

**Development of Biomonitoring Protocols for
Large Rivers in Idaho**

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SUMMARY

We selected 22 sites, from an initial 31 sites examined, on large rivers (approximately 6th or greater) throughout Idaho to develop bioassessment protocols for large rivers. The sites covered the range of environmental conditions found in Idaho rivers and geographically covered the entire state. At each site a suite of habitat variables were measured, macroinvertebrates were collected, and the fish community censused. In general, the degraded sites we examined displayed greater ion concentrations (i.e., conductivity and alkalinity) than did the reference sites. Of the habitat variables we examined, water chemistry was the most important in distinguishing the degraded sites from the reference sites.

Based on the macroinvertebrate community in each site, we developed an index for assessing the ecological condition of large rivers in Idaho. We have termed this index, the Idaho River Index (IRI) and it appears to have excellent potential for distinguishing degraded rivers from those in good to excellent condition. Five macroinvertebrate metrics were included in the IRI: taxa richness, EPT richness, % dominance, % Elmidae (riffle beetles), and % predators. Furthermore, the IRI was able to identify two sites as being in an intermediate condition; these two sites are worthy of more detailed monitoring in order to fully assess their ecological health. Future research will involve refining the IRI by examining additional sites, this may result in more metrics being included in the index.

The fish metrics provided only minimal additional insight

beyond what the macroinvertebrates provided. Several problems exist with the use of fish metrics, primarily the influence of stocking programs, the occurrence of threatened and endangered species (which hinders sample collection), and the natural lack of species diversity in the western United States. However, we found that carp (*Cyprinus carpio*) occurred only in degraded sites and may be indicators of impaired habitat conditions.

We also examined nutrient limitation and composition of the diatom community in nine of the sites (three degraded sites and six reference sites). The degraded sites were either phosphorus limited or not limited by nutrient concentrations. The reference sites tended to be either nitrogen limited or co-limited by nitrogen and phosphorus. The relative abundance of individual diatom species varied among the nine sites, with only a few species displaying a distinct pattern between degraded and reference sites. However, the potential for developing an index of river health based on diatoms will be examined further.

INTRODUCTION

Since the 1960's, aquatic biota, particularly macroinvertebrates, have become an important component of water quality monitoring programs throughout North America (see papers in Rosenberg and Resh 1993). In Idaho, biomonitoring protocols in which macroinvertebrate metrics play a central role have been developed for small, wadeable streams (Robinson and Minshall 1995). Streams of this size have been the focus of the majority of the water quality monitoring efforts in Idaho and throughout the United States. Not surprisingly then, the rapid assessment protocols currently in use have been developed for, and tested in, wadeable streams (Resh and Jackson 1993). The need for assessment and monitoring of larger river systems (6th order and greater) has long been apparent but has begun to receive attention only recently. Refinement of the current protocols (and/or the creation of new ones) for use in large river systems is the next obvious step in the evolution of monitoring programs based on biotic as well as abiotic components. In Idaho, the need for such protocols is especially timely because population growth and landuse have caused a number of rivers, once regarded to have high recreational qualities, to approach critical limits of water quality and self-sustainability. In some cases, such as the Middle Snake and Lower Coeur d'Alene Rivers, water quality limits already have been severely exceeded.

The development of assessment and monitoring procedures for large rivers is likely to be a more complex process than simply "scaling-up" the protocols used in wadeable streams. For example, adequate statistical replication of 'control' and

'treatment' streams is relatively easy when examining streams of 1-4th order, of which several may exist in a given basin. Replication of large rivers, however, often is not possible simply because they do not occur in as great a density as do smaller streams. Furthermore, unlike small streams, comparisons among large rivers are more likely to be confounded by differences in geology, geography, and climate, due to the increased distances between and along these large river systems. Sampling methodologies suitable for wadeable streams are not likely to be adequate for large rivers which are less easily sampled and require different techniques and equipment (many of which remain to be developed and/or adequately tested). Sampling large rivers is more intensive in terms of both equipment and personnel and accessibility may be limited due to the need for boat launching. Furthermore, working on large river systems presents greater inherent dangers for field personnel.

The composition of biotic communities changes with progression from headwater areas to a large river (sensu Vannote et al. 1980). Also, large rivers are not linked as tightly to the immediate terrestrial environment as are small streams. Hence, the metrics used to describe the ecological condition of a biotic community in a small stream may not indicate the same condition in a large river. For example, in forested, headwater streams an abundance of collector-filterer macroinvertebrates may be indicative of excessive organic sediment inputs to the stream, but in a large river collector-filterers may naturally occur in high densities. Thus, while the same metrics are potentially useful in both systems, the interpretation of them may differ.

The use of macroinvertebrates and fish as indicators of ecological conditions within lotic systems has become widespread and relatively standard in monitoring and assessment programs. However, these are measures of community structure only, they do not measure the ecological functions of a river system (although functional feeding group metrics occasionally have been used to infer whole-system function). Attempts to incorporate functional measures generally are hindered by increased time and expense and difficulties of interpretation, relative to structural measures. Nevertheless, functional measures need to be incorporated into large-scale bioassessments in order to determine completely and unequivocally the condition of a lotic ecosystem (Minshall 1993, 1996). We tested the potential of using functional measures in bioassessment by including estimates of nutrient limitation and community metabolism, in conjunction with our structural measures, in the bioassessment of some of the rivers we examined.

Unfortunately, our measures of ecosystem metabolism, based on measurements of dissolved oxygen, failed to detect a biological signal. The reaeration of the river water by physical processes (e.g., turbulent flow) maintained the dissolved oxygen concentration at saturation. Thus, the changes in dissolved oxygen that we measured were simply a function of changes in temperature, rather biological activity among the primary producers. Methods for accurately measuring reaeration rates in large rivers containing areas of turbulent flow need to be developed before ecosystem metabolism can be determined in these systems using open-system methods. However, closed-system procedures (i.e., recirculating metabolism chambers) do not require reaeration rates and offer an alternative means of

determining ecosystem metabolism. We plan to continue work towards refining functional measures for use in bioassessment; the results will be reported at a later date.

Our objectives in this study were to develop, test, and refine methods for use in the assessment and monitoring of the ecological condition of Idaho's large rivers. Here, we include sampling design, sampling methods and equipment, and data analysis and interpretation. The information collected during this project also provides baseline data against which the effects of future disturbances (natural or anthropogenic) can be examined. In developing these methods we have drawn upon many sources of information, including numerous published sources, recommendations from professional resource managers, and our previous work on assessment protocols (Robinson and Minshall 1995) and large river systems in Idaho (Minshall et al. 1992, Snyder and Minshall 1994, Royer et al. 1995, Thomas et al. 1995). It must be emphasized that the recommendations presented here are intended to be initial guidelines, subject to adjustment as various situations dictate, rather than as rigid formulae to be followed in all instances. Different river systems and detection of different anthropogenic impacts may require modification of the procedures we present in this report.

PART I - SITE SELECTION

In general, the selection of study sites was based primarily on river size and type. We defined large rivers as those greater than approximately 6th order (Strahler 1957). The selection process began by examining 31 large river sites for potential use

in the study (see Appendix A). Twenty-two study sites were then chosen from the initial list of 31. Selection criteria included size of the river, state and federal water quality classifications, and our initial field reconnaissance (including measures of alkalinity, conductivity, and habitat quality; see Robinson and Minshall 1995). Insufficient size was the most common reason for rejecting a given site; these sites will be reconsidered for our future analysis of medium-sized rivers (approximately 4-6th order).

The selected sites covered the range of environmental conditions found in Idaho rivers. This resulted in a continuum of conditions from pristine sites to sites that were obviously degraded. The 22 sites represented nine distinct river systems: the Bear, Snake, Owyhee, Boise, Payette, Salmon, Clearwater, St. Joe, and Coeur d'Alene (Fig. 1). For purposes of metric development, five of the sites were classified as 'degraded' and the remaining 17 sites classified as 'reference' (Fig. 2).

For clarity of presentation in this report, each site was assigned a Site Number and will be referred to throughout this report by that number (Table 1). The Snake and Salmon Rivers each were sampled at several locations to investigate potential longitudinal changes that may occur over these long river systems. The exact sampling location on each river was determined by proximity to a USGS gaging station and accessibility, particularly boat launching facilities. Three sites (Sites 5, 11, and 13) were sampled in 1994 as part of the development and testing of field methods; the other sites were sampled in 1995.

