Diagnostic Analysis of Annual Phosphorus Loading and Pelagic Primary Production in
Flathead Lake, Montana.

Flathead Lake Clean Lakes Project, Phase One

FLBS Open File Report 132-94

by

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INTRODUCTION

Flathead Lake, Montana is thought to be one of the most pristine large lakes in the temperate latitudes of the world. The water column is very transparent due to the paucity of plant growth nutrients incoming annually to the lake. Secchi disk (15 cm diameter) readings in the fall and winter usually exceed 15 meters. Owing to rapid increase in human habitation (> 2% per year since 1970) of the lake's shoreline and catchment basin, sustaining the high quality of water in the lake is an important public concern (Stanford and Ellis 1988).

Pelagic primary production at a single midlake site was quantified in relation to mass flux of nitrogen and phosphorus through the lake during the period 1977 - 1993. (Stanford et al. 1992) showed that primary production increased significantly, which strongly suggested chronic decline in water quality. Earlier analyses of the data bases (Stanford and Ellis 1988, Stanford et al. 1983) suggested that pollution from human sources was a primary factor in the observed decline. In an effort to curtail the phosphorus (P) mass reaching the lake, the State water quality regulatory agency instituted a phosphorus reduction strategy for the catchment that included P removal from sewage effluents and a ban on sale of phosphorus containing detergents (Water Quality Bureau 1985). The rationale was that lake wide primary production was determined by availability of phosphorus and that significant P reduction was achievable at the sewage treatment plants. The present study was undertaken to further examine inter-annual variation in P loading and pelagic primary production in a process - response context and to determine if the pelagic site was truly representative of lake wide conditions.

We report herein time series measures of limnological variables that describe conditions at six sites compared to the long-term data base at the midlake monitoring station. Inferences about the utility of the midlake site in describing long term trends in water quality in Flathead Lake are provided from the intersite comparisons. We also provide preliminary calculations of phosphorus mass flux through Flathead Lake, based on all data collected during 1977 - 1993. The major sources (i.e., precipitation on the lake surface, river tributaries, urban sewage discharges) of P are identified and annual loads from each source are quantified. Special consideration was given to the

importance of Ashley Creek as a source of P, because the largest sewage treatment plant in the catchment discharges into that creek. Storage and release capacity of the sediments in Ashley Creek were estimated from *in vitro* studies. Finally, preliminary analyses of annual P load as a determinant of annual pelagic primary production are provided.

STUDY SITE

Flathead Lake and its catchment basin comprise 18,379 km^2 (Figure 1). Seven major tributaries drain the catchment, but the 3 forks of the Flathead River produce most of the water that enters Flathead Lake annually (Table 1). About 65% of the annual inflow occurs from mountain snowmelt during the spring freshet, which generally peaks between 15 May and 10 June in the mainstem Flathead River at the gauging station near Columbia Falls. Minimum flows generally occur during the mid-winter as a result of ice formation in the tributaries. The pattern of water flux is controlled in part by Hungry Horse and Kerr Dams; consequences of hydromanipulation, while likely very significant, are not discussed herein (but see Stanford and Hauer 1992).

Flathead Lake is deepest on the east shore and relatively shallow on the west side (Figure 2). Owing to the large volume of water stored (Table 2), the lake has a large heat budget and rarely freezes over in winter. Seasonal cooling and heating (2° - 20°C annual amplitude at the surface) interacts with very strong coriolis circulation to produce dramatic longshore flow in a counterclockwise direction. Longshore flow is moderated by the inflow and outflow currents and complex wind-generated surges and currents. Hence, movement of river water into the lake and subsequent mixing is very complex, related to temperature and density patterns interacting with convective (internal) and advective (external) forces .

About 60,000 people live within the catchment basin including the shoreline of the lake and upstream areas. Ninety five percent of the population lives within the area of the box in Figure 1. Therefore, a large share of the nutrient pollution that reaches Flathead Lake likely is derived within that relatively small portion of the drainage. Kalipsell, Whitefish (including the Big Mountain
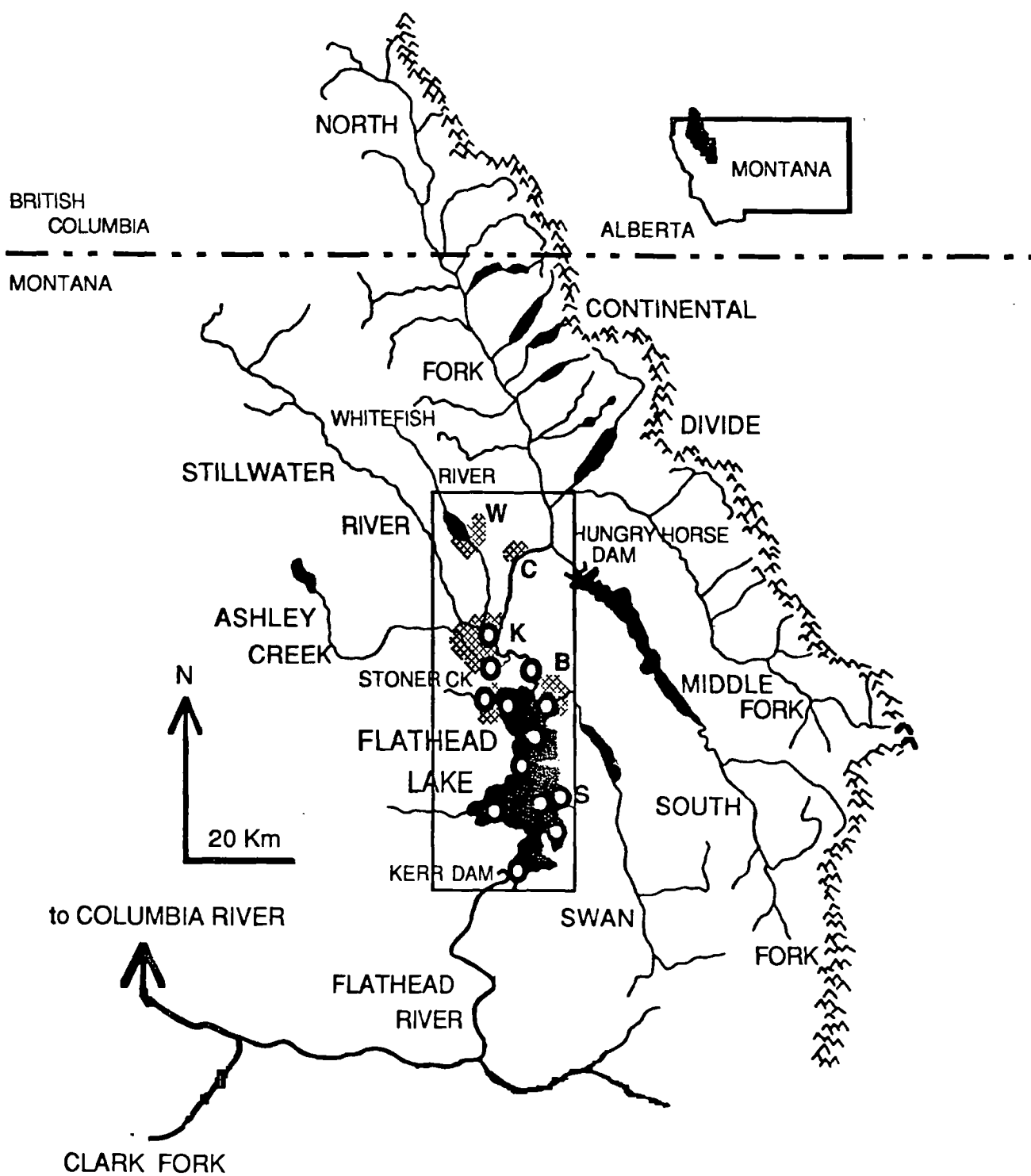
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Figure 1. Flathead River Basin, Montana and British Columbia. Major urban centers are shown by the cross hatching; urban sewage treatment plants that discharge into Flathead Lake exist at K (Kalispell), C (Columbia Falls), W (Whitefish) and B (Bigfork). Open circles are locations of field sampling sites. The Flathead Lake Biological Station is located at S. Rectangle includes 95% of the catchment's population.

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	--	--	5
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] Data collected by the Flathead Lake Biological Station
[d] Due to dam closure

winter sports complex), Columbia Falls and Bigfork are sewered communities that discharge effluent into the lake (Bigfork) or the larger tributaries. Other, much smaller sewage treatment plants exist in the basin but effluents are irrigated on crop land and assumed not to reach the lake. The Biological Station is served by a very small STP that discharges tertiary effluent directly into the lake. Rural areas include homes served by septic systems, and fertilizers are used on some crop lands. Water quality effects from these diffuse sources are problematic in the lower reaches of the Stillwater and Whitefish Rivers, Ashley Creek, Stoner Creek (and other small tributaries of
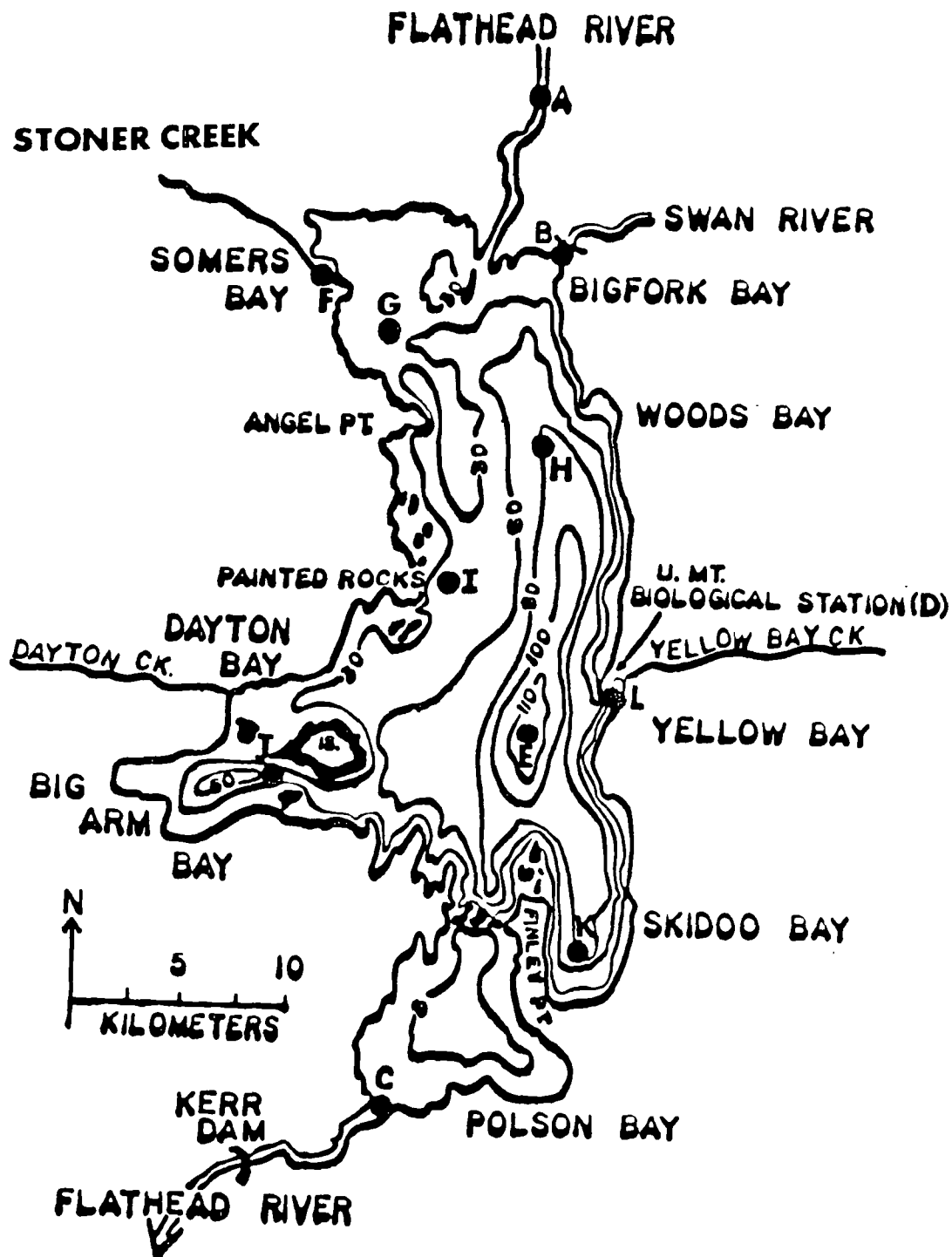
Figure 2. Bathymetry of Flathead Lake, Montana, and sampling sites. The long term sampling sites are A - E, whereas intersite comparisons were made at G - L.

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

the lake) and the alluvial aquifer of the Flathead River in the Evergreen area (Noble and Stanford 1986, Stanford et al. unpubl.). The lake shoreline is intensively developed with homes served by septic systems; shoreline sources of sewage breakout from septic systems have been documented (Hauer 1988). Timber harvest and associated road building have affected large areas of the catchment and may influence water, sediment and nutrient yield (Hauer 1991, Hauer and Blum 1991, Spencer 1991). Episodic wildfires also generate measurable nutrient loading in small tributaries and for short time periods (Spencer and Hauer 1991).

Approximately 60 percent of the land mass in the basin is included in Glacier National Park and various National Forest Wilderness Areas and roadless areas; this land mass also includes the highest elevations and highest precipitation in the basin. Hence, most of the water that reaches Flathead Lake is derived from a very pristine land mass and is therefore of very high quality. This is extremely important in terms of the mass flux of materials in the lake, since the volumetric renewal time (period required for tributary inflow and lake outflow to equal the volume of the lake basin) is less than 3 years (Stanford et al. 1983).

Intersite Comparisons

Six sites (G-L, Figure 2) on Flathead Lake were selected for collection of limnological data for comparison with the midlake site (E, Figure 2). Locations of sites (Figure 2) were determined by shipboard global positioning system (GPS):

- G, Lakeside, 20m depth;
- H, Midlake North, 90m;
- I, Painted Rocks, 50m;
- J, Ross Deep, 30m;
- K, Skidoo Bay, 60m;
- L, Yellow Bay, 20m; and,
- E, Midlake Deep, 110m, the long term monitoring site.

These sites were sampled monthly from August, 1991 - September, 1992 and April - August, 1993. All work was done from the Biological Station's research vessel, *Jessie B*. Sites were sampled within a one or two day period each month.

The sampling protocol at each site was as follows:

- discrete samples for chemistries (Table 3) at 5 m (or 1 m) and near bottom;
- a single, integrated (0 - 30 m or near bottom) sample, subsampled for:

   1.) analytical chemistries (Table 3),

   2.) chlorophyll *a*;

- duplicate chlorophyll *a* samples from the depth of maximum fluorescence (as determined *in situ* using a shipboard fluorometer);
- depth profiles using electronic instrumentation to measure relative fluorescence, photosynthetically-active radiation, specific conductance, pH, temperature, dissolved oxygen and water clarity (% transmission); and,
- secchi depth using a standard black and white secchi disk.

Precision of the analytical analyses of water samples was determined by ±1 sd of replicated analyses on individual samples, whereas accuracy was determined by 110% > x > 90% recovery of a known addition of standard solution to selected samples. These quality control criteria were tested on approximately 1 out of every 15 samples run in the Freshwater Research Laboratory at the Biological Station. Analytical performance of the lab personnel was also evaluated about every 6 months by analyses of quality control (unknown concentrations) samples provided by the U.S. Environmental Protection Agency. These performance evaluations are on file at the Biological Station. The lab was able to achieve the correct analytical answer over 90% of the time in spite of the fact that the unknown samples provided by EPA were often 1 or more orders of magnitude more concentrated than samples routinely received by the lab from collections in the solute-poor waters of the Flathead Basin; thus, errors almost always resulted from inaccurate dilution of the unknowns into the lab's normal working ranges, which approach the analytical detection limits of the various methods. All sample data, laboratory standard curves and quality control information were archived by G. Poole (FLBS Data Manager) in the Biological Station's data storage and retrieval system.

Ashley Creek Sediment Assays

The objective was to determine the extent of sorption or uptake, and desorption or release of soluble forms of phosphorus and nitrogen to and from the bottom sediments of Ashley Creek. Two experiments were done. In the first experiment, we utilized the commonly employed method of creating a synthetic equilibration solution or reagent (see Klotz 1988) which was used in combination with intact cores of sediment, rather than suspending sediments in a flask. In our second experiment we attempted to better mimic natural conditions by adding filtered creek water (collected above the STP outfall) to intact cores of sediments from Ashley Creek (below STP outfall) for same-day incubation.

All materials used in the field and laboratory were thoroughly cleaned using 10% hydrochloric acid, followed by repeated rinsing with deionized water.

A 24 cm length sediment core was extracted from the bottom of Ashley Creek, downstream of the Kalispell STP, approximately midstream. The sediment core was collected in a clear 6.4 cm diameter PVC tube. The sediment column consisted of a fine silty layer of detritus, approximately 2 cm in depth, overlying a 16 cm layer of coarse sandy loam and a 6 cm layer of sand at the bottom. The bottom of the tube was sealed (watertight) using a 6.4 cm diameter rubber stopper with a metal screw-on compressor wrapped in parafilm. The tube was filled with stream water from the same location and capped using a rubber stopper wrapped in parafilm. The sediment core remained undisturbed and intact throughout collection.

Temperature and oxygen content of the stream water were measured on site. The sediment core was transported to the laboratory within 45 minutes and immediately placed in an environmental chamber at 5° C (the temperature of Ashley Creek at the time of core sampling). The water column above the sediment core was carefully extracted using a peristaltic pump, leaving 50 ml of water in order to minimize disturbance at the sediment/water interface. The creek water was immediately replaced with 250 ml of prepared reagent by slowly pumping the solution into the top of the core incubation apparatus. An L-shaped glass tube was used to distribute the water to the side of the incubation apparatus to reduce disturbance of the upper sediment layer. In order to mimic nutrient and specific ion concentrations (Ca) in Ashley Creek a reagent was prepared with deionized, bacteria-free water and was amended with $NaHPO_4$ (final concentration of 19.2 µg/l SRP), buffered with sodium acetate/acetic acid to a pH of 8.07 and CaCl added for a final concentration of 39 mg/l Ca. For the period of record 1977-1992, the mean concentration of Ca and pH in Ashley Creek at the site of core collection was 39 mg/l and 8.04, respectively; the mean concentration of SRP in Ashley Creek above the STP was 19.2 µg/l. We chose a site on Ashley Creek above the STP which might represent a minimum level of SRP likely to occur in Ashley Creek (i.e., should the STP remove all SRP from wastewater inflow).

Initial incubation of the reagent and the sediment core proceeded for 2 hours, then 200 ml was removed by peristaltic pump and filtered for subsequent analysis of SRP, $NO_{2/3}$-N and NH_3-N. Immediately following removal of the water, another 250 ml of the reagent was added to the core incubation apparatus for continued incubation. The same removal, filtration and addition procedures were performed 4, 6, 8, 10, 17, 22, 32, 52 and 72 hours after the initial addition of the reagent to the core incubation apparatus.

Second Experiment - 25 March, 1993

Four benthic cores were collected from Ashley Creek in the same vicinity as the initial core (i.e., below Kalispell STP). The cores were obtained equi-distant from one another in a cross section of the stream. Each core was approximately 20 cm in length and included a clay layer below the silt and sandy loam layers. All field preparations were identical to the initial run. A grab sample from the water column of Ashley Creek was obtained for chemical analysis at the same site and at the same time of core sampling .

The cores were handled in much the same way as the initial run. However, instead of using a synthetic solution in the incubations, water collected from Ashley Creek below Smith Lake, a site well above the Kalispell STP, was filtered for use in the experiment. The incubation was also extended 34 hours to assess potential equilibration; samples of incubating water were collected 3, 6, 8, 12, 24, 45, 65, 85, and 106 hours after initiation of the experiment. In an attempt to diminish any effects from other sorption/desorption variables such as dissolved oxygen content, pH and ionic strength, additional water was collected from Ashley Creek (same site below Smith Lake) approximately every 48 hours for use in the incubation (a total of three new batches). Dissolved oxygen and pH of each of the filtered water samples did not change more than 5% during the period of use. All cores were incubated at a constant temperature of 4° C (the temperature of Ashley Creek at the time of core sampling).

The removal and addition of water to the core incubation apparatus and filtration and analysis of samples were the same as that described for the initial run. Filtered Ashley Creek water

samples were utilized according to the following time table: 1) first sample was used for the 3, 6, 8, 12, 24, and 45 hour incubations, 2) second sample was used for the 65 and 85 hour incubations, 3) third sample was used for the 106 hour incubation.

Long Term Phosphorus Loading and Pelagic Primary Productivity

Monitoring sites (Figs. 1 and 2) where long term phosphorus data were obtained:

- Ashley Creek below the Kalispell sewage treatment plant outfall (Figure 1);

- Stillwater River in Evergreen below the confluence of the Whitefish River (Figure 1);

- Flathead River near Holt (Sportsmen Bridge), the primary upstream tributary (A, Figure 2);

- Swan River at Bigfork, upstream from the outfall of the sewage treatment plant (B, Figure 2);

- Flathead Lake at the outlet sill near the Highway 93 bridge in Polson (C, Figure 2);

- the bulk precipitation collector located on the dock at the Flathead Lake Biological Station (D, Figure 2);

- midlake deep (110 m depth) ca. 1 mile west of Yellow Bay Point in a pelagic area of Flathead Lake (E, Figure 2);

- and, Stoner Creek near Lakeside (F, Figure 2).

These sites were selected so that all of the major inputs of water and materials into Flathead Lake could be quantified (e.g., the Kalispell sewage effluent via Ashley Creek) and related to biophysical dynamics within the lake measured in time series at the midlake deep site and at the lake outlet. Flow data were obtained from the US Geological Survey (Table 1) except for Ashley and Stoner Creeks, where we operated continuous recording devices to estimate flow.

Long term biophysical data were collected 6 - 15 times per year from October, 1977 - October, 1993. Sampling frequency was increased to twice monthly during spring in years where complete data sets were obtained. Some years funding limitations prevented collection on the regular schedule. Hence, we report herein annual estimates based only on those years in which a full set of data were obtained (i.e., at least 12 monthly samples per year).

