Assessing responses of Bonneville cutthroat trout to restoration strategies implemented under the 1997 Conservation Agreement

Scott W. Miller[1], Nira L. Salant[1], Phaedra E. Budy[1,2], John (Jack) C. Schmidt[1]

[1]Intermountain Center for River Rehabilitation and Restoration
Department of Watershed Sciences
Utah State University

[2]USGS Utah Cooperative Fish and Wildlife Research Unit
Department of watershed Sciences
Utah State University

February 2010

Endangered Species Mitigation Fund Project Number: 0809
Grant Period: July 2008 – September 2009

Suggested citation:

Miller, S.W., N.L. Salant, P. Budy, J.C. Schmidt. 2010. Assessing responses of Bonneville cutthroat trout to restoration strategies implemented under the 1997 Conservation Agreement. 2009 Annual Report. Endangered Species Mitigation Fund Project Number: 0809. ICRRR 2010(1):1-122.

Table of Contents

List of Tables

List of Figures

Executive Summary

Physical habitat restoration is one of the most common river restoration practices; however, the underlying assumption that restored habitat benefits imperiled fishes is rarely supported by empirical evidence. With habitat restoration being one of the primary conservation actions to restore Bonneville cutthroat trout populations, confirming benefits of proposed actions and causes of past failures is critical to successful implementation of the 1997 Conservation Agreement. Working towards this goal, we evaluated the effectiveness of riparian grazing exclosures (Task 1) and active instream restoration (Task 2) to restore Bonneville cutthroat trout (BCT) in the Bear River and Northern Bonneville geographic units, as well as the critical methodological and ecological variables facilitating BCT recovery. Secondly, we used our own monitoring efforts on the Strawberry River as a case study to propose a set of monitoring protocols to evaluate BCT responses to active restoration (Task 3).

In Task 1, we assessed the effectiveness of riparian grazing exclosures to restore riparian vegetation, instream physical habitat, invertebrate prey resources, and subsequent BCT populations. Specifically, we sampled ten riparian exclosures spanning a gradient of grazing intensity, exlosure size, and construction date to quantify recovery trajectories and develop hypotheses regarding the conditions whereupon passive restoration through grazing exclosures has the greatest potential. We observed significantly improved riparian vegetative condition, geomorphic conditions, prey resource availability, and subsequent BCT populations within the ten studied grazing exclosures. Observed increases in vegetative cover (40%) and stubble height (240%), overhead cover (28%), undercut banks (30%), and channel narrowing (24% decrease) are not novel and have been consistently observed following grazing cessation. In contrast, consistent, positive BCT responses (density: 210% increase; biomass: 193% increase) have not been ubiquitously observed and we know of only two studies documenting increased inputs of terrestrial arthropods (136% increase) following grazing cessation. Our results suggest that on average the studied grazing exclosures effectively improved habitat conditions and prey resource availability, which facilitate higher local population densities and biomass. Further research is needed to determine if grazing exclosures increase growth and survival rates and thus facilitate a net population increase and/or the persistence of what would otherwise be a population in danger of local extirpation. Ideally, such research would be conducted in the context of determining the

size or spatial arrangement of habitat patches needed to increase growth and survival rates. Despite consistent, positive responses for measured abiotic and biotic variables, the magnitude of change exhibited considerable variability among systems. Variability in the magnitude of BCT responses was most strongly related to grazing regime and the degree of geomorphic and invertebrate prey resource recovery following grazing cessation; differences in BCT populations between grazed and ungrazed reaches were greatest under season-long grazing regimes, whereas BCT densities were more comparable within watersheds managed for short-duration, rotational grazing. These results suggest changes in grazing regimes at large spatial scales, and not necessarily complete grazing cessation, can be more effective at restoring BCT populations than the small-scale grazing exclosures studied herein. Lastly, we demonstrate that terrestrial invertebrate prey resources are a critical component of BCT summer diets and may play a critical role in facilitating the recovery of BCT populations post-grazing.

In Task 2, we investigated whether physical changes following instream restoration activities have ecologically relevant effects on BCT. We present the framework, methodology, and initial (one year post-restoration) results of an interdisciplinary, multi-metric monitoring program on the Strawberry River, a site of an ongoing instream restoration project typical of many in the Intermountain West. Our program aims to improve the effectiveness of restoration and adaptive management strategies by targeting biologically limiting conditions, monitoring at ecologically relevant scales, and identifying the link between restoration-induced physical change and biological response. In addition to establishing important baseline data for future monitoring, two years of habitat and fish population surveys produced five main findings: 1) beaver activity can cause measureable physical changes equal to or greater than those following restoration, including an increase in flow depth and pool frequency; 2) BCT habitat quality is limited by a lack of deep water, pools, and instream cover; 3) a reduction in one limiting variable (in this case, cover) results in a large decline in the estimated proportion of high quality habitat according to habitat suitability indices, 4) the estimated proportion of high quality habitat does not increase immediately after restoration (‘as-built’ conditions), and 5) in order to determine whether apparent increases in BCT populations on restored reaches are biologically significant, it is imperative to incorporate BCT life cycle dynamics, recruitment success, and stocking information (number and ages) into the analysis. Additional monitoring of this system includes historical aerial photo analysis, high precision geomorphic surveys, water quality and suspended sediment monitoring, bed sediment sampling, and redd surveys. Our continued and comprehensive study of this system will document how physical and biological components of the system adjust over the long-term.

In Task 3, we used the Strawberry River restoration project and monitoring program to demonstrate how standard geomorphic and ecological research techniques can be used to evaluate the outcome of BCT restoration projects. Pre- and post-project monitoring are essential components of any restoration strategy. Pre-project monitoring facilitates the design of effective, economical and efficient restoration activities by identifying the problems restoration should address, documenting natural system variability, and providing baseline data for future monitoring. Effective selection and application of restoration techniques requires identifying actual problems as distinct from natural conditions. Similarly, design of post-project monitoring requires a clear articulation of purpose of restoration, the problem being addressed, performance goals of the restoration technique, and direct outcomes of restoration. Using the Strawberry River restoration project as a template, we discuss a range of techniques for pre- and post-project monitoring of two commonly used restoration strategies: bank stabilization and morphological construction. We recommend this approach for restoration managers and practitioners who seek to inform adaptive management or additional restoration strategies.

Acknowledgements

We are thankful for the financial support for this research from the Utah Department of Natural Resources Endangered Species Mitigation Fund and the Utah State University Water Initiative. Christopher Keleher and Esther Biesinger helped with contracting and administrating funds. We received help from Brad Higginson, Paul Chase, Paul Cowley, James Capurso, Helen Kempenich, and Heidi Heyrend of the USFS; Pete Cavalli and Floyd Roadifer of the Wyoming Game and Fish Department; Paul Burnett and Alan Ward of the Utah Department of Natural Resources; and Tyler Staggs of the BLM with the identification of riparian grazing exclosures and grazing regimes. On the Strawberry River, Richard Heppworth, Justin Robinson, and Alan Ward of UDWR were instrumental in getting us started, providing logistical support, and providing housing after long days of cold snorkeling. None of this work would have been possible without the advice, technical assistance, and sharp wit of Gary Thiede. Marc-Coles Ritchie and Mike Scott of the USGS were great sounding boards for the design of a riparian vegetation sampling protocol. Field work could not have been completed without the hard work of Reed Chaston, Dave Fowler, Katrina Langenderfer, Ben Marett, Paul Nicholson, Greg Parry, Magen Schifiliti, Ellen Wakeley, and Hilary Whitcomb. Marhshall Baillie and Paul Mason managed field crews and deserve special thanks. Sunni Farb, Ben Marett, Paul Mason, Magen Schifiliti, Katie Swoboda, and Jared Watts assisted with the processing of invertebrate samples and fish stomachs.

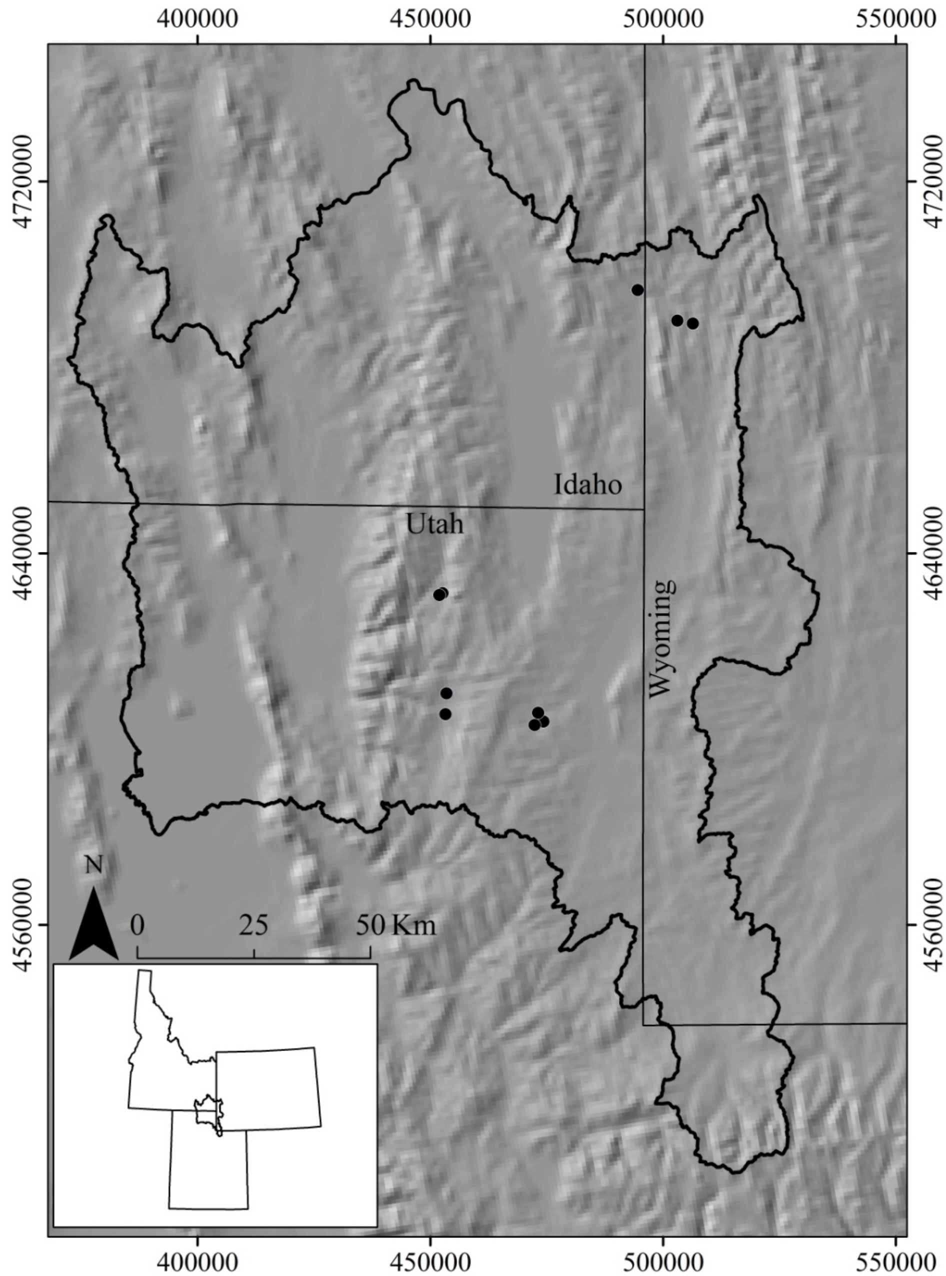
Task 1: Differential recovery trajectories of a native fish following grazing cessation: Evaluating the efficacy of riparian grazing exclosures to restore Bonneville cutthroat trout populations.

Introduction

In the western United States (U.S.), livestock grazing is one of the most pervasive causes of riparian and instream habitat degradation (Kauffman et al.1997); over 80% of riparian areas are adversely affected (Belsky et al. 1999). Livestock grazing can alter both structural and functional components of riparian and stream ecosystems (Bestcha 1997; Saunders and Fausch 2007). For example, streams and their adjacent riparian areas are intricately linked through the transfer of organic and inorganic materials in the form of nutrients, large woody debris, terrestrial arthropods, and sediment (reviewed in Gregory et al. 1991 and Baxter et al. 2005). The removal of riparian vegetation can decrease vegetative inputs to streams, thereby increasing thermal, sediment, and nutrient loadings (Platts 1991; Bestcha 1997; Belsky et al. 1999). These and other direct and indirect effects of cattle grazing have been implicated in the decline of western cold-water fishes.

Within the Bear River watershed (Fig. 1), habitat degradation, invasive species, and disease have decimated native Bonneville cutthroat trout (BCT) populations, prompting its listing as a Tier 1 sensitive species. In efforts to preclude further listing under the Endangered Species Act, BCT are protected by a multi-agency Conservation Agreement (Lentsch 1997). A primary focus of this agreement is to restore degraded habitat caused by intensive livestock grazing, one of the leading threats to BCT populations (Lentsch 1997; USFWS 2001). Consequently, riparian livestock exclosures have been widely implemented on public lands to restore degraded riparian and instream habitat and facilitate the coexistence of grazing and BCT.

Figure 1: Location of ten studied grazing exclosures within the Bear River Watershed; Utah, Idaho, and Wyoming. UTM zone 12 coordinates are provided for additional geographic reference.



Despite widespread implementation, few studies have assessed the effectiveness of riparian grazing exclosures to restore BCT populations. Results from the few studies conducted to date mirror the larger body of studies evaluating salmonid responses to grazing cessation by producing equivocal or conflicting results. For example, Binns and Remmick (1994) observed significant BCT responses following the construction of grazing exclosures on Huff Creek (WY), while Platts and Nelson (1985) failed to detect significant BCT responses to the Big Creek (UT) exclosure. Understanding the factors contributing to differential responses among systems remains one of the fundamental areas of exclosure research (Sarr 2002) and is critical to the future implementation of successful passive restoration efforts. For example, Saunders and Fausch (2007) recently identified links between alterations to riparian vegetation by livestock grazing, inputs of terrestrial arthropods, and salmonid biomass. Coupled with studies stressing the importance of terrestrial prey subsidies to stream food webs (reviewed in Baxter et al. 2005), these results suggest that physical habitat, the overwhelming target of fencing efforts, might not be the only reach-scale factor facilitating BCT recovery following grazing cessation.

We quantified vegetative, geomorphic, prey resource, and BCT responses to grazing exclosures. Our sampling spanned a gradient of grazing regimes, exclosure sizes, and construction dates in an effort to quantify recovery trajectories and develop hypotheses regarding the conditions whereupon passive restoration through grazing exclosures has the greatest effect. Specifically, we asked:

1. Do grazing exclosures improve riparian vegetative conditions and thus improve instream physical habitat and prey resources for BCT?
2. Do fish assemblages and BCT diets differ between grazed and ungrazed reaches?
3. How do grazing regime, exclosure size, and age influence results observed in questions 1 and 2?
4. What reach-scale factors (e.g., habitat versus prey resources) are related to differences in BCT populations between grazed and ungrazed reaches?

Methods

Sampling design

An intensive survey of land management agencies within the Bear River Watershed identified the large majority of 2nd and 3rd order stream reaches containing BCT, a maintained exclosure, and active grazing outside the exclosure. We subsequently screened sites to minimize geomorphic (e.g., valley slope, confinement, tributary junctions) differences between paired grazed (i.e., outside exclosure) and ungrazed (i.e., within exlcosure) reaches on the same system, such that in the absence of grazing one would expect the two reaches to be geomorphically similar. From this population of sites, we attempted to obtain a stratified random sample of exclosures by grazing regime (e.g., season-long versus rotational grazing), size, and date of construction; however, noncompareable geomorphic conditions, access constraints, and unmaintained exclosures inhibited our efforts. Subsequently, ten paired grazed and ungrazed reaches were systematically selected to represent a range of exclosure ages, sizes, and grazing regimes (Fig. 1; Table 1). We sampled the ten paired sites once during the summers of 2008 or 2009; all sampling occurred between July 15th and August 20th to coincide with peak aquatic and terrestrial invertebrate biomass and prior to the senescing of riparian vegetation; sites were generally sampled from lower to higher elevations to optimize the aforementioned variables.

The ten exclosures are found between 1790 and 2037 m in elevation and are located in relatively small watersheds ranging from approximately 14 to 75 km^2 (Table 1). All watersheds are dominated by a snowpack hydrologic regime with maximum flows occurring from April to June and baseflow predominant through the summer and winter months. Climate within the region is cold continental, characterized by hot, dry summers and cold winters. Riparian vegetative assemblages were relatively homogenous among sites with grass, sedge, rush, and willow communities dominating in minimally impacted riparian zones, which transitioned to upland grasses, forbs, and tree communities (e.g., sagebrush, rabbitbrush, juniper) laterally from the stream channel; cottonwood or other riparian gallery forests were largely absent.

Table 1: Site characteristics for the tem sampled exclosures in 2008 or 2009. Grazing regime outside the exclosure is indicated by season-long grazing (SLG) and short-duration rotational grazing (SRG). Brown trout (BNT), Bonneville cutthroat trout (BCT) and brook trout (BKT) are listed in order of abundance within exclosures.

	Land ownership	Grazing regime[a]	Date constructed	Exclosure Length (m)	Exclosure Area (km^2)	Elevation (m)	Watershed area (km^2)	Bankfull width (m)[b]	Trout species
Huff Creek	BLM	SRG	1978	2092	0.142	2005	25.62	1.7	BCT
Spawn Creek[c]	USFS	SRG	2005	3000	0.825	1838	13.69	2.4	BCT, BNT
Temple Fork	USFS	SRG	1970	110	0.002	1800	41.42	5.5	BCT, BNT
Dry Creek	USFS	SRG	1990	525	0.045	1948	23.9	2.6	BCT
Big Creek	BLM	SLG	1970	600	0.042	2013	75.1	3.1	BCT, BKT, BNT
Big Creek 2	BLM	SLG	1983	950	0.086	2036	67.8	2.6	BKT, BCT, BNT
Rock Creek[c]	State	SLG	1995	450	0.038	1790	60.9	3	BNT, BCT
Rock Creek 2[c]	State	SLG	1995	1345	0.242	1795	57.8	3.3	BNT, BCT
Randolph Creek	BLM	SLG	1983	346	0.028	2037	23.7	2.1	BNT, BKT, BCT[d]
Little Muddy Creek[c]	BLM	SLG	1979	1207	0.135	2012	25.7	1.6	BCT[d]

[a]Grazing regime determined through discussions with state and federal range specialists and needs to be verified with stocking records

[b]Bnakfull width is the average of five measurments taken within the exclosure

[c]Grazed reach located downstream from exclosure

[d]BCT were not present in sufficent numbers to permit population or diet analyses

Exclosures were generally located in low gradient, alluvial valleys where cattle have easy access to the stream. Grazing regimes for the rangelands surrounding the exclosures where preliminarily categorized as either season-long (SLG: ~60 or more days of use per year) or short-duration, rotational grazing (SRG: ~40 or less days of use per year and timing varies annually), but need to be verified by grazing records. Sampled exclosures spanned a gradient from small, reach-scale exclosures (0.002 – 0.14 km^2) to small watershed-scale exclosures and ranged in construction date from 1970 to 2005.

We sampled riparian vegetative assemblages, instream physical characteristics, fish composition, and prey availability and utilization for each of the ten paired grazed and ungrazed reaches. When possible, we located grazed reaches upstream of the exclosure. Regardless of location, we separated grazed and ungrazed reaches by a minimum of 1 km, which is greater than the average home range of BCT during the summer months (Schrank and Rahel 2004; Hilderbrand and Kershner 2000a), to maximize the independence of BCT populations between reaches. Reach lengths were determined as a function of average bankfull width within exclosures, with sample reaches equal to 20 times bankfull width (Table 1); ungrazed reaches were centered within the exclosure when possible; no sampling occurred within 50 m of the top or bottom fence boundary, as cattle are known to concentrate in these areas.

Riparian vegetation

We quantified riparian vegetative composition, height, and cover using the line-intercept method (Canfield 1991). Measurements were made along five transects per reach; transects were oriented perpendicular to the thalweg and spanned the entire width of the active riparian zone. The first transect was randomly located and the four subsequent transects systematically spaced at intervals four times bankfull width. Vegetative composition and cover were quantified along the entire transect at the genus level, while grass, forb, and shrub height was measured at 25 random locations. When present, tree composition, height, and basal area were quantified using 8 m wide belt transects. For this report, total percent cover and residual stubble height (i.e., height of grasses, forbs, shrubs etc.) are the only response variables evaluated.

Physical characteristics

Instream physical variables relevant to salmonids were measured at both the reach and point scale. At the reach scale, we continuously measured the proportion of stream bank containing undercut banks (>5 cm deep and >10 cm long), overhanging vegetation (within 1 m of water surface and >0.5 m overhang), and the length, minimum and maximum depth, and average wetted width of all habitat unit (e.g., pools, riffles, runs, etc. defined according to Hawkins et al. 1993); tailout depth was also measured for each pool. We computed reach-scale habitat diversity using Simpson's Diversity index, which was comprised of the number of unique habitat units and their relative abundance by length. For each pair of sites, the proportional abundance of habitat units was also compared between ungrazed and grazed reaches with a chi-square test.

Point-scale measurements were taken at ten cross-sections within each reach. The first cross-section was randomly located and the nine subsequent cross-sections systematically spaced at intervals two times bankfull width. Along each cross-section we measured velocity and depth at ten equidistant points across the stream channel using a Marsh McBirney Flo-Mate, the intermediate axes of 10 randomly selected particles, and percent overhead cover measured with a densiomenter held 30 cm above the water surface at left and right bank and stream center. We also surveyed channel cross-sections at each of the five vegetation transects; however, these data have yet to be analyzed.

Stream temperature was measured from approximately July 1st until September 30th of 2008 or 2009 at 60 minute intervals within each reach using Hobo data loggers deployed at the downstream end of each reach. We computed three salmonid-relevant temperature statistics: seven-day average of the maximum daily temperature ($temperature_{7\text{-}day}$), the number of days the weekly maximum temperature exceeded 18^{o}C ($temperature_{18^{o}C}$), and the average of all hourly temperature readings ($temperature_{avg}$). Three replicate in situ turbidity measurements were also obtained from each reach.

Fish prey resource availability

We quantified benthic, aerial aquatic, and terrestrial invertebrate composition and biomass to assess fish prey resource availability. Prior to fish electroshocking, benthic macroinvertebrates were sampled with a Surber sampler (0.09 m^2, 500 µm mesh) at eight random locations within riffle habitats. The eight samples from each reach were composited, preserved in 90% ethanol, and processed using a 600 count subsampling procedure (Caton 1991; Vinson and Hawkins 1996). When possible, we identified macroinvertebrates to genus (Merritt and Cummins 1996). Chironomidae midges, however, were identified to subfamily, and all non-insect taxa were identified to either order or family (Thorp and Covich 1991). Once identified and enumerated, we measured wet biomass at the order level by blotting the sample dry for 60 s and weighing them to the nearest tenth of a gram.

We quantified terrestrial and aerial aquatic invertebrate inputs to each reach using five pan traps (58.4 x 42.4 cm; total area: 1.25 m^2) located at each of the vegetation transects; left or right bank placement was randomly determined. Pan traps were located immediately adjacent to the stream bank, filled with 4 - 8 cm of water, to which a biodegradable surfactant was added to minimize surface water tension and maximize invertebrate retention, and deployed for 48 hours. Collected materials were sieved (500 µm), preserved in 90% ethanol, and processed using a 300 count subsampling procedure in addition to a big and rare search. When possible, we identified insects to family, the general taxonomic level needed to determine their origin, aquatic or terrestrial, of most taxa and non-insects to order or family. However, given the diverse life-histories of families within orders such as Diptera, Coleoptera, and Hemiptera, we adopted a conservative approach to determining origin and classified all individuals who have at least one aquatic life stage as aquatic. Blotted wet biomass was determined for all individuals at the order level. For analyses, all prey availability data was summarized as total biomass of benthic, aerial aquatic, or terrestrial organisms per reach.

Fish density, biomass and diets

We quantified differences in fish density, biomass, condition, and composition between paired reaches using a three-pass depletion technique with a single backpack electroshocker. The top and bottom of each reach were blocked with 10 mm seines to ensure a closed population.

Captured fish were anesthetized using tricaine methanesulfonate (MS 222), identified to species when possible, measured to the nearest millimeter (total length), and weighed to the nearest tenth of a gram. Population estimates were obtained using the least squares regression approach for all salmonids, as well as non-game species (e.g., mountain sucker, mountain whitefish) when present. Sculpin sampling effort and efficiency was not equal within and among reaches; therefore they are excluded from all analyses. Abundance data are reported as the number of fish per square meter of sampled stream. In addition, we also computed fish biomass per hectare by multiplying abundance estimates by median fish weights and total area of the sampled stream. We compared fish condition between paired reaches using slope and y-intercept values computed from length-weight regressions. Lastly, BCT population age structure between ungrazed and grazed reaches was compared by grouping BCT into two groups (age 1: 100-149 mm; age 3+: >225) and comparing the proportion of adult and juvenile fish between reaches. The homogeneity of size frequency distributions were also compared between paired grazed and ungrazed reaches using a Kolmogorov-Smirnov test.

We compared BCT dietary habits (prey composition, prey selection, and primary carbon source) between grazed and ungrazed reaches using two complimentary approaches: stomach content and stable isotope analysis. Stomach contents provided a short-term (< 1 week) snapshot of BCT dietary habits, while stable isotopes quantified dietary preferences over a longer timeframe (~2 months). When present, dietary habits were assessed for ten randomly selected juvenile and (<150 mm) and adult (>150 mm) BCT per reach. However, juveniles were not present in sufficient numbers to permit dietary comparisons among systems; therefore, all diet analyses focus exclusively on adults. We chose 150 mm as the size class where ontogeneteic feeding shifts are typically observed, as opposed to 225 mm which was selected to differentiate reproductively active adults (Budy et al. 2007). For the purposes of this report, our analyses focus only on the origin (aquatic, aquatic aerial, terrestrial) and utilization of dominant prey sources between ungrazed and grazed reaches and not on preferences for individual organisms within these categories.

We collected 5 mm dorsal muscle plugs for stable isotope analysis. Muscle tissue was preserved in ethanol, dried for 48 h at 60^{o}C, ground to a powder, encapsulated in tin capsules, and processes by the University of California-Davis Stable Isotope Facility for a mass-spectrometry analysis of isotopic signatures (^{13}C:^{12}C and ^{15}N:^{14}N). Isotopic ratios are expressed as δ13C and δ15N per mil (‰) relative to the ratios of the standards Pee Dee Belemnite and atmospheric nitrogen, respectively. Our main focus was on differences in the dietary carbon source (δ13C) between grazed and ungrazed reaches and not BCT trophic position (δ15N).

We obtained stomach contents through gastric lavage. Stomach contents were passed through a 500 µm sieve, preserved in ethanol, enumerated at the order level, blotted dry, and weighed. Unidentifiable organic matter was classified as vegetation or other organic matter. Individual fish whose stomach contents contained greater than 25% unidentifiable contents by mass were excluded from analyses. For analysis, we categorized diet composition as benthic aquatic, aerial aquatic, or terrestrial organisms and made comparisons between grazed and ungrazed reaches using proportional composition by weight and mass per mass of fish.

Using estimates of prey standing stocks (g/m^2), we quantified prey selection behavior for the three prey categories (benthic aquatic, aerial aquatic, and terrestrial) using Chesson's alpha (W):

$$W = \frac{\left(\frac{r_i}{p_i}\right)}{\sum_{i=1}^{n}\left(\frac{r_i}{p_i}\right)}$$

Where r_i is the proportion of a given prey category in the diet, p_i is the proportion of a given prey category in the environment, and n is equal to the total number of prey categories in the environment. W ranges from 0 – 1, with values above 1/n (0.33 for our application) indicating a preferred food item and values below 1/n indicating an avoidance of that prey category.

Analyses

We used a nonparametric Wilcoxon signed rank procedure to test for vegetative, physical habitat, prey resource, fish population, and fish diet differences between the ten paired grazed and ungrazed reaches ($\alpha = 0.05$). Nonparametric procedures were chosen because response variables were not normally distributed and transformations did not alleviate departures from normality. Average percent change between paired grazed and ungrazed reaches (n = 10) was used as the response variable, which was calculated as:

$$\text{Average percent change} = \left(\frac{Ungrazed - Grazed}{Grazed}\right) \times 100$$

We used percent change as a response variable because we were primarily interested in the relative effect of grazing exclosures and not absolute differences between the population of sampled ungrazed and grazed reaches. Furthermore, comparisons of absolute values among systems are confounded by natural differences in the abiotic and biotic potentials of individual systems unrelated to the degree of anthropogenic alteration. Response variables originally expressed as a percentage (e.g., percent cover, percent diet composed of terrestrial organisms) were analyzed by computing the difference between ungrazed and grazed reaches. Though we recognize that the probability of finding significant results by chance increases as more tests are conducted, we did not adjust alpha levels ($\alpha = 0.10$) using Bonferroni procedures because they are overly conservative and therefore increase the chance of ignoring ecologically meaningful results (Moran 2003).

We used Kendall's tau correlation coefficient to quantify how vegetative, habitat, prey resource, and BCT responses (i.e., percent change between paired reaches) varied as a function of grazing regime, exclosure size, and age; only variables exhibiting significant differences were evaluated. Furthermore, Kendall's tau was also used to identify correlations between BCT responses (density and biomass) and habitat and prey resource differences between ungrazed and grazed reaches. We used Kendall's tau because of expected nonlinearities in the above relationships and the ordinal nature of grazing regime (low: rotational; high: season-long). Correlations were computed from the percent change between ungrazed and grazed reaches and were only assessed for variables exhibiting significant differences. Lastly, we averaged Kendall's tau values across all 11 variables to assess the cumulative effects for each of grazing regime, age, and size on observed responses.

Results

Vegetative cover

On average, we found ungrazed (within exclosures) riparian areas to have greater vegetative cover (40% greater; $W_+ = 53.0$, $P = 0.011$, $df = 10$) and residual stubble height (240% greater; $W_+ = 54.0$, $P = 0.008$, $df = 10$) (Fig. 2) than grazed reaches. Conversely, percent bare ground was 6% greater in ungrazed versus grazed reaches ($W_+ = 4.0$, $P = 0.019$, $df = 10$). Increased ground cover and height of riparian vegetation within grazing exclosures translated to a 28% increase in vegetative cover overhanging the stream channel ($W_+ = 51.0$, $P = 0.019$, $df = 10$).

Physical variables

Variables related to channel geometry and the proportional area of habitat units significantly differed between ungrazed and grazed reaches, while substrate, temperature, and water clarity were relatively homogeneous (Tables 2 & 3). On average, width-to-depth ratios for grazed reaches were 24% greater ($W_+ = 18.0$, $P = 0.006$, $df = 10$), with grazed reaches generally having wider widths and shallower depths. The narrower and deeper channels found within exclosures were characterized by a greater proportion of undercut stream banks ($W_+ = 52.5$, $P = 0.013$, $df = 10$) and greater residual pool depths ($W_+ = 50.0$, $P = 0.025$, $df = 10$), 30% and 27% respectively. Habitat diversity was also significantly greater (27%) ($W_+ = 55.0$, $P = 0.006$, $df = 10$), with ungrazed reaches having a more even proportion of habitat units ($X^2 = 11.8$, $P = 0.008$, $df = 3$); grazed reaches had more than half of the linear area in riffles (Table 3).

Figure 2: Average percent riparian vegetative cover (A), percent bare ground (B), residual stubble height (C), and percent overhanging vegetation (D) compared between ungrazed and grazed reaches for each of the 10 study systems. Also presented is the average percent change (± 95% confidence interval) for each variable between ungrazed and grazed reaches; asterisks indicate statistical significance at the 0.05 α level.

