The use of aquatic macroinvertebrates as water quality indicators in mountain streams in Montana
by David C Richards

A thesis submitted in partial fulfillment of the requirements for the degree Master of Science in Entomology
Montana State University

Abstract:
The use of aquatic macroinvertebrates for monitoring water quality has become popular. This paper assessed the ability of rapid assessment using a one minute riffle kick net method to detect natural within-stream and seasonal macroinvertebrate variability in the mountain stream ecoregion of Montana. It also assessed the ability of these methods to detect water quality impairment. Results suggest that the methods used are able to reflect natural variability and that some indices or 'metrics' are potentially more useful than others for detecting water quality impairment in heavily 'impaired' streams. It is unkown how useful these methods are for detecting water quality impairment in less severely impaired streams. Suggestions are also made on how to increase the sensitivity of these methods.

THE USE OF AQUATIC MACROINVERTEBRATES AS WATER QUALITY INDICATORS IN MOUNTAIN STREAMS OF MONTANA

by

David C. Richards

A thesis submitted in partial fulfullment
of the requirements for the degree

Master of Science

in

Entomology

MONTANA STATE UNIVERSITY-BOZEMAN
Bozeman, Montana

May, 1996

APPROVAL

of a thesis submitted by

David C. Richards

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

Dr. Florence V. Dunkel — Florence V. Dunkel (Signature) — 15 May 96 Date

Approved for the Department of Entomology

Dr. Greg Johnson — Gregory Johnson (Signature) — 5-15-96 Date

Approved for the College of Graduate Studies

Dr. Robert Brown — R L Brown (Signature) — 10/4/96 Date

STATEMENT OF PERMISSION TO USE

In presenting this thesis (paper) in partial fulfillment of the requirements for a master's degree at Montana State University-Bozeman, I agree that the Library shall make it available to borrowers under rules of the Library.

If I have indicated my intention to copyright this thesis (paper) by including a copyright notice page, copying is allowable only for scholarly purposes, consistent with "fair use" as prescribed in the U.S. Copyright Law. Requests for permission for extended quotation from or reproduction of this thesis (paper) in whole or in parts may be granted only by the copyright holder.

Signature ______________________

Date 5/31/96

ACKNOWLEDGMENTS

I would like to thank the members of my graduate committee, Dr. Florence Dunkel, Dr. Kevin O' Neill, Dr. Cliff Montagne, Dr. William Kemp, Dr. Daniel Gustafson, and Rober Bukantis. I would also like to thank the Water Quality Division of the Montana Department of Environmental Quality for financial support. I would also like to thank Marni Ralston, Mary Bauman, Brenda Beaman, Li Ying Wong, and Steve and Youngsok Hamner for their field and lab assistance.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

The use of aquatic macroinvertebrates for monitoring water quality has become popular. This paper assessed the ability of rapid assessment using a one minute riffle kick net method to detect natural within-stream and seasonal macroinvertebrate variability in the mountain stream ecoregion of Montana. It also assessed the ability of these methods to detect water quality impairment. Results suggest that the methods used are able to reflect natural variability and that some indices or 'metrics' are potentially more useful than others for detecting water quality impairment in heavily 'impaired' streams. It is unkown how useful these methods are for detecting water quality impairment in less severely impaired streams. Suggestions are also made on how to increase the sensitivity of these methods.

INTRODUCTION

Freshwater is one of the most important resources in the world, however, within the last two hundred years, technology and human population growth have caused tremendous impacts on freshwater ecosystems (Bau 1995; Karr 1991). For example, Benke (1990) reported that 98% of the 3.2 million miles of rivers in the U. S. were not healthy enough to be considered high quality and worthy of federal protection.

The U.S. Clean Water Act of 1972 (CWA) reflects the concerns of the people of the United States regarding clean water and healthy aquatic ecosystems and mandates, as its goal, "...to improve and maintain the physical, chemical, and biological integrity of our nation's waters". Biological integrity can be defined as "...the ability of an aquatic ecosystem, to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats of a region" (Karr and Dudley 1981).

The U. S. Environmental Protection Agency (U.S. EPA) is the federal regulatory agency responsible for the enforcement of the Clean Water Act of 1972. The U.S. EPA has encouraged individual states to develop their own regionalized methods of assessing the biological integrity of their (states) surface waters based on a model proposed in the U.S. EPA's Rapid Bioassessment Protocols (RBP's)(U.S. EPA 1989).

