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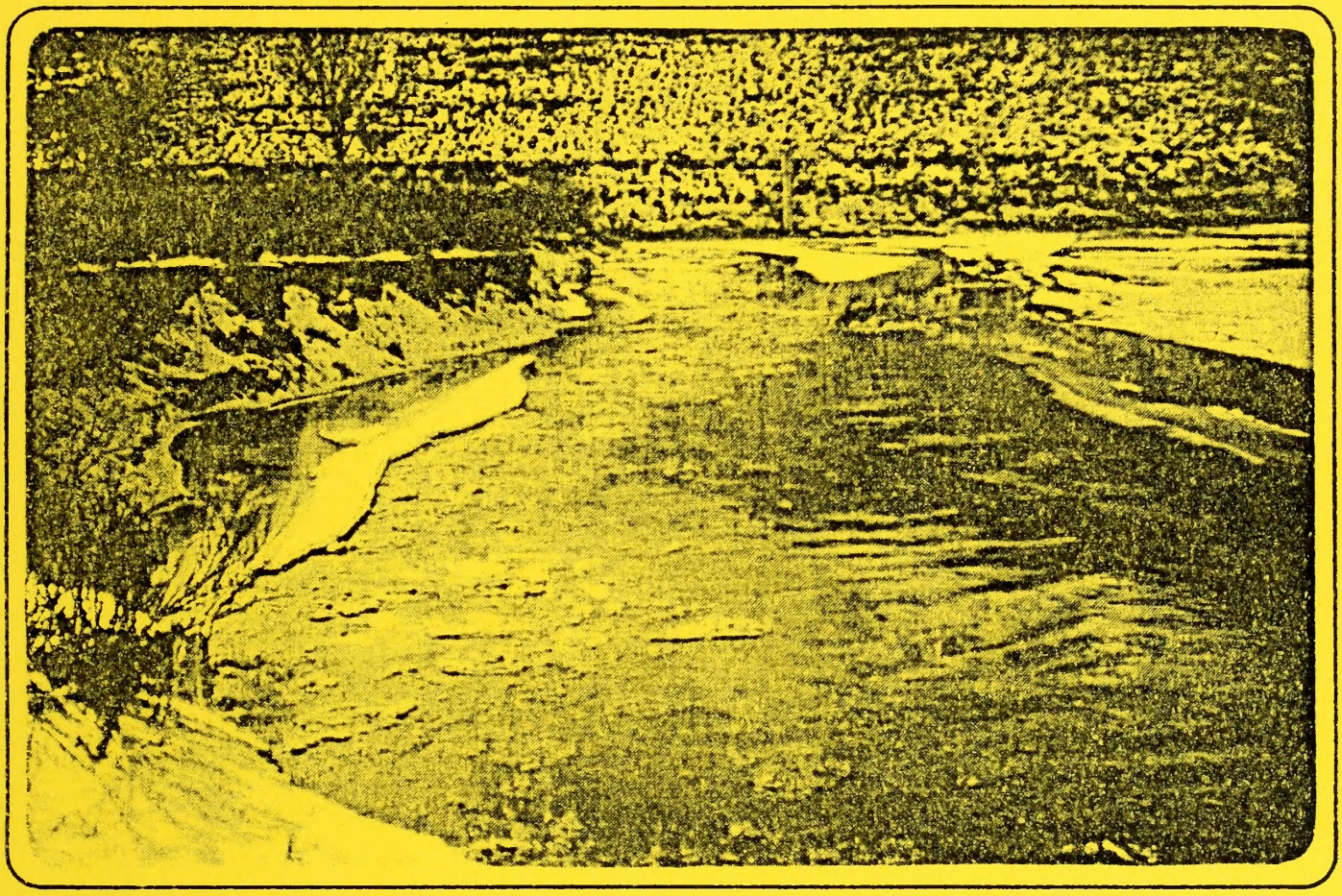
FINAL REPORT

1981 AQUATIC RESOURCES MONITORING PROGRAM



PREPARED FOR

WHITE RIVER SHALE OIL CORP.

BY

ECOSYSTEM RESEARCH INSTITUTE

APRIL 1982

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INTRODUCTION

The White River, located in the upper Colorado River basin, is a tributary to the Green River. This river has its headwaters in western Colorado (elevation 2200 M) and flows westerly through the Piceance and Uintah Basins, reaching its confluence with the Green River at Ouray, Utah (elevation 1416 M).

The White River drainage basin above federal lease tracts Ua-Ub is approximately 10,400 km^2 and contains seven major tributaries. Several of these tributaries enter the White River near Ua and Ub; however, only Evacuation Creek could be classified as perennial with the other drainages flowing only during snowmelt or periods of intense rainfall.

The White River flows near a large number of planned oil shale and tar sands developments within these two basins. Not only is the river an important source of water for domestic supply and irrigation, but it will be a critical supply link for proposed energy developments. In addition, the White River ecosystem could be altered by proposed impoundments, water withdrawals, construction activity, or by air- or water-borne pollutants related to these developments.

The design of the White River Shale Oil Corporation Aquatic Monitoring Program must then be flexible in order to address the changes, both natural and man- related, which are expected to occur within the river ecosystem. In order to be effective, this program must also be able to identify these changes while recognizing that the White River system adjacent to Tracts Ua and Ub is but a small segment of a larger stream continuum and be able to place this river segment in proper perspective within this continuum. This is important in understanding and identifying changes within the river biocenosis which are the result of upstream influences outside the control of WRSOC. Sample sites upstream of Tracts Ua and Ub can be used to identify these incoming influences, while those sites at or below the tracts can be used to separate potential WRSOC influences.

Monitoring parameters were selected for their ability to most efficiently depict the structures and functions occurring in the stream ecosystem. Their selection also recognized that natural systems have a high degree of variability and that the biota exists within constraints set by the physical environment. In the development of this program, recognition was made of ecosystem theory. That is, producers, consumers, and decomposers constitute the biota which are regulated by the physical environment and their own interactions. These interactions constitute a series of interrelationships

which can be quantified and allow not only the detection of change, but identification of the pathway of cause and effect leading to and resulting from the change. These relationships can be used in addition to the more conventional statistical comparisons of "state variables," and therefore, strengthen the ability of the program to quantify change.

The objectives of the 1981 Aquatic Monitoring Program were fivefold:

(1) To characterize the White River and Evacuation Creek habitats in order to develop the most appropriate sampling sites for long-term monitoring.

(2) Gather ecosystem-level data which would be important in tracing the fate and consequences of introduced foreign substances.

(3) To develop appropriate techniques which can be utilized efficiently in a long-term monitoring program.

(4) Determine those ecosystem parameters which have the greatest importance for long-term monitoring and the assessment of impacts.

(5) Develop statistically significant interrelationships between aquatic and other ecosystems.

Objective (1), the selection of sample sites, was addressed during an initial field investigation during April, 1981. Twenty-nine cross- stream transects were surveyed on the White River and nine on Evacuation Creek. The subsequent statistical analysis led to final site selection. The remaining objectives were addressed during monthly field investigations from May through December.

The purpose of this report is threefold: (1) To present a description of the aquatic ecosystem of the White River and Evacuation Creek, (2) to examine the usefulness of sites and parameters selected to identify change, and (3) to make recommendations for changes to the on-going monitoring program.

This report is organized into three major sections (results, discussion, and recommendations) plus an appendix. The materials and methods and an extensive literature review are presented in the appendix. This will allow the main body of the report to be read with greater clarity and continuity.

RESULTS

Initial Field Investigation

An initial field investigation was undertaken on the White River and Evacuation Creek during April, 1981. Based upon the results of Lyle (1981), sample sites were selected at 3.2 km intervals from Hell's Hole Canyon to Asphalt Wash on the White River (Figure 1). At each of these locations three transects located 200 meters apart were sampled. Three sample sites consisting of three transects spaced at 100 meters apart were selected on Evacuation Creek below the Ignatio Road Bridge. The following parameters were measured at each transect:

(1) Stream profiles were determined relative to an established, permanent bench mark. River depth was measured at two-meter intervals across the stream channel.

(2) Depth-velocity profiles were determined at four-meter increments across the stream channel. Vertical velocities were measured at 15 cm depth increments.

(3) At each two-meter increment, dominant substrates were determined (see Appendix I for substrate classifications).

(4) Interstitial sediment samples were collected at five equal intervals across the river and at three locations on transects in Evacuation Creek.

(5) Periphyton biomass estimates were determined at five equal intervals across the river and at three locations on transects in Evacuation Creek.

The above data were collected on 29 transects in the White River and nine transects in Evacuation Creek. Because of the low flows during April, 1981 (Figure 2), a complete survey of all transects was accomplished. Following laboratory analysis, all data were subjected to Clustar Analysis to determine how many different major habitat types were present in the White River and Evacuation Creek. The results of these analyses can be seen in Figures 3 and 4.

In the White River, two distinct habitat types were determined. A one-way analysis of variance (ANOVA) was used to verify these results (Table 1). In general, shallow, fast-flowing areas with large stony substrate and high chlorophyll *a* habitats were defined as "riffles", whereas deeper,

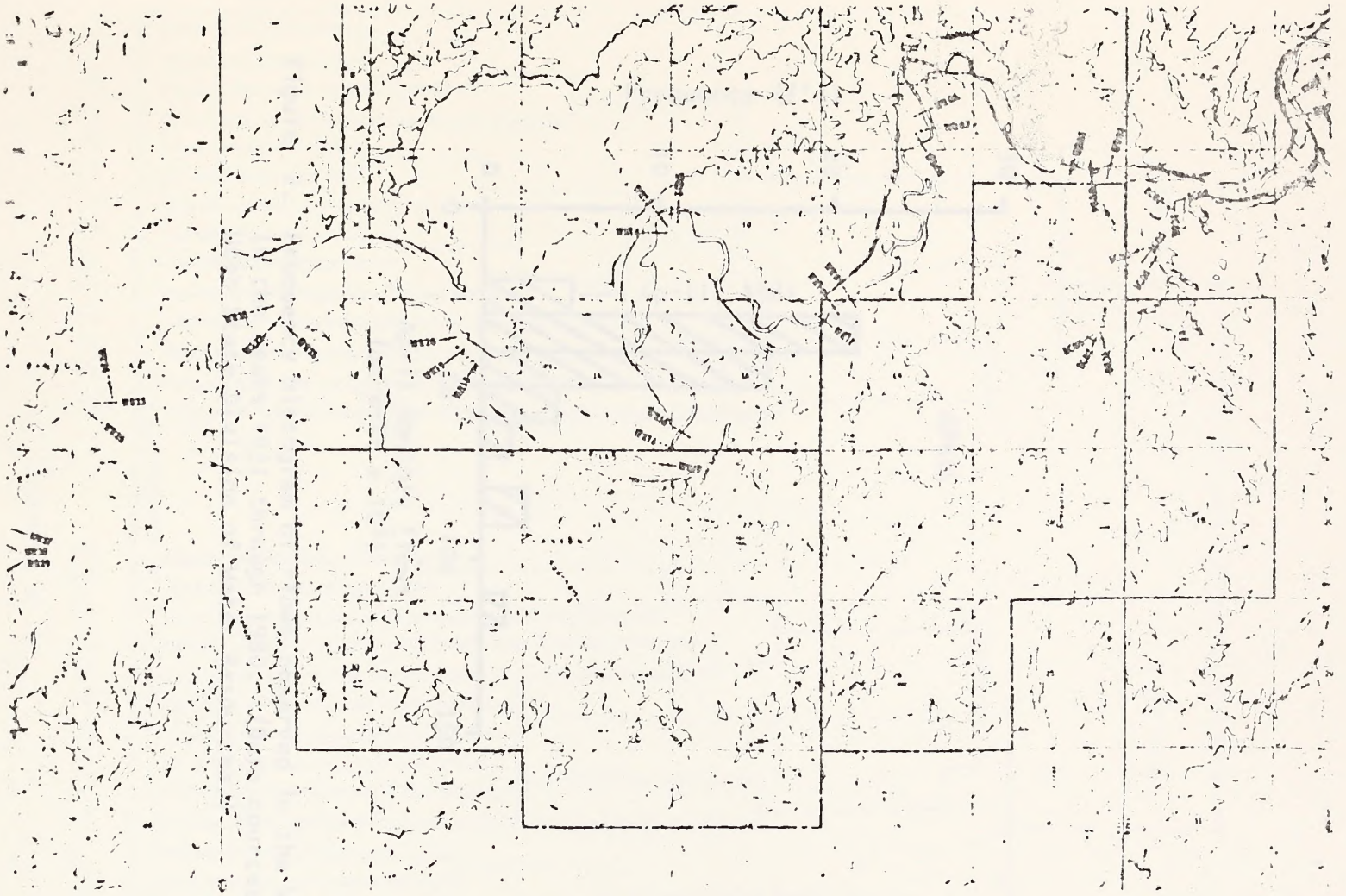


Figure 1. Map showing locations of all transects used in the initial and monthly field investigations.

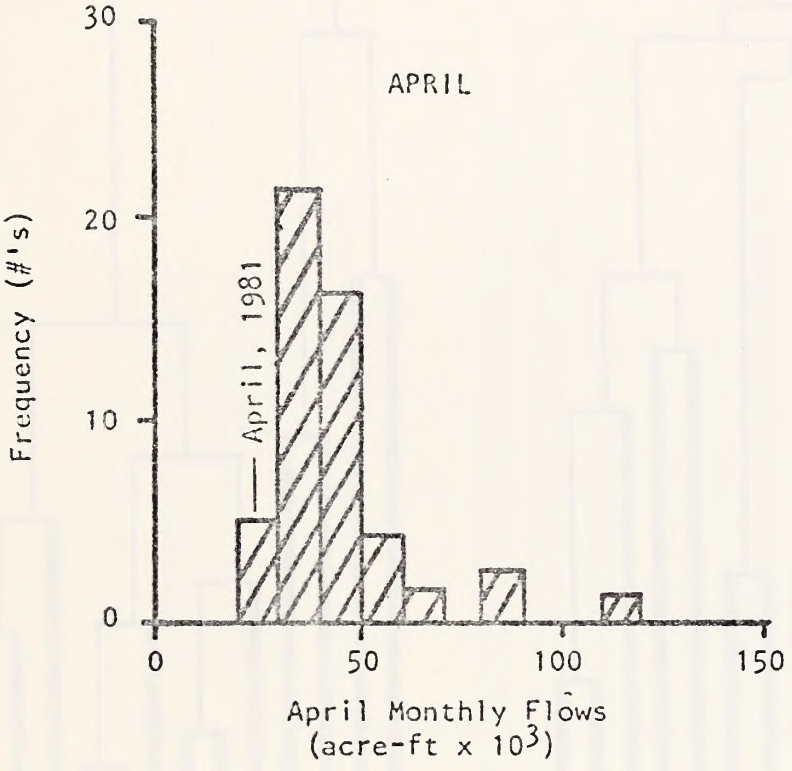


Figure 2. Frequency histogram of flows observed in the White River in the years 1931 through 1980. (Data courtesy of Utah State Division of Water Resources.)

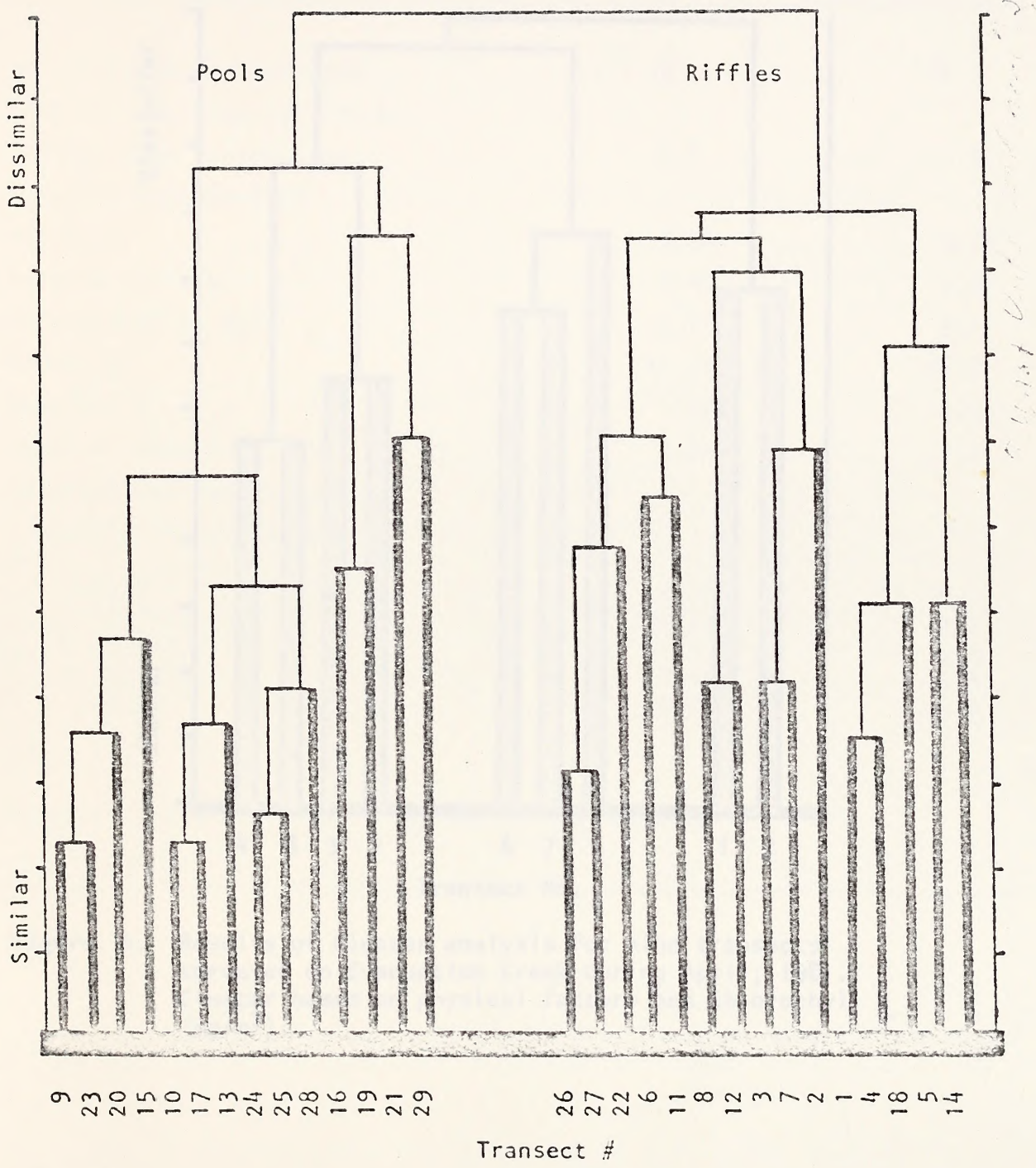


Figure 3. Results of a Clustar analysis for twenty-nine transects on the White River. Clustar based upon physical factors and Chlorophyll a data taken during April, 1981.

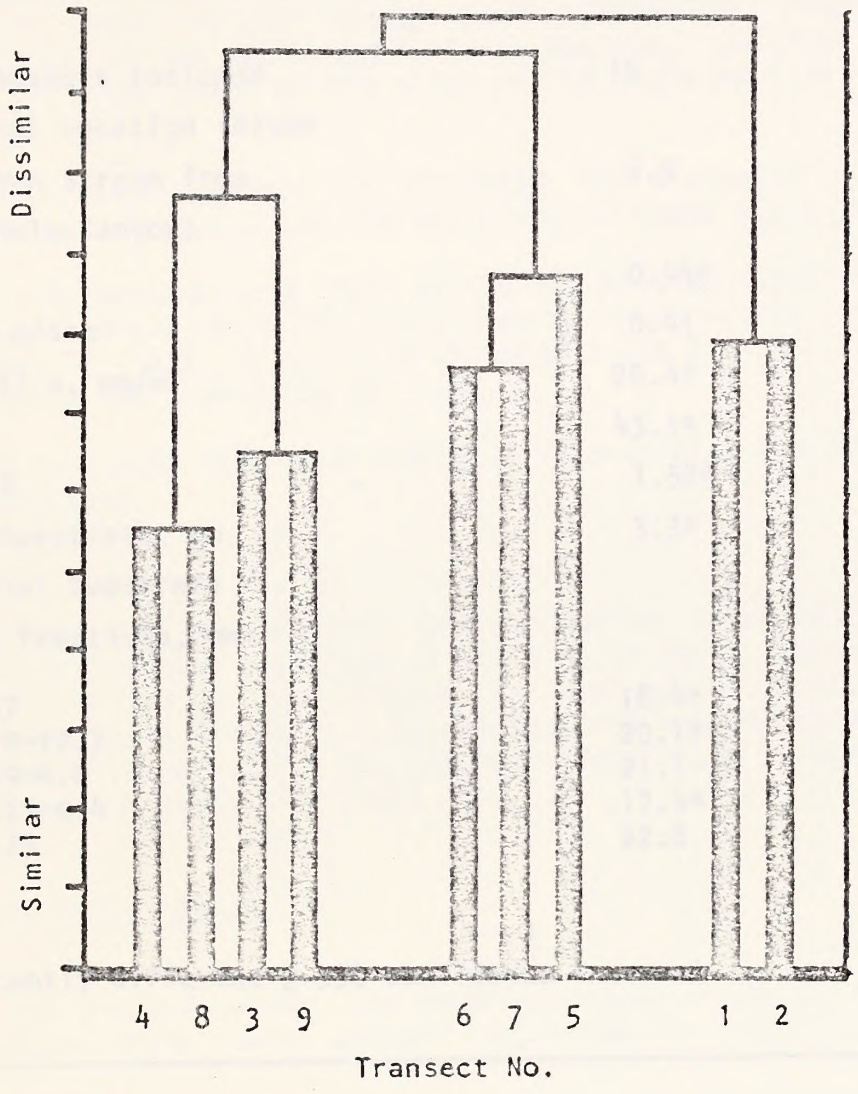


Figure 4. Results of Clustar analysis for nine transects surveyed on Evacuation Creek during April, 1981. Clustar based on physical factors and Chlorophyll a (mg/m^2).

Table 1. Mean of physical and biological parameters for riffles and pool/run type transects on the White River, Utah.

Parameters	Riffle	Pool/Run
No. of Transects Included	15	14
Mean Habitat Location (River Miles Down Stream from Hell's Hole Canyon)	6.5	11.2
Depth, m	0.44*	0.68
Velocity, m/sec	0.41	0.39
Chlorophyll a, mg/m^2	20.4*	9.1
Width, m	43.1*	28.8
Gradient %	1.58*	0.64
Dominant Substrate, cm	3.3*	2.1
Interstitial Substrate by Size Fractions, mm		
> 12.7	18.4*	12.2
4.0-12.7	20.1*	12.7
0.5-4.0	21.1	19.0
0.25-0.4	17.4*	30.6
< 0.25	22.8	24.8

*Significantly different @ 95% confidence interval

fast-flowing areas with armored clay or sand bottoms and low algal standing crops were considered pools or runs.

Over the total 17 miles sampled, the number of riffle and pool/run habitats were found to be nearly equal (Table 1); however, more riffle habitats occurred in the upper eight miles, whereas, pool/run habitats were dominant in the lower eight to ten miles of the study section.

Once statistical analysis was completed, six transects were chosen for monthly sampling. Criteria for the selection of these sites were:

(1) A representative of each of the two habitat types should be selected above the Ua-Ub federal lease tracts (River Km 102 - 93), adjacent to the tracts (Km 93 - 84), and below the tracts (Km 84 - 76).

(2) Access must have been available to economize and speed sampling.

The selection of sample sites in Evacuation Creek followed the same procedure used in the selection of White River sites. Clustar analysis was used to determine habitat type (Figure 4) and was followed by a one-way analysis of variance (Table 2). Representatives were selected from the major habitat types and were used in the intensive monitoring program. The final site selections can be seen in Table 3.

Monthly Field Investigation

After the initial survey, the selected sample sites on the White River and Evacuation Creek were sampled approximately monthly between May and December.

Physical Factors: The White River and Evacuation Creek are typical of other desert riverine systems. These systems are impacted by upper and lower basin runoff and periodic storm events. These data can be seen in Figure 5 for the White River and Figure 6 for Evacuation Creek.

Lower basin runoff occurred in May with a peak runoff of 1000 cfs. Upper basin runoff (1400 cfs) occurred at the end of June and was substantially less than the lower basin flows in 1979 and 1980. Base flow (July through August)

Table 2. Mean of physical and biological parameters for three groups of transects on Evacuation Creek. Data from initial field investigation which occurred in mid April, 1981.

	Transects		
	EC01, 02	EC05, 06, 07	EC03, 04, 08, 09
Depth (m)	0.047	0.060	0.070
Velocity (m/s)	0.38	0.24	0.32
Width (m)	1.9*	4.1*	2.9*
Gradient (m/m)	0.0072	0.0067	0.0056*
Dominant Substrate (cm)	3.83*	0.42	0.49
Interstitial Sediment by Size Fractions (mm)			
> 12.7	16.3	17.0	14.5
4.0 - 12.7	33.6	24.4	28.1
0.5 - 4.0	31.8	35.0	36.9
0.25 - 0.5	11.5	12.8	12.4
< 0.25	6.8	10.7	8.2
Chlorophyll a (mg/m^2)	5.6	3.9	6.5

* Group parameter means different at $p < 0.05$ using ANOVA.

Table 3. The sites selected for continued sampling in the White River and Evacuation Creek, 1981-82.

Location	Distance from Confluence	ERI Transect #
White River		
Above tract	KM 99	3 (riffle) 5 (pool)
Adjacent to tract	KM 84	18 (riffle) 20 (pool)
Below tract	KM 76	27 (riffle) 29 (pool)
Evacuation Creek		
On tracts	-	2 (riffle) 3 (run)

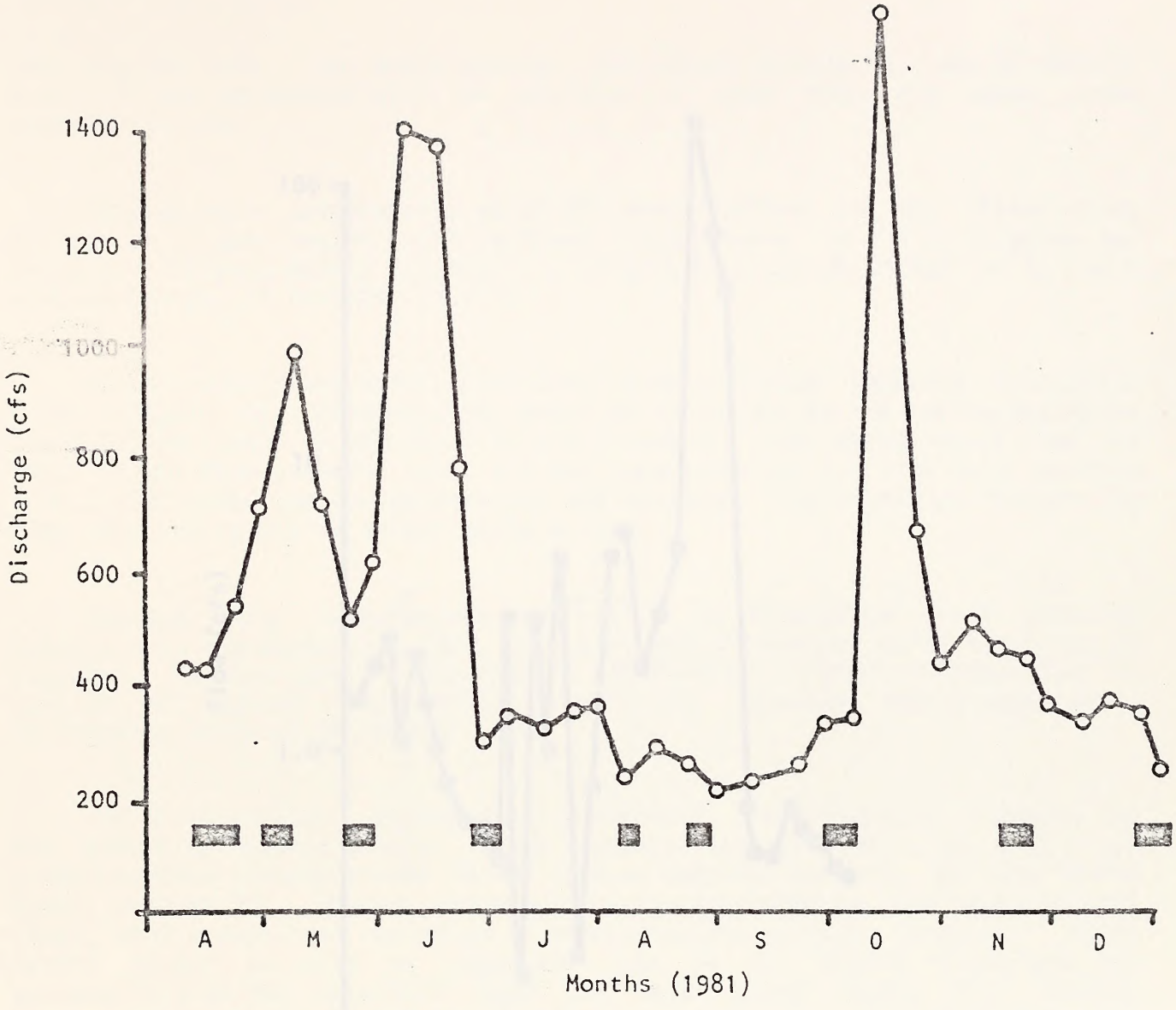


Figure 5. Discharge in the White River during 1981.

■ Indicates sample period for aquatic biology monitoring.

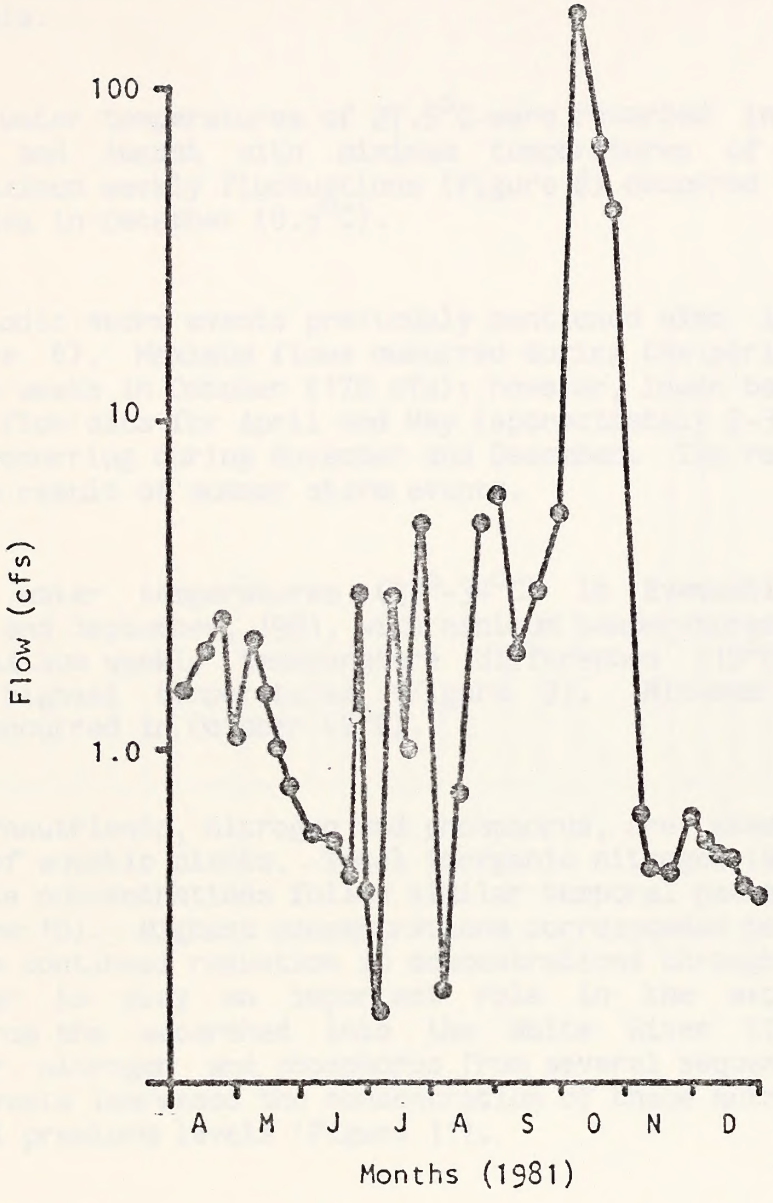


Figure 6. Discharge for Evacuation Creek during 1981.

was 250-300 cfs. In early October, 3.6 inches of rain (9.1 cms at ARA-13; Figure 7) was associated with an increase of over 1400 cfs above these baseflow levels.

Maximum water temperatures of 27.5°C were recorded in the White River during July and August with minimum temperatures of 0°C in November and December. Maximum weekly fluctuations (Figure 8) occurred in May (13°C) with minimum changes in December (0.5°C).

The periodic storm events previously mentioned also impacted Evacuation Creek (Figure 6). Maximum flows occurred during the period corresponding to the first two weeks in October (175 cfs); however, lower basin runoff can be seen in the flow data for April and May (approximately 2-3 cfs) with baseflow (.5-.8 cfs) occurring during November and December. The remaining changes in flow were the result of summer storm events.

Maximum water temperatures (33°-34°C) in Evacuation Creek occurred between July and September, 1981, with minimum temperatures (0°C) occurring in December. Maximum weekly temmperature differences (15°C) corresponded to periods of highest temperatures (Figure 9). Minimum weekly temperature differences occurred in October (7°C).

The micronutrients, nitrogen and phosphorus, are essential elements in the growth of aquatic plants. Total inorganic nitrogen ($NO_2 + NO_3 + NH_3$) and orthophosphate concentrations follow similar temporal patterns in the White River (Figure 10). Highest concentrations corresponded to lower basin runoff (May), with a continual reduction in concentrations through December. Storm events appear to play an important role in the export of nitrogen and phosphorus from the watershed into the White River (Table 4). Samples analyzed for nitrogen and phosphorus from several sequential days indicated that storm events increased the concentration of these nutrients 10-15 times their initial prestorm levels (Figure 11).

Light attenuation was determined for each sample period by calculating the coefficient of attenuation from the relationship $\ln(Ix/Io) = e^{-kx}$, where x is depth and I_x and I_o are light intensities at the depth x and the surface. The slope of the linear equation "k" was defined as the coefficient of attenuation (Figure 12). K was found to be related to the amount of total suspended solids (Figure 13) in the White River during the time light profiles were determined ($r^2 = .85$). Because of the two relationships noted above, the depth to which one percent of the surface light will penetrate (based upon TSS levels; Figure 14) can be calculated. These calculated light levels indicate the depth within the river where primary production can occur (Figure 15).

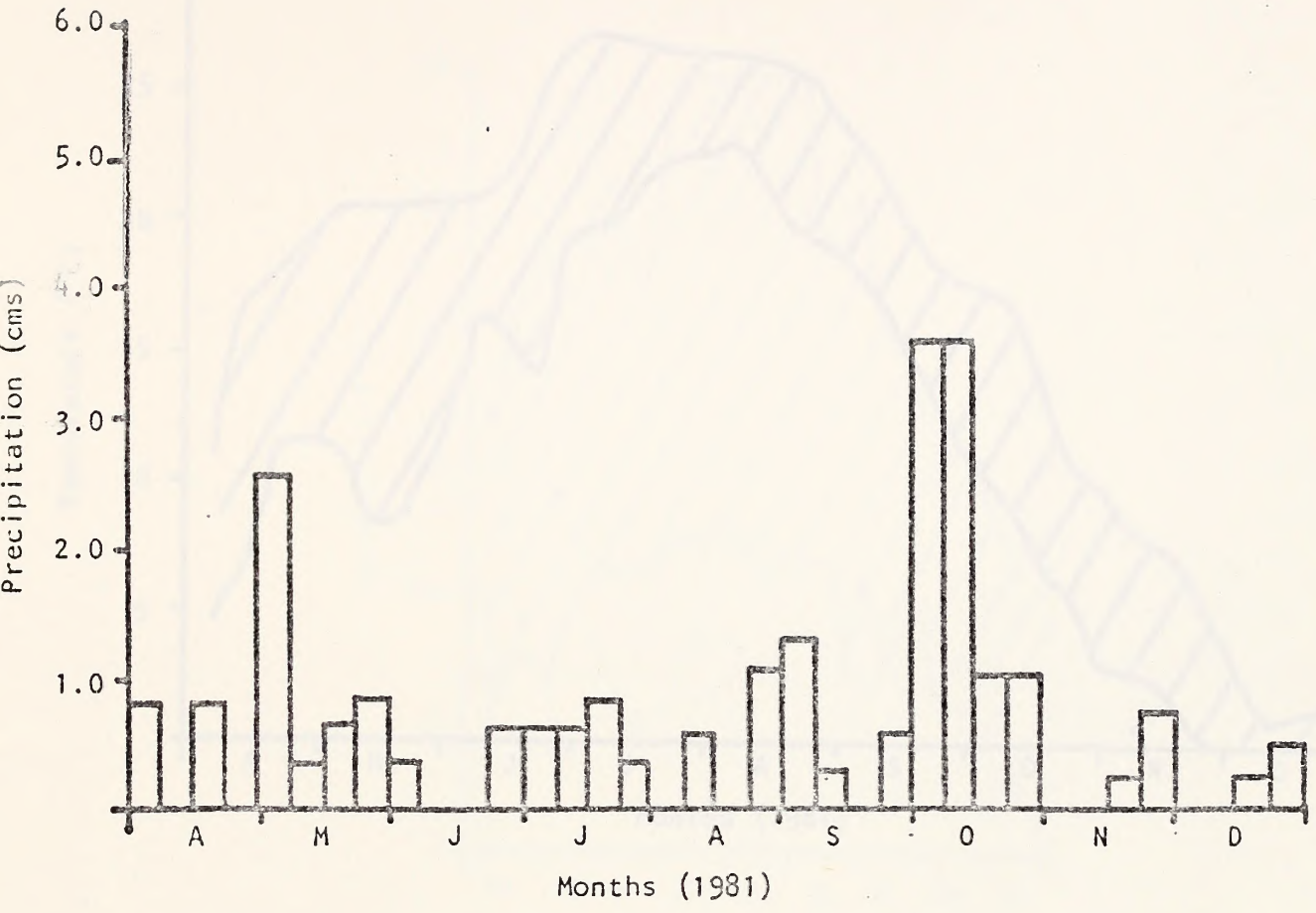


Figure 7. Weekly precipitation collected at rain guage ARA-13 during 1981.

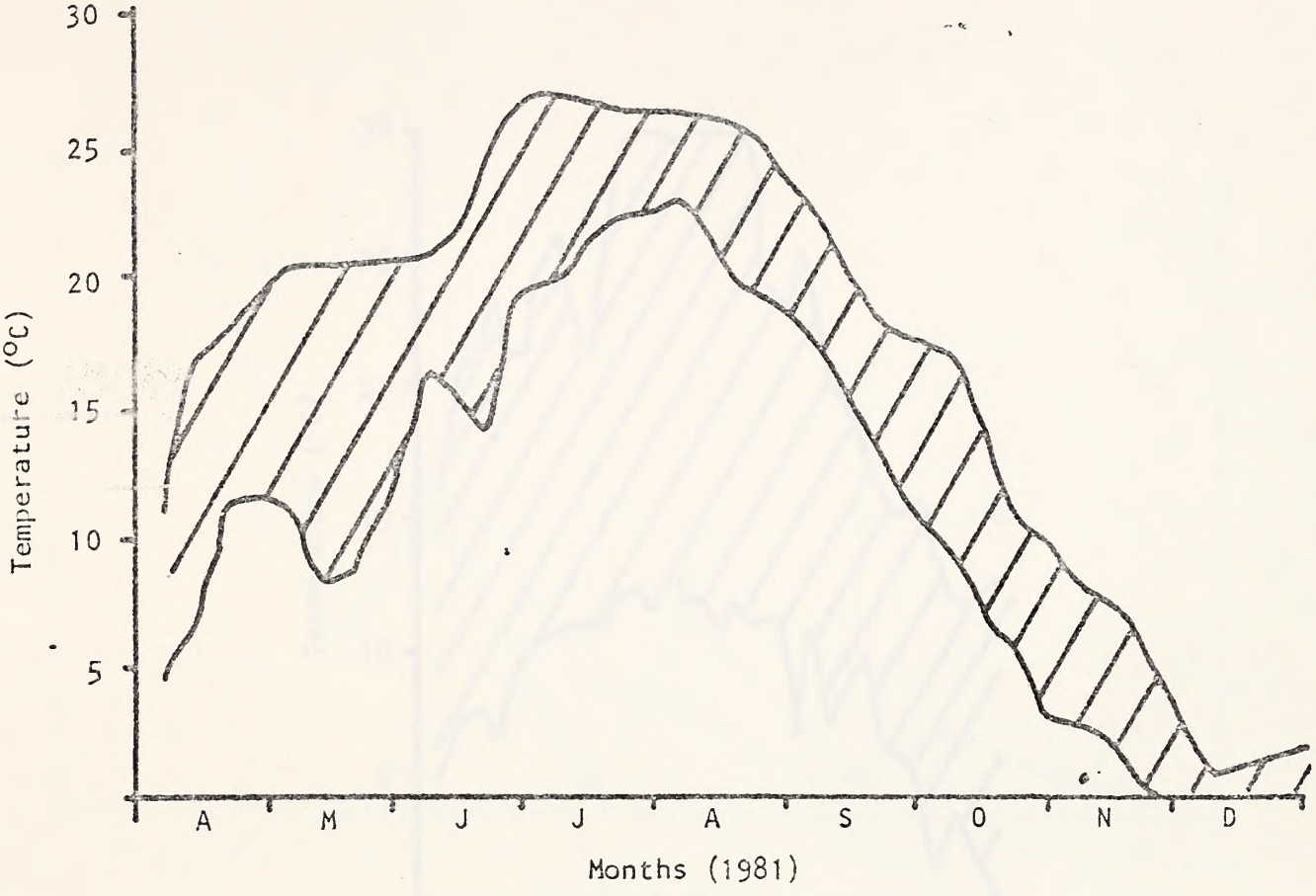


Figure 8. Minimum and maximum recorded water temperatures in the White River during 1981.

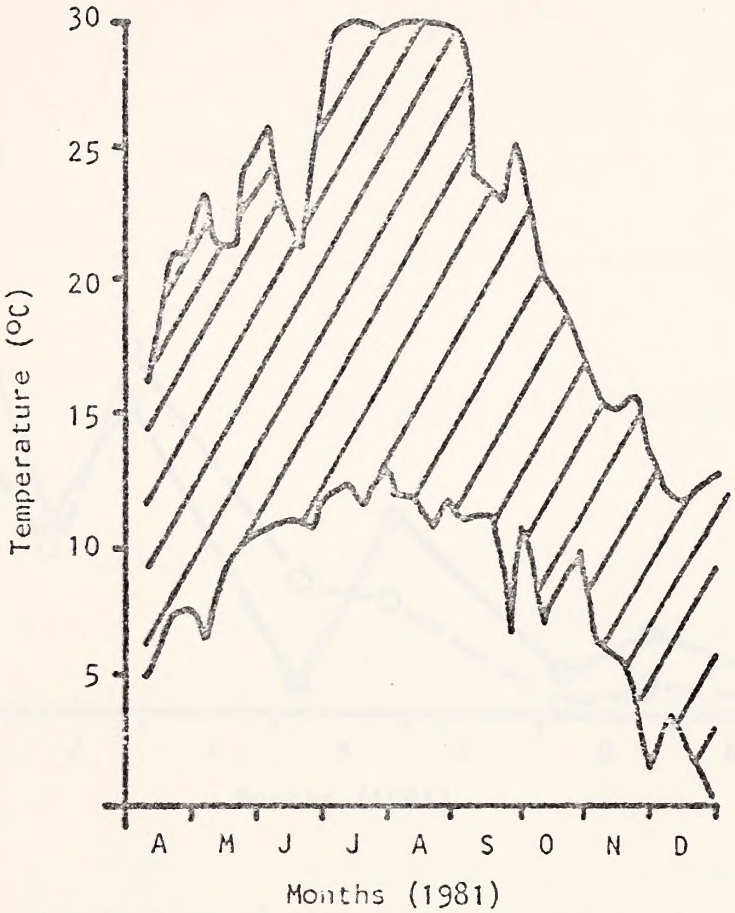


Figure 9. Minimum and maximum recorded water temperatures in Evacuation Creek during 1981.

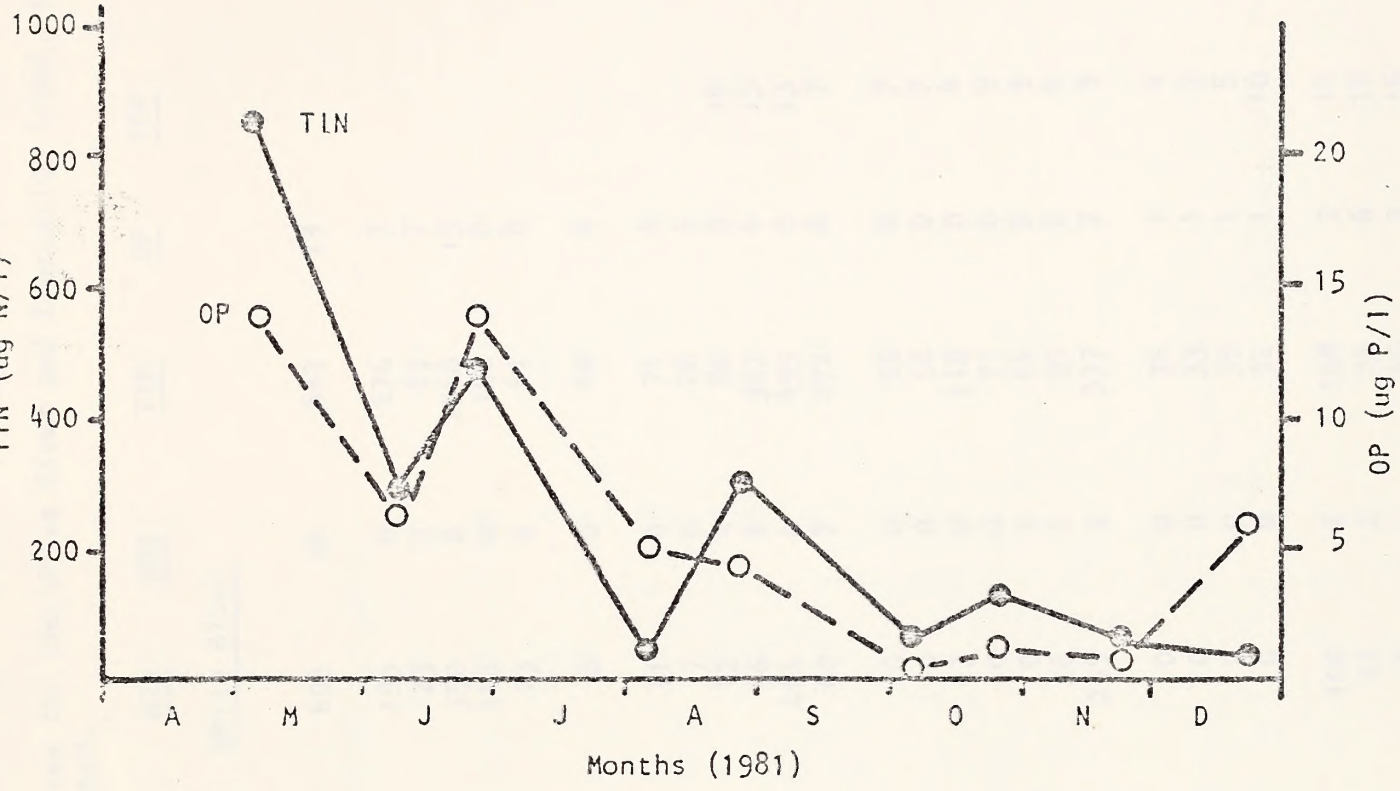


Figure 10. The concentration of orthophosphate and total inorganic nitrogen ($NH_3 + NO_2 + NO_3$) in the White River during 1981. The data represents five-day averages (monthly) at Southam Canyon (Km 79).

Table 4. Results of water chemistry analyses for the White River and Evacuation Creek, 1981. Concentrations are in ug/l unless otherwise stated.

