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THE CHEMICAL LIMNOLOGY AND LIMNETIC PRIMARY PRODUCTION OF THE TONGUE RIVER RESERVOIR, MONTANA

by

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VITA

Stephen Charles Whalen was born on 28 November 1952 in Millbrook, New York to Mr. and Mrs. Charles Whalen. He attended St. John's Preparatory High School in Danvers, Massachusetts and was graduated in June 1970. In September 1970 he enrolled at the University of Maine at Orono and received a Bachelor of Science Degree in Wildlife Management in June 1974. In September 1975 he began work toward a Master of Science Degree in Botany at Montana State University. On 10 June 1978 he married Kathi May Dick, the daughter of Mr. and Mrs. John Dick of Springfield, Virginia.

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ABSTRACT

The Tongue River Reservoir is a shallow, warmwater impoundment in southeastern Montana. Water stored in the reservoir is used primarily for irrigation, but recreational use of the reservoir is becoming increasingly popular. The Tongue River provides the only significant source of inflowing and outflowing surface water to and from the reservoir. The water chemistry and limnetic primary production of the reservoir system were studied from June 1975 through November 1976 to assess the early operational impact of surface coal mining activity on selected physical, chemical and biological parameters of the system. Data collected will also provide information against which future studies can be compared to detect any long-term changes resulting from continued and expanded mining activity.

The Tongue River Reservoir acted as a sediment trap for the inflowing Tongue River. A thermal gradient and a density current were present in the reservoir from late May through most of June in 1976. However the reservoir was typically polymictic due to the deepwater penstock of the outflow structure. Although no thermal or chemical stratification was generally evident, a mild oxygen deficit developed in the bottom water late in the summers of 1975 and 1976. The Tongue River and Tongue River Reservoir waters were a calcium-magnesium bicarbonate-sulfate type. The annual surface nutrient loading rates of 4.1 g total-P$\cdot$m^{-2} and 22.0 g total-N$\cdot$m^{-2} were indicative of a hypereutrophic system. However, the water withdrawal characteristics, temporal nutrient loading pattern and flushing rate of 7.78$\cdot$yr^{-1} held the average phytoplankton standing crop to 8.91 cm$^{3}\cdot$m^{-2} and the average photosynthetic rate to 0.77 g C$\cdot$m$^{-2}\cdot$day^{-1}. The typical algal standing crop was dominated by the Bacillariophyceae.

The dominant cation in the West Decker Mine discharge water was sodium while the most abundant anions were bicarbonate and sulfate. From June 1975 through November 1976 the mean percent contribution of the West Decker Mine discharge water to the Tongue River volume of flow was 0.09%; no change in river water quality was noted. The effluent discharged from three mines operating simultaneously should not significantly alter the water quality of the Tongue River or Tongue River Reservoir with respect to the parameters measured.

INTRODUCTION

The surface mining of coal has become an increasingly important industry in the Northern Great Plains and adjacent western states in recent years. Because the mining and combustion of fossil fuels will be essential to meet the nation's projected energy needs, future exploitation of the largely untapped coal reserves underlying this region is certain.

In the past, considerable effort has been directed toward the documentation of the ecological impact of strip mining operations on the aquatic and terrestrial resources of the eastern United States. However, the effects of surface coal mining on the semi-arid environment of the western United States are not well defined.

The Tongue River Reservoir is located near Decker, Montana, just north of the Montana-Wyoming border (Figure 1). In the summer of 1972 an open pit coal mining operation was initiated on the southwest side of the reservoir (Figure 2) by the Decker Coal Company, a subsidiary of Peter Kiewit Sons' Company. In the summer of 1977 a second mine was started on the southeast shore of the reservoir, and a future northward extension of the West Decker site is planned (Figure 2). Present mining operations at the West Decker Mine are discharging wastewater into the Tongue River immediately above the Tongue River Reservoir. When in full operation, all three mine sites will discharge effluent, either directly or indirectly, into the Tongue River

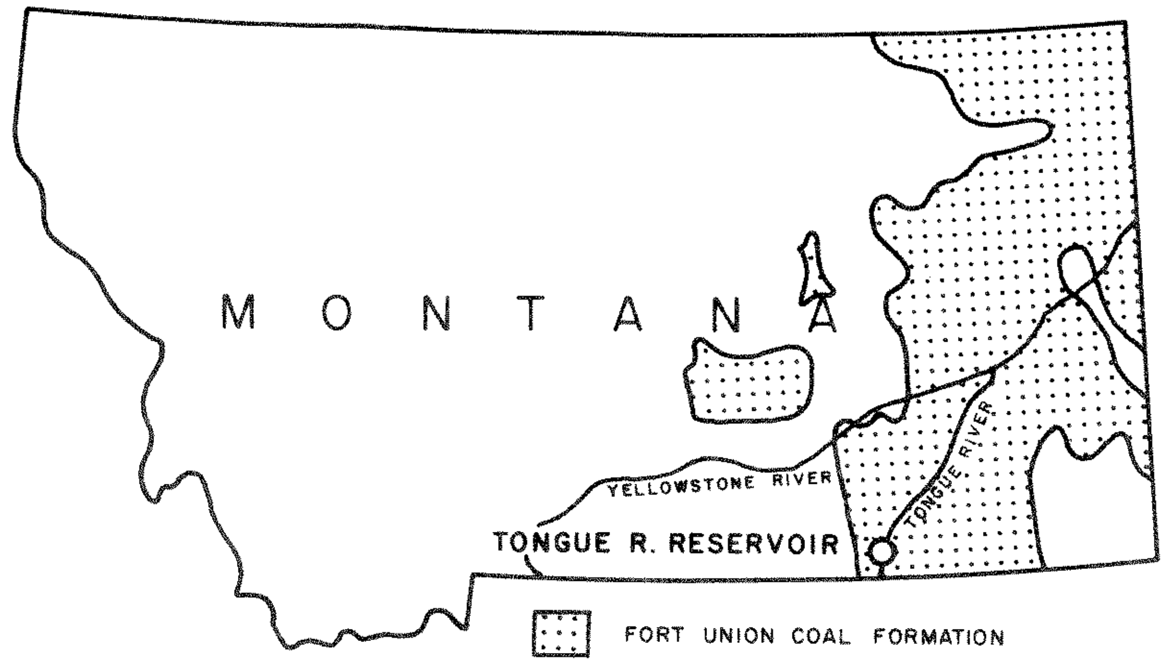

Figure 1. Location map.

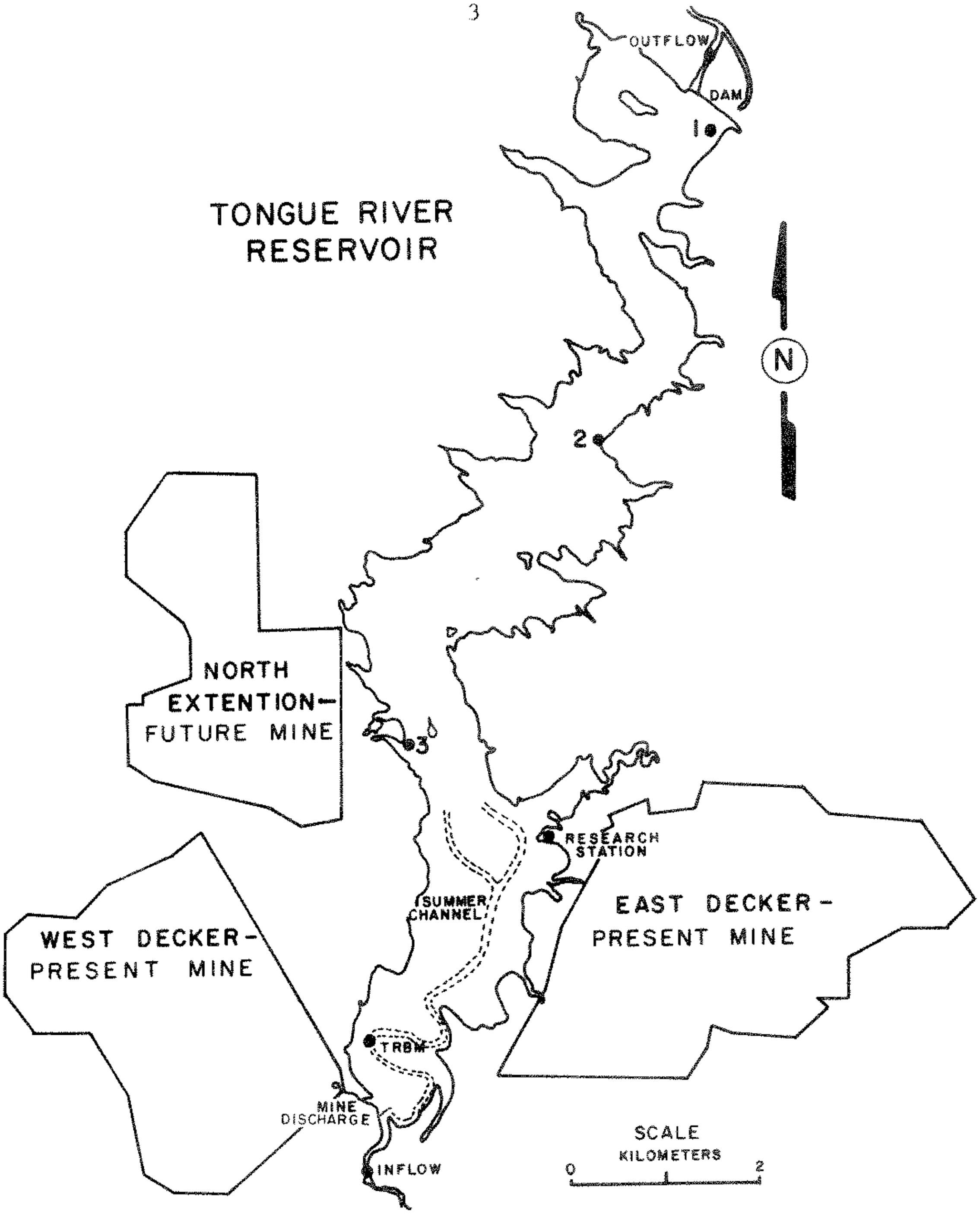

Figure 2. Map of the Tongue River Reservoir detailing the study area.

Reservoir.

In April 1975 the Montana Cooperative Fishery Research Unit initiated a program that included several detailed studies encompassing numerous aspects of the aquatic resources of the Tongue River and Tongue River Reservoir. The objective of this program is to assess the early operational impact of mining activity on selected physical, chemical and biological parameters of the river-reservoir system. Biological and chemical data are being secured to provide information against which future studies may be compared to detect any long-term changes resulting from continued and expanded mining activity. As part of the Tongue River Reservoir research program, the present study was conducted from June 1975 through November 1976 with the following specific goals:

(1) To relate the chemical, physical and hydrologic characteristics of the Tongue River-Tongue River Reservoir system and to define their effects on the limnological status of the reservoir. This goal includes; (a) characterization of the common anions and cations of the river and reservoir, (b) determination of the nutrient (nitrogen and phosphorus) loads of the Tongue River flowing in and out of the Tongue River Reservoir, (c) delimitation of the storage of these nutrients within the reservoir, (d) determination of factors affecting the fate of various forms of nutrients within the reservoir.

(2) To identify phytoplankton species present in the reservoir, to follow seasonal changes in the phytoplankton standing crop, and to examine factors affecting the size, species composition and primary productivity of the standing crop.

(3) To make a general assessment of the trophic status of the reservoir.

(4) To characterize the chemical nature of the West Decker Mine effluent water discharged into the Tongue River immediately above the reservoir, and to determine the immediate impact of this effluent on the chemical limnology of the river and reservoir.

(5) To predict possible long-term effects of continued and expanded mining on the ecology of the Tongue River Reservoir.

DESCRIPTION OF THE STUDY AREA

The Tongue River Reservoir is formed from impoundment of the Tongue River by the Tongue River Dam which is located at lat. 45°08'29", long. 106°45'15", in NE 1/4 Sec. 13, T. 8 S., R. 40 E. in Big Horn County, Montana, approximately 32 km (20 mi) north of Sheridan, Wyoming. The Tongue River originates in the Big Horn Mountains of Wyoming and flows northeasterly for 105 river kilometers (65 mi) until reaching the Tongue River Reservoir. The drainage area above the reservoir encompasses 4584 km^2 or 1770 mi^2 (U.S.G.S., 1977). The principal soils of the drainage area are sandy loam to clay loam, underlain by a bedrock of clay, shale, sandstone with coal, and limestone. The river continues for 270 kilometers (168 mi) beyond the reservoir to its confluence with the Yellowstone River at Miles City, Montana.

The Tongue River Reservoir was formed by an earthfill dam completed in May, 1939. The dam is owned by the Montana Department of Natural Resources and Conservation, but is operated by the Tongue River Water Users' Association. The Tongue River Dam was initially constructed for flood control and irrigation purposes, but water impounded by the structure provides recreational benefits as well. The reservoir had an estimated initial storage capacity of 894×10^5

m^3 or 72,450 acre-ft (Dendy and Champion, 1973). Based on calculated sedimentation rates, the 1975 usable capacity was estimated by U.S.G.S. and the Montana Department of State Lands (1977) to be 740 X 10^5 m^3 (60,000 acre-ft) between elevation 1028.5 m (3,374.4 ft) at the bottom of the outlet and elevation 1043.7 m (3,424.4 ft) at the spillway crest. Dead storage is 17.3 X 10^5 m^3 (1400 acre-ft) below elevation 1028.5 m (Mt. Dept. Nat. Res. and Cons., unpublished data).

The maximum reservoir depth is 18 m (59 ft) at storage capacity. At the maximum pool elevation the reservoir has an average depth of 5.9 m (19.5 ft), a maximum length of 12.5 km (7.8 mi) and a maximum breadth of 1.4 km (0.9 mi). Morphometric data for the Tongue River Reservoir is found in Table 1.

The Tongue River is the only significant source of inflowing and outflowing water to the Tongue River Reservoir, although several ephemeral streams make a minor contribution during spring runoff. The flow of the Tongue River above the reservoir is regulated by several small reservoirs in Wyoming with a combined storage capacity of 185 X 10^5 m^3 (15,000 acre-ft). There are diversions for irrigation of about 260 km^2 (64,300 acres) between the headwaters in the Big Horn Mountains of Wyoming and the Tongue River Reservoir (U.S.G.S., 1977). The rate of release of water held by the reservoir is regulated at the outflow structure in response to spring runoff and

Table 1. Morphometric data for the Tongue River Reservoir at maximum pool elevation (1043.7 m).[1]

Maximum depth [2]	18.0 m (59.1 ft)
Mean depth	5.9 m (19.5 ft)
Depth of outlet [3]	15.2 m (49.9 ft)
Maximum length [2]	12.5 km (7.8 mi)
Maximum breadth [2]	1.4 km (0.9 mi)
Surface area [3]	1277 ha (3156 acres)
Usable capacity [4]	740×10^5 m^3 (6.0×10^4 acre-ft)
Volume [5]	757×10^5 m^3 (6.14×10^4 acre-ft)
Length of shoreline [3]	60 km (37.3 mi)
Shoreline development [3]	4.74

[1] Definition and discussion of these parameters can be found in Ried and Wood (1976) or Welch (1948).

[2] Garrison, *et al.* (1975).

[3] Penkal (1976).

[4] U.S.G.S. and Mt. Dept. of State Lands (1977).

[5] Usable capacity and dead storage volume.

summer irrigation needs. At times when exceptionally large volumes of spring runoff enter the reservoir, excess water is discharged from the spillway at the crest of the dam. Annually, the rate of runoff from the reservoir's watershed (and hence the rate of flow of the Tongue River to the reservoir) peaks in May and June. The lowest rates of flow of the Tongue River above the reservoir are typically encountered in the winter months. River gaging stations are maintained by the United States Geological Survey at about 4 km upstream from the reservoir (Tongue River at state line, near Decker) and at about 2 km downstream from the spillway structure (Tongue River at Tongue River Dam, near Decker) to monitor inflowing and outflowing water discharges.

A total of seven permanent sampling stations were established, three of which were located within the Tongue River Reservoir (Figure 2). The reservoir stations were referred to as Station 1 (located at the north end of the reservoir, 0.4 km from the dam), Station 2 (mid-reservoir station, 4.3 km south of the dam) and Station 3. Station 3 was mobile to accomodate fluctuations of reservoir stage in the late fall. For the month of October it was necessary to move Station 3 approximately one kilometer north to facilitate sampling.

An additional sampling site was located at the east end of a culvert which passes beneath Montana Highway 314. This sampling station was named "Mine Discharge". Water collected at the Mine Discharge station was a combination of water pumped from disturbed ground water aquifers at the West Decker Mine and leachate through disturbed overburden at that mine site. The Mine Discharge effluent was collected in a settling pond and summarily discharged into the Tongue River via the Tongue River flood plain. Two permanent sampling stations were situated on the Tongue River above the Tongue River Reservoir. One station (Inflow) was located at the county bridge on the county road running northeast off Montana Highway 314. This site was approximately two kilometers south of the Tongue River Reservoir and was situated above the Mine Discharge station. The other sampling site was established on the inflowing Tongue River below the Mine Discharge station and was designated "TRBM" (Tongue River Below Mine). The single sampling location on the Tongue River below the Tongue River Reservoir (Outflow) was located about 0.5 km north of the spillway. All four non-reservoir sampling sites described above can be located in Figure 2.

METHODS AND MATERIALS

Inflow, Outflow and Mine Discharge Waters

Water samples were collected twice monthly from all previously described non-reservoir stations during the 17 June - 14 November 1975 and the 16 April - 21 November 1976 periods. In addition, samples were taken at monthly intervals from December, 1975 through March, 1976.

All samples were obtained by either lowering a rinsed 8-liter polyethylene bucket into the water or by directly filling sample containers with the water to be analyzed. Water was extracted from the main current when collecting river water samples regardless of the method used in procurring the sample. Four glass stoppered Pyrex bottles (1 each, 500-ml, 300-ml, 250-ml and 125-ml) and one 100-ml screw cap polyethylene bottle were filled with water from each station. All sample bottles were rinsed with deionized, distilled water in the laboratory, and with river water prior to sample collection.

Temperature and Dissolved Oxygen

Temperature and dissolved oxygen determinations were made immediately following sample collection on water in the 300-ml BOD bottle. The temperature of the water sample was taken with an E.H. Sargent & Co. -10° to $100^{\circ}C$ mercury thermometer. Alternatively, a Yellow Springs Instrument Co., Inc. YSI Model 54 Dissolved Oxygen Meter equipped with a thermistor probe was used.

The dissolved oxygen concentration of the water sample was analyzed with the previously mentioned YSI Model 54 Dissolved Oxygen Meter. Prior to each sampling session the instrument was calibrated against a dissolved oxygen determination by the Alsterberg Azide Modification of the Winkler Method (APHA, 1971). The dissolved oxygen concentration is reported as mg $O_2 \cdot L^{-1}$.

Specific Conductance, pH and Alkalinity

The specific conductance of the water sample was measured at a frequency of 60 cps with a Yellow Springs Instrument Co., Inc. Model 31 AC conductivity bridge equipped with a Yellow Springs Instrument Co. Model 3401 dipping cell. Samples of high electrical conductivity were measured at a frequency of 1 KC. Specific conductance is reported as $\mu mhos \cdot cm^{-1}$ @ 25°C.

The hydrogen ion concentration was determined with a Beckman Model 76 Expanded Scale pH Meter equipped with a Corning Triple Purpose pH Electrode, a Corning Calomel Reference Electrode and a Beckman Model 76 and 96 Thermocompensator.

The alkalinity of each sample was determined potentiometrically (APHA, 1971) using 0.1 N standard H_2SO_4 as a titrant. A pH of 8.3 was chosen as the carbonate endpoint and pH 4.5 was selected as the bicarbonate endpoint. Total alkalinity is expressed as $mg \cdot L^{-1}$ $CaCO_3$. Alkalinity and pH measurements were taken within a few hours of sample collection.

Turbidity

The turbidity of each sample was determined using a Hach Laboratory Turbidimeter (Model 2100) and is reported as Standard Jackson Turbidity Units (JTU). Unfiltered water from the 250-ml glass stoppered bottle was used for the analyses of turbidity, specific conductance, pH and alkalinity.

Additional Water Chemistry

Sample preparation. At the laboratory most of the water in the 500-ml glass bottles was filtered through Millipore^R membrane filters with a pore size of 0.45 microns. This water was used for the following analyses: soluble inorganic phosphorus, nitrate, nitrite, ammonia, silica, calcium, magnesium, sodium, potassium, sulfate, fluoride, chloride and soluble iron. Unfiltered water from the 125-ml glass bottles was used for the analysis of Kjeldahl nitrogen and unfiltered, acidified samples collected in the 100-ml polyethylene vials were analyzed for organic carbon content. Total phosphorus determinations were made on the unfiltered portion of water reserved from the 500-ml glass bottle.

Phosphorus. Total phosphorus and soluble inorganic phosphorus concentrations were determined using the Single Reagent Method described by the Environmental Protection Agency (1971). Soluble inorganic phosphorus was defined as that measured by direct colorometric analysis on a filtered sample. Soluble inorganic phosphorus was referred to as orthophosphorus (PO_4-P) or inorganic phosphorus, although, strictly speaking, inorganic phosphorus should include polyphosphates that would have been trapped by the 0.45 μ filter. Total phosphorus included all forms present in an unfiltered sample as measured by acid persulfate digestion followed by direct colorometric analysis. Total phosphorus included soluble inorganic phosphorus, acid hydrolyzable phosphorus (inorganic polyphosphates and some organic phosphorus) and all oxidizable organic phosphorus. Org.-P + hydro.-P was calculated as the difference between total phosphorus and soluble inorganic phosphorus and included all acid hydrolyzable and organic phosphorus of the unfiltered sample. Phosphorus concentrations are reported as $\mu g\ P \cdot L^{-1}$.

Nitrogen. The nitrite concentration was determined using Hach Chemical Company's NitriVer^R reagent. The method of Mullen and Riley as described by Barnes (1959) was used in the analysis of nitrate. The ammonia concentration was determined by the direct procedure outlined by Strickland and Parsons (1972). This technique proves satisfactory when concentrations of amino acids are low. Because the ammonia and nitrate tests include nitrite in the analyses, appropriate corrections were made to report the true nitrate and ammonia concentrations. Total soluble inorganic nitrogen was referred to as inorganic nitrogen and included soluble NO_3-N + NO_2-N + NH_3-N.

An additional nitrogen test was performed on unfiltered water by Kjeldahl digestion (Strickland and Parsons, 1972). Total Kjeldahl nitrogen (TKN) was defined as the sum of the free ammonia and organic nitrogen compounds which were converted to ammonium sulfate by acid digestion. The ammonia test described above was used to measure TKN following digestion, hence the nitrite concentration was also included in the analysis. Total nitrogen was calculated as the sum of the nitrogenous forms determined by Kjeldahl digestion plus nitrate. Total nitrogen included total soluble inorganic nitrogen, as well as dissolved and particulate organic nitrogen. Organic nitrogen (Org.-N) was computed as the difference between total nitrogen and soluble inorganic nitrogen and included dissolved and particulate forms. All nitrogenous species are reported as $\mu g\ N \cdot L^{-1}$.

Major cations. Measurements of calcium and magnesium concentrations were made using atomic absorption spectrophotometry with an Instrumentation Laboratory Inc. Model IL 151 Atomic Absorption-Flame Emission Spectrophotometer.

Sodium and potassium determinations were made using flame emission spectrophotometry and the previously mentioned instrumentation. Procedures followed are outlined by APHA (1971) and the IL Instruction

Manual. All cation concentrations are reported as $mg \cdot L^{-1}$.

Major anions. The Turbidimetric Method (APHA, 1971), utilizing Hach Chemical Company's SulfaVer^R IV, was employed to measure the sulfate concentration.

Chloride was determined titrametrically by the Mercuric Nitrate Method (APHA, 1971) utilizing Hach Chemical Company's standard 0.0141 N mercuric nitrate as the titrant and diphenylcarbazone indicator-buffer.

The fluoride concentration was measured by the SPANS Method (APHA, 1971) utilizing Hach Chemical Company's acid zirconyl SPANS reagent.

Methodology in determination of the carbonate and bicarbonate concentrations has been previously described. All anions are reported as $mg \cdot L^{-1}$.

Silica. The silica concentration was detected using the Colorimetric Molybdosilicate Method (APHA, 1971). Results are expressed as mg $SiO_2 \cdot L^{-1}$.

Organic carbon. The organic carbon content was determined by feeding an acidified, unfiltered sample into a Beckman Model IR 315 Infrared Carbonaceous Analyzer. Concentrations are reported as mg $C \cdot L^{-1}$.

Iron. Soluble iron was measured using Hach Chemical Company's FerroZine^R solution. Results are expressed as mg $Fe \cdot L^{-1}$.

Silica, sulfate and all nutrients except nitrite were analyzed with a Klett Summerson Model 900-3 Photoelectric Colorimeter with the proper filters. Analyses of sample waters were compared to standards of known concentration of the particular parameter being measured. Analyses of iron, fluoride, chloride and nitrite were performed with a Bausch and Lomb Spectronic 20 Colorimeter.

Tests for all nutrients except Kjeldahl nitrogen were performed within 12 hours of sample collection. The determination of fluoride, chloride, sulfate, soluble iron and silica concentrations was completed within 24 hours of sample collection. Analyses for calcium, magnesium, sodium, potassium, organic carbon and Kjeldahl nitrogen were performed within 96 hours of sample collection.

Water was collected at each station on each sampling date and all of the above described physical and chemical tests were performed on each sample.

Reservoir

All samples and *in situ* measurements were taken twice monthly during the periods of 17 June - 14 November 1975 and 16 April - 21 November 1976. Additionally, samples were collected at monthly intervals from December, 1975 through March, 1976 at Station 2.

Light

A vertical profile of light attenuation at each reservoir station on each sampling date was obtained by measuring the *in situ* light intensity with a Kahl Submarine Photometer. Light intensity was recorded at the surface and at 1-meter intervals to the depth at which 1% of the incident radiation penetrated.

Ruttner (1963) defined the compensation point as the depth at which assimilation was equal to dissimilation. At that point algal respiration was balanced by photosynthetic production. Verduin (1964) stated that the compensation depth was set at the depth to which 1% of the incident light penetrated. Therefore, the depth of 1% incident light penetration defined the lower limit of the euphotic zone.

The mean vertical extinction coefficient (m^{-1}) was computed as a measure of transparency of the euphotic zone at each station, on each sampling date, according to the formula of Hutchinson (1957).

Temperature and Dissolved Oxygen

Temperature and dissolved oxygen determinations were made using previously mentioned equipment. A 5-liter Van Dorn bottle was employed to obtain water samples at depth for these analyses. If a dissolved oxygen concentration proved to be <2 mg $O_2 \cdot L^{-1}$ at a given depth, a BOD Sewage Sampler was used to obtain water at that depth and a Winkler titration was performed to verify the results. Temperature and dissolved oxygen concentrations were measured at the water surface and at 1-meter intervals through the euphotic zone. Through the aphotic zone, water samples were collected at 2-meter intervals for dissolved oxygen and temperature determinations.

Specific Conductance, pH and Alkalinity

Previously discussed materials and methods were used for the measurements of specific conductance, pH and alkalinity. The vertical sampling scheme mentioned above for temperature and dissolved oxygen was followed relative to pH and alkalinity determinations. Specific conductance was measured at 2-meter intervals from the surface to the bottom of the water column.

Turbidity

Turbidity determinations were made at 2-meter intervals through the water column using previously mentioned methodology.

Additional Water Chemistry

All water samples were collected with a 5-liter Van Dorn bottle at 2-meter intervals from the surface to the bottom of the reservoir. Chemical parameters analyzed included all those tested on non-reservoir waters with the exception of the Kjeldahl nitrogen test. All methods and procedures were as previously delineated.

Timing relative to collection and subsequent analysis of samples was similar to that of non-reservoir waters. Water was collected in the prescribed manner at each reservoir station on each sampling date (except where previously noted) and all of the above mentioned physical and chemical tests were performed on each sample.

Phytoplankton Standing Crop and Chlorophyll *a*

On every sampling cruise duplicate euphotic zone composite samples were collected at each reservoir station for a total of six composites per cruise. Each composite was formed by homogenously mixing samples taken at 1-meter intervals from the surface through the euphotic zone. An aliquot of each composite was preserved with Lugol's solution and reserved for phytoplankton enumeration. A large volume of each composite sample was used for chlorophyll determinations.

In the laboratory the preserved plankton samples were shaken to homogenously resuspend the algae. Subsamples ranging in volume from 2 to 75 ml were then withdrawn and prepared for microscopic analysis according to the Membrane Filter Concentration Technique (APHA, 1971). The stained plankton were identified and counted in transects along the diameter of the filter after the water and air in the matrix of the filter were replaced with immersion oil. This method has the advantage of providing a permanent phytoplankton mount (UNESCO, 1974).

One transect was counted at each of three different magnifications: 100X, 430X and 970X. The lower magnifications were sufficient to record the occurrence of larger forms while it was necessary to use

970X to identify and enumerate the microplankton. Algae were enumerated at the species level in most cases in ten randomly selected fields along the transects defined at both 100X and 430X. Twenty random fields were counted along the 970X transect for a total count of 40 fields.

A mean volume of each morphological unit (trichome, colony, cell, etc.) was calculated for most phytoplankton species by likening the alga to the appropriate geometrical figure (i.e. sphere, cylinder, cone, etc.) and averaging the mean linear dimensions of a large population of individuals (usually 100) as measured with a Whipple ocular micrometer. Literature values were used to estimate the mean cell volumes of rarely occurring species. A series of ratios involving the volume of water filtered, the magnification and the number of fields counted were used to extrapolate to cell volume per liter of water ($mm^3 \cdot L^{-1}$). The standing crop of phytoplankton is best expressed in these terms because numbers do not give a true evaluation of biomass due to the variability in size among species (Wetzel, 1975). A discussion of the statistical validity of the volumetric determination of plankton biomass may be found in Willèn (1976) and a statistical treatment of the direct count method is presented in Lund, *et al.* (1958).

Algae were identified from living and preserved material collected throughout the course of the study. Taxonomic references used to identify most groups included Smith (1950), Tiffany and Britton (1952), Taft and Taft (1971) and Prescott (1952, 1970). Keys by Hustedt (1930), Patrick and Reimer (1966), FWPCA (1966) and Hansmann (1973) proved helpful in identifying the Bacillariophyceae, as did Drouet (1959) for the Myxophyceae. All microscopic analyses were performed with a Bausch and Lomb Dynoptic Microscope.

The chlorophyll *a* concentration was determined by filtering a known volume of each euphotic zone composite through a Millipore^R membrane filter of 0.45 micron pore diameter. The filter was then placed in a 12-ml centrifuge tube to which 5 ml of 90% acetone was added. Chlorophyll was extracted by placing the tube in complete darkness in a freezer for 24 hours. The content of the tube was then centrifuged and the chlorophyll *a* concentration of the supernatent was measured with a Beckman DU Spectrophotometer according to the methods and equation of Strickland and Parsons (1972). The equation used to calculate the chlorophyll *a* concentration eliminated some of the photometric interferences due to chlorophyll degradation products. A rough estimate of the phytoplankton standing crop was obtained from the chlorophyll *a* analysis.

Phytoplankton Primary Production

An estimate of limnetic primary production was obtained at all three reservoir stations through the use of chlorophyll and light data as outlined by Ryther and Yentsch (1957). This estimate proved to be unsatisfactory and the formula was modified to obtain a better estimate of gross primary production.

Data relative to total incident surface radiation which is a necessary component of the formula of Ryther and Yentsch (1957) and the modified empirical formula herein presented was obtained using either a Kip-Zonen Model CM-3 or a Kahl pyranometer and an Esterline Angus Model 80-M recorder. Daily radiation was converted to $\text{langley}\cdot\text{day}^{-1}$ as described by the Kip-Zonen and Kahl manuals. The pyranometer was located on the east bank of the reservoir, opposite Station 3 (see Figure 2 for location of Station 3).

Primary production was also measured at Station 2 by the more accurate carbon-14 technique first described by Steemann-Nielsen (1952). Samples were collected at 1-meter intervals from the surface to the bottom of the euphotic zone and duplicate light and opaque 300-ml BOD bottles were used to resuspend the samples to their original depths. All samples were spiked with 5 μCi or 10 μCi of $NaH^{14}CO_3$ obtained from New England Nuclear prior to resuspension. A 3/4 inch diameter iron rod of 6 m length permanently attached to two styrofoam floats at Station 2 of the reservoir was used to incubate the samples in the absence of shading. These *in situ* ^{14}C experiments lasted approximately 4 hours and were conducted during the second and/or third photoperiods in a light day divided into 5 equal photoperiods. Vollenweider (1965) and Vollenweider and Nauwerck (1961) suggested that this procedure facilitated the day rate estimation of photosynthesis.

Upon completion of the experiment, the BOD bottles were fixed with 1 ml of 1% merthiolate and an aliquot of each sample (10-25 ml; depending upon the size of the standing crop) was then filtered through a 0.45 μ Millipore^R filter. The sides of the filter funnel and filter were then rinsed with 100 ml of 1% HCl to remove possible ^{14}C precipitated extracellularly (Wetzel, 1965). Filters were then placed in a scintillation vial containing 4 ml of absolute methanol to eliminate the loss of activity during storage (Wallen and Geer, 1968). The above procedure was completed within 24 hours of sample collection. The scintillation vials were filled with a scintillation fluor similar to those suggested by Vollenweider (1969) consisting of 4 g 2,5 diphenyloxazole (PPO), 0.1 g p-Bis(2-(5-phenyloxazolyl))-benzene (POPOP) and 1 L toluene and were radioassayed within 96 hours of sample collection on a Packard Tricarb Liquid Scintillation Counter. An isotope effect of 6% was assumed in calculating activity (Steemann-Nielsen, 1955).

Statistical Analyses

Statistical methods employed in the analyses of the data collected during this study are found in Snedecor and Cochran (1967) or Steel and Torrie (1960). In the statistical analyses of all data, the 5% probability level was considered the critical value in determining whether the author rejected or failed to reject the null hypothesis (i.e. for $p \leq 0.05$ the null hypothesis was rejected). However, values of $p = 0.06$-0.10 are reported for reference purposes.

RESULTS AND DISCUSSION

One of the defined goals of the Tongue River Reservoir study program was to obtain data during the early operational stages of the West Decker Mine against which future comparisons may be made to evaluate the long range impact of coal mining on the reservoir ecosystem. As such, it was appropriate that the defined objectives of the immediate study by addressed for an "average" water year. An abnormally low reservoir stage was encountered during the first four months of this study (June - September 1975) because of the excessive reservoir drawdown undertaken to repair the outflow gate. The reservoir's hydrologic regime for the balance of the study approximated the norm. When compared with records for the previous eighteen years (unpublished data provided by Mt. Dept. Nat. Res. and Cons.), the period from ice-in (mid-November) 1975 to ice-in (mid-November) 1976 was chosen as an "average" water year. Nonetheless, some of the data collected in the summer of 1975 was used as supportive evidence for the interpretation and discussion of results. In establishing means of various parameters for the average water year, the pre-defined seasonal sampling regime has been taken into account and the data have been weighed accordingly.

Hydrology

Hydrologic data were supplied by the U.S. Geological Survey (1976, 1977 and unpublished data), the Montana Bureau of Mines and

Geology (unpublished data) and the Montana Department of Natural Resources and Conservation (unpublished data).

The Tongue River provided the only significant inflow of surface water into the Tongue River Reservoir. Ground water inflow was estimated to vary between 0.003 $m^3 \cdot sec^{-1}$ (0.1 cfs) at the southern end of the reservoir and 0.3 $m^3 \cdot sec^{-1}$ (10 cfs) at the northern end of the reservoir. The reservoir's only outflows were the gate at the bottom of the dam and the emergency spillway at the crest of the dam.

During this entire study, the maximum rate of water discharge of the Tongue River above the reservoir (near the inflow station) was 152 $m^3 \cdot sec^{-1}$ (5350 cfs), observed on 22 June 1975. The minimum rate of water discharge at this station during the study was 2.74 $m^3 \cdot sec^{-1}$ (97 cfs), recorded on 6 September 1976. At the Tongue River Dam (near the outflow station), the maximum rate of Tongue River water discharge during this study was 140 $m^3 \cdot sec^{-1}$ (4950 cfs), observed on 25 June 1975. The minimum rate of water discharge at this station was 0.42 $m^3 \cdot sec^{-1}$ (15 cfs), recorded on 5 November 1975. Averaged over the past 16 and 37 years, the mean rates of discharge of the reservoir's inflow and outflow waters were 14.5 $m^3 \cdot sec^{-1}$ (511 cfs) and 13.1 $m^3 \cdot sec^{-1}$ (461 cfs). During the two calendar years encompassing this study the mean rate of discharge of the inflow was 17.3 $m^3 \cdot sec^{-1}$ (611 cfs) while that of the outflow was 17.2 $m^3 \cdot sec^{-1}$ (607 cfs). These mean rates of discharge represent departures of +19% and +31% from the long-term average inflow and outflow water discharge rates, respectively.

Figure 3 depicts the average monthly water discharge of the Tongue River, recorded at the previously mentioned gaging stations located above and below the Tongue River Reservoir. Peak runoff typically occurred in mid-June and, on an annual basis, the highest rates of discharge of the inflow water were recorded at that time. Correspondingly high rates of discharge of outflow water were recorded during spring runoff, upon attainment of reservoir storage capacity. The lowest rates of discharge of both the inflow and outflow water were noted during the winter months.

The Tongue River Reservoir is primarily a storage reservoir and, as such, exhibits widely fluctuating pool elevations on an annual basis in response to spring runoff and downstream irrigation needs throughout the growing season. The average annual reservoir surface elevation fluctuation is 3-6 meters, with storage peaking in late June or early July and reaching a minimum in late September. Storage is held constant at about 432 X 10^5 m^3 (35,000 acre-ft) throughout the winter months. This figure represents about 60% of the storage capacity of the Tongue River Reservoir. During 1975 the pool elevation fluctuated 8 meters as a result of the previously mentioned reservoir drawdown.

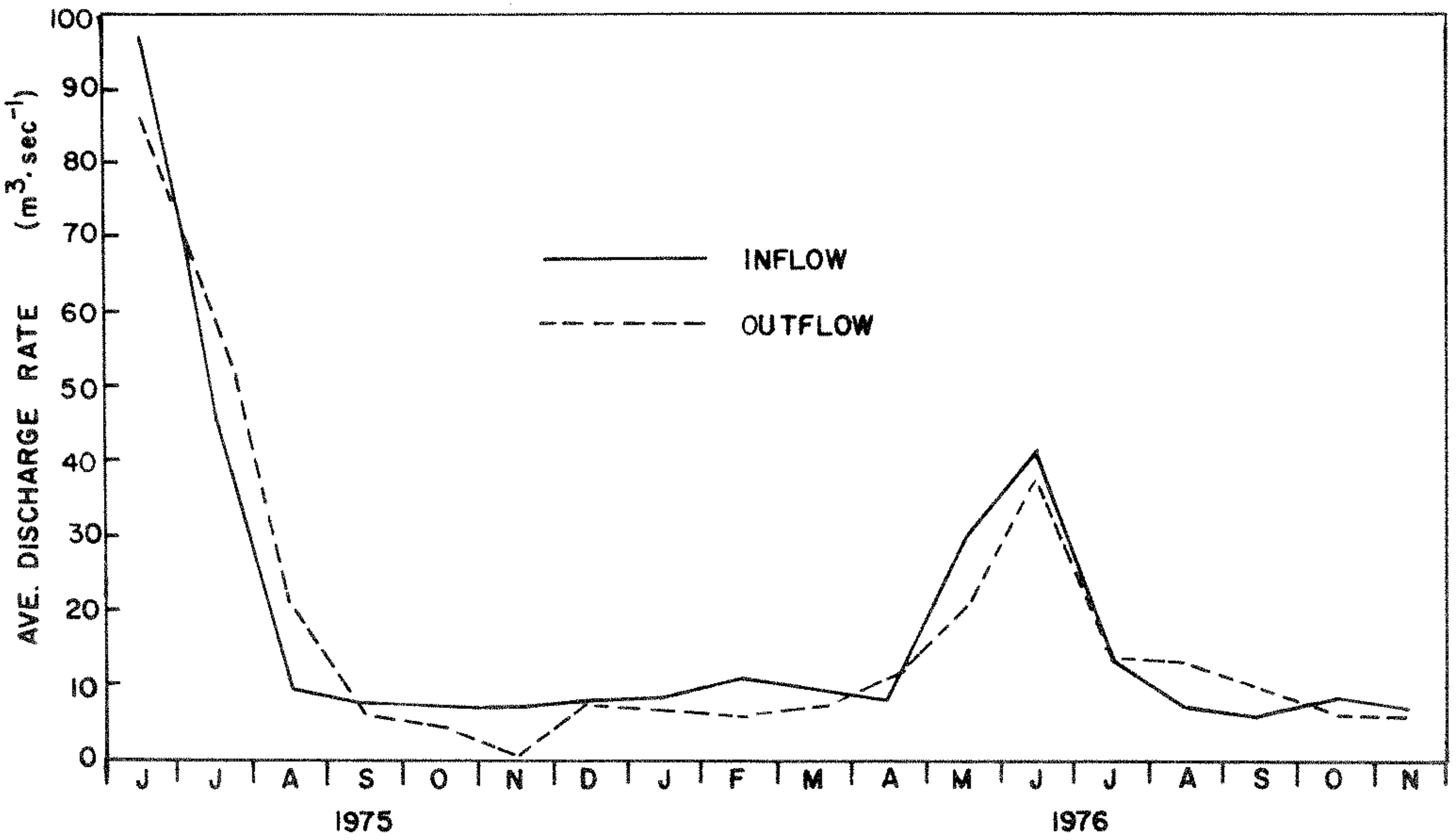

Figure 3. Mean water discharge rates ($m^3 \cdot sec^{-1}$) of the Tongue River Reservoir inflow and outflow waters. Data are averaged over monthly intervals from June 1975 through November 1976 (U.S.G.S., 1976, 1977 and unpublished data).

Figure 4 presents the annual pool elevation regime and storage capacity averaged over two week intervals for the years encompassed by this study. The maximum content observed was 889 X 10^5 m^3 (72,100 acre-ft) on 24 June 1975 and the minimum storage was 159 X 10^5 m^3 (12,900 acre-ft) on 19 September 1975. The reservoir volume on 24 June 1975 was 20.2% higher than the estimated usable capacity of 740 X 10^5 m^3 (60,000 acre-ft). This followed abnormally high early June runoff (Figure 3) catalyzed by the rapid melting of excessive snowpack in the Big Horn mountains. At that time ambient temperatures were high and water crested the reservoir spillway. The minimum storage represented only 21.5% of the usable capacity, but was considerably greater than the dead storage volume of 17.3 X 10^5 m^3 (≃1400 acre-ft).

The water storage history of the Tongue River Reservoir is depicted in Figure 5. Mean reservoir water storage from 1956-1977 was 440 X 10^5 m^3 (35,705 acre-ft). Due to the annual extremes in pool elevation, submergent and emergent aquatic vegetation have not become established.

A summary of the hydrologic data for the Tongue River Reservoir during the 1975 and 1976 calendar years is in Table 2. The data for 1976 is probably more indicative of the annual hydrologic regime of the reservoir than the 1975 data. Unusually short retention times were

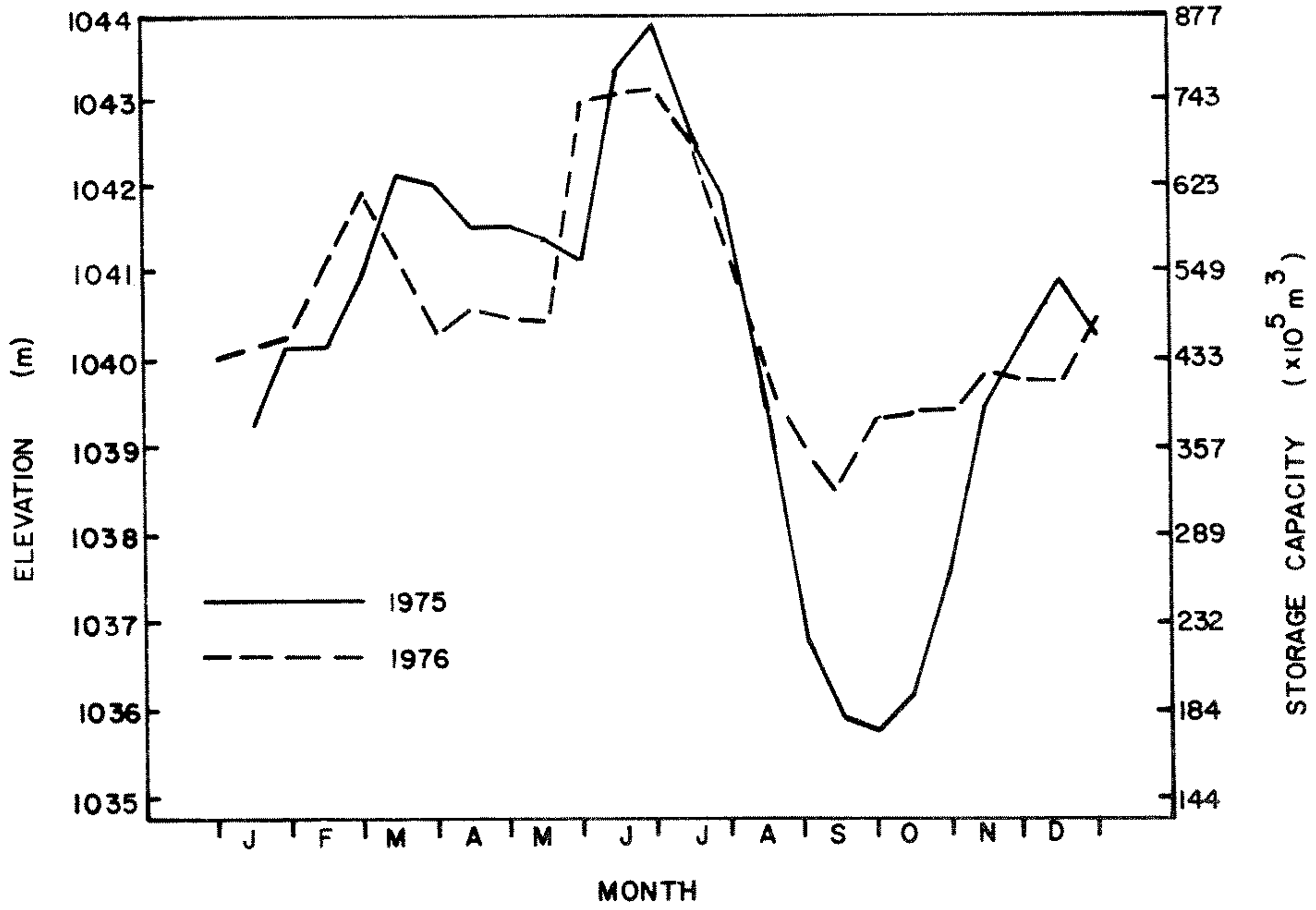

Figure 4. Pool elevation and storage capacity of the Tongue River Reservoir in 1975 and 1976 (Mt. Dept. Nat. Res. and Cons., unpublished data and U.S.G.S., 1977).