Most RBP's rely on a multi-metric approach using reference conditions from a regional framework (Davis and Simon 1995). A metric is simply a numeric index which estimates some attribute of a community, for example, the number of taxa present in a sample or the number of individual organisms in a sample. Metrics that were used in this study are in Table 1.

Table 1. Metrics used in this study.

Taxa richness[1]	EPT Richness[2]	% dominant taxa
# of organisms/ sample	% EPT individuals	modified HBI[5]
# of Ephemeroptera taxa	QSI for taxa[3]	Shannon Index[6]
# of Plecoptera taxa	# of cg and fg taxa[4]	% predator taxa
# of Trichoptera taxa	# of shredder taxa	% shredder taxa
% semivoltine taxa	# of scraper taxa	% scraper taxa
% multivoltine taxa	# of predator taxa	% cg and fg taxa
% univoltine taxa	MTI [7]	

[1]Taxa richness is the number of different taxa
[2] EPT richness is the number of Ephemeroptera, Plecoptera, and Trichoptera taxa
[3] QSI for taxa is the Quantitative Similarity Index based on percent relative abundances of taxa between samples
[4] # of cg and fg taxa is the number of collector-gatherer and filterer-gatherer taxa based on Merrit and Cummins (1984)
[5] HBI is the Hilsenhoff Biotic Index based on organic pollution tolerance values (Hilsenhoff 1987 and Bukantis 1995b)
[6] Shannon Index is a biotic diversity index
[7] MTI is a metals tolerance index (Bukantis 1995b)

Reference biological and habitat conditions are obtained from "least impaired" sites in an ecoregion or sub-ecoregion (regional framework) as defined by Omernik (1987). A biological assessment is then made for the site that is thought to be impaired within the reference ecoregion using this multi-metric approach. RBP's are used by many state and federal agencies for monitoring water quality. Regionalized RBP's are designed to be efficient, economical, easily interpretable, and accurate in assessing water quality impairment, at least on a gross level (Resh and Jackson 1993). Results of many RBP's can be implemented in less than 5 days (Lenat and Barbour 1994) although, often this may not be true for agencies that do not have the resources available (Bukantis 1995). RBP's can be used for both point and non-point sources of impairment (Resh and Jackson 1993). Aquatic macroinvertebrates are the most frequently used organism in RBP assessments (Rosenberg and Resh 1993).

The State of Montana is presently developing its own modified RBP's for wadeable rivers and streams in Montana and has delineated the state into three ecoregions: 1) the plains, 2) foothills and intermountain valleys, and 3) mountain ecoregions based on Omernik (1987) and from preliminary findings of macroinvertebrate communities in Montana using Detrended Correspondence Analysis (DECORANA) (Wisseman 1990). Water quality impairment in these three ecoregions in Montana is mostly due to non-point sources (Montana 305b Report 1994). The majority of non-point impairment in Montana to rivers and streams is from agriculture, logging and mining (particularly in the mountain ecoregion), road building, sedimentation, irrigation, and habitat loss (Montana 305b Report 1994). Of the 176,750 miles of streams in Montana, only 10% have been assessed for water quality (Montana 305b Report 1994). The State of Montana Water Quality Division of the Department of Environmental Quality has been developing RBP's since 1989 (Wisseman 1990) and is increasing its data base annually (McGuirre 1994, 1995). Bollman (1995) is presently completing an extensive study in the foothill and intermountain ecoregion in Montana. Very little data exists using macroinvertebrates to assess water quality impairment in the mountain streams ecoregion of Montana.

HYPOTHESES TESTED

I tested four hypotheses:

1) within-site variation exists in the abundance and distribution of macroinvertebrate taxa in mountain streams of Montana and can be detected with methods used in this study.

2) within-stream variation exists in the abundance and distribution of macroinvertebrate taxa in mountain streams of Montana and can be detected with methods used in this study.

3) seasonal variation exists in the abundance and distribution of macroinvertebrate taxa in mountain streams of Montana and can be detected with methods used in this study.

4) water quality impairment affects the abundance and distribution of macroinvertebrate taxa in mountain streams of Montana and can be detected with methods used in this study.

LITERATURE REVIEW

Water Quality Problems in the U.S.