	DATE	SAMPLE	TIME	NH3	NO3	NO2	TIN	OP	TDP	TP	TSS (mg/l)
					White River						
1981	May										
	3	WR		156	608	78	842	13		1577	
	June										
	3	WR		83	193	0	276	7		21	
	27	WR		38	23	2	63	7		48	158
	28	WR		2010	350	8	2368	13		826	13872
	29	WR		40	148	0	188	8		38	125
	30	WR		38	23	0	61	8		145	755
	July										
	1	WR		40	8	0	48	6		86	310
	Aug.										
	7	WR		60	11	0	71	4		19	9
	8	WR		21	7	0	28	5		21	48
	26	WR	1900	70	12	4	86	9	10	11	146
	27	WR	1930	340	16	6	362	4	15	16	135
	28	Last P/R		47	444	4	495	0	13	23	153
	28	Storm		205	60	7	272	6	7	13	346
	Oct.										
	1	WR18	1253	55	0	0	55	0	7	9	215
	2	WR27		58	0	0	58	0	7	9	236
	3	WR05		71	47	0	118	0	4	15	205
	4	WR18		71	0	0	71	0	9	48	343
	29	P/R		63	0	1	64	2	9	16	466
	30	WR		68	26	1	95	2	4	6	433
	31	WR		75	259	3	337	7	9	23	1356
	Nov.										
	21	WR		36	0	0	36	1	4	5	160
	22	WR29	1519	33	0	0	33	1	3	22	148
	23	WR18		39	0	0	39	1	5	29	210
	24	WR18	1652	32	0	0	32	1	10	27	172
	Dec.										
	29	Snowmelt		6	100	2	108	2	14	15	195
	30	Before melt	1000	0	22	2	24	6	18	19	136
	30	After melt	1530	7	38	1	46	3	16	23	237

	DATE	SAMPLE	TIME	NH3	NO3	NO2	TIN	OP	TDP	TP	TSS
	Dec.										
	30		2218	0	28	0	28	7	13	33	91
	31		0930	9	22	2	33	3	19	20	107
1982	Jan.										
	1	WR20		35	45	2	82	8	40	65	578
				Evacuation Creek							
1981		EC02		2012	2230	741	4983	4	12	26	
	May										
	3	EC		62	833	150	1045	8	-	760	
	June										
	3	EC		62	611	7	680	4	-	17	
	30	EC		64	356	16	436	8	-	40	142
	Aug.										
	8	EC		12	154	0	166	6	-	8	25
	Oct.										
	3	EC		113	490	4	607	0	4	12	302
	31	EC		41	424	4	469	7	7	18	96
	Nov.										
	24	EC01		53	151	10	214	2	7	23	47
	Dec.										
	29	EC02		20	339	7	366	11	15	20	89

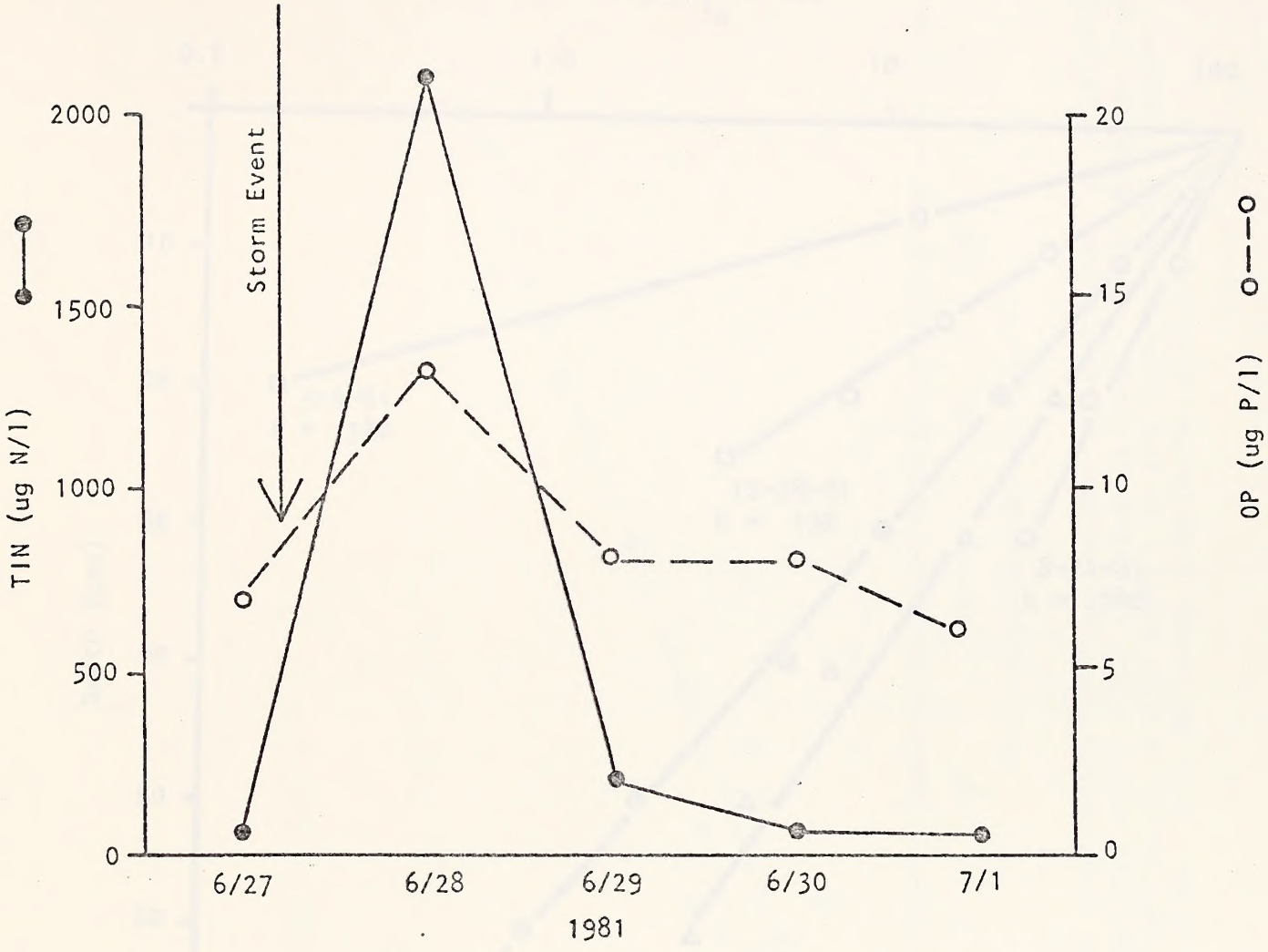


Figure 11. The effect of a storm event (6/27/81 to 7/1/81) on the concentrational changes of total inorganic nitrogen and orthophosphate in the White River. Samples were collected at Southam Canyon (km 79).

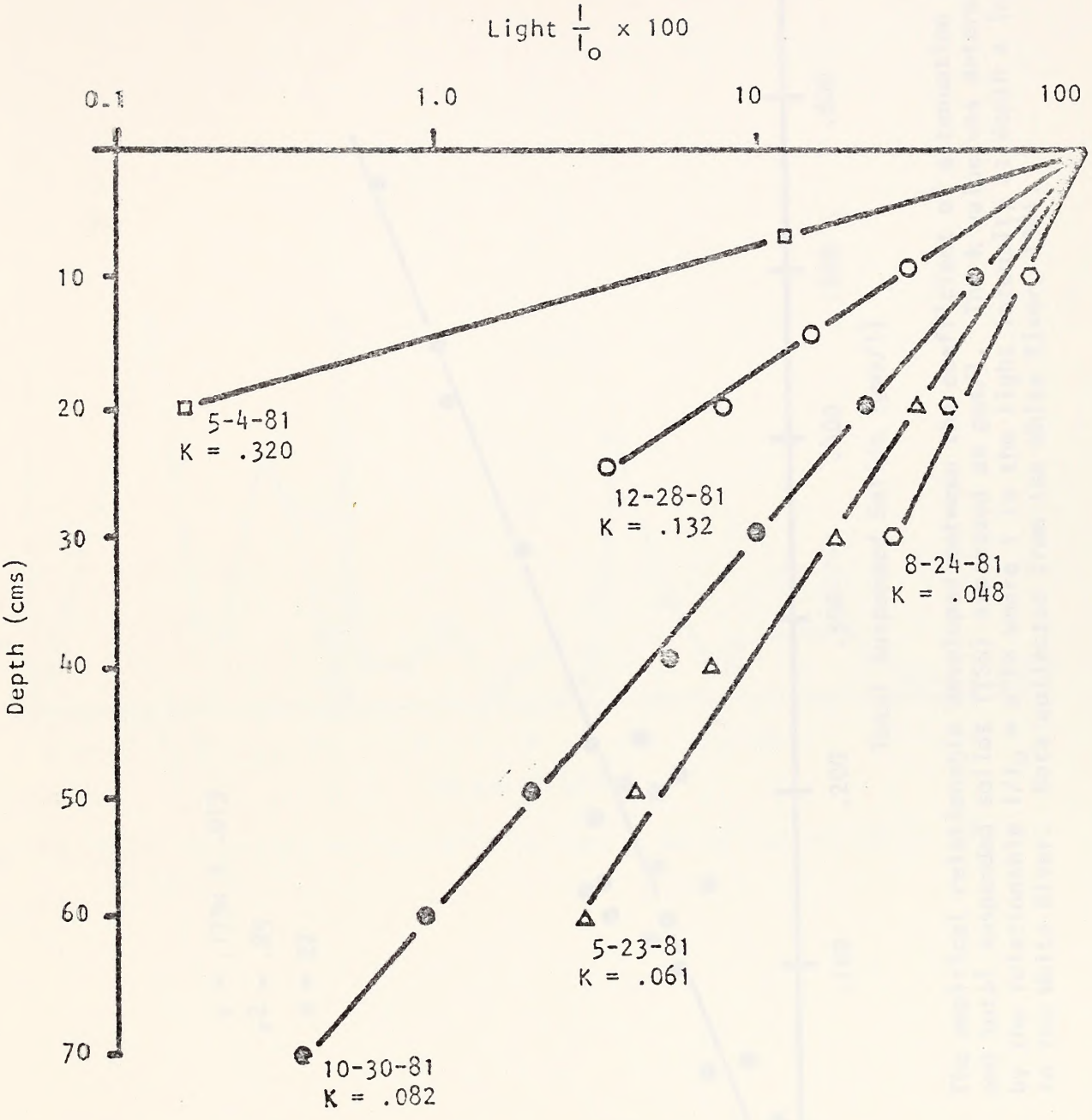


Figure 12. Attenuation coefficients and light profiles for different sample periods during 1981.

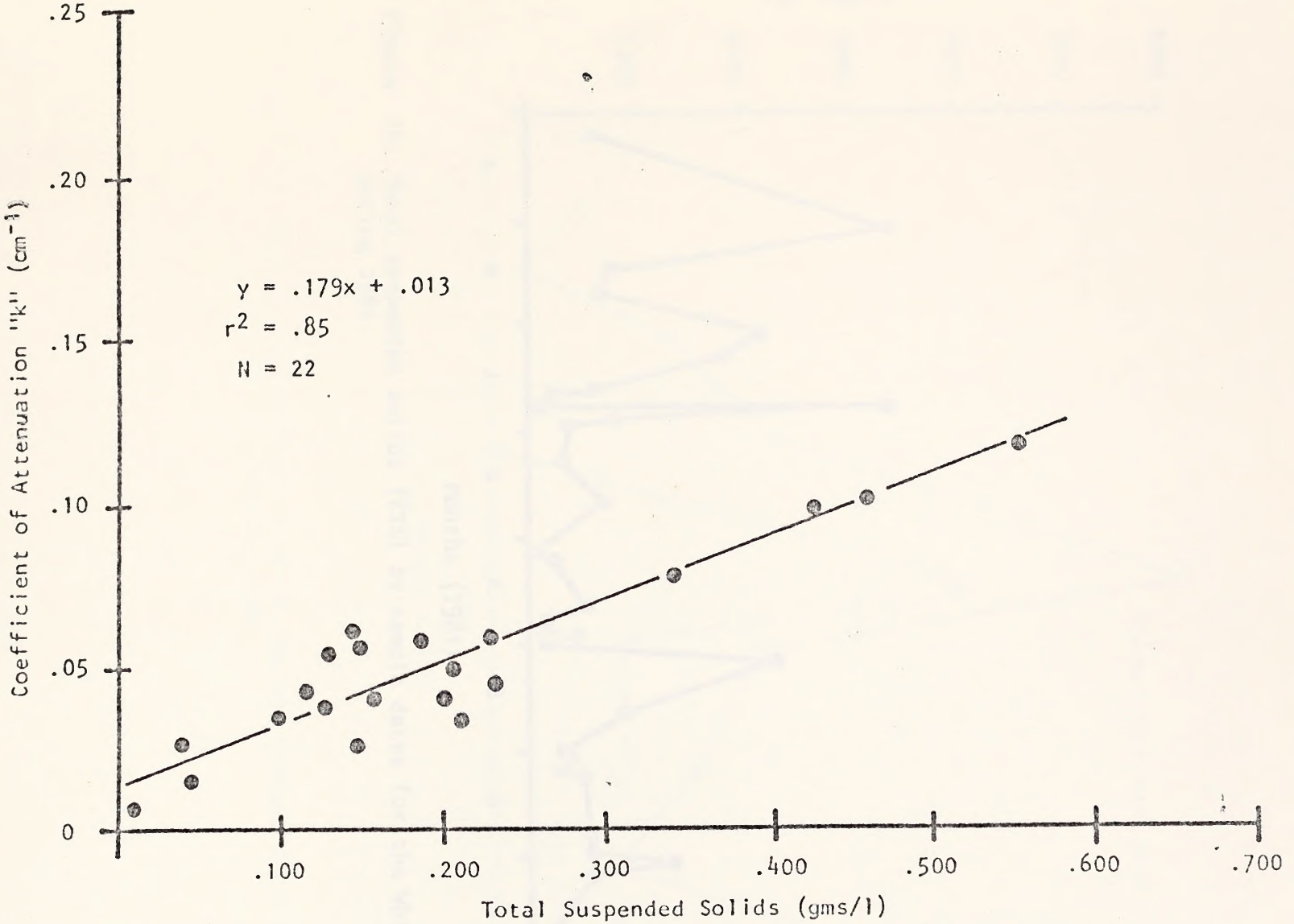


Figure 13. The empirical relationship developed between the coefficient of attenuation ("k") and total suspended solids (TSS) expressed as gms/l. The k value was determined by the relationship $I/I_o = e^{-kx}$ where I is the light intensity at depth x (cms) in the White River. Data collected from the White River, 1981.

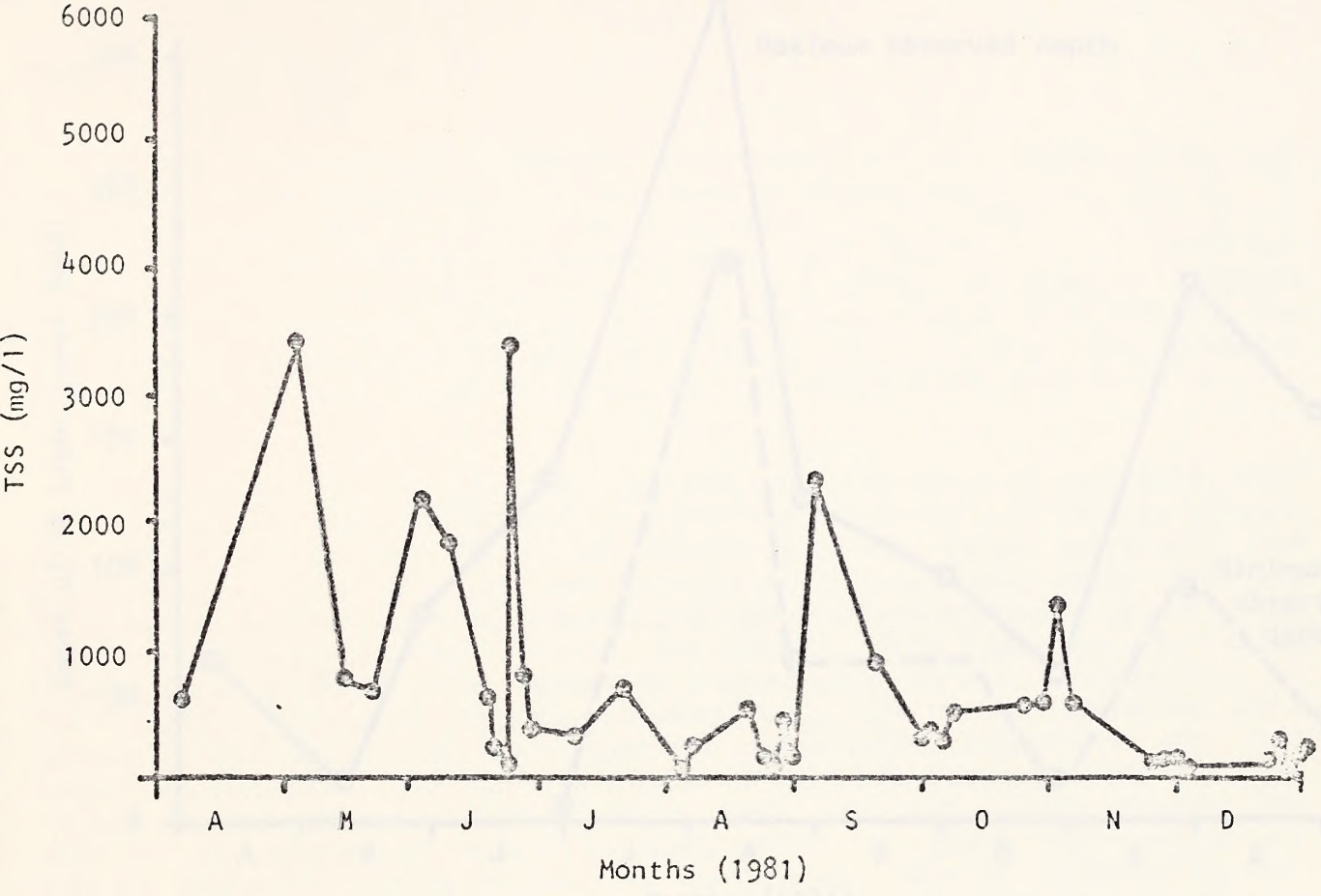


Figure 14. Total suspended solids (TSS) by sample dates for the White River during 1981.

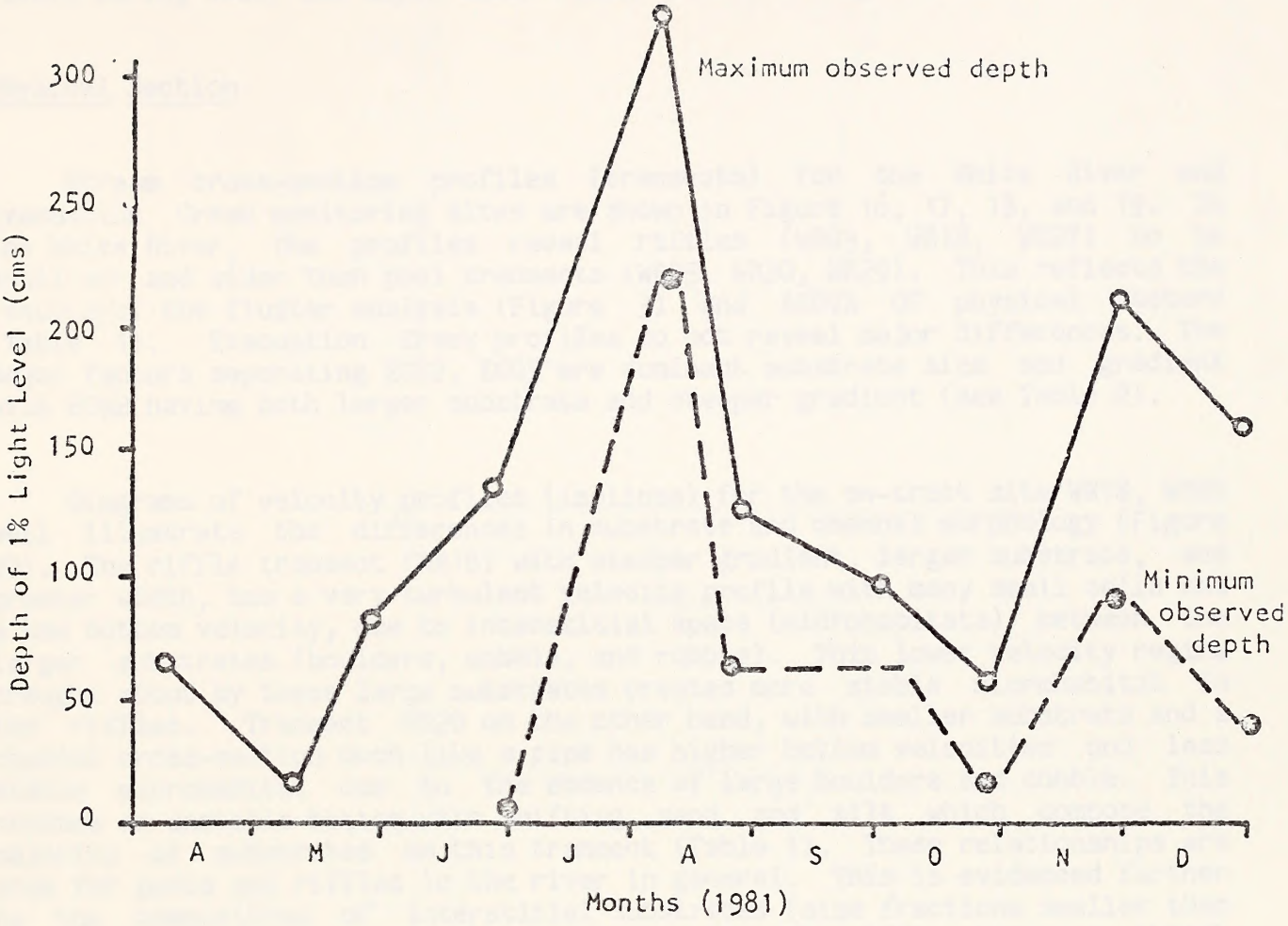


Figure 15. Depth of light penetration to the one percent light level in the White River during 1981.

Maximum light penetration occurs between June and December, with Minimum levels during lower and upper basin runoff (times of high sediment load).

Physical Section

Stream cross-section profiles (transects) for the White River and Evacuation Creek monitoring sites are shown in Figure 16, 17, 18, and 19. In the White River, the profiles reveal riffles (WR03, WR18, WR27) to be shallower and wider than pool transects (WR05, WR20, WR29). This reflects the results of the Clustar analysis (Figure 3) and ANOVA OF physical factors (Table 1). Evacuation Creek profiles do not reveal major differences. The major factors separating EC02, EC03 are dominant substrate size and gradient with EC02 having both larger substrate and steeper gradient (see Table 2).

Diagrams of velocity profiles (isolines) for the on-tract site WR18, WR20 well illustrate the differences in substrate and channel morphology (Figure 20). The riffle transect (WR18) with steeper gradient, larger substrate, and greater width, has a very turbulent velocity profile with many small cells and a low bottom velocity, due to interstitial space (microhabitats) between the larger substrates (boulders, cobble, and rubble). This lower velocity regime brought about by these large substrates creates more stable microhabitat in the riffles. Transect WR20 on the other hand, with smaller substrate and a channel cross-section much like a pipe has higher bottom velocities and less stable microhabitat due to the absence of large boulders and cobble. This creates an unstable bottom with shifting sand and silt which compose the majority of substrates on this transect (Table 1). These relationships are true for pools and riffles in the river in general. This is evidenced further by the composition of interstitial substrates (size fractions smaller than 12.7 mm in diameter) shown in Figures 21 and 22. The riffle transect (WR18) has very stable percentages of sand and silt (size fractions less than 0.25 mm) throughout the year, whereas in Transect WR20 (pool) these substrates exhibit large monthly changes.

Because stable substrates and light penetration are important to the success of in-stream primary production, the distribution of different substrate classes with depth is important. For example, during runoff and storm events with the large increase in suspended solids, light penetration can be limited to less than ten cms. In this case, it is evident from Figure 23 that very little of the larger stable substrates (boulder, cobble, rubble, gravel) fall into a light climate suitable for primary production. During the fall and winter months, light penetration is generally greater than 40 cms which greatly increases the available stable substrate upon which the periphyton community depends.

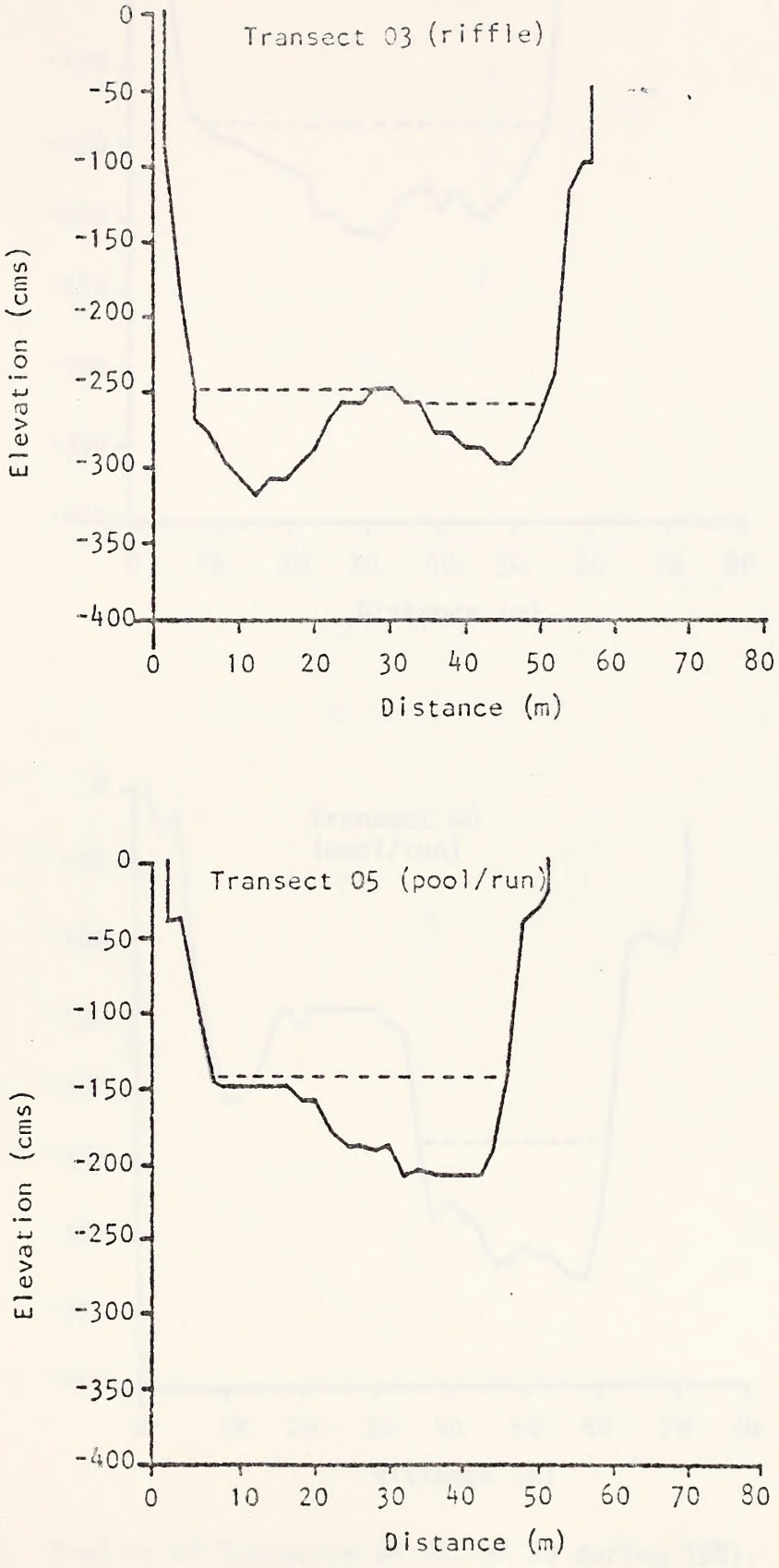


Figure 16. Profile of Transects WR 03, WR 05 during 1981.

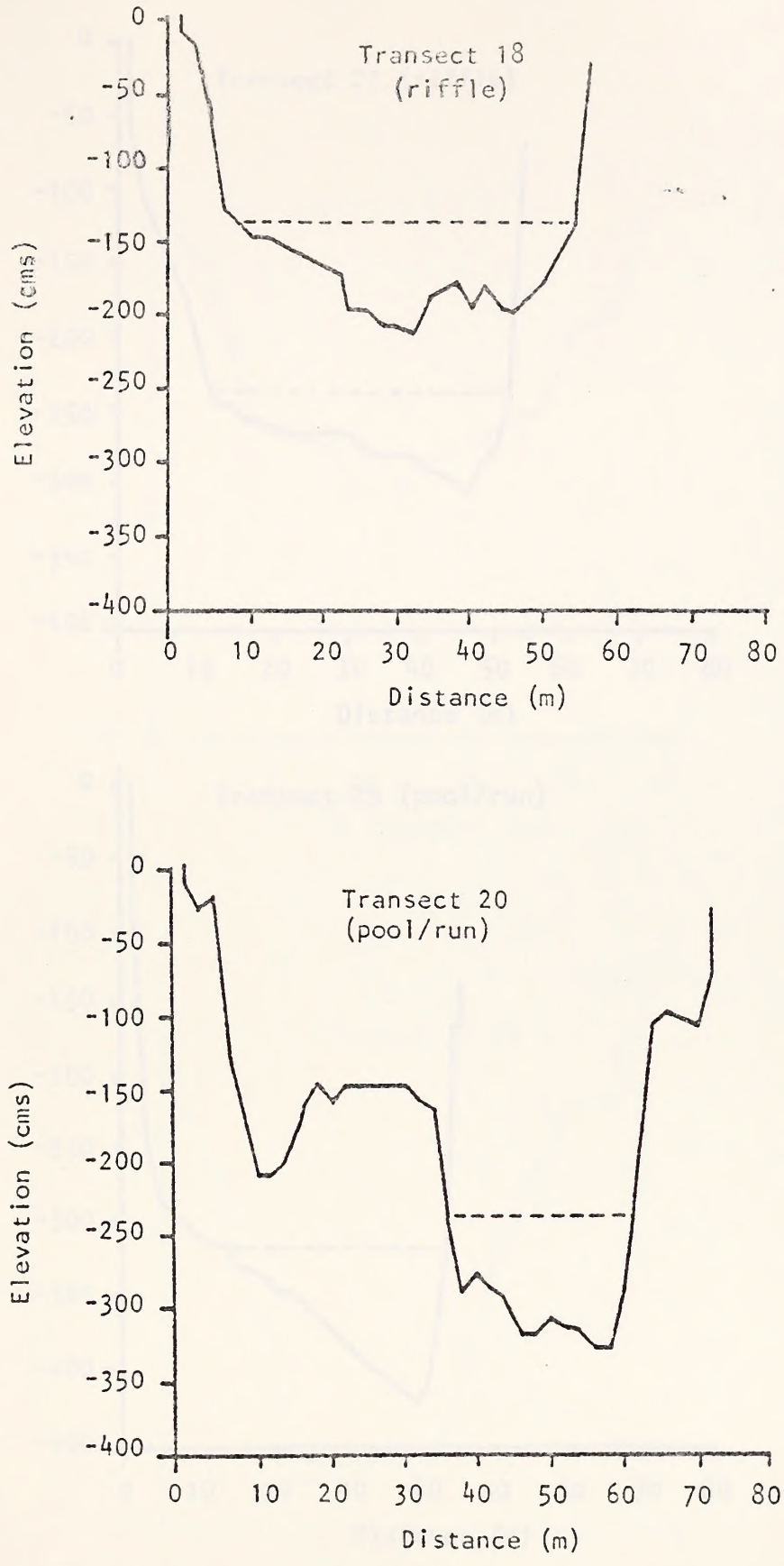


Figure 17. Profile of Transects WR 18, WR 20 during 1981.

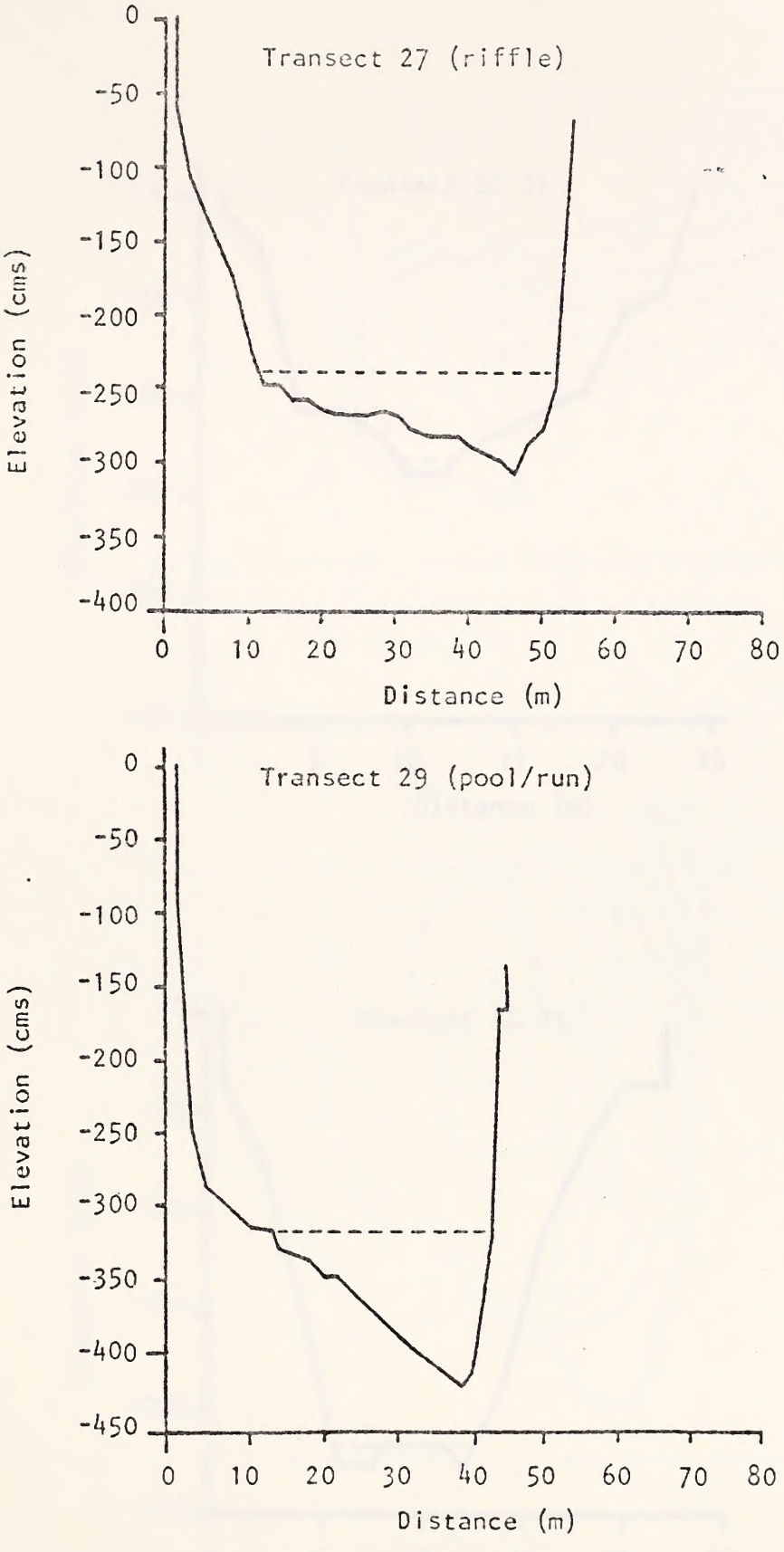


Figure 18. Profile of Transects WR 27, WR 29 during 1981.

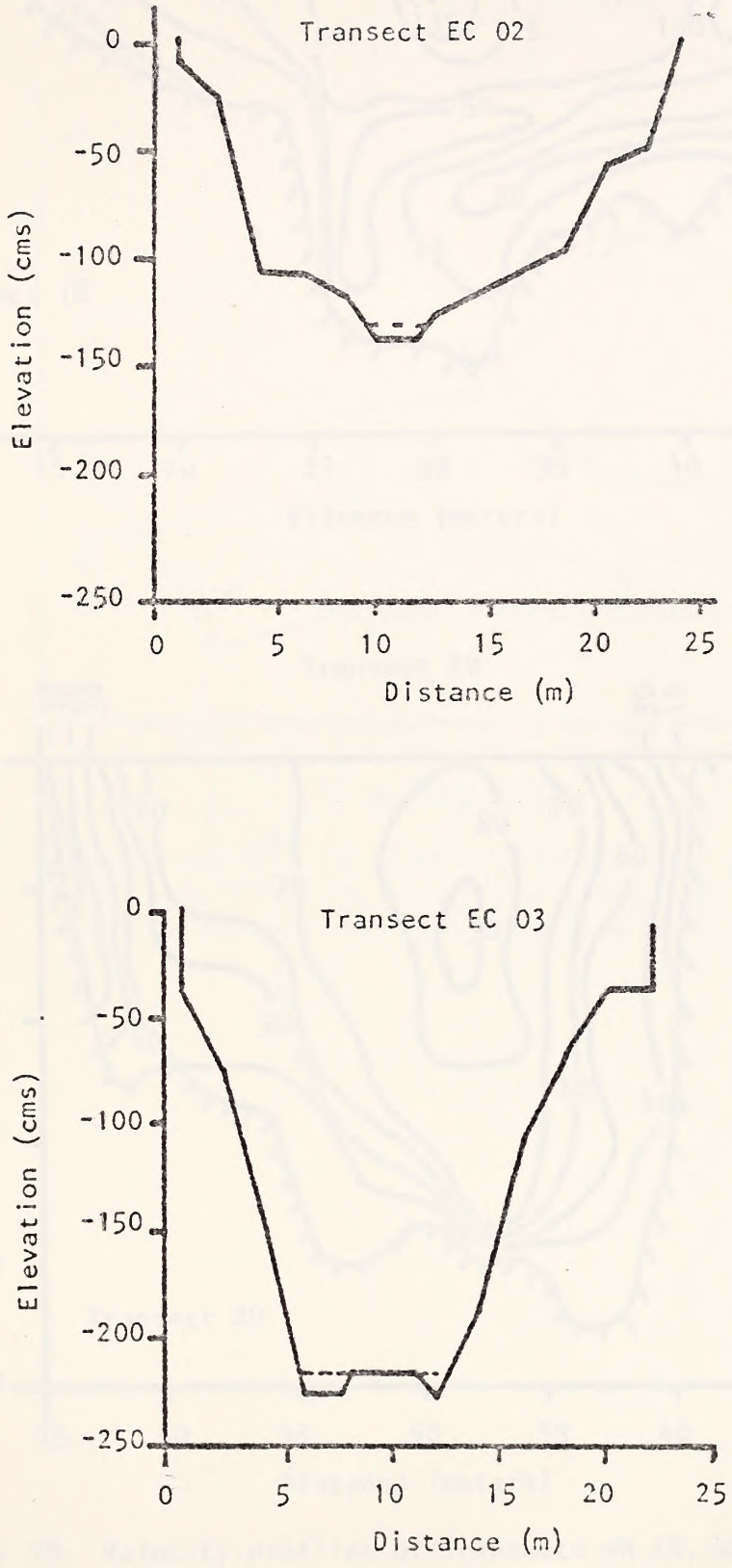


Figure 19. Profile of Transects EC 02, EC 03 during 1981.

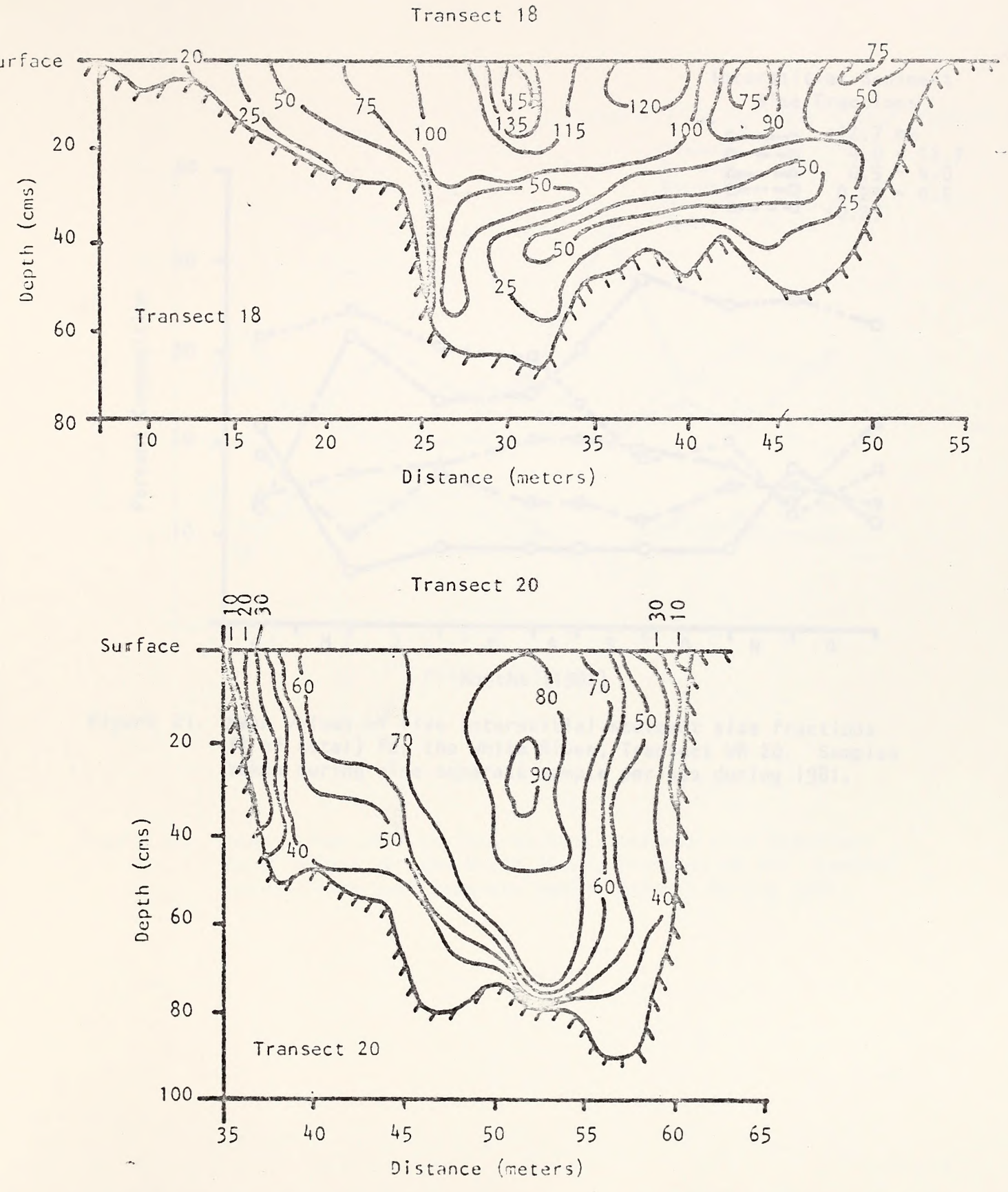


Figure 20. Velocity profiles of Transects WR 18, WR 20 showing velocity isolines.

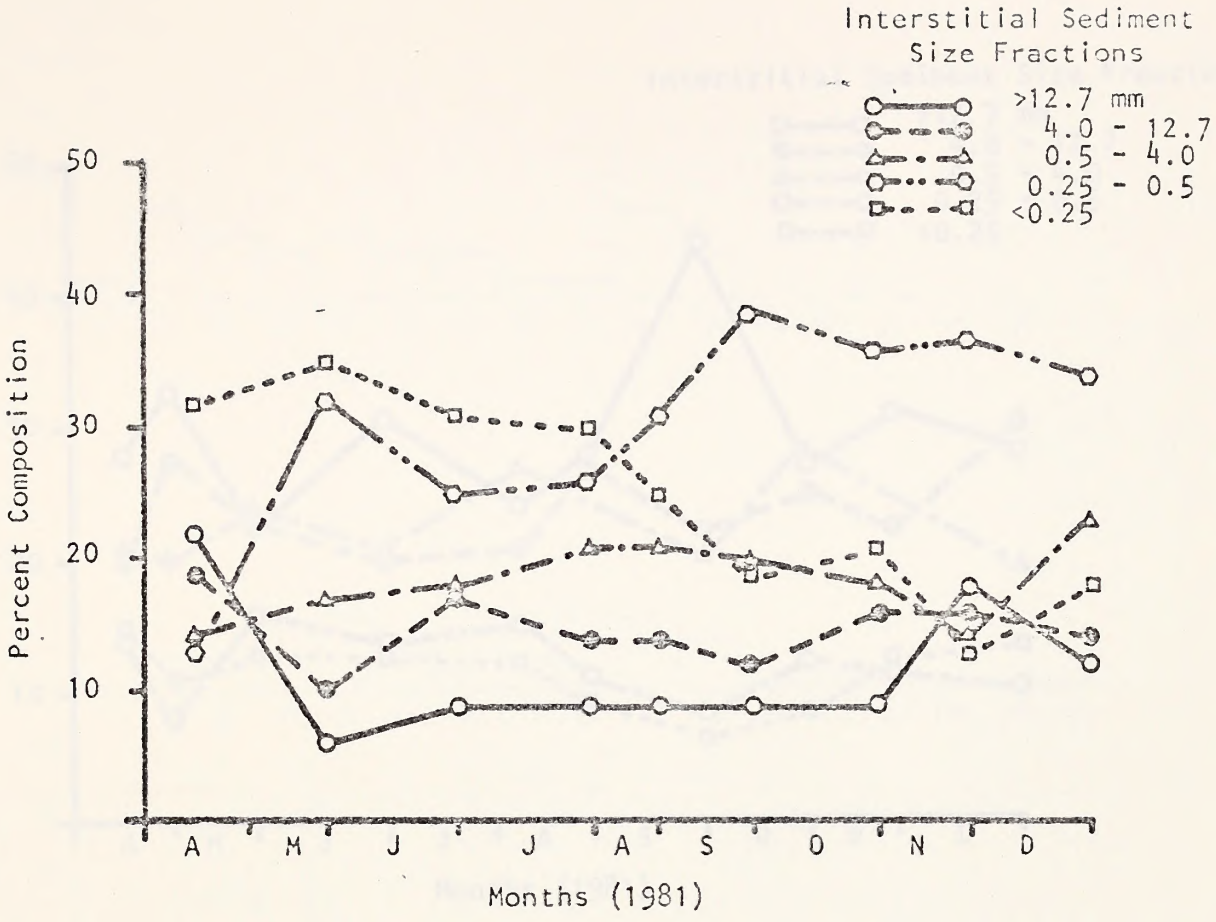


Figure 21. Mean values of five interstitial sediment size fractions (% of total) for the White River, Transect WR 20. Samples taken during nine separate sample periods during 1981.

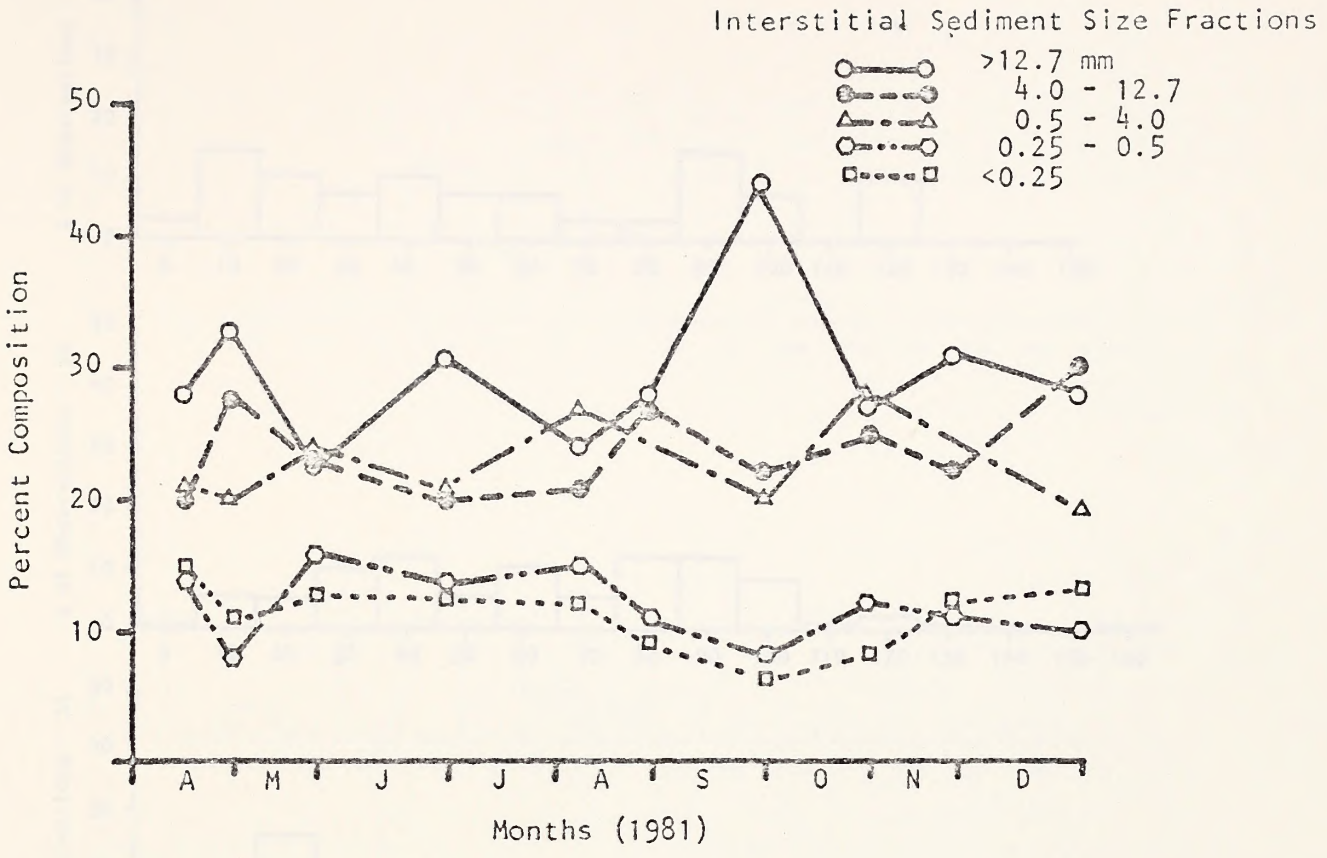


Figure 22. Mean values of five interstitial sediment size fractions (% of total) for the White River, Transect WR 18. Samples taken during nine separate sample periods during 1981.

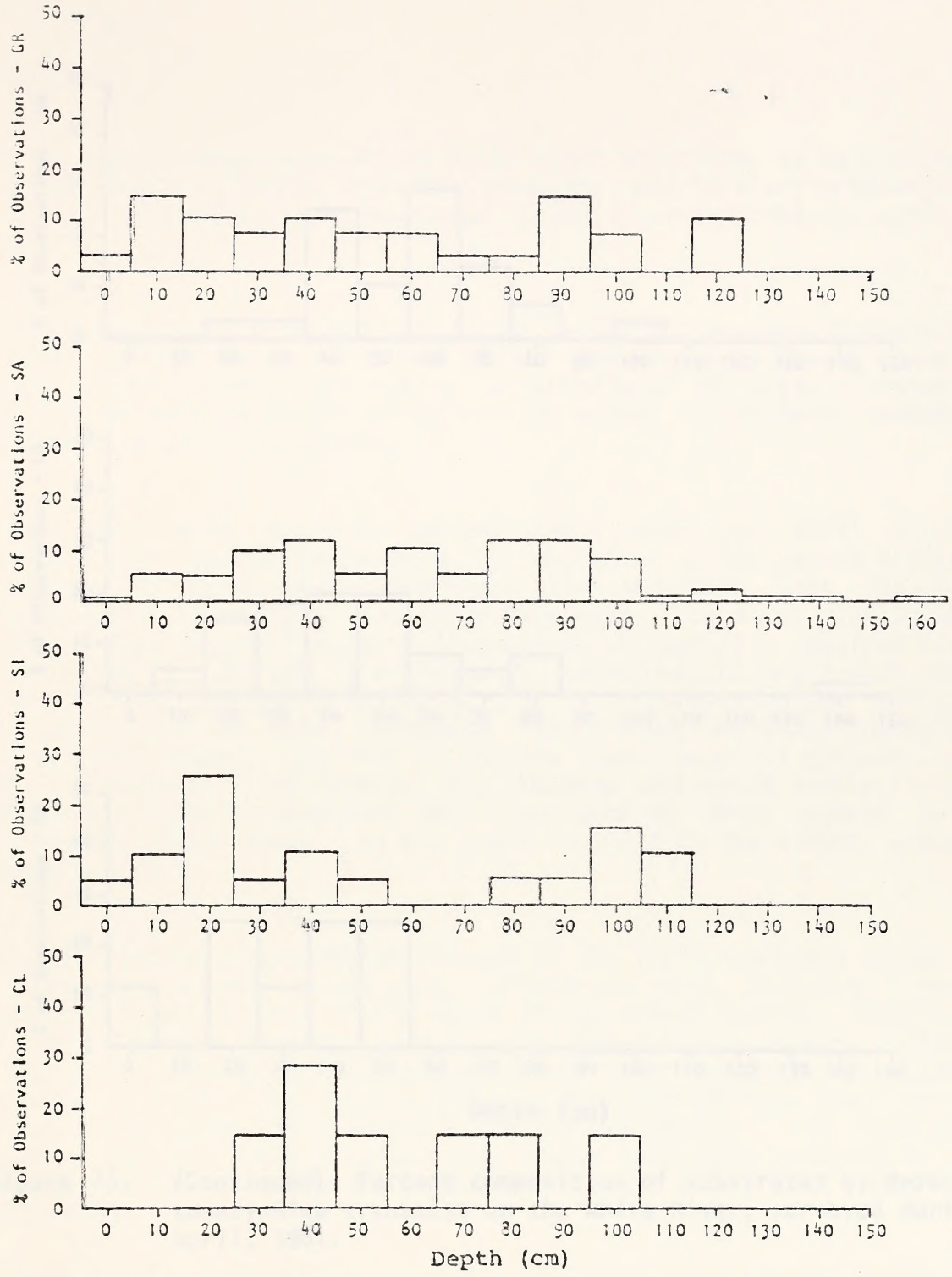


Figure 23. Percent composition of substrates by depth for twenty-nine transects on the White River, surveyed during April, 1981.