Biophysical variables were as in Table 3 with some minor modifications over the years. For example, the lab switched from long light, path spectrophotometry to an auto analyzer for colorimetric analyses in 1988. However, quality assurance criteria, described above for the intersite comparisons, remained constant throughout the period of record. Primary productivity and chlorophyll *a* were measured at the midlake site only, not on the tributaries. Estimates of input of phosphorus from the atmosphere on the surface of the lake were obtained from collections of bulk precipitation at the Biological Station. Duplicated samplers collected wet and dry deposition and the volume of precipitation was determined by averaging Polson and FLBS standard NOAA rain gauge volumes as well as the water mass obtained in the collectors.

Mass flux of phosphorus was estimated by summing the daily load values for the water year. Daily loads at the various sites were determined by multiplying water volume by TP concentrations. Concentrations for intervals between sampling dates were estimated by averaging nearest measured concentrations weighted by flow. Periods of high flow in the Flathead River at Holt were characterized by high total suspended solids (TSS) loads and correspondingly high total phosphorus concentrations (see Figure 3 for TP to TSS relationship). Ellis and Stanford (1986) previously showed that only 10% of the phosphorus in samples with high suspended sediment concentrations was biologically available. Therefore, corrections were applied to all Flathead River data where total suspended solids exceeded 10 mg/l, which is the point where TP concentration in Flathead River samples becomes a positive linear function. The same correction was used at other tributary sites when TSS values exceeded 10 mg/l. Ellis and Stanford (1986) also showed that ≥95% of the total phosphorus in bulk precipitation was bioavailable at all times of the year, so no corrections were applied to those data.
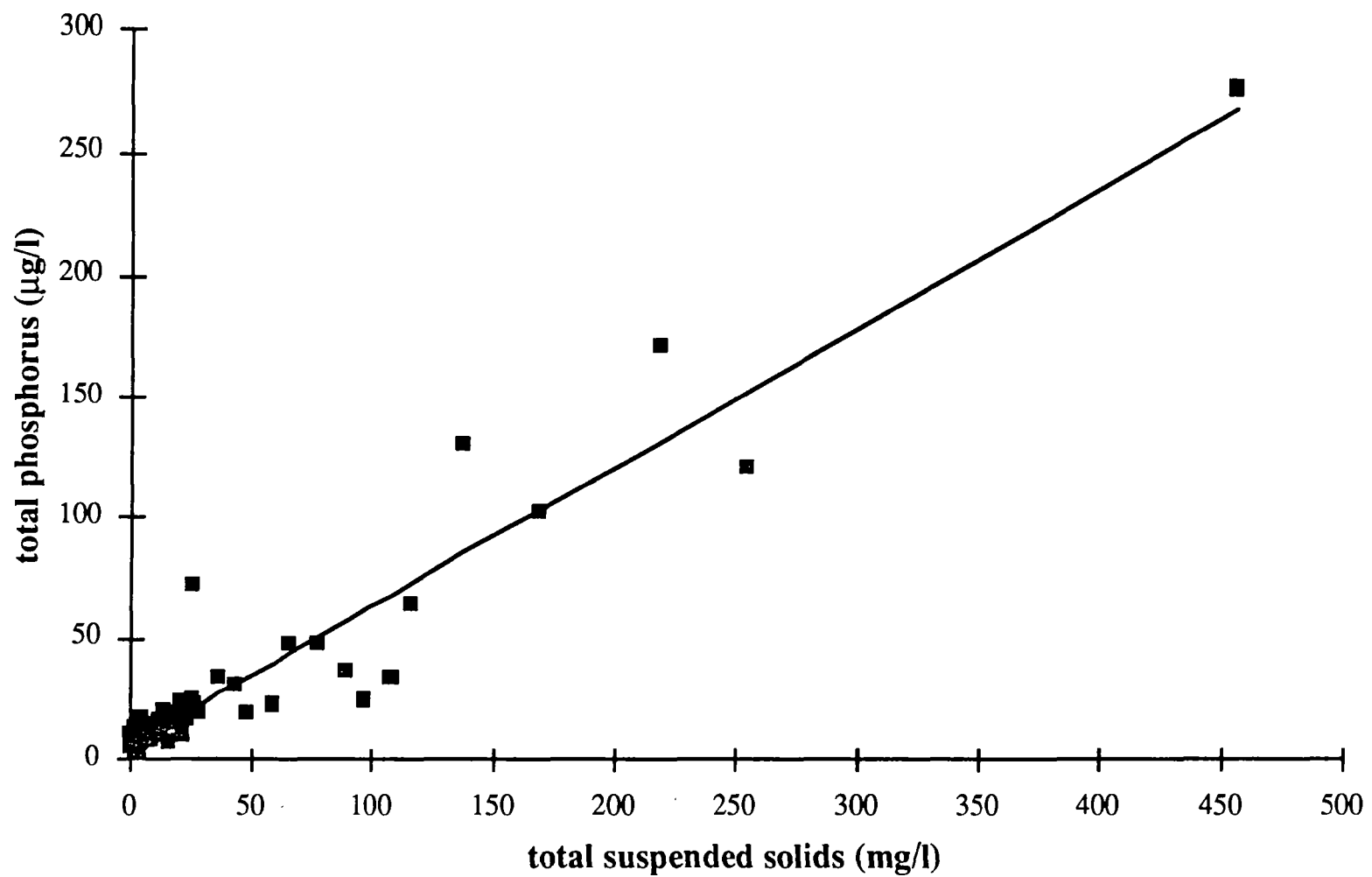
Figure 3. Total phosphorus as a linear function of total suspended solids for all data collected in the Flathead River at Holt near the confluence with Flathead Lake.

Table 3. Biophysical variables and methods used in monitoring water quality in Flathead Lake.

Variable (units)	Method (references)	Detection limit
Analyses of water samples		
phosphorus (μg/l-P)		
total	persulfate digestion; modified automated ascorbic acid (1)	0.4
soluble total	filtration; persulfate dig.; mod. auto. ascorbic acid (1)	0.4
soluble reactive	filt.; mod. auto. ascorbic acid (1)	0.4
nitrogen (μg/l-N)		
total persulfate	persulfate digestion (2); auto. cadmium reduction (1)	20.0
nitrite + nitrate	auto. cadmium reduction (1)	2.0
ammonia	auto. phenate (1)	0.5
sulfate (mg/l-SO$_4$)	ion chromatography (1)	0.05
dissolved silica (mg/l-SiO$_2$)	auto. molybdate-reactive silica (1)	0.2
carbon (mg/l-C)		
non-dissolved organic	persulfate dig.; infrared CO$_2$ detection (3)	0.10
dissolved inorganic	acid liberation; infrared CO$_2$ detection	0.10
carbonate alkalinity (mg/l-CaCO$_3$)	titration (1)	0.5
turbidity (NTU)	nephelometry (1)	0.10
soluble total metal	filtration;	
Ca; Mg; Na; K (mg/l-AW)	flame atomic absorption (4)	0.10
Biological analyses		
chlorophyll a (mg/m^3)	acetone extraction (1,5)	1.00
relative fluorescence (units)	continuous flow in situ fluorometry (6)	0.05
photosynthetically active radiation (μEinsteins/m^2/sec)	submarine/deck quantum meter (7)	0.01
phytoplankton primary productivity	^{14}C uptake in light and dark bottles; acid-bubbling technique (8)	

Table 3 (continued).

Physical profiles		
temperature (°C)	thermistor (9)	0.15
dissolved oxygen (ppm)	electrode (9)	0.20
pH (units)	electrode (9)	0.1
conductivity (μmhos/cm)	electrode (9)	1.5
secchi depth (m)	secchi disk	NA

[1]APHA, 1985
[2]D'Elia et al., 1977
[3]Menzel and Vaccaro, 1964
[4]Perkin-Elmer, 1976
[5]Marker et al., 1980
[6]Turner Designs, 1981
[7]Licor 188 integrating quantum meter
[8]Theodorssen and Bjarnason, 1975; Wetzel and Likens, 1991
[9]measured in situ using Hydrolab Surveyor III

RESULTS AND DISCUSSION

Intersite Comparisons

We observed very little intersite variation in terms of thermal patterns and stratification (e.g., Figure 4, additional profiles are given in Appendix 1). No significant differences between sites or depth categories were detected in depth profiles of pH, specific conductance, percent transmittance and photosynthetically active radiation, except during spring runoff when turbid river water was present as an overflow plume. Turbidity was always highest near the river mouth and on the west shore of the lake as reported in more detail by Stanford et al. (1983). Turbidity also persisted at the Lakeside site owing to wind resuspension of sediments at that shallow site. A multi-way MANOVA with repeated measures showed significant differences ($p < 0.05$) in turbidity at Lakeside in comparison to the other shallow sites (i.e., Yellow Bay, Ross Deep and Midlake Deep samples from $\leq$ 30m; see Table 4). Significant differences in TSS were also observed between Painted Rocks and Midlake Deep (Table 5). Only those analysis results where significant differences were found are presented in Tables 4 and 5.

Table 4. Results of a multi-way with repeated measures and post hoc Tukey HSD test for all chemical data collected at 4 sites on Flathead Lake, August, 1991 - September, 1992 and April - August, 1993. The shallow sites of Lakeside (L), Yellow Bay (Y), Ross Deep (R) and ≤30m samples from Midlake Deep (M) were included in the analyses. Significant differences in concentrations between sites, depths (t = top, i = integrated, b = bottom) or a significance due to an interaction of site and depth are noted. "Bottom" samples for the Midlake Deep site were from 30m for comparison with the other shallow sites. A description of the significance is presented by bars which overlay sites or depths with no significant difference.

Variable	Site	Depth	Site x Depth	Description
Chl a	X			$\overline{Y\ M}\ L\ \overline{R}$
NDOC	X			$\overline{M\ L\ Y}\ R$
		X		$\overline{b\ i}\ t$
$NO_{2/3}$		X		$\overline{t\ i}\ \overline{b}$
			X	For R & M: $\overline{t\ i}\ \overline{b}$
			X	For b: $\overline{M\ R}\ \overline{L\ Y}$
SiO_2	X			$\overline{L\ Y}\ \overline{M\ R}$
		X		$\overline{t\ i}\ \overline{b}$
			X	For R: $\overline{t\ i}\ \overline{b}$
			X	For b: $\overline{Y\ L\ M}\ \overline{R}$
SO_4			X	For b: $L\ \overline{Y\ R\ M}$
			X	For i: $\overline{L}\ \overline{Y\ M\ R}$
			X	For t: $\overline{L\ R\ Y\ M}$
SP		X		$\overline{t\ b}\ \overline{i}$
Turb	X			$\overline{Y\ R\ M}\ \overline{L}$
		X		$\overline{t\ i}\ \overline{b}$
			X	For t: $\overline{Y\ R\ M}\ L$
			X	For b: $\overline{Y\ R\ M}\ \overline{L}$

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Table 5. Results of a multi-way MANOVA with repeated measures and post hoc Tukey HSD test for all chemical data collected at 4 sites on Flathead Lake, August, 1991 - September, 1992 and April - August, 1993. The deep sites of Midlake North (N), Painted Rocks (P), Skidoo Bay (S) and Midlake Deep (M) were included in the analyses. Significant differences in concentrations between sites, depths (t = top, i = integrated, b = bottom) or a significance due to an interaction of site and depth are noted. A description of the significance is presented by bars which overlay sites or depths with no significant difference.

Variable	Site	Depth	Site x Depth	Description
Alk		X		$\overline{t\ i}\ \overline{b}$
NDOC		X		$\overline{b}\ \overline{i\ t}$
NO$_{2/3}$		X		$\overline{t\ i}\ \overline{b}$
			X	For b: $\overline{P\ S\ N\ M}$
SiO$_2$	X			$N\ \overline{P\ S\ M}$
		X		$\overline{t\ i}\ \overline{b}$
SO$_4$		X		$\overline{t\ i\ b}$
TPN		X		$\overline{t\ i\ b}$
TSS	X			$\overline{M\ S\ N\ P}$
			X	For N: $\overline{i\ b}\ t$
			X	For P: $\overline{i\ t}\ b$
			X	For t: $\overline{M\ S\ P\ N}$
			X	For i: $\overline{M\ S\ N}\ P$

However, during the summer period of thermal stratification, a significant dissolved oxygen sag developed in the hypolimnion at the Ross Deep site (Figure 5). The deficit was observed all three summers of the study. This phenomenon occurred no where else in the lake and, to our knowledge, dissolved oxygen values significantly less than saturation were never before reported for Flathead Lake.
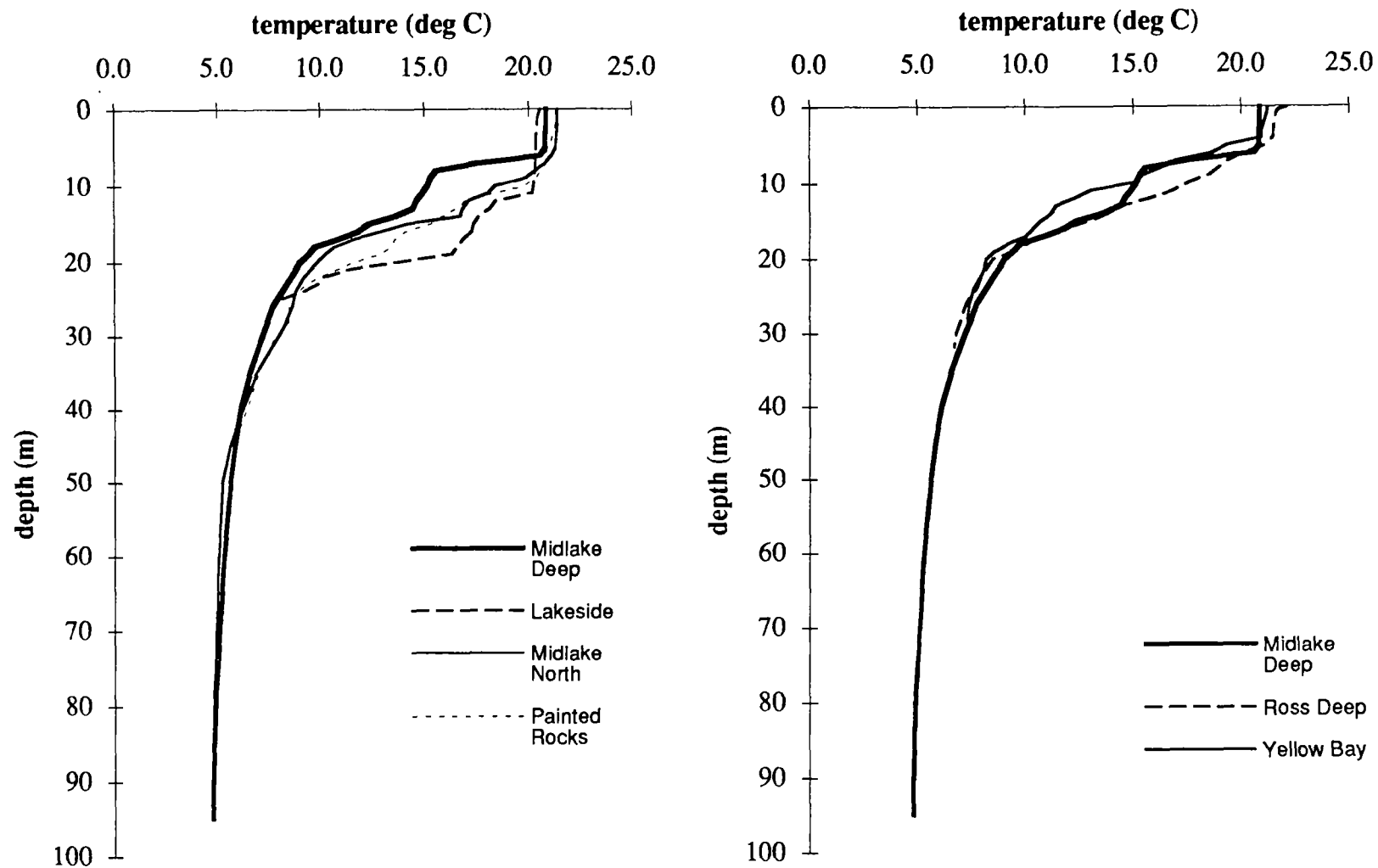
temperature (deg C)

0.0   5.0   10.0   15.0   20.0   25.0

depth (m)

0
10
20
30
40
50
60
70
80
90
100

—— Midlake Deep

— — — Lakeside

——— Midlake North

········· Painted Rocks

temperature (deg C)

0.0   5.0   10.0   15.0   20.0   25.0

depth (m)

0
10
20
30
40
50
60
70
80
90
100

—— Midlake Deep

— — — Ross Deep

——— Yellow Bay

Figure 4. Depth profiles of temperature in August, 1992 at various sites on Flathead Lake.

dissolved oxygen (mg/l)

dissolved oxygen (mg/l)
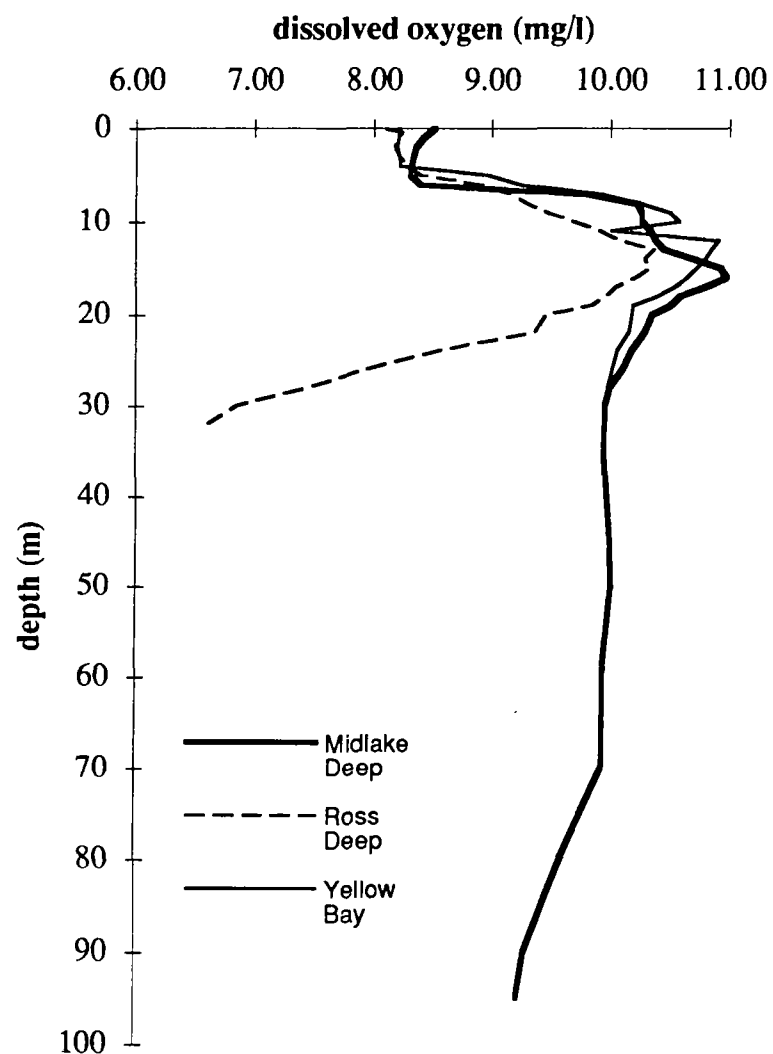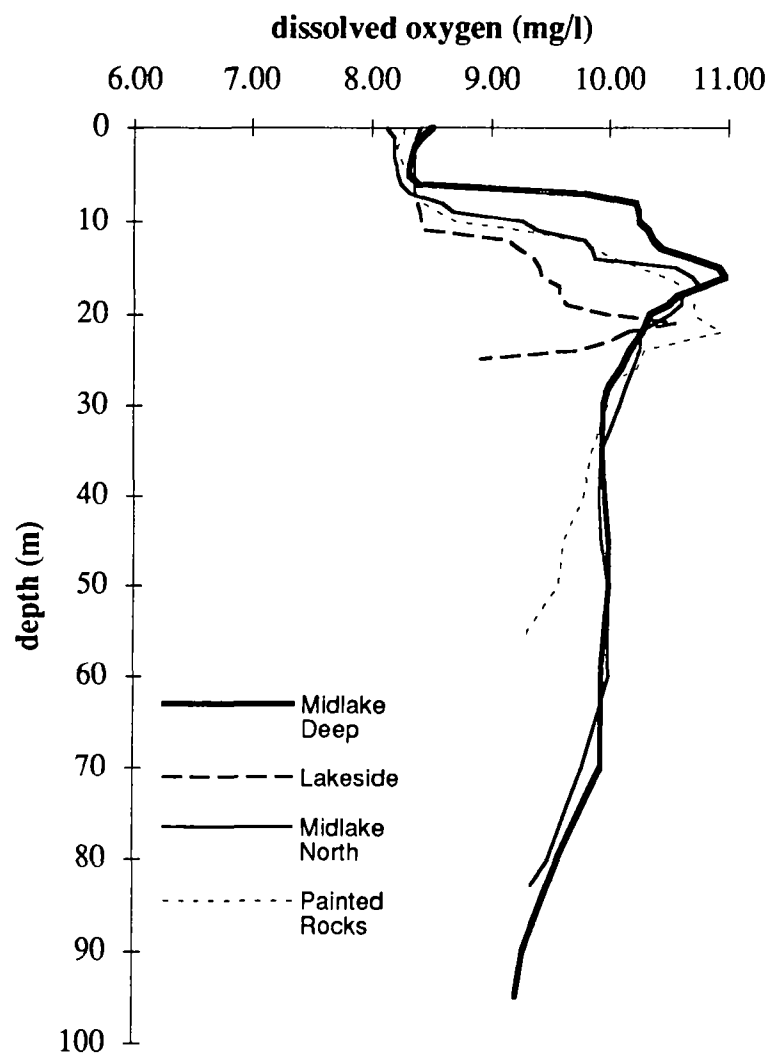
Figure 5. Depth profiles of dissolved oxygen in August, 1992 at various sites on Flathead Lake.

For easy comparison of the P and N variables over the 18 sampling dates, data for depth integrated samples from the photic zone for each site were plotted in time series with the midlake data (Figures 6 - 8). The entire data set is provided in Appendix 2. A pronounced nitrate + nitrite spike was observed at Lakeside and Painted Rocks in June, 1993 (Figure 8), but was likely due to the presence of the river plume. Summer depletion of nitrate + nitrite occurred uniformly at all sites (Figure 8), except Ross Deep where the depletion began in April. This event suggested that microbial uptake of nitrate was progressing at higher rates and for longer periods of time at that site. Site alone was not a significant predictor of the variation in $NO_{2/3}$, but for both the shallow and deep sites, depth and site x depth were significant predictors (Tables 4 and 5). Bottom $NO_{2/3}$ concentrations were significantly different from top and integrated values and the interaction of site and depth indicated that for bottom samples, Lakeside, Yellow Bay and Painted Rocks $NO_{2/3}$ values were significantly different from Midlake values. No other statistically significant differences in N or P concentrations were observed between sites. Both ammonium and soluble reactive phosphorus remained below or very near the detection limits at all sites.