Human caused degradation of river ecosystems worldwide has reached unprecedented levels. "Not one riverine system in America has been spared"(Doppelt et al. 1993). Dopplet et al. (1993) report that 50% of our nations' waters fail to meet federal water quality standards. Also, less than 2% of our nations's rivers even qualify for Wild and Scenic designation. From one-third to three-fourths of aquatic species nationwide are rare to extinct. Aquatic species are disappearing at a faster rate than terrestrial species (Doppelt et al. 1993) with twenty percent of native fishes in the western U.S. extinct or endangered (Miller et al. 1989). Two hundred and fourteen stocks of native Pacific salmon and steelhead are facing extinction (Nehlsen et al. 1991) and commercial fish harvests in major rivers throughout the U.S. have declined from 80% to 100% (Karr et al. 1985; Ebel et al. 1989; and Patrick 1992). Karr (1991) asked 'what would our country think if U. S. agricultural productivity decreased by 80%?'

It is estimated that 60% to 80% of natural riparian vegetation has already been lost or is degraded due to human activities in the U.S. (Swift 1984), while seventy percent of our nations' rivers and streams have been impaired by flow alteration (Doppelt et al. 1993). Over 600,000 miles of rivers have already been dammed in the U. S., with the Yellowstone River being the only remaining free- flowing large river (longer than 600 miles) left in the U.S., outside of Alaska (Karr 1995).

The only attempt at quantifying river channelization in the U.S. was conducted over 20 years ago in 1972 by the Council of Environmental Quality which estimated that 235,000 miles of streams and rivers in the U.S. have been channelized (Doppelt et al. 1993). The United States Environmental Protection Agency (U.S. EPA) reports that nonpoint sources affect 65% of impaired streams (Karr 1991).

Water Quality Problems in Montana

Assessments of water quality in Montana have been completed in only 10% of its stream miles (Montana Water Quality Division 1994). The State of Montana classifies its streams as fully supporting, partially supporting, or non-supporting of their designated uses. Non-supporting stream miles total 925 (about 5% of those assessed) and partially supporting streams total 75% of those assessed. It is unknown how many unassessed stream miles (159,070 miles) are fully supporting their designated uses. Agriculture has impaired 75% of the stream miles that have been assessed and 90% of stream mileage impairment in Montana comes from non-point sources (Montana Water Quality Division 1994). Nutrients, siltation, suspended solids, salinity, flow and habitat alterations, and metals are the major causes of stream impairment in Montana (Montana Water Quality Division 1994).

Biomonitoring Approaches

The idea of biological indicator organisms is not new. A kings' use of winetasters and the use of canaries by miners to detect dangerous gases are familiar examples of biological indicators (Cairns and Pratt 1993). Even the relative abundance of rats in our cities is a biological indication of environmental conditions.

The use of freshwater aquatic organisms as bioindicators of water quality appears to have its origins in Europe with the work of Kolenati in 1848. Kolenati (1848) reported that city effluents caused the disappearance of caddisflies downstream. Kolkwitz and Marsson (1908) related the degree of pollution in a river as a measure of sewage organic matter. Other researchers then began developing lists of pollution 'indicator organisms' in rivers and streams (Richardson 1925, 1929; Gaufin 1958; Cairns and Pratt 1993). Cairns and Pratt (1993) suggest that presently, "biological surveillance of communities - with special emphasis on characterizing taxonomic richness and composition - is perhaps the most sensitive tool now available for quickly and accurately detecting alterations in aquatic ecosystems".

Aquatic macroinvertebrates are the most commonly used indicator organisms for water quality assessment (Rosenberg and Resh 1993), although other aquatic organisms are routinely used, including fishes (Simon and Lyons 1995) and periphyton (Bahls 1993). Some of the reasons for using aquatic macroinvertebrates as indicators of water quality include: 1) macroinvertebrates occur in all types of freshwater environments (Merrit and Cummins 1984), 2) the many different kinds of macroinvertebrate species found in freshwater habitats allows for a wide range of responses to different sources of impairment (Hellawell 1986, Abel 1989), 3) aquatic macroinvertebrates are less mobile than fishes which allows for spatial analysis of impairment (Slack et al. 1973, Hellawell 1986) and 4) macroinvertebrates have longer life cycles compared with microorganisms and periphyton, which allows for analysis of disturbance over time (Gaufin 1973, Abel 1989). Aquatic macroinvertebrates are "on site monitoring the water, regardless of the presence or absence of the investigating biologist" (Nehrig 1976).