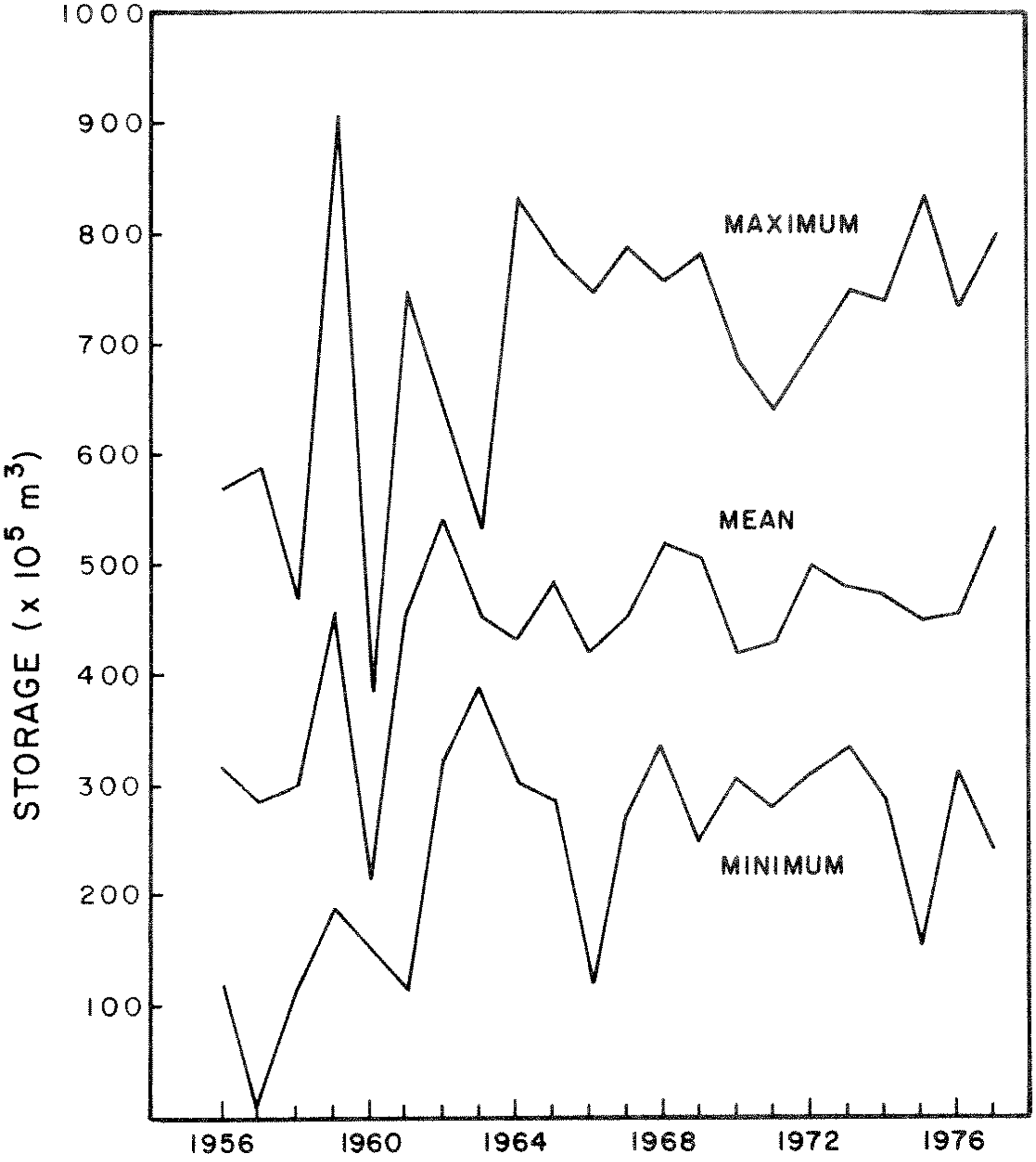

Figure 5. Water storage history of the Tongue River Reservoir from 1956 through 1977 (Mt. Dept. Nat. Res. and Cons., unpublished data derived from monthly averages).

Table 2. Summary of hydrologic data for the Tongue River Reservoir during the calendar years 1975 and 1976.

		Total Inflow $x10^5m^3$	Total Outflow $x10^5m^3$	Storage Change $x10^5m^3$	Average Storage $x10^5m^3$	Average Discharge $x10^5m^3 \cdot day^{-1}$	Retention Time (days)	Average Surface Elevation (m)
	Jan.	136.4	113.6	+ 22.8	363.3	3.7	98.1	1039.3
	Feb.	121.1	102.0	+ 19.1	429.3	3.6	119.3	1040.2
	Mar.	407.7	212.3	+258.4	618.2	6.9	89.5	1042.1
1	Apr.	302.9	361.6	- 58.7	558.0	12.1	46.1	1041.5
	May	922.2	1015.7	- 93.5	482.5	32.8	14.7	1040.8
9	June	2521.1	2238.0	+283.1	999.8	74.6	13.4	1043.3
	July	1268.8	1578.9	-310.1	724.1	50.9	14.2	1042.7
7	Aug.	253.2	581.4	-328.2	367.8	18.8	19.5	1039.5
	Sept.	318.6	175.3	- 36.7	164.1	5.8	28.2	1035.8
5	Oct.	198.5	116.0	+ 82.5	179.6	3.7	48.5	1036.2
	Nov.	186.3	29.7	+156.6	354.8	1.0	354.8	1039.3
	Dec.	205.4	194.0	+ 11.4	479.7	6.3	76.1	1040.8
	Ave.				476.7		76.8	
	Total	6725.2	6718.5	+ 6.7				
	Jan.	220.5	184.9	+ 35.9	431.7	6.0	71.9	1040.0
	Feb.	259.4	160.9	+ 98.5	441.5	5.7	77.4	1040.2
	Mar.	266.8	196.3	+ 70.5	518.1	6.3	82.2	1041.1
1	Apr.	222.2	272.8	- 50.6	466.9	9.1	51.3	1040.7
	May	789.8	545.0	+244.8	451.4	17.5	25.7	1040.5
9	June	1100.0	985.1	+114.9	706.9	32.8	21.5	1043.0
	July	344.8	379.7	- 34.9	729.9	12.3	59.3	1043.2
7	Aug.	186.5	348.7	-162.2	526.8	11.2	47.0	1041.4
	Sept.	150.3	247.2	- 96.9	342.2	8.2	41.7	1039.1
6	Oct.	212.2	169.8	+ 42.4	354.8	5.5	64.5	1039.3
	Nov.	168.7	135.7	+ 33.0	365.6	4.5	81.2	1039.4
	Dec.	166.8	116.0	+ 50.8	397.2	3.7	107.3	1039.8
	Ave.				477.7		60.9	
	Total	4088.0	3715.1	+372.9				

observed in the summer of 1975 in preparation for repairs to the outflow gate.

Inflow and Outflow Waters

Turbidity

Turbidity in the Tongue River Reservoir inflow water ranged from 3.1-53.0 JTU and averaged 18.2 JTU over the course of a year. Turbidity has been defined as the degree of opaqueness produced in water by suspended particulate matter (Reid, 1961). Suspended particulates affecting turbidity may be biogenic or abiogenic in nature. In the Tongue River turbidity was due in part to suspended organics, but primarily to siltaceous matter.

Neel (1963) and Symons, *et al.* (1964) found that reservoirs may serve as sediment traps for inflowing water, leading to increased downstream transparency. The data of Figure 6 indicate that this phenomenon occurred in the Tongue River Reservoir. During peak runoff in May and June a large volume of silt-laden water flowed into the reservoir and a strong correlation ($n \doteq 29$, $r \doteq 0.63$, $p < 0.01$) was found between inflow rate and inflow turbidity throughout this study.

The deposition of suspended abiotic matter in the upper end of the reservoir accounted for decreased turbidity of the outflowing water relative to the inflowing water. The sudden increase in tur-

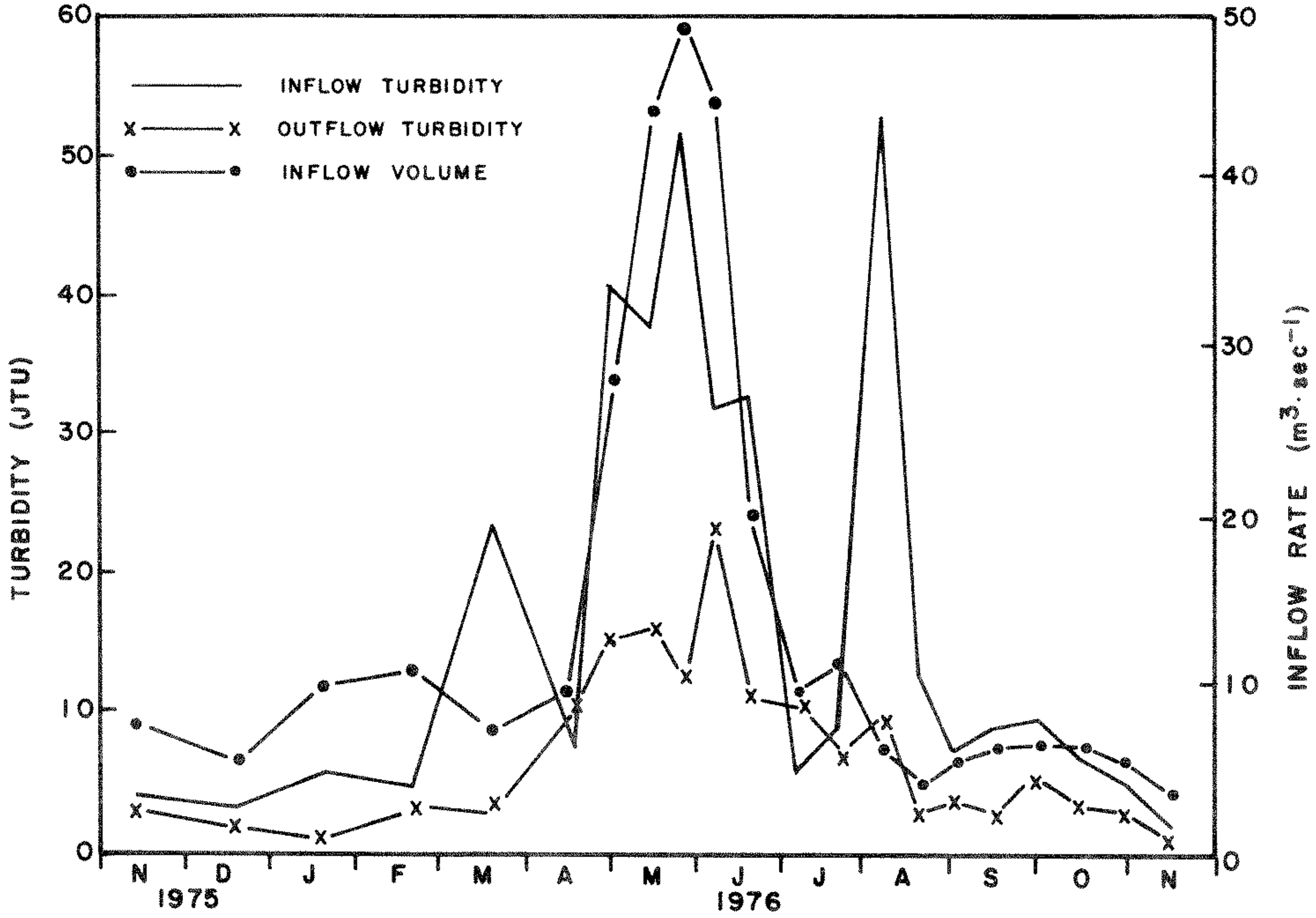

Figure 6. Turbidity of the inflow and outflow waters of the Tongue River Reservoir versus the inflow rate.

bidity of the inflow on 6 August 1976 was attributed to repair work on the County Bridge at the sampling site. The turbidity of the outflow ranged from 1.6 - 21.0 JTU and averaged 7.6 JTU. The turbidity of the outflow increased somewhat during spring runoff but the magnitude of the increase was not as substantial as that of the inflow, reflecting the deposition of silt.

Temperature and Specific Conductance

The annual temperature regime at the Inflow and Outflow stations is graphically illustrated in Figure 7. Winter water temperatures of both the inflow and outflow were close to the freezing point. A gradual warming trend began with the start of spring runoff in April and peak temperatures were reached in August at both the Inflow and Outflow stations. Due to the deepwater withdrawal, consistantly lower water temperatures were observed at the outflow than the inflow on any given sampling date during the spring and summer. The highest outflow water temperatures were observed in the months of the warmest ambient temperatures and the lowest reservoir stages. The yearly range of water temperature of the inflow was 1.3-25.8°C and the average was 13.3°C. The annual temperature range of the outflow water was 1.0-20.3°C and the average was 11.2°C.

Figure 8 presents the specific conductance regime at the Inflow and Outflow stations for a year. Specific conductance fluctuated

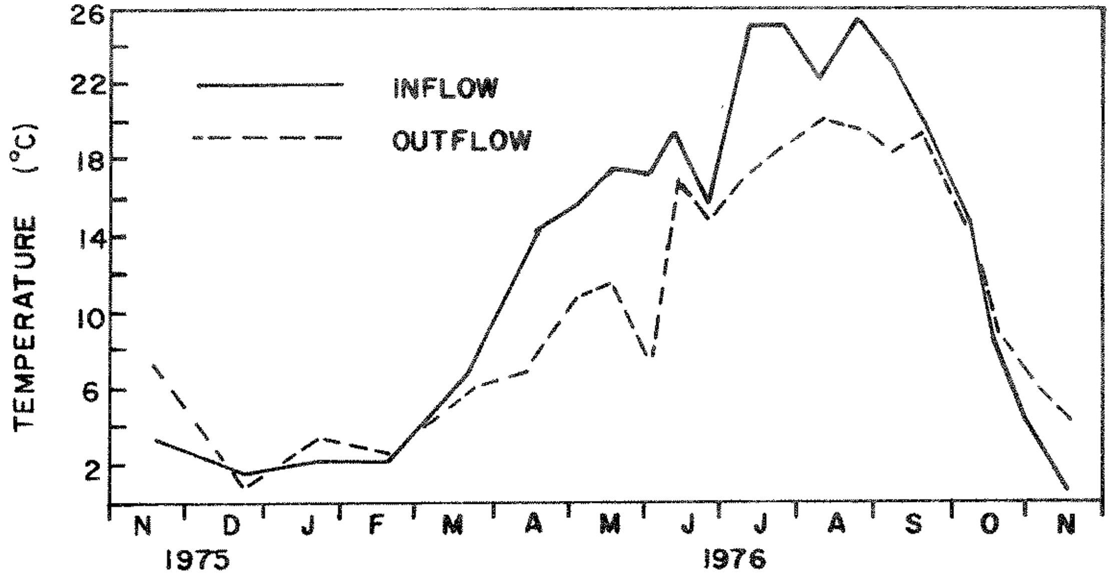

Figure 7. Annual temperature regime of the inflow and outflow waters of the Tongue River Reservoir.

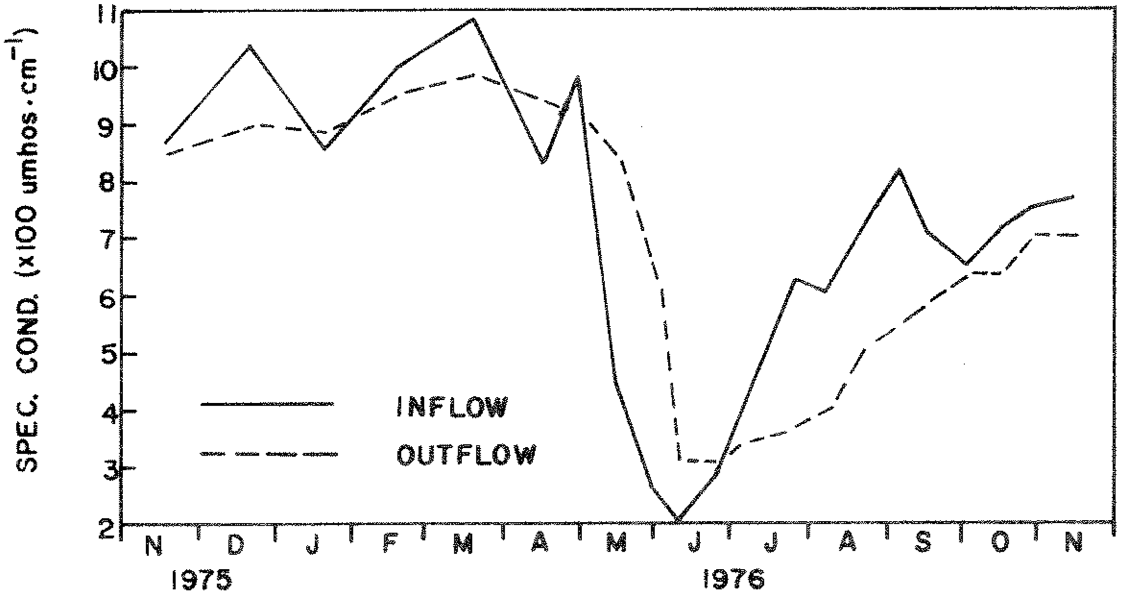

Figure 8. Annual specific conductance regime of the inflow and outflow waters of the Tongue River Reservoir.

around 875-1075 μmhos·cm^{-1} at the Inflow station during the fall and winter of 1975-1976. With the onset of spring runoff conductivity values dipped to a low of 199 μmhos·cm^{-1} on 10 June 1976 reflecting the low ionic content of snowmelt. Feth, *et al.* (1964) have recorded conductivity values of 2-42 μmhos·cm^{-1} for melted snow in the western United States. With the completion of spring runoff, specific conductance increased in the inflow water through the fall. A significant correlation ($n = 29$, $r = 0.79$, $p < 0.01$) was found between low inflow specific conductance and high inflow volume, which further illustrates the effect of spring runoff on specific conductance of the inflowing river. Specific conductance at the Outflow station followed a pattern similar to that of the Inflow, with the graph displaced one month to the right owing to the effect of impoundment. The ranges of specific conductance measured at the Inflow and Outflow stations were 199-1072 μmhos·cm^{-1} and 297-989 μmhos·cm^{-1} respectively. The mean conductivity of the inflowing water was 694 μmhos·cm^{-1}, while that of the outflowing water was 662 μmhos·cm^{-1}.

Major Cations and Anions

Table 3 includes a list of the ranges and averages of the concentrations of major anions and cations at the Tongue River Reservoir Inflow and Outflow stations. The chemical composition of soil and rock and the ionic exchange capacity of the parent material in the

Table 3. Ranges and averages of some chemical and physical parameters of the inflow and outflow waters from November 1975 through November 1976.

Parameter	Inflow	Ouflow
Ca^{++} ($me \cdot L^{-1}$)	1.05-4.79 3.04	1.40-4.24 2.84
Mg^{++} ($me \cdot L^{-1}$)	0.79-4.69 3.18	1.23-4.24 2.97
Na^{+} ($me \cdot L^{-1}$)	0.31-2.39 1.35	0.51-1.97 1.27
K^{+} ($me \cdot L^{-1}$)	0.03-0.16 0.10	0.05-0.16 0.10
Total alkalinity ($me \cdot L^{-1}$)	1.61-5.87 3.92	1.95-5.31 3.64
$SO_4^{=}$ ($me \cdot L^{-1}$)	0.65-6.25 3.56	1.12-5.68 3.46
Cl^{-} ($me \cdot L^{-1}$)	0.03-0.16 0.09	0.05-0.14 0.08
SiO_2 ($mg \cdot L^{-1}$)	3.1-12.6 7.7	1.8-11.9 6.3
NH_3-N ($\mu g \cdot L^{-1}$)	0-175 20	0-86 32
NO_3-N ($\mu g \cdot L^{-1}$)	0-173 38	0-143 32
NO_2-N ($\mu g \cdot L^{-1}$)	0-14 3	0-8 3
Total-N ($\mu g \cdot L^{-1}$)	67-620 377	67-610 417
PO_4-P ($\mu g \cdot L^{-1}$)	4-70 23	0.37 10
Total-P ($\mu g \cdot L^{-1}$)	36-164 67	12-61 40
Spec. cond. ($\mu mhos \cdot cm^{-1}$ @ 25°C)	199-1072 694	297-989 662
pH	7.9-8.8 8.4	7.8-8.6 8.3
Turbidity (JTU)	3.1-53.0 18.2	1.6-21.0 7.6
Temperature (°C)	1.3-25.8 13.3	1.0-20.3 11.2

drainage area influence the rate of weathering and ion supply to surface waters of the drainage area (Wetzel, 1975). General processes of weathering that control ion supply include solution, redox reactions, pH changes, and the formation of complexes (Gorham, 1961). Rodhe (1949) has pointed out that the ionic composition of surface water ultimately reflects the changes that occur after the initial solution of ions from the parent material as well as the lithological considerations described above.

Hutchinson (1957) stated that the interaction of the above processes led to the following cationic order of abundance in the river waters of the world: $Ca^{++} > Mg^{++} > Na^{+} > K^{+}$. In both the inflow and outflow waters the order of abundance of cations was $Ca^{++} \simeq Mg^{++} > Na^{+} > K^{+}$. This was probably due to the preponderance of dolomite $[CaMg(CO_3)_2]$ in the drainage area. Upon solution, dolomite should yield nearly equal concentrations ($me \cdot L^{-1}$) of calcium and magnesium, barring the precipitation of calcium and calcium carbonate (Hem, 1970).

An analysis of a hardwater system should yield the following anionic ranking of abundance: $HCO_3^{-} > Cl^{-} \geq SO_4^{=}$ (Hutchinson, 1957). Bicarbonate should clearly dominate the anionic chemical analysis of most waters due to the formation and subsequent dissociation of carbonic acid into bicarbonate when atmospheric CO_2 comes into contact with natural water. However, in both the inflowing and outflowing waters the order of abundance of anions was $HCO_3^- \geq SO_4^= > Cl^-$. Natural systems in contact with gypsum ($CaSO_4 \cdot 2H_2O$), anhydrite ($CaSO_4$), pyrite (FeS_2), or drainage waters through arid soils high in sodium sulfate (Na_2SO_4) could be expected to contain high concentrations of sulfate. For this reason the millequivalent concentration of sulfate was nearly equal to that of bicarbonate in the river water above and below the reservoir. The Tongue River inflow and outflow could be characterized as calcium-magnesium bicarbonate-sulfate waters.

Tables 4 and 5 illustrate the seasonal distribution of cations and anions ($me \cdot L^{-1}$) in the inflow and outflow. In both cases, the millequivalent concentrations of the anions and cations followed the same seasonal trends. The concentrations of anions and cations were high from the fall through the early spring and reached a low in the early summer, reflecting the dilution effect of spring runoff. Increases in the concentrations of anionic and cationic species were noted through the late summer and fall due to evaporative concentration and the lack of rainfall.

pH

The pH of the Tongue River directly above the reservoir ranged from 7.9-8.8 and the mean pH was 8.4. The range of pH for the outflowing Tongue River water was 7.8-8.6 with a mean of 8.3. No seasonal trends in pH values were noted at either the Inflow or Outflow

Table 4. Average seasonal distribution of cations and anions ($me \cdot L^{-1}$) in the inflow water from November 1975 through November 1976.

Date	Ca^{++}	Mg^{++}	Na^{+}	K^{+}	Total Cations	Alk.	Cl^{-}	F^{-}	$SO_4^{=}$	Total Anions
15 Nov. 75	3.80	4.15	1.69	0.12	9.76	5.55	0.10	0.02	4.32	9.99
18 Dec	4.77	4.90	1.98	0.15	11.80	5.95	0.16	0.01	5.55	11.67
17 Jan. 76	3.80	3.70	1.36	0.12	8.98	4.95	0.10	0.02	3.89	8.96
21 Feb.	4.13	4.24	1.85	0.15	10.37	5.17	0.12	0.02	5.06	10.35
22 Mar.	4.31	4.69	2.39	0.16	11.55	5.06	0.07	0.02	6.25	11.40
16 Apr.	3.23	3.60	1.67	0.10	8.60	4.26	0.10	0.02	4.45	8.83
1 May	3.69	4.19	2.20	0.14	10.22	4.49	0.12	0.02	5.05	9.68
14 May	2.00	2.22	0.85	0.07	5.14	2.73	0.07	0.01	2.17	4.98
29 May	1.26	1.07	0.39	0.04	2.76	1.68	0.05	0.01	0.84	2.58
10 June	1.09	0.79	0.31	0.03	2.22	1.58	0.03	0.01	0.66	2.28
25 June	1.31	1.17	0.41	0.04	2.93	1.94	0.04	0.01	1.05	3.04
8 July	2.04	1.52	0.61	0.05	4.22	2.61	0.01	0.05	1.45	4.12
22 July	2.84	2.73	1.23	0.10	6.90	3.90	0.02	0.05	3.15	7.12
6 Aug.	2.52	2.69	1.48	0.10	6.79	3.62	0.02	0.07	3.04	6.75
20 Aug.	3.50	3.80	1.57	0.13	9.00	4.61	0.01	0.10	4.08	8.80
2 Sept.	3.32	4.12	1.75	0.16	9.35	4.59	0.01	0.09	4.55	9.24
16 Sept.	3.32	3.50	1.55	0.12	8.49	4.52	0.03	0.08	4.02	4.65
2 Oct.	2.89	3.32	1.23	0.15	7.59	3.98	0.03	0.10	3.50	7.61
16 Oct.	3.55	3.52	1.34	0.11	8.52	4.56	0.02	0.09	3.60	8.27
30 Oct.	3.49	3.68	1.36	0.10	8.63	4.80	0.02	0.09	3.92	8.83
12 Nov.	3.55	3.68	1.43	0.10	8.76	4.76	0.03	0.08	4.02	8.89

Table 5. Average seasonal distribution of cations and anions ($me \cdot L^{-1}$) in the outflow water from November 1975 through November 1976.

Date	Ca^{++}	Mg^{++}	Na^{+}	K^{+}	Total Cations	Alk.	Cl^{-}	F^{-}	$SO_4^{=}$	Total Anions
15 Nov. 75	3.71	4.00	1.70	0.13	9.54	4.78	0.10	0.02	4.95	9.85
18 Dec.	3.80	3.95	1.67	0.14	9.56	5.05	0.10	0.01	4.80	9.96
17 Jan. 76	3.89	4.15	1.70	0.14	9.88	5.19	0.10	0.02	4.82	10.13
21 Feb.	4.01	3.95	1.66	0.14	9.76	5.35	0.10	0.01	4.46	10.28
22 Mar.	4.25	4.29	1.97	0.15	10.66	5.26	0.09	0.01	5.68	11.04
16 Apr.	3.56	4.04	1.76	0.12	9.48	4.69	0.12	0.02	4.87	9.70
1 May	3.69	4.03	1.87	0.12	9.71	4.58	0.12	0.03	4.91	9.64
14 May	3.62	4.24	1.90	0.16	9.92	4.39	0.14	0.02	5.15	9.70
29 May	2.49	2.72	1.14	0.08	6.43	3.11	0.10	0.02	3.04	6.27
10 June	1.46	1.23	0.51	0.05	3.25	1.92	0.05	0.01	1.31	3.29
25 June	1.42	1.23	0.45	0.05	3.15	1.98	0.04	0.01	1.12	3.15
8 July	1.71	1.30	0.47	0.05	3.53	2.10	0.01	0.04	1.27	3.42
22 July	1.64	1.40	0.52	0.05	3.61	2.25	0.01	0.03	1.39	3.68
6 Aug.	1.84	1.73	0.73	0.06	4.36	2.70	0.01	0.06	1.68	4.45
20 Aug.	2.26	2.10	0.87	0.09	5.32	2.95	0.01	0.05	2.15	5.16
2 Sept.	2.28	2.59	1.13	0.11	6.11	3.05	0.01	0.07	2.79	5.91
16 Sept.	2.32	2.76	1.39	0.11	6.58	3.13	0.02	0.07	3.40	6.62
2 Oct.	2.76	3.22	1.32	0.15	7.45	3.61	0.02	0.08	3.62	7.33
16 Oct.	2.89	3.16	1.38	0.12	7.55	3.81	0.02	0.09	3.70	7.62
30 Oct.	3.01	3.45	1.34	0.10	7.90	4.22	0.02	0.11	3.77	8.12
12 Nov.	3.06	3.26	1.37	0.11	7.80	4.13	0.02	0.08	4.02	8.25

stations. Rada (1974) observed reduced values of pH in the spring and rising pH values over the course of the summer in the Missouri River above Canyon Ferry Reservoir in Montana. He attributed this phenomenon to turbid runoff water and the scouring of periphyton in the spring, followed by increased periphyton productivity in the summer. Low periphyton activity and high buffering capacity in the Tongue River discourage an increase in pH during the summer growing season.

Nutrient Concentrations and Loads of the Inflowing and Outflowing Tongue River

The natural nutrient input into a water system is a function of the drainage basin geochemistry, drainage basin size, hydrology and precipitation pattern (Hynes, 1970). Additional considerations include the drainage basin land use pattern and the human population characteristics within the area (Brezonik, 1972).

Figures 9 and 10 depict the seasonal variations in the concentrations of inorganic-N, inorganic-P, organic + hydrolyzable-P, and organic-N in the inflowing water. The highest concentrations of inorganic nutrients in the inflow were observed during the winter months. No reasonable explanation is offered for this phenomenon. However, this pattern may represent the norm as Leathe and Whalen (1978) found high levels of nitrate and orthophosphorus at the Inflow

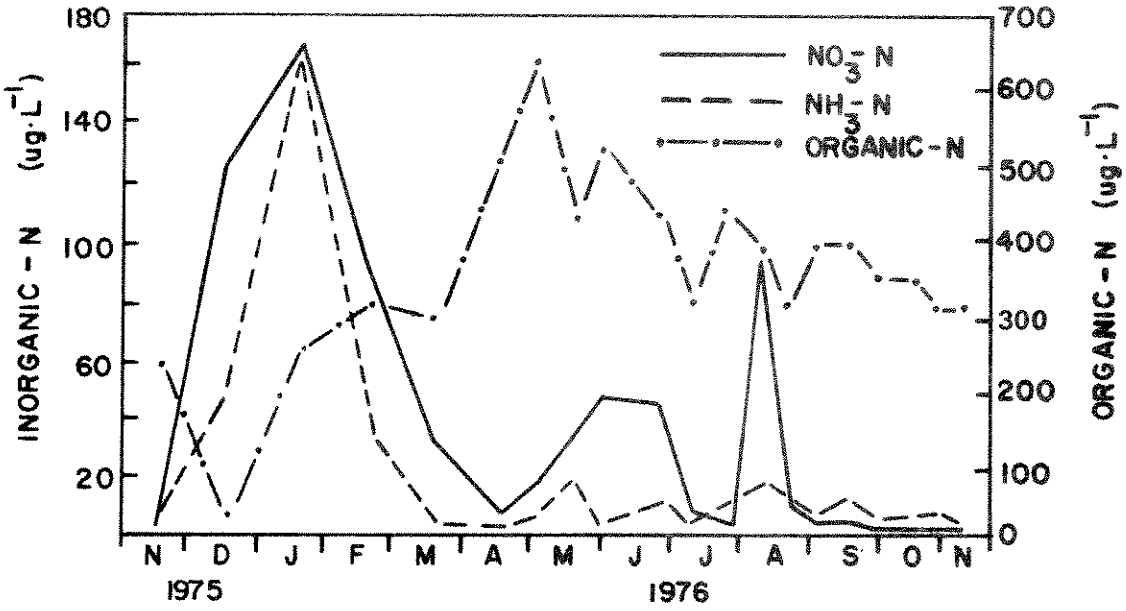

Figure 9. Seasonal concentrations of inorganic and organic nitrogen in the inflow.

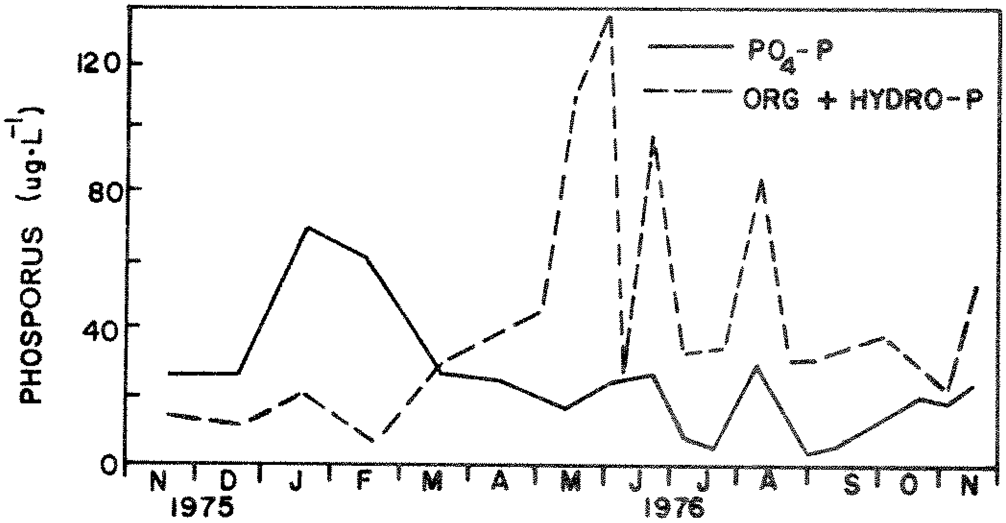

Figure 10. Seasonal concentrations of orthophosphorus and organic + hydrolyzable phosphorus in the inflow.

station during a single winter sampling session conducted on 20 March 1978.

The role of spring runoff in enhancing nutrient levels of the inflow with respect to particulate forms becomes readily apparent by comparing Figures 9 and 10 with Figure 6. The highest concentrations of organic-N and organic + hydrolyzable-P in the inflow were achieved during spring runoff. As previously discussed, the turbidity in the Tongue River was partially due to suspended organics and suspended inorganic particulates, especially silt. Hynes (1970) stressed the role of runoff in supplying organic nitrogen to lotic systems and Sawyer (1947) found that drainage from agricultural lands contributed heavily to the organic nutrient loads carried by streams. The main land uses of the Tongue River drainage above the reservoir are agriculture and grazing. A strong correlation ($n = 29$, $r = 0.56$, $p = 0.02$) was found between high turbidity values and high concentrations of organic nitrogen at the Inflow station. In contrast, no statistically significant correlations existed between inflow turbidity values and inflow ammonia concentrations ($n = 29$) or between inflow turbidity values and inflow nitrate concentrations ($n = 29$).

Wright and Soltero (1973) and Rada (1974) found significant correlations between high turbidity and high orthophosphorus in the Big Horn and Missouri Rivers respectively. A base exchange equilibrium between water and suspended sediments with the concommitant release of orthophosphorus was hypothesized to explain this observation. This mechanism apparently did not operate in the inflow water of the Tongue River Reservoir as no statistically valid relationship ($n = 29$) was noted between inflow turbidity values and inflow orthophosphorus concentrations. However, a statistically significant correlation ($n = 29$, $r = 0.36$, $p = 0.04$) was observed between high turbidity values and high concentrations of org. + hydro.-P at the Inflow station. The mode of phosphate sorption onto silt has been discussed by Syers, *et al.* (1973) and the role of silt relative to phosphorus loading in rivers has been reviewed by Golterman (1973).

The anomolous peaks displayed by several parameters of Figures 9 and 10 on 6 August 1976 are ancillary to the previously described bridge construction.

Seasonal concentrations of inorganic and organic nitrogen in the outflowing water are depicted in Figure 11. The chemistry of the outflow water necessarily reflected biogeochemical processes occurring in the reservoir sediment, interstitial water, and bottom water due to deepwater withdrawal at the outflow gate. A roughly bimodal curve (excepting the NH_3-N concentration on 2 October 1976) was observed for ammonia and nitrate concentrations in the outflow water with seasonal peaks occurring beneath ice cover and in the early summer. The first peak was attributed in part to the autochthonous microbial decomposition of sedimented organic matter, but also to the high inflow concentration of inorganic nitrogen during the winter months (Figure 9). During the winter, low water temperatures suppressed the assimilation of nutrients by algae, and, in essence, incoming nutrients flowed through the reservoir. Factors contributing to the second peak included wind-generated agitation of the sediment during reduced reservoir stage (Figure 4) and accelerated bacterial decomposition in the presence of favorable water temperatures.

Figure 12 shows the seasonal variations in the concentrations of inorganic-P and org. + hydro.-P in the outflow. Redox conditions were never conducive to the release of mineralized phosphorus trapped in the sediment. The winter peak of orthophosphorus was attributed to a high winter inflow concnentration (Figure 10) flowing through the reservoir in the absence of appreciable planktonic mediation; summer peaks were probably caused by wind activity.

Evidence for the winter inorganic nutrient flow-through theory is gained from examination of the peaks of the winter inorganic phosphorus concentration in Figures 10 and 12. The hydraulic retention time through the winter months of 1975-76 was approximately 75 days (Table 2). In the absence of biological perturbation, the highest concentration of a given nutrient in the outflow should have been noted

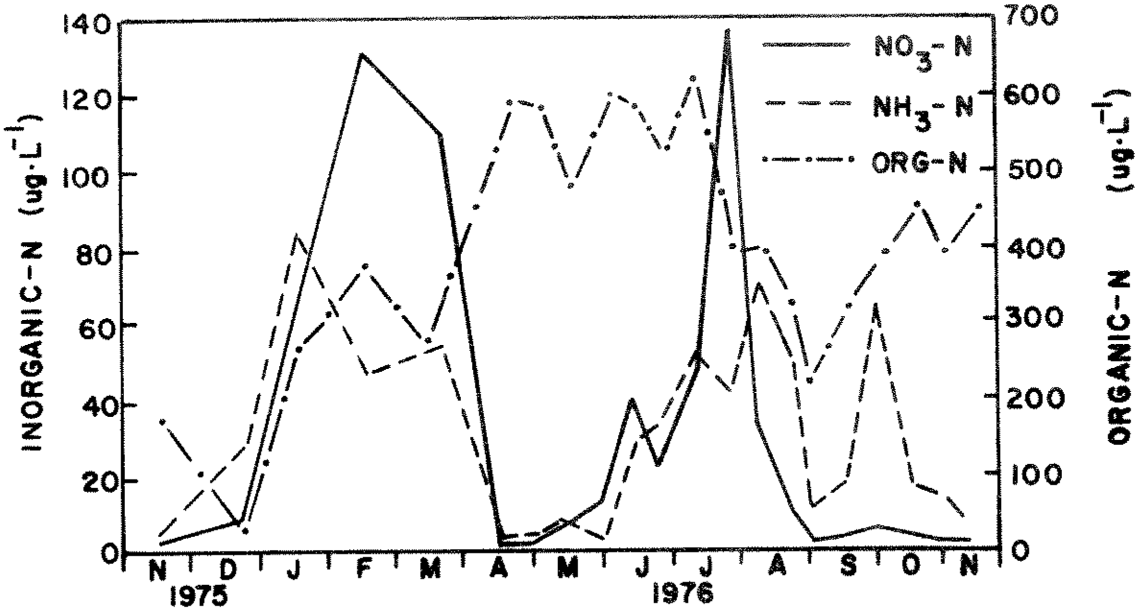

Figure 11. Seasonal concentrations of inorganic and organic nitrogen in the outflow.

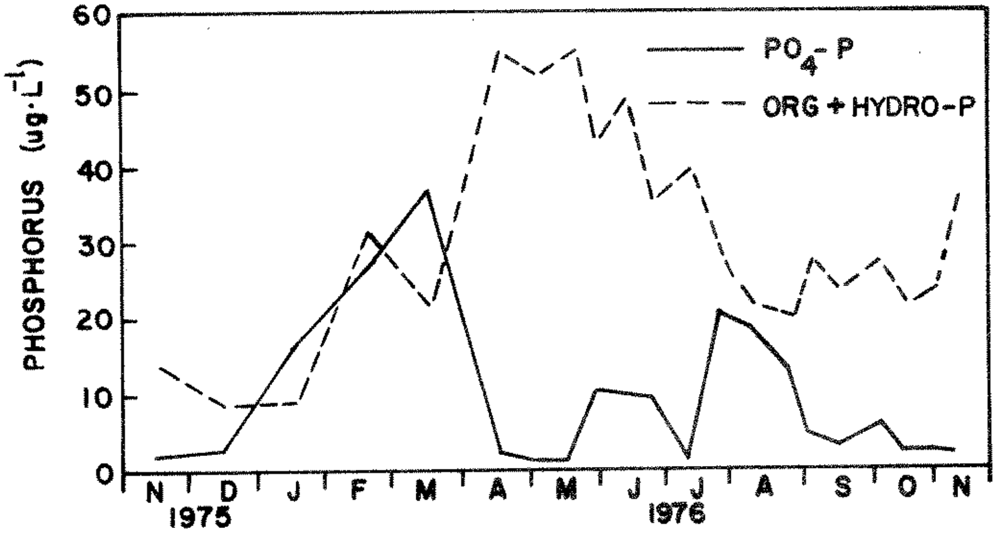

Figure 12. Seasonal concentrations of orthophosphorus and organic + hydrolyzable phosphorus in the outflow.

about 75 days following the highest recorded inflow concentration. Although a monthly sampling scheme through the winter precluded accurate detection of peak concentrations, the peak concentration of orthophosphorus in the outflow was detected 60 days following the inflow maximum (Figures 10 and 12). The outflow nitrate and ammonia maxima were noted 30 and 0 days, respectively, following the inflow peaks (Figures 9 and 11), which seemingly contradicts the theory. However, nitrogen was more mobile than phosphorus, especially in the sediment and interstitral water, which could easily explain these apparent anomalies.

Note the sharp drops encountered in Figures 11 and 12 relative to inorganic nutrient concentrations in April. The declines were attributed to increased phytoplankton activity in the reservoir upon ice-out. Considering the hydraulic retention time, it is obvious that the winter input of inorganic nutrients was not completely lost to algal consumption.

The concentrations of org. + hydro. - P and organic-N in the outflow water peaked during runoff (Figures 11 and 12) reflecting the short hydraulic retention time of the reservoir in the spring.

The nutrient load carried by a lotic system is contingent upon the volume of flow and the concentration of nutrients species in that flow. When seasonal nutrient loads borne by the reservoir's inflow and outflow waters were plotted (Figure 13), two major peaks were obtained relative to inorganic nitrogen and inorganic phosphorus. The first maxima of the inflow inorganic-N and inorganic-P plots temporally correspond with their counterparts in Figures 9 and 10. Although the flow rate was low in the inflowing water during the winter months, the concentrations of inorganic nutrient species in that flow were high enough that a significant fraction of the annual inorganic nutrient load delivered by the Tongue River to the reservoir was realized during that time. Winter loads of inorganic nitrogen and inorganic phosphorus were also high in the outflow water (Figures 13B and D) and reflect favorably upon the data presented in Figures 11 and 12 relative to winter concentrations of outflowing inorganic nutrients. Figures 13A-D lend support to the winter inorganic nutrient flow-through theory presented in the discussion of Figures 9-12.

The second peak of the inorganic nutrient loads of the inflow and outflow (Figure 13) occurred during the spring-summer months. The volume of inflowing water was greatest during spring runoff (Figure 3) but inorganic nutrient concentrations in that water were low (Figures 9 and 10). As a result the inorganic nutrient load carried by the inflowing river in the spring was of a lesser consequence when compared to the winter input (compare data from December-March with data from May-July in Figures 13A and C).

