Fisheries Investigations in the Big Blackfoot River Basin, 2011-2012

by

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Montana Fish, Wildlife & Parks

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PART III: Restoration and tributary assessments (cont.)

PART IV: Special Studies

APPENDICES

INTRODUCTION

For over two decades, the Blackfoot River Basin has been the site of a wild trout restoration and conservation initiative. This initiative began in 1988-89 when fisheries assessments identified: 1) the over-harvest of native trout, 2) basin-scale stream degradation, and 3) toxic mine waste in the headwaters of the Blackfoot River as limiting Blackfoot River fisheries (Peters and Spoon 1989, Peters 1990, Pierce et al. 1997). These early findings led to the adoption of protective angling regulations in 1990 followed by the implementation of pilot-level restoration projects. By the mid-1990s, improved fisheries and social acceptance of the restoration initiative led to the incremental development of a private lands restoration methodology for the Blackfoot River and the expansion of tributary restoration activities from the mid-1990s to the present (Aitkin 1997, Pierce et al. 1997, 2005, 2011, 2013; BBCTU 2013). While the guiding philosophy of "wild trout" conservation provides the foundation for this endeavor, the cooperation of many resource agencies, conservation groups, private landowners and a network of volunteers (i.e., *Blackfoot Cooperators* - see below) form the social and technical network necessary to fund and implement the initiative. This initiative provides a more specific framework for the recovery of dwindling stocks of imperiled native trout when integrated with targeted harvest regulations, site-specific restoration and landscape protection (i.e., conservation easements) in ecologically critical areas of the watershed.

Blackfoot River restoration is an iterative tributary-based priority-driven process whereby the scope and scale of restoration expands as information and stakeholder cooperation is generated (Pierce et al. 2005). This information-based process usually begins with fisheries assessments, which often lead to restoration activities targeting individual tributary stocks. Restoration methods include enhancing flows in rearing areas, preventing juvenile fish loss to irrigation in migration corridors, reconstructing damaged streams, fencing livestock from spawning areas, while expanding these types of actions to adjacent tributaries as human-induced limiting factors are identified and as opportunities allow. Within this process, monitoring and project evaluation provide the mechanism to identify measures of biological effectiveness, while also identifying where additional work (i.e., adaptive management) is required.

During the last 25 years, Montana Fish, Wildlife and Parks (FWP) has completed fisheries surveys and/or habitat assessment on >180 streams, including all major tributary streams within the Blackfoot River Basin. These investigations have identified human-induced fisheries impairments on a great majority (over 80%) of low-elevation water bodies (Pierce et al. 2008). With information derived from these and related investigations (e.g., biotelemetry), and with the cooperation of many stakeholders, the Blackfoot Cooperators have now targeted about 50 tributaries with >600 individual fisheries-related (Pierce et al. 2008, BBCTU 2013). Correcting environmental (riparian) damage over large tracts of mixed land ownership involves protection (e.g., conservation easements), restoration and improved management of biologically important but fisheries-impaired streams. Improving riparian/aquatic habitat involves both passive (e.g., compatible grazing) and active (e.g., channel reconstruction) measures depending on the degree of riparian degradation and a stream's recovery potential. The geographic focus of stream improvement activities has been tributaries from the North Fork down-valley and bull trout "core area*"* streams (MBTRT 2000), which includes streams classified as critical habitat for the recovery of bull trout (USFWS 2010).

Currently, fisheries restoration and related conservation measures are expanding to streams in the Lincoln Valley and Clearwater River Valley. Restoration and conservation actions in the broader Lincoln Valley are especially important because this area harbors genetically pure native westslope cutthroat trout across a broad geographic area. This region of the basin may, in fact, hold the highest potential for native cutthroat trout conservation based on sub-basin scale trout response trends to restoration (Pierce et al. 2013, Results Part IV). In addition to the Lincoln Valley, the upper North Fork Basin upstream of the North Fork Falls (within the Scapegoat Wilderness), is now being considered for the translocation of native trout. Here, historic introductions of non-native rainbow trout and Yellowstone cutthroat trout have hybridized with native westslope cutthroat trout. Hybrids are present in low abundance, seem to provide little ecological value as a result, and pose risks of hybridization to native cutthroat trout downstream of the North Fork Falls (Pierce et al. 2008).

In addition to the expansion of fisheries improvement activities, stakeholder involvement in the fisheries initiative continues to evolve among non-profit conservation groups (NPG), natural resource agencies and landowners. From the beginning of the endeavor, the Big Blackfoot Chapter of Trout Unlimited (BBCTU) has been the leading NPG involved in river conservation actions. However, the Blackfoot Challenge (BC), and Clearwater Resource Council (CRC) also coordinate fund-raising, help facilitate conservation easements and promote educational programs, drought management and related forest restoration strategies. Likewise, The Nature Conservancy (TNC), Five Valleys Land Trust (FVLT) and Clark Fork Coalition have all engaged in the development of various river-based conservation activities. The combined services of natural resource agencies or other government entities from the county to the federal level likewise contribute to river conservation. These entities [North Powell Conservation District (NPCD), Department of Natural Resources and Conservation (DNRC), U.S. Fish and Wildlife Service – Partners for Fish and Wildlife (USFWS), Natural Resource Conservation Service (NRCS), and the U. S. Forest Service (USFS)] provide a wide range of resource expertise, project funding and technical services geared to improving stream and fisheries resources. Private landowners, private foundations and volunteers also contribute significant resources to fisheries-related projects. Together this affiliation, the *Blackfoot Cooperators*, form the general support base of the Blackfoot River Fisheries Restoration Initiative.

From 2011 to the present, the Blackfoot Cooperators continued river conservation initiatives on several fronts. In addition to the ongoing restoration actions in tributaries, ecologically important activities now include the: 1) clean-up of the Mike Horse Mine, a contaminated mining area in the upper Blackfoot River, 2) the development of aquatic habitat improvements associated with the "Southwest Crown of the Continent Project" on the Lolo and Helena National Forests, and 3) the early development of a native trout restoration project in the Scapegoat Wilderness upstream of the North Fork Falls. Despite many past and current river-based initiatives, adverse human pressures upon salmonid habitat (and native trout specifically) in the Blackfoot River Basin are widespread and continue to pose a daunting future conservation challenge.

EXECUTIVE SUMMARY

The 2011-2012 reporting period was marked with two years of favorable summer flow conditions in the Blackfoot River. These conditions represent the 4th and 5th consecutive year (2008-2012) of more normal period of flow conditions. This compares with a prior seven-year period (2000-2007) of continuous drought (Figure 1) as well as a period marked by the range expansion of *Myxobolus cerebralis* (Pierce et al. 2009). Likewise, summer water temperatures between 2008 and 2012 were lower throughout the Blackfoot Basin than previous drought years, and thus more favorable to coldwater fisheries (Figure 2).
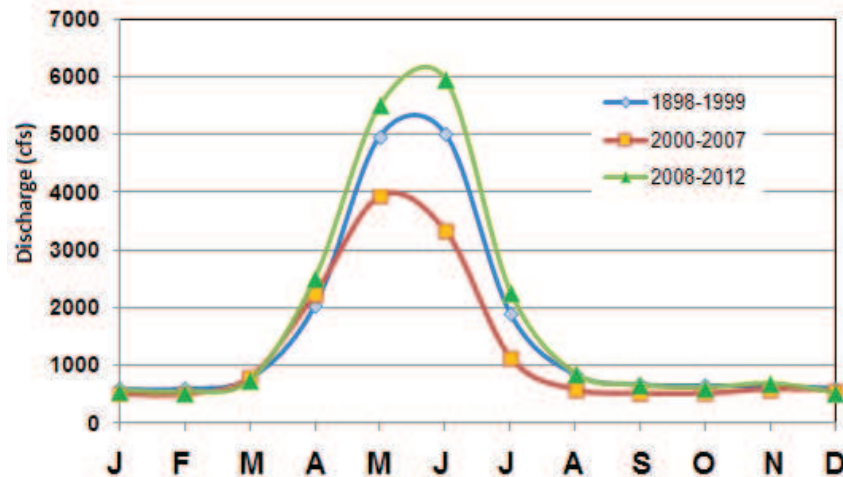
Figure 1. Three Blackfoot River hydrographs: The blue hydrograph shows the long-term (1898-1999) mean. The red hydrograph shows the 7-year mean for the 2000-2007 period of drought. The green hydrograph shows the 5-year mean for 2008-2012, a period marked by favorable river flows (USGS Bonner gauge, provisional unpublished data).
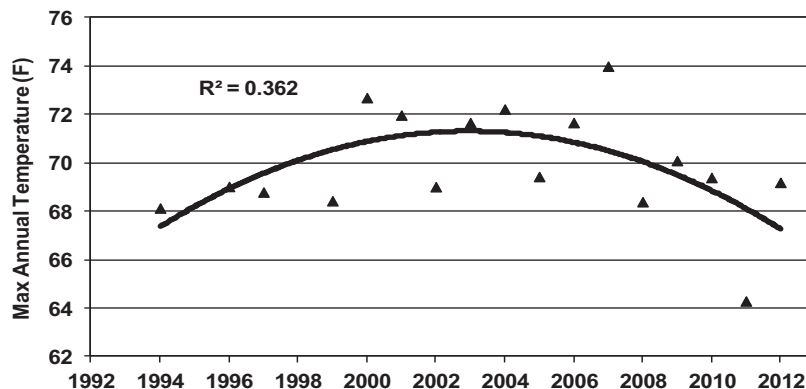
Figure 2. Maximum annual water temperatures for the lower Blackfoot River downstream of Belmont Creek, 1994-2012.

From 2011 through 2012, we resurveyed fish populations in six long-term monitoring sites of the Blackfoot River and compared these survey results to long-term trends dating to pre-restoration (i.e., 1988-89) period (Results Part II). Between 2010 and 2012 population surveys in the Johnsrud and Scotty Brown Bridge Sections indicate an increase in both total trout biomass and total trout abundance (Figures 3 and 10). Trout densities in the Wales Creek section of the Blackfoot River (downstream of Nevada Creek) continue to show a 10-year trend of low trout abundance and biomass (Figure 3 and 10, Results Part II, Appendix C). Long-term trends in trout species composition show increases in westslope cutthroat trout throughout the mainstem Blackfoot River with the exception of the upper most segment of the upper River downstream of the

Heddleston mining district where mining pollutants have led to broad ecological damage including the collapse of cutthroat trout in the upper-most Blackfoot River (Moore et al. 1991)

In addition to fisheries monitoring results from the Blackfoot River, we present the results of fisheries-related assessments for 25 tributaries involving restoration activities (Results Part III). These restoration activities show great promise for improvement of wild trout populations, including native trout, even in the presence of drought and whirling disease, once damaging land (riparian) activities and other human-related factors limiting populations are corrected. In addition to identifying many environmentally beneficial projects, assessments of restoration activities shed light on the complexities of native fish recovery as well as the inherent challenges of ensuring effective and sustainable restoration outcomes when implemented in areas of intensive land-use and/or areas of mixed private and public lands.
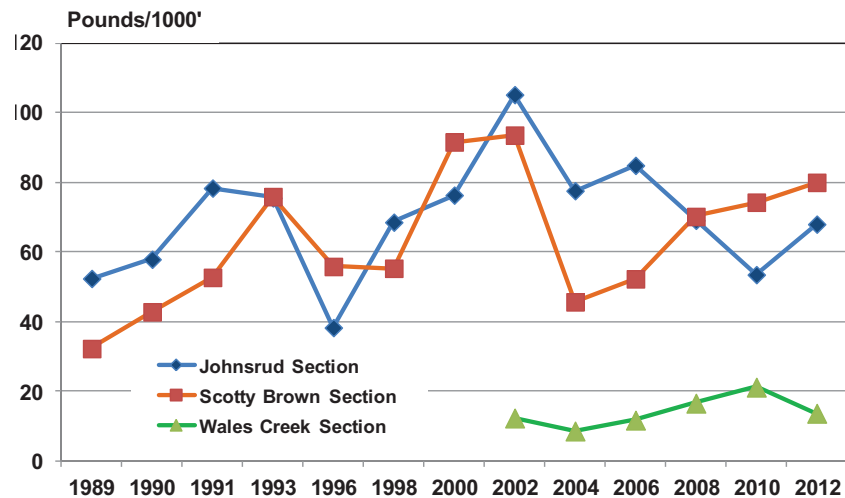
Figure 3. Total Trout biomass estimates (all trout >6.0") for three section of the lower Blackfoot River, 1989-2012.

With >20 years of restoration activities now complete (Pierce et al. 2008, 2011, 2013; BBCTU 2013), clearly one of the more pressing restoration challenges relates to the monitoring, maintenance and adaptive management needs associated with the long-term sustainability of fisheries improvement projects. This responsibility is inherent to the restoration process but poorly employed across the American West (Roni 2005). In the Blackfoot River Basin, monitoring needs now go far beyond fisheries response and include the maintenance of fish screens, fish ladders and other infrastructure as well as monitoring of instream flow projects and riparian grazing systems enacted as a habitat maintenance and/or fisheries restoration techniques. Successful riparian grazing systems are especially complex because they require and understanding of (geomorphic and vegetative) site potential (e.g., Hansen et al. 1995, Rosgen 1996), riparian healing processes and the sensitivity of target salmonid species to grazing disturbance - all conditions that vary greatly across riparian ecosystems. As a result, riparian grazing systems usually require consistent monitoring against established targets in order to effectively improve riparian condition.

In Results Part IV, we also present the results of several special studies, including the preliminary finding of three spring creek-related studies. The first is a radio-telemetry study that examines the spawning behavior of westslope cutthroat trout in the

Nevada Spring Creek complex. This study shows that strategic restoration can promote the migratory life history of westslope cutthroat trout, while improving environmental conditions for migratory fish from other areas of the upper Blackfoot Basin. The second study focuses on temperature reduction and whirling disease following the reconstruction of Kleinschmidt Creek. This second study documents water temperature reduction following active restoration but found no corresponding reduction in the severity of whirling disease. The third study examines the response of wild trout following full reconstruction of Kleinschmidt Creek. This study shows that full channel restoration can significantly increased the density and biomass of wild trout, and that the use of instream wood for habitat improvement promotes more immediate increases in wild trout in certain vegetated stream-types, but that value of instream wood is reduced with time.

Finally, we present the final results of two published studies in Results Part IV. The first study focuses on the risk of *Myxobolus cerebralis*, the cause of salmonid whirling disease, to mountain whitefish – an ecologically important but under-appreciated native salmonid. The *M. cerebralis* parasite was first detected in the Blackfoot Basin in 1995. Since then, the parasite has continued to expand its geographic range. This study shows high infection rates of *M. cerebralis* infection in age 0 whitefish in a groundwater-induced reach of the upper Blackfoot River. This environment is more conducive to the proliferation of the *M. cerebralis* parasite than the lower river. The second study is a long-term basin-scale evaluation of fisheries response to restoration activities in tributaries of the Blackfoot River over the last 20 years. This study involves 18 restoration streams, each with a minimum of 5 years post-treatment fisheries monitoring data. This evaluation shows highly variable trout responses among individual treatment streams, but positive overall long-term trends in trout abundance for the streams when examined at a basin-scale with trends towards native trout improvement in the mid- to upper-basin. In addition to evaluating the efficacy of restoration, this study should help focus native trout restoration in the mid- to upper basin (e.g., Lincoln Valley), while also prompting more comprehensive review of land-use plans in order to help sustain positive restoration outcomes.

As a continuation of 12 prior FWP fisheries reports written between 1988 and 2010, this report is intended to document various studies influencing wild trout fisheries of the Blackfoot River basin, and to specifically: 1) summarize population metrics of wild trout in the Blackfoot River over a 25 year monitoring period, 2) describe fisheries and habitat monitoring activities associated with restoration of tributaries, and 3) help guide wild trout restoration and other river conservation actions with an emphasis on the upper Blackfoot Basin.

STUDY AREA

The Blackfoot River, located in west-central Montana, begins at the junction of Beartrap and Anaconda Creeks (within the Upper Blackfoot Mining Complex), and flows west 132 miles from the base of the Continental Divide to its confluence with the Clark Fork River at Bonner, MT (Figure 4). The Blackfoot River is one of twelve renowned "blue ribbon" trout rivers in Montana with a 1972 appropriated "Murphy" in-stream flow water right of 700 cfs at the USGS Bonner (#12340000) gauging station. Mean annual discharge is 1,589 (cfs) near the mouth (USGS 2013 provisional data). This river system drains a 2,320-mile^2 watershed through a 3,700-mile stream network, of which about 1,900 miles are perennial streams capable of supporting fishes. The physical geography of the watershed ranges from high-elevation glaciated alpine meadows, timbered forests at the mid-elevations, to prairie pothole topography on the valley floor. Glacial landforms, moraines and outwashes, glacial lake sediments and erratic boulders variably cover the floor of the entire Blackfoot River valley and exert a controlling influence on the physical features of the Blackfoot River and the lower reaches of most tributaries. With the removal of Milltown Dam in 2008, the Blackfoot River is now a free flowing river to its confluence with the Clark Fork River.

The Blackfoot River is also one of the most popular, scenic, physically diverse and biologically complex rivers in western Montana. Angler pressure on the Blackfoot River was estimated at 39,365 angler days in 2011 (Montana Fish Wildlife and Parks, 2012). Despite its popularity, segments of the river system support low densities of wild trout due to an array of natural conditions and human impairments.
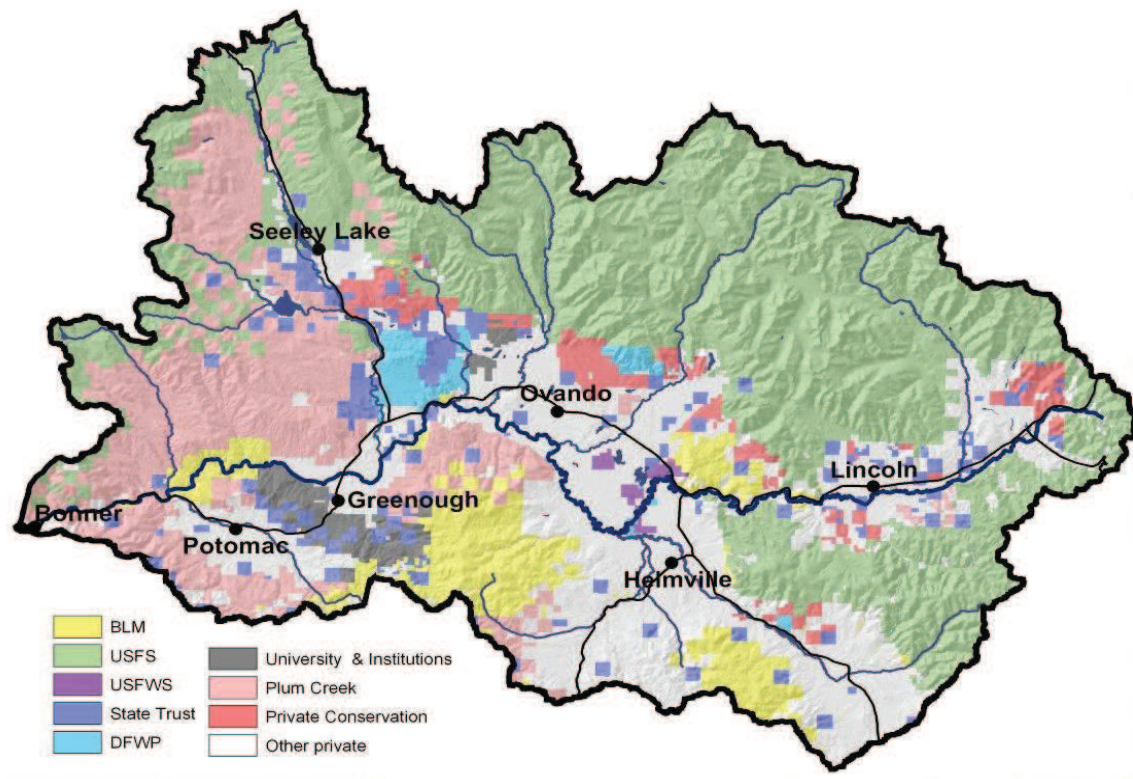

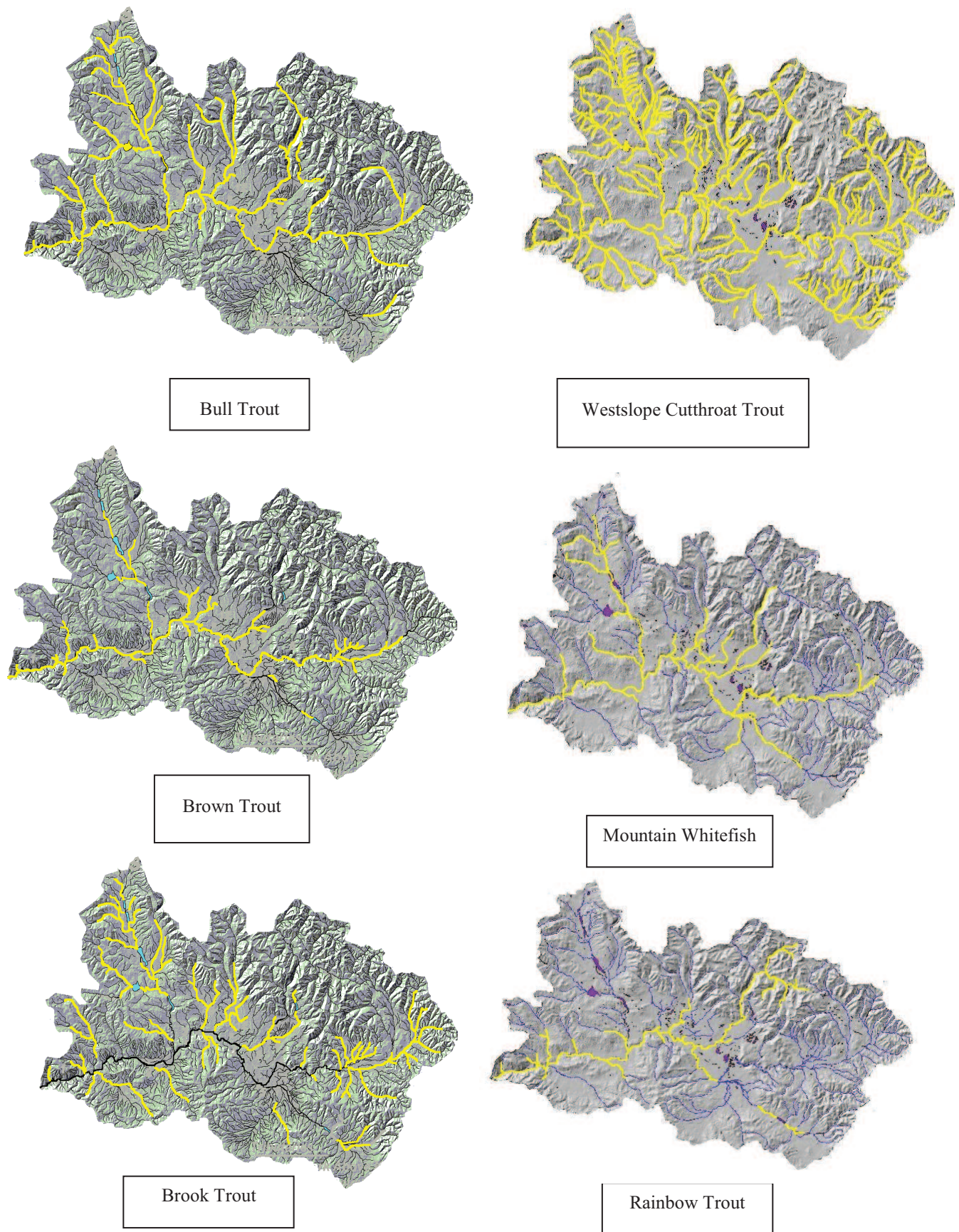Figure 4. Land ownership map of the Blackfoot River Watershed.

Figure 5. Generalized distribution of six salmonid species within the Blackfoot Basin.

Populations levels of imperiled native trout (westslope cutthroat trout - *Oncorhynchus clarkii lewisi* and bull trout - *Salvelinus confluentus*) are particularly low. Natural conditions limiting trout fisheries involve drought stressors, areas of high instream sediment loads, low instream productivity, naturally intermittent tributaries, summer warming and periods of severe icing of the lower mainstem river channel. Human impairments include mining-related contamination in the upper Blackfoot Basin, the spread of exotic organisms (e.g., *M. cerebralis* and nonnative fish) and human perturbations on >80% of tributaries (Pierce et al. 2005, 2008). The sum of natural conditions and human impairments produces an array of trout assemblages that vary regionally within the watershed and longitudinally among river and tributary reaches.

Land ownership in the Blackfoot River Basin is a mix of public and private: 27% is owned by private land owners, 9% by the Plum Creek Timber Company; 46% is managed by the USFS, 11% by the state of Montana, and 7% by the BLM. In general, public lands and large tracts of Plum Creek Timber Company properties comprise large forested tracts in mountainous areas of the watershed, whereas private lands are found in the foothills and lower valley areas (Figure 4). Traditional land-use in the basin includes mining, timber harvest, agriculture and recreation, all of which have contributed to habitat degradation and/or past fish population declines. The majority of habitat degradation occurs on the valley floor and foothills of the Blackfoot watershed and largely on private agricultural ranchlands. However, riparian degradation also extends to commercial timber lands and mining districts, as well as some state and federal public lands.

Distribution patterns of most salmonids generally conform to the physical geography of the landscape, with species richness increasing longitudinally in the downstream direction (Figure 5). Species assemblages and densities of fish can also vary greatly at the lower elevations of the watershed. Native species of the Blackfoot Watershed are bull trout (*Salvelinus confluentus*), westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), mountain whitefish (*Prosopium williamsoni*), pygmy whitefish (*Prosopium coulteri*), longnose sucker (*Catostomus catostomus*), largescale sucker (*Catostomus macrocheilus*), northern pikeminnow (*Ptychocheilus oregonensis*), peamouth (*Mylocheilus caurinus*), redside shiner (*Richardsonius balteatus*), longnose dace (*Rhinichthys cataractae*), slimy sculpin (*Cottus cognatus*) and mottled sculpin (*Cottus bairdi*). Non-native species of the Blackfoot Watershed include rainbow trout (*Oncorhynchus mykiss*), kokanee (*O. nerka*), Yellowstone cutthroat trout (*O. clarki bouvieri*), brown trout (*Salmo trutta*), brook trout (*S. fontinalis*), arctic grayling (*Thymallus arcticus*), white sucker (*Catostomus commersoni*), fathead minnow (*Pimephales pomelas*), northern pike (*Esox lucius*), brook stickleback (*Culaea inconstans*), pumpkinseed (*Lepomis gibbosus*), largemouth bass (*Micropterus salmoides*) and yellow perch (*Perca flavescens)*.

Most salmonids (westslope cutthroat trout, bull trout, rainbow trout and brown trout, mountain whitefish) in the main stem Blackfoot River system exhibit migratory (fluvial) life-history characteristics. With the exception of mountain whitefish (Results Part IV), migratory behavior usually involves spawning and rearing in tributaries. Native fishes within the Clearwater basin also exhibit migratory (adfluvial) life-histories, which include lake-dwelling behavior marked by tributary spawning. Westslope cutthroat trout has a basin-wide distribution and is the most abundant species in the upper reaches of the

tributary system; however, westslope cutthroat trout abundance decreases in lower reaches of the tributary system due to habitat impairments and interactions with nonnative trout. Bull trout distribution extends from the mainstem Blackfoot River to headwaters of larger tributaries north of the Blackfoot River main stem, including the Clearwater River Basin. Rainbow trout distribution is limited to the Blackfoot River downstream of Nevada Creek and lower reaches of the lower river tributaries. Rainbow trout occupy ~10% of the perennial streams in the Blackfoot watershed, with river populations reproducing primarily in the lower portions of larger south-flowing tributaries. The exceptions to this include the upper North Fork (upstream of the North Fork Falls and within the Scapegoat Wilderness) and Nevada Creek where historic fish stocking in lakes and reservoirs has led to establishment of rainbow trout or Yellowstone cutthroat as well as various hybrids. Brown trout inhabit ~15% of the perennial stream system with a distribution that extends from the Landers Fork down the length of the Blackfoot River and into the lower foothills of the tributary system. Mountain whitefish occupy ~20% of the basin, including the larger, colder streams, similar to bull trout distribution. Brook trout are widely distributed in tributaries, but rare in the main stem Blackfoot River below the Landers Fork.

PROCEDURES

Methods associated with Results Part II and III are identified below. Methods related to Special Studies (Part IV) are located within those reports.

Fish Population Estimators

Fish were captured using either a boat or backpack-mounted electrofishing unit. In small streams, we used a battery powered (Smith/Root) backpack mounted direct current (DC) electrofishing unit. The anode (positive electrode) was a hand-held wand equipped with a 1-foot-diameter hoop; the cathode (negative electrode), a braided steel wire. On the Blackfoot River, we used an aluminum drift boat mounted with a Coffelt Model VVP-15 rectifier and 5,000 watt generator. The hull of the boat serves as the cathode and two fiberglass booms, each with four steel cable droppers, serve as anodes. We used DC waveform with output less than 1,000 watts, which is an established method to significantly reduce spinal injuries in fish associated with electrofishing (Fredenberg 1992). Resident fish, including young-of-the year (age 0), were intensively sampled in the tributaries from August to November to enable comparisons of abundance between years and sampling sections. Captured fish were anesthetized with either tricaine methanesulfonate (MS-222) or clove oil, weighed (g) and measured (mm) for total length (TL). For this report, we converted all weights and lengths to standard units.

Fish population surveys relied on mark-and-recapture, multiple-pass depletion estimates of trout abundance and/or a catch-per-unit-effort (CPUE) statistic for small stream surveys. For the Blackfoot River below Lincoln we used a modified Petersen mark-and-recapture estimator. Using this method, estimates are considered valid if recaptures are $\geq$ four fish. Similar to tributaries, we used a depletion estimator on the upper-most mainstem of the Blackfoot River (upstream of Lincoln) to estimate trout abundance. Age class breaks (e.g., age 0 verses age 1+) were based on length-frequency histograms. All estimates of abundance in this report were calculated at the 95% level of confidence. Trout species composition for Blackfoot River was calculated as a percent of the total catch for fish $\geq$6.0". All sampling locations are referenced by river-mile or stream-mile.

For all Blackfoot River population surveys using mark-and-recapture, we also estimated biomass and calculated the Fulton condition factor (an index of "plumpness" where higher values indicate better condition; Murphy and Willis 1996) using Fisheries Analysis Plus software (FA +). The formulas for these calculations are:

$$N = \frac{(m+1)\ (c+1)\ -1}{r+1}$$

$$Biomass\ Estimate = N\ (Wt)$$

$$CF(standard) = (Wt_L\ /\ (L_L)^3)\ 100,000$$
$$CF(metric) = (Wt_L \times 3612.8)\ /\ (L_L\ /10)^3$$

N= population point estimate
m= the number of marked fish
c= the number of fish captured in the recapture sample

r = the number of marked fish captured in the recapture sample
CF = condition factor
Wt_L = average weight of length group
L_L = average length of length group

Standard deviations (SD) for the mark-and-recaputure surveys were calculated using the equation:
$SD = sqrt \{((m+1)(c+1)(m-r)(c-r)) / ((r+1)^2(r+2))\}$

The 95% confidence intervals (CI) were calculated using the equation:
$$1.96*SD$$

For fish population estimates in small streams, we used a standard two-pass depletion estimator and standard equations for calculating variance. For this estimator:
$$N = \frac{(n_1)^2}{n_1 - n_2}$$

$$P = \frac{n_1 - n_2}{n_1}$$

Where:

    N = point estimate,
    n_1 = the number of fish collected on the first pass
    n_2 = number of fish captured on the second pass
    P = probability of capture (>0.5 for n>50 or >0.6 for n<50 for valid estimates)

And,   SD   = $\dfrac{n_1 n_2 (n_1 + n_2)^{-2}}{(n_1 - n_2)^2}$

And, the 95% confidence interval for N = 1.96 (SD).

In those few cases where a three-pass estimator was necessary, we used a maximum likelihood estimator using the Lockwood and Schneider (2000) formula:

$$N = [n + 1 / n - T + 1] [kn - X - T + 1 + (k - i) / kn - X + 2 + (k - i)]_i < 1.0$$

Where n is the smallest integer satisfying Equation. Probability of capture (p) and variance of N are then estimated by:
$$p = \frac{T}{kN - X}$$

Variance of N = $\dfrac{N(N - T)}{T^{2} - N(N - T)[(kp)^{2} / (1 - p)]}$

Where,

    N = point estimate
    i = pass number,
    k = number of removals (passes),
    C_i = number of fish caught in i[th] sample,

X = an intermediate statistic used below,

T = total number of fish caught in all passes.

Standard error of N = Square root of variance of N.

95% Confidence intervals (CI) were calculated using $N + 1.96$(standard error)

For small stream fish population surveys, we commonly use an intensive single-pass electrofishing CPUE method as a simple measure of relative abundance. CPUE refers to the number of fish collected in a single electrofishing pass and is adjusted per 100' of stream (i.e. CPUE of 8 means 8 fish captured per 100' of sampled stream). For small streams surveys in this report, we refer to CPUE as Catch/100' and depletion estimates of abundance as Trout/100'. We refer to mark-and-recapture estimates of abundance in the larger water bodies (i.e., Blackfoot River and Nevada Creek) as Trout/1,000'. CPUE catch statistics are located in Appendix A. Depletion estimates are located in Appendix B. Mark-and-recapture estimates of abundance, biomass and condition factors for the Blackfoot River are located in Appendices C and D.

Water Temperatures

Water temperatures (^{o}F) were continuously recorded at either 50- or 72-minute intervals using Hobo temperature (72-minute) or tidbit (50-minute) data loggers. All raw data plotted for each station and monthly summary statistics are located in Appendix F. For this report we also standardized many temperature summaries using July (the identified period of peak warming) data and display median, quartile and minimum and maximum temperatures values consistent with other (e.g., TMDL) temperature summaries within the Blackfoot River Basin.

Stream Habitat Surveys

Habitat surveys for small streams typically begin with a Level II Rosgen (1996) geomorphic survey. Once the geomorphic setting is measured, supplemental habitat measurements are usually taken. Depending on the individual stream, habitat surveys vary by intensity (50%, 33% or 25%), begin with a randomly selected downstream habitat unit and proceed in an upstream direction. Beginning at a selected pool, we measure: 1) maximum pool depth and the downstream riffle crest depth to calculate the residual depth, 2) wetted width and bankfull width at the maximum pool depth and at the riffle crest, and 3) total pool length. Pool frequency was then calculated by measuring the survey distances using either 1:24,000 scale maps or aerial photos. A total census of large wood (> 4" in diameter and >6' in length) is usually performed for all habitat units throughout the entire length of the survey on all streams. We also note overhead canopy and under-story vegetation and anthropogenic stream degradation. In addition, Wohlman pebble counts are conducted at a minimum of one representative riffle. During these surveys, continuous water temperature recordings are taken during summer period, and stream discharges are measured using a Marsh-McBirney flow meter during low flow conditions.

Working with Private Landowners

Typically, each tributary restoration project involves multiple landowners, professional disciplines, funding sources, and involvement of a watershed group.

Restoration typically focus on correcting obvious impacts to fish populations such as migration barriers, stream de-watering, fish losses to irrigation canals, and degraded riparian areas. All projects are cooperative endeavors between private landowners and the restoration team, and occur throughout the drainage. Projects are facilitated at the local level by agency resource specialists in cooperation with two watershed groups (BBCTU, BC, CRC) or local government groups such as the North Powell Conservation District (NPCD) or state and federal agencies such as Montana Fish, Wildlife and Parks (FWP) or U.S. Fish and Wildlife Service, Partners for Fish and Wildlife (USFWS). The non-profit status (i.e., 501(c)3) of watershed groups provides a mechanism for generating tax-deductible private funds.

FWP biologists identify priorities by performing fisheries studies, communicating biological findings, reviewing proposed fisheries projects, providing funding support and monitoring fisheries on completed projects. Federal (USFWS, USFS and NRCS) biologists and other agency specialists (NRD, DNRC) help develop and fund projects usually in conjunction with watershed groups, landowners and FWP. Agency staff and project leaders generally enlist help from interagency personnel or consultants including range conservationists, hydrologists, engineers, and water right specialists as necessary. Watershed groups help with fundraising, administration of budgets, bid solicitation, application of permits, overseeing consultants and contractors, assisting with landowner contacts, coordinating volunteers, helping resolve local conflicts and addressing other social issues.

Project funding comes from many sources including landowner contributions, private donations, foundation grants, and state and federal agencies. Project managers from agencies and watershed groups jointly undertake fundraising. BBCTU generally obtains project permits on behalf of cooperating landowners. Project bids (consulting and construction) conform to State and Federal procurement policies. These policies included the development of a Blackfoot watershed *qualified vendors lists* (QVL) derived through a competitive process, which is managed primarily through BBCTU. A minimal project cost triggers use of the QVL. The watershed groups solicit bids from the QVL for both consulting and contractor services. Bid contracts are signed between the watershed group and the selected vendor upon bid acceptance.

Depending on the specific project, landowners are responsible for certain costs, construction and project maintenance once projects are completed. Addressing the source of stream degradation usually requires developing riparian/upland management options (i.e., grazing strategies) sensitive to the requirements of fish and other riparian-dependent species. Written agreements (15-30 year period) with landowners to maintain projects are arranged with cooperators on each project. Landowner awareness of the habitat requirements of fish and wildlife and their full participation and commitment to project goals and objectives are crucial to the long-term success of the restoration initiative. We encourage landowners to participate fully in all phases of restoration from fish population data collection and problem identification to project development, monitoring and adaptive management of completed projects. Although many restoration projects have been completed in the Blackfoot River watershed, this effort is still considered educational at a broad level and is far from complete in some areas of the basin.

RESULTS/DISCUSSION

PART I: Blackfoot River Environment Blackfoot River Discharge: Provisional USGS data at the Bonner gauging station #12340000

From 2011 through 2012, the Blackfoot River watershed was subject to two consecutive years of above normal runoff (Figure 6). Peak flows in 2011 were approximately double normal flow. Whereas high flows in 2012 occurred earlier than normal, however river flows remained near normal during early summer and baseflow periods (USGS provisional data).

Discharge (cfs)
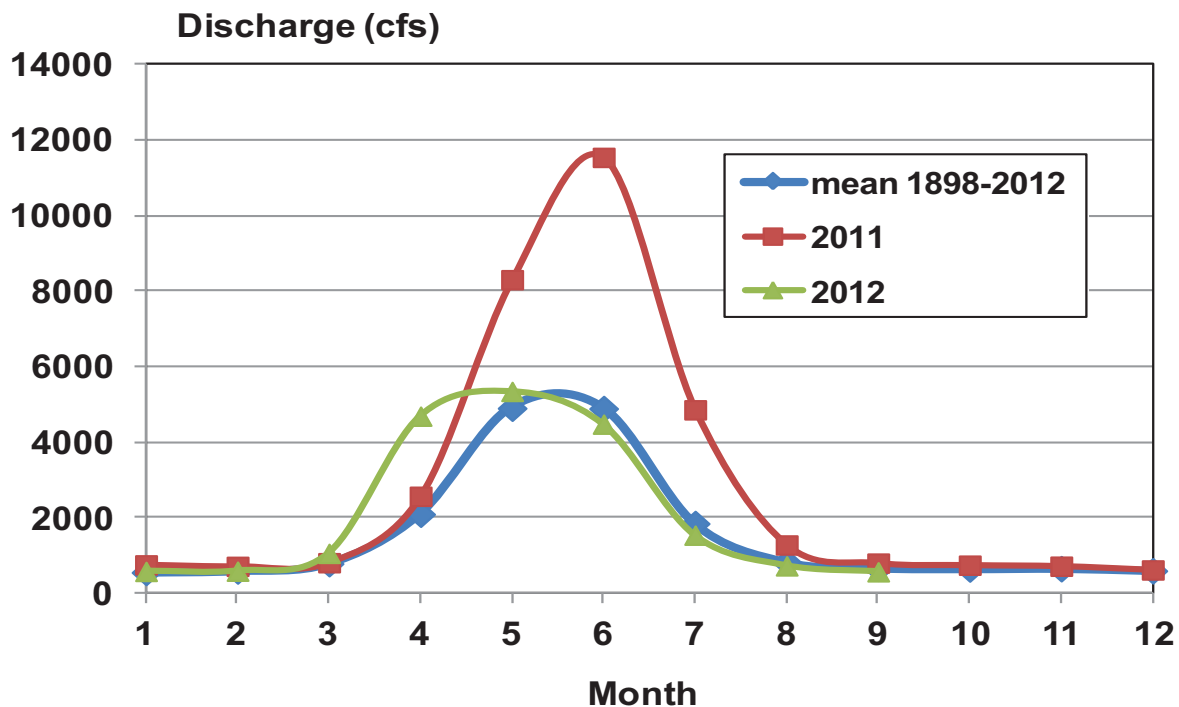
Figure 6. A comparison of the 2011 and 2012 Blackfoot River hydrograph against the long-term mean. Note the 2011 high flow year (USGS provisional data, Bonner gauge station #12340000). Also note, flow data from October through December 2012 was not available at the completion of this report.

Blackfoot River and tributary temperatures

Water temperature monitoring during 2011-12 involved 50 season-long water temperature recording at 25 tributary locations plus eight sites in the Blackfoot River (Figure 7, Results Part III, Appendix F). Temperature data were collected in order to: 1) monitor long-term trends at various sites throughout the Blackfoot watershed, 2) assess restoration projects for temperature reduction, 3) identify thermal regimes (natural and anthropogenic) favorable and unfavorable for trout' 4) monitor temperature triggers of the Blackfoot Emergency Drought Plan, and 5) study migratory and spawning behavior

of westslope cutthroat trout radio-tagged in Nevada Creek and Grantier Spring Creek. For many streams, water temperatures exceed >70 °F, which we define in this report as above the optimal range of most salmonids and temperatures >65 °F, which are considered excessively warm for bull trout.

Water temperature data is used throughout this report. Plots of all data and summaries of monthly statistics are located in Appendix F. A summary of all July water temperature data for six long-term monitoring sites of the Blackfoot River are shown on Figure 8. Similar plots of water temperatures in bull trout critical habitat is shown on Figure 20.
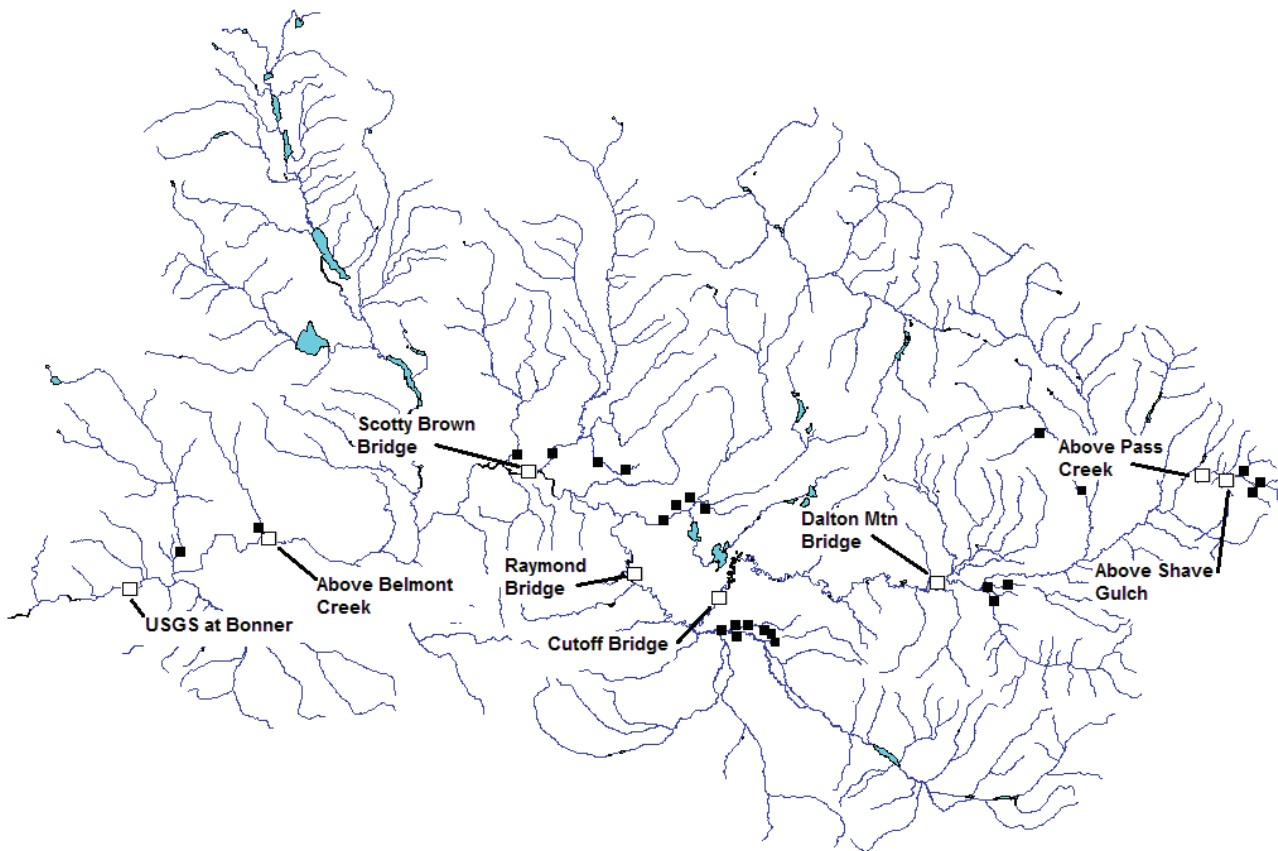
Figure 7. Temperature data collection sites in the Blackfoot Watershed for 2011-12. Names (open boxes) identify Blackfoot River monitoring sites and relate to July summary graphs in Figure 8. Black boxes show the locations of tributary water temperature monitoring sites.
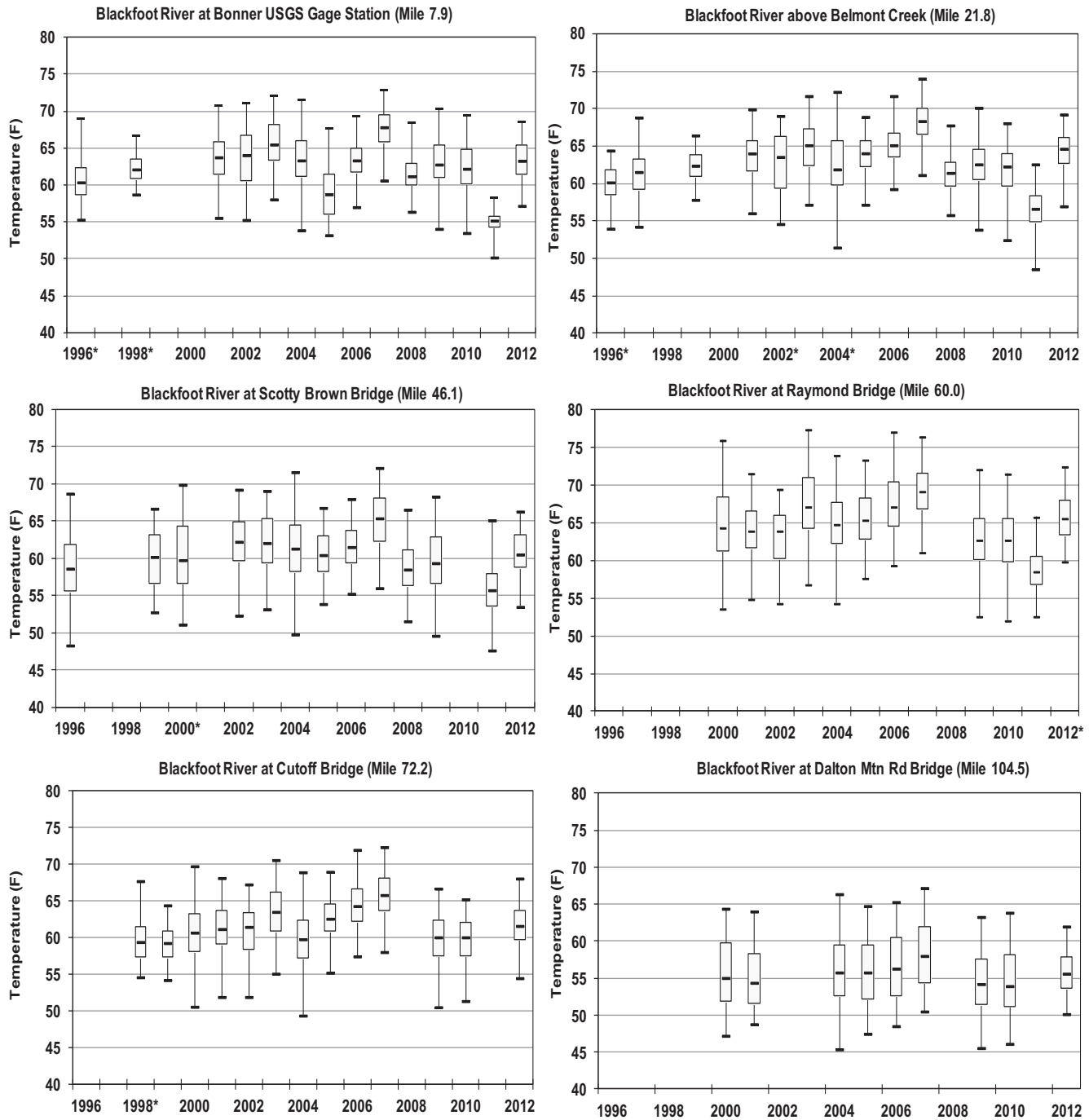
Figure 8. July water temperatures for the Blackfoot River at six long-term monitoring locations. Box plots show minimum, maximum, median and quartile values. An * denotes incomplete data for the month.

PART II: Blackfoot River Trout Populations 1988-2012

Total Trout Abundance, Biomass and Species Composition 1988-2012 - From 1988 through 2012, FWP has monitored populations of wild trout at six survey sites of the Blackfoot River (Figure 9). This monitoring period spans 25 years of wild trout conservation, which includes monitoring prior to restoration and prior to basin-wide protective angling regulations, both of which were originally initiated in 1990 (Peters and Spoon 1989, Peters 1990, Pierce et al 1997). These management actions were pursued in order to not only improve wild trout fisheries of the Blackfoot River, but also to foster the recovery of imperiled native trout that rely on tributaries as well as the Blackfoot River. This span of time also overlaps with several natural and human events that variably influenced trout populations in the Blackfoot River. These include: 1) two periods of extended drought (1988-1993, 2000-2007) followed by favorable flow years (1996-1999, 2008-2012) in both cases, 2) the introduction and range expansion of the exotic parasite *Myxobolus cerebralis* beginning in the mid-1990s, 3) a massive ice flow in the lower river in winter of 1996 that reduced trout abundance by 68 % in the Johnsrud section compared to the previous (1993) survey, and 4) the removal of Milltown Dam in 2008, which has reestablished fish passage at the mouth of the Blackfoot River.

This portion of the report summarizes population metrics for wild trout (fish >6.0") of the Blackfoot River from 1988 to 2012 using measures of total trout abundance, total trout biomass and species composition at the six long-term monitoring sites. These sites include three sites in the lower river downstream of the Nevada Creek confluence, and three sites in the upper Blackfoot River upstream of the Nevada Creek confluence (Figure 4). The three lower river sites are the: 1) Johnsrud (river-mile mid-point at 13.5, 2) Scotty Brown Bridge (mid-point at 43.9), and 3) Wales Creek (mid-point at 63.0) sections. The three upper river sites are the: 1) Canyon (mid-point at 95.3), 2) Hogum (mid-point at 119.6), and 3) Flesher (mid-point at 124.5) sections. Summaries of total trout abundance and total trout biomass for all surveys in each of the six sections are shown on Figure 10. Figure 11 shows the species composition for the survey sites. Species-specific results are located in the following section. Summaries of all statistics for 2011-12 river surveys are located in Appendices B and C.

Of the three lower Blackfoot River monitoring sites, the Scotty Brown Bridge section shows the greatest long-term improvement in total trout abundance, biomass and percent native trout. This monitoring site, located in the Ovando Valley, is an area of the Blackfoot Valley with the greatest effort to improving fisheries in adjoining spawning and rearing tributaries (Figures 9 and 10). Here, the percentage of native trout has increased from 10% in 1989 to 43% in 2012 (Figure 11). Like the Scotty Brown Bridge section, the Johnsrud section percentage of native trout increased from a low of 5% in 1989 to a high of 25% in 2012, but showed no noticeable long-term change in either total trout abundance or biomass compared to the Scotty Brown Bridge section in recent years. With the removal of Milltown dam, it appears likely the redistribution of fish between the lower Blackfoot and Clark Fork Rivers is now occurring. Population monitoring in both the Johnsrud section and the Clark Fork River will shed light on the biological response to dam removal within the next few years. The Wales Creek section of the Blackfoot River shows low total trout abundance and low total trout biomass compared to both up- and downstream monitoring sites (Figure 9 and 10). As an example of these low numbers, total trout biomass was estimated at 80 pounds/1000' in the Scotty Brown

section in 2012 compared to 12 pounds/1,000' in the Wales Creek section. Unlike the Scotty Brown Bridge section, most tributaries that enter this segment of river are fisheries impaired and provide very little, if any, access to nearby spawning streams (Pierce et al. 2008).

The three upper Blackfoot River survey sites are monitored less frequently than the lower Blackfoot River. Despite more limited data, monitoring shows increases in total trout abundance and biomass from 2006 to 2012 for fish >6.0" as well as an increase in the percentage of native trout in recent years (Figure 11). Restoration activities emphasizing native trout in the broader Lincoln valley are currently at the early stages of development. Based on tributary assessments (Results Part IV), the upper Blackfoot basin seems to have high potential native trout improvement as also indicated by recent increases in the westslope cutthroat trout in the Canyon Section of the Blackfoot River (Figure 11).
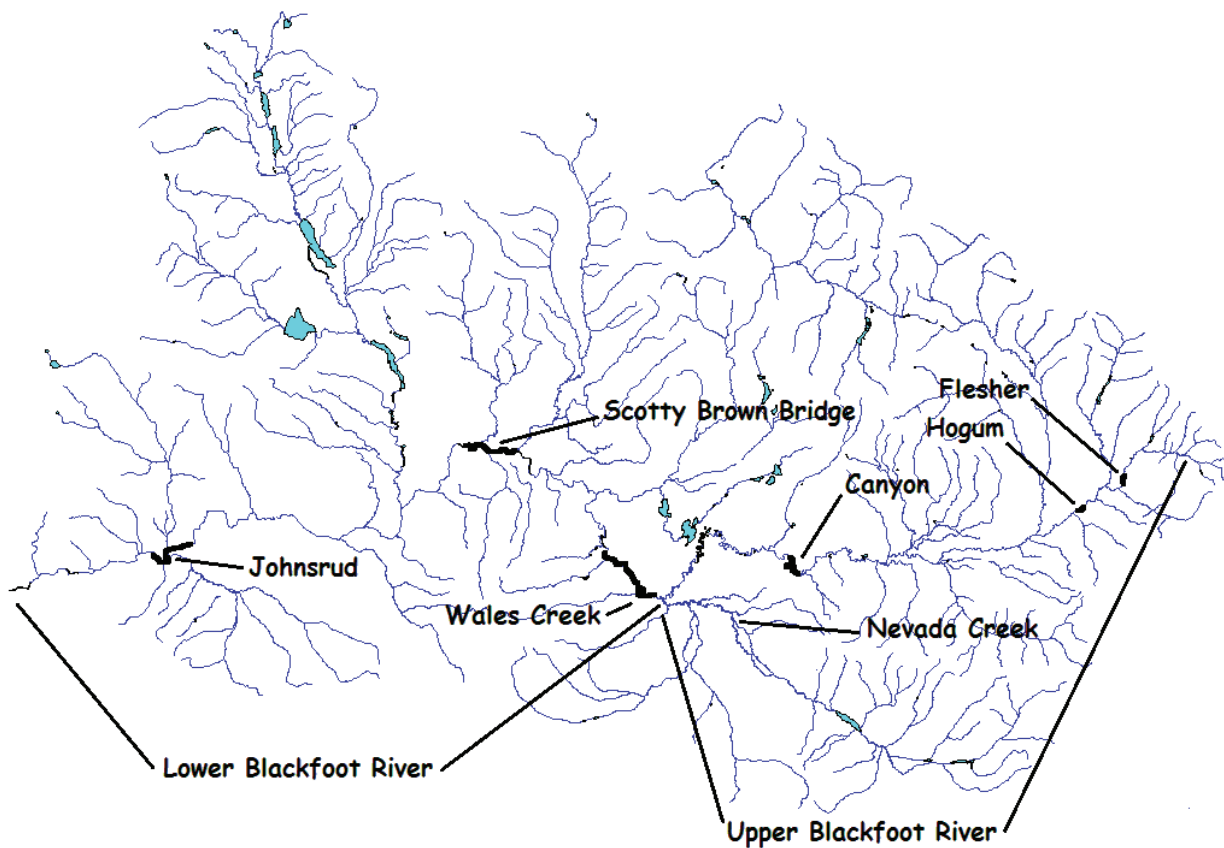
Figure 9. Location map for six long-term fish population monitoring sites on the Blackfoot River along with demarcations of the lower and upper Blackfoot River at the confluence of Nevada Creek.
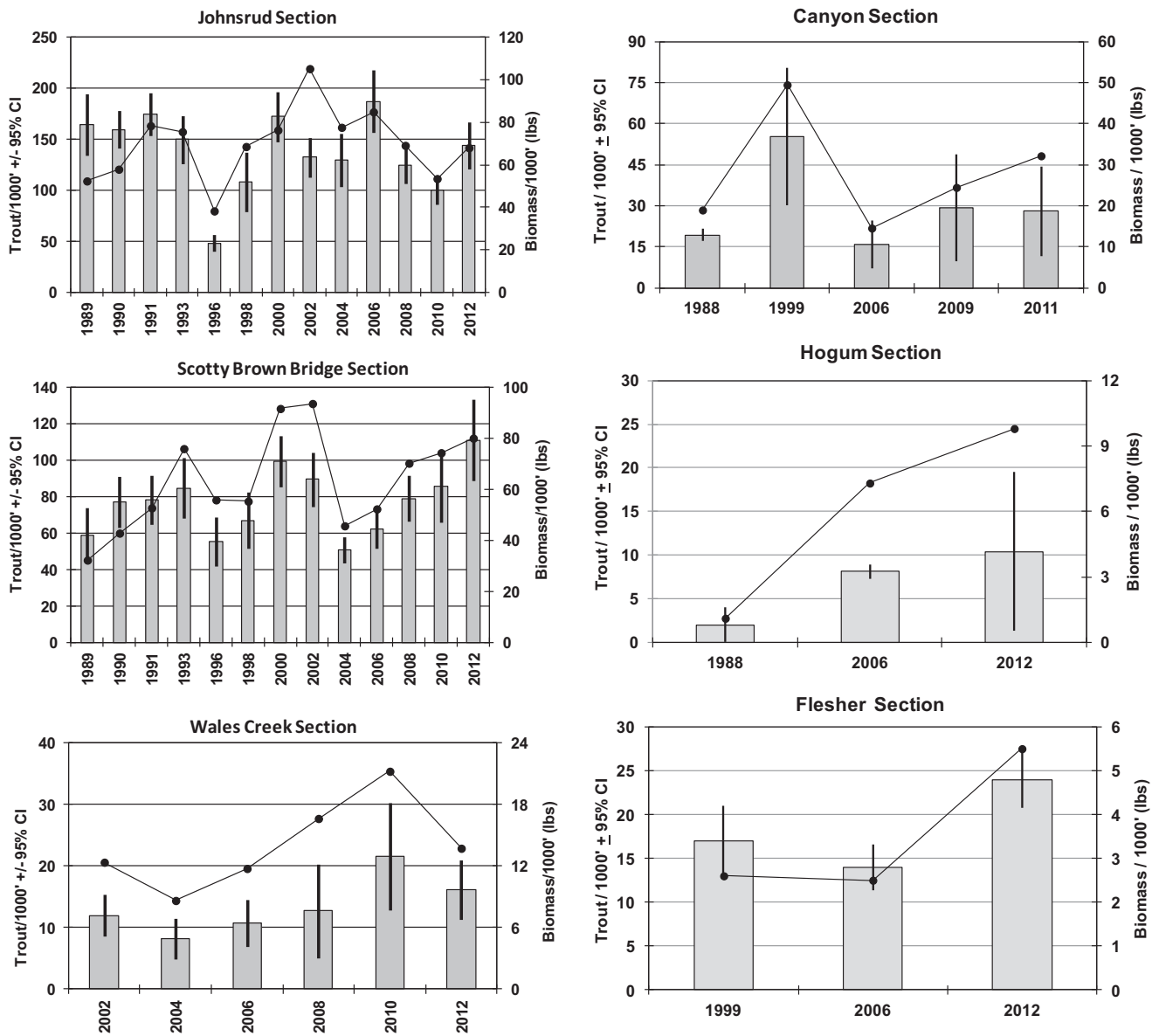
Figure 10. Estimates of total trout abundance and biomass (all trout >6.0") for six monitoring sites on the mainstem Blackfoot River for the monitoring period from 1988 to 2012. The left column shows estimates for the lower Blackfoot River downstream of Nevada Creek confluence. The right column shows estimates for the upper Blackfoot River upstream of the Nevada Creek confluence.
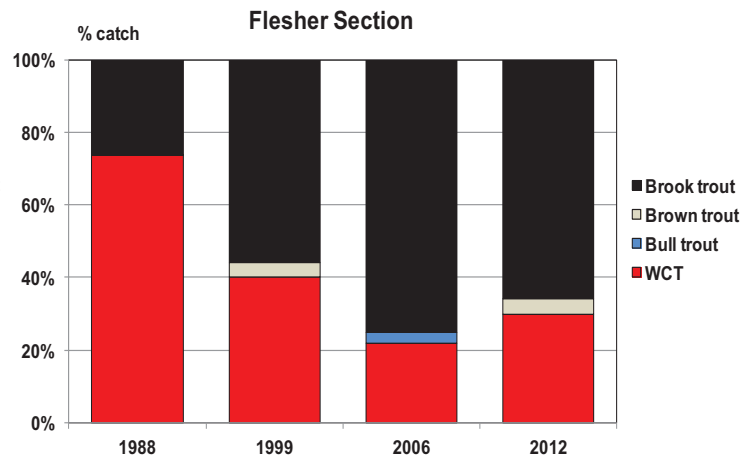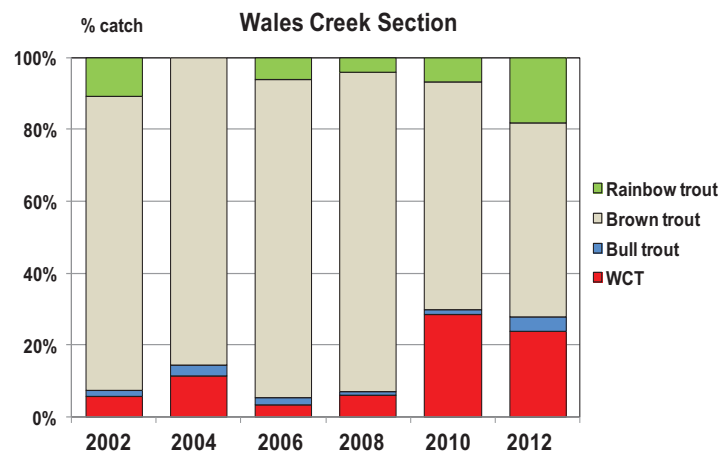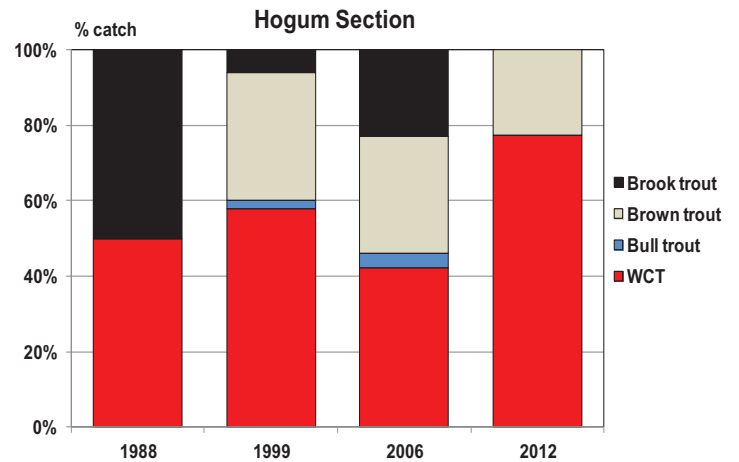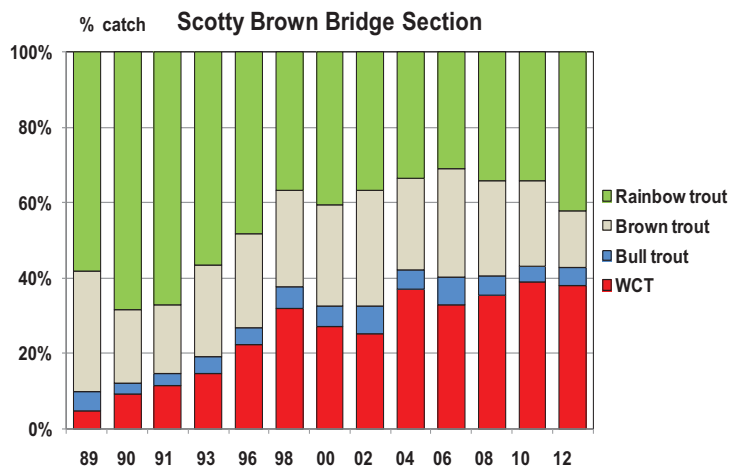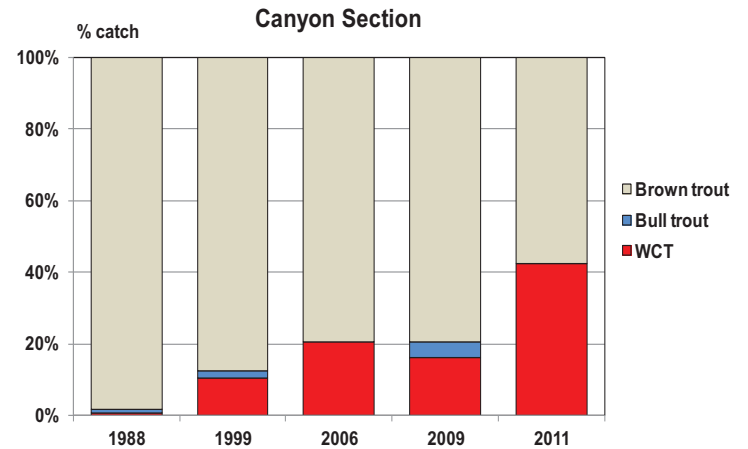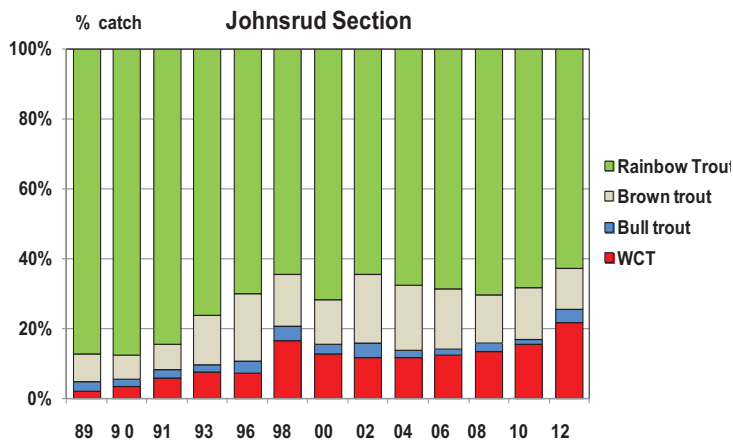
Figure 11. Percent trout species composition (fish >6.0") for six long-term fish population monitoring sites on the mainstem Blackfoot River, 1988-2012. The left column shows percent composition for monitoring sites the lower Blackfoot River and the right column shows trout composition for monitoring sites in the upper Blackfoot River.

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Lower Blackfoot River Survey Sections

Abundance and biomass for individual trout species – This section focuses on population estimates for individual trout species. The three lower river sites were sampled in the spring (May-June). The three upper river sites were all sampled in the fall (September-October). More detailed information is located in Appendices B and C.
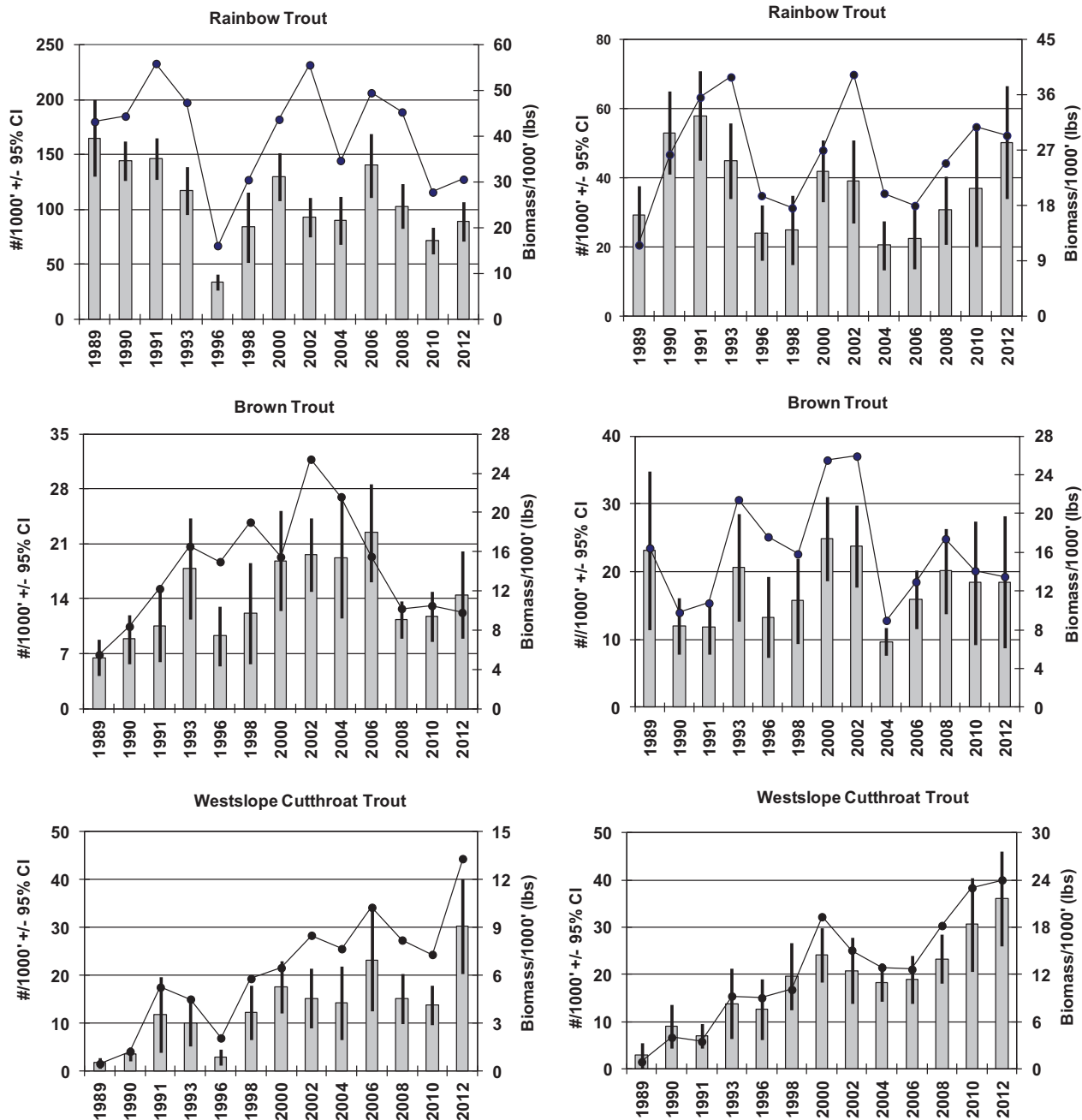
Figures 12. Estimates of trout abundance (bars) and biomass (lines) for trout in the Johnsrud section (left column) and Scotty Brown section (right column), 1989-2012.