Many types of biomonitoring of freshwater ecosystems are in use (Johnson et. al 1993). The use of individual organisms for toxicity testing for various pollutants in the laboratory, in artificial streams, and in field studies is common (Gaufin 1973, Rahel and Kolar 1990, Dunkel and Richards, submitted). These studies typically measure physiological, biochemical, behavioral, and life history changes of the individual test organisms to various types of pollutants (Marten and Zwick 1989, Nardi and Watson 1990, Johnson et al. 1993).

Population and community structure studies are also used to measure the effects of different pollutants on aquatic macroinvertebrates both in the laboratory and more commonly in field studies. These studies may use a univariate or multivariate approach or a combination of the two (Johnson et al. 1993).

Univariate studies of populations and communities can be used to develop biotic indices and scoring systems. An example of such an index is the Hilsenhoff Biotic Index which scores the sensitivities of many aquatic macroinvertebrates to organic pollution and is widely used in monitoring programs (Hilsenhoff 1987, Resh and Jackson 1993). A prerequisite to the use of biotic indices is a thorough knowledge of how the type of pollution used for each index affects the biota (Johnson et al 1993).

Often graphical analysis of macroinvertebrate data can provide more insight into biology than a simple p-value (Fore et al. 1995). For example, Fore et al. (1995) were interested in whether scores (metrics) for best and worst sites clumped together tightly or were spread more evenly. Graphs allowed them to examine each metric's range and evaluate where along the continuum of degradation the metric was most sensitive. Fore et al. (1995) suggest that the main problem with statistical correlation is that no single variable can summarize all the human activities that degrade streams. Different data may be appropriate for different types of degradation. Fore et al. (1995) doubt that any single land-use or chemical variable can reliably rank sites. Fore et al. (1995) suggest that this is a primary reason for evaluating the river condition according to the resident biota. In most cases, sites that appear to a human observer to be heavily used and visibly damaged usually are more degraded than those that appear pristine (Fore et al. 1995).

Multivariate approaches including regression analysis can be used to ordinate populations or communities along environmental gradients (Digby and Kempton 1994). Johnson et al. (1993) suggested that direct gradient analysis (regression) could be used to predict species assemblages along a known environmental gradient. This information can then later be used to infer water quality from species assemblages found in a study site.

Principal Components Analysis (PCA) is an example of indirect gradient analyses which measure species assemblages gradients regardless of any environmental gradient (Johnsen et al. 1993). For example, in Principal Components Analysis, Principal Component 1 (or eigenvector 1) contains the largest individual differences in a species assemblage. The second Principal Component (eigenvector) accounts for the largest proportion of remaining individual differences not correlated to (or orthogonal to) Principal Component 1 (Williams and Feltmate 1992). PCA calculates the line, or component, that extracts the maximum amount of statistical variance from a cloud of points (Tabachnick and Fidell 1989). For stream macroinvertebrate analysis, each point then will represent a stream site. The number of dimensions through which the line passes would equal the number of taxa collected (Fore et al.1995). Usually, PCA data uses species lists and abundances to interpret differences between stream sites (Norris and Georges 1993). If land use data are not sufficient to perform canonical correspondence analysis, then PCA may be used and site characteristics can later be identified on the principal components plots (Fore et al. 1995). Hannaford and Resh (1995) used PCA to analyze the correlation matrix of six RBP metrics used in their study of variability in rapid-bioassessment surveys. In an intensive ecological study, Gustafson (1990) was able to use principal components analysis to identify the velocity preferences of aquatic insects in the Gallatin River, MT and identify the insect species distribution to three distinct sections of the river. Gustafson (1990) also used a faunal and physical component PCA analysis to easily identify headwater insect species, lower Gallatin River species, and widespread species within the drainage.

With the many types of data analysis available, it should be pointed out that ..."Too often biological patterns in complex data sets are reported in terms of complicated, high level statistics such as principal components analysis" (Fore et al. 1995). While many times it is easier to compare sites and search for patterns in the raw data by simply noticing which taxa were present or absent in similar sites (Fore et al. 1995). All of these quantitative methods mentioned above require many replicates with detailed statistical analysis, which may be cost prohibitive given the thousands of miles of rivers and streams potentially impaired in the U. S. (Resh and Jackson 1993).

Rapid Assessment

With the high cost associated with quantitative methods, limited budgets and limited personnel in management agencies responsible for water quality, and given the increasing deterioration of water quality throughout the world, the most recent approach in biomonitoring has been "rapid assessment". Rapid assessment is a qualitative method (sometimes semi-quantitative) designed to measure water quality and can also be used to measure long-term regional changes (Resh and Jackson 1993).

