LIMNOLOGY OF FLATHEAD LAKE: FINAL REPORT

by

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Table of Contents

INTRODUCTION

Flathead Lake is one of the largest freshwater lakes in the western United States and has been a topographic focal point since early exploration of the northern Rocky Mountains. In spite of the fact that limnological investigations were initiated near the turn of the century (Forbes 1893), almost no quantitative data were available to permit an evaluation of the lake's trophic status or of changes that may have eventuated since settlement of the drainage basin. Previous investigators have consistently referred to the lake as being oligotrophic (Potter and Stanford 1975; Gaufin et al. 1976) with a phytoplankton community dominated by many species of diatoms (Moghadam 1969; Morgan 1968, 1971).

However, preliminary studies in the early 1970's suggested that complicated mechanisms, including phosphorus stripping by riverine-derived, clay sediments, may be in control of primary autotrophic pathways. Stanford and Potter (1976) proposed that flocculent aggregations of seston were caused by the inflow and subsequent settling of clay turbidity during and after spring freshet by the lake's primary tributary, the Flathead River. They further suggested that primary productivity in the lake was limited by a paucity of total phosphorus and that this limitation was maintained primarily by absorptive stripping of phosphorus via the flocculent aggregates as they settled to the lake bottom.

Similar hypotheses have been forthcoming from other investigators working on reservoirs receiving substantial sediment loads (Canfield and Bachmann 1981). Phosphorus was thought to be maintained in the lake sediments by an antagonistic redox gradient supported by continuous high concentrations of dissolved oxygen in the water column. These hypotheses were based on data which indicated standing crops of net plankton were drastically curtailed upon the onset of spring turbidity, coupled with the observation that water clarity maximized after turbidity settled through the water column (Stanford and Potter 1976). However, during the present study an alternate hypothesis was derived after Ellis and Stanford (1982) showed that ultra- and nannoplankton dominated photolithotrophy in Flathead Lake and Perry and Stanford (1982) showed that bottom sediments from the lake would stimulate substantial algal growth in cultures. The alternate hypothesis suggested that advective circulation of riverine turbidity should cause a fertilization effect, if the very small (<10um) microbes were somehow able to assimilate phosphorus and other nutrients bound to the sediment particles.

In order to test the "stripping via flocculation" hypothesis, we embarked upon a long-term study that emphasized accurate quantification of nutrient loading and associated responses in the phytoplankton community. Our approach involved mass balance measurements of lake phosphorus, nitrogen, and carbon dynamics and utilized primary productivity measurements as the primary biotic response variable. Our main objective was to determine how the autotrophic community responded to sediment inflow over a 6-year period. We utilized a variety of analytical and simulation models in an effort to relate biotic productivity to phosphorus and nitrogen loading. The study provided a long-term ecological data set with which to evaluate the various phosphorus loading models currently in vogue in limnological literature. These models were derived from rather tenuous data sets involving many lakes, as opposed to long term studies of a particular lake (Vollenweider and Kerekes 1980).

A secondary objective was to finally establish a limnological baseline for one of the most important lakes in North America. The Flathead River basin is undergoing dramatic cultural development in response to increasing demand for resources in the Flathead Basin: timber, oil and gas, coal, and recreation. Unfortunately, Flathead Lake is located downstream from all of these activities and impacts of mismanagement upstream will manifest in the lake. A basic understanding of river-lake ecosystem structure and function was needed in order to provide management alternatives to protect the heralded pristine quality of Flathead Lake and its renowned sport fishery.

STUDY AREA

Drainage Basin Morphometry and Discharge

Flathead Lake and its drainage basin comprise 18,379 km^2 in northwestern Montana and southeastern British Columbia

(Figure 1). The Flathead River, as it exits Flathead Lake, is a sixth order tributary of the Columbia River. Its tribu-

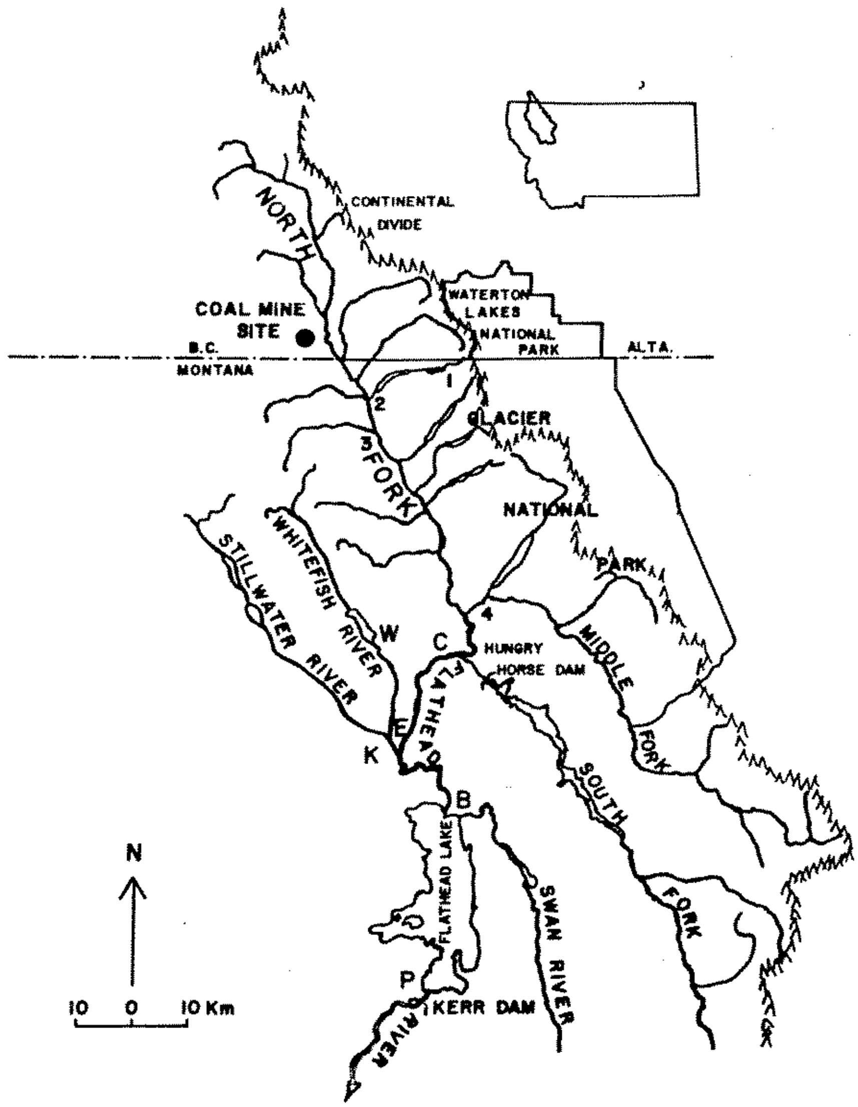

Figure 1. The upper Flathead River Basin showing the major tributaries contributing water to Flathead Lake. Locations of major towns are shown by P (Polson), B (Bigfork), K (Kalispell), E (Evergreen), W (Whitefish), and C (Columbia Falls). The numbers show where stream bank sediments were collected for algal assays (see text).

taries drain glacially sculptured mountains bounding the southern end of the Rocky Mountain Trench, in which a major mass of continental ice oscillated during Quaternary glaciation (see Richmond et al. 1965). Elevations above mean sea level range from 877.7 m at the lake outlet to over 3,000 m in the mountains of Glacier National Park and Flathead National Forest on the western side of the drainage basin. These mountains are predominantly Precambian argillites and dolomites of the Belt Series within the Lewis Overthrust zone of the northern Cordillera; some formations of more recent (Mississippian, Jurasic, Cretaceous) sedimentary rocks occur on the headwaters of the Middle and South Forks (see Mudge 1970).

Active remnants of extensive mountain glaciers and snowmelt at high elevations feed low-order tributaries which coalesce to form the 5 major forks of the Flathead River (Figure 1). These rivers discharge 10.4 km^3 of water annually into Flathead Lake (Table 1). About 65 percent of the annual inflow occurs during spring freshet, which generally peaks between 15 May and 10 June in the mainstream Flathead River above the lake. Minimum flows generally occur during the mid-winter freeze-up in late December and January. All of the tributaries are unregulated, except the South Fork (Figure 1); it is impounded by a hypolimnial-release dam, which totally regulates the flow patterns in the South Fork and, therefore, partially regulates the discharge pattern in the mainstream Flathead River.

Table 1. Basin area and discharge characteristics of major tributaries contributing flow through Flathead Lake (compiled from U. S. Geological Survey records and maps).

Tributary	Basin Area (km^2)	Total Volume[a] ($m^3 x 10^6$)	Maximum Flow (m^3/sec)	Minimum Flow (m^3/sec)	Period of Record[b] (yrs)
South Fork	4,307	3,190	1,310	0.21[d]	53
North Fork	4,009	2,670	1,960	5.61	50
Middle Fork	2,921	2,630	3,960	4.90	42
Swan	1,881	1,040	252	5.47	29
Stillwater	875	301	123	1.13	29
Whitefish	440	172	45	1.08	30
Ashley Creek[c]	520	29	--	--	--
Flathead River at Lake Outlet	18,372	10,500	2,340	0.14[d]	74

[a]Average annual discharge
[b]For calculation of mean total volume
[c]See Appendix II
[d]Due to dam closure

Numerous small (1-3 order) streams flow directly into the lake on its shoreline. All but one (Dayton Creek) of the larger streams are located on the east shore of the lake.

Lake Morphometry

The Flathead Lake basin was initially formed by glacial scour into soft sedimentary deposits, which were eroded into the graben basin of the Rocky Mountain Trench during the Tertiary epoch. Moraines at the terminus of the last ice advance dammed the western and southern outlets of the prehistoric lake. The present outlet has cut through the southern moraine to bedrock (Precambrian argillite), thus defining the base level (877 m) of the extant lake.

The present lake morphometry (Table 2; Figure 2) is predominantly of glacial origin. However, the upstream (North) end of the lake basin has been extensively modified by deposition of riverine sediments, mainly from the Flathead River. The Swan River is naturally impounded by glacial Swan Lake (Figure 1), which traps fluvial sediments. The Flathead River continues to build an extensive delta at its inlet by deposition of sediments during spring runoff. This alluvium is derived from streambank erosion in the larger tributary (4th and 5th order) and mainstream river segments. The rivers have down-cut into the Tertiary sediments, which lie below varying thicknesses of Pleistocene till, and considerable mass wasting occurs during spring runoff. Heavier sediments (silt and sand) are deposited on the delta and finer, clay-sized particles are carried far out into the lake basin in a turbidity plume during the freshet.

Table 2. Morphometric and hydrologic features of Flathead Lake, Montana, based on measurements at lake elevation of 879 m above mean sea level.

Maximum length	43.9 km
Maximum width	24.9 km
Shoreline length	301.9 km
Maximum depth	113.0 m
Mean depth	50.2 m
Area	495.9 km^2
Volume	23.2 km^3

Lake level is regulated between ca. 878 m and 882 m by Kerr Dam, a hydroelectric installation located 7 km downstream from the natural lake outlet. The dam is 62 m high and has been used to regulate lake level since 1938.

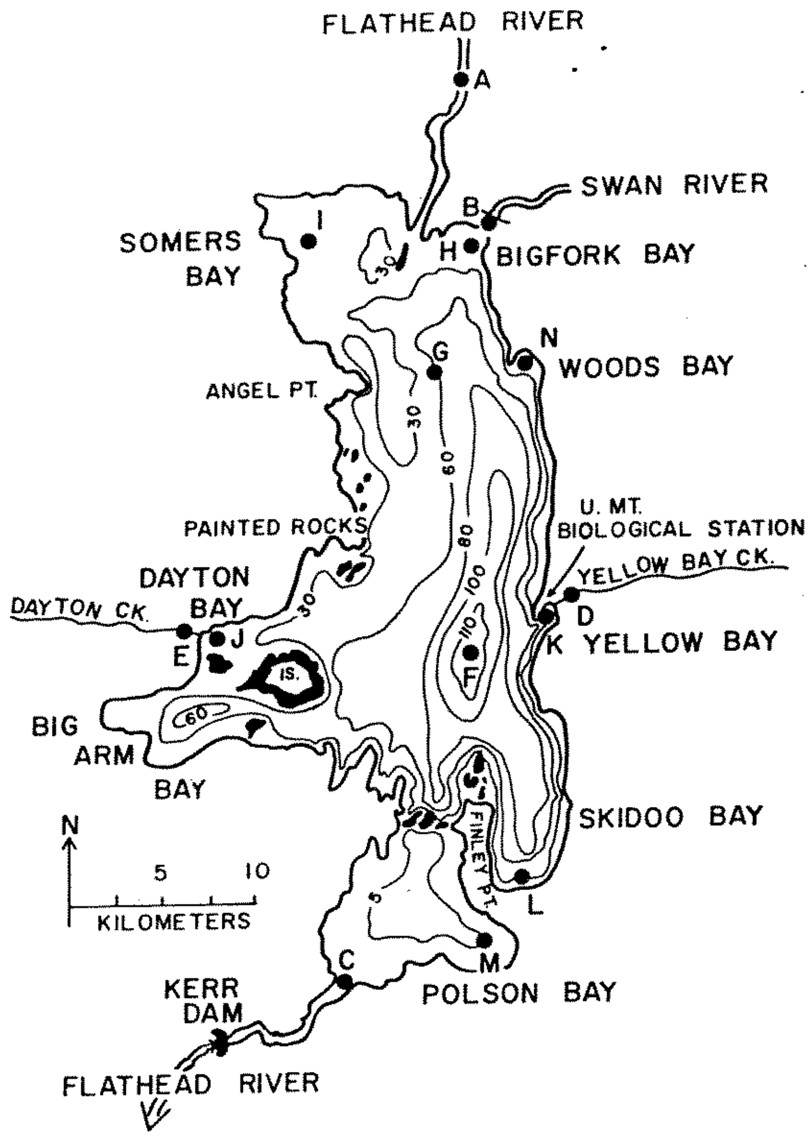

Figure 2. Bathymetric map of Flathead Lake with sampling sites shown by letters.

Land Use in the Drainage Basin

Coniferous forests occur extensively in the Flathead River Basin. Forest stands and species composition are locally determined by altitude, moisture and soil characteristics (see Pfister et al. 1977). Large acreages have been logged on the North and South Forks and Swan drainages. These areas have been extensively roaded to permit intense stand management, as well as facilitate oil and gas exploration. Also, a major coal deposit on the North Fork in Canada has been readied for production (see Figure 1). Other areas remain pristine (e.g. Glacier National Park and the Scapegoat and Bob Marshall Wilderness areas on the east side of the drainage basin).

Lands on the Flathead River flood plain and adjacent to the lower segments of the Stillwater and Whitefish Rivers are used for agriculture. Artificially fertilized crops include wheat and other grains, alfalfa, potatoes, and pasture.

About 50,000 people live in the basin, but tourism brings in many more seasonally. Major population centers occur adjacent to and on the flood plain of the Flathead River (Figure 1). Most of the urban areas are sewered (secondary) and discharge into the rivers. The Evergreen area (Figure 1) is a rapidly-growing urban area on the Flathead River flood plain that is served entirely by septic systems.

The areas adjacent to the shoreline of Flathead Lake are almost entirely developed. The steep east and west shores are utilized for recreational housing and large tracts are planted to cherry orchards. The north and south shores are mostly used for grain and pasture production. Except for the town of Polson (Figure 1), all shoreline residents presently utilize septic systems for sewage disposal.

METHODS

Time Frame and Sampling Sites

Time-series collection of biophysical data began in October, 1977, and continued through September, 1983. However, quantification of nutrient loading versus trophic status was limited to the 1978-1982 water years designated in U. S. Geological Survey discharge records (i.e. October, 1977-September, 1982). During most of the study period, monthly measurements of biophysical variables relative to mass balance calculations and analyses of lake trophic status were made at sites A-F (see Figure 2). Site F was located at the deepest point in the lake and was often referred to as the midlake site; monthly measurements of primary production, algal species composition and biomass, and other biological variables were emphasized at this site.

Additional regional data (e.g. temperature, nutrient concentration, current patterns, turbidity, and primary productivity) were gathered at sites G-M (Figure 2) on specific dates determined *a priori* or during spring freshet. These synoptic studies were aimed at accumulation of specific data for comparison to long term trends (e.g. primary productivity) at the regular sampling sites (A-F) or to more clearly delineate spatial patterns (e.g. movement of turbidity through the lake). Exact locations of the sampling sites are given in Appendix I.

Physicochemical Variables

Seasonal trends of depth profiles of temperature, dissolved oxygen, conductivity, and pH were determined by time-series plots of data gathered with either an Inter Ocean Systems, Inc. or Hydrolab, Inc. electronic metering system. Instruments were calibrated against ASTM or EPA-recommended standards prior to each sampling date; temporal records of field instrument quality control were maintained in the Freshwater Research Laboratory at the University of Montana Biological Station.

Data used for calculation of annual lake water budgets were obtained from the U. S. Geological Survey (discharge) and U. S. Weather Service (precipitation). All of the major inflow tributaries and the lake outflow (Table 1; Appendix II) were equipped with continuous recording devices.

Major surface currents were documented by plotting the movement of droghes (markers) on the lake surface in relation to various physical data (e.g. patterns of river discharge and temperature and lake temperature). Development and advective transgression of the annual turbidity plume was quantified by multiple plots of spatial measurements of water transparency made with a Kahlisco submarine photometer with a 1 m light path. These data were compared to photographs of the plume made from aircraft at various altitudes.

Characteristics of light penetration in Flathead Lake were determined *in situ* with a Licor submarine photometer equipped with a spherical sea cell (for measurement of photosynthetically active radiation with depth) and a deck cell (for measurement of incident PAR). Percent transmission at depth was routinely plotted. Additional data on water clarity were gathered with a 20 cm diameter (black and white) secchi disk.

Samples for chemical analyses were collected with Van Dorn type water bottles and transferred to high density polyethylene storage bottles for transport to the Freshwater Research Laboratory. In the lab, samples were placed in cold storage, preserved or processed immediately, depending on analyses involved (USEPA 1982).

Methodology for analyses of the variables needed for mass balance calculations of phosphorus and nitrogen are given in Table 3, along with other variables monitored in time series. These variables were utilized in attempts to order independent parameters as predictors of biological trends and trophic status using various regression routines (SPSS). Quality control of chemical analyses was based on control limits of <1 standard deviation of replicate samples for precision and 90-110 percent recovery of sample spike for accuracy. Routine analyses of EPA unknowns were also made, as an additional quality check. A detailed laboratory methods manual (Stanford et al. 1983b) was prepared by the UMBS staff and approved by EPA quality assurance personnel.

Bulk precipitation was collected at 3 sites near Yellow Bay by collecting precipitation and dry fall composits in 50 cm diameter funnel traps left in open, undisturbed areas on the UMBS grounds. Duration of composite collection was dependent on precipitation amounts. Samples were analyzed for nitrogen and phosphorus forms by routine methodology (Table 3).

Table 3. Laboratory analyses and methodology routinely made on samples collected from various locations and depths in Flathead Lake and inflowing and outflowing rivers, 1977-1982.

