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Evaluation of Rangeland Stream Condition and Recovery using Physical and Biological Assessments of Nonpoint Source Pollution

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Evaluation of Rangeland Stream Condition and Recovery using Physical and Biological Assessments of Nonpoint Source Pollution

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ABSTRACT

Livestock grazing is the most common land use in the western United States, and has caused widespread degradation of water quality as a result of impacts to stream and riparian ecosystems. The problem is especially severe in the arid Great Basin, where stream channels and associated vegetation occupy small areas and carry low runoff, but are exposed to intense grazing pressure due to congregation of livestock in areas of shade and water supply. The federal Clean Water Act requires states to assess nonpoint source water pollution, including that resulting from livestock grazing.

Water quality has traditionally been evaluated using water chemistry criteria (e.g. for point source pollution), but these measures may not be effective in detecting nonpoint source water quality problems typically caused by livestock grazing. More responsive monitoring parameters for rangeland streams may include physical characteristics of stream channels, and biological measures of the characteristics of aquatic invertebrate communities and fishes. The paucity of relevant scientific information on how grazing impacts can best be detected and ameliorated, however, has hindered efforts to improve water quality and recover stream and riparian ecosystems damaged by livestock grazing.

Our research had two major objectives. These were (1) to compare the effectiveness of different monitoring techniques in detecting livestock impacts to stream ecosystems and water quality (short-term objective); and (2) to compare the recovery of damaged streams under different livestock grazing practices (long-term objective). We addressed both of these objectives through a two year research program designed to quantify stream channel characteristics and attributes of aquatic invertebrate and fish populations on rangeland streams flowing through pastures managed under different grazing practices, including corridor fencing, rest-rotation grazing, and season-long grazing. The results presented here from an eastern Sierra Nevada Great Basin watershed address our short-term goal of comparing physical and biological monitoring

approaches, and provide a baseline for our long-term goal of evaluating stream recovery under varied grazing practices.

Physical habitat quality was calculated from composite indices of bank, channel, and substrate characteristics, and was highest in the ungrazed stream reach. The Habitat Quality Index calculated for each study reach was highly correlated between different seasons and years, and was also highly correlated with trout biomass. Correlations between the Benthic Habitat Quality Index and a multiple metric Benthic Invertebrate Community Index were high for spring surveys (before the grazing season), but low for fall surveys (at the end of the grazing season), possibly reflecting short term grazing-induced changes to the benthic invertebrate communities not detected by the stream channel monitoring. We conclude that measurements of stream channel characteristics, fish population structure, and invertebrate community composition generally provide similar information about the condition of particular stream reaches, although stream reach characterization based on fish populations and stream channel attributes were more similar to each other than those based on benthic invertebrate communities. However, while the monitoring of stream channel characteristics and fish populations will likely provide reliable information on long term trends in stream condition, benthic invertebrate community composition may provide an indication of short term grazing-related changes in stream condition and water quality that are not detectable by monitoring programs that focus solely on stream channel characteristics or fish populations. Separating natural variability in invertebrate community composition from that attributable to grazing will require long-term monitoring to develop reliable indicators. Both physical and biological measures further indicate that while impacts are evident on most grazed reaches, others appear to be unimpaired during some periods.

Key words: aquatic invertebrates, benthos, bioindicators and biomonitoring, Sierra Bioregion, fish ecology, grazing and pasture management, rangelands and range use and management, streams and stream dynamics, water quality monitoring.

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INTRODUCTION

Livestock grazing is the most widespread land use in the Intermountain Region of the western United States. Of the 288 million acres of public land administered by the U.S. Forest Service and Bureau of Land Management in the western states, over 90% is presently under use as rangeland (Armour et al. 1994). The sensitive and productive riparian zones along small streams of the Great Basin are particularly vulnerable to impacts caused by livestock, and grazing has severely impacted stream and riparian habitats in this region (Minshall et al. 1989). As a result, few pristine stream ecosystems remain, wildlife using riparian areas has been reduced or eliminated (Thomas et al. 1977), and physical habitat conditions are generally poor (Armour et al. 1994). In many areas, overgrazing has eliminated bankside and riparian vegetative cover (Kauffman et al. 1983a; Platts and Nelson 1985a; Odion et al. 1988), producing increased soil erosion (Kauffman et al. 1983b; Gamougoun et al. 1984; Marlow et al. 1987), sedimentation, organic pollution, channel incision (Odion et al. 1988), and fluctuations in stream temperature (see Kauffman and Krueger 1984, for a review). These increases in nonpoint source pollution and loss of channel stability have resulted in deteriorating water quality and diminished ability of these ecosystems to maintain healthy aquatic communities, including fisheries (Keller and Burnham 1982; Platts 1991). Rangeland production is also often lost as incised streams drain upland water tables and limit the growth of forage plants (Odion et al. 1988).

In an attempt to reduce impacts to water quality caused by nonpoint source pollution, including that resulting from livestock grazing, Section 319 of the Clean Water Act required that states assess nonpoint water pollution sources and develop Best Management Practices to improve water quality. The U.S. Environmental Protection Agency (EPA) has further recommended that states develop biological criteria through resource inventory, and identify reference areas to which waterbodies may be compared for impact assessment (Gibson 1991; USEPA 1991). The paucity of relevant scientific information on how nonpoint source pollution from livestock grazing and can best be detected and ameliorated, however, has hindered efforts to improve water quality and stream and riparian ecosystems damaged by livestock grazing.

Water quality has traditionally been evaluated using measures of water chemistry (e.g., for point source pollution). These measures, however, are unlikely to be effective in detecting non-point source water quality problems resulting from livestock grazing (MacDonald et al. 1991). Instead, researchers have suggested that nonpoint source water quality problems and stream and riparian conditions might be more effectively monitored using biologically-based parameters. These include physical characteristics of stream channels and riparian vegetation, and populations of aquatic invertebrates and fishes (Platts et al. 1983; Plafkin et al. 1989; MacDonald et al. 1991). While physical channel structure and riparian vegetation have become widely used monitoring parameters for the detection of grazing impacts on aquatic ecosystems, such measures provide only an indirect indication of the health of the in-stream ecosystem. In contrast, stream insects and fish integrate the range of chemical, physical, and biological conditions to which they are exposed, and may therefore provide a direct indication of stream health and water quality. Aquatic invertebrates and fish are known to be sensitive to a variety of grazing-related impacts, including sedimentation, changes in riparian vegetation, bank cover, substrate size, and temperature regime (Culp et al. 1986; Newcombe and MacDonald 1991; Platts 1991). The US Forest Service Region 10 monitoring guidelines (MacDonald et al. 1991) and the recent EPA manual on monitoring livestock grazing (Bauer and Burton 1993) suggest that invertebrate and fish bioassessment may be among the most sensitive and responsive parameters for gauging the impacts associated with nonpoint source pollution caused by grazing, logging, road building, and mining. In addition, monitoring stream invertebrate and fish communities may be a more rapid and cost-effective means of determining livestock grazing impacts to water quality and stream and riparian ecosystems (Plafkin, et al. 1989).

Despite the potential of these various monitoring parameters, little research has focused on the relative effectiveness of stream channel characteristics and characteristics of the stream invertebrate and fish communities in detecting impacts to water quality from nonpoint source pollution, including that caused by livestock grazing. Such an assessment could dramatically increase the effectiveness of nonpoint source pollution monitoring programs.

In addition to the need to ascertain the relative effectiveness of various monitoring techniques, there is a pressing need for information on how nonpoint source pollution resulting from livestock grazing can be reduced. Several studies have evaluated the impact of various grazing management strategies on stream ecosystems and water quality, but these studies have generally evaluated only a single grazing technique (Keller and Burnham 1982; Kauffman et al. 1983a; Platts and Nelson 1985b). The simultaneous evaluation of several grazing management techniques (e.g., corridor fencing, rest-rotation grazing, grazing subject to strict utilization standards) would provide critical information for the design of projects aimed at reducing nonpoint source pollution caused by livestock grazing.

Research Objectives

Our proposed research had two objectives. These were (1) to compare the effectiveness of different monitoring techniques in detecting livestock impacts to stream and riparian ecosystems and water quality; and (2) to compare the recovery of stream and riparian ecosystems under several different livestock grazing practices. We addressed both of these questions by conducting a two year research program to quantify stream channel characteristics and attributes of aquatic invertebrate and fish populations on rangeland streams flowing through pastures exposed to different livestock grazing practices, including corridor fencing, rest-rotation grazing, and restricted utilization grazing.

