervious cover level is approximately 21 percent. The Turkey Creek watershed is undeveloped, with an estimated impervious cover level of 1 percent.

Observed concentrations of phosphorus in stormwater runoff are summarized in Table 4-28. Mean phosphorus levels in the two urbanized watersheds were very similar at approximately 0.3 mg/l. Background and stormwater total phosphorus levels at the undeveloped site in the Turkey Creek basin were substantially lower, with an average of 0.1 mg/l phosphorus for nine runoff samples.

All of the data collected in the Austin NURP study were from relatively small storms. The maximum runoff depth was 0.18 cm (0.07 in). The runoff coefficient (runoff volume/rainfall volume) was higher for the Northwest Austin site than for the Rollingwood site, a consequence of the increased impervious cover. The increased runoff produced greater mass loads per unit of area in the Northwest Austin watershed.

Samples at the runoff detention basin were collected for only two storms. For the first event, the incoming phosphorus load was estimated at 245.2 g (0.54 lb), while the outflow load was calculated to be 204.3 g (0.45 lb) at a hydraulic detention time of 15.5 hours, a reduction of 17 percent. For the second storm event, a reduction in load of 61 percent was calculated for a set of samples at a detention time of 20.5 hours. Another data set at a detention time of 3.6 hours indicated an increase in phosphorus load (25 percent) from the basin.

Receiving water conditions were monitored as part of the NURP study. Samples were collected from Lake Austin and Town Lake to assess ambient, storm, and post-storm conditions. In general, some short-term effects upon water quality were evident, although long-term impacts were not observed in response to storm loads. Phosphorus data for the receiving waters are displayed in Tables 4-29 and 4-30, and sampling station locations are described in Table 4-31.

4.2.2.2 Watershed Delineations

A series of discrete drainage areas surrounding the Highland Lakes was delineated on USGS topographic maps (1:250,000 scale). Watersheds are displayed in Fig. 4-17. The drainage areas were defined such that all contributing areas not covered by referenced loading points for the major tributaries were accounted for. Watersheds were delineated within each of the respective drainage areas of the reservoirs which comprise the Highland Lakes. Overlap of delineated watersheds between sequential reservoir drainages was avoided. In this manner, the watersheds represent discrete sources of loads to individual reservoirs. Table 4-32 contains a description of the contiguous watersheds.

4.2.2.3 Land Use

Land use information for the adjacent watersheds delineated around the Highland Lakes was obtained from a compilation of data developed in conjunction with the water quality management planning activities under Section 208 of the Federal Water Pollution Control Act Amendments of 1972. The published information was compiled by manual interpretation of 1973 and 1976 Landsat imagery at a scale of 1:250,000 and categorized according to Level I interpretation as defined by the USGS (TDWR, 1977). This effort represents the only complete land use mapping coverage of the area contiguous to the Highland Lakes. While the resolution is limited, the land use information is sufficient for the needs of the present analysis.

Land use categories from the published information utilized for the study area watersheds include the following: urban, cropland (a combination of dry and irrigated cropland), forest, and rangeland. Land use categories are defined by the USGS (1976) as follows:

Urban or Built-up Land

Urban and built-up land comprises areas of intensive use with much of the land covered by structures. Included in this category are cities, towns, villages, strip developments along highways, transportation, power, and communications facilities, and such isolated units as mills, mines, and quarries, shopping centers, and institutions.

Agricultural Land

Agricultural land may be defined broadly as land used primarily for production of food and fiber. On high-altitude imagery, the chief indications of agricultural activity are the distinctive geometric field and road patterns on the landscape and the traces produced by livestock or mechanized equipment.

Agricultural land includes cropland, pasture, orchards, groves, vineyards, nurseries, ornamental horticultural areas, confined feeding operations and any land associated with the operations of the mentioned agricultural land uses.

Rangeland

Rangeland historically has been defined as land where the potential natural vegetation is predominantly grasses, grasslike plants, forbs, or shrubs and where natural herbivory was an important influence in its precivilization state. Management techniques which associate soil, water and forage-vegetation resources are more suitable for rangeland management than are practices generally used in managing pastureland. Some rangelands have been or may be seeded to introduced or domesticated plant species.

Forest Land

Forest lands have a tree-crown areal density (crown closure percentage) of ten percent or more, are stocked with trees capable of producing timber or other wood products, and exert an influence on the climate or water regime. Forest land generally can be identified rather easily on high-altitude imagery, although the boundary between it and other categories of land may be difficult to delineate precisely.

Land use descriptions for the immediate drainage areas of the reservoirs are presented in Table 4-33, using 1983 conditions as a reference year. The published land use data (TDWR, 1977) was assumed representative of 1976 conditions in the watershed of the Highland Lakes. Estimates of historical and projected land use characteristics were obtained by adjustment of the 1976 areas by the rate of increase or decrease in county population. The immediate drainage areas were assigned the growth rate of the county which encompasses the majority of the subbasin. With this adjustment technique, urban area was assumed to increase or decrease in accordance with population growth. Changes in urban areas were assumed to be offset by corresponding increases or decreases in rangeland area. Rangeland is the predominant land use category in the adjacent watersheds, incorporating roughly 75 percent of the total area.

4.2.2.4 Soils

Soil descriptions for the adjacent watersheds were obtained from information available on general county soil maps prepared by the SCS. The delineated watersheds encompassed portions of San Saba, Lampasas, Llano, Burnet, Blanco, and Travis Counties. Though some of these counties have available detailed soil surveys, the general soil maps were sufficient for the needs of the present analysis. A general soil map displays mapping units with a distinct pattern of soils, relief, and drainage. A mapping unit typically consists of one or more major soils and some minor soils, and it is named for the former. The soils comprising one unit can occur in other units, albeit in a different pattern. A broad perspective of the soils in the county survey areas is provided by the general soil map (SCS, 1974, 1978, 1979a, 1979b, 1982a).

General soil descriptions for the adjacent watersheds are shown in Table 4-34. The table also identifies the associated soil hydrologic group. The hydrologic classification is based upon soil infiltration rates under wetted conditions. Four hydrologic soil groups have been identified (Mockus, 1972):

A. (Low runoff potential) Soils having high infiltration rates even when thoroughly wetted and consisting chiefly of deep, well to excessively drained sands or gravels. These soils have a high rate of water transmission.

B. Soils having moderate infiltration rates when thoroughly wetted and consisting chiefly of moderately deep to deep, moderately well to well drained soils with moderately fine to moderately coarse textures. These soils have a moderate rate of water transmission.

C. Soils having slow infiltration rates when thoroughly wetted and consisting chiefly of soils with a layer that impedes downward movement of water, or soils with moderately fine to fine texture. These soils have a slow rate of water transmission.

D. (High runoff potential) Soils having very slow infiltration rates when thoroughly wetted and consisting chiefly of clay soils with a high swelling potential, soils with a permanent high water table, soils with a claypan or clay layer at or near the surface, and shallow soils over nearly impervious material. These soils have a very slow rate of water transmission.

The hydrologic classification of a soil affects the estimation of direct runoff in response to a rainfall event. As evidenced in Table 4-34, hydrologic groups C and D predominate in the adjacent watersheds.

4.2.2.5 Precipitation

The SCS hydrologic method employed in the present analysis was developed to compute runoff from a storm event. An analysis of the distribution of rainfall in the Highland Lakes watershed was conducted to define the annual frequency distribution of precipitation events, and ultimately compute an average annual runoff volume. Reported yearly precipitation characteristics were used to define historical runoff conditions. Subsets of the historical precipitation records were employed to represent projected future conditions, similar to the use of historical streamflow records.

Two long-term precipitation records were selected to represent average conditions in the Highland Lakes watershed, from stations at Austin and Llano, Texas. Complete daily precipitation records for each station were obtained from the TNRIS (1983b). A storm frequency analysis was conducted on the precipitation records for the period 1931-1982 at each station. A computer program was developed to produce the storm-frequency data. The program code discretizes the observed range of precipitation into uniform intervals to compute frequencies. Output from the program displays the precipitation intervals, the number of events within the intervals, the frequency associated with each interval, and the cumulative frequency. Storm frequency distributions for stations at Austin and Llano are displayed in Table 4-35 and Table 4-36, respectively.

The data indicate that the most frequently occurring storm events involve rainfall totals less than or equal to 0.25 cm (0.1 in). Storms of 2.54 cm (1.0 in) or less comprise 88 percent of the historical distribution at both stations. An average of 80.13 storm events occur per year at the Austin station, while the Llano data indicated a storm frequency of 63.35 storms per year. The long-term annual average rainfall at the Austin and Llano stations was determined to be 82.27 cm (32.39 in) and 68.66 cm (27.03 in), respectively.

4.2.2.6 Runoff

The SCS method for estimation of direct runoff (Mockus, 1972) from storm rainfall was designed for application to ungauged watersheds, using rainfall and watershed data that are ordinarily available. The principal application of the method is the estimation of runoff volumes in flood hydrographs or in relation to flood peak discharge rates. The SCS (Mockus, 1972) has distinguished four types of runoff:

> Channel runoff occurs when rain falls on a flowing stream or on the impervious surfaces of a streamflow-measuring installation. It appears in the hydrograph at the start of the storm and continues throughout it, varying with the rainfall intensity. It is generally a negligible quantity in flood hydrographs, and no attention is given to it except in special studies.

Surface runoff occurs only when the rainfall rate is greater than the infiltration rate. The runoff flows on the watershed surface to the point of reference. This type appears in the hydrograph after the initial demands of interception, infiltration, and surface storage have been satisfied. It varies during the storm and ends during or soon after it. Surface runoff flowing down dry channels of watersheds in arid, semiarid, or subhumid climates is reduced by transmission losses, which may be large enough to eliminate the runoff entirely.

Subsurface flow occurs when infiltrated rainfall meets an underground zone of low transmission, travels above the zone to the soil surface downhill, and appears as a seep or spring. This type is often called "quick return flow" because it appears in the hydrograph during or soon after the storm.

Base flow occurs when there is a fairly steady flow from natural storage. The flow comes from lakes or swamps, or from an aquifer replenished by infiltrated rainfall or surface runoff, or from "bank storage," which is supplied by infiltration into channel banks as the stream water level rises and which drains back into the stream as the water level falls. This type seldom appears soon enough after a storm to have any influence on the rates of the hydrograph for that storm, but base flow from a previous storm will increase the rates. Base flow must be taken into account in the design of the principal spillway of a floodwater-retarding structure.

The SCS method estimates direct runoff, which is a combination of channel runoff, surface runoff, and subsurface flow. Relative proportions of surface runoff and subsurface flow (ignoring channel runoff) can be determined with the runoff curve number (CN), which is an indicator of the probability of flow types (surface runoff is more likely with a larger CN).

A rainfall-runoff relation was developed by the SCS, suitable for use with daily precipitation totals, since such data are the most generally available in the United States (as measured at nonrecording rain gauges). Daily totals provide no information on the time distibution of the rainfall. The rainfall-runoff relation ignores rainfall intensity since time is not an explicit variable.

The relation between rainfall, runoff, and retention is expressed as

$$\frac{F}{S} = \frac{Q}{P-I_a}$$

where Q is the actual runoff, F is the actual retention, S is the potential maximum retention, P is the volume of precipitation (potential maximum runoff), and I_a is the initial abstraction (all terms are expressed in inches). The initial abstraction is that volume of precipitation at the beginning of a storm which does not appear as runoff.

The actual retention is

$$F = (P-I_a) - Q$$

The rainfall-runoff relation can then be written as

$$Q = \frac{(P-I_a)^2}{(P-I_a)+S}$$

The initial abstraction includes interception, infiltration, and surface storage. The SCS derived an empirical relationship for the estimation of the initial abstraction:

$$I_a = 0.2S$$

The relationship was developed based upon rainfall and runoff data from experimental small watersheds. The relationship for I_a is then substituted into the rainfall-runoff relation to obtain

$$Q = \frac{(P-0.2S)^2}{P+0.8S}$$

The potential maximum retention S is effected by the antecedent soil moisture condition (AMC), which is determined by the total rainfall in a 5-day period preceding a storm event. Three levels of AMC are defined: AMC-I is the lower limit of moisture (upper limit of S), AMC-II is the average condition, and AMC-III is the upper limit of moisture (lower limit of S).

The runoff curve number CN (also known as the hydrologic soil-cover complex number) has been empirically related to S:

$$CN = \frac{1000}{S+10}$$

or,

$$S = \frac{1000}{CN} - 10$$

With this transformation, the rainfall-runoff relation is reduced to a function of CN.

The curve number index represents the combined hydrologic effect of soil type, land use, and antecedent soil moisture. A compilation of curve number values is shown in Table 4-37. Curve number values are presented for a variety of land use categories, with a breakdown into the four hydrologic soil groups. In certain categories, curve numbers are developed for alternate hydrologic conditions (poor, fair, and good) which reflect the level of land management. Curve number values in Table 4-37 are associated with AMC-II. Equivalent curve numbers for other antecedent moisture conditions are displayed in Table 4-38.

The SCS has determined that AMC-II is not the average throughout the State of Texas. For the Highland Lakes area, the average condition is represented as

$$\overline{AMC} = AMC\text{-}I + 0.40\,(AMC\text{-}II - AMC\text{-}I)$$

Curve numbers obtained from Table 4-37 are adjusted using this correction factor. According to the SCS methodology, if the adjustment results in a curve number less than 60, a curve number of 60 will be selected as the minimally applicable value. If the unadjusted curve number is less than 60, the number will be used without adjustment (SCS, 1982b).

Curve numbers for the watersheds adjacent to the Highland Lakes were determined based upon the information presented in Table 4-37. For each watershed the composite curve number is a function of the soil hydrologic classification and the land use. The conventional method for determination of curve numbers involves integration of land use and soils data by overlaying land use and soil maps. Each land use-soil type complex is identified and the area obtained by a planimeter. A composite curve number for the watershed is estimated by weighting the curve numbers for each land use-soil type complex by their respective areas.

An alternative method for computing the composite curve number has been developed by Rawls et al. (1981). The method replaces the overlaying of soils and land use information with separate analyses of the two types of data. The percentage of each soil group is determined independently from the land use distribution. A weighted mean curve number is determined as follows:

$$CN_w = \sum_{i=1}^{n} \overset{4}{\underset{j=1}{S}} L_i S_j CN_{ij}$$

where CN_w is the weighted curve number for a watershed, n is the number of different land use categories, L_i is the fraction of land use category i, S_j is the fraction of soil group j (i.e., A, B, C, or D), and CN_{ij} is the curve number associated with a particular combination of L_i and S_j.

For the present analysis, it was desirable to obtain a weighted curve number for each land use category within a watershed, in order to ultimately examine runoff volumes from each category. The method of Rawls et al. (1981) was modified to weight the curve number for each land use category according to the fraction of hydrologic soil types in the subbasin:

$$CN_i = \sum_{j=1}^{4} S_j CN_{ij}$$

where CN_i is the curve number for each land use category i and the other terms are as previously defined. This approach implicitly assumes that each land use type is distributed over the existing soil types in accordance to their relative proportions. With this technique, a flow can be computed for each land use category.

For computation of curve numbers in the watersheds adjacent to the Highland Lakes, assumptions for land use categories and conditions were as follows:

Rangeland - fair condition
Forest - fair condition
Cropland - small grain, straight row - poor condition
Urban - ¼ acre average lot size

Curve numbers were adjusted as described earlier in this section. A computer program was developed to calculate the curve numbers for each land use category within each subbasin, using the compilation of land use areas and fractions of hydrologic soil types. The program code (Program HILOADS) is presented in Appendix B. Curve numbers calculated for the study watersheds under 1983 conditions are shown in Table 4-39.

To compute annual runoff depth, the frequency distribution of rainfall was discretized into a series of average precipitation depths, or representative storms. The number of events represented by each representative storm depth was obtained, such that a total depth for the storms associated with each average storm event could be calculated. Summation of the total depth for the series of average storms then approximates the average annual rainfall depth.

For the present analysis, average precipitation events of 3.8 cm (1.5 in), 6.4 cm (2.5 in), and 10.2 cm (4.0 in) were employed. With these three storm depths, the annual frequency distribution of rainfall is adequately represented, since the smaller storms which occur more frequently do not result in significant runoff. The SCS rainfall-runoff relationship was used to estimate direct runoff for each average precipitation event depth. Runoff was estimated for each land use category within the individual subbasins. Using the storm event runoff depths, average annual runoff volumes were calculated for each land use category based upon the average number of events associated with each storm depth. These calculations were carried out with the computer code HILOADS referenced previously. Calculated annual runoff volumes from the various 1983 land use categories in the series of watersheds adjacent to the Highland Lakes are presented in Table 4-40 to illustrate the magnitude of the runoff calculations.

The performance of the SCS rainfall-runoff relationship on area watersheds was investigated. Runoff estimates derived from the SCS technique were compared to observed runoff characteristics from USGS streamflow monitoring data on Shoal Creek, Bull Creek, and Sandy Creek. The evaluation is described in Appendix C. It was concluded that the SCS method adequately represents area runoff for the needs of the present analysis.

4.2.2.7 Determination of Loads

Phosphorus loads from the adjacent watersheds of the Highland Lakes were calculated using runoff volumes determined as described in the preceding section and stormwater constituent concentrations described previously. Annual loads were obtained by multiplication of flow and phosphorus concentration. To compute the loads, the following average runoff phosphorus concentrations were assumed for land use categories:

Rangeland	0.03 mg/l
Forest	0.02 mg/l
Cropland	0.10 mg/l
Urban	0.15 mg/l
Austin Urban	0.60 mg/l

The preceding assumptions for representative runoff phosphorus concentrations were based upon the water quality data compiled and reviewed in the present evaluation. The assumed rangeland runoff phosphorus concentration of 0.03 mg/l was based upon the rural watershed data reported previously. Since runoff from forest land would be expected to have lower phosphorus levels than rangeland runoff, a value of 0.02 mg/l was assumed. Cropland is often characterized by high phosphorus loadings. A phosphorus concentration of 0.10 mg/l was assumed for cropland runoff. Based upon the mixed urban-rural data described previously, runoff from urban areas was assumed to have an average phosphorus level of 0.15 mg/l. For the more intensively urbanized Austin area, an average value of 0.60 mg/l phosphorus was assumed. Phosphorus loads were calculated using the computer program HILOADS described in the preceding section. Loads are displayed in Table 4-41 for the land use categories in each subbasin, under 1983 conditions. Actual loads in any particular year will vary in accordance with the land use and storm conditions encountered.

4.2.2.7.1 Historical Loadings

Adjacent area loads were computed with the methodology described above. For each year, specific land use and precipitation characteristics were obtained. Land use information was estimated from the data described previously (TDWR, 1977). In particular, the urban and rangeland categories were adjusted in conformance with county population trends. Changes in urban areas were offset by corresponding increases or decreases in rangeland areas, since rangeland is the predominant land use throughout the study area.

Historical precipitation records for the Llano and Austin observation stations were examined for annual storm event characteristics. As with the previous analysis of the long-term frequency distribution, the number of events within the precipitation categories of 2.5-5.1 cm, 5.1-7.6 cm, and 7.6-12.7 cm was determined. The land use and precipitation characteristics were then input into the SCS computational methodology to obtain estimates of annual runoff for each of the years in the designated period of record. Phosphorus concentrations for each land use category were held constant, as described previously. Loading computations were executed with the computer code HILOADS. Results are displayed in Table 4-42. Annual phosphorus loads from adjacent areas demonstrate substantial fluctuation over the historic period of record. With the assumptions employed in the present analysis, the fluctuations in load correspond principally to the fluctuations in the precipitation cycle, since land use changes are relatively minor.

4.2.2.7.2 Projected Loadings

Phosphorus loadings from adjacent watersheds were also projected for future developmental conditions. With the recreational amenities offered by the areas near the Highland Lakes, future population growth is assured. Much of the growth will occur within the residential communities and lakeside developments which are presently in existence in the watershed, such as the Lakeway and Horseshoe Bay areas. New residential developments will also be constructed.

The county population estimates, presented in Table 4-6, provided the basis for loading projections from adjacent watersheds. The area of urban land within each subbasin was increased for each future year in accordance with the projected population increment for the appropriate county (defined as the county which comprised the majority of the subbasin). Projected areas of urban land use for 1990 and 2000 conditions are shown in Table 4-43. Projected increases in urban land area were assumed to be offset by corresponding decreases in rangeland area, which is the predominant land use throughout the watershed. The changes in urban land use area affected the calculation of runoff volumes and thus the calculation of loads. Projected phosphorus loadings from adjacent watersheds are displayed in Table 4-44 up to the year 2000. Projected loads were based upon runoff derived from repetition of precipitation characteristics from the period 1967-1983.

4.2.3 Atmospheric Loadings

The atmosphere represents a potential source of phosphorus loadings directly to the reservoirs. Little information is available concerning phosphorus deposition from the air. Atmospheric inputs have usually been assumed to be minimal under normal conditions, but sufficient data exist to establish the need to consider atmospheric contributions when evaluating eutrophication problems (Vollenweider, 1968). Atmospheric loads need only be computed for direct loadings to the reservoir surface areas, since atmospheric loads to the watershed surface are accounted for in runoff loads.

Site-specific atmospheric loading data are not available for the Highland Lakes study area. For the present analysis, the phosphorus concentration in rainfall was assumed to be 0.02 mg/l, which is within the typical range reported in Table 4-9 (Loehr, 1974). Alternatively, atmospheric loads have been reported as areal loading rates (see Table 4-8).

4.2.3.1 Historical Loadings

An average annual atmospheric phosphorus load to the Highland Lakes was computed based upon the annual rainfall totals at the Austin and Llano weather stations of 82.27 cm and 68.17 cm, respectively. At a concentration of 0.02 mg/l, the areal loading rate was computed as 0.016 $g/m^2/yr$ for the Austin climatological regime and 0.014 $g/m^2/yr$ for Llano conditions. Calculation of annual phosphorus loads of atmospheric origin requires definition of reservoir surface areas. For the constant-level reservoirs, Inks Lake, Lake LBJ, Lake Marble Falls, Lake Austin, and Town Lake, the spillway elevations were used to determine the normal surface area. To determine the "normal" surface areas of Lake Buchanan and Lake Travis, an historical average elevation was calculated using mean monthly average contents for the respective complete periods of record up to the year 1975 (TDWR, 1980). The average content data were converted to surface areas using the reservoir area and capacity curves (TDWR, 1971). Historical atmospheric phosphorus loadings, shown in Table 4-45, were estimated based upon the recorded annual precipitation totals at the Llano and Austin weather stations.

4.2.3.2 Projected Loadings

Projected atmospheric loadings are displayed in Table 4-46. Estimated loads were based upon repetition of historical precipitation totals for the period 1967-1983 to project up to the year 2000.

4.2.4 Barton Springs Loadings

Inflow from Barton Springs represents a source of phosphorus loadings directly to Town Lake. Discharge rates and water quality of Barton Springs are monitored by the USGS (Sta. 08155500). Daily discharge data, collected since March 1978, were used to estimate recent annual flows. To complete the historical record, estimates of mean annual discharge were obtained from the USGS, as shown in Table 4-47 (Slade, 1984). The mean phosphorus concentration for springs inflow was calculated as 0.018 mg/l, based upon a total of 25 measurements during the period February 1979-September 1981.

4.2.4.1 Historical Loadings

Average annual phosphorus loads to Town Lake from Barton Springs were calculated from the mean annual discharge and historical mean total phosphorus concentration. Loadings for the period 1952-1983 are displayed in Table 4-47.

4.2.4.2 Projected Loadings

Projected phosphorus loadings from Barton Springs were based upon the assumption that historical mean annual discharge rates (and thus loadings) would be repeated. Historical discharge estimates for the period 1967-1983 were employed to simulate post-1983 conditions. Projected loadings are equivalent to the historical loadings for 1967-1983 displayed in Table 4-47.

4.3 SUMMARY OF LOADS

The preceding sections described the development of phosphorus loading estimates for point and nonpoint sources in the watershed of the Highland Lakes. Historical point and nonpoint source phosphorus loadings to each reservoir are summarized in Table 4-48 for the period 1952-1983. Projected loadings for future conditions are summarized in Table 4-49. Urban and total nonpoint source phosphorus loads from adjacent watersheds and major tributary nonpoint source loads are identified along with point source estimates. Loadings to Town Lake from Barton Springs are tabulated as tributary loads in the summary tables for convenience. As indicated in the summary table, nonpoint sources account for the predominant portion of phosphorus loads to the Highland Lakes. The nonpoint source loads generally exceed the point source loads in the various watersheds by one or two orders of magnitude. Nonpoint loads from major tributaries are estimated to generally exceed nonpoint contributions from adjacent watersheds by one or two orders of magnitude. The nonpoint source loading estimates from adjacent areas are based upon runoff volumes calculated with annual storm characteristics. Major tributary historical loads are based upon recorded discharge values. Projected tributary loads were derived under the assumption that historical flow regimes would be repeated. In subsequent analyses, the phosphorus loadings associated with releases from upstream reservoirs will be determined. These loadings may represent a substantial portion of the incoming mass load.

