Chapter 6: Meadow-Stream Processes and Aquatic Invertebrate Community Structure

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Introduction

Riparian areas make up less than 1 percent of the total area of the Great Basin, yet they provide many critical ecosystem services, and they support a disproportionately large percentage of the regional biodiversity (Hubbard 1977; Saab and Groves 1992). Jenson and Platts (1990) estimate that over 50 percent of the riparian areas in the Great Basin are in poor ecological condition due to various forms of disturbance and climate change (Chambers and Miller 2004). Ongoing stream incision in the region and progressive degradation of riparian meadow complexes make meadow systems a management priority (Chambers and Miller 2004). Understanding the connections between benthic macroinvertebrate (BMI) communities and meadow-stream environmental characteristics provides managers with important information about the effects of this degradation.

Biological surveys of BMI communities have been used along with water chemistry analyses to indicate environmental conditions in other lotic ecosystems (Yoder and Rankin 1998; Karr and Chu 1999). Benthic invertebrates are relatively long-lived, diverse, and ubiquitous (Linke and others 1999). Due to these factors and because their response to disturbance is broad, they are good indicators of system changes.

Previous ecological research in Great Basin streams of Nevada has focused on links between riparian condition and aquatic invertebrate community structure (Kennedy and others 2000); invertebrate community responses to spring disturbance (Sada and others 2005); and assemblage clustering driven by natural environmental gradients (Myers and Resh 2002). Our study builds, in part, upon the work of Kennedy and others (2000), who found that community composition was strongly related to a number of environmental parameters such as total dissolved residue, fish diversity, and percent silt. Since relationships between landscape and benthic composition have been demonstrated, we sought to examine whether such relationships could be shown using common benthic measurements.

While there have been studies of the nutrient dynamics of central Great Basin streams, little research has documented the temporal and spatial patterns of environmental characteristics and the resulting implications for aquatic invertebrate communities in meadow reaches. Amacher and others (2004) demonstrated that catchment lithology is an important driver of stream water chemistry in Kingston Creek and other upland Toiyabe streams. Mast and Clow (2000) showed that early season snowmelt can dilute aqueous nutrients derived from catchment lithology.

In 2005, we initiated a study to determine if multimetric bioassessment methods that are commonly used by management agencies in the United States are sensitive to riparian meadow influences on benthic communities at Kingston Creek. Comparing multimetric and multivariate methods, we investigated whether a meadow environment affects community structure by sampling invertebrates and environmental characteristics at finer spatial and temporal scales than in previous work.

Methods

Data Collection

We collected invertebrates and environmental data at 12 sites upstream, within, and downstream of Kingston 3 meadow (figs. 1.7 and 6.1). There is a 50-m vertical drop across the 2000-m sampled area. The discharges of multiple springs within the meadow merge into two main tributaries that join the main creek at the bottom of the meadow reach. Sampling was conducted in 2005 in late spring (May), early summer (June), mid summer (July), late summer (August), and early fall (October).

All environmental parameters and invertebrate metrics that were measured are listed in table 6.1. Sites were sampled on approximately two-week intervals for environmental parameters. Depth, current velocity, and dominant substrate size were measured at five equidistant points along a stream cross-section transect for each site (Sanders 1998). Velocity was measured with a Marsh-McBirney Flo-mate 2000 current meter, and discharge was calculated using the cross-sectional area method (Sanders 1998). Dissolved oxygen, temperature, and specific conductivity were measured with a handheld probe (YSI-85). Dominant substrate size was categorized as fine sediment, sand, gravel, cobble, or boulder. Seasonal cover of vegetation over the stream was measured at the center of each transect using a densiometer (Barbour and others 1999). A water sample was obtained at each site using the depth-integrated equal-transit-rate-equal-width-increment method (Amacher and others 2004). Water samples were kept in refrigerated, dark conditions until processing. A mixed subsample was filtered using GF/F filters (0.7 μm) and measured for dissolved nutrients (nitrate, ammonium, soluble reactive phosphorous, and total dissolved phosphorous) and total phosphorus at the Aquatic Ecosystems Analysis Laboratory (AEAL) at the University of Nevada, Reno, and at the High Sierra Water Lab using standard methods (Hunter and others 1993).

