Relationships between benthic communities, land use, chemical dynamics, and trophic state in Georgetown Lake

by Paul Allen Garrett

A thesis submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy in Botany

Montana State University

Abstract:
The relationship between land use, chemical dynamics, arid benthic plant corrmunities in Georgetown Lake was investigated from 1973-1975 and 1981-1982. Experimental enclosures were used to isolate littoral macrophyte communities and evaluate effects on aquatic chemistry. Over 90% of the Georgetown Lake drainage was covered by natural vegetation, primarily coniferous forest. Cultural development is concentrated around the reservoir, but does not appear to be affecting basin water quality or recreational values due to the small relative area (<3% of the total) developed. Results of a septic system survey indicated that domestic wastes are not having a significant impact csi water quality. Macrcphyte communities were found to dominate primary productivity and strongly influence chemical dynamics in the reservoir. Canopy coverage of Potamoqeton praelonqus doubled between 1975 and 1981. An exponential growth rate of 0.1155/yr was calculated for this population. Standing crops of other macrophyte communities appeared to vary annually in response to meteorological and hydrological conditions. During July-September, 1974, benthic communities accounted for an average of 90% of gross reservoir photosynthesis and 73% of reservoir respiration. Carbon limitation was suggested as a cause of declining photosynthetic rates in late summer. Rooted aquatic vascular plants were not limited by nitrogen or phosphorus availability, as indicated by tissue analysis. Ccmmunity oxygen consumption under ice cover ranged from 0.08-0.32 g O2/m^2/day. Positive rates of oxygen production (0.089-0.174 g O2/m^2/day) were measured at shallow sites and attributed to algal photosynthesis. The majority of oxygen uptake was attributed to benthic communities. Concentrations of inorganic dissolved phosphorus remained low thoughout the ice-free periods and suggested regulation by calcium hydroxyapatite solubility equilibria. Regulation of pH by macrophyte photosynthesis is implicated in control of phosphorus solubility. Rates of internal phosphorus loading up to 3.83 mg P/m^2/day were calculated. Major internal sources of nitrogen identified were release from sediments during winter anoxia and release from littoral sediments during late summer. Sediment accumulation rate was estimated at 0.25 cm/yr, equalling 1050 g dry matter/m^2/year. Vertical distribution of sediment phosphorus indicated an upward mobilization, which was attributed to uptake and redeposition by rooted vascular plants. Mass budgets for nitrogen and phosphorus indicate that major nutrient sources are natural and within acceptable levels. Phosphorus loading rates are negative, but dissolved phosphorus supplies are being buffered by sediment phosphorus supplies.

RELATIONSHIPS BETWEEN BENTHIC COMMUNITIES, LAND USE, CHEMICAL DYNAMICS, AND TROPHIC STATE IN GEORGETOWN LAKE

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APPROVAL

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Paul Allen Garrett

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

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ABSTRACT

The relationship between land use, chemical dynamics, and benthic plant communities in Georgetown Lake was investigated from 1973-1975 and 1981-1982. Experimental enclosures were used to isolate littoral macrophyte communities and evaluate effects on aquatic chemistry. Over 90% of the Georgetown Lake drainage was covered by natural vegetation, primarily coniferous forest. Cultural development is concentrated around the reservoir, but does not appear to be affecting basin water quality or recreational values due to the small relative area (<3% of the total) developed. Results of a septic system survey indicated that domestic wastes are not having a significant impact on water quality. Macrophyte communities were found to dominate primary productivity and strongly influence chemical dynamics in the reservoir. Canopy coverage of Potamogeton praelongus doubled between 1975 and 1981. An exponential growth rate of 0.1155/yr was calculated for this population. Standing crops of other macrophyte communities appeared to vary annually in response to meteorological and hydrological conditions. During July-September, 1974, benthic communities accounted for an average of 90% of gross reservoir photosynthesis and 73% of reservoir respiration. Carbon limitation was suggested as a cause of declining photosynthetic rates in late summer. Rooted aquatic vascular plants were not limited by nitrogen or phosphorus availability, as indicated by tissue analysis. Community oxygen consumption under ice cover ranged from 0.08-0.32 g O_2/m^2/day. Positive rates of oxygen production (0.089-0.174 g O_2/m^2/day) were measured at shallow sites and attributed to algal photosynthesis. The majority of oxygen uptake was attributed to benthic communities. Concentrations of inorganic dissolved phosphorus remained low thoughout the ice-free periods and suggested regulation by calcium hydroxyapatite solubility equilibria. Regulation of pH by macrophyte photosynthesis is implicated in control of phosphorus solubility. Rates of internal phosphorus loading up to 3.83 mg P/m^2/day were calculated. Major internal sources of nitrogen identified were release from sediments during winter anoxia and release from littoral sediments during late summer. Sediment accumulation rate was estimated at 0.25 cm/yr, equalling 1050 g dry matter/m^2/year. Vertical distribution of sediment phosphorus indicated an upward mobilization, which was attributed to uptake and redeposition by rooted vascular plants. Mass budgets for nitrogen and phosphorus indicate that major nutrient sources are natural and within acceptable levels. Phosphorus loading rates are negative, but dissolved phosphorus supplies are being buffered by sediment phosphorus supplies.

INTRODUCTION

Georgetown Lake is a multiple use reservoir located between Anaconda and Philipsburg, Montana, on the Philipsburg Divide. Highway 10A parallels the shoreline for about 5 miles (8 km). Major uses of the reservoir have included water storage for industrial processing, downstream irrigation, power generation, and, as a by-product, recreation. The beautiful natural setting and absence of other similar resources in the area have lent particular importance to the recreational use aspect. Over time, secondary uses of a resource may assume primary importance under changing economic and social conditions. So it may be with Georgetown Lake. With the closure of the Anaconda Company smelting facilities in Anaconda and the development of alternative power generation facilities, recreational uses of the reservoir have assumed more importance to the local community, from a standpoint of both personal enjoyment and economic assets. It thus becomes more important to maintain and protect the conditions and qualities which endow the reservoir with its unique esthetic and recreational values.

Georgetown Lake has a history of excellent fishing, easy accessibility, and intense recreational use. Recreational activities include fishing, power-boating, sailing, water-skiing, swimming, waterfowl hunting, and in winter, snow-mobiling, cross-country skiing, and ice fishing. The reservoir also has a history of

periodic algal blooms, occasional winter fish kills of varying severity, and, to some but not all recreationists, excessive aquatic vascular plant growth.

As a result of these problems, coupled with accelerated cultural development in the area and intense public concern, Georgetown Lake has been the subject of considerable limnological investigation over the past nine years. The work now being reported began in 1973. Local concern over the problems stated relative to recreational use of the reservoir and development in the drainage resulted in a research grant from the Department of Interior's Office of Water Research to Montana State University for the investigation of limnological relationships in Georgetown Lake. Paul Garrison, William Geer, William Foris, and Jonathan Knight, and the author participated in these studies, which continued into 1976, and have been reported, in part, by Geer (1977), Garrison (1976), Foris(1976), Knight et al (1975), and Knight (1980). Several significant questions remained unanswered by these reports. The following are of particular concern, relative to lake management:

1) What are the causes of the dense macrophyte growths observed in Georgetown Lake?

2) What are the relationships between macrophytes and other aspects of recreational water quality, such as clarity, algal growth, and dissolved oxygen concentrations during ice cover?

3) What is the probable pattern of macrophyte community development and growth in the reservoir in the near future?

4) Are there any feasible management options which may bring biological conditions more in line with those desired in a multiple-use recreational reservoir?

5) What impacts are recreational use of the reservoir and cultural development of the surrounding drainage having on the condition of the reservoir, and what restrictions may be necessary to maintain or improve the esthetic qualities of the reservoir as a recreational resource?

Subsequent to 1976, a perceived rapid increase in the density and distribution of macrophytes brought renewed local concern about water quality/land use/macrophyte relationships. Under the provisions of Section 14, Public Law 92-500 (Federal Water Pollution Control Act Amendments of 1972), EPA administers the Clean Lakes Program to assist the states in maintaining the recreational water quality of lakes and reservoirs through pollution control and lake restoration projects. Through the efforts of local citizens, Montana Department of Natural Resources, and the Montana Water Quality Bureau, EPA funded a diagnostic/feasibility study for Georgetown Lake to attempt to answer the above questions. This program was initiated in August, 1980 and continued through August, 1982. The author was involved in the program as Project Scientist, and much of the data used in this report were collected during the project.

The following specific objectives were identified and research efforts throughout the project focused on answering the critical questions involved:

1. Identify cultural impacts on trophic state and macrophyte growth through analysis of land use and nutrient budgets.

2. Determine contribution of macrophytes to primary production through analysis of seasonal dynamics of biomass and community metabolism.

3. Investigate factors influencing winter oxygen budgets in Georgetown Lake relative to macrophytes, phytoplankton, and morphometry.

4. Identify changes in distribution, composition, and productivity of macrophyte communities.

5. Establish nutrient status of macrophytes relative to the availability of nitrogen and phosphorus.

6. Investigate the relationships of macrophytes to internal nutrient cycles.

7. Determine rates of sediment accumulation and evaluate sediment relative to nutrient budgets, nutrient supplies and lake management.

8. Investigate other factors affecting the expression of productivity or implementation of management programs in Georgetown Lake.

Classification systems of lakes and reservoirs based on "trophic state" have been extensively developed and almost universally accepted (Reckhow, 1979). Categories used are usually eutrophic, mesotrophic, and oligotrophic, in the sense of nutrient supply

(Hutchinson, 1957a). Parameters of classification systems include morphometry (Rawson, 1955); hypolimnetic oxygen demand (Hutchinson, 1957a); Secchi disc depth (Megard, et al, 1980); alkalinity (Wright, 1980b); phytoplankton pigment concentrations (Reckhow, 1979; Dobson et al, 1974; EPA, 1974a); phosphorus and nitrogen concentrations in lake waters (Hutchinson, 1957a; Edmondson, 1969; EPA, 1974a; Sawyer, 1947; and many others); phosphorus loading rates (Vollenweider, 1968; 1976); community metabolism (Odum, 1961; Hooper, 1969); nitrogen loading rate (Shannon and Brezonik, 1972). Suggested systems of classification containing discussion of parameters, their values, and applications are found in Reckhow (1979), Carlson (1977), EPA (1974), and Uttormark and Wall (1975). There are many other excellent references dealing with the subject of eutrophication and related limnological processes.

The relationship of surface land use to the process of cultural eutrophication is well established (Edmondson, 1969; Vollenweider, 1968). Coefficients of nutrient export have been estimated for a variety of surface land uses (Uttormark, et al, 1974; Reckhow, et al, 1980). Lake managers have had considerable success in controlling or reversing cultural eutrophication by control of point and non-point sources; examples include Shagawa Lake (Larsen and Malueg, 1981), Lake Washington (Edmondson, 1969), and Lake Onandaga (Bartsch, 1981). Although implicated in many cases of cultural eutrophication, on site treatment systems (septic tank/drainfield systems) have been shown to be much less troublesome as non-point sources than suspected

(EPA and Wapora, 1981). In many cases, nutrient loads from septic tanks have been estimated at less than 5% of the total phosphorus loading in spite of dense residential development, and in general have been much less important than point discharges from treatment facilities or nonpoint agricultural sources.

According to Pond (1905), in his pioneering paper on the relationship of rooted aquatic plants to their substrates, the importance of aquatic macrophytes in stabilizing bottom sediments, providing habitats for aquatic fauna, and maintaining dissolved oxygen concentrations had long been recognized at the turn of the century. The role of benthic macrophytic vegetation in limnological processes has been given increasing attention in recent research, in part due to the public's perception that macrophytic vegetation is in general more troublesome to many water-based recreational pursuits than algae. However, advances in methods have afforded opportunities heretofore unavailable for new directions in research on this component of the aquatic environment.

Recent research has addressed diverse aspects of the ecology of aquatic macrophytes. Pond's (1905) finding that rooted aquatic vascular plants grew better when attached to rich soil than to washed sand gave rise to a long-standing argument concerning nutrient sources for rooted aquatics. The development of isotopic tracer techniques has provided substantial evidence that such vegetation is fully capable of mobilizing nutrient ions, including P, Fe, Ca, and N from the substrate (DeMarte and Hartman, 1974; Toetz, 1974; Gentner, 1977; Nichols and Keeney, 1976; Barko, et al, 1982), and is also

capable of absorbing nutrients (particularly phosphorus) from the free water at low concentrations (Carignan and Kalff, 1980). Barko, et al (1982) found that K was not mobilized as effectively as other nutrient ions under consideration and speculated that competition for adsorption sites on the root epidermis by more mobile N compounds may be responsible. It is now widely accepted that rooted vascular aquatic plants are capable of obtaining most of their phosphorus from sediment sources and often function as nutrient pumps to the free water during senescence (Welch, et al, 1979; Carpenter, 1980).

Productivity of aquatic macrophytes is much more difficult to study than phytoplankton productivity because of sampling problems relative to accessibility of the submerged communities, spatial variation, size of the organisms, edaphic factors, and interference by periphyton. As a result, three distinct approaches are frequently employed. Much of the work on photosynthesis and primary productivity of aquatic macrophytes has been on enclosed shoot or leaf samples under laboratory conditions and is difficult to apply to community or stand productivity (Klarich, 1977).

Serial sampling techniques have been applied to production estimates for aquatic macrophytes (Wetzel, 1964; Westlake, 1969; Forsberg, 1957; Adams and McCracken, 1974; Boylen and Sheldon, 1976). Problems are encountered due to difficulty in obtaining sufficient samples, sorting samples by species, multiple cohorts in some species, partial senescence of stems, and presence of significant underground biomass. Diurnal measurements of oxygen and carbon

dioxide changes in unenclosed environments have been used to estimate aquatic productivity in planktonic and benthic communities. Jackson and McFadden (1954), Odum (1957), Talling (1957), Sugiura (1953), McConnell (1958), and Hoskin (1959) applied the use of the diurnal curve method to estimates of community productivity. This method was subsequently applied to estimates of community metabolism in estuarine environments by Odum and Hoskin (1958), Park, et al (1958), Odum and Wilson, (1961), and Beyers, et al (1963). Diurnal curve estimates of primary productivity, made on unenclosed plankton communities, have usually been several times greater than light/dark bottle estimates made on the same communities (Odum and Wilson, 1961).

Limitations of the diurnal curve method include difficulties in estimating invasion/evasion coefficients, particularly for oxygen, small scale variations in dissolved oxygen and carbon dioxide, incomplete vertical mixing, excessive buffering capacity of some waters, and equipment limitations. In addition, the method is extremely time-consuming and physically exhausting. However, it may be the method of choice in situations where it can be successfully applied.

Gerloff and Krombholz (1966) evaluated the nutrient status of aquatic vascular plants by tissue analysis. Tissue content of nitrogen and phosphorus on a dry weight basis was proposed as an indication of nutrient supplies in natural environments. They estimated the critical nitrogen content (that below which growth was reduced) to be 1.3%; the comparable value for phosphorus was 0.13%.

In the lakes they studied, they concluded that phosphorus was more likely to be limiting growth of aquatic macrophytes. Subsequently, Gerloff (1973, 1975) proposed the use of index segments near the growing tip of the stem as a more reliable indicator of nutrient availability than the entire stem. Critical levels of P, N, Fe, Cu, Z, and other nutrients for several aquatic species were established.

DESCRIPTION OF THE STUDY AREA

Georgetown Lake, latitude 46° 10' 16" N, longitude 113° 10'42" W, is formed by the impoundment of Flint Creek, in the Clark Fork River drainage (Fig. 1). Highway 10A parallels the east shore for about 8 km. Early human activity in the area centered around mining and ranching. The site now occupied by the reservoir was used as pasture for cattle as early as 1872 (Lutey, et al, 1974). The first dam at the present site was built in 1885, and subsequent reconstruction, the last in 1937, raised the full pool water level to its present elevation of 6378 feet, msl (USGS, 1971).

Geology of the area is described in Alt and Hyndman (1972) and Ingman and Bahls (1979). The Flint Creek Range is composed mostly of complexly folded Precambrian metamorphic rocks and pre-Cenozoic sedimentary rocks with large granite intrusions. The general area is characterized as folded metamorphic formations, sedimentary deposits from an inland sea, subsequent uplift and erosion, leaving a fretted upland. The higher peaks in the area have elevations up to 10,000 feet (3200 m). The North Fork of Flint Creek has known gold and silver deposits, with mining activity occurring in the area as late as 1951. There are also deposits of gravel, clay, and limestone in the surrounding drainages.

Reservoir morphometric parameters are given in Table 1. These

were derived from a registered survey conducted in 1937 for the Anaconda Company by A. J. Davidson. Maximum surface area is given as 2,990 acres (1,210 hectares). Maximum mean depth is 16.0 feet (4.88 m). Shoreline length is approximately 17.4 miles (27.8 km).

Mean annual precipitation for the area is 18.75 inches (47.6 cm), measured at the Silver Lake precipitation gauge. Precipitation is substantially greater in the higher elevations [35-45 inches (89-114 cm)/year] surrounding the reservoir. May and June are the wettest months, averaging 32 percent of the annual total. Six of the last seven years (1975-1981) have had above-average precipitation, ranging from -27.5 to +79.4 percent of normal.

Average annual total evaporation was estimated at 35.4 inches (89.9 cm). Estimated monthly net evaporation values (net evaporation = total evaporation - precipitation) are shown in Table 2, based on pan evaporation measurements at several western Montana locations (NOAA, 1979) and evaporation estimates for small lakes and reservoirs derived by Meyer, (1942).

Hydrologic relationships in the Georgetown Lake area are complex, involving inputs from Silver Lake via the Hardtla ditch, North Fork of Flint Creek, Stewart Mill Creek , and a considerable groundwater flow. Outputs include the dam discharge via Flint Creek, pumping via the Anaconda Company aqueduct, which feeds back into Silver Lake, and possibly seepage. The mean annual discharge from Georgetown Lake is 29.6 cubic feet per second (cfs) (0.84 m^3/s). Silver Lake, in addition to its natural drainage, receives water

Table 1. Morphometric parameters of Georgetown Lake. Total volume 5.9158×10^7 m^3. Storage parameters in $m^3 \times 10^6$. Area in hectares (ha). Surface elevation as feet on the Anaconda company gage. Mean depth in meters.

Surface Elev.	Surface Area	Change in Storage	Cumulative Δ Storage	Cumulative Δ Area	Mean Depth
6429.5	1210.5				4.88
		1.835	1.835	13.48	
6429.0	1197.0				4.79
		1.815	3.650	26.84	
6428.5	1183.7				4.69
		1.798	6.065	40.16	
6428.0	1170.48				4.57
		1.774	7.222	53.36	
6427.5	1157.2				4.48
		1.754	8.975	66.52	
6427.0	1144.0				4.39
		1.734	10.710	79.64	
6426.5	1130.9				4.27
		1.714	12.423	92.63	
6426.0	1117.9				4.18
		1.694	14.118	105.55	
6425.5	1105.0				4.08
		1.675	15.793	131.17	
6425.0	1092.1				3.96
		1.655	17.448	143.48	
6424.5	1179.4				3.87
		1.633	19.081	159.68	
6424.0	1063.0				3.75
		1.608	20.714	159.68	
6423.5	1046.6				3.68
		1.584	22.273	175.75	
6423.0	1030.7				3.57

diverted from Storm Lake Creek during spring runoff. Under normal conditions, water from Silver Lake is diverted by the Anaconda Company through the Hardtla ditch, which is controlled by a headgate. Three thousand (3,000) acre-feet (3.7×10^6 m^3) or more of water may be diverted into Georgetown Lake, much of which may spill down Flint Creek due to insufficient storage capacity.

The reservoir drains an area of approximately 33,900 acres

Table 2. Net evaporation estimates for Georgetown Lake.

Month	Meyer (1942)		NOAA[1]		s. d.[2]	
	in.	cm	in.	cm	in.	cm
Jan	0.75	1.91	---	---	--	--
Feb	0.75	1.91	---	---	--	--
Mar	1.30	3.30	---	---	--	--
Apr	2.30	5.84	---	---	--	--
May	3.40	8.64	4.48	11.40	1.2	3.0
Jun	4.70	11.94	6.38	16.20	0.9	2.3
Jul	6.50	16.51	6.74	17.12	1.1	2.8
Aug	7.20	18.29	5.63	14.30	0.9	2.3
Sep	4.50	11.43	3.42	8.69	0.7	1.8
Oct	2.50	6.35	---	---	--	--
Nov	1.00	2.54	---	---	--	--
Dec	0.50	1.27	---	---	--	--

[1] Data from three western Montana meteorological survey sites (NOAA, 1977, 1978, 1979).
[2] Standard deviation of NOAA data

(13,725 hectares). The North Fork of Flint Creek is the major surface stream entering the reservoir, draining approximately 10,000 acres (4048 ha). Peak flows usually occur in May or June; these two months account for 55-80 percent of the total annual discharge. Extreme fluctuations in discharge are common. Total yield for May-June has varied from less than 2,500 acre-feet (3.09×10^6 m^3) to more than 15,000 acre-feet (18.52×10^6 m^3), a seven-fold variation. Minimal flows are usually obtained in October, averaging 5-8 cfs (0.14-0.22 m^3/s).

Stewart Mill Creek originates as a spring about 300 yards (meters) upstream from its entry into the reservoir, draining an area of approximately 10,400 acres (4,210 hectares). Discharge from Stewart Mill is much more stable annually and seasonally than the North Fork, reflecting its basic nature as groundwater. Data from

Knight, et al (1975) show that the annual discharge from Stewart Mill Creek varied by less than 10 percent from 1973-74 to 1975-75, while precipitation increased by 80 percent and annual discharge from the North Fork increased by 282 percent.

The remaining 4600 ha drains the area to the northwest and southwest of the reservoir. There is no surface discharge, but presumably groundwater flows to the reservoir.

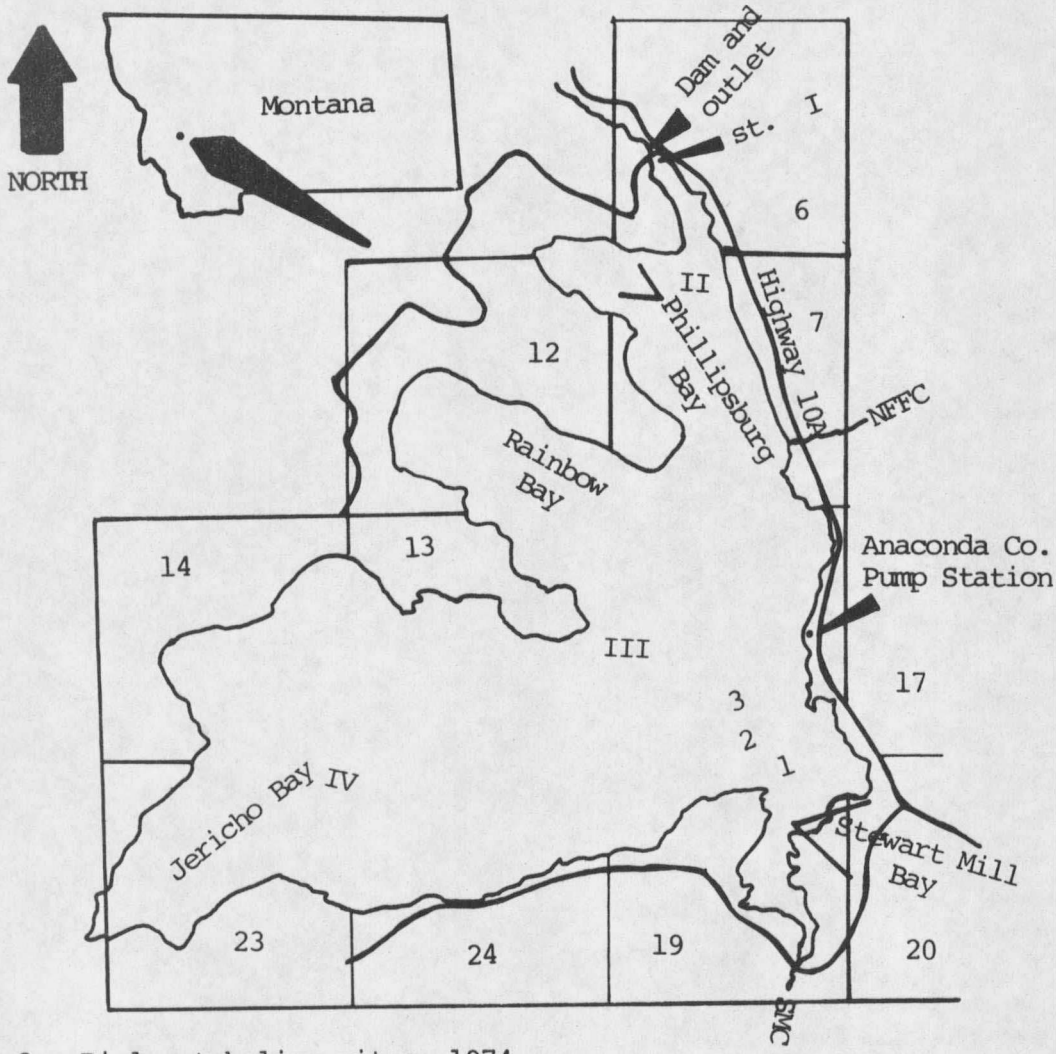
METHODS

SAMPLING SITES, SAMPLING SCHEDULE, AND METHODS OF COLLECTION

Surface water samples were collected from June 1973 through June 1975, and from June 1981 through June 1982. Four reservoir sampling stations and three stream sampling stations were used in 1973-1975. Locations are shown in Figure 1. In 1981-1982 four reservoir sampling stations were established which roughly corresponded to stations I-IV previously used (Figure 2). In addition, a littoral sampling area was established in Philipsburg Bay (Fig 2.) as station V within which five enclosures, or limnocorrals, were installed to isolate littoral macrophyte communities for experimental manipulation and monitoring of littoral chemical and biological processes.