METHODS

Our research was conducted within the Long Valley watershed, a watershed influenced to varying extents by grazing. We characterized physical and biological characteristics of stream channels and aquatic and riparian habitats at nine study sites (four on Convict Creek, three on Mammoth Creek, and two on McGee Creek) on which new grazing practices are being established. These practices included newly fenced exclosures, rest-rotation pastures, 35% forage utilization pastures, and traditional season-long grazing management. A reference reach that has been ungrazed for more than 50 years was also monitored. These different management strategies formed the treatments of a natural experiment.

Long Valley is a 7,000' high desert watershed located near the town of Mammoth Lakes in the eastern Sierra Nevada (Figure 1). Most of the land is owned by the Los Angeles Department of Water and Power (LADWP) and the U.S. Forest Service, and has been grazed by cattle to varying extents for at least the past century. Convict, McGee, and Mammoth Creeks flow through our study area. In an attempt to reduce impacts to these streams resulting from season-long grazing, LADWP and the grazing permittees instituted rest-rotation grazing and fenced exclosures on rangeland surrounding Convict and McGee Creeks in 1992, and on Mammoth Creek in 1994 (refer to Figure 1 and Table 1 for a summary of study reach treatments). In addition, cattle have been excluded from portions of Convict Creek (on a University of California Natural Reserve) for about 50 years.

<u>Stream Channel Characteristics</u>

Stream channel characteristics are commonly used to evaluate impacts of surrounding land uses on stream ecosystems. These characteristics are particularly useful because their temporal variability is low in the absence of human-caused disturbances, and they respond relatively rapidly to surrounding land management practices. Since impacts of livestock grazing on stream channels are well-documented (e.g., Kauffman and Krueger 1984), monitoring of key channel characteristics may allow direct links to be made between observed changes in channel conditions and the quality of important stream resources (e.g., coldwater fisheries). The channel characteristics we measured are widely used to assess livestock grazing effects on streams (e.g., Platts et al. 1983; Platts et al. 1987; MacDonald et al. 1991; Bauer and Burton 1993).

Stream channel characteristics were measured along a 125 m stream reach at each site. Monitoring was conducted in October 1993, April and October 1994, and May 1995. Channel characteristics of each reach were collected along 25 transects oriented perpendicular to stream flow and spaced 5 m apart. The following parameters were measured along each transect: channel width, channel depth, bankfull height, stream width, bank angle, bank overhang, and bank water depth. In addition, stream canopy cover was measured at each bank and in midstream. We defined "channel width" as the width of the active stream channel and was typically delineated by scour lines, vegetation limits, and bank slopes. This was intended as a measure of the extent of recent lateral channel migration, with unstable channels typically being wider than stable channels. "Channel depth" was measured as the distance from the top of the channel to the water surface, and provided a measure of channel incision. We defined "bankfull height" as the height above the water surface at which the channel loses its ability to contain the stream (Gordon et al. 1992). This provided a measure of the degree to which banks confine the stream, since streams with destabilized banks typically have lower bankfull heights than streams with stable banks. "Stream width" was defined as the width of water across each transect. "Bank angle" was the angle of the bank relative to the water surface and was measured with a clinometer. "Bank overhang" was the amount of bank hanging over the stream. "Bank water depth" was the water depth at the stream bank. Bank angle, bank overhang, and bank water depth all provide information on bank stability and on the amount of streamside cover for resident fishes. "Stream canopy cover" was measured as percent shading with a densiometer.

The following parameters were measured at 10 equally-spaced points along each transect: water depth, water velocity, substrate size, substrate embeddedness, depth of detritus, and depth of aquatic vegetation. Water depth at each point was measured with a top-set wading rod. After recording the water depth at a particular point, we measured water velocity with an electromagnetic current meter at four-tenths of the water depth (as measured upwards from the streambed). After measurement of depth and current velocity at a point, the substrate particle size (sieve diameter) contacted by the front of the wading rod was measured. Embeddedness of each particle was estimated as the percent of the particle buried by fine (< 1 mm diameter) sediment for particles larger than 10 mm in diameter. The depth of detritus and/or aquatic vegetation present at a sample point was also measured. The stream gradient of each reach was measured using a surveyor's level and staff. Sinuosity was calculated as the ratio of the channel distance to straight line distance over each reach.

Fish Population Monitoring

Concern over the impact of livestock grazing on resident salmonid populations has been a major stimulus for long-term, detailed studies of the effects of rangeland management on streams (Meehan 1991). Salmonid populations can provide an excellent tool for monitoring stream habitat quality and water quality for several reasons (MacDonald et al. 1991): (1) their relatively long life span allows them to indicate broad-scale and long-term habitat conditions; (2) their higher trophic position allows them to be used as an integrator of changes in the lower trophic levels; (3) they are relatively easy to collect and identify in the field; and (4) the habitat requirements of many species are relatively well-known (Plafkin et al. 1989).

We censused fish populations along the same 125 m stream reaches on which stream channel monitoring was conducted. Fish populations were censused in 7 of the 9 study stream reaches in October 1993 and all 9 study stream reaches in October 1994. We were unable to census fish populations in the two McGee Creek sites during 1993 because of an early leaf fall. We estimated the population density, biomass, and size structure of all fish species present at each site using an electrofisher and standard depletion techniques (Platts et al. 1983). We partitioned each site into five 25 m sections with block seines and removed fish from each section during three or more passes using a Smith-Root Mark 12 electrofisher. Captured fish were identified by species, measured to the nearest millimeter (fork length), and weighed to the nearest 0.1 g. All fish were returned alive to the stream. The population size in each section was estimated using maximum-likelihood techniques (MicroFish v. 3.0 software).

Aquatic Macroinvertebrate Bioassessment

Aquatic macroinvertebrates, especially insects, are central to the function of stream ecosystems. Occupying diverse roles in the food chain, these organisms consume organic matter and algae, and in turn provide the primary food source to fish populations. The structure of the invertebrate community reflects the habitat resources, water quality, and biological integrity of the stream. Invertebrate species also have different levels of tolerance for pollution that may be used to produce a composite indicator of habitat and water quality (biotic index: Hilsenhoff 1987; Plafkin et al. 1989). Since aquatic invertebrates in effect sample the chemical and physical environment they live in, they integrate the effects of various pollution sources over space and time. The composition of the invertebrate community often changes in response to degradation or improvement of conditions, thereby providing a sensitive measure of habitat quality. The utility of the technique has been demonstrated in upstream/downstream measures of point source pollution (USEPA 1991) and is currently the subject of extensive review and evaluation (Rosenberg and Resh 1993; Loeb and Spacie 1994). In addition, the EPA is in the process of developing biological criteria for water quality standards (USEPA 1990).

We collected samples of macroinvertebrate benthos from within the same 125 m stream reaches at which stream channel and fish population monitoring was conducted, and sampling was performed concurrently with stream channel monitoring during each of the four monitoring periods. Five replicate kick samples were collected from riffles in each reach using a 0.25 mm mesh net. Each kick sample consisted of a composite of three equal-effort collections from microhabitats across each riffle. Samples were preserved and stored in 80% ethanol until laboratory processing. Subsamples were processed until 100 to 500 individual organisms were counted and identified to the lowest taxonomic level possible (usually genus). Data on the relative abundance of benthic invertebrate taxa were then used to calculate various metrics of ecological structure, function, and stability (Plafkin et al. 1989; Barbour et al. 1992).