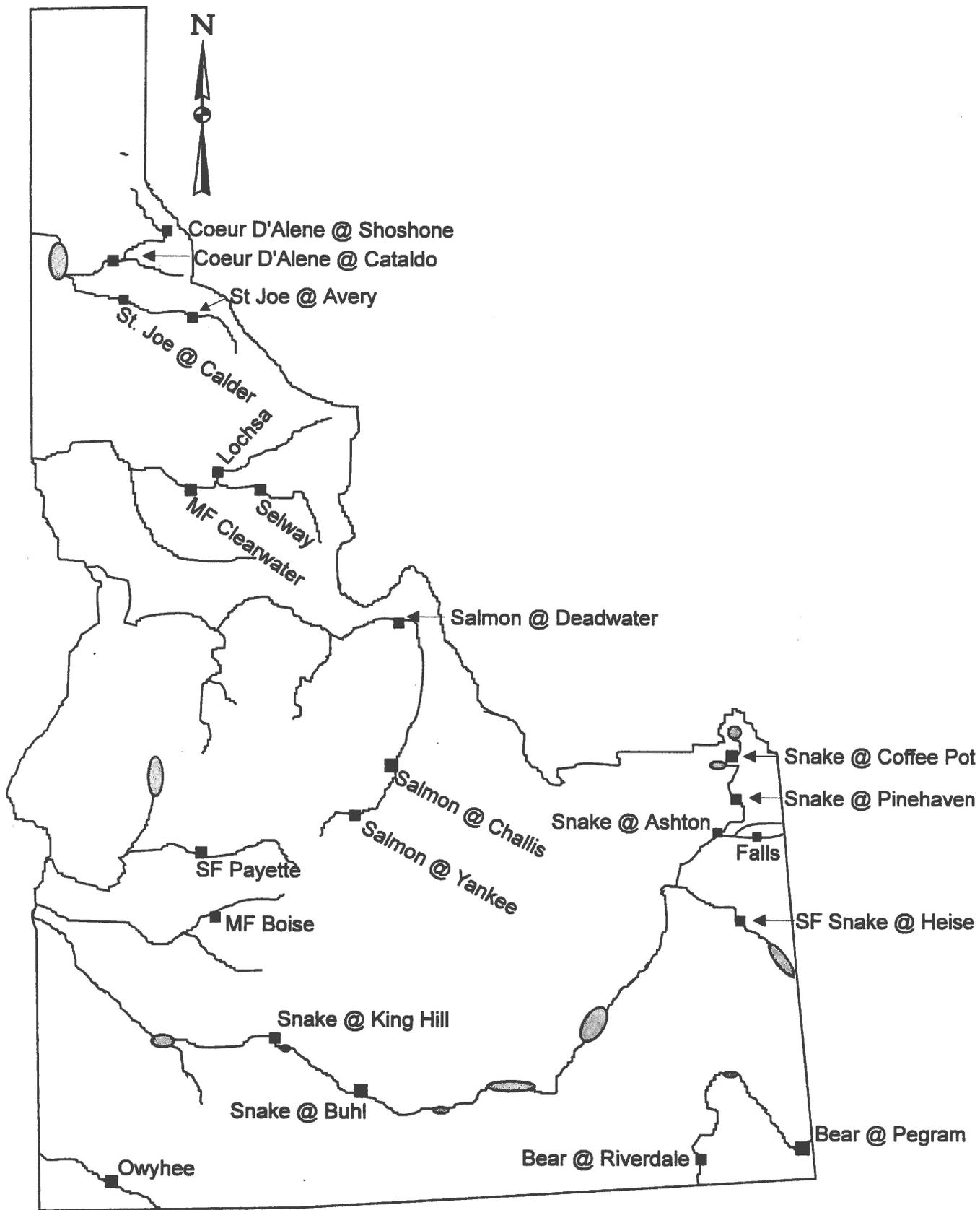


Fig. 1. Large river study sites used to develop the biomonitoring protocols

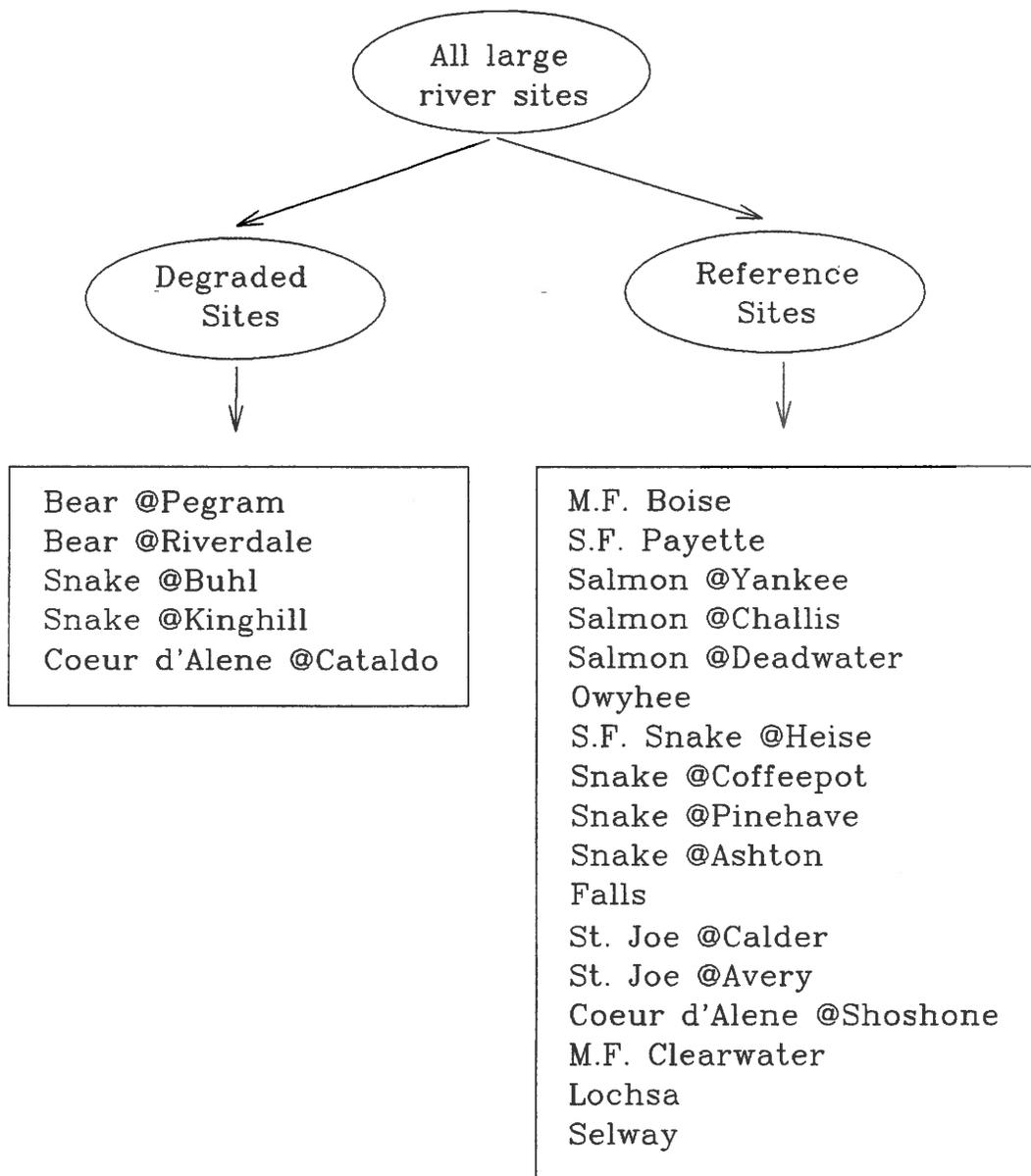


Fig. 2. Breakdown of the 22 sites used in the study as either degraded or reference.

Table 1. Site number, latitude and longitude for each of the 22 large river study sites.

Site No.	River/Location	Latitude	Longitude
D 1	Bear @Peagram	42°07'54"N	111°08'24"W
D 2	Bear @Riverdale	42°10'03"N	111°51'10"W
R 3	Falls	44°03'33"N	111°11'35"W
R 4	Snake @Coffee Pot	44°29'30"N	111°22'01"W
R 5	Snake @Pinehaven	44°17'29"N	111°27'22"W
R 6	Snake @Ashton	44°03'24"N	111°31'59"W
R 7	S.F. Snake @Heise	43°35'50"N	111°28'59"W
D 8	Snake @Buhl	42°39'38"N	114°39'04"W
D 9	Snake @King Hill	42°59'51"N	115°13'05"W
R 10	Owyhee	42°10'10"N	116°30'07"W
R 11	Salmon @Yankee	44°15'56"N	114°43'48"W
R 12	Salmon @Deadwater	45°24'26"N	114°01'26"W
R 13	Salmon @Challis	44°21'06"N	114°16'16"W
R 14	S.F. Payette	44°04'07"N	115°56'31"W
R 15	M.F. Boise	43°39'42"N	115°43'34"W
R 16	Selway	46°05'06"N	115°29'13"W
R 17	Lochsa	46°09'52"N	115°35'24"W
R 18	M.F. Clearwater	46°08'18"N	115°39'47"W
R 19	Coeur d'Alene @Shoshone	47°42'05"N	115°58'05"W
D 20	Coeur d'Alene @Cataldo	47°32'50"N	116°21'50"W
R 21	St. Joe @Calder	47°16'42"N	116°14'38"W
R 22	St. Joe @Avery	47°13'54"N	115°43'37"W

PART II - BENTHIC HABITAT, MACROINVERTEBRATES, AND FISH

This section describes the methods and results from the sampling of benthic habitat, macroinvertebrates, and fish in the study rivers. Habitat variables which were able to distinguish between the degraded and reference sites are presented. An index, based on macroinvertebrate metrics, for assessing the quality of large rivers in Idaho is presented and its usefulness discussed. Problems associated with using fish metrics in bioassessments also are discussed.

METHODS

At each site, six cross-sectional transects were located equidistant along the study reach. Transect 1 was situated in the proximity of a permanent structure located alongside the river for ease in relocating the site for future monitoring. The remaining transects were located downstream (covering a total distance of approximately 500 m). The location of each study reach was recorded on a USGS 7.5' map with longitude and latitude coordinates noted to aid in relocating the sites.

Chemical measures were recorded at transect 1 using water collected from mid-depth in the thalweg or on-site spot samples. Measures recorded included water temperature, dissolved oxygen, pH, specific conductance, alkalinity, total hardness, sulfate, turbidity, ortho-phosphorus (SRP), and nitrate nitrogen (NO_3) (APHA 1992). Secchi disk depth also was recorded at this location. Water samples were collected for estimates of coliform and fecal strep bacteria to determine the prevalence of livestock or human derived organic waste.

General physical characteristics recorded on site included aspect and discharge at one of the transects (unless a USGS gaging station was located near the study reach). At each transect, channel width, bankfull width and depth, riparian width, percent canopy cover, bank characteristics, bank and canopy angles, bank vegetation, riparian vegetation, bank materials, and bank characteristics were recorded. Riparian vegetation density was quantified using the Point Quarter Method for both trees and shrubs at each transect on alternate sides of the river (Mueller-Dombois and Ellenberg 1974). Water depth, current velocity, substrate size, and embeddedness were measured and presence of macrophytes recorded at a minimum of 20 equidistant locations along each transect; velocity was measured only at transects 1, 3, and 6. A quantitative periphyton sample (Robinson and Minshall 1986) was collected from the right, left and center at each transect (n=9), and analyzed for chlorophyll a and AFDM (APHA 1992, Robinson and Minshall 1986).

Benthic invertebrates were collected along the three most physically different transects present at a site (e.g. pool, riffle, run) selected during the habitat evaluation procedure or at transects 1, 3 and 6 if uniform habitat conditions appeared to be present. Nine quantitative samples were collected from each study reach. At each selected transect, a sample was collected from the center of the channel, and midway between the margin and center on both sides of the river. The nine quantitative samples were then composited into a single sample to facilitate processing. Because of the diverse habitats present among rivers, no single sampling technique or device could be used. Emphasis was placed on the NAWQA developed "slack sampler"

(essentially a modified Surber net), petite ponar, D-frame dip net, and when necessary, a diver-assisted dome sampler (Cuffney et al. 1993a). Mesh size was 250 μ m for all benthic sampling equipment. All results were standardized for the size and type of sampler used, and expressed per unit area.

Along with the quantitative samples, a qualitative sample was collected using a D-framed dip net. The qualitative sample was designed to collect organisms from the variety of habitats found in the study reach, such as snags, bars, macrophyte beds, areas that differ in substrate composition and flow characteristics (e.g., island margins). The qualitative sample was used in conjunction with the quantitative samples to calculate a total taxonomic richness for each site. All invertebrate samples were preserved on-site with 5% Formalin and returned to the laboratory.

In the laboratory, all large and/or rare taxa were removed from each sample, identified, and counted. Following removal of these organisms, each sample was split into eighths using an automated subsampler and all macroinvertebrates removed from one or more sub-samples until a minimum of 300 organisms was reached. Data from these two steps were combined to determine the species richness for a site. Macroinvertebrates were identified, enumerated, and dried for biomass determinations. Benthic organic matter was determined from each processed sub-sample by drying the processed material at 60°C, weighing, ashing at 550°C, rehydrating the material, drying at 60°C, and then reweighing. Values are adjusted to correct for the sub-sampling procedure.

Fish were sampled from a rubber raft (5.4 m in length) equipped with a boom-mounted anode and a trailing cathode. A 3.5 or 5.0 kw, 60 Hz portable alternator and a variable voltage pulsator (Coffelt model VVP-2C) were used to generate pulsed direct current. Water conductivity was measured at each site so that the circuit voltage could be adjusted to maintain a current density of 0.1 to 1.0 volts/cm. The raft operator controlled electrical current flow with a hand/foot-activated pressure switch, while two technicians netted fish from the bow of the boat. When conditions precluded use of the raft (e.g., numerous exposed rocks), an aluminum Jon-boat (4.5 m in length) was substituted for the rubber raft and hand-towed through the reach during electrofishing. Habitats inaccessible by boat, such as shallow riffle areas inhabited by sculpin or backwater areas, were sampled using a backpack electrofishing unit (Coffelt model BP-4).

One pass was made along one channel margin and through all backwaters found in the study reach. All fish retrieved were held in a 120-quart insulated container until processed. The container was filled to a depth of 25 cm with river water, which was recirculated using a 12-volt pump. Fish were identified to species, measured (total length), weighed (mg), and noted for anomalies (parasites, injuries, or deformities). All fish captured were returned to the stream upon completion of sampling, except that a few representatives of each species (unless threatened or endangered) were preserved in 5% Formalin and deposited in the Orma J. Smith Museum of Natural History, Albertson College of Idaho, Caldwell.

The habitat data was analyzed with principal component analysis (PCA) to determine the variables that distinguished the various sites. In the past, PCA has also been used for metric selection (e.g., Robinson and Minshall 1995). However, the use of PCA in biomonitoring has been criticized for being too complicated and difficult to interpret (Fore et al. 1996). Indeed, Barbour et al. (1996) developed an index for river health in Florida using box plots rather than PCA or other multivariate procedures. In this report we have adopted the method of Barbour et al. (1996) for analysis of the macroinvertebrate metrics. We have developed an index of river integrity for large rivers in Idaho using box plots, rather than multivariate statistics.

RESULTS

The PCA of habitat variables identified water chemistry and river size as variables that distinguished the sites (Fig. 3). The degraded sites (Sites 1, 2, 8, 9, and 20) generally had greater values of alkalinity, hardness, and conductivity than did the reference sites; the exception was Site 20 (Coeur d'Alene @Cataldo). Conductance in the disturbed sites (other than Site 20) ranged from approx. 490-560 $\mu\text{S}/\text{cm}$ @20°C. A similar pattern was observed in alkalinity and hardness. Site 7 (S.F. of the Snake @Heise), although classified as a reference site, tended to be more similar to the degraded sites than to the reference sites, at least in terms of the water chemistry parameters that we measured. The degraded site in northern Idaho (Site 20) did not differ in water chemistry from reference sites in that area. Mean river depth and water velocity tended to vary between the various sites, as indicated by the PCA, but did not distinguish the degraded sites from the reference sites (Fig. 3). Although

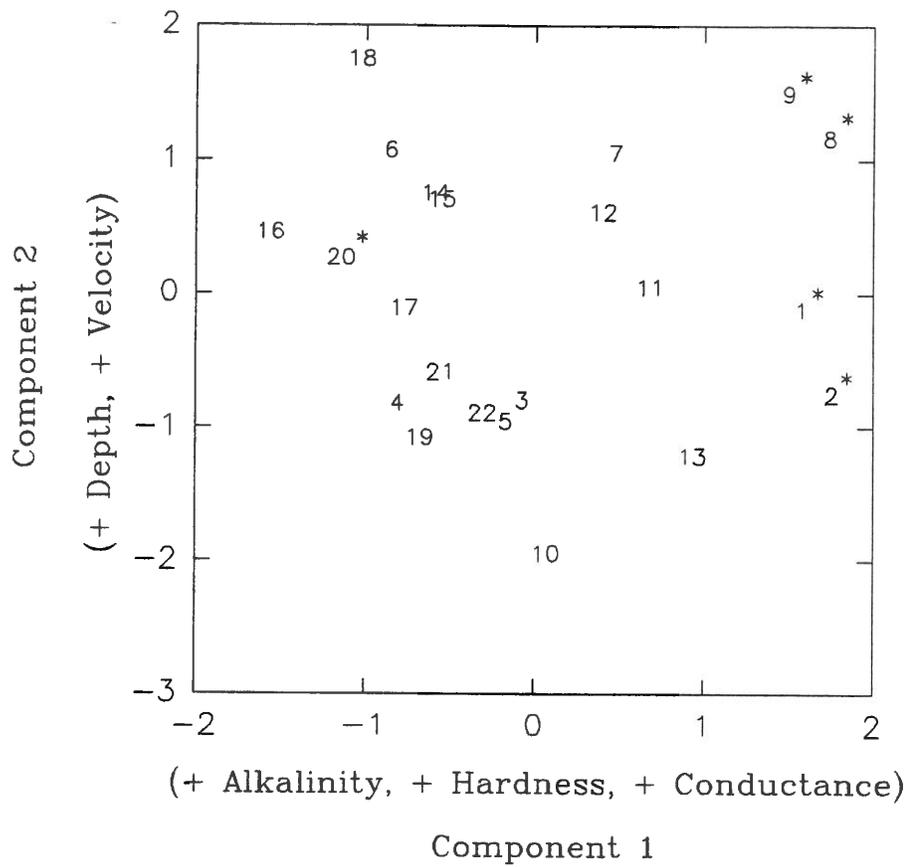


Fig. 3. PCA of the habitat measures from the study sites. The First two axes explained 53% of the variance. Degraded sites are designated with an asterisk.

not identified by the PCA as a principal factor, % macrophyte cover also is presented in Figure 4. An abundance of macrophytes may be caused by anthropogenic activities such as irrigated agriculture (e.g. Mid-Snake), or may occur naturally (e.g. Henry's Fork of the Snake). The usefulness of variables with a high range of values among reference sites, such as % macrophyte cover, is discussed below. The data for all measured habitat variables is presented in Appendix B.