These enclosures were fabricated to project specifications by Kepner Plastics, Inc. Torrance, California, from opaque, cloth-reinforced, polyvinyl fabric. They were approximately two meters in depth. Triangular frames 3.9 meters on a side were attached to the outside of the enclosures with wood screws. They were anchored in place by steel rods driven through the skirt into the bottom, by weights attached to the external frame, and by a chain which was seamed into the bottom of the skirt, causing it to sink into the bottom mud and form a seal at the mud-water interface.

Exchange of water at the surface was prevented by flotation collars, which provided about ten centimeters of freeboard. The



1-3 Diel metabolism sites, 1974

I-IV Water chemistry sampling stations, 1973-75

Figure 1. Georgetown Lake Study Area, Deer Lodge and Granite Counties, Montana.

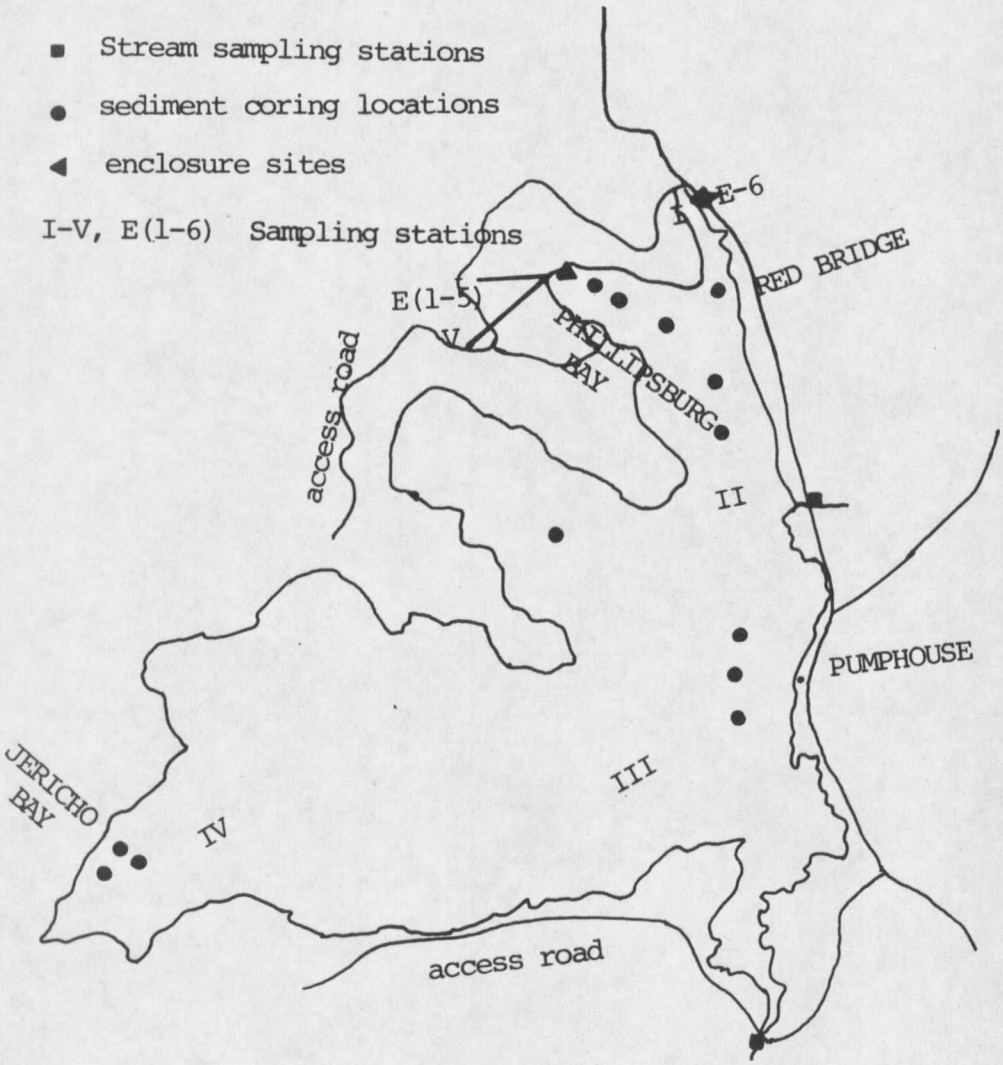


Figure 2. Sampling stations, 1981-1982.

depth of the water in this area was about 2.0 meters at reservoir full pool. These enclosures were designated E-1 through E-5. They were installed between 24-27 June, 1981, after soaking in the lake for three weeks to remove leachable chemicals. Two control sampling stations were established in unenclosed communities adjacent to the enclosures. Data collected at these sites were pooled as station V.

Experimental treatments planned for the littoral enclosures included fertilization with nitrogen and phosphorus, harvesting, and nutrient inactivation with alum. However, during the first two weeks of monitoring, which was considered an acclimation period, the condition of the enclosed aquatic communities deteriorated, as indicated by changes in turbidity, pH and dissolved oxygen. Consequently the enclosures became a means to examine the relationships of aquatic plant decay, sediment respiration, and internal nutrient loading, with essentially four replicates. In late July, one littoral enclosure, which had been designated E-5, was removed, clear vinyl windows were installed in the opaque walls to increase light transmittance to the interior, and the enclosure was reinstalled in an undisturbed area.

To evaluate differences in effects of sediments in areas not supporting macrophyte growths as compared to those on which dense macrophyte communities occurred, a deeper enclosure 6.5 m deep by 2 m in diameter similar in construction to the littoral enclosures was placed near station I. Macrophytic growth was absent at this location, and the bottom was composed of fine-grained silt and detritus.

Stream sampling stations were designated NFF (North Fork, Flint Creek), SMC (Stewart Mill Creek) and FCO (Flint Creek, reservoir Outfall). NFF was located about 100 meters above the bridge on Highway 10A; SMC was located just below the bridge on the road to Denton's Point, and FCO was located immediately below the Montana Power Company generating station (Figure 1).

In 1973-75, lake water samples were collected in a 3 liter Van Dorn-type polyvinylchloride sampling bottle. Samples were collected at two-meter vertical intervals from surface to near the bottom at station I, three-meter intervals at stations II-V. In 1981-82 lake water samples were collected using a portable, peristaltic pump, powered by a 12-volt battery. Discrete water strata were sampled at two meter vertical intervals at station I and Enclosure 6 by lowering Tygon tubing to the desired depth and filling sample bottles directly from the hose outlet. At stations II-V and E-1 through E-6, depth-integrated samples were collected by lowering the hose intake to near the bottom and raising it through the water column at an even rate while pumping, then mixing the resulting sample thoroughly for subsampling. All stream samples were collected as surface grabs directly into sample bottles, using rubber gloves to prevent sample contamination.

Sites were sampled semi-weekly throughout the summer and early fall. During ice cover, sampling of selected sites was on a monthly basis. During 1973-75, only station I on the reservoir and the stream samples were regularly sampled through the winter. In 1981-

1982, stations I-V, E-2, E-5, E-6 and the stream stations were all sampled throughout the winter months.

Samples were filtered in the field for the determination of chlorophyll a. The filtered water was retained for the determination of dissolved organic and inorganic components. Gelman type A glass fiber filters or equivalent were used and extracted in 90% acetone in accordance with procedures described by Strickland and Parsons (1972). Unfiltered water samples were also retained for the determination of particulate organic materials, pH, alkalinity, and conductivity. Samples were stored in the dark on ice until analysis, which was initiated within 24 hours.

WATER CHEMISTRY

Dissolved ortho-phosphorus (as defined by EPA. 1974b) was determined by the ascorbic acid-molybdate single reagent method (Strickland and Parsons, 1972) on filtered, unpreserved samples. Total dissolved and total phosphorus were determined on filtered and unfiltered samples, respectively, using acid-persulfate oxidation followed by the single reagent method (Strickland and Parsons, 1972; EPA, 1974b). Ammonia-N was determined by hypochlorite oxidation (Strickland and Parsons, 1972); nitrate-N was determined by hydrazine reduction with a diazotization finish as described in Barnes (1959). Total organic nitrogen was determined by a micro-Kjeldahl technique as described in Strickland and Parsons (1972), which utilizes a phenol-hypochlorite finish on digested, normalized samples.

Hydrogen ion concentration (pH) was determined in the laboratory using a pH meter with an accuracy of 0.01 unit or better. Total alkalinity was determined by titration with 0.01 N acid to an end-point of pH 4.50, which was consistently found to be the inflection point of the titration curve of Georgetown Lake water. Conductivity was determined with a Yellow Springs Instrument Co. Model 31 conductivity bridge standardized with 0.01 M KCl (EPA, 1974b). Dissolved oxygen was determined in the field using a Yellow Springs Model 54 meter calibrated according to the manufacturer's instructions, and in the laboratory by Winkler titration (Strickland and Parsons, 1972). Water temperature was measured using a standard laboratory thermometer or a thermistor in conjunction with the dissolved oxygen meter.

QUALITY ASSURANCE

General laboratory procedures followed Handbook for Analytical Quality Control in Water and Wastewater Laboratories (EPA, 1972). Reagent grade water was prepared by passing glass-distilled water through three Barnes Ultrapure ion-exchange column cartridges. Distilled water for standards and blanks was taken directly from the columns. All glassware used in nutrient analyses was acid washed, capped and stored in covered shelves to prevent contamination when not in use. Analytical standards were prepared weekly from analytical reagent grade chemicals dried at 105 C and stored in a desiccator.

Samples were replicated and spiked at a frequency recommended

by EPA (1972). Recovery rates were determined and analytical values were adjusted for percent recovery. Quality control results are reported in Table 3, Results of Spikes and Replicate Samples, and Table 4, Results of EPA Quality Control Sample Analysis.

Low recovery of nitrate occurred on one occasion. The value of 43% nitrate recovery was obtained for both spikes. The standard curve obtained was linear within the analytical range.

TABLE 3. Analytical results of spikes and replicate samples. N = number of replicates; MEAN%R = average percent recovery of spike concentrations in water samples; RANGE %R = range of percent recovery; MEAN,S-D = average difference between replicate spikes; S. D. = standard deviation; RANGE S-D = range of differences in analytical values between replicate samples.

	SPIKES				REPLICATES			
	N	MEAN%R	S.D.	RANGE%R	N	MEAN,S-D	S.D.	RANGE,S-D
NH_3	18	99	13	70-124	20	1.7	2	0-8
NO_3	19	81	19	43-109	20	1.0	3	0-7
PO_4	19	100	1.4	96-102	20	0.0	0.0	0-1

TABLE 4. Analysis of EPA quality control samples. N = 2 for each parameter analyzed.

	NOMINAL VALUE	ANALYTICAL VALUE	%ERROR	%VARIATION
NH_3	0.19	0.18	-5	0.5
NO_3	0.31	0.34	+8	2.6
PO_4	0.031	0.029	-6	0.0

The low recovery was probably due to incomplete reduction of of nitrate by hydrazine. All values are reported as ug per liter or milligrams (mg) per cubic meter. Values less than 10 ug/l for ammonia nitrogen and nitrate nitrogen are below the accepted EPA

limits for analytical precision, but were evaluated at stated concentrations for study purposes.

SEDIMENT CHEMISTRY

Sediment cores were collected in January and July, 1981, using a Hydro Products sediment corer. Locations of sediment sampling are shown in Figure 2. Cores were 3.5 cm in diameter and about 50 centimeters long. Intact cores with overlying water were stored in the plastic coring tube at -15° C until analysis. Cores were thawed, measured, and sectioned in 2 cm sections with a stainless steel knife. The volume of intact, unaltered sections was determined by displacement in a graduated cylinder, then samples were dried at 105° C and weighed to the nearest 0.01 g for the determination of sediment density. Subsamples of dried sections were analyzed for total phosphorus using persulfate digestion and acid-molybdate finish (USGS, 1979) by Montana School of Mines Analytical Laboratory.

HYDROLOGY

Discharge data for the North Fork of Flint Creek from 1973-1981 were obtained from the Deer Lodge National Forest Hydrologist Office at Butte, which maintains a recording gage on that stream about 1 kilometer upstream from station NFF. Discharge data for Flint Creek below the dam (station FCO) were obtained from USGS and Montana Power Company, who maintain daily gage records. Daily reservoir surface elevations, morphometric data, and other hydrologic data were obtained from the Anaconda Company offices in Anaconda, Montana.

Influent streams were gaged in 1981 following the USGS method (USGS, 1969), using Price AA and Pygmy pattern current meters. Staff gages were installed in June 1981 on the North Fork of Flint Creek and Stewart Mill Creek, near the sampling stations NFF and SMC, respectively, and read several times weekly through September, weekly or semi-monthly after that through May, 1982. Discharge rating curves were constructed following USGS methodology and gage data were used to calculate instantaneous and daily discharge. The volume of groundwater flow was calculated on a monthly basis as a residual from changes in gaged input, gaged output, storage, and net evaporation estimates.

Surveys for groundwater inputs were run using an Endeco Septic Leachate Detector concurrently with a septic leachate survey which was performed by Swanson Environmental, Inc. Equipment characteristics and methods are described in Garrett (1983), Swanson Environmental (1981), and Environmental Devices Co., (undated). The conductivity channel on the septic leachate detector proved sufficiently sensitive to detect major groundwater inputs at depths up to 18 feet (5.8 m), due to the lowered conductivities of lake water (150-160 micromhos), compared to 250-300 micromhos for groundwater, as indicated by values from Stewart Mill Creek. Groundwater inputs showed as spikes on the conductivity channel with no increase (or a decrease) in fluorescence. Other groundwater inputs were located during a walking shoreline reconnaissance. Locations of these

sources were noted on field maps and transferred to permanent maps in the office.

In areas not covered by the above methods, a Yellow Springs Instrument Co. portable conductivity meter was used. The conductivity cell was mounted on a submerged bracket attached to a small boat and slow shoreline cruises were made while monitoring the meter for changes in conductivity. Due to the biologically-caused low conductivity of lake water during the summer of 1981, this method was found to be effective in locating significant groundwater inputs.

Samples from submerged, aquatic springs were collected using the peristaltic pump and hose. The hose intake was positioned immediately above spring inputs using SCUBA, and samples were pumped slowly to prevent disturbance of the surrounding sediment.

SEPTIC LEACHATE SURVEY

A study of septic tank effluents using fluorescent dye was carried out July 11-19, 1981. Approximately 15 ml of Rhodamine WT dye was introduced by hand into 200 septic systems near the shoreline of Georgetown Lake on July 11. Dye tracing began on July 14, using a Turner Model 111 fluorimeter equipped with a flow-through cell and filters as recommended by the Turner Manual of Operation. Stable electrical power was provided by a 110 volt gasoline generator equipped with an inline voltage stabilizer. A small, 12-volt, submersible, bilge pump was mounted on a section of rigid plastic pipe and connected to the fluorimeter cell with Tygon tubing.

Prior to each survey, the fluorimeter was calibrated to full scale deflection with a solution of lake water containing about 0.1 ppm dye. The apparatus was placed in a small boat manned by two people, one to operate the pump, the other to monitor the fluorimeter and run the boat. Surveys were made by running the boat along the shoreline at slow speed while pumping a steady sample stream from just below the surface through the fluorimeter cell. Deflections were recorded on a portable recorder attached to the meter.

SOLAR RADIATION

Total daily solar radiation was measured in 1974 and 1976 using a Kipp and Zonen Model CM-3 pyranometer and an Esterline-Angus Model 80-M recorder, which were installed at the Montana Department of Fish, Wildlife, and Parks cabins on Highway 10A. Radiation measurements were converted to langleys per day as prescribed by the manufacturer's instructions.

Light intensity profiles within the reservoir were determined using a submarine photometer fitted with a Weston B56 photocell. Vertical extinction coefficients were calculated according to the definition and equation given by Hutchinson (1957a). A mean vertical extinction coefficient was calculated for the water column by determining the percent transmission per meter from the surface to within one meter of the bottom (or the tops of macrophytes), then taking the natural logarithm of the arithmetic average of these values. This method is analogous to the graphical method of Hutchinson (1957a) and

does not require interpretation of a non-linear graph.

AQUATIC PLANT COMMUNITIES

Distribution and Composition of Macrophyte Communities

The distribution and composition of aquatic macrophyte communities was investigated by aerial photography and transect observations using SCUBA. Aerial photography of aquatic macrophyte communities was obtained in early August, 1975, by Dr. R. L. Eng, Biology Department, Montana State University, and in early August, 1981-1982, by Jeff Ryan, ECON, Inc., Helena, Montana. A 35mm camera and ASA 200 Ektachrome color slide film were used for all aerial photography. The camera was mounted on a Meyer side-mount attached to a small, fixed-wing aircraft. Line transects covering all areas of the lake were selected and photographed at a film scale of 1:20, 000. Details of this method are found in Meyer (1973).

Interpretation of the photography was done by the author. Slides were projected onto transparent plastic film and community outlines delineated with an ink pen. Recognition of distinct community types depended on texture, color, and pattern of the photographic imagery, within the context of ground investigations to determine reference areas. Locations were found with overlapping coverage from all three years of imagery and composite maps showing changes in coverage and community distribution were generated. Areas of selected community types were determined using a dot grid. Changes in extent of communities between years were determined by subtraction.

Standing crops of selected macrophyte communities were sampled with the aid of SCUBA. Quadrat frames (0.5 m^2) were fitted with a floating skirt approximately 1 m in depth to facilitate location in deep plant beds. Frames were placed randomly in some community types; in others a grid was established with floats to obtain an evenly distributed representation of the plant community. The latter method was considered by the author to be analogous to a stratified random sampling scheme as discussed by Grieg-Smith (1964).

Above-ground portions of the plants enclosed within the frame were collected by hand, rinsed of excess sediment in lake water, stored in plastic bags on ice in the field, and deep-frozen in the laboratory until analysis. Later, samples were washed, in some cases sorted according to species, and dried at 90°C for a minimum of 48 hours. Taxonomic identifications were made using Hitchcock and Cronquist (1973), Muenscher (1944), Mason (1957), and Booth and Wright (1959). Samples were weighed to the nearest 0.1 g on an electronic balance.

Tissue Analysis of Aquatic Macrophytes

Tissues of aquatic macrophytes were analysed for nitrogen and phosphorus concentrations according to methods described in Gerloff and Krombholz (1966), Gerloff (1973), and Johnson and Ulrich (1959). Plants were washed in distilled water, index segments removed as appropriate (terminal and subterminal 2.5 cm segments) and dried at 90°C. Dry weights were determined to 0.01 mg on an analytical balance. Loss on ignition was determined by ashing at 550°C, then

rehydrating and drying at 90°C. Organic carbon content was estimated using values for percentage of ash-free dry weight provided by Westlake (1965) and Boyd (1967). Nitrogen in dried samples was determined by a semi-micro Kjeldahl procedure. Phosphorus was determined by ashing in H_2SO_4, extraction into dilute HCl, and subsequent analysis using the acid-molybdate method.

COMMUNITY METABOLISM

Community metabolism was measured in enclosed and unenclosed communities by the diurnal pH-CO_2/O_2 method as developed by Odum and Hoskin (1958), Park, et al (1958), and Beyers, et al, (1963). In 1974, sites were selected at one, two, three, and four meter depth contours in an area off the mouth of Stewart Mill Bay (Figure 2). During a diurnal measurement, water samples were collected at 1 meter depth intervals from the surface to the bottom using a BOD sewage sampler or a handpump fitted with a series of siphoning bottles and a hose into 300 ml glass stoppered bottles. Samples were stored for not more than one hour in a cooler filled with lake water to maintain ambient temperature.

Samples were taken as quickly as possible to a field laboratory located at the MDFWP facilities on Highway 10A and pH determined using a Beckman expanded-scale pH meter and standard electrodes. The pH meter was calibrated for each series of measurements with a buffer of suitable pH range which was maintained at the ambient temperature of the samples, as suggested by Beyers, et al (1963).

Sample temperatures were measured with a standard laboratory

thermometer calibrated to 0.1°C. Oxygen measurements were made in the field with a YSI Model 54 dissolved oxygen meter and appropriate probe. Oxygen measurements were made in the laboratory by Winkler titrations. Storage times for samples were about one hour. A storage correction was made for planktonic respiration based on dark bottle respiration rates determined concurrently with the experiments.

All diurnal measurements were carried out over at least 24 hours. During the daylight measurements were made at approximately four hour intervals. Measurements were not made during darkness due to navigation difficulties, however, samples were collected at dusk, usually around 10:00 p.m., and just before sunrise, about 5:00-6:00 a.m., MDT. For the calculation of carbon uptake versus pH change, empirical pH/CO_2 curves were constructed for lake water according to the CO_2 titration technique described by Beyers, et al (1963). A computer program was developed for these calculations by Dr. Dalton Burkhalter and the author. Community respiration and photosynthesis were calculated as m-moles (mg) C $(O_2)/m^2$/unit time. Carbon or oxygen changes at each depth were integrated graphically for each site to arrive at area-based values. Volume-based values were derived by dividing the area-based values by the site depth.

STATISTICAL ANALYSES

Statistical parameters and definitions are according to Snedecor and Cochran (1967) and Lamont, et al (1977).

RESULTS AND DISCUSSION

RESERVOIR HYDROLOGY, MORPHOMETRY, AND OPERATION

The hydrologic budget for 1981-1982 is shown in Table 5. The North Fork accounted for 32 percent of the total basin water yield in 1981-82. Total discharge from Stewart Mill Creek in 1981-82 contributed 22 percent of the total annual water yield for the basin. Annual discharge from Stewart Mill in 1981-82 was approximately 11 percent greater than discharge in 1974-75 (Knight, et al, 1975), while the North Fork decreased by 8 percent. Average annual discharge from Stewart Mill, calculated from data for 1973-75 and 1981-82, is 6,529 acre feet (8.059×10^6 m^3), with a standard deviation of 837 acre feet and coefficient of variation of 0.128.

Table 5. Hydrologic budget for 17 June, 1981-31 May, 1982.

Source	Inputs $m^3 \times 10^6$	Outputs $m^3 \times 10^6$	Percent Total
North Fork	12.2926		32
Stewart Mill	8.732		22
Hardtla Ck	1.194		3
Groundwater	16.686		43
Flint Ck		40.8329	92
Net Evaporation		3.6517	8
Storage	-5.5893		
Total	38.891	38.891	100

Approximately 4530 hectares are not drained by any surface stream, thus the only discharge from this area is groundwater. Evidence also points to considerable groundwater flow from Silver Lake to Georgetown when Silver is approaching capacity (Personal Communication, Phil Farnes, SCS, Bozeman, Mt). Approximate locations of groundwater inputs are shown in Figure 3, which identifies three probable areas of most importance, in terms of volume. Other areas of submerged groundwater input were located in upper Stewart Mill Bay and Jericho Bay. Some subsurface discharge also occurs along other shoreline areas during late spring and early summer. The presence of wetlands type vegetation on the east shore and adjacent area of Stewart Mill Bay and south shore of Jericho Bay indicate probable groundwater discharge.

