Fisheries Investigations in the Big Blackfoot River Basin Montana, 2013-2015

By
Ron Pierce and Craig Podner

Montana Fish, Wildlife & Parks

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3201 Spurgin Road
Missoula, MT 59804

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INTRODUCTION

The Blackfoot Basin has been the site of a *wild trout* restoration initiative since 1988-89 when fisheries assessments first identified: 1) the over-harvest of native trout, 2) basin-scale degradation of riparian and aquatic habitat in tributaries, and 3) a long history of toxic mine waste in the headwaters of the Blackfoot River as limiting Blackfoot River fisheries (Peters and Spoon 1988; Peters 1990; Moore et al. 1991). These findings triggered basin-wide protective fishing regulations in 1990 followed by pilot-level restoration actions in tributaries of the Blackfoot River. By the mid-1990s, improved fisheries and social acceptance of the restoration initiative led to the refinement of a private lands restoration methodology for the Blackfoot River and the expansion of tributary restoration from the 1990s to the present (Aitkin 1997; Pierce et al. 1997, 2005, 2011, 2013; BBCTU 2016). While aquatic habitat improvement provides the foundation for this endeavor, the cooperation of many resource agencies, conservation groups, private landowners form the social and technical network necessary to focus, fund and implement the restoration work. This initiative provides a specific framework for the recovery of dwindling stocks of imperiled native trout when integrated with protective angling regulations, site-specific restoration and landscape protection (e.g., conservation easements) in ecologically critical areas of the watershed.

Blackfoot River restoration is a voluntary, tributary-based, priority-driven process whereby the scope and scale of restoration expands as information and stakeholder cooperation is generated (Pierce et al. 2005, 2013). As an iterative process, restoration usually begins with fisheries and habitat assessments with emphasis on human-induced limiting factors, which then lead to restoration activities targeting individual tributary stocks. Restoration priorities focus on tributaries supporting migratory native trout and emphasize restoration techniques that 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 limiting factors are identified and as opportunities allow. Within this restoration process, monitoring and project evaluations are critical to identify measures of biological effectiveness, as well as areas where adaptive management is required.

After 28 years of fisheries field work, Montana Fish, Wildlife and Parks (FWP) has completed fisheries and/or habitat-related surveys on all major tributaries (n=207) within the Blackfoot River Basin, including 180 streams outside of designated wilderness/roadless areas. Of these 180 streams, fisheries investigations have identified human-induced fisheries impairments on >80% of inventoried streams (Pierce et al. 2008, Appendix H). With this and other fisheries information, and with the cooperation of many stakeholders, restoration has now targeted 78 tributaries with >500 individual fisheries improvement projects (Pierce et al. 2008; BBCTU 2016; Appendix G). Restoration emphasizes private lands at the lower elevations of the basin; however, restoration is now expanding to industrial timberlands and public lands following the transfers of former Plum Creek Timber Company lands to conservation groups and to public ownerships. Of these 78 streams, twenty eight streams now are now approaching final restoration phases (Appendix G). In addition to habitat work, the upper North Fork Basin upstream of the North Fork Falls (within the Scapegoat Wilderness) is now being considered as a possible large-scale native trout recovery area (Pierce et al. 2016).

Unlike the habitat restoration on lower elevation streams, the North Fork project would replace hybrid trout with native westslope cutthroat trout and bull trout in pristine high elevation habitat highly suited to native trout.

With this continued expansion of fisheries improvement work, river restoration has increasingly evolved from a reach/tributary-scale to a more watershed-scale conservation effort. From the beginning of the endeavor, the Big Blackfoot Chapter of Trout Unlimited (BBCTU) has been the leading proponent of river conservation. Though the Blackfoot Challenge and Clearwater Resource Council also coordinate fund-raising; promote educational programs, drought management and forest restoration. The Nature Conservancy (TNC), Five Valleys Land Trust and Montana Land Reliance, together with agencies including FWP, United States Fish and Wildlife Service (FWS), Department of Natural Resource Conservation (DNRC), United States Forest Service (FS) and Bureau of Land Management (BLM) have jointly protected 628 miles^2 of private land through easements and land exchanges. This landscape-level effort now protect 1,012 miles of riparian corridor through conservation easements and transfer of 222 miles^2 of former Plum Creek Timber Company land to public ownership during the last 20 years (Amy Pearson, TNC, personal communication). The most recent landscape conservation project involves the 2014 TNC purchase of the remaining 183 miles^2 of Plum Creek Timber Company land in the western region Blackfoot Basin (Figure 4).

Lastly, the most significant threat to ecological health of the Blackfoot River is now being addressed with the removal of toxic mine waste in the headwaters of the Blackfoot River. To this end, the cleanup has removed 400,000 cubic yards of mine waste during the 2015 calendar year, which includes the removal Mike Horse tailings dam and reconstruction 1,750 feet of new channel in Bear Trap Creek. When the clean-up is finished in 2017, a total of 860,000 yards^3 of streamside mine waste will be removed from the valley floor and placed within an off-site repository, and three miles of stream will be restored to natural form and function.

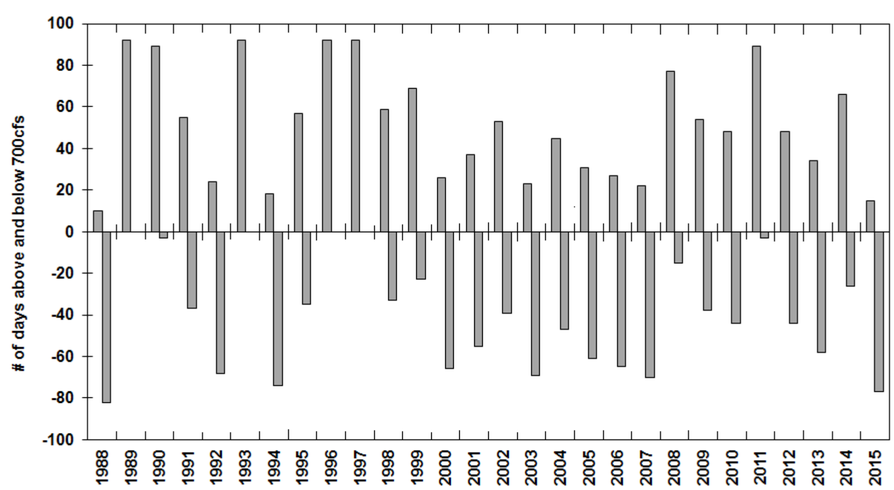


Figure 1. Summer drought index for the Blackfoot River at the USGS Bonner Gauge, 1988-2015. This graph shows the 92 day period from July 1 through September 30 when discharge was above and below the 700cfs minimum instream flow value in the lower Blackfoot River. Note the high number of low flow days in summer 2015

EXECUTIVE SUMMARY

The 2013-2015 reporting period was marked with two drought years (2013, 2015) and one year of above normal flow (2014) (Results Part I). Low flow conditions during the 2015 drought were among the lowest in the last 27 years (Figure

1). Likewise, high summer water temperatures in 2013 and 2015 were 2-3°F above average (Figure 2).

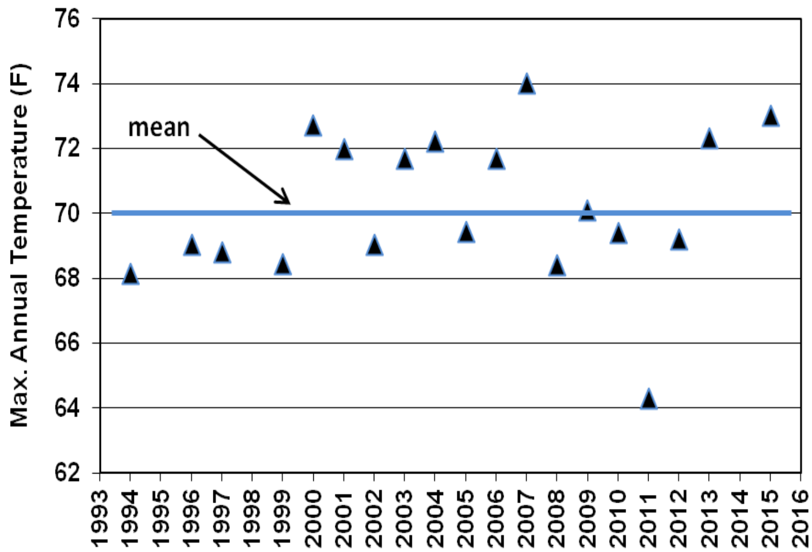


Figure 2. Maximum annual water temperatures for the lower Blackfoot River downstream of Belmont Creek, 1994-2015. The blue horizontal line shows the long-term average of 70.2°F for the temperature values.

In 2014, we continued to monitor the abundance, biomass and species composition of wild trout in four long-term monitoring sites on the Blackfoot River (Results Part II). Population surveys in the Johnsrud and Scotty Brown Bridge Sections showed overall stable to increasing biomass (Figure 3), though declines in the abundance of brown trout and rainbow trout were also noted. These declines were largely offset by corresponding increases with westslope cutthroat trout. Because of weak recruitment and degraded tributaries, wild trout in the Wales Creek Sections of the Blackfoot River continue to show a 12-year trend of low trout abundance and low biomass compared to up- and down-river monitoring sites (Figure 3 and 10, Results Part II, Appendix C). Despite variability, long-term monitoring at the four river monitoring sites shows consistent long-term improvement in the westslope cutthroat trout metapopulation throughout the mainstem Blackfoot River.

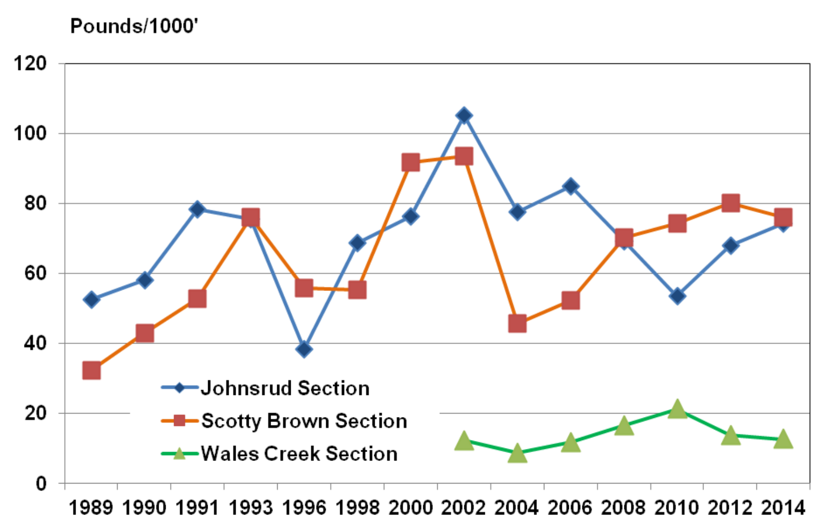


Figure 3. Total trout biomass estimates (all trout >6.0") for three section of the lower Blackfoot River, 1989-2014.

Based on bull trout redd counts and other surveys, bull trout numbers in three upper river tributaries (Monture Creek, North Fork Blackfoot River and Copper Creek) show stable to increasing trends. Likewise, lakes in the

Clearwater drainage also indicate stable to increasing trends in adfluvial bull trout. Conversely, bull trout continue to decline in three lower river tributaries (Gold Creek, Belmont Creek and Cottonwood Creek; Results Part III).

From 2013 through 2015, we surveyed fisheries on 47 tributaries outside of the Clearwater drainage (Appendix A). These surveys involved 1) long-term monitoring associated with past restoration, 2) a comprehensive assessment of fisheries in the Keep Cool Creek drainage as a baseline for future restoration (Results Part III), and 3) aquatic inventories in the Scapegoat Wilderness related to the North Fork native trout recovery project (*a separate report*). Long-term monitoring reveal population increases on many restored streams once habitat damage and other human-related factors limiting populations are corrected. However, monitoring also sheds light on the biological complexities of native fish recovery, as well as the inherent social challenges of ensuring comprehensive, effective and sustainable restoration outcomes in areas of mixed land ownership and intensive land-use.

Currently, one of the more pressing restoration challenges relates to the monitoring, maintenance and adaptive management associated with the long-term sustainability of past restoration projects. Inherent to effective restoration, monitoring obligations not only include tracking fisheries response (*as described in this report*), but also include the renewal of instream flow projects, monitoring of riparian grazing systems, revegetation and weed control along with the maintenance needs of fish screens, fish ladders, fences and other infrastructure. Successful riparian grazing systems are especially complex because they require and understanding of site potential (e.g., Hansen et al. 1995; Rosgen 1996), riparian healing processes and the sensitivity of target salmonids to grazing disturbance - all conditions that vary greatly across riparian and aquatic ecosystems. As a result, riparian grazing systems usually require consistent monitoring against site-specific targets to effectively improve riparian and aquatic habitat.

In Results Part IV, we present seven applied research studies involving restoration within the Blackfoot Valley. These include four long-term evaluations of restoration outcomes associated with spring creeks, and emphasize: 1) population response of migratory cutthroat trout to multi-scale restoration and the experimental use of Coanda diversions for fish passage and ditch entrainment; 2) water temperature reduction through active restoration and relationships of groundwater-induced temperatures to the severity of whirling disease infection; 3) the relative role of channel reconstruction and instream wood as habitat improvement techniques and relationships to long-term fisheries response; and 4) relationships of spawning site quality and benthic invertebrates in restored and unrestored streams. Additional studies in Results Part IV focus on 1) the prediction of whirling disease within the Blackfoot Basin and the predisposition of low-gradient alluvial streams to high whirling disease severity; and 2) whether whirling disease can mediate hybridization between rainbow trout and westslope cutthroat trout based on the increased susceptibility of rainbow trout and spatial overlap of rainbow trout with *Myxobolus cerebralis*. Lastly, we present a bull trout genetic assignment study that links natal tributaries with the presence of fluvial bull trout in the Blackfoot River as well as adfluvial bull trout in lakes of the Clearwater River basin.

Finally, as a continuation of 13 prior FWP fisheries reports between 1988 and 2012, this 2013-2015 report was written to guide future wild trout restoration and to promote other river conservation actions within the Blackfoot Basin.

STUDY AREA

The Blackfoot River, located in west-central Montana, begins within the *Upper Blackfoot Mining Complex* at the junction of Beartrap and Anaconda Creeks. From this junction, the Blackfoot River flows west 132 miles from the base of the Continental Divide to its confluence with the Clark Fork River at Bonner (Figure 4). With the removal of Milltown Dam in 2008, the Blackfoot River is now free-flowing over its entire length. The Blackfoot River is one of twelve renowned *blue ribbon* trout rivers in Montana with a 1972 appropriated *Murphy* in-stream flow summer water right of 700 cfs at the USGS Bonner (#12340000) gauging station. This 700 cfs value represents the minimum flow below which river productivity begins to decline. In 2015, this 700 cfs water right gained more senior (1904) status when the Montana Legislature ratified the Confederated Salish Kootenai Water Compact with Senate Bill 262.

Mean annual discharge for the lower Blackfoot River near Bonner is 1,590 cfs (USGS station 12340000, 2015 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, moraine and outwash deposits, 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.

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 66,923 angler days in 2013 (Montana Fish Wildlife and Parks, 2015).

Despite its popularity, segments of the river system support low abundance of wild trout due to an array of natural conditions and human impairments. Populations of imperiled native trout (westslope cutthroat trout - *Oncorhynchus clarkii lewisi* and bull trout - *Salvelinus confluentus*) are particularly low and continue to decline in some areas of the basin. Natural conditions limiting trout fisheries involve drought stressors, areas of high instream sediment loads, low instream productivity, naturally intermittent tributaries, warm summer temperatures and periods of severe icing of the lower mainstem river. Human-induced fisheries impairments include mining-related contamination in the upper Blackfoot Basin, the spread of exotic organisms (e.g., *Myxobolus cerebralis* - the causative agent of whirling disease and nonnative fish) and a wide array of land use perturbations (Pierce et al. 2005, 2008; Eby et al. 2015). The sum of natural conditions and human impairments produces an array of trout assemblages that vary regionally within the watershed and longitudinally across river and tributary reaches.

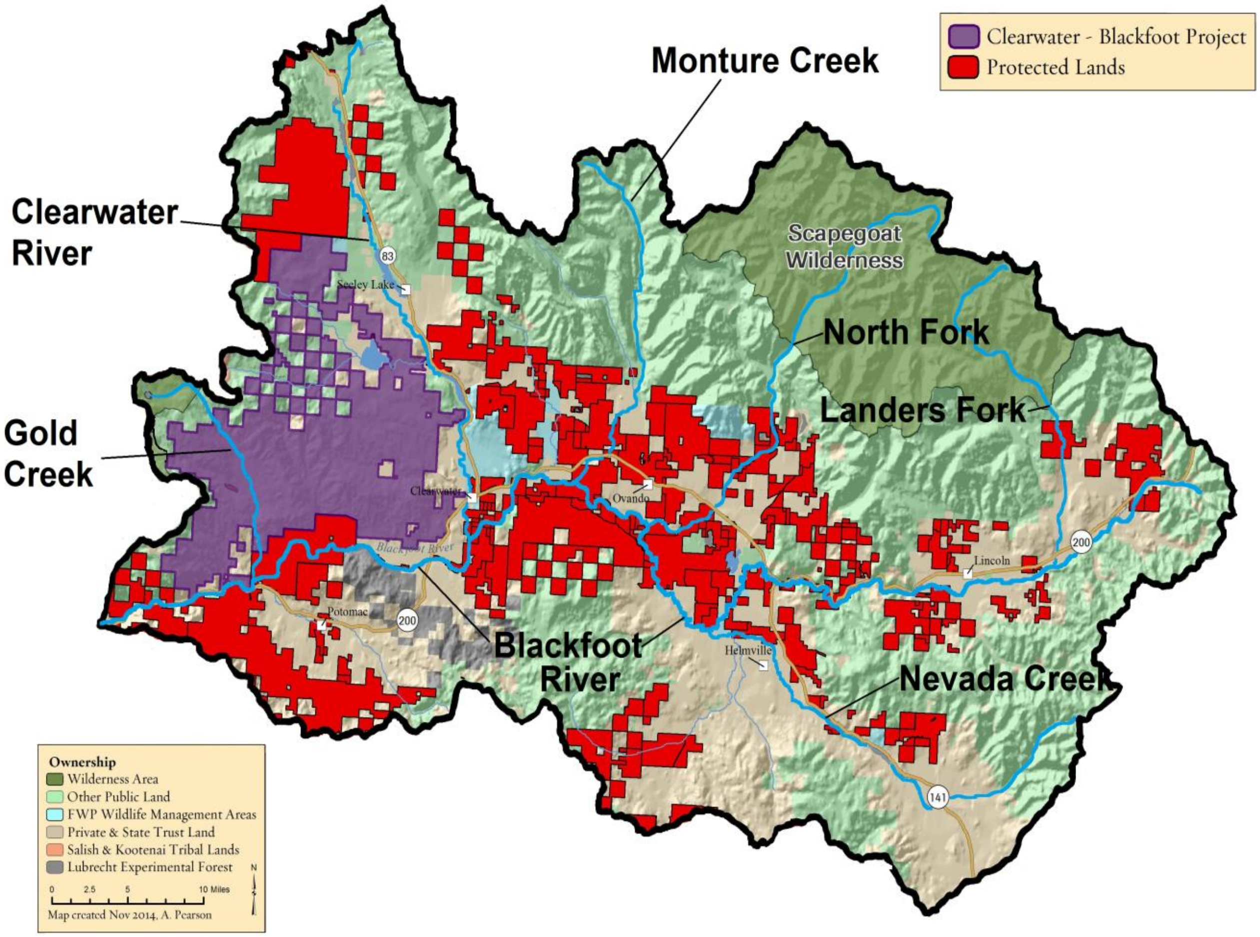


Figure 4. Blackfoot River Basin: major streams and landownership. The red areas shows conservation lands converted to either public ownership or private land with easement protection. The purple area shows the 2014 TNC purchase of remaining 117,152 acres of Plum Creek land within the Blackfoot Basin.

Land ownership in the Blackfoot River Basin is a mix of public and private: 36% private land owners; 46% USFS land, 11% by the state of Montana, and 7% by the BLM. In general, public lands and large tracts of TNC properties comprise large forested tracts in mountainous areas of the watershed, whereas private timber and agricultural 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. Currently, the majority of habitat degradation occurs on the valley floor and foothills of the Blackfoot watershed and largely on private agricultural ranchlands. However, legacy riparian and aquatic degradation also extend to commercial timber lands and mining districts, as well as state and federal 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 abundance 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 Rocky Mountain 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*), smallmouth bass (*Micropterus dolomieui)*, yellow perch (*Perca flavescens)* and central mudminnow *(Umbra limi).*

Most salmonids (westslope cutthroat trout, bull trout, and rainbow trout) in the mainstem Blackfoot River system exhibit migratory (fluvial) life-histories involving spawning and rearing tributaries (Swanberg 1997; Schmetterling 2001; Pierce et al. 2007, 2009, 2012, 2014). However other salmonids (mountain whitefish and brown trout) variously spawn in the Blackfoot River and tributary streams (Pierce et al. 2009). Native fishes within the Clearwater River and it chain of lake also exhibit migratory (adfluvial) life-histories, which include lake-dwelling behavior marked by tributary spawning (Bensen 2009). Westslope cutthroat trout has a basin-wide distribution and is the most abundant salmonid 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 larger lower river tributaries. However, rainbow trout are also established in 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 self-sustaining populations. Brown trout inhabit ~15% of the perennial stream system with a distribution that extends from about Landers Fork down the length of the Blackfoot River and into the lower foothills of the tributary system. Similar to bull trout, mountain whitefish occupy ~20% of the basin, including the larger, colder streams and lakes. Brook trout are widely distributed in tributaries, but rare in the main stem Blackfoot River below the Landers Fork.

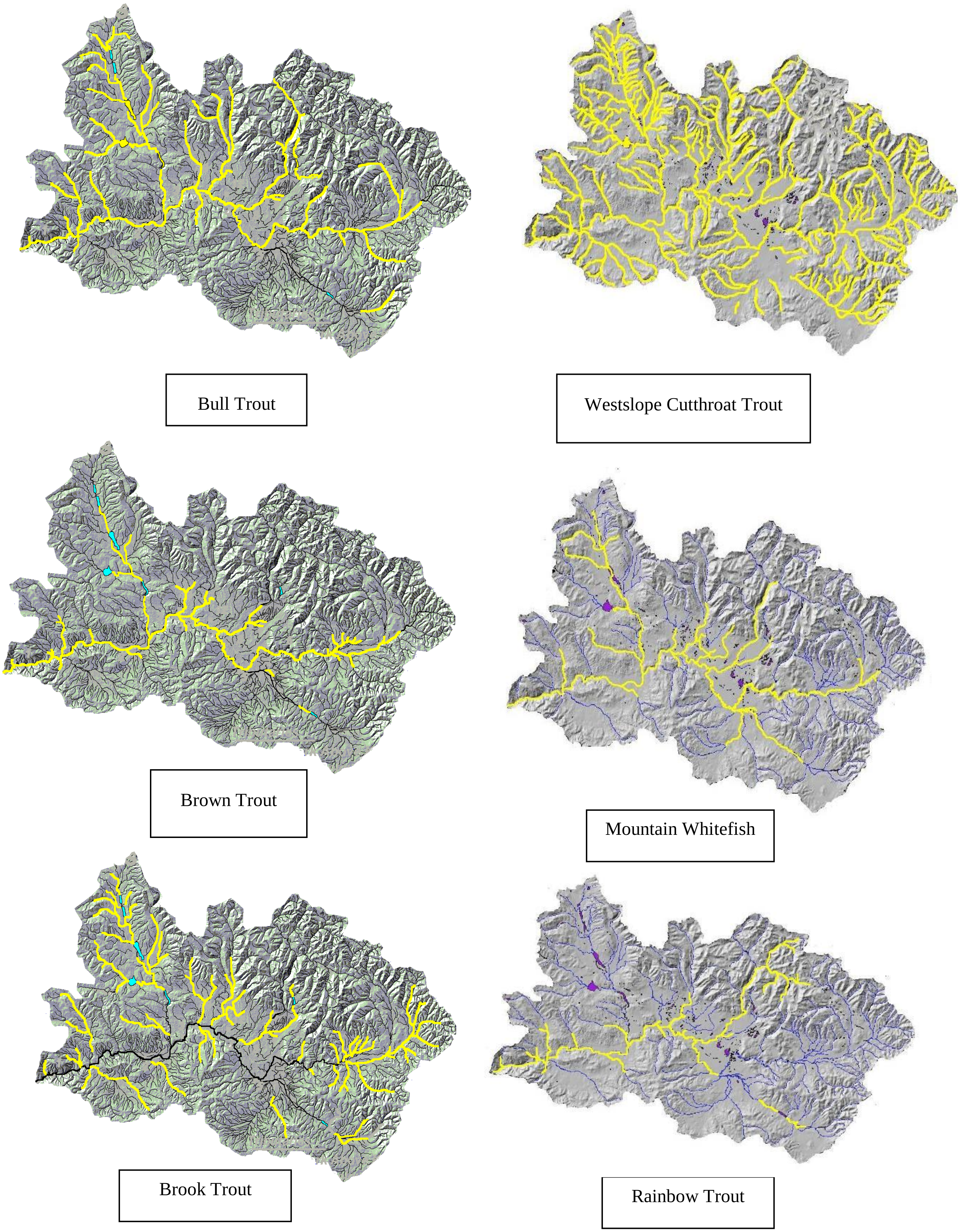


Figure 5. Generalized distribution of six salmonids in the Blackfoot Basin.

PROCEDURES

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

Fish population surveys and estimators

In 2014, we completed fish population surveys in four long-term monitoring sites on the Blackfoot River. These are the 1) Johnsrud (river-mile mid-point 13.5), 2) Scotty Brown Bridge (river mile mid-point 43.9), 3) Wales Creek (river mile mid-point at 63.0), and 4) Canyon Sections (river mile mid-point 95.3). Outside of the Clearwater drainage, from 2013 through 2015 we also completed fish population surveys at 96 sites on 48 tributaries (Figure 6).

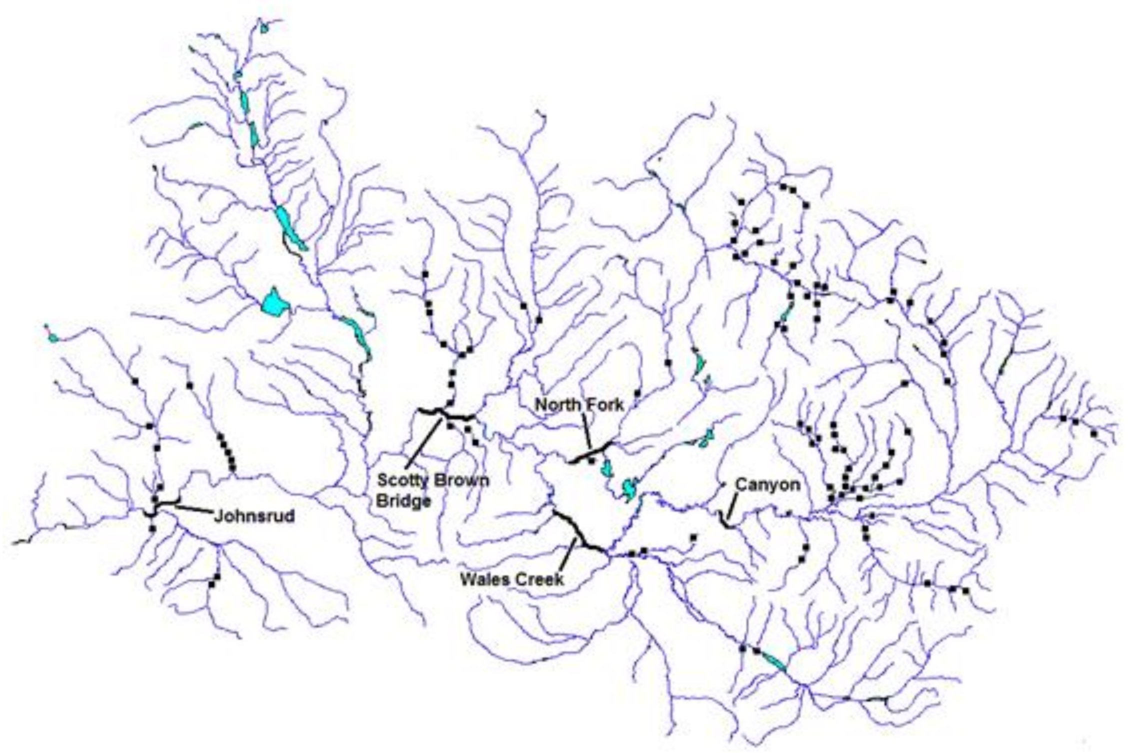


Figure 6. Fish population survey sites in the Blackfoot River basin 2013-2015. The mainstem Blackfoot River and lower North Fork Sections are labeled. The smaller stream surveys sites are shown as black squares.

Depending on the size of the individual river or stream, fish population surveys relied on either boat or backpack electrofishing methods. On larger waterbodies (Blackfoot River, North Fork Blackfoot River and Nevada Creek), the electrofishing unit was aluminum drift boat with either a Coffelt Model VVP-15 or a Smith-Root 15B rectifier and a 5,000 watt generator. The hull of the boat served as the cathode and two fiberglass booms, each with four steel cable droppers, served 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). On smaller 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) was a braided steel wire. Fish populations were intensively sampled using standard methods from August to November to enable comparisons of abundance between years and across sampling sections. All captured fish were anesthetized with either tricaine methanesulfonate (MS-222), weighed (g) and measured (mm) for total length (TL). For this report, all weights and lengths were converted to standard units.

Fish population surveys relied on mark-and-recapture for the larger streams, and multiple-pass depletion estimates of trout abundance and/or a catch-per-unit-effort (CPUE) statistic for small stream surveys. For the Blackfoot River, Nevada Creek and the North Fork Blackfoot River, we used a modified Petersen mark-and-recapture estimator. Using this method, estimates are considered valid if recaptures include $\geq$ four fish. For small streams, we used a depletion estimator to determined trout abundance. All 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 and other tributary locations are referenced by river-mile or stream-mile.