The taxonomic richness of invertebrates increased in all sites when calculated with both quantitative and qualitative samples (total richness), rather than with the quantitative samples alone (Fig. 5). This is likely due to the fact that the qualitative samples were collected from all habitat types, as opposed to the quantitative samples which were collected only from the predominant habitat types (see Methods). In general, the qualitative samples contributed a greater increase in richness in the reference sites than in the degraded sites. This suggests that the degraded sites contained less habitat diversity, which resulted in less invertebrate diversity. The difference (or % increase) in diversity between the two types of sampling may be a useful metric in identifying rivers with reduced habitat diversity. For example, in this study the average increase in richness for the degraded sites was 27%, while in the reference sites it was 43% (Fig. 5). This metric will be examined further in our analysis of medium-sized rivers.

Box plots for each invertebrate metric are presented in Figure 6. Each metric was scored based on the amount of separation observed between the degraded and reference sites.

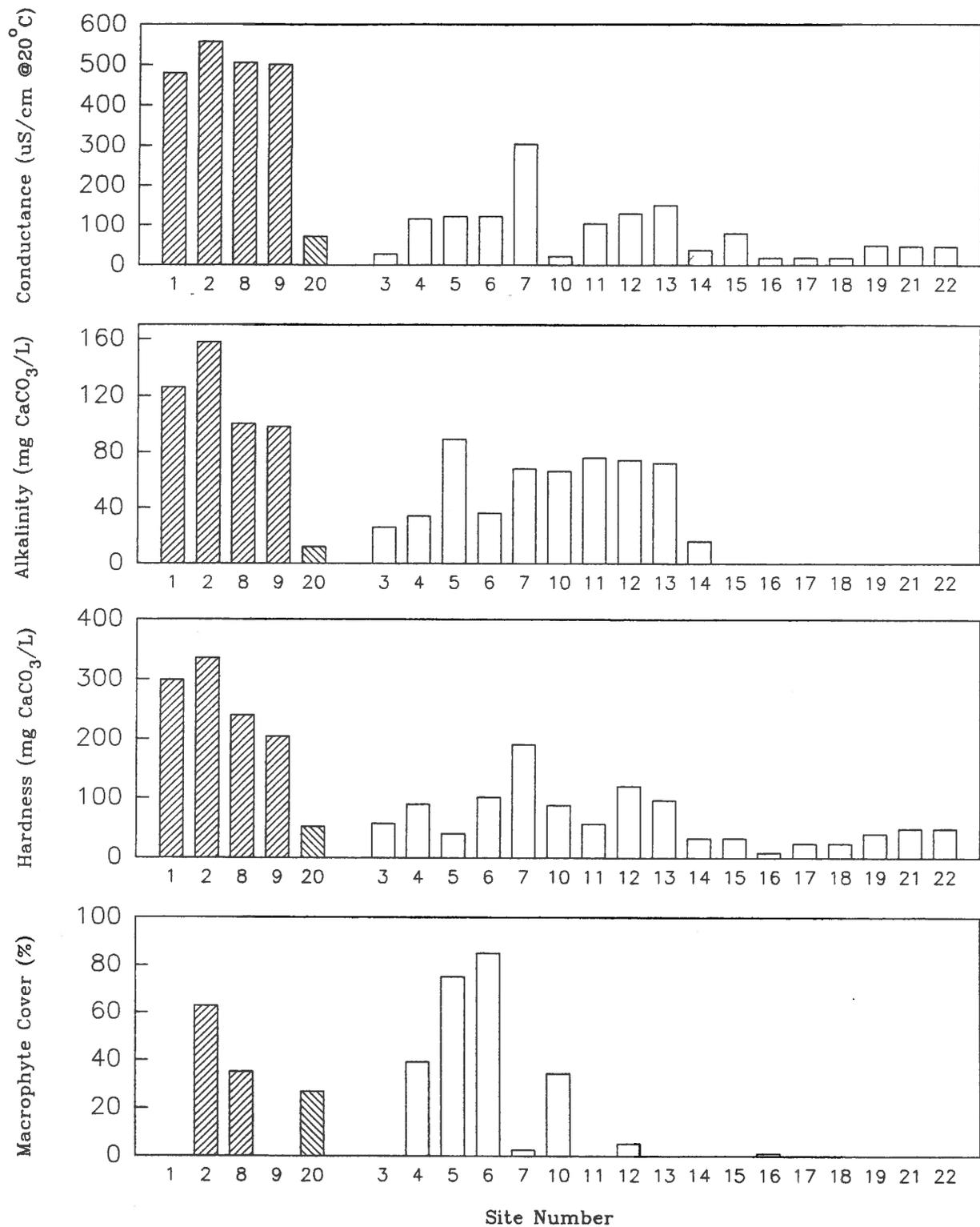


Fig. 4. Habitat variables that the PCA identified as distinguishing the degraded (▨) and reference (□) sites. For each graph a missing bar indicates a value of zero.

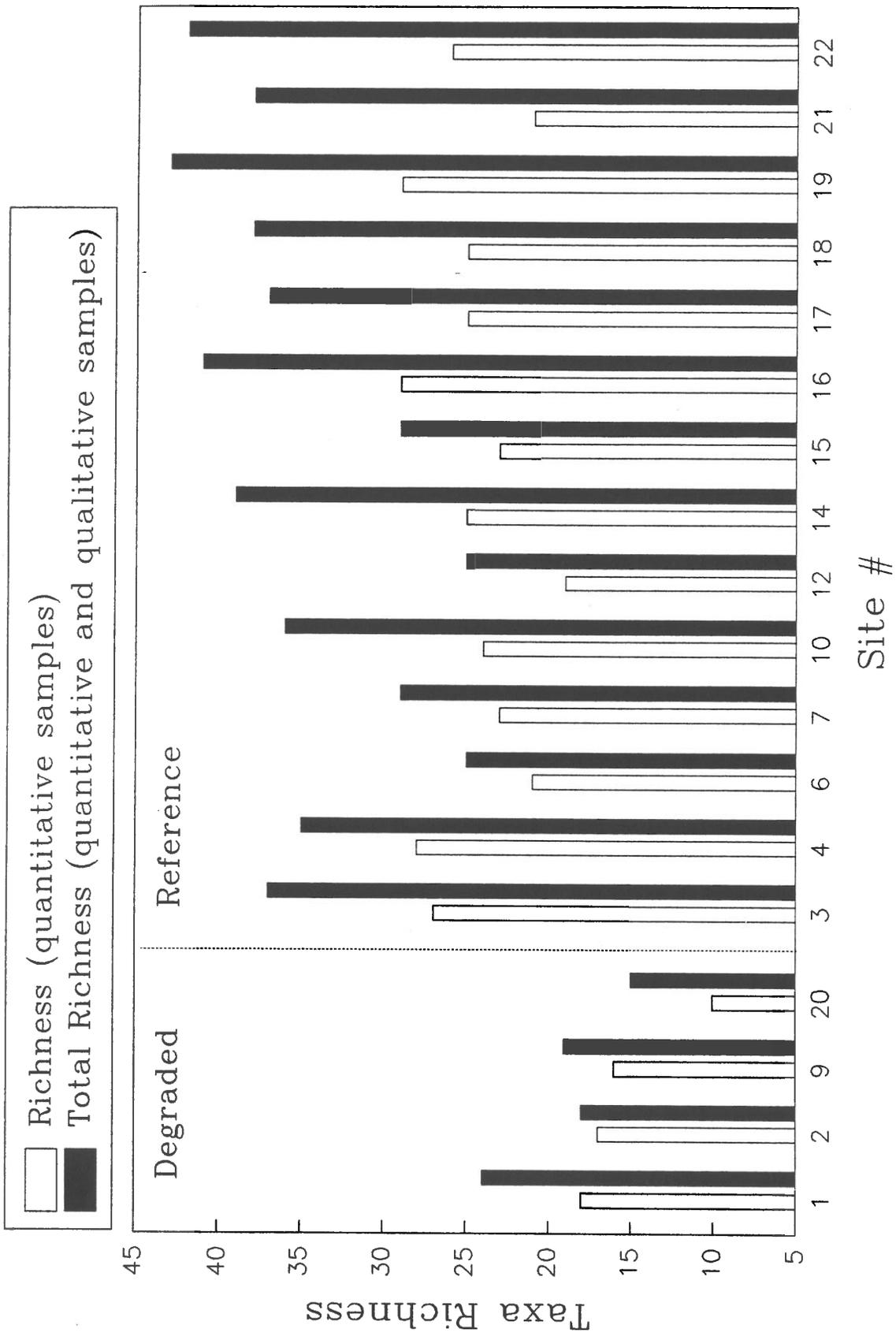


Fig. 5. Taxa richness from the quantitative samples and from the quantitative and qualitative samples combined (total richness).

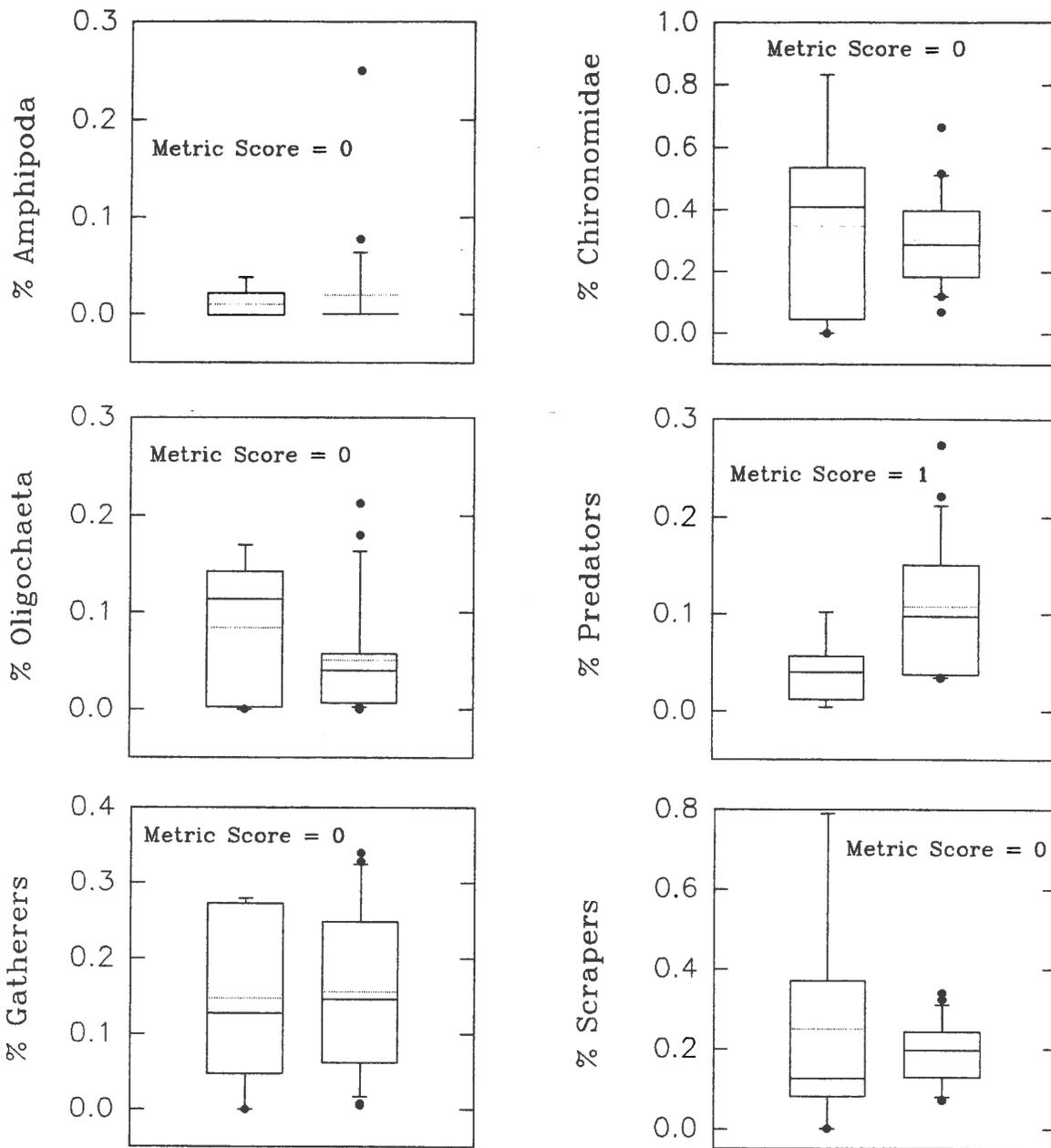


Fig. 6. Box plots of the various invertebrate metrics measured at of the degraded and reference sites. Dashed horizontal line is the mean value. Solid horizontal lines are the interquartiles. Error bars are the 10th and 90th percentiles. Dots are values that were outside the 10th and 90th percentiles. The metric score represents the discreminatory power of the metric, see text for further explanation.