Johnsrud section: The 2012 trout species composition (% of total catch for trout >6.0") in the Johnsrud section was 62.8% rainbow trout (*n*=562), 11.7% brown trout (*n*=105), 21.7% westslope cutthroat trout (*n*=194), 3.8% bull trout (*n*=34) and 0.1% brook trout (*n*=1). The total trout point estimate (fish >6.0") for the Johnsrud section increased from 111 fish/1,000' in 2010 to 157 fish/1,000' in 2012, a 41% increase (Figure 11). The total trout biomass estimate for fish >6.0" in the Johnsrud section also increased 21% from 56.4 lbs/1,000' in 2010 to 68.4 lbs/1,000' in 2012. All estimates of biomass and abundance and related size statistics for all Blackfoot River survey sites are located in Appendix B or C.

Estimates of abundance and biomass for individual trout species from 1989 through 2012 are shown on Figure 12. These data show native westslope cutthroat trout (> 6.0") increased from 13.7 fish/1,000' in 2010 to 30 fish/1,000' in 2012 (Figure 12), which represents a continued 23-year positive trend. The 2012 point estimate for brown trout (> 6.0") showed a slight increase from 11.7 fish/1,000' in 2010 to 14.5 fish/1,000' in 2012. The estimate for rainbow trout (>6.0") abundance increased from 71 fish/1,000' in 2010 to 89 fish/1.000' in 2012. Because of a small sample size and a low recapture rates, we were unable to generate a valid bull trout population estimate (Appendix C).

Scotty Brown Bridge section: The 2012 percent trout composition in the Scotty Brown Bridge section was 42% rainbow trout (*n*=292), 37% westslope cutthroat trout (*n*=263), 15.2% brown trout (*n*=106) and 4.9% bull trout (*n*=34). Total trout abundance (fish >6.0") increased 29% from 86 to fish/1,000' in 2010 to 111 fish/1,000' in 2012. The total trout biomass estimate for fish >6.0" in the Scotty Brown Section increased 54% from 52.2 lbs/1,000' in 2010 to 80.2 lbs/1,000'in 2012.

Estimates of abundance and biomass for all trout species (fish >6.0") in the Scotty Brown Bridge section are shown in Figure 10. The rainbow trout estimate of abundance increased from 37 fish/1,000' in 2010 to 50.3 fish/1000' in 2012. Westslope cutthroat trout (fish >6.0") increased from 30.6 fish/1,000' in 2010 to 36.1 fish/1,000' in 2012. Brown trout (fish >6.0") showed no change with 18.4 fish/1,000' in 2010 compared to 18.5 fish/1,000' in 2012. A valid bull trout estimate was not obtained in 2012. All metrics of abundance and biomass are located in Appendix C.

Wales Creek section: The Wales Creek section was established in 2002 and has been monitored on a biannual basis concurrent with the Johnsrud and Scotty Brown surveys (Figure 10). In May 2012, trout species composition in the Wales Creek section was 54% brown trout (*n*=109), 23.8% westslope cutthroat trout (*n*=48), 18.3% rainbow trout (*n*=27) and 4% bull trout (*n*=4). We estimated total trout abundance (fish > 6.0") for the Wales Creek section at 16.1 trout/1,000' in 2012 compared to 21.5 in 2010 (Figure 10). The total trout biomass estimate for fish >6.0" in the Wales Creek in 2012 was 13.7 lbs/1,000' compared to 21.2 lbs/1,000' in 2010 (Figure 10). Estimates of abundance and biomass are noticably lower than upriver (Canyon Section) and downriver (Scotty Brown) samples (Figure 10).

In the 2012 survey of the Wales Creek section, we doubled the sampling effort by electrofishing both banks (versus a single pass) of the Blackfoot River during mark and recapture runs. This allowed higher sampling efficiency (i.e., R/C=0.09 in 2010 versus 0.20 in 2012). As a result, we were able to estimate abundance for trout species present in low abundance (e.g., rainbow and westslope cutthroat trout). Estimates of abundance from 2002 through 2012 for individual species are shown in Figure 13. In 2010 and 2012, we also re-

Figure 13. Estimates of trout abundance for fish > 6.0" in the Wales Creek section, 2002-2012. An 'NE' indicates no estimate for that species.

surveyed mountain whitefish in the Wales Creek section. These results are located in the mountain whitefish study section located in Results Part IV.

Upper Blackfoot River Survey Sections

Canyon section: Fish populations in the Canyon section was established in 1971 and was resampled in 1988, 1999, 2006, 2009 and 2011. The long-term dataset for total trout abundance and biomass (fish >6.0") is shown in Figure 10. Trout species composition for these years is shown in Figure 11. In 2011, brown trout (*n*=34) were again the prevalent trout
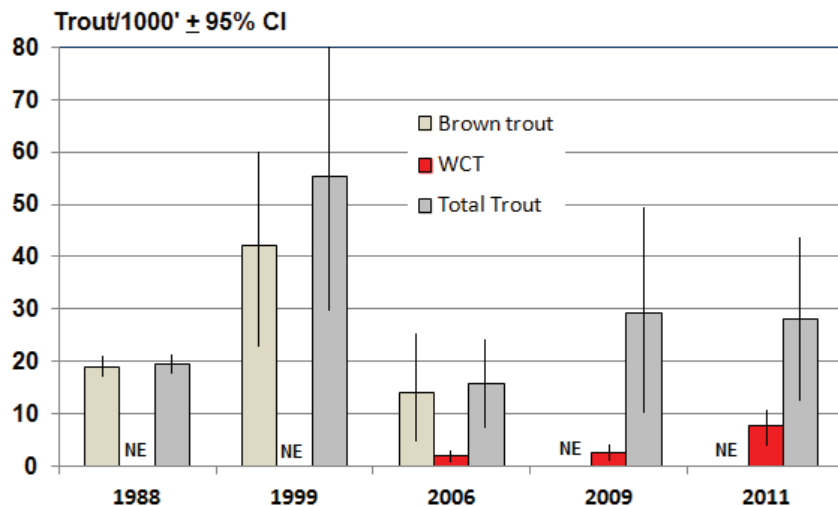
Figure 14. Estimates of trout abundance for fish > 6.0" in the Canyon section, 1988-2011. An 'NE' indicates no estimate for that species.

comprising 58% of the sample versus 80% in 2009. However, the percentage of westslope cutthroat trout increased from 16% (n=10) in 2009 to 42% (n=25) of the total

trout catch in 2011. The total trout estimate (fish >6.0") in the Canyon section showed no change between 2009 (i.e., 29.3 fish/1,000') and 2011 (i.e., 28.1 fish/1,000'). As was the case in 2009, we were unable to reliably estimate brown trout abundance in 2011 due to sampling difficulties (Appendix C). However, we did generate an estimate of abundance for westslope cutthroat trout as well as a total trout estimate. A comparison of cutthroat trout versus total trout abundance (fish >6.0") is shown in Figure 14. Similar to the Wales Creek section, we resurveyed the Canyon section for mountain whitefish in 2011. These results are located in Appendix D.

Hogum section: The Hogum section was established in 1972 and resurveyed in 1973, 1988, 1999, 2006 and 2012. For this section, total trout abundance and biomass (fish >6.0") from 1988 through 2012 is shown on Figure 10. Trout species composition (fish >6.0") for 1988-2012 is shown in Figure 11. Age 1 and older estimates of trout abundance for individual species across the entire dataset are shown in Figure 15. These data indicate westslope cutthroat remain stable, whereas brook trout have declined. Brown trout appear to be increasing in the section. All other 2012 population statistics for the Hogum section are shown in Appendix B.
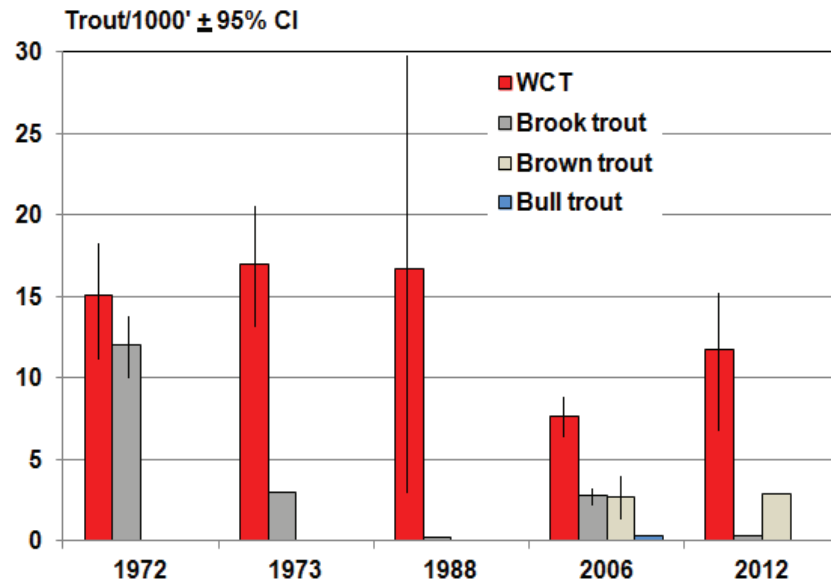
Figure 15. Estimates of trout abundance for age 1 and older fish in the Hogum section of the upper Blackfoot River, 1972-2012.

Flesher Section: The Flesher Section was established in 1973 prior to the release of toxic mine waste upstream of this section. It was resurveyed in 1975 following the release of mine waste and then again in 1988,
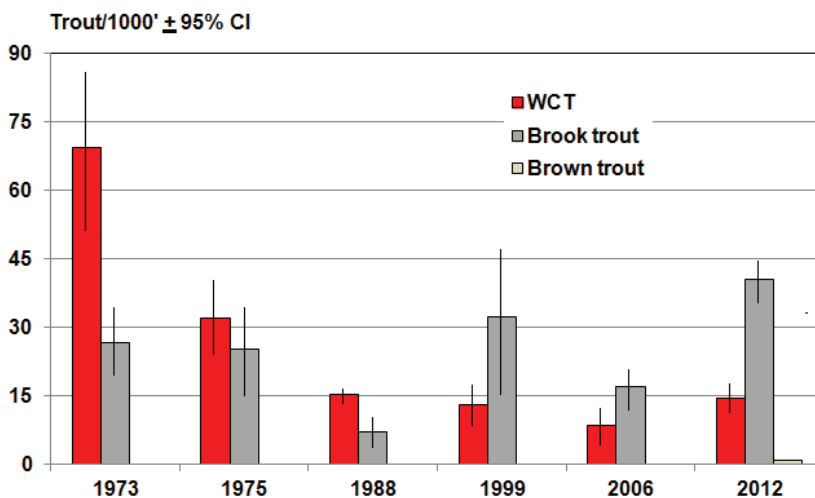
Figure 16. Estimates of trout abundance for age 1 and older fish the Flesher Section 1973-2012.

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1999, 2006 and 2012. For this section, estimates of total trout abundance and biomass (fish >6.0") from 1999-2012 are shown in Figure 10. Trout species composition (fish >6.0") for 1988-2012 is shown in Figure 11. Age 1 and older estimates of abundance for individual species across the entire 40-year dataset are shown in Figure 16. The surveys show declines in age 1 and older cutthroat between 1973 and 1988 have stabilized at low abundance from 1988-2012. Brook trout have increased since 1988. In 2012, we also identified the presence of adult brown trout in the Flesher section for the first time.

Bull Trout Conservation

       Bull trout, an inland char native to western Montana, was listed as "threatened" under the Endangered Species Act in 1998. In 2010, the United States Fish and Wildlife Service designated critical habitat for the recovery bull trout (USFWS 2010).   This designation includes streams, lakes and rivers within the Blackfoot Watershed (Figure 17), including all major bull trout spawning streams. In addition to this designation, "core areas" i.e., watersheds supporting critical habitat, were identified in 2000 by the State of Montana in order to more broadly foster restoration and protection of riparian habitat in the headwaters of these critical streams (Figure 17, MTBRT 2000).
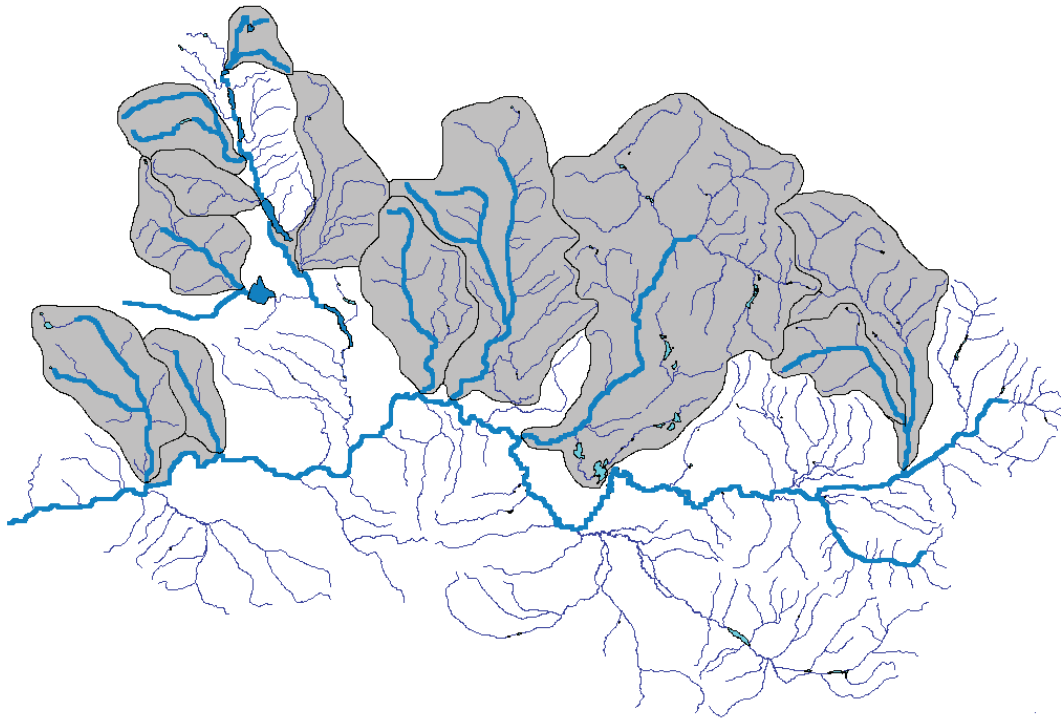
Figure 17. Bull trout recovery areas: The grey watersheds show bull trout "core areas" (MTBRT 2000). The bold blue lines show critical bull trout habitat (USFWS 2010).

       The Blackfoot Basin supports stream-resident and migratory (i.e., fluvial [river-dwelling] and adfluvial [lake-dwelling]) bull trout. The recovery of bull trout in the Blackfoot Basin fundamentally relies no-harvest angling regulation, combined with restoration and protection of critical waters with corridors connecting spawning, rearing and refugia habitat (Figure 17). Within these broader conservation areas, migratory bull trout life histories involve spawning in discrete areas, tributary use by early life-stages, large home ranges, extensive migrations at higher flows, and seasonal use of larger, more

productive river (or lake) habitats as well as refuge seeking behavior during periods of river warming (Swanberg 1997, Benson 2009). Migratory bull trout also require complex

habitats, colder water, groundwater upwelling for spawning, lower sediment levels, lower water temperatures and more tributary access than currently exists in many areas of the Blackfoot Watershed. Water temperatures of $\leq 57^{o}F$ are considered optimal for bull trout (Dunham et al. 2003).

Stream-resident bull trout require similar environments and complete their life-cycle in tributary streams. Adfluvial bull trout are rare in the upper Clark Fork Basin but occupy the Clearwater chain of lakes and migrate to tributaries for spawning and rearing (Benson 2009). The life-histories and habitat use of migratory bull trout have been extensively studied in the Blackfoot basin (Swanberg 1997, Schmetterling 2003, Pierce et al. 2005, Benson 2009). These studies, along with state and federal
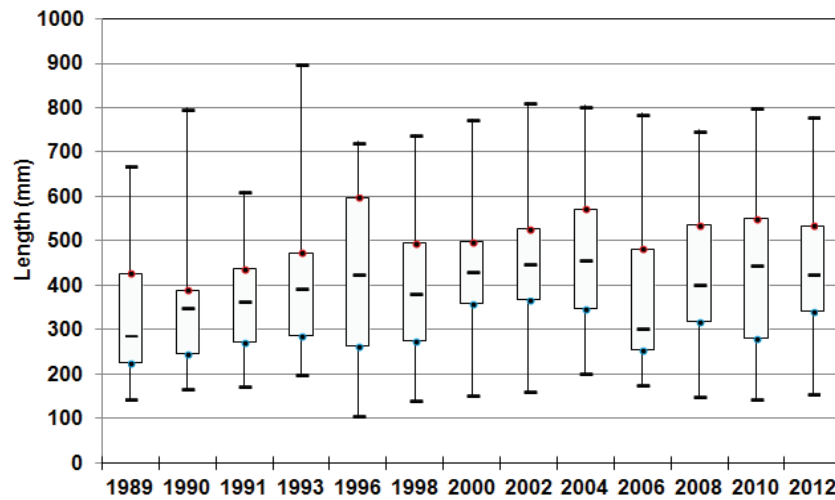
Figure 18. Range of total lengths for all individual bull trout collected in the Johnsrud and Scotty Brown Bridge monitoring sections of the Blackfoot River, 1989-2012. Box plots represent the range of total lengths (minimum and maximum) the 25th and 75th percentiles and median length values.
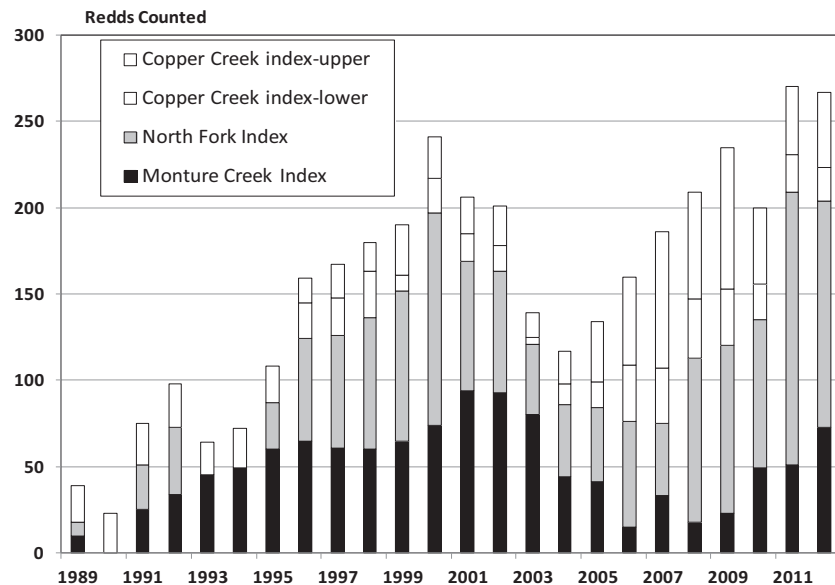
Figure 19. Bull trout redd counts for three spawning tributaries used by fluvial bull trout from the Blackfoot River. Note: Beginning in 1996, an upper spawning site was added to the Copper Creek count. Redd counts in this section are shown as the upper white bar above the original Copper Creek index count.

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recovery plans, provide the framework for restoration and recovery actions within the Blackfoot Basin, which also include the delineation of priority areas for bull trout restoration actions (MTBRT 2000; Pierce et al. 2008, USFWS 2010).

Since 1990, many restoration actions targeting the recovery of fluvial bull trout in the Blackfoot Watershed have been completed. These include: 1) enhancing instream flows and improving fish passage by screening major irrigation canals and improving road crossing in several bull trout streams, 2) flow enhancement livestock fencing and improved irrigation for fish passage on several streams, 3) the removal of Milltown Dam, 4) the placement of conservation easements on many streams, and 5) the purchase of Plum Creek lands in the Clearwater Basin in order to restore and protect bull trout habitat.

Following the 1990 adoption of protective (catch-and-release) harvest regulations, as well as early recovery actions (Pierce et al. 1997), bull trout population monitoring showed an increase in redd counts during the decade of the 1990s for the three primary spawning tributaries (Figure 19), an inclination towards larger fish in the lower Blackfoot River (Figure 18), and higher catch rates of juvenile bull trout in primary spawning
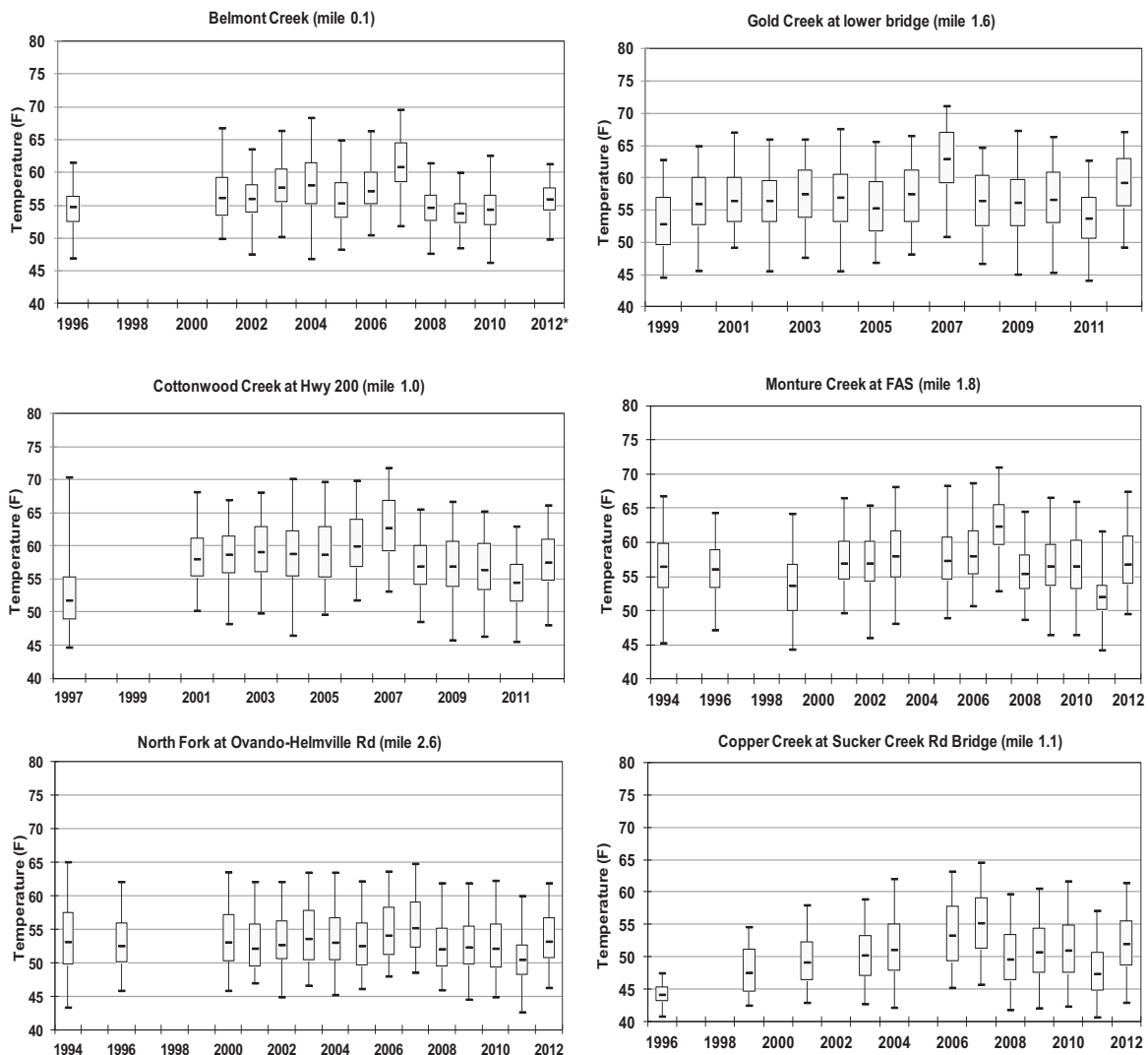
Figure 20. July water temperatures summaries six tributaries supporting bull trout spawning. Note only the North Fork and Copper Creek show consistent temperatures <65°F.

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streams (Results Part III). However, during the 2000-07 drought, these measures of population size showed declines with the exception of Copper Creek. Conversely, with a return to more favorable flow and temperature conditions (2008-12), bull trout redd counts, bull trout size in Blackfoot River and juvenile catch-rates have all increased (Figures 19, 18, Results Part III). Despite certain improvements, bull trout appear to be in decline in Gold and Belmont Creeks based on redd counts (Appendix G). Recent monitoring of reach scale restoration activities in bull trout habitat are described in Results Part III and IV.

PART III: Restoration and tributary assessments

Arrastra Creek

Restoration objectives: Restore upstream fish passage for fluvial native fish of the Blackfoot River.

Project summary

Arrastra Creek, the largest and among the coldest Blackfoot River tributaries between Beaver Creek (rm 105.2) and Nevada Creek (rm 67.8), enters the Blackfoot River at river mile 88.8. Arrastra Creek is also the only stream between Poorman Creek (rm 108) and the North Fork (rm 54.1) to support bull trout. A radio telemetry study also identified Arrastra Creek as the primary spawning tributary for fluvial westslope cutthroat trout in the middle Blackfoot River (Pierce et al. 2007). All radioed westslope cutthroat trout in this study spawned downstream from a pair of undersized culverts located at mile 3.2. In 2005, these culverts were replaced with a bridge which restored access to ~six miles of perennial stream upstream of the crossing. Other fisheries improvements along Arrastra Creek include riparian grazing changes on BLM land.

Figure 21. Electrofishing catch for age 1 and older trout in lower Arrastra Creek at mile 0.7, 1989-2011.

Fish population monitoring

Arrastra Creek supports bull trout and westslope cutthroat trout throughout the mainstem as well as brown trout and brook trout in lower reaches. Past genetic tests found genetically pure cutthroat trout in Arrastra Creek. However, a recent telemetry study identified the possibility of hybridization in the West Fork of Arrastra Creek. Fish populations in lower Arrastra Creek (mile 0.7) have been periodically monitored between 1989 and 2011. Results of these surveys are shown in Figure 21.

Ashby Creek

Restoration objectives: Protect the genetic purity of westslope cutthroat trout in the upper Ashby Creek watershed by using an existing wetland complex as a migration barrier, and improve westslope cutthroat trout habitat by creating a natural channel that provides complexity, increases riffle-pool habitat features and available spawning substrate and increase shade and small diameter wood recruitment to the stream channel. Improve and re-establish wetland functionality.

Project summary

Ashby Creek is a second-order tributary to Camas Creek located in the Union Creek basin. Ashby Creek supports a genetically pure population of westslope cutthroat along with brook trout in low numbers (Figure 22). Ashby Creek originate in forested areas on DNRC and BLM properties before entering private ranch lands near mile 3.0. Land uses considered adverse to fisheries include roads in riparian areas, undersized culverts on public lands and along with past agricultural practices on private lands that include overgrazing of riparian, channel manipulations and dewatering.

A comprehensive restoration project plan was completed on private ranchland in 2007. The project included: 1) reconstruction of three miles of stream that had been historically ditched, 2) enhanced in-stream flows, 3) improved fish passage, 4) the installation of a fish screen at a diversion point, 5) riparian grazing changes, and 6) riparian re-vegetation including shrub plantings, soil lifts and weed management. This project also connected Ashby Creek to an 80-acre wetland in a manner that is designed to inhibit the upstream movement of fish. Lastly, a conservation easement was placed on the cooperating ranch in order to preserve the rural character and natural resources of the property. More recent work has been initiated on upstream DNRC lands. This work involves removing a segment road from the riparian area, some channel reconstruction, the removal of undersized culverts and livestock management measures.

Figure 22. Estimates of trout abundance for age 1 and older fish for two reaches of Ashby Creek, 2007-2012.

Fish population monitoring

We began monitoring for fisheries response to the treatment area of Ashby Creek in 2007 and continue through 2012. The monitoring includes a downstream treatment (i.e., reconstructed) reach (mile 2.7) and an upstream reference reach (control) at mile 4.0. Results of these surveys show increased abundance of trout in the treatment reach and recent decline in the reference (Figure 22). These upstream declines in numbers may

be the result of movement, the effects of recent flooding or habitat changes involving increased grazing pressure as recently observed during surveys.

Bear Creek

Restoration Objectives: Restore habitat degraded by historical activities in the channel, restore fish passage and thermal refugia, and improve recruitment of trout to the Blackfoot River.

Project summary

Bear Creek, a small second-order tributary to the lower Blackfoot River, flows six miles north to its mouth where it enters the Blackfoot River at river mile 12.2 with a base-flow of 3-5 cfs. Bear Creek is one of the colder tributaries to the lower Blackfoot River.

Located on industrial forest and agricultural lands, Bear Creek has a long history of adverse habitat changes, which has included undersized culverts, road drainage and siltation, irrigation, channelization of the stream, excessive riparian grazing and streamside timber harvest. Prior to restoration activities, these fisheries impairments contributed to the loss of migration corridors and the simplification and degradation of salmonid habitat. Many of

Figure 23. Estimates of trout abundance for age 1and older fish in Bear Creek at mile 1.1, 1998-2011.

these impairments were corrected between the 1990s and 2011. Restoration activities included: 1) upgrading or removing culverts and addressing road-drainage problems, 2) improving water control structures at irrigation diversions, 3) reconstructing 2,000' of channel, 4) enhancing habitat complexity on an additional 2,000' of stream, 5) shrub plantings, and 6) the development of compatible riparian grazing systems for one mile of stream.

Fish population monitoring

Bear Creek supports predominately rainbow trout mixed with low numbers of brown trout and brook trout in the lower stream, westslope cutthroat trout in the upper basin and the incidental presence of juvenile bull trout. In 1998 we began monitoring trout populations in a reconstructed stream reach and continued monitoring through 2011 (Figure 23). In 2011, the survey shows a significant decline in total trout abundance, which may be the result of a 2011 high run-off event and redistribution of fish to the Blackfoot River as occurred through the Blackfoot basin during recent high water years.

Braziel Creek

Restoration objectives: Reestablish natural channel conditions, riparian area and flows capable of increasing westslope cutthroat trout numbers.

Project Summary

        Braziel Creek drains a small tributary Nevada Creek (mile 24.5) located along the southeastern foothills of Hoodoo Mountain near Helmville. Braziel Creek is about 4 miles in length and generates an estimated base-flow of 0.5-1.0 cfs prior to entering Nevada Creek about 2.0 miles downstream of the Nevada Creek Reservoir. Prior to restoration activities in 2010, lower Braziel Creek was heavily altered from channelization; dewatering and heavy riparian grazing, all of which contributed to degradation of westslope cutthroat trout habitat. Furthermore, undersized culverts limited fish passage and cutthroat trout entrainment had been indentified in one irrigation ditch. To improve conditions for cutthroat trout, a project was initiated that included reconstructing 1,500' of lower Braziel Creek, upgrading an undersized a county road culvert, installing a fish screen on the existing diversion and livestock were fenced from the project to allow vegetative recovery. The landowner has entered into a single season agreement with Trout Unlimited for 2013, which secures 0.5 cfs of minimum flows.
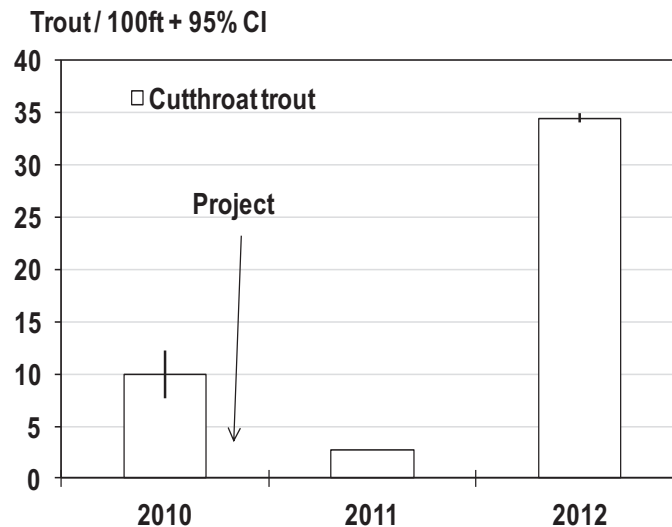
Figure 24. Estimates of trout abundance for age 1 and older westslope cutthroat trout in Braziel Creek at mile 0.2, 2010-2012.

Fish population and other monitoring activities

        Braziel Creek supports a simple fish community of westslope cutthroat trout and sculpins. Genetic testing in 2008 of the westslope cutthroat trout found mild (1.5%) introgression with rainbow trout. Prior to restoration, a fish population monitoring site was established in 2010 at mile 0.2 within the treatment area followed by two years of post-treatment monitoring (Figure 24). We also conducted a single pass electro-fishing survey of an irrigation ditch in 2011 on lower Braziel Creek to examine Coanda fish screen installed in 2010. The 2011 survey found no fish in the ditch, compared to a CPUE of 11 in 2010. Pre- and post-project flow monitoring has also been conducted through the project area, which highlighted an opportunity to secure late-season flows.

Chamberlain Creek

Restoration objectives: Improve access to spawning areas; improve rearing conditions for westslope cutthroat trout; improve recruitment of westslope cutthroat trout to the Blackfoot River.

Project summary

       Chamberlain Creek is a small Garnet Mountain tributary to the middle Blackfoot River, entering near rm 43.9 with a base-flow of 2-3 cfs. Prior to 1990, sections of lower Chamberlain Creek were dewatered and severely altered by heavy riparian grazing, road encroachment and channelization, which led to sharp declines in westslope cutthroat trout densities in lower stream reaches (Peters 1990). During the early 1990s, Chamberlain Creek was also one of the first comprehensive restoration projects within the Blackfoot Basin. Restoration emphasized road drainage repairs, riparian livestock management changes, in-stream habitat restoration, irrigation upgrades (consolidation of ditches, water conservation, elimination of fish entrainment and fish ladder installation on a diversion), and improved stream flows through water leasing. Stream restoration occurred throughout the drainage with emphasis in the lower mile of stream. Cooperating landowners also placed conservation easements on adjacent private lands. In 2010, >13,000 acres of private timberland was transferred to DNRC in 2010 with stringent conservation easement protection of the riparian areas. In addition, the conservation agreements required the removal of 5.5 miles of roads immediately adjacent to Chamberlain Creek, the West Fork of Chamberlain Creek and Bear Creek. Lastly, 10 culverts and two bridges were either removed or upgraded to meet fish passage and stream function objectives. With the completion of this project, the cooperators in this endeavor were able to address the majority of known human impacts hindering recovery of westslope cutthroat trout in the Chamberlain Creek drainage.
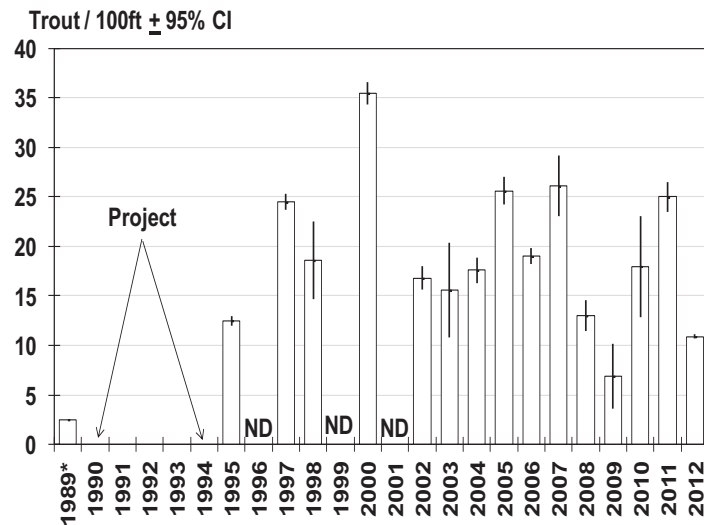
Figure 25. Estimates of trout abundance for age 1 and older westslope cutthroat trout in Chamberlain Creek at mile 0.1, 1989-2012.

Fish populations and other monitoring activities

       Chamberlain Creek is a westslope cutthroat trout dominated stream over its entire length although lower reaches also support rainbow, brown and brook trout in low abundances. We established a fish population monitoring section prior to restoration in lower Chamber Creek at mile 0.1 in 1989 and continued to monitor populations through 2012 (Figure 25). Overall, the long-term dataset continues to show relatively elevated population levels following restoration activities that occurred in the early 1990s. Following restoration and westslope cutthroat trout recovery in lower Chamberlain Creek, radio telemetry identified Chamberlain Creek as a primary spawning stream for fluvial westslope cutthroat trout from the Blackfoot River (Schmetterling 2001).

Copper Creek

<u>Introduction</u>

Copper Creek, the largest tributary to the lower Landers Fork entering at mile 3.6, is a critical spawning and rearing stream for genetically pure fluvial westslope cutthroat trout and fluvial bull trout in the upper Blackfoot River drainage. Copper Creek supports an entirely native fish community basin-wide, and provides the only major spawning site for fluvial bull trout in the upper Blackfoot River basin. Cold water from Copper Creek also moderates temperatures in the lower Landers Fork.

During August 2003, the Snow/Talon wildfire on the Helena National Forest ran through the Copper Creek drainage. This high intensity, stand replacement fire burned significant portions of the basin including a fluvial bull trout spawning site approximately three weeks prior to spawning.

<u>Fish populations and other monitoring activities</u>

FWP established a fish population monitoring sites on Copper Creek at mile 6.2 in 1989 and has continued to monitor native trout populations through 2012. We also continued long-term water temperature monitor at mile 1.1. In addition, the USFS has monitored bull trout redd counts surveys since 1989 at an index section and then began total redd count surveys beginning in 1996.
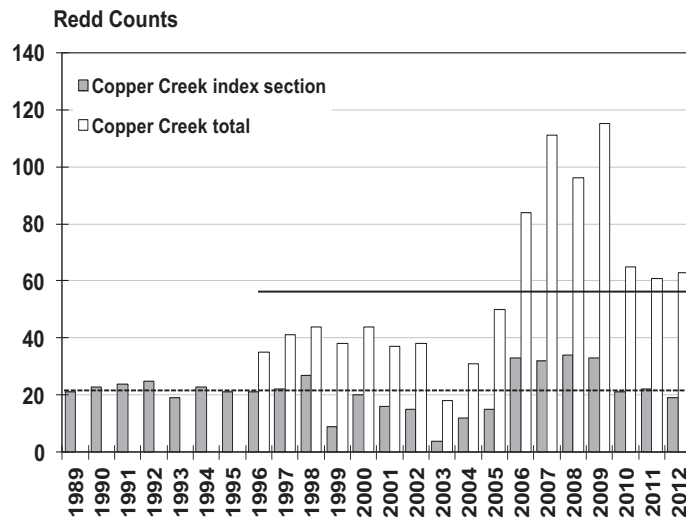
Figure 26. Bull trout redd counts for Copper Creek: Gray bars show redd counts in long-term (1989-2012) index section. White bars show the total redd counts (index plus upstream section) from 1996-2012. The dashed horizontal line shows the long-term mean for the index section and the black line shows the mean for the total bull trout redd counts from 1996 to 2012.
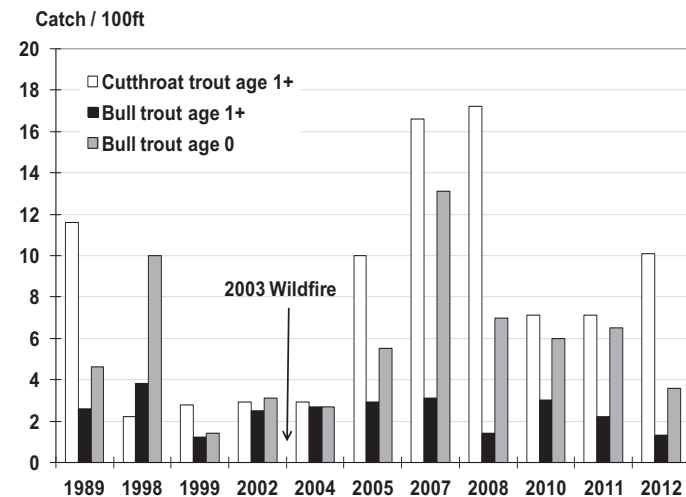
Figure 27. Electrofishing catch for native trout at stream mile 6.2 in Copper Creek, 1989-2012.

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Following the 2003 wildfire, total bull trout redd counts showed a substantial increase between 2005 and 2009 before declining in recent years (Figure 26). Likewise, electro-fishing surveys at a long-term monitoring site (mile 6.2) show similar trends (Figure 27). Long-term monitoring of water temperatures show Copper Creek as the coldest of all bull trout streams as shown on Figure 20 (Appendix F).

Cottonwood Creek
Restoration objectives: Improve degraded habitat; eliminate fish losses to irrigation ditches; and restore in-stream flows and migration corridors for native fish.

Project summary

Cottonwood Creek, a third-order stream, flows ~16 miles south from Cottonwood Lakes and enters the middle Blackfoot River at rm 43 with a base-flow of ~15 cfs. Genetically pure westslope cutthroat trout are prevalent in the headwaters along with low numbers of bull trout. Whereas rainbow trout, brook trout and brown trout are the dominant salmonids in middle to lower stream reaches. Cottonwood Creek is considered a "core area" and was designated as "critical habitat" under the ESA for the recovery of bull trout. Since 1996, Cottonwood Creek has been the focus of several restoration actions. These include the placement of fish ladders and fish screens at diversion points, instream flows enhancement throughout

Figure 28. Estimates of abundance for age 1 and older westslope cutthroat trout on Cottonwood Creek at mile 12.0. Diversion upgrades and instream flow enhancement occurred in 1997.

the mainstem of Cottonwood Creek and riparian fencing projects on the Blackfoot Clearwater Game Range to manage livestock. Currently the USFS is pursuing the road repairs and a culvert removal project to reduce sediment and improve habitat connectivity in the headwaters of the Cottonwood Creek basin.

Fish populations and other monitoring activities

In 2011 and 2012, we continued to monitor fish populations in upper Cottonwood Creek at mile 12.0 at a site established in 1997 where enhanced flow, irrigation ditch screening and diversion upgrades were completed (Figure 28). Prior to 1997, this section was completely dewatered during late summer and fall by irrigation. Following an initial increase in trout abundance during the late 1990s, long-term monitoring indicates a slight decline in recent years, which we attribute to habitat simplification caused by the reduction of instream wood within the monitoring section. Numbers of bull trout at the mile 12.0 monitoring section are too low to estimate.

July maximum water temperatures on Cottonwood Creek at the mile 1.0 monitoring site for 2011 and 2012 continue to show an average cooling of 5°F in water temperatures compared to prior years (Figure 20, Executive Summary). Summary statistics for all water temperature monitoring on Cottonwood Creek for 2011-2012 are located in Appendix F.

Enders Spring Creek

Restoration objectives: Restore the spring creek to natural conditions: reduce water temperatures to level suitable for bull trout, reduce in-stream sediment levels, enhance habitat quality utilizing in-stream structures, vegetation and provide suitable substrate for spawning.

Project summary

Prior to restoration, Enders Spring Creek was a heavily degraded 1st-order spring creek tributary to the North Fork of the Blackfoot River entering at mile 6.3. Past stream channel degradation stems from historic agricultural activities that included the loss of sinuosity, channel widening and heavy sediment loading in pools and glides. Enders Spring Creek was fully reconstructed in 2008. It was the last major spring creek to the North Fork that required active restoration.

Fish populations and other monitoring activities

Enders Spring Creek supports a trout community dominated by brook trout followed by bull trout and brown trout in low abundance. Pre-treatment fish and habitat assessments were reported in Pierce et al. (2011).

In 2011, we moved the fish population monitoring site from mile
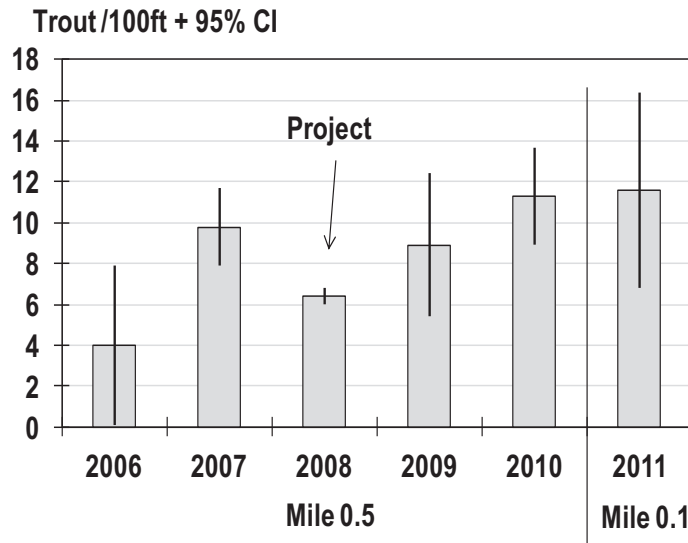
Figure 29. Estimates of total trout abundance for age 1 and older fish in Enders Spring Creek at prior survey location (mile 0.5, 2006-2010) and new survey location (mile 0.1, 2011).
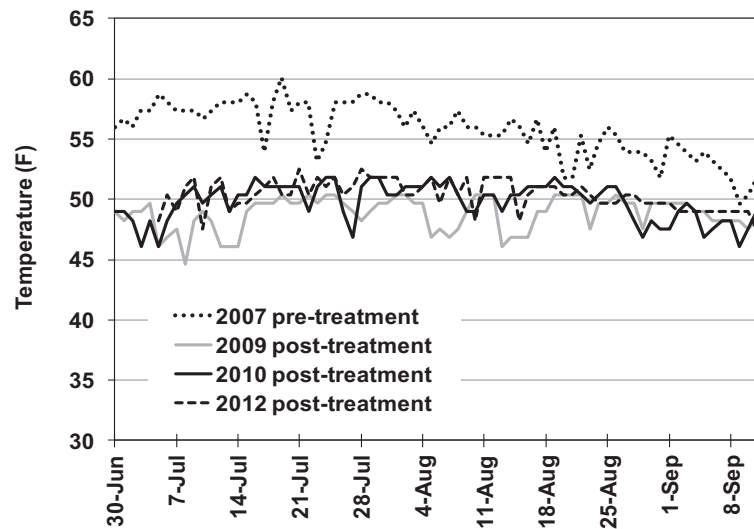
Figure 30. Summer maximum daily water temperatures for Enders Spring Creek at mile 0.1.

0.5 to mile 0.1 because large trees had fallen into the stream at the mile 0.5 monitoring site making reliable fish population surveys impossible. Pre-and post-restoration fish population sampling for monitoring sites are shown in Figure 29).

Pre-and post-treatment water temperature monitoring in Enders Spring Creek near the confluence (mile 0.1) with the North Fork show a post-restoration cooling effect with maximum temperatures declining 6-8°F to summer highs of <52°F (Figure 30, Appendix F).

Gold Creek

Restoration objectives: Restore pool habitat and morphological complexity; restore thermal refugia for Blackfoot River native fish species.

Project summary

Gold Creek is the largest tributary to the lower Blackfoot River, entering at rm 13.5. The majority of the Gold Creek watershed is industrial forest. Past harvest of riparian conifers combined with the actual removal of large wood from the channel reduced habitat complexity on the lower three miles of Gold Creek. The result of this fish habitat simplification was low densities of age 1 and older fish. In 1996, we installed 66 habitat structures made of native material (rock and wood) constructing 61 new pools in the three-mile section (Schmetterling and Pierce 1999). Prior to restoration work (1996), we established a baseline fish population survey section (mile 1.9) in the project area.
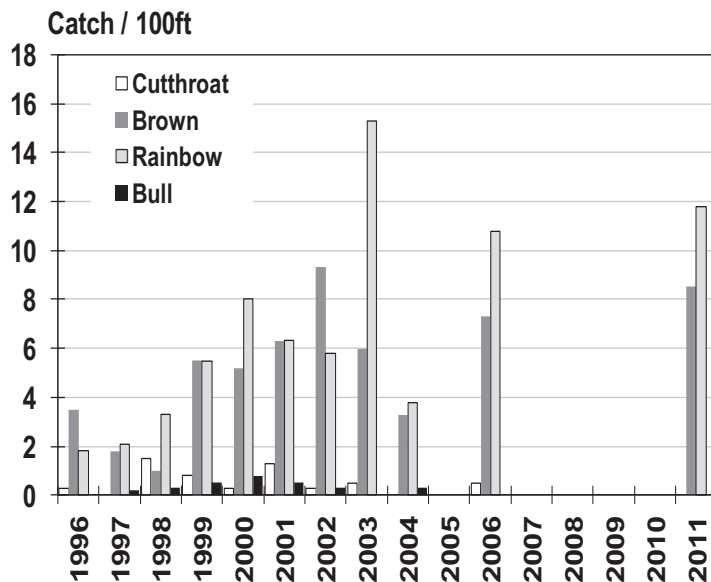
Figure 31**.** Electrofishing catch for age 1 and older salmonids in Gold Creek at mile 1.9, 1996 – 2011.

Fish populations and other monitoring activities

Gold Creek is a spawning and rearing tributary to the lower Blackfoot River for bull trout, westslope cutthroat trout, rainbow trout, and brown trout. Resident brook trout also inhabit the drainage in some areas. The mainstem and confluence area of Gold Creek provide thermal refugia for Blackfoot River bull trout during periods of river warming.

In 2011, we re-surveyed post-restoration fish populations at mile 1.9 in a reach influenced by the placement of instream habitat structure, and continued long-term water temperature monitoring at mile 1.6. Plum Creek Timber Company has performed annual bull trout redd count surveys since 2004 (Appendix G).

Long-term electrofishing survey data are shown in Figure 31. These survey results show that numbers of rainbow and brown trout are higher in recent years compared to initial surveys. Compared to prior surveys, electro-fishing sampling in 2011

failed to detect native cutthroat or bull trout in the monitoring section. Bull trout redd counts show a declining trend in adult bull trout (Appendix G).   Long-term monitoring results of water temperatures appear in Figure 20, Executive Summary and Appendix F.

Grantier Spring Creek
Restoration objective: Restore natural channel features of a degraded spring creek.

Project summary

Grantier Spring Creek is a large spring creek tributary to lower Poorman Creek, which enters the upper Blackfoot River at river mile 108. Grantier Spring Creek was the first major spring creek restoration project undertaken in the Blackfoot River Basin. Grantier Spring Creek was reconstructed in 1990. In addition, vegetation was allowed to recover by reducing livestock pressure on streambanks.

Fish populations and other monitoring activities

Prior to the restoration of Grantier Spring Creek, FWP established a fish population monitoring site at mile 1.0. Initial (1991 and 1994) surveys at this site found brook and brown trout as the only salmonids present. We returned to resurvey the reach 17 years post-treatment and repeated the surveys between 2008-2012 (Figure 32). These long-term surveys show westslope cutthroat trout are now reestablished as the prevalent trout species within the monitoring section and that both nonnative brown and brook trout are now present in relatively low abundance (Figure 32). This shift in species composition includes the presence of large (>400mm) adult westslope cutthroat trout which elevates total trout biomass relative to population abundance. Subsequent spawning surveys completed

Figure 32. Estimates of trout abundance and biomass for age 1 and older fish in Grantier Spring Creek at mile 1.0, 1991-2012.

in 2011 and 2012 identified westslope cutthroat trout spawning and redds within the upper spring creek. In addition, during fish population surveys age-0 westslope cutthroat trout were also observed but not collected throughout the survey reach.

Because this type community shift has not been documented in other areas, a series of aquatic assessments (geomorphic and in-stream habitat surveys, water temperature monitoring and in-stream sediment surveys) were conducted to help document habitat conditions (Pierce et. al 2011). To further investigate this expansion, we radio-tagged 10 adult westslope cutthroat trout in spring 2012 and tracked these fish

through summer (Table 1). Of these ten fish, four made spawning movements to tributaries to the upper Blackfoot River. Most of the radioed fish summered in the Blackfoot River. Although cutthroat trout are now reproducing in Grantier Spring Creek, this small study indicates a majority of adult cutthroat trout in lower Granter Spring Creek likely originate from other tributaries and that Grantier Spring Creek provides seasonal habitat for various stocks (Table 1). In addition to these investigations, genetic studies of this population expansion in Grantier Spring Creek are ongoing. Lastly, we also collected water temperatures at two locations (mile 0.1 and 1.0) on Grantier Spring Creek in 2012. Those results are located in Appendix F.

| Fish ID | Capture location | Pre-spawning migration | | | Tributary spawning | | | Post-spawning | | |
		Date migration started	Total km	Total Days	Tributary	Estimated spawning date	Date exited	Last live location	Last live contact date	Fate
1	Grantier Spring Creek							Grantier Spring Creek	23-Aug	Alive
2	Grantier Spring Creek	12-May-12	14.5	29	Landers Fork	14-Jun		Landers Fork	15-Jun	Unknown
3	Grantier Spring Creek							Blackfoot River	23-Aug	Alive
4	Grantier Spring Creek							Blackfoot River	11-Jul	Unknown
5	Grantier Spring Creek	30-Apr-12	0.24	1	Poorman Creek	4-May	8-May	Blackfoot River	23-Aug	Alive
6	Grantier Spring Creek							Blackfoot River	21-Jun	Unknown
7	Grantier Spring Creek	30-Apr-12	17.1	16	Copper Creek	24-May	3-Jun	Grantier Spring Creek	19-Jul	Unknown
8	Grantier Spring Creek							Blackfoot River	19-Jul	Alive
9	Grantier Spring Creek							Blackfoot River	23-Aug	Alive
10	Grantier Spring Creek	20-Apr-12	14	16	Landers Fork	9-May	10-May	Blackfoot River	23-Aug	Alive

Table 1. Summary of spawning and movements for 10 westslope cutthroat trout radio-tagged in Grantier Spring Creek in spring 2012.

Hoyt Creek

Restoration objectives: Restore floodplain elevation for channel stability, wetland values and irrigation efficiency improvement. Reduce water temperatures and sediment loading in the downstream direction and increase riparian vegetation and undercut bank habitat through the development and implementation of a grazing plan.

Project summary

Hoyt Creek, a small first-order spring-fed tributary to Dick Creek, originates from alluvial aquifers located just north of Ovando, MT and flows ~4.5 miles through private agriculture ranchland. Hoyt Creek has a history of land-use impacts that include channel instability (incision), damage to stream banks, suppressed riparian vegetation, irrigation dewatering and elevated water temperatures.

In 2006, middle Hoyt Creek (mile 1.3 to 3.4) underwent: 1) channel reconstruction to an "E4-type" channel on over 11,000' of channel, 2) the restoration of 334 acres of herbaceous wetland, and 3) the placement of rock cross-vane diversion structures at two irrigation ditches. The stream channel substrate throughout the project was lined with gravel/cobble to assure the stability of the channel features and provide habitat. In 2007, native trees and shrubs were planted to facilitate riparian vegetation recovery.
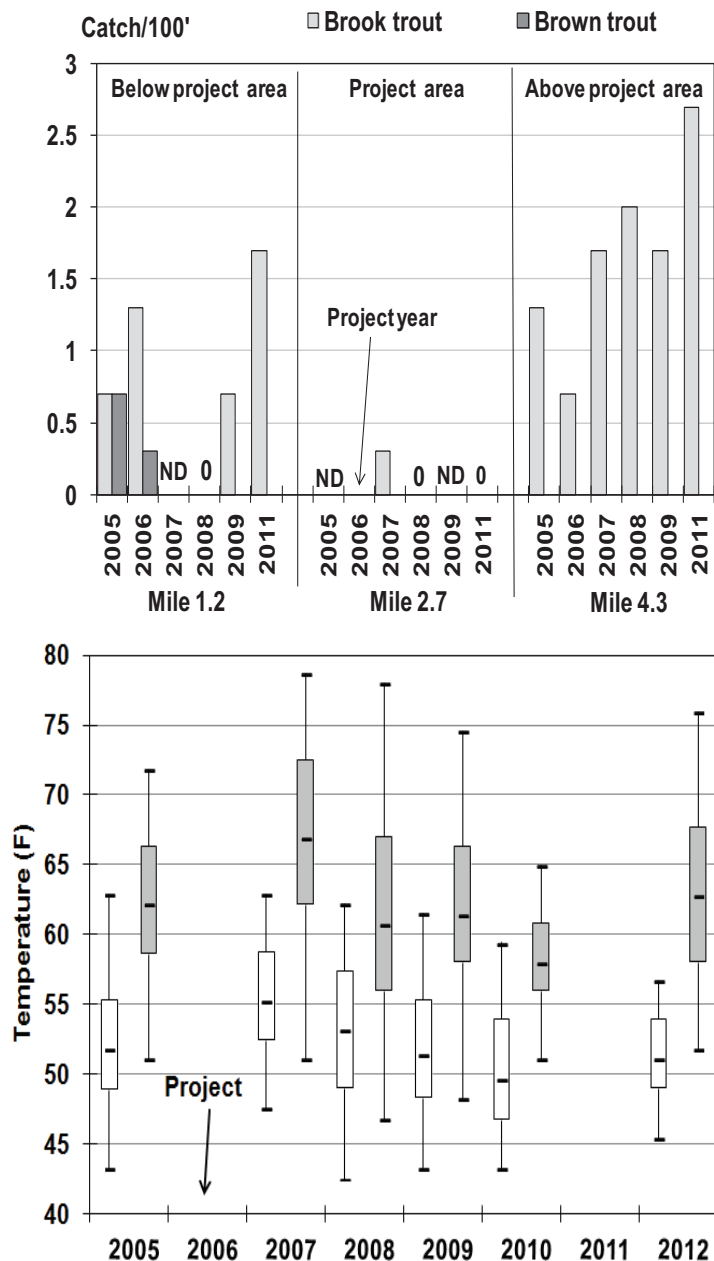
Figure 34. July water temperatures for Hoyt Creek upstream (white box, mile 4.3) and downstream (grey box, mile 1.2) of stream restoration project, 2005-2012.

Fish populations and other monitoring activities

In 2011, we continued fish population surveys and water temperature monitoring up- (mile 4.3) and downstream (mile 1.2) of the reconstructed channel. As shown on Figure 33, brook trout were the only species identified in the most recent fisheries surveys. Water temperature monitoring results for July 2005-2012 are shown in Figure 34.

Jacobsen Spring Creek

Restoration objectives: Maximize secondary in-stream productivity; maximize quality of shoreline rearing areas; restore spawning site potential by reducing levels of fine sediment in riffles to a level suitable for spawning; reduce summer water temperatures suitable for bull trout ($<60^0$F); provide high quality pools with high level of complex cover; maximize use of existing channel belt-width and existing shoreline areas.

Trout / 100ft + 95% CI

Figure 35. Estimates of total trout abundance for age 1 and older fish in Jacobsen Spring Creek at mile 0.6, 2005-2012.

Project summary

Jacobsen Spring Creek is a second-order spring creek tributary to the North Fork of the Blackfoot River that flows about 3.4 miles through private ranch land. Jacobsen Spring Creek forms from two spring creeks that merge at mile 0.7 and together these generate a base-flow of 4-7 cfs near the mouth. This small spring creek system enters the North Fork of the Blackfoot River at mile 4.7. According to landowner accounts, Jacobsen Spring Creek historically supported both bull trout and westslope cutthroat trout. Jacobsen Spring Creek was severely degraded due to historic grazing and timber harvest practices, the consequences of which included a wide and shallow channel, low sinuosity, elevated water temperatures and excessive sediment loading (Pierce et al. 2006). However, early habitat investigations identified the spring creek as possessing the basic habitat components necessary for improved fisheries, such as stable groundwater inflows, gravel substrate and a relatively dense riparian spruce forest that has potential to provide shade, complexity, and wood to the stream channel.

Starting in 2005, both channels of Jacobsen Spring Creek were reconstructed. The project emphasized a deep and narrow channel with higher sinuosity, the inclusion of backwater and shoreline rearing areas, gravel in pool tail-outs, and the placement of in-stream wood and sod mats on the stream banks to facilitate recovery. Pre- and post-treatment changes to the stream are summarized in Pierce et al. (2008). The project also included shrub plantings and the adoption of livestock management changes consistent with project objectives.

Fish populations and other monitoring activities

Jacobsen Spring Creek supports a mixed community of salmonids. Brook trout comprise >90% of the community followed by brown trout and the incidental presence of rainbow trout. In 2011 and 2012, we continued monitoring fish populations at mile 0.6, a site established in 2005 prior to restoration activities (Figure 35). Water temperatures monitoring near the mouth indicate a post-restoration cooling effect with maximum temperatures of >65°F pre-treatment to <58°F post-treatment (Figure 36, Appendix F).
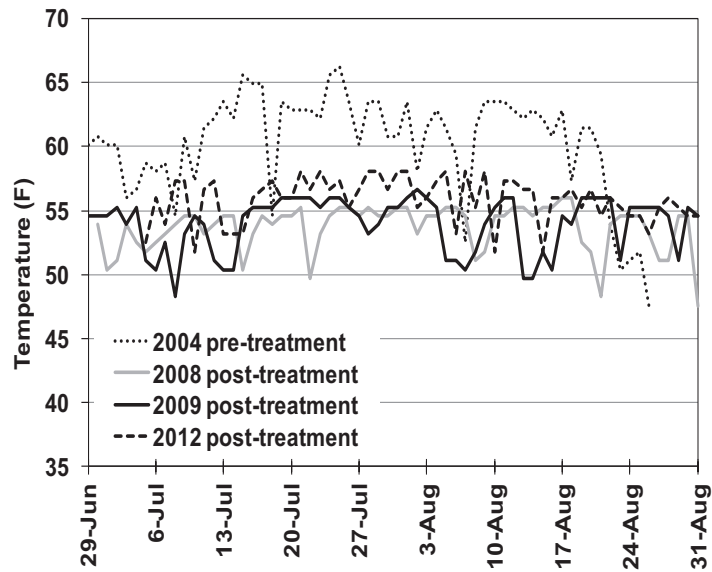
Figure 36. Maximum summer daily water temperatures for Jacobsen Spring Creek pre-treatment (2004) and post-treatment (2008-12).

Lincoln Spring Creek

Restoration objectives: Improve overall habitat conditions; improve spawning and rearing habitat for salmonids, eliminate fish passage barriers; and improve water quality conditions.

Project summary

Lincoln Spring Creek is a large spring creek tributary to Keep Cool Creek that enters the Blackfoot River at mile 105.2. This first-order, low-gradient spring creek is ~6.3 miles in length and originates from an alluvial aquifer and generates variable base-flow that seasonally rises and falls with the aquifer. Flowing west through private ranchland and the town of Lincoln, it enters Keep Cool Creek at mile 0.6. It is primarily a gravel based stream with a surrounding spruce riparian over-story.

Figure 37. Estimates of abundance for age 1 and older brown and brook trout in Lincoln Spring Creek at mile 3.8, 2007-2012.

Fisheries-related impairments include irrigation practices, heavy livestock grazing, and residential impacts and undersized culverts. These activities have suppressed riparian vegetation and contribute to an over-widened and shallow stream channel, fine sediment loading and generally simplified fish habitat.

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About 9,000' of Lincoln Spring Creek (mile 2.9 and 4.6) was reconstructed in 2008 to create a narrower and deepened channel with increased stream sinuosity. The project included the placement of in-stream wood, re-vegetation of stream banks, removal of three undersized culverts and an irrigation diversion upgrade. The project hopes to benefit salmonids by improving physical habitat and by reducing water temperature and sediment levels and restoring movement corridors.

Fish populations and other monitoring activities

Lincoln Spring Creek supports a community of brown and brook trout. Native trout have not been detected in fish population surveys undertaken between 1995 and 2010. In 2007, we established a pre-treatment fish population survey within the project area at mile 3.8 and continued to monitor fish populations through 2012. The surveys show brown trout are increasing in abundance following restoration. Conversely, brook trout appear to be decreasing (Figure 37). We also identified the incidental presence of adult westslope cutthroat trout in both 2011 and 2012 surveys. During the 2012 survey, we also observed increasing livestock grazing pressure (hoof shear of streambanks) within the riparian zone throughout the project area. Monitoring of grazing should be a priority to ensure grazing pressure is compatible with the recovery and protection of the restoration project.
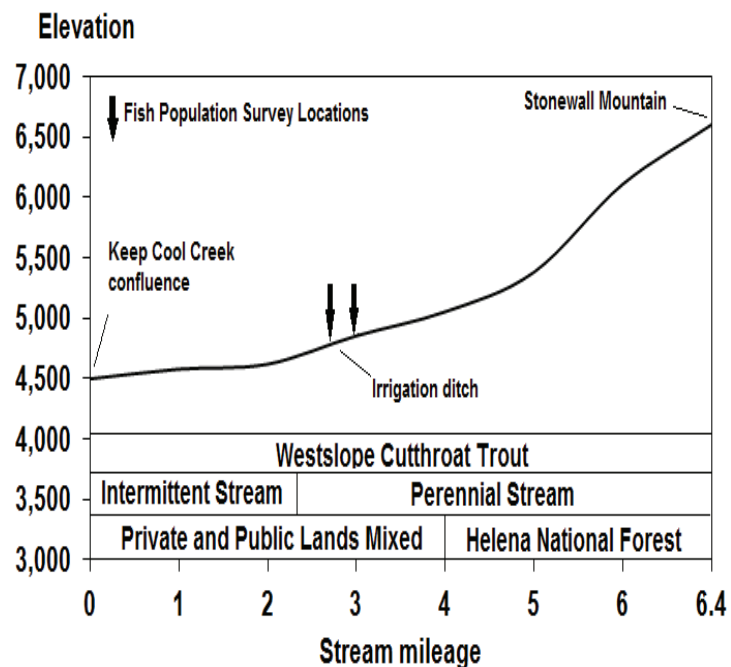
Figure 38. Longitudinal profile and fish population survey locations for Liverpool Creek, 2011.

Liverpool Creek

Introduction

Liverpool Creek is a first-order basin-fed stream that drains the southern slopes of Stonewall Mountain located on the Helena National Forest. The upper 2.4 miles flows south with a stream gradient decreasing from ~ 514 feet/mile near the headwaters to 138 feet/mile near mile 4.0 as it flows through public (State) and private ranch lands to its confluence with Keep Cool Creek at mile 2.2 (Figure 38).

The riparian vegetation consists of a mix of conifers and deciduous native shrubs. Once Liverpool Creek leaves the mountains, a majority of stream flow is diverted for irrigation. The landowner is currently working with Trout Unlimited to address dewatering and entrainment issues.

Fish population surveys

We surveyed fisheries in Liverpool Creek for the first time (miles 2.7 and 3.0) in 2011. We also sampled an un-screened irrigation ditch at mile 2.8. Our surveys found Liverpool Creek supports a resident westslope cutthroat trout population. Sampling at mile 2.7 (downstream of the irrigation ditch) identified relatively low numbers of westslope cutthroat trout compared to mile 3.0 (upstream of the ditch) (Figure 39). Sampling the irrigation ditch recorded a CPUE of 5.4.

Monture Creek

Restoration objectives: Restore habitat for spawning and rearing bull trout and westslope cutthroat trout; improve recruitment of bull trout and westslope cutthroat trout to the Blackfoot River; improve staging areas and thermal refugia for fluvial bull trout.

Figure 39. Electrofishing catch for age 1 and older westslope cutthroat trout in Liverpool Creek at miles 2.7 and 3.0, 2011.

Project summary

Monture Creek, a large tributary to the middle Blackfoot River, is a primary spawning and rearing tributary for fluvial bull trout and fluvial westslope cutthroat trout (Swanberg 1997, Schmetterling 2001). Monture Creek also serves as thermal refugia for fluvial bull trout during periods of Blackfoot River warming. Reproduction of westslope cutthroat trout and bull trout occurs primarily in the mid-to-upper basin. Lower Monture Creek supports the largest spawning run of fluvial rainbow trout upstream of Gold Creek.

Figure 40. Bull trout redd counts for Monture Creek index reach, 1989-2012.

Brook trout are absent upstream of an intermittent reach at stream mile 14 but are found in lower Monture Creek and its adjoining tributaries downstream of the intermittent reach (Pierce et al. 2008).

Riparian areas in the lower reaches of Monture Creek have a long history of riparian timber harvest and adverse grazing practices, with resulting adverse impacts to riparian habitats (Fitzgerald 1997). All lower tributaries of Monture Creek from Dunham Creek downstream likewise were identified as fisheries-impaired (Pierce et al. 2008). Many identified problems were corrected through a decade of cooperative restoration activities (Pierce et al. 1997; Pierce et al. 2001). Despite many improvements, excessive livestock access continues in certain riparian areas of Monture Creek.
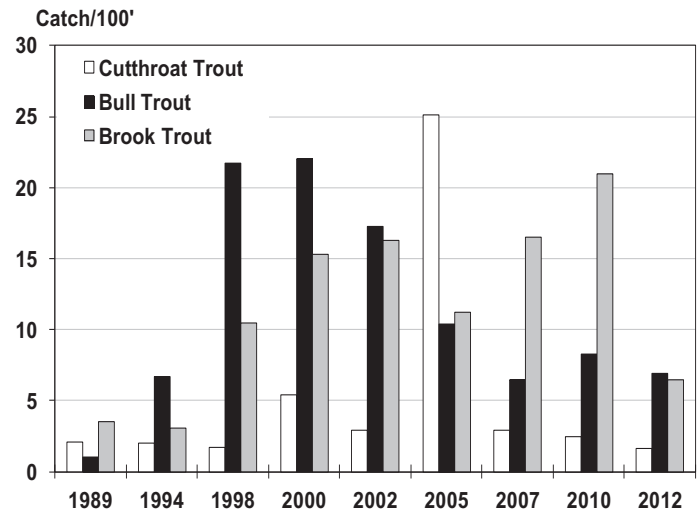
Figure 41. Electrofishing catch for age 0 and older trout in Monture Creek at mile 12.9, 1989-2012.

Fish populations and other monitoring activities

Monitoring for the 2011-2012 period included: 1) continued bull trout redd counts, 2) re-survey juvenile bull trout abundance at one long-term monitoring station (mile 12.9), and 3) water temperature monitoring (mile 1.8).