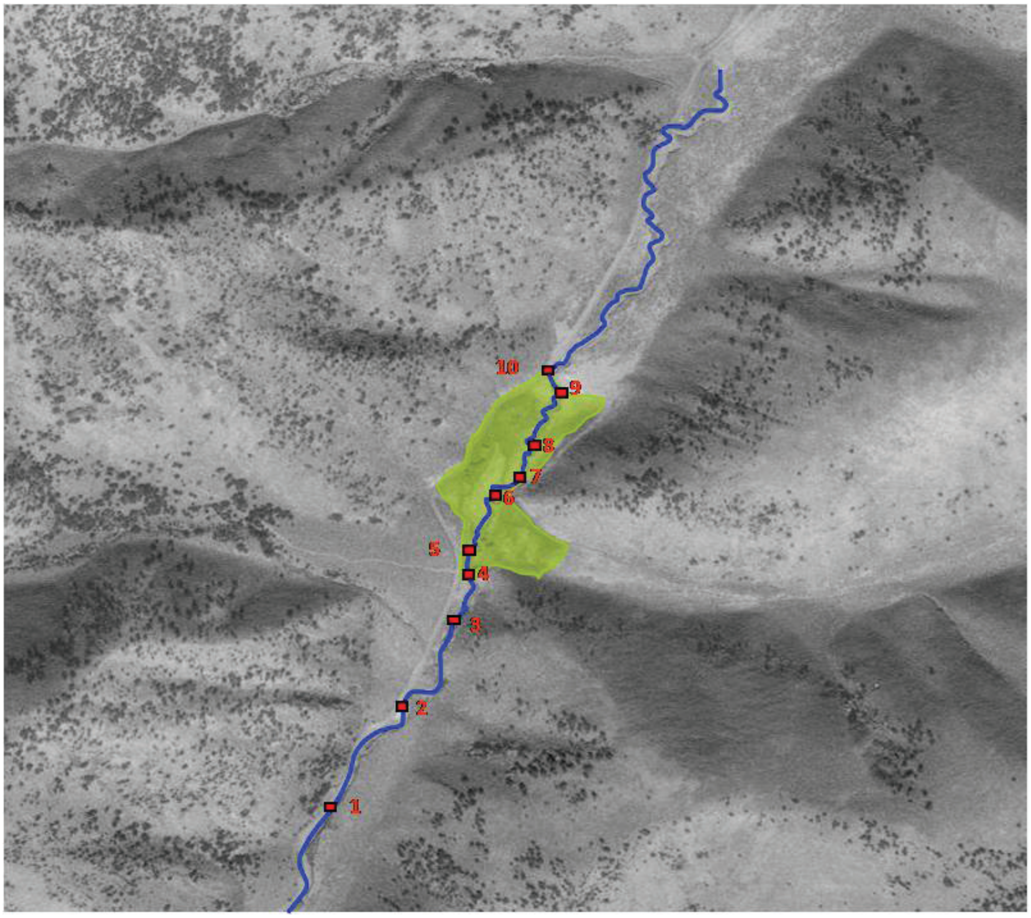

Figure 6.1. Kingston 3 meadow and sampling locations. This meadow lies at an approximate elevation of 2325 m. These sites encompassed close to 50 m in vertical gain and spanned nearly 2000 m of the creek. Sites 1 through 4 (below meadow) and 5 through 9 (meadow) had sufficient stream depths for sampling throughout the sampling season. Sites 10 through 12 (above meadow) were dropped from analysis.

Table 6.1. Environmental variables measured and bioassessment metrics calculated for Kingston Creek. EPT = Ephemeroptera, Plecotera, and Trichoptera.

Environmental parameters	Invertebrate metrics
Stream Discharge	Abundance
Substrate Size	Total Taxa
Percent Vegetative Cover	EPT Abundance
Specific Conductivity	Diptera Richness
Dissolved Oxygen (DO)	EPT Richness
Nitrate (NO_3)	Percent Tolerant Taxa
Ammonium (NH_4)	Percent Intolerant Taxa
Soluble Reactive Phosphorous (SRP)	Percent Dominance
Total Dissolved Phosphorous (TDP)	Percent Non-insect Taxa
Total Phosphorus (TP)	Percent EPT Richness
	Percent EPT Abundance
	Ephemeroptera Richness
	Plecoptera Richness
	Trichoptera Richness
	Percent Chironomidae Richness
	Hilsenhoff Biotic Index (modified)
	Shannon Diversity Index
	Percent Shredders
	Percent Scrapers
	Percent Filterer-Collectors
	Percent Gatherer-Collectors
	Percent Predators

BMI sampling and habitat characterization took place approximately every four weeks. Two invertebrate samples were taken at each site. U.S. Environmental Protection Agency (USEPA) protocols (Barbour and others 1999), UC-Sierra Nevada Aquatic Research Lab protocols (Herbst and Silldorff 2006), and California Department of Fish and Game protocols (CA DFG 2003) were modified for this study. Two BMI samples were obtained from each site using a Hess-type surber sampler (0.105 m^2, 247 µm mesh size). In order to obtain a more accurate representation of community structure within each site, samples were taken from the different microhabitat types that were present (Kerans and others 1992)—one from the thalweg and one from the stream edge. Data from thalweg and edge samples were then combined for each site because effects of environmental parameters on invertebrates are detectable across microhabitats (Parsons and Norris 1996; Rehn and others 2007). Samples were preserved in the field using 70 percent ethanol. Invertebrates were picked from substrate at the AEAL under dissecting microscopes. Invertebrates, excluding early instars, were enumerated and identified to genus—except for oligochaetes and water mites, which were identified to order, and chironomidae, which were identified to sub-family—using Merritt and Cummins (1996), Wiggins (1996), Stewart and Stark (2002), Thorp and Covich (1991), and Post (2005). The California Department of Fish and Game's Aquatic Biology Laboratory at Chico State University verified taxa.

