Effectiveness Monitoring in Whychus Creek; Benthic Macroinvertebrate Communities in 2005, 2009, and 2011-2015

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Abstract

This project uses aquatic macroinvertebrates as biomonitoring tools to assess the ecological effects of multiple restoration activities in Whychus Creek in Sisters, OR. Benthic macroinvertebrates were collected via kick net sampling from riffles in 10-13 reaches along Whychus Creek from RM 30.25 to RM 0.5 in late August of 2005, 2009, and 2011-2014. Xerces staff trained local volunteers in standardized sampling techniques for wadeable streams used by the Oregon Department of Environmental Quality (OR DEQ), and these trained teams collected macroinvertebrate samples from riffles at pre-determined sites. Samples from each reach were sub-sampled to a target count of 500 organisms and identified to the lowest practical level of taxonomic resolution (generally genus).

Analysis of the macroinvertebrate community in Whychus Creek using the PREDATOR predictive model and invertebrate-based indices of biotic integrity developed by OR DEQ, and a suite of taxonomic and ecological traits, reveal multiple changes since 2005. Similarity between duplicate samples taken by volunteers and Xerces staff at two sites in each year for quality assurance indicates that volunteers implement the protocol successfully, such that changes in the macroinvertebrate community are due more to altered habitat conditions than differences between operators. The community in 2015 appears to have been affected by the exceptionally dry hot spring and summer, as well as newly-implemented restoration work in the upstream reaches, since organismal abundance, community richness, and diversity of the sensitive EPT (Ephemeroptera, Plecoptera, and Trichoptera) were significantly lower among upstream reach sites compared to previous years. Even with these additional stressors, however, site IBI scores in 2015 were the same as or higher than the 2014 scores for seven of the 13 sampling sites, with 10 sites scoring as having slight disturbance, and three as having moderate disturbance. PREDATOR observed/expected (O/E) scores in 2015 were the same as or higher than the 2014 scores at 11 of the 13 sites, with three sites scoring as having poor biological condition, seven sites as fair, and three as good—the first “good biological condition” scores seen among any sites since 2009.

No dramatic changes in IBI or PREDATOR scores have occurred throughout this study. However, taxa identified by the PREDATOR model as “increasers” in each year include several genera of EPT and increased numbers of OR DEQ cool temperature indicators, while decreaser communities have contained increasing numbers of warm temperature and high sediment indicator taxa, and the mean percent fine sediment (%FSS) optima have been lower in the increaser community compared to the decreaser community in every year. Missing and replacement taxa identified by PREDATOR in each sampling year have included both sensitive and tolerant taxa, but a shift has occurred from earlier years in which replacement taxa had higher mean temperature optima than missing taxa, to a significantly lower temperature optima among replacement compared to missing taxa followed by no strong differences between the two. The %FSS optima of the replacement community have been significantly lower than that of the missing community in nearly every sampling year.

Overall, taxonomic differences are greatest between the 2005/2009 macroinvertebrate community and all subsequent sampling years, including 2015, with changes including a greater abundance of sensitive and/or lotic-associated taxa. This is further reflected by a sustained decrease in mean temperature optima for the communities at downstream and mid-stream sampling reaches across time, and an initial decrease in mean %FSS optima of the stream community. Analysis of functional feeding groups (FFGs) has been less informative, with more generalist feeding groups (predators and collector-gatherers) dominating the community in all years.

New taxa are collected in each year; most are in the sensitive EPT group, including all three of the new taxa found in 2015. The number of new taxa has decreased with each sampling year, suggesting that colonists from the existing pool in surrounding habitats have filled available niches under altered stream conditions and that the community may be stabilizing, although continuing restoration work and increasingly severe climate conditions are likely to impose additional stresses. Overall, since restoration began in 2005, the diversity of sensitive taxa in Whychus Creek has increased, and the community has shifted towards taxa with lower temperature and sediment optima.

Project Background and Summary

Bioassessment

Freshwater ecosystems are severely impacted by human activities, with 45% of US waters classified as endangered or impaired (US EPA, 2013). Stream restoration projects have intensified, but determining their ecological success requires baseline assessment and long-term post-project monitoring of biotic communities (biomonitoring), and an analysis of how changes in community composition relate to stream ecosystem functions (Bernhardt et al., 2005). Because biomonitoring evaluates the condition of biological communities inhabiting a water body, such as plants, amphibians, algae, diatoms, or invertebrates, it provides data about stream “health” that physical and chemical data alone do not address (Rosenberg & Resh, 1993; Karr & Chu, 1999). The structure of biological communities changes in response to habitat impairment, based on individual species’ sensitivity or tolerance to different stressors. The communities assessed must generate a biological “signal” based on human impacts that can be detected apart from the “noise” of normal variation in space and time (i.e. changes in season or stream order). Benthic macroinvertebrates (BMIs) are useful for biomonitoring because they are critical to the food web, are confined to water for most or all of their life cycle, respond in a known way to a range of human-induced stressors, have a short generation time that allows changes in community structure to be detected rapidly, are ubiquitous and abundant, and their sampling and identification is relatively straightforward, standardized, and cost-effective.

Many restoration projects are undertaken with the assumption that improved physical habitat automatically increases faunal biodiversity, which in turn restores impaired or lost ecological processes, i.e. “the field of dreams hypothesis” (Palmer et al., 1997). However, a variety of reach- and catchment-specific influences must be considered when evaluating project outcomes (Roni et al., 2002; Bond & Lake, 2003; Palmer et al., 2005; Lake et al., 2007. Restoration can improve habitat and water quality at the reach level, but streams experience watershed-wide stressors that site-specific activities may not completely remediate (Booth & Jackson, 1997; Bohn & Kershner, 2002; Bond & Lake, 2003). The composition and mobility of the regional pool of colonists must also be considered when assessing post-restoration changes in biological communities (Bond & Lake, 2003; Blakely et al., 2006), and because the recovery process can take years, a long-term monitoring plan is imperative.

Whychus Creek Project

The aquatic macroinvertebrate community in Whychus Creek was sampled in 2005, 2009, and 2011-2015 at 10-13 reaches from RM 30.25 to RM 0.5, to assess changes in community composition following multiple restoration projects. This long-term effectiveness monitoring also engages local stakeholders, as annual macroinvertebrate sampling is done by community volunteers trained by Xerces staff. This study uses multiple approaches to examine the benthic macroinvertebrate community and the ecological implications of changes in community composition. Standard biotic assessment techniques, including the general invertebrate Index of Biotic Integrity (IBI), Grande Ronde IBI (GR-IBI), and the PREDATOR predictive model (Hubler, 2008), all developed by the OR DEQ, are applied annually to assign levels of biological impairment to sampling reaches. Changes in individual IBI metrics as well as community tolerances for percent fine sediment and temperature, relative diversity and abundance of OR DEQ temperature and sediment indicator taxa, and functional feeding group composition are also assessed. Multivariate analysis is done to identify taxa contributing the most to observed community differences from year to year.

Methods

Sampling Sites

Thirteen reaches were sampled along Whychus Creek in 2015, with duplicate samples taken at two sites for quality control (Table 1). Some sites have been added or removed from the slate since 2005 based on access, changes in land use, or re-assessment of their importance, and some site names changed following more precise GIS mapping done in 2014. Sampling sites are distributed broadly into downstream (RM 0.5- 11.5; DS), mid-stream (RM 18-19.5; MS), and upstream (RM 23.5 - 30.25; US).

Table 1. Whychus Creek sampling sites, 2005-2015.

Site ID	Description	Coordinates	Year(s) sampled
WC0050	RM 0.50	44.45682, -121.34028	2005
WC0150	RM 1.5, d/s Alder Springs	44.44491, -121.34543	2009, 2011-2015
WC0300[a]	RM 3, u/s Alder Springs	44.43458, -121.35976	2005
WC0600[b,d]	RM 6, u/s Rd 6360	44.40412, -121.40259	2005, 2009, 2011-2015
WC0650	RM 6.5, u/s Rd 6360 crossing	44.396799, -121. 404268	2013
WC0850[d,f]	RM 8.75, Rimrock Ranch d/s	44.391278, -121.406182	2011-2015
WC0900	RM 9, Rimrock Ranch	44.384198, -121.407892	2005, 2009, 2011-2015
WC0925 [e]	RM 9.25, Rimrock Ranch	44.380815, -121.408592	2013
WC1025	RM 10.25, Rimrock Ranch u/s	44.371534, -121.415865	2011-2012, 2014-2015
WC1100	RM 10.25	44.364587, -121.421706	2014-2015
WC1150	RM 10.75	44.361288, -121.427525	2014-2015
WC1800	RM 18	44.328342, -121.494534	2005
WC1825	RM 18.25, d/s end DBLT property	44.32781, -121.495406	2009, 2011-2015
WC1850	RM 18.5, DBLT property	44.326601, -121.500229	2009, 2011-2015
WC1900	RM 19, DBLT property	44.321523, -121.507461	2005, 2009, 2011-2015
WC1950[g]	RM 19.5, d/s Camp Polk Bridge, DBLT	44.318741, -121.514961	2009, 2011-2015
WC2350	RM 23.5, Perit Huntington Rd.	44.29066, -121.53064	2005
WC2425[c,e,g]	RM 24.25, City Park, d/s gauge	44.287806, -121.544229	2005, 2009, 2011-2015
WC2600[c,f]	RM 26, 4606 Rd. footbridge	44.2730592, -121.555297	2005, 2009, 2011-2015
WC2650	RM 26.5, d/s TSID	44.256434, -121.550692	2011
WC2700	RM 27, u/s TSID	44.250744, -121.549892	2011
WC3025	RM 30.25, OWRD gauge	44.233647, -121.567105	2005, 2009, 2011

Superscripts indicate sites where duplicate samples were taken for quality control: [a] 2005 duplicate; [b] 2009 duplicate; [c] 2011 duplicate; [d] 2012 duplicate; [e] 2013 duplicate [f] 2014 duplicate; [g] 2015 duplicate.

Volunteer Training

Volunteer training and stream sampling was done on a single day in the same index period across all years (August 17-20). Volunteers assembled at City Park (Sisters, OR) to be trained by Xerces staff in standardized macroinvertebrate monitoring protocols for Oregon's wadeable streams (OWEB, 2003). The sampling technique was demonstrated and each item on the data sheet was explained. A handout with detailed sampling instructions and field guides to Northwest stream macroinvertebrates (Adams *et al*., 2003) were provided, although volunteers did not identify organisms. The group was divided into teams of three to four people, each equipped with a D-frame kick net with 500 μm mesh; metal 500 μm sieve, 500 μm mesh Nitex membrane square; squirt bottle; plastic spoons; forceps; thermometer; tape measure; 10-gallon plastic bucket; hand lens; 1-liter Nalgene sample jars half-filled with 80% ethanol; datasheets; jar labels; clipboard; and detailed instructions for finding sites and the upstream and downstream ends of sampling reaches. Volunteers also had cell phone numbers of Xerces and UDWC staff, and staff moved among sites throughout the day to answer questions. Samples were returned to the park at the end of the day and inspected by Xerces staff to be sure each was properly labeled and preserved.

Sampling Techniques

Benthic macroinvertebrates were collected from riffle habitats according to standardized protocols developed by OR DEQ for Oregon's wadeable streams (see OWEB, 2003). Sample reach lengths were calculated as 40 times the average wetted width of the stream at the sampling point, with a minimum of

500 feet and a maximum of 1000 feet. Watershed council staff performed these calculations and flagged the upstream and downstream extent of each reach a few days prior to sampling.

A sample from a single reach consisted of composited net sets taken from eight different randomly selected riffles within the designated reach. Each of the eight net sets was collected from a 1 ft^2 area using a 500 μm D-frame kick net. In reaches with fewer than eight riffles, two kick net samples were taken in each of four riffles. Large rocks and debris in the sampling area were first rubbed and rinsed into the net to dislodge and collect any clinging organisms and set aside. The substrate was then disturbed thoroughly using a boot heel to a depth of 6-10 cm for 1-2 minutes. The eight net sets at each site were pooled into a bucket; large debris was rinsed and removed, and any vertebrates such as fish were noted and carefully replaced in the stream. Sample material was concentrated by being poured through a 500 μm sieve lined with a flexible square of 500 μm Nitex membrane; the membrane was then lifted out and the concentrated sample carefully scooped and rinsed into the sample jar.

Samples with excessive amounts of sand and/or gravel were elutriated to separate the lighter invertebrates from heavier mineral material and allow them to be jarred separately to avoid crushing or grinding specimens. Elutriation was done by adding water to the sample bucket, swirling it thoroughly, then pouring the suspended organic material through the sieve. After two to three rinses, the organic material was placed in one sample jar and the mineral material in another; all sample material from each site was retained for subsequent examination in the lab so that heavy-bodied organisms (i.e., snails, stonecase-making caddisflies) were not lost. Jars were filled no more than halfway with sample to ensure preservation, and the ethanol was replaced with within 48 hours to maintain the 80% concentration, since water leaches from the sample and dilutes the preservative. A simple physical habitat assessment was done at each site to provide data on human use and landscape alterations, substrate composition, water temperature and appearance, and wetted width and depth at each riffle sampled (see Appendix A for datasheet).

Macroinvertebrate Identification

Sample identification was done by Cole Ecological, Inc. Each composite sample was randomly sub-sampled to a target count of 500 organisms. If the sample contained fewer than 500 organisms, the entire sample was picked and identified. All organisms were identified to the level of taxonomic resolution currently used by the OR DEQ (genus and species). If a specimen was too immature for key taxonomic characters to be fully developed or visible, identification was done only to the family.

Data analysis

Invertebrate Index of Biotic Integrity (IBI)

Biological condition of each sampling site was assessed using both the OR DEQ general IBI and the more regional northeastern (Grande Ronde) IBI (Table 2), and the correlation between the two sets of scores was determined. Ranges used to calculate the scaled score of each metric in the IBIs were created based on community composition in reference streams determined previously by OR DEQ. A higher scaled score (5) is given to metric ranges typical of a healthy stream, while a lower scaled score (3 or 1) reflects values associated with more degraded conditions. Some metrics are thus positive (a higher raw value receives a higher scaled score) while others are negative (a higher raw value receives a lower scaled score). Scaled scores for all metrics are summed to generate a single value that reflects the level of site impairment.

All sites from all sampling years were assessed using both IBI models and changes across years in mean raw metric values as well as mean scaled values among upstream, mid-reach, and downstream sites. Two-tailed t-tests were done to determine significance of differences between years.

Table 2. OR DEQ general IBI and Grande Ronde IBI metrics and scoring. Both use taxa identified to genus.

	OR DEQ IBI				Grande Ronde IBI		
	Scoring Criteria						
Metric	5	3	1		5	3	1
POSITIVE METRICS							
Taxa richness	>35	19-35	<19		>31	24-31	<24
Mayfly richness	>8	4-8	<4		>7	6-7	<6
Stonefly richness	>5	3-5	<3		>6	5-6	<5
Caddisfly richness	>8	4-8	<4		>4	2-4	<2
# sensitive taxa	>4	2-4	<2		>4	3-4	<3
# sediment-sensitive taxa	≥2	1	0		>1	1	0
NEGATIVE METRICS							
% dominance*	<20	20-40	>40		<39	39-42	>42
% tolerant taxa	<15	15-45	>45		<24	24-36	>36
% sediment-tolerant taxa	<10	10-25	>25		<10	10-15	>15
MHBI**	<4	4-5	>5		<3.9	3.9-4.3	>4.3
Summed Score & Condition							
Severely impaired	<20				<15		
Moderately impaired	20-29				15-25		
Slightly impaired	30-39				N/A		
Minimally/not impaired	>39				≥26		

**For general IBI, dominance of the top (most abundant) taxon is assessed; for Grande Ronde IBI, abundance of the top 3 taxa is assessed.*

***MHBI = Modified Hilsenhoff Biotic Index, reflecting tolerance to organic pollution/enrichment; individual taxa MHBI values may range from 1 (low tolerance) to 10 (high tolerance).*

PREDATOR model

The PREDATOR model (Predictive Assessment Tool for Oregon; Hubler, 2008) was developed for two major regions in Oregon: Marine Western Coastal Forest (Willamette Valley and Coast Range ecoregions; MWCF) and Western Cordillera and Columbia Plateau (Klamath Mountain, Cascades, East Cascades, Blue Mountains, and Columbia Plateau ecoregions; WCCP). The model calculates the ratio of taxa observed at a sampling site to taxa expected at that site if no impairment existed (O/E), based on community data collected previously at a large number of reference streams. The model incorporates environmental gradients such as elevation, slope, and longitude to select the most appropriate reference streams. An O/E value of less than one indicates loss of taxa, while values greater than one indicate taxa enrichment, potentially in response to pollution or nutrient loading.

PREDATOR scores are generated from a habitat file and a macroinvertebrate data file for each site, which are uploaded to the model software at the Western Center for Monitoring & Assessment of Freshwater Ecosystems (www.qcnr.usu.edu/wmc). Model outputs include a site test result, which indicates whether the habitat data falls within the model parameters; an O/E score for each sample, which provides a measure of biological condition; a probability matrix indicating taxa expected to occur at each site but absent, and taxa not expected to occur at the site but which are present (missing and replacement taxa, respectively); and a sensitivity index that reveals “increaser” and “decreaser” taxa in the community. Invertebrate community data were submitted to the PREDATOR WCCP model (Hubler,

2008). O/E scores associated with a probability of capture (P_c) > 0.5 were used (i.e. the model considers only invertebrates with over 50% likelihood of being collected at reference sites). In the WCCP model, site biological condition is assigned based on the following O/E scores: ≤0.78 = poor (most disturbed); 0.79 – 0.92 = fair (moderately disturbed); 0.93 – 1.23 = good (least disturbed); and >1.23 = enriched.

Predictive models are often considered to be more sensitive and accurate than IBIs, but it should be noted that the PREDATOR model has not been re-calibrated since it was created using stream survey data from 1998-2004. Periodic sampling of reference streams used to build the models and model re-calibration is needed to detect any shifts arising from altered environmental conditions such as climate change, so it is possible that the sensitivity of this model has changed over time, especially as the WCCP model applicable to the Cascades has lower precision than the MWCF model developed for the Willamette Valley (Hubler, 2008; Hubler, pers. comm.). In addition, mean annual rainfall at the sites along Whychus Creek is at the very low end of the annual rainfall range at the reference sites from which community data are drawn for analysis (Hubler, pers. comm.), which may affect model accuracy.

Temperature and Sediment Optima

Differences in macroinvertebrate community composition may be driven by stream temperature and the amount of fine sediment in the substrate. OR DEQ developed a dataset of individual taxa optima values for seasonal maximum temperature and percent fine sediments (i.e. the temperature or %FSS under which a taxon can maximize its abundance). Temperature and sediment optima of increaser vs. decreaser taxa and missing vs. replacement taxa were examined to diagnose whether the community was responding to changes in temperature and/or sediment conditions. The weighted mean temperature and sediment optima of the macroinvertebrate community at each sampling site were also examined across sites and years. The presence of taxa considered by OR DEQ to be indicators (i.e. taxa with the strongest responses to environmental gradients) of cool or warm temperatures and high or low fine sediment conditions (Hubler et al., 2008; see Appendix B for a list of OR DEQ indicator taxa) was also noted among missing, replacement, increaser, and decreaser groups.

Taxonomic and Ecological Trait Analysis

Multimetric and multivariate models routinely examine taxonomic differences among biotic communities; the identity of species within the community serves as a surrogate for the attributes they possess that are affected by changing environmental conditions (Southwood, 1977). A prime illustration is the ubiquity of the "EPT" metric, which looks at the richness and/or abundance of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera), taxa considered as a whole to be the most sensitive to decreased flows and/or increased temperature, sedimentation, and pollution. However, ecological traits, which involve properties such as trophic guild, body size, or number of generations per year, can also be examined to assess site conditions in conjunction with taxonomic traits (Pollard and Yuan, 2010; Culp et al., 2011; van den Brink et al., 2011; Lange et al., 2014). Ecological traits provide additional insights into mechanisms structuring the community at a site, and assessing a combination of taxonomy- and biology-based traits can increase the likelihood of identifying recovery of macroinvertebrate assemblages (Arce et al., 2014).

To incorporate ecological traits into bioassessment of Whychus Creek, a functional feeding group (FFG) designation was assigned to each taxon in the dataset from 2005-2015. Designation of a taxon as a predator, scraper, shredder, collector-filterer, or collector-gatherer was done according to Merritt et al. (2008). Richness, relative diversity, and relative abundances of taxa in different FFGs was determined for each year's dataset, and changes in distributions among downstream, mid-reach, and upstream sites were examined.