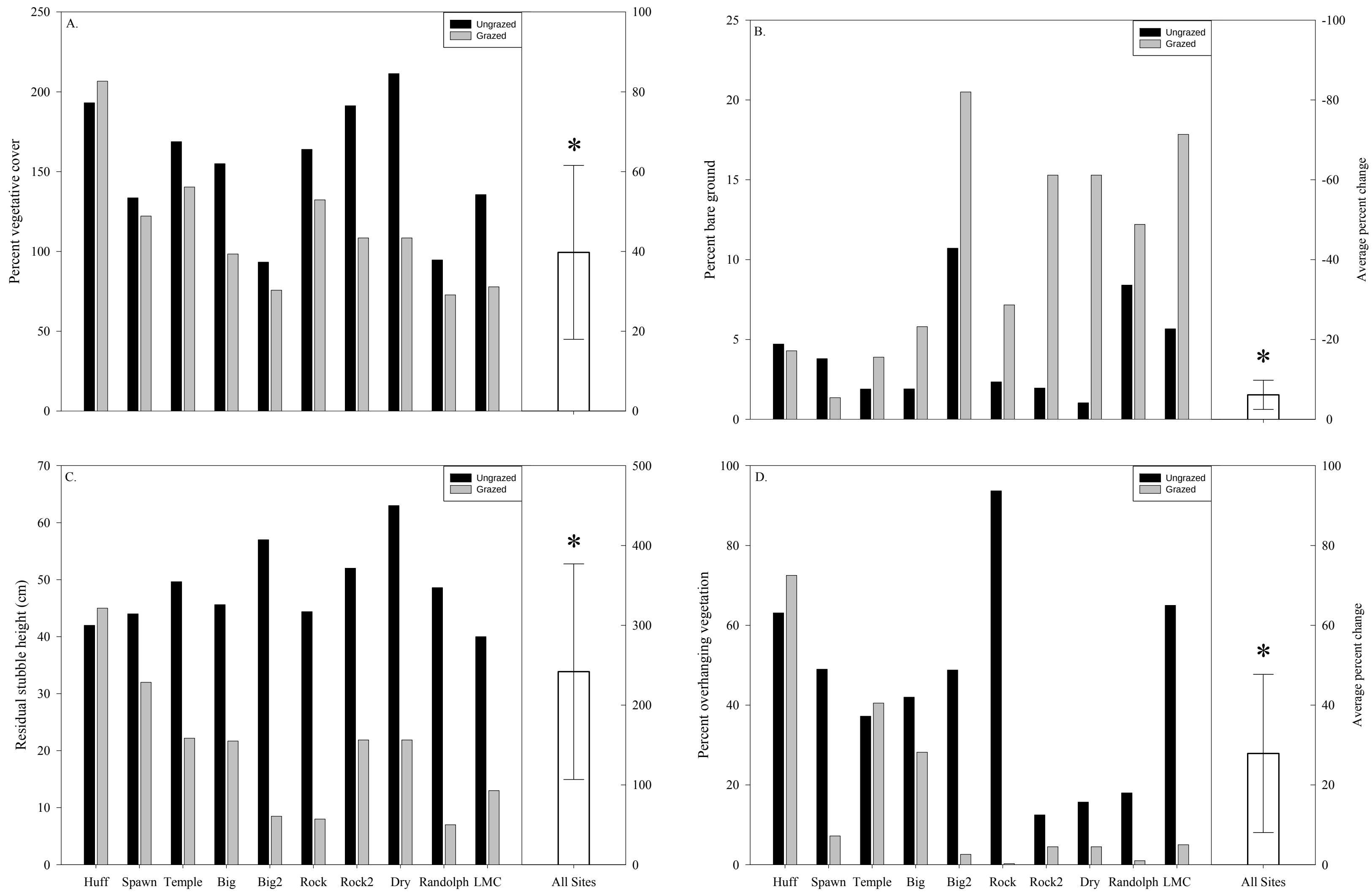


Table 2: Results of Wilcoxon signed-rank test comparing average percent change (± 95% confidence interval) in measured environmental variables between ungrazed and grazed reaches; variables exhibiting significant differences are bolded. Also displayed are average values for each system.

		D16 (mm)	D50 (mm)	Width:Depth (m)	% Undercut banks	Residual Pool Depth(m)	Turbidity (NTU)	Temperature-$_{7\text{-day}}$ (^oC)	Temperature-$_{avg.}$ (^oC)
All Sites	Ungrazed	9.5	29.8	11.1	29.9	2.6	4.1	18.4	11.8
	Grazed	11.7	28.6	14.3	14.5	2.0	4.5	18.5	11.9
	Avg. % change[a]	-12.2 (13.8)	15.7 (25.9)	**-17.5 (14.5)**	**15.5 (7.8)**	**30.7 (20.0)**	-0.41 (12.4)	-0.48 (1.9)	-0.82(1.5)
Individual Sites									
Huff	Ungrazed	7.6	42.6	2.2	16.6	2.1	6.5	18.0	10.5
	Grazed	7.3	29.4	4.2	15.6	1.7	7.5	18.2	10.4
Spawn	Ungrazed	19.5	49.2	16.4	35.0	1.6	0.7	15.8	10.1
	Grazed	19.5	47.1	16.7	17.4	1.2	0.7	16.1	10.5
Temple	Ungrazed	16.9	39.4	14.7	29.0	2.0	1.0	16.6	10.3
	Grazed	23.2	50.6	17.3	8.6	2.1	1.0	16.5	10.2
Dry	Ungrazed	6.7	35.1	10.4	27.2	3.5	6.9	15.9	12.4
	Grazed	6.5	18.0	15.3	11.1	2.4	5.3	14.9	12.1
Big	Ungrazed	6.8	19.3	7.2	38.6	3.3	4.2	21.9	9.5
	Grazed	7.2	21.0	10.6	8.7	2.3	5.5	22.3	9.8
Big2	Ungrazed	7.7	28.8	12.1	40.2	3.7	2.3	17.7	7.0
	Grazed	7.9	26.1	14.3	30.1	3.2	2.3	17.7	7.1
Rock	Ungrazed	7.0	29.3	14.8	14.4	2.3	1.5	20.0	16.1
	Grazed	16.0	36.4	18.5	15.8	2.2	1.5	20.1	16.2
Rock2	Ungrazed	9.0	24.4	11.9	17.2	3.4	1.8	19.1	15.6
	Grazed	16.0	36.4	18.5	15.8	2.2	1.5	20.1	16.2
Randolph	Ungrazed	6.6	14.7	15.6	37.4	2.0	6.8	22.1	13.4
	Grazed	6.7	12.0	21.4	12.9	0.0	10.8	22.0	13.2
LMC[b]	Ungrazed	7.6	15.0	5.5	43.6	2.6	9.6	17.0	13.4
	Grazed	7.0	9.0	6.4	8.6	2.8	8.7	17.5	13.7

[a]Average percent change values do not match those computed for reported averages because they were computed for each individual system, then averaged

[b]Little Muddy Creek is abbreviated as LMC

Table 3: Results of Wilcoxon signed-rank test comparing average percent change (± 95% confidence interval) of habitat unit proportional abundance and habitat diversity between ungrazed and grazed reaches; variables exhibiting significant differences are bolded. Also displayed are raw values for individual systems.

		Run	Riffle	Pool	Glide	Simpson's Diversity
All Sites	Ungrazed	11.8 (9.7)	33.9 (13.7)	32.3 (12.1)	17.1 (11.9)	0.60 (0.07)
	Grazed	7.4 (4.5)	58.9 (12.5)	26.0 (10.0)	8.5 (6.9)	0.50 (0.09)
	Avg. % Change[a]	4.46 (7.7)	**-25.0 (12.04)**	6.4 (7.8)	8.6 (11.8)	**27.1 (17.5)**
Individual Sites						
Huff	Ungrazed	0	31.3	49	19.7	0.62
	Grazed	0	54.1	19.4	28.9	0.61
Spawn	Ungrazed	0	83.6	16.3	2.7	0.31
	Grazed	0	89.6	10.4	0	0.19
Temple	Ungrazed	0	45.3	38.9	0	0.54
	Grazed	0	60.2	29.1	10.7	0.50
Dry	Ungrazed	20.9	19.6	13.3	46.2	0.69
	Grazed	21.3	48.5	31.9	0	0.63
Big	Ungrazed	7.9	22.9	64.6	4.6	0.59
	Grazed	8.7	39.4	49.3	2.6	0.52
Big2	Ungrazed	0	47.5	52.5	0	0.57
	Grazed	0	47.4	52.6	0	0.50
Rock	Ungrazed	25.6	36.9	31	6.5	0.70
	Grazed	11.3	62.3	19.1	7.3	0.56
Rock2	Ungrazed	47.7	28.8	16.8	7	0.66
	Grazed	11.3	62.3	19.1	7.3	0.56
Randolph	Ungrazed	2.5	23.5	4.2	35	0.54
	Grazed	8.8	94.4	0	0	0.28
LMC[b]	Ungrazed	13.5	0	37.2	49.2	0.73
	Grazed	12.1	30.7	29.3	27.9	0.60

[a]Percent change values do not always match those computed for reported averages because they were computed for each individual system, then averaged

[b]Little Muddy Creek is abbreviated as LMC

Fish: composition, abundance, biomass, condition, size structure

Eight out of the ten sampled systems contained BCT within the designated sample reaches. Additionally, we found three BCT on Randolph Creek during spot shocking, while only one BCT was found on Little Muddy Creek. For the eight systems containing BCT within sampled reaches, we found significantly greater BCT densities ($W_+ = 31.0$, $P = 0.080$, $df = 8$) and biomasses ($W_+ = 33.0$, $P = 0.042$, $df = 8$) within ungrazed reaches (Fig. 3), while BCT condition was similar between treatments (slope: $W_+ = 22.0$, $P = 0.205$, $df = 8$; intercept: $W_+ = 21.0$, $P = 0.272$, $df = 8$) (Table 4). Both density (200% higher for ungrazed reaches) and biomass (400% higher) exhibited relatively consistent directional responses, while the magnitude of change varied greatly among systems. BCT responses to grazing exclosures were consistent among juvenile and adult BCT; the proportion of juveniles and adults did not significantly differ between treatments ($W_+ = 11.0$, $P = 1.0$, $df = 8$). Furthermore, Dry Creek was the only site to exhibit differences in size frequency distributions between ungrazed and grazed reaches (Appendix 1, Fig. A1-1).

Bonneville cutthroat trout were the numerically dominant salmonid in 9 of the 20 sampled reaches (Appendix 1, Fig. A1-2). When present, non-native salmonid density and biomass were generally greater in ungrazed versus grazed reaches, whereas mountain suckers did not exhibit any apparent trends (Table 5). Furthermore, fish assemblages both inside and outside of the exclosures were largely dominated by native fishes, with no appreciable changes in the percentage of native fishes between ungrazed and grazed reaches (Appendix 1, Fig. A1-2).

Figure 3: BCT density and biomass estimates compared between ungrazed and grazed sites for the eight systems containing BCT; Randolph and Little Muddy creeks are excluded because BCT were absent. Also presented is the average percent change (± 95% confidence interval) for each variable between ungrazed and grazed reaches; asterisk indicates statistical significance at the 0.05 α level.

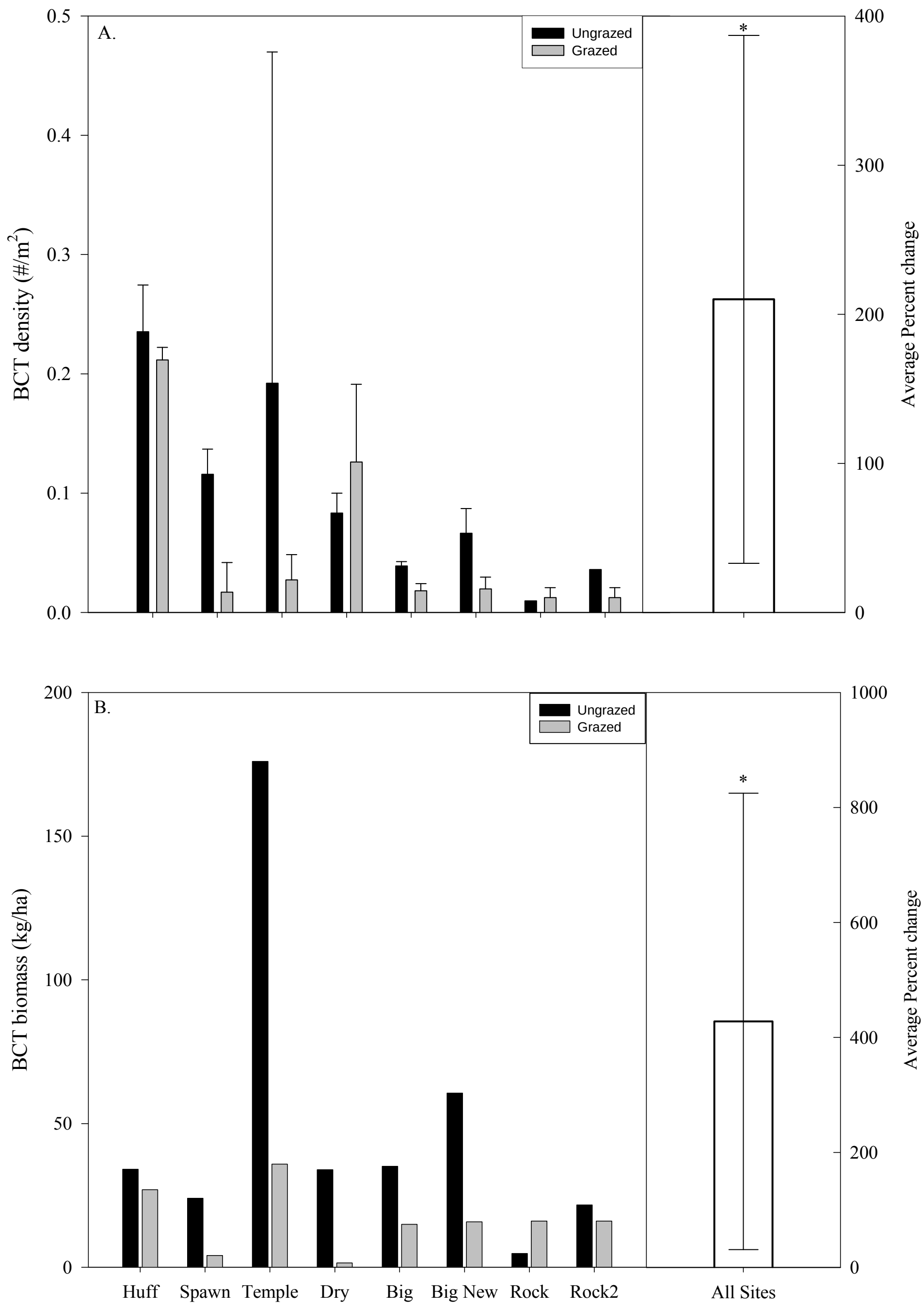


Table 4: Results of Wilcoxon sign-rank test comparing average percent change (± 95% confidence interval) for BCT condition and proportion of juveniles and adults between ungrazed and grazed reaches; variables exhibiting significant differences are bolded. Also displayed are raw values for individual study systems.

		Length-weight slope coefficient[a]	Length weight y-intercept[a]	Percent juvenile (<150 mm)	Percent Adults (>150 mm)
All Sites	Ungrazed	3.0	-5.0	36.7	63.3
	Grazed	2.9	-4.9	39.1	60.9
	Avg. % change[b]	3.9 (5.1)	3.9 (6.2)	-2.4 (16.9)	2.4 (16.9)
Individual sites					
Huff	Ungrazed	3.0	-5.1	78.4	21.6
	Grazed	2.8	-4.6	86.8	13.2
Spawn	Ungrazed	3.0	-5.1	63.2	36.8
	Grazed	3.0	-5.1	50.0	50.0
Temple	Ungrazed	2.8	-4.5	55.0	45.0
	Grazed	2.8	-4.7	25.0	75.0
Dry	Ungrazed	3.1	-5.2	30.0	70.0
	Grazed	3.0	-5.1	84.6	15.4
Big	Ungrazed	3.0	-5.0	0.0	100.0
	Grazed	2.7	-4.4	0.0	100.0
Big2	Ungrazed	2.9	-4.8	6.7	93.3
	Grazed	2.7	-4.5	0.0	100.0
Rock	Ungrazed	3.5	-5.9	33.3	66.7
	Grazed	3.1	-5.2	33.3	66.7
Rock2	Ungrazed	2.8	-4.6	27.3	72.7
	Grazed	3.1	-5.2	33.3	66.7
Randolph	Ungrazed	0.0	0.0	0.0	0.0
	Grazed	0.0	0.0	0.0	0.0
LMC[c]	Ungrazed	0.0	0.0	0.0	0.0
	Grazed	0.0	0.0	0.0	0.0

[a]Length-weight coefficients derived from logarithmic transformations of both variables

[b]Percent change values do not always match those computed for reported averages because they were computed for each individual system, then averaged

[c]Little Muddy Creek abbreviated as LMC

Prey resource availability

On average, total invertebrate prey resource availability was similar between ungrazed and grazed reaches (Fig. 4); however, the relative contributions of individual prey categories differed between treatments. Benthic biomass provided the largest prey resource among all sites, followed by biomass of aerial aquatics and terrestrial inputs. Among paired reaches, inputs of terrestrial arthropods to ungrazed reaches were 136% greater than those observed for grazed reaches ($W_+ = 50.0$, $P = 0.025$, $df = 10$). As with other abiotic and biotic response variables, the direction of change was relatively consistent among sites, while the magnitude of change was highly variable. In contrast, the biomass of aerial aquatics returning to the stream did not exhibit consistent differences between grazed and ungrazed reaches ($W_+ = 30.0$, $P = 0.838$, $df = 10$). Benthic organisms exhibited the opposite pattern as terrestrials; on average, standing stocks were 17% greater in grazed reaches ($W_+ = 3.0$, $P = 0.014$, $df = 10$).

Fish: diet

We sampled the stomachs and isotopic signatures of 136 adult BCT; juveniles (<150 mm) were not present in sufficient numbers to facilitate comparisons between ungrazed versus grazed reaches. Of the 136 stomachs examined, three were empty and four additional stomachs were removed from all analyses because they contained greater than 25% unidentifiable matter. To date, isotope samples have only been processed for five of the ten paired sites.

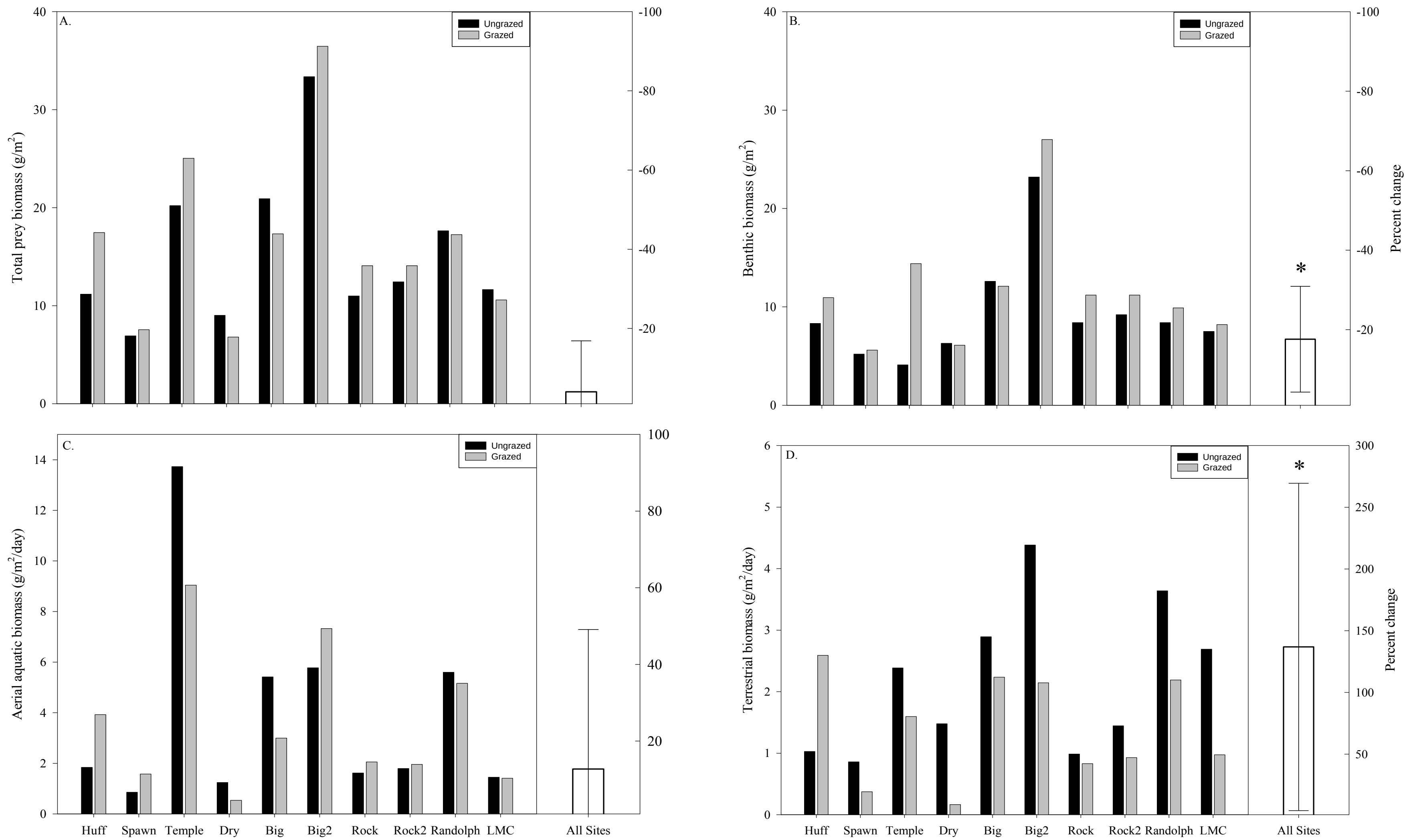
Table 5: Results of Wilcoxon sign-rank test comparing average percent change (± 95% confidence interval) for Bonneville cutthroat trout (BCT), brown trout (BNT), brook trout (BKT), and mountain suckers (MSU) densities between ungrazed and grazed reaches; variables exhibiting significant differences are bolded. Also displayed are densities (± 95% confidence interval) for individual study systems.

		Density (#/m^2)			
		BCT	BNT	BKT	MSU
All Sites	Ungrazed	0.08	0.06	0.02	0.14
	Grazed	0.04	0.03	0.01	0.14
	Avg. % change[a]	**210.1 (177)**	**66 (59)**	**156 (154)**	-0.25 (61.3)
Individual sites					
Huff	Ungrazed	0.24 (0.04)	0	0	0.01 (0)
	Grazed	0.21 (0.01)	0	0	0.01 (0)
Spawn	Ungrazed	0.12 (0.02)	0.11 (0.33)	0	0
	Grazed	0.02 (0.02)	0.10 (0.02)	0	0
Temple	Ungrazed	0.19 (0.28)	0.02 (0)	0	0
	Grazed	0.03 (0.02)	0.01 (0.01)	0	0
Dry	Ungrazed	0.08 (0.02)	0	0	0.03 (0.05)
	Grazed	0.13 (0.07)	0	0	0.03 (0)
Big	Ungrazed	0.04 (0.004)	0	0.03 (0.08)	0.08 (0.14)
	Grazed	0.02 (0.01)	0.003 (0)	0.027 (0.006)	0.10 (0.06)
Big2	Ungrazed	0.07 (0.02)	0.004 (0)	0.10 (0.008)	0.004 (0)
	Grazed	0.02 (0.01)	0.003 (0)	0.026 (0)	0.19 (0.13)
Rock	Ungrazed	0.01 (0.0)	0.07 (0)	0	0.58 (0.04)
	Grazed	0.01 (0.008)	0.04 (0.004)	0	0.45 (0.13)
Rock2	Ungrazed	0.04 (0.0)	0.08 (0.01)	0	0.62 (0.008)
	Grazed	0.01 (0.008)	0.04 (0.004)	0	0.45 (0.13)
Randolph	Ungrazed	0.0	0.35 (0.09)	0.04 (0.02)	0.013 (0)
	Grazed	0.0	0.13 (0.06)	0.01 (0)	0
LMC[b]	Ungrazed	0.0	0	0	0.02 (0.1)
	Grazed	0.0	0	0	0.12 (0.07)

[a]Percent change values do not always match those computed for reported averages because they were computed for each individual system, then averaged

[b]Little Muddy Creek abbreviated as LMC

Figure 4: Comparison of total prey resource biomass (A), benthic (B), aerial aquatic (C), and terrestrial biomass (D) between ungrazed and grazed reaches for the ten sampled exclosures. Also presented is the average percent change (± 95% confidence interval) for each variable between ungrazed and grazed reaches; asterisk indicates statistical significance at the 0.05 α level.



BCT stomach contents and δ13C values indicated a high degree of dietary overlap between ungrazed and grazed reaches. Terrestrial and benthic prey items dominated BCT diets in both reaches, while aerial aquatics comprised less than 2% of stomach contents by weight on average. A comparison of δ13C values indicated that the organic carbon source utilized by BCT did not significantly differ between grazed and ungrazed reaches for the five systems analyzed to date ($W_+ = 7.0$, $P = 1.0$, $df = 5$) (Fig. 5). Similarly, the proportional weight of the three prey categories within BCT diets exhibited only minor differences between ungrazed and grazed reaches (benthic: -8%, $W_+ = 6.0$, $P = 0.107$, $df = 8$; aerial aquatics: 4.7%, $W_+ = 36.0$, $P = 0.014$, $df = 8$; terrestrial: 4.6%, $W_+ = 28.0$, $P = 0.183$, $df = 8$) (Fig. 6). Although consumption of aerial aquatics was significantly higher within ungrazed reaches, percentage increases were nominal. Excluding Spawn Creek, the only site differing in the direction of benthic and terrestrial prey responses, greatly reduces variability and results in significant differences for the proportional use of benthic and terrestrial prey items.

Although similar from a compositional standpoint, the biomass of terrestrial ($W_+ = 31.0$, $P = 0.069$, $df = 8$) and aerial aquatic ($W_+ = 31.0$, $P = 0.069$, $df = 8$) prey items per gram of fish were significantly greater for ungrazed versus grazed reaches, 77% and 107% respectively (Fig. 7). Despite differences exceeding 25% between ungrazed and grazed reaches, variable responses among systems precluded significant differences for the biomass of benthic prey items ($W_+ = 16.0$, $P = 0.834$, $df = 8$) by mass.

Across all studied systems and treatments, BCT foraging behavior exhibited a strong preference for terrestrial insects (Fig. 8). In contrast, BCT showed weak selection for both benthic and aerial aquatics, with Chesson's alpha below 0.3 on average. The differential selection of terrestrials was slightly greater, although not significantly so, for grazed reaches because of reduced availability of terrestrial prey resources (Fig. 4D). On average, BCT within exclosures exhibited a stronger affinity for aerial aquatic insects than those in grazed reaches ($W_+ = 34.0$, $P = 0.030$, $df = 8$) (178%), although consumption was extremely low overall (Fig. 6B). In contrast, fish utilization of benthic ($W_+ = 27.0$, $P = 0.234$, $df = 8$) and terrestrial ($W_+ = 12.0$, $P = 0.441$, $df = 8$) prey resources was not significantly different between ungrazed and grazed reaches.

Figure 5: Comparison of δ13C values between ungrazed and grazed reaches; only the 5 systems having isotope samples processed to date are presented. Also presented is the average percent change (± 95% confidence interval) between ungrazed and grazed reaches; asterisk indicates statistical significance at the 0.05 α level.

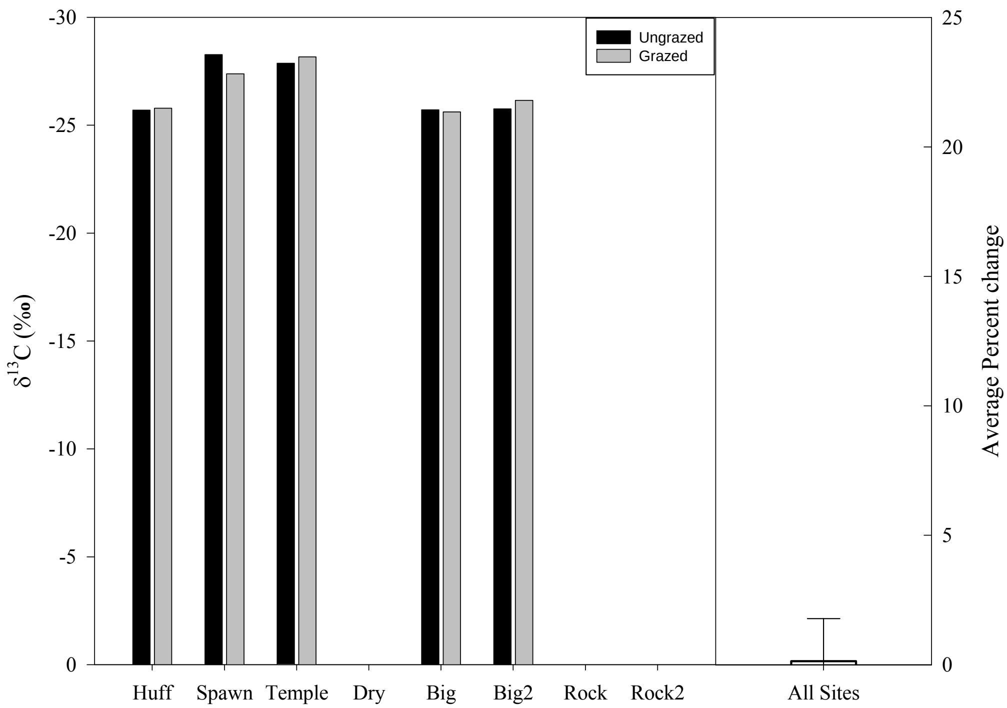


Figure 6: Comparison of proportional biomass of benthic (A), aerial aquatic (B), and terrestrial (C) prey items in BCT diets between ungrazed and grazed reaches. Also presented is the average percent change (± 95% confidence interval) for each variable between ungrazed and grazed reaches; asterisk indicates statistical significance at the 0.05 α level.

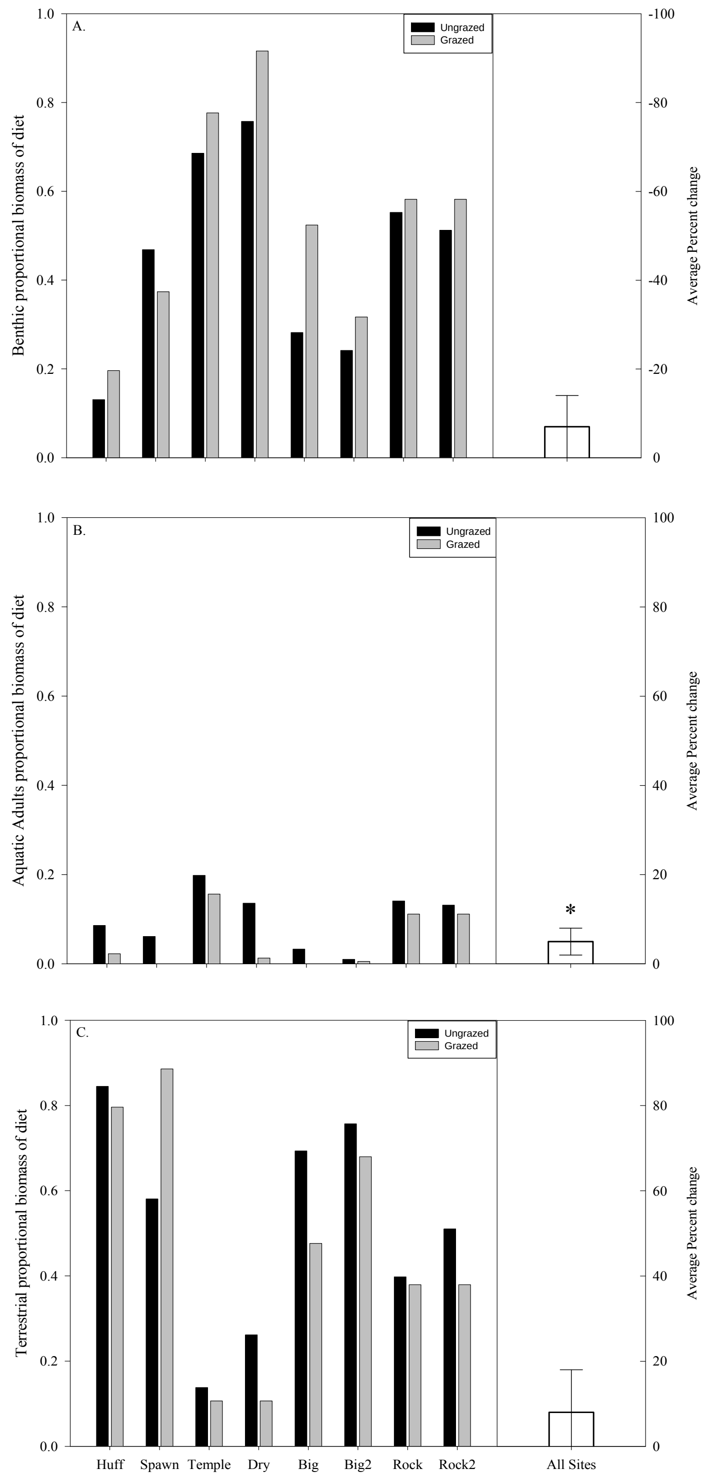


Figure 7: Comparison of benthic (A), aerial aquatic (B), and terrestrial (C) prey items in BCT diets between ungrazed and grazed reaches. Data presented as mg of prey item per mg of fish mass. Also presented is the average percent change (± 95% confidence interval) for each variable between ungrazed and grazed reaches; asterisk indicates statistical significance at the 0.05 α level.