Data Analysis

Cross-sectional stream discharge varied from 0.01 to 0.26 m^3/sec and stream depths ranged from 2 to 34 cm. Over the course of the study, the farthest upstream sites dried up. At sites where stream depths dropped below our ability to sample (5 cm), the sites were removed from our analysis. This threshold eliminated the three sites above the meadow and gradually reduced our meadow sites from six to three over the course of the field season. In addition, the earliest sample points—late spring and early summer—were removed from analysis because invertebrate totals (<500 individuals) were insufficient to meet standard USDA Forest Service/USEPA and California bioassessment criteria (Barbour and others 1999; Herbst and Silldorff 2006). Therefore, we analyzed the remaining two reaches—meadow and below meadow—in mid summer, late summer, and early fall.

For each of the three time periods, we compared invertebrate communities between the two reaches with multi-response permutation procedures (MRPP). Groups were defined by reach (meadow or below meadow) and we examined species composition at each site. The Sørensen distance method (also known as Bray-Curtis) was used in this procedure since it performs well with ecological data (McCune and Grace 2002). MRPP was used for a number of reasons. It is a nonparametric method that was developed for testing group differences (McCune and Grace 2002). It is closely related to non-parametric multivariate analysis of variance (MANOVA) used by Sada and others (2005) for assessing aquatic invertebrate community similarity across disturbance gradients in the Great Basin. Zimmerman and others (1985) applied a variant of MRPP to examine vegetation community differences in the Great Basin.

Stream reaches that MRPP demonstrated to be biologically distinct from one another ($p<0.05$) at the community scale within a given season were analyzed further for reach-specific relationships among invertebrate metrics, invertebrate taxa, and environmental variables. Abundances of individual taxa were compared between reaches using analysis of variance (ANOVA) to illustrate taxon-driven community differences. Dominant taxa were also examined for their possible role in contributing to the differences between reaches and over time.

For the multimetric analysis, we calculated 22 invertebrate metrics based on Herbst and Silldorff's (2006) benthic index of biotic integrity (B-IBI) for the eastern Sierra and western Great Basin (table 6.1). Hilsenhoff biotic index values for individual taxa that indicate tolerance/intolerance to organic pollution were taken from EPA's northwest assessment (Barbour and others 1999). Metric values were aggregated and scaled to 0 to 10 (Barbour and others 1999). B-IBI values were created from composites of scaled biological metrics (Herbst and Silldorff 2006) for each reach by season (table 6.2). We evaluated the B-IBI values for each reach over time and compared reaches. Like MRPP, this approach is designed to illustrate spatio-temporal changes in community composition (Barbour and others 1999).

Environmental variables (table 6.1) were sampled approximately every two weeks—twice as frequently as invertebrate—during mid summer and late summer. During those two seasons, environmental data were averaged from two dates within each season. One sample was taken during early fall. We performed individual ANOVAs for each environmental variable to test whether they differed across reaches. Environmental variables were assessed for normality (skew<2 standard deviations) in PC-ORD. Non-normal distributions were monotonically log transformed. To identify relationships over time, Pearson's coefficients were calculated using invertebrate metrics and environmental data for each season (Myers and Resh 2002). We defined highly correlated relationships as those with r-values greater than $|0.90|$ (Myers and Resh 2002). Linear relationships between environmental variables and invertebrate metrics would indicate the potential of fine-scale biotic and abiotic interactions.

Results

Seventy-two taxa (table 6.3) and 40,494 invertebrates were identified. The multivariate (MRPP) and multimetric (B-IBI) methods we used to compare the meadow and below-meadow communities provided differing results. MRPP indicated that significant community-level differences existed between reaches during mid summer ($p = 0.04$, $n = 9$) and late summer ($p = 0.02$, $n = 8$). No measurable community difference was detected between reaches in early fall ($p = 0.53$, $n = 6$). B-IBI scores, however, were identical

Table 6.2. Taxa list for Kingston Creek upper meadow-stream. Taxa with an asterisk are predators found in early fall. EPA tolerance values assessed for Idaho were used.

