Western North American Naturalist

Volume 60
Number 3 *Festschrift: Gary L. Vinyard*

Article 3

8-11-2000

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Western North American Naturalist 60(3), © 2000, pp. 255–272

A COMPARISON OF RIPARIAN CONDITION AND AQUATIC INVERTEBRATE COMMUNITY INDICES IN CENTRAL NEVADA

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ABSTRACT.—The importance of maintaining healthy riparian communities to sustain natural stream processes and function is well documented. Land management agencies in the West are currently developing methods to assess and monitor riparian community condition to adapt land use practices that would better protect rangeland ecosystems. To determine whether these methods also provide an indication of abiotic and biotic stream condition, we compared the classification system of riparian communities developed by the U.S. Forest Service (USFS) to physical parameters of stream condition and to aquatic invertebrate community assemblages. Thirty-three sites in 19 different streams of the Toiyabe Range in central Nevada were measured for water quality, substrate characteristics, and fish abundance and diversity. We sampled aquatic invertebrates and calculated community indices based on environmental tolerance levels, taxonomic diversity, and abundance of sensitive taxa. USFS personnel classified these sites by dominant riparian plant community type (meadow, willow, or aspen) and ecological status (low, moderate, or high) using plant abundance data, rooting depth, and soil infiltration to determine similarities to potential natural communities.

Riparian condition indices as well as community diversity were significantly correlated to proportions of fine and small-diameter substrate in streambeds. Accumulation of silt was significantly related to plant community type, with meadow sites expressing highest proportions. Further examinations indicated that 2 of 6 invertebrate community indices were significantly related to ecological status, with highest diversity levels occurring mainly in willow- and aspen-dominated sites in moderate ecological condition. Nevertheless, we show that several other environmental variables, including substrate characteristics, dissolved oxygen, water temperature, and species richness of fish communities, were more strongly and consistently related to invertebrate assemblage patterns. Our results demonstrate that information on aquatic invertebrates and stream condition could augment the existing riparian classification system and provide useful monitoring tools to more thoroughly examine ecosystem health in rangelands.

Key words: Toiyabe, Great Basin, rangeland, aquatic invertebrates, biotic indicators, riparian condition, ecological status, diversity measures.

Healthy riparian plant communities are essential components for proper stream ecosystem processes and function (Karr and Schlosser 1978, Gregory et al. 1991, Elmore 1992, Edwards and Huryn 1996, Friberg 1997). Riparian zones in rangelands provide critical sources of diversity and biomass productivity for both plant and animal species (Thomas et al. 1979). Stream bank vegetation also produces essential organic matter for headwater communities (Cummins 1974, Cummins and Spengler 1978) as well as processed material for downstream catchments (Kennedy 1977). Moreover, riparian habitat condition exerts a strong influence on stream channel morphology. Riparian plant root systems increase bank stability, and streamside vegetation attenuates peak velocities of high flows, thereby reducing energies that could otherwise erode banks, elevate sediment loads, and widen channels (Schumm and Meyer 1979). By stabilizing soils, robust vegetation also helps reduce potential damage that could result from land management activities such as livestock grazing (Platts 1981, Swanson et al. 1982).

The U.S. Forest Service (USFS) has estimated that 22% of riparian habitat under its jurisdiction is not meeting their natural resource objectives (USDI Bureau of Land Management and USDA Forest Service 1994). One recently developed method used to improve rangeland condition assessment is the Ecological Status Riparian Determination scorecard developed by the USFS Humboldt-Toiyabe

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National Forest Ecology Team (Weixelman et al. 1996, 1997, 1999). Managers using this method measure impacts of disturbance to riparian corridors by comparing existing soil and plant communities to presumed potential natural communities (PNC) at selected reference sites. These scorecards classify soil and vegetation communities into low, moderate, or high ecological status ratings. Soil and plant community ratings are based on color and permeability of soil surface layers, plant species composition and abundance, litter cover, and rooting depths. Managers use this evaluation system to identify habitat that may require rest from livestock grazing, presumably before damage becomes irreparable. Although this evaluation may provide a useful indication of riparian condition, it may not address aspects of stream condition, which is of primary importance to the maintenance of healthy rangeland ecosystems (National Research Council 1994).

Stream invertebrate communities have been routinely used and recommended as biological indicators of habitat degradation from land use practices (Plafkin et al. 1989, Rosenberg and Resh 1993, Barbour et al. 1995, Resh et al. 1995), including impacts from livestock grazing (Bauer and Burton 1993). Aquatic invertebrate communities are useful monitors because they integrate ecological conditions both temporally and spatially. Our research explores whether the USFS riparian classification system reflects abiotic and biotic components of associated streams. To address this, we compared condition ratings of riparian communities derived from the USFS Ecological Status Riparian Determination methodology to abiotic measures and community assemblages of aquatic invertebrates in streams of central Nevada.

METHODS

Fieldwork

Thirty-three sampling sites were located in 19 different drainages (Fig. 1). Region 4 staff of the USFS surveyed plant and soil communities at each of these sites using the Ecological Condition scorecard methodology (Weixelman et al. 1996). Dominant plant communities were typed and rated for percent similarity to PNC and grouped into 3 ecological status categories based on litter cover and abundance of vegetative species (low, moderate, and high; Table 1). Soil condition ratings based on color of surface layers, depth of fine roots, and infiltration tests of water absorption were used to adjust these classifications. Eight low-, 12 moderate-, and 13 high-condition sites were sampled, with meadow riparian bank vegetation occurring at 12 sites, willow species dominating at 11 sites, and aspen at 10 (Fig. 2).

We collected 5 invertebrate samples using an upper-frame Winget-modified Surber net (Winget and Mangum 1979) in June 1994 at each site from similar microhabitats reflecting dominant substrate conditions and flow regimes (Hauer and Resh 1996). All samples were preserved in 95% isopropyl alcohol. Invertebrates were identified in the lab using Usinger (1956), Edmondson (1959), Thorp and Covich (1991), and Merritt and Cummins (1996). Difficult taxonomic identifications were sent to the USDA Aquatic Ecosystem Laboratory at Brigham Young University in Provo, Utah, for verification.

At 17 sites where fish were present, we completed population estimates using 3-pass depletion sampling with an electrofisher. Captured fish were identified, weighed, measured, and released. Total reach lengths surveyed (combined upstream and downstream sections from the invertebrate sampling point) varied from 20 to 66 m, depending on stream order. We measured section widths at the upstream seine net, middle of the reach, and downstream net, and then averaged them. Estimated densities (fish $\cdot$ m^{-2}) extrapolated to zero-effort were calculated using regression equations for each of the 2 sections and averaged. Cumulative species captured from the 2 reaches were recorded with no adjustments made.