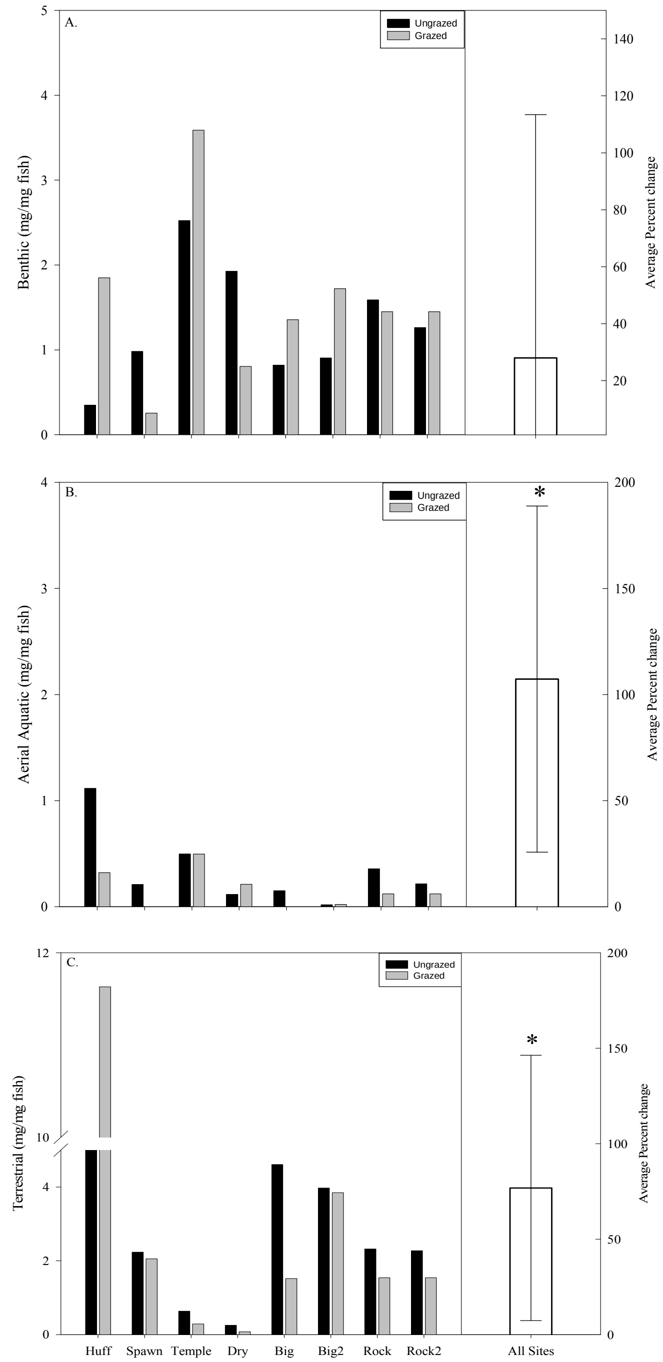
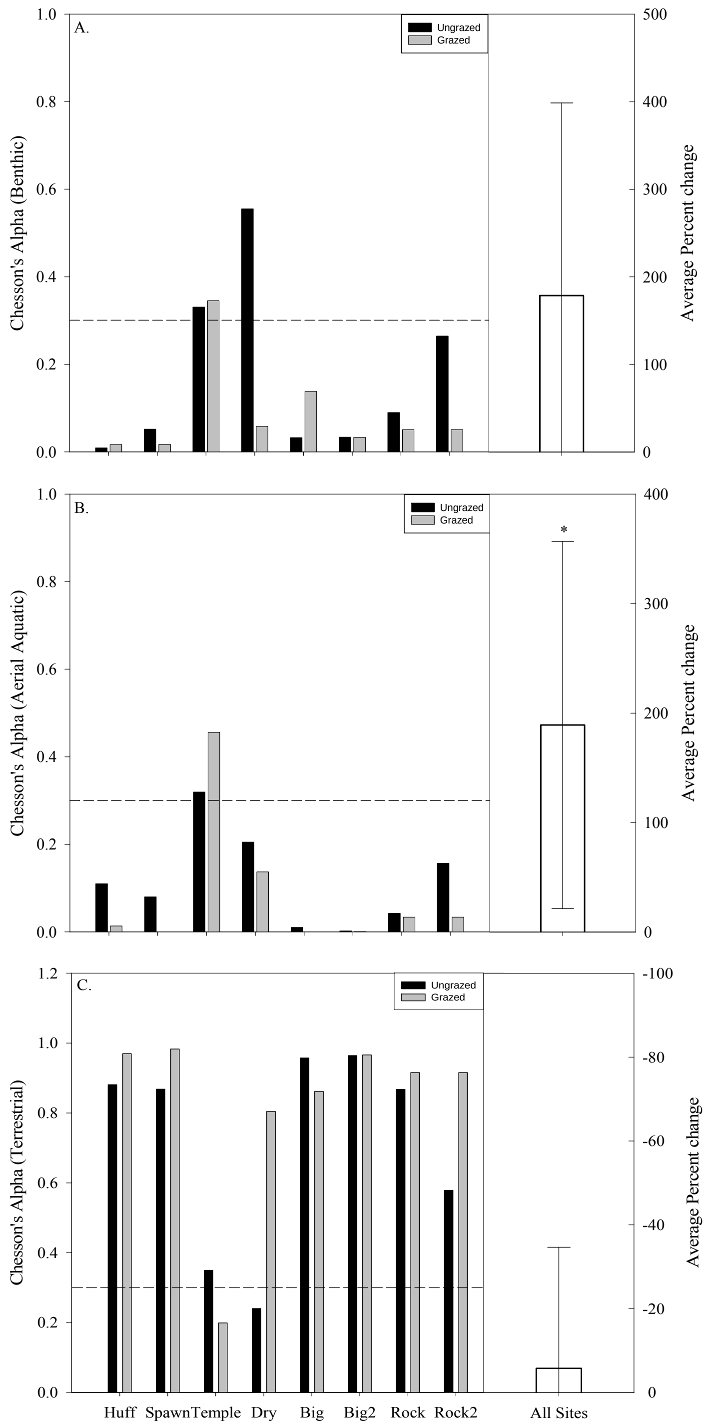


Figure 8: Comparison of Chesson's alpha values for benthic (A), aerial aquatic (B), and terrestrial (C) prey items by biomass in BCT diets between ungrazed and grazed reaches. Values above the dashed horizontal line indicate preference for a particular prey item. Also presented is the average percent change (± 95% confidence interval) for each variable between ungrazed and grazed reaches; asterisk indicates statistical significance at the 0.05 α level.



Influence of grazing regime, exclosure size, and age

Grazing regime exhibited the strongest and most consistent relationships with abiotic and biotic variables that were significantly different between ungrazed and grazed reaches (Table 6). In general, sites managed for season-long grazing were characterized by more disparate vegetative, geomorphic, and fish conditions between grazed and ungrazed reaches than those managed for short-duration rotational grazing. In contrast, variability in the magnitude of responses among systems was not consistently related to exclosure size or age.

Overall, variability in the magnitude of fish responses among systems was weakly correlated with responses of measured environmental variables (Table 6). Differences in BCT biomass and density were moderately correlated with changes in the proportion of undercut banks and overhead vegetation, while biomass and to a lesser extent density were also correlated with differences in the biomass of terrestrial arthropod inputs (i.e., paired sites with greater terrestrial arthropod inputs inside the exclosure also tended to have significantly greater BCT biomass; Fig. 9).

Discussion

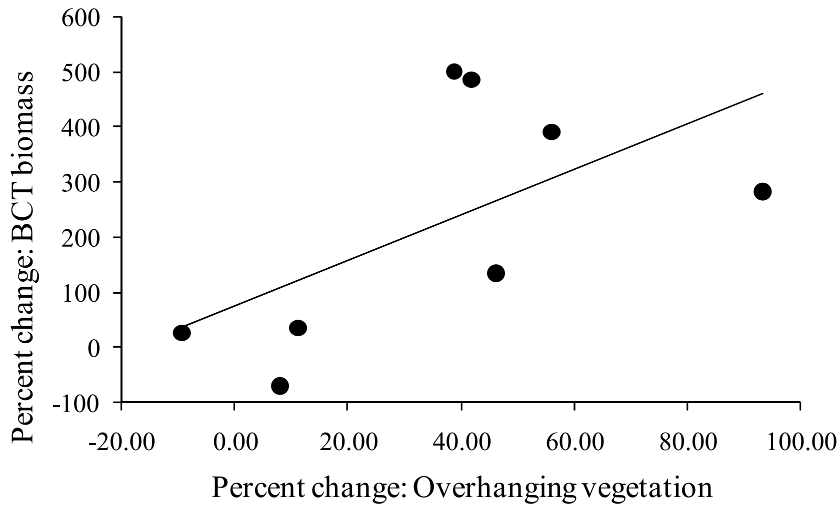
Grazing cessation within the ten studied exclosures resulted in significantly improved riparian vegetative condition, geomorphic conditions, prey resource availability, and subsequent BCT populations. Observed increases in vegetative cover and structure, bank stability, undercut banks, and channel narrowing are not novel and have been consistently observed following grazing cessation (reviewed in Sarr 2002). In contrast, consistent, positive fish responses have not been ubiquitously observed and we know of only two studies documenting increased inputs of terrestrial arthropods following grazing cessation. Despite consistent, positive responses among measured abiotic and biotic variables, the magnitude of change exhibited considerable variability among systems. In succeeding paragraphs, we address the significance of increased terrestrial prey resource availability, the nature of the fish assemblage response, factors related to fish recovery, and probable causes for the observed variability in the magnitude of recovery among systems.

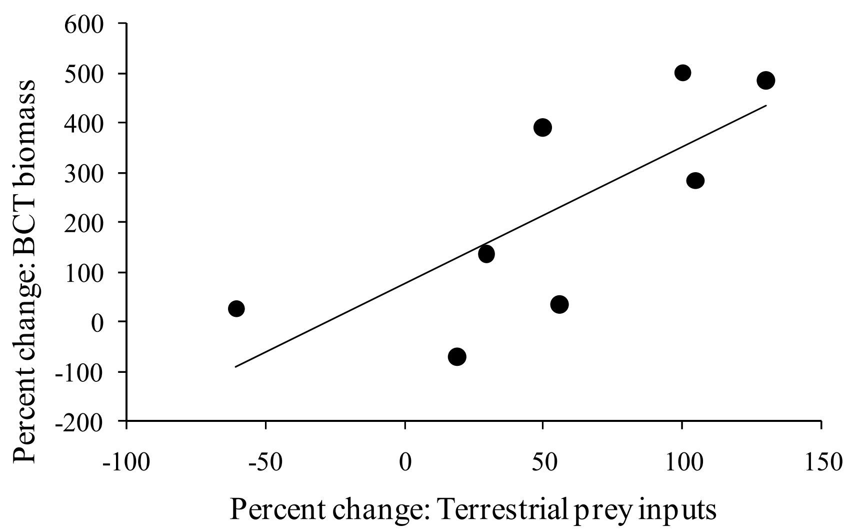
Table 6: Kendall's tau correlation coefficients between grazing regime, exclosure age, exclosure size, BCT density, BCT biomass and the percent change in measured abiotic and biotic variables between ungrazed and grazed reaches. Also presented are correlation coefficient for the relationship between the percent change in BCT density and biomass and the percent change in measured abiotic and biotic variables between ungrazed and grazed reaches.

	Grazing regime	Exclosure Age	Exclosure Size	BCT Density	BCT Biomass
Vegetative Cover	**0.43**	0.27	-0.35	-0.14	-0.07
Residual Stubble Height	**0.73**	-0.23	**-0.46**	0.07	-0.14
Benthic biomass	0.06	0.05	0.28	-0.07	**0.43**
Terrestrial biomass	0.06	0.18	0.11	0.36	**0.57**
Width-to-Depth ratio	**-0.43**	-0.1	**0.63**	0.21	0.14
Proportion undercut banks	0.12	-0.32	-0.14	**0.43**[a]	**0.5**
Proporton overhead vegetation	**0.43**	0	0.13	**0.43**	0.36
Residual Pool Depth	0.06	0.14	-0.04	-0.21	0.14
Habitat Diversity	**0.43**	**0.41**	0.17	0.21	-0.14
BCT Density	-0.1	-0.15	0.14	NA	NA
BCT Biomass	**0.47**	0	-0.07	NA	NA
Averages	0.30	0.17	0.23	NA	NA

[a]Interpretation: Differences in density between ungrazed and grazed reaches increased as differences in inputs of terrestrial arthropods increased.

Figure 9: Relationship between percent change in overhanging vegetation and terrestrial prey inputs with percent change in BCT biomass between ungrazed and grazed reaches.





Invertebrate prey resource responses

On average, total invertebrate prey resource availability was similar between ungrazed and grazed reaches; however, the relative contributions of individual prey categories differed between treatments. Increased biomass of benthic invertebrates in grazed reaches is likely a response to the direct and indirect impacts of grazing, while greater inputs of terrestrial arthropods within exclosures reflect the potential for recovery post-grazing cessation. Increased rates of primary productivity and/or nutrient loading provide the most probable explanation for elevated benthic invertebrate biomass. Light, nutrients, and temperature interact to limit rates of primary productivity, which can in turn control rates of secondary production in autochthonous systems (Allan 2007). Although not directly measured, the amount of incoming solar radiation was likely higher for grazed reaches; we observed 28% less overhead cover and 26% greater width-to-depth ratios on average. Both primary and secondary production have been shown to increase following canopy removal (Feminella et al. 1989). Nutrients also were not measured in this study, but are commonly found at elevated levels in response to cattle grazing (reviewed in Blesky 1999). Alternatively, invertebrate densities have been shown to increase in direct responses to manure inputs, as collector-gatherers consume fecal matter (Del Rosario et al. 2002). We observed active cattle grazing and/or manure inputs within or in close proximity to the stream channel for all grazed reaches. Furthermore, Chironomidae are typically the dominant taxa to respond to manure inputs and were one of the dominant taxa driving biomass increases in our study. Lastly, biomass increases could result from physical habitat alterations favoring larger bodied invertebrates or hyper-abundant taxa capable of producing multiple overlapping generations per year; additional community level analyses are forthcoming.

The strong, positive correlation between overhanging vegetation and inputs of terrestrial arthropods suggests riparian vegetative recovery played a key role in restoring terrestrial arthropod subsidies. We know of only two other studies documenting the effects of grazing on riparian arthropods and both found reduced inputs to stream systems resulting from loss of vegetative cover and structure (Edwards and Huryn 1995; Saunders and Fausch 2007). Increased arthropod diversity and biomass could result from increased vegetative diversity, biomass, and structural complexity within exclosures (Gibson 1992, Debano 2006). Different vegetative assemblages frequently harbor invertebrates with different habits, with shrub and tree communities harboring a greater proportion of climbers and flying arthropods (Collins and Thomas 1991). We observed significantly higher densities of several spider, wasp, bee, grasshopper, and katydid families within ungrazed reaches. Furthermore, greater overhanging vegetation likely increased the probability of all arthropods, including ground dwellers, passively falling into the stream channel.

Fish responses

We observed consistent, positive responses to grazing exclosures by both native and non-native salmonids. Grazing exclosures did not appear to differently benefit native or non-native salmonids, as the relative abundance of all salmonid species was consistent between treatments. Spawn Creek represented the only instance where the ratio of BCT to the non-native brown trout (BNT) changed appreciably; BCT were numerically dominant within the exclosure, while BNT dominated the grazed reach (Appendix 1, Fig. A1-2). In contrast, mountain suckers, the only non-game species present in sufficient numbers to facilitate analyses, did not differ between ungrazed and grazed reaches. The paucity of non-game fish responses combined with relatively high densities in grazed reaches suggests habitat degradation and alterations to prey resources resulting from grazing did not have adverse population effects.

On average, ungrazed reaches had a BCT density and biomass that were twice and four times that of grazed reaches, respectively, while the average condition and age structure did not consistently differ between treatments. The increased biomass response was primarily driven by one site, Dry Creek, where biomass increased by over 2,000%. Excluding this site, density and biomass responses were comparable, 210 and 193 respectively. Given similarities in size structure and condition, biomass increases appear to reflect density increases and not the presence of larger fish within exclosures. Emigration of BCT from upstream or downstream reaches into exclosures provides the most plausible explanation for the increased BCT density and biomass post-grazing. Potential mechanisms attracting BCT to exclosures include increased physical habitat quality, cover, and availability of preferred prey items.

Grazing cessation likely reduced rates of soil compaction and bank trampling, while also releasing riparian vegetation from herbivory. Removal of the direct physical disturbance and recovery of riparian vegetative assemblages have been associated with reduced width-to-depth ratios (Kauffman et al. 2002; Binns and Remmick 1994), increased overhanging vegetation (Bayley and Li 2008; Saunders and Fausch 2007), and overhanging banks (Bayley and Li 2008). Such conditions were observed within the studied exclosures and are typically correlated with optimal salmonid habitat. We observed moderate correlations between BCT density and biomass increases within ungrazed reaches and increases in the proportion of undercut banks and overhead vegetation (Table 6). In contrast, width-to-depth ratios and residual pool depth were not associated with the magnitude of BCT responses. Overhanging vegetation and undercut banks represent preferred salmonid habitat, providing cover from predatory birds, increased opportunities for inputs of terrestrial arthropods, and thus optimal foraging opportunities. The process of restoring salmonid populations through recovery of riparian and instream habitat conditions represents the traditional rationale for implementing riparian grazing exclosures and likely played a significant role within the studied exclosures. Less commonly studied is the role of restoring critical prey resources in facilitating salmonid recovery following grazing cessation (but see Saunders and Fausch 2007).

Terrestrial arthropods comprised only 11% of prey resource availability across all reaches; however, they constituted over 50% of ingested prey items by mass. The dependency of salmonids on terrestrial arthropods during summer months has been repeatedly observed throughout the western U.S. (e.g., Hilderbrand and Kershner 2004; Saunders and Fausch 2007) and has caused several authors to speculate on their role in facilitating the recovery of salmonid populations (Bayley and Li 2008). Furthermore, recent studies manipulating riparian subsidies to stream systems observed dramatic shifts in salmonid densities and foraging behaviors (reviewed in Baxter et al. 2005). In our study, BCT occupying ungrazed reaches had significantly greater biomasses of terrestrial prey items in their diet as compared to those sampled in grazed reaches. We also found correlations between the magnitude of BCT density and biomass increases within exclosures and increases in terrestrial prey resource availability (Table 6). Interestingly, elevated levels of benthic biomass outside the exclosure did not produce a compensatory response for the reduction in terrestrial inputs; either food is not limiting or all benthic biomass is not available to foraging salmonids. These results, combined with other recent studies, suggest terrestrial prey resources may play an integral role in restoring BCT populations following grazing cessation.

Unfortunately, our study did not have sufficient replication to permit modeling efforts investigating the individual and interactive roles of habitat and prey resources in facilitating BCT recovery; however, the two likely interact to increase the quality and quantity of optimal habitat and foraging opportunities for prey items of higher energy density. Consequently, ungrazed reaches were able to accommodate more fish by providing a refuge of better habitat, increased cover from predation, and more optimal foraging opportunities. Although the studied exclosures were generally small, they may harbor a core population that facilitates BCT persistence, both inside and outside the exclosure, in the face of degraded habitat, prey resource availability, and environmental stochasticity. Our study was not designed to assess where exclosures facilitate increased growth and/or a net population increase throughout the system.

Variable responses among systems

Our observation of consistent, positive fish responses across the ten studied systems is surprising given the myriad of past studies producing equivocal results. Salmonid responses to grazing exclosures have been shown to increase, exhibit no change, or be limited to specific size classes or metrics (Platts and Nelson 1985; Binns and Remmick 1994; Saunders and Fausch 2007; Bayley and Li 2008). For example, Bayley and Li (2008) observed significant responses for age-0 rainbow trout, while older age-classes exhibited no response. Despite finding significant results, our study was not immune to such variability. We generally observed consistent directional responses, while the magnitude of change was highly variable among systems; vegetative and geomorphic responses generally exhibited lower variability than prey resource availability, fish populations, and fish diets. In general, variable fish responses to restoration are frequently attributed to biological hysteresis, persistence of watershed-scale degradation, poor connectivity to the regional species pool, and/or the degree of degradation pre-restoration among other viable hypotheses (Sarr 2002; Bond and Lake 2003; Lepori et al. 2005). We attempted to sample different grazing regimes, as well as exclosures of variable size and age to develop hypotheses regarding recovery rates and differential recovery trajectories.

The grazing regime outside the exclosure had the largest impact on the magnitude of differences between grazed and ungrazed reaches (Table 6); the magnitude of change was inversely related to grazing intensity. These results are further exemplified by three observations among systems managed for season long versus short-duration, rotational grazing. The Huff and Big Creek exclosures were constructed in the late 1970's and appear to have suffered the same degree of historic riparian and instream habitat degradation (Platts and Nelson 1985; Binns and Remmick 1994); however, following grazing cessation, Huff Creek's grazing regime was changed from season-long to late-season rotational grazing, while season-long grazing persisted in Big Creek. While we were not able to detect vegetative, geomorphic, prey resource, or fish differences within Huff Creek, conditions were highly disparate between ungrazed and grazed reaches monitored on Big Creek 40 years later. The influence of grazing regime is further illustrated by comparing BCT densities between reaches managed for rotational versus season-long grazing (Table 7); densities were 263 times higher for watersheds managed under rotational grazing regimes on average. Compared to other published BCT densities, densities are similar on reaches managed for rotational grazing practices, but well below average on those managed for season-long grazing. Lastly, sites managed for rotational grazing had a more even proportion of juveniles (grazed: 61.6%; ungrazed: 56.6%) and adults (grazed: 38.4%; ungrazed: 43.3%) than those managed for season-long grazing, which were dominated by adult BCT (grazed: 83.3%; ungrazed: 83.2%) (Appendix 1, Fig. A1-1).

Table 7: Comparison of Bonneville cutthroat trout densities (fish/km) observed under different grazing regimes in this study with regional estimates of BCT densities by management unit (MU) and state. Fish were sampled via electrofishing (EF); N is the number of times the reach was sampled. Grazing managed indicated by season-long (SL) and short-duration, rotational (SDR) grazing. Table modified from Budy et al. 2007.

Location	Years	Grazing Management	Method	N	Reach length (m)	Reach Width (m)	Density (fish/km)	Source
Tributaries of the North Bonneville MU: UT, ID, WY	2008-2009	Grazed: SL	3-pass EF	1	100	2.7 (2.4-3.3)	50 (30-60)	This study
	2008-2009	Grazed: SDR	3-pass EF	1	117.5 (100-170)	3.0 (1.9-4.5)	206 (45-400)	This study
	2008-2009	Ungrazed: SL	3-pass EF	1	100	2.7 (2.4-3.1)	97.5 (30-160)	This study
	2008-2009	Ungrazed: SDR	3-pass EF	1	100	2.2 (1.5-2.8)	330 (200-540)	This study
Logan River and tributaries: UT	2001-2005	Unknown	3-pass EF	6	167 (100-200)	9.7 (7.5-13.7)	696 (66-1,339)	Budy et al. 2007
North Bonneville MU: UT	1996-2004	Unknown	2-pass EF	20	177 (66-745)	3.4 (2.7-4.3)	129 (24-350)	Cowley 1997a, 1997b; Budy et al. 2005; Thompson 2003
North Bonneville MU: WY	1999-2001	Unknown	and 3-pass	6	100	5.2 (2.0-10.3)	128 (50-302)	Schrank et al. 2003; Cowley 2001
Southern Bonneville MU: UT	1994-2005	Unknown	1-pass EF	16	161	2.0 (1.0-3.3)	228 (118-546)	Hepworth 1997

These results suggest changes in grazing regimes at large spatial scales, and not necessarily complete grazing cessation, can be more effective at restoring BCT populations than the small-scale grazing exclosures studied herein. Such findings are concordant with estimates of the mean stream length needed to maintain viable populations of cutthroat trout in small streams of the Intermountain West; Hilderbrand and Kershner (2000b) estimated that a minimum of 5 linear stream kilometers are required to maintain similar population densities to those observed in our study (0.2 fish/m), which is much smaller than the exclosures we studied. The failure of reach-scale restoration to elicit desired biotic responses is frequently attributed to discordance between the scale of restoration relative to the scales of degradation processes (Larson 2001; Bond and Lake 2003; Harrison et al. 2004). For example, reach-scale habitat manipulations often enhance fish populations when perturbations are local in nature, while ecological benefits are generally low when larger, watershed-scale degradation persists (Frissell and Ralph 1998; Lake et al. 2007). Greater BCT population densities within watersheds managed for rotational grazing suggests that larger scale processes are limiting BCT within heavily grazed watersheds, processes which cannot be compensated for with small-scale exclosures. Conversely, natural differences in the biotic potential of individual systems and/or the degree of anthropogenic alteration prior to changes in management strategies could confound our results. These very concerns are the original reason we did not base analyses on comparisons of absolute values among systems.

Qualitatively, differential responses between grazing regimes did not appear to interact with exclosure age or size; these same variables did not explain variable responses in measured biotic and abiotic variables among systems. Similarly, Coles-Ritchie and authors (2007) found weak relationships between riparian wetland index scores and exclosure age. Such findings provide further evidence for the idea of variable recovery trajectories following grazing cessation as a function of degree of degradation among other factors (reviewed in Sarr 2002).

Conclusion and necessary caveats

Grazing cessation within the ten studied exclosures resulted in significantly improved riparian vegetation, geomorphic conditions, prey resource availability, and subsequent BCT populations. However, our results should be interpreted with caution. Specifically, the short-term nature of our study necessitated a space for time substitution where paired restored and unrestored reaches were located on the same stream (i.e., control-impact design [CI]). CI study designs lack the pre-restoration data needed to assess inherent differences between control and treatment reaches that might confound responses to restoration treatments (Laasonen et al. 1998; Halpern 2003; Negishi and Richardson 2003). Furthermore, a paucity of information regarding the extent of pre-restoration degradation and larger landscape-scale conditions hindered our ability to understand variable recovery rates and trajectories.

Despite these limitations, the observed differences in BCT populations between ungrazed and grazed reaches suggest that grazing exclosures effectively create refugia of improved habitat conditions and prey resource availability. Further research is needed to determine if grazing exclosures increase growth and survival rates and thus facilitate a net population increase and/or the persistence of what would otherwise be a population threatened with local extirpation. Ideally, such research would be conducted in the context of determining the size or spatial arrangement of habitat patches needed to increase growth and survival rates. Conversely, our results suggest that when possible, a simpler approach would be to change the grazing regime at the largest possible scale to maximize benefits to BCT populations. Lastly, we demonstrate that terrestrial invertebrate prey resources are a critical component of BCT summer diets and may play a critical role in facilitating their recovery post-grazing. Further research is needed to determine how riparian arthropod assemblages respond to grazing cessation; specifically, whether their composition and biomass is influenced by the reestablishment of native versus non-native riparian vegetation or the absence of willow assemblages.

References

Allan, J.D. 1995. Stream Ecology: Structure and Function of Running Waters. Kluwer Academic, Dordrecht, Netherlands.

Bayley, P.B., & H.W. Li. 2008. Stream fish responses to grazing exclosures. North American Journal of Fisheries Management 28: 135-147.

Baxter, C.V., K.D. Fausch, and W.C. Saunders. 2005. Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. Freshwater Biology 50: 201-220.

Beschta, R.L. 1997. Riparian shade and stream temperature: an alternative perspective. Rangelands 19: 25-28.

Binns, N.A., and R. Remmick. 1994. Response of Bonneville cutthroat trout and their habitat in the drainage-wide habitat management at Huff Creek, Wyoming. North American Journal of Fisheries Management 14: 669-680.

Blesky, A.J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54: 419-431.

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: 193-198.

Budy, P., P. McHugh, G. P. Thiede, and E. VanDyke. 2005. Logan River whirling disease study: factors affecting trout population dynamics, abundance, and distribution in the Logan River, Utah. 2004 Utah Division of Wildlife Resources. Federal Aid in Sport Fish Restoration, Project F-47-R, Annual Report, Utah State University, Logan.

Budy, P., G.P. Thiede, and P.McHugh. 2007. Quantification of the vital rates, abundance, and status of a critical, endemic population of Bonneville cutthroat trout. North American Journal of Fisheries Management 27: 593-604.

Canfield, R.H. 1941. Application of the line interception method in sampling range vegetation. Journal of Forestry 39: 388-394

Caton L.W. 1991. Improving subsampling methods for the EPA “Rapid Bioassessment” benthic protocols. Bulletin of the North American Benthological Society 8: 317-319.

Coles-Ritchie, M.C., D.W. Roberts, J.L. Kershner, and R.C. Henderson. 2007. Use of a wetland index to evaluate changes in riparian vegetation after livestock exclusion. Journal of the American Water Resources Association 43: 731-743.

Collins, N.M. and J.A. Thomas. 1991. The conservation of Insects and their Habitats. Academic Press, London, UK

Cowley, P. 1997a. Fish surveys on the Uinta and Wasatch-Cache National Forest conducted during 1996. Wasatch National Forest, Salt Lake City, Utah.

Cowley, P. 1997b. Fish surveys on the Uinta and Wasatch-Cache National Forest conducted during 1997. Wasatch National Forest, Salt Lake City, Utah.

Cowley, P. 2001. Fish surveys conducted on the Uinta and Wasatch-Cache National Forest during 2001. Wasatch National Forest, Salt Lake City, Utah.

Debano, S.J. 2006. Effects of livestock grazing on aboveground insect communities in semi-arid grasslands of southeastern Arizona. Biodiversity and Conservation. 15:2547-2564.

Del Rosario, R.B., E.A. Betis, and V.H. Resh. 2002. Cow manure in headwater streams: tracing aquatic insect responses to organic enrichment. Journal of the North American Benthological Society 21: 278-289.

Edwards, E.D. and A.D. Huryn. 1996. Effect of riparian land use on contributions of terrestrial invertebrates to streams. Hydrobiologia 337: 151-159.

Feminella, J. W., M. E. Power, and V. H. Resh. 1989. Periphyton responses to invertebrate grazing and riparian canopy in three northern California coastal streams. Freshwater Biology 22: 445–457.

Frissel, C.A., and S.C Ralph. 1998. Stream and watershed restoration. Pages 599-624 in R.J. Naiman and R.E. Bilby, editors. River ecology and management. Springer Verlag, New York.

Gibson, C.W.D., V.K. Brown, L. Losito, and G.C. McGavin. 1992. The response of invertebrate assemblages to grazing. Ecography 15: 166-176.

Gregory, S.V. F.J. Swanson, W.A. McKee, and K.W. Cummins 1991. An ecosystem perspective of riparian zones. BioScience 41: 540-551.

Halpern, B.S. 2003. The impacts of marine reserves: do reserves work and does reserve size matter? Ecological Applications 13: S117-S137.

Harrison, S.S.C., J.L. Pretty, D. Shepherd, A.G. Hildrew, C. Smith, and R.D. Hey. 2004. The effect of instream rehabilitation structures on macroinvertebrates in lowland rivers. Journal of Applied Ecology 41: 1140-1154.

Hawkins, C.P., J.L. Kershner, P.A. Bisson, M.D. Bryant, L.M. Decker, S.V. Gregory, D.A. McCullough, C.K. Overton, G.H. Reeves, R.J. Steedman, and M.K. Young.

1993. A hierarchical approach to classifying stream habitat features. Fisheries 18: 3-12.

Hepworth, D. K., M. J. Ottenbacher, and L. N. Berg. 1997. Distribution and abundance of native Bonneville cutthroat trout (Oncorhynchus clarki utah) in southwestern Utah. Great Basin Naturalist 57: 11–20.

Hilderbrand, R.H., and J.L. Kershner. 2000a. Movement patterns of stream-resident cutthroat trout in Beaver Creek, Idaho-Utah. Transactions of the American Fisheries Society 129: 1160-1170.

Hilderbrand, R.H., and J.L. Kershner. 2000b. Conserving inland cutthroat trout in small streams: how much stream is enough? North American Journal of Fisheries Management 20: 513-520.

Hilderbrand, R.H., and J.L. Kershner. 2004. Influence of habitat type on food supply, selectivity, nd diet overlap of Bonneville cutthroat trout and nonnative borrk trout in Beaver Creek, ID. North American Journal of Fisheries Management 24: 33-40.

Kauffman, J.B., R.L. Beschta, N. Otting, and D. Lytjen. 1997. An ecological perspective of riparian and stream restoration in the Western United States. Fisheries 22: 12-24.

Kauffman, J. B., P. B. Bayley, H. W. Li, P. McDowell, and R. L. Beschta. 2002. Effects of livestock enclosures (corridor fencing) on riparian vegetation, stream geomorphic features, and fish populations. Final Report. Bonneville Power Administration, Projects 2000-050-00 and 2000-051-01, Portland, Oregon.

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

Larson, M.G., D.B. Booth, and S.A. Morley. 2001. Effectiveness of large woody debris in stream rehabilitation projects in urban basins. Ecological Engineering 18: 211-226.

Laasonen, P. T. Muotka, and I. Kivijarvi. 1998. Recovery of macroinvertebrate communities from stream habitat restoration. Aquatic Conservation: Marine and Freshwater Ecosystems 8: 101-113.

Lentsch, L., Y. Converse, and J. Perkins. 1997. Conservation agreement and strategy for Bonneville cutthroat trout (*Oncorhynchus clarki utah*) in the State of Utah. Utah Department of Natural Resources, Division of Wildlife Resources, Publication number 97-19.

Lepori, F., D. Palm, and B. Malmqvist. 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? Ecological Applications 15: 2060-2071.

