1984

Factors Affecting Water Quality and Microinvertebrate Distribution Within a Small Black Hills Stream

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FACTORS AFFECTING WATER QUALITY AND MACROINVERTEBRATE DISTRIBUTION WITHIN A SMALL BLACK HILLS STREAM

BY

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A thesis submitted
in partial fulfillment of the requirements
for the degree, Master of Science,
Major in Wildlife and Fisheries Sciences
Fisheries Option
South Dakota State University
1984

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WITHIN A SMALL BLACK HILLS STREAM

This thesis is approved as a creditable and independent investigation by a candidate for the degree, Master of Science, and is acceptable for meeting the thesis requirements for this degree. Acceptance of this thesis does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

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FACTORS AFFECTING WATER QUALITY AND MACROINVERTEBRATE DISTRIBUTION WITHIN A SMALL BLACK HILLS STREAM

Abstract

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A comparative evaluation of the benthic macroinvertebrate fauna was conducted concurrently with a physiochemical investigation on Slate Creek, Pennington County, South Dakota in the Black Hills. Water quality differences between years and among stations were detected in Slate Creek from both physiochemical and macroinvertebrate evaluations. The primary sources of disturbance to the Slate Creek study site during the sampling period were landscaping activities within the Deerfield Park Resort development and livestock activity. Increased runoff and elevated stream flows in 1982 were responsible for the variation in water quality between years. Water quality differences among stations indicated significantly ($P < 0.05$) higher turbidity and temperature immediately below the development site in 1981 and 1982. Significantly ($P < 0.05$) higher fecal coliform bacteria counts were observed at station 5 for both years, resulting from increased livestock activity. Phosphates and nitrates were highest at stations 4 and 5 but were not significantly ($P > 0.05$) different from the other stations in either year. Conductivity, hardness, alkalinity, and pH were significantly ($P < 0.05$) lower at station 2 for both years due to natural variation within the watershed. Physiochemical differences between years and among stations resulted in subsequent changes in the macroinvertebrate

fauna. Differences from above and below the development site were observed among both macroinvertebrate species and assemblages. Species with a higher tolerance to sedimentation were more abundant in downstream stations. Among the community indices utilized, the biotic index provided the greatest discrimination among stations and tended to group stations above and below the development site.

ACKNOWLEDGEMENTS

I would like to express my appreciation to the following people for making this study possible: Dr. Timothy Modde for his guidance and encouragement throughout the study; Mark Rumble for his help in designing the study and arranging project logistics; Dr. W. Lee Tucker for his statistical advice; Phil Haskett for his help in computer programming; Pat Hudson for verifying and identifying invertebrate specimens; and fellow graduate students for their help and criticism throughout the study.

I would especially like to thank Michele Deisch and Brian Smith for their companionship and invaluable assistance in the field and laboratory work. Finally, I would like to extend sincere thanks to my parents, Mr. and Mrs. Albert Drewes, who have given their love and support throughout my education.

Financial assistance was provided by the United States Forest Service, Rocky Mountain Forest and Range Experiment Station.

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INTRODUCTION

Early records suggest that 1,937 kilometers of Black Hills streams were potentially suitable for maintaining trout populations, however, by 1964 only 597 kilometers of stream were considered suitable for trout management (Stewart and Thilenius 1964). Decreasing stream flows, increasing sediment loads, and pollutants have been cited as the primary causes for the loss of trout stream habitat in the Black Hills (Anonymous 1967). Thicker timber stands resulting in increased evapotranspiration, wells draining aquifers on the periphery of the Black Hills, and increased agricultural water usage are factors which may contribute to loss of flow. Loss of trout stream habitat has also been attributed to mining activity, road construction, the addition of domestic wastes, and trampling of stream banks by livestock (Stewart and Thilenius 1964).

The number of private homes and recreational developments are increasing rapidly on the borders of national forests (Segall 1976). Mountain home developments vary considerably in design, but many are located near streams or lakes. Gary et al. (1981) reported that rural home development can degrade water quality in mountainous areas. Water quality problems result directly from homesite development (cutting, filling, leveling, etc.), road systems, and sewage disposal (EPA 1973; Segall 1976). The inventory of water quantity and quality is necessary, together with other elements of the environment that are known to significantly influence the production and utilization of water in the Black Hills (Orr 1975).

Despite increasing demands on the natural resources of the Black Hills, the evaluation of cultural impacts upon stream habitats has been negligible. Those investigations of stream habitats which have been conducted in the Black Hills have primarily involved physiochemical (Anderson 1980) and fisheries aspects (Stewart and Thilenius 1964). While physiochemical techniques accurately and quickly evaluate the water quality, the measurements are only indicative of perturbations at the time samples are collected (Tracy 1979). Intermittent pollution, though not readily discernible by chemical and physical tests, does effect the aquatic biota (Goodnight 1973). While more is probably known about the biology of fishes than that of any other group of aquatic organisms, their ability to avoid areas of contamination limit their usefulness in stream water quality investigations (Goodnight 1973).

Aquatic invertebrates are sensitive to subtle changes in water quality and consequently have been extensively examined as indicators of pollution (Larimore 1974). Gaufin and Tarzwell (1952) concluded that the quantitative and qualitative composition of an aquatic macroinvertebrate population constitutes a valuable index in delineating zones of pollution in a stream. Benthic macroinvertebrates are particularly suitable as ecological indicators because their habitat preference and relatively low mobility cause them to be directly affected by substances that enter the environment (Cummins 1979, Hynes 1970). Water quality measured by physiochemical methods in conjunction with the evaluation of the macroinvertebrate community can be used to indicate the degree of perturbation in lotic systems.

The objective of this study was to evaluate the water quality of a small Black Hills headwater stream as affected by agricultural practices and upstream construction activities associated with the Deerfield Park Recreation Resort. In this study a comparative evaluation of the benthic macroinvertebrate fauna was conducted concurrently with a physiochemical investigation on Slate Creek, Pennington County, South Dakota, in the Black Hills.

STUDY AREA

Slate Creek is located in the upper reaches of the Rapid Creek drainage. The section of stream investigated flows through precambriam metamorphic (schists and slate) bedrock in Pennington County, South Dakota. Under the stream order classification system devised by Horton (1945) and later modified by Strahler (1957), the Slate Creek Study area exists as two second order tributaries, South Slate Creek and Slate Creek proper which join and flow as a third order stream through the remainder of the study area. This small headwater stream flows through open meadows surrounded by timberland within the Black Hills National Forest. Within the study area, meadows adjacent to the stream have been cultivated and are also used as pastureland for cattle grazing.

Slate Creek has been categorized by the U.S. Fish and Wildlife Service and the South Dakota Department of Game, Fish and Parks (Anonymous 1978) as a high priority fishery resource. South Dakota Public Law 74:03:02 currently classifies Slate Creek as a cold water permanent fish life propagation water and as a limited contact recreational water.

METHODS

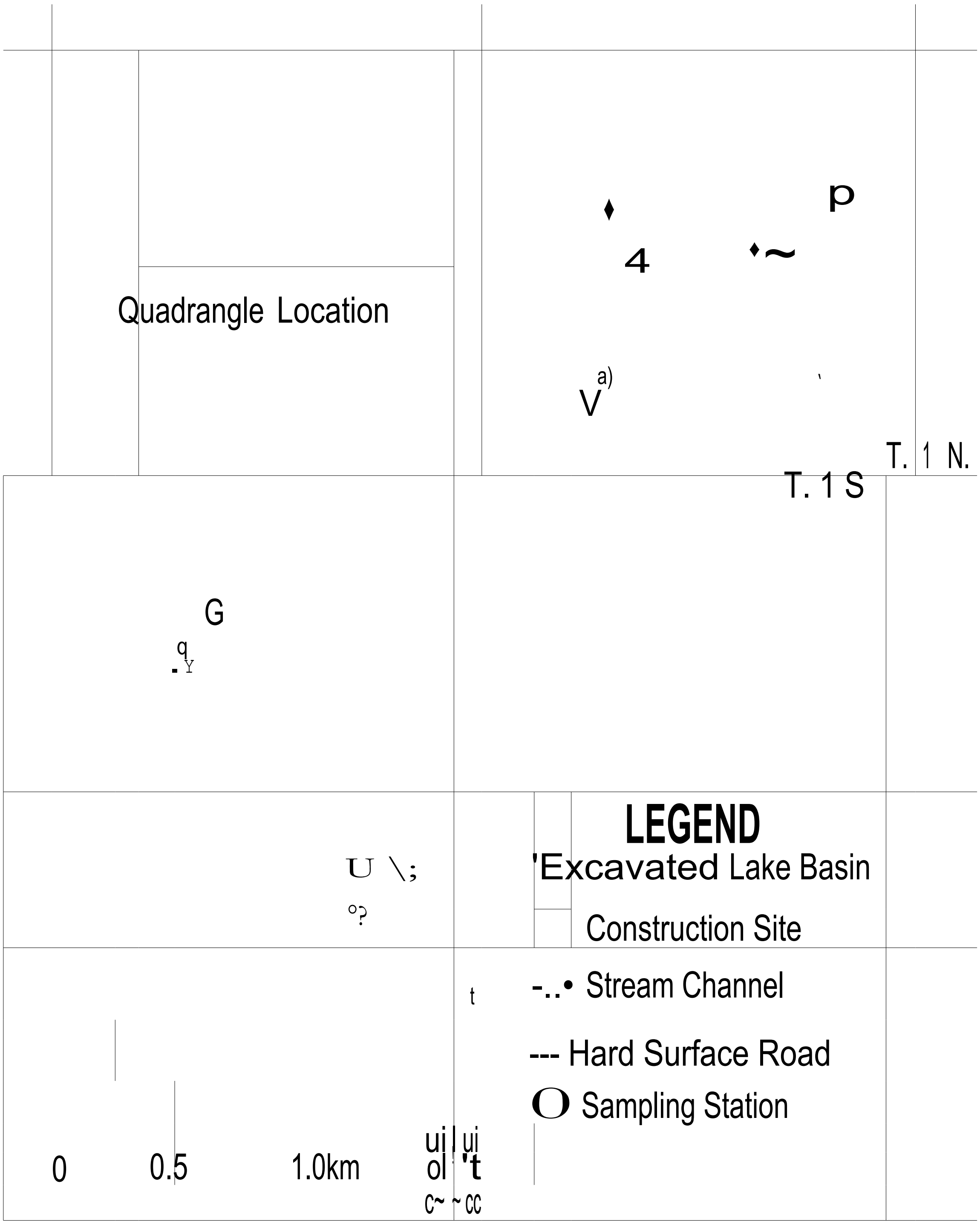
Station Selection

Five permanent sampling stations were selected for collecting water quality measurements and benthic macroinvertebrates (Figure 1). Two stations were established above Deerfield Park, one on South Slate Creek and the other on Slate Creek proper. A third sampling station was located 0.1 km downstream from the location of the resort dam. Two additional stations were established at approximately 1.0 km intervals from the resort dam. Stations were assigned numbers 1 - 5 and henceforth will be referred to as such. Stations were selected on the basis of homogenity of the aquatic habitat and accessibility.