Monthly average groundwater flow in 1981-82 varied from a high of 53 cfs in June to an average of 16 cfs during January-May, 1982. Groundwater contributed 43 percent of the total basin water yield for 1981-82, with a total yield of 1.668×10^7 m^3. Data recalculated from Knight, et al (1975) show that groundwater flow accounted for 50 and 40 percent of the total basin yield in 1973-74 and 74-75, respectively. For the three year period, the average groundwater discharge is 13,197 acre-feet (1.629×10^7 m^3), with a standard deviation of 2.19×10^6 m^3 and coefficient of variation, 0.134.

The correlation between percent groundwater flow and total yield is negative, indicating that surface flows account for most of the variation in basin yields. Over the three year period for which data are readily available, groundwater flows (which in this sense include

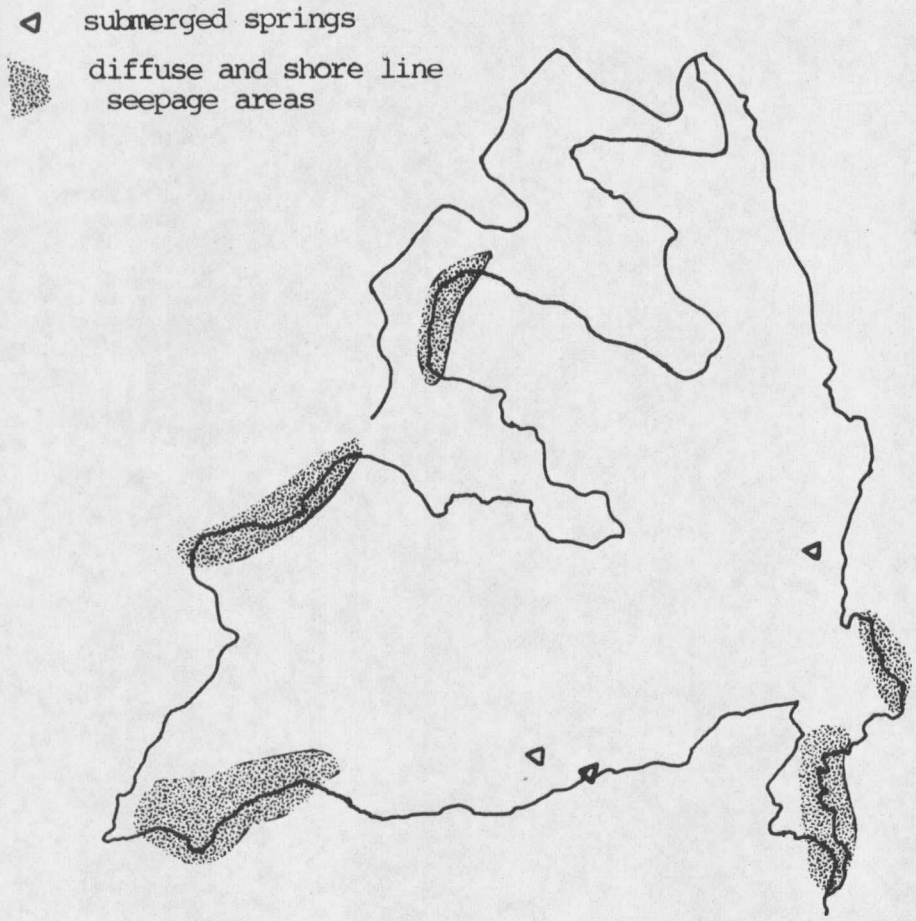


Figure 3. Approximate locations of groundwater inputs.

Stewart Mill Creek), have contributed 73, 67, and 57 percent of the total yield, with an average of 67 percent.

The average basin yield for the past 20 years from Georgetown Lake is 28,100 acre-feet (3.469×10^7 m^3) per year (Phil Farnes, SCS, Bozeman, Mt, unpublished data). Using the above estimate for groundwater contribution, the average groundwater yield is (0.67 x 28,010) 18,767 acre-feet (2.316×10^7 m^3)/year; the average surface input is 9,240 acre-feet (1.141×10^7 m^3)/year.

The percentage of total surface area between 1 meter elevation contours is shown in Table 6. Sixty-five percent of the total surface area lies above the 6.5 meter depth contour, and the largest interval is that between 6 and 7 meters.

Lake surface elevations (Table 7) were considerably lower during 1973-1975 than in 1981-1982, due to low hydrologic yield and higher industrial use. The maximum surface elevation obtained (Anaconda gage datum) in 1973 was 6427.2 feet. The minimum elevation that winter was 6424.8, a drawdown of 4.7 feet (1.43 m) below maximum stage. In 1974, the maximum surface elevation obtained was 6428.1, as compared to conditions in 1981-82, during which surface elevations remained near 6429.5 during the ice free season. Based on review of Anaconda Company water level records, the reservoir reaches its maximum surface elevation in early July. During the period from 1952-1969, water levels were drawn considerably lower than in recent years, due to greater industrial water use by the Anaconda Company Smelter. For many years, pumping by Anaconda Company was a major source of withdrawal during winter from Georgetown Lake. Water was pumped through the

pumphouse on Highway 10A to Silver Lake, stored, then pumped through an aqueduct to Warm Springs Creek, which flows south to Anaconda, where it would then be withdrawn for process water in the smelter or pumped on to Butte for use in facilities there.

Table 6. Area-depth relationships in Georgetown Lake.

Depth Contour (m)	Area (ha)	% Total	Cumulative %
0-1	67.8	5.6	5.6
1-2	70.2	5.8	11.4
2-3	70.2	5.8	17.2
3-4	110.1	9.1	26.3
4-5	124.7	10.3	36.6
5-6	143.5	12.0	48.6
6-6.5	198.5	16.4	65.0
6.5-7	335.2	27.7	92.7
7-8	59.3	4.9	97.6
8+	29.1	2.4	100.0

Table 7. Average seasonal reservoir surface elevations for 1952-1982. Anaconda Company reservoir gage datum. Elevation of spillway, 6429.5 feet.

Year	Summer Ave	Winter Ave	5 Jul	Summer Max	Annual Min	
81-82	6429.4	6429.1	6429.4	6429.5	6427.7	Mar
80-81	29.4	28.8	29.5	29.5	28.4	Mar
79-80	29.5	27.7	29.6	29.6	26.8	Apr
78-79	29.1	28.5	29.3	29.3	27.4	Apr
77-78	29.5	27.7	29.5	29.5	26.9	Apr
76-77	29.0	29.1	29.3	29.3	28.6	Mar
75-76	29.4	28.9	29.4	29.4	27.6	May
74-75	29.5	27.3	29.4	29.7	26.4	May
73-74	27.9	25.9	27.7	28.1	24.8	Mar
72-73	26.6	28.1	27.2	27.2	25.9	Sep
71-72	29.5	28.8	29.3	29.6	27.7	May
70-71	29.5	28.4	29.3	29.6	27.2	May
69-70	29.4	28.1	29.5	29.5	27.2	May
68-69	29.1	28.5	29.2	29.2	27.4	Apr
67-68	29.1	28.4	29.0	29.2	27.4	May
66-67	29.1	26.1	29.1	29.2	25.2	May
65-66	27.1	28.3	27.2	27.2	26.0	May
64-65	28.9	28.4	28.8	29.0	26.4	May
63-64	29.1	28.3	29.1	29.2	27.6	Apr
62-63	29.0	27.8	29.1	29.1	26.8	Apr
61-62	28.8	27.0	28.9	28.9	25.9	Apr
60-61	27.7	27.5	28.2	28.2	26.6	Apr
59-60	28.7	28.4	28.9	28.9	27.2	May
58-59	28.3	28.2	28.1	28.5	26.1	May
57-58	28.9	26.1	28.9	28.9	25.5	Apr
56-57	27.0	25.8	27.1	27.2	24.2	May
55-56	27.7	26.9	26.7	26.9	25.0	May
54-55	28.7	26.9	28.5	29.0	25.9	Apr
53-54	27.4	26.8	27.5	28.7	25.8	Apr
52-53	28.8	26.7	28.7	28.9	26.3	Feb

LAND USE AND STREAM WATER QUALITY

Surface Land Use/Cover Classification

Land cover/use was classified into seven classes. Classifications were made using USGS 7.5 minute quad maps, black and white aerial stereo photographs at a scale of 1:24,000, and data from the Forest Service land use classification program (Ruppert, 1980). The classifications utilized, which are based on those given by McElroy, et al. (1976) and Reckhow, et al. (1980), are given in Table 8. The area of each land use/cover type in the basin was determined by planimetry. Table 9 summarizes the total percent of each land use/cover type in the basin.

The most common land cover type in the basin is coniferous forest. For study purposes, rock outcroppings and talus slopes were included in this land cover type. Under the Forest Service preliminary inventory, 71 percent of the total area under Forest Service management is considered commercial forest with potentially harvestable timber resources. Major timber types include lodgepole pine, Douglas fir, limber pine, subalpine fir, and mixed conifers, which comprise over 90 percent of the total forested area.

As of 1981, timber had been harvested from approximately 1400 acres (567 ha). The method of harvest has been primarily clearcutting, the majority of which has occurred in the Stewart Mill Creek drainage. All clearcuts were several miles from the reservoir, so have had no measureable impact on esthetics or water quality.

Other small clearcuts have occurred in the Flint Creek drainage. Several timber sales have been completed, with projected additional harvest in the North Fork Flint Creek drainage of approximately 400 ha, which includes areas which are to be developed for ski lifts and runs at the Discovery Basin Ski Area. This will bring the total acreage in clear cuts to approximately 1750 acres (708 ha).

Table 8. Land Use and Cover Classifications used in the Georgetown Lake Clean Lakes Project.

Use/Cover Type	Symbol	Description
Coniferous Forest	(CF)	Timbered areas not adjacent to streams, or timbered areas not shown as wetland or riparian
Grassland/Meadow	(GL/M)	All grassland, including wet mountain meadows not in stream bottoms
Clearcut	(CC)	Areas from which all harvestable timber has been removed.
Riparian/Wetland	(RP/WL)	Wet areas along stream courses around lakes which were identifiable by vegetation type.
Rural Residential	(RR)	Areas upon which homes have been constructed and an one acre radius around the structures.
Recreational	(REC)	Public campgrounds, picnic areas
Commercial	(CM)	Private commercial facilities

Riparian/wetland areas tabulated were associated with Georgetown Lake or the North Fork of Flint Creek. Acreages obtained were less than the Forest Service land inventory indicated because narrow riparian zones along the stream course were not included.

Riparian/wetland land cover type was primarily willow-sedge-grass scrub or sedge-grass wet meadows (Cowardin, et al., 1979). There are several such wetland areas on the North Fork of Flint Creek (Figure 4) which should be considered critical habitats for water quality management, as they are probably acting as traps for sediment and nutrients from the upper drainage (van der Valk, et al., 1979; Boto and Patrick, 1979).

Table 9. Percent of land use/cover classes in the Georgetown Lake Basin.

Land use/cover type	% Total area	Acres	Hectares
CF	76.19	25,830	10,457
GL/M	15.97	5,414	2,192
CC	5.60	1,900	769
RP/WL	1.22	414	168
RR	2.46	834	338
REC	0.41	139	56
CM	0.04	14	6

The riparian zone bordering the North Fork of Flint Creek should also be considered critical habitat, as it stabilizes soils adjacent to the stream bed during high water and prevents erosion (Allen, 1979). The riparian type acreage tabulated also includes a narrow zone bordering the shoreline of Georgetown Lake within which vegetation is influenced by bank seepage from the reservoir or shallow groundwater inflows. In general this riparian zone was considered to be approximately 50 feet (18 m) wide, although in some areas (such as Stewart Mill Bay and the upper end of Phillipsburg and Rainbow Bays, it is considerably wider. The total area represented is approximately 78 acres (31.6 ha). It is important in intercepting nutrients

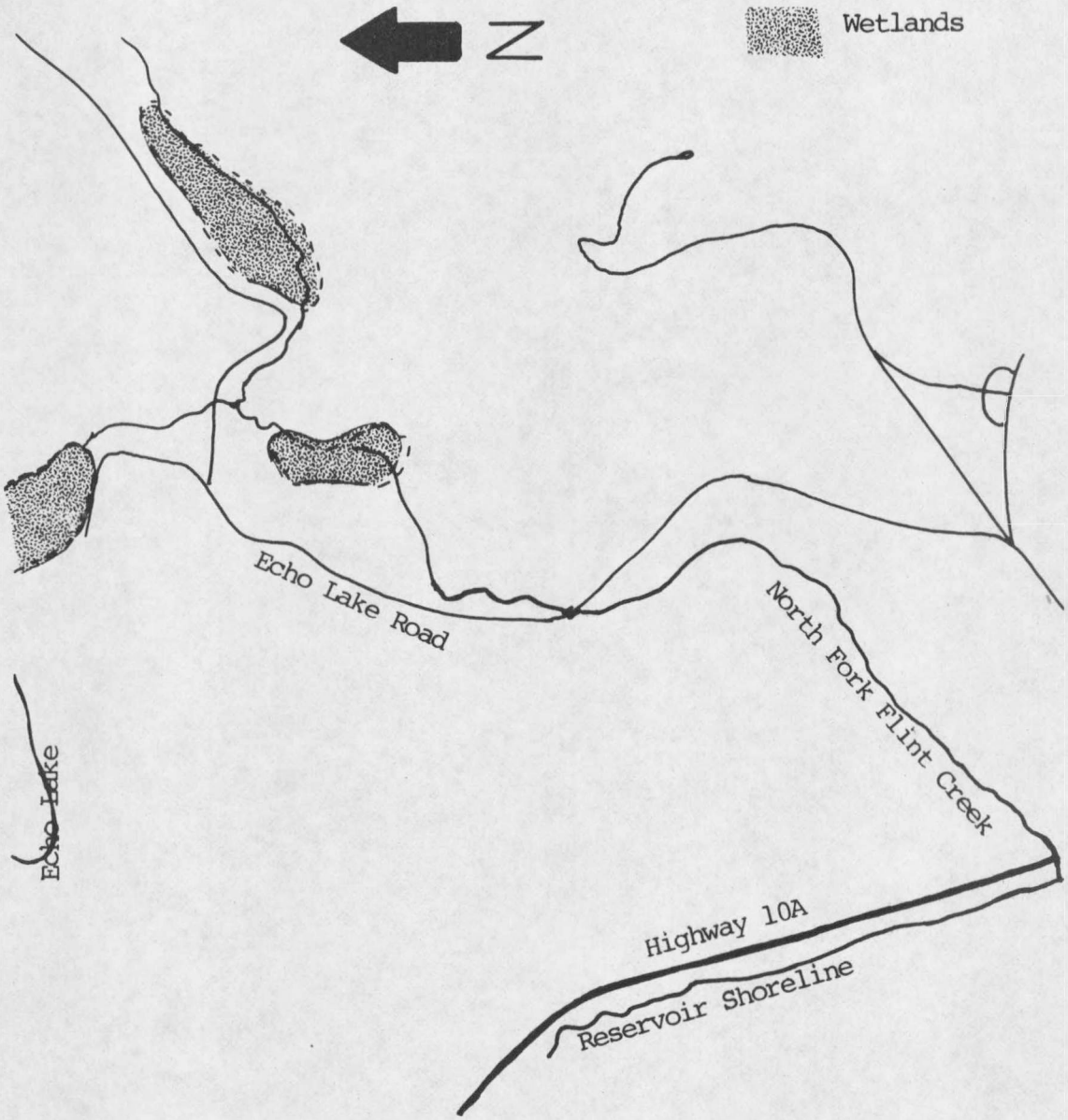


Figure 4. Wetlands associated with North Fork of Flint Creek which should be preserved to maintain water quality.

and silt from direct surface runoff around the perimeter of the reservoir and protecting the shoreline from erosion.

The total acreage tabulated in Rural Recreational type was 840 (340 ha). There are a total of ten platted subdivisions within a one mile radius of the reservoir. The average lot size, number of platted lots, and location by section are shown in Table 10. The total number of subdivision lots is 821. The overall average lot size is 0.77 acres (0.31 ha); this includes some open areas in the old Eccleston's Subdivision which are not developed, but included in the subdivision boundary. The area occupied by subdivided lots is approximately 632 acres (256 ha); this does not include access roads and open areas not platted for residential development, or isolated lots under lease or private ownership. The total area under subdivision, including access roads, is estimated at approximately 700 acres (283 ha).

Additional shoreline residential development has occurred as individual residential leases from the Anaconda Company (78) and Forest Service (37), totaling 115 individual tracts. There are approximately 10 private owners who have smaller tracts near the reservoir. Allowing approximately 1 acre per ownership under the residential category, the total residential acreage in non-subdivided tracts under private ownership is estimated at 16. Assuming the lease tracts are approximately the same size as private, subdivided tracts, or may be considered to influence approximately the same area, they represent another (115x0.31) = 36 ha, which may be classified as residential. These leases are scattered around the

Table 10. Parameters of subdivisions in the Georgetown Lake area.

Subdivision	County	Range/Township/Section	Number lots	Ave. Lot size
Georgetown	Deer Lodge	5N/13W/17	176	0.08 Ac.
Tract	Deer Lodge	5N/13W/17	39	0.60
Sunset Homesites	Deer Lodge	5N/13W/17	67	0.63
Mountain View	Deer Lodge	5N/13W/8	40	0.60
Denton's Tract	Granite	5N/14W/14	114	0.35
Georgetown Village	Granite	5N/14W/24	27	0.55
Edwards's Tract	Granite	5N/14W/24	117	0.30
			10	1-2
Lakeview	Granite	5N/14W/26	93	0.88
			1	28.1
			1	21.8
Shakopee Heights	Granite	5N/14W/15	49	2.31
Eccleston's*	Granite	5N/14W/14	87	2.1

*Approximate lot size includes unplatted open areas. The actual lots are considerably smaller.

shoreline of the reservoir and represent the majority of shoreline development. At the time they were granted, there were no restrictions on the construction of individual boat docks, which has resulted in a proliferation of small docks and piers in several areas, notably sections 24, 13, 12, and 6. A few home owners have resorted to rip-rapping short sections of the shoreline around their private boat docks to prevent erosion. The Butte Boat Club represents an additional 39 residences and approximately 11 acres of residential land use/cover type.

Moderate increases have occurred during the period from 1972 through 1981 in several subdivisions around the reservoir, totaling approximately 42 units. Total construction of new residential units was approximately 75-80; this includes estimated construction in Sunset, Mountain View, Georgetown tracts, or Gangier's tracts of an estimated 20 additional units, and construction of 20 units on leases. A review of sanitary permits issued confirmed this estimate.

The Commercial (CM) land use type occupies approximately 14 acres in the drainage. This does not include the ski runs at Discovery Basin, which were placed under CC or GL/M.

Septic Leachate Survey

Results of the Rhodamine dye study were negative. No dye was detected, indicating insignificant contributions of septic tank effluents to the reservoir. However, additional surveys for septic tank effluents performed by Swanson Environmental, Inc., assisted by the author, located five active septic leachate plumes and two others

which were considered inactive (Swanson Environmental, 1981; Garrett, 1983). Subsequent chemical and biological sampling did not indicate detectable effects on chemical or bacteriological water quality due to these plumes.

As part of the septic leachate survey, data on use and characteristics of residential development in the Georgetown Lake area were collected. A questionnaire was developed prior to the dye study, and responses filled in for cooperating homeowners by volunteers who distributed the dye.

Approximately 120 questionnaires were returned, from which 65 were selected randomly for analysis, taking care that all areas of the shoreline were represented. Results are shown in Table 11.

Over 50 percent of the residences surveyed were within 100 yards of the shoreline and over 90 percent were within 400 yards (400m) (estimated). The questionnaire results indicated that 94 per cent of the residences surveyed were of the "second home" recreational type

Table 11. Residential parameters for the Georgetown Lake area.

Age (Years)			Lot size (Acres)			Use			Septic system		
	#	%		#	%		#	%		#	%
0-5	9	15	<1/2	41	64	Seasonal	62	94	Septic tank	33	59
6-10	8	13	1/2-1	20	31	Permanent	4	6	Cesspool	12	21
11-20	15	25	1-2	2	3				Outhouse	15	27
20-40	17	28	2+	1	1				Other	3	5
40+	12	20							Unknown	1	1

(indicated by the Seasonal Use category). Cooperating homeowners were asked to estimate the average number of days their residence was

when in use. Heaviest use occurred in June, July, and August (about 90 percent of the total occupant days). The average total occupant-days per residence (average number of occupants times the total days occupied per year) was 260, equivalent to about 0.7 occupant-years per residence (260/365). The range of total occupant-days per residence went from a low of ten to a high of four hundred. There are an estimated 20 family residences occupied year-round in the immediate vicinity of Georgetown Lake, and about 21 additional residences with year-round occupants at Southern Cross.

The Recreational land use category area was 139 acres (56.3 ha). This includes three Forest Service campgrounds around the reservoir, public launching ramps at Red Bridge, Grassy Point, Piney Point, and Phillipsburg Bay, and three areas owned by Anaconda Company which are being used as public campgrounds, located on the east and west shores of Jericho Bay, and at Stewart Mill Bay, with a capacity of approximately 80-85 units. The total capacity of all available camping facilities is approximately 233 spaces [Denton's (45) + Forest Service (103) + Anaconda (85)]. This will be reduced to approximately 145 if the Anaconda Company closes their facilities now available to the public.

Water Quality of Surface Streams and Groundwater

Results of chemical analysis of surface streams draining into Georgetown Lake did not indicate a deterioration of water quality during the period from 1973-1982 (Tables 12, 13). Total phosphorus

Table 12. Means and ranges of chemical parameters analysed in 1973-1982 for surface streams flowing into or out of Georgetown Lake.

	NORTH FORK FLINT	FLINT CREEK OUTFLOW	STEWART MILL CREEK
pH			
Range	7.7-8.5	7.6-9.1	7.6-8.3
Alkalinity	2.24	2.16	2.82
meq/l Range	2.05-2.53	1.51-3.20	2.70-2.88
Spec Cond	250	228	295
Range	134-272	168-362	265-315
D. O.	9.2	9.0	9.8
mg/l Range	5.9-10.9	5.6-11.2	7.5-11.6
NO_3-N	6	5	58
ug/l Range	0-36	0-46	16-99
NH_3-N	5	43	5
ug/l Range	0-23	0-410	0-18
Total-N	168 /l	410	161
ug/l Range	0-355	44-1015	0-350
Total-P	16	21	18
ug/l Range	4-38	16-134	3-32
Calcium	31	33	41
mg/l Range	30-38	19-38	39-46
Magnesium	8	8	10
mg/l Range	8-12	6-10	8-12
Total Suspended Solids. mg/l[1]			
Range	1-20		0-3

[1]Data from U. S. Forest Service, Butte, Montana

Table 13. Monthly averages of nitrate and ammonia nitrogen concentrations in surface streams flowing into and out of Georgetown Lake. Concentrations in ug/l.