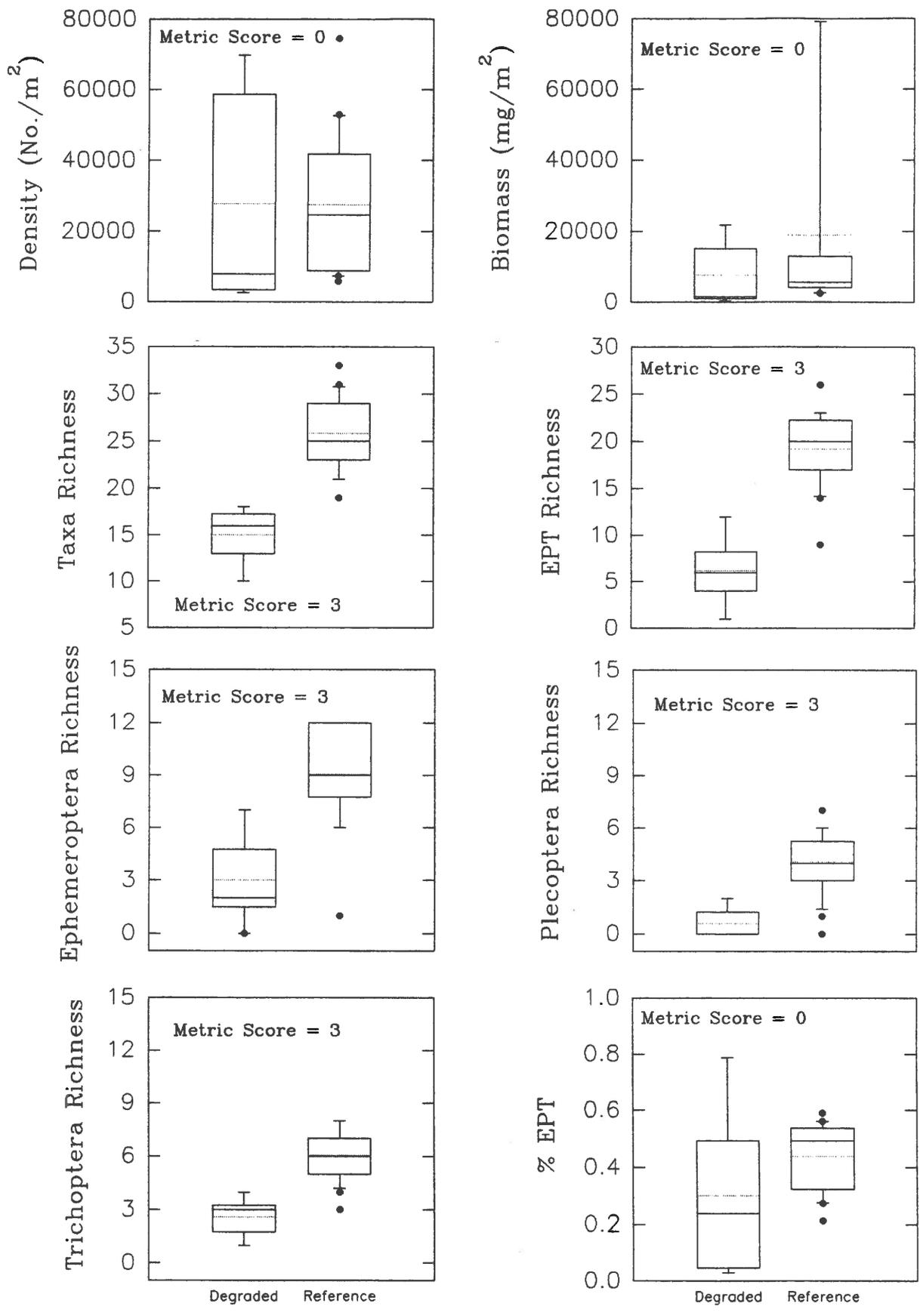


Fig. 6. Continued.

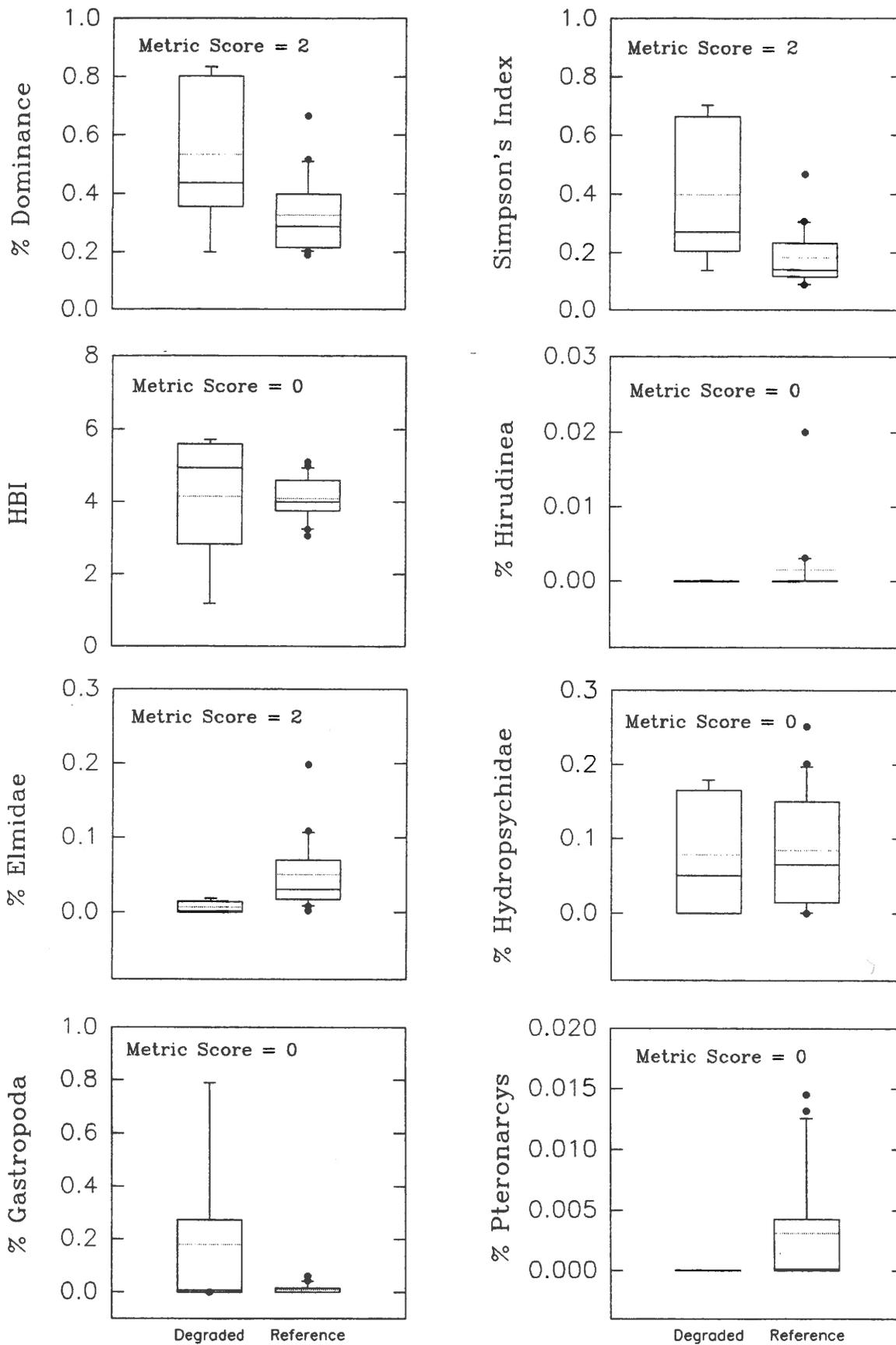


Fig. 6. Continued.

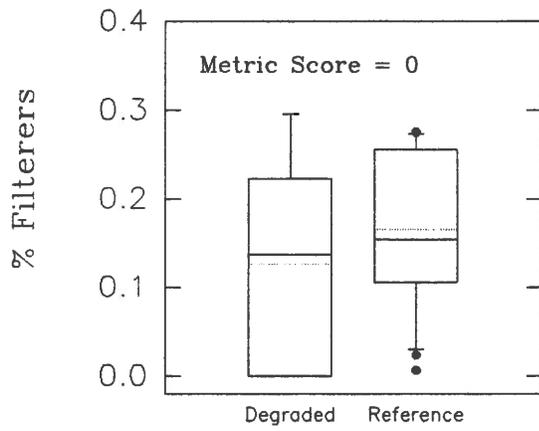
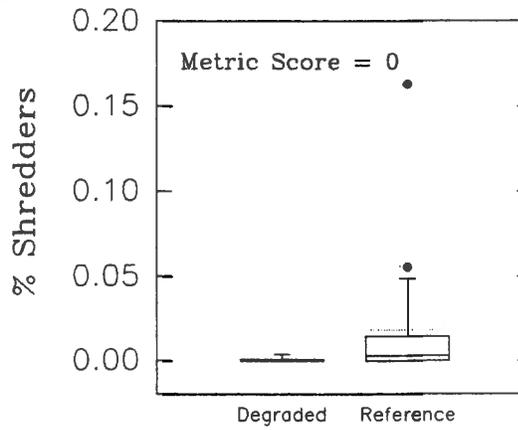
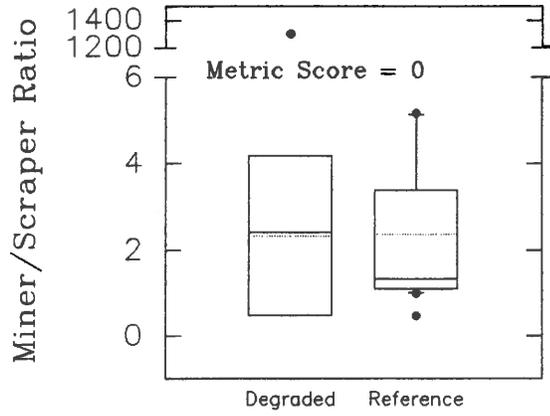
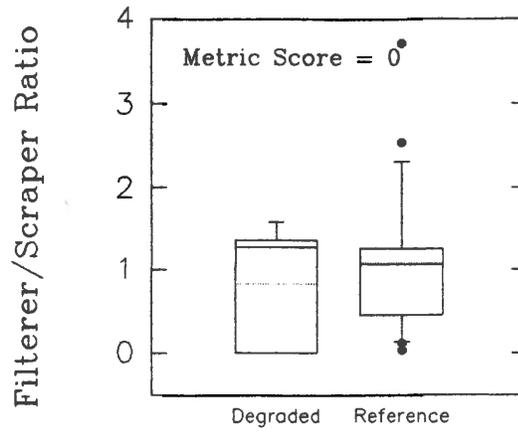
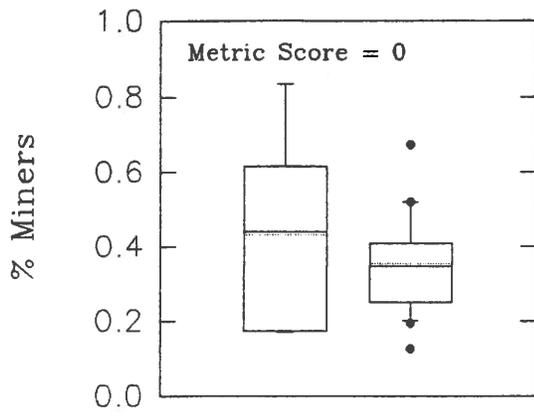


Fig. 6. Continued.

Metrics that contained no overlap in the interquartile ranges received a three, those with some overlap but with each mean outside the interquartile range received a two, those with moderate overlap and at least one of the means outside the interquartile range received a one, those with extreme overlap received a zero (Barbour et al. 1996). The score for each metric is included in Figure 6. Only those metrics that scored a one or greater were considered for inclusion in the final index.

A Spearman correlation analysis was then performed among all metrics that scored a one or greater. If two metrics were highly correlated (Spearman Correlation Coefficient of 0.750 or greater) one of the metrics was discarded, as the information was redundant (Barbour et al. 1996). For example, EPT richness and Ephemeroptera richness were strongly correlated (coefficient = 0.856) and EPT richness was selected over Ephemeroptera richness. In the end, five metrics were selected for use in developing an index of river integrity for large rivers in Idaho: taxa richness, EPT richness, % dominance, % Elmidae, and % predators.

Using the reference sites only, the minimum, maximum, and 25th, 50th, and 75th percentiles were calculated for the five metrics (Table 2). A scoring system was then designed whereby a site would be scored either a one, three, or five for each metric, based on how the site compared to the reference sites. For example, a site with a taxa richness value less than the minimum value observed in the reference sites would receive a score of one for taxa richness. If taxa richness was between the minimum and the 25th percentile, the site would score a three, if it was greater than the 25th percentile it would score a five. The exception to

Table 2. Descriptive statistics and scoring range for the metrics included in the Idaho River Index.