In addition, as the OR DEQ IBI incorporates both taxonomic and ecological (sensitivity/tolerance) metrics, changes across years in the raw scores of individual IBI metrics were examined for each site, as well as changes in the mean richness, relative diversity, and relative abundance of combined EPT taxa.

Community Similarity Analysis

Additional analyses to detect patterns in macroinvertebrate community composition were done using the PRIMER v6 ecological community statistics software package (Clarke & Warwick, 2001). CLUSTER analysis of a Bray-Curtis similarity matrix of square-root transformed abundance data was done to investigate macroinvertebrate community similarity between sites and across years. To examine the communities of increaser/decreaser and missing/replacement taxa, CLUSTER analysis was done on a presence/absence abundance dataset. Community similarity was also subjected to ordination by non-metric multidimensional scaling (MDS) to investigate site assemblage similarity based on sampling year and stream reach location. SIMPER analysis was used to assess the taxa that contributed the most to community differences between years.

Results and Discussion

Indices of Biotic Integrity

OR DEQ General IBI

In 2015, 10 of the 13 sites sampled (77%) received IBI scores indicating slight biological impairment, and three sites (23%) received scores indicating moderate biological impairment. These scores follow the trend seen in previous years, with most sampling sites scoring as slightly impaired, and the number of slightly impaired sites increasing over time while the proportion of moderately impaired sites has shown a slight downward trend (Figure 1). No sites have ever received a score indicating severe levels of impairment, but few to no sites can be considered unimpaired or minimally impaired under this IBI. The proportion of minimally impaired sites has ranged from 8-18% in past years, but since 2013, no sampling site has received a general IBI score high enough to qualify as minimally impaired (Table 3).

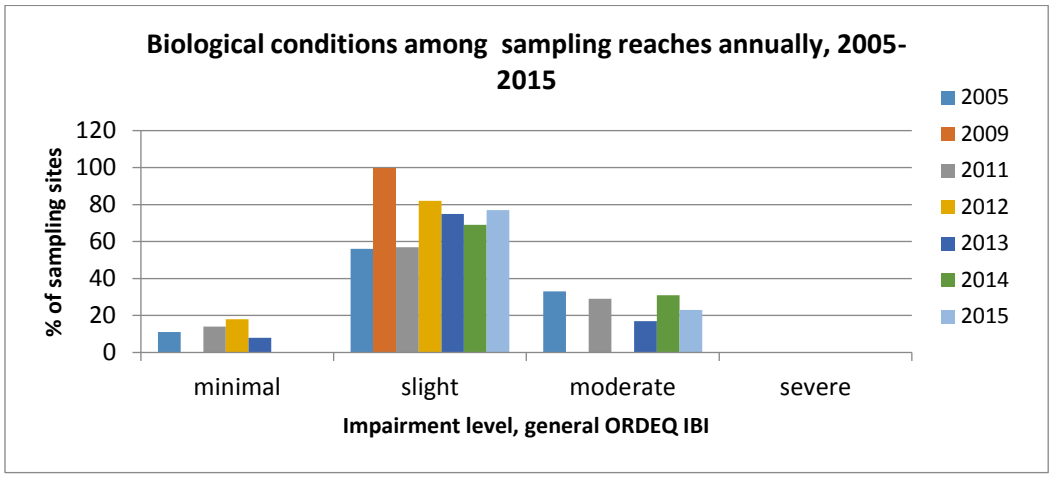


Figure 1. Comparison of site biological condition categories across time

Table 3. OR DEQ IBI scores for all sites sampled between 2005 and 2015. Colors indicate minimal (orange), slight (green), or moderate (blue) disturbance.

Site	2005	2009	2011	2012	2013	2014	2015
WC0050	30	---	---	---	---	---	---
WC0150	---	38	44	34	36	34	28
WC0300	26	---	---	---	---	---	---
WC0600	24	32	38	32	32	28	30
WC0650	---	---	---	---	34	---	---
WC0875	---	---	40	30	26	30	30
WC0900	36	34	34	32	38	32	32
WC0950*	---	---	38	34	30	24	34
WC1025 (WC1100)**	---	---	---	---	---	24	32
WC1075 (WC1150) **	---	---	---	---	---	30	34
WC1800	32	---	---	---	---	---	---
WC1825	---	36	34	34	32	32	20
WC1850	---	34	22	36	26	28	32
WC1900	40	34	28	34	36	34	32
WC1950	---	34	34	36	36	36	34
WC2325	28	---	---	---	---	---	
WC2425	28	34	26	42	40	38	24
WC2600	30	38	28	46	32	36	30
WC2650	---	---	32	---	---	---	---
WC2700	---	---	36	---	---	---	---
WC3025	38	38	36	---	---	---	---

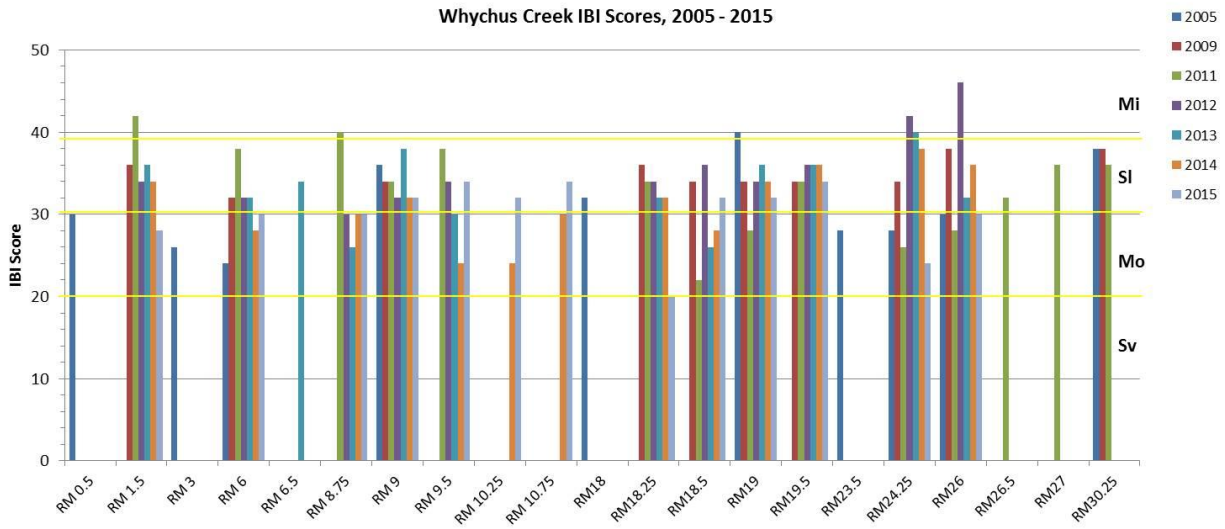
*in 2012, sampling was done at RM 9.25 instead of RM 9.5
** Site names changed for these locations based on improved GPS mapping

Individual site scores have changed across time, but no sustained trend towards increased IBI scores has been seen for individual sites (Figure 2A). When considered across different stream reaches, IBI scores have been relatively stable across time among the downstream and midstream reach sites, with greater annual differences occurring at upstream sites (Figure 2B). Among downstream sites, mean IBI scores increased from 2005 to 2011, with the mean score in 2011 significantly greater than in 2005, and the impairment level also improved from moderate to slight. The mean IBI score among downstream reaches in every subsequent year has been significantly lower than in 2011, but mean scores within the 2012-2015 span have not differed significantly for the sites, and in all years except 2014, the mean score indicated only slight impairment among downstream sampling reaches. Similarly, although there has been some annual change in mean IBI score in the midstream reach, these differences have never been significant, and in three of the seven sampling years, the mean score IBI score has reflected slightly impaired conditions among midstream sites.

Among the upstream sampling reaches, annual differences in mean IBI scores have been greater. From 2005 to 2012, mean scores increased, and site conditions showed a significant improvement from slight to minimal impairment. Mean IBI scores were lower in both 2013 and 2014; though the difference was not significant compared to the mean score in 2012, but site conditions still registered as slightly impaired. In 2015, the mean IBI score among upstream reaches was significantly lower than in 2012 and 2009, which was reflected in the corresponding moderately impaired condition category. However, two

factors may be influencing this lower mean score in 2015: 1. very low precipitation and an unusually warm winter and hot summer may have influenced flow and temperature conditions in the stream, affecting the macroinvertebrate community; and 2. one of the two upstream sampling reaches, WC2600, is the site of the Whychus Floodplain Project, and recently experienced extreme perturbations such as earth moving, flow interruption, and disturbance of the sediment regime.

a. Individual site scores across time. Mi = minimal impairment, Sl = slight, Mo = moderate, Sv = severe



b. Mean IBI scores across time among different stream reaches. Letter pairs indicate significant difference between means ($p<0.05$). Downstream = RM 0.5-11.5; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

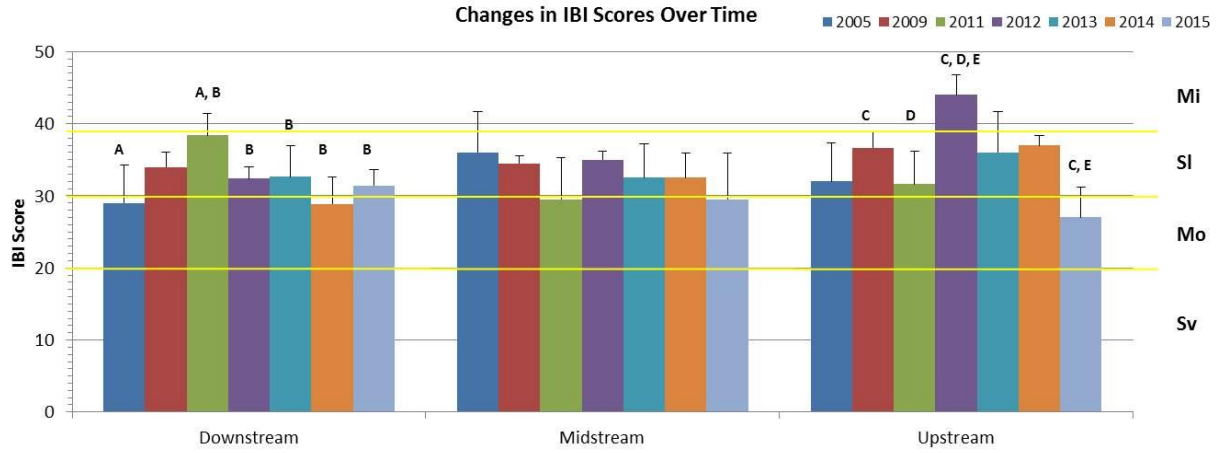


Figure 2. OR DEQ general IBI scores for Whychus Creek sampling sites.

Grande Ronde IBI

Macroinvertebrate community data were also analyzed using a more regional IBI developed by OR DEQ for northeastern Oregon (Grande Ronde IBI; Shannon Hubler, pers. comm.). The metrics in the Grande Ronde (GR) and general IBI are the same, except that the general IBI scores the percent dominance of the single most abundant taxon while the GR IBI scores dominance of the three most abundant taxa. Raw score ranges and corresponding scaled scores differ between the two IBIs, and the GR IBI scores reflect only three levels of biological condition (minimal impairment, >25; moderate impairment, 15-25;

and severe impairment, <15), in contrast to the four categories in the general IBI (minimal, slight, moderate, and severe impairment; see Table 2).

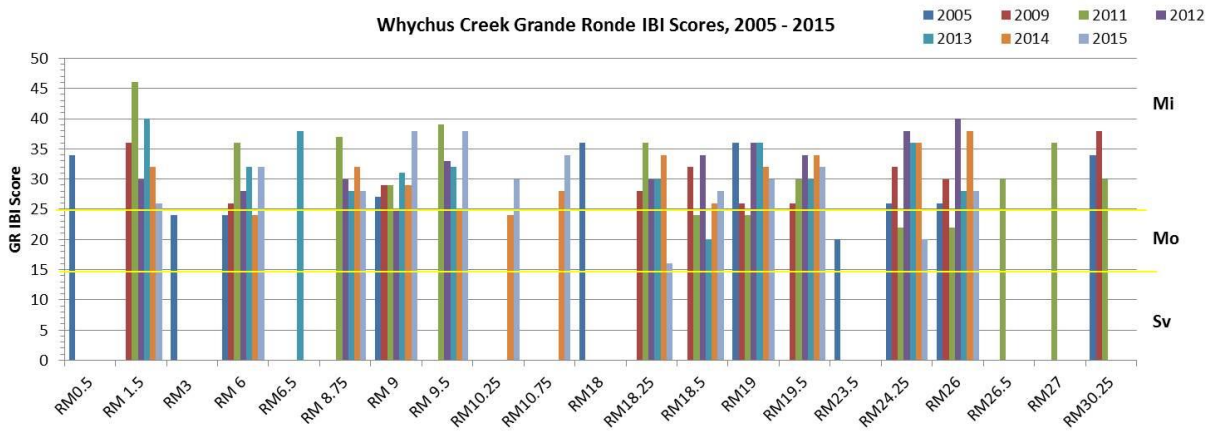
Most sites (70-100%) in all years scored as minimally impaired (Table 5; Figure 3A); the remainder scored as moderately impaired, and no site had a summed score low enough to indicate severe impairment. Annual changes in mean IBI score among the reaches varied when using the GR IBI (Figure 3B), but differences were generally not significant from year to year, and usually did not change the biological condition category. Although the GR IBI score tended to indicate better biological conditions, the scores for the general and GR IBI have a strong positive correlation ($r^2 = 0.6523$)

Table 5. Grande Ronde IBI scores for all sites sampled between 2005 and 2014. Colors indicate minimal (orange) or moderate (green) disturbance; no site scored as severely disturbed in any year.

Site	2005	2009	2011	2012	2013	2014	2015
WC0050	34	---	---	---	---	---	
WC0150	---	36	46	30	40	32	26
WC0300	24	---	---	---	---	---	---
WC0600	24	26	36	28	32	24	32
WC0650	---	---	---	---	38	---	---
WC0875	---	---	37	30	28	32	28
WC0900	27	29	29	25	31	29	38
WC0950*	---	---	39	33	32	25	38
WC1025 (WC1100)	---	---	---	---	---	24	30
WC1075 (WC1150)	---	---	---	---	---	28	34
WC1800	36	---	---	---	---	---	---
WC1825	---	28	36	30	30	34	16
WC1850	---	32	24	34	20	26	28
WC1900	36	26	24	36	36	32	30
WC1950	---	26	30	34	30	34	32
WC2325	20	---	---	---	---	---	---
WC2425	26	32	22	38	36	36	20
WC2600	26	30	22	40	28	38	28
WC2650	---	---	30	---	---	---	---
WC2700	---	---	36	---	---	---	---
WC3025	34	38	30	---	---	---	---

*in 2012, sampling was done at RM 9.25 instead of RM 9.5

a. Individual GR IBI site scores across time



b. Mean GR IBI scores among different stream reaches. Letter pairs indicate significant difference between means ($p<0.05$). Downstream = RM 0.5-11.5; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

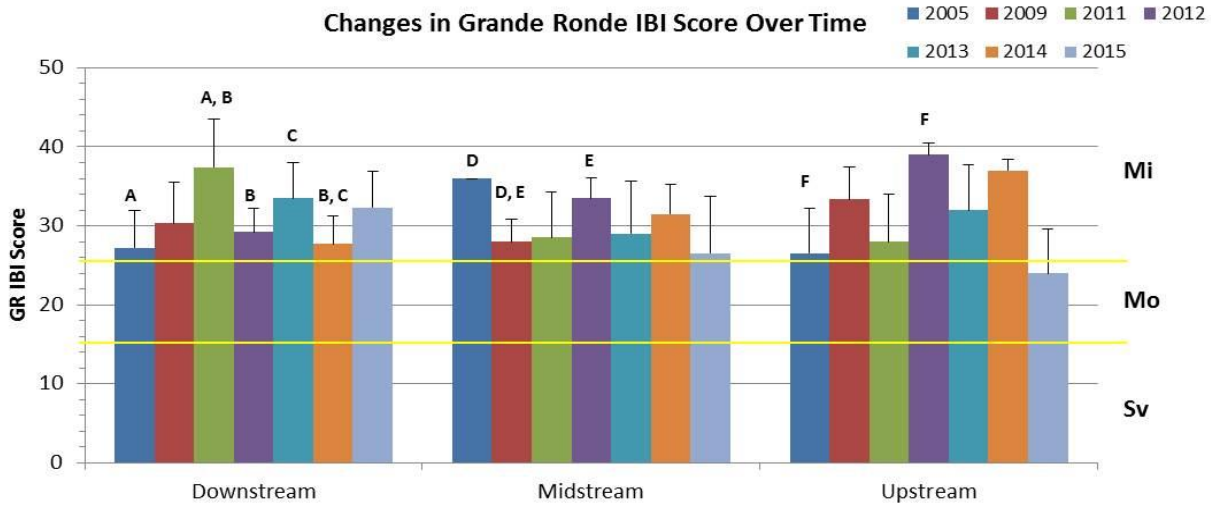


Figure 3. Grande Ronde IBI scores for Whychus Creek sampling sites.

Predator

Site O/E

PREDATOR scores were higher in 2015 compared to recent years, with 23% of the sites scoring as good, 54% as fair, and 23% as poor. This is a contrast to 2014 PREDATOR scores, where the same sites all had O/E scores reflecting poor conditions. Biological conditions reflected by PREDATOR scores have in general been worse than those reflected by IBI scores, although that may be influenced in part by the fact that PREDATOR comprises only 3 impaired condition categories apart from enriched (good, fair, poor), while the general IBI has four categories that encompass this same condition span (minimal, slight, moderate, and severe impairment). PREDATOR scores at individual sites vary annually (Table 6), but the differences generally result in little change in biological condition category, though some sites have fluctuated annually between ratings of poor and fair (Figure 4A). When considered at the reach level, mean O/E scores among downstream sites increased from 2005 to 2009, though not significantly, and remained at about the same level of fair biological condition until 2014, when the mean O/E score

was significantly lower. However, in 2015, the mean O/E score among the downstream reaches increased significantly compared to the previous year, and once again reflected fair conditions.

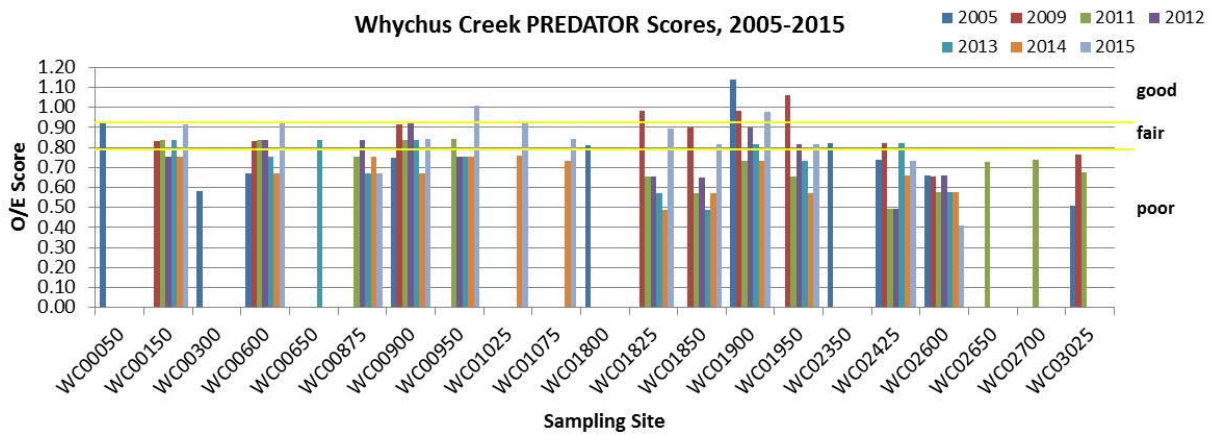
The greatest variation has been among midstream sites, with significant between-year differences in mean scores that have corresponded to good, fair, and poor conditions during this study. The highest (good) midstream reach mean O/E scores were seen in 2005; scores in subsequent years were significantly lower and reflected apparently poor biological conditions. However, the mean O/E score in 2015 was significantly greater than the means in 2011-2013, and reflected an improvement to fair biological conditions. The macroinvertebrate community may thus have recovered from perturbations caused by earlier restoration actions in this portion of the creek, and/or have improved as a result of management actions. Mean O/E scores for the upstream reach sites have consistently reflected poor biological conditions, however, and though they too vary annually there is no clear pattern and differences between years are not significant.

Table 6.
PREDATOR O/E scores. Colors indicate biological conditions that are good (orange), fair (green), or poor (blue).