For the Blackfoot River fish population surveys in this report, 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:

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

$$\text{Biomass Estimate} = N\,(Wt)$$

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

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

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:

$$N = [n + 1 / n - T + 1]\ [kn - X - T + 1 + (k - i) / kn - X + 2 + (k - i)]_{\mathrm{i}} < 1.0$$

Where n is the smallest integer satisfying Equation. Probability of capture (p) and variance of N are then estimated by:

$$\frac{p = T}{kN - X}$$

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

Where,

N = point estimate

i = pass number,

k = number of removals (passes),

C_i = number of fish caught in ith 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 intensive electrofishing pass and is adjusted per 100’ of stream (i. e., CPUE of 8 means 8 fish captured per 100’ of sampled stream). For figures in this report, we refer to CPUE as Catch/100' and depletion estimates of abundance as Trout/100' ± 95% CI. We refer to mark-and-recapture estimates of abundance in the larger water bodies (Blackfoot River, North Fork Blackfoot River and Nevada Creek) as Trout/1,000' ± 95% CI. CPUE catch statistics are located in Appendix A. Depletion estimates are located in Appendix B. Mark-and-recapture estimates of abundance, biomass and condition factor for the Blackfoot River are located in Appendix C.

Water Temperature

From 2013 through 2015, we collected continuous water temperature data (^{o}F) at 52 sites including 1) six long-term monitoring sites on the Blackfoot River, 2) six long-term monitoring sites on bull trout spawning streams, 3) twenty three sites in the Scapegoat Wilderness upstream of the North Fork Falls, and 4) eighteen sites low-elevation associated with tributary restoration (Figure 7). We used either Hobo temperature (72-minute) or tidbit (50-minute) data loggers (Onset Computer Corporation). All raw data are plotted for each station and monthly summary statistics are located in Appendix E. For the 12 long-term datasets on the Blackfoot River and bull trout streams, we standardized temperature summaries for this report using July (the identified period of peak warming) data and display median, quartile and minimum and maximum temperatures values. For this report, we also used maximum daily temperatures when comparing pre- and post-restoration monitoring.

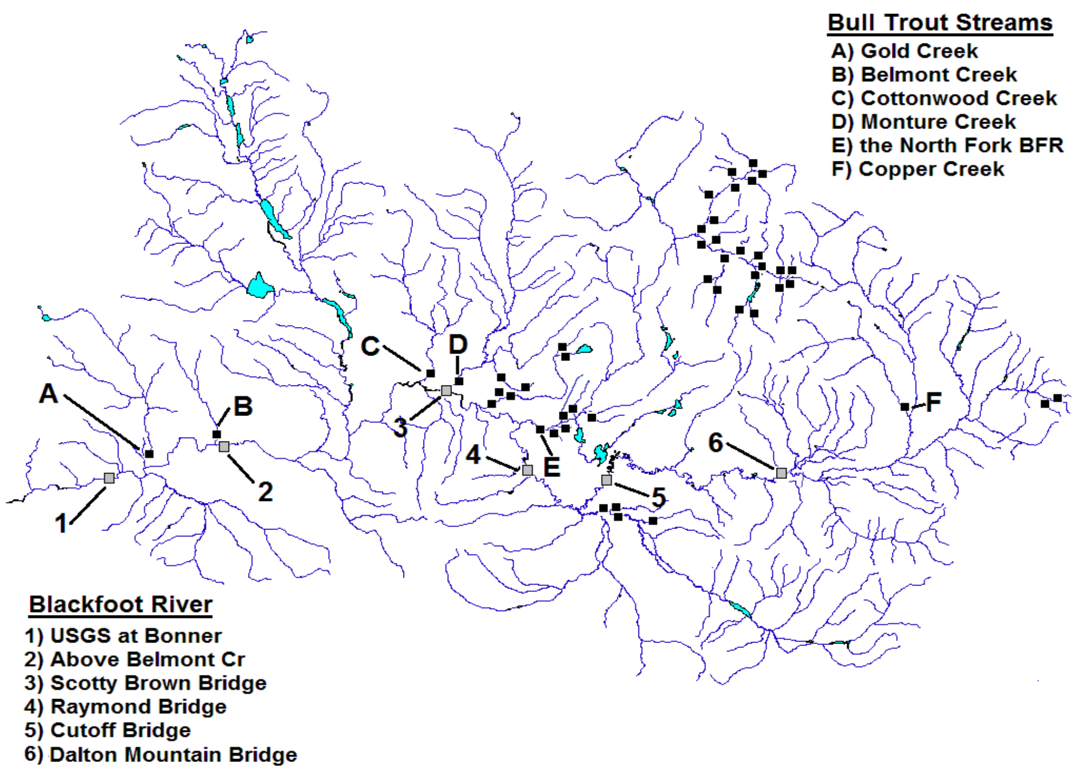


Figure 7. Water temperature sensor locations map 2013-2015. Monitoring sites on the Blackfoot River are numbered. Monitoring sites on bull trout streams are identified alphabetically.

Working with Private Landowners

Typically, each tributary restoration project involves multiple landowners, professional disciplines, funding sources and the involvement of one or more watershed groups. Restoration typically focus on correcting obvious impacts to fish populations such as migration barriers, stream de-watering, fish losses to irrigation canals, and over-grazed and degraded riparian areas. Most 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 watershed groups (BBCTU, BC, CRC) and local government, state and federal agencies such as FWP, DNRC, FWS, USFS and NRCS. The non-profit 501(c)3 status 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, assisting with funding support and monitoring fisheries response. Fisheries biologists and other agency specialists help develop projects usually in conjunction with watershed groups, consultants and landowners. Project leaders generally enlist help from interagency personnel or consultants including range conservationists, hydrologists, engineers, and water right specialists as necessary. Watershed groups, especially BBCTU, 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.

RESULTS/DISCUSSION

PART I: Blackfoot River Discharge (USGS station #12340000 provisional data) and Water Temperatures

From 2013 through 2015, the Blackfoot River watershed was subject to two years of below normal runoff (2013 and 2015) and one year (2014) of above normal runoff (Figure 8). In 2013, the magnitude of peak flows were near normal but high flows occurred early, which led to low summer base flows. In 2014, peak flows were about 50% above normal, which led to more favorable summer flows. Whereas, peak flows in 2015 were about 40% below normal and occurred much earlier than normal (Figure 8). This, combined with low spring precipitation, led to very low late summer base flow. Under these conditions, the Blackfoot River consistently fell below minimum flows of 700 cfs and recorded the lowest summer (July throughout September) flow reading (401 cfs) since 1988 (Figures 1 and 8).

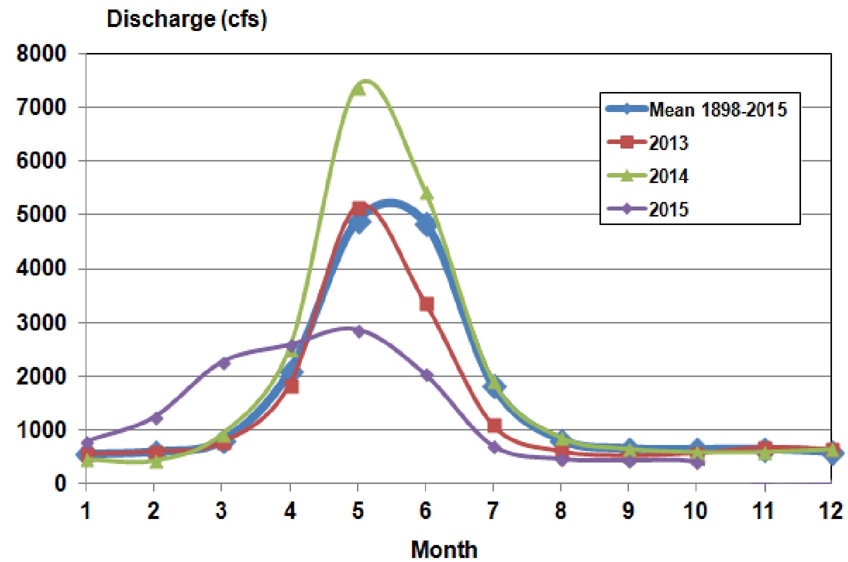


Figure 8. Mean monthly discharge for the Blackfoot River at the Bonner gauge: The bold blue line shows the long-term (monthly mean) hydrograph for the 1898-2015 period of record. The red line shows the 2013 hydrograph. The green line shows the 2014 hydrograph. The purple line shows the 2015 hydrograph.

Blackfoot River and tributary temperatures

Temperature data were collected 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, and 4) monitor temperatures associated with Blackfoot Drought Plan. A summary of all July water temperature data for six long-term monitoring sites of the Blackfoot River are shown on Figure 9. Similar plots of water temperatures in bull trout critical habitat are shown on Figure 19. Plots of all raw temperature data and summaries of monthly statistics are located in Appendix E.

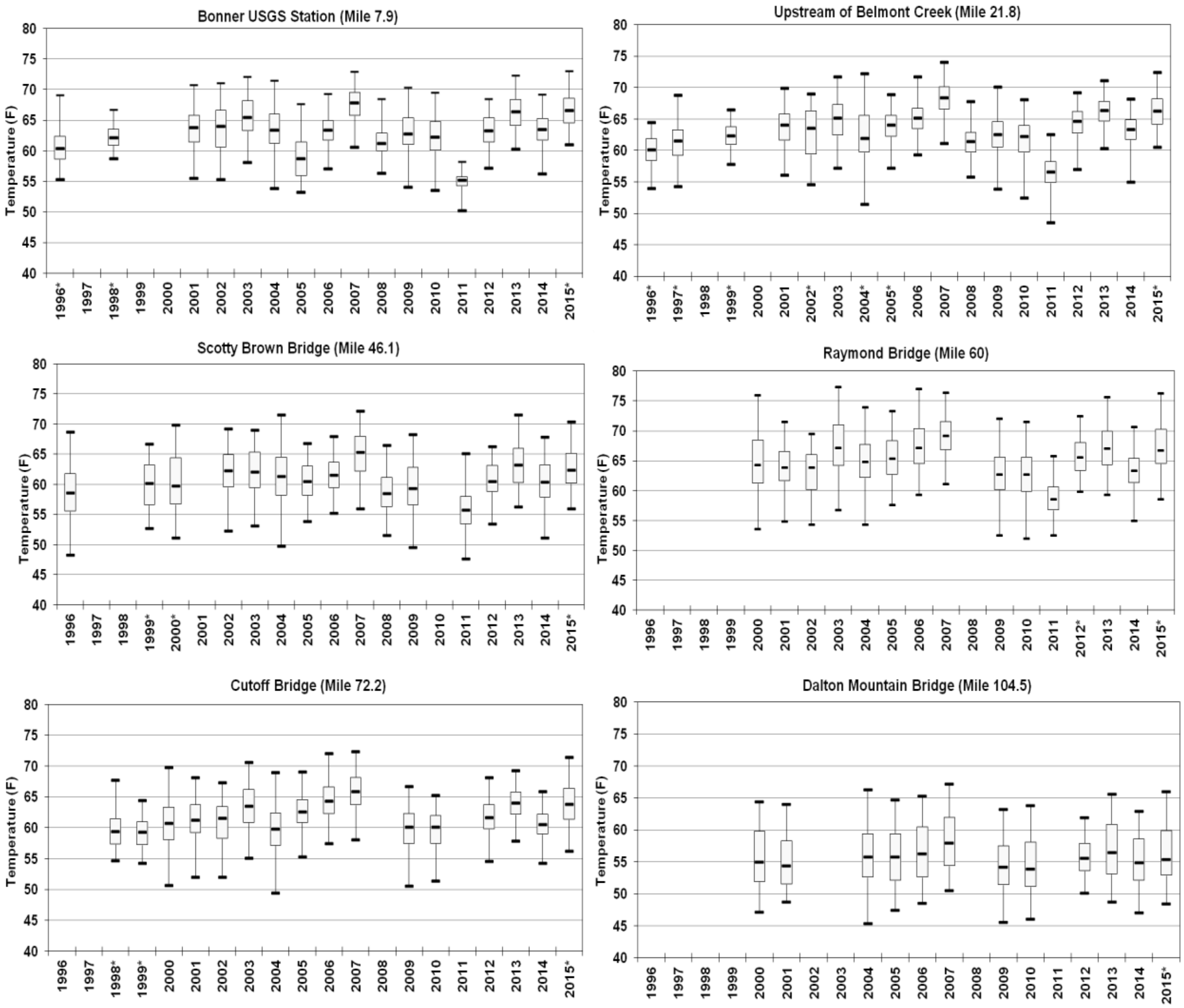


Figure 9. July water temperatures at six long-term monitoring locations on the Blackfoot River. Box plots show minimum, maximum, median and quartile values. An * denotes incomplete data for the month.

PART II: Blackfoot River Trout Populations 1988-2014

Trout Abundance, Biomass and Species Composition – Part II summarizes population survey results for wild trout (fish >6.0”) at four long-term (1989-2014) monitoring sites (Johnsrud, Scotty Brown Bridge, Wales Creek and Canyon Sections) of the Blackfoot River (Figure 6). For these monitoring sites, summaries of total trout abundance and total trout biomass (trout >6.0”) from 1989 to the present are shown on Figure 10. Figure 11 shows the trout species composition from 1988 - 2014 for the four monitoring sites. Species-specific population estimates by monitoring section are described below. All summary statistics for the 2014 Blackfoot River mark and recapture population surveys are located in Appendices C.

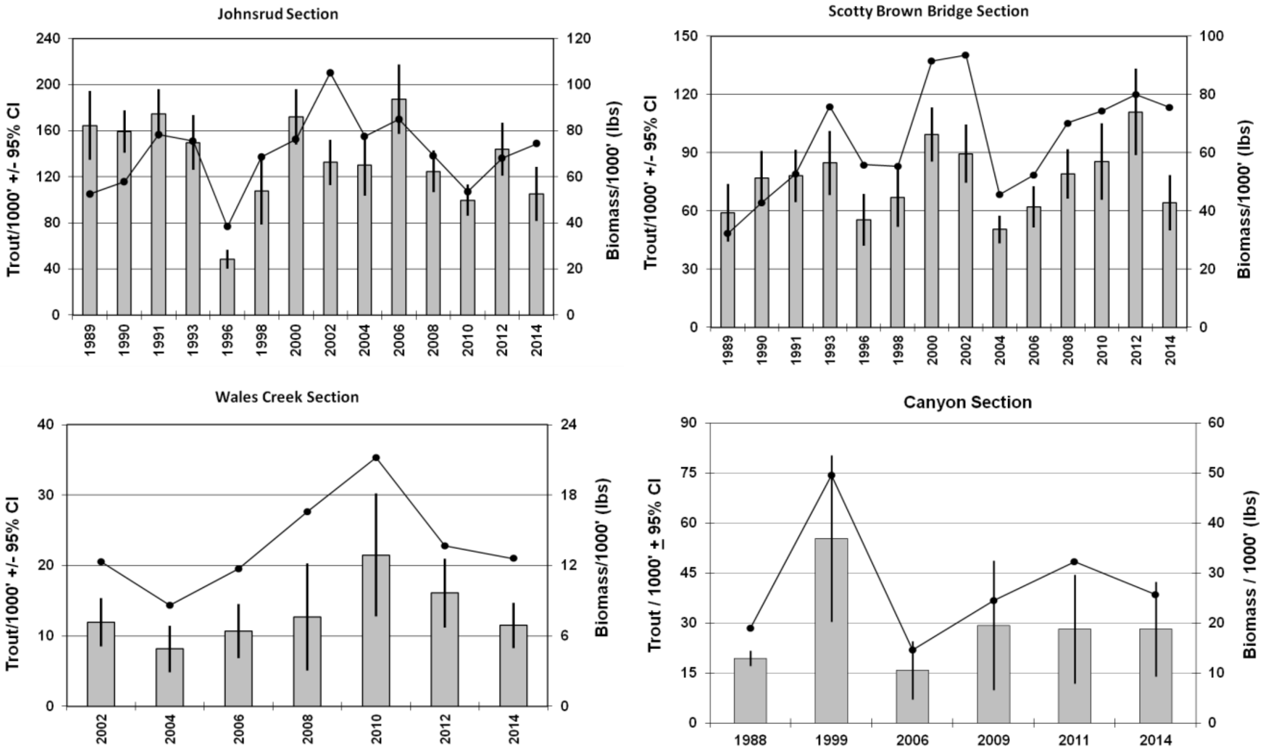


Figure 10. Estimates for total trout abundance and biomass (all trout >6.0”) at four monitoring sites on the mainstem Blackfoot River, 1988 - 2014. Survey names relate to survey sites on Figure 6. Species composition associated with these data sets are shown below on Figure 11.

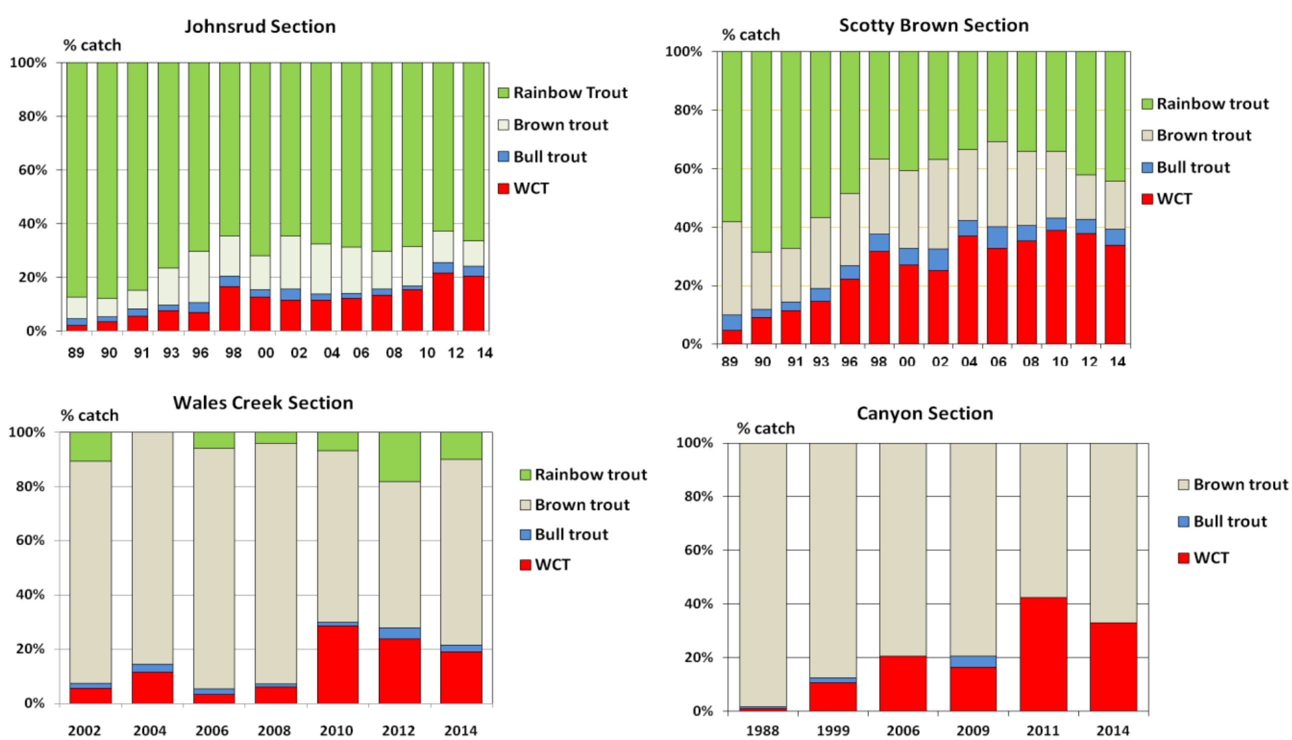
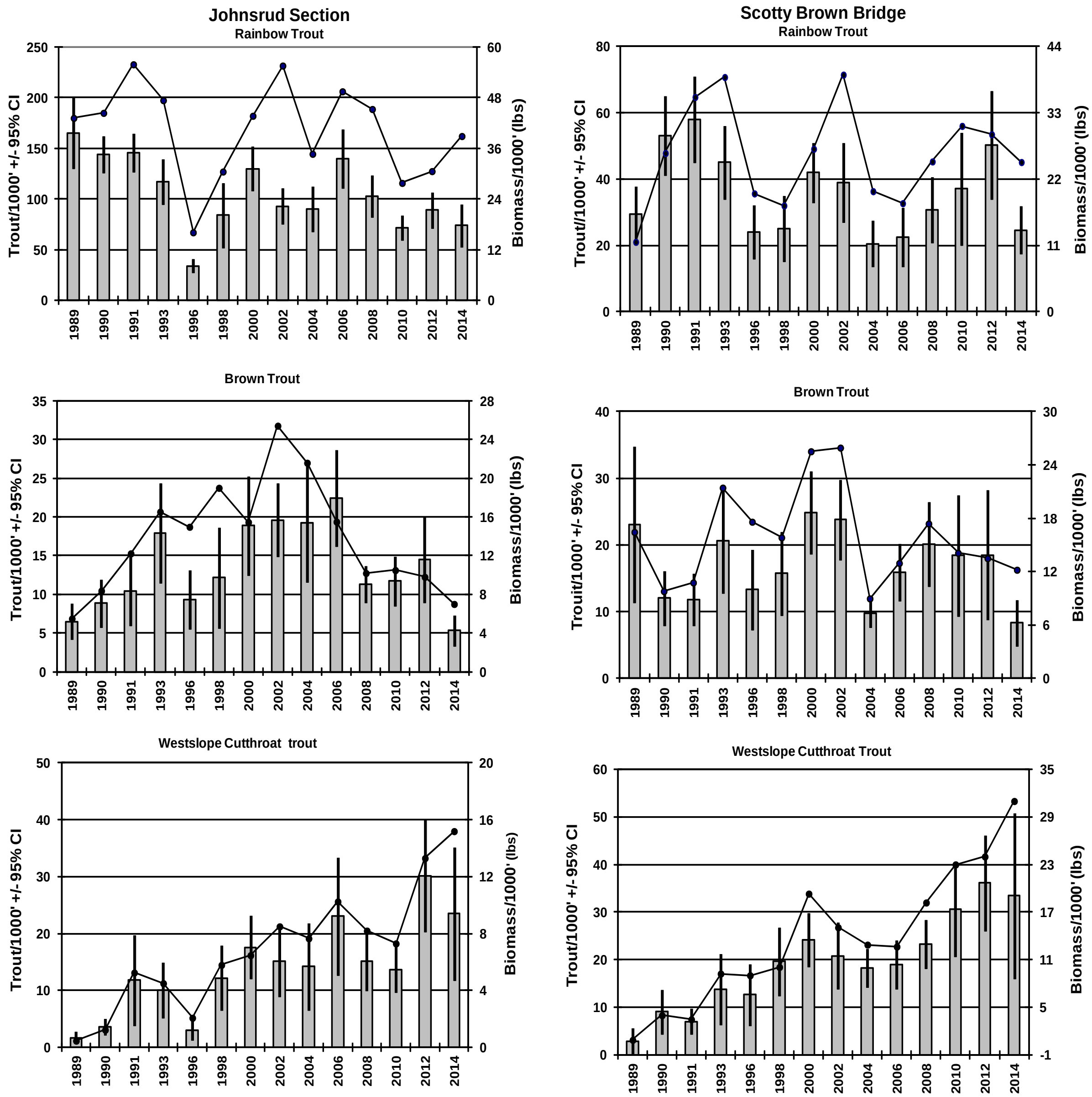


Figure 11. Percent trout species composition (fish >6.0") at four long-term fish population monitoring sites on the mainstem Blackfoot River, 1988-2014.



Figures 12. Estimates of trout (fish >6.0") abundance (bars) and biomass (lines) in the Johnsrud section (left column) and Scotty Brown section (right column), 1989-2014.

Johnsrud section: The 2014 trout species composition (fish >6.0") in the Johnsrud section was 63.6% rainbow trout (*n*=528), 8.8 % brown trout (*n*=107), 19.6% westslope cutthroat trout (*n*=151) and 3.5% bull trout (*n*=29). The total trout point estimate (all trout >6.0") for the Johnsrud Section decreased from 144 fish/1,000' in 2012 to 105 fish/1,000' in 2014, a 27% decrease (Figure 10). Despite this decrease, the total trout biomass estimate for fish >6.0" in the Johnsrud section increased 9% from 68.4 lbs/1,000' in 2012 to 74.3 lbs/1,000' in 2014.

Estimates of abundance and biomass for individual trout species from 1989 through 2014 are shown on Figure 12. The point estimate for westslope cutthroat trout (> 6.0") decreased from 30.2 fish/1,000' in 2012 to 23.5 fish/1,000' in 2014 (Figure 12). The 2012 point estimate for brown trout (> 6.0") decreased from 14.5 fish/1,000' in 2012 to 5.3 fish/1,000' in 2014. The point estimate for rainbow trout (>6.0") abundance decreased from 89 fish/1,000' in 2012 to 74 fish/1,000' in 2014. Because of a small sample size and a low recapture rates, we were unable to generate a valid bull trout population estimate in 2014. Associated biomass estimates are shown in Figure 12.

Scotty Brown Bridge section: The 2014 percent trout composition in the Scotty Brown Bridge section was 43.5% rainbow trout (*n*=201), 33.3% westslope cutthroat trout (*n*=154), 16% brown trout (*n*=74) and 5.4% bull trout (*n*=25). Total trout abundance (all trout >6.0") decreased 42% from 111 to fish/1,000' in 2012 to 64 fish/1,000' in 2014. The total trout biomass estimate for trout >6.0" in the Scotty Brown Section decreased only 5.7% from 80.2 lbs/1,000' in 2012 to 75.6 lbs/1,000'in 2014.

Estimates of abundance and biomass for all trout species (fish >6.0") in the Scotty Brown Bridge section are shown in Figure 12. The rainbow trout point estimate of abundance decreased from 50.3 fish/1,000' in 2012 to 24.6 fish/1000' in 2014. The point estimate for westslope cutthroat trout (fish >6.0") abundance decreased slightly from 36.1 fish/1,000' in 2012 to 33.4 fish/1,000' in 2014. The point estimate for brown trout (fish >6.0") decreased from 18.5 fish/1,000' in 2012 to 8.3 fish/1,000' in 2014. The bull trout point estimate was 2.3 fish/1000' in 2014.

Wales Creek section: The Wales Creek section was established in 2002 and has been monitored every two years concurrent with the Johnsrud and Scotty Brown surveys (Figure 10). In May 2014, trout species composition in the Wales Creek section was 69% brown trout (*n*=115), 18.9% westslope cutthroat trout (*n*=32), 10.1% rainbow trout (*n*=17) and 2.4% bull trout (*n*=4). Estimates of total trout abundance (all trout > 6.0") for the Wales Creek section were 11.5 trout/1,000' in 2014 compared to 16.1 in 2012 (Figure 10). The total trout biomass estimate for trout >6.0" in the Wales Creek in 2014 was 12.1 lbs/1,000' compared to 10.2 lbs/1,000' in 2012 (Figure 10). These estimates of abundance and biomass were considerably lower than upriver (Canyon Section) and downriver (Scotty Brown) samples (Figure 10).

Estimates of abundance from 2002 through 2014 for individual species are shown in Figure 13. Population estimates for mountain whitefish in the Wales Creek section are summarized below.

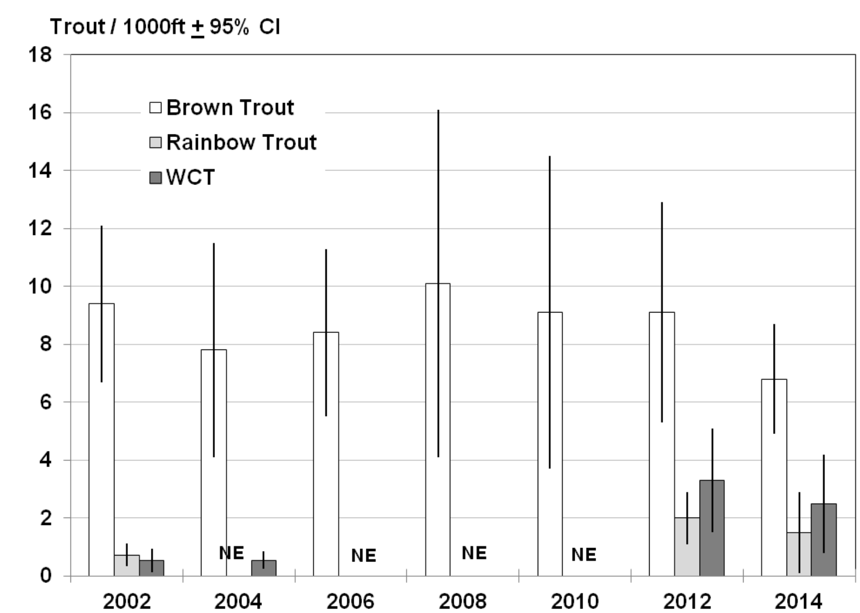


Figure 13. Estimates of trout abundance (fish > 6.0") in the Wales Creek section, 2002-2014. An 'NE' indicates no estimate for rainbow trout or westslope cutthroat trout.

Canyon section: Fish populations in the Canyon section were sampled in 1988, 1999, 2006, 2009, 2011 and 2014. The long-term dataset for total trout abundance and biomass (all trout >6.0") is shown in Figure 10. Trout species composition for these years is shown in Figure 11. In 2014, brown trout (*n*=41) were again the prevalent trout comprising 66.1% of the sample versus 57.6% in 2011. The percentage of westslope cutthroat trout decreased from 42.4% (*n*=25) in 2011 to 32.3% (*n*=20) in 2014. The total trout point estimate (all trout >6.0") in the Canyon section increased from 26.0 fish/1,000' in 2011 to 35.2 fish/1,000' in 2014. A comparison of westslope cutthroat trout abundance versus total trout abundance (fish >6.0") is shown in Figure 14. Similar to the Wales Creek section, we continued to monitor mountain whitefish in the Canyon section in 2014 as described below (Appendix C).