	Statistics					Score		
	Minimum	25%ile	50%ile	75%ile	Maximum	1	3	5
Taxa Richness	19.0	23.0	25.0	29.0	33.0	<19	19-22	>22
EPT Richness	9.0	17.0	20.0	22.5	26.0	<9	9-17	>17.0
% Dominance	0.190	0.215	0.290	0.430	0.665	>0.665	0.430-0.665	<0.430
% Elmidae	0.002	0.015	0.030	0.075	0.198	<0.002	0.002-0.014	>0.014
% Predators	0.034	0.040	0.100	0.150	0.274	<0.040	>0.040	*

* % Predators was given only two scoring ranges due to limited discriminatory power.

this procedure was for % predators, which had less discriminatory power than did the other metrics. Thus, % predators was given only two scoring possibilities, one or three to compensate for its lack of discriminatory power (Barbour et al. 1996). The scores for each of the five metrics are then summed to determine the overall index score for a site. We have termed this final score the Idaho River Index (IRI). The above process was performed on all sites, including the degraded sites to examine the ability of the IRI to distinguish reference from degraded conditions (Fig. 7).

The results show clearly that the IRI was able to distinguish the degraded sites from the reference sites, even with the low number of degraded sites that we used (n=5). Sites with a IRI score of 13 or less should be considered degraded, those with scores of 16 or greater can be considered in good to excellent condition, and those sites with IRI scores between 13-16 should be considered intermediate. Sites with intermediate IRI scores may be in the process of becoming degraded, and likely are worthy of more detailed monitoring. Two of the sites that we examined fell into this category, Sites 6 and 7 (the Snake River @Ashton, and the S.F. of the Snake @Heise, respectively). Site 7 in particular appears to be approaching a degraded state, at least in terms of the macroinvertebrate community. The values for all invertebrate metrics at each site are presented in Appendix C.

Due to the limited number of fish metrics, PCA was used rather than the box plot method. The PCA of the fish metrics resulted in distinct clustering of the reference sites separate

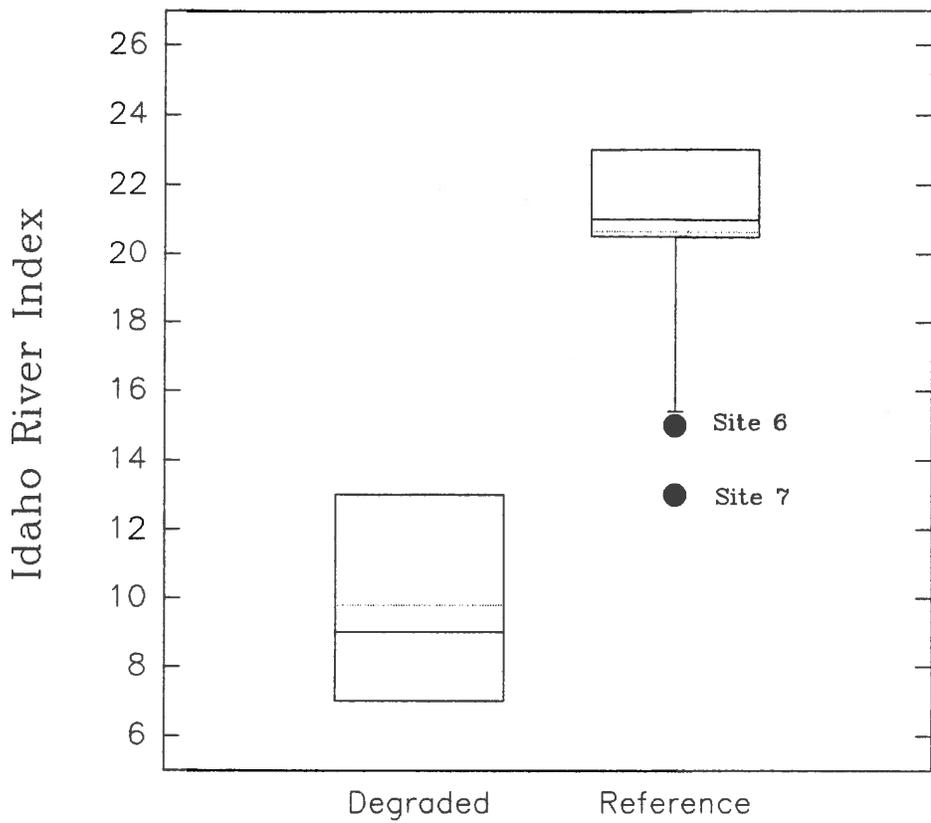


Fig. 7. Box plots of the Idaho River Index scores for the degraded and reference sites. Box plots presented in the same form as in Figure 5.

from the degraded sites (Fig. 8). Site 20, the degraded site in northern Idaho, also separated from the reference sites, but in a different direction than the other degraded sites. The first component of the PCA separated the sites based upon the abundance of intolerant (sensitive) species and insectivorous species. The fish communities in the degraded rivers appeared to be affected by both water quality and condition of the invertebrate community, as indicated by reduced abundance of intolerant species and insectivorous species, respectively (Fig. 9). The second component of the PCA was a result of Cyprinidae abundance, both native and introduced (carp). Here, the degraded sites in southern/central Idaho (Sites 1, 2, 8, and 9) were distinctly different from the degraded site in northern Idaho (Site 20). A predominance of carp (*Cyprinus carpio*) distinguished Sites 1, 2, 8, and 9 from the reference sites (Fig. 9). At these sites, carp biomass ranged from approx. 40 - 60% of the total fish community biomass. The values for all fish metrics at each site are presented in Appendix D.

Site 10 (Owhyee River, reference) contained no insectivorous or intolerant fish species. However, this site does not display other degraded characteristics, at least in terms of water quality or invertebrate community composition. Site 10 is located in the Owhyee Desert of southwest Idaho, and the different fish community found in this site may be a result of thermal conditions, rather than anthropogenic influences. This underscores the need for using appropriate reference sites and understanding the possible reasons for variability in the values of a given variable among reference sites.

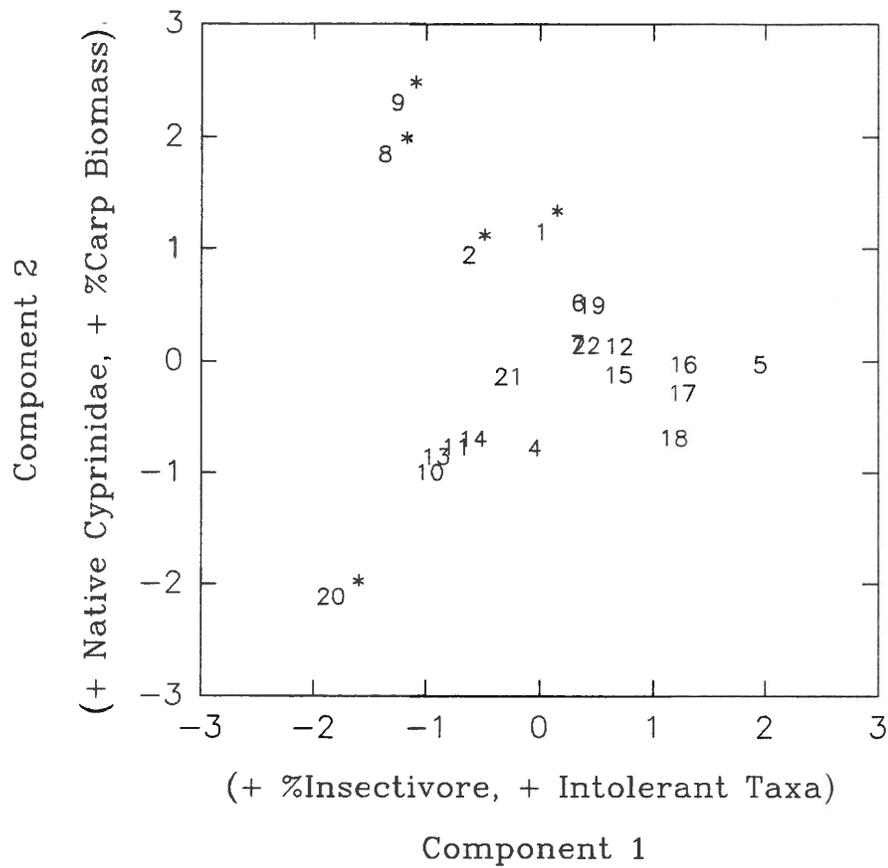


Fig. 8. PCA of the fish metrics from the study sites. The first two axes explained 56% of the variance. Degraded sites are designated with an asterisk.

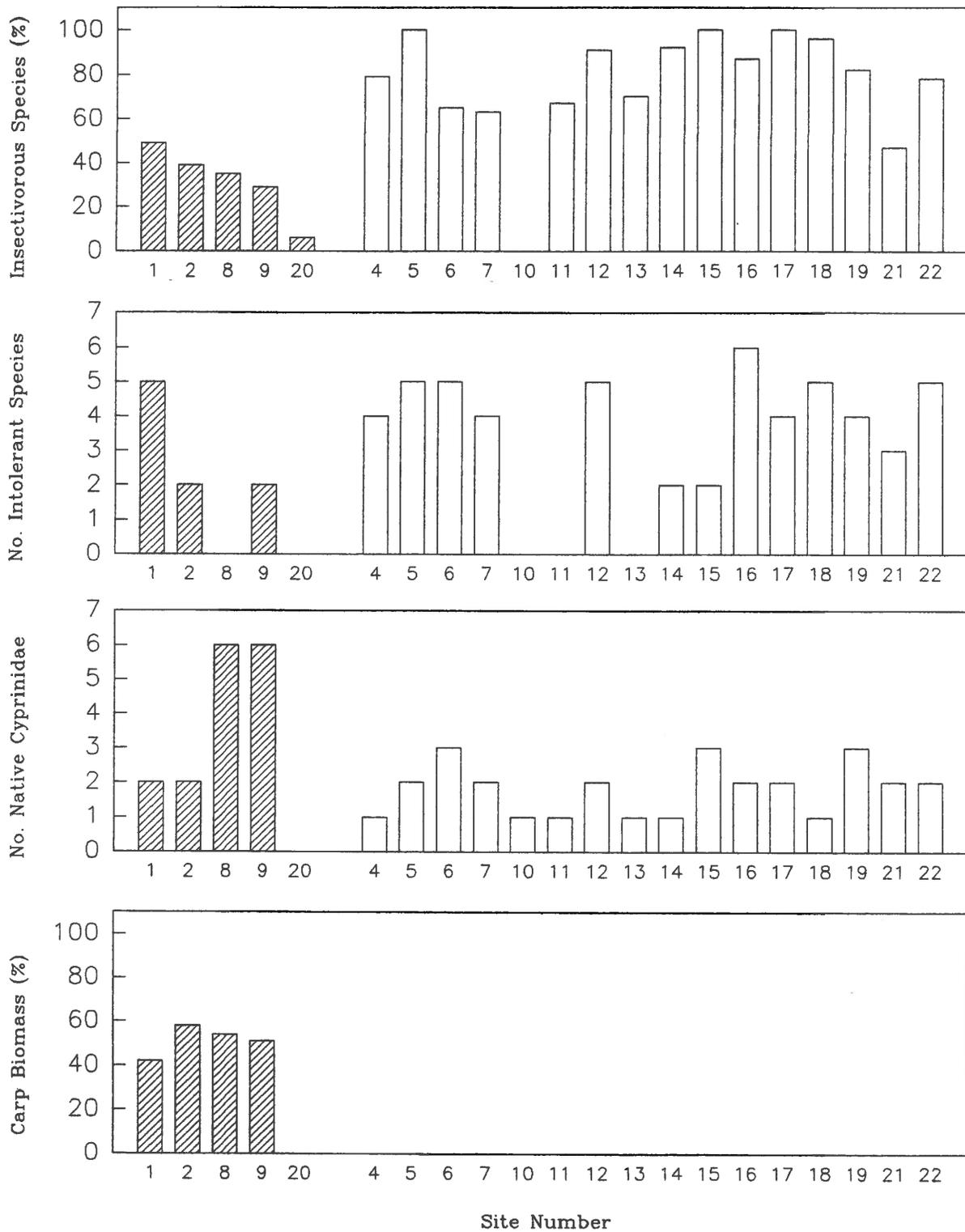


Fig. 9. Fish metrics identified by the PCA as distinguishing the degraded (▨) and reference (□) sites. For each graph a missing bar indicates a value of zero.

DISCUSSION

In general, chemical variables (alkalinity, hardness, conductance) distinguished the sites more readily than did physical variables. This was most apparent in the nutrient-rich rivers of southeast Idaho, where point and non-point sources of pollution and sediment have elevated ion concentrations in the water. In northern Idaho, the degraded site (Site 20) did not differ in terms of water chemistry from the reference sites in that region. This suggests that influences on water chemistry by an anthropogenic activity may depend on geographic region and/or type of activity. Nonetheless, the water chemistry variables used in this study are easily measured in the field and can vary greatly between degraded and reference sites, making them an essential component of assessment protocols.

