SOME EFFECTS OF MINE DRAINAGE ON PRIMARY PRODUCTION IN COEUR D'ALENE RIVER AND LAKE, IDAHO

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ABSTRACT

Variations in primary production and physiochemical measurements in the Coeur d'Alene River and Lake contaminated by mine and industrial wastes were examined from May, 1969, to November, 1970. Metal concentrations Md, Cd, Mg, Ca, Pb, Cu, Zn, Fe, Na, and K; water quality and phytoplankton composition-density were determined for thirty-five dates during this period. Additional sampling included unpolluted portions of Coeur d'Alene Lake from December, 1969, to November, 1970, and the unaffected St. Joe River during the summers of 1969-70.

Primary production ranged from 17.6 to 1337.9 mg C/m^2/day in the Coeur d'Alene River and 69.3 to 1714.5 mg C/m^2/day in the Coeur d'Alene Lake. Concentrations of zinc (0.1 to 11.2 mg Zn/l) and copper (0.0 to 0.6 mg Cu/l) in the Coeur d'Alene River indicated that heavy metals could be toxic to algae. Diatoms dominated phytoplankton in the Coeur d'Alene River, Lake, and St. Joe River.

Primary production in the southern portion of Coeur d'Alene Lake appeared controlled by discharges from the Coeur d'Alene and St. Joe rivers. Low concentrations of inorganic carbon and moderately high concentrations of nitrate and phosphate in the Coeur d'Alene River, Lake, and St. Joe River suggest the importance of carbon in regulating production. The effect of wind on eplimnetic production and poorly developed stratification in the south end of the lake appeared related to decreased depths. Wind action may control eutrophication by suppressing hypolimnetic oxygen depletion and anaerobic regeneration.

Nannoplankton from Coeur d'Alene Lake were exposed to known concentrations of Cu^{2+}, Cd^{2+}, Zn^{2+}, and dilutions of Coeur d'Alene River water under controlled light and temperature. Inhibitory effects of separate and interacting metals on carbon-14 uptake by algae were assessed with factorial designed bioassays and response surfaces. Copper, cadmium, and zinc were acutely and synergistically toxic to carbon uptake by phytoplankton. Concentrations ranged from 0.05 to 0.75 mg Cu/l, 0.1 to 0.3 mg Cd/l, and 0.1 to 1.5 mg Zn/l. Copper caused an overriding effect on two- and three-way interactions of Cu^{2+}, Cd^{2+}, and Zn^{2+}. Dilutions of Coeur d'Alene River water decreased Cu and Zn toxicity. Variable algal community structure, major cations, softwater (<60 mg/l as $CaCO_3$), and water quality appeared to affect metal toxicity.

INTRODUCTION

The Coeur d'Alene River and Lake have received mine tailings and metallic sulfide minerals for over 80 years. Toxicity of the river water to fish and plankton was documented at the mouth of the Coeur d'Alene River in 1932 (Ellis, 1940). Sappington (1970) used unpolluted headwaters of the Coeur d'Alene River and reported 96-hour TLm values for cutthroat trout of 0.09 mg Zn/l. Savage (1970) noted that deposits of mine wastes limited macrobenthic colonization of riffle areas in the Coeur d'Alene River. However, no attempts have been made to assess the impact of mine pollution on autotrophic production. Photosynthesis by phytoplankton produces the majority of the organic matter necessary for the subsequent development of heterotrophic communities.

Prior to the present study, I noted sparse plankton communities, high concentrations of zinc, copper, and lead, and a minimal pH of 6.4 at the river mouth. Maximum metal concentrations of 3.05 mg Zn/l and 0.50 mg Cu/l exceeded the tolerance limits of most phytoplankton (McKee and Wolf, 1963) and suggested inhibition of algal productivity by a synergistic mode of toxicity. Metal synergism usually results from simultaneous cooperation of separate elements which produce an effect greater than of any metal taken alone. Such combined toxic actions of metals are well known for various species of fish (Skidmore, 1964), but little information exists on cultured and <u>in situ</u> algal sensitivities to interacting metals. Fitzgerald (1971) detected no synergism between copper and zinc in tests of hardwater cultured blue-green and green algae. However, the naturally softwater of the Coeur d'Alene River and the presence of acidic and various metallic wastes increases the probability that synergism reduces phytoplankton productivity. Heavy metals have been shown to be more toxic to fish (Jones, 1938) and algae (Moyle, 1949) in softwater than in hardwater.

Mine wastes enter the main Coeur d'Alene River from the South Fork some 30 miles upstream from the lake (Figure 1). Contaminants in the South Fork include seepage from old tailings deposits, heavy metals, acids, and suspended solids-colloids from electrolytic zinc and antimony plants, milling, sulfuric-phosphoric acid, and fertilizer operations (Mink, 1972).

The Coeur d'Alene and St. Joe rivers enter the southern end of Coeur d'Alene Lake (Figure 1). The close proximity of the river mouths and differences in environments provide unique opportunities for studying the behavior of pollutants. The Coeur d'Alene River has poorly developed aquatic communities with no apparent harvestable crop of fish. Colloidal mill and industrial effluents impart a greenish cast to the river and alter light penetration and extinction while tailings sediments cover most substrates. The St. Joe River has a productive sport fishery but has been affected by sewage from the town of St. Maries, farming, logging, and boating activities.

Coeur d'Alene Lake lies in a sunken river valley 23 miles long and drains northwest through the Spokane River near the city of Coeur d'Alene (Figure 1). The lake appears mesotrophic with autumal eutrophication in developed bays.

My research examined the influence of mine drainage on seasonal primary productivity and water quality in the Coeur d'Alene

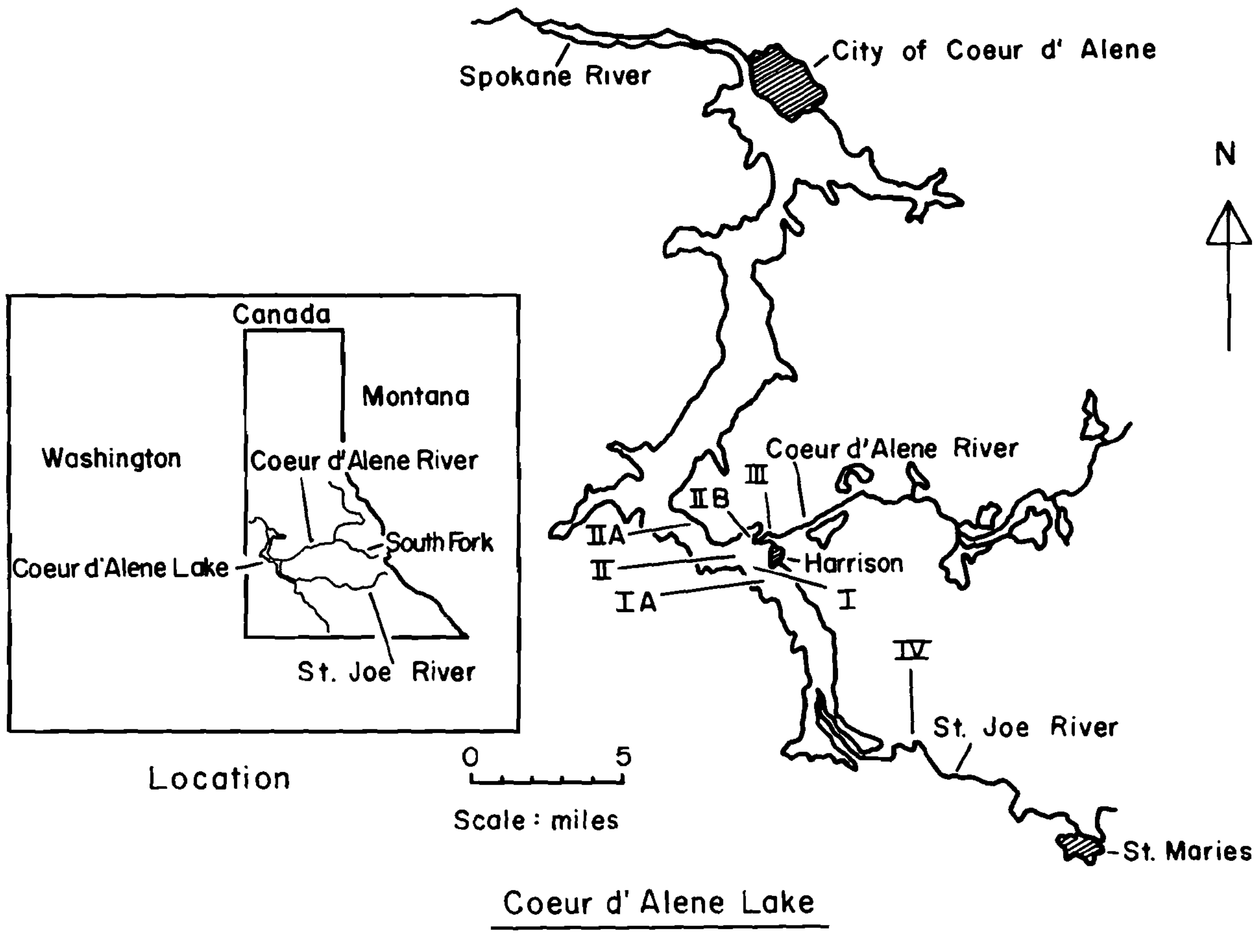

Fig. 1.--Coeur d'Alene River, Lake, and St. Joe River station locations

River and Lake, from May, 1969, to November, 1970. I made additional observations of the Coeur d'Alene Lake and St. Joe River free of mine wastes. I assessed synergistic effects of different concentrations of metals and volumes of Coeur d'Alene River water on phytoplankton production with factorial designed bioassays of carbon-14 uptake in algae. Bioassays were conducted from August to November, 1970.

The format of my research follows suppositions examined in field and bioassay experiments. If mine wastes in the Coeur d'Alene River have an effect on primary production, then measuring primary production, phytoplankton composition, and water quality in the Coeur d'Alene River, Lake, and St. Joe River will indicate that the Coeur d'Alene River has water quality characteristics and metal concentrations that alter the development of "normal" phytoplankton communities and inhibit primary production. If mine effluents inhibit primary production in the Coeur d'Alene River they will cause a decrease in carbon uptake by phytoplankton in the delta and lake. Therefore, the addition of different volumes of Coeur d'Alene River water and concentrations of toxic metals to natural phytoplankton communities of the lake which contain carbon-14 will result in decreased carbon fixation. Conversely, carbon fixation will not decrease in controls without river water and metal treatments.

METHODS

Field Data

I took water quality data, phytoplankton samples, and conducted simultaneous primary production experiments in the Coeur d'Alene River (Station III) and Lake (I) from May, 1969, to November, 1970 (Figure 1). Observations during shorter intervals describe the primary production, phytoplankton composition, and water quality in the Coeur d'Alene River mouth, Lake, and St. Joe River. Stations IIB at the mouth of the Coeur d'Alene River and II directly west of the delta were established to assess the mixing effect of river and lake waters. I measured differences in lake waters at Stations IA (south of I) and IIA (north of II) from December, 1969, to November, 1970). The Coeur d'Alene River and St. Joe River (Station IV) were sampled during July and August, 1969, and June to September, 1970 (Figure 1).

Physical and chemical measurements included estimates of light penetration by Secchi disc visibility and a submarine photometer. The photometer was equipped with a Weston 856 YR barrier layer photocell with a cosine filter. Absorbance and scattering characteristics of light in water were estimated with average vertical extinction coefficients (Vollenweider, 1961). The extinction coefficient was calculated from the average transmission per meter of the spectral region between 300 and 700 mu.

$$\bar{\varepsilon}_{\nu}^{\lambda} = -\ln \bar{T}$$

Temperature and dissolved oxygen measurements were taken with an oxygen meter at the surface and 1 meter intervals. Sampling depths for specific conductivity (solubridge readings), pH (color comparator), and bicarbonate ions (Standard Methods, 1965) were at the selected primary production incubation levels. Samples for nitrate (NO_3), phosphate (PO_4), biochemical oxygen demand (B.O.D.), and metal ions were collected with a plastic Van Dorn bottle from a mixed subsample of all carbon-14 incubation depths. Nitrates (Brucine Sulfate Method), phosphate (Stannous Chloride Method), and B.O.D. were analyzed according to Standard Methods (1965). For metal ion measurements including Ca, Cd, Cu, Fe, K, Mg, Mn, Na, Pb, and Zn, I used an atomic absorption spectrophotometer and the procedures of Mink (1972). River discharge readings (cfs) were obtained from the United States Department of Interior, Water Resource Division, Sandpoint, Idaho.

Phytoplankton collections, identification, and enumeration were from a concurrent study by R. F. Minter (1971). Phytoplankton were identified to the generic level. All species of each genus are represented collectively by generic name. Net forms were considered to be separate units and enumerated as number of units/l. Nannoplankton were classed as all phytoplankton passing through a 70 micron mesh net. Nannoplankton were counted in the same manner as net phytoplankton and expressed as cells/ml.