Bull trout redd counts were upward trending between 1989 and 2003, but then declined sharply during a period of protracted drought (2004-2009), before increasing between 2010 and 2012 (Figure 40). Long-term monitoring of juvenile bull trout (mile 12.9) indicated a declining trend since 2000, which seems to correspond with increases in brook trout (Figure 41). Water temperature monitoring, began in 1994 at mile 1.8, continued in 2011 and 2012. Summaries of water temperature monitoring are located in (Figure 20, Executive Summary and Appendix F).

Murphy Spring Creek
Restoration objectives: Restore habitat conditions suitable to westslope cutthroat trout and juvenile bull trout; prevent irrigation ditch losses; maintain minimum in-stream flows and provide rearing and recruitment for fluvial bull trout and cutthroat trout to the North Fork.
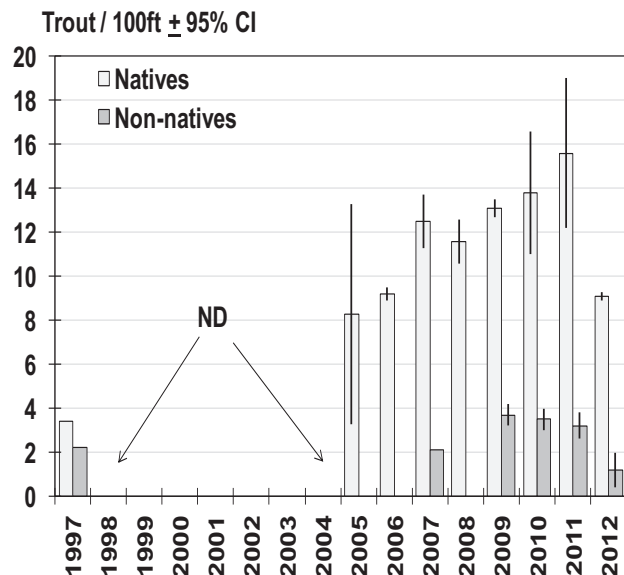
Figure 42. Estimates of abundance for age 1 and older native and nonnative trout in Murphy Spring Creek at mile 0.6, 1997-2012.

Project summary

 Murphy Spring Creek, a small westslope cutthroat trout dominated tributary, originates on the northeast side of Ovando Mountain and flows six miles south and enters the North Fork at mile 9.9. Murphy Spring Creek has a history of irrigation impacts and fish passage problems (Pierce et al. 2006). Irrigation problems involved chronic dewatering and entrainment of westslope cutthroat trout to the Murphy ditch at mile 1.8. Fish passage problems involved an undersized culvert at mile 0.5 and the poor condition of the Murphy diversion. The culvert reduced the upstream movement of native trout from the North Fork, while the diversion reduced downstream movement of westslope cutthroat trout from the headwaters to the North Fork through dewatering and entrainment.

 Restoration on Murphy Spring Creek began in 1998 with a new diversion fitted with a Denil fish ladder. In 2004-05, restoration expanded with an in-stream flow agreement that granted habitat maintenance flows as well as a 2.2 cfs minimal in-stream flow in Murphy Spring Creek. In 2006, a Coanda fish screen was placed at the diversion as a measure to eliminate losses of westslope cutthroat trout. The most recent work occurred in 2010 with: 1) an upgrade of the culvert at stream mile 0.5, and 2) the restoration of bankfull benches on the outside of stream bends and installation of toe-wood and log vanes in the stream channel on 880' of stream. A rock vane was also installed at the culvert for grade control. The landowner has also agreed to better manage cattle in the riparian area. Stream banks are now stable and from past livestock impacts.
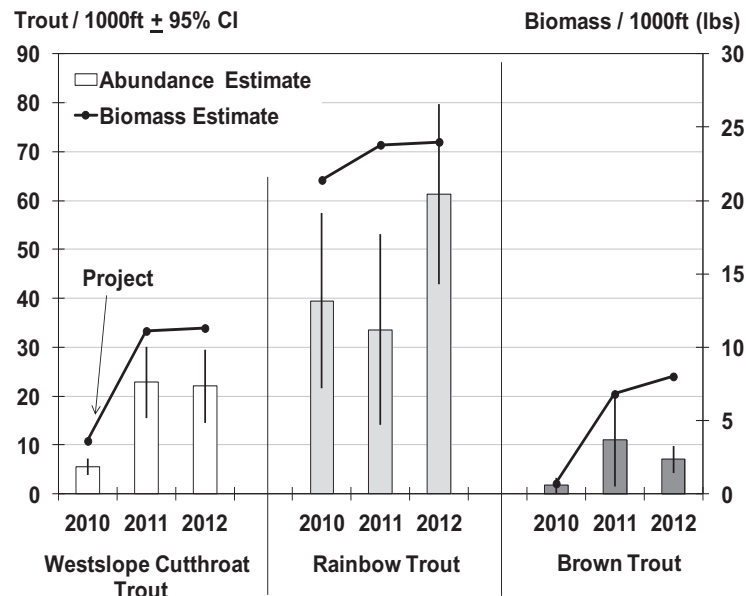
Figure 43. Estimates of abundance and biomass for age 1 and older trout in Nevada Creek at mile at mile 29, 2010-2012.
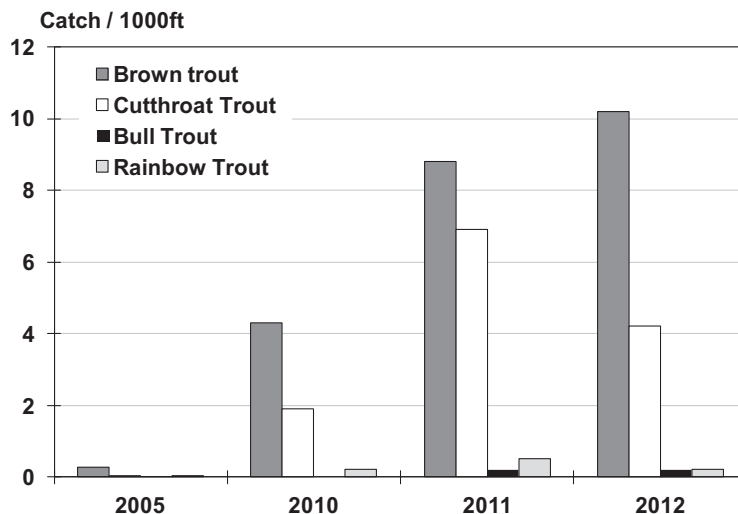
Figure 44. Electrofishing catch for age 1 and older trout in Nevada Creek immediately downstream of Nevada Spring Creek (miles 5.0-6.3), 2005-2012.

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Fish population monitoring

Murphy Spring Creek support primarily westslope cutthroat trout along with low numbers of bull trout and brook trout. Prior to restoration, we established a fish population monitoring site at mile 0.6 Following stream restoration activities, post-restoration fish population surveys between 2005 and 2012 show and increasing trend in the in the number native trout (Figure 42, Appendix B).

Nevada Creek

Restoration objectives: Restore a functioning stream and riparian area capable of maintaining complex habitat and providing environmental conditions supportive of trout.

Project summary

Nevada Creek is a large and heavily degraded tributary to the middle Blackfoot River, entering at river mile 67.8. Nevada Creek is a

Figure 45. Map showing three water temperature monitoring sites on Nevada Spring Creek and Nevada Creek. Locations relate to data summaries in Figure 46.

TMDL 303(d) water quality impaired stream (DEQ 2008), which reduces the ability of Nevada Creek to support coldwater salmonids over large reaches of the lower stream.

In the middle portion of Nevada Creek, immediately downstream of Nevada Reservoir (mile 29), a stream restoration project was completed on ~4,400' of channel in 2010 in order to restore more natural channel features to a degraded section of Nevada Creek. Here, Nevada Creek was incised, over-widened with eroding banks and lacking woody riparian vegetation. In addition to active channel work, a grazing management plan was also developed consistent with the protection of riparian resources. Finally, a diversion was replaced with cross-vane and retrofitted with a Coanda fish screen to exclude fish from the ditch.

Nevada Spring Creek near mouth

stats	Before			After		
	1994	2000	2001	2010	2011	2012
Max	76.9	77.3	74.5	63.5	67.9	64.9
75%	73.8	73.2	69.7	60.8	62.4	62.1
Median	72.0	70.4	68.3	60.1	60.8	61.2
25%	68.9	67.8	65.8	58.0	59.5	59.8
Min	55.9	64.2	54.6	50.4	58.1	55.6
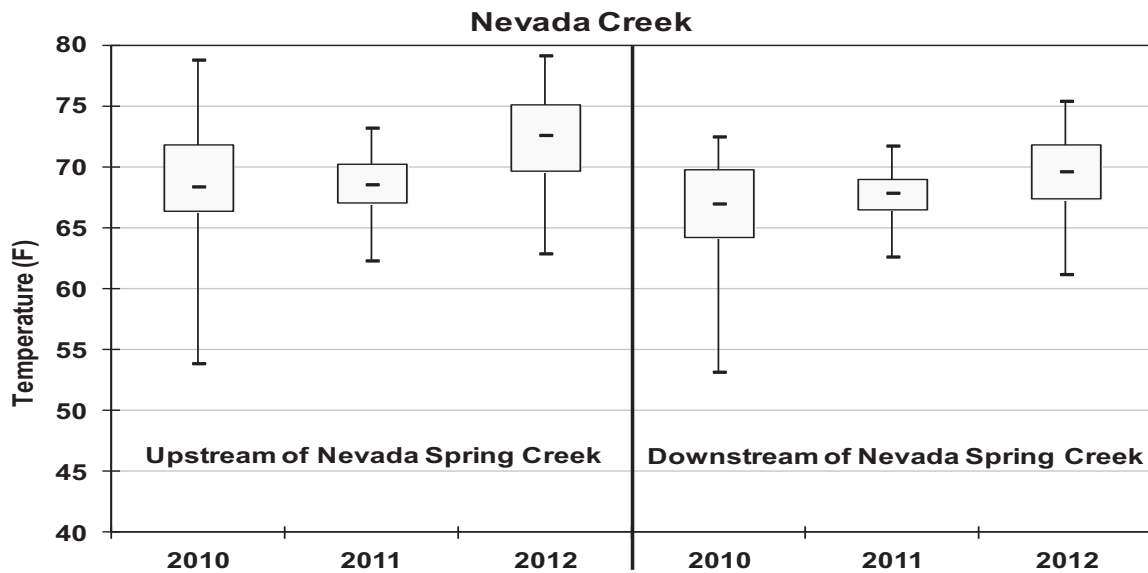
Nevada Creek

stats	Upstream			Downstream		
	2010	2011	2012	2010	2011	2012
Max	78.8	73.2	79.2	72.5	71.8	75.4
75%	71.8	70.2	75.1	69.7	69.0	71.8
Median	68.4	68.6	72.6	67.0	67.9	69.6
25%	66.3	67.0	69.6	64.2	66.4	67.4
Min	53.9	62.3	62.9	53.2	62.7	61.2

Figure 46. Temperature summary for Nevada Spring Creek pre-and post-treatment (top graph) and Nevada Creek up- and downstream of the Nevada Spring Creek (bottom) confluence 2010-2012.

Fish populations and other monitoring activities

Depending on location, Nevada Creek variously supports westslope cutthroat trout, rainbow trout and brown trout. Bull trout have also been reported in the upper creek (USFS unpublished data) and the incidental presence of bull trout has been identified by FWP in the lower Nevada Creek basin (Pierce et al. 2006). In 2011 and 2012, fish population surveys were conducted at two sites on Nevada Creek. The upper site (mile 29) is located within the reconstructed stream reach and was established in 2010 prior to restoration actions (Figure 43, Appendices A and C). We also continued to survey a section of lower Nevada Creek (mile 5.0-6.3) in 2011 and 2012 at a site originally established in 2005 immediately downstream of the Nevada Spring Creek confluence (Figure 44, Appendices A and C). During 2011-2012, we continued water temperature monitoring on Nevada Spring Creek and on Nevada Creek upstream and downstream of the Nevada Spring Creek confluence (miles 6.3 and 5.0) (Figure 45). A comparison of the pre-and post-treatment temperature dataset shows a cooling effect in Nevada Spring Creek at the mouth. Likewise, the up- and downstream comparison of summer water temperatures show a temperature reduction in Nevada Creek downstream of Nevada Spring Creek. All summary temperature data for Nevada Creek and Nevada Spring Creek are located in Appendix F.

North Fork Blackfoot River

Restoration objectives: Eliminate the loss of bull trout and westslope cutthroat trout to irrigation canals; manage riparian areas to protect habitat for native fish; improve recruitment of native fish to the Blackfoot River.

Project summary

The North Fork of the Blackfoot is the largest tributary to the Blackfoot River, with headwaters draining the Scapegoat Wilderness. Upon exiting the mountains near mile 12, the North Fork enters Kleinschmidt Flat, a large glacial outwash plain, before entering the middle Blackfoot River at river mile 54. Five irrigation canals, located on the Flat between stream miles 8.8 and 15.3, divert up to an estimated 40-60 cfs from the North Fork. In addition, this reach of
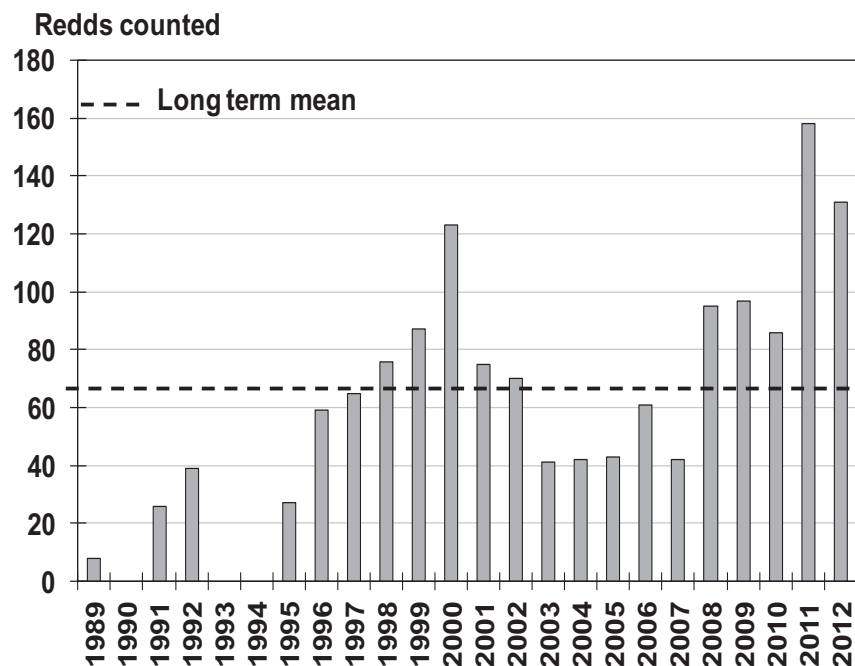
Figure 47. Bull trout redd counts in the North Fork of the Blackfoot River, 1989-2012. Redd counts were not performed in 1990, 1993 and 1994.

the North Fork naturally loses water to alluvium with flows returning as down-valley spring creek water. The North Fork is one of three primary fluvial bull trout spawning streams for the Blackfoot River. Bull trout recovery and related "core area" fisheries conservation projects (MBTRT 2000) involve developing compatible riparian grazing systems and eliminating entrainment of migratory bull trout from the five canals.

The North Fork has been the focus of extensive restoration activities, which include: 1) the screening of all irrigation canals of the mainstem North Fork, 2) instream restoration of all spring creeks (Rock Creek, Kleinschmidt Creek, Enders Spring Creek, Jacobsen Spring Creek and Murphy Spring Creeks), 3) instream flow enhancement and protection in the mainstem as well as tributaries, and 4) grazing management changes and road upgrades.

Fish Populations and other monitoring activities

The North Fork of the Blackfoot River supports one of the largest stocks of fluvial bull trout in the Blackfoot Basin. Fluvial bull trout of the North Fork are wide-ranging and migratory with a documented range extending from the upper Blackfoot River to the Clark Fork River (Swanberg 1997, Pierce et al. 2004, Schmetterling 2003). To monitor the North Fork stock of fluvial bull trout, FWP relies primarily on annual spawning (redd count) surveys as an index of population trends and abundance. These redd counts show population increases during the decade of the 1990 when protective angling regulations and the screening of all the North Fork ditches were enacted (Figure 47). This increase was followed by a seven-year decline (2001-2007) during a period of protracted drought. With the return of more normal flows and the removal of Milltown Dam, bull trout spawning has increased between 2008 and 2012.

In 2012, we surveyed juvenile bull trout numbers at one long-term monitoring site (river mile 17.2) established in 1989. Survey results show a trend similar to redd counts (Figure 48). In 2012, we also conducted electro-fishing surveys up and downstream of a fish screen as well as a by-pass channel off a large irrigation diversion at mile 15.3. In the diversion canal between the head-gate and fish screen, we recorded a bull
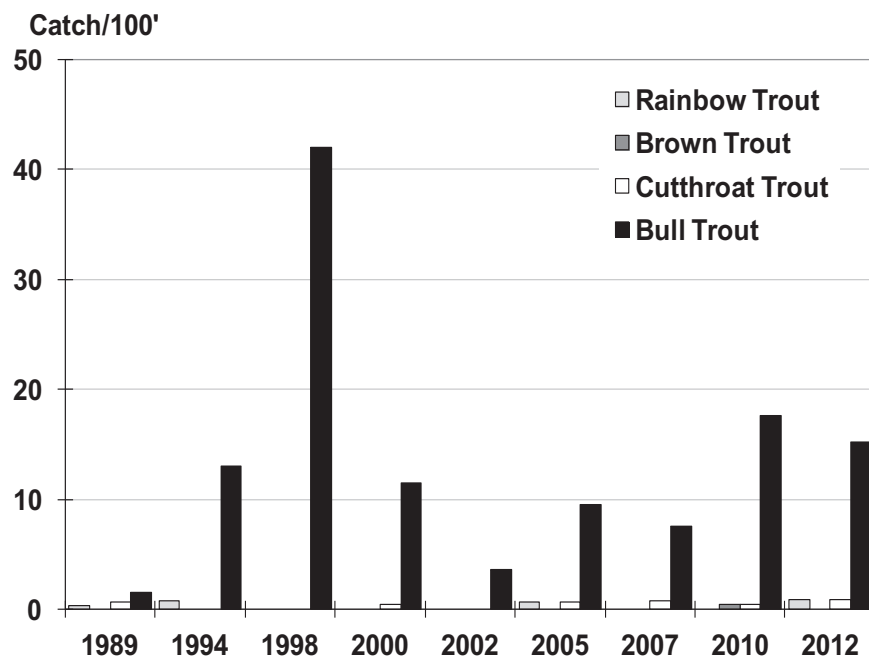
Figure 48. Electrofishing catch for age 0 and older trout in North Fork Blackfoot River at mile 17.2, 1989-2012.

trout CPUE of 7.4 compared to a CPUE of 2.9 in the canal downstream of the fish screen and a CPUE of 6.9 in the bypass (Appendix A). These survey results show the fish screen should be assessed to prevent fish losses.

Park Creek

Introduction

Park Creek is a first-order basin-fed stream that drains the southern slope of Stonewall Mountain near Lincoln. Park Creek flows 4.2 miles southwest through Helena National Forest, Plum Creek and state land to its confluence with Stonewall Creek at mile 2.5. Stream gradient varies from 1100 feet/mile near the headwaters to 320 feet/mile at the mouth (Figure 49). The riparian area in the headwaters supports a dense conifer dominated over-story with a dense under-story. Once Park Creek leaves the mountains, past timber harvest practices and public recreational road use has greatly contributed to areas of stream channel degradation. Due to past harvest of riparian conifers and the subsequent lack of large woody debris recruitment, Park Creek clearly lacks habitat complexity in lower reaches. A substantial amount of Park Creek's flow is diverted for irrigation and lost to in-stream fords during high water. Lower Park Creek has been damaged with a network of roads and fords. The lower portion of Park may be naturally intermittent.

Fish population surveys

FWP surveyed fisheries in Park Creek for the first time in 2011. Westslope cutthroat trout were the only fish present. Surveys included two sections (miles 1.1 and 1.4). We also sampled an un-screened irrigation ditch at mile 1.4. Upstream of the irrigation ditch, we recorded a CPUE of 6.1, compared to a CPUE of 2.1 in the ditch, verses a CPUE of 0 in Park Creek downstream of the diversion (Appendix A).
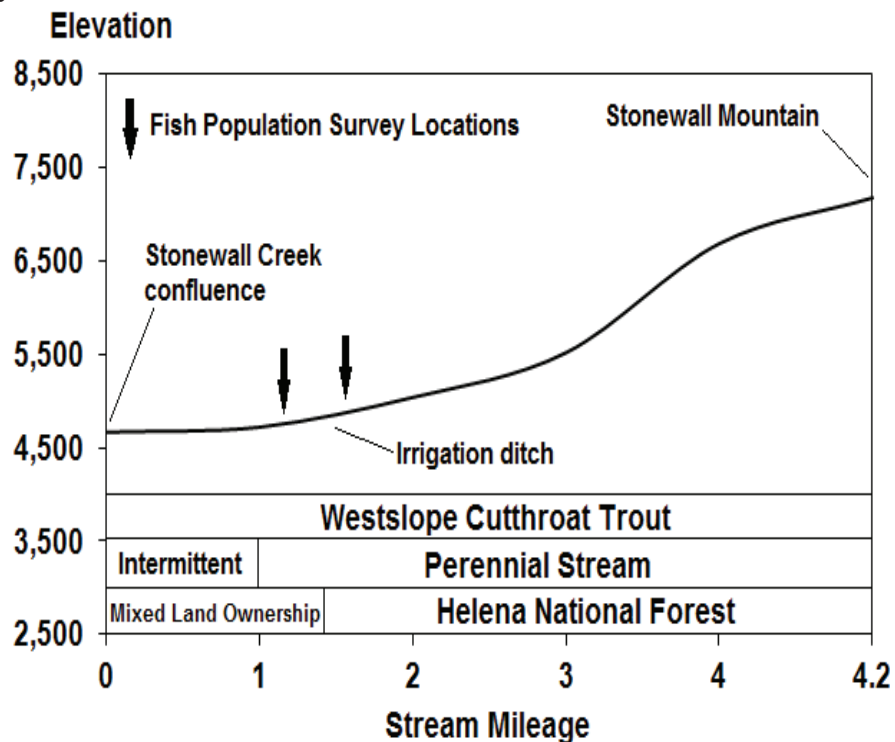
Figure 49. Longitudinal profile and fish population survey locations for Park Creek, 2011.

Pearson Creek

Restoration objectives: Improve status of westslope cutthroat trout population and increase recruitment of fluvial westslope cutthroat trout to the Blackfoot River.

Project summary

Pearson Creek is a small second-order tributary to Chamberlain Creek with a base-flow of one cfs. Pearson Creek has a history of channel alterations and adverse irrigation and riparian land management practices (grazing and timber harvest) in its lower two-miles of channel. Beginning in 1994, Pearson Creek has been the focus of a holistic restoration project involving channel reconstruction and in-stream habitat work, in-stream flow enhancement (water leasing), conservation easements and riparian grazing changes. Additional riparian grazing improvements completed in 2006 included riparian corridor fencing for the lower two miles of stream, off-stream water developments and armoring a road crossing. Despite these changes, unplanned cattle use continues to

Figure 50. Estimates of abundance for age 1 and older westslope cutthroat trout in Pearson Creek at miles 0.5 and 1.1.

hamper habitat recovery on Pearson Creek. In addition, road crossing and historical channel manipulations continue to adversely influence westslope cutthroat trout habitat in Pearson Creek. Corrective actions issues are currently ongoing.

Fish population monitoring

Pearson Creek is a fluvial westslope cutthroat trout spawning stream connected to Chamberlain Creek. In 2011 and 2012, we continued fish population monitoring at two sites on lower Pearson Creek (Figure 50). The upstream site (mile 1.1) was established in 1999 prior to in-stream restoration activities. In 2005, we established the downstream site (mile 0.5) to assess road-crossing and grazing impacts on lower Pearson Creek. These data show no improvement in the abundance following corrective grazing changes, which indicates problems at the upstream road crossing. A restoration project to correct the road crossing and related channel problems was completed in 2013.

Poorman Creek

Restoration objectives: Improve riparian habitat conditions and enhance in-stream flows; restore migration corridors; improve recruitment of native fish to the Blackfoot River.

Project summary

Poorman Creek, one of the larger tributaries from the Garnet Mountains enters the Blackfoot River at river mile 108. Poorman Creek impairments have included hardrock and placer mining, irrigation dewatering, fish losses to ditches, channel instability, excessive riparian grazing pressure, subdivision impacts and multiple undersized culverts. Corrective actions began in 2002 and continue through the present. These include a focus on lower Poorman Creek, which includes in-stream flow enhancement (water lease) and ditch screening through flood-to-sprinkler irrigation conversion, stream crossing upgrades, and riparian grazing changes (corridor fencing, off-stream water) and shrub plantings. Riparian areas and channel conditions appear to be improving with the continuation of compatible grazing practices. In addition, several road crossings have been upgraded on the mainstem of Poorman Creek to improve habitat connectivity for native trout.
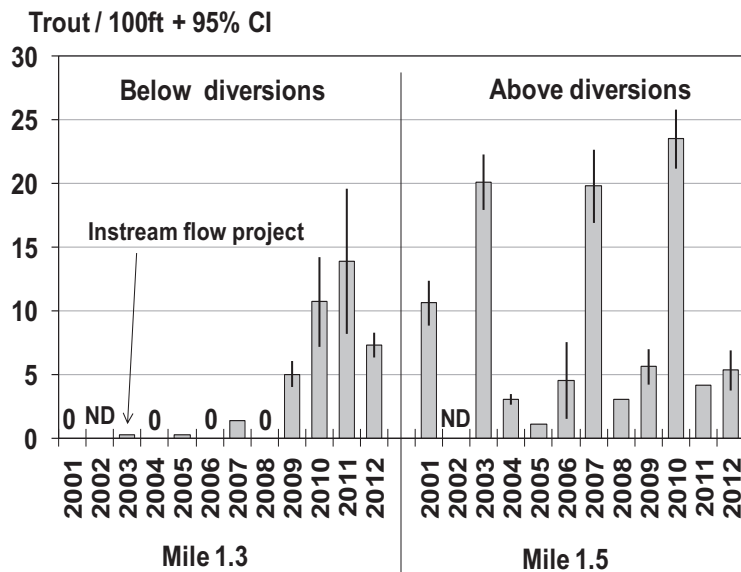
Figure 51. Estimates of abundance for age 1 and older trout in Poorman Creek at miles 1.3 and 1.5, 2001 – 2012.

Fish population and other monitoring activities

Poorman Creek supports genetically pure westslope cutthroat trout, as well as brown trout and brook trout fisheries. Poorman Creek is the only small stream south of the Blackfoot River to support bull trout reproduction. The relative abundance of native trout tends to increase in the upstream direction, whereas non-native fish occupy lower Poorman Creek. In 2001, we established two fish population monitoring sites in lower Poorman Creek. These sites are located up- and downstream of two diversions (miles 1.3 and 1.5). These monitoring sites were established prior to flow enhancement, ditch screening and passive restoration actions associated with a reduction in livestock grazing.

Survey results from both monitoring sites from 2001-2012 are shown in Figure 51. These surveys show improved numbers of trout downstream of the diversion where trout were absent prior to restoration. Trout numbers are highly variable upstream of the diversion and this may relate to the loss of instream wood during recent flow years. A summary of 2012 water temperature monitoring on Poorman Creek is located in Appendix F.

Rock Creek

Restoration objectives: Restore migration corridors for native fish; restore natural stream morphology to improve spawning and rearing conditions for all fish using the system.

Project summary

Rock Creek is the largest tributary to the lower North Fork of the Blackfoot River entering at stream mile 6.1. In the past, Rock Creek has been degraded over most of its 8.2-mile length due to a wide range of past channel alterations and riparian management activities (Pierce 1990; Pierce et al. 1997, 2006). Rock Creek has also been the focus of continued restoration since 1990. Restoration actions involved working with 13 separate landowners on grazing improvements, in-stream flow enhancement, channel reconstruction and re-vegetation. Active restoration is now completed over the entire length of Rock Creek and its primary tributaries, the South Fork of Rock Creek, Salmon Creek and Dry Creek. The sustainability of fisheries improvements now hinge on collective ability of landowners to manage the riparian area for the passive recovery of riparian plants.

Fish populations and other monitoring activities

Rock Creek supports a mixed salmonid community including brown trout and rainbow trout in lower reaches, a resident brook trout population, limited bull trout rearing and a migration corridor for fluvial westslope cutthroat trout to headwater areas.
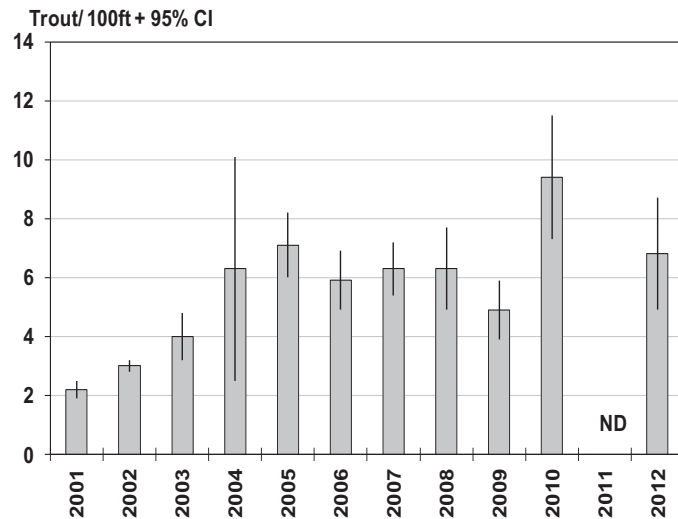
Figure 52. Estimates of abundance for age 1 and older trout in Rock Creek at stream mile 1.6, 2001-2012.

Photo 1. This 2013 aerial photo shows the recent widening of a restored segment of Rock Creek channel following excessive livestock pressure.

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In 2011 and 2012, we continued to monitor fish populations in lower Rock Creek (mile 1.6) where the stream was reconstructed in 1999 (Figure 52). This location continues to be dominated by brown trout and brook trout species and overall numbers appear to have leveled. In 2013, we observed low to moderate livestock impacts to stream banks and signs of channel widening in the monitoring section, as well as the return of excessive livestock pressure on a restored segment of Rock Creek (Photo 1).

Sauerkraut Creek

Restoration objectives: Restore natural stream morphology to improve spawning and rearing conditions for native cutthroat trout and bull trout.

Project summary

Sauerkraut Creek is a small tributary to the upper Blackfoot River, entering at river mile 102.1 with a base flow of 3-4 cfs. Sauerkraut Creek has a long history of placer mining, which has resulted in severe channel alterations, including channelization, the loss of floodplain function and contributes to intermittent flows in one section of stream. In addition, inadequate stream crossings and overgrazing by livestock has also contributed to the degraded channel conditions. Restoration of Sauerkraut Creek began in 2008 when a conservation easement intended to promote the conservation of native trout was placed on private ranchland. As part of the agreement, a stream restoration project was developed in upper Sauerkraut Creek (miles 2-3) to correct past mining impacts. Restoration involved the reconstruction of

Figure 53. Estimates of abundance age 1 and older trout in Sauerkraut Creek at treatment (mile 2.9) and reference reaches (mile 3.2), 2007-2012.

approximately 5,000 feet of Sauerkraut Creek, a grazing management plan including riparian fencing and off-site water developments, shrub transplants, seeding and weed management. In 2010-12, three undersized stream crossings (miles 0.3, 1.5 and 1.8) were upgraded from undersized culverts to bridges to accommodate fish passage and channel function. In addition, irrigation ditches were consolidated into a single diversion with a fish screen. An instream flow agreement was also secured for a minimum flow of three cfs on the lower two miles of Sauerkraut in 2012. Future restoration work in both lower and upper Sauerkraut Creek is currently in development phases.

Fish population monitoring

Sauerkraut Creek supports primarily westslope cutthroat trout along with low numbers of brook and bull trout in the headwaters and a mixed community of salmonids in the lower stream (Appendix B). Sauerkraut Creek supports a small run of migratory cutthroat trout as identified in past telemetry study (Pierce et al. 2007). Westslope cutthroat trout have been tested as genetically unaltered.

To develop a fisheries baseline for the upper Sauerkraut Creek restoration project, we established a fisheries monitoring site at an upstream reference reach site (mile 3.2) and within the treatment site (mile 2.9) beginning in 2007 (Figure 53). Reference reach surveys (mile 3.2) recorded a slight decline, probably the result of fish redistribution or increases in livestock grazing pressure on streambanks.

Snowbank Creek

Restoration objectives: Restore migration corridor for native fish; enhance in-stream flows; eliminate loss of bull trout and westslope cutthroat trout to a diversion ditch; improve recruitment of native fish to Blackfoot River.

Project summary

Snowbank Creek is a first-order tributary to Copper Creek, entering at stream mile 5.9. Snowbank Creek was identified as chronically dewatered with fish passage and entrainment problems in 2003. Following initial surveys, instream flows were enhanced to a minimum of four cfs, in 2004, and then in 2009 the diversion was replaced with one that allows fish passage and a Coanda fish screen was placed in the ditch to eliminate entrainment. Additional work is now being planned to: 1) remove an undersized culvert from lower Snowbank Creek, and 2) prevent loss of water from Snowbank Lake.

Fish populations and other monitoring activities

Fish population monitoring began in 2003 and continued through 2012 downstream of the diversion (Figure 54). Initial surveys identified increases in cutthroat trout abundance and the apparent re-colonization of bull trout from Copper Creek into Snowbank Creek after flows in lower Snowbank Creek were reestablished (Pierce et al. 2011). In 2008, the USFS documented bull trout spawning in Snowbank Creek for the first time, which included the historically dewatered stream segment. Between 2008 and 2012, bull trout redd counts in Snowbank Creek have averaged 18 (range: 1-34). Results from

Figure 54. Population estimates for age 1 and older cutthroat trout and bull trout downstream of Snowbank Creek diversion, 2003-2012.

genetic samples collected from westslope cutthroat trout in 2009 found no introgression.

Finally, we monitored water temperature during the summers of 2011 and 2012 at mile 0.2. This monitoring shows cold and stable temperatures with maximum summer temperatures ranging from 56-58°F (Appendix F).

Wasson Creek

Restoration Objectives: Restore channel maintenance and minimal in-stream flow; restore migration corridors in lower Wasson Creek in order to provide recruitment of westslope cutthroat trout to Nevada Spring Creek; restore channel conditions to support spawning and rearing conditions in lower Wasson Creek; prevent fish losses to irrigation ditches; prevent the introduction of unwanted fish into the drainage.

Project summary
      Wasson Creek is a small second-order basin-fed tributary to Nevada Spring Creek. It joins Nevada Spring Creek ~100feet below the (artesian) spring source, contributing base-flow of about 1-2 cfs. Wasson Creek has a long history of fisheries-related impairments that include fish passage barriers (culverts and diversions), irrigation dewatering and entrainment of fish to ditches, livestock damage to stream banks and channelization.

      In 2003, we began to develop a stream restoration project following and concurrent with downstream improvements on Nevada Spring Creek. Fisheries elements on the Wasson Creek project include: 1) grazing management over the length of the project, 2) irrigation changes to accommodate in-stream flows (low flows and channel maintenance) and fish passage, and 3) channel reconstruction and floodplain containment in the lower mile (Pierce et al. 2006). In 2005, a 10-year in-stream flow lease also went into effect. Since then low flows remained at or near the target of 0.75 cfs. The final element to the project was the installation of two fish screens at both irrigation diversions in the spring of 2007. With the exception of grazing changes on upstream properties, Wasson Creek is now in the recovery phase of the project.
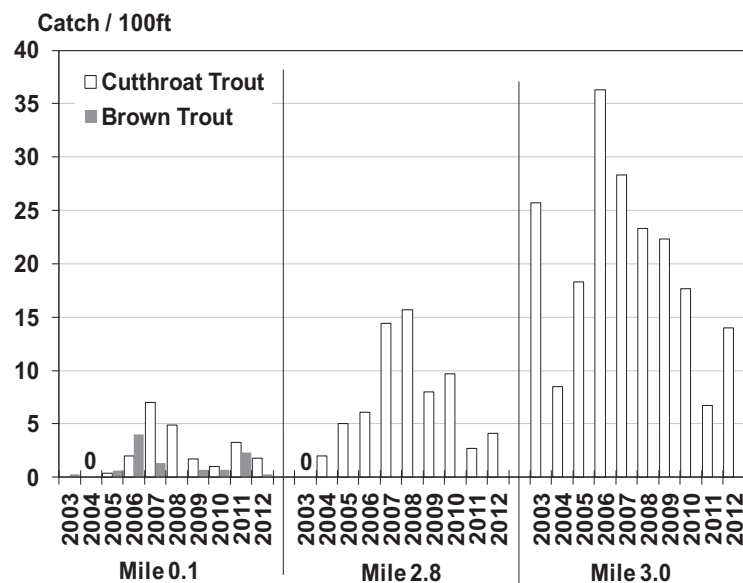
Figure 55. Electrofishing catch for age 0 and older westslope cutthroat trout and brown trout at three monitoring locations on Wasson Creek, 2003-2012.

Fish Populations and other monitoring activities

To monitor the Wasson Creek restoration project, we established three fish population monitoring sites in 2003 at miles 0.1 (near mouth), 2.8 (below diversions) and 3.0 (upstream of diversions). Results of these surveys are shown in Figure 55. In 2011-12, we also monitored water temperature near the mouth (mile 0.1). In addition, Trout Unlimited monitored stream discharge upstream (mile 2.9) and downstream (mile 2.1) of the diversions. These data are presented in Results Part IV.

Since 2004, water temperature monitoring at the mouth continues to show summer water temperatures cooling (Figure 56, Appendix F). This cooling appears to result from cumulative restoration measures including the early recovery of streamside plants, increased flows, and the passive narrowing of the channel in response to stream-side grazing changes.
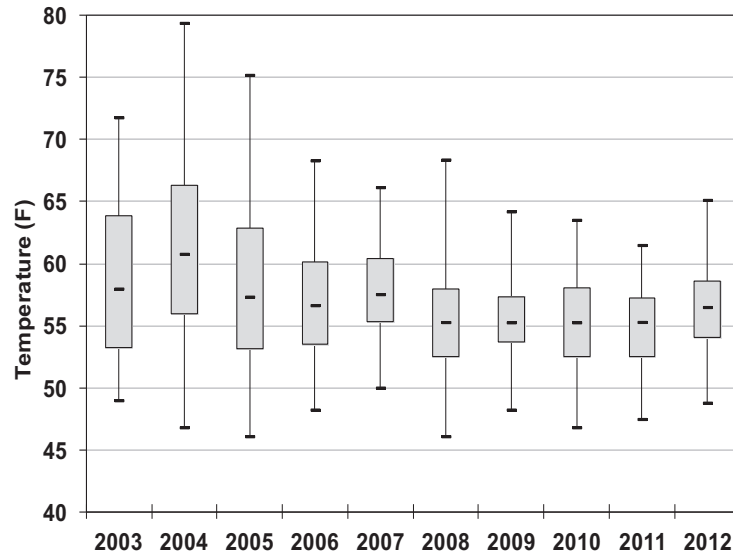
Figure 56. July water temperatures for Wasson Creek near the mouth, 2003-2012.

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Acknowledgements

PART IV: Special Studies

Westslope cutthroat trout movements through restored habitat and Coanda diversions in the Nevada Spring Creek complex, Blackfoot Basin, Montana

Draft Report

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Abstract

In the Blackfoot Basin of western Montana, the recovery of migratory westslope cutthroat trout requires landscape conservation as well as restoration of spawning tributaries. Westslope cutthroat trout are now increasing in the Blackfoot River and several streams, including Nevada Spring Creek, where natural channel, flow and temperature regimes have reestablished aquatic habitat and migration corridors. To examine whether restoration has improved corridors for migration, we tracked the movements of 14 adult westslope cutthroat trout from wintering areas in lower Nevada Creek (downstream of Nevada Spring Creek) to spawning and summering areas. Ten fish moved through Nevada Spring Creek upstream a median distance of 7.7 km (range, 7.6–16.9) to spawning sites at the headwaters of Wasson Creek through stream reaches where channels were reconstructed, instream flows enhanced and grazing practices improved. Eight of the 10 fish that entered Wasson Creek spawned in a concentrated area upstream of two experimental diversion/fish screen structures located in the main channel of Wasson Creek. Compared to Wasson Creek spawners, pre-spawning movements of the remaining four radio-tagged fish were much farther (median, 51.8 km; range, 44.9-63.1). These four fish moved downstream through Nevada Creek into the Blackfoot River and then ascended upper Blackfoot River before entering two separate spawning tributaries. This telemetry study indicates that restoration can improve migration corridors, which in turn promote the recovery of migratory westslope cutthroat trout and that spring-influenced tributaries like Nevada Spring Creek provide important over-winter habitat for westslope cutthroat trout that spawn and summer elsewhere in the basin.

Introduction

Native salmonids were once abundant and widespread across the western United States, but as natural landscapes were modified many native salmonids declined to an imperiled state (Nehlsen et al. 1991; Behnke 1992; Thurow et al. 1997). Declines were largely associated with mining activities, timber extraction, stream channelization, irrigation practices, dams, riparian grazing, over-fishing and the adverse influence of nonnative species (e.g., Meehan 1991; Behnke 1992; Thurow et al. 1997). As a result of these human-induced threats, all 14 subspecies of native cutthroat trout *Oncorhynchus clarkii* are either imperiled ($n = 12$) or extinct ($n = 2$) (Behnke 1992, 2002). In Montana, the westslope cutthroat trout (*Oncorhynchus clarkii lewisi),* a species-of-special-concern (Shepard et al. 1997, 2005), is especially imperiled east of the Continental Divide (i.e., upper Missouri Basin) where most populations are isolated above barriers in small (< 10 km) habitat fragments (Shepard et al. 1997). In Montana west of the Continental Divide, westslope cutthroat trout populations have also declined; however, populations are more widely distributed (Shepard et al. 2005; Fausch et al. 2009), present in greater abundance and possess higher levels of life history and genetic diversity (Shepard et al. 2005; Fausch et al. 2009; Drinan et al. 2011).

Westslope cutthroat trout have migratory and stream-resident life histories, both of which are often represented in the same population (Rieman and Dunham 2000). Stream-resident fish occupy small tributaries their entire lives and can persist in isolated segments of stream. Conversely, migratory fish move downstream to larger rivers (or lakes) at age 2-4, where they mature at much larger sizes before returning to natal tributaries as adults to spawn. Migratory westslope cutthroat trout thereby require much larger stream networks to fulfill their life history requirements than resident fish (Behnke 1992, 2002). In the Blackfoot River basin of western Montana, spawners often migrate >50 river kilometers (rkm hereafter) upriver in May during the rising limb of the hydrograph, enter small streams where they spawn near the peak of the hydrograph (May and June), and then move downstream to larger waters as flows decline (Schmetterling 2001; Pierce et al. 2007).

Diverse life histories of native trout allow for dispersal and genetic exchange among subpopulations (Rieman and Dunham 2000; Fausch et al. 2009), which provides resiliency to natural stressors such as wildfire and debris flows (Fausch et al. 2009; Sestrich et al. 2011). Because migratory native trout require wide-ranging and often complex movements across a river network (Swanberg 1997; Schmetterling 2001; Petty et al. 2012), their recovery often requires multi-scale conservation (Pierce et al. 2005, 2013; USFWS 2010), along with site-specific restoration techniques such as instream habitat restoration and balancing water needed for irrigation with the needs of migratory stocks (Pierce et al. 2007, 2013; Gale et al. 2008).

Although restoration is often conducted to conserve migratory native trout, few studies have examined metapopulation and life history dynamics of native trout from a restoration perspective (Rieman and Dunham 2000, Roni 2005; *but see* Petty et al. 2012). Likewise, the efficacy of restoration to mediate irrigation effects, such as managing for more natural flow regimes and using new technologies (e. g., Coanda-effect fish screens; Wahl 2001, 2003) to reconnect seasonally-occupied habitats and limit entrainment of fish in irrigation systems, are rarely evaluated and poorly understood (Moyle and Israel 2005; Gale et al. 2008; Simpson and Ostrand 2012). Multi-scale studies that document effects of restoration techniques on migratory trout are critical because migratory trout have experienced more severe declines than resident forms due to in part to greater impacts

from irrigation practices (McIntyre and Rieman 1995; Gale et al. 2008; Simpson and Ostrand 2012).

In the Blackfoot Basin of western Montana, declines of migratory westslope cutthroat trout and bull trout (*Salvelinus confluentus*) in the Blackfoot River during the 1980s triggered basin-wide no-harvest (i.e., catch-and release) regulations in 1990, combined with a program to restore degraded spawning tributaries on private ranch- and timberlands from 1990 to the present (Aitken 1997; Schmetterling 2001; Pierce et al. 2005, 2007, 2013). Following these actions, the westslope cutthroat trout have increased in abundance during the last 20 years in the Blackfoot River, where they now provide a valuable sport fishery for western Montana (Montana Fish, Wildlife and Parks 2012; Pierce and Podner 2013).

Within a context of these management strategies, restored tributaries of the Blackfoot River offer an ideal opportunity to examine the effects of multi-scale efforts to conserve migratory westslope cutthroat trout. This study expands on a prior study showing that westslope cutthroat trout increased in abundance, while documenting a community-level shift from brown trout (*Salmo trutta*) to westslope cutthroat trout following restoration of Nevada Spring Creek and Wasson Creek, a small adjoining tributary (Pierce et al. 2013). In this study, we examine the post-treatment spawning behavior of migratory westslope cutthroat trout associated with this local expansion within a context of irrigation system and multi-scale restoration activities. Specific study objectives are to: 1) examine migration behaviors of westslope cutthroat trout from their wintering areas into summer and to identify on spawning sites for fish inhabiting the Nevada Creek complex, and 2) document the efficacy of irrigation-based restoration techniques involving an experimental Coanda fish screen and diversion for passing migratory spawners in Wasson Creek. Our broader goal is to help improve management of migratory trout and to guide habitat restoration on private lands where native trout conservation often requires balancing irrigation and other land uses with the life history and habitat needs of migratory fish.

Study Area

The Blackfoot River, a fifth-order tributary (Strahler 1957) of the upper Columbia River, lies in west-central Montana and flows west 212 rkm from the Continental Divide to its confluence with the Clark Fork River in Bonner, Montana (Figure 1). The Blackfoot Basin is regionally variable with subalpine forests in the high mountains, montane woodlands at the mid-elevations and semi-arid glacial topography on the valley floor. Land ownership in the Blackfoot Basin is approximately 44% private land and 46% public land. Public lands occupy the mountainous areas, while private lands occupy the foothills and bottomlands where traditional uses of the land include mining, timber harvest and agriculture. These activities have degraded habitat or led the loss of habitat connectivity for westslope cutthroat trout in most tributaries of the Blackfoot River (Peters and Spoon 1989; Schmetterling 2001; Pierce et al. 2005, 2007).

Our study involves the Nevada Spring Creek complex (i.e., Wasson Creek, Nevada Spring Creek and lower Nevada Creek) located in the Nevada Creek drainage (Figure 1). Nevada Creek has been intensively managed for irrigation livestock production, which led to flow alterations, impaired water quality and depleted fisheries (DEQ 2007; Pierce et al. 2007). Nevada Spring Creek, located in the lower Nevada Creek drainage, originates from an artesian spring (Figure 1) that discharges 0.2-0.4 m^3/s of water with a nearly constant annual temperature ranging from 6.7-7.8 $^{\circ}$C (Pierce et al.

2002). From this spring source, Nevada Spring Creek flows 7.1 rkm and enters Nevada Creek 10.1 rkm above its mouth. Prior to 2005, Nevada Spring Creek was over-widened and heavily degraded with high summer temperatures near 25 $^{\circ}$C at its junction with Nevada Creek (Pierce and Peters 1990). Likewise, the lower 3.8 rkm of Wasson Creek, a tributary to upper Nevada Spring Creek, was dewatered and damaged by intensive agricultural practices. Electrofishing surveys found westslope cutthroat trout were incidental or absent from sampled segments of lower Wasson, lower Nevada Spring Creek and lower Nevada Creek (FWP unpublished data; Pierce et al. 2013). Indeed, an intensive 6.1 rkm electrofishing survey of Nevada Creek downstream of the Nevada Spring Creek confluence captured only a single brown trout (and no westslope cutthroat trout) in April 1990 (Montana Fish, Wildlife and Parks, unpublished data).
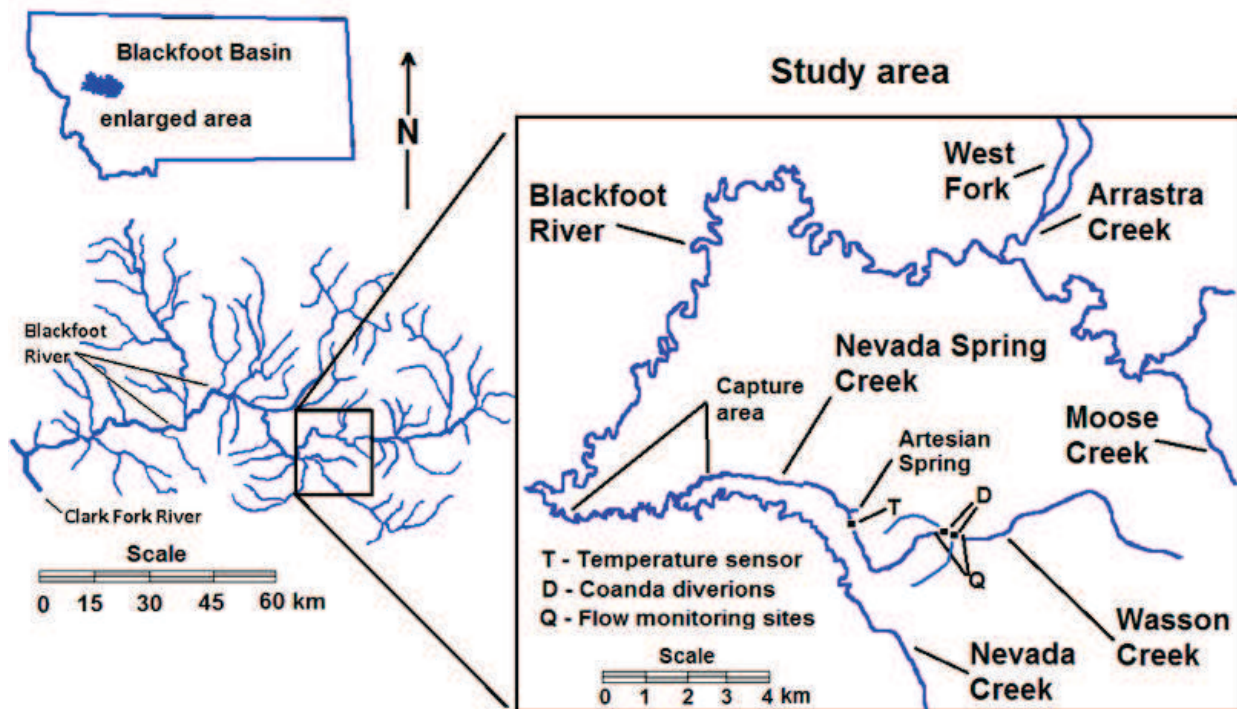
Figure 1. Location map showing the Blackfoot Basin and the study area. Also shown are the capture locations of fish, flow and temperature monitoring sites and locations of the two Coanda diversions.

Both Nevada Spring Creek and Wasson Creek were restored over a 10-year period (Pierce et al. 2013). Nevada Spring Creek was completely restored by forming a deep narrow channel, restricting livestock grazing in riparian areas, enhancing instream flows, and placing a protective conservation easement along the entire stream (Table 1). Restoration actions on Wasson Creek were similar, but also include the addition of two experimental Coanda-effect fish screens at two diversion points (photo 1a and described below) in order to eliminate entrainment and facilitate movements of fish during the irrigation season. These combined treatments were intended to recreate more natural channels and flow regimes, reduce temperatures in Nevada Spring Creek and restore habitat connectivity (Table 1). Following these activities, fisheries monitoring not only documented the down-valley expansion of westslope cutthroat (Pierce et al. 2013), but

also the increasing presence of larger adult westslope cutthroat (fish>300mm in total length) in Nevada Creek downstream of the Nevada Spring Creek confluence (Pierce and Podner 2013).

Stream name	Width/Depth ratio		Sinuosity		% pool area		Max. summer temp (C)		Min. summer flow (m³/s)		Ditch entrainment (# age 1+ trout /30m)	
	before	after	before	after	before	after	before	after	before	after	Before	After
Nevada Spring Cr.	22	3.2	1.4	1.7	51	71	25	18	0.17	0.28	na	na
Wasson Cr.	3	0.7	1	1.5	nd	nd	22	18	0	0.02	1.3	0

Table 1. Summary of stream metrics before and after restoration. An 'nd' refers to no data and 'na' means not applicable.

Irrigation improvements - Instream flow and the Coanda fish screens – Upgrades to irrigation systems in Wasson Creek enhanced instream flows and placed a pair of site-designed Coanda-effect (Coanda *hereafter*) diversion/fish screens in the mainstem of Wasson Creek at two diversion points (Figure 1, Photo 1). Instream flow enhancement was intended to mimic natural flow regimes including high and low flows, while maintaining a minimum baseflow (>0.02 m³/s) downstream of the diversion points in order to reestablish spawning, rearing and movement corridors for westslope cutthroat trout in areas of restored habitat. The Coanda in this study is an experimental structure intended to allow the uninhibited movement of fish and eliminate ditch entrainment, while also allowing diversion of water from the main channel of Wasson Creek into an irrigation ditch (Photo 1a). To accomplish these functions, the structure is slightly elevated above the bed of the channel, which allows water to flow over the screen and wash debris from the screen in a manner that provides for the upstream movement of fish, while preventing fish from entering the ditches. The Coanda-effect fish screen itself is a slightly-tilted angular "wedge wire" design (Wahl 2001, 2003) with closely spaced bars (gap = 0.5 mm), which shears water from surface of the screen and routes water into a buried pipe that then discharges into irrigation ditches.

Methods
        Radio-telemetry – Consistent with previous studies (Schmetterling 2001; Pierce et al. 2007), we captured adult westslope cutthroat in Nevada Creek and lower Nevada Spring Creek by electrofishing suspected wintering areas prior to spawning migrations. We implanted 14 fish at capture locations with continuous radio transmitters (model MST-930 miniature sensor tag; Lotek Wireless, Newmarket Ontario, Canada) between 18 and 21 April 2011 (*n* = 6) and 9 and 10 April 2012 (*n* = 8) and tracked these fish to their spawning sites and summering areas. At the time of capture, these fish ranged from 292 to 377 mm in total length (mean, 333 mm) and from 299 to 590 g in weight (mean, 438 g). We selected larger fish in this study to increase the likelihood that radio tagged fish were sexually mature. To confirm visual identification of individuals as westslope cutthroat trout, all 14 fish were tested for genetic purity by removing a small portion of fin and assessing eight microsatellite loci diagnostic between westslope cutthroat trout and rainbow trout as described by Muhlfeld et al. (2009a).
        Transmitters were distributed in fish captured over 8.7 rkm of stream, which included the lower 1.3 rkm of Nevada Spring Creek (*n* = 3) and an adjoining 7.4 rkm section of Nevada Creek downstream of the Nevada Spring Creek confluence (Figure 1). Individually coded transmitters weighed 4.0 g, had an estimated life of 278 days, emitted an individual coded signal, did not exceed 2 percent of fish weight (Winter 1996), and

were implanted following standard surgical methods (Swanberg et al. 1999). Technicians use an omni-directional whip antenna mounted on a truck, all-terrain vehicle or canoe when identifying general fish locations and then identified specific locations on foot using a hand held three-element Yagi antenna. Technicians located fish weekly prior to migrations, 3-4 times per week during migrations and spawning, once per week following spawning and generally twice per month thereafter. All river locations and movements of westslope cutthroat trout were referenced by rkm.

Fish were assumed to have spawned at their upper-most detected location if they ascended a stream with suitable spawning habitats during the spring (May-June) spawning period (Schmetterling 2001). Suitable spawning habitats were identified by observations of spawning, presence of redds and/or age-0 westslope cutthroat trout. We estimated the timing of migration and spawning events as the median date between two contacts for a given event, and the peak of spawning for the entire group was identified as the median spawning date (Pierce et al. 2007, 2009). We used Mann-Whitney rank sum test to analyze pre-spawning movement distances to spawning tributaries and migration distances up spawning tributaries for Wasson Creek versus other tributaries where tagged fish spawned (Arrastra Creek and Moose Creek). These tests were performed using R software (R Development Core Team 2012), and evaluated at the = 0.05 level of significance.

Water temperature and flows - Mean daily water temperatures and daily stream flows were also measured in Wasson Creek to explore potential relationships with westslope cutthroat trout movements and spawning events, including movements through the experimental (Coanda) diversion structures and stream reaches downstream of the diversions where instream flows were enhanced (Figure 2a and b). Stream flow and temperature measurements were taken between 1 April and 1 September in both 2011 and 2012, and began prior to irrigation use and prior to movements of radioed fish. To measure water temperatures, we used a continuous (50 minute interval) digital thermograph (Onset Computer Corporation, Pocasset, Massachusetts) located at rkm 0.2 on Wasson Creek (Figure 1). To calculate flows, we measured discharge and developed stage-discharge rating tables for staff gauges immediately up- (rkm = 3.7) and downstream (rkm = 4.3) of the two diversions (Figure 1). Estimates of mean daily discharge were then made from weekly staff gauge readings and correlations with daily flows from the USGS streamflow gauge on Nevada Creek (USGS station 1235500).

Results

Telemetry – We tracked 14 adult westslope cutthroat trout to spawning sites in this study by making a total of 374 contacts with an average of 27 contacts (range, 13-37) per fish. All individuals were successfully tracked to spawning tributaries from 24 April to 7 June (Table 2). Thirteen of the 14 fish tested as genetically unaltered westslope cutthroat trout across 8 microsatellites. One fish that entered the West Fork of Arrastra Creek in 2012 tested as 6% introgressed with rainbow trout across the eight loci examined. With the exception of the West Fork fish, these genetic tests support our visual observations of westslope cutthroat trout in this study.

As measured in lower Wasson Creek, water temperatures incrementally increased in the spring during the 2011 to 2012 cutthroat trout pre-spawning migrations. In these years, migrations began between 2 and 13 May during spring runoff. Ten westslope cutthroat trout moved upstream through Nevada Spring Creek and into Wasson Creek,

and four moved down Nevada Creek before ascending the Blackfoot River and moved up into two upper river tributaries (Arrastra and Moose Creek). Over an average of 14 days (range, 3-27), migratory westslope cutthroat trout traveled a median of 14 rkm (range, 7.6-63.1) to their respective spawning site. Westslope cutthroat trout that spawned in Wasson Creek entered the stream at 5-6°C as flows increased and spawned at temperature between 8 and 12°C as measured in lower Wasson Creek (Figure 2a and b). Of these 10 fish, eight spawned in a concentrated area upstream of the diversions (Figure 3).

Fish ID	Capture location	Pre-spawning migration			Tributary spawning			Post-spawning		
		Date migration started	Total rkm	Total days	Tributary	Estimated spawning date	Date exited	Last live location	Last live contact date	Fate
1	Nevada Spring Cr	16-May-11	7.6	16	Wasson Cr	2-Jun	4-Jun	Nevada Spring Cr	19-Jul	Unknown
2	Nevada Cr	15-May-12	10.8	2	Wasson Cr	24-May	6-Jun	Nevada Spring Cr	8-Jun	Unknown
3	Nevada Cr	26-Apr-12	11.3	11	Wasson Cr	17-May	27-May	Nevada Cr	29-May	Heron Predation
4	Nevada Spring Cr	12-May-11	14.5	7	Wasson Cr	28-May	1-Jun	Wasson Cr	27-Jun	Mortality
5	Nevada Cr	10-May-12	11.6	1	Wasson Cr	15-May	22-May	Blackfoot R	27-Aug	Alive
6	Nevada Cr	12-May-11	12.1	10	Wasson Cr	29-May	1-Jun	Wasson Cr	16-Jun	Mortality
7	Nevada Cr	5-May-12	14.6	4	Wasson Cr	15-May	24-May	Nevada Cr	27-Aug	Alive
8	Nevada Cr	7-May-12	13.4	2	Wasson Cr	14-May	10-Jun	Nevada Cr	27-Aug	Alive
9	Nevada Cr	26-Apr-12	12.7	19	Wasson Cr	23-May	27-May	Nevada Cr	29-May	Heron Predation
10	Nevada Spring Cr	11-May-11	16.9	6.5	Wasson Cr	26-May	1-Jun	Wasson Cr	25-Jul	Alive
11	Nevada Cr	13-May-11	63.1	25	Arrastra Cr	7-Jun	10-Jun	Blackfoot R	25-Jul	Alive
12	Nevada Cr	24-Apr-12	44.9	11	West Arrastra Cr	1-May	7-Jul	Blackfoot R	23-Aug	Alive
13	Nevada Cr	24-Apr-11	49.9	27	Moose Cr	30-May	2-Jun	Blackfoot R	12-Jun	Unknown
14	Nevada Cr	9-May-12	53.8	14	Moose Cr	30-May	19-Jul	Blackfoot R	23-Aug	Alive

Table 2. Summary of cutthroat trout spawning migrations for 14 migratory adults. The table includes the duration, dates and distances of spawning events as well as summering locations. These summaries relate to spawning locations on Figure 3.

Spawners spent an average of 18 days (range, 1-74) in spawning tributaries and ascended a median of 3.1 rkm (range, 0.2-6.4) to their spawning sites in low-order streams where they held for an average of 7 days (range 1-16) before returning to the Blackfoot River (*n* = 4) or Nevada Creek (n = 3) (Table 2). Based on the distance between location at the start of migration and spawning sites, fish moved a (median) distance of 14.1 rkm for the total group, and a median of 7.7 rkm (range, 7.6-16.9) for Wasson Creek fish versus 51.8 rkm (range, 44.9-63.1) for upper river spawners. The total migration distances to the mouths of Arrastra and Moose creeks were further than to Wasson Creek (*P* = 0.002). However, Wasson Creek fish spawned higher in their respective spawning stream than fish that spawned in Arrastra and Moose creeks (median, 5.3 versus 1.6 rkm, *P* = 0.02).
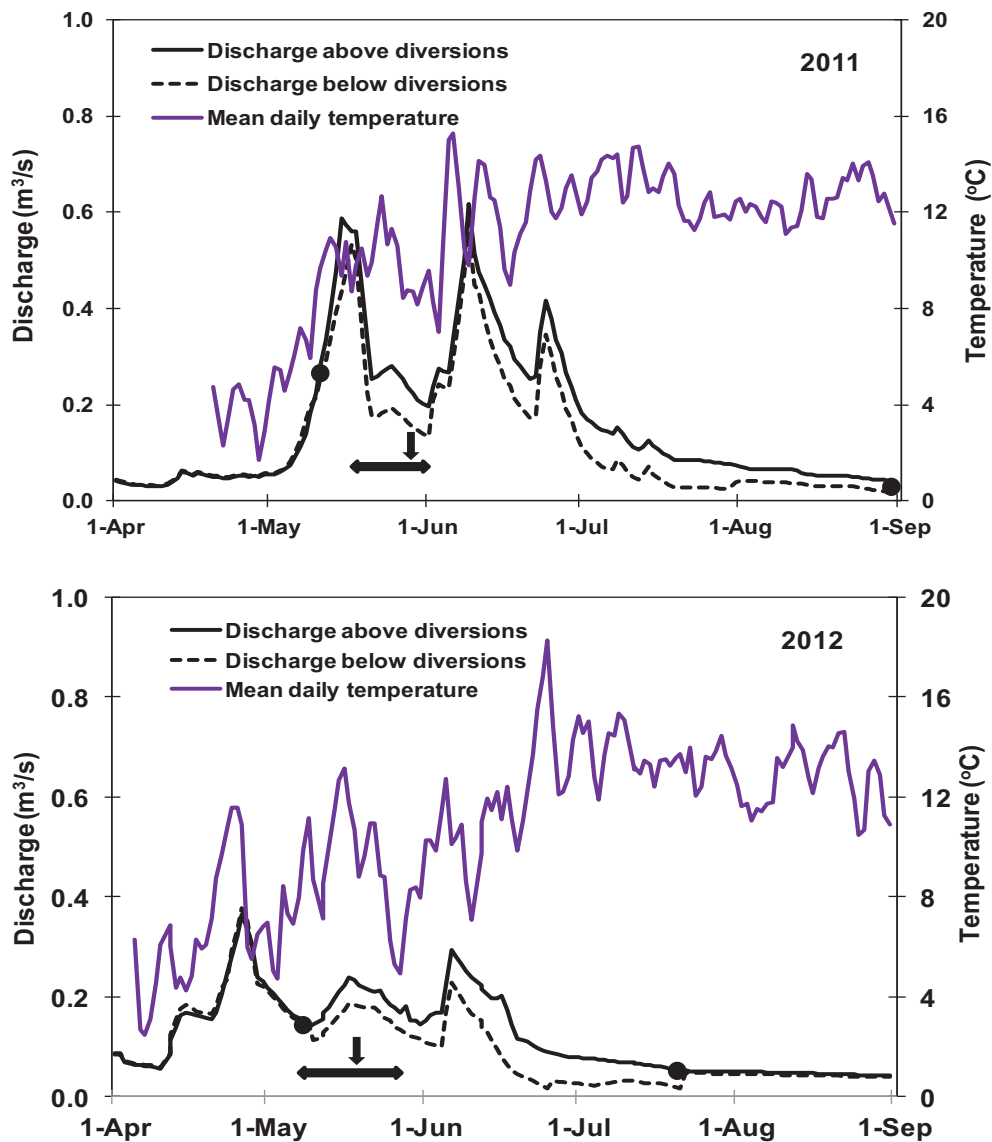
Figure 2a and b. Relationship of migration and spawning to discharge and water temperatures in Wasson Creek for 2011 and 2012. The horizontal (arrowed) bar shows the migration period through the diversions (n=3 in 2011, n=5 in 2012). The vertical arrow represents the peak (median) spawning period for all Wasson Creek fish (n=10). The dark circles show the dates irrigation was turned on and off.

When last contacted (Table 2), two post-spawning Wasson Creek fish died in Wasson Creek (# 4 and 6), two (# 3 and 9) were killed by blue heron (*Ardea herodias*) based on tags traced to a rookery, one (# 10) remained in Wasson Creek, two exited to Nevada Spring Creek (# 1 and 2), two exited to Nevada Creek (# 7 and 8), and one moved into the Blackfoot River 4.3 rkm downstream of the Nevada Creek confluence. After spawning, all Arrastra Creek (*n* = 2) and Moose Creek (n = 2) spawners returned to the Blackfoot River and moved downriver from the confluences of their spawning tributaries distances ranging from 6.0-81.4 rkm when last contacted. The Moose Creek spawner (#14) that showed the longest pre-spawning movement (53.8 rkm) also showed

and longest post-spawning downriver movement (81.4 rkm). We ended the tracking in July when migratory trout exited spawning tributaries and entered summering areas of the larger streams.
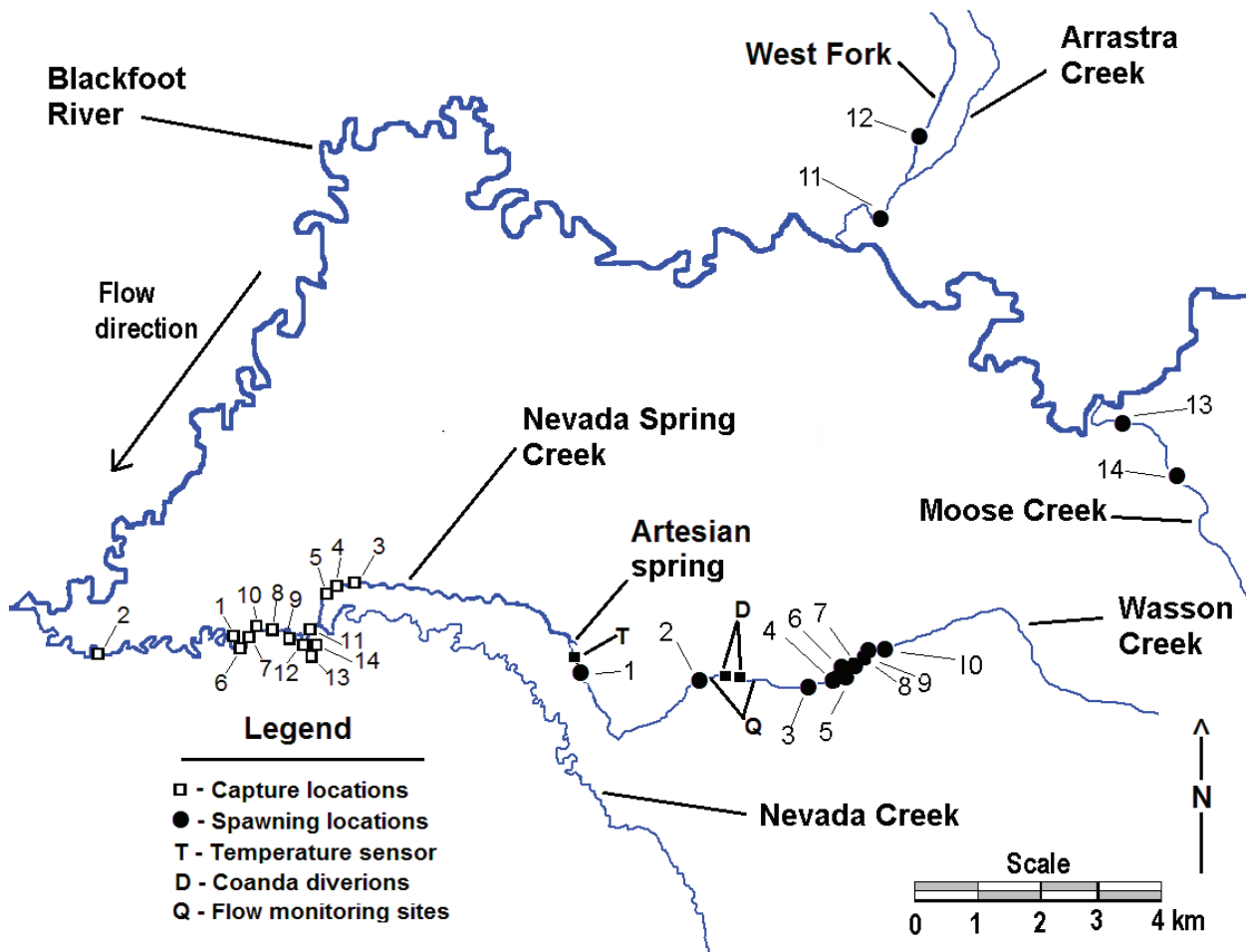
Figure 3.   Capture locations (squares) and spawning locations (black circles) for 14 migratory westslope cutthroat trout.   The numbers for spawning locations relate to summaries of individual fish movements on Table 2.