Parameter	Method (Reference)	Primary Instrument	Detection limit and units
Alkalinity	Titration to pH 4.5 (APHA, 1980)	Corning pH meter	.1 mg/ℓ-$CaCO_3$
Suspended Solids	Filtration (.45μm); Gravimetric (APHA, 1980)	Mettler Balance	.2 mg/ℓ
Turbidity	Nephelometric	Hach Model 2100A	.1 NU
Metals (Ca, Mg, Na, K, Fe)	Atomic Absorption Spectrophotometry (Parker, 1972)	PE Model 5000	.05 μg/ℓ-AW
Sulfate	Ion Chromatography (Small, 1979)	Dionex Model 16	0.1 mg/ℓ-S
Ammonia	Ion Chromatography (Small, 1979)	Dionex Model 16	0.1 mg/ℓ-N
Nitrite	Ion Chromatography (Small, 1979)	Dionex Model 16	0.1 mg/ℓ-N
Nitrate	Ion Chromatography (Small, 1979)	Dionex Model 16	0.1 mg/ℓ-N
Organic Nitrogen	Kjeldahl (APHA, 1980)	Bucchi Distillation	0.1 mg/ℓ-N
Soluble Reactive Phosphorus ($PO_4^{=}$)	Filtration (.45μm); Ascorbic Acid colorimetry (Wetzel and Likens, 1979)	PE Model 559 UV-VIS Spectrophotometer	1.0 μg/ℓ-P
Total Phosphorus	Persulfate Digestion; Ascorbic acid colorimetry (Wetzel and Likens, 1979)	PE Model 559 UV-VIS Spectrophotometer	1.0 μg/ℓ-P
Dissolved and particulate organic carbon	Filtration (.45μm); Persulfate digestion, infrared detection CO_2 (Menzel and Vacarro, 1973)	Oceanography International Inc., TOC system	.1 mg/ℓ-C
Inorganic carbon	Acid liberation, infrared CO_2 detection	Oceanography International Inc., TOC system	.1 mg/ℓ-C

Bio-availability of Sediment Phosphorus

Two separate approaches were used to assess possible fertilization or nutrient stripping effects of sediments discharged into the lake. Initially, river bank sediments were added to 2 m (width) x 6 m (length) enclosures (limno-corrals) located in Yellow Bay and various biophysical measurements followed temporally. These data were inconclusive (see Stuart 1982) and an alternate approach utilizing algal assays of sorted sediments was utilized. The assays emphasized documentation of phosphorus enrichment via microbial (mainly algae) mobilization of sediment-phosphorus.

During the spring of 1982, sediment samples were collected from eroding river banks within the Flathead River drainage (Figure 1 and Table 4) and from suspension in the

Flathead River for subsequent analyses of various phosphorus fractions. Bank sediments were stored in acid-washed polypropylene containers for transport.

Table 4. Sites for collection of eroding bank sediments and descriptions of formations (see also Figure 1).

Locations	Parent Material
1. Agassiz Creek at Upper Kintla Lake	Undesignated Tertiary siltstone
2. North Fork at Kintla Creek	Tertiary siltstone of Kishenehn Formation
3. North Fork at Ford Creek	Undesignated Tertiary siltstone
4. Middle Fork near McDonald Creek	Recent alluvium of Quaternary till

Suspended sediments from the Flathead River were concentrated using two methods. During peak runoff in 1979 and 1982, acid-washed, fiberglass tubs (dimensions 80 x 40 x 60cm) were placed on the Flathead River delta for collection of the heavier, fluvial sediments. After 1 week, the tubs were removed and the sediments transported on ice to UMBS for immediate processing. The second method involved concentration of suspended sediments >0.1 um in size using a Sharples continuous flow centrifuge. Seventy-six liters of Flathead River water were collected during peak runoff in 1982 near site A (Figure 2) in acid-washed Nalgene jugs and transported on ice to the lab for immediate centrifugation.

All sediments were dried at 50°C. Bank sediments were sieved through a 106 um mesh in order to collect the finer particles that would be carried in suspension by the river.

Total and inorganic sediment phosphorus were determined after methods of Aspila et al. (1976) and John (1970). Organic phosphorus was determined by difference.

The availability of sediment phosphorus to algae was estimated using algal bioassays (e.g. Sagher et al. 1975; Cowen and Lee 1976a,b; Williams et al. 1980; Dorich et al. 1980). The bioassay technique of Cowen and Lee (1976a,b) involved the incubation of a test algae with sediments as the only source of phosphorus in a phosphorus-free medium and direct counting of cells throughout the assay. Comparison of cell numbers resulting from sediment-phosphorus incubations with cell numbers of algae grown in solutions containing known amounts of KH_2PO_4 produced estimates of phosphorus available for algal growth.

The test algae, Selenastrum capricornutum Printz, was cultured in PAAP medium according to the S. capricornutum Printz Algal Assay Bottle Test (Miller et al. 1978). Before the start of an assay, the active culture was rinsed and then resuspended in phosphorus-free medium, with potassium supplied as KCl. The culture was incubated in continuous light (>4300 lux) until growth was less than 10 percent of controls over a 24-hour period. The phosphorus-starved culture was again rinsed and resuspended in phosphorus-free medium before use as inoculum in the assay.

Experimental flasks were prepared for each sediment sample by adding 100 ml phosphorus-free PAAP medium and 0.1g dried sediment to each 500 ml erlenmeyer flask. Control flasks were identical, except PAAP medium containing 200ugP l^{-1} was used. Standards contained known amounts of KH_2PO_4 in phosphorus-free PAAP medium, resulting in final concentrations ranging from 10 to 200ugP l^{-1}. Controls containing no phosphorus or sediment were also made. Four replicates were prepared for each experimental sediment sample or control and each of the 5 standards. All flasks were inoculated with phosphorus-starved S. capricornutum yielding an initial cell density of 4 x 10^4 cells ml^{-1}.

All flasks were incubated under assay procedure bottle test conditions (Miller et al. 1978) with daily shaking by hand. Immediately following inoculation and periodically throughout the assay, 1 ml was removed from each flask and preserved with Lugol's iodine solution. The algae were counted using an inverted microscope. For statistical validity, at least 200 cells were counted in each sample. At the conclusion of the assay (i.e. when the increase in algal biomass was less than 5% per day) the entire contents of the standard and experimental flask were filtered through Gelman glass fiber filters (0.3 um pore size) and the total and inorganic phosphorus content on each filter determined as described previously for sediments.

Chemical extraction techniques were also investigated to determine their usefulness in estimating available sediment phosphorus. A modification of the NaOH-extractable phosphorus procedure of Sagher (1976) was used in which 0.2g of sediment was extracted with 400 ml of 0.1 N NaOH/1.0 N NaCl solution for 16 hr on a shaker table. It has been determined that the higher the solution:solid ratio the more inorganic phosphorus will be extracted. Our experiments confirmed that to be true for lower ratios (with good correlations); however, beyond a ratio of 400 ml:0.2g, there was no increase in the amount of phosphorus extracted. Extracts were analyzed by the ascorbic acid method (Table 3).

In a variant of the nitrilotriacetic acid (NTA)-extractable phosphorus procedure of Golterman (1976), 0.4g of sediment was extracted with three 300 ml portions of 0.01 M NTA solution at pH 7. Each successive extraction lasted 4 hours. Between extractions the samples were centrifuged and the sample recovered for the next extraction. Preliminary analyses of the extracts using the method of Harwood et al. (1969), indicated an interference of the complexation reaction by NTA. Doubling the concentration of the ammonium molybdate solution and increasing the color development time to 40 minutes resulted in linear standard curves and good spike recovery. Analysis of each of the successive extracts showed that at least 1/3 of the NTA-extractable phosphorus was removed during the third extraction, indicating that continued extraction would remove additional phosphorus. The majority of the sediments appeared to be conditioned by the extraction, as phosphorus concentration in the third extract was substantially higher than in the others.

Biotic Variables in Flathead Lake

The major biotic variables of lake trophic status utilized throughout the 5-year study period were primary production, species composition, and standing crop of phytoplankton.

Productivity was quantified only on cloudless and generally calm days utilizing the depth-integrated light and dark bottle method ($H^{14}CO_3$ incubation) (Steeman-Nielson 1952). Duplicate incubations were made during mid-day (1000-1400 hours), usually at 6 depths (1, 2.5, 5, 10, 20, 30m) in the euphotic zone. We killed samples by acidification to pH 2 with concentrated HCl. Subsamples were purged with nitrogen gas and radioactivity was assayed via scintillation cocktail (Aquasol[c]) on a Beckman LS-7500 counter (Schindler et al. 1972; Theodorssen and Bjarnasson 1976). Daily fixation rates were calculated by Lind's (1979) formula.

On dates of synoptic studies of primary productivity, assistants were sent to each of the bays of interest (sites H-M) and midlake (site F) so that simultaneous profiles could be taken. Some sites were too shallow to permit 30 m profiles. Comparisons were made on the basis of unit area or volume integrations and on the basis of depth corrected data.

Light data (total irradiation and percent transmission at depth) were gathered with the Licor meter described above, concomittant with productivity profiles.

Phytoplankton counts and measurements for quantification of standing crops were made using a Wild M40 inverted scope fitted with Utermohl counting chambers. Samples were preserved with neutralized formalin and iodine was used to accelerate sedimentation in the 100 ml Utermohl chambers (see Lind 1979; Wetzel and Likens 1979). Cell measurements were made with the aid of stage micrometers and an occular micrometer (cf. Ellis and Stanford 1982).

Total biomass of phytoplankton and bacteria was measured by adenosine triphosphate (ATP) assays (cf. Karl 1980), using hot extractions of filtered (.45um) samples (Perry et al. 1979). Biomass-depth profiles were made at the midlake site (F) only, and during the period 1977-1979.

RESULTS AND DISCUSSION

Water Budget

Runoff in the major sub-basins (i.e. Flathead, Swan, Stillwater and Whitefish) for the 1978-80 water years was below the average of the ca. 50 year record of discharge at the U. S. Geological Survey gages (Tables 1 and 5). In 1981-82, runoff was slightly above the annual average. Thus, average riverine discharge into Flathead Lake during the study period (9.957 km^3/year) was slightly below normal (10.345 km^3/year).

Mass balance calculations for annual water movement into and out of the lake (Table 5) indicated that riverine inflow and direct precipitation on the lake surface accounted for most of the water income. Evaporation was estimated from data of Kohler et al. as given in Anderson and Jobson (1982). Also, the 12 larger (2nd and 3rd order) creeks draining into the lake along the east shore, and Dayton Creek on the west shore, were not continually gaged, forcing us to crudely estimate their contributions to total inflow. Slight inaccuracies in these estimates probably explain the slightly greater loss total compared to water income in Table 5. We also could not directly measure groundwater income or loss; but, assuming that the estimated values in Table 5 were not grossly in error, we concluded that groundwater was not a major source in the water budget. Seepage areas on the shoreline were common and some were thought to be of importance as spawning areas for Kokanee salmon (Oncorhynchus nerka) (Decker-Hess and Graham 1983). However, riverine inputs dominated the water budget to the extent that a water mass equal to the lake volume is exchanged in about 2.2 years:

$$\frac{\text{lake volume (km}^3\text{)}}{\text{annual inflow (km}^3\text{/yr)}} = \frac{23.2}{10.4} = 2.2 \text{ years.}$$

Table 5. Water budget for Flathead Lake based on mean annual additions and losses to the lake basin for the period October, 1977, through September, 1978 (see Appendix II for details of calculations).

Sources of Water Income	Volume ($m^3 x10^9$)	Sources of Water Loss	Volume ($m^3 x10^9$)
Flathead River Inlet	8.768	Flathead River Outlet	10.044
Swan River	1.217	Evaporation	0.336
Shoreline Creeks	0.152		
Precipitation	0.236		
Totals	10.373		10.380

The rate of water movement through Flathead Lake was dramatically affected by timing and volume of spring freshet, lake morphometry, and operation of Kerr Dam. During 1978-82, 58 percent (5.8 km^3/yr) of the inflow occurred during the freshet (May and June). Thirty-eight percent (2.2 km^3/yr) of the freshet flow was retained by Kerr Dam in the lake basin for discharge for power generations during winter months (November-March). Water retention by Kerr Dam raised summer elevation of the lake to about the same maximum levels reached on the average prior to regulation (i.e. before 1937) (Figure 3). The bedrock sill in the lake outlet at Polson

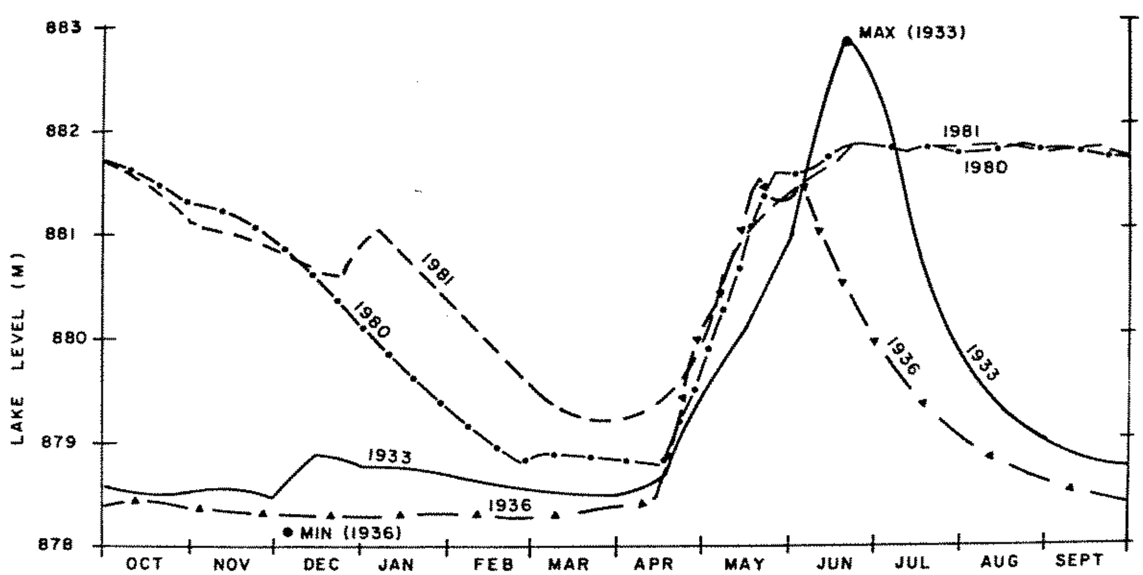

Figure 3. Dynamics of lake levels (altitude above mean sea level, m) before and after regulation by Kerr Dam.

(3 km upstream from Kerr Dam: Figure 2) determined base-level of the lake and, prior to regulation by Kerr Dam, also limited the rate of water outflow during freshet. Thus, preregulation lake level increased dramatically in spring flooding extensive shoreline areas at each end of the lake, and receded to base level by late August, depending on intensity of the freshet. Operation of Kerr Dam in concert with inflow augmentation by Hungry Horse Dam now greatly retards the receeding portion of the lake hydrograph (Figure 3). During 1978-82, average retention time of a unit volume of freshet flow was increased by 33 percent over the estimated preregulation retention time.

Physicochemistry

Potter and Stanford (1975) referred to Flathead Lake as dimictic, although they noted that the water column freely circulated throughout the winter. The lake has frozen over completely only about once in every 10-15 years. The prevailing southwesterly winds and long fetch generally prevent midwinter ice formation. Dimixis was indicated by our data (Figure 4), especially for the unusually cold winter of 1978-79. The other, milder winters during the study period did not elicit pelagic ice cover and temperatures in the water column remained vertically homeothermous through the midwinter months (December-March). In all years (including those not shown in Figure 4), the water column cooled to less than 4°C in fall and early winter (September-January) and warmed to greater than 4°C in spring (March-June). Significant thermal stratification developed in June and by late July the lake was firmly stratified each year. However, in 1979 summer stratification was less firm and surface temperatures were 2-3^{o}C cooler than other years, due to the extended cold temperatures the previous winter (Figure 4). We emphasize that wind mixing and surface currents (see below) varied location of the thermocline in the water column considerably on a spatial scale.

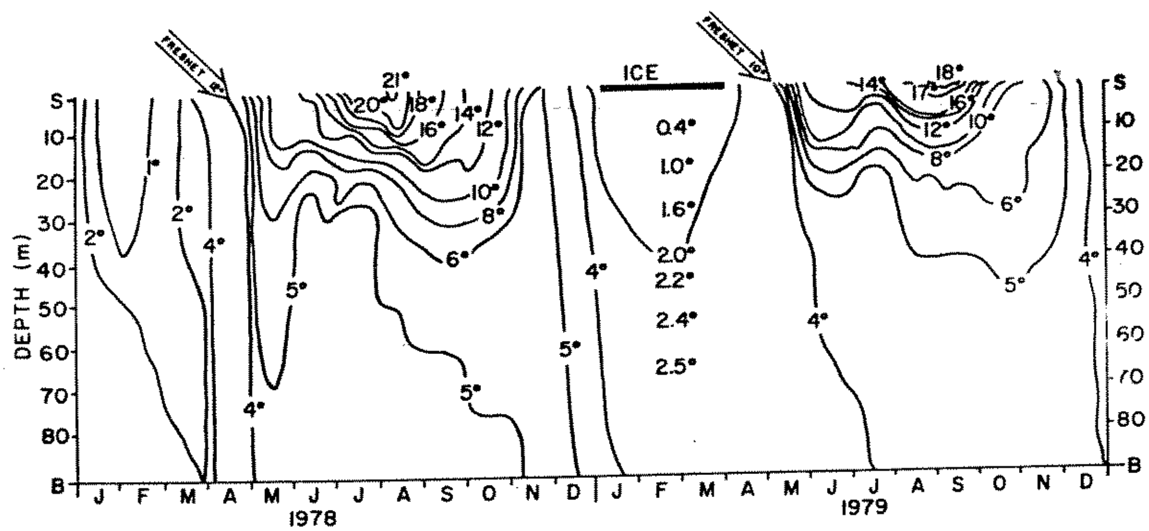

Figure 4. Temperature (^{o}C) as function of depth for period 1978-79 for the midlake (site F) area of Flathead Lake. Arrows show the date of turbidity plume overflow and temperature of Flathead River on that date.

Dissolved oxygen concentration remained at or above saturation throughout the study period, due to predominance of cold waters in the water column of the lake, generally good wind circulation of the water column and relatively low rates of organic catabolism (see below). Dissolved oxygen did not vary diurnally and depth profiles were always orthograde. Deviations from saturation (or greater) were not observed in the littoral areas, although some deficits may have occurred in Polson Bay where rooted vascular plants were abundant.

Although riverine discharge from the Flathead and Swan Rivers dominated the water budget of the lake, analysis of chemical constituents of influent and effluent waters in comparison to the pelagic area of the lake indicated that the lake is somewhat of a mixing pot and that considerable mass of solids are retained in the lake basin (Table 6). Differences observed between the Swan and Flathead Rivers were due

Table 6. Mean concentrations (µg/ℓ) of selected constituents in Flathead Lake waters 1977-1982. Numbers of measurements are given in parentheses.

Chemical Constituent	Flathead River Inflow	Swan River Inflow	Yellow Bay Ck Inflow	Flathead Lake Pelagic	Flathead River Outflow
Total Phosphorus (µg/ℓ-P)	22.7 (53)	7.8 (49)	11.3 (24)	7.2 (328)	6.3 (45)
Dissolved Inorganic Phosphorus (µg/ℓ-P)	1.35 (15)	<1.0 (19)	4.7 (15)	<1.0 (29)	<1.0 (16)
Nitrate (µg/ℓ-N)	61.3 (61)	27.6 (54)	31.9 (24)	41.7 (321)	23.0 (47)
Total Kjehdahl Nitrogen (µg/ℓ-N)	83.1 (35)	74.6 (26)	50.0 (8)	69.8 (233)	79.1 (25)
Calcium (mg/ℓ-Ca)	22.67 (22)	23.1 (22)	25.13 (19)	22.7 (63)	21.7 (22)
Magnesium (mg/ℓ-Mg)	5.69 (22)	5.35 (22)	4.86 (19)	5.30 (63)	5.27 (22)

in part to differences in sub-basin geology, but, mainly pertained to the retention of suspended and dissolved solids by Swan Lake. During freshet the Flathead River transported a heavy suspended solids load from unregulated tributaries and, except for discharge from large lakes in Glacier National Park, was not influenced by the "sink" effect of dissolved solids, since water releases from Hungry Horse Reservoir were hypolimnial. The shoreline creeks, exemplified by Yellow Bay Creek in Table 6, were fed mainly by spring brooks and, therefore, contained higher concentrations of dissolved solids.