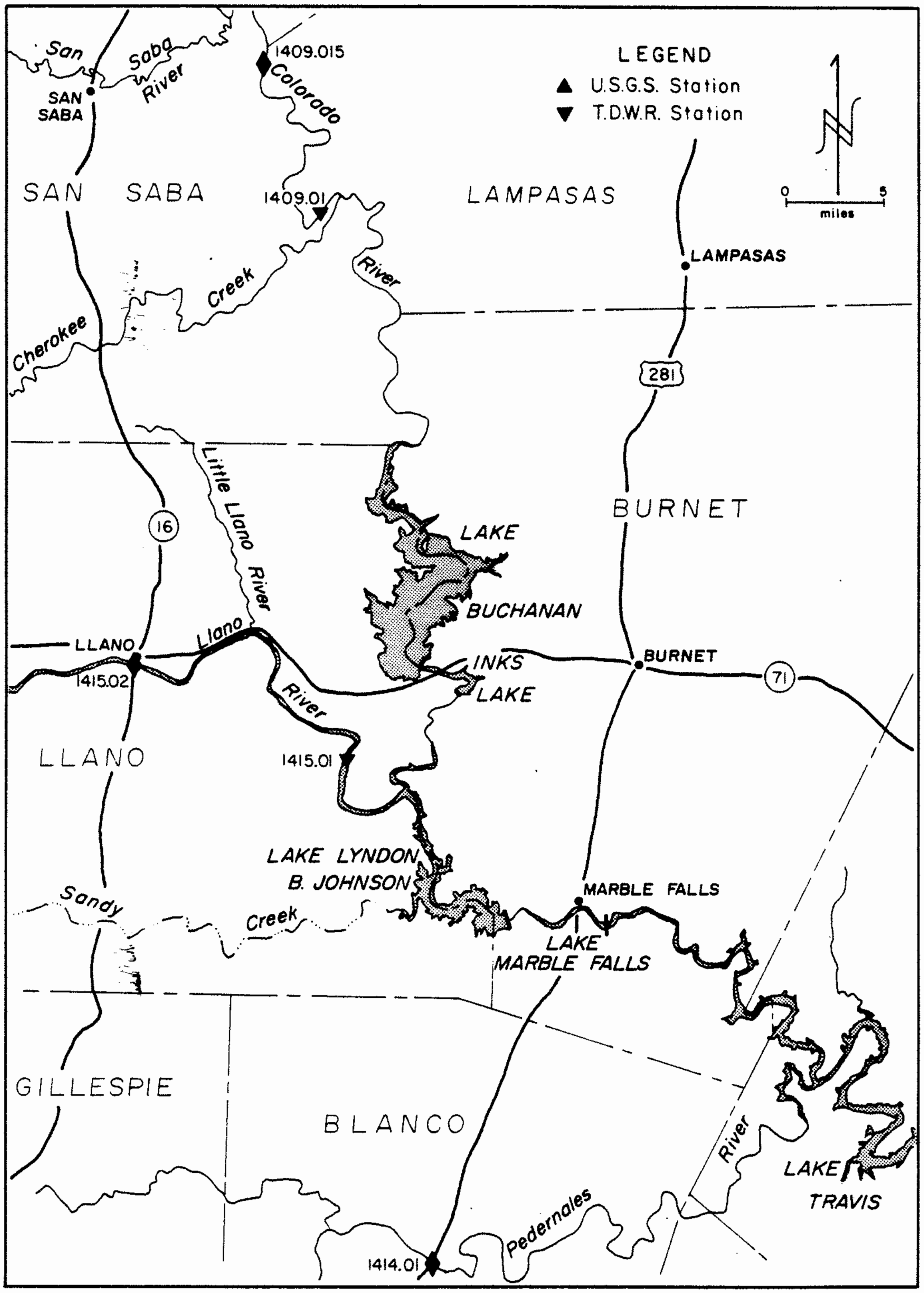


Fig. 4-1
MONITORING STATIONS ON MAJOR TRIBUTARIES

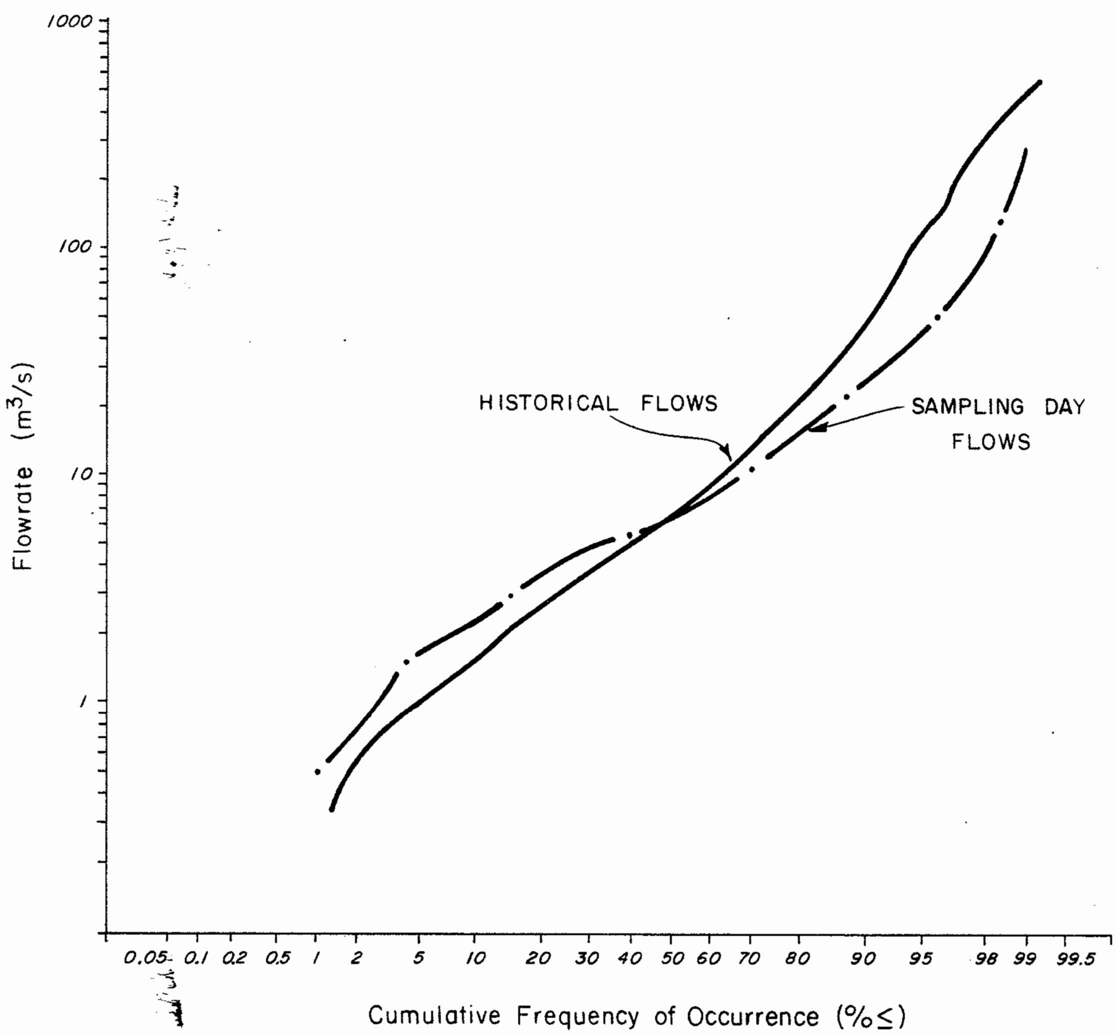


Fig. 4-2 CUMULATIVE FLOW FREQUENCY DISTRIBUTION
COLORADO RIVER AT USGS ST. NO. 08147000

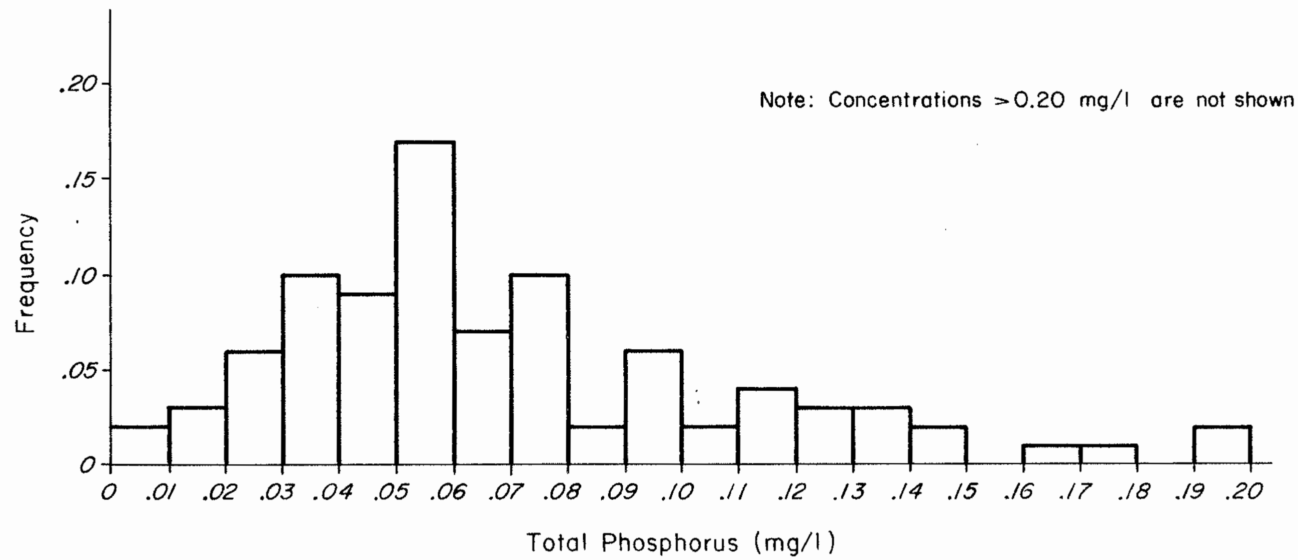


Fig. 4-3 HISTOGRAM FOR TOTAL PHOSPHORUS, COLORADO RIVER ABOVE LAKE BUCHANAN

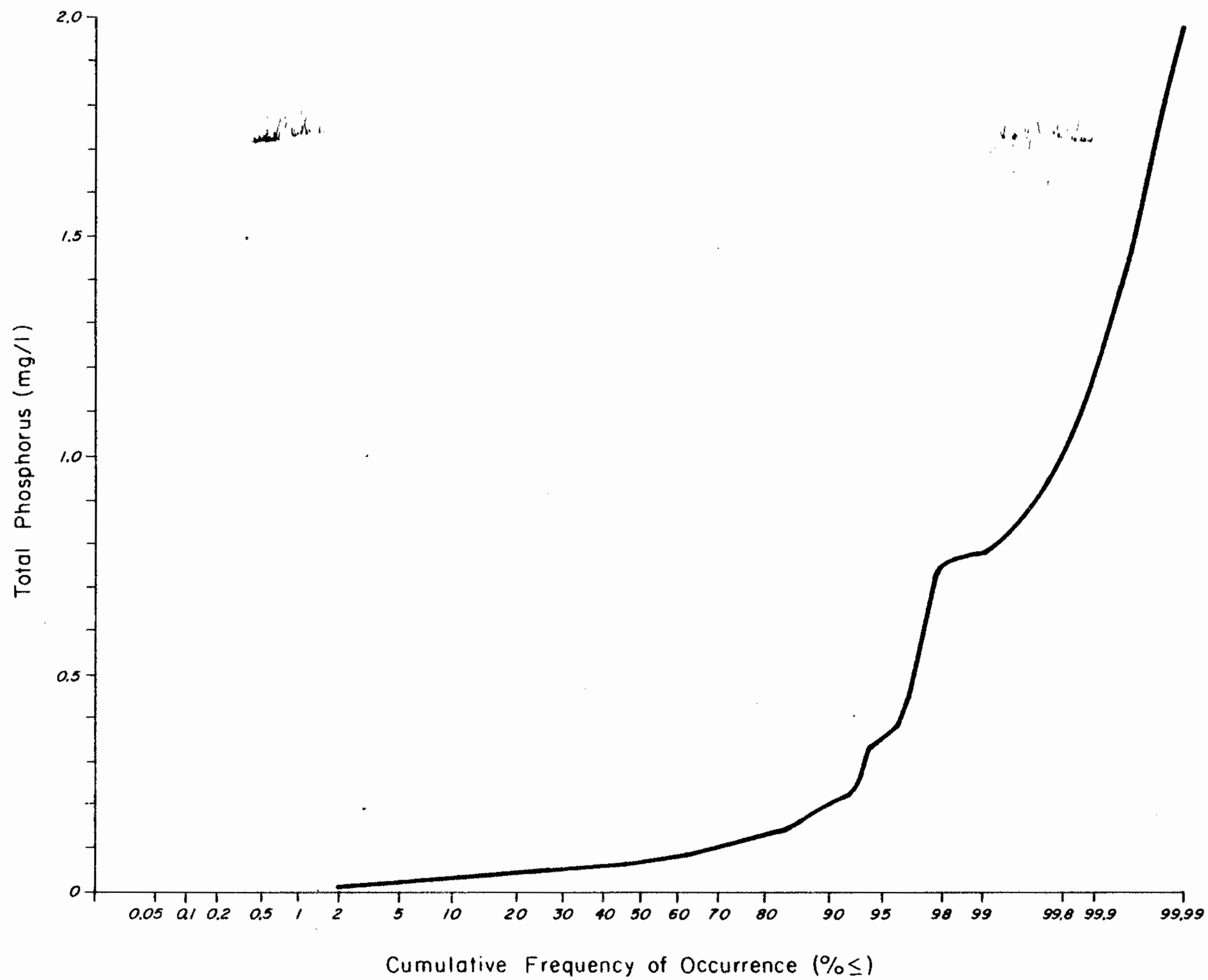


Fig. 4-4 CUMULATIVE FREQUENCY DISTRIBUTION FOR TOTAL PHOSPHORUS, COLORADO RIVER ABOVE LAKE BUCHANAN

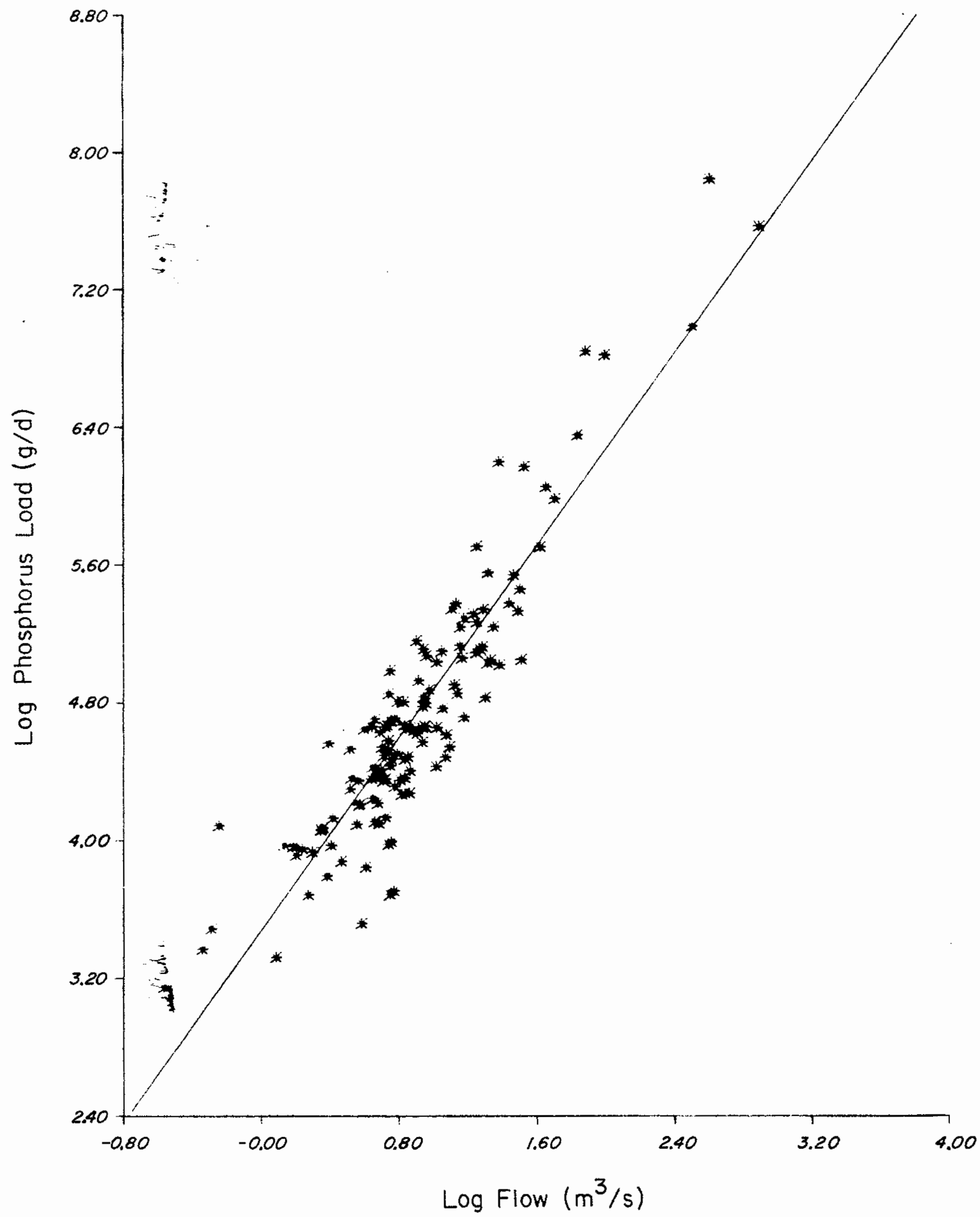


Fig. 4-5 RELATIONSHIP BETWEEN FLOW AND PHOSPHORUS LOAD
COLORADO RIVER ABOVE LAKE BUCHANAN

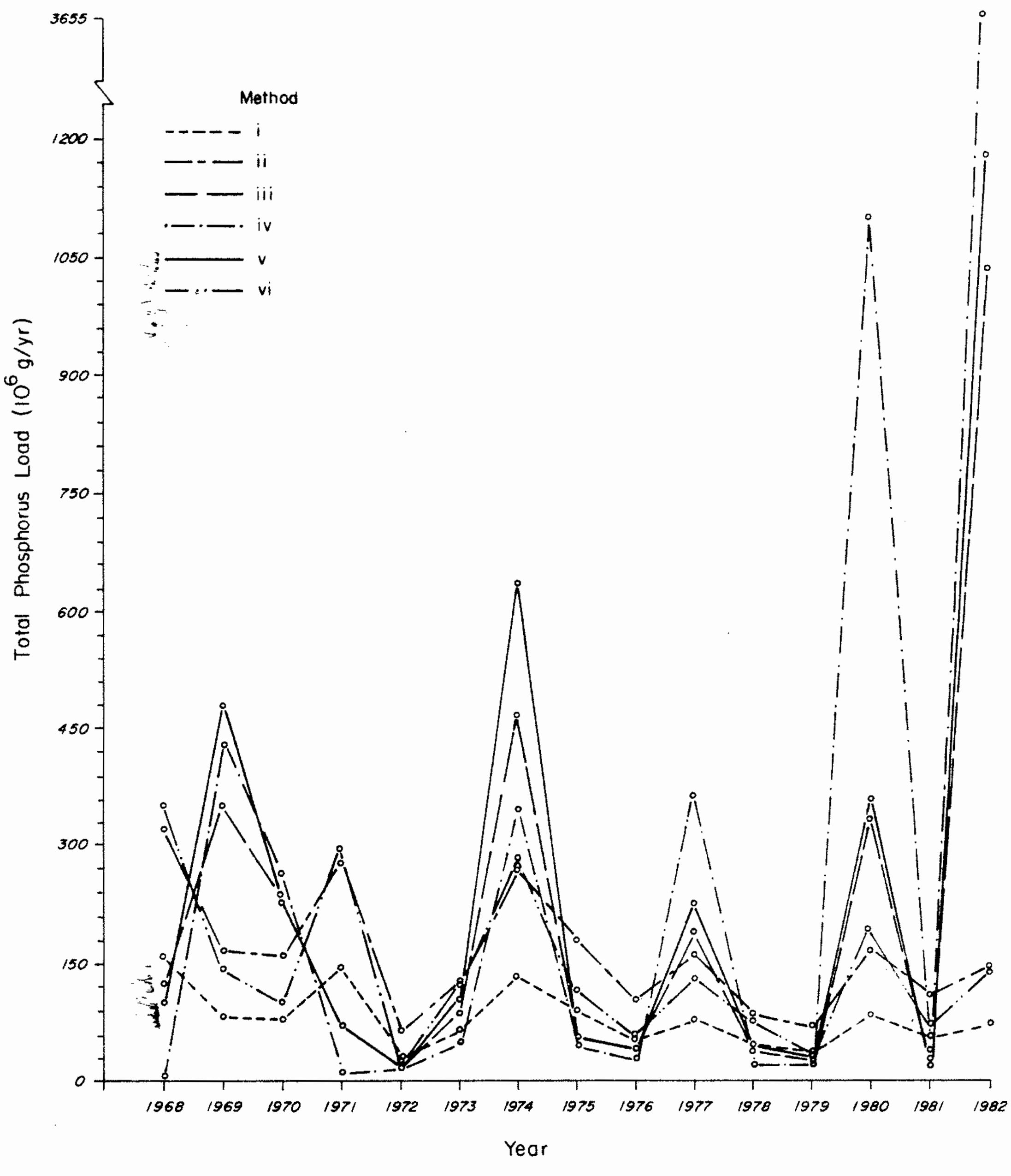


Fig. 4-6 COMPARISON OF ANNUAL PHOSPHORUS LOADS WITH ALTERNATIVE METHODOLOGIES COLORADO RIVER ABOVE LAKE BUCHANAN

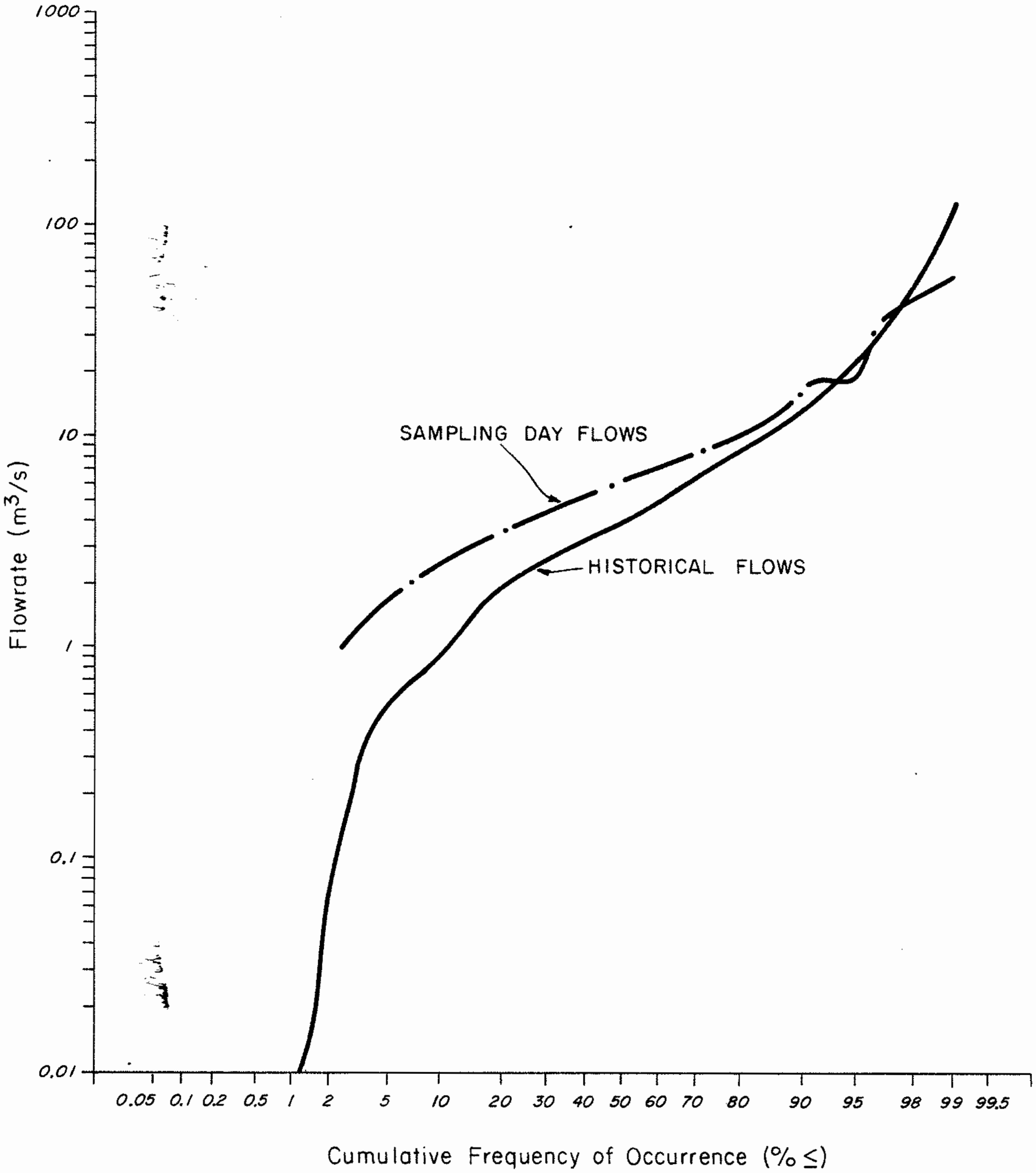


Fig. 4-7 CUMULATIVE FLOW FREQUENCY DISTRIBUTION
LLANO RIVER AT USGS ST. NO. 08151500

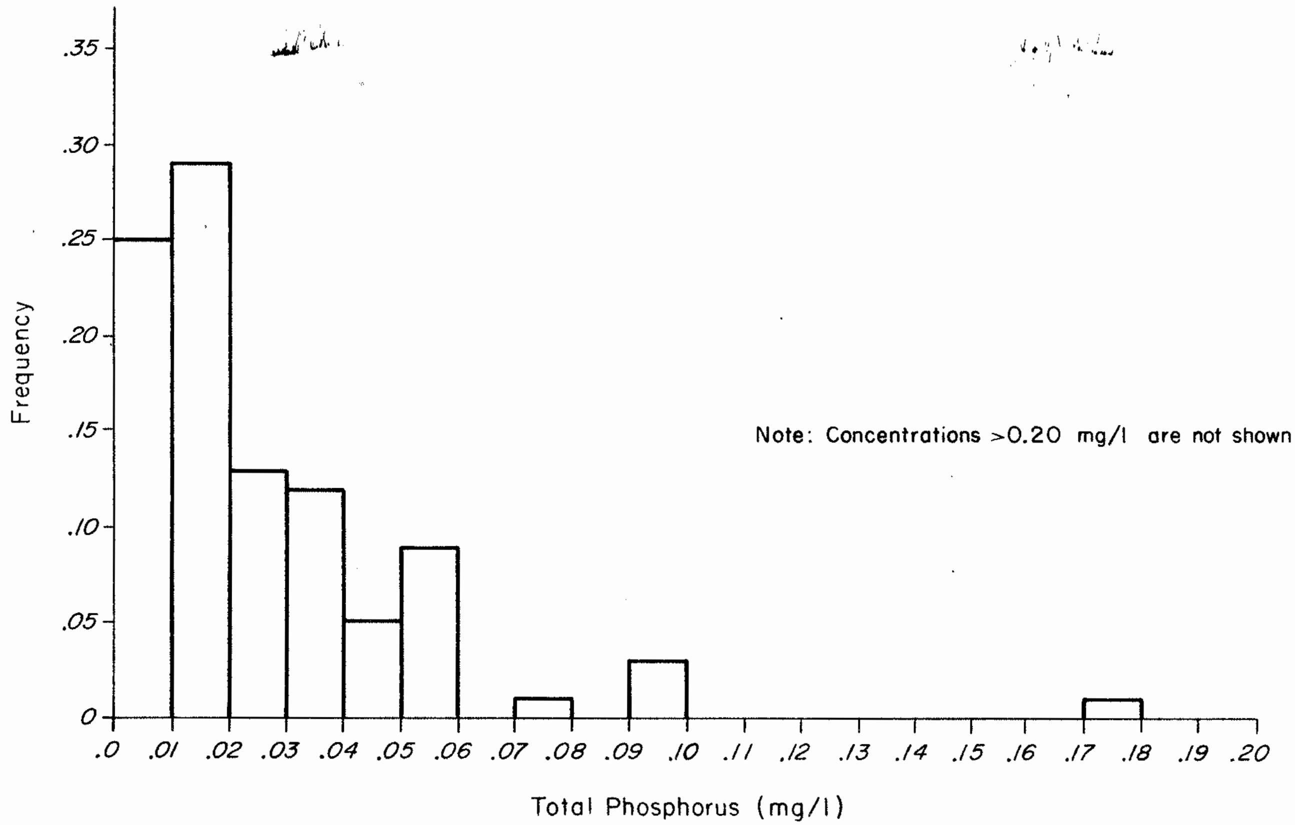


Fig. 4-8 HISTOGRAM FOR TOTAL PHOSPHORUS, LLANO RIVER ABOVE LAKE LBJ

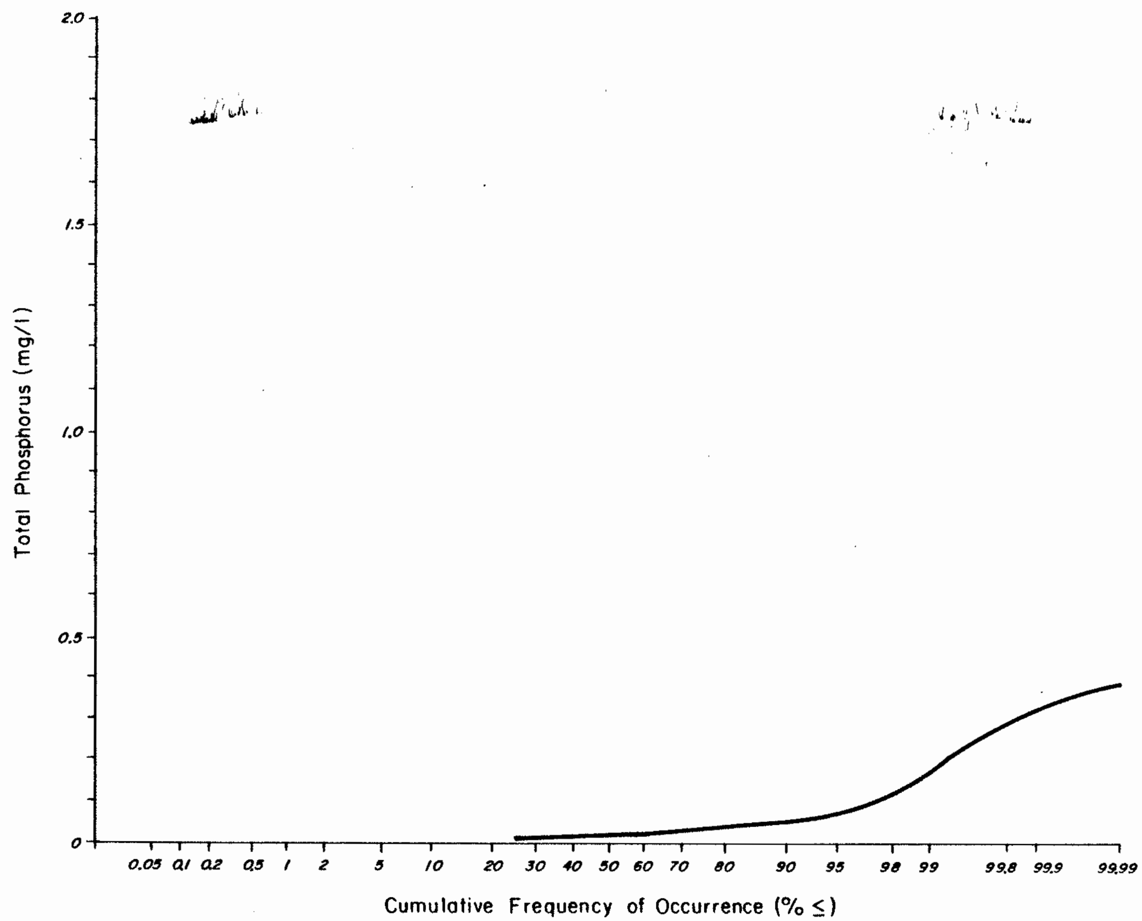


Fig. 4-9 CUMULATIVE FREQUENCY DISTRIBUTION FOR TOTAL PHOSPHORUS, LLANO RIVER ABOVE LAKE LBJ

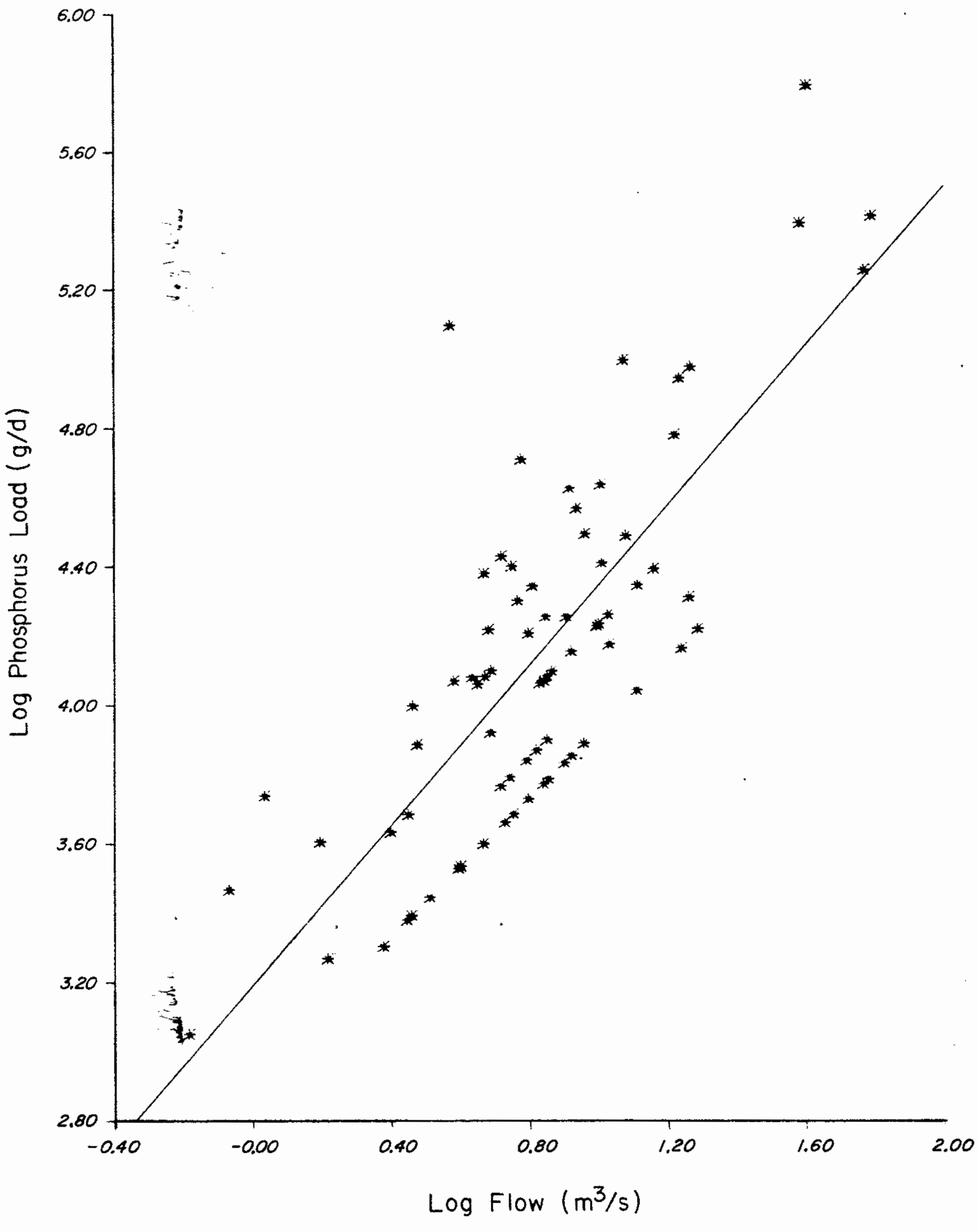


Fig. 4-10 RELATIONSHIP BETWEEN FLOW AND PHOSPHORUS LOAD LLANO RIVER ABOVE LAKE LBJ

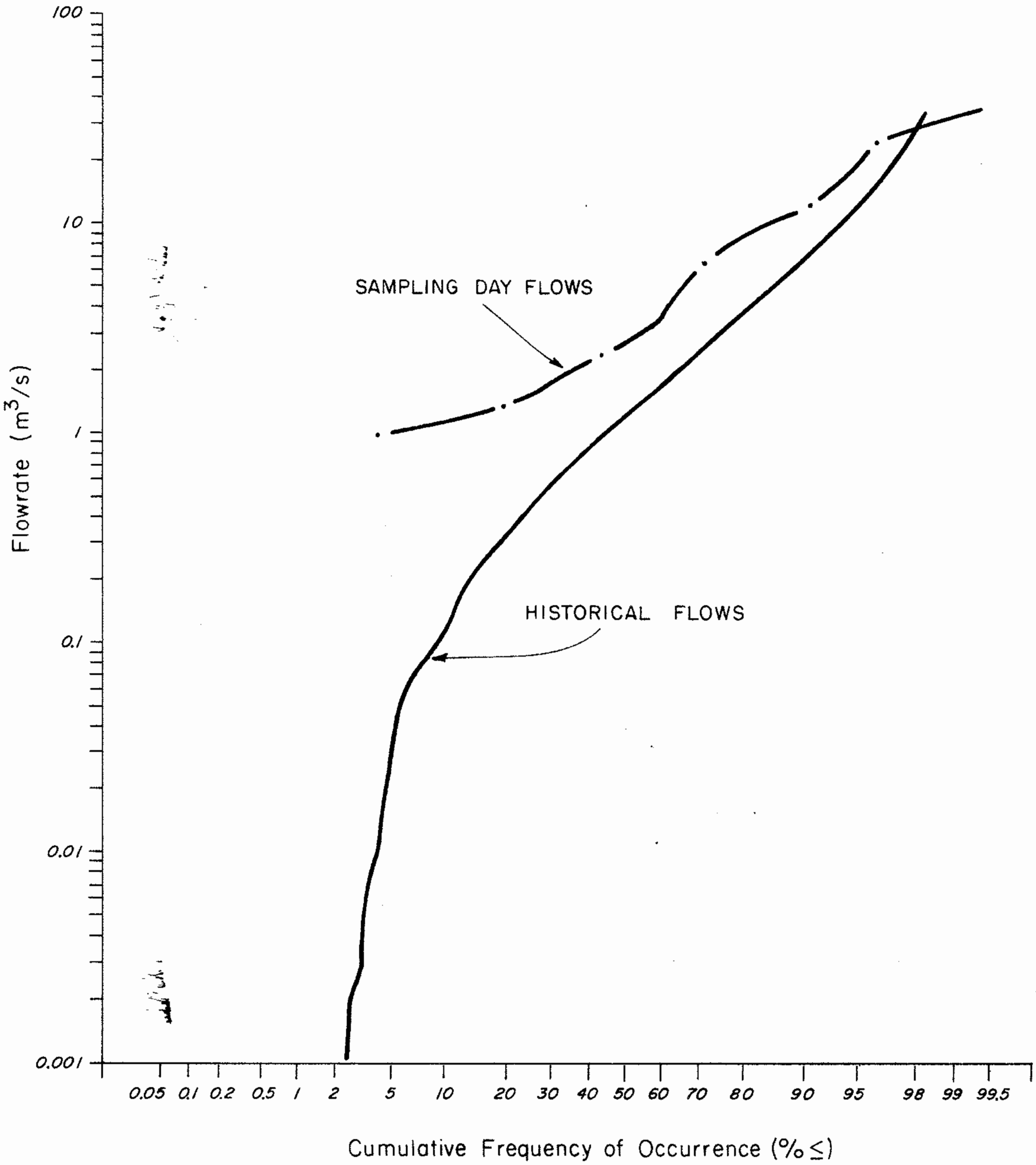


Fig. 4-11 CUMULATIVE FLOW FREQUENCY DISTRIBUTION
PEDERNALES RIVER AT USGS ST. NO. 08153500

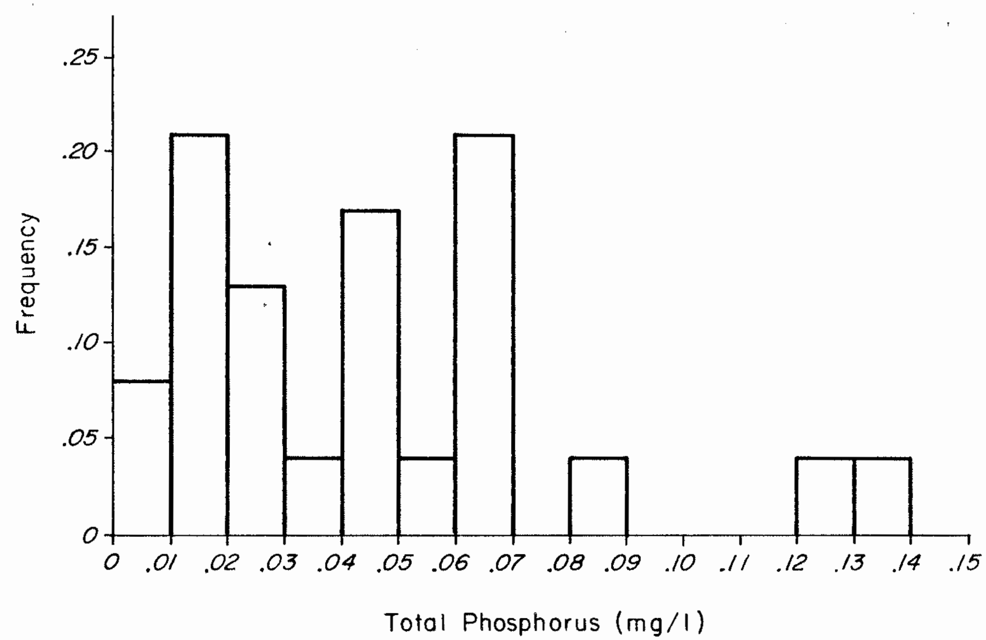


Fig. 4-12 HISTOGRAM FOR TOTAL PHOSPHORUS, PEDERNALES RIVER ABOVE LAKE TRAVIS

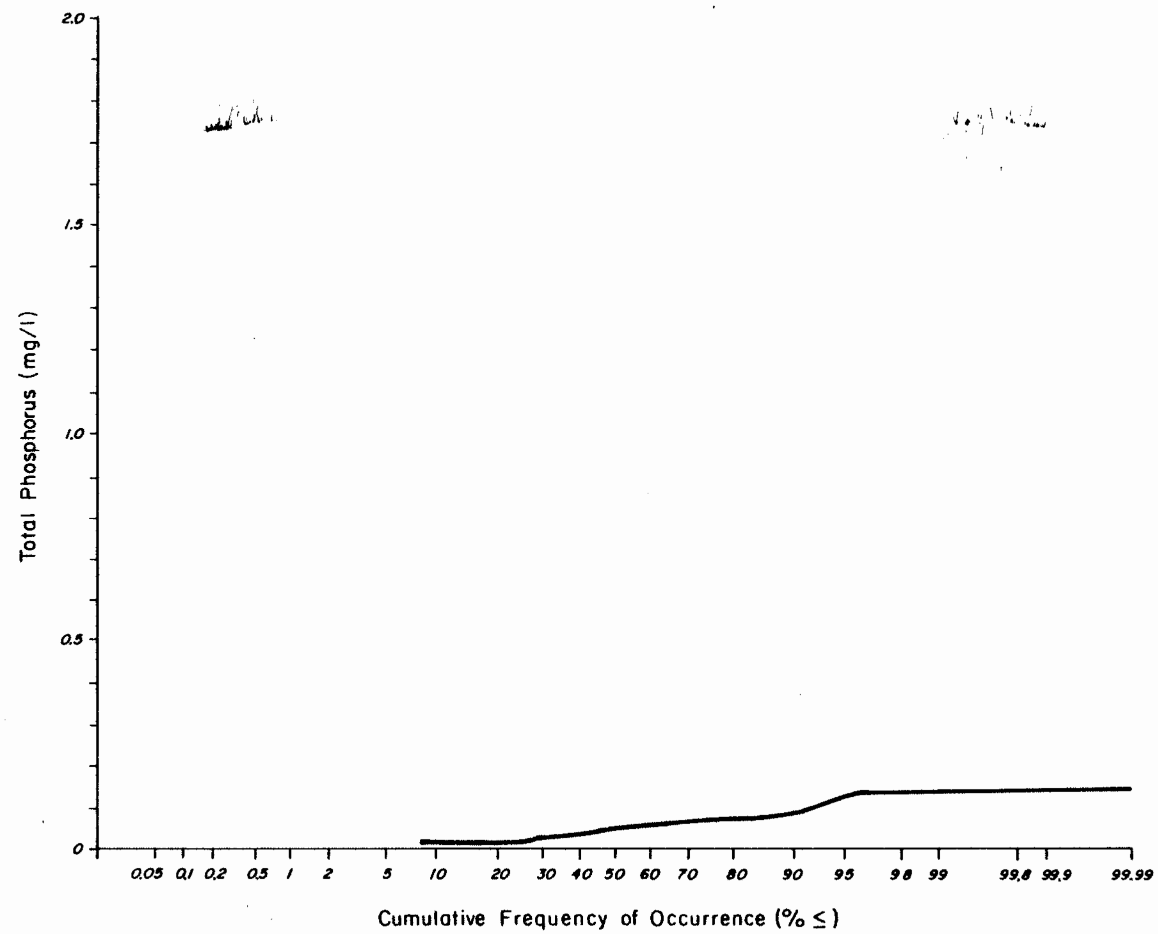


Fig. 4-13 CUMULATIVE FREQUENCY DISTRIBUTION FOR TOTAL PHOSPHORUS, PEDERNALES RIVER ABOVE LAKE TRAVIS

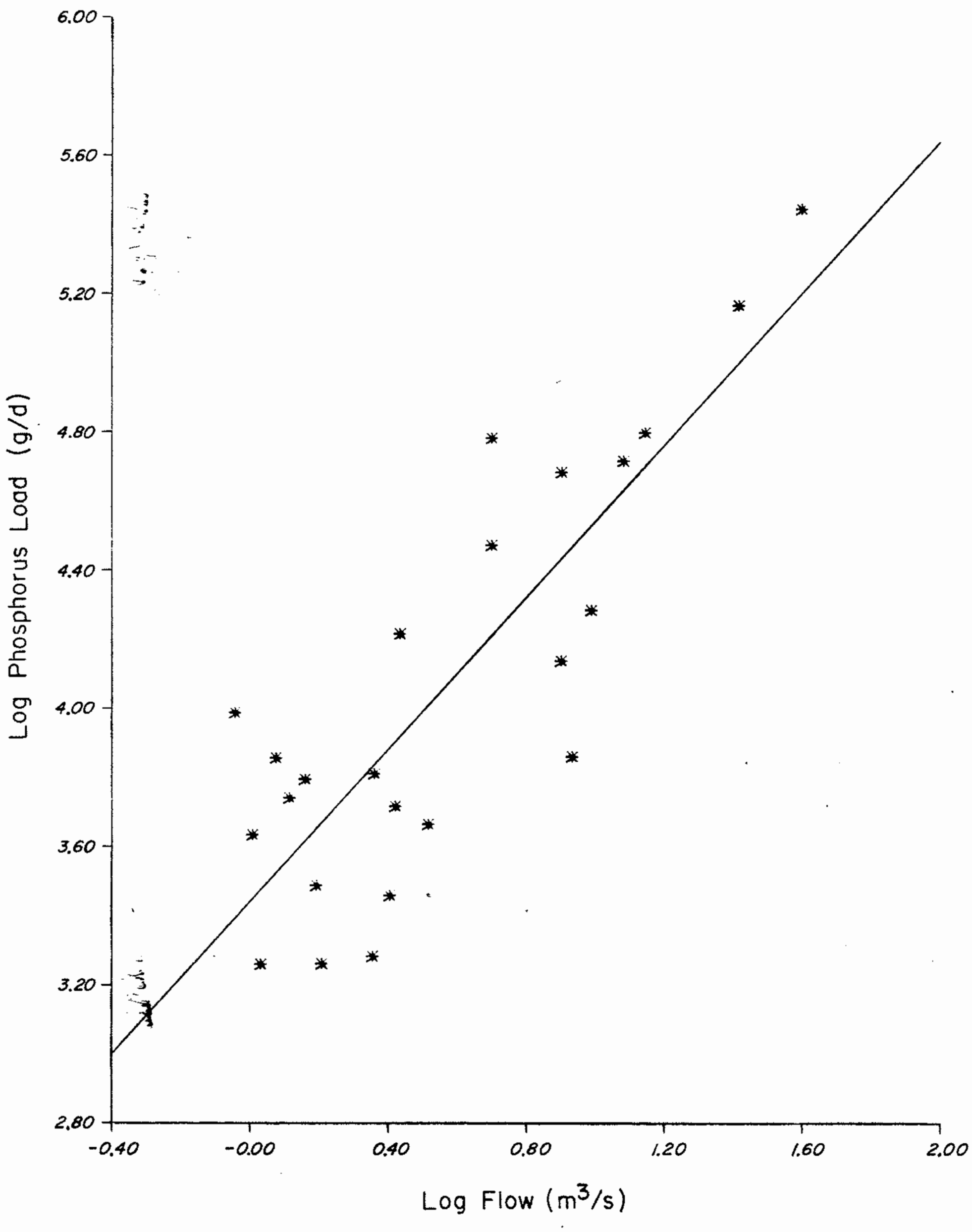


Fig. 4-14 RELATIONSHIP BETWEEN FLOW AND PHOSPHORUS LOAD PEDERNALES RIVER ABOVE LAKE TRAVIS

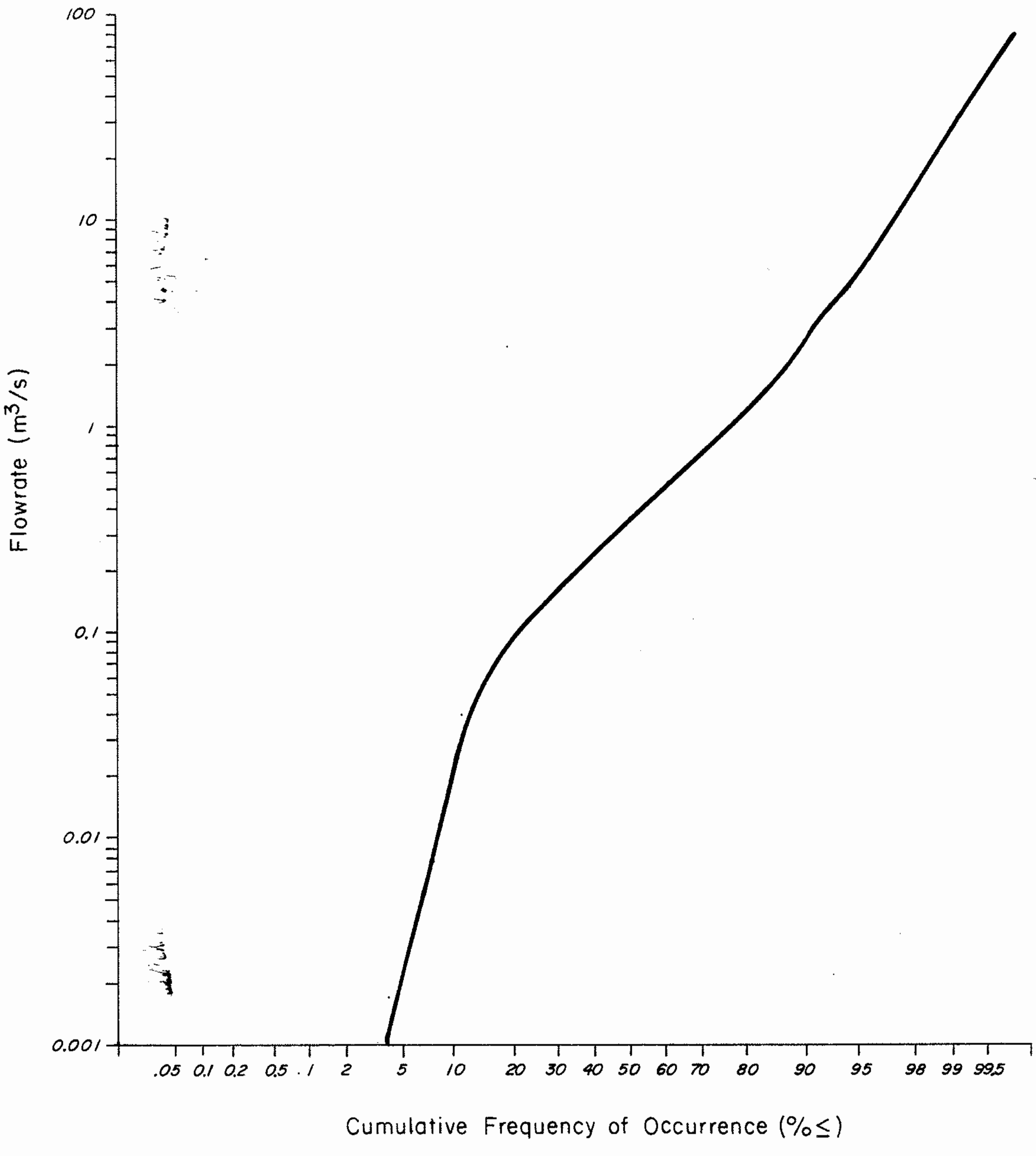


Fig. 4-15 CUMULATIVE FLOW FREQUENCY DISTRIBUTION
SANDY CREEK AT USGS ST. NO. 08152000

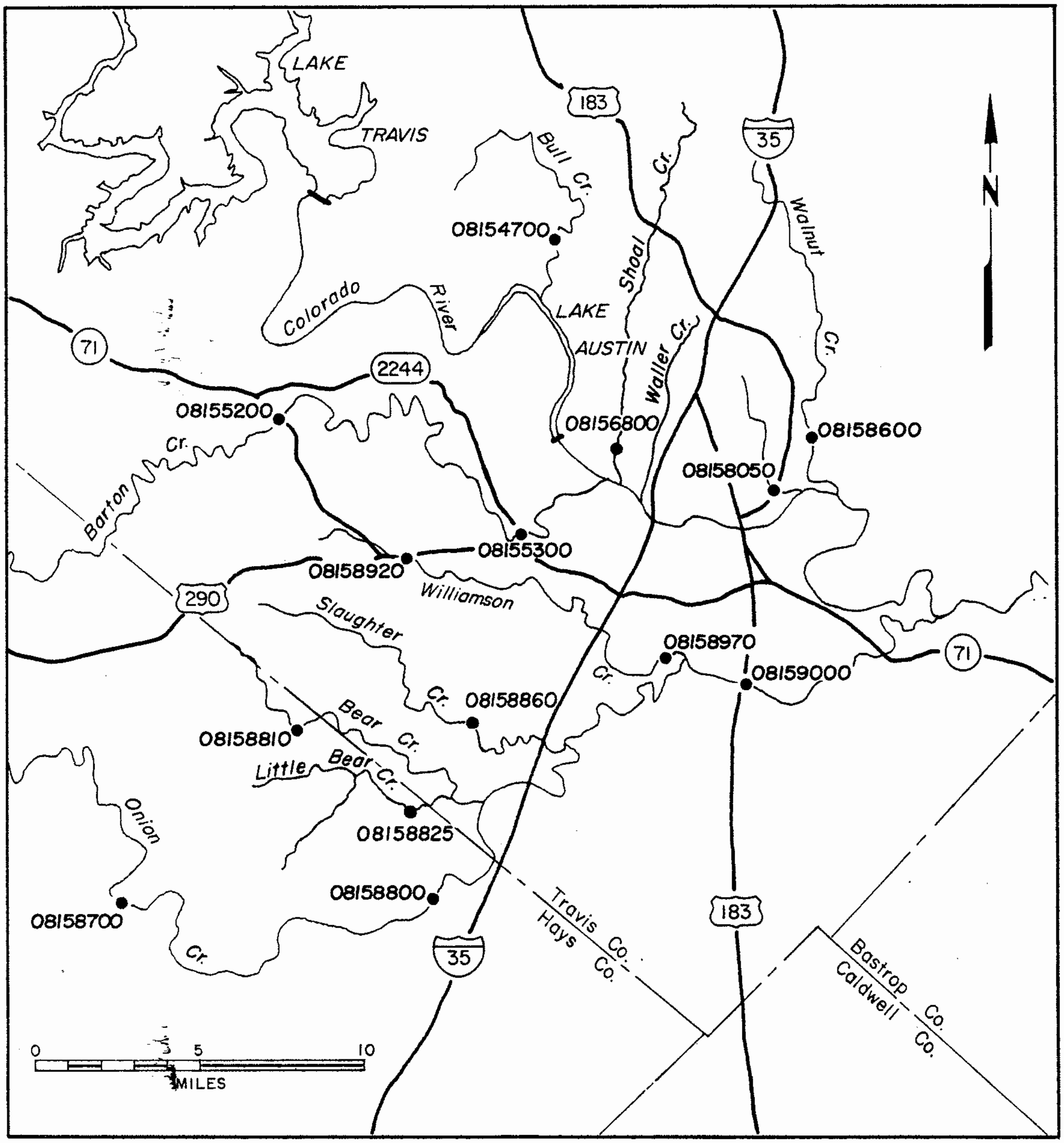


Fig. 4-16

MONITORING STATIONS ON MINOR TRIBUTARIES IN THE AUSTIN AREA

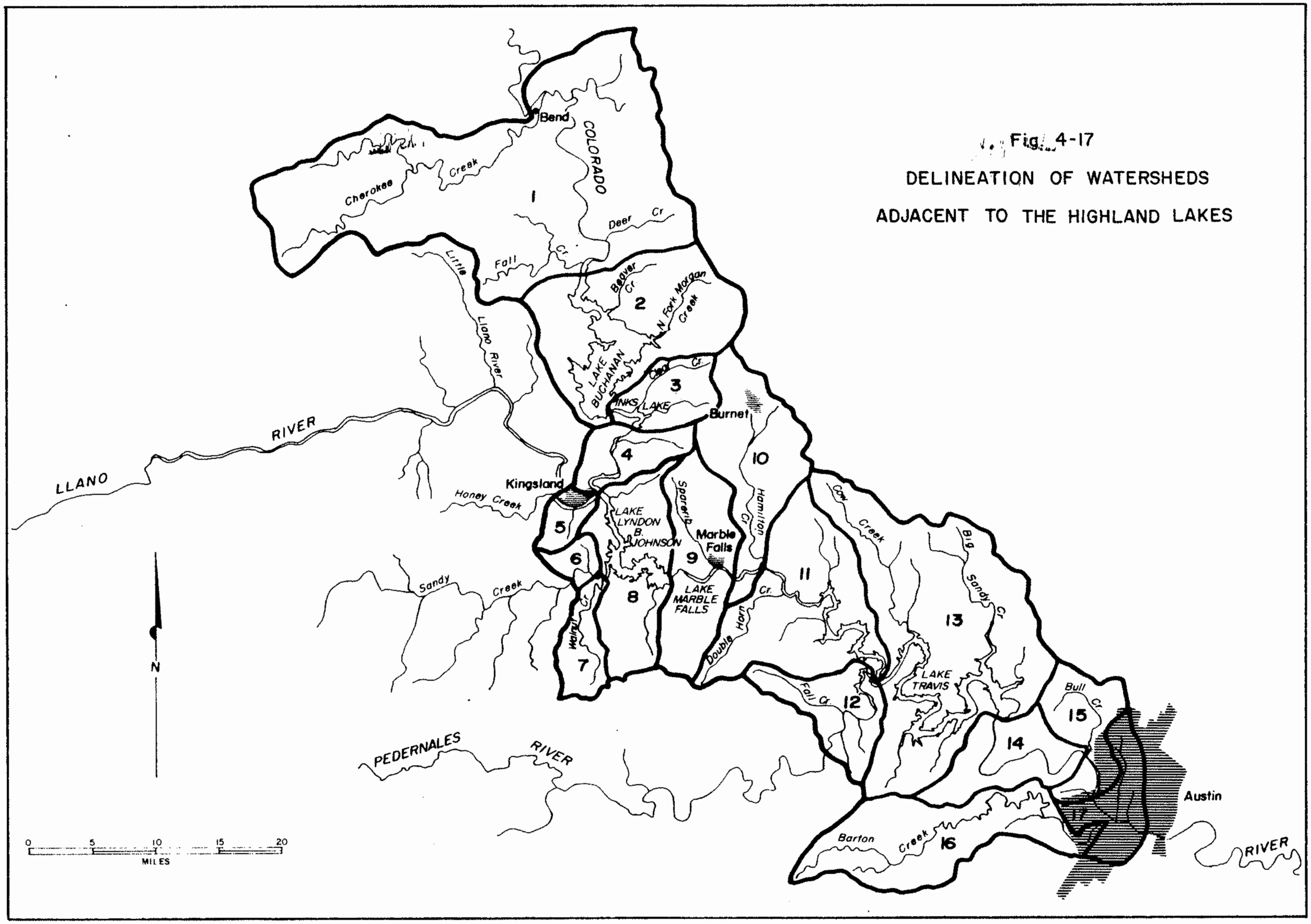


Fig. 4-17
DELINEATION OF WATERSHEDS
ADJACENT TO THE HIGHLAND LAKES

TABLE 4-1

MUNICIPAL AND INDUSTRIAL PERMIT HOLDERS

HIGHLAND LAKES WATERSHED AND MAJOR TRIBUTARIES

Segment	Permit Holder	Permit No.	Comments
1403	Texas Tumbleweed Restaurant	02584	retention
	City of Austin, Albert B. Davis Water Treatment Plant	10543	mainstrem discharge, potable water treatment plant waste
	St. Stephens Episcopal School	11202	retention, irrigation
	Spicewood Development Corp., Balcones Village	11363	retention, irrigation
	River Development Corp., Wilding STP	11514	retention, irrigation
	Davenport Ranch MUD No. 1	12363	retention, irrigation
1404	Lone Star Industries, Inc. Burnet plant	00641	discharge of limestone wastewater
	Ribera Brothers Turkey Farm (Round Mountain)	01974	retention, irrigation
	Lakeway MUD No. 1 Lakeway Inn and Marina	10531	retention, irrigation
	City of Burnet	10793	discharge, irrigation
	Pedernales Country Club	10992	retention, irrigation
	Whitecliff Services, Inc. Lake Travis Townhouse	11151	retention, irrigation
	Travis County WCID Point Venture Plant 1	11232	retention, irrigation
	Lakeway MUD No. 1 World of Tennis	11281	discharge, irrigation
	Travis County WCID Point Venture Plant 2	11385	retention, irrigation
	Allan R. Klein Commander's Point	11456	mainstem discharge
	Lakeway MUD No. 1 Central Plant	11495	retention, irrigation
	Travis Vista Subdivision	11531	mainstem discharge
	PJAP, INc., MWJ, Inc. & Tridel Villa on Travis	11532	inactive, discharge
	The Chase Corporation Windemere Plant	11694	retention, irrigation

TABLE 4-1 (Cont'd)

Segment	Permit Holder	Permit No.	Comments
	Travis County MUD No. 1 Lago Vista Plant	11752	mainstem discharge, irrigation
	Lakeway Development Co. Hurst Creek Plant 1	12215	discharge
	Jerome K. Felps Swine Feedlot	20062	retention, irrigation
1405	Calcium Carbonate Co.	02411	discharge, limestone operation
	City of Marble Falls	10654	mainstem discharge
	Meadowlakes MUD	11439	retention, irrigation
	M. O. Scott Swine	20028	retention, irrigation
1406	Lower Colorado River Authority T. C. Ferguson SES	01369	mainstem discharge, cooling water, etc.
	Lake LBJ MUD Horseshoe Bay	11217	mainstem discharge, irrigation
	B&W Gathering Co., Inc. Commanche Point	11332	mainstem discharge
	Buckner Boys Ranch	11484	retention, irrigation
	Kingsland MUD	11549	mainstem discharge
	S. L. Morris Granite Shoals	11995	discharge, inactive
	Lake LBJ MUD Horseshoe Bay West	12045	retention, irrigation
	Edgar J. Moss Swine	20035	retention, irrigation
1407	Joseph Dixon Crucible Co. Burnet Plant	00350	discharge
	Texas Parks & Wildlife Department Inks Lake State Park	11566	retention, irrigation
1408	Elam Miles Sheep Feedlot	01625	retention, irrigation
	Les Anders Silver Creek Lodge	11394	mainstem discharge
	Texas 4-H Youth Devel. Foundation Youth Center Plant	11664	retention, irrigation
1409	City of Lometa Kirby Creek Plant	11982	discharge

TABLE 4-1 (Cont'd)

Segment	Permit Holder	Permit No.	Comments
	Cen-Tex Pork Producers Swine Operation (Voca)	20752	retention, irrigation
	Herbert E. Merz, Jr. Turkey Farm (Albert)	01591	retention, irrigation, evaporation
1414	Sunday House Foods, Inc. Turkey Feedlot (Fredericksburg)	01613	retention, irrigation, evaporation
	C. A. Staats Turkey Farm Turkey Feedlot (Fredericksburg)	01658	retention, irrigation, evaporation
	Nolan Ottmers Ottmers Dairy (Stonewall)	02228	retention, irrigation
	Sunday House Foods, Inc. Turkey Farm (Fredericksburg)	02615	retention, irrigation
	City of Fredericksburg Outfall 001	10171	discharge
	City of Fredericksburg Outfall 002, County Fairground	10171	retention, irrigation
	City of Johnson City	10198	discharge
	Texas Parks & Wildlife Dept. LBJ State Park	11480	retention, irrigation
1415	The PAKS Corporation Cedar Oil Mill (Junction)	01391	discharge
	Cedar Fiber Company, Inc. (Junction)	01412	retention, evaporation
	Mason Feed Store, Inc. Swine Feedlot (Mason)	01449	retention, irrigation
	Mason Feeders, Inc. Cattle Feedlot (Mason)	01454	retention, irrigation
	Bar D Hog Company Swine Feedlot (Mason)	01467	retention, irrigation
	Tom Dean Enterprises, Inc. Poultry Raising (Cherokee)	02283	retention, irrigation
	City of Junction	10199	mainstem discharge, irrigation
	City of Llano	10209	mainstem discharge
	City of Mason	10670	discharge, irrigation
	Raymond F. Winkel Swine (Llano)	20268	retention, irrigation
	James D. Epperson Swine Operation (Valley Spring)	20284	retention

TABLE 4-1 (Concluded)

Segment	Permit Holder	Permit No.	Comments
	Sam Rabb Swine (Llano)	20290	retention, irrigation
	A. J. Hopson & A. D. Hopson Swine (Llano)	20294	retention, irrigation
	Three G Hog Company Swine Operation (Llano)	20295	retention, irrigation
	Simpson and Simpson Hog Farm Swine (Llano)	20297	retention
	Clarence Hasse Swine (Castell)	20313	retention, irrigation
	Sterling Jordan Feedlot Swine (Mason)	20469	retention, irrigation
	Miller Hog Farm Swine (Llano)	20512	retention, irrigation
	Llano Pork Products Ltd. Swine	20582	retention, irrigation
	Loyd Mitchell, Jr. Swine Operation (Rocksprings)	20631	retention
	Weber and Weber Hog Company Swine Operation (Llano)	20642	retention, irrigation

Note: Comments derived from TDWR permit listing:
mainstem discharge = discharge into mainstem of segment
discharge = discharge to tributary of segment
retention = no discharge, or discharge only during flood conditions
irrigation = effluent used for irrigation
evaporation = effluent disposed of by evaporation

Source: TDWR, 1983a.

TABLE 4-2

PERMIT LEVELS
SIGNIFICANT POINT SOURCE DISCHARGES
HIGHLAND LAKES AREA

Segment	Discharger	Permit No.	Flow* (MGD)	BOD_5 (mg/l)	TSS (mg/l)	TP (mg/l)	Comments
1403	—	—	—	—	—	—	no significant direct discharges
1404	Lakeway MUD No. 1 World of Tennis	11281	0.105	10	15	—	discharge to Hurst Cr., irrigation
	Commander's Point	11456	0.035	5	5	1	permit to discharge to Lake Travis, presently nonexistent
	Travis Vista	11531	0.006	10	15	—	discharge to Lake Travis
	Travis Co. MUD No. 1 Lago Vista Plant	11752	0.300	5	5	1	discharge to Lake Travis, irrigation
	Lakeway Development Co. Hurst Creek Plant 1	12215	0.250	5	10	2	discharge to Lake Travis
	Villa on Travis	11532	0.042	5	5	2	permit to discharge to Lake Travis, projected for start-up
1405	City of Marble Falls	10654	0.47	10	15	—	discharge to Lake Marble Falls
1406	Lake LBJ MUD Horseshoe Bay	11217	1.500	5	5	1	discharge to Lake LBJ, irrigation
	Commanche Point	11332	0.0300	5	5	1	discharge to Lake LBJ
	Kingsland MUD	11549	0.750	5	5	2	discharge to Lake LBJ
1407	—	—	—	—	—	—	no significant direct discharges
1408	Silver Creek Lodge	11394	0.004	10	15	—	discharge to Lake Buchanan

* 30-day daily average limitations

Source: TDWR, 1983b.

TABLE 4-3

SELF-REPORTING DATA

DIRECT POINT SOURCE DISCHARGES TO THE HIGHLAND LAKES

	Silver Creek Lodge Permit No. 11394			Kingsland MUD Permit No. 11549			Commanche Point Permit No. 11332			Travis Vista Permit No. 11531			City of Marble Falls Permit No. 10654		
	Flow	Phosphorus		Flow	Phosphorus		Flow	Phosphorus		Flow	Phosphorus		Flow	Phosphorus	
	(m^3/d)	(mg/l)	(kg/d)	(m^3/d)	(mg/l)	(kg/d)	(m^3/d)	(mg/l)	(kg/d)	(m^3/d)	(mg/l)	(kg/d)	(m^3/d)	(mg/l)	(kg/d)
Permit Levels:	15.0	NR	NR	2839	2.0	5.9	113.6	1.0	0.11	22.7	NR	NR	1779.0	NR	NR
Yearly Avg															
1983*	3.4	NR	NR	ND	3.14+	ND	15.5	0.54	0.008	45.4	NR	NR	1341.4	NR	NR
1982	4.2	NR	NR	ND	ND	ND	11.0	0.66	0.007	36.0	NR	NR	1414.1	NR	NR
1981	6.8	NR	NR	ND	ND	ND	7.6	0.55	0.004	49.2	NR	NR	1268.7	NR	NR
1980	3.8	NR	NR				6.8	0.67	0.004	39.7	12.5	0.52	1131.7	NR	NR
1979	4.2	NR	NR				2.0	0.52	0.001	2.1	0.55	0.02	1044.7	NR	NR
1978	9.8	NR	NR				ND	ND	ND				692.7	NR	NR
1977	6.1	NR	NR				ND	ND	ND				662.4	NR	NR
1976	5.3	NR	NR				ND	ND	ND				541.3	NR	NR
1975	9.5	NR	NR				ND	ND	ND				302.8	NR	NR
1974							ND	ND	ND				302.8	NR	NR
1973													302.8	NR	NR
1972													302.8	NR	NR
1971													261.2	NR	NR
1963-1970													ND**	ND	ND

* Partial record

NR = not required, therefore no data

ND = no data, even though required

** No data for period 1963-1970

\+ Data for 2 months only

Note: Records for each plant extend to approximate start-up date

Source: TDWR, 1983b.

TABLE 4-4

SELF-REPORTING DATA
POINT SOURCES ADJACENT TO THE HIGHLAND LAKES
WITH EFFLUENT DISPOSAL VIA IRRIGATION

	Hurst Creek Plant 1 Permit No. 12215			Lago Vista Permit No. 11752			Lakeway MUD No. 1 Permit No. 11281			Lake LBJ MUD Permit No. 11217		
	Flow	Phosphorus		Flow	Phosphorus		Flow	Phosphorus		Flow	Phosphorus	
	(m^3/d)	(mg/l)	(kg/d)	(m^3/d)	(mg/l)	(kg/d)	(m^3/d)	(mg/l)	(kg/d)	(m^3/d)	(mg/l)	(kg/d)
Permit Levels:	946.2	2.0	1.91	1135.5	1.0	1.13	397.4	NR	NR	5677.5	1.0	5.9
Yearly Avg.												
1983	3.8	1.24	0.005	ND	ND	ND	ND	NR	NR	635.5	0.55	0.36
1982	11.4	1.57	0.014	ND	ND	ND	138.9^{+}	NR	NR	936.0	0.65	0.64
1981				ND	ND	ND	249.8^{++}	NR	NR	884.2	0.89	0.79
1980				ND	ND	ND				897.0	0.89	0.80
1979				ND	ND	ND				ND	ND	ND
1978				ND	ND	ND				310.4	0.57	0.72
1977				ND	ND	ND				ND	ND	ND
1976				ND	ND	ND				ND	ND	ND
1975				ND	ND	ND				ND	ND	ND
1974										ND	ND	ND

\+ Data for one month only.

++ Data for four months only.

Note: Records for each plant extend to approximate start-up date.

Source: TDWR, 1983b.

TABLE 4-5

HISTORIC POINT SOURCE PHOSPHORUS LOADS

HIGHLAND LAKES

Year	Silver Creek Lodge Permit No. 11394		Kingsland MUD Permit No. 11549		Commanche Point Permit No. 11332		Travis Vista Permit No. 11531		Marble Falls Permit No. 10654	
	Flow (m^3/d)	Phosphorus (kg/yr)	Flow (m^3/d)	Phosphorus (kg/yr)	Flow (m^3/d)	Phosphorus (kg/yr)	Flow (m^3/d)	Phosphorus (kg/yr)	Flow (m^3/d)	Phosphorus (kg/yr)
1983	3.4	12.4	757	1464.4	15.5	3.1	45.4	165.7	1341.4	4896.1
1982	4.2	15.3	757	2763.0	11.0	2.6	36.0	131.4	1414.1	5161.5
1981	6.8	24.8	757	1381.5	7.6	1.5	49.2	179.6	1268.7	4630.8
1980	3.8	13.9			6.8	1.7	39.7	144.9	1131.7	4130.7
1979	4.2	15.3			2.0	0.4	2.1	7.7	1044.7	3813.2
1978	9.8	35.8			2.0	0.4			692.7	2528.4
1977	6.1	22.3			2.0	0.4			662.4	2417.8
1976	5.3	19.3			2.0	0.4			541.3	1975.7
1975	9.5	34.7			2.0	0.4			302.8	1105.2
1974					2.0	0.4			302.8	1105.2
1973									302.8	1105.2
1972									302.8	1105.2
1971									261.2	953.4
1970-1963									261.2	953.4

Source: TDWR, 1983b.

TABLE 4-6

POPULATION AND PROJECTIONS
FOR COUNTIES SURROUNDING THE HIGHLAND LAKES

County	1970	1975	1980	1985	1990	1995	2000
Blanco	3,567	4,086	4,681	5,484	6,425	7,320	8,340
Burnet	11,420	14,258	17,803	21,310	25,509	29,263	33,571
Hays	27,642	33,497	40,594	49,787	61,064	74,489	90,867
Llano	6,979	8,413	10,144	11,962	14,107	15,157	16,287
San Saba	5,540	5,862	6,204	6,182	6,162	6,376	6,598
Travis	295,516	352,122	419,573	494,878	583,699	666,430	760,888
Williamson	37,305	53,428	76,521	100,068	130,862	162,430	201,614

Source: Texas Department of Water Resources, 1983c.

TABLE 4-7

PROJECTED POINT SOURCE PHOSPHORUS LOADINGS

Year	Phosphorus Loads (kg/yr) Commander's Point 11456	Travis Vista 11531	Villa on Travis 11532	Marble Falls 10654	Commanche Point 11332	Kingsland MUD 11549	Silver Creek 11394
1984	0.0	165.7	0.0	5068.6	2.5	1511.7	12.8
1985	0.0	165.7	0.0	5241.2	2.6	1559.2	13.3
1986	0.0	165.7	0.0	5447.7	2.7	1615.0	13.8
1987	0.0	165.7	29.0	5654.3	2.8	1670.8	14.3
1988	12.1	165.7	58.0	5860.8	2.9	1726.7	14.9
1989	24.2	165.7	87.0	6067.4	3.0	1782.4	15.4
1990	48.3	165.7	116.0	6274.9	3.1	1838.3	15.9
1991	48.3	165.7	116.0	6459.6	3.2	1866.7	16.4
1992	48.3	165.7	116.0	6644.3	3.3	1895.1	16.9
1993	48.3	165.7	116.0	6829.0	3.4	1923.5	17.4
1994	48.3	165.7	116.0	7013.7	3.5	1951.9	17.9
1995	48.3	165.7	116.0	7198.3	3.6	1980.3	18.4
1996	48.3	165.7	116.0	7410.2	3.7	2008.7	18.9
1997	48.3	165.7	116.0	7622.2	3.8	2037.1	19.4
1998	48.3	165.7	116.0	7834.1	3.9	2065.5	19.9
1999	48.3	165.7	116.0	8046.1	4.0	2093.9	20.4
2000	48.3	165.7	116.0	8258.0	4.1	2122.3	20.9

TABLE 4-8

TYPICAL LOADING RATES REPORTED IN VARIOUS STUDIES

	Export Coefficients, $g/m^2/yr$		
	(1)*	(2)	(3)
	Total Phosphorus		
Urban	0.1	0.11	0.11-0.56
Rural/Agriculture	0.05	0.018	0.006-0.29
Forest	0.005-0.01	0.008	0.003-0.09
Atmosphere	0.025	0.044	0.005-0.006
	Total Nitrogen		
Urban	0.05(0.25)**	0.88	0.7-0.9
Rural/Agriculture	0.5(0.2)	0.85	0.01-1.3
Forest	0.3(0.1)	0.24	0.3-1.3
Atmosphere	2.4(1.0)	0.58	0.56-1.0

* (1) Rast and Lee, 1983.
(2) Shannon and Brezonik, 1972.
(3) Loehr, 1974.

**Values for western U.S. in parentheses.

TABLE 4-9

SUMMARY OF NONPOINT SOURCE CHARACTERISTICS*

Source	Concentration (mg/l)				
	COD	BOD	NO_3-N	Total N	Total P
Precipitation	9-16	12-13	0.14-1.1	1.2-1.3	0.02-0.04
Forested land	--	--	0.1-1.3	0.3-1.8	0.01-0.11
Rangeland	--	--	--	--	--
Agricultural cropland	80	7	0.4	9	0.02-1.7
Land receiving manure	--	--	--	--	--
Irrigation tile drainage, western U.S.					
Surface flow	--	--	0.4-1.5	0.6-2.2	0.2-0.4
Subsurface drainage	--	--	1.8-19	2.1-19	0.1-0.3
Cropland tile drainage	--	--	--	10-25	0.02-0.7
Urban land drainage	85-110	12-160	--	3	0.2-1.1
Seepage from stacked manure	25,900-31,500	10,300-13,800	--	1,800-2,350	190-280
Feedlot runoff	3,100-41,000	1,000-11,000	10-23	920-2,100	290-360

*Source: Loehr, 1974.

TABLE 4-10

MAJOR TRIBUTARY WATER QUALITY MONITORING STATIONS

HIGHLAND LAKES

TDWR St. No.	USGS St. No.	Description	River Mile	Drainage Area (km^2)	Drainage Area (mi^2)	Period of Record*
1409.01	--	Colorado River FM 580 at Bend, TX	458.4	51,686	19,956	TDWR: 1974-1982 USGS: NA
1409.015	08147000	Colorado River US 190 near San Saba, TX	474.3	51,331	19,819	TDWR: very limited USGS: 1968-1981
1414.01	08153500	Pedernales River Hwy 281 near Johnson City, TX	48.0	2,334	901	TDWR: 1968-1981 USGS: very limited
1414.02	08152900	Pedernales River Hwy 87 near Fredericksburg, TX	88.7	956	369	TDWR: 1973-1982 USGS: NA
1415.01	--	Llano River County Rd. near Kingsland, TX	6.5	11,399	4,401	TDWR: 1973-1982 USGS: NA
1415.02	08151500	Llano River Hwy 16 at Llano, TX	29.3	10,857	4,192	TDWR: 1972-1978 USGS: 1979-present

*Record with available phosphorus data.
NA denotes none available.

TABLE 4-11

PHOSPHORUS SUMMARY STATISTICS
MAJOR TRIBUTARY MONITORING STATIONS

TDWR St. No.	USGS St. No.	Description	TP Range (mg/l)	Average TP (mg/l)	Standard Deviation	Number of Data Points
1409.01	—	Colorado R. at Bend	0.01-1.98	0.155	0.376	26
1409.015	08147000	Colorado R. near San Saba	< 0.01-1.046	0.118	0.154	110
1414.01	08153500	Pedernales R. near Johnson City	0.0098-0.14	0.048	0.035	24
1414.02	08152900	Pedernales R. near Fredericksburg	0.02-0.70	0.137	0.116	34
1415.01	—	Llano R. near Kingsland	0.0098-0.06	0.026	0.018	29
1415.02	08151500	Llano R. at Llano	0.0-0.39	0.040	0.601	47

Data source: TNRIS, 1983a.

TABLE 4-12

REGRESSION EQUATIONS FOR
PHOSPHORUS CONCENTRATION AND FLOW
COLORADO RIVER ABOVE LAKE BUCHANAN

Equation	a Coefficient	b Coefficient	Standard Error	Correlation Coefficient
$y = a + bx$	0.09101	0.00149	0.177	0.57
$y = a + b \log x$	-0.08588	0.23554	0.179	0.55
$y = ae^{bx}$	0.07005	0.00466	0.274	0.45
$y = ax^{b}$	0.03461	0.39336	0.185	0.54

Note: Concentration in mg/l, flow in m^3/s.

TABLE 4-13

REGRESSION EQUATIONS FOR
PHOSPHORUS LOAD AND FLOW
COLORADO RIVER ABOVE LAKE BUCHANAN

Equation	a Coefficient	b Coefficient	Standard Error	Correlation Coefficient
$y = a + bx$	-445,834	67,975	4,127,278	0.80
$y = a + b \log x$	-5,056,586	6,866,487	5,958,623	0.50
$y = ae^{bx}$	38,836	0.0135	155,196,576	0.62
$y = ax^{b}$	2,989	1.3935	4,995,944	0.91

Note: Load in g/d, flow in m^3/s.

TABLE 4-14

COMPARISON OF PREDICTED LOADS WITH ALTERNATIVE REGRESSION EQUATIONS

COLORADO RIVER ABOVE LAKE BUCHANAN

Streamflow (m^3/s)	Linear Relationship (g/d)	Logarithmic Relationship (g/d)	Exponential Relationship (g/d)	Power Relationship (g/d)
1	-377,859	-5,056,586	39,365	2,989
10	233,919	1,809,901	44,460	73,977
100	6,351,700	8,676,388	150,153	1,830,603
1,000	67,529,506	15,542,875	28,986,898,000	45,299,582

TABLE 4-15

CALCULATED HISTORICAL PHOSPHORUS LOADS
WITH ALTERNATIVE METHODOLOGIES
COLORADO RIVER AT SAN SABA, 1968-1982

Year	Number of Phosphorus Measurements	Phosphorus Loads (10^6 g/yr)					
		Method (i)	Method (ii)	Method (iii)	Method (iv)	Method (v)	Method (vi)
1968	2	161.4	320.3	122.2	11.1	101.9	349.7
1969	6	83.5	165.9	349.7	431.7	481.8	143.6
1970	4	81.2	161.1	235.8	261.4	229.9	102.0
1971	6	140.0	277.8	70.1	11.6	69.0	293.9
1972	6	32.3	64.1	19.9	18.3	18.8	22.7
1973	7	64.5	127.9	83.9	44.0	108.3	120.5
1974	11	135.7	269.3	467.6	350.0	637.5	283.4
1975	10	92.1	182.8	56.2	41.9	57.8	118.3
1976	9	53.2	105.5	40.4	23.8	42.5	55.8
1977	11	81.5	161.6	189.8	365.4	225.1	131.1
1978	15	44.2	87.7	34.7	19.9	45.0	74.6
1979	14	35.1	69.6	25.9	19.3	30.8	33.7
1980	13	84.0	166.7	341.5	1,104.2	362.6	193.9
1981	15	56.0	111.1	38.5	24.8	42.6	70.7
1982	7	74.8	148.5	1,039.0	3,654.4	1,179.0	139.3

TABLE 4-16

STATISTICAL SUMMARY OF ALTERNATIVE PHOSPHORUS LOAD PREDICTION METHODS

Method	Number of Data Points	Mean Load (10^6 g/yr)	Standard Deviation (10^6 g/yr)	Confidence Range (10^6 g/yr)
i	15	81.3	38.4	60.2-102.4
ii	15	161.3	76.2	119.4-203.2
iii	15	207.7	269.5	59.4-356.0
iv	15	425.5	940.2	-91.9-942.8
v	15	242.2	318.6	66.9-417.5
vi	15	142.2	98.1	88.2-196.2

Note: Confidence range at 0.05 level of significance, t-test of means:

$$\frac{t(s)}{\sqrt{n}} = | \mu - \bar{x} |$$

TABLE 4-17

HISTORICAL MAJOR TRIBUTARY PHOSPHORUS LOADS

Year	Phosphorus Loads (10^6 g/yr)			
	Colorado River Above Lake Buchanan	Llano River Above Lake LBJ	Sandy Creek Above Lake LBJ	Pedernales River Above Lake Travis
1952	222.37	10.26	0.54	51.47
1953	160.08	14.44	0.76	2.31
1954	173.74	9.81	0.52	0.79
1955	595.24	7.56	0.40	3.67
1956	302.39	6.94	0.37	0.31
1957	1644.89	5.16	0.27	20.55
1958	115.49	5.61	0.30	18.53
1959	434.57	4.54	0.24	22.46
1960	86.31	18.11	0.96	10.83
1961	208.92	7.24	0.38	9.17
1962	72.80	3.06	0.16	3.02
1963	50.27	1.42	0.07	1.29
1964	164.98	19.12	1.01	2.35
1965	313.79	3.20	0.17	12.53
1966	82.72	0.94	0.05	2.44
1967	59.76	11.18	0.50	3.08
1968	353.15	0.75	2.99	15.07
1969	145.01	39.95	1.74	11.09
1970	103.04	16.72	1.77	12.69
1971	296.73	13.80	1.03	7.22
1972	22.88	13.57	1.05	6.19
1973	121.64	19.66	0.75	11.15
1974	286.19	2.50	2.17	20.85
1975	119.47	1.93	2.70	23.59
1976	56.31	14.61	1.37	9.36
1977	132.34	8.78	1.64	19.43
1978	75.29	6.84	0.78	17.89
1979	34.01	3.30	2.95	20.24
1980	195.84	21.18	0.72	2.57
1981	71.38	18.25	2.17	19.75
1982	140.67	12.46	0.14	2.68
1983	15.62	15.62	0.85	5.97

TABLE 4-18

PROJECTED MAJOR TRIBUTARY PHOSPHORUS LOADS

Year	Phosphorus Loads (10^6 g/yr)			
	Colorado River Above Lake Buchanan	Llano River Above Lake LBJ	Sandy Creek Above Lake LBJ	Pedernales River Above Lake Travis
1984	50.73	11.18	0.50	3.08
1985	144.64	0.75	2.99	15.07
1986	145.01	39.95	1.74	11.09
1987	103.04	16.72	1.77	12.69
1988	296.73	13.80	1.03	7.22
1989	22.88	13.57	1.05	6.19
1990	121.64	19.66	0.75	11.15
1991	286.19	2.50	2.17	20.85
1992	119.47	1.93	2.70	23.59
1993	56.31	14.61	1.37	9.36
1994	132.34	8.78	1.64	19.43
1995	75.29	6.84	0.78	17.89
1996	34.01	3.30	2.95	20.24
1997	195.84	21.18	0.72	2.57
1998	71.38	18.25	2.17	19.75
1999	140.67	12.46	0.14	2.68
2000	15.62	15.62	0.85	5.09

TABLE 4-19

REGRESSION EQUATIONS FOR
PHOSPHORUS CONCENTRATION AND FLOW
LLANO RIVER ABOVE LAKE LBJ

Equation	a Coefficient	b Coefficient	Standard Error	Correlation Coefficient
$y = a + bx$	0.02838	0.00071	0.049	0.16
$y = a + b \log x$	0.02483	0.01253	0.049	0.09
$y = ae^{bx}$	0.02016	0.01880	0.050	0.26
$y = ax^{b}$	0.01797	0.15536	0.050	0.17

Note: Concentration in mg/l, flow in m^3/s.

TABLE 4-20

REGRESSION EQUATIONS FOR
PHOSPHORUS LOAD AND FLOW
LLANO RIVER ABOVE LAKE LBJ

Equation	a Coefficient	b Coefficient	Standard Error	Correlation Coefficient
$y = a + bx$	-18,041	5,690	57,433	0.73
$y = a + b \log x$	-69,774	129,113	70,027	0.56
$y = ae^{bx}$	6,286	0.0819	121,904	0.72
$y = ax^{b}$	1,552	1.1555	66,191	0.78

Note: Load in g/d, flow in m^3/s.

TABLE 4-21

REGRESSION EQUATIONS FOR
PHOSPHORUS CONCENTRATION AND FLOW
PEDERNALES RIVER ABOVE LAKE TRAVIS

Equation	a Coefficient	b Coefficient	Standard Error	Correlation Coefficient
$y = a + bx$	0.04221	0.00081	0.035	0.21
$y = a + b \log x$	0.04321	0.00780	0.035	0.10
$y = ae^{bx}$	0.03120	0.02269	0.036	0.26
$y = ax^{b}$	0.03181	0.10229	0.037	0.14

Note: Concentration in mg/l, flow in m^3/s.

TABLE 4-22

REGRESSION EQUATIONS FOR
PHOSPHORUS LOAD AND FLOW
PEDERNALES RIVER ABOVE LAKE TRAVIS

Equation	a Coefficient	b Coefficient	Standard Error	Correlation Coefficient
$y = a + bx$	-10,743	6,526	18,256	0.96
$y = a + b \log x$	-23,346	99,848	41,986	0.75
$y = ae^{bx}$	4,993	0.1248	92,334	0.80
$y = ax^{b}$	2,748	1.1023	30,796	0.84

Note: Load in g/d, flow in m^3/s.

TABLE 4-23

MINOR TRIBUTARY WATER QUALITY MONITORING STATIONS

AUSTIN URBAN AREA

USGS St. No.	Description	Drainage Area (km^2)	Drainage Area (mi^2)	Period of Record	Watershed Type	Impervious Cover* (%)
08155200	Barton Cr. at Hwy 71	232.3	89.7	1978-1981	Rural	--
08155300	Barton Cr. at Loop 360	300	116	1979-1981	Rural/Urban	9
08158810	Bear Cr. below FM 1826	31.6	12.2	1978-1981	Rural	3
08158050	Boggy Cr. at Hwy 183	33.9	13.1	1975-1981	Urban	43
08154700	Bull Cr. at Loop 360	57.8	22.3	1978-1981	Rural/Urban	11
08158825	Little Bear Cr. at FM 1626	54.4	21.0	1978-1981	Rural	--
08158700	Onion Cr. near Driftwood	321	124	1974-1981	Rural	--
08158800	Onion Cr. at Buda	430	166	1978-1981	Rural/Urban	--
08159000	Onion Cr. at Hwy 183	831	321	1976-1981	Rural/Urban	--
08156800	Shoal Cr. at 12th St.	31.9	12.3	1975-1981	Urban	47
08158860	Slaughter Cr. at FM 2304	59.8	23.1	1978-1981	Rural	--
08158600	Walnut Cr. at Webberville Rd.	132.9	51.3	1975-1981	Urban	--
08158920	Williamson Cr. at Oak Hill	16.32	6.3	1974-1981	Urban	--
08158970	Williamson Cr. at Jimmy Clay Rd.	71.5	27.6	1975-1981	Urban	20

*Impervious cover estimates from City of Austin, 1983.

Source: USGS, 1983a.

TABLE 4-24

PHOSPHORUS SUMMARY STATISTICS
MINOR TRIBUTARY MONITORING STATIONS

USGS St. No.	Description	TP Range (mg/l)	Average TP (mg/l)	Standard Deviation	Number of Data Points
08155200	Barton Cr. at Hwy 71	0.0-0.18	0.036	0.049	20
08155300	Barton Cr. at Loop 360	0.0-0.73	0.128	0.163	34
08158810	Bear Cr. below FM 1826	0.0-0.09	0.024	0.024	20
08158050	Boggy Cr. at Hwy 183	0.0-2.60	0.594	0.746	71
08154700	Bull Cr. at Loop 360	0.0-1.70	0.207	0.304	45
08158825	Little Bear Cr. at FM 1626	0.04-0.57	0.165	0.183	8
08158700	Onion Cr. near Driftwood	0.0-0.20	0.025	0.035	66
08158800	Onion Cr. at Buda	0.0-0.30	0.066	0.091	23
08159000	Onion Cr. at Hwy 183	0.0-1.60	0.071	0.239	45
08156800	Shoal Cr. at 12th St.	0.1-3.70	0.478	0.720	58
08158860	Slaughter Cr. at FM 2304	0.01-0.39	0.094	0.116	9
08158600	Walnut Cr. at Webberville Rd.	0.0-2.00	0.238	0.452	35
08158920	Williamson Cr. at Oak Hill	0.0-0.65	0.186	0.148	45
08158970	Williamson Cr. at Jimmy Clay Rd.	0.0-1.40	0.207	0.296	52

Data source: USGS, 1983a.

TABLE 4-25

PHOSPHORUS SUMMARY STATISTICS FOR HIGH FLOW DATA
MINOR TRIBUTARY MONITORING STATIONS

USGS St. No.	Description	TP Range (mg/l)	Average TP (mg/l)	Standard Deviation	Number of Data Points
08155200	Barton Cr. at Hwy 71	0.0-0.13	0.032	0.037	11
08155300	Barton Cr. at Loop 360	0.0-0.73	0.128	0.163	34
08158810	Bear Cr. below FM 1826	0.01-0.09	0.037	0.038	6
08158050	Boggy Cr. at Hwy 183	0.02-2.60	0.999	0.778	40
08154700	Bull Cr. at Loop 360	0.0-1.70	0.293	0.333	31
08158825	Little Bear Cr. at FM 1626	0.04-0.21	0.120	0.085	3
08158700	Onion Cr. near Driftwood	0.0-0.20	0.029	0.045	40
08158800	Onion Cr. at Buda	0.01-0.30	0.118	0.112	11
08159000	Onion Cr. at Hwy 183	0.01-1.60	0.110	0.311	26
08156800	Shoal Cr. at 12th St.	0.10-3.70	0.663	0.802	40
08158860	Slaughter Cr. at FM 2304	0.02-0.39	0.105	0.120	8
08158600	Walnut Cr. at Webberville Rd.	0.0-2.00	0.404	0.566	19
08158920	Williamson Cr. at Oak Hill	0.03-0.44	0.150	0.143	10
08158970	Williamson Cr. at Jimmy Clay Rd.	0.02-1.40	0.365	0.333	22

Data source: USGS, 1983a.

TABLE 4-26

STORMWATER TOTAL PHOSPHORUS DATA
AUSTIN INTENSIVE PLANNING AREA

Station	Average Total Phosphorus* (mg/l)	Volumetric Average Total Phosphorus** (mg/l)	Number of Samples	Land Use†
Johnson Branch	.14	.15	10	72% LDR, 14% HDR, 7% COM, 7% UND
W. Bouldin Creek	.52	.53	11	46% LDR, 5% HDR, 15% COM, 34% UND
Downtown storm sewer	.19	.16	11	93% COM, 5% HDR, 2% UND
Waller Creek	.24	.29	12	56% LDR, 4% HDR, 22% COM, 5% IND, 13% UND
Unnamed tributary	.14	.15	10	53% HDR, 17% LDR, 6% COM, 24% UND
Ferguson Branch	.12	.17	7	91% UND, 9% LDR

* Mathematical average of all data points.

** Average weighted by runoff volume.

† LDR = low density residential
HDR = high density residential
COM = commercial/public
IND = industrial
UND = undeveloped/park

TABLE 4-27

STORMWATER TOTAL PHOSPHORUS DATA
SAN ANGELO INTENSIVE PLANNING AREA

Station	Average Total Phosphorus* (mg/l)	Volumetric Average Total Phosphorus** (mg/l)	Number of Samples	Land Use†
Unnamed tributary	.42	.52	15	79% SFR, 2% COM, 19% UND
North Concho River	.37	.68	15	Nonurban watershed
Sulphur Draw	.50	.69	15	72% SFR, 2% MFR, 26% COM
North Concho River	.44	.71	15	After urban inflow
South Concho River	.23	.20	15	Nonurban watershed
Red Arroyo	.61	.63	15	77% UND, 15% SFRS, 1% MFR, 6% COM, 1% IND
Concho River	.35	.58	21	After urban inflow

* Mathematical average of all data points.

** Average weighted by runoff volume.

† SFR = single family residential
MFR = multifamily residential
COM = commercial/public
IND = industrial
UND = undeveloped

TABLE 4-28

SUMMARY OF PHOSPHORUS CONCENTRATIONS
IN STORM RUNOFF
AUSTIN NURP STUDY

Site*	TP Range (mg/l)	Median TP (mg/l)	Mean TP (mg/l)	Standard Deviation	Data Points
H	0.03 - 3.40	0.2	0.31	0.42	104
W	0.08 - 0.37	0.2	0.22	0.08	34
R	0.04 - 3.60	0.22	0.29	0.39	85
Tb	<0.01 - 0.31	0.03	0.04	0.07	21
Ts	<0.01 - 0.24	0.06	0.10	0.09	9

* H = Hart Lane (Northwest Austin)
W = Woodhollow Dam
R = Rollingwood
Tb = Turkey Creek background data
Ts = Turkey Creek storm data

TABLE 4-29

PHOSPHORUS DATA FOR LAKE AUSTIN

AUSTIN NURP STUDY

Sampling Date	Constituent*	Station ES-O-A	ES-O-B	ES-O-O	ES-O-A	ES-1-B	ES-1-C	ES-2-A	ES-2-B	ES-2-C	ES-3
March 1981**	TP	—	—	—	—	—	0.01	—	—	<0.01	<0.01
	DOP	—	—	—	—	—	0.01	—	—	<0.01	<0.01
	TSS	—	—	—	—	—	6	—	—	14	6
1 May 81	TP	—	—	0.03	0.09	0.03	0.03	0.17	0.03	0.03	0.03
	DOP	—	—	0.02	0.02	0.03	0.03	0.03	0.03	0.03	0.03
	TSS	—	—	5	69	14	5	144	14	7	6
3 May 81	TP	—	—	0.01	0.03	0.03	0.03	0.06	0.03	0.01	0.01
	DOP	—	—	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
	TSS	—	—	7	38	19	4	39	17	8	5
4 May 81	TP	—	—	0.03	0.05	0.03	0.03	0.07	0.03	0.03	0.03
	DOP	—	—	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
	TSS	—	—	1	29	13	7	129	22	5	3
5 May 81	TP	—	—	0.03	0.06	0.01	0.01	0.01	0.06	0.01	0.01
	DOP	—	—	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
	TSS	—	—	1	31	17	5	27	16	4	3
16 May 81	TP	—	—	<0.01	0.09	0.12	0.01	0.06	0.11	0.03	0.02
	DOP	—	—	<0.01	0.02	0.03	0.01	0.02	0.02	0.03	0.02
	TSS	—	—	1	144	28	40	270	116	17	7
17 May 81	TP	—	—	<0.01	0.06	0.09	0.05	0.07	0.03	0.14	<0.01
	DOP	—	—	<0.01	0.03	0.03	0.03	0.03	0.02	0.02	<0.01
	TSS	—	—	7	124	26	21	87	32	8	6

TABLE 4-29 (Concluded)

Sampling Date	Constituent*	Station									
		ES-O-A	ES-O-B	ES-O-O	ES-O-A	ES-1-B	ES-1-C	ES-2-A	ES-2-B	ES-2-C	ES-3
18 May 81	TP	—	—	0.02	0.05	0.03	0.07	0.05	0.06	0.03	0.03
	DOP	—	—	0.01	<0.01	<0.01	<0.01	0.03	0.05	0.01	0.01
	TSS	—	—	2	38	20	13	29	16	8	4
20 May 81	TP	—	—	0.01	0.03	0.03	0.02	0.04	0.04	0.03	0.02
	DOP	—	—	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	TSS	—	—	3	48	30	8	56	76	8	7
26 May 81	TP	—	—	0.08	0.03	0.05	0.07	0.05	0.08	0.06	0.10
	DOP	—	—	<0.01	<0.01	0.01	0.01	<0.01	0.01	0.01	<0.01
	TSS	—	—	23	19	22	37	22	34	43	29
28 May 81	TP	—	—	0.05	0.03	0.03	0.06	0.05	0.06	0.08	0.06
	DOP	—	—	0.01	0.02	<0.01	<0.01	<0.01	0.01	0.01	0.01
	TSS	—	—	7	5	22	17	18	15	14	13
11 June 81	TP	0.03	0.03	0.03	0.04	0.05	0.05	0.04	0.05	0.05	0.04
	DOP	<0.01	0.01	<0.01	0.01	0.01	0.01	0.01	0.02	0.01	0.01
	TSS	26	5	16	22	41	53	25	25	26	24
6 July 81	TP	0.03	0.03	0.03	0.03	0.05	0.03	0.06	0.06	0.01	<0.01
	DOP	<0.01	<0.01	<0.01	0.03	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	TSS	8	8	9	33	67	23	41	37	6	9
9 July 81	TP	<0.01	0.02	<0.01	0.03	0.03	0.02	0.05	0.03	0.03	<0.01
	DOP	<0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	TSS	7	14	9	72	48	19	30	27	23	9

** March 1981 data represent ambient conditions, other dates are poststorm samples.

* TP = total phosphorus, DOP = dissolved orthophosphorus, TSS = total suspended solids, all in mg/l.