	NO_3			NH_3		
	NFFC	SMC	FCO	NFFC	SMC	FCO
Jul 73	3	60	3	1	1	17
Aug 73	6	72	20	3	2	3
Sep 73	8	62	26	0	0	6
Dec 73	6	70	7	1	2	46
Jan 74	15	34	0	0	0	75
Feb 74	20	58	8	7	4	300
Apr 74	20	79	20			
Jun 74	4	28	3	1	2	1
Jul 74	3	60	14	8	7	12
Aug 74	5	72	12	11	8	11
Sep 74	5	69	3	12	8	12
Nov 74	6	55	7	6	6	7
Dec 74	0	53	0	8	8	28
Jan 75	16	50	32	5	5	46
Feb 75	21	66	7	8	8	277
Mar 75	16	54	10	6	4	159
Apr 75	31	54	0	10	6	162
May 75	0	35	9	6	3	145
Jun 75	1	54	14			
Jun 81	0	42	0	6	4	4
Jul 81	0	40	1	4	4	1
Aug 81	0	52	0	8	4	3
Sep 81	0	51	2	6	13	12
Oct 81	0	58	0	0	0	0
Nov 81	0	64	0	0	0	8
Jan 82	12	78	5	4	0	106
Feb 82	12	73	5	0	0	139
Mar 82	13	75	12	23	11	124
Apr 82	21	99	27	0	0	68
May 82	13	73	13	0	18	156

Georgetown Lake did not indicate a deterioration of water quality during the period from 1973-1982 (Tables 12, 13). Total phosphorus and total nitrogen levels (Table 14) measured were lower in 1981-82 than in 1973-1975. This may be an analytical artifact, however, the relationship between monthly average total phosphorus concentration and monthly average stream flow for the North Fork of Flint Creek in 1981-82 closely follows the relationship predicted by nationwide data analyzed under the National Eutrophication Survey, which predicts a 0.11 % decrease in total phosphorus concentration for a 1.0 % increase in flow (EPA, 1975). The relationship for the North Fork follows the equation:

$$[TP] = -6.7477 \log_{10} (flow)_{cfs} + 19.3498$$

$$r^2 = 0.923;\ n = 5;\ 9\ ug/l < TP < 16\ ug/l;\ 4 < flow < 44.$$

Omernik (1977) gives mean total nitrogen and total phosphorus values for streams draining basins with 90 % forest cover in the western United States of 601 ug/l and 22 ug/l, respectively. Qualitative modifiers given by Omernik (1977) indicate that drainages in the Pacific northwest usually have values for total nitrogen and total phosphorus up to 50 percent below the mean. Calculation of a flow-weighted mean for total nitrogen and total phosphorus (= total annual load / total annual discharge), gives values of 16 ug/l total phosphorus and 235 ug/l total nitrogen for Stewart Mill Creek, and 18 ug/l total phosphorus and 214 ug/l total nitrogen for North Fork Flint Creek. Hydrologic data of Knight et al, (1975) were used to obtain discharge for 1973-1975. These results agree with the

Forest Service as being in the "Low" and "Moderate" erosion potential classes. This is reflected in the low total suspended solids (a measure of sediment and silt) measured in the North Fork of Flint Creek (USFS, Butte, Montana, unpublished data). Forest Service values for total suspended solids range from 1-20 mg/l, with highest values being associated with peak flows. However, there does not appear to be an apparent relationship between total suspended solids and flow (Figure 5), and the linear correlation between total suspended solids and flow is not significant ($r^2 = 0.22$; $n = 28$). This would seem to indicate that increased surface runoff is not consistently related to the external stream sediment load. Comparisons of values of total suspended solids in North Fork of Flint Creek from 1974-1982 do not show a trend of increasing concentrations with time, (U. S. Forest Service, unpublished data) which indicate that, for the present, water quality in the drainage is not deteriorating.

The nearest industry of any significance to water quality in Georgetown Lake was the Anaconda Company Reduction Works, 16 miles (25.6 km) from the reservoir in the town of Anaconda. This facility used Georgetown Lake as a storage reservoir for industrial process water. There are no point source discharges, major mining activity, or significant agricultural activities in the basin upstream from Georgetown Lake. Some mining has occurred in the past and exploration is expected in the near future, as mineral deposits of economic significant are indicated in the area (U. S. Forest Service, Deer Lodge National Forest Planning Team, unpublished data; Ingman and Bahls,

Table 14. Monthly averages of total nitrogen and total phosphorus concentrations in streams flowing into and out of Georgetown Lake. Concentrations in ug/l.

Date	Total-N			Total-P		
	NFFC	SMC	FCO	NFFC	SMC	FCO
Jul 73	200	200	250	9	12	20
Aug 73	20	70	700	11	10	30
Sep 73	260	280	650	11	11	58
Dec 73	110	290	160	6	12	22
Jan 74	300	200	440	17	16	34
Feb 74	40	60	460	36	12	72
Jun 74	180	90	280	25	16	35
Jul 74	170	140	570	10	11	30
Aug 74	120	80	450	22	16	36
Sep 74	180	220	530	20	15	30
Nov 74	350	150	550	7	8	18
Dec 74	170	180	830	12	14	21
Jan 75	260	330	800	13	13	25
Feb 75	110	160	360	18	17	49
Mar 75		40	150	9	10	69
Apr 75	30	50	340	12	13	100
May 75	80	50	750	18	22	128
Jun 75	230	190	340	28	16	39
Jun 81	110	150	250	9	12	20
Jul 81	90	150	230	11	13	23
Aug 81	80	150	120	13	15	24
Sep 81	80	170	220	14	13	23
Oct 81	80	170	220	12	13	17
Jan 82	280	280	320	22	8	22
Feb 82	280	280	340	9	10	28
Mar 82	280	280	360	8	9	25
Apr 82	280	280	390			22
May 82				14	8	20

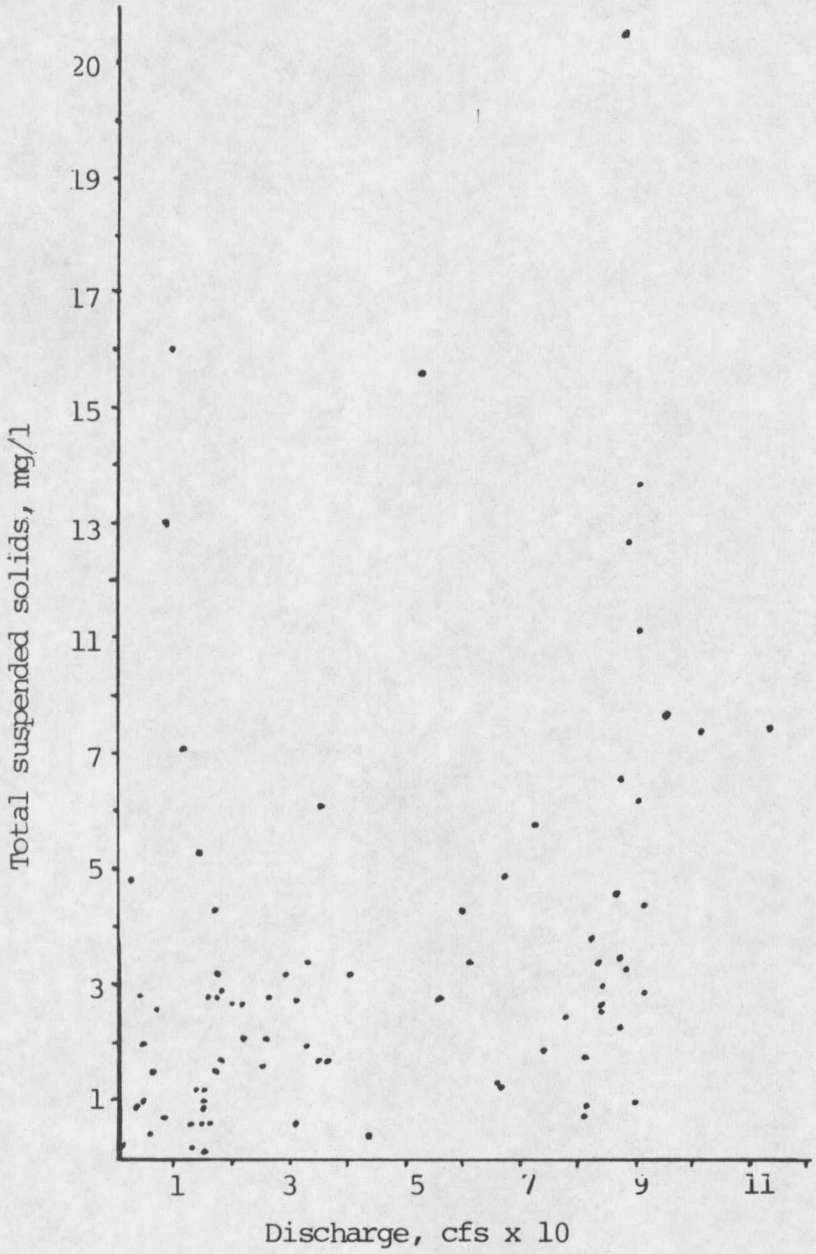


Figure 5. Total suspended solids versus Discharge in North Fork of Flint Creek, 1981-1982. Data courtesy of U. S. Forest Service, Butte, Montana.

1979). Timber harvest also has some potential for impacting water quality, primarily through increases in erosion and siltation to the North Fork of Flint Creek. The major potential impacts of further recreational, residential, mining, and agricultural developmentment are associated with erosion during construction, increased surface runoff due to removal of natural vegetation and construction of roads, parking areas, and walkways, and increased nutrient export due to domestic sewage wastes and fertilizers.

SOLAR RADIATION

Average total incident solar radiation at Georgetown Lake (latitude 46° 10" N) was calculated for bi-monthly intervals from June through mid-October (Table 15). Total daily incident radiation varied up to 628% within a single 15-day period due to cloud cover. Maximum average daily radiation occurred from 1-15 July and decreased considerably subsequent to 31 August. Monthly averages of daily total incident radiation for alpine sites given by Hutchinson (1957a), shown in Table 16, are similar to values obtained for Georgetown Lake and indicate the general significance of local weather patterns in influencing incident radiation.

Data on light penetration and water clarity in Georgetown Lake are summarized in Table 17. Mean vertical extinction coefficients in 1974 during the ice free season averaged about -0.70, while in 1981, the average was approximately -0.43. Measured at

station I, the depth of the euphotic zone, defined as the depth below the water surface at which 1% of the incident radiant energy in the spectrum from approximately 380-700 nanometers remains (Ryther, 1956), reached a maximum of 12 meters and a minimum of 9.4 meters in 1981. Values obtained in 1981 for light penetration were 2-85 percent greater than those observed in 1974.

Table 15. Bimonthly averages for daily total incident radiation (DIR) (cal/cm^2/day) at Georgetown Lake.

Period	Mean DIR	N	s	Range
16-30 Jun	605	12	140	290-773
01-15 Jul	610	14	97	290-760
16-31 Jul	620	19	107	231-780
01-15 Aug	501	9	129	334-628
16-31 Aug	534	11	93	343-610
01-15 Sep	482	12	93	290-569
16-30 Sep	301	16	152	74-465
01-15 Oct	304	2		278-331

Table 16. Monthly means of total incident radiation (cal/cm^2/day) at two alpine localities in Europe (Hutchinson, 1957a).

	Lat. N	Jun	Jul	Aug	Sep	Oct
Zugspitze	47°	562	433	509	368	277
Davos	46°	621	640	511	468	358

Table 17. Average depth of euphotic zone, 1974 and 1981.

	1974[1]	1981	% Increase
June	7.5	9.4	25
July	6.5	12.0	85
August	6.1	10.0	64
September	6.9	9.7	41
October	10.2	10.4	2

[1] Calculated from mean vertical extinction coefficients given by Knight (1980).

MACROPHYTE COMMUNITIES

Community Composition

The most significant biological feature of Georgetown Lake is the benthic plant community, which appears to be the driving force of lake metabolism and strongly influences chemical cycles of carbon, nitrogen and phosphorus. Aerial color infra-red photography obtained in 1975, 1981, and 1982 indicates that benthic macrophyte coverage is essentially complete in terms of presence, to a minimum depth contour of 6408.2 on the Anaconda gage (maximum surface elev. 6429.5), which represents a depth at full pool of about 6.5 meters. The area-contour interval relationships shown in Table 6 confirm that macrophyte communities are present over 65-70% of the total reservoir surface area, as the euphotic zone is typically 6 meters or more in depth during the growing season.

A list of aquatic vascular plants identified in Georgetown Lake is given in Table 18. In addition to the vascular plants identified, the macrophytic algae Chara and Nitella are significant components of the benthic plant community. Chara commonly occurs over a wide range of depths, while Nitella is found only in the deeper areas.

Several characteristic plant communities may be identified, although many species are widely distributed and occur across a considerable depth range. The border of the reservoir to a depth of about 0.75 m, which spans the normal zone of fluctuation in water level, is characterized by the presence of Chara sp., which occurs

in sparse to dense mats with standing crops of up to 250 g dry wt/m^2. Polygonum amphibium also occurs in several areas within this depth zone, particularly around the mouth of Stewart Mill Bay, although it is scattered around much of the protected shoreline areas. It does not occur along windy shorelines which are exposed to much wave action, such as along Highway 10A.

Depth contours from 1-2 meters include communities dominated by Myriophyllum exalbescens, Anacharis canadensis, and Potamogeton richardsonii, with a generally continuous understory of Chara spp. These three vascular plants are a dominant feature of the benthic plant community, occurring at depths up to 4 meters in many areas of the reservoir. They are major primary producers in the reservoir, obtaining high standing crops over a wide depth range. Stands of Potemogeton pectinatus and Potamogeton friesii occur around the 1-1.5 m contour and appear to be temporary features, depending on annual hydrologic and meteorologic conditions.

The most extensive plant community in the reservoir, in terms of area, is characterized by Potamogeton praelongus, which grows in "clumps" which reach the surface in midsummer, and produce aerial flowering spikes, interspersed with "meadows" of Chara, which also are colonized by Anacharis, Myriophyllum, and P. richardsonii. The basic association occurs between contours from 3.5 to 6.5 meters, occupying a total area in excess of 600 hectares.

Potamogeton zosteriformis, a species common to high conductivity waters in eastern Montana (Booth and Wright, 1959), attained

Table 18. Submerged and floating-leaved aquatic vascular plants identified in Georgetown Lake. Nomenclature according to Hitchcock and Cronquist (1973) and Muenscher (1944).

ALISMATACEAE
- *Alisma plantago-aquatica* L.

POTAMOGETONACEAE
- *Potamogeton alpinus* Balbis
  - *friesii* Rupr.
  - *gramineus* L.
  - *pectinatus* L.
  - *praelongus* Wulf.
  - *pusillus* L.
  - *richardsonii* (Bennett) Rydb.
  - *vaginatus* Turcz.
  - *zosteriformis* Fern.

SPARGANIACEAE
- *Sparganium eurycarpium* Engelm.

HYDROCHARITACEAE
- *Anacharis canadensis* (Michx.) Planchon.

LEMNACEAE
- *Lemna trisulca* L.

POLYGONACEAE
- *Polygonum amphibium* L.

CERATOPHYLLACEAE
- *Ceratophyllum demersum* L.

RANUNCULACEAE
- *Ranunculus aquatilis* L.

HALORAGACEAE
- *Myriophyllum exalbescens* Fern.

LENTIBULARIACEAE
- *Utricularia vulgaris* L.

ISOETACEAE
- *Isoetes howellii* Engelm.

high standing crops in 1973-76 along the shoreline bordering Highway 10A north of the inlet of the North Fork of Flint Creek, but was almost non-existent in 1981-82, having been replaced by P. praelongus in that area. Ranunculus aquatilis occurs at depths of 1-1.5 meters, and in some areas, notably around the mouth of Stewart Mill Bay and on the south shore of Jericho Bay, reaches densities sufficient to constitute a nuisance to boaters. Ceratophyllum demersum, although present in 1973-76, was not at any time abundant relative to the dominant species. This is not unexpected, as this species does not have a functional root system, as do Anacharis, Myriophyllum, and other common species, and must obtain its mineral nutrition from dissolved nutrients in the water, sediment sources being relatively unavailable (Schulthorpe, 1967).

Other species present are relatively minor, do not contribute significantly to primary productivity, and do not constitute a nuisance to boaters or fishermen.

Aerial photography obtained in August of 1975, 1981, and 1982, revealed significant changes in the distribution and abundance of P. praelongus (Figures 6-9). Three areas, each covering approximately 4-7 ha, for which overlapping imagery was available, were selected for analysis of canopy coverage. Each area represents a different level of colonization. All areas of study are approximately 4.5-5.5 m deep. P. praelongus is a perennial (Schulthorpe, 1967), and stands probably represent a few plants, each with many stems interconnected by rhizomes. Individual clumps

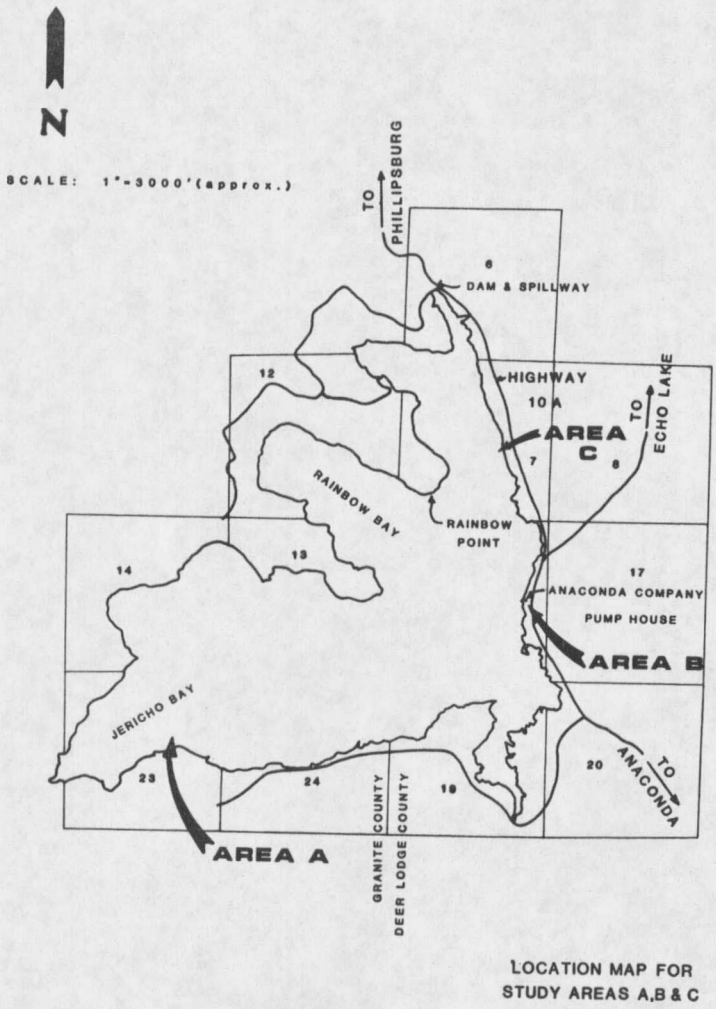


Figure 6. Location of study areas for remote sensing of P. praelongus canopy coverage, 1975-1981.

Figure 7. Study area A, canopy coverage of P. praelongus.

Figure 8. Study area B, canopy coverage of P. pralongus.

Figure 9. Study area C, canopy coverage of P. praelongus.

are recognizeable over all three years' photography, and in some cases it can be determined that several individual clumps increased in size to form a continuous stand. Observations with SCUBA showed that canopy coverage within a clump is 100%, and that the clumps are relatively pure, with little or no understory of any kind. Accumulations of detritus on the bottom form a loose layer of particulate organic matter, but accumulations of sedimentary detrital material were in general slight, with firm clay, silt or gravel often being encountered within 8-10 cm of the surface. Stem density was estimated as approximately 24 stems/m^2 (range = 14-32).

Canopy coverage increased at sites A and B (Figures 7, 8) by 100%, i. e., the extent of areal coverage (which in this case would also equal basal area of the stand) increased twofold. Total percent surface coverage in 1981 for site A was 62%, at site B, 27%. At site C (Figure 9), canopy coverage increased by 700%, but the percent total coverage was not determined (<5%). Marked increases in this plant were also observed in the Phillipsburg Bay area, the stands there being very vigorous, with standing crops greater than those observed in other areas (Table 19).

Photography of the same areas in 1982 showed no significant discernable changes in the canopy coverage of P. praelongus. The canopy coverage (as represented by stand area) doubled between 1975 and 1981, which suggests exponential population growth, equivalent to a population growth rate r (Slobodkin, 1961) = 0.1155/year. The Georgetown Lake dam was raised to its present elevation in 1937 (Lutey, et al, 1974). Assuming an exponential growth rate of

0.1155/year, the relative population density of P. praelongus in 1937, as canopy cover, can be estimated as 1/exp (44 x 0.1155) = 0.62% of the density present in 1981.

Numerous cases of such exponential population growth of species invading open habitats, as may have been created by the raising of the Georgetown Lake dam, have been documented (Hutchinson, 1957b; Slobodkin, 1961). Although the hypothesis of sustained exponential growth of P. praelongus is not verifiable at this time, continued monitoring should provide evidence for or against it.

Distribution of other species was not evaluated by use of aerial photography, other than to determine the presence of plant canopy, which was relatively continuous in all areas of the reservoir between the 1 meter to 5 m contours, the latter representing the limit of image differentiation, using color and texture as recognition criteria. Exceptions occurred along the east shore between the pumphouse and the North Fork inlet, which is very windy and exposed to the full reach of wave action. In such areas wave turbulence has often been found to limit colonization by rooted aquatic plants (Jupp and Spence, 1977).

Standing Crop and Biomass

Standing crops were estimated for several areas in the reservoir from 1974-1977 and in 1981-82 by harvesting 0.5 m^2 quadrats with aid of SCUBA. Underground parts were not included. Results are summarized in Table 19.

Table 19. Standing crops (g dry weight/m^2) of macrophyte communities in Georgetown Lake. n = number samples; s = standard deviation of mean.

Date	Site	Community Type	n	biomass	s
6 Oct 73	P'burg Bay	Anacharis	5	260	39
18 Aug 81	"	Anacharis	7	735	214
27 Aug 82	"	Anacharis	7	228	89
18 Jul 74	"	P. richardsonii	10	186	58
24 Aug 74	P'burg Bay	P. richardsonii	10	241	89
25 Aug 74	Stewart Mill Bay	M. exalbescens	6	285	97
18 Sep 74	"	M. exalbescens	9	365	107
18 Aug 74	Jericho Bay	P. praelongus	8	185	61
25 Aug 76	"	P. praelongus	9	224	78
18 Aug 81	P'burg Bay	P. praelongus	12	320	114
18 Aug 81	Jericho Bay	P. praelongus	6	156	53
27 Aug 82	"	P. praelongus	7	259	87

In 1974 and 1976, serial standing crop samples were collected in Rainbow Bay at the 2-3 meter depth contour, Stewart Mill Bay at 1.5-1.8 meters, and Philipsburg Bay between 1.7-2.2 meters, for estimates of production based on annual standing crop accrual (Westlake, 1969). In Rainbow Bay and Stewart Mill Bay, the Myriophyllum stands harvested attained maximum standing crops in early September, which decreased slightly by late September (Figures 10, 11). Relative specific growth rates were determined for both Myriophyllum stands according to the equation given by Evans (1972).

Table 20 gives values of R calculated from interpolated points on the biomass curves at intervals throughout the growing season. In both cases maximum R values were obtained between 1-15 July, which coincides with maximum rates of gross community photosynthesis measured by the diel CO_2/O_2 method. Although the slopes of the

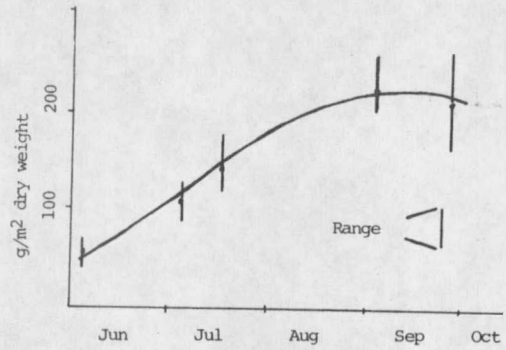


Figure 10. Standing crop (g/m^2 dry weight) of M. exalbescens in Rainbow Bay, 1974.

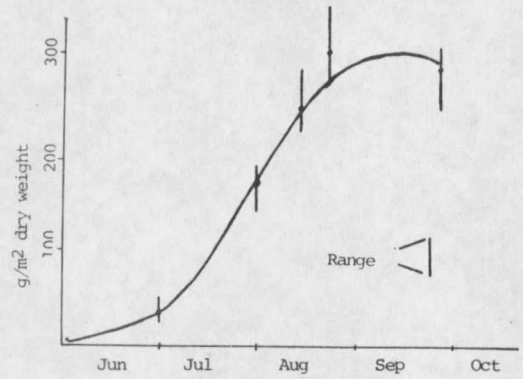


Figure 11. Standing crop (g/m^2 dry weight) of M. exalbescens in Stewart Mill Bay, 1976.