Migrations at the Coanda diversions - Of the 10 spawners that entered Wasson Creek, eight spawners migrated upstream of the Coanda diversion structures between 10 May and 1 June (Figure 2a and b). Three spawners ascended the Coandas between 21 and 25 May 2011 at flows ranging from 0.25 - 0.28 m^3/s. Five spawners ascended the diversions between 10 and 19 May 2012 at flows ranging from 0.14 - 0.24 m^3/s. The remaining two fish that spawned in Wasson Creek fish spawned in lower Wasson Creek downstream of the Coanda diversions (Figure 3). Of the eight fish that moved over the Coanda fish screens, seven migrated back downstream through the diversion structures without becoming entrained in the ditch, and one fish died after spawning about 2 rkm upstream of the upper diversion. Water was diverted into irrigation ditches during these migration periods, but instream flows were managed to emulate natural flow conditions (Figure 2a

and b). Under these conditions, the Coanda fish screens showed no observed effect on up-or downstream movements of adult fish.

Discussion:

Though human activities are broadly implicated in the loss of native salmonids, few studies evaluate the long-term efficacy of restoration for fisheries response (Bernhart et al 2005, Roni 2005; Balidgo et al. 2008), and very few, if any, published studies document the response of migratory native trout to multi-scale restoration. For this study, we chose a small sample size because we expected only local movements within the Nevada Creek complex. As expected, our small sample of spawners confirmed: 1) the migratory behavior associated with the local expansion of resident cutthroat trout following restoration actions, and 2) the efficacy of experimental Coandas for passing adult migratory westslope cutthroat trout. Interestingly, tagged fish also revealed unexpected large-scale movements to streams outside of the Nevada Creek basin. Though sample sizes were especially small for these spawners, these results were compelling because these individuals link the restoration area with increases of the broader metapopulation (Rieman ad Dunham 2000; Pierce and Podner 2013).

Photo 1a and b. Photo of a Coanda diversion and fish screen on Wasson Creek. The large photo shows two fish screens as well as a sediment sluice gate (middle slot with boards). The smaller photo in the upper right shows an adult westslope cutthroat trout ascending the Coanda (Photo by Jamie Nesbit).

Restoration, migration and spawning - Restoration and habitat connectivity are both crucial to the long-term conservation of migratory salmonids (e.g., Rieman and Dunham 2000; Shrank and Rahel 2004; Petty et al. 2012; *this study*). Compared to resident trout, migratory forms appear to have experienced large and disproportionate reductions in numbers (Gale et al. 2008). In many areas, population reductions have been broadly implicated with instream dams, diversions and dewatering that prevent or restrict the movements of fish (Pierce et al. 2007, 2013; Gale et al. 2008; Roberts and Rahel 2008). Indeed, age-1 and older westslope cutthroat trout in Wasson Creek were abundant immediately upstream of the diversions (i.e., abundance = 22 trout/30m), but absent immediately downstream of the diversions prior to restoration and irrigation upgrades when surveyed in 2003 (Montana Fish, Wildlife and Parks, unpublished data). Following restoration (Table 1), the abundance of age 1 and older cutthroat trout increased from zero to an average 11 fish/30m (range, 4.3-21) downstream of the diversions between 2004 and 20012.

In our study, spawners captured in lower Nevada Creek migrated in some cases long distances (>50 rkm) at high water through a complex range of large and small stream networks and spawned near the peak of the hydrograph in small headwater streams as temperatures increased, before returning to larger waterbodies as flows declined. This behavior conforms to the known spawning life histories of migratory westslope cutthroat trout from the Blackfoot River (Schmetterling 2001; Pierce et al. 2007), and is similar to migratory cutthroat trout behavior in other areas (Brown and Mackay 1995; Rosenfeld et al. 2002; Muhlfeld et al. 2009b).

In this study, ten of 14 spawners ascended upper Wasson Creek after the restoration and installation of the Coandas. These movements were expected given the relatively high abundance of westslope cutthroat trout above the upper diversion prior to restoration (Pierce and Podner 2013), increases in the abundance of westslope cutthroat trout into Nevada Spring Creek following restoration (Pierce et al. 2013), and assignment tests demonstrating genetic similarity between the fish in this study and the population in Wasson Creek (K. Carim, unpublished data). Though irrigation was occurring during these movements, flows were managed to emulate natural conditions, and the Coandas passed all fish with no observed disruption. One adult fish was actually observed successfully ascending the Coanda diversion (Photo 1b). In addition to passing migratory fish at the irrigation diversions, we electrofished the ditches and found no entrained fish, which further indicate the Coanda fish screens are an effective screening device.

Telemetry not only revealed concentrated spawning in the headwaters of Wasson Creek, but also identified long-distance migrations from Nevada Creek to spawning habitats outside of the focal stream network. Though small sample sizes limits our ability to interpret these results, varied movements of fish in this study suggest some recovery of metapopulation function. Specifically, the seasonal use of multiple stocks from distant natal streams using Nevada Creek where none were detected pre-treatment demonstrate the effects of restoration beyond the local population. Conversely, we identified no spawning movements to other tributaries within the Nevada Creek drainage although resident westslope cutthroat trout are distributed widely in the headwaters of nearby streams. This was expected given pervasive human alterations of aquatic habitat in lower stream reaches and very little, if any, habitat connectivity between low-elevation stream and headwater populations (Pierce et al. 2007). In the case of Wasson Creek, spawning was concentrated near the mountain/valley interface upstream of a low-gradient meadow stream, which seems to generally lack the gravel bedforms that migratory westslope

cutthroat trout typically require for spawning (Schmetterling 2000). This concentrated spawning shows the patchy nature of spawning sites common to migratory native trout (Rieman and Dunham 2000) and underscores the importance of small streams for cutthroat trout, as shown in other regions (Rosenfeld et al. 2002)

Following spawning, most cutthroat trout from Wasson Creek returned to Nevada Creek to over-summer. Conversely, spawners from both Arrastra and Moose creeks entered the Blackfoot River, though they were originally captured, and presumably wintered in Nevada Creek. Although this study was not intended to examine overwintering habitat, our findings of migrant fish from outside of the Nevada Creek basin suggest Nevada Creek may provide important habitat for westslope cutthroat trout that spawn and summer elsewhere. The Blackfoot River near the mouth of Nevada Creek is prone severe winter conditions (i.e., super-cooled ($<0^{\circ}$C)) water and anchor ice; Peters and Spoon 1989; Pierce et al. 2012), which can trigger movements of native trout to areas of groundwater upwelling where temperatures are moderated (Cunjack 1996; Jakober et al. 1998; Brown et al. 2011). In the case of Nevada Creek, the artesian spring at the head of this creek flows at a constant annual temperature of 6.7-7.8 $^{\circ}$C, cooling mainstem of Nevada Creek during the summer, while also warming the stream during the winter.

Conclusions

Westslope cutthroat trout conservation west of the Continental Divide involves managing for diverse life histories, including both stream-resident and migratory populations (Schmetterling 2001; Shepard et al. 2005; Fausch et al. 2009). Unlike resident fish that can persist in isolation (Shepard 1997; Cook et al. 2010), the recovery of migratory native trout requires large and highly connected systems. In the case of the upper Blackfoot Basin, stream systems are complex and private lands provide most of the spawning sites, migration corridors, and wintering areas for migratory cutthroat trout, as well as having the most opportunity for meaningful restoration (Pierce et al. 2007, 2013). Here, managing for migratory westslope cutthroat trout involves basin-scale conservation strategies, which integrate site-specific techniques that provide for the habitat and benefit the life history diversity of individual stocks. In the Nevada Spring Creek complex, reach-scale restoration has improved the general habitat necessary for migratory salmonids, while Coanda fish screens provide the mechanism to improve habitat connectivity in areas of suitable habitat by passing fish and reducing losses of fish to irrigations ditches even during active irrigation. This study shows that the integration of restoration techniques can not only improve specific habitat needed for migratory trout at a local scale, but can also promote the recovery of migratory fish across larger stream networks.

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The relative roles of Natural Channel Design and coarse woody debris in the recovery of wild trout populations in a reconstructed spring creek, Blackfoot Basin, Montana

Draft Report

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Abstract. Kleinschmidt Creek, a low-gradient groundwater-dominated stream in the Blackfoot Basin, Montana, was fully reconstructed using natural channel restoration design principles. Following reconstruction, densities and biomass of wild trout were monitored for 11-years in order to test the fisheries response to the new channel and the variable use of instream wood. Stream restoration increased channel length by 36%, decreased the width/depth ratio of the channel and reduced the wetted surface area by 56%. Following restoration, the density and biomass of age 1 and older wild trout (primarily Brown Trout *Salmo trutta*) showed significant continuous (linear) increases over an 11-year post-treatment monitoring period at both treatment sites. Compared to the reconstructed with minimal wood, the section with more concentrated wood showed a rapid 3-4 year initial increase in both density and biomass. After a four-year recovery period, both densities and biomass began to increase in the section with minimal wood, and ultimately, the relative rates of population increase differed little between sections. Though trout densities were consistently higher in the section with wood during the entire 11 year post-treatment monitoring period, the benefits of instream wood were more strongly expressed at the early phase of habitat recovery in this small vegetative-controlled stream.

INTRODUCTION

To reverse human-induced degradation of river ecosystems, aquatic habitat restoration is expanding across the American West where riverscapes have been widely modified to the detriment of coldwater salmonids (Nehlsen et al. 1991; Behnke 1992; Thurow et al. 1997). As restoration methods evolve, practitioners are increasingly attempting to develop and apply more natural restoration techniques in order to reestablish the physical and ecological integrity of streams in order to recover sensitive fisheries (Nagel 2007; Baldigo et al. 2008; Ernst et al. 2010; Pierce et al. 2012). Despite broad increases in restoration, very few projects document the use of natural channel design (Nagle 2007; *but see* Klein et al. 2007; Ernst et al. 2010) or monitor biological effectiveness beyond 5 years post-treatment (Ernst et al. 2010; Roni et al. 2008). As a result, the ability of restoration practitioners to develop informed, effective and more natural stream restoration techniques are clearly limited by a basic lack of published applied field studies (Bernhardt et al. 2005; Baldigo et al. 2008; Roni et al. 2008).

Though few studies document the application and/or biological effects of natural channel design, the use of instream wood as a habitat improvement technique has been successfully applied for decades (e.g., Tarzwell 1936; Hunt 1976; Binns 2004; Roni et al. 2008). The loss of instream wood is often the result of deforestation, excessive grazing, intentional clearing, road construction and other streamside development pressures (Meehan 1991; Gregory et al. 2003), all of which can lead to the simplification and loss of aquatic habitat (Schmetterling and Pierce 1999; Rosenfeld and Huato 2003), and reduce the overall ecological integrity of streams (Bilby et al. 2003). Conversely, the input of large wood either by passive or active means can influence channel morphology by controlling gradient, increasing pools and diversifying habitat necessary for spawning and rearing (Hunt 1993; Schmetterling and Pierce 1999). Woody material not only provides instream cover for fish (Roni et al. 2008; Whiteway et al. 2010; Pierce et al. 2012), but also enhances habitat diversity for aquatic communities (Benke and Wallace 2003; Dolloff and Warren Jr. 2003).

Despite the biological importance of instream wood, very few long-term (>5 years post-treatment) studies examine the response of salmonids to the placement of instream wood (Roni et al. 2005, 2008) with some exceptions on small-scale treatments on confined, stable channels with moderate gradient (Roper et al. 1998; White et al. 2011; *but see* Hunt 1976; Baldigo et al. 2008). Conversely, the long-term efficacy of wood placement in unconfined vegetated channels with lateral habitat-forming processes are less certain (Frissell and Nawa 1992; Schmetterling and Pierce 1999) and may even be unwarranted as a habitat improvement technique where meandering channel processes and rhizomatous meadow vegetation more strongly influence channel morphology (Rosgen 1996).

In the Blackfoot Basin of Western Montana, we compiled pre-and post restoration channel form data and concentrations of instream wood in Kleinschmidt Creek, a spring creek fully reconstructed in 2001. We also monitored fish populations before and after restoration over a 15-year period (1998 through 2012) in order to evaluate channel renaturalization and the placement of coarse instream wood as a restoration technique for improving wild trout. In this study, we describe restoration-induced changes to channel morphology and specifically examine the post-treatment changes in salmonid density and biomass compared to the use of added large instream wood within this reconstructed groundwater-dominated stream. Our broader aim is to gain a better understanding of

trout response in vegetative-controlled channels and promote more natural restoration techniques for degraded spring creeks in the river valleys of western North America.

STUDY AREA

Kleinschmidt Creek is a groundwater-dominated tributary to the lower North Fork of the Blackfoot River, located on the floor of the Blackfoot River Valley in west-central Montana near the town of Ovando (Figure 1). Kleinschmidt Creek originates along the southern margin of a large glacial outwash plain, and flows for about 3.2 km within a terraced alluvium before entering North Fork of the Blackfoot River at river km 9.8. Discharge in Kleinschmidt Creek ranges from 0.26 m^3/s during winter and spring to a high of about to 0.42 m^3/s during mid-summer months (Pierce et al. 2006). Streamside vegetation consists of wetland gramminoids (*Carex/Juncus/Phararis* spp.) as well as low, but increasing, riparian shrub (*Salix/Alnus* spp.) cover. Kleinschmidt Creek supports a mixed community of salmonids although brown trout, *Salmon trutta* comprise >90% of the salmonid community. Other salmonids present in the order of decreasing density include brook trout *Salvelinus fontinalis*, native westslope cutthroat trout *Oncorhynchus clarkii lewisi*, native bull trout *S. confluentus* and rainbow trout *O. mykiss*.

Prior to restoration, Kleinschmidt Creek was subject to heavy livestock grazing, channelization from highway construction, instream dams and undersized culverts (Pierce 1991; Marler 1998; Land and Water Consulting 1999). These conditions led to a wide, shallow and straightened channel with elevated water temperature and sediment and a reduction in instream habitat complexity, degraded wetlands and the complete loss of woody riparian vegetation (Photo 1). In 1994, the lower 0.5 km of Kleinschmidt Creek was reconstructed to more natural form. Then in 2001, the remaining upper 2.7 km (i. e., spring-fed) of channel was restored using natural channel design techniques in order to restore natural form and function consistent with a vegetative-controlled deep and narrow channel (Table 1, Photo 2). Channel restoration involved reconnecting the channel with its floodplain, constructing a low width/depth ratio, meandering (E4) stream type (Rosgen, 1996, 1997). As described by Rosgen (1996), the E4 channel-type constructed typically exhibit low channel gradients (<2%) very low width/depth ratios (<12), high sinuosity (>1.5) and gravel-based pool/riffle bedforms in channels that are geomorphically stable in the absence of vegetative disturbance.

To diversify pool habitat and channel bedforms and compensate for the anthropogenic loss of instream wood, a total of 208 woody stems were anchored into the bed and outer streambanks of 122 of 138 pools (mean = 1.8 stems/pool; range 1-9) of the new channel. This increased the amount of instream wood from 0.03 stems/100m pre-treatment to 6.4 stems/100m post-treatment over the length of the new channel between stream km 0.5 and 3.2. Woody stems averaged 22cm in mean diameter (range 10-41cm) and 2.9m in mean length (range 0.6-5.2m).

To test the fisheries response to the use of instream wood, sixteen pool/riffle sequences between stream km 0.7 and 0.8 were left with minimal wood (Figure 1, Table 2). Two 154 m fish population survey sites were established (Figure 1): one was located in an upstream site (km-1.3) with the presence of large wood and the other was located in the downstream site (km 0.8) with minimal wood (Table 1, Figure 2). Following this work, riparian shrubs were planted, livestock were fenced from the riparian corridor and a perpetual conservation easement was placed on a majority of the stream corridor to protect the ecological integrity the riparian area and new channel (MDT 2001). Finally,

the majority of the streams discharge (0.25 m^3/s) was dedicated to the maintenance of instream flows (DNRC 2011).
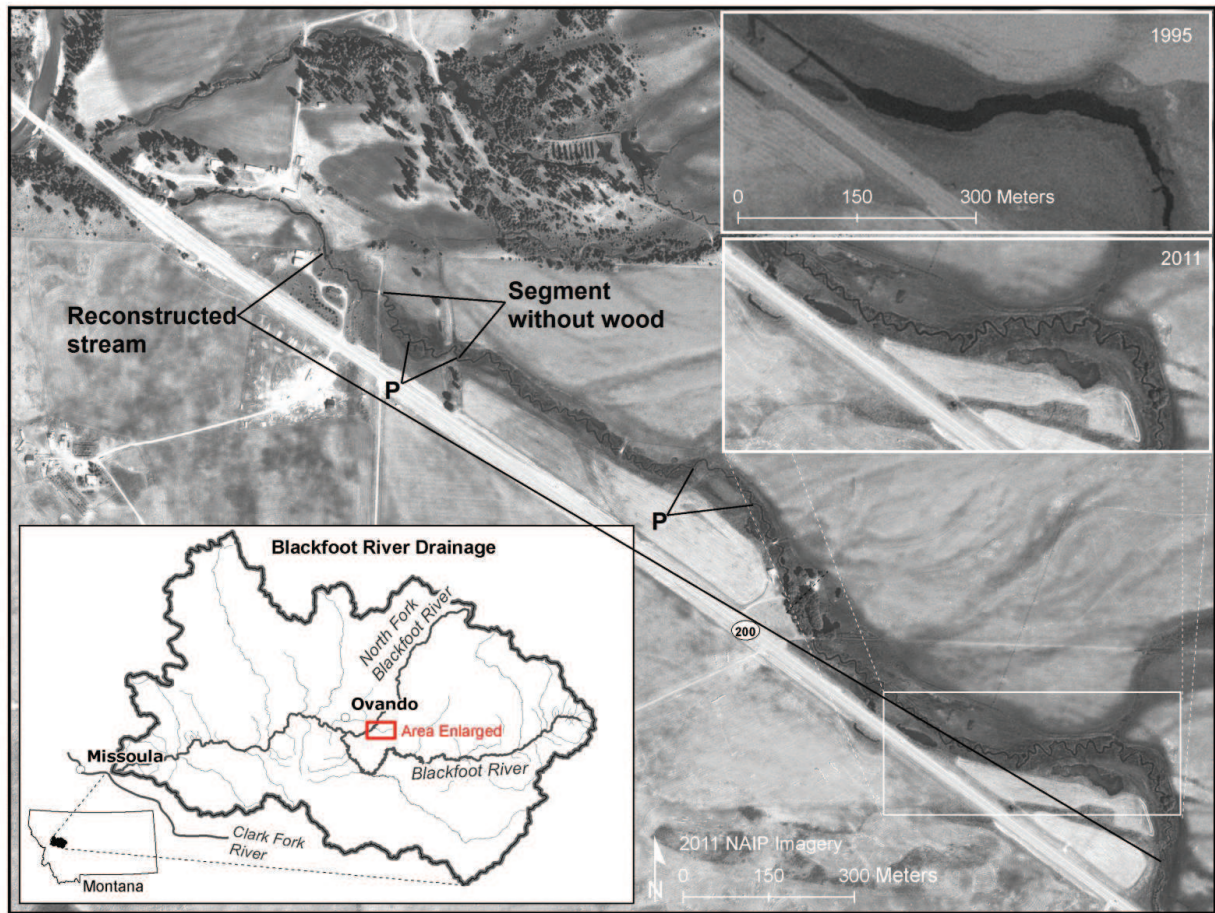
Figure 1. Study area showing the Blackfoot River basin and the Kleinschmidt Creek project area: The basin map also shows the location of the project area. The project aerial shows the fish population (P) monitoring sites and delineates the upper project area, including the stream segment with minimal instream wood. The enlarged area of upper Kleinschmidt Creek show pre- (1996-top) and post-treatment (2011-bottom) aerials of the upper-most project area.

	Channel length (km)	Sinuosity	Mean wetted-width (m)	Wetted surface area (h)	Number of pools/100m	Mean maximum pool depth (m)	Percent fine sediment (<2mm)
Before	1.9	1.06	9.5	1.81	0.58	0.67	30*
After	2.6	1.44	3.0	0.78	4.5	1.01	21

Table 1. Summary of channel changes before and after restoration. The * denotes data from Neudecker et al. 2012.

Photos 1 and 2. Kleinschmidt Creek before (top) and after (bottom) restoration in 2001. The top photo shows straightened and widened section of channel with an example of a rock dam that trapped sediment. The bottom photo was taken immediately after channel reconstruction in the late summer of 2001.
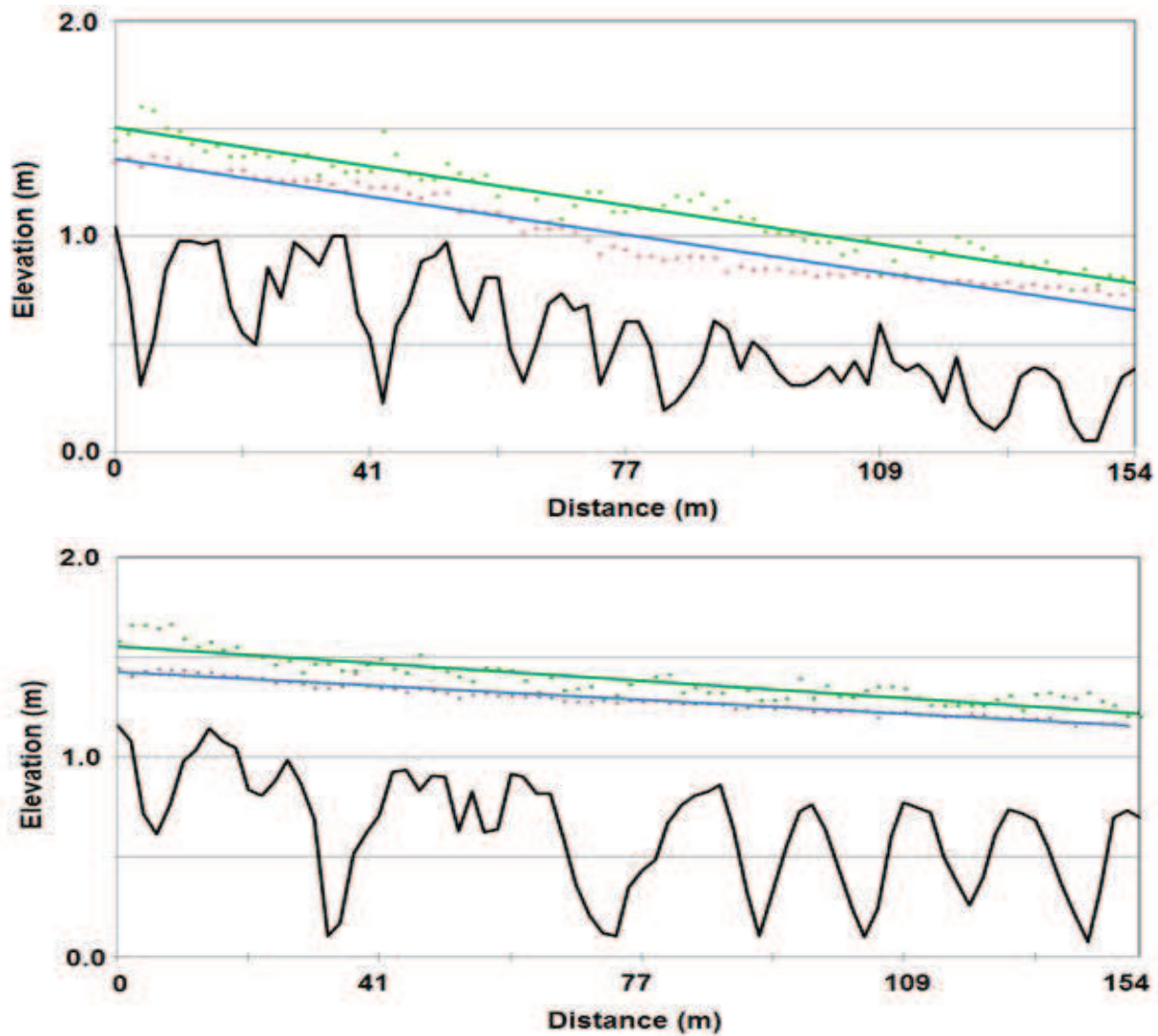
Figure 2. Longitudinal profiles of the fish population monitoring sites with floodplain elevations and trend line (green lines and symbols), water surface elevations and trend line (blue line and symbols) and thalweg profile (black line). The top graph shows the section with wood, and the bottom shows the section without minimal instream wood. Profiles relate to channel metrics and instream habitat structure in Table 2

	Pools		Riffles/glides		Instream habitat structure		% shrub cover
	Wetted depth mean (range)	Wetted width mean (range)	Wetted depth mean (range)	Wetted width mean (range)	# Woody stems	# Rock structures	
W/ wood	0.8(0.7-1.1)	3.5(3.1-4.0)	0.3(0.2-0.4)	3.1(2.7-3.5)	28	2	9
W/o wood	1.1(0.8-1.2)	3.4(3.2-3.8)	0.4(0.3-0.5)	2.8(2.4-3.6)	2	1	<1

Table 2. Summary of pool and riffle width and depth measurements, number instream habitat structures and percent shrub cover for the two fish population survey sites. Surveys were completed in 2013.

METHODS

Fisheries response to instream wood.- Prior to the reconstruction of the upper channel, 3-years of pre-calibration fish population data (1998 - 2000) was obtained in the degraded reach at km 0.8 for baseline data for the two post-treatment monitoring sites (Figure 1). Following channel reconstruction and placement of instream wood, annual fish population surveys (post-treatment) were obtained between 2002 and 2012 at km 0.8 (i. e., minimal wood section). Post-treatment fish populations were concurrently monitored at km 1.3 (i.e., concentrated wood section). Fish populations were not surveyed in 2011 due to sampling difficulties related to high water.

Trout density and biomass was determined using backpack electrofishing depletion estimates (Van Dervender and Platts 1989). Fish population estimates were repeated at both monitoring sites during post-treatment monitoring and had a mean capture probability of 0.7 (range, 0.5-1.0). All trout captured were measured for total length (mm) and weight (g). Because the weights of trout were measured using both mechanical and digital scales during the 1998-2012 monitoring period, we standardized the weight (g) data using length/weight data from a digital scale (AND model HL-3000WP) for all trout collected between 2009 and 2012. To standardize weights, we first developed weight/length regressions for the both brown trout (y = 3.0848x-5.1205, R^2 = 0.979) and brook trout (y = 3.2449x-5.479, R^2 = 0.971), the two prevalent species, which respectively comprised 92% and 6% of all age 1+ fish captured in this study. Subsequent weights were applied to total lengths of trout in the 1998 - 2008 dataset. Following Pierce et al. (2013), we removed age-0 fish from the dataset using length-frequency histograms, and used age ≥ 1 trout to estimate total trout densities and biomass for all trout in each of the two treatment reaches. Since brown trout comprised >90% of the salmonid community over the entire monitoring period, all age 1 and older trout were combined to generate estimates of total trout density (trout/m^2) and biomass (kg/m) with the recognition that changes in the brown trout population were driving community-level trends in density and biomass.

Analyses of fisheries response – To assess relative rates of change in total trout density and biomass between the segment with wood treatment (11 year post-restoration timeframe) and the segment without the wood (15 year timeframe including 3 years pre-restoration), an ANCOVA analysis with time as a continuous covariate and wood treatment as a fixed factor for both fish biomass and density was used because of the linear nature of the dataset. A separate t-test to explicitly compare only post-restoration mean density and biomass between the wood addition and no wood sites was also utilized. We calculated separate effect sizes for both density and biomass of fish as the natural log of the relative response ratio (*ln* R) of each variable in the wood addition reach to that in the control reach for each year of record. Analyses were performed at the α = 0.05 level of significance using SAS Analytics software.

RESULTS

Response of wild trout to restoration — Both density and biomass of age-1 and older salmonids increased as a function of time (effect of restoration period: density F = 23.54, p < 0.0001, Figure 3A; biomass F = 12.71, p = 0.0021, Figure 3B). Mean density and mean biomass of fish were significantly higher in the wood addition compared to the minimal wood addition reach (effect of wood treatment: density, F = 14.93, p = 0.001;

biomass, F = 9.21, p = 0.007), a pattern corroborated by assessment only during the 11 years of post-restoration data (density, t = 2.74, p = 0.007; biomass, t = 2.00, p = 0.03). Effect sizes over the entire 15 year monitoring period illustrate that fish population measures were substantially elevated in the wood addition reach for roughly the first 3-4 years following restoration; however, thereafter difference in both fish density (Figure 4A) or biomass (Figure 4B) were greatly reduced between the two restoration treatments. Two years following restoration, fish densities were roughly 11× greater and fish biomass was roughly 6 × greater in the wood addition reach compared to the no wood reach in other post-treatment years. Following this rapid increase in density and biomass (i. e., years 1-4 post treatment), effect ratios between 5 and 11 years averaged 0.55 greater densities and 0.14 greater biomass in the wooded section.
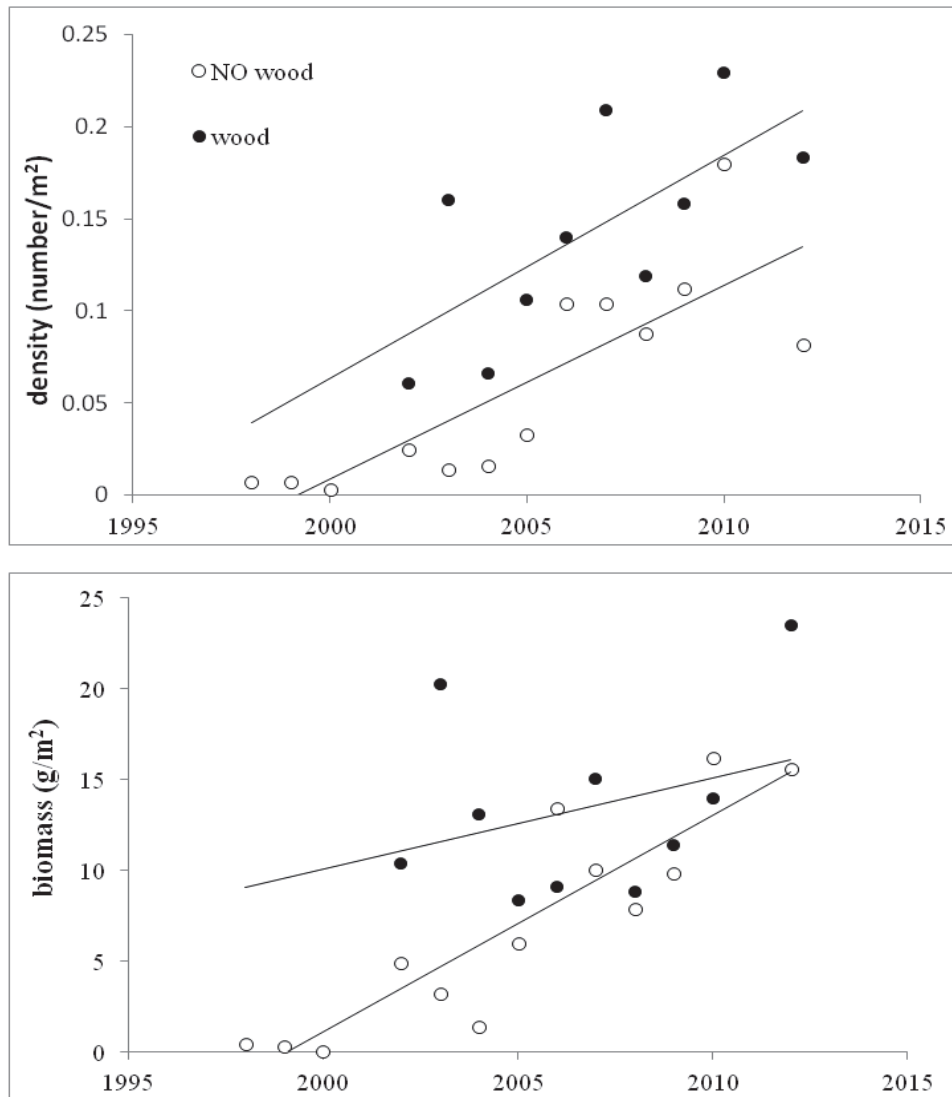
Figure 3. Response of fish (A) density (ln) (B) biomass (ln) over time in wood addition and no wood addition reaches.
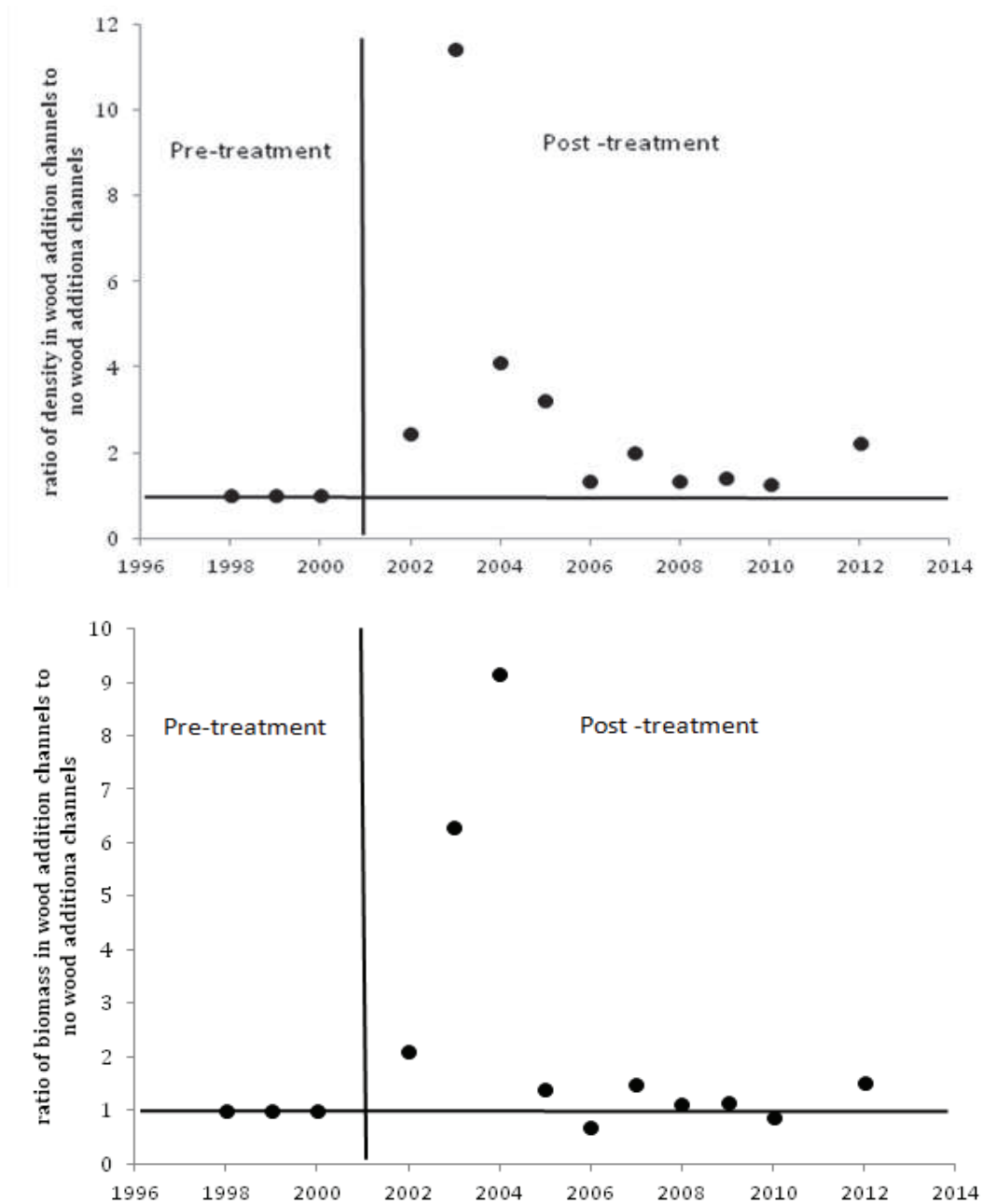
Figure 4A and B. Change in effect size ratios for wood addition against no wood addition over the entire assessment period. Graph A (top) is density effect size over time, and graph B (bottom) is biomass effect size over time. The vertical line separate pre- and post-treatment monitoring periods. The horizontal line show 1:1 ratio indicating no effect.

DISCUSSION

The reconstruction of the entrenched Kleinschmidt Creek from a degraded, over-widened channel (Photo 1) to a relatively deep and narrow meandering channel (E4 stream type, photo 2) over 2.7 km preceded significant increases in the density and biomass of wild trout at both treatment sites. The segment with wood showed sharp initial (1-4 year) post-treatment increases in both density and biomass versus a delayed response in the woodless section. Four years after treatment, densities and biomass

increased in the section with minimal wood, and eventually, delayed trends in density and biomass showed similar positive trajectories.

Trout responses and the use of instream wood - Instream wood has been widely used for habitat improvement, hydraulic function and streambank stability for decades (Binns 2003; Gregory et al. 2003). In our study area, the placement of instream wood in Kleinschmidt was used primarily for habitat improvement and diversity versus bank revetment or hydraulic function based on the low frequency of high flow (flood) events. Though the placement of instream wood clearly can increase trout carrying-capacity where bank-cover, pool quality and/or habitat complexity may be limiting (Hunt 1976; Binns 2003; White et al. 2011), the application and efficacy of instream wood as a habitat improvement technique varies widely depending geomorphic conditions (e.g., channel-types, Rosgen 1996; Schmetterling and Pierce 1999; White et al. 2011). Yet, few fisheries studies link wood to geomorphic gradients or explore long-term biological relationships of adding instream wood into deep/narrow meandering meadow streams.

In this study area, wood was biologically most beneficial during the initial (four year) post-recovery period (Figure 4); thereafter, increases in both densities and biomass at both treatment sites showed similar trajectories (Figure 3). This short-term response in areas of wood may be due to movements to pools with complex woody structure (e.g., Hunt 1976; Gowan and Fausch 1996; Dolloff and Warren Jr. 2003), which especially applies to brown trout because of their preference for streambanks with abundant cover and dim light (Behnke 2002). Whereas, long-term trends likely relate to successful spawning (i. e., redds and 0 brown and brook trout, R. Pierce, personal observation) and population growth. Though long-term trajectories were similar after 4 years recovery period, densities remained elevated in segment with wood as revealed by the response ratios between 5 and 11 years post-treatment (Figures 4A and B), which showed 55 % higher densities in the section with wood (i.e., mean annual density = 0.17 fish/m^2 for wooded segment versus 0.11 fish/m^2 for the segment with minimal wood) but only 14% higher biomass (i. e., mean annual biomass = 13.9 g/m^2 for wooded segment versus 12.2g/m^2 for the segment with minimal wood). These differences reflect a higher proportion of smaller fish (e.g., average total length = 168mm, average weight = 79g) in the section with more wood, versus larger fish (e.g., average total length = 195mm, average weight = 122g) in the section with minimal wood. Interestingly, all age-1 and older rainbow, cutthroat and bull trout sampled in this study (*n* = 10) were also sampled post-treatment in the wood section where physical habitat is more diverse.

Rosgen (1996) suggests the placement of wood within E-types channels provide little, if any, habitat benefit given the low width/depth (W/D) ratios of the stable channel form and increased bank cover typical of E-type channels where W/D ratios range from 2-12 (Rosgen 1996). To further examine the W/D relationships of Kleinschmidt Creek, we measured W/D ratios following Rosgen (1996) methods (i.e., bankfull channel width/mean area of riffle cross-section) in both fish population monitoring sites and found ratios within the upper-range (i. e., 8.7 - 9.7) of E-channel type classification (i.e., <12). Though broader trends seem to support the assertion that wood is a minor habitat feature in E-type channels, elevated densities and a more diversity within the trout community indicate wood to be an effective fish habitat improvement technique at the higher range of the E-channel classification.

Though the short-term trends in this study indicated the value of added woody structure was mostly beneficial during initial recovery period (Figure 4), long-term linear

trends indicate both density and biomass of wild trout have yet to fully reach equilibrium at either treatment (Figure 3). These long-term trends contrast with other studies that report about 5-years is required for wild trout to reach equilibrium following habitat manipulation (Hunt 1976; Roni et al 2008; Whiteway et al. 2011). Though the initial trout response in the wooded section is generally consistent with this recovery period, long-term (> 10 years) linear increases in both study sections show that response trends can span a decade or more (Kondolf and Micheli 1996; Pierce et al. 2013). This may be especially relevant in fully reconstructed streams where recovery of biotic communities, including riparian vegetation, occur over many years before upper trophic aquatic species (e.g., age 1 and older wild trout) reach equilibrium with fully restored streams. From our field observations, it appears that wild trout in the new channel continue to increase as changes in vegetative shrub-cover, streambanks, channel bedforms and the presence of instream wood continue to slowly improve and diversify aquatic habitat.

CONCLUSIONS

Channel reconstruction can not only help restore the form and function of more natural channels, but also significantly improve habitat for wild trout as shown by the broad positive response trends in this study. In the case of Kleinschmidt Creek, wild trout increased through an 11-year post-treatment monitoring period in reaches with and without the presence of instream wood. Although the section with wood supported a more rapid initial response, long-term rates of increase for density and biomass were similar between treatment sites. These findings suggest the instream placement wood in newly restored groundwater-dominated streams provides mostly short-term fisheries increases in deep narrow channels where riparian vegetation and lateral stream processes form the primary habitat features necessary for wild trout recovery. Though long-term response trends were broadly similar in both treatments, the segment with wood showed 1) more rapid response, 2) higher densities over the entire post-treatment monitoring period, 3) higher proportion of smaller fish, and 4) higher species diversity post-treatment. Considered together, these differences demonstrate that wood can benefit wild trout in reconstructed streams where width/depth ratios fall within the upper range of the E-type channel. Management implications associated with these use of wood relate to the potential for added recruitment of juvenile fish and/or managing for mixed-species assemblages as well as ascertaining recovery rates for salmonids when planning restoration actions in streams similar to Kleinschmidt Creek.

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Instream habitat restoration and water temperature reduction in a whirling disease positive spring creek in the Blackfoot Basin, Montana

Draft Report

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Abstract. Anthropogenic warming of coldwater streams and the presence of exotic diseases such as whirling disease are both contemporary threats to coldwater salmonids. We examined water temperature reduction over a 15-year pre- and post-restoration period and the severity of *Myxobolus cerebralis* infection over a 7-year pre- and post-restoration period in Kleinschmidt Creek, a fully reconstructed spring creek in the Blackfoot Basin, Montana. Stream restoration increased channel length by 36%, but narrowing of the channel reduced the wetted surface area by 56%. Following channel renaturalization, average maximum daily summer water temperatures decreased from 15°C to 12.5^{o}C and the range of summer temperatures narrowed. Despite changes in channel morphology and reductions in summer water temperature, the severity of *M. cerebralis* infection remained high at three years post-treatment as measured with field exposures of age 0 hatchery rainbow trout *Oncorhynchus mykiss* used as surrogates for wild trout. The study shows channel renaturalization in groundwater-dominated systems can reduce anthropogenic warming at low-elevations to levels more suitable to native trout. However because of continuous high *M. cerebralis* infections associated with groundwater, the restoration of Kleinschmidt Creek will likely favor brown *Salmo trutta* trout given their innate resistance to the *M. cerebralis* and the higher relative susceptibility for other salmonids.

INTRODUCTION

Degradation of salmonid habitat historically involved physical alterations of streams and rivers from land use activities such as channel degradation and overgrazing (e.g., Meehan 1991, Behnke 1992, Thurow et al. 1997; Pierce et al. 2013). However, the additive stressors of anthropogenic warming (e.g., climate change and riparian degradation) and spread of exotic organisms (e.g., *Myxobolus cerebralis*, the cause of salmonid whirling disease) have not only elevated the overall need for aquatic restoration, but also the need to refine and implement specific restoration activities imposed by these conditions (Rieman et al. 2007; Williams et al. 2009, Pierce et al. 2009, 2012; Hansen and Budy 2011). Despite the pressing need for applied studies of this nature, few, if any, published long-term field investigations link restoration to water temperature reduction (Williams et al. 2007; Pool and Berman 2001; Pierce et al. 2012), or test the ability of restoration to reduce *M. cerebralis* infection (*but see* Hansen and Budy 2011; Thompson 2011).

Because ambient climate (i.e., mean annual temperature) regulates groundwater temperatures (Meisner et al. 1988), groundwater-dominated streams (e.g., spring creeks) temperatures are more constant and less affected by seasonal, elevational and weather conditions that influence basin-fed streams. Groundwater-induced streams are thereby cooler during summer months, especially near the sources of groundwater inflows (Meisner et al. 1988; Cassie 2006). As an example of this, temperatures at the source of an artesian spring creek on the floor of the Blackfoot Valley remain at a near constant 7-8°C range of annual temperatures versus <0 to >23°C in nearby waters (Pierce et al. 2002, 2012, 2013). Given the moderating effects of groundwater, stream improvements that renaturalize channels and reduce water temperatures during the heat of summer may prove increasingly important given projections of climate warming, especially in low-elevation streams (Meisner et al. 1988; Rieman et al. 2007; Williams et al. 2009).

Likewise, the known potential of restoration to moderate the effects *M. cerebralis* are limited by localized and/or short-term nature of field studies (Hansen and Budy 2011; Thompson 2011). Habitat conditions favorable for the proliferation of *M. cerebralis* and its oligochaete host (*Tubifex tubifex)* generally include: 1) high water temperature (MacConnell and Vincent 2002; Hansen and Budy 2011); 2) fine sediment (Anlauf and Moffitt 2008; McGinnis and Kerans 2013); and 3) nutrient concentrations (Kaeser et al. 2006); all of which can be elevated with the anthropogenic degradation of streams (Zendt and Bergersen 2000; Hansen and Budy 2011; McGinnis and Kerans 2013). As a result, the potential of restoration to offset whirling disease requires reduction of one or more of these conditions, as well as recognition that the effects of these conditions can vary greatly between basin- (Anlauf and Moffitt 2008; Hansen and Budy 2011; McGinnis and Kerans 2013) and ground-fed environments (Burkhart and Hubert 2005; Neudecker et al. 2012; Pierce et al 2013).

Myxobolus cerebralis has a complex, two-host life cycle involving the aquatic oligochaete worm *T. tubifex*, and most salmonids, which include trout, whitefish and salmon (Bartholomew and Wilson 2002). Susceptibility to the pathogen depends on species (Hendrick et al. 1999; MacConnell and Vincent 2002; Vincent 2002), fish age and size (Ryce et al. 2005), and parasite dose at time of exposure (Vincent 2002). Infectious conditions often vary by season (Vincent 2000; Downing et al. 2002, De La Hoz Franco and Budy 2004; Neudecker et al. 2012) and typically peak in rivers during summer and autumn (MacConnell and Vincent 2002; De La Hoz Franco and Budy 2004;

Pierce et al. 2012) at temperatures conducive (10-15°C) to the release of TAMS (El-Matbouli et al. 1999; Kerans et al. 2005). However, recent studies show *M. cerebralis* infection can be continuous across seasons and can occur at much lower temperatures with the influence of more thermally stable ground-water (Pierce et al. 2012; Neudecker et al. 2012). For example, high infection rates in age 0 mountain whitefish *Prosopium williamsoni* were documented in April in the groundwater-induced upper Blackfoot River at mean temperatures of <5°C versus no concurrent infection in the basin-fed lower river (Pierce et al. 2012). Likewise, a recent assessment of Montana spring creeks, revealed a temporal pattern of high *M. cerebralis* infections in the autumn, winter and spring at temperatures as low as 4.5°C (Neudecker et al. 2012).

In the Blackfoot Basin of Western Montana, we monitored water temperatures associated with the restoration of Kleinschmidt Creek over a 15-year pre-and post-restoration period (1998-2012), and monitored the severity of *M. cerebralis* infection during a 7-year pre-and post-restoration period (1998-2004). This study expands on a prior study that describes changes in channel morphology and examines the variable use of instream wood (habitat structure) for increasing wild trout density and biomass in Kleinschmidt Creek (*This report*). In this study, we specifically examine summer temperature reduction before and after the reconstruction of Kleinschmidt Creek and the influence of restoration on the severity of *M. cerebralis* infection. Our broader aim of this study is to better identify the potential of restoration for water temperature reduction in low-elevation ground-water dominated streams to levels suitable to native trout and to explore whether channel renaturalization and temperature reduction in groundwater-dominated can mediate *M. cerebralis* infection.

STUDY AREA

Kleinschmidt Spring Creek, a spring creek tributary to the lower North Fork of the Blackfoot River, is located on the floor of the Blackfoot River Valley in west-central Montana (Figure 1). Discharge in Kleinschmidt Creek ranges from 0.26 m^3/s during winter and spring to a high of about to 0.42 m^3/s during mid-summer months (Pierce et al. 2006). Although Kleinschmidt Creek receives some basin runoff upstream of stream km 3.2, about 90% of summer stream flows are generated by groundwater inflows from its alluvial aquifer, most of which surface between stream km 1.6 and 3.2 (Pierce et al. 2002, 2006). To examine water temperature reduction, we also monitored water temperature in the North Fork of the Blackfoot River at USGS gauge station #12338300 as a control in this study (Figure 1). Although the discharge of the North Fork of the Blackfoot River is much large larger, like Kleinschmidt Creek, the lower 12 km of North Fork receives >80% of its base-flow from groundwater inflows (DNRC and USGS unpublished data), including several spring creeks between stream km 8 and 10.

To reestablish natural features of a relatively deep and narrow channel (Table 1), the reconstruction of Kleinschmidt Creek was completed in the autumn of 2001. Channel renaturalization increase channel length by 36% but reduced the wetted surface area of the channel by 56%.

Kleinschmidt Creek supports a mixed community of salmonids although brown trout *Salmo trutta* comprise >90% of the salmonid community, and enters the North Fork in a reach classified as critical habitat for the recovery of bull trout (USFWS 2010). Other salmonids present in the order of decreasing density include brook trout *Salvelinus fontinalis*, native westslope cutthroat trout *O. clarkii lewisi*, native bull trout *S. confluentus* and rainbow trout *O. mykiss*. Prior to restoration, the exotic parasite *M. cerebralis*, the cause of salmonid whirling disease was already present at high infection levels in Kleinschmidt Creek in 1998 (Neudecker et al. 2012). When channel reconstruction was completed in 2001, infections in Kleinschmidt were the highest among eight tested spring creeks in Western Montana (Neudecker et al. 2012). Of the salmonids present in Kleinschmidt Creek, rainbow trout are highly susceptible to whirling disease (Vincent and MacConnell 2002). Brook trout, westslope cutthroat trout and bull trout possess intermediate (i. e., partial) resistance to whirling disease (Vincent and MacConnell 2002). Whereas, nonnative brown trout are naturally much more resistant to the parasite given their coevolution on the Eurasian continent (Bartholomew and Reno 2002).
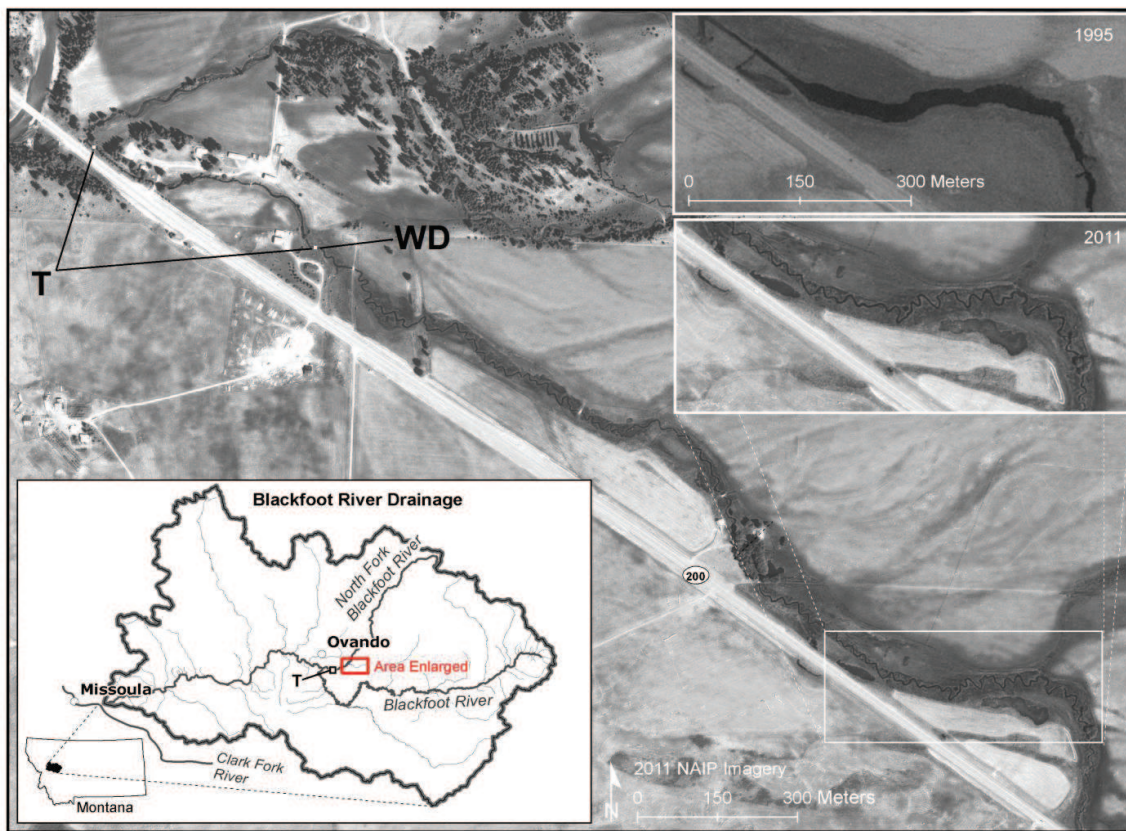
Figure 1. Study area showing the Blackfoot River drainage and the Kleinschmidt Creek project area: The basin map also shows the location of the water temperature control site (T at USGS station) on the North Fork of the Blackfoot River. The project aerial shows the temperature (T) and whirling disease (WD) monitoring sites on Kleinschmidt Creek. The enlarged area of upper Kleinschmidt Creek show pre- (1996-top) and post-treatment (2011-bottom) aerials of the upper-most project area.

	Channel length (km)	Sinuosity	Mean wetted-width (m)	Wetted surface area (h)	Number of pools/100m	Mean maximum pool depth (m)	# woody stems/100m	Percent fine sediment (<2mm)
Before	1.9	1.1	9.5	1.81	0.58	0.67	0.03	30*
After	2.6	1.4	3	0.78	4.5	1.01	6.4	21

Table 1. Channel metrics before and after restoration. An * indicates data from Neudecker et al. 2013.

METHODS

Water temperature reduction.— To examine water temperature reduction, we monitored water temperatures near the mouth of Kleinschmidt Creek and at river km 4.2 on the lower North Fork of the Blackfoot River between 1-June to 1-October over a course of seven years between 1998 and 2012 (Figure 1). Monitoring includes three years pre-treatment (1998, 1999 and 2001) and four years post-treatment (2002, 2004, 2010, 2012) on both Kleinschmidt Creek and the North Fork. During these years, temperature data were collected on Kleinschmidt Creek at two sites during different years downstream of the treatment (km 0.1 and km 0.5). Both data sets were used in this study because of their downstream location from the primary treatments and proximity to each other. For both Kleinschmidt Creek and the North Fork, we used digital thermograph recorders (Onset Computer Corporation, Pocasset, Massachusetts), which recorded continuously at intervals of 48- or 72-minutes.

We used a before-after, control-impact design to compare the mean, maximum, minimum and daily range in temperature for the control (North Fork) and impact (Kleinschmidt Creek) site for the 3 years pre-restoration and over the 11-year span of post-restoration data. Data were analyzed as a two-factor anova with pre/post restoration time period and site as fixed factors.

Whirling disease testing in Kleinschmidt Creek.- Similar to prior whirling disease studies in the Blackfoot Basin (Pierce et al. 2009, 2012, Neudecker et al. 2012), we conducted sentinel cage exposures using hatchery rainbow trout fry (diploid age-0 cohorts) in lower Kleinschmidt Creek as surrogates for infection in order to monitor the prevalence and severity of *M. cerebralis* infection. Test fish ranged in age from 66 to 151 d post-hatch and total lengths ranged from 34 - 53 mm (Table 2). Exposure trials were undertaken between late winter (March 15) and early summer (July 11) and spanned a seven-year period including two years of pre-restoration (1998-99) and three years post-restoration (2002-04). To further examine variation of infection among three trout species present in Kleinschmidt Creek, we also completed side-by-side exposure trials of age-0 brown trout, brook trout and rainbow trout in March 2002 at one year post-restoration (Table 2). This March to July span of exposure trials in this study overlaps with hatching, emergence and/or early rearing periods (i.e., periods of increased disease susceptibly) for fall spawning brown trout, brook trout, bull trout and mountain whitefish) and spring spawning rainbow trout and westslope cutthroat trout (Behnke 1992; Ryce 2005; Neudecker et al 2012; Pierce et al. 2009, 2012). According to Ryce (2005), age 0 rainbow trout are most susceptible if exposed to *M. cerebralis* at <63 d posthatch and size

<40 mm, after which time the effects of disease are dampened through increased resistance.

Following field exposures and a holding period, all fish in this study were killed and heads were histologically examined and scored using the MacConnell-Baldwin grading scale (Hedrick et al. 1999; Baldwin et al. 2000; Ryce 2004). This scale categorically ranks the severity of infection into 1 of 6 qualitative groups, ranging from (0) no infection, (1) minimal, (2) mild, (3) moderate, (4) high, to (5) severe. For each exposure trial, the severity of infection was considered high if a majority of exposed trout had histological scores of grade ≥3 severity as used in prior studies (Pierce et al. 2009, 2012; Neudecker et al. 2012). At grade ≥ 3 severity, cartilage damage and a dispersed inflammatory response can be severe in infected fish (Hedrick et al. 1999; Vincent 2002), lead to the development of whirling disease and ultimately lead to elevated mortality (Hedrick et al. 1999; Ryce et al. 2005). During analyses, we further combined grade 4 and grade 5 fish into a single "severe infection" category to enable comparisons with the original 1998 exposure when grade 5 infections were not distinguished from grade 4. We used a χ^2-test to determine differences in histological category between pre- and post-restoration time periods using SAS Analytics Software.

RESULTS

Water temperature reductions — Maximum daily temperature in Kleinschmidt Creek were significantly lower in the 10-years post-restoration period (2002-12) compared to pre-restoration temperatures (1998-2001) and lower than temperatures at the North Fork control site in both time periods (stream × time period, F = 86.2, p <0.0001; Figure 2A). This pattern was consistent for mean daily temperature (stream x time period, F = 3.78, p < 0.0001; Figure 2B),

Figure 2. Comparison of temperatures between the North Fork control and Kleinschmidt Creek: (A) maximum daily temperature, (B) mean daily temperature, and (C) mean daily temperature difference.

and the difference between maximum and minimum daily temperature (stream × time period, F = 129.4, p < 0.0001; Figure 2C). There was no interaction between stream and restoration time period for minimum daily temperature; the daily minimum temperature

in Kleinschmidt Creek was lower than that in the North Fork across the entire study period (stream, $F = 6.2$, $p = 0.013$) and the minimum daily temperature in both streams increased over the post-restoration period (time period, $F = 16.5$, $p <0.0001$).

Sentinel cage exposures.— Following restoration, over 90% of surrogate rainbow trout were infected, of which 78 - 100% had grade ≥ 3 severity during late winter to early summer exposures (Table 2). Pre-restoration infection rates for rainbow trout were distributed across the histological scale significantly different from post-restoration infection rates ($\chi^2 = 157.7$, $p < 0.0001$); post-restoration infections were more severe (Table 2). The side-by-side brown trout, brook trout and rainbow trout exposure trials showed high (grade ≥ 3) severity of infections for both rainbow (100%) and brook trout (88%) versus minimal infection for brown trout (i.e., 2%), none of which exceeded grade 3 infection. In the side-by-side species comparison, exposure trials showed differences in infection rates by species ($\chi^2 = 164.5$, $p <0.0001$). Both rainbow trout and brook trout were dominantly in the most severe infection class; whereas, brown trout showed the lowest rates and severity of infection (Table 2).

Species	Cage site (km)	Month/ year	Exposure Date	Number histologially examined	Individual histological scores						Group scores		Age of fish (d) at exposure	Length of fish (mm) at exposure
					0	1	2	3	4	5	Percent Infected	Percent > grade 3		
Rainbow trout	0.1	Jul-98	7/1-7/11	48	5	3	7	13	20	na	90	69	75	37
Rainbow trout	0.1	Jul-99	7/1-7/11	50	5	3	3	4	18	17	90	78	91	43
Rainbow trout	0.3	Mar-02	3/15-3/25	48	0	0	0	3	17	28	100	100	71	45
Brown trout	0.3	Mar-02	3/15-3/25	43	42	1	0	0	0	0	2	0	107	37
Brook Trout	0.3	Mar-02	3/15-3/25	50	2	1	3	4	18	22	96	88	115	38
Rainbow trout	0.3	Apr-03	4/23-5/3	50	1	0	0	0	9	40	98	98	151	53
Rainbow trout	0.3	Jun-03	6/20-6/30	49	0	0	0	0	3	46	100	100	111	44
Rainbow trout	0.3	Jun-04	6/21-7/1	27	1	0	1	0	0	25	96	93	66	34

Table 2. Sentinel exposure results for lower Kleinschmidt Creek before (1998-99) and after (2002-04) restoration. The table shows the timing of exposure, individual and groups histological scores, percent infection and percent of sample with $\geq$ grade 3 severity of infection as well as the age and length of the test fish.

DISCUSSION
Groundwater-dominated streams are important for wild trout because the potential for cool temperatures and stable inflows can provide optimum conditions for spawning, rearing and refugia. Yet, the same groundwater environments that foster productive fisheries also make spring creeks highly prone to the proliferation of *M. cerebralis* (MacConnell and Vincent 2002; Burkhart and Hubert 2005; Neudecker et al. 2012). Kleinschmidt Creek was actively restored to more natural form in 2001 to improve habitat for wild trout, reduce water temperatures and test the efficacy of restoration for whirling disease reduction. Though stream renaturalization clearly increased numbers of wild trout (i.e., brown trout) and reduced water temperatures, restoration showed no reduction in the severity of *M. cerebralis* infection. To the contrary, infections were higher post-treatment and severe enough ($\geq$ grade 3) to impact all salmonids present in Kleinschmidt Creek with the exception of brown trout.

Channel restoration and water temperature reduction – The importance of groundwater to salmonid habitat is widely recognized (Brown and Mackay 1995; Baxter and Hauer 2000; Chu et al. 2008), yet, very few, if any, applied studies examine the effects of

renaturalization of groundwater-dominated streams or the potential of groundwater inflows to buffer anthropogenic warming (Poole and Berman 2001; Ebersole et al. 2003). Groundwater cooling is particularly important because the conservation of migratory native trout often rely on patchy network of cold low-elevation streams for refugia (Swanberg 1997; Rieman et al. 2007; Jones et al. 2013). Indeed, this is the case in the lower 12km of North Fork of the Blackfoot River where cold-water inflows from primarily groundwater sources provides thermal refugia for migratory bull trout from the Blackfoot River during the heat of mid-summer (Swanberg 1997; USFWS 2010).

This study shows active restoration can significantly reduce high water temperature during summer. In the case of Kleinschmidt Creek, a 56% reduction in wetted surface area of the channel preceded a 5°C decline in maximum water temperatures one year post-treatment, a long-term reduction in the average maximum daily of 2.5^{o}C (Figure 2) and a narrower range of temperatures. Though lower temperatures were sustained over the monitoring period, the sharp (5 ^{o}C) temperature reduction within one year post-treatment was likely due to reduced wetted surface area versus passive vegetative (e. g., shrub) regrowth given the early stages of vegetative recovery at the time of the initial reduction.

Following restoration, temperatures on Kleinschmidt Creek declined into the thermal niche of bull trout (i. e., maximum temperatures < 13°C; Selong et al. 2001; Dunham et al. 2003) with maximum temperature 2-4°C colder those in the North Fork (range 16-18°C) of the Blackfoot River (Figure 2). Other reconstructed spring creeks to the North Fork have shown similar maximum annual temperature reductions of 4-6 ^{o}C and maximum summer temperatures of 10-14°C following full channel reconstruction (Pierce and Podner 2013). Taken together, the results from Kleinschmidt Creek and other restored spring-fed streams show active restoration can reduce temperatures elevated by riparian degradation even in low-elevation streams where coldwater fisheries maybe prone to loss of thermal habitat as currently projected (e.g., Rieman et al 2007; Williams et al. 2009; Jones et al. 2013).

Severity of M. Cerebralis infection - Despite more natural channel morphology and reductions in water temperature, exposure trials of surrogate rainbow trout showed no post-treatment moderation in the severity of *M. cerebralis* infection. To the contrary, post-treatment exposure trials indicate continuous high *M. cerebralis* infection scores ($\geq$ grade 3) between March and July (Table 2), as also shown in the groundwater-induced upper Blackfoot River (Pierce et al. 2012). Except for high infections in summer, the timing of high exposure scores in our study overlap in time with other Montana spring creeks that show infectious conditions from autumn into spring (Neudecker et al. 2012). Unlike the seasonal peak (summer to autumn) of high infections for Montana Rivers (Downing et al. 2002; Gilbert and Granath 2001; Neudecker et al. 2012; Pierce et al. 2009, 2012), exposure trials in other groundwater-dominated streams in western Montana (Neudecker et al. 2012; Pierce et al. 2012), combined with our study results show infectious conditions can occur continuously regardless of season.

Though histological scores were higher post-treatment (Table 2), this may be due to the early stages of the *M. cerebralis* enzootic, which intensified in the Blackfoot Basin between 1998 and 2005 (Pierce et al. 2011). Other factors such as stable water temperature influencing the development and release of TAMs (Neudecker et al 2012; Pierce et al. 2012) and/or higher spore loads associated with increasing brown trout abundance in Kleinschmidt Creek may also increase infections in spring creeks (Kerans

and Zale 2002; Burkhart and Hubert 2005, Neudecker et al. 2012). Though the severity of infection increased post-restoration, exposure trials also ended at three years post-treatment, which may not provide enough time for restoration-induced changes to alter tubificid lineages and/or otherwise reduce the proliferation of *M. cerebralis* through changes in benthic communities (Kerans et al. 2004; Beauchamp et al. 2005; Nehring et al. 2005).

Prior to restoration, Neudecker et al. (2012) described Kleinschmidt Creek as a localized "hot spot" for infection. Such areas of high and continuous infection contrast in time and space with many basin-fed streams (Anlauf and Moffitt 2008; Hansen and Budy 2011; McGinnis and Kerans 2013), including the North Fork in the Blackfoot River, where sentinel exposures (near the confluence of Kleinschmidt Creek) consistently show low to no infection (Neudecker et al. 2012; Pierce et al. 2009, 2012). It appears infectious conditions in spring creeks relate more to areas of stable water temperatures where *M. cerebralis* develop and release TAMs more continuously versus other stream conditions (e.g., elevated levels of fine sediment), which create habitat for *T. tubifex* (Neudecker et al. 2012; McGinnis and Kerans 2013). To further clarify sediment conditions in Kleinschmidt Creek, we performed post-treatment Wolman pebble counts and found 21% fine sediment (<2mm) compared to 30% pre-treatment (Neudecker et al 2013). Based on these reductions, Kleinschmidt Creek shows little short-term (three years post-treatment) potential for restoration-induced reductions in *M. cerebralis* infection. However, this may not be the case in basin-fed streams where human conditions, such as elevated temperature, sediment and nutrients regimes more directly influence the proliferation of *M. cerebralis* (Zendt and Bergersen 2000; Anlauf and Moffitt. 2008; Hansen and Budy 2011; McGinnis and Kerans 2013). Additional restoration and evaluations are clearly needed to better explore these questions.

With the exception of brown trout, most post-treatment exposures trials showed severe ($\geq$ grade 4) infection. At $\geq$ grade 4, cartilage degenerates and large numbers of leucocytes infiltrate necrotic cartilage, which leaves susceptible young fish weak and less able to compete for food and habitat and ultimately elevates mortality (Hedrick et al. 1999; Vincent 2002; Ryce 2004). In addition to grade $\geq$4 infections, most rainbow trout in this study exceeded the age (i.e., 63 d) and size (i.e., 40 mm) that resistance to whirling disease is conferred (Ryce 2005), which confirm infectious conditions. According to MacConnell and Vincent (2002), rainbow trout are highly susceptible to whirling disease, westslope cutthroat trout, brook trout and bull trout have an intermediate to partial susceptibly; whereas, brown trout have a high innate natural resistance to the *M. cerebralis* parasite, as demonstrated by the side-by-side exposures in this study. Following restoration brown trout numbers have increased significantly in Kleinschmidt Creek, whereas, susceptible species (rainbow trout, bull trout or westslope cutthroat trout) have shown no detectable post-treatment improvement.

CONCLUSIONS

Channel reconstruction can reduce summer water temperatures in degraded groundwater environments in low elevation streams, depending on geomorphic, groundwater and vegetative characteristics associated with site potential. Despite large changes in channel morphology and reductions in water temperature, the severity of *M. cerebralis* infection remained high for susceptible salmonids, but low for nonnative brown trout, a salmonid with natural immunity to the parasite. Because of the more continuous release of TAMS in spring creeks and increased disease resistance of brown

trout, it appears unlikely salmonids other than brown trout can reproduce and rear successfully in Kleinschmidt Creek due to highly infectious conditions that overlap with hatching and/or early rearing windows for both spring and fall spawners. Though infectious conditions clearly favor brown trout, a reduction in water temperature may favor age 1 and older native trout within Kleinschmidt, as well as improve thermal habitat in receiving waters of the North Fork where infections are low.

ACKNOWLEDGMENTS

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ARTICLE

Spawning Behavior of Mountain Whitefish and Co-occurrence of *Myxobolus cerebralis* in the Blackfoot River Basin, Montana

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Abstract

To assess the exposure of Blackfoot River mountain whitefish *Prosopium williamsoni* to the exotic parasite *Myxobolus cerebralis*, the cause of salmonid whirling disease, we investigated the spawning behavior of 49 adult mountain whitefish and their overlap with *M. cerebralis* within the Blackfoot River basin, Montana. A majority of the mountain whitefish radio-tagged in the Blackfoot River migrated upstream (range, 0.1–79.0 km) to spawning sites located primarily in the main stem of the Blackfoot River. Spawning ranged from 31 October in the lower river to 9 November in the upper river and occurred across a range of substrate and channel types. Despite later spawning in the upper river, eggs hatched earlier under the warming influence of groundwater inflows. Here, a majority of wild mountain whitefish fry (65%) tested positive for *M. cerebralis* infection during the immediate posthatch period of mid-April. Conversely, mountain whitefish fry from the lower river, downstream of the groundwater influence, showed no detectable infection. June exposure trials using surrogate rainbow trout *Oncorhynchus mykiss* in nine tributaries supporting mountain whitefish showed *M. cerebralis* infection rates ranging from 0% to 100% as well as a pattern of high triactinomyxon (TAM) exposure throughout the main-stem Blackfoot River. For mountain whitefish, the co-occurrence with *M. cerebralis* varied spatially across the basin and temporally within the main-stem Blackfoot River at the most vulnerable early life stages. This variability appears to buffer age-0 mountain whitefish from infectious conditions across large areas of the basin. However, continuous TAM release from groundwater-influenced environments coinciding with mountain whitefish hatch and early rearing may impose pathogenic conditions on mountain whitefish in the upper Blackfoot River.