Merritt R.W. and K.W. Cummins. 1996. An introduction to the aquatic insects of North America. Kendall/Hunt Publishing: Dubuque, Iowa.

Moran, M.D. 2003. Arguments for rejecting the sequential Bonferroni in ecological studies. Oikos 100: 403-405.

Negishi, J.N., and J.S. Richardson. 2003. Response of organic matter and macroinvertebrates to placements of boulder clusters in a small stream of southwestern British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Sciences 60: 247-258.

Platts, W.S. and L. Nelson. 1985. Stream habitat and fisheries responses to livestock grazing and instream improvement structures, Big Creek, Utah. Journal of Soil and Water Conservation 40: 374-379.

Platts, W.S. 1991. Livestock grazing. Pages 389-423 *in* W.R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitat. American Fisheries Society, Special Publication 19, Bethesda, Maryland.

Sarr, D.A. 2002. Riparian livestock exclosure research in the western United States: A critique and some recommendations. Environmental Management 30: 516-526.

Saunders, C.W., and K.D. Fausch. 2007. Improved grazing management increases terrestrial invertebrate inputs that feed trout in Wyoming rangeland streams. Transactions of the American Fisheries Society 136: 1216-1230.

Schrank, A. J., F. J. Rahel, and H. C. Johnstone. 2003. Evaluating laboratory-derived thermal criteria in the field: an example involving Bonneville cutthroat trout. Transactions of the American Fisheries Society 132: 100– 109.

Schrank, A.J. & F.J. Rahel. 2004. Movement patterns in inland cutthroat trout (*Oncorhynchus clarki Utah*):management and conservation implications. Canadian Journal of Fisheries and Aquatic Sciences 61: 1528-1537.

Thorp J.H. and A.P. Covich. 1991. Ecology and classification of North American freshwater invertebrates. Academic Press: San Diego, California.

Thompson, P. 2003. Bonneville cutthroat trout (Oncorhynchus clarki utah) surveys in the northern region, 2002. Utah Division of Wildlife Resources Publication 03-03, Salt Lake City.

U.S. Fish and Wildlife Service. 2001. Status review for Bonneville cutthroat trout (*Oncorhynchus clarki utah*).

Task 2: Quantify geomorphic and hydraulic alterations resulting from instream habitat management and their relevance to BCT populations.

Introduction

The predominant goal of instream habitat restoration is to increase the diversity, density and/or biomass of aquatic organisms through enhanced physical heterogeneity and increased food availability (e.g., Lassonen et al., 1998; Roni et al., 2006). In physically homogenized systems, habitat restoration is most commonly achieved at the reach-scale through the addition of instream structures, channel reconfiguration, and the revegetation of riparian zones. Instream structures and channel construction are intended to enhance hydraulic and substrate heterogeneity, in turn increasing the complexity of physical habitat for fish and other aquatic organisms (Dean and Connell, 1978; Minshall 1984; Scealy et al., 2007). Despite the completion of over 6,000 restoration projects in the United States, studies of fish responses to habitat restoration have largely produced equivocal results (Bernhardt et al., 2005; Bernhardt et al., 2007). Paradoxically, restoration monitoring typically measures a biological response (e.g., Larson et al., 2001) without understanding how this response links to the physical variables being altered. In this study, we investigate whether physical changes that have occurred following instream restoration efforts have ecologically relevant effects on the organism of interest.

We base our analysis on data from an instream restoration project on the Strawberry River, Utah, that is similar in approach to many instream restoration projects of the Intermountain West. Restoration of the Strawberry River aims to increase resident and spawning populations of Bonneville cutthroat trout (BCT) through direct and indirect changes to physical conditions. Bonneville cutthroat trout (BCT) are native to the Bonneville basin in Utah, Idaho, Wyoming, and Nevada. Historically, BCT are assumed to have occupied all suitable habitats within the Pleistocene Bonneville Lake basin (Behnke 1998); however their distribution has been greatly constrained in recent times by human alterations of aquatic systems (Lentsch 1997, USFWS 2001). As a result, BCT has been listed as a Tier 1 sensitive species in Utah. Throughout its range, BCT has been the focus of restoration activities aimed at improving habitat for the various life stages of this species. For instance, bank stabilization techniques are intended to reduce fine sediment loads to spawning gravels, while instream structures are placed to create low-velocity pools and cover for resident juveniles and adults. Despite widespread restoration, quantitative assessments of BCT response are rare. Consequently, benefits of individual actions and the critical methodology contributing to project effectiveness remain unknown. Our study of the Strawberry River restoration project aims to address this knowledge gap.

Low resident populations and limited reproduction of BCT on the Strawberry River have been attributed to several perceived problems, including high bank erosion rates, large fine sediments loads, increasing width-to-depth ratios, limited vegetative cover, and high summer daytime temperatures. (UDWR, 2007). In an attempt to address these issues, instream restoration of the study site began in 2008 and is projected to continue through 2010; we have monitored the effects of restoration for two years (2008 and 2009). In this section, we focus on the methodology and baseline results from the habitat and snorkel surveys from 2008 and 2009. Methodology and preliminary results from other components of the monitoring program are provided in the appendix. Our analysis addresses four monitoring questions:

1. How does instream habitat restoration alter geomorphic and hydraulic conditions, and are such changes persistent through time?
2. What geomorphic and hydraulic variables limit juvenile and adult BCT distribution and habitat use on the Strawberry River?
3. Do geomorphic and hydraulic changes occur at spatial scales relevant to the variables limiting BCT populations?
4. Does restoration result in a change in BCT density, size structure, or spawning habitat use?

From these four questions a similar monitoring framework could be established for other active restoration projects designed to illicit a biological response through physical change. With Q1, one can assess the magnitude, direction, and temporal trend of physical changes resulting from restoration. An answer to Q2 establishes what physical variables, if any, are limiting to the organism of interest, from which one can identify the problems that restoration should address. In Q3, needs identified in Q2 are compared with the changes measured in Q1 to determine whether restoration changes physical conditions in an ecologically relevant manner. Finally, Q4 asks if a biological response occurs that may or may not be related to the physical changes identified.

As stated in the Utah Division of Wildlife Resources (UDWR) Strawberry River Restoration Phase III project proposal, specific project objectives are as follows (UDWR, 2007):

- Restore and maintain the natural dimension, pattern, and profile of the Strawberry River (defined as a single thread meandering channel)
- Improve upstream fish migration from Strawberry Reservoir
- Stabilize eroding banks
- Reestablish a more natural riparian plant community
- Reduce stream temperatures
- Reconnect river to historic flood plain
- Improve and increase complexity of aquatic habitat
- Reduce fine sediment and improve spawning habitats

A natural channel design strategy (Rosgen, 1996) is being employed in which rock and log vanes, root wads and logs, and revegetation techniques are used to stabilize banks, create physical diversity, increase channel roughness, and provide cover for trout. Revegetation methods include the sloping of vertical banks, transplanting of willow clumps, planting of willow clippings, spreading of coconut fiber on outside bends of meanders, and reseeding of disturbed areas with native riparian species. Pools are made deeper and more frequent by digging out bed material.

In order to assess whether these restoration activities achieve project goals over the long-term, we have established an ongoing, multi-metric monitoring program on the Strawberry River. We are using a variety of geomorphic and ecological techniques to assess the physical and biological conditions of the system before and after restoration. In collaboration with UDWR, our comprehensive program includes:

- Historic analysis of channel planform geometry and migration rates using aerial photos
- Geomorphic assessment of longitudinal and cross-sectional channel morphology using high-precision GPS surveys
- Habitat surveys of micro- and reach-scale geomorphic, hydraulic, and chemical conditions
- Water quality, turbidity, and suspended sediment monitoring
- Surface and subsurface sediment sampling
- Redd surveys and micro-scale spawning habitat measurements
- Snorkel surveys of fish use and population abundance
- Depletion estimates of fish population abundance and size structure

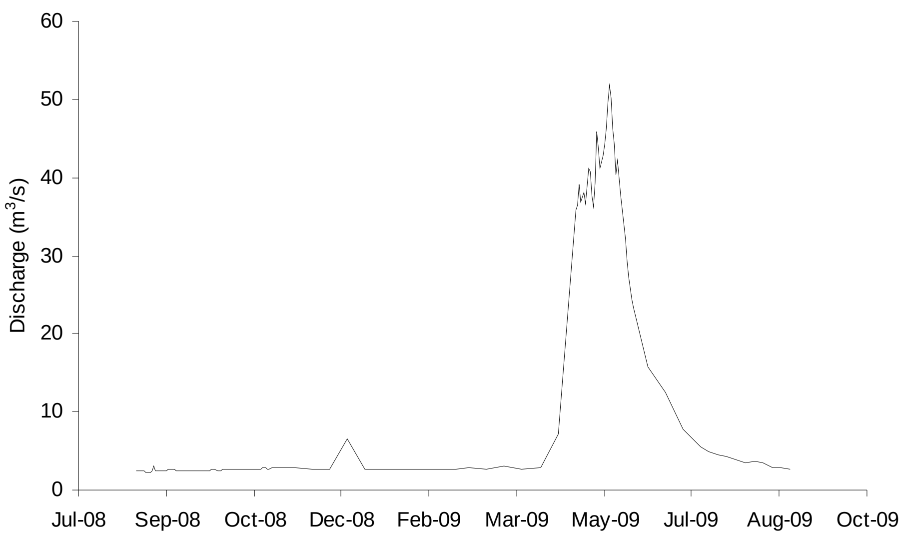
Because at this point in time we are limited to two years of data, we place less emphasis in this report on the short-term results and more emphasis on the methodology and baseline data collected during these assessments to facilitate our continued study of this system. In the long term, we will assess whether restoration has achieved projects goals and whether such changes are persistent through time.

Methods

The Strawberry River is located in Wasatch County, Utah, on the Heber Ranger District of the Uinta National Forest. Detailed information about the physiography, climate, geology, soils, vegetation, and hydrology of the Strawberry River watershed can be found in several documents (U.S.D.A. Forest Service 2004; Utah Department of Environmental Quality 2007). For this study, we focused on a ~8 km section of the Strawberry River from Bull Springs Road upstream to Highway 40, upstream of the Strawberry Reservoir. Restoration of this section of river began in July 2008 and is expected to continue until 2011, under the direction of the Utah Division of Wildlife Resources (UDWR) in collaboration with the U.S. Forest Service.

The Strawberry River is a low-gradient, meandering channel with riffle-pool sequences and a sand-gravel-cobble substrate, although upstream reaches contain beaver complexes with long, deep pools and silt substrate. Riparian vegetation is dominated by willow, grasses, sedges, and sagebrush. The flow regime is snow-melt dominated, with peak flows occurring in late April and early May, receding to base flows in late summer (Fig. 10).

Figure 10: Strawberry River hydrograph, August 2008 – August 2009.



We established survey reaches within each restored and unrestored section of the river in a modified before-after-control-impact (BACI)-type study design. Surveys were conducted in September 2008 and September 2009 at baseflow conditions (~0.24 m^3/s in both years). In September 2008, following the first phase of restoration, three 500 m-long reaches were established in an upstream progression, including the downstream-most reach that had been recently restored (1-month post-restoration) and two unrestored reaches (Table A2.I-1; Fig. 11). In July of 2009, the middle of the three reaches was restored, resulting in two restored reaches for our September 2009 surveys (1-year post-restoration and 1-month post-restoration) and the remaining upstream unrestored reach as a control. We selected a 200 m section nested within each 500 m reach for snorkel and habitat availability surveys. A BACI-type design allows for three types of comparisons:

1) before-after (pre- vs. post-restoration at a single site), 2) control-impact (unrestored site vs. restored site), and 3) temporal trends before and after restoration vs. temporal trend at control site (i.e., whether the change is due to restoration or an outside source of variability). This method is advantageous because it accommodates temporal change and natural variability, but can be problematic if paired control and impact sites differ in ways other than the treatment effect. A timeline summarizing the schedule of restoration and monitoring projects is provided in Table 1. Details of the surveys are provided in the following sections.

Figure 11: Map of survey reaches and schematic of cross-sections within a reach on the Strawberry River. Black points on map show locations of cross-sections; schematic distances are not to scale.

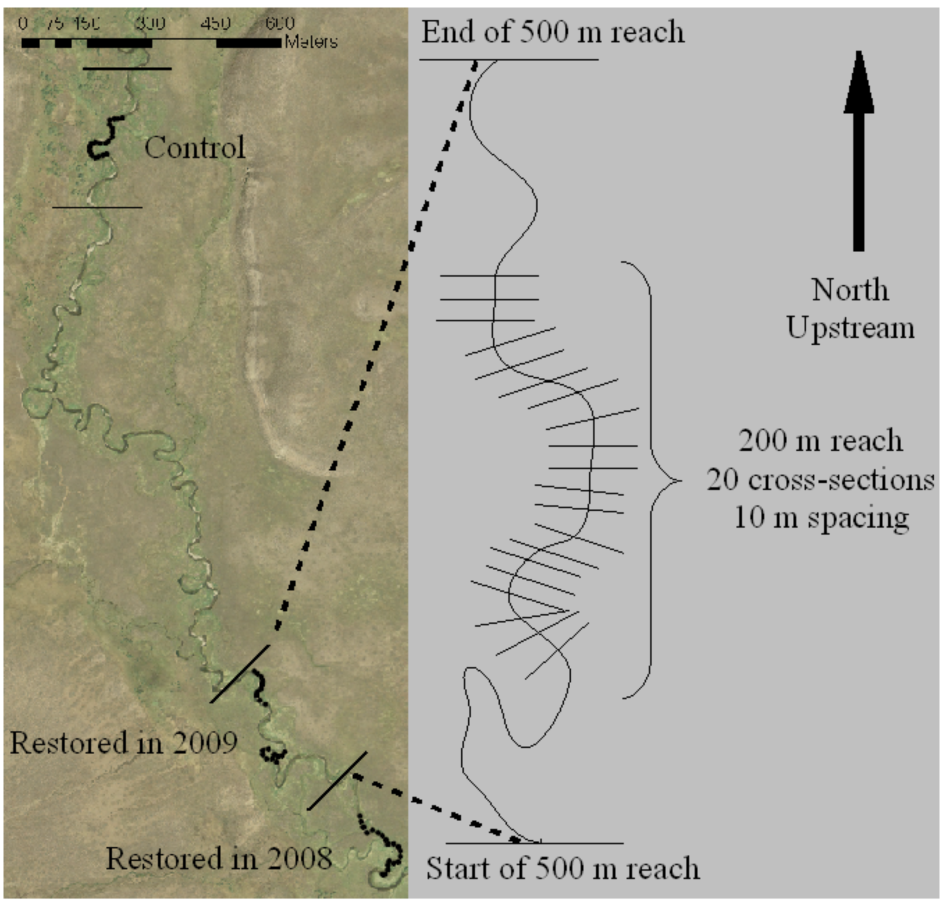


Table 8: Timeline of restoration and monitoring projects in 2008 and 2009, Strawberry River.

Year	Month	Activity	Conditions monitored
2008	July-August	Begin restoration upstream of Bull Springs	
	September	Snorkel surveys of resident trout use and availability	1 mo. post-restoration (Impact) 1 yr. pre-restoration (Before) Unrestored (Control)
2009	July-August	Continue restoration upstream of Bull Springs	
	September	Snorkel surveys of resident trout use and availability	1 yr. post-restoration (Impact) 1 mo. post-restoration (After) Unrestored (Control)

Habitat availability and morphological complexity

We used point measurements on evenly spaced cross-sections to quantify local-scale physical variability and microhabitat availability for BCT. Twenty channel-spanning cross-sections were established at regular intervals (i.e., every 10 m) along each 200 m reach. Locations of all cross-sections were identified with a high-precision Topcon RTK Pro GPS surveying system (Table A2.I-2). For each cross-section we measured the wetted width and unit type, such as pool (including lateral scour and thalweg, natural and artificially constructed), riffle, run, or glide. Unit definitions were based on previously established criteria of depth, velocity, and turbulence (Hawkins et al. 1993; Heitke et al. 2008). Pools were defined by a maximum depth ≥1.5 times the tailout depth. Glides were considered the relatively slow-moving, laminar, and shallow transitions between pools and riffles.

At twelve evenly spaced points on each cross-section (including 10 within the channel and one at each water's edge), we measured flow depth, near-bed velocity, average velocity (at 0.6 of the flow depth), overhead cover, and substrate size. Five categories of cover type adopted from the U.S. Forest Service PIBO monitoring protocol (Heitke et. al,

2008) were used: large woody debris (LWD; > 1 m long and 10 cm diameter); turbulence (bottom is not visible); undercut bank (> 5 cm deep and > 10 cm long); boulder (substrate > 125 mm); depth (> 0.7 m); overhanging vegetation (within 1 m of water surface and overhangs by > 0.5 m); and aquatic macrophytes. In restored reaches, artificial LWD and artificial boulders were also identified when apparent; in some cases it was difficult to assess whether the objects were natural or introduced, thus we combine natural and artificial structures in our analysis. Two substrate particles were randomly selected at each point and measured with a gravelometer to quantify local substrate size for a total of 24 particles along each cross-section. For the 2008 surveys, the smallest maximum particle size measured was 8 mm; given the importance of fines < 2 mm to the quality of spawning habitat, however, 2 mm was the smallest maximum particle size measured in the 2009 surveys. Stones were grouped into standard particle size classes: < 2, 2-4, 4-5.7, 5.7-8, 8-11.3, 11.3-16, 16-22.6, 22.6-32, 32-45.3, 45.3-64, 64-90.5, 90.5-128, 128-180, 180-256, and > 256 mm. Point measurements of depth, velocity, cover, and particle size were combined from all cross-sections on each reach to represent the spatial distribution of micro-scale physical conditions for comparison among reaches.

Morphological unit diversity was compared among reaches using the number and proportion of unit types in a Shannon diversity index, which accounts for both richness and evenness. We calculated the proportion of cross-sections on each reach that were identified as a particular morphological unit (pool, riffle, etc.). We calculated the width-to-depth ratio – based on the wetted width and maximum depth – of all cross-sections within a reach.

Snorkel surveys

We conducted daytime snorkel surveys to quantify BCT habitat use at the micro-, unit, and reach scale. Snorkelers began at the downstream end of each 500 m reach and moved upstream in a zigzag pattern (Thurow 1994). For each observed BCT, we estimated the length and focal elevation (i.e., bottom one-fourth, one-fourth to one-half, one-half to three-quarters, and three-quarters to top water depth) and marked trout locations with flagged washers or painted rocks. Observed fish were identified as either adults or juveniles based on their estimated length (greater or less than 150 mm, respectively). Snorkel counts were used to estimate adult and juvenile BCT density (number of fish per area of stream, computed using the reach-averaged width) on each reach. During the September habitat availability surveys, we measured local-scale physical conditions of each trout use location: depth; near-bed and average velocities; cover type; and size of 10 randomly selected surface particles. For trout use locations measured in 2008, 8 mm was the smallest maximum particle size measured; all other particle size classes were the same as the habitat availability surveys. In 2009, the smallest particle measured was 2 mm.

Point measurements of depth, velocity, cover, and particle size were combined from all trout use locations to develop microhabitat suitability indices. For each microhabitat variable (depth, velocity, and particle size), classifications were based on the frequency distribution of that variable. Classifications were defined as follows: “optimal” was the central 50% of the distribution; “useable” was the central 95%; and “unsuitable” was less than 5% or greater than 95% of the distribution. All forms of cover were considered optimal, and no cover was considered unsuitable based on previous studies and the published habitat suitability index for cutthroat trout. Composite habitat quality classes were further defined in a limiting condition framework, such that “optimal’ habitat required fully optimal conditions and “unsuitable” habitat was defined by any unsuitable conditions. Habitats were “optimal” if all microhabitat values were optimal; “useable” if all values were optimal, useable, or both; and “unsuitable” is any values were unsuitable. We then classified each point measured in the habitat availability surveys and calculated the percent of each habitat class on each reach for each year. We developed these classifications from the fish use data of the unrestored reaches in 2008 for comparison with availability on unrestored and restored reaches in both years. Suitability indices were used to quantify the effects of restoration on the proportion of “optimal” habitat.

Redd surveys

We conducted redd surveys between BCT spawning season and the predicted date of emergence. We began surveys once flows dropped to baseflow and spawning activity had ceased, completing all measurements within two weeks. The full 500 m length of all three reaches was waded at least three times by one or two individuals; BCT redds were identified by their characteristic shape, marked with a numbered metal washer, and located with a high-precision GPS (Table A2.III-1). We returned to each redd during the habitat availability surveys to measure micro-scale physical conditions: depth; near-bed and average velocities; temperature (associated with time of day); pH; electrical conductivity; cover type; and surface particle size distribution. Surface particle size distribution was quantified by randomly selecting and measuring with a gravelometer 100 stones from within a 1 m^2 grid placed over the redd; the sampler looked aside while pointing a finger at the streambed, picked up the first stone touched, and traversed the grid in a zigzag manner until 100 stones were measured (Table A2.III-2). For redds, 2 mm was the smallest maximum particle size measured.

Depletion estimates

Three-pass electrofishing depletion estimates of BCT population abundance were conducted on each reach by UDWR in August of 2008 and 2009. Lengths and weights of all BCT were measured. We used a maximum likelihood estimation of population size and total fish biomass from the depletion data to calculate fish density (linear and areal) for comparison among reaches and years. We also compared the distributions of BCT length and weight among reaches and years.

Results and Discussion

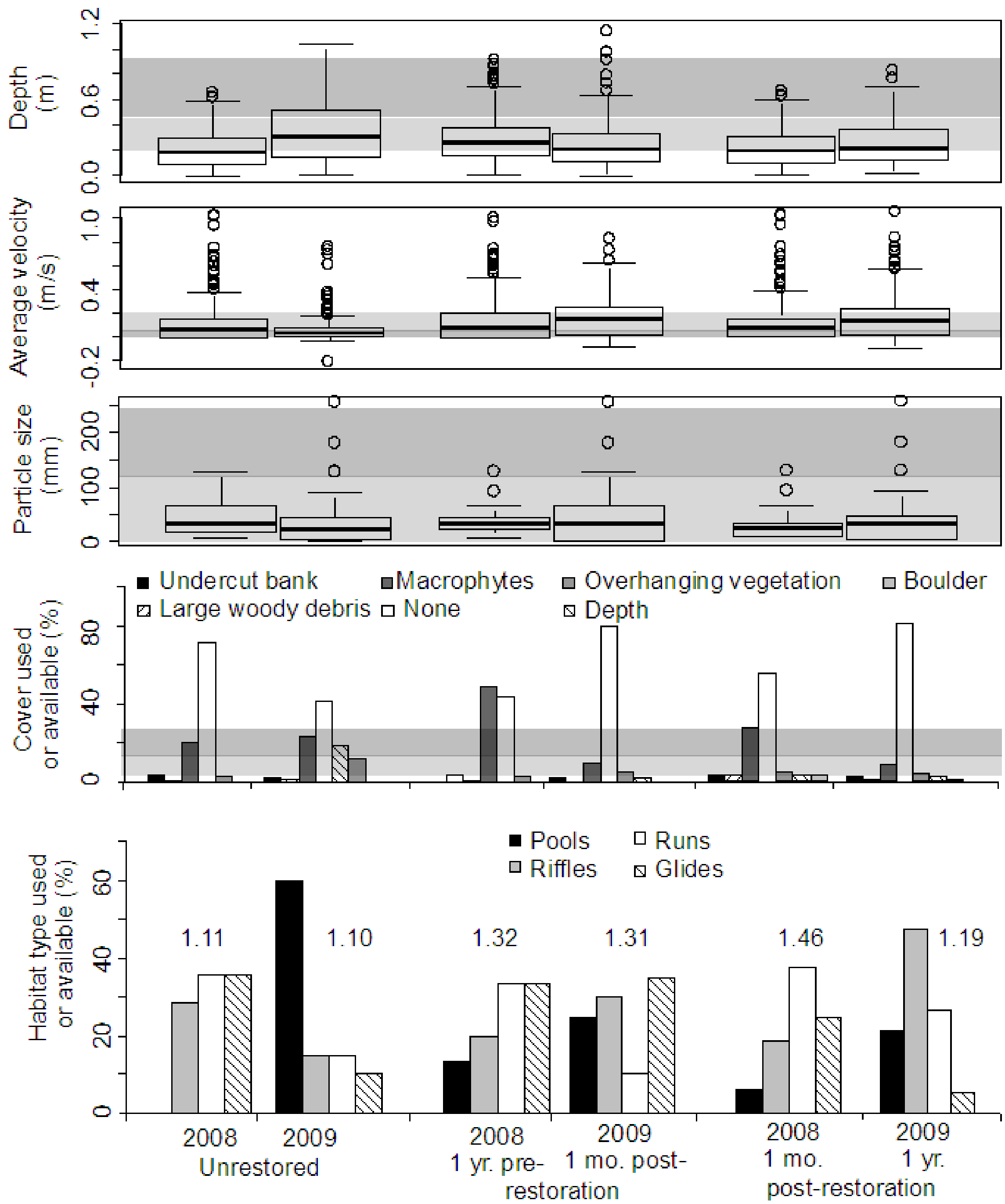
Data collected from the habitat availability surveys, snorkel surveys, and depletion estimates are used to answer each of the four monitoring questions outlined above. To determine whether restoration has altered physical conditions (Q1), we compare changes in the distribution of physical variables pre- and post-restoration to changes exhibited on unrestored reaches in the same time periods; a difference in the magnitude or direction of temporal change on restored and unrestored reaches potentially indicates a restoration effect. In order to establish which physical conditions are limiting to adult and juvenile BCT (Q2), we compare fish use of physical variables to the distribution of those conditions prior to restoration. A lack of overlap in the distribution of use and availability, or a disproportionate use relative to availability, indicates a potentially limiting variable. We then assess if changes due to restoration are relevant to BCT (Q3), if those changes increase the availability of one or all of the identified limiting variables. Finally, we assess biological response to restoration (Q4) by examining temporal trends in BCT abundance and size structure (length and weight) on restored and unrestored reaches; an increase in BCT abundance or biomass density on restored reaches without a comparable increase on unrestored reaches would indicate a positive effect of restoration.

Question 1: How does instream habitat restoration alter geomorphic and hydraulic conditions and are such changes persistent through time?

A comparison of physical conditions on unrestored and restored reaches shows little change in the distribution of point-scale physical variables (depth, velocity, and particle size) one month after restoration (Fig. 12). Increased beaver activity on the control reach in 2009 resulted in a higher median and maximum flow depth, increased proportion of depth as cover, and a high proportion of pools (60%) compared to none in the previous year. On the restored reaches, changes are most apparent in the proportions of cover and habitat unit types. Macrophyte cover declines in 2009 on both restored reaches (one month and one year after restoration) resulting in an increase in the proportion of locations without cover. Although undercut banks, overhanging vegetation, and depth as cover increase slightly one month after restoration, it appears that 'as-built' conditions do not solve the problem of limited cover. Long-term monitoring will be necessary to determine if vegetative growth overtime increases overhead cover. Despite negligible changes in depth and velocity, the proportion of pools does increase one month and one year after restoration. Artificial pool construction may explain the immediate increase in pool frequency. However, given that neither depth nor the proportion of depth as cover increased following restoration, any increase in pools did not effectively address the problem of depth as a limiting variable.

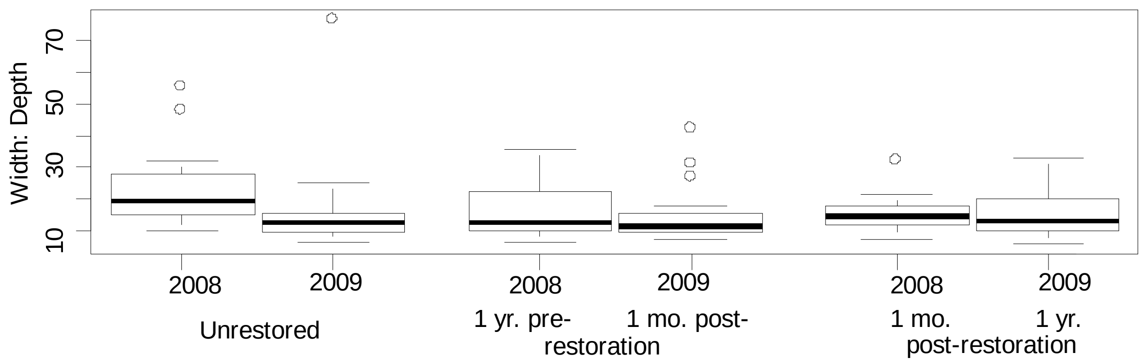
On the control reach, morphological unit diversity has a Shannon index value of ~1.1 in both years, which is lower than both restored reaches due to low richness in 2008 (no pools) and low evenness in 2009 (high proportion of pools). Meanwhile, increases in pool and riffle numbers following recent restoration do not change unit diversity (~1.3 pre- and one month post-restoration). One year after restoration, diversity decreases from ~1.5 to ~1.2 due to a large decrease in the number of glides, increase in the number of riffles, and thus lower evenness; however this change is not easily explained and may be due to observer subjectivity.

Figure 12: Physical conditions (depth, average velocity, substrate, overhead cover, and morphological unit type) on unrestored and restored reaches, Strawberry River in September, 2008 and 2009. Numbers atop columns are the Shannon diversity indices calculated for each reach.



Reach-scale changes in channel morphology are small between years (Fig. 13); width-to-depth ratios of channel cross-sections decrease slightly on the unrestored control reach due to beaver activity and the increase in deep pools, but restored reaches show virtually no change. Significant changes in channel width-to-depth ratios will likely take several years to occur if bank stabilization has been effective.

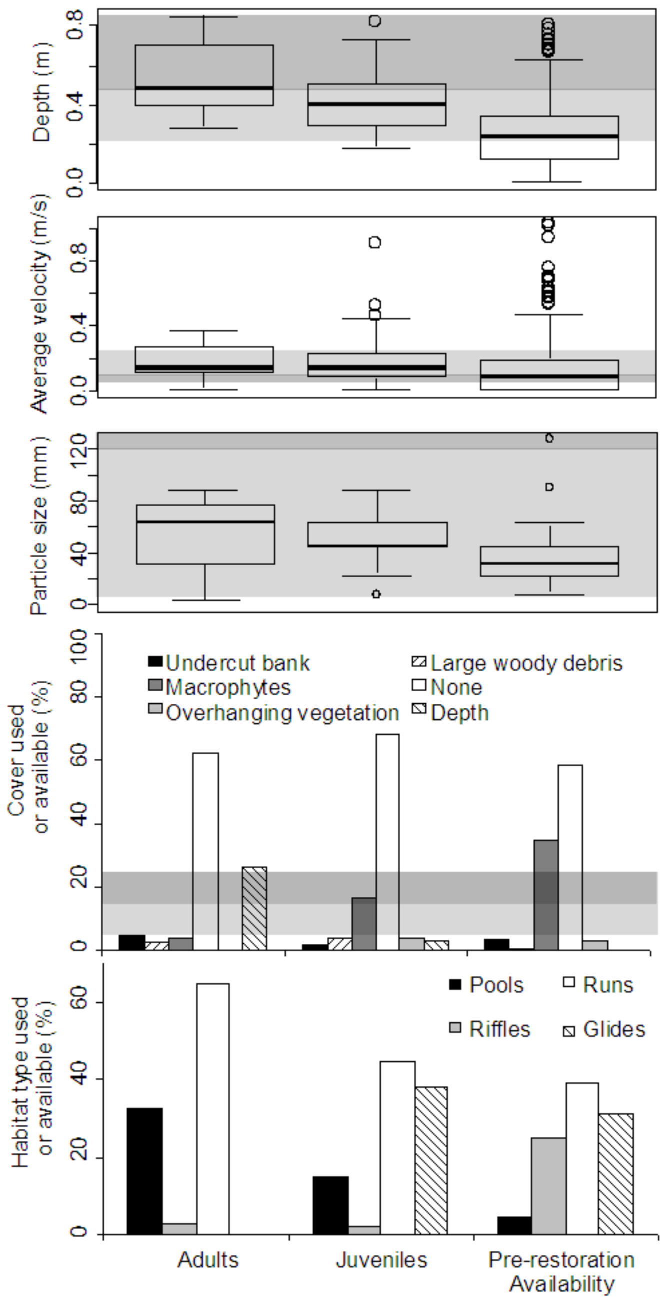
Figure 13: Width-to-depth ratios of cross-sections on restored and unrestored reaches, Strawberry River, September 2008 and 2009



Question 2: What geomorphic and hydraulic variables limit juvenile and adult BCT distribution and habitat use on the Strawberry River?