TABLE 4-30

PHOSPHORUS DATA FOR TOWN LAKE
AUSTIN NURP STUDY

Sampling Date	Constituent*	Station ES-4	ES-5	ES-6	ES-7	ES-8
March 1981**	TP	0.02	0.03	0.03	—	—
	DOP	<0.01	0.01	0.01	—	—
	TSS	4	7	11	—	—
11 June 1981	TP	0.04	0.05	0.10	0.13	0.08
	DOP	0.03	0.02	0.01	0.01	0.05
	TSS	11	74	209	300	45
6 July 1981	TP	<0.01	<0.01	0.03	0.03	0.08
	DOP	<0.01	<0.01	<0.01	<0.01	0.05
	TSS	6	11	17	25	6
9 July 1981	TP	0.03	0.03	0.02	0.01	0.03
	DOP	<0.01	<0.01	<0.01	<0.01	0.01
	TSS	12	6	3	26	11

* TP = total phosphorus, DOP = dissolved orthophosphorus, TSS = total suspended solids, all in mg/l.

** March 1981 data represnt ambient conditions, other dates are poststorm samples.

TABLE 4-31

SAMPLING STATION LOCATIONS
LAKE AUSTIN AND TOWN LAKE
AUSTIN NURP STUDY

Waterbody	Station	Description
Lake Austin	ES-O-A	Turkey Cr. above confluence
	ES-O-B	Turkey Cr. near confluence
	ES-O-O	Loop 360 bridge
	ES-1-A	Bull Cr. arm .7 mi above confluence
	ES-1-B	Bull Cr. arm .25 mi above confluence
	ES-1-C	Mouth of Bull Cr.
	ES-2-A	Dry Cr. arm .4 mi above confluence
	ES-2-B	Dry Cr. arm .25 mi above confluence
	ES-2-C	Mouth of Dry Cr.
	ES-3	Main pool near Tom Miller Dam
Town Lake	ES-4	Mopac bridge
	ES-5	South First Street bridge
	ES-6	Main pool near Longhorn Dam
	ES-7	Mouth of Barton Cr.
	ES-8	Lamar Blvd. bridge

TABLE 4-32

ADJACENT WATERSHED DELINEATIONS

HIGHLAND LAKES

Subbasin No.	Description	Area		Reservoir Watershed
		(km^2)	(mi^2)	
1	Colorado R. near Tow, TX	1064	411	Buchanan
2	Buchanan Dam	293	113	Buchanan
3	Roy Inks Dam	98	38	Inks
4	Colorado R. above Llano R.	83	32	LBJ
5	Llano R. at mouth	44	17	LBJ
6	Sandy Cr. above Walnut Cr.	31	12	LBJ
7	Walnut Cr.	65	25	LBJ
8	Alvin Wirtz Dam	186	72	LBJ
9	Max Starcke Dam	215	83	Marble Falls
10	Colorado R. below Hamilton Cr.	233	90	Travis
11	Colorado R. above Pedernales R.	342	132	Travis
12	Pedernales R. at mouth	124	48	Travis
13	Mansfield Dam	580	224	Travis
14	Colorado R. above Bull Cr.	140	54	Austin
15	Tom Miller Dam	127	49	Austin
16	Barton Cr.	287	111	Town
17	Longhorn Dam	96	37	Town

TABLE 4-33

LAND USE DESCRIPTION
HIGHLAND LAKES WATERSHED
IMMEDIATE DRAINAGE AREAS

Subbasin No.	Description	Land Use	1983 Conditions (km^2)	(mi^2)
1	Colorado R. near Tow	Rangeland	775	299
		Forest	64	25
		Cropland	217	84
		Urban	7.7	3.0
2	Buchanan Dam	Rangeland	202	78
		Forest	49	19
		Cropland	10	4
		Urban	29.6	11.4
3	Roy Inks Dam	Rangeland	97	37
		Cropland	1	1
		Urban	.6	.2
4	Colorado R. above mouth of Llano R.	Rangeland	69	27
		Cropland	9	4
		Urban	4.4	1.7
5	Llano R. at mouth	Rangeland	23	9
		Forest	12	5
		Urban	7.2	2.8
6	Sandy Cr. above Walnut Cr.	Rangeland	27	10
		Forest	4	2
7	Walnut Cr.	Rangeland	65	25
8	Alvin Wirtz Dam	Rangeland	152	59
		Urban	35.8	13.8
9	Max Starcke Dam	Rangeland	202	78
		Cropland	1	1
		Urban	9.1	3.5

TABLE 4-33 (Concluded)

Subbasin No.	Description	Land Use	1983 Conditions (km^2)	(mi^2)
10	Colorado R. below Hamilton Cr.	Rangeland	207	80
		Forest	11	4
		Cropland	9	3
		Urban	8.3	3.2
11	Colorado R. above Pedernales R.	Rangeland	260	100
		Forest	43	17
		Cropland	34	13
		Urban	4.5	1.7
12	Pedernales R. at mouth	Rangeland	124	48
		Urban	0.8	0.3
13	Mansfield Dam	Rangeland	393	152
		Forest	141	54
		Cropland	6	2
		Urban	43.1	16.6
14	Colorado R. above Bull Cr.	Rangeland	96	37
		Forest	37	14
		Urban	8.3	3.2
15	Tom Miller Dam	Rangeland	99	38
		Forest	14	5
		Urban	13.3	5.1
16	Barton Cr.	Rangeland	233	90
		Forest	43	17
		Urban	8.6	3.3
17	Longhorn Dam	Urban	96.2	37.1

TABLE 4-34

GENERAL SOILS DESCRIPTION
HIGHLAND LAKES WATERSHED
IMMEDIATE SUBBASIN DRAINAGE AREAS

Subbasin No.	Description	Soil Type	Hydrologic Group	Area (km^2)	Area (mi^2)
1	Colorado R. near Tow	Weswood-Yahola	B	2	1
		Roughcreek-Eckrant	D	418	161
		Rumple	C	301	116
		Yates-Hye	D	38	15
		Katemcy-Demona-Ligon	C	16	6
		Hensley-Eckrant	D	146	56
		Brackett-Purves-Doss	C	13	5
		Roughcreek	D	5	2
		Brackett-Dugout-Purves	C	70	27
		Callahan-Nocken	C	49	19
		Weswood	B	6	2
2	Buchanan Dam	Hensley-Eckrant	D	151	58
		Keese-Nebgen	D	48	19
		Brackett-Purves-Doss	C	2	1
		Rumple	C	5	2
		Hye-Pontotoc-Nebgen	B	17	7
		Keese-Ligon	D	40	16
		Roughcreek	D	1	1
		Castell-Click	C	26	10

TABLE 4-34 (Cont'd)

Subbasin No.	Description	Soil Type	Hydrologic Group	Area	
				(km^2)	(mi^2)
3	Roy Inks Dam	Keese-Nebgen	D	74	29
		Hensley-Eckrant	D	16	6
		Keese-Ligon	D	3	1
4	Colorado R. above Llano R.	Voca-Click	C	18	7
		Hensley-Eckrant	D	31	12
		Keese-Ligon	D	34	13
5	Llano R. at mouth	Keese-Ligon	D	12	5
		Ligon-Katemcy	D	29	11
6	Sandy Cr. above Walnut Cr.	Ligon-Katemcy	D	31	12
7	Walnut Cr.	Ligon-Katemcy	D	11	4
		Keese-Ligon	D	5	2
		Nebgen-Eckert-Ligon	D	28	11
		Brackett-Purves-Doss	C	19	7
8	Alvin Wirtz Dam	Nebgen-Eckert-Ligon	D	11	4
		Hensley-Tarpley	D	10	4
		Brackett-Purves-Doss	C	13	5
		Voca-Click	C	64	25
		Hensley-Eckrant	D	5	2
		Keese-Ligon	D	19	8
		Castell-Click	C	19	7
		Tarpley-Eckrant	D	10	4
		Ligon-Katemcy	D	36	14

TABLE 4-34 (Cont'd)

Subbasin No.	Description	Soil Type	Hydrologic Group	Area (km^2)	Area (mi^2)
9	Max Starcke Dam	Voca-Click	C	47	18
		Hensley-Eckrant	D	109	42
		Hensley-Tarpley	D	49	19
		Brackett-Purves-Doss	C	10	4
		Tarpley-Eckrant	D	7	3
10	Colorado R. below Hamilton Cr.	Brackett-Purves-Doss	C	110	42
		Hensley-Eckrant	D	123	47
11	Colorado R. above Pedernales R.	Hensley-Eckrant	D	96	37
		Brackett-Purves-Doss	C	194	75
		Hensley-Tarpley	D	3	1
		Brackett	C	50	19
12	Pedernales R. at mouth	Brackett-Purves-Doss	C	38	15
		Brackett	C	85	33
13	Mansfield Dam	Eckrant-Brackett	D	39	15
		Brackett-Purves-Doss	C	19	8
		Brackett	C	450	174
		Tarrant	C	74	28
14	Colorado R. above Bull Cr.	Brackett	C	143	55

TABLE 4-34 (Concluded)

Subbasin No.	Description	Soil Type	Hydrologic Group	Area	
				(km^2)	(mi^2)
15	Tom Miller Dam	Brackett	C	93	36
		Tarrant	C	35	13
16	Barton Cr. at mouth	Brackett	C	281	109
		Houston Black-Heiden	D	4	2
17	Longhorn Dam	Brackett	C	17	7
		Austin-Eddy	C	42	16
		Houston Black-Heiden	D	20	8
		Bergstrom-Norwood	B	13	5
		Tarrant	C	3	1

TABLE 4-35

STORM FREQUENCY DISTRIBUTION FOR AUSTIN STATION (INCHES)

INTERVAL		NO. OF EVENTS	FREQ	CUM. NO.	CUM. FREQ
0.00	0.10	1658.00	0.40	1658.00	0.40
0.10	0.20	643.00	0.15	2301.00	0.55
0.20	0.30	384.00	0.09	2685.00	0.64
0.30	0.40	264.00	0.06	2949.00	0.71
0.40	0.50	185.00	0.04	3134.00	0.75
0.50	0.60	139.00	0.03	3273.00	0.79
0.60	0.70	132.00	0.03	3405.00	0.82
0.70	0.80	104.00	0.02	3509.00	0.84
0.80	0.90	82.00	0.02	3591.00	0.86
0.90	1.00	85.00	0.02	3676.00	0.88
1.00	1.10	67.00	0.02	3743.00	0.90
1.10	1.20	49.00	0.01	3792.00	0.91
1.20	1.30	54.00	0.01	3846.00	0.92
1.30	1.40	39.00	0.01	3885.00	0.93
1.40	1.50	40.00	0.01	3925.00	0.94
1.50	1.60	36.00	0.01	3961.00	0.95
1.60	1.70	23.00	0.01	3984.00	0.96
1.70	1.80	16.00	0.00	4000.00	0.96
1.80	1.90	19.00	0.00	4019.00	0.96
1.90	2.00	14.00	0.00	4033.00	0.97
2.00	2.10	13.00	0.00	4046.00	0.97
2.10	2.20	10.00	0.00	4056.00	0.97
2.20	2.30	14.00	0.00	4070.00	0.98
2.30	2.40	8.00	0.00	4078.00	0.98
2.40	2.50	10.00	0.00	4088.00	0.98
2.50	2.60	11.00	0.00	4099.00	0.98
2.60	2.70	3.00	0.00	4102.00	0.98
2.70	2.80	7.00	0.00	4109.00	0.99
2.80	2.90	6.00	0.00	4115.00	0.99
2.90	3.00	1.00	0.00	4116.00	0.99
3.00	3.10	7.00	0.00	4123.00	0.99
3.10	3.20	3.00	0.00	4126.00	0.99
3.20	3.30	6.00	0.00	4132.00	0.99
3.30	3.40	2.00	0.00	4134.00	0.99
3.40	3.50	3.00	0.00	4137.00	0.99
3.50	3.60	1.00	0.00	4138.00	0.99
3.60	3.70	3.00	0.00	4141.00	0.99
3.70	3.80	3.00	0.00	4144.00	0.99
3.80	3.90	4.00	0.00	4148.00	1.00
3.90	4.00	1.00	0.00	4149.00	1.00
4.00	4.10	2.00	0.00	4151.00	1.00
4.10	4.20	1.00	0.00	4152.00	1.00
4.20	4.30	4.00	0.00	4156.00	1.00
4.30	4.40	0.00	0.00	4156.00	1.00
4.40	4.50	1.00	0.00	4157.00	1.00
4.50	4.60	1.00	0.00	4158.00	1.00
4.60	4.70	3.00	0.00	4161.00	1.00
4.70	4.80	1.00	0.00	4162.00	1.00
4.80	4.90	1.00	0.00	4163.00	1.00
4.90	5.00	0.00	0.00	4163.00	1.00
5.00	5.10	0.00	0.00	4163.00	1.00
5.10	5.20	0.00	0.00	4163.00	1.00
5.20	5.30	1.00	0.00	4164.00	1.00
5.30	5.40	0.00	0.00	4164.00	1.00
5.40	5.50	0.00	0.00	4164.00	1.00

TABLE 4-35 (Concluded)

5.50	5.60	1.00	0.00	4165.00	1.00
5.60	5.70	1.00	0.00	4166.00	1.00
5.70	5.80	0.00	0.00	4166.00	1.00
5.80	5.90	0.00	0.00	4166.00	1.00
5.90	6.00	0.00	0.00	4166.00	1.00
6.00	6.10	0.00	0.00	4166.00	1.00
6.10	6.20	0.00	0.00	4166.00	1.00
6.20	6.30	0.00	0.00	4166.00	1.00
6.30	6.40	0.00	0.00	4166.00	1.00
6.40	6.50	0.00	0.00	4166.00	1.00
6.50	6.60	0.00	0.00	4166.00	1.00
6.60	6.70	0.00	0.00	4166.00	1.00
6.70	6.80	0.00	0.00	4166.00	1.00
6.80	6.90	0.00	0.00	4166.00	1.00
6.90	7.00	0.00	0.00	4166.00	1.00
7.00	7.10	0.00	0.00	4166.00	1.00
7.10	7.20	0.00	0.00	4166.00	1.00
7.20	7.30	0.00	0.00	4166.00	1.00
7.30	7.40	0.00	0.00	4166.00	1.00
7.40	7.50	0.00	0.00	4166.00	1.00
7.50	7.60	0.00	0.00	4166.00	1.00
7.60	7.70	0.00	0.00	4166.00	1.00
7.70	7.80	0.00	0.00	4166.00	1.00
7.80	7.90	0.00	0.00	4166.00	1.00
7.90	8.00	0.00	0.00	4166.00	1.00
8.00	8.10	1.00	0.00	4167.00	1.00

RAIN DAYS= 4167 RMIN= 0.00 RMAX= 8.00 ANN. AVG.= 32.39

TABLE 4-36

STORM FREQUENCY DISTRIBUTION
FOR LLANO STATION
(INCHES)

INTERVAL		NO. OF EVENTS	FREQ	CUM. NO.	CUM. FREQ
0.00	0.10	1097.00	0.33	1097.00	0.33
0.10	0.20	516.00	0.16	1613.00	0.49
0.20	0.30	334.00	0.10	1947.00	0.59
0.30	0.40	251.00	0.08	2198.00	0.67
0.40	0.50	191.00	0.06	2389.00	0.73
0.50	0.60	157.00	0.05	2546.00	0.77
0.60	0.70	118.00	0.04	2664.00	0.81
0.70	0.80	96.00	0.03	2760.00	0.84
0.80	0.90	79.00	0.02	2839.00	0.86
0.90	1.00	67.00	0.02	2906.00	0.88
1.00	1.10	52.00	0.02	2958.00	0.90
1.10	1.20	43.00	0.01	3001.00	0.91
1.20	1.30	36.00	0.01	3037.00	0.92
1.30	1.40	34.00	0.01	3071.00	0.93
1.40	1.50	30.00	0.01	3101.00	0.94
1.50	1.60	26.00	0.01	3127.00	0.95
1.60	1.70	16.00	0.00	3143.00	0.95
1.70	1.80	23.00	0.01	3166.00	0.96
1.80	1.90	21.00	0.01	3187.00	0.97
1.90	2.00	12.00	0.00	3199.00	0.97
2.00	2.10	18.00	0.01	3217.00	0.98
2.10	2.20	9.00	0.00	3226.00	0.98
2.20	2.30	8.00	0.00	3234.00	0.98
2.30	2.40	9.00	0.00	3243.00	0.98
2.40	2.50	10.00	0.00	3253.00	0.99
2.50	2.60	5.00	0.00	3258.00	0.99
2.60	2.70	1.00	0.00	3259.00	0.99
2.70	2.80	2.00	0.00	3261.00	0.99
2.80	2.90	4.00	0.00	3265.00	0.99
2.90	3.00	2.00	0.00	3267.00	0.99
3.00	3.10	5.00	0.00	3272.00	0.99
3.10	3.20	5.00	0.00	3277.00	0.99
3.20	3.30	3.00	0.00	3280.00	1.00
3.30	3.40	0.00	0.00	3280.00	1.00
3.40	3.50	4.00	0.00	3284.00	1.00
3.50	3.60	1.00	0.00	3285.00	1.00
3.60	3.70	0.00	0.00	3285.00	1.00
3.70	3.80	0.00	0.00	3285.00	1.00
3.80	3.90	1.00	0.00	3286.00	1.00
3.90	4.00	0.00	0.00	3286.00	1.00
4.00	4.10	0.00	0.00	3286.00	1.00
4.10	4.20	0.00	0.00	3286.00	1.00
4.20	4.30	1.00	0.00	3287.00	1.00
4.30	4.40	2.00	0.00	3289.00	1.00
4.40	4.50	0.00	0.00	3289.00	1.00
4.50	4.60	0.00	0.00	3289.00	1.00
4.60	4.70	1.00	0.00	3290.00	1.00
4.70	4.80	0.00	0.00	3290.00	1.00
4.80	4.90	0.00	0.00	3290.00	1.00
4.90	5.00	0.00	0.00	3290.00	1.00
5.00	5.10	0.00	0.00	3290.00	1.00
5.10	5.20	0.00	0.00	3290.00	1.00
5.20	5.30	0.00	0.00	3290.00	1.00
5.30	5.40	0.00	0.00	3290.00	1.00
5.40	5.50	0.00	0.00	3290.00	1.00

TABLE 4-36 (Concluded)

5.50	5.60	1.00	0.00	3291.00	1.00
5.60	5.70	0.00	0.00	3291.00	1.00
5.70	5.80	0.00	0.00	3291.00	1.00
5.80	5.90	0.00	0.00	3291.00	1.00
5.90	6.00	0.00	0.00	3291.00	1.00
6.00	6.10	0.00	0.00	3291.00	1.00
6.10	6.20	0.00	0.00	3291.00	1.00
6.20	6.30	0.00	0.00	3291.00	1.00
6.30	6.40	1.00	0.00	3292.00	1.00
6.40	6.50	0.00	0.00	3292.00	1.00
6.50	6.60	0.00	0.00	3292.00	1.00
6.60	6.70	0.00	0.00	3292.00	1.00
6.70	6.80	1.00	0.00	3293.00	1.00
6.80	6.90	0.00	0.00	3293.00	1.00
6.90	7.00	0.00	0.00	3293.00	1.00
7.00	7.10	0.00	0.00	3293.00	1.00
7.10	7.20	0.00	0.00	3293.00	1.00
7.20	7.30	1.00	0.00	3294.00	1.00

RAIN DAYS= 3294 RMIN= 0.00 RMAX= 7.20 ANN. AVG.= 27.03

TABLE 4-37

RUNOFF CURVE NUMBERS FOR HYDROLOGIC SOIL-COVER COMPLEXES
(ANTECEDENT MOISTURE CONDITION II, AND $I_a = 0.2\ S$)

Land Use Description/Treatment/Hydrologic Condition			Hydrologic Soil Group A	B	C	D
Residential:[1]						
Average lot size	Average % Impervious[2]					
1/8 acre or less	65		77	85	90	92
1/4 acre	38		61	75	83	87
1/3 acre	30		57	72	81	86
1/2 acre	25		54	70	80	85
1 acre	20		51	68	79	84
Paved parking lots, roofs, driveways, etc.[3]			98	98	98	98
Streets and roads:						
paved with curbs and storm sewers[3]			98	98	98	98
gravel			76	85	89	91
dirt			72	82	87	89
Commercial and business areas (85% impervious)			89	92	94	95
Industrial districts (72% impervious)			81	88	91	93
Open spaces, lawns, parks, golf courses, cemeteries, etc.						
good condition: grass cover on 75% or more of the area			39	61	74	80
fair condition: grass cover on 50% to 75% of the area			49	69	79	84
Fallow	Straight row	—	77	86	91	94
Row crops	Straight row	Poor	72	81	88	91
	Straight row	Good	67	78	85	89
	Contoured	Poor	70	79	84	88
	Contoured	Good	65	75	82	86
	Contoured & terraced	Poor	66	74	80	82
	Contoured & terraced	Good	62	71	78	81
Small grain	Straight row	Poor	65	76	84	88
		Good	63	75	83	87
	Contoured	Poor	63	74	82	85
		Good	61	73	81	84
	Contoured & terraced	Poor	61	72	79	82
		Good	59	70	78	81
Close-seeded legumes[4] or rotation meadow	Straight row	Poor	66	77	85	89
	Straight row	Good	58	72	81	85
	Contoured	Poor	64	75	83	85
	Contoured	Good	55	69	78	83
	Contoured & terraced	Poor	63	73	80	83
	Contoured & terraced	Good	51	67	76	80

TABLE 4-37 (Concluded)

Land Use Description/Treatment/Hydrologic Condition			Hydrologic Soil Group A	B	C	D
Pasture or range		Poor	68	79	86	89
		Fair	49	69	79	84
		Good	39	61	74	80
	Contoured	Poor	47	67	81	88
	Contoured	Fair	25	59	75	83
	Contoured	Good	6	35	70	79
Meadow		Good	30	58	71	78
Woods or forest land		Poor	45	66	77	83
		Fair	36	60	73	79
		Good	25	55	70	77
Farmsteads		—	59	74	82	86

1 Curve numbers are computed assuming the runoff from the house and driveway is directed towards the street with a minimum of roof water directed to lawns where additional infiltration could occur.

2 The remaining pervious areas (lawn) are considered to be in good pasture condition for these curve numbers.

3 In some warmer climates of the country a curve number of 95 may be used.

4 Close-drilled or broadcast.

Source: McCuen, 1982.

TABLE 4-38

CURVE NUMBERS (CN) AND CONSTANTS FOR THE CASE $I_a = 0.2\ S$

CN for Condition II	CN for Conditions I	III	S Values* (inches)	Curve* Starts Where P = (inches)	CN for Condition II	CN for Conditions I	III	S Values* (inches)	Curve* Starts Where P = (inches)
100	100	100	0	0	60	40	78	6.67	1.33
99	97	100	.101	.02	59	39	77	6.95	1.39
98	94	99	.204	.04	58	38	76	7.24	1.45
97	91	99	.309	.06	57	37	75	7.54	1.51
96	89	99	.417	.08	56	36	75	7.86	1.57
95	87	98	.526	.11	55	35	74	8.18	1.64
94	85	98	.638	.13	54	34	73	8.52	1.70
93	83	98	.753	.15	53	33	72	8.87	1.77
92	81	97	.870	.17	52	32	71	9.23	1.85
91	80	97	.989	.20	51	31	70	9.61	1.92
90	78	96	1.11	.22	50	31	70	10.0	2.00
89	76	96	1.24	.25	49	30	69	10.4	2.08
88	75	95	1.36	.27	48	29	68	10.8	2.16
87	73	95	1.49	.30	47	28	67	11.3	2.26
86	72	94	1.63	.33	46	27	66	11.7	2.34
85	70	94	1.76	.35	45	26	65	12.2	2.44
84	68	93	1.90	.38	44	25	64	12.7	2.54
83	67	93	2.05	.41	43	25	63	13.2	2.64
82	66	92	2.20	.44	42	24	62	13.8	2.76
81	64	92	2.34	.47	41	23	61	14.4	2.88
80	63	91	2.50	.50	40	22	60	15.0	3.00
79	62	91	2.66	.53	39	21	59	15.6	3.12
78	60	90	2.82	.56	38	21	58	16.3	3.26
77	59	89	2.99	.60	37	20	57	17.0	3.40
76	58	89	3.16	.63	36	19	56	17.8	3.56
75	57	88	3.33	.67	35	18	55	18.6	3.72
74	55	88	3.51	.70	34	18	54	19.4	3.88
73	54	87	3.70	.74	33	17	53	20.3	4.06
72	53	86	3.89	.78	32	16	52	21.2	4.24
71	52	86	4.08	.82	31	16	51	22.2	4.44
70	51	85	4.28	.86	30	15	50	23.3	4.66
69	50	84	4.49	.90					
68	48	84	4.70	.94	25	12	43	30.0	6.00
67	47	83	4.92	.98	20	9	37	40.0	8.00
66	46	82	5.15	1.03	15	6	30	56.7	11.34
65	45	82	5.38	1.08	10	4	22	90.0	18.00
64	44	81	5.62	1.12	5	2	13	90.0	38.00
63	43	80	5.87	1.17	0	0	0	infinity	infinity
62	42	79	6.13	1.23					
61	41	78	6.39	1.28					

*For CN in column 1.

Source: Mockus, 1972.

TABLE 4-39

SCS CURVE NUMBERS
HIGHLAND LAKES WATERSHEDS

Subbasin	Rangeland	Forest	Cropland	Urban
1	71.85	65.97	77.33	76.33
2	72.41	67.00	77.64	76.65
3	74.00	69.00	80.00	79.00
4	72.90	67.46	78.68	77.68
5	74.00	69.00	80.00	79.00
6	74.00	69.00	80.00	79.00
7	72.55	66.97	78.26	77.26
8	71.45	65.43	76.94	75.94
9	72.70	67.18	78.44	77.44
10	71.65	65.71	77.18	76.18
11	70.45	64.03	75.74	74.74
12	69.00	62.00	74.00	73.00
13	69.35	62.49	74.42	73.42
14	69.00	62.00	74.00	73.00
15	69.00	62.00	74.00	73.00
16	69.10	62.14	74.12	73.12
17	70.79	63.88	74.80	73.79

TABLE 4-40

FLOWS FROM LAND USE CATEGORIES

Subbasin	Precip. (in)	Runoff Flow (10^6 m^3/yr) Rangeland	Forest	Cropland	Urban	Total
1	1.5	8.72	0.253	4.79	0.152	—
1	2.5	0.0	0.0	0.0	0.0	—
1	4.0	57.1	3.52	20.2	0.689	95.462
2	1.5	2.46	0.243	0.228	0.606	—
2	2.5	0.0	0.0	0.0	0.0	—
2	4.0	15.3	2.85	0.945	2.69	25.278
3	1.5	1.45	0.0	.029	0.0158	—
3	2.5	0.0	0.0	0.0	0.0	—
3	4.0	7.87	0.0	0.104	0.0598	9.5276
4	1.5	0.897	0.0	0.229	0.101	—
4	2.5	0.0	0.0	0.0	0.0	—
4	4.0	5.33	0.0	0.886	0.416	7.8593
5	1.5	0.345	0.0866	0.0	0.189	—
5	2.5	0.0	0.0	0.0	0.0	—
5	4.0	1.87	.772	0.0	0.718	3.9757
6	1.5	0.405	.0289	0.0	0.0	—
6	2.5	0.0	0.0	0.0	0.0	—
6	4.0	2.19	0.257	0.0	0.0	2.8803
7	1.5	0.806	0.0	0.0	0.0	—
7	2.5	0.0	0.0	0.0	0.0	—
7	4.0	4.94	0.0	0.0	0.0	5.7489
8	1.5	1.62	0.0	0.0	0.677	—
8	2.5	0.0	0.0	0.0	0.0	—
8	4.0	11.0	0.0	0.0	3.15	16.438
9	1.5	2.56	0.0	0.0248	0.203	—
9	2.5	0.0	0.0	0.0	0.0	—
9	4.0	1.55	0.0	0.0976	0.853	19.199
10	1.5	5.66	0.103	0.488	0.403	—
10	2.5	0.0	0.0	0.0	0.0	—
10	4.0	0.0	0.0	0.0	0.0	6.6534
11	1.5	5.94	0.258	1.57	0.185	—
11	2.5	0.0	0.0	0.0	0.0	—
11	4.0	0.0	0.0	0.0	0.0	7.9573
12	1.5	2.24	0.0	0.0	0.0263	—
12	2.5	0.0	0.0	0.0	0.0	—
12	4.0	0.0	0.0	0.0	0.0	2.2629

TABLE 4-40 (Concluded)

Subbasin	Precip. (in)	Runoff Flow (10^6 m^3/yr) Rangeland	Forest	Cropland	Urban	Flow
13	1.5	7.52	0.510	0.237	1.50	—
13	2.5	0.0	0.0	0.0	0.0	—
13	3.0	0.0	0.0	0.0	0.0	9.7696
14	1.5	1.73	0.110	0.0	0.273	—
14	2.5	0.0	0.0	0.0	0.0	—
14	4.0	0.0	0.0	0.0	0.0	2.1152
15	1.5	1.79	0.0418	0.0	0.438	—
15	2.5	0.0	0.0	0.0	0.0	—
15	4.0	0.0	0.0	0.0	0.0	2.2653
16	1.5	4.28	0.136	0.0	0.288	—
16	2.5	0.0	0.0	0.0	0.0	—
16	4.0	0.0	0.0	0.0	0.0	4.6989
17	1.5	0.0	0.0	0.0	3.51	—
17	2.5	0.0	0.0	0.0	0.0	—
17	4.0	0.0	0.0	0.0	0.0	3.5090

TABLE 4-41

PHOSPHORUS LOADS
1983 CONDITIONS

Subbasin	Phosphorus Loadings (g/yr) Rangeland	Forest	Cropland	Urban	Total
1	1.97×10^6	7.56×10^4	2.50×10^6	1.26×10^5	4.68×10^6
2	5.32×10^5	6.18×10^4	1.17×10^5	4.94×10^5	1.20×10^6
3	2.80×10^5	0.00	1.33×10^4	1.13×10^4	3.04×10^5
4	1.87×10^5	0.0	1.12×10^5	7.76×10^4	3.76×10^5
5	6.63×10^4	1.72×10^4	0.0	1.36×10^5	2.20×10^5
6	7.78×10^4	5.72×10^3	0.0	0.0	8.35×10^4
7	1.72×10^5	0.0	0.0	0.0	1.72×10^5
8	3.78×10^5	0.0	0.0	5.75×10^5	9.53×10^5
9	5.41×10^5	0.0	1.22×10^4	1.58×10^5	7.11×10^5
10	1.70×10^5	2.05×10^3	4.88×10^4	6.05×10^4	2.81×10^5
11	1.78×10^5	5.16×10^3	1.57×10^5	2.77×10^4	3.68×10^5
12	6.71×10^4	0.0	0.0	3.95×10^3	7.10×10^4
13	2.26×10^5	1.02×10^4	2.37×10^4	2.25×10^5	4.84×10^5
14	5.19×10^4	2.21×10^3	0.0	4.10×10^4	9.51×10^4
15	5.36×10^4	8.35×10^2	0.0	6.57×10^4	1.20×10^5
16	1.28×10^5	2.71×10^3	0.0	4.32×10^4	1.74×10^5
17	0.0	0.0	0.0	2.11×10^6	2.11×10^6

TABLE 4-42

HISTORICAL NONPOINT SOURCE PHOSPHORUS LOADINGS FROM ADJACENT WATERSHEDS

	Annual Phosphorus Loadings (10^6 g/yr)						
Year	Lake Buchanan	Inks Lake	Lake LBJ	Lake Marble Falls	Lake Travis	Lake Austin	Town Lake
1983	5.88	0.30	2.56	0.71	1.20	0.22	2.28
1982	1.20	0.06	0.36	0.14	2.46	0.49	4.05
1981	4.12	0.21	1.23	0.48	8.27	1.65	13.56
1980	2.14	0.11	0.63	0.24	1.28	0.23	2.28
1979	2.14	0.11	0.62	0.24	5.53	1.09	8.70
1978	3.77	0.19	1.09	0.43	3.31	0.64	5.24
1977	4.30	0.22	1.24	0.50	1.14	0.21	1.88
1976	2.81	0.14	0.79	0.31	2.38	0.45	3.62
1975	3.27	0.17	0.91	0.36	5.06	0.97	7.34
1974	7.60	0.40	2.14	0.88	4.15	0.80	5.79
1973	1.86	0.10	0.51	0.21	4.35	0.83	5.98
1972	0.93	0.05	0.25	0.10	1.89	0.36	2.53
1971	2.32	0.12	0.62	0.25	0.88	0.15	1.34
1970	3.77	0.20	1.02	0.43	2.63	0.49	3.40
1969	4.83	0.25	1.28	0.53	2.60	0.46	3.48
1968	5.06	0.26	1.34	0.56	1.84	0.30	2.51
1967	3.53	0.18	0.94	0.40	3.04	0.55	3.76
1966	1.84	0.10	0.48	0.20	1.50	0.27	1.86
1965	2.98	0.16	0.77	0.33	6.05	1.12	6.92
1964	8.65	0.46	2.26	0.98	4.94	0.91	5.49
1963	0.92	0.05	0.23	0.10	1.06	0.19	1.18
1962	3.43	0.18	0.87	0.37	2.78	0.48	3.11

TABLE 4-42 (Concluded)

Year	Lake Buchanan	Inks Lake	Lake LBJ	Lake Marble Falls	Lake Travis	Lake Austin	Town Lake
	Annual Phosphorus Loadings (10^6 g/yr)						
1961	2.52	0.13	0.63	0.27	5.18	0.93	5.46
1960	4.66	0.24	1.18	0.52	3.17	0.56	3.26
1959	7.70	0.40	1.96	0.86	3.17	0.55	3.28
1958	5.34	0.28	1.35	0.59	7.10	1.29	6.84
1957	5.34	0.28	1.35	0.59	7.28	1.29	7.03
1956	0.91	0.05	0.23	0.10	0.73	0.12	0.72
1955	3.97	0.21	1.00	0.44	0.94	0.14	0.99
1954	0.92	0.05	0.22	0.09	0.21	0.03	0.21
1953	2.28	0.12	0.57	0.24	3.37	0.61	2.96
1952	5.93	0.31	1.49	0.65	1.34	0.21	1.28

TABLE 4-43

PROJECTED URBAN LAND USE AREAS, HIGHLAND LAKES WATERSHED

Subbasin No.	Description	Predominant County	Projected Urban Area (km^2) 1983	1990	2000
1	Colorado River near Tow	San Saba	7.7	7.7	8.2
2	Buchanan Dam	Burnet	29.6	37.9	49.9
3	Roy Inks Dam	Burnet	0.6	0.7	0.9
4	Colorado River above Llano River	Burnet	4.4	5.6	7.4
5	Llano River at mouth	Llano	7.2	9.1	10.5
6	Sandy Creek above Walnut Creek	Llano	0	0	0
7	Walnut Creek	Llano	0	0	0
8	Alvin Wirtz Dam	Llano	35.8	44.9	51.8
9	Max Starcke Dam	Burnet	9.1	11.6	15.3
10	Colorado River below Hamilton Creek	Burnet	8.3	10.7	14.1
11	Colorado River above Pedernales River	Burnet	4.5	5.7	7.5
12	Pedernales River at mouth	Travis	0.8	1.0	1.3
13	Mansfield Dam	Travis	43.1	54.1	70.5
14	Colorado River above Bull Creek	Travis	8.3	10.4	13.6
15	Tom Miller Dam	Travis	13.3	16.7	21.8
16	Barton Creek	Travis	8.6	10.9	14.2
17	Longhorn Dam	Travis	96.2	96.2	96.2

TABLE 4-44

PROJECTED NONPOINT SOURCE PHOSPHORUS LOADINGS FROM ADJACENT WATERSHEDS

Year	Phosphorus Loadings (10^6 g/yr)						
	Lake Buchanan	Inks Lake	Lake LBJ	Lake Marble Falls	Lake Travis	Lake Austin	Town Lake
1984	3.68	0.19	1.13	0.44	3.34	0.65	5.64
1985	5.27	0.27	1.64	0.63	2.06	0.38	3.72
1986	5.06	0.26	1.60	0.60	2.90	0.56	4.99
1987	3.94	0.20	1.26	0.48	2.92	0.59	4.74
1988	2.44	0.12	0.79	0.29	1.01	0.19	1.82
1989	0.98	0.05	0.32	0.12	2.09	0.43	3.31
1990	1.95	0.10	0.64	0.24	4.83	0.99	6.05
1991	7.96	0.40	2.63	1.00	4.61	0.96	7.16
1992	3.45	0.17	1.15	0.42	5.66	1.17	8.83
1993	2.96	0.15	0.99	0.36	2.68	0.55	4.23
1994	4.51	0.23	1.51	0.57	1.29	0.26	2.13
1995	3.97	0.20	1.34	0.49	3.72	0.78	5.75
1996	2.26	0.11	0.77	0.28	6.18	1.32	9.29
1997	2.25	0.11	0.77	0.28	1.46	0.29	2.36
1998	4.34	0.22	1.48	0.55	9.24	1.99	13.69
1999	1.27	0.06	0.44	0.16	2.75	0.59	4.09
2000	6.18	0.31	2.12	0.80	1.39	0.28	2.31

TABLE 4-45

HISTORIC ATMOSPHERIC PHOSPHORUS LOADS

HIGHLAND LAKES

Year	Annual Precipitation Llano (m/yr)	Annual Precipitation Austin (m/yr)	Phosphorus Loads L. Buchanan (g/yr)	Inks L. (g/yr)	L. LBJ (g/yr)	L. Marble Falls (g/yr)	L. Travis (g/yr)	L. Austin (g/yr)	Town L. (g/yr)
1983	.73	.86	1.22×10^6	4.74×10^4	3.77×10^5	4.61×10^4	1.08×10^6	1.27×10^5	2.89×10^4
1982	.56	.68	9.39×10^5	3.64×10^4	2.89×10^5	3.54×10^4	8.53×10^5	1.01×10^5	2.28×10^4
1981	.74	1.16	1.24×10^6	4.81×10^4	3.82×10^5	4.68×10^4	1.45×10^6	1.72×10^5	3.90×10^4
1980	.60	.70	1.01×10^6	3.90×10^4	3.10×10^5	3.79×10^4	8.78×10^5	1.04×10^5	2.35×10^4
1979	.70	.95	1.17×10^6	4.55×10^4	3.61×10^5	4.42×10^4	1.19×10^6	1.41×10^5	3.19×10^4
1978	.71	.79	1.19×10^6	4.62×10^4	3.66×10^5	4.49×10^4	9.91×10^5	1.17×10^5	2.65×10^4
1977	.62	.56	1.04×10^6	4.03×10^4	3.20×10^5	3.92×10^4	7.02×10^5	8.30×10^4	1.88×10^4
1976	.73	1.05	1.22×10^6	4.75×10^4	3.77×10^5	4.61×10^4	1.32×10^6	1.56×10^5	3.53×10^4
1975	.71	.93	1.19×10^6	4.62×10^4	3.66×10^5	4.49×10^4	1.17×10^6	1.38×10^5	3.12×10^4
1974	.87	.92	1.46×10^6	5.66×10^4	4.49×10^5	5.50×10^4	1.15×10^6	1.36×10^5	3.09×10^4
1973	.71	1.03	1.19×10^6	4.62×10^4	3.66×10^5	4.49×10^4	1.29×10^6	1.53×10^5	3.46×10^4
1972	.53	.66	8.88×10^5	3.45×10^4	2.73×10^5	3.35×10^4	8.28×10^5	9.78×10^4	2.22×10^4
1971	.72	.63	1.21×10^6	4.68×10^4	3.72×10^5	4.55×10^4	7.90×10^5	9.34×10^4	2.12×10^4
1970	.51	.78	8.55×10^5	3.32×10^4	2.63×10^5	3.22×10^4	9.78×10^5	1.16×10^5	2.62×10^4
1969	.89	.85	1.49×10^6	5.79×10^4	4.59×10^5	5.62×10^4	1.07×10^6	1.26×10^5	2.86×10^4
1968	.96	.98	1.61×10^6	6.24×10^4	4.95×10^5	6.07×10^4	1.23×10^6	1.45×10^5	3.29×10^4
1967	.62	.85	1.04×10^6	4.03×10^4	3.20×10^5	3.92×10^4	1.07×10^6	1.26×10^5	2.86×10^4
1966	.49	.64	8.21×10^5	3.19×10^4	2.53×10^5	3.10×10^4	8.03×10^5	9.48×10^4	2.15×10^4
1965	.73	1.03	1.22×10^6	4.75×10^4	3.77×10^5	4.61×10^4	1.29×10^8	1.53×10^5	3.46×10^4
1964	.76	.84	1.27×10^6	4.94×10^4	3.92×10^5	4.80×10^4	1.05×10^6	1.24×10^5	2.82×10^4
1963	.41	.44	6.87×10^5	2.67×10^4	2.12×10^5	2.59×10^4	5.52×10^5	6.52×10^4	1.48×10^4
1962	.64	.85	1.07×10^6	4.16×10^4	3.30×10^5	4.04×10^4	1.07×10^6	1.26×10^5	2.86×10^4

TABLE 4-45 (Concluded)

Year	Annual Precipitation Llano (m/yr)	Annual Precipitation Austin (m/yr)	Phosphorus Loads L. Buchanan (g/yr)	Inks L. (g/yr)	L. LBJ (g/yr)	L. Marble Falls (g/yr)	L. Travis (g/yr)	L. Austin (g/yr)	Town L. (g/yr)
1961	.55	.93	9.22×10^5	3.58×10^4	2.84×10^5	3.48×10^4	1.17×10^6	1.38×10^5	3.12×10^4
1960	.76	.91	1.27×10^6	4.94×10^4	3.92×10^5	4.80×10^4	1.14×10^6	1.35×10^5	3.06×10^4
1959	.98	.89	1.64×10^6	6.37×10^4	5.06×10^5	6.19×10^4	1.12×10^6	1.32×10^5	2.99×10^4
1958	.85	1.04	1.42×10^6	5.52×10^4	4.39×10^5	5.37×10^4	1.30×10^6	1.54×10^5	3.49×10^4
1957	.91	1.30	1.53×10^6	5.92×10^4	4.70×10^5	5.75×10^4	1.63×10^6	1.93×10^5	4.37×10^4
1956	.32	.39	5.36×10^5	2.08×10^4	1.65×10^5	2.02×10^4	4.89×10^5	5.78×10^4	1.31×10^4
1955	.61	.57	1.02×10^6	3.96×10^4	3.15×10^5	3.86×10^4	7.15×10^5	8.45×10^4	1.92×10^4
1954	.32	.29	5.36×10^5	2.08×10^4	1.65×10^5	2.02×10^4	3.64×10^5	4.30×10^4	9.74×10^3
1953	.54	.75	9.05×10^5	3.51×10^4	2.79×10^5	3.41×10^4	9.40×10^5	1.11×10^5	2.52×10^4
1952	1.05	.70	1.76×10^6	6.82×10^4	5.42×10^5	6.64×10^4	8.78×10^5	1.04×10^5	2.35×10^4

TABLE 4-46

PROJECTED ATMOSPHERIC PHOSPHORUS LOADS

HIGHLAND LAKES

Year	Phosphorus Loads (g/yr) L. Buchanan	Inks L.	L. LBJ	L. Marble Falls	L. Travis	L. Austin	Town L.
1984	1.04×10^6	4.03×10^4	3.20×10^5	3.92×10^4	1.07×10^6	1.26×10^5	2.86×10^4
1985	1.61×10^6	6.24×10^4	4.95×10^5	6.07×10^4	1.23×10^6	1.45×10^5	3.29×10^4
1986	1.49×10^6	5.79×10^4	4.59×10^5	5.62×10^4	1.07×10^6	1.26×10^5	2.86×10^4
1987	8.55×10^5	3.32×10^4	2.63×10^5	3.22×10^4	9.78×10^5	1.16×10^5	2.62×10^4
1988	1.21×10^6	4.68×10^4	3.72×10^5	4.55×10^4	7.90×10^5	9.34×10^4	2.12×10^4
1989	8.88×10^5	3.45×10^4	2.73×10^5	3.35×10^4	8.28×10^5	9.78×10^4	2.22×10^4
1990	1.19×10^6	4.62×10^4	3.66×10^5	4.49×10^4	1.29×10^6	1.53×10^5	3.46×10^4
1991	1.46×10^6	5.66×10^4	4.49×10^5	5.50×10^4	1.15×10^6	1.36×10^5	3.09×10^4
1992	1.19×10^6	4.62×10^4	3.66×10^5	4.49×10^4	1.17×10^6	1.38×10^5	3.12×10^4
1993	1.22×10^6	4.75×10^4	3.77×10^5	4.61×10^4	1.32×10^6	1.56×10^5	3.53×10^4
1994	1.04×10^6	4.03×10^4	3.20×10^5	3.92×10^4	7.02×10^5	8.30×10^4	1.88×10^4
1995	1.19×10^6	4.62×10^4	3.66×10^5	4.49×10^4	9.91×10^5	1.17×10^5	2.65×10^4
1996	1.17×10^6	4.55×10^4	3.61×10^5	4.42×10^4	1.19×10^6	1.41×10^5	3.19×10^4
1997	1.01×10^6	3.90×10^4	3.10×10^5	3.79×10^4	8.78×10^5	1.04×10^5	2.35×10^4
1998	1.24×10^6	4.81×10^4	3.82×10^5	4.68×10^4	1.45×10^6	1.72×10^5	3.90×10^4
1999	9.39×10^5	3.64×10^4	2.89×10^5	3.54×10^4	8.53×10^5	1.01×10^5	2.28×10^4
2000	1.22×10^6	4.74×10^4	3.77×10^5	4.61×10^4	1.08×10^6	1.27×10^5	2.89×10^4

TABLE 4-47

HISTORICAL BARTON SPRINGS PHOSPHORUS LOADS
TO TOWN LAKE

Year	Spring Flow (10^6 m^3/yr)	Phosphorus Load (10^6 g/yr)	Year	Spring Flow (10^6 m^3/yr)	Phosphorus Load (10^6 g/yr)
1983	59.7	1.07	1967	32.0	0.58
1982	48.0	0.86	1966	59.0	1.06
1981	66.7	1.20	1965	76.0	1.37
1980	41.9	0.75	1964	20.0	0.36
1979	72.5	1.31	1963	38.0	0.68
1978	31.0	0.56	1962	48.0	0.86
1977	88.0	1.58	1961	101.0	1.82
1976	86.0	1.55	1960	65.0	1.17
1975	97.0	1.75	1959	60.0	1.08
1974	84.0	1.51	1958	82.0	1.48
1973	82.0	1.48	1957	46.0	0.83
1972	85.0	1.53	1956	13.0	0.23
1971	45.0	0.81	1955	17.0	0.31
1970	84.0	1.51	1954	29.0	0.52
1969	61.0	1.10	1953	45.0	0.81
1968	81.0	1.46	1952	25.0	0.45

TABLE 4-48

SUMMARY OF HISTORICAL PHOSPHORUS LOADINGS TO THE HIGHLAND LAKES

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE BUCHANAN -		
1983	0.012	5.88	15.62	1.22
1982	0.015	1.20	140.67	0.94
1981	0.025	4.12	71.38	1.24
1980	0.014	2.14	195.84	1.01
1979	0.015	2.14	34.01	1.17
1978	0.036	3.77	75.29	1.19
1977	0.022	4.30	132.34	1.04
1976	0.019	2.81	56.31	1.22
1975	0.035	3.27	119.47	1.19
1974	—	7.60	286.19	1.46
1973	—	1.86	121.64	1.19
1972	—	0.93	22.88	0.89
1971	—	2.32	296.73	1.21
1970	—	3.77	103.04	0.86
1969	—	4.83	145.01	1.49
1968	—	5.06	353.15	1.61
1967	—	3.53	59.76	1.04
1966	—	1.84	82.72	0.82
1965	—	2.98	313.79	1.22
1964	—	8.65	164.98	1.27
1963	—	0.92	50.27	0.69
1962	—	3.43	72.80	1.07
1961	—	2.52	208.92	0.92
1960	—	4.66	86.31	1.27
1959	—	7.70	434.57	1.64
1958	—	5.34	115.49	1.42
1957	—	5.34	1644.89	1.53
1956	—	0.91	302.39	0.54
1955	—	3.97	595.24	1.02
1954	—	0.92	173.74	0.54
1953	—	2.28	160.08	0.90
1952	—	5.93	222.37	1.76

TABLE 4-48 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- INKS LAKE -		
1983	—	0.30	—	0.047
1982	—	0.06	—	0.036
1981	—	0.21	—	0.048
1980	—	0.11	—	0.039
1979	—	0.11	—	0.046
1978	—	0.19	—	0.046
1977	—	0.22	—	0.040
1976	—	0.14	—	0.048
1975	—	0.17	—	0.046
1974	—	0.40	—	0.057
1973	—	0.10	—	0.046
1972	—	0.05	—	0.034
1971	—	0.12	—	0.047
1970	—	0.20	—	0.033
1969	—	0.25	—	0.058
1968	—	0.26	—	0.062
1967	—	0.18	—	0.040
1966	—	0.10	—	0.032
1965	—	0.16	—	0.048
1964	—	0.46	—	0.049
1963	—	0.05	—	0.027
1962	—	0.18	—	0.042
1961	—	0.13	—	0.036
1960	—	0.24	—	0.049
1959	—	0.40	—	0.064
1958	—	0.28	—	0.055
1957	—	0.28	—	0.059
1956	—	0.05	—	0.021
1955	—	0.21	—	0.040
1954	—	0.05	—	0.021
1953	—	0.12	—	0.035
1952	—	0.31	—	0.068

TABLE 4-48 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr)			
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE LYNDON B. JOHNSON -		
1983	1.46	2.56	16.47	0.38
1982	2.77	0.36	12.60	0.29
1981	1.38	1.23	20.42	0.38
1980	0.002	0.63	21.90	0.31
1979	0.0004	0.62	6.25	0.36
1978	0.0004	1.09	7.62	0.37
1977	0.0004	1.24	10.42	0.32
1976	0.0004	0.79	15.98	0.38
1975	0.0004	0.91	4.63	0.37
1974	0.0004	2.14	4.67	0.45
1973	—	0.51	20.41	0.37
1972	—	0.25	14.62	0.27
1971	—	0.62	14.83	0.37
1970	—	1.02	18.49	0.26
1969	—	1.28	41.69	0.46
1968	—	1.34	3.74	0.50
1967	—	0.94	11.68	0.32
1966	—	0.48	0.99	0.25
1965	—	0.77	3.37	0.38
1964	—	2.26	20.13	0.39
1963	—	0.23	1.49	0.21
1962	—	0.87	3.22	0.33
1961	—	0.63	7.62	0.28
1960	—	1.18	19.07	0.39
1959	—	1.96	4.78	0.51
1958	—	1.35	5.91	0.44
1957	—	1.35	5.43	0.47
1956	—	0.23	7.31	0.16
1955	—	1.00	7.96	0.32
1954	—	0.22	10.33	0.16
1953	—	0.57	15.20	0.28
1952	—	1.49	10.80	0.54

TABLE 4-48 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE MARBLE FALLS -		
1983	4.90	0.71	—	0.046
1982	5.16	0.14	—	0.035
1981	4.63	0.48	—	0.047
1980	4.13	0.24	—	0.038
1979	3.81	0.24	—	0.044
1978	2.53	0.43	—	0.045
1977	2.42	0.50	—	0.039
1976	1.98	0.31	—	0.046
1975	1.10	0.36	—	0.045
1974	1.10	0.88	—	0.055
1973	1.10	0.21	—	0.045
1972	1.10	0.10	—	0.034
1971	0.95	0.25	—	0.046
1970	0.95	0.43	—	0.032
1969	0.95	0.53	—	0.056
1968	0.95	0.56	—	0.061
1967	0.95	0.40	—	0.039
1966	0.95	0.20	—	0.031
1965	0.95	0.33	—	0.046
1964	0.95	0.98	—	0.048
1963	0.95	0.10	—	0.026
1962	—	0.37	—	0.040
1961	—	0.27	—	0.035
1960	—	0.52	—	0.048
1959	—	0.86	—	0.062
1958	—	0.59	—	0.054
1957	—	0.59	—	0.058
1956	—	0.10	—	0.020
1955	—	0.44	—	0.039
1954	—	0.09	—	0.020
1953	—	0.24	—	0.034
1952	—	0.65	—	0.066

TABLE 4-48 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE TRAVIS -		
1983	0.17	1.20	5.97	1.08
1982	0.13	2.46	2.68	0.85
1981	0.18	8.27	19.75	1.45
1980	0.14	1.28	2.57	0.88
1979	0.077	5.53	20.24	1.19
1978	–	3.31	17.89	0.99
1977	–	1.14	19.43	0.70
1976	–	2.38	9.36	1.32
1975	–	5.06	23.59	1.17
1974	–	4.15	20.85	1.15
1973	–	4.35	11.15	1.29
1972	–	1.89	6.19	0.83
1971	–	0.88	7.22	0.79
1970	–	2.63	12.69	0.98
1969	–	2.60	11.09	1.07
1968	–	1.84	15.07	1.23
1967	–	3.04	3.08	1.07
1966	–	1.50	2.44	0.80
1965	–	6.05	12.53	1.29
1964	–	4.94	2.35	1.05
1963	–	1.06	1.29	0.55
1962	–	2.78	3.02	1.07
1961	–	5.18	9.17	1.17
1960	–	3.17	10.83	1.14
1959	–	3.17	22.46	1.12
1958	–	7.10	18.53	1.30
1957	–	7.28	20.55	1.63
1956	–	0.73	0.31	0.49
1955	–	0.94	3.67	0.72
1954	–	0.21	0.79	0.36
1953	–	3.37	2.31	0.94
1952	–	1.34	51.47	0.88

TABLE 4-48 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr)			
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE AUSTIN -		
1983	—	0.22	—	0.13
1982	—	0.49	—	0.10
1981	—	1.65	—	0.17
1980	—	0.23	—	0.10
1979	—	1.09	—	0.14
1978	—	0.64	—	0.12
1977	—	0.21	—	0.08
1976	—	0.45	—	0.16
1975	—	0.97	—	0.14
1974	—	0.80	—	0.14
1973	—	0.83	—	0.15
1972	—	0.36	—	0.10
1971	—	0.15	—	0.09
1970	—	0.49	—	0.12
1969	—	0.46	—	0.13
1968	—	0.30	—	0.14
1967	—	0.55	—	0.13
1966	—	0.27	—	0.09
1965	—	1.12	—	0.15
1964	—	0.91	—	0.12
1963	—	0.19	—	0.07
1962	—	0.48	—	0.13
1961	—	0.93	—	0.14
1960	—	0.56	—	0.14
1959	—	0.55	—	0.13
1958	—	1.29	—	0.15
1957	—	1.29	—	0.19
1956	—	0.12	—	0.06
1955	—	0.14	—	0.08
1954	—	0.03	—	0.04
1953	—	0.61	—	0.11
1952	—	0.21	—	0.10

TABLE 4-48 (Concluded)

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- TOWN LAKE -		
1983	—	2.28	1.07	0.029
1982	—	4.05	0.86	0.023
1981	—	13.56	1.20	0.039
1980	—	2.28	0.75	0.024
1979	—	8.70	1.31	0.032
1978	—	5.24	0.56	0.026
1977	—	1.88	1.58	0.019
1976	—	3.62	1.55	0.035
1975	—	7.34	1.75	0.031
1974	—	5.79	1.51	0.031
1973	—	5.98	1.48	0.035
1972	—	2.53	1.53	0.022
1971	—	1.34	0.81	0.021
1970	—	3.40	1.51	0.026
1969	—	3.48	1.10	0.029
1968	—	2.51	1.46	0.033
1967	—	3.76	0.58	0.029
1966	—	1.86	1.06	0.022
1965	—	6.92	1.37	0.035
1964	—	5.49	0.36	0.028
1963	—	1.18	0.68	0.015
1962	—	3.11	0.86	0.029
1961	—	5.46	1.82	0.031
1960	—	3.26	1.17	0.031
1959	—	3.28	1.08	0.030
1958	—	6.84	1.48	0.035
1957	—	7.03	0.83	0.044
1956	—	0.72	0.23	0.013
1955	—	0.99	0.31	0.019
1954	—	0.21	0.52	0.010
1953	—	2.96	0.81	0.025
1952	—	1.28	0.45	0.024

TABLE 4-49

SUMMARY OF PROJECTED PHOSPHORUS LOADINGS
TO THE HIGHLAND LAKES

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE BUCHANAN -		
1984	0.013	3.68	50.73	1.04
1985	0.013	5.27	144.64	1.61
1986	0.014	5.06	145.01	1.49
1987	0.014	3.94	103.04	0.86
1988	0.015	2.44	296.73	1.21
1989	0.015	0.98	22.88	0.89
1990	0.016	1.95	121.64	1.19
1991	0.016	7.96	286.19	1.46
1992	0.017	3.45	119.47	1.19
1993	0.017	2.96	56.31	1.22
1994	0.018	4.51	132.34	1.04
1995	0.018	3.97	75.29	1.19
1996	0.019	2.26	34.01	1.17
1997	0.019	2.25	195.84	1.01
1998	0.020	4.34	71.38	1.24
1999	0.020	1.27	140.67	0.94
2000	0.021	6.18	15.62	1.22
		- INKS LAKE -		
1984	—	0.19	—	0.040
1985	—	0.27	—	0.062
1986	—	0.26	—	0.058
1987	—	0.20	—	0.033
1988	—	0.12	—	0.047
1989	—	0.05	—	0.034
1990	—	0.10	—	0.046
1991	—	0.40	—	0.057
1992	—	0.17	—	0.046
1993	—	0.15	—	0.048
1994	—	0.23	—	0.040
1995	—	0.20	—	0.046
1996	—	0.11	—	0.046
1997	—	0.11	—	0.039
1998	—	0.22	—	0.048
1999	—	0.06	—	0.036
2000	—	0.31	—	0.047

TABLE 4-49 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE LYNDON B. JOHNSON -		
1984	1.51	1.13	11.68	0.32
1985	1.56	1.64	3.74	0.50
1986	1.62	1.60	41.69	0.46
1987	1.67	1.26	18.49	0.26
1988	1.73	0.79	14.83	0.37
1989	1.78	0.32	14.62	0.27
1990	1.84	0.64	20.41	0.37
1991	1.87	2.63	4.67	0.45
1992	1.90	1.15	4.63	0.37
1993	1.93	0.99	15.98	0.38
1994	1.96	1.51	10.42	0.32
1995	1.98	1.34	7.62	0.37
1996	2.01	0.77	6.25	0.36
1997	2.04	0.77	21.9	0.31
1998	2.07	1.48	20.42	0.38
1999	2.10	0.44	12.6	0.29
2000	2.13	2.12	16.47	0.38
		- LAKE MARBLE FALLS -		
1984	5.07	0.44	—	0.039
1985	5.24	0.63	—	0.061
1986	5.45	0.60	—	0.056
1987	5.65	0.48	—	0.032
1988	5.86	0.29	—	0.046
1989	6.07	0.12	—	0.034
1990	6.27	0.24	—	0.045
1991	6.46	1.00	—	0.055
1992	6.64	0.42	—	0.045
1993	6.83	0.36	—	0.046
1994	7.01	0.57	—	0.039
1995	7.20	0.49	—	0.045
1996	7.41	0.28	—	0.044
1997	7.62	0.28	—	0.038
1998	7.83	0.55	—	0.047
1999	8.05	0.16	—	0.035
2000	8.26	0.80	—	0.046

TABLE 4-49 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr)			
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE TRAVIS -		
1984	0.17	3.34	3.08	1.07
1985	0.17	2.06	15.07	1.23
1986	0.17	2.90	11.09	1.07
1987	0.19	2.92	12.69	0.98
1988	0.24	1.01	7.22	0.79
1989	0.28	2.09	6.19	0.83
1990	0.33	4.83	11.15	1.29
1991	0.33	4.61	20.85	1.15
1992	0.33	5.66	23.59	1.17
1993	0.33	2.68	9.36	1.32
1994	0.33	1.29	19.43	0.70
1995	0.33	3.72	17.89	0.99
1996	0.33	6.18	20.24	1.19
1997	0.33	1.46	2.57	0.88
1998	0.33	9.24	19.75	1.45
1999	0.33	2.75	2.68	0.85
2000	0.33	1.39	5.09	1.08
		- LAKE AUSTIN -		
1984	—	0.65	—	0.126
1985	—	0.38	—	0.145
1986	—	0.56	—	0.126
1987	—	0.59	—	0.116
1988	—	0.19	—	0.093
1989	—	0.43	—	0.098
1990	—	0.99	—	0.153
1991	—	0.96	—	0.136
1992	—	1.17	—	0.138
1993	—	0.55	—	0.156
1994	—	0.26	—	0.083
1995	—	0.78	—	0.117
1996	—	1.32	—	0.141
1997	—	0.29	—	0.104
1998	—	1.99	—	0.172
1999	—	0.59	—	0.101
2000	—	0.28	—	0.127

TABLE 4-49 (Concluded)

Year	Phosphorus Loadings (10^6 g/yr)			
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- TOWN LAKE -		
1984	—	5.64	0.58	0.029
1985	—	3.72	1.46	0.033
1986	—	4.99	1.10	0.029
1987	—	4.74	1.51	0.026
1988	—	1.82	0.81	0.021
1989	—	3.31	1.53	0.022
1990	—	6.05	1.48	0.035
1991	—	7.16	1.51	0.031
1992	—	8.83	1.75	0.031
1993	—	4.23	1.55	0.035
1994	—	2.13	1.58	0.019
1995	—	5.75	0.56	0.026
1996	—	9.29	1.31	0.032
1997	—	2.36	0.75	0.024
1998	—	13.69	1.20	0.039
1999	—	4.09	0.86	0.023
2000	—	2.31	1.07	0.029

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EVALUATION OF THE EFFECTS OF POINT AND NONPOINT SOURCE PHOSPHORUS LOADS UPON WATER QUALITY IN THE HIGHLAND LAKES

Part II

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CHAPTER 5.0

MODEL DEVELOPMENT

For the present study, the general objective was to develop a model for the Highland Lakes system that can be utilized to project long-term trends in water quality under alternative developmental (i.e., loading) scenarios. Alternative approaches to the modeling of water quality in reservoirs were described in Chapter 2.0. Particular attention was devoted to the modeling of phosphorus, as it is frequently a primary variable in the eutrophication process. Closure of a phosphorus budget is more easily attained than a nitrogen or a carbon budget, since exchange of a gaseous phase with the atmosphere need not be considered. To satisfy the study objective, the model formulation considers the chain of reservoirs as a series of completely mixed reactors. A phosphorus budget is constructed for each reactor, which accounts for inflow and outflow from the reactor, with first-order internal kinetics.