Concentrations of chemical constituents in the Flathead River inflow exhibited marked temporal dynamics. Values for organic nitrogen and total phosphorus increased several orders of magnitude during spring freshet (Figure 5). This

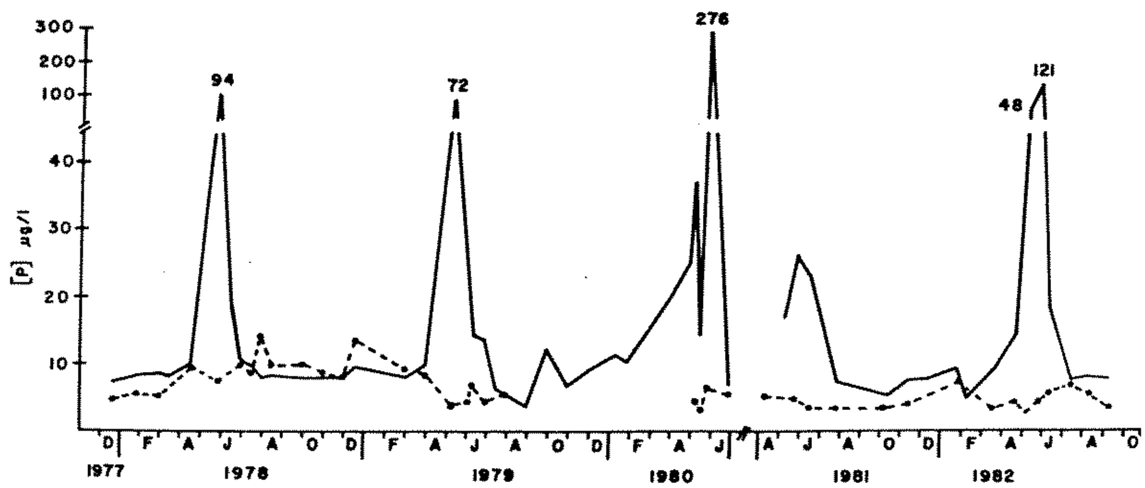

Figure 5. Time-series plot of total phosphorus (solid line) concentrations in the Flathead River (site A) during 1977-82. Soluble reactive phosphorus (broken line) dynamics are also shown.

was due primarily to the presence of high mass of suspended solids (Figure 6) composed of sediments and organic detritus in river floodwaters. Also, upstream groundwater reservoirs were probably flushed by the freshet, since concentrations of nitrate and dissolved organic carbon were elevated well above base flow conditions, especially during the rising side of the hydrograph. Other constituents, which reflect basin geochemistry (e.g. calcium, magnesium, bicarbonate, sulfate, etc.), were diluted by freshet flows and decreased in concentrations. Similar trends were observed in the Swan River inflow, but were less dramatic, due to amelioration of freshet effects by Swan Lake.

The levels of nitrogen (NO^3 and TKN) and total phosphorus increased markedly during freshet. The increase in organic nitrogen apparently was due to organic detritus in seston and to dissolved organic matter; nitrates were probably flushed from groundwater reservoirs. Nitrate loading from the alluvial aquifers under the unsewered urban area of Evergreen (Figure 1) on the Flathead River flood plain, was demonstrated in a concurrent study using computer imagery of multispectral data gathered by high altitude aircraft and correlated with a spatial series of nitrate measurements (Mace 1981). Concentrations of total phosphorus were strongly correlated with suspended solids during freshet (Figure 6).

Movement of turbidity into the lake upon the onset of spring freshet provided a remarkably clear picture of advective circulation. Measurable turbidity began to enter the lake in concert with the rising hydrograph of the Flathead River in mid-April each year. However, the concentration of suspended solids in the inflowing water was dependent on the intensity of runoff (amount and altitude of snow deposition and melt rate in head-waters) and the volume of clear water discharge from Hungry Horse Reservoir. We observed that the general runoff pattern involved: (1) a rise in the hydrograph

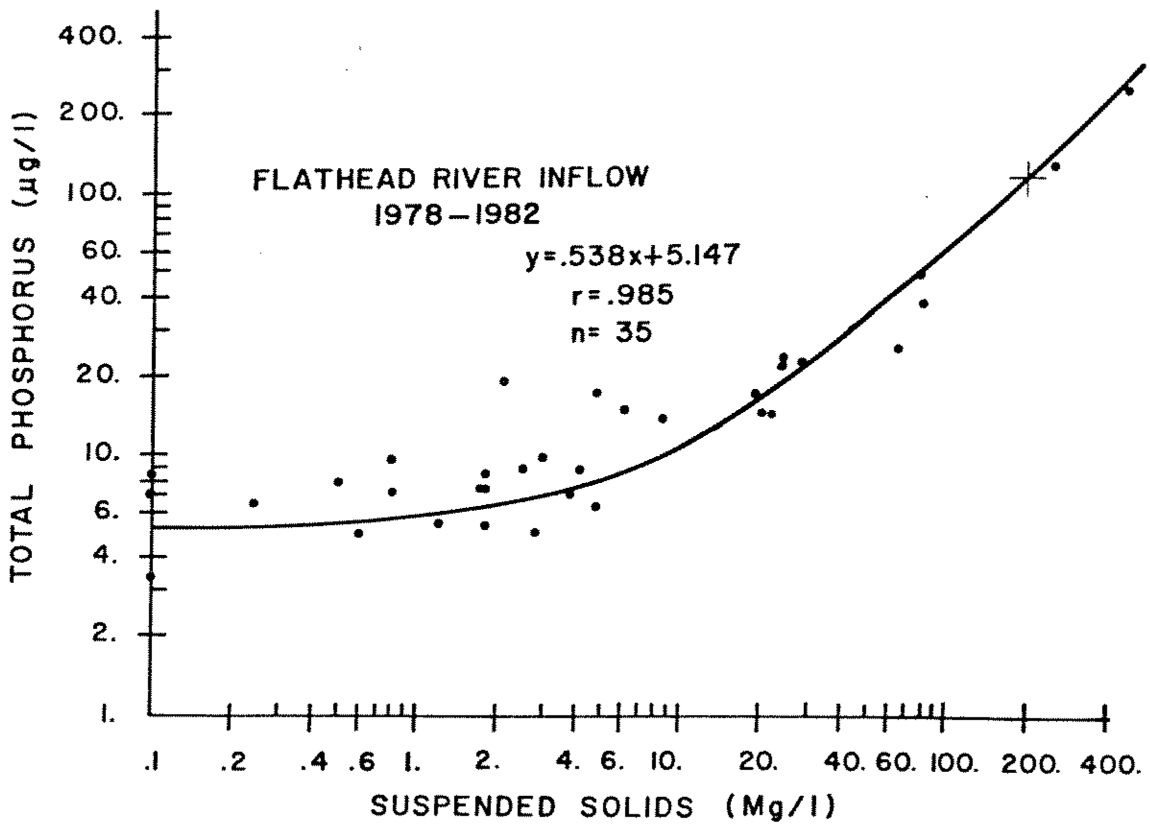

Figure 6. Relationship between suspended solids and total phosphorus concentration in the Flathead River (site A) over the period 1977-82. The linear regression accurately predicted total phosphorus in a quality control sample taken of 31/V/83 as shown by the cross. The log plot is used to show tailing of the curve at base flow (i.e. < 10 mg/l TSS).

in mid-April, corresponding to melt of low altitude snowpack; followed by (2) a period of reduced flow in response to cooler weather in early May, which maintained the high elevation snowpack (and predominate water mass) until late May. The early plume scarcely moved down-lake beyond the delta area on low flow years, but proceeded into pelagic regions on high flow years. The major freshet plume generally eventuated between 15 May and 15 June. Turbidity flows were less visual during the 1977-80 period, due to below average runoff; but, nevertheless, followed the same downlake circulation pattern observed in years of higher flow.

In all cases, the plumes moved downlake as overflows (Figures 7 and 8) and were deflected to the west by the

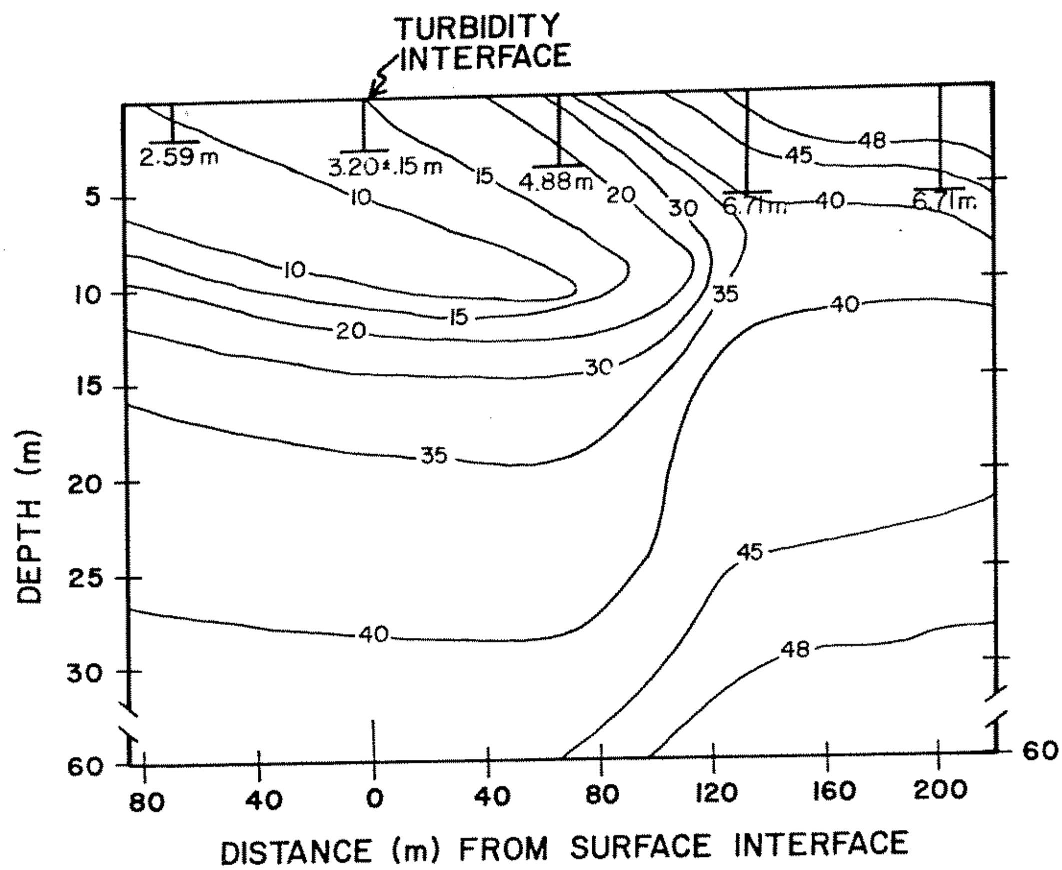

Figure 7. Isopleths of water clarity (percent transmission) showimg overflow of freshet turbidity in pelagic area of Flathead Lake. Secchi depths (m) are shown at top of the figure.

coriolis effect (Figure 9). Turbidity flowed down the west shore, deflecting into Big Arm Bay, and split at the constriction at the north end of Polson Bay into backwash flow up the east shore and outwash flow toward the outlet.

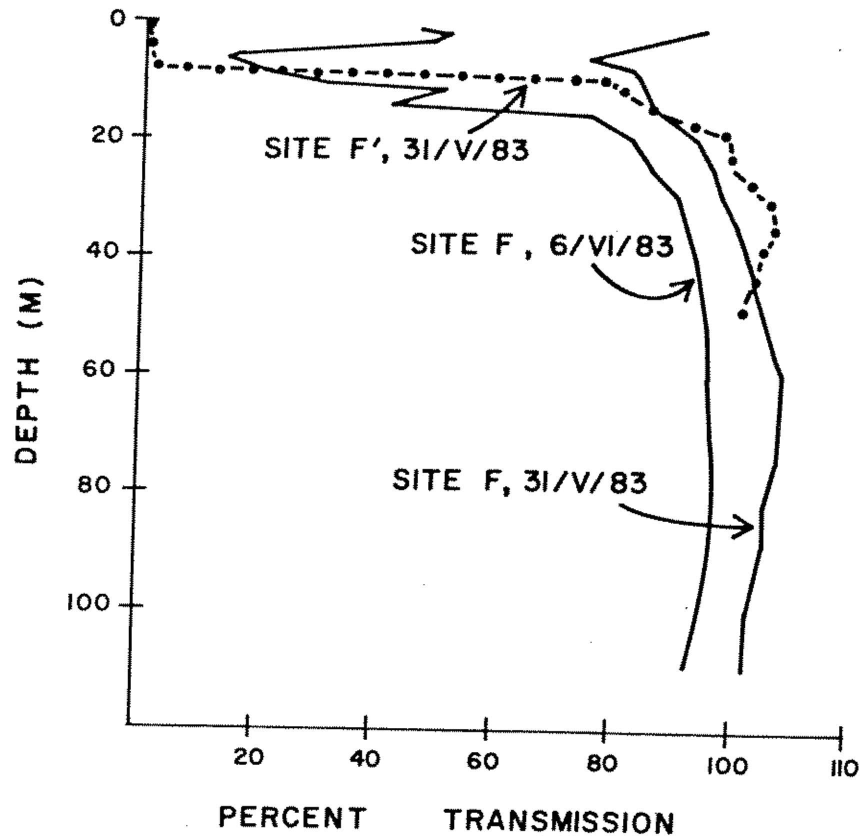

Figure 8. Depth profiles of water clarity at sites F and F[1] (shown in Figure 9), demonstrating overflow of freshet turbidity.

The general pattern of turbidity flow in Flathead Lake (Figure 9) was controlled by several simultaneous limnological phenomena. First, the plume was pushed downlake by the intensity (volume) of floodwaters, which, of course, also largely determined how far out into the lake suspended solids were carried. Second, freshet flows were ca. 4 to 6°C warmer than resident lake water (i.e. 8-10° in river versus 2-5° in pelagic areas of the lake, see Figure 4), and, therefore, the

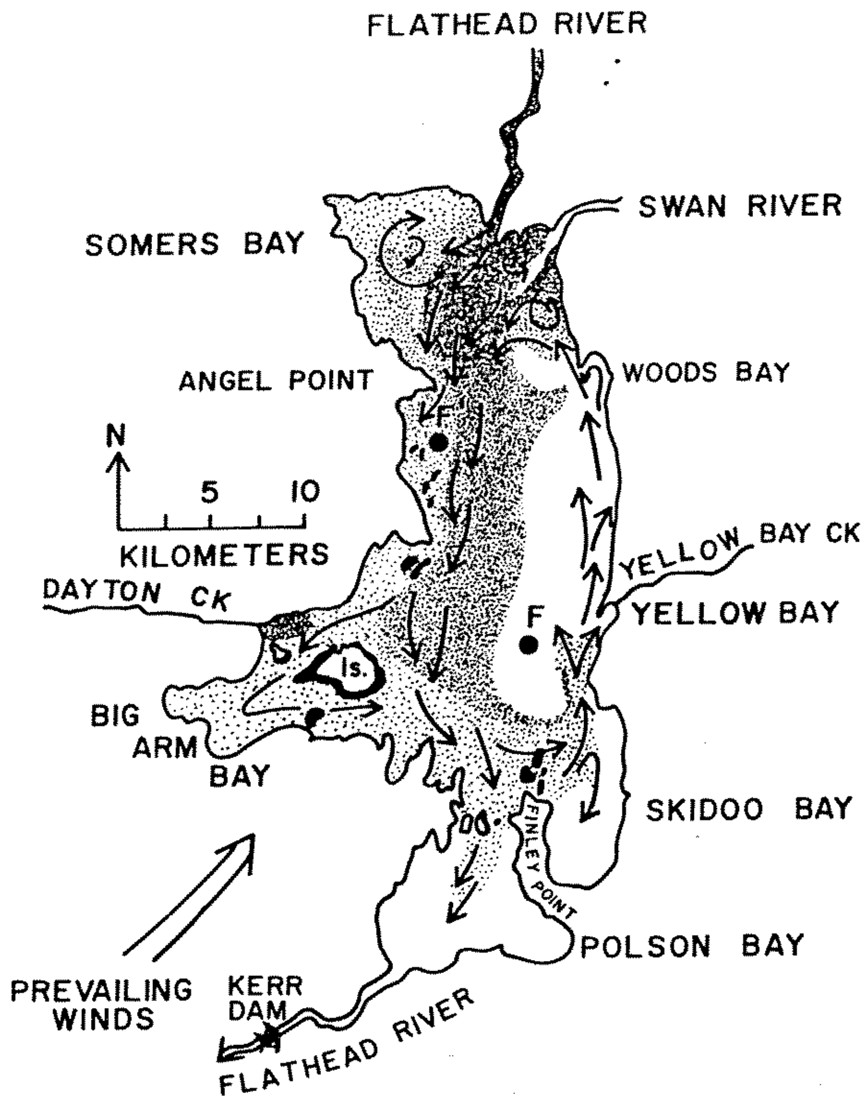

Figure 9. Advective circulation pattern (arrows) in Flathead Lake. Shaded areas describe location of freshet turbidity on 30/V/83, as drawn from aerial photographs.

less dense water mass from the river flowed over the lake surface. Third, a strong, right-bending (coriolis) current was especially pronounced in the littoral areas, as a consequence of the maturation of a thermal bar (warm resident waters (6-8°C) near shore flowed north on the east side and south on the west side of the lake around the major mass of colder (2-5°C) water in the pelagic area, cf. Wetzel 1975); this strong, internal current deflected the turbidity plume to the west shore and helped carry it toward the narrows at Polson Bay and back north up the east shore. Finally, the natural hydraulic flow was south toward the outlet, which carried a maximum of ca. 1200 m^3/sec. during freshets in 1977-83. Hydraulic flow continuously shoved the less-dense surface waters toward the outlet; the rate of flow depended on the release rate from Kerr Dam. Prior to regulation, hydraulic flow was determined by the amount of water that could pass over the bedrock sill in the outlet. Since Kerr Dam functions as a water retention device, southward hydraulic force was probably much less in the extant pattern of freshet movement through Flathead Lake than occurred in the unregulated lake. At no time did we detect discharge of measurable sediments at the outlet, although it undoubtedly occurs on very high flow years (e.g. 1964, 1974).

The generalized picture of turbidity flow through the lake (Figure 9) was less clear on low flow years (1978-80) and during periods when clear water released from Hungry Horse Reservoir augmented freshet flows. In these years sediment concentrations in the lake inflow decreased, while discharge remained high, and river temperature decreased, all of which combined to cause the plume to interflow. Also, strong, northwesterly winds, produced by occasional weather fronts or thunderstorms, occasionally pushed turbidity from the west shore toward the middle of the lake. On several occasions we observed clear water in west shore bays, as the plume slid easterly. Such winds also produced seiche movements with as much as 40 cm oscillations.

Measurable turbidity was no longer present in the water column by early August in 1979-83. Sediments apparently gradually settled to the lake bottom. Near the river delta, we observed 2-4 cm accumulation of silt and clay sized sediments in a 1 m^3 trap placed on the lake bottom for two weeks during the height of the 1982 freshet. A concurrent study (Moore et al. 1982) suggested that the average sedimentation rate was about 0.3 mm per year lakewide. It was apparent from our transmission data that considerably greater sedimentation and higher settling rates of larger particles occurred in the delta area.

Physics of particle transport and settling was not studied in detail. However, sediment particles reaching the midlake site (F) were consistently between 5 and 45 um in size. Grain size analysis of bottom sediments also showed predominance of clay-sized particles (Moore et al. 1982).

Due to the high correlation between sediment loading and phosphorus concentration in the river (Figure 6), phosphorus values in the lake were higher in areas most profoundly affected by plume overflow and sediment deposition (i.e. near the Flathead River delta). Farther downlake (e.g. at midlake site F), phosphorus concentrations were lower because lesser volumes (and smaller particle sizes) reached those areas and the plume was diluted by lake volume (Figure 10).