Depth, pools, and cover are the most limiting habitat variables for both adult and juvenile BCT, although depth is more limiting for adults than juveniles (Fig. 14). Depths used by adults fall above the upper quartile of what is available, indicating a limited supply. Furthermore, depth is the most commonly used form of cover for adults but is clearly limiting (habitat availability estimated at zero). In contrast, macrophytes are the most commonly used by juveniles but are less proportionately used relative to availability, indicating unlimited supply. While macrophytic vegetation is the most common form of cover available before restoration, large woody debris and depth are disproportionately used relative to their availability. Based on the use vs. availability comparison, macrophyte cover may be considered less desirable for juveniles and unsuitable for adults. Excluding macrophytes, total availability of cover on the Strawberry River would be rated average to poor by the HSI criteria (<10%). Among morphological unit types, pools and runs are disproportionately used by BCT relative to their availability; given that pools are characteristically deep and often formed by large woody debris, this result corroborates the limited supply of depth and cover. In contrast, shallow, high-velocity riffles are used in much lower proportions relative to their availability and shallow, low-velocity glides are not used by BCT at all despite high availability.

Figure 14: Pre-restoration adult and juvenile BCT habitat use (depth, average velocity, substrate, overhead cover, and habitat type) compared to habitat availability on the Strawberry River.



Notes: Whiskers are defined by the sample minimum and maximum; boxes indicate the lower quartile, median, and upper quartile; open circles are outliers (outside 1.5 times the interquartile range). Dark and light gray boxes indicates values rated as 'good' and 'average', respectively, based on the cutthroat trout HSI model (Hickman and Raleigh, 1982). Given the similar values reported for adults and juveniles, only the adult values are shown.

Table 9: Suitability values of flow depth, flow velocity, percent of overhead cover, and substrate for juvenile and adult cutthroat trout (determined from habitat suitability indices).

Physical variable	**Below average**		**Average**		**Good**	
	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult
Depth (m)						
≤5m width channel	<0.10		0.10-0.30		0.30	
>5m width channel	<0.22		0.22-0.45		0.45	
Velocity (m/s)	>0.22		0.12-0.22	0.14-0.22	0.10-12	0.10-0.14
Cover (%)	<2	<4	2-15	4-25	15	25
Dominant substrate	Fines, bedrock, or large boulders		Equal distribution of sizes or gravel		Small boulders	

Notes: In the HSI, variables with an index <0.4 are considered "below average" and can not be compensated for by higher values of related variables; "average" variables are those with an index of 0.4-1.0 and "good" variable have an index of 1.0. We extracted values for below average, average, and good conditions from the adult and juvenile HSI diagrams for flow depth, percent cover, and dominant substrate. We also estimated values for average velocity based on literature reviewed in the HSI publication.

Question 3: Do geomorphic and hydraulic changes occur at spatial scales relevant to the variables limiting BCT populations?

Depth, pool frequency, and overhead cover were identified as potential limiting habitat variables for adult and juvenile BCT. At the reach scale, pool frequency increased with recent restoration, but as seen in Fig. 11, this did not translate to an increase in depth or cover at the point scale. Continued monitoring will be necessary to determine whether restoration activities produce less rapid responses such as vegetation growth and channel narrowing that will eventually increase cover and channel depths.

Ranges for specifying "optimal", "useable", and "unsuitable" BCT habitat (Table 3) differed for adults and juveniles likely due to behavioral and physiological differences. For instance, smaller-sized juveniles seemed to tolerate shallower depths than adults; useable depths were those greater than 0.13 m for juveniles and greater than 0.21 m for adults. Useable average velocities were similar for adults and juveniles, although juveniles used slightly higher-velocity locations in both years. Particle size did not appear to limit habitat suitability, with useable conditions encompassing a wide range of sizes, from fine particles < 2 mm to cobble-sized particles > 90 mm. Only large cobble- to boulder-sized material was considered unsuitable.

Table 10: Characteristics of “optimal”, “useable”, and “unsuitable” habitat for BCT adults and juveniles, Strawberry River.

	Depth (m)	**u_0 (m/s)**	**U (m/s)**	**Particle size (mm)**	**Cover**
Adults					
Optimal	0.40-0.70	0.00-0.04	0.11-0.27	32.0-64	Cover
Useable	0.21-0.87	-0.19-0.28	-0.04-0.39	1.1-105.8	None
Unsuitable	<0.21, >0.87	< -0.19, >0.28	<-0.04, >0.39	<1.1, >105.8	
Juveniles					
Optimal	0.30-0.50	0.00-0.06	0.08-0.23	22.6-64.0	Cover
Useable	0.13-0.67	-0.09-0.41	-0.09-0.41	-7.8-93.9	None
Unsuitable	<0.13, >0.67	<-0.09, >0.41	<-0.09, >0.41	<-7.8, >93.9	

Notes: For each microhabitat variable (depth, velocity, and particle size), classifications were based on the frequency distribution: optimal = central 50%; useable = central 95%; unsuitable = less than 5% or greater than 95%. All forms of cover were considered optimal and no cover unsuitable based on previous studies and published the habitat suitability index for cutthroat trout.

Our habitat classification values correspond well with values used in the cutthroat trout habitat suitability index (HSI) model. For depth, our useable habitat ranges from 0.21-0.87 m for adults and 0.13-0.67 m for juveniles, similar though extending above the 0.22-0.45 m reported in the HSI (for channels >5 m wide). Our useable average velocities range from -0.04-0.39 for adults and -0.09-0.41 for juveniles, which encompasses but is wider than the HSI range for average habitat. Less than 4 and 2% cover for adults and juveniles, respectively, are considered below average in the HSI, suggesting that the ~70% of juveniles and adults that occupied locations with no cover in 2008 were either only in temporary residence or were being forced to use sub-optimal habitat due to lack of availability. Like the HSI, our unsuitable substrate conditions included very fine or very large particles. However, cutthroat in our system did not use small boulders (identified as good by the HSI), simply because these particles of this size are not naturally found in the Strawberry River; only large boulders were introduced as part of instream restoration.

Using our habitat classification criteria, we calculated the percentage of each habitat class on the unrestored and restored reaches in 2008 and 2009 (Table 4). For adults, the percentage of optimal and useable habitat decreased, and unsuitable habitat increased, on all reaches except the control reach, which had an 18% increase in useable habitat balanced by a 17% decline in unsuitable habitat, and the recently restored reach (one month post-restoration), which had no change in optimal habitat. For juveniles, the percentage of optimal and useable habitat declined on all reaches. Changes in habitat condition were greatest for the recently restored reach, with increases in unsuitable habitat of 17 and 26% for adults and juveniles, respectively. To understand the cause of this increase in unsuitable habitat, we calculated what percentage of each physical parameter used in the composite index (depth, near-bed velocity, average velocity, substrate, or cover) fell into each habitat class. We found that in both years, depth had the highest proportion of unsuitable conditions (30-50%), supporting the idea that depth is limiting in this system. However the greatest change in optimal conditions occurred for cover. In 2009, optimal cover dropped from 55 to 20% on the recently restored reach and increased from 27 to 57% on the control reach. Such changes support the idea that cover is limiting on all reaches except for the 2009 beaver-dominated control reach. Slight increases in the percentage of unsuitable conditions for average velocity and particle size (2-7%) on the restored reaches combined with this loss of cover to increase the overall proportion of unsuitable habitat. Use of a limiting factor suitability index thus demonstrates how changes in a single parameter can have a profound effect on the overall quality of instream habitat.

Table 11: Percentages of optimal, useable, and unsuitable habitat for adult and juvenile BCT habitat, Strawberry River, September 2008 and 2009.

Year	Condition	Adults			Juveniles		
		Optimal	Useable	Unsuitable	Optimal	Useable	Unsuitable
2008	1 mo. post-restoration	1.3	48.3	49.6	0.8	62.5	35.8
2009	1 yr. post-restoration	0.0	42.7	56.5	0.0	53.6	45.6
% Change	**2008 to 2009**	**-1.3**	**-5.6**	**6.9**	**-0.8**	**-8.9**	**9.8**
2008	1 yr. pre-restoration	0.8	55.8	43.3	3.8	70.8	25.4
2009	1 mo. post-restoration	0.8	39.3	60.3	0.8	48.1	51.5
% Change	**2008 to 2009**	**0.0**	**-16.5**	**17.0**	**-3.0**	**-22.7**	**26.1**
2008	Unrestored-Control	1.3	42.1	56.7	2.1	62.3	36.7
2009	Unrestored-Control	0.0	59.9	39.7	0.0	55.7	43.9
% Change	**2008 to 2009**	**-1.3**	**17.8**	**-17.0**	**-2.1**	**-6.6**	**7.2**

Question 4: Does restoration result in a change in BCT density, size class, or spawning habitat use?

Population densities of total BCT from snorkel surveys (Fig. 15c) and depletion estimates (Fig. 16a) show an increase in total BCT population density on all reaches from 2008 to 2009. Snorkel surveys demonstrated an increase in density with time since restoration. Depletion estimates also show no significant change in total density on the unrestored control reach, while both restored reaches increase both one month and one year after restoration. However, unlike snorkel surveys depletion estimates record the highest density one month after restoration. Furthermore, the increase in density is only for juveniles (Fig. 15b) and small size classes (Fig. 17b and c), results supported by the general decline in fish weights and lengths (Fig. 16c and d). A measure of fish biomass per unit area, rather than number, shows a slightly different pattern among reaches (Fig. 16b). BCT biomass density increased on the two reaches with no physical changes from 2008 to 2009 (control and restored in 2008) but decreased on the recently restored reach. All of these results point to an increase in young, small (< 150 mm) fish, particularly on the recently restored reach (one month post-restoration). An increase in young fish on the restored reaches could have been due to a) improved reproductive success on restored reaches, b) increased survival of early life stages, c) increased habitat use (i.e., via migration) of the restored reaches, d) stocking of young fish at least one year prior to these measurements or some combination of all effects.

In order to attribute these increases to a reproductive effect of restoration, we would have had to see an increase in spawning or egg survival two years prior to our surveys. Given that our most recent survey was only one year after restoration, a positive effect of restoration on reproductive success was unlikely, and redd counts in July 2009 showed little evidence of a reproductive effect. Redds were most abundant on the reach restored in September 2009 (one month prior to restoration), slightly lower on the reach restored in 2008 (one year post-restoration), and lowest on the unrestored control. All redds were found on the upstream ends of riffles. Extensive beaver dam backwater areas limited the number of riffles on the unrestored control reach, a likely cause of the lower redd count (5 redds). Meanwhile, one year after restoration the number of redds was less (8 redds) than on the reach one month before restoration (12 redds) thus spawning habitat use did not show a clear response to restoration.

A second possibility is that the increase in juvenile BCT on the restored reaches in 2009 was due to increased habitat use (i.e., the total number or density of juvenile BCT has not changed, only their distribution among reaches), either because the habitat was preferable or the reaches were simply more accessible. Further, the number of beaver dams on the upper control reach may have limited juvenile movement upstream. Given that we did not detect significant changes in habitat from 2008 to 2009 or differences between the restored reaches, it is unlikely that BCT juveniles were responding to an improvement in habitat quality.

Most likely, the observed increase in juveniles on the restored reaches was due to stocking efforts. From September-October in 2007 and 2008, UDWR stocked ~300,000 and 450,000 young-of-year BCT, respectively, to locations throughout the study site and tributaries of the Strawberry River. Fish ranged in size from ~80-90 mm (J. Robinson, pers. comm.). With an additional 150,000 fish added in 2008, it is not surprising that we detected an increase in small fish during population surveys in 2009. We are thus unable to attribute the positive response in BCT populations to restoration. Future monitoring will be needed to assess whether this increase in juvenile abundance, combined with habitat restoration, translates to more resident adult BCT, greater reproductive success, and higher natural recruitment.

Figure 15: Snorkel counts of adult (> 150 mm), juvenile (< 150 mm), and total BCT densities (fish per square meter) on restored and unrestored reaches in September 2008 and 2009, Strawberry River.

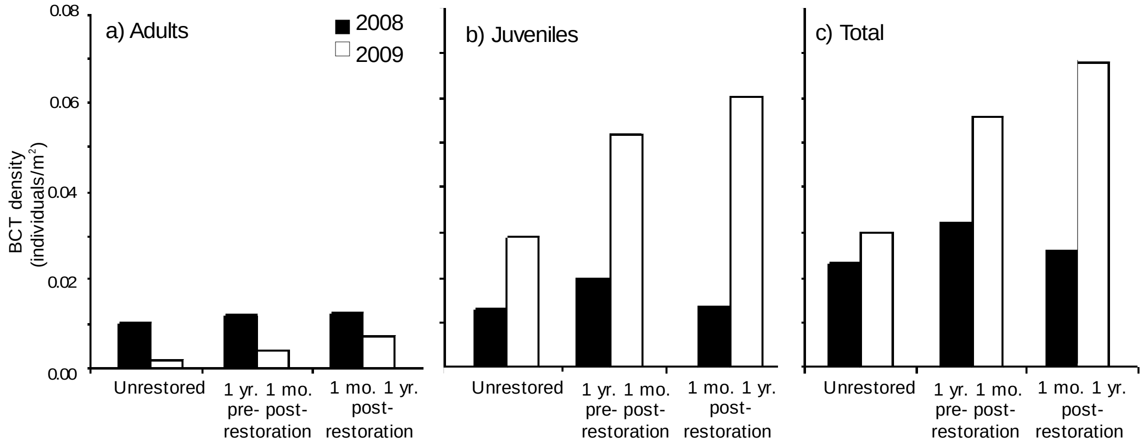


Figure 16: BCT population density (a), biomass density (b), and distributions of weight (c) and length (d) from three-pass electrofishing depletions on restored and unrestored reaches in August 2008 and 2009, Strawberry River. Error bars are the 95% confidence interval.

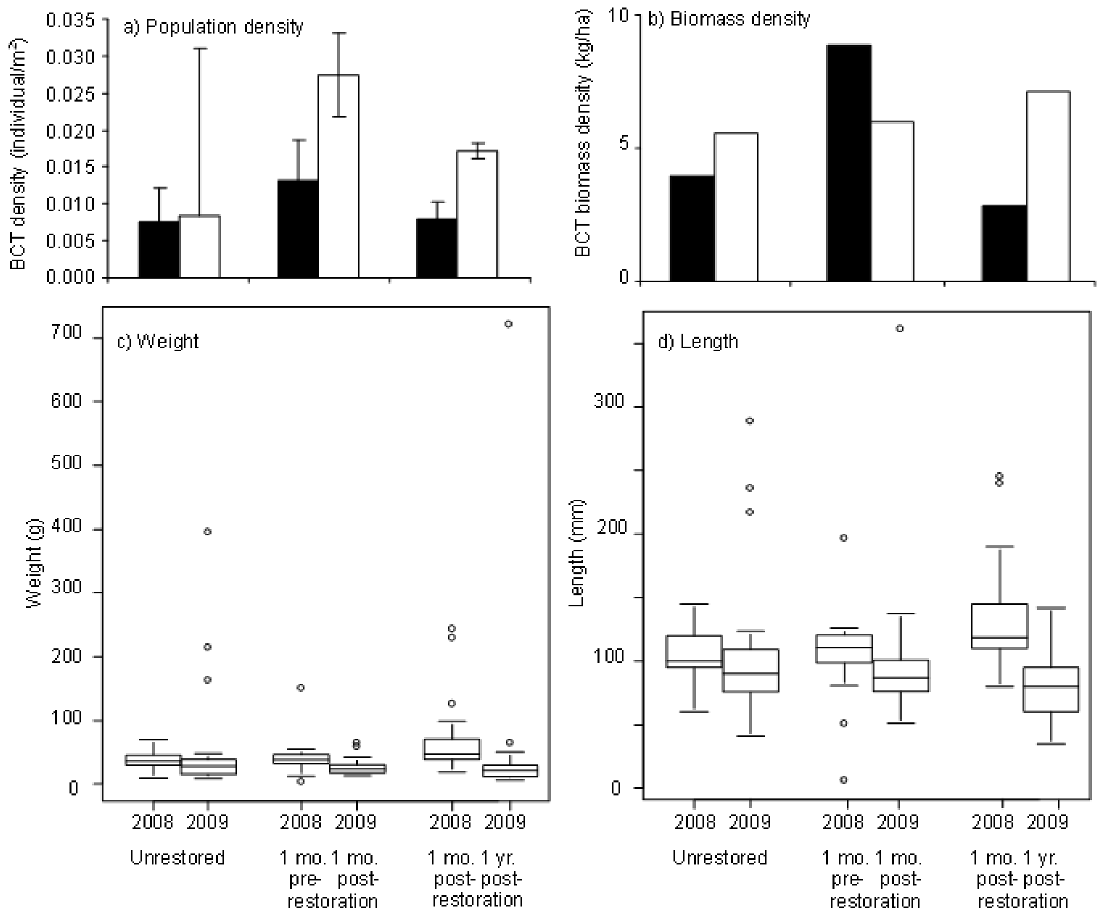
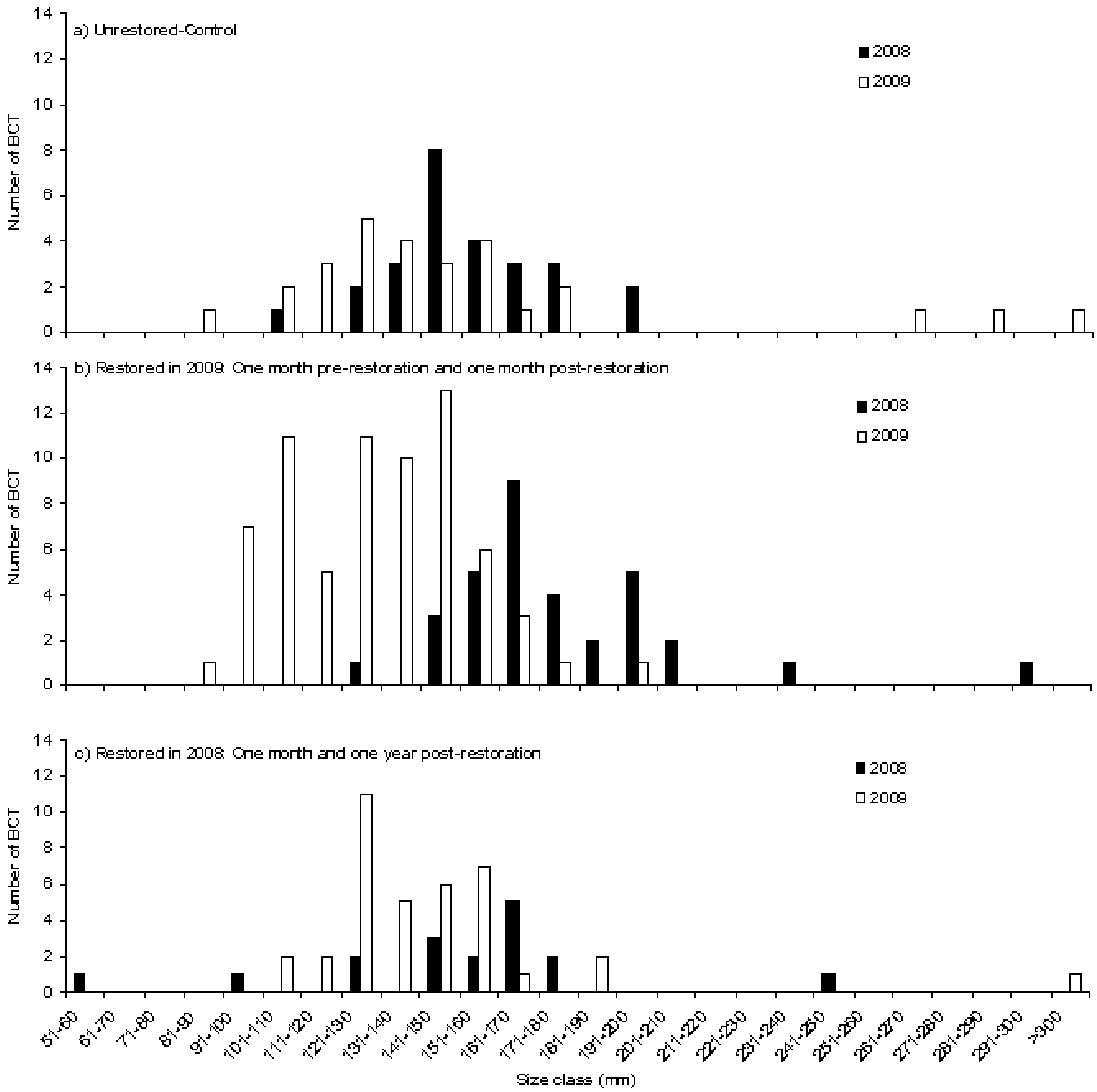


Figure 17: Size class frequency distributions for restored and unrestored reaches, Strawberry River, September 2008 and 2009**.**



Conclusions and future monitoring

We present the framework, methodology, and initial (one year post-restoration) results of an interdisciplinary, multi-metric monitoring program on the Strawberry River, a site of an ongoing instream restoration project typical of many in the Intermountain West. Our program aims to improve the effectiveness of restoration and adaptive management strategies by targeting biologically limiting conditions, monitoring at ecologically relevant scales, and identifying the link between restoration-induced physical change and biological response. In addition to establishing important baseline data for future monitoring, two years of habitat and fish population surveys have produced five main findings: 1) beaver activity can cause measureable physical changes equal to or greater than those following restoration, including an increase in flow depth and pool frequency; 2) BCT habitat quality is limited by a lack of deep water, pools, and instream cover; 3) a reduction in one limiting variable (in this case, cover) results in a large decline in the estimated proportion of high quality habitat according to habitat suitability indices, 4) the estimated proportion of high quality habitat does not increase immediately after restoration ('as-built' conditions), and 5) in order to determine whether apparent increases in BCT populations on restored reaches are biologically significant, it is imperative to incorporate BCT life cycle dynamics, recruitment success, and stocking information (number and ages) into the analysis.

Additional monitoring of this system includes historical aerial photo analysis, high precision geomorphic surveys, water quality and suspended sediment monitoring, bed sediment sampling, and redd surveys. Our continued and comprehensive study of this system will document how physical and biological components of the system adjust over the long-term. Future questions to address, in addition to more cumulative answers to the questions highlighted herein, include:

1. Do rates of lateral bank erosion decrease to equal rates of bar deposition so that the channel maintains a steady-state width and depth?
2. Do constructed channel morphologies and artificial structures persist in form and function over time?
3. What is the fine sediment content of spawning gravels; is this amount detrimental to BCT reproductive success; and does restoration reduce fine sediment loading and infiltration over time?
4. Does the abundance of spawning and resident BCT increase, and does this increase reflect improved habitat quality and/or availability?

In the next section, we use the Strawberry River monitoring program as a case study for demonstrating how these standard techniques can be used in the assessment of restored systems.

References

Bernhardt, E.S., Sudduth, E.S., Palmer, M.A., Allan, J.D., Meyer, J.L., Alexander, G., et al. 2007. Restoring rivers one reach at a time: results from a survey of U.S. river restoration practitioners. Restoration Ecology. 15:482-493.

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

.Bunte, K.; Abt, S.R. 2001. Sampling surface and subsurface particle-size distributions in wadeable gravel-and cobble-bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. Gen. Tech. Rep. RMRS-GTR-74. Fort Collins,CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 428 p.

Dean, R.L., Connell, J.H. 1978. Marine invertebrates in algal succession. III. Mechanisms linking habitat complexity with diversity. Journal of Experimental Marine Biology and Ecology. 109:249-273.

Hawkins, C.P., Kershner, J.L., Bisson, P.A., Bryant, M.D, Decker, L.M, Gregory, S.V., McCullough, D.A., Overton, C. K., Reeves, G.H., Steedman, R.J., and Young, M. K. 1993. A hierarchical approach to classifying stream habitat features. Fisheries. 18: 3-12. doi: 10.1577/1548-8446(1993)018<0003

Heitke, J. D.; Archer, E. J.: Dugaw, D. D.: Bouwes, B. A.; Archer E. A.; Henderson, R. C.; Kershner, J. L. 2008. Effectiveness monitoring for streams and riparian areas: sampling protocol for stream channel attributes. Unpublished paper on file at: http://www.fs.fed.us/biology/fishecology/emp.

Hickman, T., and R. F. Raleigh. 1982. Habitat suitability index models: Cutthroat trout. U.S.D.A. Fish and Wildlife Service. FWS/OBS-82/10.5. 38 pp.

Laasonen, P., Muotka, T., Kivijarvi, I. 1998. Recovery of macronivertebrate communities from stream habitat restoration. Aquatic Conservation: Marine and Freshwater Ecosystems. 8:101-113.

Larson, M.G., Booth, D.B, Morely, S.A. 2001. Effecitveness of large woody debris in stream rehabilitation projects in urban basins. Ecological Engineering. 18: 211-226.

McNeil, W. J. and Ahnell, W. H., 1964. Success of pink salmon spawning relative to size of spawning bed materials. United States Fish and Wildlife Service Special Scientific Report--Fisheries, No. 469 15 pp.

Minshall, G.W. 1984. Aquatic insect-substratum relationships. Pp. 358-400 in V.H. Resh and D.M. Rosenbberg [eds]. The Ecology of Aquatic Insects. Praeger Publishers, New York.

Roni, P., Bennet, T., Morely, S., Pess, G.R., Hanson, K. Van Slyke D., Olmstead, P. 2006 rehabilitation of bedrock stream channels: the effects of boulder placement on aquatic habitat and biota. River Research and Applications. 22: 967-980.

Rosgen, D. L., 1996. "Applied River Morphology", Wildland Hydrology Books, 1481 Stevens Lake Road, Pagosa Springs, Co. 81147, 385 pp.

Scealy, J., Mika, A.S. J., Boulton, A.J. 2007. Aquatic macroinvertebrate communities on wood in an Australian lowland river: experimental assessment of the interactions of habitat, substrate complexity, and retained organic matter. Marine and Freshwater Research. 58: 153-165.

Thurow, R. F. 1994. Underwater methods for study of salmonids in the Intermountain West. U.S. Forest Service General Technical Report INT-GTR-307.

UDWR, 2007. Strawberry River Restoration Phase III Project Proposal, Utah Division of Wildlife Resources.

USDA Forest Service. 2004. Strawberry Watershed Restoration Report. USDA Forest Service, Uinta National Forest, Heber Ranger District, Heber, Utah. April 2004.

Utah Department of Environmental Quality, 2007. Strawberry Reservoir TMDL, Division of Water Quality TMDL Section, July 2007.

Task 3: Monitoring protocols for evaluating the performance and success of BCT restoration strategies

Introduction

In this section, we use the Strawberry River restoration project and monitoring program to demonstrate how standard geomorphic and ecological research techniques can be used to evaluate the outcome of BCT (Bonneville cutthroat trout; *Oncorhynchus clarki utah*) restoration projects. We do not provide an exhaustive synthesis of available methodologies; instead, our goal is to highlight commonly used techniques that we consider to be both practical and instructive. We use the Strawberry River as a case study through which we discuss the design, implementation, analysis, and application of a monitoring program. The Strawberry River monitoring program provides a useful design template because a) its restoration strategies are similar to those used on many BCT-based restoration projects and b) the program is comprehensive, interdisciplinary, and ongoing. However, unique system attributes, restoration techniques, and project objectives must be incorporated into the design of other monitoring programs.

Intended audience

Our monitoring recommendations are intended for those who direct, perform, or prioritize restoration. We expect our audience to have a mixed background of biology, fisheries sciences, fluvial geomorphology or related science. Our recommendations herein are focused on projects aimed at native trout recovery in the Intermountain West, although the structure and approach we present may be applied to other types of restoration projects.

Importance of pre-project monitoring

Pre-project monitoring is an essential component of any monitoring program for three reasons. Firstly, pre-project monitoring helps managers determine whether or not certain restoration measures are needed at all. Perceived problems may be nonexistent, ecologically insignificant, or misinterpreted. In order to choose the most effective restoration technique, the perceived problem and intended effects of restoration must first be accurately identified. For example, if excessive bank erosion is a perceived problem, the existence of rapid erosion must first be confirmed, stabilization techniques then employed, and monitoring used to determine if restoration was effective at reducing erosion. But if assessments indicate no problem with erosion, bank stabilization techniques are likely unnecessary and other strategies may need to be pursued. Secondly, monitoring provides an understanding of the natural variability of the system. Short-term changes observed after restoration need to be placed in the long-term context of the system to determine if they are natural or restoration-induced. Thirdly, monitoring prior to implementation of a restoration project establishes a baseline against which post-project conditions can be compared.

Factors contributing to monitoring design

Three primary factors influence the design of a monitoring program:

1. Objectives of analysis – What is the purpose of a post-project assessment?
   a. To inform adaptive management or future restoration (i.e., what activities to change, add, or remove)
   b. To determine if project met expectations of public or other funding agency

In the case of (a), pre- and post-project monitoring should be designed to assess the functionality and sustainability of system attributes and the potential for improvement or degradation; we consider this to be the 'performance' of a restoration project; i.e., did restoration strategies and techniques function or perform as expected? The monitoring should be designed to facilitate the evaluation of outcomes of restoration activities (positive or negative), trajectories of change, whether techniques effectively achieved their intended purpose, if additional measures are needed or if certain actions should not be repeated.

Monitoring for (b) targets specific goals of a restoration project, mandated or promised at the outset. Assessments will measure outcomes and gauge whether certain criteria were met (e.g., fish population abundance or water quality standards), rather than appraise the performance of certain techniques. System conditions should be evaluated, without necessarily addressing the effectiveness of techniques used to achieve those conditions.

Our monitoring recommendations herein target project performance: the ability of restoration techniques to perform as intended and the effectiveness of those strategies to achieve project goals. Restoration practitioners must recognize, however, the potential for disconnect between performance and restoration goals; successful performance of a technique may not necessarily produce the desired outcome. Continuing with the example from above, bank stabilization techniques may be employed to decrease bank erosion, ultimately intended to reduce fine sediment loads to spawning gravels. But if fine sediment in the system is due to other factors, such as agriculture or logging in the watershed, a decrease in bank erosion will have limited effect. A thorough pre-project analysis may be required to accurately determine the causes of system degradation; doing so will greatly improve the efficiency and effectiveness of restoration.

2. Restoration design – What are the techniques and goals of a restoration project?

Monitoring techniques will depend on restoration strategies and the variables being manipulated. Restoration can involve a wide and varied spectrum of techniques, ranging from fish passage to physical construction to disease control. In this report, we focus on strategies of active instream habitat construction aimed at increasing BCT populations, such as those used on the Strawberry River, because of their ubiquity among BCT-based restoration projects. Many of these habitat restoration projects employ techniques to enhance physical heterogeneity (e.g., structure placement, pool-riffle construction), with the intention of consequently increasing BCT abundance. However, in some cases BCT abundance may not be limited by habitat quality or availability; other factors, such as disease or fish passage, could be to blame. Thus if increased BCT abundance is the ultimate goal of restoration, we strongly recommend a careful and deliberate pre-project analysis to identify the variables limiting BCT populations. Restoration strategies and pre-project monitoring should be designed to explicitly address these limiting factors. In this report, we review techniques for the assessment of changes to potentially limiting physical conditions; determination of abundance-limiting factors often require detailed population modeling analyses that are beyond the scope of this report (e.g., Budy and Schaller 2007 {and see references within}). We focus on the links between restoration actions and physical change, rather than biological response. However, we briefly review methods for assessing BCT population abundance in Appendix 2.

3. Temporal and spatial scale – What is the necessary frequency and scope of monitoring?

The breadth and frequency of monitoring efforts depend on the temporal and spatial scales of variables being measured. Strategies aimed at improving fish passage, for example, may require monitoring over long distances (kilometers) but relatively short time periods (seasonally), while strategies for revegetation may take years to perform but occur over a small area. Previous knowledge of rates of change, process scales, and natural variability must be incorporated into monitoring design to document meaningful change with minimal effort.

Monitoring for BCT recovery: case study of the Strawberry River

We begin by outlining two restoration strategies and associated objectives monitored on the Strawberry River. For each strategy, we then recommend techniques for evaluating pre-project conditions, the performance of each strategy, and the condition of the system relative to the project goal (i.e., what is the existing problem, whether or not the strategy performed as expected, and whether the project objective was achieved). We provide two levels of monitoring intensity: 1) minimum (limited resources) and 2) comprehensive. The level of monitoring chosen will fall somewhere between these extremes d logistical and financial constraints. For each technique, we list a set of pros and cons (including tradeoffs between effort and information provided), the relevant variables or processes to monitor, and the scale of monitoring (distribution, frequency), and a summary recommendation for implementation of each method.

Restoration techniques and associated goals

We focus our monitoring on two common strategies used by BCT restoration projects (bank stabilization and morphological construction), evaluating pre-project conditions, restoration strategies, expected performance, and associated goals. We recommend devising explicit questions related to each component (i.e., performance, strategy, and goal). We then suggest techniques for addressing each question. In the following sections, we make specific recommendations based on the approach and results from the Strawberry River study.

I. *Strategy*: Bank stabilization

Perceived problem: Excessive bank erosion contributing to high fine sediment in bed material

Performance: Reduced lateral streambank erosion

Goal: Reduced fine sediment and improved spawning habitat

II. *Strategy*: Structure placement and morphological construction

Perceived problem: Limited availability of suitable habitat for adult and juvenile BCT

Performance: Increase frequency and depth of pools and amount of instream cover

Goal: Improve and increase the complexity of aquatic habitat

Monitoring recommendations

I. Bank stabilization

Q1. Perceived problem: Three questions are required to address the perceived problem of rapid bank erosion contributing to high fine sediment contents of spawning gravel.

a. Are banks eroding at an accelerated (non-equilibrium) rate?
b. Do spawning gravels have fine sediment contents greater than unimpaired conditions or established thresholds for fish?

c. Are banks the source of fine sediment to spawning gravels?