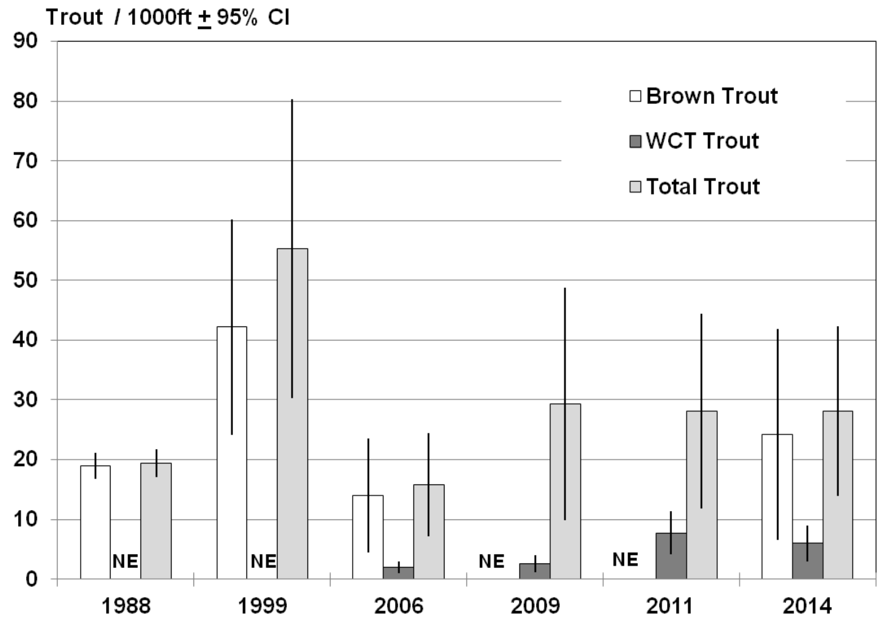


Figure 14. Estimates of trout abundance (fish > 6.0") in the Canyon Section, 1988-2014. An 'NE' indicates a valid estimate was not obtained.

Mountain whitefish surveys

Mountain whitefish (MWF) occupy the larger, low elevation streams, rivers and lakes of western Montana. In the Blackfoot River Basin, the low-elevation distribution of MWF overlaps with that of the exotic parasite *Myxobolus cerebralis*, the cause of salmonid whirling disease (Pierce et al. 2011, 2012).

To investigate MWF status and possible relationships with whirling disease, FWP began monitoring/compiling MWF population data in the Blackfoot River in 2006 during the peak of the *Myxobolus cerebralis* epizootic (Pierce et al. 2011). We followed the status review with a MWF life history (telemetry) study with emphasis on spawning behavior of adult MWF combined with *M. cerebralis* disease testing (Pierce et al. 2012; Beth MacConnell unpublished report 3-20-11). These studies found 1) both migratory and nonmigratory spawning behavior among MWF in the Blackfoot River, 2) *M. cerebralis* infected age 0 MWF in the upper Blackfoot River (near Lincoln), and 3) whirling disease in low numbers of juvenile MWF with caudal deformities in the Wales Creek section (Pierce et al. 2012; Beth MacConnell Technical Report 3-20-11).

Because of *M. cerebralis* infection and the presence of whirling disease in MWF in the upper Blackfoot River, we continued to monitor MWF populations at the Canyon and Wales Creek sections from 2006 through 2014. Population surveys found no change in abundance (MWF >8.0" in total length) at either section (Figures 15), though survey data suggests a reduction in biomass in the Wales Creek Section in recent sampling. This reduction primarily relates to a slight reduction in average total length and corresponding decline in condition factor (Appendix C). These reductions are inconsistent with past whirling disease studies in the Blackfoot Basin that show aging population, recruitment loss and density-dependant increases in condition factor (McMahon et al. 2010). Given these inconsistencies, changes in biomass are likely not whirling disease related.

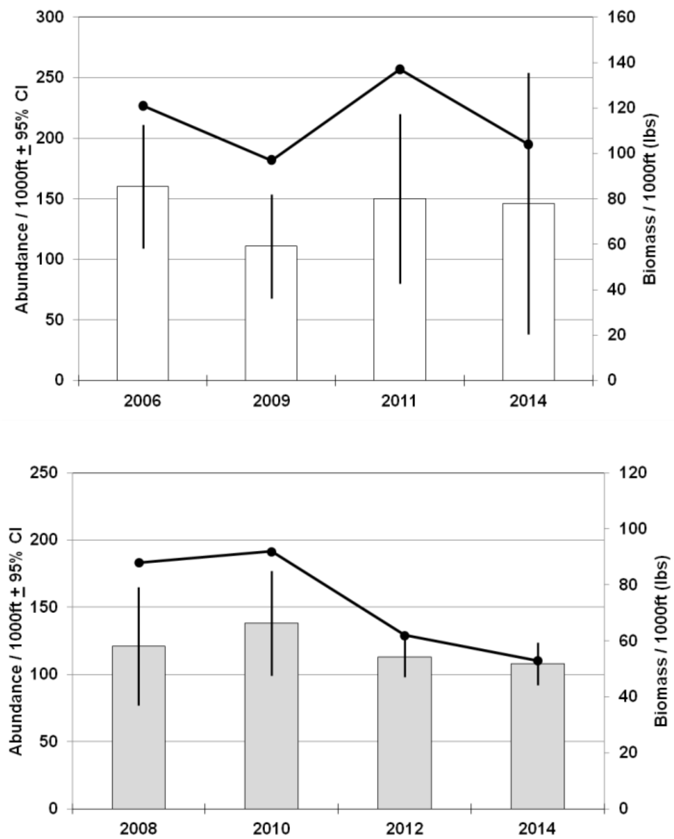


Figure 15. Estimates of abundance and biomass for age 2 and older mountain whitefish (> 8.0’’) in the Canyon Section (top) and Wales Creek Section (bottom).

Bull Trout Recovery

Bull trout, an imperiled inland char native to western Montana, was listed as "threatened" under the Endangered Species Act in 1998. In 2000, *core area* watersheds were delineated to broadly foster restoration and protection of riparian habitat in the headwaters of spawning streams (Figure 16, MTBRT 2000). In 2010, the United States Fish and Wildlife Service designated final critical habitat for the recovery of bull trout (USFWS 2010). This designation includes various streams, lakes and rivers within designated core areas the Blackfoot Watershed (Figure 16), which further includes all major bull trout spawning streams. In 2014, the USFWS completed their bull trout recovery plan that emphasizes threat-based reductions to bull trout habitat (USFWS 2015).

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 currently relies on no-harvest angling regulation, combined with restoration and protection of critical waters with corridors connecting spawning, rearing and refugia habitat (Figure 16).

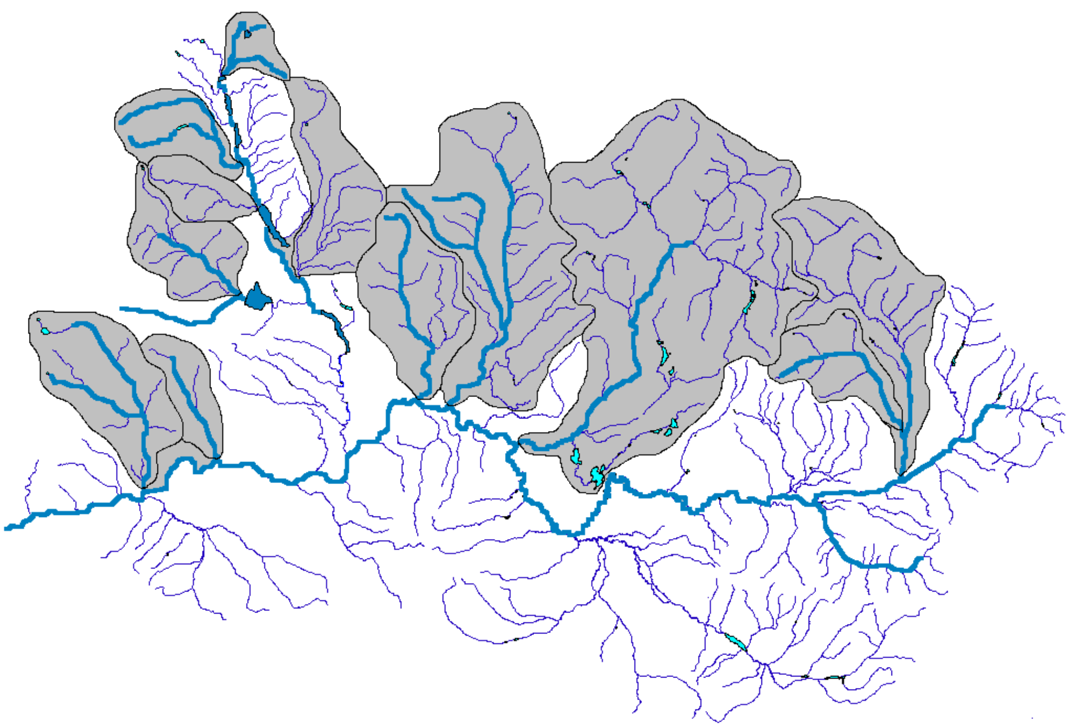


Figure 16. Bull trout recovery areas: The grey watersheds show bull trout "core areas" (MTBRT 2000). The bold blue lines show critical bull trout habitat (USFWS 2010).

Within these broader recovery areas, migratory bull trout life histories involve spawning in discrete areas, tributary use by early life-stages, large home ranges, extensive spawning 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 ≤57°F are considered optimal for bull trout (Dunham et al. 2003). Because of these requirements, bull trout are highly sensitive to human alterations of aquatic conditions.

Stream-resident bull trout require similar environments and complete their life-cycle in tributary streams. Adfluvial bull trout, rare in the upper Clark Fork Basin, 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 were extensively studied in the Blackfoot basin (Swanberg 1997; Schmetterling 2003; Pierce et al. 2005; Benson 2009). These studies, along with state and federal recovery plans, provide the framework for restoration and recovery actions (MTBRT 2000; Pierce et al. 2008; USFWS 2010).

Since 1990, many restoration actions targeting the recovery of bull trout in the Blackfoot Watershed have been completed in all core areas. Major restoration actions include: 1) enhancing instream flows and improving fish passage by screening

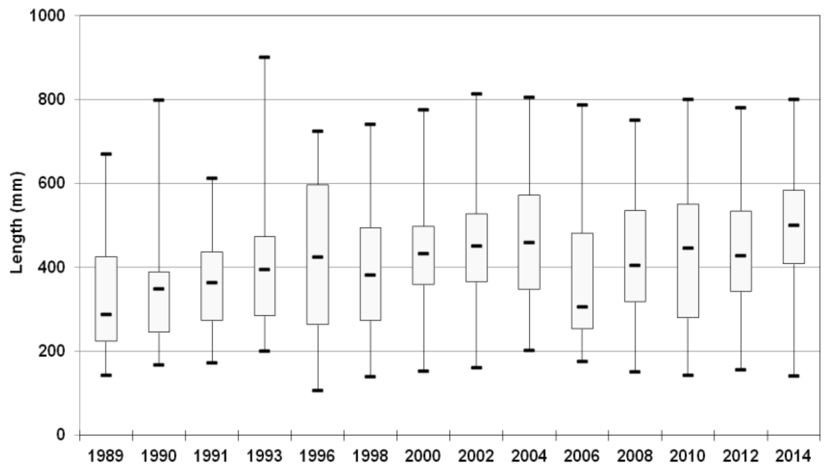


Figure 17. Range of total lengths for all individual bull trout collected in the Johnsrud and Scotty Brown Bridge monitoring sections of the Blackfoot River, 1989-2014. Box plots represent the range of total lengths (minimum and maximum) the 25th and 75th percentiles and median values.

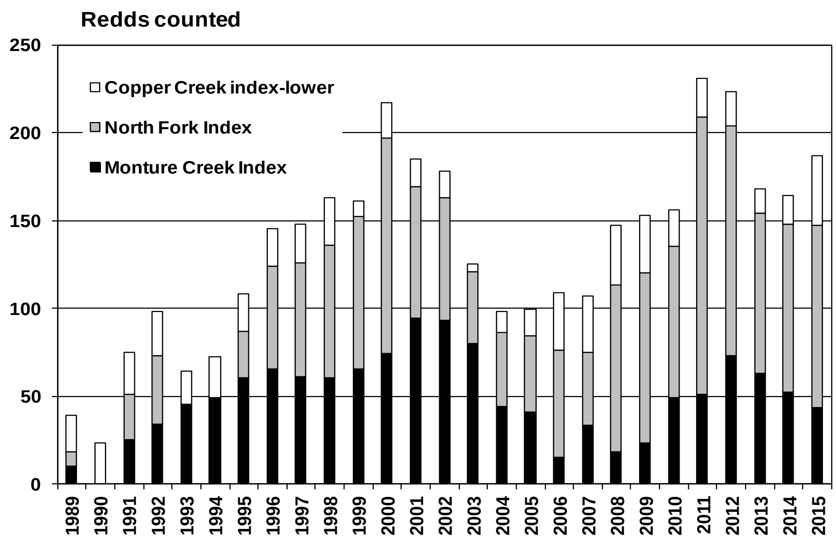


Figure 18. Bull trout redd counts for three spawning tributaries used by migratory bull trout of the Blackfoot River, 1989-2015.

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	2013	2014	2015
East Fork Clearwater	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3	21	21	18	-	6	0	6	2
West Fork Clearwater	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	19	20	65	33	66	74	49	60	47
Marshall Creek (upper)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3	0	-	-	5	5	6	5
Copper Creek (lower)	-	21	23	24	25	19	23	21	21	22	27	9	20	16	15	4	12	15	33	32	34	33	21	22	19	14	16	40
Copper Creek (upper)	-	-	-	-	-	-	-	-	14	19	17	29	24	21	23	14	19	35	51	79	62	82	44	39	44	22	29	17
Snowbank Creek	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	27	1	17	13	35	24	21	-
Belmont Creek	-	-	-	-	-	-	-	-	14	-	14	8	4	3	11	-	-	3	-	-	-	-	-	1	2	1	0	-
Gold Creek	-	-	-	-	-	-	-	-	-	-	16	30	9	17	6	4	-	7	-	1	2	1	1	0	1	1	2	-
Dunham Creek	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11	6	6	4	-	5	7	5	8	8	9	6	11	4
Monture Creek (index)	11	10	-	25	34	45	49	60	65	61	60	65	74	94	93	80	44	41	15	33	18	23	49	51	73	63	52	43
Monture Creek (upper)									14	10	7	10	6	0	8	3	9	1	0	0	14	12	3	9	3	3	2	0
Morrell Creek	-	-	-	-	-	-	-	-	-	-	-	-	-	-	24	10	22	16	26	4	33	54	37	27	38	24	28	36
North Fork Blackfoot River	12	8	-	26	39	-	-	27	59	65	76	87	123	75	70	41	42	43	61	42	95	97	86	158	131	91	96	104

Table 1. Summary of bull trout redd counts in the Blackfoot River drainage, 1988-2015.

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 to eliminate northern pike and restore fish passage, 4) the placement of conservation easements on segments of several spawning streams, and 5) fish passage enhancement on two low-head dams on the mainstem Clearwater River. The recent TNC purchase of Plum Creek Timber Company lands in the Clearwater and lower Blackfoot Basin may have potential to benefit bull trout.

From 2013 through 2015, the USFS and FWP cooperated on a genetic assignment study of all major bull trout stocks the Blackfoot Basin (Results Part IV). The study collected genetic samples (fin clips) from streams of bull trout origin (spawning and rearing streams) along with fin clip samples from migratory bull trout in the Blackfoot River and Clearwater Lakes. This study not only identified genetically distinct stocks among most spawning streams, but also identified specific stocks that provide downstream recruitment of migratory fish to the Blackfoot River and the Clearwater chain of lakes, as well as spawning streams that provide little, if any, recruitment to the Blackfoot River. This study confirmed Monture Creek, the North Fork Blackfoot River, Copper Creek and Snowbank Creek as all contributing recruitment to the Blackfoot River. This study also identified, for the first time, the biological connection of Blackfoot River bull trout (genetically assigned to the North Fork and Monture Creek) with Salmon Lake in the lower Clearwater River drainage. These specific findings point to a need to screen fish from at least two large irrigation canals on the upper Blackfoot River and one on the lower Clearwater River.

Following the 1990 adoption of protective (catch-and-release) harvest regulations, as well as early recovery actions (Pierce et al. 1997), long-term monitoring of bull trout populations showed 1) an increase in redd counts during the decade of the 1990s for the three primary spawning tributaries (Figure 18), 2) an inclination towards larger fish in the lower Blackfoot River (Figure 17), and 3) the identified the critical role of Monture Creek, the North Fork and Copper Creek in supporting migratory stocks (Swanberg 1997; Schmetterling 2003; Results Part IV). However, long-term monitoring also reveal continued bull trout declines in Gold, Belmont and Cottonwood Creeks (Table 1, Results Part III), all lower elevation streams with a legacy of intensive land use. The highly precarious status of bull trout in Gold, Belmont and Cottonwood Creeks currently indicate very low population viability and may point to near-term loss of bull trout from these streams.

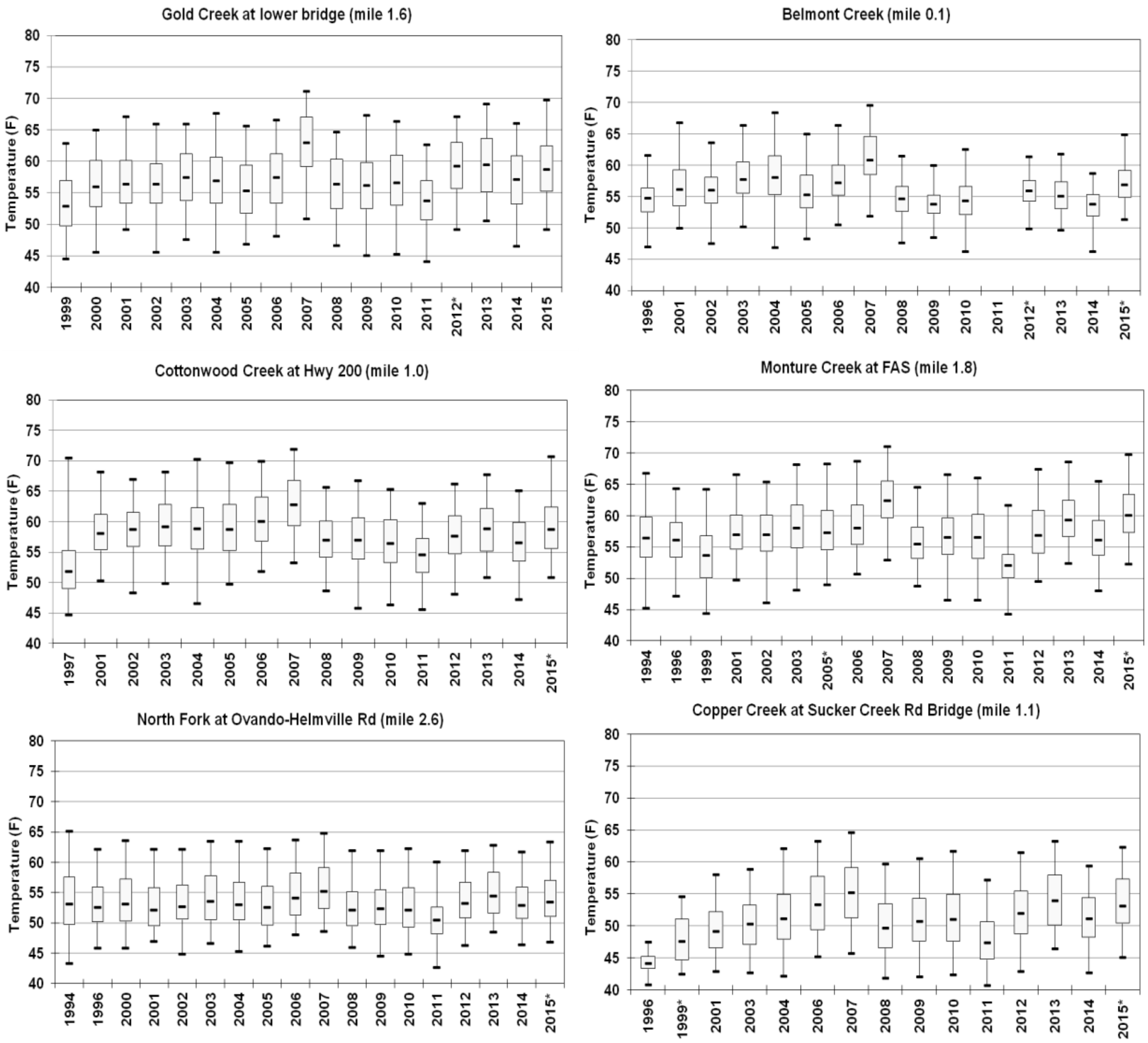


Figure 19. July water temperatures summaries on the lower reaches of six tributaries supporting bull trout spawning. Monitoring locations are shown of Figure 7.

PART III: Tributary Restoration and Fisheries Monitoring

Summaries of fisheries improvement work and fish population monitoring results for restoration streams are organized alphabetically by stream name with the exception of those streams in the Keep Cool drainage. Fish population surveys on 10 streams in the Keep Cook drainage were part of a concerted 4-year fisheries investigation. Survey summaries for those 10 steams are located in the Keep Cool Creel drainage section. These summaries are organized alphabetically by stream name. All stream locations and sampling sites (e.g., mile 4.0) refer to stream mileage.

All tributary fish population survey sites are shown of Figure 6. All supporting data are summarized in the appendices, as follows: catch and size statistics (Appendix A), population estimates (Appendix B), water chemistry readings (Appendix D), water temperature data (Appendix E), and westslope cutthroat trout genetic test results (Appendix F). Appendices G, H and I describe lists of restoration actions (Appendix G), cooperators (Appendix H), and identify fisheries impairments as observed by FWP fisheries biologists (Appendix I).

Ashby Creek

Restoration objectives: Protect the genetic purity of westslope cutthroat trout in the upper Ashby Creek watershed using an existing wetland complex as a migration barrier; 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.

<u>Project summary</u>

Ashby Creek is a small 2nd order tributary to Camas Creek located in the Union Creek basin. Ashby Creek is eight miles in length and drains a 24.8 mile^2 watershed. The stream originates near Mineral Ridge in the Garnet Mountain range and drains a forested basin with a mix private, DNRC and BLM properties before entering private ranchlands near stream mile 3.8. Major tributaries include the East Fork Ashby Creek entering at mile 4.5 and Arkansas Creek entering at mile 1.4. Stream gradients range from 570ft/mile at headwaters to 45ft/mile the lower mile of stream. In 2010, all former Plum Creek Timber Company land upstream of mile 3.8 was transferred to the DNRC. Fisheries impairments include roads in riparian areas, undersized culverts, past agricultural practices on private lands that included overgrazing of the riparian zone, channelization 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 inhibits the upstream movement of fish. The project was complete when conservation easement was placed on the cooperating ranch to preserve the rural character and natural resources of the property. More recent work was then initiated on upstream DNRC lands in 2013 following the transfer of former Plum Creek Timber Company land to the DNRC. This DNRC work removed a 0.8 mile segment road from the riparian area, restored 0.46 miles of stream, and fenced livestock from the adjacent riparian area.

Fish population monitoring

Ashby Creek supports a resident population of non-hybridized westslope cutthroat along with brook trout in low numbers. We began monitoring for fisheries response at two sites on private land treatment area of Ashby Creek in 2007 and continued through 2014. These surveys show increased abundance of trout in the upstream treatment reach and down-valley dispersion in the lower project area (Figure 20).

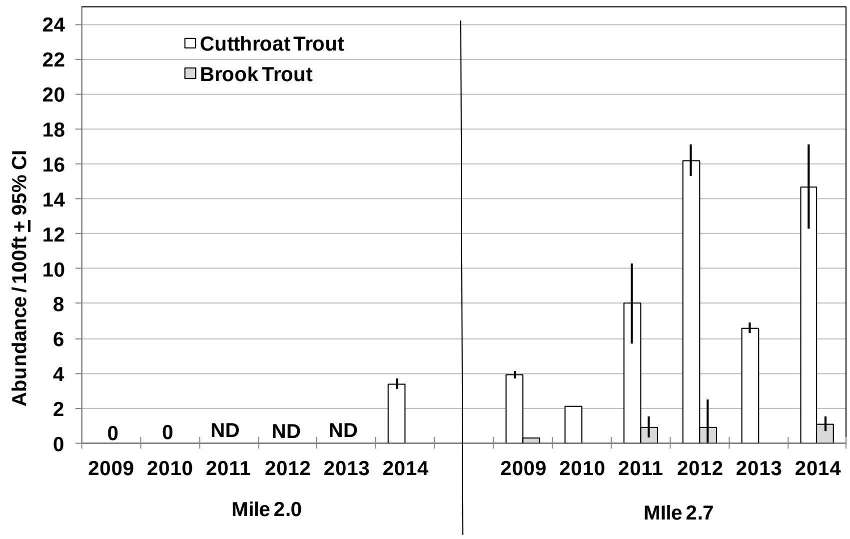


Figure 20. Estimates of trout abundance for age 1 and older fish for two treatment reaches of Ashby Creek, 2009-2014.

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 2nd order tributary to the lower Blackfoot River, flows 6 miles north to its mouth where it enters the Blackfoot River at river mile 12.2 with a base-flow of 3-5 cfs. Its headwaters drain the east and southeastern slopes of Olsen Mountain in the Garnet Mountain range with stream gradients ranging from 460ft/mile in the upper reaches to 135ft/mile in the lower mile of stream. In 2010, all former Plum Creek Timber Company land in the Bear Creek drainage was transferred to DNRC. Bear Creek is one of the colder tributaries to the lower Blackfoot River. Located on DNRC and private land, 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 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) reconstruct or enhance habitat complexity on 4,000 feet of stream, 4) shrub plantings, and 5) the development of compatible riparian grazing systems for one mile of stream.

Fish populations

Bear Creek supports predominately rainbow trout along 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 2014 (Figure 21).

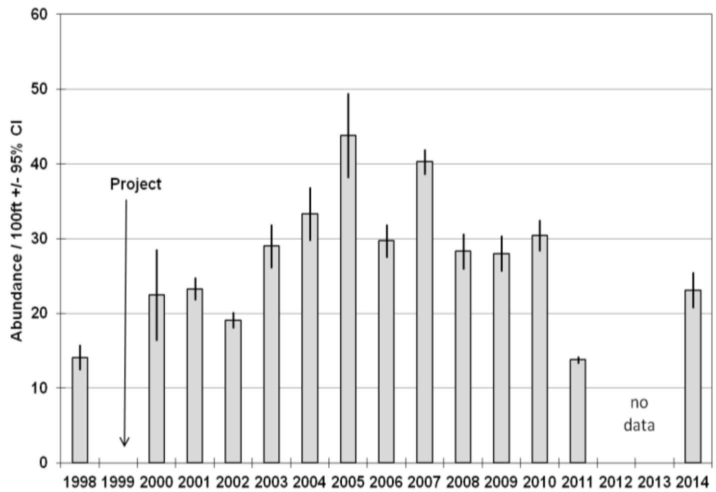


Figure 21. Estimates of total trout abundance for age 1 and older fish in Bear Creek at mile 1.1, 1998-2014.

Belmont Creek

Restoration objectives: Restore pool habitat and morphological complexity; restore thermal refugia for Blackfoot River native fish species. Reduce chronic and episodic road sediment sources and increase native fish habitat availability.

Project summary

Belmont Creek, a 2nd order stream, drains a 29.2 mi^2 watershed and flows southeast 11 miles before entering the lower Blackfoot River at mile 21.8 with a baseflow of 10-12 cfs. The headwaters of Belmont Creek originate on a checkerboard of Lolo National Forest, DNRC, and TNC lands on the slopes of Gold Creek Peak and Belmont Peak. Stream gradients range from 220 ft/mile in the upper most reaches moderating to 80 ft/mile in the middle reaches that increases to 240ft/mile between miles 3.0 and 4.0 before decreasing to 100ft/mile downstream of mile 3.0. Ninety two percent of Belmont Creek watershed was managed as industrial forest by Plum Creek Timber Company prior to The Nature Conservancy purchase. The remaining 8% consists of USFS and DNRC property in the headwaters along with BLM property and one small private inholding in the lower basin.

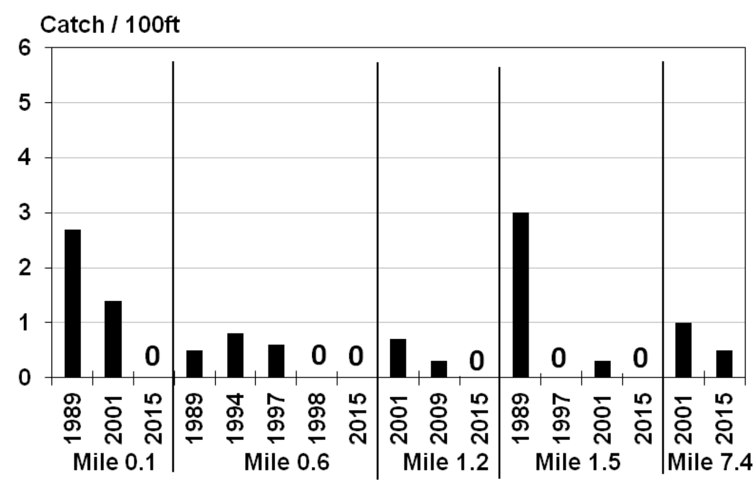


Figure 22. CPUE for all bull trout sampled at five sites on Belmont Creek, 1989 - 2015

Majority of the stream classifies as Rosgen B2-B4 stream types with the exception of higher gradient areas classifying as A1-A3 stream types with boulder, cobble, gravel substrate with areas of bedrock substrates. Instream wood is an important habitat feature that adds complexity and helps form scour pools, plunge pools and riffles. The middle reaches are heavily beaver influenced with elevated levels of instream sediment. The riparian vegetation consists of a conifer-dominated overstory with a dense understory of alder, willow, red osier dogwood, chokecherry as well as grasses and sedges along the banks.

Prior restoration actions included the removal of culverts at road/stream crossings, the replacement of a perched culvert with a bridge, reduction in grazing pressure and an instream wood project on BLM properties. There were also several sediment reduction measures associated with logging roads including the installation of rolling dips, and seeding and closing roads after logging. In 2015, TNC completed road inventories to identify additional sediment reduction along with riparian measures (Inroads Consulting 2015). There were 225.5 miles of roads inventoried in the Belmont watershed (including those on other ownerships than TNC), a road density of 7.7 mi/mi^2 and 4.3 miles of road within 50 feet of perennial streams (Amy Pearson, TNC, personal communication). There were 48 culverts inventoried at stream crossings with defined channels on TNC lands. Key road-related sediment sources include two ephemeral stream diversions, stream-side roads, and hazards associated with culverts blocking and failing. There was also one 18 inch culvert on a perennial creek that is perched and potentially blocking fish passage (Inroads Consulting 2015).