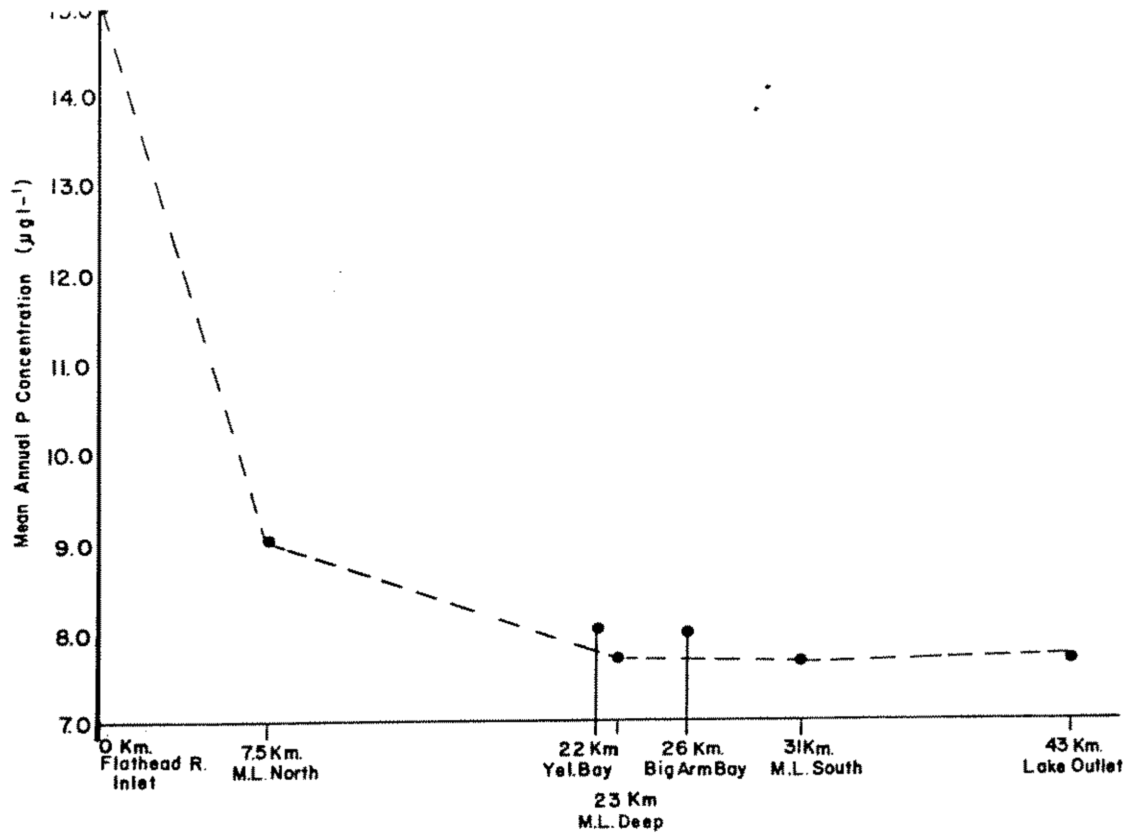

Figure 10. Average annual concentrations of total phosphorus measured at various locations downlake from Flathead River inlet during period 1977-1980.

Our time-series data of chemical variables at pelagic sites, mainly site F, did not show major inflections due to freshet overflow for constituents other than total phosphorus and, to a lesser extent, nitrate. Differences in conductivity and general ion composition could not be detected with our _in situ_ instruments very far downlake from the Flathead River inlet. Since total phosphorus was correlated with sediment loading in the Flathead River (Figure 6), we expected to measure a similar relationship at the midlake site. However, gravimetric measurements of suspended solids were only slightly above detection limit of .1mg/l, even during plume overflow, which prevented meaningful segregation of variance spatially within the water column. But, measurements of water clarity using the submarine transmissometer permitted sensitive determination of seston distribution (See Figure 7). We observed a highly-significant, linear correlation between total phosphorus concentration [P] and percent transmission (T) (i.e. water clarity): $[P] = .08T + 12.3$, $r = -.72 (P<.01)$, and $n = 44$. This regression was based only on data obtained when transmission readings were taken simultaneously (time and space) with water samples for phosphorus analysis. Phosphorus concentrations were consistently 2-10ug/l above the mean (7.2ug/l) in the presence of the overflow plume. Otherwise, there existed no statistically significant trends in the 84-month data series. Nitrate concentrations also were elevated by advective circulation of river water: $[NO_3] = -.12T + 5.1$, $r = -.61$ $(p<.1)$, although the relationship was weakened by the tendency for concentrations to remain very close to the detection limit of the analytical method. We also observed epilimnetic nitrate regeneration after fall turnover (Figure 11).

In spite of the apparent fertilization effect of advective circulation of the sediment plumes each year, phosphorus and nitrogen concentrations remained very low in the water column. Nitrogen:phosphorus mass ratios averaged 16.2:1.0 over the entire 5-year period study, supporting our *a priori* contention that we were dealing with a system in which autotrophic production was limited by a phosphorus deficit, relative to nitrogen. It is important to note that presence of nitrogen ions thought to be usable in plant (algae) metabolism was limited to nitrate; ammonium and nitrite were never detected in Flathead Lake. Also, nitrate depletion was evident in the epilimnion in late summer (Figure 11). Therefore, the trophic response to added nutrients from cultural sources may well involve nitrogen forms, at least during some periods of the year.

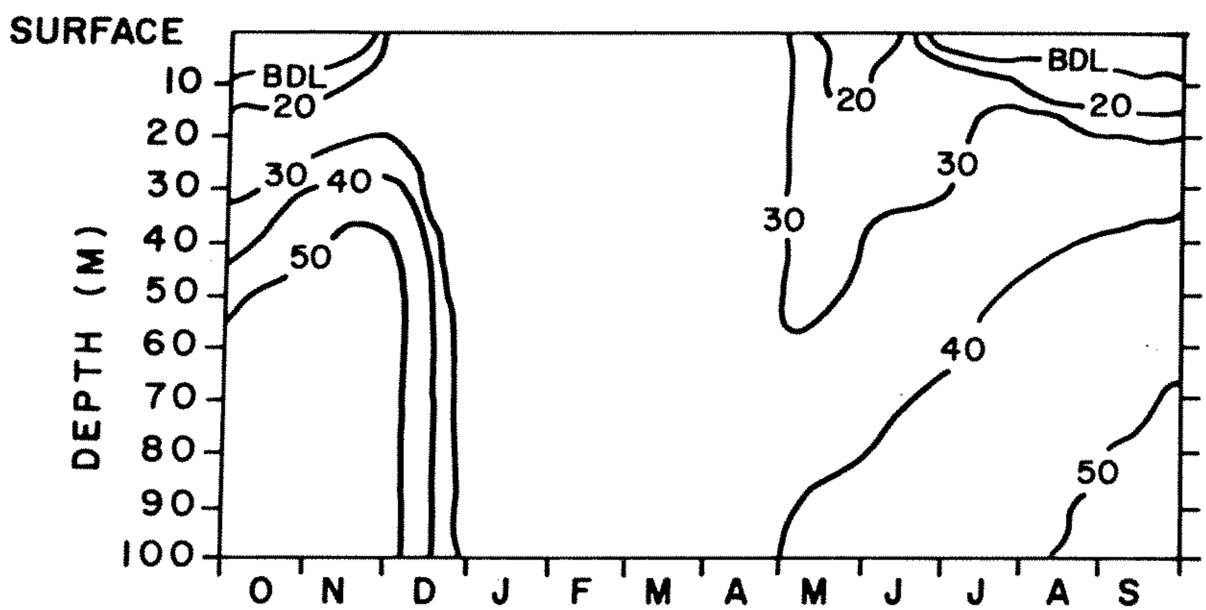

Figure 11. Nitrate concentration (isopleths) over time and depth in pelagic area of Flathead Lake, 1982 water year (data are ug/l NO^{3}-N). BDL means below detection limit.

Mass Balance of Phosphorus and Nitrogen

Annual loading rates of phosphorus and nitrogen from all known inputs were calculated by multiplying the average total annual discharge by the average annual concentrations of phosphorus and nitrogen (Tables 7 and 8). Secondary sewage effluent entered the Swan River downstream from our sampling

Table 7. Annual phosphorus budget for Flathead Lake, 1978-1982.

Phosphorus Source	Water Volume ($m^3 x 10^9$)	Phosphorus Concentration (mg P/m^3)	Phosphorus Mass (MT)
Influent			
Flathead River	8.768	8.6*	75.41
Swan River	1.217	7.8	9.49
Bigfork Sewage	.000169	6,400.0	1.08
UMBS Sewage	.0000055**	3,621.7**	0.02
Shoreline Creeks	.152	11.3***	1.72
Bulk Precipitation	.236	127.0	29.97
Total	10.373		117.69
Effluent			
Flathead River	10.044	5.9	59.26

*Corrected for bioavailability of sediment phosphorus (see text).
**Extreme seasonal variation, see Appendix III.
***Data for Yellow Bay Creek only (n = 24).

Table 8. Annual nitrogen budget for Flathead Lake, 1978-1982.

Nitrogen Source	Water Volume ($m^3 x 10^9$)	Nitrogen Concentration (mg N/m^3)	Nitrogen Mass (MT)
Influent			
Flathead River	8.768	144.4	1,266.10
Swan River	1.217	102.2	124.38
Bigfork Sewage	.000169	33,270.0	5.62
UMBS Sewage	.0000055*	9,473.3*	0.05
Shoreline Creeks	.152	82.0**	12.46
Bulk Precipitation	.236	764.3	180.38
Totals	10.373		1,588.99
Effluent			
Flathead River	10.044	102.1	1,025.49

*Extreme seasonal variation, see Appendix III.
**Data for Yellow Bay Creek only (n = 24).

point (site B, Figure 2) and UMBS discharged tertiary treated effluent into the lake on the east shore. Data for these sources were obtained from unpublished monitoring records provided by Montana Department of Health and Environmental Sciences. Other sewage treatment facilities in the urban areas shown in Figure 1 release secondary effluent into the Flathead River system well upstream from our inlet sampling site A (Figure 2). We used only data from Yellow Bay Creek to calculate loading from shoreline creeks, because most of the creek inflow into the lake was derived from the east shore and the chemical and flow dynamics of these creeks were quite similar. Estimates for loading on the lake surface from atmospheric deposition were based on our data collected at Yellow Bay (Table 9). Concentrations of phosphorus and nitrogen were surprisingly high in bulk precipitation and represent a significant contribution to the loading totals. On numerous occasions, smoke from wood stoves and slash burning created visually dirty air in the Flathead Valley, especially during winter periods of poor air circulation caused by temperature inversions. Spring winds also carried substantial amounts of pollen onto the lake surface. Microscopic inspection of particulates in precipitation samples revealed presence of pollen and what appeared to be very small ash particles intermixed with variable amounts of dirt (clay) and unidentifiable detritus. Unfortunately, no comparable bulk precipitation data exist; however, air quality studies during 1979-83 showed that particulates from wood smoke and dust were major air pollutants in the Flathead Valley (Coefield 1983). Also, we were concerned about potential loading from groundwater sources, including breakout of sewage leachate from shoreline septic systems. Because only limited and localized data concerning volume and quality of groundwater flow into the lake existed, and since our water budget balanced (Table 5), we concluded that groundwater loading was insignificant in the lakewide nutrient budgets. However, we acknowledge that localized shoreline loading from low volume groundwater or surface flow may have occurred. This problem and the lakewide distribution of nutrients from wet and dry precipitation requires additional study.

Table 9. Concentrations (mg/l) of chemical constituents (Nitrate Nitrogen = NO_3-N; Kjeldahl Nitrogen = TKN; Sulfate Sulfur = SO_4-S; Soluble Reactive Phosphorus = SRP; Total Phosphorus = TP) in bulk precipitation samples collected at 3 locations in the vicinity of Yellow Bay, Flathead Lake, from 1979-1982. Numbers of samples are indicated in parentheses.

Collection Point	NO_3-N	TKN	SO_4-S	SRP	TP
UMBS Dock[1]	.2358 (23)	.4748 (13)	.2864 (23)	.0529 (22)	.1137 (17)
Yellow Bay[1]	.3686 (19)	.4361 (12)	.3312 (18)	.0261 (19)	.0734 (19)
Residence[2]	.0986 (13)	.6793 (12)	.2803 (16)	.0608 (16)	.1941 (14)
Grand Mean	.2343 (55)	.5300 (37)	.2993 (57)	.0466 (57)	.1270 (50)

[1]sampling point at lake shore elevation
[2]sampling point 500 m above lake shore elevation

The annual phosphorus budget for Flathead Lake (Table 7) was adjusted for bio-availability of sediment phosphorus in the Flathead River. Had we used the values actually measured during spring freshet (i.e. periods when TSS exceeded 10mg/l and skewed the mean annual phosphorus concentrations to 22.7mg/l), the loading rate via the Flathead River would have exceeded 199 MT annually. This would have elevated the total load from all sources by 49 percent. Regardless, it is quite clear that on a mean annual basis, most of the phosphorus (64 percent) and nitrogen (80 percent) loads entered Flathead Lake via the Flathead River; and, of the incoming tonnage, 50 percent of the phosphorus and 36 percent of the nitrogen were retained in the lake (Tables 7 and 8).

The ratio of phosphorus to nitrogen (1:16.2) at midlake, discussed above, also was illustrated in mass balance data. The ratio of phosphorus to nitrogen loading was 1:13.5, which again supported our *a priori* contention of phosphorus limitation. The ratio was 1:17.3 in lake effluent. This downlake gradient of nitrogen surplus relative to phosphorus clearly demonstrated the importance of phosphorus retention within the lake basin.

Biotic Availability of Sediment Phosphorus

Inorganic phosphorus comprised 73 to 96 percent of the total phosphorus concentration in the Flathead River sediments, except for North Fork bank sediments collected near Ford Creek (Table 10). The organic phosphorus concentration of that sample was much higher than in the other sediments. Total phosphorus concentrations in all bank and suspended

Table 10. Phosphorus concentrations (μgP/g $\pm$ 1.0 Standard Deviation) in various alluvial clays from selected river banks compared to concentrations in sediments obtained in the Flathead Lake freshet turbidity.

Sediment Collection Site	Phosphorus Total	Inorganic	NTA Extractable	NaOH Extractable	Algal Available[a]
Stream Banks					
Kintla	602.5 ± 0.1	577.3 ± 3.5	393.0 ± 0.1	25.6 ± 1.8	24.6 ± 0.5
Ford	555.3 ± 0.0	171.6 ± 0.0	50.8 ± 0.0	10.4 ± 1.8	0.9 ± 0.1
Agassiz	533.6 ± 4.7	437.4 ± 3.4	38.6 ± 3.0	26.4 ± 3.3	5.8 ± 0.6
Middle Fork	515.2 ± 2.3	491.6 ± 14.2	38.0 ± 2.0	12.2 ± 0.1	4.4 ± 0.3
Suspended Sediments					
Trap 1979	553.8 ± 1.3	461.2 ± 4.8	134.1 ± 11.7	35.8 ± 0.5	12.0 ± 2.5
Trap 1982	591.4 ± 5.7	468.9 ± 0.8	103.6 ± 1.0	55.1 ± 1.7	18.6 ± 3.0
Trap 1982[b]	573.8 ± 3.2	418.4 ± 2.8	248.1 ± 6.6	82.8 ± 0.2	---
Centrifuged 1982	567.8 ± 13.2	461.8 ± 11.9	179.6 ± 0.4	47.3 ± 1.7	35.8 ± 0.8

[a]based on algal assays (see text)
[b]sediment suspended in trap (see text)

sediments were very similar, the mean being 562 ± 29 ug P per g sediment. The bank sediments selected for analyses appear to have been representative samples of the sediments actually carried by the river during runoff. Similar values were obtained from sediments collected in traps on the river delta and in suspended sediments from the river. We concluded that the trap was efficient in capturing fluvial sediments, at least in terms of total and inorganic phosphorus concentrations.

Standard curves generated from the algal assays were linear over the range of standards selected (cells/ml = 2.6 x

10^7 [P] + 1.7×10^4, $r = 0.99$). Numbers of S. capricornutum grown in flasks containing sediments as their only source of phosphate peaked on day 8 or 9 of the assay (Figures 12 and 13). The assays indicated that from 1 to 36 ug of phosphorus were available per g sediment (Table 10).

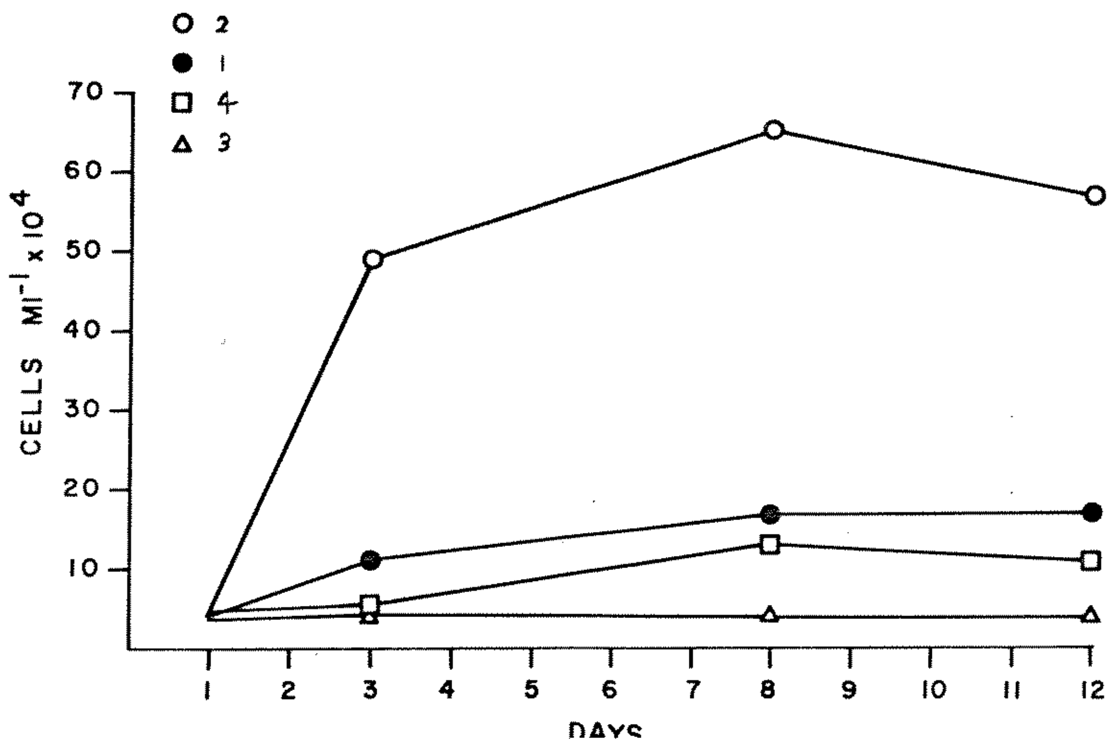

Figure 12. Growth of Selenastrum in cultures dosed with stream bank sediments from locations shown in Figure 1.

The bank sediments from the North Fork at Ford Creek (Site 3, Figure 1), which contained predominantly organic phosphorus, had the lowest availability value. Organic phosphorus compounds must be broken down by enzymatic action to liberate orthophosphate before it can be used by biota (Lund 1965). Apparently microbial mineralization of organic matter in the Ford sample was negligible. Suspended sediments from

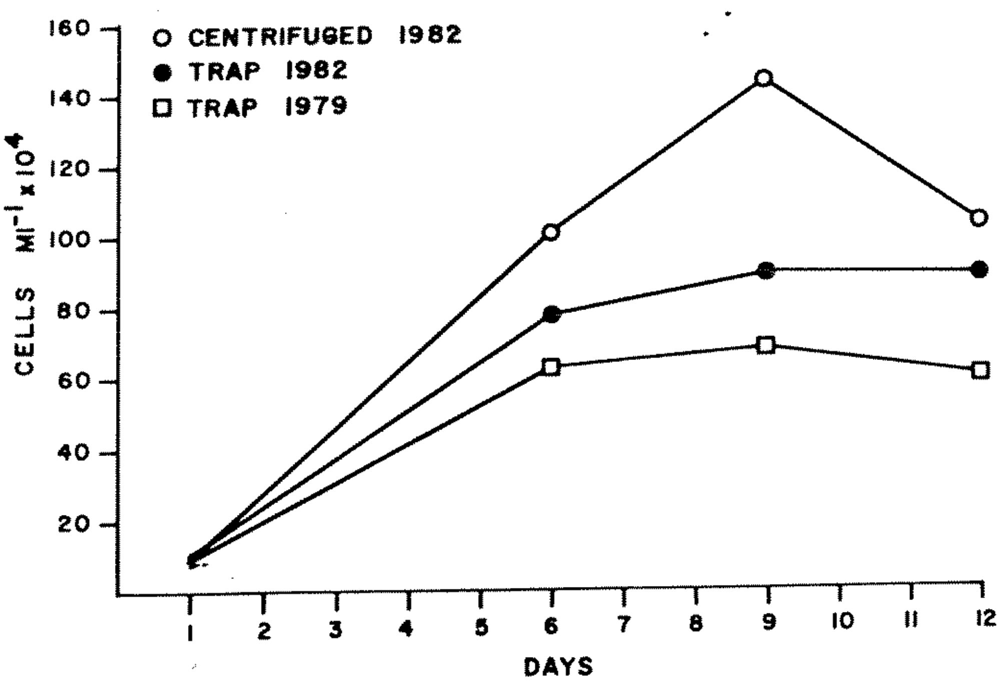

Figure 13. Growth of _Selenastrum_ in cultures dosed with sediments collected in traps placed on the Flathead River delta in 1979 and 1982 compared to growth in cultures dosed with sediments centrifuged from river samples in 1982.

the Flathead River collected by traps and by continuous centrifugation generally had greater availability values than eroding bank sediments, the exception being high available phosphorus present in bank sediments from the North Fork at Kintla Creek. Since total phosphorus values were similar for all samples (i.e. banks vs. suspended) and yet the availability values differ markedly (Table 10), we concluded that fluvial sediments were somehow "conditioned" by riverine transport to yield greater amounts of phosphorus to the algae. Also, the finer sediments collected by continuous flow centrifugation contained more available phosphorus than the larger suspended particles that settled in the sediment traps. Since the centrifugation process collects large and small particles, we were unable to determine the phosphorus availability strictly associated with fine sediments (i.e. 0.1 um to ca. 10 um). The more than 2X higher available phosphorus in the centrifuged sediments may indicate that the very fine sediments alone have very high amounts of available phosphorus. However, it is also possible that the availability of phosphorus in sediments from the trap was reduced, due to uptake and transformation to organic forms by the microbial population present during the week the trap was *in situ*.