A few other instances of significant deviation from the midlake data were observed at the other sites. Concentrations of NDOC at Ross Deep were significantly different from Midlake Deep data (Table 4). Dissolved silica concentrations at Lakeside, Yellow Bay, Midlake North and Painted Rocks were also significantly different from Midlake Deep concentrations; for bottom samples, Ross Deep SiO_2 was significantly different from Midlake Deep SiO_2 (Tables 4 and 5). The interaction of site and depth indicated that Lakeside SO_4 concentrations were significantly different from Yellow Bay, Ross Deep and Midlake Deep concentrations at each depth (Table 4).

Higher bioproduction at the Ross Deep site in Big Arm Bay was corroborated by presence of significantly higher chlorophyll values on nearly all dates (Figure 9; see also Table 4). We think that plankton fallout from the epilimnion and subsequent microbial decomposition in the hypolimnion may explain the oxygen deficit noted above. We assume this phenomenon is likely driven by shoreline nutrient loading in the Big Arm area, since other measurements were consistent with other sites. Entrainment of the overflow plume of the nutrient-rich spring runoff event may also be a factor in the observed decline in
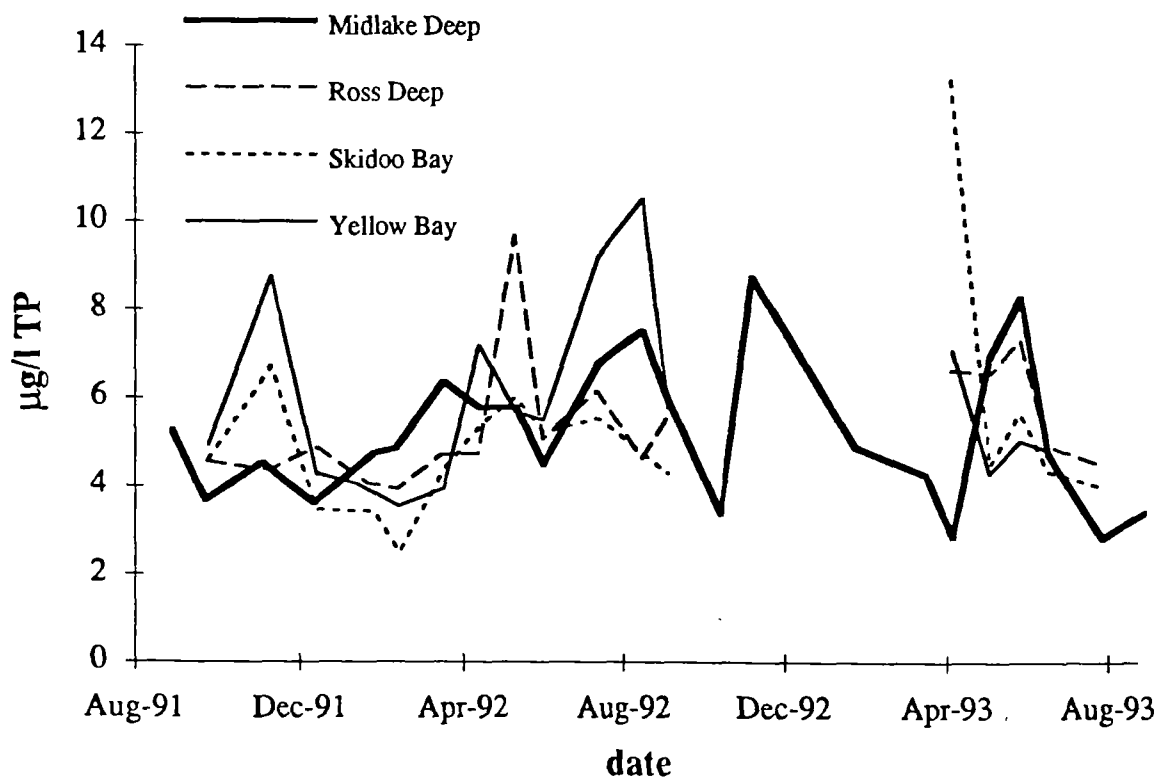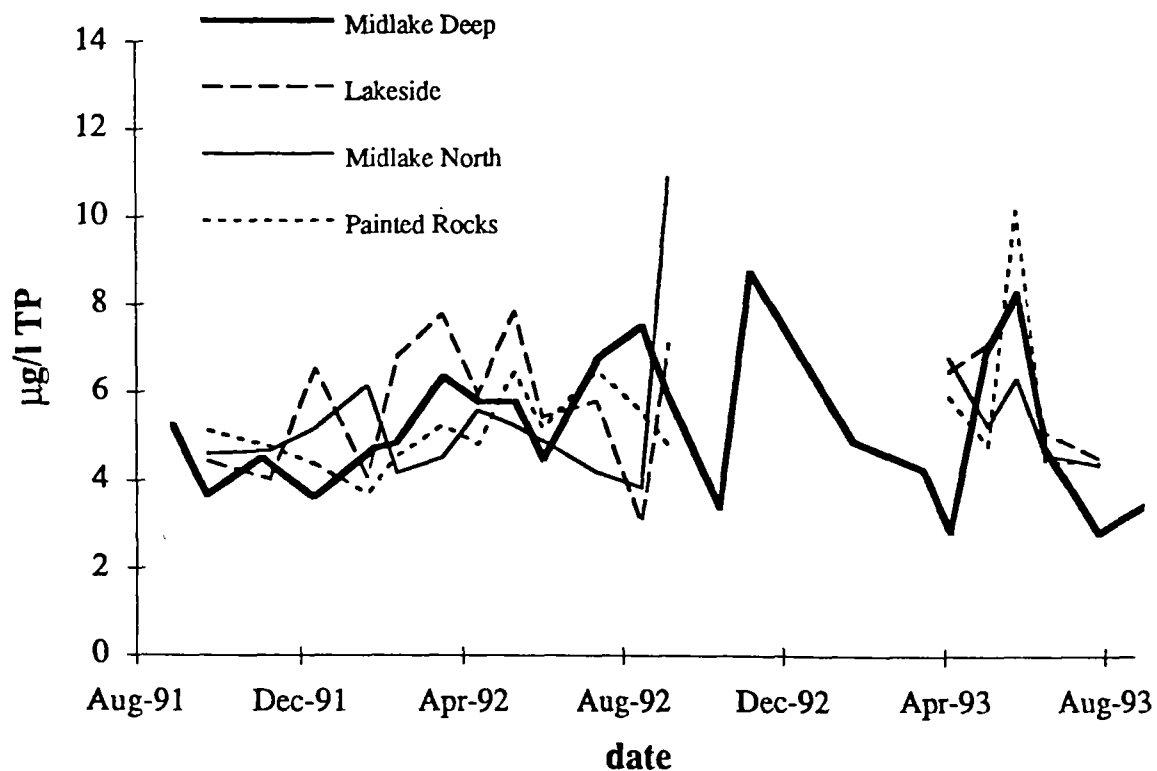
Figure 6. Intersite comparisons of total phosphorus (µg/l-P) within the photic zone of the water column (i.e., integrated samples from 0-30m, or bottom) for the period of record in the clean lakes diagnostic study.
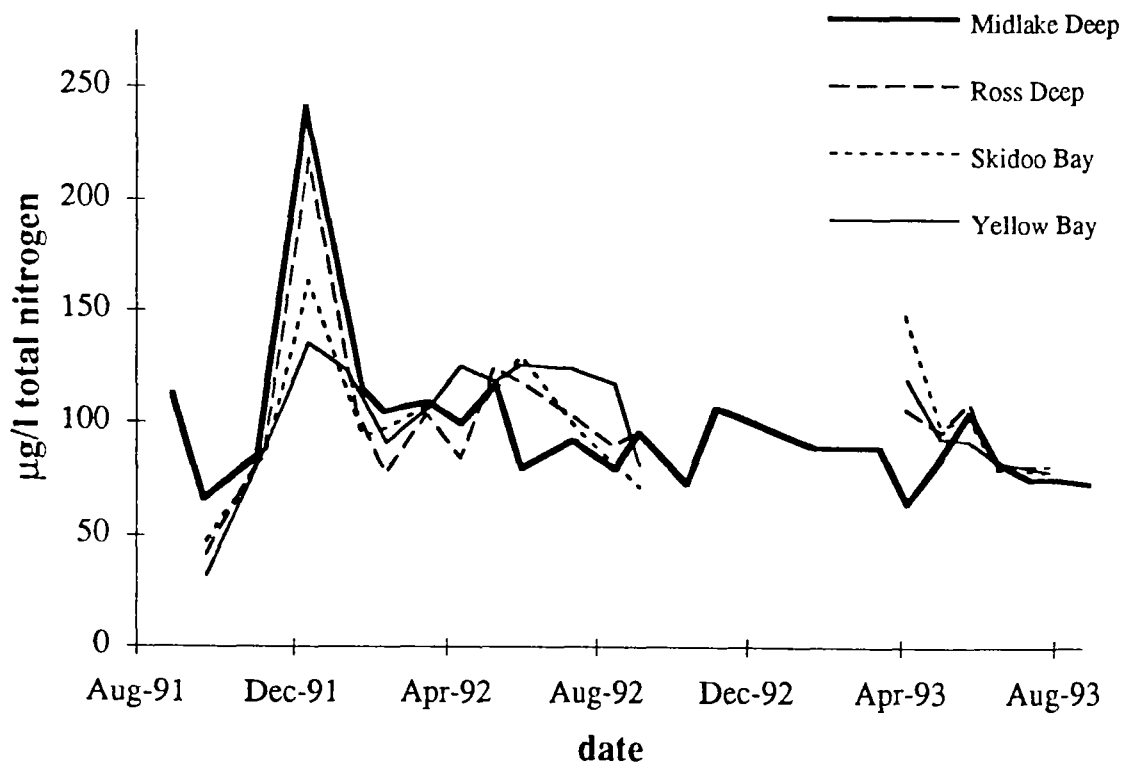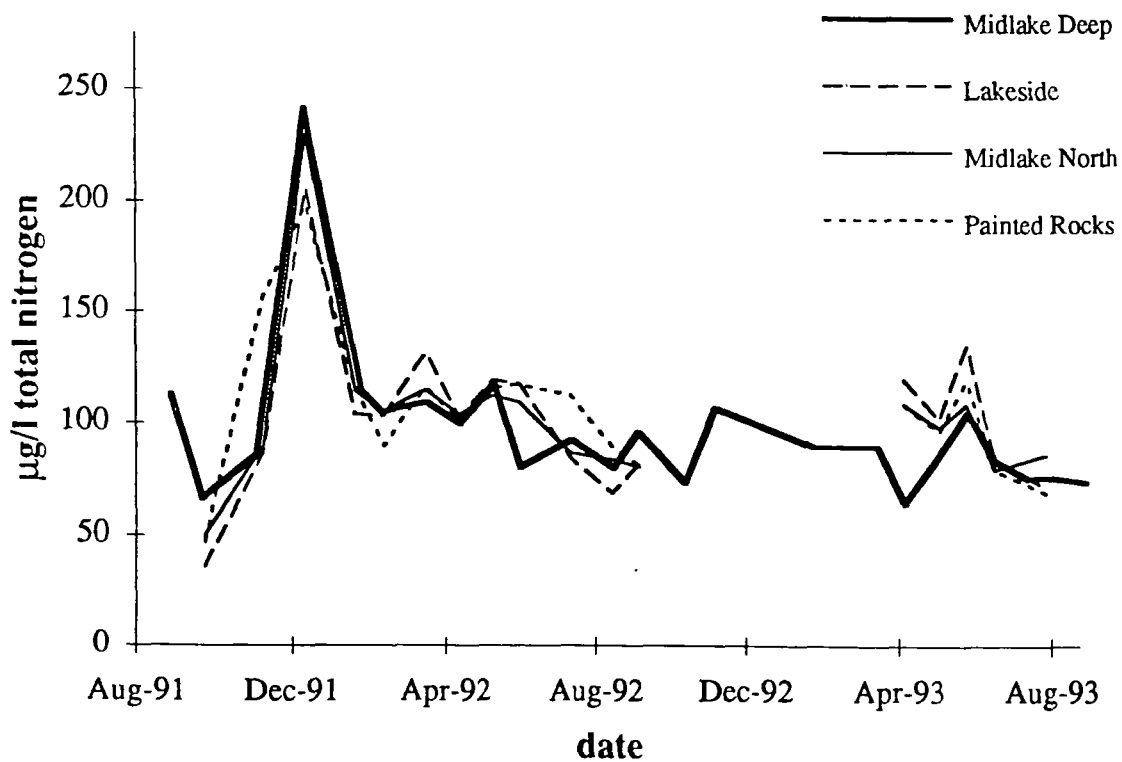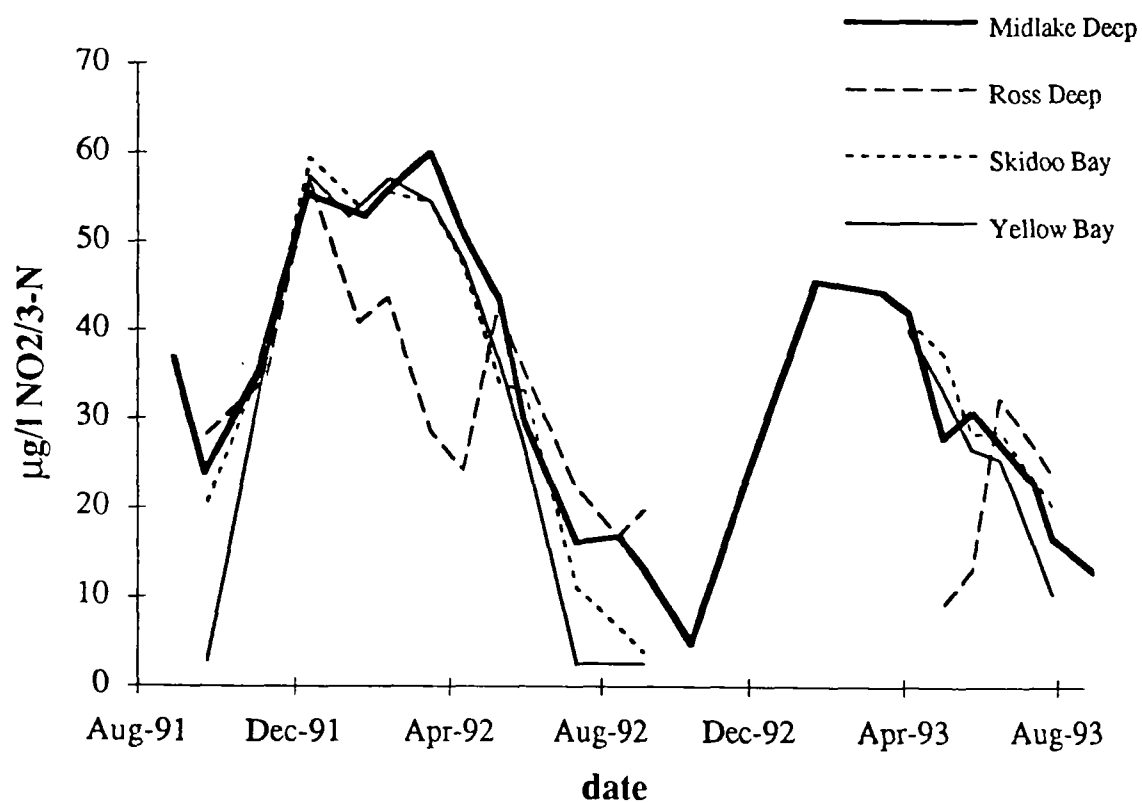
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Figure 7. Intersite comparisons of total nitrogen (μg/-N) within the photic zone of the water column (i.e., integrated samples from 0-30m, or bottom) for the period of record in the clean lakes diagnostic study.
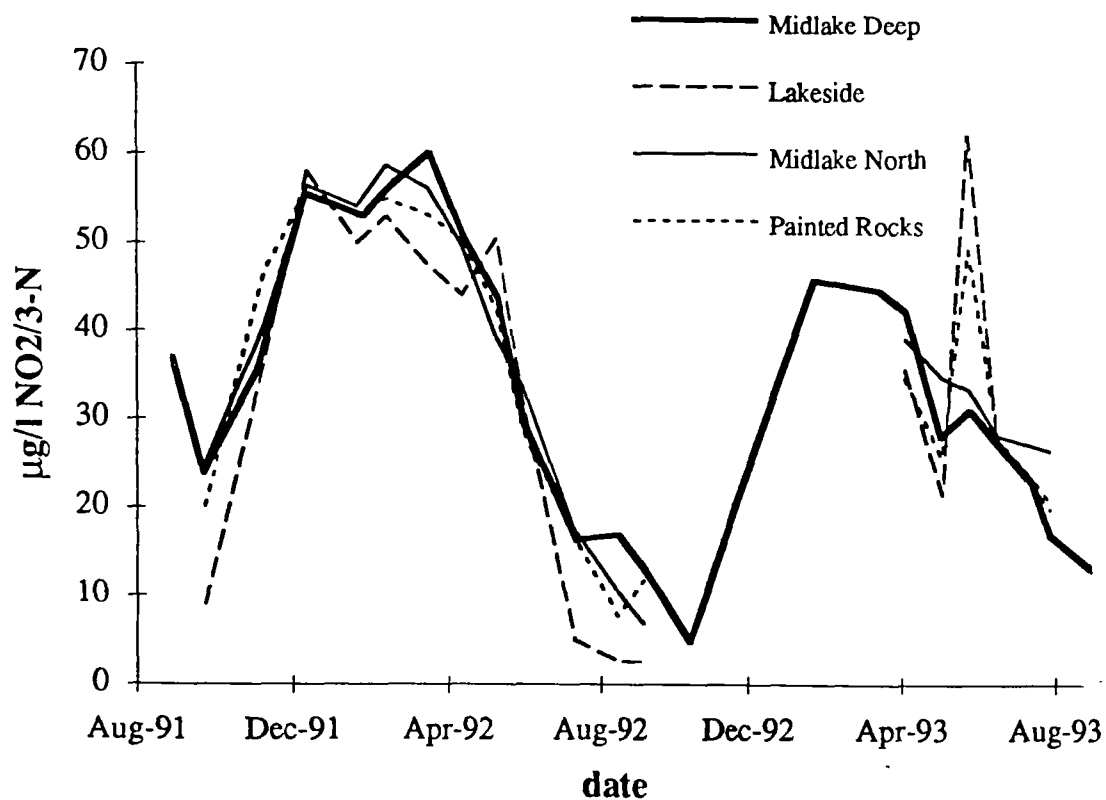
Figure 8. Intersite comparisons of nitrate+nitrite (μg/l-N) within the photic zone of the water column (i.e., integrated samples from 0-30m, or bottom) for the period of record in the clean lakes diagnostic study.
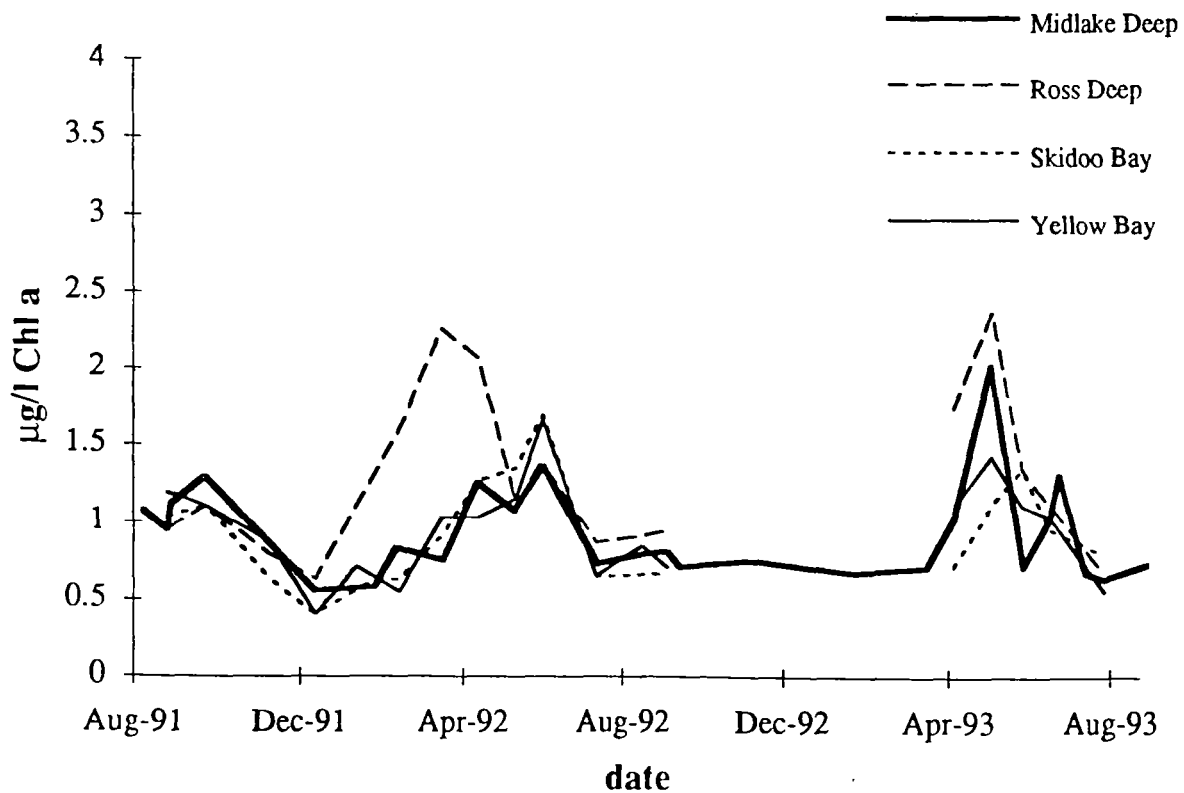
Figure 9. Intersite comparisons of chlorophyll *a* (μg/l-Chl *a*) within the photic zone of the water column (i.e., integrated samples from 0-30m, or bottom) for the period of record in the clean lakes diagnostic study.
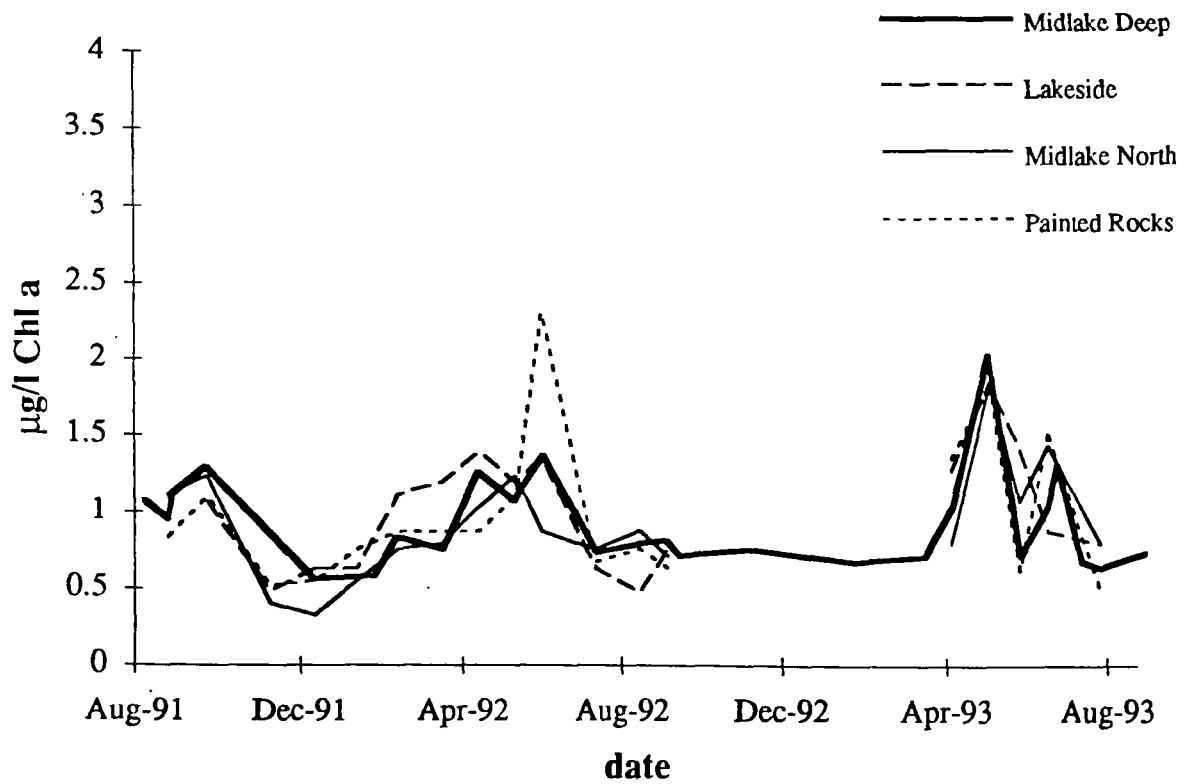
water quality. Otherwise, except for a very few instances, the chlorophyll data at other sites were remarkably similar to midlake.