Site	2005	2009	2011	2012	2013	2014	2015
WC0050	0.92	---	---	---	---	---	---
WC0150	---	0.83	0.84	0.75	0.84	0.75	0.92
WC0300	0.58	---	---	---	---	---	---
WC0600	0.67	0.83	0.84	0.84	0.75	0.67	0.92
WC0650	---	---	---	---	0.84	---	---
WC0875	---	---	0.75	0.84	0.67	0.75	0.67
WC0900	0.75	0.92	0.84	0.92	0.84	0.67	0.84
WC0950*	---	---	0.84	0.76	0.75	0.76	1.01
WC1025 (WC1100)	---	---	---	---	---	0.76	0.93
WC1075 (WC1150)	---	---	---	---	---	0.73	0.84
WC1800	0.81	---	---	---	---	---	---
WC1825	---	0.98	0.65	0.65	0.57	0.49	0.90
WC1850	---	0.90	0.57	0.65	0.49	0.57	0.81
WC1900	1.14	0.98	0.73	0.90	0.82	0.73	0.98
WC1950	---	1.06	0.65	0.82	0.73	0.57	0.81
WC2325	0.82	---	---	---	---	---	---
WC2425	0.74	0.82	0.49	0.49	0.82	0.66	0.74
WC2600	0.66	0.66	0.58	0.66	0.58	0.58	0.41
WC2650	---	---	0.73	---	---	---	---
WC2700	---	---	0.74	---	---	---	---
WC3025	0.51	0.76	0.68	---	---	---	---

*in 2012, sampling was done at RM 9.25 instead of RM 9.5

a. Individual site O/E scores across time



b. Mean PREDATOR O/E scores across time among stream reaches. Letter pairs indicate significant difference between means ($p<0.05$). Downstream = RM 0.5-11.5; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

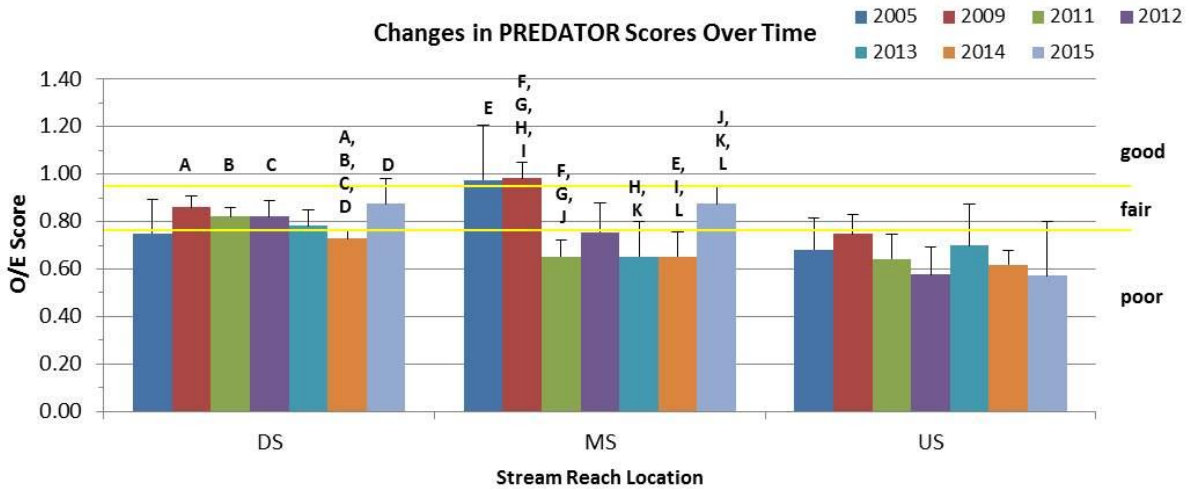


Figure 4.
PREDATOR O/E Scores for Whychus Creek sampling sites

Increaser and Decreaser Taxa

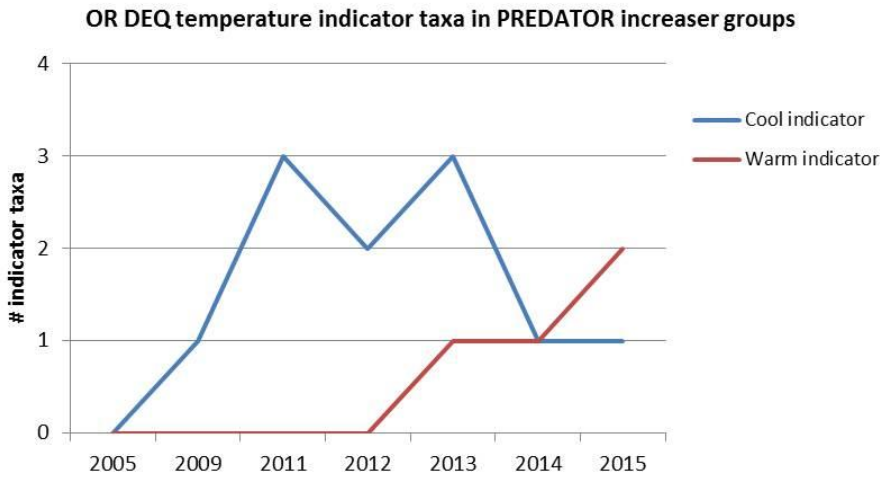
The PREDATOR model output includes a sensitivity index for each taxon, calculated as the # sites at which a taxon was observed/#sites at which it was expected (S_o/S_e), such that a single sensitivity index score is generated for each taxon across the entire sampling set. Taxa with S_o/S_e >1 are considered to be "increasers" (observed more frequently than expected); taxa with S_o/S_e <1 are "decreasers" (observed less frequently than expected). Increaser and decreaser taxa for each sampling site in Whychus Creek were analyzed across all sampling years. To avoid a bias for rare taxa, increasers were identified using $S_o/S_e \geq 1.3$, while taxa with S_o/S_e <0.8 were selected as decreasers.

Increaser and decreasers included several taxa identified by OR DEQ as indicators for temperature and/or sediment conditions (Appendix B), and changes in the numbers of indicator taxa suggest a response to altered stream conditions (Figures 5 and 6). In 2005, there were no cool temperature

indicators among the increaser group, but the number of cool indicator taxa rose among increasers through 2013 (Figure 5A). Conversely, among the decreaser taxa, no warm indicators were seen In 2005 and 2009, after which the number of warm indicators rose and remained high among the decreaser groups (Figure 5B). In both 2014 and 2015, the number of cool indicator taxa also rose among the decreaser groups, which may suggest a new temperature stressor.

A pattern of increased numbers of low sediment indicator taxa and decreased numbers of high sediment indicators is apparent in the increaser groups, while the opposite occurred among the decreasers, with increasing numbers of high sediment indicators and low numbers of low sediment indicators in all years except 2014 (Figure 6).

a. Number of cool and warm indicator taxa in increaser group



b. Number of cool and warm indicator taxa in decreaser group

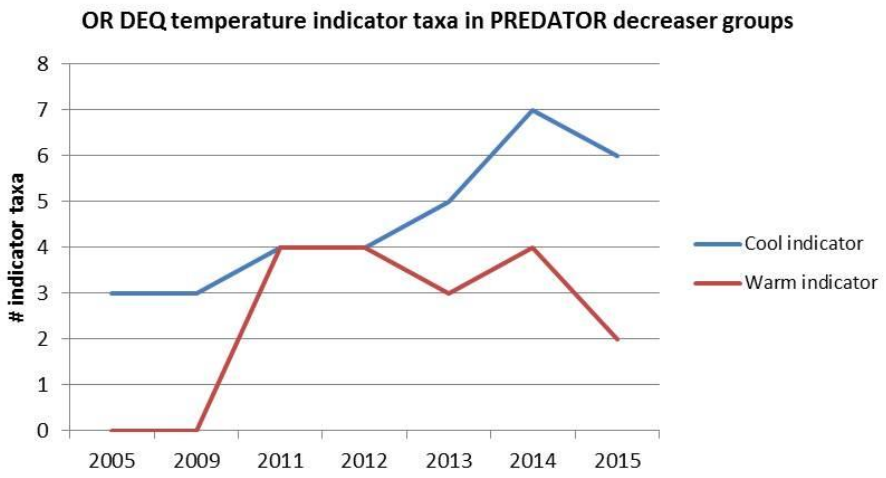
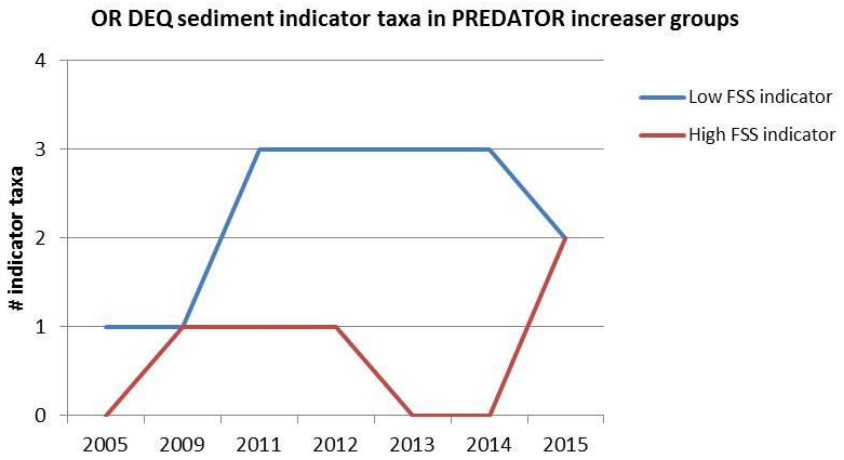


Figure 5. Indicator taxa for temperature among PREDATOR increaser and decreaser taxa groups.

a. Indicator taxa for sediment among PREDATOR increaser groups.



b. Indicator taxa for sediment among PREDATOR decreaser groups.

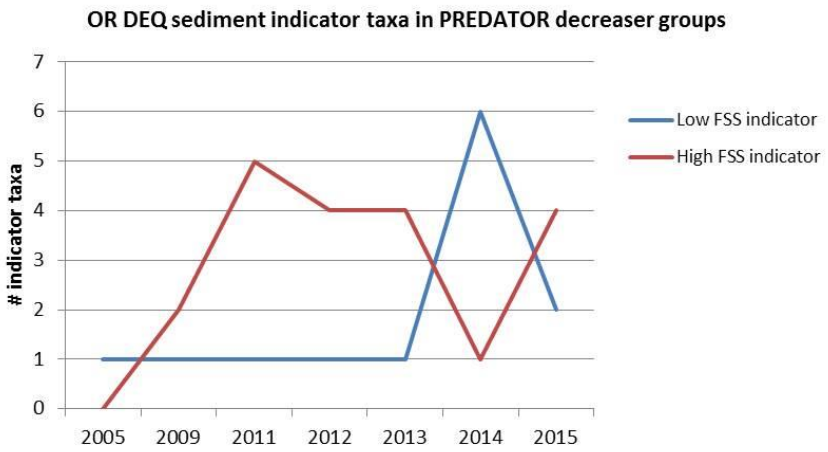
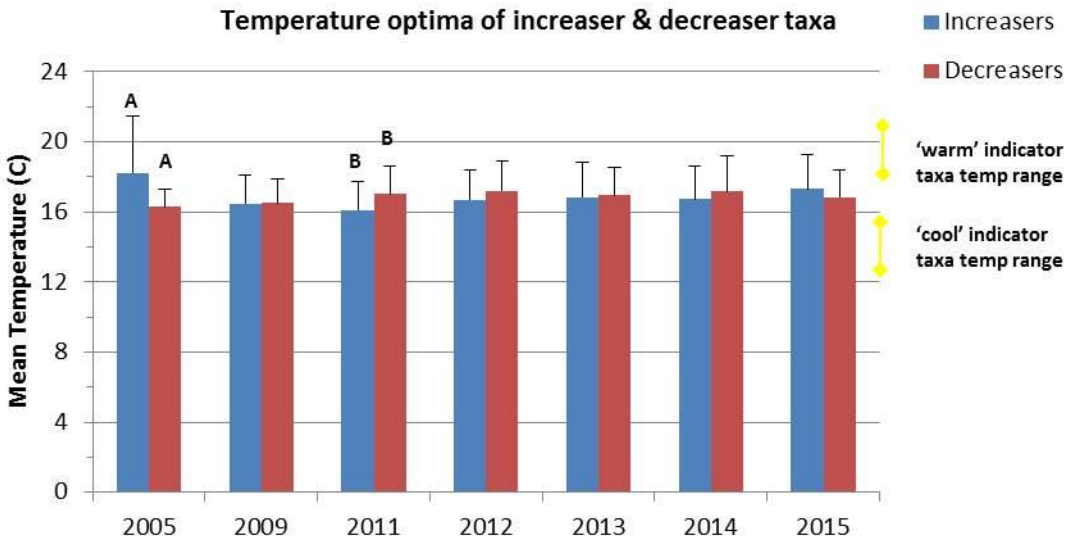


Figure 6. Indicator taxa for fine suspended sediment (FSS) among PREDATOR increaser and decreaser taxa groups.

Increaser communities tend to have lower mean temperature and sediment optima compared to decreaser communities, although differences are not significant in every year (Figure 7). The greatest differences are seen in sediment optima; the mean %FSS optima of the increaser community are lower than the decreaser community in every sampling year (Figure 7B), and this difference was significant in 2012 and 2014 ($p<0.05$) and near significant in 2011 ($p=0.06$). Temperature optima in increaser and decreaser communities have differed less; mean optima are similar within years, and no significant differences have been seen since 2011 (Figure 7A). However, the consistent pattern of higher mean sediment optima among the decreaser groups in each year suggests that stream conditions are facilitating colonization and survival of taxa with greater sensitivity to sediment levels.

a. Mean temperature optima of increaser and decreaser taxa. Letter pairs indicate significant differences (p<0.05). Temperature ranges of OR DEQ indicator taxa shown for reference.



b. Mean %FSS optima of increaser and decreaser taxa. Letter pairs indicate a significant difference (p<0.05). %FSS ranges of OR DEQ indicator taxa shown for reference.

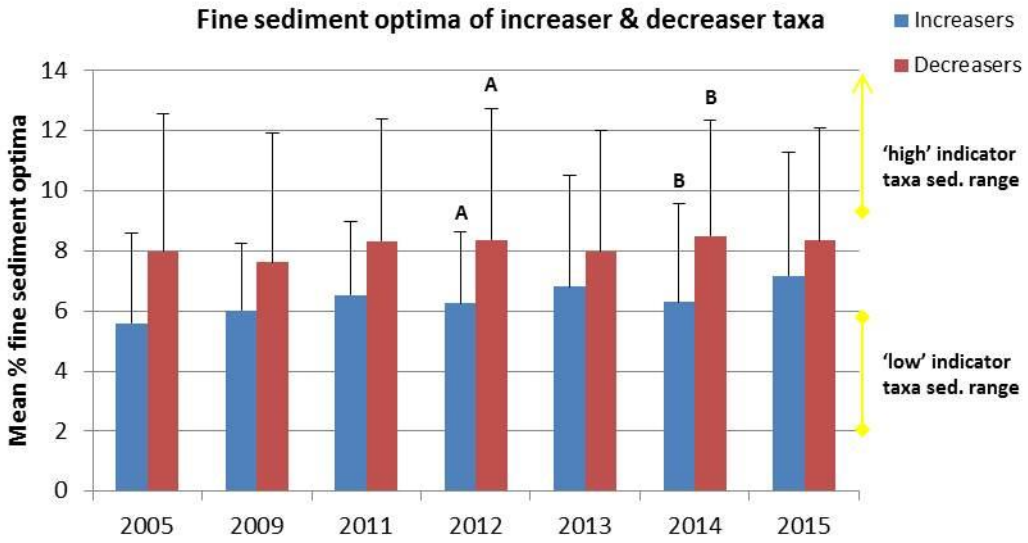


Figure 7. Mean temperature and fine sediment optima among increaser and decreaser taxa

The composition of increaser and decreaser groups varies annually, but some taxa are present consistently. *Acentrella* (small minnow mayfly), *Brachycentrus (*flatheaded mayfly), Capniidae (small winter stonefly family), and *Rhithrogena* (humpless casemaker caddisfly) are among the increaser taxa each year; all are members of the EPT (Ephemeroptera, Plecoptera, Trichoptera) group considered most sensitive to degraded stream conditions. *Rhithrogena*, Capniidae, and *Brachycentrus* have a low tolerance for organic pollution, and *Rhithrogena* is also an OR DEQ low temperature indicator taxon. Taxa that occur consistently as decreasers are more mixed and include sensitive and more tolerant taxa such as *Drunella doddsi* (a sensitive spiny crawler mayfly with low tolerance for organic pollution; DEQ cool temperature and low sediment indicator), *Heterlimnius* (riffle beetle), *Ironodes* (flatheaded mayfly; DEQ cool temperature indicator), *Lepidostoma* (casemaker caddisfly with low tolerance for organic

pollution), *Malenka* (small brown stonefly with low tolerance to organic pollution), *Maruina* (moth fly), *Micrasema* (of humpless casemaker caddisfly with low tolerance for organic pollution; DEQ cool temperature indicator) and *Neophylax* (stonecase caddisfly).

CLUSTER analysis of a Bray-Curtis similarity matrix of presence/absence abundance of decreaser taxa (Figure 8) groups the 2005 and 2009 communities separately; in later years, decreaser community composition was most similar among the 2011-2013 communities and the 2014 and 2015 communities, suggesting a continuous shift in stream conditions. CLUSTER analysis of increaser taxa also separates the communities in 2005 and 2009 from all later years.

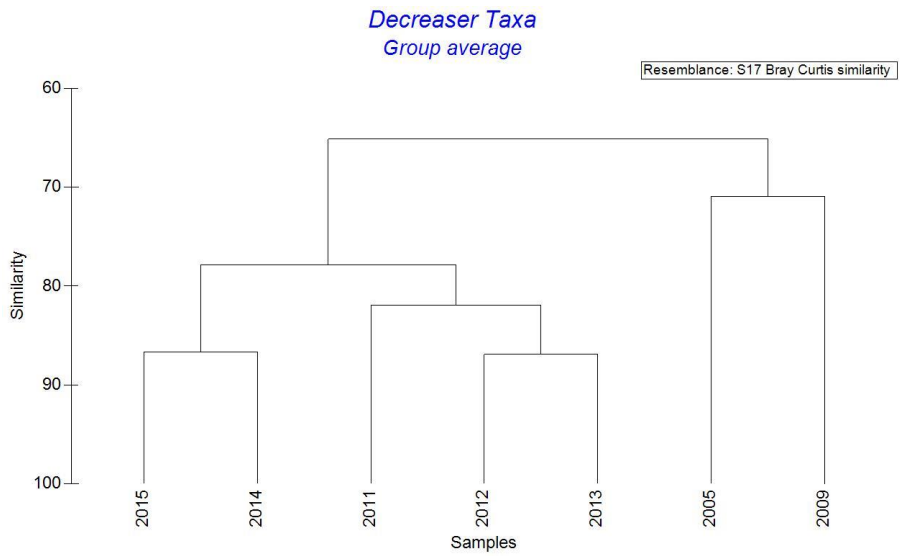


Figure 8. Similarity among PREDATOR decreaser communities

Missing and Replacement Taxa

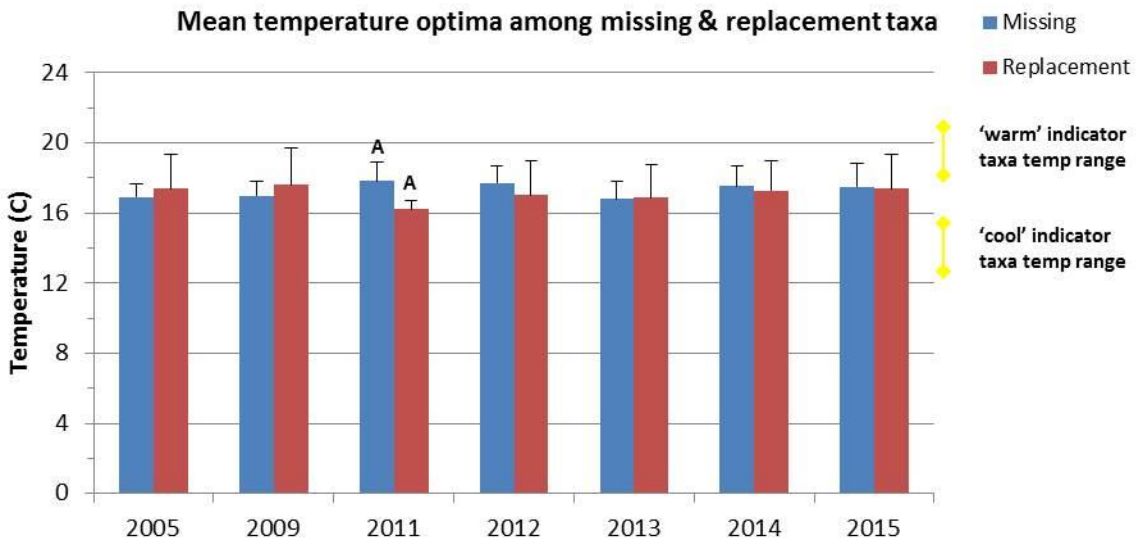
The PREDATOR model further identifies taxa expected at a site but not collected (missing taxa), and taxa that were not expected at a site but were collected in samples (replacement taxa), based on comparison to appropriate reference sites. Missing and replacement taxa are indicated for each individual sampling site. Taxa missing from a majority of expected sites (≥7) in every year included *Calineuria* (a moderately sensitive perlid stonefly), *Malenka* (small brown stonefly with low tolerance to organic pollution), Pisidiidae (a family of sediment- and pollution-tolerant fingernail clams; DEQ high sediment indicator), and Tanypodinae (a common tolerant non-biting midge group; DEQ high sediment indicator). *Malenka* and Pisidiidae were also among the decreaser taxa in each year. Taxa identified as replacements at ≥7 sites in each sampling year included *Acentrella* (a small minnow mayfly), *Atherix* (a common, tolerant watersnipe fly), *Narpus* (a common, moderately tolerant riffle beetle), *Rhithrogena* (a flatheaded mayfly with low tolerance for organic pollution; DEQ low sediment indicator), and *Serratella* (a common spiny crawler mayfly with low tolerance to organic pollution). *Rhithrogena* and *Acentrella* were also among taxa identified as increasers in each year. No new missing taxa were identified in the 2015 samples, but three replacement taxa in the 2015 samples had not been seen as replacements in any previous year: *Diphetor hageni* (small minnow mayfly; OR DEQ warm temperature indicator), *Hydroptila* (a tolerant microcaddisfly; OR DEQ high sediment indicator taxon)and Dasyheleinae (a biting midge subfamily).