Water Quality

Water quality data were collected at three-week intervals from all stations between 20 May 1981 and 30 August 1981, and again from 26 April 1982 until 4 September 1982. Water quality data were collected at six to eight week intervals through the fall, winter, and spring months between August 1981 and April 1982. A total of 12 water quality variables were measured during this study. Variables measured included temperature, total hardness, alkalinity, conductivity, turbidity, pH, phosphorous (orthophosphate and organic phosphate), nitrogen (ammonia and organic nitrogen), dissolved oxygen, and fecal coliform bacteria. On-site analysis of total hardness, alkalinity, dissolved oxygen, pH, and conductivity were determined with field analysis units (i.e. Hach kit Model DR-EL/2 and Yellow Springs Instrument S-C-T Meter Model 33). Turbidity was determined by subjecting field samples to analysis by a

Figure 1. Location of the five sampling stations in relation to the Deerfield Park Development site in Pennington County, South Dakota, 1981 and 1982.



Hach turbidimeter located in the U.S. Forest Service Experiment Station laboratory. Analytical analyses of phosphorous, nitrogen, and fecal coliform bacteria were made through contract services with Travis Laboratories in Rapid City, South Dakota.

Site Survey

Stream morphometry measurements were evaluated on 23 July 1982. A 61.0 m section of stream was designated at each station. Eleven transects at 0.61 m increments along each section were established. Stream width was recorded and water depth measured at 0.15 m intervals across each transect and at both banks. Data from the 11 transects were pooled and mean depth, width, and depth at stream-bank interface were calculated for each station. Bottom substrate at each station was also examined. A shovel was used to remove three samples of the substrate at each station. Samples were placed in containers and returned to the U.S. Forest Service laboratory. Samples were then dried and sifted through a series of U.S. Standard Testing sieves. Mean percent by weight of each particle **size** was calculated for each station.

As an index of livestock impacts, canopy coverage was examined at each station on 4 June 1982. Adjacent to each station, 50 m transects were established parallel to the stream at 1, 5, and 15 m distances from the stream. Daubenmire plots (0.1 m^2) were interpreted every 1.0 m along each transect line. Bare ground, dung (primarily from livestock), forb, grass, litter, and shrub cover in each plot were assigned a numerical value that ranged from 1 - 6 corresponding to percent ground cover by type. Data from three transects were pooled and mean values for each cover type were calculated at each station.

Macroinvertebrates

Benthic macroinvertebrates were collected from each station at three-week intervals between 1 July 1981 and 30 August 1981 and between 25 May and 4 September 1982. A single invertebrate collection was taken six weeks after the 30 August 1981 sampling date on 10 October 1981. Two aquatic invertebrate sampling methodologies were employed. An Eckman dredge was used to collect three benthic samples from each station on the designated collection dates. Invertebrates were also sampled using artificial substrate samplers (Hester-Dendy type). Three artificial substrate samplers were removed from the stream on the same dates benthic samples were collected. Prior to their removal, artificial substrate samplers had been subjected to colonization for a six week period. A total of 155 benthic and 143 artificial substrate (12 samplers not recovered) samples were collected. Contents from each dredge haul were placed in 1 liter mason jars and labeled. Artificial substrate samplers were dismantled and placed in labeled plastic sample bags. All samples were preserved in the field with 5% formalin solution and stained with rose bengal. Samples were returned to the laboratory and later sorted and picked by hand using a low power scanning lens. Organisms were counted and identified to genera using the most appropriate keys available.

To facilitate the identification procedure in samples with large numbers of chironomids, a subsampling system was devised. A grid consisting of 80, 1 cm^2 squares was placed on the bottom of a glass petri dish. The organisms were counted and placed in the dish with ample water to allow movement upon agitation. The sample was then mixed and allowed to come to rest. A random numbers table was used to select squares from which to remove a subsample of the chironomids. Organisms continued to be removed in this fashion until the subsample comprised 20% of the original sample. The organisms in the subsample were then enumerated and identified. The numbers of each chironomid genera in the subsample were then extrapolated to approximate the total number of organisms in the original sample.

Analysis

Water quality and invertebrate data were evaluated with analysis of variance and discriminant analysis. A two-way model was constructed, utilizing stations (five) and years (two), as treatments to test for differences within each water quality variable and invertebrate genera. Invertebrate data were analyzed separately by type (benthic and artificial substrate). Missing data values were calculated for lost artificial substrate samples based on mean numbers of invertebrates from the remaining samples.

Invertebrate data from the 25 May 1982 collection were not included in the analysis of variance in order to maintain the condition of balanced design between years. The Waller-Duncan k-ratio t-test was used to group station means for those water quality variables and genera exhibiting significant differences. Stepwise discriminant analysis was performed to determine: if stations could be separated statistically, which water quality variables and genera were important in separating stations, and derivation of coefficients for classifying stations.

Cluster analysis, based on the presence and absence of invertebrates in each sample collection, was used to compare the similarity of species assemblages in both temporal and spacial dimensions. Species similarity was analyzed by the unpaired group arithmetic average clustering (UPGMA) method (Sneath and Sokal 1973). The Czekanowski coefficient was employed in determining pair similarity (Bray and Curtis 1957). Only those genera which occurred in at least 5% of the total number of samples were included as variables in the cluster analysis.

Assemblages of macroinvertebrates were also evaluated using four biological indices. The Shannon-Weaver index of diversity, species richness, species evenness, and the biotic index were calculated for each sample. The Shannon-Weaver index of diversity (Wilhm and Dorris 1968) was calculated as follows:

$$\overline{H} = \sum_{i=1}^{s} (ni/N) \log_2 (ni/N);$$

when N is the total number of individuals in the sample, ni is the number of individuals in the ith species (taxon) and s is the number of species. Species richness proposed by Margalef (Wilhm and Dorris 1968) was calculated as follows:

$$d_1 = s - 1 / \text{In } N$$

where s is the number of species and N is the total number of individuals in the sample. Species evenness (Pielou 1966) was calculated as follows:

$$e = H / \log S$$

where H is the value of the Shannon-Weaver index of diversity and S is the number of species. The biotic index proposed by Chutter (1972) and modified by Hilsenhoff (1977) was the final biological index evaluated. The biotic index was calculated as follows:

$$BI = \sum_{i=1}^{s} N_i \quad a_i / N$$

where N is the total number of individuals in the sample, n_i is the number of individuals in the ith species (taxon), and a_i is the pollution tolerance value for the ith species.

Pollution tolerance values (a_i) used in the biotic index were adopted from Hilsenhoff (1982) or were assigned a_i values corresponding to the most pollution-tolerant species of that genus (Appendix Table 1). Community indices were evaluated with analysis of variance. A two-way model was constructed, utilizing stations (five) and years (two) as treatments, to test for differences within each biological index.

RESULTS

Site Survey

Stream morphometry data indicated increases in mean depth, width, and stream-shore depth with decreasing stream gradient (Table 1). Morphometry measurements from station 2 were somewhat misleading because that section of stream has been altered by beaver activity.

Analysis of sediment samples indicated that the bottom substrates at stations 1 and 2 contained much less sand and silt than did the substrates from stations 3, 4, and 5 (Table 2). Sand and silt combined comprised only 4.6 and 1.0% of the samples by weight at stations 1 and 2, compared to 37.6, 15.4, and 27.7% at stations 3, 4, and 5, respectively. Rubble and gravel combined comprised 95.4 and 99.0% of the sediments from stations 1 and 2, compared to 62.4, 84.6, and 72.3% at stations 3, 4, and 5, respectively.

Canopy coverage data indicated that percent ground cover by grass and dung were highest at stations 3, 4, and 5 (Table 3). Percent ground cover by dung at stations 1 and 2 were 0.3 and 0.5% compared to 2.4, 5.2, and 3.3% at stations 3, 4, and 5, respectively. Over 50% of the ground cover at stations 3 and 4, and 39.5% of each plot at station 5 were covered by grass. Station 2 was located in a more wooded area and had the highest percentage of ground cover by litter (56.1%) and shrub (6.5%) of the five stations.

Water Quality

Twelve physiochemical variables were evaluated on 16 dates from each of the five sampling stations (Appendix Table 2). Variation in

Table 1. Mean stream morphometry measurements for the five sampling stations collected on 23 July 1982 from Slate Creek, South Dakota.

	Station				
Parameter	1	2	3	4	5
Depth (cm)	12.1	29.1[a]	15.7	24.7	25.4
Width (cm)	93.3	260.2[a]	94.0	67.6	106.7
Stream-shore depth (cm)	5.5	11.4[a]	5.6	19.0	23.1

[a]Morphometry data from station 2 was collected from a section of stream altered by beaver activity.