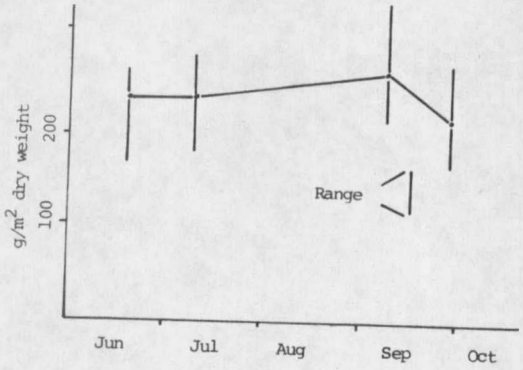


Figure 12. Standing crop (g/m^2 dry weight) of P. richardsonii-Anacharis-Chara community type in Phillipsburg Bay, 1976.

biomass curves remain uniform until late August or early September, indicating that the stands are fixing organic matter at a relatively constant rate, R declines rapidly subsequent to mid-July as biomass accumulates and such inhibiting factors as self-shading, CO_2 limitation, and growth of reproductive structures reduce growth efficiency become significant (Westlake, 1969).

Table 20. Values of R (per unit time as given) for Myriophyllum exalbescens stands in Georgetown Lake.

	R	
	Stewart Mill Bay	Rainbow Bay
01-30 Jun	0.073	0.021
01-15 Jul	0.224	0.025
15-31 Jul	0.049	0.012
01-15 Aug	0.034	0.012
15-31 Aug	0.024	0.006

A key environmental factor differentiating the chemical habitats of Stewart Mill Bay and Rainbow Bay is the input of high conductivity groundwater via Stewart Mill Creek to Stewart Mill Bay. The alkalinity and pH of Stewart Mill Creek is relatively constant at 2.8 and 8.1, respectively, while water in Rainbow Bay in middle and late summer has an alkalinity and pH of less than 2 and greater than 9, respectively, with correspondingly much lower free CO_2 concentrations. Klarich (1977) found that a decrease in the productivity and standing crops of aquatic vascular macrophytes in the Madison River followed a gradient of increasing pH and decreasing alkalinity and free CO_2 concentrations, suggesting limitation by inorganic carbon availability. The possible relationship between higher specific relative growth rate of the Stewart Mill Bay stand

in response to higher free CO_2 is clouded by possible meteorologic influences, as the data were collected in different years, and other possible site-specific influences, such as nitrogen and phosphorus concentrations in the sediment, which were not investigated.

Standing crops of a P. richardsonii, Chara community (Figure 12) demonstrated no clear peak of biomass. Thirty to fifty percent of the total standing crop was composed of Chara. Analyses of the size distribution of individual stems of P. richardsonii, which composed up to 30% of the total biomass in August, indicated that several cohorts of stems are produced annually, beginning in early July and continuing until September, thus the annual net production of the stand may be several times the standing crop at any given time (Wetzel, 1969).

Biomass development, and presumably primary production, of macrophytes appears to vary considerably from year to year. In 1981, biomass accrual in Anacharis and P. richardsonii-Anacharis communities was 2-3 times greater than in other years. In 1982, Anacharis biomass was reduced by almost 70% relative to 1981, being 229 g/m^2 on 27 August, as compared to over 700 g/m^2 the previous year. Observations of Anacharis in previous years (1973-1976) indicates that maximum standing crops are not usually attained until late September or early October. Thus the early development of high standing crops of Anacharis in 1981 is an unusual event. Direct observation and height measurements of other stands indicated that similar reductions occurred in standing crops

of other areas.

Marked differences in meteorologic events differentiated the years 1981 and 1982. In 1982, heavy snow cover in March, coupled with prolonged cool weather, reduced light penetration through the ice cover and delayed ice-out until the last week in May. In 1981, snow cover was light during most of the spring, allowing early growth of macrophytes under the ice, as indicated by the high spring standing crops obtained.

Although it is possible that other factors could have caused the biomass crash of Anacharis, such as disease (Schulthorpe, 1967), it seems most probable that the cause was meteorologic in nature. The density and production of some vascular plant communities in Georgetown Lake appears to fluctuate from year to year in response to meteorologic and hydrologic influences, which can significantly affect depth, transparency, temperature, and dissolved substances.

Nutrient Status of Macrophytes

Nutrient status of vascular plants was evaluated by tissue analysis of nitrogen and phosphorus in mixed plant samples, whole individual plants, and index segments of Anacharis canadensis, Ceratophyllum demersum, and Myriophyllum exalbescens. Gerloff and Krombholz (1966) estimated critical nitrogen concentrations for whole plants as low as 1.3% on a dry weight basis; the comparable phosphorous value was 0.13%. However, native plants from some "infertile" lakes contained from 0.15-0.30% P. Subsequently,

Gerloff (1973) estimated critical concentrations of N and P in index segments of Anacharis occidentalis at 1.4% N and 0.10% P. Critical values for Ceratophyllum demersum were 1.3% for N and 0.10% for P. Myriophyllum spicatum had critical levels of 0.75% N and 0.07% P. Wilson (1972) reported a critical phosphorus concentration of 0.9% for M. exalbescens, which is very similar to that for M. spicatum.

Mixed plant samples from Georgetown Lake contained 0.10-0.50% P (dry weight basis), with an average of 0.25%, and an average of 2.87% N. Index segments of C. demersum contained an average of 0.36% P. Index segments of E. canadensis contained 2.48% N and 0.19% P. P. praelongus contained the highest average P-concentration, 0.45%, and M. exalbescens contained 0.28%. These values indicate that aquatic vascular plants in Georgetown Lake are not nutrient limited by N or P.

COMMUNITY METABOLISM

Data summarizing community primary productivity and respiration as measured by the diel pH-CO_2/O_2 method are presented in Figures 13-15, and Table 21. Community and benthic gross photosynthesis attained maximum values in mid-July, which coincides with the period of maximum average daily incident radiation. Highest rates of gross photosynthesis were observed at Site 2 (Figure 14), which was located in a dense stand of Myriophyllum exalbescens. Average rates at Sites 1 and 3 were very similar,

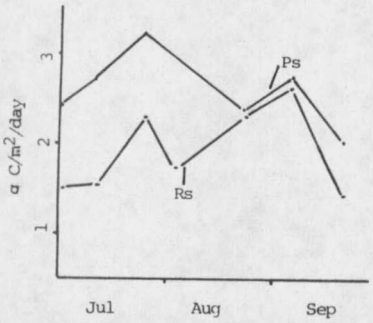


Figure 13. Community metabolism at Diurnal Site 1, 1974.

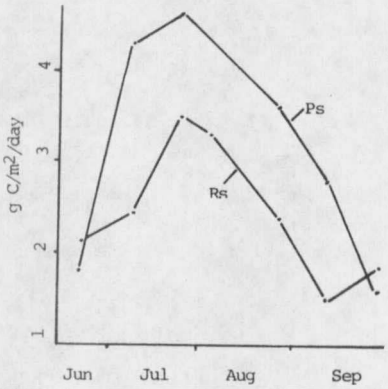


Figure 14. Community metabolism at Diurnal Site 2, 1974.

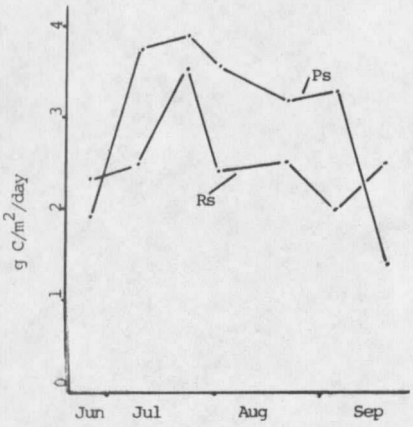


Figure 15. Community metabolism at Diurnal Site 3, 1974.

Table 21. Community metabolism, 27 June-20 September, 1974, Georgetown Lake. RQ, PQ assumed = 1.1 (Odum and Wilson, (1961).

		PG_T Ave	PG_T Max	PG_T Min	PG_P Ave	PG_B Ave	$\frac{PG_B}{PG_T}$	R_T Ave	R_P Ave	R_B Ave	$\frac{R_B}{R_T}$
Site											
1	C	2.87	3.75	1.90	0.21	2.65	0.92	1.98	0.42	1.56	0.79
	O_2	8.42	11.0	5.6	0.63	7.78		5.81	1.22	4.56	
2	C	3.50	4.65	1.65	0.31	3.19	0.91	2.50	0.61	1.89	0.76
	O_2	10.3	13.6	4.84	0.91	9.36		7.33	1.78	5.54	
3	C	3.09	3.95	1.42	0.47	2.62	0.85	2.55	0.92	1.03	0.64
	O_2	9.06	11.6	4.17	1.38	7.68		7.48	2.68	4.78	

PG_T = Total Community Gross Photosynthesis, g C $(O_2)/m^2$/day.

PG_P = Plankton Gross Photosynthesis, corrected for depth. Data from Knight, 1980.

PG_B = Benthic Gross Photosynthesis = $PG_T - PG_P$, g C $(O_2)/m^2$/day.

R_T = Total Community Respiration, g C $(O_2)/m^2$/day.

R_P = Plankton Respiration, g C $(O_2)/m^2$/day.

R_B = Benthic Respiration = $R_T - R_P$, g C $(O)/m^2$/day.

which may indicate that the rates obtained are actually an average for the upper reservoir basin, most of which is occupied by the P. praelongus community type, represented at Site 3.

A general decline in gross community photosynthesis was observed in August and September. This coincided with a decrease in incident solar radiation and free dissolved CO_2. Odum and Wilson (1961) found a linear relationship between total incident radiation and gross community photosynthesis in Texas estuaries. Klarich (1977) reported a decline in net photosynthesis and ratios of net photosynthesis to dark respiration as free dissolved CO_2 concentrations decreased from 7 mg/l to 1 mg/l. In August and September, pH in Georgetown Lake macrophyte beds often attains 9.2-9.5, with accompanying free CO_2 concentrations of <0.05 mg/l. Although most species of vascular aquatic plants are capable of utilizing bicarbonate as an inorganic carbon source, Klarich's results indicate that carbon limitation is a likely factor in reducing community photosynthesis under conditions present in Georgetown Lake in late summer.

It is not possible to derive net values of community productivity from the experimental data because sediment and plant community respiration were not measured separately. However, Klarich (1977) found ratios of net photosynthesis:respiration in the range of 1.5-2.5 for Potamogeton sp. under conditions of low free CO_2, which support estimates of efficiency in converting inorganic carbon to dry matter on the order of 30-50%.

Assuming an average organic carbon content of 0.46 x ash free

dry weight, ash content of 0.20 (Westlake, 1969; Boyd, 1967), a periphyton contribution to total community photosynthesis of 10% (Cattaneo and Kalff, 1980; Sheldon and Boylen, 1975), an estimation of annual macrophytic biomass production as dry weight/m^2 can be made (Table 22). The values obtained assuming a 90-100 day growing season are in reasonable agreement with the maximum standing crops observed in several different macrophyte communities.

Table 22. Estimated annual terminal standing crop (g/m^2 dry weight) of macrophyte communities in Georgetown Lake. Annual biomass accrual would be somewhat less (20-25%) due to sloughing of leaves and lower stems (Wetzel, 1975).

	Assumed Efficiency	Terminal Standing Crop 90 days	100 days
Site 1	30%	175	194
	50%	291	323
Site 2	30%	212	233
	50%	351	390
Site 3	30%	172	193
	50	287	320

The values of community primary productivity and metabolism observed in Georgetown Lake are within the range of productivity and community metabolism of benthic plant communities as measured by Park, et al (1958), Odum and Hoskin (1958), and Copeland and Whitworth (1963), of 7-27 g O_2/m^2/day, using the diurnal method on unenclosed communities. The ratio of community gross primary productivity:phytoplankton gross primary productivity and the negative rates of planktonic net primary production reported by Knight (1980) indicate that benthic macrophytes are the major primary

producers in Georgetown Lake, and are apparently capable of productivity rates sufficient to drive planktonic respiration to the levels reported by Knight (1980) through detrital coupling. Knight reported an average planktonic respiration of 2.91 g O_2/m^2/day in Georgetown Lake during the ice free period. Geer (1977) also found that net zooplankton production in Georgetown Lake was almost equal to net phytoplankton production. Such high efficiencies of energy transfer and incorporation between trophic levels are not possible (Golterman, 1975). The situations described by Geer and by Knight require that benthic communities contribute to substrate for planktonic respiration, which is most likely to occur through cycling of autochthonous benthic detritus (Wetzel, 1975).

EFFECTS OF MACROPHYTES ON AQUATIC CHEMISTRY IN GEORGETOWN LAKE

Seasonal Dynamics of Dissolved Inorganic Carbon and Oxygen

The high rates of carbon uptake by macrophyte communities, observed during diel metabolism measurements, strongly affect pH, alkalinity, and conductivity relationships in Georgetown Lake. Through the summers, pH increased from around 8.3-8.6 in June to 9.1-10 in late August (Tables 23-26), and reached a maximum of around 10.0 at stations located in macrophyte beds (Table 24) which occurred at depths of about 2 m. A decrease of up to 1.5 pH units within 10 cm of the sediment was observed at Station V during diurnal pH measurements in 1981 (Table 25), indicating a sharp stratification of primary zones of carbon fixation by plant photosynthesis and carbon

Table 23. Depth-integrated means of alkalinity (meq/l) and pH at Stations I-IV, 1973-75, 1981-1982.

	I		II		III		IV	
	Alk	pH	Alk	pH	Alk	pH	Alk	pH
10 Jul 73	1.85	8.6	1.82	8.7	1.87	8.8	1.86	8.8
25 Jul 73	1.72	8.8	1.68	9.0	1.73	9.0	1.75	9.2
28 Aug 73	1.62	9.2	1.63	9.1	1.65	9.2	1.64	9.1
12 Sep 73	1.61	8.8	1.64	8.9	1.67	9.0	1.65	9.1
08 Oct 73	1.69	9.2						
21 Jan 74	2.24	7.9						
18 Feb 74	2.41	7.6					2.33	7.6
30 Mar 74	2.72	7.6						
25 Apr 74	2.80	7.5						
05 Jun 74	2.16	8.4	2.14	8.1	2.21	8.0	2.19	8.1
17 Jun 74	2.10	8.2	1.98	8.2	2.16	8.4	2.24	8.3
15 Jul 74	2.08	8.4	2.11	8.1	2.10	8.2	2.15	8.0
29 Jul 74	2.07	8.8	2.03	8.8	1.98	8.6	2.04	8.7
26 Aug 74	1.84	8.9	1.88	8.6	1.88	8.6	1.88	8.6
24 Sep 74	1.69	9.1						
13 Jan 75	2.16	8.1						
23 Feb 75	2.44	7.6						
29 Mar 75	2.53	7.7						
03 May 75	2.69	7.6						
16 Jun 75	2.04	8.0						

Table 23, continued. Depth-integrated mean values for pH-Alkalinity for Georgetown Lake sampling stations. Alkalinity is given in milli-equivalents per liter. E-2, E-3, and E-4 were not sampled after 13 October, 1981.

Station	I		II-IV		V		E (1-4)		E-5		E-6	
	Alk	pH	Alk	pH	Alk	pH	Alk	pH	Alk	pH	Alk	pH
29 Jun 81	1.78	8.4	1.76	8.6	1.69	8.9	1.56	9.0	1.56	9.0		
08 Jul 81	1.72	8.7	1.70	9.0	1.65	9.1	1.90	8.1	1.90	7.9	1.72	8.6
12 Aug 81	1.54	9.1	1.51	9.1	1.48	9.5	2.45	8.3			1.76	7.8
09 Sep 81	1.60	9.2	1.53	9.4	1.46	9.6	2.58	8.0		9.0	1.80	7.8
22 Sep 81	1.54	9.1	1.52	9.1	1.57	9.4	2.51	8.0		8.1	1.80	7.8
13 Oct 81	1.59	9.1	1.58	9.1	1.57	9.2	2.72	7.9	1.72	8.2	1.79	8.3
24 Nov 81			1.70	8.4								
30 Jan 82	2.20	7.8			2.13	7.9	2.55	7.4	2.13	7.9	1.80	7.6
26 Feb 82	2.28	7.7	2.08	7.7	2.23	7.8	2.78	7.7	2.23	7.4	1.80	7.6
26 Mar 82	2.35	7.7	2.13	7.8	2.48	7.7	3.45	7.6	2.48	7.5	2.13	7.8
01 May 82	2.34	7.6	2.05	7.8	2.87	7.4	2.04	7.5	2.87	7.5	2.10	7.7

Table 24. Representative vertical profiles of pH at Station I in Georgetown Lake.

			1973			
Depth (m)	27 Jun	10 Jul	25 Jul	06 Aug	28 Aug	12 Sep
0	8.21	8.79	8.53	9.23	9.21	8.89
2	8.35	8.80	8.92	9.27	9.20	8.91
4	8.39	8.70	9.03	9.22	9.25	8.72
6	8.34	8.58	8.92	9.22	9.25	8.72
8	8.08	7.98	8.74	9.12	9.26	8.52
			1974			
	28 Jun	15 Jul	29 Jul	12 Aug	26 Aug	10 Sep
0	8.41	8.76	9.00	8.49	9.03	9.08
2	8.55	8.42	8.99	8.44	8.92	9.02
4	8.55	8.38	8.95	8.43	8.94	9.02
6	7.80	8.26	8.82	8.43	8.92	9.05
8	7.55	7.95	8.32	8.44	8.64	9.09
			1981			
	03 Jul	17 Jul	23 Jul	03 Aug	27 Aug	18 Sep
0	8.81	9.12	9.11	9.25	9.10	9.49
2	8.78	8.96	9.11	9.25	9.00	9.50
4	8.76	8.99	9.19	9.25	9.00	9.46
6	8.73	8.97	9.21	9.24	9.10	9.49
8	8.36	8.60	9.20	9.25	9.03	9.44
10	8.23	8.25	8.73	9.12	9.00	9.35

Table 25. Representative vertical profiles of pH in macrophyte beds in 1973-1974. Depth = 2 m.

Depth	8 Aug 73	5 Sep 73	26 Jul 74	5 Sep 74
0.0	10.10	9.18	9.65	9.13
1.0	9.85	9.21	9.80	9.21
2.0	9.30	9.25	9.30	9.10

Table 26. Vertical profiles of pH at Station V (littoral macrophyte bed) in summer, 1981.

Depth (m)	3 Jul	17 Jul	3 Aug	27 Aug	2 Sep	18 Sep
0.0	8.85	9.17	9.38	9.50	9.80	9.80
0.5	8.80	9.21	9.47	9.69	9.80	9.80
1.0	8.91	9.20	9.60	9.68	9.84	9.89
1.5	8.75	9.30	9.53	9.68	9.85	9.67
1.8(Bottom)	7.84	7.40	8.57	8.54		

recycling by sediment-related respiration.

Significant differences in pH conditions occurred in 1981 as compared to those observed in 1973 and 1974. Although pH values comparable to those of 1981 were attained in 1973-74 for areas of dense macrophyte growth (Tables 24-25), late summer pH values at Station I increased from 8.9-9.0 in 1974 to around 9.5 in 1981, which indicates an increase in community net primary production.

Alkalinity decreased throughout the summer of all sampling years from around 2 milli-equivalents (meq) per liter in late June to about 1.5-1.6 in late August-early September (Table 23). Coupled with the high pH's observed in late summer, this represents a condition of undersaturation of CO_2 far removed from equilibrium with the atmosphere, which reflects the high rates of net primary productivity and the low rate of invasion of CO_2 from the atmosphere (Stumm and Morgan, 1970). The summer decrease in alkalinity and increase in pH occurred earlier in 1981 than in 1973 or 1974, and to a greater extent, which indicates higher rates of net removal of dissolved inorganic carbon from the water, i. e., greater net community primary production, in 1981.

Due to the death and decomposition of the enclosed macrophytes, which had already attained standing crops of 80-100 gm/m^2 when the enclosures were put in place in late June, 1981, Enclosures 1-4 showed a different pattern of pH, alkalinity, and conductivity changes as compared to unenclosed stations. The pH decreased from 8.6 in June to between 7-8.2 in July and remained near that level. Alkalinity also increased to an average of 2.71 meq per liter, which reflects increased microbial respiration from decomposition of dead macrophytes. This represents more favorable conditions for plant growth in terms of carbon availability due to the accompanying increase in free CO_2 (Ruttner, 1963), and may have been a factor in the tremendously increased phytoplankton standing crop observed in Enclosures 1-4 (Boveng, unpublished data). Concurrently, dissolved oxygen decreased from around 11 mg/l to minimum values of approximately 2 mg/l.

Alkalinity increased at all sampling sites during winter ice cover as community respiration exceeded carbon fixation and released CO_2 into the water (Table 23). In 1981-82, Enclosures 1, 5, and 6, and Station V were sampled throughout the winter. Subsequent to 26 March, alkalinity in E-1 appeared to decrease somewhat, which would indicate an increase in the ratio of net oxygen production:community respiration, although this may have been due to sampling error. Alkalinity in Enclosure 5 continued to increase throughout the winter to a maximum of 2.87 by 1 May. In contrast, Station V, the shallow control, showed the effects of profile-bound density currents, which would tend to dilute the expected increase in alkalinity (Hutchinson,

1941). The inorganic carbon exported from such shallow areas, which amounts to approximately 1.45 mole-equivalents/m^2 (17.4 g inorganic C or 88 g HCO_3^-), could be estimated as the difference between total inorganic carbon at Station V and Enclosure 5.

Dissolved oxygen concentrations remained near or above saturation during the late spring and summer sampling period of 1981 (Table 27) except near the sediment, where moderate decreases of 2-3 mg/l were observed. Dissolved oxygen profiles at Station I comparing late spring-early summer data for 1981 and 1982 are shown in Figure 16. In 1981 dissolved oxygen was 1-2 mg/l higher at Station I than in 1982, being 13 to 17 percent supersaturated near the surface. Although ice-out did not occur until several weeks later in the spring in 1982, the water column at Station I was close to saturation (95-101 percent) by 15 June. According to Hutchinson (1957a), physical processes are sufficient to bring this about within the observed time period (three weeks). The supersaturation observed in 1981 is attributed to unusually heavy macrophytic growth occurring under the ice in late spring.

Dissolved oxygen concentrations at the diurnal sampling Stations 2 and 3, which were located at the three and four meter depth contours in mixed macrophyte communities near Station III, and at Station V, which was located in a dense stand of Anacharis (Table 28), frequently attained levels of oxygen supersaturation of up to 125-150 percent during the ice-free season. Measurements of community metabolism showed diel variations of 2-3 mg/l. At Station V, dissolved oxygen remained high throughout the plant canopy but

Table 27. Depth-integrated mean dissolved oxygen concentrations in mg/l in the water column at stations in Georgetown Lake, 1981-82.

Station	I	II-IV	IV	V	E (1-4)[1]	E-5	E-6
Date							
30 Jun 81	9.8			12.8	11.5		
08 Jul 81	8.5			9.0	5.0		
14 Jul 81	8.5			8.1	2.9		7.9
07 Aug 81				10.4	3.0		
27 Aug 81	8.1			11.3	6.3	6.1	4.9
02 Sep 81	7.4			10.8	6.2	4.9	4.2
17 Sep 81	9.6			12.5	10.7	10.0	5.6
13 Oct 81	8.4	9.6		9.6	10.4	8.8	6.2
24 Nov 81		9.2					
30 Jan 82	3.5			2.1	0.2	1.3	5.8
26 Feb 82	1.6	4.5		3.9	5.5	3.7	4.5
25 Mar 82	1.7	3.4		0.4	1.3	0.4	2.8
01 May 82	1.7	4.6		3.1	2.6	3.2	2.6
15 Jun 82	8.4	8.6					

[1] Average values for 1-4. Enclosures 2,3, and 4 were not sampled after 13 Oct 81.

Table 28. Vertical profiles of dissolved oxygen (mg/l) at littoral sampling sites, 1974-1981.