The concept of simulation of a natural system as a completely mixed reactor, also referred to as a continuously-stirred tank reactor (CSTR), is a practicable approach to water quality modeling. Chemical reactors and processing units have long been analyzed with CSTR principles, and the concept has thus permeated the environmental engineering field. The principles are straightforward: a constituent introduced into the reactor is instantaneously dispersed, effecting a higher concentration within the contents of the vessel; the constituent mass within the vessel may be reduced by either internal reactions or external outflow, with the constituent concentration uniform in the vessel and outflow. The CSTR concept has been successfully employed in the analysis of natural water systems. As described in Chapter 2.0, O'Connor and Mueller (1970) used a CSTR model for simulation of chloride levels in the Great Lakes. Of particular interest, successful applications have been achieved with phosphorus budget analyses. For example, Chapra (1977) developed a CSTR model for simulation of historical and future phosphorus concentrations in the Great Lakes, and similar models have been applied to numerous other waterbodies (see Chapter 2.0). At the simplest level, the Vollenweider-type approach (Vollenweider, 1968, 1975) incorporates the complete-mix assumption at steady state in development of a graphical analysis of trophic state; other CSTR models examine time-variant water quality behavior. The CSTR model is an appropriate water quality management tool to support the objectives of the study, namely, for use in analyzing the long-term annual average phosphorus concentrations within the series of reservoirs, where mass inputs are expressed as annual loadings from point and nonpoint sources. A more complex model is not warranted for the present investigation because of severe limitations in the available water quality data base.

5.1 CONCEPTUAL FRAMEWORK

The development of a mathematical model is initiated with the concept of a material mass budget on a control volume, wherein sources, sinks, and transformations of mass are evaluated. A schematic diagram of a simple completely mixed system is presented in Fig. 5-1. A mass budget for the system may be expressed in a general form as

$$V\frac{dC}{dt} = W(t) - QC \pm S_i \tag{1}$$

where V = volume, L^3 (assuming $\frac{dV}{dt} = 0$)
C = material concentration, ML^{-3}
t = time, T
Q = throughflow, L^3T^{-1}
$W(t)$ = mass inflow, MT^{-1}
S_i = internal source/sink term, MT^{-1}

As a reasonable approximation, it is assumed that the source/sink term may be represented by first-order kinetics, so that

$$V\frac{dC}{dt} = W(t) - QC - kCV \tag{2}$$

where k = decay rate, T^{-1}. In terms of phosphorus dynamics, the decay term represents loss of phosphorus from the water column.

Eqn (2) may be rewritten as

$$\frac{dC}{dt} = \frac{W(t)}{V} - \frac{C}{\tau} - kC \tag{3}$$

where τ = hydraulic residence time $(\frac{V}{Q})$, T

To this basic framework of a phosphorus budget, certain refinements can be added. Refinements typically address internal phosphorus mechanisms. The first-order decay term may be expressed as k_1A_sC, where k_1 = the effective settling velocity, LT^{-1}, and A_s = the surface area of sediments. In addition to this transfer of mass to the sediments, release of phosphorus from the sediments can be considered as $k_2A_sC_s$, where k_2 = reaction coefficient, LT^{-1}, and C_s = concentration of exchangeable phosphorus in the sediments. With sediment transfer and release terms, the system equation can be written as

$$V\frac{dC}{dt} = W(t) - CQ + k_2A_sC_s - k_1A_sC \tag{4}$$

A further refinement would entail the development of a mass budget on the sediment system:

$$V_s \frac{dC_s}{dt} = k_1 A_s C - k_2 A_s C_s - k_1 k_3 A_s C \tag{5}$$

where V_s = volume of the sediment reservoir, L^3
k_3 = coefficient to account for the fraction of phosphorus input unavailable for exchange (dimensionless)

Alternate formulations may be developed to consider additional physical components, for example, epilimnion and hypolimnion compartments, and exchange across the thermocline. A variety of biological processes, including plankton uptake and production, or bacterially-mediated conversions, may also be included. Increases in model complexity, however, are accompanied by increasingly complex solutions and more extensive data requirements. To a certain extent, complexity is a burden sustained more with analytical solution techniques, since numerical solution techniques are facilitated with computer modeling. A more serious limitation is typically the need for an extensive input data set.

To demonstrate an analytical solution for the case of constant mass inflow, replace W(t) in eqn (2) with QC_o and rearrange to obtain

$$\frac{dC}{dt} + aC = \frac{Q}{V} C_o \tag{6}$$

where $a = \frac{Q}{V} + k$, T^{-1}

C_o = inflow concentration, ML^{-3}

The term 1/a represents the time constant of the system, a measure of the response time of the system to a change in input concentration. The ratio Q/V, referred to as the dilution rate or flushing rate of the system, expresses the number of volume replacements by throughflow occurring in the system per unit time. The retention time is defined as V/Q, which describes the idealized time that the fluid is retained in the system. An integrating factor can be used to solve the nonhomogeneous eqn (6). The equation is multiplied by e^{at}:

$$(C' + aC)\, e^{at} = \frac{Q}{V} C_o\, e^{at}$$

The left side may be written as a differential:

$$(C\,e^{at})' = \frac{Q}{V} C_o\, e^{at}$$

Integrate to obtain (C_o is constant)

$$C = \frac{Q}{V}\,\frac{C_o}{a} + Ae^{-at}$$

For initial conditions, assume that $C = C_i$ at $t = 0$. Thus

$$A = C_i - \frac{Q}{V}\,\frac{C_o}{a}$$

Substitution of A yields

$$C = \frac{Q}{V}\,\frac{C_o}{a}\,(1 - e^{-at}) + C_i e^{-at} \qquad (7)$$

This is the nonsteady-state analytical solution for the case of constant mass input into the system.

For a steady-state solution, $t \rightarrow \infty$, such that

$$C = \frac{Q}{V}\,\frac{C_o}{a} \qquad (8)$$

which can be rewritten as

$$C = \frac{Q}{V}\,\frac{C_o}{(k+Q/V)} = \frac{C_o}{1+k(V/Q)} = \frac{C_o}{1+k\tau} \qquad (9)$$

The general solution of eqn (6) is

$$C = \frac{Q}{V}\, e^{-at} \int_o^t C_o(t) e^{at}\, dt + C_i e^{-at} \qquad (10)$$

where $C_i = C$ at $t = 0$

Concentrations for cases of constant, linear, and exponential input functions are presented in Table 5-1 (after O'Connor and Mueller, 1970).

Solutions may also be developed for a series of completely mixed reactors. Consider, for example, the second reactor of a hypothetical series. The concentration of a particular constituent within this reactor is governed by four major influences: inflow of mass from the first reactor, influx of mass from other sources, outflow of mass, and internal reactions. In the analogy with a chain of reservoirs, mass loadings to the second reservoir originate from the upstream reservoir, as well as tributaries and runoff directly into the reservoir. We begin with the general solution written for the second reactor:

$$C_2 = \frac{e^{-a_2 t}}{V_2} \int_o^t W(t)\, e^{a_2 t}\, dt + C_{2i}\, e^{-a_2 t} \tag{11}$$

The initial concentration in the reactor is represented by C_{2i}. W(t) represents the load from the first reactor, $W_{1,2}(t)$, plus other external loads into the second reactor, $W_2(t)$.

To explore a solution, consider the case of a constant mass input into the first reactor, so that (from eqn (7))

$$W_{1,2}(t) = Q_1 \left[\frac{C_o Q_1}{a_1 V_1} (1-e^{-a_1 t}) + C_{1i} e^{-a_1 t} \right] \tag{12}$$

Assume there is an additional mass load into the second reactor, constant in time:

$$W_2(t) = Q_R C_R \tag{13}$$

Now, the integral takes the following form:

$$\int_o^t W_{1,2}(t) e^{a_2 t} dt + \int_o^t W_2(t) e^{a_2 t} dt \tag{14}$$

The contribution from the first reactor, $W_{1,2}(t)$, is obtained as follows:

$$\frac{e^{-a_2 t}}{V_2} \int_o^t Q_1 \left[\left(\frac{C_o Q_1}{a_1 V_1} (1-e^{-a_1 t}) + C_{1i} e^{-a_1 t} \right) \right] e^{a_2 t} dt$$

$$= \frac{C_o Q_1^2}{a_1 a_2 V_1 V_2} \left[(1-e^{-a_2 t}) + \frac{a_2 (a_1 C_{1i} V_1 - C_o Q_1)}{C_o Q_1 (a_2 - a_1)} (e^{-a_1 t} - e^{-a_2 t}) \right] \tag{15}$$

The effect of the additional mass load, $W_2(t)$, is

$$\frac{e^{-a_2 t}}{V_2} \int_o^t Q_R C_R \, e^{a_2 t} \, dt$$

$$= \frac{C_R \, Q_R}{V_2 \, a_2} \left(1 - e^{-a_2 t}\right) \qquad (16)$$

Decay of the initial mass present in the second reactor may be represented by

$$C_{2i} e^{-a_2 t} \qquad (17)$$

Then by superposition, the concentration in the second reactor, with the assumptions noted above, is obtained by combination of terms (15), (16), and (17). In a similar manner, the solution for any reactor in a series can be developed.

5.2 FORMULATION OF MODEL HILAKES

The computer program HILAKES was developed to compute phosphorus concentration in individual reservoirs within a series of reservoirs in response to various mass inputs. A flow chart for the program is displayed in Fig. 5-2, and a complete listing of the code is presented in Appendix D. The system differential equation describes a mass budget on phosphorus for each reservoir, which accounts for phosphorus inputs, output, and internal reactions. Phosphorus concentration is computed by solution of the system differential equation using a fourth-order Runge-Kutta integration technique (which is basically an iterative process).

The computer code is structured to analyze conditions in a series of seven reactors or reservoirs. Required input data sets describe mass loadings, physical data, and rate coefficients for each reservoir. In addition, multiple input data sets for individual reservoirs can be accommodated, which facilitates simulation of time-varying input conditions. Within the core of the program, the terms of the system differential equation are defined for each time step, after which the terms of the recursion equation of the integration technique are defined. Within each time step, the solution block is engaged for all reservoirs, and concentrations are computed under the specified loading conditions.

5.2.1 System Equation

The phosphorus budget concepts described in the preceding section were used to develop the water quality model for the Highland Lakes system. The chain of reservoirs was treated as a series of completely mixed reactors. For each reactor, phosphorus inflow and outflow, and first-order internal reaction kinetics were considered. The general system equation for reactor n is

$$V_n \frac{dC_n}{dt} = W_n(t) - QC_n - kV_nC_n \tag{18}$$

where $W_n(t)$ = mass load (MT^{-1}) from preceding reactor n-1 and from direct inputs to reactor n, or,

$$W_n(t) = Q_{n-1} C_{n-1} + \sum_{j=1}^{m} W_j \tag{19}$$

where m is the total number of separate loads into lake n.

The assumptions inherent in the modeling analysis are appropriate for strategic management-level decisions, particularly for projection of long-term water quality trends. For the present application, the model was employed to project long-term phosphorus concentrations in the reservoirs. Mass inputs, described in Chapter 4.0, were structured as annual average loadings of phosphorus. (In application, the annual average values may be subdivided into smaller time elements, in order to improve the accuracy of the numerical method.) In turn, the appropriate output from the model consists of projected annual average constituent concentrations within each reservoir.

While simple in concept, the modeling approach is unique in that the mass budget for each reservoir is coupled to adjacent reservoirs, such that no reservoir is analyzed as a single entity. This approach avoids a shortcoming common to most previous studies of water quality in the Highland Lakes, which evaluated water quality aspects in a single reservoir without due consideration of the interrelationships extant within the chain.

5.2.2 Mass Input

Emphasis in the analysis was upon external mass loadings of total phosphorus. Prediction of long-term trends involves the simulation of annual input loads and reservoir response. Mass inputs for the long-term analysis were described in Chapter 4.0. Input loads were determined for point and nonpoint sources in the watershed of each reservoir. The nonpoint source load consisted of contributions from major tributaries, runoff from immediate drainage areas, and atmospheric input. Runoff from immediate drainage areas was broken out into land use categories of urban, rangeland, forest, and cropland. Annual phosphorus loadings from the various sources were summarized in Tables 4-48 and 4-49.

A key phosphorus input may be associated with releases from an upstream reservoir. These loads cannot be adequately defined with historical water quality data, since it is virtually nonexistent. The phosphorus content of reservoir releases will be described within the context of the mathematical modeling analysis. Releases from a reservoir will be assumed to have the same phosphorus level as the lower pool of that reservoir, which is assumed to be completely mixed. In actuality, the lower pools of the reservoirs are thermally stratified during warm-weather periods, and releases from the dams typically originate from the hypolimnion. Since concentrations of phosphorus are likely to be elevated in the hypolimnion (as discussed in Sec. 3.2.3), the actual mass loads exiting the reservoir in the summer are probably greater in magnitude than those calculated with the present methodology (see also Sec. 5.5.3). However, the assumption of completely mixed conditions should be appropriate for the evaluation of long-term trends in average annual concentrations, with mass input in terms of annual loads.

5.2.3 Internal Reactions

Within the context of the present analysis, the internal reaction coefficient, k, represents the net rate of loss of phosphorus from the water column within the reservoir. For particulate phosphorus, sedimentation may occur through direct deposition through gravity settling. Alternatively, removal of phosphorus may occur via biotic uptake by plankton, with subsequent settling. Phosphorus may be released from the sediments, principally under anaerobic conditions. Resuspension of sediment-bound phosphorus may also be induced by turbulence. For the present analysis, the various dynamic processes which affect the phosphorus levels in the water column are incorporated within the kinetic term k. The rate k characterizes a first-order reaction, that is, the rate of removal of phosphorus from the water column is directly proportional to the amount of phosphorus present.

Reported first-order reaction rates for phosphorus have varied widely. Depending upon the formulation of the internal decay term, the rate coefficient can assume different forms. In the present analysis, the term is formulated as

$$-kV_nC_n$$

in eqn (18), such that k is in terms of a time rate of change (T^{-1}). The rate k is also referred to as the sedimentation coefficient, σ. Alternatively, the term can be expressed as

$$-k_1A_sC_n$$

as shown in eqn (4), where k_1 represents mass transport (or settling velocity, v_s) in unit length per unit of time (LT^{-1}). The second formulation is based upon the assumption of a constant settling velocity and incorporates the characteristic depth of a reservoir into the rate term. The first formulation was adopted for the present analysis in order to remove the intrinsic relationship with depth, although the settling velocity is thus depth-dependent. In this way, a reaction rate can theoretically be selected on physical principles and subsequently applied to a reservoir of any depth.

Historically, evaluation of the phosphorus reaction rate (Vollenweider, 1975, 1976; Chapra, 1975) has frequently been based upon the steady-state solution displayed in eqn (8), which can be rewritten as

$$C = \frac{L}{z(1/\tau + \sigma)}$$

or

$$C = \frac{L}{z/\tau + v_s}$$

where
- L = areal phosphorus loading rate, $ML^{-2}T^{-1}$
- z = mean depth, L
- τ = hydraulic retention time, T
- σ = phosphorus sedimentation coefficient, T^{-1}
- v_s = apparent settling velocity ($=z\sigma$), MT^{-1}
- C = steady-state concentration, ML^{-3}

An alternate formulation (Dillon and Rigler, 1974; Chapra, 1975) incorporates a phosphorus retention coefficient:

$$C = \frac{L\,(1-R)}{z/\tau}$$

where

$$R = \frac{\sigma}{(1/\tau + \sigma)} = \frac{v_s}{(q_s + v_s)}$$

and
- R = phosphorus retention coefficient, dimensionless
- q_s = overflow rate (= Q/A), LT^{-1}

Vollenweider (1975) determined empirically that the sedimentation coefficient (yr^{-1}) was a function of depth (m):

$$\sigma = 10/z$$

This formulation implies a constant settling velocity, $v_s = 10$ m/yr. Chapra (1975) examined data from Canadian lakes and obtained an apparent settling velocity v_s of 16 m/yr. Based upon data for 51 natural lakes, Jones and Bachmann (1976) estimated σ at 0.65 yr^{-1}. Canfield and Bachmann (1981) examined data for 704 lakes and reservoirs, resulting in estimates of the phosphorus sedimentation coefficient of

$$\sigma = 0.162\ (L/z)^{0.458} \text{ for natural lakes}$$
$$\sigma = 0.114\ (L/z)^{0.589} \text{ for reservoirs}$$

where L represents the areal phosphorus loading rate (mg/m^2/yr). Vollenweider's formulation (1975) was applied to ten Tennessee Valley Authority reservoirs with positive phosphorus retention to obtain

$$\sigma = 92/z$$

which implies an apparent settling velocity of 92 m/yr (Higgins and Kim, 1981). The high settling velocity was attributed to the pronounced surface extension of the reservoirs.

Examples of phosphorus reaction rates reported in the literature are presented in Table 5-2. The tabulated values for sedimentation coefficient and apparent settling velocity were either reported directly or calculated from reported data in the references cited. Substantial variation is evident in the reported rates, since they are generally based upon individual phosphorus budget analyses.

For the present analysis, examination of the mass budget of a single reservoir could potentially provide definition of an internal reaction rate, which could then be applied to other reservoirs characterized by more limited data bases. Decay coefficients can be estimated based upon the assumed steady-state concentration of total phosphorus within a waterbody. The steady-state solution to the system equation was shown in eqn (8). To apply this method, a steady-state phosphorus concentration or load must be assumed and the equation solved for k. The results of the calculation are highly sensitive to the equilibrium conditions selected. Alternatively, a period of time exhibiting stable external loadings and a monotonic variation in phosphorous concentration within the water column can be analyzed for estimation of a reaction rate. With this approach, trial and error calculations can be performed using the analytical solution shown in eqn (7) in order to define the k rate.

5.2.4 Solutions

Where a differential equation is not amenable to an analytical solution, or where the solution is cumbersome, numerical integration methods can be employed. The system of differential equations comprising the series of reactors was solved numerically with a fourth-order Runge-Kutta technique. The Runge-Kutta method is a numerical integration technique based upon an iterative approach to the solution of a differential equation. For the differential equation

$$\frac{dy}{dx} = f(x,y)$$

where $y(x_o) = y_o$

approximations to the solution are obtained with the recursion formula

$$y_{n+1} = y_n + \frac{1}{6}(\alpha_1 + 2\alpha_2 + 2\alpha_3 + \alpha_4)$$

where

$$\alpha_1 = hf(x_n, y_n)$$

$$\alpha_2 = hf(x_n + \frac{h}{2}, y_n + \frac{1}{2}\alpha_1)$$

$$\alpha_3 = hf(x_n + \frac{h}{2}, y_n + \frac{1}{2}\alpha_2)$$

$$\alpha_4 = hf(x_n + h, y_n + \alpha_3)$$

and h is a fixed step size. Four function evaluations are required per step, which is a disadvantage of the method. Runge-Kutta methods have the advantage that they are self-starting, i.e., only the value of y at an initial point is required in order to compute y and the derivative of y at an incremental point. Another disadvantage is that the method provides no estimate of the accuracy of the solution and thus the adequacy of the step size h. This problem is overcome by trial iterations with several step sizes for comparison of results (Conte and deBoor, 1972).

For the present analysis, solutions were in the form of mean concentrations of phosphorus for each reactor (or reservoir), and were obtained on an annual basis. The time step required for convergence was determined empirically.

5.3 WATER BUDGET

The present analysis of phosphorus trends in the Highland Lakes is based upon mass loadings. Phosphorus mass loadings are input directly into the mathematical model, as described in a subsequent section, in order to compute concentrations within the reservoirs. With the use of mass loadings to construct the phosphorus mass budget, flowrates, or inflow volumes, are incorporated implicitly. Specifically, the present formulation of the mass budget implicitly accounts for annual inflows from major tributaries, runoff from contiguous watersheds, and the volume of direct precipitation.

The volumetric components of the mass loads developed in the previous chapter were isolated for comparison with historic reservoir hydrologic budget data reported by the LCRA (1983). The LCRA has available annual hydrologic summaries for each reservoir in the chain of Highland Lakes. The summaries describe monthly storage, turbine releases, spillway releases, evaporation, and withdrawals for the period 1966-1982. Annual average reservoir storage, releases, evaporation losses, and pumped withdrawals are presented in Tables 5-3 through 5-6, respectively.

The first step in the comparison was the calculation of runoff inflow to each reservoir. Runoff was calculated for each reservoir using the change in storage, releases from the dam, releases from the reservoir upstream, and evaporation reported by the LCRA. Reported evaporation was corrected to lake evaporation with a coefficient of 0.7. Inflow was computed from the data reported by the LCRA with the following formulation of the water budget:

$$Q_i = \frac{dS}{dt} + Q_o + Q_e - Q_{ur} + Q_{po} - Q_{pi}$$

where Q_i = runoff inflow, including major tributary and contiguous runoff

Q_o = releases from the dam, including turbine and spillway releases

Q_e = evaporation

Q_{ur} = releases from upstream reservoir

Q_{po} = pumped outflow

Q_{pi} = pumped inflow

dS/dt = change in storage

Water budget computations were performed with the computer program HYDROB, which is listed in Appendix E. Hydrologic data at monthly intervals were input to the model, with output provided in the form of monthly and annual budget terms. Inflow computed in this manner is a combination of tributary inflow, runoff from the adjacent watershed and precipitation. Annual inflow computed from the reported water budget is presented for selected reservoirs in Table 5-7 through 5-10. (Lakes Buchanan, Inks, LBJ and Travis were selected for analysis, since these represent the various types of waterbodies in the Highland Lakes chain.) In addition, annual inflows debited for precipitation are displayed.

Also shown in Table 5-7 through 5-10 are annual inflow data isolated from the mass loadings developed in the present study. These inflow data were calculated independently of the water budget data reported by the LCRA, using recorded streamflow data from the USGS and runoff from adjacent watersheds derived from the SCS method. As an accuracy check, the differences between the inflow values calculated with the two approaches were determined, as well as the ratio of the difference to the total inflow, including precipitation and releases from upstream reservoirs, derived from the LCRA data.

Tributary inflow obtained from USGS records is probably the most accurate variable in the preceding analysis. To a certain extent, then, the comparison of inflow calculated by the two alternative approaches is a comparison of runoff from contiguous watersheds obtained from the data reported by the LCRA and estimated with the SCS curve number technique. This is not strictly the case, however, since tributary inflow alone often exceeds the total inflow computed from the water budget. Inspection of Table 5-7 through 5-10 indicates that runoff from adjacent watersheds typically represents a very small portion of the total inflow to the reservoirs of the Highland Lakes chain. The total inflow is dominated by major tributary inputs and releases from upstream reservoirs. The estimated differences in runoff between the water budget method and the method employing recorded streamflow and calculated runoff are relatively small, within the same order of magnitude as either value of runoff. The ratio of the calculated difference in inflow to the total water budget inflow is usually less than 0.1, but is as high as 0.35.

Limitations of the SCS technique were described previously. The observed difference between the runoff computation methods is well within the accuracy of the water budget data reported by the LCRA. Potential sources of error in the water budget are numerous. Storage data are potentially the source of very significant errors, since relatively small changes in water surface elevation can represent large volumetric changes. Turbine release data may incorporate errors due to variation in calibration from pump curves. Spill release data are approximated, and can involve large volumes of water and thus provide substantial errors. Finally, errors may be associated with reported evaporation data, which are measured at only three sites along the series of reservoirs and are uncorrected for reservoir surface area. The volume of inflow to the Highland Lakes from contiguous watersheds, regardless of the computational method, can only be estimated at the present time.

In summary, the phosphorus mass balance model described in the preceding sections of this chapter does not explicitly perform a water budget calculation. Instead, hydrologic inputs are implicitly incorporated within specified mass loading inputs. Water budget terms that are input directly into the model include reservoir volume and outflow obtained from LCRA data. Atmospheric mass loads account for the volume of direct precipitation, obtained from recorded data. Tributary inflow is incorporated within tributary mass loadings, but it is based upon recorded USGS streamflow data. Similarly, runoff from adjacent watersheds is contained within calculated mass loadings, which are based upon runoff volumes generated by the SCS method. The present analysis demonstrates that the calculated runoff data occasionally exhibit a substantial difference with runoff derived from reported water budget data.

5.4 CHLORIDE BUDGET

The concentration of a substance within a reservoir is affected by a multitude of factors. At the simplest level, the concentration is determined by the mass of the substance and the volume within which it is mixed. Input of additional mass or quantities of water will affect the concentration. With some materials, additional processes act upon the concentration, such as sedimentation, bacterially-mediated conversions, oxidation, and biological uptake. Materials subject to these processes are designated as nonconservative. Conversely, conservative substances are not susceptible to decay-type influences (at least not significantly). Conservative substances are therefore commonly utilized as hydrodynamic tracers, since internal reaction kinetics need not be specified. Unlike phosphorus dynamics, which are subject to a complex of processes (represented in the present analysis as an apparent sedimentation rate), the concentration of chloride is influenced solely by the hydrologic budget and the magnitude of mass loadings.

The objective of the chloride budget analysis was to evaluate the performance of the water quality model HILAKES with a conservative constituent. To accomplish this objective, historical chloride loadings were estimated for the period 1968-1982, which corresponds approximately to the period with chloride concentration data available for the Highland Lakes and their major tributaries. The model HILAKES was then employed to simulate chloride concentrations in the series of reservoirs for the 15-year period, based upon the historical chloride mass loading data.

5.4.1 Chloride Mass Loadings

The procedures for estimation of chloride mass loadings were analagous to the methods employed for determination of phosphorus loads, described in detail in Chapter 4.0. Point source loadings were discounted as significant sources of historical chloride loadings to the Highland Lakes, based upon the analysis of phosphorus loads. Nonetheless, loads were approximated with the assumption that domestic wastewater is characterized by a chloride addition of 50 mg/l above the chloride concentration in the carriage water (Metcalf and Eddy, 1982). Estimating a typical carriage water chloride level of 50 mg/l, effluent chloride levels of 100 mg/l were obtained, and applied to historical discharge rates. To define major tributary loads, relationships between streamflow rate and chloride load were determined through a linear regression analysis. Relationships for each tributary are described in Table 5-11. The relationships were then applied to historical streamflow records for the Colorado, Llano, and Pedernales Rivers, as well as Sandy Creek. Barton Springs loads were calculated from estimates of the annual discharge (Slade, 1984) at an average chloride concentration of 25 mg/l. Loadings from adjacent watersheds were based upon runoff volumes derived from the SCS curve number technique. The concentration of chloride in runoff was estimated from historical data for monitoring stations in the Austin area, summarized in Table 5-12. Chloride data for high flows (greater than or equal to 0.28 m^3/s) are displayed in Table 5-13. For phosphorus concentrations, distinctions were made between rural, intermediate development, and urbanized land use conditions. No significant differences in chloride concentration were identified among the sampling station groupings comprising the land use categories. Monitoring stations on key tributaries to Lake Austin and Town Lake did, however, display higher chloride concentrations. Therefore, in a manner analogous to the specification of typical phosphorus concentrations described in Chapter 4.0, the chloride concentration of runoff from adjacent watersheds was assumed to be 15 mg/l for all development conditions in the watersheds of Lake Austin and Town Lake, and 12 mg/l for watersheds surrounding the other reservoirs. The average chloride concentration was applied to the runoff volume to obtain estimates of annual loads from adjacent watersheds. Atmospheric loads were estimated from recorded annual precipitation totals with an assumed chloride concentration of 0.5 mg/l. Historical chloride loadings are summarized in Table 5-14. Loadings associated with releases from upstream reservoirs were calculated within the mass budget analysis in the mathematical model, and are not included in Table 5-14.

5.4.2 Simulation of Reservoir Chloride Levels

5.4.2.1 Periods of Stable Inflow

The performance of the model HILAKES was investigated under input chloride mass loading conditions representative of periods of relatively stable tributary inflow to the Highland Lakes. A period of stable inflow is a necessary, but not sufficient, prerequisite to define steady-state conditions within a reservoir. At least, stable inflow can define a steady-state hydraulic condition.

Historical streamflow records were examined for periods with stable flowrates. As an additional criterion, at least one concentration sampling run was required within the period of stable inflow. Ideally, periods of stable inflow in cold-weather conditions were preferred, in order to avoid complications associated with thermal stratification in the reservoirs.

The following periods were selected to represent steady-state hydraulic conditions:

1 November 1973 - 30 April 1974
1 July 1975 - 30 June 1976
1 October 1975 - 31 March 1976
1 November 1980 - 28 February 1981

Input data were developed for each period, including reservoir volume, outflow, and chloride mass loadings from point and nonpoint sources. Chloride budgets were analyzed with the model HILAKES and resulting chloride concentrations were calculated for each reservoir.

Results of the chloride simulation are presented in Table 5-15 for each steady-state period. Also tabulated are observed chloride concentrations characteristic of the end of each period. The period-ending concentrations are in general approximated from measurements collected within one month of the termination date of each period.

Calculated concentrations conform relatively well to observed data. Certain trends warrant discussion. Calculated concentrations in Lake Buchanan tend to be too high. In fact, calculated concentrations exhibit perhaps more correlation to observed upper reservoir concentrations. This trend could indicate an over-estimation of mass loadings from the Colorado River (which dominates the total mass loading to the reservoir). Alternatively, the correlation to the upper reservoir chloride levels may indicate that the time frames of the designated steady-state periods are too short for the reservoir contents to equilibrate in response to a steady mass input. To elucidate the time required, the residence time for each reservoir was calculated under the hydraulically stable conditions as the ratio of volume to outflow rate. Estimated residence times are shown in Table 5-16. Indeed, residence times for Lake Buchanan are extremely large, on the order of 80-160 months (6.7-13.3 yrs), for the stable conditions. Lake Travis residence times are substantially lower, due to the magnitude of releases from the reservoir, with accumulated additional inflow. Thus, calculation of assumed steady-state chloride levels in Lake Buchanan (and probably Lake Travis) may be affected adversely by residence time. That is, the designated periods of stable inflow are too short to allow acquisition of a steady-state concentration.

Predicted concentrations in Inks Lake are similar to concentrations in Lake Buchanan, as expected. Lake LBJ displays good agreement between observed and calculated chloride concentrations. This is a key result, since the simulation correctly describes the observed substantial drop in chloride levels in Lake LBJ, relative to upstream reservoirs. Predicted concentrations in Lake Marble Falls have a tendency to fall slightly below observed levels. General agreement was realized between predicted and observed concentrations in Lake Travis. Calculated concentrations in Lake Austin and Town Lake were generally consistent with observed data. Because the residence times in Lake Austin and Town Lake are very short, stable hydraulic conditions develop over much shorter time frames than those associated with the four steady-state periods selected. As a result, comparison of predicted concentrations with observed period-ending concentrations may be inappropriate.

Calculated and observed concentrations exhibit the best agreement with the period 1 October 1975 - 31 March 1976. This may indicate that mass loadings are more accurately defined for this period. The performance may be enhanced by the fact that the steady-state period is encompassed by a longer period of relatively stable inflow.

Overall, calculated concentrations differ from observed data by roughly 5-20 percent. Because observed data are severely limited, the actual accuracy of the simulation cannot be assessed. Water budget terms are believed to be accurate to within only 10-30 percent of recorded values. Given the limitations of the data base, the model HILAKES appears to perform adequately.

5.4.2.2 Long-Term Period

The performance of the model HILAKES was investigated with annual chloride mass loading inputs. Chloride levels in the Highland Lakes for the period 1968-1982 were simulated. Input data for the simulation included the chloride mass loads developed in a previous section. Simulation results are displayed in Table 5-17 for the series of reservoirs. Also shown in Table 5-17 are mean annual chloride concentrations obtained from the limited historical data base (TNRIS, 1983a). Mean annual concentrations were calculated as the average of all stations within each reservoir. Historical average concentrations are based upon as few as three measurements per year. The most substantial historical data base was available for Lake Austin, where, typically, monthly measurements were available. Generally, the simulated chloride levels exhibit reasonable agreement with historical concentrations. With the limited historical data base, it is impossible to assess the accuracy of the predicted annual concentrations. The model HILAKES assumes constant mass input over each one-year period. In reality, the sequencing and timing of tributary flows and storm events during a year-long period can greatly affect reservoir chloride levels at any point in time. When a data collection program with limited sampling frequency is superimposed over the naturally occurring variability of the source terms, the use of historical data for estimation of reservoir concentrations for prolonged periods of time (and, in fact, at any time) is tenuous, at best.

5.5 ANALYSIS OF PHOSPHORUS BUDGET

Phosphorus budgets for distinct periods of time were analyzed for the Highland Lakes. As opposed to the analysis of chloride budgets described in the preceding section, phosphorus budgets were not evaluated to simulate concentrations, but rather, were evaluated for estimation of phosphorus reaction kinetics.

Two methods were employed for estimation of reaction rates under periods of stable inflow to the reservoirs. In the first method, reaction rates were calculated under the assumption of steady-state conditions. With the second technique, rates were calculated using the analytical solution to the system mass budget equation for periods of decreasing concentration.

5.5.1 Periods of Steady-State Conditions

Phosphorus reaction rates were investigated under input mass loading conditions representative of periods of relatively stable tributary inflow. Stable inflow periods were described in Sec. 5.4.2.1 in the chloride budget analysis.

Input data were developed for each period of stable inflow, including reservoir volume, outflow, and phosphorus mass loadings from point and nonpoint sources. Phosphorus reaction rates were estimated under the assumption of steady-state conditions for each period, as implied in eqn (9):

$$k_1 = \frac{W_1}{V_1 Css_1} - \frac{Q_1}{V_1}$$

$$k_2 = \frac{W_2 + Q_1 Css_1}{V_2 Css_2} - \frac{Q_2}{V_2}$$

etc., for additional reactors,

where k = reaction rate, T^{-1}
W = mass input, MT^{-1}
Q = outflow rate, L^3T^{-1}
V = volume, L^3
Css = steady-state concentration, M L^{-3}

and the numerical subscript refers to a specific reactor.

Periods with stable external loadings were examined in an attempt to eliminate as much variation as practicable in the complex of mass budget terms. Results of the reaction rate calculations are displayed in Table 5-18. The results illustrate a fundamental difficulty in the analysis: with the extremely limited available data base, accurate specification of a steady-state concentration is not possible because of the sparseness in time of the samples and the intrinsic error in measurement. Hence, several alternatives were examined as representative of the steady-state concentration: the period-ending concentration, average of all reservoir stations; the period-average concentration, average of all reservoir stations; the period-ending concentration from the lower pool station; and the period-average concentration from the lower pool station. The variation in calculated reaction rates is substantial, both within the study periods for individual reservoirs and among the reservoirs. Positive reaction rates indicate a concentration decrease; negative rates a concentration increase. With the Highland Lakes, overall phosphorus decay rates were anticipated, since the reservoirs should act as sinks for phosphorus. Calculated negative rates therefore are suspect. Variability or inconsistencies in the calculated rates may be indicative of a number of problems, including errors in the water budget and mass budget terms, and sampling and analytical errors connected with phosphorus measurements.

There is another, fundamental, consideration that may potentially contribute to difficulties in the preceding calculations. The reaction rates were calculated under the assumption of steady-state conditions, i.e., concentrations have attained a state of equilibrium. The estimated phosphorus reaction rates for Lake Buchanan, for example, range from 0.26 to 1.41 yr^{-1}, and 0.8 yr^{-1} can be considered a representative rate. The time constant of a reservoir was previously defined as:

$$\frac{1}{a} = \frac{1}{\frac{Q}{V} + k}$$

where $\frac{1}{a}$ = time constant, T
Q = outflow rate, L^3T^{-1}
V = volume, L^3
k = reaction rate, T^{-1}

For Lake Buchanan, the time constant can be estimated as approximately 1.05 years, using a reaction rate of 0.8 yr^{-1}. At a length of time equal to three time constants under steady external conditions, the reservoir can be assumed to establish a concentration within 5 percent of the equilibrium value. By this criterion, equilibrium conditions in Lake Buchanan can be assumed to materialize after a 3.15-year period, at the given reaction rate. Thus, the calculation indicates that for Lake Buchanan, the reaction rate cannot be rigorously computed using a steady-state formulation for assumed steady-state periods of one-year (or less) duration.

Representative reaction rates for each reservoir derived from the steady-state analysis and associated time constants and equilibrium times are shown in Table 5-19. Based upon the calculated time constants, the durations of the assumed steady-state periods are technically adequate for reaction rate computations using a steady-state formulation for the smaller reservoirs. For Lakes Buchanan and Travis, however, the time periods examined appear to be inappropriate for rigorous calculation of rates with a steady-state formulation.

5.5.2 Periods of Decreasing Concentration

In the preceding section, the calculation of reaction rates under the assumption of steady-state conditions was described. The limited availability of data resulted in considerable uncertainty as to the applicability of the steady-state assumption. In addition, the time constant criterion for the larger reservoirs cast doubt upon the assumption of steady-state conditions.

An alternate approach to the calculation of internal phosphorus reaction rates involves the analysis of periods of decreasing concentration within a reservoir under stable external conditions. Theoretically, a reaction rate can be calculated for a reservoir that exhibits a monotonic variation in concentration over time, using the analytical solution to the system mass balance equation described in eqn (7):

$$C = \frac{W}{Va}\ (1 - e^{-at}) + C_i e^{-at}$$

where

a	$= \frac{Q}{V} + k$
t	= time, T
w	= mass load, $M\ T^{-1}$
V	= volume, L^3
Q	= outflow, $L^3 T^{-1}$
k	= reaction rate, T^{-1}
C_i	= initial concentration, $M\ L^{-3}$
C	= concentration at time t, ML^{-3}

With this formulation, it is assumed that the mass loading, volume, and outflow are constant over the period examined.

For the periods examined, all terms in the equation were specified and k was determined by trial and error. Historical periods were selected for each reservoir that displayed a decreasing total phosphorus concentration in the lower pool station, accompanied by relatively stable inflow. Input data were developed for each period, including reservoir volume, outflow, and phosphorus loadings from point and nonpoint sources.

Results of the reaction rate calculations are shown in Table 5-20. As with the steady-state calculations, substantial variability in reaction rates is displayed within each reservoir and among reservoirs. The calculated reaction rates for Lake Buchanan are comparable, with values of 5.8 and 2.3 yr^{-1}. Returning to Table 5-18, these rates are greater than those calculated under the steady-state assumption. Of the two choices in Table 5-20, more confidence could perhaps be assigned to the value of 2.3 yr^{-1}, since it was calculated under cold-weather conditions, and the final concentration is greater than the detection limit. Because the final concentration was below the detection limit, the calculated rate of 5.8 yr^{-1} represents a lower bound to the actual reaction rate.

Calculated rates for Inks Lake vary an order of magnitude, 0.2 to 7.3 yr^{-1}, and lie outside the range of positive rates calculated with the steady-state assumption. For Inks Lake, the calculation is affected by the large throughflow, relative to the reservoir volume. Water replacement times of 46 days during the 100-day period used for the calculation of the 7.3 yr^{-1} rate and 141 days during the 112-day period used for the calculation of the 0.2 yr^{-1} rate were observed. In the former case then, the final concentration may have had little association with the initial concentration observed over the 100-day period, since the reservoir contents were theoretically replaced twice. More confidence is thus placed upon the calculated rate of 0.2 yr^{-1} for Inks Lake. This calculation also had the benefit of cold-weather conditions.

Lake LBJ reaction rates are generally consistent, 2.8 and 6.7 yr^{-1}. Both calculation periods had the benefit of cold-weather conditions. The rate of 2.8 yr^{-1} was calculated based upon a period-ending concentration of 0.0098 mg/l total phosphorus, which represented the level of detection. Reaction rates calculated under the assumption of steady-state conditions were generally lower than those obtained in this analysis.

Calculated rates differed markedly for Lake Marble Falls: 2.9 and -15.9 yr^{-1}. Both calculation periods were affected by several water volume replacements, based upon the large throughflow-to-volume ratio, and may have been affected by complications of summer conditions. The steady-state assumption yielded large negative and large positive reaction rates, as shown in Table 5-18. None of the calculated rates for Lake Marble Falls are particularly valid, in view of the complication of the replacement time.

A reaction rate of 3.1 yr^{-1} was computed for Lake Travis, with only a single period of time suitable for the analysis. This value is higher than the rates calculated under the steady-state assumption.

Nearly identical reaction rates of 15.2 and 15.4 yr^{-1} were computed for two periods on Lake Austin. These values are within the range of rates obtained with the assumption of steady-state conditions, shown in Table 5-18. The magnitude of the rates calculated for Lake Austin may be influenced by uptake of phosphorus by the dense growth of aquatic macrophytes. However, the rates displayed in Table 5-20 (and 5-18 as well) are corrupted by the water replacement time, with several replacements of the reservoir contents occurring over the time frames of analysis.

Town Lake reaction rates shown in Table 5-20 are highly variable. A large negative rate was calculated for one period, and negative rates were also evident in the steady-state calculation. A range of possible reaction rates is displayed for two periods of analysis on Town Lake. The range of values was presented because of large outflow-to-volume ratios, which relegated the reaction rate to a relatively minor effect. All rates calculated for Town Lake are affected by the water replacement time, with numerous volumetric replacements for each period of analysis.

The variability in calculated reaction rates may be attributable to inaccuracies in water budget or mass budget terms, as well as errors in the phosphorus measurements. Obvious potential problems with the concentration data include sampling and analytical errors. In addition, a substantial error may be incorporated by the assumption that the observed decrease in phosphorus concentration occurred over the time period between measurements. This problem is related to the water replacement time for the period of calculation. The problem is mainly one of a lack of sufficient temporal density in the sampling data: there is no information available to determine if fluctuation in the phosphorus level occurred within the intervals between samples, which generally are on the order of three months in duration. The cycling time of phosphorus within a reservoir is influenced by a complex of processes, including (see, for example, Hutchinson, 1957) liberation of phosphorus into the epilimnion from littoral areas, uptake of phosphorus by littoral vegetation, uptake of phosphorus by phytoplankton, sedimentation of phytoplankton into the hypolimnion, and diffusion of phosphorus from hypolimnetic sediments in the absence of an oxidized microzone.

5.5.3 CRWR Data for Lake LBJ

Previous sections lamented the limited historical data base available for the Highland Lakes. A singular exception is the comprehensive data set available for Lake LBJ (see Fruh, et al., 1974a, 1974b, 1974c, 1975a, 1975b) collected over the period 1971 - 1975 by the CRWR. Vertical profiles of total phosphorus and other constituents were measured at typically monthly intervals in the lower pool and at several other sampling locations throughout the reservoir. The CRWR data were compiled and analyzed for estimation of a first-order reaction rate in Lake LBJ.

The CRWR measurements of total phosphorus display generally higher concentrations than the TDWR data set. Comparison of several samples from the two data sources collected at nearly simultaneous times indicated substantial differences in magnitude. The source of the discrepancy cannot be discerned. For the present analysis, it was assumed that the CRWR data exhibited systematic behavior, regardless of whether or not it is correct. The data were not combined with other data sources but instead were analyzed separately.

Average phosphorus levels were calculated for both the upper water column and the complete water column for each sampling date at the lower pool station. The upper water column was delimited wherever a sharp increase in phosphorus concentration was evident, which usually encompassed a depth of 9-15 m (30-50 ft), compared to a total depth of 24 m (80 ft). Annual averages were calculated for surface, upper water column, and total water column data. The analysis indicated that average surface and upper water column concentrations were virtually identical, which has key implications for analyses throughout this investigation that are based upon surface phosphorus measurements, the only data generally available.

The period of stable inflow of 1 November 1973-30 April 1974 described in a previous section was examined for estimation of a reaction rate under a steady-state assumption. Calculated rates indicated generation of phosphorus. The failure of the calculation may be attributed to two problems. First, the loading methodology developed in this study may very well be incompatible with the CRWR data. Secondly, the assumption of a steady-state condition in the reservoir may have been erroneous, a conclusion in fact supported by the CRWR phosphorus data.

Several periods of decreasing phosphorus concentration were identified and examined for reaction rates, similar to the analysis described in the preceding section. Results are summarized in Table 5-21. Calculated k rates range from 3.8 to 27.4 yr^{-1}. The rates are comparable to those presented in Table 5-20 for Lake LBJ, though there is some indication of rates of greater magnitude.

Examination of CRWR data as an independent data set neither proved nor disproved previous analyses. It simply confirmed that a first-order reaction rate for Lake LBJ can only be estimated at the present time.

5.6 CALIBRATION

Calibration of a model entails comparison of calculated results against measured values. Input variables and reaction coefficients within the model may be adjusted to improve performance, though the parameters should conform to theoretical formulations. In the verification process, the calibrated model is tested against an independent data set (or sets) of external conditions. If the calibrated model displays acceptable performance with the independent data set, the model is said to be verified. Optimally, a calibrated model should exhibit successful performance under a variety of external conditions or verification data sets.

With many of the more complex models currently in use, calibration may be a tenuous process, due largely to the absence of a substantial, detailed input data base. In some cases, "the mathematical formulations have run ahead of the conceptual understanding of the underlying processes and the measurement of data on these processes" (Rogers, 1982). In other words, the most appropriate mathematical model formulation for a particular application is circumscribed by the nature of the available data base as well as the objectives of the study.

The model formulation for the Highland Lakes is based upon a one-dimensional series of completely mixed reactors. The expressed objective of the modeling analysis is to evaluate long-term trends in water quality within the reservoirs and assess the effects of waste loads to upstream reservoirs upon water quality in lower reservoirs. The proposed predictive time-scale is in terms of annual values. For this modeling approach, calibration requires either a long-term history of external input and internal reservoir characteristics, or a series of detailed mass budgets for individual years. The long-term history is preferable, since calibration to a single year is subject to the potential use of an unrepresentative set of conditions. For the Highland Lakes, detailed data for description of an annual mass budget are extremely limited. A time-series of historic annual external phosphorus loads was developed for application to model calibration, as described in Chapter 4.0.

The series of historic annual external phosphorus loads was input directly into the program HILAKES for calibration. In addition to data characterizing external loads, reservoir volume and outflow data are also required. All external inflows and associated mass loadings to the reservoirs are accounted for in the loading data set except for the inputs associated with releases from upstream reservoirs. While the recorded releases were input directly into the calibration data set, the phosphorus concentration of the discharged water was calculated with the mass budget analysis in order to define the corresponding phosphorus loads.

Input parameters which drive the model HILAKES were described in Sec. 5.2. In most respects, the historical data base is extremely limited. These limitations may have affected the accuracy of the estimated input variables, though the extent cannot be ascertained. Calibration of the model involved simulation of historical phosphorus levels in the Highland Lakes for the period 1952 through 1982. Theoretically, any of the myriad of input variables could be adjusted to effect improvement in simulated versus observed phosphorus concentrations. From a practical standpoint, however, there is no basis for indiscriminate adjustment of most of the parameters. For the present analysis, the first-order reaction rate was adjusted based upon observed concentration data to effect calibration. Initial estimates of reaction rates were described in the preceding sections.

The calibration procedure involved a sequential examination of the reservoirs to evaluate the effects on predicted concentration of variation of the reaction rate. Predicted results were compared to historical data. Measurements of phosphorus concentrations within the reservoirs which constitute the historical data base are extremely limited in temporal and spatial coverage. This lack of data severely handicapped the calibration procedure.

The analysis commenced with Lake Buchanan. The k rate was varied over a range of 2.0 to 9.0 yr^{-1} and the effects on concentration were observed. Simulation results are displayed in Table 5-22 for the years 1968 to 1982. Available concentration data are included in the table in two forms: as annual averages of all mainstem reservoir data and lower pool data. In effect, the only years with data even remotely sufficient for determination of annual average conditions are 1974, 1975, 1976, and 1981. With the limited historical data base, identification of the most appropriate reaction rate was not straightforward. No single k rate provided precise conformance to all observed concentration data. In fact, the data indicated that k could potentially vary over a fairly significant range. To assist in the selection of an appropriate rate, allowable error bands of ± 50 percent of the observed average pool concentrations were visualized. Simulation results were then examined with respect to the overlapping error bounds. Data for the pool stations were emphasized because they theoretically are representative of a major portion of the reservoir contents; however, whole-lake averages were also taken into consideration to supplement the sparse pool measurements. For Lake Buchanan, the whole-lake average concentration can be substantially higher than the average lower pool value, due to the influence of the Colorado River in the headwater reach. For Lake Buchanan, a k rate of 6.0 yr^{-1} fell within the error bounds of the majority of the data. At least one data set (e.g., 1968) indicated that a higher reaction rate would be appropriate. The whole-lake average data generally indicated that a lower reaction rate (e.g., 3.0 to 5.0 yr^{-1}) could be utilized. In addition to the limitations in the historical data set already described, many of the available measurements of phosphorus concentration were at or below the analytical detection limit. This could have a substantial effect upon the calculated average concentrations (values at the detection limit were assumed equal to the limit for averaging purposes), and was part of the rationale for employing an error bound to the averages. A reaction rate calculated from average concentrations that include values below detection limits may represent a lower bound to the actual rate.

With the k rate for Lake Buchanan set at 6.0 yr^{-1}, the calibration exercise was repeated for Inks Lake. In Table 5-23 are shown the results of the simulation over a range of k rates from 0.01 to 10.0 yr^{-1}. It is apparent that the reaction rate exerts relatively little effect upon the concentration in the reservoir, since the conditions within Inks Lake are frequently dominated by the throughflow originating from Lake Buchanan, rather than internal kinetics. None of the k rates examined provided systematic agreement with the reported data. Supposition of error bounds provided little benefit, due to the wide divergence among observed average concentrations. Quantitative analysis of the concentration data for Lake Inks for definition of a k rate is tenuous, due to the domination of throughflow (as represented in the ratio of throughflow to volume). The logical choice was to adopt the k rate previously determined for Lake Buchanan for consistency.

The calibration exercise was subsequently focused upon Lake LBJ. Simulated concentrations for a range of reaction rates are presented in Table 5-24, with the k rates for Lakes Buchanan and Inks set at 6.0 yr^{-1}. Taking into consideration assumed error bounds on the average concentrations, a range of k rates from approximately 3.0 to 9.0 yr^{-1} could be employed for representation of historical conditions. Recognizing the analysis for the upstream reservoirs, a reaction rate of 6.0 yr^{-1} was determined to be appropriate for Lake LBJ.

The simulation for Lake Marble Falls is described in Table 5-25. As with Inks Lake, Lake Marble Falls is a throughflow-dominated reservoir, and the k rate has little effect upon the predicted concentrations. The observed concentrations indicate that a positive reaction rate may be inappropriate. A rate of zero, which implies behavior as a conservative constituent, was examined but the simulation was not improved. A negative reaction rate, indicating that the reservoir acts as a source of phosphorus, would be required to obtain agreement with much of the observed data. For the present analysis, it was hypothesized that the reservoir constitutes a phosphorus sink, and the irreproducibility of observed average concentrations was attributed to errors in quantification of loading terms or phosphorus measurements. A rate of 6.0 yr^{-1} was therefore selected for Lake Marble Falls, consistent with the upstream reservoirs.

The simulation of phosphorus levels in Lake Travis at varying k rates is described in Table 5-26. The reaction rates for all upstream reservoirs were set at 6.0 yr^{-1} for the analysis. Over the range of k rates examined, predicted concentrations were relatively low, compared to upstream reservoirs. The observed concentrations indicate that a reaction rate of 1.0 yr^{-1} or lower may be appropriate for Lake Travis. However, comparison of simulation results to historical data is difficult, since many of the phosphorus measurements in Lake Travis were at or below the detection limit. Thus, a rate of 1.0 yr^{-1} probably constitutes a lower bound to the actual k rate, and with this uncertainty, a precise range of k rates cannot be defined. A reaction rate of 6.0 yr^{-1} was selected for use in conformance with upstream reservoirs, since specification of an alternate rate cannot be justified with cognizance of the limitations in the data base.

Simulation results for Lake Austin and Town Lake are shown in Tables 5-27 and 5-28, respectively. Both reservoirs are throughflow-dominated, with little effect demonstrated by the reaction rate. Predicted concentrations were relatively low, and were influenced primarily by predicted concentrations in Lake Travis upstream. The situation for Lake Austin and Town Lake is similar to that for Lake Marble Falls described above: a k rate of zero or a negative rate is indicated by the average concentration data. Examination of reaction rates may only indicate an appropriate lower bound, since much of the available concentration data displayed phosphorus levels at or below the limit of detection. In addition, with the rapid replacement times in these two lowermost reservoirs, an average or typical concentration cannot be accurately determined on the basis of infrequent sampling data. For consistency, a reaction rate of 6.0 yr^{-1} was adopted for both Lake Austin and Town Lake.

With the completion of the calibration exercise, a uniform k rate for phosphorus was selected for application within the annual simulation methodology of the model HILAKES. As described above, the available data set for the Highland Lakes was limited, which affected the accuracy of the analysis. A rate of 6.0 yr^{-1} demonstrated a reasonable approximation to the observed data sets. Selection of a reaction rate of 6.0 yr^{-1} was not precise. Given the limited data base, a range of k rates would have been an appropriate conclusion. The acceptable range would probably extend from roughly 1.0 to 10.0 yr^{-1}. While this range may appear large, this is actually a significant conclusion, since it defines the correct order of magnitude for the reaction rate. In other words, while the analysis did not unequivocably set the appropriate k rate for the reservoirs at precisely 6.0 yr^{-1}, the analysis did demonstrate that the rate is nearer 6.0 yr^{-1} than 0.6 or 60.0 yr^{-1}.

The selected reaction rate can be compared to reported values for the phosphorus sedimentation coefficient shown in Table 5-2. To facilitate the comparison, Table 5-29 displays the mean depths and flushing rates (reciprocal of retention time) for the Highland Lakes, along with the calculated apparent settling velocity for phosphorus. Schnoor and O'Connor (1980) reported a particulate phosphorus settling rate of 2.92 yr^{-1} for Lake LBJ. With respect to the present analysis, the most pertinent data in Table 5-2 are the values for the TVA reservoirs (Higgins and Kim, 1981), which display some similarities to the Highland Lakes. The phosphorus sedimentation coefficient for the TVA reservoirs ranged from 0.16 to 17.06 yr^{-1}, and the apparent settling velocity displayed a range of 2.4 to 268.8 m/yr. Both ranges encompass the phosphorus removal rates estimated for the Highland Lakes.