Table 2. Mean percent composition by particle size of sediment samples collected from the five sampling stations on 23 July 1982 from Slate Creek, South Dakota.

	Station				
Particle **size**	1	2	3	4	5
Rubble (> 76.2 mm)	15.4	41.2	3.0	32.8	30.9
Medium and coarse gravel (4.7 - 76.2 mm)	71.9	56.4	45.2	44.9	36.0
Fine gravel (2.0 - 4.7 mm)	8.1	1.4	13.9	6.9	5.4
Course sand (1.0 - 2.0 mm)	2.0	0.3	8.3	3.8	5.0
Medium sand (0.5 - 1.0 mm)	1.4	0.3	10.0	4.5	8.8
Fine sand and silt (< 0.5 mm)	1.2	0.4	19.3	7.1	13.9

Table 3. Mean percent canopy coverage by cover type from transect lines established adjacent to the five sampling stations on 4 June 1982 from Slate Creek, South Dakota.

	Station				
Cover type	1	2	3	4	5
Bare ground	29.4	8.4	20.6	7.7	13.8
Forb	21.7	21.4	4.9	26.1	29.9
Grass	22.2	30.2	54.5	51.6	39.5
Dung	0.3	0.5	2.4	5.2	3.3
Litter	22.2	56.1	23.8	16.1	12.5
Shrub	1.3	6.5	0.0	0.0	0.0

precipitation between 1981 and 1982 resulted in water quality differences between years. Based on long-term averages from Lead and Spearfish, South Dakota, precipitation in the Black Hills in 1982 was slightly higher than normal while 1981 was a year of lower than normal precipitation (Figure 2). Increased runoff resulted in elevated stream flows throughout the summer of 1982. Mean values for dissolved oxygen and organic phosphorous were significantly ($P < 0.05$) higher in 1982, while mean conductivity and pH were significantly ($P < 0.05$) lower. Mean fecal coliform number, ammonia nitrogen, and total (Kjeldahl) nitrogen values were higher from all stations in 1982, however, these differences were not significant ($P > 0.05$).

Analysis of variance indicated that seven physiochemical variables varied significantly ($P < 0.05$) among stations. The Waller-Duncan k-ratio t-test revealed where differences among station means occurred for those variables (Table 4). Mean turbidity was significantly ($P < 0.05$) higher at stations 3, 4, and 5 than upstream at stations 1 and 2. The highest mean turbidity value was at station 3 (35.7 ntu) just below the development site. Turbidity decreased with gradient downstream from station 3 to a mean of 26.9 ntu at station 5, however, these differences were not significant ($P > 0.05$). Mean temperatures at stations 3 and 4 were significantly ($P < 0.05$) higher than upstream at stations 1 and 2. Temperature decreased significantly ($P < 0.05$) from a mean of 12.5 C at station 3 to 10.4 C downstream at station 5. Mean temperature at station 5 was not significantly ($P > 0.05$) different from mean values from stations 1 and 2. Conductivity, hardness, alkalinity, and pH were all significantly ($P < 0.05$) lower at station 2 than at any of the other stations. Mean fecal coliform number at

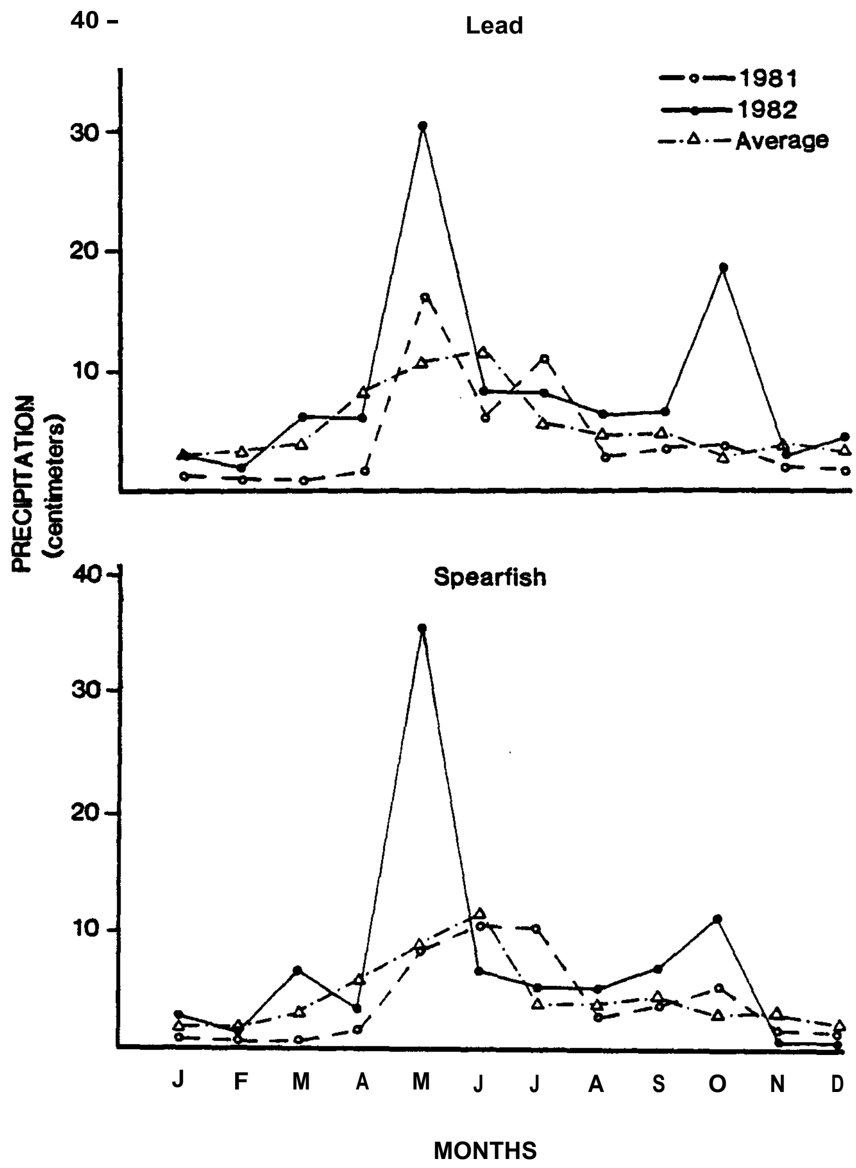


Figure 2. Monthly precipitation levels for the Lead and Spearfish, South Dakota, gauging stations for 1981 and 1982.

Table 4. Waller-Duncan's k-ratio t-test for water quality variables exhibiting significant ($P \leq 0.05$) differences among stations. Mean values for each station appear in parentheses.

Parameter	Station				
			X		
Turbidity (ntu)	3 (35.7)	4 (28.4)	5 (26.9)	1 (7.1)	2 (3.9)
Temperature (C)	3 (12.5)	4 (11.5)	5 (10.4)	1 (10.1)	2 (9.1)
pH (units)	1 (7.5)	4 (7.5)	3 (7.4)	5 (7.3)	2 (7.1)
Conductivity (pmhos)	5 (167.6)	3 (163.3)	4 (163.2)	1 (161.0)	2 (115.4)
Hardness (mg/1 as $CaCO_3$)	1 (110.3)	4 (105.3)	5 (105.0)	3 (102.5)	2 (82.5)
Alkalinity (mg/1 as $CaCO_3$)	1 (101.2)	5 (100.0)	4 (97.8)	3 (94.1)	2 (81.9)
Fecal coliforms (#/100 ml)	5 (309.4)	1 (173.7)	4 (163.9)	3 (100.2)	2 (78.75)

station 5 was significantly ($P < 0.05$) higher than at stations 2 and 3. No significant ($P > 0.05$) differences were detected among the five stations for phosphorous (orthophosphate and organic phosphate) or nitrogen (ammonia and organic), however, nutrient levels were highest at stations 4 and 5 (Appendix Table 2).

Stepwise discriminant analysis could not statistically separate the five stations based upon the limited number of physiochemical variables measured. Four discriminant functions were derived from the water quality data. The variables hardness, conductivity, organic phosphorous, and dissolved oxygen were the most important discriminating variables in the model (Appendix Table 3). The four water quality variables accounted for 54.0% of the variation in the discriminant model. The reclassification procedure of the discriminant analysis correctly reclassified only 42.5% of the station dates.

Macroinvertebrates

Fifty-four genera representing 16 orders of aquatic macroinvertebrates were collected from the five stations in the Slate Creek study area (Appendix Table 4). Analysis of variance indicated that eight invertebrate genera from the artificial substrate samples and eight invertebrate genera from the benthos samples varied significantly ($P < 0.05$) in abundance between years (Table 5). Seven of the eight genera from the artificial substrate samples were collected in significantly ($P < 0.05$) higher numbers in 1982, while seven of the eight genera from the benthos samples were collected in significantly ($P < 0.05$) higher numbers in 1981. *Pisidium* was the only invertebrate

Table 5. Invertebrate genera exhibiting significant ($P \leq 0.05$) differences in abundance between 1981 and 1982. Numerical values represent mean numbers of organisms per sample collected from all stations.

Artificial Substrate			Benthos		
Genera	1981	**1982**	Genera	1981	1982
Baetis	4.6	30.8	Alloperla	0.1	1.0
Diamesa	10.1	27.9	Gammarus	0.9	0.1
Heterotrissocladius	8.7	22.9	Paraleptophlebia	1.4	0.1
Nais	6.6	16.8	Pisidium	27.3	4.3
Optiacervus	0.3	1.7	Procladius	18.4	2.3
Physa	4.8	1.7	Sialis	1.4	0.1
Pisidium	3.5	13.3	Tabanus	3.1	0.2
Polypedilum	0.2	2.5	Zavrelimyia	5.2	0.3

genus that varied significantly ($P < 0.05$) in abundance between years from both sample types.