Depth (m)	09 Jul 74 8:00 p.m.	09 Jul 74 7:00 a. m.	24-25 Jul 74 8:00 p. m.	24-25 Jul 74 7:00 a. m.
Site 3				
0	8.6	8.2	9.3	9.2
1	8.6	8.2	9.3	9.2
2	8.6	8.2	9.2	9.2
3	8.5	7.6	10.0	9.2
4	9.9	7.4	10.3	8.5
	22 Aug 74		19 Sep 74	
0	9.0	8.9	11.2	10.8
1	8.9	8.9	11.1	10.9
2	8.9	8.8	11.1	10.4
3	8.9	8.8	10.9	10.4
4	9.3	7.9	10.9	9.5
	17 Sep 81			
0	10.1	9.7		
1	10.4	9.8		
2	10.4	9.8		
3	11.4	9.7		
4	8.1	6.3		
Station V				
	17 Jul 81		02 Sep 81	
0	7.9	7.6	10.2	9.8
0.5	8.0	8.0	10.1	9.8
1.0	8.1	7.5	10.7	9.8
1.5	8.4	7.6	12.0	9.9
1.8			2.5	0.7
	17 Sep 81			
0	12.8	9.9		
1	12.9	9.9		
1.5	12.2	9.9		
Enclosure 1				
	17 Jul 81		02 Sep 81	
0	6.3	4.0	9.0	6.8
1	6.4	3.9	8.8	6.8
1.5	6.4	2.8	4.6	6.5

sometimes decreased near the sediment-water interface to 1-2 mg/l, indicating the high rates of sediment respiration.

Enclosures 1-4 showed an increase in dissolved oxygen within a week of installation, obtaining maximum concentrations of 164% saturation, but oxygen concentrations declined rapidly after 30 June, decreasing to minimum values on or about 7 July of 2-3 mg/l at the surface and less than 1 mg/l near the sediment. Oxygen concentrations remained depressed for most of August at 4-6 mg/l, then recovered in early September to 8-10 mg/l at the surface as algal populations increased. Diel variations of 2-3 mg/l were observed during community metabolism measurements in September, concentrations being reduced to 6-8 mg/l by early morning. Diel swings of lesser amplitude were observed at Station V, where even in early morning oxygen remained above saturation.

In 1981, dissolved oxygen profiles were determined at Stations II and III on 24 November. Air temperature was -1° C, water temperature was 0.5-1° C. The reservoir was partially frozen over with a thin sheet of ice which extended from the dam to a line connecting Piney Point to the shoreline along Highway 10A just north of the mouth of North Fork, Flint Creek. Oxygen concentration at the surface was 9.1 mg/l and increased slightly with depth to 9.3 mg/l near the bottom. The average concentration was 9.1 mg/l, which represents 82 percent saturation at the surface. The 18 percent saturation deficit is evidence of intense respiratory activity and high biochemical oxygen demand caused by senescing and decaying macrophyte growth, which had maintained high standing crops in some areas into late October.

Severe oxygen deficits developed at all sampling stations during

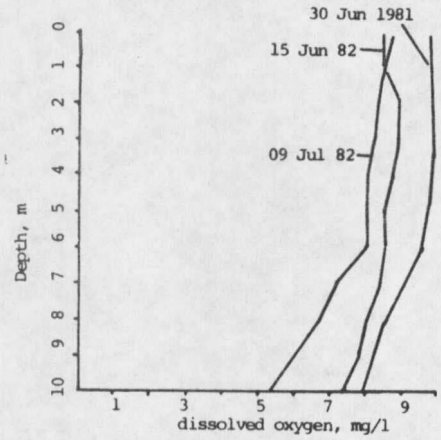


Figure 16. Dissolved oxygen profiles at station I, late spring and early summer.

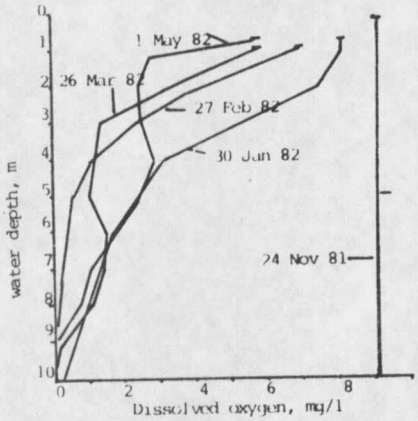


Figure 17. Dissolved oxygen profiles at station I, winter, 1981-1982.

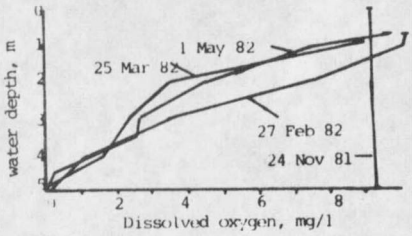


Figure 18. Average dissolved oxygen profiles at stations II-IV, winter, 1981-1982.

winter ice cover (Table 27). In all sampling years, oxygen concentrations continued to decline at Station I through 1 May, although some evidence of recovery was present subsequent to 26 March in 1982 (Figure 17). Deficits at Station I were greater during the winter of 1974-75 than 1981-82, probably due to the lower water level in 1974-75, which resulted in a higher sediment surface/water volume ratio. In 1982, dissolved oxygen concentrations were lower (and thus dissolved oxygen deficits greater) at Station I than at Stations II-IV (Tables 29-30, Figures 17-18). Due to the deep-water withdrawal at the dam, winter dissolved oxygen profiles at Station I are probably atypical as a result of withdrawal-created density currents (Wright, 1967).

Net oxygen consumption (or production) $/m^2$ was estimated by planimetric integration of the dissolved oxygen concentration (as g O_2/m^3) profiles for each site, which yields dissolved oxygen mass /unit area, then subtracting the oxygen mass (g O_2/m^2) obtained on subsequent dates. An increase in mass/unit area on subsequent dates indicates net production of dissolved oxygen; a decrease indicates dissolved oxygen consumption.

Dissolved oxygen profiles in Enclosures 1 and 5 during ice cover are show in Figures 19 and 20. Four independent estimates of net oxygen consumption at these sites , encompasssing the period between 24 November, 1981 and 26 March, 1982 May gave a mean oxygen uptake rate of 0.16 g O_2/m^2/day (s.d. = 0.015; range = 0.138-0.166). Rates varied only slightly between time periods and different enclosures. The same uptake rate (0.16 g O_2/m^2) was obtained for

Table 29. Dissolved oxygen profiles at Station I during ice cover, 1974-1982. Concentration in mg/l.

Depth(m)	21 Jan	18 Feb	30 Mar	25 Apr
		1974		
1	8.2	4.8	4.0	3.9
2	4.9	3.8	0.4	3.0
4	1.4	0.4	0.0	0.0
6	0.3	0.0	0.0	0.0
8	0.0	0.0	0.0	0.0
		1975		
	13 Jan	20 Feb	29 Mar	03 May
1	9.3	8.4	7.1	1.5
2	8.5	4.0	0.9	0.7
4	5.0	1.2	0.3	0.0
6	2.8	0.6	0.0	0.0
8	1.7	0.1	0.0	0.0
		1982		
	30 Jan	27 Feb	25 Mar	01 May
1	8.1	6.9	5.7	2.8
2	7.4	4.0	3.3	2.4
4	3.1	1.1	1.2	2.8
6	1.7	0.4	1.5	1.7
8	1.0	0.2	1.0	0.8
10	0.3	0.0	0.1	0.0

Table 30. Average dissolved oxygen concentrations at stations II-IV during ice cover, 1982. Concentration in mg/l.

Depth(m)	27 Feb	25 Mar	1 May
1	10.1	8.8	7.5
2	7.4	3.6	4.4
3	3.5	2.4	3.7
4	1.2	1.7	1.5
5	0.2	0.1	0.1
6	<0.1	<0.1	<0.1

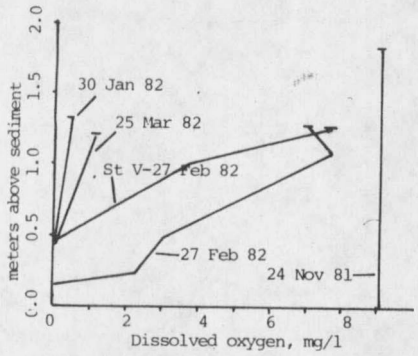


Figure 19. Dissolved oxygen profiles in Enclosure 1, winter, 1981-1982.

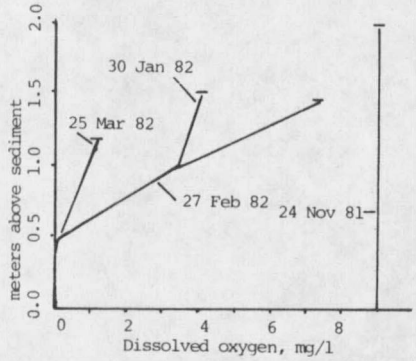


Figure 20. Dissolved oxygen profiles in Enclosure 5, winter, 1981-1982.

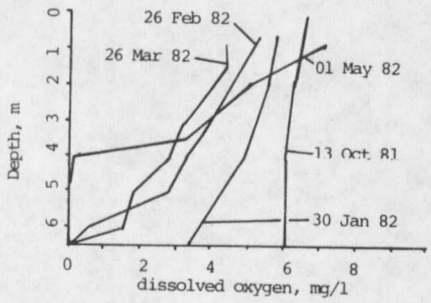


Figure 21. Dissolved oxygen profiles in Enclosure 6, winter, 1981-1982.

Station V from 24 November to 30 January, and a rate of 0.080 g O_2/m^2/day was obtained for Station V from 27 February to 26 March.

At Stations II-IV, calculated net oxygen consumption varied only slightly between early and late winter. Prior to 27 February, net community oxygen consumption was estimated at 0.272 g/m^2/day. Subsequently to 1 May, the consumption rate was estimated at 0.322 g/m^2/day. During the period from 27 February to 26 March, heavy snow accumulations of up to 16 inches blocked much of the incident light, creating conditions for maximum oxygen depletion. The estimated uptake rate of 0.32 g O_2/m^2/day during that period is considered the most reasonable estimate of average oxygen uptake in the upper lake basin. Mathias and Barica (1980) give a mean oxygen depletion rate for shallow, eutrophic prairie lakes of 0.23 g O_2/m^2/day, with maximum observed rates about 0.40 g/m^2/day. The Georgetown Lake basin falls in the upper end of this range, and is clearly eutrophic by this criterion.

From 27 February through 26 March, oxygen consumption in Enclosure 6 was estimated at 0.15 g O_2/m^2/day (Figure 21). During this time, a sharp stratification was established at the 3 meter level, below which oxygen declined abruptly. The uptake rate was calculated based on changes from 3 meters to the bottom. The range of oxygen uptake rates calculated for Enclosure 6 was 0.080 to 0.310 g O_2/m^2/day. The maximum rate was estimated from 24 November through 30 January, and may be an over-estimate.

Shallow littoral areas are usually considered the greatest

oxygen sinks in lake/reservoir basins, due to high sediment oxygen demand, high sediment surface:water volume ratios and consequent frequent turbulent renewal of water at the sediment-water boundary (Hutchinson, 1957a). However, between 30 January and 26 February, 1982, dissolved oxygen increased in Enclosures 1 and 5, by 4.87 g /m^2 (0.174 g/m^2/day) and 2.42 g/m^2 (0.089 g/m^2/day), respectively (Figures 20-21). During the same period Station V showed an increase of dissolved oxygen of 2.45 g/m^2 (0.089 g/m^2/day). This rate can be taken to express net community oxygen production, as the enclosures were sealed at the top and bottom, thus preventing any exchange of dissolved gases, other than through the ice. These rates are higher than any estimates of net production reported for Georgetown Lake by Knight (1980) during the winter of 1974-1975, and show that oxygen metabolism at Station I does not represent whole-lake events.

The average chlorophyll a concentrations at Stations I-IV were 28% of the average chlorophyll a concentrations at Station V and in Enclosures 1 and 5 (Boveng, unpublished data) under ice cover. This indicates the influence of bottom profile, which tends to keep sinking phytoplankton in the euphotic zone, and nutrients released from the sediment on growth and standing crop of phytoplankton populations under the ice (Harris, 1978). The high phytoplankton standing crop (Boveng, unpublished data) and absence of macrophytes at these stations indicate that algae, either planktonic or benthic, are responsible for the oxygen production.

The apparent low uptake rates observed for Enclosures 1 and 5

may be a result of oxygen limitation of sediment-related metabolism, proposed by Mathias and Barica (1980), or the result of significant oxygen production in these areas, as was observed between 30 January and 26 February. The same reasoning may be applied to Station V, although mass transfer by density currents and profile-bound currents (Hutchinson, 1941) are also likely explanations.

It should not be inferred that these values represent actual respiration rates in littoral areas, or that respiration in shallow littoral areas is less than in deeper areas; however, the data do demonstrate that under certain conditions the littoral zone may act as an oxygen source rather than a sink during ice-cover. In this case the oxygen production observed was certainly by algae, as no standing macrophytes were present before or after ice cover.

Oxygen data collected over a period of several years by the Montana Department of Fish, Wildlife, and Parks (unpublished data) show the same pattern, with minima occurring in middle to late March, and recovery by early April. The rates obtained at Station IV indicate the possibility of winterkill of fish due to oxygen stress should prolonged heavy snow cover and cloudy weather occur during late March and April under a drawdown condition, which would probably tend to cause an increase in the rate of net consumption. Obviously, the maintenance of positive oxygen concentrations is dependent on periods of thawing and clear weather, which allow sufficient light penetration for positive net photosynthesis, and oxygen inputs from ground water, of which several major areas are known. One is located near the pumphouse, where a large spring enters (with a dissolved

oxygen concentration of about 7-8 mg/l). Another similar spring is located in the Denton's Point Marina. These two areas provide refuges for oxygen-stressed fish during critical period of oxygen depletion during extremely heavy snow cover. Other smaller spring inputs are probably present throughout the reservoir basin.

If changes in pH and alkalinity are used to estimate respiratory metabolism, Enclosure 5 exhibited a net carbon metabolism of 0.111 g $C/m^2/day$ from 30 January through 26 March. If aerobic metabolism were responsible for oxidation of this organic material, the oxygen consumption rate would be 0.296 g $O_2/m^2/day$, which is approximately equal to that obtained for Stations II-IV. The same calculation for Enclosure 1 gives an estimate of net carbon respiration of 0.238 g $C/m^2/day$, equivalent to 0.762 g $O^2/m^2/day$. Net carbon oxidation for Enclosure 6, estimated from changes in alkalinity, averaged 0.246 g $C/m^2/day$, equivalent to 0.656 g $O_2/m^2/day$. Theses ratios of oxygen uptake to carbon mineralization indicate a large component of anaerobic metabolism in regeneration of inorganic carbon under the low oxygen tensions present during ice cover.

A significant question to be answered during the 1981-82 study concerned macrophytic oxygen production during winter. Knight, et al (1975), calculated a hypothetical oxygen budget based on light/-dark bottle measurements of planktonic metabolism. Those measurements ranged from -0.01 to -5.56 g $O_2/m^2/day$, with a mean of -1.51. This estimate was subsequently changed (Knight, 1980) to a mean value of -0.85 g $O_2/m^2/day$. The latter rate is approximately 8 times the average winter plankton respiration rate for eutro-

phic lakes given by Mathias and Barica (1980), and 2.5 times the average observed rate of depletion in the upper lake basin measured during winter, 1981-82. To explain this discrepancy, Knight invoked oxygen production by macrophytes, which he proposed was sufficient to provide the difference. Presumably the major oxygen producer would be Potamogeton praelongus, which is a perennial and may be seen green and apparently viable under the ice cover. Anacharis may also be responsible for significant winter oxygen production, as considerable standing crops accumulate during late spring of some years, such as 1981, in which standing crops of Anacharis were 50-80 g/m^2 in early June.

To determine if such rates of oxygen production were present, several diurnal measurements of community metabolism of unenclosed communities were made through the ice in April, 1982. Two areas were sampled; one near the Grassy Point access site at the 4 meter contour, a location known to have dense stands of Potamogeton praelongus. The other sampling site was near Red Bridge access, at the 7 meter contour, which is known to be free of macrophytes and was assumed to be isolated from macrophyte effects over 24 hours. Measurements at the Red Bridge site were confined to the upper 4 meters of the water column, to minimize sediment influence and bias due to additional respiration in the water column below the mean depth of the reservoir. Clear days were selected for measurements to maximize potential photosynthesis. At this time, there was no snow cover on top of the ice, which was approximately 0.7 m thick. Between 25 and 29 April, net community oxygen production at the Grassy Point Site

was -1.13 g O_2/m^2/day. Community oxygen consumption was estimated at 2.46 g O_2/m^2/day. Gross daytime oxygen production was 1.33 g/m^2/day.

At the Red Bridge Site, gross oxygen production was 2.34 g O_2/m^2/day. Community oxygen uptake was 0.37 g O_2/m^2/day. Net oxygen production was 1.94 g O_2/m^2/day. The plankton respiration rate was equivalent to approximately 0.012 g O_2/m^3/day, which agrees closely with the value of 0.01 g O_2/m^2/day given for average winter planktonic respiration rates in eutrophic prairie lakes by Mathias and Barica (1980). Applying this value to community respiration at the Grassy Point Site gives a benthic respiration estimate of 2.46-0.30 = 2.16 g O_2/m^2/day. Benthic respiration at that particular time accounted for 88% of the total community respiration measured at the Grassy Point Site.

One set of measurements does not prove that macrophytes are not the major oxygen producers. However, the diunal data agree with the observed oxygen profiles, which indicate that the benthic macrophyte community is essentially a winter oxygen sink, and not a major oxygen source to upper layers. In situ measurements of the photosynthetic potential of macrophytes under ice cover (Boylen and Sheldon, 1976) indicate that potential oxygen production is about 10% of summer rates per unit biomass. They also found that substantial reductions in biomass occurred during ice-cover. Considering the potential oxygen production observed, their results provide additional evidence that the rates of oxygen production attributed to macrophytes in Georgetown Lake by Knight, et al (1975) are not

likely to exist.

Further evidence of underestimation of oxygen production by phytoplankton during the winter by Knight (1980) is found in the chlorophyll distribution during winter, 1981-82, when Stations I-IV had an average chlorophyll a concentration equal to 28% of that observed at Station V and Enclosures 1 and 5 (Boveng, unpublished data).

In view of the observed increases of oxygen in the littoral zone at Station V, the rates of oxygen consumption in macrophyte communities near Grassy Point in April, and other evidence presented, the author must conclude that the macrophyte communities, although producing some oxygen during winter ice-cover, are not the major source of oxygen input during winter stagnation, and that a well planned harvesting program which eliminated up to 50% of the major macrophytic vegetation would not cause a deterioration of the winter dissolved oxygen conditions in Georgetown Lake.

Dynamics of Phosphorus in Georgetown Lake

Nutrient chemistry in Georgetown Lake during 1973-75 has been discussed by Knight, et al (1975), Knight (1980), Garrison (1976), and Foris, (1976). The aquatic chemistry of phosphorus is complex and involves several solubility and equilibrium constants. Suggested solubility-control mechanisms include conversion of inorganic phosphate and calcite to hydroxyapatite and subsequent precipitation, particularly at high pH's, as are often encountered in productive fresh waters (Halstead and Tash, 1982). At a pH of 9.0 in

the presence of 2 meq/l Ca^{2+} (conditions approximated in Georgetown Lake during early summer), the limit of inorganic phosphate solubility is given as approximately 10 ug/l P; at a pH of 10.0, inorganic phosphate solubility is given as approximately 1 ug/l P (Golterman, 1982).

Inorganic phosphorus concentrations in Georgetown Lake during the ice-free season (Table 30) are within the range consistent with control due to calcium hydroxyapatite solubility. The maximum inorganic phosphorus concentrations observed at unenclosed sampling stations were 10 ug P/l at Station I, 24 September, 1974, which represents a condition of saturation relative to hydroxyapatite solubility at a pH of approximately 9.1 and alkalinity 1.7 meq/l, 9 ug P/l at Station I on 17 June, 1974, and at Stations III and IV on 15 July, 1974 (undersaturated at pH 8.2), and 9 ug P/l at Station V on 25 August, 1981, which is slightly oversaturated with respect to hydroxyapatite solubility (pH 9.4, alkalinity 1.5 meq/l).

Particulate and dissolved organic phosphorus comprised over 90% of the average total phosphorus concentration during the summer of 1981. During the summer of 1981, dissolved organic phosphorus (equivalent to dissolved unreactive phosphorus) comprised 55% of the total phosphorus through October sampling, averaging about 10 ug/l at Stations I-III and 12 ug/l at Station IV. Bieleski (1973) has suggested that some algae and aquatic vascular plants exhibit a significant leakage of organic phosphorus, while able to take up inorganic phosphorus to very low levels, less than 0.1 uM. The equilibrium concentration of organic phosphorus was indicated to be

Table 31. Average phosphorus concentrations at sampling stations in Georgetown Lake. Concentration in ug/l (mg/m^3.) I = inorganic P; D = total dissolved phosphorus; T = total phosphorus; P = particulate phosphorus (total less total dissolved).

Station	I				II				III				IV			
	I	D	T	P	I	D	T	P	I	D	T	P	I	D	T	P
Date																
25 Jul 73	0		20		0		20		0		25		0		27	
06 Aug 73	0		41		0		25		0		16		0		24	
28 Aug 73	5		36		4		36		3		37		4		50	
12 Sep 73	0		30		0		27		0		27		0		31	
08 Oct 73	1		20													
21 Jan 74	4		24													
18 Feb 74	18		30												27	
25 Apr 74	33		55													
05 Jun 74	5		44		4		32		4		35		3		28	
17 Jun 74	9		38		8		40		7		36		8		38	
01 Jul 74	0		21		2		16		3		15		4		24	
15 Jul 74	1		38		1		22		9		36		8		30	
29 Jul 74	3		26		4		23		3		23		2		30	
12 Aug 74	5		26													
26 Aug 74	2		28		2		25		2		31		3		31	
10 Sep 74	4		25													
24 Sep 74	10		33													
13 Jan 75	6		24													
23 Feb 75	12		35													
28 Mar 75	21		44													
03 May 75	28		63													
26 May 75	34		67													
16 Jun 75	3		50													

Table 31, continued.

Station	I				II				III				IV			
	I	D	T	P	I	D	T	P	I	D	T	P	I	D	T	P
Date																
17 Jun 81	0		20		0		20		0		21		0		34	
30 Jun 81	0	13	20	07	0	12	18	06	0	12	26	14	0	12	33	21
15 Jul 81	0	13	24	11	0	11	26	15	1	13	21	8	0	14	24	10
28 Jul 81	1	14	17	2	0	14	18	4	0	14	14	0	3	21	29	5
12 Aug 81	1	13	19	5	3	9	14	2	3	10	17	4	5	12	20	3
25 Aug 81	1	5	17	11	2	8	21	11	1	4	14	10	2	3	20	15
09 Sep 81	0	8	16	8	0	9	10	1	0	7	13	6	0	12	22	10
13 Oct 81	3	4	19	12	3	5	17	11	3	5	19	11	3	5	34	26
24 Nov 81					4	6	17	7	3	5	18	10				
30 Jan 82	5	3	14	6												
27 Feb 82	4	9	19	6	0	10	14	4	0	10	17	7	0	11	17	6
26 Mar 82	5	4	15	6	2	4	13	7	0	6	12	6	2	4	20	14
01 May 82	2	7	16	7	0	8	15	7	0	7	16	9	0	8	17	9

approximately 0.3 uM, which would be about 8-10 ug/l. The concentration of dissolved organic phosphorus, which is near this value in Georgetown Lake, may be due to phosphorus leakage from macrophytes or algae, and represent an equilibrium condition for exchange of organic phosphorus between plant cells and the free water.

Particulate phosphorus averaged 7.4 ug/l at Stations I-III, and 13.1 at Station IV, comprising 39% of the average total phosphorus. Average total phosphorus fluctuated around 20 ug/l throughout the summer of 1981 at Stations I-IV. From 17 June through 15 July increases in total phosphorus at Stations I and II were balanced by decreases at Stations III and IV, indicating a relatively steady state of phosphorus mass in the water column outside the shallow littoral areas. Subsequent to 15 July, particulate and total phosphorus declined after 29 July, which represented a net loss of approximately 10 ug/l, or 66 mg/m^2.

In 1981, total phosphorus concentrations were slightly higher at Station V, which is located in a macrophyte stand with a total depth of 2 m, than at Stations I-IV, which are deeper and less strongly influenced by the water volume:sediment ratio. Most of the increase observed at Station V occurred as particulate phosphorus, which indicates the source of the material is the sediment or senescing macrophytes (Lie, 1979; Carpenter, 1980).