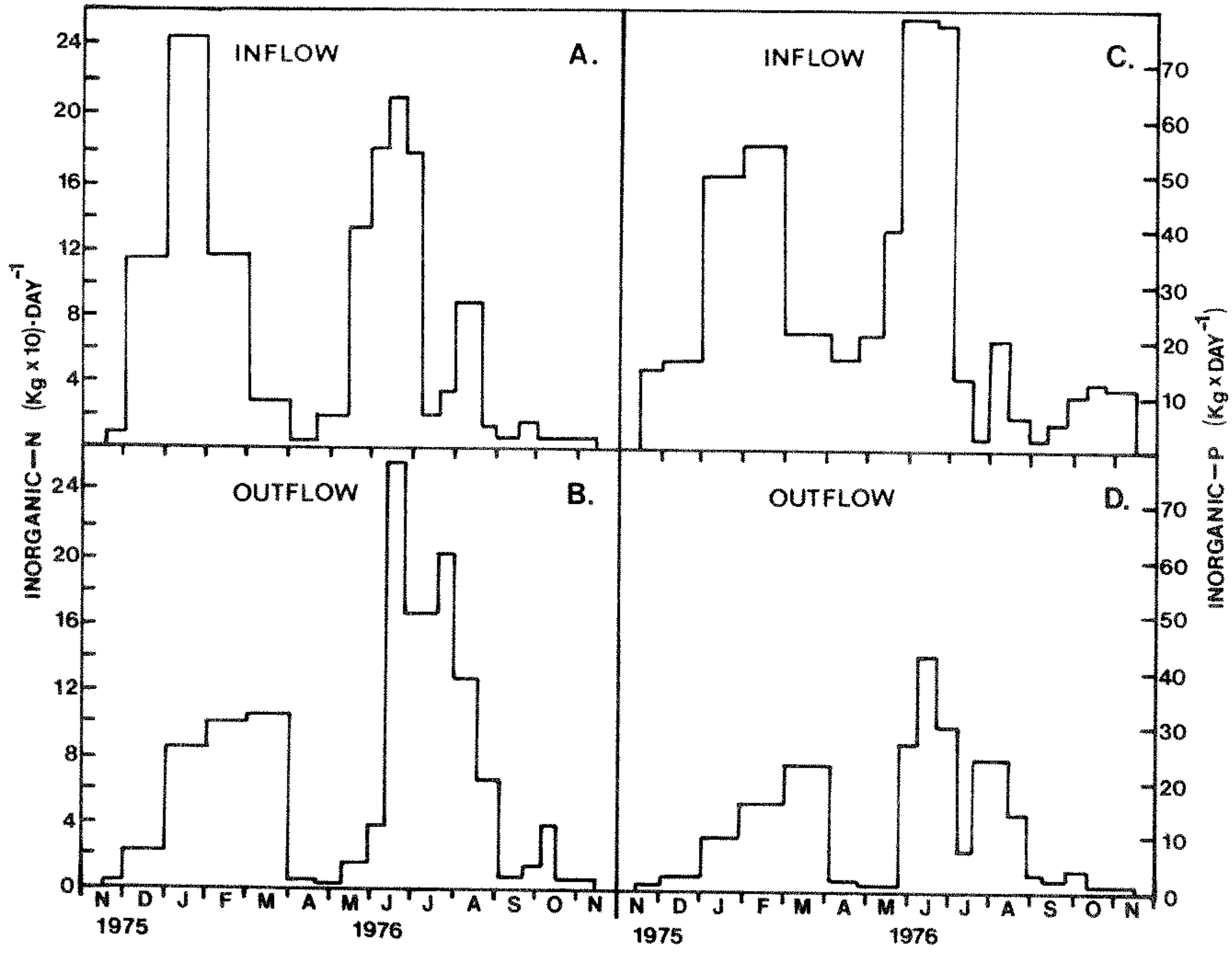

Figure 13. Seasonal loads of inorganic nitrogen (NO_2 + NO_3 + NH_3-N) and inorganic phosphorus (PO_4-P) carried by the inflow and outflow waters.

Due to the short hydraulic retention time characteristic of spring runoff, a "perturbed flow-through" situation is hypothesized for the inorganic nutrient load received by the reservoir from the river during that period. If an average retention time of 30 days (Table 2) is assumed for the inflow water received by the reservoir from 15 April - 30 June (spring runoff), the data in Figures 13A and B relative to inorganic nitrogen indicate that much of the import received by the reservoir during spring runoff was exported in the 15 May - 30 July period. The inorganic phosphorus data (Figure 13C and D) are not as good, but initiation of high output loading was 15 days subsequent to the initiation of high input loading. Certainly much biological cycling of the spring inorganic nutrient import occurred (as opposed to the winter import), but the deepwater outlet and turbulent conditions facilitated the rapid outflow of the inorganic nutrient input received during spring runoff.

Peak loading of organic-N and org. + hydro.-P in the inflowing water occurred during spring runoff (Figure 14A and C). On an annual basis, volumetric concentrations of organic-N and org. + hydro.-P were highest during spring runoff for previously discussed reasons. This, combined with the high volume of flow encountered during spring runoff, led to the observed pattern of loading.

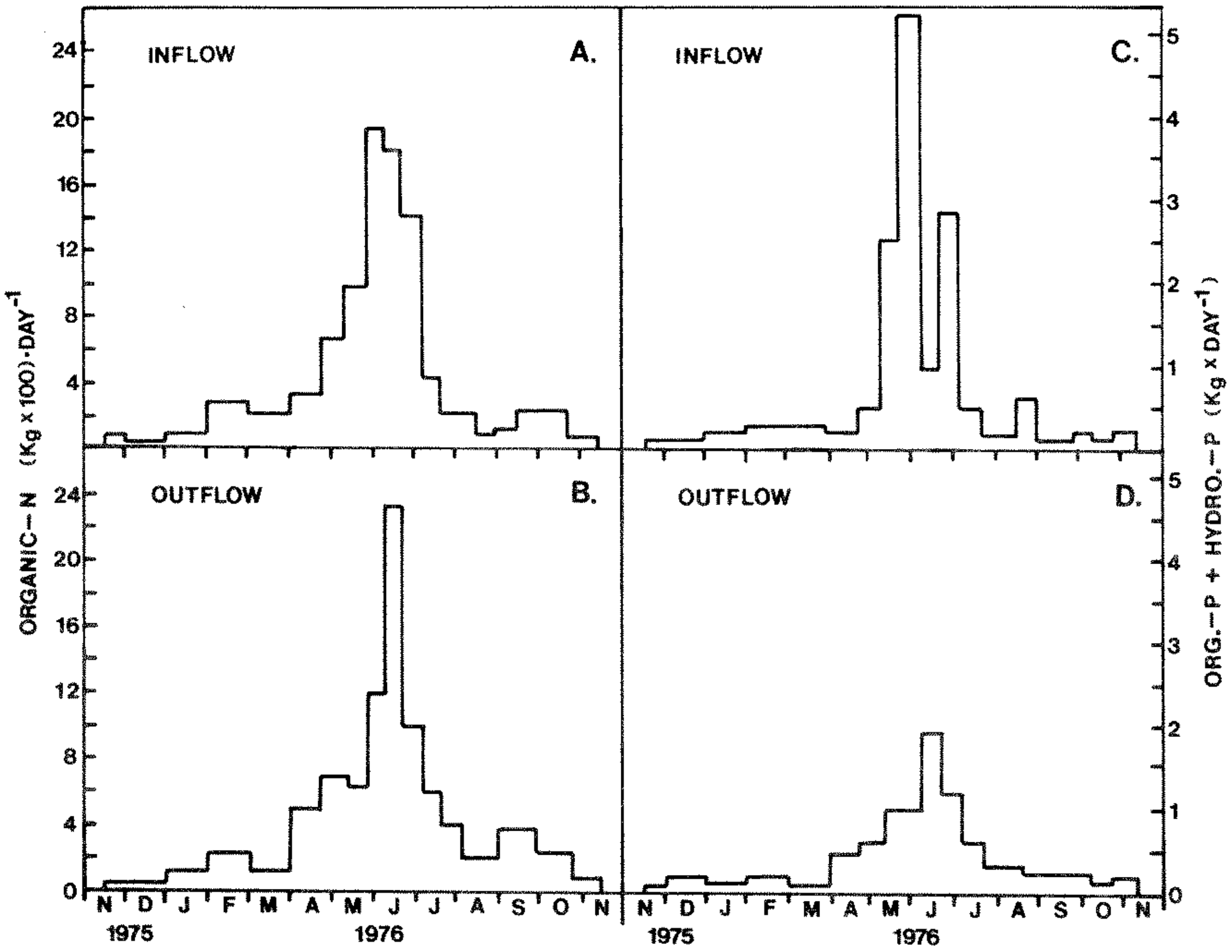

Figure 14. Seasonal loads of organic nitrogen and organic + hydrolyzable phosphorus carried by the inflow and outflow waters.

Further support for the "perturbed flow-through" theory in regard to the spring nutrient input to the reservoir is gained from an examination of the outflow loading patterns of organic-N and org. + hydro.-P, depicted in Figure 14B and D. The outflow loads of these nutrients were generally displaced by one month relative to the inflow loads. The importance of the seasonal loading patterns of inorganic and organic nutrients will be emphasized in a later discussion concerning the trophic status of the Tongue River Reservoir.

Tables 6 and 7 contain information concerning the annual (November 1975-November 1976) nutrient loads of the inflow and outflow of the Tongue River Reservoir. Of the inorganic nitrogen entering the reservoir, nearly all (91%) was exported. The same pattern was observed for organic nitrogen as 100% of the import was exported. This led to a net export of 99% of the total nitrogen entering the reservoir on an annual basis. The data are consistant with information presented by Wright (1967) suggesting that reservoirs with deep-water withdrawal tend to prevent the accumulation of nutrients, especially nitrogenous forms.

An examination of the species composition of nitrogen import and export in the Tongue River Reservoir (Table 6) reveals that 144% of the ammonia imported was exported while 68% of the nitrate imported was exported. Apparently ammonification outweighed nitrification

Table 6. A summary of the nitrogenous nutrient loads (kg of nitrogen) of the inflow and outflow waters from 15 November 1975 through 15 November 1976.

Location	NO_2-N	NO_3-N	NH_3-N	Inorganic-N	Organic-N	Total-N
Inflow	1607	18,270	8,430	28,307	153,800	182,107
Outflow	1221	12,361	12,121	25,703	153,873	179,576
Difference	+386	+5,909	-3,691	+2,604	-73	+2,531

Table 7. A summary of the phosphorus loads (kg of phosphorus) of the inflow and outflow waters from 15 November 1975 through 15 November 1976 and a comparison of molar ratios of some inorganic nutrients.

Location	Ortho-P	Organic + Hydrolyzable-P	Total-P	NO_2+NO_3:NH_3	Inorganic N:Ortho-P
Inflow	10,391	23,383	33,774	2.36	6.0
Outflow	3,990	13,196	17,186	1.12	14.4
Difference	+6,401	+10,187	+16,588		

within the reservoir's tropholytic zone prior to withdrawal through the deepwater gate. The synchronous conservation of oxidized species led to a drop in the molar ratio of oxidized inorganic nitrogen to reduced inorganic nitrogen from 2.36 at the inflow to 1.12 at the outflow (Table 7). This compares favorably with data presented by Pomeroy (1970) for similar situations.

The above discussion is a gross simplification and disregards various modes of nitrogen cycling within the reservoir. Denitrification has been implicated as a significant nitrogen sink in lakes (Chen, *et al.*, 1972) and nitrogen fixation by heterocystous blue-green algae has been demonstrated to contribute significantly to the nitrogen budget of some lakes (Kuznetsov, 1968). The potential allochthonous terrestrial and atmospheric nitrogen contributions to aquatic systems have been previously defined.

Of the various forms of phosphorus in the inflowing water, only 38% of the orthophosphorus and 56% of the org. + hydro.-P were discharged (Table 7). This resulted in a net discharge of 51% of the total phosphorus entering the reservoir. A simultaneous increase in the molar ratio of inorganic nitrogen to orthophosphorus from 6.0 at the inflow to 14.4 at the outflow was noted. This indicates that the Tongue River Reservoir acted as a phosphorus sink, with approximately half of the influent phosphorus being sedimented and retained within the reservoir. This is probably a reflection of the temporal intake of phosphorus by the reservoir (sorbed phosphorus associated with turbid spring runoff) and the absence of reducing conditions at the sediment-water interface of the reservoir's tropholytic water. The role of oxidation-reduction reactions in the phosphorus cycle of a lake has been well documented (Mortimer, 1941, 1942) and will be discussed in regard to the Tongue River Reservoir in the next section.

Reservoir

Turbidity and Light

The average turbidity profile of the Tongue River Reservoir is shown in Figure 15. Turbidity progressively decreased from the upper (Station 3) to the lower end of the reservoir (Station 1) on each sampling date. A Randomized Complete-Block Analysis of Variance revealed that the differences among average turbidity values of the reservoir stations were statistically significant ($n = 23$, $p < 0.01$). A Newman-Keul's test indicated that the mean turbidity value at Station 3 differed significantly ($p < 0.01$) from that of Station 1 or Station 2. However, the test failed to reject the null hypothesis when comparing the mean turbidity values of Stations 1 and 2. The data indicate that the bulk of the silt carried by the inflow settled out before it reached Station 2. This is consistant with information presented previously regarding seasonal and mean annual inflow and outflow turbidity values. Mean annual turbidity values were 20.3, 8.6 and 7.3 JTU measured at the three sampling sites from the upper to the lower end of the reservoir.

The mean extinction coefficients (Figure 16) decreased progressively from Station 3 to Station 1. This corresponded to increasing euphotic depth, decreasing turbidity, and deeper water. The mean euphotic depths progressing from Station 3 to Station 1 were 2.2, 3.4, and 3.9 meters.

The attenuation of light in the Tongue River Reservoir was due to the water itself, chlorophyll, suspended solids and color (resulting from dissolved organic particulate substances). These factors operated to varying degrees temporally and spatially within the reservoir. Owing to its proximity to the inflow, the extinction of light through the water column at Station 3 was due primarily to siltaceous matter, especially in the spring (Figure 6). During the period of low reservoir stage in the late summer, flocculent autochthous matter brought into suspension by wind activity also decreased light penetration through the water mass. Late in the growing season, phytoplankton blooms contributed sporadically to the attenuation of light incident upon the water's surface at Station 3.

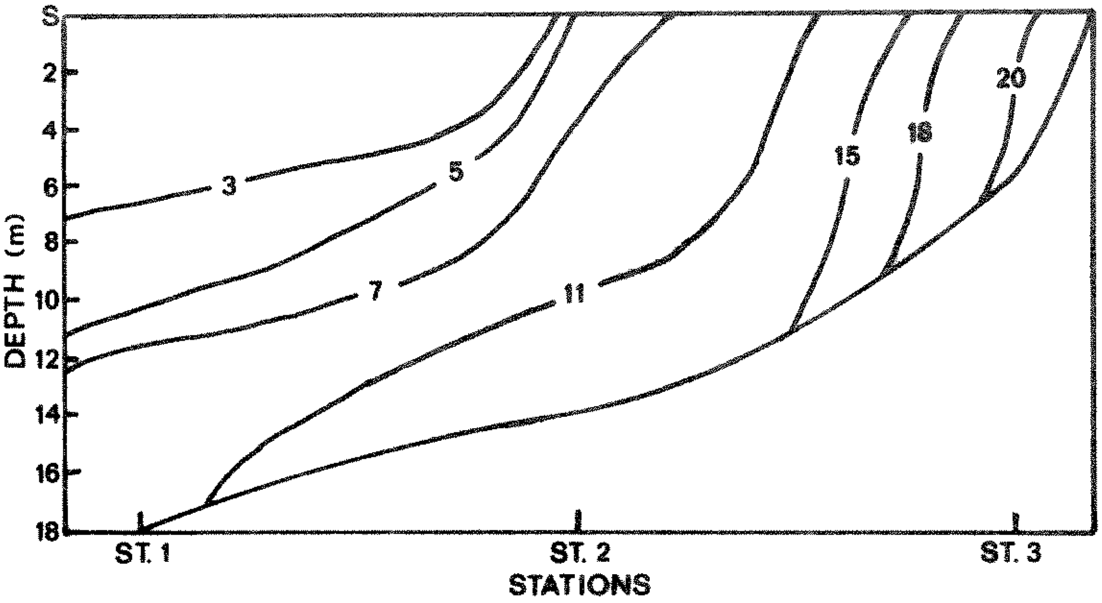

Figure 15. Average turbidity profiles in the Tongue River Reservoir from June 1975 through November 1976.

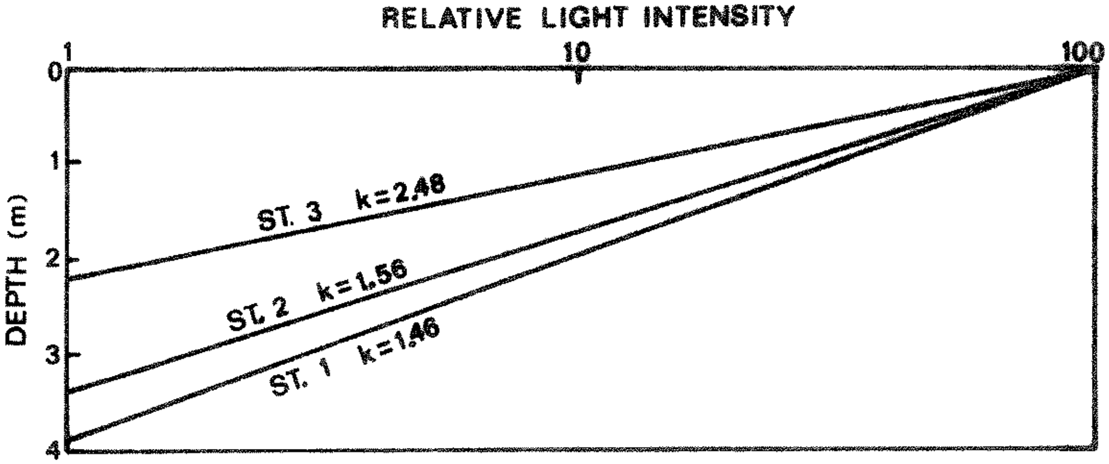

Figure 16. Penetration of total visible light, mean extinction coefficients (k) and euphotic zone depths in the Tongue River Reservoir from 6/75 - 11/76.

Since most of the influent siltaceous matter was settled out before reaching Station 2, dissolved organics, color, and, to a varying extent, chlorophyll influenced turbidity and penetration of total visible light at Stations 1 and 2. The role of biogenic absorption of photosynthetically active light has been discussed by Talling (1971). Steemann-Nielsen (1962) has shown that the absorption and attenuation of light per unit chlorophyll can be highly significant in productive lakes and varies among algal groups.

Table 8 presents the mean daily solar input, extinction coefficients, and euphotic depths averaged monthly for the entire reservoir over the entire study period. Values of mean daily incident radiation compare favorably with values observed by Odum (1971), Hutchinson (1957) and Kimball (1928) at roughly the same latitude. Generally the highest monthly mean extinction coefficients and lowest monthly mean euphotic depths were encountered beneath ice cover when ice and snow pack led to the rapid attenuation of the incident solar radiation.

Temperature and Specific Conductance

At the onset of ice formation in 1975, an orthograde temperature profile was evident in the Tongue River Reservoir (Figure 17). A relatively homothermous condition was maintained beneath the winter ice cover. The bottom water was slightly warmer than the surface water, due to the heat convected from the sediment and the anomolous density-

Table 8. Monthly mean daily solar radiation (langley·day^{-1}) from several sources compared with that recorded at the Tongue River Reservoir. Also, monthly mean extinction coefficients and monthly mean euphotic depths for the entire Tongue River Reservoir. All data were collected from June 1975 through November 1976.

	Tongue R. Res. (1975-76)	Northwest U.S. (Odum, 1971)	Europe (Lat. 47^{o}N) (Hutchinson, 1957)	Northwest U.S. Kimball (1928)	Mean Extinction Coefficient (m^{-1})	Mean Euphotic Depth (m)
January (11)[1]	224	150	167	183[2]	2.81	2.0
February (10)	257	225	278	286	2.20	2.0
March (11)	389	350	422	417	1.50	5.0
April (10)	450	475	575	538	1.69	5.0
May (18)	560	550	684	619	1.90	3.0
June (17)	643	600	729	641	1.80	3.8
July (19)	678	650	700	580	1.58	3.5
August (20)	581	550	614	503	1.70	3.1
September (17)	500	450	484	392	2.59	2.7
October (10)	305	275	332	272	1.64	3.3
November (17)	201	175	206	178	1.51	4.3
December (10)	138	125	144	148	3.30	2.0

[1] Number in parentheses indicates number of observations on which average incident solar input was calculated for Tongue River Reservoir.

[2] Values computed by averaging direct and diffuse solar radiation received in the presence and absence of cloud cover at 42^{o} N latitude and 48^{o} N latitude.

temperature relationship characteristic of water.

With increasing intensity of solar input, a clinograde temperature curve began to develop in late May and early June, but rapidly disintegrated in July as a result of wind-generated mixing and deepwater withdrawal. These factors, in combination with lowered reservoir stage, served to maintain a homothermous temperature profile in the Tongue River Reservoir into the fall months. Water temperatures increased almost uniformly throughout the summer to a maximum of 23°C in August, and decreased to 4°C by mid-November.

Seasonal conductivity isoclines for the Tongue River Reservoir are depicted in Figure 18. In the late fall of 1975, no vertical stratification in regard to specific conductance was evident. Beneath ice cover, a pronounced inverse clinograde conductivity profile was established. In the absence of wind-generated mixing beneath ice cover, suspended electrolytes with a specific gravity approximating unity could have settled to deeper water, affording the observed winter conductivity pattern. Convection between the sediment and its overlying water is often accompanied by the release of ions into the water (Mortimer, 1956). This mechanism could also have contributed to the observed increase in conductivity of the bottom water of the reservoir during the winter months. In March a decrease in specific conductance was noted immediately beneath the ice as a result of the

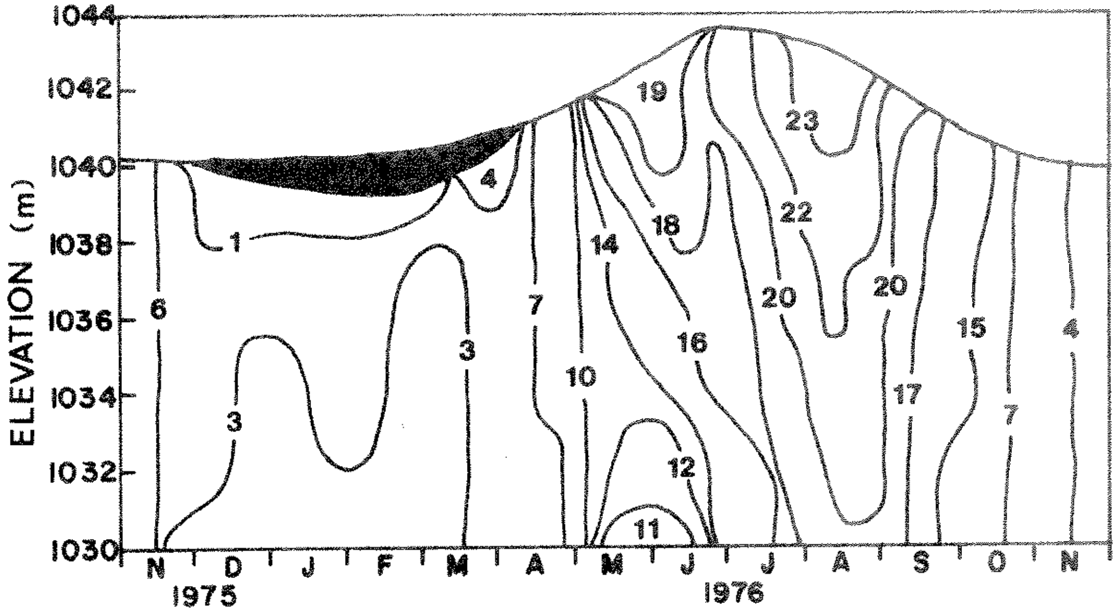

Figure 17. Seasonal isotherms (^{o}C) in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

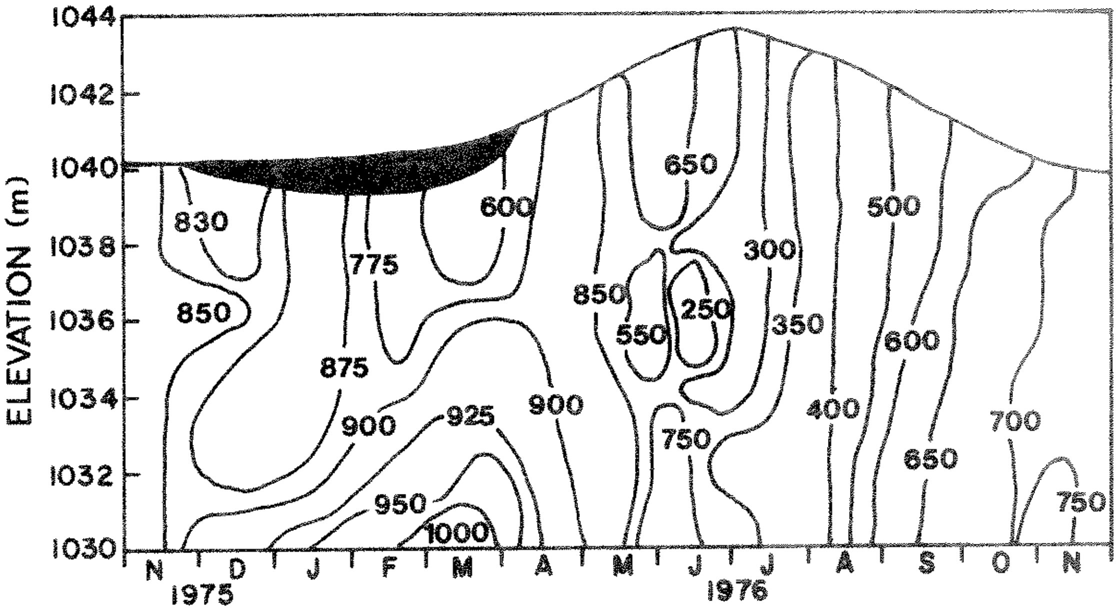

Figure 18. Seasonal conductivity ($\mu mhos \cdot cm^{-1}$) isoclines in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

melting process taking place above.

With the advent of open water and the initiation of the mixing process in April, an orthograde conductivity profile was established. This was replaced with a negative heterograde vertical pattern of specific conductance in late May and June concommitant to the formation of the clinograde temperature profile discussed above (compare Figures 17 and 18). This development was also ephemeral, and no vertical heterogeneity was noted in the conductivity profile of the Tongue River Reservoir from July through mid-November, 1976. Evaporative losses from both the reservoir and inflowing river led to progressively increasing values of specific conductance through the summer and fall months.

From late May through most of June, a density current through a thermal gradient was found in the reservoir. Evidence for this phenomenon was garnered from the aberrant conductivity profile observed in the Tongue River Reservoir at that time. Inflowing water with a temperature of 14-15°C and low specific conductance (250-550 μmhos$\cdot$cm^{-1}) moved through the reservoir midway in the water column (Figures 17 and 18). The existence of a density current at that time was further substantiated by chemical observations and turbidity profiles at all three reservoir stations. Upon dissolution of the vertical thermal gradient previously discussed, the framework for a density current was destroyed. Soltero, *et al.* (1974) presented similar proof of a density current in Bighorn Lake, a man-made impoundment with deepwater withdrawal in south-central Montana.

Dissolved Oxygen and pH

Figure 19 presents seasonal time-depth isopleths of the dissolved oxygen concentration in the Tongue River Reservoir. An orthograde dissolved oxygen profile was evident in the early fall, precipitated by low reservoir stage, wind activity, and a homothermous water mass. This condition was maintained under the quiescent conditions of ice cover and into the spring, largely as a result of deepwater withdrawal and the eddy effect of the inflowing water.

A distinctive summer hypolimnion and accompanying clinograde dissolved oxygen profile is characteristic of moderately productive dimictic lakes in temperate latitudes. Lack of sufficient depth and withdrawal characteristics of the Tongue River Reservoir precluded the establishment of the thermocline necessary for the development of a hypolimnion. However, the interplay between degradative oxidation of autochthonous organic detritus in the bottom water and moderately high planktonic production in the trophogenic water served to establish a clinograde summer oxygen profile in the absence of a hypolimnion.

Although no thermal resistance to mixing was established, microbial respiratory demand under favorable temperatures was sufficient to maintain low tropholytic dissolved oxygen concentrations for an extended period late in the summer. From mid-August through mid-September dissolved oxygen concentrations as low as 0.2 $mg{\cdot}L^{-1}$ were encountered in the bottom water. At the same time, dissolved oxygen concentrations reached 12 $mg{\cdot}L^{-1}$ at the surface, reflecting high rates of primary production.

With the advent of cooler weather in mid-October and the deceleration of biochemical processes, the clinograde oxygen profile was replaced with an orthograde curve which persisted through the balance of the open water period.

A lack of bathymetric data and the absence of a distinct hypolimnion in the Tongue River Reservoir vitiate the calculation of a relative areal oxygen deficit for the summer months. Seasonal oxygen saturation profiles for the Tongue River Reservoir are depicted in Figure 20 in an attempt to elucidate the relationship between photosynthetic production in the euphotic zone and heterotrophic decomposition of organic matter in the underlying aphotic zone.

The surface water was generally supersaturated with dissolved oxygen, indicating a high order of photosynthetic activity. Percent saturation could be interpreted as a rough estimate of relative primary productivity in the euphotic zone. In a similar manner, the degree of saturation in the bottom water could estimate the relative

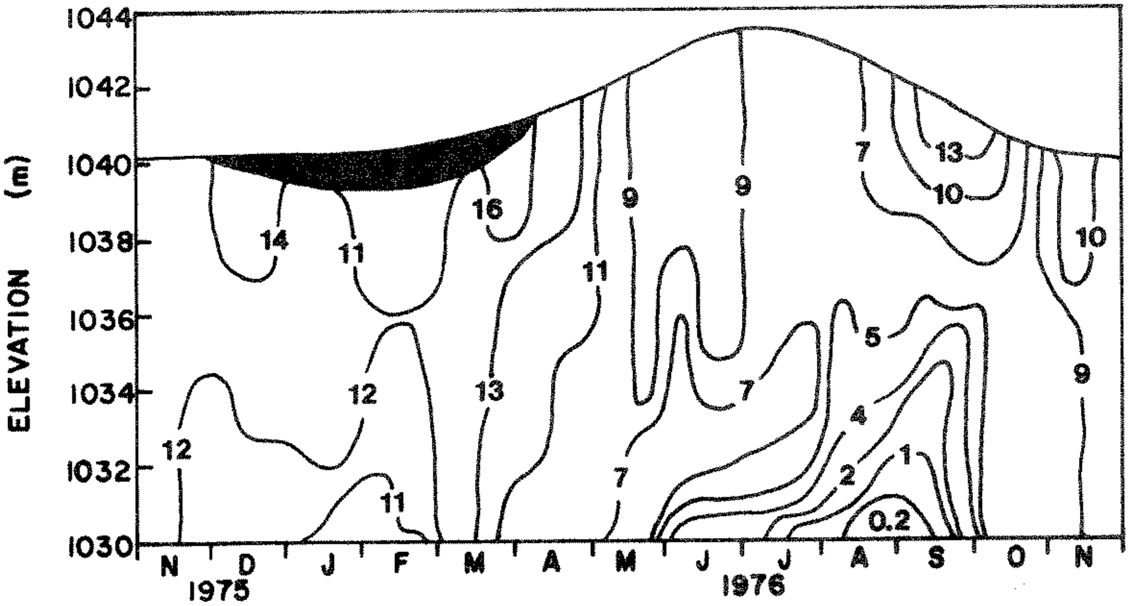

Figure 19. Seasonal dissolved oxygen ($mg \cdot L^{-1}$) isopleths in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

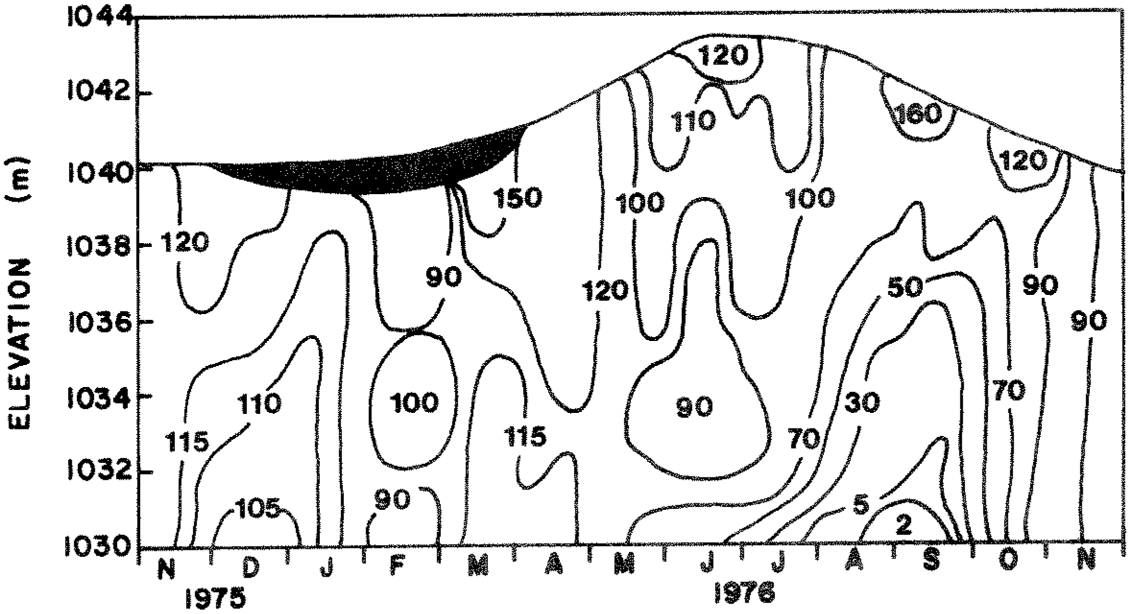

Figure 20. Seasonal oxygen saturation (%) isopleths in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

microbial respiratory demand and the difference between the two could give an indication of the relative magnitude of assimilative and degradative processes. Microbial activity appeared greatest in August and September, when high water temperature facilitated bacterial decomposition of organics. Assimilative photosynthetic processes were also maximized at that time. It should be noted that wind generated turbulence and diffusion could also affect the degree of oxygen saturation of the surface and bottom waters.

The pH of natural water is governed largely by the interaction of hydrogen ions arising from the dissociation of carbonic acid and from hydroxyl ions resulting from the hydrolysis of bicarbonate. Biological processes can influence the dynamic equilibrium within the CO_2-HCO_3^--$CO_3^=$ system and effect a shift in pH. The photosynthetic removal of carbon dioxide from bicarbonate can increase the pH of the surface water, while oxidative degradation of organics in the tropholytic water can increase the CO_2 concentration and decrease the pH of that water.

The seasonal pattern of pH change in the Tongue River Reservoir (Figure 21) was analogous to that of dissolved oxygen. An orthograde pH profile was maintained through the fall and winter of 1975-76. Photosynthetic activity just prior to ice-out in March increased the pH to 8.7 immediately beneath the ice. An orthograde pH profile persisted until the summer months. At that time, microbial activity in the bottom water combined with planktonic photosynthesis in the euphotic water effected a maximum pH differential of 1.5 pH units between the surface and bottom waters of the Tongue River Reservoir. Reduced biological activity, low reservoir stage, and continuously circulating water in the early fall of 1976 re-established vertical homogeneity with regard to pH in the Tongue River Reservoir.

The pH values encountered in the Tongue River Reservoir ranged from 7.5 - 9.0 with a mean value of 8.4 (Table 9). Calcareous hardwater lakes such as the Tongue River Reservoir are commonly well buffered at pH values above 8, which accounts for the narrow range of pH values observed.

Major Cations and Anions

The order of abundance of anions and cations in the reservoir was the same as that of the inflow and outflow (compare Tables 3 and 9). The Tongue River Reservoir could be characterized as a calcium-magnesium bicarbonate-sulfate type water. The semblance in chemical composition of the Tongue River above the reservoir and the Tongue River Reservoir reflects the source of inflowing surface water to the reservoir. It also indicates a similarity in geology, ion exchange reactions and weathering patterns of the drainage area above the reservoir and that of the reservoir itself.

Table 9. Ranges and averages of some chemical and physical parameters of the Tongue River Reservoir from November 1975 through November 1976.

Parameter	Station 1	Station 2	Station 3
Ca^{++} ($me \cdot L^{-1}$)	1.26-3.80	1.30-4.31	1.09-3.63
	2.86	2.79	2.81
Mg^{++} ($me \cdot L^{-1}$)	0.96-4.29	0.84-4.59	0.79-3.81
	2.99	2.91	2.99
Na^{+} ($me \cdot L^{-1}$)	0.34-1.92	0.29-22.7	0.28-2.02
	1.25	1.23	1.26
K^{+} ($me \cdot L^{-1}$)	0.04-0.16	0.04-0.16	0.03-0.13
	0.10	0.10	0.10
Total alkalinity ($me \cdot L^{-1}$)	1.92-4.70	1.68-5.62	1.56-4.71
	3.70	3.59	3.69
$SO_4^{=}$ ($me \cdot L^{-1}$)	0.81-5.18	0.66-6.24	0.54-5.27
	3.44	3.35	3.37
Cl^{-} ($me \cdot L^{-1}$)	0.03-0.12	0.03-0.13	0.03-0.12
	0.08	0.08	0.08
SiO_2 ($mg \cdot L^{-1}$)	1.4-11.8	1.1-10.0	2.5-13.0
	5.6	5.7	6.8
NH_3-N ($\mu g \cdot L^{-1}$)	0-236	0-142	0-220
	24	18	21
NO_3-N ($\mu g \cdot L^{-1}$)	0-204	0-187	0-47
	27	26	27
NO_2-N ($\mu g \cdot L^{-1}$)	0-20	0-10	0-10
	3	3	3
PO_4-P ($\mu g \cdot L^{-1}$)	0-100	0-77	0-27
	10	8	12
Total-P ($\mu g \cdot L^{-1}$)	16-144	10-109	37-260
	40	41	71
Spec. cond. ($\mu mhos \cdot cm^{-1}$@25°C)	246-929	221-1032	197-948
	660	645	654
pH	7.5-8.9	7.5-9.0	7.9-9.0
	8.4	8.5	8.4
Turbidity (JTU)	1.9-24	1.3-32	5.5-62
	7.3	8.6	20.3
Temperature (°C)	1.2-23.5	1.2-23.8	1.2-23.9
	10.6	10.9	11.4
Dissolved Oxygen ($mg \cdot L^{-1}$)	0.2-13.4	0.8-19.6	2.5-17.6
	8.5	9.3	10.1

Table 10 presents the average seasonal distribution of cations and anions in the Tongue River Reservoir. The temporal concentration of anions and cations followed a pattern similar to that of the inflow (Table 4). Millequivalent concentrations of cations and anions were high (9-10 $me \cdot L^{-1}$) in the fall and winter, decreased to a low of around 3 $me \cdot L^{-1}$ following spring runoff, and steadily increased through the late summer and fall. A one month lag period existed before the inflow of dilute runoff water in the spring was reflected in the millequivalent concentrations of ionic species in the reservoir. Owing to the length of mid-summer retention times (≃ 50 days), relatively dilute reservoir water persisted for some time after water more concentrated in ionic species flowed into the reservoir.

Figure 22 illustrates characteristic seasonal total alkalinity (predominately HCO_3^-) profiles for the Tongue River Reservoir. Winter profiles were orthograde (Figure 22-I) and negatively heterograde profiles (Figure 22-II) were obtained while the previously described density current flowed through the Tongue River Reservoir in the late spring and early summer. An inverse clinograde profile (Figure 22-III) was ordinarily found in the summer. As the rate of microbial decomposition increases in the tropholytic zone, the concentrations of carbon dioxide and especially bicarbonate show a marked increase (Wetzel, 1975). Bacterial activity in part probably accounted for the

Table 10. Average seasonal distribution of cations and anions ($me \cdot L^{-1}$) in the Tongue River Reservoir from November 1975 - November 1976.

Date	Ca^{++}	Mg^{++}	Na^{+}	K^{+}	Total Cations	Alk.	Cl^{-}	F^{-}	$SO_4^{=}$	Total Anions
15 Nov. 75	3.75	3.90	1.72	0.13	9.50	4.80	0.10	0.02	4.85	9.77
18 Dec.	3.72	3.95	1.66	0.14	9.47	4.89	0.09	0.01	4.80	9.79
17 Jan. 76	3.95	4.19	1.68	0.14	9.96	5.25	0.10	0.02	4.82	10.19
21 Feb.	4.01	3.95	1.62	0.14	9.72	5.22	0.12	0.02	4.44	9.80
22 Mar.	4.31	4.29	1.77	0.14	10.51	5.10	0.10	0.01	5.24	10.45
16 Apr.	3.65	3.95	1.73	0.12	9.45	4.64	0.12	0.02	4.95	9.73
1 May	3.72	4.03	1.90	0.12	9.77	4.56	0.11	0.02	4.97	9.66
14 May	3.74	4.10	1.85	0.15	9.84	4.44	0.14	0.02	5.15	9.75
29 May	2.64	2.76	1.13	0.09	6.62	3.18	0.08	0.02	3.02	6.30
10 June	1.76	1.43	0.65	0.06	3.90	2.21	0.04	0.02	1.53	3.80
25 June	1.46	1.15	0.42	0.05	3.08	1.92	0.04	0.01	1.16	3.13
8 July	1.64	1.35	0.48	0.05	3.52	2.29	0.03	0.01	1.26	3.59
22 July	1.65	1.38	0.51	0.05	3.59	2.21	0.03	0.01	1.31	3.56
6 Aug.	1.99	1.78	0.68	0.06	4.51	2.78	0.05	0.01	1.69	4.53
20 Aug.	2.20	2.00	0.83	0.08	5.11	2.89	0.06	0.01	2.01	4.97
2 Sept.	2.26	2.44	1.00	0.10	5.80	2.98	0.07	0.02	2.61	5.68
16 Sept.	2.35	2.88	1.29	0.12	6.64	3.19	0.08	0.02	3.54	6.83
2 Oct.	2.72	3.32	1.32	0.16	7.52	3.83	0.09	0.03	3.81	7.76
16 Oct.	2.83	3.19	1.37	0.12	7.51	3.76	0.10	0.02	3.64	7.52
30 Oct.	3.21	3.45	1.34	0.10	8.10	4.31	0.08	0.01	3.80	8.20
12 Nov.	3.07	3.18	1.36	0.11	7.72	3.99	0.08	0.03	3.76	7.86

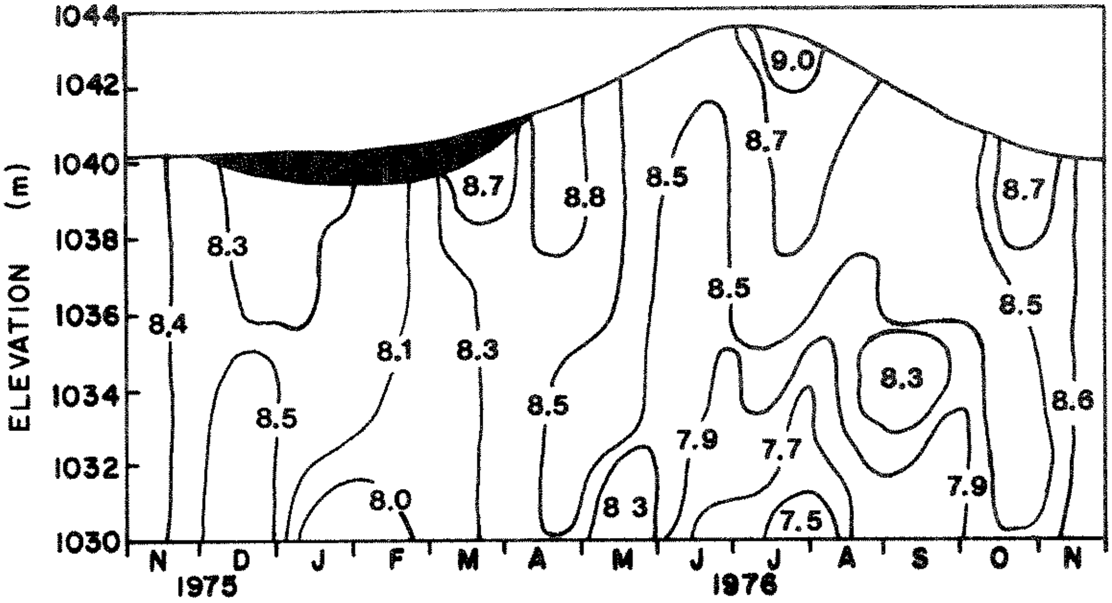

Figure 21. Seasonal pH isopleths in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

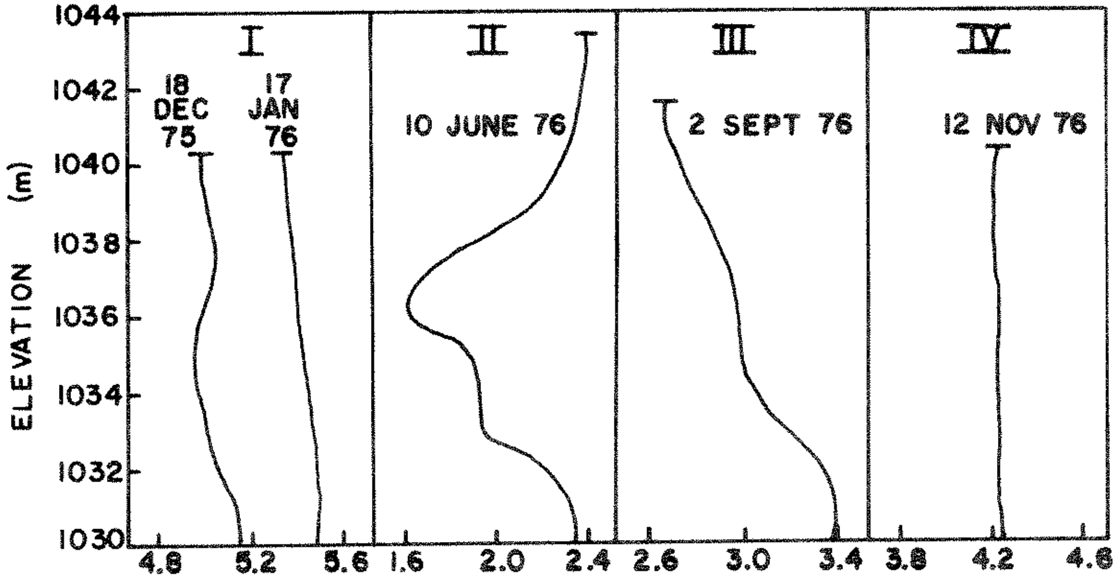

Figure 22. Total alkalinity ($me \cdot L^{-1}$) profiles in the Tongue River Reservoir on selected dates.

increase in total alkalinity of the bottom water of the Tongue River Reservoir during the summer. In the fall, orthograde total alkalinity profiles (Figure 22-IV) were encountered.