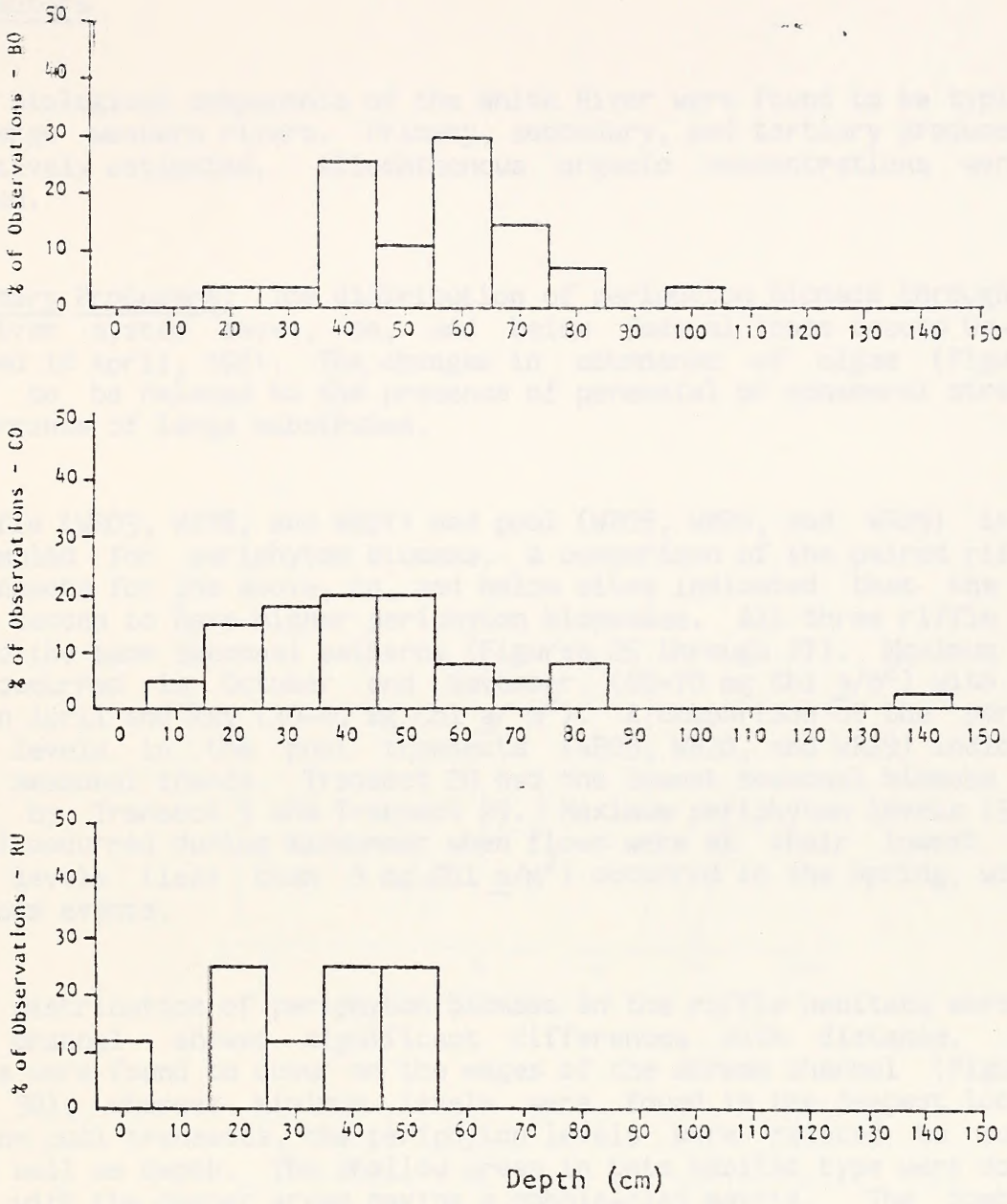


Figure 23. (Continued). Percent composition of substrates by depth for twenty-nine transects on the White River, surveyed during April, 1981.

Biotic Factors

The biological components of the White River were found to be typical of other large western rivers. Primary, secondary, and tertiary producers were quantitatively estimated. Allochthonous organic concentrations were also determined.

Primary Producers: The distribution of periphyton biomass throughout the White River system above, on, and below federal lease tracts Ua-Ub were determined in April, 1981. The changes in abundance of algae (Figure 24) appeared to be related to the presence of perennial or ephemeral streams and the occurrance of large substrates.

Riffle (WR03, WR18, and WR27) and pool (WR05, WR20, and WR29) transects were sampled for periphyton biomass. A comparison of the paired riffle and pool transects for the above, on, and below sites indicated that the riffle habitats tended to have higher periphyton biomasses. All three riffle habitat types had the same seasonal patterns (Figures 25 through 27). Maximum biomass levels occurred in October and November (60-70 mg Chl a/m^2) with minimum values in April and May (30-40 mg Chl a/ m^2). A comparison of the periphyton biomass levels in the pool transects (WR05, WR20, and WR29) indicated no apparent seasonal trends. Transect 20 had the lowest seasonal biomass levels followed by Transect 5 and Transect 29. Maximum periphyton levels (30-40 mg Chl a/m^2) occurred during midsummer when flows were at their lowest levels. Minimum levels (less than 5 mg Chl a/m^2) occurred in the spring, winter or after storm events.

The distribution of periphyton biomass in the riffle habitats across the stream channel showed significant differences with distance. Maximum biomasses were found to occur on the edges of the stream channel (Figures 28 through 30), whereas minimum levels were found in the deepest locations. Within the pool transects, the periphyton levels were related to substrate type as well as depth. The shallow areas in this habitat type were dominated by sand, with the deeper areas having a cobble-clay matrix. The combination of unstable shallow substrates and deep stable substrates resulted in these pool habitats having lower overall periphyton biomass levels when compared to the riffle areas.

An example of the spatial and temporal pattern of periphyton biomass within a riffle transect (WR18) can be seen in Figure 31. Although the density of algae change over time, the general distribution of biomass by distance (or depth) is relatively constant.

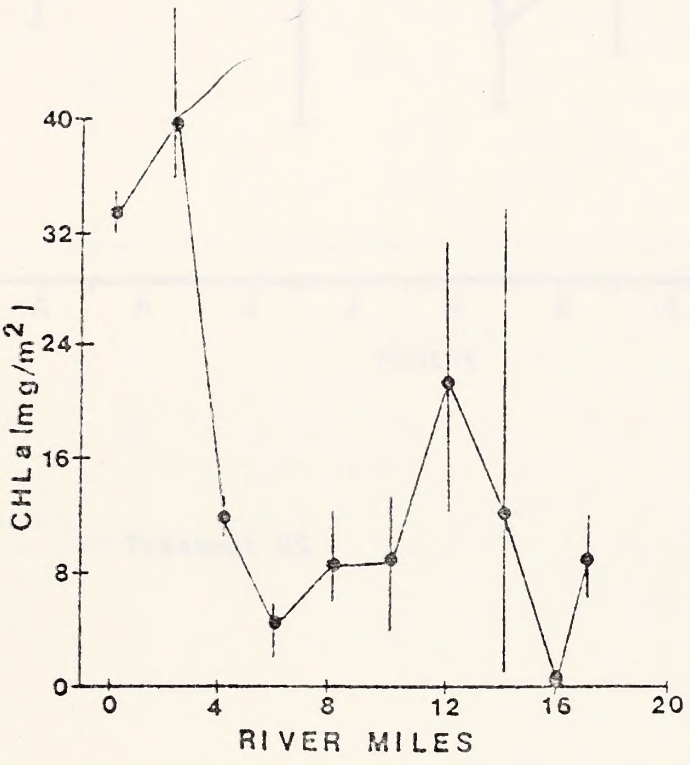
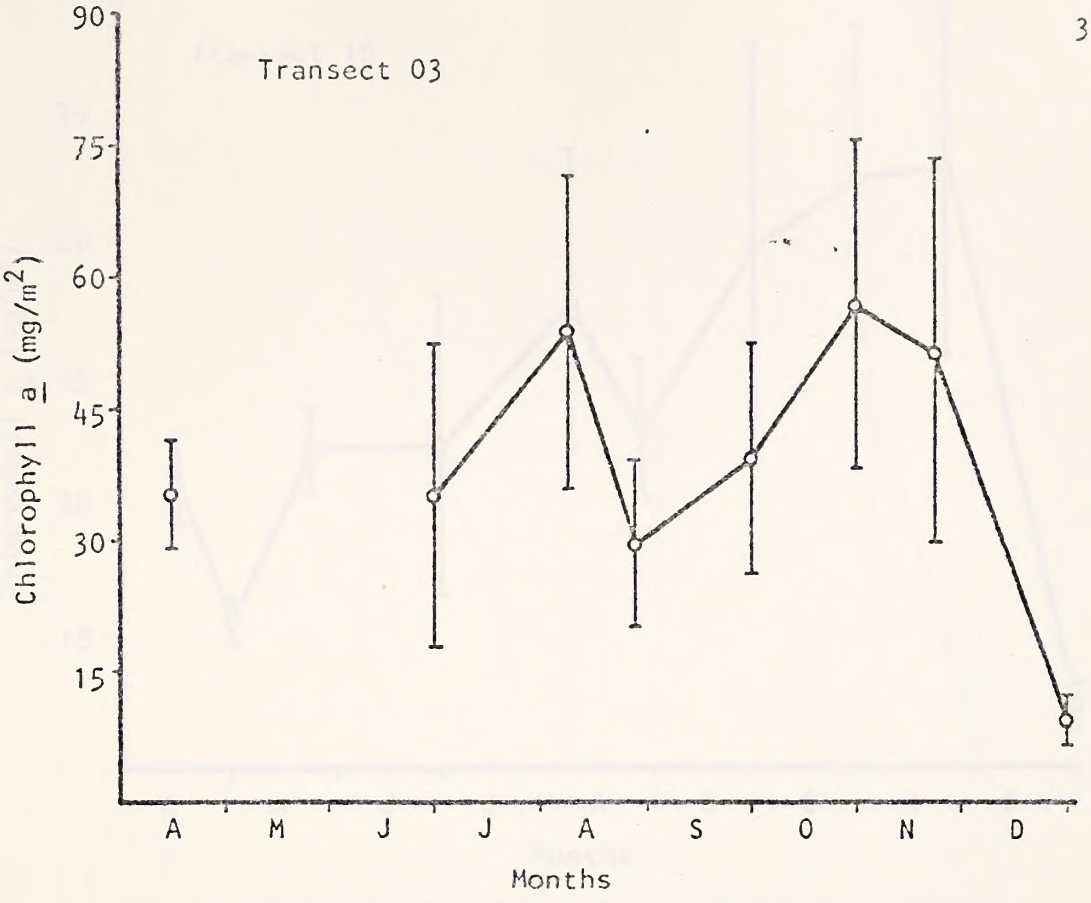


Figure 24. Mean periphyton biomass (Chl a mg/m^2) by river mile between Hells' Hole Canyon (Mile 0) and Asphalt Wash (Mile 17) during April, 1981.



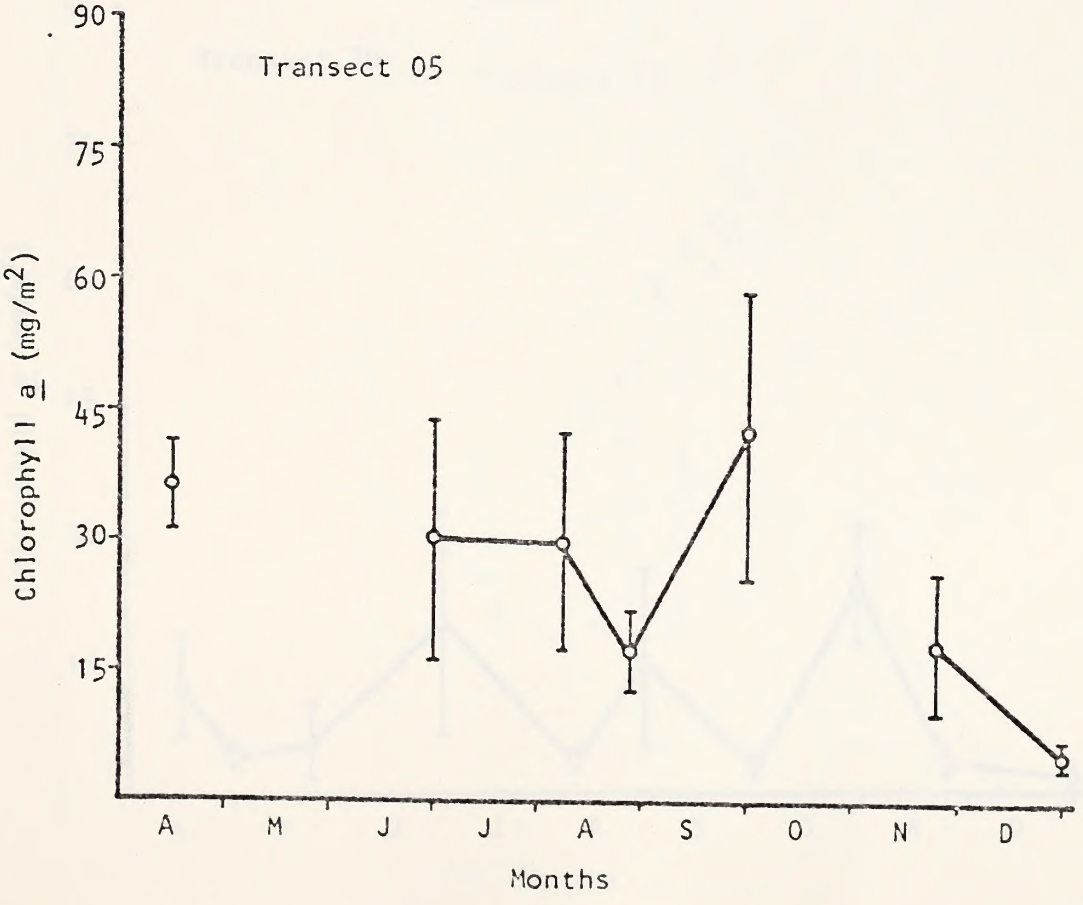


Figure 25. Mean periphyton biomass (Chl a mg/m^2) for Transect WR 03, WR 05 during 1981.

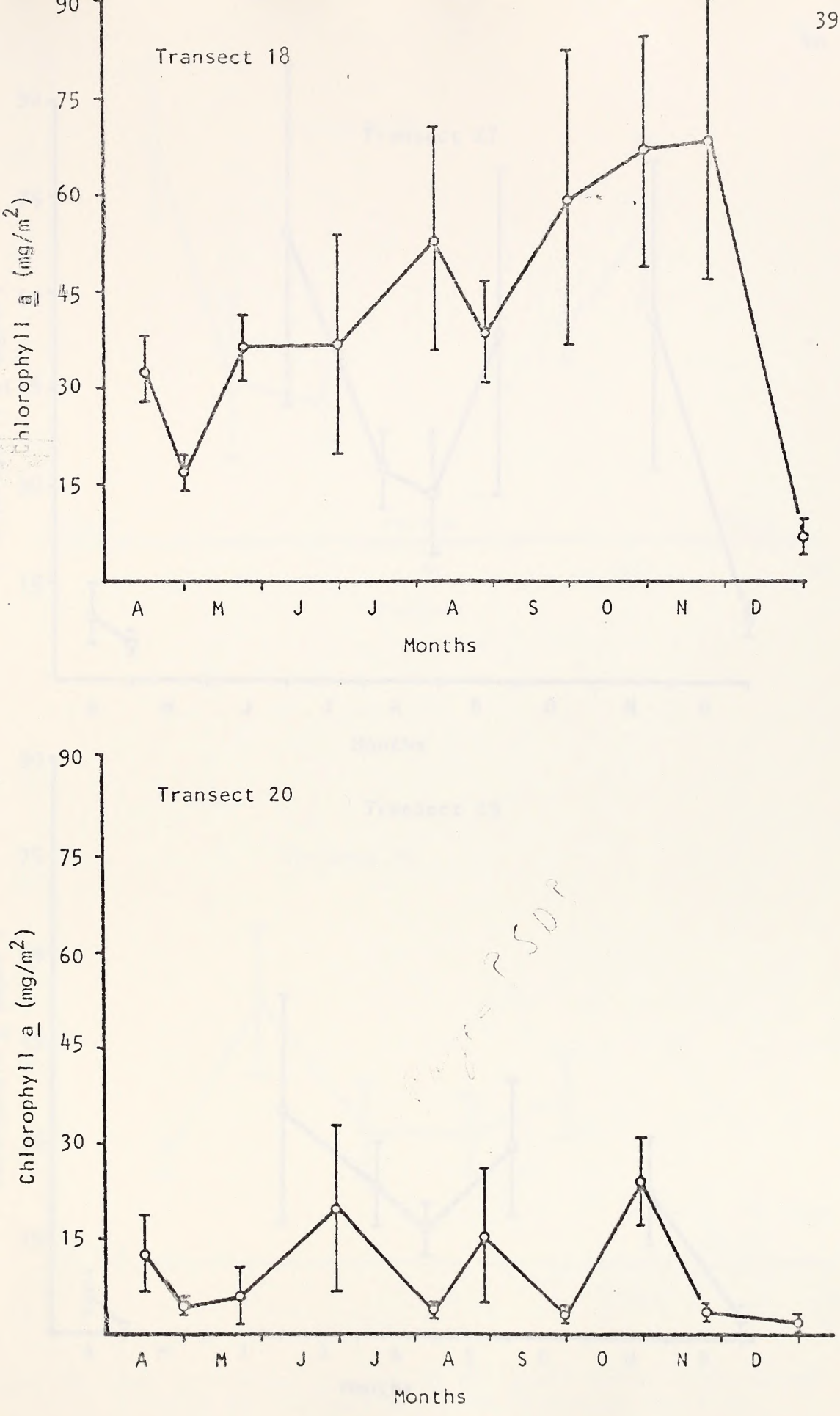


Figure 26. Mean periphyton biomass (Chl a mg/m²) for Transects WR 18, WR 20 during 1981.

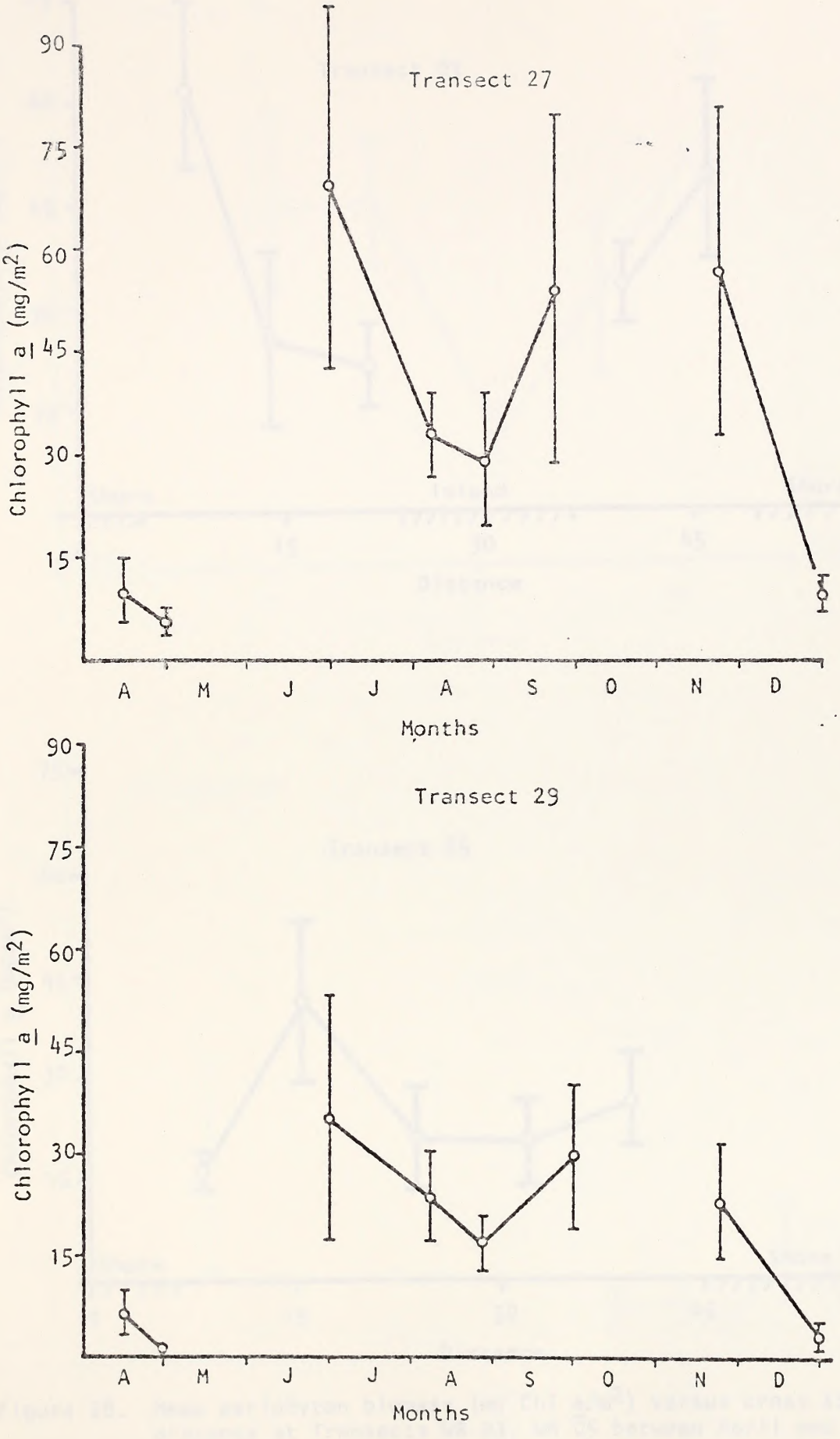
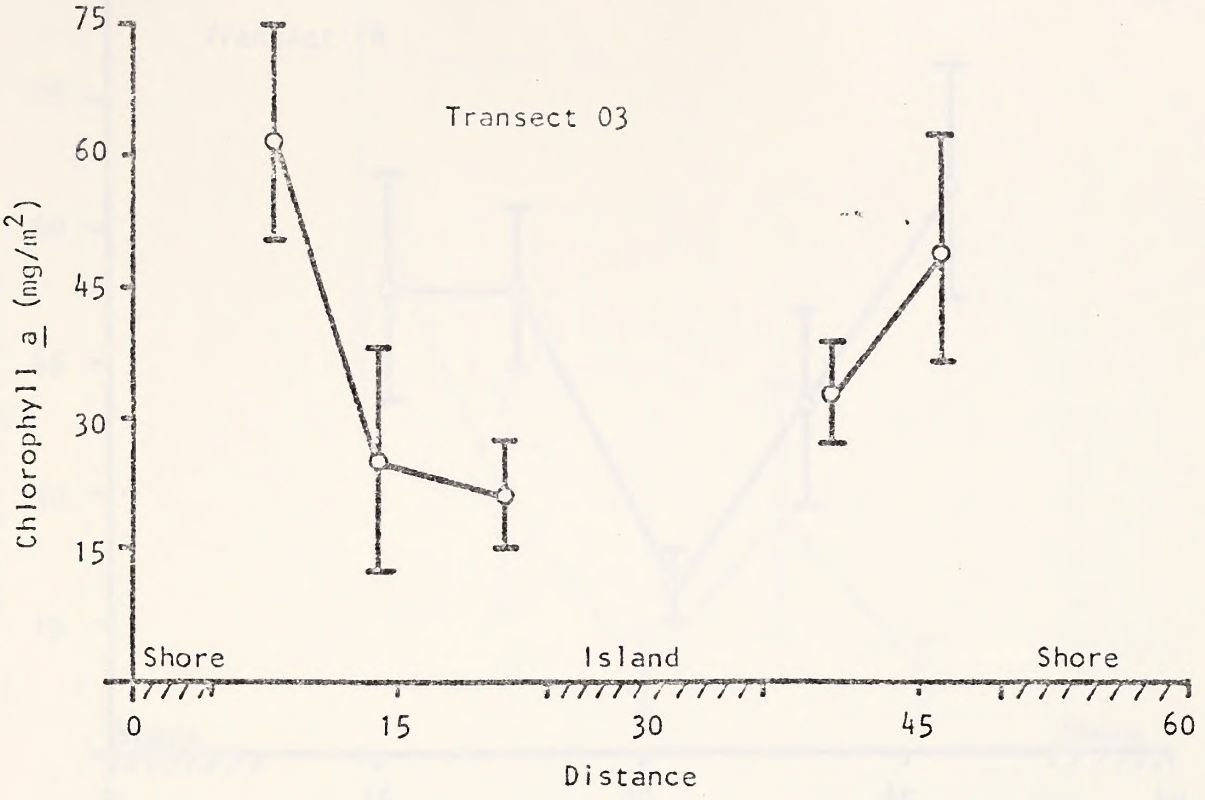


Figure 27. Mean periphyton biomass (Chl a mg/m^2) for Transects WR 27, WR 29 during 1981.



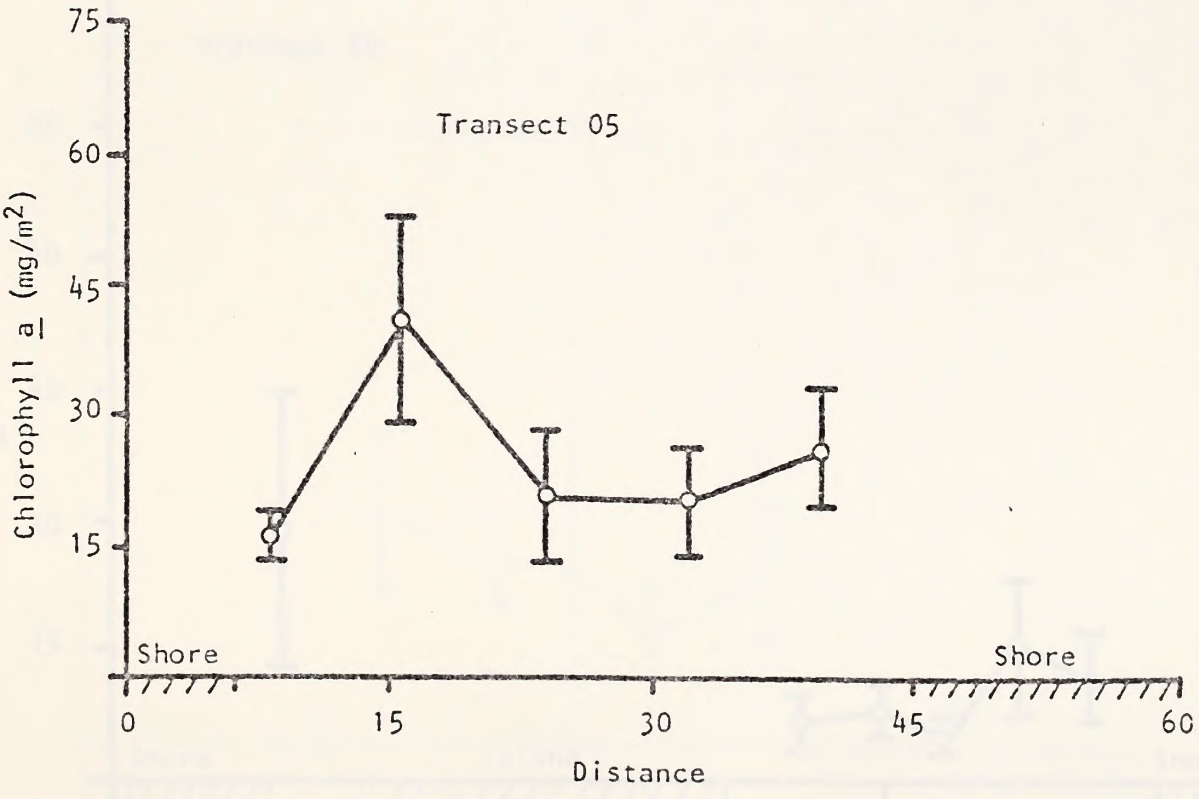
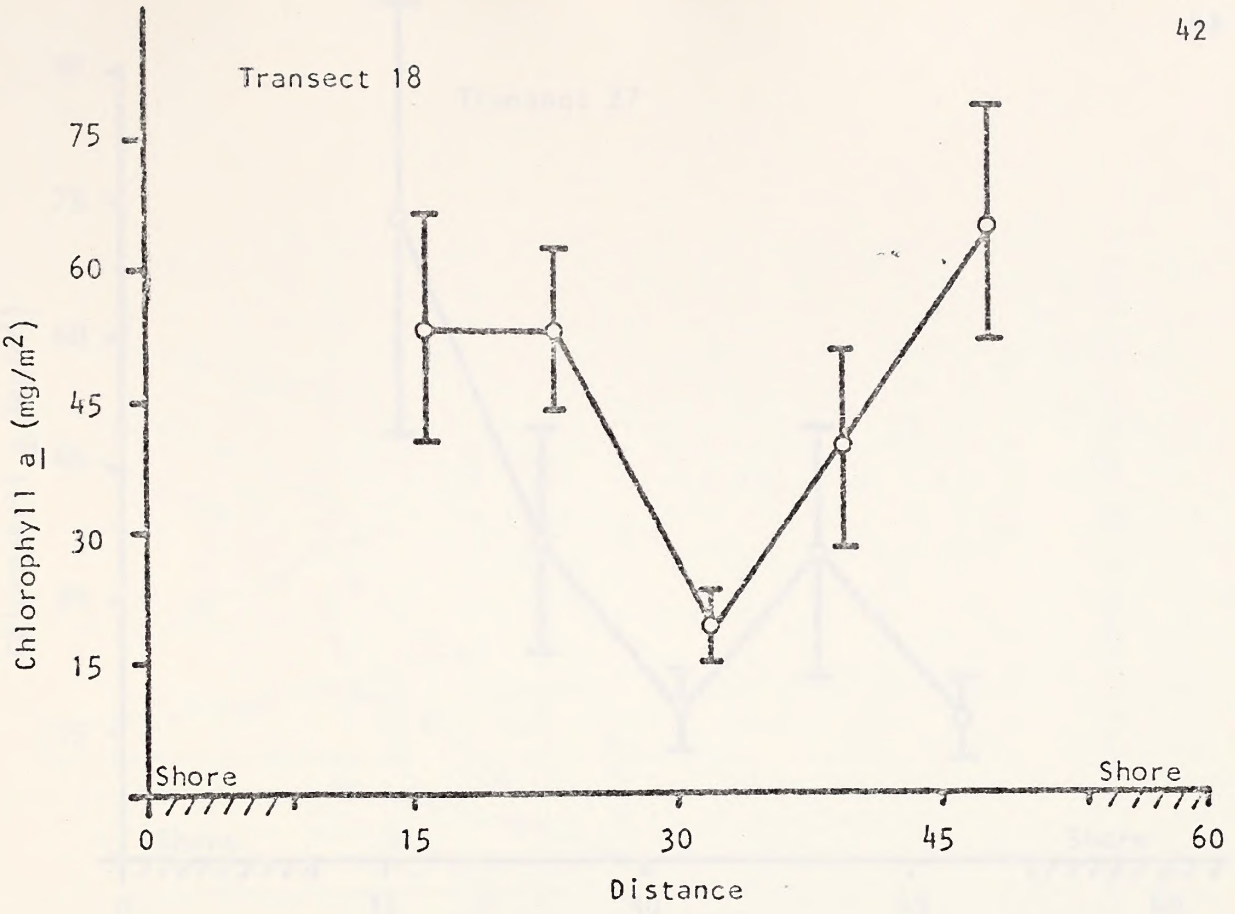


Figure 28. Mean periphyton biomass (mg Chl a/m^2) versus cross stream distance at Transects WR 03, WR 05 between April and December, 1981.



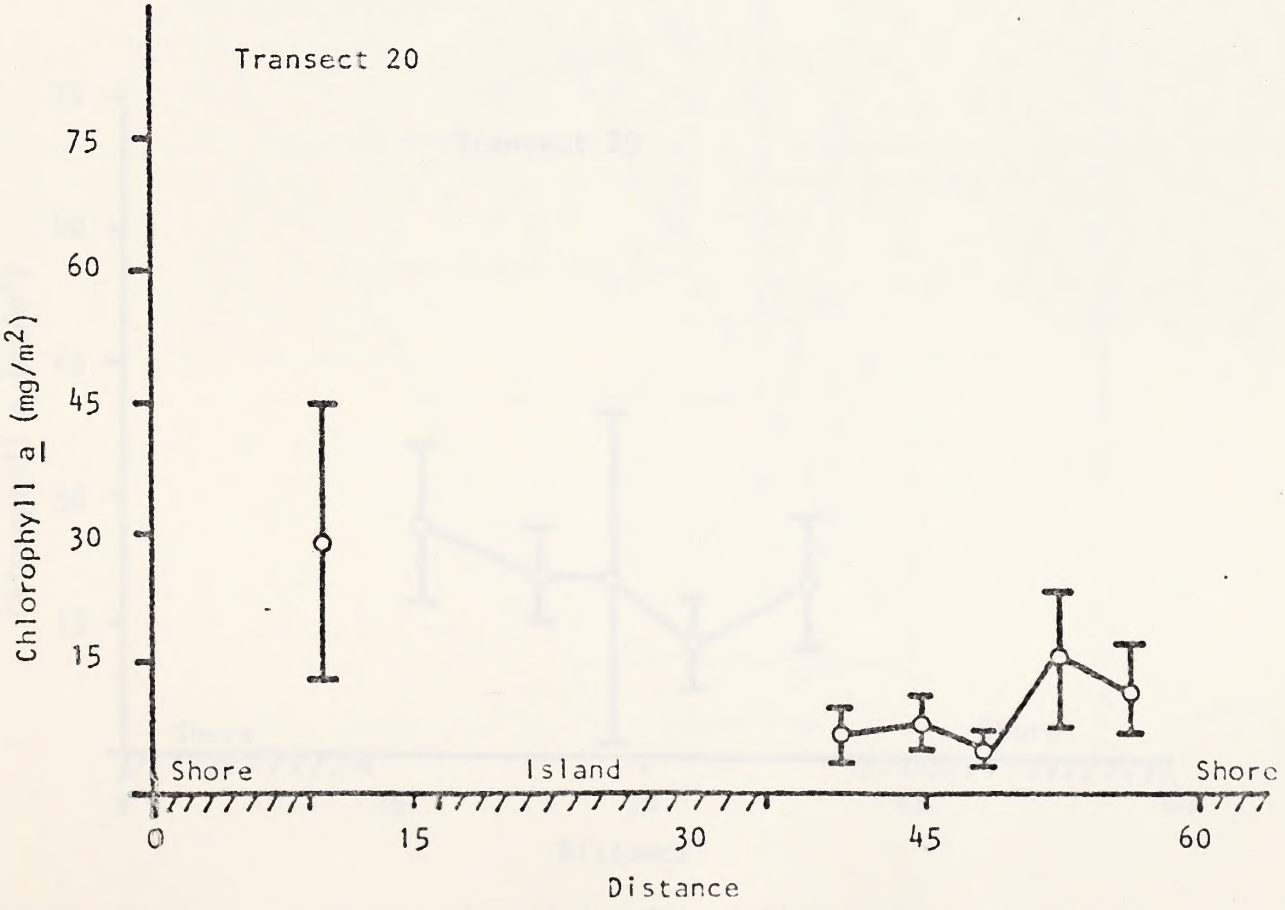
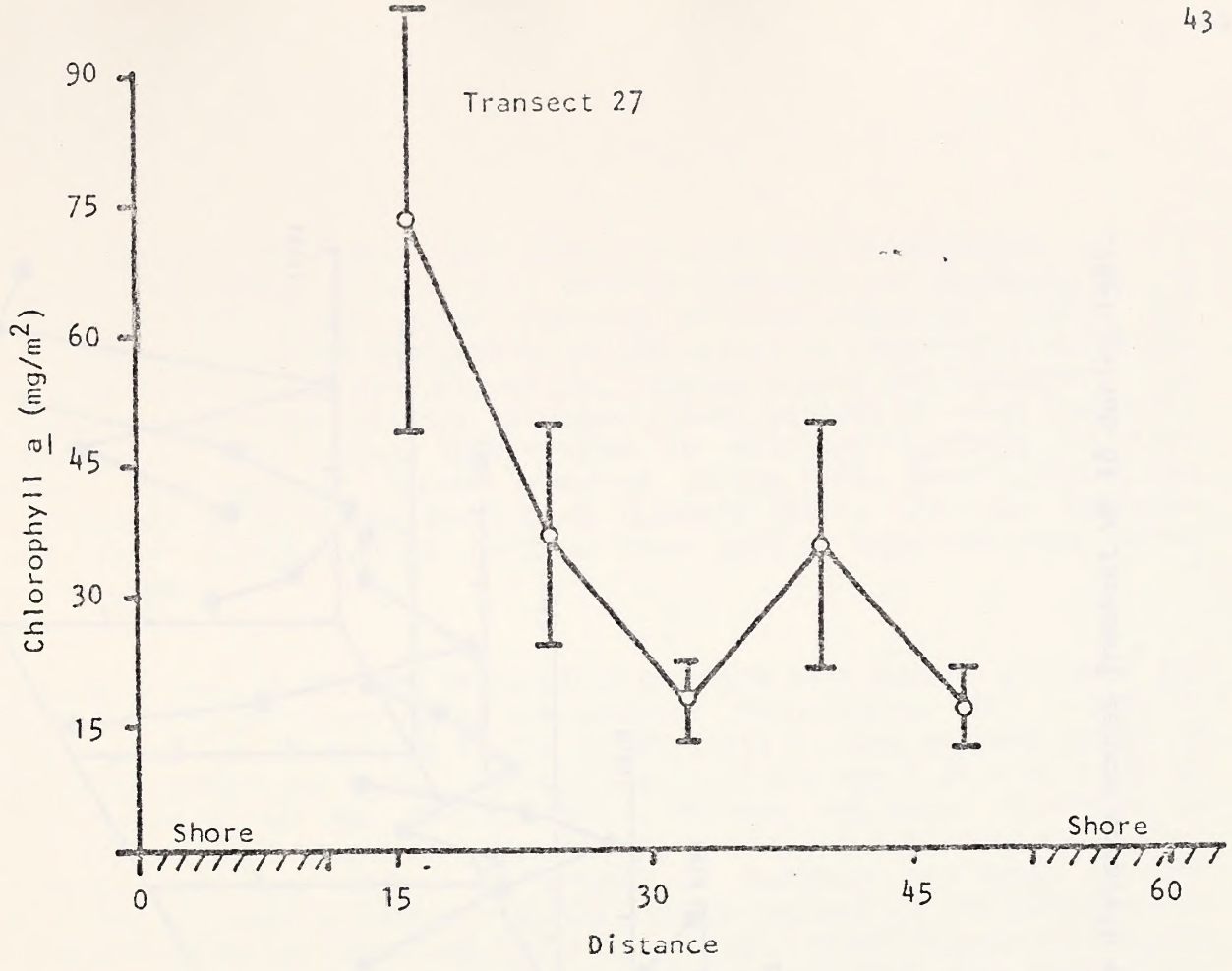


Figure 29. Mean periphyton biomass (mg Chl a/m^2) versus cross stream distance at Transect WR 18, WR 20 between April and December, 1981.



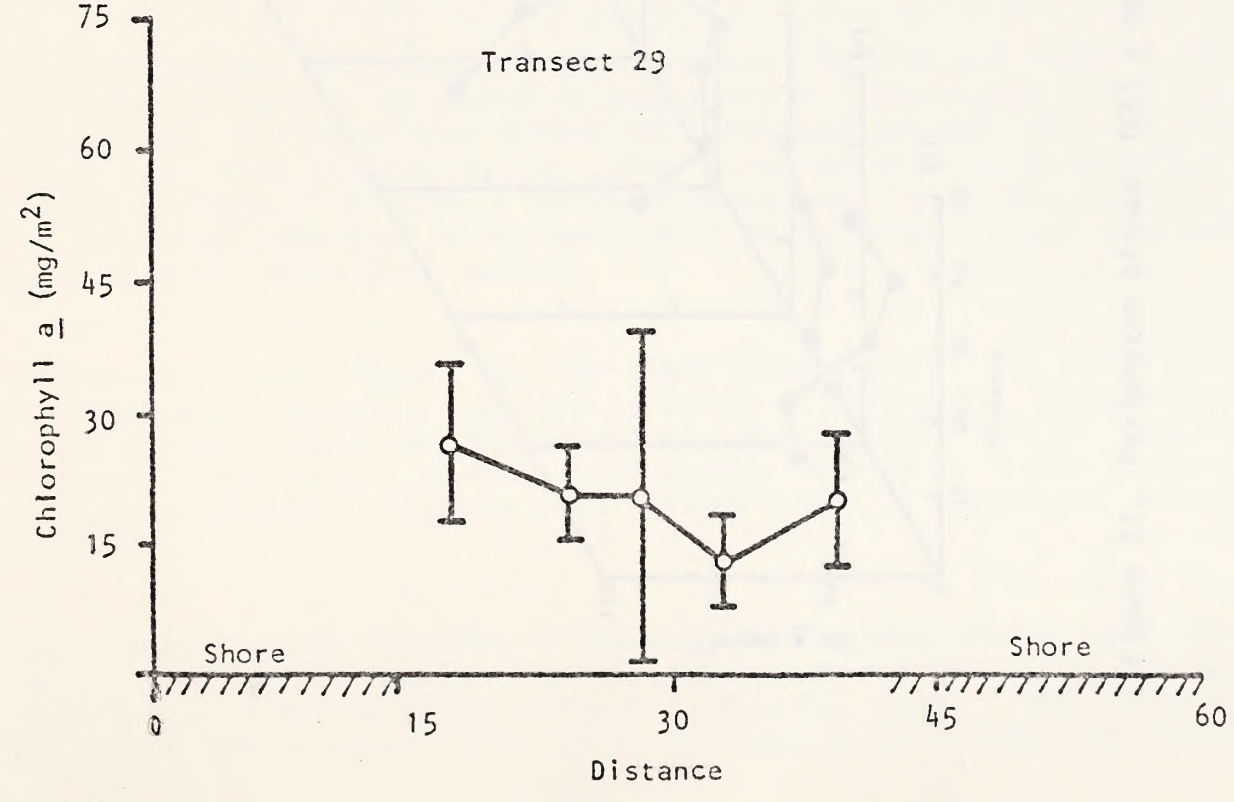


Figure 30. Mean periphyton biomass (mg Chl a/m^2) versus cross stream distance at Transect WR 27 and WR 29 between April and December, 1981.

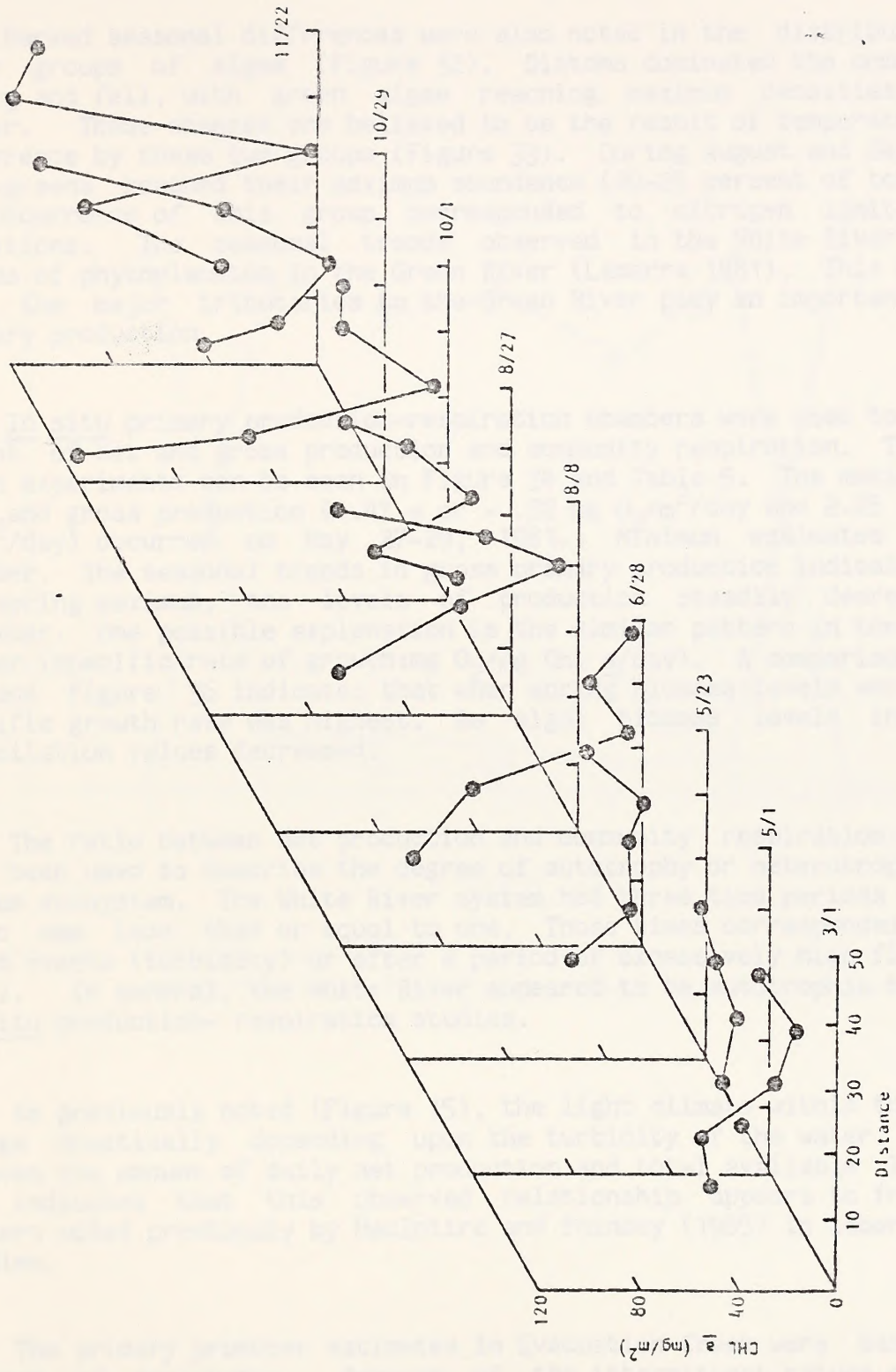


Figure 31. Periphyton biomass (Chl a mg/m^2) by distance across Transect WR 18 during 1981.

Marked seasonal differences were also noted in the distribution of the major groups of algae (Figure 32). Diatoms dominated the community in the spring and fall, with green algae reaching maximum densities during the summer. These changes are believed to be the result of temperature and light preference by these two groups (Figure 33). During August and September, the blue-greens reached their maximum abundance (20-25 percent of total biomass). The occurrence of this group corresponded to nitrogen limiting chemical conditions. The seasonal trends observed in the White River parallel the trends of phytoplankton in the Green River (Lamarra 1981). This may indicate that the major tributaries to the Green River play an important role in its primary production.

In situ primary production-respiration chambers were used to estimate the amount of net and gross production and community respiration. The results of these experiments can be seen in Figure 34 and Table 5. The maximum amount of net and gross production (1.47 + or - .22 mg O_2/m^2/day and 2.25 + or - .34 mg O_2/m^2/day) occurred on May 27-29, 1981. Minimum estimates occurred in October. The seasonal trends in gross primary production indicated that after the spring maximum, the levels of production steadily decreased through December. One possible explanation is the similar pattern in the assimilation number (specific rate of growth:mg O_2/mg Chl a/day). A comparison of Figure 35 and Figure 36 indicates that when spring biomass levels were lowest, the specific growth rate was highest. As algal biomass levels increased, the assimilation values decreased.

The ratio between net production and community respiration (Figure 36) has been used to describe the degree of autotrophy or heterotrophy within the stream ecosystem. The White River system had three time periods where the P/R ratio was less than or equal to one. Those times corresponded to extensive storm events (turbidity) or after a period of excessively high flows (October, 1981). In general, the White River appeared to be autotrophic based upon the in situ production- respiration studies.

As previously noted (Figure 15), the light climate within the river can change drastically depending upon the turbidity of the water. A comparison between the amount of daily net production and total available light (Figure 37) indicates that this observed relationship appears to follow the same pattern noted previously by MacIntire and Phinney (1965) in laboratory stream studies.