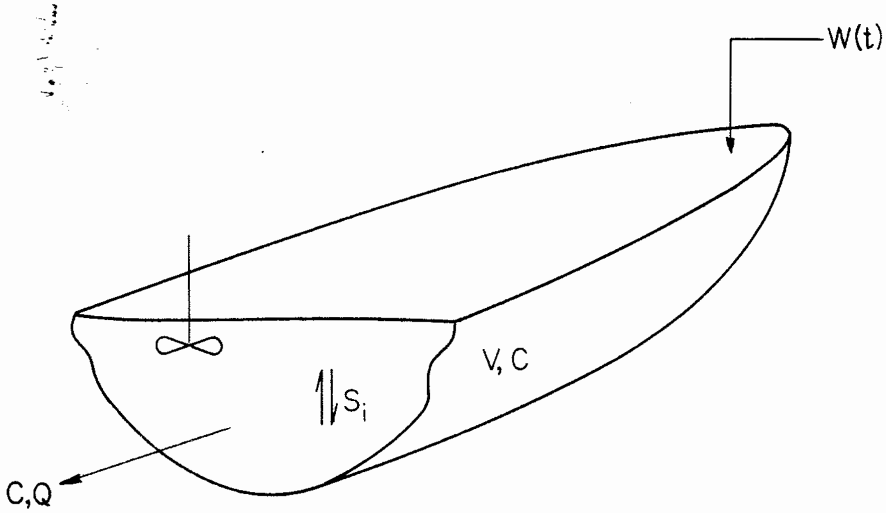

Fig. 5-1 COMPLETELY MIXED SYSTEM

FIGURE 5-2

FLOW CHART OF MODEL HILAKES

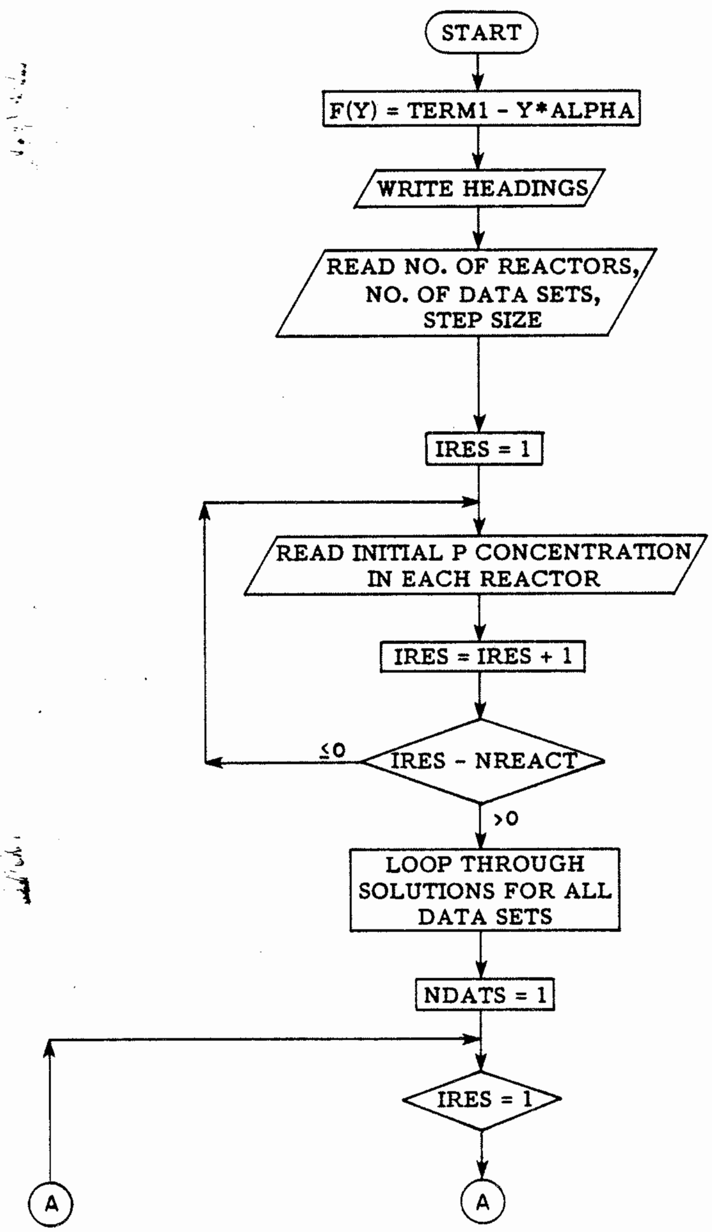

FIGURE 5-2 (Cont'd)

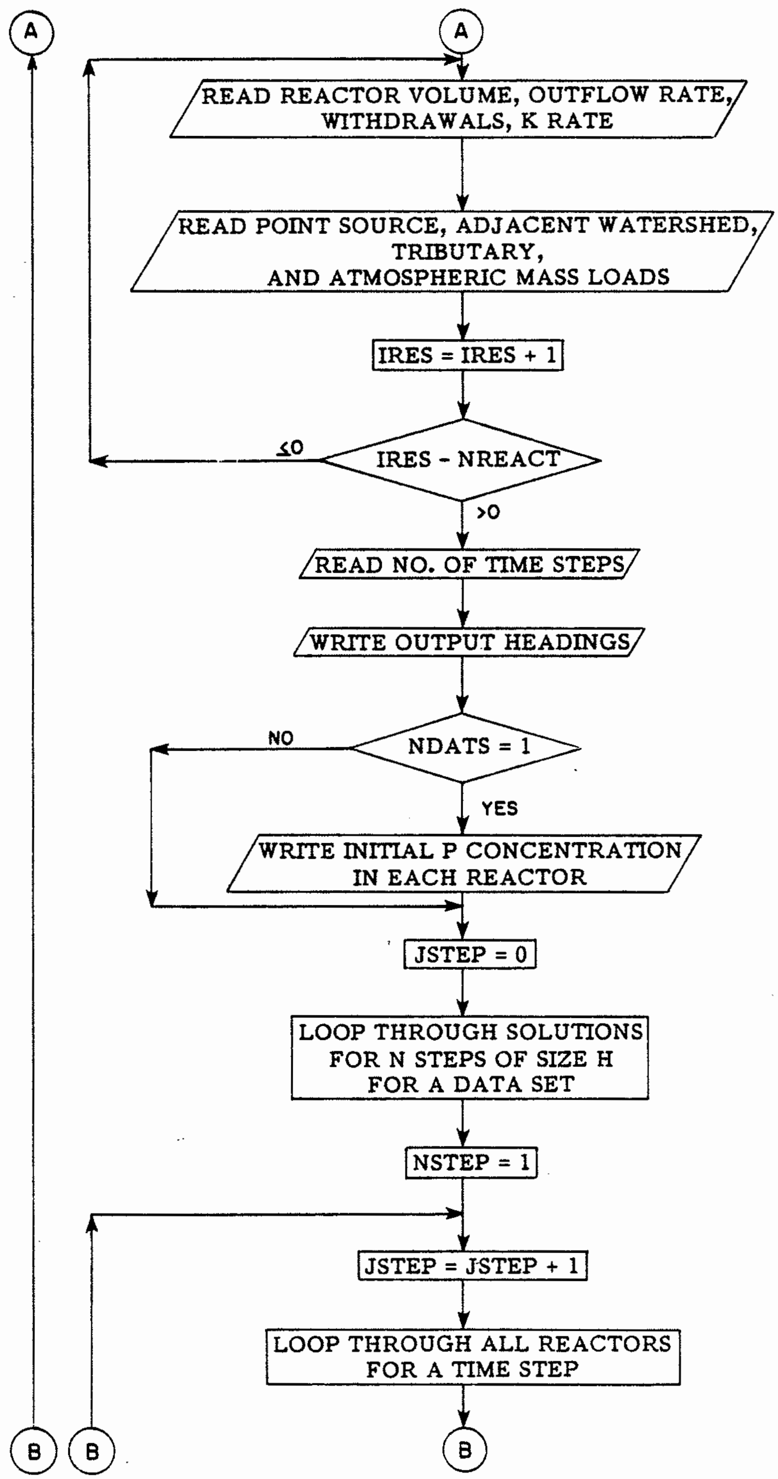

FIGURE 5-2 (Cont'd)

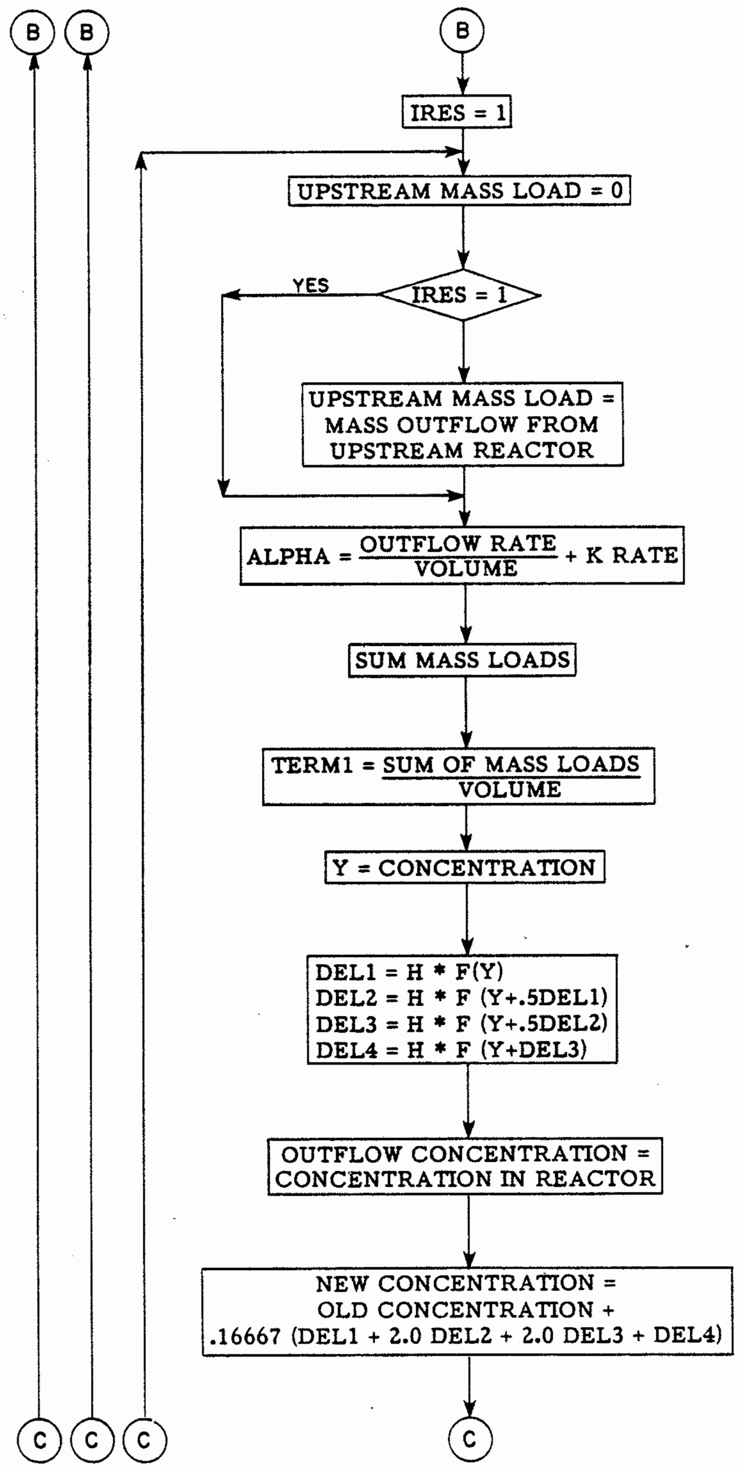

FIGURE 5-2 (Concluded)

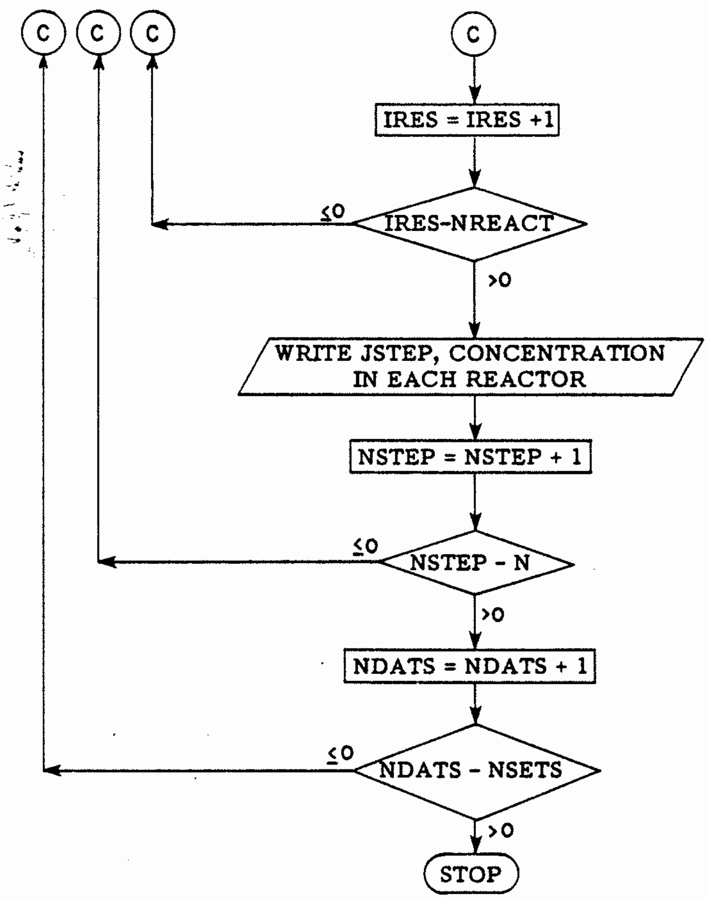

TABLE 5-1

CONCENTRATIONS FOR SELECTED MASS INPUT FUNCTIONS

Input	Form	Concentration
Constant	W_o	$\frac{W_o}{aV}(1 - e^{-at})$
Linear	$W_o \pm wt$	$\frac{W_o}{aV}(1 - e^{-at}) \pm \frac{w}{a^2V}(1 - e^{-at} - at)$
Exponential	$W_o e^{\pm we}$	$\frac{W_o}{(a \pm w)V}(e^{-wt} - e^{at})$

Source: O'Connor and Mueller, 1970

TABLE 5-2

EXAMPLES OF PHOSPHORUS REMOVAL RATES

Waterbody	Sedimentation Coefficient (1/yr)	Apparent Settling Velocity (m/yr)	Mean Depth (m)	Flushing Rate (1/yr)	Note	Source
Aegerisee	0.24	11.97	49	0.015	--	Dillon, 1975
Turlersee	1.86	26.04	14	0.465	--	Dillon, 1975
Hallwilersee	0.15	4.10	28	0.260	--	Dillon, 1975
Bodensee-Obersee	0.38	38.07	100	0.205	--	Dillon, 1975
Pfaffikersee	1.29	23.20	18	0.385	--	Dillon, 1975
Zurichsee	0.23	11.33	50	0.680	--	Dillon, 1975
Greifensee	0.80	15.19	19	0.490	--	Dillon, 1975
Baldeggersee	0.34	11.70	34	0.220	--	Dillon, 1975
Tahoe	0.02	5.64	303	0.001	--	Dillon, 1975
Norrviken	1.68	9.08	5.4	1.750	--	Dillon, 1975
Sebasticook	0.36	2.16	6.0	0.390	--	Dillon, 1975
Clear	0.52	6.50	12.5	0.130	--	Dillon, 1975
Erie	2.10	37.80	18	0.400	--	Dillon, 1975
Ontario	0.54	45.27	84	0.152	--	Dillon, 1975
Superior	0.05	7.06	148	0.005	--	Dillon, 1975
Michigan	0.29	24.19	84	0.032	--	Dillon, 1975
227 (ELA)	1.93	8.47	4.4	0.238	--	Dillon, 1975
Okanagan	0.32	24.32	75.3	0.017	--	Dillon, 1975
Skaha	1.65	43.80	26.5	0.890	--	Dillon, 1975
Kalamalka	0.08	4.70	58	0.009	--	Dillon, 1975
Wood	0.08	1.74	21	0.009	--	Dillon, 1975
Menona	2.04	15.91	7.8	0.833	--	Dillon, 1975

TABLE 5-2 (Cont'd)

Waterbody	Sedimentation Coefficient (1/yr)	Apparent Settling Velocity (m/yr)	Mean Depth (m)	Flushing Rate (1/yr)	Note	Source
Kegonsa	0.28	1.30	4.6	2.860	--	Dillon, 1975
Leman	0.02	3.22	155	0.083	--	Dillon, 1975
Cameron	8.30	58.93	7.1	16.35	1971, 1972 avg	Dillon and Rigler, 1974
Four Mile	1.20	11.16	9.3	0.235	1971, 1972 avg	Dillon and Rigler, 1974
Raven	16.86	12.31	0.73	13.55	1971, 1972 avg	Dillon and Rigler, 1974
Talbot	10.91	9.27	0.85	4.900	1972 only	Dillon and Rigler, 1974
Bob	0.90	16.20	18.0	0.335	1971, 1972 avg	Dillon and Rigler, 1974
Twelve Mile-Boshkung	1.21	21.90	18.1	2.125	1971, 1972 avg	Dillon and Rigler, 1974
Halls	1.01	27.47	27.2	0.895	1971, 1972 avg	Dillon and Rigler, 1974
Beech	1.30	12.69	9.8	19.75	1971, 1972 avg	Dillon and Rigler, 1974
Maple	3.14	36.42	11.6	6.950	1971, 1972 avg	Dillon and Rigler, 1974
Pine	0.13	0.96	7.4	17.40	1971, 1972 avg	Dillon and Rigler, 1974
Eagle-Moose	0.90	11.58	12.8	1.925	1971, 1972 avg	Dillon and Rigler, 1974
Oblong-Haliburton	0.82	14.51	17.7	0.320	1972 only	Dillon and Rigler, 1974
Bay of Naples	3.28	14.10	4.3	14.0	--	Larsen and Mercier, 1976
Canadaigua	0.13	5.07	39.0	0.07	--	Larsen and Mercier, 1976
Carlos	0.34	4.45	13.1	0.27		Larsen and Mercier, 1976
Carry Falls	3.74	20.20	5.4	9.61		Larsen and Mercier, 1976
Cass	1.99	15.12	7.6	1.17		Larsen and Mercier, 1976
Charlevoix	0.53	8.90	16.8	0.31		Larsen and Mercier, 1976
Higgins	0.14	2.09	14.9	0.06		Larsen and Mercier, 1976
Houghton	0.68	1.56	2.3	0.76		Larsen and Mercier, 1976
Long-Aroostook	0.39	5.23	13.4	0.31		Larsen and Mercier, 1976
Long-Cumberland	0.83	8.63	10.4	0.83		Larsen and Mercier, 1976

TABLE 5-2 (Concluded)

Waterbody	Sedimentation Coefficient (1/yr)	Apparent Settling Velocity (m/yr)	Mean Depth (m)	Flushing Rate (1/yr)	Note	Source
Mattawamkegg	3.90	14.43	3.7	8.69		Larsen and Mercier, 1976
Moosehead	0.28	4.59	16.4	0.33		Larsen and Mercier, 1976
Pelican	0.25	0.68	2.7	0.30		Larsen and Mercier, 1976
Rangeley	0.35	5.00	14.3	0.36	--	Larsen and Mercier, 1976
Sebago	0.25	7.70	30.8	0.19	--	Larsen and Mercier, 1976
Winnipesaukee	0.84	11.00	13.1	0.25	--	Larsen and Mercier, 1976
Wilson	5.84	71.6	12.3	76.1	TVA reservoir	Higgins and Kim, 1981
Guntersville	12.64	53.5	4.2	40.6	TVA reservoir	Higgins and Kim, 1981
Nickajack	3.76	25.3	6.8	130.4	TVA reservoir	Higgins and Kim, 1981
Watts Bar	13.69	99.9	7.3	28.5	TVA reservoir	Higgins and Kim, 1981
Chatuge	0.73	6.9	9.5	2.3	TVA reservoir	Higgins and Kim, 1981
Cherokee	17.06	237.5	13.9	4.6	TVA reservoir	Higgins and Kim, 1981
Douglas	8.85	95.1	10.7	8.5	TVA reservoir	Higgins and Kim, 1981
Fontana	7.11	268.8	37.8	3.5	TVA reservoir	Higgins and Kim, 1981
Hiwassee	1.54	31.3	20.2	6.5	TVA reservoir	Higgins and Kim, 1981
Norris	4.59	74.9	16.3	2.7	TVA reservoir	Higgins and Kim, 1981
South Holston	3.80	89.0	23.4	1.8	TVA reservoir	Higgins and Kim, 1981
Tims Ford	0.16	2.4	14.9	1.8	TVA reservoir	Higgins and Kim, 1981
Watauga	4.72	115.5	24.5	1.5	TVA reservoir	Higgins and Kim, 1981
Washington	0.95	36.0	38	--	TVA reservoir	Lorenzen et al., 1976
Lyndon B. Johnson	2.92	19.6	6.7	--	Particulate P settling	Schnoor and O'Connor, 1980
Natural lakes	24	--	13	5	290 lakes avg	Canfield and Bachman, 1981
Artificial lakes	145	--	9	35	433 reservoirs avg	Canfield and Bachman, 1981

TABLE 5-3

ANNUAL AVERAGE RESERVOIR STORAGE

HIGHLAND LAKES

Year	L. Buchanan ($10^6 m^3$)	Inks L. ($10^6 m^3$)	L. LBJ ($10^6 m^3$)	L. Marble Falls ($10^6 m^3$)	L. Travis ($10^6 m^3$)	L. Austin ($10^6 m^3$)	Town L. ($10^6 m^3$)
1983	1064.8	20.5	169.9	9.5	1138.2	24.7	4.3[2]
1982	1103.9	21.4	169.9	10.1	1306.3	23.5	4.3
1981	1163.8	20.5	170.2	10.2	1407.3	24.7	4.3
1980	1119.9	21.0	168.8	10.2	1277.6	23.4	4.3
1979	1107.1	21.0	169.7	10.2	1300.3	25.0	4.3
1978	1000.2	20.5	171.0	10.3	997.6	25.0	4.3
1977	1124.2	21.5	168.9	10.3	1336.9	24.6	4.3
1976	1167.3	21.1	169.2	10.3	1373.3	24.8	4.3
1975	1164.2	21.0	169.9	9.9	1432.7	23.3	4.3
1974	1084.9	21.4	169.3	10.2	1450.6	24.7	4.3
1973	917.3	21.1	171.3	10.1	1467.3	22.4	4.3
1972	1091.0	21.1	169.4	10.0	1304.6	23.6	4.3
1971	1056.5	21.0	119.9	9.7	1216.5	25.0	4.3
1970	1120.1	20.3	128.2	10.3	1432.4	23.6	4.3

TABLE 5-3 (Cont'd)

Year	L. Buchanan ($10^6 m^3$)	Inks L. ($10^6 m^3$)	L. LBJ ($10^6 m^3$)	L. Marble Falls ($10^6 m^3$)	L. Travis ($10^6 m^3$)	L. Austin ($10^6 m^3$)	Town L. ($10^6 m^3$)
1969	1133.1	20.4	169.7	10.4	1363.9	21.3	4.3
1968	1154.0	21.0	169.8	10.4	1450.9	20.9	4.3
1967	1044.6	21.1	166.7	10.1	1162.2	22.8	4.3
1966[1]	1159.9	21.0	169.8	10.4	1383.1	22.9	4.3
1965	1045.3	21.0[3]	168.6	10.2[3]	1099.5	23.6[3]	4.3
1964	847.8	21.0	167.8	10.2	621.5	23.6	4.3
1963	688.3	21.0	168.4	10.2	799.3	23.6	4.3
1962	770.6	21.0	168.4	10.2	1126.0	23.6	4.3
1961	938.0	21.0	170.8	10.2	1215.7	23.6	4.3
1960	1040.9	21.0	169.0	10.2	1340.2	23.6	4.3
1959	1050.8	21.0	165.9	10.2	1373.5	23.6	4.3
1958	1075.4	21.0	166.5	10.2	1387.7	23.6	4.3
1957	1065.5	21.0	160.4	10.2	1272.7	23.6	4.3
1956	990.2	21.0	161.6	10.2	1253.0	23.6	4.3
1955	979.1	21.0	161.6	10.2	1231.5	23.6	4.3
1954	1017.6	21.0	162.8	10.2	1184.0	23.6	4.3

TABLE 5-3 (Concluded)

Year	L. Buchanan ($10^6 m^3$)	Inks L. ($10^6 m^3$)	L. LBJ ($10^6 m^3$)	L. Marble Falls ($10^6 m^3$)	L. Travis ($10^6 m^3$)	L. Austin ($10^6 m^3$)	Town L. ($10^6 m^3$)
1953	952.6	21.0	168.4	10.2	1354.4	23.6	4.3
1952	667.7	21.0	114.7	10.2	964.7	23.6	4.3

1 Storage for 1966-1982 computed from monthly storage data; storage for 1952-1965 computed from beginning and ending storage during the year.

2 Town Lake storage assumed constant.

3 Storage for 1952-1965 estimated from average of preceding years.

TABLE 5-4

ANNUAL RESERVOIR RELEASES

HIGHLAND LAKES

Year	L. Buchanan ($10^6 m^3/yr$)	Inks L. ($10^6 m^3/yr$)	L. LBJ ($10^6 m^3/yr$)	L. Marble Falls ($10^6 m^3/yr$)	L. Travis ($10^6 m^3/yr$)	L. Austin ($10^6 m^3/yr$)	Town L. ($10^6 m^3/yr$)
1983	143.6	167.5	403.5	453.0	698.4	555.2	592.4
1982	599.2	602.7	759.0	790.8	1167.1	1047.2	1088.7
1981	493.6	502.9	1280.6	1352.3	1890.4	1899.1	2088.7
1980	593.2	604.6	1003.7	1023.5	919.2	837.2	866.0
1979	106.3	127.4	503.7	516.0	795.8	740.4	835.1
1978	358.7	355.5	666.0	683.3	900.6	801.4	836.7
1977	740.0	740.9	1253.0	1312.3	2040.5	2046.2	2120.8
1976	462.0	473.7	829.0	867.5	1096.2	1040.7	1121.6
1975	860.7	839.0	1477.2	1564.6	2380.9	2441.3	2557.3
1974	700.1	634.2	1295.9	1335.1	2162.2	2022.5	2119.2
1973	408.7	421.2	927.4	964.8	1122.5	1086.0	1188.0
1972	425.8	434.7	642.3	678.1	797.6	766.8	849.2
1971	875.2	873.6	1206.1	1226.2	1326.9	1249.1	1285.1
1970	804.7	836.9	1409.6	1430.0	1855.6	1814.0	1906.2
1969	600.0	604.4	1074.1	1102.3	1336.1	1301.0	1367.3
1968	1325.6	1326.0	2117.3	2118.7	2642.4	2710.9	2783.2

TABLE 5-4 (Concluded)

Year	L. Buchanan ($10^6 m^3/yr$)	Inks L. ($10^6 m^3/yr$)	L. LBJ ($10^6 m^3/yr$)	L. Marble Falls ($10^6 m^3/yr$)	L. Travis ($10^6 m^3/yr$)	L. Austin ($10^6 m^3/yr$)	Town L. ($10^6 m^3/yr$)
1967	376.7	391.9	493.6	505.8	770.8	772.9	797.0
1966	456.6	475.8	722.2	727.4	1016.1	1030.3	1086.7
1965	832.7	856.5	1393.9	1416.5	1179.8	1230.5	1300.1
1964	412.7	440.0	807.1	829.1	577.3	554.2	562.5
1963	67.8	90.0	147.5	152.6	763.4	726.4	752.5
1962	610.4	607.9	695.3	707.4	733.6	749.6	784.6
1961	965.4	979.5	1562.0	1603.8	2099.6	2101.6	2196.0
1960	755.1	727.3	1205.8	1208.8	1694.2	1673.4	1734.8
1959	1123.8	1133.0	1796.8	1796.8	2245.9	2240.2	2300.2
1958	915.0	925.9	1548.7	1548.7	2398.7	2413.0	2487.2
1957	3560.8	3571.2	4631.3	4658.9	5038.4	5015.5	5052.2
1956	847.0	670.4	640.0	640.0	799.3	771.4	764.0
1955	1309.6	1255.5	1663.5	1662.5	1331.8	1331.8	1328.0
1954	674.4	674.4	678.0	664.4	758.1	758.1	767.7
1953	319.5	325.5	442.9	440.6	661.7	684.8	708.6
1952	425.0	435.1	1038.5	1038.5	578.4	562.6	566.3

Notes:

L. Buchanan = discharge - pump back.
Town L. = L. Austin release + Barton Springs - City withdrawal + adjacent runoff + precipitation - evaporation.

TABLE 5-5

ANNUAL EVAPORATION FROM HIGHLAND LAKES

Year	L. Buchanan ($10^6 m^3/yr$)	Inks L. ($10^6 m^3/yr$)	L. LBJ ($10^6 m^3/yr$)	L. Marble Falls ($10^6 m^3/yr$)	L. Travis ($10^6 m^3/yr$)	L. Austin ($10^6 m^3/yr$)	Town L. ($10^6 m^3/yr$)
1982	108.3	4.0	36.8	4.1	88.4	9.2	2.1
1981	112.8	4.0	35.5	3.9	88.3	7.9	2.0
1980	134.1	4.9	39.8	4.3	94.5	9.0	2.3
1979	119.8	4.3	38.2	4.2	95.4	8.8	2.2
1978	126.4	4.8	39.4	4.4	77.7	9.2	2.3
1977	132.8	4.8	40.7	4.5	105.4	9.6	2.5
1976	115.6	4.1	34.2	3.8	93.6	8.2	2.1
1975	116.7	4.2	35.7	3.6	101.5	8.2	2.2
1974	107.0	4.7	39.5	4.3	97.3	6.9	2.2
1973	75.0	3.0	27.7	3.0	71.4	6.7	1.6
1972	92.8	3.4	25.3	2.8	76.7	8.0	1.8
1971	94.7	3.7	22.8	3.2	67.3	7.4	1.8
1970	90.4	3.3	22.2	3.0	79.1	7.1	1.7
1969	94.8	3.4	27.5	2.9	75.5	5.4	1.7
1968	87.3	3.1	24.2	2.7	70.8	6.0	1.6
1967	102.2	3.9	30.3	3.1	74.2	7.6	2.0

TABLE 5-5 (Concluded)

Year	L. Buchanan ($10^6 m^3/yr$)	Inks L. ($10^6 m^3/yr$)	L. LBJ ($10^6 m^3/yr$)	L. Marble Falls ($10^6 m^3/yr$)	L. Travis ($10^6 m^3/yr$)	L. Austin ($10^6 m^3/yr$)	Town L. ($10^6 m^3/yr$)
1966	96.1	3.4	26.7	3.0	73.4	6.5	1.7
1965	84.2	3.1	26.1	3.1	74.8	7.9	1.8
1964	78.7	3.8	30.8	3.6	47.0	9.3	2.1
1963	87.7	4.1	33.4	3.8	68.8	9.8	2.2
1962	88.9	4.0	25.6	3.3	68.7	8.2	1.9
1961	97.1	3.6	23.9	3.0	74.9	7.8	1.8
1960	88.1	3.5	21.3	2.8	64.3	7.2	1.6
1959	96.6	3.6	18.8	2.9	72.6	7.5	1.7
1958	85.4	3.2	18.3	2.8	72.8	7.3	1.7
1957	78.1	2.9	21.8	2.7	69.9	6.9	1.6
1956	110.9	3.8	24.2	4.0	96.3	10.2	2.3
1955	105.3	3.9	31.9	3.6	88.1	9.3	2.1
1954	112.6	4.2	34.4	4.0	95.3	10.6	2.4
1953	95.0	3.8	30.0	3.4	88.0	8.7	2.0
1952	77.2	3.9	27.5	3.5	56.4	9.6	2.2

Note: Pan evaporation reported by LCRA.

TABLE 5-6

ANNUAL PUMPED WITHDRAWALS FROM LAKE AUSTIN AND TOWN LAKE
CITY OF AUSTIN WATER SUPPLY

Year	L. Austin ($10^6 m^3/yr$)	Town L. ($10^6 m^3/yr$)	Year	L. Austin ($10^6 m^3/yr$)	Town L. ($10^6 m^3/yr$)
1982	90.1	24.1	1966	39.8	13.5
1981	78.9	20.5	1965	42.7	11.1
1980	86.0	21.2	1964	38.2	15.0
1979	68.9	20.0	1963	37.5	18.8
1978	71.7	20.0	1962	26.5	21.1
1977	71.5	20.1	1961	31.8	10.5
1976	51.6	23.0	1960	29.5	13.2
1975	54.0	19.6	1959	27.8	9.5
1974	60.6	19.1	1958	23.1	13.6
1973	59.2	14.1	1957	19.7	15.8
1972	58.8	16.8	1956	21.0	24.5
1971	61.8	14.5	1955	11.9	26.4
1970	55.2	12.0	1954	2.9	32.4
1969	49.6	14.1	1953	--	30.7
1968	31.3	22.1	1952	—	30.4
1967	41.7	16.7			

Note:

1982 - 1967 from LCRA hydropower records.

1966 - 1951 from City of Austin file data.

TABLE 5-7

LAKE BUCHANAN WATER BUDGET

Year	Runoff Inflow from Water Budget ($10^6 m^3/yr$)	Rainfall ($10^6 m^3/yr$)	Budget Inflow Less Rainfall ($10^6 m^3/yr$)	Tributary Inflow from USGS Data ($10^6 m^3/yr$)	Runoff Inflow from SCS Method ($10^6 m^3/yr$)	Tributary Plus Runoff Inflow ($10^6 m^3/yr$)	Budget Minus Calculated Inflow ($10^6 m^3/yr$)	Difference in Inflow to Total Inflow
1982	594.8	46.9	547.9	602.2	21.8	624.0	-76.1	.13
1981	574.2	62.0	512.2	450.1	82.1	532.2	-20.0	.03
1980	754.8	50.3	704.5	675.2	40.9	716.1	-11.6	.02
1979	409.0	58.7	350.3	279.8	39.2	319.0	31.3	.08
1978	397.7	59.5	338.2	358.9	72.9	431.8	-93.6	.24
1977	751.8	52.0	699.8	656.9	88.0	744.9	-45.1	.06
1976	609.9	61.2	548.7	425.8	53.8	479.6	69.1	.11
1975	896.7	59.5	837.2	742.1	62.5	804.6	32.6	.04
1974	1010.6	72.9	937.7	1094.9	158.5	1253.4	-315.7	.31
1973	540.8	59.5	481.3	517.0	36.4	553.4	72.1	.13
1972	278.0	44.4	233.6	261.6	17.3	278.9	-45.3	.16
1971	1113.7	60.3	1053.4	1125.3	43.3	1168.6	-115.2	.10
1970	737.2	42.7	694.5	650.8	79.2	730.0	-35.5	.05
1969	768.4	74.6	693.8	669.1	96.1	765.2	-71.4	.09
1968	1536.7	80.4	1456.3	1301.7	102.2	1403.9	52.4	.03
1967	344.1	52.0	292.1	340.6	73.1	413.7	-121.6	.35
1966	526.3	41.1	485.2	462.3	36.3	498.6	-13.4	.03

TABLE 5-8

INKS LAKE WATER BUDGET

Year	Total Inflow from Water Budget ($10^6 m^3/yr$)	Runoff Inflow from Water Budget ($10^6 m^3/yr$)	Rainfall ($10^6 m^3/yr$)	Budget Inflow Less Rainfall ($10^6 m^3/yr$)	Runoff Inflow from SCS Method ($10^6 m^3/yr$)	Budget Minus Calculated Inflow ($10^6 m^3/yr$)	Difference in Inflow to Total Inflow
1982	609.1	7.8	1.8	6.0	1.9	4.1	< .01
1981	507.1	13.5	2.4	11.1	6.6	4.5	< .01
1980	609.5	16.3	2.0	14.3	3.4	10.9	.02
1979	131.4	25.1	2.3	22.8	3.4	19.4	.15
1978	360.2	1.4	2.3	-0.9	6.0	-6.9	.02
1977	746.7	6.6	2.0	4.6	7.0	-2.4	< .01
1976	477.0	15.0	2.4	12.6	4.5	8.1	.02
1975	843.8	16.9	2.3	14.6	5.3	-24.5	.03
1974	638.5	61.5	2.8	58.7	12.6	-76.9	.12
1973	424.2	15.5	2.3	13.2	3.0	10.2	.02
1972	438.0	12.3	1.7	10.6	1.5	9.1	.02
1971	891.8	16.6	2.3	14.3	3.7	10.6	.01
1970	850.0	35.3	1.7	33.6	6.3	27.3	.03
1969	615.4	15.4	2.9	12.5	7.9	4.6	< .01
1968	1337.1	11.4	3.1	8.3	8.3	0	0
1967	417.9	19.4	2.0	17.4	5.9	11.5	.03
1966	534.6	22.4	1.6	20.8	3.0	17.8	.03

TABLE 5-9

LAKE LBJ WATER BUDGET

Year	Total Inflow from Water Budget ($10^6 m^3/yr$)	Runoff Inflow from Water Budget ($10^6 m^3/yr$)	Rainfall ($10^6 m^3/yr$)	Budget Inflow Less Rainfall ($10^6 m^3/yr$)	Tributary Inflow from USGS Data ($10^6 m^3/yr$)	Runoff Inflow from SCS Method ($10^6 m^3/yr$)	Tributary Plus Runoff Inflow ($10^6 m^3/yr$)	Budget Minus Calculated Inflow ($10^6 m^3/yr$)	Difference in Inflow to Total Inflow
1982	795.0	192.2	14.4	177.8	172.0	6.7	178.7	-0.9	<.01
1981	1317.7	814.8	19.1	795.7	712.1	25.1	737.2	58.5	.04
1980	1044.0	439.4	15.5	423.9	455.1	12.5	467.6	-43.7	.04
1979	539.9	412.5	18.1	394.4	375.4	12.0	387.4	7.0	.01
1978	704.5	349.0	18.3	330.7	372.8	22.3	395.1	-64.4	.09
1977	1294.5	553.6	16.0	537.6	579.4	26.8	606.2	-68.6	.05
1976	861.6	387.8	18.8	369.0	380.1	16.4	396.5	-27.5	.03
1975	1513.7	674.8	18.3	656.5	634.2	18.9	653.1	3.4	<.01
1974	1339.5	705.3	22.4	682.9	801.2	37.7	838.9	-156.0	.12
1973	951.9	530.7	18.3	512.4	549.9	11.0	560.9	-48.5	.05
1972	669.2	234.5	13.7	220.8	263.3	5.2	268.5	-47.7	.07
1971	1371.2	497.6	18.6	479.0	503.4	13.1	516.5	-37.5	.03
1970	1289.5	452.6	13.2	439.4	482.2	24.0	506.2	-66.8	.05
1969	1103.9	499.5	23.0	476.5	556.0	29.0	585.0	-108.5	.10
1968	2139.2	813.2	24.8	788.4	704.0	30.8	734.8	53.6	.03
1967	523.9	132.0	16.0	116.0	135.9	22.1	158.0	-42.0	.08
1966	747.3	271.4	12.6	258.8	237.6	10.9	248.5	10.3	.01

TABLE 5-10

LAKE TRAVIS WATER BUDGET

Year	Total Inflow from Water Budget $(10^6 m^3/yr)$	Runoff Inflow from Water Budget $(10^6 m^3/yr)$	Rainfall $(10^6 m^3/yr)$	Budget Inflow Less Rainfall $(10^6 m^3/yr)$	Tributary Inflow from USGS Data $(10^6 m^3/yr)$	Runoff Inflow from SCS Method $(10^6 m^3/yr)$	Tributary Plus Runoff Inflow $(10^6 m^3/yr)$	Budget Minus Calculated Inflow $(10^6 m^3/yr)$	Difference in Inflow to Total Inflow
1982	960.6	169.8	42.6	127.2	74.5	62.0	136.5	-9.3	.01
1981	1970.1	617.8	72.7	545.1	403.2	211.5	614.7	-69.6	.04
1980	1182.1	158.6	43.9	114.7	70.1	30.3	100.4	14.3	.01
1979	1085.0	569.0	59.6	509.4	433.9	142.7	576.6	-67.2	.06
1978	959.4	276.1	49.5	226.6	322.1	84.3	406.4	-179.8	.19
1977	1809.6	497.4	35.1	462.3	370.3	27.6	397.9	64.4	.04
1976	1282.0	414.5	65.8	348.7	223.5	60.8	284.3	64.4	.05
1975	2329.8	765.3	58.3	707.0	504.0	131.6	635.6	71.4	.03
1974	2174.8	839.6	57.7	781.9	398.8	109.3	508.1	273.8	.13
1973	1440.4	475.6	64.6	411.0	260.8	114.5	375.3	35.7	.02
1972	885.6	207.5	41.4	166.1	149.0	50.1	199.1	-33.0	.04
1971	1333.1	107.0	39.5	67.5	151.2	21.0	172.2	-104.7	.08
1970	1905.5	475.6	48.9	426.7	276.1	69.6	345.7	81.0	.04
1969	1495.0	392.7	53.3	339.4	243.2	66.5	309.7	29.7	.02
1968	2916.9	798.2	61.4	736.8	293.6	45.8	339.4	397.5	.14
1967	627.0	121.2	38.9	82.3	72.[illegible]	80.0	152.3	-70.0	.11
1966	991.0	263.6	30.7	232.9	67.9	39.0	106.9	126.0	.13

TABLE 5-11

REGRESSION EQUATIONS FOR
CHLORIDE LOAD AND FLOW FOR
MAJOR TRIBUTARIES

Tributary	Equation	a Coefficient	b Coefficient	Standard Error	Correlation Coefficient
Colorado	$y = a + bx$	128.316	1.2588	260.349	0.58
Llano	$y = a + bx$	1.774	1.9176	10.114	0.90
Pedernales	$y = a + bx$	13.863	0.3296	16.510	0.92

Note: Load in 10^6 g/d, flow in m^3/s.

TABLE 5-12

CHLORIDE SUMMARY STATISTICS

MINOR TRIBUTARY MONITORING STATIONS

USGS St. No.	Description	Chloride Range (mg/l)	Average Chloride (mg/l)	Standard Deviation	Number of Data Points
08155200	Barton Cr. at Hwy 71	6-21	12	5	17
08155300	Barton Cr. at Loop 360	4-15	9	4	22
08158810	Bear Cr. below FM 1826	5-22	14	5	17
08158050	Boggy Cr. at Hwy 183	3-59	28	20	31
08154700	Bull Cr. at Loop 360	9-55	29	14	34
08158825	Little Bear Cr. at FM 1626	2-9	5	2	7
08158700	Onion Cr. near Driftwood	5-18	12	3	62
08158800	Onion Cr. at Buda	4-22	10	5	18
08159000	Onion Cr. at Hwy 183	10-99	39	24	44
08156800	Shoal Cr. at 12th St.	3-86	18	18	35
08158860	Slaughter Cr. at FM 2304	11-31	18	10	8
08158600	Walnut Cr. at Webberville Rd.	6-57	28	17	24
08158920	Williamson Cr. at Oak Hill	8-38	18	7	45
08158970	Williamson Cr. at Jimmy Clay Rd.	3-96	32	25	30

Note: Data from USGS (1983a), period of record 1975-1982.

TABLE 5-13

CHLORIDE SUMMARY STATISTICS FOR HIGH FLOW DATA MINOR TRIBUTARY MONITORING STATIONS

USGS St. No.	Description	Chloride Range (mg/l)	Average Chloride (mg/l)	Standard Deviation	Number of Data Points
08155200	Barton Cr. at Hwy 71	3-19	11	4	10
08155300	Barton Cr. at Loop 360	4-15	9	4	22
08158810	Bear Cr. below FM 1826	8-15	11	3	5
08158050	Boggy Cr. at Hwy 183	3-15	10	4	13
08154700	Bull Cr. at Loop 360	9-41	22	10	23
08158825	Little Bear Cr. at FM 1626	2-4	3	1	3
08158700	Onion Cr. near Driftwood	5-17	12	2	38
08158800	Onion Cr. at Buda	2-16	7	4	9
08159000	Onion Cr. at Hwy 183	10-93	26	19	26
08156800	Shoal Cr. at 12th St.	3-32	10	7	24
08158860	Slaughter Cr. at FM 2304	2-31	18	10	8
08158600	Walnut Cr. at Webberville Rd.	3-40	16	12	13
08158920	Williamson Cr. at Oak Hill	6-24	12	6	11
08158970	Williamson Cr. at Jimmy Clay Rd.	3-32	13	10	11

Note: Data from USGS (1983a), period of record 1975-1982.

TABLE 5-14

SUMMARY OF HISTORICAL CHLORIDE LOADINGS
TO THE HIGHLAND LAKES

Year	Chloride Loadings (10^6 g/yr)			
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE BUCHANAN -		
1982	0.2	261.6	55,625.6	23.5
1981	0.2	985.2	53,431.0	31.0
1980	0.1	490.8	56,732.6	25.2
1979	0.2	470.4	50,987.0	29.2
1978	0.4	874.8	52,061.4	29.8
1977	0.2	1056.0	56,428.1	26.0
1976	0.2	645.6	53,105.1	30.5
1975	0.4	750.0	57,687.8	29.8
1974	--	1902.0	62,801.2	36.5
1973	--	436.8	54,434.0	29.8
1972	--	207.6	50,640.1	22.2
1971	--	519.6	63,300.4	30.2
1970	--	950.4	56,395.7	21.5
1969	--	1153.2	56,662.6	37.2
1968	--	1226.4	65,756.1	40.2
		- INKS LAKE -		
1982	--	22.8	--	0.9
1981	--	79.2	--	1.2
1980	--	40.8	--	1.0
1979	--	40.8	--	1.2
1978	--	72.0	--	1.2
1977	--	84.0	--	1.0
1976	--	54.0	--	1.2
1975	--	63.6	--	1.2
1974	--	151.2	--	1.4
1973	--	36.0	--	1.2
1972	--	18.0	--	0.8
1971	--	44.4	--	1.2
1970	--	75.6	--	0.8
1969	--	94.8	--	1.4
1968	--	99.6	--	1.6

TABLE 5-14 (Cont'd)

Year	Chloride Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE LYNDON B. JOHNSON -		
1982	27.7	80.4	10,681.6	7.2
1981	13.8	301.2	13,880.8	9.5
1980	0.02	150.0	15,269.0	7.8
1979	0.004	144.0	6,343.4	9.0
1978	0.004	267.6	7,110.9	9.2
1977	0.004	321.6	8,857.8	8.0
1976	0.004	196.8	11,059.8	9.5
1975	0.004	226.8	5,528.3	9.2
1974	0.004	452.4	5,386.6	11.2
1973	—	132.0	14,226.1	9.2
1972	—	62.4	11,366.4	6.8
1971	—	157.2	10,644.2	9.2
1970	—	288.0	14,449.8	6.5
1969	—	348.0	24,111.3	11.5
1968	—	369.6	4,400.8	12.5
		- LAKE MARBLE FALLS -		
1982	51.6	42.0	—	0.9
1981	46.3	157.2	—	1.2
1980	41.3	78.0	—	1.0
1979	38.1	74.4	—	1.1
1978	25.3	139.2	—	1.1
1977	24.2	167.9	—	1.0
1976	19.8	102.0	—	1.2
1975	11.0	118.8	—	1.1
1974	11.0	301.2	—	1.4
1973	11.0	69.1	—	1.1
1972	11.0	33.0	—	0.8
1971	9.5	82.2	—	1.2
1970	9.5	150.0	—	0.8
1969	9.5	182.2	—	1.4
1968	9.5	193.7	—	1.5

TABLE 5-14 (Cont'd)

Year	Chloride Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- LAKE TRAVIS -		
1982	1.3	744.0	5,348.1	21.2
1981	1.8	2,538.0	6,605.0	36.2
1980	1.4	363.6	5,332.0	22.0
1979	0.8	1,712.4	6,723.0	29.8
1978	—	1,011.6	6,295.3	24.8
1977	—	331.2	6,480.2	17.5
1976	—	729.6	5,912.0	33.0
1975	—	1,579.2	6,984.9	29.2
1974	—	1,311.6	6,583.2	28.8
1973	—	1,374.0	6,058.5	32.2
1972	—	601.2	5,632.3	20.8
1971	—	252.0	5,636.4	19.8
1970	—	835.2	6,118.0	24.5
1969	—	798.0	5,990.6	26.8
1968	—	549.6	6,257.4	30.8
		- LAKE AUSTIN -		
1982	—	176.1	—	2.5
1981	—	607.5	—	4.2
1980	—	79.5	—	2.5
1979	—	409.5	—	3.5
1978	—	236.2	—	3.0
1977	—	72.0	—	2.0
1976	—	169.5	—	4.0
1975	—	370.0	—	3.5
1974	—	310.5	—	3.5
1973	—	323.9	—	3.8
1972	—	142.0	—	2.5
1971	—	51.1	—	2.2
1970	—	194.5	—	3.0
1969	—	176.6	—	3.2
1968	—	116.0	—	3.5

TABLE 5-14 (Concluded)

Year	Chloride Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric
		- TOWN LAKE -		
1982	—	279.5	1,200.0	0.6
1981	—	952.5	1,667.5	1.0
1980	—	138.0	1,047.5	0.6
1979	—	641.5	1,812.5	0.8
1978	—	379.2	775.0	0.6
1977	—	124.5	2,200.0	0.5
1976	—	273.0	2,150.0	0.9
1975	—	588.0	2,425.0	0.8
1974	—	487.5	2,100.0	0.8
1973	—	509.9	2,050.0	0.9
1972	—	223.5	2,125.0	0.6
1971	—	93.5	1,125.0	0.5
1970	—	309.5	2,100.0	0.6
1969	—	295.0	1,525.0	0.7
1968	—	202.5	2,025.0	0.8

TABLE 5-15

SIMULATION OF CHLORIDE CONCENTRATIONS FOR STEADY-STATE PERIODS

		1 Nov 73-30 Apr 74		1 Jul 75-30 Jun 76		1 Oct 75-31 Mar 76		1 Nov 80-28 Feb 81	
Reservoir	Station	Observed Conc. (mg/l)	Calculated Conc. (mg/l)	Observed Conc. (mg/l)	Calculated Conc. (mg/l)	Observed Conc. (mg/l)	Calculated Conc. (mg/l)	Observed Conc. (mg/l)	Calculated Conc. (mg/l)
Buchanan	1408.03	120		136-113		136		101	
	1408.01	102		100-97		109		85	
	avg.	111	135	≈110	125	122	119	93	111
Inks	1407.03	102		—		—		—	
	1407.01	102		109-84		109		85	
	avg.	102	106	≈92	119	109	107	85	98
LBJ	1406.03	50		—		—		60	
	1406.02	46		49-65		49		—	
	1406.01	44		55-66		55		55	
	avg.	47	46	≈61	57	52	56	58	56
Marble Falls	1405.02	43		—		—		—	
	1405.01	43		55-66		55		56	
	avg.	43	39	≈62	53	55	51	56	54
Travis	1404.06	45		55-44		55		55	
	1404.03	55		52-47		52		50	
	1404.01	45		51		51		51	
	avg.	48	44	≈49	42	53	56	52	58

TABLE 5-15 (Concluded)

		1 Nov 73-30 Apr 74		1 Jul 75-30 Jun 76		1 Oct 75-31 Mar 76		1 Nov 80-28 Feb 81	
Reservoir	Station	Observed Conc. (mg/l)	Calculated Conc. (mg/l)	Observed Conc. (mg/l)	Calculated Conc. (mg/l)	Observed Conc. (mg/l)	Calculated Conc. (mg/l)	Observed Conc. (mg/l)	Calculated Conc. (mg/l)
Austin	1403.03	55		54-53		54		52	
	1403.012	54		55-54		54		—	
	1403.01	54		55-53		55		52	
	avg.	54	45	≈54	42	54	52	52	54
Town	1402.09	≈53		51-54		51		—	
	1402.08	≈52		16-53		51		50	
	avg.	≈52	44	≈53	41	51	50	50	54

NOTE: Observed concentrations from sampling data near the end of each period; ranges are presented where data coincident with the period end are unavailable.

TABLE 5-16

ESTIMATED RESIDENCE TIMES
UNDER STEADY-STATE CONDITIONS

Reservoir	Residence Time (mo.)			
	1 Nov 73-30 Apr 74	1 Jul 75-30 Jun 76	1 Oct 75-31 Mar 76	1 Nov 80-28 Feb 81
Buchanan	162.0	80.8	146.9	121.9
Inks	2.8	1.4	2.6	2.0
Lyndon B. Johnson	9.2	5.0	7.1	6.2
Marble Falls	0.5	0.3	0.4	0.4
Travis	18.5	19.4	68.0	46.5
Austin	0.3	0.3	1.0	0.7
Town	0.04	0.05	0.2	0.08

TABLE 5-17

SIMULATION OF HISTORICAL ANNUAL
AVERAGE CHLORIDE CONCENTRATIONS

	Chloride Concentration (mg/l)													
	L. Buchanan		Inks L.		L. LBJ		L. Marble Falls		L. Travis		L. Austin		Town L.	
Year	Calc.	Obs.	Calc.	Obs.	Calc.	Obs.	Calc.	Obs.	Calc.	Obs.	Calc.	Obs.	Calc.	Obs.
1968	66	—	67	—	45	—	45	—	46	—	45	49	44	—
1969	79	—	78	—	66	—	64	—	51	—	53	49	51	—
1970	75	—	72	—	53	—	53	—	47	—	48	54	47	—
1971	74	—	74	—	63	—	62	—	56	—	60	66	59	—
1972	88	—	86	—	74	—	70	—	60	—	62	60	59	—
1973	105	—	101	—	60	—	58	—	58	—	60	57	57	—
1974	99	98	110	100	58	50	57	46	44	47	47	51	46	50
1975	83	98	85	96	53	66	50	66	40	52	39	49	38	45
1976	94	112	91	100	64	58	62	61	47	52	49	53	48	46
1977	86	—	86	—	59	—	56	—	42	—	42	58	42	56
1978	105	—	105	—	65	—	63	—	49	—	55	64	54	60
1979	139	—	112	—	41	—	40	—	45	—	49	—	46	43
1980	122	—	120	—	89	—	87	—	73	—	79	—	78	58
1981	118	97	116	90	57	50	54	52	53	50	53	50	51	49
1982	107	—	107	—	100	—	96	—	61	—	68	—	67	—

Note: Calculated versus observed chloride concentrations; observed concentrations are whole lake averages, minimum of 3 measurements per year.

Data from TNRIS (1983a).

TABLE 5-18

CALCULATED PHOSPHORUS REACTION RATES
STEADY-STATE ASSUMPTION

Reservoir	Assumed Steady-State Concentration (mg/l)	Reaction Rate (yr^{-1}) 1 Nov 73-Apr 74	1 Jul 75-30 Jun 76	1 Oct 75-31 Mar 76	1 Nov 80-28 Feb 81
Buchanan	i*	.26	.86	.46	.60
	ii	.36	.82	.64	.60
	iii	.44	1.41	.46	.96
	iv	.56	1.12	.64	.96
Inks	i	4.48	.97	-1.66	2.31
	ii	4.02	-.70	-1.42	.66
	iii	1.58	-4.07	-1.66	-.39
	iv	1.56	-2.46	-1.42	-1.50
LBJ	i	1.02	1.00	2.36	-.12
	ii	.76	1.86	1.72	.06
	iii	1.94	1.11	2.36	-.12
	iv	1.50	1.47	1.72	.06
Marble Falls	i	-11.06	-11.85	8.48	18.36
	ii	-9.02	-8.08	8.48	7.98
	iii	-16.08	-18.71	8.48	18.36
	iv	-13.90	-5.32	8.48	7.98
Travis	i	-.14	.76	.40	.48
	ii	-.20	.55	.34	.60
	iii	.16	1.02	.40	.48
	iv	-.16	.66	.40	.60
Austin	i	17.42	-6.22	-2.10	-.48
	ii	11.26	-27.30	-10.06	-.48
	iii	1.18	-6.22	.81	-.48
	iv	3.96	-.68	.81	-.48
Town	i	-16.56	-159.74	-58.06	44.94
	ii	-154.38	-34.07	114.10	44.94
	iii	-33.92	-14.60	-39.46	44.94
	iv	-182.92	-129.25	-36.70	44.94

*Notes:
i = period-ending concentration, average of all reservoir stations;
ii = period-average concentration, average of all reservoir stations;
iii = period-ending concentration, pool station;
iv = period-average concentration, pool station;

TABLE 5-19

EVALUATION OF TIME CONSTANTS FOR REPRESENTATIVE REACTION RATES

Reservoir	Estimated Reaction Rate (yr^{-1})	Outflow ($10^6 m^3/yr$)	Volume ($10^6 m^3$)	Outflow/ Volume (yr^{-1})	Time Constant (yr)	Three Time Constants (yr)
Buchanan	0.8	171.1	1,152.0	0.15	1.05	3.15
Inks	2.0	89.8	21.1	4.26	0.16	0.48
LBJ	1.5	410.1	169.8	2.42	0.26	0.77
Marble Falls	8.5	317.8	10.4	30.6	0.026	0.077
Travis	0.5	368.7	1,429.0	0.26	1.32	3.96
Austin	10.0	972.6	22.9	42.5	0.02	0.06
Town	45.0	420.6	4.3	97.8	0.007	0.021

TABLE 5-20

CALCULATED PHOSPHORUS REACTION RATES
PERIODS OF DECREASING CONCENTRATION

Reservoir	Period	Initial Conc. (mg/l)	Final Conc. (mg/l)	Volume ($10^6 m^3$)	Outflow ($10^6 m^3/t$)*	Mass Load ($10^6 g/t$)*	k (yr^{-1})
Buchanan	11 Apr 74-19 Jul 74	0.0261	<0.0098	1,028.8	37.5	10.719	5.8
	14 Jan 76-21 Apr 76	0.0261	0.016	1,150.3	13.3	4.719	2.3
Inks	11 Apr 74-19 Jul 74	0.0163	<0.0098	21.2	46.2	.868	7.3
	14 Jan 76-21 Apr 76	0.0392	0.0261	21.3	16.9	0.3	0.2
LBJ	14 Nov 73-23 Jan 74	0.0523	0.0131	171.7	67.5	1.39	6.7
	23 Jan 74-10 Apr 74	0.0131	0.0098	171.3	40.4	.999	2.8
Marble Falls	14 Nov 73-27 Jan 74	0.0784	0.0261	10.0	79.4	2.241	2.9
	11 Apr 74-18 Jul 74	0.0327	0.0163	10.3	163.7	2.163	-15.9
Travis	24 Jan 74-16 Apr 74	0.0196	0.0098	1,465.8	159.0	2.019	3.1
Austin	14 Nov 73-24 Jan 74	0.0196	0.0131	24.6	258.5	4.326	15.2
	24 Jan 74-16 Apr 74	0.0131	<0.0098	24.2	160.0	2.419	15.4
Town	17 Jan 74-24 Jan 74	0.1046	0.0654	4.3	34.5	.443	-283.1
	24 Jan 74-28 Mar 74	0.0654	0.0131	4.3	111.9	1.46	0.6-5.7
	22 May 74-08 Jul 74	0.0163	<0.0098	4.3	198.9	1.997	0.8-7.6

* t refers to the length of each period.

TABLE 5-21

CALCULATED PHOSPHORUS REACTION RATES
FOR LAKE LBJ USING CRWR DATA
PERIODS OF DECREASING CONCENTRATION

Period	Initial Conc. (mg/l)	Final Conc. (mg/l)	Volume ($10^6 m^3$)	Outflow ($10^6 m^3 t$)	Mass Load ($10^6 g/t$)	k Rate (yr^{-1})
24 Oct 71-20 Nov 71	0.123	0.075	168.4	124.0*	9.638	4.8
18 Dec 71-14 Jan 72	0.092	0.038	168.2	31.5	2.430	12.4
10 Jun 72-24 Jun 72	0.045	0.026	169.2	41.6	1.191	12.2
6 Aug 73-17 Aug 73	0.072	0.028	170.4	32.5	1.062	27.4
20 Oct 73-17 Nov 73	0.179	0.118	173.5	24.7*	0.674	3.8
11 Jan 74-13 Apr 74	0.160	0.050	171.3	48.4	1.801	3.9

* estimated, could be higher.

TABLE 5-22

SIMULATION OF PHOSPHORUS LEVELS IN LAKE BUCHANAN
AT VARYING K RATES

Year	Observed Conc.[1] (mg/l)	Observed Conc.[2] (mg/l)	Predicted Conc. for Various k Rates ($year^{-1}$)							
			2.0	3.0	4.0	5.0	6.0	7.0	8.0	9.0
1968	—	$.026^{5}$	.096	.074	.060	.051	.044	.038	.034	.031
1969	—	$.004^{1}$	.056	.039	.030	.024	.020	.018	.016	.014
1970	—	—	.037	.026	.020	.017	.014	.012	.011	.010
1971	—	—	.097	.073	.059	.049	.042	.036	.032	.029
1972	—	—	.018	.009	.006	.004	.004	.003	.003	.002
1973	$.041^{2}_{1}$	$.023^{1}_{1}$	.052	.038	.030	.025	.021	.018	.016	.014
1974	$.054^{8}_{2}$	$.016^{4}_{2}$	.099	.074	.058	.048	.041	.036	.032	.028
1975	$.019^{8}_{1}$	$.012^{4}_{1}$	.043	.030	.023	.019	.016	.014	.012	.011
1976	$.023^{6}_{1}$	$.017^{3}_{1}$	.024	.016	.012	.010	.008	.007	.006	.006
1977	$.035^{2}$	$.020^{1}$	.044	.033	.026	.022	.018	.016	.014	.013
1978	—	—	.035	.024	.018	.015	.013	.011	.010	.009
1979	—	—	.018	.012	.008	.007	.006	.005	.004	.004
1980	—	$.020^{1}$	.066	.049	.039	.032	.027	.024	.021	.019
1981	$.028^{6}_{1}$	$.017^{3}_{1}$	.031	.020	.015	.012	.010	.009	.008	.007
1982	—	—	.049	.036	.028	.023	.020	.017	.015	.014

Note: Observed Concentration[1] = average of stations within main body of reservoir for the year.

Observed Concentration[2] = average of pool station for the year.

Superscript indicates total number of measurements

Subscript indicates number of measurements at or below detection limit

TABLE 5-23

SIMULATION OF PHOSPHORUS LEVELS IN INKS LAKE
AT VARYING K RATES

Year	Observed Conc.[1] (mg/l)	Observed Conc.[2] (mg/l)	Predicted Conc. for Various k Rates ($year^{-1}$) .01	.1	1.0	2.0	3.0	6.0	8.0	10.0
1968	—	$.014^{5}$	.044	.044	.043	.042	.042	.040	.030	.024
1969	—	$.007^{1}$	.021	.021	.020	.020	.019	.017	.013	.010
1970	—	—	.014	.014	.014	.013	.013	.012	.009	.007
1971	—	—	.042	.042	.041	.040	.039	.037	.027	.021
1972	—	—	.004	.004	.004	.003	.003	.003	.002	.002
1973	$.046^{2}$	$.026^{1}$	.021	.021	.020	.019	.018	.016	.011	.009
1974	$.021^{8}_{2}$	$.016^{4}_{2}$	.046	.046	.044	.043	.042	.038	.028	.022
1975	$.014^{5}_{2}$	$.013^{4}_{2}$	.016	.016	.016	.016	.015	.014	.011	.008
1976	—	$.025^{3}_{1}$	.008	.008	.008	.008	.007	.006	.005	.004
1977	—	$.030^{1}$	.019	.019	.018	.018	.017	.016	.012	.009
1978	—	—	.013	.013	.013	.012	.011	.010	.007	.005
1979	—	—	.006	.006	.005	.004	.004	.003	.002	.002
1980	—	$.030^{1}$	.027	.027	.026	.025	.024	.022	.016	.012
1981	—	$.023^{3}$	.011	.011	.010	.010	.009	.009	.006	.005
1982	—	—	.020	.020	.019	.018	.018	.016	.012	.009

Note: **Observed Concentration [1] = average of stations within main body of reservoir for the year.**

Observed Concentration [2] = average of pool station for the year.

Superscript indicates total number of measurements.

Subscript indicates number of measurements at or below detection limit.

TABLE 5-24

SIMULATION OF PHOSPHORUS LEVELS IN LAKE LBJ
AT VARYING K RATES

Year	Observed Conc.[1] (mg/l)	Observed Conc.[2] (mg/l)	Predicted Conc. for Various k Rates ($year^{-1}$) 1.0	3.0	5.0	6.0	7.0	9.0	12.0
1968	$.012^{5}$	$.018^{5}$	.026	.022	.020	.019	.018	.016	.014
1969	$.002^{2}$	$.000^{1}$	.043	.034	.028	.026	.024	.021	.017
1970	—	—	.020	.017	.015	.014	.013	.012	.010
1971	—	—	.036	.030	.026	.025	.023	.021	.018
1972	—	—	.021	.014	.011	.010	.009	.008	.006
1973	$.042^{3}$	$.052^{1}$	.025	.019	.016	.014	.013	.011	.009
1974	$.016^{12}_{2}$	$.013^{4}_{2}$	.021	.017	.015	.014	.013	.011	.009
1975	$.012^{9}_{3}$	$.011^{4}_{3}$	.011	.009	.008	.007	.007	.006	.005
1976	$.010^{6}_{2}$	$.011^{3}_{2}$	.020	.015	.012	.011	.010	.009	.007
1977	$.030^{2}$	$.030^{1}$	.017	.014	.011	.010	.010	.009	.007
1978	—	—	.015	.011	.008	.007	.007	.006	.005
1979	—	—	.011	.008	.006	.005	.004	.004	.003
1980	$.020^{2}$	$.020^{1}$	.031	.024	.020	.018	.017	.014	.012
1981	$.023^{6}$	$.020^{3}$	.019	.016	.013	.012	.011	.010	.008
1982	—	—	.028	.020	.016	.014	.013	.011	.009

Note: Observed Concentration [1] = average of stations within main body of reservoir for the year.

Observed Concentration [2] = average of pool station for the year.

Superscript indicates total number of measurements.

Subscript indicates number of measurements at or below detection limit.

TABLE 5-25

SIMULATION OF PHOSPHORUS LEVELS IN LAKE MARBLE FALLS AT VARYING K RATES

Year	Observed Conc.[1] (mg/l)	Observed Conc.[2] (mg/l)	Predicted Conc. for Various k Rates ($year^{-1}$) .1	5.0	6.0	7.0	12.0
1968	—	$.001^{5}$	.019	.019	.019	.019	.018
1969	—	$.003^{1}$	.026	.025	.025	.025	.024
1970	—	—	.015	.014	.014	.014	.013
1971	—	—	.025	.024	.024	.024	.023
1972	$.027^{4}$	—	.011	.010	.010	.010	.010
1973	$.046^{4}$	$.078^{1}$	.015	.014	.014	.014	.014
1974	$.020^{8}_{1}$	$.024^{4}$	.015	.014	.014	.014	.014
1975	$.012^{4}_{2}$	$.011^{3}_{2}$	.008	.008	.007	.007	.007
1976	—	$.015^{3}_{2}$	.013	.012	.012	.012	.012
1977	—	$.020^{1}$	.012	.012	.012	.012	.011
1978	—	—	.012	.011	.011	.011	.010
1979	—	—	.012	.012	.012	.012	.012
1980	—	$.030^{1}$	.022	.021	.021	.020	.020
1981	—	$.020^{3}$	.015	.015	.015	.014	.014
1982	—	—	.021	.020	.019	.019	.018

Note: Observed Concentration [1] = average of stations within main body of reservoir for the year.

Observed Concentration [2] = average of pool station for the year.

Superscript indicates total number of measurements.

Subscript indicates number of measurements at or below detection limit.

TABLE 5-26

SIMULATION OF PHOSPHORUS LEVELS IN LAKE TRAVIS
AT VARYING K RATES

Year	Observed Conc.^{1} (mg/l)	Observed Conc.^{2} (mg/l)	Predicted Conc. for Various k Rates (year^{-1})				
			1.0	3.0	5.0	6.0	7.0
1968	$.008^{11}$	$.009^{6}$	.013	.008	.006	.005	.004
1969	$.010^{2}$	$.006^{1}$	.016	.008	.005	.004	.004
1970	—	—	.012	.006	.004	.003	.003
1971	—	—	.014	.008	.005	.004	.004
1972	—	—	.009	.004	.002	.002	.002
1973	$.021^{4}$	$.013^{1}$	.011	.005	.004	.003	.003
1974	$.038^{15}_{5}$	$.016^{4}_{1}$	.012	.007	.005	.004	.004
1975	$.011^{13}_{9}$	$.011^{4}_{3}$	.011	.006	.004	.004	.003
1976	$.010^{9}_{8}$	$.010^{3}_{3}$	.010	.004	.003	.002	.002
1977	$.017^{3}_{2}$	$.010^{1}_{1}$	.011	.006	.004	.004	.003
1978	—	—	.015	.008	.005	.004	.004
1979	—	—	.016	.007	.004	.004	.003
1980	$.010^{3}_{3}$	$.010^{1}_{1}$	.012	.005	.004	.003	.003
1981	$.012^{9}_{7}$	$.010^{3}_{3}$	.015	.008	.006	.005	.004
1982	—	—	.009	.004	.003	.002	.002

Note: Observed Concentration ^{1} = average of stations within main body of reservoir for the year.

Observed Concentration ^{2} = average of pool station for the year.

Superscript indicates total number of measurements.

Subscript indicates number of measurements at or below detection limit.

TABLE 5-27

SIMULATION OF PHOSPHORUS LEVELS IN LAKE AUSTIN
AT VARYING K RATES

Year	Observed Conc.[1] (mg/l)	Observed Conc.[2] (mg/l)	Predicted Conc. for Various k Rates ($year^{-1}$) 0.1	1.0	3.0	6.0	7.0
1968	—	$.004^{5}$	.005	.005	.005	.005	.005
1969	—	$.022^{1}$	.005	.005	.005	.005	.004
1970	—	—	.004	.004	.004	.004	.003
1971	—	—	.005	.005	.004	.004	.004
1972	—	—	.003	.002	.002	.002	.002
1973	$.015^{3}_{1}$	$.020^{1}$	.004	.004	.004	.004	.003
1974	$.014^{12}_{5}$	$.011^{4}_{2}$	.005	.005	.005	.005	.004
1975	$.043^{9}_{3}$	$.012^{4}_{2}$	.004	.004	.004	.004	.004
1976	$.013^{6}_{3}$	$.011^{3}_{2}$	.003	.003	.003	.003	.003
1977	$.015^{4}_{2}$	$.010^{1}_{1}$	.004	.004	.004	.004	.003
1978	$.025^{4}_{1}$	$.025^{2}_{1}$	.006	.006	.005	.005	.004
1979	$.012^{10}_{8}$	$.012^{3}_{2}$	.006	.006	.005	.005	.004
1980	$.010^{17}_{17}$	$.010^{4}_{4}$	.004	.004	.003	.003	.003
1981	$.012^{6}_{5}$	$.013^{3}_{2}$	.006	.006	.006	.005	.005
1982	—	—	.003	.003	.003	.003	.003

Note: Observed Concentration [1] = average of stations within main body of reservoir for the year.