Ashley Creek Sediment Assays

Phosphorus and nitrogen loading to Ashley Creek from the Kalispell sewage treatment plant was very high prior to 1989; thereafter concentrations decreased (Figure 10) in response to the ban on phosphorus containing detergents and better technology at the treatment plant. Although no measurements have been made of the P and N content of the Ashley Creek sediments, they are also believed to be very high. The concern in this portion of the study was that a reduction in the ambient nutrient concentrations in the water column due to improved removal of phosphorus and possibly nitrogen by the Kalispell STP may accelerate the desorption of nutrients from the streambed and offset the gains made by improving the phosphorus removal technology.

Initial Experiment - 13 November 1992

The greatest concentrations of SRP, $NO_{2/3}$-N and NH_3-N were observed at the end of the first 2 hour incubation (Figs. 11 and 12) indicating an initial dramatic release of both nitrogen and phosphorus from the sediments. SRP concentrations remained above the synthetic water reagent concentration of 19.2 μg/L until the 10 hour interval. The reduction of SRP below the level of added SRP indicated that either adsorption or microbial uptake of SRP occurred 8 - 10 hours after addition of the reagent to the sediment core. Release of SRP into the water column increased after 10 hours and continued to steadily increase throughout the experiment. The concentration of SRP was 15 μg/l higher than reagent concentration at the final 72-hour interval. This discovery led to an increase in the duration of the second experiment to assess the extent of desorption.

Release of NH_3 from the sediments into the water column was extremely high throughout the incubation (Figure 11). Unlike the SRP incubation, no NH_3 was added to the reagent water; analysis confirmed levels of NH_3-N below the detection limits (<0.5 μg/l) in the reagent water.
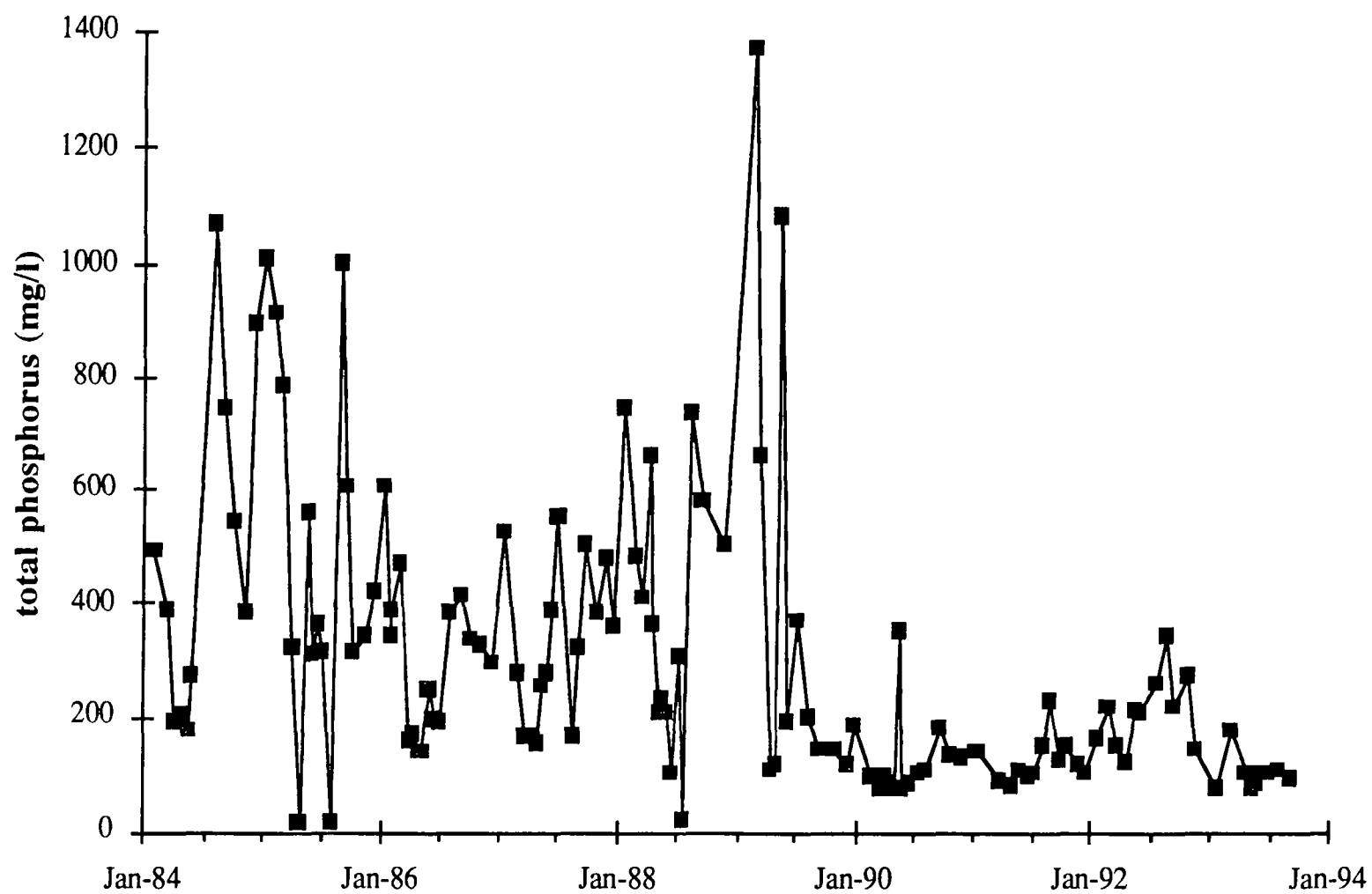
Figure 10. Total phosphorus (mg/l-P) in Ashley Creek downstream from the Kalispell sewage treatment plant.

After only 2 hours of incubation, 3538 μg/l NH_3-N was detected in the reagent water and levels never dropped below 1500 μg/l during the experiment. The release of NH_3 decreased after the initial pulse and did not begin to increase until after the 8-hour interval, similar to the pattern observed for SRP. NH_3-N concentrations were still increasing in the water column at the end of the incubation, with a final value of 2646 μg/l measured after 72 hours.

Nitrate+nitrite nitrogen was not added to the reagent solution and analysis confirmed a concentration below detection limits (<2.0 μg/l). Approximately 239 μg/l $NO_{2/3}$-N was released from the Ashley Creek sediments during the first 2 hours of the incubation (Figure 12). A rapid reduction in the release of $NO_{2/3}$ was observed after 2 hours and concentrations in the water column were down to 13 μg/l by 8 hours. From the 17 to 72 hour intervals, $NO_{2/3}$-N was concentrations were steady at approximately 14 μg/l. Although the intervals of incubation varied greatly in length (i.e., 2 - 20 hours), a fairly constant level of $NO_{2/3}$ was released into solution.

Second Experiment - 25 March 1993

The experimental design for the second core incubation was modified to include an appropriate equilibration solution by using Ashley Creek water collected above the wastewater treatment facility. By using filtered creek water collected during the period of core sampling, ambient concentrations of SRP, $NO_{2/3}$ and NH_3 as well as important sorption variables such as ionic strength, specific ionic concentrations, dissolved oxygen, and pH were represented. Such a change in conditions served to dramatically affect the uptake and release of nutrients observed in the second experiment.

The ambient concentration of SRP in Ashley Creek at the site below the Kalispell STP during the time of sediment core sampling was 119 μg/l. The ambient concentration of SRP in the reagent water prepared for incubation (i.e., filtered water from Ashley Creek above the Kalispell STP) ranged from 6.3 to 7.6 μg/l (Figure 13). Release of SRP from the sediment cores was somewhat variable within the first 24 hours . Release of SRP from core A into the reagent creek water peaked at 147 μg/l at 6 hours while core D water peaked at 47 μg/l at 24 hours. Levels in the
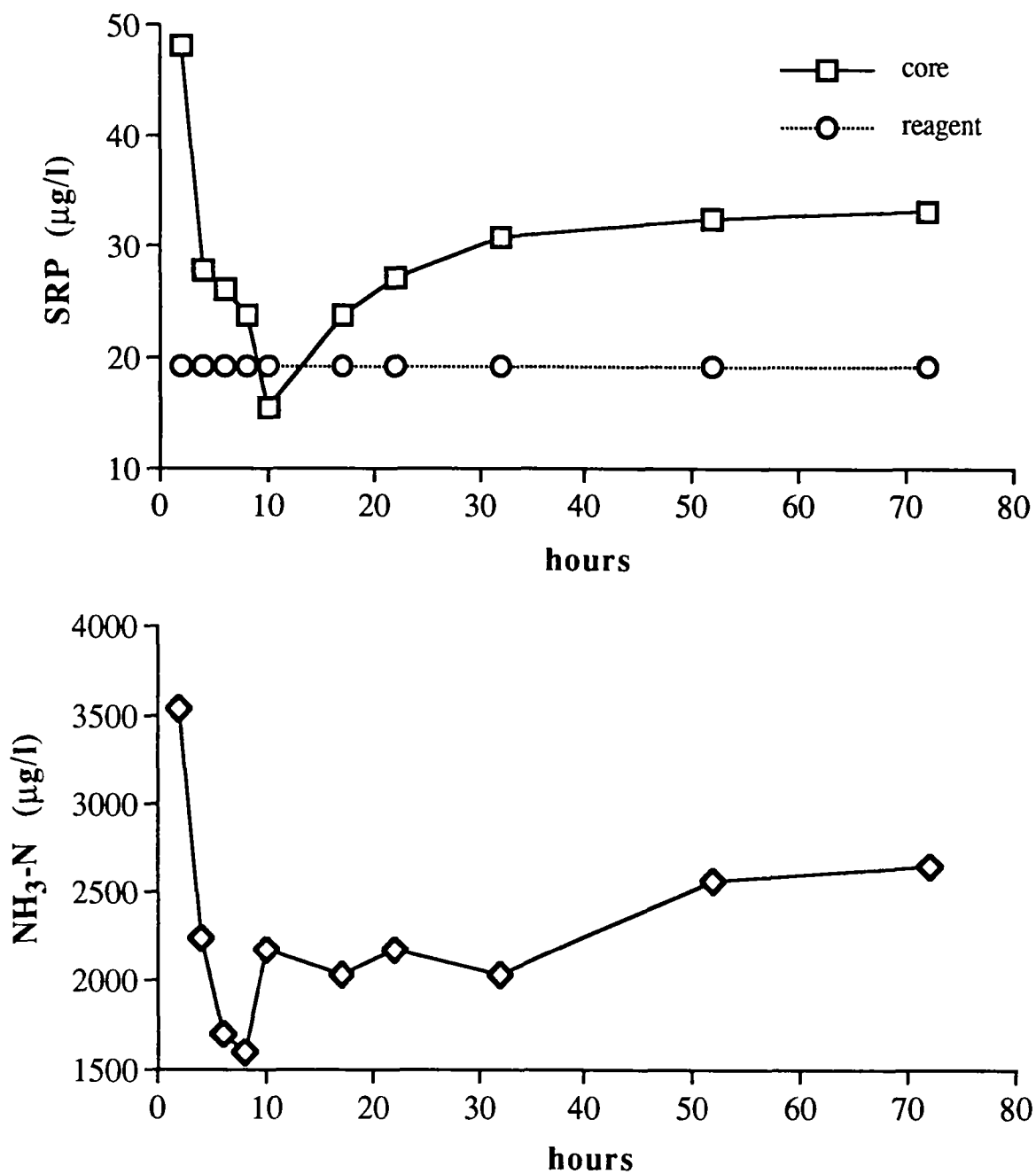
Figure 11. Concentration of SRP (upper graph) and NH3-N (lower graph) in water samples collected at varying intervals from the water column above a core of Ashley Creek sediments. The water reagent was amended with 19.2 μg/l SRP.
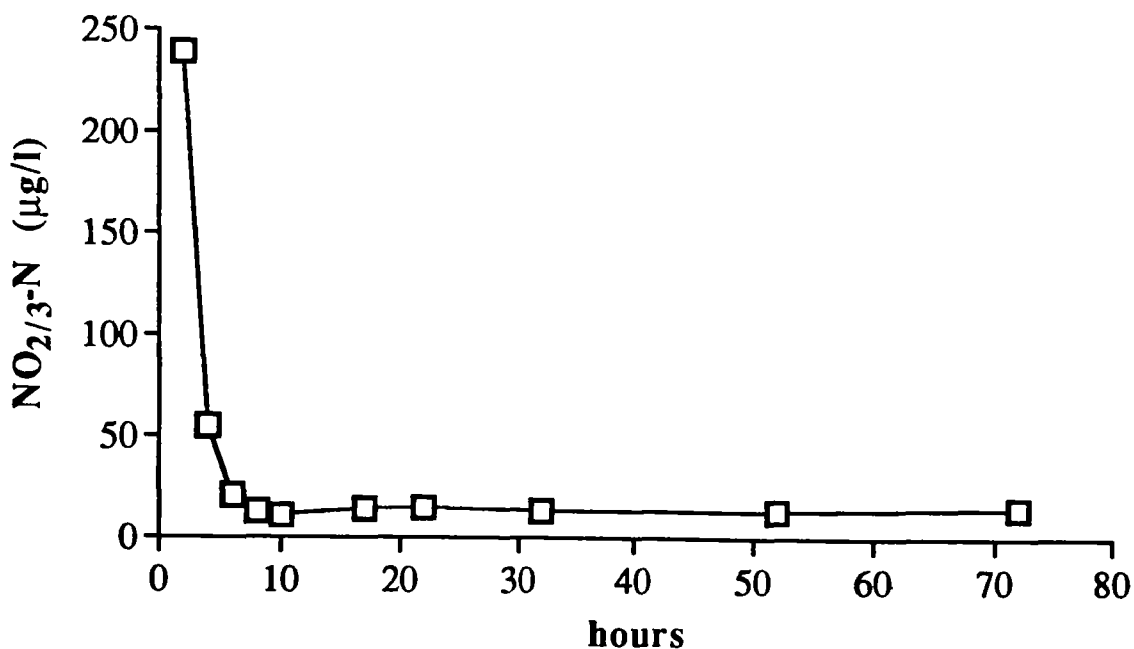
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Figure 12. Concentration of NO$_{2/3}$-N in water samples collected at varying intervals from the water column above a core of Ashley Creek sediments.

other two cores peaked at only twice the level present in the reagent creek water. Concentrations

of SRP in water from all cores decreased to concentrations below or near ambient conditions (i.e.,

reagent creek water concentration) as early as 45 hours after initial incubation. SRP concentrations

in cores C and D remained below ambient levels throughout the remainder of the incubation,

indicating possible adsorption or uptake of SRP by the sediment column. Core A concentrations

remained above ambient SRP, while core B concentrations fluctuated below and then above

ambient levels.

The ambient concentration of NH$_3$-N in Ashley Creek below the STP at the time of

sediment core sampling was 167 µg/l. Reagent creek water collected three times during the period

of incubation indicated quite variable NH$_3$-N levels at the site above the STP, ranging from 103

µg/l for the first 45 hours, to 83.7 during the next 48 hours, to 52.7 by the 106 hour interval

(Figure 13). During the first 45 hours of the incubation, when the reagent creek water had a high
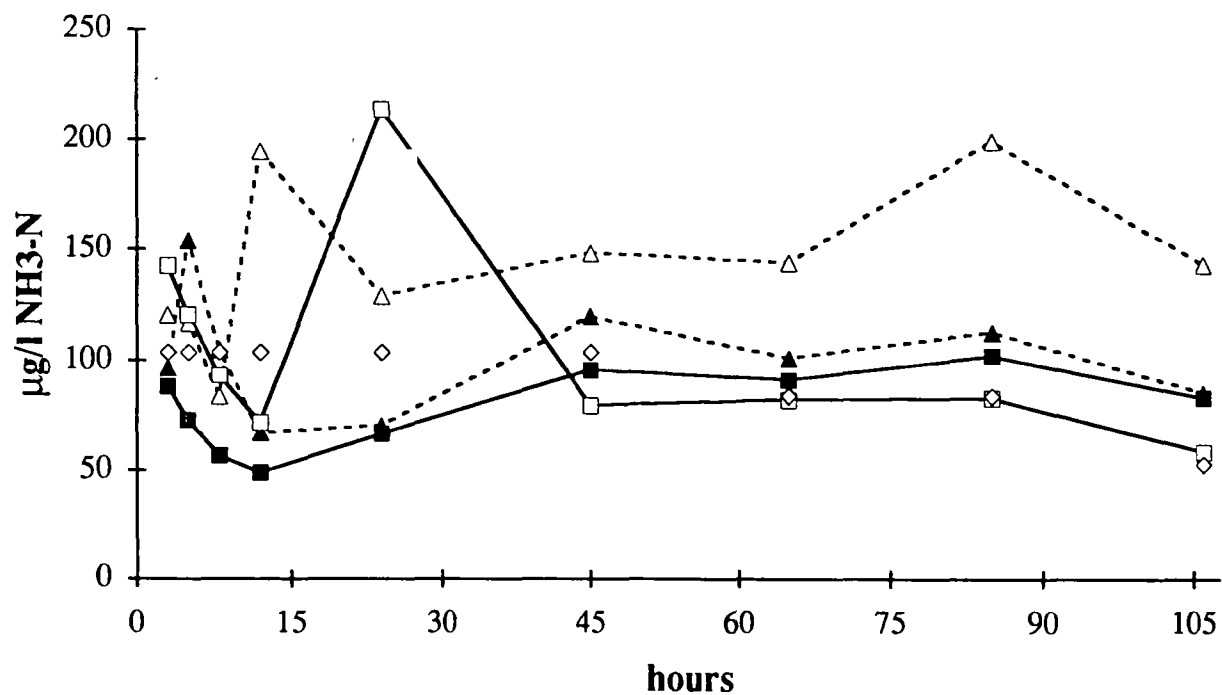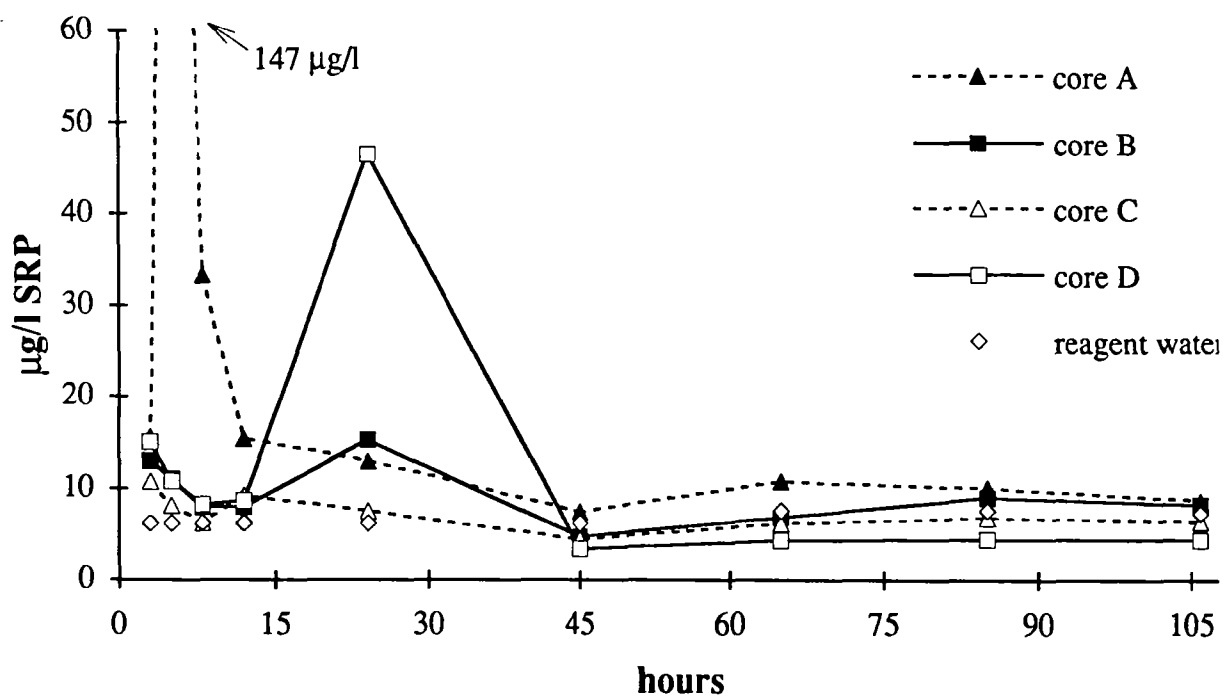
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Figure 13. Concentration of SRP (upper graph) and NH3-N (lower graph) in water samples collected at varying intervals from the water column above 4 cores of Ashley Creek sediments. The concentration of filtered Ashley Creek water (reagent water) used in the experiment is also given.

ambient level of NH_3-N (103 µg/l), levels of NH_3 increased above ambient concentrations in some

core incubations and decreased below ambient in others. After 45 hours, the reagent creek water

used in the incubations contained less NH_3 and the response appeared to be less variable, as three

of the four cores exhibited a release of NH_3 from the sediments (i.e., 7 to 90 µg/l above ambient,

excluding core C at 85 hours). The peak in NH_3 release in core C at the 85 hour interval was

likely due to accidental disturbance of the sediment/water interface during extraction of the sample

from the core incubation apparatus.