We also measured water quality parameters that affect habitat conditions for aquatic organisms. Mean daily water temperature (°C) was recorded at 2-h intervals for the month using Hobo digital data loggers. Replicate water samples were collected with sterilized Nalgene containers and analyzed that same day under laboratory conditions. Ammonia (NH_3–N), nitrite (NO_2–N), nitrate (NO_3–N), and orthophosphate phosphorus (PO_4–P) concentrations (μg L^{-1}) were measured using a portable spectrophotometer. Alkalinity (μeq L^{-1}) was measured using titration with a phenolphthalein indicator (Wetzel and Likens 1991). Total dissolved residues (mg L^{-1}) were quantified by filtering 200-mL samples onto

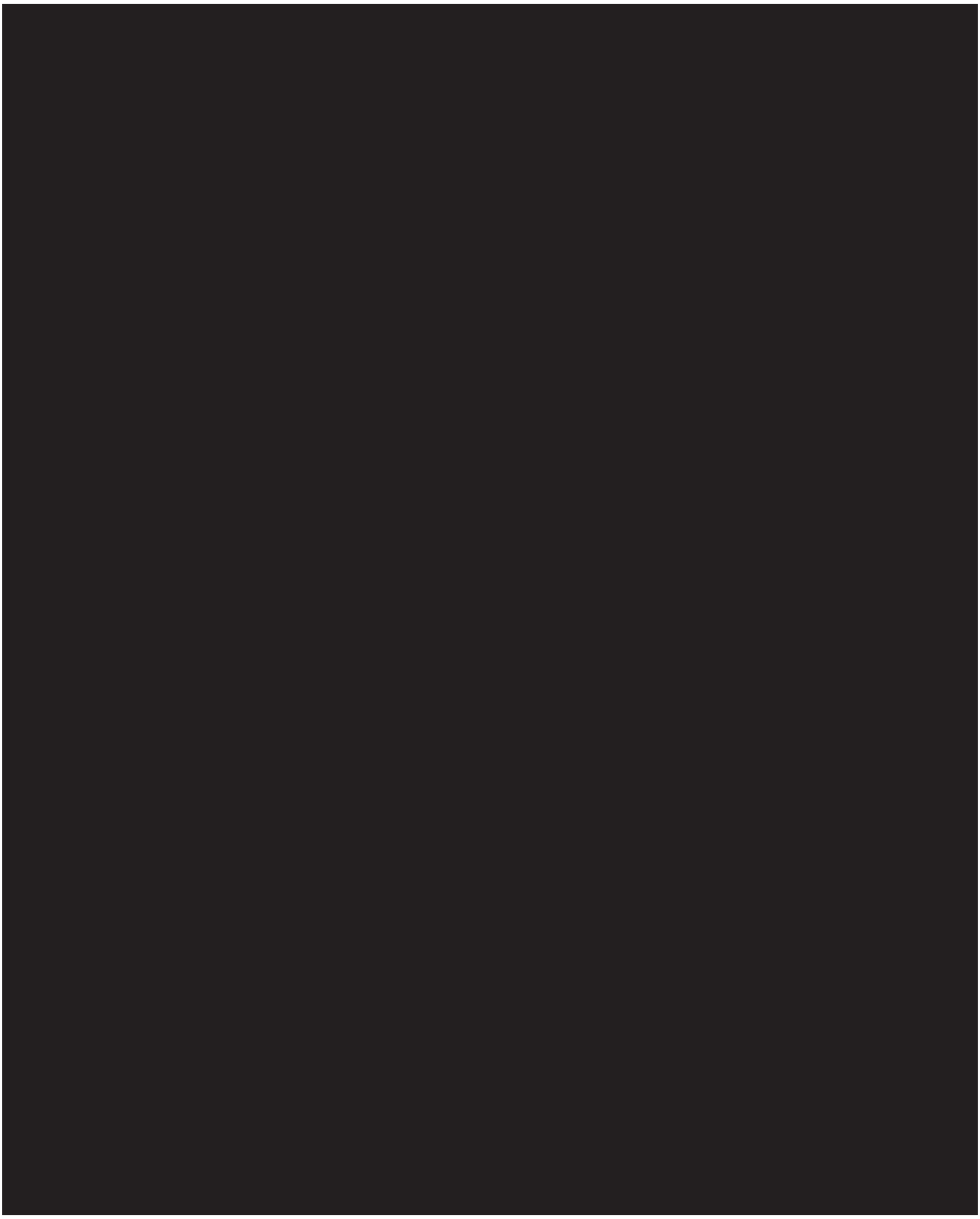

Fig. 1. Study site locations in the Toiyabe Range, Nevada.

pre-weighed 0.50-μm glass fiber filters that were evaporated to dryness for 1 d, folded in foil wraps, dried overnight in an oven set to 103°C, and reweighed (Lind 1985). Dissolved oxygen concentrations (mg L^{-1}), conductivity (μmhos), and pH were measured on-site using YSI meters. Primary production (chlorophyll *a*) was assessed by placing 10 glass microscope slides into fitted plexiglas frames that were tied to rebar stakes and oriented horizontally to water current for 3 wk prior to retrieval. Mean chlorophyll *a* concentrations (μg L^{-1})

TABLE 1. Study site locations in the Toiyabe Range, Nevada. Elevation, latitude, and longitude were recorded in the field. Plant community type (meadow, willow, aspen) and riparian condition (L: low, M: moderate, H: high) using percent similarity to potential natural communities were developed by U.S. Forest Service professionals (methods in Weixelman et al. 1996). Riparian condition acronyms are as follows: percent similarity to potential natural communities (% PNC) for plant communities and fine root depths (FRD) for soil communities. Soil condition ratings (L, M, and H) were developed from data based on color of surface layers, rooting depth, and infiltration tests of water absorption. Sites where data were not collected for that parameter are signified with "nd" (no data). Overall ecological status ratings are in bold and listed under Plant Condition.

			Latitude			Longitude			Elevation			Condition			
Site	Drainage	Order	$x°$	y'	z''	$x°$	y'	z''	(m)	Plant community	Type	Plant (% PNC)		Soil (FRD)	
1	San Juan	2	117	16	21.7	39	7	10.1	2213	*Carex nebrascensis*	Meadow	**M**	(46.8)	M	(118.8)
2	San Juan	2	117	16	28.7	39	7	9.5	2226	*Salix boothi lutea*	Willow	**H**	(60)	H	(nd)
3	San Juan	2	117	16	32.5	39	7	24.1	2204	*Populus tremuloides*	Aspen	**M**	(31)	H	(nd)
4	Cottonwood	2	117	16	28.3	39	8	55.1	2143	*Carex nebrascensis*	Meadow	**H**	(51.5)	M	(79.4)
5	Cottonwood	2	117	15	58	39	8	47.4	2186	*Populus tremuloides*	Aspen	**H**	(67.4)	H	(nd)
6	Washington	2	117	13	51.7	39	9	14.7	2345	*Populus tremuloides*	Aspen	**H**	(57.6)	M	(nd)
7	Cahill	1	117	2	17.7	39	28	0.8	2271	*Carex nebrascensis*	Meadow	**M**	(48)	L	(14.7)
8	Birch-N.fork	1	117	2	37.9	39	25	37.6	2329	*Populus tremuloides*	Aspen	**H**	(59.5)	L	(nd)
9	Birch-N.fork	1	117	2	45.8	39	25	25.3	2284	*Carex nebrascensis*	Meadow	**L**	(13.4)	M	(105.9)
10	Birch-S.fork	1	117	5	30.5	39	22	26.3	2399	*Salix boothi lutea*	Willow	**M**	(37.9)	L	(18.9)
11	Birch-S.fork	2	117	2	48.9	39	23	57.5	2165	*Carex nebrascensis*	Meadow	**H**	(54.8)	H	(37.7)
12	Reese	3	117	25	19.8	38	48	24.9	2250	*Salix boothi lutea*	Willow	**H**	(62.9)	L	(37.7)
13	Reese	3	117	25	53.1	38	48	49	2238	*Salix boothi lutea*	Willow	**H**	(66.2)	L	(54.7)
14	Indian	3	117	30	11.8	38	48	49.4	2155	*Carex nebrascensis*	Meadow	**M**	(35)	M	(108.8)
15	Kingston	2	117	9	37.2	39	14	8	2134	*Carex nebrascensis*	Meadow	**L**	(42.7)	L	(58.8)
16	Kingston	2	117	9	57.3	39	15	3	2159	*Salix boothi lutea*	Willow	**M**	(28)	M	(98.6)
17	Big Creek	1	117	7	5.6	39	17	58.3	2317	*Carex nebrascensis*	Meadow	**L**	(41.8)	H	(59.4)
18	Big Creek	1	117	7	14.8	39	18	17.4	2226	*Salix boothi lutea*	Willow	**M**	(33.4)	L	(30.2)
19	Stewart	1	117	22	3.9	38	53	33.2	2735	*Carex aquatilis/nebrascensis*	Meadow	**M**	(43)	H	(nd)
20	Stewart	1	117	21	35.1	38	53	16.2	2838	*Salix geyeriana*	Willow	**L**	(23)	L	(64.2)
21	Stewart	2	117	23	40.5	38	55	34.3	2323	*Populus tremuloides*	Aspen	**H**	(39)	H	(nd)
22	Clear	3	117	22	26.9	38	55	52.1	2402	*Populus tremuloides*	Aspen	**H**	(35)	L	(nd)
23	Illinois	1	117	26	28.8	38	53	37.9	2216	*Carex nebrascensis*	Meadow	**M**	(27.9)	M	(64.7)
24	Marysville	1	117	19	30	39	2	12	2165	*Populus tremuloides*	Aspen	**M**	(33.7)	H	(nd)
25	Marysville	1	117	20	9.4	39	2	8.6	2204	*Salix exigua lutea*	Willow	**H**	(63)	M	(34.4)
26	Veatch	1	117	5	0.5	39	27	17.7	2040	*Populus tremuloides*	Aspen	**H**	(67.8)	M	(nd)
27	Willow	2	117	0	46.8	39	34	17	2104	*Carex nebrascensis*	Meadow	**L**	(24.1)	M	(38.4)
28	Willow	2	116	59	28.5	39	32	59.3	2049	*Salix exigua lutea*	Willow	**L**	(25.2)	M	(63.9)
29	Blackbird	1	116	58	43.8	39	25	2.4	2055	*Populus tremuloides*	Aspen	**M**	(31)	L	(nd)
30	Summit	1	117	16	48.2	38	59	3	2470	*Salix boothi lutea*	Willow	**L**	(21)	L	(38.4)
31	Summit	1	117	15	48	38	58	34.2	2226	*Populus tremuloides*	Aspen	**H**	(111)	M	(nd)
32	Peavine	3	117	18	12.2	38	37	0.8	1875	*Salix boothi lutea*	Willow	**L**	(21)	M	(78.1)
33	Marysville	1	117	20	38.5	39	2	27.6	2131	*Deschampsia cespitosa*	Meadow	**M**	(29)	L	(35.9)