Although total phosphorus concentrations during the summer of 1981 were higher at Station V than at Stations I-IV, particularly subsequent to 4 August, the phosphorus mass per unit of sediment area

was often greater at Station I because of the greater water volume:unit area ratio. On 12 August, Station V had 112 mg P/m^2; Station I, 190. Station II had 84 mg/m^2, and Station III, 102. Enclosure 6, which was located near the dam on the 6.5 meter contour in an area uncolonized by macrophytes, attained a total phosphorus mass of 132 mg/m^2 (average concentration 22 ug/l) on the same date. Thus it does not appear necessary to postulate export of phosphorus from shallower littoral areas of heavy macrophyte growth to account for phosphorus supplies to areas in Georgetown Lake which are not subject to macrophyte colonization. This suggests a normalizing process (sedimentation) which would tend to equilibrate chemical conditions in surficial sediments. Garrison (1976) found some evidence of this in his analysis of Georgetown Lake sediments. In addition Enclosure 6 exhibited average concentrations of dissolved phosphorus equal to or greater than those observed at Stations I-V, indicating that the concentration of dissolved organic phosphorus may be controlled by influences other than benthic macrophytes.

Total phosphorus concentrations at Stations I-IV were up to 50 % lower in 1981 relative to comparable dates in 1973 and 1974. Approximately 6-10 % of the decrease can be accounted for by dilution due to the higher water level in 1981. Another possible factor is an increased flushing rate in 1981 due to higher spring runoff. However, water yields for a comparable period in 1981-82 were only 19% higher than in 1973-74, thus it does not appear that flushing due to spring runoff is directly responsible.

Internal loading has been well established as a factor in

maintaining dissolved phosphorus concentrations in the absence of external sources. Lean and Charlton (1977) documented internal phosphorus loading rates of up 18 mg P/m^2/day under oxygenated conditions. Stauffer (1981) also presented data relative to internal phosphorus loading under oxygenated conditions.

Vollenweider (1968, 1976), Golterman (1980, 1982) and others have discussed the relationship between external phosphorus loading, phosphorus concentrations in receiving waters, and sedimentation. The quantitative expression (Golterman, 1980) relating phosphorus balance is simple due to the absence of a gas phase:

$$dP_w = P_{in} - P_{sed} - P_{out} + P_{reg}$$

P_{in} = P input

P_{out} = P output

P_{sed} = P sedimented out in receiving water

P_{reg} = P regenerated from sediments

dP_w = change in P concentration in receiving water

All units expressed as mass/unit area or mass/unit volume.

The two components of internal cycling, P_{sed} and P_{reg}, representing sedimentation and regeneration, respectively, combined give a net rate of phosphorus exchange between water and sediment. Thus it is not possible to calculate actual rates of nutrient (in this case, phosphorus) loss and gain relative to bottom sediments from loading and concentration data. However, net rates can be calculated from positive (loading) or negative (depletion) changes in

phosphorus concentrations in the receiving water, which in both cases will be conservative due to the fact that both sedimentation and regeneration processes are occurring simultaneously.

In summers of 1973, 1974, and 1981, significant increases in total phosphorus concentration occurred which cannot be explained by calculated external loading rates (Table 39, Nutrient Budget section). Table 32 presents a sample calculation of net internal phosphorus loading in terms of P_{out}, P_{in}, and dP_w.

Table 32. Calculations of internal phosphorus loading. Hydrological loading based on average P concentration in inflow of 18 ug/l; evaporation rates given in Table 2; outflow = 0.85 m^3/s; P concentration in outflow given in Table 14. Non-point and atmospheric loading rates calculated from loading coefficients given in Nutrient Budget section, corrected for fractional year. Further details for calculation of hydrologic loading given in Reckhow (1979).

06 Aug-28 Aug, 1973 = 22 days.

Stations II-IV: average dP_w = 19.3 mg/m^3
x 4.3 m mean depth

83 mg P/m^2 net loading

$P_{in} = P_{hydrological} + P_{non\text{-}point} + P_{atmospheric}$

$P_{hydrological}$ = 0.184 m^3/m^2 (evaporation/m^2 + surface elev.)
x 18 mg/m^3

3.3 mg/m^2

$P_{non\text{-}point} + P_{atmospheric}$ = 2 mg/m^2

P_{in} = 5.3 mg/m^2

P_{out} = 6.6 mg/m^2 (outflow)

$-P_{sed} + P_{reg}$ = 83 -5.3 (+ 6.6) = 84.3 mg P/m^2

Positive Net internal loading = 84.3 mg P/m^2/22 days
= 3.83 mg P/m^2/day

Similar calculations for Stations I-IV from 25 July-28 August, 1974, yield a net internal loading rate of 74 mg P/m^2, and for 09 September-13 October, 1981, 30 mg P/m^2. Actual rates would be higher due to the sedimentation component. It appears that internal loading is sufficient to account for higher phosphorus concentrations observed in 1973-74.

Enclosures 1-4, in which most macrophyte growth (99%) died during the summer, exhibited large increases in phosphorus through late September, and obtained a maximum average of 224 ug/l on September 9, 1981, most of which was dissolved organic or particulate phosphorus. This was 5-10 times greater than other Stations, and is attributable to decay of macrophytic vegetation, representing a net release of 400 mg P/m^2 of lake bottom. Assuming an approximate standing crop of 100 g macrophyte biomass, mostly Anacharis, which has a mean phosphorus concentration of 0.30 % in samples from Georgetown Lake, macrophyte decomposition could have accounted for 300 mg/m^2 of the phosphorus released in the enclosures.

Subsequent to 22 September, total phosphorus in the enclosures declined rapidly from 209 ug/l to 61 ug/l, representing a loss from the water column of approximately 280 mg/m^2. This coincides with a temperature decrease from 11.5° to 6° C. Stauffer (1981) has suggested that temperature strongly affects the regeneration of phosphorus from sediments under oxic conditions by influencing invertebrate activity as well as microbial metabolism.

Enclosure 5, the "windowed" macrophyte enclosure, was removed in July for the installation of clear vinyl inserts to allow more light

into the interior. After it was replaced, phosphorus concentrations increased slightly above those observed at Station V, but the difference in means was not significant in an unpaired t-test. Particulate and dissolved phosphorus were the major components of total phosphorus at Station V and Enclosure 5, with inorganic phosphorus ranging from 0-14 ug/l. The non-significant difference in phosphorus in Enclosure 5 and Station V leads to a conclusion that phosphorus released in protected littoral areas during the summer growing season is not exported to other parts of the reservoir. This may not represent actual conditions for other more exposed littoral areas of high macrophyte growth, which are subjected to considerably more turbulence than the experimental site. All forms of phosphorus were present at Station V and Enclosure 5 at higher concentrations than at Stations I-IV from 12 August through 13 October, 1981. The rise in phosphorus at Station V and Enclosure 5 as macrophyte growth and temperatures increased seems to indicate mobilization of phosphorus from the littoral zone, which surely occurs in shallow macrophyte beds exposed to strong wave turbulence. The distribution of sediments, of which the greatest accumulations were found in protected areas and least were found in exposed areas, also supports this conclusion.

Through the winter of 1981-82, in contrast to the winters of 1973-74 and 1974-75, the average total phosphorus concentrations at Stations I-IV decreased slightly, to a mean of 16 ug/l on 1 May, 1982. At Station I in 1982, the only site at which stratified samples were collected, concentrations very near the sediment increased sharply. Both particulate and dissolved fractions

decreased during winter, while inorganic phosphorus increased only slightly at Station I (range 2-5 ug/l), and remained very low in depth-integrated samples at Stations II-IV (range 0-2 ug/l).

At Station V total phosphorus decreased in early winter to levels observed at Stations I-IV, then steadily increased throughout the winter to a maximum of 43 ug/l on 30 April. This compares to 46 ug/l in Enclosure 1 and 47 ug/l in Enclosure 5 on the same date. Similar trends were observed for total phosphorus in the two shallow enclosures sampled through the winter, as concentrations in both decreased in early winter, then increased through 30 April. The highest concentrations of inorganic phosphorus were observed in Enclosure 1 (3-14 ug/l) and Enclosure 6 (2-6 ug/l). Preconditioning of the sediment during the previous summer may have had some influence on the subsequent release of inorganic phosphorus, as it remained very low at Station V and Enclosure 5, neither of which were enclosed through the summer. Enclosing the sediment and subjecting the system to reduced oxygen tensions may have affected the sediment- water interface, facilitating release of inorganic phosphorus, although the contained waters were well oxygenated on 13 October. The similarity of concentrations in the three shallow sites on 30 April suggests an equilibrium between water and sediment, but may be an artifact of sampling. However, the similarity between concentrations in Enclosure 6 and Stations I-IV also indicates the possibility of a steady state relationship which tends to normalize fluctuations.

Subsequent to 30 April, Enclosure 5 was not sampled, as it was

damaged during the ice break-up. Enclosure 1 was sampled 15 June, 1982, at which time total phosphorus had increased to 74 ug/l. Water temperature had increased to 12 degrees C, dissolved oxygen was 10.5 mg/l to a depth of 1.5 meters. Inorganic phosphorus was 2 ug/l, comprising 3 % of the total. These results support recycling of organic phosphorus by invertebrate activity under oxic conditions.

Aquatic Chemistry of Nitrogen

Inorganic nitrogen concentrations during the summer of 1981 were similar to those found in 1973-75, with the exception of EPA samples collected during the summer of 1975 (20 ug/l nitrate N). Nitrate (Table 33) remained undetectable through most of the summer in 1981, a maximum concentration of 4 ug/l being observed at Stations I-IV on 29 June. In the experimental enclosures, nitrate was undetectable in all but 2 samples, the maximum concentrations observed being 4 ug/l on 29 June, 1981. Subsequent to 15 July, 1981 nitrate was not detected in any samples from the lake until after freeze-up. The same general pattern occurred in 1974, with the exception of a slight pulse of nitrate on 12 August, when the average concentration at Station I was 4-5 ug/l. In 1973, however, nitrate increased in samples from Station I to an average concentration of 6-8 ug/l on 28 August, and it remained detectable at Station I at concentrations ranging from 7-13 ug/l distributed throughout the water column until December.

During the winter of 1981-82, nitrate increased at Station I to

Table 33. Average nitrate concentrations at stations in Georgetown Lake during 1973-75 and 1981-82. Concentrations in ug/l N.

	I	II	III	IV	V	E(1-4)	E-5	E-6
10 Jul 73	0	1	1	1				
25 Jul 73	1	1	1	0				
06 Aug 73	1	1	3	3				
28 Aug 73	9	1	0	4				
12 Sep 73	12	1	3	3				
08 Oct 73	6							
21 Jan 74	0							
18 Feb 74	7							
25 Apr 74	15			4				
05 Jun 74	5	5	5	5				
17 Jun 74	0	0	0	0				
01 Jul 74	3	0	0	0				
15 Jul 74	1	0	0	0				
29 Jul 74	0	0	0	0				
12 Aug 74	5							
28 Aug 74	1	0	1	1				
10 Sep 74	0							
24 Sep 74	1							
13 Jan 75	15							
23 Feb 75	31							
28 Mar 75	18							
03 May 75	22							
26 May 75	12							
16 Jun 75	10							

Table 33, continued.

	I	II	III	IV	V	E(1-4)	E-5	E-6
17 Jun 81	0	0	0	0	0			
30 Jun 81	4	3	4	4	4	4		
15 Jul 81	2	2	2	2	0	0		2
28 Jul 81	0	0	0	0				0
12 Aug 81	0	0	0	0	0	0	0	0
25 Aug 81	0	0	0	0	0	0	0	0
09 Sep 81	0	0	0	0	0	0	0	0
22 Sep 81	1	0	0	4	0	0	0	0
13 Oct 81	0	0	0	0	0	0	0	0
09 Nov 81		0	0					
30 Jan 82	5				0	0	0	1
27 Feb 82	4	1	1	1	1	4	4	3
26 Mar 82	9	10	4	3	5	4	3	3
01 May 82	31	7		4	1	6	0	6
15 Jun 82	2	2	2	2	2	2	2	2

an average concentration of 31 ug/l at Station I on 30 April, 1982. At this time, the highest concentrations occurred just under the ice, which led to the conclusion that its origin was from snow and ice melt. Nitrate concentrations during the winter were higher at Station I than at Stations II-IV, which obtained a maximum concentration of 10 ug/l at Station II on 25 March. In Enclosures 1, 5 and 6, nitrate remained low throughout the winter (less than 5 ug/l). Nitrate at Station V also remained very low, obtaining a maximum concentration estimated at 5 ug/l on 26 March.

The low concentrations obtained for nitrate in lake samples, compared with concentrations obtained for Stewart Mill Creek of up to 100 ug/l and samples obtained for small shoreline springs up to 170 ug/l, raises the possibility that some plants in the reservoir may be obligate nitrate users, or that the aquatic flora in general may be nitrate-starved. Other forms of inorganic nitrogen (ammonia, nitrite) were also reduced to low concentrations during June and July, 1981. Algal assays performed by EPA on Georgetown Lake water collected in July and September, 1975, indicated nitrogen limitation (EPA, 1976).

Ammonia nitrogen concentrations are shown in Table 34. Patterns of ammonia distribution at Station I in 1981 were very similar to those observed in 1974, concentrations gradually increasing from a June minimum to maximum concentrations in September. In 1981, Station IV obtained the highest ammonia concentration of the deeper Stations, 41 ug/l in September. Ammonia concentrations increased rapidly in Enclosures 1-4 subsequent to macrophyte die-off, reaching

Table 34. Average ammonia-N concentrations at sampling stations in Georgetown Lake. Concentration in ug/l N.

Station	I	II-III	IV
Date			
10 Jul 73	6	4	6
25 Jul 73	12	3	3
06 Aug 73	5	6	5
28 Aug 73	34	0	0
12 Sep 73	5	23	2
08 Oct 73	5		
21 Jan 74	29		
18 Feb 74	109		50
05 Jun 74	0	1	1
17 Jun 74	0	0	0
01 Jul 74	2	1	3
15 Jul 74	7	6	11
29 Jul 74	6	5	5
12 Aug 74	8		
26 Aug 74	8	6	8
10 Sep 74	8		
24 Sep 74	15		
13 Jan 75	33		
23 Feb 75	29		
29 Mar 75	87		
03 May 75	91		
16 Jun 75	1		

Table 34, continued.

Station	I	II-III	IV	V	E (1-4)	E-5	E-6
Date							
17 Jun 81	0	2	0	0			
29 Jun 81	2	6	11	0			
15 Jul 81	6	4	2	0	1		2
28 Jul 81	3	2	3	1	2		10
04 Aug 81	6	8	7	8	26		13
25 Aug 81	0	1	2	14	3	13	5
09 Sep 81	14	5	10	25	260	405	16
22 Sep 81	9	6	41	9	99	17	21
24 Nov 81		6					
30 Jan 82	94			6	174	25	42
27 Feb 82	92	37	38	0	64	15	46
25 Mar 82	110	38	30	26	198	20	69
30 Apr 82	81	61	26	172	232	318	81

a maximum of 260 ug/l (494 mg/m^2) on 9 September. Ammonia concentrations then rapidly decreased in Enclosures 1-4, being 99 ug/l (188 mg/m^2) on 22 September.

In Enclosure 5, ammonia concentrations increased from 13 ug/l (24.7 mg/m^2) to 405 ug/l (770 mg/m^2) between 25 August and 9 September. At the same time, Station V showed an increase in ammonia from 14 to 25 ug/l (48mg/m^2), and Enclosure 6, an increase to 21 ug/l (137 mg/m^2). From results in Enclosure 5, it appears that bioactivity in littoral communities is a major source of ammonia to the plankton, and may be a key factor in triggering the observed late summer algal pulse.

During ice cover in 1981-82, ammonia concentrations increased through the winter at Stations I-III, attaining a maximum of 110 ug/l at Station I on 26 March. Again, concentrations were higher at Station I than at Stations II-IV, and increased with depth, the highest concentrations being observed just above the sediment. In Enclosures 1 and 5, ammonia increased through April to maximum concentrations of 232 and 318 ug/l (190 and 260 mg/m^2), respectively. Enclosure 6 showed a steady increase in ammonia through April, to a maximum concentration of 81 ug/l (467 mg/m^2). It appears the potential release of ammonia is as great in areas not colonized by macrophytes as in the littoral macrophyte zone, and that ammonia release is limited by other factors, most likely the presence of oxygen (Mortimer, 1971), which was absent from the deeper strata of Enclosure 6 during late winter, while primary production of oxygen in

Enclosures 1 and 5 was considerable, and oxygen was measurable to within 0.1 meters of the bottom.

Two main internal sources of nitrogen can be identified from the observed patterns of occurrence and distribution: 1) release from the sediments during winter stagnation under anoxic conditions, and 2) release from littoral sediments during late summer due to invertebrate bio-activity, bacterial metabolism, and macrophyte senescence. The strong pulse of ammonia observed at Enclosure 5 and Station V in September both suggest the significance of the latter source to late summer algal production. External sources readily identified include groundwater, in which concentrations run up to at least 100 ug/l, and precipitation, which adds a significant mass of nitrate at ice melt in the spring.

SEDIMENT ACCUMULATION AND CHEMICAL RELATIONSHIPS

The location of sediment core sampling sites is shown in Fig 2. The vertical profile of average sediment density is given in Table 35. The mean total depth of sediment cores obtained was 24 cm (range 10 -35 cm). Most cores showed a definite vertical physical stratification, or profile, within which could be distinguished features which defined the extent of deposited lake sediments. The presence of coarse gravel, the predominance of clay, and/or the absence of animal debris, such as snail shells, were taken as evidence that the origin of the material below the occurrence of such characteristics was not due to deposition of aquatic sediments. Based on these criteria, the average depth of aquatic sediment deposition was calculated as 20 cm (n = 20, range = 10-32 cm), or 7.8 inches. The deepest sediment deposits occurred in protected areas, such as Phillipsburg Bay, Stewart Mill Bay, upper Jericho Bay, and in deeper areas near the dam. The shallowest deposits occurred in areas exposed to maximum turbulence, such as Rainbow Point, the area off Highway 10A, and areas off Stewart Mill Bay. Intermediate sediment depths occurred in protected areas too deep for macrophyte growth, such as Red Bridge.

Table 35. Sediment mass-density, dry weight gm/cm^3.

Sediment Depth	N	Mean	Range
0-5 cm	5	0.42 g/cm^3	0.17-0.58
5-10 cm	5	0.74 g/cm^3	0.49-1.06
10+ cm	5	0.79 g/cm^3	0.59-0.91

Taking the approximate age of the reservoir to be 80 years (1901-1982), the average overall rate of deposition is about 0.25 cm (= 0.10 inches)/year, equivalent to about 1050 gms dry matter /m^2/year. The maximum observed rate was estimated to be about 0.40 cm (= 0.16 inches)/year, representing the deposition of approximately 1680 gms of dry matter per m^2 per year. The minimum rate calculated was about 0.13 cm (= 0.05 inches)/year. By comparison, Prentki (1979) calculated the rate of sediment deposition in Myriophyllum spicatum beds of Lake Wingra to be about 0.7 cm/year, representing the deposition of over 1600 grams dry weight per square meter per year of plant and animal detritus and mineral deposits. This is directly comparable to the value for deposition of dry matter in high rate deposition sites for Georgetown Lake, although the rate of vertical accretion is less in Georgetown, possibly due to a greater compaction of the surface sediments. Prentki also obtained similar values for the deposition of dry matter throughout the remainder of Lake Wingra, approximately 1200 gms dry weight/m^2 /year. Stefan and Hanson (1980) reported sediment accrual rates of 0.12 cm/yr in Hall Lake, Minnesota, as determined by ^{14}C dating, and Dunst (1980) reported sediment accrual rates in a small eutrophic lake in Wisconsin to be about 0.40 cm/year.

Based on the distribution of core depths obtained, sedimentation rates in deeper macrophyte beds are approximately equal to those in shallower areas, apparently due to export of sedimented organic detritus. Rich and Wetzel (1978) found a similar situation and postulated that wind-driven turbulence was responsible for redistribution

of organic sediments.

Values of total sediment phosphorus for Georgetown Lake are compared to sediment phosphorus and other nutrient parameters for some other eutrophic lakes from which data are available in Table 36. The average total phosphorus concentration obtained in surficial sediments in 1981 was 571 ug/g, which is within the range obtained by Garrison (1976) for surface sediment material collected from five sites around the reservoir during 1973-74.

The mean total phosphorus concentration in surficial sediments from Georgetown Lake (calculated from data of Garrison, 1976) is 950 ug/g. In Garrison's data, variations in total sediment phosphorus appear to be randomly distributed on a seasonal basis. Sampling error is a likely cause of the lack of any consistent relationship between sediment phosphorus and dissolved/suspended phosphorus.

The average total phosphorus concentrations obtained in surficial sediment from Georgetown Lake (950 ug/g, 1974; 571 ug/g, 1981) is within the range of sediment total phosphorus concentrations from other shallow, eutrophic lakes (range 575-1231 ug/g) (Wildung and Schmidt, 1973). Sediments from Lake Mendota, which is somewhat deeper than other lakes considered, contained 1950 ug P/g dry sediment, and other deeper eutrophic lakes have had values reported of over 5000 ug P/g (Stauffer, 1981).

Sediment phosphorus in Shagawa Lake showed no statistically significant spatial variation, indicating some normalizing processes in sediment phosphorus distribution (Wildung and Schmidt, 1973). This is in spite of a point source effluent from Ely, Minnesota via

Table 36. Comparisons of sediment chemistry for Georgetown and other eutrophic lakes.

Lake		$\%C_T$	C/P	N_T	C/N	P_T	N/P	P_I	P_I/P_T
Georgetown[1]									
1974	St. 1	12.6	412	0.90	16.3	789	25.3	522	66%
	St. 2	16.6	373	1.35	14.4	1150	26.0	731	64%
	St. 3	15.0	413	1.21	14.5	939	28.5	555	59%
	St. 4	27.6	599	2.12	15.2	1190	39.4	646	54%
	St. 5	9.1	346	0.75	11.1	680	24.4	414	61%
Average		16.1	438	1.27	14.9	950	29.6	571	60%
Georgetown 1981	Average					571		363	
Shagawa Lake[3]		12.2	256	1.20	11.9	1231	21.6	691	57%
Upper Klamath Lake[3]	St. 1	7.4	185	0.88	9.8	1033	18.9	574	56%
	St. 2	6.6	271	1.01	7.6	629	35.6	295	47%
	St. 3	6.5	254	0.95	8.0	662	31.8	349	53%
Average						775	28.9		
Erie[3] (grand mean)						865		651	75
Agency[3]	"					575		328	57
Diamond[3]	"					659		252	38
Mendota[2]						1950			

Data from Garrison (1976)[1], Stauffer (1981)[2], Wildung and Schmidt (1973)[3]. Total carbon (C_T) and total nitrogen (N_T) are expressed as percent dry weight of sediment. Phosphorus (P) is expressed as ug/g dry weight of sediment. P_T = Total phosphorus; P_I = inorganic phosphorus.

the Burntside River. Highest total phosphorus values were obtained in deep "holes" near the sewage outfall, reflecting the possible influence of turbulence and sedimentation in influencing the redistribution of phosphorus. A trend of increasing phosphorus with increasing water depth in upper Klamath Lake was explained through agricultural runoff, although total loading from cultural sources was

estimated at only 25 percent. Upper Klamath Lake was described as a shallow, naturally eutrophic lake, subject to periodic dense algal blooms, which in some cases reduced Secchi disc depths to <0.5 M. Phosphorus leachates from Upper Klamath Lake were fully active in promoting algal growth when compared to inorganic phosphorus (Wildung and Schmidt, 1973).

Total phosphorus concentrations in sediment cores from Georgetown Lake showed a sharp decrease with increasing depth (Table 37).

Table 37. Vertical distribution of total phosphorus in sediment cores from Georgetown Lake. Sampling locations shown in Figure 2.