The NaOH and nitrilotriacetic acid (NTA) extractions were employed as alternative methods for estimating phosphorus availability, because some researchers have suggested a strong correlation between these mild extractions (as opposed to strong acid extractions, like we used to determine inorganic phosphorus) and availability determined by algal assays (Cowen and Lee 1976b; Golterman 1976; Grobler and Davies 1979). We also observed a strong correlation between NTA and algal assay data across varying sediment concentrations and sources, but the absolute values were signficantly different. The details of this work were presented elsewhere, along with those of tracer studies designed to document the mechanisms of algal mobilization of sediment-phosphorus (Ellis and Stanford, in preparation). In Table 10, NTA and NaOH extraction data are presented only to demonstrate the absolute differences between these methods and the algal assay. We considered the phosphorus availability values derived from algal assays to be the most relevant to our objective of correcting the phosphorus loading rate from the Flathead River for available sediment phosphorus.

Available phosphorus comprised a low percentage of the total phosphorus, the highest being 6 percent for the riverine suspended sediments (Table 11). Cowen and Lee

Table 11. Percentage of total phosphorus in various alluvial sediments available for microbial metabolism, based on chemical extractions (NaOH and NTA) compared to algal assays.

Sediment Collection Site	Method for Estimating P-Availability		
	NaOH Extraction	NTA Extraction	Algal Assay
Stream Banks			
Agassiz	5	7	1
Kintla	4	65	4
Ford	2	9	<1
Middle Fork	2	7	1
Suspended Sediments			
Trap 1979	6	24	2
Trap 1982[a]	14	43	-
Trap 1982	9	18	3
Centrifuged 1982	8	32	6[b]

[a]Sediment suspended in trap (see text)
[b]Value used to correct phosphorus loading rate from Flathead River (see text)

(1976b) found that 23 to 45 percent of total phosphorus in suspended particulate material in urban runoff was available to S. capricornutum, but found lower percentage availability in other particulate materials: <25 percent for suspended sediments from the Genesee River, N.Y. and <6 percent for other New York rivers. Similar methods were employed in the study by Cowen and Lee (1976b) as were used here, except they used filtration to concentrate sediments. Continuous centrifugation was used here in an effort to collect the smallest particles carried by the river, which would have been lost within a filter matrix if collected by filtration. This was based on our observation that only very small particles were transported by advective circulation within the turbidity plume.

Algal assays are done under conditions of optimal light intensity, temperature, nutrient status and time that would allow the algal cells to reach their maximum population level. Thus, phosphorus availability values are often described as potentially available phosphorus. The length of time a particle is accessible to algae before it sinks out of the photic zone can limit phosphorus utilization by algae in a natural situation. However, in Flathead Lake the very fine-grained materials in the plume may travel over 40 km from the inlet to the outlet or back up the east shore in the backwash current. Thomas (in Williams et al. 1980) has shown that up to 70 percent of total sediment in some rivers is finer than 0.25 um and that Canadian rivers draining into Lake Ontario and Lake Erie have an average of about 16 percent finer than 0.25 um. Clay particles of nominal diameter of 2 um or less have a settling rate of not more than 10 cm in 8 h and thus in calm water would remain in a photic zone 5 m thick for about 16 days (Williams et al. 1980), about a week longer than our algal assays. Also, we had no way of determining if *S. capricornutum* was more or less able to mobilize sediment phosphorus than the native algae in Flathead Lake. We assumed that the latter community would be most adapted for utilization of sediment-phosphorus and that the greater percentage of fine clays in the plume would yield greater phosphorus to lake microbiota. We, therefore, used the 6 percent value (Table 11) to correct the riverine phosphorus loading rates for availability to phytoplankton. More recent study, yet incomplete (Ellis and Stanford, in preparation), utilized radioactive tracers to document phosphorus turnover in turbid lake samples; the data suggest that the bio-availability of sediment-phosphorus may be about 10 percent.

Correction of phosphorus data for bio-availability was done by applying a 6 percent correction factor to temporal phosphorus values obtained at the Flathead River inlet which were correlated with suspended solids (i.e. sediment) values greater than 10 mg/l. We concluded that, based on the significant inflection in the highly significant TSS-total phosphorus regression at ca. 10 mg/l TSS (Figure 6), flows carrying less than 10 mg/l suspended solids contained insignificant amounts of sediment phosphorus (i.e. the TSS values less than 10 mg/l contained primarily labile organic materials and all or most of the associated total phosphorus was ultimately available to phytoplankton in the lake). Therefore, phosphorus loading via the Flathead River (site A) during freshet (i.e. that period when suspended solids load was greater than 10 mg/l) was calculated by:

$[P]^c = 0.06\ ([P]^m - [P]^b) + [P]^b$, where

$[P]^c$ = total phosphorus concentration corrected for bio-availability,

0.06 = percent bio-availability of sediment phosphorus,

$[P]^m$ = measured concentration of total phosphorus, and,

$[P]^b$ = base flow average phosphorus concentration (= 8.0 ug/lP).

This relationship yielded a corrected mean for annual total phosphorus concentration of 8.6 ug/l, compared to the uncorrected value of 22.7 ug/l.

Phytoplankton Dynamics

Ellis and Stanford (1982) showed that standing crops and production of the phytoplankton community in Flathead Lake was dominated by very small species, < 50 um in size (i.e. ultra- and nannoplankton). This was true throughout our study, although the larger forms, particularly certain diatoms (e.g. *Tabellaria fenestrata* and *Asterionella formosa*), did bloom seasonally. The smaller species of green (Chlorophyta) and blue-green (Cyanophyta) algae (Table 12) contributed the largest share of the total phytoplankton standing crop throughout the year (Figure 14). In general, the green algae were abundant in spring and the blue-greens became very important during the stratified period. The abundant blue-green species (e.g. *Aphanocapsa*, *Chroococcus* and *Gomphosphaeria*) were not forms normally associated with conditions of eutrophy, although *Anabaena flos-aquae* was present in very low concentrations during late summer and fall all years. The overflow of the freshet plume was always

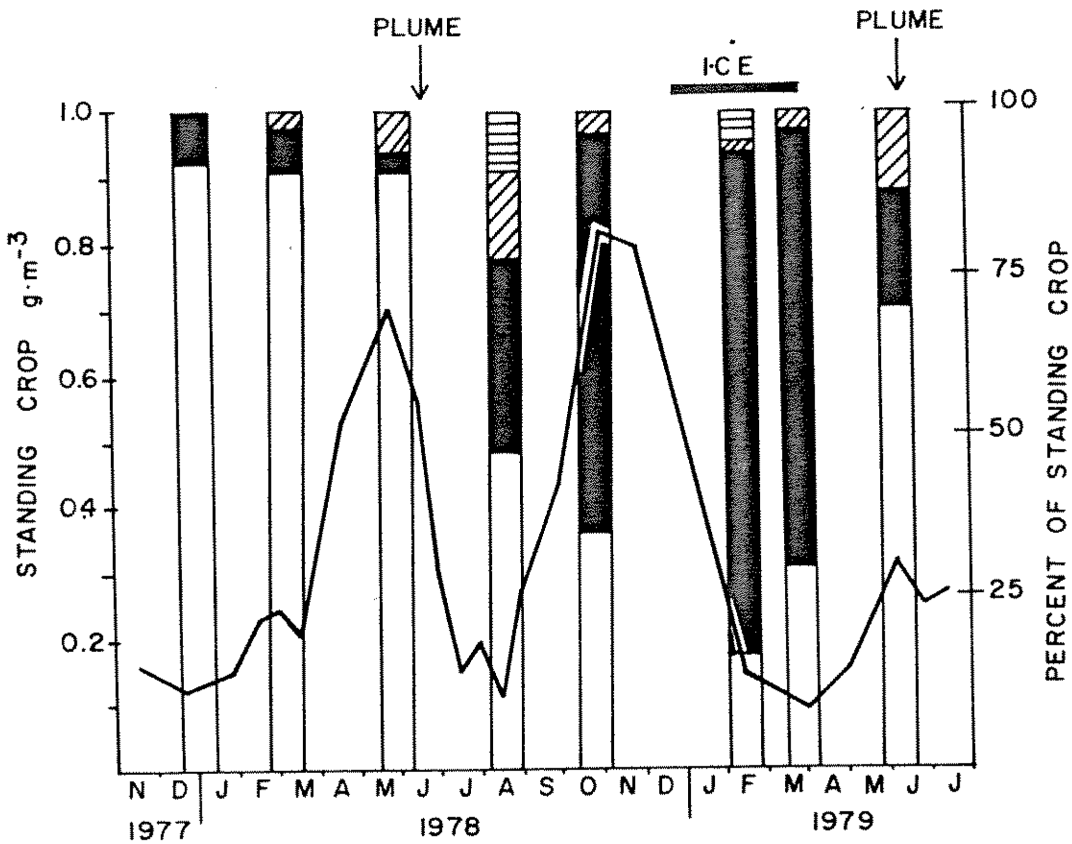

Figure 14. Temporal dynamics of the phytoplankton community at the midlake site (F), expressed as mean mass (g/m^3) in the water column (solid line). Histograms show percentage abundance of major groups: Chlorophyta (open), Cyanophyta (solid), Chrysophyta (lateral slash), and Pyrrhophyta (horizontal slash). Arrows show dates of plume overflow.

accompanied by a decrease in phytoplankton standing crops, but during low runoff years the decline was less noticeable (e.g. 1979, Figure 14). It was not completely clear why such dramatic declines in phytoplankton standing crops occurred with the onset of turbidity. We could not demonstrate any sort of flocculation process which may have stripped cells from the water column as hypothesized _a priori_ (see below). Also, recovery of standing crops was not immediate, indicating that any fertilization effect associated with advective circulation necessarily involved a variable lag period, if such an effect existed at all. The onset and maintenance of thermal stratification may have been a major factor in the observed pattern, by simply favoring the more buoyant cyanophytes in less dense, warmer waters (cf. Kalff and Knoechel 1978).

Table 12. Species and average cell size (um^3) of the most abundant phytoplankton across all seasons at the midlake site (F) during 1977-82.

Species	Size
CHLOROPHYTA	
Crucigenia quadrata	14
Diceras sp.	267
Elakatothrix gelatinosa	251
Goelenkinia radiata	100
Oocystis pusilla	1,436
Radiococcus sp.	-
Sphaerocystis sp.	14,101
Stylosphaeridium stipitatum	-
Unknown Coccoid forms	10-65
PYRRHOPHYTA	
Ceratium hirudinella	130,786
Gymnodinium sp.	8,120
CHRYSOPHYTA	
Asterionella formosa	1,411
Centrales sp.	1,735
Dinobryon setularia	1,507
Fragilaria crotonensis	1,237
Mallomonas caudata	267
Melosira sp.	1,757
Navicula sp.	150
Rhizosolenia eriensis	2,000
Synedra sp.	<400
Tabellaria fenestrata	912
CYANOPHYTA	
Anabaena flos-aquae	9
Anabaenopsis sp.	112
Aphanocapsa elachista	22,393
Chroococcus limneticus	112
Chroococcus prescottii	150
Coelosphaerium naegelianum	285,000
Gomphosphaeria lacustris var *compacta*	14,137
Oscillatoria aquatica	35
CHLOROBACTERIALEAE	
Pelogloea bacillifera	3

The pattern shown in Figure 14 for the period November, 1977 through October, 1978, recurred each year through 1982, with the exception of more complete cyanophyte dominance during the period of ice-cover in 1979. Other years the biota shifted back to dominance by Chlorophyta and diatoms in fall and winter. A major shift in this generalized pattern occurred in the summer of 1983, however. Advective circulation produced the most dense plumes at the midlake site observed since the very intense freshet turbidity overflow in 1974 (Stanford et al. unpubl.). Also, 13.4 cm of rainfall was recorded at Yellow Bay during June, more than 8 cm above average. Intense bloom conditions were recorded by mid-July, which included large standing crops of several species of green algae not previously reported in Flathead Lake: an unidentified, small-celled (1-2 um long) filamentous form dominated the community at the midlake site, and *Botryococcus* was windrowed on the west shore. Although green algae were quantitatively most abundant, the pollution algae, *Anabaena flos-aquae*, reached an alarming 24 percent of total standing crops in surface samples. Unfortunately, we did not collect chemical data during this period.

The picture of seasonal succession of phytoplankton species and biomass presented here is an overview, because a great many species were involved and it will be relevant to examine species specific biomass dynamics. These analyses will be presented elsewhere. However, we concluded that the onset of spring circulation and warming created conditions favoring small-celled algae and diatoms, which declined after (or in response to) freshet overflow in favor of small-celled cyanophytes. The pattern of succession appeared to be sensitive to the intensity of advective circulation (e.g. fertilization by turbidity overflow), but this could not be verified solely on the basis of phytoplankton dynamics.

Phytoplankton Primary Productivity

Time-series plot of primary productivity values obtained at the midlake site (F) (Figure 15) confirmed that the peaks in phytoplankton standing crops observed above (Figure 14) were preceeded by periods of high-rate carbon fixation. The highest rates (e.g. 50-85 $gCm^{-2}day^{-1}$) of primary productivity consistently occurred immediately after overflow of freshet turbidity, contrary to our *a priori* expectations. Ellis and Stanford (1982) showed that ultraplankton (i.e. < 10 um in size) contributed as much as 65-75 percent of the total primary productivity in late summer and fall, 1978. A single plankter, *Pelogloea bacillifera*, was responsible for over 75 percent of the species-specific primary productivity during the post plume period and into fall. *Pelogloea bacillifera* is an extremely small species (mean cell volume < 3 um^3) and was probably mistaken for bacteria in the Utermohl analysis of standing crops during early years of the study (1977-80). Reexamination of 1982-83 crop dynamics indicated that *Pelogloea* was extremely abundant in post-plume populations. We concluded that the post-plume spikes in production were due to *Pelogloea* and a variety of the smaller Cyanophyta.

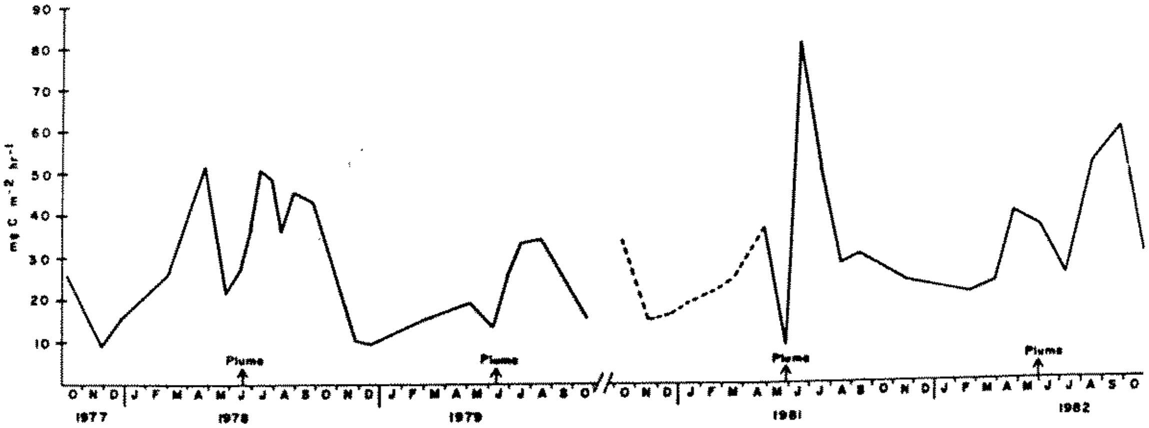

Figure 15. Temporal dynamics of primary productivity in pelagic areas of Flathead Lake (site F), 1977-1982. Dates of turbidity plume overflow are shown by arrows.

The absolute value of the post-plume productivity increase each year was clearly associated with the intensity of advective circulation. Years of high turbidity at midlake (1978, 81, 82, 83) were associated with maximum primary productivity (Figure 15). This was supported (but only circumstantially) by a multiple regression analysis of primary productivity as a function of various physicochemical variables; water clarity (and, hence, phosphorus concentration) explained about 40% of the variance. The spring bloom was probably related to water column circulation and onset of the warming trend and is a typical occurrence in temperate lakes (cf. Wetzel 1983). Productivity was depressed in 1979, apparently due to cooler conditions in the epilimnion as a result of the preceding cold winter and below normal freshet.

Concerning the temporal plot of standing crops as discussed above, we have no direct evidence of why the production rates dropped so dramatically in concert with turbidity overflow. We assumed that light limitation and species-specific adaptations for mobilization of sediment phosphorus (and possibly nitrogen) elicited intense competition within the phytoplankton community. The fertilization effect indicated by sediment bioassays, in combination with conditions of summer stratification, apparently favored *Pelogloea* and the small Cyanophytes, although in 1983, larger species (e.g. the unidentified filamentous green algae and *Anabaena flos-aquae*) probably contributed significantly to primary productivity. Our estimates of annual primary productivity reflected these trends (Table 13).

Table 13. Annual rates of primary productivity measured at the midlake site.

Water Year	$gC\ m^{-2}\ yr^{-1}$	$gC\ m^{-3}\ yr^{-1}$
1978	152	4.6
1979	98	3.6
1981	145	5.6
1982	152	5.2
	mean [1] = 137 ± 70	mean = $4.8 \pm .1$

[1] mean $\pm$ 95% confidence limit for monthly data (n=48)

The rates of primary productivity measured at midlake (sites F and G) were compared to other areas of the lake (sites H-N, Figure 2) in several synoptic studies. In October, 1979 and April, 1980, the bay sites were located well off shore (i.e. greater than 1 km) and no statistically significant values for bays compared to midlake were obtained (Mann-Whitney test, $P >> .05$), causing us to conclude that productivity in the pelagic areas of the lake, including deeper areas of the bays, was relatively uniform. In subsequent synoptics, we moved close to shore in the bay areas and the resultant values for these littoral areas were significantly different in comparison to mid-lake (pelagic) (Tables 14 and 15). We concluded that the littoral areas in the bays were more productive, due to wind-induced resuspension of nutrients and less water volume for dilution of creek and groundwater inflows. We measured high nutrient loading in Dayton Creek (site E) (i.e. mean concentrations of N and P

Table 14. Primary productivity (mg C m^{-3} day^{-1}) in Flathead Lake bays compared to midlake (F), as calculated in photic zone and upper 10m; 11/IX/81.

Bay	Depth of Photic Zone(m)	Primary Production: Photic Zone	Primary Production: 10m Column
Somers (I)	20	16.2	18.3
Skidoo (L)	20	19.2	19.2
Yellow (K)	20	24.9	28.5
Bigfork (H)	10	28.9	28.9
Woods (N)	20	31.2	32.6
Dayton (J)	10	33.4	33.4
Polson (M)	5	61.8	---
Midlake Deep	30	13.1	8.0

Table 15. Primary productivity (mg C m^{-3} day^{-1}) in Flathead Lake bays compared to midlake (F), as calculated in photic zone and upper 10m; 23/IV/82.