Structural measures of the community included diversity and evenness indices such as species richness (total taxa = S), Shannon Index (H'), Simpson diversity (D), equitability (D/S), and dominance (% contribution of the dominant taxon = Dom). Functional indicator measures included a modified Hilsenhoff Biotic Index (HBI, Hilsenhoff 1987) and the EPT index (total number of the generally sensitive Ephemeroptera, Plecoptera and Trichoptera taxa). One weakness of the HBI metric is the absence of adequate tolerance data for many western species of aquatic insects. Therefore, we adjusted some of the tolerance values based on natural history and unpublished sources (listed by species in Table 5). Other indicator group metrics included the proportion of relatively tolerant *Hydropsyche* within the Trichoptera (H/T) and *Baetis* within the Ephemeroptera (B/E), percent of the relatively tolerant Chironomidae (%C), and the ratio of EPT to Chironomidae (EPT/C). In addition to these measures, several indicator metrics specific to the local fauna were also developed. Tolerant members of the EPT taxa were identified based on natural history and adjusted tolerance values and included *Baetis* and *Tricorythodes* among the Ephemeroptera, *Isogenoides* and *Pteronarcella* among the Plecoptera, and *Hydropsyche*, *Oligoplectrum*, *Hydroptila*, *Agraylea*, and *Oxyethira* among the Trichoptera (= % tolerant EPT). The relative abundance of a high pollution-tolerant group consisting of leeches (Hirudinoidea), the amphipod *Hyallela*, the snail *Physa* and the fingernail clam *Pisidium* was summed to calculate the HHPP metric. To explicitly reflect some of the expected consequences of livestock grazing on functional feeding groups and energy flow in streams, two other metrics were created. Loss of riparian vegetation and opening of the canopy along with increased sedimentation should favor an increase in grazers (G) on periphyton (increased algae resulting from increased sunlight) and sediment-feeding and sediment-adapted collector-filterer (CF) guilds. Reduced inputs of leaf and wood litter should also result in a decline in the abundance of shredders (S). As grazing impacts increase, the proportion of G+CF to the total should increase (relative to reference conditions) while the ratio of S/G+CF should decrease. Except for dominance, all structural metrics would be expected to show positive correlations with increasing habitat quality, while all functional indicator measures (with the exception of the EPT/C ratio and Shredder to Grazer+Filterer ratio) should show negative correlations with habitat quality.

Analyses

In order to provide integrated measures of physical habitat quality, we developed several composite indices. This approach is similar to the physical habitat assessment scores of Platts et al. (1983) but is based on the ranking of direct measures of physical features rather than a subjective visual assessment. The "Habitat Quality Index" (HQI) we used integrated nine stream channel characteristics, including bank, channel, and substrate features known to provide important habitat to trout. HQI was calculated by first ranking the range of values for each parameter from one to four and then summing these ranks (Table 2). The HQI was used to assess differences in stream condition between study reaches over the three seasons and two years of the study, and for correlation analyses with trout population data.

To evaluate the relationship between benthic invertebrate communities and physical habitat, we developed the "Benthic Habitat Quality Index" (BHQI) that integrated six benthic habitat and substrate features. We developed this second index of stream condition because stream invertebrates inhabit the benthic environment and are more likely to be influenced by substrate characteristics than by channel morphology. Therefore, the BHQI incorporates parameters that emphasize substrate features, including measures of substrate size, embeddedness, % fines (<1 mm), sum of detrital organic matter deposits and vegetative cover, shading, and stream width to depth ratios. We calculated the BHQI as the average of standardized scores for these parameters (Table 3). Scores were standardized for each parameter within each sample period by assigning 0 to the lowest value and 1 to the highest value among the nine study sites. Values between the lowest and highest were scored in relation to this scale and then averaged over all six parameters to compute a BHQI between 0 and 1.

Standardized scores were also used to compute a composite index of 10-15 metrics of invertebrate community structure and function (Table 3). Such a multi-metric approach has been used in many biological monitoring programs (e.g., Plafkin et al. 1989; Barbour et al. 1995) and may provide a more accurate assessment of the condition of stream habitats than what is possible using individual metrics. The "Benthic Invertebrate Community Index" (BICI) we used included both widely-used metrics and several we developed specifically for detection of grazing impacts and for the local fauna (e.g., % tolerant EPT). We conducted correlation and regression analyses with individual metrics as well as with the standardized index (BICI).

RESULTS

The short-term goal of this research was to examine the correspondence between physical and biological measures of stream habitat quality. Since complete data for all sites was available only for Fall 1993, Spring 1994, and Fall 1994, our interpretations are limited to the conditions during these periods. In addition, because newly-instituted grazing practices have had little or no time to produce environmental or ecological changes, our comparisons of grazing treatments are limited to contrasts among grazed sites having varied levels of physical habitat quality as defined by our measures. Continued long-term studies that compare stream condition to the baseline conditions described here will provide the data needed to discriminate patterns of recovery in stream reaches under different grazing practices.

The physical habitat characteristics that we measured showed consistent differences between the study reaches, and these differences were related to grazing intensity. HQI's for individual reaches were highly repeatable, as demonstrated by the significant correlations between seasons and between years (Figure 2). Correlations were strongest when comparing data collected during the same season but in different years (r=0.91), lower for comparisons for HQI's from different seasons in the same year (r=0.86), and lowest for HQI's from different seasons and different years (r=0.82). Of the nine study reaches, the ungrazed control reach (SNARL) always had the highest HQI score (Figure 3), indicating that it had the highest habitat quality. The rankings of the HQI's for grazed stream reaches were also very stable over the study period (Figure 3). In addition, the HQI was highly correlated with several fish population characteristics, including total trout biomass, total brown trout biomass, and total adult trout biomass (Figure 4). The strength of these correlations indicates that monitoring of both stream channel features and fish populations would result in similar assessments of stream condition.

The benthic habitat quality index (BHQI) also showed highest values associated with the ungrazed SNARL reference site and an overall significant correlation ($r=0.48$, $p<.01$) with the composite benthic invertebrate community index for pooled data over all seasons (BICI, Figure 5). Correlations within seasons were also positive but significant only for Spring 1994 ($r=0.88$, $p<.005$; Figure 6). The correlation matrix for the BHQI with the individual invertebrate community metrics comprising the BICI is given in Table 4 and shows that the 10 metrics combined to yield a BICI with the highest correlation included species richness (S), EPT taxa index, Simpson diversity (D), Shannon diversity (H'), modified Hilsenhoff biotic index (HBI), pollution-indicator taxa (HHPP), percent Chironomidae (%C), percent EPT numbers comprised of tolerant local taxa (%tol.EPT), ratio of shredders to grazers + collector-filterers (S/G+CF), and percent grazers + collector-filterers of the total (G+CF/T). The diversity indices tended to show high positive correlations with the BHQI in the spring sample, while indicator taxa groups often produced high negative correlations in the fall though were not consistent.

Seasonal differences appeared to have marked influences on the macroinvertebrate community. Most BICI values were at or below 0.5 during the fall periods while most were above 0.5 in the spring (except the SNARL reach which was near or above 0.6 during all sample periods; Figure 5 and 6). This was particularly evident on Convict Creek where grazed sites with spring BICI values of near 0.8 were usually below 0.4 in fall (Figure 6). While reaches with the highest and lowest benthic habitat quality (BHQI) consistently had among the best or worst BICI ratings, respectively, sites of intermediate habitat quality showed variable BICI values during the fall sampling periods. Although cumulative downstream degradation was evident on Mammoth Creek (poorer conditions downstream by all measures), McGee Creek (two sites only) showed this pattern in the fall but a reversal in the spring (associated with algal cover on the upper reach), and Convict Creek sites showed no consistent gradient during any sample period.

in channel adjustment to grazing exclosure has been noted in the nearby White Mountains of California (Kondolf 1993). Though some features such as embedded sediments may reflect the cumulative effects of erosion from problems such as overgrazing, some impacts may be episodic in nature and not be reflected in any of the physical habitat features we measured. By contrast, stream invertebrate community metrics showed distinctive shifts between spring and fall sample periods. The BICI correlated well with physical conditions in the spring before grazing had occurred, but became poorly correlated in the fall after the grazing season. Positive correlations with diversity measures in the spring may be indicative of a recovery phase while negative correlations with pollution-indicator metrics in fall suggest deleterious impacts have occurred at this time. The absence of significant correlations with physical habitat in fall may be related to degraded biotic conditions occurring without similar changes in the more stable (unresponsive) physical features. During the fall some of the grazed Convict Creek sites appeared to be over-rated in terms of physical habitat quality while some sites on Mammoth and McGee were under-rated. This suggests that biological conditions on some sites may not attain the potential habitat quality in fall after grazing has occurred while others may actually be more healthy than is apparent. Though sites with BICI values in the same range as those observed on the SNARL reach may for that season have attained the ecological status of the reference, no grazed site remained within this range in all seasons compared. Long-term monitoring will be required to determine the extent to which seasonal changes represent natural variability or whether they are in fact indicative of grazing-related alterations in stream condition and water quality.