Primary production was measured with the carbon-14 light and dark bottle technique of Goldman (1960). Stock solutions of carbon-14 were prepared for each sampling period from one milliliter serum bottles containing 20 μc of $NAH^{14}CO_3$ (Amersham Corp.). A micropipette was used to subsample stock solutions for calibration. I used inoculations of 1.0 ml per bottle with an activity of 2.5 μc/ml except 5.0 μc/ml in the winter. Four-hour incubations (1000 to 1400 hours except for 900 to 1300 hours in the winter) were made at depths of 0, 1, 2, 3, 5, 7, 9, 12, and 15 m. After I removed the samples from the lake, I membrane-filtered them (pore size 0.45μ) at a shore laboratory. Carbon-14 activity of algae on filters and quench corrections were determined by liquid scintillation counting (Lind and Campbell, 1969). Available carbon used for calculations of the amount of carbon fixed was obtained from a conversion table of pH and alkalinity (R. W. Bachmann,[1] unpublished data). Calculations of carbon uptake by phytoplankton (Strickland and Parsons, 1968) were integrated with planimetry to obtain midday exposure values of mg C/m^2/4 hr. I calculated daily primary production (mg C/m^2/day) by multiplying 4-hour areal estimates by the ratio between areas under a pyrheliograph curve for the day in question and the fraction of the curve representing exposure time (Vollenweider, 1969).

I used a polyregression program to regress areal production values of the Coeur d'Alene River and Lake upon separate physical and chemical factors. The coefficient of determination (R^2), variable coefficients up to the third degree polynomial, standard error of estimate, and "F" values were calculated. The coefficient of determination (R^2) describes what proportion of the total variation in

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[1] R. W. Bachmann, The calculation of available carbon. Mimeographed report, Department of Zoology, University of Michigan, Ann Arbor, Michigan, 1961.

the midday exposure (mg $C/m^2/4$ hr) was due to regression with each physiochemical variable. The midday exposure values of mg $C/m^2/4$ hr were used as the dependent variables because of time and water mass similarities with independent variable samples. When required, independent and dependent variables were normalized by an appropriate transformation. Transformations were obtained from a first-four moments computer program. Temperature, bicarbonate ions, specific conductivity, and dissolved oxygen values used in analysis were the means of data from the euphotic zone; pH was taken as the mode pH of the euphotic zone.

In this study phytoplankton photosynthesis was considered a reliable index for measuring the impact of mine drainage on organic production in the river and lake. Therefore, it was necessary to estimate what amount of the variation in production could be attributed to separate extrinsic environmental factors of both areas. In addition, an exploratory analysis was needed to estimate the likelihood of high significance levels of temperature and light. These two factors could mask the "true" relationship of other important variables and of treatments to be used in bioassays. The attempt to analyze these relations consisted of single factor regression analyses calculated to the third power of X. Although these equations provide estimates of the relative value of each factor for predicting production, they are considered more adequate than multiple regression equations because of the plausible prior importance of light and temperature to phytoplankton production, the non-linearity of some relationships, the possibility that some variables have obscure and accidental relationships that interact or covary with other variables, incompleteness of observations, and the large number of factors to be examined.

Bioassays

I conducted bioassays with carbon-14 and phytoplankton to determine the inhibitory effects of metals to carbon fixation in algae. Zinc (zinc sulfate), Cu (copper sulfate), and Cd (cadmium sulfate) were added to phytoplankton to assess single and combined effects of metals. Additional bioassays examined the effects of different volumes (with and without metal treatments) of Coeur d'Alene and St. Joe river waters.

Bioassays were conducted in a controlled light and temperature tank at the shore laboratory from August to November, 1970. The laboratory was located at Heyburn State Park, 30 minutes from the mouth of the Coeur d'Alene River. The bottles were incubated in a 0.6 m^3 stainless steel tank. Tank temperatures were controlled by a separate submerged refrigeration unit and circulated by self-priming centrifugal pumps. A shaker plate in the tank held the incubation bottles and facilitated suspension of algal cells. Four 500-w incandescent bulbs illuminated the bioassay tank. A submarine photometer measured light at the neck level of submerged incubation bottles. The light source and temperature were regulated to simulate average light and temperature conditions in the lake.

Light intensity readings were at 260 ± 10 foot candles. The average midday intensities at the two-meter depth in the lake during the falls of 1969 and 1970 were 220 and 240 foot candles respectively.

I examined the variation in light intensity within the incubation tank with a Latin square designed experiment and an "F" test.

Division of the entire bottle rack into columns and rows facilitated blocking in two directions. Three main treatments, consisting of the same concentrations on Zn and Cu used in bioassays, were assigned to each block so that responses (CPM) to any light changes could be compared. Sub-treatments of Coeur d'Alene Lake and River aliquots were assigned to the main treatments in order to assess within-block variability. The only significant difference was for main treatments at the P .01. Only one of the three main treatments caused this difference. Apparently fewer replications of main treatments made the comparison less accurate than that of the sub-treatments. Since columns, rows, sub-treatments, and the two-way main treatment x sub-treatment interaction were not significantly different, I concluded that the variation in light intensity within the tank was not significant and therefore adequate for the tests.

Water temperatures were maintained at 20 ± 2 C for tests from August through October. During November, the test temperature was reduced to 5.7 C in accordance with *in situ* observations.

During late October, the temperature of the lake and river waters dropped below 10 C. In order to assess the effects of temperature on carbon fixation, simultaneous incubations with Coeur d'Alene and St. Joe River dilutions were conducted at 19 C and 5 C. Each temperature level and river source were assigned Cu, Zn, and two-way Cu x Zn treatments. An "F" test for the St. Joe River indicated that all treatments were significantly different at P .01. An increase in temperature for the St. Joe increased carbon fixation by approximately 40 percent. The above treatments in the Coeur d'Alene water were not significantly different. Since the amount of carbon fixed at 5 C was similar for both tests, a low temperature was used in G and H bioassays.

For each bioassay, a sample of 20 l of water was collected from the lake at a depth of 2 m. The sample was filtered through No. 20 mesh to obtain nannoplankton (cells less than 70 microns in size) and to eliminate grazing effects of zooplankton. Carbon-14 was added to the 20 l of nannoplankton, the sample mixed, and subsamples of 750 ml drawn off into 2 l premix bottles. This procedure assured that all treated premix bottles as well as controls had equal dosages of carbon-14 and algal cells. The metals zinc, copper, and cadmium in various concentrations and combinations were then added to premix bottles. Metals added from stock solutions ranged from 0.005 to 0.14 percent of the total volume (750 ml). Replicate subsamples were then taken from premix bottles into 250 ml clear-pyrex reagent bottles. The bottles were incubated for 8 or 12 hours, depending on the date of the test. After incubation, subsamples of nannoplankton from bottles were filtered onto 0.45μ membrane filters and activity of carbon-14 assimilated was counted.

Separate tests of the effect of various volumes of river water were conducted by adding aliquots of membrane filtered (pore size 0.45μ) river water to premix bottles containing lake nannoplankton and carbon-14. Prior to river water additions, each premix bottle (2 l) received uniform subsamples (750 ml) of algae and carbon-14 as described above. Volumes of river water added to premix bottles ranged from 75 ml (5 percent) to 750 ml (50 percent) of the total volume (1500 ml) in premix bottles. The volumes of river water additions of less than 50 percent (and controls) were made up with 0.45μ membrane filtered lake water (not river water). Metal additions ranged from 0.005 to 0.14 percent of the total volume in premix bottles (1500 ml). Final concentrations of Zn, Cu, Cd, Ca, K, Mg, Na, Mn, Fe, and Pb in all experimentals and controls were determined using an atomic absorption spectrophotometer. Replicate subsamples from premix bottles were taken and treated as above.

The filtration to obtain nannoplankton and the addition of carbon-14 to 20 l permitted comparison of treatments under homogeneous experimental conditions. The two replications of an entire experiment formed uniform blocks which decreased the variance, increased sample size, and procured the degrees of freedom for estimating the variance. Random selection of subsamples to be filtered from incubation bottles within each replication tended to randomly assign error effects to the treatments. I used factorial designs for bioassays because they increase the sample size by centralizing responses in the experiment on factor interactions and main effects. The treatment effects in the bioassays were not additive but multiplicative; therefore a log transformation was used in "F" tests.

RESULTS

Physical and Chemical Characteristics

(1) Coeur d'Alene River and Lake

The ionic composition and concentrations of the Coeur d'Alene River differed from that of the lake (Table 1). The order of abundance of cations in the river was Ca > Na > Mg > Zn > K > Mn > Fe > Pb > Cu > Cd which was similar in the lake except for K > Zn > Fe > Mn. Metal concentrations in the river were generally higher than in the lake, except for similar concentrations of Fe, Cu, and Pb. The concentrations of Zn in the river increased during low flow in the summer and fall months with ranges of 2.4 - 11.2 mg Zn/l in 1969 and 2.9 - 5.2 mg Zn/l in 1970. These measurements exceeded values reported toxic to fresh-water algae (Report of the Committee on Water Quality Criteria, 1968).

The pH values of the river were usually lower than those of the lake. The lowest pH values for the river (6.20 - 7.60) and lake (6.60 - 7.45) occurred in the winter and spring. The highest pH measurements were during the summer and fall with 6.40 - 7.30 in the river and 6.80 - 8.15 in the lake.

The concentrations of inorganic carbon and bicarbonate ions were similar in the river and lake. Total CO_2 for both areas ranged from 3.9 to 11.1 mg CO_2/l and bicarbonate ions from 13.0 to 25.0 mg HCO_3/l. The low levels of total CO_2 and bicarbonate ions may approach limiting conditions for primary production, but these concentrations vary with photosynthetic activity of phytoplankton.

Table 1.--Physical and chemical characteristics of the Coeur d'Alene River and Lake, May, 1969, to November, 1970

Factor	N[c]	Coeur d'Alene River			Coeur d'Alene Lake		
		Mean	S.E.[b]	Range	Mean	S.E.	Range
Sodium (mg Na/l)	26	3.40	±0.20	1.4 - 5.1	1.7	±0.1	1.0 - 2.5
Potassium (mg K/l)	26	0.90	±0.05	0.5 - 1.4	0.7	±0.03	0.5 - 1.1
Magnesium (mg Mg/l)	32	3.20	±0.30	0.4 - 6.4	1.5	±0.1	0.2 - 2.5
Calcium (mg Ca/l)	32	7.80	±0.70	0.6 - 13.1	4.4	±0.3	0.5 - 7.1
Copper (mg Cu/l)[a]	28	0.10	±0.02	0.0 - 0.6	0.10	±0.02	0.0 - 0.4
Iron (mg Fe/l)	32	0.30	±0.09	0.0 - 2.0	0.3	±0.09	0.3 - 2.0
Manganese (mg Mn/l)[a]	32	0.60	±0.09	0.01 - 1.90	0.10	±0.01	0.0 - 0.3
Zinc (mg Zn/l)[a]	32	2.70	±0.40	0.1 - 11.2	0.40	±0.05	0.01- 1.00
Cadmium (mg Cd/l)[a]	18	0.02	±0.005	0.01 - 0.05	0.010	±0.001	0.01- 0.02
Lead (mg Pb/l)[a]	32	0.20	±0.07	0.0 - 1.9	0.20	±0.07	0.0 - 1.7
Nitrate (mg NO_3/l)	30	0.50	±0.10	0.0 - 3.1	0.20	±0.06	0.0 - 1.1
Phosphate (mg PO_4/l)	30	0.30	±0.10	0.0 - 2.8	0.10	±0.03	0.0 - 1.7
Bicarbonate (mg HCO_3/l)	35	17.10	±0.40	13.0 - 25.0	17.2	±0.3	13.8 - 20.0
Total CO_2 (mg CO_2/l)	35	5.50	±0.20	3.9 - 11.1	5.0	±0.1	3.6 - 8.2
Dissolved oxygen (mg/l)	29	8.70	±0.50	4.8 - 11.6	8.0	±0.4	5.2 - 13.6
B.O.D. (mg/l)	26	1.50	±0.20	0.3 - 3.8	1.6	±0.2	0.6 - 4.9
pH	35	----	----	6.4 - 7.3	----	----	6.6 - 8.2
Conductance (umho/cm at 25 C)	35	111	±8	<50 - 315	46	±3	<50 - 75
Temperature (Centigrade)	35	13.7	±1	2.0 - 21.7	12.7	±1.0	2.6 - 19.7
Sicche disc (meters)	35	2.3	±0.9	1.5 - 3.1	2.8	±0.1	0.8 - 4.9
Extinction coefficient ($\bar{\varepsilon}_{\nu}^{\lambda}$)	34	0.744	±0.046	0.437- 1.537	0.717	±0.049	0.408-1.857

[a]Trace values of <0.01 mg/l treated as 0.01 mg/l.

[b]Standard error

[c]Number of observations

Nitrates and phosphates were higher in the river than in the lake with maximum ranges for the river occurring during high flow in the spring months (0.00 - 3.08 mg NO_3/l and 0.05 - 2.75 mg PO_4/l). The lowest concentrations of nitrate (0.00 - 0.30 mg NO_3/l) and phosphate (0.00 - 0.08 mg PO_4/l) were observed in the lake during periods of high primary production in late summer and fall.

The levels of dissolved oxygen and B.O.D. were similar in the river and lake. The low biochemical oxygen demand (0.30 - 4.90 mg/l) of the area waters indicates minimal oxygen requirements by bacteria for decomposing organic matter. Temperatures were lower in the river except during the summer months when they ranged from 1 - 4 C higher than the lake. Secchi disc readings and extinction coefficients indicated that deeper light penetration and less scattering and absorbance of light occurred in the lake during the summer and fall months while the opposite conditions existed in the river during the winter and early spring.