Nutrients

The ranges and means of various nutrient species in the Tongue River Reservoir are presented in Table 9.

Figures 23 and 24 depict the seasonal concentrations of NO_3-N and NH_3-N, respectively, in the Tongue River Reservoir. Nitrite concentrations are not shown as relatively small quantities of NO_2-N were detected. Nitrite is ephemeral in nature because it acts as an intermediate in nitrification, the oxidative conversion of ammonia and ammonium salts to nitrate (Brezonik, 1972). Figures 23 and 24 therefore represent the essence of the annual inorganic nitrogen regime in the Tongue River Reservoir.

The concentrations of various nitrogenous species in a lake system are a function of autochthonous nitrogen immobilization, mineralization, nitrification and denitrification, in conjunction with export and allochthonous input (Keeney, 1973). Beneath ice cover the lowest concentrations of nitrate were observed midway through the water column. Ammonia concentrations seemed to increase with depth during the winter. Just prior to ice-out, an intense phytoplankton bloom depleted the inorganic nitrogen in the euphotic zone

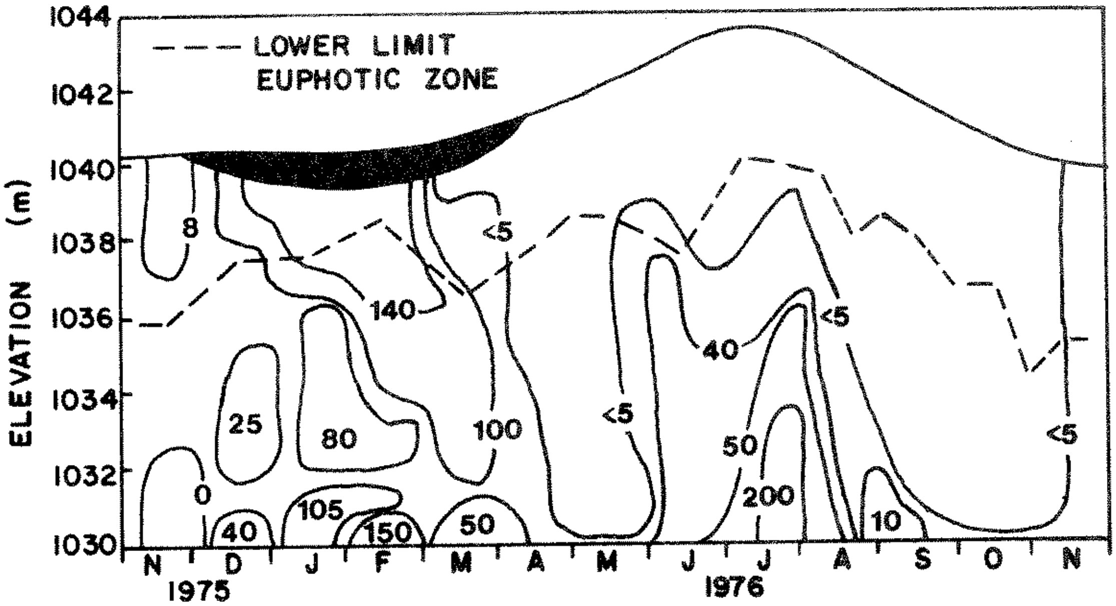

Figure 23. Seasonal NO_3-N ($\mu g \cdot L^{-1}$) isopleths in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

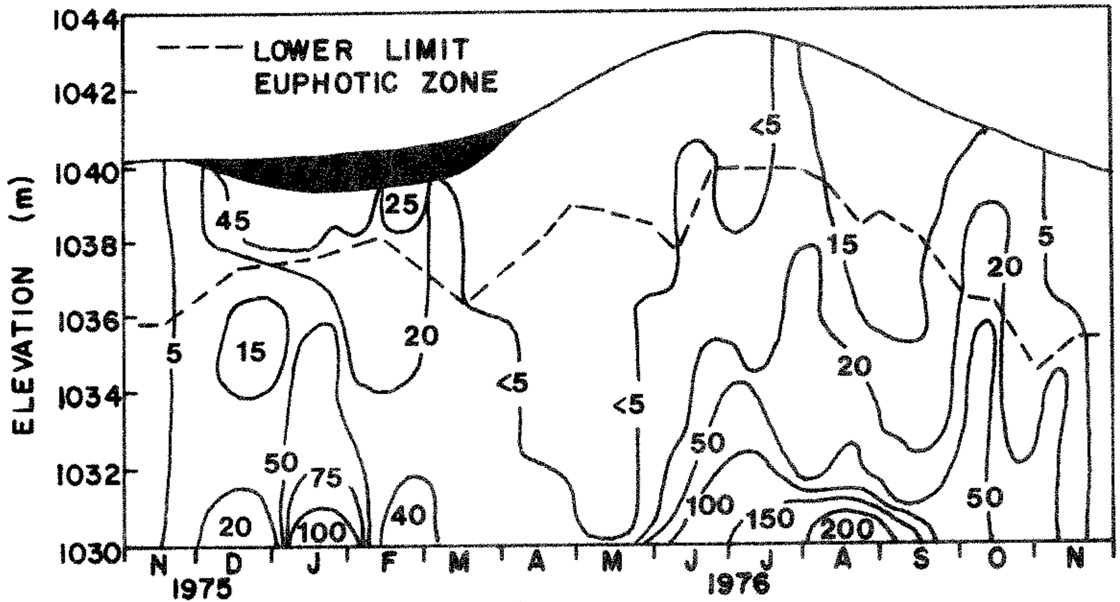

Figure 24. Seasonal NH_3-N ($\mu g \cdot L^{-1}$) isopleths in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

such that nitrate and ammonia concentrations declined to nearly indetectable levels. Very low concentrations of inorganic nitrogen were noted throughout the entire water column during April and May, presumably due to photoautotrophic uptake and a flushing effect, as high volumes of water were released through the deepwater gate during spring runoff.

Beginning in June, warmer water temperatures enhanced microbial decomposition of particulate organics, releasing NH_3-N into the bottom water via the process of ammonification (Figure 24). In the presence of adequate dissolved oxygen (Figure 19), much of the mineralized ammonia was nitrified and also appeared in the tropholytic water (Figure 23).

In contrast to the early summer, from mid-August through September there was a conspicuous absence of nitrate in the aphotic water (Figure 23). This was coincidental with reduced dissolved oxygen concentrations ($0.2\ mg \cdot L^{-1}$) in the deeper water (Figure 19) and increased ammonia concentrations (Figure 24). The diffusion rate of oxygen to bacteria becomes limiting at dissolved oxygen concentrations below $0.3\ mg \cdot L^{-1}$ and nitrification becomes greatly reduced or absent (Chen, *et al.*, 1972). This explains the pattern of occurrence of nitrogenous species observed in the tropholytic water of the Tongue River Reservoir in the late summer months.

In October and November reduced concentrations of nitrate and ammonia were observed in the aphotic water. With the onset of colder temperature, degradative processes decelerated and the release of inorganic nitrogen from the interstitial water declined.

Notably low concentrations of all major forms of inorganic nitrogen were encountered in the euphotic zone and, to a varying extent, the upper reaches of the aphotic zone, throughout the open water period (Figures 23 and 24). Constant planktonic assimilation obviously withdrew inorganic nutrients from the euphotic zone. Nutrients liberated from the interstitial water by bacterial activity migrated upward in the water column along a concentration gradient because of wind-generated agitation in the absence of a distinct thermocline. In this manner nutrients of the interstitial water were made available for the algal community. This hypothesis is offered to account for the reduced concentrations of ammonia and nitrate observed to varying depths in the aphotic zone and will be discussed relative to algal productivity in a later section.

In natural lakes there are commonly several different forms of inorganic nitrogen depending upon the redox state of the system. In contrast, the only significant form of inorganic phosphorus in lakewater is orthophosphate (PO_4-P). Because phosphorus readily chelates or complexes with metal ions and has a high affinity for adsorption onto clays and organic matter, as much as 90% of the phosphorus in natural water is associated with inorganic or humic particles (Wetzel, 1975). Allochthonous sources of phosphorus to lake systems include that derived from mechanical weathering of the earth's crust, input from influent water, and atmospheric contributions. Phosphorus solubilized from decomposition at depth and subsequently released from sediment comprises the phosphorus of autochthonous origin in aquatic systems.

Figures 25 and 26 illustrate the seasonal time-depth isopleths of orthophosphorus and total phosphorus, respectively, in the Tongue River Reservoir. Before the onset of ice formation in mid-November, orthophosphorus concentrations throughout the water column were at nearly indetectable levels ($<5\ \mu g \cdot L^{-1}$). During the winter months, concentrations of orthophosphorus were consistantly greater than the levels observed in November, and the lowest concentrations of PO_4 were often detected midway through the water column. In a series of papers, Groterud (1972a, 1972b, 1972c) stressed the importance of ice as a nutrient trap and subsequent source of inorganic nitrogen and phosphorus to aquatic systems upon thawing. The seemingly high levels of inorganic nutrients sometimes noted just beneath ice cover (Figures 23-25) could be due in part to this mechanism during the alternate periods of freezing and thawing common to southeastern

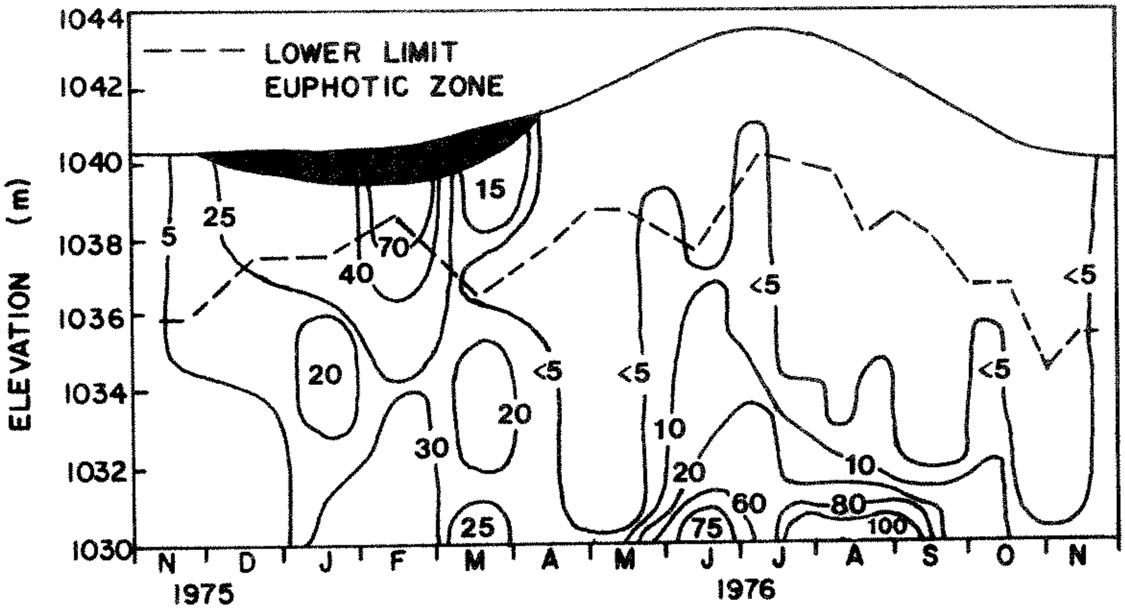

Figure 25. Seasonal PO$_4$-P ($\mu g \cdot L^{-1}$) isopleths in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

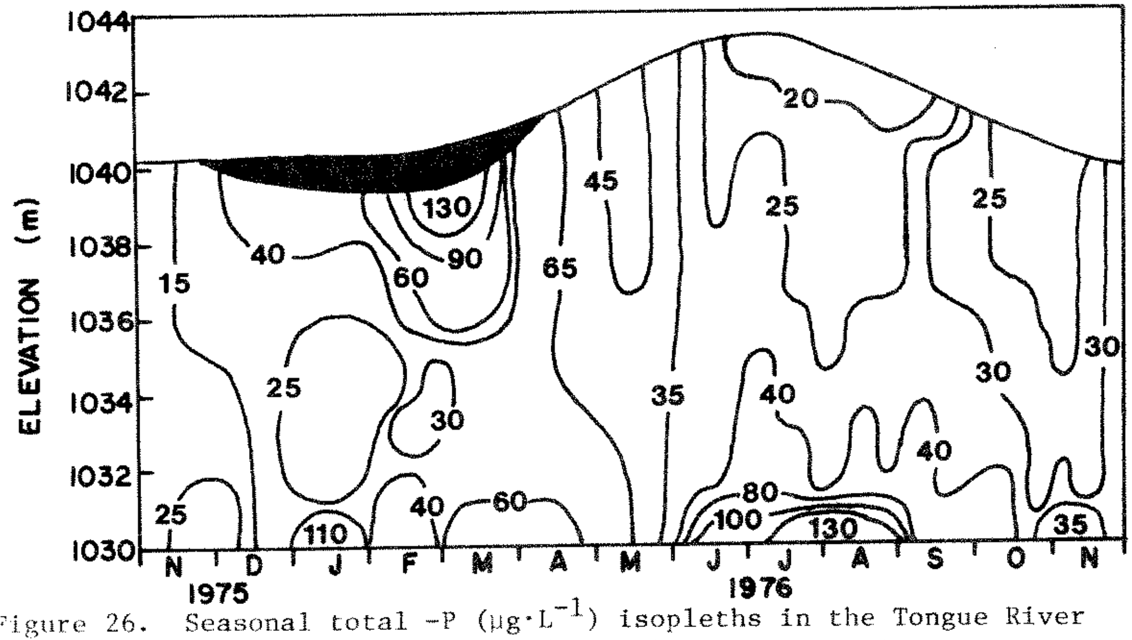

Figure 26. Seasonal total -P ($\mu g \cdot L^{-1}$) isopleths in the Tongue River Reservoir. Ice cover (opaque) drawn to scale.

Montana in the winter months. Very low concentrations of inorganic phosphorus were observed throughout the entire water column during April and May. This was attributed to the same factors mentioned earlier relative to spring concentrations of inorganic nitrogen.

Throughout the entire summer notably low ($<5\ \mu g \cdot L^{-1}$) concentrations of inorganic phosphorus were found in the trophogenic water and in all but the deepest reaches of the tropholytic zone. This corresponded with the period of time when water temperatures were conducive for both planktonic assimilative and microbial degradative processes. Phosphorus liberated from the sediment and interstitial water presumably was made available to the phytoplankton community in the same manner that inorganic nitrogen was made available, i.e. turbulent diffusion along a concentration gradient in the absence of a hypolimnetic barrier.

Total phosphorus behaved seasonally in a manner analagous to that of inorganic phosphorus (compare Figures 25 and 26). Factors influencing the temporal distribution of total phosphorus are congruent to those controlling orthophosphorus. The lowest concentration of total phosphorus ($\approx 20\ \mu g \cdot L^{-1}$) was recorded in the euphotic water during the summer, while the greatest concentration of total phosphorus ($130\ \mu g \cdot L^{-1}$) was reported just above the lake sediment during July and August.

In the absence of appreciable allochthonous input, internal cycling of phosphorus plays a dominant role in the overall phosphorus budget of lake systems. In any natural lake where the bottom water is well oxygenated year-round, little autochthonous cycling is expected and the distribution of phosphorus with depth is uniformly at a low level. Of the allochthonous input, most of the phosphorus entering a well oxygenated lake becomes organically bound into the planktonic and macrophytic biomass, and is temporarily cycled before reaching the lake sediment where it is decomposed and hydrolyzed (Sommers, *et al.*, 1970).

When exposed to adequate dissolved oxygen, the sediment surface of a lake exhibits a redox potential of about +500 mV. The deepwater sediment, however, is strongly reducing, showing a potential of around 0 mV. Iron in the oxidized state at the sediment-water interface exists as hydrated ferric phosphate (Hutchinson, 1957). Phosphorus may also be co-precipitated with manganese and carbonates (Harter, 1968). If turbulent transport maintains a dissolved oxygen concentration of about 0.5 $mg \cdot L^{-1}$ at the sediment-water interface, an oxidized microzone is established in the few millimeters of surface sediment permeated with oxygen. This will prevent the upward migration of phosphorus from the reduced sediment below (Wetzel, 1975). In eutrophic systems, a microzonal oxygen deficit may develop as the bottom water becomes anoxic during summer or winter stagnation. In this event, the integrity of the oxidized microzone is destroyed and the oxidized iron and manganese are reduced with the coincidental release of phosphorus (and other elements, depending upon the redox potential reached) into the hypolimnetic water mass. Drastic seasonal and spatial variations of phosphorus would then be expected.

From the above discussion and a cursory examination of Figures 25 and 26 it could be supposed that this stage was reached in the Tongue River Reservoir during the summer of 1976. Although concentrations of orthophosphate and total phosphorus increased markedly in the bottom water during July and August, other data (Figure 27-II) do not support the theory that the redox state of the reservoir's sediment-water interface favored the liberation of ferrous iron and phosphorus from the sediment in the summer of 1976. Dissolved oxygen concentrations (Figure 27-IIA) dipped to 0.2 $mg \cdot L^{-1}$ during August and September of 1976, a level at which ferric iron may start to become reduced. If reducing conditions were present at the sediment-water interface, marked increases in CO_2 and an attendant decrease in pH should have been noted in the bottom water, resulting from increased microbial respiratory demands as the system shifted toward anoxia. No major changes in pH or free CO_2 concentrations were observed during the summer of 1976.

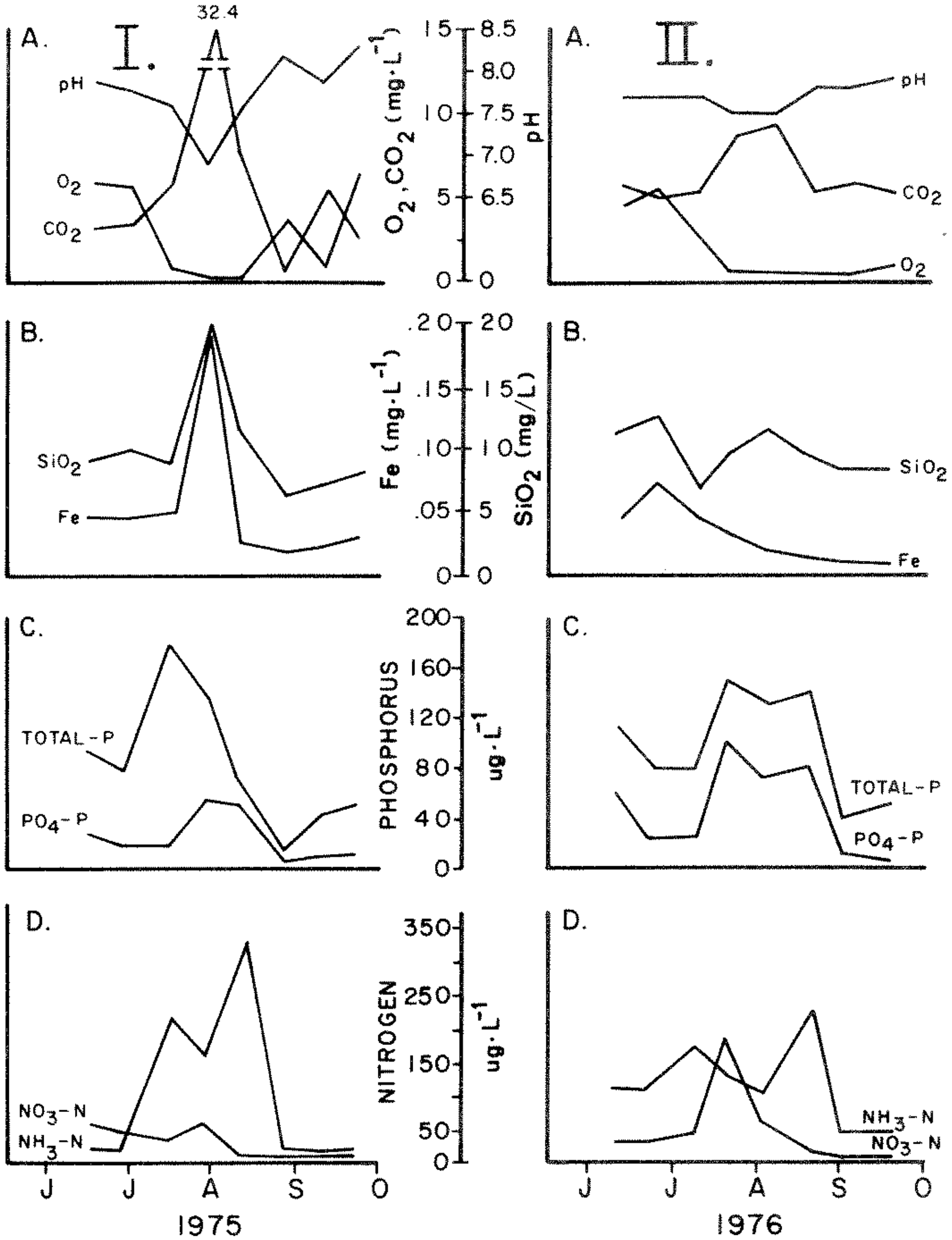

Figure 27. Variation in chemical composition of water overlying sediment in the Tongue River Reservoir during the summers of 1975 and 1976.

Oxidized iron in natural water (ferric hydroxide) is only sparingly soluble (0.018 $mg \cdot L^{-1}$), although iron in the ferrous (reduced) state is very soluble. The 1976 summer concentration of soluble iron was often greater than the calculated solubility equilibrium (Figure 27-IIB). This was attributed to minute suspended organic iron complexes seemingly not trapped by the 0.45 μ glass fiber filter. Note, however, that no marked increase in soluble iron was observed, as would have been the case if reducing conditions had been attained.

Mortimer (1941) and Hutchinson (1957) believed that the silica cycle was intimately associated with the iron cycle in the respect that SiO_2 was released from a ferri-silicic-humate complex in the sediment upon solution of iron as Fe^{++}. Golterman (1975) discounted that theory and felt that silicates were derived from the accelerated mineralization of diatoms following a typical spring diatom pulse. In any event, it is agreed that silica frequently appears in high concentrations in the hypolimnion of many eutrophic lakes and that the silicate-time curve closely mirrors the iron and phosphate curves. No noteworthy increase in the silica concentration was found in the bottom water at any time in the summer of 1976 (Figure 27-IIB).

Ancillary to some of the previously mentioned changes to be expected in a reducing environment is the increase in concentration of inorganic nitrogenous species, particularly ammonia, at the sediment-water interface. These increases are brought about by accelerated microbial activity and by the decreased adsorptive capacity of the sediment surface. Increases in both the ammonia and the nitrate concentrations were observed to some degree in the course of the summer of 1976 (Figure 27-IID).

The bulk of the chemical evidence (pH, CO_2, O_2, Fe, SiO_2) leads to the conclusion that reducing conditions were not present in the summer of 1976 to the extent that ferric iron was reduced (effecting the release of phosphorus) at the sediment surface of the Tongue River Reservoir. The lone evidence in support of reducing conditions is the increased level of phosphorus detected in July and August (Figure 27-IIC). Zicker, *et al.* (1956) found that under anaerobic conditions the rate of phosphorus release from the sediment increased two-fold if the sediment was disturbed by agitation from turbulence. Wetzel (1975) felt that agitation of this nature in shallow lakes was an important source of extractable phosphorus for algal consumption. The severe windstorms that frequently occur in the Sheridan, Wyoming area, the homothermous water mass, and the slight current in the reservoir could have created sufficient turbulence to catalyze the release of some phosphorus from the thin oxidized layer.

The preceding argument is in contrast to the data presented in Figure 27-I for the summer of 1975. In this case the evidence weighs heavily in favor of the occurrence of reducing conditions to the extent of the release of ferrous iron on 28 July 1975. Marked increases in free CO_2, SiO_2, soluble Fe, and nutrient concentrations were observed on that date as well as a decrease in the dissolved oxygen concentration and pH value.

The Tongue River Reservoir reached Phase II of Hutchinson's (1957) scheme of summer stagnation in 1976. The reservoir proceeded no further because continued circulation provided sufficient oxygen to the bottom water to prevent the reduction of hydrated ferric phosphate bound in the sediment surface. In 1975 conditions were reduced enough to liberate ferrous iron but not sulfide from the Tongue Reservoir sediment, hence Phase III of Hutchinson's scheme was realized. Based upon further observation (Whalen and Leathe, unpublished data), the data presented for 1975 probably approximates the norm more closely than that of 1976 and the reservoir typically reaches Phase III of summer stagnation. Despite the lack of a hypolimnetic barrier to prevent the diffusion of oxygen to the sediment surface, the rate of metabolic oxygen removal in the bottom water is apparently sufficient to create a summer oxygen deficit at the sediment-water interface.

Figure 28 illustrates the seasonal storage pattern of inorganic nitrogen, orthophosphorus and org. + hydro.-P in the Tongue River Reservoir. The concentration of nutrients in the reservoir water was

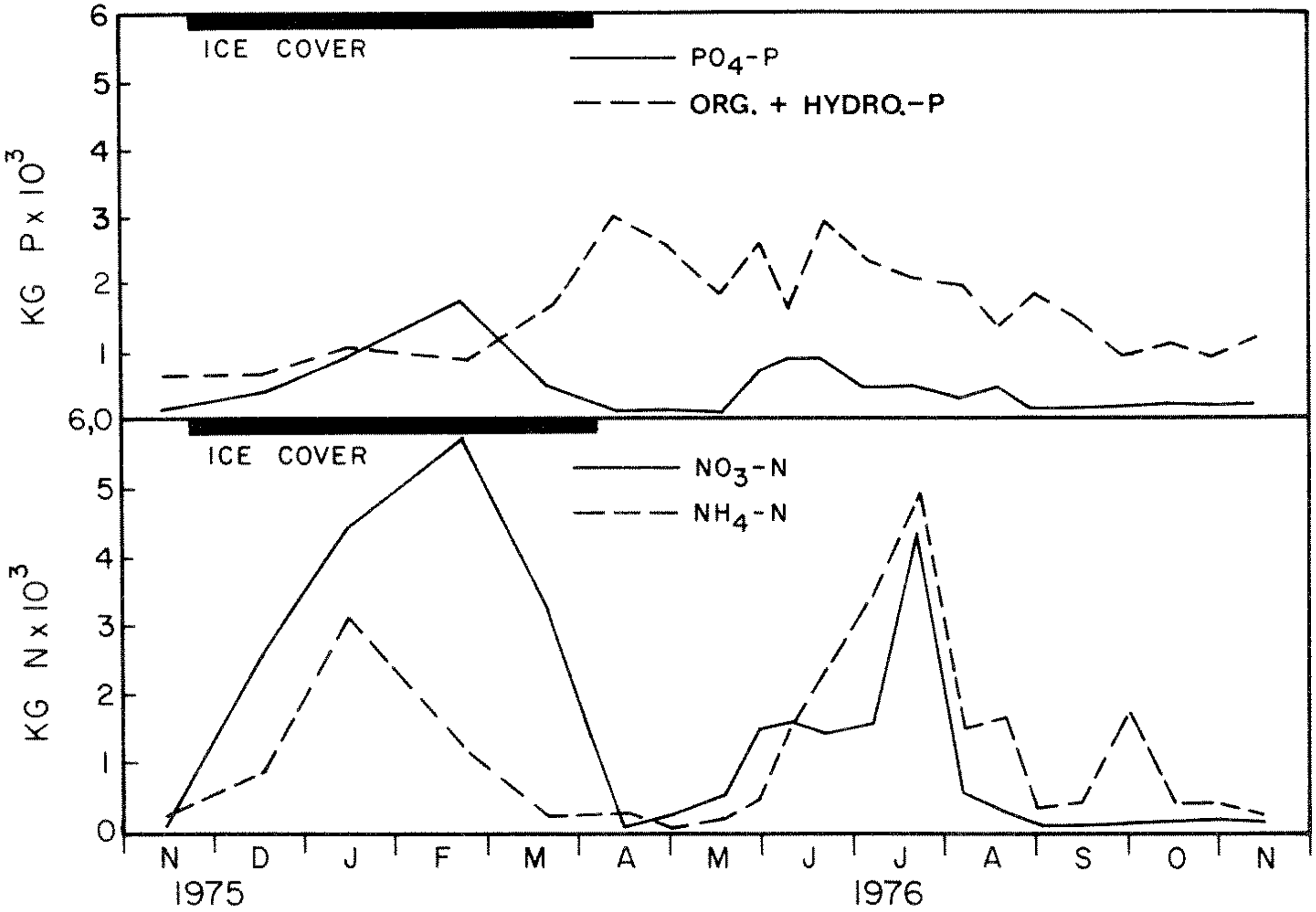

Figure 28. Seasonal storage of inorganic nitrogen, orthophosphorus and organic + hydrolyzable phosphorus in the Tongue River Reservoir.

dependent to varying degrees seasonally on physical, biological, and biogeochemical processes. The seasonal storage of inorganic phosphorus and inorganic nitrogen within the reservoir was greatest in the winter months. This again reflects upon the flow-through situation illustrated in Figures 13A and C relative to high winter loads of inorganic nutrients carried by the inflowing Tongue River.

The seasonal storage of inorganic nitrogen and inorganic phosphorus in the Tongue River Reservoir declined during April and May. On a yearly basis, minimal loading of inorganic nitrogen as well as depressed loading of orthophosphorus in the inflowing Tongue River were observed during April and May (Figure 13). However, the reservoir acted in a conservative nature toward inorganic nutrients in the early spring as small quantities were exported (Figure 13). With the advent of open water, the freely mixing inorganic nutrients of the Tongue River Reservoir were apparently incorporated into the algal biomass.

Maximum reservoir stage (Figure 4) and greatly increased phosphorus loading in the inflow compared to the outflow (Figure 13) accounted for the increased storage of PO_4 in the reservoir from mid-May through June (Figure 28). Greater export relative to import caused a decline in PO_4-P storage in July when data from June and July were compared. Although this import-export pattern persisted in August, storage levels stabilized as a result of the previously discussed migration of PO_4-P from the sediment. Low import, less export (Figure 13), reduced reservoir stage (Figure 4), and biological processes interacted to stabilize the seasonal storage or inorganic phosphorus at low levels from September through the balance of the growing season (Figure 28).

From June through November the export of inorganic nitrogen from the reservoir approximated the import (Figure 13). However, seasonal peaks and declines of inorganic nitrogenous parameters were noted throughout the summer (Figure 28). These were attributed to the interplay between biologically mediated assimilation and degradation combined with the influence of reservoir stage.

Phytoplankton Standing Crop and Chlorophyll *a*

General Taxonomic Analysis. Six classes of algae were identified in samples collected from the Tongue River Reservoir during 1975 and 1976: Bacillariophyceae (diatoms), Chlorophyceae (green algae), Cryptophyceae (cryptomonads), Dinophyceae (dinoflagellates), Euglenophyceae (euglenoids) and Myxophyceae (blue-green algae).

Within these groups 125 species of algae were identified and an additional 20 phytoplankton were taken to the generic level. Class Bacillariophyceae was the most diverse with 81 representative species. This group was followed by Class Chlorophyceae in which 28 species were identified. Six species of Class Myxophyceae were observed, while Class Euglenophyceae was represented by four species. Classes Cryptophyceae and Dinophyceae were the least diverse, each containing three specific representatives. Table 11 contains a list of all phytoplankton species observed in the Tongue River Reservoir during this study.

Horizontal Distribution. The seasonal variations in the total volumetric phytoplankton standing crop ($mm^3 \cdot L^{-1}$) and chlorophyll *a* concentration ($\mu g \cdot L^{-1}$) at all three Tongue River Reservoir stations are depicted in Figure 29. Except for a brief period immediately following ice-out, the total volumetric phytoplankton standing crop at Stations 1 and 2 followed the same trend during the 1976 growing season (ice-free period). However, the standing crop was generally smaller at Station 1. Oscillations of a far greater magnitude were encountered at Station 3 than at Stations 1 and 2 during August and September. An intense bloom of *Aphanizomenon flos-aqua* was noted at Station 3 on 16 September, which limited the euphotic zone to the upper 0.5 m of the water column. During the growing season the mean volumetric algal standing crops were 2.88, 3.53 and 8.94 $mm^3 \cdot L^{-1}$ at Stations 1, 2, and 3. The average for Station 3 was calculated exclusive of the anomalous observation on 16 September, which would have raised the mean to 39.60 $mm^3 \cdot L^{-1}$ if included. Statistical analysis (Randomized Complete-Block ANOV) revealed that the differences among

Table 11. Phytoplankton species observed at all sampling stations in the Tongue River Reservoir from June 1975 through November 1976 with the corresponding mean volume of each organism.

DIVISION Class *Species*	MEAN VOLUME (μm^3)
DIVISION CHLOROPHYTA (Green Algae)	
Class Chlorophyta	
Ankistrodesmus braunii Naegeli	861
A. falcatus (Corda) Ralfs	94
A. falcatus var. *mirabilis* West & West	773
Botryococcus braunii Kuetzing (colony)	2,453
Carteria sp. Diesing	268
Chlamydomonas sp. Ehrenberg	112
Closterium gracile de Brébisson	5,652
Coelastrum microsporum Naegeli *in* A. Braun (colony)	4,808
Cosmarium formulosum Taft	5,298
C. subcostatum Nordstedt	3,679
C. sp. unk. 1	680
Dictyosphaerium pulchellum Wood (colony)	1,308
Eudorina elegans Ehr. (colony)	18,608
Gloeocystis vesiculosa Naeg. (colony)	410
Oocystis gigas Archer (single cell)	4,710
O. pusilla Hans. (single cell)	98
O. sp. unk. 1 (single cell)	1,130
Pandorina morum (Mueller) Bory (colony)	1,766
Pediastrum boryanum (Turpin) Meneghini (colony)	8,352
P. boryanum var. *longicorne* Raciborski (colony)	8,352
P. duplex Meyen (colony)	12,262
P. duplex var. *reticulatum* Lagerheim (colony)	12,262
Phacotus lenticularis Ehr. Stein	549
Planktosphaeria gelatinosa G. M. Smith (colony)	1,595
Scenedesmus arcuatus Lemm. (colony)	565
S. dimorphus (Turp.) Kutzing (colony)	339
S. quadricauda (Turp.) de Bréb. (colony)	151
Schroederia setigera Schroeder	63
Sphaerocystis schroeteri Chodat (colony)	4,287
Spirogyra sp. unk. 1 Link (1 mm)	22,625

Table 11. Continued.

DIVISION Class *Species*	MEAN VOLUME (μm^3)
Chlorophyta (continued)	
Staurastrum paradoxum Meyen	7,480
Tetraedron sp. unk. 1 Kutz	360
T. sp. unk. 2	56
Ulothrix subtillissima Rabenhorst (1 mm)	12,560
Chlorophyta sp. unk. 1	35
Chlorophyta sp. unk. 2	60
Chlorophyta sp. unk. 3	85
Chlorophyta sp. unk. 4	25
DIVISION EUGLENOPHYTA (Euglenoids)	
Class Euglenophyceae	
Euglena elastica Prescott	3,925
E. sp. unk. 1	2,055
Trachelemonas hispida (Perry) Stein	4,186
T. robusta Swirenko	2,571
T. volvocina Ehr.	1,766
T. sp. unk. 1	1,766
DIVISION CHRYSOPHYTA (Yellow-green Algae)	
Class Bacillariophyceae	
Achnanthes lanceolata (Bréb.) Grunow	523
Amphora ovalis Kutz	502
A. ovalis var. *pedicularis* Kutz	340
A. veneta Kutz	370
Asterionella formosa Hassall (single cell)	188
Bacillaria paradoxa Gmel	-
Cocconeis diminuta Pant.	55
C. placentula Ehr.	569
C. placentula var. *euglypta* (Ehr.) Cleve	569
C. placentula var. *lineata* (Ehr.) Cl.	569

Table 11. Continued.

DIVISION Class *Species*	MEAN VOLUME (μm^3)
Bacillariophyceae (continued)	
Cyclotella comta (Ehr.) Kutz.	529
C. meneghiniana Kutz.	1,256
C. michiganiana (?)	151
C. sp. unk. 1	57
C. sp. unk . 2	48
Cymatopleura solea (Bréb.) W. Sm.	3,768
C. solea var. *regula* (Ehr.) Grun.	2,260
Cymbella affinis Kutz	1,850
C. cistula (Hemprich) Grun.	6,800
C. laevis Naeg.	630
C. prostrata (Berkeley) Cl.	788
C. turgida (Greg.) Cl.	488
C. ventricosa Kutz	261
C. sp. unk. 1	125
Denticula elegans Kutz.	235
Diatoma elongatum Agard.	490
D. hemiale Roth.	1,219
D. hemiale var. *mesodon* (Ehr.) Grun.	308
D. vulgare Bory	1,450
Diploneis puella (Schumann) Cl.	208
Epithemia sorex Kutz.	840
E. turgida Kutz.	3,225
Eutonia pectinalis var. *minor* (Kutz.) Rab.	700
Fragilaria brevistriata Grun.	200
F. capunica Desmazieres	212
F. capunica var. *mesolepta* Grun.	212
F. construens (Ehr.) Grun.	300
F. crotenensis Kitton (1 mm)	63,000
F. leptostauron (Ehr.) Hustedt	320
F. vaucheriae (Kutz) Peters	125
Gomphonema angustatum (Kutz) Rab.	452
G. angustatum var. *producta* Grun.	430
G. constrictum Ehr.	1,450
G. constrictum var. *capitata* (Ehr.) Cl.	1,450

Table 11. Continued.

DIVISION Class *Species*	MEAN VOLUME (μm^3)
Bacillariophyceae (continued)	
G. olivaceum (Lyngb.) Kutz	348
G. parvulum (Kutz) Grun.	348
G. sp. unk. 1	210
Gyrosigma acuminatum Kutz.	5,735
G. spencerii (W. Smith) Cl.	5,735
Hannaea arcus (Ehr.) Patrick	1,501
Melosira granulata (Ehr.) Ralfs (1 mm)	19,625
M. varians Agarhd (1 mm.)	2,500,000
Meridion circulare Ag.	490
Navicula accomoda Hust.	1,043
N. anglica Ralfs	153
N. arvensis Hust.	69
N. capitata Eht.	169
N. cryptocephala Kutz	263
N. cryptocephala var. *veneta* (Kutz) Grun.	188
N. cuspidata Kutz.	4,710
H. hungarica Grun.	226
N. miniscula Grun.	24
N. protracta Grun.	188
N. pupula Kutz.	1,004
N. salinarum Grun.	467
N. tripunctata (Mull.) Bory	753
N. sp. unk. 1	102
N. sp. unk. 2	68
N. sp. unk. 3	91
Nitzschia amphibia Grun.	113
N. dissipata (Kutz) Grun.	325
N. filiformis (W. Sm.) Hust.	986
N. fonticola Grun.	166
N. palea (Kutz) W. Sm.	122
N. sublinearis Kutz.	2,497
N. sp. unk. 1	88
N. sp. unk. 2	120

Table 11. Continued.

DIVISION Class *Species*	MEAN VOLUME (μm^3)
Bacillariophyceae (continued)	
Rhoicosphenia curvata (Kutz.) Grun.	291
Rhopalodia gibba (Ehr.) Mull.	15,187
Stauroneis crucicula (W. Sm.) Donk.	196
Stephanodiscus astraea (Ehr.) Grun.	3,039
S. hantzschii Grun.	151
S. niagarae Ehr.	19,270
S. sp. unk. 1	150
Surirella angustata Kutz.	1,570
S. linearis W. Sm.	2,800
S. ovata Kutz.	942
Synedra cyclopum Brutschy	2,260
S. ulna (Nitz.) Ehr.	3,787
S. ulna var. *ramesi* (Heribaud & Peragalla) Hust.	2,115
S. ulna var. *spathulifera* Grun.	6,038
DIVISION PYRROPHYTA (Dinoflagellates)	
Class Dinophyceae	
Ceratium hirundinella (Mull.) Dujardin	4,000
Glenodinium gymnodinium Penard	6,489
Peridinium gatunense Nygaard	84,780
DIVISION CYANOPHYTA (Blue-green Algae)	
Class Myxophyceae	
Anabaena circinalis Rab. (1 mm)	12,000
A. spiroides var. *crassa* Lemm. (1 mm)	11,100
Aphanizomenon flos-aqua (L.) Ralfs (1 mm)	4,906
Coelosphaerium naegelianum Unger (colony)	18,000
Microcystis aeruginosa Kutz (colony)	22,500
M. flos-aqua (Wittr.) Kirchner (colony)	17,300
Oscillatoria sp. unk. 1 (1 mm)	20,000

Table 11. Continued.

DIVISION Class *Spceies*	MEAN VOLUME (μm^3)
DIVISION CRYPTOPHYTA (Blue and Red Flagellates)	
Class Cryptophyceae	
Cryptomonas erosa Ehr.	662
C. ovata Ehr.	3,463
Rhodomonas lacustris Pascher and Ruttner	190

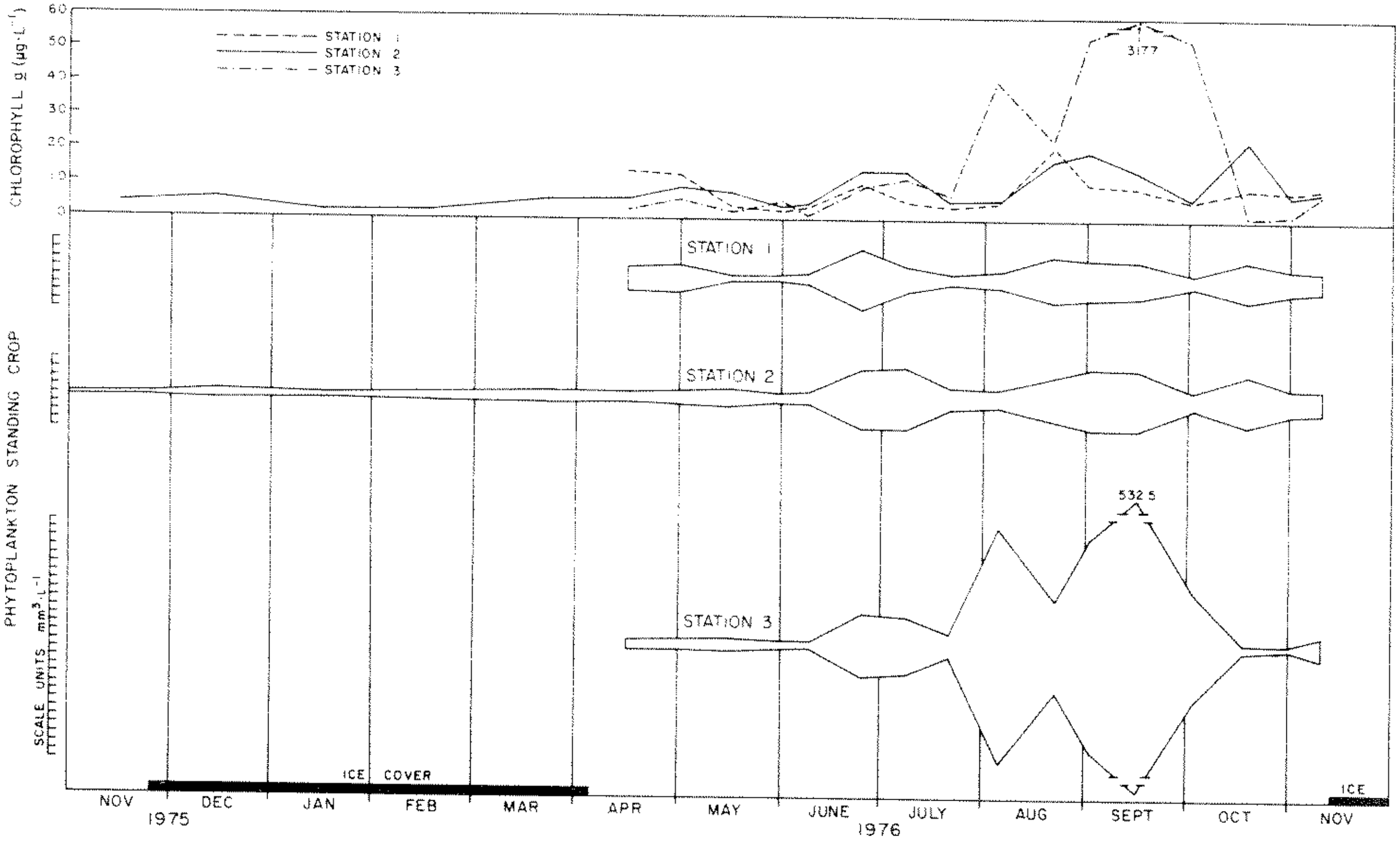

Figure 29. Seasonal variations in the volumetric phytoplankton standing crop and chlorophyll *a* concentration at three Tongue River Reservoir sampling sites. Data were collected from November 1975 through November 1976.

mean volumetric growing season algal standing crops were not significant whether inclusive (n = 16) or exclusive (n = 15) of the 16 September data.

Phytoplankton standing crop data was available for the entire average water year (November 1975 - November 1976) at Station 2 (Figure 29). The volumetric phytoplankton standing crop was maintained at a uniformly low level throughout the winter months as a result of reduced biological activity in the presence of colder water temperatures and low euphotic light intensities. The weighted mean annual volumetric phytoplankton standing crop was 2.62 $mm^3 \cdot L^{-1}$, compared with a mean of 3.53 $mm^3 \cdot L^{-1}$ for the growing season. The mean algae biomass beneath ice cover was 0.71 $mm^3 \cdot L^{-1}$. This figure represents 20% of the mean cell volume encountered during the growing season (i.e. the volumetric phytoplankton standing crop averaged five times larger in the ice-free period than during the winter).

The seasonal variation in volumetric chlorophyll *a* concentration (Figure 29) closely followed that of the algal standing crop at each respective station. The determination of chlorophyll *a* concentrations has generally been regarded as one of the more manageable means of estimating algal biomass because chlorophyll *a* is a unique component of plant matter (Hallegraeff, 1977).