The ambient concentration of $NO_{2/3}$ -N in Ashley Creek below the STP at the time of

sediment core sampling was 468 µg/l, while the ambient concentration in the reagent water

collected above the STP was significantly lower but did increase throughout the period, ranging

from 39.9 to 57.1 µg/l (Figure 14). Concomitant with the increase in $NO_{2/3}$ concentrations

observed at the upper site was a steady decrease in NH_3 (Figure 13). The greatest release of

$NO_{2/3}$ from the sediment cores was observed during the first 3 hours, when $NO_{2/3}$ -N in the

reagent creek water ranged from 25 to 46 µg/l above ambient concentrations. A significant release

of $NO_{2/3}$ from the sediments continued in all core incubations throughout the 45 hour interval.

Levels of $NO_{2/3}$ in the core incubations were at or near ambient reagent water concentrations after

45 hours. However, during this period higher $NO_{2/3}$-N was present in the reagent water. The

slight increase in $NO_{2/3}$ level in core C at 85 hours may have been due to disturbance of the

sediment/water interface during extraction of the water sample.

Implications of the Sediment Assays

Bottom sediments are a potential source for regulation of soluble nitrogen and phosphorus

concentrations in freshwater streams. Several mechanisms regulate the exchange of nutrients

across the sediment/water interface, which may include the ambient nutrient concentration in both

sediment and water, oxygen-dependent interactions, ion-exchange processes and activities of

bacteria, fungi, invertebrates and algae in both the sediments and the water column. The nitrogen

dynamics of the sediments are not well understood. Several studies have characterized the
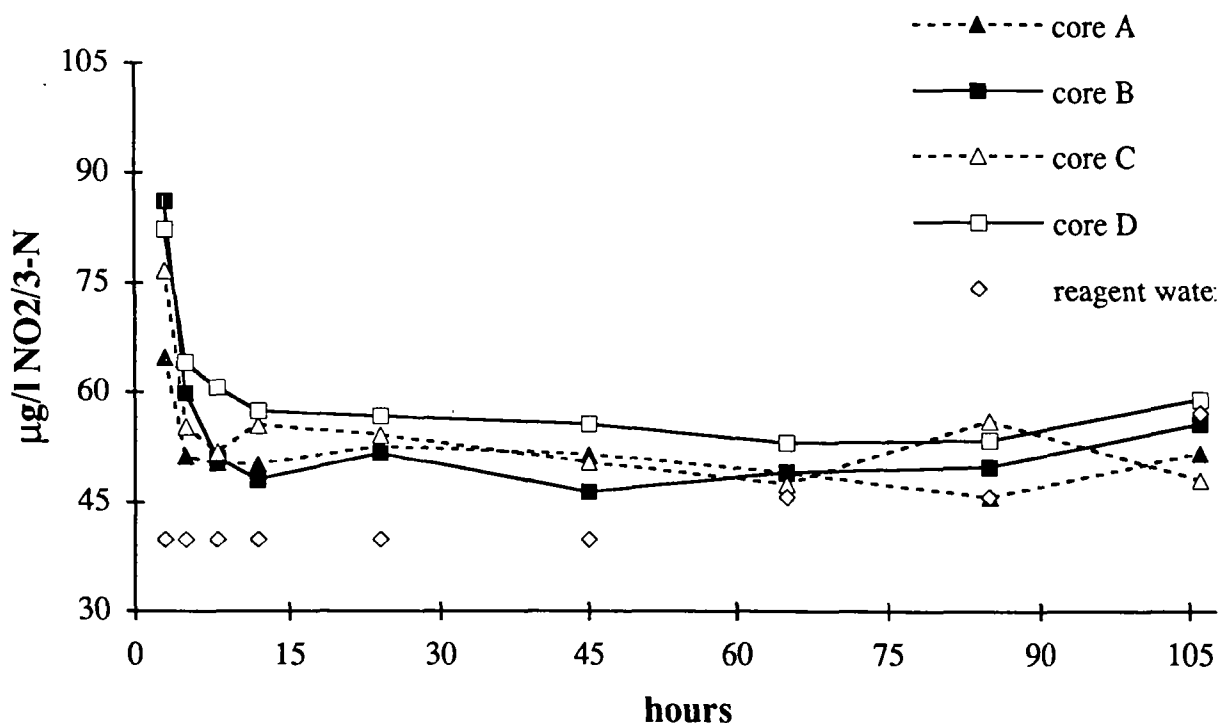
Figure 14. Concentration of $NO_{2/3}$-N in water samples collected at varying intervals from the water column above 4 cores of Ashley Creek sediments. The concentration of filtered Ashley Creek water (reagent water) used in the experiment is also given.

phosphorus sorption-desorption processes in specific stream sediments (Green et al. 1978, Klotz 1988). Most studies have quantified the phosphate sorption index or the equilibrium phosphorus concentration by suspending sediments in various solutions over a given time period (usually 1 hour) and measuring the concentration of phosphorus at the end of the incubation. Few studies have attempted to characterize sorption and release of both nitrogen and phosphorus using intact streambed cores.

The initial core incubation using a synthetic water reagent indicated that the bottom sediments from Ashley Creek were capable of releasing significant quantities of soluble phosphorus and nitrogen. The greatest release of all the nutrients was within the first 2 hours of the incubation. Although the ambient concentration of SRP in the synthetic reagent was higher than that in the reagent creek water used in the second incubation (i.e., 19 versus 7 µg/l), the sediments of the preliminary experiment continued to release SRP (15 µg/l higher than ambient at

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72 hours), whereas release and adsorption or uptake of SRP was near equilibrium in the second incubation within 45 hours. The initial release of SRP was 30 $\mu g/l$ higher than ambient using the synthetic reagent, while initial release was as high as 147 $\mu g/l$ using natural stream water. The dissolved oxygen concentration in the preliminary incubation using the synthetic reagent ranged from 69% at 0 hours to 34.7% saturation at 72 hours, whereas the dissolved oxygen concentration in the second incubation was 49.3% ($\pm$ 5%) saturation. Perhaps the greater reduction in dissolved oxygen in the preliminary experiment resulted in greater release of SRP from the sediment core.

The effectiveness of the oxidized microzone of the surface layer of bottom sediments in preventing significant release of phosphorus from the porewaters of the sediments to the overlying water column has been well documented (Mortimer 1971). The oxidized layer serves as a trap for manganese, iron and phosphate, and thus greatly reduces transport of materials into the water. At the sediment surface, a difference of a few millimeters in oxygen penetration is critical in regulating the exchange of materials between the sediment and water. Oxygen penetration into the sediments is governed by the oxygen content of the overlying water, turbulent mixing of superficial sediments and the oxygen demand of the sediments. Adsorbed phosphate, manganese and iron are quickly mobilized when the redox potential decreases, as the oxidized microzone barrier weakens. If the core incubations were repeated under anaerobic conditions, it is very likely that enormous loading of phosphorus would occur. Continuous recording of dissolved oxygen concentrations in Ashley Creek in combination with additional core incubation experiments under anaerobic conditions would provide valuable information concerning possible loading of SRP from Ashley Creek sediments. However, under aerobic conditions the sediments may serve to regulate concentrations of SRP in the water column. Such regulation has been suggested in other studies (e.g., Klotz 1988).

The initial core incubations utilizing the synthetic reagent may provide an indication of the maximum amount of NH_3 and $NO_{2/3}$ that would be released from Ashley Creek sediments. The synthetic reagent did not contain any NH_3 or $NO_{2/3}$ and it appeared that under these conditions a massive amount of NH_3 was released (3538 $\mu g/l$ initially and 2646 $\mu g/l$ at 72 hours). When a high

ambient concentration of NH_3 was present in the water column, as in the second incubation, the response was variable; but when the ambient level of NH_3 was reduced during that incubation, more NH_3 was released from the sediments (i.e., up to 90 µg/l).

Although the magnitude was not the same as that observed for NH_3, the response of $NO_{2/3}$ to ambient levels of nutrients in the water column was similar. In the incubation with the synthetic reagent (no $NO_{2/3}$), 239 µg/l was released in the first 2 hours, then a steady release of about 14 µg/l was observed from 17 to 72 hours. However, in the incubation utilizing natural stream water (40-57 µg/l ambient $NO_{2/3}$) the initial release was only 25-46 µg/l in the first 3 hours. Continual release of $NO_{2/3}$ was observed while ambient concentrations in the water column were at their lowest, but when the levels increased, much less $NO_{2/3}$ was released. In fact, when ambient concentrations were at their highest, three of the cores showed loss of $NO_{2/3}$ from solution.

Much of the nitrogen present in sediments of fresh waters is immobilized and sorbed to inorganic particles. The porewaters of the sediments usually contain a much higher concentration of NH_3-N and organic N than that of the overlying water. The oxidized microzone of the sediments is crucial to the solubility and sorption properties of the sediments for ammonia and can alter the rates of microbial transformations (Wetzel 1983). NH_3 can accumulate when appreciable amounts of organic matter are deposited. Under anaerobic conditions, bacterial nitrification of NH_3 to $NO_{2/3}$ ceases and with the loss of the oxidized microzone a marked release of NH_3 from the sediments can occur. Nitrate may diffuse into the water column following bacterial nitrification in well-oxygenated surficial sediments. The massive release of ammonia observed in the incubations is perplexing if the microzone was indeed oxygenated. Water samples from the core indicated an oxygenated water column, but lack of circulation as would occur in the creek may have resulted in a reduction of oxygen at the interface. However, a greater release of SRP would be expected under such conditions.

The regulation of NH_3 and $NO_{2/3}$ by sediments is poorly understood (Wetzel 1983). Conditions in a flowing stream are obviously quite different than the conditions presented in this experiment. The degree of turbulence at the sediment/water interface caused by stream flow would

no doubt affect the extent of oxygenation of the surface sediments and thus the many chemical, physical and biological mechanisms controlling the release, sorption or uptake of nutrients.

Long Term Phosphorus Loading and Pelagic Primary Productivity in Flathead Lake

Over the period of record, pelagic primary production measured annually at the midlake site increased (Figure 15). Until 1989, the rate of increase was very steep corresponding to at least two major limnological phenomena. First, note that inputs of P and N from the sewage treatment plants in the basin were very high during 1977 - 1988 and runoff was below normal most years. Second, establishment of *Mysis relicta* in the lake in 1981 and subsequent, rapid expansion of the population to peak numbers in 1986 and 1987 reduced the crustacean zooplankton to very low levels (Spencer *et al.* 1991). Crustacean zooplankton numbers remained low in comparison to pre-*Mysis* years and significant shifts in the abundance of zooplankton species occurred, with the disappearance of two species in 1988 and the appearance of a new species in April 1991 (Chess and Stanford, unpubl.). Hence, substantially reduced herbivory related to the trophic cascade caused by the establishment of *Mysis* may explain the high level of primary production in 1988. Chlorophyll values reached a maximum in 1988 as well (Figure 16). After 1987, the *Mysis* populations declined dramatically and numbers stabilized, possibly in relation to size selective predation by fishes or a density-dependent effect from low zooplankton numbers. Primary productivity also declined and stabilized (Figure 15). Indeed, inclusion of the 1993 primary production data reduced the statistical significance of the linear relation in the long term data to $P < 0.1$. Prior to 1993 the increasing trend in primary production over the period of record was highly significant ($P < 0.01$).

Were the observed dynamics in pelagic primary productivity due to food web effects or nutrient loading or both? We examined the annual phosphorus flux through the lake for clues to this question.
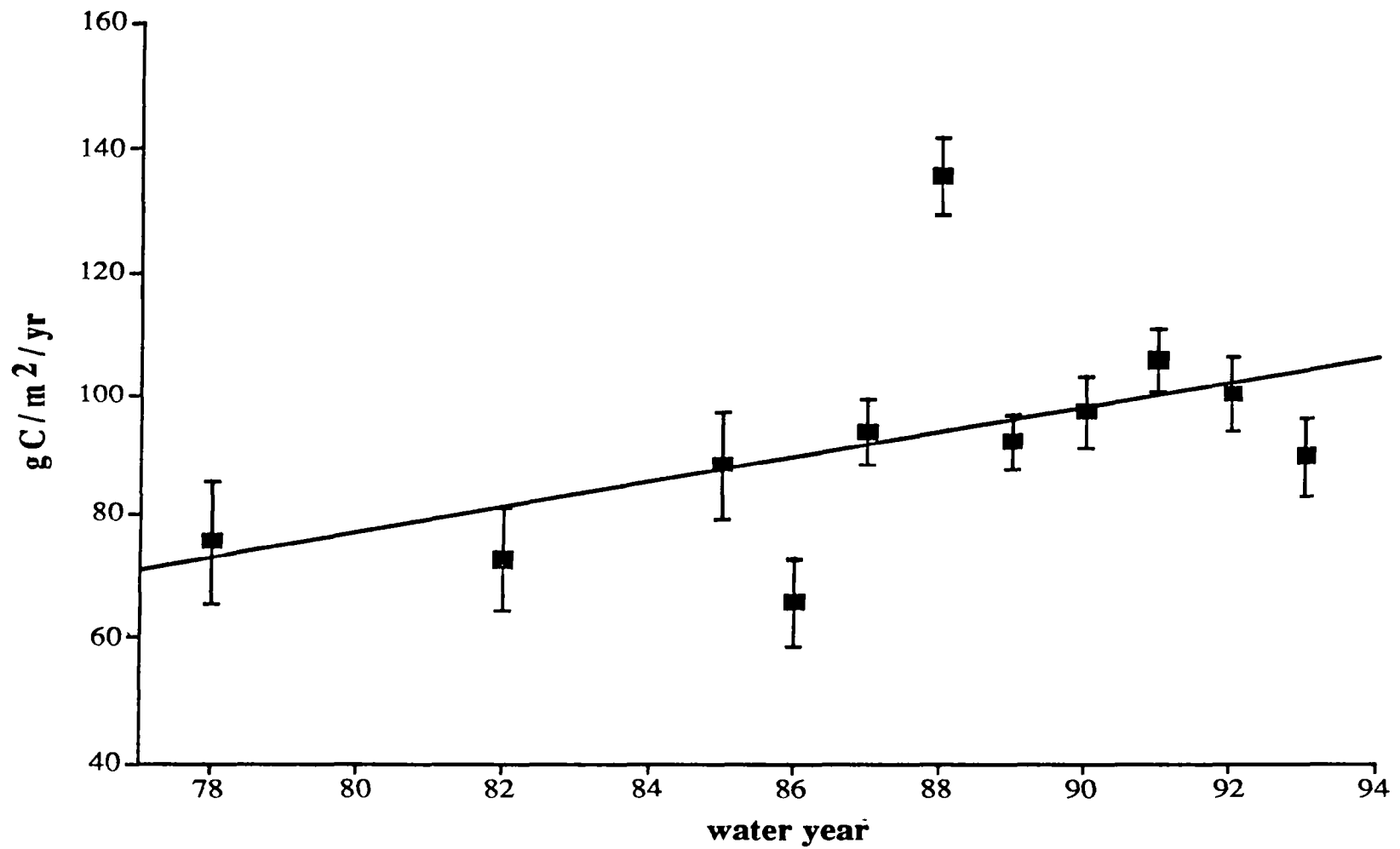
Figure 15. Mean annual pelagic primary productivity over the period of record at the midlake site (E, Figure 2). Bars represent minimum and maximum yearly estimates.
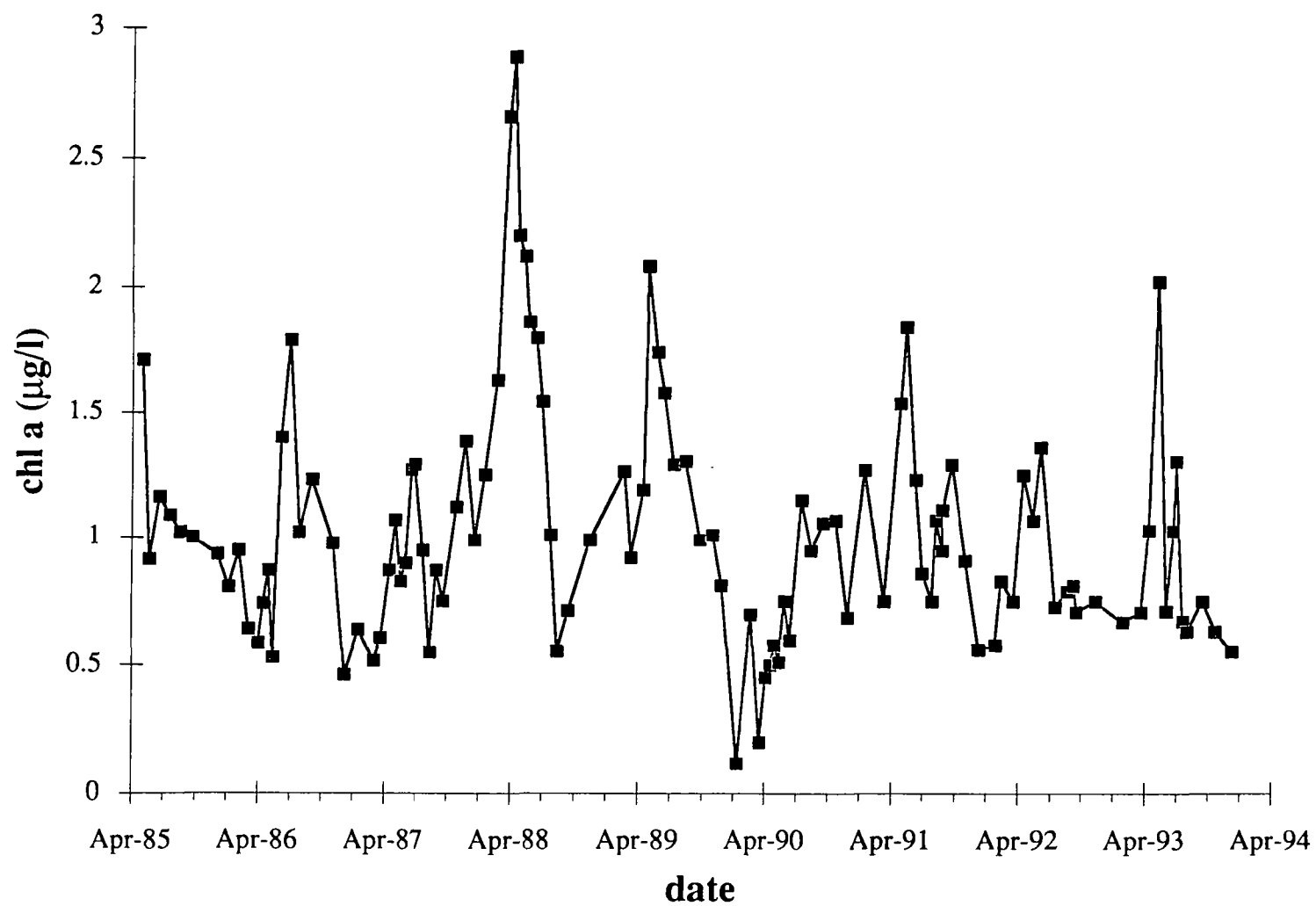
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Figure 16. Concentrations of chlorophyll *a* in the photic zone (0-30m, integrated samples) over the period of record of Flathead Lake at the midlake site (E, Figure 2).

Daily input of bioavailable phosphorus from the Flathead River was distinctly correlated with flood flows (Figure 17), as expected. And, a general tendency, although not statistically significant, for higher TP values in the lake to occur in years of high inflow was observed, especially in the surface waters (Figure 18).

In all years the Flathead River was the largest contributor of bioavailable phosphorus to Flathead Lake (Figure 19). However, bulk precipitation was a major source as well (10 - 38% of the total mass input); and, while total load was clearly related to the Flathead River flow (i.e., total load was highest in the years of highest runoff), inputs from bulk precipitation were independent of riverine water yield. In fact, bulk precipitation load was highest in low water years.

Over half of the incoming load of bioavailable phosphorus each year was retained in the lake (Figure 20). Phosphorus annually retained in the lake is stored in the bottom sediments at the mud-water interface and is held there by the oxidation equilibrium (Ellis and Stanford 1988), as noted above for Ashley Creek. Should the water column above the sediments become anoxic, thereby reversing the redox gradient, a substantial amount of the stored phosphorus would be released back into the lake and water quality could decline precipitously (Wetzel 1983). This further underscores the concern for the hypolimnetic oxygen deficit observed in Ross Deep, although values observed to date do not approach anoxia.