CLUSTER analysis of a Bray-Curtis similarity matrix of presence/absence abundances of missing taxa showed two main groups, with the 2005 and 2009 communities most closely related to each other (91%

similarity) and then to the 2013 missing community (71% similar). The remaining years clustered into a second group, with greater similarity between the 2011 and 2012 missing community (94% similarity) and 2014 and 2015 community (89% similarity). However, the two main groups had an overall similarity of 69%, showing a high degree of commonality among missing taxa throughout all sampling years. Less overall similarity was seen among replacement communities in earlier sampling years, with the replacement communities from 2005-2011 having only 53-66% similarity. The replacement communities in recent years were more closely related, with 85% similarity between the 2014 and 2015 replacement communities and between the 2012 and 2013 replacement communities.

A comparison of mean temperature and sediment optima among missing and replacement communities reveals a pattern similar to that seen for decreasers and increasers (Figure 9). In 2005 and 2009, mean temperature optima were higher among replacement taxa, but in 2011, the mean temperature optima of the replacement community was significantly lower than the missing community, and in subsequent years the temperature optima of the two groups are similar. A stronger trend occurs for %FSS optima, which are significantly lower in replacement versus missing communities in every year except 2009 (and the difference in 2009 is close to significant, at p=0.056).

a. Mean temperature optima of missing and replacement taxa. Letter pairs indicate a significant difference (p<0.05). Temperature ranges spanned by ORDEQ indicator taxa shown for reference.



b. Mean %FSS optima of missing and replacement taxa. Letter pairs indicate a significant difference (p<0.05). The OR DEQ indicator taxa range is shown for reference.

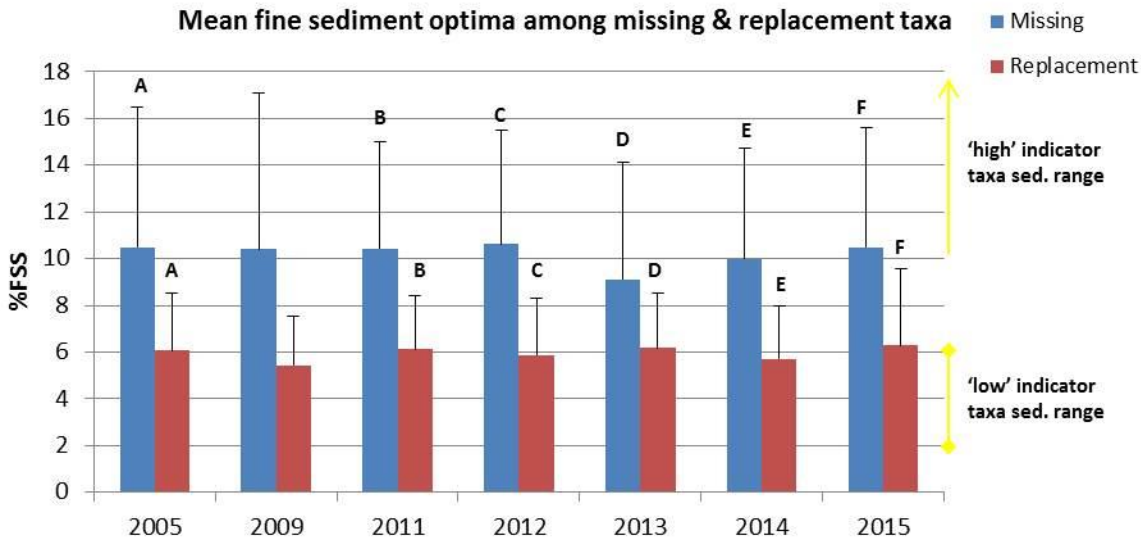


Figure 9. Differences in temperature and fine sediment tolerances among missing and replacement taxa

Taxonomic and Ecological Trait Analysis

Community analysis

Community composition

Sample organismal abundance has been high enough that the target subsampling count of 500 was achieved for most samples in most years; samples with lower abundances have generally come from more upstream sites, which is not unusual for small cold headwater streams (Crunkilton & Duchrow 1991, Lillie *et al*. 2003). From 2005-2013, the 500 organism target was not attained for only one to three samples from within a given year. In 2015, however, abundances seemed lower overall, as not only was the 500-organism subsampling target not met for four of the 13 composite site samples, but these samples had unusually low numbers of invertebrates, ranging from 67 to 264. The reason for this is not known, and differences in sampling technique between volunteers cannot be ruled out, but these lower abundances are likely influenced by the abnormally hot dry weather in 2015 and the occurrence of significant recent restoration-related disturbances at a flood plain project site (RM26; composite sample contained 67 organisms).

Seventy-two unique taxa were collected among all sites in 2015, which is in the same range as other years (76-83 taxa per year in 2005–2014) but is the lowest number of taxa collected in a single sampling year. Despite lower overall abundances and diversity, three new taxa were added to the list in 2015, bringing the number of unique taxa collected among all sampling sites since 2005 to 141 (see Appendix C for taxa list). New taxa not seen in previous years were *Labiobaetis*, *Acentrella insignificans*, and *Attenella delantala*. *Labiobaetis* is a small minnow mayfly genus that is widely distributed across a variety of lotic habitats; it tends to occur in low abundance when present (only one individual was found at a single site in 2015) and is thus not unusual as an occasional record in longer-term data sets (Mike Cole, pers. comm.). *Acentrella insignificans* is also in the small minnow mayfly family; one to two individuals of this species were present at three sampling sites in the lower reaches in 2015, and additional species in this genus were present in samples in other years. A single individual of *Attenella*

delantala, a spiny crawler mayfly, was found at one sampling site. Both *Labiobaetis* and *Acentrella insignificans* have low %FSS optima (3% and 4%, respectively), in keeping with the trend for low sediment optima seen among increaser and replacement taxa.

The rate of addition of new taxa to the species list has slowed dramatically. In 2009, 35 new taxa were found among all sites, 69% of which were in the sensitive EPT group. This dramatic increase in 2009 occurred in part because four years had passed since the first sampling was done and restoration activities that increased stream flows had been implemented in the interim, and because several groups that had been left at genus in 2005 were identified to species in 2009 (and later years). Fewer new taxa were seen in every subsequent year, but except for 2014, the majority of new species are EPT, with 10 new taxa in 2011 (80% EPT), three in 2012 (100% EPT), seven in 2013 (71% EPT), five in 2014 (20% EPT), and three in 2015 (100% EPT). This suggests a larger initial shift in the macroinvertebrate community in response to restoration activities that favored colonization by sensitive taxa, with increased stabilization in recent year.

Agreement between replicate samples

In each year, duplicate samples are taken at two sites by Xerces staff as a quality control measure to assess operator differences between newly-trained volunteers and experienced stream samplers. A CLUSTER analysis showed that in 2015, as in almost every previous year, each duplicate sample community was most similar in composition to the sample taken by volunteers at the same site. This provides assurance that volunteers correctly implement sampling protocols, and that differences seen between sites and years are due more to changes in the macroinvertebrate community than differences among practitioners.

Community similarity analysis

CLUSTER analysis and MDS ordination of the 2015 dataset revealed greatest macroinvertebrate community similarity among samples taken in the same reaches (Figure 10), with downstream (46% similar), mid-reach (49% similar), and upstream (55% similar) sites forming three separate groups, as expected based on the River Continuum Concept (Vannote et al., 1980). Similar relationships are evident in all years among sites in the same sampling reaches, with average community similarities of 66% among all downstream and mid-reach sampling sites, and 58% among all upstream sampling sites. The greatest between-year differences continue to be seen for the 2005 sampling community compared to all other years (Figure 11).

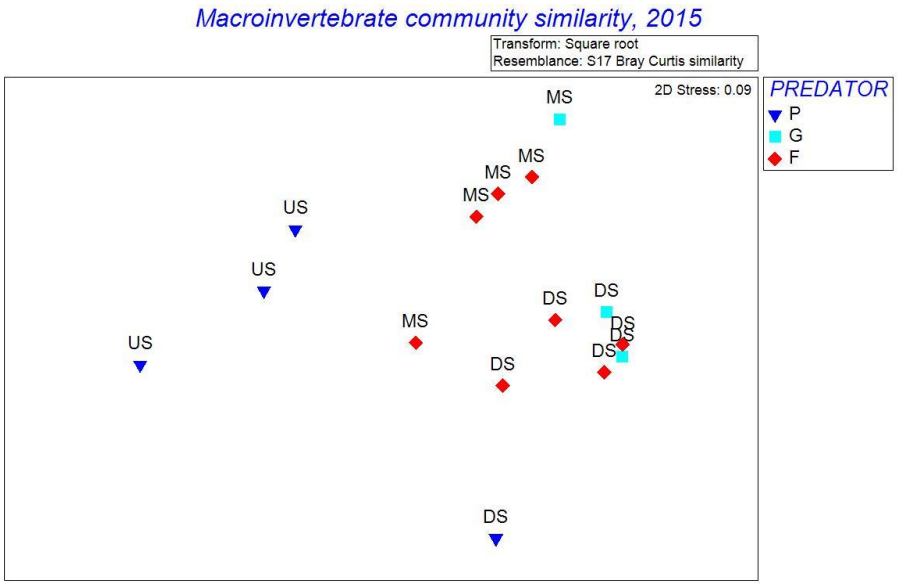


Figure 10. MDS ordination of macroinvertebrate communities in Whychus Creek, 2015. Downstream (DS) reach = RM 0.5 – 11.5; mid-stream (MS) = RM 18.0 – 19.5; upstream (US) reach = RM 23.5 – 30.25.Site PREDATOR score (Poor, Fair, Good) is also indicated.

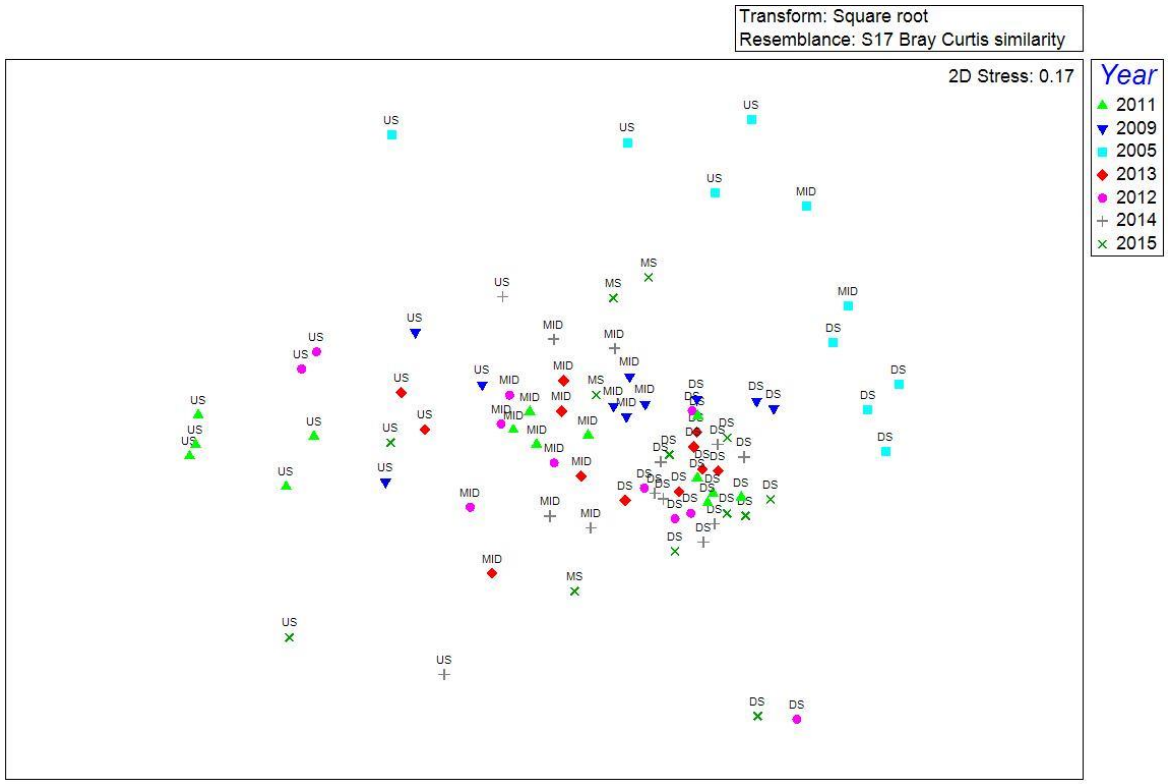


Figure 11. MDS ordination of macroinvertebrate communities in Whychus Creek, 2005-2015. Downstream (DS) reach = RM 0.5 – 11.5; mid-stream (MS) = RM 18.0 – 19.5; upstream (US) reach = RM 23.5 – 30.25. Site PREDATOR score (Poor, Fair, Good) is also indicated.

Downstream sites

MDS ordination of downstream sampling sites showed a distinct separation between the 2005 community compared to later years (Figure 12). The lowest average between-year dissimilarity (35%) was seen in 2013 and 2014, and the communities in 2005 and 2012 had the greatest average dissimilarity (55%). The macroinvertebrate community in 2009 was the most similar to the 2005 community (48% dissimilarity), while those in all subsequent years were 50-55% dissimilar to the 2005 community. The downstream macroinvertebrate community in 2015 was 52.4% dissimilar to the 2005 community, and the taxa that contributed most to the differences were *Baetis* (common widespread genus of small minnow mayfly; includes species that are OR DEQ cool temperature and low sediment indicator taxa; more abundant in 2015), *Brachycentrus* (sensitive humpless case-maker caddisfly; more abundant in 2015), and *Zaitzevia* (tolerant riffle beetle; OR DEQ warm temperature and high sediment indicator; more abundant in 2005). The downstream communities in 2014 and 2015 were only 43% dissimilar, due primarily to differing abundances of *Simulium* and *Baetis* (both more abundant in 2014), as well as *Brachycentrus* (more abundant in 2015).

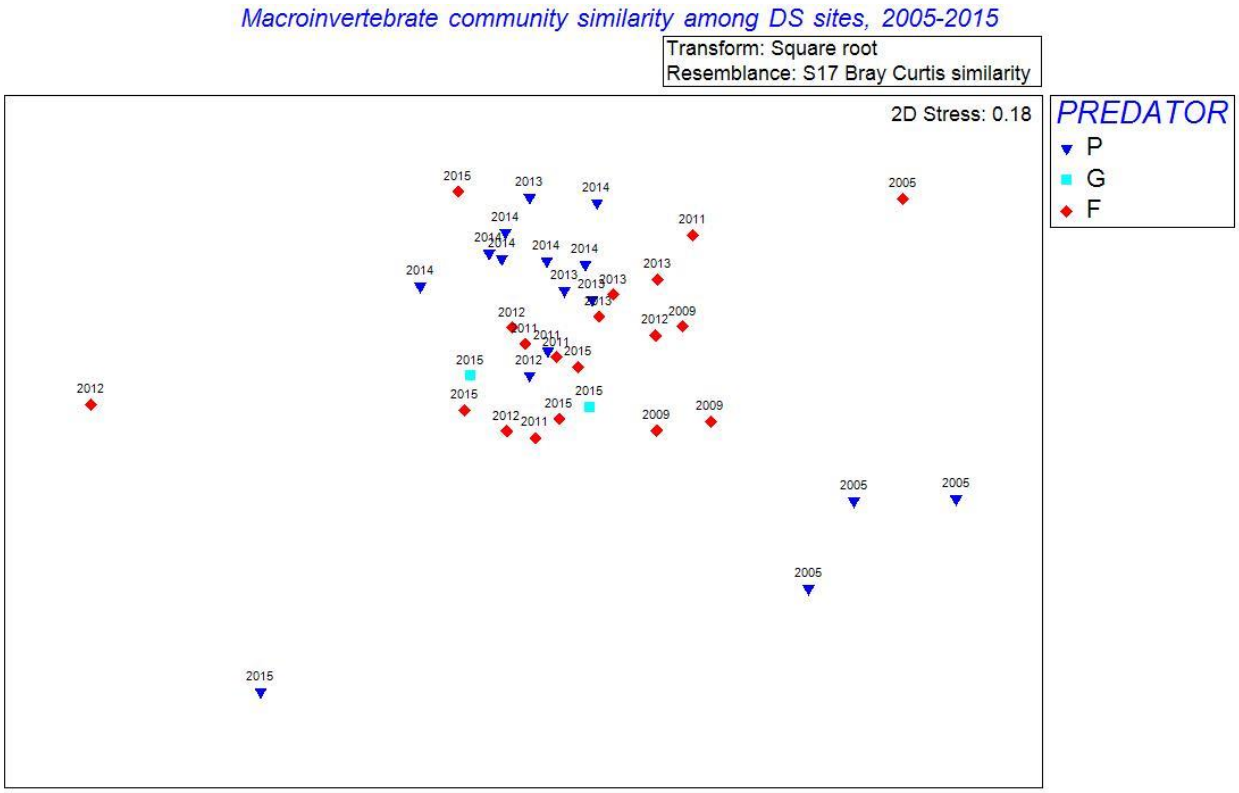


Figure 12. MDS ordination of macroinvertebrate communities among downstream sampling sites (RM 0.5 – 11.5) in Whychus Creek, 2005-2015. Site PREDATOR score (Poor, Fair, Good) is also indicated.

Mid-reach sites

MDS ordination of mid-reach sampling sites also separates the 2005 community from later years (Figure 13). The lowest between-year community dissimilarity was seen for 2011 and 2012 (37% average dissimilarity), and the greatest average dissimilarities were seen between the community in 2005 versus each year in the 2011-2014 span (62-70% average dissimilarity).

Taxa that contributed the most to differences with the 2005 community were similar in different years, and included Baetis (common genus of small minnow mayfly; lower abundance in 2005), Zapada (relatively sensitive genus of small brown stonefly; greater abundance in 2005); Simulium (moderately tolerant cosmopolitan genus of black fly; greater abundance in 2005); and Chironominae (diverse subfamily of non-biting midge with many tolerant genera; greater abundance in 2005). Taxa that contributed most to differences between the 2014 and 2015 mid-reach communities, which were 48% dissimilar, also included Baetis (more abundant in 2014) as well as Chironominae and Oligochaeta, tolerant groups that were more abundant in the 2015 community, which may indicate stressors related to abnormally low precipitation and high temperatures in that year.