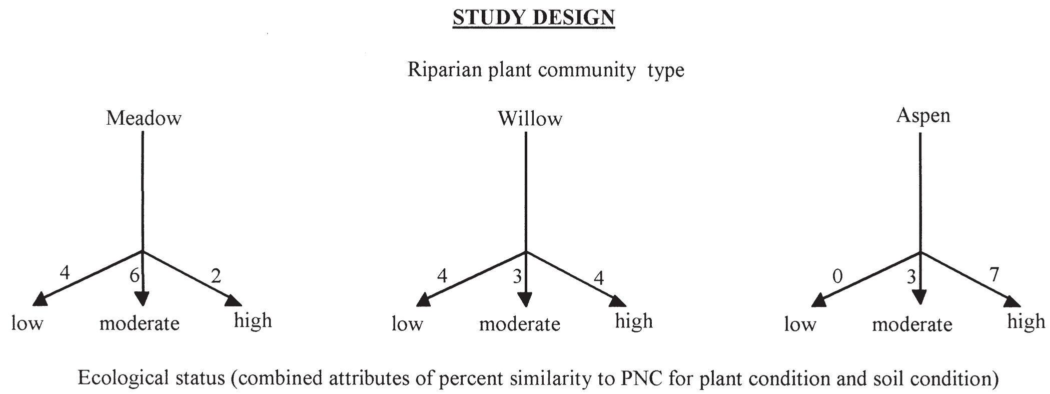

Fig. 2. Classifications of aquatic study sites according to riparian plant community types and ecological status. Sample sizes are provided next to each branch of the study design forks.

were calculated from periphyton colonizations on 3 of these slides (2nd from either end and the middle) using spectrophotometric procedures described in Wetzel and Likens (1991). Mean discharge was measured by dividing 3 representative transects into equal intervals depending on total stream width and microhabitat complexity. Velocity (dm sec^{-1}), depth (dm), and width (dm) of each interval were recorded, summed, and then averaged (L sec^{-1}; Platts et al. 1983). We also estimated mean proportion of substrate at 5 transects that best represented overall site characteristics. Two observers independently quantified and concurred on substrate composition present at each transect (silt, mud [clay], sand, cobble, gravel, boulder, and woody debris; particle size list adopted from Platts et al. 1983). Fresh cow dung (number per acre) was counted from three 0.01-acre plots in riparian vegetation on both banks on-site, upstream, and downstream (3 replicate counts) to indirectly quantify grazing pressure that has occurred in recent years. Elevation (m) was recorded in the field using a portable Trimble GPS unit. Stream order and slope (%) for each site location were derived from 7.5′ topographic quadrangles (USGS) and digital elevation models (DEM) in Arc/Info software, respectively.

Sampling Effort Estimates

Standard deviations (s) in number of taxa and organisms collected with the 5 Surber samples from each site were used to evaluate sampling effort. We used the equation for estimating invertebrate densities (Resh 1979):

$$E_{Ts,Os} = [\,(t * s_{T,O}) / (D_{T,O} * \bar{x}_{T,O})\,]^2 \qquad (1)$$

where E_{Ts} is the number of samples estimated from s_T for number of taxa, and E_{Os} is the number of samples estimated from s_O for number of organisms, t is Student's t distribution ($t = 2.13$, df = 4), D is the relative error (D_T equals ± 20% because taxa not collected at a site where they do occur would impair interpretations, and D_O equals ± 40%, a realistic variance for number of organisms collected), and $\bar{x}_T$, $\bar{x}_O$ are the mean numbers of taxa and organisms collected from the 5 samples, respectively.

Aquatic Invertebrate Community Indices

We developed biotic components of the same index used by USFS and BLM in western states based on published tolerance quotients (TQ; Vinson 1999). Dr. Fred Mangum (Aquatic Ecosystem Laboratory, Provo, UT) also provided some TQ values not available in the literature. Tolerance quotients ranged from 2 (taxa found in only high-quality unpolluted waters) to 108 (taxa found in severely polluted waters). Values are based on tolerances to elevated alkalinity and sulfate levels, as well as selectivity for or against fine substrate and low stream gradients (Winget and Mangum 1979, Platts et al. 1983).The community tolerance quotient (CTQ_a) was calculated with the equation:

$$CTQ_a = \Sigma\,(TQ / S) \quad (2)$$

where TQ is the taxonomic tolerance and S is the total number of taxa collected per site. The dominance-weighted community tolerance quotient (CTQ_d) was calculated as:

$$CTQ_d = \Sigma\,(n_i\,TQ / N) \quad (3)$$

where n_i is the number of individuals collected of taxa *i*, TQ is the same as above, and N is the total number of individuals collected at a site. A high score would indicate invertebrates collected from that site had a disproportionate number of species tolerant to the low water-quality conditions described previously.

We modeled diversity of the invertebrate communities using the logarithmic series (Fisher et al. 1943). Selection of this model was based on preliminary inspections of species abundance plots and consideration of attributes that could influence our investigation (good ability to discriminate between sites, low sensitivity to sample size, and common use in the ecological literature; Magurran 1988). Shannon diversity (H′) was also included to allow comparisons with other published studies.

The log series diversity measure is calculated through an iterative process to determine 'x', the parameter constant of the logarithmic series. This was done with cumulative numbers of taxa and organisms collected from each site with the equation (Krebs 1989):

$$S / N = (1 - x) / x\,[-\ln (1 - x)] \quad (4)$$

where S is the total number of taxa and N is the total number of individuals collected.

Alpha diversity (α) was then calculated using the equation:

$$\alpha = N\,(1 - x) / x \quad (5)$$

with expected values of α generally twice that of the Shannon diversity index (H′) and ranging from slightly <2.0 to >6.0, with higher numbers indicating higher levels of diversity. Goodness-of-fit (χ^2) tests were conducted for each site to examine whether expected number of taxa in each abundance class differed from observed values.