Q2. Performance: Did bank stabilization techniques reduce rates of lateral bank erosion?

Q3. Goal: Did a reduction in bank erosion rate reduce the fine sediment content of spawning gravels?

A. Bank erosion rates (*Q1a, Q2*)

Similar techniques can be used to quantify both historic and recent erosion rates as a means of assessing whether erosion rates were high prior to restoration (*Q1a*) and whether restoration reduced erosion rates (*Q2*) (Table 1). Suitable techniques will depend on the spatial and temporal scale of study. Remote techniques and historical analyses will improve the study of large-scale systems or long time periods. Conversely, changes occurring rapidly over a short time frame or limited to a small area may not be detected if measurements are too dispersed in space or time. We recommend techniques for both large and small-scale analyses; it is essential that the practitioner match the appropriate method with the scale of the question, area, and system under study.

Frequently, bare or vertical banks are interpreted as evidence of excessive bank erosion. However, lateral erosion and channel migration are natural processes and may not be occurring in response to disturbance. Nevertheless, bank erosion may be a problem if recent erosion rates are higher than in the past, the channel is eroding on both sides, or if erosion rates exceed deposition rates, resulting in unnatural rates of channel widening or incision.

Table 12: Summary of recommended techniques for quantifying bank erosion rates

Technique	Variable/Process	Scale/Timing	Pros	Cons
Aerial photography	Bank erosion rate Planform metrics	Wide distribution (watershed) Low frequency (yearly to decadal)	Provides data over large scale and long time period	Sufficient photo records not commonly available Only provides 2D information at coarse scale May require specialized training
Recommendation	Use spatial analysis tools to compute historic and recent channel migration rates and planform statistics from available photographic records; establish and/or continue photography pre- and post-restoration			
Repeat cross-section surveys	Bank erosion rate Channel morphology	Local or reach scale High frequency (1-2 times per year)	Low cost Minimal training	Localized changes Requires high density measurements
Recommendation	Establish permanent cross-sections for repeat surveys to document changes in channel morphology and location			
Erosion pins	Bank erosion rate	Local or reach scale High frequency (2-5 times per year)	Low cost Minimal training Direct measurement Vertical variability	Localized changes Potential for disturbance, loss, hazard Potential for reinforcement of banks (bias)
Recommendation	Install pins at a high density within representative reaches and measure throughout year to calculate direct erosion rates and spatial (lateral, vertical) variation in erosion			

i. Minimal approach: Aerial photography and channel cross-sections

Any historical photographs or maps that are available should be used in an analysis of bank erosion rates. An aerial photo record with several years of identical coverage is ideal; if possible, a photo record should be established and continued before, during, and after restoration. Aerial photo and map records can be visually assessed for evidence of channel change (more quantitative methods are described below). To begin, find unchanging landmarks on successive photos in order to directly compare channel shape and location between years. If discernible, calculate basic planform metrics such as width and sinuosity for each year. Depending on the area covered and the length of record, calculate changes in width and sinuosity between time periods for different sections of river. Track the location of the channel over time; a rough estimate of channel migration rate can be calculated from the distance between channel locations and the time elapsed.

Rapid erosion or migration relative to historic rates or channel widening over time may indicate currently accelerated erosion rates. Aerial photos are thus useful for documenting long-term changes over a large spatial scale; however, they are not commonly available, may require special training to analyze, and only provide two-dimension information (i.e., no information about channel depth)

Repeat surveys of channel cross-sections can provide information about channel shape not available from aerial photos, although at a more limited spatial scale. Low-cost standard survey techniques can be used to measure channel morphology at permanent locations; repeated over time these measurements can provide evidence of bank erosion and other types of morphological change. Endpoints of cross-sections should be monumented and flagged so that exact locations can be resurveyed in subsequent years; semipermanent structures should be established as local benchmarks. Each cross-section should show the benchmark elevation and location, terraces, floodplain, natural or artificial levees, left and right edge of water (at various water stages), thalweg, and variation in channel shape. Instructions for establishing and conducting cross-section surveys can be found in several sources (e.g. Harrelson et al., 1994; Rosgen, 1996). When possible, surveys should incorporate historic cross-sections such as abandoned gage station surveys or surveys from previous studies. For instance, gage station records from the U.S. Geological survey are readily available for many rivers; a detailed guide to obtaining these records and using them for historic analysis can be found in Smelser and Schmidt (1998). Repeated surveys can be used to quantify changes in channel width, depth, bed elevation, and location, as well as changes in alluvial features such as bars, benches, or banks. Bank erosion rates can be calculated from an extended period of surveys by the change in bank location over time. Similarly, bed incision or aggradation can be assessed from changes in bed elevation. Cross-section surveys are relatively low-cost and require minimal specialized training. However, information about channel change is localized to the cross-section and may not reflect reach- or watershed-scale processes. A high density of cross-sections along a river segment (e.g., one cross-section per length of wetted width) and across a range of morphological units is needed to provide information about reach-scale changes in morphology.

ii. Comprehensive approach: Aerial photography, high precision channel cross-sections, erosion pins

Spatial analysis tools can be used for a more rigorous analysis of historic and recent aerial photographs. Digitized and georeferenced photographs from single or multiple time periods can be analyzed using geographical software such as ArcGIS. We recommend two recently developed approaches for quantifying channel widths and lateral migration rates. In the method developed by Constantine et al. (2009), digitized thalweg centerlines from two years are intersected to create polygons that represent the area flooded over the time period. A minimum average migration rate is calculated by dividing the polygon area by half its perimeter and time elapsed. A free planform software tool developed by J.W. Lauer at the National Center for Earth Surface Dynamics (NCED) requires only the digitization of the two channel banks, from which the software calculates the channel centerline, width, and curvature at evenly spaced points along the channel and the lateral distance between two points in time. Software downloads and details of the NCED planform statistics tool can be found at the link: http://www.nced.umn.edu/content/tools-and-data.

On-site measurements of erosion rate can supplement remote techniques by providing measurements at a smaller scale and in the vertical plane. Cross-section surveys as described above can also be conducted using GPS tools that provide absolute elevations and locations and precise repeat measurements. GPS surveys can then be spatially linked to georeferenced aerial imagery to compare estimates of erosion rate.

Example: Strawberry River channel migration. Monitoring on the Strawberry River includes a combination of aerial photography analyses with GPS-surveyed channel cross-sections to estimate bank erosion rates in portions of the watershed over different time periods. Migration rates were calculated using image analysis software (ArcMap, ArcView, Imagine) for ten-year periods of a photo record dating back to 1938. Lateral migration was most rapid from 1953 to 1963, coinciding with herbicidal removal of *riparian willows. Migration has been minimal since 1993 and restricted to upstream reaches, suggesting that bank erosion is not currently affecting the system but legacy effects from historic erosion may persist.*

Finally, direct estimates of erosion rate can be made using erosion pins. In this method, a metal rod is hammered horizontally into the steep face of an eroding bank until it is flush; two or more may be installed along a vertical profile for information about vertical variability in erosion rate; pins should be spaced ~1-5 m apart along the bank. The length of pin exposed is measured after each flood, rainfall, and/or frost event and then hammered back into the bank, indicating the amount of bank eroded in the time elapsed. Although the most direct method of measuring erosion rate, pins can be disturbed or lost and present possible hazards for livestock or humans. Under some conditions, pins may also reinforce bank material and limit erosion, potentially biasing measurements.

B. Fine sediment content of gravels (*Q1b, Q3*)

By directing measuring the fine sediment content of bed material, we can assess whether fines pose a problem to the quality of spawning habitat (*Q1b*) and whether restoration activities have reduced fine sediment infiltration (*Q3*). Before selecting an approach and technique, however, it is important to define the size and quantity of particles relevant to the target organism. A comprehensive review of techniques for measuring and analyzing the quality of spawning gravels is provided by Kondolf (2000); here we focus on the problem of interstitial clogging by fines (Table 2).

For decades, fish biologists have used the percentage of particles (by mass) < 0.83, 4, or 10 mm to assess the quality of spawning gravels (e.g, McNeil and Ahnell 1964, Tappel and Bjornn 1983, Kondolf 2000). Most studies show that interstitial particles < 1 mm reduce gravel permeability and slow intragravel flow, in turn restricting the supply of oxygen to embryos and the removal of metabolic wastes; particles 1-10 mm may block fry emergence (Everest et al., 1987). According to the habitat suitability index for cutthroat trout (Hickman and Raleigh, 1982), gravels with more than ~30% fines will reduce survival of embryos and fry; optimal spawning substrates contain only ~5% fines and gravel particles ~1.5-6.0 cm. Interpretation of fine sediment contents can also be made by comparison with previous studies of embryo survival or emergence requirements (e.g., Kondolf et al. 2000 Table 1; Kondolf et al. 2008 Table 2). Studies show that finer particles have negative effects at much lower percentages; for instance, 50% emergence occurs at 7.5-21% for particles < 0.83 but at 15-40% for particles < 6.35 mm.

Finally, predictive and quantitative assessments of salmonid egg or embryo survival rates could be computed from functions relating survival to the percentage of fine sediments of various sizes (e.g., < 9.5, 6.35, or 0.85 mm) (e.g., Tappel and Bjornn 1983; Lisle and Lewis 1992; Wu and Wang 2000; McHugh et al. 2004), but these predictions may be inaccurate without calibration for the species of interest or reference to natural system variability (e.g., embeddedness and percent fines may be naturally high). Negative relations between embryo survival and the percentage of fine particles has been demonstrated in numerous studies (e.g., Julien and Bergeron, 2006; Levasseur et al. 2006).

Table 13: Techniques for measuring fine sediment infiltration and supply.

Technique	Variable/Process	Scale/Timing	Pros	Cons
Subsurface samplers (MacNeil, pipe, or flow isolation cell)	Percent bed fines Subsurface particle size distribution	Riffle or reach-scale Several times per year	Low cost/expertise Large sample sizes Actual substrate conditions	Loss of fines Require wadeable conditions
Recommendation	Bulk sample riffle substrates at different times of year to assess spatial and seasonal variability in fine sediment content using techniques to minimize loss of fines			
Freeze-core samplers	Percent bed fines Subsurface particle size distribution	Riffle or reach-scale Several times per year	Minimal loss of fines Actual substrate conditions	High cost/expertise Small sample sizes Misrepresents coarse fraction
Recommendation	Pair freeze cores with bulk samplers to supplement information and correct biases			
Sediment collectors or infiltration traps	Fine sediment supply to bed	Riffle or reach-scale Several times per year	Easy to install Inexpensive No loss of fines Measures supply rate	Relative comparisons Sediment exposure, not actual infiltration Biased by contained gravel composition
Recommendation	Use throughout year in multiple locations to assess accumulation rates and compare sediment supply (amount and composition) between sites			
Suspended sediment monitoring	Suspended sediment concentration in flow	Reach-scale Continuous to monthly	Measure of fine sediment supply and transport	Not direct measure of substrate conditions
Recommendation	Measure during periods of high flow to assess supply and transport and in combination with sediment samples or traps to evaluate relation between suspended sediment transport and infiltration			

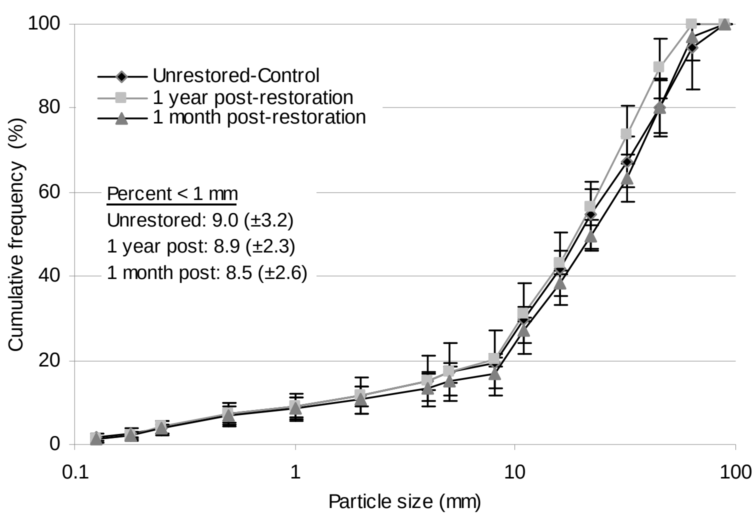
i. Minimal approach: Subsurface bulk samplers

Several relatively low-cost, low-expertise samplers exist for collecting bed sediment samples. Selection of an appropriate sampling method should consider flow conditions, man-power, budget, and study question; a thorough and detailed review of available techniques is provided in Bunte and Abt (2001), while Kondolf et al. (2008) summarizes techniques specific to assessing spawning gravels. For assessing fine particle infiltration with limited resources, we recommend using a bulk subsurface sampler such as a MacNeil sampler (MacNeil and Ahnell, 1964; Tussing, 2008) or flow isolation cell and shovel (Zimmerman and Lapointe, 2005). Both methods are low-cost, simple, moderately labor-intensive, produce large samples, and provide information on bulk (rather than just surface) particle size distributions; easier pebble counts only measure surface conditions, not subsurface characteristics relevant to spawning. A disadvantage of these samplers, however, is the potential loss of fine particles during removal; in addition, MacNeil and pipe samplers are unable to sample particles larger than their opening, limiting their use in large cobble- to boulder-bedded systems. Fine particles suspended from the bed during sampling should be collected by either filtering the water in the field, subsampling a specified volume of the suspension, or allowed to settle and collected with the coarser fraction. When using a MacNeil sampler, a cap can be used to seal the water within the sampler and all suspended material can easily be collected (MacNeil and Ahnell, 1964).

For the purpose of this question, samples can be restricted to riffle substrates (where BCT are likely to spawn). Number of riffles within a study reach and individual samples on each riffle will depend on riffle spacing, width of stream, bed particle size, and sampler size. High variability among riffles within a study reach will necessitate greater numbers of samples to detect significant differences between treatment reaches. Samples from a single riffle may need to be combined into a composite sample for analysis if samples are small relative to the largest particle size. A simple and widely accepted rule for sample size is that the largest particle should not be >1 % of the total sample weight (Church et al., 1987). Bulk sampling methods are only feasible at wadeable flow levels, but should be spaced temporally to capture seasonal variation in fine sediment content (e.g., fluctuations with seasonal use or flow regime) and biologically relevant events (e.g., incubation, emergence). Once collected, samples should be sieved into standard size fractions; we recommend sorting and weighing fractions > 8 mm in the field and processing finer fractions in the laboratory, minimizing the amount of material that must be moved. Field processing can be done with a set of bucket, rocker or hanging sieves, a tripod, and spring-loaded scale for particles 8-32 mm, and a gravelometer for particles > 32 mm; water retained on these larger particles adds a trivial mass relative to their weight. Return particles < 8 mm to the laboratory to be oven-dried, sieved, and weighed into standard size fractions, including fractions < 1, 1-2, 2-4, and 4-8 mm. From this data, the percentage of different fine particle fractions (e.g., < 1 mm) can be calculated and assessed for each riffle or reach.

Example: Strawberry River fine sediment content. MacNeil samples were collected from riffle substrates in the fall of 2009 to determine the sediment content of gravels following a long period of low flow (July – October) and the conditions experienced by emerging BCT fry (assuming little sediment mobilization between August and October). Two evenly spaced samples were collected from three riffles on three 500 m reaches, representing three stages of restoration status (restored 1 year prior, restored 1 month prior, and unrestored). Suspended particles were collected by capping the MacNeil sampler and collecting all water and sediment in a bucket; the material was then allowed to settle, the water was filtered through a very fine (4 micron) mesh, and all particles < 8 mm were returned to the laboratory for processing. Particles > 8 mm were sieved and weighed in the field using bucket sieves and a spring-loaded scale on a tripod. Particle size distributions and percent fines indicate similar and suitable spawning gravel conditions (percentage of particles < 1 mm ranges between 5-14%) on all reaches (Fig. 18), being less than the 30% deemed unsuitable by the cutthroat HSI though slightly more than the 5% fines considered optimal. Sampling will be repeated in June 2010 to determine conditions experienced by spawning BCT and assess seasonal variability in fine sediment content.

Figure 18: Particle size distributions of riffle substrates on unrestored (control) and restored reaches of the Strawberry River. Values are the mean (± standard deviation) of composite samples from three replicate riffles on each reach.



ii. Comprehensive approach: Bulk samplers, freeze cores, sediment collectors, and suspended sediment monitoring

As described above, bulk samplers provide information about subsurface particle size distribution, but can underestimate fine sediment content if suspended particles are lost during collection (Zimmerman et al., 2005). Freeze core samplers can more precisely measure fine sediment, but imprecisely sample coarse sediment due to small sample sizes (Rood and Church, 1994), physical displacement during installation, and misrepresentative attachment of coarse grains to the core (Klingeman, 1987; Petts and Thoms, 1992). Freeze core and bulk samples can thus be paired and combined to produce a composite particle size distribution (Fripp and Diplas, 1992, 1993; Rice and Haschenburger, 2004) that accounts for the biases introduced by each method. Freeze cores are costly and require expertise, but are also quick to collect, precisely estimate fines, and can be segmented into layers for information about substrate variability with depth. If resources allow, we recommend using freeze cores in combination with bulk samples whenever possible to avoid potentially large biases (Zimmerman et al., 2005).

Sediment collectors or infiltration traps provide a means of assessing fine sediment loads to spawning gravels (e.g. Carling, 1984; Lachance and Dube 2004). Previously used traps vary in size and construction, but most are composed of a mesh container filled with clean, finely graded gravels. Trap structure and gravel composition differs from natural substrates; as a result, traps measure the amount and composition of fine sediment entering the bed, not actual subsurface conditions. Traps should not be used to estimate the fine sediment content of substrates encountered by spawning fish or infiltration rates into natural gravels (Zimmerman and Lapointe, 2005), but can be used to compare between sites or evaluate controls on sediment accumulation, such as suspended sediment or bedload transport (e.g., Carling, 1984; Acornley and Sear, 1999). Infiltration traps can be used to assess fine sediment supply to the bed under different conditions (e.g. restoration status, season, flow level). Several types of traps are available, but the bucket type used by Zimmerman and Lapointe (2005) is relatively simple, inexpensive, and effective for relative comparisons. Gravel sizes used in the trap will influence infiltration rate (Frostick et al., 1984) and should be carefully selected; we recommend using a gravel composition similar to the size distribution of the natural substrate (determined from bulk samples) but constant between sites so that comparisons are not biased. Traps should be installed and removed throughout the year to assess rates of accumulation during periods of different flow levels or biological processes; for example, installation during spawning and removal at hatching or emergence will provide information on sediment accumulation during the critical egg incubation period. In contrast, installation during incubation would provide less meaningful measurements, being unable to capture the sediment that accumulated during the early stages of incubation.

Although not a direct measure of substrate condition, suspended sediment measurements provide information about sediment supply, transport, and deposition. When spatially and temporally linked to bed samples or sediment traps, suspended sediment monitoring helps explain the factors influencing infiltration; infiltration will be greatest when supply is high but transport out of the system is low. Most suspended sediment transport occurs during snowmelt periods and single floods; transport is strongly related to discharge and often exhibits hydrograph hysteresis, with concentrations at a given flow on the rising limb different than the corresponding flow on the falling limb. Patterns of hysteresis in the relation between suspended load and water discharge are related to types and locations of active sources (Lenzi and Marchi, 2000; Nistor and Church, 2005). For instance, lower concentrations on the falling limb (clockwise hysteresis) indicate depleted sediment supplies, while high concentrations on the falling limb (counterclockwise hysteresis) may reflect an addition of sediment such as a bank collapse. Concentrations may be highest during lower flows following peak snowmelt as sediment sources become exposed. Patterns in suspended sediment thus reflect source and supply mechanisms and how much sediment is retained in the system. Suspended sediment concentrations can be measured from water samples collected manually or automatically; automatic samplers (http://www.isco.com/) permit continuous measurements and sampling during high flows when wading is not feasible. Optical backscatter sensors are used extensively to monitor turbidity as a surrogate for suspended sediment; the relationship between turbidity and suspended sediment must be developed from numerous samples at various flow levels.

Regardless of method used, samples should be taken frequently (e.g., multiple times per day) during the rising and falling limbs of high flow events, when discharge and sediment concentrations change rapidly. Less frequent (e.g., daily or weekly) sampling is needed during low flow periods when sediment transport is minimal. Substrate sampling should occur throughout the period of suspended sediment monitoring to link concentrations and rates of infiltration (Zimmerman and Lapointe, 2005).

C. Linking bank erosion and bed sediment (*Q1c*)

High rates of bank erosion may increase sediment loads and subsurface fines, but explicitly linking these two processes requires additional information. In some cases, eroded bank material may be transported out of the system before significant infiltration can occur or after bed mobilization and sediment flushing; in other cases, fine sediment contents may be high despite limited bank erosion due to supply from other parts of the watershed. Furthermore, evidence of both rapid bank erosion and excess bed fines does not prove they are related. Thus restoration measures to stabilize banks may not reduce fine sediment loads or improve spawning gravels. Determining whether banks are the source of bed sediment may require a comparison of sediment composition between banks, bed, and transported material, as well as an evaluation of other sediment sources in the watershed. A comprehensive review of common approaches to source identification and the problems associated with each is provided most recently by Collins and Walling (2004); discussion of these methods is beyond the scope of this report. Although establishing the link between bank erosion and bed composition is essential to determining causation, it may not be necessary if bank stabilization is an independent goal of restoration. However, if a reduction in bed fines is not achieved despite bank stabilization, additional study is needed to determine the source of fines or whether this is a natural condition of the system.

II. Structure placement and morphological construction

Q1. Perceived problem: We ask two related questions regarding the perceived problem of limited suitable habitat for juvenile and adult BCT:

a. What are the characteristics of suitable juvenile and adult BCT habitat?
b. Are suitable physical conditions used in proportion to their availability?

Q2. Performance: Did structure placement and pool construction increase pool depth and frequency and the amount of instream cover?

Q3. Goal: Did morphological changes improve and increase the complexity of aquatic habitat?

A. Assessing limiting factors, habitat suitability, and habitat complexity (*Q1a, Q1b, Q2, Q3*)

Structure placement and channel construction are designed to improve habitat for juvenile and adult BCT, based on knowledge of BCT habitat needs. By synthesizing the results of previous studies, the cutthroat trout habitat suitability index identifies physical conditions considered suitable for each life stage and thus provides a initial framework for habitat improvement. An increase in physical heterogeneity (often termed 'habitat complexity') is generally expected to promote BCT success by addressing the diverse habitat requirements of each life stage. However, restoration may be more effective and efficient if BCT habitat use relative to availability is first measured, so that suitability and limiting conditions can be identified for a given system (Table 3).

Table 14: Techniques for assessing habitat suitability and complexity

Technique	Variable/Process	Scale/Timing	Pros	Cons
Unit surveys	Number and dimensions of distinct morphological units and structures; bed and bank conditions Habitat complexity	Reach-scale Once or twice per year	Low cost/expertise Low time investment	Subject to observer bias Highly approximate and qualitative
Recommendation	Visually identify distinct habitat units and measure unit dimensions and bank conditions along a specified study reach; compute metric of habitat diversity			
Snorkel and habitat availability surveys	Point-scale physical conditions of BCT locations Distributions of	Point, cross-section and reach scale Once or twice per	Low cost/expertise Provides detailed, quantitative, multi-scale, and multi-	Time-consuming May underestimate fish use

	physical variables Limiting variables Habitat quality Habitat complexity	year	metric data Can be used in a variety of metrics	
Recommendation	Compare use and availability of physical variables to categorize and quantify the proportion of habitat quality classes (optimal, suitable, and unsuitable) before and after restoration			
Longitudinal profile	Longitudinal variability in bed elevation and flow depth Average water surface slope Maximum pool/riffle depth Pool/riffle slopes Riffle-pool spacing Pool/ riffle frequency Habitat complexity	Reach-scale Once or twice per year (high and low flow)	Provides detailed, quantitative data Can be used in computer modeling Provides a variety of morphological data Low cost if surveyed with level	High cost/expertise if surveyed with GPS Does not provide information about lateral variability, hydraulic conditions, substrate, instream structures, or bank conditions
Recommendation	Use to assess longitudinal unit variability; link with cross-section surveys to describe channel morphology in three dimensions			
Morphological maps (three methods):	Areas and locations of morphological units, structures, riparian vegetation			
(1) High density cross-section surveys	Cross-section morphology	Unit and reach scale Once or twice per year	Low cost/expertise if surveyed with level	Incomplete coverage Requires high density of cross-sections (may be time consuming) High cost/expertise if surveyed with GPS
(2) High resolution aerial photos	Planform morphology Vegetative cover Locations/sizes of alluvial features and structures	Watershed and reach scale May provide long-term/historic data	Uniform spatial coverage	Certain features may not be detectable depending on quality of photos
(3) LiDAR	High resolution three-dimensional morphology Vegetative cover Locations/sizes of alluvial features and structures	Particle to watershed scale Frequency dependent on available resources, at least once before and after restoration	Detailed, three-dimensional data Can be used in hydrodynamic, habitat suitability, and sediment mobility models	High cost/expertise Data may be too detailed
Recommendation	Construct morphological maps using one or more of the three methods, depending on available funds and resources; apply LiDAR only if spatial modeling is necessary			

i. Minimal approach: Cutthroat habitat suitability index and habitat unit assessment

The cutthroat trout suitability index (HSI; Hickman and Raleigh, 1982) can be used to assess the suitability of existing physical conditions, including temperature, depth, velocity, instream cover, and substrate. Structural physical conditions (depth, velocity, cover, and substrate) can be assessed quickly and easily by a survey of morphological habitat units that have distinct physical characteristics. For each morphological unit within a section of river, a surveyor can measure unit dimensions, assess substrate and bank conditions (e.g., substrate size, percent bare bank, and percent overhanging vegetation), and count instream structures. Units can be visually identified using qualitative assessments of depth, velocity, and flow structure, although these factors will vary with discharge and should be assessed throughout the year. Although many detailed classification schemes exist for reach and unit types (e.g., Montgomery and Buffington, 1997), common morphological units relevant to fish habitat include riffles, pools, glides and runs. Descriptions of these unit types can be found in several sources (e.g., Hawkins et al., 1993; Ramos, 1996), which typically provide ranges of depths and velocities for each unit type. However, visual habitat designation is subject to observer bias and should be verified when possible with depth and velocity measurements or by a second observer.

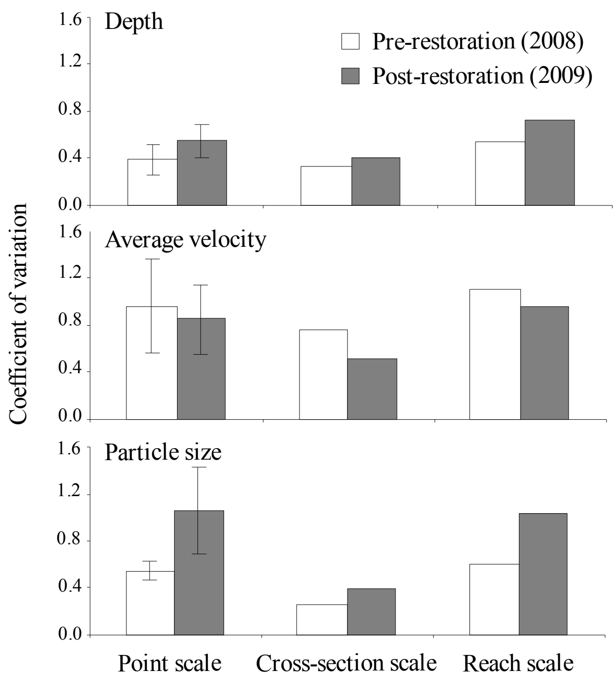
Along each river section, unit types should be counted and dimensions measured, including thalweg length, up- and downstream width, average depth, and pool maximum and tailout depth (Heitke et al. 2008); the length of bare bank, overhanging vegetation, and undercut bank should be recorded and substrate conditions should be visually assessed. Number and residual depths of pools can be compared to habitat suitability requirements of the cutthroat HSI to assess whether depth or pool frequency are limited; amount of overhanging vegetation or instream structures can be used to roughly calculate the reach-scale percentage of overhead cover for comparison with the HSI requirements; and the amount of suitable substrate type can be assessed (see Task 2; Table 2). Habitat complexity can be calculated with a diversity index (e.g., Shannon diversity index) using the number of different unit types. Although approximate, relatively qualitative, and subject to observer bias, this survey will at least indicate what proportion of available habitat is suitable and which structural variables are likely to be most limiting to BCT success.

ii. Comprehensive approach: Snorkel and habitat availability surveys, longitudinal profile, reach morphology maps, and LiDAR

Snorkel and habitat availability surveys allow for the measurement of BCT habitat use and the distribution of physical variables along a reach. A detailed methodology for these surveys can be found in the preceding section of this report (Task 2, Methods: *Habitat availability and morphological complexity* and *Snorkel surveys*). In habitat availability surveys, point measurements on evenly spaced cross-sections measure the distribution of physical parameters laterally across the channel and longitudinally along the reach. Snorkel surveys record the locations of BCT individuals; point-scale measurements at each location characterize the physical conditions used by BCT. Availability can then be compared with BCT use by plotting the distribution of physical parameters from cross-sections with the distribution of parameters from BCT locations (e.g., Task 2, Fig. 11). Preference for certain physical conditions can be determined from BCT use, assuming that the habitat is not saturated and BCT are not forced to use less preferential conditions. Conversely, if BCT densities are low, high quality habitat may be left unused. Nevertheless, physical conditions used in greater proportion to their availability can be considered limited, even if other (i.e., non-habitat) factors limit BCT population abundance. Habitat use surveys can also be used to develop composite microhabitat suitability indices by the method described in the preceding section (Task 2, Methods: *Snorkel surveys*); applied to habitat availability data, these indices can be used to compute the proportion of different levels of habitat quality (e.g., ‘optimal’ or ‘unsuitable’) before and after restoration. Finally, spread in the availability distribution of each parameter (e.g., depth, velocity, substrate, and cover) provides a measure of geomorphic variability at the point, cross-section and reach scale (see example below)

Example: Strawberry River physical heterogeneity at three scales. Point measurements of depth, velocity, and substrate on evenly spaced cross-sections along three reaches can be *used to determine physical variability at the point, cross-section, and reach scale. We calculated the coefficient of variation (standard deviation divided by mean) between points on a cross-section, cross-sections on a reach, and reaches (using all measurement points) as a measure of habitat variability. At all scales, the variability in depth and particle size increases immediately after restoration, but the variability in velocity decreases (Fig. 19), possibly due to pool dredging (increasing maximum depths) and boulder placement (increasing maximum particle size).*

Figure 19: Coefficient of variation between points, cross-sections, and reaches for three physical parameters before and after restoration. Point-scale values are the mean ± standard deviation of 20 cross-sections on each reach.



A longitudinal survey of thalweg bed elevation, or longitudinal profile, can also be used to assess morphological complexity. Like cross-section surveys, a longitudinal profile should be tied into a semi-permanent benchmark for repeat measurements and can be surveyed with a level or high-precision GPS. Bed and water surface elevations along the thalweg should be measured at each distinct change in bed morphology, including all high and low points. Data obtained from a profile include: average water surface slope, maximum pool/riffle depth, pool/riffle slopes, riffle-pool spacing, and pool/ riffle frequency. Habitat complexity can be assessed from the number of distinct morphological units (frequency) and the variability in depth.

Reach morphology maps show all distinct units (size and location), instream structures, riparian vegetation, and alluvial features (e.g., bars, islands). Maps may be constructed from low-flying oblique or high resolution aerial photos, high density channel cross-section surveys, or high resolution optical remote sensing technology such as LiDAR. Once constructed, maps can be used to compute an index of unit diversity (e.g., Shannon diversity index) and the aerial coverage of unit types, instream structures, or vegetation. Photos and LiDAR imagery provide uniform spatial coverage, while cross-sections are limited by their spatial density. Cross-sections and LiDAR also provide data in the vertical dimension, which can be used in three-dimensional mapping. Cross-section surveys are relatively inexpensive and simple, but provide coarse spatial data. In contrast, a LiDAR scan provides extremely high resolution data, but can be expensive and difficult to utilize. LiDAR imagery can be applied to two-dimensional hydrodynamic models, habitat suitability models, and sediment mobility models to assess alternative restoration scenarios and determine what activities will enhance physical habitat. A framework for use of this technology for adaptive management and monitoring has been developed by Wheaton et al. (2004a, b) and Pasternack (2008); more information on the Spawning Habitat Integrated Rehabilitation Approach (SHIRA) can be found at the website: http://shira.lawr.ucdavis.edu/.