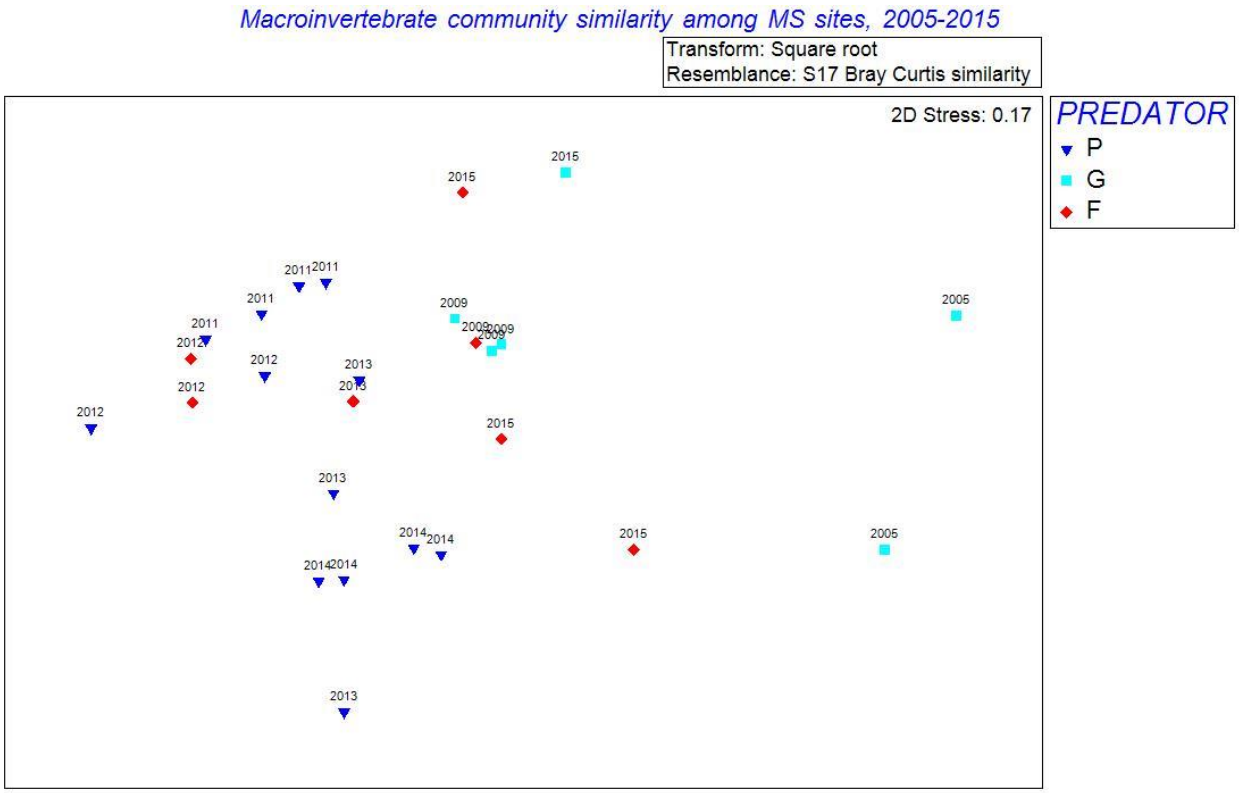


Figure 13. MDS ordination of macroinvertebrate assemblages at all mid-stream reach sampling sites (RM 18.25 – 19.5). Site PREDATOR score (Poor, Fair, Good) is also indicated.

Upstream sites

MDS ordination of upstream sampling sites separated the 2005 community from all later years (Figure 14). The lowest average between-year dissimilarity was in 2009 and 2013 (39%), and the greatest average dissimilarities were seen between the upstream community in 2005 and each year in the 2011-2015 span (64-75% average dissimilarity). Taxa that contributed most to differences with the 2005 community were similar between different years; those that were consistently more abundant in the 2005 community included the tolerant non-biting midge subfamilies Orthocladiinae and Chironominae, and taxa that were more abundant in subsequent years included more sensitive groups and/or taxa more associated with lotic waters, such as Rhithrogena (flatheaded mayfly genus with low tolerance for organic pollution; DEQ low sediment indicator), Simulium (moderately tolerant genus of black fly), Prosimulium (black fly genus; OR DEQ cool temperature indicator), Baetis (common small minnow mayfly), and Suwallia (sensitive genus of green stonefly).

The upstream macroinvertebrate communities present in 2014 and 2015 had an average dissimilarity of 62%, due primarily to differing abundances of Baetis and Orthocladiinae, both of which were more abundant in the 2014 samples, as well as Rhithrogena, which had greater abundances in 2015 samples.

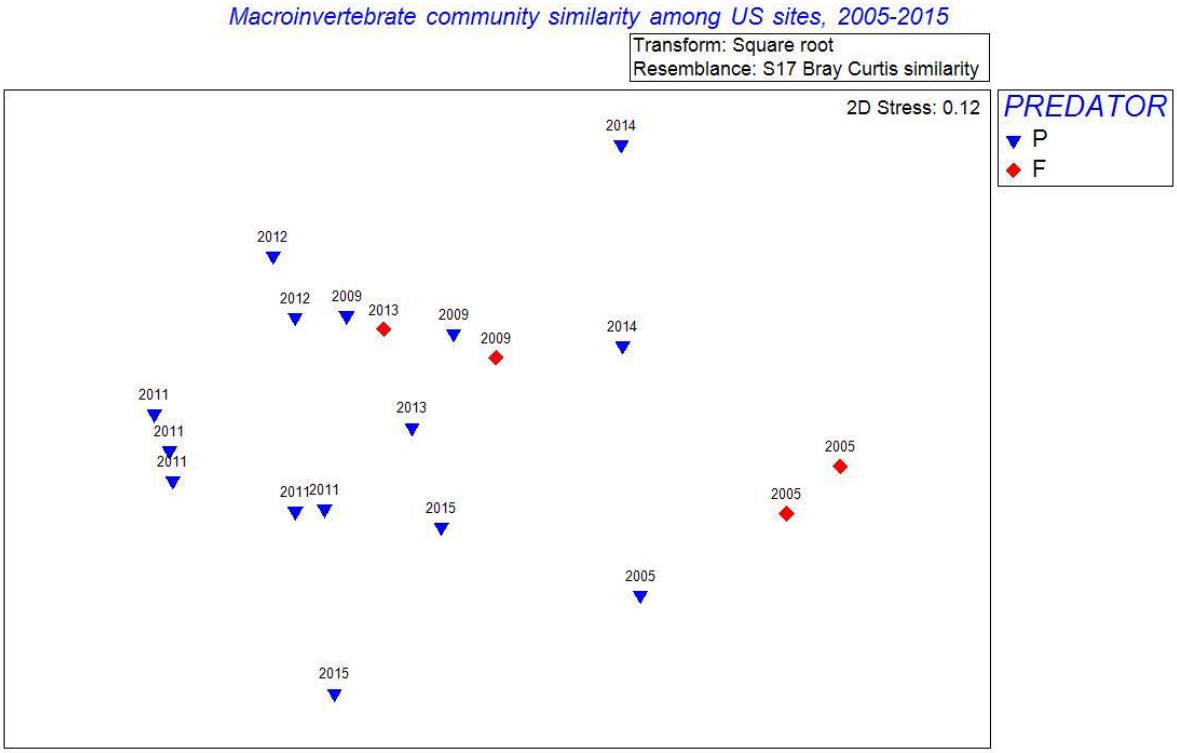


Figure 14. MDS ordination of macroinvertebrate assemblages at all upstream sampling sites (RM 23.5 – 30.25). Site PREDATOR score (Poor, Fair) is also indicated (no upstream site had an O/E score reflecting Good condition in any year).

Changes in individual IBI metrics

The OR DEQ IBI comprises different classes of metrics: richness (# taxa), tolerance (# or % tolerant or sediment-sensitive taxa), and composition (dominance of top taxon). Total IBI scores vary among sites and years, but since these scores are scaled from raw values, it can be informative to examine changes in individual metrics at the reach level.

Diversity measures

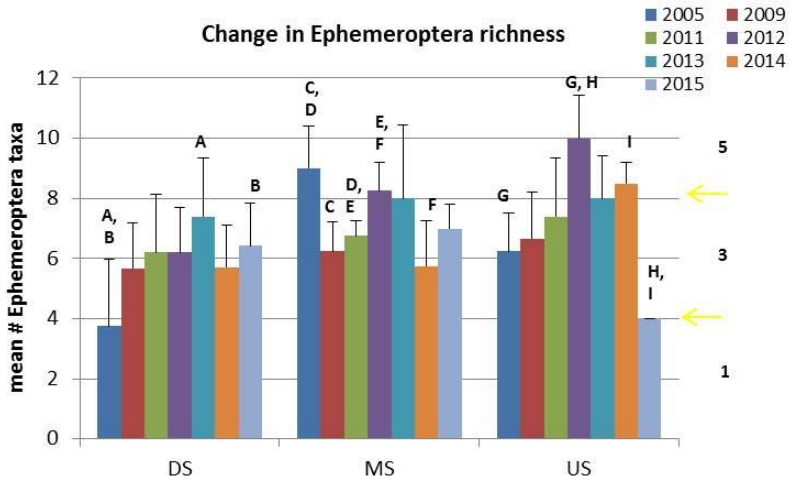
The first four metrics in the IBI are based on the rationale that a less disturbed, healthier stream system has more biodiversity with higher overall taxa richness (Norris & Georges, 1991; Barbour et al., 1996) and greater diversity of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera). The EPT are a focus of bioassessment metrics as they are widely considered to be the most sensitive to changes in flow, temperature, and sediment (although individual genera differ in their sensitivities), and they may be considered separately or as a group (EPT) in bioassessment.

Changes in taxa richness are not pronounced in most reaches (Figure 15). Mean richness oscillates between years, but differences are rarely significant. The greatest variation occurred among upstream reach sites, and diversity in 2015 was significantly lower than most previous years. Other years have seen significant increases and decreases in mean richness among upstream sites, and this decrease in 2015 likely reflects substantial disturbance both in-stream and in the riparian zone that occurred as part of the Whychus Floodplain project at RM26, as it was not apparent at downstream or mid-reach sites.

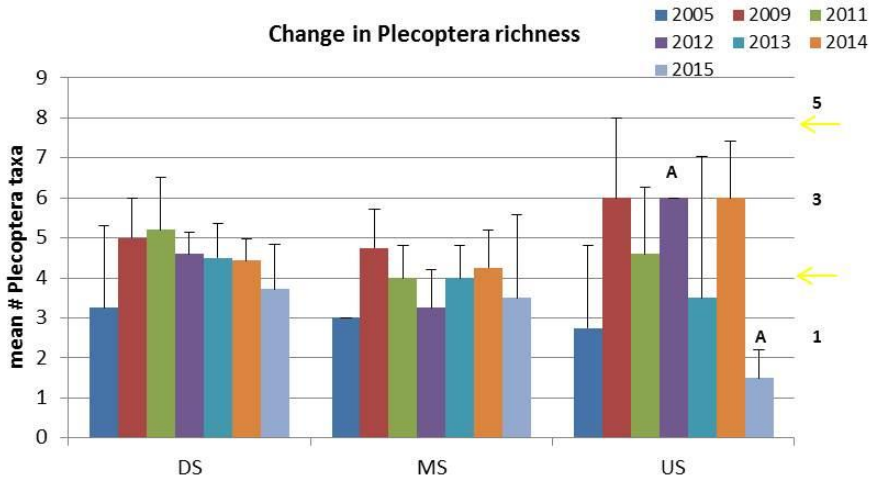
Figure 15. Changes in diversity metrics among Whychus Creek sampling sites. Letter pairs indicate significant difference between means (P<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25. Ranges for IBI scoring (5, 3, 1) are indicated.

EPT richness was examined for each order and for EPT as a group (Figures 16 and 17). An apparent trend toward increased mayfly diversity at upstream and downstream sites continued in 2015 for the downstream reach, where the mean number of Ephemeroptera taxa remained higher than in several previously years and was significantly greater than in 2005 (Figure 16A). However, the disturbance of upstream sampling sites in 2015 resulted in a dramatic decrease in mean number of Ephemeroptera taxa among these sites. Mayfly diversity has fluctuated most among mid-reach sites, and after a significant decrease in mayfly richness among these sites in 2014, the number of taxa increased again in 2015 to levels close to those seen in 2012 and 2013. Mean stonefly diversity has generally been greater among all reaches since 2005, although these differences have not been significant (Figure 16B). The greatest variation occurred among upstream sampling sites, but until 2015, which saw the lowest mean number of stonefly taxa among upstream sites since the study began, these differences have not been significant. The mean number of caddisfly taxa has oscillated the most between sites and years, with no clear trends emerging (Figure 16C).

a. Changes in Ephemeroptera richness



b. Changes in Plecoptera richness



c. Change in Trichoptera richness

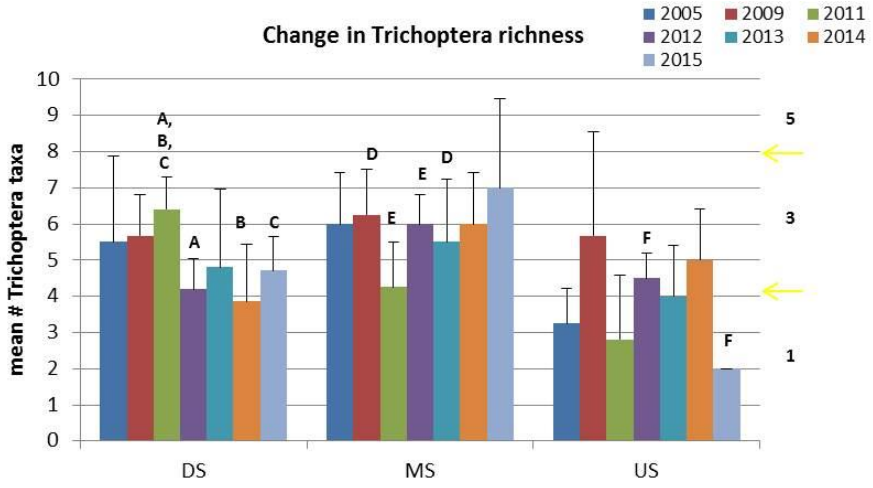
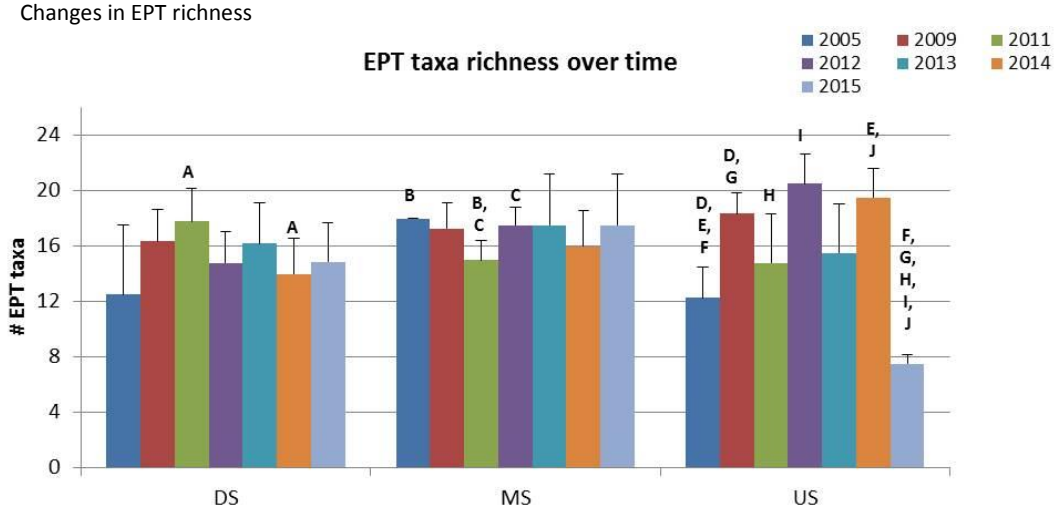


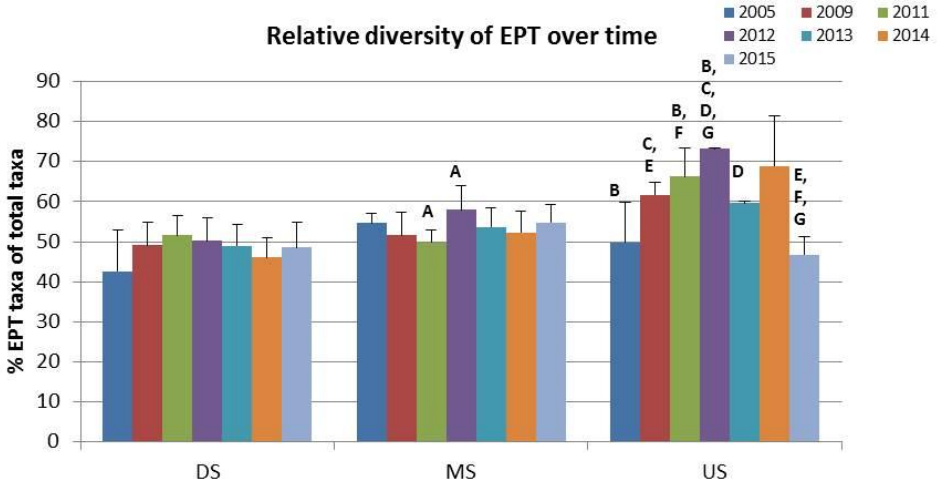
Figure 16. Changes in richness of Ephemeroptera, Plecoptera, and Trichoptera among Whychus Creek sampling sites. Letter pairs indicate significant difference between mean values (P<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25. Ranges for IBI scoring (5, 3, 1) are indicated.

Total EPT richness and relative diversity have changed little in downstream and midstream reaches (Figure 17A), but a steady increase in EPT taxa among upstream sites was sharply reversed in 2015. The proportion of all taxa comprised of EPT also rose steadily among upstream sites for several years, but this trend was reversed in 2015 (Figure 75B). The relative abundance of EPT individuals among samples has increased overall among all reaches; this increase has been most sustained in downstream and upstream reaches, with more annual variation in mid-reach sites (Figure 17C).

a. Changes in EPT richness



b. Changes in EPT relative diversity



c. Changes in EPT relative abundance

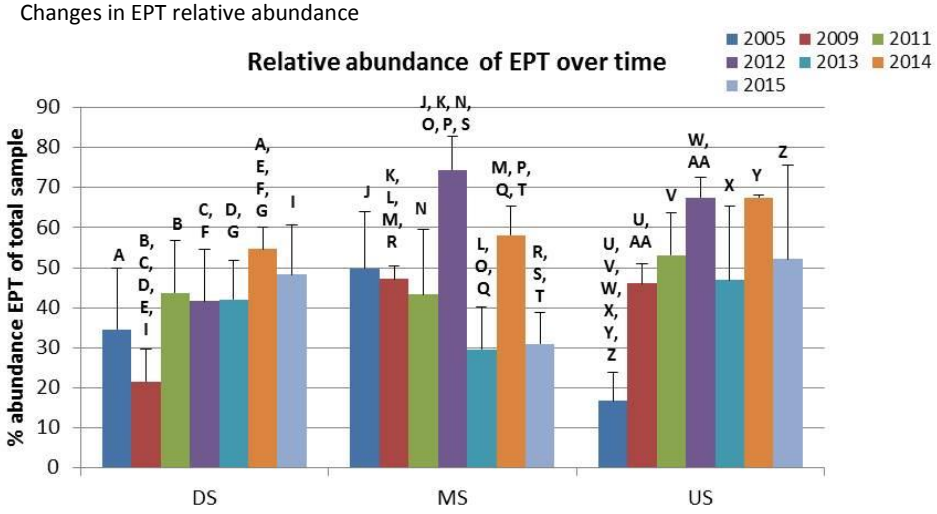


Figure 17. Changes in EPT (Ephemeroptera, Plecoptera, Trichoptera) diversity and relative abundance among Whychus Creek sampling sites. Letter pairs indicate significant difference ($p<0.05$). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Composition measures

The % dominance of the top (most abundant) taxon is a composition metric related to diversity. A healthy system is expected to have a more balanced composition, and a large abundance of a small number of taxa is considered indicative of impaired conditions and environmental stressors, as the macroinvertebrate community becomes dominated by one or a few more tolerant groups (Plafkin *et al*., 1989; Barbour *et al*., 1999). In the OR DEQ IBI, the most abundant taxon at a site must comprise less than 20% of the total abundance to receive the highest scaled score.