Shannon diversity was calculated as:

$$H' = -\sum_{i=1}^{S} (p_i \ln p_i) \quad (6)$$

where H′ is an index of diversity, S is the total number of taxa occurring in a sample, and p_i is the proportional abundance of the i^{th} taxa. Values usually range from 1.5 to 3.5, with higher numbers indicating higher levels of diversity.

Simple counts of total number of taxa (total taxa) and number of Ephemeroptera, Plecoptera, and Trichoptera (EPT taxa) collected were also included in the analysis since impacted aquatic systems often exhibit reduced numbers of EPT taxa and taxonomic richness (Resh et al. 1995).

Data Analysis

We used Canonical Correspondence Analysis (CCA) to examine the relative importance of measured environmental parameters to numbers of individuals collected for each taxa ($\log_2$-transformed). The categorical variables, riparian community type (meadow = 1, willow = 2, aspen = 3) and ecological status (low = 1, moderate = 2, high = 3), were not included in the ordination. These were instead examined as treatment effects for both the invertebrate assemblage metrics and proportion of silt in the streambed. A 2-level nested ANOVA model (ecological status nested within riparian community type) was used since ranges within each ecological classification differed depending on riparian community type. We used the Tukey multiple-comparison test (adjusted for unequal sample sizes) with significance levels maintained at the 0.05 level if factor effects were detected.

Percent similarities to PNC of the riparian community and plant community diversity measures (H′ and α) were included in the CCA along with the 22 environmental variables previously described, as well as E_{Ts} (equation 1). Chlorophyll *a* concentrations, soil community condition (FRD), stream order, conductivity, and E_{Os} were not input into the data set because of missing values (chlorophyll *a* and soil community condition) and redundancy. Taxonomic abundance and environmental data were analyzed using the Cornell Ecology program CANOCO (Ter Braak 1987–1992). CCA uses reciprocal averaging of species compositions that is constrained by data on environmental factors. Linear combinations of measured environmental variables are used to construct the 3 ordination axes with an associated eigen value (λ) that describes the explained variation in taxonomic composition. Options selected included centering and standardizing axis scores to unit variances with scales optimized for representation of invertebrate taxa.

Correlations between axes scores and each environmental variable (including those environmental parameters not originally included in CCA) were determined using Statistical Analysis Software (SAS). Environmental parameters that were significantly correlated with at least 1 canonical axis were then used as independent variables in regression analysis. Dependent variables were the 6 invertebrate community indices. Multiple regression models were screened using a 2% or greater improvement to R^2 values when each parameter was subsequently added (RSQUARE option using PROC REG model construction). Highly correlated environmental variables ($r > 0.50$) were restricted from model entry (Cody and Smith 1991). These 6 models were further reduced by inspecting changes to the index of determination (R^2_a) to ascertain whether they were improved by the addition of another parameter compared to preceding models.

RESULTS

Sampling Effort and Aquatic Insect Community Indices

Eighty-five percent (89 of 105) of aquatic invertebrate taxa collected were identified to genus and to species when possible (Table 2). Variability in number of taxa collected using a predetermined relative error of 20% suggested that 5 replicates per site reasonably described the community, as 3 Surber samples was the most frequent number calculated for the 33 sites (Table 3). In fact, estimates of the required sampling effort were between 1 and 5 samples for 45% of the sites. However, variability in number of taxa captured did elevate estimates of sampling effort required to very high levels at several sites. Our samples were more variable in number of organisms collected than for taxonomic units. As a result, estimated sampling effort for capturing a representation of abundance was higher, with a mode of 6 samples (Table 3). Six sites required ≤5 samples to determine abundance of aquatic invertebrate taxa with a relative error of 40%.

Aquatic invertebrate community indices were highly correlated with one another. Community tolerance quotients (CTQ_a and CTQ_d) were significantly positively correlated to each other ($r = 0.58$, $p < 0.001$) and negatively correlated to number of EPT taxa (CTQ_a: $r = -0.74$, $p < 0.0001$; CTQ_d: $r = -0.44$, $p \leq 0.05$). CTQ_d was also significantly correlated to the diversity metrics (H′: $r = -0.57$, $p < 0.001$; α: -0.51, $p \leq 0.05$). Diversity measures ($r = 0.73$, $p < 0.0001$) as well as total taxa and EPT taxa ($r = 0.63$, $p < 0.0001$) were significantly positively correlated to one another.

The logarithmic series model (the basis for determining α-diversity levels) demonstrated an overall good fit for a majority of our sites (27 of 33), with significant departures from expected distributions occurring at Reese 13 ($\chi^2 = 24.8$, df = 10), Indian 14 ($\chi^2 = 32.8$, df = 10), Kingston 15 ($\chi^2 = 27.0$, df = 10), Marysville 25 ($\chi^2 = 26.7$, df = 11), Blackbird 29 ($\chi^2 = 1278.6$, df = 10), and Peavine 32 ($\chi^2 = 22.6$, df = 10).

Plant Community Type and Ecological Status

Four of 6 ANOVA models for aquatic invertebrate community indices were not significant, either among groups (riparian community type) or within groups (ecological status) (CTQ_a: $F_{2,5} = 4.09$, $F_{5,25} = 1.87$; CTQ_d: $F_{2,5} = 2.63$, $F_{5,25} = 2.06$; total taxa: $F_{2,5} = 0.02$, $F_{5,25} = 1.12$; EPT taxa: $F_{2,5} = 2.07$, $F_{5,25} = 1.07$). In contrast, both diversity indices were significantly related to ecological status (H′-diversity: $F_{2,5} = 2.14$, $F_{5,25} = 2.84$; α-diversity: $F_{2,5} = 1.34$, $F_{5,25} = 2.92$). Tukey mean comparisons of Shannon diversity levels were significant for willow in moderate and high condition (0.78 ± 0.17, $q_{25,12} = 4.60$), for willow and meadow in moderate condition (0.60 ± 0.16, $q_{25,12} = 3.82$), for meadow and willow in high condition (0.65 ± 0.19, $q_{25,12} = 3.42$), and for aspen in both moderate (0.60 ± 0.17, $q_{25,12} = 3.54$) and high (0.56 ± 0.14, $q_{25,12} = 4.02$) condition compared to willow communities in high condition. Mean comparisons for α-diversity of the aquatic invertebrate fauna were similar, with significant differences occurring for willow in moderate condition compared to both meadow in moderate condition (1.92 ± 0.53, $q_{25,12} = 3.63$) and willow in high condition (2.4 ± 0.57, $q_{25,12} = 4.21$), as well as for aspen in moderate condition compared to both meadow in moderate condition (1.89 ± 0.53, $q_{25,12} = 3.53$) and willow in high condition (2.35 ± 0.57, $q_{25,12} = 4.12$).

TABLE 2. Aquatic invertebrate taxa collected from 33 stream sites in the Toiyabe Range sampled in June 1994. Tolerance quotients (TQ) that were available either from the literature or from consultations with Fred Mangum (Aquatic Ecosystem Laboratory, Provo, UT) are provided. Voucher specimens and samples are maintained at the University of Nevada–Reno, Biology Department.