Stoner Creek, the largest shoreline tributary, was not a significant source of bioavailable phosphorus on a lake wide basis. We made no attempt to add other lake shore sources into the P mass flux calculations because the data were insufficient for accurate load calculations and the very small additional loads clearly would not change the observed pattern in annual loading. We also assumed that ground water input was minimal. However, inputs of labile phosphorus and nitrogen from shoreline sources, while small in comparison to lake wide mass flux, do produce localized bursts of bioproduction, particularly in the form of benthic algae mats. Indeed, on several instances sewage breakouts have been discovered by presence of these mats (Stanford *et al.* unpubl.). Shoreline nutrient pollution is a problem in Flathead Lake in some areas, such as Big Arm Bay (Hauer 1988), and should not be trivialized by the fact that shoreline sources of
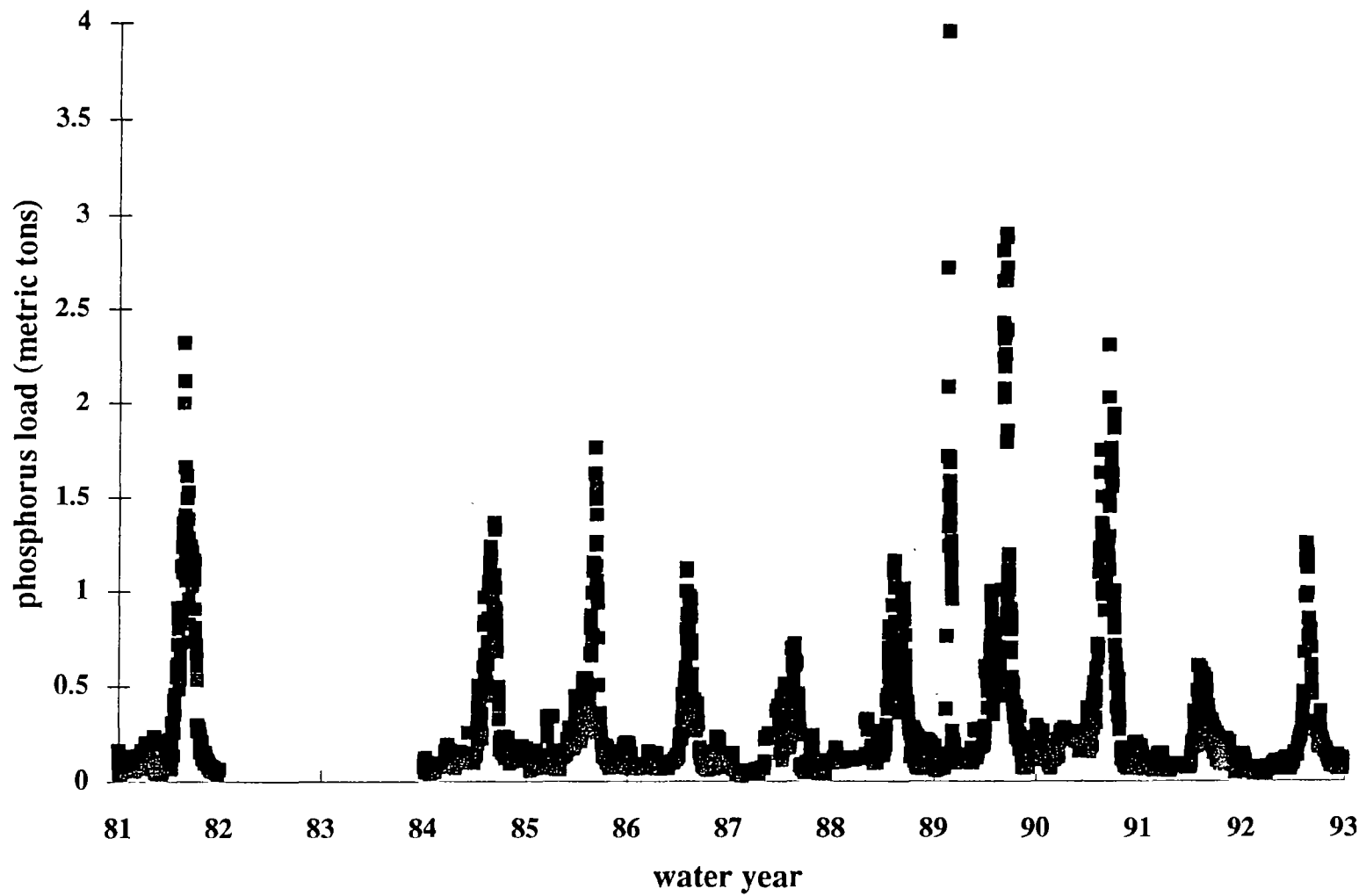
Figure 17. Mass of biologically active phosphorus discharged daily over the period of record into Flathead Lake by the Flathead River, based on measures made in the river at the Holt sampling site.
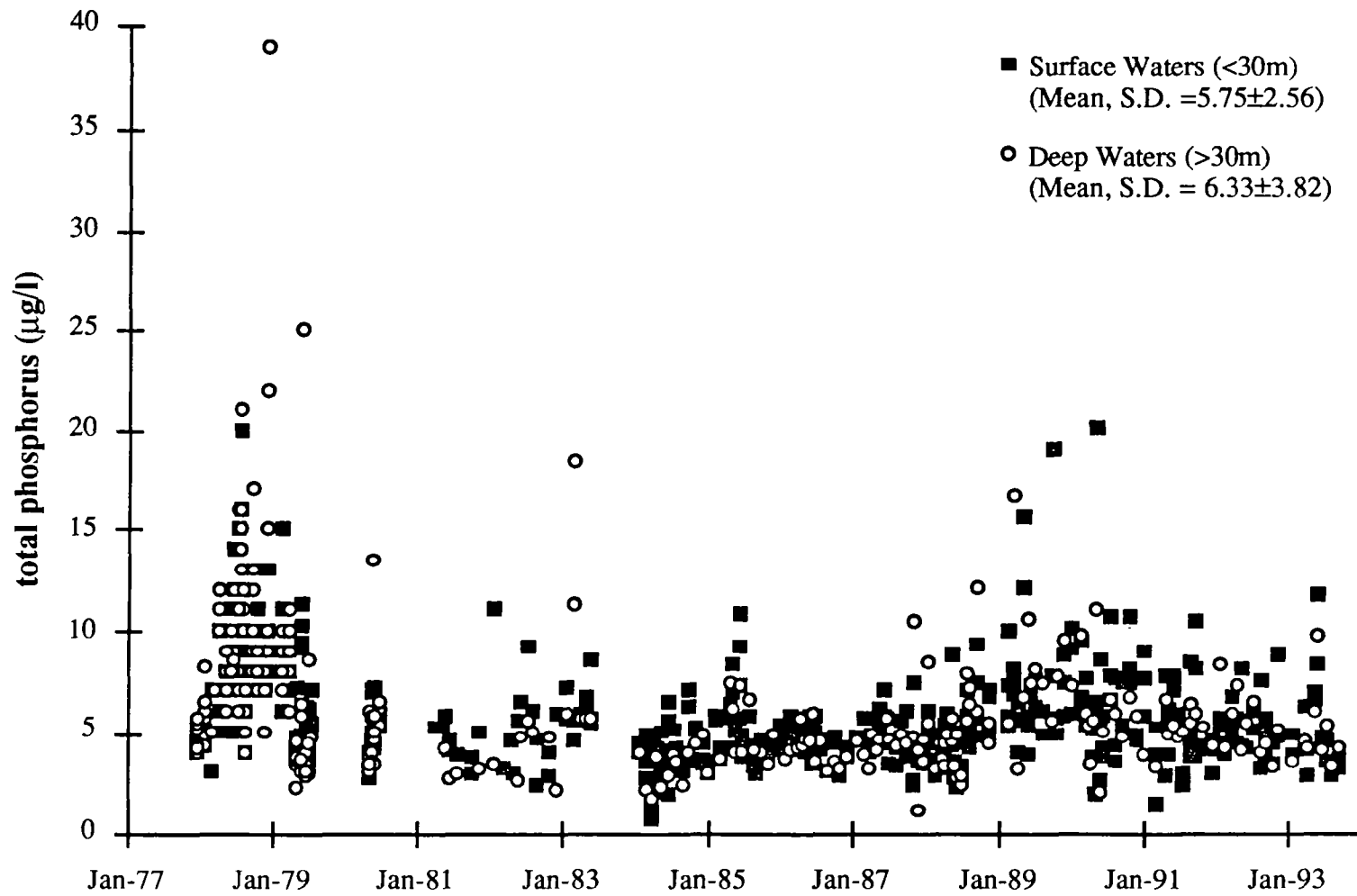
Figure 18. Concentrations of total phosphorus observed in the photic and aphotic zones of Flathead Lake over the period of record. Box shows the long-term means and standard deviations.
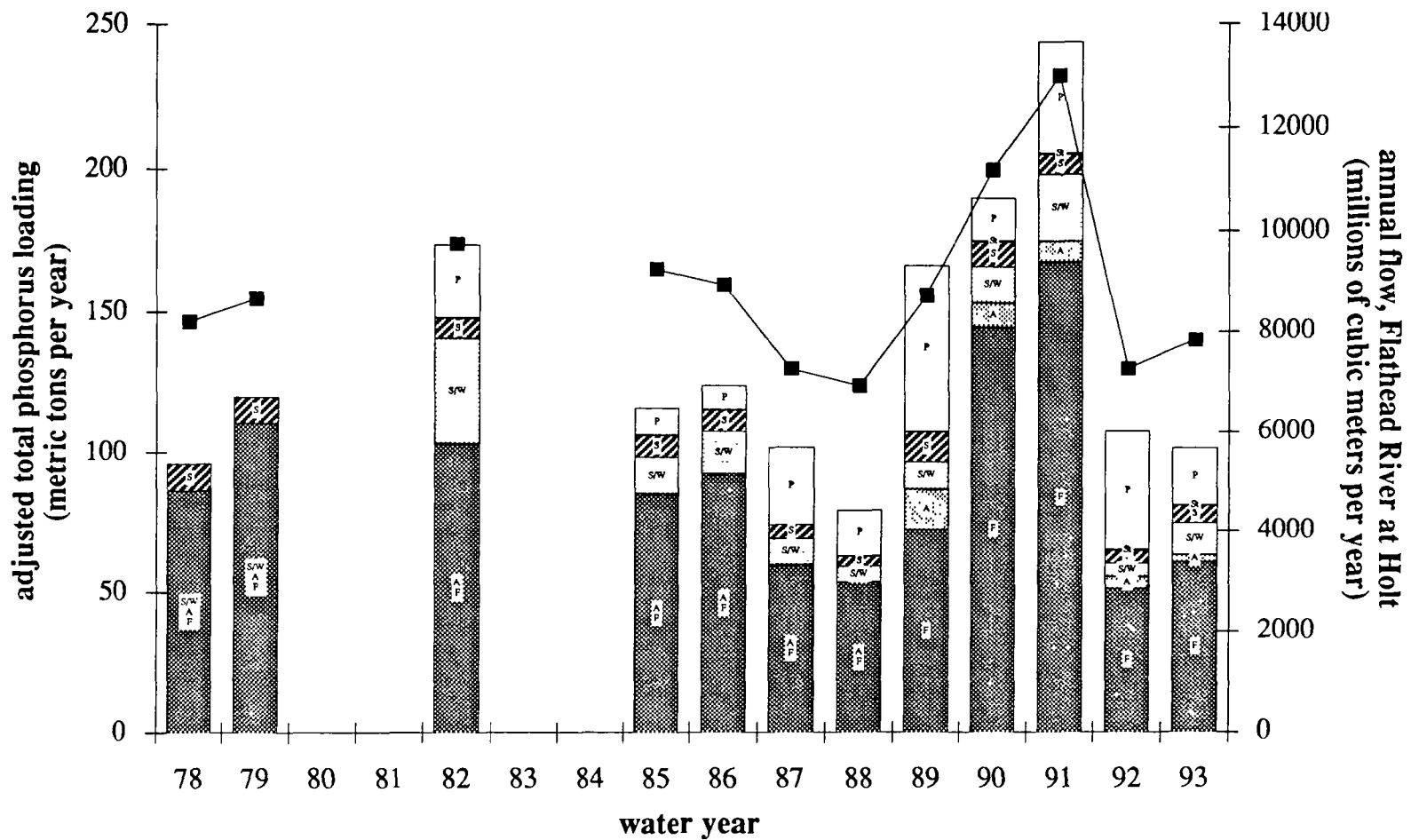
Figure 19. Mass of biologically available phosphorus by source (histograms) reaching Flathead Lake annually in relation to annual inflow from the Flathead River (closed squares). S = Swan River, S/W = Stillwater River (below confluence with the Whitefish River), F = Flathead River (at Holt), A = Ashley Creek (from 1978 - 1988 Ashley Creek values were included in the Flathead River values because flow data were unavailable for the Creek), P = bulk precipitation, St = Stoner Creek.

Figure 20. Phosphorus (corrected for bioavailability) mass balance for Flathead Lake (i.e., solid histograms represent annual phosphorus export whereas the open histograms represent retention in the lake). Asterisk denotes a mass balance calculation which lacked precipitation data.

bioavailable P are very small in terms of whole lake flux. Moreover, shoreline periphyton growth in the littoral zone of Flathead Lake is limited by the quantity of phosphorus in the water column (Bauman 1982, Marks and Lowe 1993) and lake shore residents often express concern over the very visible "ring a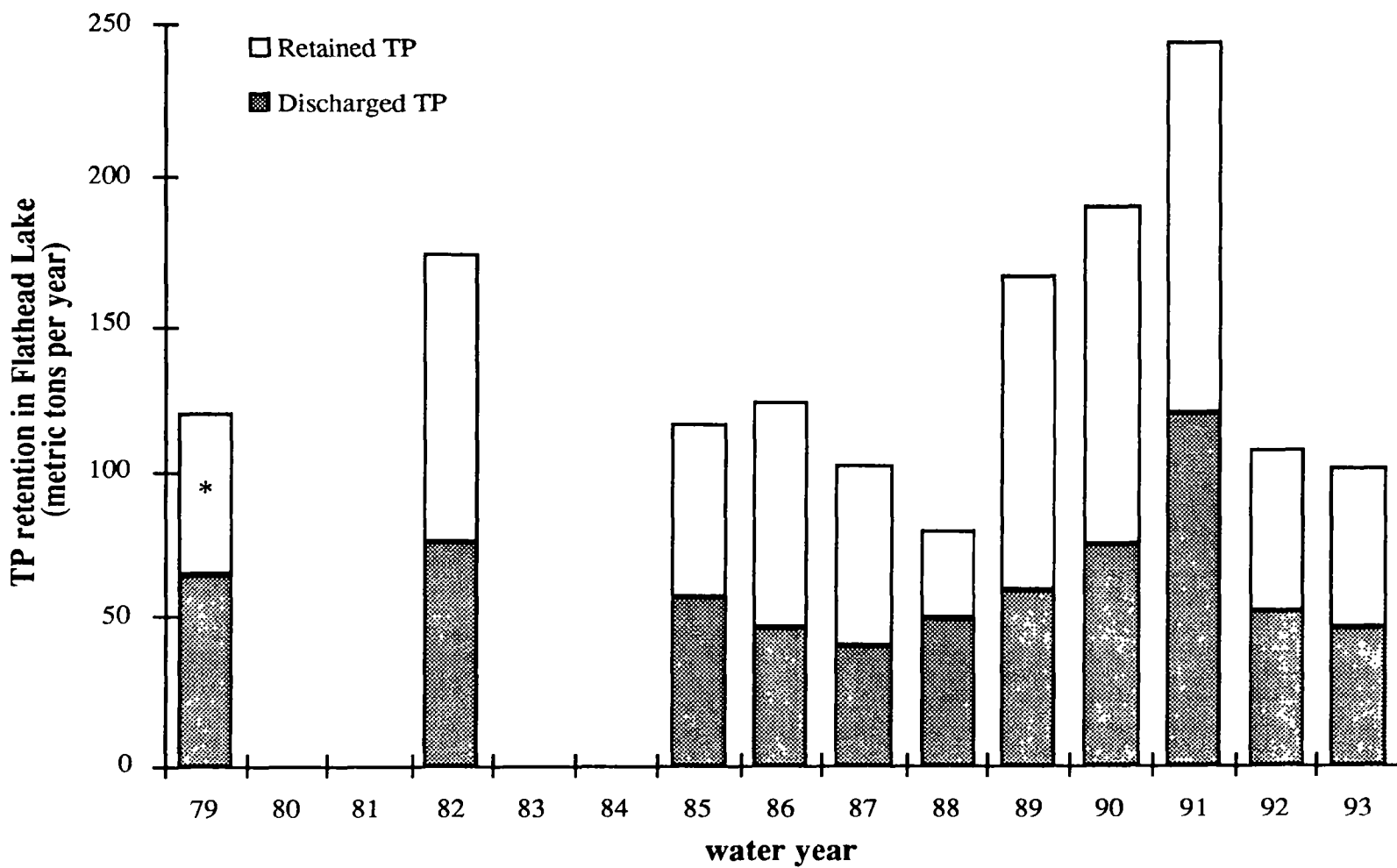round the lake" caused by periphyton growths. Long term data are not available for shoreline periphyton productivity but should be expected to respond to annual P flux in similar manner to pelagic primary productivity. An often-expressed complaint by long time shoreline residents is that the ring around the lake is significantly more visible in the last decade (comments made at Flathead Lakers annual meetings).

The dramatic reduction in point source loading of bioavailable phosphorus from the urban sewage treatment plants as a consequence of the P-reduction strategy (Figure 21) seems particularly important. Indeed, the percent of total load declined from well over 20% in 1978 to less than 5% since 1989 (Figure 21, top panel). Phosphorus in point source sewage discharges is 95+% bioavailable (Ellis and Stanford 1986). Reducing these sources of P loading may be a major reason why primary productivity stabilized in recent years. However, note the increase in total phosphorus loading from the Whitefish STP from 1989 to 1993. Although advances in the removal of P at the Whitefish STP have greatly reduced TP concentration in the effluent, population expansion in the Whitefish area has resulted in an increase in the total volume of sewage and thus the Whitefish STP accounts for about half the TP load from Flathead Valley sewage treatment plants.

We were unable to demonstrate a statistical relationship between pelagic primary production and bioavailable phosphorus load on the basis of annual load calculations (Figure 22). However, if elevated levels of productivity in 1988 were largely a result of the *Mysis* reducing crustacean zooplankton that eat algae, a relationship does exist. On the other hand, none of the many bioassays done *in vitro* suggest top down (consumer mediated) controls on primary productivity (Spencer 1992). As noted above with respect to the declining P loads from the sewage treatment plants, the pattern and timing of water delivery to the lake may also be important. In the 1990's the operation of Hungry Horse Dam was such that large volumes of water were delivered to the lake

during the summer stratification period (Stanford and Hauer 1992). Since the water was discharged from the bottom of the reservoir, the water mass arriving at Flathead Lake was significantly colder (7 - 10°C) and more dense than the surface layers of the lake (10 - 12°C). Hence, the river water had to flow under the epilimnion into the hypolimnion. Since the lake was full during the summer, considerable water from Kerr Dam had to be released to prevent lake shore flooding as summer releases from Hungry Horse arrived in the lake. That water had to come from the epilimnion of the lake. The result could have been enrichment of the hypolimnion which would produce effects after breakdown of thermal stratification or perhaps even the following spring. Clearly, the loading relationship is more complicated than can be discerned from annual loading calculations.

Moreover, primary production in Flathead Lake is influenced by processes other than water and nutrient flux. Depth and duration of the mixed layer in relation to patterns of thermal stratification, food web dynamics related to the establishment of opossum shrimp, *Mysis relicta*, (Spencer *et al.* 1991) and seasonal or shorter term dynamics in bioavailability of P, N and inorganic carbon in relation to photosynthetically active radiation (Dodds *et al.* 1989, Dodds *et al.* 1991, Dodds 1989, Ellis and Stanford 1986, Spencer and Ellis 1990, Stanford *et al.* unpubl. data) are some of the more important processes that have been examined quantitatively to date. Final determination of process and response relationships requires synthesis of these interactive determinants in relation to P loading in both short (seasonal) and long (annual) time scales. Nonetheless, based on all previous study and inferences of the data presented herein, our preliminary conclusion is that P flux likely is a primary determinant of pelagic primary production (and hence water quality), though clearly other biophysical factors are important.