Large rivers in Idaho range from low gradient systems, such as those in southeast Idaho, to high gradient mountain rivers in northern Idaho. Natural, physical differences (e.g., gradient, confinement, width, depth) among rivers can result in different biotic conditions among the same rivers. For example, Gregg and Rose (1985) demonstrated that aquatic macrophytes influence the composition of stream invertebrate communities. In the present study, the amount of coverage of the river channel by macrophytes ranged from 0 - 85%. Variables with such high variability among sites are not well suited for distinguishing degraded sites from reference sites. However, these variables may aid in interpretation of the invertebrate and fish metrics and, therefore, should be measured during all river assessments, at least qualitatively. In Idaho, such variables include, but are not limited to, water chemistry (Kootsier et al. 1996), substrate

characteristics (Minshall and Minshall 1977, Minshall 1984) and aquatic macrophyte abundance (Gregg and Rose 1982, 1985).

The Idaho River Index (IRI) developed in this study appears to have excellent potential for classifying large river sites as being in either a degraded condition, good condition, or some intermediate condition. The IRI is rather simple, based on only five invertebrate metrics. However, the metrics include not only diversity measures (taxa richness, EPT richness, % dominance), but also the abundance of Elmidae (riffle beetles) and predators (see Table 2). The index developed by Barbour et al. (1996) for use in Florida streams contained eight metrics, but was based on a much larger number of samples (22-89 sites, depending on ecoregion). Thus, as the IRI is refined and tested with additional sites, the number of metrics included in the index may increase.

The IRI included only one functional feeding group (FFG) metric, % predators. In general, FFG metrics did not display a distinct pattern between reference and degraded sites. However, predominance of one FFG in a benthic invertebrate community is likely indicative of reduced diversity following habitat change. For example, increased sedimentation may create a community composed primarily of miners, such as observed at Site 20. At Site 8, the predominance of scrapers is a result of the introduction, and rapid spread, of an exotic snail, *Potamopyrgus antipodarum* (Royer and Minshall, in review). In general, FFG metrics are best used as coarse-scale indicators of community integrity. Rivers exhibiting an extreme abundance of one type of functional group may require more detailed analyses to (1)

confirm that the aberrant values are not artifacts of sampling, and (2) identify potential causal factors for the predominance of one functional group.

All five of the degraded sites separated clearly from the reference sites in terms of the fish metrics. Within the degraded sites, those in southern Idaho were distinctly different from Site 20 in northern Idaho. The abundance of carp was a strong indicator of degraded habitat conditions, at least in southern/central Idaho rivers. In northern Idaho, the abundance of insectivorous and intolerant species was greatly reduced in the degraded site, relative to the reference sites; no site in northern Idaho contained carp. Many factors confound accurate interpretation of the fish data. For example, rivers in different regions contain naturally distinct fish assemblages. Knowledge of historic fish assemblages is needed before concluding that habitat degradation has altered the fish community of a given river. Furthermore, the stocking of large rivers with hatchery populations may result in the presence of intolerant, insectivorous fish at the time of sampling. These fish, however, may not be able to feed or reproduce in the rivers in which they are planted, due to a lack of insects or poor habitat conditions. Thus, spawning success of Salmonid populations may be a more sensitive, and useful, parameter than simple Salmonid presence or density.

Site 7 (S.F. of the Snake @Heise) was initially classified as a reference site. However, the analyses presented here indicate that Site 7 is displaying early signs of habitat degradation. For example, water chemistry at Site 7 was more

similar to Sites 8 and 9 than to other reference sites. Sites 8 and 9 (Mid-Snake @Buhl and @Kinghill, respectively) are considered water quality-limited by the U.S. Environmental Protection Agency (IDHW-DEQ 1995). Moreover, the IRI indicated that the invertebrate community of Site 7 was quite similar to those found in the degraded sties. For example, Site 7 exhibited the second greatest values of Simpson's Index and % dominance, exceeded only by Site 8. Over 60% of the invertebrate community at Site 7 was represented by the miner functional group. Continued monitoring, and possibly more detailed research, at this site appear warranted, given the importance of the native cutthroat trout fishery in this river.

Although conditions at Site 7 indicate that habitat degradation may be occurring, this conclusion may be premature given the lack of temporal sampling in this study. This study has developed protocols for the assessment of large rivers based on spatial differences among the study sites. Expansion of the methods to include temporal sampling is highly recommended, particularly for the invertebrate metrics. In temperate biomes, including Idaho, the composition of benthic invertebrate communities may change significantly from one season to the next. This temporal variation needs to be recognized and incorporated into bioassessment programs for large rivers (Minshall 1993).

Indeed, routine monitoring of biotic conditions in all large rivers is justified for the following reasons:

1. Ecological conditions in large rivers are a function of the conditions found in upstream reaches and tributaries; changes in

the integrity of a large river may indicate disturbances further up in the basin.

2. Once degraded, large rivers are more expensive and difficult to restore than are wadeable streams. Early detection of habitat degradation may prevent the need for costly restoration programs.

3. Monitoring programs establish long-term, baseline data which allow disturbance effects to be more accurately quantified, should a disturbance occur within the basin.

In summary, we have developed an index for assessing the ecological condition of large rivers in Idaho, primarily based on macroinvertebrate communities. This index, the IRI, in conjunction with water chemistry and other habitat data (e.g., substrate characteristics) provides an objective means of classifying the ecological condition of large rivers in Idaho. The use of fish metrics for assessing large rivers in Idaho is hindered due to the presence of stocking programs, the naturally low number of species, the large variation in community composition among rivers, and restrictions on fish collection in some rivers. In general, only minimal insight was gained from the fish metrics, above that obtained from the habitat and macroinvertebrate measures.

PART III - NUTRIENT LIMITATION

This section presents the results from our incorporation of measurements of nutrient limitation in bioassessment protocols. Although not as strong an indicator of degradation as the macroinvertebrate indices, monitoring of nutrient limitation appeared useful for detecting nutrient loading in rivers. The potential for using diatoms as indicators of ecological condition is also discussed.

METHODS

Small plastic containers with permeable ceramic lids were used as nutrient-diffusing substrata (hereafter, diffusers). The diffusers were filled with 4% agar solutions enriched with four nutrient treatments: nitrogen (N) as 0.1 mol/L NaNO_3 , phosphorus (P) as 0.1 mol/L KH_2PO_4 , nitrogen and phosphorus (N+P) as 0.1 mol/L NaNO_3 + 0.1 mol/L KH_2PO_4 , and unenriched 4% agar (C) (Tate 1990). These concentrations were selected to ensure diffusing nutrient levels were greater than ambient concentrations in each study reach (Fairchild and Lowe 1984).

The diffusers were attached to wooden boards and the treatment locations randomly assigned on each board (Bushong and Bachmann 1989). Boards with diffusers attached were placed between 0.5 and 1 m in depth at each site. Data for the sites used in this portion of the study are presented in Table 3 along with the initiation and completion date at each site. On each collection day, all periphyton was removed from the top of each diffuser for determination of chlorophyll a and periphyton ash-free dry mass (AFDM). The material was then filtered through a

0.45 μm pre-ashed glass fibre filter (Whatman GF/F), and frozen at -25°C for analysis in the laboratory (Robinson and Minshall 1986). In the laboratory, samples were extracted in 100% reagent-grade methanol for 24 hrs with chlorophyll a determined spectrophotometrically (Gilford model 2200) (APHA 1992). AFDM was determined using the remaining material from each sample used for chlorophyll analysis. This material was dried at 60°C , weighed, ashed at 550°C for at least 3 hrs, rewet, redried, and reweighed. The difference in weights equaled the AFDM. Differences among dates and treatments for each site by river were tested using ANOVA and Tukey's post-hoc multiple comparison test (Zar 1984).

Qualitative diatom samples were collected from additional diffusers at each site following the procedure of Robinson and Rushforth (1987). Samples were composited by treatment to facilitate analysis and preserved with 5% formalin. Each composite sample was boiled in concentrated nitric acid, rinsed, mounted in Naphrax mountant, and examined under 1000X oil immersion using a Zeis RA microscope with Nomarski optics (St. Clair and Rushforth 1976). Counts of a minimum 550-1000 diatom valves were made from each slide for estimates of relative density.

RESULTS

In general, molar N/P ratios less than 10 indicate N limitation and ratios greater than 20 indicate P limitation (Bothwell 1989, Lohman et al. 1991, Morris and Lewis 1988). The transition from N to P limitation occurs at ratios of 10-20. Which nutrient is actually limiting at ratios of 10-20 is

dependent on the species composition of the algal community; co-limitation also is possible at these ratios. Based on the molar N/P ratios, the two Mid-Snake sites (Sites 8 and 9) appeared to be P-limited (Table 3). However, actual concentrations of N and P at these sites were above saturating levels for periphyton and it is unlikely that either N or P were limiting the periphyton community in the Mid-Snake. The molar N/P ratios indicated N limitation at the other sites, with possible exceptions at Sites 1 and 6 (Bear @Peagram and Snake @Ashton, respectively). Molar ratios at Sites 1 and 6 were 10 and 17, respectively, suggesting that either N or P could be limiting, depending on the composition of the algal community.

Based on the nutrient addition experiments, only Site 1 (Bear River @Peagram) showed evidence of P limitation (Figs. 10 and 11). In northern Idaho, Site 17 (Lochsa) clearly displayed N limitation, while Sites 16 and 18 (Selway and M.F. Clearwater, respectively) appeared either N-limited or possibly co-limited, but the results were not statistically significant. At Site 4 (Snake @CoffeePot) the greatest response was observed on the N+P treatment, indicating co-limitation. The periphyton community at sites 6, 7, 8, and 9 (Snake @Ashton, S.F. Snake @Heise, Snake @Buhl, and Snake @Kinghill, respectively) did not exhibit a response to the nutrient enrichment, suggesting some factor(s) other than nitrogen or phosphorus availability was limiting these communities. These sites also had the greatest biomass of periphyton under ambient conditions.

The relative abundance of individual diatom species varied among all sites, with only a few species displaying a distinct

Table 3. Physical and chemical measures at each site used for a nutrient limitation study.

Site	Date Started/ Date Ended	Velocity (cm/sec)	Depth (cm)	Turbidity (NTU)	NO3+NO2 (mg/L as N)	NH4 (mg/L as N)	TPO4 (mg/L as P)	Molar N/P ratio
Degraded Sites								
1	09/07/95		16	38.8				
1	10/07/95	29	11	10.7	0.089	0.021	0.024	10
8	08/24/95	1	30					
8	09/25/95	0	30	9.5	1.610	0.037	0.100	36
9	08/24/95	5	30					
9	09/25/95	13	40	8.4	1.360	0.020	0.062	49
Reference Sites								
4	09/08/95	30	27	1.5				
4	10/07/95	22	25	2.2	0.002	0.010	0.009	3
6	09/08/95	32	30	1.2				
6	10/06/95	35	20	1.9	0.090	0.017	0.014	17
7	09/08/95	32	30	2.0				
7	10/06/95	20	4	1.3	0.018	0.012	0.012	6
16	08/27/95	25	30					
16	09/24/95	32	21	0.4	0.002	0.010	0.014	2
17	08/27/95	23	30					
17	09/24/95	21	27	5.1	0.003	0.006	0.016	1
18	08/27/95	25	25					
18	09/24/95	10	10	4.1	0.003	0.020	0.028	2

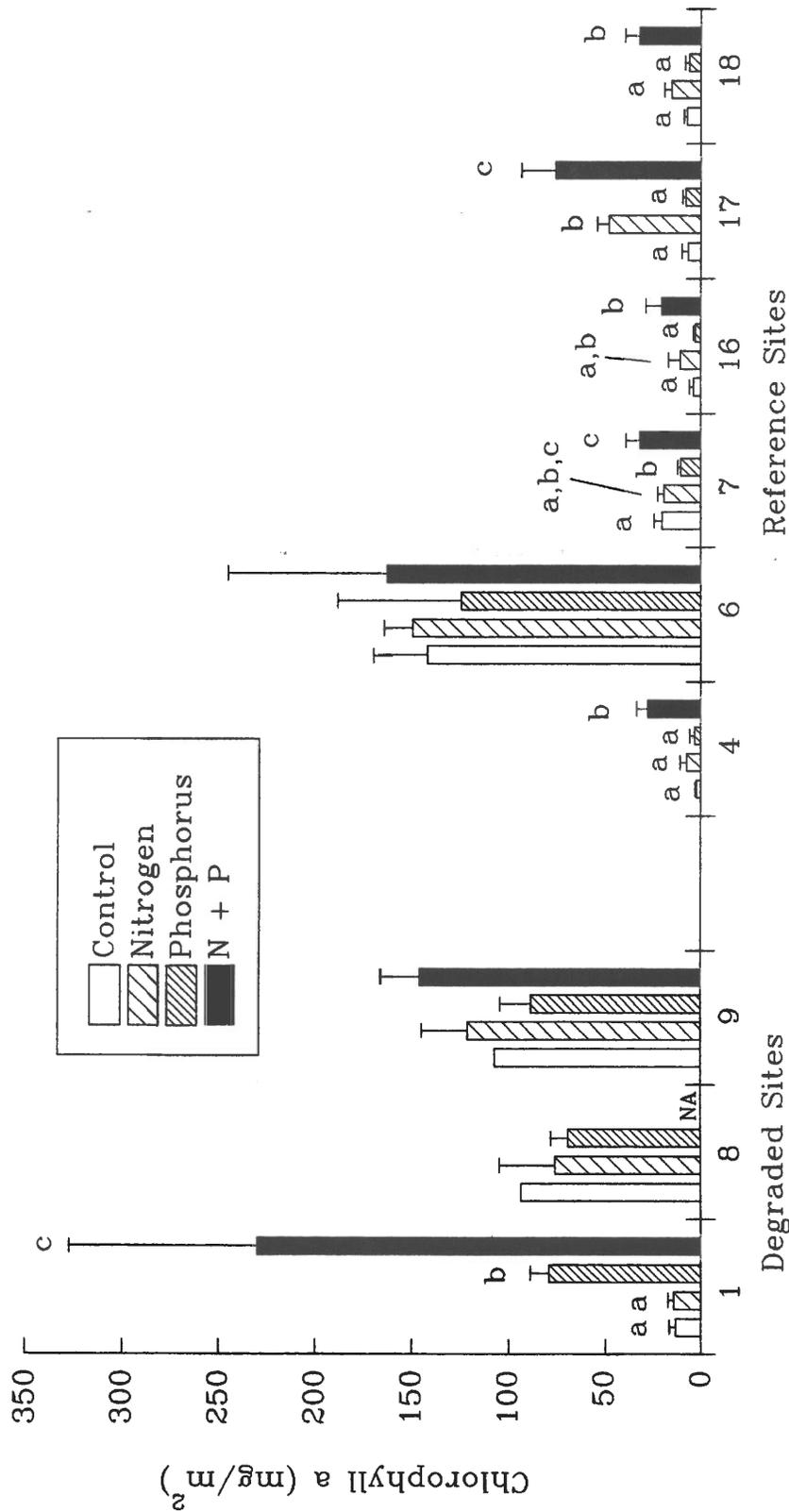


Fig. 10. Chlorophyll a levels measured on the nutrient diffusers containing the various treatments. Error bars equal +1SD from the mean. Within a site, bars with different letters were significantly ($\alpha=0.05$) different; sites without letters displayed no treatment effect. The N+P treatment was lost at Site 8 due to high flows.