Order	Family	*Genus* sp.	TQ
Amphipoda	Talitridae	*Hyalella azteca*	98
Anisoptera	Aeshnidae	*Aeshna* sp.	72
		Triacanthogyna trifida	72
Coleoptera	Amphizoidae	*Amphizoa* sp.	24
	Chrysomelidae	*Donacia* sp.	—
	Curculionidae	*Hyperodes* sp.	—
		Listronotus sp.	—
		Lixus sp.	—
		Steremnius sp.	—
	Dryopidae	*Helichus* sp.	72
	Dytiscidae	*Hydaticus* sp.	72
		Dytiscus sp.	72
		Eretes sticticus	72
		Hydroporus sp.	72
		Hydrovatus sp.	72
		Nebrioporus sp.	72
		Rhantus sp.	72
	Elmidae	*Dubiraphia* sp.	—
		Optioservus sp.	104
		Zaitzevia sp.	104
	Gyrinidae	*Dineutus* sp.	108
		Gyrinus sp.	108
	Haliplidae	*Peltodytes* sp.	54
	Hydraenidae	*Hydraena* sp.	72
	Hydrophilidae	*Ametor* sp.	72
		Helophorus sp.	72
		Laccobius sp.	72
		Tropisternus sp.	72
	Lampyridae		—
	Limnichidae		—
	Noteridae	*Suphisellus* sp.	—
Collembola			—
Diptera	Ceratopogonidae	*Bezzia* sp.	96
	Chironomidae		108
	Culicidae		108
	Deuterophlebiidae	*Deuterophlebia coloradensis*	4
	Dixidae		108
	Dolichopodidae		108
	Empididae	*Chelifera* sp.	108
	Muscidae	*Limnophora* sp.	108
	Psychodidae	*Pericoma* sp.	86
	Ptychopteridae		100
	Rhagionidae	*Atherix* sp.	66
	Simuliidae	*Simulium* sp.	108
	Stratiomyidae	*Euparyphus* sp.	108
	Tabanidae		108
	Tipulidae		72
Ephemeroptera	Ameletidae	*Ameletus* sp.	72
	Baetidae	*Baetis* sp.	72
	Ephemerellidae	*Drunella doddsi*	2
		Ephemerella inermis	92
	Heptageniidae	*Cinygmula* sp.	30
		Epeorus sp.	18
		Rhithrogena sp.	21

TABLE 2. Continued.

Order	Family	*Genus* sp.	TQ
Ephemeroptera	Leptophlebiidae	*Choroterpes* sp.	60
		Leptophlebia sp.	36
		Paraleptophlebia sp.	30
Gastropoda			108
Hemiptera	Corixidae	*Corisella* sp.	—
		Hesperocorixa sp.	108
Hydracarina			98
Lepidoptera	Cosmopterigidae	*Pyroderces* sp.	72
Nematoda			108
Oligochaeta	Lumbricidae	*Lumbricus* sp.	108
Ostracoda			108
Pelecypoda			108
Plecoptera	Capniidae	*Eucapnopsis brevicauda*	—
	Chloroperlidae	*Paraperla* sp.	24
		Suwallia sp.	24
		Sweltsa sp.	24
	Nemouridae	*Malenka* sp.	6
		Prostoia besametsa	24
		Zapada sp.	16
	Perlidae	*Doroneuria baumanni*	18
		Hesperoperla pacifica	30
	Perlodidae	*Cultus* sp.	12
		Isoperla sp.	48
		Megarcys signata	30
	Pteronarcyidae	*Pteronarcella* sp.	30
Trichoptera	Brachycentridae	*Amiocentrus aspilus*	24
		Brachycentrus sp.	24
		Micrasema sp.	24
		Oligoplectrum echo	24
	Glossosomatidae	*Anagapetus* sp.	24
		Glossosoma sp.	24
	Hydropsychidae	*Cheumatopsyche* sp.	108
		Hydropsyche sp.	108
		Parapsyche almota	10
	Hydroptilidae	*Hydroptila* sp.	108
		Ochrotrichia sp.	108
		Oxyethria sp.	108
	Lepidostomatidae	*Lepidostoma* sp.	24
	Limnephilidae	*Chyranda centralis*	18
		Ecclisomyia sp.	24
		Hesperophylax sp.	108
		Neophylax sp.	24
	Odontoceridae	*Namamyia* sp.	—
	Philopotamidae	*Dolophilodes* sp.	24
	Phryganeidae		64
	Polycentropidae	*Polycentropus* sp.	72
	Rhyacophilidae	*Rhyacophila acropedes*	72
		R. hyalinata	24
Tricladida			108
Zygoptera	Coenagrionidae	*Amphiagrion* sp.	72
		Argia sp.	108

TABLE 3. Sampled biota surveyed in June 1994 in the Toiyabe Range, Nevada. Estimated samples required using standard deviations of either number of taxa (E_{Ts}) or number of organisms (E_{Os}) collected from 5 Surber samples at 33 study sites (see equation 1 in text for details), calculated indices based on the Biotic Condition Index (BCI), diversity measures (H′ and α), total counts of all taxa (total taxa) and enumerations of taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT taxa) as well as total number of invertebrates collected (*N*) are presented. Results from fish surveys are shown for density estimates ($\bar{x} \pm s$) and cumulative species collected. Fish species acronyms are: *Catostomus tahoensis* (ct), *Oncorhynchus clarki henshawi* (och), *Oncorhynchus mykiss* (om), *Rhinichthys osculus* (ro), *Salvelinus fontinalis* (sf), and *Salmo trutta* (st).

Site	$E_{Ts,Os}$ (± % $D_{T,O}$)		BCI components		Diversity indices		Taxa and individuals collected			Fish surveys	
	E_{Ts} D_T: 20%	E_{Os} D_O: 40%	CTQ_a	CTQ_d	Shannon (H′)	Alpha (α)	Total taxa	EPT taxa	*N*	Density (no./m^2)	Species collected
1	5	26	73.0	101.2	1.80	4.50	27	11	1832	0	0
2	4	11	61.2	88.0	2.00	3.85	20	13	689	0	0
3	3	2	60.3	75.1	2.16	5.29	27	17	1269	1.22 ± 0.19	om, sf, st
4	3	8	53.4	75.3	2.32	4.74	23	16	587	0.17 ± 0.01	sf, st
5	2	10	53.9	89.7	1.72	4.21	24	18	1461	0.89 ± 0.48	om, sf, st
6	1	3	59.5	74.0	2.09	3.56	22	14	1690	0.02 ± 0.02	och
7	39	23	81.8	101.4	1.80	1.85	11	0	716	0	0
8	17	16	73.4	100.4	1.99	4.20	23	9	1988	0	0
9	31	12	61.6	76.5	1.62	4.72	19	6	257	0	0
10	1	3	55.1	78.0	2.57	6.30	27	16	532	0	0
11	8	6	73.5	97.7	1.82	4.59	22	10	699	0.72 ± 0.05	sf, st
12	13	42	54.3	96.9	1.15	2.06	14	10	1840	0.65 ± 0.00	om, ro, sf, st
13	6	24	56.6	95.9	1.25	2.11	13	9	1005	0.40 ± 0.01	ct, och, om, sf, st
14	25	32	80.2	99.5	1.19	1.88	9	2	216	60.7 ± 0.00	ct, ro
15	66	36	96.7	106.7	1.24	2.04	15	3	2836	0.16 ± 0.00	om, st
16	11	15	64.4	92.6	1.95	3.87	23	14	2137	0.91 ± 0.67	om, st
17	7	8	70.8	94.8	2.25	4.10	24	13	1949	0	0
18	21	28	56.0	62.8	2.06	4.65	23	13	710	0	0
19	10	25	69.8	84.5	2.01	3.17	24	13	2658	0	0
20	3	10	65.4	79.9	1.67	3.06	17	11	758	0	0
21	3	7	62.4	78.3	1.99	3.53	17	10	393	0.72 ± 0.02	sf
22	4	5	60.5	86.5	2.33	4.99	23	15	683	0.28 ± 0.13	sf
23	9	8	58.5	100.2	1.44	3.80	22	14	1260	0.74 ± 0.37	om, sf, st
24	6	2	63.3	80.2	1.87	3.35	19	10	832	0.48 ± 0.00	sf
25	3	19	60.0	96.5	1.27	2.13	19	11	5826	0.60 ± 0.01	sf
26	22	6	73.0	97.9	1.61	4.13	16	6	312	0	0
27	4	6	79.3	96.9	2.04	4.08	27	6	3713	0	0
28	2	10	87.6	96.3	1.88	5.75	27	5	565	0	0
29	56	16	65.6	73.5	2.02	6.03	11	6	37	0	0
30	5	11	67.7	82.4	1.94	4.59	23	12	906	0	0
31	3	31	65.4	93.3	2.09	3.23	24	14	2716	0	0
32	7	12	67.1	103.5	1.45	2.35	17	10	3299	0.93 ± 0.23	st
33	9	1	72.6	97.5	1.36	2.94	19	8	2467	0.42 ± 0.15	st