Analysis of variance indicated that 12 invertebrate genera collected from the artificial substrate samples varied significantly ($P < 0.05$) in abundance among stations. The Waller-Duncan k-ratio t-test revealed where differences among station means occurred for those invertebrate genera (Table 6). Mean numbers of *Malenka* collected at station 1 were not significantly ($P > 0.05$) different from station 2, however, *Malenka* numbers from both stations 1 and 2 were significantly ($P < 0.05$) higher than at stations 3, 4, and 5. Numbers of *Nais* collected from stations 3, 4, and 5 were significantly ($P < 0.05$) higher than at stations 1 and 2. *Hesperophylax* was most abundant at station 2 and was significantly greater ($P < 0.05$) there than at any of the other stations. *Physa* was collected in highest numbers from station 1, and was significantly ($P < 0.05$) greater than at the remaining four stations. No trends in macroinvertebrate distribution among stations were detected using data from the artificial substrate samples.

Analysis of variance indicated that 10 invertebrate genera collected from the benthos samples varied significantly ($P < 0.05$) in abundance among stations. The Waller-Duncan k-ratio t-test revealed where differences among station means occurred for those invertebrate genera (Table 7). *Alloperla,* *Optiocervus,* and *Physa* were collected in significantly ($P < 0.05$) higher numbers at station 1 than from the remaining four stations. Mean numbers of *Gammarus* and *Simulium* were significantly ($P < 0.05$) greater at station 3 than numbers collected from the other four stations. *Hesperophylax* and *Tabanus* were collected

Table 6. Waller-Duncan's k-ratio t-test for macroinvertebrates exhibiting significant (P < 0.05) differences among stations from artificial substrate collections. Mean values for each station appear in parentheses.

Invertebrate	Station x					Invertebrate	Station x				
Dubiraphia	4	3	5	1	2	Parapholox	4	1	2	3	5
	(0.8)	(0.1)	(0.1)	(0.0)	(0.0)		(1.0)	(0.1)	(0.0)	(0.0)	(0.0)
Gammarus	5	3	4	1	2	Physa	1	5	4	3	2
	(2.3)	(1.2)	(0.6)	(0.1)	(0.0)		(10.4)	(3.0)	(1.7)	(1.o)	(0.1)
Hesperophylax	2	1	3	5	4	Pisidium	4	1	3	2	5
	(3.4)	(0.8)	(0.4)	(0.3)	(0.3)		(24.3)	(12.0)	(3.4)	(1.2)	(1.2)
Malenka	2	1	3	5	4	Procladius	3	5	4	1	2
	(9.7)	(6.7)	(0.7)	(0.2)	(0.1)		(16.4)	(16.2)	(15.5)	(7.2)	(7.0)
Nais	5	3	4	1	2	Sialis	4	5	1	3	2
	(22.2)	(18.1)	(14.5)	(2.4)	(1.1)		(2.9)	(2.5)	(1.3)	(0.5)	(0.1)

Table 6. Continued

Invertebrate	Station $\bar{x}$					Invertebrate	Station $\bar{x}$				
Paraleptophlebia	4 (19.3)	2 (7.7)	1 (7.2)	5 (6.8)	3 (6,1)	Zaitzevia	1 (4.7)	4 (2.4)	3 (1.6)	5 (0.8)	2 (0.0)

Table 7. Waller-Duncan's k-ratio t-test for macroinvertebrates exhibiting significant ($P \leq 0.05$) differences among stations from benthos collections. Mean values for each station appear in parentheses.

Invertebrate	Station $\bar{x}$					Invertebrate	Station $\bar{x}$				
Alloperla	1	2	3	4	5	*Optiocervus*	1	3	4	5	2
	(2.5)	(0.2)	(0.0)	(0.0)	(0.0)		(2.2)	(1.0)	(0.8)	(0.3)	(0.1)
Cryptochironomus	5	4	3	1	2	*Physa*	1	5	4	3	2
	(1.3)	(0.4)	(0.1)	(0.1)	(0.0)		(4.0)	(0.5)	(0.2)	(0.1)	(0.1)
Diamesa	2	3	5	1	4	*Pisidium*	4	1	2	5	3
	(9.1)	(7.2)	(4.5)	(4.0)	(1.4)		(25.2)	(22.1)	(19.5)	(8.0)	(4.1)
Gammarus	3	5	4	1	2	*Simulium*	3	2	1	4	5
	(1.4)	(0.5)	(0.5)	(0.1)	(0.0)		(16.1)	(3.3)	(2.2)	(0.2)	(0.1)
Hesperophylax	2	5	1	3	4	*Tabanus*	2	1	5	3	4
	(3.4)	(0.5)	(0.4)	(0.4)	(0.3)		(5.5)	(1.0)	(0.9)	(0.5)	(0.5)

in significantly ($P < 0.05$) higher numbers at station 2 than from the remaining four stations. Cryptochironomus was most abundant at station 5, and was significantly greater ($P < 0.05$) than at the other four stations. Gammarus, Hesperophylax, Physa, and Pisidium were the only invertebrate genera that varied significantly ($P. < 0.05$) in abundance among stations from both sample types.

Stepwise discriminant analysis showed that statistical separation of the five stations was not possible based on analysis of the invertebrate community assemblages for either the artificial substrate or the benthos collections. Four discriminating functions were derived for each set of data. The genera Physa, Dubiraphia, Gammarus, Sialis, and Hesperophylax were the most important discriminating variables from the artificial substrate data (Appendix Table 4). The genera Physa, Simulium, Gammarus, Tabanus, and Procladius were the most important discriminating variables from the benthos data (Appendix Table 5).

The five variables from the artificial substrate data accounted for 67.1% of the total variation in invertebrate numbers among stations. The five variables from benthic data explained 56.8% of the total variation in invertebrate numbers among stations. The classification procedure correctly reclassified 51.5% of the station-dates with the artificial substrate data and 46.1% of the station-dates with the benthic data. Physa was the most important discriminating variable from both the artificial substrate and the benthos collections. Physa accounted for 29.8% of the total variation in invertebrate numbers among stations for the artificial substrate data and 25.4% of the total variation from the benthos data.

In 1981 and 1982, 34 and 33 invertebrate genera, respectively, were included as variables for the cluster analysis program. Percent similarity dendograms constructed for each sample type, for each year, (Appendix Figures 1-4) indicated that station-dates grouped more in a temporal dimension than in a spatial dimension. Distinct groupings were not achieved with any greater efficiency from one particular sample type in either year.

Analysis of variance indicated that only two of the four biological indices for each sample type differed significantly ($P < 0.05$) among stations. The Waller-Duncan k-ratio t-test revealed where differences among station means occurred for those indices (Table 8). The biotic index detected similar differences among stations from both artificial substrate and benthic data. For the artificial substrate data, station 2 had a significantly ($P < 0.05$) lower biotic index value than stations 3, 4, and 5, but was not significantly ($P > 0.05$) different than station 1. For the benthic data, station 1 had a significantly lower biotic index value than stations 3, 4, and 5, but was not significantly ($P > 0.05$) different from station 2. The Shannon-Weaver index of species diversity calculated from the artificial substrate collections and species evenness calculated from the benthos collections were the other biological indices exhibiting significant ($P < 0.05$) differences among stations. The biotic index calculated from the benthos data was the only biological index that differed significantly ($P < 0.05$) between years. The mean for all stations in 1981 was 2.76, this was significantly ($P < 0.05$) higher than the 1982 mean of 2.39.

Table 8. Waller-Duncan's k-ratio t-test for biological indices exhibiting significant ($P < 0.05$) differences among stations. Mean values for each station appear in parentheses.

Artificial substrate						Benthos					
Biological index	Station					Biological index	Station				
Shannon-Weaver species diversity	4 (2.25)	1 (2.19)	3 (2.09)	2 (2.09)	5 (2.00)	Species richness	1 (3.12)	3 (2.90)	2 (2.60)	4 (2.56)	5 (2.43)
Biotic index	5 (2.63)	3 (2.54)	4 (2.47)	1 (2.33)	2 (2.17)	Biotic index	3 (2.74)	4 (2.70)	5 (2.66)	2 (2.47)	1 (2.20)

Biotic index values, based upon the combined samples collected in late spring, early summer, late winter, and late autumn, were evaluated using the criteria proposed by Hilsenhoff (1977) (Table 9). Using this criteria, only station 2 considering artificial substrate data, and station 1 considering benthic data, would be classified as good quality water. The remaining stations would be classified as fair quality water.

Table 9. Evaluation of water quality using biotic index values of samples collected in late spring, early summer, late summer, and late autumn (Hilsenhoff 1977).

Biotic index	Water quality	State of stream
<1.75	Excellent	Clean undisturbed
1.75-2.25	Good	Some enrichment or disturbance
2.25-3.00	Fair	Moderate enrichment or disturbance
3.00-3.75	Poor	Significant enrichment or disturbance
>3.75	Very poor	Gross enrichment or disturbance

DISCUSSION

Water quality differences between years and among stations were discernible in Slate Creek from both physiochemical and macroinvertebrate evaluations. Water temperature, turbidity, fecal coliform number, and nutrient levels were influenced by activities within the Slate Creek watershed, while conductivity, hardness, alkalinity, and pH differed due to natural variation within the watershed.