The major cation concentrations of Ca, Na, Mg, and Zn in the Coeur d'Alene River increased with higher specific conductance values from 50 - 315 μmho/cm at 25 C. The relationships between concentrations of major metal ions and conductance are expressed as regression equations (Figure 2). It is evident that a record of conductivity can be used to predict the Ca, Na, Mg, and Zn concentrations in the river except at high flow in the spring. Specific conductance values during high flows were usually 50 μmho/cm at 25 C.

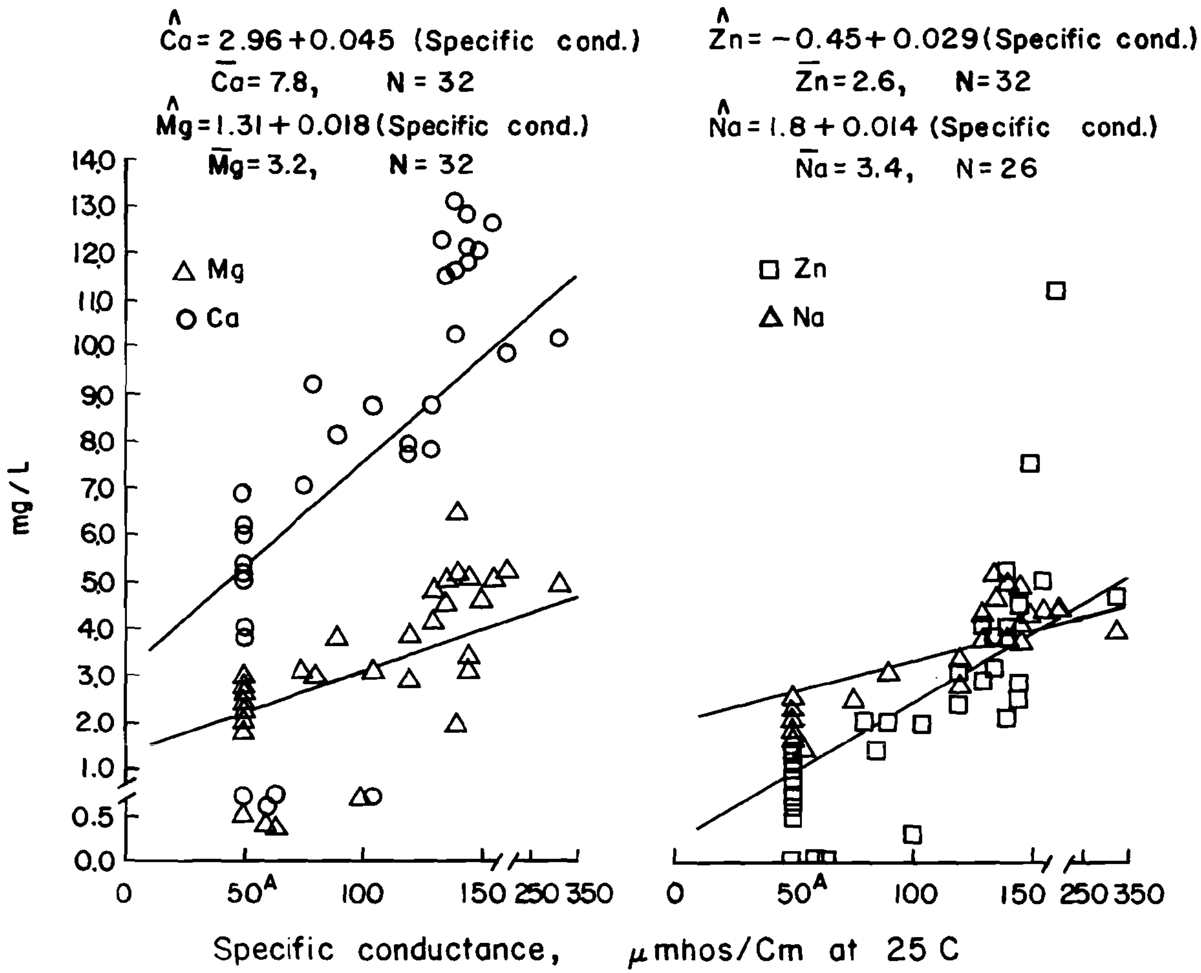

Fig. 2.--Regression of major cation concentrations on specific conductance in the Coeur d'Alene River from May, 1969, to November, 1970

[A] Conductance of <50 treated as 50 μmhos/cm at 25 C.

(2) Coeur d'Alene River and St. Joe River

Water quality during the summers of 1969 and 1970 differed for the Coeur d'Alene River and St. Joe River in ionic composition, metal concentrations, specific conductivity, pH, bicarbonates, total CO_2, and light conditions (Table 2). Both rivers had homothermous waters with no oxygen stratification. Other physical and chemical characteristics followed the trends expressed in the Coeur d'Alene River and Lake comparison. The order of abundance of cations in the Coeur d'Alene River followed the same sequence as in the preceding river data. The only exception occurred in 1970 with Mg > Na. The cations in the St. Joe River were in the following order of abundance: Ca > Na > Mg > K > Mn > Fe > Pb > Zn > Cd > Cu. Metal ion concentrations in the St. Joe River were lower except for similar levels of K, Fe, Pb, Cd, and Cu. The specific conductivity for the Coeur d'Alene River (50 - 145 μmho/cm at 25 C) usually exceeded the St. Joe River with 50 - 65 μmho/cm at 25 C. The pH (6.6 - 7.2), total CO_2 (3.9 - 7.0 mg/l), HCO_3 (13.0 - 20.0 mg/l), and river discharge (355 - 2220 cfs) of the Coeur d'Alene River tended to be lower than the St. Joe River. Light penetration by Secchi disc readings in the Coeur d'Alene River (1.8 - 3.1 m) was lower than the St. Joe River. Extinction coefficients of 0.45 - 1.26 indicated greater light scattering and absorbance in the Coeur d'Alene River.

Phytoplankton Composition

A total of 59 phytoplankton genera were identified in the Coeur d'Alene River, Lake, and St. Joe River. Thirty-nine genera were

Table 2.--Physical and chemical characteristics of the Coeur d'Alene River and St. Joe River, July and August, 1969, and June to September, 1970. Means of four samples in 1969 and six samples in 1970

Factor	Coeur d'Alene River				St. Joe River			
	1969		1970		1969		1970	
	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Sodium (mg Na/l)	3.7	2.9-4.0	3.3	1.8-4.8	1.7	1.6-2.0	1.4	1.1-1.6
Potassium (mg K/l)	0.9	0.7-1.0	0.8	0.5-1.1	0.8	0.6-0.9	0.8	0.6-0.9
Magnesium (mg Mg/l)	2.8	1.9-3.4	3.8	2.2-5.0	1.2	0.9-1.4	1.5	1.0-1.7
Calcium (mg Ca/l)	10.5	7.9-12.1	8.3	5.0-12.8	7.4	5.0-8.9	5.4	3.8-7.1
Copper (mg Cu/l)[a]	0.03	0.00-0.10	0.09	0.05-1.00	0.03	0.00-0.10	0.03	0.01-0.10
Iron (mg Fe/l)	0.3	0.2-0.3	0.2	0.1-0.3	0.2	0.1-0.3	0.1	---
Manganese (mg Mn/l)[a]	0.4	---	0.7	0.2-1.1	0.03	0.00-0.10	0.1	---
Zinc (mg Zn/l)[a]	2.5	2.1-2.9	2.7	1.1-4.5	0.08	0.00-0.10	0.05	0.01-0.10
Cadmium (mg Cd/l)[a]	---	---	0.02	0.01-0.04	---	---	0.01	---
Lead (mg Pb/l)[a]	0.3	0.0-0.5	0.1	---	0.10	0.0-0.5	0.1	---
Nitrate (mg NO_3/l)	0.5	0.4-0.6	0.3	0.0-0.9	---	---	0.3	0.0-0.6
Phosphate (mg PO_4/l)	0.09	0.00-0.10	0.1	0.03-0.20	---	---	0.09	0.00-0.20
Bicarbonate (mg HCO_3/l)	19.2	18.8-20.0	16.3	13.0-18.0	29.3	26.0-35.0	25.0	18.8-29.0
Total CO_2 (mg CO_2/l)	5.5	5.0-7.0	5.0	3.9-6.3	---	---	7.5	5.3-11.0
Dissolved oxygen (mg/l)	10.9	7.4-11.6	10.0	7.1-11.5	---	---	8.6	7.5-9.6
B.O.D. (mg/l)	2.1	1.7-2.6	1.0	0.6-2.4	---	---	0.5	0.2-0.7
pH	---	6.6-7.2	---	6.6-7.2	---	7.0-7.5	---	6.6-7.4
Conductance (umho/cm at 25 C)	138	115-145	103	50-145	58	<50-65	<50	---
Temperature (Centigrade)	19.4	17.1-22.4	19.9	18.3-21.7	20.3	20.1-20.5	19.3	17.1-20.6
River discharge (cfs)	485	355-662	912	438-2220	617	461-861	1788	616-4978
Sicche disc (meters)	2.4	1.8-3.1	2.5	1.8-3.0	3.3	2.9-3.8	3.2	2.6-3.7
Extinction coefficient ($\bar{\varepsilon}_{\nu}^{\lambda}$)	0.60	0.45-0.94	0.85	0.64-1.26	---	---	0.59	0.40-0.69

[a]Trace values of <0.01 mg/l treated as 0.01 mg/l.

observed in the Coeur d'Alene River, 38 genera in the St. Joe River, and 30 genera in the Coeur d'Alene Lake (Table 3, Minter, 1971).

(1) Coeur d'Alene River and Lake

The average density of net phytoplankton was 63,460 units/l in the lake as compared to 4,640 units/l in the river. The most abundant diatoms in both areas were Melosira, Tabellaria, Asterionella, Synedra, and Fragillaria. Maximum seasonal pulses of Melosira, Tabellaria, and Asterionella in the lake and river occurred during the fall months.

Nannoplankton comprised from 90-97 percent of the total phytoplankton in the study area. The most abundant forms were Melosira and Tabellaira with 1955 cells/ml in the lake and 83 cells/ml in the river. Seasonal pulses were similar in both the river and lake with peaks of Melosira in the spring and Tabellaria in the fall.

(2) Coeur d'Alene River and St. Joe River

Net phytoplankton counts in the St. Joe River averaged four times higher than in the Coeur d'Alene River with 1,770 and 470 units/l respectively. The most abundant forms in both rivers were diatoms dominated by Melosira. The diatoms Melosira, Tabellaria, Synedra, Ceratoneis, Dinobryon, and Frustulia comprised 50 percent of the net phytoplankton in the St. Joe River. Other abundant forms were the green algae Ulothrix, Mougeotia, Chaetophora, and Volvox. Diatoms comprised over 90 percent of the net phytoplankton in the Coeur d'Alene River. The dominant forms were Melosira, Tabellaria, and Asterionella.

The average number of nannoplankton in the St. Joe and Coeur d'Alene Rivers were 69 and 24 cells/ml, respectively. Diatoms comprised

Table 3.--Phytoplankton genera in the Coeur d'Alene Lake, River, and St. Joe River (Minter, 1971)

Genus	Station[a]			Genus	Station		
	I	III	IV		I	III	IV
			Chlorophycophyta				
Ankistrodesmus	X		X	*Mougeotia*	X	X	X
Cerasterias			X	*Palmodictyon*		X	
Chaetophora			X	*Pediastrum*		X	X
Cladophora		X		*Rhizoclonium*		X	
Closterium	X	X	X	*Spirogyra*	X	X	X
Cosmarium		X		*Spondylosium*			X
Desmidium			X	*Staurastrum*	X	X	
Docidium	X	X		*Stigeoclonium*	X	X	
Eudorina	X	X	X	*Ulothrix*	X	X	X
Gleocystis		X		*Volvox*	X	X	X
Hydrodictyon		X		*Zygnema*	X	X	
Micrasterias		X	X				
			Chrysophycophyta				
Amphora		X	X	*Gomphonema*			X
Asterionella	X	X	X	*Gyrosigma*		X	X
Bumilleria			X	*Mastogloia*			X
Camplodiscus			X	*Melosira*	X	X	X
Ceratoneis	X	X	X	*Navicula*	X	X	X
Chrysosphaerella			X	*Opephora*	X		
Cocconeis			X	*Rhopalodia*			X
Coscinodiscus		X	X	*Stauroneis*			X
Cyclotella	X			*Surirella*			X
Denticula			X	*Synedra*	X	X	X
Diatoma			X	*Synura*		X	X
Dinobryon	X	X	X	*Tabellaria*	X	X	X
Fragillaria	X	X	X	*Tribonema*	X	X	X
Frustulia	X	X	X				
			Cyanophycophyta				
Anabaena	X	X	X	*Marssoniella*	X		
Aphanizomenon	X	X	X	*Oscillatoria*	X	X	X
Calothrix		X		*Phormidium*		X	
Coelospharium	X			*Spirulina*	X	X	X
			Pyrrhophycophyta				
Ceratium	X	X					

[a]Station I, Coeur d'Alene Lake; Station III, Coeur d'Alene River; Station IV, St. Joe River.

80 percent of the nannoplankton in the St. Joe River and approached 100 percent in the Coeur d'Alene River. The major forms in the St. Joe River in decreasing order of abundance were Melosira, Tabellaria, Synedra, Ceratoneis, and Frustulia. The dominant forms in the Coeur d'Alene River were Melosira, Tabellaria, and Asterionella.