Bay	Depth of Photic Zone(m)	Primary Production: Photic Zone	Primary Production: 10m Column
Somers (I)	10	33.3	33.3
Skidoo (L)	20	36.3	32.1
Yellow (K)	20	25.9	16.9
Bigfork (H)	20	28.7	30.9
Woods (N)	20	31.7	28.9
Dayton (J)	10	27.8	27.8
Polson (M)	2.5	30.6	---
Midlake Deep	30	18.6	17.0

were 240 and 26 ug/l) and frequently observed resuspension of delta sediments after storms in the Sommers Bay area (site I). Also, shoreline breakouts of septic leachates possibly coupled with localized natural flows of low-volume groundwaters, were thought to occur in Yellow Bay (site K), Skidoo Bay (site L), and Sommers Bay (site I). Site M in Polson Bay was located well out of the riverine outflow current and our data probably reflected the naturally more eutrophic conditions of a shallow lake in that area.

Carbon Dynamics

Suspended solids (seston) carried by the Flathead River consisted of silt and clay particles and variable amounts of particulate organic matter. During high flows, riverine seston was composed primarily (>99%) of inorganic materials, whereas, in baseflow conditions organic matter was more important (i.e. 20-50% of the seston was organic). Particulate organic carbon (POC) concentrations were significantly higher during freshet (.5 - 1.6mg/l) than at baseflow (.05 - .30 mg/l. Therefore, the freshet plume contained both inorganic and organic materials as it moved over the lake surface each year.

Since we were concerned about an apparent fertilization effect from freshet overflow in the lake and wanted to know if allochthonous organic materials were labile or were carried to the bottom with the freshet sediments, we attempted to separate the seston crop in the water column at midlake into inorganic and organic components, and further separate the POC into living and dead portions in time-series. This work was limited to the 1978 and 1979 water years only, because it was extremely labor intensive.

We have previously pointed out that attempts to accurately measure suspended solids in the water column were unsuccessful, because the freshet plume was considerably diluted by the time it reached midlake. Similar problems were encountered in interpreting organic carbon data (Figure 16), because a large share was produced autochthonously via phytoplankton primary productivity. In general, POC levels were not elevated in concert with plume overflow and we concluded that POC dynamics generally reflected phytoplankton dynamics. Also, there was no consistent evidence of POC accumulation with depth (Figure 17), providing no support for our a priori idea that organic materials were stripped from the water column by clay particles as they settled to the lake bottom. Rather, Figure 17 suggested that POC was oxidized as it passed through the tropholytic portion of the water column.

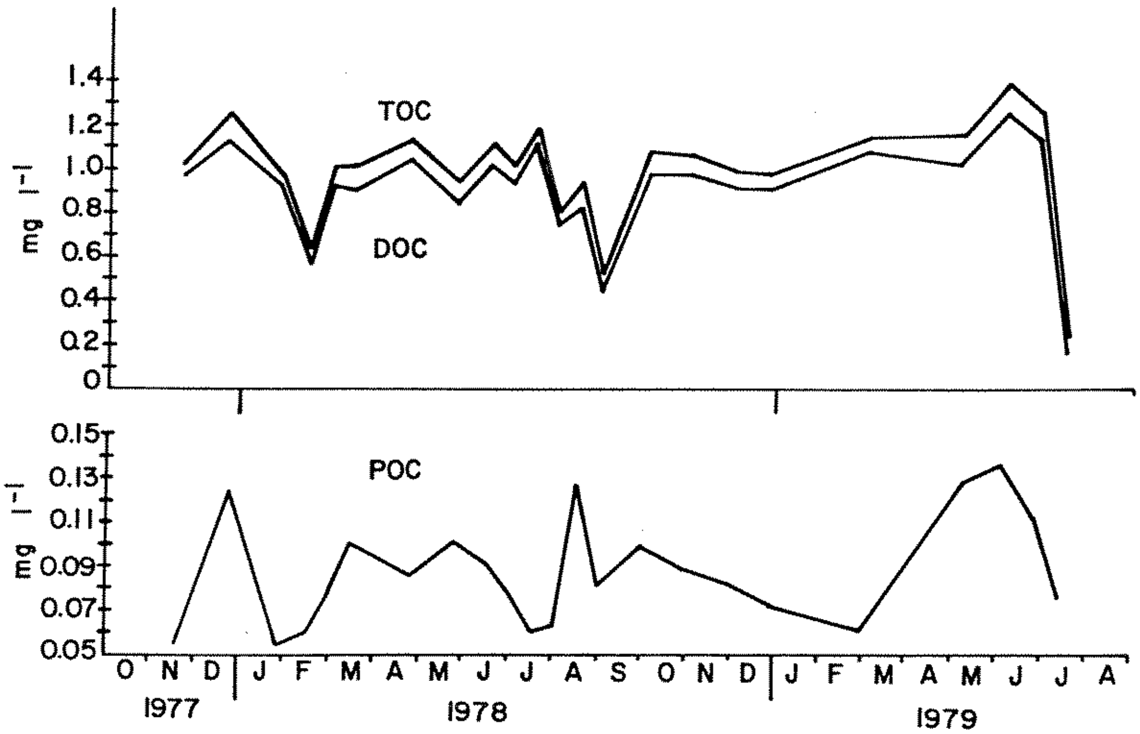

Figure 16. Temporal dynamics of total (TOC) and dissolved (DOC) and particulate (POC) organic carbon concentrations in pelagic area of Flathead Lake (site F). Data are means for the 110 m water column.

This was supported by plotting of depth profiles of adenosine triphosphate (ATP) concentrations in seston (Figure 18). Biomass (i.e. living POC determined from ATP assays, Boswell et al. 1980; Karl 1980) was concentrated primarily in the photic zone (above 40 m), which indicated that, in general, little labile material remained in the bottom half of the water column.

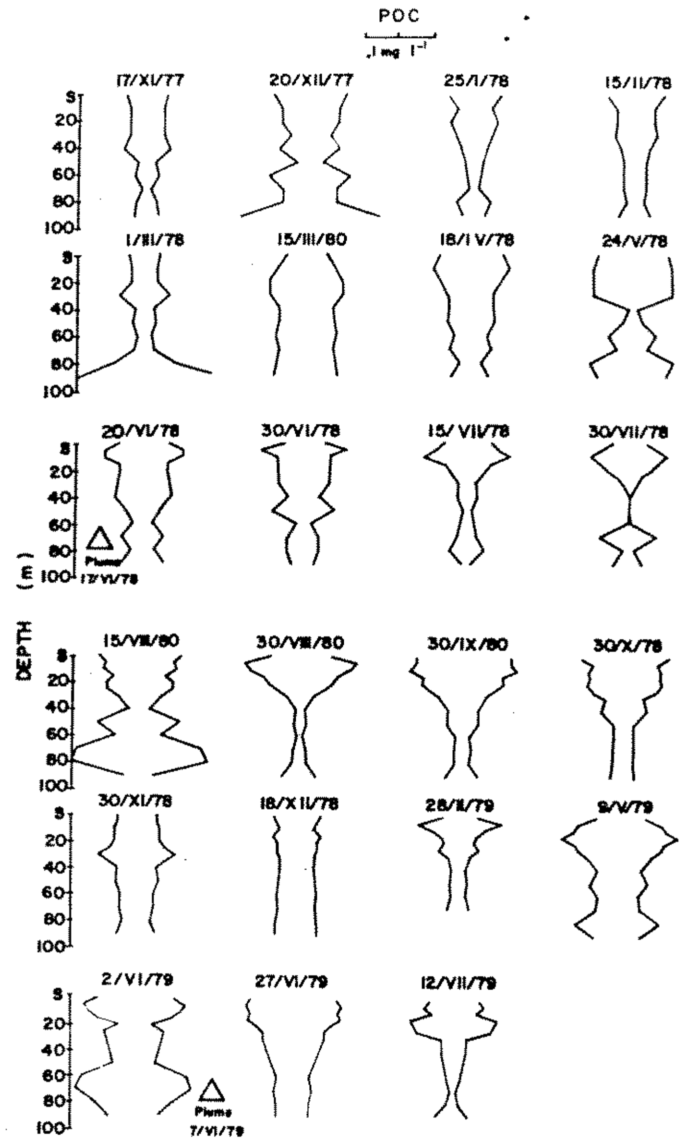

Figure 17. Depth profiles of particulate organic carbon (POC) concentrations in pelagic area (site F) of Flathead Lake, 1977-1979. Triangles indicate dates of turbidity overflow.

The fertilization effect, indicated by sediment bioassays and primary productivity dynamics discussed above, was apparent when the POC data at midlake were related to ATP concentrations in the seston crops over time (Figure 19).

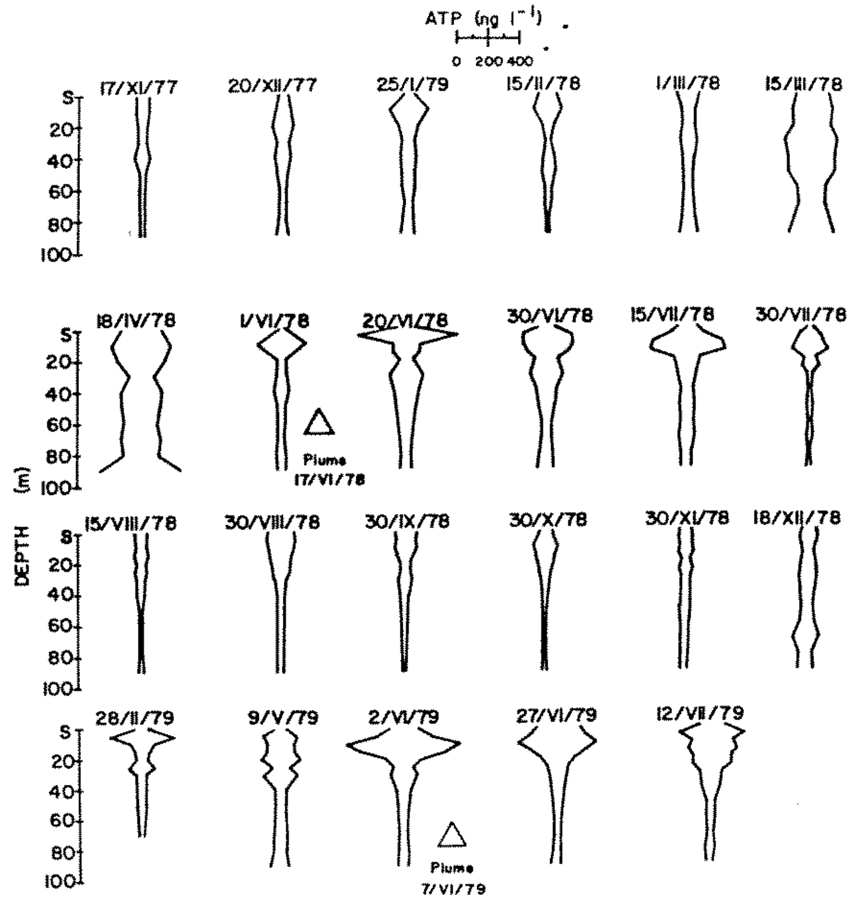

Figure 18. Depth profiles of microbial biomass, expressed in ng/l of adenosine triphosphate, 1977-1979. Triangles indicate dates of turbidity overflow.

Planktonic biomass dynamics in Figure 19 were significantly ($P < .05$) correlated with primary production data for the same time period. We concluded that the major effect of turbidity overflow at midlake was introduction of nutrients associated with sediments which stimulated phytoplankton production. Particulate organic substrates for heterotrophic utilization were apparently limited to very small sizes (i.e. < 10 um) or involved dissolved organic components (see Ellis and Stanford 1982).

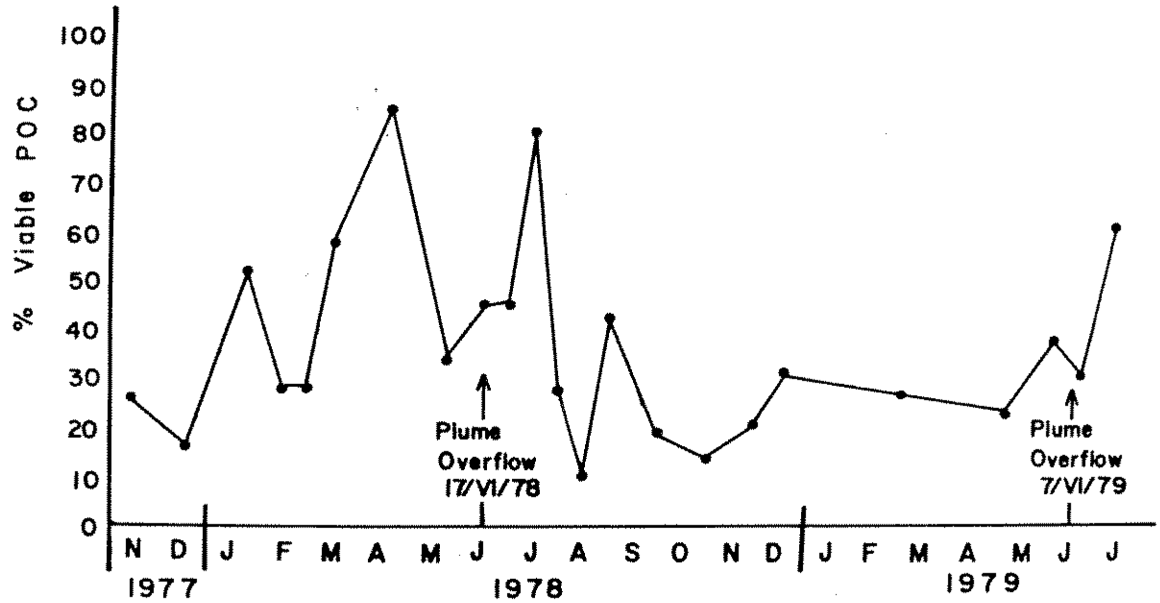

Figure 19. Temporal dynamics of microbial biomass expressed as percent of particulate organic carbon concentrations, 1977-1979, at midlake site (F). Data are water column means.

This implied that particulate organic material in Flathead Lake is extremely labile. The quantitative relationships between dissolved carbon and particulate materials in the water column were demonstrated to scale in Figure 20. These relationships identified several important attributes of Flathead Lake. Most obvious was the 20:1:.1 ratio of inorganic carbon:DOC:POC, which is fairly typical of deep, temperate lakes (Wetzel 1983). About one-third of the POC was living (i.e. viable organic carbon, VOC); the remaining POC mass was very labile organic detritus. This large reservoir of detrital POC relative to VOC concentrations supports the contention that lakes are essentially detritus-based systems (Wetzel 1983). On a mass balance basis, we estimated that 56% of the annual total organic carbon pool was derived from allochthonous sources (including POC from advective circulation), while 44% was produced in the lake by phytoplankton. (The autochthonous contribution from rooted plants and periphyton was assumed to be nil; we conducted 21 transects with SCUBA and determined that macrophytes and aufwuchs were virtually non-existent below the 25 m contour.) The seston was about 12% organic (POC/TSS). Thirty-two percent of the POC was viable (VOC/POC) and, if based on phytoplankton alone (i.e. excluding heterotrophic fixation), the VOC turned over (i.e. completely regenerated) in less than 3 days (VOC/PP). However, Ellis and Stanford (1982) demonstrated that light-mediated heterotrophy (photoheterotrophy) by planktonic bacteria averaged 365% of dark bottle uptake (chemoheterotrophy), which strongly suggested a close coupling of autotrophic and heterotrophic pathways in plankters < 10 um in size. Ultraplankton were likely responsible for most of the primary productivity in the present study and turnover of the VOC may have been much shorter than indicated by Figure 20. However, should future measurements demonstrate substantial deviation from the basic structure of Figure 20, it must be concluded that major watershed perturbations have altered key (e.g. total phosphorus, quantity and lability of organic detritus, etc.) loading rates controlling carbon dynamics.

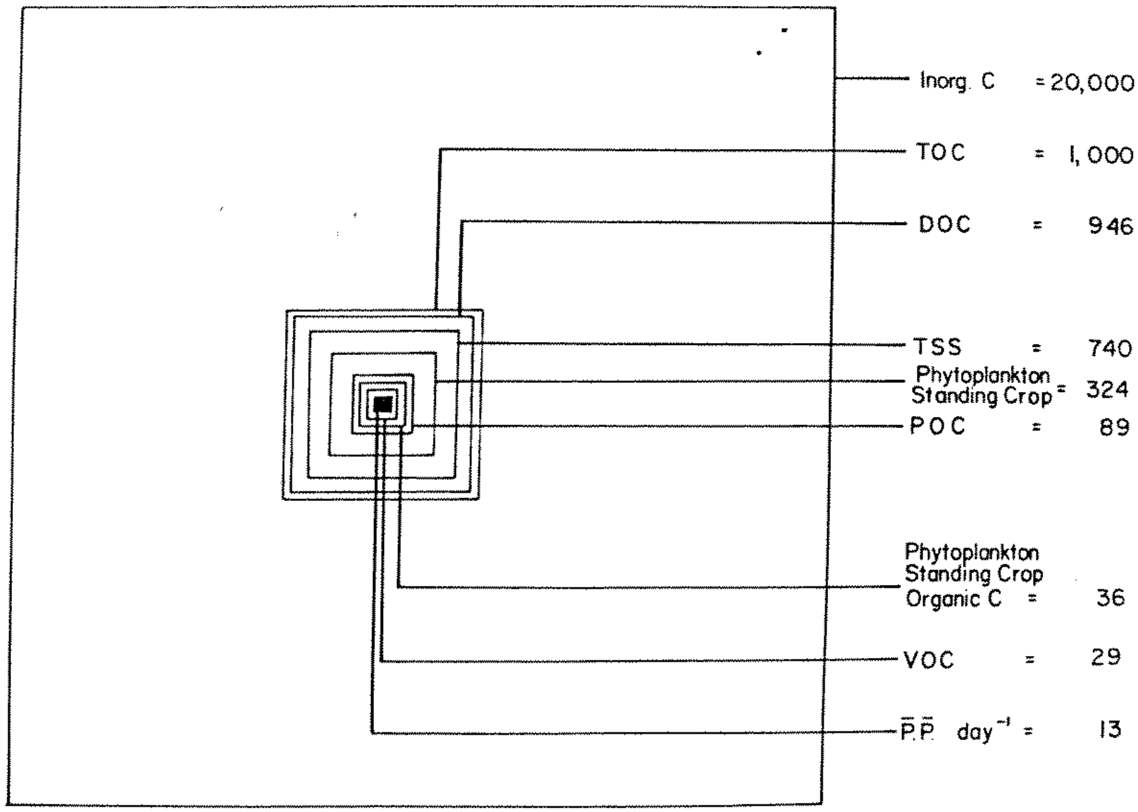

Figure 20. Comparative mass of components of lake seston and organic carbon expressed in ug C/l. Biomass (VOC) was converted to carbon using ATP:C of 1:250 (Boswell et al. 1980).

Lake Trophic Status

That chemical characteristics and productivity in lakes are largely controlled by geochemistry and hydrology of associative drainage basins was first proposed on an inter-regional basis by Naumann (1932) and has become a unifying concept in limnology (cf. Wetzel 1983). The basic theme of lake typology is that trophic components (e.g. bacteria, phytoplankton, zooplankton, fish) dynamically interact within constraints imposed by morphometry of the basin and the laws of thermodynamics, to utilize a given nutrient load from the water- and airsheds. Natural or unnatural acceleration of nutrient loading, which in most systems translates to accelerated phosphorus loading (Schindler 1977), almost immediately changes the lake's internal nutrient cycling processes by altering transfer rates and, hence, the size of each trophic compartment (Hutchinson 1973; Vollenweider and Kerekes 1980). The trophic status (i.e. ranking on a scale from ultra-oligotrophic to hypereutrophic) is a fundamental and encompassing description, which permits comparisons (typology) of lake ecosystems (the lake and its drainage basin).

Flathead Lake has classically been thought of as oligotrophic, although concerns about abnormal nutrient loading were expressed 20 years ago (Gaufin et al. 1976). Our data placed the lake closer to mesotrophy than oligotrophy (Table 16) and we were concerned that our data reflected symptoms of changing trophic status toward eutrophy.