The presence of livestock and landscaping activities within the Deerfield Park Resort development were the primary sources for water quality variation among stations. The upper range of values for those water quality variables affected, did not exceed standards established under South Dakota Public Law 94:03:02, for waters classified as cold water permanent fish life propagation water and as a limited contact recreation water. Increased runoff and elevated stream flows were responsible for the variation in water quality between years. Subsequent changes in the macroinvertebrate community were evident in response to variations in water quality between years and among stations

Livestock grazing can affect all four components of the aquatic system: streamside vegetation, stream channel morphology, shape and quality of the water column, and the structure of the soil portion of the streambank (Platts 1978). Nutrient enrichment and increased bacterial concentrations, resulting from livestock grazing practices have been recognized as primary sources for water quality deterioration on western rangelands (Meehan and Platts 1978; Stephenson and Street 1978; EPA 1979; Robbins 1979). Nutrient accrual is generally regarded as having detrimental effects on lotic systems by increasing productivity, turbidity, temperature, and the biochemical oxygen demand of the aquatic environment (Hynes 1970). In regard to bacterial concentrations, they do not relate directly to the suitability of fish habitat; they are important to water quality and, therefore, relate indirectly to fish habitat (Meehan and Platts 1978).

The highest grazing intensity within the Slate Creek study area occurred at stations 3, 4, and 5 where meadows adjacent to the stream were utilized as winter pastures. During the remainder of the year the study area was not grazed with the exception of a couple of access locations where cattle were permitted to water at the creek. Non-point source contamination from livestock grazing resulted in elevated, but non-signficant differences, in nitrogen and phosphorous levels and fecal coliform number. These contaminants tended to accumulate downstream from the areas where grazing intensity was the highest.

Stream channel sedimentation caused by soil erosion on rangelands was regarded by Moore (1976) as a major problem associated with livestock grazing. Stream bank degradation by livestock was not a major problem in the Slate Creek study area, however, turbidity measurements and percent composition of the channel substrate by sand and silt were consistently higher in Slate Creek at the downstream stations. Excavati-'n of the lake basin, clearcutting of an adjacent hillside, and other related landscaping activities within the Deerfield Park Resort development increased the silt load of the stream below the resort dam.

While the development site was the primary factor affecting the elevated turbidity in Slate Creek below stations 1 and 2, it is not known to what extent the activity of livestock contributed to maintaining increased turbidity levels downstream.

Large quantities of fine sediment change the structure of aquatic communities, diminish productivity, and reduce the permeability of channel bottom materials used by fish for spawning (Meehan and Platts 1978). Silt alters aquatic environments chiefly by screening out light, increasing heat radiation, blanketing the stream bottom, and retaining organic material and other substances which create unfavorable bottom conditions (Ellis 1936). Sedimentation is a natural process in all lotic systems with fluctuating flow regimes. It is difficult to assess the effects of sedimentation due to the inherent variation in stream characteristics and because watershed practices often have multiple effects on stream ecosystems (Murphy et al. 1981). The erosional nature of headwater streams provides a degree of resiliency in maintaining gravel-rubble substrates in the stream channel. Murphy et al. (1981) researched small streams in the Cascade Mountain range in Oregon and found that small streams have a high capacity to flush introduced sediment downstream because of their steep gradient. They found that most sediment less than 1 mm in size apparently was removed from the surface layers of the stream bed within one to two years. When water behind the Deerfield Park Resort dam is impounded and downstream flows are regulated, the natural capacity of the stream to scour its channel materials free from fine sediments may be altered.

Faunal differences within the macroinvertebrate community of Slate Creek varied among stations and can be divided among the four functional trophic groups. The specific factors determining the distribution of an invertebrate population within a section of stream having a suitable food supply may often be controlled by such factors as sediment, particle **size**, current, competition for space, and predation (Cummins 1975). The rubble-gravel substrate found at stations 1 and 2 favored the organisms that function as shredders and scrapers. Two plecopterans, Alloperla and Malenka, and one trichopteran, Hesperophylax, belong in this functional group (Merrit and Cummins 1978) and were most numerous at stations 1 and 2. The gastropod, Physa, functions as a scraper and is sensitive to increases in siltation rates (Hart and Fuller 1974). Stream habitat was most suitable for Physa at station 1. Hard water is favorable for molluscs (Hart and Fuller 1974) and lower hardness at station 2 may have been the cause for reduced numbers of Physa within that section of stream. The fine sediments that have accumulated downstream from the Deerfield Park Resort dam created favorable conditions for the oligocheate, Nais. Nais is adapted to burrowing in the soft sediments (Hart and Fuller 1974) and obtains nutrients by ingesting quantities of the substrate (Pennack 1978).

Two common approaches toward assessing stream environmental quality by means of aquatic macroinvertebrates are the use of the animals as indicator organisms and the evaluation of community indices (Jones et al. 1981). The discriminating ability of the macroinvertebrate data was better than that of the water quality data. Cluster analysis utilizing presence and absence data did not separate stations as well as did biological indices. Of the indices utilized, the biotic index provided the best separation of stations in relation to physiochemical variables. The biotic index summarizes the deviation of the observed community of animals from the community that would be expected if the water were unenriched (Chutter 1972). Biotic index values are determined by integrating the biology, the natural history, and tolerance to organic pollution of individual species collected (Jones et al. 1981). The biotic index is only sensitive to the effects of organic enrichment and strongest interpretations are made using the index when identifications are to the species level (Tracy 1979). Unfortunately, due to time and monetary constraints, identification of invertebrates in this study was accomplished only to the genus. It is believed that greater sensitivity in station separation may have been achieved with identification to the species level.

Livestock grazing, construction activities associated with the Deerfield Park Resort development, and natural variation in the watershed combined to create distinct water quality differences among the five stations in the Slate Creek study area. It was demonstrated that variations in the flow regime resulting from yearly precipitation differences, will also affect certain water quality variables. All these factors created environmental conditions which determined benthic macroinvertebrate distribution and abundance.

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APPENDIX

Appendix Table 1. Pollution tolerance values (a_1) of macroinvertebrates collected from Slate Creek, South Dakota. Values of a_1 range from 0 (pollution-intolerant) to 5 (very pollution tolerant).

Taxon	a_1	Taxon	a_1
Ephemeroptera		Megaloptera	
Baetidae		Sialidue	
Baetis	2	Sialis	2
Leptophlebiidae			
Paraleptophlebia	1	Coleoptera	
Tricorythidae		Elmidae	
Tricorythodes	2	Dubiraphia	3
		Optiocervus	2
Plecoptera			
Chloroperlidae		Amphipoda	
Alloperla	0	Gammaridue	
Nemouridae		Gammarus	2
Malenka	0		
Perlidae		Diptera	
Acroneuria	0	Ceratopogonidae	
Perlodidae		Probezzia	3
Isoperla	0	Chironomidae	
		Chironomus	5
Trichiptera		Corynoneura	2
Helicopsychidae		Cryptochironomus	4
Helicopsyche	2	Diamesa	2
Hydropsychidae		Heterotrissocladius	2
Cheumatopsyche	3	Microspectra	3
Lepidostomatidae		Microtendipes	3
Lepidostoma	1	Paratendipes	2
Limnephilidae		Polypedilum	3
Hesperophylax		Procladius	3
Limnephilus		Prodiamesa	2
		Zavrelimyia	4
Odonata			
Aeshnidae			
Aeshna	3		
Corduliidae			
Somatochlora	0		

Appendix Table 2. Water quality data collected from Slate Creek between 20 May 1981 and 4 September 1982.

Station/ Date	Temperature (C)	Dissolved oxygen (mg/1)	pH	Turbidity (ntu)	Conductivity (pmho)	Total hardness (mg/1 $CaCO_3$)	Alkalinity (mg/1 $CaCO_3$)	Fecal califorms (#/100 ml)	Organic phosphate (mg/1)	Ortho phosphate (mg/1)	Ammonia nitrogen (mg/1)	Kjeldahl nitrogen (mg/1)
STATION 1												
05/20/81	6.0	10.0	7.5	5.0	161	120	75	10	<0.010	0.020	<0.03	0.28
06/10/81	12.0	10.0	7.8	20.0	170	110	95	18	0.005	0.033	<0.03	0.26
07/01/81	18.0	7.5	7.4	3.0	200	120	120	208	0.015	0.040	<0.03	0.17
07/22/81	11.0	9.2	8.5	5.0	210	130	100	480	0.005	0.065	<0.03	<0.03
08/11/81	14.0	8.2	8.2	27.0	180	110	90	70	0.023	0.056	<0.03	0.09
08/30/81	15.0	8.3	7.8	2.0	190	110	110	205	0.008	0.033	<0.03	0.07
10/10/81	7.0	9.2	7.9	2.0	160	110	100	20	<0.001	0.021	<0.03	0.73
12/14/81	1.0	10.6	8.0	8.0	122	130	125	30	0.030	0.020	<0.03	1.13
02/16/82	3.0	8.5	7.5	6.0	130	125	125	110	0.019	0.063	0.06	0.55
04/06/82	1.0	10.0	7.7	1.7	110	105	110	300	0.016	<0.010	<0.03	0.33
05/26/82	9.0	9.8	7.3	2.5	148	90	90	165	0.015	0.020	<0.03	0.36
06/08/82	13.0	10.0	7.0	4.5	160	95	90	30	0.042	0.020	<0.03	0.32
06/29/82	12.0	11.8	6.5	5.1	160	100	90	114	0.055	0.022	<0.03	0.44
07/21/82	15.0	11.0	6.6	5.0	160	70	70	230	0.074	0.035	<0.03	0.47
08/09/82	11.0	11.0	7.0	11.0	155	120	120	590	0.032	0.024	0.08	0.30
09/04/82	13.0	11.6	7.5	6.0	160	120	110	200	0.007	0.045	<0.03	0.42
STATION 2												
02/20/81	7.0	9.5	7.2	0.0	105	70	60	<2	<0.01	0.015	<0.03	0.18
06/10/81	8.0	9.5	7.6	7.0	105	65	60	22	0.001	0.029	<0.03	0.08
07/01/81	11.0	9.0	6.9	2.0	120	90	70	<2	0.140	0.045	<0.03	0.48
07/22/81	11.0	7.0	6.8	18.0	120	110	90	8	0.005	0.040	<0.03	<0.03
08/11/81	12.0	8.4	7.8	2.0	130	80	70	<2	<0.001	0.047	<0.03	0.09
08/30/81	11.0	8.0	7.7	3.0	130	80	70	9	0.017	0.033	<0.03	0.04
10/10/81	7.0	8.7	7.7	1.5	140	80	90	<2	0.013	0.075	<0.03	0.29
12/14/81	3.0	9.6	7.7	0.5	90	90	105	<2	<0.001	0.029	<0.03	0.08
02/16/82	3.0	10.4	7.1	0.8	88	95	90	<2	<0.010	0.028	0.03	0.10
04/06/82	4.0	11.5	7.4	0.8	70	75	95	<1	0.037	0.010	<0.03	0.19
05/26/82	11.0	9.0	6.5	2.0	110	90	60	874	0.015	0.020	0.03	0.20
06/08/82	10.0	9.1	6.6	3.0	118	90	80	40	0.031	0.019	0.04	0.41
06/29/82	12.0	11.0	6.0	5.3	120	65	100	32	0.048	0.033	<0.03	0.44
07/20/82	12.0	12.1	6.7	3.2	130	70	70	8	0.024	0.040	<0.03	0.18
08/09/82	12.0	7.0	6.7	6.5	140	90	100	58	0.022	0.030	<0.03	0.15
09/04/82	12.0	11.8	7.2	7.0	130	80	100	196	0.004	0.044	<0.03	0.28