63% of the 123 U.S.D.A. National Forest Districts and 61% of the 56 Department of Interior, Bureau of Land Management Districts surveyed by Angradi and Vinson 1995, reported conducting some type of aquatic macroinvertebrate monitoring. Mining, timber harvest, and grazing management were the most often cited reasons for using aquatic macroinvertebrate monitoring. Protocol III is the most rigorous tier level approach for the RBP's suggested by the U.S. EPA and is used by over 20 states (Resh et al. 1995). Protocol III identifies macroinvertebrates to the genus or species level and incorporates a habitat assessment for each site.

Many of the assessment methods used throughout the U.S. rely on a multimetric approach, while in Europe a multivariate approach is more common. Norris (1995) discussed the use of multivariate methods in Europe and suggested that multivariate methods are seriously under used in the U.S. Gerritsen (1995) on the other hand, argues that multimetric methods used in the U.S. function well and that multivariate methods "..are more complex, require additional specialization by practitioners, and are more difficult to convey to managers and the public". Fore et al. (1995) state that, "multimetric indices formalize what any good biologist familiar with local biota, knows about the biological condition of a stream". Indices, thus, become important tools for communication with non specialists. A multimetric approach can be compared to familiar economic indexes such as the consumer price index which has been widely used for over 50 years (Fore et al. 1995).

Rapid bioassessments are frequently compared to the use of thermometers in assessing human health (Karr 1995; Resh and Jackson 1993). Results are easily obtained and can be compared to the "normal" condition. Resh and Jackson (1993) point out that the key questions then become: what populations and community measures are biologically relevant (the thermometers)?, what are the thresholds against which they are being compared (the normal body temperature)? and, how much of a deviation from the threshold is a sign of "ill health"? They suggest that a critical and often controversial decision is the selection of the most appropriate measures for use in multimetric rapid assessment approaches.

Many metrics or indices are presently being tested and used in rapid assessment programs throughout the U.S. including the Pacific Northwest (Fore et al. 1995). Resh and Jackson (1993) gave an excellent overview of the many metrics that have been proposed for use, as well as the various collection methods, sampling strategies and taxonomic resolution used by various researchers and agencies.

Problems with Rapid Assessments

Even though rapid assessments are becoming very popular, their ability to detect water quality impairment is often questioned (Norris 1995, Southerland and Stribling 1995, Resh and Jackson 1993). Although rapid bioassessments were designed to be cost effective, several Forest Service Districts and Bureau of Land Management Districts surveyed by Angradi and Vinson (1995) reported that they could not afford to conduct assessments properly. Several districts also reported that natural variability is so great in aquatic macroinvertebrate communities that current methods are unreliable or insensitive to land management impacts.

Another problem using RBP's concerning natural within-site variability is that most RBP's, including Protocol III of the U.S. EPA, suggest collection of only one sample per site (Resh and Rosenberg 1989). Research has shown variability in the replication of many RBP metrics (Resh et al. 1990, Steven and Szczytko 1990, Barbour et al. 1992, Hannaford and Resh 1995, and Resh 1994).

The choice of sampling methods and the choice of the most appropriate habitat to be sampled may influence results (Rosenberg and Resh 1993). Even the choice of the portion of a riffle to be sampled may be biased due to the distribution of benthic organisms within the riffle. Brown and Brown (1984) showed a strong upstream biased distribution of insects within riffles. With all of the problems seemingly associated with rapid assessment, many researchers and mangers feel that these problems can be addressed and eventually overcome with an increased understanding of macroinvertebrate community responses to water quality impairment.

Rapid Assessment in Montana

The Water Quality Division (WQD) of the Montana Department of Environmental Quality (DEQ) is presently assessing the applicability of its modified version of the U. S. Environmental Protection Agencies (EPA) Rapid Bioassessment Protocols (RBP's) using macroinvertebrates for monitoring water quality in mountain streams of Montana. These methods use a multi-metric approach based on reference conditions within a regional framework (Davis and Simon 1995). The Montana Water Quality Division recommends the collection of macroinvertebrate samples during the summer season (June 21 to September 21), following runoff. This is because the Montana WQD has the most reference data from this season and because stream flows and weather conditions are most suitable for field work. The Montana WQD also recommends that for monitoring trends at a particular site over time, sampling as close as possible on the same date will produce the best results (Bukantis 1995).