Fish populations

We resurveyed Belmont Creek in 2015 at five monitoring sites (miles 0.1, 0.6 1.2, 1.5 and 7.4) where bull trout were present in prior surveys. The 2015 surveys failed to find bull trout in the four lower sample locations and captured bull trout only at the upstream (mile 7.4) site (Figure 22). The 2015 surveys also indicate brown trout are expanding in the upstream direction. Consistent with bull trout declines in these electrofishing surveys, bull trout redd counts, conducted by Plum Creek Timber Company, show a similar declining trend (Table 1). With the sale of Belmont Creek forest land to TNC, Plum Creek Timber Company ceased bull trout redd counts in 2014. The BLM will assume future bull trout redd counts.

Braziel Creek

Restoration objectives: Reestablish natural channel conditions, improve riparian area and enhance flows to increase westslope cutthroat trout numbers.

Project Summary

Braziel Creek, a small 2nd order tributary, enters Nevada Creek at mile 24.5 about two miles downstream of the Nevada Creek Reservoir with base flow of <1.0 cfs. The stream is 3.7 miles in length drains the foothills of Hoodoo Mountain. The upper 1.9 miles of stream are located on BLM property and the lower 1.8 miles flow through private ranchlands. Stream gradients range from 77ft/mile the lower 0.5 mile of stream but increases significantly to an average of 405ft/mile the upper 3.2 miles of stream.

Fisheries impairments on Braziel Creek include road drainage and grazing pressure. Prior to restoration in 2010, lower Braziel Creek was heavily altered from channelization, dewatered and subject to heavy riparian grazing. Furthermore, undersized culverts limited fish passage, and westslope cutthroat trout entrainment had been indentified in one irrigation ditch at mile 0.26 (Pierce et al. 2011). To improve conditions for westslope cutthroat trout, a comprehensive restoration project was initiated in 2010, which included reconstruction of 1,500 feet of channel and re-vegetation upgrade of an undersized county road culvert, installation of a Coanda fish screen at one diversion and a grazing management plan. The grazing plan excluded livestock to recover riparian vegetation and stabilize the new channel. The landowner has entered into an agreement with Trout Unlimited in 2013 for minimum flows of 0.5 cfs. In 2011, the immediate downstream reach of Braziel Creek (540') was also restored. The reach suffered from channelization, bank erosion and simplified habitat.

Fish populations

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 five years of post-treatment monitoring (Figure 23). Low abundance in 2014 may be the result of severe winter flooding and severe icing of the channel.

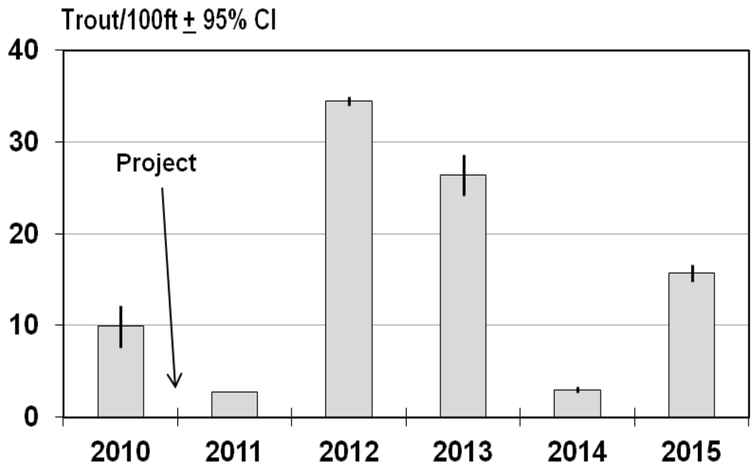


Figure 23. Estimates of trout abundance for age 1 and older westslope cutthroat trout in Braziel Creek at mile 0.2, 2010-2015.

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, a small 2nd order tributary, drains a 22.4 mile^2 basin, enters to the Blackfoot River at river mile 43.9 with a base-flow of 2-3 cfs. Chamberlain Creek is 11.5 miles in length, and originates on the eastern slopes of Lost Horse Mountain near Chamberlain Meadows in the Garnet Mountain range. Following recent land exchanges, the majority of the upper basin is now BLM land, most of the middle basin is DNRC land, and the lower basin is privately owned. Stream gradients average 118ft/mile from the mouth to stream mile 5.2. Upstream of mile 5.2, stream gradient increases significantly to an average of 301ft/mile to approximately stream mile 8.0. Gradient moderates to 150ft/mile upstream of mile 8.0.

Prior to 1990, sections of lower Chamberlain Creek were dewatered, damaged by heavy riparian grazing, road encroachment and channelization, which led to low westslope cutthroat trout abundance in lower stream reaches (Peters 1990). Initiated in 1990, Chamberlain Creek was one of the first comprehensive restoration projects within the Blackfoot Basin. Restoration emphasized road drainage repairs, removal of livestock from riparian areas, in-stream habitat restoration, irrigation upgrades (consolidation of two ditches into one and the install of a fish ladder on the diversion), enhanced stream flows through water leasing and the placement of a conservation easements on all ranchlands in the lower basin. In 2010, over 13,000 acres of former Plum Creek Timber Company land was transferred to DNRC with special easement provisions to remove 5.5 miles of roads adjacent to Chamberlain Creek, the West Fork of Chamberlain Creek and Bear Creek, as well as the removal of culverts to meet fish passage and natural stream function. A similar land exchange of 6,080 acres of former Plum Creek Company checkerboard ownership in the middle basin were later transferred to the BLM. With the completion restoration projects, conservation easements and land exchanges, the Chamberlain Creek project has now addressed all known primary impairments to fisheries, while achieving landscape-level conservation for the entire Chamberlain Creek drainage.

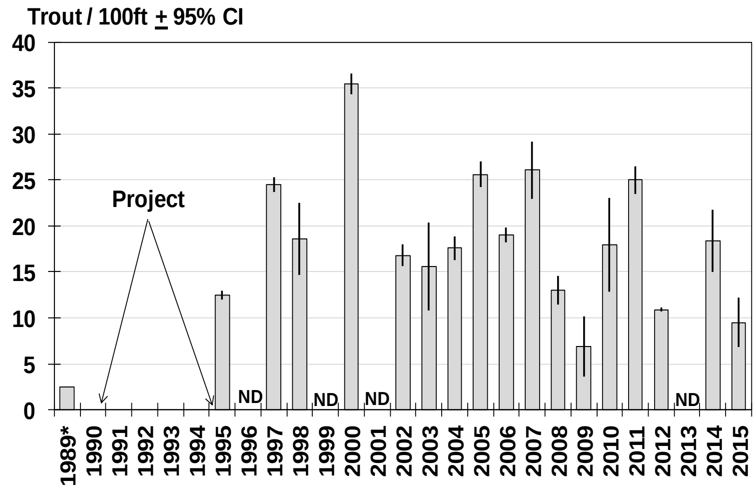


Figure 24. Estimates of trout abundance for age 1 and older westslope cutthroat trout in Chamberlain Creek at mile 0.1, 1989-2015.

Fish populations

Chamberlain Creek is a westslope cutthroat trout dominated stream over its entire length, though 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 2015 (Figure 24). 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

Copper Creek is a large 3rd order tributary to the lower Landers Fork entering near mile 4.1. Copper Creek, 17.6 miles in length, drains a 40.6 mile^2 forested watershed and generates a base flow of about 20-25 cfs. The mainstem originates from two small cirque lakes (Upper and Lower Copper Lakes) within the Helena-Lewis and Clark (H-LC, hereafter) National Forest. Stream gradients range from 940ft/mile near the headwaters to 78ft/ mile at the mouth. Headwater tributaries are Red Creek (mile 11.6), Cotter Creek (mile 11.5) and the North Fork of Copper Creek (mile 8.8). Snowbank Creek enters Copper Creek at mile 6.2. The upper 13.8 miles of Copper Creek lies within the H-LC National Forest; whereas, the lower 3.8 stream miles of Copper Creek flows through a mix of Sieben Ranch, State, private, and H-LC National Forest lands.

Copper Creek is among the coldest of all bull trout streams in the Blackfoot Basin (Figure 19, Appendix E), which help moderate temperatures in the lower Landers Fork.

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

Fish populations

Copper Creek supports an entirely native fish community basin-wide. The mainstem provides critical bull habitat as well as spawning and rearing for genetically pure fluvial westslope cutthroat trout that also inhabit the upper Blackfoot River.

Following the 2003 wildfire, electrofishing surveys and redd counts both showed a substantial increase in bull trout from 2006 through 2009 followed by a decline in recent years (Figure 25). Likewise, electro-fishing surveys at a long-term monitoring site (mile 6.2) showed a similar pattern for westslope cutthroat trout (Pierce et al. 2013). These trends, which extend into Snowbank Creek, are likely the result of increases, followed by decreases, in basic stream productivity following the Snow Talon Fire in 2003. In addition, the USFS has conducted annual bull trout redd counts surveys since 1989 at an index section, then began conducting total redd count surveys in 1996 when a telemetry study identified a second bull trout spawning area. Redd counts are shown in Figure 25.

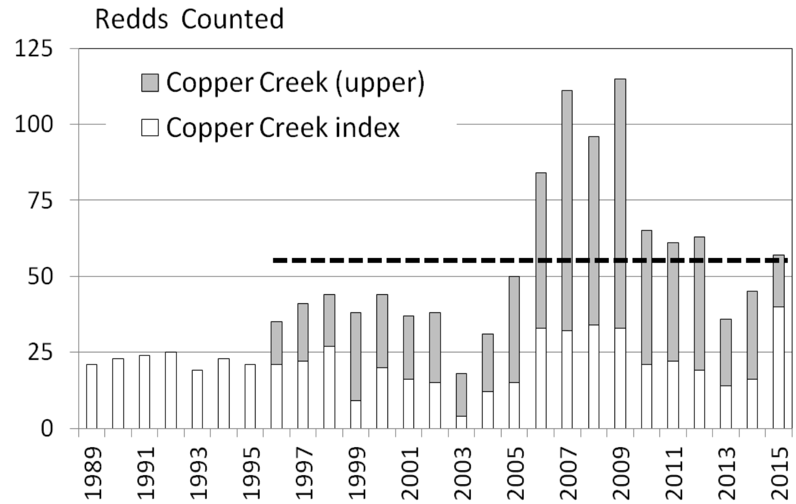


Figure 25. Bull trout redd counts for Copper Creek: White bars show redd counts in long-term (1989-2015) index section. Grey bars show the redd counts in the upstream section monitored from 1996-2015. The dashed horizontal line shows the long-term mean of 55 redds for the total bull trout redd count, 1996-2015.

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 3rd order stream, drains 70 mile^2 watershed, and flows approximately 18 miles south from Morrell Mountain and enters the Blackfoot River at mile 43 with a baseflow of about 15-20 cfs. The North Fork of Cottonwood Creek, entering a mile 13.4, is the largest tributary to upper mainstem, and Shanley Creek, entering at mile 5.6, is the largest tributary to the lower mainstem. Stream gradients range from 830ft/mile in the headwaters to 40ft/mile in the lower mile of stream. The stream originates on the Lolo National Forest before entering State (FWP, DNRC and University of Montana) and small sections of private land beginning at mile 11.9.

Cottonwood Creek has been the focus of ongoing restoration since 1996. Fisheries improvements began with a fish-friendly irrigation project at mile 12.0 that enhanced flows, improved fish passage and screened fish from an irrigation ditch. Prior to this work, a middle section of Cottonwood Creek was completely dewatered during late summer and fall by irrigation. Later work included the removal of two diversions, instream flow enhancement on lower Cottonwood Creek and riparian fencing projects on the Blackfoot Clearwater Game Range to reverse livestock degradation of the channel. Currently, the USFS is completed 1.0 miles of channel reconstruction (mile 14.6 to 15.5) to reduce sediment levels and has upgraded two culverts to a bridge and bottomless arch culvert. Together these improvements have reconnected approximately 4.0 miles of habitat.

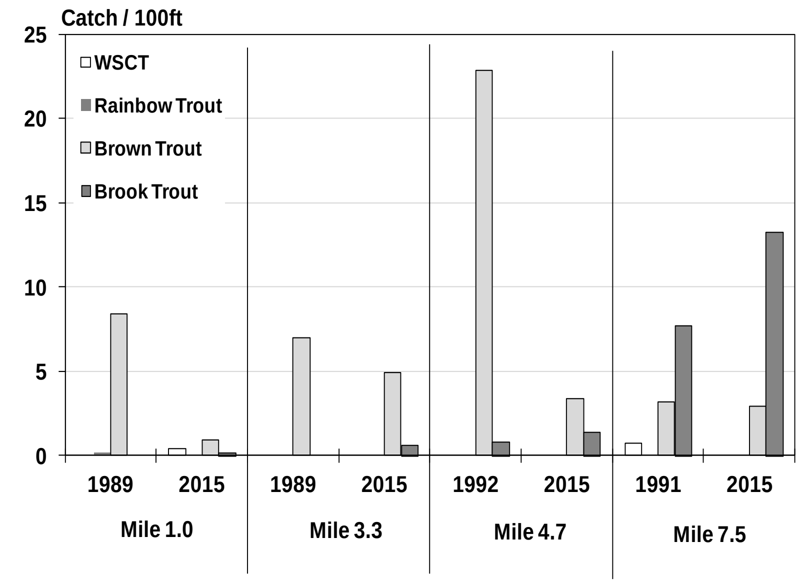


Figure 26. CPUE for age 1 and older trout at four locations in Cottonwood Creek.

<u>Fish populations</u>

The headwaters of Cottonwood Creek support non-hybridized westslope cutthroat trout, low numbers of brook and bull trout in very low and declining abundance. Cottonwood Lakes supports a small population of rainbow trout. Rainbow trout, brook trout and brown trout are prevalent in middle to lower stream reaches. Cottonwood Creek is designated as critical habitat for the recovery of bull trout under the ESA.

From 2013 through 2015, we continued annual long-term monitoring of fisheries in upper Cottonwood Creek at mile 12.0. We also resurveyed four sites (1.0, 3.3, 4.7 and

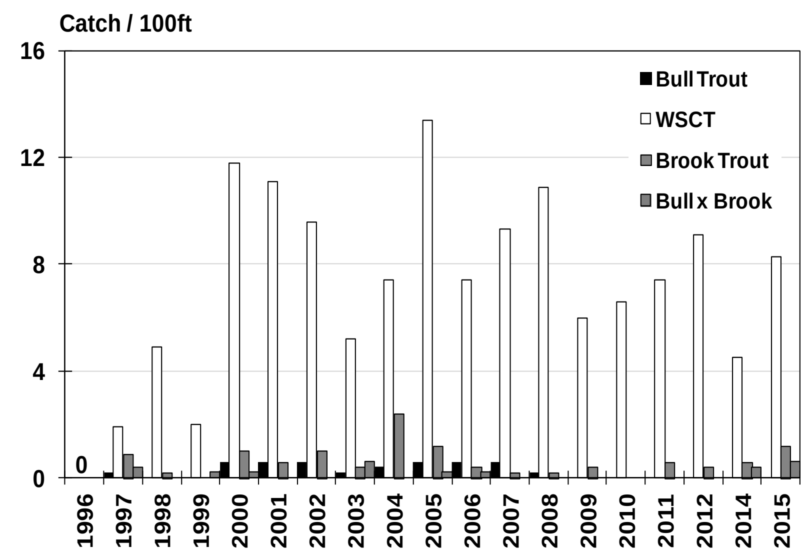


Figure 27. CPUE for age 1 and older trout in Cottonwood Creek at mile 12.0, 1996-2015. Diversion upgrades and instream flow enhancement occurred in 1997.

7.5) originally established in lower Cottonwood Creek between 1989 and 1991(Figure 26). Lastly, we collected genetic samples (fin clips) from bull trout (n=13) in upper Cottonwood Creek drainage for a genetic assignment study (Results Part IV).

The survey site at mile 12.0 shows an initial increase in westslope cutthroat trout following irrigation upgrades, followed by stable westslope cutthroat trout at increased levels (Figure 27). Though consistently present between 1997 and 2008, more recent surveys (2009-2015) failed to detect bull trout at mile 12.0 (Figure 27). Survey results for the four sites in lower Cottonwood Creek are shown in Figure 26.

Despite intensive sampling for bull trout in the headwaters, surveys indicate very low and declining numbers of bull trout. One survey revealed a CPUE of 0.5 for age 1 and older fish on North Fork Cottonwood Creek (Appendix A). The presence of bull trout x brook trout hybrids are consistently found in population surveys.

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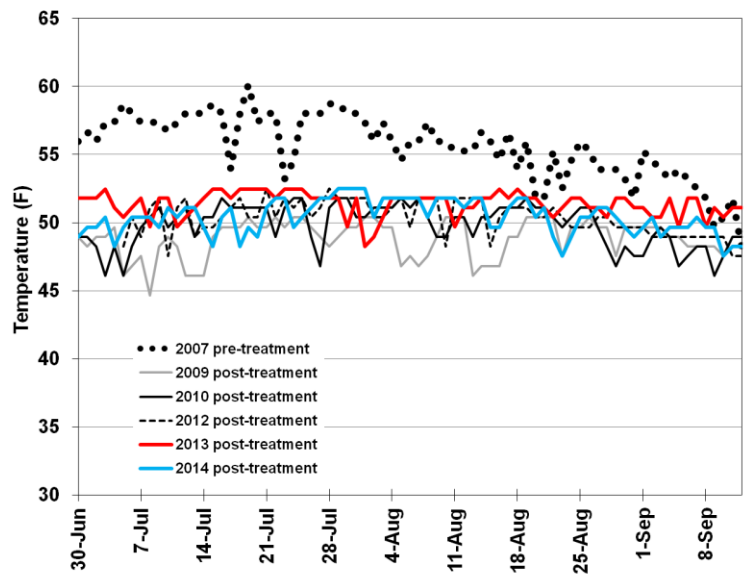


Figure 28. Summer maximum daily water temperatures for Enders Spring Creek at mile 0.1 pre restoration (2007) and post restoration (2009-2014).

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.

<u>Project summary</u>

Enders Spring Creek is a small 1st order spring creek tributary to the North Fork of the Blackfoot River entering at mile 6.3 with a discharge of <2 cfs. The stream is approximately 2.6 miles in length and flows entirely though a forested riparian area located on private land. Stream channel gradient averages 15ft/mile. 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.

<u>Fish populations and temperature monitoring</u>

Enders Spring Creek supports a trout community of 97% brook trout followed by bull trout and brown trout in low abundance. Pre- and post treatment fish and habitat assessments are summarized in prior monitoring reports (Pierce et al. 2008).

In 2013 and 2014, we continued to monitor post-treatment water temperatures near the confluence (mile 0.1) with the North Fork at a site established in 2007 prior to restoration. These data indicate a post-restoration cooling effect with maximum temperatures 6-8^{o}F lower than pre-treatment temperatures (Figure 28, Appendix E).

Gold Creek

Restoration objectives: Restore pool habitat and morphological complexity; restore thermal refugia for Blackfoot River native fish species. Reduce chronic and episodic road sediment sources and increase native fish habitat availability

Project summary

Gold Creek, a large 3rd order tributary, originates from Fly Lake in the upper Rattlesnake National Wilderness-Recreational Area. Gold Creek drains a 62.6 mi^2 watershed from the western slopes of Gold Creek Peak and Black Mountain and flows 19.9 miles east and southeast and enters the lower Blackfoot River at mile 13.5 with a baseflow of 20-25 cfs. The West Fork of Gold Creek, the largest tributary to Gold Creek, entering near mile 6.8. Stream gradients and stream types vary from 315ft/mile A2-A3 stream type in the upper reaches to 73 ft/mile B4-C4 stream type in the middle reaches increasing to 105ft/mile B3-C3 stream type in the lower reaches.

Approximately 66% of the Gold Creek watershed was managed as industrial forest (Plum Creek Timber Company) prior to 2014 when these lands were purchased by TNC (Figure 4). Lolo National Forest administers 29% of upper drainage, whereas the BLM, DNRC and private land, located in the lower drainage, comprise the remaining 5% of the drainage. Following the TNC acquisition, road inventories identified 330.5 miles of road in the entire watershed (all ownerships) with a road density of 5.3 mi/mi^2, which includes 5.4 miles of road within 50 feet of perennial streams (InRoads Consulting 2015; Amy Pearson, TNC, personal communication). There were also 126 culverts inventoried at stream crossings with defined channels. Road-related sediment issues include hazards associated with undersized culverts and a failing log culvert. These problems were exacerbated when much of the watershed burned in 2003 and was

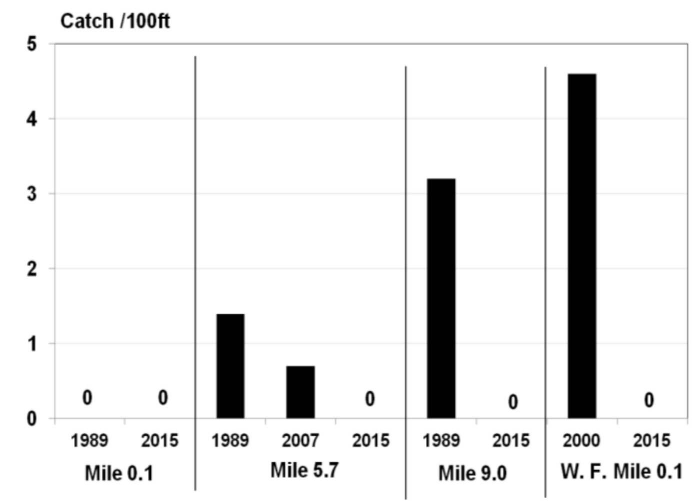


Figure 29. CPUE for age 0 and older bull trout at three sites on Gold Creek and one site on the lower West Fork of Gold Creek, 1989-2015

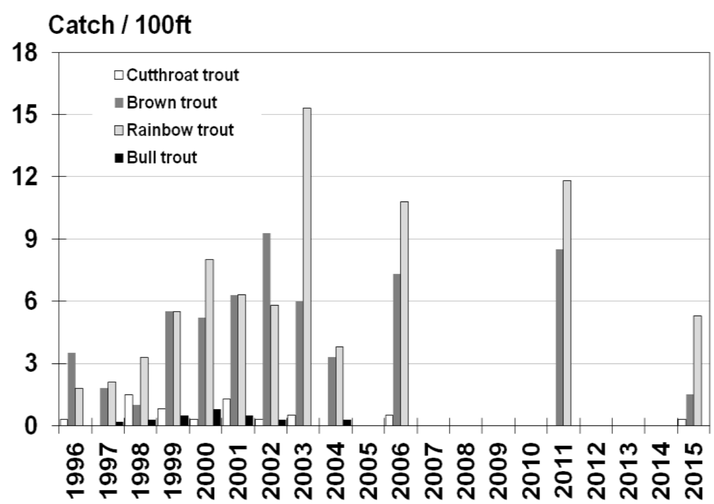


Figure 30. CPUE for age 1 and older salmonids in Gold Creek at mile 1.9, 1996 – 2015.

subsequently salvage-logged. There are ten culverts on perennial creeks in Gold Creek that were perched more than one foot that may impede fish passage (InRoads Consulting 2015).

Prior road restoration actions include removal of several culverts at road/stream crossings and mechanical ripping of some roadbeds. There were also a number of sediment reduction measures associated with logging roads including the installation of rolling dips, and seeding and closing roads after logging was completed.

Past harvest of riparian conifers combined with the actual removal of large wood from the channel has also reduced habitat complexity on the lower three miles of Gold Creek. The result of this fish habitat simplification was low abundance of age 1 and older fish. In 1996, a cooperative project installed 66 habitat structures made of native material (rock and wood) constructing 61 new pools in the three-mile section (Schmetterling and Pierce 1999).

Fish populations

Gold Creek is a spawning and rearing tributary to the lower Blackfoot River for westslope cutthroat trout, bull trout, rainbow trout and brown trout. Resident brook trout also inhabit the drainage.

In 2015, we resurveyed fisheries at five sites in the Gold Creek drainage, including four mainstem sites (mile 0.1, 1.9, 5.7, 9.0) and one site on the lower West Fork of Gold Creek (mile 0.1). Three of the four mainstem Gold Creek sites (mile 0.1, 5.7 and 9.0) were established in 1989, and one site (mile 1.9) was establish just prior to the 1996 habitat improvement project (Schmetterling and Pierce 1999). The West Fork site was established in 2000. None of the 2015 surveys detected bull trout where they were present in prior surveys (Figure 29). Fish population survey results for the 1996 habitat project are shown on Figure 30. Consistent with bull trout declines at all monitoring sites, redd counts conducted by Plum Creek Timber Company from 1998 through 2014 show a similar declining trend in bull trout (Table 1). Along with long-term declines in bull trout, the surveys show increasing numbers of nonnative trout in the upstream direction. All 2015 fisheries summary data is shown in Appendix A and B.

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 mile 108. Grantier Spring Creek was the first major spring creek restoration project undertaken in the Blackfoot River Basin.

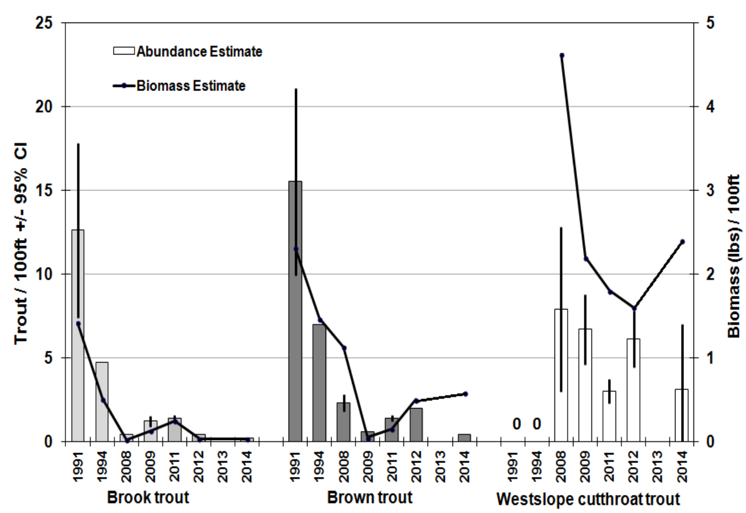


Figure 31. Estimates of trout abundance and biomass for age 1 and older fish in Grantier Spring Creek at mile 1.0, 1991-2014.

Grantier Spring Creek was reconstructed in 1990. In addition, vegetation was allowed to recover by reducing livestock pressure on streambanks.

Fish populations

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 resurveyed the mile 1.0 site annually between 2008 and 2014 and found westslope cutthroat trout are now prevalent (Figure 31). Subsequent surveys completed in 2011-2015 revealed westslope cutthroat trout spawning (redds) and age 0 westslope cutthroat trout within the upper spring creek. Post-treatment habitat survey and telemetry study results of adult westslope cutthroat trout are described in prior studies (Pierce et al. 2011, 2013).

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 (<60^0F); provide high quality pools with high level of complex cover; maximize use of existing channel belt-width and existing shoreline areas.

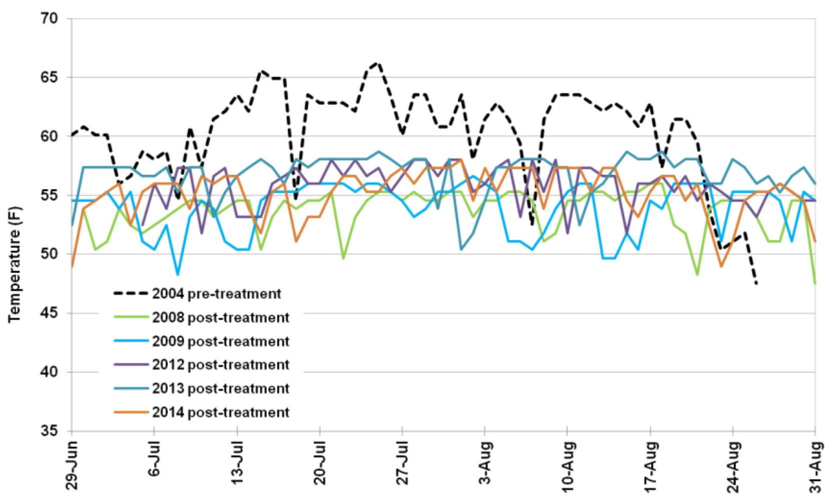


Figure 32. Maximum summer daily water temperatures for Jacobsen Spring Creek pre-treatment (2004) and post-treatment (2008-14).

Project summary

Jacobsen Spring Creek is a 2nd 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. Jacobsen Spring Creek was severely degraded from past grazing and timber harvest practices, which contributed to a wide, shallow channel with 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 and final work was completed in 2015. The project reestablished 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 (Pierce et al. 2008, 2013). The project also included shrub plantings and the adoption of improved livestock management in riparian areas. In total, 18,320 feet of restoration was completed including 1,100 feet of the West Fork of Jacobsen Spring Creek in 2015.

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

According to landowner accounts, Jacobsen Spring Creek historically supported both bull trout and westslope cutthroat trout. Jacobsen Spring Creek currently supports a mixed community of salmonids. Brook trout comprise >90% of the community followed by brown trout and the incidental presence of rainbow trout, bull trout and westslope cutthroat trout (Pierce et al. 2013). In 2013 and 2014, we continued water temperatures monitoring at mile 0.1. This monitoring shows a post-restoration cooling effect with maximum temperatures of >65^{o}F pre-treatment compared to <58^{o}F post-treatment (Figure 32, Appendix E).