Primary Production

(1) Coeur d'Alene River and Lake

Primary production of Coeur d'Alene Lake and River over 19 months averaged 476.6 and 269.2 mg C/m^2/day with relatively large fluctuations on a seasonal basis (Figure 3). The annual mean productivities during 1970 (includes December, 1969) for the lake and river were 533.3 and 312.3 mg C/m^2/day, respectively. The depth of the trophogenic layer in the lake varied from 5 to 15 m and from 5 to 9 m in the river.

The mean productivities for optimal growth periods during summers and falls of 1969 and 1970 were 418.6 and 727.0 mg C/m^2/day in the lake and 255.6 and 470.0 mg C/m^2/day for the river. The above values were calculated from synchronous dates in 1969 and 1970. The lower areal production of both the lake and river in 1969 as compared to 1970 agrees with observations in 1969 of shallower trophogenic layers, colder epilimnetic temperatures, and greater irregularities in solar radiation.

Univariate polyregression analyses of areal production (mg/C/m^2/4 hr) and physiochemical factors in the Coeur d'Alene River indicated that bicarbonate ion concentrations produced the largest R^2 value (Table 4). The regression with the best fit for bicarbonate ions

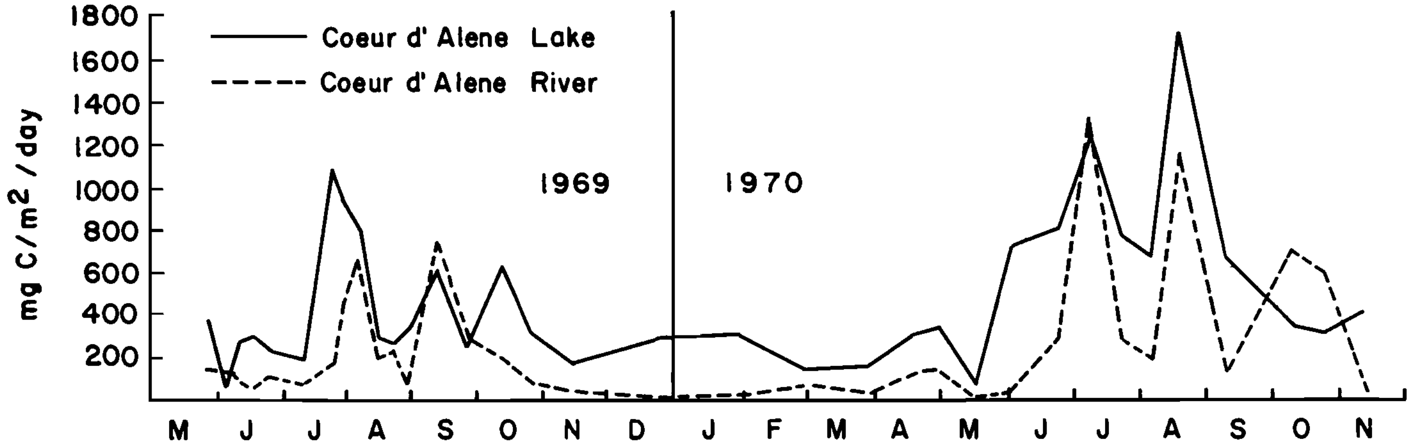

Fig. 3.--Seasonal variation of primary productivity (mg C/m^2/day) in the Coeur d'Alene Lake (Station I) and River (Station III)

Table 4--Polyregression equations and coefficients of determination (R^2) of the total variation in primary production due to association between mg $C/m^2/4$ hr and physiochemical factors in the Coeur d'Alene River, May, 1969, to November, 1970.

Factor	N	Constant	X	X^2	X^3	R^2	s.y.x.[f]	F
Sodium (mg Na/l)	26	1.303	0.1611			0.133	0.481	3.690*
Potassium (mg K/l)	26	11.374	-34.378	39.554	-14.584	0.145	0.499	1.247
Magnesium (mg Mg/l)	32	1.875	-0.587	0.274	-0.030	0.171	0.449	1.928
Calcium (mg Ca/l)	32	1.490	0.040			0.110	0.449	3.694*
Copper (mg Cu/l)	28[b]	0.192	-0.104			0.016	0.458	0.436
Iron (mg Fe/l)	32[b]	1.395	1.634	-1.167		0.088	0.462	1.403
Manganese (mg Mn/l)	32[b]	1.109	1.951	-1.122		0.106	0.458	1.728
Zinc (mg Zn/l)	32[c]	1.270	2.244	-1.807		0.168	0.442	2.903*
Cadmium (mg Cd/l)	18[d]	0.112	3.337	-1.376		0.245	0.528	2.433
Lead (mg Pb/l)	32[b]	1.492	1.472	-1.157		0.071	0.473	1.106
Nitrate (mg NO_3/l)	30[b]	1.791	-0.002			0.0001	0.480	0.003
Phosphate (mg PO_4/l)	30[d]	1.920	-0.037			0.017	0.476	0.491
Bicarbonate (mg HCO_3/l)	35[a]	-3.849	4.652			0.280	0.404	12.830***
pH	35[d]	-1.708	0.198			0.022	0.471	0.751
Total CO_2 (mg CO_2/l)	35[a]	1.940	-0.110			0.061	0.461	2.137
Dissolved oxygen (mg/l)	29	1.551	2.732			0.025	0.465	0.691
B.O.D. (mg/l)	26[c]	1.933	-0.150			0.012	0.464	0.300
Conductance (umho/cm at 25 C)	35[a]	-0.738	0.461	-0.019		0.206	0.431	4.157**
Temperature (Centigrade)	35[e]	0.988	0.230			0.212	0.422	8.885***
River discharge (cfs)	35[a]	3.385	-0.514			0.206	0.424	8.586***
Sicche disc (meters)	35[d]	1.033	0.243	-0.016		0.056	0.469	0.958
Extinction coefficient ($\vec{\varepsilon}_{\nu}^{\lambda}$)	34[b]	-3.669	12.033	-6.356		0.153	0.447	2.795*

[a,b,c,d,e] Represent transformations used to normalize data. a = log (X); b = $\sqrt{X}$; c = log (X + 1); d = Arcsin $\sqrt{X}$; e = $\sqrt{X + 1}$.

[f] Standard error of estimate

*,**,*** Regression coefficient significantly different from 0 at the 0.10, 0.05, and 0.01 probability levels, respectively.

was linear and highly significant (P .01). Linear equations were chosen for temperature and river discharge and the quadratic equation for specific conductivity. These regressions showed significance at P .05. Significant regressions at P .10 included linear equations for Na and Ca and quadratic for Zn and light extinction. Although K, Mg, Mn, and Cd produced no significant regressions, they displayed higher R^2 values than remaining, non-significant equations.

In polyregression analyses of areal production and physio-chemical parameters in Coeur d'Alene Lake, a maximum R^2 value of .448 was caused by dissolved oxygen (Table 5). The dissolved oxygen equation was cubic and highly significant (P .01). A cubic equation was selected for Secchi disc readings and linear equations for temperature, K, and Fe. Each regression equation of the preceding four variables produced significance at P .05. Significant regressions at P .10 included light extinction (cubic) and linear for Cu and nitrate. The cubic regression for bicarbonate ions was not significant but yielded an R^2 value of .150.

The quadratic and cubic equations were used only when they indicated a better fit by an improvement in R^2 and a relatively large decrease in the standard error of the estimate. Although R^2 values were low for the important factors, they were considered adequate for such a highly variable environment. They explain a small, but significant, portion of the total variance in primary production.

Primary production values, phytoplankton densities, and metal concentrations from additional stations showed the major differences between the Coeur d'Alene River and Lake (Figure 4). These supple-mentary stations (IA, IIA, and II in the lake and IIB at the river

Table 5.--Polyregression equations and coefficients of determination (R^2) of the total variation in primary production due to association between mg C/m^2/4 hr and physiochemical factors in the Coeur d'Alene Lake, May, 1969, to November, 1970

Factor	N	Constant	X	X^2	X^3	R^2	s.y.x.[e]	F
Sodium (mg Na/l)	26	21.648	-4.187			0.102	4.406	2.721
Potassium (mg Mg/l)	26	2.358	-12.213			0.185	4.200	5.461**
Magnesium (mg Mg/l)	32	12.360	0.878			0.013	4.631	0.403
Calcium (mg Ca/l)	32	9.112	2.888	-0.366		0.095	4.511	1.521
Copper (mg Cu/l)	28[a]	16.757	-9.934			0.102	4.570	2.965*
Iron (mg Fe/l)	32[a]	16.128	-5.380			0.137	4.331	4.760**
Manganese (mg Mn/l)	32[a]	16.354	-8.834			0.026	4.600	0.809
Zinc (mg Zn/l)	32[b]	15.391	-12.986			0.054	4.533	1.726
Cadmium (mg Cd/l)	18	11.075	277.400			0.083	4.624	1.440
Lead (mg Pb/l)	32[a]	14.768	-2.754			0.028	4.596	0.862
Nitrate (mg NO_3/l)	30[a]	12.444	0.620			0.121	4.287	3.845*
Phosphate (mg PO_4/l)	30[a]	15.254	-7.592			0.087	4.370	2.656
Bicarbonate (mg HCO_3/l)	35	296.990	-53.452	3.288	-0.066	0.154	4.384	1.881
pH	35	419.894	-112.368	7.759		0.072	4.520	1.239
Total CO_2 (mg CO_2/l)	35	9.382	6.376			0.083	4.600	0.277
Dissolved oxygen (mg/l)	28	-122.733	50.410	-5.872	0.216	0.448	3.410	6.495***
B.O.D. (mg/l)	25[b]	15.928	-4.934			0.075	4.869	1.852
Conductance (umho/cm at 25 C)	35	14.288	-0.010			0.0002	4.619	0.005
Temperature (Centigrade)	35[c]	8.066	0.243			0.115	4.347	4.269**
Sicche disc (meters)	35[d]	375.038	-633.541	357.945	-65.516	0.246	4.138	3.375**
Extinction coefficient ($\bar{\varepsilon}_{\nu}^{\lambda}$)	34[c]	-238,044	117.066	-17.572	0.849	0.200	4.237	2.500*

[a,b,c,d]Represent transformations used to normalize data. a = $\sqrt{X}$; b = log (X + 1); c = Arcsin $\sqrt{X}$; d = $\sqrt{X + 1}$.

[e]Standard error of estimate.

*, **, ***Regression coefficient significantly different from 0 at the 0.10, 0.05, and 0.01 probability levels, respectively.

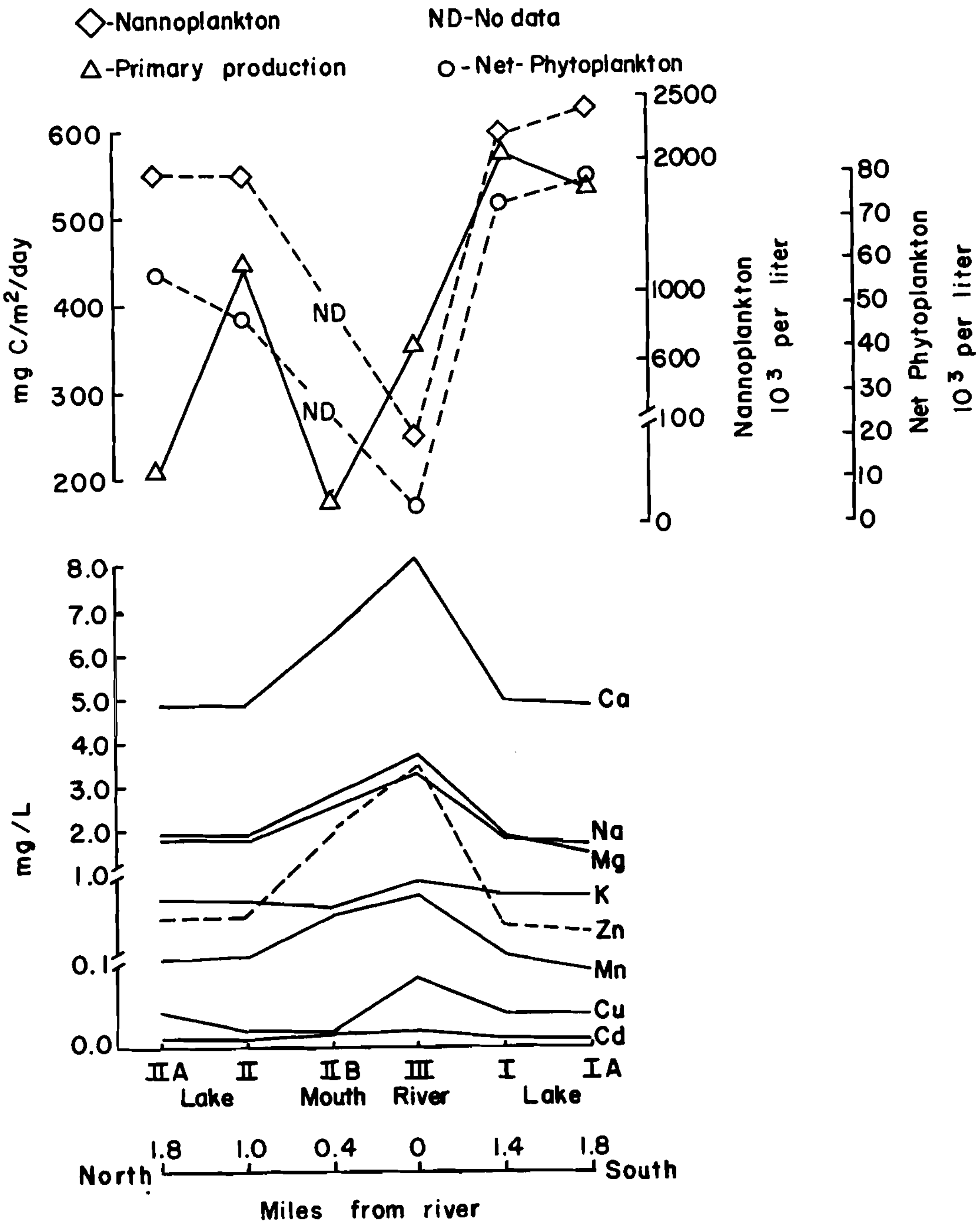

Fig. 4.--Differences in primary production, phytoplankton density, and ionic concentrations between the Coeur d'Alene River and Lake. Values for each station are means of eight dates from December, 1969, to November, 1970

mouth) assessed waters adjacent the more frequently sampled stations of I (lake) and III (river). These intermittent stations were sampled bimonthly from December, 1969, to November, 1970, along with Stations I and III (May, 1969, to November, 1970). The lowest primary production values occurred at the river mouth (IIB). The delta was shallow in depth and almost continually under the influence of upstream winds and wave action. Such conditions suspended tailings sediments and decreased light penetration and increased extinction. Metal concentrations at the mouth were lower than those of the river channel, indicating dilution by lake waters. The low to moderate production levels of the Coeur d'Alene River usually corresponded with low phytoplankton densities and high metal concentrations. The only pronounced differences between lake stations were lower primary production values and phytoplankton densities at the northern stations while the opposite situation occurred to the south at I and IA. Concentrations of metals at lake stations were relatively homogeneous but less than levels in the river and mouth.