The primary producer estimates in Evacuation Creek were based upon the biomass of periphyton. Because of the intermittent nature of the stream, samples were only collected during periods of flowing water. Mean monthly biomass values are given in Figure 38 for Transects 2 and 3. Maximum values

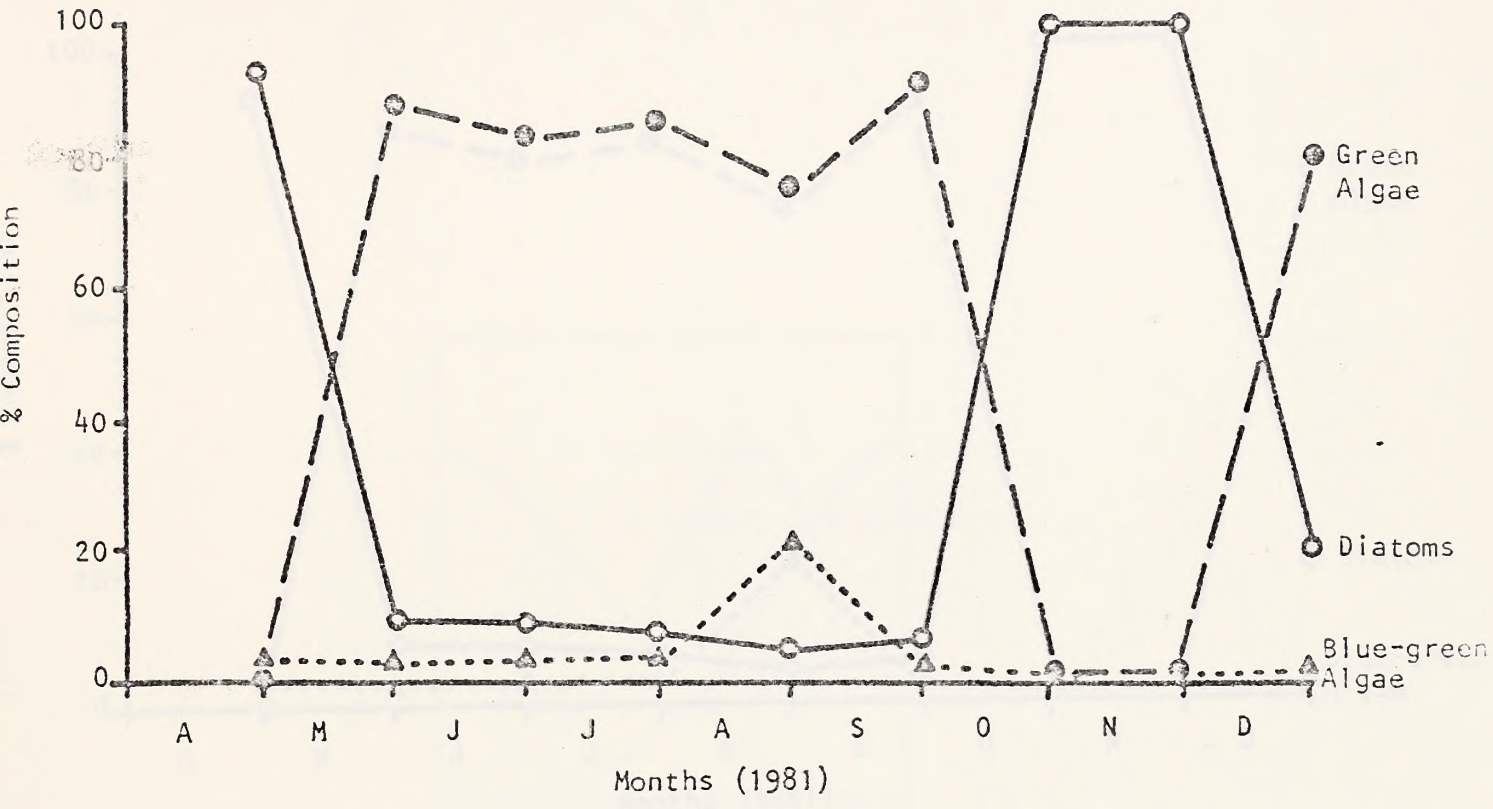


Figure 32. Percent composition of the major taxonomic groups of periphyton community based on relative volume observed in microscopic counts.

Figure 32. Percent composition of the major taxonomic groups of periphyton community based on relative volume observed in microscopic counts.

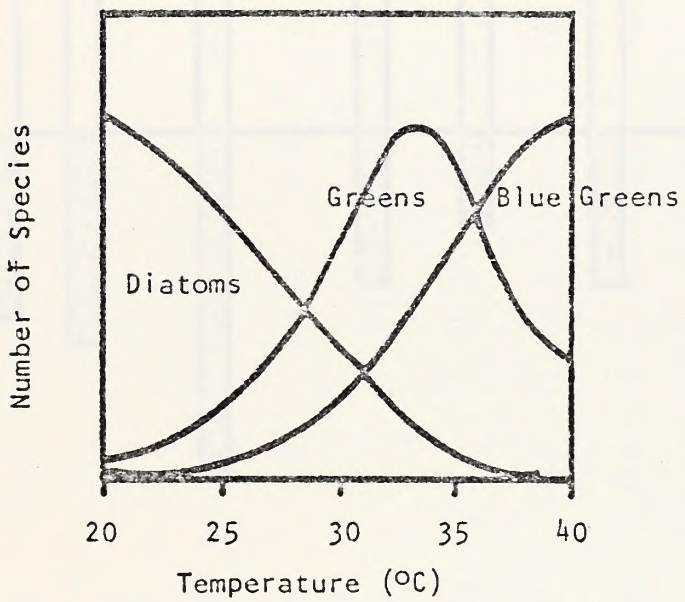


Figure 33. Qualitative shifts in algal population composition with temperature (Cairns 1970).

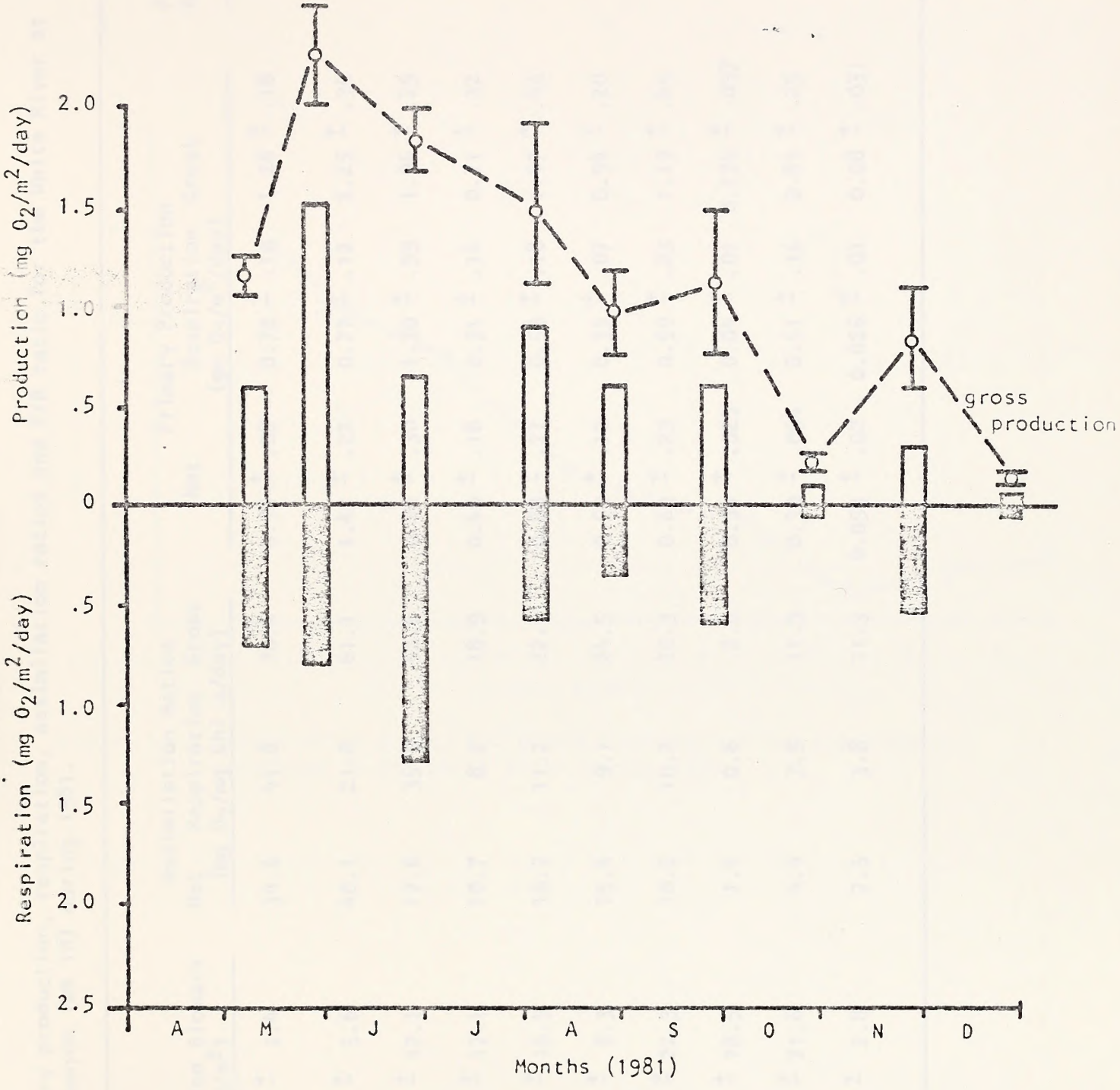


Figure 34. Gross, net production, and respiration for P/R experiments in the White River during 1981.

Table 5. The primary production, respiration, assimilation ratios and P/R ratio for the White River at Southam Canyon (WR 18) during 1981.

Date	Periphyton Biomass (mg/m^2)	Assimilation Ratios (mg O_2/mg Chl a/day) Net	Respiration	Gross	Primary Production ($gm\ O_2/m^2$/day) Net	Respiration	Gross	Net Production Respiration Ratio
5/1/81	16.7 ± 2.4	34.6	41.8	76.4	0.58 ± .08	0.70 ± .10	1.28 ± .18	$\frac{1}{1.2}$
5/27/81	36.8 ± 5.6	40.1	21.0	61.1	1.47 ± .22	0.77 ± .12	2.25 ± .34	$\frac{1.9}{1}$
6/27/81	37.6 ± 17.1	17.6	35.0	52.6	0.66 ± .30	1.30 ± .59	1.96 ± .29	$\frac{1}{1.9}$
6/29/81	37.6 ± 17.1	10.7	8.2	18.9	0.40 ± .18	0.31 ± .14	0.71 ± .32	$\frac{1.3}{1}$
8/7/81	52.8 ± 16.5	16.7	11.2	22.9	0.88 ± .27	0.59 ± .18	1.47 ± .46	$\frac{1.5}{1}$
8/26/81	38.5 ± 8.3	15.4	9.1	24.5	0.59 ± .12	0.35 ± .07	0.94 ± .20	$\frac{1.6}{1}$
9/30/81	58.9 ± 22.3	10.2	10.2	20.3	0.60 ± .23	0.59 ± .23	1.19 ± .45	$\frac{1}{1}$
10/29/81	67.3 ± 18.5	1.4	0.6	2.0	0.94 ± .025	0.04 ± .01	0.134 ± .037	$\frac{2.3}{1}$
11/22/81	68.8 ± 21.6	4.4	7.5	11.9	0.30 ± .094	0.51 ± .16	0.81 ± .25	$\frac{1}{1.7}$
12/29/81	7.1 ± 2.8	7.5	3.8	11.3	0.053 ± .02	0.026 ± .01	0.08 ± .031	$\frac{2.0}{1}$

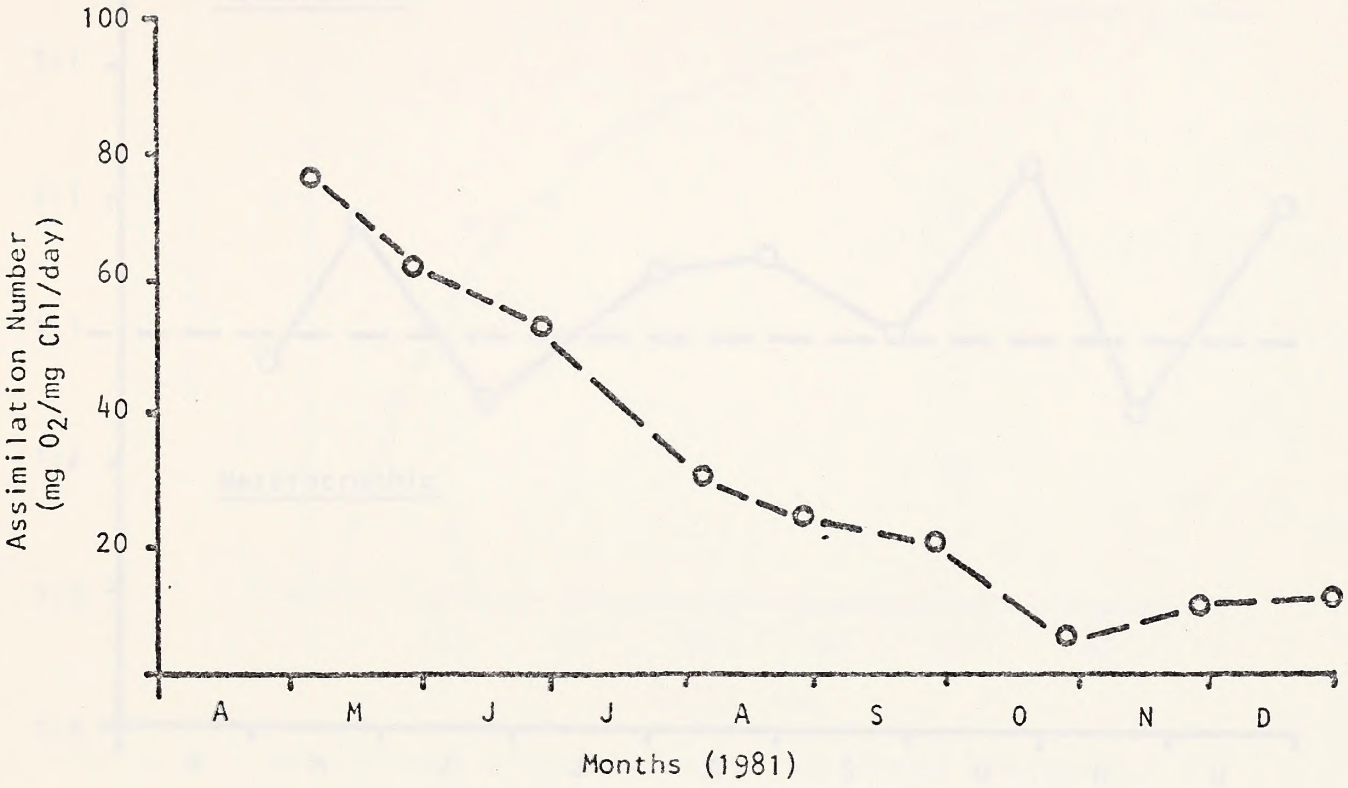


Figure 35. Assimilation number for production-respiration experiments conducted in the White River during 1981.

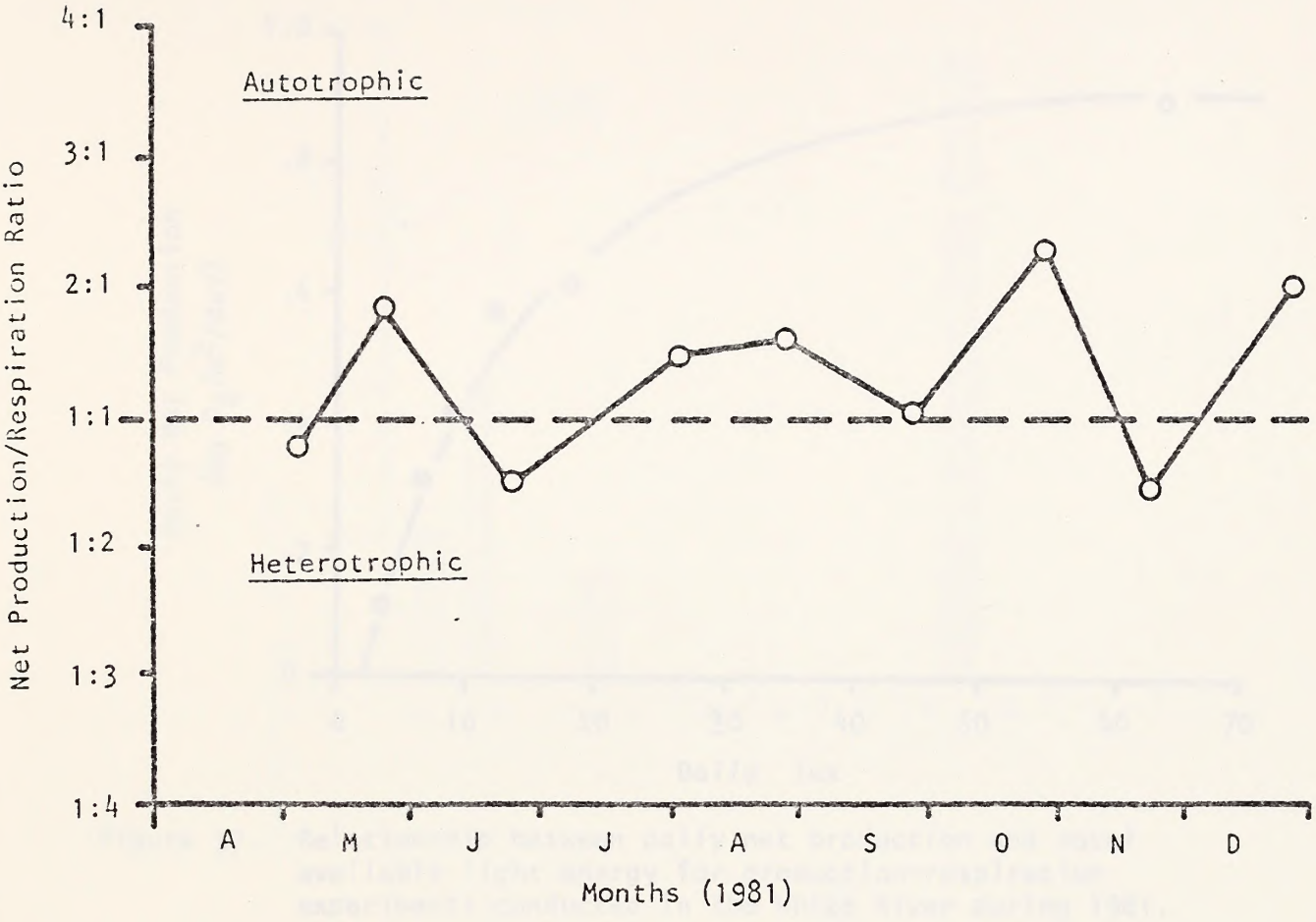


Figure 36. Net production to respiration ratio for experiments conducted in the White River during 1981.

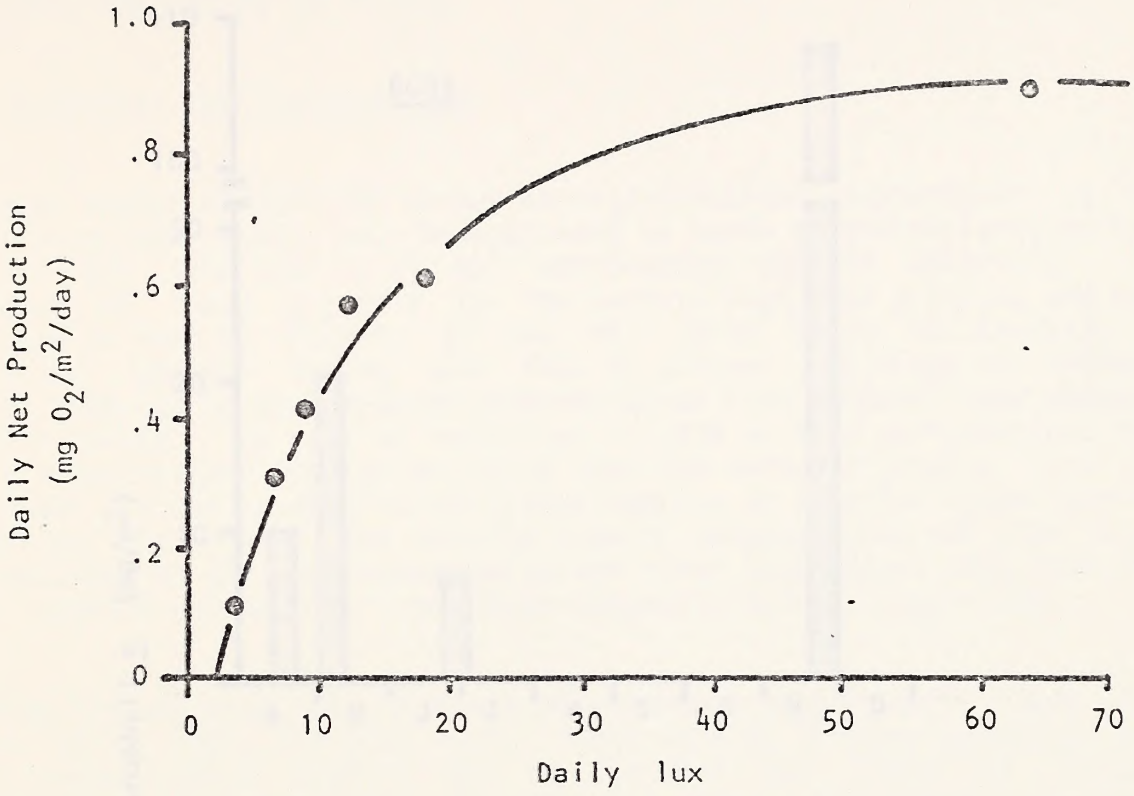


Figure 37. Relationship between daily net production and total available light energy for production-respiration experiments conducted in the White River during 1981.

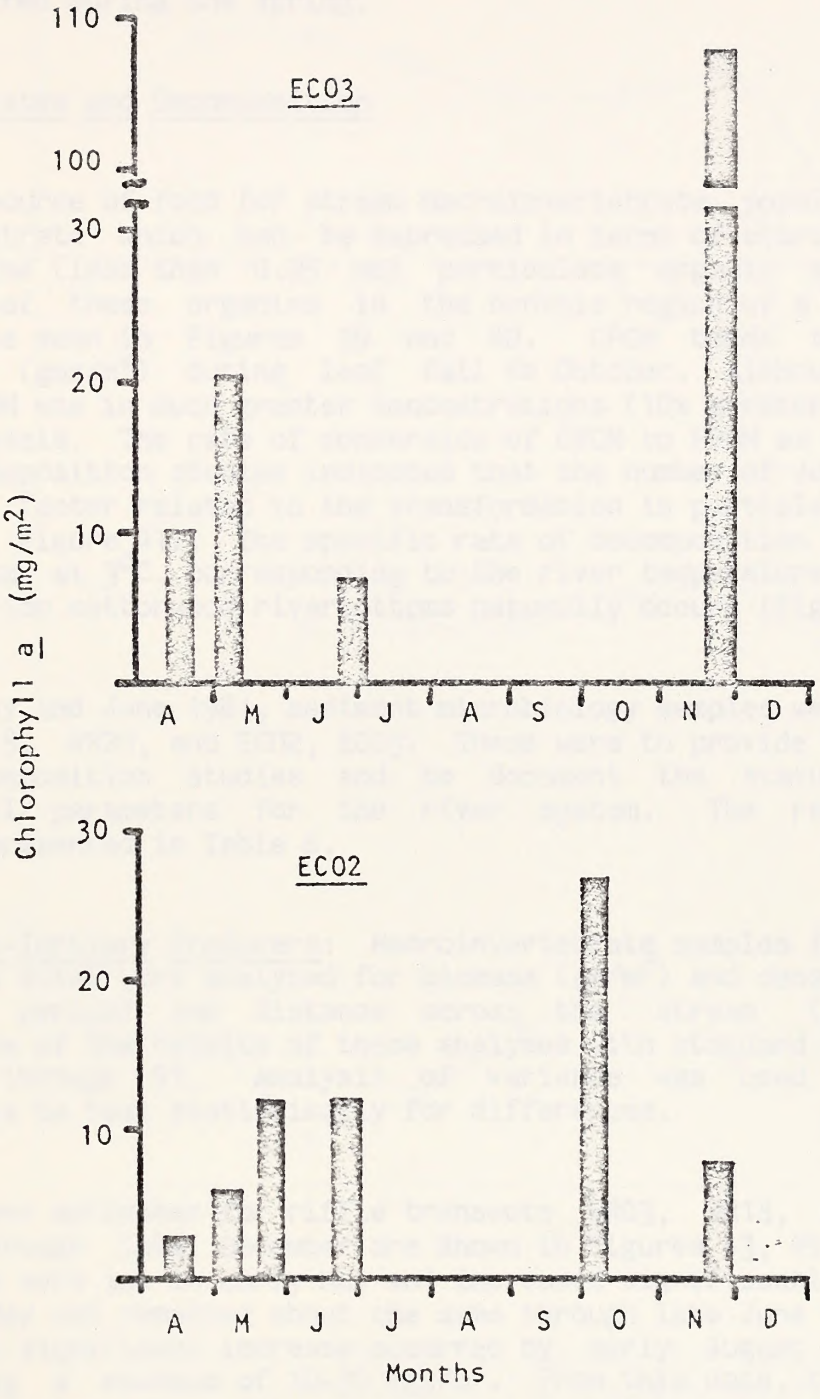


Figure 38. Mean periphyton biomass (Chl a mg/m^2) for Evacuation Creek between April and December, 1981.

occurred in November (EC03) and in October (EC02). Minimum values (excluding drought) occurred during the spring.

Organic Substrates and Decomposition

A major source of food for stream macroinvertebrate populations is the detrital substrate which can be expressed in terms of coarse (greater then 0.25 mm) or fine (less than 0.25 mm) particulate organic materials. The distribution of these organics in the benthic region of a riffle and pool transect can be seen in Figures 39 and 40. CPOM tends to increase in concentration (gms/m^2) during leaf fall in October. Although not changing sesonally, FPOM was in much greater concentrations (10x greater) when compared to the CPOM levels. The rate of conversion of CPOM to FPOM as determined from leaf pack decomposition studies indicated that the number of degree days was an important factor related to the transformation in particle sizes (percent decomposition, Figure 41). The specific rate of decomposition was also found to be maximum at $3^{o}C$, corresponding to the river temperature when leaf fall from the riparian cottonwood riverbottoms naturally occurs (Figure 42).

During May and June 1981, sediment microbiology samples were taken from Transects WR18, WR20, and EC02, EC03. These were to provide background data for the decomposition studies and to document the status of certain microbiological parameters for the river system. The results of these analyses are presented in Table 6.

Secondary-Tertiary Producers: Macroinvertebrate samples from the above, on, and below sites were analyzed for biomass (gm/m^2) and density ($\#'s/m^2$) by time (sample period) and distance across the stream (meters). Log transformations of the results of these analyses with standard error are shown in Figures 43 through 51. Analysis of variance was used on these log transformations to test statistically for differences.

The biomass estimates for riffle transects WR03, WR18, and WR27 from early May through late December are shown in Figures 43, 45, and 47. Mean biomass levels were low in early May and increased significantly ($p < .01$) by late May and remained about the same through late June ($p > .05$). Another significant increase occurred by early August ($p < .01$), reaching a maximum of 10-30 mgs/m^2. From this date, biomass declined significantly ($p < .01$) each month through late October and fluctuated without significant difference for the remaining two sample periods.

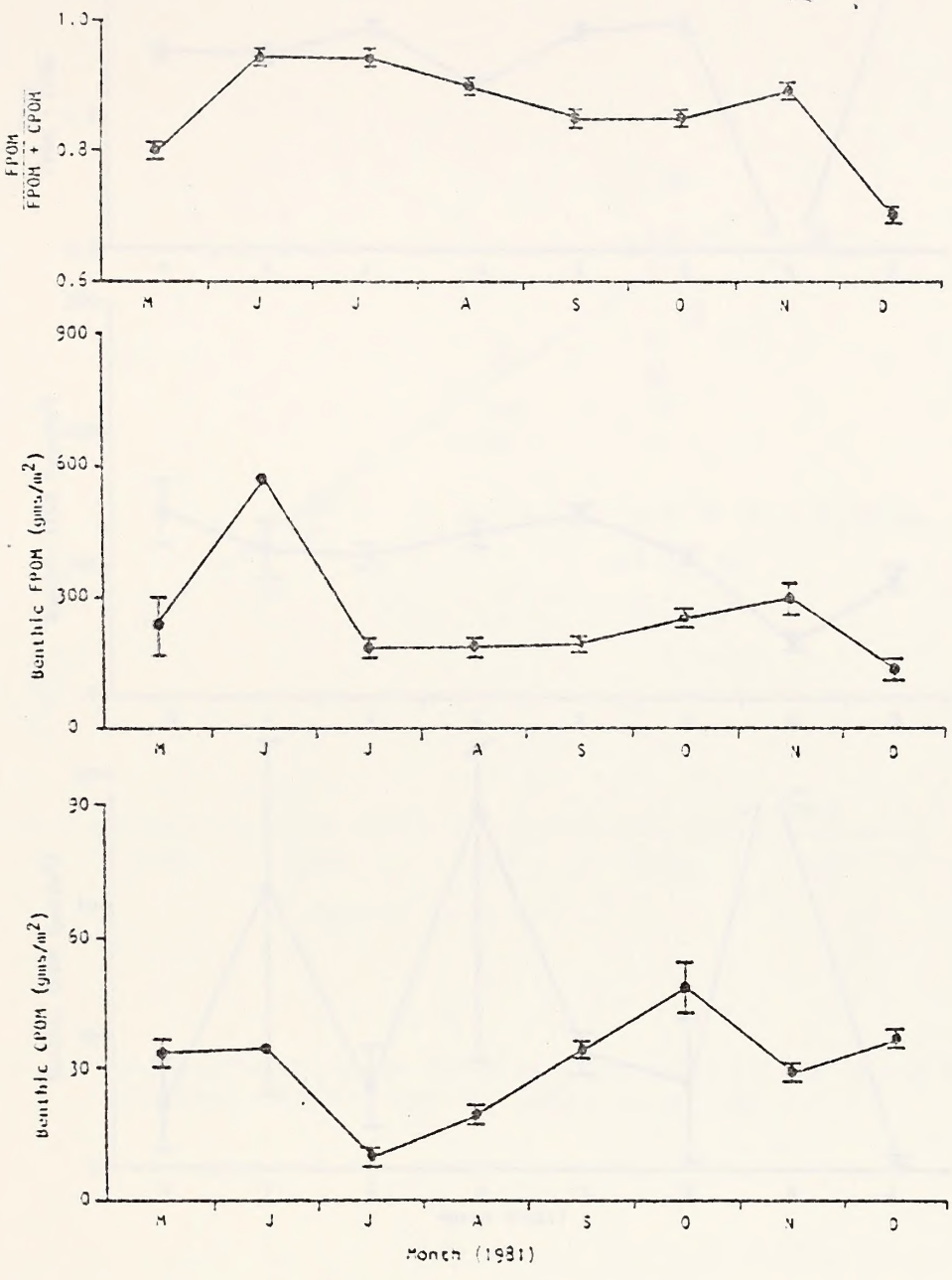


Figure 39. The monthly changes in benthic CPOM (gms/m^2) and FPOM (gms/m2) at a riffle transect (km 79) in the White River.

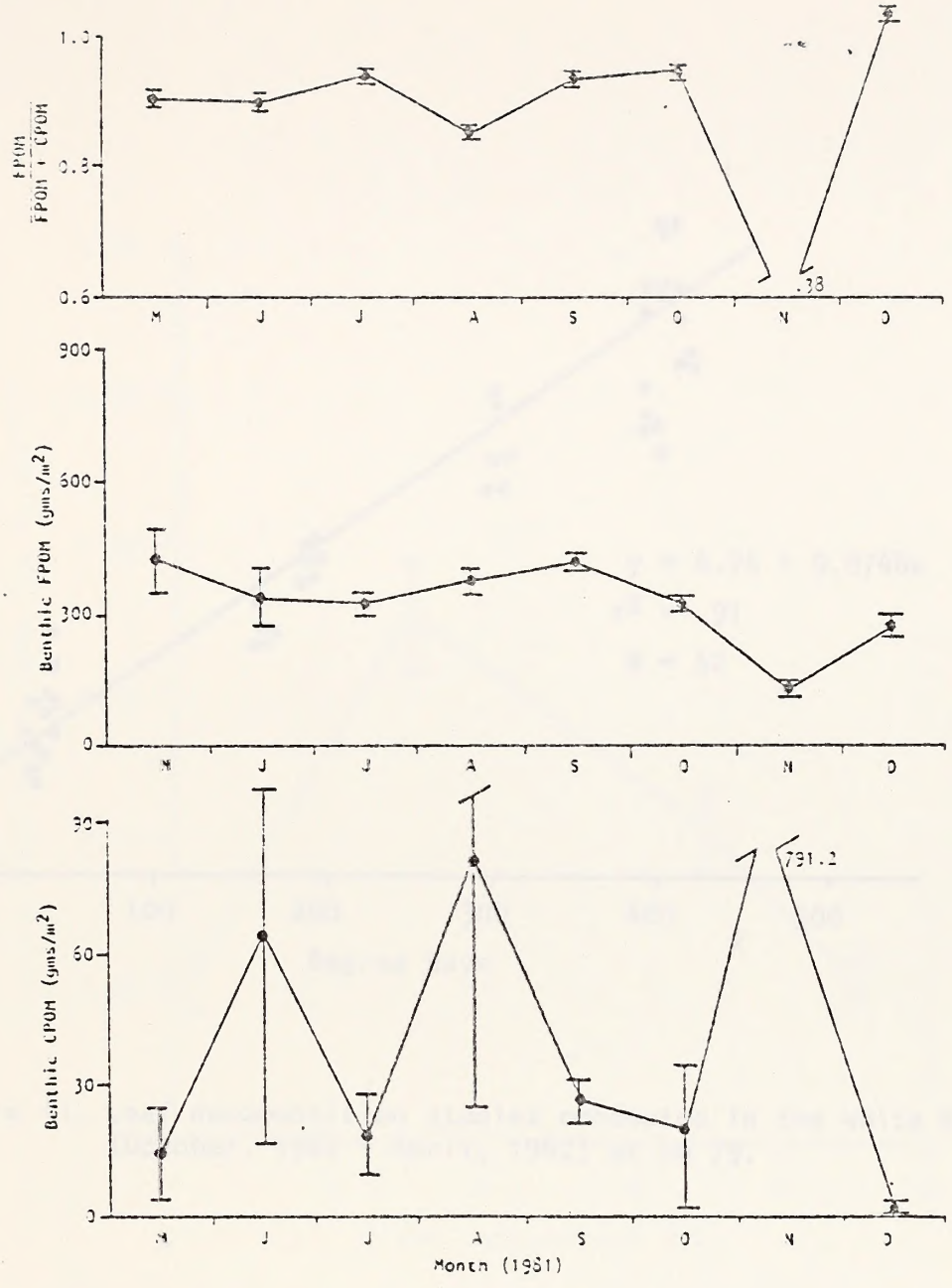


Figure 40. The monthly changes in benthic CPOM (gms/m^2) and FPOM (gms/m^2) at a pool transect (km 79) in the White River.

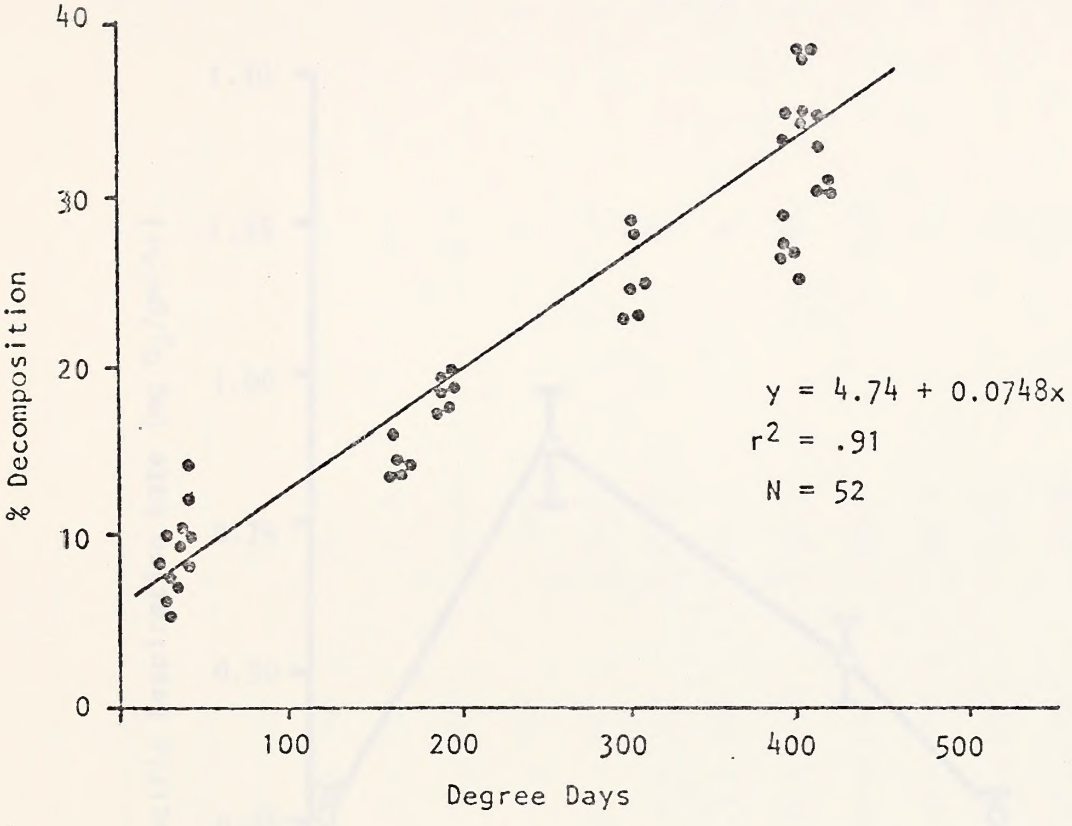


Figure 41. Leaf decomposition studies conducted in the White River (October, 1981 - April, 1982) at km 79.

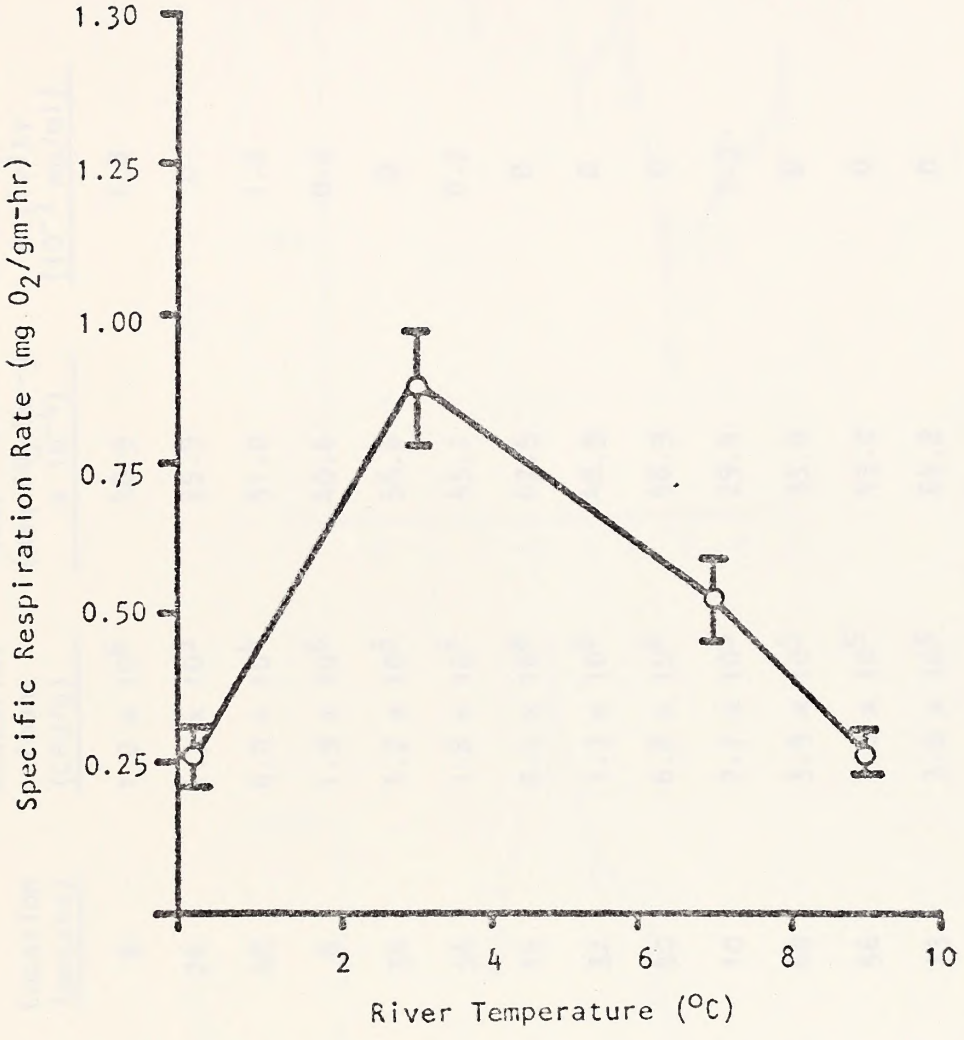


Figure 42. Specific respiration rate (mg O_2/gm-hr) of decomposing leafpacks as a function of river temperature.

Table 6. Microbiology samples collected at the intensive sites (WR 18, WR 20) on the White River during 1981.

Date	Transect	Location (meters)	Bacterial Count (CFU/g)	Respiration (meq CO_2/hr-g $\times 10^{-4}$)	Dehydrogenase Activity (10^{-3} mg/ml)	Organic Carbon (%)	Organic Nitrogen (%)
May 1, 1981	WR18	8	1.0×10^6	48.9	1.8	0.47	0.050
		24	1.3×10^5	25.9	0	0.25	0.028
		48	6.0×10^4	51.0	1.2	0.23	0.007
	WR20	8	1.9×10^6	40.6	0.2	0.61	0.035
		34	1.7×10^5	56.6	0	0.52	0.027
		56	1.8×10^5	45.1	0.2	0.84	0.029
June 30, 1981	WR18	16	2.4×10^6	67.5	0	0.33	0.018
		32	1.3×10^6	48.9	0	0.36	0.031
		40	6.8×10^6	48.9	0	0.31	0.039
	WR20	10	7.7×10^5	29.4	7.3	0.68	0.048
		48	3.9×10^5	35.0	0	0.40	0.011
		56	1.8×10^5	43.0	0	0.28	0.023
	EC02	13	3.0×10^5	64.8	0	0.52	0.036
	EC03	6.5	1.3×10^6	33.3	0	0.40	0.027

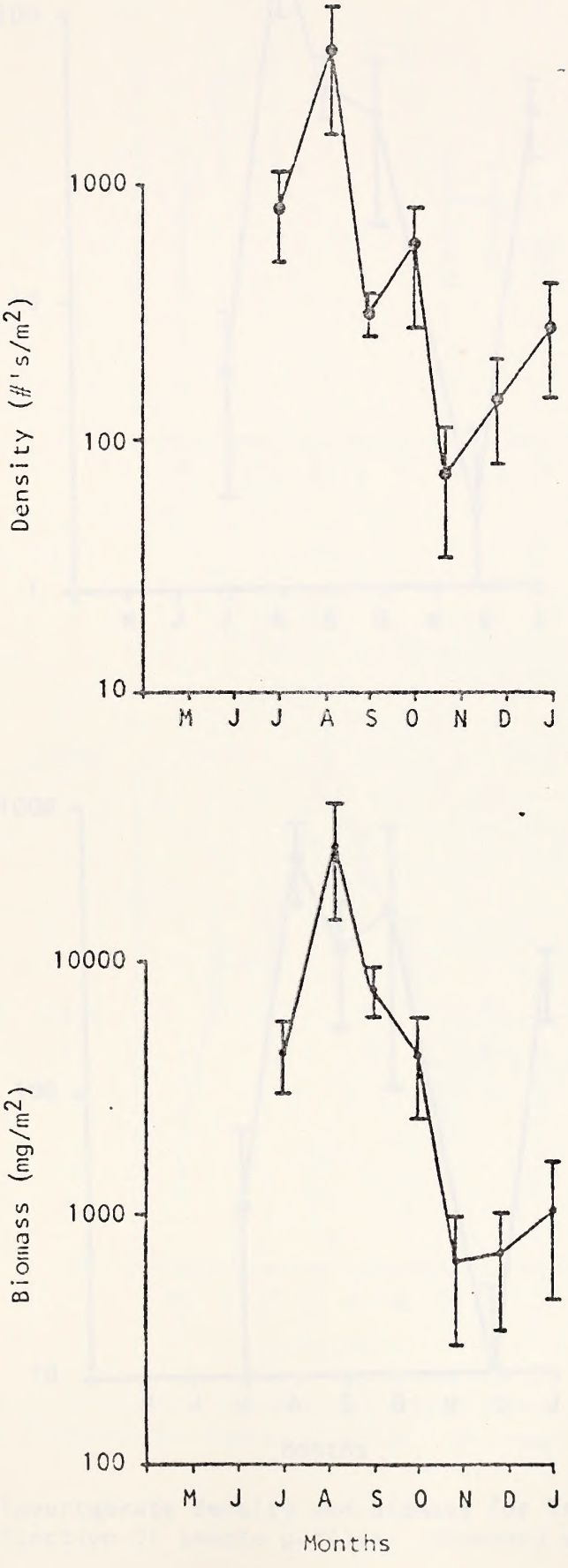


Figure 43. Macroinvertebrate density and biomass for Transect WR 03 as a function of sample periods. Standard error bars shown.

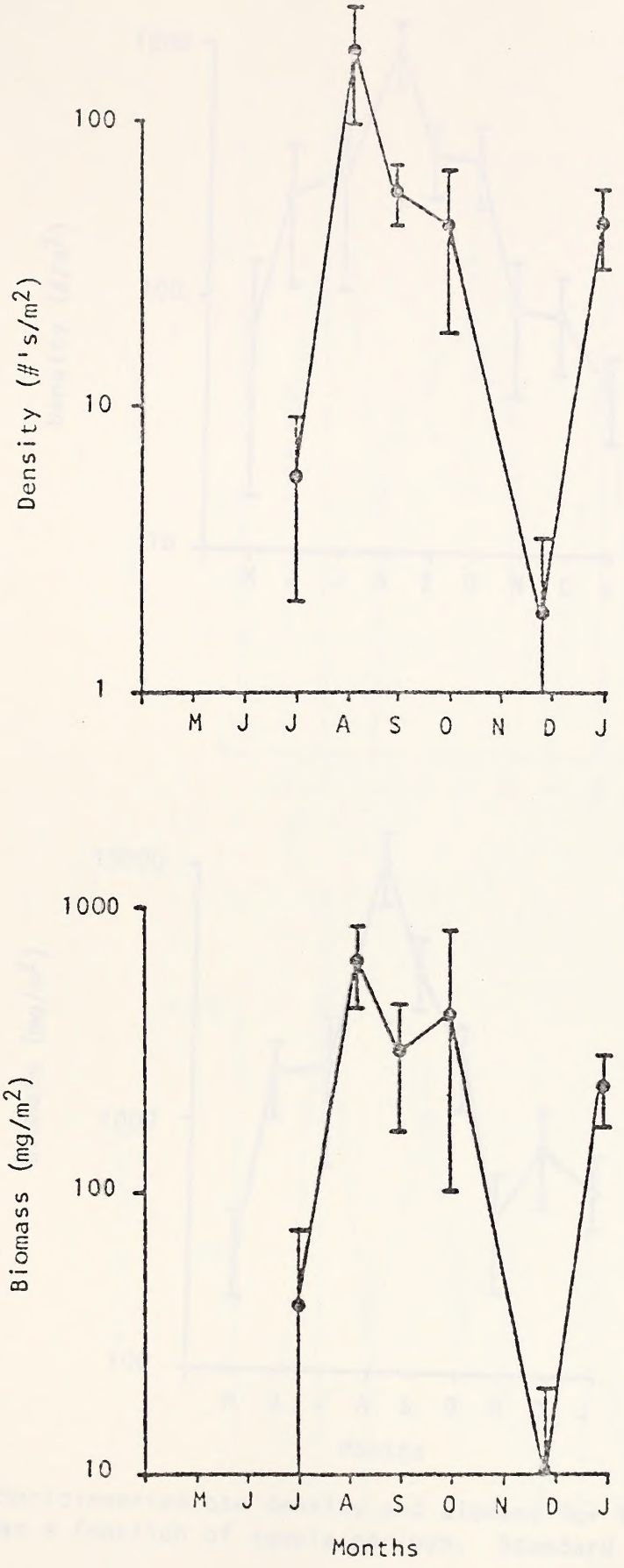


Figure 44. Macroinvertebrate density and biomass for Transect WR 05 as a function of sample periods. Standard error bars shown.

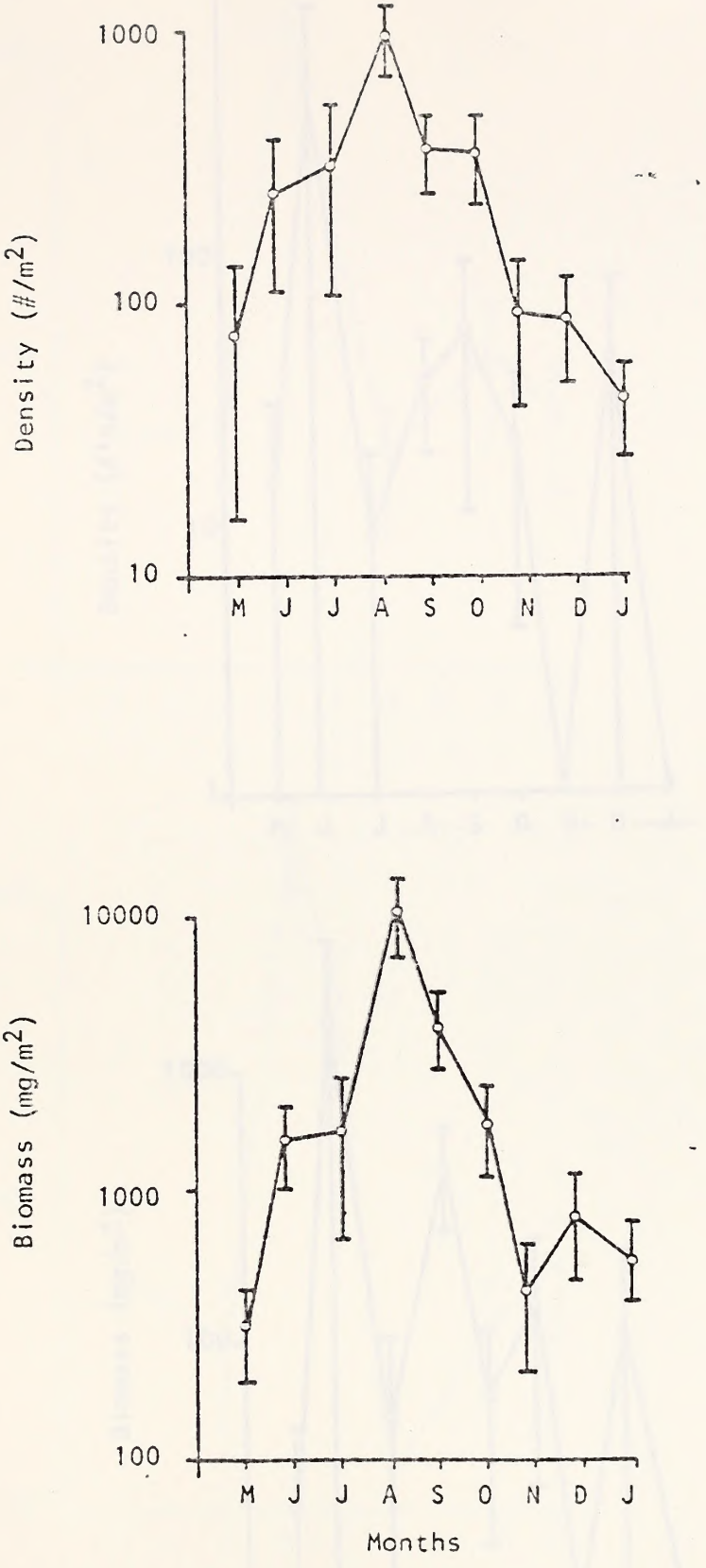


Figure 45. Macroinvertebrate density and biomass for Transect WR 18 as a function of sample periods. Standard error bars shown.