Keep Cool Creek Drainage

Keep Cool Creek drains a large (54.7mi^2) 3rd order watershed situated in the northern Lincoln Valley. The watershed extends from Arrastra Creek basin divide near Black Mountain east to the Landers Fork divide and includes the southern slopes of Stonewall Mountain. Nine south-flowing basin-fed tributaries and one large west-flowing spring creek drain into Keep Cool Creek (Figure 33). With these inflows, Keep Cool Creek is the largest tributary to the Blackfoot River within the Lincoln Valley with

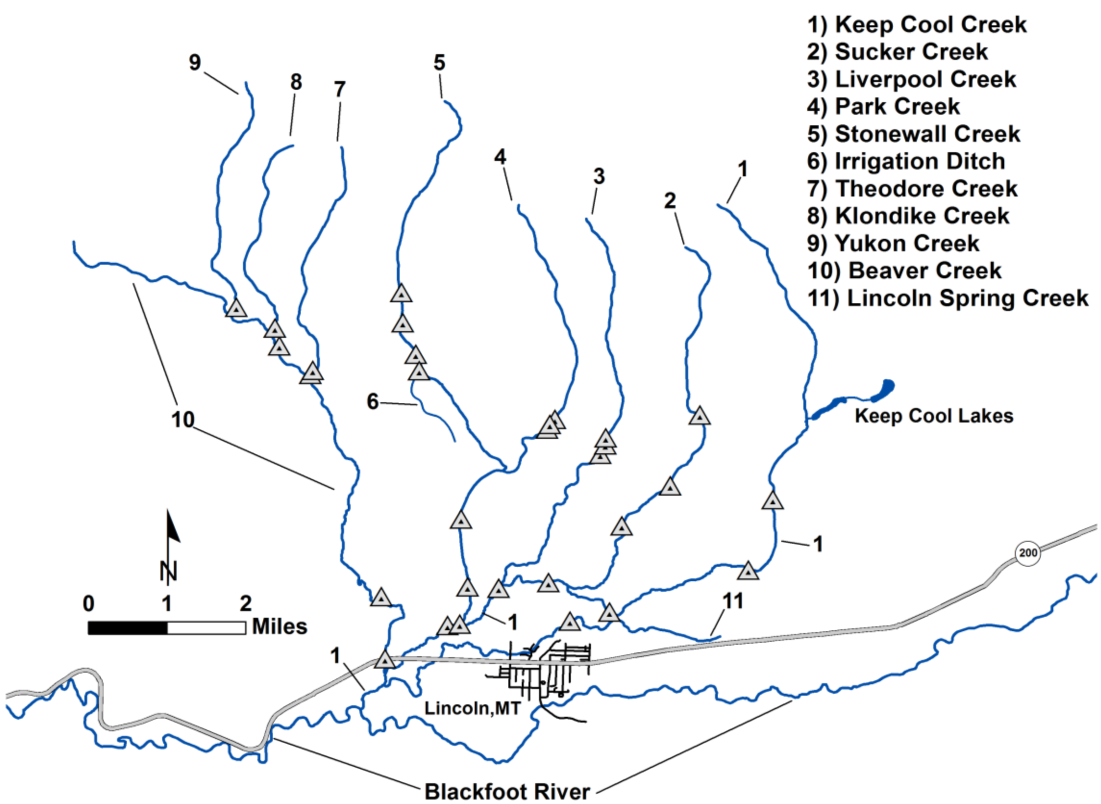


Figure 33. Location map of the Keep Cool drainage and 10 primary tributaries along with fish population survey sites conducted from 2011 through 2015.

a base flow estimated at 40 cfs. Keep Cool Creek enters the Blackfoot River about two miles west of Lincoln (Figure 33).

FWP conducted fish population surveys at 28 sites on all 10 primary streams (Keep Cool Creek, Sucker Creek, Liverpool Creek, Stonewall Creek, Park Creek, Beaver Creek, Theodore Creek, Klondike Creek, Yukon Creek and Lincoln Spring Creek) in the Keep Cool Creek drainage from 2011 through 2015 (Figure 33). This drainage-wide fisheries investigation was completed to assist with local restoration planning.

Keep Cool Creek

The mainstem of Keep Cool Creek flows 14.8 miles southwest to its junction with the Blackfoot River at mile 105.2. Its headwaters drain the H-LC National Forest, the north slope of Black Mountain and eastern slopes of Stonewall Mountain, and the outlet channel from Keep Cool Lakes near mile 9.6 (Figure 34). Stream gradient decreases from an average 183ft / mile in its headwaters on H-LC National Forest to 80 ft/ mile where it enters private ranchland near mile 7.4 to 17ft/mile near the mouth. The three largest tributaries to Keep Cool Creek are Stonewall Creek, Beaver Creek and Lincoln Spring Creek; all of which enter lower Keep Cool Creek downstream of mile 2.2 (Figure 34).

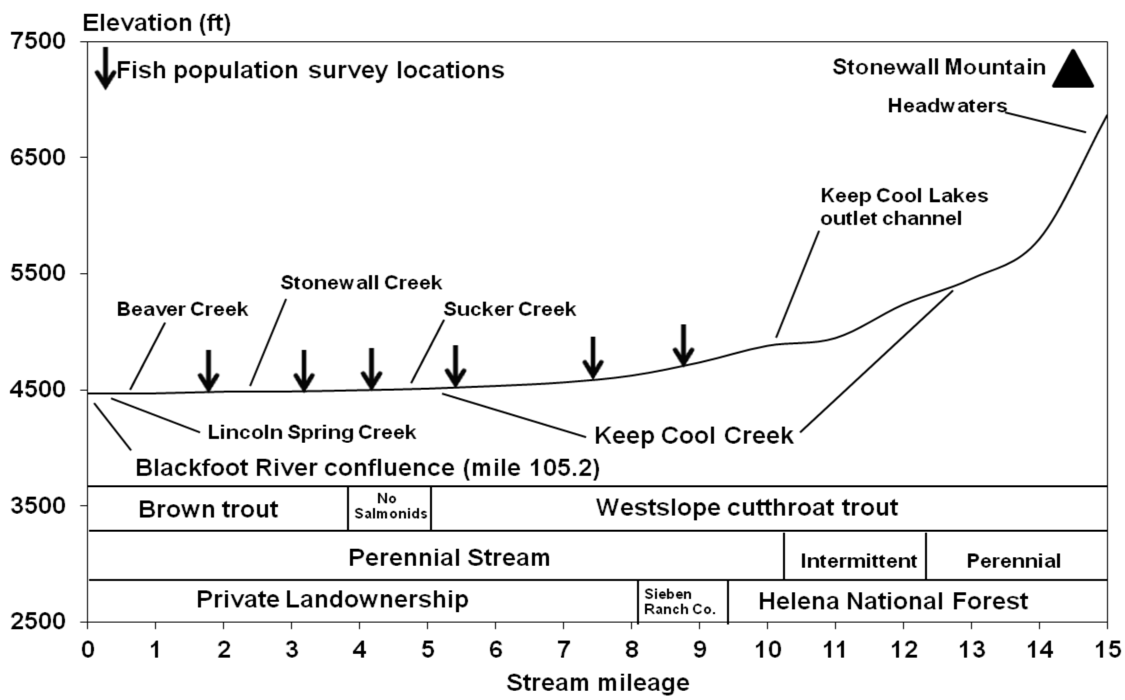


Figure 34. Longitudinal profile of Keep Cool Creek with fish population survey sites, the names and locations of direct tributaries, general trout species distribution and landownership.

While on the H-LC National Forest, upper Keep Cool Creek occupies B3-B4 stream types with an overstory of lodgepole pine, aspen and black cottonwood, a dense riparian understory of shrubs and sedge-lined streambanks. Here, instream wood provide habitat structure that form deep scour pools and complex trout habitat. Once Keep Cool

Creek enters private land near mile 7.4, Keep Cool Creek attenuates to a C4-E4 stream types. At this junction, the stream loses water to at least six unscreened irrigation diversion and then gradually gains flow in the downstream direction from groundwater and tributary inflows.

In addition to the effects of irrigation, other fisheries impairments vary by location and include channelization, dewatering and excessive streamside grazing pressure; whereas, other portions of stream and riparian areas are managed for riparian health. Degraded reaches are typically wide and shallow and lack shrub and bank cover when compared to deeper, narrower and more sinuous and vegetated channels where managed for riparian health. Though predominately a gravel-bed stream, high level of fine sediment are present in the mid to lower reaches of the stream. Beaver variously occupy segments of Keep Cool Creek.

To begin to offset fisheries impairments on the mainstem Keep Cool Creek, several restoration actions have been completed. These include upgrades of three undersized culverts, water conservation and reductions in riparian grazing pressure between mile 2.6 and 5.0.

Fish populations

Past telemetry studies in the upper Blackfoot River found incidental bull trout use in the lower-most reach of Keep Cool Creek, but no evidence of migratory westslope cutthroat trout use of the drainage (Pierce et al. 2004, 2007).

In 2014, FWP surveyed fish populations at six locations (miles 1.8, 3.3, 4.3, 5.5, 7.7 and 8.9) in Keep Cool Creek drainage (Figure 35). Sampling found trout in low abundance with westslope cutthroat trout prevalent in the headwaters and brown trout prevalent in lower reaches. Brook trout were variably present in upper and lower stream reaches. A fish population survey in the middle reach Keep Cool Creek (mile 4.3) failed to detect trout in a livestock-degraded segment of channel. This section of channel was fenced in 2015 and a set of undersized culverts at a road crossing were replaced with a bridge to help improve riparian habitat. The mile 1.8 site, just downstream of Stonewall Creek confluence, was originally surveyed in 2004 prior to passive restoration. The 2014 resurvey at this site found low numbers of westslope cutthroat trout where none were detected in 2004.

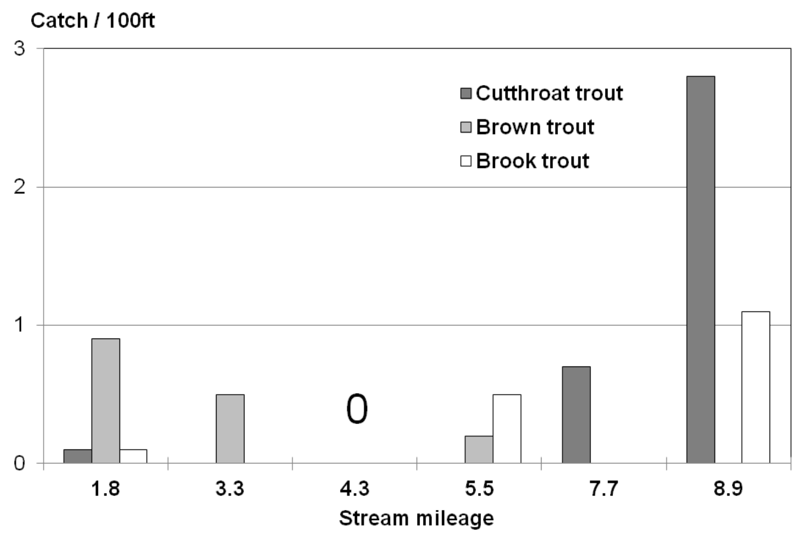


Figure 35. CPUE for trout at six locations on Keep Cool Creek, 2014.

During stream surveys, a westslope cutthroat trout genetic sample (*n*=11) was collected at miles 7.7 and 8.9. Ten fish tested as non-hybridized westslope cutthroat trout; however, rainbow trout alleles were detected in one fish (Appendix F).

Sucker Creek

Sucker Creek, a 1st order tributary to upper Keep Cool Creek, drains the southeastern slopes of Stonewall Mountain, and flows southerly 6.8 miles to its confluence with Keep Cool Creek at mile 4.2. Sucker Creek enters DNRC land at mile 2.8 and private ranchland at mile 2.4. Average stream gradients range from 492ft/mile in the headwaters to 57ft/mile near the mouth. The riparian vegetation in the headwaters supports a dense conifer overstory of Douglas fir, ponderosa pine and Englemann spruce that contribute instream wood to a stable Rosgen B3-B4 stream type. One perched culvert on the National Forest at mile 3.9 that inhibits fish passage barrier is scheduled to be upgraded in 2016. The middle reaches support the same coniferous overstory, though it is relatively more open from past timber harvest, which allows for the denser understory of alders and sedges. Once Sucker Creek enters agricultural ranchlands, the channel attenuates to an E4 stream type. Here, fisheries impairment includes channelization, excessive grazing pressure and numerous perched culverts as well as irrigation dewatering from at least one unscreened diversion.

Fish populations

Fish population surveys were conducted by FWP at three sites (miles 1.6, 2.6 and 3.8) in the Sucker Creek in 2014 (Figure 36). Survey results found westslope cutthroat trout abundance decreased in the downstream direction, which included the absence of trout at mile 1.6. Westslope cutthroat trout genetic samples (n=9) taken at miles 2.6 and 3.8 found no evidence of hybridization (Appendix F).

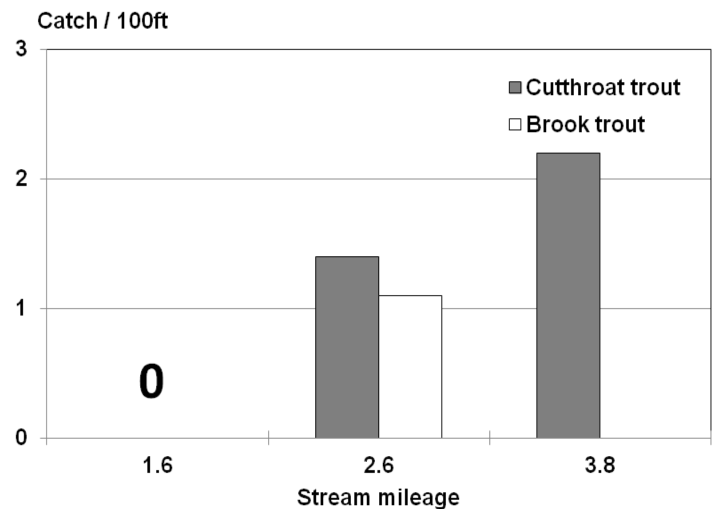


Figure 36. CPUE for age 1 and older trout at three sites on Sucker Creek, 2014.

Liverpool Creek

Liverpool Creek, a 1st order basin-fed stream, drains a small (4.1 mile^2) watershed and flows from Stonewall Mountain and joins upper Keep Cool Creek at mile 3.1 through an intermittent channel. The stream is 6.6 miles in length and begins on the H-LC National Forest where it flows through a B4 stream type. Stream gradients decrease from about 514 feet/mile in the headwaters to about 64 feet/mile near the mouth. The stream exits the National Forest land at mile 3.6 and enters mixed DNRC, TNC and private ranch land.

The riparian vegetation consists of a mix of conifers and dense shrub understory. Once Liverpool Creek leaves the mountains, a majority of stream flow was diverted for irrigation when inventoried in 2011. From 2011 through 2014, the private landowner has been working with Trout Unlimited to improve irrigation practices and fisheries by consolidating two irrigation ditches into one screened diversion, replacing an undersized culvert with a bridge and water conservation actions to provide more natural channel function. The DNRC has also upgraded one undersized downstream culvert with a larger inset culvert to improve fish passage and channel function.

Fish populations

Liverpool Creek supports non hybridized westslope cutthroat trout according to USFS genetic tests completed in 1988. FWP surveyed fisheries in Liverpool Creek for the first time in 2011 prior to restoration. Our surveys found resident westslope cutthroat trout and no other fish species. Two surveys were completed up- and downstream of two diversions (miles 2.7 and 3.0) and a third survey was completed in an un-screened irrigation ditch located at mile 2.8 (Figure 37). The survey upstream of the diversion recorded a CPUE of 11.7 compared to 4.7 downstream of the diversions, and a CPUE of 5.4 in the irrigation ditch.

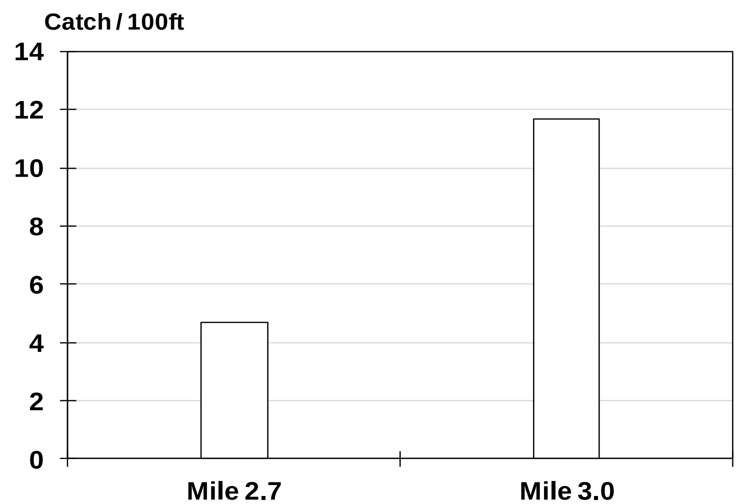


Figure 37. CPUE for age 1 and older westslope cutthroat trout in Liverpool Creek at miles 2.7 and 3.0, 2011.

Stonewall Creek

Stonewall Creek, a 2nd order tributary, drains an 11.3 mile^2 watershed and enters Keep Cool Creek at mile 2.2. From the western slopes of Stonewall Mountain, Stonewall Creek flows south 7.6 miles through a mix of public (H-LC National Forest and DNRC) and private lands. Park Creek, the primary tributary to upper Stonewall Creek, enters at mile 2.4 though an intermittent channel and beaver complex. Stonewall Creek channel ranges from B2/B3 stream types in the headwaters to E4 channel/wetland bog near the mouth with gradients that range from 600ft/mile in the headwaters to 48ft/mile in the lower mile of stream (Figure 38).

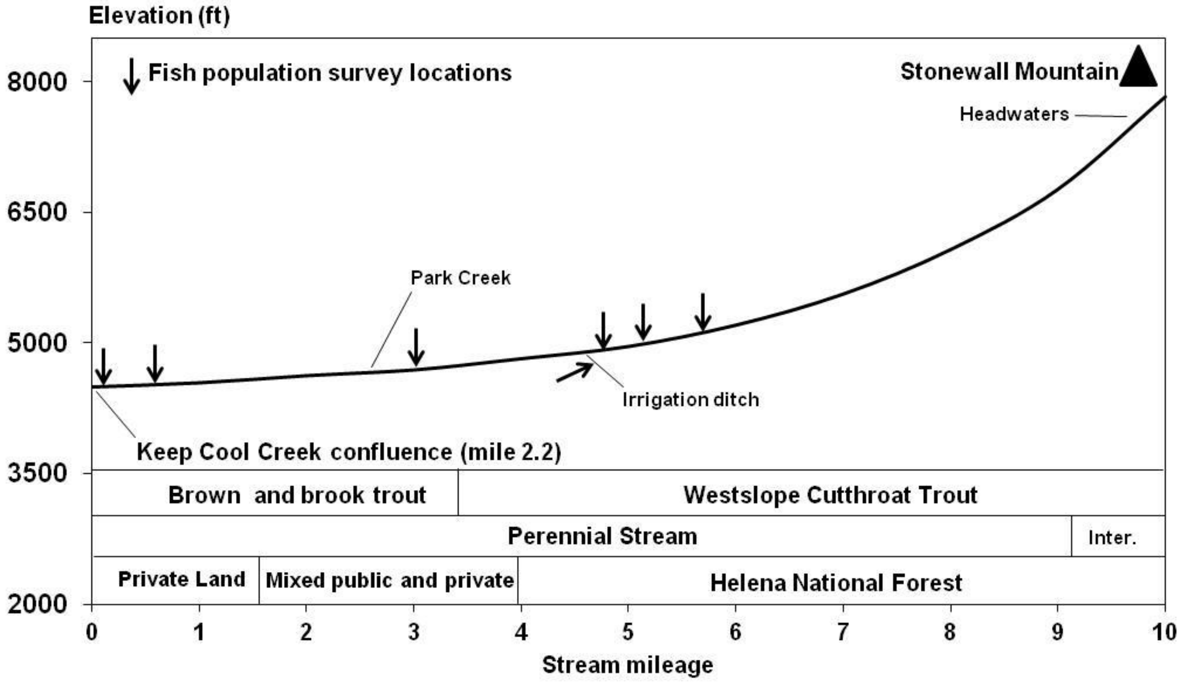


Figure 38. Profile of Stonewall Creek along with fish population survey sites, distribution of prevalent trout and landownership.

The Stonewall riparian area on the National Forest has been highly altered near mile 5.7 from the past placement of massive piles of placer mining spoil. Near the confluence with Park Creek, Stonewall Creek flow through a beaver complex from about mile 2.7 to 1.1. There is one large irrigation diversion at mile 4.3, plus five small diversions including one at mile 1.0 that directs water to Smith Lake. The lower mile of Stonewall Creek variables flows within channelized stream segments including captured irrigation ditches. This reach has been managed for riparian health since 2004.

The riparian vegetation in the upper reaches on the H-LC National Forest consists of rocky mountain maple, alder, red osier dogwood, willow and snowberry mixed with grasses beneath a conifer overstory of ponderosa pine Douglas fir and black cottonwood. Habitat features in upper reaches include plunge pools and pocket pools created by the boulder/cobble substrate. In the area of mine waste, the stream loses complexity as it attenuates to a riffle-dominated channel from mile 5.7-6.5 due in part to the loss of large wood from the riparian area. A reach near the forest boundary is seasonally intermittent. Downstream of the beaver complex near mile 1.0, overhanging willows and sedges above undercut banks generally provide high quality trout fish habitat, though elevated levels of fine sediment are present near the mouth.

In 2015, the H-LC National Forest and BBCTU developed a project to increase habitat complexity and improve pool quality on 4,200 feet of stream by 1) removing 35,000 yards of mine waste rock from the floodplain and riparian area, 2) actively reconstructing pools and adding instream wood to the channel, and 4) allowing the recovery of native riparian vegetation. Following the identification of entrained westslope cutthroat trout in the irrigation ditch at mile 4.6, the diversion was upgraded in 2015 to facilitate fish passage, and the ditch was screened to prevent fish losses.

Fish populations

Stonewall Creek supports non hybridized westslope cutthroat trout, along with brown trout and brook trout in lower reaches. In 2014, we surveyed fish populations at six sites (miles 0.1, 0.65, 3.0, 4.7, 5.2 and 5.7) on the mainstem of Stonewall Creek, plus one site in the unscreened irrigation ditch at mile 4.3. Four of the six, survey sites were originally surveyed in 2004 prior to improved riparian grazing strategies in the lower mile of stream.

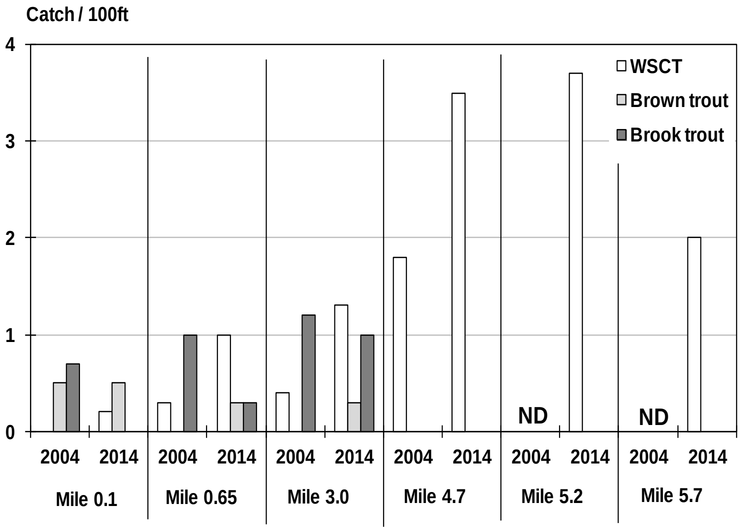


Figure 39. CPUE for age 1 and older trout at six locations on Stonewall Creek in 2014 and a comparison to four monitoring sites in 2004.

The 2014 surveys identified westslope cutthroat trout as the only salmonid in the headwaters with abundance decreasing downstream of mile 4.7 (Figure 39). Tailed frogs were also common in headwater surveys, which indicate high water quality and cold water. The mile 5.7 site was completed adjacent to the mine waste in a reach characterized by long riffles and low habitat complexity. The mile 5.2 and mile 4.7 sites were surveyed in reference sites with higher habitat complexity (Figure 39). Compared to 2004 survey results, the 2014 surveys revealed higher westslope cutthroat trout abundance in all four lower sample locations. Brown trout were present at miles 0.65 and 3.0 in low abundance where absent in 2004. In 2014, brook trout absent at mile 0.1 where they were prevalent in 2004 (Figure 39). A survey in the un-screened irrigation ditch at mile 4.3 recorded a CPUE of 2.1 for westslope cutthroat trout (Appendix A). Additional genetic samples collected from westslope cutthroat trout during the 2014 surveys are pending.

Park Creek

Park Creek, a 1st order stream, originates the southern slope of Stonewall Mountain and flows 2.5 miles south through H-LC National Forest, 1.2 miles through TNC land then another 0.3 miles within a beaver complex on DNRC land to its confluence with Stonewall Creek at mile 2.4. Stream gradient varies from 660 ft/mile near the headwaters to 55ft/mile on the lower mile of stream. The riparian area in the headwaters supports a dense conifer forest. Once Park Creek leaves the mountains, past timber harvest practices and recreational road use on DNRC land contribute to areas of stream degradation, a lack of instream wood, and low habitat complexity. All of the Park Creek baseflow is diverted for irrigation at mile 1.4 at an unscreened diversion with no fish passage consideration. One primary road crossing was upgraded on DNRC land in 2014. Similar to Liverpool Creek, the lower portion of Park may be naturally intermittent.

Fish populations

FWP surveyed fisheries in Park Creek for the first time in 2011. Westslope cutthroat trout were the only fish present in surveys. Surveys were completed up- and downstream (miles 1.1 and 1.4) of an irrigation diversion, plus on site within the irrigation ditch near mile 1.4. Upstream of the irrigation ditch, we recorded a CPUE of 6.1, compared to 2.1 within the ditch, verses none in Park Creek downstream of the diversion. Westslope cutthroat trout genetic samples (*n*=10) collected by the USFS in 1988 tested as non-hybridized.

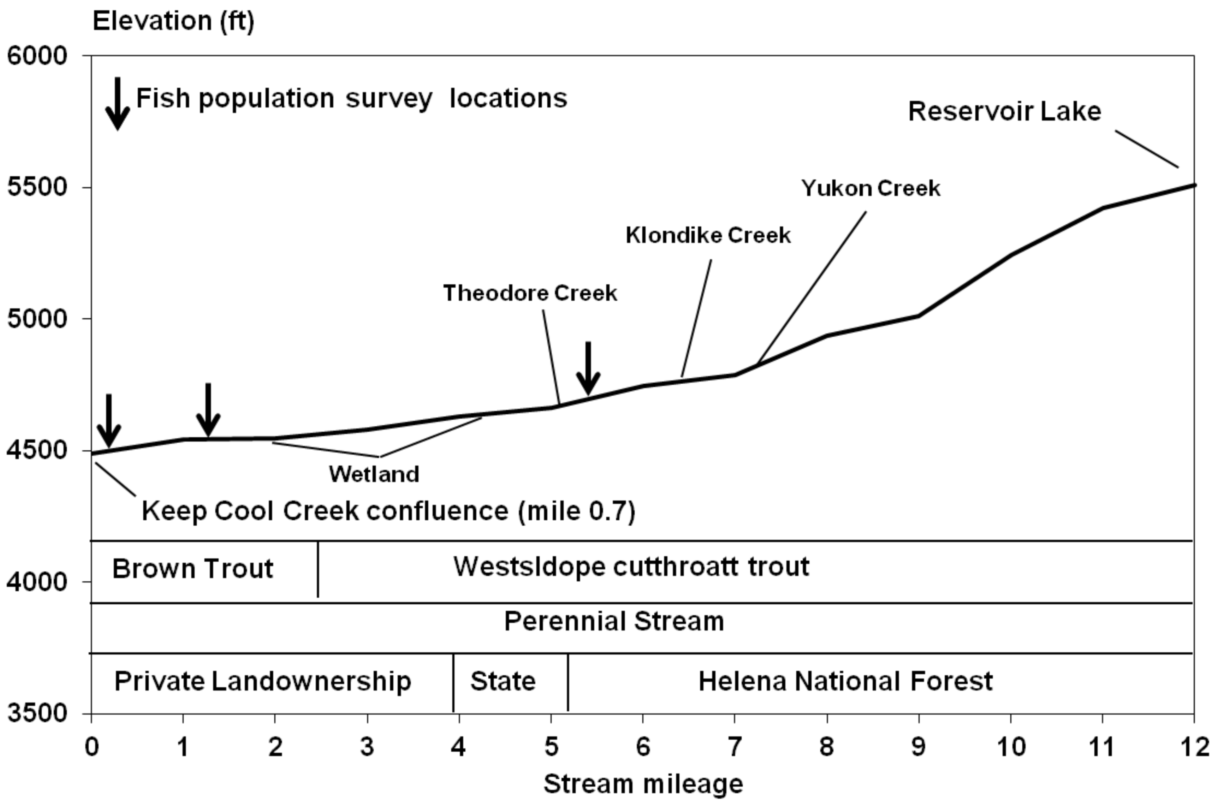


Figure 40. Longitudinal profile of Beaver Creek with sampling sites, tributary locations, general species composition and land ownership.

Beaver Creek

Beaver Creek, a 2nd order tributary, drains an 18.1mi^2 watershed and enters Keep Cool Creek at stream mile 0.7. Mainstem Beaver Creek originates at Reservoir Lake (mile 10.3) on the H-LC National Forest on the northern side of Black Mountain. It gains flow from Theodore, Klondike and Yukon Creeks and enters DNRC land at mile 5.3 and private agricultural land at mile 4.1 before entering small residential ownerships in the lower basin (Figure 40).

Upper Beaver Creek flows with a stable, boulder and cobble-dominated Rosgen B3-B4 stream below a heavily forested overstory and dense understory before entering agricultural ranchland downstream. Beaver Creek has losing reach downstream of Theodore Creek where the headwaters meet the main Lincoln Valley. Here, the stream attenuates to C4 and E4 stream types and enters a large wetland complex with willow and sedge vegetation. There are seven diversions (six active) serving irrigation on private land. There are also areas of trampled and sloughing banks that contribute to streambank degradation and elevated levels of fine instream sediment. Near the mouth, Beaver Creek supports had a mixed cottonwood and conifer overstory above an understory of willow, alder and other shrubs. This vegetation provides stable channel with undercut streambanks, deep pools and substrates that consist of cobble, gravel with areas of elevated fine sediment.