Mountain whitefish *Prosopium williamsoni*, an endemic salmonid in the Pacific Northwest, occupy a range of environments, including medium to large rivers as well as lake and reservoir environments. In the Blackfoot River basin of western Montana, mountain whitefish occupy streams and rivers and interconnected natural lakes at the low elevations of the basin, a distribution that broadly overlaps with that of the parasite *Myxobolus cerebralis*. Despite the ubiquitous and often abundant presence of mountain whitefish in the large river systems, the life histories, population status, and potential effects of *M. cerebralis* on mountain whitefish populations are rarely studied and poorly understood. Even so, mountain whitefish are ecologically important as forage for upper trophic predators such as native bull trout *Salvelinus confluentus*, a species listed

as threatened under the Endangered Species Act (USFWS 2010).

Whirling disease, a parasitic infection caused by the myxosporean *M. cerebralis*, is known to infect six genera of salmonids, including the genus *Prosopium*, which includes mountain whitefish. *Myxobolus cerebralis* has a complex, two-host life cycle involving the aquatic oligochaete worm *Tubifex tubifex* and a salmonid host. Salmonid susceptibility to the pathogen varies by species (Hedrick et al. 1999; MacConnell and Vincent 2002; Vincent 2002), fish age and size (Ryce et al. 2005), and parasite dose at time of exposure (Vincent 2002; Schisler 2010). Mountain whitefish are considered less susceptible to severe infection than other susceptible salmonids (MacConnell and Vincent 2002). However, age-0 mountain whitefish

are highly susceptible to injury-related mortality when exposed to *M. cerebralis* at a very young age (MacConnell and Vincent 2002; Schisler 2010).

Following the detection of *M. cerebralis* in the Blackfoot River basin in 1995, Montana Fish, Wildlife and Parks (FWP) began monitoring the extent of the range of *M. cerebralis* using sentinel exposures of age-0 hatchery rainbow trout *Oncorhynchus mykiss* as surrogates for infection in wild salmonids. Between 1998 and 2009, exposures of rainbow trout at 32 fixed monitoring sites identified the range expansion of *M. cerebralis* among certain low-elevation streams of the Blackfoot River valley (Pierce et al. 2009), including the upper Blackfoot River, where summer exposures have consistently demonstrated a high severity of infection since 2005 (FWP, unpublished data). Concurrent with the expansion of *M. cerebralis*, reports of possible disease-related mountain whitefish declines across the American West have been mounting (Burkhardt 2002; Vincent 2009; Schisler 2010), laboratory research has demonstrated high *M. cerebralis*–induced mortality of age-0 mountain whitefish (MacConnell et al. 2000; Schisler 2010), and field-based research has suggested similar high age-0 mortality in the wild (Hubert et al. 2002a; Schisler 2010).

Because *M. cerebralis* poses the greatest threat to salmonids during the early life stages (MacConnell and Vincent 2002; Ryce et al. 2005), the timing and location of spawning and early rearing sites and the co-occurrence of *M. cerebralis* essentially determine susceptibility to whirling disease (Bartholomew and Wilson 2002; Koel et al. 2006). Fish are most vulnerable if they hatch during the peak release of *M. cerebralis* triactinomyxons (TAMs), which typically occurs during the months of June through September (Thompson and Nehring 2000; Gilbert and Granath 2001; Downing et al. 2002) at water temperatures near 12–15°C (El-Matbouli et al. 1999; Kerans et al. 2005). Conversely, species that spawn in the fall and hatch during late winter or early spring (e.g., mountain whitefish) prior to the seasonal peak in TAMs, are usually older, larger and more resistant when they first encounter high parasite abundance at conducive temperatures and thus are less likely to develop whirling disease than spring spawners (Vincent 2000; Ryce et al. 2004). However, in groundwater-influenced environments, where water temperatures are moderated and more constant, high infection can occur in the late winter and early spring (Anderson 2004). This early exposure elevates the infection potential for fall spawners in general as well as injury-related mortality in the case of mountain whitefish (Hubert et al. 2002a, 2002b; Schisler 2010).

Although the distributions of mountain whitefish and *M. cerebralis* overlap at the low elevations of the Blackfoot River basin (Figure 1), the exposure risk of age-0 mountain whitefish to *M. cerebralis* at the critical early rearing stages is poorly understood. To investigate this exposure, we assessed the spawning behavior of mountain whitefish and the overlap with *M. cerebralis* within the main-stem Blackfoot River and several tributaries supporting mountain whitefish. The study objectives were to (1) identify the spawning movements, locations of spawning sites, and hatching periods for mountain whitefish in the Blackfoot River, (2) test for *M. cerebralis* infection at the early life stages across distinct spawning and early rearing areas of the Blackfoot River, and (3) examine the spatial overlap of *M. cerebralis* across mountain whitefish habitat within the basin. Our broader purpose was to gain a better understanding of mountain whitefish life history as well as the risks of *M. cerebralis* exposure in order to better manage mountain whitefish within parasite-positive rivers of western Montana.

STUDY AREA

The Blackfoot River, a free-flowing fifth-order tributary (Strahler 1957) of the upper Columbia River, lies in west-central Montana and flows west 212 km from the Continental Divide to its confluence with the Clark Fork River at Bonner, Montana (Figure 1). The River drains a 5,998-km^2 watershed through 3,038 km of perennial streams and generates a mean annual discharge of 44.8 m3/s (U.S. Geological Survey, unpublished data). The physical geography of the watershed is regionally variable, with subalpine forests dominating the high mountains, montane woodlands at the mid elevations, and semiarid glacial (pothole and outwash) topography on the valley floor. The primary tributaries of the Blackfoot River include the Clearwater River, North Fork, and Nevada Creek. Public lands and large tracts of industrial forestlands generally comprise the mountainous areas, while private lands comprise most of the foothills and bottomlands where traditional uses of the land include mining, timber harvest, cattle ranching, and recreation.

Within the Blackfoot River basin, the distribution of mountain whitefish includes the main-stem Blackfoot and Clearwater rivers, the larger, colder tributaries, the glacially formed lakes on the floor of the Clearwater River valley, and the lower reaches of small streams where age-0 mountain whitefish tend to concentrate during summer (Pierce et al. 2008). The total distribution of mountain whitefish in the Blackfoot basin spans about 450 km of rivers and streams. Although mountain whitefish occupy only about 15% of streams in the Blackfoot basin, they support a majority of the salmonid biomass in the main-stem Blackfoot River and comprise as much as 70% of the salmonid community (Pierce and Podner 2011). The mountain whitefish distribution overlaps with that of *M. cerebralis* at the lower elevations of the Blackfoot basin as well as with the general distribution of bull trout (USFWS 2010; Figure 1), where the mountain whitefish is considered an important forage fish (Bjornn 1991; McPhail and Troffe 2001).

For this study, we divided the main-stem Blackfoot River into three reaches downstream of river kilometer (rkm) 174 based on morphological features of the river environment. The lower river reach includes the lower 55.8 km of the main-stem Blackfoot River between the mouth and its confluence with the Clearwater River. This lower reach has a confined, higher-gradient channel with gravel to boulder substrate and deep bedrock and boulder-formed pools through a narrow canyon and is fed primary by higher-gradient tributaries. The middle reach extends
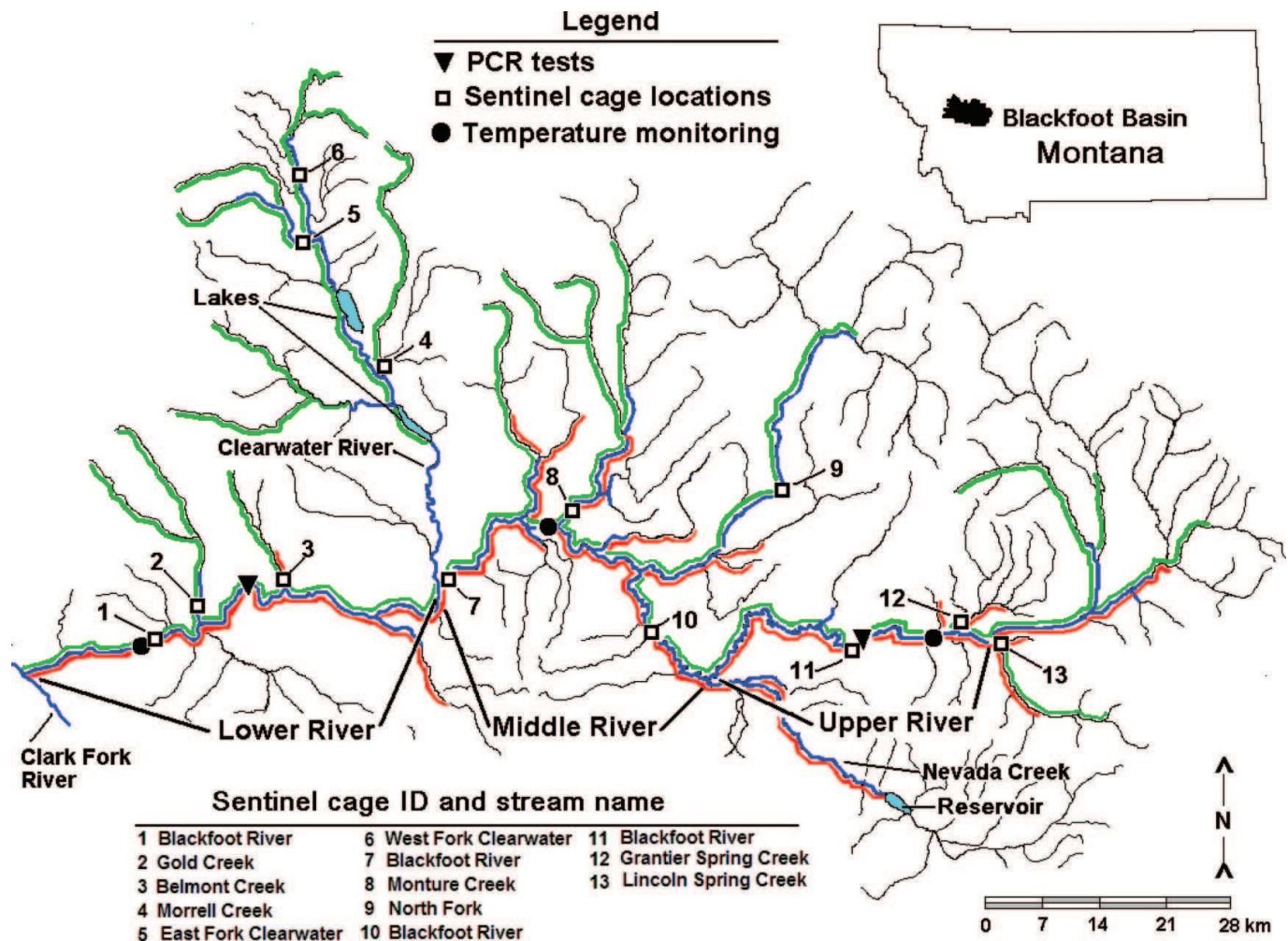
FIGURE 1. Map of the Blackfoot River basin showing the approximate overlap of bull trout (bold green lines), mountain whitefish (bold blue lines), and *M. cerebralis* (bold red lines) distributions (USFWS 2010; Pierce and Podner 2011). Also shown are the sentinel cage sites, water temperature monitoring sites, and locations of fish sampled by PCR for *M. cerebralis*, along with the demarcations of the lower, middle, and upper reaches of the Blackfoot River. The numbers assigned to the sentinel cages correspond to those in Table 2. [Figure available online in color.]

from the mouth of the Clearwater River 53.3 km upstream to the mouth of Nevada Creek. Here the channel is less confined, with deep pools with similar coarse substrate and is fed by larger lower-gradient tributaries. The upper reach extends 64.9 km from Nevada Creek to an intermittent (seasonally dry) section of the Blackfoot River located at km 174 (Figure 1). This upper reach is a sinuous, lower-gradient, unconfined alluvial channel with a primarily gravel substrate. The upper portion of the upper reach is groundwater induced and fed via several spring creeks and groundwater seeps, which collectively create stable river flows and moderate water temperatures during base flow (August–May) periods.

METHODS

Radiotelemetry: Migration and spawning.—To identify the spawning migrations and the timing and location of spawning sites, we tracked 49 adult mountain whitefish in the three study reaches of the Blackfoot River using radiotelemetry. These fish

were captured using electrofishing in the Blackfoot River, implanted with continuous (12 h on : 12 h off) Lotek (Lotek Wireless, Newmarket, Ontario) radio transmitters (model NTC-6-2) on 10–11 June 2008 (*n* = 13) and between 4 and 17 June 2009 and tracked from early June (*n* = 36) through the end of the spawning period in late November (in 2009) or December (in 2008). These fish ranged from 305 to 485 mm in total length (mean, 388 mm) and from 274 to 1,146 g in weight (mean, 623 g). Transmitters were evenly allocated among the three reaches. Fish were captured in the spring during high, turbid flow conditions at water temperatures ranging from 6.1°C to 13.8°C. Transmitters weighed 4.5 g, and each emitted a unique coded signal. Transmitters weighed less than 2% of the mass of recipient fish (Winter 1996) and were implanted following standard surgical methods (Swanberg et al. 1999).

We located fish on foot using a handheld three-element Yagi antenna or by truck using an omnidirectional whip antenna. We located fish weekly prior to migrations

(June–August), 3–4 times per week during migrations and spawning (September–November), and once per week following spawning (November–December). All river locations and movements of mountain whitefish were referenced by rkm. As in a previous study (Pierce et al. 2009), we assumed that fish spawned if they followed a prespawning migration pattern common to other migrants in this study. The most upstream or downstream location for each fish expressing movement during the spawning window was the assumed spawning site. Spawning was visually confirmed in the upper Blackfoot River where viewing conditions allowed but not in the lower river due to poor viewing conditions. For individual fish, we estimated the timing of spawning movement and spawning as the central date between two contacts. Among reaches, the peak of spawning was identified as the median spawning date (Pierce et al. 2009).

Of the 49 radioed mountain whitefish, 18 nonmigrants showed no movement beyond the boundary of the habitat unit and were removed from the analyses of migration and spawning. Because of small sample sizes between reaches and similar spawning dates in both 2008 and 2009 (i.e., median spawning date = 5 November in both years), we grouped the remaining 31 mountain whitefish and used linear regressions to explore the relationships between the start date of migration and (1) the distance (km) to the spawning sites and (2) the total duration (number of days) of spawning-related migrations. To compare spawning dates among the three river reaches, we used an analysis of variance test (Kruskal–Wallis one-way ANOVA by ranks). These tests were performed in Statistica (version 7) software and evaluated at the $\alpha = 0.05$ level of significance.

Water temperature and hatching.—To further assess the timing of mountain whitefish migration and spawning and to estimate the timing of the mountain whitefish egg hatch, we used mean daily water temperatures measured with digital thermograph recorders (Onset Computer Corporation, Pocasset, Massachusetts) located in each of the three reaches (km 12.7, 73.5, and 169.0) of the Blackfoot River (Figure 1). Following Pierce et al. (2009), all thermographs recorded at 48-min intervals. To estimate the timing of the hatch for each of the three main-stem study reaches, we first averaged daily temperature readings from each of the three thermographs to calculate degree-days. Because the total degree-days ($°C$) necessary for incubating mountain whitefish eggs varies with temperature regime, we then used two calculations to estimate the timing of the hatch based on the thermal conditions specific to each river reach. We used a total of $258°C$ degree-days for mountain whitefish eggs incubated at $2°C$ mean daily temperature (Schisler 2010) to estimate the timing of the hatch in both lower river reaches where colder winter water temperatures prevail. For the upper reach, where winter water temperatures are higher, we used a total of $320°C$ degree-days for mountain whitefish eggs incubated at a mean daily temperature of $3.5°C$ (Jody Hupka, Pony Fish Hatchery, FWP, personal communication). Depending on the reach, we then estimated the hatching date of each fish within each reach of the Blackfoot River using an accumulation

of either $258°C$ or $320°C$ degree-days, beginning with the estimated spawning date of each migrant mountain whitefish. The two migrant mountain whitefish that ascended the lower North Fork during the spawning period were excluded from estimates of hatching dates due to temperature data gaps during the winter incubation period.

Overlap of Myxobolus cerebralis with mountain whitefish.— We used two methods to investigate the overlap of *M. cerebralis* with mountain whitefish. One method was a basin-scale assessment of *M. cerebralis* in streams supporting mountain whitefish using sentinel cage exposures of hatchery rainbow trout as surrogates for *M. cerebralis* infection. Following Pierce et al. (2009), 50 hatchery rainbow trout of the Fish Lake strain (diploid age-0 cohorts; mean total length = 34 mm) were placed in sentinel cages at 13 sites 36–39 d posthatch to test for parasite exposure (Figure 1). Field exposures ran between 15 and 24 June 2009, a time that corresponds with the typical peak TAM production period for rivers of western Montana (Downing et al. 2002; Krueger et al. 2006; Pierce et al. 2009). Following sentinel exposures, test fish were held in pathogen-free water for another 186–217 d to allow *M. cerebralis*, if present, to mature, at which time all surviving fish were killed and sent to the Washington State University Animal Disease Diagnostic Laboratory at Pullman, where fish heads were histologically analyzed and scored using the MacConnell–Baldwin grading scale to determine infection and disease severity (Baldwin et al. 2000). This scale classifies infection into one of six categorical groups, ranging from 0 (nondetected) to 5 (severe). For this study, TAM exposure was considered high if a majority (>50%) of exposed trout had histological scores of at least grade 3. At grade 3 or higher, cartilage damage and inflammation of tissue can be severe in infected fish (Hedrick et al. 1999; Vincent 2002; Ryce et al. 2004).

The second method involved a polymerase chain reaction (PCR) test specific to *M. cerebralis* infection in wild mountain whitefish from the Blackfoot River. For this test, we collected 20 age-0 mountain whitefish from lower reach (km 29.1) downriver of the groundwater discharge area and 20 age-0 mountain whitefish from the upper reach (km 153.7) within the groundwater influence area (Figure 1). Fish were collected on 19–20 April 2010 during the early posthatch period. We did not test mountain whitefish in the middle reach due to geomorphic and water temperature similarities with the lower reach. These 40 fish (average total length = 18 mm; range, 14–24 mm) were placed in 95% ethanol and sent to the Colorado Division of Wildlife fish health laboratory for PCR analyses. For the PCR test, the head of each fish was removed and placed in an individual centrifuge to extract DNA. The sample DNA was examined for the presence of the *M. cerebralis Hsp70* gene segment by single-round PCR amplification (Schisler et al. 2001). Based on positive correlations between PCR amplification and spore counts, *M. cerebralis* infection of individual mountain whitefish was then grouped into one of following five categories: (1) below detection levels, (2) weak positive signal, (3) positive
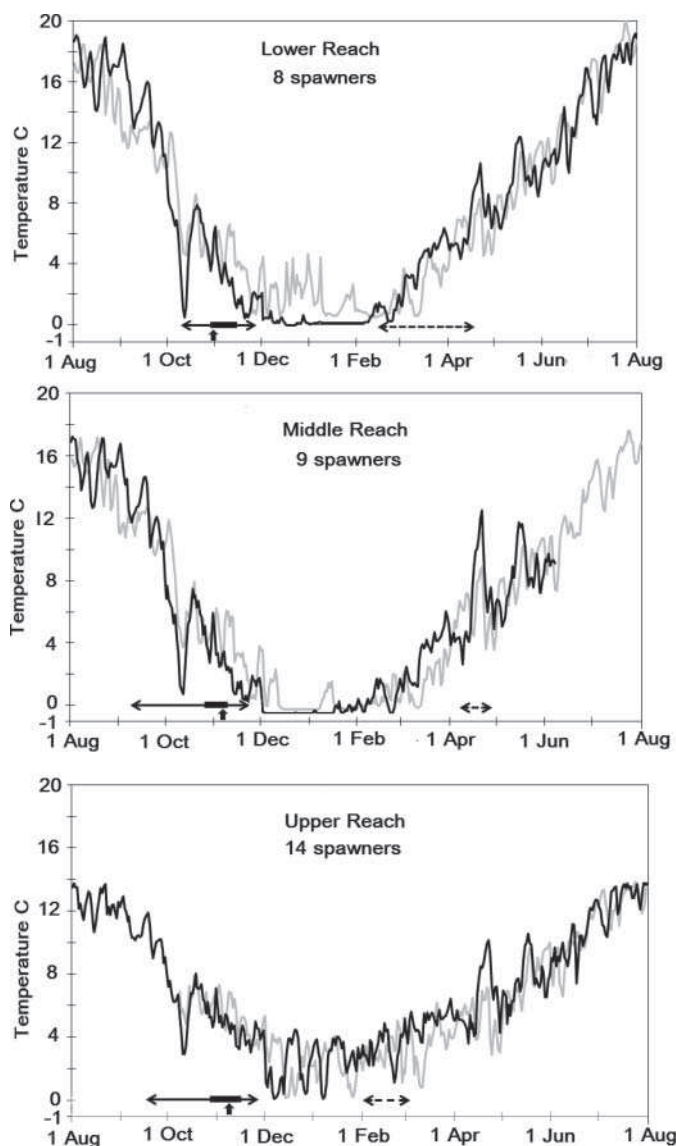
FIGURE 2. Mean daily water temperatures for the three reaches of the Blackfoot River from 1 August 2008 to 1 August 2009 (grey lines) and from 1 August 2009 to 1 August 2010 (black lines). The total period of spawning migrations is shown by the horizontal black arrowed lines. The thickened areas on the migration lines show the spawning windows, and the vertical arrows show the median spawning dates. The horizontal dashed lines show the estimated hatching period for an accumulated 258°C temperature units (lower and middle reaches) and 320°C temperature units (upper reach). Note the relatively warm winter temperatures in the upper reach.

signal, (4) strong positive signal, and (5) very strong positive signal (Schisler et al. 2001).

RESULTS

Spawning Behavior

To identify migrations and the locations and timing of spawning events, we tracked 49 radio-tagged mountain whitefish from early June though the postspawning period in late November. We made a total of 1,887 contacts, with an average of 39 contacts (range, 22–47) per fish. Of these 49 fish, most ($n = 31$) mountain whitefish expressed spawning-related migrations (0.1–79.0 km) beginning in early September, which culminated with spawning between 25 October and 26 November. Of these 31 mountain whitefish, most ($n = 29$) spawned within the main stem of the Blackfoot River and only 2 ascended a tributary (i.e., the North Fork; Figure 1). Over the course of this spawning, the remaining 18 mountain whitefish were identified as nonmigrants based on movements within the habitat unit but no detectable migration beyond the habitat unit. Across the three river reaches, the number of migrant mountain whitefish relative to the total number of radio-tagged mountain whitefish was 9 of 17 (53%), 11 of 15 (73%), and 11 of 17 (65%) for the lower, middle, and upper reaches, respectively.

Migrant mountain whitefish began their prespawning migrations as daily average water temperatures declined to 12°C (Figure 2). With the onset of migration, 27 mountain whitefish traveled upriver and 4 traveled downriver (Figure 3). These 31 fish migrated a median of 3.2 km (average = 12.2 km, range = 0.1–79.0 km) to spawning sites, where they spent an average of 12 d (range, 1–98). Fish that began spawning migrations earlier traveled a longer total (pre- and postspawning) distance ($R^2 = 0.14$, $P = 0.036$) over a longer period of time ($R^2 = 0.66$, $P < 0.0001$) than fish that began their migrations later. Mountain whitefish from the lower and upper reaches migrated short (median) distances of 1.0 km (range, 0.1–79.0 km) and 2.5 km (range, 0.4–10.2 km), respectively, compared with 8.4 km (range, 0.8–73.0 km) for mountain whitefish in the middle reach. As daily average water temperatures decreased from 12°C to 6°C, migrations attenuated to staging in large schools followed by evening spawning in aggregates of typically 4–10 fish (based on observations in the upper reach), which ensued at an average daily temperature of 5.0°C (range, 3.2–7.1°C). The peak of spawning was 31 October in the lower reach, versus 6 and 9 November in the middle and upper reaches (ANOVA: df = 2, $P = 0.15$).

After spawning, most migrant mountain whitefish ($n = 21$) migrated to downriver wintering areas, six moved short distances upriver, and four remained in the habitat unit used for spawning. Of the 21 fish that migrated downriver, most ($n = 11$) returned to their original premigration start location, including 1 that returned downriver 73.2 km to its original premigration location. Twelve others returned to within 1.6 km of their original start locations.

Water Temperatures and Hatching

For 29 migrant mountain whitefish that spawned in the Blackfoot River, the estimated hatch varied between 1 February and 27 April. However, this timing varied by river reach (Figure 2; Table 1). Despite later spawning in the upper reach, the eggs in the upper reach hatched earlier (1 February to 4 March) due to higher winter water temperature in this groundwater-influenced reach. In the middle reach, where water temperatures were consistently colder, the hatch occurred later (10–27 April). Winter temperatures were more variable in the lower reach, and the
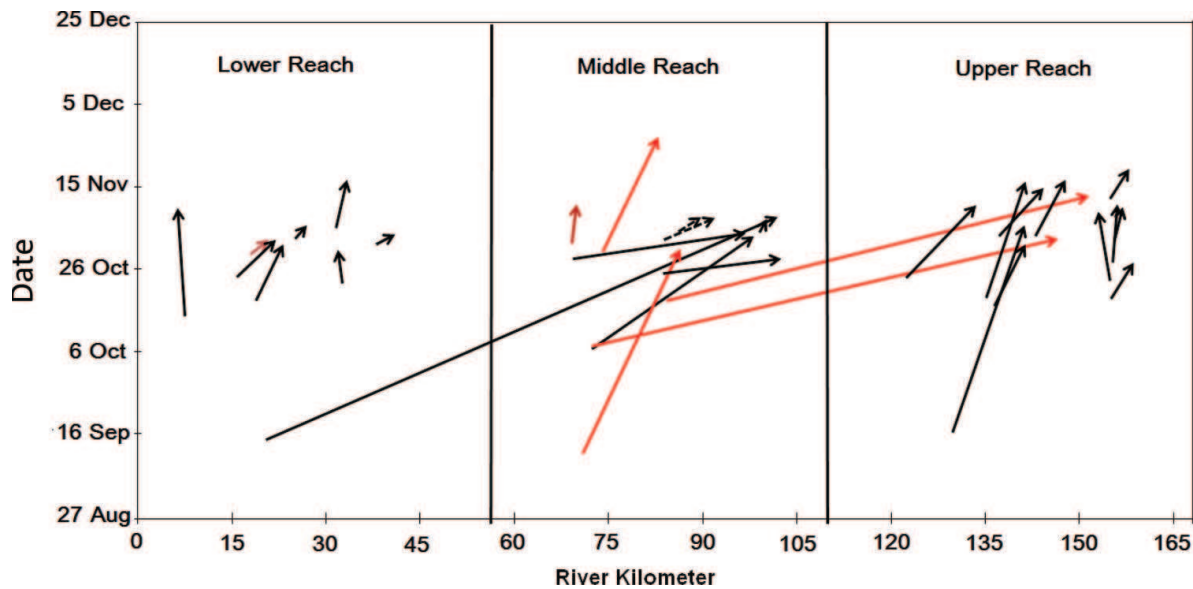
FIGURE 3. Migrations to spawning reaches of 31 mountain whitefish radio-tagged in three reaches of the main-stem Blackfoot River in 2008 (red arrows) and 2009 (black arrows). The arrows show where the migrations began and ended. The dashed lines in the middle reach represent two tributary (North Fork) spawners. [Figure available online in color.]

estimated hatching window was relatively wide (9 February to 22 April).

Histological scores and *M. cerebralis* Infection in Mountain Whitefish

Sentinel exposures and histological examinations of surrogate rainbow trout were completed for all 13 monitoring sites (Table 2). Histological examinations identified infection rates for exposure groups ranging from 0% to 100% during the June 2009 exposure. Seven of 13 exposures groups had high histological scores with ≥50% of the individual exposures scoring at grade ≥3 severity (Table 2). High exposure scores were recorded in all four monitoring sites on the Blackfoot River and one tributary entering each of the three reaches. Four exposure groups did not detect *M. cerebralis* at the three sites in the Clearwater River drainage and Gold Creek. The remaining two streams (Grantier Spring Creek and the North Fork) had low histological scores, with a majority of fish at <grade 3 severity. The PCR tests of newly hatched mountain whitefish tested during April did not detect *M. cerebralis* in the 20 fish collected in the lower Blackfoot River. However, a majority (65%; $n = 13$) of the 20 mountain whitefish from the upper Blackfoot River tested positive for *M. cerebralis*, of which 25% ($n = 5$) of the upper-river sample scored a strong positive signal.

DISCUSSION

Spawning Behavior

Brown (1952) was the first to identify nonmigratory spawning behavior in the larger rivers of Montana. Yet other studies report resident mountain whitefish in smaller streams (Wydoski 2001), residents among migratory populations (Baxter 2002),

and highly migratory behavior across larger river systems (Pettit and Wallace 1975). In our study, migrations ranged from very short distances (<1 km) to long distances across river reaches (Figure 3). Within the larger metapopulation of the Blackfoot basin, additional spawning life history variation (e.g., resident fish) is expected across tributaries where mountain whitefish are consistently sampled (Pierce et al. 2008) but were not clearly linked in this study, with the exception of the lower North Fork. The Clearwater River chain of lakes also support lake-dwelling mountain whitefish; however, that life history has not been studied. Life history variation across tributaries, rivers, and lakes is recognized as high across the range of mountain whitefish (Brown 1952; Pettit and Wallace 1975; Northcote and Ennis 1994; McPhail and Troffe 1998; Wydoski 2001), which includes metapopulation function (migration and genetic exchange) at a broad regional scale (Whitely et al. 2006).

In addition to highly variable spawning movements within the main-stem Blackfoot River, mountain whitefish spawned across a diversity of physical channel features. In the upper study area, we observed aggregates of mountain whitefish broadcast spawning along the margins of pools and glides of an alluvial channel with gravel substrate and groundwater inflow. However, spawning areas in the mid- and lower Blackfoot River were morphologically variable and included large boulder-laden and bedrock pools with cobble to boulder substrate and little, if any, direct groundwater influence. Although we were unable to observe spawning in the larger, deeper confined channels of the mid- and lower Blackfoot River, radio-tracking indicated an increasing intensity of movement of both migrant and nonmigrant mountain whitefish within the large runs and pools (similar to the upper river) during peak spawning periods. Observations and collections of age-0 fry during the immediate posthatch period

TABLE 1. Spawning locations, estimated hatching dates for 258°C and 320°C degree-days, and average daily hatching temperatures for individual fish from the three study reaches and the North Fork (NF).

Spawning reach	River km	Spawning date	Temperature at spawning	Hatching date		Average daily incubation temperature (°C)
				At 258°C	At 320°C	
Lower	20.9	30 Oct 2008	4.8	19 Feb 2009		2.3
	32.7	28 Oct 2009	4.4	3 Apr 2010		1.7
	41.2	31 Oct 2009	6	5 Apr 2010		1.6
	20.9	30 Oct 2009	6.7	5 Apr 2010		1.7
	21.2	31 Oct 2009	6	5 Apr 2010		1.7
	6.8	8 Nov 2009	4	12 Apr 2010		1.7
	33	15 Nov 2009	1.1	15 Apr 2010		1.7
	26.2	4 Nov 2009	3.2	22 Apr 2010		1.7
Middle	83	29 Oct 2008	5.4	10 Apr 2009		1.6
	69	8 Nov 2008	6.9	19 Apr 2009		1.6
	81.4	26 Nov 2008	6.7	27 Apr 2009		1.7
	100.3	25 Oct 2009	5.5	13 Apr 2010		1.5
	97.8	27 Oct 2009	5.2	15 Apr 2010		1.5
	95.6	1 Nov 2009	4.9	17 Apr 2010		1.6
	139	6 Nov 2009	5.4	18 Apr 2010		1.6
	98.8	9 Nov 2009	4.2	19 Apr 2010		1.6
NF	1	6 Nov 2009	6.2			
	1.3	6 Nov 2009	6.2			
Upper	150.5	11 Nov 2008	5.9		28 Feb 2009	2.9
	144.5	1 Nov 2008	6.2		8 Feb 2009	3
	144.5	15 Nov 2009	3.2		3 Mar 2010	2.9
	139	5 Nov 2009	4.8		19 Feb 2010	3
	139.9	30 Oct 2009	7.1		10 Feb 2010	3.1
	143.2	14 Nov 2009	3.2		3 Mar 2010	2.9
	138.9	15 Nov 2009	3.2		3 Mar 2010	2.9
	132	10 Nov 2009	4.6		27 Feb 2010	2.9
	155	16 Nov 2009	3.2		4 Mar 2010	2.9
	151.8	2 Nov 2009	6		15 Feb 2010	3
	156.1	25 Oct 2009	5.5		1 Feb 2010	3.2
	152.6	8 Nov 2009	4.7		25 Feb 2010	2.9
	154.5	4 Nov 2009	4.6		17 Feb 2010	3

seem to confirm local spawning. Consistent with our tracking and fry observations, mountain whitefish spawn across a range of habitat types with little, if any, selection for stream substrate composition (Brown 1952; Daily 1971).

Parasite Distribution and the Influence of Groundwater

The 2009 sentinel cage exposures of surrogate rainbow trout showed highly variable histological scores in June across mountain whitefish habitat. Infection rates ranged from 0% to 100% in the tributaries occupied by mountain whitefish as well as a pattern of high TAM exposure (67–76% of group scores ≥ grade 3) throughout the main stem of the Blackfoot River. This variation among streams conforms to a basin-scale pattern of increasing infection in the downstream direction (Pierce et al. 2009), as broadly observed across the Intermountain West (Sandell et al. 2001; de la Hoz Franco and Budy 2004; Anlauf and Moffitt 2008). In our study area, the low-elevation presence of *M. cerebralis* largely overlaps with the distribution of mountain whitefish with the exception of the Clearwater River, which flows through a series of glacially formed lakes as well as cold, rocky, basin-fed, forested streams with low levels of fine instream sediment, such as Gold Creek and the North Fork Blackfoot River.

Unlike Montana rivers in which TAMs are typically released during summer (Vincent 2000; Downing et al. 2002), groundwater-induced stream (i.e., spring creek) environments

TABLE 2. Severity of *Myxobolus cerebralis* infection based on histology of sentinel cage exposures of surrogate rainbow trout in 13 locations in the Blackfoot basin. Scores in bold italics denote high TAM exposures with a majority of the exposed fish at ≥grade 3 severity. Sentinel cage site locations are indicated in Figure 1.

Cage ID	Stream name	Exposure period	Number rainbow histologically examined	Individual histological scores						Group scores	
				0	1	2	3	4	5	Percent infected	% ≥grade 3
1	Blackfoot River	15–24 June	46	5	6	4	12	17	2	89	*67*
2	Gold Creek	15–24 June	50	50	0	0	0	0	0	0	0
3	Belmont Creek	15–24 June	38	4	0	0	0	4	30	89	*89*
4	Morrell Creek	15–24 June	50	50	0	0	0	0	0	0	0
5	East Fork Clearwater	15–24 June	43	43	0	0	0	0	0	0	0
6	West Fork Clearwater	15–24 June	44	44	0	0	0	0	0	0	0
7	Blackfoot River	15–24 June	38	2	2	5	7	16	6	94	*71*
8	Monture Creek	15–24 June	45	2	2	6	6	22	6	100	*95*
9	North Fork	15–24 June	33	32	0	0	0	1	0	3	3
10	Blackfoot River	15–24 June	45	2	3	6	6	22	6	95	*76*
11	Blackfoot River	15–24 June	43	4	3	7	6	23	0	90	*67*
12	Grantier Spring Creek	15–24 June	34	1	1	2	0	6	24	97	*88*
13	Lincoln Spring Creek	15–24 June	39	28	4	3	3	1	0	28	14

can release TAMS from late winter through spring and into summer (Hubert et al. 2002b; Anderson 2004). This early release relates to stable flow and temperature regimes. However, the low channel gradients, high sediment loading, and organic enrichment typical of spring creeks also tend to create ideal habitat for *T. tubifex* and thus foster high TAM production (Hiner and Moffitt 2002; Hubert et al. 2002a). Similar to reports of early TAM release in smaller spring creeks (Anderson 2004), the April PCR test confirmed positive infection prevalence in the upper study reach in a larger groundwater-induced river environment. In the upper study reach, the temperature sensor recorded a mean daily temperature of 4.6°C (range, 1.1–8.7°C) between the estimated start of the mountain whitefish hatch on 1 February and the 19 April and PCR fry collection date. These low temperatures contrast with reports of TAM viability occurring at higher water temperatures of 7–15°C (El-Matbouli et al. 1999; Sandell et al. 2001; Hiner and Moffitt 2002) and reports of much warmer (12–15°C) temperatures during the typical peak in TAM release for river environments (Vincent 2000; Downing et al. 2002; de la Hoz Franco and Budy 2004), including waters (>20°C) influenced by geothermal input (Koel et al. 2006). Though it may be that high worm abundance can increase TAM release at colder water temperatures (Hiner and Moffitt 2002), high TAM release in spring creeks during winter and spring may also relate to an accumulation of degree-days versus a range of water temperatures (Anderson 2004). With the upper river being warmer, accumulated temperature units would occur faster than in the cooler temperatures of the lower Blackfoot River. Consistent with these mechanisms, most PCR-tested age-0 mountain whitefish (65%) in the upper Blackfoot River tested positive for *M. cerebralis* in early spring. Conversely, the lack of *M. cerebralis* detection in the lower-river PCR test indicates consistency with the typical water temperatures and seasonality of TAM release in a rivers unaffected by direct groundwater inflow.

As shown by the PCR test and sentinel cage exposure in the upper Blackfoot River, groundwater environments can extend parasite exposure from the early spring into summer. With earlier, more continuous exposure, newly hatched mountain whitefish are prone not only to early infection but also to a heightened potential of injury-induced mortality relative to other salmonids (Schisler 2010). This heightened sensitivity to injury reflects the more fragile nature of the newly hatched fry and the invasive nature of the parasite, which causes injury when the sporoplasm penetrates the epithelium. This injury causes osmotic imbalance, plasma leaks, and avenues for secondary infection, which ultimately increases the potential for elevated mortality (MacConnell et al. 2000; Schisler 2010).

Avoidance of the Parasite and Other Mechanisms of Risk Reduction

With the exception of those in groundwater-induced streams, mountain whitefish appear to be separated from *M. cerebralis* over large areas of the basin during the critical early posthatch period. This separation can either help them avoid exposure or slow the progression of infection prior to the onset of seasonally high TAM releases, depending on the spawning and hatching windows and/or the early dispersion of age-0 mountain whitefish.

Though this study identifies spawning and hatching windows in the main-stem Blackfoot River, the early dispersion and other aspects of age-0 mountain whitefish life history are poorly understood. Elsewhere, mountain whitefish fry seek protected backwaters along stream margins once the eggs hatch and then passively disperse downstream during early summer when the water is warmer and food availability is higher (Brown 1952; Grove and Johnson 1978; Northcote and Ennis 1994). In our study area, downriver dispersion of this type would likely place age-0 mountain whitefish from the upper reach in more continuously contact with *M. cerebralis*, first during the spring (posthatch) in groundwater areas and then during summer, when TAM concentrations are seasonally elevated (Table 2). Conversely, downstream dispersion of age-0 mountain whitefish from the mid to lower reaches would avoid early exposure associated with groundwater. In addition, several "clean" mountain whitefish–bearing tributaries enter all study reaches, and these seem to provide more continuous refugia from the parasite.

As with other susceptible salmonids, the time between hatching and parasite exposure may allow mountain whitefish to reach a size or age that is less susceptible to *M. cerebralis* infection and the secondary effects of disease. Though the ability of mountain whitefish to develop physiological resistance to *M. cerebralis* requires further research (MacConnell and Vincent 2002), a reduction in infection prevalence has been detected in mountain whitefish exposed after 5 months of age (Schisler 2010). Other species (e.g., rainbow trout) develop an immune response as early as 9 weeks of age (Ryce et al. 2005). For mountain whitefish, a salmonid with relatively large, platy scales, it may be that scale development at 3–4 months posthatch or 30–45 mm in fork length (Thompson and Davies 1976) provides some protection (i.e., armor) from infection or injury. Under these conditions, early hatching (e.g., February–March) could reduce parasite contact in basin-fed streams prior to the typical summer peak in TAM production.

As shown at a basin scale in this study, the co-occurrence of mountain whitefish and *M. cerebralis* can vary broadly across both time and space. In other areas, spawning windows, for example, extend from late September through February across the range of mountain whitefish depending on elevation and latitude (Brown 1952; Thompson and Davies 1976; McPhail and Troffe 1998; Wydoski 2001). Likewise, fry emergence varies from early February (this study) to as late as early June (McPhail and Troffe 2001). Winter water temperatures greatly influence the timing of the hatch, as described by Schisler (2010), who reported that the degree-days for mountain whitefish egg incubation ranged from 258°C at an average temperature of 2°C to about 444°C at an average temperature of 6°C. In our study area, the mean daily water temperature during the mountain whitefish incubation period varied by year and by up to 4°C depending on the river reach and the influx of groundwater (Figure 2). Similarly, interannual temperature variation can either accelerate or delay the hatch, as shown by two mountain whitefish in the lower reach, one that spawned on 30 October 2008 and the other on 30 October 2009. Under milder winter temperatures, the estimated hatch of the 2008 spawning event occurred on 19 February 2009, as opposed to 5 April 2010 for the 2009 spawner, a difference of 45 d. Interestingly, natural hydrologic events such as spring flooding may also trigger early hatching (McPhail and Troffe 1998).

Conclusions

Mountain whitefish are considered common in many rivers of western Montana, widespread elsewhere, and secure over large areas of their geographic range (McPhail and Troffe 1998; Baxter 2002; Meyer et al. 2009). However, populations are also in decline in many areas of western North America, and in some river systems the declines have been dramatic (Meyer et al. 2009; Vincent 2009; Schisler 2010). Despite their high ecological value, mountain whitefish rarely receive much attention from anglers or resource managers. As a result, evaluations of perceived mountain whitefish declines are generally insufficient to determine whether *M. cerebralis* is causing population-level declines. In our study area, mountain whitefish appear to be separated from *M. cerebralis* over large areas of the basin during the critical early posthatch period, with the exception of the groundwater-induced upper Blackfoot River. Here, winter water temperatures are higher than those in the lower river and remain largely above freezing. This seems to result in earlier and ongoing parasite exposure relative to other areas, such as the lower Blackfoot River and cold, rocky basin-fed streams with low levels of fine instream sediment. Our results suggest that groundwater-induced areas make newly hatched mountain whitefish more likely to be exposed to *M. cerebralis* before the development of scales or other avoidance mechanisms that lower the risk of parasite contact.

While *M. cerebralis* may be deleterious at a local scale in our study area, several other, more direct human-mediated conditions are clearly contributing to the declining mountain whitefish populations. As reported in Idaho and Colorado, dam building and other water projects as well as the recent introduction of exotic predators are all implicated in mountain whitefish declines (Meyer et al. 2009; Schisler 2010). Within the Clearwater River basin of our study area (where *M. cerebralis* has not been detected), sharp declines in the abundance of lake-dwelling mountain whitefish followed the illegal introduction the northern pike *Esox lucius* in the Clearwater lakes. Likewise, the construction of Nevada Reservoir in the Blackfoot Valley preceded the local extirpation of mountain whitefish from upper Nevada Creek prior to the introduction of *M. cerebralis*. Mountain whitefish have also been identified in irrigation canals from the larger, low-elevation streams of the Blackfoot basin (FWP, unpublished data).

In the Blackfoot Valley, the distribution of mountain whitefish overlaps that of bull trout, an imperiled native salmonid (USFWS 2010), and species with moderate susceptibility (MacConnell and Vincent 2002). Interestingly, bull trout also spawn

in the fall in areas of groundwater inflow (Baxter and Hauer 2000), which could elevate the *M. cerebralis* infection potential for this fish in parasite-positive waters. Like those of bull trout, the life histories of mountain whitefish require clean, cold water and open migratory corridors to a variety of habitat conditions. Ongoing recovery activities targeting bull trout, such as the screening of irrigation canals, stream flow improvement projects, and improvements to water quality likely benefit mountain whitefish.

In addition to continued stream improvements in mountain whitefish–bull trout habitat, we recommend (1) expanded disease testing of age-0 mountain whitefish in groundwater environments, (2) evaluations of age-0 life histories within the context of *M. cerebralis* overlap, and (3) expanded monitoring of mountain whitefish populations in parasite-positive waters in order to elucidate long-term population trends.

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ARTICLE

Response of Wild Trout to Stream Restoration over Two Decades in the Blackfoot River Basin, Montana

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Abstract

Anthropogenic degradation of aquatic habitats has prompted worldwide efforts to improve or restore stream habitats for fisheries. However, little information exists on the long-term responses of salmonids to restoration in North American streams. To recover wild trout populations in the Blackfoot River in western Montana, a collaborative approach to stream restoration began in 1990 to improve degraded stream habitats, primarily on private land. To assess the efficacy of various restoration techniques (channel reconstruction and placement of instream habitat structures, restoration of natural instream flows, installation of fish ladders and screens at irrigation diversions, and modification of grazing practices) in the recovery of wild trout, we examined long-term (>5 years) trends in trout abundance on 18 tributaries treated between 1990 and 2005 and subsequently monitored between 1989 and 2010. At pretreatment conditions, average trout abundance was significantly lower in treatment versus reference sites (0.19 versus 0.62 trout/m; $P = 0.0001$). By 3 years posttreatment, trout abundance had increased significantly to an average of 0.47 trout/m across treatment sites ($P = 0.01$) and was no longer significantly different from the reference average ($P = 0.12$). These initial rapid increases were sustained over the long term (5–21 years) in 15 streams. However, trout abundance declined below pretreatment levels on three streams presumably due to the return of human impacts from heavy riparian grazing and detrimental irrigation practices. Although long-term (12 year) average response trends were positive, trends varied spatially and native trout responded more strongly in the upper portion of the basin. Study results indicate that restoration should focus in the mid to upper basin and emulate features of natural channels to promote life history diversity and the recovery of native trout. Finally, long-term monitoring led to adaptive management on most (10 of 18) projects, and thus proved vital to the overall sustainability of wild trout fisheries throughout the basin.

Native salmonids were once abundant and widespread across the western United States, but as natural landscapes were modified many populations declined dramatically to imperiled status (Nehlsen et al. 1991; Behnke 1992; Thurow et al. 1997). Declines are largely associated with mining activities, timber extraction, stream channelization, irrigation practices, dams, riparian grazing, overfishing, and the influence of introduced exotic species (e.g., Meehan 1991; Behnke 1992; Thurow et al. 1997). These anthropogenic activities often destroy and degrade aquatic habitats (Meehan 1991; Waters 1995), disrupt fish migrations (Rieman and McIntyre 1993; Thurow et al.

1997), and can alter environments in favor of exotic organisms (Bartholomew and Wilson 2002; Shepard 2004). As a result, many public and private organizations have developed strategies to improve recovery, management, and protection of native salmonids (e.g., Williams et al. 1997; MBTRT 2000; USFWS 2010).

Despite widespread increases in stream restoration projects, strategies to restore the ecological integrity of river ecosystems remain chronically challenged due to a lack of project monitoring and evaluation (Bernhardt et al. 2005; Roni 2005; Reeve et al. 2006). Consequently, resource managers often lack basic

information to assess the biological effectiveness of techniques or identify where, when, and how to apply adaptive management strategies (Platts and Rinne 1985; Meehan 1991; Wissmar and Bisson 2003; Reeve et al. 2006). While only about 10% of all stream improvement projects implemented in the United States are evaluated (Bernhardt et al. 2005), those that are evaluated generally report increases in stream-dwelling salmonid populations (Reeves et al. 1991; Binns 2004; Roni et al. 2008). However, most studies examined short-term (<5 years), small-scale (i.e., reach level) responses and emphasized traditional (i.e., artificial) habitat enhancement structures (Roni 2005, Roni et al. 2008). Few studies have evaluated fish population responses associated with restoration techniques, such as those that attempt to return streams to undistributed ecological conditions (Baldigo et al. 2008, 2010). Furthermore, few restoration studies have reported community-level shifts in favor of native trout (Behnke 1992), related increases in tributary stocks to metapopulation function (Williams et al. 1997; Reeve et al. 2006; Roni et al. 2008), or examined restoration activities on private lands where traditional land uses, such as livestock production and irrigation, often conflict with sustainable fisheries values (Meehan 1991; Pierce et al. 2005, 2007). Information gaps such as these clearly complicate the ability of fisheries managers and other stakeholders to develop and ensure effective and sustainable conservation strategies within and across ecological landscapes (Platts and Rinne 1985; Wissmar and Bisson 2003; Reeve et al. 2006).

In the Blackfoot River basin of western Montana, most tributaries possess some level of human-induced habitat modification (i.e., channelization, riparian timber extraction, road building, or agricultural practices) land-use activities (Pierce et al. 1997, 2005, 2007, 2008). Because tributary alterations have depleted wild trout fisheries in the Blackfoot River (Peters and Spoon 1989; Peters 1990; Pierce et al. 1997), fisheries biologists working together with willing natural resource agencies, conservation groups, and private landowners have developed a basin-scale, voluntary strategy to improve the ecological integrity of tributaries (Aitken 1997; Pierce et al. 2005; BBCTU 2012). Since 1990, this strategy has focused on the restoration of streams with emphasis on the recovery of federally threatened Bull Trout *Salvelinus confluentus* (USFWS 2010) and Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi*, a Montana species of special concern (Shepard et al. 2005). Both native and nonnative trout of the Blackfoot River exhibit fluvial life histories and rely on tributaries for spawning, rearing, and migration (Swanberg 1997; Schmetterling 2001; Pierce et al. 2007, 2009). To improve tributaries for supporting wild trout, a variety of overlapping restoration techniques that include the core principles of natural channel design (Dunne and Leopold 1978; Rosgen 1994, 1996, 2007) are applied, primarily on lower reaches of small streams and usually on agricultural ranchlands.

The purpose of this study was to evaluate and improve restoration strategies for the recovery of native trout in small tributaries of the Blackfoot River. The primary objectives were to (1) assess the long-term efficacy of restoration techniques for increasing the abundance of wild trout (i.e., native and naturalized nonnative trout) for 18 small tributaries of the Blackfoot River, and (2) examine variation in response of native and nonnative trout at a subbasin scale.

STUDY AREA

Geography and land ownership.—The Blackfoot River, a free-flowing, fifth-order tributary (Strahler 1957) of the upper Columbia River, lies in west-central Montana and flows west 212 river kilometers from the Continental Divide to its confluence with the Clark Fork River at Bonner, Montana (Figure 1). The river drains a 5,998-km^2 watershed through 3,038 km of perennial streams that generate a mean annual discharge of 44.8 m^3/s near Bonner, Montana (USGS 2010 gauge 12340000 field data). Flowing among three mountain ranges, the Blackfoot River drains a diverse range of ecosystems from high-elevation glaciated peaks and alpine meadows, midelevation boreal and montane forests and foothills, to semiarid prairie-pothole and glacio-alluvial plains on the valley floor. Land ownership in the Blackfoot River basin is a mix of public and private lands: 46% is managed by the U.S. Forest Service, 11% by the state of Montana, 7% by the U.S. Bureau of Land Management; 9% by the Plum Creek Timber Company, and 27% is privately owned. Public lands and industrial forestland generally comprise mountainous areas, whereas private lands dominate the foothills and bottomlands where traditional land uses, such as mining, riparian timber harvest, cattle grazing, irrigation, and roads, have contributed to fisheries impairments on a majority (>80%) of tributaries to the Blackfoot River (Pierce et al. 2005, 2008).

Wild trout of the Blackfoot River basin.—Since 1974, the Blackfoot River has been managed for wild trout populations (Zackheim 2006), most of which reproduce in tributaries (Swanberg 1997; Schmetterling 2001, 2003; Pierce et al. 2007, 2009). Nonnative Rainbow Trout *O. mykiss* are prevalent in the lower Blackfoot River and lower reaches of adjacent tributaries where they express both resident and fluvial life histories (Pierce et al. 2009). Conversely, nonnative Brown Trout *Salmo trutta* are prevalent in the upper Blackfoot River and lower reaches of many adjacent tributaries (Pierce et al. 2011). Nonnative Brook Trout *Salvelinus fontinalis* typically occupy the lower reaches of small tributary streams and rarely occupy the main-stem Blackfoot River or steeper headwater areas. Native Westslope Cutthroat Trout, in contrast, are present basin-wide, but most prevalent in streams of the mid-to-upper elevations of the basin. Likewise, Bull Trout are present basin-wide predominately within larger, colder streams (Swanberg 1997; MBTRT 2000; USFWS 2010). Both native Westslope Cutthroat Trout and Bull Trout express stream-resident and fluvial life histories (Swanberg 1997; Schmetterling 2001; Pierce et al. 2007). Compared with nonnative trout, fluvial native Bull Trout and Westslope Cutthroat Trout occupy the main-stem Blackfoot River in relatively low but increasing abundance (Pierce et al. 2011).
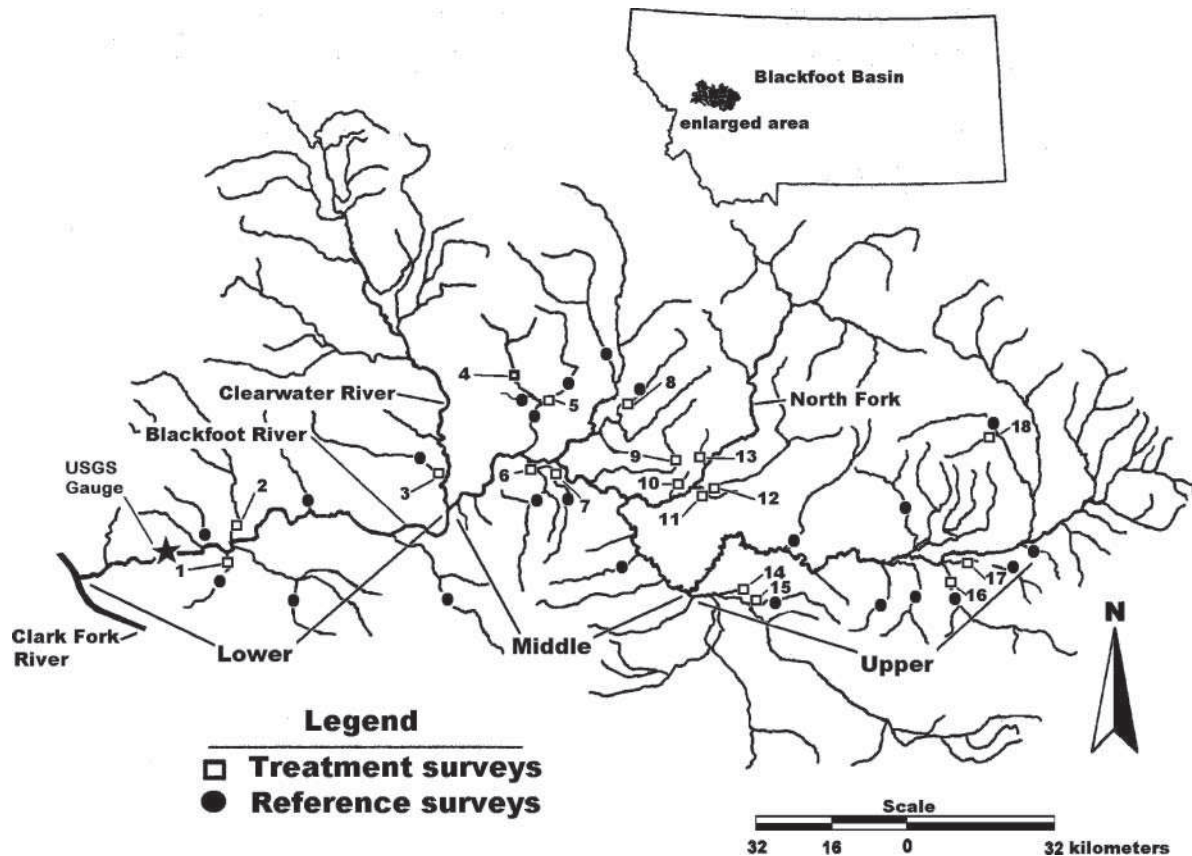
FIGURE 1.   The Blackfoot River basin in western Montana showing treatment and reference sites and demarcations of the lower, middle, and upper Blackfoot River basin. Treatment sites (open squares with site numbers from 1 to 18) relate to restoration sites and techniques in Table 1 and fisheries response on Figure 2.

Small stream restoration techniques.—Small stream restoration in the Blackfoot River basin is an iterative multiscale process, whereby the scope and scale of restoration expands as information and stakeholder support are generated (Aitken 1997; Pierce et al. 2005). Each stream restoration project typically begins with a tributary assessment of fish populations and aquatic habitat conditions within the context of land uses, such as riparian timber harvest, livestock grazing, and irrigation practices. Projects are then prioritized based on native fisheries values (MBTRT 2000; USFWS 2010), water quality benefits, and the importance of tributary populations to the Blackfoot River, as well as funding and landowner interest in potential stream improvements (Aitken 1997; Pierce et al. 2005, 2008). Once a stream reach is selected for fisheries improvement, multiple restoration techniques are individually tailored (Table 1) to correct habitat impairments (Table 2).

Natural channel restoration techniques were employed in the most degraded streams to return them to geomorphically stable and natural states that are capable of maintaining habitat-forming processes. These methods incorporated bankfull theory (Dunne and Leopold 1978) and the core principles of natural channel design (Rosgen 1994, 1996, 2007), and relied on geomorphic indicators of the bankfull channel, mea-

sured reference reaches, and design validation using empirically derived regional curves of channel geometry for western Montana streams (Lawlor 2002). In the Blackfoot River basin, these methods further incorporated the placement of instream habitat features suited to the geomorphic potential (Rosgen 1996; Schmetterling and Pierce 1999), vegetative setting (Manning et al. 1989; Hansen et al. 1995), and local fisheries resource of the site (MBTRT 2000; Brown et al. 2001; Pierce et al. 2005).

Depending on the specific land-use conflicts with fisheries, most restoration projects also required retrofitting irrigation diversions with fish ladders and screening ditches to prevent fish losses within migratory corridors (e.g., Schmetterling et al. 2002; Pierce et al. 2003), while restoring instream flows to minimal flow standards using water leases or other voluntary methods (Tennant 1976; Wesche and Rechard 1980; MUSWC 2006). Because most of our stream improvement work is undertaken on private ranchland, treatment streams that supported intensive livestock grazing also required development of alternative riparian livestock grazing practices consistent with the maintenance of natural channel form and vegetative stability (Meehan 1991; Armour et al. 1994; Bengeyfield and Svoboda 1998).

TABLE 1. Summary of wild trout restoration techniques for 18 treatment streams. Stream name and identification (ID) refer to project locations in Figure 1 and summary of treatments in Table 2.

Stream name	Stream ID	Subbasin location	Project length (km)	Channel reconstruction	Instream habitat structure	Increase instream flow	Fish screens–ladders	Riparian grazing changes	Revegetation
Bear Creek	1	Lower	2.4	X	X			X	X
Gold Creek	2	Lower	4.8		X				
Blanchard Creek	3	Lower	1.8			X	X		
Cottonwood Creek	4	Middle	1.6			X	X		
Shanley Creek	5	Middle	1.3					X	
Chamberlain Creek	6	Middle	4	X	X	X	X	X	X
Pearson Creek	7	Middle	3.2	X	X	X		X	X
McCabe Creek	8	Middle	4	X	X	X	X	X	X
Warren Creek	9	Middle	1.3					X	
Jacobsen Spring Creek	10	Middle	5.2	X				X	X
Kleinschmidt Creek	11	Middle	4.5	X	X			X	X
Rock Creek	12	Middle	3.2	X	X			X	X
Murphy Spring Creek	13	Middle	4.8			X	X		
Nevada Spring Creek	14	Upper	7.1	X		X		X	X
Wasson Creek	15	Upper	4.5	X		X	X	X	X
Poorman Creek	16	Upper	1			X	X	X	X
Grantier Spring Creek	17	Upper	2.4	X	X	X		X	X
Snowbank Creek	18	Upper	0.8			X	X		

METHODS

Data collection and organization.—To determine the response trends of wild trout to small stream restoration in the Blackfoot River basin, we compiled fish population monitoring data on 18 treatment and 23 reference sites surveyed between 1989 and 2010 (Figure 1). Treatment surveys were located directly within restored reaches, whereas reference sites included a similar range of low- to midelevation small stream valleys where riparian and aquatic habitat were unaffected by direct human activities (Table 3). Reference sites included surveys for all years (1989–2010) in this study. Most reference sites were located in separate nearby streams ($n = 14$); however, nine were located on the same stream an average distance of 4.2 km from the treatment monitoring sites, and eight of these were located upstream from the treatment areas (Figure 1).

All treatment sites had at least 1 year of preproject fish population data, although only data from the year immediately preceding treatment was used in this study to standardize the analysis. In addition, each site had 5–21 years (mean = 12 years) of posttreatment monitoring data. For treatment streams with more than one reach-scale project ($n = 5$ streams), the project site with the most complete long-term data set was selected for this study.

Surveys of age-0 trout were completed at all monitoring sites; however, sampling efficiencies were often low or inconsistent for the purposes of generating population estimates. As a result,

we removed age-0 fish from the data set using length-frequency histograms, and used trout of age >1 to determine response trends in the analyses. For most population surveys at reference (54 of 76 sites) and treatment sites (144 of 155 sites), we estimated trout abundance using backpack electrofishing depletion techniques (Van Deventer and Platts 1989). For sites with only a single-pass intensive electrofishing survey (i.e., 11 treatment surveys and 22 reference surveys), estimates of abundance were calculated using a single-pass and multiple-pass linear regression equation derived from data in this study (i.e., abundance = 1.2206 (catch) + 1.8723, $r^2 = 0.91$, $P < 0.0001$) similar to Kruse et al. (1998).

Because of small sample sizes and an inability to reliably estimate the abundance of individual trout species in many sites, we categorized trout as native, nonnative, and total trout groups. Estimates of abundance were then calculated for each group as number of trout per linear stream meter (trout/m). We removed eight estimates at five sites from the analyses because of low capture probabilities (i.e., the 95% confidence interval [CI] of the estimate overlapped with zero). In the case of McCabe Creek, this included the removal of the pretreatment nonnative trout population estimate. As a result, we did not analyze McCabe Creek for trends in nonnative trout response. Prior to statistical analyses, all estimates of abundance were natural log ($\log_e$) transformed to meet assumptions of normality and homogeneity of variance. Before transformation, we added a value of "1" to

TABLE 2. Summary of stream pre- and posttreatment habitat conditions associated with trends in trout response. Channel type refers to Rosgen (1994) stream classification. Posttreatment water temperature refers to the maximum summer temperature recorded during the last monitoring year. NC = no change resulting from the treatment, ND = no data.

Stream name	Channel type		Width : depth ratio		Sinuosity		Pool area (%)		Maximum summer temperature (°C)		Minimum summer flow (m³/s)	
	Before	After	Before	After	Before	After	Before	After	Before	After	Before	After
Bear Creek	G	B	<9	13.3	1.1	1.3	4	66	17	17	0.06	NC
Gold Creek	B–C	B–C	20.3	NC	1.3	NC	1.5	13	17	NC	0.82	NC
Blanchard Creek	C	C	ND	NC	ND	1.1	ND	12	ND	25	0	0.085
Cottonwood Creek	C	C	19.5	NC	ND	NC	ND	NC	ND	20	0	0.22
Shanley Creek	C	C	ND	ND	ND	ND	ND	ND	ND	16	0.085	NC
Chamberlain Creek	C	C	ND	19.2	1.1	1.1	9	ND	ND	21	0.009	0.085
Pearson Creek	B	B–E	variable		1.1	1.1–1.3	ND	ND	ND	ND	0	0.028
McCabe Creek	G	B	3	9	1.2	1.3	ND	31	ND	13	0.11	0.22
Warren Creek	F	E	>20	13.7	ND	1.3	ND	45	ND	ND	ND	NC
Jacobsen Spring Creek	C	C	40	11	1.2	1.4	22	32	19	14	0.11	NC
Kleinschmidt Creek	C	E	10.8	2.8	1.1	1.4	28	53	18	15	0.43	NC
Rock Creek	C	E	55	6.2	1.1	1.7	3	15	21	15	0.82	NC
Murphy Spring Creek	B	B	ND	16.7	ND	NC	ND	NC	ND	ND	0.007	0.085
Nevada Spring Creek	C	E	22	3.2	1.4	1.7	51	71	25	18	0.17	0.28
Wasson Creek	F	E	3	0.7	1	1.5	ND	ND	22	18	0	0.02
Poorman Creek	C	C	ND	17.1	ND	NC	ND	NC	16	NC	0	0.085
Grantier Spring Creek	C	C	19.7	14.6	1.2	2.3	ND	34	ND	13	0.28	0.37
Snowbank Creek	C–B	C–B	NC	17.8	NC	1.2	NC	44	ND	13	0	0.11

TABLE 3. Comparison of physical channel features associated with both treatment and reference streams.

Stream group	Stream order [mode (range)]	Elevation (m) [mean (range)]	Bankfull area (m^2) [mean (range)]	Valley slope [mean (range)]
Treatment streams ($n = 18$)	2 (1–3)	1,279 (1,075–1,620)	2.0 (0.4–9.0)	0.019 (0.003–0.04)
Reference streams ($n = 23$)	2 (1–3)	1,320 (1,059–1,611)	2.1 (0.2–7.3)	0.026 (0.006–0.05)

each estimate to avoid generating a value of negative infinity when attempting to transform values of zero.

Analyses of trout response at individual reach scale.—We used a before–after study design to explore the individual trends of native, nonnative, and total trout groups for each of the 18 treatment sites (Table 4). We performed linear regressions of estimates on all monitoring years (1 year pretreatment and all monitoring years posttreatment) to determine trends in all treatment sites except for nonnative trout in McCabe Creek as noted above. Increases in trout abundance after treatment were considered significant if the slope of the trend line was significantly different from zero.

Analyses of trout response in all treatment and control sites.—To analyze the collective trend of total trout abundance across treatment sites, we used before–after and control–impact comparisons (Table 4). For these comparisons, we organized the treatment and reference data as follows. For treatment data, we averaged estimates at 1-year intervals from pretreatment surveys through a 12-year posttreatment monitoring period across 17 of 18 treatment sites. Grantier Spring Creek was not included in these analyses because posttreatment monitoring occurred at 3, 17, and 18 years posttreatment and thus did not fit the 5–12-year time frame associated with these analyses. Analyses of these overall trends did not extend beyond 12 years due to small sample size in the small number of sites with longer monitoring data sets.

We were not able to use paired sites, and the number of reference sites varied annually (range, 1–6) over the 22-year study period. Therefore, we used linear regression to test for trends in the reference sites across the region during the study period (1989–2010). We performed a linear regression of trout/m versus calendar year to test for trends in the reference sites over time. Since no trend was found, we averaged trout/m across all years for each reference site, and then across all reference sites to obtain a single nested average value for comparison with treatment data (see description of *t*-tests below). Variation in the nested average represents variation between sites but not across years. Additionally, we also calculated a single grand average value for all reference site data by collectively averaging all reference observations without organizing by years or site. Thus, the variance around this average incorporates both the spatial and temporal variance into a single estimate of variance for our comparison to treatment sites. This average is used for visualization of the reference data in Figure 2b. Finally for treatment sites, estimates of abundance were organized by year posttreatment and averaged across all streams.

To analyze the initial changes in total trout abundance before and after treatment, we used a paired sample Wilcoxon signed-rank test to compare the average trout/m in treatment streams at pretreatment and 3 years posttreatment ($n = 12$ sites; 6 of 18 sites did not have monitoring data at 3 years posttreatment). To examine the initial pattern of total trout response in treatment versus reference (control) sites, we performed two independent two-sample *t*-tests to compare average total trout abundance at both pretreatment and also 3 years posttreatment.

Analyses of subbasin scale trout response.—To explore spatial variation in the response trends of native and nonnative trout at a subbasin scale, we first sorted each site by location (i.e., lower, middle, and upper basins: Figure 1; Table 1) and then calculated the average trout/m of native and nonnative trout for each site with data at both pretreatment and 5 years posttreatment. Sample sizes for this comparison were $n = 3$, 6, and 4 sites for the lower, middle, and upper subbasins, respectively (Table 4). We chose 5 years posttreatment for this comparison owing to small sample sizes for monitoring years beyond 5 years posttreatment. To statistically compare changes in community composition within each subbasin, we performed a paired sample Wilcoxon signed-rank test between the proportions of native trout to wild trout per site at pretreatment and at 5 years posttreatment. All reach and subbasin scale statistical analyses were performed at the $P = 0.05$ level of significance using the computer programming language R (R Development Core Team 2009).

RESULTS

Reach-Scale Trout Response (Before–After Comparisons for Individual Sites)

Response patterns of total trout abundance varied widely among individual treatment sites (Table 5; Figure 2). Of the 18 sites individually analyzed, 15 sites showed positive trends in total trout abundance, of which seven were statistically significant. Conversely, the remaining three sites in this study (Blanchard, Pearson, and Grantier Spring creeks) declined during the monitoring period, but none of these declines were statistically significant.

Several patterns emerged when examining response trends in native and nonnative trout groups for individual sites (Table 5; Figure 2). Of the seven sites with significant increases in total trout abundance, four sites (Bear, Jacobsen Spring, Kleinschmidt, and Rock creeks) showed significant increases in nonnative trout abundance, and three (Murphy Spring, Nevada

TABLE 4.   Summary of reach- and subbasin-scale study design and methods of analyses.

Scale	Study group (number of sites)	Study design	Analysis method
Individual reach	Total trout ($n = 18$) Native trout ($n = 18$) Nonnative trout ($n = 17$)	Before–after	Linear regression
All streams	Total trout (treatment, $n = 12$)	Before–after	Paired Wilcoxon signed rank test
All streams	Total trout (treatment, $n = 18$) Total trout (reference, $n = 23$)	Before–after, control–impact (Before treatment vs reference, 3 years after treatment vs reference)	Independent two-sample t-test
Subbasin	Lower basin ($n = 3$) Middle basin ($n = 6$) Upper basin ($n = 4$)	Before–after	Paired Wilcoxon signed rank test

Spring, and Poorman creeks) displayed significant increases in native trout abundance. Statistically significant declines in both native and nonnative trout groups were observed at only two sites. Native trout declined significantly in Gold Creek concurrent with an increasing trend in nonnative trout. Conversely, nonnative trout decreased in Grantier Spring Creek concurrent with a significant increase in native trout. While increases in native trout were not significant in Cottonwood or Chamberlain creeks, data from both streams show native trout abundance increased quickly and remained elevated for several years fol-

lowing treatment (Figure 2). As shown in these examples, linear regressions can mask short-term nonlinear responses and reduce statistical rigor compared with best-fit regression models (e.g., Akaike information criterion models, Akaike 1974; Burnham and Anderson 2002).