One of the difficulties of evaluating the effectiveness of the benthic invertebrate metrics in detecting impacts resulting from livestock grazing is the lack of information of how grazing might be expected to influence invertebrate community structure. Although numerous studies have documented the effects of livestock grazing on stream channel characteristics and fish populations (see Introduction for a review), few studies of invertebrates in grazed and ungrazed rangeland streams have been published. Though grazing impacts were detected in one study in New Mexico, these could not be separated from natural longitudinal changes on downstream grazed areas (Rinne 1988). This study did not, however, utilize a multiple metric approach or include baseline data and treatment replication. Grazed eastern Oregon streams with open riparian canopy showed increased invertebrate biomass attributable to the large algal-grazing caddisfly *Dicosmoecus*, and decreased salmonid abundance (Tait et al. 1994). Changes in indicator assemblage metrics or structural diversity were not reported in this study either. Studies of sedimentation from surrounding land use practices have shown losses in the benthic fauna resulting from impacts to benthic habitat (Lenat, et al. 1981; Lemly 1982; Lenat 1984). Experimental studies of the effects of suspended sediments have produced results suggesting acute responses such as increased drift (Rosenberg and Wiens 1978), and mortality resulting from chronic exposure (Brusven and Hornig 1984), although recovery through recolonization may be rapid.

Another difficulty in evaluating the effectiveness of benthic invertebrate metrics in detecting impacts caused by livestock grazing is the lack of ungrazed sites for each stream in our study against which to compare the biological condition of grazed sites. Differences in species pools and potential colonization sources over the short-term suggest the need for ungrazed reference reaches from each stream and a wider spectrum of habitat conditions overall to form a more reliable basis for regression analyses. With additional reference reaches it would also be possible to calculate similarity indices which form an important basis for bioassessment analyses. Similarity indices typically used in bioassessment analyses, such as the Community Loss Index or Jaccard Index, were not used in this study because we could not justify such comparisons based on the single reference reach at SNARL. Our data were also limited in that the analysis in this report was based on only three seasons over two years. With longer-term monitoring of stream channel and biological characteristics of our study reaches, we will be able to better discern the reasons underlying the fluctuations in invertebrate community structure observed in this two year study. Longer-term monitoring will also allow us to improve the composite indexes used in this study (HQI, BHQI, and BICI). In calculating our indexes, we assumed that relationships between habitat quality and biological parameters were linear. It may be more reasonable, however, to expect a sigmoid response (e.g. Plafkin, et al. 1989) where below or above certain thresholds further change in habitat quality has no effect on the biological response. More data over a wider range of conditions is needed to determine whether it would be more appropriate to use non-linear regression and differential weighting of physical habitat measures.

Further research on the rangeland streams under study is planned. In addition to expanding the number of grazed and ungrazed stream reaches, we also plan to incorporate more detailed descriptions of the physical environment. Temperature is known to have a controlling influence on rates of growth, survival and mortality, especially among poikilotherms such as insects and fish. For example, increased solar radiation resulting from a grazing-caused reduction in riparian vegetation has been shown to have a substantial negative impact on salmonids by causing increases in temperature to lethal levels and by reducing the amount of suitable habitat (Li, et al. 1994). We plan to set up hourly temperature-logging devices for daily and seasonal data at all our study sites. We also plan to quantify invertebrate food resources using core samples to determine coarse and fine particulate organic matter availability (CPOM/FPOM), chlorophyll density on rock surfaces (algal biomass and types), and vegetation cover along stream transects. Measurements of dissolved oxygen and substrate sulfide levels will also be made to evaluate potential limiting anoxic conditions.

Although the streams of Long Valley examined in this study are typical of many Great Basin streams, other regions of the Great Basin should be examined to develop a more accurate representation of the potential fauna and range of habitat conditions. Further studies will incorporate additional grazed and ungrazed watersheds in the western Great Basin.

SUMMARY

Our research had two major objectives. These were (1) to compare the effectiveness of different monitoring techniques in detecting livestock impacts to stream ecosystems and water quality (short-term objective); and (2) to compare the recovery of the physical environment and ecological integrity of streams under different livestock grazing practices (long-term objective). We addressed both of these objectives through a two year research program designed to quantify stream channel characteristics and attributes of aquatic invertebrate and fish populations on rangeland streams flowing through pastures managed under different grazing practices, including corridor fencing, rest-rotation grazing, and season-long grazing. The results presented here from an eastern Sierra Nevada Great Basin watershed address our short-term goal of comparing physical and biological monitoring approaches, and provide a baseline for our long-term goal of evaluating stream recovery under varied grazing practices.

Indices of habitat quality were calculated from composite measures of bank, channel, and substrate characteristics, and were highest in the ungrazed stream reach. The Habitat Quality Index calculated for each study reach was highly correlated between different seasons and years, and was also highly correlated with trout biomass. Correlations between the Benthic Habitat Quality Index and the multiple metric Benthic Invertebrate Community Index were high for spring surveys (before the grazing season), but low for fall surveys (at the end of the grazing season), possibly reflecting short term grazing-induced changes to the benthic invertebrate communities not detected by the stream channel monitoring.

We conclude that measurements of stream channel characteristics, fish population structure, and invertebrate community composition generally provide similar information about the condition of particular stream reaches, although stream reach characterization based on fish populations and stream channel characteristics were more similar to each other than those based on benthic invertebrate communities. However, while the

monitoring of stream channel characteristics and fish populations will likely provide useful information on long term trends in stream condition, benthic invertebrate community composition may provide an indication of grazing season changes in stream condition and water quality that are not detectable by monitoring programs that focus solely on stream channel characteristics or fish populations.

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Table 1. New livestock grazing practices on the Long Valley study reaches

	GRAZING MANAGEMENT PRACTICE			
Stream (Reach Name)	**Season-long** or traditional use	**Rest-rotation (1 year on / 2 off)**	**35% Utilization**	**Exclosure** (or **Ungrazed)**
Convict (SNARL)				**Reference**
Convict (Church)				
Convict (Upper)				
Convict (Middle)				
McGee (S. swamp)				
McGee (W. Convict)				
Mammoth (Front)				
Mammoth (Back)				
Mammoth (Spring)				

Note: Conditions measured for all sites in this study were under season-long grazing (except SNARL reference areas). These changes in grazing practices have only recently occurred so the data presented in this report (1993-94) represents a baseline for future comparison of changes related to the new grazing practices shown in this table.

Table 2. Values given to parameter ranges for nine physical habitat variables used to construct the Habitat Quality Index (HQI).

Bank angle (degrees)		**Bankfull height (cm)**		**Embeddedness (%)**	
Range	Value	Range	Value	Range	Value
121-140	1	5-11.25	1	21-25+	1
101-120	2	11.26-17.5	2	16-20	2
81-100	3	17.51-23.75	3	11-15	3
61-80	4	23.76-30	4	5-10	4
Bank overhang (cm)		**Channel width:depth**		**Percent fines (% < 1 mm)**	
Range	Value	Range	Value	Range	Value
0-4.5	1	18.6-22	1	16-20+	1
4.6-9	2	15.1-18.5	2	11-15	2
9.1-13.5	3	11.6-15	3	6-10	3
13.6-18	4	8-11.5	4	0-5	4
Bank water depth (cm)		**Stream width:depth**		**Stream shading (%)**	
Range	Value	Range	Value	Range	Value
5-10	1	29.6-36.0	1	0-10	1
11-15	2	23.1-29.5	2	11-20	2
16-20	3	16.6-23.0	3	21-30	3
21-25+	4	10.0-16.5	4	31-40	4

Table 3. Variables used to calculate the Benthic Habitat Quality Index (BHQI) and the Benthic Invertebrate Communities Index (BICI)

<u>BHQI (each variable standardized as 0 to 1 from lowest to highest value, and averaged)</u>

Substrate size

Embeddedness

% Fines (<1mm)

Sum of detrital organic matter deposits and vegetative cover

Shading (densiometer readings of canopy cover)

Width to depth ratios of the wetted perimeter

<u>BICI (standardized/ averaged 10 metrics producing highest correlation with BHQI)</u>

Species Richness (S)

Shannon Diversity (H')

Simpson Diversity (D)

EPT Index (EPT)

modified Hilsenhoff Biotic Index (HBI)

High Pollution Tolerant Hirudinoidea, *Hyallela*, *Physa*, *Pisidium* (HHPP)

Percent Chironomidae (%C)

local Tolerant EPT (% tolerant EPT)

Ratio of Shredders to Grazers + Collector-Filterers (S/G+CF)

Percent Grazers + Collector-Filterers (%G+CF)

Table 4. Correlation matrix for the Benthic Habitat Quality Index and invertebrate metrics.