Fish populations

Beaver Creek supports non hybridized westslope cutthroat trout, brown trout, and brook trout and once supported a run of bull trout (Leo Fleming, personal communication). However, our surveys failed to detect bull trout in any Beaver Creek surveys or any other surveys in the Keep Cool drainage.

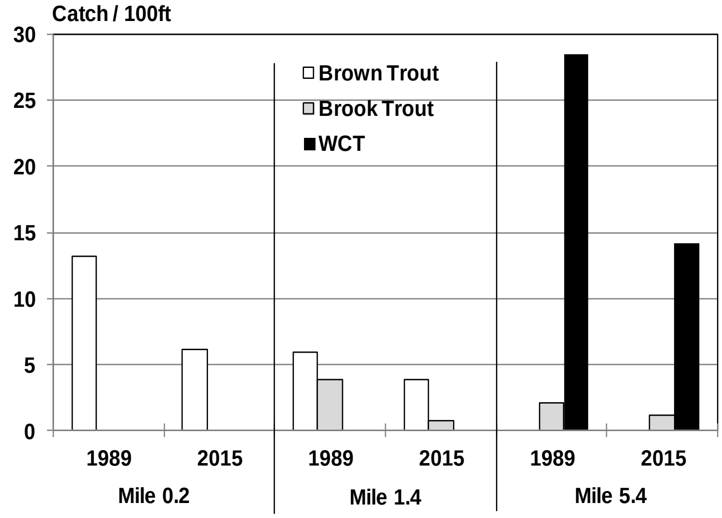


Figure 41. CPUE for age 1 and older trout at three locations on Beaver Creek, 1989 and 2015.

In 2015, we resurveyed Beaver Creek in 2015 at three sites (miles 0.2, 1.4 and 5.4) originally established in 1989 (Peters 1990). Results from the 2015 surveys were very similar to those reported in original surveys (Figure 41). Similar to 1989, brown trout were prevalent at the two lower survey sites (mile 0.2 and 1.4), and westslope cutthroat trout were prevalent in the upper (mile 5.4) sample. Trout species composition was similar though abundance was proportionally lower at all sites in 2015 compared to 1989. Similar to 1989, the CPUE was higher in the headwaters and lowest in the middle reach.

Theodore, Klondike and Yukon Creeks

Theodore, Klondike and Yukon Creeks are all small, 1st order tributaries to upper Beaver Creek. They are located entirely on the H-LC National Forest, and all drain the southern flanks of un-named mountain peaks between Arrastra Mountain to the west and Stonewall Mountain to the east. These small tributary streams range from 2.9-4.0 miles in length, flow within B3-B4 channels with gradients ranging from 215-351 ft/mile near their junctions with Beaver Creek. Theodore, Klondike and Yukon Creeks enter upper Beaver Creek at miles 5.3, 6.1, and 6.8, respectively (Figure 40). Riparian areas on all three tributaries are heavily forested with a conifer overstory above an alder understory. The vegetation generates stable channels with excellent shade and recruitment of large wood to the streams. All surveys were associated with road crossings on FS road #4106. The three road crossings were all upgraded in 2013-2015. Bridges replace undersized culverts on Theodore and Klondike Creek, while a bottomless arch was installed to provide fish passage on Yukon Creek.

Fish populations

FWP surveyed fish populations on the lower segments of Theodore (mile 0.1 and 0.15), Klondike (mile 0.1) and Yukon Creeks (mile 0.1) for the first time in 2014. Surveys focused on baseline species composition, CPUE and westslope cutthroat trout genetic samples. Westslope cutthroat trout were present in low to moderate numbers in all three tributaries (CPUE range 3.8-8.4, Appendix A). Brook trout were also found in low abundance in Theodore Creek. Genetic testing results (*n*=10 in Klondike, *n*=9 in Yukon, *n*=10 in Theodore) found non-hybridized westslope cutthroat trout in all three tributaries (Appendix F). However, allele characteristics also identified possible rainbow trout hybridization in one fish in Theodore Creek.

Lincoln Spring Creek

Lincoln Spring Creek is a large, low-gradient, 1st order spring creek that originates from the alluvial aquifer underlying the Lincoln Valley. The stream is 5.3 miles in length and flows west entirely through private land. Lincoln Spring Creek enters the town of Lincoln at mile 3.4. At mile 2.9, the spring creek splits into two separate channels. The south channel continues to flow through residential neighborhoods and exits the town at mile 2.1. The north channel flows through willow-dominated wetlands. The two channels rejoin at mile 1.0 before entering Keep Cool Creek at mile 0.5. Spring creek flows tend to seasonally rise and fall with the underlying aquifer and the influence of an irrigation diversion off the Blackfoot River. The upper portion of the spring creek is intermittent upstream of mile 4.5 from fall into early spring; whereas, the lower portion of the spring creek continuously gains water and maintains perennial flow downstream of mile 4.5.

Lincoln Spring Creek was reconstructed from mile 5.3 to 3.4 to a more natural narrow and deeper, gravel-based channel with increased stream sinuosity in 2008. This project, located upstream of the town of Lincoln, included the placement of in-stream wood, re-vegetation of stream banks, removal of three undersized culverts and one irrigation diversion upgrade. The project improves physical habitat by restoring natural channel form, enhancing habitat complexity with wood, reducing water temperature and sediment levels, while reestablishing movement corridors and improving water quality.

Within the town of Lincoln, fisheries-related impairments relate to residential developments and so include channel alterations, undersized culverts, artificial grade control (rock dams) and the removal of woody riparian vegetation, all of which contribute to a

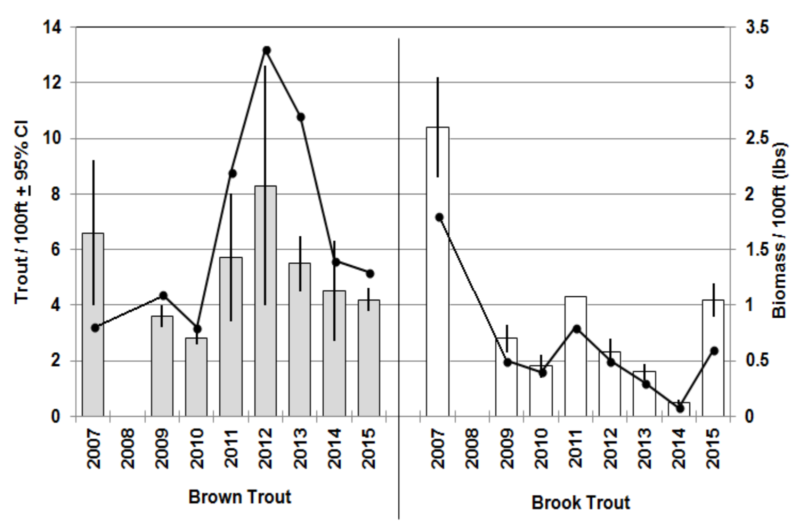


Figure 42. Estimates of abundance and biomass for age 1 and older brown and brook trout in Lincoln Spring Creek at mile 3.8, 2007-2015.

wide, shallow channel, fine sediment loading and generally low quality trout habitat.

Fish populations

Lincoln Spring Creek supports brown and brook trout and the incidental presence of westslope cutthroat trout. 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 2015 (Figure 42). The surveys show brown trout biomass increasing until 2012 followed by a biomass decrease in recent years. The recent declines may be flow/drought-related.

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.

Project summary

Monture Creek, a 4th order stream, drains a 152.1 mile^2 basin and flows south 29.5 miles before entering the Blackfoot River at river mile 45.9 with a baseflow of 30-40 cfs. The headwaters drain the southern slopes of Monahan, Foolhen and Youngs Mountains. Major tributaries include Dunham Creek entering at mile 11.5 and Dick Creek entering at mile 4.2. Gradients in the upper reaches of the mainstem average 420ft/mile and attenuate to 22ft/mile near the mouth. Monture Falls is located at mile 25, which also delineates the upper distribution of bull trout (Pierce et al. 2008). However, genetically pure westslope cutthroat trout reside upstream of the falls. There is an intermittent reach of Monture Creek between mile 14.5 and 13.5. The majority of the drainage (73%), including the headwaters, consist Lolo National Forest land; whereas, the remaining 16% of the lower basin consists of private ranchland along with 11% TNC and DNRC land.

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 (Pierce et al. 1997;

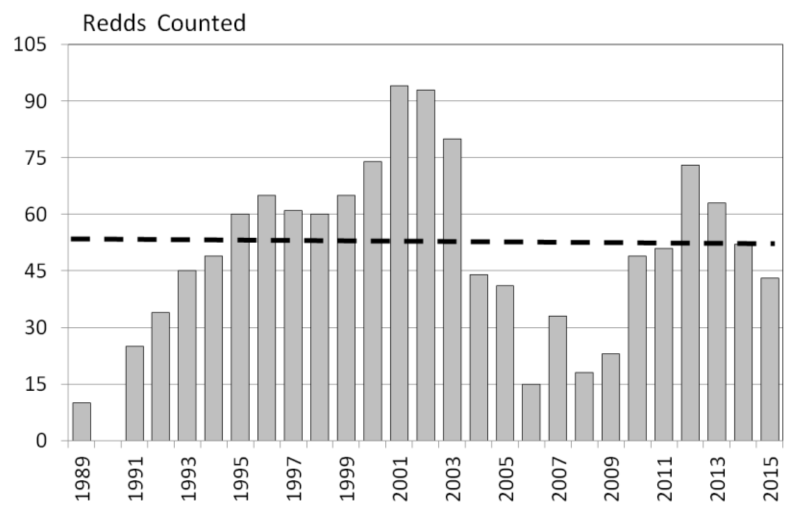


Figure 43. Bull trout redd counts for Monture Creek index reach, 1989-2015. The dashed line is the long-term mean.

Pierce et al. 2001). Despite many improvements, excessive livestock access continues to degrade certain riparian areas of Monture Creek.

Fish populations

Monture Creek 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 and rearing 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 (Pierce et al. 2009). Brook trout are absent upstream of an intermittent reach at mile 14 but are found in lower Monture Creek and its adjoining tributaries downstream of the intermittent reach (Pierce et al. 2008). Dunham Creek is the largest tributary to Monture Creek and like Monture Creek provides spawning and rearing for fluvial bull trout and fluvial westslope cutthroat trout and rainbow trout (Swanberg 1997; Schmetterling 2001; Pierce et al. 2009).

Monitoring for the 2013-2015 period included: 1) continued bull trout redd counts, 2) water temperature monitoring (mile 1.8) and 3) a genetic assignment study for bull trout (Results Part IV)

Bull trout redd counts were upward trending between 1989 and 2003, then declined sharply during a period of protracted drought (2004-2009), before increasing between 2010 and 2012 (Figure 43). Water temperature monitoring, began in 1994 at mile 1.8, continued through 2015 (Figure 19, Appendix F). The bull trout genetic assignment study shows a distinct stock of bull trout in the Monture drainage and the presence of Monture Creek bull trout in the Blackfoot River. In 2015, genetic assignment tests in Clearwater River drainage found Monture bull trout in Salmon Lake (Results Part IV)

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.

Project summary

Murphy Spring Creek, a small 1st order tributary, drains a 4.4 mile^2 basin and flows south 6.7 miles before entering the North Fork of the Blackfoot River at mile 9.9, with a baseflow of 2-3 cfs. The stream originates on Lolo National Forest land on the northeast side of Ovando Mountain, and then enters DNRC land at mile 2.3, before entering private land near mile 1.0. Stream gradients range from 749ft/mile near the headwaters to 91ft/mile the lower mile of stream.

Prior to restoration, Murphy Spring Creek had a history of dewatering from irrigation 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 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 880ft of stream.

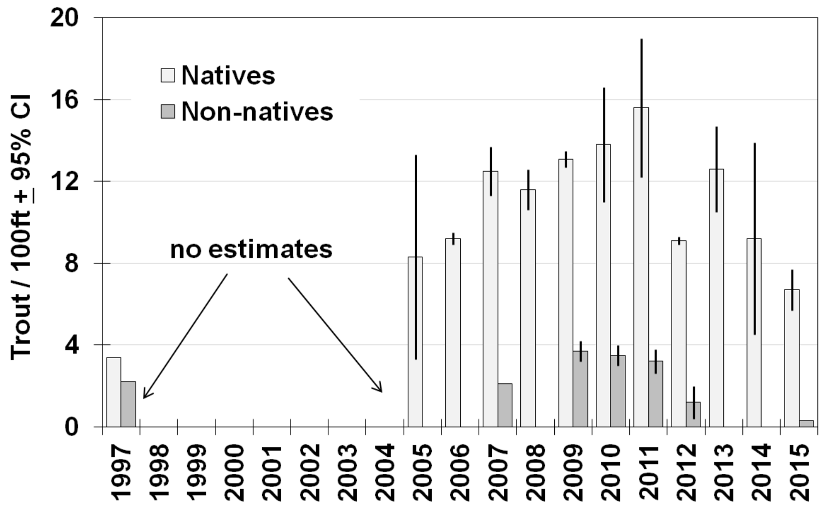


Figure 44. Estimates of abundance for age 1 and older native and nonnative trout in Murphy Spring Creek at mile 0.6, 1997-2015.

Fish populations

Murphy Spring Creek supports primarily westslope cutthroat trout and low numbers of bull trout and brook trout. Prior to restoration, we established a fish population monitoring site at mile 0.6. Post-restoration fish population surveys between 2005 and 2015 show a 7 year increasing trend, followed by a four year decline (Figure 44, Appendix A and 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, a 4th order stream, drains a 355 mile^2 watershed and flows 53 miles west-northwest from the Continental Divide north of Nevada Mountain and enters the Blackfoot River at river mile 67.8. At mile 45.6, Nevada Creek leaves the H-LC National Forest enters private ranchlands then flows another 11.9 miles where it enters Nevada Creek Reservoir at mile 33.7. The reservoir is managed primarily for irrigation water storage. Classified as a Rosgen B3-C4 stream types, gradients on upper Nevada Creek range from 320ft/mile at the headwaters to 53ft/mile immediately upstream of the reservoir. Downstream of the reservoir, the lower 31.7 miles of Nevada Creek flows through primarily private ranchland in a channel that ranges from Rosgen C3-E6 stream types. Stream gradients range from 40ft/mile below the reservoir decreasing to <10ft/mile near the mouth. Major tributaries include Nevada Spring Creek entering at mile 6.3 and Douglas Creek at mile 5.1.

Downstream of the National Forest, Nevada Creek is intensively managed for irrigated hay and livestock production. Nevada Creek is a 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. Downstream of the Reservoir, there are two large unscreened canals (mile 28.5 and 25.7) and several unscreened smaller ditches that divert a majority of the baseflows.

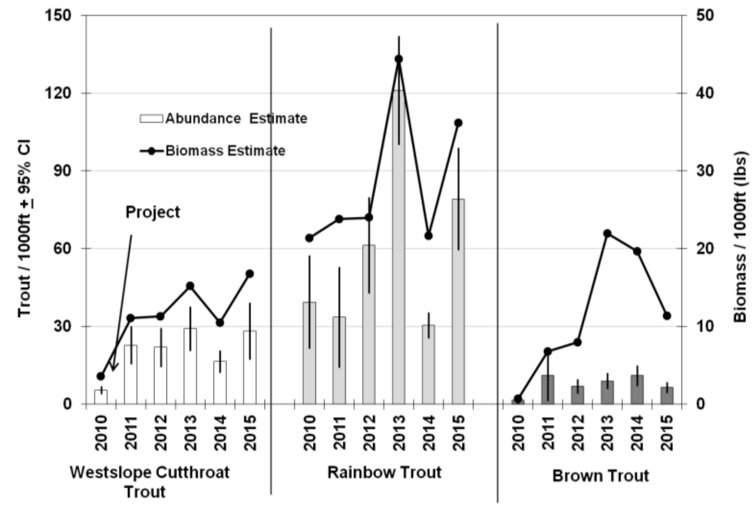


Figure 45. Estimates of abundance and biomass for age 1 and older trout in Nevada Creek at mile 29, 2010-2015.

Immediately downstream of Nevada Reservoir, a stream restoration project was completed on ~4,400feet of channel in 2010 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. In lower Nevada Creek a 3,200feet streambank restoration project was completed between the junction of Nevada Spring Creek and Douglas Creek (mile 5.1-6.3). This project reestablished a vegetated bankfull bench in a reach with highly erosive and vertical streambanks.

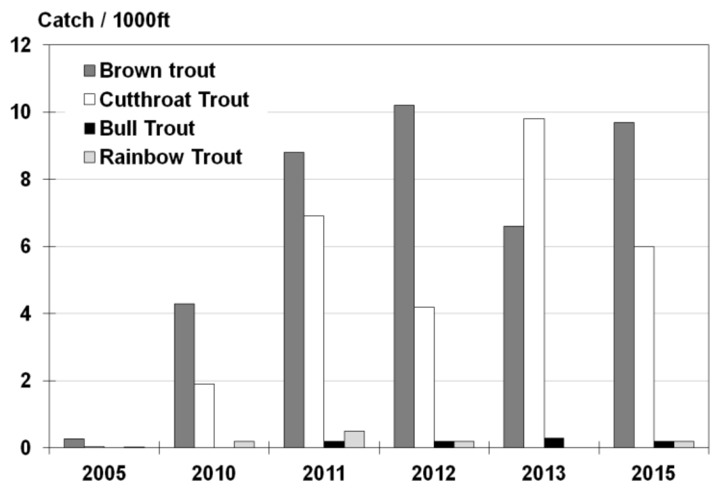
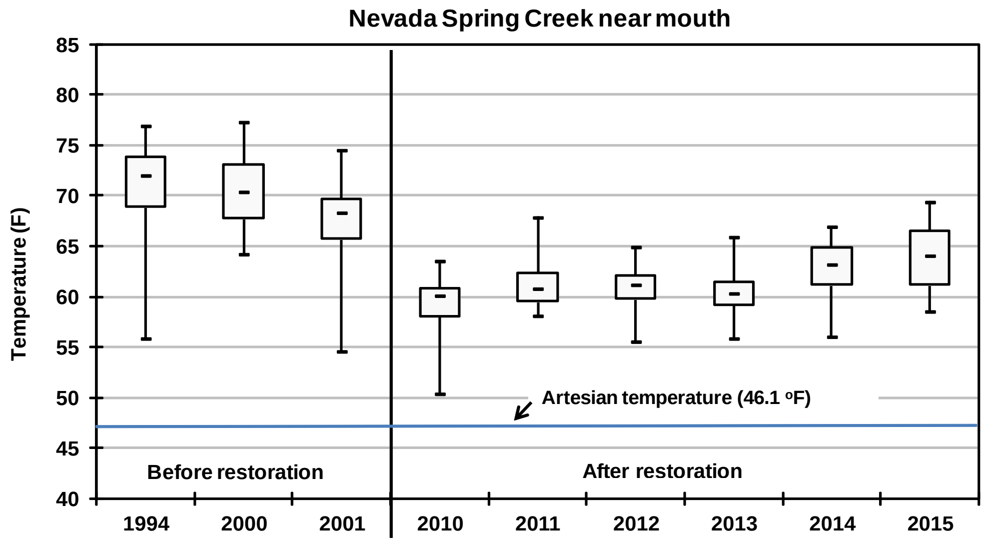


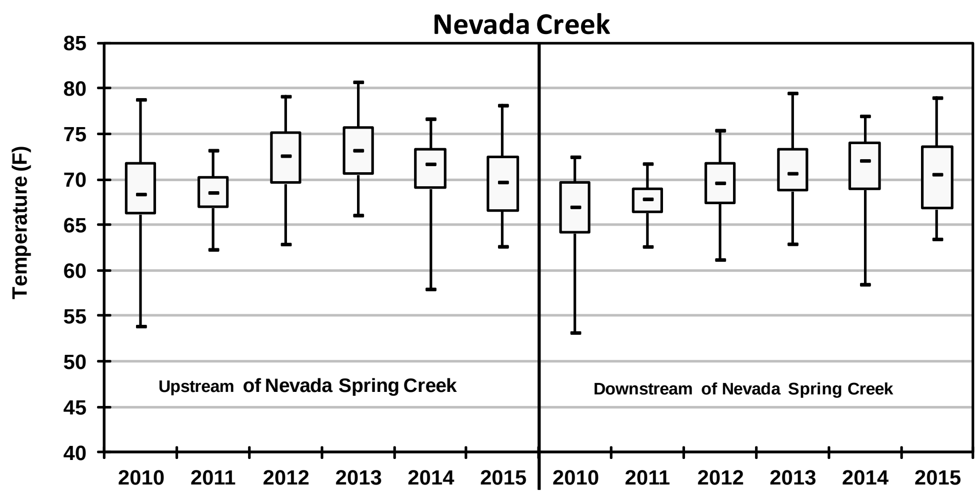
Figure 46. CPUE for age 1 and older trout in Nevada Creek immediately downstream of Nevada Spring Creek (mile 5.1-6.3), 2005-2015.

<u>Fish populations and water temperature monitoring</u>

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 and within Nevada Spring Creek (Pierce et al. 2006, Appendix C). In 2011 and 2012, fish population surveys were conducted at two



	Before			After					
	1994	2000	2001	2010	2011	2012	2013	2014	2015
Min	55.9	64.2	54.6	50.4	58.1	55.6	55.9	56.0	58.5
25%	68.9	67.8	65.8	58.0	59.5	59.8	59.1	61.2	61.2
Median	72.0	70.4	68.3	60.1	60.8	61.2	60.3	63.2	64.1
75%	73.8	73.2	69.7	60.8	62.4	62.1	61.5	64.9	66.6
Max	76.9	77.3	74.5	63.5	67.9	64.9	65.9	66.9	69.4



	Upstream						Downstream					
	2010	2011	2012	2013	2014	2015	2010	2011	2012	2013	2014	2015
Min	53.9	62.3	62.9	66.1	58.0	62.7	53.2	62.7	61.2	62.9	58.5	63.5
25%	66.3	67.0	69.6	70.6	69.1	66.6	64.2	66.4	67.4	68.8	69.0	66.9
Median	68.4	68.6	72.6	73.2	71.7	69.7	67.0	67.9	69.6	70.7	72.1	70.6
75%	71.8	70.2	75.1	75.7	73.3	72.5	69.7	69.0	71.8	73.3	74.0	73.6
Max	78.8	73.2	79.2	80.7	76.7	78.2	72.5	71.8	75.4	79.5	77.0	79.0

Figure 47. The top graph shows a before/after temperature summary for Nevada Spring Creek at the mouth from1 July to 1 September. The bottom graph shows a comparison of water temperatures up and downstream of the Nevada Spring Creek confluence for the same period.

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 45, Appendices A and C). We also continued to monitor lower Nevada Creek (mile 5.0-6.3) in a site originally established in 2005 immediately downstream of the Nevada Spring Creek confluence (Figure 46, Appendices A and C).

From 2013 through 2015, we continued water temperature monitoring on Nevada Spring Creek and on Nevada Creek upstream (mile 6.3) and downstream (mile 5.0) of the Nevada Spring Creek confluence (Figure 47). A comparison of the pre-and post-treatment temperature dataset shows the cooling of Nevada Spring Creek near the mouth, as well as only a slight temperature reduction in Nevada Creek downstream of Nevada Spring Creek. The downstream temperature sensor is also located downstream of the Douglas Creek confluence, which may limit the cooling influence of Nevada Spring Creek due to high temperatures (FWP unpublished temperature data). Future monitoring should identify the temperature effects of both streams. 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 River, a large 4th order tributary, drains a 313 mile^2 and flows 40.3 miles south from the Continental Divide near Scapegoat Mountain and enters the Blackfoot River at mile 54. Average monthly discharge range from a low of about 100 cfs in mid winter to high of about 1,300 cfs in late May and early June (USGS provisional data station 123389300). The upper 23.8 miles of stream drain the Scapegoat Wilderness (Lolo National Forest) with an average gradient of 135ft/mile. At stream mile 26.2, the North Fork flows over the North Fork falls is joined by the Dry Fork junction. The North Fork enters private land near stream mile 16.5. Upon exiting the mountains near mile 13.0, the North Fork enters Kleinschmidt Flat, a large glacial outwash plain where stream gradients decrease to about 33ft/mile. Upon entering Kleinschmidt Flat, the North Fork loses water to alluvium between mile 8.3 and 6.1 before gaining groundwater inflows, including those from several spring creeks. Five irrigation canals, located on the Flat between miles 15.3 and 8.8, divert up to an estimated 40-60 cfs from the North

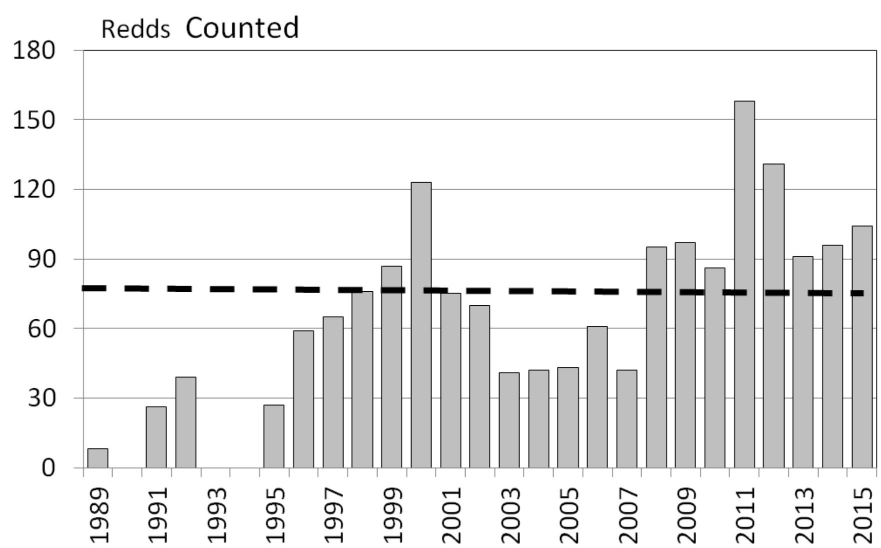


Figure 48. Bull trout redd counts in the North Fork of the Blackfoot River, 1989-2015. Redd counts were not performed in 1990, 1993 and 1994. The dashed line represents the long-term mean.

Fork.

The North Fork has been the focus of comprehensive private land restoration project, 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 on the mainstem and its tributaries, 4) improved riparian grazing practices, and 5) conservation easements on a majority of the riparian areas found on private land.

In addition to this work, a possible large-scale native trout conservation project is now being investigated in the Scapegoat Wilderness area of the upper North Fork drainage upstream of the North Fork Falls (Pierce et al. 2016).

Fish Populations

Depending on location, the North Fork of the Blackfoot River supports migratory westslope cutthroat trout and bull trout, mountain whitefish, migratory and resident rainbow trout, brown trout and brook trout. The North Fork supports the largest run of migratory (fluvial) bull trout in the upper Clark Fork River Basin. The known geographic range of the North Fork bull trout extends from spawning sites downstream of the North Fork Falls to the upper Blackfoot River near Lincoln to the Clark Fork River (Swanberg 1997, Pierce et al. 2004, Schmetterling 2003). A 2015 genetic assignment study not only identified North Fork stock as distinct (Results Part IV), but also identified the North Fork bull trout as the prevalent stock in the lower Blackfoot River. This study also connected the North Fork stock with Salmon Lake within the Clearwater drainage for the first time (Results Part IV).

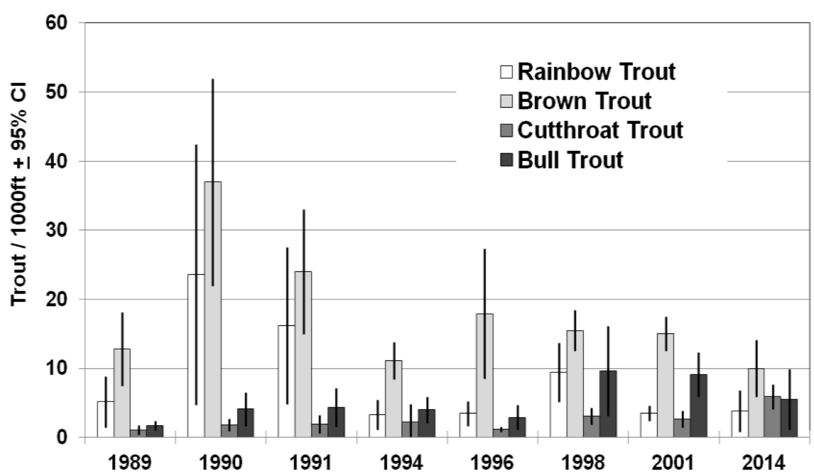


Figure 49. Estimates of trout abundance (fish >6.0’) in the lower North Fork of the Blackfoot River, 1989-2014.

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. Redd counts increased during the decade of the 1990 after protective angling regulations and the screening of all the North Fork ditches were enacted (Figure 48). Redd count then showed a seven-year decline (2001-2007) during a protracted drought. With the return of more normal flows and the removal of Milltown Dam in 2008, bull trout spawning has again increased between 2008 and 2015.

In addition to bull trout redd counts, we conducted fish population survey in 2014 at a long-term monitoring (electrofishing) site in the lower North Fork established in 1989 (Figure 49, Appendix C). These surveys indicate incremental increases in native trout abundance during the last 16 years.

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 1st order tributary to lower Chamberlain Creek with a base-flow of about one cfs. Approximately 9.4 in length, Pearson Creek begins on BLM and DNRC lands and drains the northern and western slopes of Chamberlain and Granite Mountains. It flows north and enters private agricultural land at mile 2.9 and enters Chamberlain Creek at mile 0.1 with a baseflow of about 1.0 cfs. Stream gradient decreases from 173ft/mile in the headwaters to about 60ft/mile the lower mile of stream near.

Pearson Creek has a history of channel alterations and adverse irrigation, riparian grazing and timber harvest practices in its lower two-miles of channel. From 1994 through 2013, the lower two miles of Pearson Creek have been the focus of comprehensive restoration involving channel reconstruction and in-stream habitat improvement and revegetation, flow enhancement (water leasing), riparian grazing changes and conservation easements. In 2013 an undersized culvert at mile 0.8 was replaced with a bridge and immediately upstream 1,500 feet of channel was reconstructed to facilitate up- and downstream fish movement and improve habitat. Upstream improvements include the replacement of an undersized culvert with a bridge and land exchanges in the headwaters that transfer all former Plum Creek Timber Company lands to BLM and DNRC ownership. With the completion of this work, all major fisheries impairments have been corrected.