Order/Suborder	Family	Genera
Trombidformes/Hydracarina		
Amphipoda	Crangonyctidae	*Crangonyx*
	Gammaridae	*Gammarus*
	Pontoporeiidae	*Monoporeia*
	Gastropoda	*Gastropoda*
Nematoda		
Nematomorpha		
Bivalvia		
Oligochaeta		
Ostracoda		
Ephemeroptera	Baetidae	*Baetis*
		Acerpenna
	Ephemerellidae	*Drunella*
	Heptageniidae	*Cinygmula*
		Epeorus
	Sipholonuridae	*Parameletus*
Plecoptera	Chloroperlidae	*Haploperla*
	Nemouridae	*Malenka*
		Zapada
Trichoptera	Brachycentridae	*Micrasema*
		Brachycentrus
	Glossosomatidae	*Glossosoma*
	Hydropsychidae	*Parapsyche**
		Arctopsyche
		Leptonema
		Hydropsyche
	Hydroptilidae	*Hydroptila*
		Orthotrichia
		Metrichia
	Lepidostomatidae	*Lepidostoma*
		Psycopglypha
	Limnephilidae	*Limnephilus*
		*Hesperophylax**
		Clostocea
	Odontoceridae	*Namamyia*
	Philopotamidae	*Dolophilodes*
	Polycentropodidae	*Polycentropus**
	Rhyacophilidae	*Rhyacophila**
Diptera	Ceratopogonidae	*Leptoconops*
		Culicoides
		*Probezzia**
	Chironomidae	*Orthocladinae*
		Diamesinae
		Chironomini
		Tanypodinae
		Tanytarsini
	Dixidae	*Dixa*
	Psychodidae	*Pericoma/ Telmatoscopus*
	Simuliidae	*Simulium*
		Prosimulium
	Tipulidae	*Antocha*
		*Dicranota**
		Pedicia
Coleoptera	Dytiscidae	*Dytiscus*
	Elmidae	*Stenelmis*
		Atractelmis/ Cleptelmis
		Zaitzevia
		Optioservus
		Gonielmis
		Heterlimnius
		Narpus
		Ordobrevia
		Ampumixis
	Hydrophilidae	*Helobata*
		Laccobius
Lepidoptera	Pyralidae	*Pyralidae*

during mid summer, late summer, and early fall, indicating no substantial metric level difference between the reaches (table 6.2). While the index scores were identical across reaches within season, the scores gradually increased from three to five from mid summer to early fall (table 6.2). To investigate patterns in the taxon-driven distinctions between reaches, we used ANOVA to examine dynamics in individual taxa beginning with the dominant taxa.

The midge subfamily *Orthocladiinae* was the dominant taxon for all sites, regardless of reach, during mid summer (range 43 to 85%). The mayfly *Baetis* dominated sample composition for all sites during late summer (range 45 to 86%). In mid summer, the mayfly *Drunella* and *Orthocladiinae* were found in higher numbers in the meadow reach compared to below meadow, though at marginally significant values ($p = 0.056$ and $p = 0.063$, respectively). During late summer, the mayfly genus *Baetis* occurred in significantly higher numbers at the meadow reach compared to below meadow ($p = 0.042$). Environmental variable comparisons provide additional context for variations found in the invertebrate community.

By fall, there is no flow above the meadow, and the flow within and below the meadow is supported by groundwater inputs. Consequently, discharge was greater at the below-meadow reach during both mid summer and late summer (table 6.4). In mid summer, substrate size was greater at the below-meadow reach of the stream (table 6.4). Specific conductivity values ranged from 353 to 467 µS/cm. Dissolved oxygen levels varied between 4.77 and 7.36 mg/l. Aquatic nutrient ranges are illustrated in fig. 6.2. Specific conductivity and total phosphorous were greater in the meadow reach than below the meadow during late summer (table 6.4; fig. 6.3). This pattern suggests that the two springs within the meadow have distinct influences on water chemistry, most noticeably on specific conductivity and total phosphorous.

The two spring tributaries had distinct impacts on water quality that varied by season (fig. 6.3). Total phosphorous concentrations in the stream increased downstream of the upper spring tributary during late summer (table 6.4; fig. 6.3). Immediately downstream from the lower spring, specific conductivity dropped in mid summer, stabilized in late summer, and increased during early fall (fig. 6.3). Specific conductivity levels typically increased from mid summer to late summer to early fall as the influence of groundwater increased.

Irrespective of reach, a number of relationships between invertebrate metrics and environmental parameters for mid summer, late summer, and early fall were notable. Correlation patterns of those relationships changed across seasons. The number of highly correlated relationships between environmental parameters and invertebrate metrics (with $r>|0.90|$) increased from zero in mid summer to five in late summer to eight in early fall (tables 6.5 through 6.7). During late summer, discharge, dissolved oxygen (DO), and overstory cover were all correlated with three metrics (table 6.6). Discharge and DO both correlated with the percent of intolerant taxa and percent of shredder taxa. Overstory, however, was correlated with the percent of Ephemeroptera, Plecotera, and

Table 6.3. Scaled metric values and index scores for meadow and below-meadow reaches during three seasons.

	Mid summer		Late summer		Early fall	
	Below-meadow	**Meadow**	**Below-meadow**	**Meadow**	**Below-meadow**	**Meadow**
Density	0	3	1	4	3	3
Total Taxa	5	6	4	5	8	7
EPT Richness	2	3	2	3	8	7
EPT Abundance	0	1	1	5	3	2
Diptera Richness	6	8	5	6	6	6
Percent Dominance	8	7	8	8	6	6
Percent Non-insect	1	1	1	1	1	6
Shannon Index	6	6	5	5	8	7
Percent EPT Richness	3	4	5	4	9	8
Percent EPT Abundance	1	1	8	8	7	7
E. Richness	5	7	4	4	6	6
P. Richness	1	0	4	6	5	8
T. Richness	1	2	1	1	8	5
Percent Chironomid Richness	5	4	6	6	3	3
Percent Intolerant Taxa	0	0	1	2	5	5
Percent Tolerant Taxa	1	1	0	0	1	5
Percent Shredders	0	0	1	2	2	5
Percent Scrapers	3	0	0	0	1	1
Percent Filterer-Collectors	1	4	2	2	6	1
Percent Gatherer-Collectors	9	8	9	8	6	8
Percent Predators	2	2	2	1	7	2
Hilsenhoff Biotic Index	7	8	8	8	7	8
Index scores	**3**	**3**	**4**	**4**	**5**	**5**
	n = 4	n = 5	n = 4	n = 4	n = 4	n = 2

Table 6.4. Results from ANOVA among variables between meadow and below-meadow reaches for each season. Single asterisks indicate transformed distributions. Numbers in bold are significant at p<0.05. See table 6.1 for abbreviated variables.