Site	Phillipsburg Bay	Jericho Bay	Red Bridge	Pump House
Depth (cm)		Concentration (ug/g dry sediment)		
0	670	925	615	335
5	560	700	475	285
10	420	440	328	232
15	290	350	230	275
20	200	329	180	265
25		370	230	205
30		260		160
Depth (cm)		Mass density (ug P/cm^3 sediment)		
0	370	400	230	135
5	360	365	230	150
10	325	320	230	175
15	223	265	170	215
20	170	255	140	215
25		290	139	165
30		220		125

However, the lower density of surface sediment alters the apparent concentration gradient. "Mass-density" profiles for phosphorus shown in Table 37 are very similar to those obtained for unfertilized Montana rangeland (Table 38) by Wight and Black (1977).

Concentrations of phosphorus (ug/gm dry sediment) for Georgetown Lake sediments and surface layers of sedimentary rangeland soils are also similar (950 and 775 ug/gm respectively).

Table 38. Concentrations of total phosphorus (ug/g soil dry weight) in unfertilized sedimentary Montana rangeland soil (from Wight and Black, 1977).

Depth in Soil Profile (cm)	P Concentration ug/g dry weight
0-3	780
5-8	750
9-15	700
16-21	560
23-30	500

The phosphorus profile for unfertilized Montana range soil indicates that natural biogeochemical processes are sufficient to create a substantial phosphorus gradient in terrestrial soils. There are several natural processes which would tend to create the observed gradients in Georgetown Lake sediments. It has been shown that rooted aquatic vascular plants are capable of extracting most of their required phosphorus from sediments at depths up to 20 cm (Carignan, 1982; Barko, et al, 1982). The uptake of phosphorus from deeper sediments and subsequent redeposition in autochthonous detritus, which has been shown to be accumulating at a slow rate in Georgetown Lake (present data and Garrison, 1976), would tend to cause a gradual depletion of phosphorus in deeper sediments, which is indicated by the apparent low phosphorus concentrations at sediment depths of 20-30 cm, and enrichment at the surface. The

average phosphorus concentration in surface sediments, which is 0.05-0.1%, is consistent with the average phosphorus content of aquatic macrophytes (0.25-0.3%), when other components of autochthonous sediments, primarily silicates, carbonates, and organic detritus from plankton, are considered. Mobilization of deeper sedimentary phosphorus supplies by roots of aquatic vascular plants, which were observed in the deepest layers of sediment cores from Georgetown Lake and which other researchers have observed to depths of 50 cm and more (Reimold, 1974), and subsequent deposition on the surface provides a mechanism for concentration near the surface layers.

Several investigators have reported an increase of phosphorus in aquatic sediments due to increased external phosphorus loading (Wagner, 1972; Golterman 1980; Larsen and Mercier, 1976). Golterman (1980) has described a relationship between external loading and phosphorus sedimentation rates as follows:

$$A = k\ (P_w)^v, \text{ where}$$

A = P concentration in sediment
P_w = P concentration in lake water
k = approximately 0.62, a constant
v = approximately 0.34, a constant

Combining this equation with a form of the phosphorus balance equation, given by Golterman (1980) as

$$P_{in} = P_{out} + P_{sed} + dP_w$$

P_{in} = P input
P_{out} = P output
P_{sed} = P sedimented out in receiving water
dP_w = change in P concentration in receiving water

the resulting equation is obtained:

$$P_{in} = P_{out} + A\ K\ (P)^v + dP_w\ z \quad (z = \text{mean depth})$$

The values for k and v were calculated by Golterman from experimental data on natural sediments. This relationship predicts that phosphorus content in sediment will approximately double as phosphorus concentrations in the overlying water increase by a factor of 10 in the range of 1-500 ug P/l. Data of Wagner (1972), which showed a 2.2-fold increase of sediment phosphorus content in response to a 10-fold increase in phosphorus concentrations in the overlying water, verified the relative accuracy of the predicted response.

Using experimental values obtained for the vertical distribution of phosphorus and rate of sediment deposition in sediments of Georgetown Lake, Golterman's equation predicts that the concentration of phosphorus in Georgetown Lake water would be increasing at a rate of about 10-fold every 40 years to cause the observed four-fold increase in phosphorus near the sediment surface. This requires the assumptions of a background phosphorus concentration of 225-250 ug P/gm dry sediment at a depth of 20 cm below the sediment surface, a sediment phosphorus concentration of 950 ug/gm dry sediment at the surface (approximated by value obtained by Garrison, 1976), and an average annual sedimentation rate of 0.25 cm/year over the life of the reservoir (approximately 80 years).

Total phosphorus concentrations in the reservoir, inflowing streams, and outflows do not support the hypothesis that the increase in sediment phosphorus content nearer the sediment surface is due to increases in phosphorus sedimentation from external sources. There does not appear to be a trend toward increasing external phosphorus loading, or an increase in net phosphorus throughput (see Nutrient

Budgets). Total phosphorus concentrations in 1973-75 and 1981 indicate a relatively stable or decreasing average phosphorus concentration in the reservoir during summer.

By integrating the phosphorus mass density for the top 20 cm of sediment, a rough approximation of the total phosphorus supply in the lake sediments can be estimated, although it must be realized that the entire mass does not represent phosphorus which is readily exchangeable with water or available to rooted aquatic plants for growth (Stauffer, 1981). The mean phosphorus mass per centimeter square of sediment surface was calculated at 5120 ug, which is equal to 51.2 g per square meter and 619,500 kg for the entire reservoir. At the estimated loss rate of 35 mg $P/m^2/yr$ (see Nutrient Budget), the sediment supply in the top 20 cm of sediment and soil represents approximately 1500 times this value. Apparently the phosphorus mass in the sediment is acting as a stabilizing buffer in the aquatic system, tending to hold dissolved phosphorus near the same levels from year to year in spite of a negative phosphorus budget (see Nutrient Budgets). Both the estimated total mass present and loss rates should be considered minimum values, as the data from which they were derived were the most conservative available.

Yoshida (1981) presented evidence based on laboratory experiments and theoretical considerations that phosphorus release from sediments asymptotically approached a maximum value at a sediment depth of about 30 cm. His release rates were about 75% of maximum at a total sediment depth of 20 cm. Based on this evidence, internal phosphorus loading in Georgetown Lake (due to direct

exchange between water and sediment) should be at or approaching maximum values and would not be expected to increase without additional external inputs.

Garrison (1976) estimated atomic N:P ratios in the sediment to be 31:1. If this ratio holds throughout the sediment profile, the total nitrogen mass present in the shallow sediments is estimated at 1.37×10^6 kg. Nitrogen in aquatic systems is much more labile than phosphorus (Brezonik, 1972), due to the various processes of transformation. Nitrogen:phosphorus ratios in the water might be expected to fluctuate seasonally and annually according the specific local or environmental conditions which favored one process or the other. It does not seem that N:P ratios in Georgetown Lake sediments provide an accurate indication of relative availability of nutrients to the biosphere, due to the interference of specific chemical processes which affect chemical species present in the free water. N:P ratios in nutrient budgets and sediments predict that phosphorus should be in short supply relative to nitrogen, yet N:P ratios during part of the growing season (late June-early August) and algal bioassay (EPA, 1976) show nitrogen to be apparently limiting, although removal of the nitrogen limitation by addition of inorganic nitrogen resulted in phosphorus limitation at a slightly higher standing crop. Late summer increases of dissolved ammonia in Enclosure 5 and at Station V indicate the potential for internal loading of inorganic nitrogen and the significance of internal nutrient sources in stimulating late-summer algal growth.

NUTRIENT BUDGETS

Nutrient budgets for nitrogen and phosphorus were calculated for 17 June 1981-31 May 1982. Inputs and outputs from surface streams were based on the product of gaged stream discharge and analytical concentrations of all forms of the nutrient under consideration for a given unit of time. In cases of missing data on nutrient concentrations, values were used from U. S. Forest Service data or data obtained under similar flow conditions. Concentrations of total nitrogen and phosphorus in surface streams are given in Table 14.

Analyses of groundwater samples from shoreline seeps and submerged springs indicated that the chemical constitution of Stewart Mill Creek approximates that of other groundwater sources in the reservoir (Knight, 1980; Garrett, 1983). Therefore, flow-weighted averages of total nitrogen and phosphorus in Stewart Mill Creek were used to approximate groundwater inputs.

Non-point contributions were based on nutrient export and runoff coefficients given in Reckhow, et al, (1980), and McElroy, et al, (1976), for land use and land cover types similar to those present in the drainage. Only direct surface runoff was considered in the non-point estimate, as contributions from septic tanks, groundwater, and atmospheric sources were included separately. It was assumed that direct surface runoff was contributed from a radius 0.5 km around the reservoir, the area of which is approximately 1,360 ha.

From data given in Table Ala of Reckhow, et al (1980) a

linear regression for phosphorus export from western coniferous forest was calculated, resulting in the following equation:

$$P \text{ (export) kg/ha/yr} = 0.3699 \ln \text{runoff (cm)} - 0.9035.$$
$$r = 0.8997 \quad n = 4$$

Due to the small number of data points, it is difficult to attach statistical significance to this equation. However, from data of Frederickson (1979), with a runoff of 76 cm/yr a phosphorus export of 0.698 kg/ha/yr was calculated, which agrees well with his experimental value of 0.680 kg/ha/yr. Precipitation at Georgetown Lake is approximately 50 cm/yr, of which 30% was assumed to run off (Reckhow, et al, 1980), giving a total runoff of 15 cm/unit area. Substituting this into the regression equation, a total phosphorus yield of approximately 0.1 kg/ha/yr was obtained. Less than 10% of the total runoff occurs as surface runoff (McElroy, et al, 1976). However, for study purposes, a conservative value of 25% was assumed. Therefore, the export of phosphorus in direct surface runoff was estimated as 0.25 x 0.1 kg/ha/yr = 0.025 kg/ha/yr. This value was also used for native grasslands and residential areas, most of which have been left in a natural state. Data from Reckhow, et al, (1980) indicate that this is a conservative estimate for grasslands, which have low ratios of runoff:precipitation and correspondingly low phosphorus yields. Thus the total non-point phosphorus load was calculated as 0.025 kg P/ha/yr x 1360 ha = 34 kg P/yr.

A similar calculation for total nitrogen export gave the equation

$$N \text{ (export) kg/ha/yr} = 3.404 \ln \text{runoff (cm)} - 4.7748$$
$$r = .879 \quad n = 4$$

Substituting a value of 15 cm total runoff, total nitrogen export was estimated at 4.4 kg/ha/yr, and that due to surface runoff at 1.1 kg/ha/yr. This gives a calculated non-point nitrogen load of 1.1 x 1,360 = 1496 kg N/yr.

Inputs from septic tanks were based on export coefficients given by EPA (1975). The values given for annual export coefficients for nitrogen and phosphorus (0.1134 kg P/capita/yr and 4.263 kg N/capita/yr) were modified by residential use data obtained during the septic leachate survey accomplished in 1981. Results of the residential survey indicated that the average residence was used at a rate equivalent to 0.7 per capita years, i. e., occupancy by one person for 0.7 years/calendar year. Thus the average load per residence was estimated at 0.08 kg P and 2.98 kg N per year x 125 near-shore residences equals 10 kg P and 373 kg N/year.

Loading due to campground facilities was calculated on the same basis. With a total capacity of approximately 235 spaces, and assuming 70% occupancy by an average of 3 persons for 4 months a year, the potential loadings were calculated as 18.5 kg P and 427 kg N per year. Commercial facilities (restaurants, bars) were assumed to contribute approximately the same as campgrounds (18.5 kg P and 427 kg N per year).

Fish harvest data were provided by Jim Vashro, Montana Department of Fish, Wildlife, and Parks fisheries biologist. Nitrogen and phosphorus concentrations in fish flesh were taken from Neel, et al, (1973) as 2.4% N and 0.50% P on a wet weight basis. Fish harvest is composed primarily of Kokanee salmon, which reproduce

naturally in the reservoir, and rainbow trout, of which about 250,000-350,000 are planted annually as fry or fingerlings. The biomass and phosphorus content of the planted fish is thus negligible in terms of N and P input. Approximately 125,000 rainbow trout (average weight 0.27 kg) and 175,000 Kokanee (ave. weight 0.16 kg) are harvested annually, representing a net output of approximately 310 kg P and 1550 kg N per year. This constitutes 25 % and 12% of the phosphorus and nitrogen outputs, respectively.

Comparisons of the phosphorus and nitrogen budgets calculated for 1981-82 and previous nutrient budgets (Knight, et al, 1975; EPA, 1976) are shown in Tables 39 and 40, respectively. The budget of Knight, et al, (1975), has been recalculated to include estimates of atmospheric and non-point inputs (septic tanks, direct runoff, groundwater). The EPA budgets lump hydrologic sources other than the North Fork of Flint Creek and direct surface runoff into non-point sources.

In comparing phosphorus budgets, the most significant aspect is that all result in a negative phosphorus loading relationship, which indicates that the phosphorus status of Georgetown Lake is relatively stable and more than likely the reservoir is losing phosphorus annually. Also significant is the small contribution calculated for septic tanks, which have been repeatedly implicated in cases of cultural eutrophication, but which recent work (EPA and Wapora, Inc., 1981) has shown to be much less significant than previously thought. Considering the phosphorus mass available in the sediment and the potential for internal loading to the biological community via

Table 39. Comparison of phosphorus budgets for Georgetown Lake. Inputs and outputs in Kilograms per year. Net balance in kilograms/lake.

INPUTS	Knight, et al, 1974-75	EPA 1975	Garrett 1981-82
NFFC	262	95	175
Stewart Mill Ck	198	(*1)	101
Groundwater	200 (*3)	200-380	200
Non-point	34 (*3)	435	34
Hardtla Creek	103	(*1)	30
Septic tanks	47 (*3)	55	47
Atmospheric	218 (*3)	195	224
Total	1062	980-1160	811
OUTPUTS			
Fish Yield	250-350	250-350	250-350
Flint Creek	1060	730	920
Anaconda aqueduct	621	500	-0-(*2)
Total	1931-2031	1480-1580	1170-1270
NET BALANCE			
Kg/lake/yr	-869>X>-969	-320>X>-600	-359>X>-459
$mg/m^2/yr$	-72>X>-80	-37>X>-76	-30>X>-38

*1 Included in non-point sources.
*2 Personal communication, Ron Eccleston, Anaconda Company
*3 Calculated from 1981-1982 data.

macrophytes and detritus cycling, it is unlikely that foreseeable increases in phosphorus loading from cultural sources will cause significant increases in total primary productivity.

Differences in nitrogen budgets are larger, for which there may be several reasons. The high input estimate for nitrogen by the EPA is due to their non-point source component (45.4%), which, considering the evidence of nitrogen/phosphorus ratios in water and sediment, would appear to be an overestimate. It is noteworthy that the atmospheric component is the next largest source of nitrogen, and is not subject to management. EPA estimates of inputs of nitrogen

and phosphorus are at a ratio of approximately 70:1, while N:P ratios in the water range from 10:1 to a maximum of about 50:1, that observed during late winter (Knight, 1980). N:P ratios in the sediment (31:1, Garrison, 1976) also run much lower than one would expect if such a high input ratio were real.

The high nitrogen output estimated by EPA may be the result of a sampling artifact. It was found during 1981 that samples from the dam outflow collected several hundred yards below its discharge from the power station results in high values of nitrate, probably from groundwater contamination. Increases from 50-75 ug/l were observed on several occasions. Additionally, total nitrogen in reservoir water was shown to have been considerably higher in 1974-1976, possibly due to nitrogen fixation by phytoplankton or enhanced release of ammonia and organic nitrogen from benthic sediments. It appears that the nitrogen budget is much less stable than phosphorus and may fluctuate considerably from year to year.

The average nitrogen content of macrophytic vegetation in Georgetown Lake is around 2.8%. Assuming a standing crop of 250 g dry weight/m^2 over 60 % of the reservoir surface results in approximately 50,000 kg of nitrogen being tied up in macrophytic vegetation. This amounts to approximately twice the estimated annual nitrogen input for 1981-1982, and reflects the importance of internal nutrient supplies to aquatic plant growth. The calculation is hypothetical, but represents easily obtainable conditions in the reservoir, and corresponds with the very low dissolved inorganic nitrogen concentrations found during the summer of 1981 and 1982.

TABLE 40. Comparison of nitrogen budgets for Georgetown Lake. Units for input and output and net balance are kg/lake/year.

INPUTS	KNIGHT, et al, 1974-75	EPA/NES 1975-76	GARRETT, 1981-82
NFFC	2330	3920	2558
SMC	2130	(*1)	1804
GROUNDWATER	3450 (*2)	(*1)	3450
NON-POINT	1496 (*2)	17960	1496
HARDTLA CK	1730	(*1)	350
SEPTIC SYSTEMS	1227 (*2)	2125	1227
ATMOSPHERIC	12090 (*2)	12090	13888
TOTAL	24453	36095	24773
OUTPUTS			
FISH YIELD	1550	1550	1550
FLINT CK	11200	22290	11426
AQUEDUCT	6650	15255	-0-(*3)
TOTAL	19400	39095	12976
BALANCE			
Kg/lake/yr	5053	-3000	11797
$gm/m^2/yr$	0.42	-0.25	0.97

*1 Included in nonpoint estimate.
*2 Calculated from 1981-1982 data.
*3 Personal communication, Ron Eccleston, Anaconda Company.

Considering the relative magnitude of cultural sources, it does not appear that any significant changes in the current nitrogen balance can be effected by external management. Major changes in land use patterns, such as a shift to agricultural uses of the basin, or introduction of a point source effluent of considerable size would be required to significantly alter the nitrogen budget.

According to the Vollenweider loading model (Reckhow, 1979),

Georgetown Lake is oligotrophic based on phosphorus loading, and in the upper end of the safe range relative to nitrogen loading. A nitrogen loading rate of about 1 gm/m^2/year was calculated for 1981-82, which is right at the limit of "safe" loading, but well below the criteria of 2 $g/m^2/yr$ for "dangerous" loading (definitions by Reckhow, 1979). Considering the variability in nitrogen output, demonstrated by the estimate for 1974-75 compared to 1981-1982, it appears that net nitrogen loading can vary considerably from year to year, and 1981-82 may represent a maximum loading situation.

The average concentrations of inorganic and total phosphorus in the discharge were considerably lowere in 1981-1982 than in 1973-1974 (20 and 31 ug/l, respectively). A possible cause is found in the relationship between reservoir elevation, dissolved oxygen and phosphorus in the discharge, which is withdrawn approximately 1.5 m above the sediment, during the winter. Representative elevations, dates, and maximum concentrations of inorganic and total phosphorus at Station I and in the outflow are given in Table 41. Linear correlation coefficients for surface elevation against phosphorus in the outflow, maximum phosphorus concentration near the bottom at station I are highly significant (Table 42). The relationship apears to be mediated by dissolved oxygen concentrations, as comparison of dissolved oxygen data of Knight (1980) indicate that deficits were much more severe in 1973-74 than in 1981-82.

Other factors may affect phosphorus concentrations in the outflow. Snow cover enhances the development of oxygen deficit, thus

Table 41. Relationship between water surface elevation and phosphorus (ug/l at deepest sampling depth) at station I and the outflow during winter stagnation.

Date	Elev	site	Inorg-P	Total-P
18 Feb 74	25.6	I	54	54
		Outflow	62	72
25 Apr 74	25.2	I	114	134
		Outflow	72	100
23 Apr 75	26.4	I	57	107
		Outflow	51	72
25 Mar 82	28.4	I	18	39
		Outflow	3	25
1 May 82	28.5	I	13	30
		Outflow	2	20

Table 42. Correlation coefficients for inorganic and total phosphorus at Station I and the outflow against water level during winter stagnation of Georgetown Lake.

	Inorg P	Total P
Station I	-0.894*	-0.767
Outflow	-0.767	-0.975

N = 5; P(.05) = 0.811; P(.02) = 0.882

heavy snow cover will encourage phosphorus release. Total phosphorus release from the sediment is a time-sequenced event, thus one would expect greater phosphorus concentrations later in winter stagnation, but recent data do not indicate that this is necessarily the case, depending the further development of the dissolved oxygen deficit. Mathias and Barica (1979) found mean depth to be highly significant in influencing the total oxygen deficit. It appears that phosphorus in Georgetown Lake may be amenable to manipulation by water level control within the limits necessary for fish survival.

Nitrogen does not appear to show the same relationship. Inorganic nitrogen concentrations were higher near the sediment-water interface in 1981-82, possibly because the sampling device allowed sampling closer to the substrate.

SUMMARY AND CONCLUSIONS

Groundwater contributes an average of about 67% of the total water yield of the Georgetown Lake basin. Annual fluctuations in water yield are primarily due to differences in surface runoff. Reservoir operation has been modified in recent years, resulting in a reduction of fluctuations in water levels.

Most of the drainage remains covered by natural vegetation. Areas from which timber has been harvested are well removed from the reservoir and do not appear to be impacting water quality or esthetics. The area developed for recreation and residential use is relatively small, although localized around the reservoir, and no measurable impacts on water quality could be detected. Water quality in the surface streams remains high, and no evidence was found of deterioration over the past 8 years (1973-1981).

Macrophyte communities dominate community metabolism and strongly influence nutrient cycles and dynamics of other dissolved substances, such as oxygen and carbon. The *Potamogeton praelongus* community type has undergone extensive change from 1975 to 1981. Canopy coverage of *P. praelongus* has increased 100% in several areas and up to 700% in some. Standing crops of macrophytes attain maximum levels in late summer or early fall, and vary from year to year due to the influence of meteorologic conditions.

Nitrogen and phosphorus do not appear to be limiting macrophyte growth; however, inorganic carbon was implicated in limitation of macrophyte photosynthesis. Due to the high measured rates of benthic photosynthesis, most of which was attributed to vascular plants, and negative rates observed for the planktonic community (Knight, 1980), it was concluded that benthic primary production is coupled to zooplankton production through the cycling of autocthonous benthic detritus.

Dissolved oxygen concentrations often reach supersaturation during late spring and summer due to high rates of photosynthesis by macrophyte communities. Diel measurements of dissolved oxygen did not indicate significant sag in dissolved oxygen concentrations due to dark respiration of benthic communities, although concentrations did, on occasion, show drastic decreases very near the sediment-water interface.

All stations showed severe oxygen depletion under ice-cover. Significant oxygen production was measured at littoral sites, which appear to contribute a net input of dissolved oxygen to the aquatic community under favorable weather conditions. Macrophytes were not found to be a source of dissolved oxygen during most of the winter. The benthic community accounted for 89% of the dissolved oxygen uptake during diurnal measurements of community oxygen metabolism under the ice. Algae were determined to be the source of photosynthetic oxygen input to the reservoir during most of the winter.

Evidence was found for nutrient limitation of phytoplankton by pH regulation of inorganic phosphorus solubility due to macrophyte photosynthesis. Dissolved organic phosphorus was found to comprise a substantial fraction (40-50%) of the total suspended phosphorus, while dissolved inorganic phosphorus frequently was reduced to very low or undetectable levels. Macrophytes were also implicated in the depletion of dissolved inorganic nitrogen. Results of the enclosure experiments showed that benthic communities are capable of releasing large quantities of nitrogen and phosphorus to the water and stimulating algal growth. Rates of internal phosphorus loading up to approximately 3 mg phosphorus/m^2/day were calculated for the reservoir.

Although phytoplankton are evidently limited by dissolved nutrients in Georgetown Lake during some years, it is not relevant to discuss limitation of primary production by nutrients. The productivity rates attained by macrophytes indicate that Georgetown Lake is a eutrophic basin. The reservoir will express its productive potential either through macrophyte or algal growth, depending to some extent on meterological and hydrological events, and to some extent on management, which could modify the productivity toward phytoplankton by artificially recycling nutrients in macrophytes through harvesting and in-lake composting.

Sediment accumulations were similar to those observed in other eutrophic lakes and reservoirs, but were not indicative of excessive external loading with nutrients or silt. The distribution of phos-

phorus observed in benthic sediments was attributed to uptake and subsequent deposition by rooted vascular plants. Sediment phosphorus and nitrogen concentrations appear sufficient to maintain the reservoir in its present state of productivity for the foreseeable future without any changes in external loading due to management of cultural or natural sources.

Comparisons of nutrient budgets for nitrogen and phosphorus prepared by several investigators indicate that phosphorus dynamics of the reservoir are in a relatively stable state, with a net negative balance. The nitrogen balance is positive but within acceptable limits for external loading. Cultural sources of nutrients appear to be relatively insignificant, in comparison to natural external and internal sources.

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