Type	Metric	R - Values All Seasons	Fall 1993	Spring 1994	Fall 1994
Structural	S (Richness)	.133	-.147	**.711**	.036
	H'(Shannon)	.286	.352	**.776**	-.021
	EPT taxa	.132	.173	.504	.036
	Dominance	-.215	-.347	-.615	.167
	D (Simpson)	.310	.396	**.745**	.392
	Equitability(D/S)	.213	.654	.392	-.126
Functional	HBI (biotic index)	-.225	-.130	-.386	-.442
	Hydropsyche/T	-.069	-.228	.591	-.351
	HHPP Index	**-.522**	**-.807**	**-.684**	-.347
	%Tolerant EPT	-.316	-.632	-.626	-.327
	EPT/Chironomidae	.157	-.166	-.019	.616
	%Chironomidae	-.245	.106	-.182	**-.670**
	Baetis/E	.011	.387	-.205	-.190
	S/G+CF	.328	-.262	**.913**	.326
	%G+CF	-.288	.218	**-.814**	-.496
Integrated	BICI - 15	**.442**	.382	**.807**	.377
	BICI - 13	**.438**	.406	**.825**	.313
	BICI - 11	**.462**	.507	**.816**	.301
	BICI - 11a	**.475**	.381	**.869**	.353
selected	**BICI - 10**	**.488**	.355	**.876**	.402
	BICI - 10a	**.468**	.438	**.836**	.330

Bold R-Values represent significant correlations at $p<.05$.

BICI - 15 uses all metrics above

BICI - 13 excludes *Hydropsyche*/T and EPT/C

BICI - 11 excludes *Hydropsyche*/T, EPT/C, *Baetis*/E and S/G+CF

BICI - 11a excludes *Hydropsyche*/T, EPT/C, *Baetis*/E and D/S

BICI - 10 excludes *Hydropsyche*/T, EPT/C, *Baetis*/E, D/S and Dominance

BICI - 10a excludes *Hydropsyche*/T, EPT/C, *Baetis*/E, D/S and S/G+CF

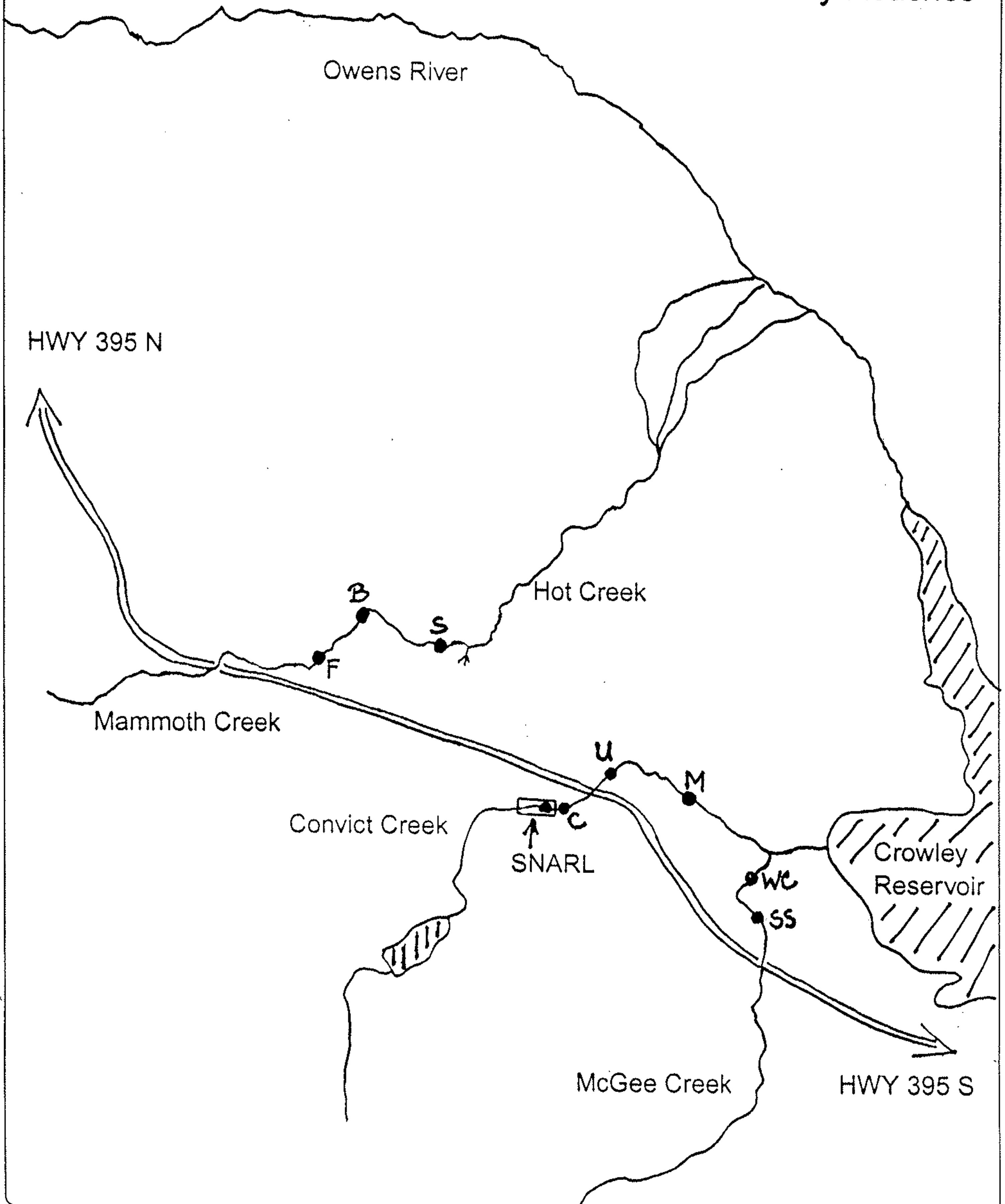
Table 5. Relative abundance and tolerance values of all invertebrates collected during 1993-94.