The proportion of the community dominated by the most abundant taxon shows a U-shaped pattern among downstream and upstream sites (Figure 18), decreasing from 2005-2011 then increasing 2013-2015, while the proportion of the dominant taxon increased at mid-reach sites though 2013 and then dropped again in recent years. The identity of the dominant taxon among sampling reaches has changed as overall macroinvertebrate community composition has changed, with the most noticeable shift occurring among downstream sampling sites. In 2005, *Glossosoma* (sensitive saddle case-maker caddisfly; OR DEQ low sediment indictor) and *Zaitzevia* (tolerant riffle beetle; OR DEQ indicator for warm temperatures and high sediment) were the dominant taxa at downstream sites, but in 2009 and 2011, dominant taxa consisted primarily of different common cosmopolitan taxa of non-biting and biting midges (Chironomidae and Ceratopogonidae, respectively), as well as Oligochaeta (disturbance- and sediment-tolerant aquatic earthworms; DEQ high sediment indicator). In 2013-2014, dominant taxa were comprised mainly of more lotic-associated taxa such as *Simulium* (common and moderately tolerant black fly) and *Baetis tricaudatus* (common cosmopolitan small minnow mayfly), and in 2015 they were joined by two caddisflies: *Brachycentrus* (humpless case-maker caddisfly sensitive to organic pollution) and *Hydropsyche* (common free-living caddisfly tolerant of organic pollution; warm temperature indicator).

Dominant taxa among mid-reach sites also changed across time. *Zapada* (small brown stonefly sensitive to organic pollution), Oligochaeta, and Orthocladiinae (subfamily of non-biting midge common in rock and gravel substrates) were the common dominant taxa in 2005-2009 among mid-reach sites, but from 2011-2014, *Baetis tricaudatus* (small minnow mayfly; DEQ low sediment indicator) and *Simulium* dominated the communities, and in 2015, *Simulium* was joined by the first instance of Tanytarsini as a dominant taxon. Tanytarsini is a tribe of non-biting chironomid midge in the subfamily Chironominae, a tolerant cosmopolitan group found in all types of aquatic habitats in sand and silt sediment (Merritt et al.; Pinder, 1986). The dominant taxa among upstream sampling sites have been more similar across time. Orthocladiinae was a dominant group in every sampling year except 2015, and Simulium and *Rhithrogena* (a flatheaded mayfly genus with low tolerance for organic pollution; low sediment indicator) occur in multiple years.

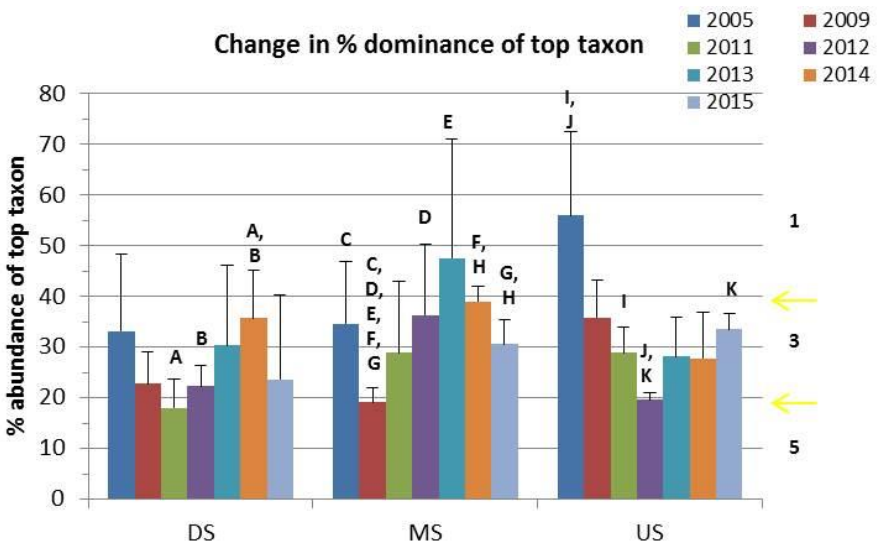


Figure 18. Dominance of the top taxon in Whychus Creek. Ranges for IBI scoring (5, 3, 1) are indicated. Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

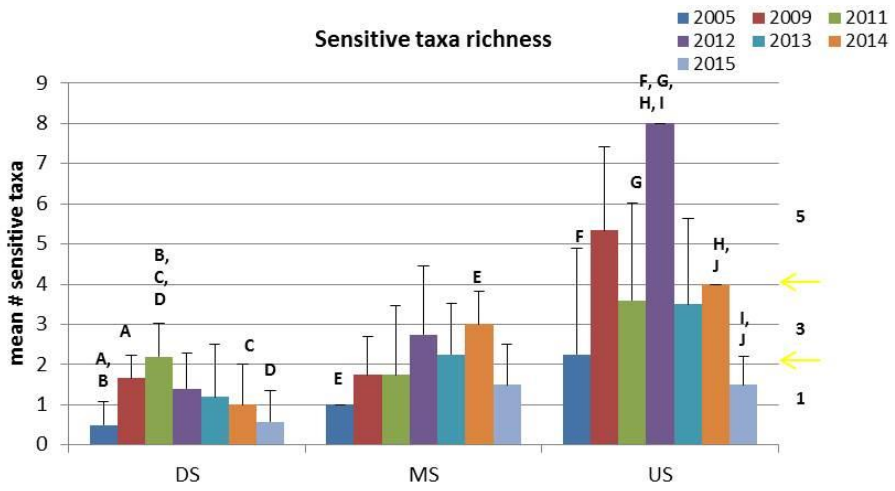
Sensitivity/tolerance measures

The remaining metrics in the IBI reflect diversity and dominance of groups that are sensitive to or tolerant of warm temperatures and increased sediment loads and include # of sensitive and sediment-intolerant taxa, %tolerant taxa (tolerant to high levels of disturbance and pollution) and sediment-tolerant taxa, and sample weighted mean of the MHBI (Modified Hilsenhoff Biotic Index) value. MHBI reflects tolerance for organic pollution (Hilsenhoff, 1987); values range from 0-10, with lower values indicating greater sensitivity. Tolerant taxa occur in both healthy and impaired habitats, but their ability to persist and even thrive under conditions with low dissolved oxygen, high turbidity, or heavy siltation means that as stressors increase and sensitive taxa drop out of the community, their relative abundance in the community increases.

Sensitive taxa diversity is highest among upstream sampling sites (Figure 19A) and increased steadily from 2005 to 2012, but sensitive taxa diversity at upstream sampling sites has decreased in recent years, and in 2015 the mean number of sensitive taxa was significantly lower than in both 2012 and 2014. Mid-reach sampling sites also showed a sustained increase in mean number of sensitive taxa since 2005, and although diversity was lower in 2015, the difference was not significant. Downstream sites have the lowest sensitive taxa diversity overall, and though sensitive taxa diversity rose among downstream sites from 2005 to 2011, this was followed by a continued decrease, and the mean number of sensitive taxa among downstream sites was significantly lower in 2014 and 2015 compared to 2011.

The mean numbers of sediment-intolerant taxa have shown an overall downward trend across all reaches over time, which does not reflect the observed pattern of lower sediment optima among the total community and among replacement and increaser taxa (Figure 19B). However, mean values in each year have had large standard deviations, and since the overall range of sediment-intolerant taxa is very limited in each year, ranging from only 0-2 among all sites, this metric has limited information content.

a. Number of sensitive taxa



b. Number of sediment-sensitive taxa

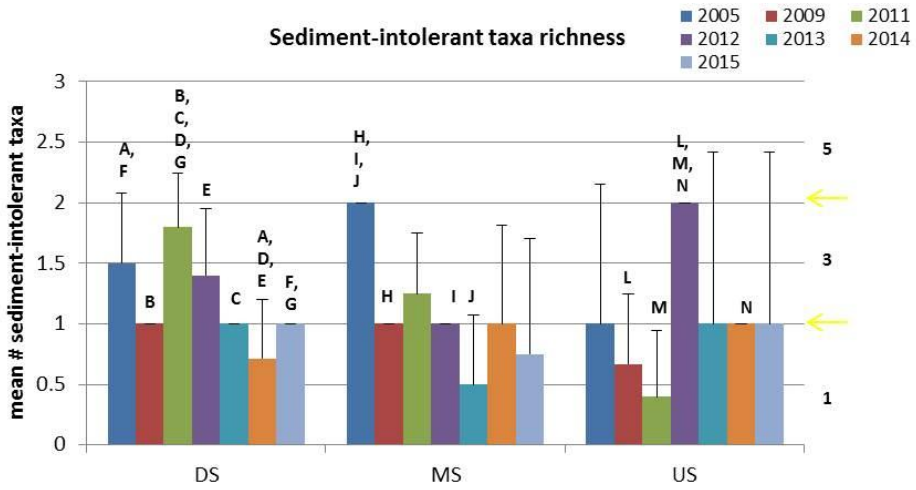
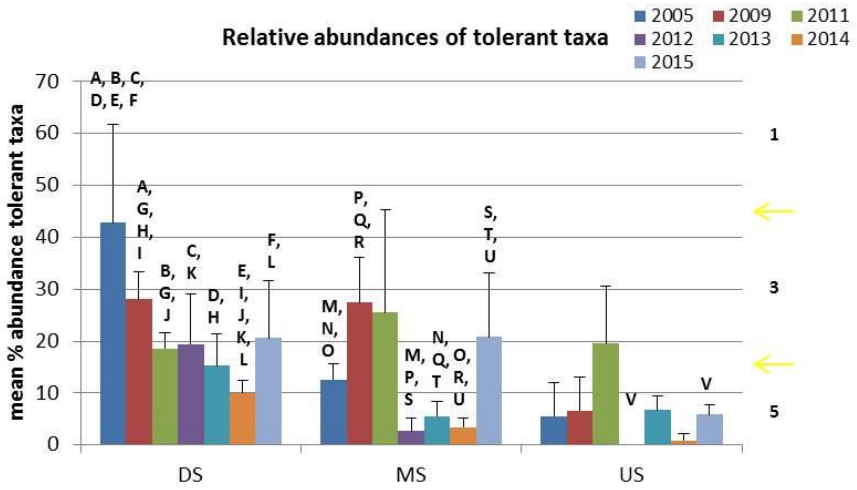


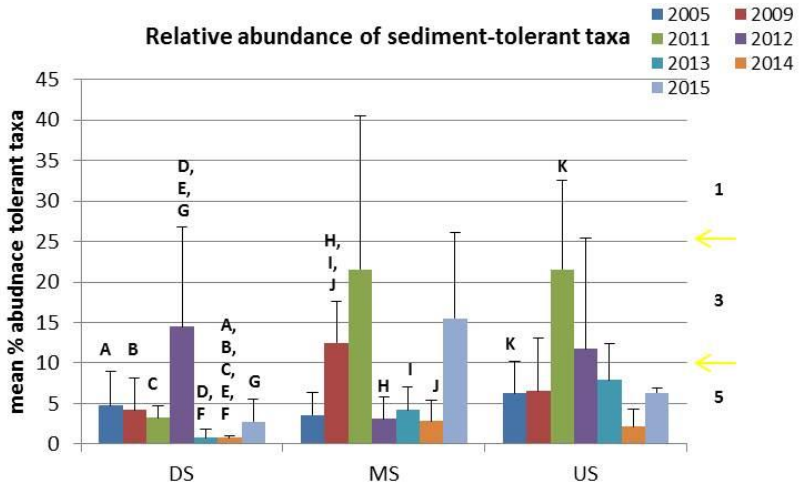
Figure 19. Richness of sensitive taxa among Whychus Creek sampling sites. Ranges for IBI scoring (5, 3, 1) are indicated. Letter pairs indicate significant difference between means. Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Greater changes have occurred among tolerant taxa in the sampling communities. Relative abundance of both tolerant and sediment-tolerant taxa decreased overall among all sampling reaches (Figure 20A, B), suggesting a response to improved stream conditions. However, in 2015 there were significantly greater proportions of tolerant taxa among all sampling reaches; relative abundance of sediment-tolerant taxa was higher as well but not significant. Mean assemblage MHBI in 2015 was similar to or lower than values seen in previous years (Figure 20C), and while mean community MHBI has varied across time, values have remained below or close to the range that receives the highest scaled score in the IBI (MHBI <4).

a. Changes in relative abundance of tolerant taxa



b. Changes in relative abundance of sediment-tolerant taxa



c. Changes in weighted mean of assemblage MHBI

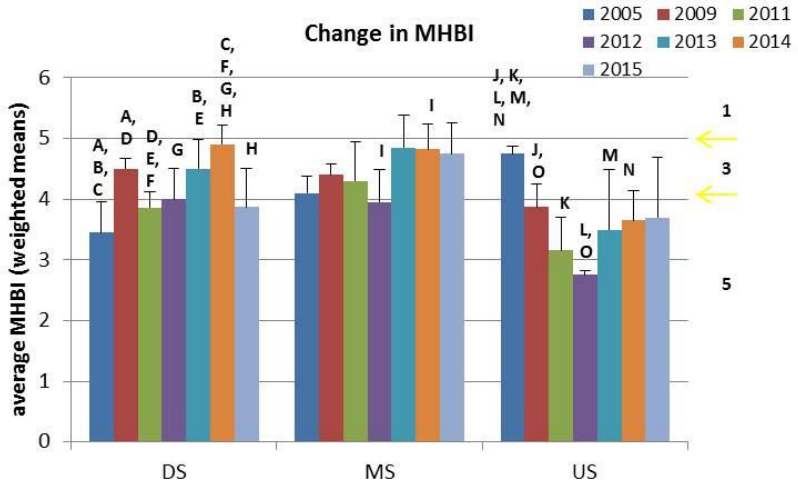


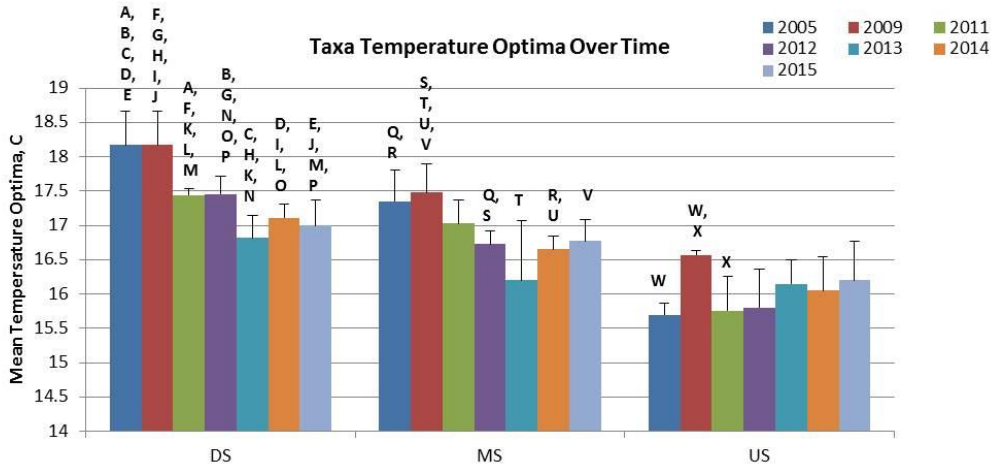
Figure 20. Changes in tolerant taxa measures among Whychus Creek sampling sites. Ranges for IBI scoring (5, 3, 1) are indicated (note that these are negative metrics, i.e. a higher raw value receives a lower scaled IBI score). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Community temperature and sediment optima

The weighted mean of temperature and sediment optima of macroinvertebrate communities collected in all sampling years was determined (Figure 21). Macroinvertebrate assemblages at upstream sites have the lowest mean temperature optima, which then increased through mid-reach and downstream sites. These differences are expected based on predictions of the River Continuum Concept (Vannote et al. 1980) and are observable in the stream itself, as colder, faster, deeper headwaters transition to shallower stretches flowing through a landscape with more human disturbance and impervious surfaces. However, within this framework, the mean community temperature optima have decreased significantly during the course of this study among downstream and mid-reach sampling sites (Figure 21A). Mean temperature optima of downstream sites show a stepwise pattern of decrease; optima in 2013-2015 are significantly lower than those of all prior years, but not significantly different from each other, suggesting that response to changed temperature regimes may have stabilized. Mid-reach communities also showed a significant decrease in mean temperature optima through 2013, and though the community means in 2014-2015 were higher than in 2013, differences were not significant, again suggesting potential stabilization. In contrast, mean temperature optima of upstream communities have changed little; the community optima in 2009 was significantly higher than 2005, but in 2011 it fell again and there are no significant differences in community temperature optima from 2011 through 2015.

Community fine sediment optima are relatively low overall and similar among all reaches, but %FSS optima in the 2005 & 2009 communities are significantly higher than the communities in 2013-2015. Among downstream and upstream sites, differences in mean %FSS optima are not significant across 2013-2015. Among mid-reach sites, the pattern of lower sediment optima seen in 2012-2014 was reversed in 2015, when mean %FSS optima increased significantly to levels similar to the earliest years of the study.

a. Macroinvertebrate community temperature optima (weighted means).



b. Macroinvertebrate community % fine sediment optima (weighted means).

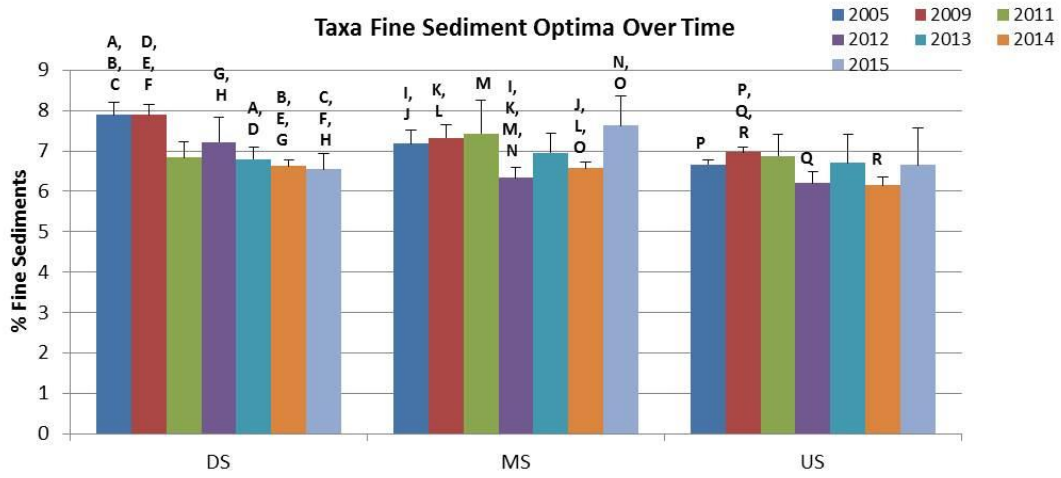


Figure 21. Temperature and % fine sediment optima among macroinvertebrate assemblages collected in Whychus Creek. Letter pairs indicate significant difference between means ($p<0.05$). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25

Functional feeding groups

Because food sources and availability play a large role in structuring aquatic macroinvertebrate communities, examining different functional feeding groups (FFGs) can be informative. The proportion of different FFGs varies naturally based on stream order, reach location, and different inputs of allochthonous (terrestrial-derived, i.e. leaves, seeds, wood, carcasses, feces) and autochthonous (stream-derived, i.e. photosynthesis by primary producers, decomposition of dead organisms, feces) nutrients into the stream (Vannote et al., 1980). However, FFG distributions are further influenced by surrounding land uses, hydrologic alterations, and excess nutrients, sediment, or contaminants.

The main FFGs of aquatic macroinvertebrates are shredders, collectors, scrapers, and predators (reviewed in Wallace & Webster, 1996). Shredders (SH) rely on terrestrial organic input such as leaf litter and wood (coarse particulate organic matter, CPOM, >1 mm diameter), and are often dominant in headwater reaches, especially where there is substantial riparian shading. Their reliance on CPOM

makes them particularly sensitive to riparian conditions and surrounding land uses that impact allochthonous inputs to the stream; because SH are more specialized in their feeding and less tolerant of disturbance, their proportions are expected to be lower in more impaired habitats.

Collectors include gatherers (CG), which take in fine particulate organic matter (FPOM, <1 mm diameter) deposited on surfaces and sediment, and filterers (CF), which take in suspended particles of FPOM. Collector-gatherers are often an abundant component of the stream biota and an important prey resource for invertebrate and vertebrate predators. Because collectors are generalist feeders with a fairly broad food range, they can be more tolerant of disturbances that might alter food availability, and changes in their abundances with impairment vary.

Scrapers (SC), also called grazers, rasp algae and diatoms off mineral and organic surfaces, and can become dominant where primary production is maximized. Scrapers are a more specialized feeding group that require abundant periphyton; their proportions can be lower in impaired habitats, as the filamentous algae and mosses that increase as an early response to stream pollution interfere with their ability to feed.

Predators (PR) consume other animals by mastication or using piercing mouthparts to suck out body contents. Their proportions within the community remain relatively steady with stream order but because they rely on an abundant food base, and different life stages of a single taxon may require prey of different sizes, the PR proportions are expected to be lower at more disturbed or impaired sites.