Observed Concentration [2] = average of pool station for the year.

Superscript indicates total number of measurements.

Subscript indicates number of measurements at or below detection limit.

TABLE 5-28

SIMULATION OF PHOSPHORUS LEVELS IN TOWN LAKE
AT VARYING K RATES

Year	Observed Conc.[1] (mg/l)	Observed Conc.[2] (mg/l)	Predicted Conc. for Various k Rates ($year^{-1}$) .10	1.0	3.0	6.0	9.0
1968	—	$.009^{5}$	.006	.006	.006	.006	.006
1969	—	$.064^{1}$	.008	.008	.008	.008	.008
1970	—	—	.006	.006	.006	.006	.006
1971	—	—	.006	.006	.006	.006	.006
1972	—	—	.007	.007	.007	.007	.006
1973	$.034^{2}$	$.039^{1}$	.010	.010	.010	.009	.009
1974	$.027^{11}_{4}$	$.038^{6}_{1}$	.008	.008	.008	.008	.008
1975	$.017^{11}_{5}$	$.017^{7}_{2}$	.007	.007	.007	.007	.007
1976	$.046^{10}_{3}$	$.026^{7}_{2}$	.007	.007	.007	.007	.007
1977	$.016^{6}_{5}$	$.017^{4}_{3}$	.005	.005	.005	.005	.005
1978	$.013^{6}_{4}$	$.019^{3}_{1}$	.012	.012	.011	.011	.011
1979	$.202^{6}_{2}$	$.267^{3}_{1}$	.016	.016	.016	.016	.016
1980	$.022^{9}_{6}$	$.023^{5}_{3}$	.007	.007	.006	.006	.006
1981	—	$.010^{3}_{3}$	.012	.012	.012	.012	.012
1982	—	—	.007	.007	.007	.007	.007

Note: Observed Concentration [1] = average of stations within main body of reservoir for the year.

Observed Concentration [2] = average of pool station for the year.

Superscript indicates total number of measurements.

Subscript indicates number of measurements at or below detection limit.

TABLE 5-29

PHOSPHORUS REMOVAL RATES CALCULATED FOR

THE HIGHLAND LAKES

Reservoir	Sedimentation Coefficient (1/yr)	Apparent Settling Velocity (m/yr)	Mean Depth (m)	Flushing Rate (1/yr)
Buchanan	6.0	78.6	13.1	0.41
Inks	6.0	40.2	6.7	22.64
LBJ	6.0	39.6	6.6	5.82
Marble Falls	6.0	20.4	3.4	97.83
Travis	6.0	123.0	20.5	0.98
Austin	6.0	12.6	2.1	54.79
Town	6.0	15.6	2.6	301.53

CHAPTER 6.0

SIMULATION RESULTS AND DISCUSSION

6 1 ALTERNATIVE CONDITIONS

The model HILAKES was employed for simulation of future phosphorus concentrations in the Highland Lakes under several sets of alternative conditions for the period 1984-2000. The scenarios represent a range of future development conditions, formulated to depict conditions that may potentially occur, as well as demonstrate the effects of loadings to upstream reservoirs upon water quality downstream. Alternative scenarios are described below.

Scenario I

The baseline scenario for projected future conditions (that which is anticipated to occur) was described in detail in Chapter 4.0. The basis for Scenario I was projected population growth in the counties contiguous to the Highland Lakes. Projected population growth was used to estimate increases in urbanized area within the adjacent watersheds of the series of reservoirs. The increase in urbanized area produced an increase in the phosphorus loading originating from runoff from adjacent watersheds. Point source phosphorus loads were also incremented in accordance with projected population growth. As discussed in Chapter 4.0, hydrologic conditions for the period 1967-1983 were assumed to be repeated for the period 1984-2000 to formulate the baseline conditions. With this assumption, phosphorus loadings from major tributaries and atmospheric loads did not increase over the historic conditions.

Scenario II

The second scenario was a variation of Scenario I, in that increases in urbanized area in the Highland Lakes watershed were based upon projected population growth. However, for Scenario II, the phosphorus concentration in runoff from urban areas was set at 0.30 mg/l, a 100 percent increase over the previously assigned value of 0.15 mg/l. The assumptions for Scenario I were retained for all other loads and parameters.

Scenario III

In Scenario III, all assumptions in Scenario I concerning urbanized area and external loadings were retained with a single exception. A point source load equivalent to a population of 10,000 (2.8×10^6 g/yr, at an assumed effluent phosphorus level of 2 mg/l) was added to Lake LBJ. This scenario was designed to examine the effects of an intensive residential development in the watershed of one of the Highland Lakes, both upon phosphorus levels within that reservoir and within reservoirs downstream.

Scenario IV

The fourth scenario retained the assumptions of Scenario III. In addition to the 10,000-person point source load to Lake LBJ, a supplemental point source load equivalent to a population of 50,000 (13.8×10^6 g/yr) was added to Lake Travis. With this scenario, the effects of hypothetical intensive residential developments on Lake LBJ and Lake Travis were examined, with respect to phosphorus levels within those reservoirs as well as downstream waterbodies.

Scenario V

Scenario V represented a substantial departure from the baseline assumptions for urbanization of the watersheds of the lower reservoirs. Instead of the baseline assumption that urbanized area will increase in accordance with projected population growth, it was hypothesized that the entire adjacent watershed of Lake Travis, Lake Austin, and Town Lake would become completely urbanized. All other loading assumptions were compatible with Scenario I. This scenario was formulated to evaluate the potential effects of a hypothetical maximum degree of residential development in the area surrounding the lower reservoirs.

Scenario VI

The sixth scenario was identical to Scenario V with one exception. In addition to the assumed complete urbanization of the watersheds of the lower reservoirs, a supplemental point source load was introduced into Lake Travis, equivalent to the phosphorus load from a facility or facilities treating the wastewater from a population of 100,000 (27.6×10^6 g/yr). This scenario thus examined the effects of a very intensive level of urbanization, supplemented with a substantial point source discharge to Lake Travis.

Scenario VII

Scenario VII was devised to investigate the increment in phosphorus loading required to effect a specific increase in predicted concentration within each reservoir. The required loading was defined as the magnitude of the mass input that resulted in an increase in predicted phosphorus concentration of at least 0.010 mg/l within the reservoir directly receiving the load for a majority of the years in the period 1984-2000. Except for the supplemental phosphorus loading, all other input parameters were identical to the assumptions in Scenario I. The supplemental phosphorus loadings were input as point source loads; however, the simulation results are independent of the loading categorization, and the supplemental load can be considered as originating from any of the potential sources. For Scenario VII, the reservoirs were analyzed in sequential order, and the model HILAKES was exercised in a trial and error process for each waterbody to identify the loading required to achieve the target level of effects. The supplemental loadings were not accumulated; that is, only a single supplemental load was considered in the sequential analysis of the chain of reservoirs.

Scenario VIII

In a manner similar to the previous scenario, the reservoirs were subjected to supplemental phosphorus loadings above the baseline conditions. Instead of a target level increase of 0.010 mg/l phosphorus within the reservoir receiving the load, the loading was adjusted to determine the mass input required to elevate concentrations in the reservoir immediately downstream by 0.010 mg/l. As with the previous scenario, the target effect was required for a majority of the years in the period 1984-2000.

Scenario IX

Scenario IX was formulated to investigate the effects of imposition of urban runoff controls upon the baseline predictions. The assumptions of Scenario I were retained, thus, the urbanized area within the various watersheds was estimated in accordance with projected population growth. In the only exception to the baseline assumptions, the runoff phosphorus loads derived from urban areas were reduced by 50 percent. (The runoff load from the lower of the two subwatersheds defined for Town Lake was not reduced, since urbanization of the watershed is already virtually complete.) The reduction in urban loads constitutes the simulated imposition of runoff control measures. For example, detention basins and filtration ponds represent feasible control measures.

Scenario X

In a manner similar to the preceding scenario, the imposition of runoff control measures was evaluated in Scenario X. This scenario retained the basic assumptions of Scenario V, which evaluated the complete urbanization of the adjacent watersheds of Lake Travis, Lake Austin, and Town Lake. To simulate runoff controls, runoff phosphorus loads from urban areas were reduced by 50 percent. (As with the preceding scenario, the runoff load from the lowermost watershed of Town Lake was not reduced.)

6.2 PROJECTED PHOSPHORUS CONCENTRATIONS

Predicted average phosphorus concentrations in the Highland Lakes are described below for each alternative scenario formulated in the preceding section.

Scenario I

Simulated phosphorus concentrations in the Highland Lakes for Scenario I are displayed in Table 6-1. Scenario I represents projected development conditions based upon a systematic increase in urbanization in accordance with projected population growth, and constitutes the future baseline predictions. The projected phosphorus concentrations for the period 1984-2000 typically show nominal increases above the simulated levels for 1967-1983, the base period for simulation of future conditions, which are presented in Table 6-2. In Lake Buchanan and Inks Lake, predicted future phosphorus concentrations are essentially identical to the historical baseline levels. Increments of 0.000-0.002 mg/l phosphorus are projected for Lake LBJ. In Lake Marble Falls, predicted increases range from 0.002-0.008 mg/l. Phosphorus concentrations in Lake Travis are projected to increase by 0.000-0.001 mg/l above historical conditions. In Lake Austin, increases of 0.000-0.002 mg/l are predicted. Projected increases in phosphorus levels in Town Lake range from 0.000-0.003 mg/l. Predicted phosphorus ranges are as follows:

Lake Buchanan	0.004 - 0.042 mg/l
Inks Lake	0.003 - 0.038 mg/l
Lake LBJ	0.006 - 0.027 mg/l
Lake Marble Falls	0.012 - 0.030 mg/l
Lake Travis	0.002 - 0.005 mg/l
Lake Austin	0.003 - 0.007 mg/l
Town Lake	0.006 - 0.018 mg/l

There is no indication of a persistently substantial elevation in phosphorus concentrations within the series of reservoirs under the Scenario I conditions. The projected increases in urbanized area for future conditions resulted in increases in the runoff mass loading from adjacent watersheds. However, the projected increase in urban loading is generally insignificant when compared to the overall runoff mass loading from all existing types of land uses. Further, the total runoff mass loading does not usually comprise the major fraction of the total external mass loading, with the exception of loadings to Town Lake. Phosphorus loads from major tributaries and releases from upstream reservoirs typically account for the major portion of the total load within the chain of reservoirs, and are often an order of magnitude greater than runoff loads and atmospheric loads. In addition, the point source loading of phosphorus represents a substantial fraction of the total load to Lake Marble Falls.

Scenario II

Scenario II was formulated to examine the effects of increased urban mass loads upon reservoir phosphorus levels. Simulated phosphorus concentrations under Scenario II assumptions, presented in Table 6-3, display little departure from the results shown in Table 6-1 for Scenario I. In response to the increase in urban runoff phosphorus concentration to 0.30 mg/l, small elevations in reservoir phosphorus levels over baseline (Scenario I) conditions were occasionally predicted. Phosphorus concentrations were not elevated more than 0.001 mg/l over baseline levels.

The limited effect of the increased phosphorus concentration in urban runoff is attributable to two factors. First, runoff loads from adjacent watersheds do not usually comprise the major portion of the total external mass loadings to the reservoirs. Second, of the components of the total phosphorus loading from runoff, urban area generally represents a small fraction of the entire watershed area.

Scenario III

The effects of a hypothetical moderate-size point source load to Lake LBJ, with respect to phosphorus concentrations in Lake LBJ and downstream reservoirs, were investigated in Scenario III. Simulation results for Scenario III are displayed in Table 6-4. Insertion of the 10,000-person point source load to Lake LBJ increased predicted phosphorus concentrations above baseline (Scenario I) conditions by 0.001-0.002 mg/l within the reservoir. In addition, phosphorus concentrations downstream in Lake Marble Falls showed comparable elevations. Slight elevations were also occasionally observed farther downstream.

The effect of the additional point source loading was generally small. The extent of the effect is dependent upon the magnitude of the point source contribution, relative to the other sources of mass loadings. For example, in years with low tributary inflow and little rainfall, direct point source loads can comprise the major portion of the mass loading to Lake LBJ. However, loadings associated with releases from upstream reservoirs usually represent the phosphorus input of greatest magnitude.

Scenario IV

The effects of hypothetical moderate-size point source loads to Lake LBJ and Lake Travis were examined in Scenario IV. Simulated phosphorus concentrations in the Highland Lakes under Scenario IV are presented in Table 6-5. The effects of the supplemental point source load to Lake LBJ were described for Scenario III. Addition of a substantial point source load to Lake Travis, equivalent to a population of 50,000, resulted in projected increases in phosphorus levels within the reservoir of 0.001-0.003 mg/l above Scenario I levels. Concomitant increases in concentration of 0.001-0.002 mg/l were predicted for Lake Austin and Town Lake.

The effects of the hypothetical point source discharge to Lake Travis can be interpreted in terms of the percent increase in ambient levels, as well as the ultimate magnitude of the predicted concentrations. The supplemental mass load resulted in generally 20-100 percent elevations in phosphorus concentration in Lake Travis above baseline (Scenario I) levels. However, even with the elevations in concentration, the predicted average phosphorus levels within Lake Travis are consistently below 0.010 mg/l. The large volume of the reservoir enables it to assimilate substantial mass loadings yet remain at a relatively low phosphorus concentration. Phosphorus levels in Lake Austin downstream also remain below 0.010 mg/l, with increments in concentration of 15-70 percent.

Despite the increases in point source loading, releases from upstream reservoirs usually constitute the major fraction of the external mass loading. However, the hypothetical point source load to Lake Travis is generally on the same order of magnitude as the loading from the upstream reservoir. In years with high inflow, tributary loads can also achieve the order of magnitude of point source and upstream release loadings.

Scenario V

The effects of widespread urbanization of the watersheds of Lake Travis, Lake Austin, and Town Lake were investigated in Scenario V. Results of the simulation are displayed in Table 6-6. The assumption of complete urbanization produced increases in phosphorus concentration of 0.000- 0.004 mg/l within Lake Travis, 0.001-0.008 mg/l within Lake Austin, and 0.001-0.012 mg/l within Town Lake. In terms of percent increases over baseline levels, the results indicate elevations of 0-100 percent in Lake Travis, 15-130 percent in Lake Austin, and 15-80 percent in Town Lake. In terms of the magnitude of the predicted levels, phosphorus concentrations in Lake Travis remain below 0.010 mg/l. The maximum predicted concentration in Lake Austin is 0.014 mg/l, though the levels are usually below 0.010 mg/l. Concentrations are generally higher in Town Lake, with a predicted maximum phosphorus level of 0.030 mg/l.

Of the total external mass loading to Lake Travis releases from the upstream reservoir usually provide the major portion under the Scenario V assumptions. Runoff loads from the adjacent watershed or tributary loads are the largest contributor of phosphorus in other years. These three sources are typically of the same order of magnitude for Lake Travis. Upstream releases generally constitute the major source of phosphorus loading to Lake Austin, through runoff loads from the contiguous watershed are frequently of the same order of magnitude. For Town Lake, the major source of phosphorus loading is usually the release from the upstream reservoir. Runoff loads from the adjacent watershed are occasionally the largest contributor, and are typically of the same order of magnitude as the upstream release phosphorus loads. With the assumed complete level of urbanization in the watersheds of the lower reservoirs, the magnitude of the runoff load is primarily dependent upon the assumed precipitation characteristics -- the largest loads are associated with the largest storm events. Thus, runoff loads can comprise a substantial fraction of the total external mass loading in years with a large number of runoff-generating storm events.

Scenario VI

The effects of a large point source discharge to Lake Travis, equivalent to a population of 100,000, in conjunction with widespread urbanization of the watersheds of Lake Travis, Lake Austin, and Town Lake, were examined with Scenario VI. Simulation results are shown in Table 6-7. The combination of the intensified urbanization and the supplemental point source load produced the following ranges of phosphorus concentration increments above baseline levels: 0.003-0.007 mg/l in Lake Travis, 0.004-0.011 mg/l in Lake Austin, and 0.004-0.015 mg/l in Town Lake. Recognizing the contributions from the urbanization assumption discussed under the preceding scenario, it is evident that the supplemental point source accounted for concentration increments of approximately 0.003 mg/l in Lake Travis and the reservoirs downstream. Percent increases in baseline concentrations under Scenario VI include 75-300 percent in Lake Travis, 70-270 percent in Lake Austin, and 70-120 percent in Town Lake. In terms of the concentrations ultimately realized, a maximum average phosphorus level of 0.012 mg/l was predicted for Lake Travis, with most of the predicted concentrations below 0.010 mg/l. For Lake Austin, a maximum phosphorus concentration of 0.017 mg/l was predicted. The maximum concentration in Town Lake was estimated to be 0.033 mg/l.

With the large supplemental load, the point source contribution usually accounts for the major portion of the total external mass loading to Lake Travis, and typically displays the same order of magnitude as the tributary and runoff loadings. Upstream releases generally constitute the major fraction of the total phosphorus loading to Lake Austin and Town Lake. During years with substantial precipitation events, runoff loads can potentially represent the largest source of phosphorus.

Scenario VII

The magnitudes of supplemental loadings required to effect a 0.010 mg/l minimum increase in phosphorus concentration above baseline levels within each reservoir were evaluated in Scenario VII. Reservoirs were examined sequentially, with the loading supplement assigned to only one reservoir at a time. The criterion for specifying the required phosphorus load was that the 0.010 mg/l minimum increment in concentration be achieved in a majority of the years comprising the 17-year simulation period. Phosphorus loads effecting the target concentration increment were determined as follows:

Lake Buchanan	69×10^6 g/yr
Inks Lake	7×10^6 g/yr
Lake LBJ	19×10^6 g/yr
Lake Marble Falls	11×10^6 g/yr
Lake Travis	85×10^6 g/yr
Lake Austin	13×10^6 g/yr
Town Lake	12×10^6 g/yr

For perspective, the point source load added to Lake Travis under Scenario VI was 27.6×10^6 g/yr, and was equivalent to an effluent phosphorus load from a population of 100,000 (at an effluent concentration of 2 mg/l). Under the supplemental loadings identified above, the predicted average phosphorus concentrations in the respective reservoirs displayed the following ranges:

Lake Buchanan	0.014 - 0.051 mg/l
Inks Lake	0.015 - 0.047 mg/l
Lake LBJ	0.016 - 0.036 mg/l
Lake Marble Falls	0.017 - 0.051 mg/l
Lake Travis	0.012 - 0.019 mg/l
Lake Austin	0.009 - 0.022 mg/l
Town Lake	0.010 - 0.032 mg/l

Predicted concentrations for each reservoir under its respective supplemental phosphorus loading are shown in Table 6-8 (note that this tabulation does not include consideration of the cumulative effects of a loading upon reservoirs downstream).

Under the supplemental loads described above, measurable effects upon phosphorus concentration in downstream reservoirs were observed. Examples of the sequential effects are displayed in Tables 6-9 through 6-11, presenting the results of supplemental loads to Lake Buchanan, Lake LBJ, and Lake Travis, respectively.

The effects of the supplemental loading to Lake Buchanan on downstream reservoirs are shown in Table 6-9. As expected, the greatest effect is exerted upon phosphorus concentrations in Inks Lake immediately below Lake Buchanan. Phosphorus concentrations are elevated by 0.004- 0.009 mg/l above the baseline (Scenario I) predictions in Inks Lake. Observed effects in reservoirs farther downstream include increases of 0.001-0.004 mg/l in Lake LBJ and 0.000-0.004 mg/l in Lake Marble Falls. Elevations of phosphorus concentration of 0.001 mg/l are predicted for roughly half of the years in the simulation period in Lake Travis, and similar increments are occasionally observed in Lake Austin and Town Lake. The magnitude and longitudinal extent of the effects on downstream reservoirs is dependent upon the total external loading to each reservoir and the outflow through the series of reservoirs. Where the total external loading is dominated by the loading associated with releases from an upstream reservoir, the effect is potentially more substantial. In addition, the downstream effect is more substantial with larger releases from the upstream reservoir.

In Table 6-10 are displayed the effects of the supplemental load to Lake LBJ upon downstream reservoirs. Phosphorus concentrations in Lake Marble Falls show increases of 0.006-0.011 mg/l. Increases in reservoirs below Lake Marble Falls were less substantial, ranging from 0.000-0.002 mg/l in Lake Travis, Lake Austin, and Town Lake. The elevations in concentration in Lake Marble Falls are equal to or slightly less (by usually 0.001 mg/l) than the concentration increases predicted in Lake LBJ as a consequence of the supplemental load. Concentrations within Lake Marble Falls are influenced primarily by the loads associated with releases from Lake LBJ upstream.

The effects of a large supplemental loading to Lake Travis upon phosphorus levels in Lake Austin and Town Lake are presented in Table 6-11. Concentrations in Lake Austin resemble closely the phosphorus concentrations in Lake Travis, due to the throughflow-dominated nature of the lower reservoir. Predicted concentration increments in Lake Austin are in the range of 0.008-0.013 mg/l, and range from 0.007-0.012 mg/l in Town Lake. External phosphorus loadings to both Lake Austin and Town Lake are dominated by upstream releases.

Scenario VIII

Under Scenario VIII, the magnitudes of supplemental loadings to individual reservoirs required to produce a 0.010 mg/l increase in phosphorus concentration within the reservoir immediately downstream were determined. As with Scenario VII, the concentration increment was required in a majority of the years comprising the 17-year simulation period. Magnitudes of the phosphorus loads were as follows:

Lake Buchanan	84×10^6 g/yr
Inks Lake	25×10^6 g/yr
Lake LBJ	22×10^6 g/yr
Lake Marble Falls	90×10^6 g/yr
Lake Travis	85×10^6 g/yr
Lake Austin	13×10^6 g/yr

The phosphorus inputs listed above are generally larger than the loadings required to produce increases of 0.010 mg/l within each reservoir receiving the direct load supplement, as described for Scenario VII. Considering the results of Scenario VII and VIII, two alternative mechanisms have been defined for increasing the phosphorus level in a particular reservoir by 0.010 mg/l: either the load identified in Scenario VII may be input directly to the particular reservoir, or, the loads calculated under this scenario may be input to the reservoir upstream.

Scenario IX

The effects of runoff control measures upon the predicted baseline (Scenario I) phosphorus concentrations in the reservoirs were evaluated in Scenario IX. As described for Scenario I, the predicted baseline phosphorus levels in the reservoirs were generally low, without substantial elevation over historical conditions. With the assumed 50 percent reduction in runoff loads from urban areas, only very small decreases in baseline phosphorus levels were occasionally observed within the Highland Lakes, except for Town Lake, where more substantial decreases were prevalent. The maximum reduction in concentration was 0.001 mg/l in reservoirs other than Town Lake; phosphorus levels in Town Lake were reduced by 0.000-0.006 mg/l, as shown in Table 6-12.

Under the baseline conditions, runoff control measures generally have little effect upon predicted reservoir phosphorus levels. The effect is negligible due to the relatively small proportion of urbanized area within the entire adjacent watersheds of the Highland Lakes. The urban runoff loads are generally small compared to external phosphorus loadings from other sources. An exception is Town Lake, where reduction in urban loads demonstrated a more substantial effect due to the large degree of urbanization and the relative magnitude of runoff loadings.

Scenario X

Under Scenario X, the effects of runoff control measures were examined with a hypothetical intensive level of urban development. As with Scenario V, it was assumed that the adjacent watersheds of Lake Travis, Lake Austin, and Town Lake were completely urbanized. Control measures were simulated via a 50 percent reduction in urban runoff loads. Predicted concentrations are displayed in Table 6-13. Compared to the baseline conditions of Scenario I, the average reservoir concentrations increased by 0.000-0.002 mg/l in Lake Travis, 0.000-0.003 mg/l in Lake Austin, and 0.000-0.004 mg/l in Town Lake. Compared to the predicted concentrations under complete urbanization without runoff controls, imposition of a 50 percent reduction in urban loads generally resulted in a reduction of roughly 50 percent or less in the magnitude of the reservoir concentration increases. Reductions in concentration vis-a-vis the case of complete urbanization without controls displayed the following ranges: 0.000-0.002 mg/l in Lake Travis, 0.000-0.005 mg/l in Lake Austin, and 0.001-0.008 mg/l in Town Lake.

The effect of runoff control measures upon reservoir phosphorus levels is dependent upon the relative magnitude of the urban loading. Where the urban load comprises a substantial fraction of the total external mass load, runoff controls can exert a more significant effect. Under the urbanization assumptions of Scenario X, phosphorus loads associated with releases from upstream reservoirs usually constitute the primary source of external loadings to Lake Travis and Lake Austin for the simulation period. Runoff loads from adjacent watersheds are frequently of the same order of magnitude as the upstream release loads, and on Lake Travis occasionally represent the major contribution of the total external loading. On Town Lake, upstream release loads and runoff loads typically exhibit a similar order of magnitude under Scenario X, with the runoff loads constituting the primary source of mass slightly more than half of the time.

6.3 DISCUSSION OF RESULTS

6.3.1 Implications on Water Quality

A case-by-case presentation of the predicted phosphorus concentrations for each scenario was provided in the previous section. The alternative scenarios represented a range of future developmental conditions. A summary of the simulation results is displayed in Table 6-14. The historical data indicates that the reservoirs comprising the Highland Lakes are usually characterized by relatively low concentrations of total phosphorus. As a general conclusion, projected concentrations for future conditions indicate that phosphorus will remain at relatively low levels.

The effects of projected future population growth and urbanization were examined in Scenario I, which constituted the baseline case for future conditions. Projected phosphorus concentrations for 1984-2000 under the Scenario I assumptions displayed increases of 0.000-0.008 mg/l above the simulated historical concentrations for 1967-1983 (see Table 6-1 and Table 6-2). Since the underlying hydrologic conditions for the periods 1967-1983 and 1984-2000 were identical, projected changes in phosphorus concentrations are attributable to increases in external mass loadings, primarily point source and runoff loads from the adjacent watersheds. No increases in phosphorus concentrations above historical levels are projected for Lake Buchanan and Inks Lake under the Scenario I assumptions, due to negligible increases in external loadings. Increments of 0.000-0.002 mg/l are projected for Lake LBJ, in response to increases in point source and urban runoff loads. Projected point source and runoff loads to Lake Marble Falls under the Scenario I assumptions resulted in phosphorus elevations of 0.002-0.008 mg/l. Loadings to Lake Travis were projected to increase phosphorus levels 0.000-0.001 mg/l above historical levels. In Lake Austin, increases of 0.000-0.002 mg/l were predicted. Increases in phosphorus of 0.000-0.003 mg/l were projected for Town Lake under Scenario I assumptions.

In Scenario II, the effects of increased urban loadings were examined. It was assumed that the phosphorus concentration in urban runoff was doubled above the baseline value. This assumption investigated the possibility that the average phosphorus concentration in urban runoff would be higher than anticipated, as a consequence of either an inaccurate forecast of the baseline phosphorus level or the realization of a more intense level of urban development. In response to the increase in urban runoff loadings, elevations in reservoir phosphorus levels above the baseline Scenario I results of 0.001 mg/l were occasionally predicted.

Hypothetical supplemental point source loads were examined in Scenarios III and IV. Addition of a moderate-size point source load to Lake LBJ, equivalent to a population of 10,000, increased predicted phosphorus concentrations above baseline (Scenario I) conditions by 0.001-0.002 mg/l. Addition of a larger point source load to Lake Travis, equivalent to a population of 50,000, resulted in projected increases in phosphorus levels of 0.001-0.003 mg/l above baseline conditions.

Complete urbanization of the watersheds of Lake Travis, Lake Austin, and Town Lake was examined in Scenario V. Projected phosphorus levels showed increases of 0.000-0.004 mg/l within Lake Travis, 0.001- 0.008 mg/l within Lake Austin, and 0.001-0.012 mg/l within Town Lake. A large supplemental point source load to Lake Travis, equivalent to a population of 100,000, was coupled with the complete urbanization assumptions in Scenario VI. Projected increases in phosphorus levels of 0.003- 0.007 mg/l within Lake Travis, 0.004-0.011 mg/l within Lake Austin, and 0.004-0.015 mg/l within Town Lake were obtained.

In Scenario VII, the magnitudes of supplemental loadings required to produce elevations in phosphorus concentration of 0.010 mg/l were defined. Required supplemental inputs were substantial, for example, 85×10^6 g/yr for Lake Travis. In a similar manner, the magnitudes of supplemental loadings required to effect a 0.010 mg/l increase in phosphorus in the reservoir immediately downstream were examined in Scenario VIII. A load of 85×10^6 g/yr to Lake Travis, for example, was sufficient to elevate concentrations in Lake Austin by the target level.

The effects of urban runoff control measures were investigated in Scenarios IX and X. In Scenario IX, urban loadings were reduced by 50 percent from the baseline conditions in Scenario I. With the reduction in urban loads, decreases in projected phosphorus levels of 0.001 mg/l were occasionally observed in the series of reservoirs, though concentrations in Town Lake decreased by 0.000-0.006 mg/l. Urban loadings predicted with the complete urbanization assumption of Scenario V were reduced by 50 percent for the Scenario X exercise. Compared to the complete urbanization results, the reductions in loading resulted in decreases in predicted phosphorus concentrations of 0.000-0.002 mg/l in Lake Travis, 0.000-0.005 mg/l in Lake Austin, and 0.001-0.008 mg/l in Town Lake.

Generally, the projected increases in average ambient phosphorus concentrations do not appear to be large enough to cause a significant water quality impact. The primary concern is that additional phytoplankton growth may be stimulated by increased phosphorus levels. To estimate roughly the potential magnitude of the effect, biological activity can be approximated with chlorophyll-a levels. Chlorophyll-a levels can demonstrate a strong correlation with algal biomass and are commonly employed as a relative measure of biological activity, especially when more detailed phytoplankton data are not available. In some systems, correlations have been demonstrated between total phosphorus and chlorophyll-a levels. As an example, the following empirical relationship is frequently employed (Dillon and Rigler, 1974):

$$\log(\text{Chl-a}) = 1.449 \log(P) - 1.136$$

where Chl-a = concentration of chlorophyll-a, μg/l
P = concentration of total phosphorus, μg/l

Potential increases in chlorophyll-a levels, though, have little intuitive value as a perceptible change in water quality. Ideally, the projected total phosphorus or chlorophyll-a concentrations should be related to a readily discernible indicator of water quality. An approach that has been used in other investigations is to relate chlorophyll-a, as an indicator of algal density, to transparency in the water column. For example, Rast and Lee (1978) obtained the following relationship between Secchi depth and chlorophyll-a:

$$\log(SD) = -0.473 \log(\text{Chl-a}) + 0.803$$

where SD = Secchi depth, m
Chl-a = chlorophyll-a, μg/l

Using the two relationships described above, increments in phosphorus concentration can be interpreted in terms of chlorophyll-a and Secchi depth, as shown in Table 6-15.

According to data from Fruh and Davis (1969a, 1972), chlorophyll-a concentrations in the Highland Lakes were in the approximate range of 1-60 μg/l, and Secchi depths typically ranged from 0.3-3 m. The data in Table 6-15 display a strong sensitivity in Secchi depth at low chlorophyll-a levels. Data in the lower range of chlorophyll-a values do not appear to be representative of the Highland Lakes. In the higher range of chlorophyll-a values, increments in concentration result in smaller changes in Secchi depth: increases of 0.001 mg/l in phosphorus effect roughly 0.1 m decreases in Secchi depth.

The empirical relationships employed in the calculation were not developed specifically for the Highland Lakes, and it is probable that the relationships do not hold precisely for the series of reservoirs. Though the relationships may not accurately predict the magnitude of chlorophyll-a levels or Secchi depths in the Highland Lakes, they could possibly predict the correct systematic trends. However, this cannot be assured, since additional factors such as turbidity and particulate levels may interfere with the relationship. The Highland Lakes may have higher levels of turbidity and thus lower Secchi depths that those used in the development of the empirical relationships. The extant turbidity levels may be due primarily to erosional material as opposed to algal densities. A higher level of background turbidity could diminish the effects of increased algal growth and preclude serious impairment of water quality (Lee, 1977). The effect of higher background turbidity would be most noticeable in the empirical relationships at the lower range of chlorophyll-a values.

Numerous assumptions concerning mass loadings, hydrologic parameters, and reaction rates comprised the present simulation of phosphorus concentrations in the Highland Lakes. Uncertainty in the variables could affect the accuracy of the predicted phosphorus levels. Confidence in the predicted concentrations is enhanced by the fact that various simulation scenarios demonstrated that substantial augmentation of the baseline external mass loadings was generally required to effect a material change in phosphorus concentrations. Recognizing this fact, the predicted average baseline phosphorus concentrations can be assumed to be accurate within reasonable limits, probably within ± 0.010 mg/l, and perhaps within ± 0.005 mg/l. However, localized fluctuations in phosphorus levels will occur. For example, there exists a degree of longitudinal variation in concentration within certain reservoirs, primarily the larger waterbodies. In addition, discrete zones or embayments within a reservoir may exhibit elevated phosphorus levels as a result of localized tributary, runoff, or point source loadings.

As a general conclusion then, projected reservoir phosphorus concentrations for future conditions appear to be in line with historical levels. In terms of increments in concentration, low or nominal increases are generally projected, and future phosphorus concentrations should remain at relatively low levels. However, the implications of the projected increases in phosphorus levels, albeit small, cannot be precisely ascertained at the present time. The relationship between phosphorus concentration and algal growth, or chlorophyll-a as an indicator, has not been defined in the Highland Lakes. Further, the relationship between chlorophyll-a and transparency has not been defined. Based upon commonly used relationships, though, the projected phosphorus increments would entail no substantial stimulation of algal growth or deterioration of transparency.

6.3.2 Sources of Phosphorus Loadings

Within Chapter 4.0, external mass loadings to the Highland Lakes from point sources, runoff from adjacent watersheds, major tributaries, and precipitation were quantified for historical and projected future conditions. It was apparent that the nonpoint source loads exceeded the point source loads in the multitude of watersheds, usually by orders of magnitude. An exception was the load to Lake Marble Falls; because of its limited drainage basin and the existence of the discharge from the City of Marble Falls, the point source load comprised the majority of the total external loading from the aforementioned sources.

An additional source of phosphorus loading, releases from upstream reservoirs, was examined through exercise of the model HILAKES. The loads associated with releases were estimated under the assumption that the outflow from a reservoir was characterized by the same concentration as calculated within the reservoir, one of the principles of the analysis of completely mixed systems. With this technique, the annual loads were calculated in conjunction with the modeling analyses, subsequent to the determination of the reservoir phosphorus level. The simulation results from the alternative future scenarios indicate that the loadings associated with releases from upstream reservoirs are frequently the major source of phosphorus loading, on an annual basis, to the reservoirs comprising the Highland Lakes. These upstream release loads are often orders of magnitude greater than loadings from other sources. As an illustration of the relative magnitudes of the various sources of phosphorus loadings, a year-by-year inventory of source loads for the period 1984-2000 under the anticipated future baseline conditions (Scenario I) is displayed in Table 6-16. For the baseline scenario, loads associated with upstream releases usually constitute the most substantial source of phosphorus on an annual basis for Inks Lake, Lake Marble Falls, Lake Travis and Lake Austin. Major tributary loads most frequently represent the largest fraction of the total phosphorus load on Lake Buchanan and Lake LBJ. On Town Lake, runoff loads from adjacent watersheds are usually the largest component of the total external mass loading under the baseline scenario, just slightly more often than phosphorus loads from upstream releases. A summary description of the primary mass loading sources for each simulation scenario is presented in Table 6-17.

Mean loads from the various sources for the future simulation conditions were calculated for each scenario and are compared in Table 6-18. On the basis of these calculations, the mean load from upstream releases is usually the largest contributor of phosphorus mass to most of the reservoirs, frequently an order of magnitude in excess of loads from other sources. Major tributary loads usually constitute the largest source of phosphorus loads for Lake Buchanan and Lake LBJ. Runoff loads from adjacent watersheds are occasionally substantial, sometimes of the same order of magnitude as other sources.

The magnitude and concomitant effects of runoff loads were enhanced under certain scenarios. With the calculation technique employed in the present analysis, the magnitude of runoff loads was dependent upon the land use categories indigenous to each watershed and the volume of stormwater runoff. The annual runoff load is thus a function of the number and magnitude of large storm events in any given year, and is potentially a more substantial component of the total external load in years with a number of large storm events. Loads associated with direct precipitation input are consistently low.

The general system equation was described in Chapter 5.0 (eqn 18):

$$\frac{dC}{dt} = \frac{W(t)}{V} - \frac{QC}{V} - kC$$

According to this formulation, the various sources of mass are summed together for input. The effect of the mass load upon concentration is determined by the magnitude of the total load and the volume of the reservoir. A specific mass load of phosphorus would have markedly different effects upon Lake Buchanan and Inks Lake, for example, due to the tremendous difference in volume. Year-to-year fluctuations in the average volume within a particular reservoir also influence the effects of the mass load. As shown in Table 5-3, the annual average storage on Lake Travis, for example, ranged from 621.5×10^6 - 1467.3×10^6 m^3 during the period 1952-1983. If the mass load was held constant, such a fluctuation in volume could account for a variation in reservoir phosphorus concentration in excess of 100 percent.

As described above, the load associated with releases from an upstream reservoir can comprise a major portion of the total external loading to a reservoir. The system equation indicates that the releases also have a substantial effect upon the phosphorus mass within the reservoir from which the outflow originates. The concentration within the reservoir is further affected by the first-order decay term.

The reservoir concentrations observed under the simulation scenarios are thus determined not only by the magnitude of external mass loadings, but also by the volume, outflow, and decay terms. In particular, the loss of mass via the outflow term can have a substantial effect upon reservoir concentration. The combined actions of the various terms can mitigate the effects of what appear to be large increases in mass loading to a reservoir. As an example, large point source loads of phosphorus were introduced into the reservoirs under Scenario VII. Large supplemental loads, as much as 85×10^6 g/yr to Lake Travis, were required to effect a small increase of 0.010 mg/l in reservoir concentration. Though the absolute magnitude of the loads appeared large, the effects upon reservoir concentration were mitigated by the often large volume and outflow terms.

Having established the importance of the outflow term, the implications should be further examined. In the present analysis, the outflow is input as an annual total volume. As such, the magnitude of the outflow term directly affects the calculation of the annual average concentration within a reservoir. If the volume of water released is substantial, the effects of large phosphorus loadings upon concentration within the reservoir will be diminished. Though not accounted for analytically in the present analysis, reservoir releases in the Highland Lakes are not constant in time, but instead vary substantially in response to downstream demands. As described in Chapter 3.0, much of the water for satisfaction of downstream irrigation demands is released between March and November of each year. The timing of releases can have a significant effect upon reservoir phosphorus levels at any particular time. If, for example, a large storm event delivers a mass influx of phosphorus to a reservoir during a period with substantial outflow, the effects upon concentration within the reservoir will be less than if the influx occurred at a time when releases were negligible. Similar reasoning applies to sources of continual mass inflow: during any given year, the effects of a point source load upon average reservoir phosphorus levels may be greater under periods of negligible release. It is interesting to note that maximal outflow usually occurs during warm-weather periods. If indeed a substantial influx of mass is received during a period of negligible outflow, it is usually coincident with cold-weather conditions. Since biological activity, including phytoplankton growth, is typically reduced during cold weather, influx of a substantial phosphorus load under such conditions may have relatively little effect upon biota.

6.3.3 Interrelationships Among Reservoirs

The concentration of phosphorus within a reservoir of the chain of Highland Lakes is influenced not only by external mass loadings directly to the reservoir, but also by loadings and characteristics in upstream reservoirs. The effects of a substantial influx of mass to a particular reservoir may increase the concentration within that reservoir and may also propagate downstream. This propagation of effects defines the interrelationships among the reservoirs in terms of water quality.

The interrelationships among reservoirs were elucidated by several of the simulation scenarios. Specifically, the results of Scenarios III, IV, VI, VII, and VIII directly address the propagation of water quality effects downstream. With Scenario III, insertion of a supplemental point source load to Lake LBJ resulted in nominal increases in phosphorus concentration within the reservoir. Comparable concentration increments were observed downstream in Lake Marble Falls, and slight elevations were occasionally displayed farther downstream. Addition of a substantial point source load to Lake Travis in Scenario IV produced projected increases in phosphorus levels within the reservoir of 0.001-0.003 mg/l. Increments of 0.001-0.002 mg/l were predicted downstream in Lake Austin and Town Lake. The effects of a larger point source load to Lake Travis were examined in Scenario VI. Concentration increments up to 0.003 mg/l phosphorus were predicted for Lake Travis and the reservoirs downstream. With Scenario VII, the effects of large supplemental loads to each of the reservoirs, sufficient to elevate phosphorus concentration by 0.01 mg/l, were analyzed. Insertion of a large supplemental load to Lake Buchanan resulted in increments in phosphorus concentration of 0.006-0.012 mg/l. In Inks Lake immediately below Lake Buchanan, phosphorus concentrations were elevated by 0.004-0.009 mg/l. Downstream effects included concentration increments of approximately 0.001-0.004 mg/l in Lake LBJ and Lake Marble Falls. Increments of 0.001 mg/l were occasionally predicted for Lake Travis, Lake Austin, and Town Lake. Simulation of a large supplemental load to Lake LBJ resulted in downstream increases in concentration of 0.006-0.011 mg/l in Lake Marble Falls and 0.000-0.002 mg/l in the lower reservoirs. The addition of a large supplemental load to Lake Travis in Scenario VII produced concentration increments of 0.008-0.014 mg/l within the waterbody and roughly similar increases in Lake Austin and Town Lake. Downstream propagation of effects was similarly observed in Scenario VIII.

The magnitude and longitudinal extent of the propagation of effects on downstream reservoirs is dependent upon the total external mass loading to each reservoir and the outflow through the series of reservoirs. In situations where the total external mass loading to a reservoir is dominated by the loading associated with releases from an upstream reservoir, the effect is potentially more substantial. In addition, downstream effects are generally more substantial with larger releases from the upstream reservoir. In the extremely throughflow-dominated reservoirs--Inks Lake, Lake Marble Falls, Lake Austin, and Town Lake--effects upon concentration from upstream loadings are potentially much greater. In essence, whatever increment in concentration is achieved in the upstream reservoir will also be realized in the throughflow-dominated waterbody below. Conversely, a larger, less throughflow-dominated reservoir such as Lake Travis is less likely to demonstrate substantial increases in concentration in response to a mass input to an upstream waterbody.

The phosphorus reaction rate can have a substantial influence on the propagation of effects downstream. In the context of the present analysis, the reaction rate prescribes a loss of phosphorus from the water column. Thus, the reaction rate affects the mass of phosphorus available for transport downstream. The nature of the present modeling analysis also affects the simulated interrelationships among the reservoirs. By analyzing the reservoirs as a series of CSTR's, the model assumes instantaneous propagation of the effects of mass loadings, representing a fraction of the original load, down the chain of impoundments. In reality, there is a time delay for the downstream transport.

6.3.4 Implications Upon Development and Controls

The watersheds of the Highland Lakes are being subjected to increasing developmental pressures. Proposed developments are primarily residential with light commercial areas, and research/office parks are currently under consideration in some areas. Accompanying the projected development trends are the prospects of increased wastewater discharges and urban runoff loadings to the reservoirs. The prospect of additional discharges to the reservoirs has generated heated debate within the Central Texas area. Proponents of development, and subsequently discharge of treated wastewater, have argued that wastewater effluent subjected to relatively high levels of treatment will have no demonstrable impact upon water quality nor impair the uses of the waters. Conversely, others view the prospect of wastewater discharges as a bestial act that will totally degrade the present state of water quality. Much of the debate has been largely emotional, notably devoid of comprehensive quantitative analyses. In 1983, the Texas Water Development Board enacted a rule prohibiting additional discharges of municipal wastewater to Lake Travis and Lake Austin pending the results of a comprehensive evaluation of potential effects upon water quality, presently being conducted by the Center for Research in Water Resources at The University of Texas at Austin.

The prospect of increased urban runoff loadings has also generated considerable debate, as well as attempts at regulation, in the Austin area. Debate has often centered upon the means for effecting control of runoff loads, namely, limitations in developmental density versus structural control measures. The City of Austin has enacted ordinances for the control of development within the watersheds of Lake Travis, Lake Austin, and Town Lake that generally prescribe a mixture of density and structural controls. Ostensibly, the City of Austin promulgated the regulations on the basis of maintenance of water quality for protection of the municipal water supply.

The continuing debates over point source discharges and runoff loads raise a variety of issues, for example, maintenance of water quality, continuation of existing uses, rights of landowners, rights of municipal government, appropriate control measures, and assessment of impacts. Opposing viewpoints have been offered by concerned (and contracted) laymen and professionals--all have in common redoubtable sincerity. With the exception of the aforementioned comprehensive evaluation of the potential effects of point source discharges upon water quality currently underway, the technical analyses supporting the issues have been limited. Typically, the tendency has been to focus upon a single reservoir when a specific issue is raised. As an example, debate over the efficacy of runoff controls for maintenance of water quality in Lake Austin generally focused upon conditions in that particular waterbody, with little regard for conditions upstream in Lake Travis.

The results of the present analysis indicated a potentially significant shortcoming for studies that focus upon a single reservoir. It has been demonstrated that loads associated with releases from upstream reservoirs are frequently the major contributor of phosphorus in the total mass budget of a particular reservoir. Further, the interrelationships among the reservoirs, with respect to phosphorus concentrations, have been shown to be potentially significant. These two aspects are themselves closely related; a pertubation in concentration or mass within one reservoir directly affects the magnitude of the outflow load, which in turn dictates the magnitude and longitudinal extent of the downstream propagation of effects.

The results of the future developmental scenarios examined in the present analysis have direct implications upon the issues of development and controls. Historically, the Highland Lakes have been characterized by relatively low concentrations of total phosphorus. Projected future conditions indicated that phosphorus will remain at relatively low levels under a variety of potential development scenarios. Under the baseline Scenario I, which incorporated the most likely prospects for growth in the period 1984-2000, projected phosphorus concentrations generally did not differ substantially from the simulated historical levels for the period 1967-1983, which provided the hydrologic basis for simulation of future conditions. The projected maximum elevation above historical levels was typically 0.001 mg/l in Lake LBJ, Lake Travis, and Lake Austin, and up to 0.003 mg/l in Town Lake. Larger increases in phosphorus concentrations were demonstrated in Lake Marble Falls, ranging from 0.002-0.008 mg/l. Even with a doubling of the projected baseline urban runoff loads, predicted phosphorus concentrations within the reservoirs displayed only occasional increases and were never elevated more than 0.001 mg/l over baseline (Scenario I) levels. In essence, these results indicated that projected long-term future trends in phosphorus concentration for the Highland Lakes demonstrate no substantial increases over historical levels.

Complete urbanization of the watersheds of Lake Travis, Lake Austin, and Town Lake was postulated for Scenario V. The assumption of complete urbanization produced increases in phosphorus concentration of 0.000-0.004 mg/l within Lake Travis, 0.001-0.008 mg/l within Lake Austin, and 0.001-0.012 mg/l within Town Lake. In terms of the ultimate magnitude of the predicted phosphorus levels, phosphorus concentrations in Lake Travis remained below 0.010 mg/l (which is the current analytical limit of detection routinely obtained by commercial water chemistry laboratories). Similarly, predicted concentrations for Lake Austin were usually below 0.010 mg/l, with a projected maximum of 0.014 mg/l phosphorus under the Scenario V assumptions. Predicted concentrations within Town Lake were generally higher, with a maximum phosphorus level of 0.030 mg/l. The predicted effects of urban runoff are relatively small, with respect to the magnitude of projected average phosphorus concentrations. Institution of runoff control measures was addressed by a simulated fifty percent reduction in urban loadings. Compared to the predicted concentrations realized in the absence of runoff controls, imposition of a fifty percent reduction of urban loads resulted in reduction in the magnitude of the reservoir phosphorus concentration increases. However, since the projected effects of urban loads without controls were generally small, the net benefits of the simulated control measures were not readily ascertained.

The results of this analysis question the efficacy of urban runoff controls in terms of realization of an improvement in phosphorus concentrations within the reservoirs. The effectiveness of the various control measures in reduction of urban phosphorus loads is not at issue--control of urban phosphorus loads can be achieved by some technique or combination of techniques. Instead, the issue raised within the context of the results of the present analysis is whether or not a reduction of urban loads will have a significant beneficial effect upon reservoir phosphorus concentrations. It is apparent that reduction of urban runoff phosphorus loads will have little material benefit upon long-term reservoir phosphorus concentrations, due to the relative magnitude of the urban runoff mass loading compared to other sources. This is not a general condemnation of urban runoff control measures. The benefits of mitigation of urban runoff loads transcend the implications on phosphorus concentrations.

The effects of point source loadings were examined in several of the future development scenarios. A phosphorus load to Lake LBJ equivalent to a discharge from a population of 10,000 (2.8×10^6 g/yr) was considered in Scenario III and Scenario IV, the latter including a load from a population of 50,000 (13.8×10^6 g/yr) directed to Lake Travis. Within Lake LBJ, projected phosphorus levels displayed increments of 0.001-0.002 mg/l in response to the loading, with comparable elevations downstream in Lake Marble Falls. The additional point source load to Lake Travis (13.8×10^6 g/yr) produced predicted concentration increments of 0.001-0.003 mg/l, with the propagation of similar effects in Lake Austin and Town Lake. Simulation of a larger point source load to Lake Travis, equivalent to a 100,000-person discharge (27.6×10^6 g/yr), was coupled with the complete watershed urbanization assumption in Scenario VI. The supplemental point source resulted in projected increases in phosphorus concentration of approximately 0.003 mg/l within Lake Travis.

Supplemental phosphorus loadings required to produce a 0.010 mg/l rise in phosphorus concentration within each reservoir were determined in Scenario VII. Required phosphorus loads ranged from 7×10^6 g/yr for Inks Lake to 85×10^6 g/yr for Lake Travis. For reference, the required load to Lake Travis represents the point source discharge from a population of roughly 300,000. The predicted supplemental loadings required to effect a nominal increase in phosphorus concentration are substantial and are not likely to materialize within the realm of projected future conditions up to the year 2000. The projected point source phosphorus loading equivalent to a population of 100,000 described above, 27.6×10^6 g/yr, was based upon an assumed effluent total phosphorus concentration of 2 mg/l. Without the incorporation of specific phosphorus removal measures, the uncontrolled phosphorus concentration in a municipal effluent can range from roughly 3-14 mg/l as discussed in Chapter 4.0. If it is assumed that an effluent value of 10 mg/l is representative, then the corresponding phosphorus loads from populations of 10,000, 50,000, and 100,000 would be 14×10^6 g/yr, 69×10^6 g/yr, and 138×10^6 g/yr, respectively. This level of loading would constitute a substantial increase over the assumed loading at a phosphorus concentration of 2 mg/l. It is apparent that limitation of the phosphorus level in point source effluent is prudent.

In terms of the results of the present analysis, future phosphorus concentrations within the Highland Lakes are predicted to remain at relatively low levels, consistent with historical conditions. The results indicate that ambient phosphorus levels will generally be maintained, even under intensive developmental pressure. To a large extent, the predicted results are attributable to the relative magnitudes of the various sources of phosphorus mass loads, as discussed in Sec. 6.5, and the substantial volume and throughflow characteristic of the series of reservoirs. However, the results are also a consequence of the techniques employed. The present analysis was designed to evaluate long-term water quality trends, thus, mass loadings and hydrologic parameters were defined in terms of annual values. Simulation results were subsequently defined as projected average annual phosphorus concentrations. Projected phosphorus levels should be interpreted with cognizance of the analytical methodology: long-term average phosphorus levels were predicted. With this qualifier, the results described throughout this section are defensible and are likely representative of the typical concentration characteristics of the series of reservoirs that will be observed over the period 1984-2000.

At a smaller temporal scale or more intensive spatial scale of analysis, more significant elevations in phosphorus concentration could possibly be projected. For example, influx of phosphorus mass loads under conditions of reduced throughflow in the series of reservoirs (typically cold-weather conditions) could comprise a situation where phosphorus levels are elevated above the concentration ranges projected in the present analysis. However, effects at smaller temporal or spatial scales are more likely to be transitory, with respect to the long-term water quality characteristics of a reservoir. For the objectives of the present analysis, then, the simulation of long-term phosphorus concentration trends represented a practicable approach.

6.3.5 Analysis of Simple Trophic Models

Simple trophic models frequently employed in reservoir studies were described in Chapter 2.0. Three of the most commonly used models were selected for application with the data developed in the present modeling analysis: the Vollenweider (1968) phosphorus loading-mean depth model, Vollenweider (1976) phosphorus loading-hydraulic loading model, and the Dillon (1975) algorithm relating loading, phosphorus retention, hydraulic residence time, and mean depth. The models are used to determine the trophic state of a reservoir, given fundamental mass loading and hydrographic data.

The trophic models were applied to the data and results of the Scenario I simulation, which constitutes the baseline projection for future conditions in the reservoirs. The mean loadings for each reservoir for the future simulation period were employed, as well as average values for volume, outflow, and area, as necessary. The data are summarized in Table 6-19. The plotting positions of each reservoir, indicating the trophic state, are displayed in Figs. 6-1 through 6-3 for the three models. Ranges are delineated on the loading ordinates for each reservoir. The maximum and minimum values were determined from the extremes of the total external mass load only, under the assumption that all other parameters, including surface area, volume, and outflow, remained at average levels. This simplification was based upon the realization that the projected individual annual loadings are in fact independent of the area, volume, and outflow, and could occur under a variety of combinations of these parameters. The ranges of loading parameters under the Scenario I assumptions are displayed in Table 6-20.

Vollenweider Loading-Mean Depth Model

Vollenweider's (1968) original model simply examined the areal phosphorus loading rate and the mean depth of a reservoir as an indication of trophic state. Data for the Highland Lakes are shown in Fig. 6-1. The graph classifies all of the reservoirs as eutrophic. Further, all projected ranges of loading rates are situated within the eutrophic zone, with the exception of the minimum loading rate for Lake Travis, which crosses into the mesotrophic region. According to these results, all reservoirs are strongly eutrophic, with virtually no potential for attainment of a lower trophic state through a reduction of the phosphorus mass loading.

Vollenweider Mass Loading-Hydraulic Loading Model

Vollenweider's (1968) original model suffered from a lack of sensitivity to a reservoir's hydrologic character. As a refinement, Vollenweider (1976) incorporated the hydraulic retention time into the analysis in a ratio with the mean depth, a term which constitutes the hydraulic loading to the reservoir. Projected data for the Highland Lakes are presented in Fig. 6-2. Mean values for Lake Buchanan, Lake LBJ, and Inks Lake indicate a state of eutrophy. The lower range of loading rates for Lake Buchanan falls into the mesotrophic region. For Lake LBJ, the lower bound of the range extends through the mesotrophic region and into the oligotrophic zone. The range of values for Lake Inks is exceptionally large, extending well into the oligotrophic region.

A mesotrophic state is indicated for the mean loading characteristics of Lake Travis and Lake Marble Falls. Maximum and minimum loading rates for Lake Travis extend into the eutrophic and oligotrophic zones, respectively. For Lake Marble Falls, the range extends into the eutrophic region.

Mean values for Lake Austin and Town Lake fall within the oligotrophic region. The Lake Austin range remains within the oligotrophic state. The upper end of the loading rate range for Town Lake extends into the mesotrophic zone.

The effects of the hydraulic retention time are evident in the graph. Values for the smaller reservoirs that are characterized by shorter retention times (or greater flushing rates) plot toward the right, where higher phosphorus loading rates are required to denote a state of eutrophy. Conversely, the larger reservoirs with longer retention times are situated within the region of the graph where lower phosphorus loading rates define a eutrophic condition.

Dillon Loading Parameter-Mean Depth Model

The Dillon (1975) model evaluates trophic state on the basis of mean depth and a loading parameter that includes the areal phosphorus loading rate, phosphorus retention coefficient, and hydraulic retention time. Data for the Highland Lakes are shown in Fig. 6-3. With the Dillon (1975) method, none of the predicted mean values are indicative of eutrophic conditions. The plotting positions of Lake Buchanan, Inks Lake, Lake LBJ, and Lake Marble Falls indicate mesotrophic conditions. For these reservoirs, the maximum and minimum values representing the range of mass loadings extend into the eutrophic and oligotrophic regions, respectively.

Oligotrophic states are indicated with predicted mean values for Lake Travis, Lake Austin, and Town Lake. The ranges of values for Lake Travis and Lake Austin are encompassed within the oligotrophic region. The maximum loading parameter for Town Lake extends into the mesotrophic zone.

Lake Travis falls within the oligotrophic category due to its relatively low areal loading rate and relatively high mean depth. By comparison, the loading parameter for Lake LBJ is similar to Lake Travis, but the substantially lower mean depth for Lake LBJ places it within the mesotrophic region. An oligotrophic state is indicated for Lake Austin as a result of its relatively low areal loading rate and short hydraulic retention time. Though having the same mean depth, the larger areal loading rate of Lake Marble Falls places it within the mesotrophic category.

The effects of alternative future scenarios can be ascertained by examination of the Scenario I results displayed in Figs. 6-1 through 6-3. Plotting positions for each reservoir can be determined by the mean, maximum, and minimum phosphorus loadings, with the underlying hydraulics remaining unchanged from Scenario I conditions. As an example, results for Scenario V, which assumed complete urbanization of the lower watersheds of the Highland Lakes, were plotted on Vollenweider's (1976) mass loading-hydraulic loading model and Dillon's (1975) model. (Vollenweider's (1968) original model was not examined further since all reservoirs were positioned within the eutrophic region under Scenario I conditions.) The plotting positions of Lake Buchanan, Inks Lake, Lake LBJ, and Lake Marble Falls were unchanged from the Scenario I assumptions. Results for Lake Travis, Lake Austin, and Town Lake on the Vollenweider (1976) graph under the Scenario V assumptions are shown in Fig. 6-4. The effects of the increased phosphorus loadings are evident. The mean value for Lake Travis moved from the mesotrophic to the eutrophic zone. The mean value for Town Lake moved from the oligotrophic to the mesotrophic zone. For Lake Austin, the mean loading rate remained within the oligotrophic region.

Means and ranges on the Dillon (1975) plot are shown in Fig. 6-5. With the Dillon (1975) analysis, mean loading parameters for Lake Travis and Lake Austin remained within the oligotrophic region. A move from oligotrophic to mesotrophic was indicated for Town Lake under the Scenario V assumptions.

In an analogous manner, results for any of the future simulation scenarios can be examined, and the plotting positions can be ascertained from the mass loading data. The plots themselves can be examined for an indication of the loads required to effect a change in trophic classification. In Fig. 6-5, depicting the Dillon (1975) plot, for example, the mean loading parameter for Lake Travis remained within the oligotrophic region under the Scenario V conditions. Inspection of the vertical distance required to elevate the mean value to the line delimiting the mesotrophic zone indicates that a total phosphorus loading of approximately 85×10^6 g/yr to Lake Travis would be needed to effect the revision in classification. Similarly, a phosphorus load of 170×10^6 g/yr is indicated for elevation of Lake Travis to the eutrophic region on the Dillon (1975) plot. From a review of the loadings summarized in Table 6-18, it is evident that none of the projected future scenarios develop a mean phosphorus loading sufficient to effect a change in trophic classification for Lake Travis on the basis of the Dillon (1975) model. (Scenarios VII and VIII, which defined loads necessary to elevate phosphorus concentrations by 0.010 mg/l, involved loadings sufficient to change the reservoir classification to mesotrophic. However, these scenarios were formulated for investigative purposes only, and do not represent projected potential future conditions.)

For Lake Austin, the Dillon (1975) plot indicates that a mean phosphorus load of 16×10^6 g/yr is required to achieve a mesotrophic classification, and a load of

32×10^6 g/yr would change the status to eutrophic. Returning to Table 6-18, the mean total phosphorus loading under the Scenario VI assumptions is sufficient to denote a mesotrophic state. Similarly, mean phosphorus loads of 15×10^6 g/yr and 30×10^6 g/yr are required to effect classifications of mesotrophic and eutrophic, respectively, for Town Lake, based upon the Dillon (1975) model. The fact that the required loads double in magnitude from the lower limits of the mesotrophic and eutrophic regions is not coincidence. The boundaries delineating the trophic states were based upon permissible phosphorus concentrations of 0.01 mg/l and 0.02 mg/l for the lower and upper lines, respectively.

The preceding discussions focused for the most part upon changes in trophic classification brought about by changes in the phosphorus mass loading. While this is certainly a valid concern, loadings that are not sufficient to effect an alteration in trophic state may still incur undesirable changes in water quality. Delineations among trophic states are not precise, in fact, it is impossible to define unequivocably characteristics of oligotrophic, mesotrophic, or eutrophic reservoirs that are universally applicable. Characteristics to be considered could include nutrient loadings, phytoplankton populations, dissolved oxygen depletion in the hypolimnion, water clarity, and a multitude of additional factors. General characteristics of oligotrophic and eutrophic categories are described in Table 6-21.

Having examined the predictions of the simple trophic models, a discussion of their limitations is warranted. The models are simplistic in nature, and are based on the premise that trophic state can be determined from a few readily available parameters. The models were derived empirically, using classifications for lakes outside of the southwestern United States. It may be significant to note that manmade reservoirs were generally absent from the data set. The Vollenweider (1976) and Dillon (1975) models refer to phosphorus concentrations of 0.01 mg/l and 0.02 mg/l as indicative of changes in trophic classification. These concentrations may be inappropriate for most reservoirs in Texas, including the Highland Lakes. The Highland Lakes may be able to assimilate more or less phosphorus than suggested by Vollenweider and Dillon. Putnam et al. (1972), for example, determined that Florida lakes could assimilate more phosphorus than indicated by Vollenweider's (1968) analysis before becoming mesotrophic or eutrophic, and subsequently revised the critical loading delineations. Lee (1977) cautioned against application of the 0.01 mg/l phosphorus concentration as critical in Texas impoundments, citing high levels of background turbidity due to erosional material. Factors which could influence critical phosphorus concentrations include turbidity levels and the depth of the photic zone, phosphorus forms and availability for algal nutrition, temperature and the effects of thermal stratification, and reservoir operational characteristics, including effects on hydraulic residence time and the configuration of outlet structures. An accurate definition of critical phosphorus levels is simply not available for the Highland Lakes.