The ultimate goal of the Clean Lakes Program at Flathead Lake is to develop and implement a plan for management of daily nutrient loads (N and P) from point (e.g., sewage treatment plants) and non-points (diffuse) sources for the purpose of preventing cultural eutrophication of Flathead Lake. The phosphorus reduction strategy implemented in 1984 by the Water Quality Bureau clearly is working (cf. Figure 21). However, volume of sewage reaching
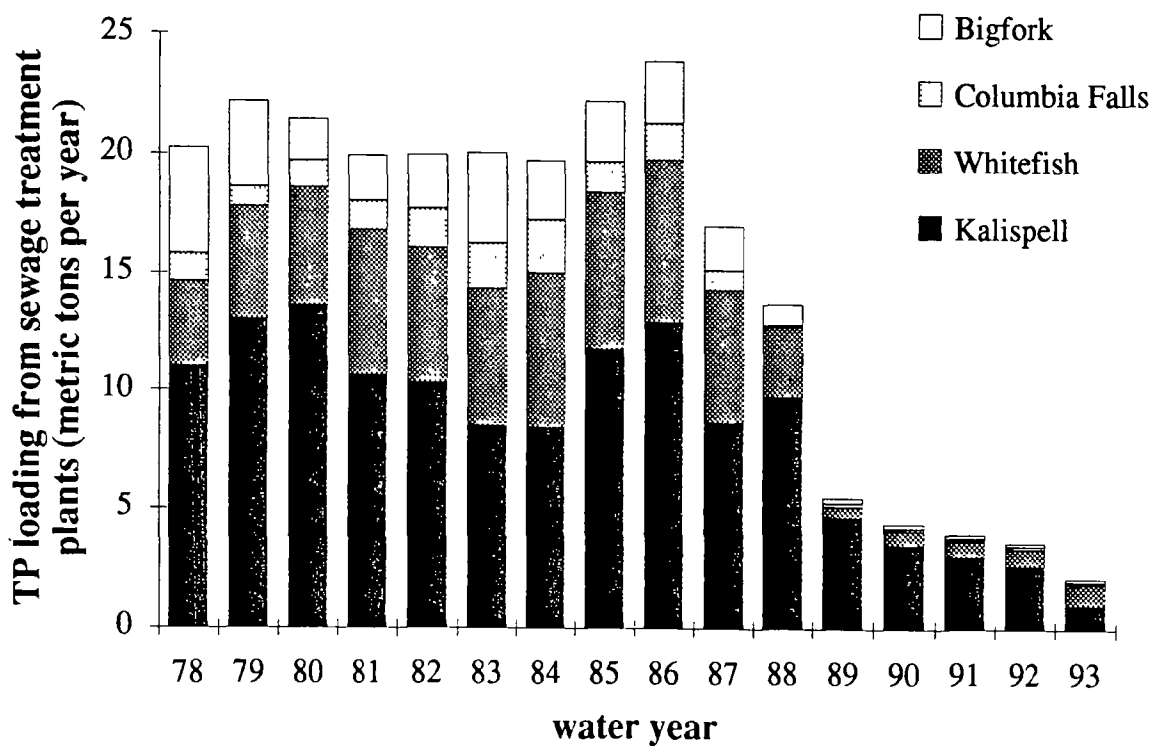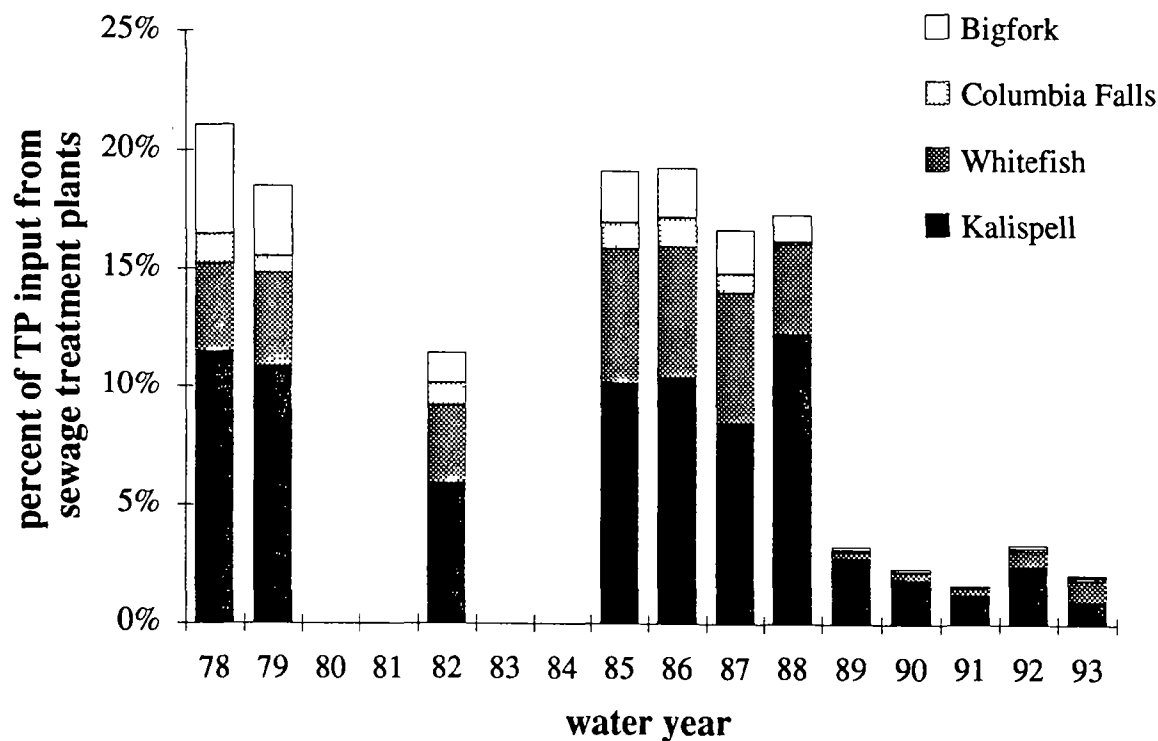
Figure 21. Mass of bioavailable phosphorus from the urban sewage treatment plants expressed as percent of the total annual mass input from all sources (top graph). Mass of bioavailable phosphorus from the urban sewage treatment plants discharged annually into Flathead Lake or its tributaries (bottom graph).
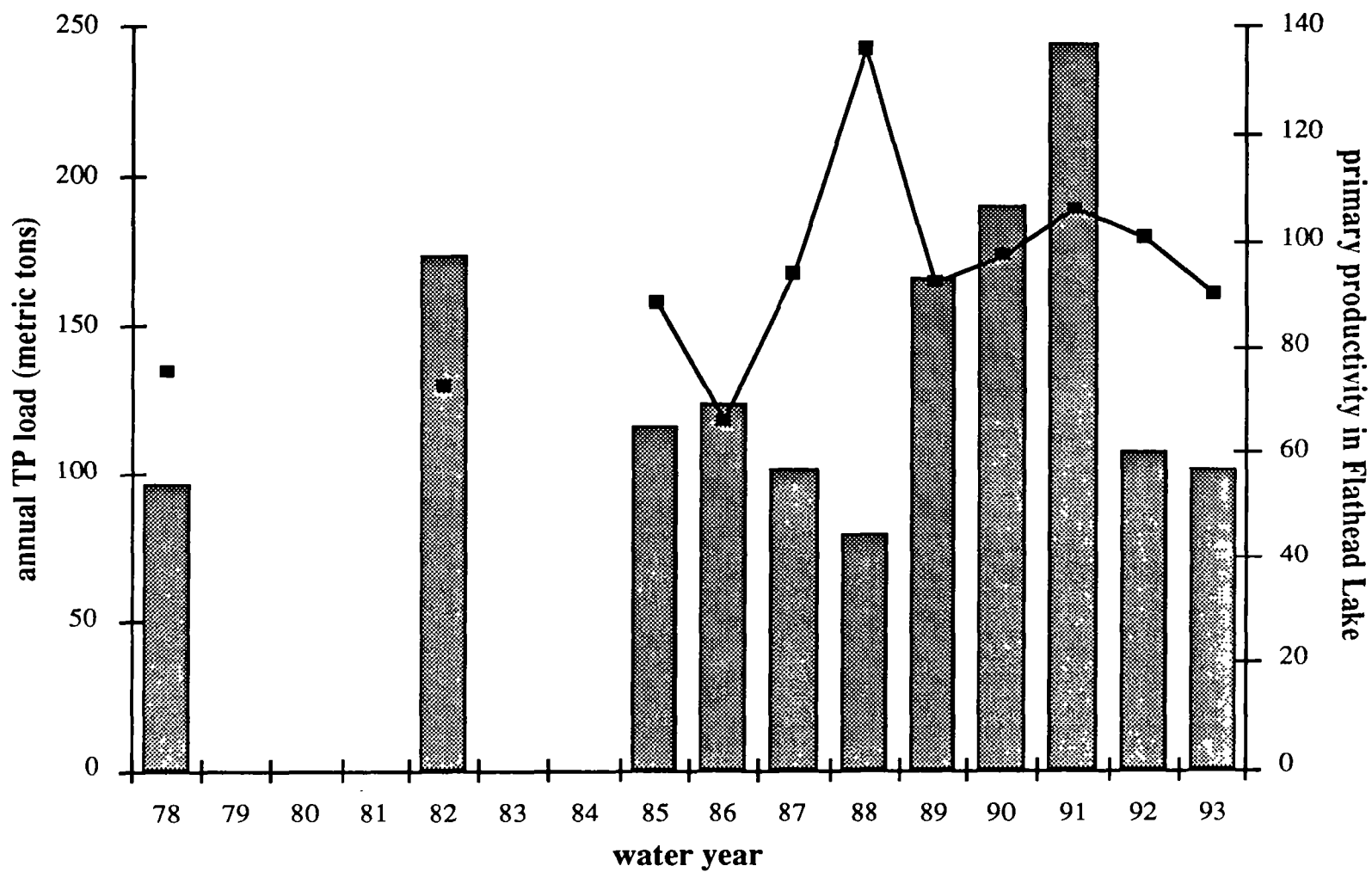
Figure 22. Annual primary productivity (solid squares) plotted in relation to annual input of biologically active phosphorus from all sources (histograms).

the waste treatment plants and non-point discharges likely is increasing in relation to increasing human population and may be offsetting the gains made by reducing N and P concentrations in treatment plant effluents. Data bases at the Flathead Lake Biological Station are currently being used to more precisely allocate the loads to upstream point and diffuse sources (i.e., Flathead Lake Total Maximum Daily Load Study).

CONCLUSIONS

- Limnological parameters vary in similar ways at the midlake site in comparison to other sites, except Ross Deep, and to a lesser extent, at the shallow Lakeside site. Long term data collected at the midlake site are representative of the lake as a whole. Continuation of the long term collection at the midlake monitoring site is critically important to assessment of water quality in the lake and its tributaries.

- Parts of the lake, such as Ross Deep, appear to be changing and routine monitoring is warranted. Other sites are important too because, like Ross Deep, they may show signs of deterioration before Midlake Deep.

- Pollution of Ashley Creek by the Kalispell Sewage Treatment Plant has significantly abated as a result of installation of new treatment technology and the ban on phosphorus containing detergents, but the sediments of Ashley Creek contain a great deal of potentially exchangeable phosphorus and nitrogen. Hence, sediment release may somewhat offset improvements in water quality attributable to better sewage treatment, at least for several years. Maintaining oxic conditions in the creek (e.g., by eliminating pollution sources above the effluent from the Kalispell Treatment Plant) is very important because P storage in the sediments of the creek is accelerated by oxidation equilibrium at the mud-water interface.

- Bioavailable phosphorus mass from the urban sewage plants declined more than 15% after 1988 as a result of the implementation of the phosphorus reduction plan of the Montana Water

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Quality Bureau. Reduced discharge of P from the waste treatment facilities in the basin may be responsible, at least in part, for stabilization of annual primary productivity in recent years. Shorter term (seasonal) analyses of loading are required to verify a relationship.

- Bioavailable phosphorus mass reaching Flathead Lake is largely determined by river flow (i.e., annual catchment runoff) and no significant changes have occurred over the period of record with respect to the allocation of load by tributary.

- Bulk precipitation is a significant source of bioavailable phosphorus (10 - 38% of the total mass input) that varies independently of river inflow. Contrary to river input which occurs predominantly in the spring runoff period, sporadic events producing deposition of bulk precipitation occur year around (though peak P inputs are common in both spring and fall). Bioavailable P input via airshed deposition occurs lake wide, while river inflow occurs at the north end of the lake.

- Annual input of bioavailable phosphorus alone is not enough to predict primary productivity with any accuracy. However, algae growth rates do increase in response to added labile phosphorus in bioassays of Flathead Lake water. Interactions with food web (*Mysis* establishment) and other biophysical dynamics (depth and duration of the mixed layer; photosynthetically active radiation, DOM and DON concentrations in the photic zone, epilimnetic withdrawal of water by Kerr Dam) may influence primary production measures in ways that mask lake wide responses to phosphorus loading. Careful analysis of intra-annual variation is warranted.

RECOMMENDATIONS

Based on the results of this Phase I diagnostic study, the following activities are recommended for protection of the high values of water quality in Flathead Lake and its catchment basin.

1. The long term water quality monitoring program for Flathead Lake should be continued in order to document any new changes in the lake's status and to assist the ongoing TMDL effort. New monitoring is needed to more accurately demonstrate ground water pollution from septic systems on the lake shore and the chemistry of bulk precipitation on a lake wide basis (i.e., the one bulk precipitation site at the Biological Station may be insufficient to accurately determine airshed sources of nutrient pollution.

2. Research should be funded to determine the cause of the hypolimnetic oxygen deficit in Big Arm Bay. Shoreline sources of pollution and the possibility of entrainment of river water in the bay are possible explanations that should be investigated thoroughly.

3. The P reduction strategy of the Montana Water Quality Board should continue to guide management actions in the Flathead Basin to reduce nutrient pollution. Greater attention should be paid to documentation and reduction of non-point sources, especially with respect to increasing fertility of the Stillwater and Whitefish Rivers and the Evergreen alluvial aquifer as they flow through the Flathead Valley and discharge into the Flathead River. Additional water quality monitoring sites are needed to determine how land use activities are associated with non-point source inputs into these water bodies. Predictive knowledge of sewage leachates from septic systems and soil capacities for new systems is badly needed in view of very rapid growth of homes and businesses outside of the sewer districts in the Flathead Valley.

4. All federal, state and local statutes that control water quality should be strictly enforced.

5. Introduction or planting of non-native biota into the waters of the Flathead Basin should be strictly forbidden.

6. Reduce the full pool level in Flathead Lake one foot to curtail shoreline erosion as recommended by (Lorang 1993).

7. Efforts by the Flathead Basin Commission to develop a water quality management program for the Flathead Basin should be responsive to the conclusions and recommendations of this study. Control of non-point sources of nitrogen and phosphorus should be emphasized in all planning processes. The Basin Commission should encourage best management practices for all land use activities in the basin. The main goal in all cases should be to conduct activities in such manner to minimize sediment and nutrient inputs to surface and ground waters that reach Flathead Lake. Use of development setbacks, buffer strips, limitations on fertilizer uses and animal waste and road runoff management systems (e.g., settling ponds and wetlands to retain sediments and nutrients) should be emphasized.
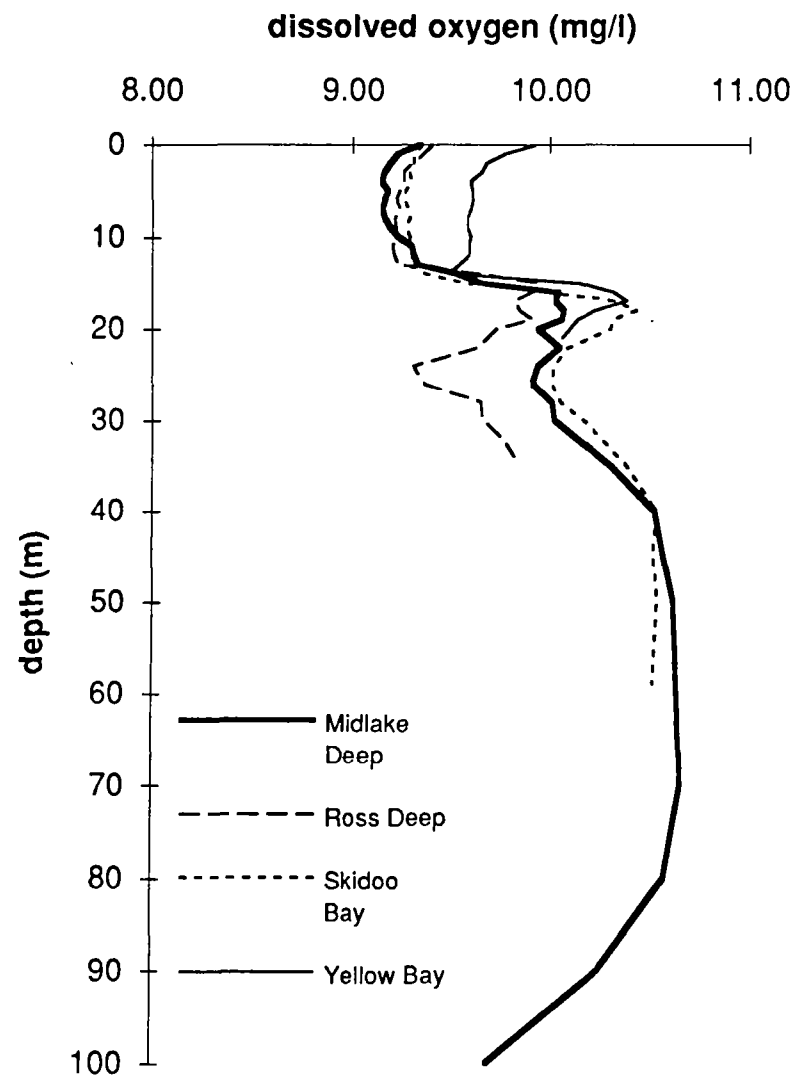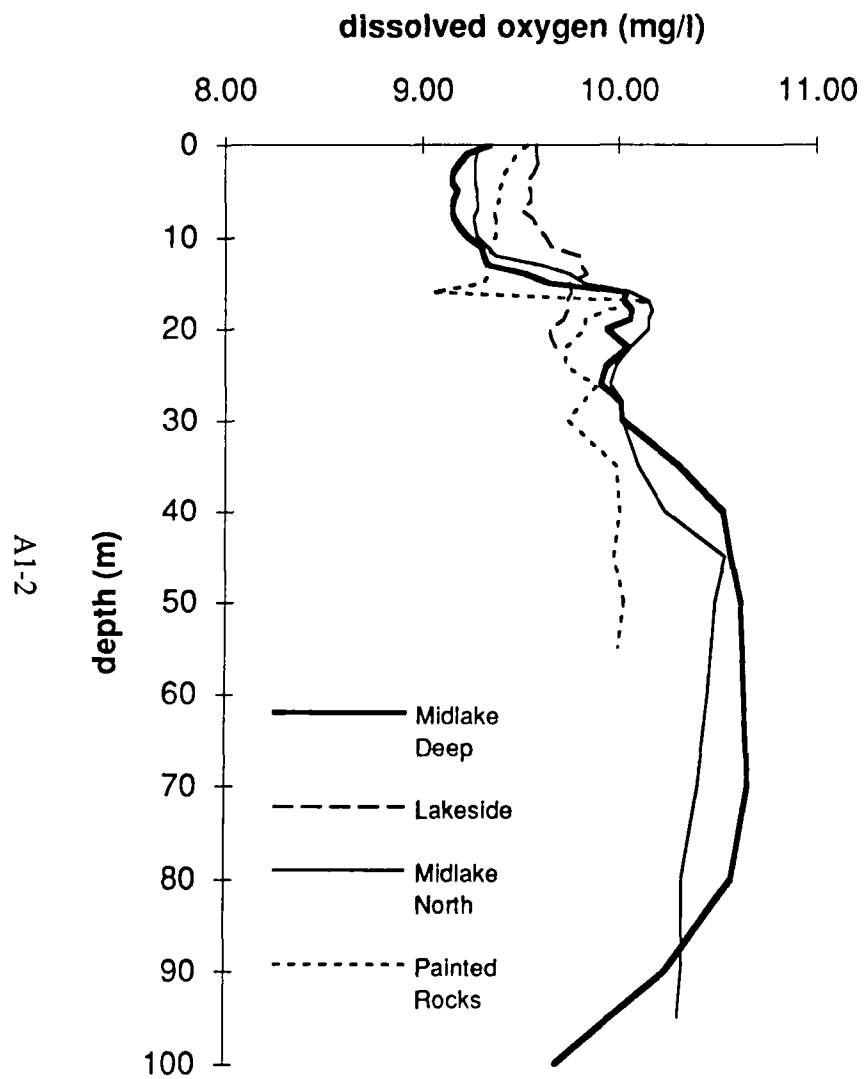
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APPENDIX 1. Depth profiles of dissolved oxygen and temperature.

Sep-91

dissolved oxygen (mg/l)

dissolved oxygen (mg/l)

A1-2

A1-3

dissolved oxygen (mg/l)

8.00   9.00   10.00   11.00

depth (m)

Midlake
Deep

Lakeside

Midlake
North

Painted
Rocks

dissolved oxygen (mg/l)

8.00   9.00   10.00   11.00

depth (m)

Midlake
Deep

Ross
Deep

Skidoo
Bay

Yellow
Bay

Jul-92

dissolved oxygen (mg/l)

6.50   7.50   8.50   9.50   10.50   11.50

depth (m)

0
10
20
30
40
50
60
70
80
90

Midlake Deep

Lakeside

Midlake North

Painted Rocks

dissolved oxygen (mg/l)

6.50   7.50   8.50   9.50   10.50   11.50

depth (m)

0
10
20
30
40
50
60
70
80
90

Midlake Deep

Ross Deep

Skidoo Bay

Yellow Bay

Sep-92

dissolved oxygen (mg/l)

dissolved oxygen (mg/l)

Midlake Deep

Lakeside

Midlake North

Painted Rocks

Midlake Deep

Ross Deep

Skidoo Bay

Yellow Bay

A1-5

Jun-93

dissolved oxygen (mg/l)

dissolved oxygen (mg/l)

A1-6

Aug-93

dissolved oxygen (mg/l)

dissolved oxygen (mg/l)

depth (m)

depth (m)

A1-7

Legend (left):
- Midlake Deep
- Lakeside
- Midlake North
- Painted Rocks

Legend (right):
- Midlake Deep
- Ross Deep
- Skidoo Bay
- Yellow Bay

Sep-91

A1-8

Nov-91

A1-9

Jul-92

Sep-92

temperature (deg C)

temperature (deg C)

A1-11

Jun-93

Aug-93

temperature (deg C)

A1-13

APPENDIX 2. Summary of chemical characteristics.

Table 1. Chemical characteristics of grab samples from a depth of 1m (September 1991 to January 1992) or 5m (February 1992 to August 1993) at Flathead Lake Clean Lakes water quality monitoring sites. Means, standard deviations (±1.0 S.D.) and ranges (minimum - maximum) are given for n samples.