Conclusions

Pre- and post-project monitoring are essential components of any restoration strategy. Pre-project monitoring facilitates the design of effective, economical and efficient restoration activities by identifying the problems restoration should address, documenting natural system variability, and providing baseline data for future monitoring. Effective selection and application of restoration techniques requires identifying actual problems as distinct from natural conditions. Similarly, design of post-project monitoring requires a clear articulation of purpose of restoration, the problem being addressed, performance goals of the restoration technique, and direct outcomes of restoration. Using the Strawberry River restoration project as a template, we discuss a range of techniques for pre- and post-project monitoring of two commonly used restoration strategies: bank stabilization and morphological construction. We recommend this approach for restoration managers and practitioners who seek to inform adaptive management or additional restoration strategies.

References

Acornley, R.M. and Sear, DA. 1999. Sediment transport and siltation of brown trout (Salmo trutta L.) spawning gravels in chalk streams. Hydrological Processes 13: 447–458.

Al-Chokhachy, R., and P. Budy. 2008. Demographic characteristics, population structure, and vital rates of a fluvial population of bull trout in Oregon. Transactions of the American Fisheries Society. 137: 1709-1722. doi: 10.1577/T07-247.

Al-Chokhachy, R., P. Budy, and H. Schaller. 2005. Understanding the significance of redd counts: a comparison between two methods for estimating the abundance of and monitoring bull trout populations. North American Journal of Fisheries Management 25: 1505-1512.

Al-Chokhachy, R., P. Budy, and M. Conner. 2009. Detecting declines in the abundance of a bull trout (*Salvelinus confluentus*) population: understanding the accuracy, precision, and costs of our efforts. Canadian Journal of Fisheries and Aquatic Sciences 66: 649-658.

Budy, P., and H. Schaller. 2007. Evaluating tributary restoration potential for pacific salmon recovery. Ecological Applications 17(4):1068-1086.

Bunte, K.; Abt, S.R. 2001. Sampling surface and subsurface particle-size distributions in wadeable gravel-and cobble-bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. Gen. Tech. Rep. RMRS-GTR-74. Fort Collins,CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 428 p.

Burnham, K. P., D. R. Anderson, and G. C. White. 1995b. Selection among open population capture-recapture models when capture probabilities are heterogenous. Journal of Applied Statistics 22:611-624.

Burnham, K. P., G. C. White, and D. R. Anderson. 1995a. Model selection strategy in the analysis of capture-recapture data. Biometrics 51:888-898.

Carling PA. 1984. Deposition of fine and coarse sand in an open-work gravel bed. Canadian Journal of Fisheries and Aquatic Sciences 41:263–270.

Collins, A.L., Walling, D.E., Documenting catchment suspended sediment sources: problems, approaches and prospects. Progress in Physical Geography, 28(2), pp. 159-196, 2004.

Dunham, J. B., A. E. Rosenberger, R. F. Thurow, A. Dolloff, and P. J. Howell. In review. Coldwater fish in wadeable streams. Chapter 8 in Standard Sampling Methods for North American Freshwater Fishes, S.A. Bonar, D.W. Willis, W.A. Hubert. American Fisheries Society, Bethesda, MD.

Dunham, J.B., B.E. Rieman, and K. Davis. 2001. Sources and magnitude of sampling error in redd counts for bull trout Salvelinus confluentus. North American Journal of Fisheries Management 21:343-352.

Fox, R. 2002. Stream Geomorphology: Laboratory Excercises for Limnology. water.washington.edu/Protocols/surveying.html. Accessed 15 October 2009.

Fripp J, Diplas P. 1992. Surface sampling of dry and underwater sediment deposits. In Hydraulic Engineering: Saving a Threatened Resources – in Search of Solutions, Jennings M, Bhowmik NG (eds). ASCE: Baltimore, MD; 853–858.

Fripp JB, Diplas P. 1993. Surface sampling in gravel streams. Journal of Hydraulic Engineering, ASCE 119(4): 473–490. Rice S, Haschenburger J. 2004. A hybrid method for size characterization of coarse subsurface fluvial sediments. Earth Surface Processes and Landforms 29(3): 401–421.

Frostick, L.E., Lucas, P.M., and Reid, I. 1984. The infiltration of fine matrices into coarse-grained alluvial sediments and its implications for stratigraphical interpretation. The Journal of the Geological Society 141: 955–965.

Harrelson, Cheryl C., Rawlins, C. L., Potyondy, John P. 1994. Stream Channel Reference Sites: An Illustrated Guide to Field Technique. Gen. Tech. Rep. RM-245. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 61 p.

Haro, A. and B. Kynard. 1997. Video evaluation of passage efficiency of American shad and sea lamprey in a modified Ice Harbor fishway. North American Journal of Fisheries Management 17:981-987.

Hawkins, C. P, J. F. Kershner, P A. Bisson, M. D. Bryant, L. M. Decker, S. V Gregory, D. A. McCullough, C. K. Overton, G. H. Reeves, R. J. Steedman, and M. K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18:3-12.

Heitke, J. D.; Archer, E. J.: Dugaw, D. D.: Bouwes, B. A.; Archer E. A.; Henderson, R. C.; Kershner, J. L. 2008. Effectiveness monitoring for streams and riparian areas: sampling protocol for stream channel attributes. Unpublished paper on file at: http://www.fs.fed.us/biology/fishecology/emp.

Hemmingsen, A.R., Gunckel, S.L., and Howell, P.J. 2001. Bull trout life history, genetics, habitat needs, and limiting factors in Central and Northeast Oregon. Bonneville Power Administration. Project No. 1994-05400.

Hickman, T., and R. F. Raleigh. 1982. Habitat suitability index models: Cutthroat trout. U.S.D.A. Fish and Wildlife Service. FWS/OBS-82/10.5. 38 pp.

Hiebert, S., L. A. Helfrich, D. L. Weigmann, and C. Liston. 2000. Anadromous salmonid passage and video image quality under infrared and visible light at Prosser Dam, Yakima River, Washington. Fisheries 20:827-832.

Homel, K., P. Budy, M. Pfrender, T. Whitesl, L. Bjerregaard, and K. Mock. 2008. Evaluating the consequences of life-history variation on genetic structure within a population of bull trout (Salvelinus confluentus) in Northeast Oregon. Ecology of Freshwater Fish: 17(3**):** 465-474

Homel, K. and P. Budy. 2008. Temporal and spatial variability in the migration patterns of juvenile and subadult bull trout (*Salvelinus confluentus*) in Northeast Oregon. Transactions of the American Fisheries Society.

Julien, H. P. and Bergeron, N. E. 2006. Effect of fine sediment infiltration during the incubation period on Atlantic salmon (Salmo salar) embryo survival. Hydrobiologia (2006) 563:61–71. DOI 10.1007/s10750-005-1035-2.

Klingeman PW. 1987. Discussion of Church MA, McLean DG, Wolcott JF 1987. In *Sediment Transport in Gravel-Bed Rivers*, Thorne CR, Bathurst JC, Hey RD (eds). Wiley: Chichester; 81–83

Kondolf, G.M. 2000. Assessing salmonid spawning gravels. Transactions of the American Fisheries Society 129:262-281.

Kondolf, G.M., Williams, J.G., Horner, T.C., and Milan, D. 2008. Assessing physical quality of spawning habitat. American Fisheries Society Symposium 65:000-000, 26 p.

Lachance, S. and Dube M. 2004. A new tool for measuring sediment accumulation with minimal loss of fines. North American Journal of Fisheries Management 24: 303–310.

Lenzi, M.A., Marchi, L. 2000. Suspended sediment load during floods in a small stream of the Dolomites (northeastern Italy). Catena, 39(4), pp. 267-282.

Levasseur, M., Bergeron, N.E., Lapointe, M. and Bérubé. 2006. Effects of silt and very fine sand dynamics in Atlantic salmon (Salmo salar) redds on embryo hatching success. Canadian Journal of Fish and Aquatic Sciences. **63**: 1450–1459

Lisle, T. E., and J. Lewis. 1992. Effects of sediment transport on survival of salmonid embryos in a natural stream: a simulation approach. Can. J. Fish. Aquat. Sci. 49: 2337-2344.

Maxell, B.A. 1999. A prospective power analysis on the monitoring of bull trout stocks using redd counts. North American Journal of Fisheries Management 19:860–866.

McHugh, P. M., Budy. P, and Schaller, H. 2004. A model-based assessment of the potential response of Snake River spring-summer Chinook salmon to habitat improvements. Transactions of the American Fisheries Society 133: 622-638.

McNeil, W. J. and Ahnell, W. H., 1964. Success of pink salmon spawning relative to size of spawning bed materials. United States Fish and Wildlife Service Special Scientific Report--Fisheries, No. 469 15 pp.

Moore, T.L., Starcevich, S.J., S. Jacobs, and P.J. Howell. 2005. Migratory patterns, structure, abundance, and status of bull trout populations from subbasins in the Columbia Plateau. 2004 Annual Report. Project 199405400. Bonneville Power Administration, Portland, OR

Montgomery, D.R., and J.M. Buffington, 1997, Channel-reach morphology in mountain drainage basins, Geological Society of America Bulletin, 109:596-611

Nistor, C.J., Church, M., Suspended sediment transport regime in a debris-flow gully on Vancouver Island, British Columbia. Hydrological Processes, 19(4), pp. 861-885, 2005.

Pasternack, G. B. 2008. Spawning habitat rehabilitation: advances in analysis tools. In (D.A. Sear, P. DeVries, S. Greig, Eds) Salmonid spawning habitat in rivers: physical controls, biological responses, and approaches to remediation. Symposium 65, American Fisheries Society, Bethesda, MD.

Peterson, J. T., R.F. Thurow, and J. Guzevich. 2004 An evaluation of multi-pass electrofishing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133:462-475.

Peterson, J., J. Dunham, P. Howell, R. Thurow, and S. Bonar. 2002. Protocol for determining bull trout presence. Report to the Western Division of the American Fisheries Society. Available: www.fisheries.org/wd/committee/bullptrout/protocolFinal-2–02.doc.

Peterson, J. T., R.F. Thurow, and J. Guzevich. 2004 An evaluation of multi-pass electrofishing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133:462-475. ow 2004

Petts GE, Thoms MC. 1986. Channel aggradation below Chew Valley Lake, Somerset, UK Catena 13: 305–320.

Pollock, K.H. 1981. Capture-recapture models allowing for age-dependent survival and capture rates. Biometrics 37: 521-529.

Pollock, K.H. 1982. A capture-recapture design robust to unequal probabilities of capture. Journal of Wildlife Management 46: 757-760.

Ramos CE. 1996. Quantification of stream channel morphological features: recommended procedures for use in watershed analysis and TFW ambient monitoring. TFW-AM 9-96-006, 89 p.

Rood K, Church M. 1994. Modified freeze-core technique for sampling the permanently wetted streambed. North American Journal of Fisheries Management 14: 852–861.

Rosenberger, A.E., and J.B. Dunham. 2005. Validation of abundance estimates from mark-recapture and removal techniques for rainbow trout by electrofishing in small streams. North American Journal of Fisheries Management 25:1395-1410.

Rosgen, D. L., 1996. “Applied River Morphology”, Wildland Hydrology Books, 1481 Stevens Lake Road, Pagosa Springs, Co. 81147, 385 pp.

Sankovich, P.M., S.L. Gunckel, A.R. Hemmingsen, I.A. Tattam, and P.J. Howell. 2003. Migratory patterns, structure, abundance, and status of bull trout populations from subbasins in the Columbia Plateau. 2002 Annual Report. Project 199405400. Bonneville Power Administration, Portland, OR.

Snyder, D.E. 2003. Electroshocking and its harmful effects on fish. Information and Technology Report USGS/BRD/ITR-2003-0002. U.S. Government Printing Office, Denver, Colorado. 149 pages.

Snyder, D.E. 2004. Invited overview: conclusions from a review of electrofishing and its harmful effects on fish. Reviews in fish biology and fisheries 13:445-453.

Tappel, P. D., and T. C. Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. N. Am. J. Fish. Manage. 3: 123-135.

Thoms M.C. 1992. A comparison of grab- and freeze-sampling techniques in the collection of gravel bed river sediment. Sedimentary Geology 78: 191–200.

Thurow, R.F., J.T. Peterson, and J.W. Guzevich. 2006. Utility and validation of day and night snorkel counts for estimating bull trout abundance in first to third order streams. North American journal of Fisheries management 26:117-132.

Thurow, R. F. and D. J. Schill. 1996. Comparison of day snorkeling, night snorkeling, and electrofishing to estimate bull trout abundance and size structure in a second-order Idaho stream. N. Amer. J. Fish. Manage. 16: 314-323.

Wheaton, J.M., Pasternack, G.B., and Merz, J.E. 2004a. Spawning habitat rehabilitation – I. Conceptual Approach & Methods. International Journal of River Basin Management 2:1:3-20.

Wheaton, J. M., Pasternack, G. B., and Merz, J. E. 2004b. Spawning Habitat Rehabilitation – II. Using hypothesis development and testing in design, Mokelumne River, California, U.S.A. International Journal of River Basin Management 2:1:21-37.

White, G. C., D. R. Anderson, K. P. Burnham and others. 1982. Capture-recapture and removal methods for sampling closed populations. UC-11. Prepared by U.S. Department of Energy, Los Alamos National Laboratory, Los Alamos, New Mexico.

Wu, F. C. and C. F. Wang. 2000. Modeling embryo survival affected by sediment deposition into salmonid spawning gravels: Application to flushing flow prescriptions. Water Resources Research 36: 1595-1603.

Zimmerman, A., Coulombe-Pontrbriand, M. and Lapointe, M. 2005. Biases of submerged bulk and freeze core alluvial sediment samples. Earth Surface Processes and Landforms 30 (11), 1405-1417.

Zimmerman, A. and Lapointe, M. 2005. Sediment infiltration traps: their use and potential misuse to monitor salmonid spawning habitat. Hydrological Processes 19, 4161-4177.

Appendix 1: Supplemental results from Task 1 (effects of grazing exclosures)

Figure A1-1: Size-frequency distribution compared between ungrazed and grazed reaches for each of the eight watersheds containing Bonneville cutthroat trout. Also presented are results from Kolmogorov-Smirnov tests comparing the distributional homogeneity between ungrazed and grazed reaches.

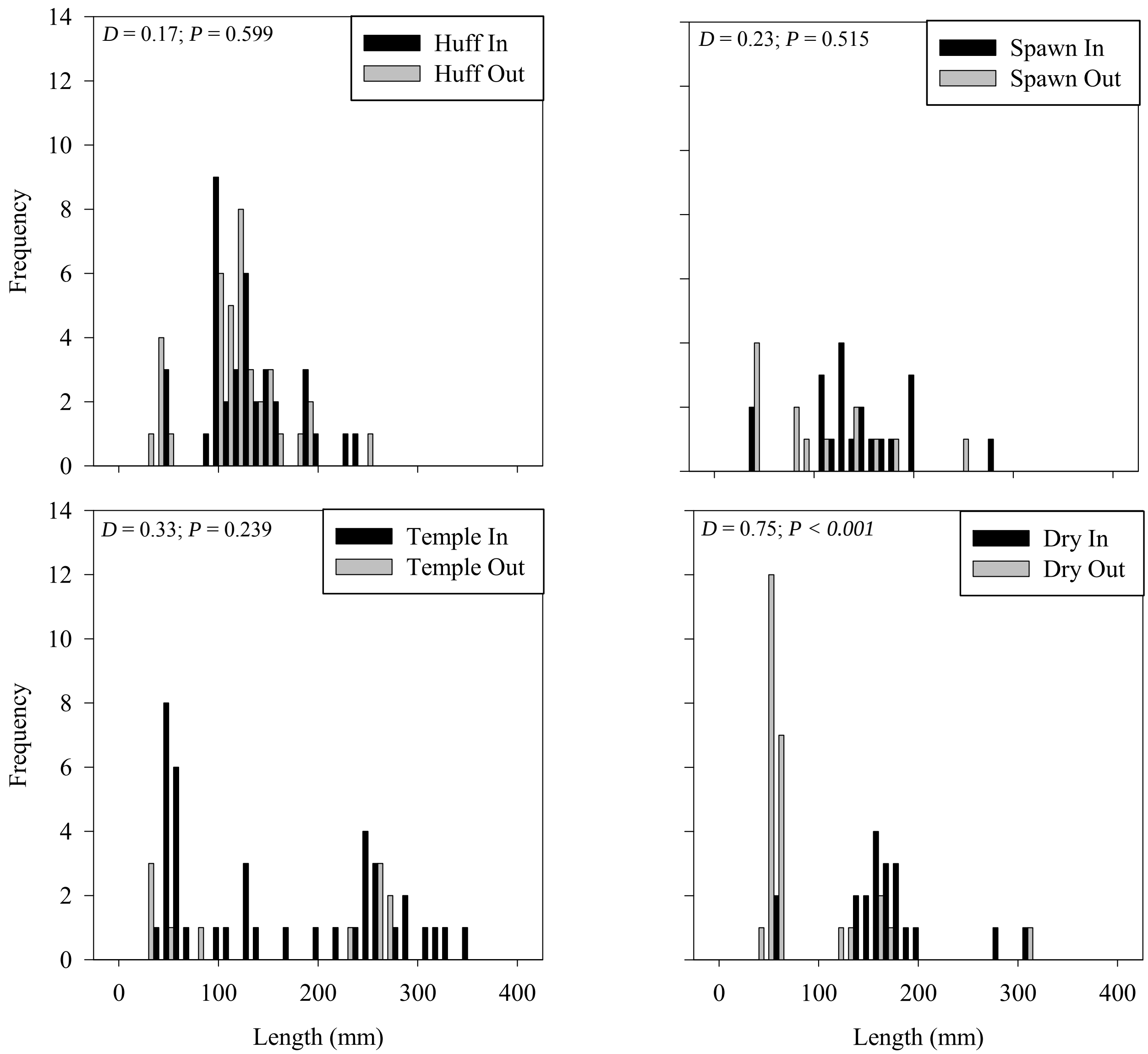


Figure A1-1 Cont.

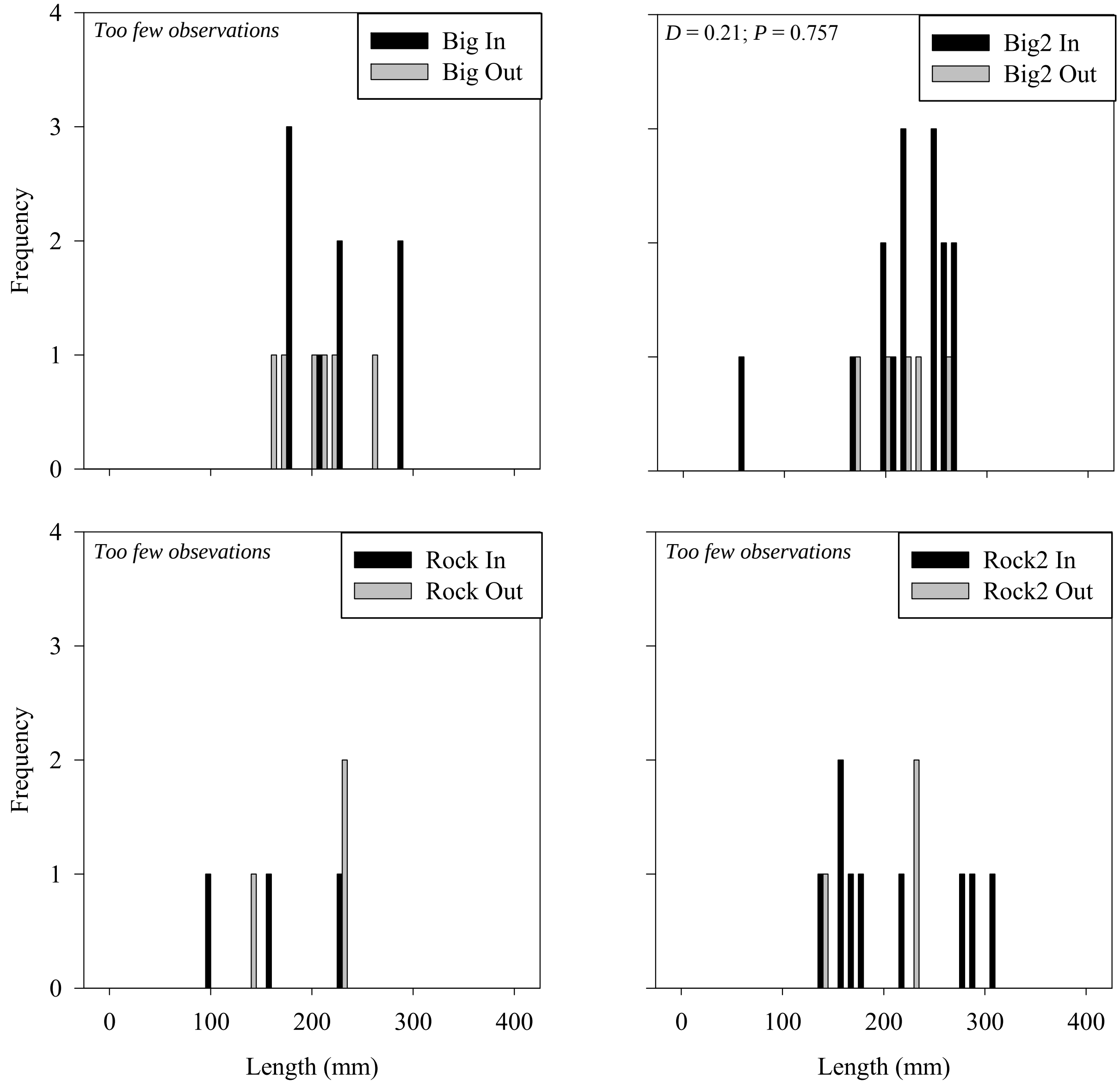
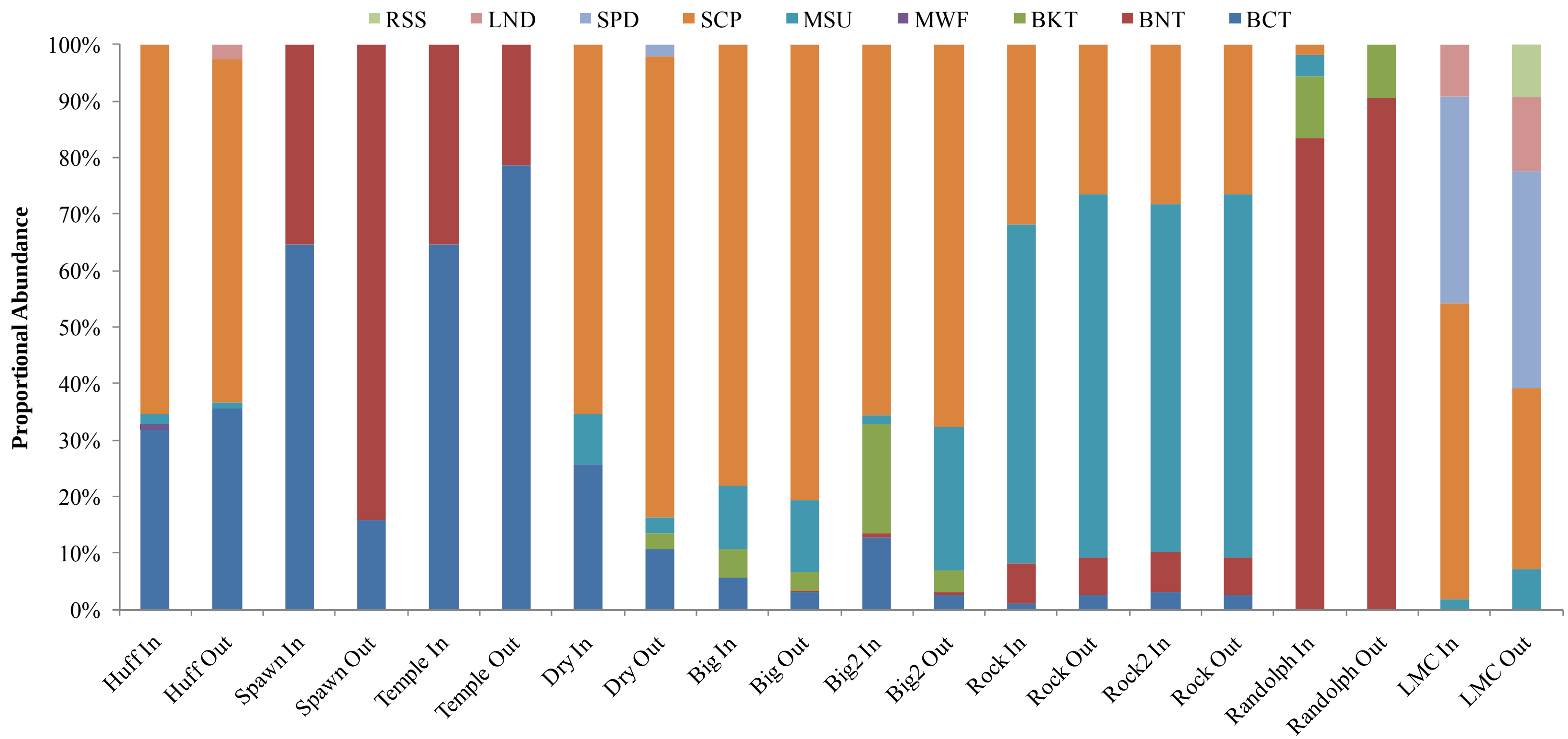


Figure A1-2: Proportional abundance of fish assemblages for the ten paired reaches. Fish abbreviations include: redside shiner (RSS), longnose dace (LND), speckled dace (SPD), sculpin (SCP), mountain sucker (MSU), mountain whitefish (MWF), brook trout (BKT), brown trout (BNT), Bonneville cutthroat trout (BCT). Note, sculpin were not sampled at Spawn Creek and Temple Fork.



Appendix 2: Supplemental results and methodology from Task 2 (Strawberry River monitoring program)

I. Habitat availability and snorkel surveys: Methodology and supplemental results

Table A2.I-1: Locations of the 500 m-long monitoring reaches on the Strawberry River, established September 2008. All coordinates are in UTM grid zone 12.

	Downstream		Upstream		Access point
Location	Northing	Easting	Northing	Easting	
Restored in 2008	4456555	0481428	4456782	0481279	Bull Springs
Restored in 2009	4456823	0481131	4457017	0481006	Bull Springs
Unrestored (Control)	4458175	0480704	4458495	0480803	Highway 40 at bridge

Table A2.I-2: Locations of habitat availability cross-sections (XS) on monitoring reaches of the Strawberry River, established September 2008. All coordinates are in UTM grid zone 12.

	Restored in 2008		Restored in 2009		Unrestored (Control)	
XS	N	E	N	E	N	E
1	4456833.90	481141.30	4456564.74	481388.65	4458289.00	480746.57
2	4456840.15	481132.10	4456564.20	481395.50	4458286.17	480737.71
3	4456841.06	481119.57	4456574.93	481401.46	4458283.45	480728.00
4	4456850.60	481111.89	4456585.67	481402.29	4458281.20	480718.16
5	4456862.54	481111.08	4456596.54	481405.93	4458287.23	480710.65
6	4456864.24	481124.09	4456590.64	481415.29	4458296.70	480706.52
7	4456850.64	481136.01	4456595.00	481427.26	4458306.41	480705.20
8	4456847.38	481145.04	4456606.27	481432.63	4458314.81	480710.67
9	4456848.57	481156.15	4456619.16	481431.94	4458317.79	480719.94
10	4456857.06	481161.14	4456631.14	481426.51	4458314.66	480730.40
11	4456972.68	481115.47	4456639.12	481415.37	4458315.62	480741.75
12	4456979.41	481101.68	4456636.35	481401.90	4458320.96	480750.79
13	4456992.90	481094.84	4456646.53	481397.52	4458329.76	480757.69
14	4457002.39	481098.83	4456656.58	481388.41	4458341.26	480757.88
15	4457009.91	481105.09	4456665.26	481378.16	4458351.95	480757.31
16	4457018.91	481109.28	4456671.57	481366.16	4458361.16	480752.70
17	4456665.71	481352.747	4457028.87	481110.646	4458370.74	480754.06
18	4456677.38	481345.408	4457037.44	481105.606	4458373.78	480763.56
19	4456690.95	481340.582	4457044.86	481098.899	4458373.06	480772.78
20	4456704.35	481336.765	4457050.08	481090.918	4458374.49	480782.94

Table A2.I-3: Surface composition and hydraulic parameters from surveys of adult and juvenile cutthroat trout habitat use and habitat availability, Strawberry River, September 2008 and 2009.

Reach	Restored in July 2008		Restored in July 2009		Unrestored (Control)		Adults		Juveniles	
Date and condition	Sep. 2008 1 mo. post	Sep. 2009 1 yr. post	Sep. 2008 1 mo. pre	Sep. 2009 1 mo. post	Sep. 2008	Sep. 2009	Sep. 2008	Sep. 2009	Sep. 2008	Sep. 2009
Surface composition										
N	401	480	404	480	398	480	103	360	132	6590
D_{16}*	6.9	1.1	12.6	1.0	9.9	1.3	25.2	16.7	7.3	8.8
D_{50}	18.5	23.9	27.2	23.4	28.9	18.3	47.1	52.8	36.6	39.7
D_{64}	25.3	32.5	34.8	35.9	38.1	31.3	55.6	62.9	47.9	53.3
D_{84}	39.8	55.8	52.0	60.0	55.0	55.4	71.4	88.3	61.8	80.8
D_{90}	46.0	68.5	47.6	73.7	61.2	70.2	78.4	102.7	69.8	90.5
%< 2mm	--	28.8	--	31.9	--	24.9	--	3.1	--	9.5
Geometric SD $(D_{84} / D_{16})^{-0.5}$	0.4	0.1	0.5	0.1	0.4	0.2	0.6	0.4	0.3	0.3
Hydraulic parameters										
N	240	240	240	240	240	240	107		160	
Depth (m)	0.28(0.19)	0.26(0.11)	0.29(0.16)	0.25(0.1)	0.21(0.15)	0.35(0.15)	0.54(0.17)	0.55(0.12)	0.40(0.14)	0.50(0.16)
u_0 (m/s)	0.04(0.10)	0.07(0.07)	0.04(0.09)	0.07(0.07)	0.05(0.11)	0.01(0.04)	0.18(0.11)	0.11(0.10)	0.16(0.13)	0.04(0.07)
U (m/s)	0.13(0.17)	0.17(0.09)	0.14(0.15)	0.17(0.09)	0.12(0.18)	0.07(0.05)	1.40(1.98)	0.14(0.10)	1.55(2.48)	0.11(0.09)
BCT numbers										
BCT Adults	36	20	34	12	37	3	107	36	--	659

BCT Juveniles	54	175	59	145	47	105	--	--	160	--

Notes: D_i is the size of the particle in the i^{th} percentile; % < 2 mm is the percent of particles less than 2 mm in size. *Surface composition measurements differed between years that bias the D_{16} estimate in 2008 for the reach restored in 2009. The smallest maximum particle size measured was 8 mm in 2008 and 2 mm in 2009, thus a D16 of 8 mm in 2008 and 2 mm in 2009 for the reach restored in 2009 is likely an artificially imposed difference. For the same reason, percent fines (%<2 mm) estimates are not available for 2008. -- = Data not available or not applicable. Values in parentheses are standard deviation of all cross-sections or fish locations on each reach. 'n' is the number of points use to calculate a given statistic (e.g., number of particles measured, number of depth measurements); particle distributions of adults and juvenile locations are based on single substrate measurements at each location for the 2008 data and 10 particles per location in the 2009 data. u_0 = near-bed velocity; U = average velocity

II. Water quality and riparian conditions: Field methodology

At the left bank, right bank, and center of each habitat availability cross-section we recorded temperature, pH, electrical conductivity, and vegetative cover. Water quality metrics were measured with a handheld multiprobe system; cover was measured with a densitometer. We measured the angle of both banks using a stadiorod and compass (Heitke et al, 2008).

Five temperature loggers were installed in September 2008 to continuously monitor stream temperature (one hour intervals) in restored and unrestored reaches. Loggers are located ~ 2 km upstream of Highway 40 (unrestored), ~ 500 m upstream of Highway 40 (unrestored), at the Highway 40 crossing (unrestored), downstream of the section restored in July 2009 (pre- and post-restoration), and downstream of the section restored in July 2008 (post-restoration).

III. Redd surveys: Methodology and supplementary results

Table A2.III-1: Locations of cutthroat redds surveyed in July 2009, Strawberry River. All coordinates are in UTM grid zone 12.

	Restored in 2009		Restored in 2008		Unrestored (Control)	
Redd	Northing	Easting	Northing	Easting	Northing	Easting
1	4456861	481106.8	4456531	481406.5	4458346	480755.1
2	4456887	481125.2	4456591	481397.3	4458317	480711.6
3	4456909	481100.9	4456591	481398.5	4458194	480703.2
4	4456930	481108.2	4456600	481408.9	4458385	480787.3
5	4456943	481123.2	4456600	481412	4458402	480812.8
6	4457049	481087.3	4456620	481426	--	--
7	4457029	481046.5	4456663	481355.6	--	--
8	4457021	481001.3	4456795	481322.1	--	--
9	4457021	481003.8	4456785	481287.3	--	--
10	4457022	481003.2	--	--	--	--
11	4457021	481005.5	--	--	--	--
12	4457026	481005.3	--	--	--	--

BCT select substrates with a narrower distribution but similar D_{50} to reach-scale and riffle substrates (Table A2.III-2, Fig. A2.III-1 and Fig. A2.III-2). Redd depths are on average shallower and velocities faster than reach-scale conditions (Table A2.III-2).