Variables	Mid summer	Late summer	Early fall
Discharge (m3/s)	**<0.01**	**<0.01**	0.46
DO (mg/L)	0.61	0.71*	0.22
Specific Conductivity (μS/cm)	0.17	**<0.01**	0.55
TDP (μg/L)	0.06	0.07	0.08
SRP (μg/L)	0.07	0.1	0.07
NH4 (μg/L)	0.36*	0.75	0.54
NO3 (μg/L)	0.96	0.1	0.1
TP (μg/L)	0.37	**<0.01**	0.08
Overstory Cover (percent)	0.89	0.87	0.37
Substrate Size	**0.02**	1	0.2
Number of sites	n = 9	n = 8	n = 6

Trichoptera (EPT) richness. Four environmental variables were found to correlate with six metrics in the early fall (table 6.7). Three environmental variables (discharge, TP, and NH4) were highly correlated with multiple invertebrate metrics. These results reveal that correlations between environmental variables and community metrics increase from mid summer to early fall. Only in early fall did we see aquatic nutrients correlate with invertebrate metrics. Total phosphorous, ammonia, and nitrate correlated with percent EPT, non-insect abundance, plecoptera richness, percent shredder taxa, and tolerant/intolerant taxa. Pearson's coefficient calculations examined data on a site-by-site basis, indicating the influence of the environmental variables on community structure, irrespective of possible meadow effects.

Discussion

The degradation of central Great Basin meadow ecosystems presents an important management challenge. These meadows are highly productive habitat islands (Sada and others 2005) but they are prone to channel incision and desiccation. Understanding the relationship between changing environmental characteristics and BMI communities

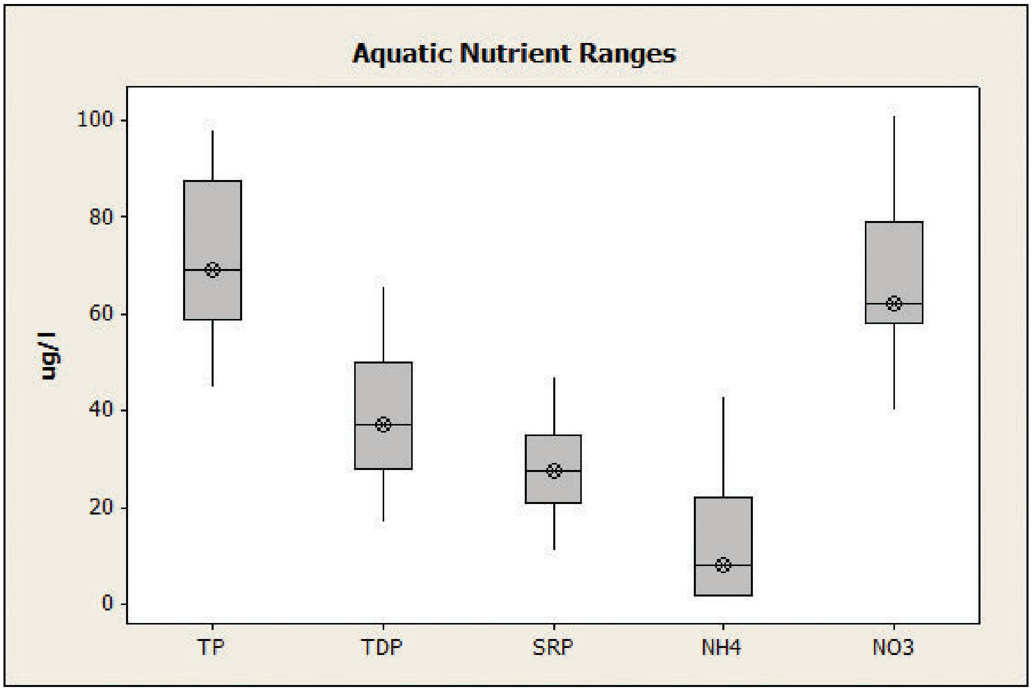

Figure 6.2. Aquatic nutrient ranges for nine sites at Kingston Creek from mid summer through early fall. The 25th and 75th percentiles are shown as a box around the 50th percentile. The error bars represent the highest and lowest values within the upper and lower limits where the upper limit is Q3 + 1.5 (Q3 - Q1) and the lower limit is Q1 - 1.5 (Q3 - Q1).

provides managers with important information about the impact of meadow degradation on aquatic ecosystems. In Kingston Creek, we found strong relationships between environmental characteristics and BMI communities. There were, however, important seasonal differences and differences in the results of the two different analytical methods we used (MRPP and multimetric bioassesments).