Appendix Table 2. (continued)

Station/ Date	Temperature (C)	Dissolved oxygen (mg/1)	pH	Turbidity (ntu)	Conductivity (mho)	Total hardness (mg/1 $CaCO_3$)	Alkalinity (mg/1 $CaCO_3$)	Fecal califorms (11/100 ml)	Organic phosphate (mg/1)	Ortho phosphate (mg/1)	Ammonia nitrogen (mg/1)	Kjeldahl nitrogen (mg/1)
STATION 3												
05/20/81	10.0	9.2	7.6	40.0	180	100	80	26	<0.010	0.035	<0.03	0.37
06/10/81	14.5	9.2	7.7	45.0	174	85	75	40	0.016	0.051	<0.03	0.39
07/01/81	22.0	8.1	7.4	66.0	210	100	90	10	0.007	0.283	<0.03	0.47
07/22/81	18.0	7.0	7.5	200.0	190	120	100	100	0.057	0.128	0.08	0.75
08/11/81	15.0	9.6	8.1	24.0	190	120	60	30	0.015	0.050	<0.03	0.25
08/30/81	19.0	9.6	7.6	60.0	220	105	100	40	0.024	0.073	<0.03	0.23
10/10/81	10.0	8.4	7.4	18.0	180	100	90	10	0.013	0.057	<0.03	0.38
12/14/81	1.0	10.0	7.8	8.1	125	115	125	<2	0.014	0.029	<0.03	0.16
02/16/82	2.0	3.5	7.4	17.0	138	130	110	8	0.019	0.083	0.06	0.50
04/06/82	2.5	11.0	8.1	4.5	100	100	115	<1	0.039	0.011	<0.03	0.24
05/25/82	12.0	8.0	7.3	12.0	165	110	80	208	0.025	0.025	<0.03	0.35
06/08/82	13.0	10.4	6.7	6.3	156	85	85	70	0.045	0.033	<0.03	0.44
06/29/82	13.0	7.5	6.0	35.0	150	100	100	440	0.160	0.057	0.04	0.81
07/20/82	16.0	10.3	7.1	7.9	115	70	100	30	0.067	0.030	<0.03	0.27
08/09/82	17.0	**9.8**	7.4	6.6	150	100	95	8	0.022	0.032	<0.03	0.15
09/04/82	15.0	10.2	7.5	21.0	170	100	110	580	0.033	0.047	<0.03	0.44
STATION 4												
05/20/81	12.0	9.6	7.4	28.0	195	100	75	222	<0.010	0.016	<0.03	0.36
06/10/81	15.0	9.6	7.7	25.0	170	100	85	40	0.012	0.055	<0.03	0.43
07/01/81	17.0	12.2	7.7	38.0	180	100	100	90	0.023	0.047	<0.03	0.18
07/22/81	16.0	7.8	7.1	200.0	190	120	110	220	0.017	0.103	<0.03	0.55
08/11/81	12.0	9.8	7.8	18.0	170	120	50	80	0.020	0.070	<0.03	0.20
08/30/81	13.0	9.3	7.9	45.0	185	110	90	110	<0.001	0.051	<0.03	0.23
10/10/81	7.0	9.7	9.4	8.5	160	110	110	30	<0.001	0.037	<0.03	0.43
12/14/81	2.0	9.8	7.8	4.5	130	120	140	2	0.003	0.020	<0.03	0.07
02/16/82	2.0	10.0	7.5	6.1	150	120	120	320	0.095	0.315	1.00	4.25
04/06/82	2.0	10.6	7.9	3.0	100	105	115	5	0.040	0.010	<0.01	0.38
05/25/82	14.0	7.5	7.5	13.0	165	120	80	170	0.020	0.025	<0.03	0.39
06/08/82	13.0	9.3	6.5	13.0	156	105	90	50	0.055	0.035	<0.03	0.50
06/29/82	13.0	8.2	6.0	27.0	150	90	100	330	0.158	0.044	<0.03	0.80
07/20/82	16.0	**8.6**	7.4	8.5	175	60	100	10	0.058	0.030	<0.03	0.33
08/09/82	16.0	12.5	6.8	7.4	165	95	100	84	0.028	0.023	0.10	0.57
09/04/82	14.0	9.2	7.4	9.0	170	110	100	860	0.040	0.047	<0.03	0.41

Appendix Table 2. (continued)

Station/ Date	Temperature (C)	Dissolved oxygen (mg/1)	pH	Turbidity (ntu)	Conductivity (mho)	Total hardness (mg/1 $CaCO_3$)	Alkalinity (mg/1 $CaCO_3$)	Fecal califorms (11/100 ml)	Organic phosphate (mg/1)	Ortho phosphate (mg/1)	Ammonia nitrogen (mg/1)	Kjeldahl nitrogen (mg/1)
STATION 5												
05/20/81	13.0	9.3	7.5	28.0	199	105	70	212	0.010	0.017	<0.03	0.43
06/10/81	12.5	9.3	7.6	37.0	450	100	90	625	0.001	0.070	<0.03	0.35
07/01/81	12.0	7.8	7.7	14.0	170	110	90	230	0.033	0.070	<0.03	0.21
07/22/81	15.0	8.0	7.2	210.0	190	120	100	470	0.009	0.097	<0.03	0.39
08/11/81	10.0	9.2	7.9	14.0	170	120	100	230	0.003	0.020	<0.03	0.16
08/30/81	11.0	8.8	8.0	30.0	75	110	120	320	0.030	0.045	<0.03	0.10
10/10/81	6.0	9.4	7.6	5.6	160	100	90	70	0.002	0.105	<0.03	0.47
12/14/81	1.0	3.6	7.6	3.9	132	120	120	28	<0.001	<0.026	<0.03	0.15
02/16/82	2.0	9.5	7.4	6.5	85	115	140	960	0.069	0.655	2.30	8.00
04/06/82	1.0	10.6	7.5	2.9	110	95	110	<10	0.001	0.043	<0.03	0.23
05/25/82	11.0	13.0	6.7	13.0	160	100	85	146	0.030	0.025	<0.03	0.45
06/08/82	13.0	10.4	6.7	6.3	156	85	85	90	0.070	0.046	<0.03	0.51
06/29/82	13.0	7.2	6.2	29.0	160	100	90	250	0.170	0.042	<0.03	0.96
07/20/82	15.0	11.4	7.2	14.0	130	100	100	130	0.073	0.038	<0.03	0.42
08/09/82	16.0	10.8	6.7	7.9	165	100	110	160	0.038	0.032	<0.03	0.26
09/04/82	15.0	10.8	7.4	8.0	170	100	100	1.020	0.038	0.052	<0.03	0.45

Appendix Table 3. Standardized canonical discriminant coefficients for the four derived functions, and percent variation explained for water quality variables included in the stepwise discriminant analysis.

Water quality parameter	Standardized canonical discriminant coefficients[a]				Cumalative percent variation explained
	Function 1	Function 2	Function 3	Function 4	
Hardness	0.98	-0.34	-0.05	-0.41	34.1
Conductivity	0.60	0.46	-0.07	0.71	42.7
Organic phosphorous	0.46	0.62	0.65	-0.37	48.4
Dissolved oxygen	0.46	-0.58	0.72	0.35	54.0

[a]Absolute value indicates relative importance in the function derived.

Appendix Table 4. Composite list of macroinvertebrates collected between 1 July 1981 and 4 September 1982 from Slate Creek, South Dakota. Asterisk indicates from which sample stations organisms were collected.

Taxon	Station 1	2	3	4	5
Ephemeroptera					
Leptophlebiidae					
Paraleptophlebia	***	***	***	***	***
Tricarythidae					
Tricorythodes					
Baetidae					
Baetis	***	***	***	***	***
Plecoptera					
Perlodidae					
Tsoperla	***	***	***	***	***
Perlidae					
Acroneuria	***	***			
Nemouridae					
Malenka	***	***	***	***	***
Chloroperlidae					
Alloperla	***	***	***		
Trichoptera					
Limnephillidae					
Hesperophylax	***	***	***	***	***
Limnephilus	***	***	***	***	***
Hydropsychidae					
Cheumatopsyche	***	---	***	***	***
Lepidostomatidae					
Lepidostoma	***	***	***	***	***
Helicopsycheidae					
Helicopsyche	-		***	***	***
Hemiptera					
Gerridae					
Gerris			***		
Metrobates		***			
Mesoveliidae					
hesovel is			***		

Appendix Table 4 continued.