Figure 30 presents the same standing crop data as Figure 29

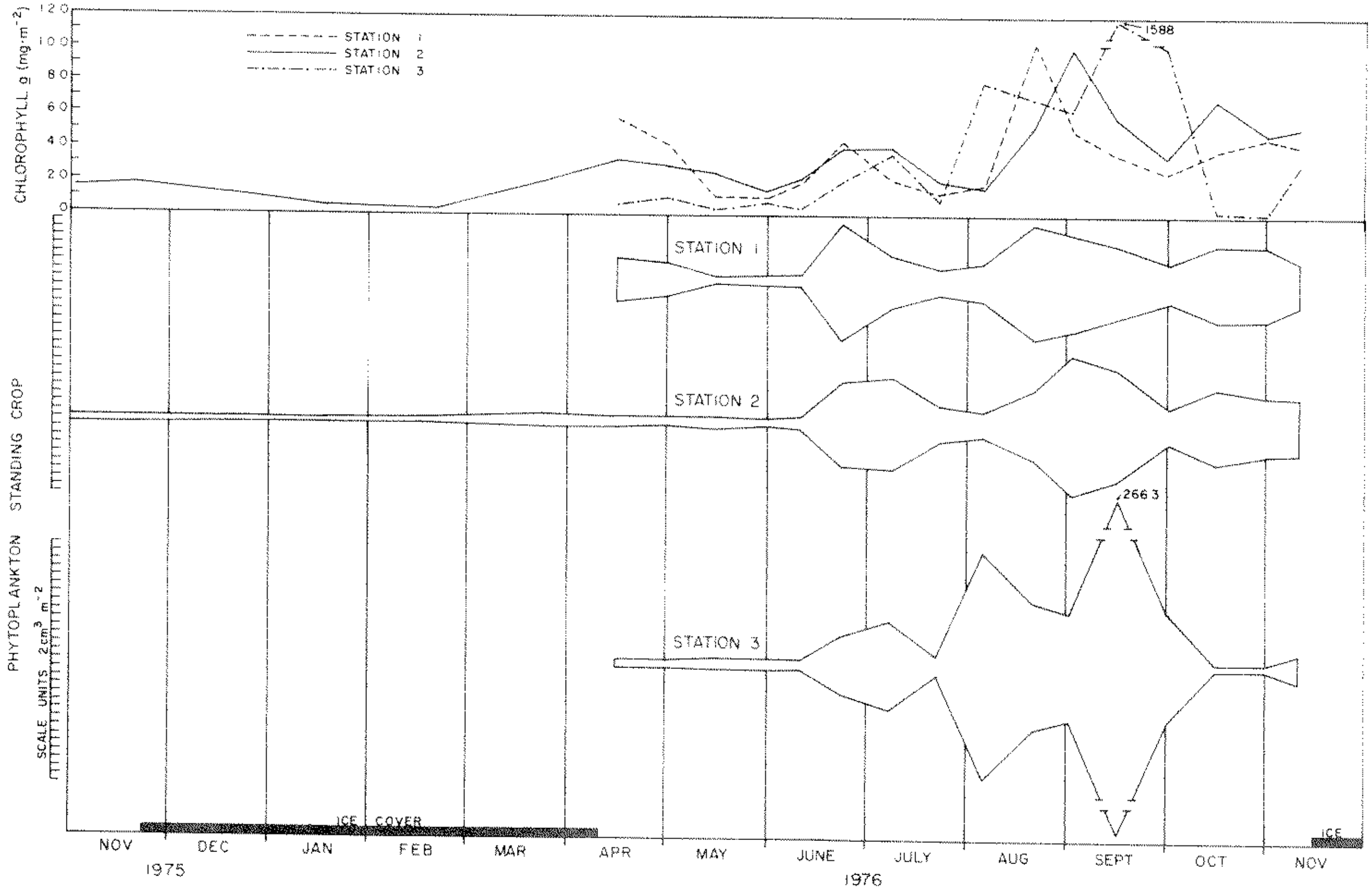

Figure 30. Seasonal variations in the areal phytoplankton standing crop and chlorophyll a concentration at three Tongue River Reservoir sampling sites. Data were collected from November 1975 through November 1976.

except that phytoplankton cell volumes were computed on an areal basis (i.e. cm^3 of algae per m^2 of lake surface area). Comparisons of biomass in this manner eliminate ambiguities resulting from differing horizontal (same date, different stations) and vertical (same station, different dates) euphotic depths. Phytoplankton standing crops reported as $cm^3 \cdot m^{-2}$ are expressed in absolute terms. A congruent argument applies to chlorophyll *a* data.

Expressed on an areal basis, the phytoplankton standing crops at Stations 1 and 2 followed the same trend during the 1976 growing season except for an apparent bloom at Station 1 subsequent to ice-out. Expressed on a volumetric basis (Figure 29) the standing crop at Station 2 generally appeared larger than at Station 1, but these differences were eliminated when the euphotic depth (typically greater at Station 1) was accounted for (Figure 30). In a similar manner, the magnitude of difference between Station 3 and the other two reservoir stations was dampened somewhat when biomass was expressed on an areal basis (compare Figures 29 and 30).

The mean areal standing crops for the entire growing season were 12.67, 12.55 and 14.76 $cm^3 \cdot m^{-2}$ for Stations 1, 2 and 3. The anomalous biomass observed at Station 3 on 16 September was eliminated from the calculation of the average standing crop. If it were included, the mean standing crop at Station 3 would have been 28.35 $cm^3 \cdot m^{-2}$. The differences in the mean areal phytoplankton standing crop among stations for the entire growing season were not statistically significant (Randomized Complete-Block ANOV), whether inclusive ($n = 16$) or exclusive ($n = 15$) of the 16 September data. As expected, the variation in areal chlorophyll *a* concentrations closely followed that of the areal standing crops at all reservoir sampling sites (Figure 30).

Although not considered significant to reject the null hypothesis, the p-values obtained when Friedman's Test was performed to identify differences among seasonal (arbitrarily defined: spring-temperature $<17.5^{\circ}C$, $n = 5$; summer temperature $>17.5^{\circ}C$, $n = 7$; fall-temperature $<17.5^{\circ}C$, $n = 4$) areal and volumetric standing crop means often approached ($p \simeq 0.07$-0.09) the defined level of significance. When Wilcoxson's Signed Rank Test was run on the same data, the trend was for the differences among means to more closely approach significance ($p \simeq 0.08$-0.10) when Station 3 was compared with either Station 1 or Station 2. The difference between the seasonal areal or volumetric standing crop means of Stations 1 and 2 never approached statistical significance.

The weighted mean areal phytoplankton standing crop at Station 2 was 8.91 $cm^3 \cdot m^{-2}$, compared with a mean of 12.55 $cm^3 \cdot m^{-2}$ for the growing season. The mean algal biomass beneath ice cover was 1.64 $cm^3 \cdot m^{-2}$, or, on the average, only 13% as large as the typical open water areal phytoplankton standing crop.

Class Composition of the Standing Crop. Table 12 presents data relative to the broad composition of the phytoplankton standing crop in the Tongue River Reservoir. Class Bacillariophyceae was the only ubiquitous group of algae and comprised the bulk of the average areal phytoplankton standing crop during the 1976 growing season. On a whole lake basis, 59% of the mean areal phytoplankton standing crop during the growing season was composed of diatoms.

Species of Class Cryptophyceae were encountered in nearly all of the phytoplankton samples collected during the study. The Cryptophyceae were never a dominant group, but did contribute significantly to the algal biomass in the fall, winter and early spring. The average areal standing crop on a reservoir-wide basis could be expected to be composed of 6.4% cryptomonads during the growing season.

The Chlorophyceae were never abundant in the Tongue River Reservoir during the growing season. The mean annual whole lake phytoplankton standing crop during that time contained only 1.9% green algae.

The Dinophyceae and Myxophyceae contributed seasonally to the areal phytoplankton standing crop in the Tongue River Reservoir. The relative contributions of the Dinophyceae and Myxophyceae to the areal algal biomass were 20.1% and 12.6% during the growing season. The figure quoted for the Myxophyceae disregards the previously des-

Table 12. A comparison of the areal class composition of the phytoplankton standing crops ($cm^3 \cdot m^{-2}$) at three Tongue River Reservoir stations. All values were calculated from samples collected during the open water period of 1976 unless otherwise specified.

Class	Station 1		Station 2		Station 3	
	Max.	Ave.	Max.	Ave.	Max.	Ave.
Bacillariophyceae	27.42	7.81	18.35	6.58 (4.75)[1]	45.15	9.24
Chlorophyceae	1.35	0.38	1.41	0.26 (0.20)	0.79	0.12
Cryptophyceae	3.78	1.27	2.65	0.81 (0.69)	1.99	0.48
Dinophyceae	12.51	1.54	24.66	3.06 (2.04)	19.95	3.41
Myxophyceae	6.76	1.67	6.88	1.84 (1.23)	219.28 (19.20)[2]	15.10 (1.51)[2]
Total		12.67		12.55 (8.91)		28.35 (14.76)[2]

[1] Averages in parentheses are weighted means of data collected from November 1975 - November 1976 at Station 2.

[2] Numbers in parenthesis represent maximum and means of data collected at Station 3 during the open water period of 1976 exclusive of sample collected on September 1976, when an extraordinary bloom of *Aphanizomenon flos-aqua* occurred.

cribed anomaly.

The Euglenophyceae were encountered sporadically and comprised an insignificant portion of the plankton standing crop. Consequently they have been ignored in the preceding discussion.

Beneath the 1975-76 winter ice cover, the following relative areal class composition of the standing crop was realized at Station 2: Bacillariophyceae-67%, Cryptophyceae-29%, Chlorophyceae-5%. This reflects the importance of diatoms and crytpomonads during the winter months. The annual (November 1975-November 1976) average areal phytoplankton standing crop at Station 2 was 8.91 $cm^3 \cdot m^{-2}$ (Table 12) and was composed of 53.3% Bacillariophyceae, 2.2% Chlorophyceae, 7.7% Cryptophyceae, 22.9% Dinophyceae and 13.8% Myxophyceae.

Seasonal Succession and Periodicity. Factors affecting the seasonal variation in phytoplankton species were defined by Hutchinson (1967) as follows:

1) partly independent physical factors (temperature, illumination, turbulence)
2) interdependent chemical factors (inorganic nutrients, antibiotics)
3) biological factors (predation, parasitism, competition).

Moss (1973) felt that residual populations of all algal species of a given body of water were omnipresent and that the dynamic interaction of the above factors created a myriad of ever-changing niches favoring various species at different times. Species successional patterns reflected the environmental conditions. In the absence of undue perturbation from outside influences, the successional periodicity of the planktonic biomass for a given body of water is reasonably constant from year to year (Wetzel, 1975). Figures 31 and 32 were constructed to illustrate the typical annual (November 1975-November 1976) periodicity and successional pattern of algae at Station 2, the mid-reservoir sampling site.

The bulk of the standing crop at Station 2 during the winter months was composed of the Bacillariophyceae and the Cryptophyceae (Figures 31 and 32). Of the cryptomonads present, *Cryptomonas ovata* dominated, while *C. erosa* and *Rhodomonas lacustris* contributed little to the planktonic biomass. The Cryptophyceae are often regarded as cold-water, low-light adapated species (Wright, 1964; Rodhe, 1955). Class Bacillariophyceae was represented by numerous species that occurred sporadically and in low numbers (Figure 32). A few species dominated and contributed significantly to the winter standing crop. These included *Gomphonema olivaceum*, *G. parvulum*, *Diatoma hemiale*, *Synedra ulna*, and an unknown species of *Cyclotella*. The Chlorophyceae were usually present during the winter, although always in such low numbers that no particular species were outstanding. Typically,

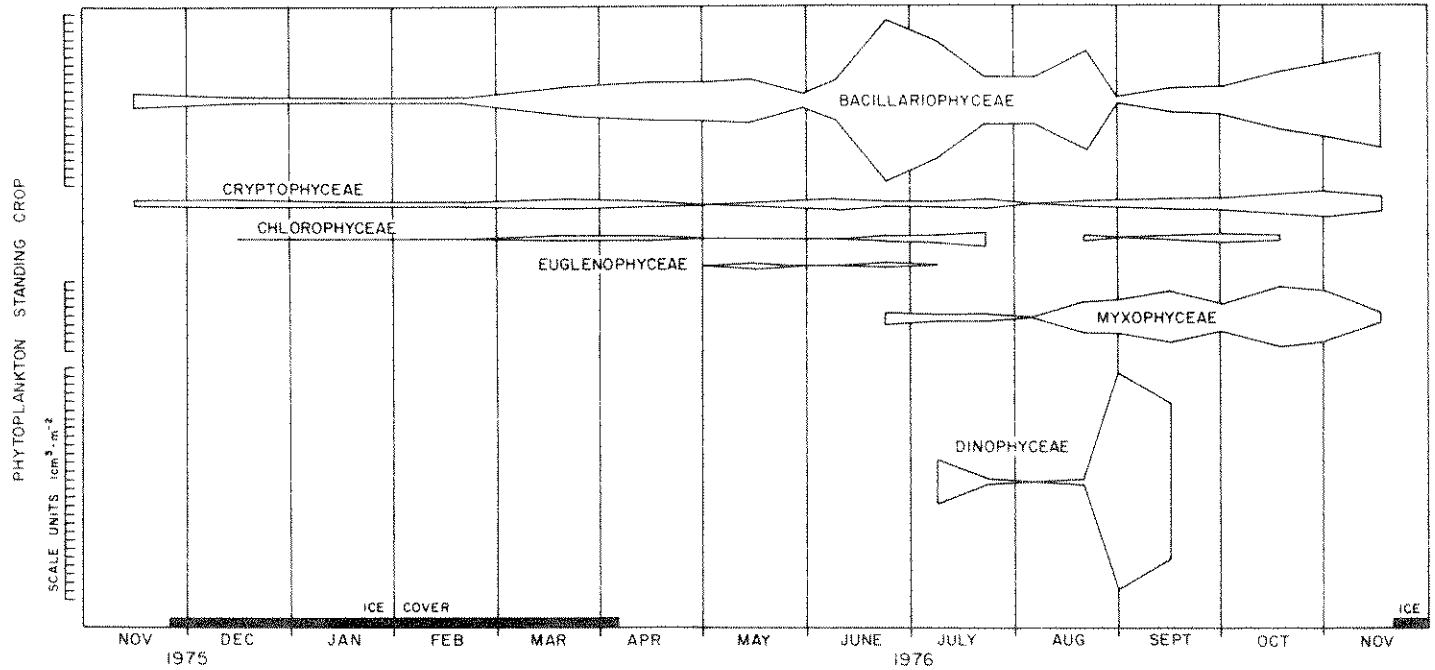

Figure 31. Seasonal variations in class composition of the phytoplankton standing crop at Station 2 in the Tongue River Reservoir. Data were collected from November 1975 through November 1976.

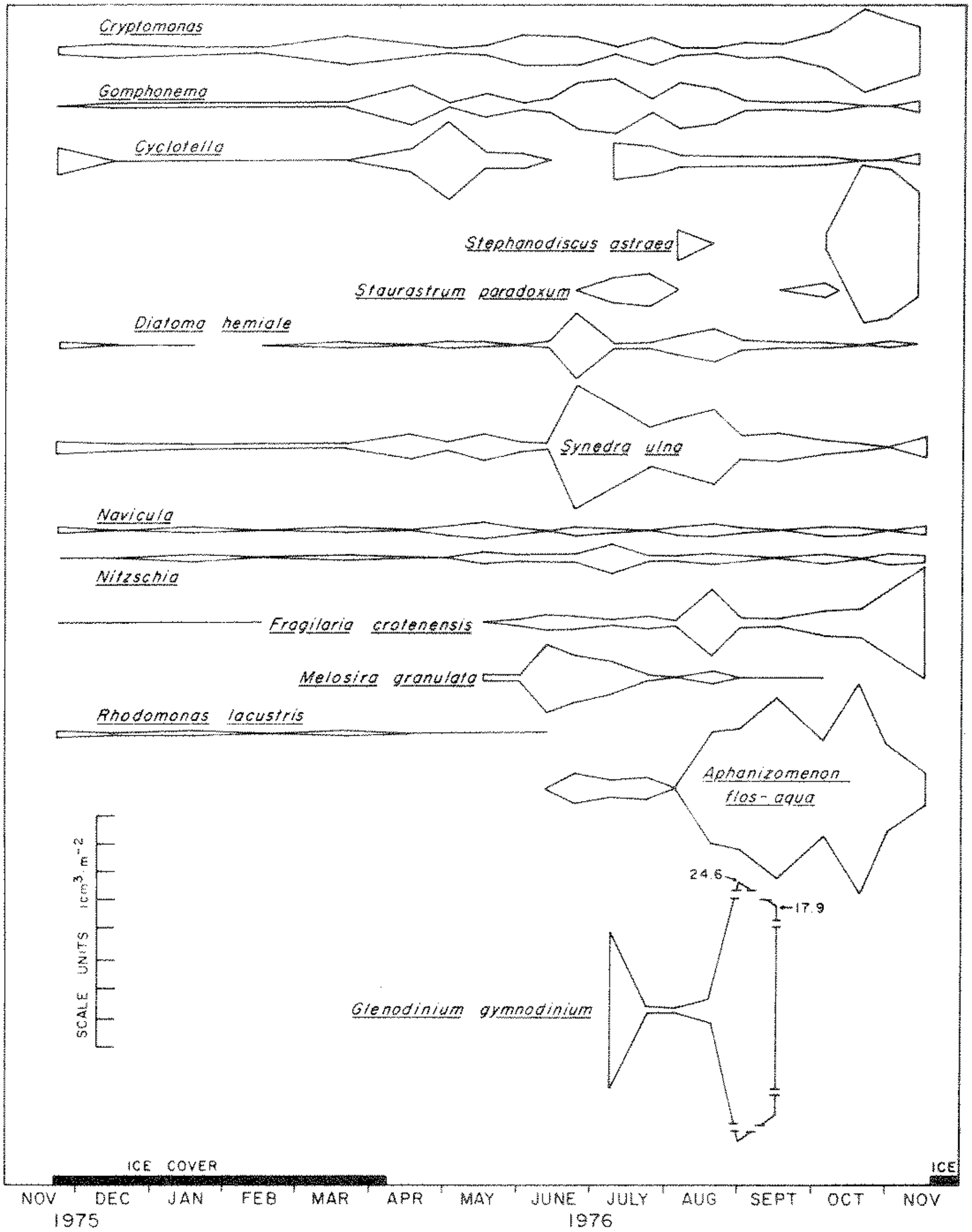

Figure 32. Seasonal variation and successional pattern of phytoplankton at Station 2 in the Tongue River Reservoir.

flagellated microplankton such as *Chlamydomonas* sp. and *Carteria* sp. were encountered. *Chlamydomonas* has also been identified as a low-light, cold-water alga (Wetzel, 1975). Winter algal standing crops were maintained at low levels until just prior to ice-out in late March. The bloom that occurred at that time was composed predominately of diatoms (Figure 31).

Upon ice-out a gradual increase in the total plankton biomass was noted (Figures 31 and 32), although a vernal maximum was never realized. This increase was due almost wholly to an increase in diatoms, such as *Synedra ulna, Gomphonema olivaceum* and most notably *Cyclotella bodanica*. Algal pulses subsequent to ice-out are often dominated by centric diatoms, although *Stephanodiscus* is more often implicated than *Cyclotella* (Pechlaner, 1970). *Asterionella* has been classically studied as a spring bloomer (Lund, 1949, 1950), but has also been noted to develop a fall maximum (Tucker, 1957). At no time during the course of this study did *Asterionella* comprise an appreciable segment of the biomass.

In June and early July a phytoplankton pulse composed principally of the Bacillariophyceae was observed (Figure 31). Dominant algal forms included *Synedra ulna, Diatoma hemiale, Gomphonema constrictum, G. olivaceum* and *Melosira granulata* (Figure 32). Additionally, a large cell volume of the cryptomonad *C. ovata* was evident, and the blue-green alga *Aphanizomenon flos-aqua* first appeared, although in relatively low volume.

From early July through early August a gradual decrease in total planktonic biomass was detected (Figure 31). At that time a minor bloom of an unknown species of *Cyclotella* (probably *michiginiana*) was realized and *S. ulna* continued to contribute heavily to the total plankton standing crop (Figure 32). During the 1976 growing season, the largest standing crop of the genus *Nitzschia* (predominately *N. palea* and *N. dissipata*) was observed in July. In early July a minor bloom and rapid crash of the dinoflagellate *Glenodinium gymnodinium* was noted while in early August the Myxophyceaen *A. flos-aqua* decreased to a nearly indetectable level. The Chlorophyceae, although never contributing significantly to the total standing crop, made their largest contribution in July, largely because of the thermophilic (Hutchinson, 1967) *Staurastrum paradoxum*. Other common species of green algae included *Oocystis pusilla*, *Eudorina elegans*, *Pandorina morum*, *Pediastrum duplex* and *P. simplex*. The Bacillariophyceae continued to clearly constitute the bulk of the phytoplankton biomass through early August.

In late August the standing crop started to increase toward an early September maximum which expired by 1 October. The largest areal phytoplankton standing crop achieved at Station 2 was 32.99

$cm^3 \cdot m^{-2}$, recorded on 2 September. This was attributed predominately to *G. gymnodinium* which reached a maximum of 24.66 $cm^3 \cdot m^{-2}$ on that date (Figure 32). *G. gymnodinium* decreased to 17.9 $cm^3 \cdot m^{-2}$ on 16 September and was completely absent from Station 2 on 2 October. In early August the Mxyophycean *A. flos-aqua* began to build toward a fall maximum. *M. granulata* is characteristic of lakes that produce significant blue-green maxima, although the development of this diatom need not necessarily coincide with that of the Myxophyceae (Hutchinson, 1967). Note the pulse of *M. granulata* immediately preceding that of *A. flos-aqua* in Figure 32. Although this late-summer pulse was dominated by the Myxophyceae and the Dinophyceae, the Bacillariophyceae were also present. Through the beginning of September *S. ulna* continued to play an integral role in the total standing crop. In August, a moderate pulse of *Fragilaria crotenensis* and the first appearance of the centric diatom *Stephanodiscus astraea* were observed. An increase in *Gomphonema* at that time was attributed to *G. olivaceum* and *G. constrictum. Navicula cryptocephala* and *N. cryptocephala* var. *veneta* were observed in relatively high numbers through early September.

In the month prior to ice-in a final phytoplankton maximum was realized (Figure 31). A high cell volume of *A. flos-aqua* persisted almost until ice-in. A major pulse of *S astraea* appeared in early

October and flourished until ice-in, while *F. crotenensis* progressively increased in biomass from mid-September through mid-November (Figure 32). Both of these algae are considered to be cold-water, low-light forms, although *Stephanodiscus* typically dominates vernal as opposed to autumnal pulses (see previous discussion). A marked increase in the population of cryptomonads (notably *C. ovata*) was also attendant to the cooling of the water mass and waning of the solar input. Just prior to ice-in the Cryptophyceae-Bacillariophycean association characteristic of the winter months was restored.

In a classical temperate system, a spring bloom of fast-growing diatoms is followed by a reduction in standing crop when the nutrients in the euphotic zone of the epilimnion become depleted. Often the availability of silica becomes limiting, although obviously many interacting factors determine the pattern of species succession. Thermophilic green algae with a lower growth rate than their predecessors are common in the summer months, although their biomass is usually not great. If the nitrogen concentration becomes limiting, a preponderance of heterocystous blue-green algae appears late in the summer, followed by an autumnal pulse of the Bacillariophyceae with fall overturn. Because of its shallow nature and morphometric character, the Tongue River Reservoir deviated somewhat from this mold: no vernal phytoplankton bloom was noted immediately following ice-out, although a minor pulse was observed at Station 1; the Chlorophyceae were distinctly lacking in the euphotic zone throughout the summer at all sampling sites; a major pulse of Dinophyceae was evident late in the summer.

Perhaps the most notable observation relative to the algal succession and standing crop was the temporal differences in appearance of various algal groups among reservoir stations. The lack of bays and arms, small size, and narrow shape of the reservoir indicate that a homogenous reservoir-wide phytoplankton population should have existed on a given sampling date. However, this was not always the case. Exclusive of the minor pulse observed at Station 1 in April, three distinct phytoplankton maxima were noted during the growing season at each reservoir sampling site (Figure 30). The pulses were often discordant temporally and in terms of phytoplankton class composition when Station 3 was compared to Stations 1 and 2.

The initial pulse in June and July (Figure 30) was of a Bacillariophyceae-Cryptophycean nature at all three reservoir stations, with no notable differences in species composition of the pulse among sites. The second peak at Station 3 occurred in early August when minima were being reached at Stations 1 and 2 and was composed almost wholly of Dinophyceae (*G. gymnodinium*). The corresponding pulse at Stations 1 and 2 occurred in late August and early September, during which time a minimum was realized at Station 3. Pulse II was predominantely Myxophyceae-Dinophyceae and bi-specific (*A flos-aqua* and *G. gymnodinium*) in nature at both sampling sites. Pulse III appeared in mid-September at Station 3 and was a Myxophycean monoculture of *A. flos-aqua*. The sister pulses at Stations 1 and 2 took place from mid-October until ice-in and were Myxophyceae-Bacillariophyceae-Cryptophycean relationships, with a gradual phasing out of the Myxophyceae (*A flos-aqua*). Again the species composition of the standing crop was identical between Stations 1 and 2.

The most obvious example of horizontal limnoplanktonic heterogeneity in the reservoir involves the appearance and disappearance of *Glenodinium gymnodinium*. An analysis of the temporal abundance of this alga at the various sampling sites indicates that conditions favored *G. gymnodinium* at Station 3 late in August and that the slight current moving through the narrow reservoir carried *G. gymnodinium* to the lower reservoir stations (Figure 33). The peak of the pulse occurred later and the magnitude of the pulse generally weakened as it progressed to the lower end of the reservoir.

Little is known about the ecology of the freshwater dinoflagellates. *G. gymnodinium* is considered to be a warmwater form favoring hard waters of moderate humic content (Hutchinson, 1967). Conditions relative to iron, inorganic carbon, nitrogen, and phosphorus were

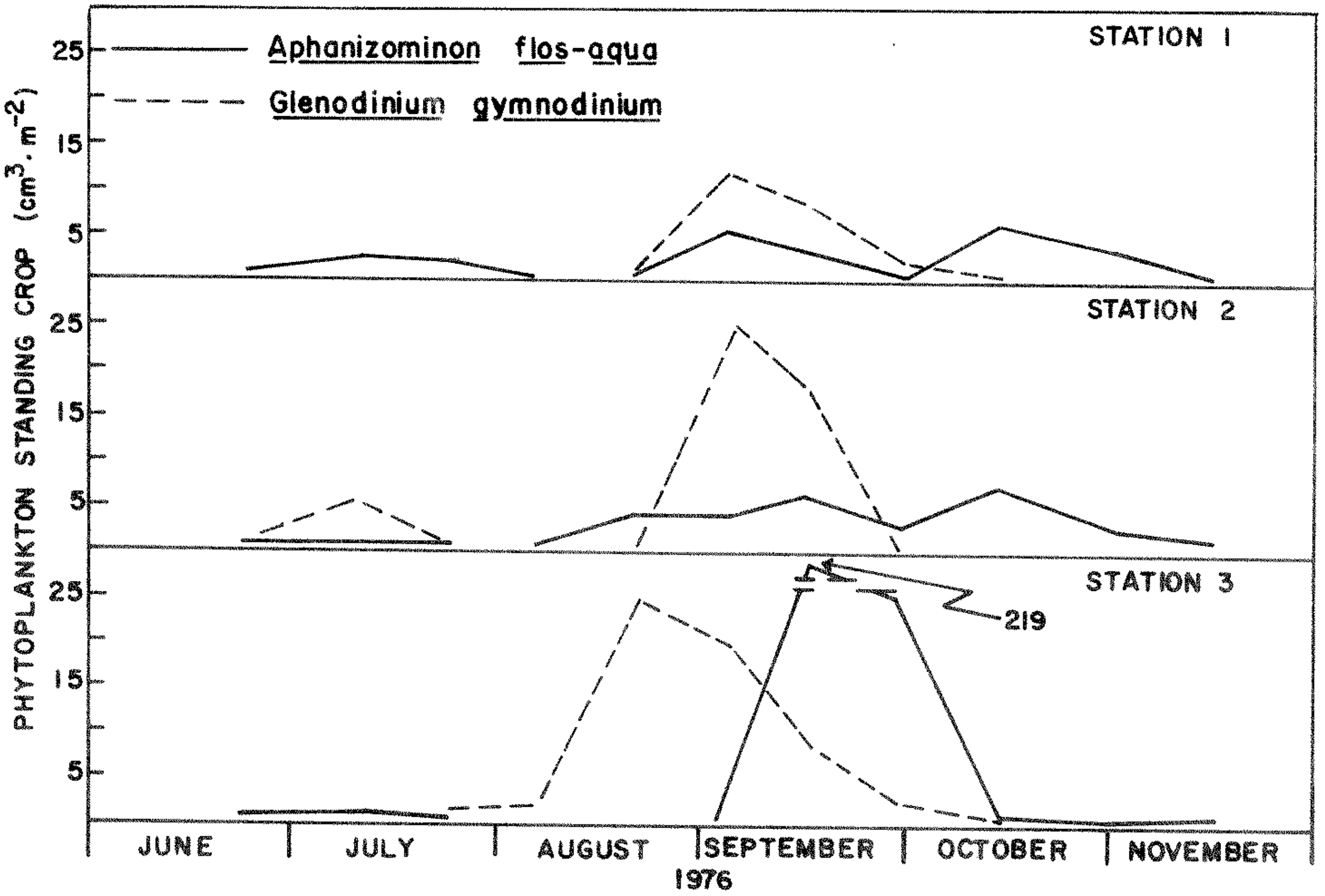

Figure 33. Seasonal succession of *Aphanizomenon flos-aqua* and *Glenodinium gymnodinium* in 1976 at three sampling sites in the Tongue River Reservoir.

similar at all three reservoir sampling stations in the period preceding the bloom as well as during the bloom. Hence, it is believed that other factors were responsible for the initiation of the bloom at Station 3.

Prakash and Rashid (1968) found that the growth of marine dinoflagellates was enhanced by humic substances which were believed to have a stimulatory effect, increasing the permeability of the cell membrane to nutrients. Golterman (1975) hypothesized that the observed growth stimulation was due entirely to chelation.

The humic content of the euphotic water at Station 3 was probably higher than that of the other two reservoir stations, due to its shallow nature (especially during reduced reservoir stage in late summer), proximity to the inflowing river, and wind-generated agitation of the sediment. However, it seems unlikely that the humic content alone triggered the bloom, irrespective of its mode of action. It is more likely that an unidentified combination of many biological, physical and chemical factors (possibly including humic content) favored *G. gymnodinium* late in the summer at Station 3. The presumed high humic content of Station 3 may have been contributory or merely coincidental to the bloom.

To further illustrate relative differences among sampling sites for a given sampling date, the Bray-Curtis Measure of Dissimilarity

(Clifford and Stephenson, 1975) was applied to the 1976 growing season phytoplankton standing crop data (Table 13). Because of the manner in which this index was used, differences in the total algal standing crop and relative class composition of the standing crop were reflected in the values obtained. A value of zero reveals that two stations were identical while maximum dissimilarity was indicated by a value of 1. For values greater than 0.5 (arbitrarily chosen), the author's explanation for the dissimilarity is noted in the table. Most of the dissimilarity among stations was reflected in the size of the total standing crop, especially in the fall. During the dinoflagellate bloom at Station 3, differences in the relative class composition became important, while on 16 September both factors were involved (see Figure 33). From the means in Table 13, note that Stations 1 and 2 were generally more similar to each other than to Station 3 as a result of their coincidental pulses.

Table 14 ranks some of the major phytoplankton species on a reservoir-wide basis according to absolute mean cell volume. Table 15 ranks several of the major plankton species according to frequency of occurrence. *Staurastrum paradoxum* is the only green alga to appear in either table and both tables are clearly dominated by diatoms. Several forms were virtually holoplanktonic (Table 15) and *Cymbella turgida* (Tables 14 and 15), strictly a benthic form, was unexpectedly

Table 13. The Bray-Curtis Measure of Dissimilarity applied to the 1976 Tongue River Reservoir growing season standing crop data. Causative factors are indicated in parenthesis for values greater than 0.5.

Date	Station 1:2	Station 1:3	Station 2:3
16 April 1976	0.34	0.70 (1)	0.49
1 May	0.33	0.62 (1)	0.37
14 May	0.42	0.18	0.34
29 May	0.17	0.29	0.25
10 June	0.43	0.30	0.47
25 June	0.21	0.31	0.15
8 July	0.43	0.59 (1)	0.31
22 July	0.34	0.27	0.37
6 August	0.15	0.65 (1)	0.71 (1)
20 August	0.45	0.71 (2)	0.61 (2)
2 September	0.32	0.29	0.15
16 September	0.28	0.89 (1,2)	0.88 (1,2)
2 October	0.26	0.62 (1)	0.55 (1)
16 October	0.11	0.96 (1)	0.96 (1)
30 October	0.10	0.91 (1)	0.90 (1)
11 November	0.17	0.21	0.36
$\bar{x}$	0.28	0.53	0.49

(1) = differences in size of total standing crop.

(2) = differences in relative class composition of standing crop.

common in the euplankton. Other diatoms listed in Tables 14 and 15 have been variously described as being planktonic, benthic or epiphitic and can occupy a number of niches. However, the dominant diatoms (*S. ulna*, *S. astraea*, and *F. crotenesis*) are strict euplankters.

Figure 34 plots the mean relative standing crop of the six most

Table 14. Rank of the major phytoplankton taxa in the euphotic zone of the Tongue River Reservoir according to absolute mean cell volume ($cm^3 \cdot m^{-2}$). The table is constructed from samples collected at all stations during the open water period of 1976.

Rank	Taxon	Cell Volume
1	*Glenodinium gymnodinium*	2.539
2	*Aphanizomenon flos-aqua*[1]	1.400
3	*Synedra ulna*	1.156
4	*Stephanodiscus astraea*	0.846
5	*Cryptomonas ovata*	0.659
6	*Fragilaria crotenensis*	0.635
7	*Gomphonema olivaceum*	0.442
8	*Hannaea arcus*	0.381
9	*Diatoma hemiale*	0.350
10	*Melosira granulata*	0.330
11	*Cyclotella bodanica*	0.265
12	*Cyclotella* sp.	0.255
13	*Cocconeis placentula*	0.238
14	*Gomphonema constrictum*	0.224
15	*Staurastrum paradoxum*	0.190
16	*Diatoma elongatum*	0.156
17	*Asterionella formosa*	0.129
18	*Rhoicosphenia curvata*	0.119
19	*Cymbella turgida*	0.071
20	*Navicula cryptocephala*	0.037

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[1] Data excludes extraordinary observation of 266.3 $cm^3 \cdot m^{-2}$ at Station 3 on 16 September 1976.

Table 15. Rank of the major phytoplankton taxa in the euphotic zone of the Tongue River Reservoir according to frequency of occurrence (%). The table is constructed from samples collected at all stations during the open water period of 1976.

Rank	Taxon	Frequency of Occurrence
1	*Cryptomonas ovata*	94
2	*Gomphonema olivaceum*	92
3	*Synedra ulna*	88
	Fragilaria crotenensis	88
5	*Navicula cryptocephala*	80
6	*Cocconeis placentula*	78
7	*Diatoma hemiale*	75
8	*Cymbella turgida*	74
9	*Hannaea arcus*	70
10	*Rhiocosphenia curvata*	68
	Asterionella formosa	68
12	*Cyclotella* sp.	61
13	*Aphanizomenon flos-aqua*	58
14	*Diatoma elongatum*	50
15	*Stephanodiscus astraea*	49
16	*Melosira granulata*	45
17	*Gomphonema constrictum*	41
18	*Rhodomonas lacustris*	36
19	*Cyclotella bodanica*	31
20	*Glenodinium gymnodinium*	30

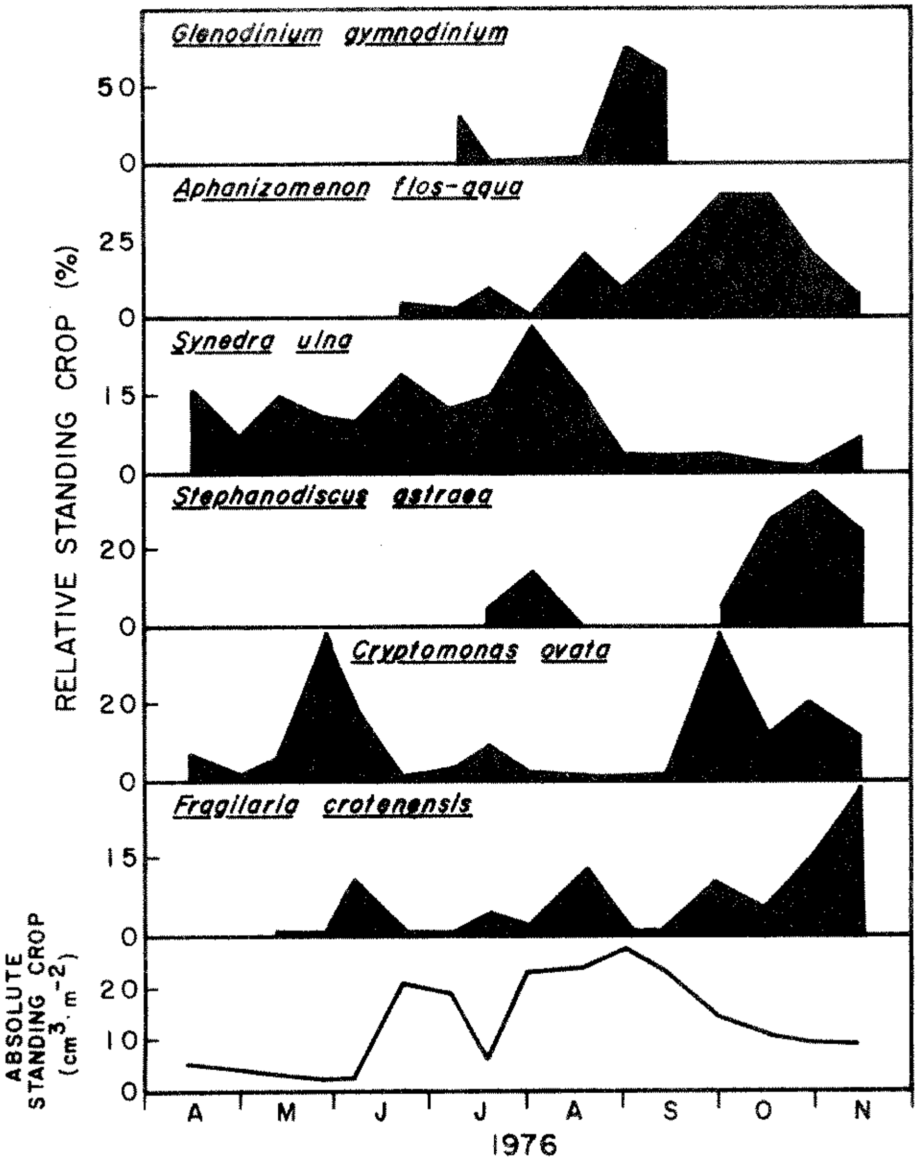

Figure 34. Mean relative standing crops of the five major phytoplankton species in the Tongue River Reservoir during the 1976 growing season plotted against the absolute areal phytoplankton standing crop.

common plankton species in the Tongue River Reservoir against the total areal standing crop. The two most dominant species on an annual basis, *G. gymnodinium* and *A. flos-aqua* , owe their dominance to seasonal pulses in which they composed the bulk of a large areal standing crop. *S. astraea* behaved in an analogous manner. *S. ulna, C. ovata* and *F. crotenensis* were holoplanktonic and rarely composed a significant portion of a large areal standing crop.

Chlorophyll *a* and Cell Volume Relationships. The chlorophyll content per unit of algal biomass can be expected to vary interspecifically as well as intraspecifically. The age of the culture, nutrient availability, light intensity and temperature all have an effect on the chlorophyll content of phytoplankton (Paasche, 1960). Diel fluctuations in chlorophyll content have also been recorded (Yentsch and Ryther, 1957). Figure 35 presents seasonal average chlorophyll *a*/cell volume ratios ($\mu g \cdot mm^{-3}$) for all three Tongue River Reservoir stations. Throughout the summer the ratio remained relatively constant and was similar at all three sampling sites. Higher chlorophyll/cell volume ratios were recorded in late November and in the spring at all three sites and throughout the winter at Station 2. Jorgensen (1969) postulated that there were two types of algal adaptation to low light intensities; an increase in the chlorophyll/cell volume ratio (presumably an increase in chlorophyll/photosynthetic

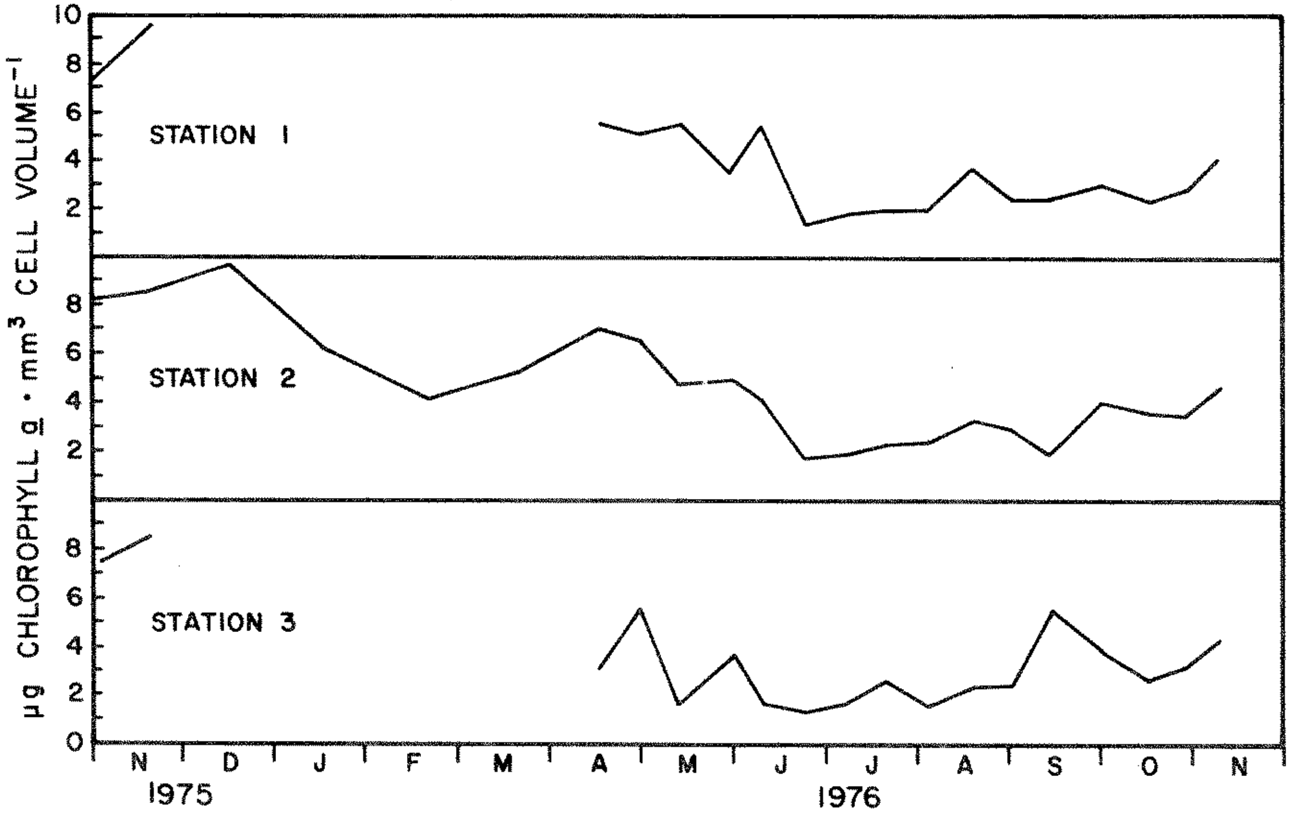

Figure 35. Seasonal euphotic zone phytoplankton chlorophyll *a*/cell volume ratios in the Tongue River Reservoir. Data were collected from November 1975 through November 1976.

unit) or a decrease in the light intensity necessary to reach the light-saturated rate of photosynthesis. He also admitted that there were probably transition states between these extremes. It is possible that the increased chlorophyll/cell volume ratios observed in late fall and winter were a result of planktonic adaptation to reduced light intensity, i.e. an increase in chlorophyll/photosynthetic unit.

A strong correlation ($n = 51$, $r = 0.99$, $p < 0.01$) was found between large phytoplankton standing crops and high chlorophyll *a* concentrations when data collected during the entire study was considered. A simple linear regression was run between 1976 growing season whole lake phytoplankton cell volumes and chlorophyll *a* concentrations (Figure 36), and the following highly significant ($p < 0.01$) regression equation was obtained:

$$\mu\text{g Chl } a = 1.93\ (\text{mm}^3 \text{ Cell Volume}) + 3.27 \quad ; \quad n = 47,\ r^2 = 0.78.$$

During the growing season a change in 1.0 $\text{mm}^3 \cdot \text{L}^{-1}$ cell volume was equivalent to a change of 1.93 μg chlorophyll *a*. The weighted mean annual (November 1975-November 1976) whole lake chlorophyll *a*/cell volume ratio was 2.97 while the whole lake ratio during the 1976 growing season was 2.66.