Environmental Variables

While only 1 of 3 riparian plant community measures (H′-plant) contributed somewhat to CCA results, 12 other environmental variables were more effective in distinguishing invertebrate community assemblages from one another (Figs. 3a, 3b). Particularly influential parameters to axis construction were percent silt and cobble in the stream channel, total dissolved residue, and PO_4–P concentrations. All 6 regression models using aquatic invertebrate community indices as community descriptors were significantly related to reduced subsets of the 15 environmental parameters (Table 4). The number of parameters used in each model ranged from 5 (EPT taxa) to 9 for both diversity measures (H′ and α). The most common environmental parameters used in the models were percent silt, percent cobble, and temperature. Percent cobble was the only significant factor of these 3 for the H′-diversity model. Dissolved oxygen was used in 4 models, but was a significant factor for only CTQ_a and α-diversity. Sample variability in number of organisms collected (E_{Os}) was used in 4 models and was a significant component for all except CTQ_d. Variability in number of taxa collected (E_{Ts}) was a significant component for both CTQ_d and EPT taxa models. Fish diversity helped to explain variability of both diversity measures of invertebrate assemblages. Nitrate concentration was used to describe variability in 3 models, but was significant for only CTQ_a. Although a count of cow dung was included in 3 regression models, it was not a significant factor for any.

Riparian condition measures were significantly negatively correlated to percent silt in the substrate (plant community type: $r = -0.45$, $p \le 0.05$, ecological status: $r = -0.40$, $p \le 0.05$, and H′-plant: $r = -0.42$, $p \le 0.05$). Both riparian classification indices were positively correlated to percent sand (plant community type: $r = 0.44$, $p \le 0.05$; ecological status: $r = 0.37$, $p \le 0.05$), and H′-plant diversity showed a significant positive relationship to percent cobble ($r = 0.42$, $p \le 0.05$).

Proportion of silt in the stream channel was significantly related to riparian community type ($F_{2,5} = 7.64$) but not to ecological status ($F_{5,25} = 1.63$). Mean comparisons illustrated that silt comprised a greater proportion of substrate composition in channels bordered by meadow communities than in sections bordered by either willow (26.0 ± 5.08, $q_{25,3} = 5.12$) or aspen (22.03 ± 5.21, $q_{25,3} = 4.23$; Fig. 4).

DISCUSSION

Riparian and Biotic Stream Condition

Our results show that aquatic invertebrate assemblages in central Nevada are strongly related to a suite of environmental parameters. However, measurable effects of riparian community condition on stream condition were not clearly detectable. This could be due to effects of integrated influences operating at larger spatial scales, such as land use practices in upstream sections of a watershed (Roth et al. 1996). The USFS Ecological Status Riparian Determination scorecard measures plant community composition and soil condition in a small plot along the stream bank, whereas aquatic invertebrate communities may be responding to processes and conditions occurring upstream and between watersheds (Richards et al. 1997). Therefore, while intact riparian corridors are an essential component of a properly functioning stream, vegetation composition and structure may not be directly linked to aquatic communities or habitat conditions at the same sampling sites (Kondolf 1993).

Although our data on aquatic invertebrate communities are limited to 1 yr, they represent the most extensive effort to collect and identify aquatic invertebrates in central Nevada. This collection and the differences we found between sites provide valuable information for those interested in designing future stream condition monitoring programs in this area. Future studies should address the effects of temporal variability on community compositions. A preliminary examination of a reduced set of samples (3 per site) collected in May suggests that community composition differed between the 2 months. Effects of seasonal patterns in the interpretation of bioassessments in other systems have been documented (Furse et al. 1984, Ormerod 1987). Data collected in multiple years would also be useful for determining appropriate sampling time intervals to monitor changes in riparian condition, which could have resulted from either natural fluctuations