Taxon	Stat ion				
	1	2	3	4	5
Odonata					
Aeshnidae					
Aeshna			***	***	
Corduliidae					
Somatochlora	***	---	***	***	***
Amphipoda					
Gammaridae					
Gammarus	***	***	***	***	***
Megaloptera					
S ial idae					
S ial is	***	***	***	***	***
Nematomorpha					
Gordiidae					
Gordius		***	---	***	
Hirudinea					
Glossiphoniidae					
Helobdella	***	***	***	***	***
Glossiphonia			***	***	***
Pelecypoda					
Sphaeriidae					
Pis idium	***	***	***	***	***
Gastropoda					
Planorbidae					
Parapholox	***	***	***	***	
Gyraulus	***	***	---	***	
Physidae					
Physa	***	***	***	***	***
Oligochaeta					
Naididae					
Nais	***	***	***	***	***
Hydracarina					
Hydrachnidae					
Hydrachna	***	***	***	***	***
Sperchonidae					
Sperchon			***	***	

Appendix Table 4 continued.

Taxon	Station 1	2	3	4	5
Arrenuridae					
Arrenurus	***	***		***	***
Coleoptera					
Elmidae					
Optiocervus	***	***	***	***	***
Zaitzevia	***	***	***	***	***
Dubiraphia		***	***	***	***
Dyt isc idae					
Eretes	***		***		
Agabus	***	***	***	***	
Halipidae					
Halipus	***				
Coelenterata					
Hydridae					
Hydra	***	***	***	***	***
Diptera					
Chironomidae					
Microspectra	***	***	***	***	***
Procladius	***	***	***	***	***
Prodiamesa	***	***	***	***	***
Diamesa	***	***	***	***	***
Synposiocladius	***	***	---	--	---
Heterotrissocladius	***	***	***	***	***
Zavreiimyia	***	***	***	***	***
Polypedilum	***	***	***	***	***
Paratendipes	***	***	***	***	***
Cryptochironomus	***		***	***	***
Corynoneura	***	***	***	***	***
Chironomus	---	---	***	***	
Microtendipes	***	***	***	***	***
Dixidae					
Dixa	***	***	***	***	***
Ceratopogonidae					
Probezzia	***	***	***	***	***
S imul Adae					
Simulium	***	***	***	***	***
Tabanidae					
Tabanus	***	***	***	***	***

Appendix Table 5. Standardized canonical discriminant coefficients for the four derived functions, and percent variation explained for macroinvertebrate genera from artificial substrate collections, included in the stepwise discriminant analysis.

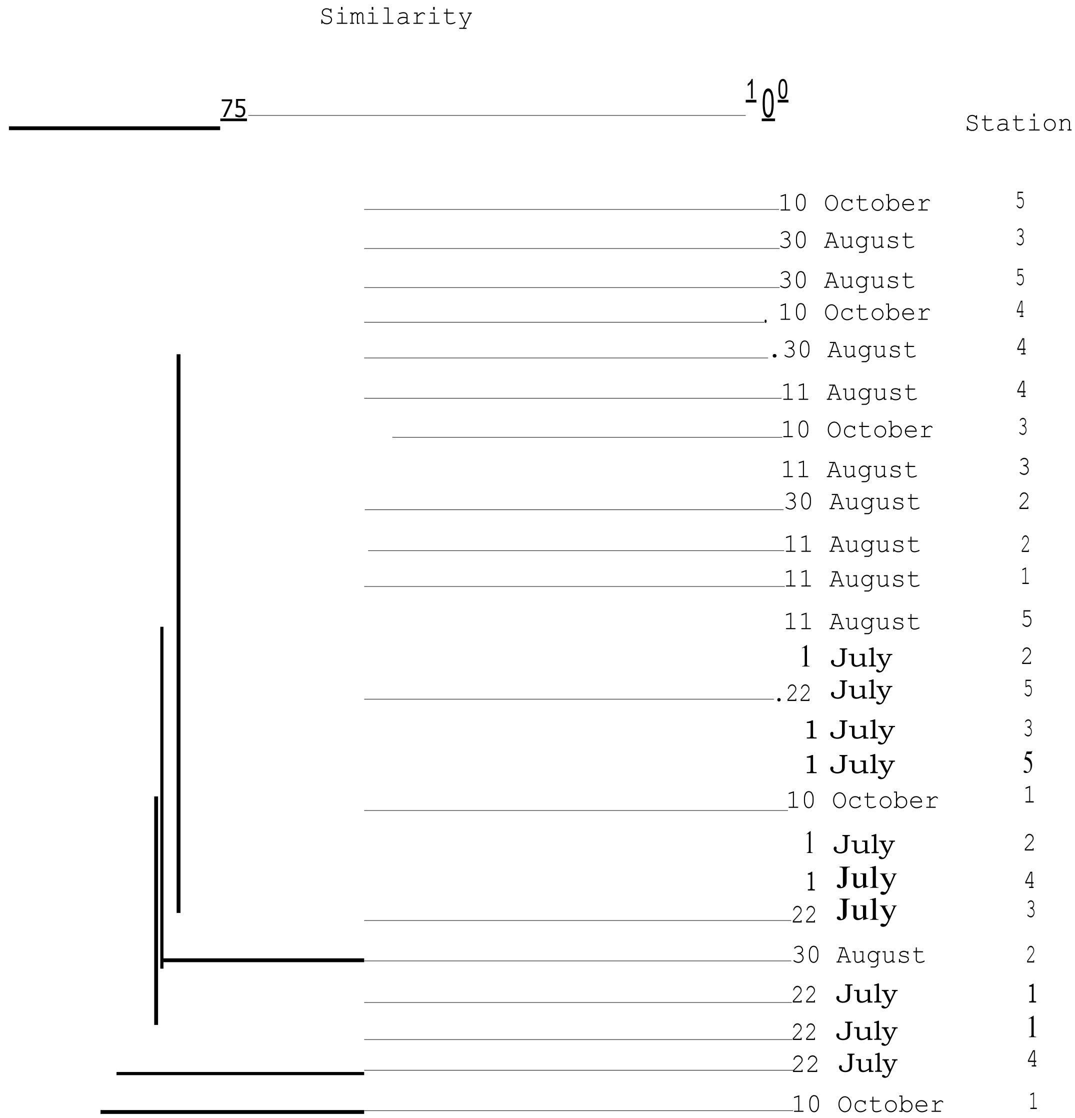
Invertebrate genera	Standardized canonical discriminant coefficients[a] Function 1	Function 2	Function 3	Function 4	Cumulative percent variation explained
Physa	-0.09	0.95	0.17	0.07	29.8
Dubiraphia	0.38	-0.25	0.75	-0.05	42.8
Gammarus	0.45	0.12	-0.71	0.10	53.1
Sialis	0.66	0.11	0.07	0.64	61.6
Hesperophylax	-0.54	-0.20	0.20	0.73	67.1

[a]Absolute value indicates relative importance in the function derived.

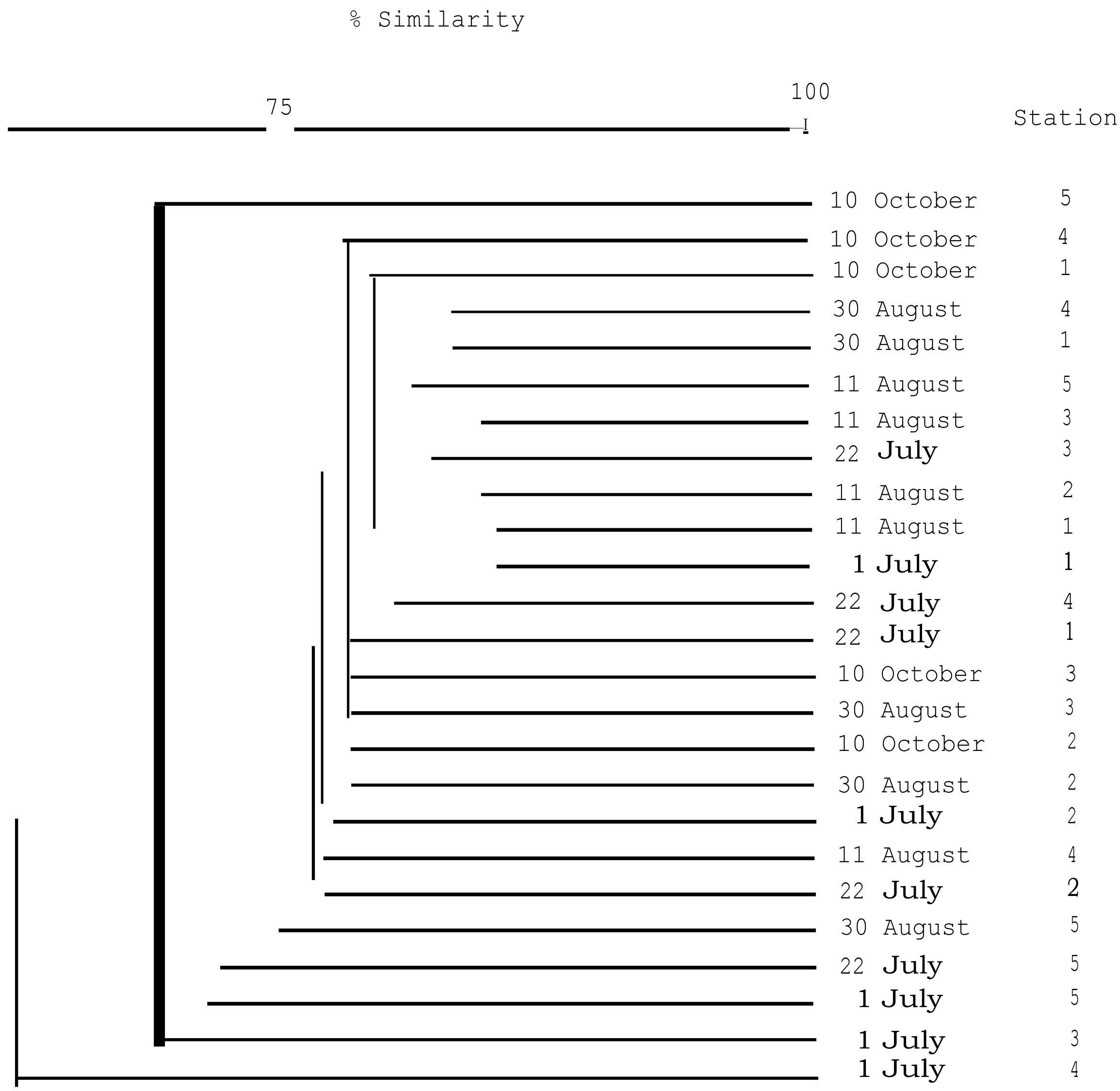
Appendix Table 6. Standardized canonical discriminant coefficients for the four derived functions, and percent variation explained for macro invertebrate genera from benthic collections, included in the stepwise discriminant analysis.