Numerous researchers have suggested that within-site, within-stream, seasonal, and yearly variability of stream macroinvertebrate communities may affect rapid assessment metrics enough to make them unreliable for assessing water quality impairment (Resh and Jackson 1993; Bode and Novak 1995; Angradi and Vinson 1995). To be effective, rapid assessment metrics must be able to discriminate between impairment and natural variability (Bode and Novak 1995; Barbour et al. 1995). Very little analyzed data exists concerning metric variability using rapid assessment in mountain streams of Montana.

Macroinvertebrate Variability in Mountain Streams

There has been much documentation of within-stream and seasonal variability of macroinvertebrates in mountain streams and these types of variability are well understood concepts (Ward and Kondratieff 1992; Resh and Rosenberg 1989; Yoder and Rankin 1995). For example, longitudinal macroinvertebrate community changes within a stream from headwaters downstream have fostered many classification and zonation schemes including the River Continuum Concept (Vanote et al. 1980) and Illies' and Botosaneanu's (1963) zonation system.

Seasonal and yearly variability studies are numerous. Richards et al. (1995) reported a 3 to 4- fold increase in abundance of adult *Pteronarcys californica* (Plecoptera) in the Madison River in SW Montana between 1994 and 1995. Gustafson (1990) conducted an intensive study of the ecology of aquatic insects in the Gallatin River drainage in SW Montana. From his work on the mainstem of the Gallatin River, Gustafson (1990) reported that total macroinvertebrate density was very low the first summer but was at an all time high during the following summer, which he attributed to an early spring run-off before the second summer. Gustafson (1990) also reported that in autumn, the species numbers peak due to the appearance of species with winter life cycles. A winter depression of species numbers occurred due to winter mortality or downward migration from the top layer of the benthos. The peak in species number during the spring was associated with the appearance of species with summer life cycles, which occurs before all species with winter cycles have emerged. Gustafson (1990) stated that spring peaks in species numbers occurred during a depression in total invertebrate density. A decline in the average number of species after the spring peak was associated with an increased total density because many of the species with winter life cycles were absent from the benthos and the species with summer life cycles became very abundant (Gustafson 1990). Gustafson's (1990) conclusion was that there were low macroinvertebrate densities in summer, then an increase during autumn, a decline in winter, a brief increase in the spring, followed by a decline to the summer low.

The State of Ohio suggested that fundamental decisions need to be made early in the development of biocriteria, including accounting for seasonal variability (Yoder and Rankin 1995). The State of Ohio confines their collection times to June through September and is presently analyzing temporal variability from a special Ohio EPA methods study conducted in 1981 (DeShon 1995). Bode and Novak (1995) tested seasonal variability by sampling two streams monthly for a year. They suggested that between-month comparisons should not be done. Reash (1995) reported that seasonal and yearly variability must be accounted for when deriving biocriteria. Reash (1995) based this by monitoring fish populations in the Ohio River, where 50% of the electrofishing and gill net samples showed significant (5% level) temporal variability in catch per unit effort or in total biomass. Resh and Jackson (1995) reported that in their study of the accuracy of macroinvertebrate rapid assessment measures, 16 of the 22 measures tested found significant (5% level) seasonal differences. A detailed knowledge of life histories, especially voltinism and phenology, of macroinvertebrates encountered in biomonitoring samples should help alleviate many problems associated with seasonal variability (Rosenberg and Resh 1995; Johnson et al. 1995).

Functional feeding groups are also known to follow seasonal patterns of abundance (Cummins 1974). The functional feeding group classification system "distinguishes macroinvertebrate taxa that perform different functions within aquatic ecosystems with respect to processing of nutritional resource categories" (Merrit and Cummins 1984). Functional feeding groups are analogous to "guilds". Many stoneflies that are in the functional feeding group 'shredders' have short life cycles and may be present or absent from samples collected at certain times of the year (Gustafson 1990). Gustafson (1990) reported that predators, shredders, and collectors generally increased in density as stream order increased, while the density of scrapers peaked in the mid-order streams in the Gallatin River drainage. Scrapers had a high variance at all stream orders and headwater streams had more species of shredders than streams of intermediate size (Gustafson 1990).