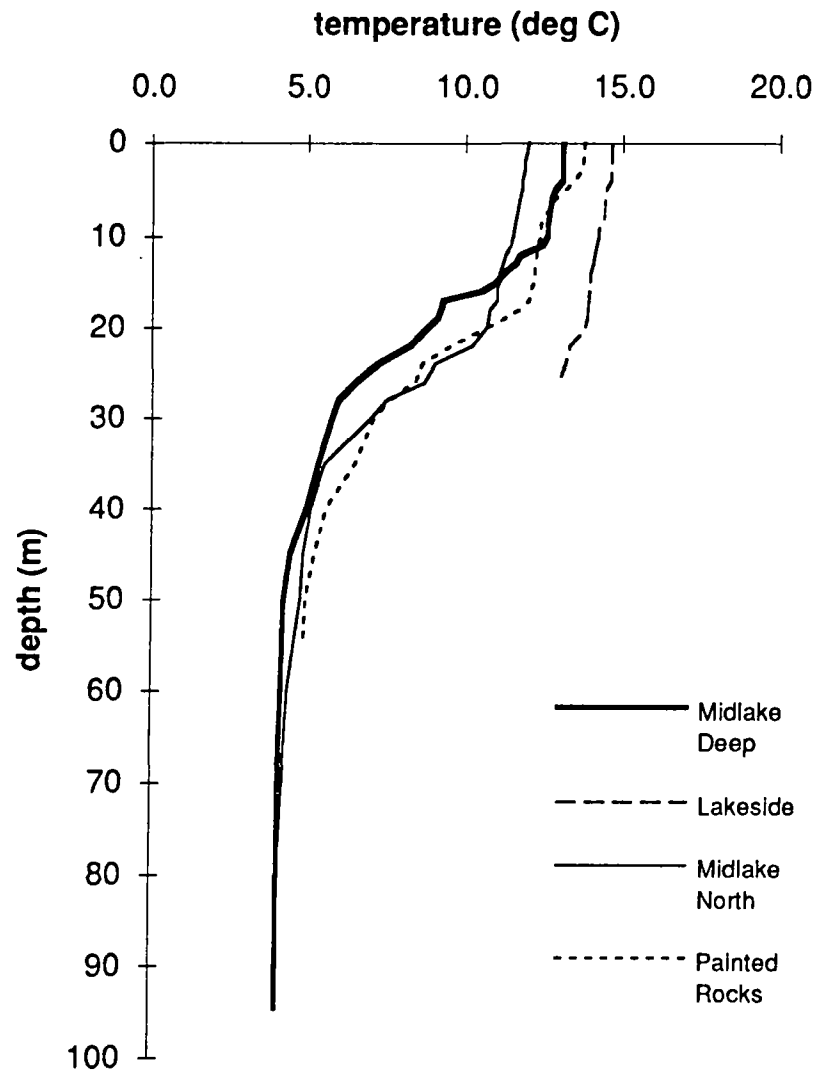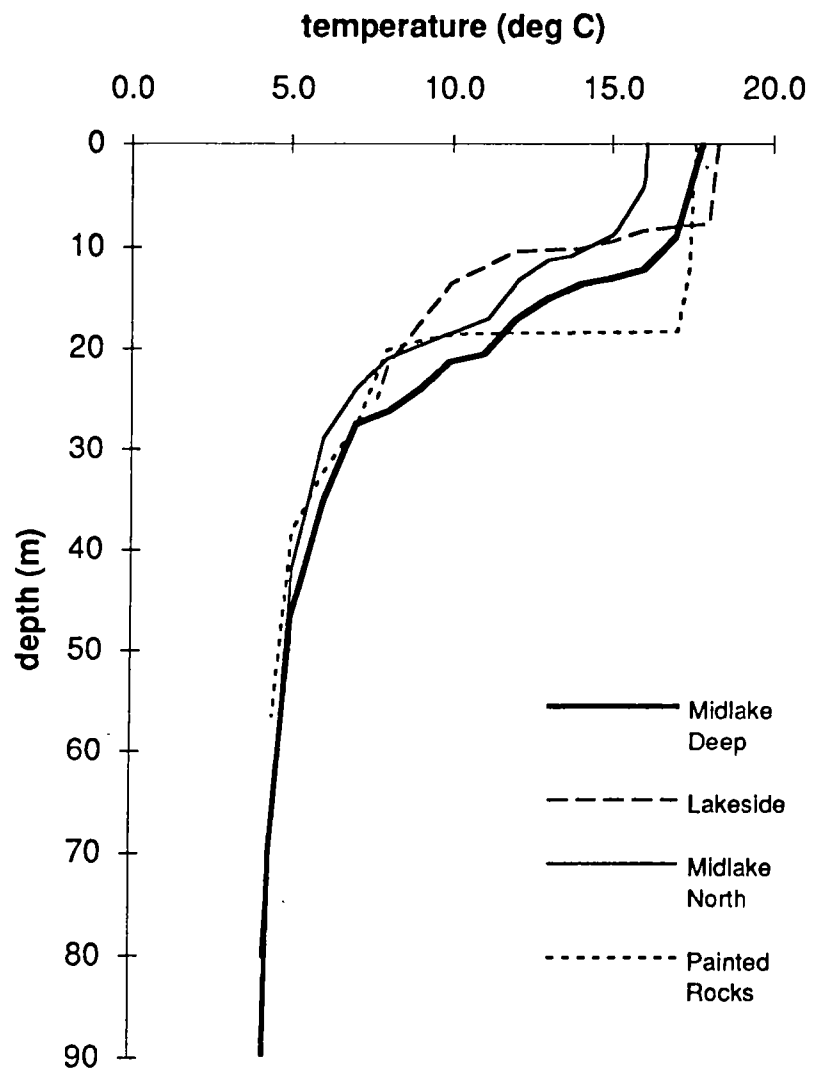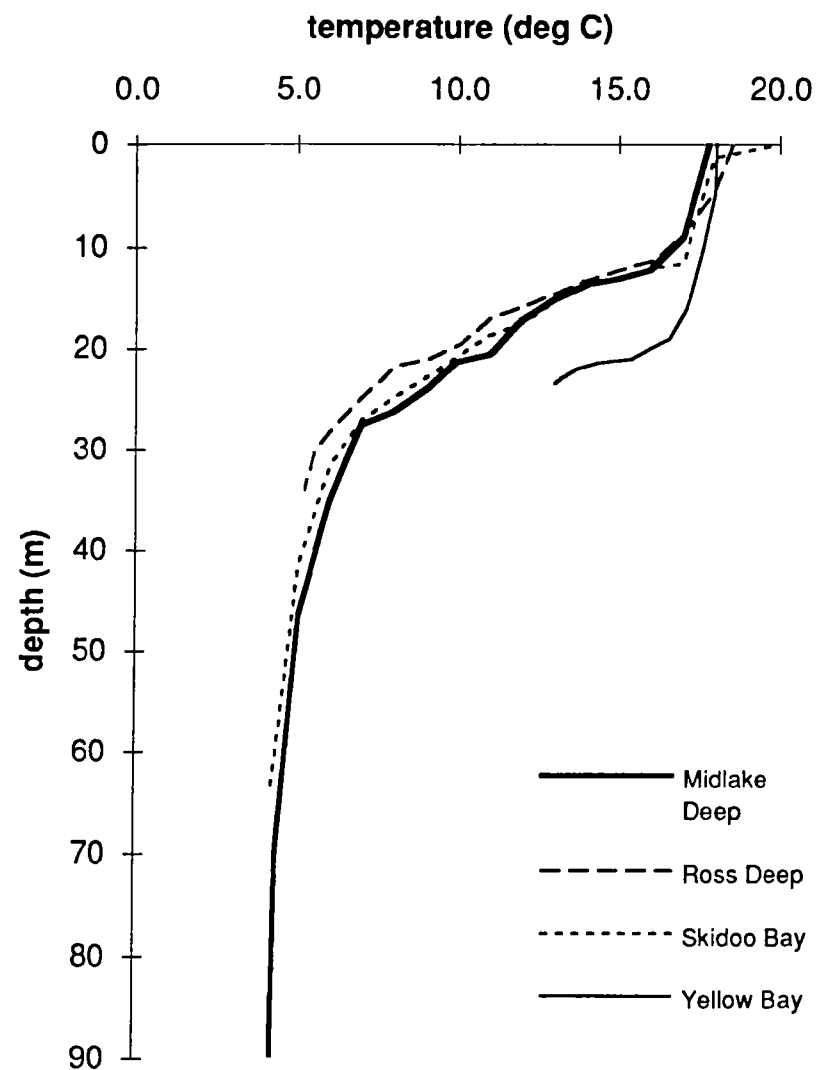| Site (n)		Alk mg/l-CaCO3	Turb NTU	TSS mg/l	SiO2 mg/l-SiO2	SO4 mg/l-SO4	SRP µg/l-P	SP µg/l-P	TP µg/l-P	NH3 µg/l-N	NO2/3 µg/l-N	TPN µg/l-N	NDOC mg/l-C
Midlake Deep (27)	Mean	84.7	1.2	0.7	4.8	3.05	1.0	2.9	5.1	5.1	26.9	88.1	0.25
	S.D.	2.8	3.0	0.4	0.4	0.27	0.7	1.3	1.8	3.2	21.2	35.1	0.07
	Min.	80.3	0.2	<0.5	4.0	2.71	<0.4	1.0	3.1	1.2	<2.0	39.4	0.14
	Max.	90.2	16.0	1.1	5.5	3.37	3.4	7.4	11.7	14.1	62.2	215.0	0.39
Midlake North (18)	Mean	85.5	1.6	1.1	4.8	3.04	0.9	2.9	5.2	3.6	30.8	101.0	0.24
	S.D.	2.9	3.9	0.5	0.3	0.23	0.5	1.1	1.4	1.9	21.1	44.3	0.06
	Min.	80.3	0.3	0.5	4.0	2.60	<0.4	1.6	3.2	0.5	<2.5	31.4	0.13
	Max.	90.2	17.0	1.6	5.3	3.42	2.6	5.7	7.6	7.3	57.4	236.0	0.38
Lakeside (17)	Mean	84.9	4.2	2.3	4.8	2.99	0.8	3.0	6.3	3.1	31.4	104.7	0.26
	S.D.	4.5	13.6	1.7	0.4	0.28	0.5	1.1	2.6	1.4	21.3	41.6	0.06
	Min.	73.7	0.4	0.8	4.1	2.60	<0.4	1.1	2.3	<1.2	<2.6	35.5	0.17
	Max.	92.4	57.0	5.3	5.5	3.42	2.8	5.8	12.6	6.3	63.8	224.0	0.43
Painted Rocks (18)	Mean	84.7	2.5	1.0	4.8	3.04	0.8	3.4	6.4	2.9	29.6	97.8	0.26
	S.D.	3.7	7.6	0.7	0.4	0.20	0.4	2.3	3.7	1.4	20.8	41.1	0.07
	Min.	77.0	0.2	<0.5	3.9	2.58	<0.4	1.3	3.2	0.5	<2.6	35.7	0.17
	Max.	90.2	33.0	2.2	5.3	3.27	1.6	10.4	19.0	5.4	55.2	213.0	0.44
Ross Deep (18)	Mean	85.3	1.2	0.8	4.8	3.02	0.7	2.7	5.7	2.8	20.3	92.0	0.30
	S.D.	3.7	2.5	0.2	0.4	0.16	0.2	1.1	1.4	1.6	16.7	32.5	0.07
	Min.	77.0	0.2	<0.5	3.9	2.67	<0.4	1.3	3.3	<0.5	<2.5	26.1	0.15
	Max.	90.2	11.0	0.9	5.7	3.30	1.4	5.2	8.0	7.4	53.7	195.0	0.40
Yellow Bay (18)	Mean	85.5	0.6	0.6	4.8	3.04	0.7	2.4	5.4	2.7	28.0	93.3	0.25
	S.D.	3.1	0.3	0.1	0.3	0.15	0.2	0.9	1.8	1.3	20.9	31.7	0.11
	Min.	80.3	0.3	<0.5	4.0	2.79	<0.4	1.0	2.9	<1.2	<2.5	20.8	0.14
	Max.	89.7	1.2	0.8	5.2	3.30	1.0	4.7	9.5	5.8	59.5	140.0	0.59
Skidoo Bay (17)	Mean	85.6	1.3	0.8	4.8	3.03	0.7	2.8	5.5	3.0	29.0	96.3	0.25
	S.D.	3.2	3.0	0.2	0.4	0.20	0.2	1.2	1.9	1.6	20.4	34.2	0.11
	Min.	80.3	0.3	0.6	3.9	2.56	<0.4	1.6	3.2	<1.4	<2.5	33.3	0.07
	Max.	90.2	13.0	1.0	5.6	3.27	1.3	6.3	10.6	7.0	59.2	154.0	0.57
Osprey Nest (1)		79.2	0.5	-	4.8	2.93	0.5	4.2	5.1	1.5	<5.0	38.9	0.30
Somers Bay (1)		67.1	0.6	-	4.7	3.21	0.6	3.3	11.0	3.6	<5.0	41.5	0.27
Narrows (1)		80.3	0.5	-	4.8	2.93	0.8	3.8	4.4	1.4	<5.0	37.3	0.30
East Bay (1)		78.1	1.4	-	5.3	2.80	0.9	5.1	8.2	3.0	<5.0	92.5	0.37

Table 2. Chemical characteristics of grab samples from a depth of about 1m from the bottom at each Flathead Lake Clean Lakes water quality monitoring sites. Data are also given for an additional mid-water site at Midlake Deep (30m). Means, standard deviations (±1.0 S.D.) and ranges (minimum - maximum) are given for n samples.

A2-3

Site (n)		Alk mg/l-CaCO3	Turb NTU	TSS mg/l	SiO2 mg/l-SiO2	SO4 mg/l-SO4	SRP µg/l-P	SP µg/l-P	TP µg/l-P	NH3 µg/l-N	NO2/3 µg/l-N	TPN µg/l-N	NDOC mg/l-C
Midlake Deep (12) 30m	Mean	86.4	0.8	0.5	5.0	3.12	0.68	2.9	5.8	3.26	55.6	126.8	0.17
	S.D.	2.3	0.5	0.0	0.2	0.10	0.17	0.8	1.5	2.32	8.4	55.2	0.02
	Min	81.4	0.3	<0.5	4.7	3.03	<0.40	2.2	4.5	0.58	45.3	80.6	0.14
	Max	89.1	1.9	0.5	5.4	3.22	1.01	4.6	8.7	9.30	71.2	247.0	0.22
Midlake Deep(27) 90m	Mean	86.4	0.7	0.5	5.5	3.35	0.89	3.1	5.4	4.79	64.8	116.3	0.13
	S.D.	1.8	0.3	0.0	0.5	0.08	0.42	1.4	1.6	3.63	15.5	43.9	0.04
	Min	82.5	0.3	<0.5	4.6	3.31	0.40	1.3	3.3	<0.5	36.9	66.9	0.10
	Max	91.3	1.4	0.5	7.1	3.47	2.17	7.4	9.8	14.97	95.5	254.0	0.29
Midlake North (18)	Mean	87.3	1.0	0.7	5.2	3.10	0.87	3.3	5.6	4.34	63.4	116.2	0.14
	S.D.	1.5	1.0	0.3	0.4	0.17	0.3	1.2	1.4	3.3	14.6	36.9	0.04
	Min	85.5	0.3	<0.5	4.6	2.88	0.5	<1.6	3.7	<1.2	39.8	79.6	0.08
	Max	91.3	4.6	1.1	5.8	3.52	1.8	5.8	9.3	13.2	87.3	238.0	0.24
Lakeside (17)	Mean	86.1	3.7	1.9	5.0	2.97	0.73	3.3	6.3	4.51	39.1	104.1	0.24
	S.D.	2.8	9.9	1.2	0.3	0.30	0.4	1.2	2.6	4.4	14.4	35.0	0.06
	Min	81.4	0.4	0.8	4.4	2.31	<0.4	1.6	3.7	<1.2	<2.6	42.6	0.17
	Max	90.2	42.0	3.4	5.5	3.39	2.1	5.7	14.4	20.2	55.1	216.0	0.42
Painted Rocks (18)	Mean	87.9	1.9	0.9	5.2	3.08	0.70	3.0	5.7	3.95	59.3	114.8	0.17
	S.D.	3.8	3.6	0.4	0.3	0.21	0.2	1.8	1.9	3.0	15.4	27.6	0.03
	Min	83.6	0.5	<0.5	4.5	2.67	0.4	1.2	3.7	<1.2	33.8	73.1	0.12
	Max	101.2	16.0	1.3	6.0	3.50	1.3	8.4	12.0	11.5	82.2	196.0	0.24
Ross Deep (18)	Mean	86.9	0.7	0.8	5.4	3.06	0.72	3.1	6.3	3.64	50.7	116.6	0.25
	S.D.	2.0	0.3	0.4	0.5	0.19	0.2	1.1	2.1	2.5	18.7	27.9	0.08
	Min	82.5	0.3	0.5	4.6	2.76	<0.4	1.7	3.5	1.0	21.0	72.5	0.16
	Max	90.2	1.4	1.4	6.4	3.34	1.1	5.9	12.6	11.3	77.9	165.0	0.49
Yellow Bay (18)	Mean	86.2	0.6	0.7	4.9	3.10	0.69	2.6	5.2	4.11	37.2	95.7	0.23
	S.D.	2.8	0.3	0.4	0.3	0.10	0.3	0.8	1.2	4.1	17.4	23.4	0.10
	Min	80.3	0.3	<0.5	3.9	2.91	<0.40	1.4	3.4	0.7	7.9	36.3	0.12
	Max	90.2	1.3	1.2	5.3	3.22	1.6	4.9	7.6	17.7	60.6	148.0	0.50
Skidoo Bay (17)	Mean	87.1	0.6	0.6	5.3	3.14	0.72	3.4	6.0	4.17	59.8	116.1	0.14
	S.D.	2.1	0.2	0.3	0.5	0.16	0.2	1.9	2.9	3.9	14.6	27.0	0.03
	Min	83.6	0.4	<0.5	4.9	2.79	<0.4	1.5	4.0	0.9	35.3	68.5	0.11
	Max	91.3	1.2	1.0	6.5	3.43	1.3	9.2	14.3	14.9	86.1	177.0	0.21
Osprey Nest (1)		84.7	0.9	-	5.4	3.25	1.6	8.2	8.6	3.3	62.2	93.9	0.15
Somers Bay (1)		84.7	2.3	-	5.2	3.31	0.6	5.1	8.1	2.7	46.7	89.6	0.34
Narrows (1)		81.4	0.7	-	4.8	2.93	0.8	4.5	7.4	2.8	<5.0	45.0	0.37

Table 3. Chemical and biological characteristics of integrated photic zone samples collected from 0 to 30m (except Lakeside and Yellow Bay, 0-20m; Somers, 0-15m; Narrows 0-8m; East Bay 0-2m) at Flathead Lake Clean Lakes water quality monitoring sites. Means, standard deviations (±1.0 S.D.) and ranges (minimum - maximum) are given for n samples.

Site (n)		Alk mg/l-CaCO3	Turb NTU	TSS mg/l	SiO2 mg/l-SiO2	SO4 mg/l-SO4	SRP µg/l-P	SP µg/l-P	TP µg/l-P	NH3 µg/l-N	NO2/3 µg/l-N	TPN µg/l-N	NDOC mg/l-C	Chl a µg/l	Phaeo µg/l
Midlake Deep (26)	Mean	85.1	1.3	0.5	4.9	2.70	0.7	2.7	5.1	4.3	31.9	96.3	0.23	0.91	0.44
	S.D.	2.5	3.3	0.0	0.3	0.91	0.4	0.9	1.6	3.0	16.6	34.5	0.06	0.32	0.26
	Min.	80.3	0.3	0.5	4.2	0.00	<0.4	<0.8	2.8	<0.5	4.8	64.0	0.13	0.56	0.09
	Max.	89.7	17.5	0.6	5.6	3.17	1.8	4.6	8.7	12.3	59.9	241.0	0.33	2.02	0.97
Midlake North (18)	Mean	86.1	1.7	0.8	4.8	2.87	0.7	3.1	5.4	3.7	35.7	103.1	0.22	0.92	0.52
	S.D.	2.7	3.7	0.2	0.3	0.81	0.2	1.5	1.6	1.5	15.8	36.7	0.05	0.36	0.28
	Min.	81.4	0.3	0.5	4.1	0.00	<0.4	1.5	3.8	<1.2	6.8	49.2	0.13	0.32	0.11
	Max.	90.2	16.0	1.0	5.1	3.27	1.1	8.5	10.9	6.1	58.6	229.0	0.34	1.82	1.11
Lakeside (17)	Mean	85.1	3.8	1.8	4.8	3.02	0.8	3.4	5.9	3.1	32.6	102.6	0.25	1.00	0.51
	S.D.	4.1	11.4	1.5	0.4	0.23	0.3	1.6	1.5	1.1	19.9	36.4	0.06	0.39	0.40
	Min.	74.8	0.4	0.5	4.1	2.58	0.4	1.1	3.1	<1.2	<2.6	35.4	0.15	0.48	0.07
	Max.	91.3	48.0	4.4	5.3	3.32	1.6	8.6	8.1	5.0	62.0	205.0	0.39	1.85	1.41
Painted Rocks (18)	Mean	85.6	2.5	1.2	4.9	2.80	0.7	2.6	5.3	3.3	35.1	106.3	0.24	0.98	0.40
	S.D.	3.5	6.8	0.8	0.3	0.80	0.3	0.7	1.4	1.3	16.4	34.0	0.06	0.49	0.30
	Min.	79.2	0.3	<0.5	4.1	0.00	<0.4	1.6	3.7	1.3	7.6	46.8	0.15	0.48	0.05
	Max.	90.8	29.0	2.3	5.3	3.24	1.6	4.0	10.2	5.4	55.7	198.0	0.36	2.30	1.37
Ross Deep (18)	Mean	86.3	1.4	0.6	5.0	2.89	0.7	2.7	5.4	3.2	29.3	101.3	0.27	1.28	0.46
	S.D.	2.6	2.6	0.1	0.3	0.81	0.2	0.7	1.5	2.0	12.1	35.4	0.06	0.53	0.23
	Min.	81.4	0.3	<0.5	4.1	0.00	0.3	1.8	3.9	<0.5	9.3	41.4	0.16	0.63	0.16
	Max.	90.2	11.5	0.8	5.5	3.24	1.0	3.9	9.7	7.7	56.6	218.0	0.40	2.38	0.94
Yellow Bay (17)	Mean	86.0	0.7	0.8	4.8	2.89	1.0	3.4	5.7	3.3	30.4	101.9	0.25	0.95	0.33
	S.D.	2.9	0.4	0.4	0.4	0.81	1.0	2.3	2.1	1.6	20.2	25.7	0.11	0.32	0.36
	Min.	81.4	0.3	<0.5	4.0	0.00	<0.4	1.2	2.9	<1.2	<2.5	32.0	0.16	0.40	0.01
	Max.	90.2	1.7	1.4	5.4	3.33	4.8	10.1	10.5	7.3	57.3	135.0	0.57	1.66	1.66
Skidoo Bay (17)	Mean	86.0	1.5	0.7	4.9	2.88	0.7	2.9	5.1	3.3	35.4	100.9	0.23	0.93	0.35
	S.D.	3.1	3.4	0.2	0.4	0.84	0.3	1.8	2.3	1.8	16.3	28.4	0.09	0.34	0.22
	Min.	79.2	0.3	<0.5	4.1	0.00	<0.4	1.6	2.5	<1.2	3.9	47.3	0.12	0.40	0.09
	Max.	90.2	14.0	1.0	5.6	3.90	1.4	8.9	13.2	8.0	59.5	163.0	0.49	1.70	1.08
Osprey Nest (1)		-	-	-	-	-	-	-	-	-	-	-	-	1.19	0.21
Somers Bay (1)		-	-	-	-	-	-	-	-	-	-	-	-	0.71	0.22
Narrows (1)		-	-	-	-	-	-	-	-	-	-	-	-	0.71	0.10
East Bay (1)		-	-	-	-	-	-	-	-	-	-	-	-	0.95	0.25

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