Invertebrate genera	Standardized canonical discriminant coefficients[a]				Cumulative percent variation explained
	Function 1	Function 2	Function 3	Function 4	
Physa	-0.73	0.45	0.49	0.05	25.4
Simulium	0.49	0.22	0.55	0.52	35.7
Gammarus	0.55	0.48	0.09	-0.13	44.4
Tabanus	-0.02	-0.77	0.52	0.26	52.0
Procladius	-0.12	0.23	-0.69	0.74	56.8

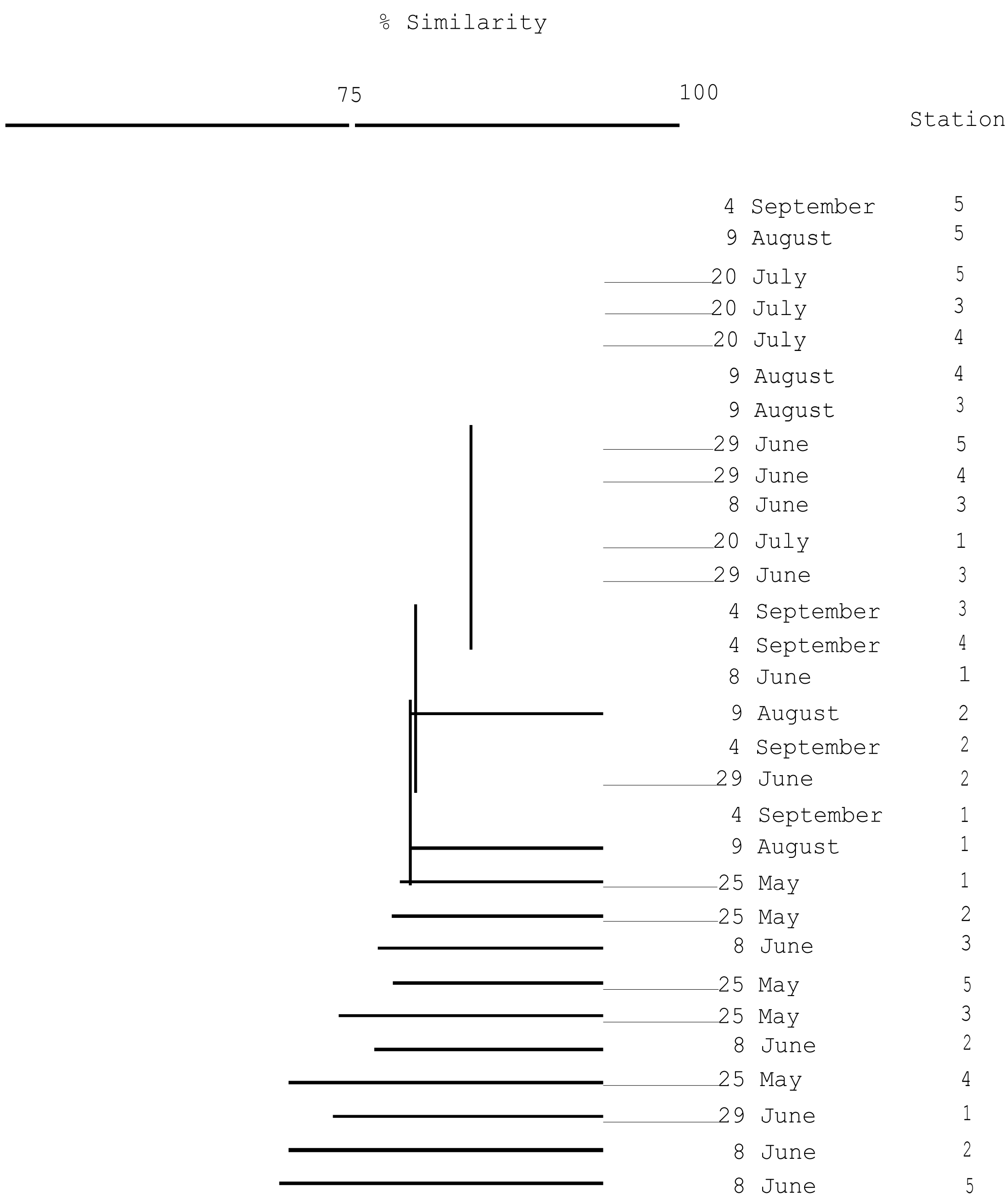
[a]Absolute value indicates relative importance in the function derived.



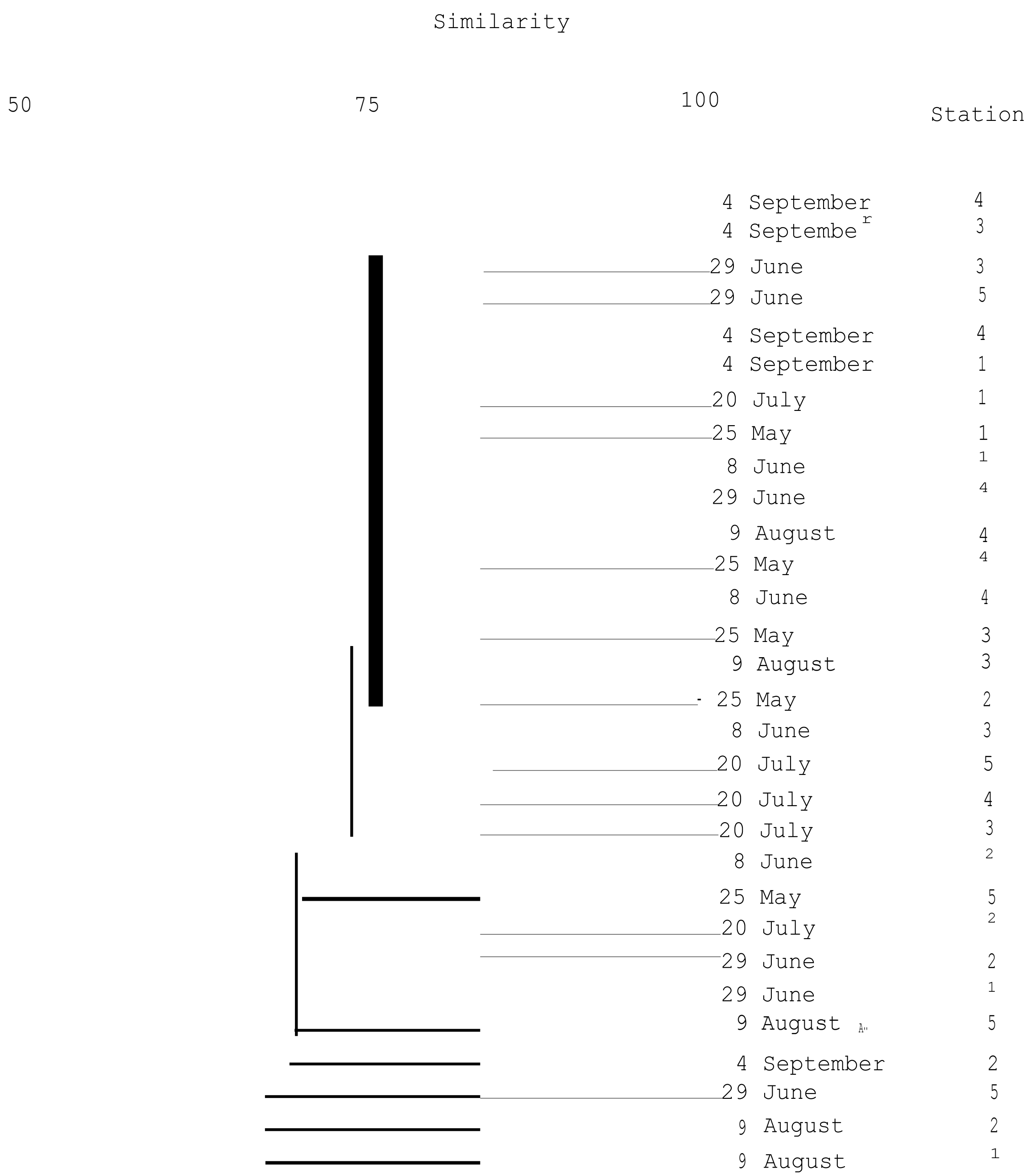
Appendix Fig. 1. Percent similarity dendogram for artificial substrate collections by station-date, based on presence or absence data, for invertebrate genera collected from Slate Creek, 1981.



Appendix Fig. 2. Percent similarity dendogram for benthic collections by station-date, based on presence or absence data, for invertebrate genera collected from Slate Creek, 1981.



Appendix Fig. 3. Percent similarity dendogram for artificial substrate collections by station-date, based on presence or absence data, for invertebrate genera collected from Slate Creek,1982.



Appendix Fig. 4. Percent similarity dendogram for benthic collection by station-date, based on presence or absence data, for invertebrate genera collected from Slate Creek, 1982.