6.3.6 Utility of the Model

The model HILAKES developed in the present investigation was formulated to analyze annual phosphorus mass budgets in a series of reservoirs. A reservoir was treated as a completely mixed system, with the concentration of phosphorus in the outflow equal to the concentration within the reactor. Internal phosphorus kinetics were represented by a single first-order decay reaction, which assumes a net loss of phosphorus from the water column to the sediment layer.

The assumptions inherent in this modeling approach are numerous and should be closely examined. The reservoirs in reality are not completely mixed; there exist longitudinal and vertical phosphorus concentration gradients, particularly in the larger impoundments. Under warm weather conditions, many of the reservoirs exhibit marked thermal stratification, with distinct epilimnion and hypolimnion layers. Phosphorus dynamics entail a cycle of sedimentation and deposition to the benthal layer, coupled with release of phosphorus under anaerobic conditions. The assumption that the plethora of internal and external processes can be characterized on an annual basis is critical.

Alternative modeling approaches are available. The completely mixed model could be employed to simulate phosphorus levels on a smaller time scale. A multi-layer model could be used to represent the epilimnion and hypolimnion compartments. Or, a model with longitudinal segmentation could be employed. With any of these approaches, additional phosphorus forms and kinetic terms can be incorporated. However, the use of a more sophisticated model places additional demands upon the available data base.

Selection of an appropriate modeling approach is dictated by the objectives of a particular analysis and the nature and extent of available data. For the Highland Lakes, the historical phosphorus data set is severely limited, in terms of both frequency and locations of measurements. The limitations of the data set discourage the use of an overly complex analysis. For the present investigation, the fundamental objective was to evaluate long-term characteristics in reservoir phosphorus concentrations. Therefore, the completely mixed analysis was selected as the basis for the modeling approach. The simplifying assumptions inherent in the analysis are warranted due to the limited data base for the Highland Lakes. The effectiveness of the modeling approach employed in the present investigation has been demonstrated in other waterbodies, as described in Chapter 2.0. For example, the completely mixed model was successfully used to simulate long-term phosphorus trends in Lake Washington (Lorenzen, et al., 1976). Similarly, Snodgrass and Dillon (1983) determined that a single-layer model demonstrated performance comparable with a two-layer model in certain types of lakes. In the present investigation, the single-layer model provided estimates of reservoir phosphorus concentration in general agreement with historical observations. The utility of the model for long-range projection of average phosphorus levels has been demonstrated.

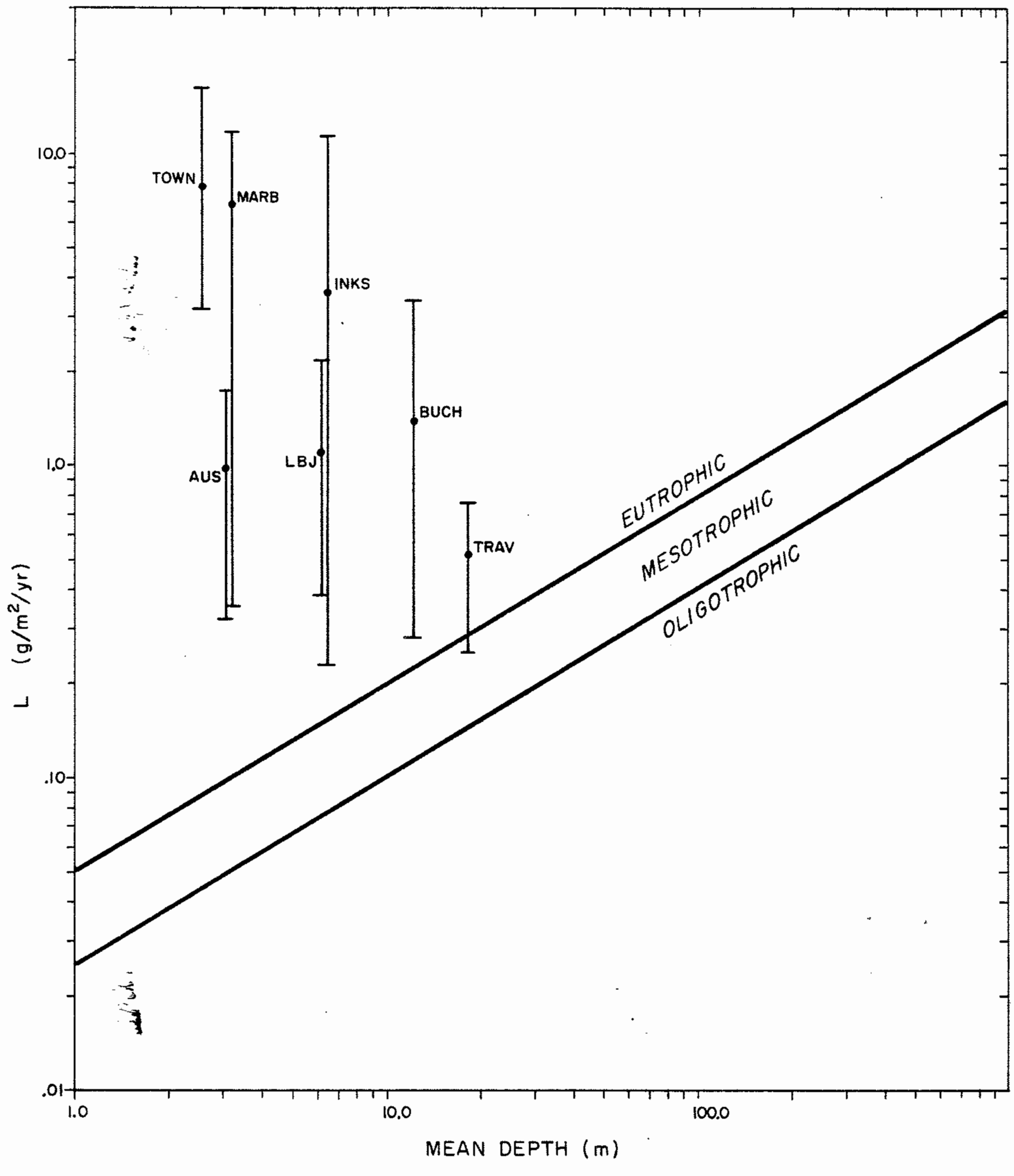

Fig. 6-1 VOLLENWEIDER PHOSPHORUS LOADING – MEAN DEPTH MODEL, SCENARIO I RESULTS

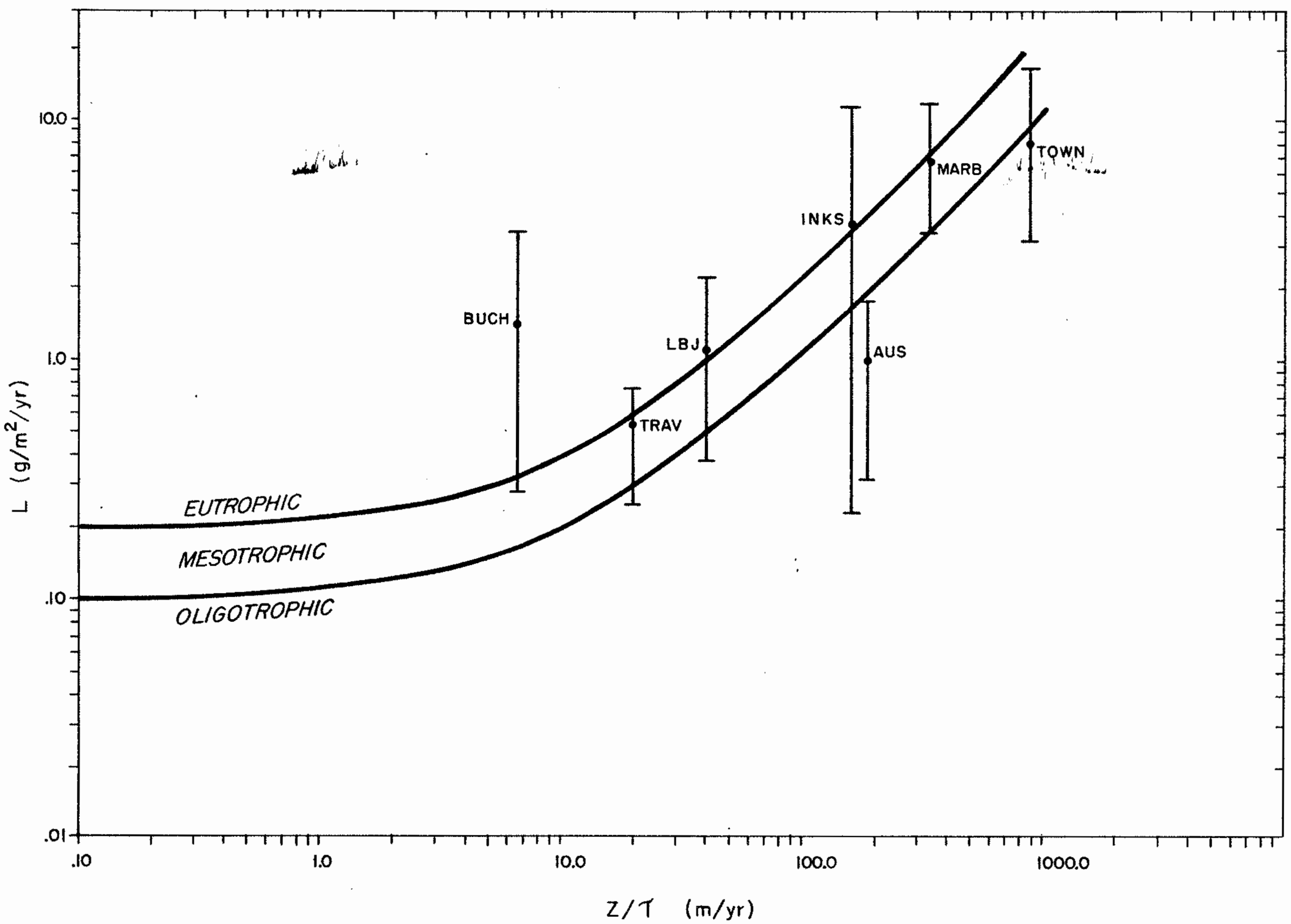

Fig. 6-2 VOLLENWEIDER PHOSPHORUS LOADING - HYDRAULIC LOADING MODEL, SCENARIO I RESULTS

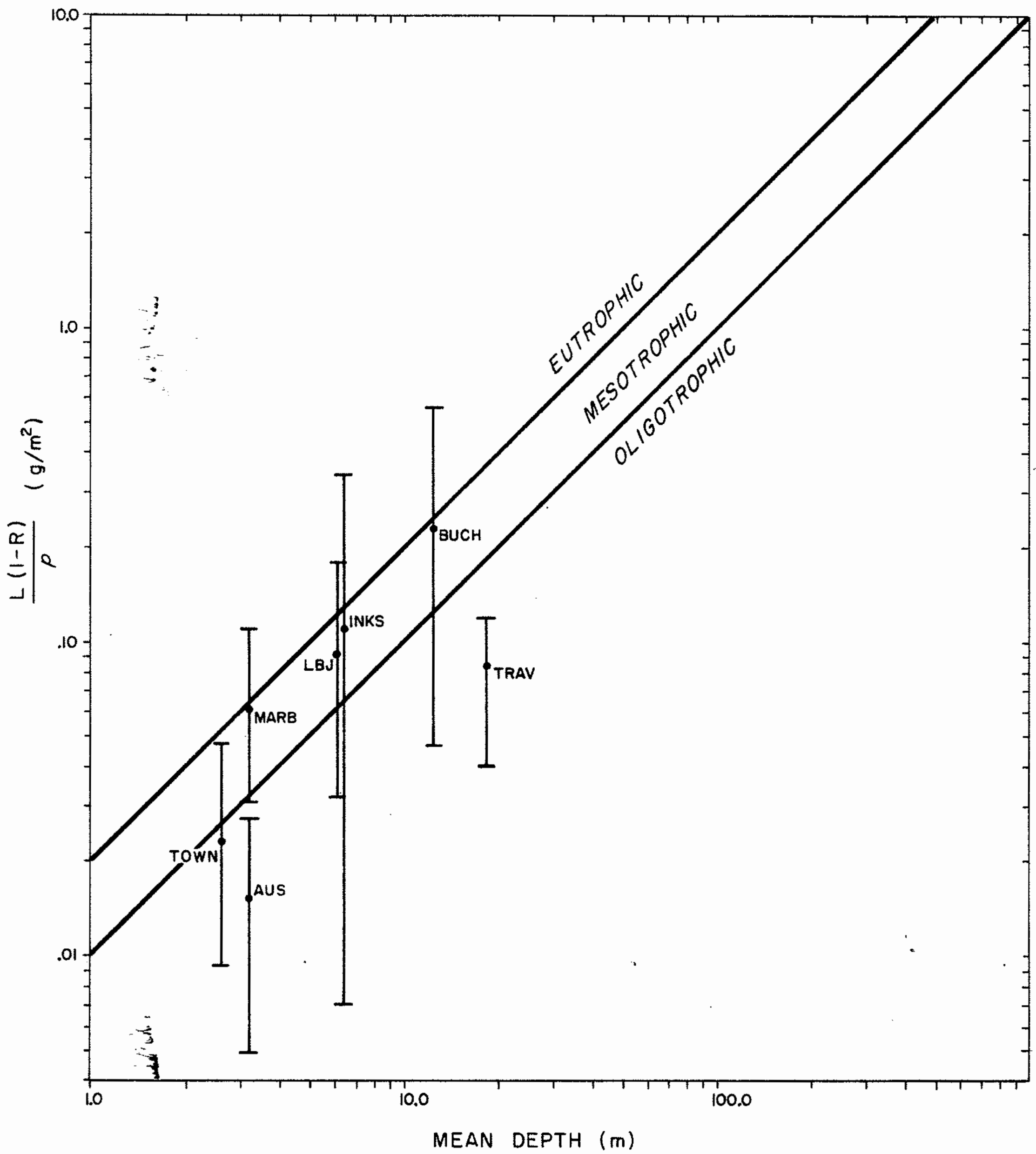

Fig. 6-3 DILLON LOADING PARAMETER - MEAN DEPTH MODEL, SCENARIO I RESULTS

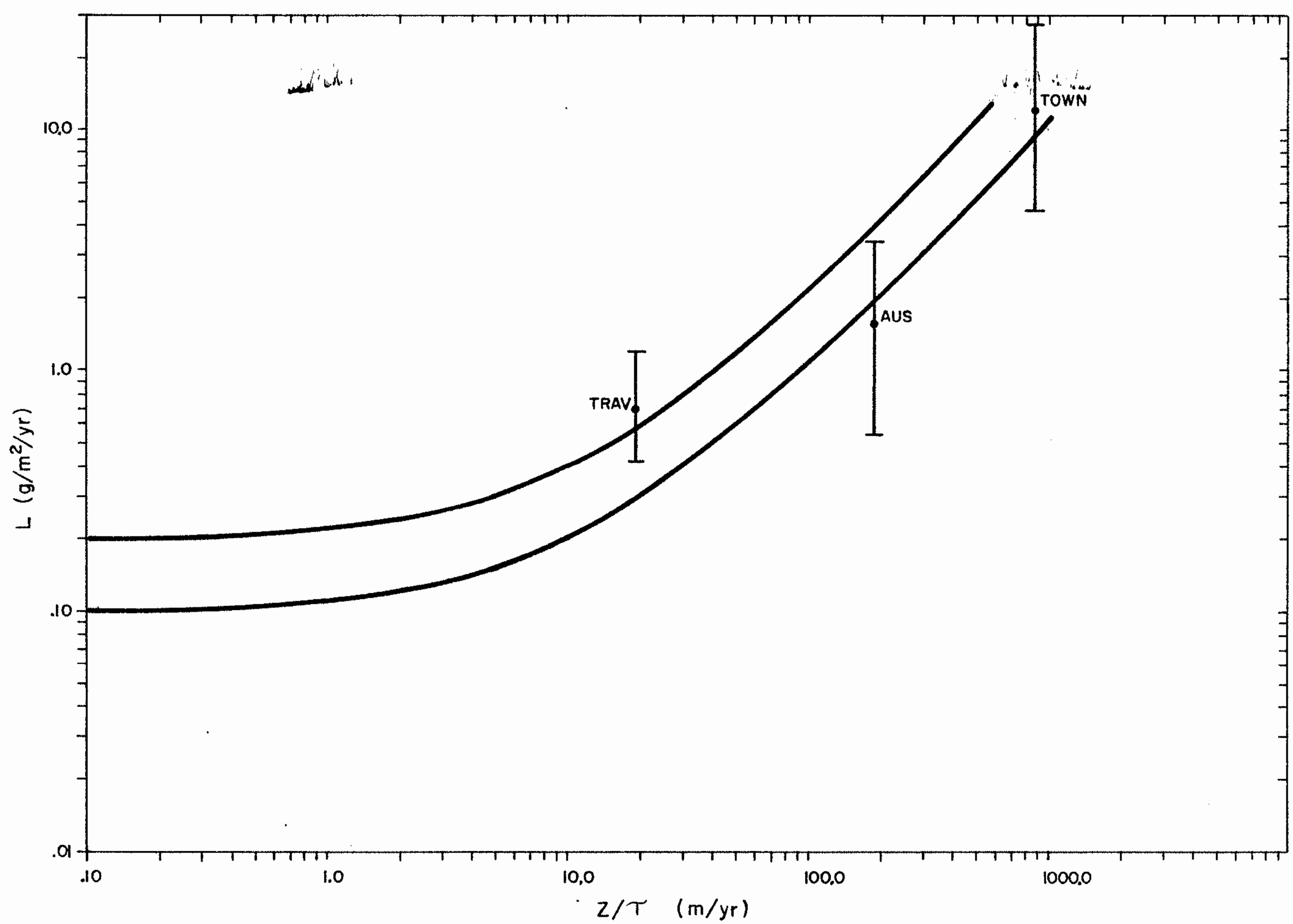

Fig. 6-4 VOLLENWEIDER PHOSPHORUS LOADING - HYDRAULIC LOADING MODEL, SCENARIO V RESULTS

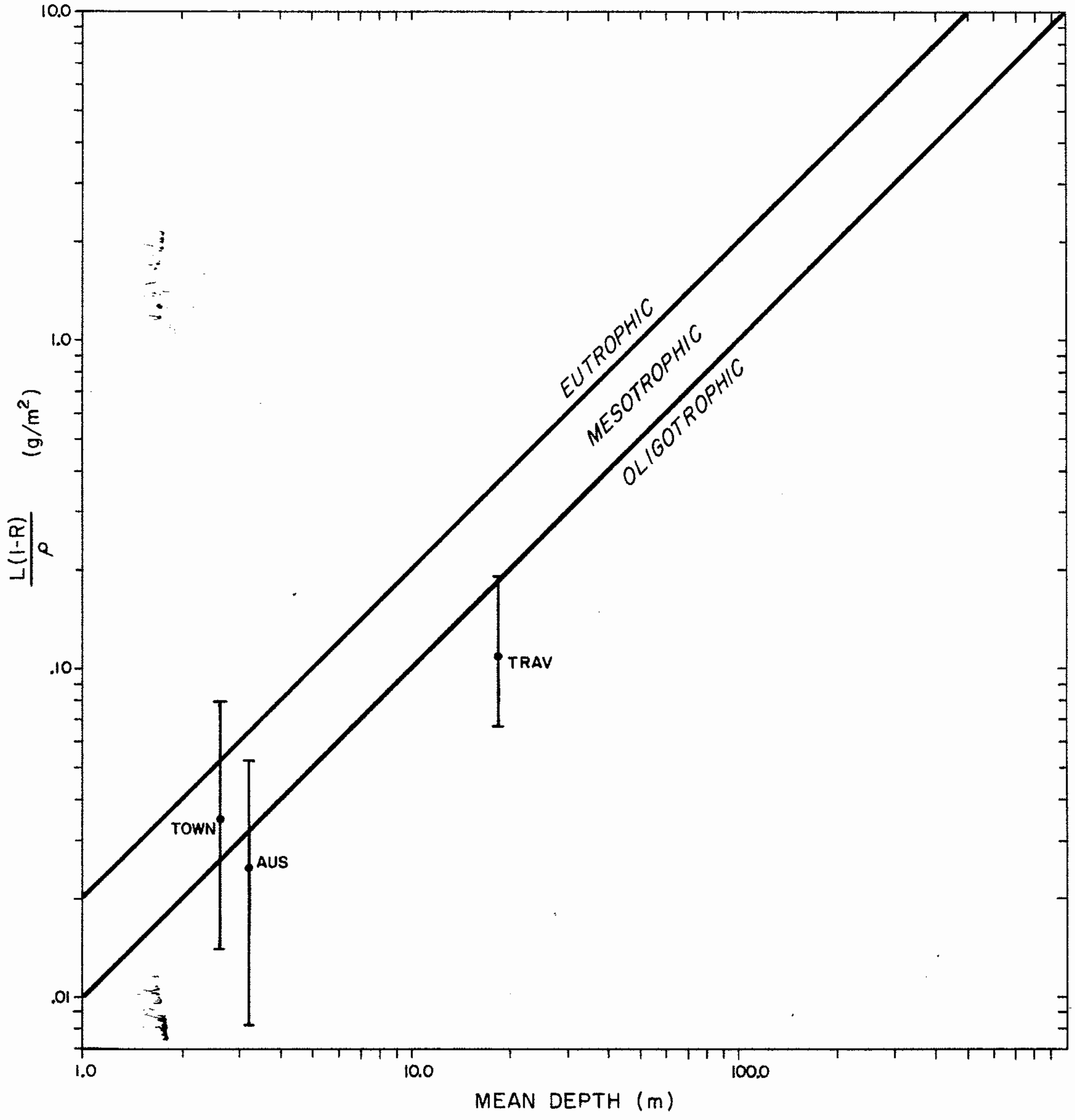

Fig. 6-5 DILLON LOADING PARAMETER-MEAN DEPTH MODEL, SCENARIO V RESULTS

TABLE 6-1

SIMULATION OF PHOSPHORUS CONCENTRATIONS
SCENARIO I
(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.012	.020	.002	.003	.010
1985	.018	.017	.010	.012	.004	.004	.006
1986	.020	.017	.027	.030	.005	.006	.010
1987	.014	.012	.015	.018	.004	.004	.008
1988	.042	.037	.026	.029	.005	.006	.007
1989	.004	.003	.011	.018	.003	.003	.008
1990	.021	.016	.015	.020	.004	.005	.010
1991	.041	.038	.015	.019	.005	.006	.009
1992	.016	.014	.008	.012	.005	.005	.009
1993	.008	.007	.012	.019	.003	.004	.009
1994	.018	.016	.012	.016	.004	.004	.006
1995	.013	.010	.009	.018	.005	.007	.014
1996	.006	.003	.006	.019	.005	.006	.018
1997	.027	.022	.019	.025	.004	.004	.008
1998	.010	.009	.012	.017	.005	.006	.013
1999	.020	.016	.014	.022	.003	.004	.008
2000	.004	.003	.015	.030	.003	.003	.009

TABLE 6-2

SIMULATION OF HISTORICAL PHOSPHORUS CONCENTRATIONS FOR 1967-1983

(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1967	.010	.007	.011	.012	.002	.002	.007
1968	.044	.040	.019	.019	.005	.005	.006
1969	.020	.017	.026	.025	.004	.005	.008
1970	.014	.012	.014	.014	.003	.004	.006
1971	.042	.037	.025	.024	.004	.004	.006
1972	.004	.003	.010	.010	.002	.002	.007
1973	.021	.016	.014	.014	.003	.004	.009
1974	.041	.038	.014	.014	.004	.005	.008
1975	.016	.014	.007	.007	.004	.004	.007
1976	.008	.007	.011	.012	.003	.003	.007
1977	.018	.016	.010	.012	.004	.004	.005
1978	.013	.010	.007	.011	.004	.005	.011
1979	.006	.003	.005	.012	.004	.005	.016
1980	.027	.022	.018	.021	.003	.003	.006
1981	.010	.009	.012	.015	.005	.005	.012
1982	.020	.016	.014	.019	.002	.003	.007
1983	.004	.003	.015	.023	.002	.003	.006

TABLE 6-3

SIMULATION OF PHOSPHORUS CONCENTRATIONS

SCENARIO II

(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.012	.020	.002	.003	.011
1985	.018	.017	.010	.012	.004	.004	.006
1986	.021	.017	.027	.030	.005	.006	.010
1987	.014	.012	.015	.018	.004	.005	.008
1988	.042	.037	.026	.029	.005	.006	.008
1989	.004	.003	.011	.018	.003	.003	.009
1990	.021	.016	.016	.020	.004	.005	.011
1991	.041	.038	.015	.020	.005	.006	.010
1992	.016	.014	.008	.012	.005	.005	.009
1993	.008	.007	.012	.019	.004	.004	.009
1994	.018	.016	.012	.016	.004	.005	.006
1995	.013	.010	.009	.019	.006	.007	.014
1996	.006	.003	.007	.020	.005	.007	.019
1997	.027	.022	.019	.025	.004	.004	.008
1998	.010	.009	.013	.018	.006	.007	.014
1999	.020	.016	.014	.023	.003	.004	.008
2000	.004	.003	.016	.031	.003	.004	.009

TABLE 6-4

SIMULATION OF PHOSPHORUS CONCENTRATIONS

SCENARIO III

(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.013	.021	.003	.003	.010
1985	.018	.017	.010	.013	.004	.004	.006
1986	.020	.017	.028	.031	.005	.006	.010
1987	.014	.012	.016	.019	.004	.005	.008
1988	.042	.037	.027	.030	.006	.006	.008
1989	.004	.003	.013	.020	.003	.003	.008
1990	.021	.016	.017	.022	.004	.005	.011
1991	.041	.038	.015	.020	.005	.006	.010
1992	.016	.014	.009	.013	.005	.005	.009
1993	.008	.007	.014	.020	.004	.004	.009
1994	.018	.016	.013	.017	.005	.005	.006
1995	.013	.010	.010	.020	.006	.007	.014
1996	.006	.003	.008	.021	.005	.006	.018
1997	.027	.022	.020	.026	.004	.004	.008
1998	.010	.009	.014	.018	.006	.006	.013
1999	.020	.016	.016	.024	.003	.004	.008
2000	.004	.003	.017	.031	.003	.004	.009

TABLE 6-5

SIMULATION OF PHOSPHORUS CONCENTRATIONS
SCENARIO IV
(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.013	.021	.004	.005	.012
1985	.018	.017	.010	.013	.005	.005	.007
1986	.020	.017	.028	.031	.007	.007	.011
1987	.014	.012	.016	.019	.006	.006	.009
1988	.042	.037	.027	.030	.007	.007	.009
1989	.004	.003	.013	.020	.004	.005	.010
1990	.021	.016	.017	.022	.005	.006	.012
1991	.041	.038	.015	.020	.006	.007	.011
1992	.016	.014	.009	.013	.006	.006	.010
1993	.008	.007	.014	.020	.005	.006	.010
1994	.018	.016	.013	.017	.006	.006	.007
1995	.013	.010	.010	.020	.008	.009	.016
1996	.006	.003	.008	.021	.006	.008	.019
1997	.027	.022	.020	.026	.006	.006	.009
1998	.010	.009	.014	.018	.007	.008	.015
1999	.020	.016	.016	.024	.005	.005	.010
2000	.004	.003	.017	.031	.005	.005	.010

TABLE 6-6

SIMULATION OF PHOSPHORUS CONCENTRATIONS

SCENARIO V

(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.012	.020	.004	.008	.018
1985	.018	.017	.010	.012	.005	.006	.008
1986	.020	.017	.027	.030	.007	.009	.014
1987	.014	.012	.015	.018	.005	.007	.011
1988	.042	.037	.026	.029	.006	.007	.009
1989	.004	.003	.011	.018	.004	.006	.013
1990	.021	.016	.015	.020	.006	.009	.018
1991	.041	.038	.015	.019	.007	.009	.014
1992	.016	.014	.008	.012	.007	.009	.014
1993	.008	.007	.012	.019	.004	.006	.013
1994	.018	.016	.012	.016	.005	.005	.007
1995	.013	.010	.009	.018	.008	.012	.022
1996	.006	.003	.006	.019	.007	.014	.030
1997	.027	.022	.019	.025	.004	.006	.011
1998	.010	.009	.012	.017	.009	.012	.022
1999	.020	.016	.014	.022	.004	.006	.012
2000	.004	.003	.015	.030	.004	.006	.013

TABLE 6-7

SIMULATION OF PHOSPHORUS CONCENTRATIONS

SCENARIO VI

(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.012	.020	.008	.011	.022
1985	.018	.017	.010	.012	.007	.008	.010
1986	.020	.017	.027	.030	.010	.012	.017
1987	.014	.012	.015	.018	.008	.009	.014
1988	.042	.037	.026	.029	.009	.010	.012
1989	.004	.003	.011	.018	.007	.009	.015
1990	.021	.016	.015	.020	.009	.012	.021
1991	.041	.038	.015	.019	.009	.012	.017
1992	.016	.014	.008	.012	.010	.011	.017
1993	.008	.007	.012	.019	.007	.010	.016
1994	.018	.016	.012	.016	.008	.008	.010
1995	.013	.010	.009	.018	.012	.016	.026
1996	.006	.003	.006	.019	.011	.017	.033
1997	.027	.022	.019	.025	.008	.010	.014
1998	.010	.009	.012	.017	.012	.015	.025
1999	.020	.016	.014	.022	.007	.010	.015
2000	.004	.003	.015	.030	.007	.009	.016

TABLE 6-8

SIMULATION OF PHOSPHORUS CONCENTRATIONS
UNDER DIRECT SUPPLEMENTAL LOADINGS
SCENARIO VII
(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.019	.020	.024	.039	.014	.017	.025
1985	.027	.022	.016	.017	.012	.009	.010
1986	.030	.027	.036	.039	.015	.015	.018
1987	.024	.020	.023	.025	.013	.011	.014
1988	.051	.044	.036	.038	.016	.015	.016
1989	.014	.015	.022	.033	.013	.017	.022
1990	.033	.029	.025	.031	.013	.015	.020
1991	.051	.047	.023	.027	.013	.012	.015
1992	.025	.022	.016	.018	.013	.010	.014
1993	.017	.018	.022	.030	.013	.015	.019
1994	.028	.024	.020	.024	.013	.010	.012
1995	.024	.025	.020	.033	.019	.020	.027
1996	.016	.031	.019	.038	.015	.021	.032
1997	.037	.032	.028	.035	.014	.017	.021
1998	.020	.020	.021	.025	.014	.013	.019
1999	.029	.026	.025	.035	.013	.014	.019
2000	.014	.027	.028	.051	.014	.022	.028

TABLE 6-9

SIMULATION OF PHOSPHORUS CONCENTRATIONS
SCENARIO VII, SUPPLEMENTAL LOAD TO LAKE BUCHANAN
(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.019	.014	.014	.022	.003	.003	.010
1985	.027	.025	.013	.015	.005	.005	.006
1986	.030	.025	.029	.032	.006	.006	.010
1987	.024	.020	.018	.021	.005	.005	.008
1988	.051	.045	.030	.033	.006	.006	.008
1989	.014	.010	.013	.020	.003	.003	.008
1990	.033	.025	.017	.022	.004	.005	.011
1991	.051	.047	.017	.021	.005	.006	.010
1992	.025	.022	.011	.014	.005	.005	.009
1993	.017	.014	.014	.020	.004	.004	.009
1994	.028	.024	.014	.018	.005	.005	.006
1995	.024	.018	.010	.020	.006	.007	.014
1996	.016	.007	.007	.019	.005	.006	.018
1997	.037	.030	.021	.027	.004	.004	.008
1998	.020	.016	.014	.019	.006	.007	.014
1999	.029	.024	.017	.025	.003	.004	.008
2000	.014	.008	.016	.030	.003	.003	.009

TABLE 6-10

SIMULATION OF PHOSPHORUS CONCENTRATIONS
SCENARIO VII, SUPPLEMENTAL LOAD TO LAKE LBJ
(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.024	.031	.003	.004	.011
1985	.018	.017	.016	.018	.005	.005	.007
1986	.020	.017	.036	.038	.006	.007	.011
1987	.014	.012	.023	.026	.005	.006	.009
1988	.042	.037	.036	.038	.007	.007	.009
1989	.004	.003	.022	.028	.003	.004	.009
1990	.021	.016	.025	.029	.005	.006	.011
1991	.041	.038	.023	.027	.006	.007	.010
1992	.016	.014	.016	.019	.006	.006	.010
1993	.008	.007	.022	.028	.004	.005	.009
1994	.018	.016	.020	.024	.005	.005	.007
1995	.013	.010	.020	.028	.007	.008	.015
1996	.006	.003	.019	.030	.005	.007	.018
1997	.027	.022	.028	.034	.005	.005	.009
1998	.010	.009	.021	.025	.007	.007	.014
1999	.020	.016	.025	.032	.004	.004	.009
2000	.004	.003	.028	.040	.003	.004	.009

TABLE 6-11

SIMULATION OF PHOSPHORUS CONCENTRATIONS
SCENARIO VII, SUPPLEMENTAL LOAD TO LAKE TRAVIS
(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.012	.020	.014	.014	.020
1985	.018	.017	.010	.012	.012	.012	.013
1986	.020	.017	.027	.030	.015	.015	.018
1987	.014	.012	.015	.018	.013	.013	.015
1988	.042	.037	.026	.029	.016	.016	.017
1989	.004	.003	.011	.018	.013	.013	.017
1990	.021	.016	.015	.020	.013	.014	.018
1991	.041	.038	.015	.019	.013	.014	.017
1992	.016	.014	.008	.012	.013	.013	.016
1993	.008	.007	.012	.019	.013	.014	.017
1994	.018	.016	.012	.016	.013	.013	.014
1995	.013	.010	.009	.018	.019	.020	.026
1996	.006	.003	.006	.019	.015	.017	.027
1997	.027	.022	.019	.025	.014	.015	.018
1998	.010	.009	.012	.017	.014	.015	.021
1999	.020	.016	.014	.022	.013	.014	.018
2000	.004	.003	.015	.030	.014	.015	.019

TABLE 6-12

SIMULATION OF PHOSPHORUS CONCENTRATIONS

SCENARIO IX

(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.006	.011	.020	.002	.003	.007
1985	.018	.017	.009	.012	.004	.004	.005
1986	.020	.017	.026	.030	.005	.005	.010
1987	.014	.012	.014	.018	.004	.004	.006
1988	.042	.037	.026	.029	.005	.005	.007
1989	.004	.003	.011	.018	.003	.003	.006
1990	.021	.016	.015	.020	.004	.004	.008
1991	.041	.038	.014	.019	.005	.005	.008
1992	.016	.014	.008	.012	.004	.005	.007
1993	.008	.006	.012	.018	.003	.004	.007
1994	.018	.016	.011	.016	.004	.004	.005
1995	.013	.010	.008	.018	.005	.006	.010
1996	.006	.003	.006	.019	.004	.006	.012
1997	.027	.022	.019	.025	.004	.004	.006
1998	.010	.008	.012	.017	.005	.006	.010
1999	.020	.016	.014	.022	.003	.004	.006
2000	.003	.003	.015	.030	.003	.003	.007

TABLE 6-13

SIMULATION OF PHOSPHORUS CONCENTRATIONS
SCENARIO X
(mg/l)

Year	Buchanan	Inks	LBJ	Marble Falls	Travis	Austin	Town
1984	.008	.007	.012	.020	.003	.005	.014
1985	.018	.017	.010	.012	.004	.005	.007
1986	.020	.017	.027	.030	.006	.007	.012
1987	.014	.012	.015	.018	.005	.005	.009
1988	.042	.037	.026	.029	.006	.006	.008
1989	.004	.003	.011	.018	.003	.004	.010
1990	.021	.016	.015	.020	.005	.006	.013
1991	.041	.038	.015	.019	.006	.007	.011
1992	.016	.014	.008	.012	.006	.006	.011
1993	.008	.007	.012	.019	.004	.005	.010
1994	.018	.016	.012	.016	.005	.005	.006
1995	.013	.010	.009	.018	.006	.008	.017
1996	.006	.003	.006	.019	.006	.009	.022
1997	.027	.022	.019	.025	.004	.005	.009
1998	.010	.009	.012	.017	.007	.008	.017
1999	.020	.016	.014	.022	.003	.004	.009
2000	.004	.003	.015	.030	.003	.004	.010

TABLE 6-14

SUMMARY OF SIMULATION RESULTS

Scenario	Description	Results
I	Baseline future urban growth in accordance with population projections	Total phosphorus levels generally low; generally no substantial elevation above historical conditions.
II	Similar to Scenario I; increase urban runoff phosphorus concentration 100 percent to 0.30 mg/l	Occasional small elevations in baseline phosphorus, maximum of 0.001 mg/l increase.
III	Similar to Scenario I; add 10,000-person point source load to Lake LBJ	Phosphorus in Lakes LBJ and Marble Falls increased by 0.001-0.002 mg/l.
IV	Similar to Scenario III; add 50,000-person point source load to Lake Travis	Phosphorus in Lake Travis increased 0.001-0.003 mg/l; increased 0.001-0.002 mg/l in Lake Austin and Town Lake.
V	Assume complete urbanization in watersheds of Lake Travis, Lake Austin, and Town Lake	Phosphorus increased 0.000-0.004 mg/l in Lake Travis, 0.001-0.008 mg/l in Lake Austin, 0.001-0.012 mg/l in Town Lake.
VI	Similar to Scenario V; add 100,000-person point source load to Lake Travis	Phosphorus increased 0.003-0.007 mg/l in Lake Travis, 0.004-0.011 mg/l in Lake Austin, 0.004-0.015 mg/l in Town Lake above baseline conditions.
VII	Add supplemental load to each reservoir, based on Scenario I, to elevate phosphorus by 0.01 mg/l the majority of the time	Average phosphorus concentrations 0.014-0.051 mg/l in Lake Buchanan, 0.015-0.047 mg/l in Inks Lake, 0.016-0.036 mg/l in Lake LBJ, 0.017-0.051 mg/l in Lake Marble Falls, 0.012-0.019 mg/l in Lake Travis, 0.009-0.022 mg/l in Lake Austin, 0.010-0.032 mg/l in Town Lake due to loads in those reservoirs.
VIII	Similar to Scenario VII, but add load to achieve 0.01 mg/l phosphorus increase in reservoir immediately downstream	Average phosphorus concentrations 0.008-0.049 mg/l in Lake Inks, 0.015-0.037 mg/l in Lake LBJ, 0.014-0.042 mg/l in Lake Marble Falls, 0.012-0.018 mg/l in Lake Travis, 0.012-0.020 mg/l in Lake Austin, 0.010-0.030 mg/l in Town Lake due to loads to upstream reservoirs.
IX	Similar to Scenario I, but decrease urban runoff loads by 50 percent	Occasional reductions in phosphorus of 0.001 mg/l, except for Town Lake with reductions of 0.000-0.006 mg/l.
X	Similar to Scenario V, but decrease urban runoff loads by 50 percent	Compared to Scenario V results without controls, attained reductions in phosphorus of 0.000-0.002 mg/l in Lake Travis, 0.000-0.005 mg/l in Lake Austin, 0.001-0.008 mg/l in Town Lake.

Note: Results describe increases relative to baseline (Scenario I) conditions, unless otherwise noted.

TABLE 6-15

RELATIONSHIPS BETWEEN PHOSPHORUS, CHLOROPHYLL-a, AND SECCHI DEPTH

Phosphorus (mg/l)	Chlorophyll-a (μg/l)	Secchi Depth (m)
.001	.073	21.9
.002	.20	13.6
.003	.36	10.3
.004	.54	8.5
.005	.75	7.3
.006	.98	6.4
.007	1.23	5.8
.008	1.49	5.3
.009	1.76	4.9
.010	2.06	4.5
.011	2.36	4.2
.012	2.68	4.0
.013	3.01	3.8
.014	3.35	3.6
.015	3.70	3.4
.016	4.06	3.3
.017	4.44	3.1
.018	4.82	3.0
.019	5.21	2.9
.020	5.61	2.8

TABLE 6-16

SUMMARY OF PROJECTED PHOSPHORUS LOADINGS TO THE HIGHLAND LAKES

SCENARIO I

Year	Phosphorus Loadings (10^6 g/yr)					
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric	Loading from Upstream Reservoir	Total Load
			- LAKE BUCHANAN -			
1984	0.013	3.68	50.73	1.04	--	55.46
1985	0.013	5.27	144.64	1.61	--	151.53
1986	0.014	5.06	145.01	1.49	--	151.57
1987	0.014	3.94	103.04	0.86	--	107.85
1988	0.015	2.44	296.73	1.21	--	300.40
1989	0.015	0.98	22.88	0.89	--	24.76
1990	0.016	1.95	121.64	1.19	--	124.80
1991	0.016	7.96	286.19	1.46	--	295.63
1992	0.017	3.45	119.47	1.19	--	124.13
1993	0.017	2.96	56.31	1.22	--	60.51
1994	0.018	4.51	132.34	1.04	--	137.91
1995	0.018	3.97	75.29	1.19	--	80.47
1996	0.019	2.26	34.01	1.17	--	37.46
1997	0.019	2.25	195.84	1.01	--	199.12
1998	0.020	4.34	71.38	1.24	--	76.98
1999	0.020	1.27	140.67	0.94	--	142.90
2000	0.021	6.18	15.62	1.22	--	23.04

TABLE 6-16 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr)					
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric	Loading from Upstream Reservoir	Total Load
			- INKS LAKE -			
1984	—	0.19	—	0.040	3.15	3.38
1985	—	0.27	—	0.062	24.30	24.63
1986	—	0.26	—	0.058	12.30	12.62
1987	—	0.20	—	0.033	11.50	11.73
1988	—	0.12	—	0.047	36.40	36.57
1989	—	0.05	—	0.034	1.54	1.62
1990	—	0.10	—	0.046	8.61	8.76
1991	—	0.40	—	0.057	28.70	29.16
1992	—	0.17	—	0.046	13.60	13.82
1993	—	0.15	—	0.048	3.75	3.95
1994	—	0.23	—	0.040	13.60	13.87
1995	—	0.20	—	0.046	4.54	4.79
1996	—	0.11	—	0.046	.59	.75
1997	—	0.11	—	0.039	16.10	16.25
1998	—	0.22	—	0.048	5.10	5.37
1999	—	0.06	—	0.036	11.80	11.90
2000	—	0.31	—	0.047	.51	.87

TABLE 6-16 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr) Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric	Loading from Upstream Reservoir	Total Load
			- LAKE LYNDON B. JOHNSON -			
1984	1.51	1.13	11.68	0.32	2.55	17.19
1985	1.56	1.64	3.74	0.50	22.50	29.94
1986	1.62	1.60	41.69	0.46	10.50	55.87
1987	1.67	1.26	18.49	0.26	10.30	31.98
1988	1.73	0.79	14.83	0.37	32.00	49.72
1989	1.78	0.32	14.62	0.27	1.26	18.25
1990	1.84	0.64	20.41	0.37	6.73	29.99
1991	1.87	2.63	4.67	0.45	24.20	33.82
1992	1.90	1.15	4.63	0.37	12.10	20.15
1993	1.93	0.99	15.98	0.38	3.12	22.40
1994	1.96	1.51	10.42	0.32	11.80	26.01
1995	1.98	1.34	7.62	0.37	3.56	14.87
1996	2.01	0.77	6.25	0.36	.38	9.77
1997	2.04	0.77	21.9	0.31	13.50	38.52
1998	2.07	1.48	20.42	0.38	4.31	28.66
1999	2.10	0.44	12.6	0.29	9.84	25.27
2000	2.13	2.12	16.47	0.38	.50	21.60

TABLE 6-16 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr)					
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric	Loading from Upstream Reservoir	Total Load
			- LAKE MARBLE FALLS -			
1984	5.07	0.44	—	0.039	5.68	11.23
1985	5.24	0.63	—	0.061	20.20	26.13
1986	5.45	0.60	—	0.056	28.70	34.81
1987	5.65	0.48	—	0.032	20.70	26.86
1988	5.86	0.29	—	0.046	31.10	37.30
1989	6.07	0.12	—	0.034	7.09	13.31
1990	6.27	0.24	—	0.045	14.20	20.76
1991	6.46	1.00	—	0.055	19.00	26.52
1992	6.64	0.42	—	0.045	11.90	19.00
1993	6.83	0.36	—	0.046	10.10	17.34
1994	7.01	0.57	—	0.039	14.40	22.02
1995	7.20	0.49	—	0.045	5.85	13.58
1996	7.41	0.28	—	0.044	3.23	10.96
1997	7.62	0.28	—	0.038	19.10	27.04
1998	7.83	0.55	—	0.047	16.00	24.43
1999	8.05	0.16	—	0.035	10.80	19.04
2000	8.26	0.80	—	0.046	6.13	15.24

TABLE 6-16 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr)					
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric	Loading from Upstream Reservoir	Total Load
			- LAKE TRAVIS -			
1984	0.17	3.34	3.08	1.07	10.00	17.66
1985	0.17	2.06	15.07	1.23	25.40	43.93
1986	0.17	2.90	11.09	1.07	32.90	48.13
1987	0.19	2.92	12.69	0.98	25.70	42.48
1988	0.24	1.01	7.22	0.79	35.60	44.86
1989	0.28	2.09	6.19	0.83	12.20	21.59
1990	0.33	4.83	11.15	1.29	19.50	37.10
1991	0.33	4.61	20.85	1.15	25.30	52.24
1992	0.33	5.66	23.59	1.17	18.30	49.05
1993	0.33	2.68	9.36	1.32	16.20	29.89
1994	0.33	1.29	19.43	0.70	21.00	42.75
1995	0.33	3.72	17.89	0.99	12.50	35.43
1996	0.33	6.18	20.24	1.19	9.80	37.74
1997	0.33	1.46	2.57	0.88	25.60	30.84
1998	0.33	9.24	19.75	1.45	23.30	54.07
1999	0.33	2.75	2.68	0.85	17.70	24.31
2000	0.33	1.39	5.09	1.08	13.50	21.39

TABLE 6-16 (Cont'd)

Year	Phosphorus Loadings (10^6 g/yr)					
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric	Loading from Upstream Reservoir	Total Load
			- LAKE AUSTIN -			
1984	—	0.65	—	0.126	2.03	2.81
1985	—	0.38	—	0.145	10.90	11.42
1986	—	0.56	—	0.126	7.43	8.12
1987	—	0.59	—	0.116	8.16	8.87
1988	—	0.19	—	0.093	7.56	7.84
1989	—	0.43	—	0.098	2.31	2.84
1990	—	0.99	—	0.153	4.70	5.84
1991	—	0.96	—	0.136	11.20	12.30
1992	—	1.17	—	0.138	11.60	12.91
1993	—	0.55	—	0.156	4.02	4.73
1994	—	0.26	—	0.083	9.41	9.75
1995	—	0.78	—	0.117	5.43	6.33
1996	—	1.32	—	0.141	4.14	5.60
1997	—	0.29	—	0.104	3.76	4.15
1998	—	1.99	—	0.172	10.90	13.06
1999	—	0.59	—	0.101	3.54	4.23
2000	—	0.28	—	0.127	1.99	2.40

TABLE 6-16 (Concluded)

Year	Phosphorus Loadings (10^6 g/yr)					
	Point Source	Adjacent Watershed Nonpoint Source	Major Tributary Nonpoint Source	Atmospheric	Loading from Upstream Reservoir	Total Load
			- TOWN LAKE -			
1984	—	5.64	.58	0.029	2.39	8.64
1985	—	3.72	1.46	0.033	10.90	16.11
1986	—	4.99	1.10	0.029	7.40	13.52
1987	—	4.74	1.51	0.026	8.23	14.51
1988	—	1.82	.81	0.021	7.01	9.66
1989	—	3.31	1.53	0.022	2.40	7.26
1990	—	6.05	1.48	0.035	5.20	12.76
1991	—	7.16	1.51	0.031	11.40	20.10
1992	—	8.83	1.75	0.031	12.20	22.81
1993	—	4.23	1.55	0.035	4.14	9.96
1994	—	2.13	1.58	0.019	9.10	12.83
1995	—	5.75	.56	0.026	5.34	11.68
1996	—	9.29	1.31	0.032	4.67	15.30
1997	—	2.36	.75	0.024	3.56	6.69
1998	—	13.69	1.20	0.039	12.10	27.03
1999	—	4.09	.86	0.023	3.73	8.70
2000	—	2.31	1.07	0.029	1.89	5.30

TABLE 6-17

SUMMARY DESCRIPTION OF MASS LOADINGS FOR SIMULATION SCENARIOS

Scenario	Description	Results
I	Baseline future urban growth in accordance with population projections	Buchanan: tributary loads dominate Inks: upstream release loads dominate LBJ: tributary loads usually largest, upstream release loads comparable Marble Falls: upstream release loads largest, point source loads substantial Travis: upstream release loads usually largest, tributary loads comparable Austin: upstream release loads largest, runoff loads occasionally substantial Town: runoff loads usually largest, upstream release loads comparable
II	Similar to Scenario I; increase urban runoff phosphorus concentration 100 percent to 0.30 mg/l	Similar to Scenario I, no substantial changes in relative magnitude
III	Similar to Scenario I; add 10,000-person point source load to Lake LBJ	Buchanan: same as Scenario I Inks: same as Scenario I LBJ: tributary and upstream release loads largest, point source loads occasionally comparable Marble Falls: similar to Scenario I Travis: similar to Scenario I Austin: similar to Scenario I Town: similar to Scenario I
IV	Similar to Scenario III; add 50,000-person point source load to Lake Travis	Buchanan: same as Scenario I Inks: same as Scenario I LBJ: same as Scenario III Marble Falls: same as Scenario III Travis: upstream release loads usually largest, tributary loads comparable, point source loads occasionally largest Austin: upstream release loads largest Town: upstream release loads usually largest, runoff loads comparable
V	Assume complete urbanization in watersheds of Lake Travis, Lake Austin, and Town Lake	Buchanan: same as Scenario I Inks: same as Scenario I LBJ: same as Scenario I Marble Falls: same as Scenario I Travis: upstream release loads usually largest, tributary loads and runoff loads are comparable Austin: upstream release loads largest, runoff loads occasionally comparable Town: upstream release loads usually largest, runoff loads comparable

TABLE 6-17 (Cont'd)

Scenario	Description	Results
VI	Similar to Scenario V; add 100,000-person point source load to Lake Travis	Buchanan: same as Scenario I Inks: same as Scenario I LBJ: same as Scenario I Marble Falls: same as Scenario I Travis: point source loads usually largest, upstream release loads and tributary loads often comparable Austin: upstream release loads largest, runoff loads occasionally comparable Town: upstream release loads usually largest, runoff loads occasionally comparable
VII	Add supplemental load to each reservoir, based on Scenario I, to elevate phosphorus by 0.01 mg/l the majority of the time	Buchanan: tributary loads usually largest, point source loads comparable and occasionally largest Inks: upstream release loads usually largest, point source loads comparable and occasionally largest LBJ: point source loads usually largest, tributary and upstream release loads often comparable Marble Falls: point source loads usually largest, upstream release loads comparable or larger Travis: point source loads dominate Austin: point source loads largest, upstream release loads often comparable Town: point source loads usually dominate, upstream release loads and runoff loads occasionally comparable
VIII	Similar to Scenario VII, but add load to achieve 0.01 mg/l phosphorus increase in reservoir immediately downstream (results for reservoir downstream of supplemental load)	Buchanan: not applicable Inks: upstream release loads dominate LBJ: upstream release loads largest, tributary loads often comparable Marble Falls: upstream release loads largest Travis: upstream release loads dominate Austin: upstream release loads dominate Town: upstream release loads dominate
IX	Similar to Scenario I, but decrease urban runoff loads by 50 percent	Buchanan: similar to Scenario I Inks: similar to Scenario I LBJ: similar to Scenario I Marble Falls: similar to Scenario I Travis: similar to Scenario I Austin: upstream release loads largest Town: upstream release loads largest, runoff loads often comparable

TABLE 6-17 (Concluded)

Scenario	Description	Results
X	Similar to Scenario V, but decrease urban runoff loads by 50 percent	Buchanan: same as Scenario I Inks: same as Scenario I LBJ: same as Scenario I Marble Falls: same as Scenario I Travis: upstream release loads usually largest, tributary and runoff loads are often comparable Austin: upstream release loads largest, runoff loads occasionally comparable Town: upstream release loads usually largest, runoff loads comparable

TABLE 6-18

MEAN PHOSPHORUS LOADS FOR SIMULATION SCENARIOS

Case	Phosphorus Loadings (10^6 g/yr)					
	Mean Point Source	Mean Runoff	Mean Trib.	Mean Atmos.	Mean Upstream	Mean Total
			- Lake Buchanan -			
I	.017	3.67	118.34	1.17	--	123.21
II	.017	4.22	118.34	1.17	--	123.75
III	.017	3.67	118.34	1.17	--	123.21
IV	.017	3.67	118.34	1.17	--	123.21
V	.017	3.67	118.34	1.17	--	123.21
VI	.017	3.67	118.34	1.17	--	123.21
VII	69.00	3.67	118.34	1.17	--	192.19
VIII	.017	3.67	118.34	1.17	--	123.21
IX	.017	3.41	118.34	1.17	--	122.95
X	.017	3.67	118.34	1.17	--	123.21
			- Inks Lake -			
I	--	.19	--	.045	11.53	11.77
II	--	.20	--	.045	11.60	11.84
III	--	.19	--	.045	11.53	11.77
IV	--	.19	--	.045	11.53	11.77
V	--	.19	--	.045	11.53	11.77
VI	--	.19	--	.045	11.53	11.77
VII	7.00	.19	--	.045	11.53	18.77
VIII	--	.19	--	.045	18.20	18.43
IX	--	.18	--	.045	11.51	11.74
X	--	.19	--	.045	11.53	11.77
			- Lake LBJ -			
I	1.86	1.21	14.50	.36	9.95	27.88
II	1.86	1.85	14.50	.36	9.98	28.55
III	4.62	1.21	14.50	.36	9.95	30.66
IV	4.62	1.21	14.50	.36	9.95	30.66
V	1.86	1.21	14.50	.36	9.95	27.88
VI	1.86	1.21	14.50	.36	9.95	27.88
VII	20.86	1.21	14.50	.36	9.95	46.88
VIII	1.86	1.21	14.50	.36	29.66	47.59
IX	1.86	.89	14.50	.36	9.92	27.53
X	1.86	1.21	14.50	.36	9.95	27.88

TABLE 6-18 (Cont'd)

Case	Phosphorus Loadings (10^6 g/yr)					
	Mean Point Source	Mean Runoff	Mean Trib.	Mean Atmos.	Mean Upstream	Mean Total
			- Lake Marble Falls -			
I	6.64	.46	--	.044	14.36	21.50
II	6.64	.59	--	.044	14.70	21.97
III	6.64	.46	--	.044	15.70	22.84
IV	6.64	.46	--	.044	14.36	21.50
V	6.64	.46	--	.044	14.36	21.50
VI	6.64	.46	--	.044	14.36	21.50
VII	17.65	.46	--	.044	14.36	32.51
VIII	6.64	.46	--	.044	24.60	31.74
IX	6.64	.38	--	.044	14.19	21.26
X	6.64	.46	--	.044	14.36	21.50
			- Lake Travis -			
I	.29	3.42	12.23	1.06	20.26	37.26
II	.29	4.39	12.23	1.06	20.72	38.69
III	.29	3.42	12.23	1.06	21.55	38.54
IV	14.14	3.42	12.23	1.06	21.55	52.40
V	.29	16.61	12.23	1.06	20.26	50.46
VI	27.94	16.61	12.23	1.06	20.26	78.11
VII	85.27	3.42	12.23	1.06	20.26	122.25
VIII	.29	3.42	12.23	1.06	104.51	121.50
IX	.29	2.93	12.23	1.06	20.03	36.54
X	.29	8.30	12.23	1.06	20.26	42.15
			- Lake Austin -			
I	--	.70	--	.13	6.42	7.25
II	--	1.05	--	.13	6.66	7.83
III	--	.70	--	.13	6.61	7.44
IV	--	.70	--	.13	8.86	9.69
V	--	3.16	--	.13	8.63	11.92
VI	--	3.16	--	.13	13.09	16.38
VII	13.00	.70	--	.13	6.42	20.25
VIII	--	.70	--	.13	20.09	20.92
IX	--	.53	--	.13	6.29	6.94
X	--	1.58	--	.13	7.24	8.95

TABLE 6-18 (Concluded)

Case	Phosphorus Loadings (10^6 g/yr) Mean Point Source	Mean Runoff	Mean Trib.	Mean Atmos.	Mean Upstream	Mean Total
			- Town Lake -			
I	--	5.30	1.21	.028	6.57	13.11
II	--	5.44	1.21	.028	7.13	13.81
III	--	5.30	1.21	.028	6.77	13.31
IV	--	5.30	1.21	.028	8.78	15.32
V	--	8.16	1.21	.028	10.78	20.18
VI	--	8.16	1.21	.028	14.78	24.18
VII	12.00	5.30	1.21	.028	6.57	25.11
VIII	--	5.30	1.21	.028	18.91	25.42
IX	--	2.87	1.21	.028	6.31	10.41
X	--	6.44	1.21	.028	8.10	15.78

TABLE 6-19

MEAN SCENARIO I CHARACTERISTICS FOR SIMPLE TROPHIC MODELS

Reservoir	Phosphorus Load (10^6 g/yr)	Area (10^6 m^2)	L (10^6 $g/m^2/yr$)	Volume (10^6 m^3)	Outflow (10^6 m^3/yr)	τ (yr)	ρ (yr^{-1})	z (m)	Outflow Load (10^6 g/yr)	R	$\frac{z}{\tau}$ (m/yr)	$\frac{L(1-R)}{\rho}$ (g/m^2)
Buchanan	123.21	89.0	1.38	1,095.1	580.8	1.89	.53	12.3	11.53	.91	6.51	.23
Inks	11.77	3.25	3.62	20.9	584.5	.04	27.97	6.43	9.95	.15	160.0	.11
LBJ	27.88	25.8	1.08	164.2	1,020.1	.16	6.21	6.36	14.36	.48	40.0	.090
Marble Falls	21.50	3.16	6.80	10.1	1,054.4	.0096	104.4	3.2	20.26	.06	333.3	.061
Travis	37.26	71.2	0.52	1,318.7	1,406.1	.94	1.07	18.5	6.42	.83	19.68	.083
Austin	7.25	7.41	0.98	23.7	1,360.7	.017	57.41	3.2	6.57	.09	188.2	.015
Town	13.11	1.68	7.80	4.3	1,435.4	.003	333.8	2.6	12.95	.01	866.7	.023

Notes: L = areal loading rate
τ = hydraulic retention time, volume/outflow
ρ = flushing rate, outflow/volume
z = mean depth
R = phosphorus retention coefficient

TABLE 6-20

RANGES OF SCENARIO I LOADING PARAMETERS

Reservoir	Phosphorus Load max (10^6 g/yr)	L max ($g/m^2/yr$)	Phosphorus Load min (10^6 g/yr)	L min ($g/m^2/yr$)	$\frac{L(1-R)}{\rho}$ max (g/m^2)	$\frac{L(1-R)}{\rho}$ min (g/m^2)
Buchanan	300.4	3.38	24.76	.28	.56	.047
Inks	36.57	11.3	.75	.23	.34	.0070
LBJ	55.87	2.17	9.77	.38	.18	.032
Marble Falls	37.3	11.8	10.96	3.47	.11	.031
Travis	54.07	.76	17.66	.25	.12	.040
Austin	13.06	1.76	2.4	.32	.027	.0049
Town	27.03	16.1	5.3	3.15	.047	.0093

TABLE 6-21

TROPHIC CLASSIFICATION

Parameter	Oligotrophic	Mesotrophic	Eutrophic
GENERAL DESCRIPTION			
Aquatic plant production	low	--	high
Algal blooms	rare	--	frequent
Algal species variety	many	--	variable to few
Characteristic algal groups	varied	--	blue-green
Littoral zone aquatic plant growth	sparse	--	abundant
Number of plant and animal species	many	--	few
Oxygen inthe hypolimnion	present	--	absent
NUMERICAL DESCRIPTION			
Chlorophyll-a (μg/l)	< 7	7-12	> 12
Total phosphorus (μg/l)	< 10	10-20	> 20
Secchi depth (m)	> 3.7	2.0-3.7	<2.0

From Rast and Lee (1978), EPA (1974).

CHAPTER 7.0

SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

7.1 SUMMARY OF ANALYSIS

7.1.1 Methodology

Phosphorus concentrations within the Highland Lakes were estimated with the model HILAKES. The model treated the Highland Lakes as a series of completely mixed reactors. For each reactor, or reservoir, a mass budget equation was evaluated. The mass budget equation required the definition of all phosphorus mass inputs to the reservoirs, volume and outflow terms, and reaction rates. For the present analysis, phosphorus dynamics were approximated by a first-order decay term, which can be construed as an apparent sedimentation rate. Average annual conditions were simulated with the model HILAKES, therefore, all input parameters were defined as yearly totals.

Annual mass loadings were calculated for point sources, runoff from adjacent watersheds, major tributaries, and atmospheric contributions. Historical phosphorus loadings were defined for the period 1952-1983. Future conditions were projected for the years 1984-2000. The fundamental hydrologic characteristics for the period 1967-1983 were assumed to be repeated for the future simulation period. Point source loads were estimated from current wastewater discharge permit limitations and historical data. Projected point source loads were based upon forecasts of population growth for area counties and discharge permit limitations. The nonpoint source loading estimates from adjacent watersheds were based upon runoff volumes calculated with annual storm characteristics. The SCS curve number technique was employed for calculations of runoff volumes. Runoff phosphorus concentration was estimated from analysis of minor tributary data collected in the Austin area. Runoff loads were calculated from the runoff volume and concentration characteristics. For determination of major tributary loads, historical instream phosphorus data were analyzed and load-discharge relationships were developed. Major tributary historical loads were derived by application of the loading relationships to recorded daily discharge values. Projected tributary loads were based upon the assumption that historical flow regimes for the period 1967-1983 would be repeated for the period 1984-2000. Atmospheric loads were based upon recorded precipitation volumes and an assumed phosphorus level.

7.1.2 Simulation Results

The model HILAKES was employed for simulation of future phosphorus concentrations in the Highland Lakes under several alternative scenarios for the period 1984-2000. The scenarios were formulated to depict anticipated conditions and conditions that may potentially occur, and investigate the interrelationships of water quality through the series of reservoirs. Simulation results were summarized in Table 6-14.

Under the anticipated Scenario I conditions, projected phosphorus levels for the period 1984-2000 generally did not display substantial increases over the simulated historical levels for 1967-1983. Projected concentrations in Lake Buchanan and Inks Lake remained unchanged. Projected concentrations in Lake LBJ displayed increases in phosphorus concentration of 0.000- 0.002 mg/l, while increases of 0.002-0.008 mg/l were estimated in Lake Marble Falls. In Lake Travis, increases of 0.000-0.001 mg/l above historical levels were projected. Increases of 0.000-0.002 mg/l phosphorus were projected for Lake Austin. Phosphorus increases of 0.000-0.003 mg/l above historical levels were projected for Town Lake under Scenario I assumptions.