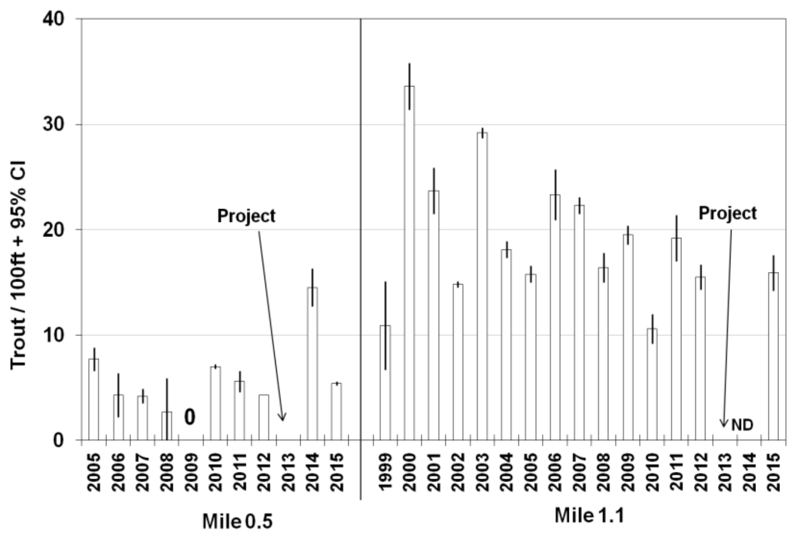


Figure 50. Estimates of abundance for age 1 and older westslope cutthroat trout in Pearson Creek at miles 0.5 and 1.1. The road crossing project at 0.8 was completed in 2013.

Fish populations

Pearson Creek is a fluvial westslope cutthroat trout spawning stream connected to Chamberlain Creek. From 2014-2015, 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 (Figure 50). Additional future monitoring should shed light on downstream effects of the road project.

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, a large 3rd order tributary drains a 43 mile^2 watershed and flows 14.1 miles west-northwest and enters the Blackfoot River at mile 108 with a baseflow of about 10-15 cfs. The stream originates on the Continental Divide in the H-LC National Forest near Stemple Pass. Stream gradients vary from 325ft/mile in the headwaters to 29ft/mile near the mouth. Landownership in the upper 11.6 miles of Poorman Creek consist primarily of H-LC National Forest land mixed with small parcels of privately owned land adjacent to the stream channel. The lower 2.5 miles of stream flow entirely through private ranchland.

Poorman Creek impairments stem from hardrock and placer mining, irrigation dewatering, fish losses to ditches, channel instability, excessive riparian grazing pressure, subdivision impacts, road encroachment, sedimentation and undersized culverts. Corrective actions began in 2002 and continue through the present. Fisheries-related improvements initially focused on lower Poorman Creek and included 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. In addition, several road crossings on the mainstem of Poorman Creek have been upgraded to improve habitat connectivity for native trout. More recent work involves the relocation of two miles of country road from the riparian area of the South Fork of Poorman Creek to an upland site. This relocation removed four fords, one undersized culvert and decommissioned 2,200 feet of streamside road.

Fish populations

Poorman Creek supports genetically pure westslope cutthroat trout, as well as brown trout and brook trout. It is the only small stream south of the Blackfoot River that still supports 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 located up- and downstream of two diversions (miles 1.3 and 1.5) prior to restoration. Survey results from both monitoring sites from 2001-2015 are shown in Figure 51. These surveys show improved numbers downstream of the diversions where

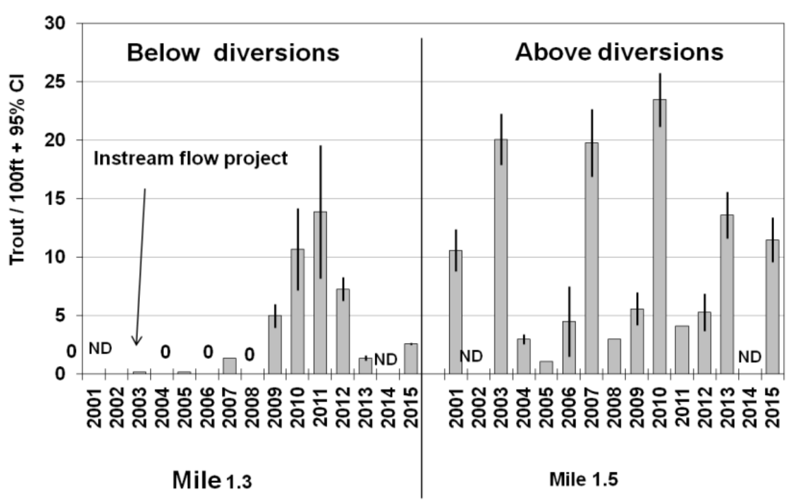


Figure 51. Estimates of abundance for age 1 and older trout in Poorman Creek at miles 1.3 and 1.5, 2001 – 2015.

trout were absent prior to restoration. However, surveys also reveal high year-to-year variation in abundance (Figure 51), which is likely flow-related. Interpretation of recent monitoring data is complicated by 1) unplanned livestock-induced streambank damage in 2014, and 2) the loss of high flows (and likely fish) at an unmanaged irrigation ditch located at mile 1.5 immediately downstream of upstream survey site.

In addition to this monitoring, we sampled the headwaters of Poorman Creek at two sites (mile 8.4 and 9.9) where we also collected fin clips for a bull trout genetic assignment study. Catch and size statistics are located in Appendix A. The assignment found genetically distinct bull trout in the headwaters and no evidence of a fluvial component in the mainstem Blackfoot River (Results Part IV).

Sauerkraut Creek

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

<u>Project summary</u>

Sauerkraut, a 2nd order north flowing stream, drains a 13.3 mile^2 watershed on the eastern slopes of Ogden Mountain. The mainstem is 7.6 miles in length and enters the upper Blackfoot River at mile 102.1. The headwaters are located on the H-LC National Forest land and the lower 3.2 miles of stream are located on private land. Stream gradients average 331ft/mile in the headwater and 91ft/mile near the mouth. Sauerkraut Creek loses water at mile 2.9 and becomes intermittent, then begins to gain water at mile 2.7 and produces 3-4cfs during baseflow periods.

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, undersized culverts, overgrazing by livestock and dewatering by irrigation have also contributed to fisheries impairments. 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 easement, a stream restoration project was developed in upper Sauerkraut Creek (miles 2-3) to correct past mining and grazing impacts. Restoration involved the reconstruction of approximately 5,000 feet of Sauerkraut Creek, a grazing management plan involving riparian fencing and off-site water developments, shrub transplants, seeding and weed control. In 2010-12, three undersized stream crossings (miles 0.3, 1.5 and 1.8) were upgraded to bridges to accommodate fish passage and channel function. In addition, irrigation ditches were consolidated into a single screened diversion in 2014. An instream flow agreement was also secured for a minimum flow of three cfs on the lower two miles of Sauerkraut in 2012. In 2015, an additional 770 feet of channel restoration and road decommissioning was completed on the H-LC National Forest to reduce sediment delivery and improve fish habitat.

Fish populations

Sauerkraut Creek supports non-hybridized westslope cutthroat trout along with low numbers of brook and the incidental presence of bull trout in the headwaters and a mixed community of salmonids in the lower stream (Appendix A and B). Sauerkraut Creek also supports a small run of migratory westslope cutthroat trout (Pierce et al. 2007) along with brown trout in lower Sauerkraut Creek. Western pearlshell mussels are also present in lower Sauerkraut Creek.

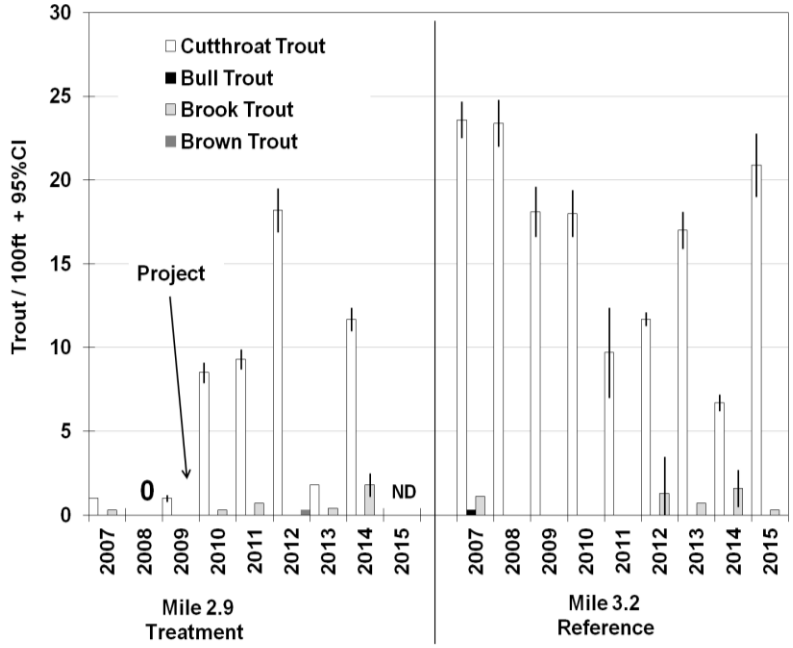


Figure 52. Estimates of abundance age 1 and older trout in Sauerkraut Creek at treatment (mile 2.9) and reference reaches (mile 3.2), 2007-2015. The mile 2.9 site was dry in 2015.

To develop a fisheries baseline for the upper Sauerkraut Creek restoration project, we established a fisheries monitoring site at an upstream reference reach (mile 3.2) and within the treatment site (mile 2.9) beginning in 2007 (Figure 52). In 2013, flows were very low at the mile 2.9 monitoring section. In 2015, this site was dry due to drought and water loss to alluvium.

Shanley Creek

Restoration objectives: restore habitat for all fish species; restore migration corridors for native fish; reduce loss of fish to irrigation ditches; maintain minimal instream flows.

Project Summary

Shanley Creek, a 2nd order tributary, drains a 13.9 mile^2 watershed and flows south 11.6 miles before entering Cottonwood Creek at mile 5.6 with an estimated baseflow of 3-5 cfs. Shanley Creek begins on the Lolo National Forest near Dunham Point then enters State and private ranchland at mile 6.3. Stream gradients range from 580ft/mile near the headwaters to 81ft/mile the lower mile of stream. Channel runs though B3 to E4 stream types.

Shanley Creek has been the focus of several riparian improvement projects, plus the placement of conservation easements on private and State (University of Montana) land. Since 1994, most of the restoration work focused on improving riparian grazing practices and upgrading irrigation systems to reduce fish losses and conserve water. Currently, the lower 1.8 miles of Shanley Creek are under riparian grazing management strategies. In 2015, a project was completed that included removing a road in the riparian area, removing to two undersized culverts and upgrading at a ford to a bridge. Upstream of mile 1.8, excessive livestock grazing in riparian areas continues to degrade aquatic habitat on both private and State properties. The Lolo National Forest in cooperation with TNC and BBCTU removed three culvert fish passage barriers from Shanley Creek between 2012 and 2014 that collectively reconnected about 2.1 miles of stream on the National Forest.

Fish Populations

Shanley Creek once supported bull trout (Brett Bodecker, personal communication). However FWP surveys have not detected bull in any surveys since fisheries surveys began in 1993.

In 2015, we resurveyed fish populations at two locations (0.2. and 1.6) influenced by restoration activities. The mile 0.2 survey site was established in 1993 in a livestock-degraded section of Shanley Creek prior to livestock exclusion. The upper sample site was established immediately downstream of the Bandy Reservoir diversion ditch prior to the installation of a screen fish within the ditch (Figure 53). The lower sites shows increased abundance of trout (primarily brown trout) following riparian fencing project; whereas, the upper survey shows increased westslope cutthroat trout and a corresponding decline in nonnative trout (primarily brook trout) in recent samples.

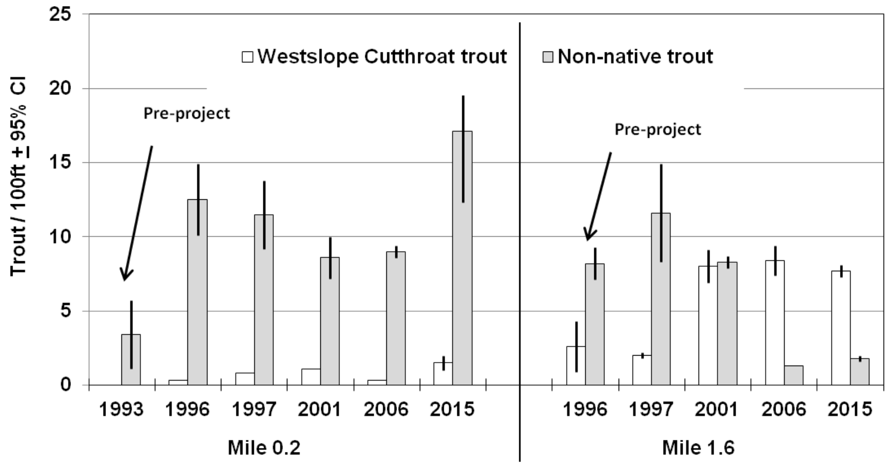


Figure 53. Estimates of trout abundance for age 1 and older trout at two project monitoring sites on Shanley Creek. The lower site (mile 0.2) is a reach influenced by riparian fencing. The upstream reach (mile 1.6) is influenced by a screened irrigation diversion.

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 1st order tributary to Copper Creek, entering at mile 6.2 with a base flow of about four cfs. The mainstem of Snowbank Creek is 5.1 miles in length and drains a small (7.6 mile^2) watershed on the northeast slopes of Stonewall Mountain within the H-LC National Forest. Stream gradients range from 917ft/mile in its headwaters to 159ft/mile the lower mile of stream. In 2003, the Snow Talon wildlife swept through the Copper Creek drainage. Prior to 2003, lower Snowbank Creek was chronically dewatered downstream of a diversion at mile 0.5, which also created fish passage and entrainment problems. Following the identification of these issues, baseflows were restored to a target four cfs in 2004, and then in 2009, the diversion was replaced with one that allowed improved fish passage and a Coanda fish screen was placed at the head of the ditch to eliminate entrainment. In 2013 an undersized culvert at mile 0.2 was replaced with a bridge to facilitate upstream movement of fish.

<u>Fish populations</u>

Snowbank Creek supports genetically pure westslope cutthroat trout and bull trout. From 2003 through 2015, we monitored fisheries at mile 0.4 at a site established prior to both the Snow Talon wildfire and restoration actions. Prior to restoration, westslope cutthroat trout sampled in low abundance, and bull trout were absent from three electrofishing survey sites in 2003 (Pierce et al. 2004, 2006). Following restoration, westslope cutthroat trout abundance increased sharply. Bull trout were then detected in 2005 (Figure 54), followed by documented spawning within and upstream of the dewatered stream segment in 2008.

Recent (2013-15) monitoring revealed a significant decline in trout abundance and a community shift from westslope cutthroat to bull trout. Changes in abundance are consistent with the rise and fall of native trout abundance in Copper Creek before and after the Snow Talon wildfire (Pierce et al. 2012). Bull trout redd counts in Snowbank Creek are shown in Table 1.

Lastly, we completed genetic assignment tests for bull trout in Snowbank Creek in 2013 (Results Part IV). As expected, Snowbank bull trout are genetically similar to Copper Creek. Assignment tests identified Copper Creek and Snowbank Creek bull trout as far down stream as the Johnsrud Section of the lower Blackfoot River a distance of 113 stream miles (Results Part IV). Similar to past telemetry studies (Pierce et al. 2006) this assignment study also confirmed Copper Creek drainage as the primary bull trout recruitment source of upper mainstem Blackfoot River upstream of the North Fork Blackfoot River.

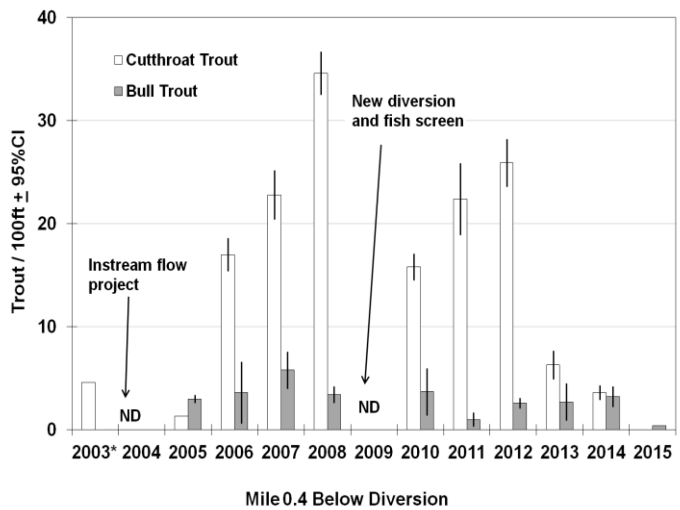


Figure 54. Population estimates for age 1 and older cutthroat trout and bull trout downstream of Snowbank Creek diversion, 2003-2015.

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 2nd order basin-fed tributary to Nevada Spring Creek. A small drainage of 6.2 miles^2 its 8.4 mile length drain the northwestern slopes of Ogden Mountain and flows west-northwest 3.7 miles through the H-LC National Forest before entering private ranchland at mile 4.7 then continues on to join Nevada Spring Creek ~100feet below the (artesian) spring source, contributing base-flow of about one cfs. Stream gradients range from about 368ft/mile at the headwaters to a low 5ft/mile in the lower mile of stream. 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, a stream restoration project was implemented concurrent with restoration 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 2006, a 10-year in-stream flow lease also went into effect. Since then, habitat maintenance high flows have been allowed and low flows have been managed at or near the target of 0.75 cfs. A final element to the project was the installation of two Coanda fish screens at both irrigation diversions in the spring of 2007.

Fish Populations and water temperature monitoring

Fish population surveys are described in prior FWP reports (Pierce et al. 2004-2013). Following the restoration of Wasson Creek, a comprehensive telemetry study emphasizing movements of westslope cutthroat trout was completed involving Wasson Creek and receiving waters of Nevada Spring Creek and Nevada Creek in 2014. That study is found in Results Part IV.

Since 2003, water temperature monitoring at the mouth continues to show summer water temperatures cooling (Figure 55, Appendix E). This cooling likely relates to the cumulative effects involving 1) the recovery of streamside plants, 2) increased flows, and 3) the passive narrowing of the channel in response to stream-side grazing changes.

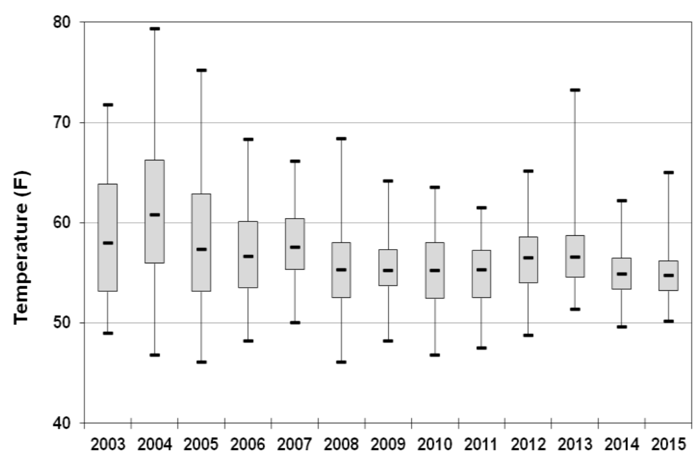


Figure 55. July water temperatures for Wasson Creek near the mouth, 2003-2015.

RECOMMENDATIONS

1) Continue to develop and implement restoration projects on high priority stream with willing landowners, land managers and conservation groups. Emphasize streams with past and current projects.

2) Continue to support the *Southwest Crown of the Continent Restoration Project* on USFS lands with fisheries information and restoration funding.

3) Develop specific restoration strategies to help recover and protect native trout on TNC properties.

4) Encourage watershed groups and agencies to provide for the monitoring and long-term maintenance needs of restoration projects they promote.

5) Pursue the targeted replacement of nonnative hybrid *Oncorhynchus* trout with native westslope cutthroat trout and bull trout upstream of the North Fork Falls.

7) Expand bull trout genetic assignment tests to include all natal streams in the Clark Fork River. Collect fin clips for bull trout during routine population surveys on the Blackfoot River and Clark Fork River to help monitor populations and to help identify restoration opportunities within the larger landscape.

8) Seek long-term solutions to tributary dewatering in the Blackfoot River Basin within a context of drought management, climate change and the 1904 water rights now jointly held by FWP and the Confederated Salish and Kootenai Tribes.

9) Begin to monitor the biological effectiveness of the remediation and restoration in the upper Blackfoot Mining Complex.

LITERATURE CITED

Aitken, G. 1997. Restoration of trout waters in the West; Blackfoot River of Montana. Pages 402-424 *in* Williams, J. E., C. A. Wood and M. P. Dombeck, *editors*. Watershed Restoration: Principles and Practices. American Fisheries Society. Bethesda, Maryland.

Bartholomew, J. L. and L. C. Wilson. 2002. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Benson, A. 2009. Effects of barriers on migratory bull trout and application of a conceptual framework to evaluate tradeoff associates with dam removal in the Clearwater River Drainage, Montana. Masters of Science Thesis in Wildlife Biology, University of Montana, Missoula, MT.

BBCTU [Big Blackfoot Chapter of Trout Unlimited]. 2016. Available at http://www.blackfoottrout.org/index2.html.

Carim, K, L. Eby and R. Pierce. 2015. Does whirling disease mediate hybridization between a native and nonnative trout? *Transactions of the American Fisheries Society*. 35:337-351.

DEQ [Montana Department of Environmental Quality]. 2008. Middle Blackfoot Nevada Creek total maximum daily loads and water quality improvement plan, Helena, Montana.

Eby, L. A., O, Helmy, L. M. Holsinger, and M. K. Young. 2014. Evidence of climate-induced range contractions in bull trout *Salvelinus confluentus* in a Rocky Mountain watershed, U.S.A. PLoS ONE, 9, e98812.

Eby, L. A., R. Pierce, M. Sparks, K. Carim and C. Podner. 2015. Multiscale prediction of whirling disease risk in the Blackfoot River Basin, Montana: A useful consideration of restoration prioritization? *Transactions of the American Fisheries Society*. 144:753-766.

Dunham, J., B. Rieman, and G. Chandler. 2003. Influences of temperature and environmental variables on the distribution of bull trout within streams at the southern marigin of its range. *North American Journal of Fisheries Management* 23:894-904.

Fitzgerald G. 1997. Analysis and inventory of riparian vegetation along Nevada and Monture Creeks using ADAR imagery. MS. Thesis University of Montana, Missoula.

Fredenberg, F. 1992. Evaluation of electrofishing-induced spinal injuries resulting from field electrofishing surveys in Montana. Montana Department of Fish, Wildlife and Parks, Bozeman, Montana.

Hansen, P. L., R. D. Pfister, K. Boggs, B. J. Cook, J. Joy, and D. L. Hinkley. 1995. Classification and management of Montana's riparian and wetland sites. Miscellaneous Publication No. 54. Montana Forest and Conservation Experimental Station, School of Forestry, University of Montana, Missoula, Montana.\

InRoads Consulting. 2015. The Nature Conservancy in Montana Clearwater-Blackfoot Project Road and Stream Crossing Inventory Report, Missoula, Montana.

Isaak, D. J., M. K. Young, D. Nagel, D. A. Horan and M. C. Groce. 2015. The cold-water climate shield: delineating refugia for preserving salmonid fisheries through the 21st century. Climate Change Biology (2015), doi: 10.111/gcb.12879.

FWP - Montana Fish, Wildlife and Parks. 2015. Montana Stagtewide Angling Pressure 2013. Helena, Montana.

MBTRT [Montana Bull Trout Restoration Team]. 2000. Restoration plan for bull trout in the Clark Fork River basin and Kootenai River basin. Montana Fish, Wildlife and Parks. Helena, Montana.

McMahon, T., J. Robison-Cox, J. Rotella, T. Horton, and B. Kerans. 2010. Trout population responses to whirling disease epizootics in Montana rivers. Pages 195-202 in Proceedings of the Wild Trout X Symposium, Sept 2010, West Yellowstone, Montana.

Moore, J. N., S. N. Louma, and D. Peters. 1991. Downstream effect of mine effluent on intermontane riparian system. *Canadian Journal of Fisheries and Aquatic Sciences* 48:222-232.

Murphy B. R. and D. W. Willis, editors. 1996. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Peters and Spoon. 1989. Preliminary fisheries inventory of the Big Blackfoot River. Montana Fish, Wildlife and Parks, Missoula, Montana.

Peters, D. 1990. Inventory of fishery resources in the Blackfoot River and major tributaries to the Blackfoot River. Montana Department of Fish, Wildlife and Parks, Missoula, Montana.

Pierce, R., and D. Peters. 1990. Aquatic investigations in the the middle Blackfoot River, Nevada Creek and Nevada Spring Creek corridors, Montana Department of Fish, Wildlife and Parks, Missoula, Montana.

Pierce, R., D. Peters, and T. Swanberg. 1997. Blackfoot River Restoration Progress Report, Montana Fish, Wildlife and Parks, Missoula, Montana.

Pierce, R. and D. Schmetterling. 1999. Blackfoot River Restoration Project: Monitoring and progress report, 1997-1998. Montana Fish Wildlife and Parks, Missoula, MT.

Pierce, R., C. Podner, and J. McFee. 2001. Blackfoot River fisheries inventory, monitoring and restoration report. Montana Fish, Wildlfie and Parks, Missoula, Montana.

Pierce, R., R. Anderson, and C. Podner. 2004. The Big Blackfoot River restoration progress report for 2002 and 2003. Montana Fish, Wildlfie and Parks, Missoula, Montana.

Pierce, R., R. Aasheim, and C. Podner. 2005. An integrated stream restoration and native fish conservation strategy for the Blackfoot River basin. Montana Fish, Wildlife and Parks, Missoula, Montana.

Pierce, R., and C. Podner. 2006. The Big Blackfoot River fisheries restoration report for 2004 and 2005. Montana Fish, Wildlife and Parks, Missoula, Montana.

Pierce, R., R. Aasheim, and C. Podner. 2007. Fluvial westslope cutthroat trout movements and restoration relationships in the upper Blackfoot Basin, Montana. Intermountain Journal of Sciences 13(2):72–85.

Pierce, R., C. Podner, M. Davidson, L. Knotek, and J. Thabes. 2008. Big Blackfoot River fisheries and restoration investigations for 2006 and 2007. Montana Fish, Wildlife and Parks. Missoula, Montana.

Pierce, R., C. Podner, and M. Davidson. 2009. Correlation of fluvial rainbow trout spawning life history with the severity of infection by *Myxobolus cerebralis* in the Blackfoot River Basin, Montana. *Transactions of the American Fisheries Society* 138:251-263.

Pierce, R., and C. Podner. 2011. Fisheries investigations in the Big Blackfoot River Basin, 2008-2010. Montana Fish, Wildlife and Parks, Missoula, Montana.

Pierce, R., M. Davidson and C. Podner. 2012. Spawning behavior of mountain whitefish and co-occurrence of *Myxobolus cerebralis* in the Blackfoot River Basin, Montana. *Transactions of the American Fisheries Society* 141:720-730.

Pierce, R., C. Podner, and K. Carim. 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River Basin, Montana. *Transactions of the American Fisheries Society* 142:68-81.

Pierce, R., C. Podner, L. Marczak and L. Jones. 2014. Instream habitat restoration and stream temperature reduction in a whirling disease positive spring creek in the Blackfoot River Basin, Montana. *Transactions of the American Fisheries Society* 143:1188–1198.

Pierce, R., C. Podner, T. Wendt, K. Carim and R. Shields. 2014. Westslope cutthroat trout movements through restored habitat and Coanda diversions in the Nevada Spring Creek complex, Blackfoot Basin, Montana. *Transactions of the American Fisheries Society* 143:230–239.

Pierce, R., C. Podner and L. Jones. 2015. Long-Term Increases in Trout Abundance following Channel Reconstruction, Instream Wood Placement, and Livestock removal from a Spring Creek in the Blackfoot Basin, Montana. *Transactions of the American Fisheries Society* 144:184-195.

Roni, P., editor. 2005. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Rosgen, D. 1996. Applied Fluvial Geomorphology. Wildlands Hydrology, Pagosa Springs, CO. International Standard Book, Number 0-9653289-0-2.

Schmetterling, D. A. and R. W. Pierce. 1999. Success of instream habitat structures after a 50-year flood in Gold Creek, Montana. *Restoration Ecology* 7(4):369–375.

Schmetterling, D. A. 2001. Seasonal movements of fluvial WCT in the Blackfoot River drainage, Montana. *North American Journal of Fisheries Management* 21:507–520.

Schmetterling, D. A. 2003. Reconnecting a fragmented river: Movements of westslope cutthroat trout and bull trout after transport upstream of Milltown Dam, Montana. *North American Journal of Fisheries Management* 23:721–731.

Swanberg, T., and L Burns. 1997. Movements and habitat use of radio-tagged bull trout in the upper Blackfoot River Drainage, Special Report Region 2 Fisheries, Missoula, Montana.

Swanberg, T. R. 1997. Movements of and habitat use by fluvial bull trout in the Blackfoot River. *Transactions of the American Fisheries Society* 126:735–746.

Sugden, B. 1996. Belmont Creek watershed analysis. Plum Creek Timber Company. Clearwater Unit, Missoula, Montana.

USGS provisional discharge data the North Fork Blackfoot River, station 123389300. 2016. Available at http://waterdata.usgs.gov/MT/nwis/current/?type=flow.

USGS provisional discharge data. Blackfoot River near Bonner, station 12340000. 2016. Available at: http://waterdata.usgs.gov/MT/nwis/current/?type=flow.

USFWS [United States Fish and Wildlife Service]. 2010. Endangered and threatened wildlife and plants; revised designation of critical habitat for bull trout in the contimnerous United States; final rule. Federal Register: 75 FR 2269. Available at http://.fws.gov/pacific/bulltrout/FinalCH2010.html#FinalCH.