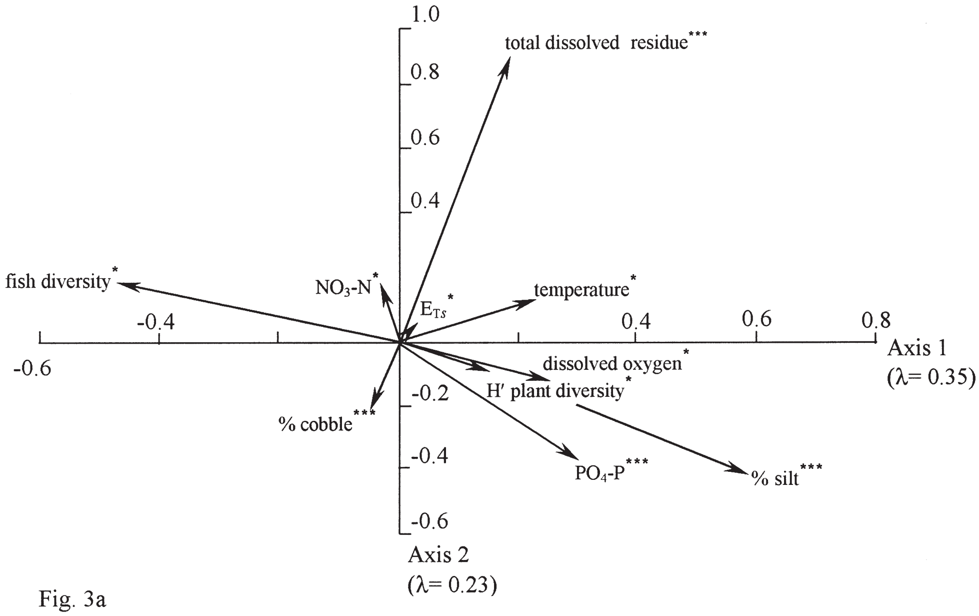

Fig. 3a

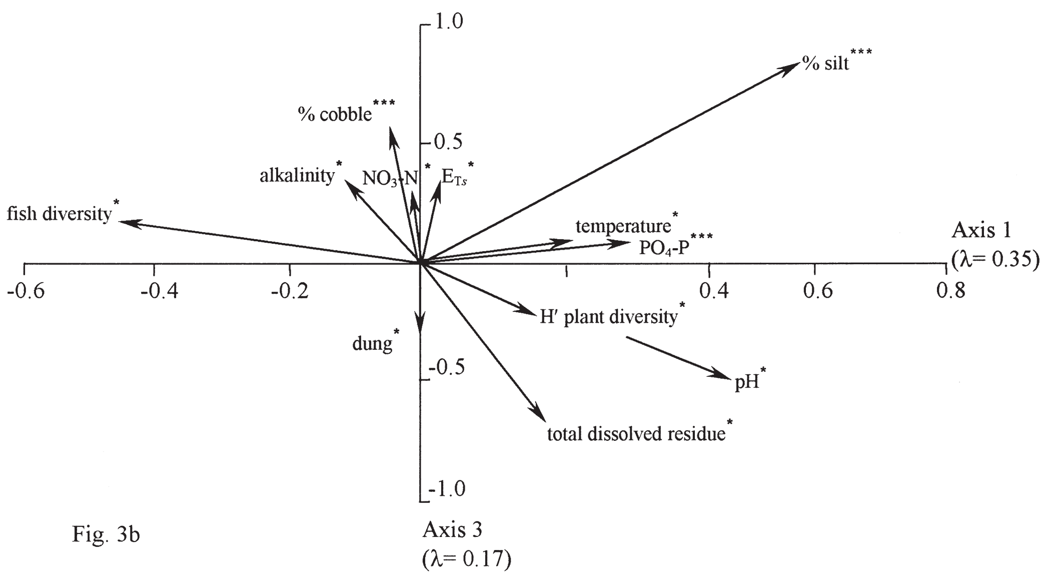

Fig. 3b

Figs. 3a, 3b. Canonical Correspondence Analysis of environmental variables and taxonomic occurrences of sampled aquatic invertebrates, Toiyabe Range, June 1994. Environmental variables used in the data set were percent similarity to PNC, α-plant diversity of riparian communities, H′-plant diversity of riparian communities, discharge (L sec^{-1}), slope (%), % silt, % mud (clay), % sand, % pebble, % cobble, % boulder, % woody debris, NH_3–N (μg L^{-1}), NO_2–N (μg L^{-1}), NO_3–N (μg L^{-1}), PO_4–P (μg L^{-1}), cow dung density (number per acre), temperature (°C), dissolved oxygen (mg L^{-1}), pH, total dissolved residue (mg L^{-1}), alkalinity (μeq L^{-1}), E_{Ts}, fish density (number per m^2) and diversity (number of species collected). Environmental variables that were significantly correlated with at least 1 axis are plotted using standardized canonical coefficients of calculated multiple regression lines. Arrow lengths denote explanatory power of each parameter in describing variability of the invertebrate community structure between sites. Significance levels of Pearson correlations (r) describe associations between measured environmental variables and CCA axis scores ($p < 0.05^{*}$, $p < 0.001^{**}$, $p < 0.0001^{***}$).

TABLE 4. Reduced regression models for aquatic invertebrate community indicators. Independent variables are those parameters that were correlated with at least 1 canonical axis score and which contributed to improvements of R^2 values of >2% upon addition. Intracorrelated parameters ($r > 0.50$) were eliminated from the models. Angular transformations were used for analysis of percentage data, but are reported in original units.

Model Dependent variable	Independent variables	($\beta_i \pm s$)	R^2	(R^2_a)	F-value T-value	P-value
CTQa	7		0.84	(0.79)	18.31	0.0001
	% silt	(0.19 ± 0.036)			4.16	0.0003
	dissolved oxygen	(2.99 ± 0.745)			4.01	0.0005
	temperature	(1.51 ± 0.490)			3.09	0.005
	% cobble	(–0.14 ± 0.041)			–2.80	0.01
	NO_3-N	(0.07 ± 0.024)			2.69	0.01
	ecological status	(–2.31 ± 1.142)			–2.03	0.054
	dung	(–0.01 ± 0.006)			–1.65	0.11
CTQd	7		0.52	(0.38)	3.83	0.006
	% silt	(0.22 ± 0.072)			3.36	0.003
	temperature	(2.12 ± 0.813)			2.61	0.02
	E_{Ts}	(–0.26 ± 0.126)			–2.08	0.048
	NO_3-N	(0.07 ± 0.045)			1.58	0.13
	E_{Os}	(0.24 ± 0.169)			1.43	0.17
	dissolved oxygen	(1.80 ± 1.419)			1.27	0.22
	alkalinity	(–0.01 ± 0.05)			–0.24	0.81
Diversity (H′)	9		0.67	(0.54)	5.22	0.0007
	fish diversity	(–0.19 ± 0.048)			–3.89	0.0008
	E_{Os}	(–0.02 ± 0.005)			–3.68	0.001
	% cobble	(0.01 ± 0.003)			2.30	0.03
	temperature	(–0.06 ± 0.028)			–2.02	0.06
	alkalinity	(0.01 ± 0.002)			1.99	0.06
	dung	(0.01 ± 0.001)			1.80	0.09
	ecological status	(0.13 ± 0.065)			1.94	0.06
	dissolved oxygen	(0.06 ± 0.042)			1.44	0.16
	% silt	(0.01 ± 0.002)			1.26	0.22
Diversity (α)	9		0.63	(0.48)	4.26	0.002
	fish diversity	(–0.61 ± 0.164)			–3.71	0.001
	E_{Os}	(–0.05 ± 0.017)			–3.03	0.006
	dissolved oxygen	(0.37 ± 0.143)			2.59	0.02
	H′-plant diversity	(0.50 ± 0.212)			2.37	0.03
	% cobble	(0.02 ± 0.012)			2.05	0.052
	dung	0.01 ± 0.001)			1.96	0.06
	conductivity	(0.01 ± 0.002)			1.55	0.13
	NO_3–N	(–0.01 ± 0.005)			–1.40	0.17
	E_{Ts}	(0.02 ± 0.014)			1.24	0.23
Total taxa	8		0.62	(0.50)	4.98	0.001
	% silt	(0.0 ± 0.038)			2.52	0.02
	pH	(5.18 ± 2.175)			2.38	0.03
	total dissolved residue	(–29.6 ± 13.09)			–2.26	0.03
	E_{Os}	(–0.16 ± 0.073)			–2.21	0.04
	temperature	(–0.79 ± 0.381)			–2.06	0.050
	fish diversity	(–1.17 ± 0.664)			–1.76	0.09
	E_{Ts}	(–0.08 ± 0.063)			–1.26	0.22
	% cobble	(0.07 ± 0.058)			1.18	0.25
EPT taxa	5		0.78	(0.74)	19.05	0.0001
	temperature	(–0.67 ± 0.194)			–3.47	0.002
	E_{Ts}	(–0.09 ± 0.031)			–2.86	0.008
	% cobble	(–0.06 ± 0.022)			2.69	0.01
	% silt	(–0.06 ± 0.019)			–2.68	0.01
	conductivity	(0.01 ± 0.004)			2.06	0.05

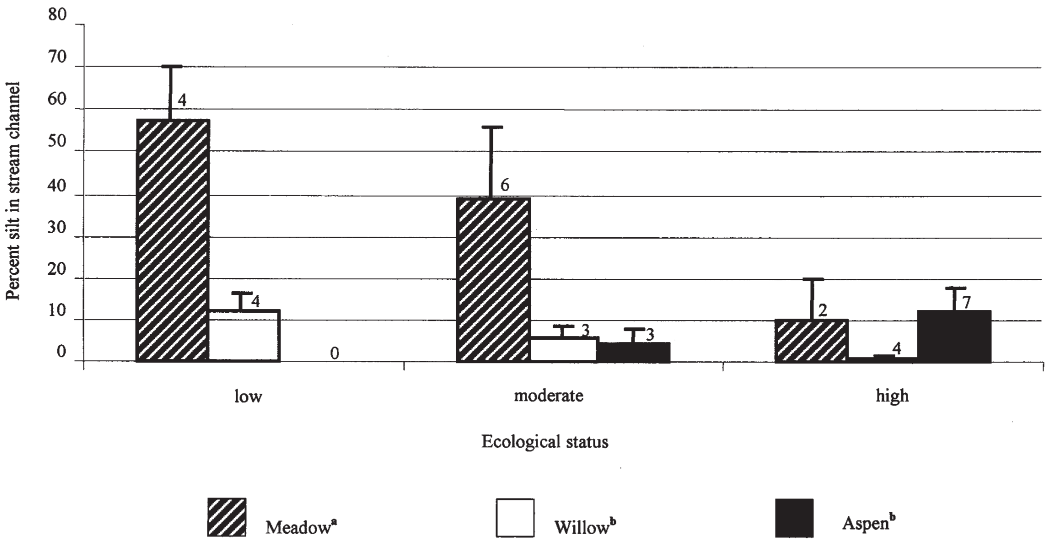

Fig. 4. Proportions of silt classified into a 2-level nested ANOVA: riparian community type (group) and ecological status (within group). Percent silt was normalized using arcsin transformations for analysis, but original units are shown. Sample sizes are listed next to each standard error bar ($\pm 1s_{\bar{x}}$). Riparian communities with the same letter do not differ significantly.

or from restoration efforts and management adjustments. Flooding events in arid regions such as the Toiyabe Range may be particularly influential to associations of riparian community condition and aquatic invertebrates (Fisher et al. 1982, Molles 1985, Grimm and Fisher 1989). Understanding this level of natural variability will assist managers in recognizing background levels of expected changes that result from environmental perturbations (Landres et al. 1999, Swetnam et al. 1999).