Typically, individual treatments supported increases in the dominant trout species present before treatment. When examined at a basin-wide scale, increases in native trout generally occurred in the mid to upper basin, whereas, increases in nonnative trout occurred in the mid to lower basin. Interestingly,

TABLE 5.   Model results for linear regressions on total trout, native trout, and nonnative trout for each individual treatment stream. Total trout regression lines and data plots for trout groups are shown in Figure 2.

Stream name	Total trout			Native trout			Nonnative trout		
	Slope	P-value	r^2	Slope	P-value	r^2	Slope	P-value	r^2
Bear Creek	0.024	0.026	0.40	–0.003	0.249	0.13	0.025	0.023	0.42
Gold Creek	0.020	0.138	0.29	–0.008	0.004	0.73	0.026	0.060	0.42
Blanchard Creek	–0.016	0.293	0.14	0.001	0.180	0.21	–0.017	0.273	0.15
Cottonwood Creek	0.010	0.353	0.07	0.010	0.322	0.08	0.0005	0.787	<0.01
Shanley Creek	0.005	0.668	0.07	0.001	0.604	0.10	0.004	0.713	0.05
Chamberlain Creek	0.005	0.535	0.07	0.007	0.393	0.06	–0.004	0.043	0.32
Pearson Creek	–0.011	0.361	0.08	–0.010	0.394	0.07	–0.002	0.185	0.17
McCabe Creek	0.023	0.224	0.34	0.024	0.135	0.47	NA	NA	NA
Warren Creek	0.020	0.242	0.32	0.0004	0.944	<0.01	0.019	0.259	0.3
Jacobsen Spring Creek	0.016	0.043	0.68	–0.001	0.158	0.43	0.017	0.038	0.70
Kleinschmidt Creek	0.033	0.006	0.63	0.001	0.213	0.19	0.033	0.007	0.62
Rock Creek	0.015	0.008	0.60	0.001	0.551	0.05	0.015	0.011	0.57
Murphy Spring Creek	0.026	<0.001	0.99	0.021	<0.001	0.95	0.003	0.319	0.46
Nevada Spring Creek	0.057	0.002	0.78	0.060	0.002	0.73	–0.003	0.864	0.00
Wasson Creek	0.023	0.350	0.15	0.025	0.298	0.18	–0.002	0.858	0.01
Poorman Creek	0.029	0.030	0.64	0.011	0.045	0.59	0.020	0.064	0.53
Grantier Spring Creek	–0.016	0.215	0.62	0.013	0.020	0.96	–0.028	0.071	0.86
Snowbank Creek	0.076	0.135	0.47	0.076	0.135	0.47	NA	NA	NA
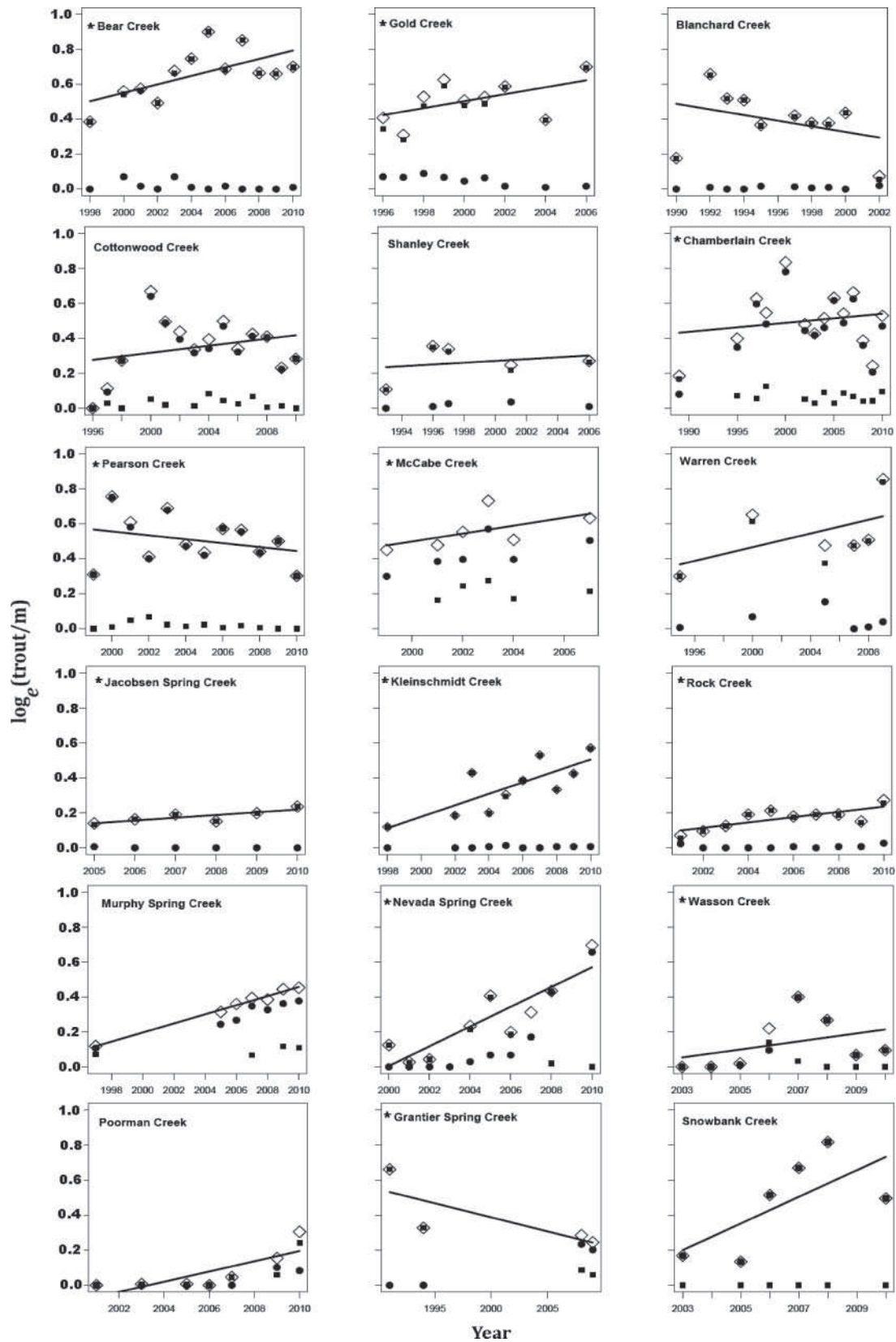
FIGURE 2. Wild trout response for 18 individual treatment streams, 1989–2010. Diamonds represent estimates of total trout abundance, circles represent estimates of native trout abundance, and squares represent estimates of nonnative trout abundance. Black line represents the linear trend line for total trout abundance. The first year on the *x*-axis denotes the pretreatment year. An asterisk (*) denotes a stream with active instream habitat treatments.
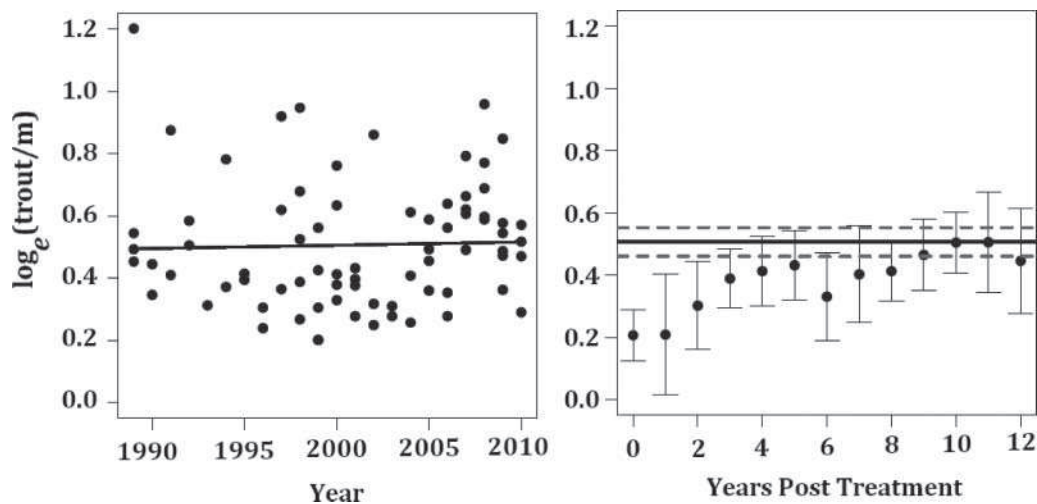
FIGURE 3.   (**a**) Estimates of total trout abundance at reference sites by calendar year. Linear regression analysis indicates a long-term stable trend with a slope not significantly different from zero during the study period (slope = 0.001, $P = 0.78$). (**b**) Average total trout abundance by years posttreatment. The solid black line represents the grand average of total trout abundance for all monitoring observations in reference sites (0.65 trout/m). Gray dashed lines represent the 95% confidence interval around the reference average (0.61–0.69 trout/m). Note: this grand average incorporates both year-to-year and stream-to-stream variation in the reference data set. Grantier Spring Creek is not included in these data.

three sites located in the mid to upper basin (Chamberlain, Nevada Spring, and Grantier Spring creeks) were exceptions with posttreatment shifts in community composition from nonnative to native trout. Additionally, posttreatment monitoring detected individual native trout species (i.e., Westslope Cutthroat Trout or Bull Trout) in four treatment reaches in the mid to upper basin (Cottonwood, Wasson, Grantier Spring, and Snowbank creeks) where they were not detected during pretreatment population surveys.

Reach-Scale Trout Response (Before–After and Control–Impact with Aggregate Site Data)

Reference sites showed wide variation but no increasing or decreasing trend in average abundance throughout the 1989–2010 monitoring period ($r^2 = 0.003$, $P = 0.78$; Figure 3a), indicating that annual variation in trout abundance is not confounding the response of trout at treatment sites. Before restoration, total abundance across all sites was significantly lower than at reference sites ($P = 0.0001$), with an average of 0.19 trout/m (95% CI = 0.12–0.30) at pretreatment sites compared with 0.62 trout/m (95% CI = 0.54–0.71) for reference sites. A paired comparison between trout/m at pretreatment and 3 years posttreatment showed a significant increase in average total trout/m in treatment streams ($P = 0.01$). Additionally, by 3 years posttreatment, average abundance in treatment sites had reached 0.47 trout/m (95% CI = 0.35–0.63) and were no longer statistically different from reference sites ($P = 0.12$). The grand average for all reference observations is 0.65 trout/m (95% CI = 0.61–0.69). Following this initial increase, total trout densities for all treatment sites remained elevated near the average reference between 4 and 12 years posttreatment (Figure 3b).

Subbasin Scale Trout Response

The analysis of community composition showed large differences among lower, middle, and upper subbasins with native trout comprising 6% of the pretreatment trout community in the lower basin compared with 58% in the upper basin (Figure 4). The lower and middle basins showed little to no change in the proportion of native trout to wild trout in treatment sites between pre- and 5 years posttreatment ($P = 1.0$ and 0.86, respectively). Conversely, tributaries in the upper basin increased from 58% native trout pretreatment to 77% native trout at 5 years posttreatment, however, this change was not statistically significant ($P = 0.37$).
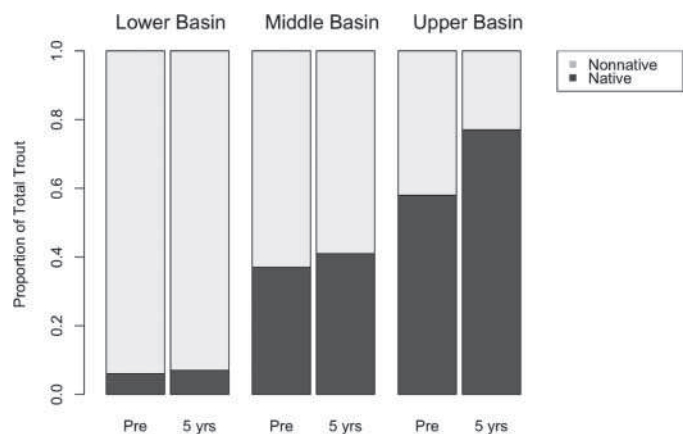
FIGURE 4.   Average proportion of native and nonnative trout in restored sites at pretreatment (pre) and 5 years posttreatment across the three subbasins of the Blackfoot River watershed.

DISCUSSION

Blackfoot River Restoration: a Riverscape Conservation Strategy

Though reach-scale restoration projects are ideally evaluated using highly controlled experimental studies (Roni 2005), such studies often fail to accommodate constraints of applied fisheries field work and the iterative nature of multiscale riverscape conservation endeavors (Aitken 1997; Fausch et al. 2002; Roni 2005). In our study area, project tributaries were identified with basin-scale fisheries and habitat assessments (e.g., Peters 1990; Pierce et al. 1991, 1997, 2005, 2008) and biotelemetry studies emphasizing the spawning life histories of free-ranging wild trout (Swanberg 1997; Schmetterling 2000, 2001, 2003; Pierce et al. 2007, 2009). With this information, restoration treatments were intended to ameliorate larger-scale human disturbance in order to ultimately meet management goals that emphasize the recovery of fluvial and native trout of the Blackfoot River (e.g., Aitken 1997; Pierce et al. 2005; Fausch et al. 2009; USFWS 2010). Monitoring efforts for small sites in this study were carried out pragmatically with emphasis on landowner education and adaptive management to help ensure sustainability in areas of intensive land use. Given this basin-scale management approach and unique nature of each treatment, fisheries data sets in this study were standardized and analyzed regionally against reference-reach data in order to elicit broader trends. Though limited in its ability to examine individual treatments, the strength of this study lies in the long-term nature of the data set, a large number ($n = 17$) of replicate sites, and strong spatial trends that help identify focal areas for native trout recovery.

Restoration Techniques and Reach-Scale Response

In our study, average total trout abundance for 17 sites increased rapidly, approached reference conditions about 3 years posttreatment, and remained elevated near reference conditions (Figure 3b). The initial rapid increase in total trout abundance can be attributed to several projects involving instream flow enhancement (e.g., Blanchard, Cottonwood, and Snowbank creeks) or enhanced fish passage, entrainment reduction, or both (Figure 2; Table 2). As intended, these projects encouraged short-term redistribution of fish older than age 1 into treatment reaches as described elsewhere (Gowan and Fausch 1996; Roni and Quinn 2001). Yet, examination of all survey data from treatments sites also revealed increased production of age-0 trout as well as diverse community-level responses in the posttreatment environments, depending on the treatment and its location within the basin. As one example, increased downstream recruitment of juvenile Rainbow Trout from an upstream population was the target of the treatment in Blanchard Creek, the lowermost instream flow project in this study (Pierce et al. 1997). Here, Rainbow Trout of age >1 showed a rapid and sustained increase. Likewise, estimates of age-0 Rainbow Trout abundance also increased from an average of 0.17 trout/m (range, 0.03–0.38) during the first 3 years of monitoring (1989–1992)

to a 7-year average of 0.94 trout/m (range, 0.46–1.57) between 1992 and 2002 (R. Pierce, unpublished data). In addition, five native fishes (Westslope Cutthroat Trout, Mountain Whitefish *Prosopium williamsoni*, Northern Pikeminnow *Ptychocheilus oregonensis*, Longnose Dace *Rhinichthys cataractae*, Largescale Sucker *Catostomus macrocheilus*, and sculpin *Cottus* sp.) were present in posttreatment surveys but were not detected in pretreatment surveys (Pierce et al. 1997). As a second example, the Snowbank Creek instream flow project (i.e., the uppermost treatment) was intended to foster a community response by restoring flows and habitat connectivity with a downstream tributary. Here, monitoring showed a sharp initial increase in Westslope Cutthroat Trout of age >1 along with the upstream expansion of Bull Trout into the project area, which included successful spawning (i.e., redds and age-0 fish present) within 3 years of treatment (Pierce et al. 2011; U.S. Forest Service, unpublished data;).

Though habitat improvements can clearly increase salmonid abundance, biomass, and species richness (e.g., Hunt 1976; Baldigo et al. 2008; White et al. 2011), movement of individuals into areas of habitat improvement may, in some cases, provide limited biological benefits (e.g., growth and enhanced juvenile production) according to Gowan and Fausch (1996). However, the Gowan and Fausch (1996) findings were reported from small, high-elevation streams supporting a simple nonnative trout community with no quantitative pretreatment assessment of life histories or limiting factors. Other studies indicate that movement to areas of improved habitat relate to competition for space or foraging areas (White et al. 2011), whereby dominant fish vacate habitat that is later occupied by subdominant fish (Hansen and Closs 2009), ultimately leading to an overall increase in population abundance. In our study area, restoration focused on lower reaches of the tributary system where habitat fragmentation, degradation, and simplification have diminished fish communities (Peters 1990; Pierce et al. 2005, 2007), including spawning and rearing and migratory habitat required for free-ranging trout of the Blackfoot River (Swanberg 1997; Schmetterling 2000, 2001; Pierce et al. 2007, 2009). In these areas, restoration-induced movement can lead to higher abundance over the long term, facilitate community-level recolonization processes, and promote the recovery of imperiled native trout depending on the individual treatment.

In addition to irrigation-related treatments, we implemented natural channel design techniques along with riparian grazing changes on the most treatment sites (Table 1). Compared with habitat enhancement techniques that rely heavily on structures (Roni 2005; Roni et al. 2008; Stewart et al. 2009), natural channel design integrates the geomorphic, hydrologic, and vegetative setting of the site and its valley in a manner that emulates natural (e.g., reference) channel conditions (Rosgen 1996; Baldigo et al. 2008; this study). Natural channel design methods are more natural and resilient than traditional methods (Schmetterling and Pierce 1999; Baldigo et al. 2008, 2010; Whiteway et al. 2010), yet few fisheries studies have documented the efficacy of this

approach. However, one study (Baldigo et al. 2008) demonstrated that both community biomass and species richness increased following natural channel design treatments over short-term (i.e., <5 years) monitoring periods. Consistent with those findings, 9 of 11 active treatments in our study showed positive trends in total trout abundance over a 6–21-year monitoring period (Figure 2). For certain sites requiring full reconstruction (e.g., Bear, Kleinschmidt, and Nevada Spring creeks), estimates of total trout abundance showed continuous linear increases 10–12 years posttreatment (Figure 2). With the exception of Gold Creek, most (8 of 9) active treatments with positive trends also required multiple techniques (Table 1). These incremental long-term increases contrast with the rapid increases observed in instream flow projects (this study), as well as with other studies that suggest about 5 years is required for the full effects of habitat manipulation alone to be realized (Hunt 1976; Whiteway et al. 2010).

To effectively apply a restoration-based strategy in areas of multiple land use, land-use practices must be consistent with processes that form and maintain natural aquatic and riparian habitat (e.g., Meehan 1991; Schmetterling and Pierce 1999; Baldigo et al. 2008). Of the treatment sites described in this study, 17 of 18 sites applied riparian grazing or irrigation methods, or both, to reverse human-induced degradation of wild trout habitat (Table 1). Depending on specific habitat objectives, various types of pre- and posthabitat monitoring (e.g., water temperature, flow, channel measurements) were applied to individual treatments (Table 2). These habitat data indicate trends toward natural geomorphic stability (Rosgen 1996), higher sinuosity, more pool habitat, cooler summer water temperatures, and higher summer flows following treatments. Under these conditions, total trout abundance increased at 15–18 sites; however, declines occurred at three sites (Blanchard, Pearson, and Grantier Spring creeks). For Blanchard and Pearson creeks, estimates of total trout abundance increased initially, but then declined after the return of dewatering practices and livestock incursions. Interestingly, total trout abundance declined in Grantier Spring Creek after treatment despite consistent instream flows and vegetative recovery of riparian areas. In this case, we also observed a community shift toward native trout 17–19 years postrestoration as well as an increase in total trout biomass relative to abundance. Given these results, we consider the Grantier Spring Creek response a positive step toward native trout conservation.

Subbasin Trends and the Response of Native Trout

Although the wild trout response varied widely among individual treatments, strong differences in trout composition were also revealed at a basin scale. Similar to other observations across the Rocky Mountains of western North America (Paul and Post 2001; Wood and Budy 2009), we observed a strong increasing trend towards native trout in the up-valley direction (Figure 4). More specifically, treatments generally favored nonnative Rainbow Trout and Brown Trout in low elevations and the valley bottom of the Blackfoot River basin; whereas, treatments in the foothills of the mid to upper basin generally favored Westslope Cutthroat Trout as the prevalent native trout.

Though most restoration activities favored the prevalent pretreatment salmonid, community-level shifts from nonnative to native trout occurred within three tributaries (Grantier Spring, Nevada Spring, and Chamberlain creeks) located in the mid to upper basin. Contrary to widespread reports of Westslope Cutthroat Trout displacement by Brook Trout and Brown Trout (Griffith 1972; Peterson et al. 2004; Shepard 2004), these results indicate that Westslope Cutthroat Trout can expand population abundance at the reach scale in the presence of nonnative trout competitors under certain favorable conditions. Community shifts from fishes with a broad range of environmental tolerances to species with more specific requirements have been observed through riparian restoration actions (Behnke 1992; Baldigo et al. 2008). Yet beyond the data presented here, we are unaware of any studies showing restoration-related shifts from nonnative Brook Trout and Brown Trout to native Cutthroat Trout without the active removal of these nonnative trout. In the case of Grantier Spring Creek, subsequent surveys documented evidence of spawning (i.e., redds) and the presence of age-0 to adult Westslope Cutthroat Trout associated with this expansion (Pierce et al. 2010). We hypothesize these community shifts relate to the prevailing regional influences that favor native trout (e.g., Paul and Post 2001; Wood and Budy 2009), short distances to source populations, and restoration techniques that emulate the natural conditions to which native trout have adapted, including reductions in water temperatures (e.g., 4°C, Nevada Spring Creek project; Table 2). In the case of both Grantier Spring and Nevada Spring creeks, the expansion of native trout was traced to nearby source populations in upstream tributaries based on genetic assignment tests (K. Carim, unpublished data).

Though this study emphasizes the response of trout to reach-scale restoration in small tributaries, many reach-scale projects were specifically undertaken to promote the recovery of fluvial native trout of the Blackfoot River. Chamberlain Creek is an example of this. Here, channel degradation in the 1980s led to a 94% reduction in Westslope Cutthroat Trout abundance between upstream reference sites and downstream disturbed areas, as well as a loss of migratory connection between Chamberlain Creek and the Blackfoot River by instream dams, diversions, and dewatering (Peters 1990; Pierce 1991). Following treatment, surveys showed that age-0 Westslope Cutthroat Trout increased from a pretreatment estimate of zero to a long-term (13-year) average of 0.83 trout/m (R. Pierce, unpublished data). Moreover, 7 years after treatment, biotelemetry confirmed migratory reconnection, as 73% of fluvial Westslope Cutthroat Trout spawners radio-tagged in the Blackfoot River between Gold Creek and the North Fork (a distance of 65 km) ascended Chamberlain Creek to access spawning areas within and upstream from the treatment reach (Schmetterling 2000, 2001).

Because regulations governing the harvest of trout have remained consistent for native and nonnative trout in small streams trout since 1990, it appears unlikely direct angling pressure

strongly influenced reach-scale trends in this study. This appears evident given a common pattern in which trout abundance increases soon after habitat treatments (Figure 3b), which developed 8 years after angling regulation changes were enacted. Most treatment (and reference) reaches in this study are, in fact, located on small, brushy streams that provide limited access and support very little angling pressure (MFWP 2011).

Monitoring and Adaptive Management

While a majority of reach-scale projects showed positive trends in the abundance of wild trout, we believe sustained increases were strongly influenced by a long-term monitoring presence followed by adaptive management on most treatments. Adaptive management eventually involved 10 of 18 treatments in this study and included (1) active channel work on 2 of 11 sites that initially received this treatment, (2) corrections to design or maintenance deficiencies with fish ladders or fish screens on six of eight sites, and (3) attempts to reduce livestock-induced streambank damage on 7 of 13 grazing-related projects. The high incidence of irrigation adjustments reflects primarily technological advancements to reduce maintenance of fish ladders and fish screens. Whereas, the high incidence of grazing-related adjustments reflects the inherent complexities and reduced probability of success of riparian grazing systems compared with livestock exclusion (Platts 1991; Roni 2005). In our experience, successful riparian grazing systems require a clear but undersupported need for consistent and specific monitoring to ensure the recovery of both riparian function and instream trout habitat (Platts and Rinne 1985; Platts 1991; Bengeyfield and Svoboda 1998).

Though long-term monitoring information is one of the most pressing needs in restoration ecology (Roni 2005), monitoring and evaluations are rarely applied (Bernhardt et al. 2005; Reeve et al. 2006; Baldigo et al. 2010). In our study area, monitoring has proven to be critical to measures of effectiveness, but equally important in areas of multiple land use is a monitoring and evaluation process that improves restoration techniques and fosters communication and working relationships among individual landowners and stakeholder groups. This strengthening of communication ultimately increases the long-term success and sustainability of improved fisheries while enabling the recovery of imperiled native trout on private lands. This process is particularly important because stream restoration on private lands is considered vital (Aitken 1997; Pierce et al. 2007) but inherently complex and challenging to effectively apply in the absence of consistent monitoring presence.

Conclusions

Though no single management tool can fully correct problems afflicting wild salmonids, reach-scale restoration activities on small streams have improved habitat conditions and the status of wild trout in tributaries of the Blackfoot River over the past 20 years. Our evaluation shows that a majority of sites displayed sustained increases in total trout abundance following restoration activities. Furthermore, projects on 9 of the 18 sites (Cottonwood, Chamberlain, McCabe, Murphy Spring, Nevada Spring, Wasson, Poorman, Grantier Spring, and Snowbank creeks), all located in the mid to upper basin, are helping managers meet their goals of increasing stocks of native trout. As stream processes and characteristics return to a more natural condition, it also appears that some salmonid communities in the mid to upper basin area are shifting towards native trout assemblages, which also promotes life history diversity and metapopulation function within the Blackfoot River.

Where restoration failed to sustain initial population increases, this was usually linked with the return of human impacts to the stream environment. Because the recovery of coldwater fisheries relates to a broad range of both ecological and social uncertainties through the entire restoration process, strategic planning at a subbasin scale, stakeholder collaboration, dedicated monitoring, and adaptive management continue to define both the effectiveness and sustainability of wild trout restoration in the Blackfoot River basin.

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Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2011-2012.

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Total Number Captured	Number Captured 1st Pass	YOY Captured 1st Pass	Range of Lengths (in)	Mean Length (in)	CPUE YOY	CPUE Age 1+
Anaconda Creek	0.5	15N,6W,22C	2-Aug-11	372	CT	126	97	3	1.9 - 8.7	4.6	0.8	25.3
					EB	1	1	0	8	8	0.0	0.3
			13-Aug-12	542	CT	162	136	0	2.1 - 7.7	4.6	0.0	25.1
					EB	3	2	1	2.0 - 7.4	3.8	0.2	0.2
Arrastra Creek	0.7	14N,10W,29B	8-Sep-11	407	CT	30	23	2	2.2 - 7.5	7.8	0.5	5.2
					DV	1	1	0	11	11.0	0.0	0.2
					LL	41	32	6	2.2 - 11.7	5.2	1.5	6.4
					EB	56	42	4	2.4 - 9.0	5.2	1.0	9.3
					MWF	8	8	8	3.4 - 3.9	3.6	2.0	0.0
					Sculpins	abundant						
Ashby Creek	2.7	13N,16W,26D	1-Aug-11	513	CT	32	16	0	3.0 - 8.6	5.4	0.0	3.1
					EB	4	3	0	4.5 - 8.3	5.9	0.0	0.6
			30-Jul-12	513	CT	68	58	0	3.5 - 10.5	5.6	0.0	11.3
					EB	6	3	1	3.2 - 6.8	4.4	0.2	0.4
	4	12N,16W,3A	1-Aug-11	300	CT	56	42	0	2.7 - 8.7	4.7	0.0	14.0
					EB	3	2	1	2.0 - 5.3	4	0.3	0.3
					Spotted frogs & Tail frogs							
			30-Jul-12	340	CT	39	32	0	2.8 - 8.0	5.4	0.0	9.4
					EB	1	1	0	9.5	9.5	0.0	0.3
Bear Creek (lower river tributary)	1.1	13N,16W,18B; 13N,16W,7C	1-Aug-11	374	CT	2	2	0	3.9 - 5.7	4.8	0.0	0.5
					RB	27	23	0	1.3 - 7.7	4.7	0.0	6.1
					LL	18	16	0	4.1 - 11	6.7	0.0	4.3
					EB	7	7	1	2.1 - 6.7	5.4	0.3	1.6
					Sculpins	abundant						
					Painted turtle	present						
Brazil Creek	0.2	12N,10W,10D	29-Aug-11	300	CT	17	17	9	1.6 - 7.1	4.0	3.0	2.7
					Sculpins	common						
			4-Sep-12	300	CT	139	127	30	2.0 - 7.6	3.8	10.0	32.3
Wade Stits irrigation ditch at 0.3			29-Aug-11	210	No fish found							
Chamberlain Creek	0.1	15N,13W,32A	12-Sep-11	358	CT	86	72	12	1.9 - 8.0	4.4	3.4	16.8
					LL	8	7	1	3.2 - 7.1	5.7	0.3	1.7
					RB	2	2	0	4.5 - 4.8	4.7	0.0	0.6
					Sculpins	common						
			12-Sep-12	358	CT	71	62	25	1.7 - 8.6	3.4	7.0	10.3
					LL	5	4	4	3.2 - 3.5	3.3	1.1	0.0
					RB	1	1	0	4.4	4.4	0.0	0.3
					Sculpins	abundant						

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Total Number Captured	Number Captured 1st Pass	YOY Captured 1st Pass	Range of Lengths (in)	Mean Length (in)	CPUE YOY	CPUE Age 1+
Copper Creek	6.2	15N,8W,9A	22-Aug-11	536	DV	39	35	23	2.0 - 12.2	3.5	4.3	2.2
					CT	54	38	0	2.6 - 15.4	4.8	0.0	7.1
					Sculpins	common						
			20-Aug-12	535	DV	28	19	12	2.2 - 5.3	3.4	2.2	1.3
					CT	70	55	1	1.7 - 14.2	5.0	0.2	10.1
					Sculpins	common						
Cottonwood Creek	12.0	16N,14W,24D	21-Sep-11	515	CT	51	38	0	1.0 - 12	3.9	0.0	7.4
					EB	3	3	0	4.3 - 8.5	6.4	0.0	0.6
					Sculpins	common						
			17-Sep-12	515	CT	55	49	2	1.7 - 12.3	5.1	0.4	9.1
					EB	7	7	5	2.8 - 6.6	3.8	1.0	0.4
					Sculpins	common						
Enders Spring Creek	0.1	14N,11W,6B	24-Aug-11	381	EB	43	29	7	1.8 - 9.2	4.4	1.8	5.8
					LL	4	2	0	4.6 - 6.5	5.3	0.0	0.5
					Sculpins	common						
Gleason Creek below culvert	0.1	12N,8W,22B	3-Oct-12	162	CT	44	44	2	1.5 - 9.9	4.3	1.2	25.9
					Sculpins	present						
above culvert	0.15	12N,8W,22B	3-Oct-12	690	CT	29	29	2	1.5 - 8.3	4.3	0.3	3.9
					Sculpins	present						
Gold Creek	1.9	14N,16W,30D	31-Aug-11	400	RB*	66	52	5	1.9 - 9.8	5.4	1.3	11.8
					LL	49	34	0	4.3 - 12.4	6.3	0.0	8.5
					Sculpins	abundant	RB YOY's	abundant	Spotted frog	observed		
Grantier Spring Creek	1.0	14N,9W,25A	20-Jul-11	507	CT	14	11	0	3.2 - 19.1	8.1	0.0	2.2
					LL	47	42	36	2.1 - 7.6	3.1	7.1	1.2
					EB	13	10	4	2.5 - 9.1	5.0	0.8	1.2
					Sculpins	abundant						
			8-Aug-12	507	CT	27	20	0	4.2 - 15.9	7.3	0.0	3.9
					LL	21	21	11	2.6 - 12.4	5.4	2.2	2.0
					EB	4	3	1	2.6 - 7.5	4.3	0.2	0.4
					Sculpins	abundant						
Hoyt Creek	1.2	15N,12W,19C	31-Aug-11	300	CT	1	0	0	4.9	4.9	0.0	0.0
					EB	8	6	1	3.9 - 7.8	5.0	0.3	1.7
					Sculpins	present						
	2.7	15N,12W,29C	31-Aug-11	300	No fish found							
	4.3	15N,12W,28C	31-Aug-11	300	EB	11	8	0	3.7 - 7.5	5.3	0.0	2.7
					Sculpins	present						

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Total Number Captured	Number Captured 1st Pass	YOY Captured 1st Pass	Range of Lengths (in)	Mean Length (in)	CPUE YOY	CPUE Age 1+
Jacobsen Spring Creek	0.6	14N,12W,1CD	9-Aug-11	525	ONC	1	1	1	1.2	1.2	0.2	0.0
					LL	4	3	2	2.1 - 8.5	4.7	0.4	0.2
					EB	69	53	40	1.6 - 7.8	3.2	7.6	2.5
					Sculpins	common						
			22-Aug-12	460	RB	1	1	1	3.7	3.7	0.2	0.0
					DV	1	0	0	6.7	6.7	0.0	0.0
					LL	3	2	0	3.8 - 11.8	7.0	0.0	0.4
					EB	56	45	21	2.2 - 7.8	3.9	4.6	5.2
					Sculpins	common						
Kleinschmidt Creek	0.5	14N,11W,5C	21-Aug-12	355	LL	31	25	4	2.2 - 14.8	8.2	1.1	5.9
					EB	2	2	2	2.5 - 3.0	2.8	0.6	0.0
					Sculpins	abundant						
	0.8	14N,11W,5C	21-Aug-12	500	CT	1	0	0	7.7	7.7	0.0	0.0
					LL	70	49	11	2.0 - 15.3	7.3	2.2	7.6
					RB	1	1	0	7.1	7.1	0.0	0.2
					EB	13	9	4	2.8 - 11.6	6.1	0.8	1.0
					Sculpins	abundant						
Lincoln Spring Creek	3.8	14N,9W,13D	17-Aug-11	396	CT	1	0	0	10.9	10.9	0.0	0.0
					LL	18	13	0	4.8 - 15.4	8.9	0.0	3.3
					EB	16	9	5	2.2 - 9.5	4.6	1.3	1.0
					Sculpins	common						
			14-Aug-12	385	CT	1	0	0	5.8	5.8	0.0	0.0
					LL	25	17	0	4.3 - 14.3	9.2	0.0	4.4
					EB	10	7	1	2.6 - 10.4	6.8	0.3	1.6
					Sculpins	common						
Liverpool Creek	2.7	14N,8W,6B	16-Aug-11	300	CT	14	14	0	2.1 - 5.4	3.1	0.0	4.7
Irrigation ditch at stream mile 2.8	0.1	14N,8W,6B	16-Aug-11	258	CT	14	14	0	2.1 - 4.3	2.5	0.0	5.4
	3	14N,8W,6B	16-Aug-11	180	CT	21	21	0	2.2 - 6.0	4.0	0.0	11.7
Monture Creek	12.9	16N,12W,29B	31-Jul-12	795	DV	55	55	47	2.3 - 5.5	3.0	5.9	1.6
					CT	13	13	2	2.9 - 6.0	3.7	0.3	0.0
					EB	52	52	19	2.0 - 6.1	3.8	2.4	1.0
					Sculpins	abundant						
Murphy's Spring Creek	0.6	15N,11W,21B	30-Aug-11	385	DV	13	10	0	4.0 - 5.0	4.6	0.0	2.6
					CT	41	31	0	2.4 - 7.3	4.6	0.0	8.1
					EB	13	11	1	2.0 - 6.3	3.9	0.3	2.6
					Sculpins	common		Spotted frogs	present			
			28-Aug-12	385	DV	2	0	0	4.3 - 7.2	5.7	0.0	0.0
					CT	43	40	7	1.7 - 7.6	4.3	1.8	8.6
					EB	4	3	0	3.9 - 6.9	5.1	0.0	0.8
					Sculpins	common	Boreal toad	observed	Tadpoles	observed		

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Total Number Captured	Number Captured 1st Pass	YOY Captured 1st Pass	Range of Lengths (in)	Mean Length (in)	CPUE YOY	CPUE Age 1+
Nevada Creek												
	0.0 - 4.9	13N,11W,7CD, 18A & 8CD	14-Apr-11	24816	CT	3	3	0	5.6 - 13.5	10.7	0.0	0.0
					DV	1	1	0	19.1	19.1	0.0	0.0
					RB	1	1	0	11.3	11.3	0.0	0.0
					LL	17	17	0	9.4 - 19.4	13.3	0.0	0.1
					MWF, NPM & LSS	present						
	5.0 - 6.3	13N,11W,9C & 8D	20-Sep-11	6500	CT	45	45	0	7.8 - 16.3	11.1	0.0	0.7
					DV	1	1	0	19.9	19.8	0.0	0.0
					RB	3	3	0	5.1 - 12.1	7.6	0.0	0.0
					LL	57	57	0	4.6 - 18.2	9.7	0.0	0.9
					MWF & LSS	common						
			25-Sep-12	6500	CT	50	27	0	8.3 - 16.3	11.4	0.0	0.4
					DV	1	1	0	15.1	15.1	0.0	0.0
					RB	1	1	0	10.8	10.8	0.0	0.0
					LL	100	66	0	5.4 - 17.2	9.6	0.0	1.0
					MWF & LSS	common						
"Wade Stits project"	29	12N,10W,11C	21-Sep-11	3695	CT	53	53	0	5.5 - 15.7	10.0	0.0	1.4
					RB	50	50	2	3.2 - 20.4	11.0	0.1	1.3
					LL	17	17	0	7.6 - 20.6	10.2	0.0	0.5
					EB	9	9	0	4.4 - 7.1	5.7	0.0	0.2
					Sculpins	common	MWF, RSS, YP LSS & LNS		observed			
			10-Sep-12	3695	CT	52	52	0	6.2 - 15.7	10.4	0.0	1.4
					RB	135	135	10	3.1 - 17.9	8.6	0.3	3.4
					LL	19	19	0	8.9 - 21	13.6	0.0	0.5
					EB	11	11	0	4.7 - 10.6	7.9	0.0	0.3
					Sculpins	common	MWF, RSS, YP LSS & LNS		observed			
North Fork Blackfoot River												
Lund Ditch at N.F.mile 15.5	0.1	15N,11W,2C	9-Aug-12	273	DV	8	8	8	1.9 - 2.7	2.2	2.9	0.0
In ditch -downstream of fish screen					Sculpins	present						
between headgate and fish screen			9-Aug-12	323	DV	24	24	24	1.9 - 2.7	2.2	7.4	0.0
					CT	1	1	0	6.1	6.1	0.0	0.3
					Sculpins	present						
Fish by-pass canal			9-Aug-12	188	DV	13	13	13	1.9 - 3.1	2.5	6.9	0.0
					CT	2	2	1	3.8 - 7.1	5.5	0.5	0.5
					Sculpins	present						
North Fork Blackfoot River	17.2	16N,11W,35B	8-Aug-12	223	DV	34	34	28	1.9 - 7.0	3.1	12.6	2.7
small side channel below USFS bridge					CT	2	2	0	5.7 - 6.7	6.2	0.0	0.9
					RB	2	2	1	2.9 - 5.1	4.0	0.4	0.4
					Sculpins	common						

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Total Number Captured	Number Captured 1st Pass	YOY Captured 1st Pass	Range of Lengths (in)	Mean Length (in)	CPUE YOY	CPUE Age 1+
Park Creek												
above diversion	1.1	14N,9W,1B	17-Aug-11	375	No fish observed							
Irrigation ditch at stream mile 1.4	1.4	14N,9W,1B	17-Aug-11	165	CT	10	10	0	2.9 - 5.7	4.4	0.0	6.1
	0.1	14N,9W,1B	17-Aug-11	195	CT	4	4	0	3.1 - 7.0	5.2	0.0	2.1
Pearson Creek	0.5	15N,13W,33D	7-Sep-11	300	CT	16	13	0	5.2 - 8.3	6.4	0.0	4.3
					EB	3	3	3	3.6 - 3.7	3.6	1.0	0.0
			12-Sep-12	300	CT	13	13	0	5.0 - 9.9	7.2	0.0	4.3
					EB	6	4	1	3.5 - 6.6	5.0	0.3	1.0
	1.1	14N,13W,3B	7-Sep-11	370	CT	66	52	0	3.0 - 8.1	4.8	0.0	14.1
			12-Sep-12	370	CT	55	46	0	3.4 - 9.3	5.1	0.0	12.4
					EB	6	5	5	2.8 - 3.5	3.3	1.4	0.0
Poorman Creek	1.3	14N,9W,36A	10-Aug-11	490	CT	12	9	0	3.6 - 7.3	5.6	0.0	1.8
					LL	51	31	6	2.0 - 8.0	4.4	1.2	5.1
					Sculpins	present						
			15-Aug-12	510	CT	14	11	0	2.8 - 11.5	5.2	0.0	2.2
					LL	26	19	4	2.3 - 11.7	5.1	0.8	2.9
					EB	3	3	1	2.8 - 8.7	5.6	0.2	0.4
					Sculpins	common						
	1.5	14N,9W,36A	10-Aug-11	270	CT	2	1	0	3.5 - 6.1	4.8	0.0	0.4
					LL	10	5	0	2.0 - 11.6	5.0	0.0	1.9
					Sculpins	present						
			15-Aug-12	270	CT	5	3	0	1.7 - 9.5	4.7	0.0	1.1
					LL	9	7	0	5.2 - 16.6	9.1	0.0	2.6
					Sculpins	present						
Rock Creek	1.6	14N,11W,5A	17-Sep-12	510	LL	18	15	2	3.0 - 12.4	7.5	0.4	2.5
					EB	17	11	9	2.6 - 7.9	3.9	1.8	0.4
					Sculpins	common						
	3.9	15N,11W,35B	9-Aug-12	366	CT	3	3	0	6.0 - 7.2	6.5	0.0	0.8
					RB	1	1	0	8	8.0	0.0	0.3
					LL	1	1	0	5.9	5.9	0.0	0.3
					EB	9	9	1	3.7 - 7.2	6.3	0.3	2.2

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Total Number Captured	Number Captured 1st Pass	YOY Captured 1st Pass	Range of Lengths (in)	Mean Length (in)	CPUE YOY	CPUE Age 1+
Sauerkraut Creek	2.9	13N,9W,5D	15-Aug-11	294	CT	27	24	0	2.7 - 7.2	4.1	0.0	8.2
					EB	2	2	0	4.8 - 6.4	5.6	0.0	0.7
			16-Aug-12	295	CT	60	51	9	1.4 - 6.1	3.6	3.1	14.2
					LL	1	1	0	9.4	9.4	0.0	0.3
					EB	4	2	0	4.5 - 6.6	5.3	0.0	0.7
					Sculpins	common						
	3.2	13N,9W,8A	15-Aug-11	318	CT	28	21	1	1.2 - 6.4	4.4	0.3	6.3
					EB	1	0	0	4.1	4.1	0.0	0.0
					Sculpins	common						
			16-Aug-12	303	CT	36	33	1	1.5 - 6.5	3.7	0.3	10.6
					EB	3	2	0	4.7 - 6.1	5.5	0.0	0.7
					Sculpins	common						
Snowbank Creek	0.4	15N,8W,9A	23-Aug-11	450	DV	5	4	1	2.0 - 7.9	5.3	0.2	0.7
					CT	96	72	7	2.1 - 11.0	4.3	1.6	14.4
					Sculpins	present						
			14-Aug-12	450	DV	19	15	6	2.2 - 6.5	3.8	1.3	2.0
					CT	108	85	0	2.4 - 11.8	4.6	0.0	18.9
					Sculpins	present	CT YOY	observed				
Wasson Creek	0.1	13N,11W,11D	11-Aug-11	300	CT	12	10	0	4.7 - 7.8	6.0	0.0	3.3
					LL	8	7	5	2.9 - 4.4	3.6	1.7	0.7
					Spotted frogs	present						
			7-Aug-12	327	CT	7	6	0	5.7 - 8.2	6.8	0.0	1.8
					LL	1	1	0	6.5	6.5	0.0	0.3
	2.8	13N,10W,7C	11-Aug-11	300	CT	9	8	0	2.9 - 6.9	4.1	0.0	2.7
			7-Aug-12	320	CT	20	13	0	3.5 - 7.7	5.4	0.0	4.1
	3	13N,10W,7C	11-Aug-11	300	CT	38	20	0	3.0 - 7.0	4.2	0.0	6.7
			7-Aug-12	300	CT	46	42	6	1.4 - 7.7	4.3	2.0	12.0
Wilson Creek	2.4	13N,10W,17C	7-Aug-12	200	CT	21	21	6	3.0 - 7.6	5.1	3.0	7.5

* Sample may include rainbow trout / cutthroat trout hybrids

** Sample may include bull trout / brook trout hybrids

*** Sample maybe Yellowstone cutthroat- genetics pending

CT = Cutthroat trout
DV = Bull trout (Dolly Varden)
LL = Brown trout (Loch Leven)
RB = Rainbow trout
EB = Eastern brook trout
MWF = Monutain whitefish
LNS = Longnose sucker
LSS = Largescale sucker
LND = Longnose dace
RSS = Redside shiner
ONC = Oncorhynchus (Belonging to trout family)

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2011-2012.

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Size Class (in)	1st Pass	2nd Pass	3rd Pass	Prob. of Capture	Total Estimate ± CI	Estim/100' ± CI
Anaconda Creek	0.5	15N,6W,22C	2-Aug-11	372	CT	Age 1+	94	27		0.71	132 + 12.2	35.3 + 3.3
					EB	Age 1+	1	0		1.00	1.0 + 0.0	0.3 + 0.0
					All trout	Age 1+	95	27		0.72	133 + 12	36 + 3.2
			13-Aug-12	542	CT	Age 1+	136	26		0.81	168 + 7.3	31.0 + 1.3
					EB	Age 1+	1	0		1.00	1.0 + 0.0	0.2 + 0.0
					All trout	Age 1+	137	26		0.81	169 + 7.2	31.2 + 1.3
Arrastra Creek	0.7	14N,10W,29B	8-Sep-11	407	CT	YOY	2	0		1.00	2.0 + 0.0	0.5 + 0.0
						Age 1+	21	7		0.67	31.5 + 7.8	7.7 + 1.9
					DV	Age 1+	1	0		1.00	1.0 + 0.0	0.2 + 0.0
					EB	YOY	4	2		0.50	8.0 + 9.6	2.0 + 2.4
						Age 1+	38	12		0.68	55.4 + 9.3	13.6 + 2.3
					LL	YOY	6	2		0.67	9.0 + 4.2	2.2 + 1.0
						Age 1+	26	7		0.73	35.6 + 5.7	8.7 + 1.4
					All trout	YOY	12	4		0.67	18.0 + 5.9	4.4 + 1.4
						Age 1+	86	26		0.70	123.3 + 12.9	30.3 + 3.2
Ashby Creek	2.7	13N,16W,26D	1-Aug-11	513	CT	Age 1+	16	11	5	0.50	35.3 + 5.6	6.9 + 2.5
					EB	Age 1+	3	1		0.67	4.5 + 2.9	0.9 + 0.6
					All trout	Age 1+	19	12	5	0.53	39 + 5.1	7.6 + 2.3
			30-Jul-12	432	CT	Age 1+	58	10		0.83	70.1 + 4.1	16.2 + 0.9
					EB	Age 1+	2	1		0.50	4.0 + 6.8	0.9 + 1.6
					All trout	Age 1+	60	11		0.82	73.5 + 4.5	17.0 + 1.1
	4	12N,16W,3A	1-Aug-11	300	CT	Age 1+	42	14		0.67	63 + 11	21 + 3.7
					EB	YOY	1	0		1.00	1.0 + 0.0	0.3 + 0.0
						Age 1+	1	1		0.00		
					All trout	YOY	1	0		1.00	1.0 + 0.0	0.3 + 0.0
						Age 1+	43	15		0.65	66 + 12.3	22 + 4.1
			30-Jul-12	340	CT	Age 1+	32	6		0.81	39.4 + 3.4	11.6 + 1.0
					EB	Age 1+	1	0		1.00	1.0 + 0.0	0.3 + 0.0
					All trout	Age 1+	33	6		0.82	40.3 + 3.3	11.9 + 1.0
Bear Creek lower river tributary	1.1	13N,16W,18B & 7C	1-Aug-11	374	RB	Age 1+	23	3		0.87	26.5 + 1.7	7.1 + 0.5
						YOY	1	0		1.00	1.0 + 0.0	0.3 + 0.0
						Age 1+	1	0		1.00	1.0 + 0.0	0.3 + 0.0
					LL	Age 1+	16	2		0.88	18.3 + 1.4	4.9 + 0.4
					EB	YOY	1	0		1.00	1.0 + 0.0	0.3 + 0.0
						Age 1+	6	0		1.00	6.0 + 0.0	1.6 + 0.0
					All trout	YOY	2	0		1.00	2.0 + 0.0	0.5 + 0.0
						Age 1+	46	5		0.89	51.6 + 1.9	13.8 + 0.5

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Size Class (in)	1st Pass	2nd Pass	3rd Pass	Prob. of Capture	Total Estimate ± CI	Estim/100' ± CI
Blackfoot River (above Hogum Cr Rd)	119.6	14N,7W,5D	5-Sep-12	3140	CT	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	21	9		0.57	36.8 ± 14.1	11.7 ± 4.6
					LL	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	3	2		0.33	9.0 ± 26.3	2.9 ± 8.4
					EB	YOY	2	2				
						Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					All trout	YOY	4	2		0.50	8.0 ± 9.6	2.5 ± 3.1
						Age 1+	25	11		0.56	44.6 ± 16.5	14.2 ± 5.3
Estimates/CI for this section are based on 1000ft												
Blackfoot River (above Flesher Pass Rd)	124.3	15N,7W,35B & 26C	6-Sep-12	2141	CT	YOY	7	0		1.00	7.0 ± 0.0	3.3 ± 0.0
						Age 1+	20	7		0.65	30.8 ± 8.4	14.4 ± 3.9
					LL	Age 1+	2	0		1.00	2.0 ± 0.0	0.9 ± 0.0
					EB	YOY	17	8		0.53	32.1 ± 16.5	15 ± 7.8
						Age 1+	63	17		0.73	86.3 ± 8.9	40.3 ± 4.1
					All trout	YOY	24	8		0.67	36.0 ± 8.3	16.8 ± 3.9
						Age 1+	85	24		0.72	118.4 ± 11.2	55.3 ± 5.2
Estimates/CI for this section are based on 1000ft												
Braziel Creek	0.2	12N,10W,10D	29-Aug-11	300	CT	Age 1+	8	0		1.00	8.0 ± 0.0	2.7 ± 0.0
			4-Sep-12	300	CT	YOY	30	6		0.80	37.5 ± 3.7	12.5 ± 1.2
						Age 1+	97	6		0.94	103.4 ± 1.4	34.5 ± 0.5
Chamberlain Creek	0.1	15N,13W,32A	12-Sep-11	358	CT	YOY	12	0		1.00	12 ± 0.0	3.4 ± 0.0
						Age 1+	60	14		0.77	78.3 ± 6.7	21.9 ± 1.9
						All CT	72	14		0.81	89.4 ± 5.4	25 ± 1.5
					RB	Age 1+	2	0		1.00	2.0 ± 0.0	0.6 ± 0.0
					LL	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	6	1		0.83	7.2 ± 1.2	2.0 ± 0.3
					All trout	YOY	13	0		1.00	13 ± 0.0	3.6 ± 0.0
						Age 1+	68	15		0.78	87.3 ± 6.5	24.4 ± 1.8
			12-Sep-12	358	CT	YOY	25	7		0.72	34.7 ± 6.0	9.7 ± 1.7
						Age 1+	37	2		0.95	39.1 ± 0.7	10.9 ± 0.2
						All CT	62	9		0.85	72.5 ± 3.3	20.3 ± 0.9
					RB	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					LL	YOY	4	1		0.75	5.3 ± 1.9	1.5 ± 0.5
					All trout	YOY	29	8		0.72	40.1 ± 6.3	11.2 ± 1.8
						Age 1+	38	2		0.95	40.1 ± 0.7	11.2 ± 0.2
Copper Creek	6.2	15N,8W,9A	22-Aug-11	536	DV	YOY	23	2		0.91	25.2 ± 1.0	4.7 ± 0.2
						Age 1+	12	2		0.83	14.4 ± 1.8	2.7 ± 0.3
					CT	Age 1+	38	16		0.58	65.6 ± 18.1	12.2 ± 3.4
					All trout	YOY	23	2		0.91	25.2 ± 1.0	4.7 ± 0.2
						Age 1+	50	18		0.64	78.1 ± 14.2	14.6 ± 2.7
			20-Aug-12	535	DV	YOY	12	2		0.83	14.4 ± 1.8	2.7 ± 0.3
						Age 1+	7	7				
					CT	YOY	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
						Age 1+	54	15		0.72	74.8 ± 8.7	14.0 ± 1.6
					All trout	YOY	13	2		0.85	15.4 ± 1.6	2.9 ± 0.3
						Age 1+	61	22		0.64	95.4 ± 15.8	17.8 ± 2.9

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Size Class (in)	1st Pass	2nd Pass	3rd Pass	Prob. of Capture	Total Estimate ± CI	Estim/100' ± CI
Cottonwood Creek	12.0	16N,14W,24D	12-Sep-11	515	CT	Age 1+	38	12		0.68	55.5 + 9.3	10.8 ± 1.8
					EB	Age 1+	3	0		1.00	3.0 + 0.0	0.6 + 0.0
					All trout	Age 1+	41	12		0.71	58 + 8.3	11.3 + 1.6
			17-Sep-12	515	CT	YOY	2	1		0.50	4.0 + 6.8	0.8 + 1.3
						Age 1+	47	5		0.89	52.6 + 1.9	10.2 + 0.4
					EB	YOY	5	0		1.00	5.0 + 0.0	1.0 + 0.0
						Age 1+	2	0		1.00	2.0 + 0.0	0.4 + 0.0
					All trout	YOY	7	1		0.86	8.2 + 1.1	1.6 + 0.2
						Age 1+	49	5		0.90	54.6 + 1.8	10.6 + 0.4
Enders Spring Creek (trib to NFBLKFT)	0.1	14N,11W,6B	24-Aug-11	381	LL	Age 1+	2	2				
					EB	YOY	7	5				
						Age 1+	22	9		0.59	37.2 + 12.8	9.8 + 3.4
					All trout	YOY	7	5				
						Age 1+	24	11		0.54	44.3 + 18.1	11.6 + 4.8
Gold Creek	1.9	14N,16W,30D	31-Aug-11	400	RB*	YOY	5	0		1.00	5.0 + 0.0	1.3 + 0.0
						Age 1+	47	14		0.70	67 + 9.2	16.7 + 2.3
					LL	Age 1+	34	15		0.56	60.8 + 19.4	15.2 + 4.8
					All trout	YOY	5	0		1.00	5.0 + 0.0	1.3 + 0.0
						Age 1+	81	29		0.64	126.2 + 17.9	31.5 + 4.5
Grantier Spring Creek	1.0	14N, 9W, 25A	20-Jul-11	507	CT	Age 1+	11	3		0.73	15.1 + 3.8	3.0 + 0.7
					LL	YOY	36	4		0.89	40.5 + 1.7	8.0 + 0.3
						Age 1+	6	1		0.83	7.2 + 1.2	1.4 + 0.2
					EB	YOY	4	2		0.50	8.0 + 9.6	1.6 + 1.9
						Age 1+	6	1		0.83	7.2 + 1.2	1.4 + 0.2
					All trout	YOY	40	6		0.85	47.1 + 2.8	9.3 + 0.5
						Age 1+	23	5		0.78	29.4 + 3.7	5.8 + 0.7
			8-Aug-12	507	CT	Age 1+	20	7		0.65	30.8 + 8.4	6.1 + 1.7
					LL	YOY	11	0		1.00	11.0 + 0.0	2.2 + 0.0
						Age 1+	10	0		1.00	10.0 + 0.0	2.0 + 0.0
					EB	YOY	1	1				
						Age 1+	2	0		1.00	2.0 + 0.0	0.4 + 0.0
					All trout	YOY	12	1		0.92	13.1 + 0.7	2.6 + 0.1
						Age 1+	32	7		0.78	41 + 4.4	8.1 + 0.9
Hoyt Creek	1.2	15N,12W,19C	31-Aug-11	300	CT	Age 1+	0	1				
					EB	YOY	1	0		1.00	1.0 + 0.0	0.3 + 0.0
						Age 1+	5	2		0.60	8.3 + 5.8	2.8 + 1.9
					All trout	YOY	1	0		1.00	1.0 + 0.0	0.3 + 0.0
						Age 1+	5	3		0.40	12.5 + 20.8	4.2 + 6.9
	2.7	15N,12W,29C	31-Aug-11	300	No fish found							
	4.3	15N,12W,28C	31-Aug-11	300	EB	Age 1+	8	3		0.63	12.8 + 6.2	4.3 + 2.1

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Size Class (in)	1st Pass	2nd Pass	3rd Pass	Prob. of Capture	Total Estimate ± CI	Estim/100' ± CI
Jacobsen Spring Creek	0.6	14N,12W,1CD	9-Aug-11	525	ONC	YOY	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
					LL	YOY	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
						Age 1+	1	1				
					EB	YOY	40	13		0.68	59.3 ± 10.2	11.3 ± 1.9
						Age 1+	13	3		0.77	16.9 ± 3.1	3.2 ± 0.6
					All trout	YOY	43	13		0.70	61.6 ± 9.1	11.7 ± 1.7
						Age 1+	14	4		0.71	19.6 ± 4.7	3.7 ± 0.9
			22-Aug-12	460	DV	Age 1+	0	1				
					LL	YOY	0	1				
						Age 1+	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
					RB	YOY	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
					EB	YOY	21	5		0.76	27.6 ± 4.1	6.0 ± 0.9
						Age 1+	24	6		0.75	32.0 ± 4.8	7.0 ± 1.0
					All trout	YOY	22	6		0.73	30.3 ± 5.3	6.6 ± 1.2
						Age 1+	26	7		0.73	35.6 ± 5.7	7.7 ± 1.2
Kleinschmidt Creek	0.5	14N,11W,6D & 5C	21-Aug-12	355	LL	YOY	4	0		1.00	4.0 ± 0.0	1.1 ± 0.0
						Age 1+	21	6		0.71	29.4 ± 5.7	8.3 ± 1.6
					EB	YOY	2	0		1.00	2.0 ± 0.0	0.6 ± 0.0
					All trout	YOY	6	0		1.00	6.0 ± 0.0	1.7 ± 0.0
						Age 1+	21	6		0.71	29.4 ± 5.7	8.3 ± 1.6
	0.8	14N,11W,5C	21-Aug-12	500	CT	Age 1+	0	1				
					LL	YOY	11	1		0.91	12.1 ± 0.7	2.4 ± 0.1
						Age 1+	38	20		0.47	80.2 ± 35	16 ± 7.0
						All LL	49	21		0.57	85.8 ± 22	17.2 ± 4.3
					RB	Age 1+	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
					EB	YOY	4	1		0.75	5.3 ± 1.9	1.1 ± 0.4
						Age 1+	5	3		0.40	13 ± 21	2.5 ± 4.2
					All trout	YOY	15	2		0.87	17.3 ± 1.4	3.5 ± 0.3
						Age 1+	44	23		0.48	92.2 ± 37	18.4 ± 7.4
Lincoln Spring Creek	3.8	14N,9W,13D	17-Aug-11	369	CT	Age 1+	0	1				
					LL	Age 1+	13	5		0.62	21.1 ± 8.4	5.7 ± 2.3
					EB	YOY	5	4		0.20	25 ± 117.6	6.8 ± 31.9
						Age 1+	4	3		0.25	16 ± 62.2	4.3 ± 16.9
					All trout	YOY	5	4		0.20	25 ± 117.6	6.8 ± 31.9
						Age 1+	17	9		0.47	36.1 ± 23.9	9.8 ± 6.5
			14-Aug-12	385	CT	Age 1+	0	1				
					LL	Age 1+	17	8		0.53	32.1 ± 16.5	8.3 ± 4.3
					EB	YOY	1	1				
						Age 1+	6	2		0.67	9.0 ± 4.2	2.3 ± 1.1
					All trout	YOY	1	1				
						Age 1+	23	11		0.52	44.1 ± 20.1	11.5 ± 5.2
Murphy's Spring Creek (trib to NFBLKFT)	0.6	15N,11W,21B	30-Aug-11	385	DV	Age 1+	10	3		0.70	14.3 ± 4.3	3.7 ± 1.1
					CT	Age 1+	31	10		0.68	45.8 ± 8.8	11.9 ± 2.3
					EB	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	10	2		0.80	12.5 ± 2.1	3.2 ± 0.6
					All trout	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	51	15		0.71	72.3 ± 9.4	18.8 ± 2.4
			28-Aug-12	385	DV	Age 1+	0	2				
					CT	YOY	7	1		0.86	8.2 ± 1.1	2.1 ± 0.3
						Age 1+	33	2		0.94	35.1 ± 0.8	9.1 ± 0.2
					EB	Age 1+	3	1		0.67	4.5 ± 2.9	1.2 ± 0.8
					All trout	YOY	7	1		0.86	8.2 ± 1.1	2.1 ± 0.3
						Age 1+	36	5		0.86	41.8 ± 2.4	10.9 ± 0.6

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Size Class (in)	1st Pass	2nd Pass	3rd Pass	Prob. of Capture	Total Estimate ± CI	Estim/100' ± CI
Pearson Creek	0.5	15N,13W,33D	7-Sep-11	300	CT	Age 1+	13	3		0.77	16.9 + 3.1	5.6 + 1.0
					EB	YOY	3	0		1.00	3.0 + 0.0	1.0 + 0.
			12-Sep-12	300	CT	Age 1+	13	0		1.00	13.0 + 0.0	4.3 + 0.0
					EB	YOY	1	2				
					EB	Age 1+	3	0		1.00	3.0 + 0.0	1.0 + 0.0
					All trout	YOY	1	2				
					All trout	Age 1+	16	0		1.00	16 + 0.0	5.3 + 0.0
	1.1	14N,13W,3B	7-Sep-11	370	CT	Age 1+	52	14		0.73	71.2 + 8.0	19.2 + 2.2
			12-Sep-12	370	CT	Age 1+	46	9		0.80	57.2 + 4.4	15.5 + 1.2
					EB	YOY	5	1		0.80	6.3 + 1.5	1.7 + 0.4
Poorman Creek	1.3	14N,9W,36A	10-Aug-11	490	CT	Age 1+	9	3		0.67	13.5 + 5.1	2.8 + 1.0
					LL	YOY	6	1	0	0.87	7.0 + 0.0	1.4 + 0.0
					LL	Age 1+	25	15	4	0.60	46.1 + 3.8	9.4 + 1.7
					All trout	YOY	6	1		0.83	7.2 + 1.2	1.4 + 0.0
					All trout	Age 1+	34	18	4	0.64	57.9 + 3.4	11.8 + 1.5
					All trout	Total	40	18	4	0.67	63.4 + 2.8	12.9 + 1.2
			15-Aug-12	510	CT	Age 1+	11	3		0.73	15.1 + 3.8	3.0 + 0.7
					LL	YOY	4	3		0.25	16.0 + 62.2	3.1 + 12.2
					LL	Age 1+	15	4		0.73	20.5 + 4.2	4.0 + 0.8
					EB	YOY	1	0		1.00	1.0 + 0.0	0.2 + 0.0
					EB	Age 1+	2	0		1.00	2.0 + 0.0	0.4 + 0.0
					All trout	YOY	5	3		0.40	12.5 + 20.8	2.5 + 4.1
					All trout	Age 1+	28	7		0.75	37.3 + 5.2	7.3 + 1.0
					All trout	Total	33	10		0.70	47.4 + 8.0	9.3 + 1.6
	1.5	14N,9W,36A	10-Aug-11	270	CT	Age 1+	1	1				
					LL	YOY	0	1				
					LL	Age 1+	5	3	1	0.64	9.0 + 0.0	3.3 + 0.0
					All trout	YOY	0	1				
					All trout	Age 1+	6	4	1	0.64	11 + 0.0	4.1 + 0.0
					All trout	Total	6	5	1	0.63	12 + 0.0	4.4 + 0.0
			15-Aug-12	270	CT	YOY	0	1				
					CT	Age 1+	3	1		0.67	4.5 + 2.9	1.7 + 1.1
					LL	Age 1+	7	2		0.71	9.8 + 3.3	3.6 + 1.2
					All trout	YOY	0	1				
					All trout	Age 1+	10	3		0.70	14.3 + 4.3	5.3 + 1.6
					All trout	Total	10	4		0.60	16.7 + 8.1	6.2 + 3.0

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Size Class (in)	1st Pass	2nd Pass	3rd Pass	Prob. of Capture	Total Estimate ± CI	Estim/100' ± CI
Rock Creek	1.6	14N,11W,5A	17-Sep-12	510	LL	YOY	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
						Age 1+	13	3		0.77	16.9 ± 3.1	3.3 ± 0.6
					EB	YOY	2	1		0.50	4.0 ± 6.8	0.8 ± 1.3
						Age 1+	9	5		0.44	20.3 ± 20.6	4.0 ± 4.0
					All trout	YOY	4	1		0.75	5.3 ± 1.9	1.0 ± 0.4
						Age 1+	22	8		0.64	34.6 ± 9.6	6.8 ± 1.9
	3.9	15N,11W,35B	9-Aug-12	366	CT	Age 1+	3	0		1.00	3.0 ± 0.0	0.8 ± 0.0
					LL	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					RB	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					EB	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	8	0		1.00	8.0 ± 0.0	2.2 ± 0.0
					All trout	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	13	0		1.00	13.0 ± 0.0	3.6 ± 0.0
Sauerkraut Creek	2.9	13N,9W,5D	15-Aug-11	294	CT	Age 1+	24	3		0.88	27.4 ± 1.7	9.3 ± 0.6
					EB	Age 1+	2	0		1.00	2.0 ± 0.0	0.7 ± 0.0
					All trout	Age 1+	26	3		0.88	29.4 ± 1.6	10 ± 0.5
			16-Aug-12	295	CT	YOY	7	1		0.86	8.2 ± 1.1	2.8 ± 0.4
						Age 1+	44	8		0.82	53.8 ± 3.8	18.2 ± 1.3
					LL	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					EB	Age 1+	2	2				
					All trout	YOY	7	1		0.86	8.2 ± 1.1	2.8 ± 0.4
						Age 1+	47	10		0.79	59.7 ± 5.1	20.2 ± 1.7
	3.2	13N,9W,8A	15-Aug-11	318	CT	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	20	7		0.65	30.8 ± 8.4	9.7 ± 2.7
					EB	Age 1+	0	1				
					All trout	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	20	8		0.60	33.3 ± 11.5	10.5 ± 3.6
			16-Aug-12	303	CT	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	32	3		0.91	35.3 ± 1.3	11.7 ± 0.4
					EB	Age 1+	2	1		0.50	4.0 ± 6.8	1.3 ± 2.2
					All trout	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	34	4		0.88	38.5 ± 1.8	12.7 ± 0.6
Snowbank Creek	0.4	15N,8W,9A	23-Aug-11	450	DV	YOY	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
						Age 1+	3	1		0.67	4.5 ± 2.9	1.0 ± 0.7
					CT	YOY	7	1		0.86	8.2 ± 1.1	1.8 ± 0.2
						Age 1+	65	23		0.65	100.6 ± 15.6	22.4 ± 3.5
					All trout	YOY	8	1		0.88	9.1 ± 1.0	2.0 ± 0.2
						Age 1+	68	24		0.65	105.1 ± 15.8	23.4 ± 3.5
			14-Aug-12	450	DV	YOY	6	2		0.67	9.0 ± 4.2	2.0 ± 0.9
						Age 1+	9	2		0.78	11.6 ± 2.4	2.6 ± 0.5
					CT	Age 1+	85	23		0.73	116.5 ± 10.4	25.9 ± 2.3
					All trout	YOY	6	2		0.67	9.0 ± 4.2	2.0 ± 0.9
						Age 1+	94	25		0.73	128.1 ± 10.6	28.5 ± 2.3

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2011-2012 (cont'd).

Stream	River Mile	Location (T,R,S)	Date Sampled	Section Length (ft)	Species	Size Class (in)	1st Pass	2nd Pass	3rd Pass	Prob. of Capture	Total Estimate ± CI	Estim/100' ± CI
Wasson Creek	0.1	13N,11W,11D	11-Aug-11	300	CT	Age 1+	10	2		0.80	12.5 ± 2.1	4.2 ± 0.7
					LL	YOY	5	1		0.80	6.3 ± 1.5	2.1 ± 0.5
						Age 1+	2	0		1.00	2.0 ± 0.0	0.7 ± 0.0
					All trout	YOY	5	1		0.80	6.3 ± 1.5	2.1 ± 0.5
						Age 1+	12	2		0.83	14.4 ± 1.8	4.8 ± 0.6
			7-Aug-12	327	CT	Age 1+	6	1		0.83	7.2 ± 1.2	2.2 ± 0.4
					LL	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					All trout	Age 1+	7	1		0.86	8.2 ± 1.1	2.5 ± 0.3
	2.8	13N,10W,7C	11-Aug-11	300	CT	Age 1+	8	1		0.88	9.1 ± 1.0	3.0 ± 0.3
			7-Aug-12	320	CT	Age 1+	13	7		0.46	28.2 ± 22.2	8.8 ± 6.9
	3.0	13N,10W,7C	11-Aug-11	300	CT	Age 1+	20	11	7	0.50	42.6 ± 6.9	14.2 ± 4.0
			7-Aug-12	300	CT	YOY	6	0		1.00	6.0 ± 0.0	2.0 ± 0.0
						Age 1+	36	4		0.89	40.5 ± 1.7	13.5 ± 0.6

* Sample may include rainbow trout / cutthroat trout hybrids

** Sample may include bull trout / brook trout hybrids

*** Sample maybe Yellowstone cutthroat- genetics pending

CT = Cutthroat trout
DV = Bull trout (Dolly Varden)
LL = Brown trout (Loch Leven)
RB = Rainbow trout
EB = Eastern brook trout
MWF = Monutain whitefish
LNS = Longnose sucker
LSS = Largescale sucker
LND = Longnose dace
RSS = Redside shiner
ONC = Oncorhynchus (Belonging to trout family)

Appendix C: Mark and recapture estimates of abundance and biomass for Blackfoot River and Nevada Creek, 2011 - 2012.