	CONVICT CREEK				McGEE CREEK		MAMMOTH CREEK			Tolerance
	SNARL	Church	Upper	Middle	SSwamp	WConvict	Front	Back	Spring	Values
Drunella flavilinea	0.019	0.013	0.042	0.042	0.006	0.025	0.039	0.079	0.005	1
Drunella spinifera	0.000	0.000	0.000	0.000	0.005	0.004	0.001	0.000	0.000	0
Drunella doddsi	0.000	0.000	0.000	0.000	0.012	0.004	0.000	0.000	0.000	0
Drunella grandis	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0
Ephemerella (infrequens)	0.001	0.001	0.002	0.000	0.002	0.000	0.017	0.007	0.000	1
Serratella (tibialis)	0.002	0.002	0.002	0.005	0.001	0.004	0.015	0.003	0.000	2
Caudatella hystrix	0.004	0.001	0.000	0.000	0.008	0.001	0.000	0.001	0.000	1
Cinygmula sp.	0.001	0.000	0.000	0.000	0.061	0.173	0.005	0.000	0.000	4
Rhithrogena sp.	0.000	0.000	0.000	0.000	0.005	0.005	0.003	0.002	0.000	0
Epeorus sp.	0.036	0.049	0.017	0.015	0.019	0.013	0.028	0.014	0.000	0
Ironodes sp.	0.016	0.009	0.004	0.001	0.004	0.000	0.007	0.004	0.000	4
Tricorythodes sp.	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.010	7
Ameletus sp.	0.004	0.001	0.001	0.000	0.001	0.005	0.001	0.001	0.003	0
Paraleptophlebia sp.	0.045	0.038	0.033	0.020	0.017	0.003	0.057	0.091	0.005	1
Baetis spp.	0.118	0.129	0.138	0.100	0.237	0.177	0.102	0.230	0.129	4
Doroneuria baumanni	0.005	0.004	0.007	0.011	0.049	0.019	0.003	0.000	0.000	1
Calineuria californica	0.001	0.001	0.002	0.000	0.000	0.000	0.001	0.000	0.000	1
Hesperoperla pacifica	0.000	0.001	0.000	0.000	0.016	0.000	0.000	0.000	0.000	1
Malenka (californica)	0.011	0.006	0.000	0.000	0.008	0.001	0.000	0.001	0.000	2
Zapada (columbiana group)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	2
Skwala (curvata)	0.000	0.000	0.000	0.003	0.006	0.016	0.000	0.000	0.000	2
Isogenoides (colubrinus)	0.000	0.000	0.000	0.000	0.000	0.000	0.016	0.079	0.005	3
Isoperla sp.	0.000	0.000	0.000	0.005	0.000	0.002	0.000	0.025	0.006	2
Sweltsa sp.	0.050	0.022	0.021	0.002	0.008	0.067	0.015	0.003	0.001	2
Pteronarcella sp.	0.001	0.000	0.000	0.000	0.000	0.026	0.003	0.009	0.000	1
Pteronarcys sp.	0.006	0.003	0.001	0.000	0.003	0.000	0.001	0.000	0.000	0
Yoraperla (brevis)	0.000	0.000	0.000	0.000	0.005	0.000	0.000	0.000	0.000	2
Capnia sp.	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.000	0.000	1
Brachycentrus (americanus)	0.002	0.005	0.002	0.020	0.014	0.022	0.000	0.009	0.001	1
Oligoplectrum echo	0.000	0.000	0.000	0.003	0.000	0.017	0.105	0.048	0.116	3
Micrasema sp.	0.008	0.009	0.003	0.002	0.002	0.000	0.001	0.000	0.000	1
Amiocentrus sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	1
Rhyacophila spp.	0.012	0.011	0.004	0.009	0.006	0.000	0.002	0.000	0.000	0
Lepidostoma (sand case) sp.	0.025	0.026	0.026	0.048	0.001	0.000	0.030	0.022	0.014	1
Lepidostoma (panel case) sp.	0.000	0.000	0.000	0.001	0.000	0.000	0.001	0.000	0.000	1
Lepidostoma (turret/chimney case) sp.	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	1
Wormaldia sp.	0.000	0.001	0.012	0.000	0.000	0.000	0.003	0.000	0.000	3
Dolophilodes sp.	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	3
Glossosoma sp.	0.003	0.003	0.010	0.046	0.009	0.005	0.000	0.001	0.000	0
Protoptila sp.	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	1
Neophylax sp.	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	3
Dicosmoecus (gilvipes)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	2
Onocosmoecus sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	1
Hydropsyche sp.	0.061	0.059	0.059	0.057	0.001	0.006	0.044	0.034	0.118	4
Arctopsyche grandis	0.008	0.016	0.010	0.011	0.089	0.016	0.004	0.000	0.000	2
Hydroptila sp.	0.000	0.001	0.003	0.001	0.001	0.000	0.000	0.000	0.005	6
Ochrotrichia sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	4
Agraylea saltesa	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	8
Oxyethira sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	8
Corisella (inscripta)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	8
Orohermes crepusculus	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0
Optioservus (divergens)	0.233	0.174	0.246	0.274	0.003	0.154	0.327	0.205	0.222	4
Optioservus (quadrimaculatus)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	4
Cleptelmis (addenda)	0.001	0.000	0.000	0.001	0.000	0.000	0.001	0.000	0.000	4
Zaitzevia (parvula)	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.002	0.000	4
Narpus (concolor)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	4
Lara (avara)	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	4
Helichus sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	5
Amphizoa sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0
Agabus sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	5
Tropisternus (lateralis)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	5

TABLE 5. (continued)

	SNARL	CONVICT CREEK Church	Upper	Middle	McGEE CREEK SSwamp	WConvict	MAMMOTH CREEK Front	Back	Spring	Tolerance Values
Paracymus sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	5
Antocha (monticola)	0.009	0.007	0.006	0.014	0.004	0.001	0.002	0.000	0.000	3
Hexatoma sp.	0.061	0.046	0.022	0.001	0.000	0.000	0.000	0.000	0.000	2
Limnophila sp.	0.007	0.002	0.003	0.000	0.000	0.001	0.000	0.000	0.000	3
Dicranota sp.	0.000	0.000	0.001	0.000	0.006	0.016	0.002	0.000	0.000	3
Pedicia sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	6
Hesperoconopa sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	1
Tipula (Yamatotipula) sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.003	0.008	6
Deuterophlebia nielsoni	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0
Agathon (comstocki)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0
Dixa sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	1
Chelifera sp.	0.000	0.000	0.001	0.002	0.002	0.011	0.003	0.000	0.000	6
Clinocera sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	6
Oreogeton sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	6
Pericoma sp.	0.008	0.012	0.006	0.000	0.000	0.000	0.001	0.001	0.001	4
Bezzia (or Palpomyia) sp.	0.003	0.003	0.003	0.000	0.000	0.000	0.000	0.000	0.000	6
Limnophora sp.	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.000	6
Simulium sp.	0.027	0.047	0.009	0.021	0.023	0.008	0.027	0.003	0.012	6
Chironomidae *(mixed species)*	0.060	0.215	0.247	0.144	0.224	0.143	0.098	0.109	0.246	6
Oligochaeta *(unidentified species)*	0.145	0.081	0.049	0.125	0.131	0.042	0.024	0.012	0.040	4
Helobdella (stagnalis)	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.003	10
Hirudinoidea *(unidentified species)*	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	10
Nematomorpha *(unidentified)*	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	5
Dugesia tigrina	0.000	0.000	0.007	0.013	0.001	0.001	0.002	0.003	0.024	4
Gammarus lacustris	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.016	5
Hyallela azteca	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.004	8
Physa sp.	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.000	8
Helisoma sp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	6
Hydracarina *(unidentified)*	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	5
Pisidium sp.	0.003	0.002	0.001	0.001	0.000	0.002	0.001	0.000	0.003	8
TOTAL TAXA	42	43	38	44	46	45	51	33	35	

Figure 1. Long Valley Watershed and Stream Study Reaches



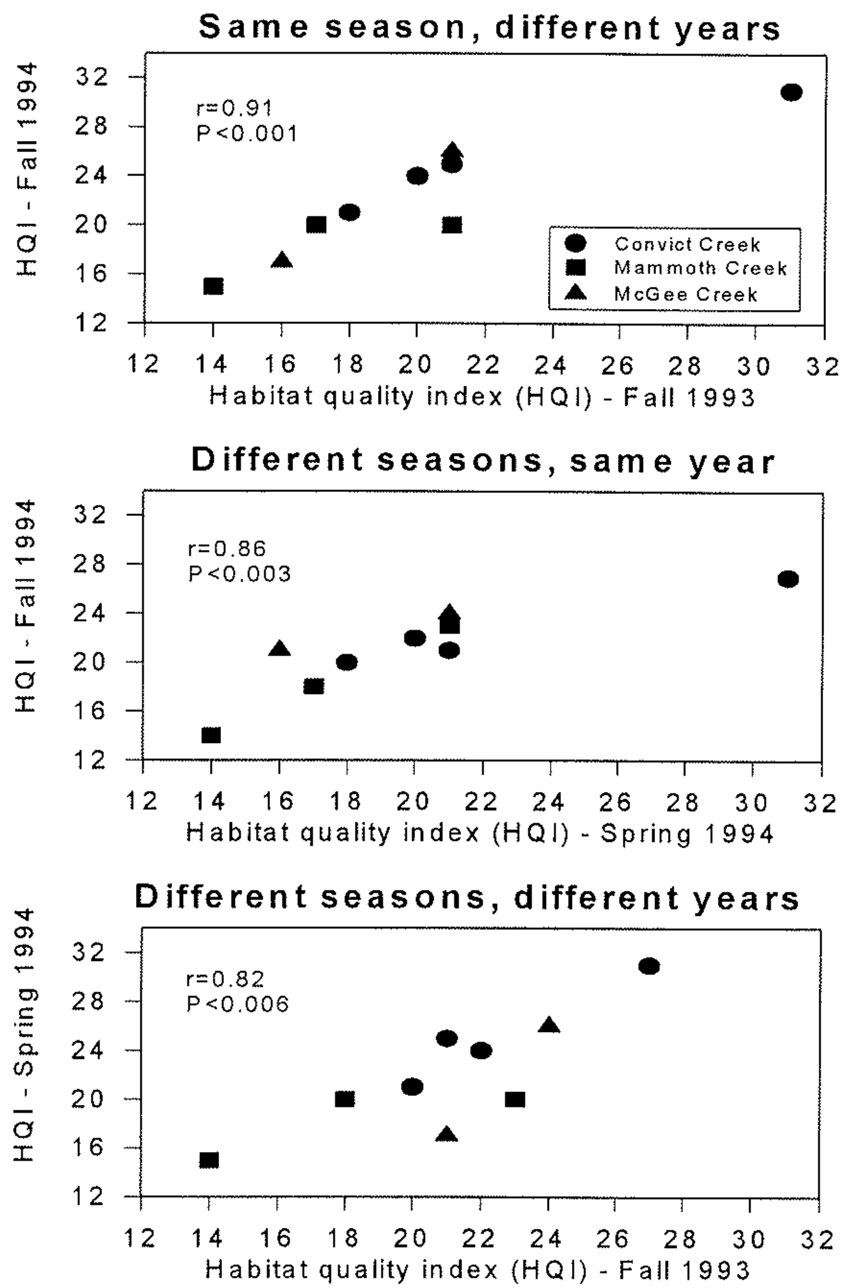


Figure 2. Correlations between the Habitat Quality Index calculated from data collected in the same season but in different years (top) , in different seasons but in the same year (middle), and in different seasons and different years (bottom). The correlation is strongest between HQI's collected in the same season but different years, and lowest between HQI's collected in different seasons and different years.