Livestock Grazing

The Ecological Status Riparian Determination scorecard was established by the USFS as a method of evaluating the ability of riparian communities to support continued livestock grazing and to recommend rest if necessary for ecosystem recovery. It is well documented that grazing by livestock has been a significant factor in the decline of riparian forests (Keller and Burnham 1982, Platts and Wagstaff 1984, Knapp and Matthews 1996). Livestock can compact soils, exacerbate bank erosion, and consume seedlings and saplings of woody riparian species (Platts 1991, Fleischner 1994). Riparian degradation in the western United States has contributed to the decline of native fisheries and has prompted efforts to restore and protect these resources (Meehan et al. 1977, NRC 1992). A direct measure of historic and current grazing intensity in the Toiyabe Range would have permitted a more thorough analysis of the effects of grazing pressure on stream condition. However, large allotments on the Toiyabe Forest and incomplete information on grazing history made such direct and quantitative measures of livestock use very difficult. We attempted to quantify grazing pressure using fresh cow dung density along stream banks. This proved to be a difficult parameter to measure accurately because excrement from herbivores decomposes gradually. Nonetheless, this parameter did provide some utility in relating invertebrate assemblages to environmental conditions.

Environmental Effects

The information on aquatic invertebrate community metrics was used to evaluate associations between assemblage structure, stream channel characteristics, and water quality. Our analysis demonstrates that aquatic invertebrate community composition has direct relations to sedimentation and smaller-diameter substrate. These results are supported by other investigations where community compositions were affected by substrate characteristics (Lenat et al. 1981, Lenat 1984, Richards and Host 1994). The synergistic relationships we describe suggest that an overabundance of particular types of substratum may effectively reduce or enhance numbers of resident taxa whose abundance is promoted by degraded conditions.

Higher mean tolerance quotients (CTQ_a and CTQ_d) were related to higher stream temperatures, illustrating that more environmentally tolerant taxa can also dominate sites with elevated thermal regimes. EPT taxa were also significantly negatively associated with increases in temperature, providing further evidence that this parameter strongly influences community assemblage structure. Lethal levels for some species of Plecoptera and Ephemeroptera have been recorded at 20°C (Whitney 1939, Nebeker and Lemke 1968), whereas temperatures as high as 40°C can be tolerated by some Odonates (Garten and Gentry 1976, Cherry et al. 1979). While scant documentation on the effects of stream temperature to invertebrate species exists based on field studies, it has been demonstrated that riparian vegetation in the West can strongly influence stream thermal regimes and consequently impact coldwater fish species (Marcuson 1977).

We also demonstrate that the relative amount of silt differs between plant community type and is not as strongly linked to ecological status. Our analysis was based on broad divisions of plant community type and ecological condition categories which may not have discriminated between riparian communities or conditions that exist in the Toiyabe Range, particularly with respect to soil communities. Other factors that could affect sediment deposition rates that were not accounted for include the effects of dissimilar geologic histories. For example, drainage basins originating from either volcanic or bedrock formations may exhibit different aqueous properties due to the unique physical and chemical characteristics of parent soils and water table depths inherent in these land forms (Chambers et al. 1999). Meadow communities of the Toiyabe Range are associated with alluvial fan deposits (Chambers et al. 1998) and are characterized by low gradients, a sandy loam soil type with significant amounts of fine sediment and organic matter, and higher sediment deposition rates (Rundel et al. 1988). Cattle spend more time in open riparian habitat such as this, where bunchgrasses (Gramineae), low shrubs (*Salix* sp.), and sedges (*Carex* sp.) predominate (Northwest Resource Information Center 1993). The USFS may have more frequently found meadow sites to be in low to moderate ecological status due to these combined effects of disproportionate livestock grazing use and soil type.

Community Metrics

Many researchers have speculated on the difficulties of using species richness calculations to detect changes in ecosystem health or disturbance effects (Schluter and Ricklefs 1993, Conroy and Noon 1996). We propose that compositional structure may be a more sensitive indicator of site condition than metrics based on relative abundance, which is consistent with results from other investigations (Lenat 1988, Herbst and Knapp 1999).

Diversity measures may not correctly model extant invertebrate communities because of spatial and temporal aspects of particular study designs as well as taxonomic levels used in identifications (Hughes 1978). The Shannon diversity index (H') reaches its maximum level when all taxa are distributed evenly, which contradicts assemblage patterns based on lognormal models (Goodman 1975). Pielou (1975) asserted that diversity indices should be selected only after species abundance curves show good fit to empirical data. Our χ^2 analysis showed that expected distributions were not significantly different from observed data for all but 6 sites, suggesting that α-diversity was indeed an appropriate metric to use for examining patterns of taxonomic diversity. Explanations of these 6 deviants from expected distributions may not be intuitively obvious, however, since several unique ecological factors may govern taxonomic diversity at these sites. These include the lowest ecological status rating of an aspen-dominated site (Blackbird 29), the highest estimated sample size requirements to reasonably represent taxonomic richness (Kingston 15, Blackbird 29), the highest density of native fish (Indian 14), and the highest level of combined introduced and native fish diversity (Reese 13).

Biotic Interactions

Negative associations of fish diversity and diversity metrics of invertebrate assemblages underscore the importance of quantifying predation levels for community assessments such as this. These results suggest that measuring invertebrate communities for stream condition assessments without any information on associated fish communities may be misleading. Although effects of fish predation on invertebrate communities have been examined for high alpine lakes in the Sierra Nevada Range (Knapp 1996), relatively little is known about the influence of fish species on native aquatic invertebrates in freshwater streams. This information would also help to inform managers of food web dynamics that may be affecting populations of native fishes. Recovery efforts of Lahontan cutthroat trout in drainages of the Toiyabe Range, for example, could benefit from an understanding of cutthroat trout feeding preferences and food availability that may be complicated by possible impacts of competition from introduced fish species.

Summary

The USFS riparian classification system has been shown to be a useful method for managers to assess how land use activities, such as livestock grazing, have affected riparian conditions. However, based on our work in central Nevada, we would recommend the incorporation of a more comprehensive stream condition assessment that would include a sampling program to monitor invertebrate communities and abiotic measures. This information would complement existing methods used to evaluate riparian plant and soil condition and provide critical information on the aquatic component of rangeland ecosystems. An approach such as this would generate a more comprehensive measure of rangeland ecosystem health and better equip resource managers to monitor effects and make reliable adjustments to prevailing land use practices.

Acknowledgments

We thank Ann Dillemuth, Angie Berg, and Libby Nance for help with field and lab work. Kim Gubanich, Candice Brown, and Matthew Setti also helped with specimen sorting. Collaboration with staff of the Region 4 Humboldt-Toiyabe National Forest was essential, and we especially wish to thank Jim Nelson, Kerry Heise, Dave Weixelman, and Desi Zamudio. Our gratitude goes out to Peter Brussard and Dennis Murphy of the Nevada Biodiversity Initiative, which supported this research, and Carol Boggs, Debbie Levoy, Fred Mangum, Andy Weiss, and Colin Brooks who also assisted with various aspects of this work. We also appreciate recommended changes to this manuscript from Jeanne Chambers, Jeff Opperman, Kerry Heise, and 2 anonymous reviewers of the *Western North American Naturalist.*

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Received 7 July 1999
Accepted 10 April 2000