The phosphorus load, for example, was relatively high for a system draining large areas of relatively pristine country. Although the Flathead Rivers carried a high volume of phosphorus-rich sediments into the lake during freshet, we showed that most of the sediment-phosphorus was unavailable to the lake's biota. However, the freshet plume did elicit dramatic responses in the phytoplankton community and we concluded that, if substantial amounts of deforestation and other land disturbances in some portions of the drainage basin (e.g. the North Fork) have in recent years increased sediment loading (due to more erosive freshets caused by

Table 16. Characteristics of trophic status compared to mean values in Flathead Lake, 1977-1982 (modified from Vollenweider 1969, 1975; Wetzel 1983).

Trophic Status	Mean Primary Productivity ($mg\ C\ m^{-2}\ yr^{-1}$)	Phytoplankton Density ($cm^3\ m^{-3}$)	Phytoplankton Biomass ($mg\ C\ m^{-3}$)	Chlorophyll a ($mg\ m^{-3}$)	Dominant Phytoplankton
Ultra-Oligotrophic	<50	<1	<50	0.01-0.5	Chrysophyceae,
Oligotrophic	50-300	1-1.5	20-100	0.3 -4.5	Cryptophyceae, Dinophyceae,
Oligo-mesotrophic	100-500	1-3	50-150	1.0 - 10	Bacillariophyceae
Mesotrophic	250-1000	2-3	100-300	2 - 15	Bacillariophyceae,
Meso-eutrophic		3-5		7 - 10	Chlorophyceae,
Eutrophic	>1000	5-10	>500	10 -500	Cyanophyceae, Euglenophyceae
Hypereutrophic	>1500	>10	<50-200	>500	
FLATHEAD LAKE	137	3.4	325	—	Chlorophyceae, Cyanophyceae

Trophic Status	Light Extinction Coefficients ($\eta\ m^{-1}$)	Total Organic Carbon ($mg\ \ell^{-1}$)	Total P ($\mu g\ \ell^{-1}$)	Total N ($\mu g\ \ell^{-1}$)	Total Inorganic Solids ($mg\ \ell^{-1}$)
Ultra-Oligotrophic	0.03-0.8	<1	<1-5	<1-250	2-15
Oligotrophic	0.05-1.0	<1-3	8	660	10-100
Oligo-mesotrophic	0.05-1.5	1-5	5-10	250-600	10-200
Mesotrophic	0.1 -2.0	<1-5	27	750	—
Meso-eutrophic	.1 -3.0	5-20	10-30	500-1100	100-500
Eutrophic	0.5 -4.0	5-30	30-100	1875	—
Hypereutrophic	1.0 -4.0	5-30	>100	500->15000	400-60000
FLATHEAD LAKE	1.2	1.0	7.2	1112	~ 100

higher snow accumulations in clear-cut areas), an unknown portion of the observed sediment enrichment could be considered an anthropogenic nutrient subsidy. Also, the various sewage treatment facilities within the urban areas (Figure 1) in the drainage basin were known to continually discharge directly into the river-lake system. These sources accounted for 17.4 percent of the phosphorus load reaching Flathead Lake annually (Table 17). Non-point sources of phosphorus were not directly quantified, but the Stillwater River (including the Whitefish River), which drains most of the agricultural areas (see Figure 1), carried 10 percent of the total annual phosphorus load (Stanford 1984). We, therefore, estimated that at least 20 percent of the phosphorus entering Flathead Lake each year was of anthropogenic origin.

Table 17. Mass (MT) of nitrogen and phosphorus discharged annually into Flathead lake from the 5 sewage treatment facilities within the drainage basin, 1978-1982.

Source	Nitrogen	Phosphorus
Kalispell	48.33	11.71
Whitefish	20.38	5.00
Columbia Falls	5.30	2.70
Bigfork	5.62	1.10
UM Biological Station	0.07	0.02
Total	79.7	20.53

Origins of nitrogen forms were less clearly segregated. About 8 percent of the lake's nitrogen budget was contributed from sewage treatment facilities. The remaining amount was derived primarily from the Flathead River and bulk precipitation. However, nitrate discharge from the fluvial aquifer in the Evergreen area into the Flathead River was evident during the 1981 freshet. Multi-spectral imagery (remote sensing) from high altitude aircraft demonstrated inflow of cold waters in the Flathead River near Evergreen. These locations were also areas of high (.5-5 mg l^{-1}) nitrate concentrations in the imagery map generated for nitrates (i.e. data trends on the nitrate and temperature maps overlapped at specific areas in the Flathead River channel, see Mace, 1981). Apparently, waters containing high concentrations of nitrate from septic systems were flushed out of the aquifer and into the river, but well upstream from the lake. No evidence of shoreline inputs of groundwater existed in the remote sensing exercise. The groundwater source of nitrate may have been responsible for the high nitrate values present in the freshet flows from the Flathead River and, possibly, in the resultant overflow plumes.

Consequences of anthropogenic nutrient loading on lake trophic status were based on our prepossession that all of the anthropogenic phosphorus and nitrogen reached Flathead Lake in bio-available form. It is quite likely that these foreign nutrients enter biophysical cycles within the Flathead River before it discharges into the lake. For example, phosphorus loading in the river near Kalispell may influence phosphorus sorption gradients involving river sediments eventually transported into the lake. Clearly, the relationships between anthropogenic nutrient loading and complex analytical relationships between microbial mobilization and sediment sorption of nutrients needs further study.

Unfortunately, no records were available from which to make long term comparisons of Flathead Lake microbial communities, particularly algae, except that qualitative descriptions indicated no noticeable growths of aufwuchs on shoreline substrata until recent years. Benthic algae, such as <u>Gomphonema</u> sp., <u>Ulothrix</u> <u>zonata</u> and <u>Mougeotia</u> sp. formed conspicuous mats on the littoral rocks and to about the 20 m depth contour lakewide, below which algae were apparently limited by presence of unstable mud bottom. Coupled with our phytoplankton data, these observations led us to conclude that the algae communities in general were more indicative of mesotrophic than oligotrophic conditions. However, the dominance of small species of Cyanophyta in Flathead Lake was probably less indicative of trophic status than one might conclude from Table 16. Rather, these forms have probably been overlooked because of their small size and bacteria-like appearance (Kalff 1972; Ellis and Stanford 1982). The presence of Anabaena flos-aquae and nuisance blooms of chlorophyte plankton and aufwuchs suggested to us that chronic degredation of water quality was in progress and that it could be explained by abnormally accelerated nutrient loading.

Based on these considerations and measured values for the various characteristics of lake trophic status (Table 16), we labeled Flathead Lake as oligo-mesotrophic. Prior to human settlement, the lake was probably well within the constraints of oligotrophy or even may have been ultra-oligotrophic. Due to cultural eutrophication, we concluded that accelerated nutrient (phosphorus) loading had altered the internal transfer rates and, thus, enhanced the autotrophic compartments of the food chain. The impact of eutrophication on higher-level consumers (e.g. zooplankton and fish) was less clear from this perspective, because these communities have been dramatically affected by introduction of exotic species. For example, a formerly very abundant and native planktavore, Salmo clarki, was largely replaced by introduced kokanee salmon, Onchorynchus nerka and the nonnative lake trout, Salvelinus namycusch, has become the dominant piscivore. Perhaps very significantly, the opossum shrimp, Mysis relicta, established a lakewide population by 1981 from introduced populations in upstream lakes (Bukantis and Stanford, in preparation).

Modelling the Eutrophication Process

In recent years a number of empirical models have been used to predict lake phosphorus concentrations from data on annual phosphorus loading rates, lake morphometry, and hydraulic flushing time (Vollenweider 1969, 1975; Dillon and Rigler 1974a; Kirchner and Dillon 1975; Chapra 1975; Jones and Bachmann 1976; Larsen and Mercier 1976; Reckhow 1977, 1979). Similar empirical models have been developed to utilize phosphorus concentration data to predict lake trophic status from algae chlorophyll a concentrations (Dillon and Rigler 1974b; Jones and Bachmann 1976), water clarity (Bachmann and Jones 1974; Dillon and Rigler 1975), and primary productivity as measured by the $H^{14}CO^{3}$ light and dark bottle method (Vollenweider and Kerekes 1980). Some of these models were based on data from only a few lakes within a localized area (e.g. Vollenweider 1975), others have been examined and reformulated using large data sets involving lakes of widely differing trophic states (see Rast and Lee 1978; Vollenweider and Kerekes 1980).

All of the analytical models were derived from a general model proposed by Vollenweider (1969):

$$TP = \frac{L}{Z(r + p)}$$

where

- TP = concentration of total phosphorus in the lake water, mg m^{-3};
- L = annual phosphorus loading per unit of lake surface area, mg m^{-2} yr^{-1};
- Z = mean depth of the lake, m;
- r = phosphorus sedimentation coefficient, yr^{-1}; and,
- p = hydraulic flushing rate of the lake, yr^{-1}.

The major problem in verifying the Vollenweider phosphorus loading model has been in accurately estimating the rate of phosphorus loss to the lake bottom sediments (i.e. estimating r). Different approaches to estimating r have resulted in the differing formulations of the above equation (Canfield and Bachmann 1981). Some researchers have reformulated the Vollenweider equation and utilized a phosphorus retention coefficient (i.e. input, minus output, divided by input) (Dillon and Rigler 1974a), because the phosphorus sedimentation coefficient is very difficult to measure. This is particularly true in lakes and reservoirs receiving substantial phosphorus loading in association with riverine sediments (Canfield and Bachmann 1981). Thus, the model becomes:

$$TP = \frac{L(1-R)}{qs}$$

where

- qs = annual water loading (lake outflow/lake surface area), m yr^{-1}; and,
- R = phosphorus retention coefficient.

The more recent model reformulations have utilized a volumetric phosphorus loading rate $[P]^i$; (input phosphorus tonnage/inflowing water volume, mg m^{-3} yr^{-1}), rather than the areal loading rate (L). Vollenweider and Kerekes (1980) used a volumetric reformulation, which expressed phosphorus loading as $[P]^i/(1 + \ p\ )$ mg m^{-3}. Using a very large set of data for North American and European lakes (the OECD Programme) they were able to accurately predict primary productivity from this loading factor. Rast and Lee (1978) used the same loading factor to predict chlorophyll a concentration.

An alternative approach to prediction of lake trophic status involves use of lake simulation models in which primary biotic and abiotic components are compartmentalized and intercompartment transfer rates of phosphorus are manipulated mathematically in order to predict lake phosphorus concentration. These models have the advantage of permitting close scrutinization of phosphorus flow and the outcome of changes in loading rates, but are very time consuming and often are based on weak or oversimplified assumptions.

We used both approaches in an attempt to verify and amplify conclusions reached above and to more accurately evaluate the apparent decline of Flathead Lake water quality in response to accelerated phosphorus and nitrogen loading. We wished also to examine our loading rates as corrected for sediment-phosphorus availability (L^c or $[P^c]^i$ versus uncorrected (L^o or $[P^o]^i$).

Table 18. Prediction of total phosphorus, TP, in Flathead Lake from various analytical models using loading rates corrected, L_c or $[P_c]_i$ and uncorrected, L_o or $[P_o]_i$, for sediment phosphorus availability.

	Model				
	Vollenweider (1969)[a]	Dillon and Rigler (1974a)[b]	Vollenweider (1975)[c]	Jones and Bachmann (1976)[d]	Larsen and Mercier (1976)[e]
corrected, TP	1.9	5.2	7.2	3.6	5.7
uncorrected, TP	3.7	5.4	14.9	7.4	5.8

[a] $TP = \frac{L}{Z(r + p)}$
using r = .330 and .460 (Table 19)

[b] $TP = \frac{Lp}{Z} (1 - R)$

[c] $TP = \frac{L}{(10 + Z/p)}$

[d] $TP = \frac{0.84L}{Z(0.65 + 1/p)}$

[e] $TP = [P]_i (1 - R)$

Flathead Lake data used:

- mean areal loading, L,
  - $L_o = 487$ mg m^{-2}
  - $L_c = 237$ mg m^{-2}
- mean hydraulic loading, $[P]_i$,
  - $[P_o]_i = 23.26$ mg m^{-3}
  - $[P_c]_i = 11.35$ mg m^{-3}
- sedimentation coefficient, R,
  - $R_o = .75$
  - $R_c = .25$
- Z = 50 m; p = 2.2 yrs.

The various empirical models consistently underestimated the measured total phosphorus concentration (7.2 mg m^{-3}) in Flathead Lake, except for the Vollenweider (1975) equation (Table 18). We expected that use of corrected versus uncorrected loading rates might yield under- and overestimates, respectively. The consistent underestimate apparently was related to poor approximation of the phosphorus sedimentation rate in Flathead Lake, whether we used coefficients derived for natural or artificial lakes (Table 19, see also Canfield and Bachmann 1981; Mueller 1982). Inexplicably, the Vollenweider (1975) estimation of r, which is based on a small sampling of Canadian shield lakes, seemed to work perfectly for Flathead Lake. The simulation model (Henry and Stanford 1983) also underestimated total phosphorus concentration (5.7 mg m^{-3}), but was well within the confidence ranges predicted by the analytical models. Both the empirical and simulation models confirmed that lake total phosphorus concentration was dependent on external phosphorus loading and the phosphorus sedimentation rate. However, sedimentation of phosphorus in Flathead lake more likely involved one volumetric relationship associated with freshet advective circulation during April-July, and a second sedimentation rate associated with internal cycles unaffected by riverine turbidity (i.e. August through March). A more realistic model, which incorporates the differential effect of freshet turbidity was, therefore, derived from Canfield and Bachmann (1981):

$$TP = \frac{f(Lc)}{Z((a(Lc/Z)^{b} + C(Lc/Z)^{d}) + P)}$$

where

- f = fraction of sediment-phosphorus lost upon initial rapid sedimentation of freshet plume;
- $a(Lc/Z)^{b}$ = sedimentation coefficient that varies with volumetric loading during freshet; and
- $c(Lc/Z)^{d}$ = sedimentation coefficient that varies with volumetric loading during base flow in the Flathead River.

These sedimentation coefficients remain to be quantified on a lakewide basis.

Table 19. Derivation of phosphorus sedimentation coefficient, r, for Flathead Lake using loading rates, corrected, L_c, and uncorrected, L_o, for sediment phosphorus availability.

	Calculation Basis		
	Emperical[a]	Natural Lakes[b]	Artificial Lakes[c]
Corrected, r_c	-1.52	.330	.285
Uncorrected, r_o	-0.85	.460	.437

[a]re-arrangement of terms of the Vollenweider (1969) equation:

$$TP = \frac{L}{Z(r + P)}, \quad r = \frac{L/Z}{TP} - p$$ (Jones and Bachmann 1976).

[b]$r = 0.162\ (L/Z)^{0.458}$ (Canfield and Bachmann 1981)

[c]$r = 0.114\ (L/Z)^{0.589}$ (Canfield and Bachmann 1981)

Our long term primary productivity estimate (137 g C m^{-2} yr^{-1}) was well within the 95% confidence limits for the relationship between primary productivity and volumetric phosphorus loading described by Vollenweider and Kerekes (1980) (Figure 21). The uncorrected (for bio-availability) rate (Lo) fit the emperical model (Figure 21) better than did the corrected (Lc) rate. These data led us to conclude that our corrected loading rates may have been low (i.e. our algal assays underestimated *in situ*, biotic mobilization of sediment-phosphorus) and that the real value lay between the corrected and uncorrected points in Figure 21. It may also be important to point out that our primary production estimates were high, due to the fact that all *in situ* measure-

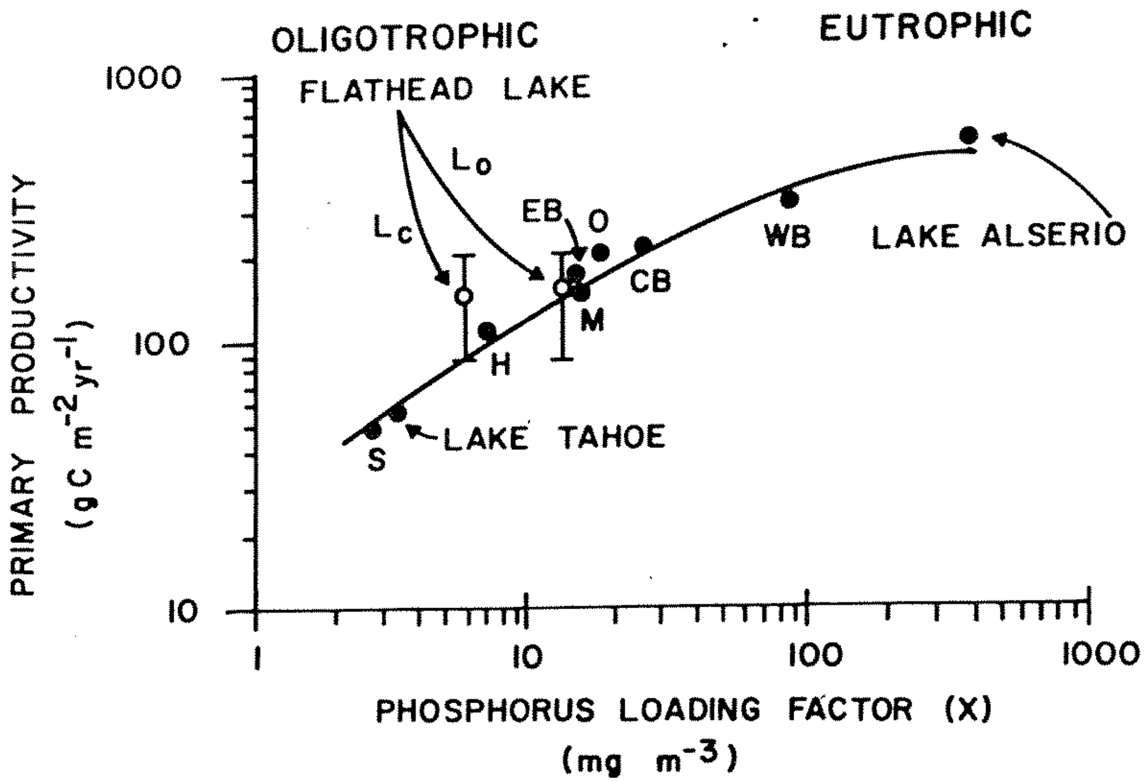

Figure 21. Observed fit of Flathead Lake data (Lc, corrected for sediment-phosphorus availability; Lo, uncorrected; bars are 95% confidence intervals) to the Vollenweider and Kerekes (1980) eutrophication model for predicting primary production from a phosphorus loading factor, $(x) = [P]^{1} / (1 + TW)$. The prediction curve included data (closed circles) for the Laurentian Great Lakes (Superior, S; Huron, H; Michigan, M; Ontario, O; the eastern, EB, central, CB, and western, WB, of Erie), Lake Tahoe, California-Nevada, and a very eutrophic European lake, Alserio. The curve is fitted by:

$$F(x) = \frac{x^{.76}}{.29 + .011\ x^{.76}}$$

from Vollenweider and Kerekes (1980).

ments of primary productivity were made on cloudless, calm days. This likely produced an overestimate, since cloudy days and rough water conditions undoubtedly reduced the rate of primary productivity on the majority of days throughout the years.

A more relevant question, however, concerns the consequences of accelerated loading of phosphorus from cultural sources, particularly point discharges of sewage. Rast and Lee (1978) and Vollenweider and Kerekes (1980) showed that loading rates above 10 in Figure 21 were likely to produce chronic symptoms of impending eutrophication and that rates above 25 resulted in grossly polluted conditions. They referred to these volumetric loading limits as "permissible" and "excessive". In other words, lakes progressively approaching a loading factor of 10 were judged to be shifting from oligotrophic to mesotrophic or eutrophic conditions. We concluded that indeed Flathead Lake was in the process of deteriorating toward eutrophy. All of the lakes to the right of Flathead Lake in Figure 21 were known to experience nuisance blooms of algae. Since at least 20 percent of the phosphorus load in Flathead lake is derived from anthropogenic sources, we concluded that the lake has progressed from an ultra-oligotrophic condition (i.e. loading factor ((x) = 3.66) to its present state of oligo-mesotrophy. Eliminating point sources of sewage and reducing the sediment load in the Flathead River would reverse this assumed trend.

CONCLUSIONS AND RECOMMENDATIONS

This study showed that the trophic status of Flathead Lake is largely controlled by the timing and biotic availability of phosphorus delivered via riverine inflow. The fertilization effect of seasonal freshet turbidity was augmented by continuous loading from urban sewage and airshed fallout (i.e. bulk precipitation on the lake surface). We believe that the lake biota will continue to respond to increased phosphorus loading by proportionately elevating primary production. Even though nitrogen may be approaching a deficit relative to phosphorus from the classical viewpoint, recent literature (cf. Schindler 1977; Wetzel 1983) supports our contention that additional phosphorus loading will favor nitrogen-fixing plankters (e.g. *Anabaena flos-aquae*) and proportionately larger blooms of pollution algae will manifest in warm waters during late summer.