Analysis of Factors Governing Standing Crop. The most common sources of nitrogen for algal consumption are inorganic even through a limited uptake or organic nitrogen has been observed in some algal

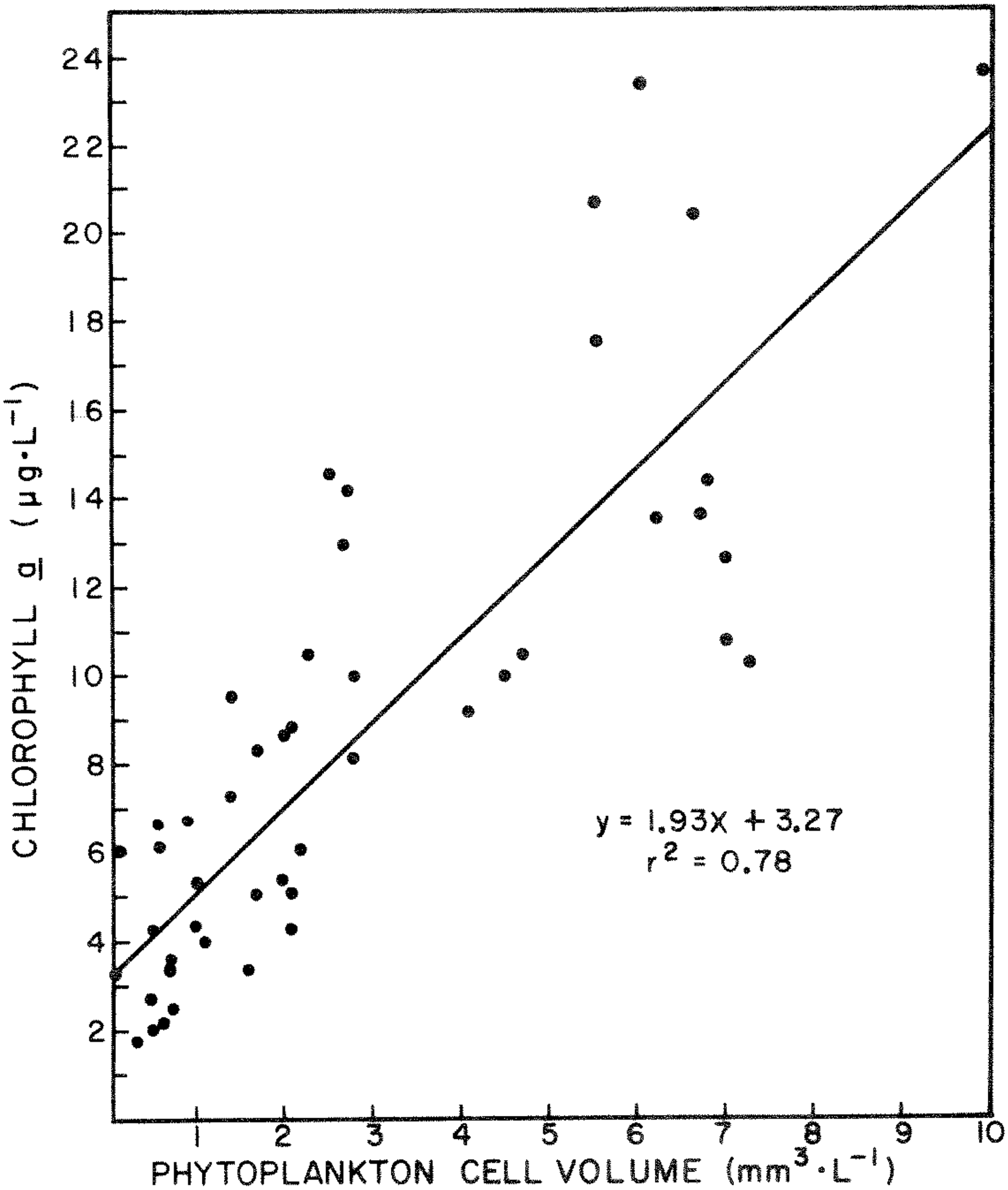

Figure 36. Regression line correlating euphotic zone phytoplankton cell volumes and euphotic zone chlorophyll *a* concentrations in the Tongue River Reservoir during the 1976 growing season.

groups (Syrett, 1962). Of the inorganic forms available, the most reduced form, ammonium, is preferentially utilized (Syrett, 1962), although some evidence indicates approximately equivalent uptake of ammonium and nitrate (Eppley, *et al.*, 1969). Inorganic orthophosphorus is the principal source of phosphorus for algal growth (Kuhl, 1974). However, it has been proven that inorganic polyphosphates can be utilized to a limited extent by some groups, possibly via the release of exoenzymes into the water for catalytic dissociation (Krauss, 1958).

An examination of Figures 23-25 suggests that inorganic nutrients were in short supply in the euphotic zone of the Tongue River Reservoir through most of the growing season. Saraceni and Gerletti (1968) demonstrated that a large increase in algal production and cell volume could be obtained by culturing a planktonic community in hypolimnetic water. The stimulation of growth was ascribed to the increased availability of inorganic nutrients (N and P) in the hypolimnetic water. A thermocline never developed in the Tongue River Reservoir, and, in effect "hypolimnetic" water was freely available to the standing crop during the summer via wind agitation and vertical migration along a concentration gradient. For this reason, the typical summer lag in standing crop caused by the depletion of nutrients in the epilimnion was never realized. Instead, a triacmic algal standing crop was observed during the growing season. Even the minima represented healthy populations of phytoplankton because the Tongue River Reservoir was in a constant state of overturn.

Table 16 lists correlation coefficients (Spearman's Rank Correlation Test) between the 1976 growing season areal whole lake standing crop and several environmental measures. Some of the factors tested proved to be significantly correlated with the areal standing crop, but the correlation coefficients of statistical significance were generally rather low. This lends supportive evidence to the statement that no few measured parameters or combination of parameters had an overriding influence on phytoplankton standing crop.

The question of the nature of the relationship (cause, effect, or coincidence) arises relative to the statistically significant correlation coefficients. There is little doubt statistically that the observed r-value associated with the whole lake HCO_3^- concentration is significant. However, it is likely that the (-) correlation obtained is coincidental in the sense that the highest bicarbonate concentrations were observed in the early spring and late fall, when standing crops were generally low. Considering the overall total alkalinity and pH, inorganic carbon was always abundant and certainly never a factor limiting the standing crop.

Table 16. Results of Spearman's Rank Correlation Test as applied between the 1976 growing season areal whole lake standing crop ($cm^3 \cdot m^{-2}$) and several environmental parameters of the Tongue River Reservoir.

Parameter		r_s	p
Areal standing crop–	Whole lake N/P (moles)	0.45	<0.01
	Euphotic zone N/P (moles)	0.41	<0.01
	Euphotic temperature (°C)	0.36	0.01
	Whole lake HCO_3 ($moles \cdot m^{-2}$)	-0.34	0.01
Significant	Whole lake Σ inorganic-N ($mmoles \cdot m^{-2}$)	0.31	0.03
	Euphotic PO_4 ($\mu moles \cdot m^{-2}$)	-0.28	<0.05
	Whole lake PO_4 ($mmoles \cdot m^{-2}$)	0.24	0.09
	Inflow PO_4 load ($kg \cdot day^{-1}$)	-0.21	-
	Spec. cond. ($\mu mhos \cdot cm^{-1}$)	-0.20	-
Non-significant	Euphotic depth (m)	0.14	-
	Incident solar input ($ly \cdot day^{-1}$)	0.12	-
	Extinction coefficient (m^{-1})	-0.07	-
	Euphotic Σ inorganic-N ($\mu moles \cdot m^{-2}$)	0.07	-
	Inflow inorganic-N load ($kg.day^{-1}$)	0.01	-

n = 48 in all cases.

It is probable that the observed (-) correlation between the phytoplankton standing crop and the areal euphotic PO_4 concentration was the effect of the standing crop, rather than the cause. The areal whole lake inorganic nitrogen concentration was positively correlated with the standing crop, yet the standing crop was indifferent to the areal euphotic inorganic nitrogen concentration. This demonstrates that nitrogen from the aphotic zone was avilable for algal uptake and implies a mild causal relationship between the areal whole lake inorganic nitrogen concentration and the standing crop. There was no statistically significant correlation between the areal whole lake PO_4 concentration and the standing crop, which indicates that inorganic phosphorus was usually in adequate supply. It then follows that increased N/P ratios were probably a cause of increased standing crops rather than an effect, although the argument for either case (cause or effect) could be developed.

Certainly numerous ecological considerations enter into the development of any phytoplankton maximum, and it appears that no few overriding factors influenced the volume of algae present in the Tongue River Reservoir.

Phytoplankton Primary Production

Ryther and Yentsch (1957) determined a relationship between the photosynthetic rate at light saturation and the chlorophyll content of a natural phytoplankton population. This was used in conjunction with the known influence of the total daily surface radiation and the extinction coefficient upon the photosynthetic rate to derive the following empirical formula for calculating day rate photosynthesis beneath a m^2 of lake surface:

$$P = R/k \times \mathrm{Chl} \times 3.7 \quad (1)$$

where P = photosynthesis of the plankton population in $g\ C \cdot m^{-2} \cdot day^{-1}$,

R = relative photosynthesis, obtained from Figure 1 in Ryther and Yentsch (1957),

k = extinction coefficient (m^{-1}),

Chl = g chlorophyll $a \cdot m^{-3}$

3.7 = average assimilation ratio (number) at light saturation expressed in g C·g Chl $a^{-1} \cdot hr^{-1}$.

The above empirical relationship assumes a homogenously distributed phytoplankton population in the euphotic zone.

In an analysis of several techniques for the estimation of day rate photosynthesis, Saunders, *et al.* (1962) compared the more accurate O_2 technique of measuring algal productivity to the Ryther and Yentsch empirical approach. An average assimilation ratio of 2.35 (determined via *in situ* ^{14}C experiments) was used. Although the mean difference expressed in percentage between 10 pairs of estimates was slightly greater than 40%, the Wilcoxson Matched-Pairs Signed-Ranks Test did not reject the null hypothesis at $p = 0.05$. Curl and Small (1965) found that the *in situ* determination of an assimilation ratio for a small homogeneous area yielded an estimate of production superior to that predicted when using the average assimilation ratio of 3.7 cited by Ryther and Yentsch. The fixed value of 3.7 is probably the weakest component of the equation (Ryther and Yentsch, 1957).

Throughout this study, the standard formula of Ryther and Yentsch (Equation 1) consistantly underestimated daily photosynthetic rates when compared to more sensitive short-term *in situ* ^{14}C determinations

(Figure 37). The mean estimates of diel photosynthesis from 29 paired observations taken from June 1975–November 1976 were 0.91 and 0.54 g $C \cdot m^{-2} \cdot day^{-1}$ for the ^{14}C experiments and the empirical approach respectively. The mean difference expressed on a percentage basis was 41%, comparable to the results of Saunders, *et al.* (1962) who worked in Lake Michigan with a locally determined assimilation ratio. However, a Paired t-test rejected the null hypothesis ($n = 29$, $p < 0.01$), and it was concluded that significant differences existed between the two mean estimates of production.

The short-term *in situ* ^{14}C incubations were conducted in a manner that facilitated the estimation of an assimilation ratio as well as the extrapolation of the results to predict day rate photosynthesis (see Methods and Materials). As estimation of the hourly light-saturated rate of carbon fixation per unit chlorophyll was obtained through an examination of the photosynthesis-depth curve constructed from the ^{14}C experiment of each cruise. In this manner, a locally determined average assimilation ratio of 4.85 ($n = 29$) was realized. This was substituted for the standard 3.7 g $C \cdot g$ Chl a^{-1} hr^{-1} of Equation 1, as per Saunders, *et al.* (1962), and the empirical estimates of day rate photosynthesis were recalculated. A comparison of 29 paired observations yielded means of 0.91 and 0.71 g $C \cdot m^{-2} \cdot day^{-1}$ for the ^{14}C experiments and the modified formula of Ryther and

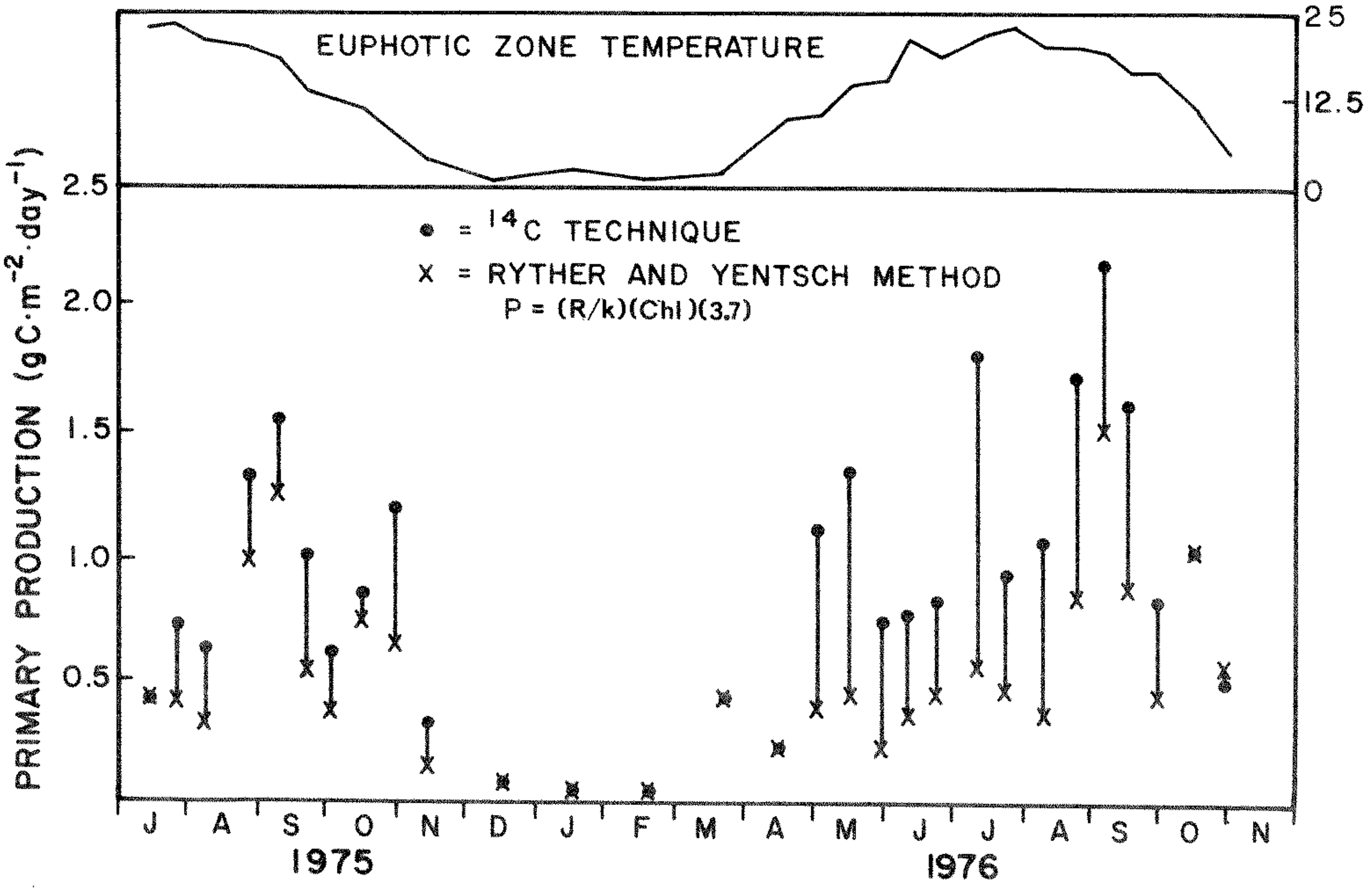

Figure 37. Differences in day rate estimates of photosynthesis at Station 2 in the Tongue River Reservoir as estimated by the ^{14}C technique and the method of Ryther and Yentsch (1957). Results are plotted against the euphotic zone temperature.

Yentsch, respectively. The mean difference expressed on a percentage basis was 22%. Although the modified Ryther and Yentsch formula was a better approximation of the ^{14}C results than the standard equation, the difference between sample means, determined by a Paired t-test, was statistically significant ($n = 29$, $p < 0.01$). The use of the locally derived assimilation ratio of 4.85 would not yield an adequate estimate of day rate photosynthesis on an annual basis.

In situ ^{14}C estimates of the assimilation ratio obtained beneath ice cover were eliminated from the data to determine if a locally derived assimilation ratio would produce an adequate estimate of primary production during the growing season only, when incorporated into Equation 1. The average *in situ* estimate of the assimilation ratio during the growing season ($5.40\ g\ C \cdot g\ Chl\ a^{-1} \cdot hr^{-1}$; $n = 25$) was substituted for the standard of 3.7 in Equation 1. An analysis of 25 paired experiments resulted in means of 1.02 and 0.88 $g\ C \cdot m^{-2} \cdot day^{-1}$ for the ^{14}C technique and the modified Ryther and Yentsch equation. The mean difference expressed in percentage was 14%. This empirical formula more closely estimated day rate photosynthesis during the combined 1975 and 1976 growing seasons than the previous two modifications estimated day rate photosynthesis for the entire study. However, the difference between means still proved to be statistically significant (Paired t-test; $n = 25$, $p = 0.05$). It is interesting to note that when the Wilcoxson Matched-Pairs Signed-Ranks Test (see previous discussion concerning experiments of Saunders, *et al.*) was applied to the data from the above three experiments in place of Paired t-tests, the results were $n = 29$, $p < 0.01$; $n = 29$, $p < 0.01$; $n = 25$, $p = 0.06$ for the three respective experiments. No fixed assimilation ratio, whether locally determined or extracted from the literature, resulted in adequate seasonal or annual estimates of day rate photosynthesis in the Tongue River Reservoir when incorporated into Ryther the Yentsch's (1957) empirical formula.

Curl and Small (1965) found no correlation between maximum assimilation ratios and total daily solar radiation. Because the highest assimilation ratios were associated with recently upwelled waters, the authors felt that the availability of nutrients was the underlying factor affecting the assimilation ratio. Spearman's Rank Correlations were calculated between *in situ* ^{14}C assimilation ratios and several environmental measures (Table 17). The results indicate that the euphotic temperature and solar input, rather than the nutrients, had a strong influence on the assimilation ratio in the Tongue River Reservoir.

An examination of Figure 37 reveals that at lower water temperatures (< 10°C) estimates via Equation 1 adequately predicted daily photosynthesis. However, the model became increasingly less accurate as the water temperature increased. In conjunction with the data of Table 17, this observation indicates that the modification of Equation 1 to incorporate a temperature dependent assimilation ratio could yield satisfactory estimates of production. The argument for an assimilation ratio that varies with euphotic temperature appears logical, because the dark reactions of photosynthesis are enzymatic, and therefore temperature dependent. A similar approach was successfully followed by Martin (1967) in Hebgen Lake, Montana.

Table 17. Results of Spearman's Rank Correlation Test as applied between *in situ* ^{14}C determined assimilation ratios (g C·g Chl $a^{-1}\cdot hr^{-1}$) and several environmental parameters. Data were collected throughout the study.

Parameter		r_s	P
Assimilation ratio-	Euphotic temperature (^{o}C)	0.54	<0.01
	Total solar radiation (ly·day^{-1})	0.43	0.02
	Whole lake Σ inorganic-N (μmoles·m^{-2})	-0.05	N.S.
	Euphotic Σ inorganic-N (μmoles·m^{-2})	-0.08	N.S.
	Whole lake PO_4 (mmoles·m^{-2})	-0.09	N.S.
	Euphotic PO_4 (μmoles·m^{-2})	-0.07	N.S.

N.S. = Non-significant; n = 29 in all cases.

In pursuit of the above discussion, a significant linear regression ($n = 19$, $r^2 = 0.58$, $p < 0.01$) was obtained between *in situ* ^{14}C determined assimilation ratios and euphotic zone temperatures for data collected between November 1975 and November 1976 (Figure 38).

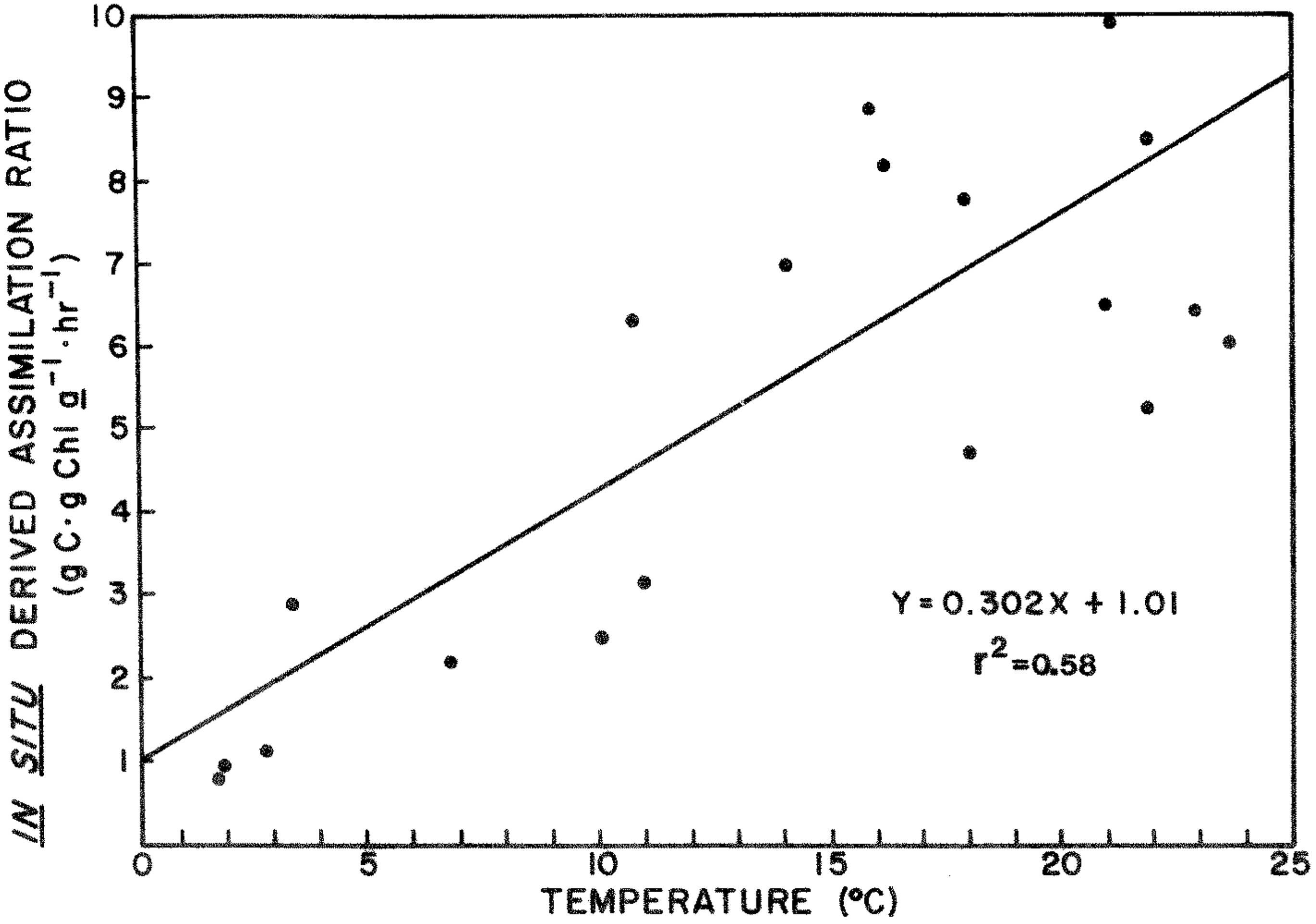

Figure 38. Regression line correlating *in situ* derived assimilation ratios and euphotic zone temperatures at Station 2 in the Tongue River Reservoir from November 1975 through November 1976.

Equation 1 was then modified as follows to incorporate a temperature dependent assimilation ratio:

$$P_w = (R/k)(Chl)(0.302\ T + 1.01) \quad (2)$$

where P_w = day rate photosynthesis in g $C \cdot m^{-2} \cdot day^{-1}$,

R, k and Chl are as previously defined,

T = euphotic temperature in °C.

The term 0.302T + 1.01 defines the assimilation ratio at light saturation in g $C \cdot g$ Chl $a^{-1} \cdot hr^{-1}$.

As a corollary, a "theoretical" temperature dependent assimilation ratio, A_{Th}, was determined for each sampling session via the following formula:

$$A_{Th} = \frac{P_{14}}{(R/k)(Chl)} \quad (3)$$

where P_{14} = day rate photosynthesis of the plankton population in g $C \cdot m^{-2} \cdot day^{-1}$ as determined from *in situ* ^{14}C incubations,

R, k and Chl are as previously defined,

A_{Th} = theoretical assimilation ratio, i.e. what value in g $C \cdot g$ Chl $a^{-1} \cdot hr^{-1}$ must the assimilation ratio assume for the Ryther and Yentsch formula to yield an estimate of production equivalent to that of the ^{14}C technique.

A statistically significant linear regression ($n = 19$, $r^2 = 0.38$, $p < 0.01$) was obtained between A_{Th} and the euphotic zone temperature

(Figure 39) for data collected during the predefined water year. As a result, Equation 1 was modified as follows:

$$P_{Th} = (R/k)(Chl)(0.284T + 2.92) \qquad (4)$$

where P_{Th} = day rate photosynthesis in g $C \cdot m^{-2} \cdot day^{-1}$,

R, k, Chl and T are as previously defined.

The term 0.284T + 2.92 defines the assimilation ratio at light saturation in g C·g Chl $\alpha^{-1}.hr^{-1}$.

Figure 40 presents the comparative estimates of photosynthetic production from November 1975 through November 1976 at Station 2 via the *in situ* ^{14}C technique and the two empirical formulas of Equations 2 and 4. Both empirical formulas yielded estimates of production that more closely coincided with the ^{14}C estimate than Equation 1 (compare Figures 37 and 40). Paired t-tests were performed to statistically compare the mean estimates of photosynthetic production taken at Station 2 on 19 dates from November 1975 through November 1976. The estimates were obtained by the ^{14}C technique and by the formulas of Equations 1, 2 and 4. The results (Table 18) indicate that Equations 2 and 4 adequately estimated primary production when the ^{14}C technique was used as a standard. Note that the regression lines incorporated into Equations 2 and 4 have similar slopes but different y-intercepts (Figures 38 and 39). The two lines should, in fact, be described by the same equation because A_{Th} should be equal to the *in situ* deter-

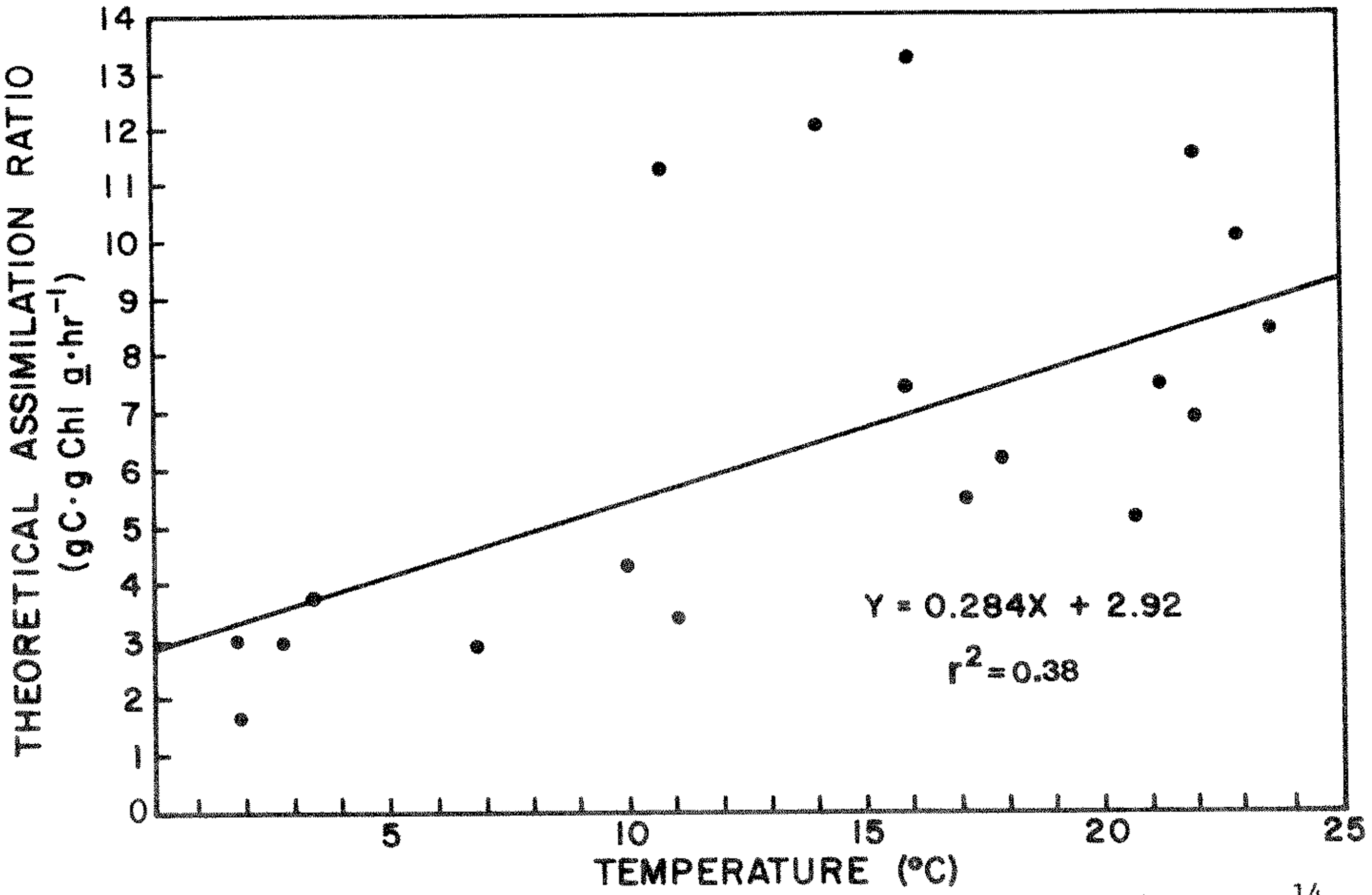

Figure 39. Regression line correlating theoretical assimilation ratios (based on ^{14}C estimates of diel photosynthesis) with euphotic zone temperatures at Station 2 in the Tongue River from November 1975 through November 1976.

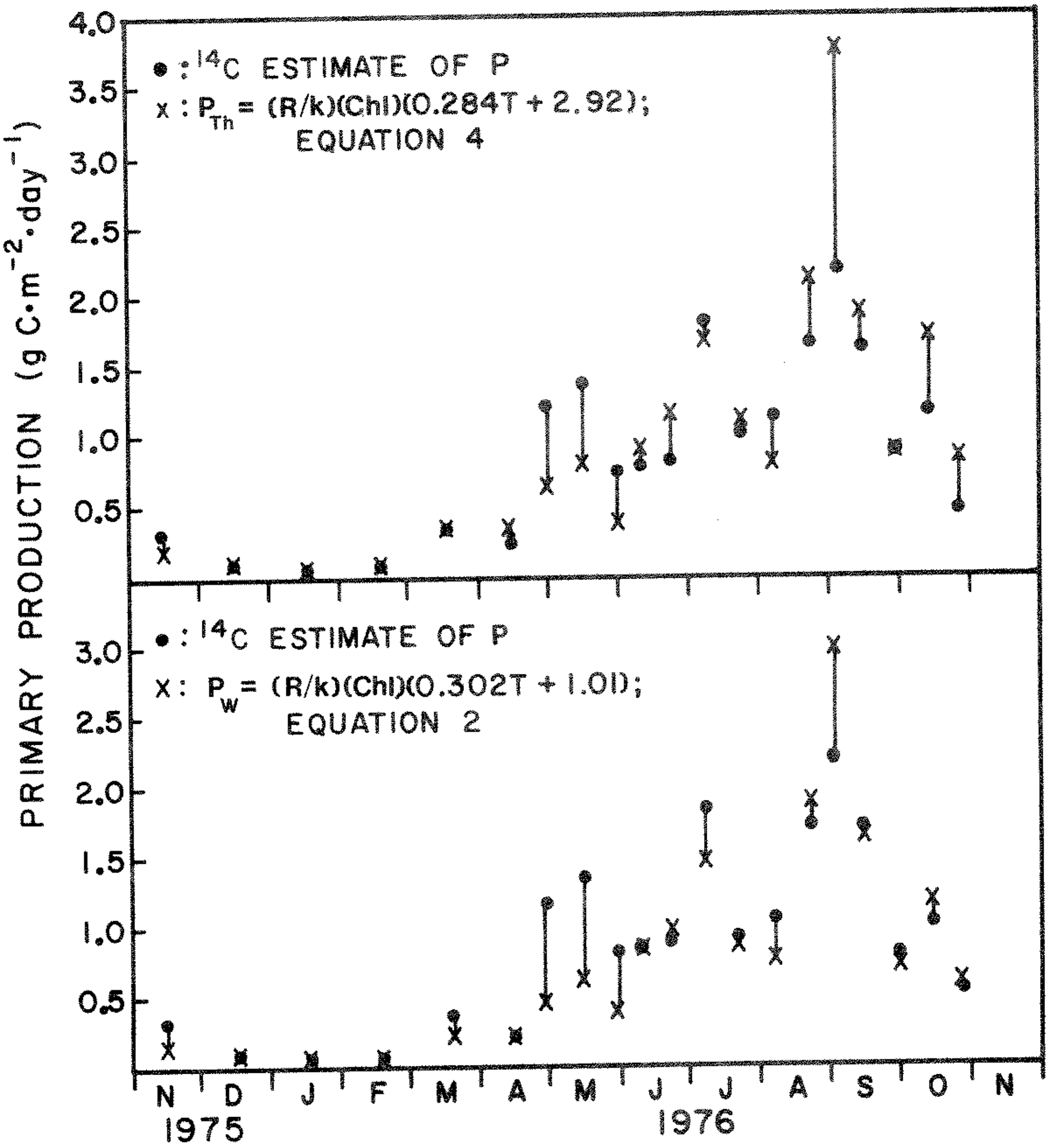

Figure 40. Comparison of ^{14}C estimates of diel photosynthesis with estimates obtained by two modifications of the Ryther and Yentsch (1957) formula. Modifications incorporate a temperature dependent assimilation ratio.

mined assimilation ratio on any given date. However, because of three anomalous A_{Th} values calculated for spring samples, the regression equation in Figure 39 yielded a line displaced upward from (although parallel to) that of Figure 38. Although Equations 2 and 4 both successfully approximated the ^{14}C mean estimate of production, the null hypothesis was rejected when comparing the mean photosynthetic estimates of the two equations (Table 18). This is because Equation 2 slightly underestimated production while Equation 4 slightly overestimated production relative to the ^{14}C estimates (Figure 40). Equation 2 was chosen over Equation 4 to calculate production at Stations 1 and 3 because the regression line describing the assimilation ratio (Figure 38) was derived from *in situ* experiments. Additionally, the fit of the regression line to the data in Figure 38 was better than that of Figure 39 ($r^2 = 0.58$ vs. $r^2 = 0.38$).

Table 19 contrasts the ^{14}C and Equation 2 estimates of photosynthetic production in several manners at Station 2 using weighted data collected from November 1975 through November 1976. Over the course of the growing season, the two estimates corresponded remarkably well, as only a 9.5% difference in estimates was noted. On an annual basis, the estimate of Equation 2 could be expected to average 12.5% lower than the ^{14}C estimate. The model tended to break down in the late winter and spring (Figure 41A) when predicted assimilation ratios were lower than the actual values determined *in situ*. Additional observations may prove that a curvilinear or semi-log function would provide a better fit to data similar to that of Figure 38. The assimilation ratios generally were higher at 15-20°C than at 20-25°C, although this may have been due to factors other than temperature.

Table 18. A statistical analysis (Paired t-tests) of several mean estimates of photosynthetic production (g $C \cdot m^{-2} \cdot day^{-1}$) at Station 2 in the Tongue River Reservoir. Data were collected from November 1975-November 1976.

	^{14}C	P_w	P_{Th}	P
$\bar{x}$ (g $C \cdot m^{-2} \cdot day^{-1}$)	0.93	0.83 (-11)[1]	1.05 (+13)	0.52 (-44)
^{14}C	-	N.S.	N.S.	$p < 0.01$
P_w	N.S.[2]	-	$p < 0.01$	$p < 0.01$
P_{Th}	N.S.	$p < 0.01$	-	$p < 0.01$
P	$p < 0.01$	$p < 0.01$	$p < 0.01$	-

[1] Number in parenthesis represents the difference in percent between the sample mean and the ^{14}C mean estimate of production.

[2] n = 19 in all cases; N.S. = Non-significant.

Figure 41B illustrates the open-water horizontal pattern of primary production at all three reservoir stations during 1976 as estimated by Equation 2. The mean daily photosynthetic rates were 0.88, 1.01, and 17.42 g $C \cdot m^{-2} \cdot day^{-1}$ at Stations 1, 2 and 3. Eliminating

Table 19. A comparison of two estimates of photosynthetic production (g $C \cdot m^{-2}$) at Station 2 of the Tongue River Reservoir from November 1975 through November 1976.

	Method		
	^{14}C technique	$P_w = (R/k)(Chl)(0.302T + 1.01)$	% Difference
Whole year	281.2	246.1	12.5
Daily-whole year	0.77	0.67	-
Winter	21.3	11.0	47.0
Daily-winter	0.16	0.08	-
Growing season	259.9	235.1	9.5
Daily-growing season	1.11	1.01	-
% annual production in growing season	92.4	95.5	-

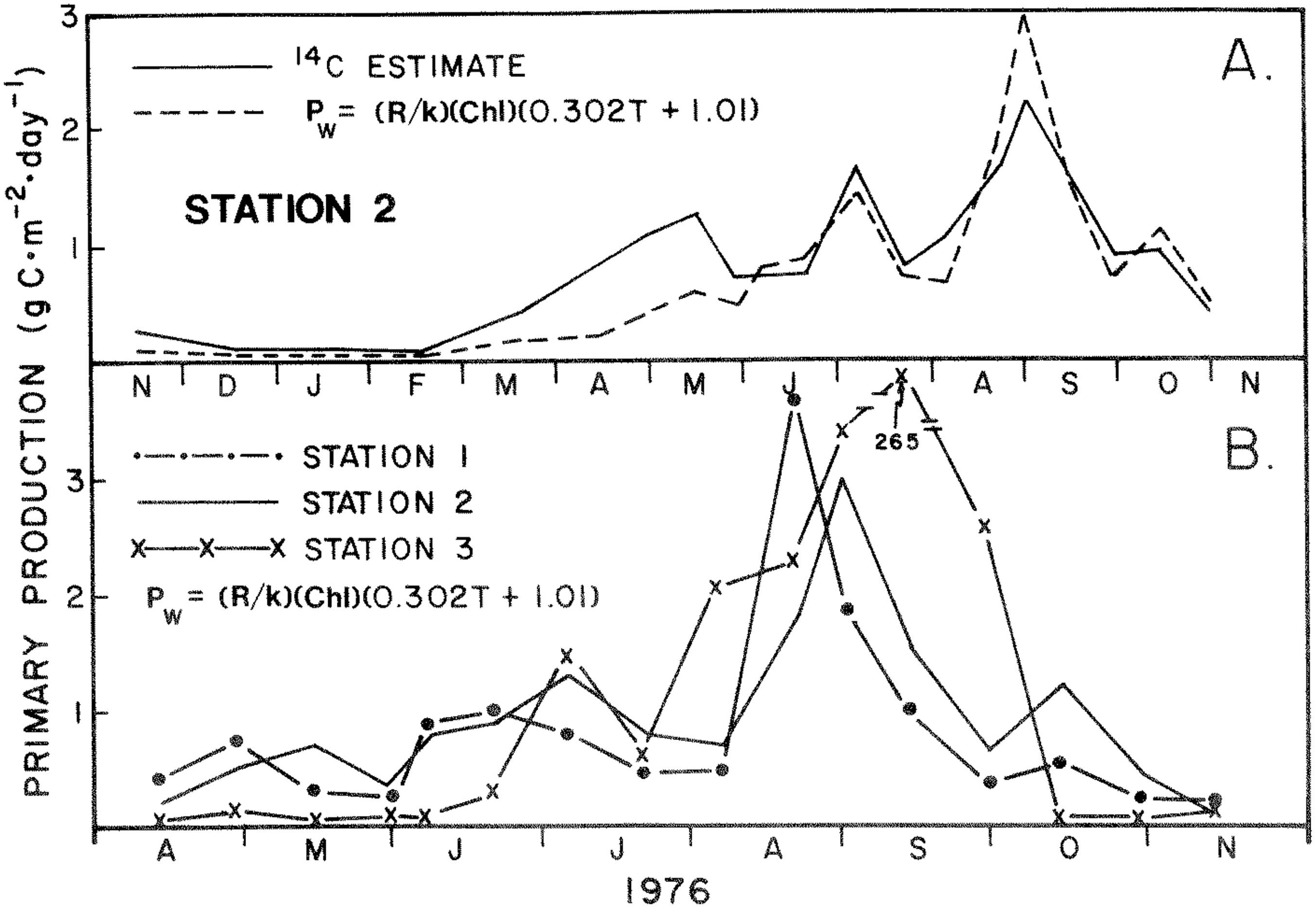

Figure 41. A. Comparison of estimates of diel photosynthesis at Station 2. Estimates were obtained by the ^{14}C method and by a modified Ryther and Yentsch (1957) formula. B. Diel photosynthesis at three Tongue River Reservoir sampling sites during the growing season of 1976 as estimated by Equation 2.

the previously mentioned irregularity at Station 3 on 16 September, the means became 0.88, 1.01, and 0.91 g $C \cdot m^{-2} \cdot day^{-1}$ at the three respective sampling sites for a whole lake average of 0.93 g $C \cdot m^{-2} \cdot day^{-1}$. A Randomized Complete-Block ANOV failed to reject the null hypothesis in all cases when used to compare daily station means of photosynthetic production, whether inclusive ($n = 16$) or exclusive ($n = 15$) of the 16 September data.

Gross photosynthesis for the entire 1976 growing season, exclusive of the anomalous data point at Station 3, was 205.9, 235.1, and 212.9 g $C \cdot m^{-2}$ at Stations 1, 2 and 3 for a whole lake average of 217.9 g $C \cdot m^{-2}$. All three reservoir stations followed the same trend relative to photosynthetic activity (Figure 41B). A gradual increase in daily production was noted from ice-out through July, followed by a sharp rise in August and September. In October and November photosynthesis declined as water temperatures dropped.

Assuming a photosynthetic quotient of unity, values of primary production in terms of g $O_2 \cdot m^{-2} \cdot day^{-1}$ can be obtained by multiplying g $C \cdot m^{-2} \cdot day^{-1}$ by a factor of 2.67.

Several assumptions were made in the use of the ^{14}C technique and the modification of Equation 1 to make predicted day rate photosynthesis closely approximate the ^{14}C estimate. For purposes of the preceding discussion, the ^{14}C method was considered to yield an estimate of gross photosynthesis. This is probably not entirely true because some respiratory and excretory losses undoubtedly occur, even during a short term incubation. The major excretory product of algal cultures is glycolate. The rate of excretion is a function of several factors, including the concentrations of CO_2 and O_2, the stage of algal development, and the intensity of photorespiration. Hence, as used in these experiments, the ^{14}C method probably measured something between net and gross photosynthesis (Steemann-Nielsen, 1963).

The enclosure of a plankton community introduces several variables into the estimation of carbon fixation. Schwoerbel (1970) discussed the differing transparencies of light at depth through various types of glass. The reduction of both turbulence and water circulation may lead to cell sedimentation or bouyant rise, thus altering the relationship of the plankton to the light and nutrient climate (Vollenweider, 1969).

As performed, the ^{14}C experiments measured the exposure rate of radiocarbon uptake. The extrapolation of results to day rate photosynthesis is subject to some criticism. Vollenweider (1965) demonstrated that in a light day divided into five equal parts, 25% of the total day rate photosynthesis would be measured during the second photoperiod and 30% during the third, assuming no nutrient limitations. This assumption was probably valid for the most part in the Tongue

River Reservoir for reasons outlined in this and a previous section. Other researchers have made the assumption that productivity is directly proportional to incident light, and have calculated day rate photosynthesis from short term experiments accordingly (see discussion in Vollenweider, 1965). The method outlined by Vollenweider (1965) was followed in extrapolating short term incubations to day rate estimates. However, a periodic incremental check of photosynthetic activity in the Tongue River Reservoir over a full light day would critically evaluate Vollenweider's generalization.

Because Equation 2 was in part derived from the results of *in situ* radiocarbon experiments, its day rate estimates are inherently subject to the same criticisms levelled against the ^{14}C technique.

There were several sources of potential error that could explain the differences between day rate estimates obtained via the ^{14}C method and Equation 2. The empirical formula assumes a homogenously distributed plankton population. Although the Tongue River Reservoir circulated freely, the presence of a significant population of Myxophyceae and/or Dinophyceae may have invalidated this assumption periodically.

The chlorophyll content of the water was used as an estimate of the standing crop in the formula. Therefore, daytime variations in chlorophyll content would affect Equation 2. The impact of this variable was minimized, however, because chlorophyll estimates were made on samples collected at the initiation of the ^{14}C experiment, i.e. the samples used to estimate chlorophyll content were consistently collected at the same time of the light day from cruise to cruise. Errors introduced by daytime variations in chlorophyll content were probably consistant from cruise to cruise. Small, *et al.* (1972) recognized this problem and accounted for variable chlorophyll content on a diel basis and at depth by estimating chlorophyll content at 2-hr increments for a whole light day and incorporating an appropriate factor into the original equation of Ryther and Yentsch.

The inability to totally extract pigments from cells was thought to be a source of error in the original equation (Ryther and Yentsch, 1957). Chlorophyll degradation products have been known to interfere with the spectrophotometric determination of chlorophyll content. Although the formula of Strickland and Parsons (1972) eliminates interferences due to magnesium-void decomposition products, the magnesium chelate chlorophyllide is measured as viable chlorophyll *a* in the calculations. High concentrations of chlorophyllide may have introduced error in the estimation of chlorophyll *a*, and hence in Equation 2.