(2) Coeur d'Alene River and St. Joe River

Primary production experiments were conducted from June to September, 1970. The mean areal production for the summer was higher for the St. Joe River than the Coeur d'Alene River. Daily production averaged 1262.3 mg C/m^2/day for the St. Joe River and 498.3 mg C/m^2/day for the Coeur d'Alene River. Maximum values of primary production occurred in July for the St. Joe River and in July and August for the Coeur d'Alene River (Figure 5). The depth of the trophogenic layer

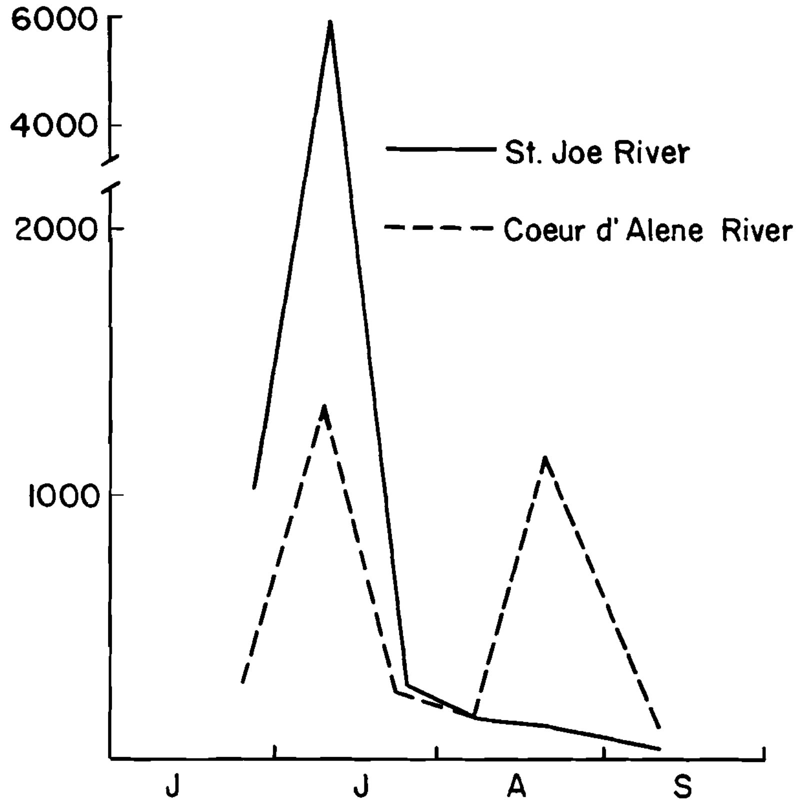

Fig. 5.--Variation in primary productivity (mg C/m^2/day) for the Coeur d'Alene River and St. Joe River during the summer of 1970

in the St. Joe River was 12 -18 m compared to 7 - 9 m in the Coeur d'Alene River.

Bioassays

Zinc, copper, and cadmium treatment levels were chosen on the basis of concentrations in the Coeur d'Alene River because of potential zinc toxicity and synergism with copper and cadmium. Additions of river water simulated the dilution effect of the lake in the delta area. In all experiments "F" tests for replications indicated no significant differences. The response characteristics of carbon-14 fixation (CPM or counts per minute) by nannoplankton are graphed as functions of metal concentrations and river dilutions. Bicarbonate ion concentrations, pH, and hardness (softwater < 60 mg $CaCO_3$/l) remained relatively constant throughout the test period and are included with temperature and incubation times for each test on respective response surfaces.

(1) Bioassays A, B, and C: metal toxicity tests

Metal additions of 0.10, 0.30, 0.50, 0.75, 1.00, 1.50 mg Zn/l; 0.10, 0.30, 0.50, 0.75 mg Cu/l; and 0.10, 0.30 mg Cd/l produced a wide range of inhibitory effects (Figure 6). Algal assays A and C had the same treatment levels for metals. Test B had higher concentrations of all metals. All experiments except Test C were conducted under controlled light and temperature conditions. Test C was incubated in the lake to verify response trends obtained in laboratory Test A. The nannoplankton at the lake site (Station I) constituted the source of algae used in all bioassays.

	Test		
	A	B	C
Date	8-27	8-29	9-15
Water	Lake	Lake	Lake
mg HCO_3/l	19.0	18.0	17.0
mg $CaCO_3$/l	19.0	19.0	19.0
pH	6.8	6.9	7.1
Ave. Temp.--C	20.0	20.2	15.6
Incubation--hrs	12	12	8

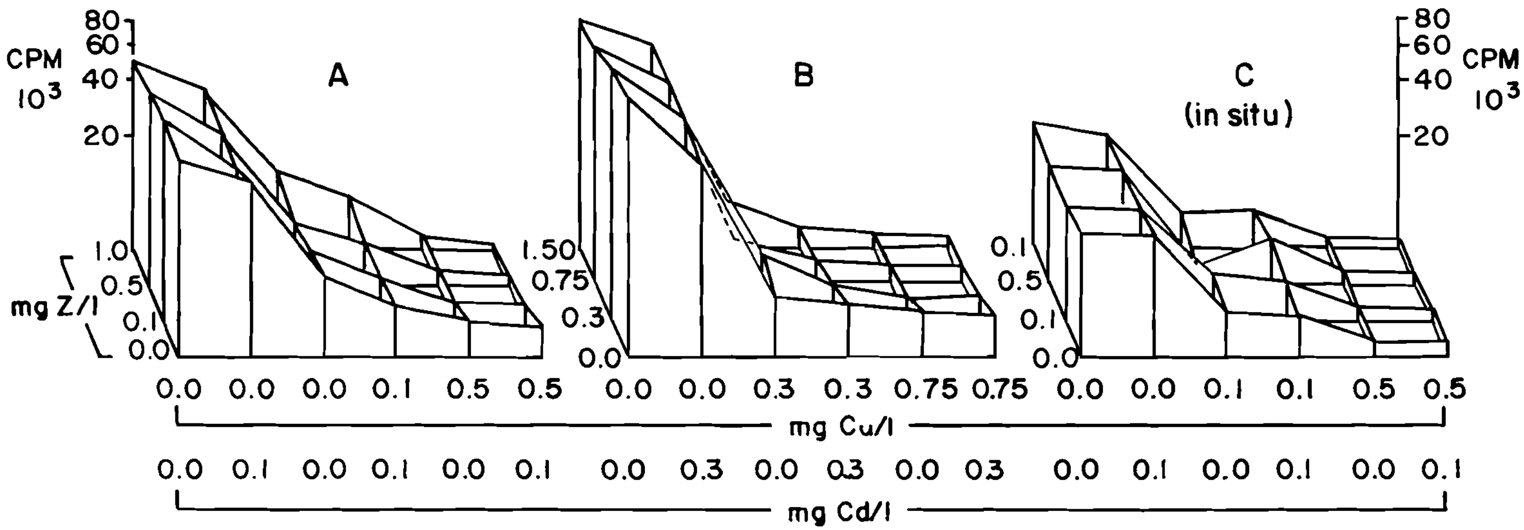

Fig. 6.--The effect of separate and interacting concentrations of zinc, copper, and cadmium (mg/l) on carbon-14 uptake (CPM) by phytoplankton

Treatments of separate metals in each of the three tests produced the following toxic order Cu > Cd > Zn. The "F" values for main metal effects indicated significant differences at P .01. The only exception was Cd in experiment C at P .05. The "F" values for Cu were so large that they indicate Cu caused an overriding effect in interactions.

The two factor interactions of Cu x Zn for each of the three tests produced significant differences (P .01). The two-way Cd x Zn interactions showed no significant differences. The remaining "F" values for two-way Cu x Cd interactions and three-way Cu x Cd x Zn interactions for experiments B and C yielded significant differences at P .01. In experiment A, the two-way Cu x Cd interaction indicated a significant difference at P .05, while three-way Cu x Cd x Zn interactions showed no significant difference.

The overall characteristics for the three tests were quite similar. The largest mean differences in carbon fixed (CPM) were caused by Cu treatments. The two-way Cu x Zn interactions produced the second highest mean differences. Inhibitory effects increased with the addition of more than one metal. High concentrations in two-factor interactions of Cu x Zn yielded greater inhibitory effects than low concentrations in two-way interactions of Cu x Zn, Cu x Cd, Cd x Zn, and three-way interactions of Cu x Cd x Zn. Maximum inhibition of carbon fixation was caused by high concentrations in the three-way Cu x Cd x Zn interactions.

(2) Bioassays D, E, and F: Coeur d'Alene River dilutions

Copper and zinc treatments were used in combination with river dilutions because of the apparent high toxicity of copper and two-way Cu x Zn interactions (Figure 7). Treatment levels in experiment D consisted of 0, 5, 15, and 25 percent river water; 0.00, 0.10, and 0.50 mg/l zinc; and 0.00 and 0.10 mg/l of copper. Results indicate that increased volume of river water decreases the inhibitory effects of metals. The low valley-like depression in Figure 7 expresses the extreme toxic effects of copper and two-way Cu x Zn interactions. The high point on the response surface at the intersection of 15 percent river water and the two-way Cu x Zn treatment (0.10 mg/l of Cu and 0.10 mg/l of Zn) resulted from omission of Cu. All treatments indicated significant differences at the P .01.

Bioassays E and F were designed to assess the effects of increased volumes of river water on carbon fixation. Treatment levels were set at 0, 10, 20, 30, 40, and 50 percent river water; 0.00 and 0.50 mg/l of zinc; and 0.00 and 0.10 mg/l of copper. The response surface for Test E indicates that increased volumes of river water stimulated carbon fixation while mean differences between copper- and non-copper-containing treatment levels remained relatively the same (Figure 7). All but the following treatments caused significant differences at P .01, the two-way dilution x Zn interaction at P .05, and the three-way dilution x Cu x Zn interaction at the P .10. No significant difference was indicated for the two-way Cu x Zn interaction.