Prior to human settlement of the basin, productivity of Flathead Lake was probably entirely under the control of sediment-phosphorus loading via advective circulation. The mechanisms by which microbiota mobilize phosphorus from riverine sediments remain to be fully elucidated. Quite likely, the indigenous biota in Flathead lake are much more adapted for the utilization of sediment-phosphorus than our algal assays with *Selenastrum* indicated. But, even if the sediment-phosphorus source was underestimated, the effect of advective loading lakewide is seasonal. Anthropogenic contributions, primarily via sewage, are continuous and probably enhance the baseflow impact of riverine inputs, because of the extreme lability of sewage-phosphorus.

Little understanding exists concerning the enhanced retention of sediment-phosphorus in the epilimnion, due to storage effect caused by operation of Kerr Dam during freshet. Likewise, we know little about how the discharges from Hungry Horse Dam affect the nutrient budgets for Flathead Lake. We believe that the release of higher dissolved phosphorus and nitrogen concentrations from the hypolimnion of Hungry Horse Reservoir, probably offset the retention of South Fork sediments (and associated phosphorus) in the reservoir. Both of these regulation mechanisms (for hydropower production) should be considered potential pollution problems for Flathead Lake. Documentation of the ways stream and lake regulation is impacting the internal nutrient delivery, and ultimately the eutrophication process, are urgently needed, especially since new discharge schedules are presently being proposed by the area hydropower distributor, the Bonneville Power Administration.

However, our mass balance data clearly show that sewage from urban point sources contribute 17.5% of the phosphorus load. Elimination of these sources would be quite costly, but, our modeling exercises indicated that reduction of 17.5% in the phosphorus load would quickly return the lake to an oligotrophic status. Nuisance blooms of algae and other chronic symptoms of eutrophication (eg. many shoreline homes can no longer obtain potable water from the lake, because coliform bacteria counts exceed state standards), would likely disappear if the net increase in phosphorus deficit (i.e. via sewage controls) were not offset by increased phosphorus loading from other sources.

With this in mind, we recommend that land disturbances which increase riverine turbidity be curtailed or eliminated. Reforestation of clear-cut areas would enhance evapotranspiration in the drainage basin, thus reducing the erosiveness of spring freshet. Processes such as mining and road building should be carefully planned to prevent additional sediment loading in the Flathead River tributaries. Also, no new discharge permits for sewage effluents into Flathead waters should be granted. The Evergreen area should be sewered immediately and additional development leading to additional urbanization of the Flathead River flood plain and its fluvial aquifer should be carefully and conservatively regulated.

The nutrient budgets (especially phosphorus) of Flathead Lake should be continuously monitored, as should the seasonal dynamics of the plankton community. It will be difficult, if not impossible, to evaluate the success or failure of control mechanisms without such data. We detail our recommended monitoring program in Appendix IV.

We were unable to demonstrate phosphorus stripping and/or clearing of the water column by flocculation associated with the freshet turbidity, and accepted the alternative hypothesis that the advective circulation of riverine turbidity actually fertilizes Flathead Lake. The lake is essentially an allochthonously-based system, in terms of nutrient recruitment, but most of the organic carbon in the lake is produced by phytoplankton *in situ*. The fertilization effect of the freshet sediments may be thought of as a natural nutrient subsidy in a nutrient (phosphorus) poor system, if one excludes the cultural sources. Conglomerations of heterotrophic bacteria, very small (ultraplankton) photolithotrophic plankters and clay particles do occur, within a sestonic microhabitat dimensioned by a few ($\leq$10) microns (Ellis and Stanford 1982). Future contributions to the understanding of factors controlling the trophic status of Flathead Lake will likely involve the physiological mechanisms by which microbes mobilize sediment-phosphorus, if the natural nutrient subsidy is not swamped by cultural effluents.

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APPENDIX I

Location of Sampling Stations

A. Flathead River below Sportsman's Bridge, Montana highway 202.
B. Swan River below Montana highway 35 bridge at Bigfork.
C. Flathead River below U. S. highway 93 bridge at Polson.
D. Yellow Bay Creek at culvert under Montana highway 35 at Yellow Bay State Park.
E. Dayton Creek at first bridge upstream from Flathead Lake confluence.
F. Midlake deep site, ca. 3.2 km south west of Yellow Bay.
G. Midlake north site, ca. midway between Yenne and Angel Points.
H. In Bigfork Bay ca. 1/2 km north west of Wayfarers State Park.
I. In Somers Bay ca. 1/2 km east of Montana Department of Fish, Wildlife and Parks fish hatchery.
J. In Dayton Bay ca. 1/4 km south east of old trestle at Dayton.
K. In Yellow Bay midway between Yellow Bay Point and Yellow Bay State Park.
L. In Skidoo Bay ca. 1/4 km from south shore.
M. In Polson Bay ca. 1/2 km north east of Ducharm access.
N. In Woods Bay ca. 1/4 km from north shore.

APPENDIX II

Calculation of Water Budget

A. Annual discharge at U.S.G.S. gages

1. Flathead River at Columbia Falls (12363000)

2. Whitefish River near Kalispell (12366000)

3. Stillwater River near Whitefish (12365000)

4. Swan River near Bigfork (12370000)

5. Flathead River at Polson (12372000) (below Kerr Dam)

Year	Annual Discharge (acre ft)				
	Site 1	Site 2	Site 3	Site 4	Site 5
1978	6,224,000	156,900	274,000	1,053,000	7,871,000
1979	6,670,000	133,400	227,700	869,700	7,890,000
1980	5,686,000	112,500	211,600	799,000	6,720,000
1981	7,718,000	179,000	287,700	995,000	9,266,200
1982	7,764,000	147,500	287,800	996,000	8,918,000
Total	33,162,000	729,300	1,288,800	4,712,700	40,665,200
$M^3 (10^9)$	40.922	0.900	1.590	5.815	50.181
$\bar{x}m^3 (10^9)$	8.184	0.180	0.318	1.163	10.036

(U.S. Geological Survey, 1978; 1979; 1980; 1981; 1982)

B. Annual discharge at ungaged sites

1. Ashley Creek

(a) mean discharge at old gaging site for period of record = 30.8 ft^3/sec = .0275 X $10^9 m^3$/yr (Stahly, 1978).

(b) sewage effluent Kalispell treatment plant (1978-1982 mean flow) = 1.85 ft ^{3}/sec = .0017 X $10^9 m^3$/yr (Montana Department Health and Environmental Science, unpublished records)

(c) total estimated discharge = .029 m^3 X 10^9/yr.

2. Spring creek average annual flow = 13.3ft^3/sec = .012 X $10^9 m^3$/yr. Stanford et al. 1983).

3. Mill-Rose Creek system average annual flow

estimated = .045 X $10^9 m^3$/yr. (Potter, 1978)

4. Side flows below gage on Swan River contribute an estimated discharge of .054 X $10^9 m^3$/yr (Potter, 1978).

5. Creeks along lake shoreline contribute an estimated .152 X $10^9 m^3$/yr. (Potter, 1978; Stanford et al. 1983).

C. Precipitation on lake surface

1. mean annual precipitation 1978-1982:

(a) Kalispell 44.04 cm

(b) Yellow Bay 57.69 cm

(c) Polson 47.96 cm

$\overline{X}$ = 49.89 cm (U.S. Weather Service, 1978-1982

2. (.4989m) (472.6 X $10^6 m^2$ lake area) = 0.236 X $10^9 m^3$/yr. At median between high and low volume.

D. Evaporation from lake surface estimated at .336 X 10^9 m^3/yr (from contour map of Kohler et al. given in Anderson and Jobson 1982).

E. An estimated .008 X $10^9 m^3$ is removed annually for irrigation above Kerr Dam (Flathead Irrigation Project, unpublished records.)

F. Influent via Flathead River at Sportsman Bridge = A1+A2+A3+B1+B2+B3= 8.768 X $10^9 m^3$/yr.

G. Influent via Swan River at Bigfork = A4+B4= 1.217 X 10^9 m^3/yr.

H. Total influent = B5+C+F+G= 10.388 X $10^9 m^3$/yr.

I. Total effluent = A5+D+E= 10.380 X $10^9 m^3$/yr.

However, for nutrient budget calculations evaporation is not included and sampling was done upstream of the irrigation intake; therefore, effluent via outlet = A5 + E = 10.044 x 10^9 m^3/yr for budget calculations.

K. Conversions used in above calculations:

1. million gallons/day X 1.547 = ft^3/sec.
2. ft^3/sec X 365 X 1.984 = acre ft/yr.
3. acre ft X 1.234 X 10^3 = m^3/yr.

APPENDIX III

Nitrogen and phosphorus loading into Flathead Lake from UMBS (Yellow Bay) sewage treatment facility

In 1974 the University of Montana Biological Station began operation of a tertiary sewage treatment plant built with funds provided by the U. S. Environmental Protection Agency. The facility serves the Biological Station and Yellow Bay State Park, under cooperative agreement with the Montana Department of Fish, Wildlife and Parks. A discharge permit (No. MT-ØØ23388) was received by the University from the Montana Department of Health and Environmental Sciences, which allows release of treated effluent into the lake (5Ø m off east shore). The permit specifies, among other things, that effluent will contain less than Ø.5 mg/l total phosphorus.

Until March, 1983, operation of the facility was supervised by the University's Physical Plant in Missoula. In process of documenting all nutrient sources to Flathead Lake, we discovered that the average effluent phosphorus concentrations (Table AIII-1) had remained well above the permit limit during our study period for nutrient mass balance (1977-1982). Analysis of plant efficiency revealed that the nutrient removal process and other features were poorly designed and had been operated incorrectly since 1974 (i.e. improper pH adjustment of phosphorus precipitation on alum). To help solve this problem, the University of Montana placed supervision of the sewage treatment plant with the UMBS Director in March, 1983, and corrective measures were initiated. While full compliance with permit requirements have not yet been met (largely because the dosages for nutrient removal are flow dependent and the recreational use of the State Park caused severe flow pulses), it is clear that effluent TP concentrations of at least .5 mg/l are sustainable. On a mass basis, the UMBS effluents contained less than .002 percent of the total phosphorus and nitrogen income to Flathead Lake (see Tables 6 and 7 in text). However, it is a point source discharge that may elicit local shoreline symptoms of eutrophication and should be limited to maximum plant capability. The history and future performance of the UMBS plant should be referred to in the event that nutrient removal options are considered for other lake influents.

Table AIII-1. Discharge (liters) and nutrient concentrations (mg/l) in effluent discharged during peak (July-August) and base (September-June) flow periods from UM Biological Station sewage treatment facility into Flathead Lake, 1978-1982. Data are means derived from 7 and 16 samples obtained from high and low flow periods (from Montana Department of Health and Environmental Sciences, unpublished records).

Monitoring Period	Total Flow	Nitrate	Ammonia	Kjeldahl Nitrogen	Total Phosphorus
July-August	3.2×10^6	1.25	6.90	9.34	4.83
Sept.-June	2.3×10^6	2.82	2.27	2.78	3.38

APPENDIX IV

Recommended monitoring of limnological (water quality) conditions in Flathead Lake

We have shown in this report that the trophic status (quality) of Flathead Lake has deteriorated significantly in recent years. The six-year data base accumulated to date will be less useful in determining future degredation of the Lake (or improvement, for example, as a result of limitations on quantity and quality of point source discharges of sewage) if monitoring of key parameters (e.g. phosphorus loading, primary productivity) is not continued. The work of C. A. Goldman and colleagues on Lake Tahoe has profoundly shown that collection of long-term (20 years or more) ecological data tempers the cultural eutrophication problem by maintaining public awareness of changing lake conditions, while at the same time providing a basis for merging water quality management and research goals (see Goldman and Horne, 1983). If the present trend toward conditions of eutrophy (involving severe algal blooms and associated toxins and drastically decreased water clarity) is to be arrested and hopefully reversed, a continuous record of nutrient loading and correlated biotic responses must be maintained. Otherwise, there will be no accurate frame of reference for judging the results of corrective measures. We herein recommend a monitoring program, based on a set of ecological parameters which our present study showed were most indicative of the eutrophication problem.

The goal of a monitoring program should be to maintain a continuous record of lake water quality. Phosphorus and nitrogen loading, along with several other chemical parameters (e.g. pH and alkalinity), which describe the geochemical nature of Flathead Lake, should be correlated with temporal dynamics of primary production and plankton standing crops.

The mass balance (ecosystem) approach (i.e. measurement of input, output, and ambient concentrations) will likely yield the most useful data, since we continue to be primarily interested in the health of the lake as a whole, not just pollution of particular bays or shoreline areas.

We recommend that costly laboratory analyses of major chemical constituents (i.e. nitrogen, phosphorus, carbon, alkalinity, silica, calcium, magnesium and iron) be limited to samples obtained monthly (plus an additional 4 sample sets during freshet) from the input and output sites and at four depths at the midlake location (Table AIV-1). Depth profiles of pH, conductivity, dissolved oxygen, water clarity (percent transmission), chlorophyll, and temperature should be taken at the same time water samples are collected. Additional profiles of these parameters should be made in the major bays (Table AIV-2) in order to relate lakewide conditions.

Table AIV-1. Recommended sampling sites for temporal analyses of chemical constituents (see Table AIV-4) in water samples returned to Freshwater Research Laboratory

Location

1. Midlake, 5 km southwest of Yellow Bay Point (4 depths: 5, 10, 25, 80 m).
2. Flathead River inlet, 1 km upstream from lake.
3. Swan River inlet, 50 m upstream from Bigfork sewer outlet.
4. Flathead River outlet under Highway 93 bridge at Polson.
5. Bulk (wet and dry fall) precipitation collected in a sampler located on Horseshoe (or Little Bird) Island (UMBS owned).

Total number of samples/year = 128

TableAIV-2. Recommended sampling sites for monthly *in situ* measurement of pH, conductivity, dissolved oxygen, water clarity (%transmission), light, chlorophyll and temperature profiles. These same sites are recommended for monthly plankton samplings.

	Location	Depth
1.	Sommers Bay 1 km west of MFWP Hatchery	20m
2.	Midlake North site between Yenne (eastshore) Angel (westshore) Points.	60m
3.	Big Arm Bay at the Ross Deep	60m
4.	Midlake 5 km southwest of Yellow Bay Point	100m
5.	Narrows area 0.5 km west of Bird Islands	80m
6.	Skidoo Bay 0.5 km from south shore	30-60m
7.	Yellow Bay 100 m northeast of Yellow Bay Point	30-60m
8.	Polson Bay 1 km north of south east shore (Ducharm access)	6m

Biotic variables can be limited to quantification of phytoplankton diversity and standing crops (Utermohl sedimentation technique) at one depth (5m; duplicate samples) and zooplankton density and community composition in duplicate, 30-m hauls with a 64um-mesh net (30cm diameter). These samples should be taken monthly at 8 locations (Table AIV-2). All samples should be curated in the Freshwater Research Lab plankton museum at UMBS for future reference.

Primary productivity should be measured monthly at the midlake site only, and using methodology of the present study (duplicated $H^{14}CO_3$ incubations in light and dark bottles at 1, 2.5, 5, 10, 20, and 30 m depths). We feel these data can be related to *in situ* measurement of chlorophyll (Turner Designs, Inc. fluorometer system) at other main lake sites (Narrows and Midlake North) and the bays (see Table A5-2); and, therefore, the detailed, synoptic (and very costly) measurements of primary productivity in the bays, as conducted in the present study, are not needed in order to document spatial differences lakewide.

We believe it is very important to maintain continuity of methodology, if a monitoring program is initiated. A methods manual in the Freshwater Research Laboratory at UMBS has been developed and approved by the U. S. Environmental Protection Agency for this project.

The cost for the monitoring program proposed here is summarized in Table AIV-3 and cost categories are itemized in Table AIV-4. We emphasize that this is the minimum amount of work that should be done. Other important monitoring considerations include:

- establishment of stream gauging stations for flow measurement on the larger shoreline creeks;
- establishment of stream gauging site on Stillwater River below confluence of Spring Creek and on Ashley Creek 2 km upstream from confluence with Flathead River;
- time-series monitoring of chemical parameters (e.g. Table AIV-4) on Stillwater River and Ashley Creek;
- monthly collection of periphyton species composition and productivity data (e.g. chlorophyll) to generate a firm estimate of the seasonal biomass dynamics and responses of benthic algae to changes in water levels and nutrient loading; and,
- collection of precipitation chemistry on the west shore.

These objectives should be seriously considered for implementation into the monitoring program, if wide interagency support for the monitoring goal can be forseen. We have proposed here a monitoring program that will do no more than assure continuity with existing data.

Table AIV-3. Annual budget[1] (FY 84 costs) for limnological monitoring of Flathead Lake, based on utilization of University of Montana Biological Station facilities and staff. See Table AIV-4 for itemization of cost elements.

A.	Salaries	$14,040
B.	Fringe Benefits (18% of A)	2,527
C.	Expendable Supplies	2,500
D.	Boat Maintenance	1,920
E.	Chemical Analyses in FRL	15,991
F.	Total Direct Costs	36,978
G.	Indirect Costs (53.4% of A)	7,497
H.	Total Costs	44,475

[1]This budget has been approved by the Office of Sponsored Programs and Research Administration, Main Hall, University of Montana, Missoula, Montana, 59812 (phone: (406) 243-6670).

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Associate Vice President for Research

Table AIV-4. Price breakdown of budget items for Flathead Lake monitoring, based on FY 84 costs.

A. Salaries - Time allocation

Duties	P.I. @ $130	Co. P.I. @ $90	Office @ $70
Field Sampling	3.0	3.0	--
Algae Counts	--	1.0	--
Zooplankton Counts	1.0	--	--
Primary Productivity work	--	1.0	--
Equipment Maintenance	0.5	0.5	--
Data Analysis	0.5	0.5	1.0
	5.0	5.0	1.0
		= 11 days/month	

B. Fringe benefits are calculated by multiplying salaries by 18 percent.

C. Expendable supplies include:
1. radioactive tracer ($H^{14}CO_3$), scintillation cocktail, counter maintenance, pipets and glassware for analysis of primary productivity samples;
2. bottles and preservative for algae and zooplankton;
3. miscellaneous supplies such as film, office supplies, computer disks, xerox, and bottles for bulk precipitation samplers.

D. Boat maintenance is based on 16 hours/month on lake use of the MonArk outboard ($10/hr.) owned by UMBS, which is equipped with the following field instruments:
- *in situ* sonde and deck readout for measuring depth profiles, pH, conductivity, dissolved oxygen, and temperature;
- submarine transmissometer for measuring water clarity;
- recording sonar for depth contouring and qualitative analysis of fish distribution;
- *in situ* sonde and deck readout for measuring light penetration (photosynthetically active radiation);
- flow through fluorometer for chlorophyll analysis

Total capital investment in this boat and equipment by UMBS is $67,000.

E. The following analyses of chemical constituents will be made in FRL on samples obtained in time series from the 5 major sites (Table AIV-1):

Constituent	Method of Analysis	Price/sample
1. Total phosphorus	Persulfate Digestion/ Ascorbic Acid	12.17
2. Soluble reactive phosphorus	Ascorbic Acid	12.17
3. Soluble total phosphorus	Persulfate Digestion/ Ascorbic Acid	12.17
4. Nitrate & Nitrite	Cadmium reduction	14.75
5. Ammonium	Phenate	6.13
6. Organic Nitrogen	Bucchi modified Kjeldahl	21.97
7. Total Inorganic Carbon	Oceanography Int. TOC	
8. Particulate Organic Carbon	Oceanography Int. TOC	25.76
9. Dissolved Organic Carbon	Oceanography Int. TOC	
10. Dissolved Silica	Molybdate reduction	8.10
11. Dissolved calcium	Atomic Absorption Spec.	2.00
12. Dissolved magnesium	Atomic Absorption Spec.	2.00
13. Dissolved iron	Atomic Absorption Spec.	2.00
14. Total alkalinity	Titration to pH 4.5	2.71
15. Turbidity	NTU meter	3.00
Total		124.93