The seasonal variation we observed was largely driven by surface water run-off and the relative contribution of groundwater springs. Throughout the season, the water chemistry changed as surface water runoff decreased and the relative contribution of groundwater increased. The two spring tributaries had distinct impacts on water quality that varied by season (fig. 6.3). Total phosphorous concentrations in the stream increased downstream of the upper spring tributary during late summer (table 6.4; fig. 6.3). In addition, Amacher and others (2004) suggest that plant community type may subtly influence water quality on a seasonal basis in these systems. The MRPP analysis illustrated that meadow and below-meadow reaches are biologically distinct from each other during mid and late summer but their populations homogenize during the early fall when surface flows are lowest and the number of correlations between metrics and environmental parameters is greatest. Hannah and others (2007) also found that as spatio-temporal heterogeneity of water sources for mountain streams decreased, invertebrate diversity decreased. Benthic communities in glacial streams also show seasonal variation that is related to the dominant water source (Burgherr and others 2002).

There are several analytical issues that also may have contributed to the fall community homogeneity detected by the MRPP and multimetric method. Two alternative effects may explain this apparent homogeneity. First, because some sites could not be measured in fall, the smaller sample size led to decreased statistical power to detect community differences. Second, sites that were not measured in the fall may have been driving community variability within the meadow. Though both MRPP and the multimetric method describe population homogeneity in the fall, these other factors may be affecting that observation.

Comparing the results of the two analyses demonstrates that these methods analyze different components of community data. Since multimetric bioassesment is based on indices (i.e., diversity, functional feeding groups, and EPT richness), numeric fluctuations in individual taxa can be masked within the metrics. Conversely, because the MRPP tests for community differences at the taxon scale, it does not detect evenly occurring changes in abundance like multimetric bioassessments. An inspection of the dominant taxa in Kingston Creek shows why the two methods differ. While there were taxonomic differences between the two reaches with more *Orthocladiinae* in the meadow in mid summer and more *Baetidae* in the meadow in late summer, raw percentage values of taxonomic dominance remained unchanged. Functional feeding group and tolerance/intolerance metrics also were unable to recognize the community change since *Orthocladiinae* and *Baetidae* have identical values (filterer-collectors and Hilsenhoff biotic index values of five). However, MRPP does detect differences in composition between the reaches during this time period.

Kennedy and others (2000) examined 19 Great Basin streams in early summer and found that substrate size was one of several in-stream characteristics related to invertebrate indices. The strength of the observed relationships may have been, in part, a function of the season in which the data were collected. Studies of both mountain and desert streams have recognized that spatio-temporal variation of invertebrate communities across seasons are driven by environmental variables (Boulton and others 1992; Robinson and others 2001). Beche and Resh (2007) also documented