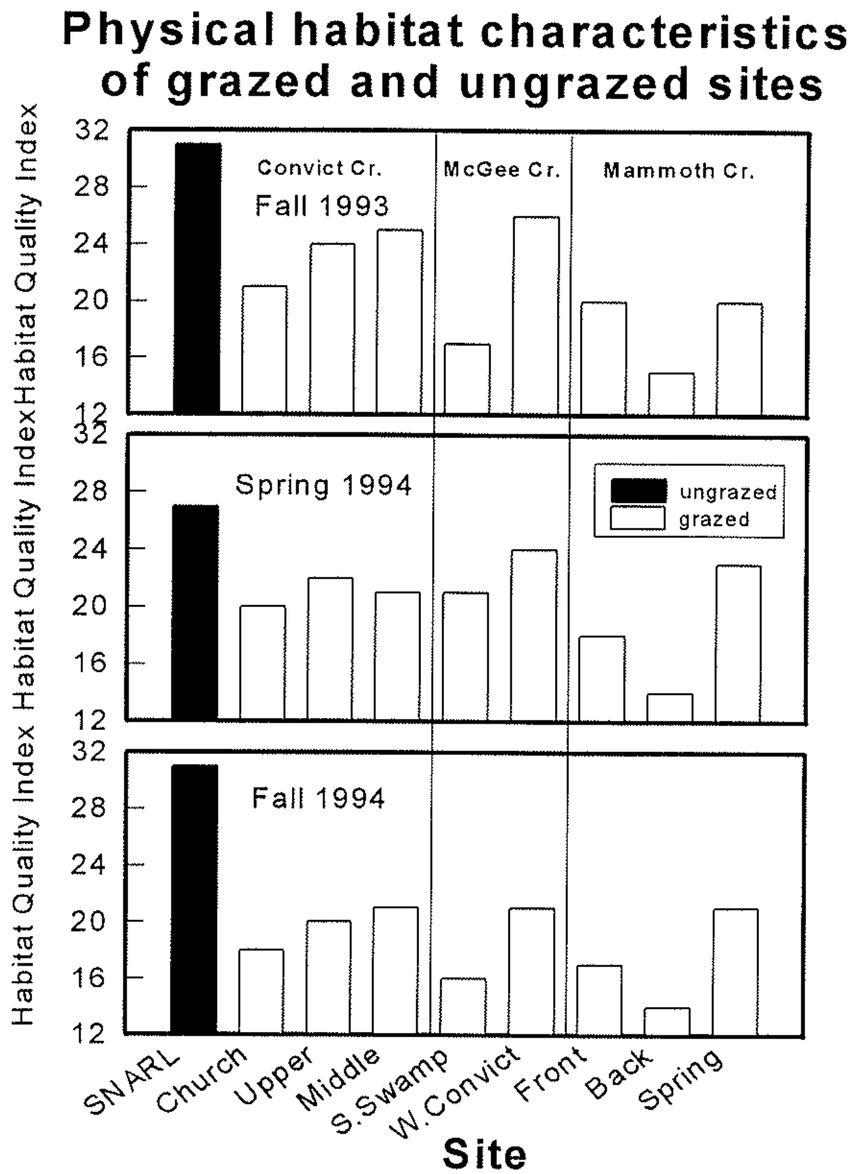


Figure 3. Habitat quality index (HQI) values for all nine study reaches on Convict, McGee, and Mammoth Creeks in Fall 1993, Spring 1994, and Fall 1994. The ungrazed SNARL reach had a higher HQI value than the grazed reaches during all three monitoring periods.

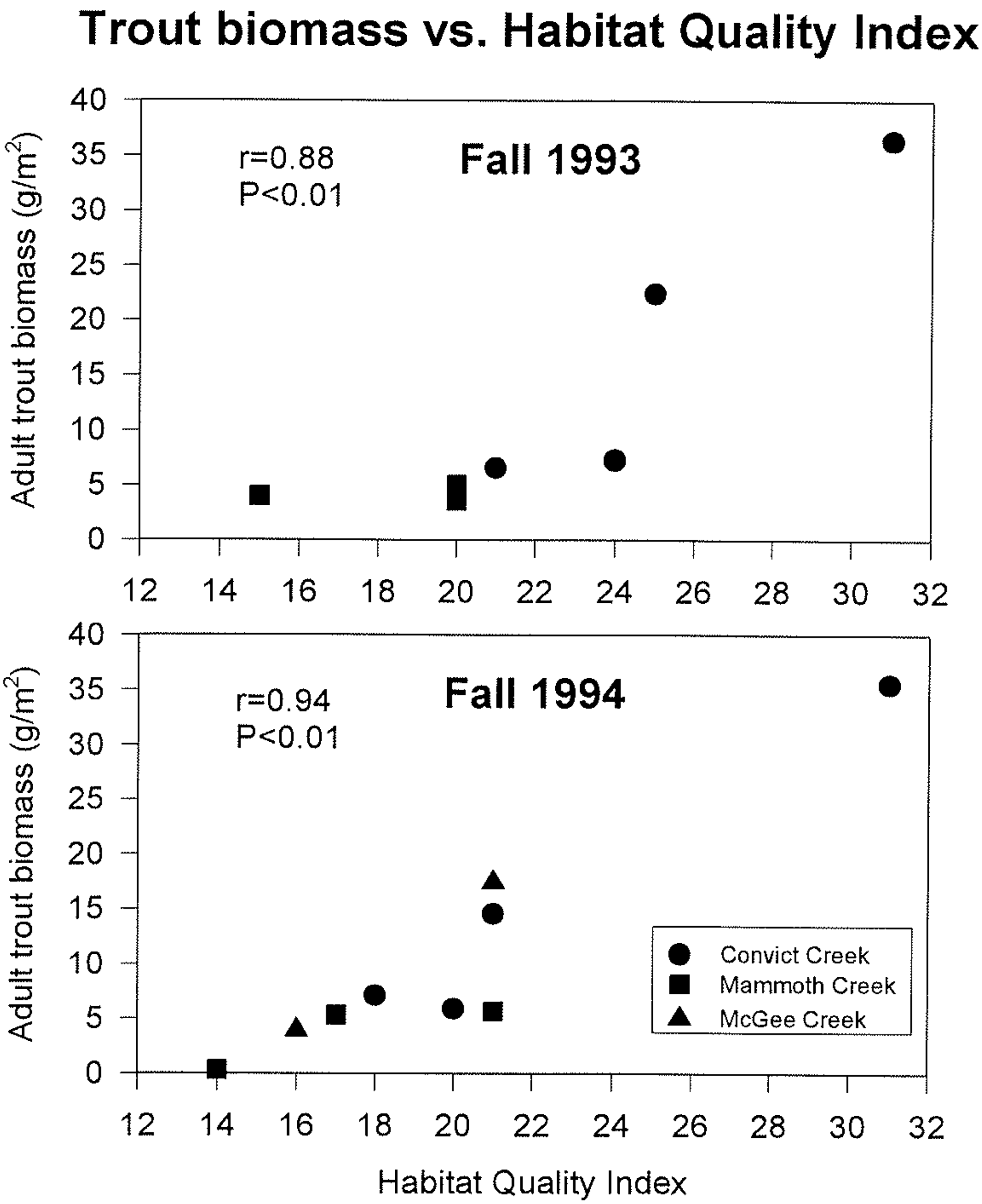


Figure 4. Correlations between adult trout biomass (trout > 150 mm) and the Habitat Quality Index for all nine study reaches in Fall 1993 and Fall 1994. Correlations are highly significant for both years.