Additional scenarios examined increases in urban mass loadings and supplemental point source loadings. Increasing the assumed urban runoff phosphorus concentration from 0.015 to 0.030 mg/l throughout the adjacent watersheds (Scenario II) had only small effects upon predicted concentrations, as elevations in reservoir phosphorus levels of 0.001 mg/l above the baseline Scenario I results were occasionally obtained. The hypothetical case of complete urbanization of the adjacent watersheds of Lake Travis, Lake Austin, and Town Lake was examined in Scenario V. Projected phosphorus levels showed increases of 0.000-0.004 mg/l in Lake Travis, 0.001-0.008 mg/l in Lake Austin, and 0.001-0.012 mg/l in Town Lake.

Under Scenario III, a supplemental point source load to Lake LBJ, equivalent to a population of 10,000, increased predicted phosphorus concentrations above baseline (Scenario I) conditions by 0.001-0.002 mg/l. Addition of a point source load to Lake Travis, equivalent to a population of 50,000, resulted in projected increases in phosphorus levels of 0.001-0.003 mg/l above baseline conditions (see Scenario IV). A point source load equivalent to a population of 100,000, coupled with the complete urbanization assumption, was simulated in Scenario VI. Projected increases in phosphorus levels of 0.003-0.007 mg/l in Lake Travis, 0.004-0.011 mg/l in Lake Austin, and 0.004-0.015 mg/l in Town Lake were obtained.

Supplemental loadings required to effect a 0.010 mg/l increase in ambient phosphorus concentrations were defined in Scenario VII. A supplemental load of 85×10^6 g/yr was defined for Lake Travis, for example. In Scenario VIII, loadings required to produce a 0.01 mg/l increase in phosphorus concentration within the reservoir immediately downstream were investigated.

The effects of urban runoff control measures were investigated in Scenarios IX and X, assuming a 50 percent reduction in urban loadings. Under Scenario IX, the effects on the baseline conditions of Scenario I were examined. With the reduction in urban loads, decreases in projected phosphorus levels of 0.001 mg/l were occasionally observed in the series of reservoirs, though concentrations in Town Lake decreased by 0.000-0.006 mg/l. Under Scenario X, the urban loadings generated in Scenario V with complete urbanization were reduced, resulting in decreases in predicted phosphorus concentrations of 0.000-0.002 mg/l in Lake Travis, 0.000- 0.005 mg/l in Lake Austin, and 0.001-0.008 mg/l in Town Lake, compared to the projections for Scenario V.

As a general conclusion, projected reservoir phosphorus concentrations for future conditions appear to be in line with historical levels. In terms of elevations in phosphorus concentration, low increases were generally projected, and future concentrations should remain at relatively low levels, consistent with historical trends.

The projected phosphorus concentrations were interpreted in terms of chlorophyll-a levels and Secchi depths with empirical relationships that are frequently used. It is probable that the relationships do not hold precisely for the Highland Lakes, due to background levels of turbidity. Based upon the empirical relationships, the projected phosphorus increments would entail no substantial elevation of chlorophyll-a levels or deterioration of transparency.

Mean loadings for the various sources under the future simulation scenarios were calculated. The mean load from upstream releases was usually the largest contributor of phosphorus mass to most of the reservoirs, frequently an order of magnitude in excess of loads from other sources. Major tributary loads usually constituted the largest source of phosphorus loads for Lake Buchanan and Lake LBJ. Runoff loads from adjacent watersheds were occasionally substantial, sometimes of the same order or magnitude as other sources. Loads associated with direct precipitation input were consistently low.

The interrelationships of water quality among the reservoirs were exhibited by several of the scenarios. The magnitude and longitudinal extent of the propagation of effects on downstream reservoirs was dependent upon the total external mass loading to each reservoir and the outflow through the series of reservoirs. In situations where the total external mass loading to a reservoir is dominated by the loading associated with releases from an upstream reservoir, the effect is potentially more substantial. Downstream effects are generally more substantial with larger releases from the upstream reservoir. Effects upon concentration from upstream loadings are potentially much greater in the extremely throughflow-dominated reservoirs: Inks Lake, Lake Marble Falls, Lake Austin and Town Lake. Any increment in concentration achieved in the upstream reservoir will in effect be realized in the throughflow-dominated impoundment below. Conversely, a larger, less throughflow-dominated reservoir such as Lake Travis is less likely to demonstrate substantial increases in concentration in response to a mass input to an upstream waterbody.

The efficacy of urban runoff controls in terms of realization of an improvement in phosphorus concentrations within the reservoir was addressed. Compared to the predicted concentrations obtained without runoff controls, imposition of a fifty percent reduction in urban loadings resulted in a reduction in the magnitude of reservoir phosphorus concentration increases. However, since the projected effects of urban loads without controls were generally small, the net benefits of the simulated control measures were not readily ascertained. This analysis did not constitute a condemnation of urban runoff control measures, as the benefits of mitigation of urban runoff loads transcend the implications on phosphorus concentrations.

The simulation results demonstrated that limitation of the phosphorus level in point source effluent is prudent. The effects of a supplemental discharge to Lake Travis, for example, equivalent to the discharge from a population of 100,000, were examined. Under an assumed effluent phosphorus level of 2 mg/l, the point source load was estimated at 27.6×10^6 g/yr. If phosphorus levels were not limited in the effluent, the point source load would be estimated at 138×10^6 g/yr, assuming an effluent concentration of 10 mg/l phosphorus. Such differences in loading would have a marked effect upon predicted reservoir phosphorus concentrations.

Mass loading and hydrologic data for Scenario I were applied to three empirical models that are commonly used to assess the trophic state of a waterbody. With the Vollenweider loading-mean depth model, all of the reservoirs were classified as eutrophic. Under the Vollenweider phosphorus loading-hydraulic loading model, mean characteristics for Lake Buchanan, Lake LBJ, and Inks Lake indicated a state of eutrophy, a mesotrophic state was indicated for Lake Travis and Lake Marble Falls, while mean values for Lake Austin and Town Lake were situated within the oligotrophic region. With the Dillon loading parameter-mean depth model, mesotrophic conditions were indicated for Lake Buchanan, Inks Lake, Lake LBJ, and Lake Marble Falls, while Lake Travis, Lake Austin, and Town Lake were categorized as oligotrophic. Classification results for alternative scenarios were also described. The analysis indicated that the simple empirical models provide widely varying predictions of trophic state for the Highland Lakes. These models may have significant limitations in their applicability to the Highland Lakes. The models were derived with data for lakes outside of the southwestern United States; manmade reservoirs were generally absent from the data base. Among the limitations of the simple trophic state models for application to the Highland Lakes, an important factor may be the effects of higher background turbidity levels on light availability for phytoplankton, which in turn may affect the critical nutrient concentrations. The effects of thermal stratification and reservoir operational characteristics are additional potential limitations.

7.2 CONCLUSIONS

The present analysis comprised a detailed inventory and projection of phosphorus loadings to the Highland Lakes and simulation of average phosphorus concentrations within the series of reservoirs for the period 1984-2000. It is appropriate to review several key assumptions which permeated the analysis. The analysis was based upon simulation of the impoundments as a series of completely mixed reactors; as such, it was assumed that mass loadings to each reservoir were instantaneously dispersed, and the contents of each reservoir displayed a homogeneous constituent concentration at all times. With the assumed instantaneous dispersal of mass and homogeneous concentration, calculated effects of loadings to a specific reservoir--representative of a fraction of the original load--are propagated instantaneously downstream, subject to attenuation through the series of reservoirs. Annual mass input loads of phosphorus were determined for use in the modeling analysis, and results were obtained in terms of annual average phosphorus concentrations within the reservoirs. (The terminology of annual average concentration denotes a temporally and spatially averaged value over a one-year period.) Thus, transient or localized fluctuations in mass loadings or phosphorus concentrations were not included in the present evaluation. It should be noted that fluctuations (for example, seasonal) in phosphorus concentrations are frequently of great importance in the determination of the biological dynamics of natural lakes and reservoirs. These integral assumptions are appropriate for satisfaction of the fundamental objective of this investigation, which was to analyze long-term trends in water quality in the impoundments, and more specifically, to evaluate the implications of point and nonpoint source control measures upon long-term, annual average reservoir phosphorus concentrations. The approach and assumptions of this investigation may not be appropriate for other research or management objectives.

With the present level of knowledge of the water quality characteristics of the Highland Lakes, and in particular the existing limitations in the available data base, completion of this analysis constitutes a necessary and appropriate first step to securing a more comprehensive understanding of the potential effects of point and nonpoint source mass loadings upon long-term phosphorus concentrations in the reservoirs. There are many other water quality-related issues for which alternative analytical approaches may be more appropriate. For example, evaluation of localized effects of a specific point source discharge would require a more refined spatial and perhaps temporal resolution. Similarly, a more intensive temporal scale would be necessary to evaluate the transient effects of stormwater runoff from a single precipitation event. However, a more sophisticated or detailed analysis will require a more comprehensive input data set.

Recognizing the objectives of the present analysis and the underlying assumptions and data availability, the following conclusions were formulated:

1. The historical data base of phosphorus measurements for the Highland Lakes available for the present analysis was extremely limited, both in terms of temporal and spatial coverage. Data were generally available over the period 1968-1981. The routine monitoring program of the Texas Department of Water Resources was the source of the majority of the historical data base, providing quarterly measurements for several years, supplemented by less frequent sampling in several other years. These data consisted of surface measurements of phosphorus concentration collected at a lower pool station and usually one or more additional sites within each reservoir. Other data sources included the Center for Research in Water Resources at The University of Texas at Austin and the City of Austin. Limitations in the data base constrained the analyses conducted in this study.

2. Profiles of historical phosphorus concentration through the series of reservoirs were presented for several sampling surveys, indicating that

concentrations generally decrease from the upper to the lower reservoirs. The analyses were based upon surface layer measurements of phosphorus concentration, because these were the only data generally available. The highest concentrations usually were encountered in Lake Buchanan, at a typical range of 0.01-0.05 mg/l, with higher values occasionally detected in the headwater reach. In the reservoirs below Lake Buchanan, the majority of observations ranged from 0.01-0.02 mg/l total phosphorus. Values below the analytical detection limit (0.01 mg/l) frequently were encountered, particularly in the lower reaches of Lake Travis. Sporadic increases in phosphorus concentrations observed within the series of reservoirs appeared to be localized, probably in response to tributary inputs.

3. The historical phosphorus concentrations through the series of reservoirs were affected by hydraulic retention time and internal reactions. The retention time is in effect the time required for inflow loads at the head of a reservoir to demonstrate effects upon concentrations in the lower pool, based upon hydraulics only. Thus, phosphorus concentrations observed in headwater reaches are reflective of recent incoming mass loads, but the effects require longer periods of time to become manifest in the lower pool, particularly in the larger reservoirs. Lower pool concentrations in the larger impoundments would be expected to be influenced primarily by internal reactions rather than incoming mass loads. In the smaller, throughflow-dominated impoundments, the observed concentrations were influenced predominantly by recent mass loadings, rather than internal kinetics.

4. There was no indication of marked temporal or seasonal trends in the limited historical phosphorus data base. The analysis of seasonal trends may have been deleteriously influenced by an inadequate number of samples and insufficient sampling frequency, effects of hydraulic retention time, and effects of thermal stratification and mixing conditions.

5. Phosphorus loadings to the Highland Lakes from point and nonpoint sources were estimated as annual mass inputs for historical (1952-1983) and projected future (1984-2000) conditions. Point sources included several domestic wastewater treatment facilities. Inflow from major tributaries, stormwater runoff from adjacent watersheds, and direct precipitation were included as nonpoint sources. Releases of water from upstream reservoirs constituted an additional type of nonpoint source loading. Mass loadings associated with releases from an upstream reservoir were usually the largest component of the total external phosphorus load to most of the reservoirs, frequently an order of magnitude in excess of loads from other sources. Major tributary loadings usually constituted the largest source of phosphorus to Lake Buchanan and Lake LBJ. Runoff loadings from adjacent watersheds and loadings from point sources were occasionally substantial, sometimes of the same order of magnitude as other source

loads. Loadings from point sources were substantial only to Lake Marble Falls and, to a lesser extent, Lake LBJ. Atmospheric loads were relatively minor.

The mass loadings entailed characteristic phosphorus concentrations for the respective sources. Point sources were typically characterized by a phosphorus concentration of 2 mg/l (as phosphorus) where permit limitations were imposed, and an assumed concentration of 10 mg/l (as phosphorus) for uncontrolled discharges. A concentration of 0.02 mg/l phosphorus was assumed for direct precipitation to the reservoirs. Stormwater runoff loads from adjacent watersheds represented a composite of phosphorus contributions of 0.03 mg/l from rangeland, 0.02 mg/l from forest land, 0.10 mg/l from cropland, 0.15 mg/l from urban areas, and 0.60 mg/l from the Austin urban area. A phosphorus concentration of 0.018 mg/l was assumed for Barton Springs inflow. For inflow from major tributaries, the projected future mean loadings constituted approximate long-term average phosphorus concentrations of 0.19 mg/l in the Colorado River, 0.03 mg/l in the Llano River and Sandy Creek, and 0.05 mg/l in the Pedernales River. The phosphorus concentrations associated with releases to an impoundment downstream were defined by the simulated average concentrations within the upstream reservior. For example, a long-term average phosphorus concentration of 0.02 mg/l characterized releases from Lake Buchanan under the projected future baseline conditions, consistent with the projected average concentration within the reservoir.

6. A variety of modeling approaches can be utilized for simulation of water quality in natural systems. The alternative approaches vary significantly in complexity and sophistication, ranging from relatively simplistic empirical models to extremely detailed models of complete ecosystems. Associated with increases in complexity are additional demands upon the input data base necessary for formulation of model parameters and rate coefficients. There is no single model that is universally suitable for all potential applications. Selection of a modeling approach must be based upon an assessment of the ultimate objectives of a study and the nature and extent of the available data base. For the present analysis, a modeling approach that examined the Highland Lakes as a series of continuously-stirred tank reactors (CSTR) was employed. This approach constitutes a relatively convenient, economical, and straightforward analysis of a complex system. The continuously-stirred tank reactor model has enjoyed successful and widespread usage for a variety of environmental studies, and is particularly useful for evaluation of long-term water quality management strategies. This approach was appropriate for the objective of simulating long-term average reservoir phosphorus concentrations in response to point and nonpoint source mass loadings expressed on an annual basis, particularly with recognition of the limitations in the available data base. More sophisticated modeling techniques require a much more comprehensive set of input data than is presently available for the Highland Lakes.

7. A water quality model was developed for the Highland Lakes and was applied to the simulation of historical and projected future reservoir phosphorus concentrations. The model analyzed the reservoirs as an interdependent series of continuously-stirred tank reactors and considered influx of mass from various sources, outflow of mass in association with reservoir releases, and an internal first-order decay term for removal of phosphorus from the water column. The first-order decay term constituted an apparent sedimentation rate.

8. Prior to its application for simulation of reservoir phosphorus concentrations, the water quality model was calibrated with the historical data base for the Highland Lakes. The calibration exercises entailed adjustment of the first-order decay rate coefficient in order to achieve agreement between simulated concentrations and measured data. Data describing annual average concentrations of phosphorus in the reservoirs were employed, although the limited availability of historical data severely handicapped the calibration exercises. A first-order decay rate of 6.0 yr^{-1} was obtained in the calibration process; however, recognizing the limitations in the data base, a value in the range of 1.0-10.0 yr^{-1} could be used.

9. With the continuously-stirred tank reactor modeling approach, projected annual average phosphorus concentrations in the reservoirs for anticipated future conditions were similar to simulated average historical levels. Small increases in phosphorus concentration were generally predicted, and future concentrations were projected to remain at relatively low levels, consistent with simulated and measured historical trends.

10. Projected phosphorus concentrations represented long-term average values for completely mixed reservoir contents. In reality, localized fluctuations in phosphorus levels did occur, based upon the historical data set. For example, there existed a degree of longitudinal variation in concentration within certain reservoirs, primarily the larger impoundments. Thermal stratification markedly affected the vertical distribution of phosphorus in the water column under warm weather conditions. In addition, discrete zones or embayments within a reservoir sometimes exhibited higher phosphorus levels as a result of localized tributary, runoff, or point source loadings.

11. With a diverse set of simulation scenarios, alternative future conditions were examined, including hypothetical increases in urban runoff and point source loadings. Under the various scenarios, increases in annual average reservoir phosphorus levels above projected future baseline conditions were realized. Projected increases in annual average phosphorus concentration with the continuously-stirred tank reactor model were typically on the

order of several micrograms per liter, and only rarely was an increment of 0.010 mg/l or greater observed.

12. The interrelationship among reservoirs was an important consideration under certain conditions. The magnitude and longitudinal extent of the propagation of effects on downstream impoundments was dependent upon the total external mass loading to each reservoir and the outflow through the series of reservoirs. Downstream effects were substantial where the external mass loading was dominated by loadings associated with upstream releases. The effects of upstream loadings displayed greatest propagation downstream in the extremely throughflow-dominated reservoirs: Inks Lake, Lake Marble Falls, Lake Austin, and Town Lake. Reservoirs that were less throughflow-dominated were less likely to demonstrate substantial increases in concentration in response to a mass input to an upstream impoundment.

13. Investigation of urban runoff control measures was accomplished with an assumed 50 percent reduction in phosphorus loads from urban areas. Imposition of the reduction in urban loadings resulted in generally small decreases in the magnitude of projected annual average phosphorus concentrations within the series of reservoirs. Decreases in projected phosphorus levels were most substantial in the more heavily urbanized impoundments, Lake Austin and Town Lake. The observed overall effects were small due to the relative magnitude of the urban loading compared to other sources of phosphorus. Since the projected effects of urban loads without controls were generally small, the net benefits of the simulated control measures were not readily apparent. This analysis did not constitute a condemnation of urban runoff control measures per se, as the benefits of control of urban runoff transcend the implications on phosphorus concentrations alone.

14. The effects of hypothetical additional point source loadings were examined under the assumption that effluent phosphorus levels would be limited to 2 mg/l. Phosphorus loads at an assumed uncontrolled level of 10 mg/l were compared. The differences in phosphorus loading between controlled and uncontrolled effluents were substantial, and such differences could have a marked effect upon predicted reservoir phosphorus concentrations under future scenarios that examined intensive levels of development.

15. Simple trophic classification models were applied to the simulation results. Vollenweider's phosphorus loading versus mean depth model indicated a state of eutrophication for all of the reservoirs. With Vollenweider's phosphorus loading versus hydraulic loading model, Lake Buchanan, Lake LBJ, and Inks Lake were classified as eutrophic; Lake Travis and Lake Marble Falls were classified as mesotrophic; and Lake Austin and Town Lake were classified as oligotrophic. Dillon's loading parameter versus mean depth model indicated a mesotrophic state for Lake Buchanan, Inks

Lake, Lake LBJ, and Lake Marble Falls, and an oligotrophic state for Lake Travis, Lake Austin, and Town Lake. It was apparent that the simple classification models have significant limitations for application to the Highland Lakes. The trophic models are simplistic in nature, assume equilibrium conditions exist, and are based on the premise that trophic state can be determined from a few readily available parameters. The models were derived empirically, using trophic classifications for lakes outside of the southwestern United States, and in fact, reservoirs were generally absent from the data set. Phosphorus concentrations employed in the models for delineation of trophic categories may be inappropriate for the Highland Lakes. Factors which could influence critical phosphorus concentrations in the Highland Lakes and affect the propriety of the simple trophic models include background turbidity levels and the depth of the photic zone, temperature and the effects of thermal stratification, and reservoir operational characteristics.

16. The implications of projected increases in reservoir phosphorus levels cannot be precisely ascertained at the present time. The relationship between phosphorus concentration and algal growth, or chlorophyll-a as a surrogate indicator, has not been defined for the Highland Lakes. Further, the relationship between chlorophyll-a and transparency has not been defined. Algal growth may also be limited by other factors. Determination of these relationships was not within the scope of the present analysis.

17. The present investigation demonstrated the utility of the continuously-stirred tank reactor modeling approach for estimation of annual average reservoir phosphorus concentrations, with mass input represented as annual loads, and internal kinetics represented as a first-order decay. The continuously-stirred tank reactor model is a useful and appropriate tool for the long-term, time-dependent projection of average reservoir phosphorus concentrations and evaluation of the long-term effects of changes in external phosphorus loadings, including the implications of control measures on point and nonpoint sources. The modeling approach is based upon the assumption of completely mixed conditions within a reservoir, therefore, mass loadings are assumed to be immediately dispersed throughout the reservoir into which they are introduced. Consequently, the effects of mass loadings-- representative of a fraction of the original load--propagate instantaneously downstream. In actuality, longitudinal and vertical gradients in phosphorus concentration, which can have significant influence on biological dynamics, exist within the reservoirs, and the hydraulic retention time varies from several days to several years, depending upon the specific impoundment under examination and the extant flow regime. With the present formulation of input data as annual loads for use in simulation of annual average reservoir phosphorus concentrations, the modeling approach cannot be employed to examine the effects of localized or transient input loads.

7.3 RECOMMENDATIONS

The following recommendations were developed pursuant to the present analysis of reservoir phosphorus concentrations:

1. The existing historical data set for the Highland Lakes is limited at best. Additional data are needed to develop a comprehensive long-term data base for support of future water quality analyses, modeling activities, and management decisions. Most importantly, water quality data should be collected at frequent intervals in time. At a minimum, a routine data collection schedule should be implemented, with sampling at monthly, bimonthly, or quarterly intervals. The frequency of sampling in individual reservoirs could be influenced by the hydraulic retention time: for the purposes of estimating average characteristics, less frequent sampling is required for reservoirs with longer retention time. However, the sampling frequency should also recognize the importance of internal kinetics. The primary sampling stations should be situated in the lower pool of each reservoir, and additional spatial coverage is desirable. Stations should also be established to monitor releases from each reservoir and major tributary inflows. Samples should be collected both above and below the thermocline, when present. With respect to phosphorus, samples should be tested for both total phosphorus and dissolved orthophosphate (reactive) phosphorus, at a minimum. Necessary ancillary data would include temperature, dissolved oxygen, conductivity, pH, suspended solids, Secchi depth, chlorophyll-a, the nitrogen series, and a variety of additional constituent analyses.

   It would be prudent to single out one or two reservoirs for more intensive routine surveys, for example, Lake LBJ and Lake Travis would be attractive candidates, representing a large and an intermediate-size impoundment. For the intensive surveys, enhanced temporal and spatial coverage would be achieved. Periodically, diurnal sampling surveys would be conducted. In addition, a more comprehensive network of sampling stations would be established, sufficient to evaluate the existence of longitudinal concentration gradients and investigate localized areas of concern.

2. An effort should be made to establish the relationship between phosphorus concentration and chlorophyll-a and between chlorophyll-a and transparency, as measured by Secchi depth or photometer, for the Highland Lakes. Relationships between chlorophyll-a and other potentially limiting factors for algal growth (nitrogen, for example) should also be examined. Additional data will be required to establish these relationships. Initially, the investigation could focus upon only one or two reservoirs. After relationships are developed for these reservoirs, the results of a long-term data collection program covering all of the impoundments could be employed to

periodically revise the formulations. These investigations are necessary to ascertain the effects on water quality of projected increases or decreases in reservoir phosphorus concentrations.

3. Though the present analysis of long-term average phosphorus concentrations is useful, transitory and localized effects of mass loadings upon concentration should also be examined. To develop such an analysis, phosphorus levels should be simulated on a daily, or smaller time scale, basis. Input data on compatible temporal and spatial scales will be required for the analysis. With a more intensive simulation, the localized effects of mass loadings, such as point source inputs, can be examined. In addition, transitory loadings, such as stormwater runoff inputs, can be analyzed.

4. As additional data become available, the continuously-stirred tank reactor model developed in the present analysis should perhaps be revised to incorporate epilimnetic and hypolimnetic layers within each reservoir. Phosphorus cycling and release from the sediments may also be included in future analyses. Consideration should also be given to analysis of the series of reservoirs with a plug-flow model, in order to incorporate the effects of hydraulic retention time.

APPENDIX A

Evaluation of Historical Colorado River Flows

APPENDIX A

EVALUATION OF HISTORICAL COLORADO RIVER FLOWS

The streamflow gauge at San Saba went into operation in 1915. However, hydrologic modifications in the Colorado watershed have continually impacted the streamflow observations. The most recent hydrologic modification was the completion of Robert Lee Dam on E. V. Spence Reservoir in the upper watershed in 1968. Subsequent to this modification, only 14 years of data are available for computation of the average flowrate. An average discharge of 19.17 m^3/s (677 ft^3/s) was reported for the period 1969-1981 (water years), while an average of 37.95 m^3/s (1,340 ft^3/s) was reported for the period 1917-1968 (USGS, 1983a). Thus, a significant difference exists between the average flowrates for the two periods of record. The difference may indicate that the closure of Robert Lee Dam did indeed have a very substantial impact upon downstream streamflow rates at San Saba. Alternatively, a 14-year period of record may be insufficient for representation of average conditions. The latter possibility would seem to be indicated by the fact that the drainage area above the dam constitutes only 25 percent of the watershed area downstream at San Saba, while the difference in average discharge before and after construction of the dam is 49 percent.

The data base for existing conditions at the San Saba gauge was artificially extended back to previous years in order to lengthen the period of record. The technique employed involves correlation of annual discharge rates between stations in the watershed (Beard, 1983). The San Saba River is a major tributary of the Colorado River, with a confluence above gauging station no. 08147000 on the Colorado. The drainage area at the San Saba River gauging station no. 08146000 comprises approximately 15 percent of the Colorado River watershed at gauge 08147000. Annual discharge rates for the Colorado River at gauge no. 08147000 were plotted versus the corresponding rates for station no. 08146000, located on the San Saba River. (Both stations are located near San Saba, Texas.)

Discharge rates were plotted for the period 1969-1982 to establish the relationship between the two gauges under the existing scheme of hydrologic modifications. The relationship is displayed in Fig. A-1. Under the assumption that the relationship for the period 1969-1982 and the attendant hydrologic modifications is representative of the long-term correspondence between the two stations, historical discharge rates for the San Saba River were used to synthesize estimated discharge rates for the Colorado River at station no. 08147000. These estimated discharge rates are assumed to approximate the rates which would have occurred had the existing hydrologic modifications been present over the long-term period. Discharge rates were estimated for station no. 08147000 with this method for the period 1932-1968, as presented in Table A-1. Using the synthesized data base with the actual observations for the period 1969-1982, an average discharge rate of 18.5 m^3/s (652 ft^3/s) was calculated for station no. 08147000 for a 51-year period of record. For the present analysis, Colorado River loads to Lake Buchanan are referenced to the monitoring station at Bend, Texas. A factor of 1.007 was applied to the observed discharge rate at station no. 08147000 to correct for increased drainage area. With the drainage area correction, an average discharge rate for the Colorado River above Lake Buchanan of 18.6 m^3/s (657 ft^3/s) is obtained.

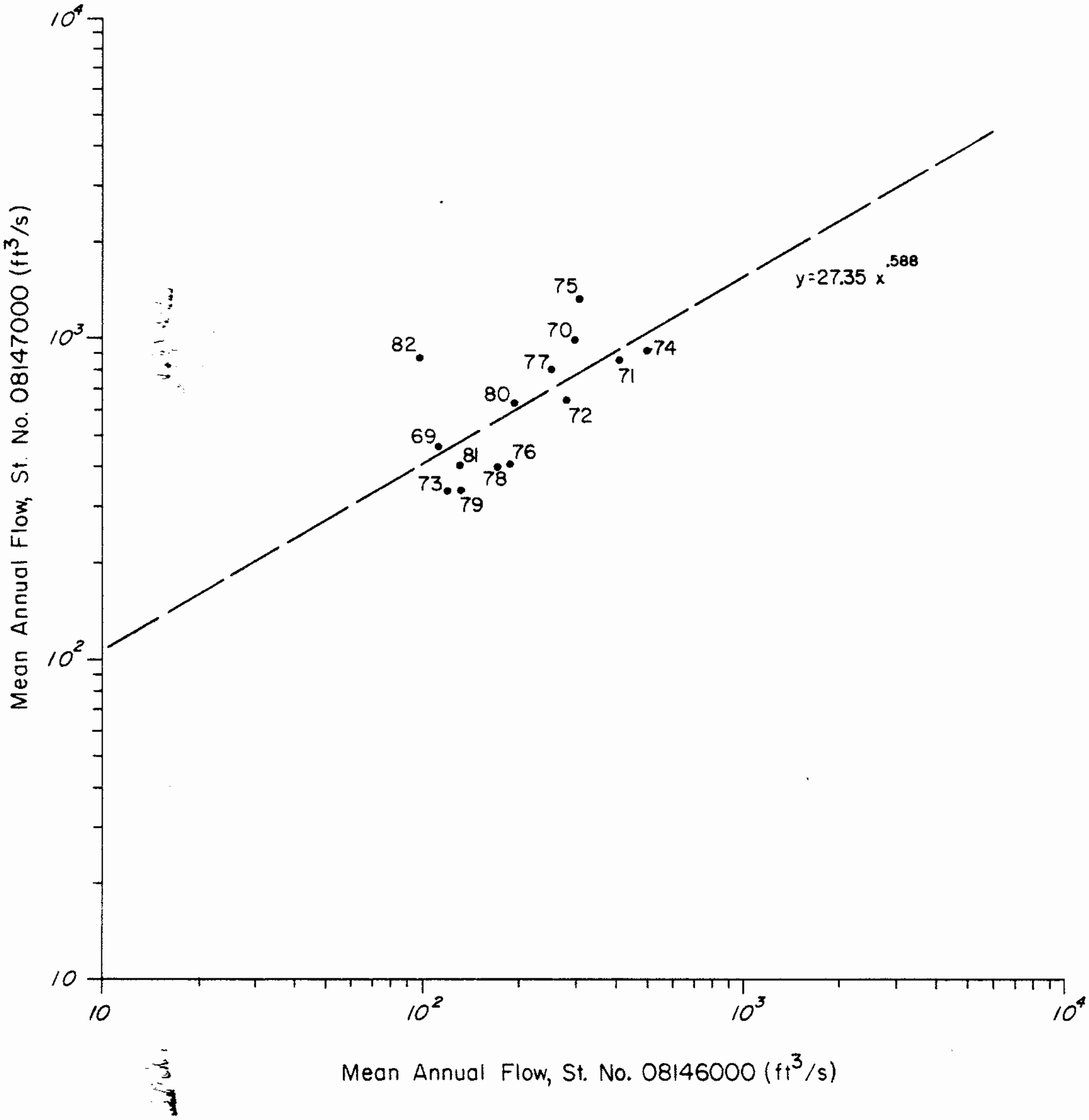

Fig. A-1

RELATIONSHIP BETWEEN ANNUAL MEAN DISCHARGE AT USGS ST. NO. 08147000 AND ST. NO. 08146000

TABLE A-1

SYNTHESIZED HYDROLOGIC DATA BASE, COLORADO RIVER NEAR SAN SABA, TEXAS

USGS ST. NO. 08147000

Water Year	Average Discharge (m^3/s)	Average Discharge (ft^3/s)	Water Year	Average Discharge (m^3/s)	Average Discharge (ft^3/s)	Water Year	Average Discharge (m^3/s)	Average Discharge (ft^3/s)
1982	24.72	873	1965	19.86	701	1948	10.90	385
1981	11.44	404	1964	16.41	580	1947	10.85	383
1980	18.04	637	1963	7.97	281	1946	13.49	476
1979	9.54	337	1962	13.06	461	1945	17.82	629
1978	11.19	395	1961	19.67	695	1944	17.87	631
1977	23.39	826	1960	19.39	685	1943	17.61	622
1976	11.24	397	1959	10.52	371	1942	21.86	772
1975	38.49	1,359	1958	27.09	956	1941	27.80	981
1974	26.11	922	1957	28.53	1,007	1940	15.87	560
1973	9.52	336	1956	15.70	555	1939	16.30	576
1972	18.44	651	1955	21.99	776	1938	52.91	1,868
1971	24.38	861	1954	9.57	338	1937	19.63	693
1970	28.80	1,017	1953	9.19	324	1936	29.18	1,030
1969*	13.03	460	1952	18.27	645	1935	31.38	1,108
1968	21.50	759	1951	8.79	311	1934	17.41	615
1967	9.51	336	1950	10.37	366	1933	12.93	457
1966	11.24	397	1949	14.57	514	1932	27.05	955

*Data for 1982-1969 are actual reported discharge rates.

APPENDIX B

Model HILO ADS

```
C       PROGRAM HILOADS


C       PROGRAM HILOADS
C
        DIMENSION CN(20,20),AREA(25,4),ASOIL(25),BSOIL(25),CSOIL(25),
       .DSOIL(25),CNLU(25,4),PREC(25,3),XSTOR(25,3),STOR(25,4),
       .QLU(25,4,3),SUMQ(25,4,3),CONC(25,4),SLOAD(25,4,3),TLOAD(25,4),
       .BLOAD(25),BSUMQ(25),TSUMQ(25,4)
C
C       INPUT NUMBER OF SUBBASINS,STORM EVENT CATEGORIES
C
        READ(5,1)NSB,NPREC
      1 FORMAT(2I5)
C
C       READ MATRIX OF CN(I,J),I=LAND USE,J=SOIL TYPE
C
        WRITE(6,110)
    110 FORMAT(/,10X,'MATRIX OF CN(I,J),I=LAND USE,J=SOIL TYPE',//,
       .5X,'LAND USE I',10X,'SOILS 1-4',/)
        DO 100 I=1,4
        READ(5,2)(CN(I,J),J=1,4)
      2 FORMAT(4F5.0)
        WRITE(6,9)I,(CN(I,J),J=1,4)
      9 FORMAT(5X,I5,10X,4F5.0)
    100 CONTINUE
C       INPUT LAND USE AREAS FOR RANGELAND,FOREST,CROPLAND,URBAN
C      .IN SQ KM
C
        WRITE(6,11)
     11 FORMAT(/,20X,'LAND USE (KM2)',//,3X,'SUBBASIN',3X,'RANGELAND',
       .3X,'FOREST',3X,'CROPLAND',2X,'URBAN',/)
        DO 20 IW=1,NSB
        READ(5,3)(AREA(IW,I),I=1,4)
      3 FORMAT(4F10.0)
        WRITE(6,12)IW,(AREA(IW,I),I=1,4)
     12 FORMAT(2X,I5,4X,4F10.0)
     20 CONTINUE
C       INPUT FRACTION OF EACH HYDROLOGIC SOIL TYPE IN SUBBASIN
C
        WRITE(6,101)
    101 FORMAT(/,10X,'FRACTION OF EACH HYDROLOGIC SOIL TYPE',//,
       .3X,'SUBBASIN',6X,'ASOIL',5X,'BSOIL',5X,'CSOIL',5X,'DSOIL',/)
        DO 21 IW=1,NSB
        READ(5,4)ASOIL(IW),BSOIL(IW),CSOIL(IW),DSOIL(IW)
      4 FORMAT(4F10.2)
        WRITE(6,102)IW,ASOIL(IW),BSOIL(IW),CSOIL(IW),DSOIL(IW)
    102 FORMAT(2X,I5,4X,4F10.2)
     21 CONTINUE
C
C       COMPUTE CN FOR EACH LAND USE CATEGORY
C
        WRITE(6,103)
    103 FORMAT(/,20X,'CURVE NUMBERS',//,3X,'SUBBASIN',3X,'RANGELAND'
       .,3X,'FOREST',3X,'CROPLAND',3X,'URBAN',/)
        DO 22 IW=1,NSB
        DO 23 I=1,4
        CNLU(IW,I)=ASOIL(IW)*CN(I,1)+BSOIL(IW)*CN(I,2)+
       .CSOIL(IW)*CN(I,3)+DSOIL(IW)*CN(I,4)
     23 CONTINUE
```

```
C     PROGRAM HILOADS


      WRITE(6,13)IW,(CNLU(IW,I),I=1,4)
   13 FORMAT(2X,I5,5X,4F10.2)
   22 CONTINUE
C
C     INPUT STORM DATA
C
      WRITE(6,104)
  104 FORMAT(/,20X,'STORM DATA',//,3X,'SUBBASIN',9X,'PRECIPITATION(IN)'
     .,12X,'NUMBER OF EVENTS PER YEAR',/)
      DO 40 IW=1,NSB
      READ(5,7)(PREC(IW,K),K=1,NPREC),(XSTOR(IW,K),K=1,NPREC)
    7 FORMAT(3F10.2,3F10.2)
      WRITE(6,105)IW,(PREC(IW,K),K=1,NPREC),(XSTOR(IW,K),K=1,NPREC)
  105 FORMAT(2X,I5,3X,3F10.2,3X,3F10.2)
   40 CONTINUE
C
C     COMPUTE STORM RUNOFF WITH SCS EQUATIONS
C
      WRITE(6,14)
   14 FORMAT(/,20X,'STORM EVENT RUNOFF(IN)',//,3X,'SUBBASIN',3X,
     .'PRECIP(IN)',2X,'RANGELAND',3X,'FOREST',3X,'CROPLAND',
     .3X,'URBAN',/)
      DO 24 IW=1,NSB
      DO 25 I=1,4
      STOR(IW,I)=1000./CNLU(IW,I)-10.
      DO 26 K=1,NPREC
      TOP=(PREC(IW,K)-.2*STOR(IW,I))**2.
      BOT=(PREC(IW,K)+.8*STOR(IW,I))
      QLU(IW,I,K)=TOP/BOT
   26 CONTINUE
   25 CONTINUE
      DO 106 K=1,NPREC
      WRITE(6,15)IW,PREC(IW,K),(QLU(IW,I,K),I=1,4)
   15 FORMAT(2X,I5,3X,F10.2,3X,4F10.2)
  106 CONTINUE
   24 CONTINUE
C
C     COMPUTE FLOWS FROM EACH SUBBASIN CATEGORY
C
      WRITE(6,17)
   17 FORMAT(/,10X,'FLOWS FROM LAND USE CATEGORIES(CU.M/YR)',//,
     .2X,'SUBBASIN',2X,'PRECIP(IN)',2X,'RANGELAND',4X,'FOREST',7X,
     .'CROPLAND',5X,'URBAN',5X,'TOT FLOW',/)
      DO 32 IW=1,NSB
      BSUMQ(IW)=0.0
      DO 31 I=1,4
      TSUMQ(IW,I)=0.0
      DO 30 K=1,NPREC
      SUMQ(IW,I,K)=QLU(IW,I,K)*XSTOR(IW,K)*AREA(IW,I)*2.54E4
      TSUMQ(IW,I)=TSUMQ(IW,I)+SUMQ(IW,I,K)
   30 CONTINUE
      BSUMQ(IW)=BSUMQ(IW)+TSUMQ(IW,I)
   31 CONTINUE
      DO 107 K=1,NPREC
      WRITE(6,16)IW,PREC(IW,K),(SUMQ(IW,I,K),I=1,4),BSUMQ(IW)
   16 FORMAT(2X,I5,3X,F10.2,4(3X,E10.3),3X,E12.5)
  107 CONTINUE
```

```
C       PROGRAM HILOADS


   32 CONTINUE
C
C       INPUT RUNOFF CONCENTRATION FOR EACH LAND USE CATEGORY
C
        WRITE(6,19)
   19 FORMAT(/,10X,'TOTAL P CONCENTRATION IN RUNOFF(MG/L)',//,
     .2X,'SUBBASIN',3X,'RANGELAND',3X,'FOREST',3X,'CROPLAND',3X,
     .'URBAN',/)
        DO 99 IW=1,NSB
        READ(5,8)(CONC(IW,I),I=1,4)
    8 FORMAT(4F10.0)
        WRITE(6,18)IW,(CONC(IW,I),I=1,4)
   18 FORMAT(2X,I5,3X,4F10.3)
   99 CONTINUE
C
C       COMPUTE MASS LOADS(G/YR)
C
        WRITE(6,61)
   61 FORMAT(/,20X,'PHOSPHORUS LOADS(G/YR)',//,2X,'SUBBASIN',2X,
     .'RANGELAND',5X,'FOREST',5X,'CROPLAND',5X,'URBAN',8X,'TOTAL',/)
        DO 42 IW=1,NSB
        BLOAD(IW)=0.0
        DO 41 I=1,4
        TLOAD(IW,I)=0.0
        DO 44 K=1,NPREC
        SLOAD(IW,I,K)=SUMQ(IW,I,K)*CONC(IW,I)
        TLOAD(IW,I)=TLOAD(IW,I)+SLOAD(IW,I,K)
   44 CONTINUE
        BLOAD(IW)=BLOAD(IW)+TLOAD(IW,I)
   41 CONTINUE
        WRITE(6,60)IW,(TLOAD(IW,I),I=1,4),BLOAD(IW)
   60 FORMAT(2X,I5,4(3X,E10.3),3X,E10.3)
   42 CONTINUE
        CALL EXIT
        END
```

APPENDIX C

Evaluation of SCS Method

APPENDIX C
EVALUATION OF SCS METHOD

The SCS curve number technique for runoff calculation has enjoyed widespread usage. The method was originally developed empirically and has been tested and proven reliable under a wide range of conditions. In conjunction with its use in this study, the performance of the SCS method was evaluated for local watershed conditions.

The SCS curve number technique was selected for the present study because of its explicit consideration of land use, which is a key variable in the projection of future developmental conditions. Of course, any predictive methodology is only as reliable as the input data available for use. Key variables for the SCS technique include soils, land use and precipitation data. The accuracy of the precipitation data directly affects the prediction of runoff. This is a critical shortcoming in many areas, due to an absence of rain gauges. Indeed, the watershed of the Highland Lakes features an acute absence of rain gauges. To calculate runoff for a specific volume of rainfall, the SCS technique should perform admirably, given the assumption that the rainfall is dispersed uniformly over the test watershed. However, the absence of a comprehensive network of rain gauges for estimation of rainfall volumes constitutes a significant shortcoming when the SCS technique (or any other technique, for that matter) is employed for the simulation of actual historical conditions. Without knowledge of actual rainfall volumes, accurate determination of runoff volumes cannot be achieved.

Tests were run on selected watersheds in the Highland Lakes drainage basin in order to assess the general performance of the SCS method against observed runoff characteristics. Selected test areas included the Shoal Creek watershed in Austin, Texas, representing urban conditions; the Bull Creek watershed near Austin, representing a combination of rural and low density urban conditions; and the Sandy Creek watershed near Llano, Texas, representing a large, undeveloped drainage basin. Data were compiled from USGS (1983b) reports for observed runoff to rainfall ratios for discrete storm events on the Bull Creek and Shoal Creek basins. For the Sandy Creek watershed, runoff volumes were estimated from observed hydrographs in order to develop runoff to rainfall ratios.

Soil and land use information were tabulated for the test watersheds and a composite SCS curve number was calculated for each basin. Curve numbers were corrected for local conditions in accordance with SCS guidelines. The SCS curve number technique was then employed to calculate runoff for reported storms in the test basins over the period of roughly 1976 to 1982.

Results derived from the SCS technique are compared to observed data in Table C-1. The calculated and reported runoff to rainfall ratios exhibit reasonable agreement for the Shoal and Bull Creek watershed, but poor correlation for the Sandy

Creek basin. The performance is not surprising, since much more extensive data bases were available for the Bull and Shoal Creek watersheds. In particular, these two watersheds featured a relatively comprehensive network of recording rain gauges. Conversely, little data were available for the Sandy Creek watershed. The nearest rain gauges are situated in the cities of Llano and Burnet, both a considerable distance from the test watershed. The Sandy Creek watershed is quite large, thus rainfall is less likely to be uniform throughout the watershed, increasing the importance of accurate rainfall data. Errors may also have been introduced into the analysis of runoff to rainfall ratios in Sandy Creek due to a lack of intuition on the response of the runoff hydrograph to the duration and intensity of rainfall.

In summary, the SCS method performs adequately where sufficiently accurate input data are available. The method should thus be suitable for application to the watershed of the Highland Lakes. Input data for estimation of runoff loads in the Highland Lakes watershed are limited. This lack of data is a handicap for any runoff-estimation technique.

TABLE C-1

COMPARISON OF REPORTED AND CALCULATED RUNOFF

Storm	Rainfall (in)	Reported[1] Runoff (in)	Reported[1] Runoff/ Rainfall Ratio	Calculated[2] Runoff (in)	Calculated[2] Runoff/ Rainfall Ratio
		- BULL CREEK -			
12-31-78	2.64	.13	.05	.38	.14
04-29-79	1.30	.08	.06	.01	.01
05-13-82	5.42	2.13	.39	2.02	.37
03-27-80	3.58	.33	.09	.84	.23
04-25-80	1.67	.12	.07	.07	.04
10-16-80	2.61	.21	.08	.37	.14
03-03-81	3.01	.55	.18	.55	.18
05-23-81	6.09	2.02	.33	2.51	.41
06-10-81	8.29	3.45	.42	4.25	.51
		- SHOAL CREEK -			
03-03-81	3.08	1.12	.36	1.51	.49
05-23-81	8.30	4.41	.53	6.26	.75
06-10-81	6.42	5.75	.90	4.48	.70
05-13-82	5.47	2.83	.52	3.60	.66
12-31-78	2.42	.69	.28	1.00	.41
05-21-79	3.46	1.99	.57	1.82	.53
07-19-79	3.48	1.22	.35	1.84	.53
03-27-80	3.26	.96	.29	1.66	.51
05-12-80	1.80	.70	.39	.56	.31
04-15-77	2.45	.48	.20	1.02	.42

TABLE C-1 (Concluded)

Storm	Rainfall (in)	Reported Runoff (in)	Reported Runoff/ Rainfall Ratio	Calculated Runoff (in)	Calculated Runoff/ Rainfall Ratio
		- SHOAL CREEK - (Concluded)			
04-16.77	1.06	.32	.30	.16	.15
04-19-77	1.78	.40	.23	.55	.31
05-02-78	1.82	.37	.21	.57	.31
05-11-78	1.42	.28	.17	.33	.23
04-18-76	3.98	1.04	.26	2.27	.57
05-25-76	3.02	.54	.18	1.46	.48
09-02-76	1.37	.12	.09	.31	.23
04-28-75	4.02	.93	.23	2.30	.57
		- SANDY CREEK -			
12-31-78	2.19	0.09	0.04	0.24	.11
02-05-79	1.54	0.26	0.17	0.06	.04
03-20-79	1.77	0.37	0.21	0.11	.06

[1] Reported by USGS.

[2] Calculated with SCS method.

APPENDIX D

Model HILAKES

```
C     PROGRAM HILAKES


C     PROGRAM HILAKES
C     SIMULATION OF PHOSPHORUS CONCENTRATION IN A SERIES OF RESERVOIRS
C     INPUT IS IN THE FORM OF ANNUAL PHOSPHORUS LOADS
C     VARIABLES ARE AS FOLLOWS:
C     NREACT=NUMBER OF RESERVOIRS
C     NSETS=NUMBER OF DATA SETS
C     H=TIME STEP,YEARS
C     N=NUMBER OF TIME STEPS FOR A PARTICULAR DATA SET
C       VOL(IRES)=RESERVOIR VOLUME,CUBIC METERS
C       QOUT(IRES)=RESERVOIR RELEASES,CUBIC METERS/YEAR
C       QWD(IRES)=RESERVOIR WITHDRAWALS,CUBIC METERS/YEAR
C       DK(IRES)=MASS KINETIC COEFFICIENT,1/YEAR
C       WLPS(IRES)=POINT SOURCE MASS LOAD,GRAMS/YEAR
C       WLAD(IRES)=MASS LOAD FROM ADJACENT WATERSHED,GRAMS/YEAR
C       WLUP(IRES)=MASS LOAD FROM UPSTREAM RESERVOIR,GRAMS/YEAR
C       WLTR(IRES)=MASS LOAD FROM TRIBUTARIES,GRAMS/YEAR
C       WLPR(IRES)=PRECIPITATION MASS LOAD,GRAMS/YEAR
C       CONC(IRES)=CONSTITUENT CONCENTRATION,MILLIGRAMS/LITER
C     KEY COUNTERS ARE AS FOLLOWS:
C       IRES=RESERVOIR NUMBER
C       JSTEP=TIME STEP,YEAR
C       JJ=CALCULATION CALENDAR YEAR
C     THE SYSTEM EQUATION IS SOLVED NUMERICALLY WITH A FOURTH-ORDER
C       RUNGE KUTTA TECHNIQUE
C
      DIMENSION CONC(10),VOL(10),DK(10),
     1AL(10),CONCUP(10),QOUT(10),WLUP(10),WLAD(10),WLTR(10),
     2WLPS(10),WLPR(10),QWD(10),TOTWL(10)
C     FUNCTION STATEMENT
      F(Y)=TERM1-Y*ALPHA
C
C     PRINT HEADING
C
      WRITE(6,21)
C
C     READ NUMBER OF REACTORS, NO. OF DATA SETS, STEP SIZE,
C
      READ(5,2)NREACT,NSETS,H,JJ
C
C     READ INITIAL VALUES FOR EACH REACTOR
C
      READ(5,3)(CONC(IRES),IRES=1,NREACT)
C
C     LOOP THROUGH SOLUTIONS FOR ALL DATA SETS
C
      DO 20 NDATS=1,NSETS
C
C     LOOP THROUGH INPUT DATA SETS FOR ALL REACTORS
C
      READ(5,4)NHEAD
      DO 8 IRES=1,NREACT
      READ(5,11)VOL(IRES),QOUT(IRES),QWD(IRES),DK(IRES)
      READ(5,109)WLPS(IRES),WLAD(IRES),WLTR(IRES),WLPR(IRES)
    8 CONTINUE
C
C     READ NUMBER OF TIME STEPS FOR THIS DATA SET
C
```

```
C      PROGRAM HILAKES


       READ(5,4)N
C
C      PRINT HEADINGS
C
       WRITE(6,122)JJ,H,N
       WRITE(6,123)
       WRITE(6,124)(VOL(I),I=1,NREACT)
       WRITE(6,127)(QOUT(I),I=1,NREACT)
       WRITE(6,128)(DK(I),I=1,NREACT)
       WRITE(6,131)(WLTR(I),I=1,NREACT)
       WRITE(6,132)(WLAD(I),I=1,NREACT)
       WRITE(6,133)(WLPS(I),I=1,NREACT)
       WRITE(6,134)(WLPR(I),I=1,NREACT)
       WRITE(6,112)
  112  FORMAT(//,13X,'TIME',5X,'CONC(1)',5X,'CONC(2)',5X,'CONC(3)',5X,
      1'CONC(4)',5X,'CONC(5)',5X,'CONC(6)',5X,'CONC(7)',/)
       IF(NDATS.GT.1)GO TO 34
       WRITE(6,101)JSTEP,(CONC(I),I=1,NREACT)
   34  CONTINUE
       JSTEP=0
       JPRINT=100
C      LOOP THROUGH SOLUTIONS FOR N STEPS OF SIZE H FOR A DATA SET
C
       DO 10 NSTEP=1,N
       JSTEP=JSTEP+1
C
C      LOOP THROUGH ALL REACTORS FOR A TIME STEP
C
       DO 9 IRES=1,NREACT
       WLUP(IRES)=0.0
       IF(IRES.EQ.1)GO TO 26
       WLUP(IRES)=CONCUP(IRES-1)*QOUT(IRES-1)
   26  CONTINUE
       ALPHA=(QOUT(IRES))/VOL(IRES)+DK(IRES)
       TOTWL(IRES)=(WLUP(IRES)+WLAD(IRES)+WLPS(IRES)+WLPR(IRES)+
      1WLTR(IRES))
       TERM1=TOTWL(IRES)/VOL(IRES)
       Y=CONC(IRES)
       DEL1=H*F(Y)
       DEL2=H*F(Y+.5*DEL1)
       DEL3=H*F(Y+.5*DEL2)
       DEL4=H*F(Y+DEL3)
       CONCUP(IRES)=CONC(IRES)
       CONC(IRES)=CONC(IRES)+.16667*(DEL1+2.0*DEL2+2.0*DEL3+DEL4)
    9  CONTINUE
       IF(JSTEP.NE.JPRINT)GO TO 157
       WRITE(6,102)JSTEP,(CONC(IRES),IRES=1,NREACT)
       JPRINT=JPRINT+100
  157  CONTINUE
   10  CONTINUE
       WRITE(6,129)(WLUP(IRES),IRES=1,NREACT)
  129  FORMAT(/,1X,'UPSTR LOAD(G/YR)',1X,7(2X,E10.3),//)
       JJ=JJ+1
   20  CONTINUE
   40  CONTINUE
    3  FORMAT(7F10.5)
   11  FORMAT(5X,3E10.6,10X,E10.6)
```

```
C       PROGRAM HILAKES

  109 FORMAT(5X,4E10.6)
  101 FORMAT(14X,I3,1X,7(2X,E10.3))
  102 FORMAT(13X,I4,1X,7(2X,E10.3))
    2 FORMAT(2I10,1F10.0,I10)
    4 FORMAT(5X,I10)
   21 FORMAT(20X,'SIMULATION OF RESERVOIR PHOSPHORUS LEVEL',/)
  122 FORMAT(/,30X,'INPUT CONDITIONS ',I4,/,1X,'TIME STEP(YR)',
     1F10.5,/,1X,'NO. OF STEPS',I9)
  123 FORMAT(1X,'REACTOR',17X,'1',10X,'2',12X,'3',10X,'4',11X,'5',
     .10X,'6',12X,'7')
  124 FORMAT(1X,'VOL(M3)',10X,7(2X,F10.3))
  127 FORMAT(1X,'OUTFLOW(M3/YR)',3X,7(2X,E10.3))
  128 FORMAT(1X,'RXN RATE(1/YR)',3X,7(2X,E10.3))
  131 FORMAT(1X,'TRIB LOAD(G/YR)',2X,7(2X,E10.3))
  132 FORMAT(1X,'BASIN LOAD(G/YR)',1X,7(2X,E10.3))
  133 FORMAT(1X,'PT SRC LOAD(G/YR)',7(2X,E10.3))
  134 FORMAT(1X,'PRECIP LOAD(G/YR)',7(2X,E10.3))
      CALL EXIT
      END
```

APPENDIX E

Model HYDROB

```
C      PROGRAM HYDROB/INFLOW,OUTFLOW,RETENTION TIME


C      PROGRAM HYDROB/INFLOW,OUTFLOW,RETENTION TIME
C
       DIMENSION QOUTR(7,205),QTURB(7,205),QSPILL(7,205),
      .QOUTT(7,205),QEVAP(7,205),DELSTO(7,205),STOR(7,205),
      .QRUNOF(7,205),QIN(7,205),VOLAVG(7,205),RTIME(7,205),
      .QMISC(7,205),QVAR(7,205),QOVAR(7,205),CUMVOL(7,205),
      .RTIME3(7,205),ANNVOL(7,205),ANNRT(7,205),
      .ANNCS(7,205),ANNRO(7,205),ANNINF(7,205)
C
C *** SET CONSTANTS
C
       CF = 0.7
       NUMDAT = 205
C
       DO 20 I=1,6
       DO 10 J=1,NUMDAT
       READ(5,9)STOR(I,J),QTURB(I,J),QSPILL(I,J),QEVAP(I,J),QMISC(I,J),
      .QVAR(I,J)
     9 FORMAT(3X,6F10.0)
C
C      CORRECT PAN EVAP DATA
C
       QEVAP(I,J)=QEVAP(I,J)*CF
    10 CONTINUE
    20 CONTINUE
C
       DO 40 I=1,6
       L=0
       DO 30 J=2,NUMDAT
       L=L+1
C
       QOUTR(I,J)=QTURB(I,J)+QSPILL(I,J)
       QOUTT(I,J)=QOUTR(I,J)+QEVAP(I,J)+QVAR(I,J)
       QOVAR(I,J)=QOUTR(I,J)+QVAR(I,J)
       JM1=J-1
       JM2=J-2
       JM3=J-3
       JM4=J-4
       JM5=J-5
       JM6=J-6
       JM7=J-7
       JM8=J-8
       JM9=J-9
       JM10=J-10
       JM11=J-11
       JM12=J-12
       IM1=I-1
       DELSTO(I,J)=STOR(I,J)-STOR(I,JM1)
C
C      CALCULATE RUNOFF INFLOW WITH HYDRO BUDGET FOR EACH MONTH
       IF(I.EQ.1)QRUNOF(I,J)=DELSTO(I,J)+QOUTT(I,J)-QMISC(I,J)
       IF(I.GT.1)QRUNOF(I,J)=DELSTO(I,J)+QOUTT(I,J)-QOUTR(IM1,J)-
      .QMISC(I,J)
C
C      CALCULATE TOTAL INFLOW WITH HYDRO BUDGET PER MONTH
       IF(I.EQ.1)QIN(I,J)=QRUNOF(I,J)+QMISC(I,J)
       IF(I.GT.1)QIN(I,J)=QRUNOF(I,J)+QMISC(I,J)+QOUTR(IM1,J)
```

```
C       PROGRAM HYDROB/INFLOW,OUTFLOW,RETENTION TIME


C
C       CALCULATE AVG VOLUME FOR EACH MONTH
        VOLAVG(I,J)=(STOR(I,J)+STOR(I,JM1))/2.0
C
C       IF OUTFLOW IS .LE. ZERO, SET = 1.0
        IF(QOVAR(I,J).LE.0)QOVAR(I,J)=1.0
C
C       CALCULATE RETENTION TIME FOR MONTH
        RTIME(I,J)=30.*VOLAVG(I,J)/(QOVAR(I,J))
C       THREE MONTH AVERAGE RET TIME
        IF(J.LT.4)CUMVOL(I,J)=0.0
        IF(J.GE.4)CUMVOL(I,J)=(VOLAVG(I,J)+VOLAVG(I,JM1)+VOLAVG(I,JM2))/3.0
        IF(J.LT.4)RTIME3(I,J)=0.0
        IF(J.GE.4)RTIME3(I,J)=91.0*CUMVOL(I,J)/(QOVAR(I,J)+QOVAR(I,JM1)+
       .QOVAR(I,JM2))
C       ANNUAL AVERAGE RET TIME
C
C       CALCULATE AVG ANNUAL VOLUME
        IF(L.EQ.12)ANNVOL(I,J)=(VOLAVG(I,J)+VOLAVG(I,JM1)+VOLAVG(I,JM2)+
       .VOLAVG(I,JM3)+VOLAVG(I,JM4)+VOLAVG(I,JM5)+VOLAVG(I,JM6)+
       .VOLAVG(I,JM7)+VOLAVG(I,JM8)+VOLAVG(I,JM9)+VOLAVG(I,JM10)+
       .VOLAVG(I,JM11))/12.
        IF(L.EQ.12)ANNRT(I,J)=365.0*ANNVOL(I,J)/(QOVAR(I,J)+QOVAR(I,JM1)+
       .QOVAR(I,JM2)+QOVAR(I,JM3)+QOVAR(I,JM4)+QOVAR(I,JM5)+QOVAR(I,JM6)+
       .QOVAR(I,JM7)+QOVAR(I,JM8)+QOVAR(I,JM9)+QOVAR(I,JM10)+
       .QOVAR(I,JM11))
C
C       CALCULATE ANNUAL RUNOFF INFLOW
C
        IF(L.EQ.12)ANNRO(I,J)=(QRUNOF(I,J)+QRUNOF(I,JM1)+QRUNOF(I,JM2)+
       .QRUNOF(I,JM3)+QRUNOF(I,JM4)+QRUNOF(I,JM5)+QRUNOF(I,JM6)+
       .QRUNOF(I,JM7)+QRUNOF(I,JM8)+QRUNOF(I,JM9)+QRUNOF(I,JM10)+
       .QRUNOF(I,JM11))
C
C       CALCULATE ANNUAL TOTAL INFLOW
C
        IF(L.EQ.12)ANNINF(I,J)=(QIN(I,J)+QIN(I,JM1)+QIN(I,JM2)+
       .QIN(I,JM3)+QIN(I,JM4)+QIN(I,JM5)+QIN(I,JM6)+QIN(I,JM7)+
       .QIN(I,JM8)+QIN(I,JM9)+QIN(I,JM10)+QIN(I,JM11))
C
C       CALCULATE AVG ANNUAL CHANGE IN STORAGE
C
        IF(L.EQ.12)ANNCS(I,J)=(DELSTO(I,J)+DELSTO(I,JM1)+DELSTO(I,JM2)+
       .DELSTO(I,JM3)+DELSTO(I,JM4)+DELSTO(I,JM5)+DELSTO(I,JM6)+
       .DELSTO(I,JM7)+DELSTO(I,JM8)+DELSTO(I,JM9)+DELSTO(I,JM10)+
       .DELSTO(I,JM11))
        IF(L.LT.12)GO TO 29
        L=0
   29   CONTINUE
   30   CONTINUE
   40   CONTINUE
C
        WRITE(6,51)
   51   FORMAT(1X,'RES',2X,'MO',3X,'DEL STOR',3X,'QIN RUN',3X,
       .'QIN TOT',2X,'QOUT TOT',2X,'RET TIME',2X,'3-MONTH',
       .2X,'ANNUAL RTIME',2X,'DEL STO',2X,'AVG VOL',
       .2X,'RUNOFF',2X,'TOT INFLO')
```

```
C       PROGRAM HYDROB/INFLOW,OUTFLOW,RETENTION TIME


C
        DO 90 I=1,6
        DO 80 J=1,NUMDAT
        WRITE(6,79)I,J,DELSTO(I,J),QRUNOF(I,J),QIN(I,J),QOUTT(I,J),
       .RTIME(I,J),RTIME3(I,J),ANNRT(I,J)
       .,ANNCS(I,J),ANNVOL(I,J),ANNRO(I,J),ANNINF(I,J)
   79   FORMAT(2X,I2,1X,I3,2X,7F10.0,4F10.0)
   80   CONTINUE
   90   CONTINUE
C
C       CONVERT TO METRIC (CU.M/YR)
C
        WRITE(6,103)
  103   FORMAT(1X,'RES',2X,'MO',3X,'DEL STO',3X,'AVG VOL',4X,
       .'RUNOFF',4X,'TOT INFLO')
        DO 101 I=1,6
        K=0
        DO 102 J=2,NUMDAT
        K=K+1
        ANNRO(I,J)=ANNRO(I,J)*1233.49
        ANNINF(I,J)=ANNINF(I,J)*1233.49
        ANNCS(I,J)=ANNCS(I,J)*1233.49
        ANNVOL(I,J)=ANNVOL(I,J)*1233.49
        IF(K.LT.12)GO TO 110
        K=0
        WRITE(6,104)I,J,ANNCS(I,J),ANNVOL(I,J),ANNRO(I,J),ANNINF(I,J)
  104   FORMAT(2X,I2,1X,I3,2X,4E12.5)
  110   CONTINUE
  102   CONTINUE
  101   CONTINUE
        CALL EXIT
        END
```

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