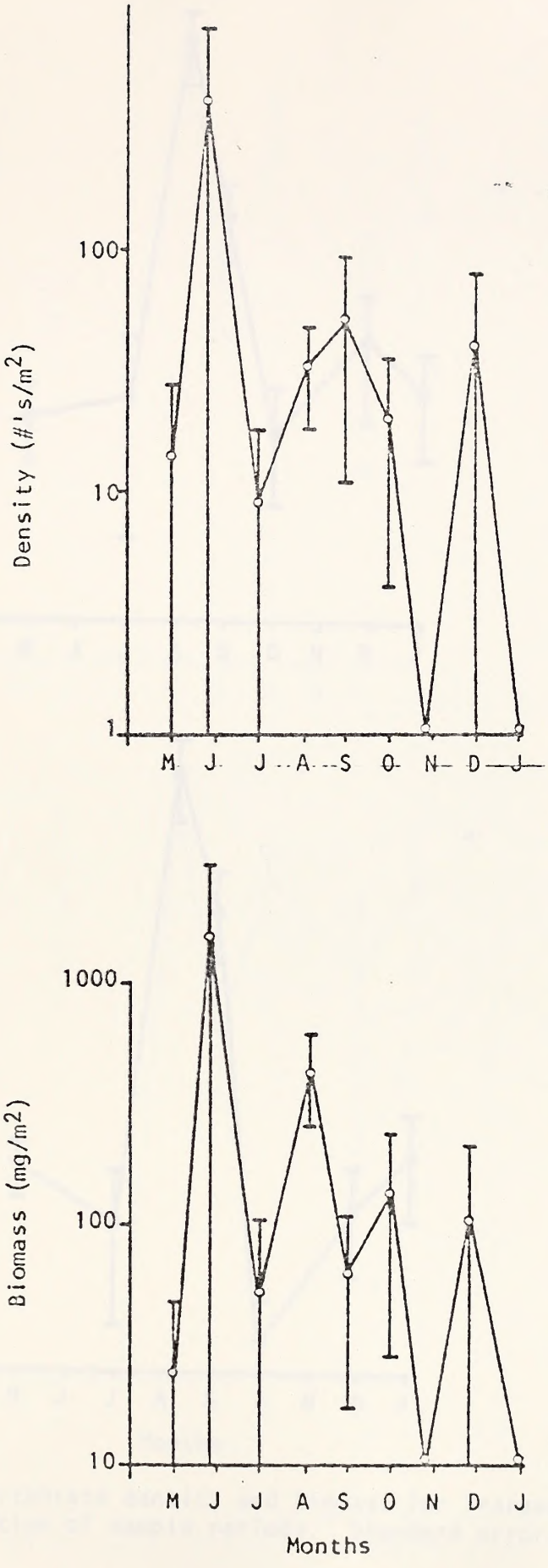


Figure 46. Macroinvertebrate density and biomass for Transect WR 20 as a function of sample periods. Standard error bars shown.

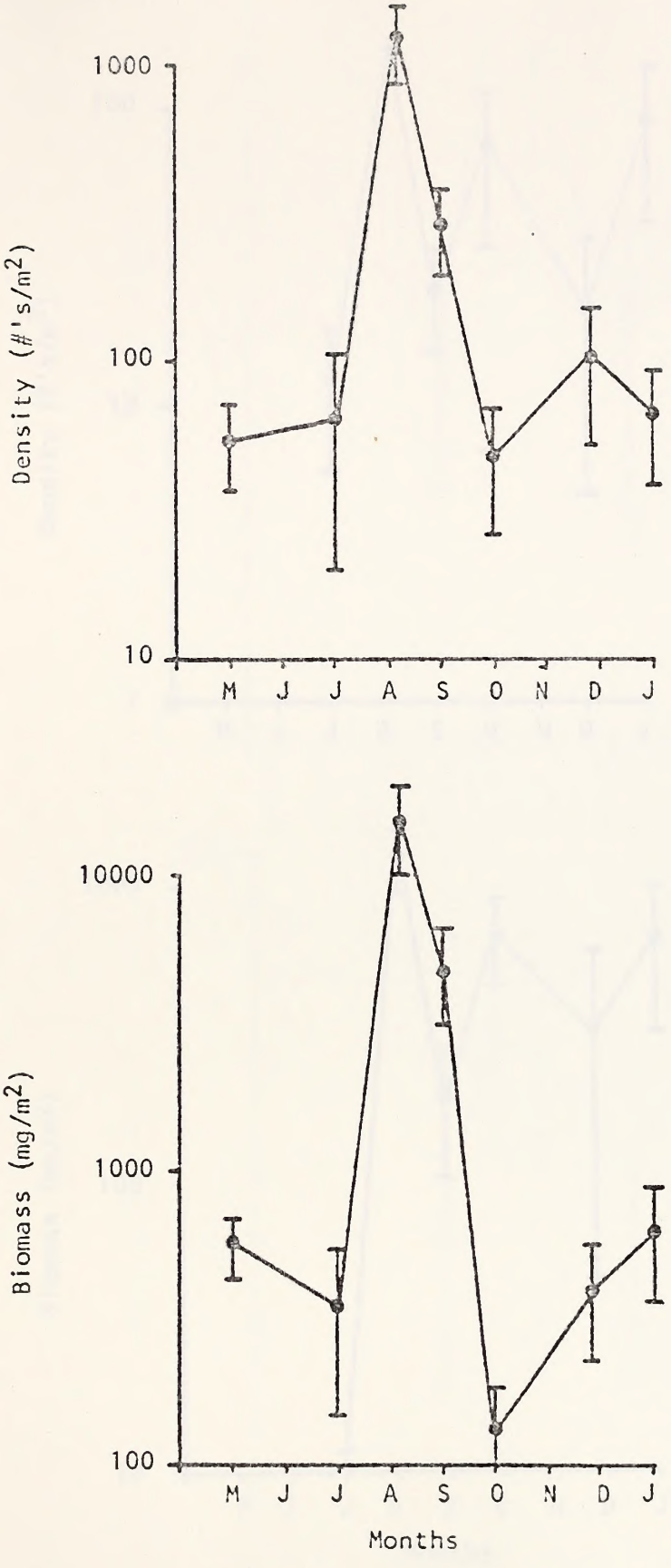


Figure 47. Macroinvertebrate density and biomass for Transect WR 27 as a function of sample periods. Standard error bars shown.

Figure 48. Macroinvertebrate density and biomass for Transect WR 29 as a function of sample periods. Standard error bars shown.

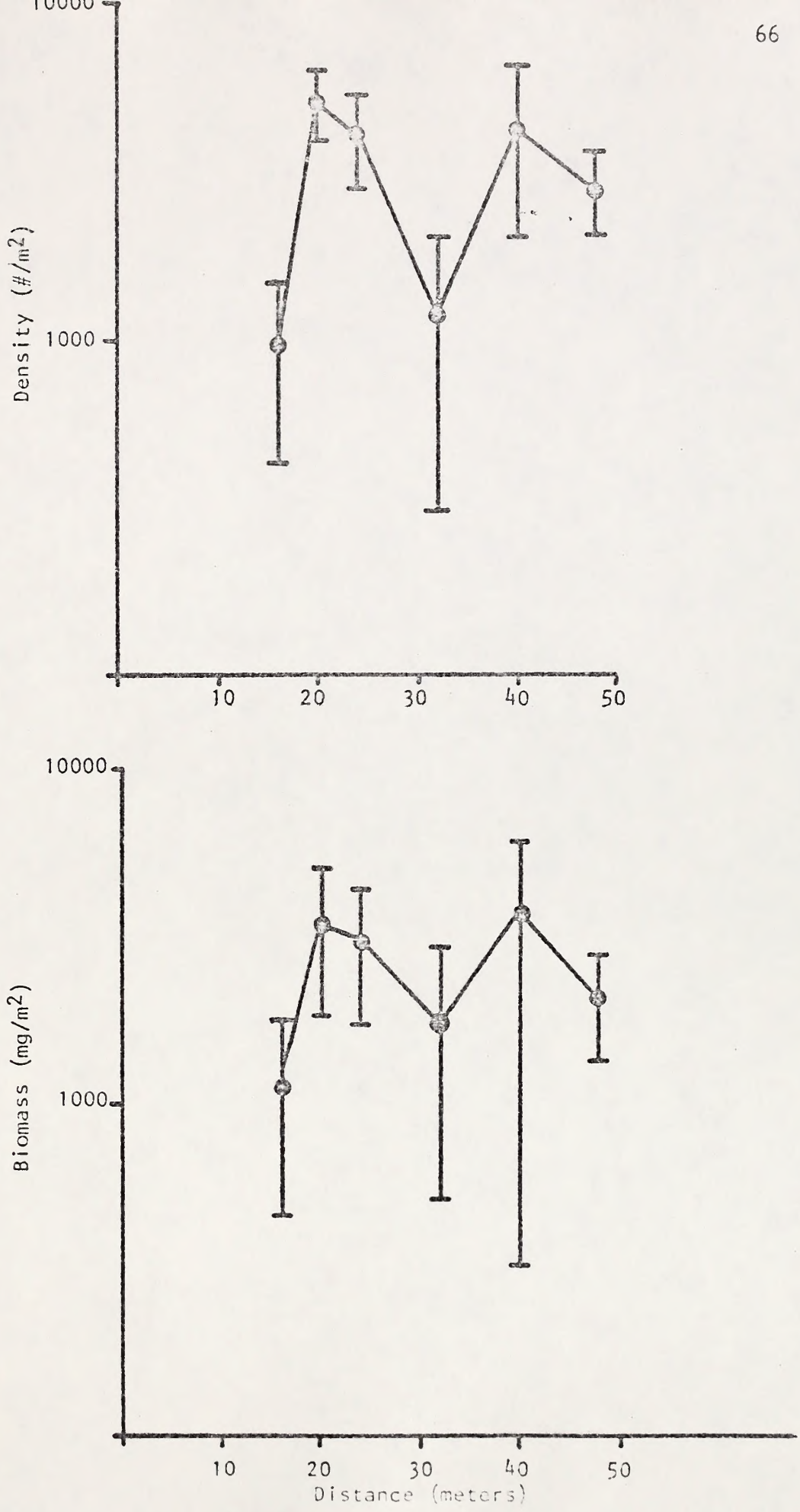


Figure 49. Macroinvertebrate density and biomass as a function of distance across the river for Transect WR 18 between April and December, 1981.

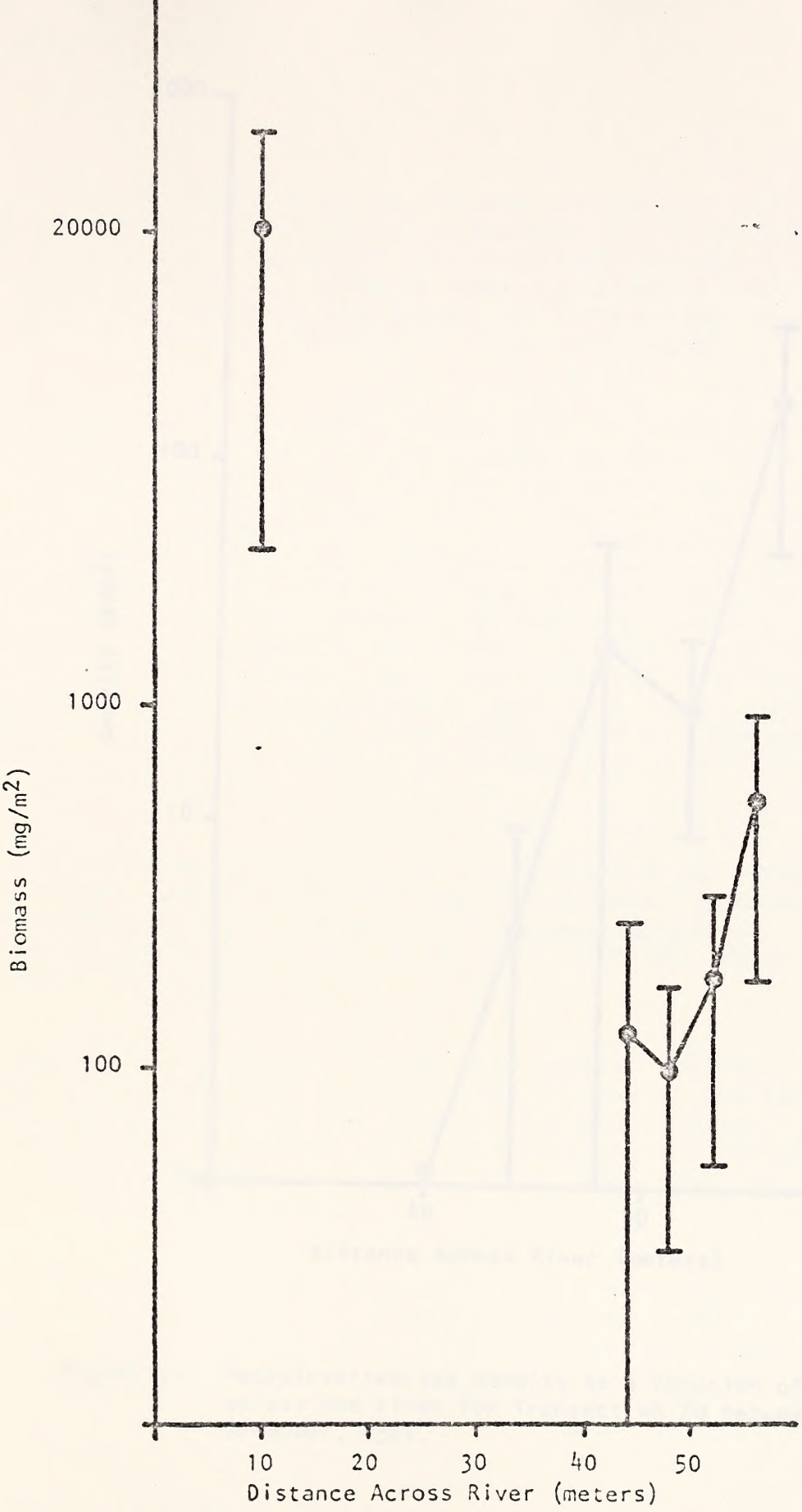


Figure 50. Macroinvertebrate biomass as a function of distance across the river for Transect WR 20 between April and December, 1981.

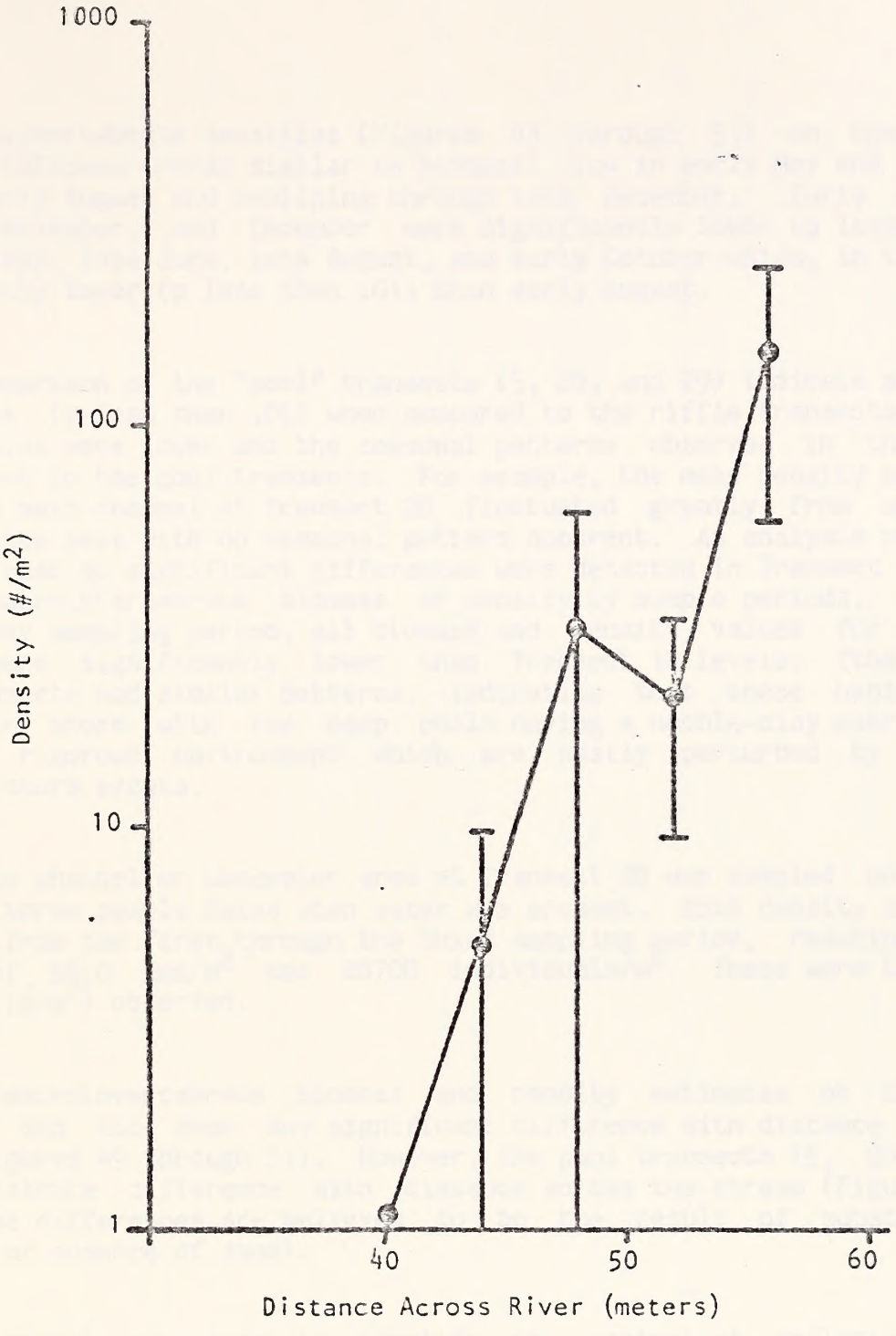


Figure 51. Macroinvertebrate density as a function of distance across the river for Transect WR 20 between April and December, 1981.

Macroinvertebrate densities (Figures 43 through 51) on these riffle transects followed trends similar to biomass: low in early May and reaching a peak in early August and declining through late December. Early May, late October, November, and December were significantly lower (p less than .01) than late May, late June, late August, and early October which, in turn, were significantly lower (p less than .01) than early August.

A comparison of the "pool" transects (5, 20, and 29) indicate significant differences (p less than .05) when compared to the riffle transects. Biomass and densities were lower and the seasonal patterns observed in the riffles were absent in the pool transects. For example, the mean density and biomass within the main channel of Transect 20 fluctuated greatly from one sample period to the next with no seasonal pattern apparent. An analysis of variance indicates that no significant differences were detected in Transect 20's main channel macroinvertebrate biomass or density by sample periods. Except for the late May sampling period, all biomass and density values for the main channel were significantly lower than Transect 18 levels. Tthe above and below transects had similar patterns, indicating that these habitat types (sand near shore with the deep pools having a cobble-clay matrix) are an extremely rigorous environment which are easily perturbed by naturally occurring storm events.

A side channel or backwater area at Transect 20 was sampled only during the first three sample dates when water was present. Both density and biomass increased from the first through the third sampling period, reaching a June maximum of 56.0 gms/m^2 and 26700 individuals/m^2. These were the highest densities (#/m^2) observed.

The macroinvertebrate biomass and density estimates on the riffle transects did not show any significant difference with distance across the stream (Figures 49 through 51). However, the pool transects (5, 20, and 29 did demonstrate difference with distance across the stream (Figures 50 and 51). These differences are believed to be the result of substrate type (presence or absence of sand).

An attempt was made to simplify the ecological analysis of the macroinvertebrate community by assigning each species into a functional group. Four major groups and seven subdivisions of these four groups further refined the ecological functions of the macroinvertebrates present within the White River. Descriptions of these groups are shown in Table 7. This system was modified from Cummins (1978) with a greater emphasis on the trophic relationships rather than on feeding mechanisms. A list of all species or groups collected, numbers collected at each site, and the functional group assignments are shown in Table 8.

Table 7. General classification system for aquatic insect trophic relations (applicable only to immature and adult stages that occur in water). Modified from Cummins (1978).

Functional Group	Subdivision of Functional Group: Dominant Food	Subdivision of Functional Group: Feeding Mechanism	Representative Types from the White River
Detritivores	Decomposing vascular plant tissue - coarse particulate organic matter (CPOM)	Chewers and wood borers	Limnephilus, Tipula
	Decomposing fine particulate organic matter (FPOM)	Gatherers or deposit (sediment) feeders (includes surface film feeders)	Tubiflex, Ephemerella, Elmidae, Chironomidae, Notophila
Herbivores	Periphyton - attached algae and associated material	Scrapers of mineral and organic surfaces	Rhithrogena, Parargyractis
	Diatoms, microscopic algae, and algal fragments	Gatherers from surfaces and deposits (sediments)	Baetis, Tricorythodes, Hydroptila
	Vascular hydrophytes	Chewers and piercers of plant tissues	Not found in the study area. 15 specimens (Trichoptera, Leptoceridae) collected elsewhere on the White River
Filterers (Omnivores)	Detritivores, herbivores, omnivores	Filters suspended materials from the current	Traverella, Hydropsyche, Brachycentrus, Simulidae
Predators	Living animal tissue	Engulfers and piercers of animal tissue	Gomphus, Isogenus, Notonectidae, Dytiscidae

Table 8. A list of all species or groups of macroinvertebrates collected on the White River and Evacuation Creek, Utah from May - December, 1981, with number collected at each sample site and functional group assignment.

Species or Group	Benthic Samples # Collected per Transect EC02	EC03	WR01	WR02	WR03	WR05	WR06	WR18	WR20	WR27	WR28	WR29	WR90	WR98	Drift	Total	Functional Group	References*
Nematoda								1								1	Detritivore (FPOM)	15
Annelida																		
Oligochaeta																		
Tubifex tubifex	33				8	2		45		3		22	32			145	Detritivore (FPOM)	15
Terrestrial sp.															1	1	Terrestrial	15
Arachnida																		
Hydracrina					1			2							1	4	Predator	15
Terrestrial sp.					1							1			32	34	Terrestrial	15
Insecta																		
Collembola																		
Isotomurus sp.					1			1							6	8	Detritivore (FPOM)	12, 15
Ephemeroptera																		
Siphlonuridae																		
Analetris exima														1		1	Predator	6, 15
Isonychia sicca					8				4						138	150	Filterer	4, 5, 6, 8, 12, 15, 16
Baetidae																		
Baetis sp.	1			2	239	18		190	3	65		3			1499	2020	Herbivore (Diatoms)	3, 5, 6, 8, 10, 12, 13, 14, 15, 16, 17

Table 8. (Continued).

Species or Group	ECO2	ECO3	WR01	WR02	WR03	WR05	WR06	WR18	WR20	WR27	WR28	WR29	WR90	WR98	Drift	Total	Functional Group	References*
Oligoneuriidae																		
Lachlaria sp.					5			6		1					384	396	Filterer	6, 12, 15
Heptageniidae																		
Rhithrogena sp.					39	3		56	3	49		13		3	333	499	Herbivore (Periphyton)	5, 6, 8, 10, 12, 15, 16
Ephemerellaiidae																		
Ephemerella mermis				3	2			20		2				1	583	611	Detritivore (FPOM)	2, 3, 5, 6, 8, 10, 12, 13, 15, 16, 19
Tricorythidae																		
Tricorythodes sp.					75	51		105	4	15		35			485	770	Herbivore (Diatoms)	6, 8, 9, 16
Leptophlebiidae																		
Paraleptophlebia sp.					1	3		31		58		5			19	117	Detritivore (FPOM)	3, 4, 5, 6, 8, 12, 15, 16
Traverella albertana					734	9		573	2	611		8			2056	3993	Filterer	1, 5, 6, 12, 15
Polymitarcyidae																		
Ephoron album					1	1						2			37	41	Detritivore (FPOM)	6, 12, 15, 16
Odonata																		
Anisoptera																		
Gomphidae					5			1				6		2	89	103	Predator	12, 15
Libellulidae					1											1	Predator	12, 15
Zygoptera																		
Lestidae															20	20	Predator	12, 15
Orthoptera															6	6	Terrestrial	

Table 8. (Continued).

Species or Group	EC02	EC03	WR01	WR02	WR03	WR05	WR06	WR18	WR20	WR27	WR28	WR29	WR90	WR98	Drift	Total	Functional Group	References*
Isoptera					1											1	Terrestrial	
Plecoptera																		
Perlidae																		
Acroneuria sp.								1								1	Predator	3, 4, 12, 15
Perlodidae																		
Isogenus sp.				1	20	12		65	1	31		19		9	304	462	Predator	12, 13, 14, 15
Chloroperlidae																		
Kathroperla perdita					2					1					1	4	Detritivore (FPOM)	3, 12, 15
Thysanoptera																		
Thripidae										1					37	38	Terrestrial	
Hemiptera																		
Veliidae																		
Rhagovelia sp.								1								1	Predator	12, 15
Gerridae															1	1	Predator	12, 15
Naucoridae															1	1	Predator	12, 15
Corixidae															4	4	Predator	12, 15
Notonectidae															23	23	Predator	12, 15
Terrestrial sp.															1	1	Terrestrial	
Homoptera																		
Cicadellidae									1	1					90	92	Terrestrial	
Aphididae	1							1							3	5	Terrestrial	
Others															1	1	Terrestrial	
Neuroptera																		
Myrmeleontidae															1	1	Terrestrial	

Table 8. (Continued).

Species or Group	EC02	EC03	WR01	WR02	WR03	WR05	WR06	WR18	WR20	WR27	WR28	WR29	WR90	WR98	Drift	Total	Functional Group	References[a]
Megaloptera																		
Corydalidae																		
Corydalus corydalus															18	18	Predator	12, 15
Coleoptera																		
Dytiscidae																		
Hydaticus sp.	1	2													121	124	Predator	12, 15
Hydrophilidae																		
Hydrophilus sp.	1														2	3	Predator	12, 15
Elmidae	1		1		57	2		29	1	29		10			168	298	Detritivore (FPOM)	12, 15
Coccinellidae															3	3	Terrestrial	
Other Terrestrials	1				1										44	46	Terrestrial	
Trichoptera																		
Hydropsychidae																		
Hydropsyche sp.			3	2	577	16		91	4	30		34		1	469	1227	Filterer	4, 5, 12, 13, 15, 18
Hydroptilidae																		
Hydroptila sp.					27	5		8		4		1			2	47	Herbivore (Diatoms)	12, 13, 15, 18
Brachycentridae																		
Brachycentrus sp.			1		26	2		24	1	11		9			49	123	Filterer	11, 12, 15, 18
Limnephilidae					2	3		2		1		12			46	66	Detritivore (CPOM)	12, 15, 18
Lepidoptera																		
Pyralidae																		
Parargyractis sp.					3			1							1	5	Herbivore (periphyton)	12, 15

Table 8. (Continued).

Species or Group	EC02	EC03	WR01	WR02	WR03	WR05	WR06	WR18	WR20	WR27	WR28	WR29	WR90	WR98	Drift	Total	Functional Group	References
Terrestrial sp.															10	10	Terrestrial	
Hymenoptera																		
Formicidae	1							1							209	211	Terrestrial	
Other Terrestrial															145	145	Terrestrial	
Diptera																		
Tipulidae																		
Tipula sp.		2			2	4		18	4	18		2		3	10	63	Detritivore (CPOM)	12, 15
Psychodidae															1	1	Detritivore (FPOM)	12, 15
Heleidae																		
Palpomyia sp.					5			101	5	1		6		27	2	147	Predator	12, 15
Simuliidae																		
Simulium sp.	2				144	1		108	2	186		1			37	481	Filterer	7, 12, 15
Chironomidae	208	272	1	2	276	69		107	45	96	1	37	110	20	2759	4003	Detritivore (FPOM)	3, 12, 15
Stratiomyidae																		
Stratiomys sp.	1															1	Detritivore (FPOM)	12, 15
Tabanidae	2							1							16	19	Predator	12, 15
Rhagionidae																		
Atherix sp.					5		1	2		2					5	15	Predator	12, 15
Empidldae																		
Hemerodromia sp.		1			1			1		1		1			35	40	Predator	12, 15
Ephydridae																		
Notophila sp.		34						3							373	410	Detritivore (FPOM)	12, 15

Table 8. (Continued).

Species or Group	EC02	EC03	WR01	WR02	WR03	WR05	WR06	WR18	WR20	WR27	WR28	WR29	WR90	WR98	Drift	Total	Functional Group	References*
Syrphidae					1											1	Terrestrial	
Other Terrestrial								1							121	122	Terrestrial	
Mollusca																		
Pelecypoda																		
Sphaeriidae															2	2	Filterer	15
Totals	251	313	6	10	2271	201	1	1597	80	1217	1	227	142	67	10843	17227		

* References

1 Allen, R. K. (1973)
2 Anderson, N. H. et al. (1978)
3 Blum, J. L. (1957)
4 Chapman, D. W. and R. Demory (1963)
5 Coffman, W. P. et al. (1971)
6 Cummins, K. W. (1973)
7 Edmunds, G. F. et al. (1976)
8 Fredeen, F. H. J. (1964)
9 Gilpin, B. R. and M. A. Brusven (1970)
10 Hall, R. J. et al. (1975)
11 Jones, J. R. E. (1950)
12 Mecom, J. O. and K. W. Cummins (1964)
13 Merrit, R. W. and K. W. Cummins (eds) (1978)
14 Minckley, W. L. (1963)
15 Minshall, G. W. (1967)
16 Neel, J. K. (1968)
17 Pennak, R. W. (1978)
18 Shapas, T. J. (1976)
19 Waters, T. F. (1972)
20 Wiggins, G. B. (1977)
21 Woodall, W. R., Jr. and J. B. Wallace (1972)

Seasonal changes in the major functional groups for a typical riffle (Transect 18) and a pool (Transect 20) are shown in Figures 52 and 53. At Transect 18, course particulate detritivores are predominant in the early summer, low in midsummer, and high again in the fall. This pattern is similar to the input of CPOM within the White River. Predators follow a nearly identical trend; high early and late in the year and low in midsummer. Herbivores were never predominant, but did have maximum levels in early and late summer, corresponding to periods of high production and biomass. The filter-feeding functional groups dominated the community structure from late June through early October. The breakdown of CPOM into FPOM, the scouring activity upon the actively growing periphyton community, and the attachment provided by filamentous algae may provide an abundant amount of food and substrate for this group of organisms at this time.

In Transect 20 (pool), no seasonal patterns in functional group composition was found. A comparison as a percent composition over sample periods varied greatly. The major cause of these highly variable results are the low species richness found in the invading community after a natural perturbation (Table 9). In general, the pool transects were significantly lower in functional group and species richness when compared to the riffle transects.

The recolonization of the pool transects is believed to be caused by macroinvertebrate drift. During each sample period, diel drift samples were collected on the White River. The number of macroinvertebrates drifting through a m^2 cross-section of river between May and December can be seen in Figure 54. Maximum drift densities were in early summer (100-700/m^2/hour) with a decline through October and November (14-16/m^2/hour). The diel pattern observed in the White River (Figure 55) indicated highest drift densities occurred between sunset (2000 hours) and sunrise (0600 hours) with lowest densities during daylight (1200 and 1800 hours).

The macroinvertebrate community in Evacuation Creek was found to be similar to the backwater community at Transect 20. The results for biomass, density, functional group composition (percent biomass), and species richness are shown in Table 10. Seasonal trends in biomass and density fluctuated greatly with highest values observed at these sites being similar to peak values from the White River. The detritivore functional group dominated in all sample periods with predators and terrestrial insects being the only other groups present.

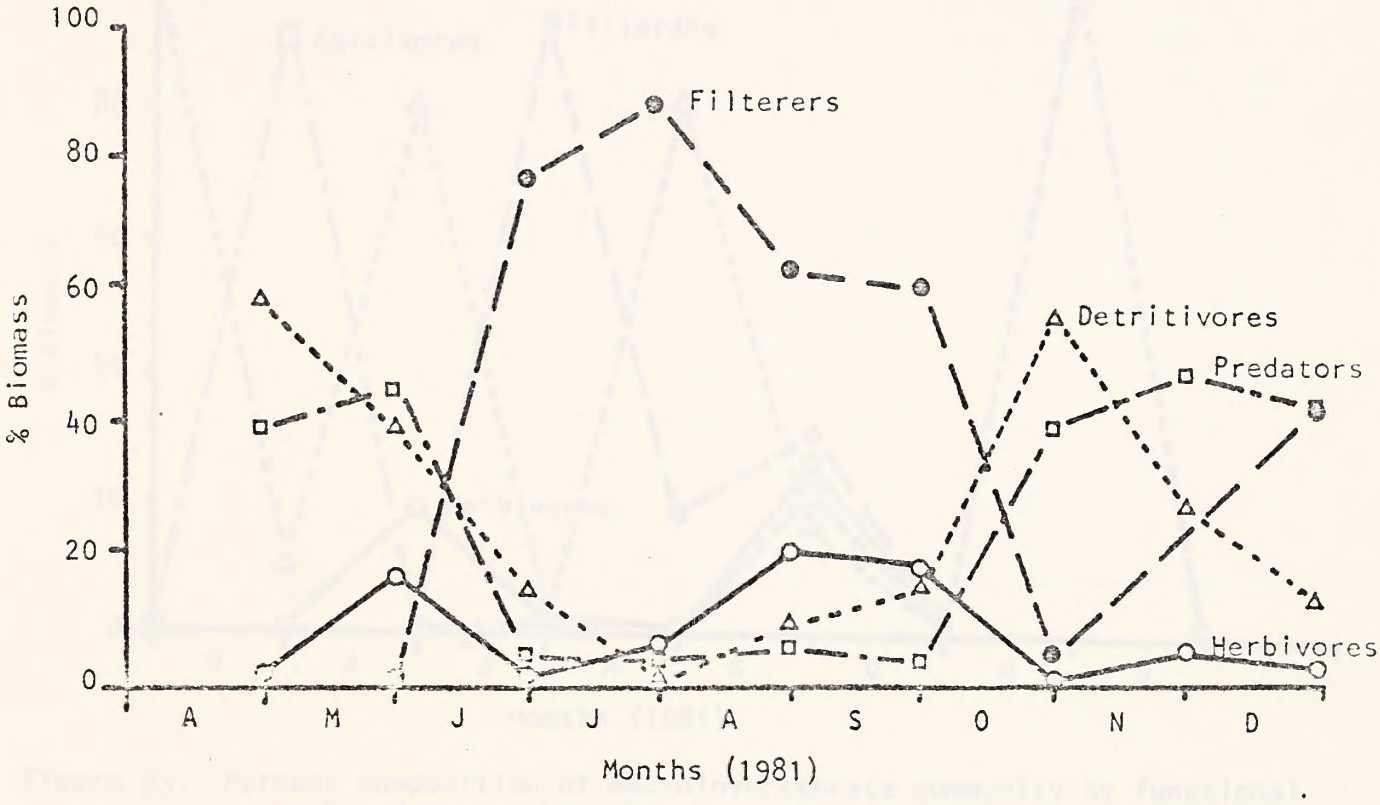


Figure 52. Percent composition of macroinvertebrate community by functional group for Transect WR 18 during 1981.

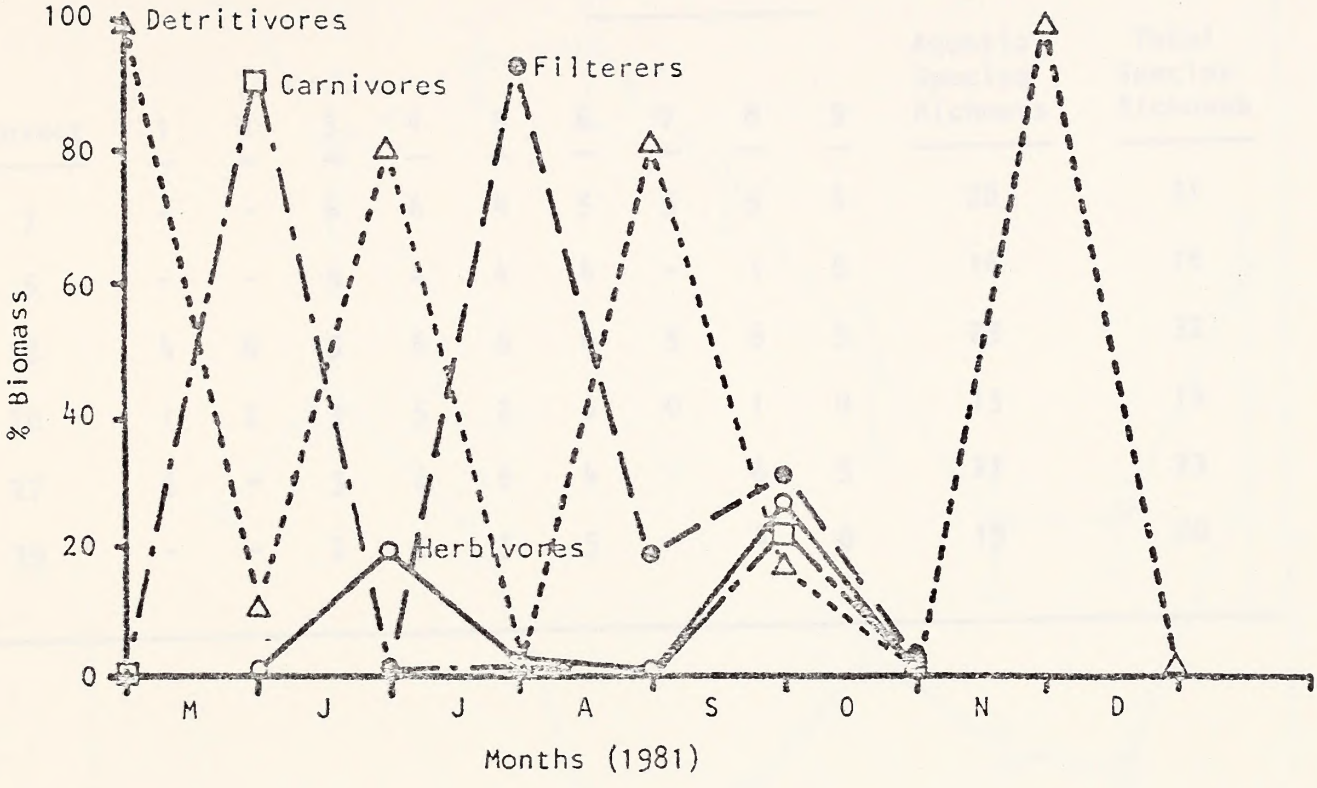


Figure 53. Percent composition of macroinvertebrate community by functional group for Transect WR 20 during 1981.

Table 9. Functional group richness for each sample period and total species richness for Transects 3, 5, 18, 20, 27, 29, based upon benthic samples collected May - December, 1981.

	Sample Period										
Transect	1	2	3	4	5	6	7	8	9	Aquatic Species Richness	Total Species Richness
3	-	-	6	6	4	5	5	5	4	28	31
5	-	-	3	4	4	4	-	1	6	16	16
18	4	6	5	6	6	6	5	6	5	29	32
20	1	2	3	5	2	6	0	1	0	13	14
27	6	-	3	6	6	4	-	6	5	21	23
29	-	-	2	5	5	5	-	4	6	19	20

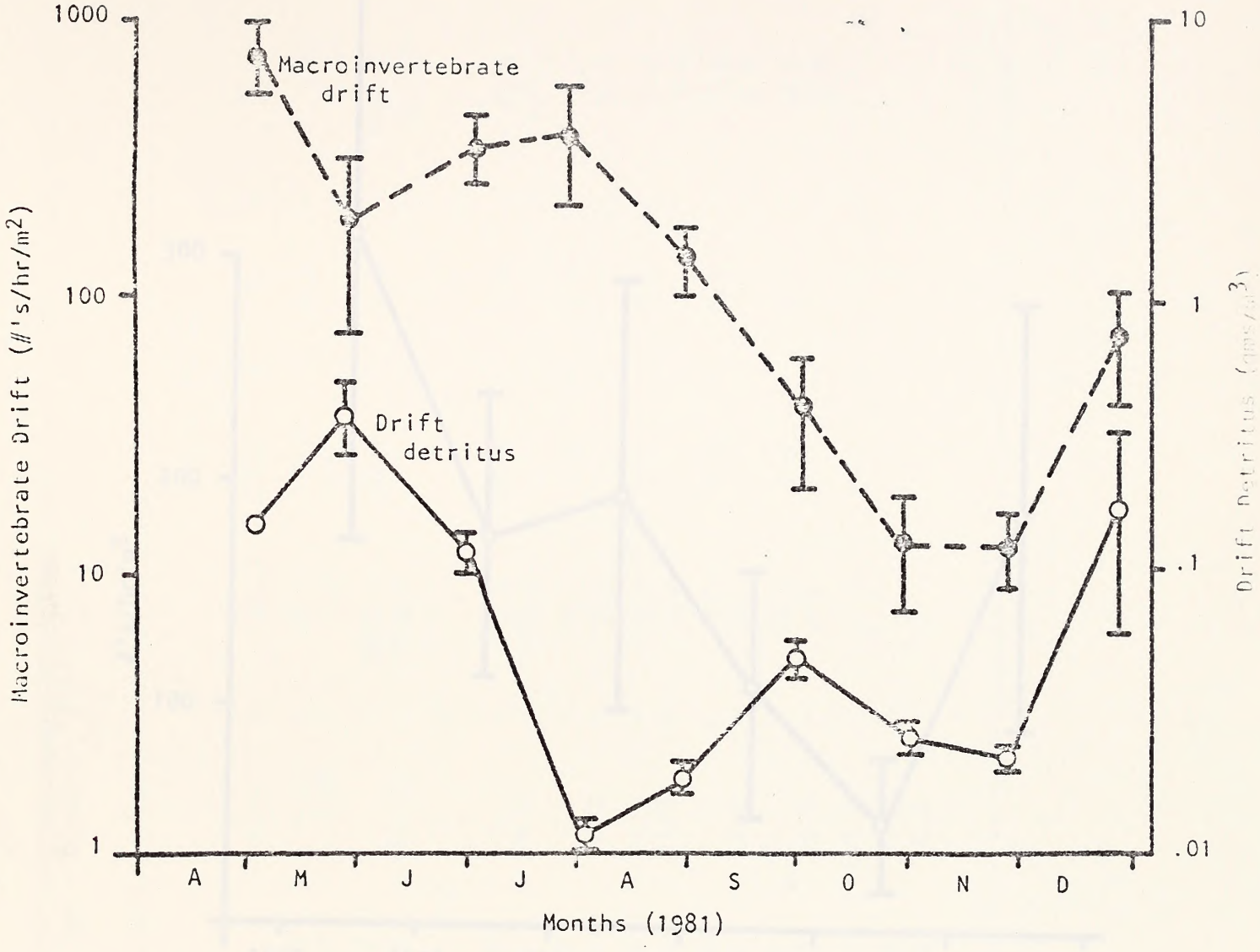


Figure 54. The macroinvertebrate drift (#'2/hr/m^2) and drift detritus (gms/m^3) in the White River during 1981. Sample means and standard errors are given.

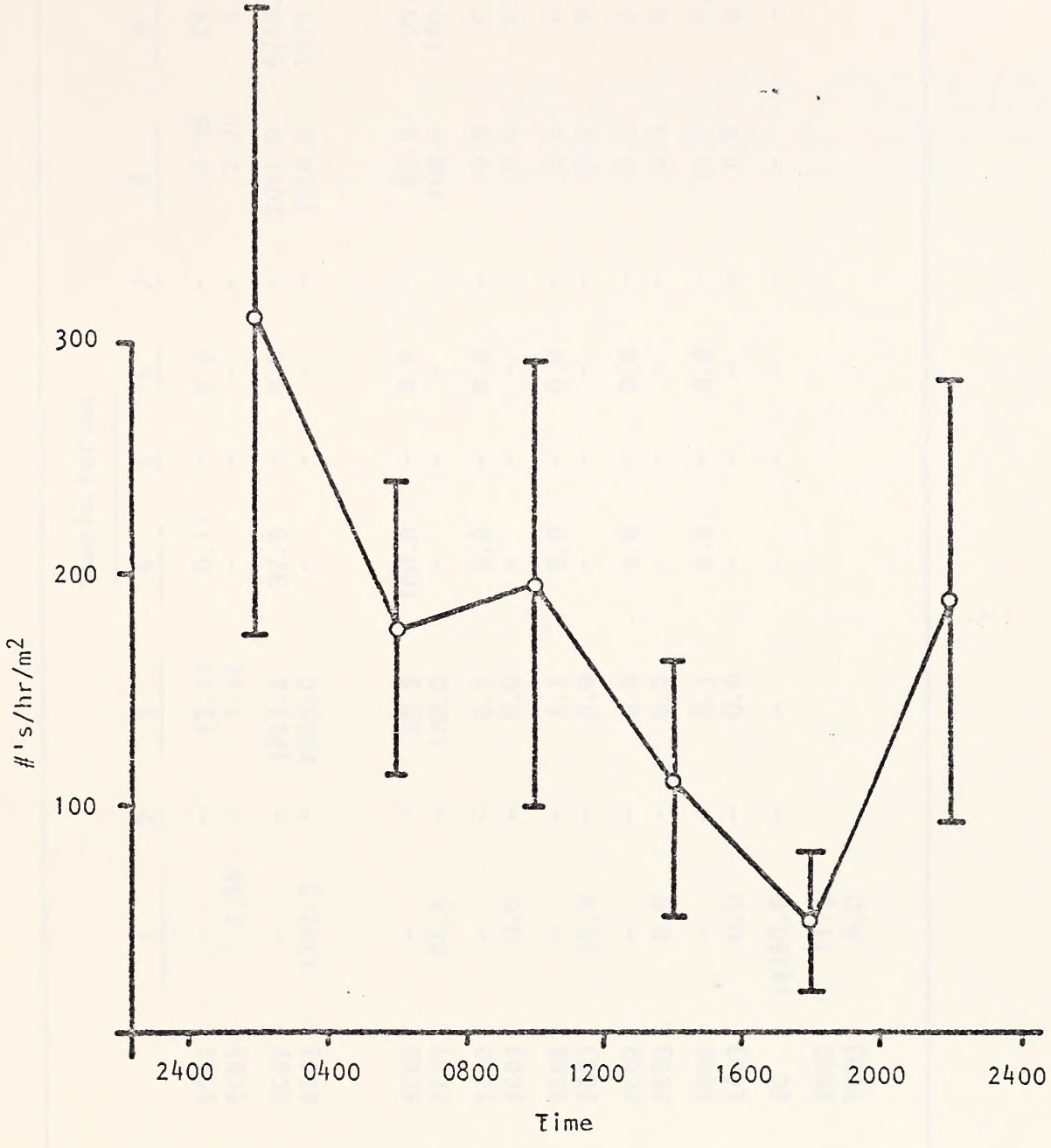


Figure 55. Numbers of macroinvertebrates per hour per m^2 of sampling area as a function of time of day. Based on drift samples taken June - December, 1981 on the White River, Utah.

Table 10. A summary of the macroinvertebrate community of Evacuation Creek, Utah, based upon benthic and drift samples collected May - December, 1981.

Parameter		Sample Period 1	2	3	4	5	6	7	8	9
Biomass (gms/m^2)	EC02	-	-	17.11	0.11	-	0.0	-	0.98	24.35
	EC03	2.84	-	1.41	-	-	-	-	2.70	5.88
Density (#/m^2)	EC02	-	-	3827.2	37.9	-	0.0	-	2011.0	6760.7
	EC03	1707.9	-	2665.0	-	-	-	-	1910.0	1470.0
Functional Group Composition (% biomass)										
Detritivores	EC02	-	-	98.5	100.0	-	0.0	-	89.9	99.5
	EC03	84.9	-	100.0	-	-	-	-	100.0	100.0
Herbivores	EC02	-	-	0.1	0.0	-	0.0	-	0.0	0.0
	EC03	0.0	-	0.0	-	-	-	-	0.0	0.0
Predators	EC02	-	-	1.1	0.0	-	0.0	-	0.0	0.0
	EC03	15.0	-	0.0	-	-	-	-	0.0	0.0
Filterers	EC02	-	-	0.0	0.0	-	0.0	-	0.0	0.5
	EC03	0.0	-	0.0	-	-	-	-	0.0	0.0
Terrestrial	EC02	-	-	0.3	0.0	-	0.0	-	10.1	0.0
	EC03	0.0	-	0.0	-	-	-	-	0.0	0.0
Drift (#/m^2/hr)	EC	14268.0	-	-	-	-	-	-	-	-
Species Richness	EC02	11.0								
	EC03	6.0								

Drift samples taken during the first sample period indicated that Evacuation Creek had higher numbers of drifting insects per m^2 cross-section per hour than did the White River. Species richness in both the benthic and drift samples were lower in Evacuation Creek compared to the White River.

DISCUSSION

The White River ecosystem has been extensively studied by the WRSOC (1974-1981). These studies have included the physical, chemical, and biological components of the river between 72 and 102 kms from the confluence. Several components of the ecosystem have been shown to have dramatic longitudinal changes as one moved downstream from Rio Blanco Lake. This section of the report is intended to demonstrate the specific seasonal changes that have been observed above, on, and below the federal lease tracts Ua-Ub. Furthermore, this discussion will attempt to provide sufficient information such that the function of this section of river within the stream continuum can be adequately assessed.

Physical Properties

Over certain reaches of the river, particularly those located from the confluence with the Green River to Rio Blanco Lake (240 kms), the White River appears to be a meandering river flowing within a meandering valley (Lyle 1980). This would suggest that the river may be under fit (flowing in a valley that was produced by a much larger river (Dury 1964). The ratio of the White River valley width to the valley meander wavelength (from USGS topographic maps) has been shown to match the empirical relationships documented by Leopold and Wolman (1957), and thus supports the contention that this river valley is, in fact, an old stream bed (Figure 56). Because of this under fitting, the present floodplain within the river valley is five to six times the river width. This has resulted in an extensive amount of river bottom land and the development of a major riparian community. In total, 2450 ha of riparian floodplain exist from the Colorado border to the White River's confluence with the Green River. Approximately 1500 ha are located adjacent to the Ua-Ub federal lease tracts.

Within this geomorphological setting, certain physical abiotic factors play a dominant role in shaping the structure and function of the ecosystem. Within this strata of the White River, these dynamic factors are:

(1) Surface flows from adjacent watersheds into the river, and

(2) upstream contributions to the river segment.

The influence of these external sources of water and materials on the aquatic system has been shown to be both direct and indirect.