Stream	River Mile Mid-point	Date Sampled	Section Length (ft)	Species	Size Class (inches)	M	C	R	(R/C)	Total Estimate ± 95%CI	Total Biomass (lb/section)	Estimate/1000' ± 95%CI	Biomass (lb/1000')	Condition Factor/1000'
Blackfoot River, Johnsrud Section	13.5	14-May-12	17680	RB	5 - 9.9	245	250	48	0.19	1259 ± 280	223.00	71.2 ± 15.9	12.60	40.30
					10 -11.9	31	34	6	0.18	159 ± 87.7	73.80	9.0 ± 5.0	4.20	36.30
					≥ 5	328	338	62	0.18	1769 ± 352	576.20	100 ± 20	32.60	38.90
					≥ 6	304	319	61	0.19	1573.2 ± 312	541.20	89 ± 17.6	30.60	38.30
					≥12	52	54	8	0.15	323 ± 167.3	297.00	18.3 ± 9.5	16.80	33.70
				LL	≥ 6	60	58	13	0.22	256.1 ± 99.7	173.20	14.5 ± 5.6	9.80	36.60
					6 - 11.9	34	39	6	0.15	199 ± 112.6	42.20	11.3 ± 6.4	2.40	36.90
					≥12	26	19	7	0.37	67 ± 28.7	99.40	3.8 ± 1.6	5.60	35.99
				CT	≥ 6	106	109	21	0.19	534 ± 174.3	236.00	30.2 ± 9.9	13.30	37.20
					6 - 11.9	85	81	14	0.17	469.1 ± 189.2	152.00	26.5 ± 10.7	8.60	37.40
					>12	21	28	7	0.25	78.8 ± 35.4	68.40	4.5 ± 2.0	3.90	36.40
				DV	≥ 6	15	20	1	0.05	167 ± 169.2	372.80	9.5 ± 9.6	21.10	35.20
				All trout	≥ 5	511	530	97	0.18	2773.2 ± 443.8	1263.20	157 ± 25.1	71.40	38.20
					≥ 6	485	506	96	0.19	2539 ± 405	1208.50	144 ± 23	68.40	37.80
Blackfoot River, Scotty Brown Bridge	43.9	15-May-12	20064	RB	4 - 10.9	82	104	11	0.11	725.3 ± 344	175.80	36.2 ± 17.1	8.80	42.00
					≥ 4	152	168	24	0.14	1033.3 ± 337	598.10	52 ± 17	29.80	39.50
					≥ 6	151	165	24	0.15	1008.3 ± 326.8	591.00	50.3 ± 16.3	29.40	39.50
					11 - 13.9	37	30	7	0.23	146.3 ± 73.6	108.60	7.3 ± 3.7	5.40	37.30
					≥ 14	33	34	6	0.18	169 ± 94	234.00	8.4 ± 4.7	11.70	34.20
				LL	≥ 6	53	61	8	0.13	371 ± 195	271.10	18.5 ± 9.7	13.50	36.60
					6 - 11.9	29	32	1	0.03	494 ± 525	145.10	24.6 ± 26.1	7.20	37.00
					≥12	24	29	7	0.24	93 ± 43.3	120.60	4.6 ± 2.2	6.00	36.00
				CT	≥ 6	145	148	29	0.20	724.1 ± 203.3	480.30	36.1 ± 10.1	24.00	39.20
					6 - 11.9	82	78	16	0.21	385 ± 141	133.10	19.2 ± 7.0	6.60	40.80
					>12	63	70	13	0.19	324 ± 130.1	338.00	16.1 ± 6.5	16.80	37.30
				DV	≥ 6	21	16	3	0.19	93 ± 65	213.20	4.6 ± 3.2	10.60	34.50
				All trout	≥ 6	370	390	64	0.16	2231 ± 446.5	1609.00	111 ± 22.3	80.20	38.70

Appendix C: Mark and recapture estimates of abundance and biomass for Blackfoot River and Nevada Creek, 2011 - 2012 (cont'd).

Stream	River Mile Mid-point	Date Sampled	Section Length (ft)	Species	Size Class (inches)	M	C	R	(R/C)	Total Estimate ± 95%CI	Total Biomass (lb/section)	Estimate/1000' ± 95%CI	Biomass (lb/1000')	Condition Factor/1000'
Wales Creek Section	63	17-May-12	31635	RB	≥6	29	14	6	0.43	63.3 ± 28.5	35.10	2.0 ± 0.9	1.10	34.5
					6 - 11.9	15	10	3	0.30	43 ± 27	12.70	1.4 ± 0.8	0.40	34.6
					>12	14	4	3	0.75	17.8 ± 6.3	16.60	1.0 ± 0.2	0.50	34.2
				LL	≥6	66	55	12	0.22	288 ± 119	295.00	9.1 ± 3.8	9.30	36.2
					6 - 11.9	19	20	4	0.20	83 ± 51	28.40	2.6 ± 1.6	0.90	38.7
					>12	47	35	8	0.23	191 ± 93	258.00	6.0 ± 3.0	8.10	34.96
				CT	≥6	38	15	5	0.33	103 ± 56	55.50	3.3 ± 1.8	1.80	37.7
					6 - 11.9	27	11	3	0.27	83 ± 57	33.80	2.6 ± 1.8	1.10	38.3
					>12	11	4	2	0.50	19 ± 11	17.00	1.0 ± 0.3	0.50	36
				DV	≥6	5	3	0	0.00	23 ± 26.3	39.00	1.0 ± 0.8	1.20	31.3
					6 - 11.9	1	1	0	0.00	3.0 ± 2.8	1.00	0.1 ± 0.1	0.00	30
					>12	4	2	0	0.00	14 ± 15.2	29.90	0.44 ± 0.5	1.00	31.7
				All trout	≥6	138	87	23	0.26	508.7 ± 155	432.40	16.1 ± 4.9	13.70	36
					6 - 11.9	62	42	10	0.24	245.3 ± 109	87.20	7.8 ± 3.5	2.80	37.4
					>12	76	45	13	0.29	252 ± 97	323.00	8.0 ± 3.1	10.20	34.8
Wales Creek "MWF estimates"	63.6	17-May-12	23760	MWF	≥6	685	566	135	0.24	2859 ± 374	1532.00	120.3 ± 15.7	64.50	34.8
					6 - 11.9	421	295	71	0.24	1734 ± 315.3	602.60	73 ± 13.3	25.40	35.7
					≥12	264	271	64	0.24	1108 ± 203	879.40	47 ± 8.5	37.00	33.5
					≥8	657	551	134	0.24	2689.5 ± 350.4	1482.50	113.2 ± 14.8	62.40	34.7
Canyon Section	95.3	29-Sep-11	5422	CT	≥6	16	14	5	0.36	41.5 ± 19.6	54.10	7.7 ± 3.6	10.00	37.9
					6 - 11.9	2	0	0		2.0 ± 0.0	0.90	0.4 ± 0.0	0.20	31.1
					>12	14	14	5	0.36	36.5 ± 16.7	50.20	6.7 ± 3.1	9.30	38.5
				LL	≥6	29	5	0	0.00	179 ± 224	184.40	33 ± 41.3	34.00	39.3
					6 - 11.9	10	2	0	0.00	32 ± 36	11.00	6.0 ± 6.6	2.00	43.5
					>12	19	3	0	0.00	79 ± 94	111.00	15 ± 17.3	20.50	37
				Age1+	≥4.5	29	5	0	0.00	179 ± 224	184.40	33 ± 41.3	34.00	39.3
				Age2+	≥8.5	24	4	0	0.00	124 ± 152	150.30	23 ± 28	27.70	39.1
				All trout	≥6	45	19	5	0.26	152.3 ± 88.6	174.60	28.1 ± 16.3	32.20	38.7
					6 - 11.9	12	2	0	0.00	38 ± 42.4	13.70	7.0 ± 7.8	2.50	41.8
					>12	33	17	5	0.29	101 ± 56	140.50	18.6 ± 10.3	26.00	37.8
				MWF	≥6	182	54	11	0.20	838 ± 390	750.50	155 ± 72	138.40	34.8
					6 - 11.9	31	6	0	0.00	223 ± 283	86.40	41.1 ± 52.2	15.90	32.3
					≥12	151	48	11	0.23	620 ± 281.4	617.20	114.3 ± 52	113.80	35.2
					≥8	177	54	11	0.20	815 ± 379	745.00	150.3 ± 70	137.40	35

Appendix C: Mark and recapture estimates of abundance and biomass for Blackfoot River and Nevada Creek, 2011 - 2012 (cont'd).

Stream	River Mile Mid-point	Date Sampled	Section Length (ft)	Species	Size Class (inches)	M	C	R	(R/C)	Total Estimate ± 95%CI	Total Biomass (lb/section)	Estimate/1000' ± 95%CI	Biomass (lb/1000')	Condition Factor/1000'
Nevada Creek above H2-O	5.0 - 6.3	20-Sep-11	6500	CT	>4.0	23	31	9	0.29	76 ± 29	43.2	11.7 ± 4.4	6.6	39.3
				DV	>4.0	0	1	0	0.00	1.0 ± 0.0	2.6	0.2 ± 0.0	0.4	33.5
				RB	>4.0	2	1	0	0.00	5.0 ± 4.8	1.4	0.8 ± 0.7	0.2	54
				LL	>4.0	31	33	7	0.21	135 ± 67.3	60.0	20.8 ± 10.4	9.2	42.4
				All trout	>4.0	56	66	16	0.24	223.7 ± 75.1	114.6	34.4 ± 11.6	17.6	41.3
		25-Sep-12		CT	>4.0	27	23	10	0.43	60.1 ± 19.8	33.3	9.2 ± 3.1	5.1	35.9
				DV	>4.0	1	0	0		1.0 ± 0.0	0.9	0.2 ± 0.0	0.1	26.1
				RB	>4.0	0	1	0	0.00	1.0 ± 0.0	0.4	0.2 ± 0.0	0.1	29.4
				LL	>4.0	66	34	15	0.44	146 ± 45	57.9	22.4 ± 6.9	8.9	37.4
				All trout	>4.0	94	58	25	0.43	215 ± 51.8	96.7	33 ± 8.0	14.9	36.8
Nevada Creek "Stit project"	29	21-Sep-11	3695	CT	>4.0	36	29	12	0.41	84.4 ± 27.1	41.1	22.8 ± 7.3	11.1	45.3
				RB	>4.0	29	24	5	0.21	124 ± 72.2	87.8	33.6 ± 19.5	23.8	39.6
				LL	>4.0	13	5	1	0.20	41 ± 36	25.3	11.1 ± 9.7	6.8	41
				All trout	>4.0	78	58	18	0.31	244.3 ± 77.2	145.5	66.1 ± 21	39.4	42.4
		10-Sep-12	3695	CT	>4.0	20	42	10	0.24	81.1 ± 28	41.6	22 ± 7.5	11.3	37.9
				RB	>4.0	38	104	17	0.16	227 ± 35	88.5	61.3 ± 18.5	24.0	37.9
				LL	>4.0	9	15	5	0.33	25.7 ± 9.9	29.4	7.0 ± 2.7	8.0	39.3
				All trout	>4.0	67	161	32	0.20	333 ± 72	165.2	90.1 ± 19.4	44.7	38

CT = Cutthroat trout
DV = Bull trout (Dolly Varden)
LL = Brown trout (Loch Leven)
RB = Rainbow trout
EB = Eastern brook trout
MWF = Mountain whitefish

Appendix D: Mark and recapture estimates of abundance and biomass for mountain whitefish for Blackfoot River, 2006 - 2012.

Stream	River Mile Mid-point	Date Sampled	Section Length (ft)	Species	Size Class (inches)	M	C	R	(R/C)	Total Estimate ± 95%CI	Total Biomass (lb/section)	Estimate/1000' ± 95%CI	Biomass (lb/1000')	Condition Factor/1000'
Blackfoot River	95.3	20-Sep-06	5422	MWF	≥6	177	121	24	0.20	868 ± 276	654	160 ± 51	121	35
Canyon Section					6 - 11.9	28	19	3	0.16	144 ± 106	71	27 ± 20	13	36
					≥ 12	149	102	21	0.21	701 ± 235	563	129 ± 43	104	35
					≥ 8	177	121	24	0.20	868 ± 276	654	160 ± 51	121	35
					8 - 11.9	28	19	3	0.16	144 ± 106	71	27 ± 19	13	36
		24-Sep-09	5422	MWF	≥6	109	92	16	0.17	602 ± 231	525	111 ± 43	97	37
					6 - 11.9	19	18	2	0.11	126 ± 105	74	23 ± 19	14	41
					≥ 12	90	73	14	0.19	448 ± 180	426	83 ± 33	79	36
					≥ 8	109	92	16	0.17	601 ± 231	525	111 ± 43	97	37
					8 - 11.9	20	18	2	0.11	132 ± 111	74	24 ± 20	14	41
		29-Sep-11	5422	MWF	≥6	182	54	11	0.20	838 ± 390	751	155 ± 72	138	35
					6 - 11.9	31	6	0	0.00	223 ± 283	86	41 ± 52	16	32
					≥ 12	151	48	11	0.23	620 ± 281	617	114 ± 52	114	35
					≥ 8	177	54	11	0.20	815 ± 379	745	150 ± 70	137	35
					8 - 11.9	26	6	0	0.00	188 ± 240	82	35 ± 44	15	34
Blackfoot River	65	24-May-06	7603	MWF	≥6	80	85	4	0.05	1392 ± 1048	707	183 ± 138	93	38
Wales Creek					6 - 11.9	52	47	3	0.06	635 ± 513	220	84 ± 68	29	39
Section					≥ 12	28	38	1	0.03	565 ± 601	436	74 ± 79	57	37
					≥ 8	74	78	4	0.05	1184 ± 887	650	156 ± 117	85	37
					8 - 11.9	40	40	3	0.08	419 ± 332	191	55 ± 44	25	38
	63.6	9-Jun-08	23760	MWF	≥6	265	254	22	0.09	2948 ± 1076	2128	124 ± 45	90	38
					6 - 11.9	131	96	11	0.11	1066 ± 518	527	45 ± 22	22	38
					≥ 12	134	158	11	0.07	1788 ± 892	1602	75 ± 38	67	38
					≥ 8	261	252	22	0.09	2881 ± 1050	2100	121 ± 44	88	38
					8 - 11.9	127	94	11	0.12	1012 ± 490	512	43 ± 21	22	38
		19-May-10	23760	MWF	≥6	365	366	36	0.10	3629 ± 1038	2307	153 ± 44	97	37
					6 - 11.9	156	180	12	0.07	2185 ± 1057	951	92 ± 45	40	40
					≥ 12	209	186	24	0.13	1570 ± 528	1273	66 ± 22	54	35
					≥ 8	348	346	36	0.10	3272 ± 930	2177	138 ± 39	92	37
					8 - 11.9	139	160	12	0.08	1733 ± 829	827	73 ± 35	35	40
		17-May-12	23760	MWF	≥6	685	566	135	0.24	2859 ± 374	1532	120 ± 16	65	35
					6 - 11.9	421	295	71	0.24	1734 ± 315	603	73 ± 13	25	36
					≥ 12	264	271	64	0.24	1108 ± 203	879	47 ± 9	37	34
					≥ 8	657	551	134	0.24	2690 ± 350	1483	113 ± 15	62	35
					8 - 11.9	393	280	70	0.25	1558 ± 282	564	66 ± 12	24	36

Appendix E: Summary of water chemistry readings for 2011.

Stream name	Date	River Mile	pH	Conductivity (uS)	TDS (ppm)	Temp (°F)	Lat	Long	TRS
Anaconda Creek	2-Aug-11	0.5	7.2	160	160	50	N47.03495	W112.35246	15N,6W,22C
Arrastra Creek	8-Sep-11	0.7	8.36	173	85	50.4	N46.94577	W112.89927	14N,10W,29B
Ashby Creek	1-Aug-11	2.7	7.2	375	266	61.5	N46.85028	W113.58542	13N,16W,26D
Ashby Creek	1-Aug-11	4	6.3	387	275	54.3	N46.83116	W113.59613	12N,16W,3A
Bear Creek	1-Aug-11	1.1	6.6	101	na	52.5	N46.83211	W113.59530	13N,16W,18B
Blackfoot River (Canyon Section)	29-Sep-11	95.3	8.6	316	224	51.6	N46.93893	W112.84408	14N,10W,27D
Braziel Creek	29-Aug-11	0.2	8.1	122	60	62.8	N46.80900	W112.84003	12N,10W,10D
Chamberlain Creek	12-Sep-11	0.1	8.2	128	63	61.7	N47.01407	W113.26826	15N,13W,32A
Copper Creek	22-Aug-11	6.2	7.3	182	93	56.7	N47.07146	W112.61197	15N,8W,9A
Cottonwood Creek	12-Sep-11	12	8.7	152	76	48	N47.12143	W113.30469	16N,14W,24D
Enders Spring Creek	24-Aug-11	0.1	7.9	198	99	54.1	N47.00069	W113.0489	14N,11W,6B
Gold Creek	31-Aug-11	1.9	9.2	216	107	50	N46.93935	W113.66874	14N,16W,30D
Grantier Spring Creek	20-Jul-11	1	7.5	292	na	50	N46.94083	W112.67336	14N,9W,25A
Hoyt Creek	31-Aug-11	1.2	8.9	446	220	57.2	N47.03936	W113.17004	15N,12W,19C
Hoyt Creek	31-Aug-11	2.7	8.9	410	206	52.5	N47.02270	W113.14390	15N,12W,29C
Hoyt Creek	31-Aug-11	4.3	8.7	375	188	49.8	N47.02404	W113.13259	15N,12W,28C
Jacobsen Spring Creek	9-Aug-11	0.6	6.5	275	195	47.7	N46.99401	W113.06123	14N,12W,1CD
Lincoln Spring Creek	17-Aug-11	3.8	7.6	281	138	49.5	N46.96301	W112.67539	14N,9W,13D
Liverpool Creek	16-Aug-11	4	na	245	172	44.8	N46.99452	W112.66775	14N,8W,6B
Liverpool Creek	16-Aug-11	4.3	na	243	173	46.9	N46.99880	W112.66553	14N,8W,6B
Murphy Spring Creek	30-Aug-11	0.6	8.3	143	71	46.9	N47.04343	W113.00662	15N,11W,21B
Nevada Creek	20-Sep-11	5.3	8.7	382	267	61.7	N47.89553	W112.99860	13N11W,9C
Nevada Creek (Stit's project)	21-Sep-11	29	9	225	157	60.3	N46.80296	W112.81900	12N,10W,11C
Nevada Spring Creek	20-Sep-11	1.1	8.5	382	262	49.3	N46.89839	W112.98492	13N,11W,10B
Nevada Spring Creek	20-Sep-11	3.5	8.3	388	276	49.3	N46.89502	W112.95020	13N,11W,11D
Park Creek	17-Aug-11	1.1	6.7	151	73	49.6	N46.99877	W112.68233	14N,9W,1B
Pearson Creek	7-Sep-11	0.5	8.9	70	35	53.8	N47.01216	W113.24538	15N,13W,33D
Pearson Creek	7-Sep-11	1.1	8.9	62	31	50.2	N47.00516	W113.23421	14N,13W,3B
Poorman Creek	10-Aug-11	1.5	8.7	248	176	47.7	N46.92713	W112.67093	14N,9W,36A
Sauerkraut Creek	15-Aug-11	2.9	6	77	55	58.1	N46.90546	W112.75568	13N,9W,5D
Sauerkraut Creek	15-Aug-11	3.2	6.6	78	55	51.6	N46.90018	W112.75793	13N,9W,8A
Snowbank Creek	23-Aug-11	0.4	8.2	140	70	52.9	N47.07184	W112.61593	15N,8W,9A
Wasson Creek	11-Aug-11	0.1	6.4	293	209	50.9	N46.89363	W112.94876	13N,11W,11D
Wasson Creek	11-Aug-11	2.8	6.1	235	167	54.1	N46.89134	W112.92165	13N,10W,7C
Wasson Creek	11-Aug-11	3	6.5	236	167	58.5	N46.89074	W112.91902	13N,10W,7C

Summary of water chemistry readings for 2012.

Stream name	Date	River Mile	pH	Conductivity (uS)	TDS (ppm)	Temp (°F)	Lat	Long	TRS
Anaconda Creek	13-Aug-12	0.5	8.4	159.2	108	51.1	N47.03483	W112.35326	15N,6W,22C
Ashby Creek	30-Jul-12	2.7	87	344	201	56.3	N46.85028	W113.58542	13N,16W,26D
Ashby Creek	30-Jul-12	4	8.7	347	238	53.8	N46.83203	W113.59608	12N,16W,3A
Blackfoot River (Flesher Pass Section)	6-Sep-12	124.3	8.9	201	143	50	N47.01177	W112.45406	15N,7W,35B
Blackfoot River (Hogum Cr Section)	5-Sep-12	119.6	9	215	124	53.6	N46.98899	W112.51138	14N,7W,5D
Braziel Creek	4-Sep-12	0.2	8.9	126	89	53.8	N46.80819	W112.33969	12N,10W,10D
Copper Creek	20-Aug-12	6.2	8.8	205	146	52	N47.07279	W112.61221	15N,8W,9A
Cottonwood Creek	17-Sep-12	12	7.9	154	77	45.1	N47.12143	W113.30469	16N,14W,24D
Gleason Creek	3-Oct-12	0.1	8.7	161	80	41.5	N46.78444	W112.59582	12N,8W,22B
Grantier Spring Creek	8-Aug-12	1	8.4	276	196	48.4	N46.94056	W112.67442	14N,9W,25A
Jacobsen Spring Creek	22-Aug-12	0.6	8.8	238	169	45.7	N46.99443	W113.05993	14N,12W,1CD
Lincoln Spring Creek	14-Aug-12	3.8	8.4	282	200	53.1	N46.98354	W112.67421	14N,9W,13D
Murphy Spring Creek	28-Aug-12	0.6	8.7	155	112	49.5	N47.04343	W113.00662	15N,11W,21B
Nevada Creek (Stit's project)	10-Sep-12	29	9.3	191.2	136	57.7	N46.80296	W112.81900	12N,10W,11C
Nevada Spring Creek	30-Aug-12	3.5	8.5	338	193	46.8	N46.89498	W112.95019	13N,11W,11D
North Fork Blackfoot River	8-Aug-12	17.2	8.9	184	131	51	N47.10236	W112.96127	16N,11W,35B
Pearson Creek	12-Sep-12	0.5	6.6	63	31	41	N47.01216	W113.24538	15N,13W,33D
Rock Creek	17-Sep-12	1.6	8.3	216	109	55.2	N47.00051	W113.01546	14N,11W,5A
Rock Creek	9-Aug-12	3.9	8.2	144	103	60.6	N47.01229	W112.96445	15N,11W,35B
Sauerkraut Creek	16-Aug-12	2.9	7.7	76	88	56.7	N46.90415	W112.75489	13N,9W,5D
Sauerkraut Creek	16-Aug-12	3.2	6.8	76	37	53.8	N46.90018	W112.75793	13N,9W,8A
Snowbank Creek	14-Aug-12	0.4	8.8	148	105	51.6	N47.07184	W112.61593	15N,8W,9A
Wasson Creek	7-Aug-12	0.1	8.7	213	194	48.4	N46.89363	W112.94876	13N,11W,11D
Wasson Creek	7-Aug-12	2.8	8.7	220	156	55.6	N46.89134	W112.92165	13N,10W,7C
Wasson Creek	7-Aug-12	3	8.5	218	155	61.7	N46.89074	W112.91902	13N,10W,7C
Wilson Creek	7-Aug-12	2.4	8.7	283	169	53.4	N46.88054	W112.89152	13N,10W,17C

Appendix F: Temperature sensor locations in the Blackfoot drainage, 2011.

| Stream Name	Location (stream mile)	Legal Description	Duration	Sensor Type	Recording Interval
Anaconda Creek	0.3	15N,6W,27A	7/6/11-10/18/11	Tidbit	50mins
Blackfoot River above Belmont Creek	21.8	14N,16W,24C	1/1/11-12/31/11	Tidbit	50mins
Blackfoot River @ Cuttoff Rd Bridge	72.2	14N,11W,32D	10/4/11-12/31/11	Tidbit	50mins
Blackfoot River @ Dalton Mtn Rd Bridge	104.5	14N,9W,28B	10/18/11-12/31/11	Tidbit	50mins
Blackfoot River @ Raymond Bridge	60	14N,12W,28D	1/1/11-10/3/11	Tidbit	50mins
Blackfoot River @ Scotty Brown Bridge	46.1	15N,13W,33A	4/26/11-12/31/11	Tidbit	50mins
Blackfoot River @ USGS Gage Station	7.9	13N,17W,9B	1/1/11-12/31/11	Tidbit	50mins
Copper Creek @ Sucker Creek Rd Bridge	1.1	15N,8W,25C	1/1/11-12/31/11	Tidbit	50mins
Cottonwood Creek @ Hwy 200	1	15N,13W,29B	4/26/11-12/31/11	Tidbit	50mins
Gold Creek	1.6	14N,16W,30C	1/1/11-10/4/11	Tidbit	50mins
Monture Creek @ FAS	1.8	15N,13W,22D	1/1/11-12/31/11	Tidbit	50mins
Nevada Creek above Nevada Spring Creek	6.3	13N,11W,9C	4/20/11-9/27/11	Tidbit	50mins
Nevada Creek below Nevada Spring Creek	5	13N,11W,8D	4/20/11-9/27/11	Tidbit	50mins
Nevada Spring Creek @ mouth	0.1	13N,11W,9C	4/20/11-9/27/11	Tidbit	50mins
North Fork Blackfoot River	2.6	14N,12W,10D	1/1/11-12/31/11	Tidbit	50mins
Snowbank Creek	0.4	15N,8W,9B	7/6/11-10/18/11	Tidbit	72mins
Wasson Creek @ mouth	0.1	13N,11W,11D	4/20/11-9/28/11	Tidbit	50mins

## Temperature sensor locations in the Blackfoot drainage, 2012.

Stream Name	Location (stream mile)	Legal Description	Duration	Sensor Type	Recording Interval
Anaconda Creek	0.3	15N,6W,27A	7/5/12-10/9/12	Hobo	72mins
Beartrap Creek	0.2	15N,6W,28A	7/5/12-10/9/12	Hobo	72mins
Belmont Creek @ mouth	0.1	14N,16W,24C	7/10/12-10/10/12	Tidbit	50mins
Blackfoot River above Belmont Creek	21.8	14N,16W,24C	1/1/12-10/10/12	Tidbit	50mins
Blackfoot River @ Cuttoff Rd Bridge	72.2	14N,11W,32D	1/1/12-10/9/12	Tidbit	50mins
Blackfoot River @ Dalton Mtn Rd Bridge	104.5	14N,9W,28B	1/1/12-10/9/12	Tidbit	50mins
Blackfoot River above Pass Creek	130.8	15N,6W,20A	7/5/12-10/9/12	Hobo	72mins
Blackfoot River @ Raymond Bridge	60	14N,12W,28D	7/10/12-10/9/12	Tidbit	50mins
Blackfoot River @ Scotty Brown Bridge	46.1	15N,13W,33A	1/1/12-10/10/12	Tidbit	50mins
Blackfoot River above Shave Creek	131.8	15N,6W,21C	7/5/12-10/9/12	Hobo	72mins
Blackfoot River @ USGS Gage Station	7.9	13N,17W,9B	1/1/12-10/10/12	Tidbit	50mins
Copper Creek @ Sucker Creek Rd Bridge	1.1	15N,8W,25C	1/1/12-10/9/12	Tidbit	50mins
Cottonwood Creek @ Hwy 200	1	15N,13W,29B	1/1/12-10/10/12	Tidbit	50mins
Ender's Spring Creek	0.1	14N,11W,6B	7/5/12-10/11/12	Hobo	72mins
Gold Creek	1.6	14N,16W,30C	7/10/12-10/10/12	Tidbit	50mins
Grantier Spring Creek	0.1	14N,9W,25A	4/2/12-10/9/12	Tidbit	50mins
Grantier Spring Creek	1	14N,9W,25A	4/3/12-10/9/12	Tidbit	50mins
Hoyt Creek	1.2	15N,12W,19C	7/5/12-10/10/12	Hobo	72mins
Hoyt Creek	4.3	15N,12W,28C	7/5/12-10/10/12	Hobo	72mins
Jacobsen's Spring Creek	0.1	14N,12W,1C	7/5/12-10/10/12	Hobo	72mins
Kleinschmidt Creek	0.1	14N,11W,6A	7/5/12-10/10/12	Hobo	72mins
Monture Creek @ FAS	1.8	15N,13W,22D	1/1/12-10/10/12	Tidbit	50mins
Nevada Creek above Nevada Spring Creek	6.3	13N,11W,9C	4/5/12-10/9/12	Tidbit	50mins
Nevada Creek below Nevada Spring Creek	5	13N,11W,8D	4/5/12-10/9/12	Tidbit	50mins
Nevada Spring Creek @ mouth	0.1	13N,11W,9C	4/5/12-10/9/12	Tidbit	50mins
Nevada Spring @ lower bridge	1.1	13N,11W,10B	4/5/12-10/9/12	Hobo	72mins
Nevada Spring @ lower fenceline	2.1	13N,11W,10A	4/5/12-10/1/12	Hobo	72mins
Nevada Spring @ upper fenceline	3.5	13N,11W,11C	4/5/12-10/1/12	Hobo	72mins
North Fork Blackfoot River	2.6	14N,12W,10D	1/1/12-10/9/12	Tidbit	50mins
Poorman Creek	0.25	14N,9W,25B	4/3/12-10/9/12	Tidbit	50mins
Shave Creek	0.1	15N,6W,21B	7/5/12-9/29/12	Hobo	72mins
Snowbank Creek	0.2	15N,8W,9A	7/5/12-10/1/12	Hobo	72mins
Wasson Creek @ mouth	0.1	13N,11W,11D	4/5/12-10/9/12	Tidbit	50mins

Anaconda Creek (Mile 0.3) - 2011
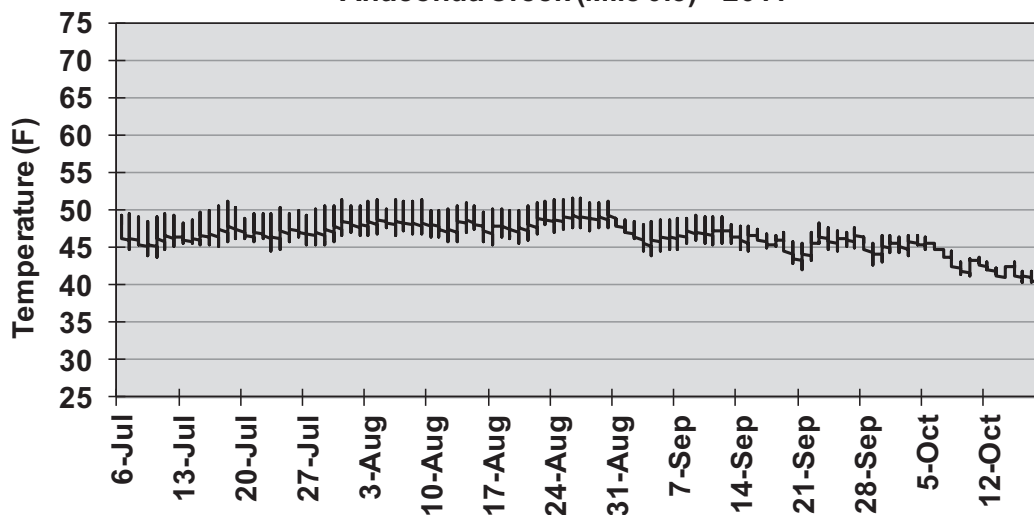
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	51.2	43.7	47	1.6	2.4
August	51.6	45.4	48.3	1.45	1.9
September	49.2	41.9	46	1.4	1.9
October	46.6	39.2	43.2	1.8	3.3

Blackfoot River above Belmont Creek (Mile 21.8) - 2011
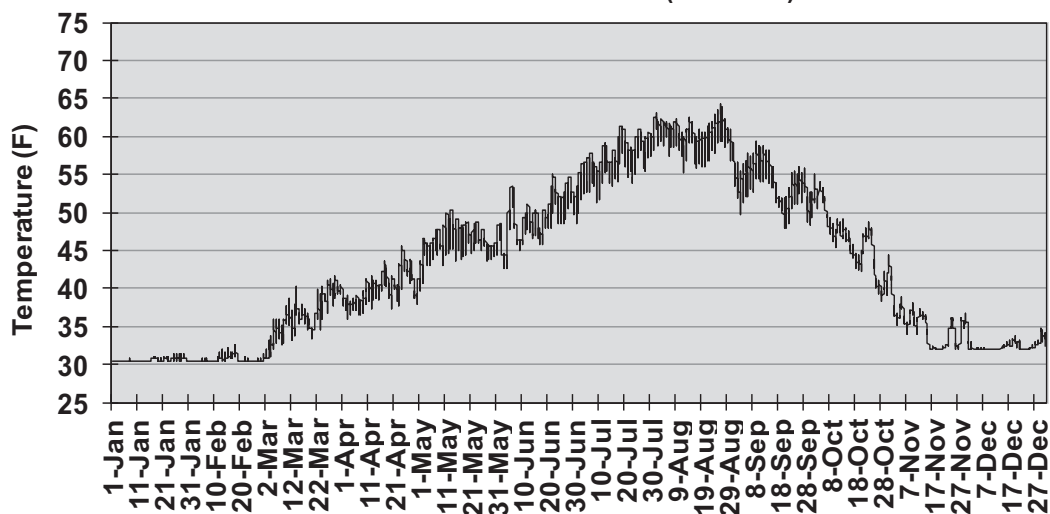
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	31.4	30.5	30.6	0.2	0.04
February	32.5	30.5	30.6	0.4	0.14
March	41.6	30.5	36.1	2.8	7.7
April	45.5	36	40	1.9	3.6
May	50.2	40	45.9	1.8	3.2
June	54.9	42.7	49.1	2.6	6.9
July	62.6	48.5	56.5	2.6	6.9
August	64.3	55.2	60	1.7	2.8
September	59.4	48	53.7	2.5	6.2
October	54.9	38.3	46.2	3.8	14.5
November	42.9	32	35	2.3	5.1
December	35.5	32	32.5	0.6	0.4

Blackfoot River at Cutoff Bridge (mile 72.2) - 2011
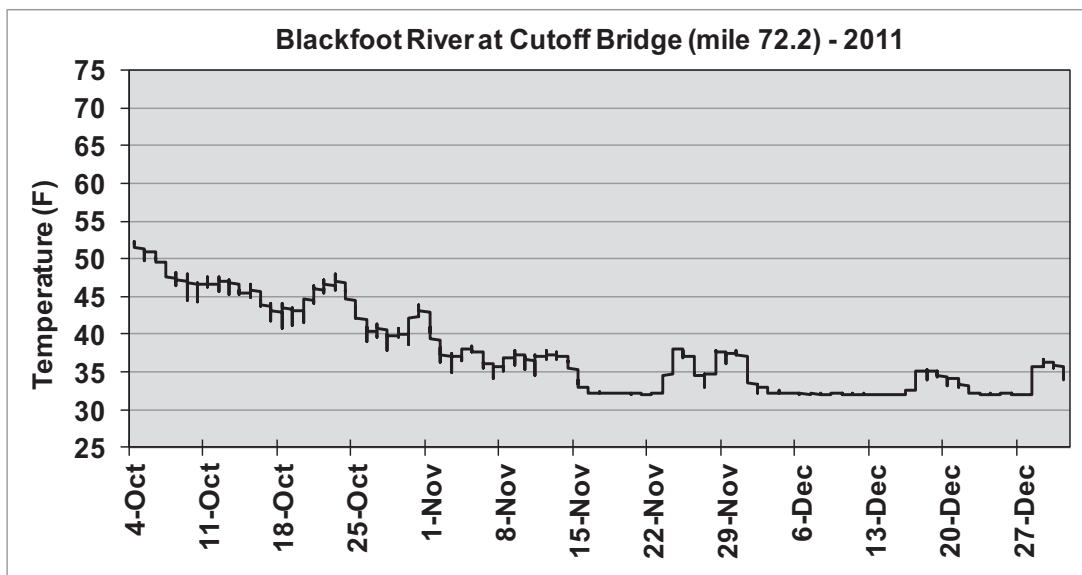
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
October	52.2	37.8	44.8	3.2	10.2
November	43	32	35.4	2.4	5.6
December	37.1	31.9	31.9	1.3	1.7

Blackfoot River at Dalton Mtn Rd (mile 104.5) - 2011
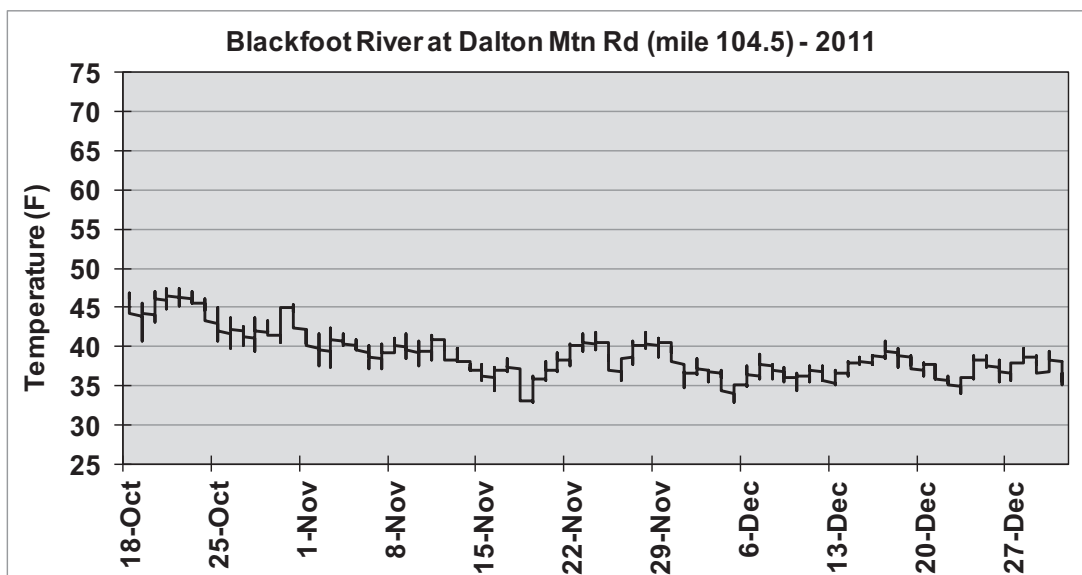
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
October	47.5	39.3	43.7	2.1	4.6
November	42.3	33	38.8	1.9	3.6
December	40.7	32.8	37	1.4	1.8

Blackfoot River at Raymond Bridge (Mile 60) - 2011
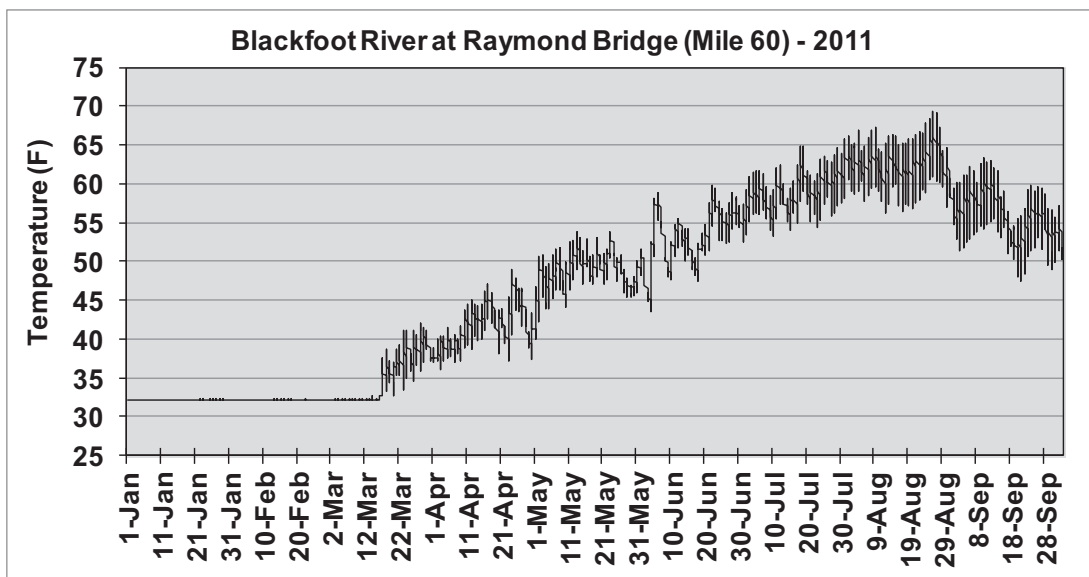
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	32.3	32	32.1	0.05	0.002
February	32.3	32	32.1	0.05	0.002
March	42	32.1	34.7	3	9.2
April	49	36.1	41.4	2.7	7.4
May	53.9	39.9	48.6	2.4	5.8
June	59.7	43.6	52.7	3.5	12
July	65.8	52.6	58.7	2.6	7
August	69.3	56.3	62.3	2.8	7.9
September	63.3	47.5	55.8	3.3	10.9
October	57.2	49.9	53.1	2	4.1

Blackfoot River at Scotty Brown Bridge (Mile 46.1) - 2011
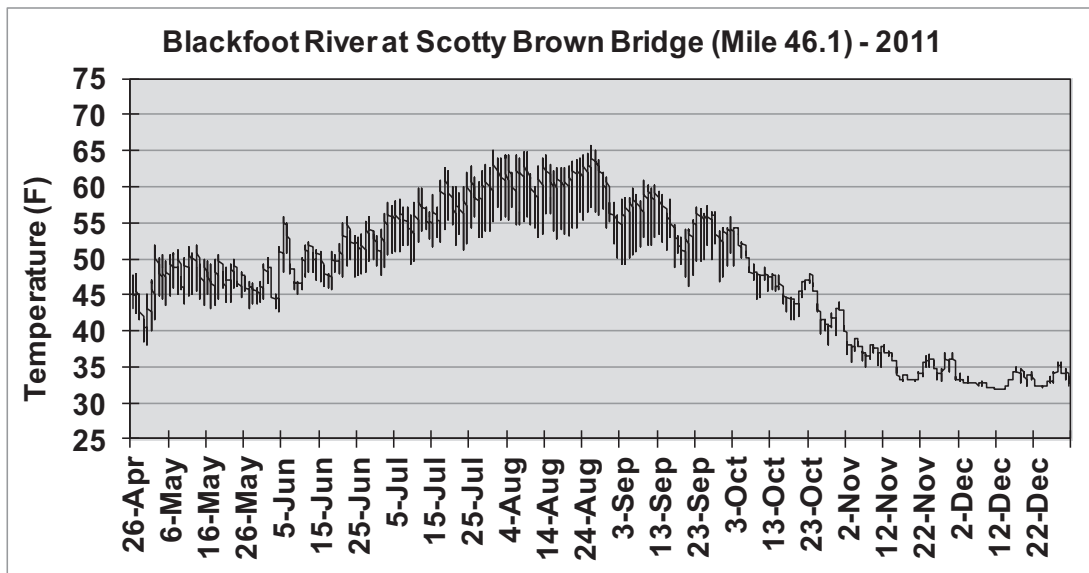
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	47.9	38.1	42.9	2.7	7.3
May	51.8	40.1	46.8	2.2	4.9
June	55.8	42.6	49.5	2.8	7.8
July	65.1	47.7	55.9	3.3	10.8
August	65.7	52.7	59.5	3.1	9.6
September	60.8	46.3	54.1	3.1	9.7
October	55.8	38.1	46.1	3.8	14.5
November	43	33	35.9	1.9	3.6
December	35.9	31.8	33.1	0.9	0.9

Blackfoot River at USGS Gauge Station (Mile 7.9) - 2011
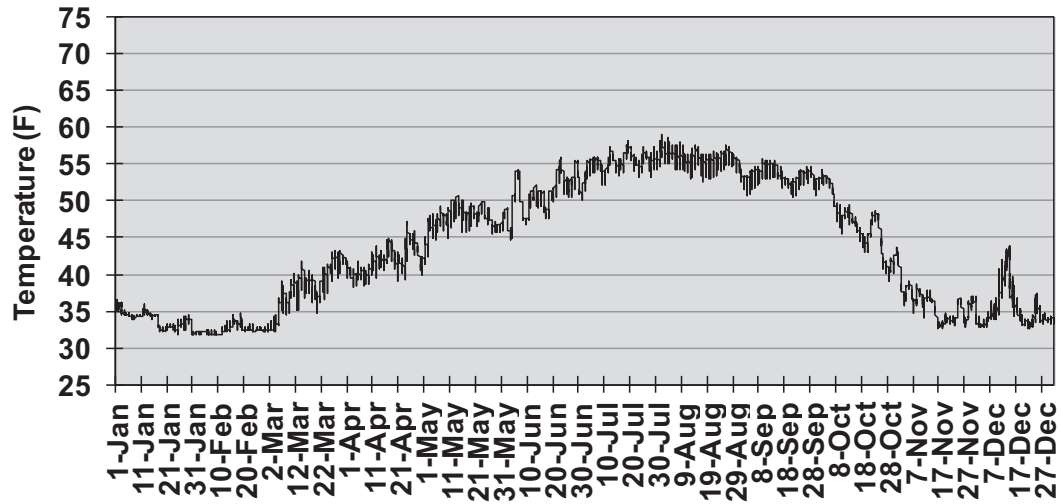
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	36.5	31.9	33.8	1.1	1.2
February	34.8	31.9	32.5	0.6	0.3
March	43.2	32.2	37.9	2.9	8.2
April	47.1	38.2	41.7	1.8	3.2
May	50.7	41.3	47.3	1.6	2.5
June	56	44.6	50.5	2.4	6
July	58.2	50.2	55	1.4	2
August	59.1	52.7	55.5	1.2	1.5
September	55.7	50.5	53.2	1.1	1.2
October	54.3	39.2	47	3.8	14.4
November	43	32.7	35.8	2.1	4.4
December	43.9	32.6	35.2	2.6	6.5

Copper Creek at Sucker Creek Rd bridge (Mile 1.1) - 2011
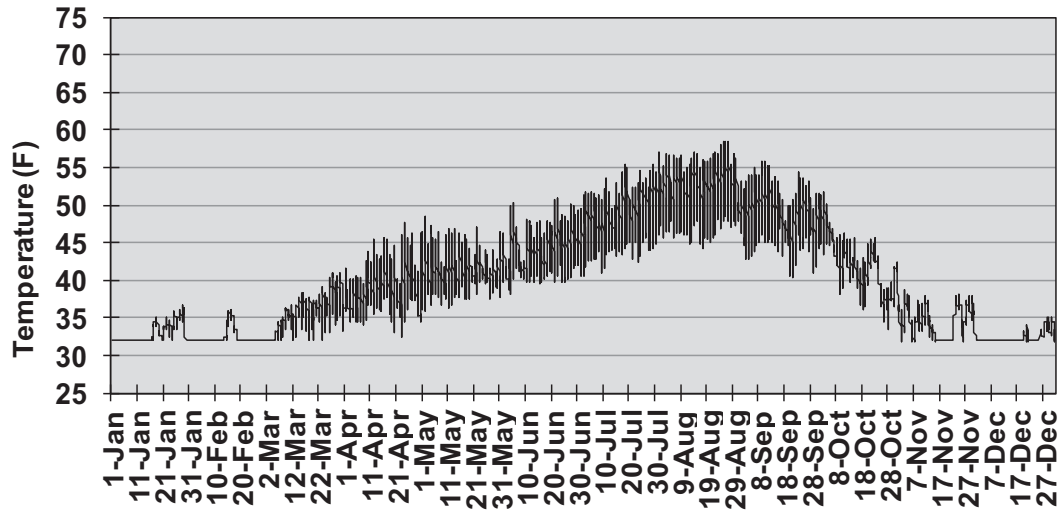
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	36.7	32.1	32.8	1.2	1.5
February	36.1	32.1	32.4	0.9	0.8
March	40.9	32.1	35.1	2.3	5.2
April	47.6	32.4	38.6	3.1	9.4
May	48.5	35.8	41	2.6	6.8
June	51	38.1	43.5	2.9	8.6
July	57.1	40.6	47.8	3.6	12.8
August	58.5	44.3	51.3	3.5	12.2
September	55.7	40.4	48.1	3.4	11.6
October	51.8	33.5	42.2	3.7	13.4
November	38.7	31.8	34.4	2	4.1
December	35	31.8	32.4	0.7	0.5

Cottonwood Creek at Hwy 200 (Mile 1.0) - 2011
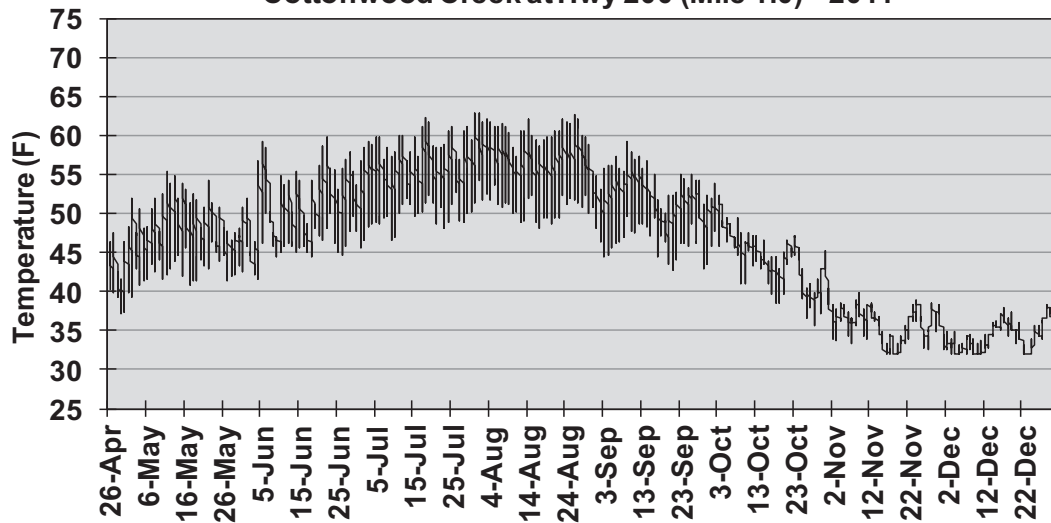
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	47.5	37.1	41.8	2.8	8.1
May	55.3	39.4	46.7	3.3	10.7
June	59.8	41.6	49.9	3.8	14.6
July	63	45.6	54.4	3.6	13.3
August	63	48.1	55.7	3.5	12.4
September	59.3	42.8	50.6	3.3	11.1
October	53.9	35.7	44.3	3.5	12.5
November	41.4	32	35.9	2	4.2
December	38.3	32	34.3	1.7	3

Gold Creek (Mile 1.6) - 2011
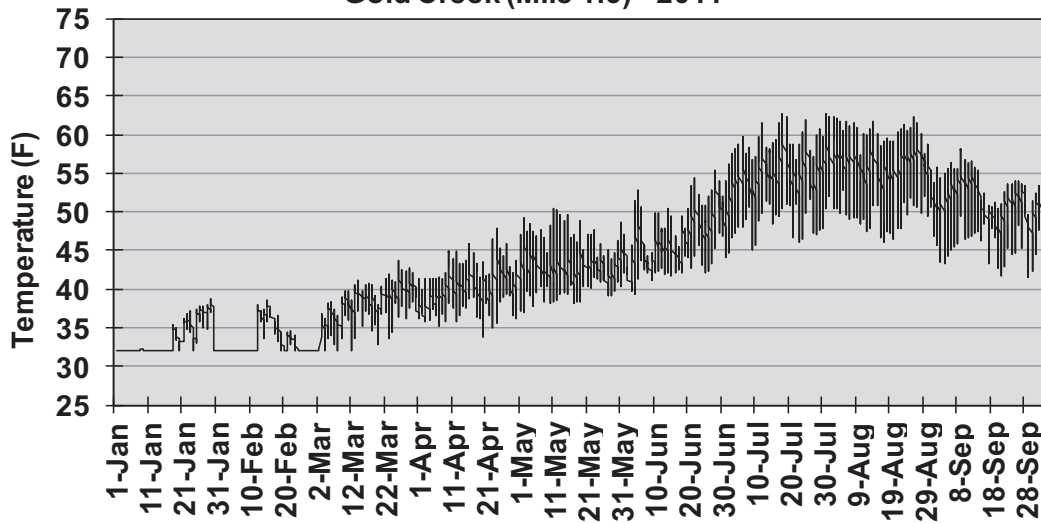
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	38.7	32	33.3	1.9	3.6
February	38.5	32	33.1	1.8	3.3
March	43.7	32.1	37.3	2.7	7.5
April	47.9	33.9	39.8	2.7	7.1
May	50.4	36.9	42.9	2.9	8.6
June	55.3	39.4	46	3.4	11.6
July	62.7	44	53.7	4.1	17.3
August	62.4	46	54.9	4	15.8
September	58.1	41.5	49.9	3.5	12.6
October	53.4	44.6	49.3	2.4	6

Monture Creek at FAS (Mile 1.8) - 2011
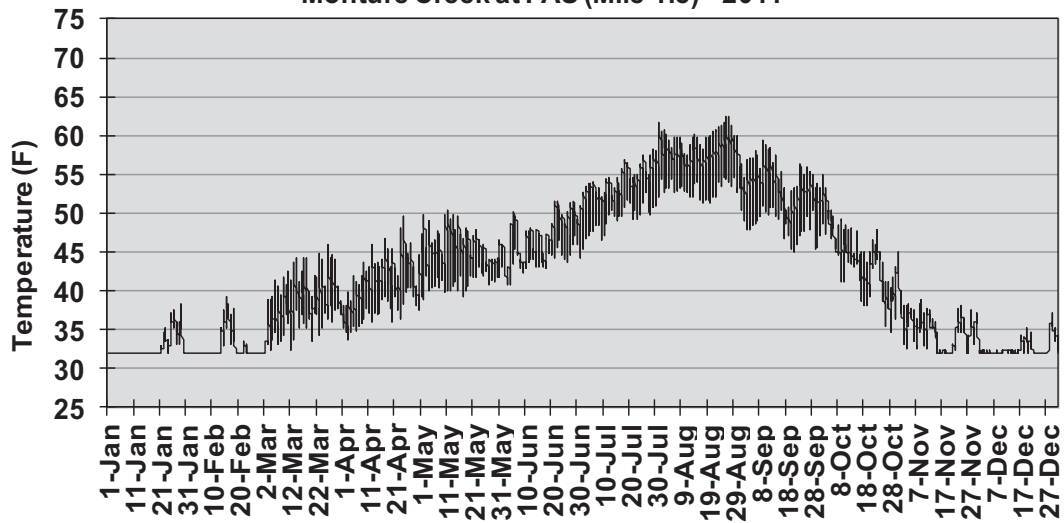
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	38.4	32	32.7	1.4	1.9
February	39.2	32	32.8	1.7	2.8
March	45.9	32	38	3.1	9.6
April	49.6	33.8	40.2	3.1	9.4
May	50.4	38.9	43.9	2.4	5.7
June	51.5	40.9	45.9	2.5	6
July	61.7	44.3	52	2.8	8
August	62.5	51.2	56.5	2.8	6.1
September	59.4	45.1	51.9	3	9.2
October	54.9	34.6	44.2	4	15.8
November	40	32	34.8	2	4.2
December	37.2	32	32.6	1	1.1

Nevada Creek above Nevada Spring Creek (Mile 6.3) - 2011
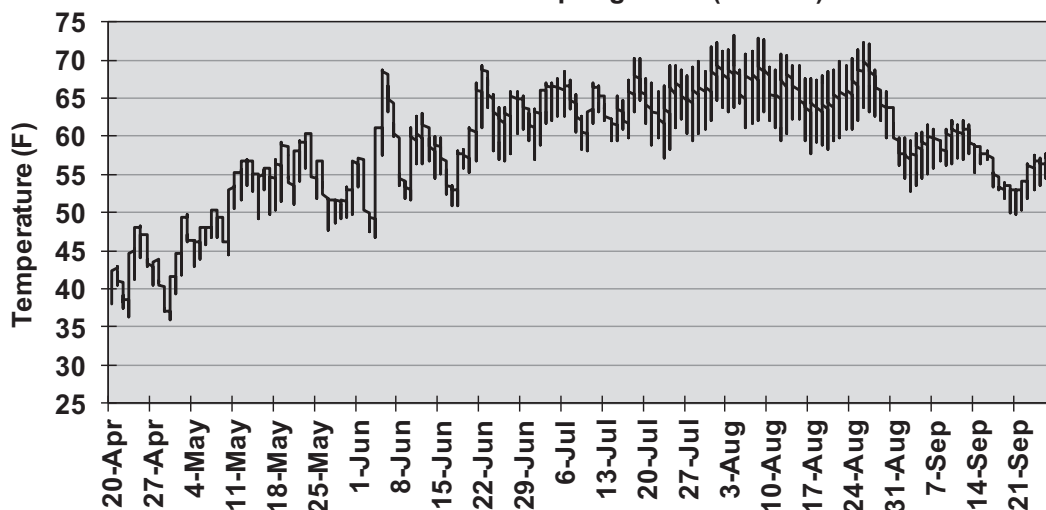
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	48.3	36	41.5	3.1	9.4
May	60.3	39.3	51.4	4.4	19.3
June	69.3	46.6	58.5	5	24.7
July	71.7	56.9	63.9	2.9	8.4
August	73.2	57.8	65.5	3.3	11.2
September	62	49.7	56.5	2.8	8

Nevada Creek below Nevada Spring Creek (Mile 5.0) - 2011
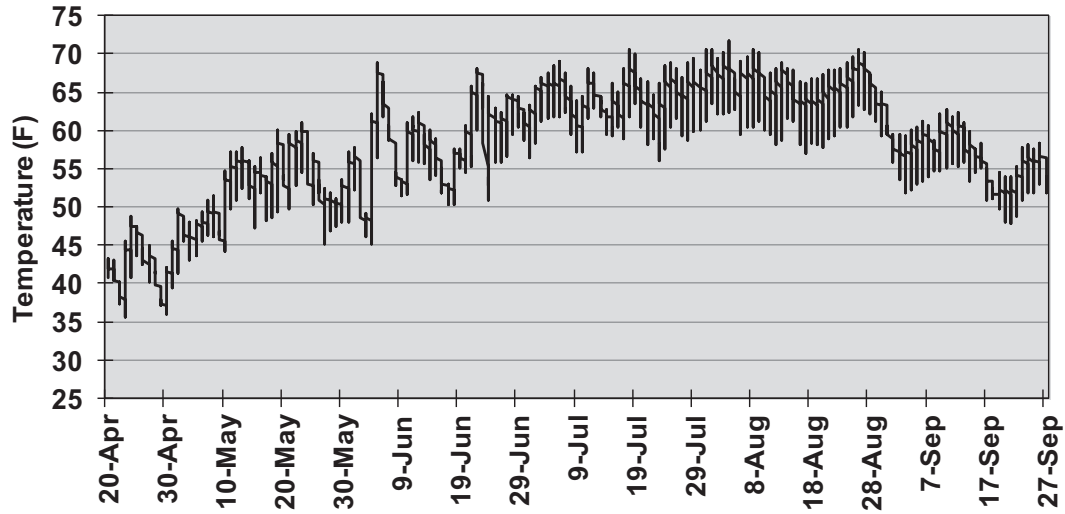
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	48.9	35.6	41.7	3.2	10
May	60.9	39.3	51.1	4.4	19
June	68.8	45.2	57.6	5	24.8
July	70.6	56.1	63.4	3.1	9.3
August	71.8	57	64.3	3.2	10.1
September	62.7	47.7	55.6	3.1	9.9

Nevada Spring Creek at mouth (Mile 0.1) - 2011
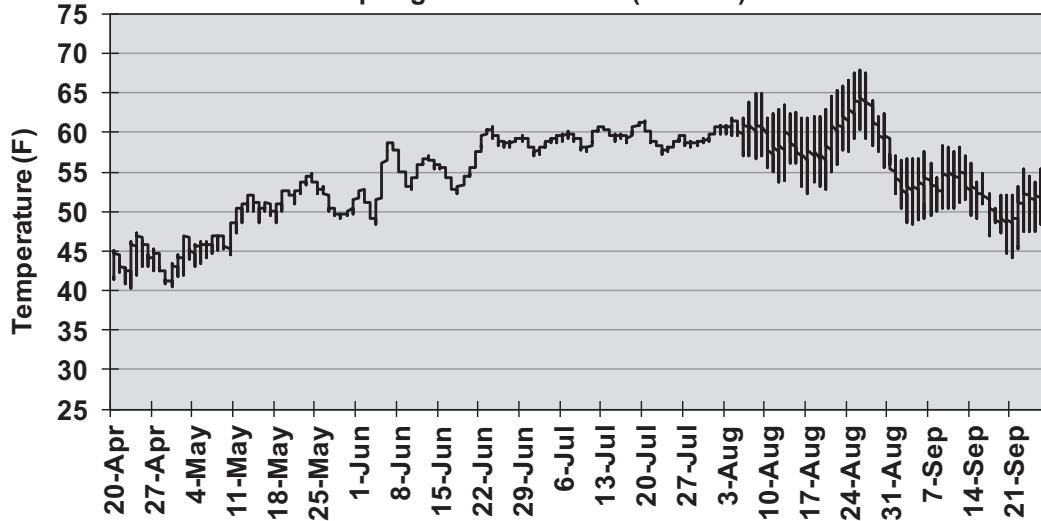
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	47.3	40.3	43.1	1.6	2.6
May	54.8	41.7	49.1	3.2	10.2
June	60.7	48.4	55.7	2.9	8.7
July	61.5	57	59.1	0.9	0.9
August	67.9	52.2	59.7	3.1	9.5
September	58.4	44.2	51.9	2.8	8.1

North Fork Blackfoot River at Ovando-Helmville Rd (Mile 2.6) - 2011
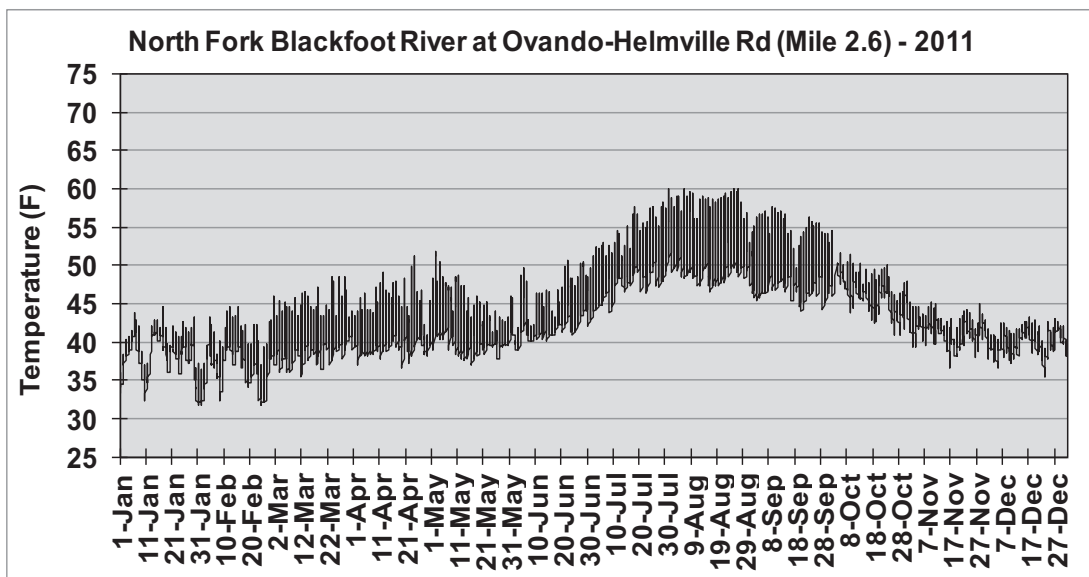
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	44.6	31.8	38.8	2.5	6.4
February	44.6	31.8	38	2.8	8
March	48.5	35.5	40.8	2.7	7.5
April	51.3	36.7	41.6	2.7	7.5
May	51.8	37	42.2	2.7	7.5
June	50.7	39	43.9	2.7	7.2
July	60	42.6	50.6	3.3	10.9
August	60	46.5	52.9	3.5	12.1
September	57.7	44	50.1	3.2	102
October	54.6	40.8	46.5	2.4	5.7
November	45.2	36.7	41.5	1.5	2.4
December	43.3	35.3	40	1.5	2.3

Snowbank Creek (Mile 0.4) - 2011
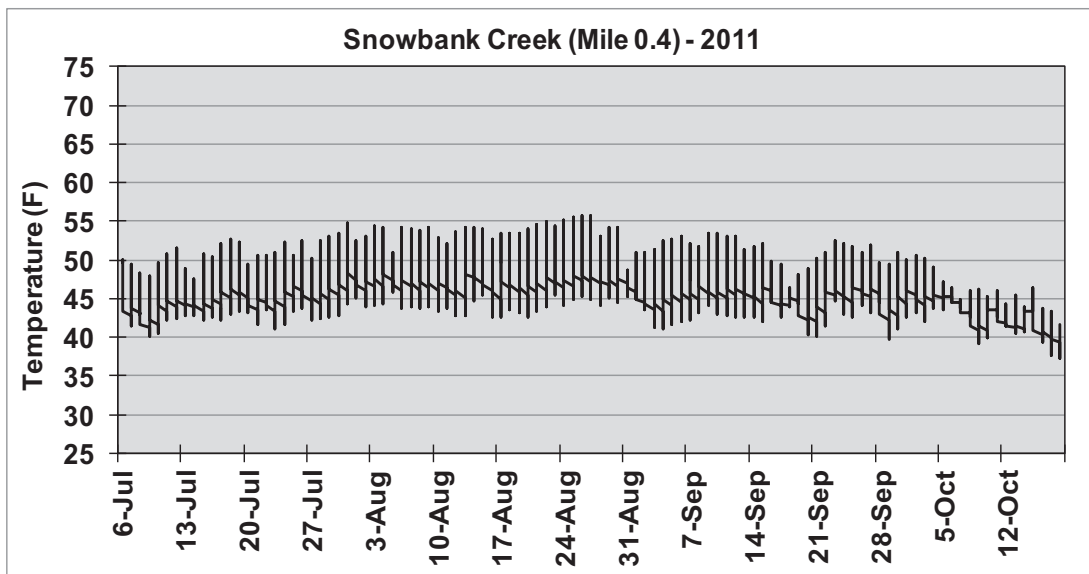
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	54.8	40.1	45.8	3.2	10
August	55.7	42.5	48	3.4	11.6
September	53.5	39.8	46.1	3.2	10.2
October	50.7	37.3	43.4	2.6	6.9

Wasson Creek at mouth (Mile 0.1) - 2011
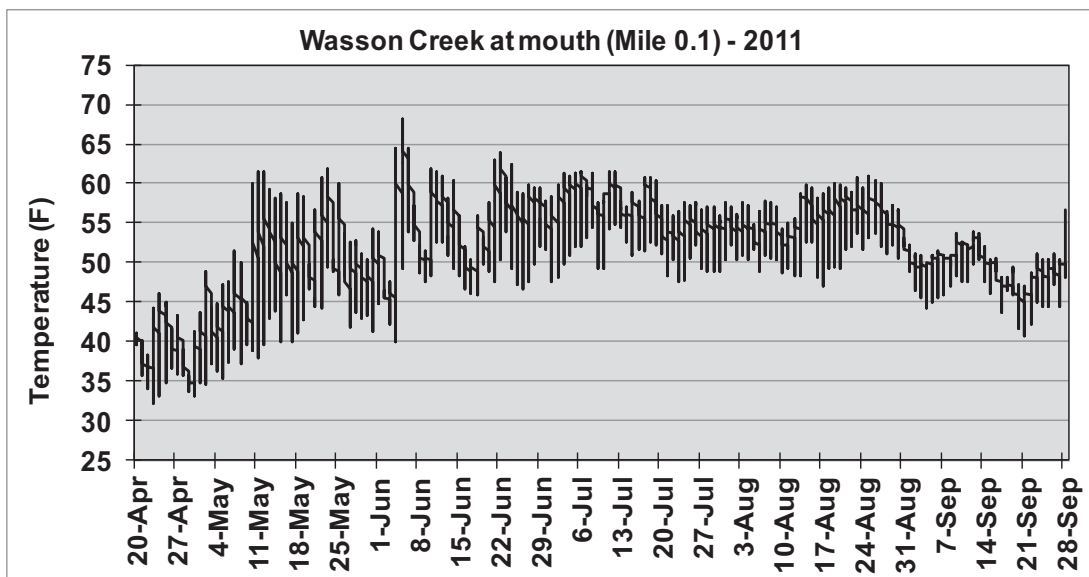
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	46.1	32.2	38.3	3.3	10.6
May	61.8	34.5	47.5	6.2	37.8
June	68.2	40	53.2	5.2	27.1
July	61.5	47.5	55.1	3.4	11.4
August	60.9	46.9	54.4	2.9	8.7
September	56.7	40.8	48.6	2.6	6.5

Anaconda Creek (Mile 0.3) - 2012
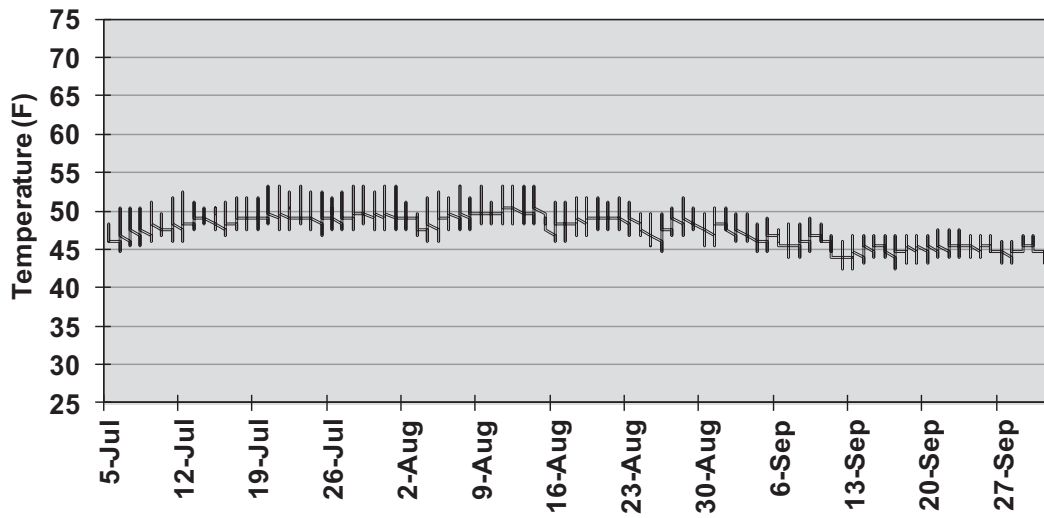
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	53.2	44.7	49	1.7	3
August	53.2	44.7	49	1.7	2.8
September	50.4	42.5	45.6	1.5	2.3
October	44.7	43.2	43.9	0.6	0.3

Beartrap Creek (Mile 0.1) - 2012
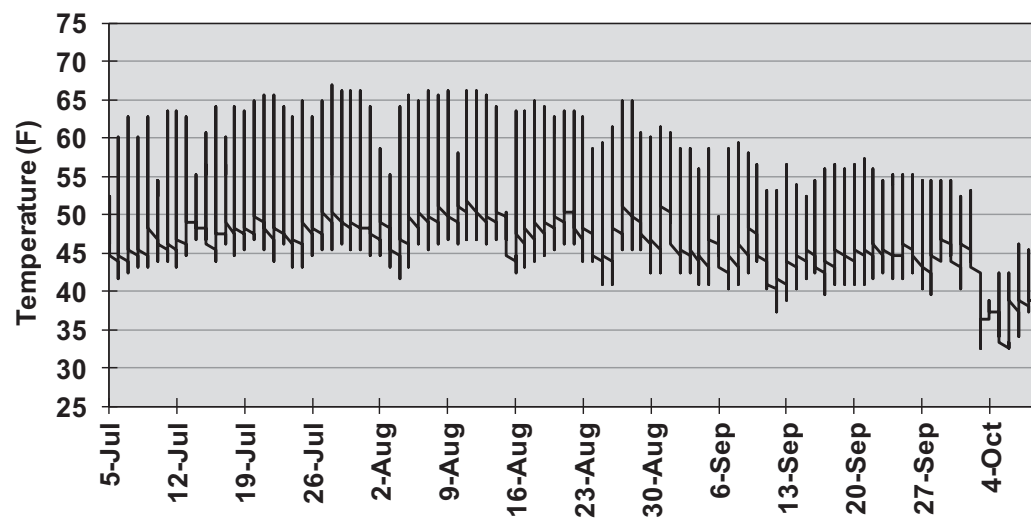
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	67	41.7	51.4	6.4	40.3
August	66.3	41	51.6	6.4	41.2
September	60.8	37.2	47.2	5.1	26.1
October	53.2	32.5	39.7	4.9	24.3