Overall, FFG examination was not revealing. Predator taxa were present in the greatest proportions among all sampling sites and years, followed by CG, SC, and SH, with CF comprising the lowest proportion of all taxa (Fig 22). The relative diversity and abundance of both predator and shredder taxa varied greatly across years and stream reaches (Figures 23 and 25). More significant changes were seen among the more specialized SC taxa (Figure 24), with overall decreased diversity and relative abundance across time. Among collectors, CF changed more over time than CG, with increased diversity and relative abundance over time (Figure 26).

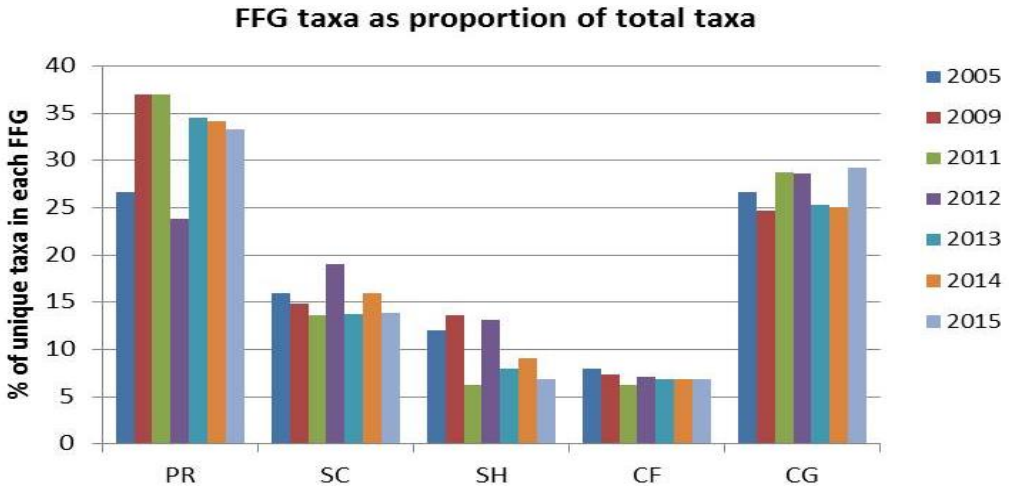
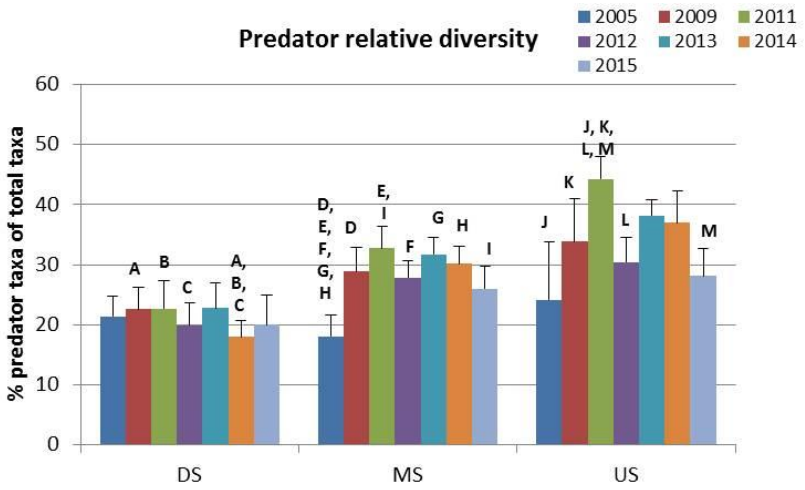


Figure 22. Proportion of total taxa collected across all Whychus Creek sampling reaches comprised by each functional feeding group. PR, predator; SC, scraper; SH, shredder; CF, collector-filterer; CG, collector-gatherer.

a. Relative diversity of predator taxa



b. Relative abundance of predator taxa

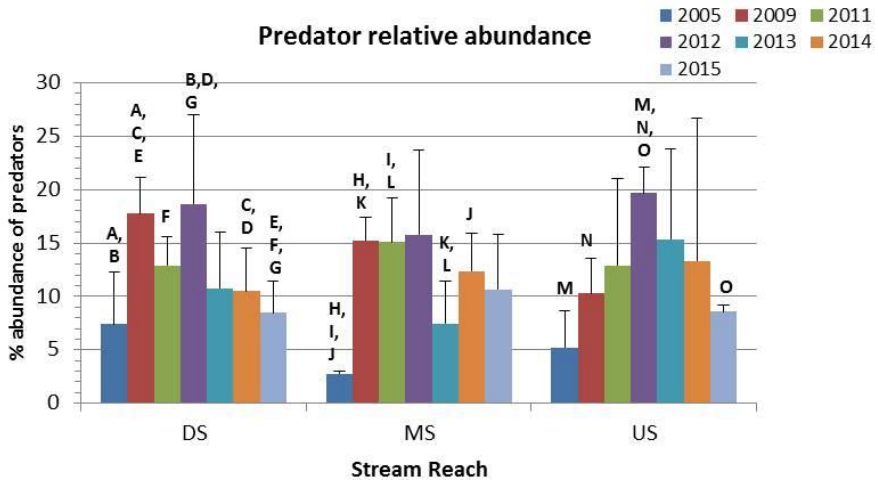
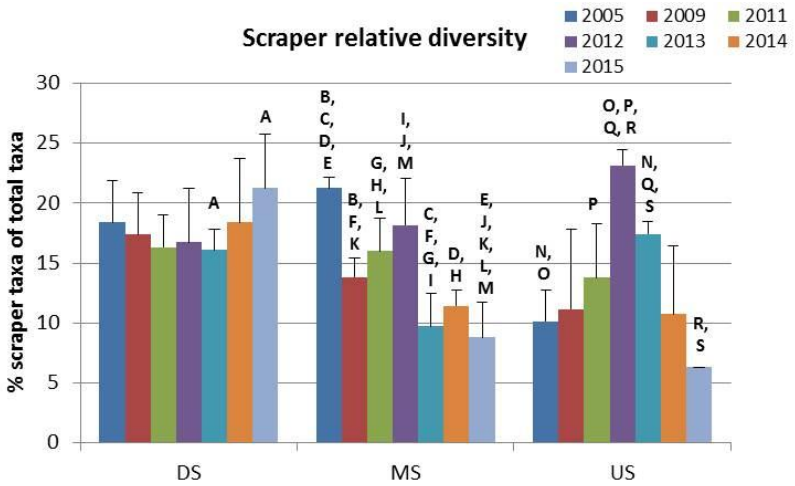


Figure 23. Changes in predator taxa over time among Whychus Creek sampling sites. Letter pairs indicate significant difference between means (P<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

a. Relative diversity of scraper taxa



b. Relative abundance of scraper taxa

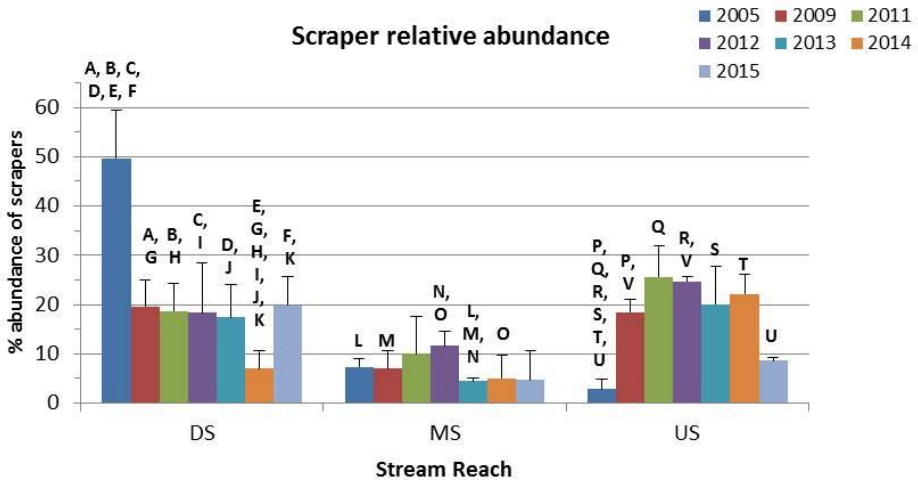
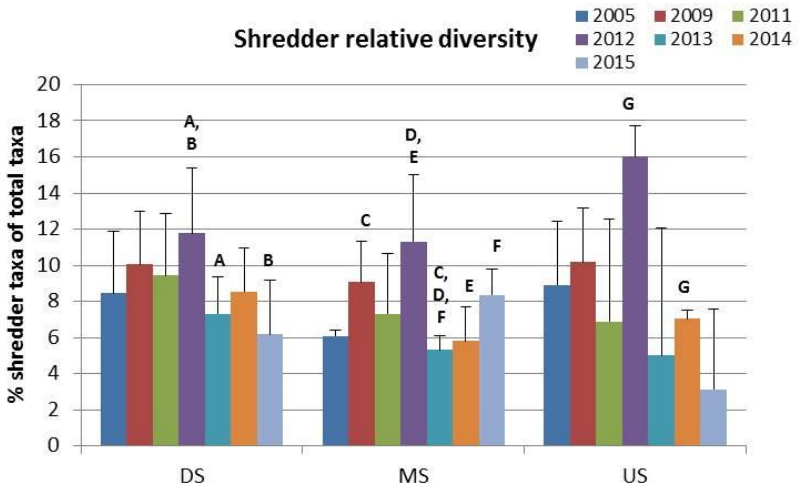


Figure 24. Changes in scraper taxa among Whychus Creek sampling sites. Letter pairs indicate significant difference between means ($p<0.05$). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

a. Relative diversity of shredder taxa



b. Relative abundance of shredder taxa

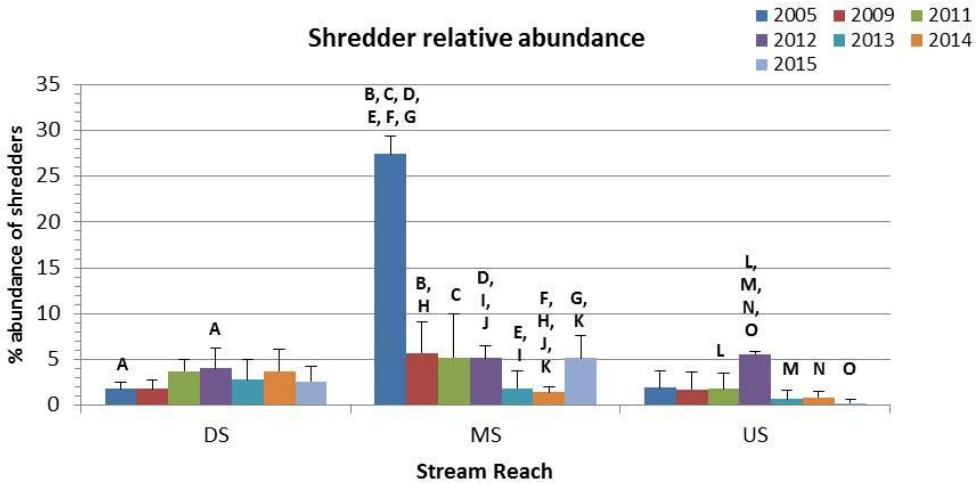
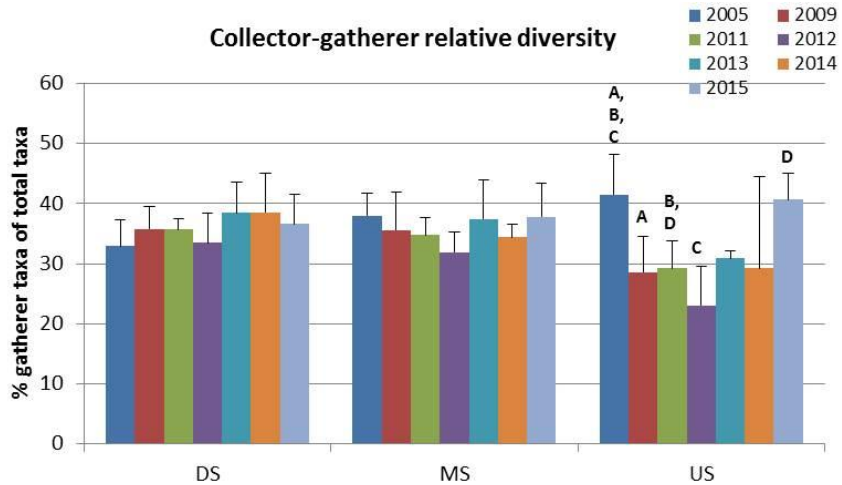
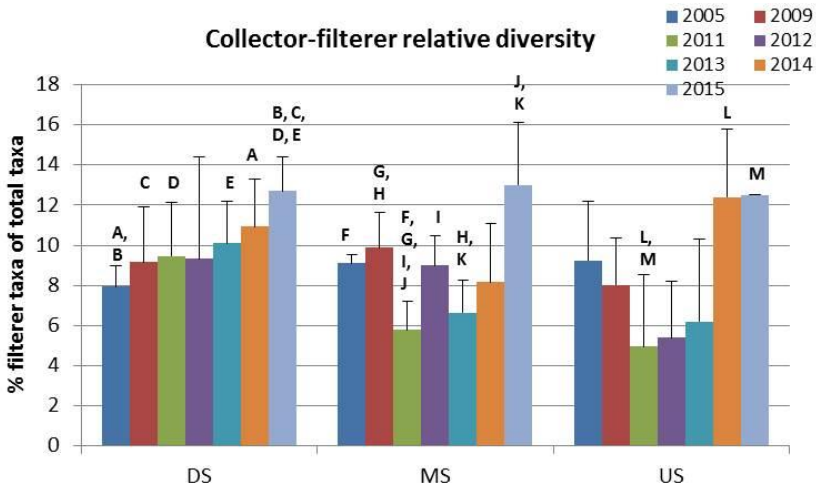


Figure 25. Changes in shredder taxa over time among Whychus Creek sampling sites. Letter pairs indicate significant difference between means ($p<0.05$). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

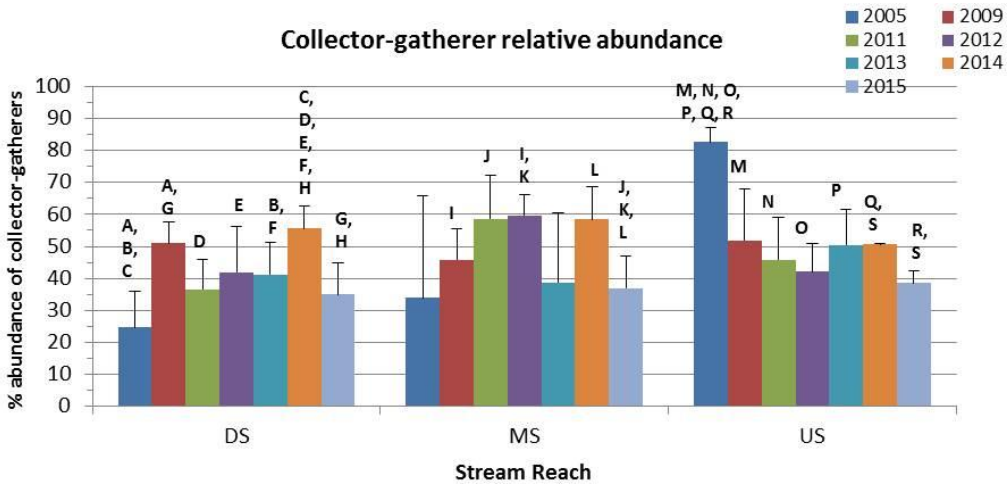
a. Relative diversity of collector-gatherer taxa



b. Relative diversity of collector-filter taxa



c. Relative abundance of collector-gatherer taxa



d. Relative abundance of collector filterer taxa

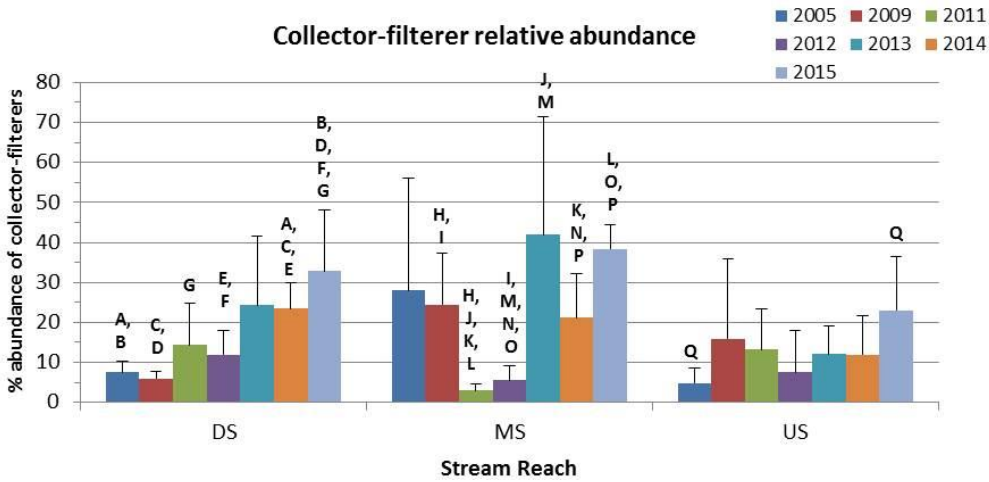


Figure 26. Changes in collector taxa (CG and CF) over time among Whychus Creek sampling sites. Letter pairs indicate significant difference between means ($p<0.05$). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Conclusions

The recovery trajectory of aquatic invertebrates in restored streams is neither linear nor instantaneous, and is influenced by a range of variables, including degree of continuing change within microhabitats in restored reaches; additional reach- or basin-scale stressors that continue to impact the restored system; distance and composition of the regional pool of colonists; interactions between species and microhabitats that 'filter' species with suitable traits from the regional species pool ; and resistance of entrenched tolerant taxa to replacement by more sensitive colonists. Thus, effectiveness monitoring of stream restoration projects must be long-term to be ecologically meaningful. In addition, studies that span only a few years may underestimate the natural variability of communities by missing disturbances that occur infrequently (i.e., major floods or fires) or as part of extended climate cycles (i.e., drought). The need for long-term biomonitoring is generally recognized (Bernhardt et al., 2005; Palmer et al., 2005), but limitations in staff, funding, and time still act to make multi-year monitoring projects the exception rather than the norm.

The Whychus Creek project study is thus noteworthy for its 10-year span; even though restoration work is still occurring, this time frame has enabled examination of long-term changes in the macroinvertebrate community within a relevant span for colonization and establishment of new taxa. It has also covered periods of different climatic conditions, with the overall lower taxa richness and organismal abundance among sampling sites likely reflecting the impacts of a dry winter and spring and an abnormally hot summer. This project has increased in complexity since its inception, when it was anticipated that biological condition scores generated by PREDATOR and IBI models would show steadily improving biological conditions resulting from stream restoration activities. These scores have been less revealing than hoped for, however; IBI scores have consistently reflected better biological conditions that PREDATOR O/E scores, but annual variations have not revealed any strong trends. PREDATOR scores have additionally changed little across the years, although O/E scores were higher among the majority of sites in 2015 compared to previous years. However, analysis of the temperature and sediment optima of the sampled community as a whole, along with that of taxa identified by PREDATOR as missing/replacement and decreaser/increasers, reveal consistently and significantly lower %FSS optima with the community over time, and among replacement and increaser taxa groups. Temperature optima within the community have also decreased significantly over time among downstream and mid-reach sampling sites.

Because similarity analyses of community assemblages showed between-year changes in community composition, additional aspects of the macroinvertebrate community have been characterized over time, many of which are more informative. Examination of individual taxonomic and ecological traits of the community suggest a measureable response to restoration activities, with increasing diversity and abundance of taxa adapted for lower temperatures and percent fine sediment loads. Community similarity analyses consistently group the 2005 and 2009 sampling communities separately from the 2011-2015 communities, indicating that earlier restoration activities had the greatest overall impact on the community. Examination of the taxa that contribute most to these community differences generally reveals groups that are more tolerant and/or associated with slower waters present in greater abundance in the earlier years, while taxa that are more abundant in later years comprise more sensitive groups (including Ephemeroptera, Plecoptera, and Trichoptera; EPT), many of which are also OR DEQ indicators for cool temperatures or low sediment conditions.

New taxa are added to the overall dataset every year, indicating that regional colonists are finding the restored reaches, and that dominance shifts may have occurred among existing resident taxa. The rate

of addition of new taxa to the overall dataset has slowed with each year of sampling, but the majority of new taxa seen in each year continue to be in the sensitive EPT group.