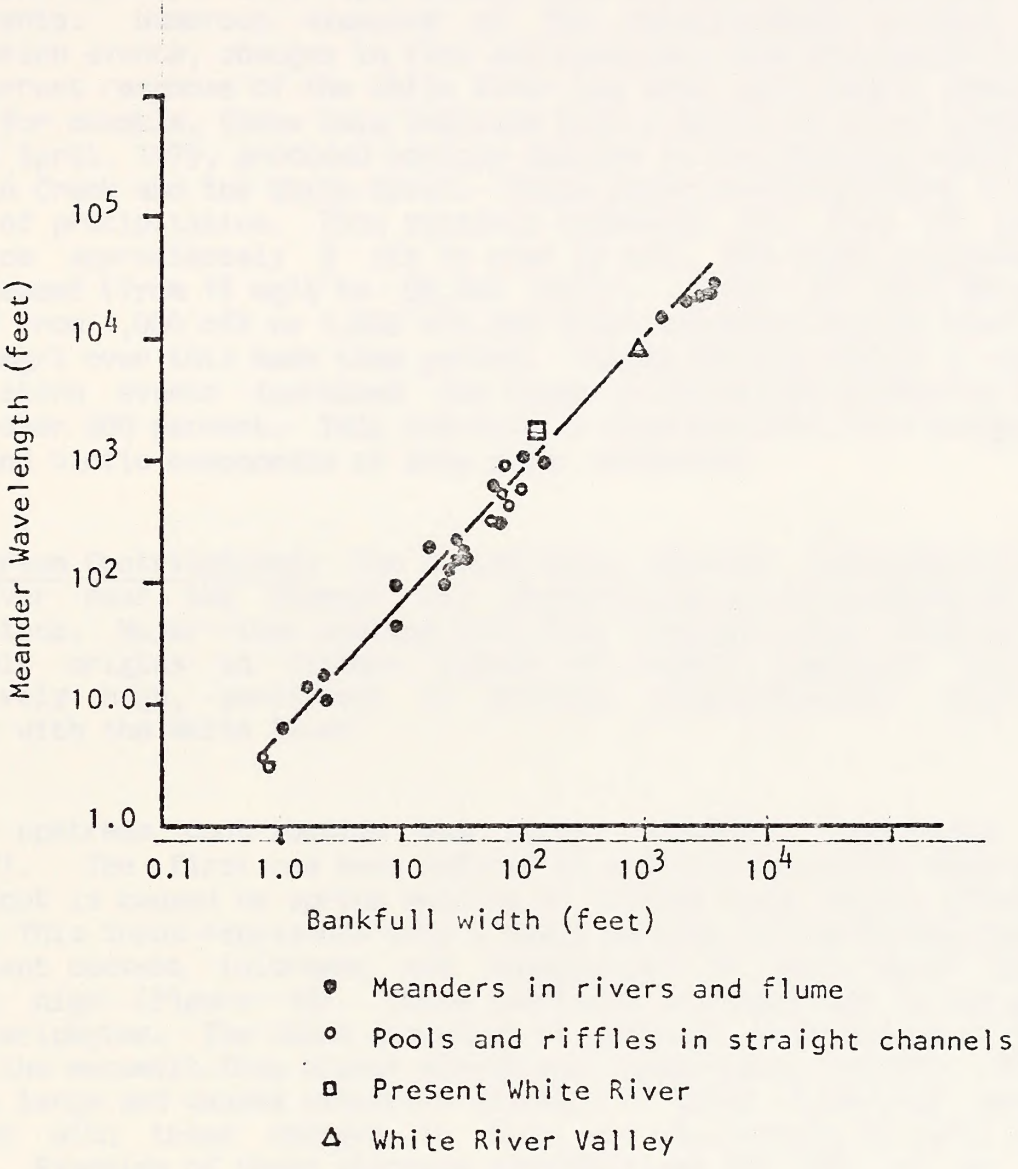


Figure 56. Relationship between bankfull width of channel to channel meander wavelength (Leopold and Wolman 1957).

Watershed Contributions: A major abiotic input into this stream segment is the surface flow from adjacent watersheds. The contribution of water and materials to the White River from Evacuation Creek and the dry washes has been studied extensively. This input occurs from periodic, intense, and localized storm events. Numerous examples of the relationships between seasonal precipitation events, changes in flow and materials from Evacuation Creek, and the concurrent response of the White River has been previously shown (WRSOC 1982). For example, these data indicate that a series of storm events during March and April, 1979, produced obvious changes in the physical conditions of Evacuation Creek and the White River. These storm events produced 1.02 inches (2.6 cm) of precipitation. This rainfall increased the flow of Evacuation Creek from approximately 8 cfs to over 32 cfs. The total suspended solids also increased (from 16 mg/l to 65,000 mg/l). Flows in the White River increased from 1,000 cfs to 1,800 cfs and total suspended solids from 200 mg/l to 4,500 mg/l over this same time period. During October 1981, a series of similar storm events increased the flows in Evacuation Creek and the White River by over 500 percent. This increase in flow resulted in a change of the abiotic and biotic components of this river ecosystem.

Upstream Contributions: The second major external contribution to the White River near the federal oil shale tracts is represented by upstream contributions. Major river systems that flow through arid regions usually have their origins in distant places of higher elevation that receive comparatively high, persistent or seasonal precipitation. Such is the situation with the White River.

The upstream contribution has three distinct components (WRSOC, 1974-1982). The first has been defined as normally occurring baseflow. The second input is caused by spring melting of lowland snow packs (lower basin runoff). This input represents only a small portion of the spring runoff, but the nutrient content (nitrogen and phosphorus) of this water source is extremely high (Figure 10). These nutrients are important in the growth of aquatic periphyton. The third component of upstream contribution of the White River is the snowmelt from higher elevations (upper basin runoff). This water source is large and causes extensive dilution of total dissolved substances. Concurrent with these changes in flow, are the changes in total suspended sediment. Examples of these upstream contributions for 1981 can be seen in Figure 5. The two dynamic factors previously mentioned are extremely important because they determine the abiotic or physical structure of this section of the river. Based upon the existing limits of these dynamic factors, this abiotic structure can be defined.

Abiotic Structure: The penetration of light (defined as the extinction coefficient "k") has been shown by WRSOC to be related to total suspended solids (Figure 13). This interrelationship combined with the total suspended

solids in the river system has allowed the calculation of the depth to which one percent of the surface light will penetrate based on TSS levels (Figure 12). This light level has been empirically determined to be the level at which primary production will be limited. Using all available data, it has been determined that between 1974 and 1981, the one percent light level has varied between a depth of 0.8 and 25.8 cms. In general, the calculated light penetration is poor between April and July (less than 10 cms in depth) and greatest between September and January (greater than 40 cms). Because light is related to primary production, the present seasonal turbidity trends are extremely important in the interpretation of production-respiration trends.

The description of the White River in terms of its abiotic structure and the associated biological interrelationships must also include the two major fluvial geomorphologic components: (1) The river must be defined in terms of its hydraulic geometry, and (2) its water characteristics. The factors necessary for these descriptions and their quantification are discussed below.

The hydraulic geometry or the determination of the physical factors which shape and form the morphology of the river bed are extremely important. The major physical impact of current and turbulence is the shifting and restabilization of the bed configuration in the White River. This process not only produces molar action which may physically damage animals and plants, but also modifies certain habitat types (pools, runs, riffles, and substrate configuration). Periods of high discharge or storm events (dynamic abiotic inputs) will produce greater habitat modifications than the normal, baseflow water discharge levels. The discharge patterns found in the White River combined with channel geomorphology have produced several basic habitat types.

As noted previously, twenty-nine cross-stream transects were surveyed from bank to bank in the White River during April, 1981. These transects were located in groups of three at two-mile intervals between Hell's Hole Canyon (km 102) and Asphalt Wash (km 72). At each transect, complete profiles of the streambed were surveyed and combined with data collected on velocity, substrate, gradient, sediment, percent light transmission, and chlorophyll a (Table 1).

Following this survey, data were subjected to "Clustar" analysis using the technique of Marshall and Romesburg (1979). The objective was to determine the distribution of basic stream habitat types. The transects were divided nearly evenly between two major categories of which 14 transects are designated as "pool" and 15 as "riffle." This implied that this section of the river is evenly divided between the two types of habitats. In order to verify the Clustar analysis, an ANOVA was conducted on these major types (Table 1).

Those parameters which differed significantly between the two transect types were designated by (*) in the first column. These results verified the output of the Clustar analysis. The riffle transects were significantly shallower, steeper in gradient, wider, and had larger substrate and algal standing crops (indicated by the chlorophyll values).

An analysis of dominant substrate for the 478 locations measured along the 29 transects indicated that 45.2 percent of the substrate occurred in the coarser size fractions of boulder, cobble, rubble, and gravel, whereas the remaining 54.8 percent were found in the fine size fractions of sand, silt, and clay.

Transect 18 (riffle) and Transect 20 (pool) are examples of the two types of habitats present within the White River. Because the magnitude and distribution of water velocities are important in determining substrate size (see Appendix II), the velocity isolines in Figure 20 describe the major differences between these habitat types. The normal tubular flow through Transect 20 indicates small uniform substrate with only a small degree of turbulence. However, in Transect 18 the velocity distributions produced a series of high and low velocity cells, therefore, indicating turbulent water movement. These higher, more turbulent velocities have resulted in larger substrate in these habitat types.

Water characteristics for rivers in general have been described by Gibbs (1970) as being distinguished by three basic origins for loads of dissolved substances. These are: atmospheric precipitation, rock weathering, and evaporation-crystallization processes. These processes affect the relative distribution and concentrations of the major macroelements found in the White River ($HCO_3^{-1} > SO_4^{-2} > Cl^{-1} > CO_3^{-2}$, and Ca^{+2} and $Na^{+1} > Mg^{+2} > K^{+1}$). In general, the highest concentrations of these macroelements occurs between July and February, with lowest concentrations occurring during runoff (April through June). Data from the White River indicate that both nitrogen and phosphorus are often below detectable levels and could, therefore, become limiting to primary production. During 1981, the seasonal changes of these components was extensive (Figure 10). This seasonal trend in total inorganic nitrogen (TIN) and orthophosphate was, however, periodically changed by storm events. An example of the impacts of one single event can be seen in Figure 11. In the past, it has also been shown that similar storm events can drastically increase the concentration of certain heavy metals in the White River. For example, during a storm event in March, 1975, the dissolved copper content of Evacuation Creek increased from 6 ug/l to over 600 ug/l. The total suspended sediments also increased (200 mg/l to 10,200 mg/l). These temporal changes were also observed in the White River. Flow increased from 500 to 1,100 cfs, total suspended solids from 200 mg/l to 2,500 mg/l, and the dissolved copper increased from 4 ug/l to 340 ug/l. This phenomenon has also been noted for

other metals.

Temperature is an important physical factor in that it can both directly (biological metabolism) and indirectly (chemical precipitation) affect the structure and function of the White River. Temperatures reached a minimum of 0°C in December through February with ice over much of the stream. Maximum temperatures (25-28°C) occur during baseflow (July-September). It is important to note that the daily fluctuations in temperatures in the White River were as high as 13°C (Figure 8). These diel changes in temperature have been shown to be an important physical factor in the ecology and lifehistory of several macroinvertebrate species. These changes in daily temperature result in higher growth efficiencies and higher species richness.

Biological Properties

Within the White River continuum, the interrelationships between gross production and the storage, processing, and export of terrestrial organics is critical in defining the structure and function of this stream ecosystem. Furthermore, the spatial and temporal differences of these defined biological processes acting in concert with the previously described physical factors, determine the distribution of terrestrial organics and aquatic primary production. The specific food webs and their abiotic and biotic interrelationships will be discussed in more detail.

Autochthonous Food Web (Grazing): This food web centers around primary production by periphytic algal communities and grazing on these communities by aquatic macroinvertebrates. An example of the spatial and temporal changes in the attached algal community can be seen in Figure 31. The monthly averages for periphyton biomass in a riffle area were significantly higher during each monthly sample period when compared to a pool habitat. The impact of runoff causing a decrease in species within the periphyton community in April was evident in both transects; however, the riffle habitat demonstrated the highest rate of recolonization and growth when compared to the pool transects. The distribution of algae across the stream channel also showed significant differences. The shallower areas in Transect 18 had significantly higher algal standing crops when averaged over the whole year. However, in the pool transect, the opposite trend was observed (Figure 29). The shallow areas had extremely low chlorophyll a values, whereas the deeper sections of the stream had the highest values. The difference between these two habitat types may be related to substrate stability. The distribution of periphyton as related to substrate size will be discussed later in more detail.

The distribution of periphyton in the riffle habitat between March and December, 1981, further demonstrates that the shallow water communities have substantially higher biomass levels when compared to the deeper portions of the transect. These changes in biomass occurred on uniform substrates. Seasonal changes in species distribution in the riffle area indicate (Figure 32) that diatoms reach their maximum numbers in early spring and late fall, with Cladophora and other filamentous green algae dominating in mid- to late summer. The blue-green algae also reach their maximum abundance in late summer, although the percentage of the total community biomass is less than ten percent. As previously noted, the seasonal distribution of algae in the White River is identical to the seasonal changes of algae taken from whole water samples in the Green River. The seasonal trends in species composition and biomass levels are similar to the results of MacIntire and Phinney (1964) (see Appendix II).

The macroinvertebrate distributions (expressed as biomass) in two separate habitat types near the federal lease tracts were biologically different. For example, the distribution of macroinvertebrate biomass across the stream in the riffle transect (mostly gravel, rubble, cobble, and boulder) was not found to be significantly different throughout the year ($p=.05$, Figure 49). However, if average densities ($N=5$) for each sample period are compared over time, significant temporal differences ($p=.01$) were found. Mean biomass was low (.31 gms/m^2) in early May and increased significantly through late June. Average size also increased dramatically during this time period. Another significant increase occurred by early August ($p<.01$) reaching a maximum biomass of 10.9 gms/m^2. Biomass decreased through October after this increase. This habitat type proved to be extremely stable, with a well-defined community structure and function.

The pool transects in the White River are characterized by shifting sand along each side of the thalweg, with armored substrate in the deepest location (clay with a cobble or rubble matrix). Because of this varied habitat, significant differences in biomass (Figure 51) were found across the stream. Maximum biomass levels (.55 gms/m^2) were much lower than riffle habitats.

These differences were extremely transitory between sample periods. The data indicates that periodic storm events or runoff removes the majority of the macroinvertebrate community. The reestablishment of this community after these unpredictable events occurs almost immediately with increasing species richness developing the longer the period between storms.

In general, biomass and density estimates for the riffle transects were significantly higher than those for the pool transects with maximum biomass and density values in the pools only 6.1 and 12.7 percent of the riffle

values, respectively.

An ephemeral side channel was also sampled because of its proximity to the pool habitat (Figure 17). Water was in the channel from May to July, 1981. The richness of macroinvertebrates was low (one or two functional groups); however, the biomass was the highest recorded at any sample site (56.0 gms/m^2 in late June). This habitat type, although highly transitory, is extremely productive and may play an important role in the river system.

The structure of the macroinvertebrate community is also diverse, with all functional groups present. An example of the temporal change in these groups in a riffle habitat can be seen in Figure 52. The relative densities of the herbivores reach a maximum in September and October, and minimum in July and November. It is interesting to note that these periods of low densities correspond to periods of changing algal community structure. The biomass of herbivores exceeds that of detritivores during August, September, and October. Low detrital input and high periphyton production (P/R>1) are believed to be the possible causes. The filter feeding macroinvertebrates had their distributions almost identical to that of the green algae. This functional group which has the highest biomass may have been eating algae or detritus caught in the long filamentous green algae (primarily Cladophora), which dominated the primary producers community at that time.

The gross community production and respiration at this riffle site (WR18; km 79) indicated that the P/R ratio was usually greater than one. However, this was not directly reflected in the macroinvertebrate community. The dominant algae (Cladophora) was usually not directly eaten but could have indirectly affected the community structure by entrapping food particles and providing a holdfast for the gathering detritivores and filter feeders. Although never dominant (by biomass), the macroinvertebrate herbivore population was greatest in early summer and fall (Figure 52). This suggests a greater reliance on diatoms (rather than Cladophora) as a food source.

Several fish species found in the White River also utilize the algal community as a food source. Food habit data indicates that three species of endemic fish (speckled dace, flannelmouth, and bluehead sucker) appear to be feeding on benthic substrates. These species have 29-62 percent of their diet volume in the form of periphyton. The roundtail chub (the other common endemic) had only nine percent of its diet volume as algae. Fishes captured with low concentrations of algae as diet items (less than ten percent of stomach volume) also appeared to have the remaining diet items in the same proportion as these items were found within the macroinvertebrates drifting in the open water.

Allochthonous Food Web (Detritus): The detrital food web in the White River changed dramatically during the year (Figure 52). It has been shown (ERI 1982) that the coarse particle organic load (>1 mm) increased from 7 mg/m^3 to 70 mg/m^3 just above the federal lease tracts. The input of these organics also changed by season. The highest mass of CPOM entered the stream during the periods of runoff (May-June), being occasionally supplemented by storm events (Figure 54). The input of the CPOM during leaf fall (September-October, 1981) was reflected by the change in the CPOM standing crop at the transects studied (primarily Transect WR20; km79) as well as the amount obtained in the drift nets. These high values during September-October are believed to be the result of cottonwood leaves settling in the slower water of this habitat type. The input and processing of these food stuffs were evident by the relative densities of the detritivores (Figure 52). A temporal change to the filter-feeding functional group of macroinvertebrates may also be the result of the processing of CPOM to FPOM. This period of time also had the highest number of degree days, and thus, an increased rate of decomposition (Figure 41).

Physical-Biological Interactions

The interrelationships between the abiotic and biotic components of the river system are important in determining the structure and function of the White River. A brief description of how the abiotic or physical environment affects the biological community and how this physical environment may change because of naturally occurring conditions is presented below.

The stability and composition of the substrates described in the White River are dependent upon current velocity and influences the type of organisms which use the benthic region for attachment (periphyton and macroinvertebrates). Substrate size also determines the size and density of organic particles (food for higher organisms) caught within pore spaces. The effect of substrate size upon the distribution of detritus and species richness of organisms within the White River can be seen in Figures 57 and 58. The distribution of all organisms (algae, macroinvertebrates, and fish) appears to be similar. Sand is avoided by all organisms while the intermediate substrates are preferred (gravel, rubble, and cobble).

Habitats characterized by fine grained substrates (silt and sand) support the fewest number of species. For example, a burrowing species, Chironomus, comprises more than 50 percent of the invertebrate populations in areas characterized by these soft substrates. Filterers are least important on the fine substrates, probably due to a lack of flow or anchor points on the substrate. Shredders of large detrital particles, although not common, reach greater densities on finer substrates than on other substrates, probably due

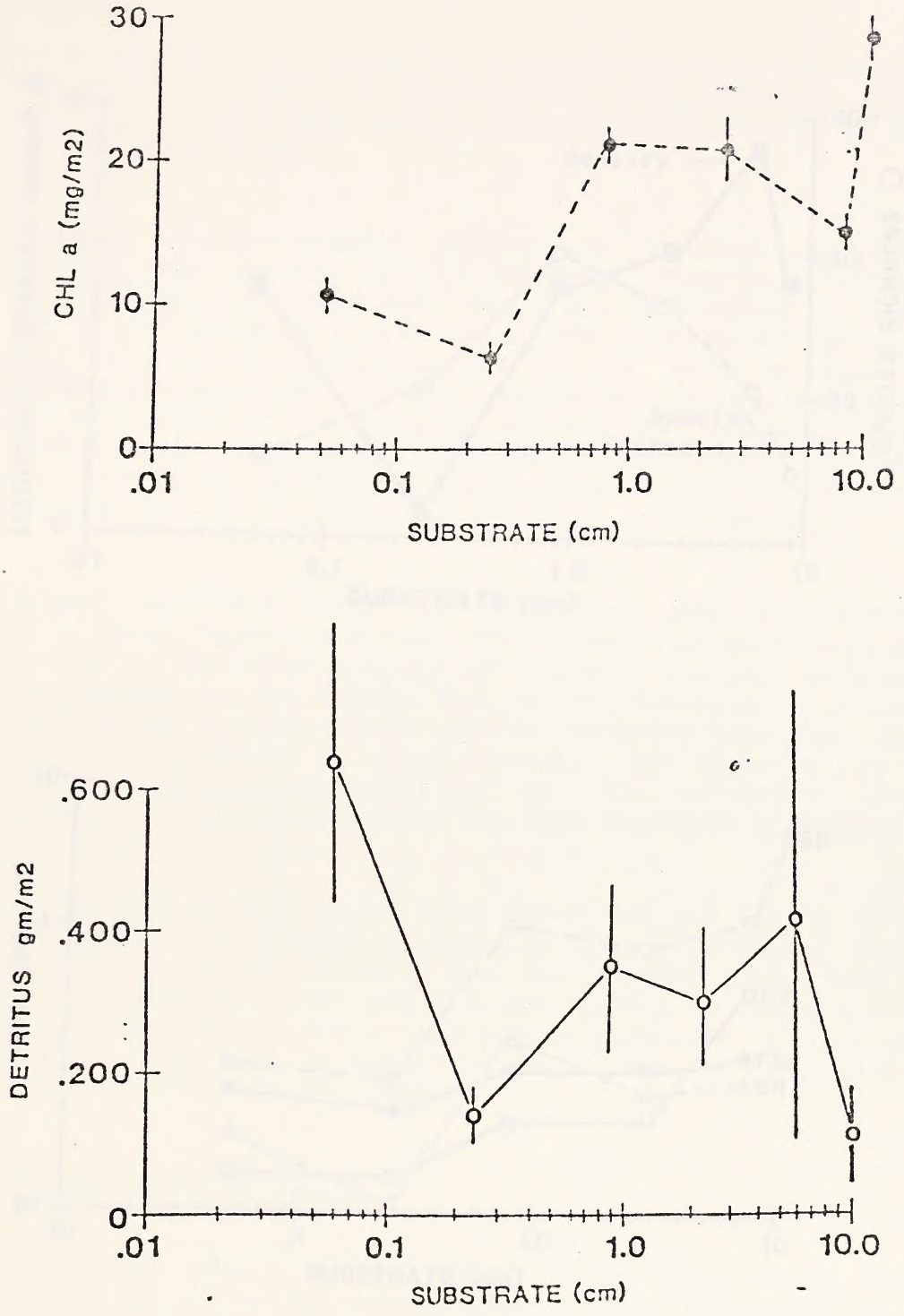


Figure 57. The distribution of detritus (gms/m^2) and chlorophyll a (mg/m^2) standing crop on different substrate sizes in the White River. Bars denote 95% confidence intervals.

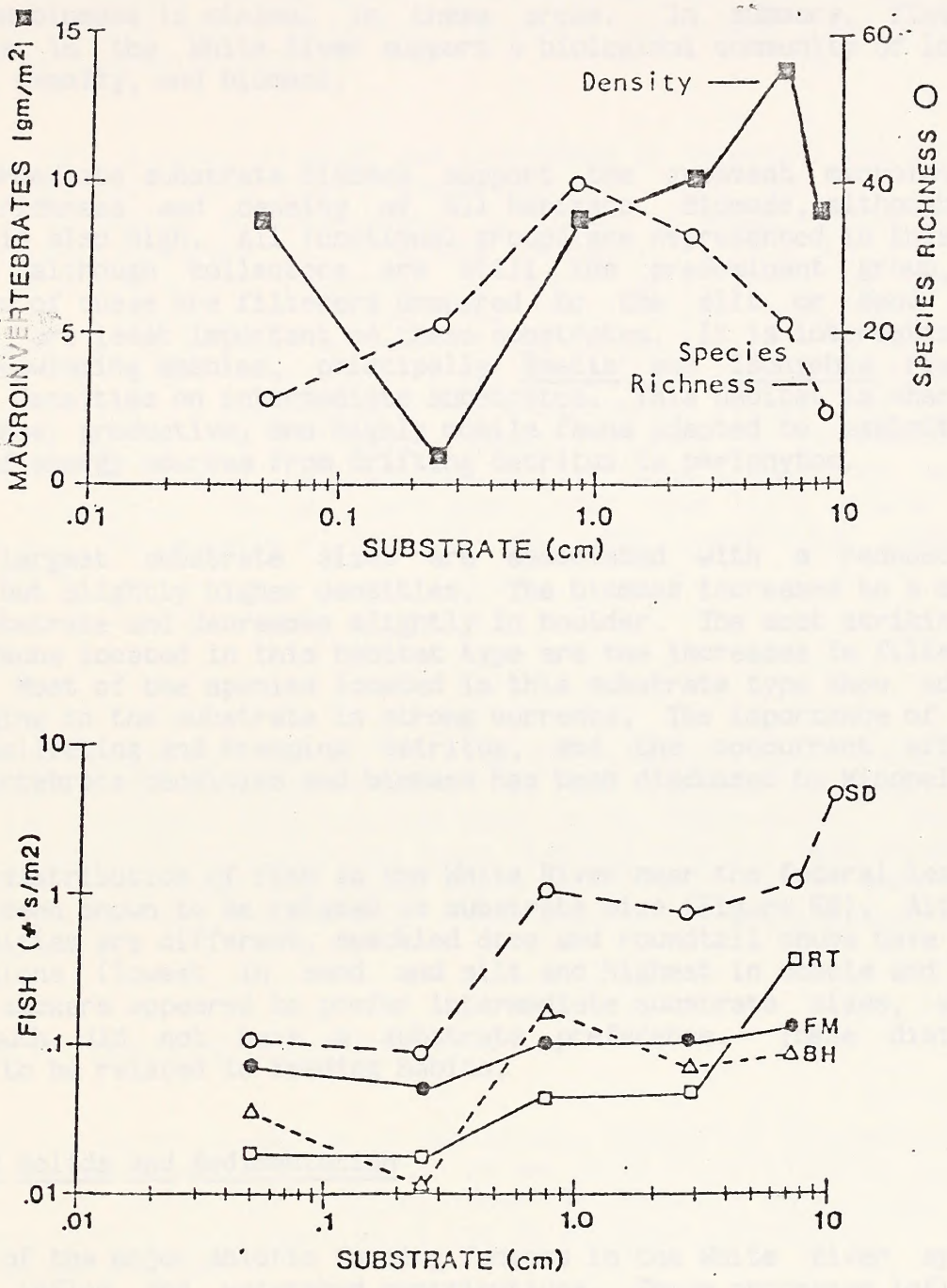


Figure 58. The distribution of macroinvertebrates (gms biomass/m^2) and fish (#'s/m^2) by substrate sizes in the White River, Utah. SD = speckled dace, RT = roundtail chub, FM = flannelmouth sucker, BH = bluehead sucker.

to collection of detrital particles settling out in these slower waters. Periphyton biomass is minimal in these areas. In summary, fine grained substrates in the White River support a biological community of low species richness, density, and biomass.

Intermediate substrate classes support the greatest macroinvertebrate species richness and density of all habitats. Biomass, although not at a maximum, is also high. All functional groups are represented in this habitat type and although collectors are still the predominant group, a larger proportion of these are filterers compared to the silt or sand habitats. Shredders are least important on these substrates. It is interesting to note that free-swimming species, principally Baetis and Isonychia reach their greatest densities on intermediate substrates. This habitat is characterized by a diverse, productive, and highly mobile fauna adapted to exploit a wide variety of energy sources from drifting detritus to periphyton.

The largest substrate sizes are associated with a reduced species richness but slightly higher densities. The biomass increases to a maximum in cobble substrate and decreases slightly in boulder. The most striking change in the fauna located in this habitat type are the increases in filter-feeding insects. Most of the species located in this substrate type show adaptations for clinging to the substrate in strong currents. The importance of substrate size in collecting and trapping detritus, and the concurrent effect upon macroinvertebrate densities and biomass has been discussed by Minchell (1969).

The distribution of fish in the White River near the federal lease tracts has also been shown to be related to substrate size (Figure 58). Although the fish densities are different, speckled dace and roundtail chubs have identical distributions (lowest in sand and silt and highest in cobble and boulder). Bluehead suckers appeared to prefer intermediate substrate sizes, while the flannelmouth did not have a substrate preference. These distributions appeared to be related to feeding habits.

Suspended Solids and Sedimentation

Two of the major abiotic forcing factors in the White River system are upstream inflow and watershed contributions. These processes influence the silt load and sedimentation rate in the White River.

The effect of sediment brought into the White River is related to current velocity and sediment transport. For example, changes in water velocity in the absence of any change in sediment load have been shown to increase the

amount of periphyton production. Conversely, an increase in sediment load without a consequent change in current velocity, can reduce light penetration, and hence, periphyton production.

During 1981, a detailed study was undertaken to determine the impacts of suspended sediment upon the transparency of light and the concurrent relationship in benthic photosynthesis. A summary of these results indicated that suspended sediments alone could indirectly affect the rate of algal production by a concurrent reduction in light penetration to the stream bottom. The relationship between the total daily production rate of photosynthesis (mg O_2/m^2/day) and daily light intensity (lux) can be seen in Figure 37. As noted previously, light penetration and turbidity were also found to be related. A comparison of the assimilation ratios (mg O_2/mg Chl a/hr) for a spring and fall algal community (Figure 59) indicated that the spring communities (7-1-81) were adapted to high light intensities while the midsummer communities (10-1-81) were grown under lower light levels. This is consistant with the patterns of turbidity and light penetration observed in the White River.

Flow

The relationship of flow and scouring effect of suspended sediments has been documented by McIntire (1965). It is apparent that high suspended sediments are associated with elevated flows in the White River. The impact of these abiotic factors upon the algal standing crop has been documented by the reduction in chlorophyll a during April and October, 1981. Such scouring or "molar action" may cause similar reductions in the populations of benthic macroinvertebrates.

Sample Site and Sample Size Justification

Two of the major objectives of the Aquatic Monitoring Program during the 1981 program were to (1) characterize the White River habitats in order to develop the most appropriate sampling sites for long-term monitoring, and (2) develop appropriate techniques which can be utilized efficiently in the long-term monitoring program.

The initial field investigation and subsequent site selection was based upon the assumption that the physical factors used in this initial survey would adequately define the biological community. This selection of above (WR03, WR05), on (WR18, WR20), and below (WR27, WR29) sites was important in the potential detection of future project- related impacts. To test the

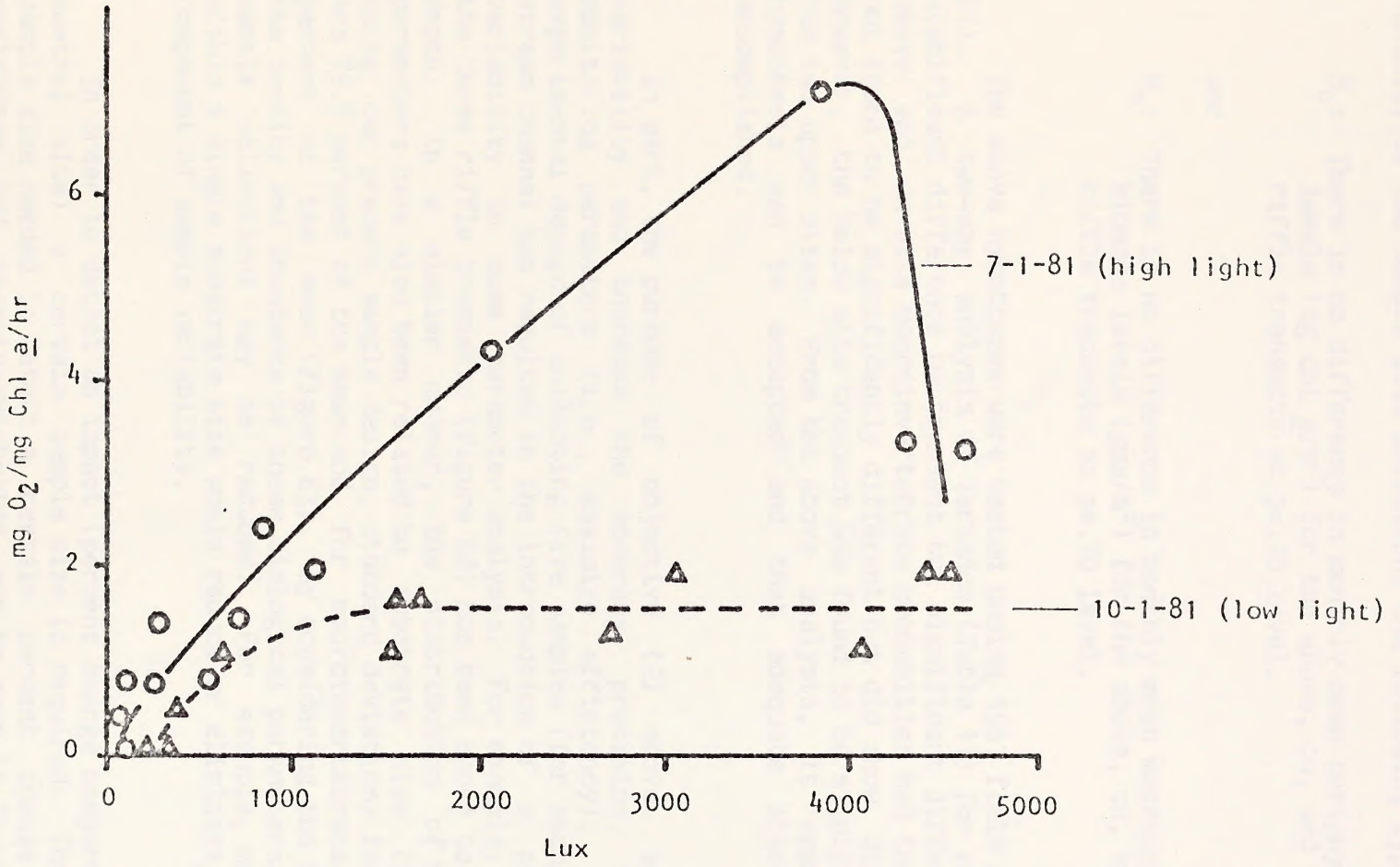


Figure 59. The specific rate of photosynthesis related to light intensity on Transect WR 18 in the White River. The data is selected from production experiments run on 7-1-81 and 10-1-81. The P/R chambers were in 10 cms of water.

validity of the sample site selection, the following hypotheses were tested:

H_o: There is no difference in monthly mean periphyton biomass levels (mg Chl a/m^2) for the above, on, and below riffle transects at p=.10 level.

and

H_o: There is no difference in monthly mean macroinvertebrate biomass levels (gms/m^2) for the above, on, and below riffle transects at p=.10 level.

The above hypotheses were tested uusing 1981 field data (Figures 60 and 61). A two-way analysis of variance (Table 11) for periphyton indicates no significant difference by transect but significant differences over time. The above and on site macroinvertebrate communities had the same trends and were not found to be significantly different but did show differences over time. However, the below site transect was found to be significantly different from the two upper sites. From the above analysis, it appears that the above hypothesis can be accepted and that adequate site selection has been accomplished.

In part, the purpose of objective (2) above, was to reduce sample variability and increase the accuracy, precision, and sensitivity of the monitoring parameters (i.e., maximize efficiency). For example, the experimental design of collecting five samples (for each parameter) across the stream channel has resulted in the introduction of a predictable degree of variability in some parameter analysis. For example, periphyton biomass in the three riffle transects (Figure 62) has been shown to be related to water depth. In a similar manner, the distribution of many of the biological parameters have also been related to substrate size (Figures 57 and 58). Using our present sample design, standard deviations for periphyton biomasses are 75.9 percent of the mean and for macroinvertebrate biomasses are 89.8 percent of the mean (Figure 63). By considering the factors which regulate the density and abundance of these biological parameters, the variability in sample collections may be reduced. For example, more intensive sampling within a single substrate size would reduce or eliminate the habitat-related component of sample variability.

In order to detect an impact (percent change compared to the above or control site) a certain sample size is required. The relationship between sample size needed to detect a certain percent change in the biomass of periphyton and macroinvertebrates can be seen in Figure 64. These data are based upon a two-tailed test using the existing variance at the above site (WR03). As stated previously, an attempt is being made to decrease the

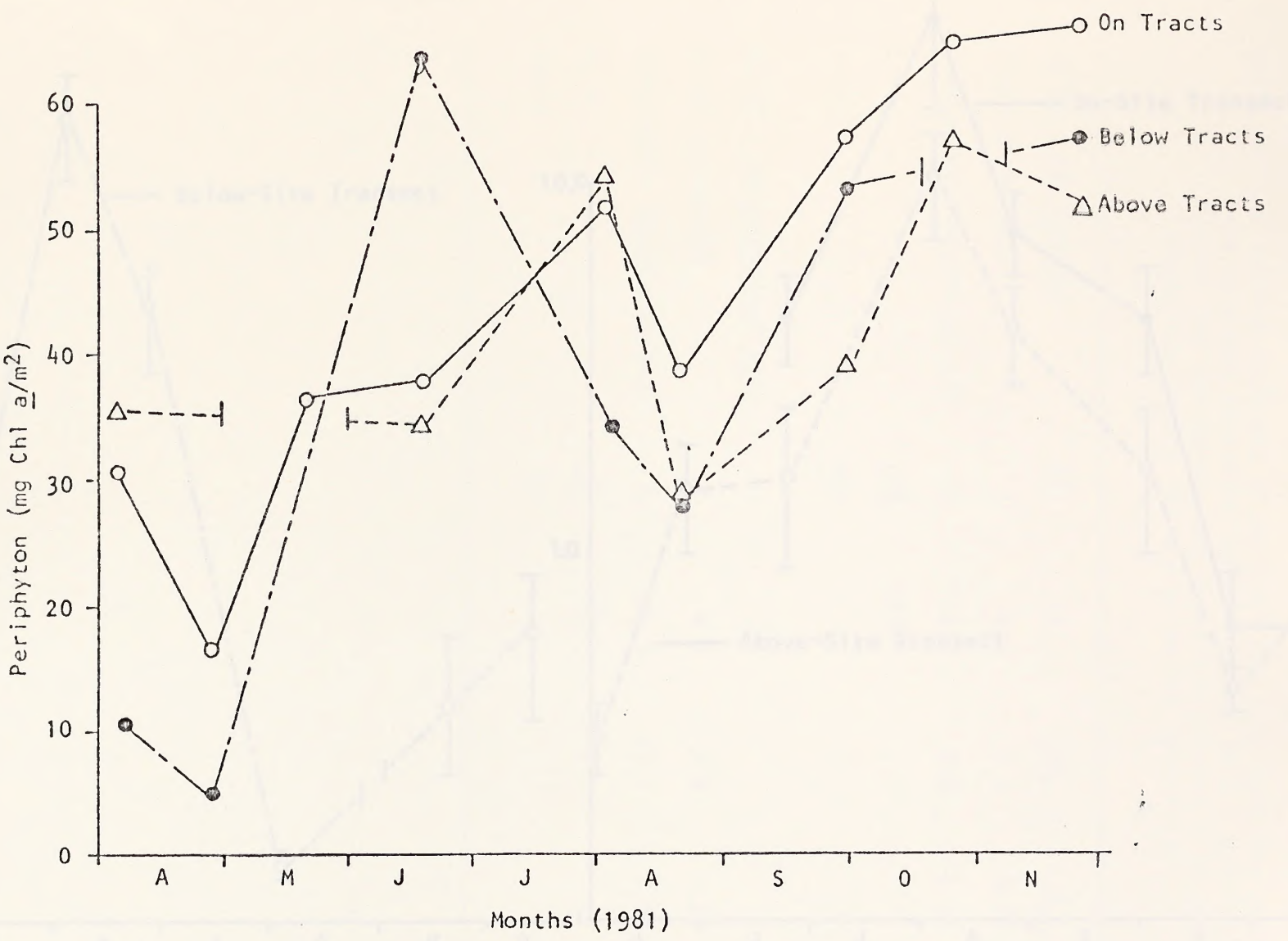


Figure 60. The monthly mean values (N=5) for periphyton biomass at the above, on-site, and below riffle transects on the White River, 1981. Standard errors are not given.

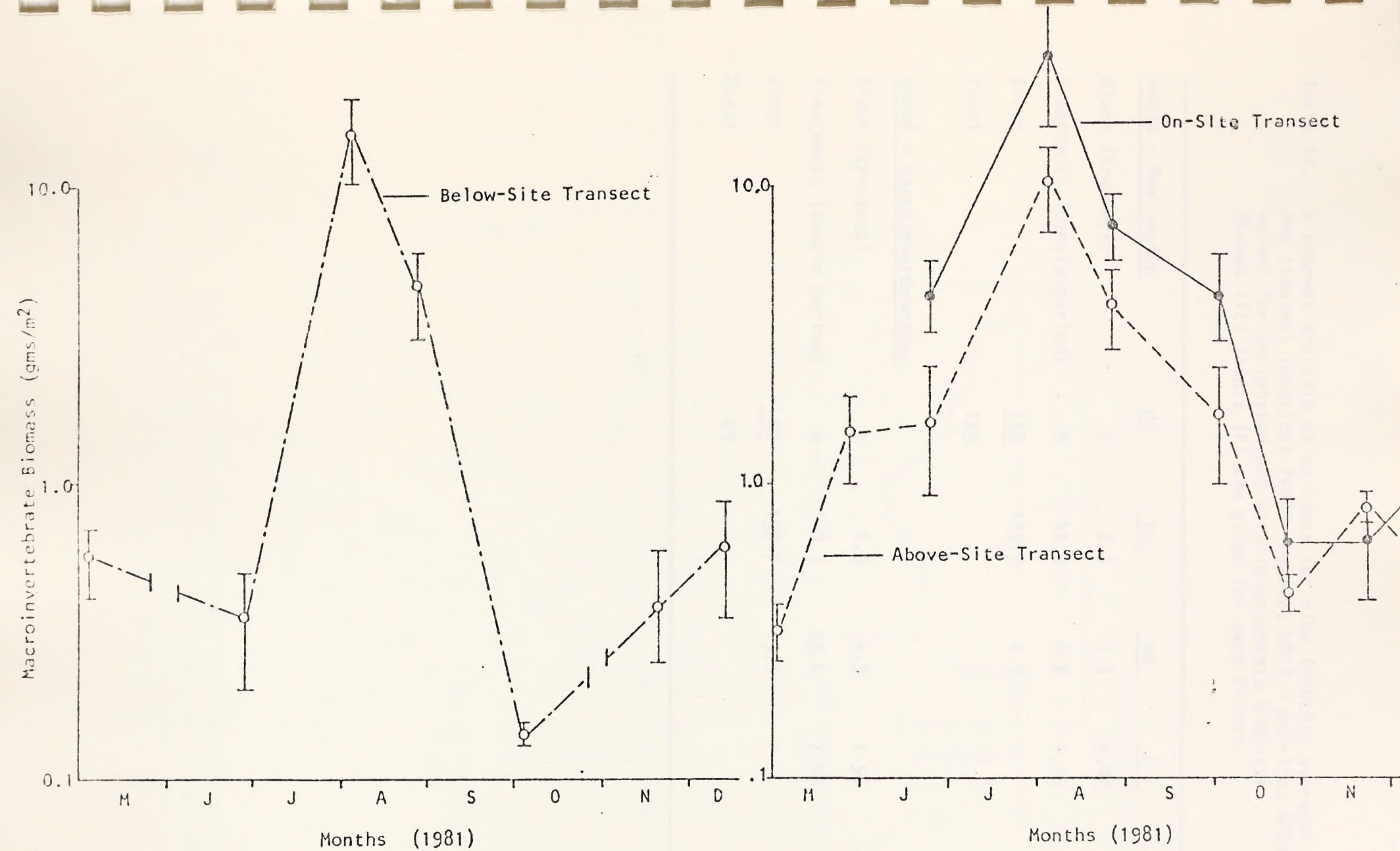


Figure 61. The monthly mean values (N=5) for macroinvertebrate biomass at the above, on-site, and below riffle transects on the White River, 1981. Standard errors are given for each transect.

Table 11. A two-way analysis of variance for time (monthly averages N=5) and transect locations (WR03 - above; WR18 - on-site; WR27 - below) for periphyton and macroinvertebrate biomasses. Probability levels (P) are given for each F-test.

ANOVA - Periphyton	df	ss	ms	F	P
Block (transect)	2	2.3	1.1	1.02	0.361
Treatments (sample period)	9	42.9	4.8	4.35	0.000
Error	118	129.5	1.1		
Total	129				
ANOVA - Macroinvertebrates					
Block (transect)	1	4.9	4.9	1.56	0.216
Treatments (sample period)	6	140.3	23.4	7.48	0.000
Error	62	193.7	3.1		
Total	69				

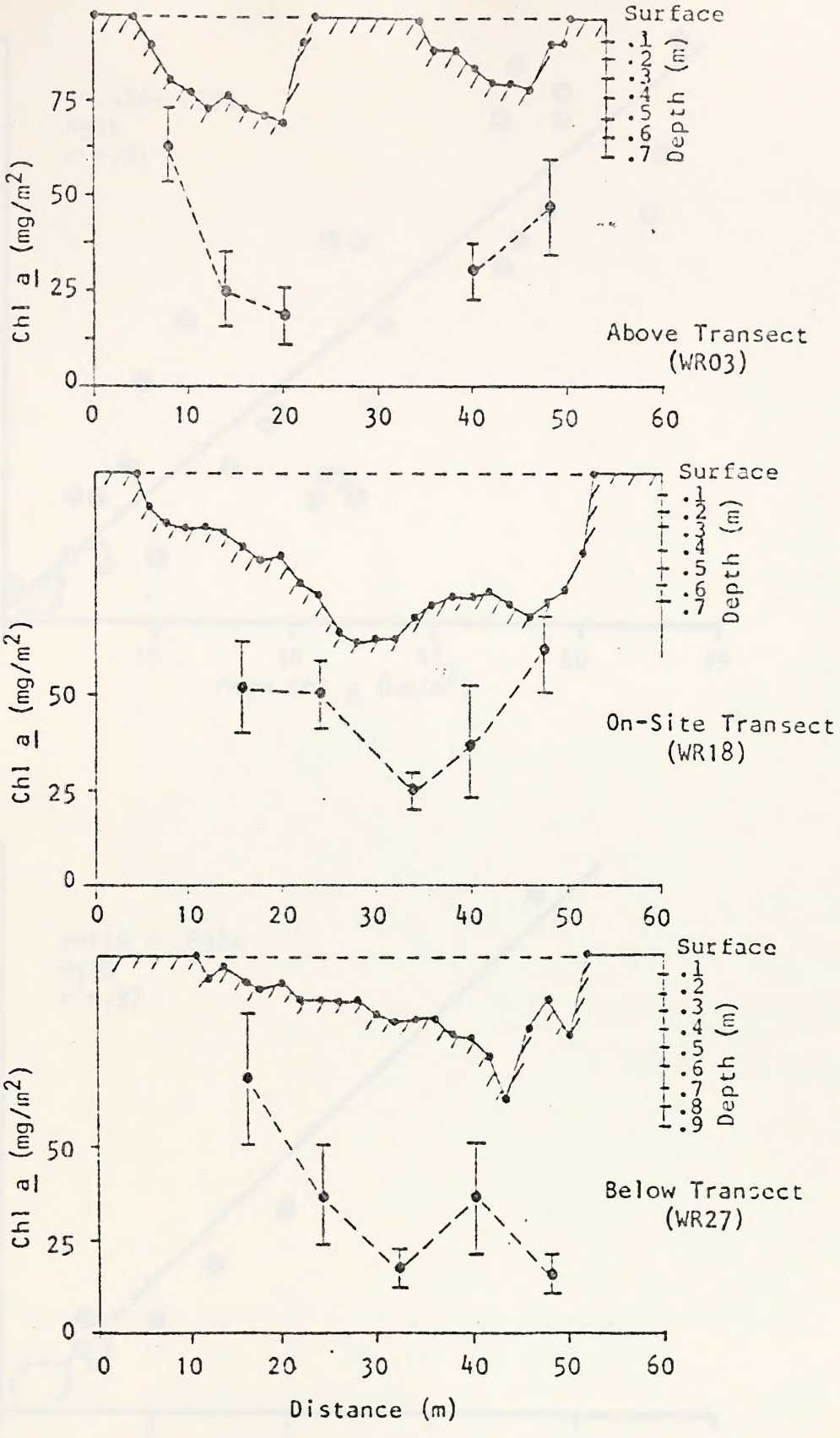
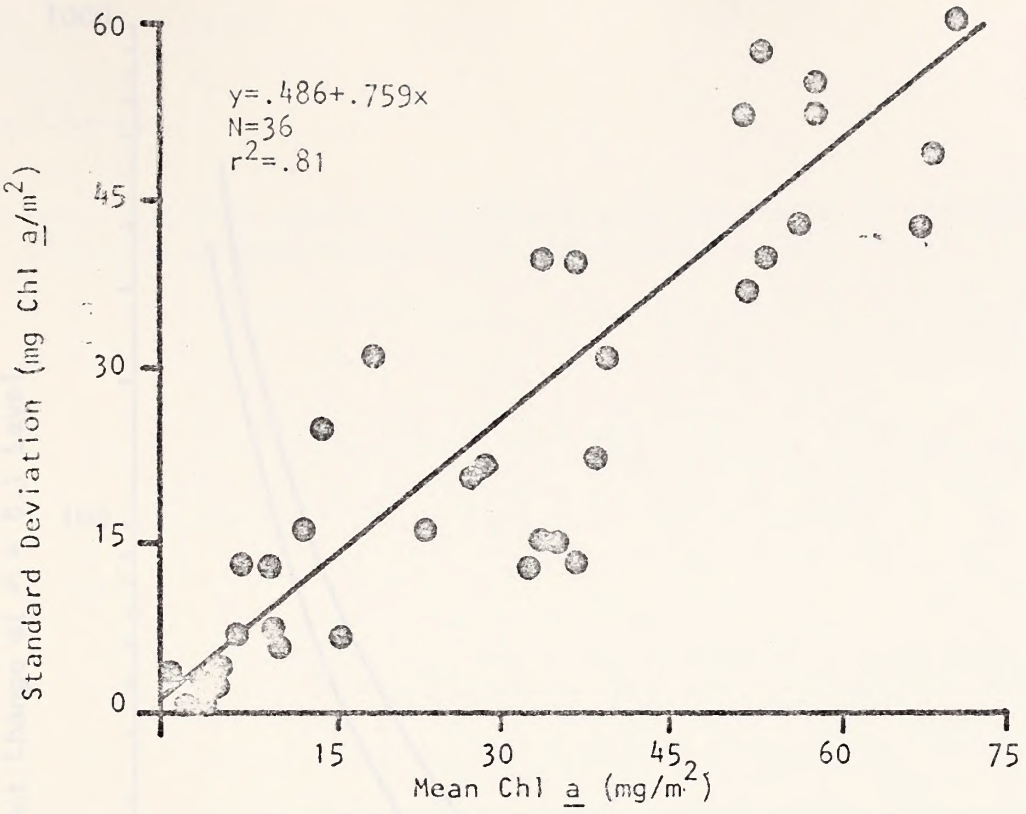


Figure 62. The distribution of annual mean (N=9) periphyton biomass estimates by distance across the stream for three riffle transects in the White River. Data collected in 1981. Standard errors are given.