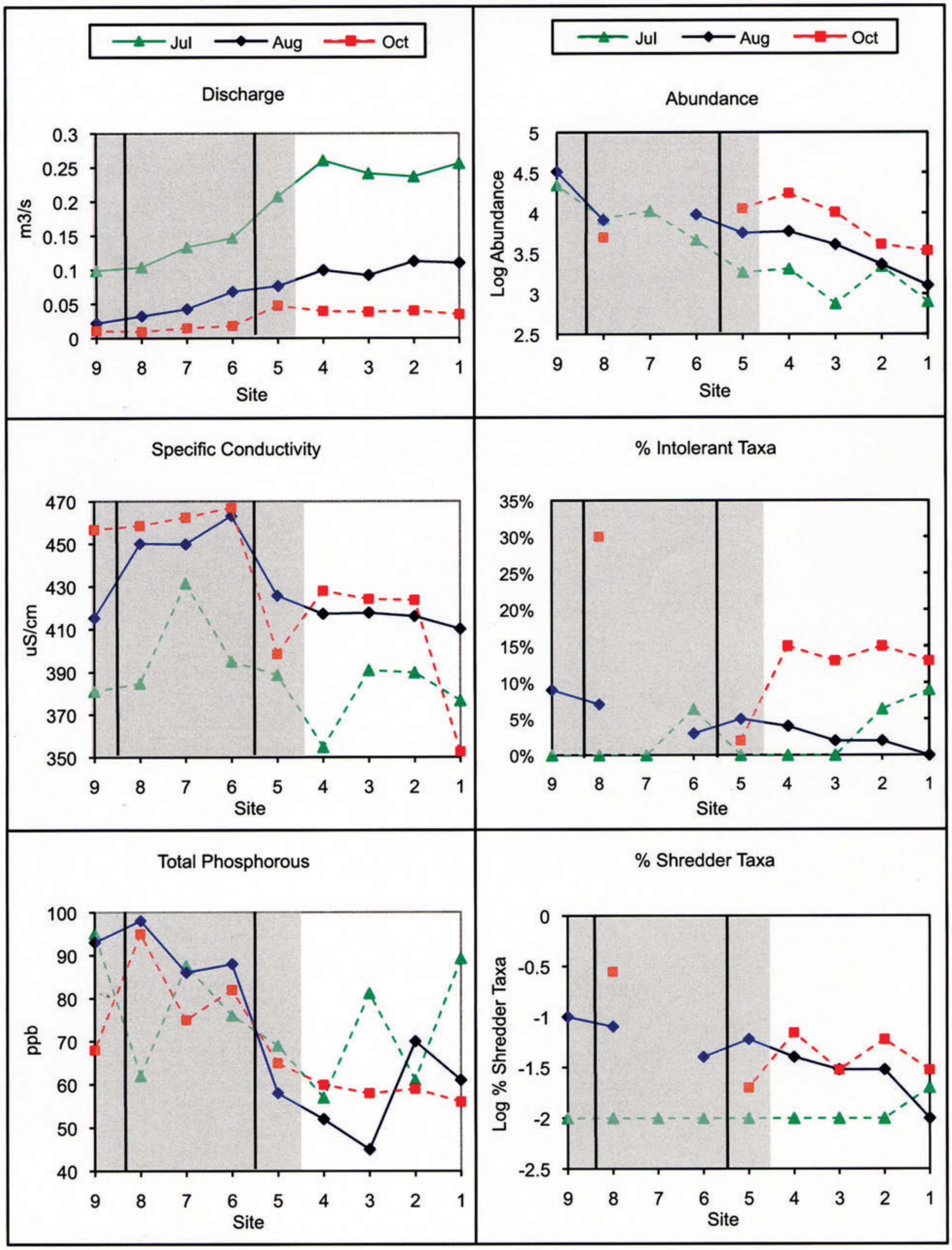

Figure 6.3. Panel illustrating the metrics and environmental variables that were significant in distinguishing meadow and below meadow reaches during late summer. Stream flow is from left to right. Solid trend lines indicate p<0.05. Gray shading represents meadow extent. Vertical lines represent noted springs.

Table 6.5. Pearson's Coefficients for metrics and environmental variables for mid summer. Values representing highly correlated relationships (r>|0.90|) between metrics and environmental variables are boxed and bold. Highly correlated relationships increase in number from mid summer to early fall. See table 6.1 for abbreviations of variables.

	Discharge	DO	Specific conductivity	NH4	NO3
Abundance	-0.81	0.18	0.01	-0.33	0.85
Total Taxa	-0.56	0.56	-0.28	0.05	0.68
EPT Richness	-0.43	0.40	-0.09	-0.23	0.58
EPT Abundance	-0.83	0.23	-0.02	-0.29	0.81
Shannon Diversity	-0.10	0.43	-0.08	-0.03	-0.17
Percent Dominant Taxa	0.66	-0.37	-0.03	0.26	-0.17
Percent Non-insect Taxa	0.43	0.44	-0.22	0.11	-0.08
Percent EPT Richness	-0.26	0.53	-0.06	-0.13	0.31
Percent EPT Abundance	-0.01	0.80	-0.52	0.43	-0.04
Ephemeroptera Richness	-0.34	0.62	-0.21	-0.05	0.37
Trichoptera Richness	-0.49	0.21	-0.04	-0.37	0.68
Percent Chironomid Richness	0.42	-0.69	0.30	-0.03	-0.59
Percent Intolerant Taxa	0.30	0.29	-0.07	-0.03	-0.39
Percent Tolerant Taxa	0.37	0.17	-0.16	-0.10	-0.27
Percent Shredders	0.44	0.17	-0.11	0.03	-0.34
Percent Scrapers	0.40	0.14	-0.13	-0.05	-0.36
Percent Filterer-Collectors	-0.84	0.06	0.05	-0.30	0.52
Percent Gatherer-Collectors	0.71	-0.14	0.02	0.30	-0.40

Table 6.6. Pearson's coefficients for metrics and environmental variables for late summer. Values representing highly correlated relationships (r>|0.90|) between metrics and environmental variables are boxed and bold. Highly correlated relationships increase in number from mid summer to early fall. See table 6.1 for abbreviated variables.

	Discharge	DO	Overstory	TDP	SRP	TP
Abundance	-0.79	-0.73	0.48	0.51	0.68	0.58
Total Taxa	-0.61	-0.63	0.34	0.43	0.64	0.42
EPT Richness	-0.41	-0.53	0.88	-0.04	0.12	-0.11
EPT Abundance	-0.87	-0.82	0.41	0.57	0.73	0.66
Percent Dominant Taxa	0.39	0.49	-0.71	-0.32	-0.44	-0.18
Percent Non-insect Taxa	0.15	0.11	-0.64	0.39	0.48	0.26
Percent EPT Richness	-0.08	-0.27	**0.91**	-0.35	-0.28	-0.50
Percent EPT Abundance	0.22	0.32	-0.66	-0.25	-0.35	-0.05
Ephemeroptera Richness	-0.13	-0.39	0.77	-0.41	-0.25	-0.46
Plecoptera Richness	-0.69	-0.87	0.57	0.37	0.60	0.43
Trichoptera Richness	-0.30	-0.28	0.85	-0.05	0.03	-0.17
Percent Intolerant Taxa	**-0.91**	**-0.91**	0.45	0.45	0.57	0.63
Percent Tolerant Taxa	0.48	0.48	-0.62	0.15	0.18	0.01
Percent Shredders	**-0.92**	**-0.91**	0.40	0.49	0.61	0.68
Percent Scrapers	-0.87	-0.66	0.29	0.65	0.54	0.77
Percent Filterer-Collectors	-0.48	-0.49	0.83	0.16	0.22	0.09
Percent Gatherer-Collectors	0.57	0.59	-0.81	-0.22	-0.31	-0.21