Variations in assimilation ratios have been noted among algal groups (Strickland, 1960). Curl and Small (1965) and Small, *et al.*

(1972) felt that most of the variation in assimilation ratios can be explained by differing nutritional states. Certainly the physiological condition and age of the culture has an influence as well. The results of this study indicate that 58% of the variation in assimilation ratios in the Tongue River Reservoir from November 1975 through November 1976 was due to temperature. This was accounted for by Equation 2. The relative contributions of these other factors accounted for the deviations in day rate estimates of Equation 2 from those of the radiocarbon experiments. It is possible that the lower than expected assimilation ratios observed at high water temperatures (Figure 39) reflected a nutrient deficiency. However, the results *generally* indicate that it is unlikely that nutrient limitations were an overriding factor affecting productivity (assimilation ratio).

A favorable set of environmental circumstances facilitated the modification of the Ryther and Yentsch formula to adequately predict day rate photosynthesis in the Tongue River Reservoir using the ^{14}C technique as a standard. It is improbable that a similar modification can be extended to any other body of water other than a shallow polymictic lake. Equation 2 estimates neither net nor gross photosynthesis. Rather, it estimates some quantity between the two, probably closer to gross photosynthesis. The use of Equation 2 does not compare relative limnetic production among the three reservoir stations any more accurately than Equation 1. *For the period studied,* however, it does contrast relative photosynthesis at the three sampling sites in terms closer to absolute production, as *approximated* by *in situ* ^{14}C experiments. Fluctuating environmental conditions may necessitate the recalculation of the regression equation of Figure 38 if this concept is extended to future studies on the same body of water.

Experimental results obtained through radiocarbon experiments are reproducable to 10% (Cassie, 1962), hence Equation 2 predicted daily photosynthetic rates in the Tongue River Reservoir remarkably well relative to the ^{14}C method (Table 19).

Evaluation of the Trophic Status of the Tongue River Reservoir

There are no generally accepted guidelines for evaluating the trophic status of a body of water. Various attempts have been made to critically evaluate lentic and lotic systems relative to species composition, standing crop, rate of productivity, and nutrient (N + P) concentrations. The results of these determinations were then used as criteria by which the degree of eutrophy of a system was estimated. Several of the important results of this study relative to the nutrients, standing crop, and primary productivity of the river and reservoir are summarized in Table 20. These and other measured parameters were compared to guidelines suggested by several researchers for assessing the degree of eutrophy of aquatic systems.

To prevent the establishment of biological nuisances and to control cultural eutrophication, the EPA (1976) recommended that total phosphorus should not exceed 50 $\mu g \cdot L^{-1}$ in any river at the point where it enters a lake. The EPA (1976) further stated that total-P concentrations lower than 100 $\mu g \cdot L^{-1}$ would prevent the establishment of plant nuisances in streams not discharging directly into lakes. The mean concentration of total-P in the inflow water during the sample year exceeded the EPA guideline (Table 20). However, the average concentration of total phosphorus in the outflow water was well within the recommended upper limit for prevention of downstream nuisance plant growth.

Sawyer (1947) defined critical levels of inorganic nitrogen (300 $\mu g \cdot L^{-1}$ N) and inorganic phosphorus (10-20 $\mu g \cdot L^{-1}$ P) at the time of spring overturn in Wisconsin lakes. Levels exceeding these concentrations led to nuisance blooms of algae during the summer. Although there was no spring overturn, spring levels of inorganic nitrogen and phosphorus were low enough to meet the defined standards (Table 20).

Vollenweider (1970) agreed with Sawyer's (1947) observation and further stated that a body of water was in danger relative to its trophic status if the specific supply loading per unit lake surface area exceeded 5-10 g total-N$\cdot m^{-2} \cdot yr^{-1}$ or 0.2-0.5 g total-P$\cdot m^{-2} \cdot yr^{-1}$.

The annual surface loading of N and P in the Tongue River Reservoir far exceeded these values (Table 20).

Table 20. A summary of the means of several parameters measured in the Tongue River and the Tongue River Reservoir from November 1975 through November 1976.

Parameter	Inflow	Outflow	Whole Lake
Total phosphorus ($\mu g \cdot L^{-1}$)	67	40	-
Inorganic nitrogen ($\mu g \cdot L^{-1}$)	61	67	-
Inorganic nitrogen-spring mean ($\mu g \cdot L^{-1}$)	-	-	25
Inorganic phosphorus-spring mean ($\mu g \cdot L^{-1}$)	-	-	8
Total phosphorus-spring mean ($\mu g \cdot L^{-1}$)	-	-	54
Surface loading rate-total-N ($g\ N \cdot m^{-2} \cdot yr^{-1}$)	-	-	22.0
Surface loading rate-total-P ($g\ P \cdot m^{-2} \cdot yr^{-1}$)	-	-	4.1
Regressed molar whole lake ratio-inorganic N/inorganic P	-	-	7/1
Inorganic-N/inorganic-P, by weight (ratio derived from regressed molar ratio)	-	-	3.2/1
Total-N/total-P, by weight (ratio derived from surface loading rates)	-	-	5.4/1
Summer chlorophyll *a* ($mg \cdot m^{-3}$)	-	-	15.3
Growing season chlorophyll *a* ($mg \cdot m^{-3}$)	-	-	11.5
Daily productivity ($g\ C \cdot m^{-2}$)	-	-	0.93
Growing season productivity ($g\ C \cdot m^{-2}$)	-	-	217.9
Growing season standing crop ($mm^3 \cdot L^{-1}$)	-	-	5.12
Growing season standing crop ($cm^3 \cdot m^{-2}$)	-	-	13.33

Rawson (1955 and 1960) pointed out the influence of morphometry on productivity, especially relative to fish production in large lakes. Vollenweider (1970) attempted to improve upon his previously mentioned generalization by integrating data concerning mean depth and permissible loading rates. Following this approach, the Tongue River Reservoir was classified as highly eutrophic (Figures 42 and 43).

Shannon and Brezonick (1972) developed a trophic status index (TSI) for shallow Florida lakes by manipulating volumetric rather than areal loading rates. Fitting the Tongue River Reservoir data to this formula yielded a TSI of 21.6, compared with a value of 7.0 cited by the authors as the lower limit of eutrophy.

The ratio of nitrogen/phosphorus has received much attention relative to eutrophication. Stumm and Morgan (1970) cited an atomic ratio of 16 atoms of nitrogen to one of phosphorus in algal protoplasm. Golterman (1975) made reference to a higher figure of 30/1. The molar ratio of N/P in water has often been used to indicate which of the two macronutrients is likely to be limiting in terms of relative algal demand. A simple linear regression was calculated between molar whole lake inorganic nitrogen and molar whole lake inorganic phosphorus during the 1976 growing season. The following regression equation was obtained:

$\text{mmoles N} = 7.03\ (\text{mmoles P}) + 0.58;\ n = 48,\ r^2 = 0.32,\ p < 0.01.$

The equation predicted a molar whole lake N/P ratio of about 7/1. Similar regression equations were obtained when comparing euphotic and aphotic molar nitrogen and phosphorus concentrations. The calculated ratio does not indicate that nitrogen was necessarily limi-

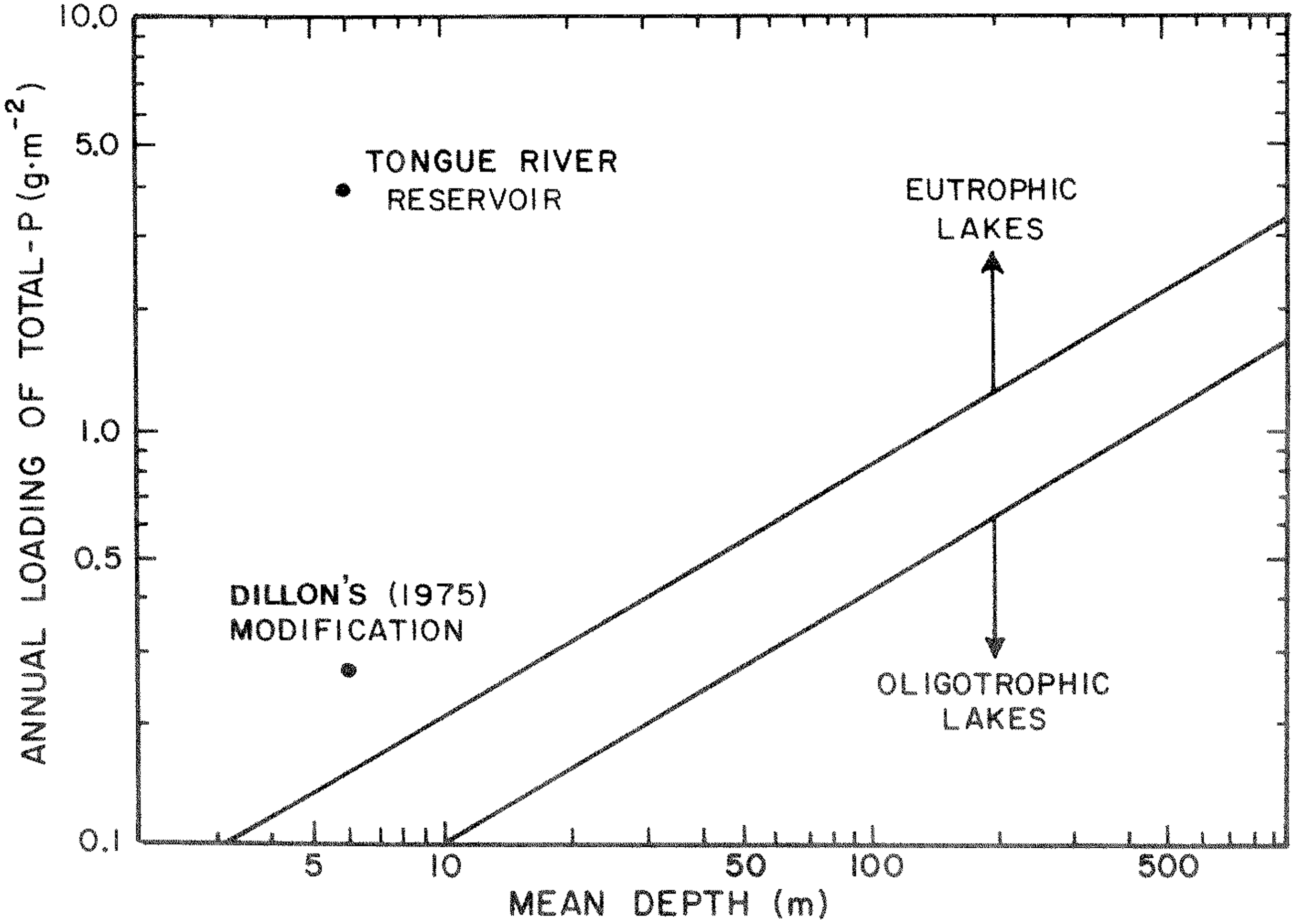

Figure 42. Vollenweider's (1970) model for the trophic classification of lakes based on annual surface loading of total phosphorus as a function of mean depth.

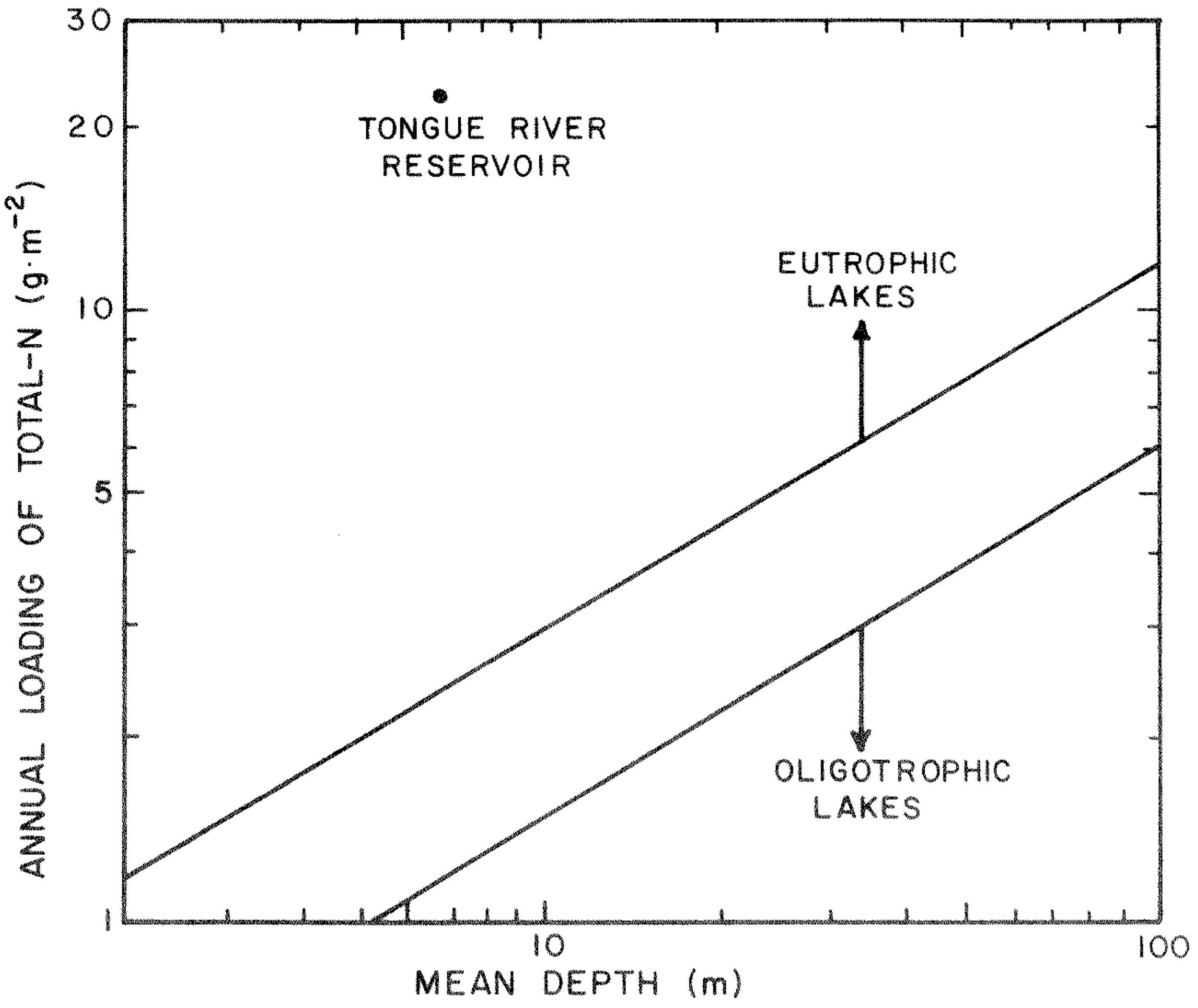

Figure 43. Vollenweider's (1970) model for the trophic classification of lakes based on annual surface loading of total nitrogen as a function of mean depth.

ting algal growth because the ratio yields no information relative to the concentrations of N and P. It does bear evidence that nitrogen was not as abundant in the water relative to phosphorus in terms of algal requirements. These observations must be viewed with caution in light of possible differences in rates of algal uptake of various nutrient species and the potential luxury uptake of both nitrogen (Foree and Scorggin, 1973) and phosphorus.

Phosphorus has long been regarded as a crucial factor influencing the size and rate of increase of phytoplankton populations. In a summary of several case studies, Schindler (1977) suggested that the C/N/P ratio in a given lake would remain constant from year to year in the absence of excessive outside perturbation. Because there were natural mechanisms that more easily compensated for deficiencies of nitrogen and carbon, Schindler (1977) felt that phosphorus was the dominant factor in controlling biomass and productivity. Based upon this reasoning, phosphorus levels have often been critically evaluated relative to eutrophication. Vollenweider (1970) founded his maximum permissible loading rates on the assumption that a tolerable N/P ratio was 15/1, by weight. The ratio of the annual loading rate of total nitrogen to total phosphorus in the Tongue River Reservoir, by weight, was 5.4/1 (Table 20), which was clearly unacceptable by the defined standards. Golterman (1975) presented evidence that in English lakes winter inorganic N/inorganic P ratios of 10/1, by weight, were indicative of nutrient-rich waters with a large summer standing crop of algae. The growing season molar inorganic N/inorganic P ratio (Table 20) was 3.2/1 when the data was converted to milligrams. The ratio for the winter months, by weight, averaged 6.4/1. Following this argument, the nutritional status of the Tongue River Reservoir was conducive to nuisance blooms of algae during the growing season. If defining permissable N/P ratios, however, the authors failed to include a discussion of nutrient concentrations.

The mean summer chlorophyll *a* concentration in the Tongue River Reservoir was 15.3 $mg \cdot m^{-3}$ (Table 20) which placed the reservoir in the "eutrophic" category of the EPA (1972). When the growing season mean of 11.5 $mg \cdot m^{-3}$ (Table 20) was used in place of the summer mean, the reservoir was still rated as eutrophic.

In an assessment of eutrophy based upon volumetric phytoplankton standing crops, Vollenweider (1970) defined the transition point of mesotrophy as a phytoplankton density of 3-5 $cm^3 \cdot m^{-3}$. The mean whole lake growing season algal standing crop in the Tongue River Reservoir was 5.12 $mm^3 \cdot L^{-1}$ ($cm^3 \cdot m^{-3}$), which indicated that the reservoir was slightly eutrophic. The author admits, however, that it is impossible to set a clear dividing line.

Rodhe (1969) presented a scheme of evaluating the trophic status of lakes by measuring mean day rates and total growing season rates of photosynthesis. The mean day rate of 0.93 g $C \cdot m^{-2}$ (Table 20) indicated that the Tongue River Reservoir was oligotrophic, yet the 1976 growing season rate of 217.9 g $C \cdot m^{-2}$ was evidence that the reservoir was naturally eutrophic.

Of five diatoms cited by the EPA (1972) as typically inhabiting nutrient-rich water, one (*Navicula cryptocephala*) appears in Tables 14 and 14 as a major specie in the reservoir, and all five were present at some time during the study. According to Palmer's (1969) algal species and algal genera pollution indices, the Tongue River Reservoir was not organically polluted. However, *Synedra ulna*, a major specie of the Tongue River Reservoir (see Tables 14 and 15) was rated by Palmer's (1969) composite species index as moderately tolerant of organic pollution. All species of limmetic algae listed as dominants in mesotrophic lakes of western Canada by Rawson (1956) were found in the Tongue River Reservoir. Two (*Aphanizomenon flos-aqua* and *Fragilaria crotenensis*) were dominant in the reservoir seasonally.

By the previously discussed standards, the Tongue River Reservoir appeared to be eutrophic. The inflowing concentration of inorganic phosphorus exceeded the recommended standard and the N/P ratio of the lake water was unacceptable (ignoring concentrations). The surface loading rates of total nitrogen and total phosphorus were too high, as was the mean summer chlorophyll concentration. The mean volumetric growing season algal standing crop bordered acceptability.

However, the previously mentioned guidelines were established for lakes with surface outflows and disregarded the influence of flushing. Dillon (1975) pointed out this shortcoming in Vollenweider's (1970) assessment of critical surface loading rates. He noted that the rapid passage of a large volume of water low in nutrients through a lake would yield a high loading rate, but would not result in eutrophy.

For the water year examined, the total outflow of the reservoir was 3767.4 X 10^5 m^3 and the average reservoir storage was 484.2 X 10^5 m^3 which led to a calculated flushing rate of $7.78 \cdot yr^{-1}$. Dillon's formula accounted for the flushing rate and the retention coefficient (0.49 in this case) of the influent phosphorus load. The application of Dillon's (1975) model to the Tongue River Reservoir resulted in a calculated annual total-P loading rate of 0.25 $g \cdot m^{-2}$. This drastically reduced the predicted degree of eutrophy of the reservoir (Figure 42) if the vertical distance from the line of dangerous loading was semi-quantitatively related to the degree of eutrophy, as Vollenweider (1970) suggested.

Dillon and Rigler (1974) presented an empirical relationship to predict the mean summer chlorophyll *a* content of a lake based upon the concentration of total phosphorus in the water at spring overturn. When applied to data of this study, the formula predicted a mean summer chlorophyll *a* concentration of 23.5 $mg \cdot m^{-3}$ in the Tongue River Reservoir. Compared to the measured mean of 15.3 $mg \cdot m^{-3}$ (Table 20), the estimate was in error by +54%, partially as a result of the rapid flushing rate of the reservoir.

Other extenuating factors limit the usefulness of even Dillon's (1975) formula when applied toward the evaluation of a body of water exhibiting the morphometric and hydraulic characteristics of the Tongue River Reservoir. The long, narrow nature of the reservoir (Figure 2), its deepwater gate, and the short hydraulic retention time (Table 2) weakened the effect of a high nutrient load.

Oglesby (1977) and Schaffner and Oglesby (1978) felt that the total phosphorus concentration was inadequate when used to predict phytoplankton response to phosphorus loading. It was suggested that biologically available phosphorus (BAP), defined as the sum of soluble reactive (herein designated as orthophosphorus), soluble unreactive (inferred to as dissolved organic-P, i.e. persulfate digestion of a filtered sample) and potentially desorbable phosphorus was more highly correlated to the algal standing crop.

Data presented earlier showed that turbidity was correlated with organic + hydrolyzable-P in the inflowing water of the reservoir and that the reservoir acted as a sediment trap. Of the total phosphorus load annually received by the reservoir, it is likely that the bulk of the phosphorus was tightly sorbed to siltaceous matter and was therefore unavailable for algal consumption.

An evaluation of the temporal pattern of nutrient loading is imperative to clearly evaluate the trophic status of the Tongue River Reservoir. Data in Figure 13 showed that a high winter loading of inorganic nutrients occurred in the Tongue River inflow, but that a high winter export was also characteristic of the outflow (Figure 13). Figures 13 and 14 demonstrated that inorganic-N, inorganic-P, organic + hydrolyzable-P and organic -N loads were substantial in the inflow during runoff, but that when a 30 day hydraulic retention time was considered, outflow loading was also great.

Table 21 elucidates the seasonal loading patterns of various nutrients in the Tongue River Reservoir inflow and outflow waters. Of the annual inorganic nitrogen load in the inflow, 55% was realized in the winter months (December through March, inclusive), when biological activity was at a minimum. Sixty-two percent of the import was exported within the same time span. In essence, a maximum of 38% of the high winter import was available for algal consumption upon ice-out. Of the inorganic nitrogen input received during the 15 April-

Table 21. An analysis of the seasonal loading patterns of inorganic and organic nutrients in the inflow and outflow waters. Data were collected from November 1975 through November 1976.

Inorganic-N load (kg)

Period	Inflow	Outflow	Outflow/Inflow (%)
1 Dec-31 Mar	15,631	9,644	62
4/15-6/30; 5/15-7/30	9,979	11,526	116
Annual	28,307	25,703	91
Winter/Annual (%)	55	38	-
Runoff/Annual (%)	35	44	-
Winter % + Runoff %	90	82	-

Inorganic-P load (kg)

Period	Inflow	Outflow	Outflow/Inflow (%)
1 Dec-31 Mar	4,365	1,563	36
4/15-6/30; 5/15-7/30	4,186	1,658	40
Annual	10,391	3,990	38
Winter/Annual (%)	42	39	-
Runoff/Annual (%)	40	42	-
Winter % + Runoff %	82	81	-

Organic-N load (kg)

Period	Inflow	Outflow	Outflow/Inflow (%)
1 Dec-31 Mar	21,202	18,121	86
4/15-6/30; 5/15-7/30	92,620	78,140	84
Annual	153,800	153,873	100
Winter/Annual (%)	14	12	-
Runoff/Annual (%)	60	51	-
Winter % + Runoff %	74	63	-

Org + Hydro-P load (kg)

Period	Inflow	Outflow	Outflow/Inflow (%)
1 Dec-31 Mar	1,935	1,248	65
4/15-6/30; 5/15-7/30	16,785	7,257	43
Annual	23,383	13,196	56
Winter/Annual (%)	8	9	-
Runoff/Annual (%)	72	55	-
Winter % + Runoff %	80	64	-

30 June spring runoff period (35% of the annual), 116% was exported after a short retention time. In total, 90% of annual inorganic nitrogen influent to the Tongue River Reservoir was received during times when its biological availability was at a minimum due to winter conditions or a reduced hydraulic retention time.

A congruent argument applies to inorganic phosphorus (Table 21) as 82% of the import was realized during the above mentioned periods. In contrast to inorganic-N, the reservoir acted in a conservative nature toward the inorganic phosphorus input (compare outflow/inflow ratios). However, phosphorus was demonstrated to have limited mobility within the reservoir.

A disproportionate fraction of the annual influent organic-N and org. + hydro.-P (60 and 72% respectively) loads were received during spring runoff (Table 21). Although these nutrients may not have been readily available for algal assimilation, bacterial transformations certainly could have increased their availability. As it were, 84 and 43% of the spring input of organic-N and org. + hydro.-P were exported within 30 days of import, limiting their potential algal benefit.

Golterman (1975) stated that two different definitions of eutrophication exist: An increase in nutrient supply, and the increase in plant growth automatically associated with the increased nutrient supply. In the ensuing discussion, the Tongue River Reservoir will be characterized according to both definitions. The growing season phytoplankton standing crop of the Tongue River Reservoir was moderately large. Blooms of *Aphanizomenon flos-aqua* and *Glenodinium gymnodinium* were noted at Station 3 late in the summer. The rates of daily and seasonal euplanktonic carbon assimilation were indicative of a moderately productive system. Considering the lack of macrophytic, epiphytic and periphytic activity, the Tongue River Reservoir had a low rate of total primary production.

Heavy annual surface loading of nutrients was characteristic of the reservoir. However, the short hydraulic retention time and temporal loading pattern dampened the predicted effect of the nutrients. Typically moderate concentrations of inorganic nutrients were observed free in the water column on a whole-lake basis (Figures 23-25). Predictions of a lake's trophic status that are based upon observed nutrient concentrations must always be viewed with caution because such concentrations only indicate the difference between amounts absorbed in biomass and the initial amount biologically available (EPA, 1972).

The calculated N/P ratio of the reservoir was exceedingly low (Table 20) due to the low inflow ratio, the conservation of phosphorus within the reservoir, and the total export of the annual nitrogen import. The Myxophyceae are generally considered to be thermophils, appearing late in the summer. Their competitive advantage in the presence of low inorganic-N concentrations lies in their ability to fix atmospheric nitrogen. This is apparently confined primarily to heterocystous forms, as non-heterocystous blue-greens have been observed to fix molecular nitrogen only under anaerobic conditions (Stewart, 1977).

Although evidence presented in previous sections indicates that nutrient conditions did not *seriously* limit the total algal biomass or production rate, conditions certainly favored some forms over others. Rhee (1974, 1978) discussed the role of the N/P ratio in influencing competition and species succession in algae. A low N/P ratio, reduced nutrient concentrations, and warm water temperatures late in the summer favored heterocystous blue-green algae in the Tongue River Reservoir. The blue-greens observed at all three reservoir stations were predominately the heterocystous *Aphanizomenon flos-aqua*. The bloom of *A. flos-aqua* (Figure 33) would have been a nuisance had it occurred farther down-reservoir, or had it drifted to the lower end of the reservoir. Instead, the bloom was confined to the Station 3 area which had a lesser recreational value. The pulse of *G. gymnodinium* was of a lesser magnitude and of no concern from a recreational standpoint. However, a nuisance bloom did develop in 1977 (Leathe, personal communication), and it spread to the lower reaches of the reservoir.

The classical parameters used to judge the degree of eutrophy have limited applicability to the Tongue River Reservoir. Based upon all data collected, the Tongue River Reservoir was characterized as naturally eutrophic to a mild degree. However, the potential exists for the development of whole lake pulses of heterocystous blue-green algae that could impair the recreational value of the lake. No deleterious effects would be realized from the standpoint of downstream irrigation as a result of algal pulses in the reservoir.

Mine Discharge

Current surface mining operations at the West Decker Mine are exploiting the uppermost (Dietz-1) coal seam. Of the various parent materials in the geological structure of the Decker area, only the coal beds are known to provide continuous aquifers of any significance. The Dietz-1 (D-1) coal seam serves as one of the more important groundwater aquifers (Van Voast, 1974). Lateral and vertical recharge to the disturbed mine site are via the D-1 and D-2 coal beds, respectively, along a hydraulic gradient. Ground water draining into the mine site and leachate to the mine site from disturbed overburden are collected in a settling pond. This wastewater (herein called mine discharge water or mine effluent) is subsequently discharged into the Tongue River flood plain or is used for dust abatement within the mine site. A schematic representation of the origin and fate of the West Decker Mine discharge water appears in Figure 44.

Figure 45 presents ionic polygonic diagrams of the relative equivalent concentrations of major ionic species at several sampling sites. Presented in parenthesis is the concentration of total cations ($me \cdot L^{-1}$) for each sampling site. The sum of the cations agreed with the sum of the anions at each respective site to within 3%. The relative and absolute concentrations of ionic constituents was the same at the Inflow and TRBM stations. Although the relative ionic composition of the Tongue River Reservoir was identical to that of the river stations, the absolute ionic concentration was slightly lower than the river values.

The ionic polygonic diagram of the Ground Water was based upon data presented by Van Voast and Hedges (1975- p . 13) for observation wells in the Decker area. Data from 14 wells in the D-1 and D-2 coal seams were averaged in the construction of the diagram. The ground water of the coal bed observation wells was predominantly a sodium bicarbonate type. Ground water samples containing high concentrations of calcium, magnesium, and sulfate were taken from areas where the coal beds were in close hydrologic connection with sandstone or alluvium

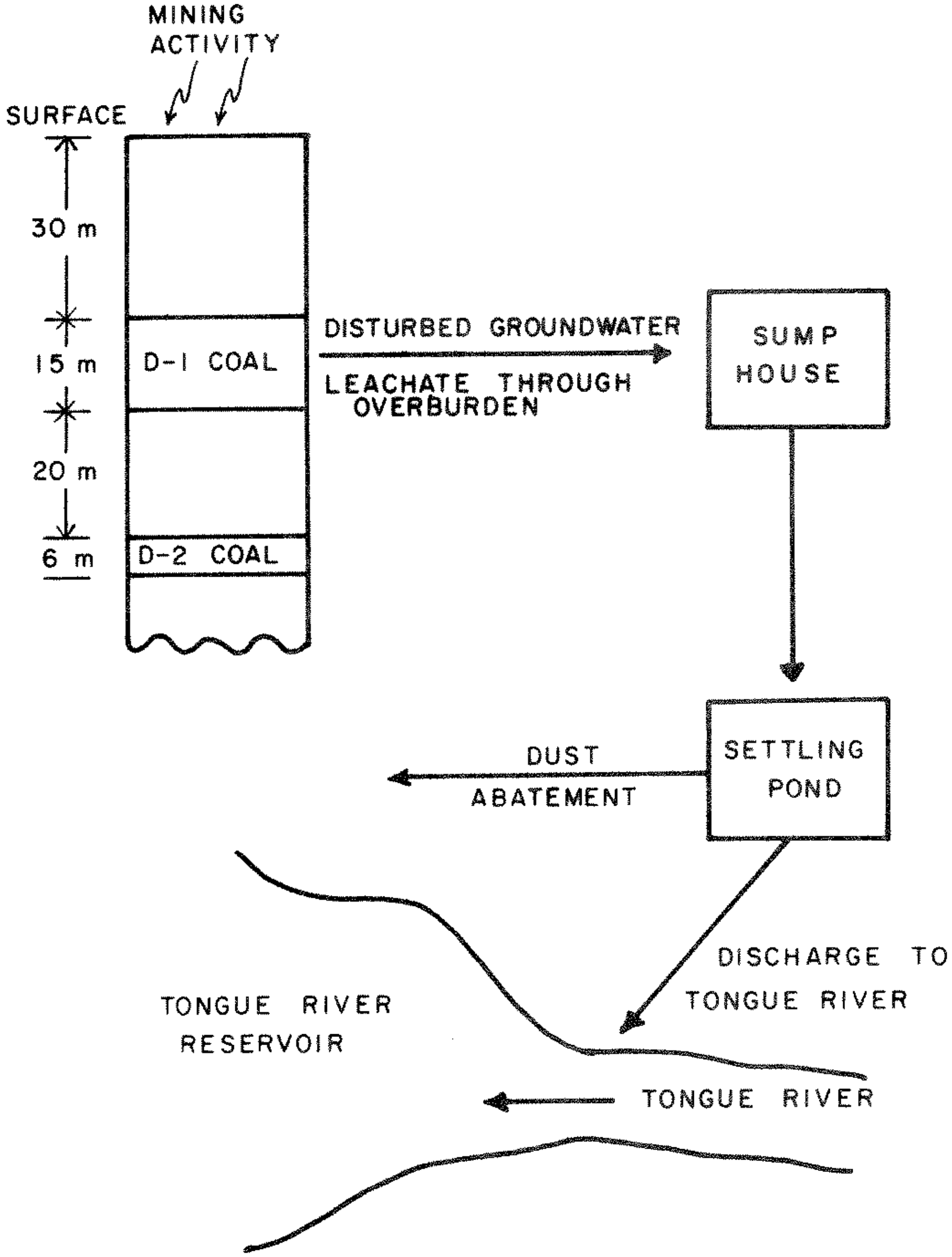

Figure 44. Schematic representation of the origin and fate of the West Decker Mine discharge water.

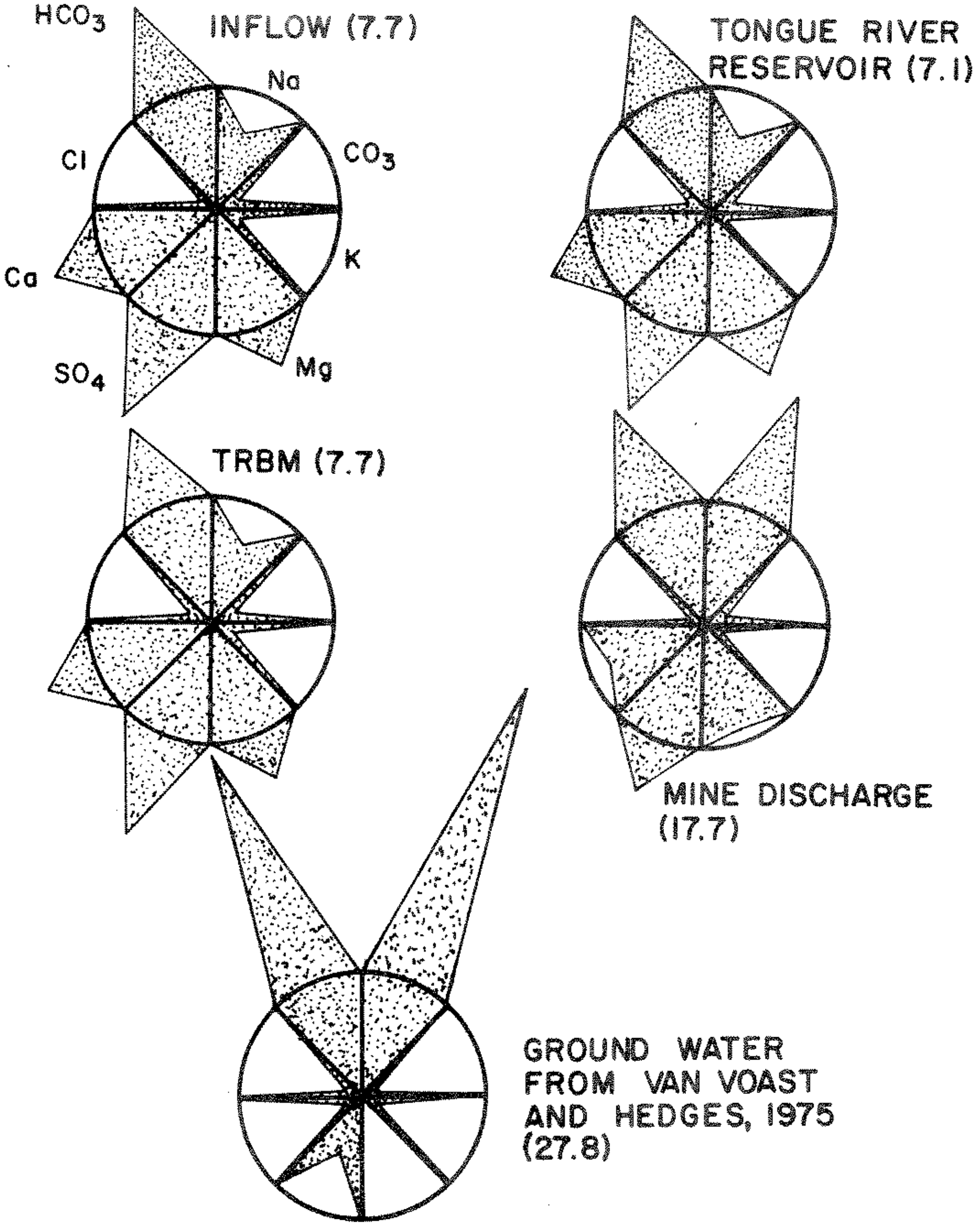

Figure 45. Ionic polygonic diagrams of the mean relative concentrations in equivalent percentages of the major cations and anions at several sampling sites. The number in parentheses represents the sum of the cations ($me \cdot L^{-1}$).

(Van Voast and Hedges, 1975). Because of the potential variability of the ground water analyses with sampling location, it is difficult to predict how closely the mine discharge water resembled the actual ground water cell of the area. As a consequence, the influence of percolation through the overburden on the mine water chemistry cannot be ascertained. The only generalization that can be made is that the mine discharge water did not chemically resemble other surface waters or the *average* ground water of the area in terms of relative or absolute ionic content. Rather, the mine discharge water was chemically somewhere between the two.

The means of several physical and chemical measurements taken on samples collected throughout the study at the Inflow and Mine Discharge stations are compared in Table 22. The dominant cation in the mine discharge water was sodium, while the dominant anions were bicarbonate and sulfate. Significant differences were noted between the inflow and mine discharge waters, especially with regard to mean values of specific conductance, total alkalinity, sodium adsorption ratio (SAR), sodium, sulfate, nitrate, and ammonia.

The use of nitrate explosives sporadically introduced nitrogenous compounds into the spoils of the mining area (Figure 46) from which they were subsequently leached by ground water and added to the mine effluent (U.S.G.S. and Mt. Dept. State Lands, 1977).

Table 22. The calculated and measured effects of the mine discharge upon the Tongue River with respect to several parameters. Data were collected from June 1975 through November 1976.

Parameter	Tongue River above discharge (Inflow)	Mine Discharge station	Hypothetical river below discharge	Tongue River below discharge (TRBM)
Spec·cond. (μmhos·cm^{-1})	693	1498	694	696
Total alk. (as $CaCO_3$)	202	509	203	203
Ca (mg·L^{-1})	61.2	33.1	61.2	61.0
Mg (mg·L^{-1})	37.8	42.4	37.8	37.8
Na (mg·L^{-1})	30.5	253.3	30.6	30.4
SO_4 (mg·L^{-1})	166.4	295.1	166.4	166.4
pH	8.6	8.5	8.6	8.4
SAR[1]	0.78	6.88	0.78	0.78
NH_3-N (μg·L^{-1})	16	282	16	16
NO_3-N (μg·L^{-1})	31	287	31	30

[1] Sodium adsorption ratio.

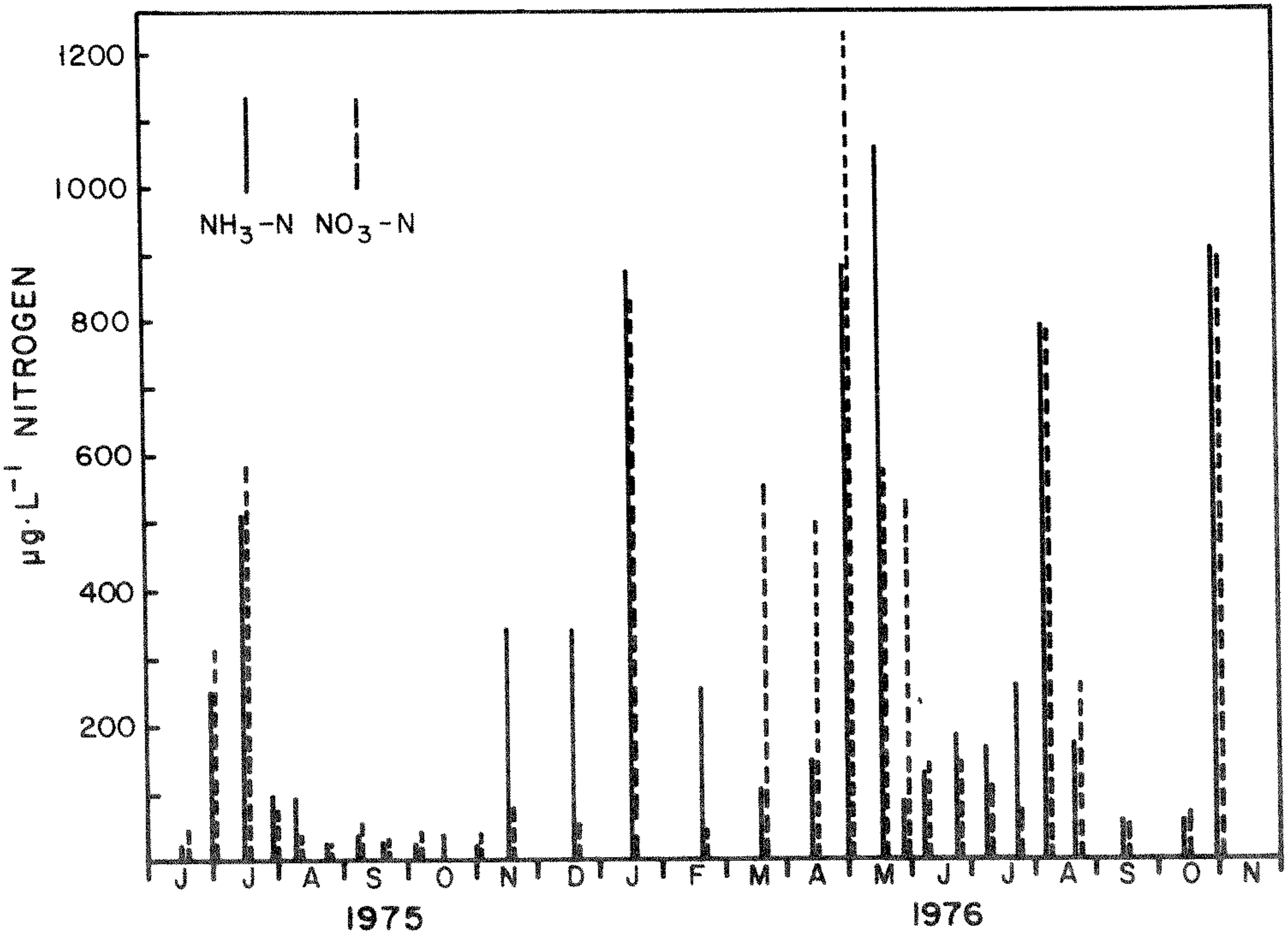

Figure 46. Ammonia and nitrate concentrations measured in the mine discharge water from June 1975 through November 1976.

The increased sulfate concentration of the mine discharge water relative to the ground water and river water was possibly due to the leaching of sulfate derived from pulverized gypsum and microbially oxidized pyrite in the spoils. Recharge to the disturbed site by ground water of high sulfate concentration was also possible.

All ground water analyses of observation wells in coal beds (Van Voast and Hedges, 1975) were high in sodium content relative to calcium and magnesium. For this reason, the mine discharge water could be expected to have a high sodium content, and consequently a high sodium adsorption ratio.

Table 23 depicts the average biweekly percent contribution of the mine discharge to the Tongue River flow over the course of this study. Data are presented in cubic feet per second as opposed to $m^3 \cdot sec^{-1}$ becuase of the low rate of flow associated with the mine discharge. The maximum percent contribution of the mine discharge to the Tongue River rate of flow was 0.35% and the mean percent contribution was 0.09% for the entire study. Although the mine discharge and Tongue River waters were chemically different, the rate of flow of the mine discharge was low enough that no significant changes should have been effected in the river water. This observation was substantiated by calculating hypothetical "pollution" loads based upon the mean chemical analysis and rate of flow of the mine discharge. The cal-

Table 23. Contribution of the mine discharge to the Tongue River flow rate from June 1975 through November 1976.

	Average mine discharge rate (cfs) for period ending..	Average Tongue River flow rate (cfs) for period ending...	% contribution of mine discharge to Tongue River flow
25 June 1975	0.68	3175	0.02
9 July	1.21	3957	0.03
23 July	1.12	1894	0.06
6 August	0.68	608	0.11
20 August	0.45	346	0.13
3 September	0.27	317	0.09
17 September	0.06	185	0.03
30 September	0.28	192	0.15
15 October	0.17	205	0.08
30 October	0.39	286	0.14
13 November	0.35	295	0.12
25 November	0.44	255	0.17
11 December	0.52	267	0.19
23 December	0.37	247	0.15
8 January 1976	0.31	248	0.13
21 January	0.29	286	0.10
4 February	0.34	316	0.11
18 February	0.30	337	0.09
4 March	0.30	420	0.07
17 March	0.23	404	0.06
31 March	0.17	336	0.05
14 April	0.13	236	0.06
28 April	0.32	314	0.10
12 May	0.11	425	0.03
26 May	0.15	1037	0.02
9 June	0.27	1612	0.02
23 June	0.61	1565	0.04
7 July	0.66	1267	0.05
21 July	0.67	559	0.12
4 August	0.83	236	0.35
18 August	0.31	313	0.10
2 September	0.21	192	0.06
15 September	0.01	150	0.01
29 September	0.43	217	0.20
14 October	0.05	283	0.02
27 October	0.15	283	0.06
17 November	0.00	248	0.00
Mean	0.37	425	0.09

culated chemical composition of the hypothetical river below the mine effluent was compared to measured parameters at the river station below (TRBM) the site of effluent discharge to the river (Table 22). Measured chemical and physical differences between the two river stations were slight, certainly beyond the limits of analytical detection. Deviations between the actual analysis and the hypothetical value for a given parameter at the TRBM Station were due to random error.