	TEST					
	D		E		F	
Date	9-29		10-17		10-22	
Water	lake	river	lake	river	lake	river
mg HCO_3/l	17.0	18.0	17.5	18.0	18.1	16.0
mg $CaCO_3$/l	24.2	51.5	24.2	50.3	28.2	51.0
pH	7.0	7.1	7.2	7.1	7.3	6.8
Ave. Temp.- C	19.4		18.8		19.5	
Incubation - hrs	12		12		10	

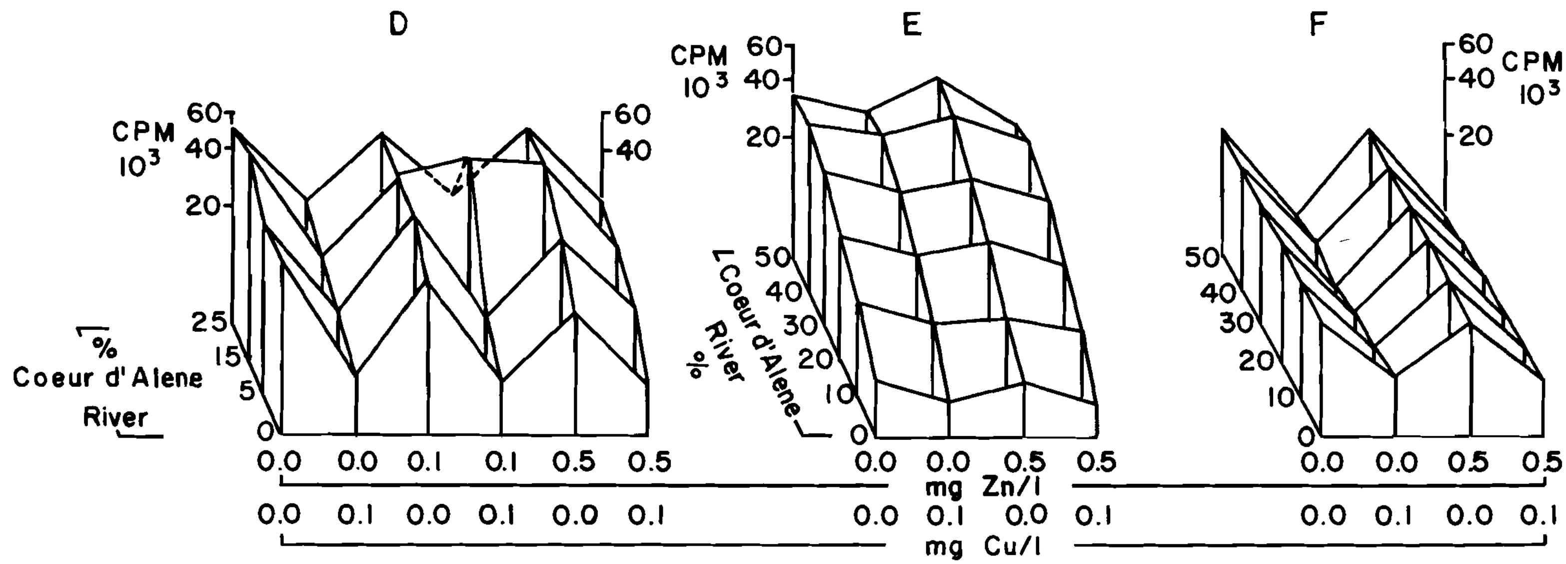

Fig. 7.--The effect of dilutions of Coeur d'Alene River water (%) with separate and interacting concentrations on zinc and copper (mg/l) on carbon-14 uptake (CPM) by phytoplankton

Test F indicated that the inhibitory effects of metals were antagonized but not to the same extent as in Test E. In Test F, increased volumes of river water stimulated carbon fixation but only at the non-copper treatment levels. When copper and two-factor Cu x Zn treatments were present and inhibition occurred, mean differences from non-copper treatments increased with higher river dilutions (Figure 7). All treatments produced significant differences at P .01 except for no significance in the Zn treatment and the two-way Cu x Zn interaction.

(3) Bioassays G and H: simultaneous Coeur d'Alene River and St. Joe River

In algal assays G (Coeur d'Alene River) and H (St. Joe River), I assessed the effects of dilutions from different rivers. Treatment levels were the same as in Tests E and F. The temperature of these bioassays was reduced to 5.7 C in accordance with the *in situ* observations.

In Test G, the non-copper treatments and increased volumes of Coeur d'Alene River water resulted in relatively uniform levels of carbon fixation. When Cu and two-factor Cu x Zn treatments were present and inhibition occurred, the mean differences from non-copper treatment levels decreased with higher river dilutions (Figure 8). This was opposite to what was observed in Test F. The two-way dilution x Zn interaction produced significant differences at P .05 and the three-way dilution x Cu x Zn interaction at P .10 while the two-way Cu x Zn interaction was not significant. All remaining treatments caused significant differences at P .01.

	Test		
	G		H
Date	11/26		
Water	lake	river	river
mg HCO_3/l	15.5		
mg CaCO /l	20.9	30.3	21.0
pH	6.8		
Ave. temp.- C		5.7	
Incubation - hrs		8	

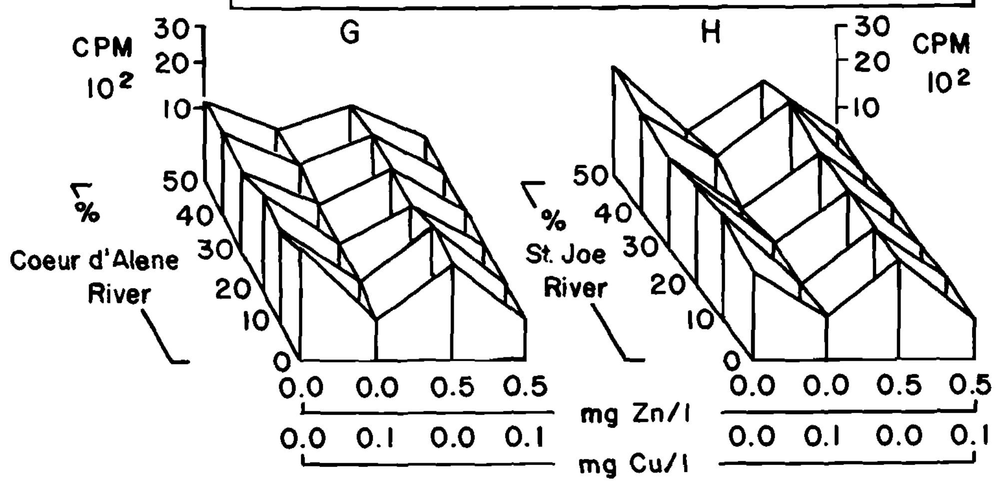

Fig. 8.--The effect of dilutions of Coeur d'Alene and St. Joe River waters (%) with separate and interacting concentrations of zinc and copper (mg/l) on carbon-14 uptake by phytoplankton

In Test H, increased St. Joe River dilutions with all metals present caused relatively no difference in the amount of carbon fixed. The presence of Cu and two-factor Cu x Zn treatments caused inhibition with mean differences remaining relatively unchanged (Figure 8). The two-way Cu x Zn and dilution x Cu interactions were the only treatments that showed no significant differences. All other treatments indicated significant differences at P .01.

The comparison of the Coeur d'Alene and St. Joe River assays by an "F" test indicated that only the two-way Cu x Zn interaction produced no significant difference. All remaining treatments caused significant differences at P .01.

Changes in primary production and phytoplankton composition of the lake and water quality of both the lake and river were considered the major causes of discrepancies observed in response surfaces. Metal concentrations of Ca, Na, Mg, K, Mn, Fe, Pb, Cd, and treatments Zn and Cu varied with different river dilution levels (Figure 9). In all river dilution tests the Cu ion occurred at lower concentrations than at the treatment addition of 0.1 mg Cu/l. These Cu concentrations were indicative of low environmental levels in the Coeur d'Alene and St. Joe Rivers and of uptake by different materials. The zinc ion occurred at higher concentrations in the Coeur d'Alene River dilution tests than at the treatment addition of 0.50 mg Zn/l. Zinc concentrations increased with increased volumes of Coeur d'Alene River water and consequent higher environmental levels of Zn. The opposite occurred with increased volumes of St. Joe River water (Test H) and concurrent declines in Zn levels. Accordingly, major cation

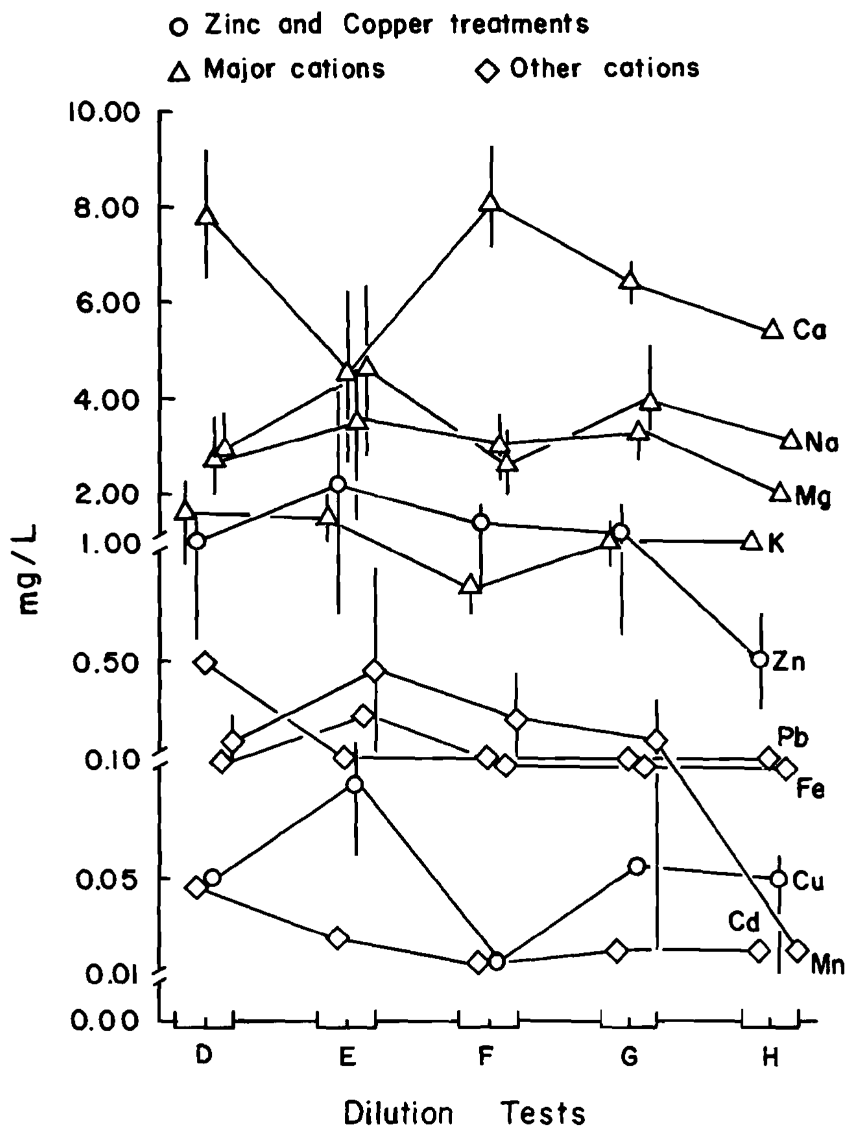

Fig. 9.--Means and ranges of metal ion concentrations in dilution tests

concentrations corresponded with Zn increases in the Coeur d'Alene River dilutions and decreases in St. Joe River dilutions.

The most dominant and fluctuating genera present during the fall test period were Melosira, Tabellaria, Mougeotia, and Ulothrix (Figure 10). Melosira was the most abundant algae during August while Tabellaria was dominant from September to November. Densities of Mougeotia and Ulothrix exceeded Melosira in September and October. Synedra and Asterionella counts were highest in August and October.

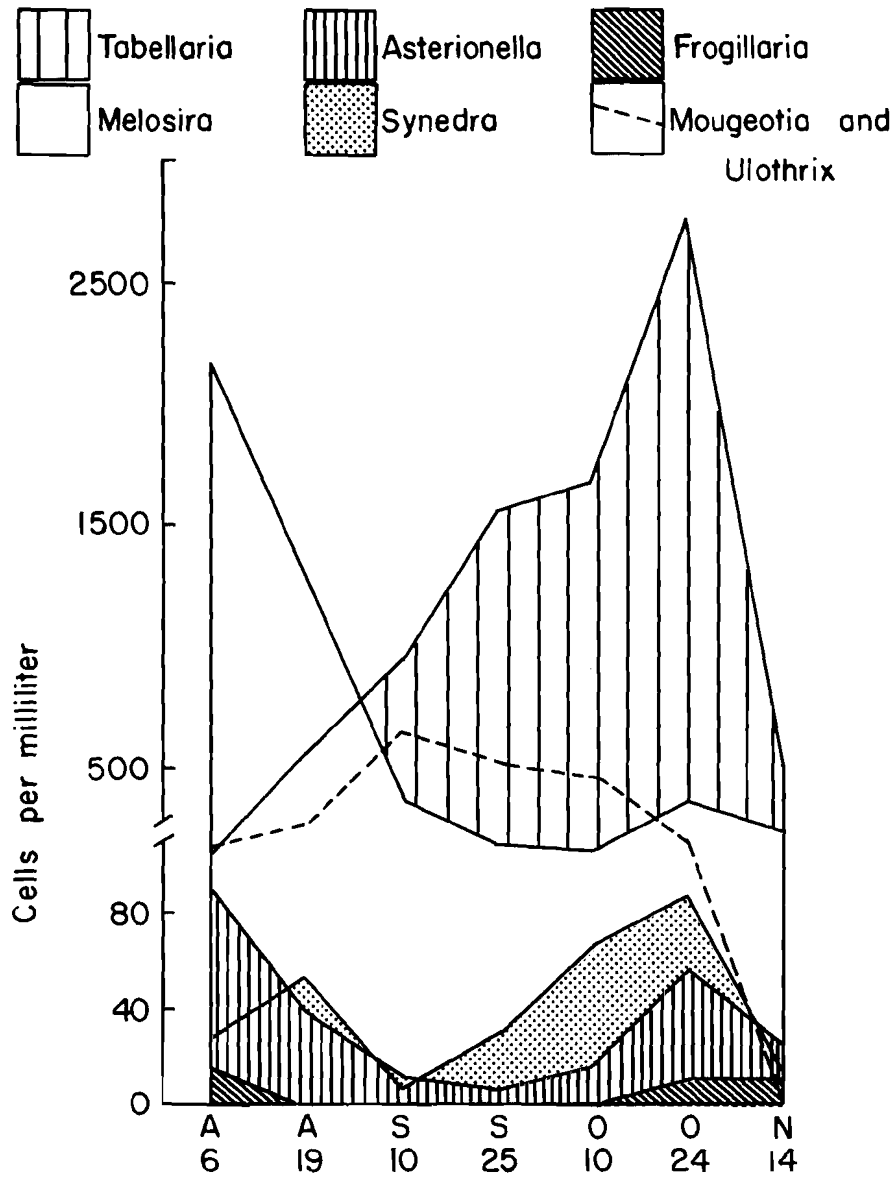

Fig. 10.--Nannoplankton community structure during test period, August to November, 1970

DISCUSSION

Mine and industrial pollution is the main cause of differences in water quality between the Coeur d'Alene River, the Lake, and the St. Joe River. Mine wastes and increased metal concentrations in the river and delta can be related to various milling operations, extraction processes, sulfuric acid and phosphoric acid fertilizer plants. Zinc sulfate and sodium cyanide are used in zinc flotation and processing, soda ash for buffering pH in metallurgy, sodium in leaching associated with electrolytic antimony operations, and calcium sulfate in phosphoric acid processes. The sources of low pH, high conductivity, high levels of Zn, Ca, Na, Mn, and lower quantities of Cd, Cu, Pb, Fe, and F are associated with acidic pond effluents and seepages adjacent to the zinc and phosphoric acid plants. Zinc, F, Cd, Cu, and Pb appear to be the major toxic materials in the South Fork. Maximum levels of 21.0 mg Zn/l (5241 kg/day), 2.3 mg F/l, 0.45 mg Cd/l (109 kg/day), and 0.1 mg/l (25 kg/day) of Cu and Pb respectively have been noted near industrial discharges. The maximum sodium concentrations occur with high conductivity in pond effluents from antimony operations. The metals of Ni, As, Cr, and Hg are present but occurred below reported limits of detectability (Mink et al., 1971; Mink, 1972).