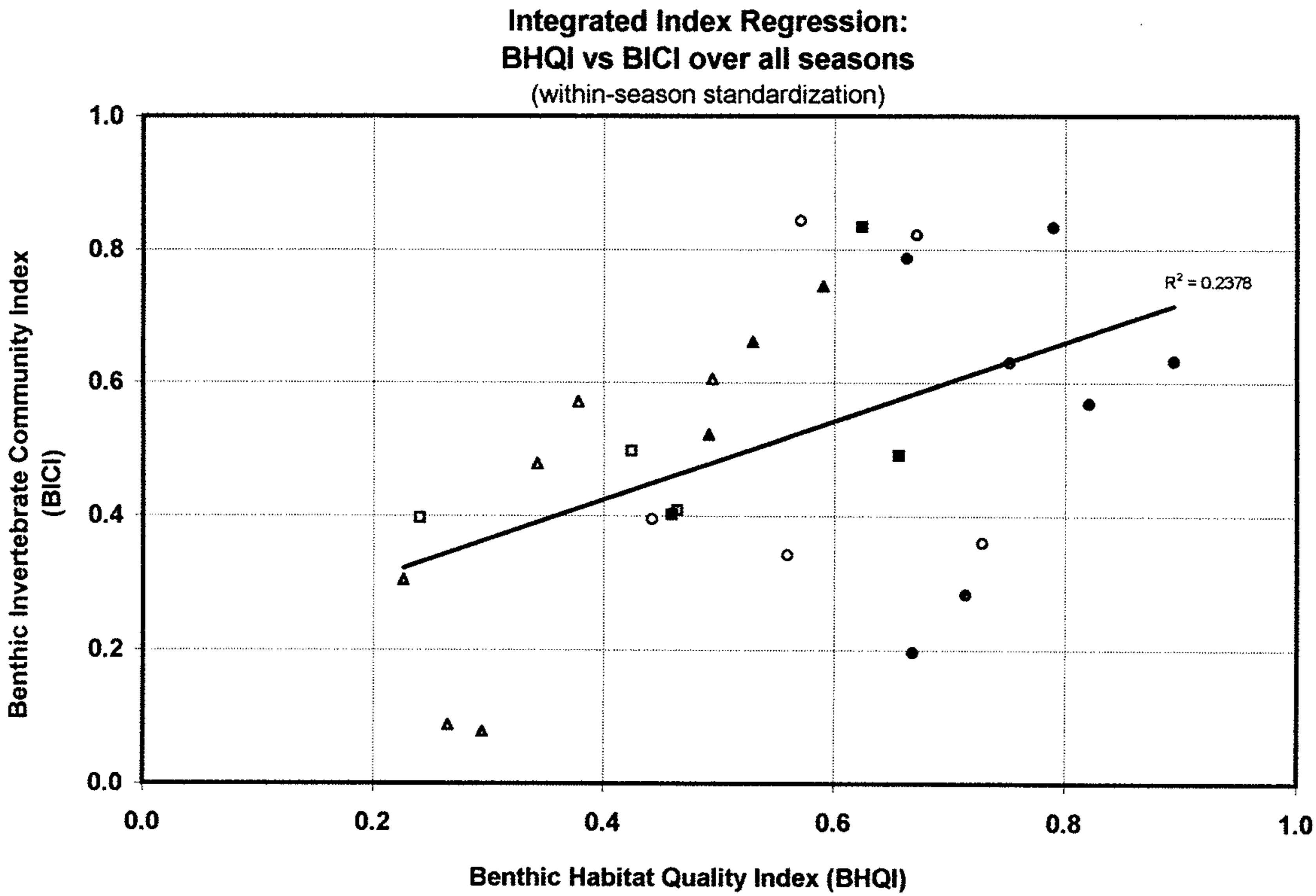


Figure 5. Regression of the Benthic Invertebrate Community Index (BICI) against the Benthic Habitat Quality Index (BHQI) for all nine study reaches for Fall 1993, Spring 1994, and Fall 1994 combined. Circles: Convict Creek (black=SNARL, blue=Church, yellow=Upper, red=Middle); squares: McGee Creek (blue=S. Swamp, red=W. Convict); triangles: Mammoth Creek (blue=Front, yellow=Back, red=Spring). Multiple metric indices provided a more reliable approach to evaluating overall physical and biological habitat quality

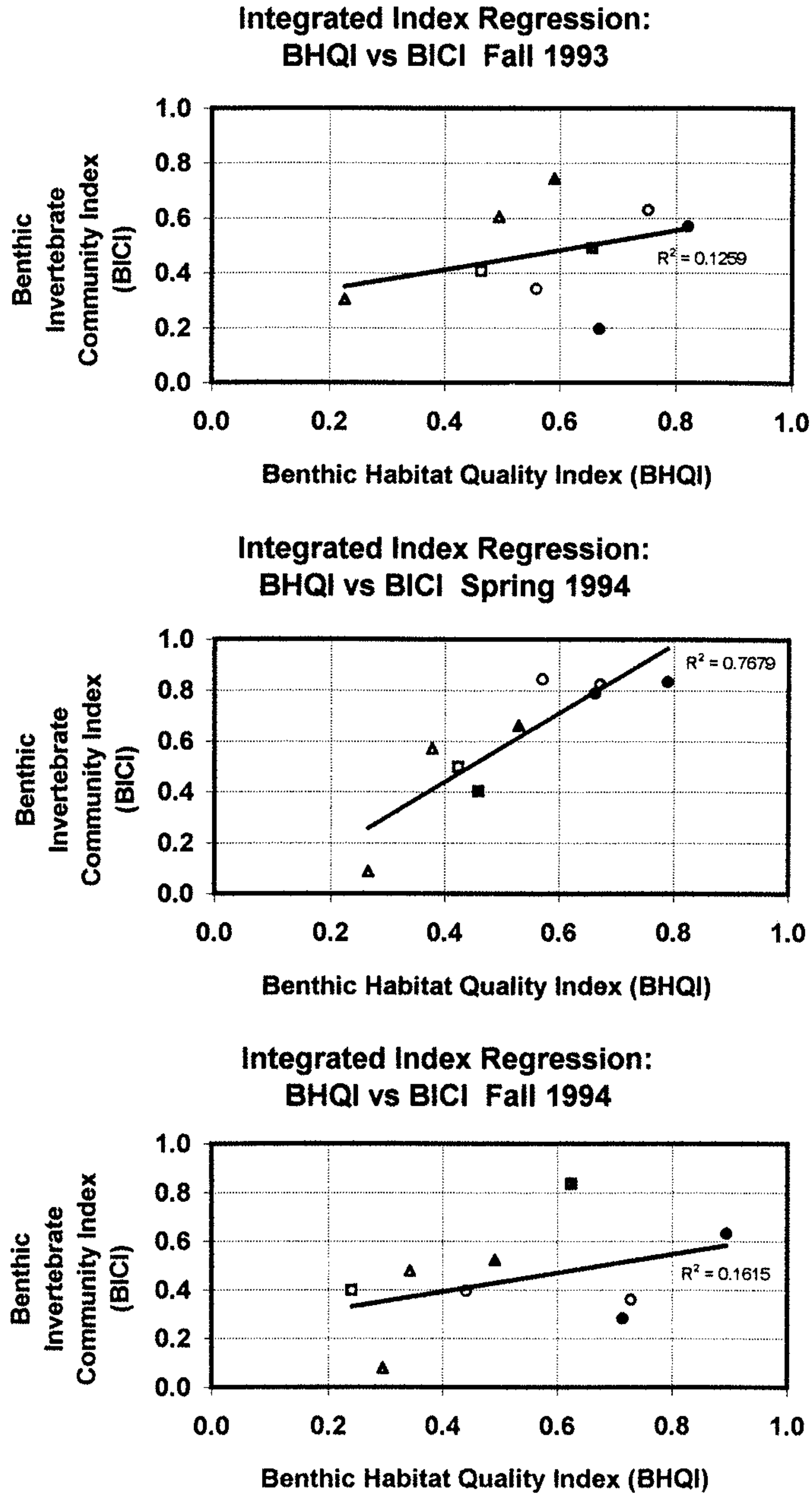


Figure 6. Regression of the Benthic Invertebrate Community Index (BICI) against the Benthic Habitat Quality Index (BHQI) for all nine study reaches for Fall 1993 (top), Spring 1994 (middle), and Fall 1994 (bottom). Symbols the same as in Figure 5. The BHQI explains little variation in BICI during the Fall periods, but a significant amount of the variation in the Spring.