Table 6.7. Pearson's coefficients for metrics and environmental variables for early fall. Values representing highly correlated relationships (r>|0.90|) between metrics and environmental variables are boxed and bold. Highly correlated relationships increase in number from mid summer to early fall. See table 6.1 for abbreviated variables.

	Discharge	Overstory	DO	TDP	TP	NH4	NO3
Abundance	0.42	0.82	-0.32	-0.53	-0.22	0.72	0.64
Total Taxa	0.06	0.86	-0.74	-0.62	0.04	0.32	0.52
EPT Richness	-0.16	0.76	-0.88	-0.54	0.10	0.03	0.40
EPT Abundance	0.06	**0.93**	-0.63	-0.76	-0.03	0.31	0.52
Diptera Richness	0.29	0.78	-0.37	-0.71	-0.19	0.41	0.56
Percent Dominant Taxa	-0.15	-0.80	0.51	0.46	-0.06	-0.57	-0.47
Percent Non-insect Taxa	0.49	-0.26	0.69	0.47	0.00	0.81	-0.02
Percent EPT Abundance	-0.69	-0.43	-0.07	0.08	0.37	**-0.96**	-0.59
Plecoptera Richness	**-0.95**	-0.22	-0.04	0.37	**0.98**	-0.40	-0.84
Trichoptera Richness	0.07	0.84	-0.89	-0.68	-0.13	0.11	0.61
Percent Intolerant Taxa	**-0.94**	0.04	-0.48	-0.07	0.74	-0.72	-0.55
Percent Tolerant Taxa	0.51	-0.23	0.70	0.47	-0.03	0.84	0.00
Percent Shredders	**-0.95**	-0.10	-0.20	0.21	**0.95**	-0.47	-0.74
Percent Filterer-Collectors	0.28	0.83	-0.70	-0.64	-0.46	0.24	0.75
Percent Gatherer-Collectors	0.23	-0.80	0.88	0.57	-0.05	0.04	-0.38
Biotic Index	0.68	0.03	0.53	0.26	-0.28	**0.97**	0.28

inter- and intra-region temporal variability of invertebrate assemblages in California streams. Maloney and Feminella (2006) found high seasonal variability in individual metrics when examining stream disturbance gradients and concluded that such metrics would not be appropriate to use alone. We found that in Kingston Creek, the relationship between environmental variables and invertebrate communities changed throughout the seasons and the reach differences were largest in mid summer when there is a mix of surface and groundwater.

Management Summary

These findings have important implications for managing aquatic resources in Kingston Creek and other Great Basin stream systems. The fact that the stream systems are incision prone makes establishing reference sites a formidable task. As a result, effective assessment and monitoring of these stream and meadow ecosystems require the following:

1. Locate monitoring sites for aquatic invertebrates based on an understanding of the past and present disturbance regime and of surface and groundwater dynamics. Because many Great Basin streams are incising, sites should be located in areas with minimal risk of incision (Chapters 3, 5, and 7). Also, to determine the relative influence of groundwater inputs on aquatic invertebrates, sites ideally would be paired and would include locations with and without groundwater inputs (Chapter 4). Finally, sites should be located in areas that are known to have perennial flow.
2. Replicate sites and include a minimum of three samples per site. Because watersheds differ both in the tendency to incise (Chapters 3, 4, and 5) and in water chemistry (Amacher and others 2004), sites should be replicated across watersheds. Spatial variability in groundwater inputs, even within seemingly similar stream reaches, necessitates multiple samples per site.
3. Collect data across multiple years and seasons. Stream flow is highly variable both among and within years in Great Basin watersheds. Our data show that aquatic invertebrates are highly responsive to the variability in stream flow. To ensure an accurate representation of the diversity and abundance of different taxa, it is necessary to sample aquatic invertebrates both among and within years.
4. Use multivariate tests such as MRPP or non-parametric MANOVA to ecologically and statistically validate multimetric index results. Multimetric and multivariate methods have different means of examining benthic communities. Since multimetric bioassessment methods can fail to recognize fundamental benthic community dynamics that are critical for informed managerial decisionmaking, multivariate testing provides a crucial technique for verifying multimetric results. Multimetric methods compare metrics that are commonly composed of groups of taxa while the multivariate method used in this study examines the entire population at greater resolution—taxon by taxon.
5. Generalized linear mixed models (GLMM) may offer another tool for describing population responses to environmental effects and for describing those population

distributions. GLMM could be used in place of the multiple ANOVAs and correlation calculations performed in this study and may not require distribution transformations.

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