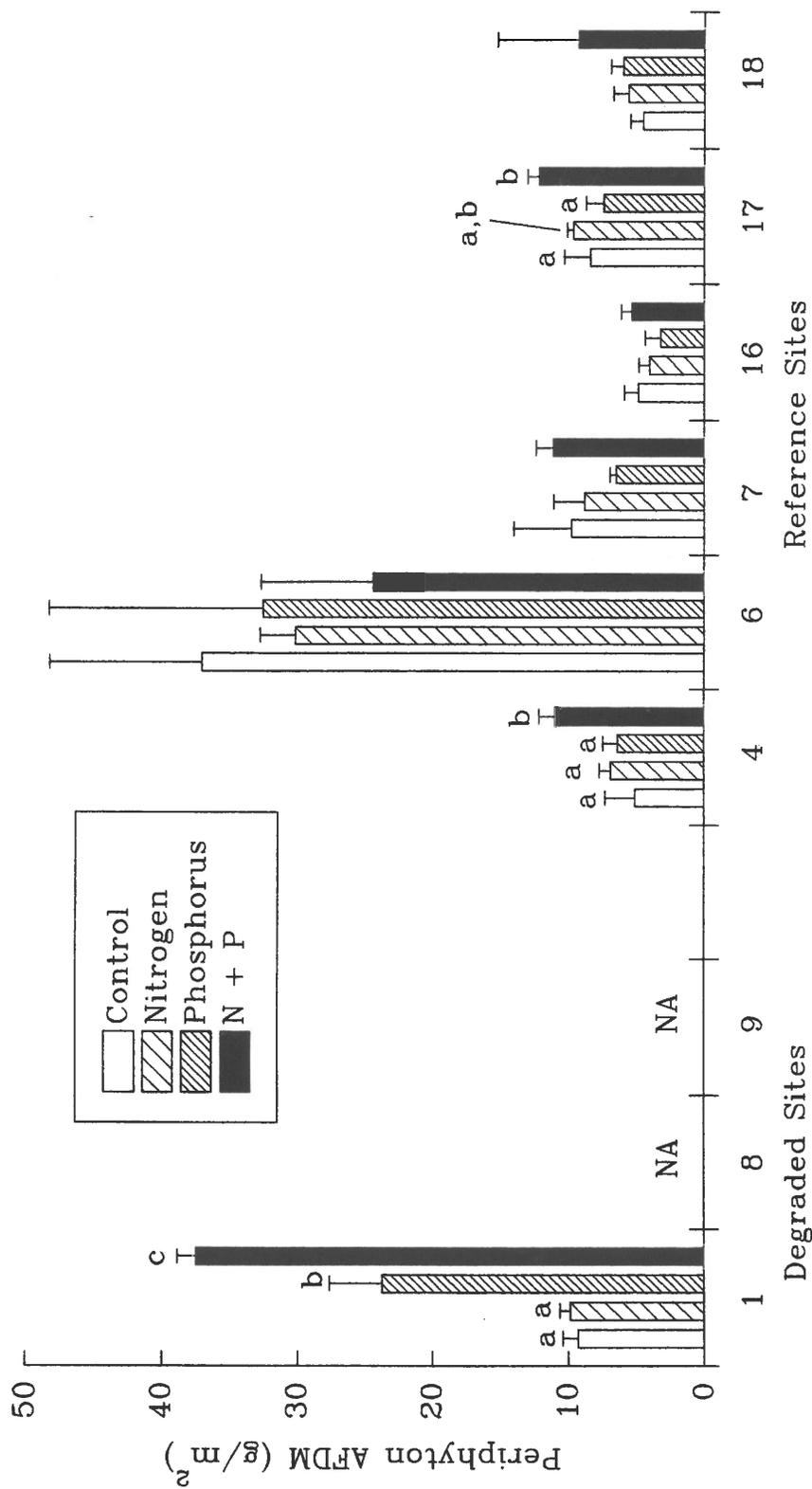


Fig. 11. Periphyton ash-free dry mass (AFDM) measured on the nutrient diffusers containing the various treatments. Error bars equal +1SD from the mean. Within a site, bars with different letters were significantly ($\alpha=0.05$) different; sites without letters displayed no treatment effect. Samples for Sites 8 and 9 were lost due to high flows.

pattern between degraded and reference sites (Fig. 12). For example, *Cymbella minuta* ranged from 1-3% of the community in the reference sites, was <0.5% of the community at Site 1 (Bear @Peagram), but did not occur in the Mid-Snake. Conversely, *Navicula menisculus* was present in each of the degraded sites, but was not found in the reference sites.

DISCUSSION

Natural sources of N to rivers include leaching from organic inputs (e.g., leaves) and overland runoff following precipitation events. Weathering of P-containing rock is the primary natural source of phosphorus for rivers. Anthropogenic sources of N and P include point-source inputs, atmospheric deposition, and runoff from fertilized agricultural fields and paved surfaces. In general, the periphyton communities in the reference sites appeared to be N-limited, while the degraded sites appeared either P-limited or limited by something other than nutrient concentration. Although P limitation is not necessarily a result of degradation, it may be indicative of N loading to the river, particularly if the surrounding rivers are N-limited.

Thus, annual monitoring of nutrient limitation and N/P ratios may provide early indications of nutrient loading in large rivers. Furthermore, these experiments provide the opportunity to efficiently examine the diatom assemblage of a river. In this regard, our results suggest the possibility of developing an index similar to that developed for macroinvertebrates, based on the composition of diatom communities. For example, some diatom species tended to be more prevalent in the degraded rivers (e.g., *Nitzschia amphibia* and *Navicula menisculus*) than in the reference

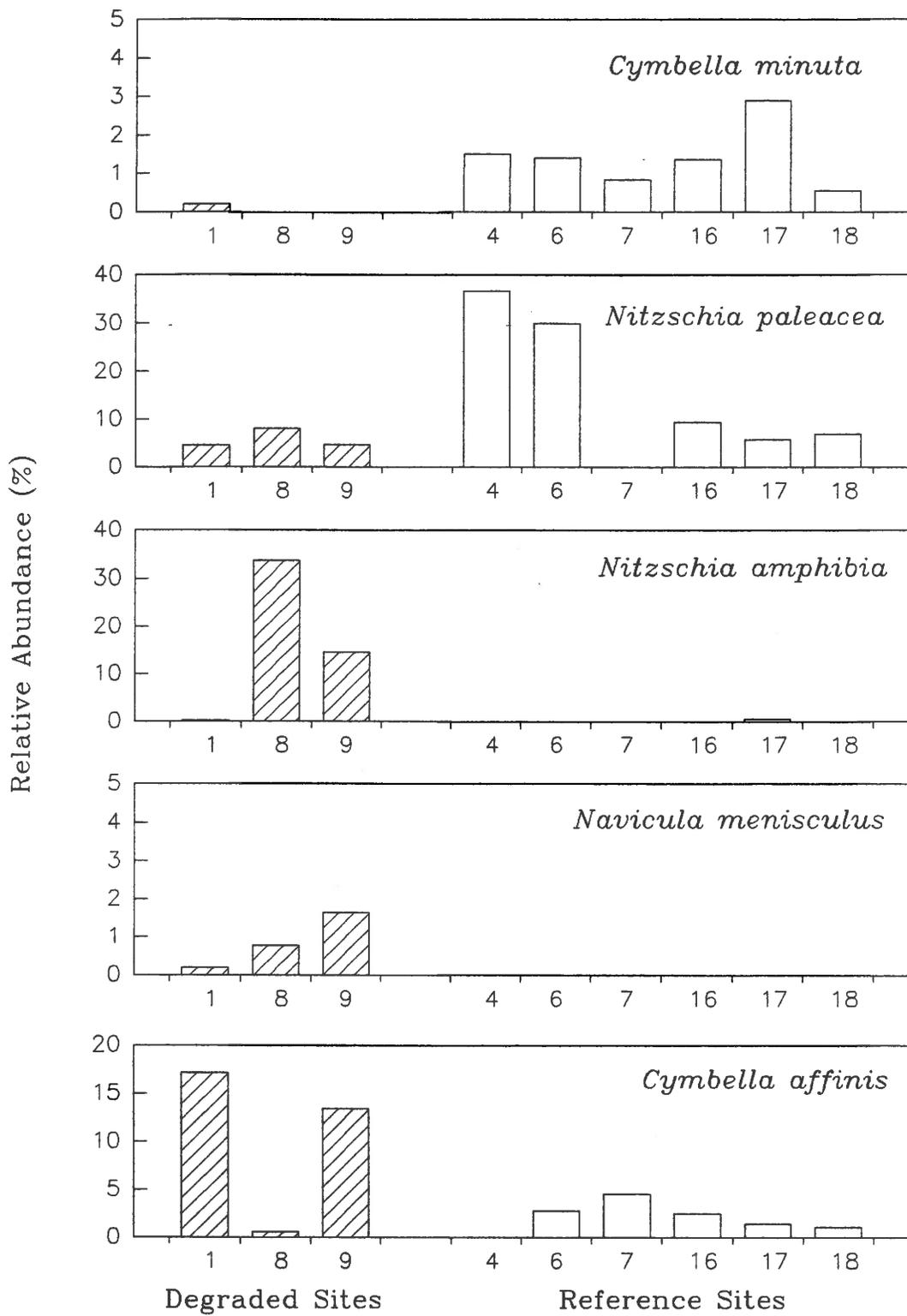


Fig. 12. Relative abundance of selected diatom species collected from the nutrient diffusers (control treatment). Note changes in scale of the Y-axis among graphs.

sites (see Fig. 12). Development of such an index will be presented at a later date, with the inclusion of data from medium-sized rivers (to increase sample size and discriminatory power).

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Appendix A. The 31 sites initially considered for use as study sites.

Bear River @Peagram
Bear River @Riverdale
Blackfoot River
Falls River
Big Lost River
Snake River @Coffeepot
Snake River @Pinehaven
Snake River @Ashton
S.F. Snake River @Heise
Snake River @Buhl
Snake River @Kinghill
Owyhee River
Bruneau River
Salmon River @Yankee
Salmon River @Challis
Salmon River @Deadwater
Salmon River @Whitebird
Little Salmon River
S.F. Salmon River
E.F. of S.F. Salmon River
Johnson Creek
S.F. Payette River
M.F. Boise River
Selway River
Lochsa River
M.F. Clearwater River
S.F. Clearwater River
Coeur d'Alene River @Shoshone
Coeur d'Alene River @Cataldo
St. Joe River @Calder
St. Joe River @Avery

Appendix B. Habitat data collected from each of the study sites. Site numbers correspond to those in Table 1.