Table A2.III-2: Surface composition and hydraulic parameters from surveys from cutthroat trout redds and restored and unrestored reaches, Strawberry River, July 2009.

Location	Unrestored Control	Restored in 2009 1 mo. pre-restoration	Restored in 2008 1 yr. post-restoration	Redds
Surface composition				
n	480	480	480	2900
D_{16}	3.3	2.0	1.5	9.9
D_{50}	27.0	27.4	18.0	24.4
D_{64}	38.1	40.7	33.6	28.7
D_{84}	63.7	65.4	58.2	40.4
D_{90}	76.4	77.9	77.9	45.0
%< 2mm	17.6	16.3	31.7	3.0
Geometric S $(D_{84}/D_{16})^{-0.5}$	0.23	0.17	0.16	0.49
Hydraulic parameters				
n	240	240	240	29
Depth (m)	0.32(0.07)	0.34(0.06)	0.33(0.07)	0.25(0.07)
Near-bed velocity (m/s)	0.11(0.07)	0.13(0.09)	0.16(0.09)	0.27(0.15)
Average velocity (m/s)	0.30(0.13)	0.36(0.12)	0.34(0.11)	0.54(0.19)

Figure A2.III-1: Surface particle size distributions of BCT redds compared to reach-scale substrate composition for each reach, Strawberry River, July 2009.

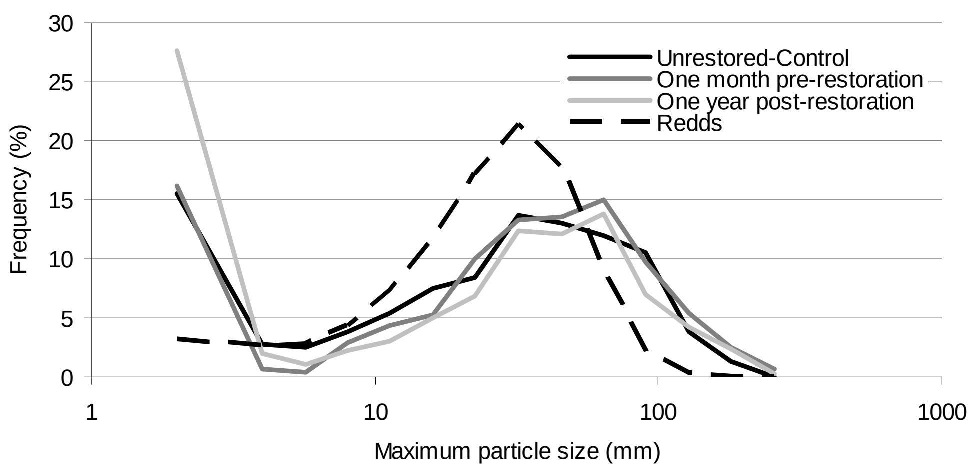
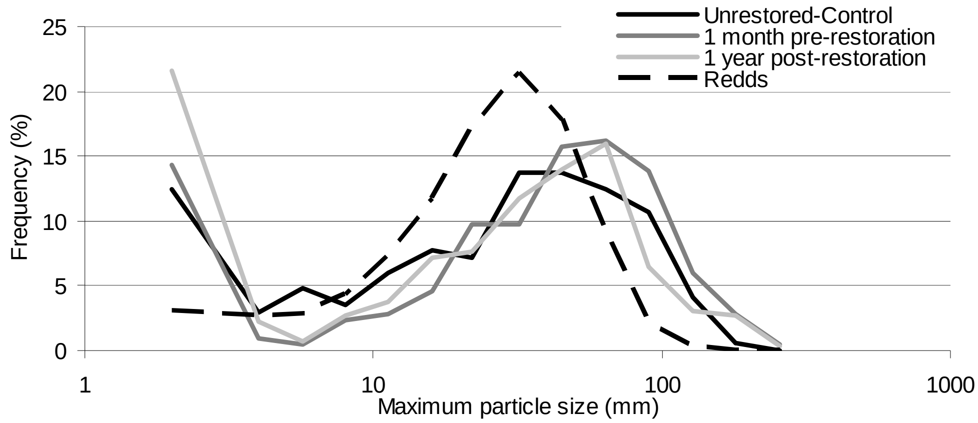


Figure A2.III-2: Particle size distributions of riffles on restored and unrestored reaches compared to BCT redds, Strawberry River, July 2009.



IV. Geomorphic assessments: Aerial photo analysis and field surveys

A long-term aerial photo record (1938-present) is being used to assess historic changes in planform geometry and riparian vegetation of the Strawberry River Watershed. Image analysis software (ArcMap, ArcView, Imagine) is being used to orhtorectify photos, delineate channel location, and quantify planform statistics (e.g. width, curvature, lateral migration rate) at different time steps. Preliminary evaluations suggest that the mid-century loss of riparian willows contributed to decreasing bank stability and channel widening.

A comprehensive field survey of channel morphology began in September 2008 following the first phase of restoration. Channel surveys were performed with a Topcon RTK Pro GPS, which provides high precision locations and elevations for the computation of vertical, lateral, and longitudinal changes in channel morphology. A longitudinal profile of the entire study section was surveyed, including the centerline bed elevation and water surface elevations at high and base flows. On each reach, a high density (10-35) of channel cross-sections (left to right valley flat) spaced a varying intervals were used to measure the cross-sectional geometry of different morphological units (e.g. riffles, pools) and river locations (e.g. meander bends). Each cross-section was marked at the left bank location with a piece of rebar driven into the bank and the geographic coordinates recorded for future repeat surveys (Table A2.IV-1). For bed profiles and cross-sections, survey points were shot at each change in elevation, thus capturing the longitudinal distribution of morphological units (Fig. A2.IV-1) and channel characteristics such as bank height, bar size, channel width and depth, channel shape, and floodplain elevation. Immediate and persistent morphological changes due to restoration can be seen by comparing repeat cross-sections immediately prior to and at intervals following restoration, such as the sloped banks seen on a cross-section one month post-restoration compared to the same cross-section surveyed one month before restoration (Fig. A2.IV-2).

Table A2.IV-1: UTM coordinates of channel cross-sections on restored and unrestored reaches of the Strawberry River. All coordinates are in UTM grid zone 12.

	Restored in 2008		Restored in 2009		Control (Unrestored)	
Cross-section	Northing	Easting	N	E	N	E
1	4456622	481434.8	4456924	481107.4	4458401	480822.6
2	4456615	481436.7	4456922	481106.5	4458384	480809.2
3	4456607	481435.7	4456919	481105.5	4458380	480797.6
4	4456600	481435.2	4456915	481104.3	4458373	480786
5	4456594	481431.8	4456910	481105.7	4458371	480774
6	4456588	481421	4456907	481107.6	4458371	480763.9
7	4456591	481411.4	4456902	481111.2	4458367	480757.7
8	4456594	481407	4456899	481113.7	4458365	480757.6
9	4456593	481406.4	4456897	481117.3	4458356	480758.2
10	4456591	481405.9	4456896	481119.9	4458347	480761.5
11	4456590	481405	4456893	481126.2	--	--
12	4456586	481406.3	4456892	481133.2	--	--
13	4456575	481405	4456890	481139.2	--	--
14	4456559	481397.2	4456892	481146.1	--	--
15	4456559	481388.3	4456887	481153	--	--
16	4456561	481382	4456880	481159.9	--	--
17	4456565	481374.4	4456869	481166.1	--	--
18	4456564	481370.7	4456863	481165	--	--
19	4456566	481371.1	4456854	481160.3	--	--
20	4456562	481369.3	4456846	481150.8	--	--
21	4456564	481368.2	4456847	481140.5	--	--
22	4456562	481368.1	4456854	481132.1	--	--
23	4456559	481366.6	4456857	481128.7	--	--
24	4456557	481365.7	4456861	481125.8	--	--
25	4456553	481366	4456863	481121.1	--	--
26	4456549	481367.1	4456863	481118.4	--	--

27	4456546	481366.9	4456862	481114.2	--	--
28	4456545	481367.2	4456859	481112	--	--
29	4456543	481370.1	4456858	481112	--	--
30	4456542	481373.6	4456855	481112	--	--
31	4456539	481378.4	4456850	481113.8	--	--
32	4456540	481381.6	4456846	481116.5	--	--
33	4456541	481386	4456843	481118.6	--	--
34	4456538	481389.9	4456843	481123.3	--	--
35	4456538	481395.2	4456842	481126.7	--	--
36	---	--	4456840	481138.8	--	--
37	--	--	4456833	481143.8	--	--
38	--	--	4456828	481147.3	--	--

Figure A2.IV-1: Bed and water surface (at high flow and base flow) elevation profiles of the Strawberry River from Highway 40 (distance = 0) to Bull Springs, 2008.

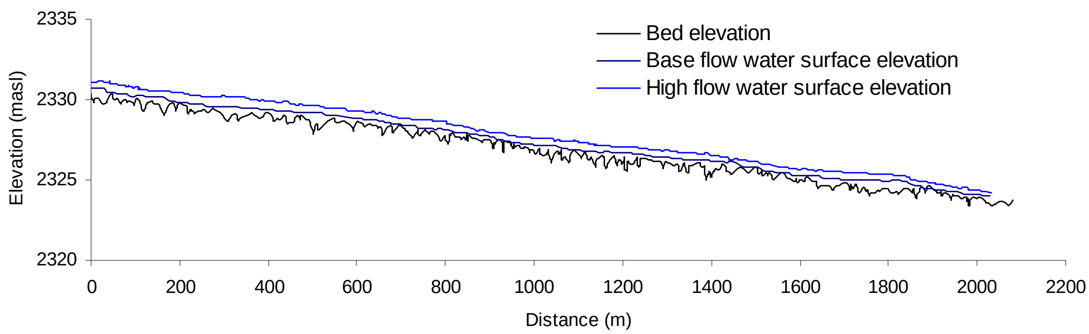
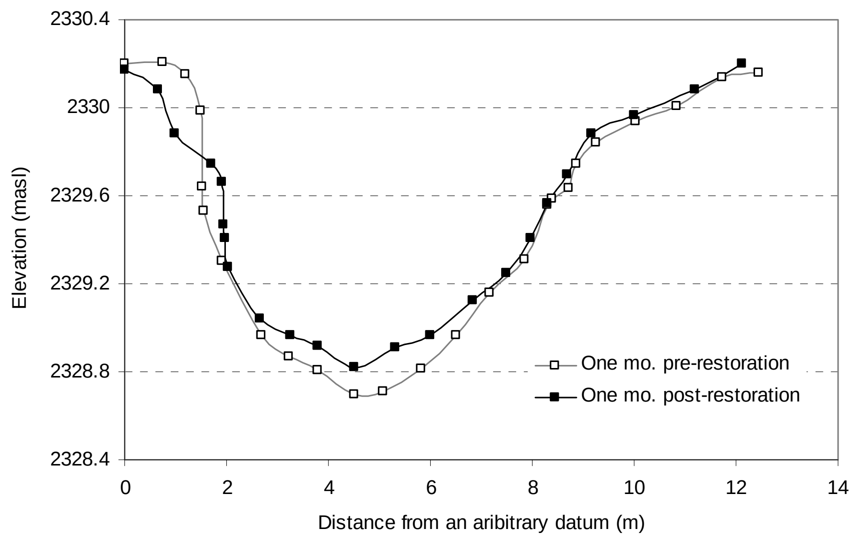


Figure A2.IV-2: A sample cross-section from a restored reach of the Strawberry River one month prior to restoration (July 2009) and one month following restoration (September 2009). Distance goes from left to right floodplain.



Surveys will be repeated in future years and used to quantify morphological changes following restoration. Longitudinal bed elevation profiles can be used to quantify the spacing and size of morphological units. Cross-sectional surveys provide width-to-depth ratios, locations and sizes of channel deposits, bank heights, and floodplain elevation, characteristics that can be used to assess the geomorphic effects of restoration. For instance, the effectiveness of restoration at reducing bank erosion and channel incision will be reflected in channel width-to-depth ratios and differences between channel and floodplain elevations.

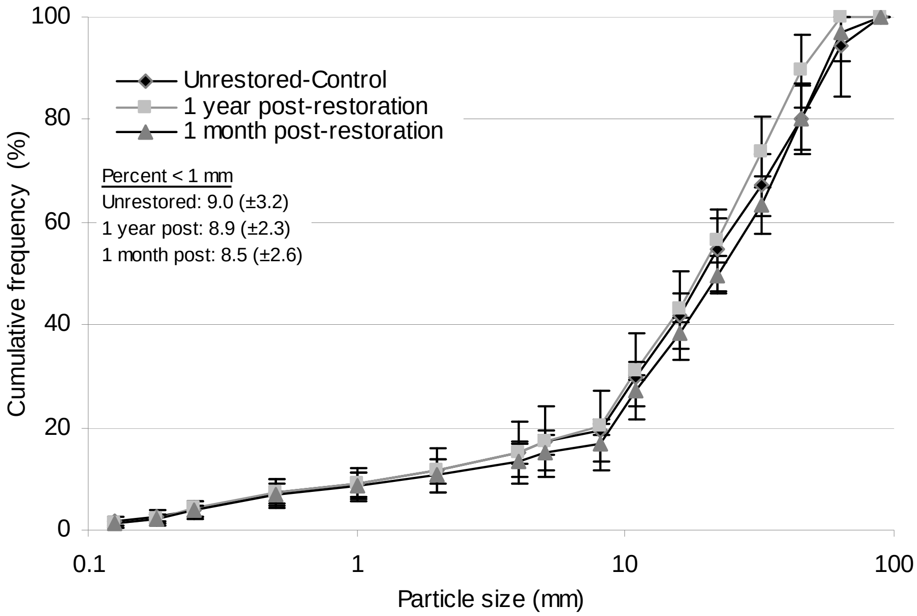
V. Subsurface sampling: Field methodology

In order to assess the particle size distribution of spawning gravels and the effects of restoration on bed sediment, we collected subsurface sediment samples on control and restored reaches in October of 2009. We used October samples to determine the sediment content of gravels following a long period of low flow (July – October) and the conditions experienced by emerging BCT fry (assuming little sediment mobilization between August and October). We sampled only riffle substrates similar to those used by spawning BCT (identified by the redd surveys) following a long period of base flow. We plan to repeat these measurements during the spawning season of 2010, immediately following high spring flows. In this manner, we can determine how much fine sediment accumulates during low flows and what bed conditions are experienced by spawning trout.

Samples were collected with a modified McNeil sampler. Details of the McNeil sampler methodology can be found in several sources (e.g. McNeil, and Ahnell, 1964; Bunte and Abt, 2001). We elected to use a McNeil sampler because of its relative ease-of-use, low cost, and ability to retain fine sediments suspended during sampling. Two evenly spaced composite samples were collected from three riffles within the three 500 m reaches, representing three stages of restoration status (restored 1 year prior, restored 1 month prior, and unrestored All material suspended during sampling was collected and filtered in the field such that loss of fines was minimal; some loss may have occurred at the base of the sampler or during filtering. All particles > 8 mm were sieved into standard size classes and wet-weighed in the field. Material < 8 mm was bagged and returned to the lab for particle size analysis. Samples were oven-dried, sieved and weighed; particle size classes included size classes 0.25-0.5, 0.5-1.0, 1.0-2.0, 2.0-4.0, and 4.0-8.0 mm. Lab- and field-weighed samples were combined into one particle size distribution. Percentage of particles < 2 mm was calculated for each riffle for comparison between sites and sampling dates.

Particle size distributions and percent fines indicate similar and suitable spawning gravel conditions (percentage of particles < 1 mm ranges between 5-14%) on all reaches (Fig. A2.V-1), being less than the 30% deemed unsuitable by the cutthroat HSI though slightly more than the 5% fines considered optimal.

Figure A2.V-1: Particle size distributions of riffle substrates on unrestored (control) and restored reaches of the Strawberry River. Values are the mean (± standard deviation) of composite samples from three replicate riffles on each reach.


Unrestored-Control
1 year post-restoration
1 month post-restoration
Percent < 1 mm
Unrestored: 9.0 (±3.2)
1 year post: 8.9 (±2.3)
1 month post: 8.5 (±2.6)
Cumulative frequency (%)
Particle size (mm)
0
20
40
60
80
100
0.1
1
10
100

Appendix 3: Methods for estimating juvenile and adult BCT population abundance

For this section, we borrowed heavily from a synthesis completed by the Bull Trout Recovery Monitoring and Evaluation Technical Group that recommends monitoring and analytical tools for evaluating bull trout recovery objectives (U.S. Fish and Wildlife Service. 2008. Bull Trout Recovery: Monitoring and Evaluation Guidance. Report prepared for the U.S. Fish and Wildlife Service by the Bull Trout Recovery and Monitoring Technical Group (RMEG). Portland, Oregon. 74 pp.). Accuracy, precision, effort and cost vary widely among techniques for assessing trout population abundance (Table A3-1). Techniques provide either an index or estimate of abundance; some also provide information on size structure, relative abundance of other species, or migration timing and allow for tagging or additional measurements (e.g. length, vital rates, or hybridization). Most techniques provide information at a reach scale (some provide watershed-scale estimates) and are typically employed one or more times per year for multiple years.

Table A2-1: Techniques for assessing habitat suitability and complexity

Technique	Pros	Cons
Redd counts	Low cost/effort No fish handling or stress Long time series available for some populations Allows estimation of population trend	Positive bias for larger, migratory fish Negative bias for small, likely resident fish Potentially high observer error Effects of superimposition and test digs Expansion to an estimate of abundance requires an estimate of adults/redd
Recommendation	Use trained observers and couple with a more precise and accurate technique in a sub-set of populations within core areas.	
Snorkel counts	Includes migratory and resident BCT Low cost and effort Low fish handling and stress Moderate field implementation difficulty Provides size structure Provides index of abundance of other species	May miss migratory fish depending on timing of sampling Strong and variable negative bias Low precision Does not allow tagging or other measurements
Recommendation	Evaluate bias across size classes, habitat types, and time periods (diel) couple with a more precise and accurate technique in a sub-set of local populations within core areas.	

Mark-recapture	Estimate of population abundance (migratory and resident BCT) Flexibility in field sample design (e.g., active or passive recapture, trap, resight) Ability to account variable recapture rates Multiple analytical models available (open and closed) Allows direct estimation of population growth rate independent of abundance estimate (e.g., temporal symmetry model) Provides size structure and potential for population structure Allows other measurements (e.g., length, vital rates, hybridization)	Extremely high cost and effort High difficulty of field implementation difficulty Moderate to high fish handling and stress Use limited in large or warm (>16 °C) rivers and small populations (e.g., handling stress) Precision sensitive to low capture and recapture rates
Recommendation	Evaluate violation of assumptions and model sensitivity analytically. Couple a comprehensive mark-recapture program in key indicator populations with a simpler, more cost-effective technique in other populations within core area.	
Depletion estimates	Estimate of abundance of juveniles and resident adults Estimate of abundance of other species Allows other measurements (e.g., length, vital rates, hybridization)	Not appropriate for migratory adults High cost and effort Moderate to high field implementation difficulty Potentially high fish handling and stress Precision varies with fish size and number of passes Negative bias possible Use limited in large or warm (>16 C) rivers and small populations (e.g., handling stress)
Recommendation	Do a minimum of 3 passes and test for variable detection probability. Evaluate bias across size classes and habitat types in a sub-set of local populations within core areas.	
Single-pass removal	Index of abundance of juveniles and resident adults Moderate cost and effort Moderate field implementation difficulty Provides index of abundance of other species Allows other measurements (e.g., length, vital rates, hybridization)	Not appropriate for migratory adults Variable and negative bias Low precision Use limited in large or warm (>16 C) rivers and small populations (e.g., handling stress)
Recommendation	Evaluate bias across size classes and habitat types, and couple with a more precise and accurate technique in a sub-set of local populations within core areas.	
Weir-trap counts	Low fish handling and stress Allows tagging or other measurements (e.g., length) Provides information on migration timing	High field implementation difficulty High cost and effort Positive bias for larger, migratory fish Negative bias for small, likely resident fish
Recommendation	Couple with a technique (e.g., MR) for quantifying non-migratory adults.	

I. Redd counts

Redd counts provide a feasible and relatively inexpensive estimate of reproductive adult abundance (Table A3-1). Redd surveyors typically visit the spawning grounds several times over the duration of the spawning event and count redds based on conditions such as the presence of spawning fish, disturbance of gravel, and nest structure (Dunham et al., *in review*). The number of redds in a local population or core area may be estimated by a census (i.e., a complete count of all redds throughout the spawning distribution and period), or a sub-sample at index sites or a random sample of potential sites (e.g., Sankovich et al. 2003).

However, in terms of assessing abundance and ultimately trends, redd counts are frequently limited by some combination of: 1) strong observer variability (Maxell 1999; Dunham et al. 2001; Hemmingsen et al. 2001), 2) redd superimposition, 3) delineation between test digs and redds (Maxell 1999; Dunham et al. 2001), and 4) a high proportion of fine substrate (Hemmingsen et al. 2001). In addition, when multiple life-history forms coexist within a single population, both redd and spawner counts are limited by the potential for a positive bias towards large (likely migratory) trout and a corresponding negative bias towards small (likely resident) trout (Moore et al. 2005), a bias that varies substantially across core areas in magnitude (Al-Chokhachy et. al. 2005). Ultimately, if these types of surveys are used to assess abundance, it will be necessary to evaluate different sources of variability and uncertainty and correct for observer error or bias (Dunham et al. 2001; Muhlfeld et al. 2006) as well as which portion of the population (migratory, resident) is primarily represented by redd count data (Al-Chokhachy et al. 2005). This latter issue can be especially critical for assessing status relative to recovery goals; those goals currently require a threshold number of reproductive (but not necessarily migratory) adults.

Snorkel counts can provide a relatively inexpensive, non-invasive technique estimate of BCT population abundance, the abundance of fish by size class, and the relative abundance of other species (Table A3-1). Cutthroat trout snorkel surveys should generally occur during the summer months when migratory adults are present

(Hemmingsen et al. 2001; Homel 2007). Recent research has demonstrated several limitations of this approach for monitoring abundance including: 1) a consistent negative bias (i.e., underestimate), which varies across size class and habitat (Thurow et al. 2006); 2) low precision due to the frequent low densities of fish present and high spatial variability in fish distribution within streams (Al-Chokhachy et al. 2009; Al-Chokhachy and Budy 2008*)* and 3) in response to 1-2, a long temporal commitment required to detect modest changes in abundance (Al-Chokhachy et al. 2009). In addition, snorkeling may be physically infeasible in small, shallow streams and can be extremely biased at low temperatures, when fish are using interstitial spaces. Despite these limitations, snorkel surveys can be an effective means for monitoring abundance if formal evaluations of bias are conducted across size classes, habitat types, and across diel periods (e.g., Thurow et al. 2006). However, further research may be necessary to evaluate the consistency of documented biases across large spatial scales (i.e., local populations within core areas/subbasins etc.).

II. Weir-trap counts

Migratory adults can be counted using a trap in conjunction with a weir or fish ladder as they are moving upstream prior to spawning. This technique is labor intensive, as it requires frequent inspection over several months while fish are migrating. Weirs can also be difficult to maintain during higher flows and periods of heavy litter accumulations. In addition, trap counts do not account for adults that do not migrate below the trap (e.g., Sankovich et al. 2003). A more complete count of adult abundance can be estimated by marking fish captured in the trap and subsequently making mark-recapture estimates (see below) (Sankovich et al. 2003; Dunham et al. 2001). In low-productivity watersheds of relatively good water clarity, the use of video techniques to count concentrations of adults also has promise (Haro and Kynard 1997; Heibert el al. 2000).

III. Electrofishing techniques

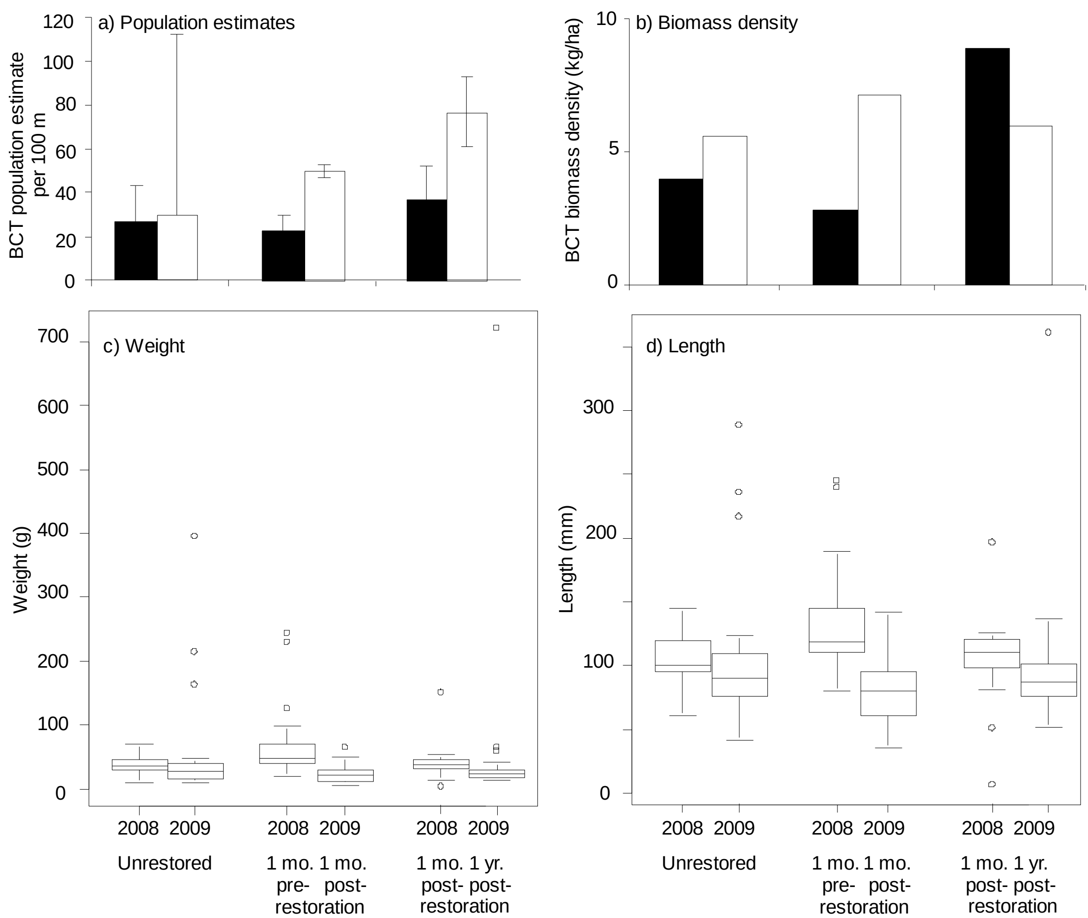
Single-pass electrofishing provides an estimate of BCT abundance that is less biased than snorkel surveys due to higher sampling efficiency, but it is also more invasive (Table A3-1; Thurow and Schill 1996; Peterson et al. 2004). Single-pass removal estimates are usually conducted during summer, base-flow conditions with backpack electroshocker units, with or without block nets, to avoid upstream migrating adults and downstream migrating juveniles. Although single-pass removal techniques are more efficient than snorkeling surveys, similar issues also apply, including: 1) a tendency to be negatively biased (e.g., Rosenberger and Dunham 2005), 2) high variance across reaches within a population (Al-Chokhachy et al. 2009), 3) higher relative cost and effort, and 4) limited feasibility in large rivers, small populations, or where densities are very low, and/or water temperatures exceed 16°C. Nevertheless, like snorkel surveys, single-pass removal techniques may offer an effective technique for monitoring populations across large spatial scales, if the bias between these indices of abundance and population abundance is evaluated *a priori* or simultaneously as part of the same effort (e.g., Rosenberger and Dunham 2005) and relative to habitat characteristics (Peterson et al. 2002; Peterson et al. 2004) and fish size (Peterson et al. 2004).

In contrast to the techniques discussed above, depletion estimates provide an absolute estimate of population abundance. Depletion techniques are generally conducted with multiple personnel and require the use of block nets, thus necessitating a considerably greater degree of effort and a higher cost. The precision of depletion techniques generally increases with an increasing number of passes (e.g., Rosenberger and Dunham 2005), and can vary substantially across size classes of fish (Peterson et al. 2004). Timing considerations are the same as described for snorkel and single-pass techniques. While depletion techniques can provide fairly accurate and precise estimates of abundance, relative to snorkeling or single pass techniques (Thurow and Schill 1996), this technique is sensitive to a similar list of limitations as described above for single-pass techniques, albeit usually not to as great of a degree. The commitment of personnel (typically a minimum crew of three must complete at least three passes and process fish) presents a potential limitation for many monitoring and evaluation programs. As with snorkel or single-pass removal estimates, there is a potential for a negative bias as a function of fish distribution (and factors affecting fish distribution behavior) (Peterson and Thurow 2004). Limitations discussed above for single-pass removal estimates associated with river size, population size and handling/stress effects, and water temperature all apply to depletion estimates.

Because larger fish such as BCT are more responsive to electric fields, they have been expected to be more susceptible to electrofishing-induced injury or mortality. However, laboratory and field data relating the incidence of electrofishing injury and fish size have been equivocal (Snyder 2004). Regardless, the use of electrofishing should be limited to periods prior to spawning because of the potential for significant damage to gametes in ripe fish and developing embryos (Snyder 2003). In addition, electrofishing can be relatively ineffective in extremely low conductivity waters. Nevertheless, despite the higher potential for injury compared to other techniques, electrofishing does permit the collection of other important monitoring data that requires having the fish in-hand (e.g., precise lengths, sex, maturity, genetic tissue samples, etc.) and may be preferable for sampling mixed populations.

Example: Strawberry River population estimates. Three-pass electrofishing depletions were conducted on restored and unrestored reaches of the Strawberry River in September 2008 and 2009. Data from the depletion estimates were used to assess the effects of restoration and time on population abundance, biomass density, and the distributions of fish weights and lengths (Fig. A3-1). Inclusion of fish size in the assessment lends greater insight into the nature of biological response. Population estimates increase one month and one year after restoration, suggesting a positive effect of restoration activities. However this result alone may be misleading. Trout weights and lengths decrease post-restoration, resulting in a decrease in biomass density one year after restoration. Together these results indicate that smaller fish increased in abundance – possibly due to the stocking of young fish in 2008 and 2009 – while the number of larger fish declined. More years of population monitoring are needed to determine whether short-term increases of small fish will produce sustained increases of all size classes.

Figure A3-1: BCT population estimates (a), biomass density (b), and distributions of weight (c) and length (d) from three-pass electrofishing depletions on restored and unrestored reaches in August 2008 and 2009, Strawberry River. Error bars are the 95% confidence interval.



Mark-recapture techniques provide one of the most efficient techniques for estimating abundance and trend; however, this technique is also typically the most costly and requires a high degree of effort and handling of fish (Table A3-1; Al-Chokhachy et al. 2009). Nevertheless, in addition to increased accuracy and precision relative to the other techniques, mark-recapture techniques can simultaneously provide additional information critical for identifying limiting factors and monitoring population status including vital rates (e.g., survival; Al-Chokhachy and Budy 2008), movement patterns (Homel and

Budy 2008), population structure (Al-Chokhachy and Budy 2008), and ultimately population trend (see below). A mark-recapture estimate of population abundance can be accomplished via a range of analytical estimators. Simple closed-capture population abundance estimates (e.g., Lincoln-Petersen type models) are based on an initial marking event (e.g., single-pass removal) and a subsequent recapture or resight event (e.g., snorkeling) in a closed population (i.e., no immigration or emigration over time interval), whereas more elaborate “robust sampling designs” use multi-year trapping periods with a set of closely spaced sampling occasions within each year (Pollock 1982; White et al. 1982; Burnham et al. 1995a). Alternatively, population trend can be estimated based on individual recapture information (e.g., PIT tags) from active or passive (antennae) recaptures using open estimator, temporal symmetry models (e.g., Al-Chokhachy and Budy 2008). Temporal symmetry models can account for the reduced capture probabilities generally associated with cryptic species and potential differences in capture probabilities across groups within populations (e.g., Peterson et al. 2004). Further, recaptures obtained via passive PIT-tag antennae can provide a means for additional recaptures of previously PIT-tagged individuals without further harassment and continuous sampling within and across years.

Despite these advantages of mark-recapture data for assessing abundance and trend, the information provided by mark-recapture techniques (abundance or trend) can be limited by: 1) low capture and/or recapture rates which affect precision (Al-Chokhachy et al. 2009); 2) patchy distribution and low-densities of fish; 3) high costs associated with mark-recapture techniques (including expensive passive antennae); and 4) feasibility issues in large rivers, as block nets not possible resulting in violations of model assumptions (Burnham et al. 1995b). Nevertheless, mark-recapture techniques allow the simultaneous estimation of key vital rates like growth and survival, as well as relatively accurate and precise estimates of abundance or trend.