Tailings sedimentation in the Coeur d'Alene River and delta occurred prior to the 1968 construction of settling ponds when all tailings and wastes were discharged directly into the South Fork. It has been estimated that approximately 500,000 tons were discharged to the river annually (Ellis, 1940).

The water of the Coeur d'Alene River and Lake can be characterized as softwater (<60 mg as $CaCO_3$). Concentrations of zinc appear to be the only ion that consistently occurred at high levels. If zinc is excluded, the Coeur d'Alene River (0.1 - 11.2 mg Zn/l) and Lake (<0.1 - 1.0 mg Zn/l) are comparable to other softwater lakes and rivers. Lake Tahoe, California, produced zinc concentrations of <0.014 mg Zn/l (Goldman, 1965) while the Clearwater and Snake rivers of northern Idaho showed 0.04 mg Zn/l (Kopp and Kroner, 1968).

Inorganic carbon concentrations were low for regional waters with the Coeur d'Alene River and Lake being similar but less than the St. Joe River. Levels of inorganic carbon were evidently related to the softwaters of the area and could be indicative of the inert geochemical character of the drainage. Bicarbonate and pH of the Coeur d'Alene River and Lake were lower than the St. Joe River and probably resulted from mine wastes. Evidently various acid seepages and inflows are diluted but not effectively neutralized due to the poor buffering capacity of the river. This could be important to the carbon uptake by the predominately diatom community since HCO_3 ions were apparently the main form of inorganic carbon available. The scarcity of inorganic carbon and the relatively low pH (6.4 to 7.3) in the Coeur d'Alene River and mine-contaminated portions of the delta and lake could be associated with a low solubility for CO_2 and its consequent loss to the atmosphere. The nature of the tailings deposits may also interfere with CO_2 regeneration from bottom sediments.

Most nutrients seemed in ample supply and not limiting to algae growth. Nitrates and phosphates appeared at higher levels in the Coeur d'Alene and St. Joe rivers than in the lake. My pilot studies during the summer of 1969 on silica in the Coeur d'Alene River and Lake indicated concentrations of 3.6 to 7.9 mg SiO_2/l. Sulfate levels of 20 to 50 mg SO_4/l from above the Coeur d'Alene River mouth were reported by Stokes and Ralston (1971).

My regression analysis of lake production indicated positive equations for light extinction and temperature while light penetration was negative (Figure 5). This suggests that as temperature and particulate scattering-absorbance of light increased, algal production decreased. The reverse condition existed for light penetration which probably involved interrelationships of light quality, intensity, and extinction. The significant relationships for NO_3, Fe, Cu, and K were most likely complicated by different nutrient requirements of planktonic algae (Chu, 1942).

Primary production in the Coeur d'Alene River appeared controlled by shallow depths, river discharge, light extinction, temperature, nutrients, and mine drainage. My regression equations suggested that productivity in the river was positively influenced by bicarbonate ions and temperature, and negatively by river discharge, conductivity, and light extinction (Figure 4). Other relationships of lesser importance apparently existed for Zn, Ca, and Na. However, consideration of cations must include their partial dependence on flow rate as suggested by conductivity relationships (Figure 2). Observations of high Zn concentrations, low pH, and bicarbonate, softwater, and unstable tailings deposits imply that mine wastes suppressed algal productivity. Bioassay assessments of Cu, Cd, and Zn interactions indicate that metals in the river water could be acutely and synergistically toxic to phytoplankton.

The occurrence of occasional high production pulses in the Coeur d'Alene River precludes the possibility of algae being chronically suppressed by metals. Such increases in production evidently resulted from upstream nutrient and plankton sources. The lower river drainage has numerous farms and feelots while the upper reaches receive raw sewage from nearly all municipal areas. Inputs of nitrates and phosphates are associated with pond effluents and seepages near zinc and acid plants. Of equal importance to production is the possibility that the river above the delta is subject to nutrient and plankton replenishment from adjoining small side lakes.

The potential for algal productivity in the St. Joe River appeared greater than that of the Coeur d'Alene River. Primary production was facilitated by denser algae populations, deeper photic and trophogenic zones, and less light extinction. Nutrient sources for the St. Joe River were attributed to greater vegetative cover and consequent allochthonous input. Extreme high production values suggested cultural eutrophication from upstream farming, logging, and sewage operations. Phytoplankton production in both the St. Joe and Coeur d'Alene rivers appeared aided by long reaches of slow moving backwaters.

Seasonal responses of primary production to different environmental factors in the Coeur d'Alene River, Lake, and St. Joe River can be explained by changes in depth profiles of carbon assimilation. The significance of production-depth curves is the shape, which can be informative of developing trophic states and influences of such major factors as light, temperature, nutrients, phytoplankton density, and mine wastes.

Three types of production-depth curves were discussed by Findenegg (1964). The first type exhibits a production maximum in the epilimnion which declines in depth with decreasing light intensity (Figure 11). Type I curves were the most frequent production profiles in the Coeur d'Alene River and Lake and prevailed during all seasons except winter. The majority of these profiles appeared light-dependent because of light extinction by suspended particles other than phytoplankton. The key qualities associated with this type of curve were adequate nutrients and dense algal communities. Production ranges for Type I curves included the highest values observed in the river with 185 - 1338 mg C/m^2/day while values observed in the lake were moderately high at 180 - 1080 mg C/m^2/day. Characteristics of Type I curves in both the river and lake tended to be representative of mesotrophic stages in eutrophication.

Findenegg's Type I curves were usually characteristic of eutrophic lakes with dense self-shading phytoplankton, high nutrient concentrations, and high production values. In contrast, Schindler and Holmgren (1971) described opposite conditions for Type I curves in Shield lakes of Canada. These lakes were considered oligotrophic to mesotrophic with production ranging from 179 - 1103 mg C/m^2/day. The principal controlling factor for this type of production curve was apparently low light penetration due to water color. The nutrient poor quality and sparse phytoplankton densities of these lakes were

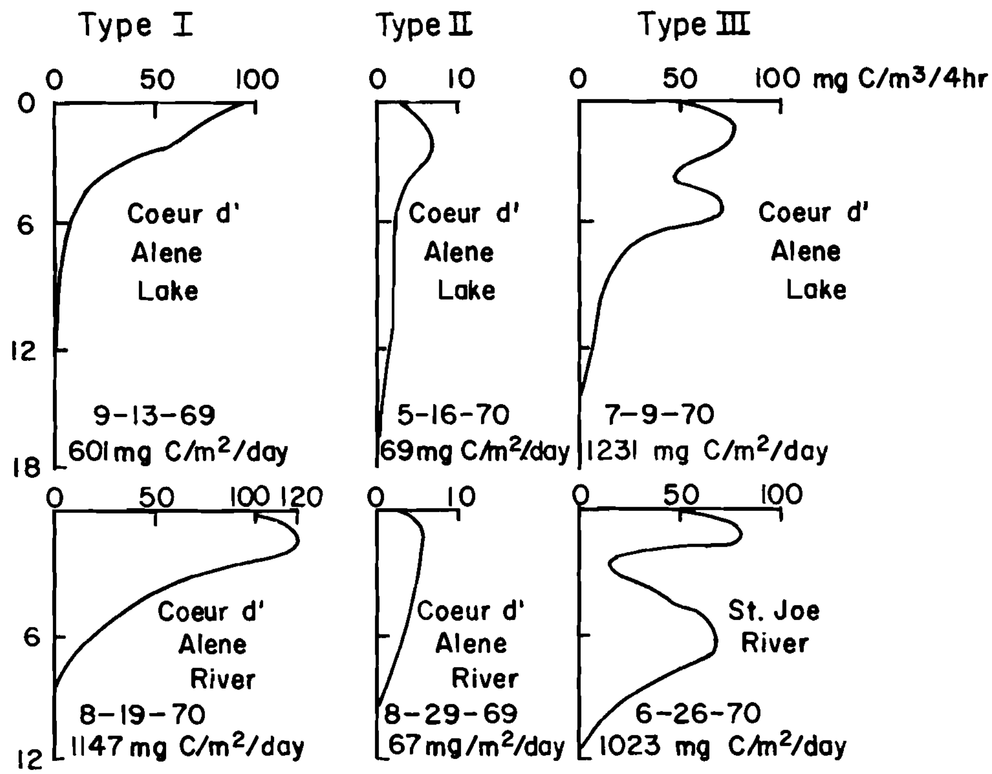

Fig. 11.--Types of production-depth curves observed in the Coeur d'Alene River, Lake, and St. Joe River

considered relatively unimportant in determining the shape of the curve.

The frequent occurrence of Type I curves in the Coeur d'Alene Lake appeared related to nutrient replenishment from the Coeur d'Alene and St. Joe rivers. My observations of relatively high nitrate and phosphate concentrations in the rivers (Tables 1 and 2) and increases in phytoplankton production and densities at southern lake stations (Figure 4) favors such nutrient renewal. Parker (1972), in studying the entire length of Coeur d'Alene Lake, found the highest primary production, algal densities, and concentrations of nitrates and phosphates to the south of the Coeur d'Alene River mouth while opposite conditions existed to the north.

Type I curves appeared during fall turnovers of Coeur d'Alene Lake in 1969 and 1970. These profiles had large epilimnetic production maxima, hence an adequate supply of bottom nutrients. The occurrence of this type of curve in the Coeur d'Alene River usually increased with productivity during low flows in the summer and early fall months (Figure 3).

Type II curves usually remain constant with depth and yield low areal production values (Figure 11). Findenegg attributed this type of curve to oligotrophic waters that were limited by nutrients rather than light. Similar curves were reported in Ontario lakes polluted with mine wastes (Johnson et al., 1969). These waters had low inorganic concentrations and production values of less than 70 mg C/m^2/day.

The production-depth relationship in Coeur d'Alene River and Lake were of Type II in the winter and intermittently during the spring and late summer. Production values for Type II profiles were the lowest in the entire study with ranges of 18 - 172 mg C/m^2/day in the river and 69 - 191 mg C/m^2/day in the lake. Nutrient limitation appeared contributory, but not in all instances. The only times when the scarcity of nutrients seemed obvious was during appearance of Type II curves in the lake following summer algal blooms.

The winter production II curves in the river and lake seemed related to minimal solar radiation and low phytoplankton densities. The appearance of slight metalimnetic production pulses corresponded with observations of dense profundal algae and rich nutrients. Heterotrophic carbon uptake may have been important in maintaining such communities. Carbon-14 uptake in dark bottles was higher than usual and suggested dark fixation of CO_2 by algae or bacteria.

The Coeur d'Alene River and adjacent lake waters produced Type II curves during high flows and low light penetration in the spring months of 1969 and 1970. Nutrients did not appear limiting except for low concentrations of inorganic carbon. Low production appeared related to short retention times of plankton in the river, delta, and lake. Accordingly, my regression analysis of production and river discharge indicated that as the flushing rate increased, phytoplankton production decreased (Table 4). The flow of water at lake stations was evident with the direction of currents being toward the Spokane River outfall.

Type II curves of the Coeur d'Alene River were observed during low production periods of the late summer. These profiles could have reflected inhibition by mine wastes and metal synergism but most likely relate to changes in nutrient levels and algal composition.

Type III curves exhibited two production maxima, one in the epilimnion and one in the metalimnion (Figure 11). The Coeur d'Alene Lake incubations indicated that this type of curve occurred intermittently in the spring and summer of 1970. Summer production values of 1231 and 1715 mg C/m^2/day were the highest in the study except for estimates in the St. Joe River at 1023 and 5943 mg C/m^2/day. The high production of the St. Joe River and the persistence of slight metalimnetic maximums throughout the summer suggested the presence of nutrient reserves and shade-cold temperature adapted algae.

During the spring, the majority of the Type III production in the lake was in the epilimnion, suggesting an ample supply of nutrients from spring circulation. The epilimnetic production persisted during the summer as the metalimnetic maximum increased. The upper production maxima usually occurred above poorly defined thermoclines and hypolimna without pronounced oxygen deficits. This suggests that primary production of the summer epilimnion was maintained not only by adequate light conditions but by frequent winds that circulated nutrient-laden metalimnetic waters through the epilimnion. In addition, observations of large individual cells of Melosira and Tabellaria indicated that production was enhanced by phytoplankton suspension through wind action.