SITE:	1	2	3	4	5	6	7	8	9	10	11
Water Velocity (m/s)	0.18	0.19	0.26	0.12	0.10	1.17	0.95	0.27	0.59	0.14	0.82
Gradient (%)	0.01	0.02	0.02	0.01	0.01	0.02	0.01	0.01	0.01	0.02	0.04
Channel Sinuosity	1.35	1.56	1.05	1.03	1.09	1.04	1.13	1.03	1.03	1.01	1.05
River Depth (cm)	86	63	55	52	63	83	87	178	178	32	57
Width:Depth Ratio	0.88	1.03	0.45	1.45	1.44	0.97	0.96	1.09	1.07	1.14	0.56
Median Bank Material Size (cm)	0.01	0.05	45.90	0.00	45.90	0.00	8.05	0.00	16.00	16.00	16.00
Bank Angle (degrees)	13.0	36.5	38.5	24.0	24.8	32.3	23.2	15.8	24.0	31.2	44.3
Median Substrate Size (cm)	7.0	10.0	25.0	2.0	6.7	11.0	11.5	0.0	13.0	20.0	40.0
Substrate Embeddedness (%)	0.41	0.72	0.41	0.06	0.37	0.10	0.13	0.93	0.48	0.46	0.45
Macrophyte Cover (%)	0.00	0.63	0.00	0.39	0.75	0.85	0.03	0.35	0.00	0.34	0.00
Alkalinity (mg CaCO ₃ /L)	126	158	26	34	89	36	68	100	98	66	76
Hardness (mg CaCO ₃ /L)	299	336	57	89	40	101	190	240	204	88	56
Conductivity (micro S/cm)	480	557	27	115	121	121	302	505	500	21	103
Phosphorus (mg/L)					0.04			0.20	1.10	0.06	0.01
Nitrate (mg/L)			0.00		1.50			1.70	1.40	1.70	1.20
Coliform Bacteria (#colonies/100mL)	50	220	10	10		240	10	10	10	10	48
Streptococci Bacteria (#colonies/100mL)	90	180	10	0	120	120	0	120	120	50	
Periphyton Chl-a (mg/m ²) mean	31.2	239.5	5.4	27.5	6.0	146.8	39.0	108.8	79.9	86.3	9.0
SD	20.4	225.4	3.9	11.6	7.0	124.0	32.7	28.4	81.0	49.2	6.0
Periphyton AFDM (g/m ²) mean	19.7	117.8	7.5	16.7	35.0	39.6	20.2	26.1	32.9	28.5	29.7
SD	17.5	104.2	5.9	6.9	30.0	25.1	9.7	4.2	28.6	19.5	16.2
Benthic Organic Matter (g/m ²)	10.4	12.0	7.7	17.9	33.2	7.5	9.0	1036.5	8.2	8.2	1012.9
Transported FPOM (g/m ³)	2.39	52.64	31.07	0.69	0.00	0.71	2.15	0.49	0.28	0.09	0.04
Transported CPOM (g/m ³)	0.83	5.15	1.66	1.81	0.02	0.94	0.46	0.66	0.06	0.18	0.01
Riparian Width (m)	118	176	32	3	1	7	213	41	11	9	5
Canopy Angle (degrees)	171	149	113	140	144	170	148	148	152	95	94
Solar Input (KJoules/m ² /yr)	19	94	27	255	281	5	92	16	21	381	53
Tree Density (#/km ²)	555	2122	2346	3898	474	2438	9565	9635	834	400	1063
Shrub Density (#/km ²)	5114	7814	294834	7963	773	7434	339335	44438	2611978	2623	2213

Appendix B. Continued.

SITE:	12	13	14	15	16	17	18	19	20	21	22
Water Velocity (m/s)	0.83	0.13	0.92	0.85	0.62	0.48	0.99	0.37	0.11	0.36	0.50
Gradient (%)	0.02	0.02	0.02	0.02	0.02	0.02	0.01	0.02	0.01	0.01	0.03
Channel Sinuosity	1.04	1.33	1.02	1.05	1.00	1.00	1.00	1.08	1.41	1.17	1.00
River Depth (cm)	78	45	73	68	82	67	159	33	119	48	35
Width:Depth Ratio	0.69	1.17	0.58	0.64	0.53	0.54	0.32	0.57	0.46	0.84	1.30
Median Bank Material Size (cm)	0.00	0.00	0.10	16.00	16.00	16.00	87.80	8.05	0.00	8.45	51.90
Bank Angle (degrees)	33.3	24.3	41.3	24.3	14.3	14.8	27.3	12.5	31.8	18.7	22.0
Median Substrate Size (cm)	12.0	17.0	15.0	12.0	40.0	27.0	24.0	17.0	5.0	10.0	28.0
Substrate Embeddedness (%)	0.26	0.48	0.34	0.33	0.36	0.39	0.28	0.14	0.13	0.16	0.29
Macrophyte Cover (%)	0.05	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.27	0.00	0.00
Alkalinity (mg CaCO3/L)	74	72	16	12	4	12	12	20	12	26	20
Hardness (mg CaCO3/L)	120	96	32	32	8	24	24	40	52	48	48
Conductivity (micro S/cm)	128	149	37	79	17	17	17	48	71	46	46
Phosphorus (mg/L)	0.01	0.05	0.12	0.16	0.23	0.03	0.10	0.09	0.17	0.03	0.22
Nitrate (mg/L)	4.40	0.08	2.10	1.60	1.10	3.40	1.20	3.80	0.90	2.50	1.00
Coliform Bacteria (#colonies/100mL)	10	50	0	40	0	0	10	80	0	10	0
Streptococci Bacteria (#colonies/100mL)	10	0	0	0	0	30	0	700	0	190	190
Periphyton Chl-a (mg/m2) mean	40.1	5.0	10.8	38.1	38.5	32.9	19.6	7.1	14.2	40.9	40.8
SD	20.9	3.0	9.4	22.1	34.7	32.7	16.2	14.6	8.8	37.1	23.6
Periphyton AFDM (g/m2) mean	29.4	27.0	16.6	16.5	17.0	13.0	8.5	4.1	8.0	14.8	15.7
SD	11.3	13.0	16.5	6.1	11.5	8.9	7.4	3.6	3.6	8.7	7.3
Benthic Organic Matter (g/m2)	7.5	34.9	5.6	10.3	9.0	1.7	3.3	8.7	2.0	2.8	10.3
Transported FPOM (g/m3)	2.43	0.00	5.28	12.63	1.87	1.51	2.22	0.47	0.14	0.35	2.44
Transported CPOM (g/m3)	0.18	0.21	0.26	0.21	0.02	0.01	0.04	0.15	0.61	0.01	0.06
Riparian Width (m)	68	7	11	19	18	10	2	29	13	74	6
Canopy Angle (degrees)	131	120	104	134	91	131	101	107	146	133	99
Solar Input (KJoules/m2/yr)	186	62	53	61	500	317	430	532	38	281	617
Tree Density (#/km2)	955	808	2578	629	1284	890	639	1404	768	843	1294
Shrub Density (#/km2)	3871	1571	18916	15243	1809	5636	1164	4962	795	3245	1433

Appendix C. Invertebrate metrics calculated for each study site. Site numbers correspond to Table 1.

SITE:	1	2	3	4	5	6	7	8	9	10	11
Total Richness	24	18	37	35		25	29		19	36	
Taxa Richness (qualitative)	24	18	37	34		25	29		19	36	
Taxa Richness (quantitative)	18	17	27	28	33	21	23	14	16	24	31
Density (#/m2)	7982	54905	52999	39339	9201	7563	74553	69839	2667	12316	10181
Biomass (mg/m2)	1545	13018	12584	4215	121699	2599	93467	21817	1132	5033	21984
EPT richness	12	5	18	20	17	9	15	1	7	17	20
Ephemeroptera Richness	7	2	9	11	8	1	7	0	4	10	6
Plecoptera Richness	1	0	3	1	4	3	4	0	0	0	7
Trichoptera Richness	4	3	6	8	5	5	4	1	3	7	7
% EPT	0.79	0.40	0.21	0.39	0.30	0.53	0.29	0.03	0.24	0.51	0.51
% Dominance	0.20	0.44	0.52	0.32	0.25	0.30	0.67	0.79	0.41	0.21	0.19
% Hirudinea	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00
% Elmidae	0.01	0.02	0.02	0.01	0.04	0.01	0.00	0.00	0.00	0.03	0.09
% Hydropsyche	0.16	0.05	0.00	0.01	0.04	0.07	0.12	0.00	0.18	0.06	0.15
% Gastropoda	0.01	0.00	0.00	0.02	0.06	0.04	0.00	0.79	0.10	0.03	0.00
% Pteronarcys	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01
% Amphipoda	0.00	0.02	0.00	0.08	0.25	0.00	0.00	0.00	0.04	0.01	0.00
% Chironomidae	0.06	0.44	0.52	0.32	0.07	0.30	0.67	0.00	0.41	0.16	0.19
% Oligochaetae	0.11	0.00	0.00	0.05	0.18	0.05	0.01	0.17	0.13	0.21	0.04
% Predator	0.01	0.04	0.09	0.04	0.08	0.03	0.04	0.04	0.00	0.05	0.12
% Gatherer	0.28	0.27	0.05	0.33	0.34	0.01	0.01	0.00	0.13	0.15	0.19
% Scraper	0.23	0.11	0.20	0.13	0.22	0.32	0.13	0.79	0.13	0.34	0.19
% Shredder	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01
% Filterer	0.30	0.14	0.14	0.14	0.11	0.28	0.15	0.00	0.20	0.09	0.26
% Miner	0.17	0.44	0.52	0.37	0.25	0.35	0.67	0.17	0.54	0.37	0.23
Filterer/Scraper Ratio	1.28	1.27	0.72	1.06	0.51	0.85	1.18	0.00	1.58	0.27	1.39
Miner/Scraper Ratio	0.75	4.07	2.64	2.89	1.11	1.06	5.16	0.22	4.30	1.09	1.20
Simpson's Index	0.14	0.27	0.30	0.14	0.12	0.19	0.47	0.65	0.23	0.13	0.09
HBI	3.39	4.94	5.11	4.21	4.25	3.77	4.99	1.18	5.55	3.90	3.90

Appendix C. Continued.

SITE:	12	13	14	15	16	17	18	19	20	21	22
Total Richness	25		39	29	41	37	38	43	15	38	42
Taxa Richness (qualitative)	24		37	29	41	36	38	43	15	38	42
Taxa Richness (quantitative)	19	30	25	23	29	25	25	29	10	21	26
Density (#/m2)	36935	7328	8017	32306	49408	5860	25707	24740	3686	19591	51321
Biomass (mg/m2)	4802	6438	3916	6169	14270	2486	8921	5659	460	2559	5270
EPT richness	14	20	23	18	21	22	26	23	6	20	23
Ephemeroptera Richness	8	6	12	8	9	11	12	12	2	12	12
Plecoptera Richness	3	6	4	4	5	6	6	5	2	3	5
Trichoptera Richness	3	8	7	6	7	5	8	6	2	5	6
% EPT	0.33	0.59	0.49	0.27	0.40	0.55	0.53	0.56	0.05	0.56	0.44
% Dominance	0.37	0.21	0.29	0.49	0.49	0.20	0.25	0.22	0.83	0.27	0.32
% Hirudinea	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
% Elmidae	0.03	0.03	0.01	0.04	0.05	0.06	0.11	0.20	0.00	0.02	0.10
% Hydropsyche	0.02	0.15	0.13	0.01	0.25	0.20	0.02	0.00	0.00	0.00	0.18
% Gastropoda	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
% Pteronarcyis	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
% Amphipoda	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
% Chironomidae	0.37	0.21	0.29	0.49	0.49	0.12	0.25	0.12	0.83	0.27	0.32
% Oligochaetae	0.01	0.04	0.01	0.03	0.00	0.08	0.10	0.00	0.00	0.00	0.05
% Predator	0.03	0.15	0.12	0.14	0.10	0.27	0.03	0.22	0.10	0.17	0.15
% Gatherer	0.06	0.16	0.06	0.10	0.06	0.13	0.23	0.31	0.06	0.31	0.16
% Scraper	0.26	0.19	0.23	0.11	0.07	0.19	0.20	0.26	0.00	0.24	0.07
% Shredder	0.00	0.01	0.02	0.00	0.01	0.00	0.16	0.02	0.00	0.00	0.06
% Filterer	0.25	0.23	0.28	0.14	0.27	0.21	0.02	0.06	0.00	0.01	0.19
% Miner	0.38	0.25	0.30	0.52	0.49	0.19	0.35	0.13	0.83	0.27	0.37
Filterer/Scraper Ratio	0.97	1.24	1.21	1.31	3.71	1.08	0.11	0.21	0.00	0.03	2.54
Miner/Scraper Ratio	1.46	1.34	1.31	4.89	6.72	1.01	1.69	0.48	1298	1.15	5.05
Simpson's Index	0.22	0.09	0.13	0.26	0.31	0.10	0.14	0.11	0.70	0.15	0.16
HBI	4.59	3.24	3.92	4.73	4.65	4.00	3.30	3.70	5.72	3.07	4.27

Appendix D. Fish metrics calculated for each study site. Site numbers correspond to Table 1.

SITE:	# Native Species	# Cottidae	# Native Cyprinidae	# Catastomidae	# Intolerant Species	# Native Salmonidae	% Carp (biomass)	% Omnivores	% Insectivores	% Introduced Species
1	7	2	2	1	5	2	42	35	49	29
2	4	1	2	1	2	1	58	32	39	46
3										
4	2	1	1	1	4	0	0	0	79	50
5	5	1	2	0	5	2	0	0	100	88
6	7	1	3	1	5	2	0	33	65	10
7	6	1	2	1	4	2	0	1	63	36
8	6	0	6	1	0	0	54	55	35	14
9	8	1	6	2	2	0	51	44	29	7
10	3	1	1	0	0	0	0	0	0	45
11	5	1	1	1	0	2	0	0	67	11
12	7	0	2	1	5	4	0	9	91	11
13	5	1	1	1	0	2	0	0	70	0
14	4	1	1	2	2	2	0	7	92	0
15	4	1	3	0	2	0	0	0	100	4
16	8	2	2	0	6	4	0	0	87	0
17	6	1	2	0	4	3	0	0	100	0
18	6	1	1	0	5	4	0	0	96	0
19	7	1	3	1	4	2	0	10	82	16
20	1	0	0	1	0	0	0	63	6	38
21	5	1	2	1	3	2	0	19	47	1
22	7	1	2	1	5	3	0	11	78	4