Belmont Creek (Mile 0.1) - 2012
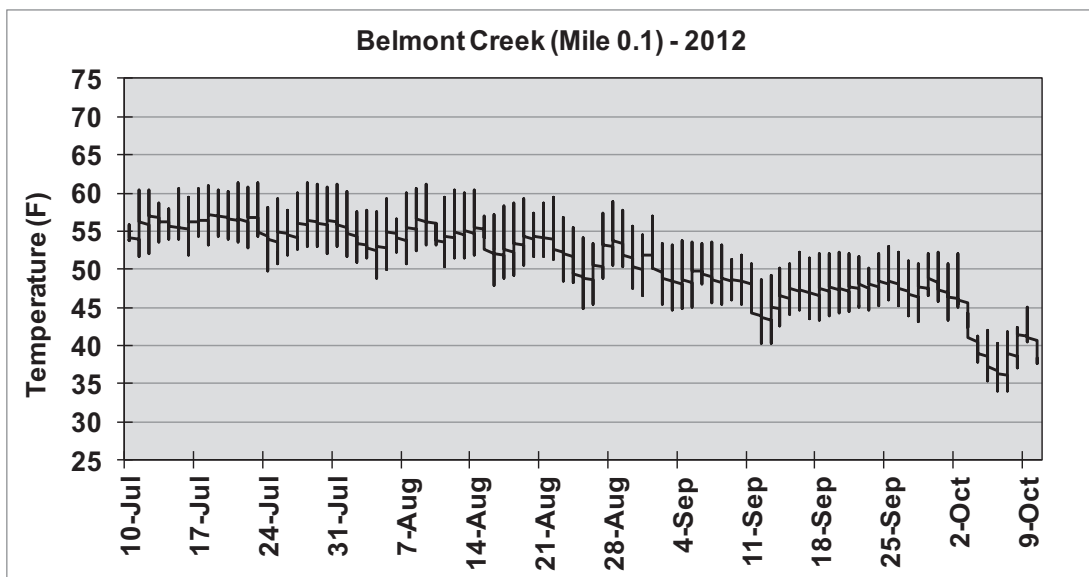
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	61.3	49.8	56	2.4	5.9
August	61.2	44.8	53.6	3.2	10.1
September	57	40.2	48	2.9	8.4
October	52	34	41.1	4.1	17.2

Blackfoot River upstream of Belmont Creek (Mile 21.8) - 2012
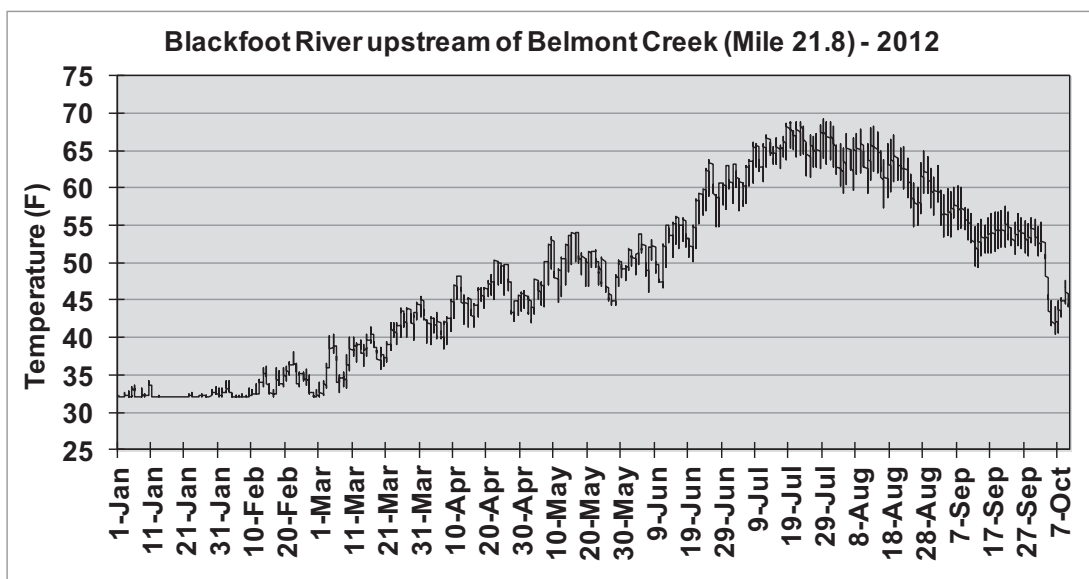
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	34.1	32	32.3	0.4	0.1
February	38.1	32	33.6	1.4	2.1
March	45.4	32	38.6	3.1	9.9
April	50.3	38.4	44.4	2.7	7.3
May	54	42	48.2	2.7	7.3
June	63.7	46.1	53.9	4.1	16.7
July	69.2	56.9	64.3	2.7	7.2
August	68.3	54.8	62.5	2.7	7.5
September	63	49.3	54.8	2.4	5.7
October	55.3	40.4	46.5	4.2	18

Blackfoot River at Cutoff Rd Bridge (Mile 72.2) - 2012
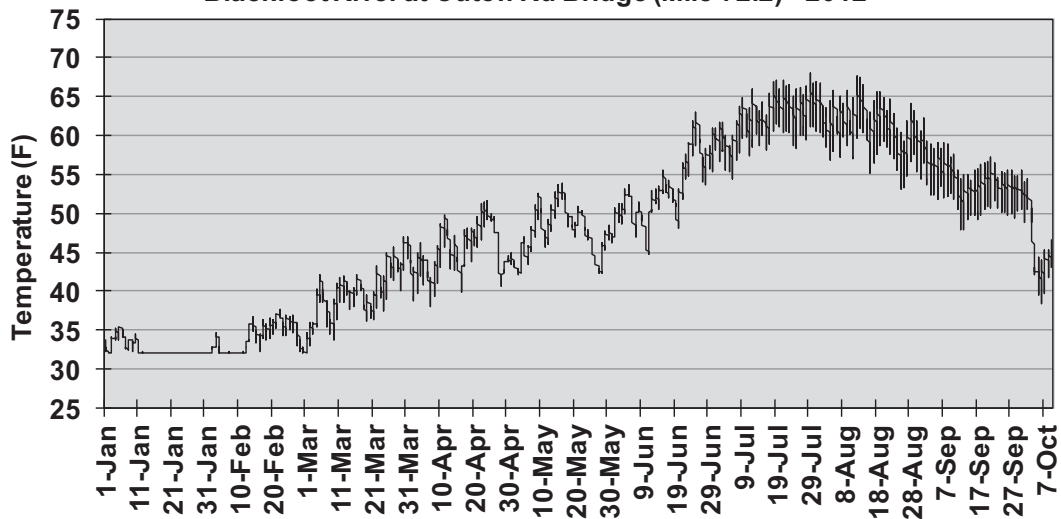
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	35.5	32	32.5	0.9	0.8
February	38	32	33.8	1.7	2.9
March	47	32	39.5	3.1	9.6
April	51.5	38.1	45	3.1	9.5
May	54	42.2	47.4	3	8.9
June	63	45	53	3.9	15
July	68.1	54.5	61.7	2.8	7.8
August	67.6	53.2	61	2.9	8.2
September	62.2	48	53.9	2.6	6.6
October	54.4	38.4	45.7	4.4	19.4

Blackfoot River at Dalton Mtn Rd (Mile 104.5) - 2012
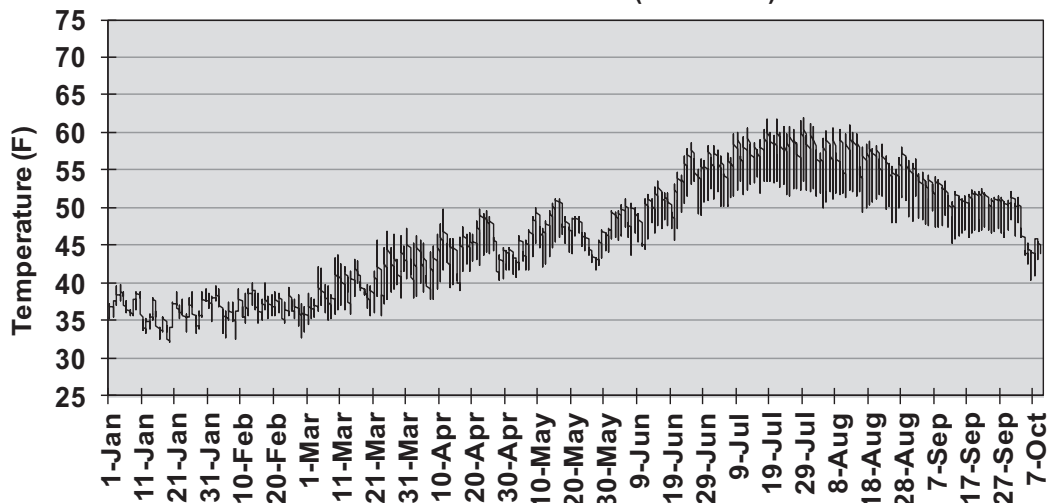
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	39.7	32	36	1.7	2.9
February	40	32.5	37	1.5	2.2
March	47.1	34.6	40	2.7	7.5
April	49.8	37.7	43.4	2.7	7.2
May	51.2	40.7	45.4	2.3	5.2
June	58.6	43.8	50.1	3.3	10.7
July	62	50.1	55.8	2.7	7.5
August	61	48	54.5	2.9	8.2
September	56.5	45.3	50	2.1	4.6
October	51.4	40.4	45.4	2.7	7.4

Blackfoot River upstream of Pass Creek (Mile 130.8) - 2012
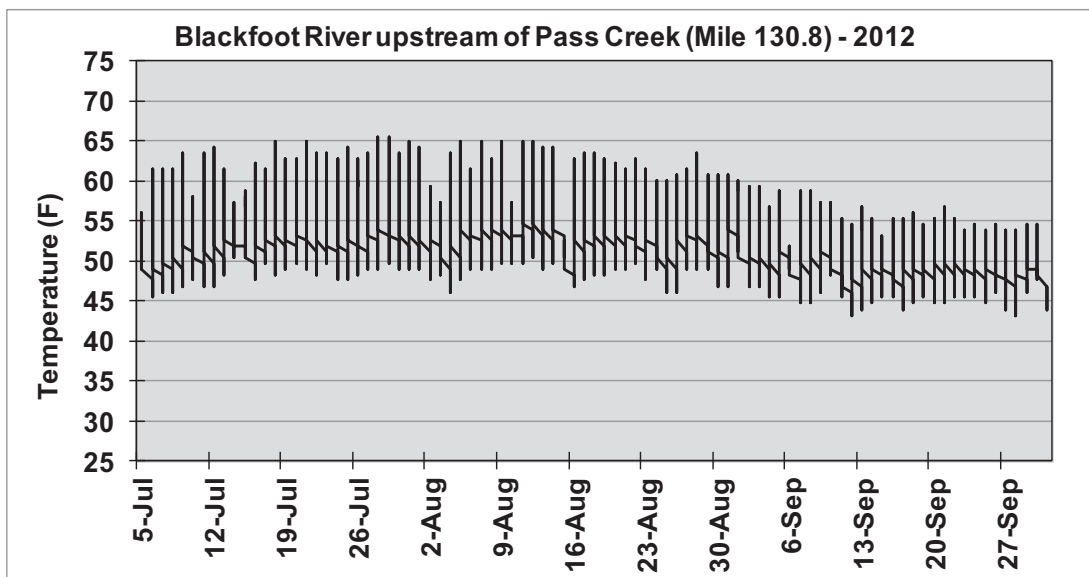
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	65.6	45.4	54.1	5.1	26.5
August	65	46.1	54.1	4.9	23.8
September	60.1	43.2	50	3.8	14.4

Blackfoot River at Raymond Bridge (Mile 60) - 2012
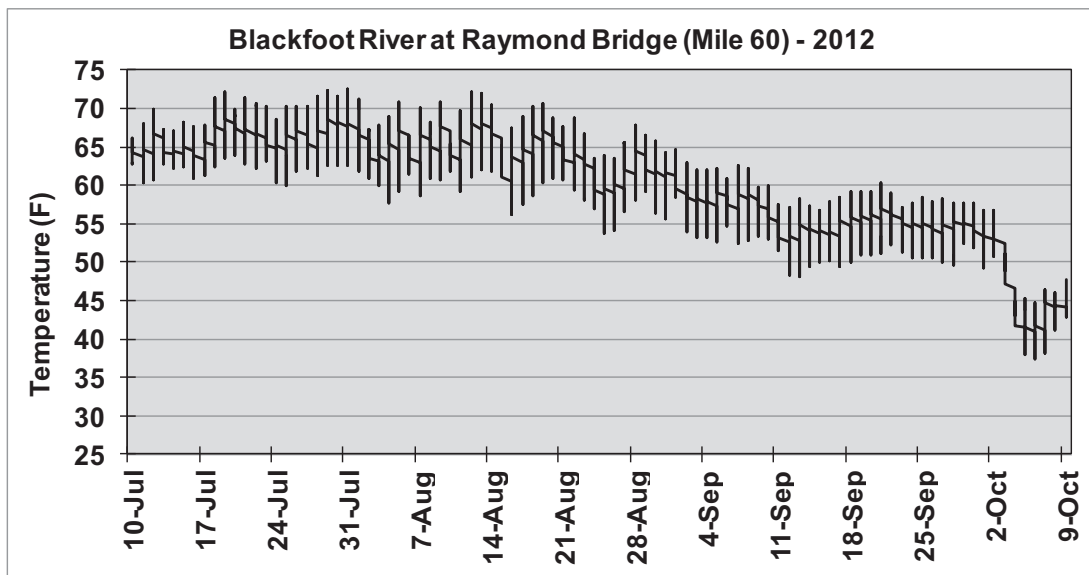
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	72.5	59.9	65.9	3	9
August	72.1	53.8	63.6	3.8	14.3
September	64.7	48	55.4	3.3	10.7
October	56.7	37.4	46	5.1	26.5

Blackfoot River at Scotty Brown Bridge (Mile 46.1) - 2012
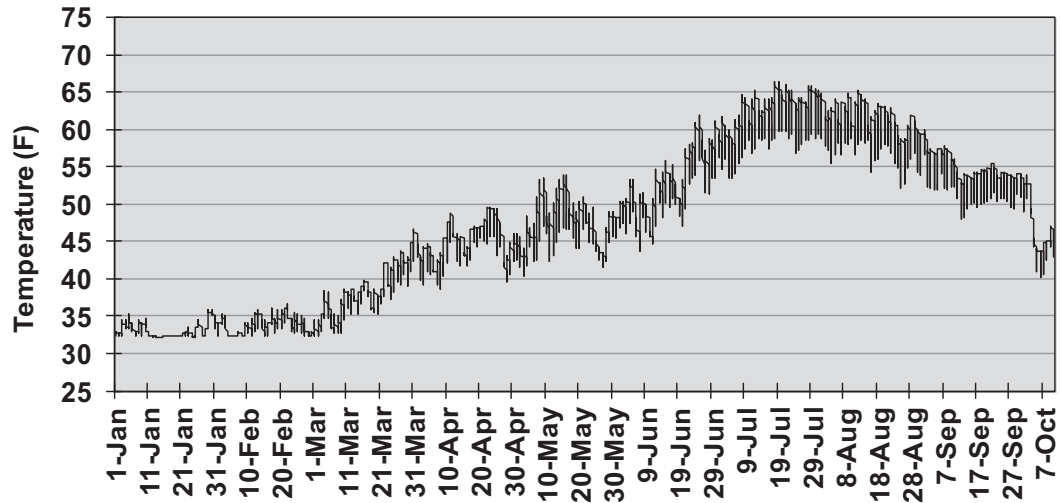
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	35.9	32.1	33	0.91	0.84
February	36.7	32.4	33.6	1.1	1.2
March	46.5	32.4	37.7	3.1	9.5
April	49.6	38.7	44.2	2.5	6.1
May	53.8	40.4	46.7	2.9	8.4
June	62	43.8	51.9	3.9	15.5
July	66.3	53.5	60.7	2.9	8.2
August	65.1	52.1	59.7	2.6	6.9
September	60	48	53.5	2.2	5
October	53.8	40.1	46.2	3.8	14.3

Blackfoot River upstream of Shave Creek (Mile 131.8) - 2012
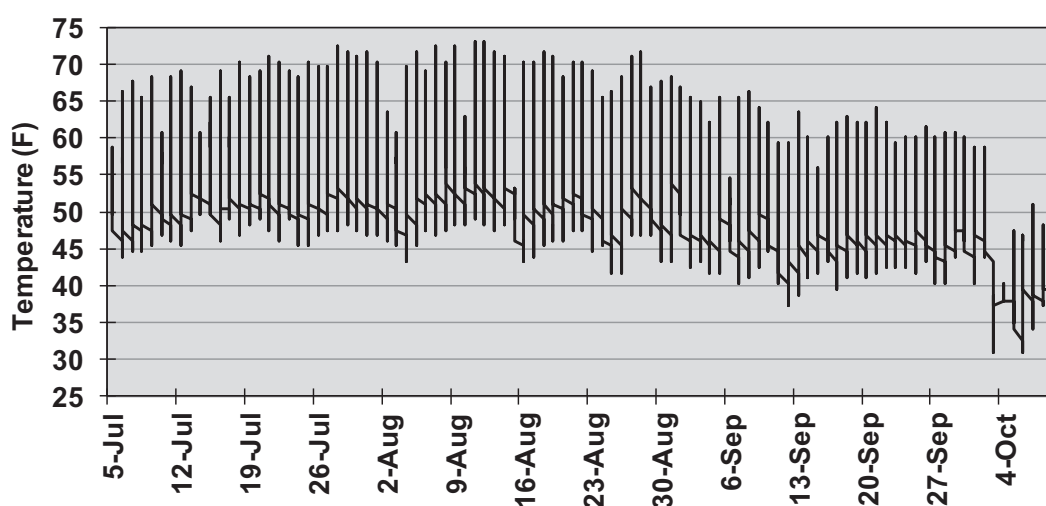
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	72.5	44	55.1	7.5	56.2
August	73.2	41.7	55.2	8.3	68.3
September	67	37.2	49.8	7.1	50.3
October	58.7	30.9	41.2	6.3	40

Blackfoot River at USGS Gauge Station (Mile 7.9) - 2012
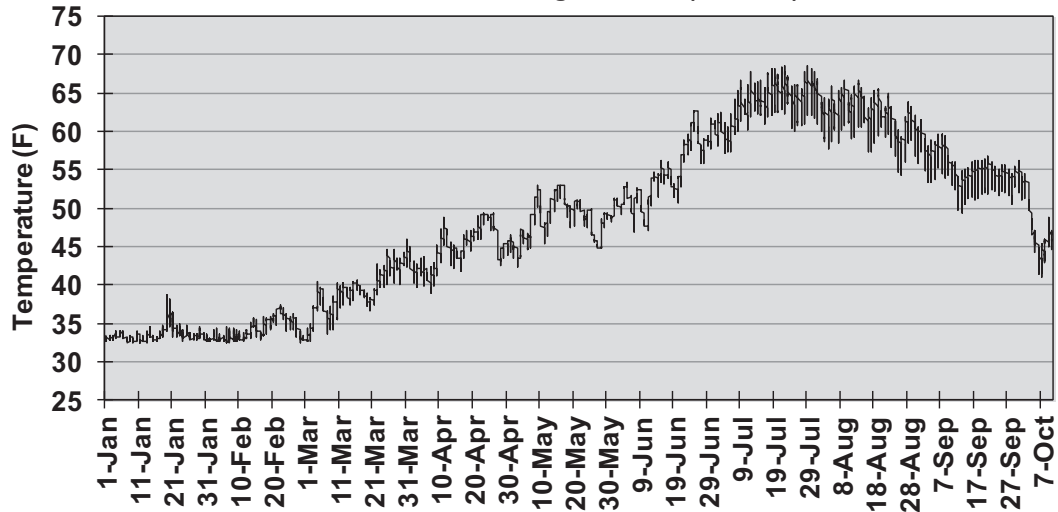
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	38.7	32.4	33.5	1	0.9
February	37.3	32.5	34.1	1.3	1.6
March	45.9	32.6	39	2.8	8.1
April	49.3	39	44.6	2.6	6.6
May	60	42.3	48.3	2.5	6.3
June	62.7	47	54	3.8	14.8
July	68.5	57.2	63.3	2.6	6.7
August	66.8	54.3	61.6	2.5	6.3
September	61.6	49.3	54.8	2.2	4.9
October	54.6	41	47.6	3.7	13.7

Copper Creek at Sucker Creek Rd (Mile 1.1) - 2012
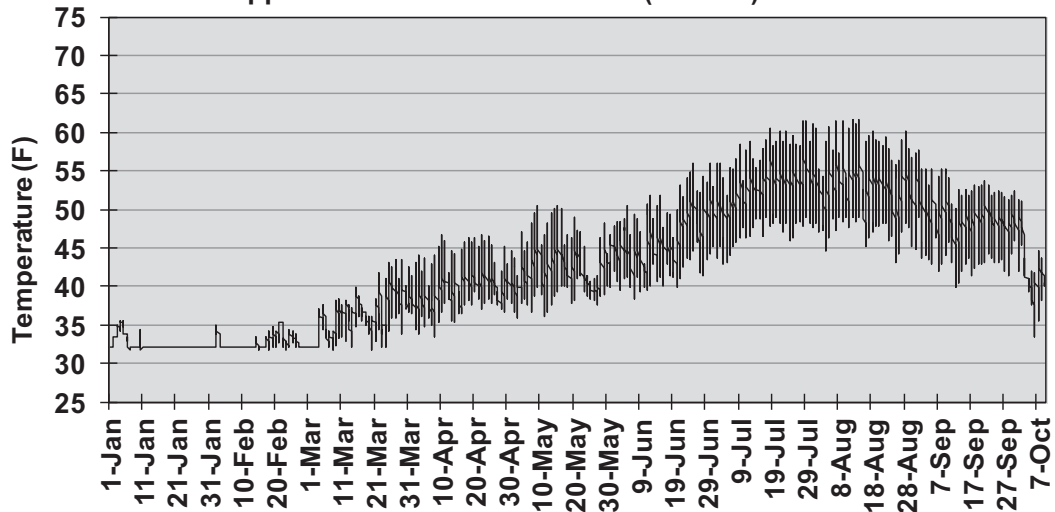
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	35.6	31.8	32.4	0.82	0.68
February	35.3	31.8	32.5	0.73	0.53
March	43.5	31.8	35.9	2.8	7.6
April	46.8	33.5	40	3	8.9
May	50.4	35.8	42.1	3.3	10.8
June	56	38.4	45.6	3.9	14.9
July	61.4	43	52.2	4.2	18
August	61.7	43.2	52.6	4.3	18.5
September	57.7	40	48.1	3.7	13.5
October	51.3	33.5	42	4	15.8

Cottonwood Creek at Hwy 200 (Mile1.0) - 2012
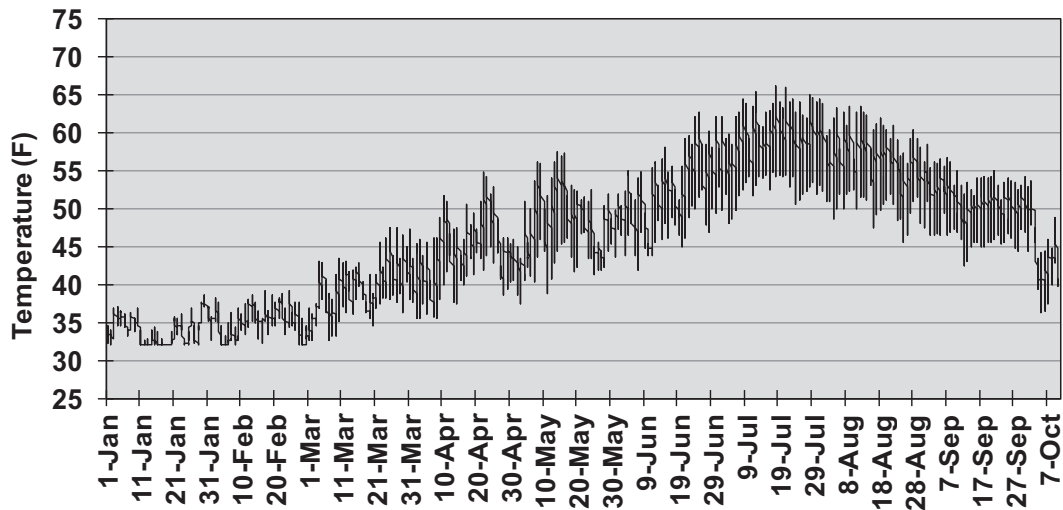
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	38.6	32	34	1.7	2.9
February	39.1	32	35	1.9	3.6
March	47.5	32.5	39.2	3.1	9.8
April	54.8	35.4	43.8	4	16.1
May	57.6	37.4	46.9	4.2	17.3
June	62.7	42	51.2	4.3	18.5
July	66.2	48.1	57.7	3.9	15.4
August	63.8	45.6	55.4	3.9	14.9
September	58.4	42.5	50.2	3.1	9.4
October	53.7	36.3	44.1	4.2	17.4

Enders Spring Creek (Mile 0.1) - 2012
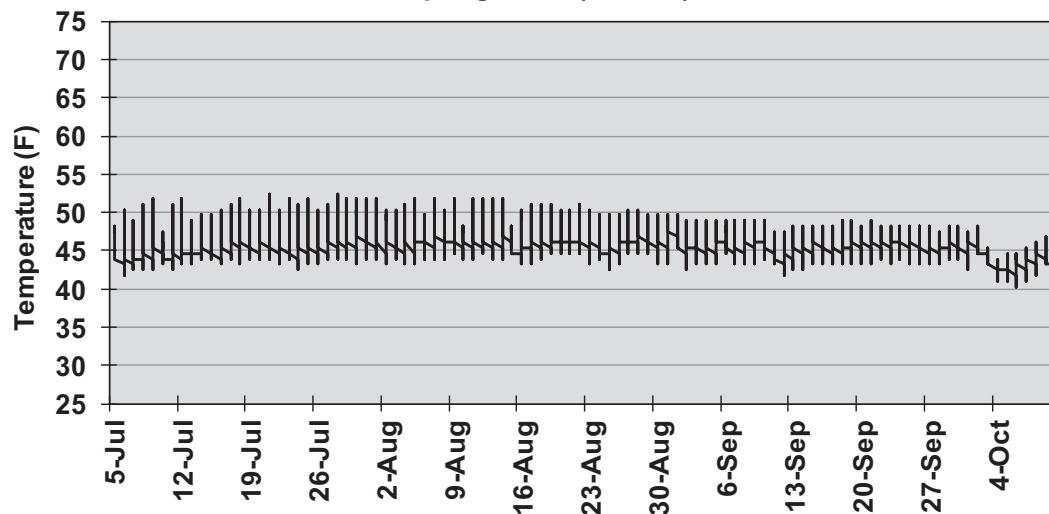
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	52.5	41.7	46.4	2.8	7.8
August	51.8	42.5	46.8	2.4	6
September	49.7	41.7	45.8	1.9	3.8
October	48.3	40.2	43.7	1.8	3.2

Gold Creek (Mile 1.6) - 2012
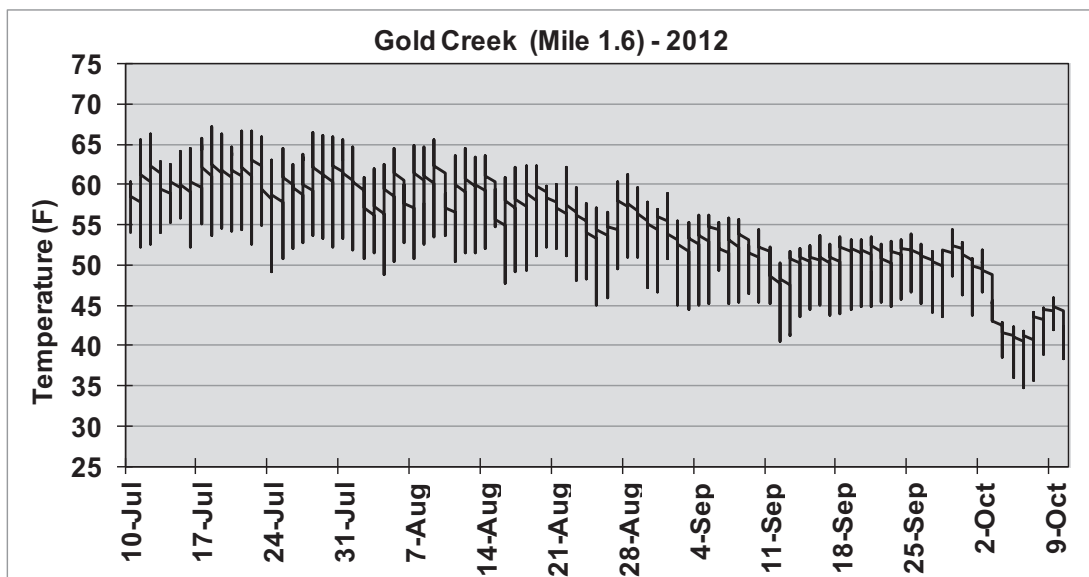
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	67.1	49.2	59.2	4.3	18.2
August	65.5	45	56.1	4.4	19.2
September	58.8	40.5	49.8	3.4	11.5
October	51.8	34.7	43	4.1	16.5

Grantier Spring Creek at mouth (Mile 0.1) - 2012
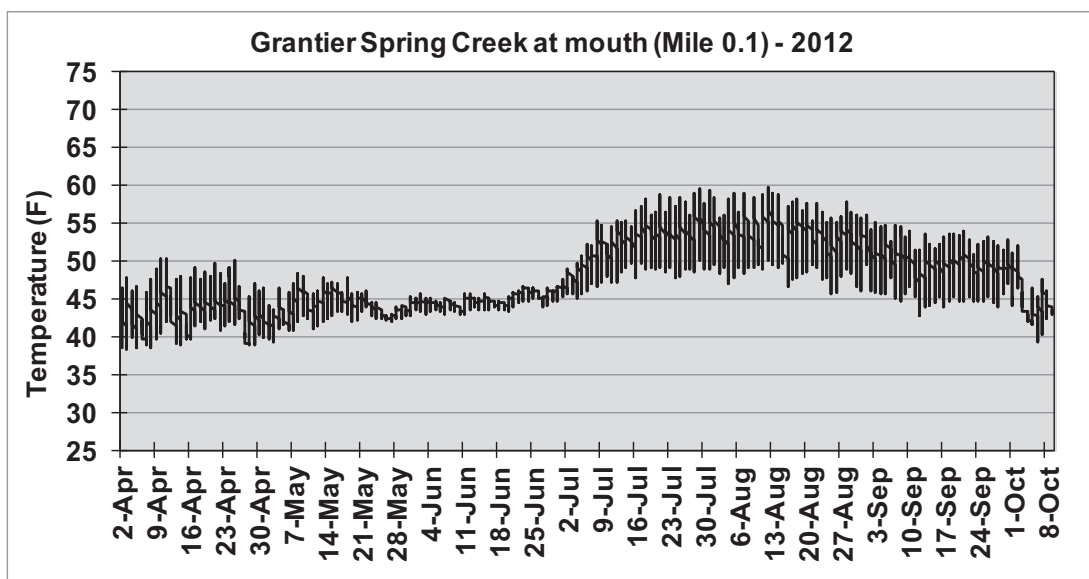
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	50.3	38.3	43.3	2.8	8
May	48.3	39.4	43.7	1.7	2.8
June	46.7	43	44.5	0.83	0.7
July	59.5	45	51.3	3.4	11.8
August	59.8	45.6	52.5	3.2	10.2
September	56.1	42.7	49	2.9	8.4
October	52	39.4	44.6	2.8	8

Grantier Spring Creek (Mile 1.0) - 2012
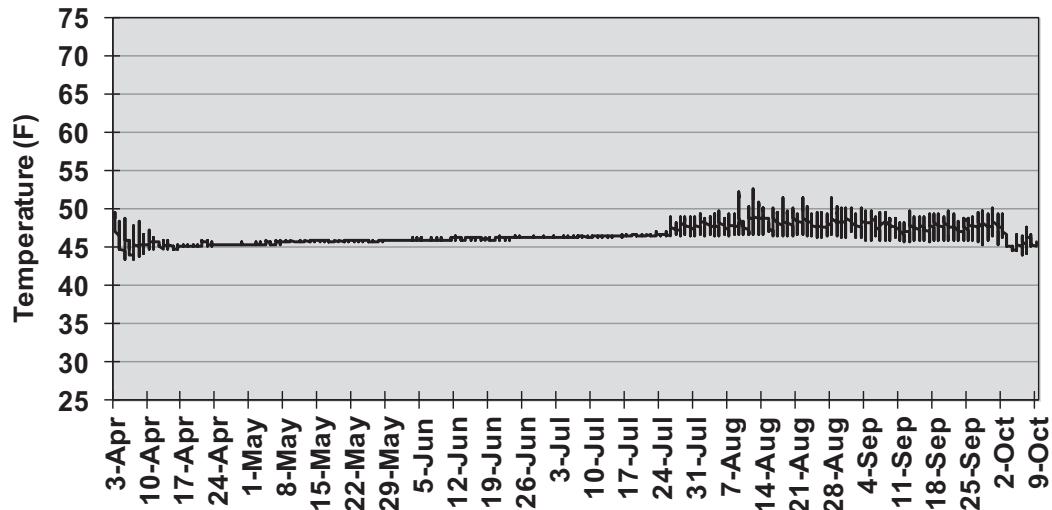
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	49.5	43.4	45.4	0.82	0.7
May	46	45.4	45.7	0.19	0.04
June	46.5	46	46	0.15	0.02
July	49	46.2	46.6	0.59	0.34
August	52.6	46.2	48	1.2	1.5
September	50.1	45.4	47.5	1.2	1.4
October	49.3	44	45.9	1.2	1.5

Hoyt Creek (Mile 1.2) - 2012
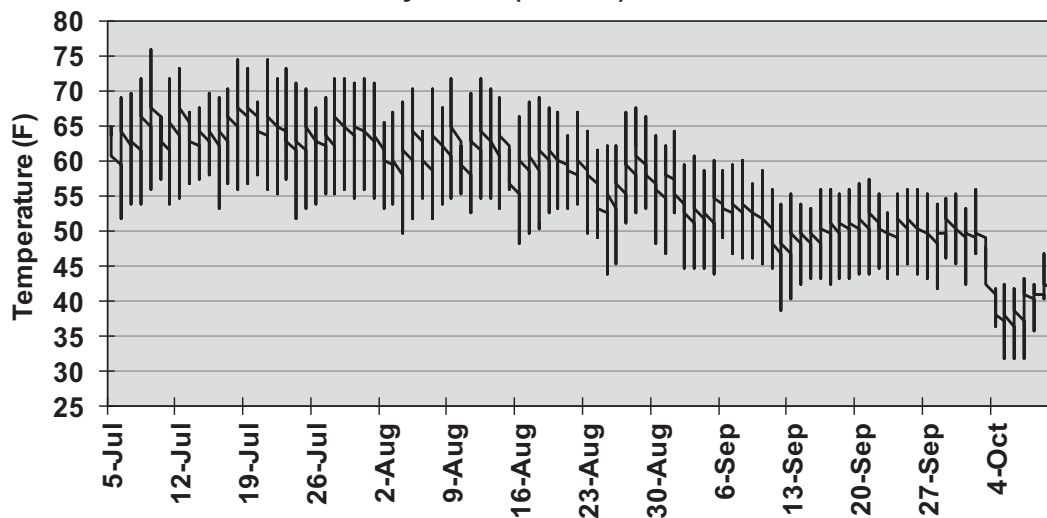
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	76	51.8	63	5.5	30.7
August	71.8	44	59	5.7	32.6
September	64.2	38.7	50.4	4.7	22.1
October	56	31.7	41.7	5.7	32.9

Hoyt Creek (Mile 4.3) - 2012
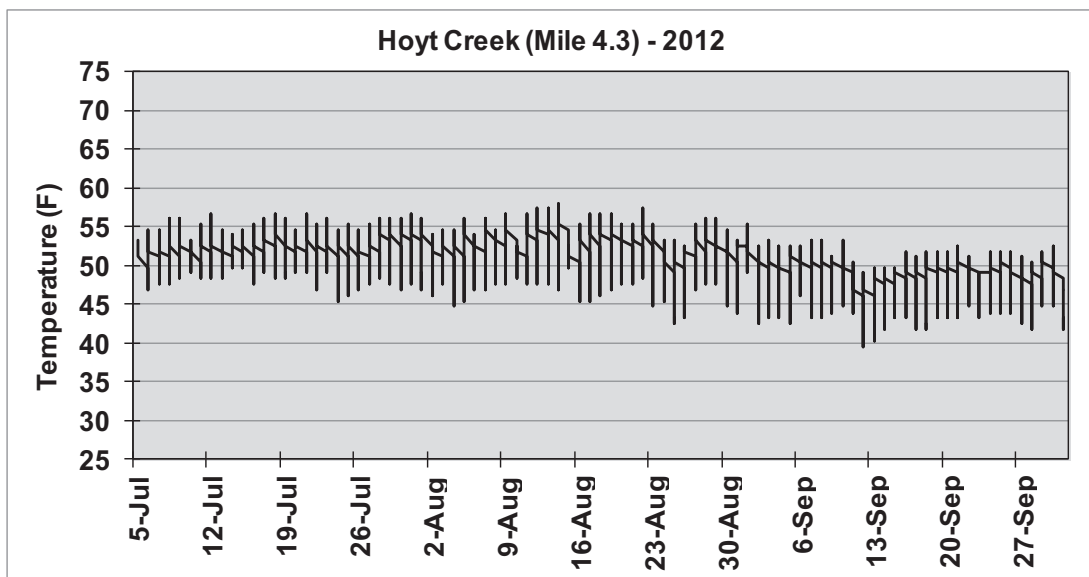
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	56.7	45.4	51.4	2.7	7.5
August	58	42.5	51	3.4	11.3
September	55.3	40	47.6	3.2	9.9

Jacobsen's Spring Creek (Mile 0.1) - 2012
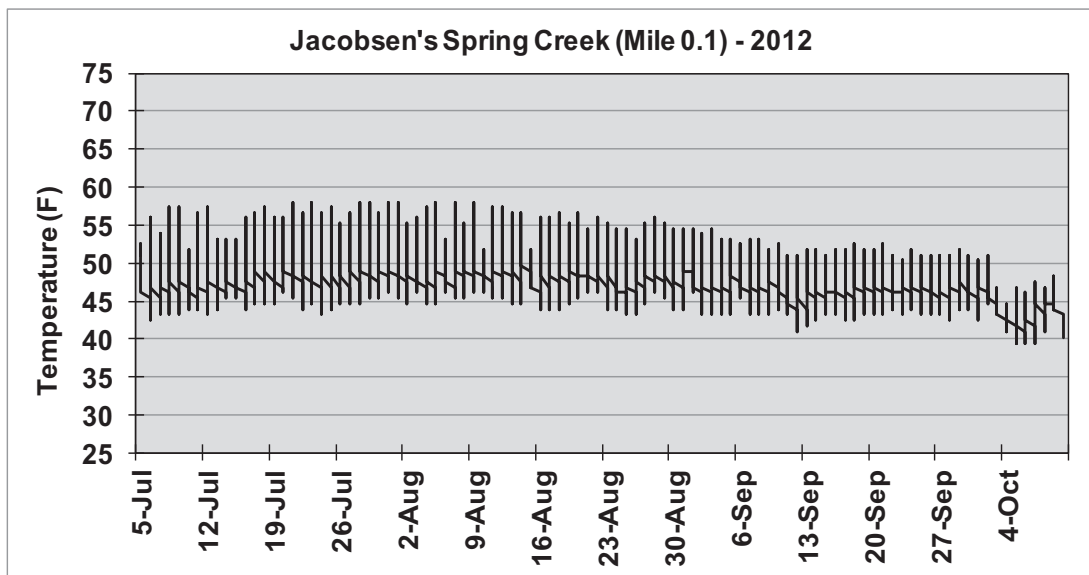
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	58	42.5	49.3	4.1	17.2
August	58	43.2	49.3	3.8	14.3
September	54.6	41	47	3.1	9.5
October	51.1	39.5	44.1	2.7	7.3

Kleinschmidt Creek (Mile 0.1) - 2012
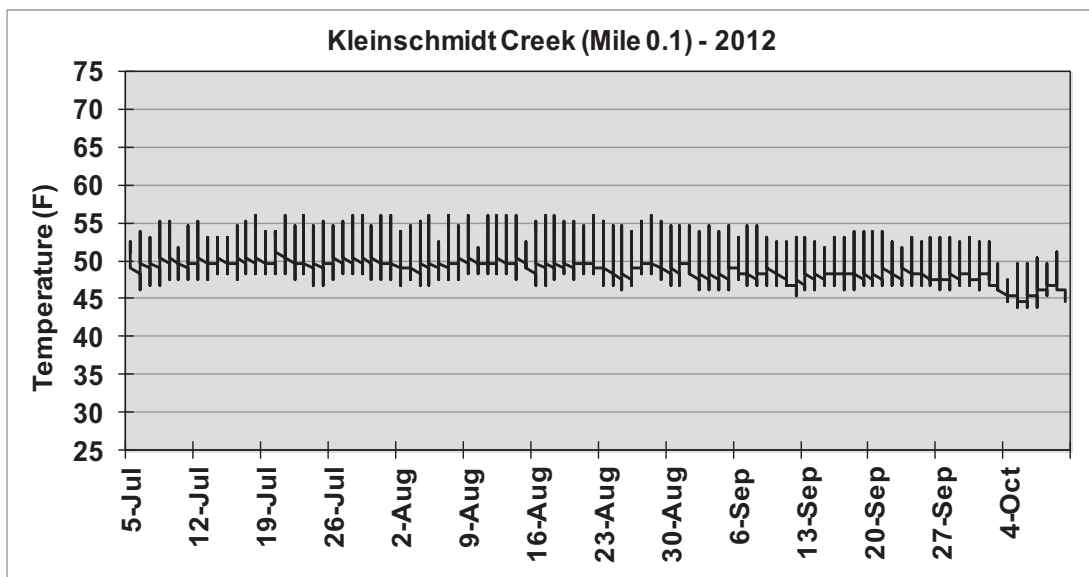
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	56	46.1	50.7	2.5	6.2
August	56	46.1	50.6	2.7	7
September	54.6	45.4	49.3	2.4	5.7
October	52.5	44	47.1	2.1	4.3

Monture Creek at FAS (Mile 1.8) - 2012
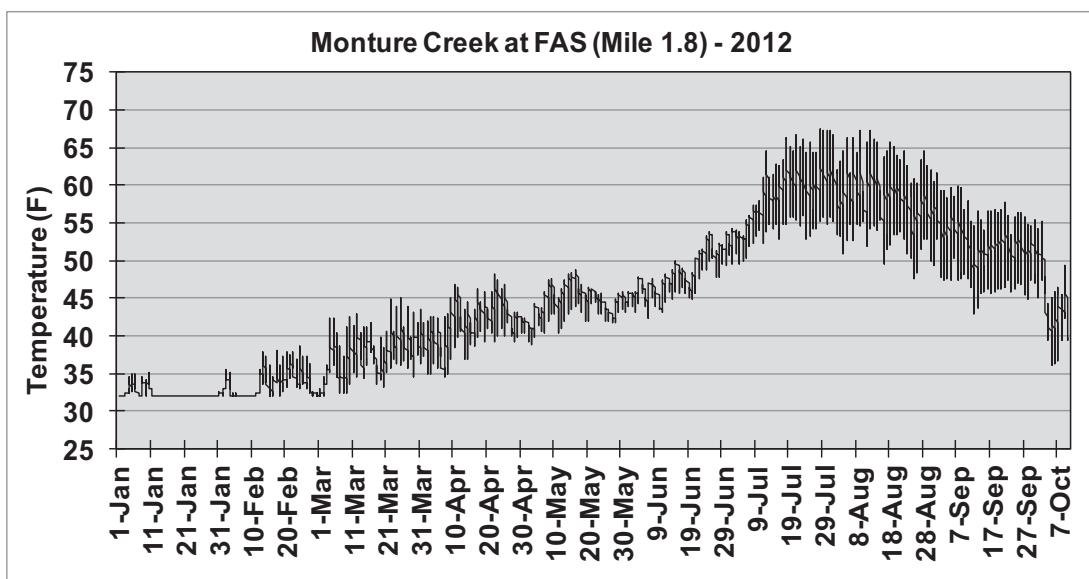
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	35.2	32	32.3	0.72	0.52
February	38.6	32	33.7	1.8	3.1
March	45.1	32	37.5	2.9	8.3
April	48.2	34.6	41.1	2.8	7.7
May	48.7	39	43.7	2	4
June	53.8	42.3	47.3	2.6	7
July	67.5	49.6	57.6	4.4	19.4
August	67.2	47.6	58.1	4.2	18
September	61.7	42.9	51.8	3.8	14.4
October	55.2	36.1	44.4	4.5	20.6

Nevada Creek above Nevada Spring Creek (Mile 6.3) - 2012
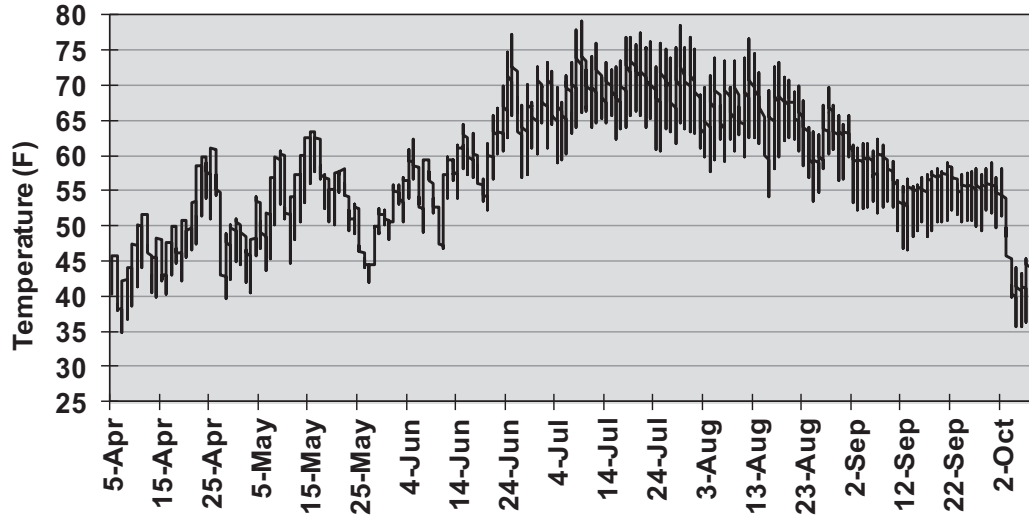
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	61.1	34.8	46.9	5.6	31.3
May	63.5	40.5	51.7	5.2	26.7
June	77.3	46.9	59	6	35.6
July	79.2	59	69	4.2	17.4
August	76.6	53.6	64.7	4.5	20
September	65.8	46.6	55.4	3.4	11.4
October	58.1	35.7	45	6.1	37.4

Nevada Creek below Nevada Spring Creek (Mile 5.0) - 2012
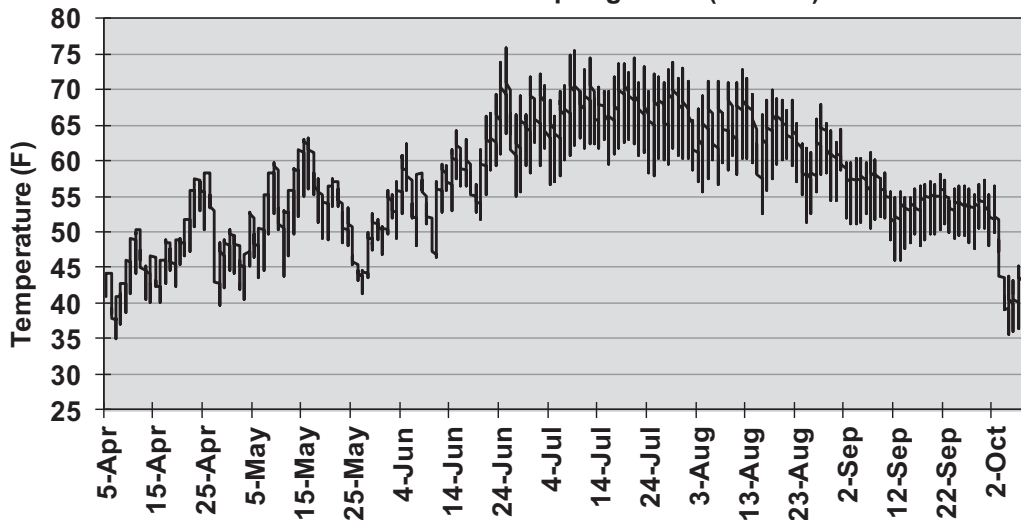
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	58.4	35.1	46.5	5.1	26.1
May	63.2	40.5	51.1	5.1	26.2
June	75.7	46.3	58.2	5.7	33.4
July	75.4	56.7	66.1	4.1	16.6
August	72.7	51.4	62.5	4.2	18.1
September	64.4	46.1	53.9	3.3	10.6
October	56.4	35.6	44.2	5.5	30

Nevada Spring Creek at mouth (Mile 0.1) - 2012
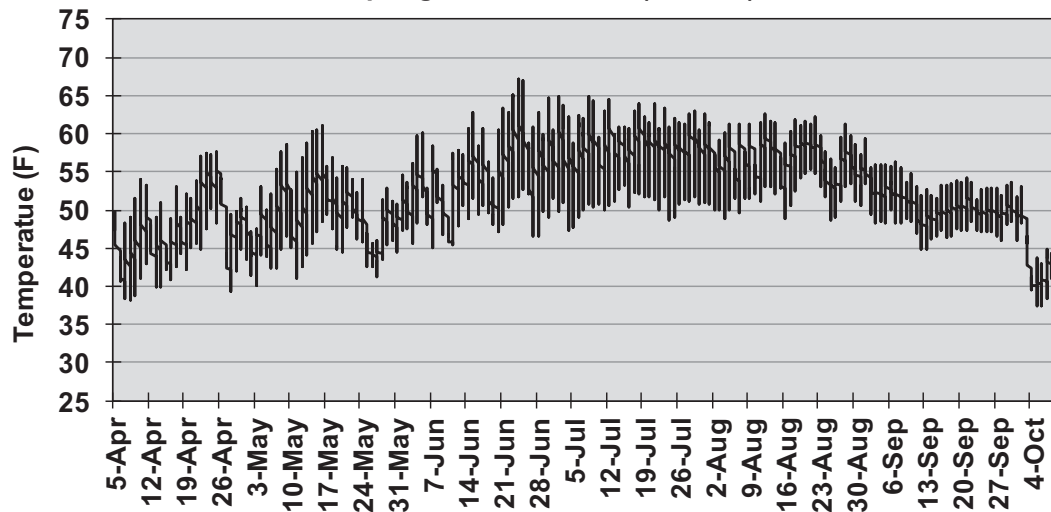
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	57.7	38.2	47	4.5	20.1
May	61	40.1	48.9	4.3	18.7
June	67.2	45.1	53.7	4.7	22.8
July	64.9	47.2	56.6	4.1	16.9
August	62.6	48.7	55.8	3.2	10
September	59.4	44.9	50.6	2.5	6.4
October	53.1	37.5	43.8	4	15.8

Nevada Spring Creek at lower bridge (Mile 1.1) - 2012
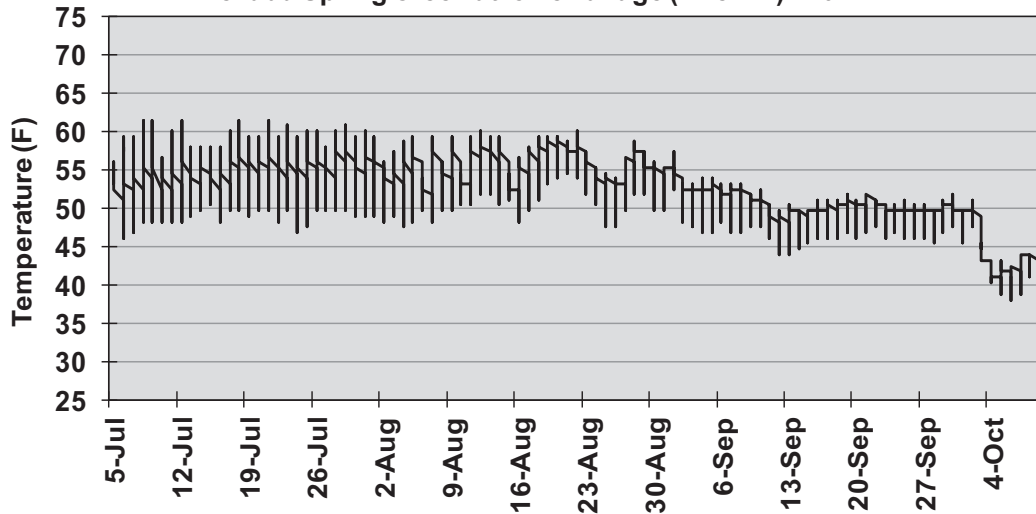
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	61.5	46.1	54.1	3.9	15.1
August	60.1	47.5	54.2	3.1	9.3
September	57.4	44	49.6	2.2	5
October	51.1	38	43.7	3.5	12

Nevada Spring Creek at lower fenceline (Mile 2.1) - 2012
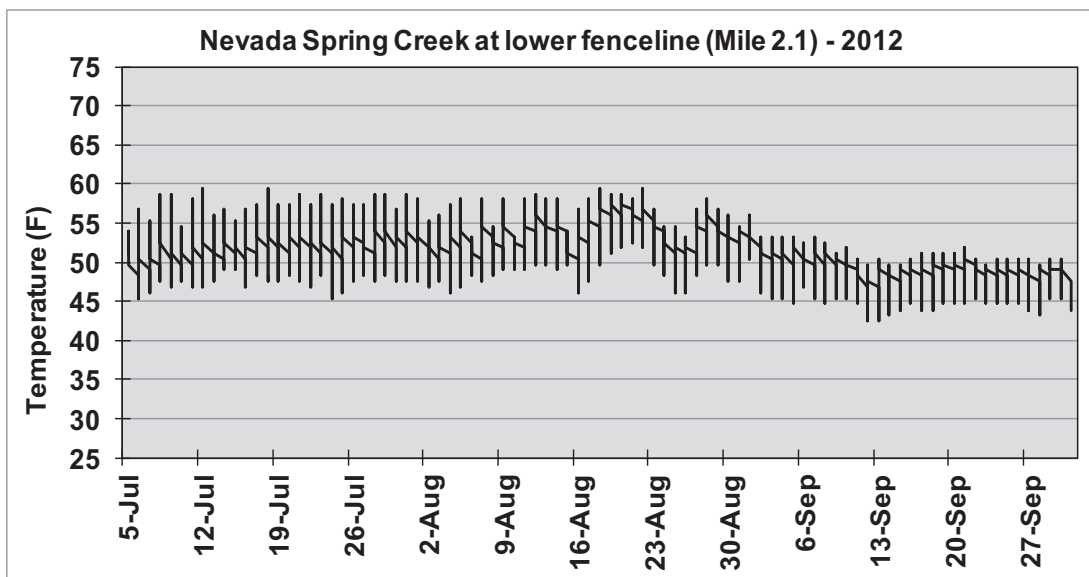
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	59.4	45.4	51.9	3.7	13.4
August	59.4	46.1	52.5	3.3	10.8
September	56	42.5	48	2.6	7

Nevada Spring Creek at upper fenceline (Mile 3.5) - 2012
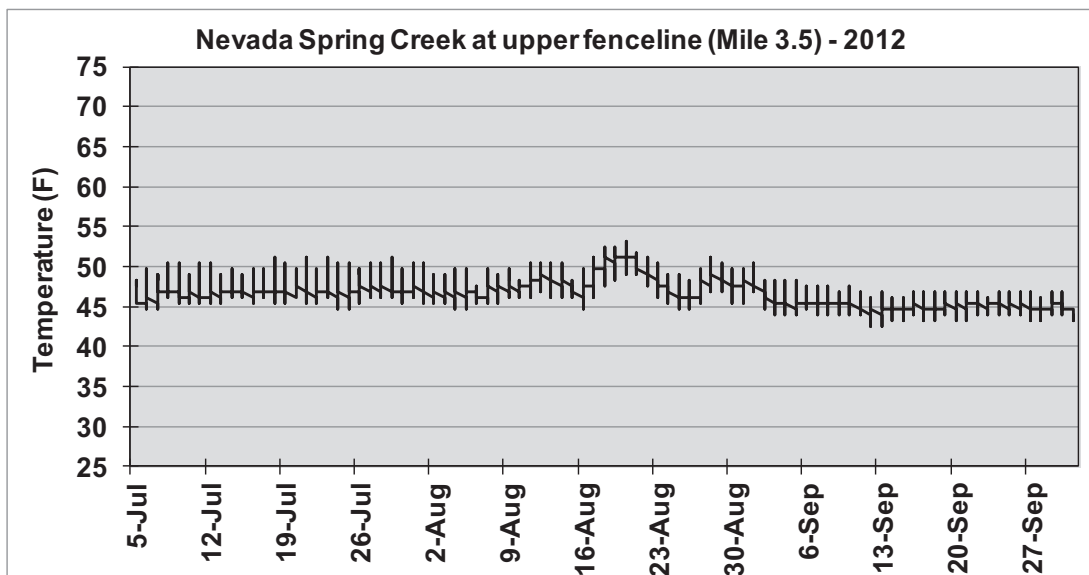
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	51.1	44.7	47.4	1.6	2.6
August	53.2	44.7	47.9	1.8	3.2
September	50.4	42.5	45.3	1.3	1.8

North Fork Blackfoot River at Ovando-Helmville Rd (Mile 2.6) - 2012
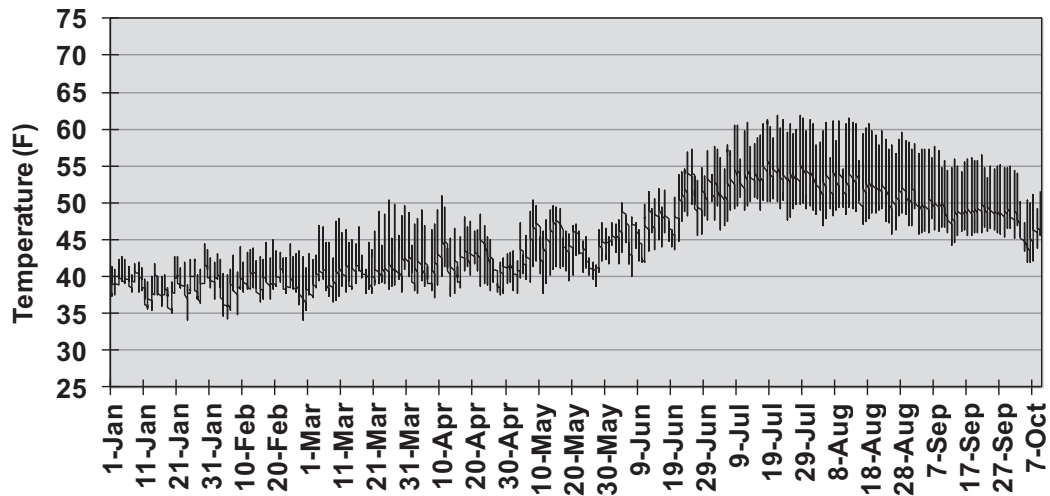
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
January	44.4	34.1	39.2	1.8	3.2
February	45	34.1	39.5	2.2	4.8
March	50.3	36.5	41.3	2.8	8.1
April	51	37.2	42.1	2.8	8.1
May	50.3	37.7	43.1	2.6	6.8
June	57.2	40	47.6	3.5	12.6
July	61.9	46.3	53.8	3.6	13
August	61.5	45.8	53	3.8	14.3
September	58	44.2	50.1	3.3	11
October	54.3	41.9	46.7	2.9	8.3

Poorman Creek (Mile 0.2) - 2012
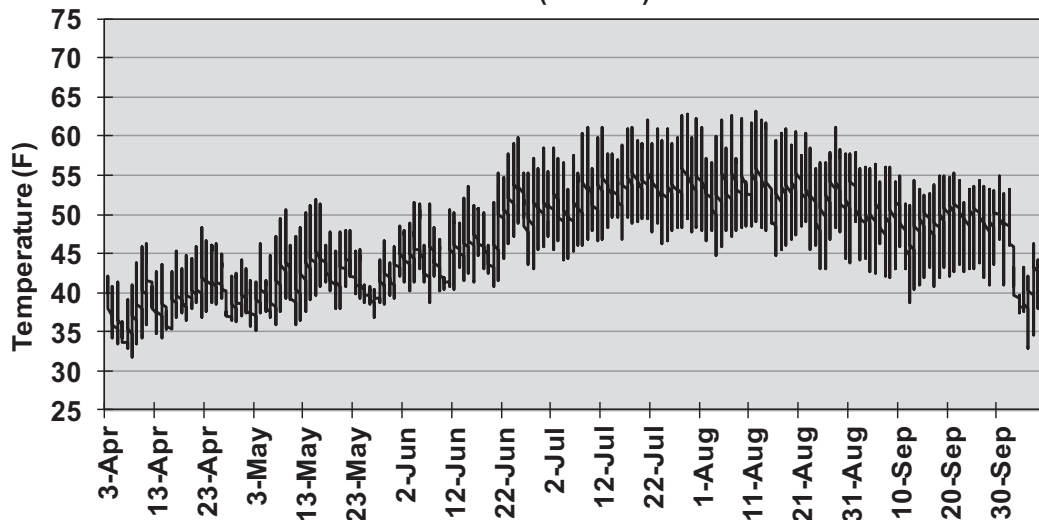
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	48.3	31.7	39.1	3.2	10.4
May	51.9	35.1	41.7	3.4	11.4
June	59.7	38.8	46.8	4.4	19.8
July	62.8	44.1	52.7	4.3	18.1
August	63.1	43	52.4	4.5	20.5
September	58	38.8	48.3	4.1	17.1
October	53.3	32.8	41.7	4.5	20

Shave Creek (Mile 0.1) - 2012
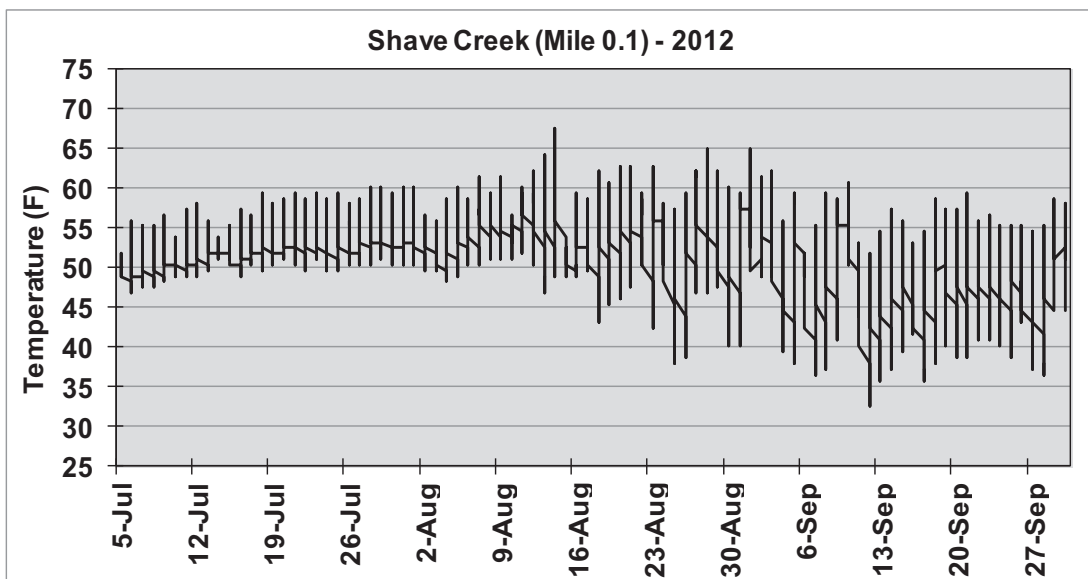
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	60.1	46.8	52.7	2.9	8.2
August	67.7	38	53.8	5.2	27.4
September	65	32.5	49.4	6.7	45.1

Snowbank Creek (Mile 0.2) - 2012
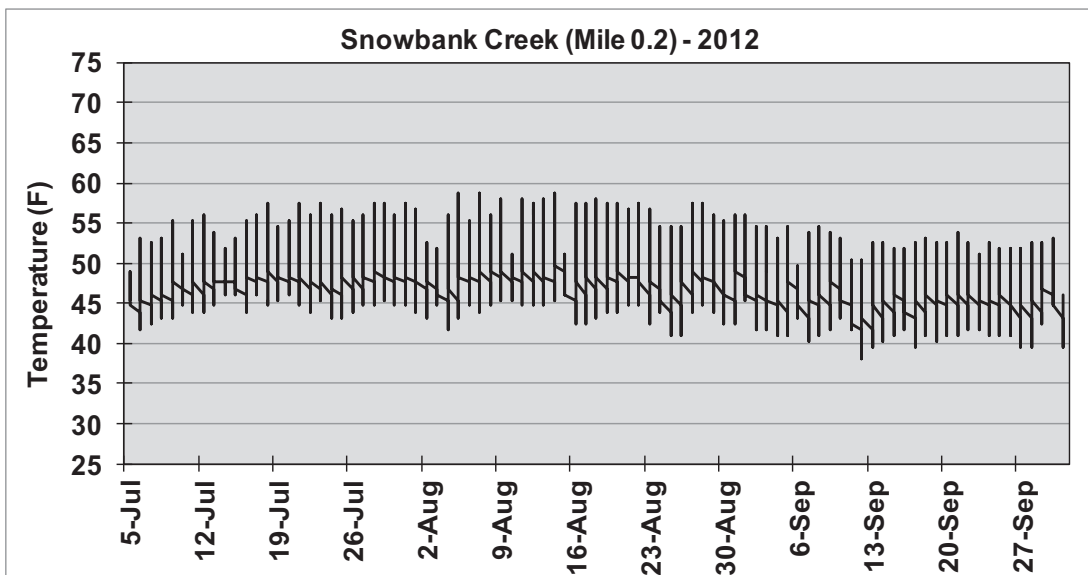
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
July	57.4	41.7	48.8	3.9	14.9
August	58.7	41	49.1	4.4	19.8
September	56	38	46.2	4.2	17.4

Wasson Creek (Mile 0.1) - 2012
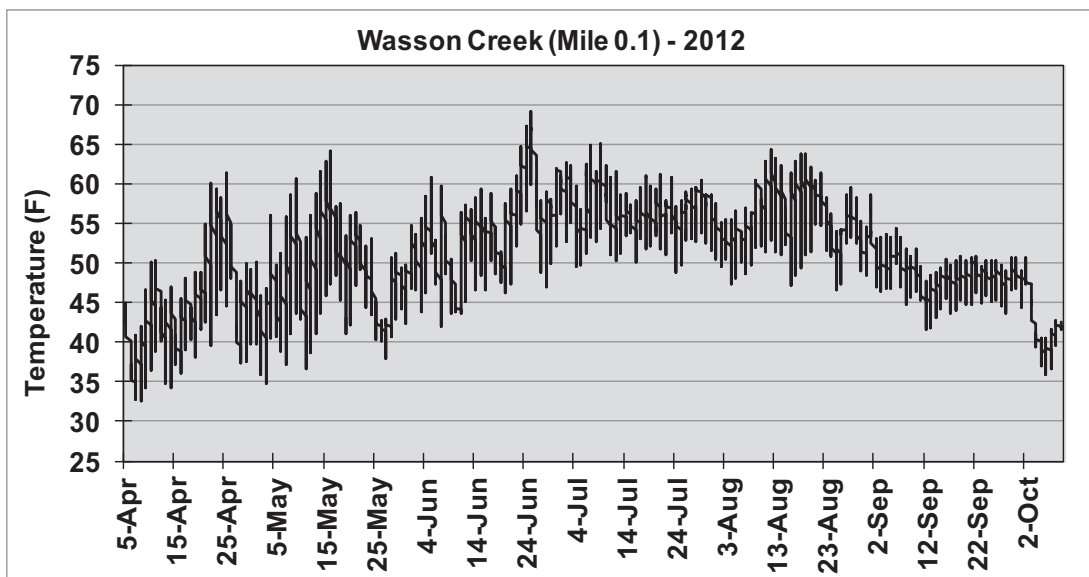
Month	Max Temp	Min Temp	Avg Temp	StDev Temp	Var Temp
April	61.3	32.6	43.7	6	35.9
May	64.1	34.9	47.5	5.8	34.1
June	69.2	42.1	53.2	5.4	28.9
July	65.1	49	56.3	3.1	9.5
August	64.3	17.6	54.7	3.8	14.1
September	58.6	41.6	48.3	2.7	7
October	50.7	35.9	42.3	3.8	14.4

Appendix G: Summary of bull trout redd counts for Blackfoot tributaries,1988-2012.

Stream	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012
East Fork Clearwater[1]	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3	21	21	18	-	6
West Fork Clearwater[1]	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	19	20	65	33	66	74
Copper Creek[2]	-	21	23	24	25	19	23	21	21	22	27	9	20	16	15	4	12	15	33	32	34	33	21	22	19
Snowbank Creek[2]	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	27	1	17	13	35
Belmont Creek[3]	-	-	-	-	-	-	-	-	14	-	14	8	4	3	11	-	1	3	1	4	3	2	1	1	2
Gold Creek[3]	-	-	-	-	-	-	-	-	-	-	16	30	9	17	6	4	4	9	1	4	2	1	1	0	1
Dunham Creek[1]	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11	6	6	4	-	5	7	5	8	8	9
Monture Creek[1]	11	10	-	25	34	45	49	60	65	61	60	65	74	94	93	80	44	41	15	33	18	23	49	51	73
Morrell Creek[1]	-	-	-	-	-	-	-	-	-	-	-	-	-	-	24	10	22	16	26	4	33	54	37	27	38
North Fork Blackfoot River[1]	12	8	-	26	39	-	-	27	59	65	76	87	123	75	70	41	42	43	61	42	95	97	86	158	131

1-Surveyed by FWP
2-Surveyed by USFS
3-Surveyed by Plum Creek Timber Company
4- Surveyed by FWP and Plum Creek Timber Company