The Type III curves reported by Findenegg usually occurred in mesotrophic lakes recently exposed to cultural eutrophication. These waters had metalimnetic maxima which developed during the summer as nutrients were impoverished in the epilimnion. Thermal stratification and oxygen stagnation tended to trap nutrients which stimulated metalimnetic production.

My observations on production profiles and regression equations give some indication of the environmental effects of mine pollution. Interrelationships of these effects and masking by light and temperature appeared to confound the effects of various metals on algal growth. The appearance of low R^2 values in most of the regression analyses could be indicative of such conditions. These problems, which are inherent in any investigation involving discrete field data, were partially dealt with by the use of bioassays. Metallic poisoning of algae in carbon-14 assays in which I controlled light and temperature supplies strong evidence of toxicity by interacting metals.

The bioassays indicated that copper was the most toxic metal tested. The largest mean differences in carbon fixation were due to copper concentrations of 0.10 and 0.30 mc Cu/l. Copper concentrations of 0.75 mg Cu/l were only slightly more inhibitory than 0.30 mg Cu/l. The toxicity of copper sulfate in these tests can be attributed to its success as an algicide under conditions of both soft- and hardwater. Although copper quantities in the Coeur d'Alene River compare closely to those in non-polluted waters, the relatively softwater of the river may enhance toxic effects of copper on phytoplankton. Most algae in softwaters can be controlled with concentrations of copper sulfate between 0.12 and 0.50 mg Cu/l while hardwater requires 1.20 to 2.00 mg Cu/l (Moyle, 1949).

Differences between my tests indicate that the toxicity of copper to natural phytoplankton could vary continually with the population structure of the community. Support for this supposition can be taken from the different concentrations recommended for control of specific algal populations (McKee and Wolf, 1963; Jordan et al., 1962). Mandelli (1969) studied the inhibitory effects of Cu on marine phytoplankton and found that the growth of dinoflagellates and diatoms was reduced at 20 C in media containing 0.055 and 0.265 mg Cu/l, respectively. He studied a total of nine species and concluded that Cu had a selective toxicity rather than a general effect at concentrations between 0.03 and 0.50 mg Cu/l. Maloney and Palmer (1956) reported similar conclusions for the toxicity of copper sulfate to 30 freshwater algal cultures at concentrations between 0.05 and 0.50 mg Cu/l. I believe that different concentrations of copper as an algicide and its selective toxicity suggest that similar conditions exist for Cd and Zn. In addition, selective toxicity may be indicative of algae adapting to varying metal concentrations. Whitton (1970) reported that *Stigeoclonium* from a stream polluted by metals had some resistance to zinc.

Cadmium in my tests inhibited carbon fixation at 0.1 and 0.3 mg Cd/l. Bringmann and Kühn (1959), in studying *Scenedesmus* from a river habitat, found threshold concentrations for detrimental effects of Cd as chloride at 0.1 mg Cd/l. McKee and Wolf (1963) reported that cadmium acts synergistically with other substances to increase toxicity.

Zinc decreased phytoplankton carbon uptake at 0.1 and 0.5 mg Zn/l. In general, the effect of zinc decreased as the concentration of zinc increased. At 1.0 mg Zn/l the amount of carbon fixed was slightly higher than in controls. These results can be explained partly by information from Bachmann (1963). He suggested that algae cells take up Zn as a direct response to concentration and at levels exceeding those ordinarily found in nature, which indicates a "luxury" consumption above immediate metabolic needs.

The metal interactions that were most inhibitory to carbon fixation occurred within the following concentrations: 0.1 - 0.3 mg Cu/l, 0.1 - 0.3 mg Cd/l, and 0.3 - 0.5 mg Zn/l. The tests indicate that all interactions of Cu x Zn, Cu x Cd, Cd x Zn, and Cu x Cd x Zn acted synergistically to produce toxic effects. The overriding effect of Cu and the two-way Cu x Zn interaction appear to be the principal toxic components in these treatments. Fitzgerald (1971) found that 0.05 to 0.075 mg Cu/l as copper sulfate was required to prevent the growth of *Microcystis* regardless of the addition of zinc at Cu:Zn ratios of 1:1, 1:2, 1:4, and 1:6. Likewise, the growth of *Chlorella* was prevented at 1.0 mg Cu/l as copper sulfate whether 0.0, 0.5, or 2.0 mg Zn/l as zinc sulfate were present. Although Fitzgerald did not detect synergism between Cu and Zn the possibility that softwater increases this toxic action was still in question.

The high concentrations of Zn and Cu and the softwater in my tests possibly explain the observed synergism. Although the levels of Zn and Cu were relatively high, they were within the range of concentrations observed in the Coeur d'Alene River and suggest that a similar action could occur. Concentrations used in these tests indicated that each metal contributed to the toxicity of the mixture in proportion to their individual toxicities. This suggests that the inhibitory effects of the interactions could be estimated by summation of the fractional toxicities of each metal.

Differences in response surfaces for the tests of river dilution could be related to other metals in the test waters. The occurrence of decreased inhibitory effects of Zn and Cu to carbon fixation corresponded with increases in concentrations of Ca, Mg, Na, K, and Mn. Bachmann (1963) found that Zn^{65} uptake by algae was decreased as the concentrations of other cations in solution were increased. The order of effectiveness of ions in reducing zinc uptake by the green alga Golenkinia was H > Ca > Mg > Na > K. The accumulation of these metals by suspended living and dead material and bottom sediments was characterized by ion exchange or non-specific adsorption. These reactions were attributed to competition between ions for available exchange sites.

Somewhat similar observations have been reported for K (Nielsen et al., 1969) and Mn reduction of copper inhibition. In Mn antagonisms of Cu, the action was after cellular uptake. The Mn responses were obtained from reactions in large chloroplasts of pokeweed (Phytolacca americana)(Habermann, 1969). Additional experiments by Habermann with chloroplast reactions indicated that Zn and Mg were slightly stimulating but not as effective as Mn in reversing copper inhibition. The cation Hg was found to be a more potent inhibitor than Cu. The mode of these reactions was considered as competition for the same site on an enzyme. These observations and the occurrence of relatively high Mn concentrations could explain the reduction of copper inhibition in some of my dilution tests. In general, cation concentrations of the Coeur d'Alene River were moderately high as compared to other softwaters and could reverse Zn, Cu, and Cd inhibition of algal carbon uptake.

The amount of a metal available to planktonic algae can depend upon the presence of soluble organic and inorganic complexing agents. Investigators have found that different chemical forms of a metal do not necessarily have the same uptake rates. Bernard and Zattera (1967) reported that when Zn^{65} was added as ionic Zn proportionately more Zn^{65} was taken up by algae than stable Zn. However, when Zn^{65}-EDTA-complex was added to algae containing seawater and a chelating resin, stable Zn was initially taken up at a higher rate than Zn^{65}.

Other studies have suggested that natural waters contain substances that complex metal ions. Nielsen and Winn-Anderson (1970) found ionic Cu at concentrations occurring in natural waters (1 to 2μg/l) was poisonous to photosynthesis and growth of unicellular algae. Nielsen suggested that this indicated that Cu is not ordinarily present in the ionic form but is complexed by organic matter such as polypeptides. He found that complexed Cu was not toxic to algae, but it was not possible to distinguish between such chelates as Cu-EDTA-complexes and natural organo-copper complexes.

Extracellular products (polypeptides) of freshwater algae and organic decay have been found to form complexes with ions of Cu, Zn, Fe, phosphate, and certain organic substances. Fogg and Westlake (1955) noted that such complex formation may be important in reducing Cu toxicity to some Cyanophyceae, Chlorophyceae, Xanthophyceae, and

Bacillariophyceae. Examples of inorganic complexing would be binding of Cu by colloidal $Fe(OH)_3$ (Nielsen and Winn-Anderson, 1970) and Zn as $ZnCO_3$ and $Zn(OH)_2$ in seawater (Alberto and Healty, 1970).

The presence of complexing agents in the Coeur d'Alene River could suggest that current levels of Cu, Cd, and Zn are less toxic to algae than my tests indicated. The decrease in inhibition with increased volumes of river water implies binding by pollutants or natural materials. The principal arguments against the importance of chelation in the river and delta are the levels of major cations, the high Zn concentrations, and evidence of metal toxicity by synergism.

The impact of metals on aquatic life in the Coeur d'Alene drainage cannot be adequately dealt with until we have knowledge of the contribution of tailings deposits to metal concentrations in solution. Prior to about 1930 stamp-mill-jig-table processes put coarse gravity tailings into the river. The tailings could not be economically smelted and therefore carried high concentrations of silver, lead, and zinc. The presence of these metallic sulfides in river sediments constitutes a pollution hazard since physical disturbances and oxidation to sulfates may increase the toxicity of these elements. The high solubility of sulfate can produce sulfuric acid, lower the pH, and release metals into solution. Accordingly, tailings deposits could affect the biota in the river valley and lake for many years.

Research on the effect of tailings deposits on metals in solution suggests that a pollution problem does exist. Studies of mine tailings indicate that the cation exchange capacities were low as compared to most soils (Toukan, 1971). Tailings had their highest exchange capacity at pH 10. Equilibrium equations at various pH values indicated that Zn was released from tailings to increase the concentration of Zn in settling pond effluents. These results suggest that tailings deposits in contact with water of pH 7.0 and lower in the Coeur d'Alene River and delta region would release Zn into solution.

Sediment studies in the Coeur d'Alene River delta have revealed the magnitude of metal concentrations in tailings deposits (Maxfield,[2] unpublished data). The mean Zn concentration for approximately 45 cm cores was 4207 mg/kg. The average levels of Mn, Pb, Sb, Cu, Cd, and Ag were 211 percent, 117 percent, 7 percent, 3 percent, 1 percent, and 0.3 percent, respectively, of that of zinc. A mercury concentration of 0.4 percent of the preceding zinc value was reported by Sceva[3] (unpublished data). The amount of metals associated with core partitions took the order of clay > organic matter > sand > silt. Metal concentrations in delta sediments usually increased with distance from the river mouth, indicating transport by fine suspended particles. equilibrium equations at pH 7.8 and 2.5 indicated zinc concentrations released at 1.8 and 22.0 mg Zn/l, respectively.

Based on the information gathered through this investigation, the following recommendations are presented for future management and research.

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[2]D. Maxfield, Biological, chemical, and geological study of the Coeur d'Alene River delta. Student originated studies, NSF, University of Idaho, Moscow, Idaho, 1972.

[3]Jack E. Sceva, Geologist, FWPCA, Room 501, Pittock Block, Portland, Oregon 97205. Mercury in sediment samples from the Coeur d'Alene River Basin, Idaho, April 7, 1971.

The effect of Coeur d'Alene River water and metal ions on primary production should be monitored at low flow periods in the fall by acute bioassays developed in this investigation.

Chronic algal bioassays should be conducted so that application factors can be developed which would indicate safe concentrations of metals and wastes in the receiving waters (Report of the Committee on Water Quality Criteria, 1968). The objective would be to establish maximal allowable values below which exist a range of concentrations that do not inhibit production. Recommended values should be adequate where more than one adverse factor exists (Zn, Cu, and Cd) and be within suitable limits of pH, hardness, and temperature.

Additional experiments should be conducted to explore the possibility that some river constituents antagonize heavy metal toxicity and to continue bioassays of the synergistic effects of metals in sediment suspensions and river water. Simultaneous bioassays should be conducted on natural algal communities, isolated populations, and recommended indicator organisms. Such tests would define the effects of metal interactions and sensitivities of community components. These studies should be expanded to endemic and harvestable fish of the area. Fish are usually more susceptible to metal toxicity than algae and could provide greater sensitivity in multivariate bioassays.

The trophic status of regional waters should be continuously surveyed by production-depth profiles and accompanying physiochemical data. Additional information on nutrient, metal, and algal biomass-density depth regimes would aid such a program.

CONCLUSIONS

1. Algal bioassays of heavy metals provide acute tests that have possible utility in management of water resources. Factorial designs, carbon-14 uptake, and simple restrictions placed upon algae communities make it possible to screen the simultaneous impact of several pollutants on autotrophic production in aquatic systems.

2. Zinc, copper, and cadmium were acutely and synergistically toxic to carbon uptake by phytoplankton.

3. Coeur d'Alene River dilutions and treatments of Zn and Cu indicate that increased volumes of river water decrease metal toxicity. Variable algal community structure and the presence of major cations, softwater, low pH values, and bicarbonate concentrations and possibly chelation appeared to affect metal toxicity.

4. Low inorganic carbon and moderately high nitrate and phosphate levels in the Coeur d'Alene River, Delta, Lake, and St. Joe River suggest the importance of carbon rather than other nutrients in regulating growth of autotrophic populations.

5. Primary production in the southern end of Coeur d'Alene Lake is evidently under the control of discharges from the Coeur d'Alene and St. Joe rivers. Changes in water quality of the lake appear to follow seasonal fluctuations in the rivers.

6. The effect of wind action on epilimnetic production and the lack of balance in stratification appear related to the decreased depth of the southern end of the Coeur d'Alene Lake. Mixing of lake

waters may be influential in deaccelerating eutrophication by suppressing hypolimnetic oxygen depletion and anaerobic nutrient regeneration.

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