At present, the Montana WQD RBP's for the mountain stream ecoregion do not fully address within-site, within-stream, seasonal, and yearly variability of macroinvertebrate communities, although variability is taken into account by setting wide confidence limits on impairment (Bukantis 1996). The Montana WQD RBP's also do not incorporate the potential beneficial uses of seasonal variability into their assessments. For example, the higher abundance of certain taxa at different seasons, other than summer, that may be more sensitive to various sources of impairment.

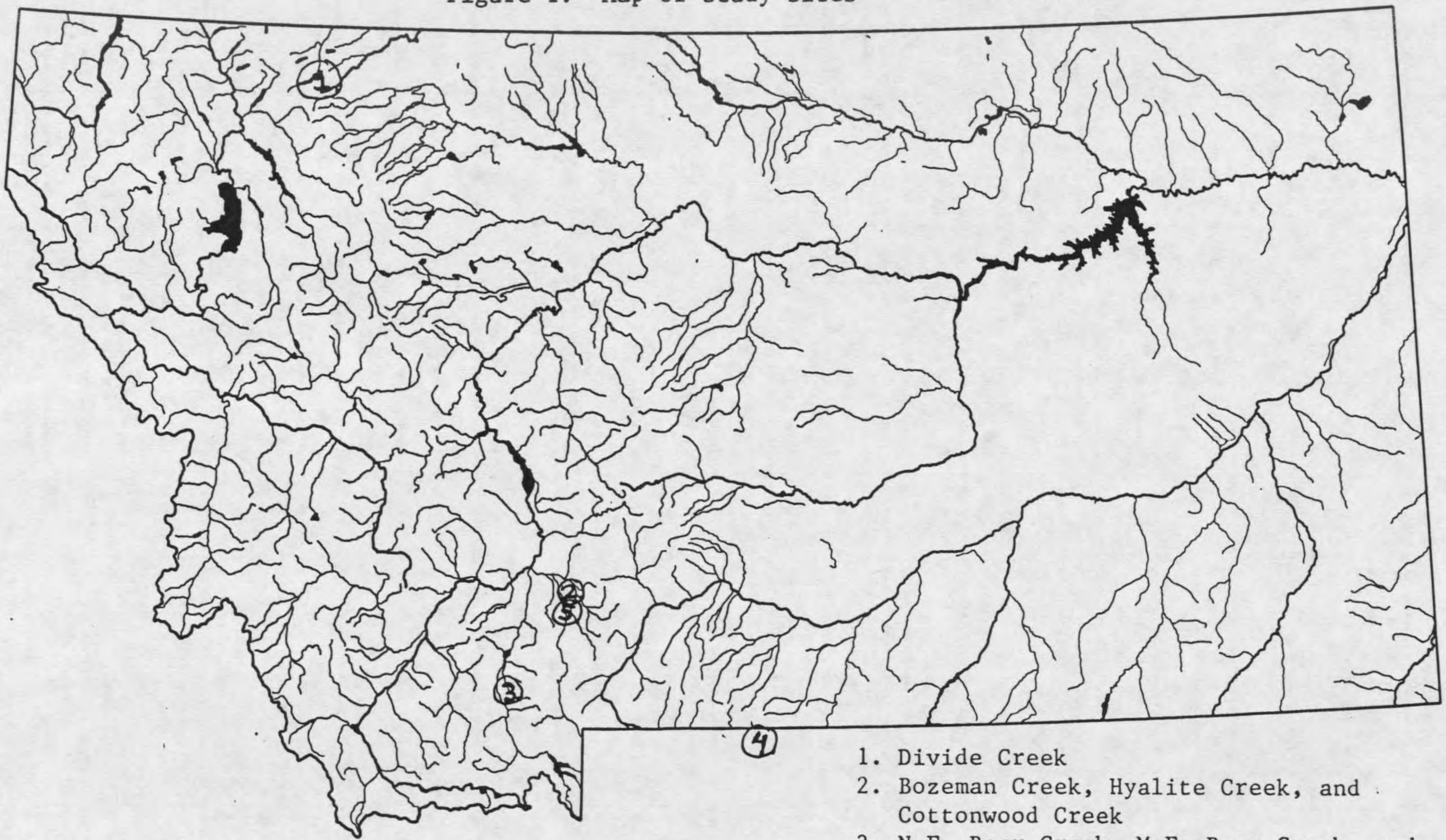
MATERIALS AND METHODS

Within-site Variability Collection Sites

Twenty six sites from ten mountain streams in western Montana and NW Wyoming were used to test within-site variability. The ten mountain streams were Bozeman Creek, Hyalite Creek, Cottonwood Creek, Squaw Creek, Pebble Creek, Soda Butte Creek, North Fork Bear Creek, Middle Fork Bear Creek, Bear Creek, and Divide Creek (Figure 1). A riffle sequence was defined as a series of consecutive riffles and pools. Three macroinvertebrate samples were collected from each riffle sequence (site), one sample per riffle, in 1995. Hyalite Creek, 1994 and Bozeman Creek, August, 9, 1994 had four samples collected from a riffle sequence (1 sample per riffle). Collection dates, physical and chemical data, and descriptions of the streams are presented in Table 2.