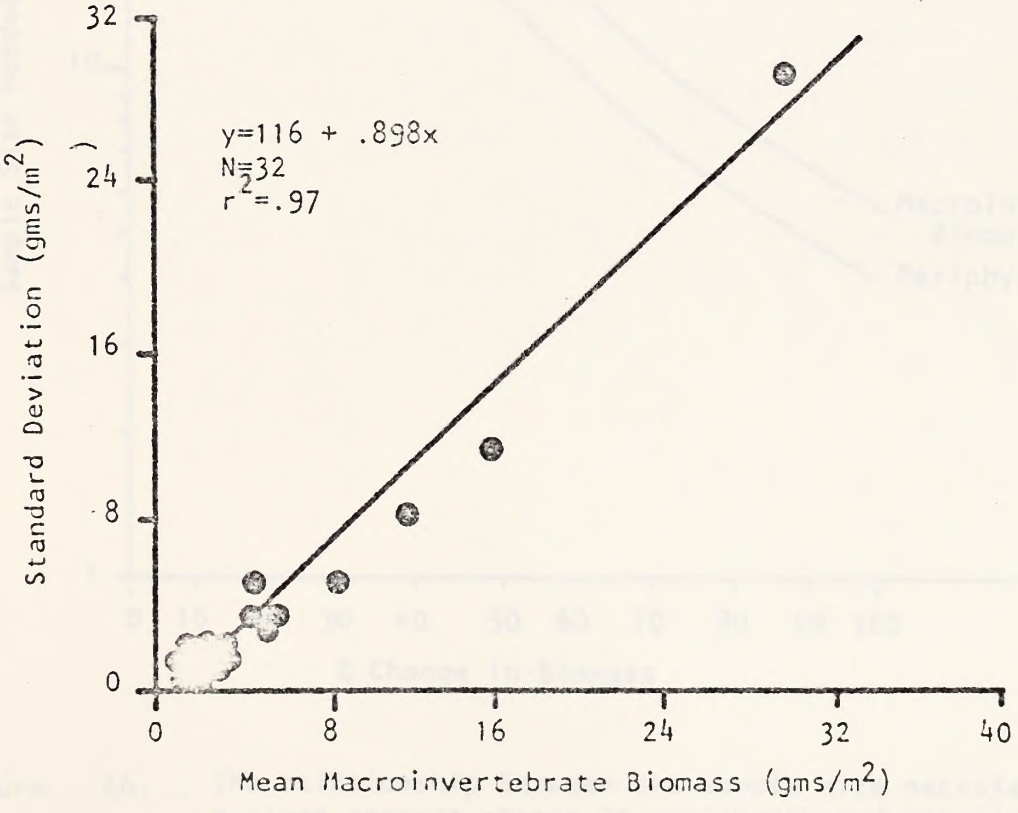


Figure 63. The relationships between biomass levels and the standard deviations for macroinvertebrate and periphyton biomass estimates in the White River, 1981.

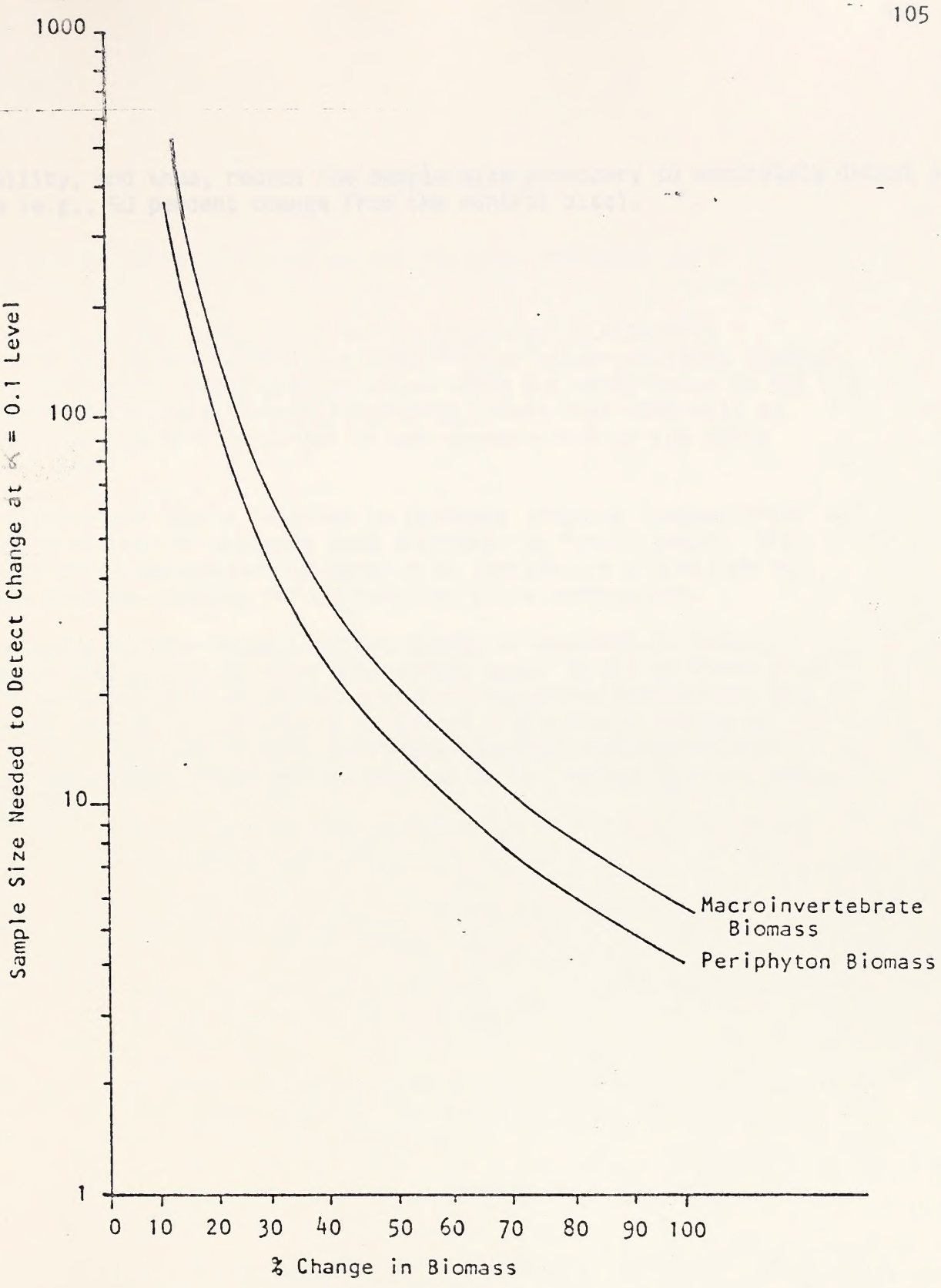


Figure 64. The relationship between the sample size necessary to detect a given percent change in periphyton and macroinvertebrate biomass in the White River. These curves are based upon data collected in 1981 with the variance of the control site. A two-tailed test was used in this analysis.

variability, and thus, reduce the sample size necessary to accurately detect a change (e.g., 50 percent change from the control site).

RECOMMENDATIONS

1. The present program design should continue unchanged until August, 1982.

2. Investigate new above tract and on tract riffle transects during the summer and fall of 1982. These sites should be sampled and calibrated to the present sites (WR03 and WR18) prior to the impact of dam construction activities. Note that WR03 will be inundated and WR18 destroyed by the construction of the White River Dam.

3. Consideration should be given to reducing sampling frequency for pool transects or dropping them entirely for future years. This would allow the monitoring program to concentrate efforts on the more diverse, stable, and productive riffle communities.

4. Beginning in late summer, effort should be expended to reduce sample variability by more intensively sampling all parameters in riffle habitats in different substrate types and determining the actual sample size necessary to detect a 50 percent change at $\alpha = 0.10$. The cost of this increased sampling could be offset by a concurrent reduction in sampling in the highly variable pool habitats.

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APPENDIX I

MATERIALS
AND
METHODS

Materials and Methods

Physical

Field

a. Stream Cross-sections: During the initial stream survey, permanent benchmarks were established above the flood plain and a stream profile was surveyed at two-meter intervals from benchmark to benchmark. A Leitz 2B level, tripod, and metric stadia rod were used in the survey. In the stream, dominant substrate was recorded at two-meter intervals and classified according to the size classes listed in Table I-1. At four-meter intervals, velocity at surface, bottom, and ten-cm depth intervals were recorded. During the initial stream survey, velocities were recorded at 30-cm intervals. At five equidistant locations on each river transect and three locations in Evacuation Creek, periphyton and interstitial sediment samples were collected using a 4.6-cm diameter plastic core tube. Sediment samples were stored in labeled Ziploc bags and transported to the laboratory for size fractioning and organic content analysis of the smallest size fraction. These were coordinated with survey points and velocity readings to insure a complete data set. Velocities were determined in cm/sec, using a Marsh-McBirney current meter with two, six, and 20 second time constants.

b. Light transmission was determined with depth using a one cm silicon blue photo cell (4 pi collector) embedded in an acrylic hemisphere with a white translucent covering. Light output was read on an ammeter with light transmission recorded as milli- or micro-amperes.

Laboratory

Interstitial sediment samples are dried at 105°C for 24 hours and sieved with U.S.A. Standard Testing Sieves into five size fractions (see Table I-2). The size fractions are weighed to the nearest 0.01 grams. The smallest size fractions were saved for organic content analysis.

Table I-1. The substrate size classification used in the field to estimate the dominant substrate.

Size Class	Largest Dimension
BOulder	> 10 cm
CObble	5 - 10 cm
RUbble	1 - 5 cm
PEa Gravel	0.5 - 1.0 cm
SAnd	0.1 - 0.5 cm
SIlt	0.01 - 0.1 cm
CLay	< 0.01 cm and compacted

Table I-2. The interstitial sediment size classifications used to determine the percent distribution of various sediment fractions.

#	Size (mm)
1	>12.7
2	4.0 - 12.7
3	0.5 - 4.0
4	0.25 - 0.50
5	<0.25 (silt)

Chemical

Field

One liter bottles were filled with water, frozen, and transported to the laboratory for analysis.

Laboratory

One hundred twenty-five mls of water was filtered for analysis of nitrite, nitrate, total dissolved phosphorous, and orthophosphate. The filters were wrapped in foil and refrigerated for later chlorophyll analysis.

Reactive nitrite was measured by the diazotization method and reactive nitrate content was obtained using the cadmium-reduction method in Strickland and Parsons (1968). The ascorbic acid technique, also described in Strickland and Parsons (1968), was followed for reactive orthophosphate determination. Total phosphorous was measured as described in Standard Methods (1971). Ammonia concentration was measured by the indophenol method by Solorzano (1969). Total suspended matter was obtained following the procedure described in Standard Methods (1981).

Turbidity was measured using a Bausch and Lomb Spectronic 88 spectrophotometer with a one cm cell. Transmittance was read at wavelengths of 435, 630, 645, 660, and 750 nm after shaking the sample.

Chlorophyll was measured by the fluorometric and trichromatographic procedures in Standard Methods (1980).

Organic

Field

Organic materials were collected from the White River with drift nets, benthic samples, and sediment samples.

Drift nets were set for approximately four-hour intervals. Floating debris collected, including invertebrates, in the drift nets was preserved in ten percent formalin and transported to the laboratory.

Benthic samples were taken at five equidistant locations along cross-stream transects in the river and three locations in Evacuation Creek with core tubes or a Hess Sampler. During a Hess sample, the river bottom was manually scraped for macroinvertebrates and organic debris present in the water column or on the bottom to an approximate maximum depth of ten cm. After collection, samples were preserved in ten percent formalin and transported to the laboratory. The procedures for sampling macroinvertebrates is discussed in more depth in the following invertebrates section.

Sediment core samples were collected corresponding with benthic and chlorophyll samples. A plastic core tube (4.6 cm dia) was used and the sample was stored in Ziploc bags and transported to the laboratory.

Laboratory

Drift and invertebrate samples were floated in a sugar-water solution and poured through a U.S.A. Standard Testing Sieve (No. 60) to separate the organic and inorganic materials. Samples greater than eight ounces weight were subsampled with a mechanical sampler. The organic material was then picked for insects and fishes. The remaining organic debris was dried at 105°C for 24 hours and weighed on a top-loading Sartorius Balance.

Substrate samples were dried at 105°C for 24 hours and sieved into five size fractions as described above. Approximately one ml of the silt fraction was redried in a crucible for 24 hours at 105°C, cooled in a dessicator to room temperature, and weighed to the nearest 0.00001 g on an analytical balance. The sample was then burned in a preheated muffle furnace at 600°C for 20 minutes, cooled, and reweighed. The weight loss on ignition was calculated as total organic content.

Primary Producers

Field

Rocks measuring at least 4.6 cm in diameter and exhibiting what was judged to be average algal coverage for each site were collected and frozen.

Where rocks were not found, i.e., gravel, sand, silt, etc., 4.6 cm diameter clear, plexiglas tubes were used for collection of specimens.

Laboratory

Rocks were thawed and a 4.6 cm diameter circle was scraped clean and suspended in 50 mls tap water and six drops of Lugol's solution. Clear tube periphyton samples were suspended in 50 mls of tap water, agitated, and allowed to settle until most large sediment particles settled out. The supernatant was decanted into a sample bottle to which six drops of Lugol's solution was added. Samples were then refrigerated in the dark until analysis. A blender was used to break calcareous and filamentous algae into countable units.

An automatic micropipette was used to withdraw a 50 microliter sample. The sample was placed on a glass slide with a cover slip. There was minute leakage around the cover slip but the advantage gained in being able to use a higher power objective was felt to make up for this possibility of error in count. Identification to lowest practical taxon was made and recorded for 50 to 110 fields of view per sample. Where filamentous forms prohibited this method, a Sedgwick Rafter counting cell was used at a lower power magnification.

Materials included Zeiss phase microscope, glass slides, cover slips, Sedgwick Rafter counting chamber, 50 lambda Eppendorf automatic pipette, five ml graduated pipette, and Lugol's solution.

Invertebrates

Field

Benthic macroinvertebrates were sampled along each transect at the above specified intervals across the river. These corresponded to chlorophyll (periphyton) and interstitial sediment sampling sites. Wherever possible, a Hess Sampler was used, sampling 1320 cm^2 of bottom with a net mesh size of 0.3 mm. Occasionally, deep water and/or slow current restricted use of the Hess Sampler. At these sites, sediment core samples were taken using a plastic core tube. Two sizes of tubes were used at different times, one sampling an area of 88.2 cm^2 and the other an area of 16.6 cm^2. Generally, when employing the smaller tube, three cores were taken at each site for a total area of 49.8 cm^2.

Insects floating in the water column were sampled using drift nets of two sizes (27.5 cm diameter with a 1.0 mm mesh size and 30.5 cm x 45.7 cm with a 0.5 mm mesh size) which were anchored to the river bottom and set in the water column at each transect during the time in which other samples were collected. Starting in July, drift samples were taken over a 24 hour period at a site below the Ignatio Bridge, and sampling of drift at each transect was discontinued. These nets were emptied at five or six regular intervals over a 24-hour sampling period.

All invertebrate samples were preserved in the field in ten percent formalin or 70 percent ethanol.

Laboratory

Drift and benthic samples were floated in a sugar-water solution and poured through a U.S.A. Standard Testing Sieve (No. 60) to separate organic and inorganic portions. This process was repeated three times. Samples greater than eight ounces and those with very large numbers of insects were subsampled using a mechanical subsampler (Waters 1969). The remaining organic materials were manually picked to remove all invertebrates and fish which were preserved in 70 percent ethanol. All organisms were then identified to lowest practical taxon, counted, and measured to the nearest mm (except exoskeletons). The remaining debris was then dried at 105°C for 24 hours and weighed on a Sartorius balance.

In order to determine insect biomass, a total of 300 insects from three orders (134 Trichoptera, 111 Diptera, and 55 Ephemeroptera) of one mm length increments were selected from samples taken on the White River. Each size group of each order was dried at 105°C for four hours and weighed to the nearest 0.01 mg on a Mettler analytical balance.

Production-Respiration

Field

Periphyton productivity in the White River was measured using Production/Respiration Chambers (Figure I-1). The chambers were made of 1/4" thick Lexan plastic. They were 29 cm x 29 cm and 19.7 cm deep. Water circulation through the chambers was accomplished with a Model 1-42 Little

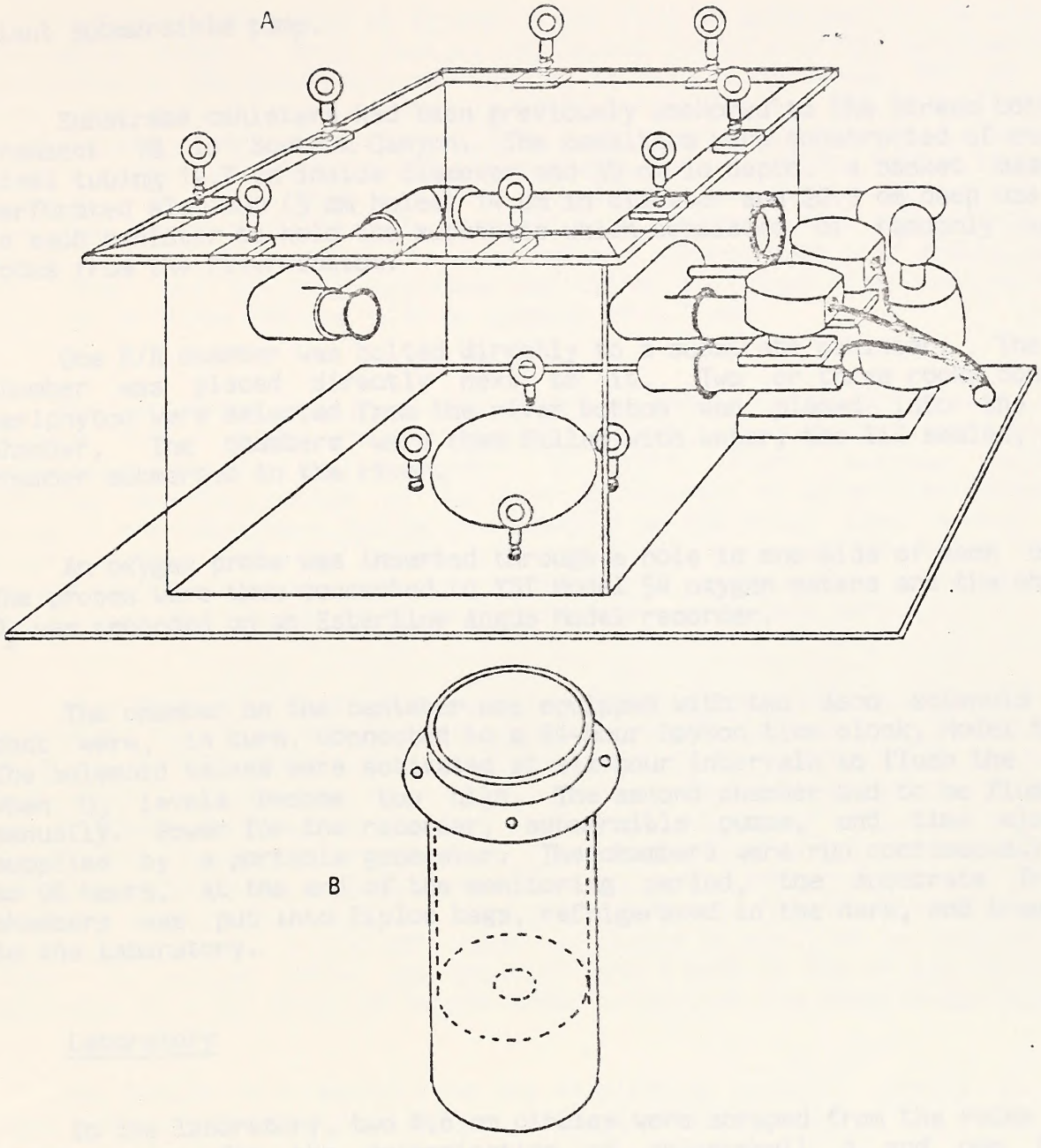

Figure I-1. A - Production-Respiration chamber used in the White River at Transect WR 18 to measure periphyton productivity. B - Substrate canister which is anchored to the stream bottom and bolted directly to the P/R chamber.

Giant submersible pump.

Substrate canisters had been previously anchored to the stream bottom at Transect 18 in Southam Canyon. The cansiters were constructed of stainless steel tubing 14.7 cm inside diameter and 35 cm in depth. A basket made from perforated aluminum (3 mm holes) 14 cm in diameter and 20.5 cm deep was placed in each canister to hold the substrate which consisted of randomly selected rocks from the river bottom.

One P/R chamber was bolted directly to a substrate canister. The other chamber was placed directly next to it. Two or three rocks containing periphyton were selected from the river bottom and placed into the second chamber. The chambers were then filled with water, the lid sealed, and the chamber submersed in the river.

An oxygen probe was inserted through a hole in one side of each chamber. The probes were then connected to YSI Model 54 oxygen meters and the change in O_2 was recorded on an Esterline Angus Model recorder.

The chamber on the canister was equipped with two Asco solenoid valves that were, in turn, connected to a 24-hour Dayton time clock, Model 6 x 758. The solenoid valves were activated at six-hour intervals to flush the chamber when O_2 levels became too high. The second chamber had to be flushed out manually. Power for the recorder, submersible pumps, and time clock was supplied by a portable generator. The chambers were run continuously for 72 to 96 hours. At the end of the monitoring period, the substrate from the chambers was put into Ziploc bags, refrigerated in the dark, and transported to the laboratory.

Laboratory

In the laboratory, two 4.6 cm circles were scraped from the rocks in each chamber, one for the determination of chlorophyll a and one for the identification of periphyton. The remaining area of the rocks was scraped to determine the total mass of chlorophyll a in each chamber. The volume that was displaced by the rocks in each chamber was also measured so that dissolved oxygen levels could be standardized to the volume of water present.

Decomposition

Leaf packs were used to determine the rate of decomposition of leaves in the White River. Dry, fallen, cottonwood leaves from the surrounding area were taken to the laboratory and leached in distilled water for 24 hours. They were then dried in an oven at 105°C for 24 hours. The dried leaves were weighed into groups of approximately 4 g on a Sartorius balance. The actual weights were recorded and the leaves were bound together with a brass paper fastener. They were then placed into individual containers (Figure I-2). The leaf pack containers were made from PVC pipe with an inside diameter of 6.5 cm and an overall length of 15 cm. One end was covered with 1/8" plexiglass and the other end of the container had either stainless steel mesh wire (3 mm opening) or Nitex netting with an opening of 0.7 mm.

The containers were placed in stainless steel racks in groups of six (Figure I-2). Each group contained three large mesh containers and three containers with the smaller mesh. To start, four six-group packs were anchored to the bottom of the river at Transect 18 in Southam Canyon. One group of six containers was to be removed from the river each month. Every other month, four new racks of containers were to be added and the decomposition rate monitored over a four-month period. This would give four repetitions and a total of 96 leaf packs over a one-year period.

When the containers were removed from the river, each pack of leaves was monitored for a period of 10-15 minutes to determine the respiration rate of organisms on the leaves. This was accomplished by taking the leaves from individual containers and placing them in a respiration chamber of approximately the same size as the leaf pack container (Figure I-2). The respiration chamber was made from a piece of PVC pipe 6.5 cm inside diameter and approximately 15 cm long. On either end was a PVC cap with a piece of plastic tubing connected to a Model P-AAA Little Giant submersible pump for water circulation. A 1/4" NPT valve was placed in the line to control water flow. An oxygen probe was inserted into a hole in one of the PVC caps and connected to a YSI Model 54 oxygen meter to monitor the respiration rate.

The leaves were removed from the respiration chamber, placed in Ziploc bags, and refrigerated for transportation to the laboratory.

In the laboratory the leaves were once again leached in distilled water for a period of 24 hours, then oven dried at 105°C for an additional 24 hours. They were then weighed to determine the amount of weight loss.

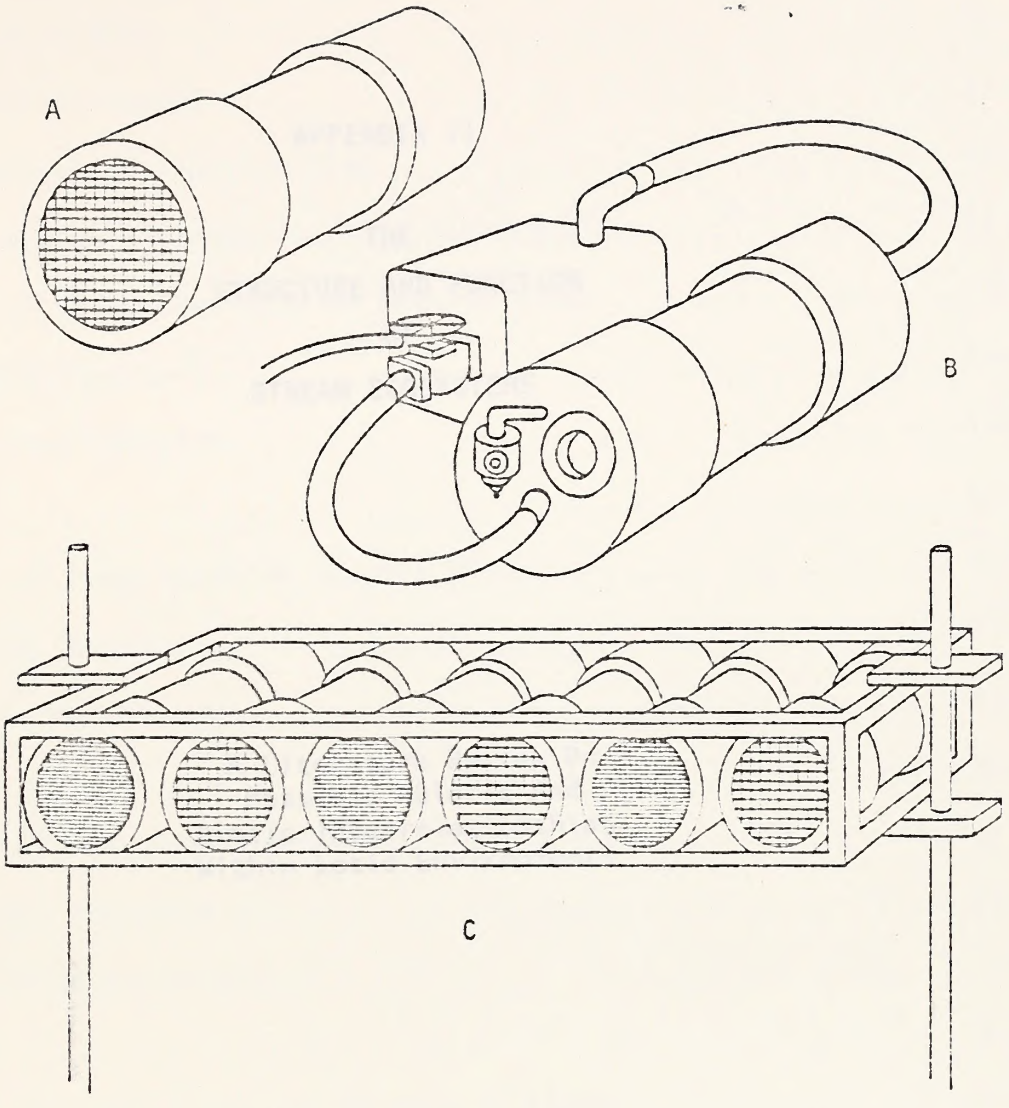

Figure I-2. A - Individual leaf pack container used on the White River at Transect WR 18 to determine leaf decomposition. B - Leaf respiration chamber. C - Group of six leaf packs in a rack which is anchored to the stream bottom. Note: Small and large mesh sizes.

APPENDIX II

THE STRUCTURE AND FUNCTION OF STREAM ECOSYSTEMS

A Literature Review On
Possible Abiotic And
Biotic Interrelationships
Within Lotic Environments

Prepared by

ECOSYSTEM RESEARCH INSTITUTE

Prepared for

WHITE RIVER SHALE OIL CORPORATION

May, 1982

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Influence of Current Velocity on Abiotic and Biotic Factors

Substrate Size

The velocity of flowing water is determined by the interaction of two factors, gravity and friction (Leopold et al. 1964). The gravitational force moves the water downslope, whereas resistance to this movement is provided by the friction between the water and the channel bed or banks. Dominance by either of these factors may cause the water to accelerate or decelerate. In most natural systems there is no change in the average water velocity over long distances, although local variations have been shown to occur (Hynes 1970).

Water velocity is not distributed evenly from the stream surface to bed. Drag, exerted by the streambanks and surface tension of the water, decreases the velocity of the water near the surface and the interface with the stream bed. The maximum velocity is, therefore, usually below the surface (Figure II-1), below which the velocity decreases logarithmically to the bottom where it approaches zero (Leopold et al. 1964). Obstructions and turbulent flow increase the thickness of the layer of dead water (velocities near zero). This layer of dead water is important biologically, providing resting areas for fish, attachment sites for periphyton and aquatic insects, and catchment areas for detritus.

Water flowing through a channel exerts a shear stress on the bed and banks (Leopold et al. 1964). The extent of material removed depends upon the velocity of the water, substrate size, and the cohesiveness of the material (Hynes 1970; Novak 1973). Cohesiveness of the material is a major factor in determining the water velocity necessary to move particles. It has been shown that initiating movement in fine sand is easier than in clay, although the particles are larger (Table II-1). Turbulence aids in erosion by lifting particles from the streambed into the main flow, where they are suspended until the velocity decreases and deposition occurs (Hynes 1970). At some point in time, an equilibrium between removal and deposition results and a stable channel is formed (Leopold et al. 1964). The energy dissipation equations presently used predict that stable, natural channels will be sinusoidal. This sinusoidal shape tends to remain constant, although the exact position of the channel within the flood plane will change. As the stream flows within this sinusoidal pattern, material is eroded from the insides of the river bends and is deposited along the outside edges. This erosion and deposition generates alternating shallow and deep sections along the streambank (Figure II-2), corresponding to the typical pool and riffle areas.

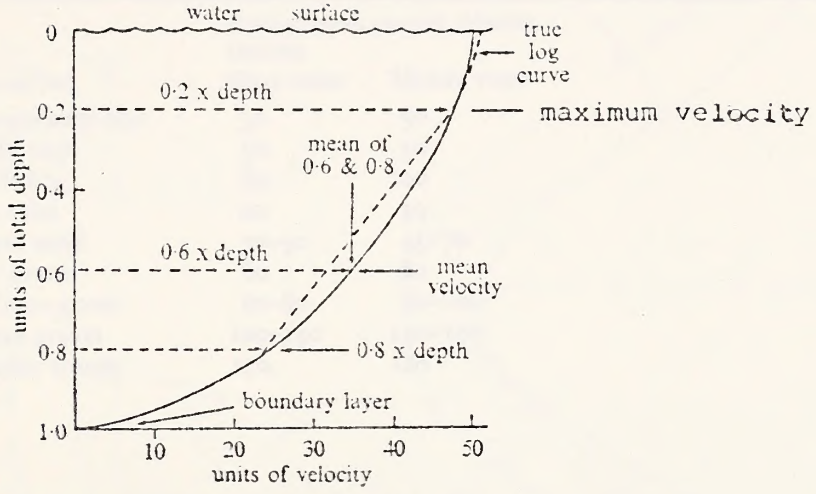


Figure 11-1. The rate of flow of water at different depths in an open channel, and depths at which the mean flow can be measured (from Leopold et al. 1964).

Table II-1. The mean current velocity of clear and muddy water required to initiate movement along a stream bed of various types of bottom deposit (from Hynes, 1970).

Type of bed	Critical mean current velocity cm./sec. Clear water	Muddy water
Fine-grained clay	30	50
Sandy clay	30	50
Hard clay	60	100
Fine sand	20	30
Coarse sand	30–50	45–70
Fine gravel	60	80
Medium gravel	60–80	80–100
Coarse gravel	100–140	140–190
Angular stones	170	180

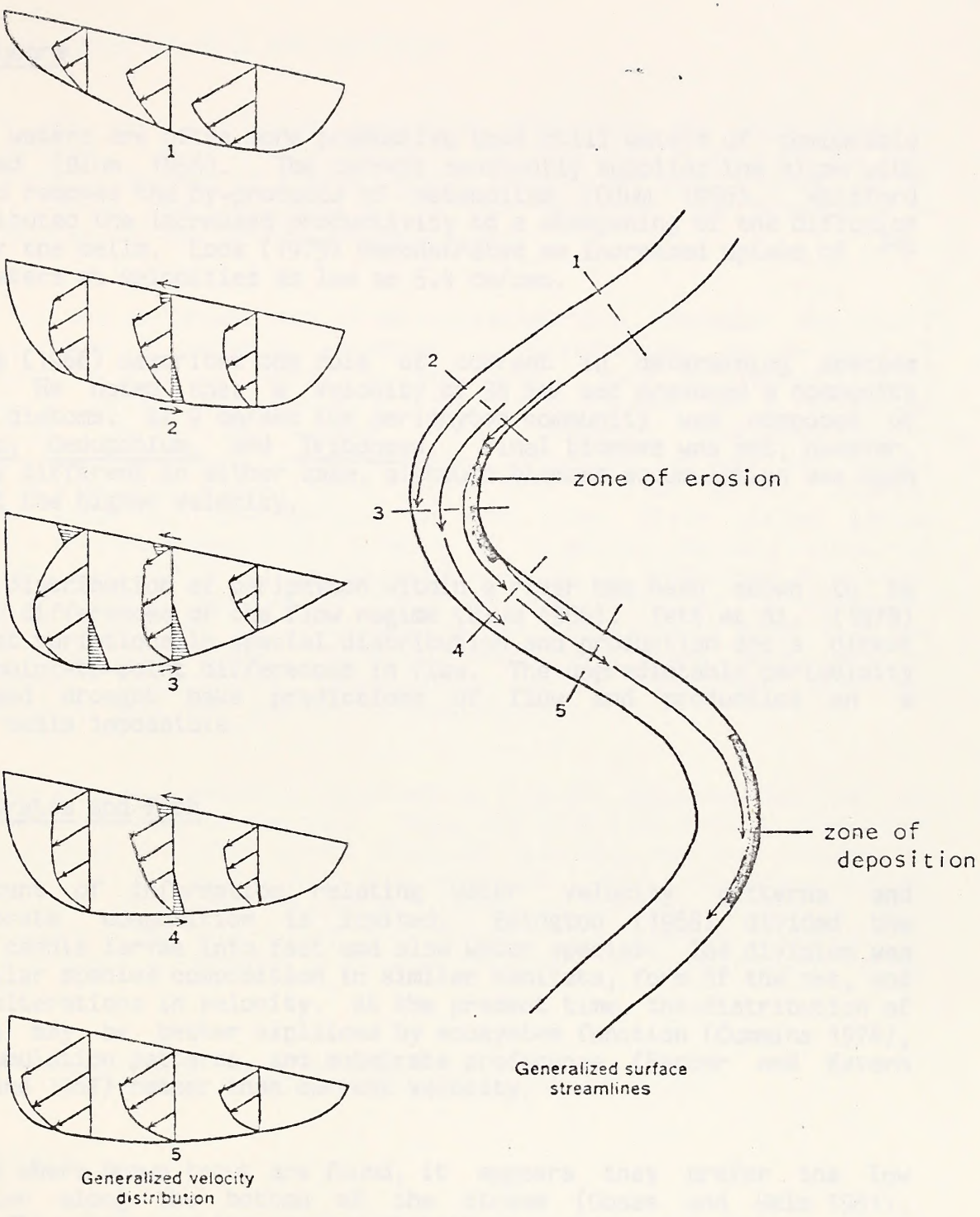


Figure II-2. Isometric view of generalized diagram of flow distribution in a meander (modified from Leopold et al. 1964).

Primary Producers

Flowing waters are often more productive than still waters of comparable nutrient load (Blum 1956). The current constantly supplies the algae with nutrients and removes the by-products of metabolism (Odum 1956). Whitford (1960) attributed the increased productivity to a steepening of the diffusion gradient near the cells. Lock (1979) demonstrated an increased uptake of ^{32}P in flowing waters at velocities as low as 5.4 cm/sec.

McIntire (1966) described the role of current in determining species composition. He noted that a velocity of 38 cm/ sec produced a community dominated by diatoms. At 9 cm/sec the periphyton community was composed of Stigeoclonium, Oedogonium, and Tribonema. Final biomass was not, however, significantly different in either case, although biomass accumulation was much more rapid at the higher velocity.

Spatial distribution of periphyton within a river has been shown to be determined by differences of the flow regime (Blum 1956). Tett et al. (1978) concluded that variations in spatial distribution and production are a direct result of point-to-point differences in flow. The unpredictable periodicity of floods and drought make predictions of flow and production on a year-to-year basis impossible.

Macroinvertebrates and Fish

The amount of information relating water velocity patterns and macroinvertebrate composition is limited. Edington (1968) divided the net-spinning caddis larvae into fast and slow water species. The division was based on similar species composition in similar habitats, form of the net, and response to alterations in velocity. At the present time, the distribution of invertebrates may be better explained by ecosystem function (Cummins 1974), detrital accumulation patterns, and substrate preference (Barber and Kevern 1973; Ulfstrand 1967) rather than current velocity.

In areas where brown trout are found, it appears they prefer the low velocity water along the bottom of the stream (Gosse and Helm 1981). Variation in flow explained the clumped distribution observed in this species. An alteration in habitat by channelization resulted in biomass changes. Seethaler (1978) found no apparent velocity preference in adult squawfish except during spawning. During this time, males captured were predominately in backwaters and eddies, while females were assumed to be in deep pools.

Interaction of Substrate Size and Organisms

Much of the work involving substrate preference has been done on macroinvertebrates. An excellent review of the available literature was done by Cummins (1966). Studies by Thorup (1966), Cummins and Lauff (1969), Barber and Kevern (1973), and Williams (1978) all describe the preference of macroinvertebrates for different substrate sizes.

A study of six different taxa of macroinvertebrates (Cummins and Lauff 1969) demonstrated a definite selection of microhabitat on the basis of substrate particle size in three of the taxa. Work by Thorup (1966) demonstrated a certain dependence of macroinvertebrates on a particular substrate size. Barber and Kevern (1973) compared macroinvertebrate distribution with substrate particle size throughout the year and found a shift in preference with season (Table II-2). In a similar manner, Pearson (1967) noted that the invertebrates of the Green River showed marked differences in species richness and numbers of organisms between substrate types (Tables II-3 and II-4). Since substrate size is determined by current velocity, the effects of the two on organisms are difficult to separate. Substrate size selection in artificial stream channels with equal flow was demonstrated by Williams (1978). Three different gravel sizes were used in the experiment. Significant relationships between substrate size and abundance of major taxa were calculated (Table II-5).

Substrate size is apparently important to fish at spawning but may be unimportant the remainder of the year. Gosse and Helm (1981) found that brown trout (Salmo trutta) preferred gravel areas for spawning, but were nonspecific at other times of the year. Data from Seethaler (1978) indicates a similar preference pattern in Colorado squawfish. Young- of-the-year squawfish were found in backwater areas with a firm silt bottom. This apparent preference of juveniles is probably a response to current velocity or temperature rather than substrate.

Influence of Water Chemistry on Aquatic Biota

Primary Producers

Cladophora glomerata is one of the dominant algal species worldwide in both lotic and lentic ecosystems (Whitton 1970a). This filamentous green algae has an apparent need for a steep diffusion gradient provided by flowing water (Whitford 1960), because it is rarely found in completely still water. Whitton (1970a) reports an avoidance of waters low in phosphorus and nitrate

Table II-2. Average estimate of individuals/m^2 at each time period (TP) associated with small and large substrates from a riffle area of the Pine River (n = number of samples) (from Barber and Kevern 1973).

n	6/3 - 7/6 TP1		7/22 - 8/2 TP2		8/19 - 9/2 TP3		9/16 - 10/6 TP4		10/4 - 11/29 TP5		3/15 - 4/11 TP6		4/25 - 5/12 TP7		5/23 - 6/7 TP8		6/24 TP9	
	Small 2	Large 3	Small 6	Large 1	Small 2	Large 3	Small 2	Large 1	Small 2	Large 4	Small 1	Large 4	Small 4	Large 3	Small 4	Large 3	Small 3	Large 3
Ephemeroptera	3,252	589	1,790	3,767	2,571	1,918	5,521	3,818	2,471	2,811	3,159	1,874	4,318	7,586	2,676	6,755	6,085	7,752
All Diptera	26,340	15,851	58,811	93,919	59,889	23,665	63,876	60,765	97,279	85,682	44,010	68,022	46,819	47,361	35,449	38,459	36,671	41,704
Chironomidae	22,692	10,628	57,065	89,571	59,119	22,667	61,595	47,143	95,125	81,151	42,546	63,385	43,550	43,944	29,926	33,959	34,568	37,516
Simuliidae	343	2,595	45	36	0	4	514	42	566	280	271	716	5,068	1,059	2,719	908	50	635
Other Diptera	3,305	608	1,700	4,312	770	994	2,168	3,578	3,588	2,248	1,193	3,921	1,400	2,358	2,805	2,602	2,073	3,554
Trichoptera	8,661	3,250	6,256	9,277	1,000	5,434	4,117	9,905	2,262	3,469	2,668	4,312	3,052	5,195	2,233	3,987	2,918	8,949
H. borealis	343	75	1,479	0	882	2,496	924	2,239	468	278	201	581	565	1,003	412	485	371	683
Coleoptera	1,220	1,517	1,427	3,364	2,802	1,918	1,170	1,624	1,868	2,709	4,923	2,717	1,364	1,864	2,219	1,061	1,299	778
Hydracarina	1,048	166	1,266	1,955	3,122	1,667	2,414	3,964	2,542	2,102	299	3,912	2,037	3,019	1,646	4,465	766	1,406
Plecoptera	18	181	111	213	495	52	1,060	433	220	886	308	270	1,017	174	658	895	154	31
TOTAL	40,559	19,534	69,661	112,495	69,879	34,663	66,958	70,505	106,442	95,659	55,587	81,107	58,607	65,199	44,881	55,600	47,883	60,620

Table II-3. Abundance of benthic invertebrates on silt or sand substrates, Island Park, Green River, 1964-1965 -- expressed as mean number per square foot (from Pearson, 1967).

Item	1964						1965				
	June 16	July 4	July 30	Aug. 17	Sept. 9	Nov. 28	May 25	June 20	July 15	Aug. 12	Sept. 1
Oligochaeta	0					2			12	1	
Plecoptera											
Isoperla						1					
Ephemeroptera											
Baetis sp. I						1			1	3	
Tricorythodes minutus			1								
Hemiptera											
Corixidae				1							
Trichoptera											
Cheumatopsyche					1				1		
Diptera											
Tipulidae								1			
Simuliidae								1			
Chironomidae			22	6	3	14	1	2	8	5	4
Ceratopogonidae		1				1		20			3
Pupae (all families)		1						1	1		1
Number of Samples	1	2	3	2	3	1	3	4	4	3	4

Table 11-4. Abundance of benthic invertebrates on rubble or debris substrates, Island Park, Green River, 1964 -- expressed as mean number per square foot (from Pearson, 1967).

Item	June 16	July 4	July 30	Aug. 17	Sept. 9	Nov. 1	Nov. 28
Oligochaeta		1		1	1	1	
Hydracarina		6	1	1			
Plecoptera							
Isoperla	1					5	
Isogenus					2		
Ephemeroptera							
Heptagenia elegantula	2		3		1		
Rhithrogena undulata			1				
Baetis sp. I	2		7		1	2	3
Baetis sp. IV			3				
Baetis sp. V	1		3				2
Baetis sp. VI	1						
Baetis sp. VIII	1		13				
Baetis sp. XIV		1					
Tricorythodes minutus		6	31	6	4		
Tricorythodes sp.			1		1		
Ephemerella inermis		3			2		
Choroterpes albiannulata			1	1	1		1
Traverella albertana			4				
Ephoron album				1	1		
Trichoptera							
Cheumatopsyche	1	2	5	1	10		
Hydropsyche					2		
Leptocella		2					
Brachycentrus		2			1		
Coleoptera							
Dytiscidae					1		
Elmidae		1					
Diptera							
Simuliidae		13					
Chironomidae	5	22	22	89	65	53	367
Ceratopogonidae		2		1		1	
Pupae (all families)	1	1	2	34	11	12	22
Number of Samples	3	2	3	3	5	2	1

Table II-5. Significant relationships between major taxa and substrate size. S-small (11.5 mm), M-medium (24.2 mm), L-large (40.8 mm). First, second, and third choices are indicated. ($P < 0.05$).

	S	M	L	Size preference
Baetis sp. A	2	1	1	upper
Prosimulium sp.	2	1	1	range
Paracamptus ?reggiae	1	1	2	lower
Brachyptera sp.	1	1	2	range
Micropsectra/ Tanytarsus spp.	1*	1	2*	
Thienemanniella sp. B	1	1	2	
Nematoda	2	1	3	medium
Nais variabilis	2	1	3	range
Chaetogaster diaphanus	2	1	2	
Rheotanytarsus sp.	3	1	2	
Phaenopsectra/ Polypedilum spp.	2	1	3	
Thienemanniella sp. C	2*	1*	2	
Eukiefferiella sp. C	2	1	2	
Eukiefferiella sp. A	1	1	1	none
Eukiefferiella sp. B	1	1	1	
Baetis parvus	1	1	1	

*$P < 0.10$.

by Cladophora and an increase of standing crop in those waters where nutrient enrichment has occurred. In the Great Lakes, Cladophora filaments of 25 cm in length have been observed near the sewage outfall areas of several major cities (Herbst 1969).

In lotic systems, nutrient limitation rarely occurs because the supply of nutrients is constantly being replenished (Blum 1956). Phosphorus enters flowing waters through a variety of mechanisms. For example, Kemp (1969) noted that phosphorus compounds can enter the system by groundwater, from atmospheric particulate material, and by surface drainage from surrounding land. In developed areas, the amount of phosphorus entering the stream through surface drainage is related to the degree of urbanization. In general, denser populations added more phosphorus to the surface water (Keup 1968).

Blum (1956) described the flora of waters low in calcium, (a requirement by a majority of algae), as "peculiar" with the community composition containing many Myxophyceae. High concentrations of calcium and magnesium were not inhibitory, but low concentrations of these ions resulted in cell lysing and death. Cladophora is apparently favored by hard or very hard water (Whitton 1970a).

Blum (1956) states that neutral or slightly elevated pH values appear to be required for the majority of algal species. Values of 7.0 to 10.0 appear to be optimum for Cladophora. Whitton (1970a) reports that Cladophora species are rarely found in waters where the pH is outside of this range.

Heavy metals, even at very low concentrations, are apparently toxic to Cladophora glomerata (Whitton 1970a). Zinc and copper appear to be the most toxic of the metals examined. Whitton (1967) found zinc to be toxic at concentrations of <0.1 mg/l and copper to be toxic at 0.2 mg/l. Whitton (1970b) examined 37 populations of Chlorophyta for responses to zinc, copper, and lead. Of the species studied, Cladophora glomerata was one of the most sensitive. The toxicity of metals to Cladophora glomerata was found to be reduced by EDTA or high salinity. Betzer and Kott (1969) conducted bioassays using saline water and found that 10 mg/l of copper sulfate over a four-day period was necessary to effect a complete kill. This represented elevated levels of the metal in saline water for the same toxic response noted in waters with less TDS.

Macroinvertebrates

Three chemical factors appear to influence macroinvertebrate distribution: (1) cation concentrations, (2) pH, and (3) dissolved oxygen. Minshall and Minshall (1978) found distribution of Gammarus pulex significantly correlated with potassium concentrations. Low potassium concentrations have been shown to cause a cessation of feeding in Gammarus pulex. Minshall and Minshall (1978) speculated that "the uptake of water, in an effort to maintain its internal ion content had given Gammarus the feeling of being 'full,' and therefore canceled its stimulus to feed." Various authors (e.g., Egglishaw 1968) have attempted to correlate macroinvertebrate distribution with calcium concentrations. The results of these studies are inconclusive.

Low pH is apparently not a problem in unperturbed natural waters. Flow-through bioassays conducted by Gaufin (1973) found that the test organisms died at pH values below those normally encountered (Table II-6). Low toxic pH levels are possible in streams draining coal mine areas. Mayflies (Ephemeroptera) appear to be most sensitive to low pH followed by stoneflies (Plecoptera) and caddisflies (Trichoptera). Sensitivity is increased during molting and emergence (Gaufin 1973).

Dissolved oxygen may be the most important factor regulating macroinvertebrate distribution (Gaufin 1973). The level at which dissolved oxygen becomes critical depends on current velocity, community structure, and metabolic needs. Aquatic insects have been shown to survive at reduced dissolved oxygen levels if the current velocity is increased. Gaufin (1973) reports a mean TL_M for 11 species of 3.64 mg/l at a flow of 500 cc/min and 2.55 mg/l at 1000 cc/ min. Similar increased survival was reported by Eriksen (1966). Different populations of the same species apparently had different dissolved oxygen requirements. For example, insects from Montana waters had markedly higher TL_M concentrations than similar insects from Utah (Table II-7). At low dissolved oxygen concentrations, some aquatic invertebrates are able to anaerobically metabolize food stuffs (Davis 1975). Additionally, certain organisms are able to reduce their oxygen consumption to match the concentrations in the water (Eriksen 1966; Davis 1975). These two factors make the establishment of dissolved oxygen criteria for aquatic invertebrates a difficult task.

Fish

Trout and other cold water fish have many of the same preferences as macroinvertebrates. Data from Davis (1975) indicates a dissolved oxygen

Table II-6. pH values at which 50% of the test species died after 96 hours exposure (TLm^{96}), Flathead Lake, Montana, 1968-69 (from Gaufin, 1973).

Species tested	pH Values	Mean
EPHEMEROPTERA		
Ephemerella doddsi Needham	4.95	5.13
	5.35	
Leptophlebia sp.	5.30	5.21
	5.11	
Hexagenia limbata Guerin	6.40	5.90
	5.40	
Cinygmula par Eaton	6.25	6.04
	6.00	
Rhithrogena robusta Dodds	6.35	6.35
Heptagenia sp.	6.25	6.18
	6.11	
PLECOPTERA		
Arcynopteryx parallela Frison	5.50	5.33
	5.16	
Pteronarcys californica Newport	5.12	4.60
	4.19	
Pteronarcella badia (Hagen)	4.90	4.37
	4.19	
Isogenus aestivalis (Needham and Claassen)	5.40	5.15
	4.90	
TRICHOPTERA		
Limnephilus ornatus Banks	2.72	2.83
	2.94	
Hydropsyche sp.	3.60	3.34
	3.10	
DIPTERA		
Simulium vittatum Zetterstadt	3.68	3.64
	3.59	
AMPHIPODA		
Gammarus limnaeus Smith	7.31	7.29
	7.28	
	7.20	7.27
	7.34	

Table II-7 . Average minimum dissolved oxygen requirements of different groups of aquatic invertebrates* (from Gaufin 1973).