Of special note is the high mean pH of the mine discharge water (Table 22). Acid leachate through spoils has sometimes been associated with mining activity in the eastern United States. The high buffering capacity of the mine effluent and the low sulfur content of the sub-bituminous B-grade coal being mined obviate this problem at the West Decker Mine.

Table 24 was constructed to determine if the mine effluent could significantly affect the chemistry of the river water at some time during the year. The mean monthly averages of selected chemical parameters of the Tongue River and the calculated mean monthly changes generated by the effluent water as depicted. No noteworthy changes in river water quality were predicted *for the parameters measured* based upon present flows and concentrations.

The problem of sodic-soils has long been recognized in the arid

Table 24. Calculated chemical influence of mean monthly mine discharge on mean monthly Tongue River flow.

TONGUE RIVER: MONTHLY AVERAGES OF SELECTED PARAMETERS (6/75-11/76) WITHOUT MINE EFFLUENT AND *WITH MONTHLY AVERAGE* (6/75-11/76) *MINE EFFLUENT.*

Month	Mean (1965-76) Tongue River flow(cfs)	Mean mine discharge-cfs (6/75-11/76)		Monthly averages of analyses (6/75-11/76) Tongue River							
				Na (mg·L^{-1})		SO$_4$ (mg·L^{-1})		Spec. cond.		SAR	
January	216	0.28	(0.13)[1]	31.2	*31.5*	187.0	*187.1*	860[2]	*861*	0.70	*0.71*
February	283	0.30	(0.11)	42.6	*42.8*	243.0	*243.0*	1001	*1002*	0.90	*0.91*
March	425	0.26	(0.06)	54.9	*55.0*	300.0	*300.0*	1072	*1072*	1.12	*1.12*
April	419	0.23	(0.05)	38.4	*38.5*	213.6	*213.7*	822	*823*	0.90	*0.90*
May	1215	0.13	(0.01)	26.4	*26.4*	137.4	*137.4*	577	*577*	0.66	*0.66*
June	2035	0.66	(0.03)	9.3	*9.4*	38.6	*38.7*	227	*227*	0.32	*0.32*
July	592	0.91	(0.15)	19.0	*19.4*	76.7	*77.2*	422	*424*	0.56	*0.57*
August	205	0.43	(0.21)	33.6	*34.1*	177.0	*177.1*	725	*727*	0.82	*0.84*
September	265	0.17	(0.06)	38.7	*38.8*	213.5	*213.5*	839	*839*	0.87	*0.87*
October	295	0.28	(0.10)	31.6	*31.8*	185.2	*185.2*	746	*747*	0.74	*0.75*
November	295	0.40	(0.15)	35.3	*35.6*	212.0	*212.1*	843	*844*	0.82	*0.83*
December	215	0.42	(0.20)	45.6	*46.0*	266.5	*266.5*	1043	*1044*	0.90	*0.91*
Annual	536	0.37	(0.07)	33.8	*34.0*	287.5	*187.6*	765	*766*	0.78	*0.78*

[1] Percent contribution of mine discharge to total river flow in parentheses.

[2] Specific conductance reported as μmhos·cm^{-1} @ 25° C.

regions of the western United States. As a result, attention has focused upon the sodium adsorption ratios (SAR's) of effluent waters associated with mining activity in the west and the Northern Great Plains. The sodium adsorption ratio of a soil has generally been regarded as an effective indicator of the suitability of that soil as a medium for plant growth. Bernstein (1967) stated that irrigation water having an SAR value lower than 8 will not cause the breakdown of soil structure. He also noted that SAR values less than 4 will not cause nutritional disturbances in crops. The calculated SAR values of the river water remained virtually unaffected by the input of the mine water (Table 24) despite the high SAR values (range 5-12) of the mine effluent. The mean annual SAR value of the Tongue River was 0.78, well below the defined upper limit of acceptability for the purpose of irrigation. It should also be noted that soil salinity (total ionic constituents) and soil texture also play an integral role in plant nutrition.

With the three mines (Figure 2) operating simultaneously, Van Voast and Hedges (1975) predict a maximum effluent flow rate of 6 cfs. Chemical analyses of effluent waters from the North Decker and East Decker mines are not predicted to differ appreciably from that of the West Decker Mine (Van Voast and Hedges, 1975). As an example of a hypothetically constructed situation, assume the following: all three mines operate simultaneously; the maximum effluent flow rate with three mines in operation (6 cfs) is observed; this effluent consists of the most highly mineralized effluent water detected during this study; the effluent is channeled directly into the river. Even under these conditions, little change in river water quality was calculated (Table 25). SAR values were still well below the critical limit. If this situation were to occur, no detrimental effects should be noted from the slight increases in sodium and sulfate. In the absence of any mine effluent whatsoever, year to year variability in the river's chemical composition would more than account for these differences.

When the 1965-76 flow data for the Tongue River were examined, it was found that unusually low rates of flow were recorded in 1966 in the Tongue River above the reservoir. Table 26 was constructed under the same assumptions as Table 25, but Table 26 contains one major change: to simulate extremely severe conditions, the 1966 monthly average flow rates (cfs) were substituted for the long-term (1965-1976) monthly average flow rates (cfs) of Table 25.

If the hypothetical situation created in Table 26 were to take place, major increases in the concentrations of sulfate and sodium would be noted in the Tongue River below the site of mine effluent discharge (assuming a highly mineralized mine effluent). These

Table 25. Calculated chemical influence of maximum predicted flow of highly mineralized mine effluent on average Tongue River flow.

TONGUE RIVER: MONTHLY AVERAGES OF SELECTED PARAMETERS (6/75-11/76) WITHOUT MINE EFFLUENT AND *WITH MOST HIGHLY MINERALIZED MINE EFFLUENT RECORDED* (7/1/75).

Month	Mean (1965-76) Tongue River flow(cfs)		Mine discharge (cfs) (6/75-11/76)	Monthly averages of analyses (6/75-11/76) Tongue River							
				Na (mg·L^{-1})		SO$_4$ (mg·L^{-1})		Spec. cond.		SAR	
January	216	*222*	6 (2.7)[1]	31.2	*39.7*	187.0	*193.8*	860[2]	*889*	0.70	*0.90*
February	283	*289*	6 (2.1)	42.6	*48.9*	243.0	*247.1*	1001	*1020*	0.90	*1.04*
March	425	*431*	6 (1.4)	54.9	*58.9*	300.0	*301.9*	1072	*1084*	1.12	*1.21*
April	419	*425*	6 (1.4)	38.4	*42.7*	213.6	*216.8*	822	*838*	0.90	*1.01*
May	1215	*1221*	6 (0.5)	26.4	*27.9*	137.4	*138.8*	577	*583*	0.66	*0.70*
June	2035	*2041*	6 (0.3)	9.3	*10.3*	38.6	*39.8*	227	*232*	0.32	*0.35*
July	592	*598*	6 (1.0)	18.0	*21.3*	76.7	*80.3*	422	*437*	0.56	*0.67*
August	205	*211*	6 (2.8)	33.6	*42.5*	177.0	*184.5*	725	*760*	0.82	*1.07*
September	265	*271*	6 (2.2)	38.7	*45.5*	213.5	*218.5*	839	*863*	0.87	*1.03*
October	295	*301*	6 (2.0)	31.6	*37.9*	185.2	*190.3*	746	*770*	0.74	*0.90*
November	266	*272*	6 (2.2)	35.3	*42.1*	212.0	*217.0*	843	*867*	0.82	*1.00*
December	215	*221*	6 (2.7)	45.6	*53.7*	266.5	*271.2*	1043	*1067*	0.90	*1.08*
Annual	536	*542*	6 (1.1)	33.8	*37.3*	187.5	*190.4*	765	*778*	0.78	*0.87*

[1] Percent contribution of mine discharge to total river flow in parentheses.

[2] Specific conductance reported as µmhos·cm^{-1} @ 25° C.

Maximum predicted discharge with three mines operating: 6 cfs.
Most highly mineralized mine discharge recorded 7/1/75:

Spec. cond. = 1940 µmhos·cm^{-1} @ 25° C
Na = 345 mg·L^{-1}
SO$_4$ = 440 mg·L^{-1}
SAR = 9.09

Table 26. Calculated chemical influence of maximum predicted flow of highly mineralized mine effluent on minimum Tongue River flow.

TONGUE RIVER: MONTHLY AVERAGES OF SELECTED PARAMETERS (6/75-11/76) WITHOUT MINE EFFLUENT AND *WITH MOST HIGHLY MINERALIZED MINE EFFLUENT RECORDED* (7/1/75).

Month	1966 Tongue River flow (cfs)		Mine discharge rate (cfs)	Monthly averages of analyses, Tongue River (6/75-11/76)							
				Na (mg·L^{-1})		SO$_4$ (mg·L^{-1})		Spec. cond.		SAR	
January	108	*114*	6 (5.3)[1]	31.2	*47.7*	187.0	*200.3*	860[2]	*917*	0.70	*1.08*
February	156	*162*	6 (3.7)	42.6	*53.8*	243.0	*250.2*	1001	*1036*	0.90	*1.16*
March	242	*248*	6 (2.4)	54.9	*61.9*	300.0	*303.4*	1072	*1093*	1.12	*1.27*
April	265	*271*	6 (2.2)	38.4	*45.2*	213.6	*218.6*	822	*847*	0.90	*1.07*
May	762	*768*	6 (0.8)	26.4	*28.9*	137.4	*139.9*	577	*588*	0.66	*0.72*
June	342	*348*	6 (1.7)	9.3	*15.1*	38.6	*45.5*	227	*257*	0.32	*0.52*
July	90	*96*	6 (6.3)	18.0	*38.4*	76.7	*99.4*	422	*517*	0.56	*1.20*
August	56	*62*	6 (9.7)	33.6	*63.7*	177.0	*202.5*	725	*842*	0.82	*1.60*
September	74	*80*	6 (7.5)	38.7	*61.6*	213.5	*230.5*	839	*922*	0.87	*1.40*
October	174	*180*	6 (3.3)	31.6	*42.0*	185.2	[illegible]	746	*796*	0.74	*1.00*
November	182	*188*	6 (3.2)	35.3	*45.2*	212.0	[illegible]	843	*878*	0.82	*1.07*
December	145	*151*	6 (4.0)	45.6	*57.5*	266.5	[illegible]	1043	*1079*	0.90	*1.16*
Annual	216	*222*	6 (2.7)	33.8	*42.2*	187.5	[illegible]	765	*797*	0.78	*0.98*

Maximum predicted discharge with three mines operating: 6 cfs.
Most highly mineralized mine discharge recorded 7/1/75:

Spec. cond. = 1940 μmhos·cm^{-1}
Na = 345 mg·L^{-1}
SO$_4$ = 440 mg·L^{-1}
SAR = 9.09

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[1] Percent contribution of mine discharge to total river flow in parentheses

[2] Specific conductance reported as μmhos·cm^{-1} @ 25° C.

increases would be most evident in the summer and fall. However, the most important measure, the SAR value, would still remain well within the defined standard, if the Tongue River water were used for irrigation.

Table 27 lists mean values of most of the parameters measured in the mine discharge water and in the Tongue River above and below the site of discharge. A total of 32 sampling sessions were conducted from June 1975 through November 1976. Chemical differences between the two river stations were not within the limits of analytical detection. Comparing several of the mean mine discharge parameters with those at the bottom of Table 26, it becomes obvious that the worst conditions possible deviate greatly from the mean and that, under present mining conditions, this situation should rarely occur.

The prospect of the entire mine effluent reaching the Tongue River is unlikely. The East Decker Mine dumps effluent directly into the Tongue River. However, the West Decker Mine effluent passes beneath Montana Highway 314, meanders in an easterly direction, and collects in a swamp. It is probable that the effluent reached the Tongue River only under conditions of high water.

Van Voast and Hedges (1975) found that the volume of effluent discharged from the West Decker Mine has decreased since mining began. It was also found that the concentrations of chemical constituents and

Table 27. Average values of selected parameters measured in the mine discharge water and in the Tongue River above and below the site of discharge from June 1975 through November 1976.

Parameter[1]	Inflow	Mine Discharge	TRBM
pH	8.6	8.5	8.4
Dissolved oxygen	10.1	9.8	9.8
Specific conductance[2]	693	1498	696
Turbidity (JTU)	15	18	14
Temperature (°C)	14.7	14.9	14.7
Organic carbon	4.9	5.2	5.0
CO_3	4	12	4
HCO_3	238	597	240
Total alkalinity[3]	202	509	203
SiO_2	7.4	13.0	7.4
Fe	0.028	0.021	0.024
Cl	2.8	6.8	2.7
F	0.33	1.22	0.33
SO_4	166.4	295.1	166.4
Ca	61.2	33.1	61.0
Mg	37.8	42.4	37.8
Na	30.5	253.3	30.4
K	3.9	7.7	3.9
NO_2-N[4]	3	37	3
NO_3-N	31	287	30
NH_3-N	16	282	16
TKN[5]	326	674	333
PO_4-P	20	6	18
Total-P	62	38	60
SAR	0.78	6.88	0.78

[1] All parameters expressed as $mg \cdot L^{-1}$ unless otherwise noted.

[2] $\mu mhos \cdot cm^{-1}$ at 25°C.

[3] $mg \cdot L^{-1}$ as $CaCO_3$.

[4] All nutrients expressed as $\mu g \cdot L^{-1}$.

[5] Total Kjehldahl nitrogen.

dissolved solids in that effluent have decreased. All of the above mentioned decreases were attributed to the reduced rate of flow entering the mine from coal bed and overburden aquifers relative to the rate of inflow from the reservoir and alluvium.

In light of the above discussion, it is safe to predice that *under existing conditions*, the West Decker Mine discharge will have no measurable effect on the *surface water* of the Tongue River *with respect to the parameters measured. Barring any extreme deviations from present prognostications,* it seems reasonable to predict that effluent discharged from the North Decker and East Decker mines will not significantly alter the *surface water* quality of the Tongue River or Tongue River Reservoir *with respect to the parameters measured.* Unforseen changes in effluent water quality, effluent discharge rate, or Tongue River flow rate may alter the above predictions.

SUMMARY AND CONCLUSIONS

(1) The Tongue River provided the only major inflowing and outflowing surface water to and from the Tongue River Reservoir. Lowest volumes of flow were recorded during the fall and winter months of 1975-76 at both the inflow and outflow stations. Peak runoff in the Tongue River drainage above the reservoir occurred in mid-June, and the highest annual volume of inflowing water was noted at that time. Correspondingly high discharge was also observed at the dam outlet during spring runoff, and water crested the reservoir spillway for a short time at the height of runoff. Peak reservoir storage (740 X 10^5 m^3) was noted late in June and the reservoir was progressively drawn down through the late summer in response to downstream irrigation needs. Storage was held constant at about 432 X 10^5 m^3 throughout the winter months.

The highest inflow and outflow turbidities were observed during spring runoff when high silt loads were carried by the Tongue River. The mean extinction coefficient and turbidity of the reservoir decreased progressively from Station 3 to Station 1. This corresponded with increasing depth and decreasing proximity to the inflow. Turbidity was due predominately to allochthonous (during the spring) and autochthonous (during the summer) wind-agitated silt at Station 3. The reservoir acted as a sediment trap with most of the suspended particulates settling out before reaching Station 2. Therefore, the extinction coefficients and turbidities of Stations 1 and 2 were heavily influenced by biogenic factors.

Annually, the lowest concentrations of major cations and anions in the river water were recorded during spring runoff, reflecting the low ionic content of snowmelt. Concentrations of major ionic species increased constantly through the summer, fall, and winter months due to evaporative concentration and a lack of rainfall. Reservoir salinity followed the same seasonal trend. The Tongue River Reservoir and its inflow and outflow waters were a calcium-magnesium bicarbonate-sulfate type.

Analyses of the temperature, conductivity, and turbidity profiles indicated the presence of a thermal gradient and a density current in the reservoir during the period from late May through most of June. The density current was ephemeral as deepwater withdrawal destroyed the vertical thermal gradient which provided the framework for the density current. However, during most of the year the reservoir was polymictic as evidenced by an orthograde temperature profile. This polymictic situation was caused by the slight current in the long, narrow reservoir, the reservoir water withdrawal characteristics, and the wind-generated agitation of the reservoir's water.

Despite the lack of thermal stratification in the reservoir, a mild oxygen deficit was evident at the sediment-water interface during part of the summer. In the summer of 1975 the oxygen deficit was apparently sufficient to effect the liberation of ferrous iron, but not sulfide, from the reservoir sediment. This situation was analagous to Phase III of Hutchinson's (1957) scheme of summer stagnation (although the Tongue River Reservoir did not stagnate in the strict sense). Available evidence indicated that in the following summer, 1976, reducing conditions at the sediment-water interface were not conducive to the reduction of ferric iron. Hence, the reservoir did not proceed past Phase II of Hutchinson's scheme. The only observation supporting the attainment of Phase III in 1976 was the presence of increased concentrations of PO_4-P and total-P in the late summer; these increases were attributed to wind-generated mixing of the sediment-water interface. Further observations (Whalen and Leathe, unpublished data) led to the conclusion that the data for 1975 were more indicative of the norm, and that the Tongue River Reservoir usually reached Phase III of Hutchinson's summer stagnation scheme.

During the time period mid-November 1975 to mid-November 1976, 182,107 kg total-N was delivered to the Tongue River Reservoir via the Tongue River, and 179,576 kg total-N was exported. The inflow and outflow total-P loads were 33,774 and 17,186 kg.

The highest nutrient concentrations in the inflow and outflow were recorded in the winter and during spring runoff. It was hypothesized that much of the winter nutrient input to the reservoir flowed through the reservoir because planktonic utilization was minimized due to the low water temperature and reduced light penetration through the ice. Of the annual (November 1975-November 1976) inorganic nitrogen input from the Tongue River to the reservoir, 55% was realized during the four winter months from December through March, yet 62% of this import was exported during this same time span. Forty-two percent of the river's annual inorganic phosphorus input to the reservoir was recorded from December through March. The reservoir acted in a more conservative manner toward phosphorus than toward nitrogen during this period, as only 36% of the winter input was expelled. Most of the winter phosphorus load to the reservoir was probably immobilized in the sediment.

It was hypothesized that most of the nutrient load delivered by the river to the reservoir during spring runoff also flowed through the system. Due to the short hydraulic retention time during spring runoff (≃ 30 days), algal cycling of the reservoir's nutrient import at that time was kept to a minimum. Sixty percent of the annual (November-November) organic-N load carried by the river to the reservoir was delivered during the 15 April-30 June period (spring runoff). In the same time span, 72% of the annual organic + hydrolyzable-P load carried by the river to the reservoir was realized. However, 84% of the organic-N and 43% of the organic + hydrolyzable-P delivered to the reservoir during spring runoff was exported in the 15 May-30 July period. The reservoir again acted in a conservative manner toward phosphorus. However, it was concluded that immobilization in the sediment neutralized much of the river's phosphorus input to the reservoir during spring runoff. Although inorganic nutrient loads in the spring were not as great as the organic loads, their behavior was analogous.

Storage of nutrients within the reservoir strongly reflected the pattern of surface water nutrient input to the reservoir, which emphasizes the importance of allochthonous nutrients to the reservoir's ecology. Typically low concentrations of inorganic nutrients were found in the euphotic water during the entire ice-free period, although moderate concentrations were recorded when the entire water column was averaged. Diffusion along a concentration gradient, wind-generated mixing, and the absence of a hypolimnetic barrier were instrumental in providing a continuous supply of nutrients for algal consumption.

(2) Six classes of algae were identified in samples collected in the Tongue River Reservoir during this study: Bacillariophyceae, Chlorophyceae, Cryptophyceae, Dinophyceae, Euglenophyceae, and Myxo-

phyceae. Within these groups, a total of 125 species were identified and an additional 20 phytoplankters were taken to the generic level. During the growing season (ice-free period) of 1976, the mean algal standing crop at Station 2 (mid-reservoir station) expressed on a volumetric basis was 3.53 $mm^3 \cdot L^{-1}$. The weighted mean annual (November 1975–November 1976) phytoplankton standing crop at Station 2 was 2.62 $mm^3 \cdot L^{-1}$, expressed on the same basis. The mean areal growing season phytoplankton standing crop at Station 2 was 12.55 $cm^3 \cdot m^{-2}$, compared with a weighted mean annual areal standing crop of 8.91 $cm^3 \cdot m^{-2}$ at the same sampling site. The differences among the station means for the growing season algal standing crops were not statistically significant when data were analyzed on either a volumetric or an areal basis.

On a reservoir-wide basis, the mean areal class composition of the growing season standing crop in 1976 was 59% Bacillariophyceae, 20.1% Dinophyceae, 12.6% Myxophyceae, 6.4% Cryptophyceae, and 1.9% Chlorophyceae. Beneath ice cover in the winter of 1975-76 the following relative areal class composition was found: Bacillariophyceae - 67%, Cryptophyceae - 28%, Chlorophyceae - 5%. The Myxophyceae and Dinophyceae contributed seasonally to the algal biomass, while the Bacillariophyceae and Cryptophyceae were virtually holoplanktonic.

Although the areal growing season phytoplankton standing crop in

1976 was generally moderate, three distinct pulses were noted at each sampling site. The pulses were discordant both temporally and in terms of phytoplanktonic class composition, despite the fact that the reservoir was long and narrow with a paucity of arms and bays.

In 1976 the mean whole lake growing season chlorophyll *a*/cell volume ratio was 2.66, while the weighted mean annual (November 1975-November 1976) whole lake ratio was 2.97. The higher winter value reflected algal adaptation to low light intensities.

The size and the relative species composition of the total phytoplanktonic biota were governed by several interrelated ecological factors rather than a few dominant factors.

Phytoplankton primary production was measured by the ^{14}C technique at Station 2 of the reservoir. Sufficient data were collected at all three reservoir stations to estimate primary production using the formula of Ryther and Yentsch (1957). A statistical analysis of mean estimates of day rate photosynthesis obtained via these two approaches rejected the null hypothesis. Because the ^{14}C technique is generally considered to be the more accurate of the two approaches, a modification was made to the Ryther and Yentsch empirical formula so that it would adequately estimate phytoplankton promary production when *in situ* ^{14}C experiments were used as a standard. *In situ* determined assimilation ratios (dependent variable) were regressed against euphotic zone temperatures (independent variable) and the results were used to obtain a temperature dependent assimilation ratio. This assimilation ratio was then incorporated into the empirical formula of Ryther and Yentsch. Despite the many factors affecting the assimilation ratio (productivity) of a given algal standing crop, temperature accounted for 58% of the variability during the November 1975-November 1976 study year. When subjected to statistical scrutiny, the modified formula yielded an estimate of primary production that compared favorably with the ^{14}C estimate. This formula was then extended to data collected at the other two reservoir sampling stations to estimate more accurately the algal productivity at those sites.

The mean daily gross photosynthetic rate during the 1976 growing season at Station 2 was 1.11 g $C \cdot m^{-2}$ while these same data, on a weighted annual (November 1975-November 1976) basis, indicated a photosynthetic rate of 0.77 g $C \cdot m^{-2} \cdot day^{-1}$. Total annual photosynthesis was estimated to be 281.2 g $C \cdot m^{-2}$ at Station 2, with 92.4% of the yearly gross photosynthesis taking place during the open water period. The differences among estimated daily photosynthetic rates during the growing season at the three sampling sites were not statistically significant.

(3) Several popular methods of evaluating the trophic status of the body of water were applied unsuccessfully to the Tongue River Reservoir data. The annual nutrient surface loading rates (4.1 g total-P$\cdot$m^{-2} and 22.0 g total-N$\cdot$m^{-2}) were indicative of a hypereutrophic system. Values of several other key nutrient parameters generally led to the conclusion that the Tongue River Reservoir was eutrophic. The temporal pattern of nutrient loading, deepwater withdrawal at the reservoir's outflow gate, and the rapid flushing rate (7.78$\cdot$yr^{-1}) of the reservoir reduced the predicted availability of nutrients. As a result, the phytoplankton productivity and standing crop were held at moderate levels. Historically, nuisance blooms of algae have developed late in the summer, but only at Station 3. However, based upon phosphorus levels and the N/P ratio, the potential exists for the initiation of reservoir-wide nuisance blooms of heterocystous blue-green algae. Such a development could mar the recreational value of the reservoir.

(4) Current surface mining operations at the West Decker Mine are exploiting the uppermost (D-1) coal seam. This coal seam provides one of the major continuous ground water aquifers of the region adjacent to the reservoir. The average ground water sample in the West Decker vicinity was primarily a sodium bicarbonate type. Ground water draining into the mine site and leachate from disturbed over-

burden are collected and discharged into the Tongue River flood plain directly above the Tongue River Reservoir. Water from the collection pond is also used for dust abatement within the mine site. The dominant cation in the mine discharge water was sodium, while the dominant anions were bicarbonate and sulfate. It was impossible to predict the relative importance on the chemistry of the mine discharge water of percolation through disturbed overburden versus disruption of the D-1 coal seam (aquifer) for two reasons: (1) Naturally occurring ground water aquifers are common in the Decker area (2) The degree of proximity of the D-1 coal seam (aquifer) to sandstone and alluvium varies.

Although the mine discharge water differed chemically from the Tongue River water, it did not measurably affect the Tongue River water with regard to the parameters considered. The mean percent contribution of the mine discharge water to the river water flow was 0.09% percent during the entire study.

(5) When the data were manipulated to produce extremely severe conditions (i.e. maximum discharge of highly mineralized mine effluent directly to the river), the concentrations of several chemical constituents of the river water were calculated to increase below the site of discharge. If these increases were to take place, they would not impair the river water quality from a recreational or irrigational

standpoint.

The effluent discharged from three mines in simultaneous operation should not alter significantly the surface water quality of the Tongue River and Tongue River Reservoir with respect to the parameters measured. Unforseen changes in effluent water quality, effluent discharge rate, or Tongue River flow rate may alter this prediction.

REFERENCES CITED

American Public Health Association. 1971. Standard methods for the examination of water and wastewater. 13th ed. APHA, New York. 874 pp.

Barnes, H.H. 1959. Apparatus and methods of oceanography. I. Chemical. G. Allen and Unwin, Ltd., London. 341 pp.

Bernstein, L. 1967. Quantitative assessment of irrigation water quality. Pages 51-65 *in* Water Quality Criteria. Am. Soc. Testing and Materials Spec. Tech. Pub. STP-416.

Brezonik, P.L. 1972. Nitrogen: sources and transformations. Pages 1-50 *in* H.E. Allen and J.R. Kramer, eds. Nutrients in natural waters. John Wiley and Sons, New York.

Cassie, R.M. 1962. Microdistribution and other error components of C^{14} primary production estimates. Limnol. Oceanogr. 7:121-130.

Chen, R.L., D.R. Keeney, D.A. Graetz and A.J. Holding. 1972. Denitrification and nitrate reduction in lake sediments. J. Environ. Qual. 1:158-162.

Clifford, H.T. and W. Stephenson. 1975. An introduction to numerical classification. Academic Press, New York. 229 pp.

Curl, H., Jr. and L.E. Small. 1965. Variations in photosynthetic assimilation ratios in natural marine phytoplankton communities. Limnol. Oceanogr. 10:R67-R71 (Supplement).

Dendy, F.E. and W.A. Champion. 1973. Summary of reservoir sediment deposition surveys made in the United States through 1970. U.S. Agr. Res. Service Misc. Pub. No. 1266. 82 pp.

Dillon, P.J. 1975. The phosphorus budget of Cameron Lake, Ontario: The importance of flushing rate to the degree of eutrophy in lakes. Limnol. Oceanogr. 20:28-39.

Dillon, P.J. and F.H. Rigler. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19:767-773.

Drouet, F. 1959. Myxophyceae. Pages 95-114 *in* W.T. Edmondson, ed. Freshwater biology. John Wiley and Sons, Inc., New York.

Environmental Protection Agency. 1971. Methods for chemical analysis of water and wastes. EPA, Cincinnati. 312 pp.

Environmental Protection Agency. 1972. Water quality criteria. EPA, Washington, D.C. 594 pp.

Environmental Protection Agency. 1976. Quality criteria for water. EPA, Washington, D.C. 256 pp.

Eppley, R.W., J.L. Coatsworth and L. Solorzano. 1969. Studies of nitrate reductase in marine phytoplankton. Limnol. Oceanogr. 14:194-205.

Federal Water Pollution Control Administration. 1966. A guide to the common diatoms at water pollution surveillance system stations. FWPCA, Cincinnati. 101 pp.

Feth, J.H., S.M. Rogers and C.E. Roberson. 1964. Chemical composition of snow in the Northern Sierra Nevada and other areas. USGS Water Supply Paper 1535-J. 39 pp.

Foree, E.J. and C.R. Scroggin. 1973. Carbon and nitrogen as regulators of algal growth. J. Environ. Eng. 99:639-653.

Garrison, P.J., S.C. Whalen and R.W. Gregory. 1975. Limnology of the Tongue River Reservoir: Existing and potential impact of coal strip mining. First progress report. Montana CFRU, Montana State University, Bozeman. 32 pp.

Golterman, H.L. 1973. Natural phosphate sources in relation to phosphate budgets: A contribution to the understanding of eutrophication. Water Res. 7:3-17.

Golterman, H.L. 1975. Physiological limnology. Elsevier Scientific Publishing Co., New York. 489 pp.

Gorham, E. 1961. Factors influencing supply of major ions to inland waters, with special reference to the atmosphere. Bull. Geol. Soc. Amer. 72:795-840.

Groterud, O. 1972a. Nutrients in ice. Some data from a high mountain lake. Verh. Internat. Verein. Limnol. 18:327-333.

Groterud, O. 1972b. Ice analysis. Data from three Norwegian lakes. Hydrobiol. 41:371-391.

Groterud, O. 1972c. Hydrographical data from two soft water lakes with special reference to precipitation (melt water). Arch. Hydrobiol. 70:277-324.

Hallegraeff, G.M. 1977. A comparison of different methods used for the quantitative evaluation of biomass of freshwater plankton. Hydrobiol. 55:145-165.

Hansmann, E.W. 1973. Diatoms of the streams of Eastern Connecticut. Conn. Geol. and Nat. Hist. Surv. Bull. 106. 119 pp.

Harter, R.D. 1968. Adsorption of phosphorus by lake sediments. Proc. Soil Sci. Soc. Amer. 32:514-518.

Hem, J.D. 1970. Study and interpretation of the chemical characteristics of natural water. USGS Water Supply Paper 1473. 364 pp.

Hustedt, F. 1930. Bacillariophyceae (Diatomae). Heft. 10 *in* A. Pascher. Die Süsswasser-flora Mitteleuropas. Gustav-Fisher, Jena, Germany. 466 pp.

Hutchinson, G.E. 1957. A treatise on limnology I. Geography, physics, and chemistry. John Wiley and Sons, Inc., New York. 1015 pp.

Hutchinson, G.E. 1967. A treatise on limnology II. Introduction to lake biology and the limnoplankton. John Wiley and Sons, Inc., New York. 1115 pp.

Hutchinson, G.E. 1973. Eutrophication. Am. Sci. 61:269-279.

Hynes, H.B.N. 1970. The ecology of running water. University of Toronto Press, Toronto. 555 pp.

Jorgensen, E.G. 1969. The adaptation of plankton algae IV. Light adaptation in different algal species. Physiol. Plant. 22:1307-1315.

Kimball, H.H. 1928. Amount of solar radiation that reaches the surface of the earth on the land and on the sea, and methods by which it is measured. Mon. Weath. Rev. 56:393-398.

Krauss, R.W. 1958. Physiology of freshwater algae. Ann. Rev. Plant Physiol. 9:207-244.

Kuhl, A. 1974. Phosphorus. Pages 636-655 *in* W.D.P. Stewart, ed. Algal physiology and biochemistry. University of California Press, Berkeley.

Kuznetsov, S.I. 1968. Recent studies on the role of microorganisms in the cycling of substances in lakes. Limnol. Oceanogr. 13:211-224.

Leathe, S.A. and S.C. Whalen. 1978. Limnology of the Tongue River Reservoir: Existing and potential impact of coal strip mining. Sixth progress report. Montana CFRU, Montana State University, Bozeman. 12 pp (mimeo).

Lund, J.W.G. 1949. Studies on *Asterionella* I. The origin and nature of the cells producing seasonal maxima. J. Ecol. 37: 389-419.

Lund, J.W.G. 1950. Studies on *Asterionella formosa* Hass. II. Nutrient depletion and the spring maximum (Parts I and II). J. Ecol. 38:1-35.

Lund, J.W.G., C. Kipling and E.O. LeCren. 1958. The inverted microscope method of estimating algal numbers and the statistical basis of estimation by counting. Hydrobiol. 11:143-170.

Martin, D.B. 1967. Limnological studies on Hebgen Lake, Montana. Ph.D. Thesis. Montana State University, Bozeman. 126 pp.

Montana Bureau of Mines and Geology. Unpublished data.

Montana Department of Natural Resources and Conservation. Unpublished data.

Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water in lakes. (Parts I and II). J. Ecol. 29:280-329.

Mortimer, C.H. 1942. The exchange of dissolved substances between mud and water in lakes (Parts III and IV). J. Ecol. 30:147-201.

Mortimer, C.H. and F.J.H. Mackereth. 1956. Convection and its consequences in ice-covered lakes. Verh. Internat. Verein. Limnol. 13:923-932.

Moss, B. 1973. Diversity in freshwater plankton. Amer. Midl. Nat. 90:341-355.

Neel, J.K. 1963. Impact of reservoirs. Pages 579-593 *in* D.G. Frey, ed. Limnology in North America. University of Wisconsin Press, Madison.

Odum, E.P. 1971. Fundamentals of ecology. 3rd ed. W.B. Saunders Company, Philadelphia. 574 pp.

Oglesby, R.T. 1977. Phytoplankton summer standing crop and annual productivity as functions of phosphorus loading and various physical factors. J. Fish. Res. Bd. Can. 34:2250-2270.

Paasche, E. 1960. On the relationship between primary production and standing crops of phytoplankton. J. Cons. Int. Explor. Mer. 26:33-48.

Palmer, C.M. 1969. A composite rating of algae tolerating organic pollution. J. Phycol. 5:78-82.

Patrick, R. and C.W. Reimer. 1966. The diatoms of the United States I. Philadelphia Academy of Natural Science. 688 pp.

Pechlaner, R. 1970. The phytoplankton spring outburst and its conditions in Lake Erken (Sweden). Limnol. Oceanogr. 15:113-130.

Penkal, R.F. 1977. Black bass populations of the Tongue River Reservoir, Montana. M.S. Thesis. Montana State Univ., Bozeman. 111 pp.

Pomeroy, L.E. 1970. The strategy of mineral cycling. Ann. Rev. Ecol. and Systematics 1:171-190.

Prakash, A. and M.A. Rashid. 1968. Influence of humic substances on the growth of marine plankton: Dinoflagellates. Limnol. Oceanogr. 13:598-606.

Prescott, G.W. 1962. Algae of the Western Great Lakes area. Wm. C. Brown Co., Dubuque, Iowa. 997 pp.

Prescott, G.W. 1970. How to know the freshwater algae. Wm. C. Brown Co., Dubuque, Iowa. 348 pp.

Rada, R.G. 1974. An investigation into the trophic status of Canyon Ferry Reservoir, Montana. Ph.D. Thesis. Montana State Univ., Bozeman. 126 pp.

Rawson, D.W. 1955. Morphometry as a dominant factor in the productivity of large lakes. Verh. Int. Verein. Limnol. 12:164-175.

Rawson, D.S. 1956. Algal indicators of trophic lake types. Limnol. Oceanogr. 1:18-25.

Rawson, D.S. 1960. A limnological comparison of twelve large lakes in Northern Saskatchewan. Limnol. Oceanogr. 5:195-211.

Reid, G.K. 1961. The ecology of inland waters and estuaries. Van Nostrand Reinhold Co., New York. 375 pp.

Reid, G.K. and R.D. Wood. 1976. Ecology of inland waters and estuaries. 2nd ed. Van Nostrand Reinhold Co., New York. 485 pp.

Rhee, G-Y. 1974. Phosphate uptake under nitrate limitation by *Scenedesmus* sp. and its ecological implications. J. Phycol. 10:470-475.

Rhee, G-Y. 1978. Effect of N:P atomic ratios and nitrate limitation on algal growth, cell composition and nitrate uptake. Limnol. Oceanogr. 23:10-25.

Rodhe, W. 1949. The ionic composition of lake waters. Verh. Int. Verein. Limnol. 10:377-386.

Rodhe, W. 1955. Can plankton production proceed during winter darkness in subarctic lakes? Verh. Int. Verein. Limnol. 12:117-122.

Rodhe, W. 1969. Crystallization of eutrophication concepts in Northern Europe. Pages 50-64 *in* Eutrophication: Causes, consequences and correctives. Natural Academy of Sciences, Washington, D.C.

Ruttner, F. 1963. Fundamentals of limnology (Translat. D.G. Frey and F.E.J. Fry). University of Toronto Press, Toronto. 295 pp.

Ryther, J.H. 1956. Photosynthesis in the ocean as a function of light intensity. Limnol. Oceanogr. 1:61-70.

Ryther, J.H. and C.S. Yentsch. 1957. The estimation of phytoplankton production in the ocean from chlorophyll and light data. Limnol. Oceanogr. 2:281-286.

Saraceni, C. and M. Gerletti. 1968. How deep waters in a large lake (Lake Maggiore, Northern Italy) may affect primary production. Mem. Ist. Ital. Idrobiol. 23:141-159.

Saunders, G.W., F.B. Trama and R.W. Bachmann. 1962. Evaluation of a modified C-14 technique for shipboard estimation of photosynthesis in large lakes. Great Lakes Research Division Pub. No. 8. Ann Arbor, Michigan. 61 pp.

Sawyer, C.N. 1947. Fertilization of lakes by agricultural and urban drainage. J. New Engl. Water Works Assoc. 61:109-127.

Schaffner, W.R. and R.T. Oglesby. 1978. Phosphorus loading to lakes and some of their responses. Part 1: A new calculation of phosphorus loading and its application to 13 New York lakes. Limnol. Oceanogr. 23:120-134.

Schindler, D.W. 1977. Evolution of phosphorus limitation in lakes. Science 195:260-262.

Schwoerbel, J. 1970. Methods of hydrobiology (freshwater biology). Pergammon Press, Inc., New York. 200 pp.

Shannon, E.E. and P.L. Brezonik. 1972. Relationship between lake trophic state and nitrogen and phosphorus loading rates. Environ. Sci. Technol. 6:719-725.

Small, L.F., H. Curl and W.A. Glooschenko. 1972. Estimates of primary production off Oregon using an improved chlorophyll-light technique. J. Fish. Res. Bd. Can. 29:1261-1267.

Smith, G.M. 1950. The freshwater algae of the United States. 2nd ed. McGraw-Hill, New York. 719 pp.

Snedecor, G.W. and W.G. Cochran. 1967. Statistical methods. 6th ed. Iowa State Univ. Press, Ames. 593 pp.

Soltero, R.A., J.C. Wright and A.A. Horpestad. 1974. The physical limnology of Bighorn Lake - Yellowtail Dam, Montana: Internal density currents. Northwest Sci. 49:107-124.

Sommers, L.E., R.F. Harris, J.D.H. Williams, D.E. Armstrong and J.K. Syers. 1970. Determination of total organic phosphorus in lake sediments. Limnol. Oceanogr. 15:301-304.

Steel, R.G.D. and J.H. Torrie. 1960. Principles and procedures of statistics. McGraw-Hill, New York. 481 pp.

Steemann-Nielsen, E. 1952. The use of radioactive carbon (C-14) for measuring organic production in the sea. J. Cons. Int. Explor. Mer. 18:117-140.

Steemann-Nielsen, E. 1955. Production of organic matter in the oceans. J. Mar. Res. 14:374-386.

Steemann-Nielsen, E. 1962. On the maximum quantity of plankton chlorophyll per surface unit of a lake or the sea. Int. Rev. ges. Hydrobiol. 47:333-338.

Steemann-Nielsen, E. 1963. Productivity, definition and measurement. Pages 129-164 *in* M.N. Hill, ed. The sea II. Wiley Interscience, New York.

Stewart, W.D.P. 1977. A botanical ramble among the blue-green algae. Br. Phycol. J. 12:89-115.

Strickland, J.D.H. 1960. Measuring the production of marine phytoplankton. Bull. 122. Fish. Res. Bd. Can. 172 pp.

Strickland, J.D.H. and T.R. Parsons. 1972. A practical handbook of seawater analysis. 2nd ed. Bull. 167 Fish. Res. Bd. Can. 310 pp.

Stumm, W. and J.J. Morgan. 1970. Aquatic chemistry. An introduction emphasizing chemical equlibria in natural water. Wiley Interscience, New York. 583 pp.

Syers, J.K., R.F. Harris and D.E. Armstrong. 1973. Phosphate chemistry in lake sediments. J. Environ. Qual. 2:1-14.

Symons, J.M., S.R. Weibel and G.G. Robeck. 1964. Influence of impoundments on water quality. USPHS Pub. No. 999-WP-18. Cincinnati. 78 pp.

Syrrett, P.J. 1962. Nitrogen assimilation. Pages 171-188 *in* R.A. Lewin, ed. Physiology and biochemistry of algae. Academic Press, New York.

Taft, C.E. and C.W. Taft. 1971. The algae of Western Lake Erie. Bull. Ohio Biol. Survey (New Series). 4(1). Columbus, Ohio. 185 pp.

Talling, J.F. 1971. The underwater light climate as a controlling factor in the production ecology of freshwater phytoplankton. Mitt. Int. Ver. Limnol. 19:214-243.

Tiffany, L.H. and M.E. Britton. 1952. The algae of Illinois. Univ. of Chicago Press, Chicago. 407 pp.

Tucker, A. 1957. The relation of phytoplankton periodicity to the nature of the physiochemical environment with reference to phosphorus. Amer. Midl. Nat. 57:300-370.

UNESCO. 1974. A review of methods used for quantitative phytoplankton studies. UNESCO Tech. Pap. Mar. Sci. 18.

United States Geological Survey. 1976. Water resources data for Montana. Water year 1975. 604 pp.

United States Geological Survey. 1977. Water resources data for Montana. Water year 1976. 766 pp.

United States Geological Survey and Montana Department of State Lands. 1977. Proposed plan of mining and reclamation: East Decker and North Extention mines, Decker Coal Co., Big Horn County, Montana. Final Environmental Impact Statement. I. 871 pp.

VanVoast, W.A. 1974. Hydrologic effects of strip coal mining in southeastern Montana - Emphasis: one year of mining near Decker. Mt. Bur. Mines and Geol. Bull. 93. 24 pp.

VanVoast, W.A. and R.B. Hedges. 1975. Hydrologic aspects of existing and proposed strip coal mines near Decker, southeastern Montana. Mt. Bur. Mines and Geol. Bull. 97. 31 pp.

Verduin, J. 1964. Principles of primary productivity: Photosynthesis under completely natural conditions. Pages 221-238 *in* D.F. Jackson, ed. Algae and man. NATO Advanced Study Institute. Plenum Press, New York.

Vollenweider, R.A. 1965. Calculation models of photosynthesis-depth curves and some implications regarding day rate estimates in primary production measurements. Pages 425-457 *in* C.R. Goldman, ed. Primary productivity in aquatic environments. Mem. Ist. Ital. Idrobiol., 18. Suppl., Univ. of California Press, Berkeley.

Vollenweider, R.A., ed. 1969. A manuel on methods for measuring primary production in aquatic environments. IBP Handbook No. 12. Blackwell Scientific Publications, Oxford. 213 pp.

Vollenweider, R.A. 1970. Scientific fundamentals of eutrophication of lakes and flowing waters, with special reference to nitrogen and phosphorus as factors in eutrophication. Organization for Economic Cooperation and Development, Paris. 254 pp.

Vollenweider, R.A. and A. Nauwerck. 1961. Some observations on the ^{14}C method for measuring primary production. Verh. Int. Ver. Limnol. 14:134-139.

Wallen, D.G. and G.H. Green. 1968. Loss of radioactivity during storage of ^{14}C-labelled phytoplankton on membrane filters. J. Fish. Res. Bd. Can. 25:2219-2224.

Welch, P.S. 1948. Limnological methods. McGraw-Hill, New York. 381 pp.

Wetzel, R.G. 1965. Necessity for decontamination of filters in ^{14}C measured rates of photosynthesis in freshwaters. Ecology 46:540-542.

Wetzel, R.G. 1975. Limnology. W.B. Saunders, Co., Philadelphia. 743 pp.

Whalen, S.C. and S.A. Leathe. 1976. Limnology of the Tongue River Reservoir: Existing and potential impact of coal strip mining. Third progress report. Montana CFRU, Montana State Univ., Bozeman. 64 pp.

Whalen, S.C. and S.A. Leathe. Unpublished data.

Willén, E. 1976. A simplified method of phytoplankton counting. Br. Phycol. J. 11:265-278.

Wright, J.C. 1959. The limnology of Canyon Ferry Reservoir II. Phytoplankton standing crop and primary production. Limnol. Oceanogr. 4:235-245.

Wright, J.C. 1967. Effects of impoundment on productivity, water chemistry and heat budgets of reservoirs. Pages 188-199 *in* Reservoir Fishery Resources Symposium, Reservoir Committee Southern Division, Amer. Fish. Soc., New York.

Wright, J.C. and R.A. Soltero. 1973. Limnology of Yellowtail Reservoir and the Bighorn River. EPA Report R3-73-002 (Ecological Research Series). Washington, D.C. 105 pp.

Wright, R.T. Dynamics of a phytoplankton community in an ice-covered lake. Limnol. Oceanogr. 9:163-178.

Yentsch, C.S. and J.H. Ryther. 1957. Short-term variations in phytoplankton chlorophyll and their significance. Limnol. Oceanogr. 2:140-142.

Zicker, E.L., K.C. Berger and A.D. Hasler. 1956. Phosphorus release from bog lake muds. Limnol. Oceanogr. 1:296-303.