Of these 26 sites (riffle sequence), 14 were from 7 'least impaired' reference mountain streams. The seven reference streams included 5 sites (riffle sequences) from Pebble Creek, (1995) and 1 site from each of the following streams, Cottonwood Creek (1995), N. F. Bear Creek (1995), M. F. Bear Creek (1995), Bear Creek (1995), Soda Butte Creek (km 1, 1995), Bozeman Creek (summer 1994) and Bozeman Creek (summer 1995)(Table 2). Mean, variance, standard deviation (s.d.), coeffiecient of variation (CV) and 95% confidence intervals (CI) were derived for each metric used.

Figure 1. Map of study sites



1. Divide Creek
2. Bozeman Creek, Hyalite Creek, and Cottonwood Creek
3. N.F. Bear Creek, M.F. Bear Creek, and Bear Creek
4. Pebble Creek and Soda Butte Creek
5. Squaw Creek

Table 2. Descriptions of the 10 mountain streams sampled in this study including, mountain range, drainage, longitude, latitiude, township, range 1/4 section, stream order, link magnitude, elevation direction of stream flow, pH, conductivity, Rosgen stream type, estimated discharge, and date of collection.

	Bozeman Creek	**N. F. Bear Creek**	**M. F. Bear Creek**	**Bear Creek**	**Divide Creek**	**Soda Butte Creek**	**Cotton-wood Creek**	**Squaw Creek**	**Hyalite Creek**
Mountain range	Gallatin range	Madison range	Madison range	Madison range	Lewis range	Absaroka range	Gallatin range	Gallatin range	Gallatin range
drainage	Gallatin River	Madison River	Madison River	Madison River	St. Mary River	Lamar River (Yellow-stone River)	Gallatin River	Gallatin River	Gallatin River
longitude	111^{0} 1' W	111^{0}33'W	111^{0} 33' W	111^{0} 33' W	113^{0} 25' W	110^{0} 00' W	111^{0} 05'	111^{0} 13'	111^{0} 02'
latitude	45^{0} 35' N	45^{0} 10' N	45^{0} 10' N	45^{0} 09' N	48^{0} 45' N	45^{0} 00' N	45^{0} 34'	45^{0} 25'	45^{0} 33'
township range 1/4 section	T3S/R6E Sec.18 NE	T7S/R1E sec. 36 SW	T8S/R1E sec. 1 NW	T8S/R1E sec. 12 NW	T35N/R14 W sec. 34 SW	T9S/R15E sec 29	T3S/R5E sec 35	T4S/R4E sec 35	T3S/R5E sec 32
stream order	3	3	3	3	3	2, 3	3rd	3rd	3rd
link mag-nitude	34	14	17	9	18	2, 8	9	32	32
elevation (meters)	1627	1948	1887	1932	1380	2370, 2332	1744	1634	1990
direction of stream flow	NW	WSW	WSW	W	NW	W	NW	WNW	N
pH	8.5	8.4	8.4	8.4	8.6	8.6, 7.3	8.6	8.4	8.4
conduc-tivity	160	110	140	210	180	210, 300	170	180	120

Table 2. Continued.

	Bozeman Creek	N. F. Bear Creek	M. F. Bear Creek	Bear Creek	Divide Creek	Soda Butte Creek	Cotton-wood Creek	Squaw Creek	Hyalite Creek
Rosgen stream type	C4b	A2/A3b	B3c	B3/B4	B3/B4	B3, B3	C4	C4	B2c
estimated discharge (m^3/s)	na	2.84	1.48	0.35	na	na	na	na	na
date sampled	Sept 20, 1995	July 5, 1995	July 5, 1995	July 5, 1995	August 30, 1995	August 16, 1995	July 15, 1995	July 12, 1995	June 27, 1994