	Montana species	Average survival (days)	Utah species	Average survival (days)
Plecoptera	4.9 mg/l	62	2.8 mg/l	14
Ephemeroptera	4.6 mg/l	30	3.3 mg/l	10
Trichoptera	4.0 mg/l	85	3.1 mg/l	48
Diptera	2.4 mg/l	40	2.2 mg/l	92
Odonata			2.2 mg/l	39
Amphipoda	2.8 mg/l	20		

* Averages based on 50% + survival for time indicated.

requirement for most trout at or above approximately 5 mg/l. Below this level physiological responses such as increased breathing amplitude and elevated buccal pressure occurred (Hughes and Saunders 1970). The presence of respiratory inhibitors raises the dissolved oxygen requirement considerably. Downing (1954) found that rainbow trout (Salmo gairdneri) exposed to cyanide died much faster if the dissolved oxygen concentration was below 9.74 mg/l. The toxic effects of zinc, copper, lead, and phenols were increased at dissolved oxygen concentrations below 5.78 mg/l (Lloyd 1961). Salmonid eggs also were found to require greater than or equal to 7 mg/l O_2 to develop normally (Davis 1975).

Fish apparently are not as sensitive to low pH as macroinvertebrates. Brook trout (Salvelinus fontinalis) appeared unaffected by water with pH values of 4.1 (Creaser 1930). Butler et al. (1973) conducted bioassays with acid mine water using a variety of species and found no significant behavioral response at a pH of 4.0. No data were available for the endemic Colorado River fishes.

Influence of Temperature on Aquatic Biota

Primary Producers

Increased temperature affects the metabolism of aquatic organisms. Phinney and McIntire (1965) found that a temperature increase from 11.9 to 20.0°C increased gross community oxygen evolution 30 percent (from 335 to 447 mg O_2/ m^2/hr). When the intensity of the light was reduced from 22,000 lux to less than 11,000 lux, oxygen evolution did not increase with temperature.

A study of the effects of nuclear power plant thermal effluents (Foerster et al. 1974) noted increased algal standing crops between pre- and postintroduction of effluent. They also noted a significant difference in summer standing crops between the control station and the thermal outfall. The response of unialgal cultures of Cladophora glomerata to increased temperature was studied by Bellis (1968). Cladophora grew increasingly well at temperatures between 15 and 30°C. Above 30°C vegetative cells were killed; below 15°C, little or no growth occurred.

Temperature-related shifts in community composition and species dominance have also been shown to occur. The classic diatom, green, blue-green succession has been described by many authors (e.g., Wright et al. 1970) and is illustrated in Figure II-3. A similar shift in species dominance from red algae to blue-green was described by Wilde and Tilly (1981). Squires et al.

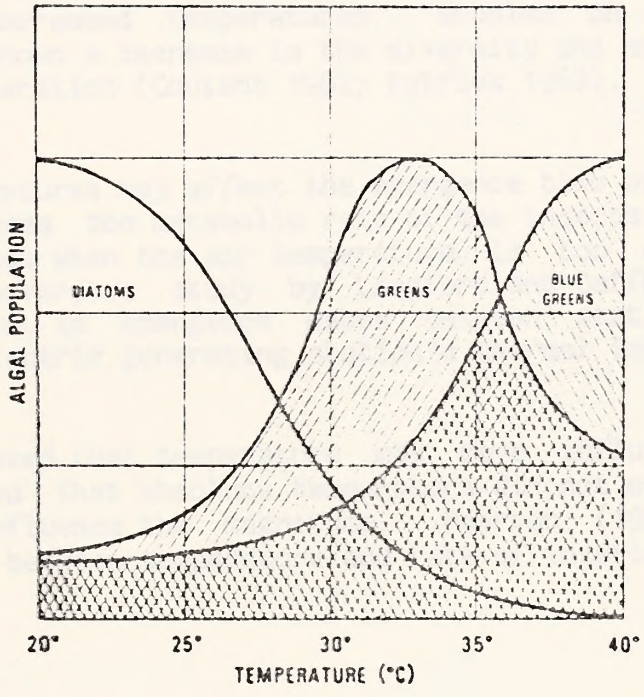


Figure 11-3. Algal population shifts with temperature (from Wright et al. 1970).

(1979) noted reduced diversity in the Provo River diatom community below a thermal discharge. He noted an increased production of Cladophora glomerata and elimination of Hydrurus foetidus from the community.

Macroinvertebrates

Gaufin (1973) studied the effects of increased temperature on a variety of aquatic insect species. The Ephemeroptera were most susceptible to temperature increases, followed by Plecoptera, Trichoptera, and Diptera (Table II-8). It was also noted that species obtained from cold, running waters had lower tolerances to increased temperatures. Studies on the Delaware and Potomac Rivers have shown a decrease in the diversity and numbers of organisms after temperature alteration (Coutant 1962; Patrick 1968).

Increased temperatures may affect the emergence time of aquatic insects. Warmer water increases the metabolic rate of the insects (more degree-days) and may cause emergence when the air temperature is too cold for survival (Gaufin 1973). However, a study by Langford and Daffern (1975) found no significant difference in emergence dates between stations upstream and downstream from an electric generating station's thermal input.

Waters (1962) noted that temperature may have influenced invertebrate drift. He concluded that absolute temperature did not affect the timing of the drift, but did influence the magnitude. Pearson (1967) also found a positive correlation between temperature and rate of invertebrate drift (Table II-9).

Fish

Temperature changes may be a direct cause of fish mortality or may also enhance the toxicity of other pollutants (Cairns et al. 1975). Every aquatic organism has a temperature range where life functions can be maintained with a preferred optimum temperature (within this range). This optimum may be affected by acclimation, but lethal limits are genetically fixed. Theories about the cause of temperature-related mortality in fish include: (1) tissue anoxia, (2) failure of osmoregulatory processes, (3) protein denaturation, (4) release of coagulating enzymes, and (5) a reduction in enzyme activity (Cairns et al. 1975). Based upon available literature, tissue anoxia and suboptimum enzyme activity appear to be the most plausible explanations.

Table II-8. Temperatures (°C) at which over 50% of the test species survived after 96 hours exposure (TLm^{96}), University of Utah, Salt Lake City, Utah, 1970 (from Gaufin 1973).

Species tested	% Survival	Temperature	
PLECOPTERA			
Acroneuria pacifica Banks	70%	27.0	
Isogenus aestivalis (Needham and Claassen)	50%	24.2	
Arcynopteryx parallela Frison	70%	23.0	
	45%	18.0	(Winter test)
EPHEMEROPTERA			
Ephemerella doddsi Needham	50%	22.0	
	60%	16.0	
TRICHOPTERA			
Brachycentrus occidentalis Banks	60%	29.0	
	70%	28.0	
Arctopsyche grandis (Banks)	40%	20.0	
DIPTERA			
Atherix variegata Walker	100%	29.0	
Holorusia grandis	80%	26.0	
	0%	28.0	

Table II-9. Correlation matrix of environmental factors and drift rates of Baetis sp. I and Simuliidae, Little Hole, Green River, 1965 (N=77) (from Pearson 1967).

	Illumination	Density	Temperature	Turbidity	Water Level	Date	Dissolved Oxygen	Depth	Baetis Drift	Simuliidae Drift
Illumination	1.00									
Density	.07	1.00								
Temperature	-.25**	.64**	1.00							
Turbidity	-.13	-.16	-.22*	1.00						
Water Level	-.10	.01	-.10	-.13	1.00					
Date	.05	.31**	.30**	-.27*	-.10	1.00				
Dissolved Oxygen	-.63**	.05	.48**	-.10	-.09	-.25*	1.00			
Depth	-.45**	-.44**	-.45**	.34**	-.15	-.50**	-.08	1.00		
Baetis Drift	.37**	.68**	.48**	-.06	.07	.03	-.12	-.13	1.00	
Simuliidae Drift	.20	.64**	.66**	-.23*	.01	.30**	-.06	-.39**		1.00

* Significant at the .05 level.
** Significant at the .01 level.

The combination of tissue anoxia and depressed dissolved oxygen concentrations at high temperatures could enhance the effects of any toxicant that either increased metabolic demand (e.g., copper) or blocked oxygen uptake at the gill level (e.g., zinc) (Cairns et al. 1975). A summary of the interaction of temperature and various toxicants is presented in Table II-10.

The preferred temperatures of native juvenile fish located in the Colorado Basin was assumed to be usually higher than for adults of the same species . Recent laboratory work by Bulkley et al. (1981) showed that this may not be true for Colorado squawfish (Ptychocheilus lucius). They found no significant difference in the preferred temperature of juvenile squawfish (24.6°C) and adults (25.4°C). The temperature range was 17-32°C and 14-27°C for juveniles and adults, respectively (Table II-11). These data differ slightly from field observations. Vanicek and Kramer (1969) found the greatest abundance of adults in water with mean summer temperatures of 11.1-12.1°C. Holden (1977) found juveniles in waters ranging from 12-28°C.

Egg temperature tolerances for various native Colorado fishes were determined by Bulkley et al. (1981). Colorado squawfish eggs were placed into tanks of 5, 10, 14, 20, and 26°C water. Normal hatch occurred only at 20°C. Field observations by Vanicek and Kramer (1969) and Holden (1973) noted ripe Colorado squawfish in waters of temperatures greater than 20°C. In a similar study, temperatures of 20-26°C apparently were optimum for the eggs of bonytail chub and humpback chub (Bulkley et al. 1981).

Influence of Suspended Sediments on Aquatic Biota

Primary Producers

Suspended sediments affect several physical properties of water. Turbidity is incresed, light penetration reduced, and the abrasiveness of the water increased. All of these factors will influence primary productivity.

Ellis (1936) showed selective absorbance of the blue and green wavelengths in highly turbid water, regardless of the color of the suspended material. Decreased light penetration would result in a corresponding decrease in primary production, assuming production is light-limited. However, Farnworth et al. (1979) noted that reduced light intensity eliminated surface inhibition of primary production and produced conditions favorable to many species of algae. Increased sediment loads have also been shown to be accompanied by an increase in nutrient levels. Work on the Arkansas River reported by Farnworth et al. (1979) described a decrease in

Table II-10. A summary of the effects of temperature upon the toxicity of chemicals to aquatic organisms (Cairns et al. 1975).

Substance	Effect
Ammonia	Direct relationship between temperature and the percent unionized ammonia. An increase in the temperature from 10° C to 20° C raises the percent by a factor of 1.3 to 1.6 depending on the pH.
Cyanide	Increases the effect of cyanide. The time to death is decreased, but the acute threshold concentration is relatively unaffected.
Trace Metals	
Zinc	Acute threshold concentration stayed the same, but the fish died almost five times faster when subjected to rapidly changing temperatures. Survival time was also decreased in constant temperatures.
Cadmium	Temperature had no effect on the survival of freshwater fish, some effect on marine species.
Chromium	Lowered resistance time in the presence of high temperatures.
Copper	Unknown, possibly some effect similar to zinc.
Mercury	Temperature did not have an effect, low dissolved oxygen might.
Pesticides	
DDT	The toxic effect of DDT was lowered by 50 percent over the range of 0.7° C to 13° C.
Malathion	Response varies with species. May depend on acclimation temperature.
Herbicides	Little data. Temperature may have some effect.

Table II-11. Temperature selection of Colorado River fishes (from Bulkley et al. 1981).

Species	Acclimation Temperature °C	n	Range of Individual Modes	Mean of Modes[a/]	Pooled Data Mode	Pooled Data Mean	Pooled Data SD	Skewness[b/]
Razorback sucker	8	4	22 - 27	25.0 ± 0.96	26	24.8	3.42	-0.4
	14	10	27 - 31	29.0 ± 0.52	28	28.3	2.47	0.1
	20	20	12 - 28	22.7 ± 1.07	24	22.5	2.67	-0.6
	26	20	13 - 29	23.1 ± 0.90	26	23.0	2.60	-1.2
Humpback chub	14	9	17 - 26	21.0 ± 1.26	18	20.3	3.88	0.6
	20	8	21 - 31	24.4 ± 1.20	23	23.1	3.78	0.03
	26	7	21 - 27	23.5 ± 1.72	17	21.3	4.37	1.0
Hybrid	20	8	18 - 28	23.8 ± 1.48	26	22.3	4.26	-0.8
Squawfish								
Juveniles	14	18	17 - 27	21.9 ± 0.74	21	22.1	2.97	0.4
Adults	14	4	14 - 27	21.5 ± 3.07	26	20.9	5.28	-1.0
Juveniles	20	20	24 - 32	27.6 ± 0.44	28	27.4	2.46	-0.3
Adults	20	2	20 - 25	22.5 ± 2.50	20	22.1	3.68	0.6
Juveniles	26	19	21 - 31	23.7 ± 0.64	23	24.1	3.46	0.3
Adults	26	6	24 - 27	25.7 ± 0.50	27	24.7	3.53	-0.7

[a/] ± standard error of mean

[b/] Pearson's skewness coefficient = (mean-mode)/standard deviation. A negative value indicates a longer tail on the low temperature side of the distribution curve.

blue-green and diatom populations with increased turbidity.

Abrasion by suspended solids may physically damage algae, and macrophytes, and therefore, reduce primary production. Substrate composition may also change during the natural process of erosion and deposition. Erosion removes substrate, carrying or detaching associated organisms; whereas deposition results in the burying of primary producers.

Macroinvertebrates

Aquatic insects are affected by increased sediment loads in a variety of ways. The most prominent is the reduction of available habitat through a change in substrate size. Lenat et al. (1981) studied the effects of road construction on stream benthos and found a decrease in available rock habitat which resulted in a decrease in macroinvertebrate density. There was, however, little change in overall community structure. Opposite results were found by Chutter (1969) working on a South African river. He reported a decrease in the number of taxa as sediment load increased with negligible change in the faunal density.

Changes in benthic density and species composition may be more related to drift than mortality. Farnworth et al. (1979) reported the findings of a study that found a direct relationship between the amount of sediment added to the stream (up to 160 mg/l) and the rate of invertebrate drift.

Fish

Depending on particle size, suspended sediments may cause fish mortality through the clogging of gill filaments (fine particles) or abrasion of gill tissue (larger particles). Suspended solid concentrations must be extremely high to cause death (100,000 mg/l), but behavioral changes do occur at much lower levels (Cordone and Kelley 1961). Green sunfish (Lepomis cyanellus) responded to a turbidity of 900 FTU (Formazin Turbidity Units) at 25°C. However, at 5°C the response was not apparent until 2100 FTU's. Response to these high turbidities included increased ventilation rates and decreased activity (Horkel and Pearson 1976). A study reported by Cordone and Kelley (1961) found that cutthroat trout ceased to feed and moved to cover when exposed to total suspended solids of 35 mg/l.

Reduction in reproductive success may be the most significant impact of increased sedimentation (Cordone and Kelley 1961). Siltation of suitable

gravel areas may prevent nest building. Sedimentation may also result in the direct mortality of eggs and fry from decreased water flow through the gravel and a corresponding drop in dissolved oxygen concentrations.

In summary, because of the high concentrations necessary to cause death, fish populations will more likely be affected through destruction of food supply, alteration of habitat, or reduction in reproductive success (Farnworth et al. 1979).

The Influence of Light Intensity on Aquatic Biota

Primary Producers

Primary production is directly related to light intensity. At low light levels, a linear relationship exists between light intensity and photosynthetic rate. The curve flattens at higher illuminations with a slight decline in the photosynthetic rate at extremely high intensities (Bannister 1974). There is evidence to suggest that photosynthetic rates in the morning are higher than those in the afternoon, although the intensities are the same (Harris and Lott 1973).

Light levels and chlorophyll concentrations have been used by various authors to estimate gross primary production (Lorenzen 1963; Duffer and Dorris 1966; Gallegos et al. 1980). Lorenzen (1963) correlated changes in light intensity to changes in gross and net photosynthesis, respiration, and chlorophyll concentrations for different dates throughout the summer.

Laboratory experiments by Gallegos et al. (1980) produced the series of curves shown in Figure II-4. The curves indicate a smooth relationship between irradiance and gross photosynthesis except when the rate of oxygen evolution was depressed by CO_2 depletion.

McConnell and Sigler (1959) estimated primary production by extracting chlorophyll from algal-coated rocks. Artificial substrates were also used and treated in a similar manner. Removal of the artificial substrates too early has resulted in an underestimate of primary production (Waters 1961a). Bannister (1974) employed the relationship between light intensity, chlorophyll concentrations, and primary production to formulate a series of predictive equations.

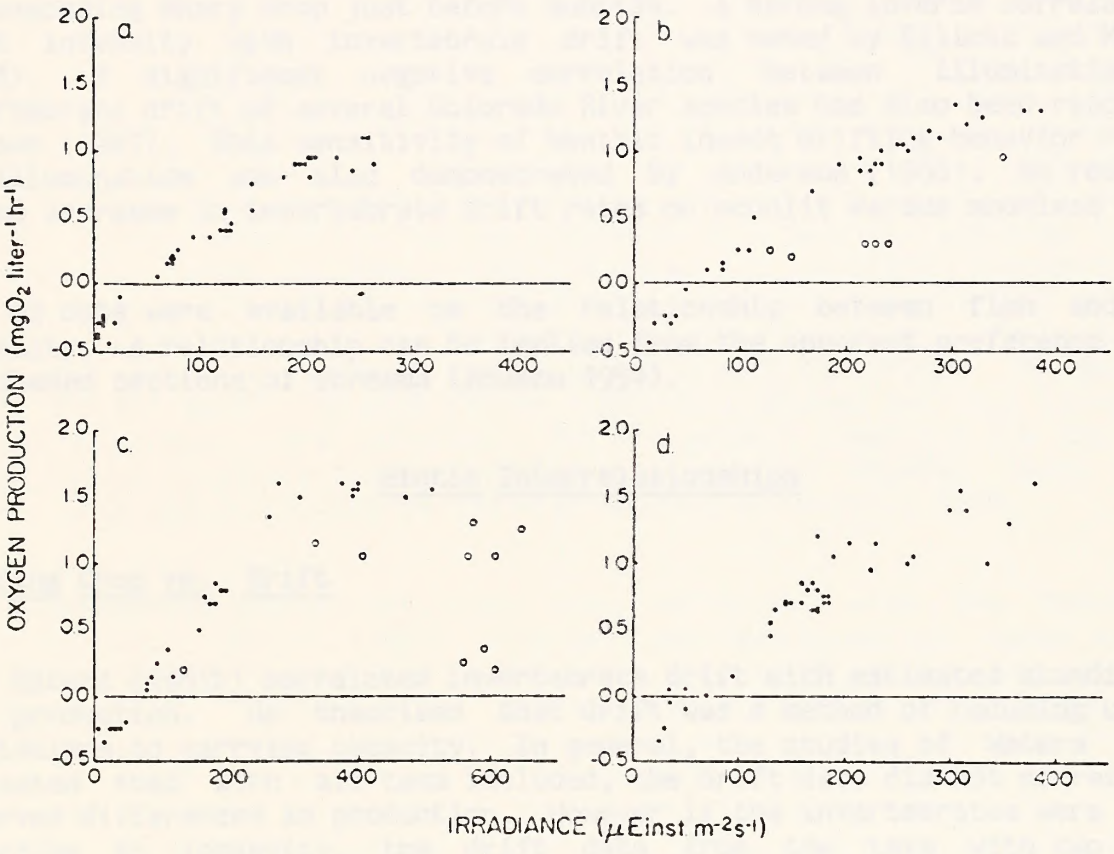


Figure 11-4. Net rate of oxygen production plotted against irradiance. O - Affected by afternoon depression (from Gallegos et al. 1980).

Macroinvertebrates and Fish

Light intensity affects aquatic insects primarily through the timing of invertebrate drift. Diel periodicity of invertebrate drift has been well-demonstrated. One of the first authors to describe the increased night-time drift was Waters (1962). He concluded that higher night-time drift rates were a result of increased activity by the organisms. An abrupt increase in drift rate occurred approximately one hour after sunset with a corresponding sharp drop just before sunrise. A strong inverse correlation of light intensity with invertebrate drift was noted by Elliott and Minshall (1968). A significant negative correlation between illumination and invertebrate drift of several Colorado River species has also been reported by Pearson (1967). This sensitivity of benthic insect drifting behavior relative to illumination was also demonstrated by Anderson (1966). He recorded a marked decrease in invertebrate drift rates on moonlit versus moonless nights.

No data were available on the relationship between fish and light intensity. A relationship can be implied from the apparent preference of fish for shaded sections of streams (Boussu 1954).

Biotic Interrelationships

Standing Crop vs. Drift

Waters (1961b) correlated invertebrate drift with estimated standing crop and production. He theorized that drift was a method of reducing upstream populations to carrying capacity. In general, the studies of Waters (1961b) indicated that with all taxa included, the drift data did not correspond to observed differences in production. However if the invertebrates were grouped according to longevity, the drift data from the taxa with two or more generations per year agreed remarkably well with production data. Another study by Waters (1966) compared the drift method of estimating annual production with the instantaneous growth method. The two methods gave similar results, but the drift method did have a greater variation for the daily estimates.

This theory of density-related emigration (drift) was verified by Peckarsky (1979). Experimentally increased benthic densities resulted in a negative linear decrease in immigration and a positive linear increase in emigration.

Invertebrate Numbers Versus Detritus

A large number of stream invertebrate taxa feed on plant detritus. Because of this relationship, various authors have correlated the amount of detritus with invertebrate numbers (Egglishaw 1964, Buscemi 1966, Barber and Kevern 1973). Egglishaw (1964) placed pans with differing amounts of detritus within similar sites in a stream riffle. Results indicated a significant positive correlation between detrital amounts and invertebrate numbers (Figure II-5). Egglishaw (1969) found that community structure to a large extent depended upon the amount of detritus present. Faunal diversity increased with an increase in detritus. Buscemi (1966) studied the effect of detrital amounts on invertebrate populations in the Palouse River, Idaho. The sites with the lowest amounts of detritus had the greatest species richness and abundance. This negative correlation was attributed to the high degree of organic pollution at the downstream station. A critical upper limit to total organic matter concentrations was suggested as a possible limiting factor to populations. Barber and Kevern (1973) altered the substrate composition and detrital amounts in a Michigan trout stream. Substrate was divided into small (less than 8 mm mean diameter) and large (greater than 8 mm). Three different levels of dried Potamogeton were added to each substrate: 121 gm/m^2, 484 gm/m^2, and 847 gm/m^2. No significant difference between detritus levels and macroinvertebrates were detected, but the general trend was for the low and medium levels to have a higher standing crop than the highest level (Figure II-6). This was attributed to the high amounts of detritus used.

Invertebrate numbers have also been correlated with the rapidity of decomposition. Osborn (1981) examined the decomposition rates of a high alkalinity stream (Temple Fork, Utah) and a low alkalinity creek (Gunbarrel Creek, Wyoming). Invertebrate and algae production were much higher in the high alkalinity stream. The average coefficients of decay (K) were .0173 for spring and .0057 for autumn- winter in the high alkalinity water and .0005 for spring and .0023 for autumn-winter in the low alkalinity stream (Table II-12). The rate of material loss was linear over time in both streams with the high alkalinity water having the steeper slope (Figure II-7). Benfield et al. (1977) reported K values similar to those found by Osborn (1981) but noted the absence of the shredder functional group. Initial leaf breakdown was accomplished wholly by microbial activity.

Reice (1977) conducted decomposition experiments using leaf packs of white ash at different current velocities, invertebrate abundances, species richness, and substrate sizes. Species richness and invertebrate abundance increased initially, reached a maximum during the second week, and then declined as decomposition progressed (Figure II-8). After an initial leaching period, loss through decomposition was linear. Conclusions from the experiment indicated that the only factor to show a significant relationship

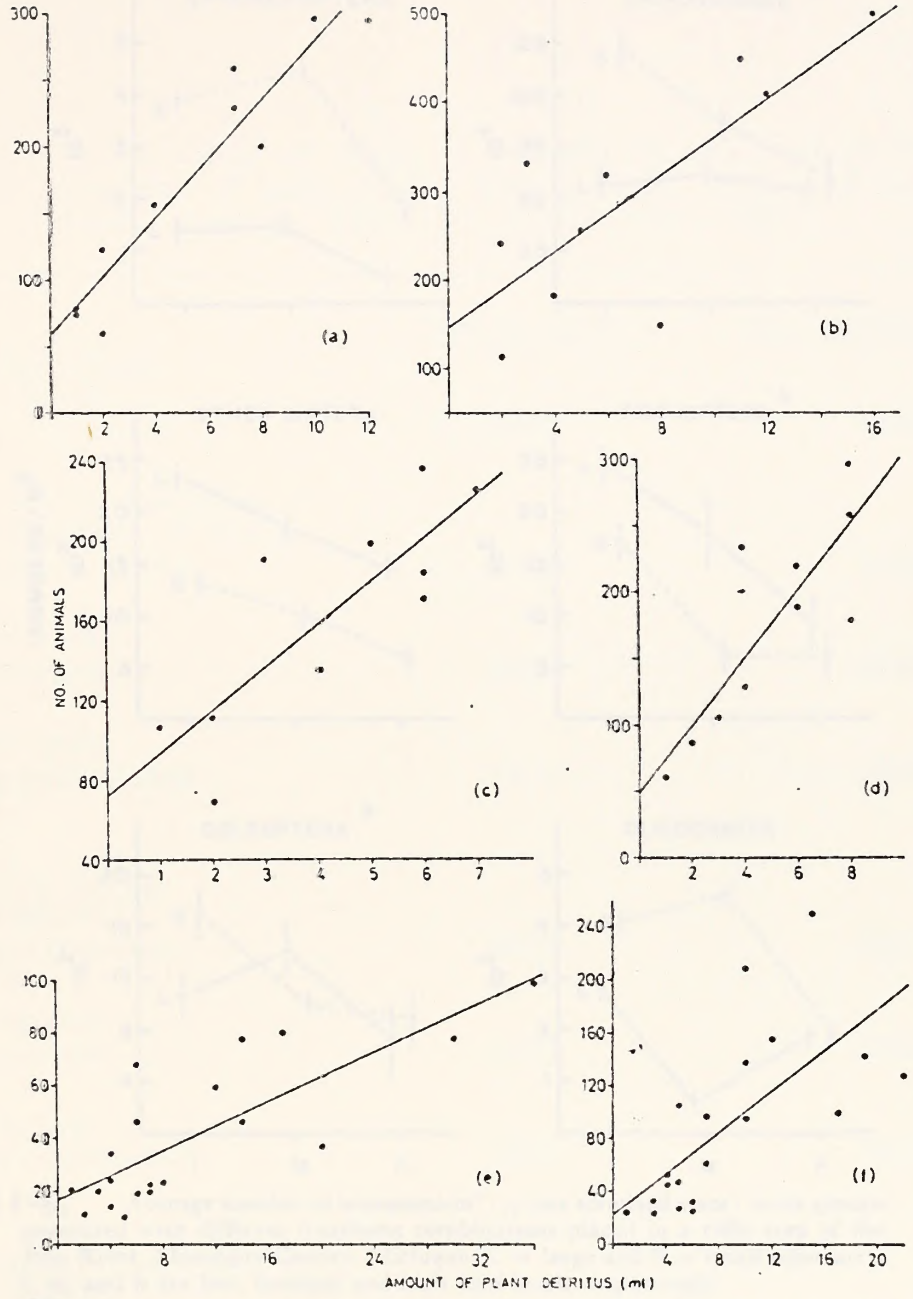


Figure 11-5. Graphs showing the relationship between the number of animals and the amount of plant detritus in samples from stream riffles with the calculated regression lines. (a)–(d) Sheiligan Burn: (a) April 1962, $y = 59 + 22{\cdot}1x$; (b) October 1962, $y = 146 + 21{\cdot}2x$; (c) March 1963, $y = 72 + 21{\cdot}6x$; (d) April 1963, $y = 49 + 25{\cdot}4x$. (e) and (f) Allt dos Mhuicarain: (e) August 1962, $y = 17 + 2{\cdot}3x$; (f) November 1962, $y = 23 + 7{\cdot}8x$ (from Egglishaw 1964).

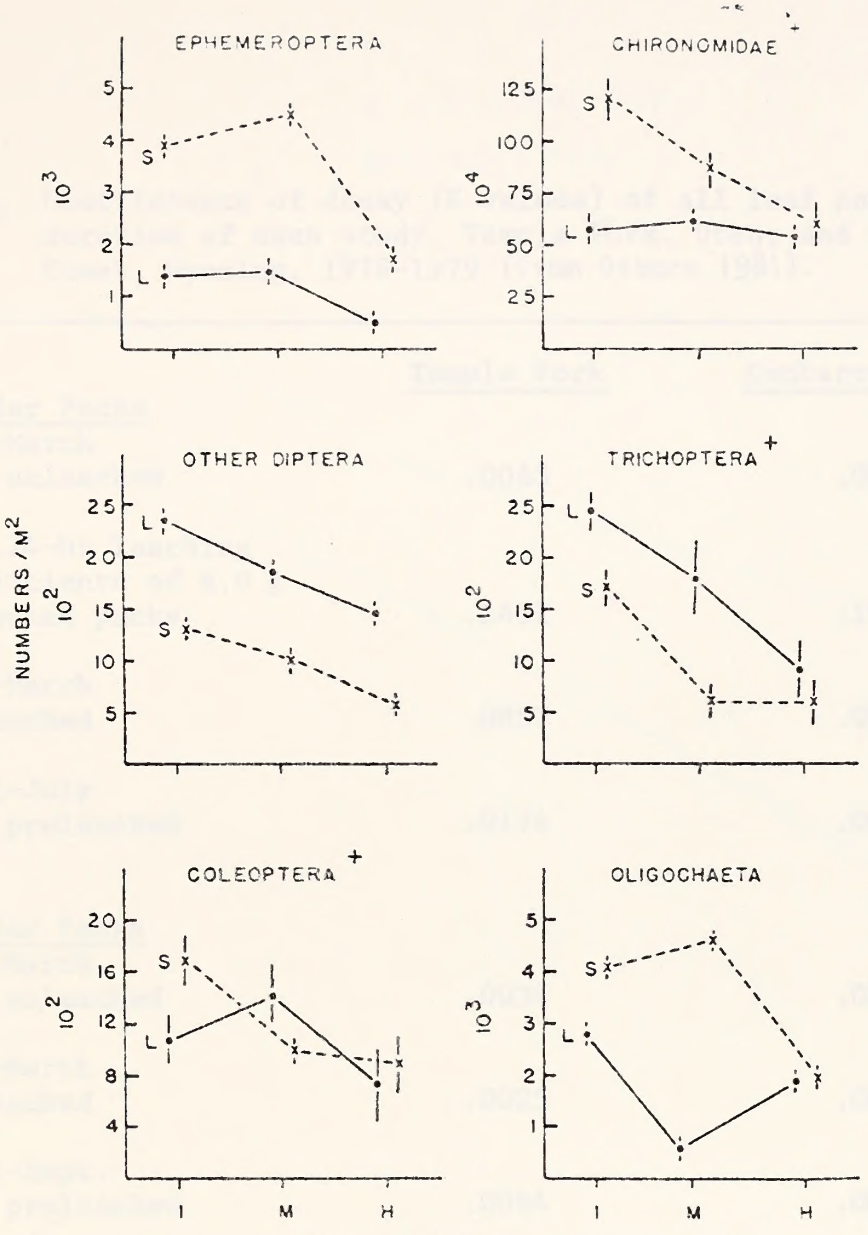
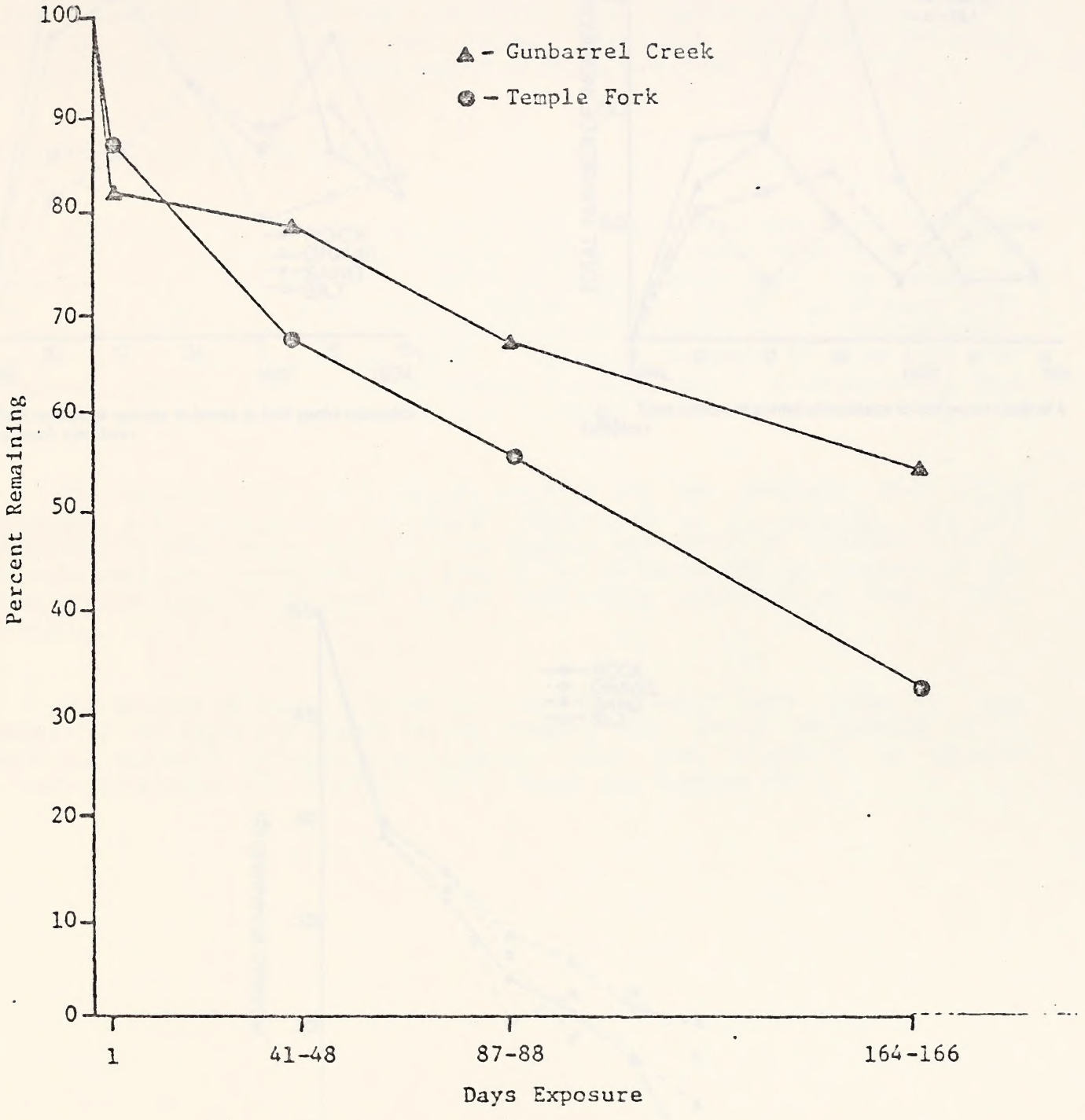


Figure 11-6. Average number of organisms/m² (± one standard error) in six groups associated with different treatment combinations placed in a riffle area of the Pine River, Montcalm County, Michigan. L = large and S = small substrates; l, m, and h are low, medium and high food levels, respectively.

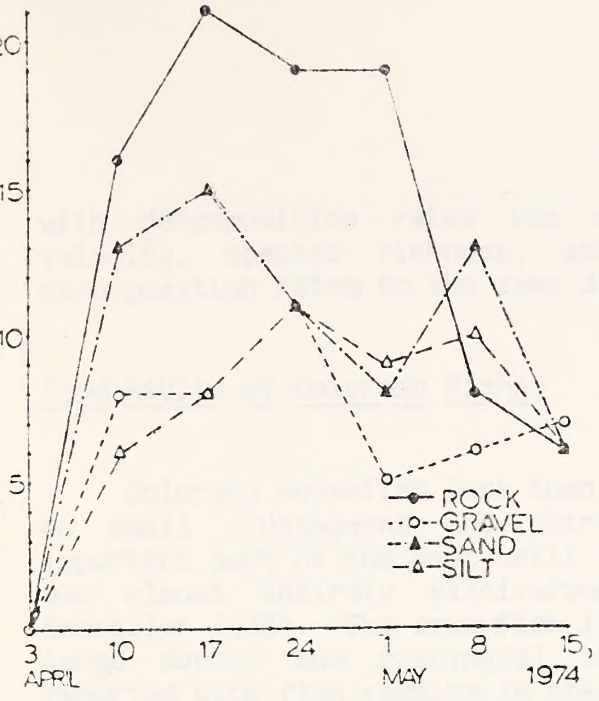
* Data for these groups demonstrated variance heterogeneity between treatment combinations, thus standard errors would have been much smaller if homogeneity existed between them (from Barber and Kevern 1973).

Table 11-12. Coefficients of decay (K values) of all leaf packs for duration of each study, Temple Fork, Utah, and Gunbarrel Creek, Wyoming, 1978-1979 (from Osborn 1981).

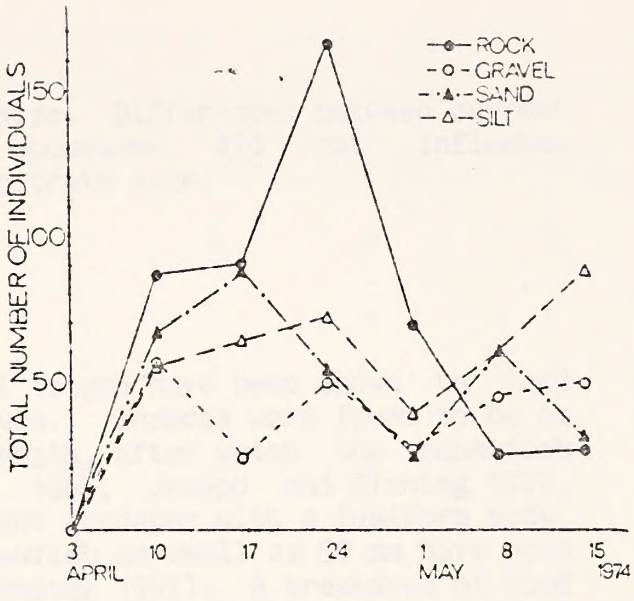
	Temple Fork	Gunbarrel Creek
Open Alder Packs		
Oct.-March 6.0 g unleached	.0065	.0035
Oct. 24-hr leaching coefficients of 6.0 g unleached packs	.1431	.1924
Oct.-March preleached	.0057	.0023
April-July 2.0 g preleached	.0174	.0005
Mesh Alder Packs		
Oct.-March 6.0 g unleached	.0034	.0033
Oct.-March preleached	.0025	.0021
April-Sept. 2.0 g preleached	.0064	.0078
Mesh Douglas-Fir Packs		
Oct.-March 6.0 g unleached	.0020	.0019



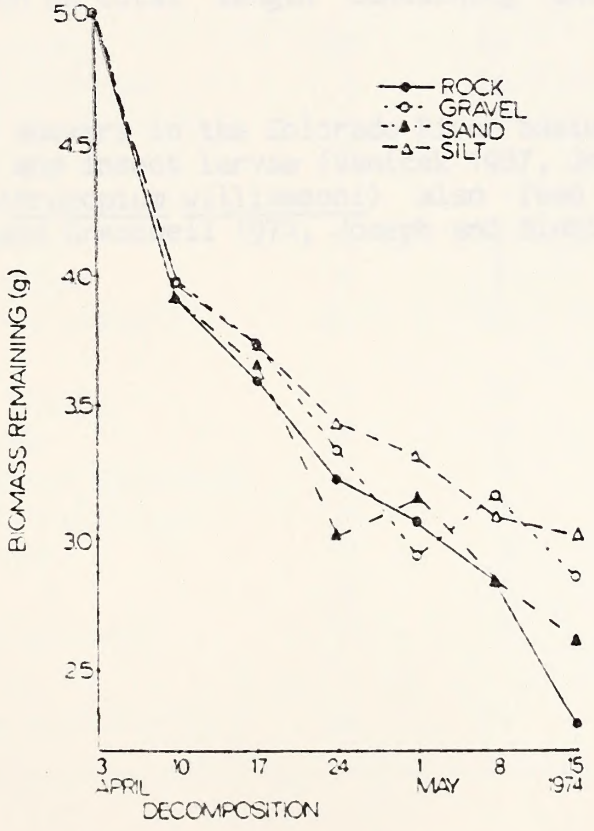
ure II-7. Breakdown of unleached 6.0 g open alder leaf packs in Temple Fork, Utah, and Gunbarrel Creek, Wyoming, October 1978 to March 1979 (from Osborn 1981).



A. Time course of species richness in leaf packs (samples pled for each site-date).



B. Time course of animal abundance in leaf packs (sum of 4 samples).



C. Decomposition data, Spring 1974. Biomass remaining (g) vs. sample date.

Figure 11-8. (From Reice 1977).

with decomposition rates was substrate size. Differences between current velocity, species richness, and animal abundance did not influence decomposition rates to the same degree as substrate size.

Food Habits of Colorado Fishes

Colorado squawfish less than 50 mm total length have been shown to feed on small crustaceans and chironomid larvae. Insects were found to be an important part of the diet until 200 mm in length, after which the squawfish was almost entirely piscivorous (Vanicek 1967, Joseph and Sinning 1977, Seethaler 1978). The squawfish is an efficient predator with a fusiform body, large mouth, and pharyngeal teeth. Squawfish as small as 50 mm have been reported with fish remains in the stomach (Vanicek 1967). A breakdown of food occurrence by total length is presented in Table II-13.

The roundtail chub (Gila robusta robusta) and the bonytail chub (Gila elegans) were originally considered subspecies of G. robusta. The only quantitative food habits work on these species was done by Vanicek (1967). Insects were the predominant food item in the stomachs with six percent of the fish greater than 100 mm total length containing unidentified fish (Table II-14).

All species of suckers in the Colorado River basin were shown to feed primarily on algae and insect larvae (Vanicek 1967, Joseph and Sinning 1977). Mountain whitefish (Prosopium williamsoni) also feed primarily on aquatic insects (Stalnaker and Gresswell 1974, Joseph and Sinning 1977).

Table 11-13. Percentage occurrence of food items in Colorado squawfish stomachs, Green River, 1964-1966 (from Vanicek 1967).

Item	Total length (mm)				
	15-25	26-50	51-100	101-200	201-598
Nematodes	0	0	5	0	0
Crustaceans					
Cladocera (Bosmina sp.)	27	14	0	0	0
Copepoda	45	14	0	0	0
Insects					
Ephemeroptera					
Nymph	5	0	8	9	0
Adult	0	0	3	9	0
Plecoptera (nymph)	0	0	5	0	0
Thysanoptera (Adult)	0	5	0	0	0
Hemiptera					
Corixidae (Adult)	0	0	11	18	0
Coleoptera					
Larvae	9	5	2	0	0
Adult	0	0	2	5	0
Trichoptera (Larvae)	0	5	3	5	0
Diptera					
Chironomidae	60	48	29	5	0
Ceratopogonidae	5	14	0	0	0
Unidentified larvae	0	10	3	0	0
Unidentified Adult	0	0	2	0	0
Hymenoptera (Adult)					
Formicidae	0	0	3	5	0
Unidentified	0	0	2	0	0
Unidentified insects	23	29	23	18	0
Fish					
Cyprinus carpio	0	0	0	0	3
Gila robusta	0	0	0	0	4
Richardsonius balteatus	0	0	2	0	6
Pantosteus delphinus	0	0	0	0	1
Unidentified	0	5	19	36	49
Empty	5	5	13	27	39
Parasitized (Cestoda)	0	0	11	27	65
Total number of stomachs	22	21	62	22	71

Table 11-14. Percentage occurrence of food items in Colorado chub stomachs, Green River, 1964-1966 (from Vanicek 1967).

Item	Colorado chub				Roundtail	Bonytail
	Total length of fish (mm)					
	15-25	26-50	51-100	101-200	201-370	201-390
Plant debris	0	2	3	4	27	37
Filamentous algae	5	0	3	4	17	26
Nematodes	5	4	1	1	0	3
Oligochaetes	0	2	0	0	0	0
Arachnids						
Araneae	0	0	0	0	0	3
Hydracarina	0	0	3	2	0	0
Copepods	5	7	0	0	0	0
Insects						
Orthoptera (Adult)						
Locustidae	0	2	0	2	10	24
Unidentified	0	0	0	0	8	14
Ephemeroptera						
Adult	0	2	3	4	8	3
Nymph	35	14	4	8	0	3
Plecoptera						
Adult	0	0	1	0	8	3
Nymph	0	0	1	0	0	0
Odonata (Adult)	0	0	0	2	0	0
Thysanoptera (Adult)	0	2	0	2	0	0
Hemiptera (Adult)						
Corixidae	0	0	7	4	8	6
Unidentified	5	5	1	0	0	3

INTERRELATIONSHIPS

Based upon the literature available, the following is a list of possible interrelationships (abiotic-abiotic, abiotic-biotic, biotic-biotic) which might exist in the White River Ecosystem.

I. Substrate Composition

A. Current velocity/Substrate composition (Leopold et al. 1964, Hynes 1970, Novak 1973).

II. Primary Producers

A. Current Velocity

1. Current velocity/Nutrient uptake (Blum 1956, Odum 1956, Whitford 1960, Lock 1979).

2. Current velocity/Species composition (McIntire 1966).

3. Flow patterns/Periphyton distribution (Blum 1956, Tett et al. 1978).

B. Nutrients

1. Nutrient concentrations/Algal growth (Blum 1956, Whitford 1960).

2. Nutrient concentrations/*Cladophora* (Herbst 1969, Whitton 1970a).

3. Divalent cation concentrations/Algae (Blum 1956, Whitton 1970a).

C. pH

1. pH (7.0 - 10.0)/Algae (Blum 1956, Whitton 1970a).

D. Heavy Metals

1. Zinc toxicity (0.1 mg/l)/*Cladophora* (Whitton 1967, 1970b).

2. Copper toxicity (0.2 mg/l)/*Cladophora* (Whitton 1967, 1970b).

3. Lead toxicity (3 mg/l)/*Cladophora* (Whitton 1970b).

E. EDTA

1. EDTA + Cu toxicity (0.6 mg/l)/*Cladophora* (Whitton 1967).

2. EDTA + $CuSO_4$ toxicity (10 mg/l)/*Cladophora* (Betzer and Kott 1969).

3. EDTA + Zn toxicity (0.5 mg/l)/Cladophora (Whitton 1967).

F. Temperature

1. Increased temperature/Standing crop (Foerster et al. 1974).

2. Increased temperature/Species composition (Wright et al. 1970, Squires et al. 1979, Wilde and Tilly 1981).

3. Temperature range (15 - 30° C)/Cladophora (Bellis 1968).

G. Suspended Sediments

1. Suspended sediments/Light penetration (Ellis 1936, Farnworth et al. 1979).

2. Suspended sediments/Species composition (Farnworth et al. 1979).

3. Suspended sediments/Periphyton abrasion (Farnworth et al. 1979).

III. Primary Production

A. Light Intensity

1. Light intensity/Primary production (Lorenzen 1963, Bannister 1974, Gallegos et al. 1980).

2. Light intensity/Chlorophyll concentrations (Bannister 1974, Gallegos et al. 1980).

B. Chlorophyll Concentration

1. Chlorophyll concentration/Primary production estimate (McConnell and Sigler 1959, Waters 1961a).

IV. Macroinvertebrates

A. Current Velocity

1. Current velocity/Caddis larvae (Edington 1968).

B. Substrate

1. Substrate/Invertebrate preference (Cummins 1966, Thorup 1966, Cummins and Lauff 1969, Barber and Kevern 1973, Williams 1978).

2. Substrate size/Green River insects (Pearson 1967).

3. Substrate size/Decomposition rate (Reice 1977).

B. Cation Concentrations

1. Potassium/*Gammarus pulex* distribution (Minshall and Minshall 1978).

2. Calcium/Invertebrate distribution (Egglishaw 1968).

C. Total Alkalinity

1. Total alkalinity/Invertebrate production (Osborn 1981).

D. pH

1. pH tolerance level/Ephemeroptera, Plecoptera, Trichoptera (Gaufin 1973).

E. Dissolved Oxygen

1. Dissolved oxygen/Invertebrate distribution (Gaufin 1973).

2. Dissolved oxygen requirements/Population location (Gaufin 1973).

3. Dissolved oxygen requirements/Current velocity (Eriksen 1966, Gaufin 1973).

4. Low dissolved oxygen concentrations/Metabolic rate (Eriksen 1966, Davis 1975).

F. Temperature

1. Lethal temperature limits/Ephemeroptera, Plecoptera, Trichoptera, Diptera (Gaufin 1973)

2. Increased temperature/Insect emergence (Gaufin 1973, Langford and Daffern 1975).

3. Temperature/Invertebrate drift (Waters 1962, Pearson 1967).

G. Light Intensity

1. Light intensity/Drift rate (Waters 1962, Pearson 1967, Elliott and Minshall 1968).

2. Moonlight/Drift rate (Anderson 1966).

H. Standing Crop

1. Standing crop/Drift rate (Waters 1961b, Peckarsky 1979).

2. Annual production/Drift rate (Waters 1966).

3. Standing crop/Decomposition rate (.0173 high alkalinity stream, .0005 low alkalinity stream) (Osborn 1981).

I.

1. Loss of functional group/Decomposition rate (.0057) (Benfield et al. 1977).

J. Detritus

1. Amount of detritus/Standing crop (Egglishaw 1964, Buscemi 1966, Barber and Kevern 1973).

2. Amount of detritus/Species diversity (Egglishaw 1969).

V. Fish

A. Current Velocity

1. Current velocity/Brown trout (Gosse and Helm 1981).

2. Current velocity/Colorado squawfish (Seethaler 1978).

B. Substrate Size

1. Substrate size/Brown trout (Gosse and Helm 1981).

2. Substrate size/Colorado squawfish (Seethaler 1978).

C. Water Chemistry

1. Dissolved oxygen

a. Dissolved oxygen (5 mg/l)/Trout (Davis 1975).

2. Pollutants

a. Cyanide/Dissolved oxygen requirements (9.74 mg/l) (Downing 1954).

b. Heavy metals/Dissolved oxygen requirements (5.78 mg/l) (Lloyd 1961).

3. pH

a. pH levels (4.1)/Brook trout (Creaser 1930).

b. pH levels (4.0)/Various species (Butler et al. 1973).

D. Temperature

1. Laboratory temperature range (17-32° C)/Juvenile Colorado squawfish (Bulkley et al. 1981).

2. Field observations (12-28° C)/Juvenile squawfish (Holden 1977).

3. Laboratory temperature range (14-27° C)/Adult squawfish (Bulkley et al. 1981).

4. Field observations (11.1-12.1° C)/Adult squawfish (Vanicek and Kramer 1969).

5. Temperature (20° C)/Squawfish hatching (Bulkley et al. 1981).

6. Temperature (20-26° C)/Humpback and bonytail chubs hatching (Bulkley et al. 1981).

7. Temperature/Heavy metal toxicity (Cairns et al. 1975).

E. Suspended Sediments

1. Suspended sediment concentrations/Fish mortality (100,000 mg/l) (Cordone and Kelley 1961).

2. Suspended sediment concentrations/Green sunfish response (Horkel and Pearson 1976).

3. Suspended sediment concentrations/Cutthroat trout response (Cordone and Kelley 1961).

F. Stream Cover

1. Shade/Fish numbers (Boussu 1954).

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