Thus, although standard assessments such as PREDATOR and IBI have not indicated substantial improvements in biological conditions in Whychus Creek, analysis of multiple attributes and characteristics strongly suggests a community whose composition has changed in response to changing stream conditions, with a greater abundance and/or diversity of taxa that thrive in conditions with cooler temperatures and lower percent fine sediments. In recent years it seemed that the community changes were slowing and community composition stabilizing, but implementation of a new floodplain restoration project in the upstream reaches, and possibly the impacts of anomalous weather in 2015, led to decreased abundance ad taxa richness and lower values for several of the metrics used in analysis, although PREDATOR O/E scores were higher than in previous years. Increasing frequency and severity of altered precipitation patterns and wildfires may contribute to community changes in the coming years, along with other stressors likely present in the watershed that have not been remediated by recent restoration activities. Integrated analysis of macroinvertebrate community data along with water chemistry, surrounding landscape use, and climate variables in the future could provide additional insights about drivers of community changes.

Literature Cited

Adams J., M. Vaughan, and S. H. Black. 2003. Stream bugs as biomonitors: guide to Pacific Northwest macroinvertebrate monitoring (CD-ROM). The Xerces Society for Invertebrate Conservation, Portland OR.

Arce, E., V. Archimbault, C.P. Mpondy, and P. Usseglio-Polatera. 2014. Recovery dynamics in invertebrate communities following water quality improvement: taxonomy- vs trait-based assessment. Freshwater Science 33(4): 10560-1073.

Barbour M.T., J. B. Stribling, J. Gerritsen, and J. R. Karr. 1996. Biological criteria: technical guidance for streams and small rivers. EPA 822-B-96-001, United States Environmental Protection Agency, Office of Water, Washington DC.

Bernhardt, E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Barnas, S. Brooks, et al. 2005. Synthesizing US river restoration efforts. Science 308: 636-637.

Blakely, T. J., J. S. Harding, A. R. McIntosh, and M. J. Winterbourn. 2006. Barriers to the recovery of aquatic insect communities in urban streams. Freshwater Biology 51(9): 1634-1645.

Bohn, B. A. and J. L. Kershner. 2002. Establishing aquatic restoration priorities using a watershed approach. Journal of Environmental Management 64: 355-363.

Bond N. R. and P. S. Lake. 2003. Local habitat restoration in streams: constraints on the effectiveness of restoration for stream biota. Ecological Management & Restoration 4(3): 193-198.

Booth, D. B. and C. R. Jackson. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association 33(5): 1077-1090.

Clarke K. R. and R. M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation, 2nd ed. PRIMER-E, Plymouth, United Kingdom.

Crunkilton, R. L. and R. M. Duchrow. 1991. Use of stream order and biological indices to assess water quality in the Osage and Black River basins of Missouri. Hydrobiologia 224: 155-166.

Culp, J.M., D.G. Armani, M.J. Dunbar, J.M. Orlofske, N.L. Poff, A.I. Pollard, A.G. Yates, and G.C. Hose. 2011. Incorporating traits in aquatic biomonitoring to enhance causal diagnosis and prediction. Integrated Environmental Assessment and Management 7(2): 187-197.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. Great lakes Entomologist 20: 31-39.

Hubler, S., B. Lamb, and D. Drake. 2009. Identifying causes of impairment for the Crooked River Basin TMDL: an application of Oregon's Stressor ID models for temperature and fine sediment. Northwest Biological Assessment Workgroup presentation, unpublished.

Hubler S. 2008. PREDATOR: development and use of RIVPACS-type macroinvertebrate models to assess the biotic condition of wadeable Oregon streams. DEQ08-LAB-0048-TR, State of Oregon Department of Environmental Quality, Laboratory Division, Watershed Assessment Section. 51 pp.

Karr J.R. and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington D. C. 206 pp.

Lake, P. S., N. Bond, and P. Reich. 2007 Linking ecological theory with stream restoration. Freshwater Biology 52: 597–615.

Lange, K., C.R. Townsend, and C.D. Matthaei. 2014. Can biological traits of stream invertebrates help disentangle the effects of multiple stressors in an agricultural catchment? Freshwater Biology 59: 2431-2446.

Lillie, R. A., S. W. Szczytko, and M. A. Miller. 2003. Macroinvertebrate data interpretation guidance manual. Wisconsin Department of Natural Resources, Madison WI. 64 pp.

Merritt, R.R., K.W. Cummins, and M.B. Berg. 2008. An introduction to the aquatic insects of North America, 4th edition. Kendall/Hunt Publishing Company, Dubuque, Iowa. 1158 pp.

Oregon Watershed Enhancement Board. 2003. OWEB water quality monitoring technical guidebook. 152 pp. Available at http://www.oregon.gov/OWEB/docs/pubs/wq_mon_guide.pdf.

Palmer, M.A., R. Ambrose, R. and N.L. Poff. 1997. Ecological theory and community restoration ecology. Journal of Restoration Ecology 5: 291–300.

Palmer, M.A., E.S. Bernhardt, J.D. Allan, P.S. Lake, G. Alexander, S. Brooks, J. Carr, S. Clayton, C.N. Dahm, et al. 2005. Standards for ecologically successful river restoration. Journal of Applied Ecology 42: 208-217.

Pinder, L.C.V. 1986. Biology of freshwater Chironomidae. Annual Review of Entomology 31: 1-23.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. EPA/444/4-89-001, U.S. Environmental Protection Agency, Washington DC.

Pollard, A.I. and L.L. Yuan. 2010. Assessing the consistency of response metrics of the invertebrate benthos: a comparison of trait- and identity-based measures. Freshwater Biology 55(7): 1420-1429.

Roni, P., T.J. Beechie, R.E. Bilby, F.E. Leonetti, M.M. Pollock and G.R. Pess. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in pacific Northwest watersheds. North American Jounral of Fisheries Management 22: 1-20.

Rosenberg D. M. and V. H. Resh. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York, NY.

Southwood, T.R.E. 1977. Habitat, the templet for ecological strategies? Journal of Animal Ecology 46: 337-365.

U.S. EPA. 2013. National rivers and streams assessment 2008-2009: a collaborative survey. EPA/841/D-13/001, U.S. Environmental Protection Agency Office of Wetlands, Oceans and Watersheds Office of Research and Development Washington, DC. 124 pp.

Van den Brink, P.J., A.C. Alexander, M. Desrosiers, W. Goedkoop, P.L.M. Goethals, M. Liess, and S.D. Dyer. 2011. Traits-based approaches in bioassessment and ecological risk assessment: strengths, weaknesses, opportunities, and threats. Integrated Environmental Assessment and Management 7(2): 198-208.

Vannote, R.L.,Minshall G. W., Cummins K. W., Sedell J.R., and Cushing, E. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Science 37: 130-137.

Wallace, J.B. and J.R. Webster. 1996. The role of macroinvertebrates in stream ecosystem function. Annual review of Entomology 41: 115-139.

APPENDIX A Whychus Creek field data sheet

Site ID: Date: Sampled by:

Start time: _____ End time: _____ Air temp _____ °C Water temp. _____ °C

Sample Information:

# of riffles sampled: _____ Field duplicate collected: ___ yes ___ no

# of kicks composited ___ 8 x 1 ft² OR ___other (describe): total # field duplicate jars _____

Total # sample jars _____

Human use & influence (*check **all** that apply; right & left bank relative to observer facing downstream*)

A = absent	**B** = on bank	**C** = ≤ 30 ft from bank	**D** = > 30 ft from bank			
Disturbance	**Left bank**	**Right bank**		**Disturbance**	**Left bank**	**Right bank**
Riprap/wall/dike/armored bank				Landfill/trash		
Buildings				Park/lawn		
Industrial				Row crops		
Rural residential				Pasture/range/hay field		
Urban residential				Livestock w/stream access		
Pavement/cleared lot				Logging within last 5 years		
Road/railroad				Sand or gravel mining		
Pipes (inlet/outlet)				Forest/woodland		
Other:						

Qualitative observations:

Water odors: none **/** organic **/** rotten eggs **/** fishy **/** chlorine **/** petroleum **/** other (describe):
Water appearance: clear **/** turbid **/** milky **/** dark brown **/** foamy **/** oily sheen **/** other (describe):
Dominant land use: Forest **/** agriculture (crops / pasture) **/** urban (industrial / residential) **/** other:
Extent of algae covering submerged materials: none **/** 1-25% **/** 25-50% **/** 50-75% **/** 75-100%
Type of algae: none **/** filamentous (i.e. strands >2”) **/** close-growing **/** floating clumps

Physical characteristics:
(*if your reach has fewer than 8 riffles, record characteristics only for the number of riffles sampled*)

Substrate

% composition of riffle	**Riffle1**	**Riffle2**	**Riffle3**	**Riffle4**	**Riffle5**	**Riffle6**	**Riffle7**	**Riffle8**
Bedrock (continuous rock)								
Boulder (> 12 in.; larger than basketball)								
Cobble (2.5-12 in.; tennis ball to basketball)								
Gravel (0.6-2.5 in.; marble to tennis ball)								
Sand (< 0.6 in.; smaller than marble)								
Silt/clay/muck (fine suspended particles)								
Woody debris								
Other (describe)								

Water depth
Please record units used (*check measuring tape—one side is metric, the other English*)

Parameter	**Riffle1**	**Riffle2**	**Riffle3**	**Riffle4**	**Riffle5**	**Riffle6**	**Riffle7**	**Riffle8**
Wetted width								
Depth @ ¼ wetted width								
Depth @ ½ wetted width								
Depth @ ¾ wetted width								

Additional notes or observations (including other wildlife noted):

APPENDIX B OR DEQ indicator taxa for temperature and fine sediment stressors. Values in parentheses indicate temperature (oC) or sediment (% fine sediment) optima for each taxon.

Taxon	Temperature indicator	Fine sediment indicator
Prosimulium	Cool (12.2)	---
Baetis bicaudatus	Cool (12.3)	---
Zapada columbiana	Cool (12.9)	---
Neothremma	Cool (12.9)	---
Parapsyche elsis	Cool (13.5)	Low (4)
Caudatella	Cool (13.6)	Low (4)
Megarcys	Cool (13.6)	Low (4)
Visoka	Cool (13.7)	---
Epeorus grandis	Cool (14.2)	Low (2)
Yoraperla	Cool (14.2)	---
Ephemerella	Cool (14.4)	---
Drunella coloradensis/flavilinea	Cool (14.5)	---
Doroneuria	Cool (14.5)	---
Despaxia	Cool (14.5)	---
Turbellaria	Cool (14.6)	---
Ironodes	Cool (14.9)	---
Drunella doddsi	Cool (15.2)	Low (3)
Ameletus	Cool (15.2)	---
Rhyacophila Brunnea Gr.	Cool (15.5)	Low (4)
Cinygmula	Cool (15.5)	Low (6)
Micrasema	Cool (15.6)	---
Diphetor hageni	Warm (17.9)	---
Antocha	Warm (18.3)	---
Hydropsyche	Warm (18.5)	---
Juga	Warm (18.6)	High (15)
Chironomini	Warm (18.8)	High (10)
Zaitzevia	Warm (19.0)	High (9)
Optioservus	Warm (19.6)	High (12)
Dicosmoecus gilvipes	Warm (20.6)	---
Physa	Warm (21.1)	High (21)
Arctopsyche	---	Low (2)
Rhyacophila Hyalinata Gr.	---	Low (3)
Rhyacophila Angelita Gr.	---	Low (3)
Drunella grandis	---	Low (3)
Epeorus longimanus	---	Low (4)
Rhithrogena	---	Low (5)

Rhyacophila Betteni Gr.	---	Low (5)
Glossosoma	---	Low (5)
Baetis tricaudatus	---	Low (6)
Oligochaeta	---	High (10)
Paraleptophlebia	---	High (11)
Tanypodinae	---	High (12)
Ostracoda	---	High (17)
Hydroptila	---	High (17)
Lymnaeidae	---	High (18)
Cheumatopsyche	---	High (20)
Sphaeriidae	---	High (21)
Coenagrionidae	---	High (25)

APPENDIX C Macroinvertebrate taxa list for Whychus Creek, 2005-2015

Phylum or subphylum	Class or Subclass	Order	Family	Genus	Species	2005	2009	2011	2012	2013	2014	2015
Platyhelminthes	Turbellaria					√	√	√	√	√		√
Annelida	Oligochaeta					√	√	√	√	√	√	√
Annelida	Hirudinea	Rhynchobdellida	Glossiphoniidae	Helobdella	stagnalis					√		
Nematoda						√	√	√	√	√	√	√
Arthropoda/ Crustacea	Malacostraca	Amphipoda						√		√		
Arthropoda/ Crustacea	Malacostraca	Decapoda	Astacidae	Pacifasticus			√					√
Arthropoda/ Crustacea	Copepoda							√				
Arthropoda/ Crustacea	Ostracoda					√		√	√	√		√
Arthropoda	Arachnoidea	Trombidiformes				√	√	√	√	√	√	
Arthropoda	Insecta	Coleoptera	Elmidae	Narpus		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Optioservus		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Zaitzevia		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Cleptelmis		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Ampumixis		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Lara	avara		√	√	√	√	√	
Arthropoda	Insecta	Coleoptera	Dytiscidae			√		√		√	√	√
Arthropoda	Insecta	Coleoptera	Dryopidae	Helichus			√					√
Arthropoda	Insecta	Coleoptera	Hydrophilidae	Ametor			√	√		√		√
Arthropoda	Insecta	Odonata	Coenagrionidae			√						

Arthropoda	Insecta	Diptera	Empididae	Neoplasta			√	√	√	√		√
Arthropoda	Insecta	Diptera	Empididae	Hemerodromia		√	√					√
Arthropoda	Insecta	Diptera	Empididae	Chelifera		√					√	√
Arthropoda	Insecta	Diptera	Empididae	Clinocera		√	√	√		√	√	√
Arthropoda	Insecta	Diptera	Empididae	Roederoides						√		
Arthropoda	Insecta	Diptera	Empididae	Wiedemannia		√						
Arthropoda	Insecta	Diptera	Tipulidae	Antocha		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Cryptolabis		√						√
Arthropoda	Insecta	Diptera	Tipulidae	Dicranota			√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Hesperoconopa		√	√				√	√
Arthropoda	Insecta	Diptera	Tipulidae	Hexatoma		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Limnophila			√	√				
Arthropoda	Insecta	Diptera	Tipulidae	Rhabdomastix			√					
Arthropoda	Insecta	Diptera	Athericidae	Atherix		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Dixidae	Dixa			√		√			
Arthropoda	Insecta	Diptera	Chironomidae	Tanypodinae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Chironominae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Diamesinae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Ceratopogoninae			√					√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Atrichopogon							√	
Arthropoda	Insecta	Diptera	Ceratopogonidae	Dasyhelea				√	√	√	√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Forcipomyia				√				
Arthropoda	Insecta	Diptera	Blephariceridae	Bibiocephala					√			

Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicera		√	√	√		√	√	√
Arthropoda	Insecta	Diptera	Deuterophlebiidae								√	
Arthropoda	Insecta	Diptera	Simuliidae	Prosimulium		√	√	√	√		√	
Arthropoda	Insecta	Diptera	Simuliidae	Simulium		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Ephydridae			√	√					√
Arthropoda	Insecta	Diptera	Psychodidae	Pericoma		√	√			√		
Arthropoda	Insecta	Diptera	Psychodidae	Maruina			√			√		
Arthropoda	Insecta	Diptera	Tabanidae				√	√				
Arthropoda	Insecta	Diptera	Sciomyzidae							√		
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella		√	√	√	√	√		
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella	turbida		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella	insignificans							√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetis		√	√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetis	tricaudatus		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Diphetor	hageni	√	√		√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Labiobaetis								√
Arthropoda	Insecta	Ephemeroptera	Ameletidae	Ameletus		√	√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella		√	√		√			√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella	delantala							√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella	margarita			√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Serratella		√						
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella		√	√	√	√			√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella (Serratella)	tibialis		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella	dorothea			√				

Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella	excrucians		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	edmundsi						√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	hystrix	√	√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	spinifera	√		√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	flavilinea				√			
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	doddsi					√	√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	coloradensis		√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus		√	√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	albertae			√	√	√		√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	deceptivus/ hesperus			√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	grandis		√	√	√		√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	longimanus		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Rhithrogena		√	√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygmula		√	√	√	√	√		√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygma					√			
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	Tricorythodes		√						
Arthropoda	Insecta	Ephemeroptera	Leptophlebiidae	Paraleptophlebia		√	√	√	√	√	√	√
Arthropoda	Insecta	Hemiptera	Corixidae								√	
Arthropoda	Insecta	Hemiptera	Gerridae								√	
Arthropoda	Insecta	Megaloptera	Sialidae	Sialis		√						
Arthropoda	Insecta	Plecoptera	Perlidae			√	√	√				
Arthropoda	Insecta	Plecoptera	Perlidae	Calineuria	californica	√	√					
Arthropoda	Insecta	Plecoptera	Perlidae	Hesperoperla		√						
Arthropoda	Insecta	Plecoptera	Perlidae	Doroneuria						√		

Arthropoda	Insecta	Plecoptera	Perlodidae			√	√	√	√	√	√	
Arthropoda	Insecta	Plecoptera	Perlodidae	Isoperla			√	√			√	
Arthropoda	Insecta	Plecoptera	Perlodidae	Megarcys			√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Perlodidae	Rickera	sorpta		√	√			√	
Arthropoda	Insecta	Plecoptera	Perlodidae	Kogotus/Rickera		√				√	√	√
Arthropoda	Insecta	Plecoptera	Perlodidae	Skwala		√	√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Paraperla		√			√		√	
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Suwallia			√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Sweltsa		√	√	√		√	√	√
Arthropoda	Insecta	Plecoptera	Leuctridae				√	√				
Arthropoda	Insecta	Plecoptera	Leuctridae	Despaxia	augusta		√					
Arthropoda	Insecta	Plecoptera	Nemouridae	Amphinemura					√			
Arthropoda	Insecta	Plecoptera	Nemouridae	Visoka	cataractae	√	√	√	√		√	√
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada		√	√	√	√			
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	cinctipes		√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	columbiana		√		√	√		
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	oregonensis			√			√	
Arthropoda	Insecta	Plecoptera	Pteronarcyidae	Pteronarcys		√	√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Yoraperla		√						
Arthropoda	Insecta	Plecoptera	Capniidae			√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Apataniidae	Pedomoecus			√			√		
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Agapetus		√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Glossosoma		√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Arctopsyche	grandis	√	√					√
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Hydropsyche		√	√	√	√	√	√	√

Arthropoda	Insecta	Trichoptera	Hydropsychidae	Parapsyche	elsis		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Brachycentridae	Micrasema		√	√	√	√		√	√
Arthropoda	Insecta	Trichoptera	Brachycentridae	Brachycentrus	americanus	√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Helicopsychidae	Helicopsyche		√						
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila		√	√	√	√	√	√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Angelita Gr.			√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	arnaudi		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	atrata complex					√		
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Betteni Gr.		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Brunnea/ Vemna Gr.		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	coloradensis						√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Hyalinata Gr.		√	√			√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	narvae		√	√				
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Nevadensis Gr.			√				
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	grandis		√					
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Vagrita Gr.		√	√			√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	valuma		√	√	√			
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	vetina complex					√		
Arthropoda	Insecta	Trichoptera	Sericostomatidae	Gumaga					√			
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Agraylea		√						
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Hydroptila		√	√	√		√	√	√
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Metrichia		√						

Arthropoda	Insecta	Trichoptera	Hydroptilidae	Ochrotrichia		√	√			√		√
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Stactobiella				√	√			
Arthropoda	Insecta	Trichoptera	Lepidostomatidae	Lepidostoma			√			√	√	
Arthropoda	Insecta	Trichoptera	Philopotamidae	Wormaldia		√			√	√		
Arthropoda	Insecta	Trichoptera	Philopotamidae	Dolophilodes		√	√	√				
Arthropoda	Insecta	Trichoptera	Limnephilidae			√		√	√			
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecus		√						√
Arthropoda	Insecta	Trichoptera	Limnephilidae	Onocosmoecus		√						
Arthropoda	Insecta	Trichoptera	Limnephilidae	Psychoglypha		√			√	√		
Mollusca	Gastropoda	Basommatophora	Ancylidae	Ferrissia		√					√	√
Mollusca	Gastropoda	Basommatophora	Physidae	Physa		√			√		√	
Mollusca	Gastropoda	Neotaenioglossa	Pleuroceridae	Juga			√		√	√	√	
Mollusca	Gastropoda	Basommatophora	Planorbidae			√			√			
Mollusca	Bivalvia	Pelecypoda	Pisidiidae							√	√	√