USFWS [United States Fish and Wildlife Service]. 2015. Recovery plan for the coterminous United States population of bull trout (*Salvelinus confluentus*). Portland Oregon. xii=179 pages. Available at http://www.fws.gov/endangered/species/recovery-plans.html

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ARTICLE

Westslope Cutthroat Trout Movements through Restored Habitat and Coanda Diversions in the Nevada Spring Creek Complex, Blackfoot Basin, Montana

Ron Pierce,* Craig Podner, and Tracy Wendt
Montana Fish, Wildlife, and Parks, 3201 Spurgin Road, Missoula, Montana 59804, USA

Ron Shields
Trout Unlimited National, 6184 Head Lane, Helena, Montana 59602, USA

Kellie Carim
College of Forestry and Conservation, University of Montana, 32 Campus Drive, Missoula, Montana 59812, USA

Abstract
In the Blackfoot basin of western Montana, the recovery of migratory Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* 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. Prespawning movements of the remaining four radio-tagged fish were much farther than those of Wasson Creek spawners (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 overwinter habitat for Westslope Cutthroat Trout that spawn and summer elsewhere in the basin.

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, overfishing, 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 *O. c. 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

*Corresponding author: rpierce@mt.gov
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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) 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 multiscale 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). Multiscale 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, 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 basinwide 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 (MFWP 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 multiscale 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 posttreatment spawning behavior of migratory Westslope Cutthroat Trout associated with this local expansion within a context of irrigation system and multiscale restoration activities. Specific study objectives are to (1) examine migration behaviors of Westslope Cutthroat Trout from their wintering areas into summer and to identify 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 semiarid 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 to 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

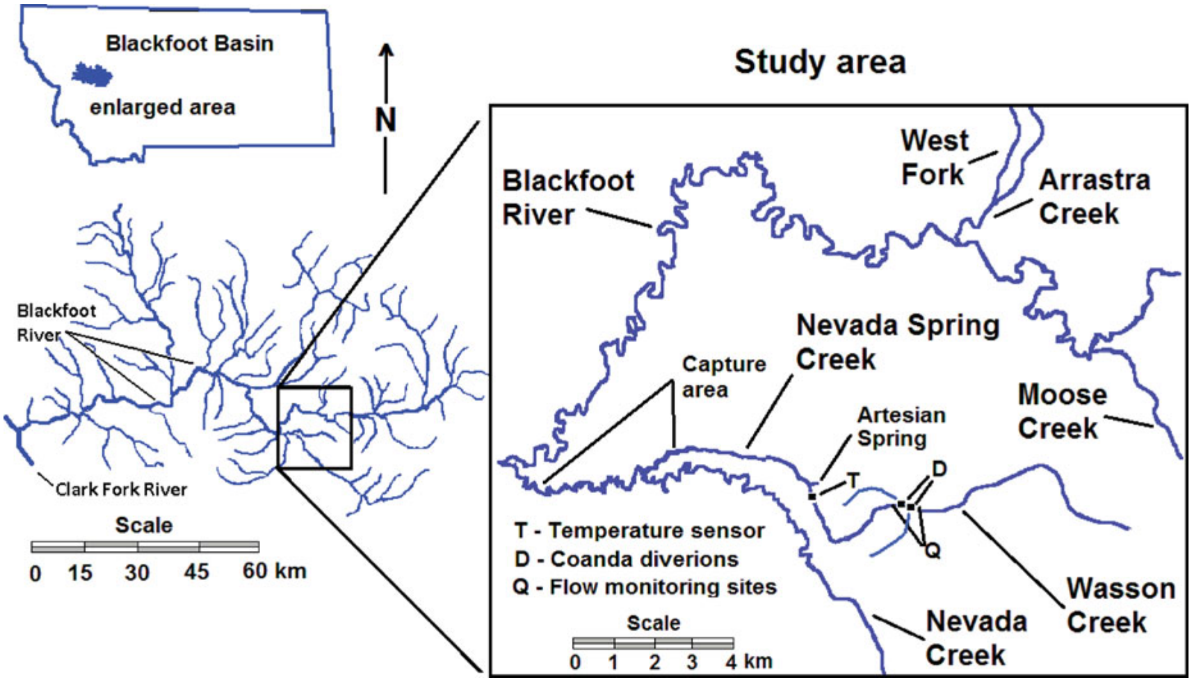

FIGURE 1. Location map showing the Blackfoot basin and the study area. Also shown are the capture locations of fish, flow and temperature monitoring sites, and locations of the two Coanda diversions.

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°C to 7.8°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 overwidened and heavily degraded with high summer temperatures near 25°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 Creek, lower Nevada Spring Creek, and lower Nevada Creek (Montana Fish, Wildlife, and Parks, 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).

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 (Figure 2 [top] 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

TABLE 1. Summary of stream metrics before and after restoration; nd = no data, na = not applicable (modified from Pierce et al. 2013).

	Width-to-depth ratio		Sinuosity		Percent pool area		Maximum summer temperature (°C)		Minimum summer flow (m^3/s)		Ditch entrainment (number of age-1 + trout/30 m)	
Stream name	Before	After	Before	After	Before	After	Before	After	Before	After	Before	After
Nevada Spring Creek	22	3.2	1.4	1.7	51	71	25	18	0.17	0.28	na	na
Wasson Creek	3	0.7	1.0	1.5	nd	nd	22	18	0	0.02	1.3	0

FIGURE 2. Picture of a Coanda diversion and fish screen on Wasson Creek. The photo shows two fish screens as well as a sediment sluice gate (middle slot with boards). The smaller photo (top) shows an adult Westslope Cutthroat Trout ascending the Coanda. (Photo by Jamie Nesbit.)

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 Trout (Pierce et al. 2013) but also the increasing presence of larger adult Westslope Cutthroat Trout (fish > 300-mm TL) in Nevada Creek downstream of the Nevada Spring Creek confluence (Pierce and Podner 2013).

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 (hereafter, "Coanda") diversion–fish screens in the main stem of Wasson Creek at two diversion points (Figures 1, 2). Instream flow enhancement was intended to mimic natural flow regimes including high and low flows, while maintaining a minimum base flow (>0.02 m^3/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 (Figure 2 [top]). 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

Radiotelemetry.—Consistent with previous studies (Schmetterling 2001; Pierce et al. 2007), we captured adult Westslope Cutthroat Trout 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) 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 TL (mean, 333-mm TL) 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 *O. mykiss* 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 d, emitted an individual coded signal, did not exceed 2% of fish weight (Winter 1996), and were implanted following standard surgical methods (Swanberg et al. 1999). Technicians use an omnidirectional 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 handheld, three-element Yagi antenna. Technicians located fish weekly prior to migrations, 3–4 times/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 river kilometer.

Fish were assumed to have spawned at their uppermost 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, age-0 Westslope Cutthroat Trout, or a combination thereof. 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 prespawning 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 $\alpha = 0.05$ 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 3). Streamflow and temperature measurements were taken between 1 April and

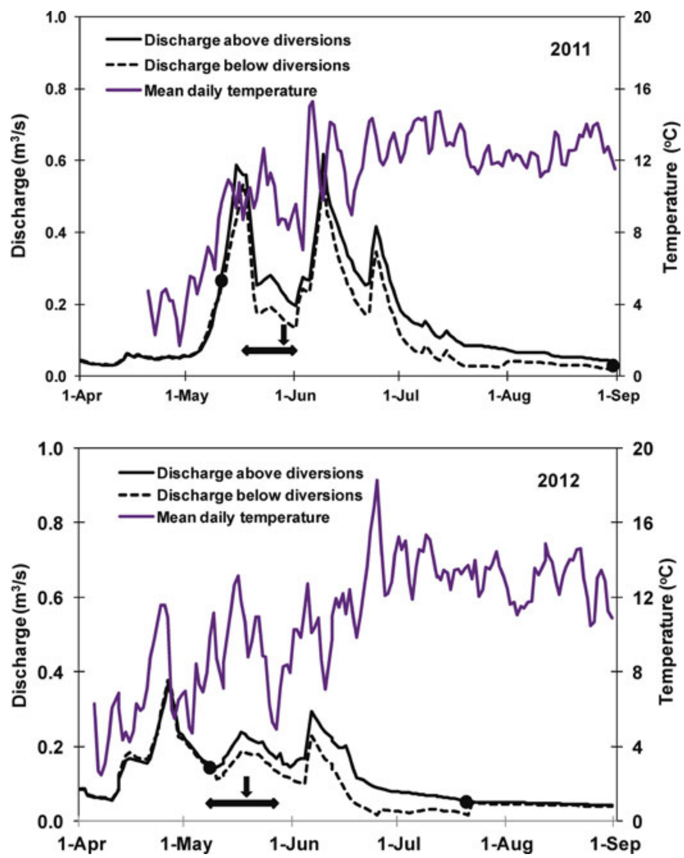

FIGURE 3. Relationship of migration and spawning to discharge and water temperatures in Wasson Creek for 2011 and 2012. The horizontal (arrowed) bar shows the migration period through the diversions (n = 3 in 2011, n = 5 in 2012). The vertical arrow represents the peak (median) spawning period for all Wasson Creek fish (n = 10). The dark circles show the dates irrigation was turned on and off.

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-min interval) digital thermograph (Onset Computer, 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 upstream (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 2013).

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 eight 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 2011–2012 Cutthroat Trout prespawning 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 moving up into two upper river tributaries (Arrastra and Moose creeks). Over an average of 14 d (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°C and 12°C as measured in lower Wasson Creek (Figure 3). Of these 10 fish, 8 spawned in a concentrated area upstream of the diversions (Figure 4).

Spawners spent an average of 18 d (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 d (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

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 in Figure 4.

		Prespawning migration			Tributary spawning			Postspawning		
Fish ID	Capture location	Date migration started	Total kilometers	Total days	Tributary	Estimated spawning date	Date exited	Last live location	Last live contact date	Fate
1	Nevada Spring Creek	16 May 2011	7.6	16	Wasson Creek	2 Jun	4 Jun	Nevada Spring Creek	19 Jul	Unknown
2	Nevada Creek	15 May 2012	10.8	2	Wasson Creek	24 May	10 Jun	Nevada Spring Creek	8 Jun	Unknown
3	Nevada Creek	26 Apr 2012	11.3	11	Wasson Creek	17 May	27 May	Nevada Creek	29 May	Heron Predation
4	Nevada Spring Creek	12 May 2011	14.5	7	Wasson Creek	28 May	1 Jun	Wasson Creek	27 Jun	Mortality
5	Nevada Creek	10 May 2012	11.6	1	Wasson Creek	15 May	22 May	Blackfoot River	27 Aug	Alive
6	Nevada Creek	12 May 2011	12.1	10	Wasson Creek	29 May	1 Jun	Wasson Creek	16 Jun	Mortality
7	Nevada Creek	5 May 2012	14.6	4	Wasson Creek	15 May	24 May	Nevada Creek	27 Aug	Alive
8	Nevada Creek	7 May 2012	13.4	2	Wasson Creek	14 May	10 Jun	Nevada Creek	27 Aug	Alive
9	Nevada Creek	26 Apr 2012	12.7	19	Wasson Creek	23 May	27 May	Nevada Creek	29 May	Heron Predation
10	Nevada Spring Creek	11 May 2011	16.9	6.5	Wasson Creek	26 May	1 Jun	Wasson Creek	25 Jul	Alive
11	Nevada Creek	13 May 2011	63.1	25	Arrastra Creek	7 Jun	10 Jun	Blackfoot River	25 Jul	Alive
12	Nevada Creek	24 Apr 2012	44.9	11	West Arrastra Creek	1 May	7 Jul	Blackfoot River	23 Aug	Alive
13	Nevada Creek	24 Apr 2011	49.9	27	Moose Creek	30 May	2 Jun	Blackfoot River	12 Jun	Unknown
14	Nevada Creek	9 May 2012	53.8	14	Moose Creek	30 May	19 Jul	Blackfoot River	23 Aug	Alive

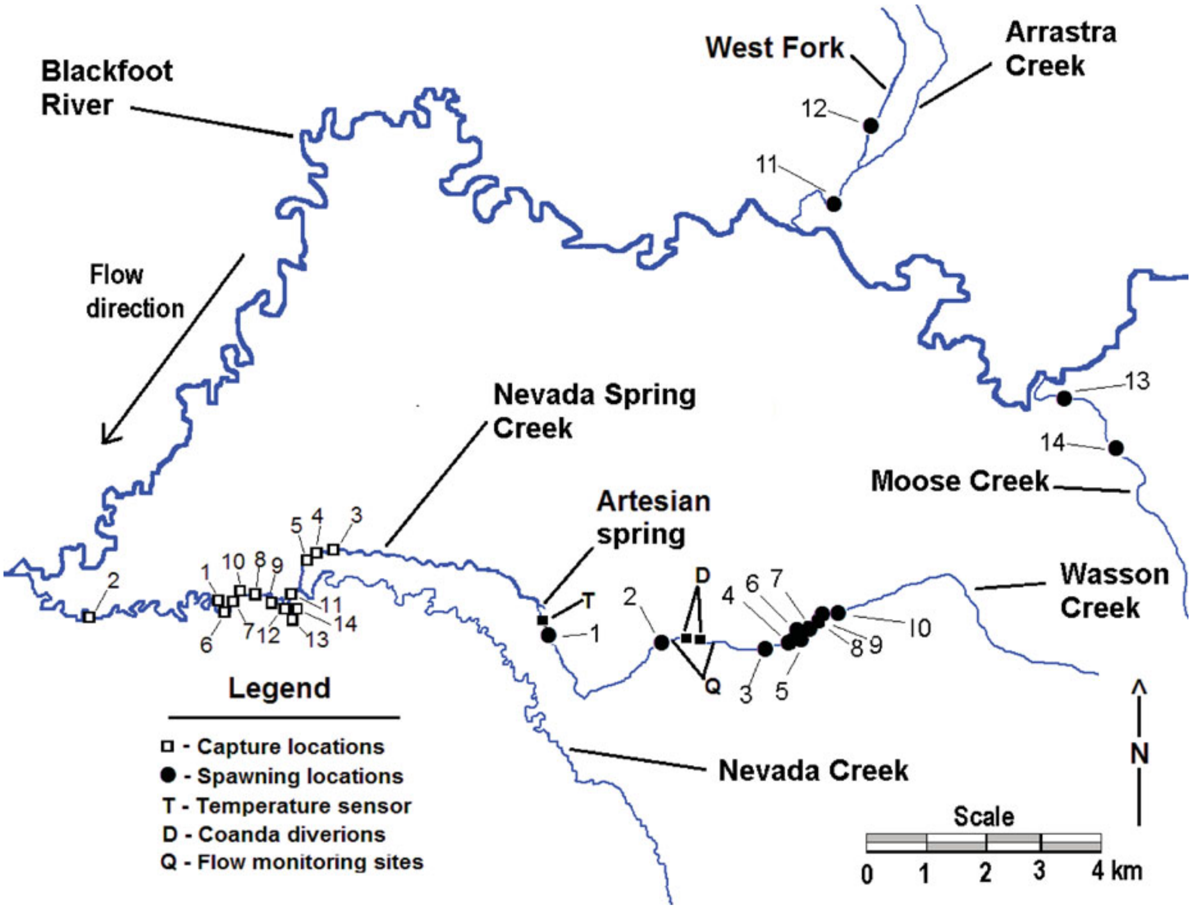

FIGURE 4. Capture locations (squares) and spawning locations (black circles) for 14 migratory Westslope Cutthroat Trout. The numbers for spawning locations relate to summaries of individual fish movements on Table 2.

($P = 0.002$). 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; $P = 0.02$).

When last contacted (Table 2), two postspawning Wasson Creek fish died in Wasson Creek (numbers 4 and 6), two (numbers 3 and 9) were killed by great blue heron *Ardea herodias* based on tags traced to a rookery, one (number 10) remained in Wasson Creek, two exited to Nevada Spring Creek (numbers 1 and 2), two exited to Nevada Creek (numbers 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 to 81.4 rkm when last contacted. The Moose Creek spawner (number 14) that showed the longest prespawning movement (53.8 rkm) also showed the longest postspawning 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.

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 3). Three spawners ascended the Coandas between 21 and 25 May 2011 at flows ranging from 0.25 to 0.28 m^3/s. Five spawners ascended the diversions between 10 and 19 May 2012 at flows ranging from 0.14 to 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 4). 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 3). Under these conditions, the Coanda fish screens showed no observed effect on upstream 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 (Bernhardt et al. 2005;

Roni 2005; Baldigo et al. 2008), and very few, if any, published studies document the response of migratory native trout to multiscale 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).

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; Schrank and Rahel 2004; Petty et al. 2012; this study). Compared with 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/30 m) 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/30 m (range, 4.3–21) downstream of the diversions between 2004 and 2012.

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 water bodies 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, 10 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 Westslope Cutthroat Trout was actually observed successfully ascending the Coanda diversion (Figure 2 [bottom]). 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 limit our ability to fully interpret these results, varied movement 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 pretreatment demonstrate the added benefits 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 oversummer. 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 to severe winter conditions (i.e., super-cooled [<0°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 (Cunjak 1996; Jakober et al. 1998; Brown et al. 2011). In the case of Nevada Spring Creek, the artesian spring at the head of this creek flows at a constant annual temperature of 6.7–7.8°C, cooling the main stem 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 et al. 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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REFERENCES

Aitken, G. 1997. Restoring trout waters in the West: Big Blackfoot River of Montana. Pages 402–424 *in* J. E. Williams, C. A. Wood, and M. P. Dombeck, editors. Watershed restoration: principles and practices. American Fisheries Society, Bethesda, Maryland.

Baldigo, B. P., D. R. Warren, A. G. Ernst, and C. I. Mulvihill. 2008. Response of fish populations to natural channel design restoration in streams of the Catskill Mountains, New York. North American Journal of Fisheries Management 28:954–969.

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society, Monograph 6, Bethesda, Maryland.

Behnke, R. J. 2002. Trout and salmon of North America. Free Press, New York.

Bernhardt, E. S., M. A. Palmer, J. D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G. M. Kondolf, P. S. Lake, R. Lave, J. L. Meyer, T. K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Synthesizing U.S. river restoration efforts. Science 308:636–637.

Brown, R. S., W. A. Hubert, and S. F. Daly. 2011. A primer on winter, ice, and fish: what fisheries biologists should know about winter ice processes and stream-dwelling fish. Fisheries 36:8–26.

Brown, R. S., and W. C. Mackay. 1995. Spawning ecology of Cutthroat Trout (*Oncorhynchus clarki*) in the Ram River, Alberta. Canadian Journal of Fisheries and Aquatic Sciences 52:983–992.

Cook, N., F. J. Rahel, and W. A. Hubert. 2010. Persistence of Colorado River Cutthroat Trout populations in isolated headwater streams of Wyoming. Transactions of the American Fisheries Society 139:1500–1510.

Cunjak, R. A. 1996. Winter habitat of selected stream fishes and potential impacts from land-use activity. Canadian Journal of Fisheries and Aquatic Sciences 53(Supplement 1):267–282.

DEQ (Department of Environmental Quality). 2007. Draft middle Blackfoot–Nevada Creek total maximum daily loads and water quality improvement plan: sediment, nutrient, trace metal and temperature TMDLs. Montana DEQ, Helena.

Drinan, D. P., S. T. Kalinowski, N. V. Vu, B. B. Shepard, C. C. Muhlfeld, and M. R. Campbell. 2011. Genetic variation in Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi:* implications for conservation. Conservation Genetics 12:1513–1523.

Fausch, K. D., B. E. Rieman, J. B. Dunham, M. K. Young, and D. P. Peterson. 2009. Invasion versus isolation: trade-offs in managing native salmonids with barriers to upstream movement. Conservation Biology 23:859–870.

Gale, S. B., A. V. Zale, and C. G. Clancy. 2008. Effectiveness of fish screens to prevent entrainment of Westslope Cutthroat Trout into irrigation canals. North American Journal of Fisheries Management 28:1541–1553.

Jakober, M. J., T. E. McMahon, R. F. Thurow, and C. G. Clancy. 1998. Role of stream ice on fall and winter movements and habitat use by Bull Trout and Cutthroat Trout in Montana headwater streams. Transactions of the American Fisheries Society 127:223–235.

McIntyre, J. D., and B. E. Rieman. 1995. Westslope Cuttthroat Trout. U.S. Forest Service General Technical Report RM 256:1–15.

Meehan, W. R., editor. 1991. Influences of forest and rangeland managment on salmonid fishes and their habitats. American Fisheries Society, Special Publication 19, Bethesda, Maryland.

MFWP (Montana Fish, Wildlife and Parks). 2012. Montana angler pressure estimates. MFWP, Bozeman.

Moyle, P. B., and J. A. Israel. 2005. Untested assumptions: effectiveness of screening diversions for conservation of fish populations. Fisheries 30(5):20–28.

Muhlfeld, C. C., S. T. Kalinowski, T. E. McMahon, M. L. Taper, S. Painter, R. F. Leary, and F. W. Allendorf. 2009a. Hybridization rapidly reduces fitness of a native trout in the wild. Biology Letters 5:328–331.

Muhlfeld, C. C., T. E. McMahon, D. Belcer, and J. L. Kershner. 2009b. Spatial and temporal spawning dynamics of native Westslope Cutthroat Trout, *Oncorhynchus clarkii lewisi,* introduced Rainbow Trout, *Oncorhynchus mykiss,* and their hybrids. Canadian Journal of Fisheries and Aquatic Sciences 66:1153–1168.

Nehlsen, W., J. E. Williams, and J. A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. Fisheries 16(2):4–21.

Peters, D., and R. Spoon. 1989. Preliminary fisheries inventory of the Big Blackfoot River. Montana Department of Fish, Wildlife and Parks, Missoula.

Petty, J. T., J. L. Hansbarger, B. M. Huntsman, and P. M. Mazik. 2012. Brook Trout movement in response to temperature, flow, and thermal refugia within a complex Appalachian riverscape. Transactions of the American Fisheries Society 141:1060–1073.

Pierce, R., R. Aasheim, and C. Podner. 2005. An integrated stream restoration and native fish conservation strategy for the Blackfoot River basin. Montana Fish, Wildlife and Parks, Missoula.

Pierce, R., R. Aasheim, and C. Podner. 2007. Fluvial Westslope Cutthroat Trout movements and restoration relationships in the upper Blackfoot basin, Montana. Intermountain Journal of Sciences 13:72–85.

Pierce, R., M. Davidson, and C. Podner. 2012. Spawning behavior of Mountain Whitefish and co-occurrence of *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 141:720–730.

Pierce, R., and D. Peters. 1990. Aquatic investigations in the middle Blackfoot River, Nevada Creek and Nevada Spring Creek corridor. Montana Fish, Wildlife and Parks, Missoula.

Pierce, R., and C. Podner. 2013. Fisheries investigations in the Blackfoot River basin 2011–2012. Montana Fish, Wildlife and Parks, Missoula.

Pierce, R., C. Podner, and K. Carim. 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 142:68–81.

Pierce, R., C. Podner, M. Davidson, and E. R. Vincent. 2009. Correlation of fluvial Rainbow Trout spawning life history with severity of infection by *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 138:251–263.

Pierce, R., C. Podner, and J. McFee. 2002. Blackfoot River fisheries inventory, restoration and monitoring report for 2001. Montana Fish, Wildlife and Parks, Missoula.

R Development Core Team. 2012. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna. Available: www.R-project.org/. (June 2013).

Rieman, B. E., and J. B. Dunham. 2000. Metapopulations and salmonids: a synthesis of life history patterns and empirical observations. Ecology of Freshwater Fish 9:51–64.

Roberts, J. J., and F. J. Rahel. 2008. Irrigation canals as sink habitat for trout and other fishes in a Wyoming drainage. Transactions of the American Fisheries Society 137:951–961.

Roni, P., editor. 2005. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Rosenfeld, J. S., S. Macdonald, D. Foster, S. Amrhein, B. Bales, T. Williams, F. Race, and T. Livingstone. 2002. Importance of small streams as rearing habitat for Coastal Cutthroat Trout. North American Journal of Fisheries Management 22:177–187.

Schmetterling, D. A. 2000. Redd characteristics of fluvial Westslope Cutthroat Trout in four tributaries to the Blackfoot River, Montana. North American Journal of Fisheries Management 20:776–783.

Schmetterling, D. A. 2001. Seasonal movements of fluvial Westslope Cutthroat Trout in the Blackfoot River drainage, Montana. North American Journal of Fisheries Management 21:507–520.

Schrank, A. J., and F. J. Rahel. 2004. Movement patterns in inland Cutthroat Trout (*Oncorhynchus clarki utah*): management and conservation implications. Canadian Journal of Fisheries and Aquatic Sciences 61:1528–1537.

Sestrich, C. M., T. E. McMahon, and M. K. Young. 2011. Influence of fire on native and nonnative salmonid populations and habitat in a western Montana basin. Transactions of the American Fisheries Society 140:136–146.

Shepard, B. B., B. E. May, and W. Urie. 2005. Status and conservation of Westslope Cutthroat Trout within the western United States. North American Journal of Fisheries Management 25:1426–1440.

Shepard, B. B., B. Sanborn, L. Ulmer, and D. C. Lee. 1997. Status and risk of extinction for Westslope Cutthroat Trout in the upper Missouri River basin, Montana. North American Journal of Fisheries Management 17:1158–1172.

Simpson, W. G., and K. G. Ostrand. 2012. Effects of entrainment and bypass at screened irrigation canals on juvenile steelhead. Transactions of the American Fisheries Society 141:599–609.

Strahler, A. N. 1957. Quantitative analysis of watershed geomorphology. Transactions, American Geophysical Union 38:913–920.

Swanberg, T. R. 1997. Movements of and habitat use by fluvial Bull Trout in the Blackfoot River, Montana. Transactions of the American Fisheries Society 126:735–746.

Swanberg, T. R., D. A. Schmetterling, and D. H. McEvoy. 1999. Comparison of surgical staples and silk sutures for closing incisions in Rainbow Trout. North American Journal of Fisheries Management 19:215–218.

Thurow, R. F., D. C. Lee, and B. E. Rieman. 1997. Distribution and status of seven native salmonids in the interior Columbia River basin and portions of the Klamath River and Great basins. North American Journal of Fisheries Management 17:1094–1110.

USFWS (U.S. Fish and Wildlife Service). 2010. Endangered and threatened wildlife and plants; revised designation of critical habitat for Bull Trout in the coterminous United States; final rule. Federal Register 75:200(18 October 2010):63898–64070.

USGS (U.S. Geological Survey). 2013. Current conditions for Montana: streamflow—gauging station 1235500 provisional unpublished data. USGS, Reston, Virginia. Available: waterdata.usgs.gov/MT/nwis/current/?type=flow. (June 2013).

Wahl, T. L. 2001. Hydraulic performance of Coanda-effect screens. Journal of Hydraulic Engineering 127:480–488.

Wahl, T. L. 2003. Design guidance for Coanda-effect screens. U.S. Bureau of Reclamation, Water Resources Research Laboratory, Report R-03-03, Denver.

Winter, J. 1996. Advances in underwater biotelemetry. Pages 555–590 *in* B. R. Murphy and D. W. Willis, editors. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

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ARTICLE

Instream Habitat Restoration and Stream Temperature Reduction in a Whirling Disease-Positive Spring Creek in the Blackfoot River Basin, Montana

Ron Pierce* and Craig Podner
Montana Fish, Wildlife and Parks, 3201 Spurgin Road, Missoula, Montana 59804, USA

Laurie Marczak
Division of Biological Science, University of Montana, 32 Campus Drive, Missoula, Montana 59812, USA

Leslie Jones
Division of Biological Science, University of Montana, 32 Campus Drive, Missoula, Montana 59812, USA; and U.S. Geological Survey, Northern Rocky Mountain Science Center Field Station, Glacier National Park, West Glacier, Montana 59936, USA

Abstract
Anthropogenic warming of stream temperature and the presence of exotic diseases such as whirling disease are both contemporary threats to coldwater salmonids across western North America. We examined stream temperature reduction over a 15-year prerestoration and postrestoration period and the severity of *Myxobolus cerebralis* infection (agent of whirling disease) over a 7-year prerestoration and postrestoration period in Kleinschmidt Creek, a fully reconstructed spring creek in the Blackfoot River basin of western Montana. Stream restoration increased channel length by 36% and reduced the wetted surface area by 69% by narrowing and renaturalizing the channel. Following channel restoration, average maximum daily summer stream temperatures decreased from 15.7°C to 12.5°C, average daily temperature decreased from 11.2°C to 10.0°C, and the range of daily temperatures narrowed by 3.3°C. Despite large changes in channel morphology and reductions in summer stream temperature, the prevalence and severity of *M. cerebralis* infection for hatchery Rainbow Trout *Oncorhynchus mykiss* remained high (98–100% test fish with grade > 3 infection) versus minimal for hatchery Brown Trout *Salmo trutta* (2% of test fish with grade-1 infection). This study shows channel renaturalization can reduce summer stream temperatures in small low-elevation, groundwater-dominated streams in the Blackfoot basin to levels more suitable to native trout. However, because of continuous high infections associated with groundwater-dominated systems, the restoration of Kleinschmidt Creek favors brown trout *Salmo trutta* given their innate resistance to the parasite and the higher relative susceptibility of other salmonids.

Degradation of salmonid habitat historically involved physical alterations of streams and rivers from land use activities such as channel degradation, dewatering, and overgrazing (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) have not only elevated the overall need for aquatic restoration, but also the need to refine, implement, and evaluate specific restoration activities associated with these conditions (Rieman et al. 2007; Williams et al. 2009; Pierce et al. 2014). Despite the pressing need for applied studies of this nature, few, if any, long-term field investigations link restoration to stream temperature reduction (Poole and Berman 2001; Williams et al. 2009).

Ambient climate (i.e., air temperature) contributes to variability in stream temperatures through heat exchange near the surface of the water (Meisner et al. 1988; Mote 2006). However, groundwater-dominated systems are thermally regulated by groundwater inflow and are therefore less affected by seasonal, elevational, and climatic conditions than basin-fed streams (Melinna et al. 2002). Groundwater-induced streams are thereby cooler during summer and warmer during winter, especially near the sources of groundwater inflows (Meisner et al. 1988; Cassie 2006; Pierce et al. 2012). As an example of this, stream 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°C in winter to >23°C during summer in nearby basin-fed streams (Pierce et al. 2002, 2012, 2013). Given the moderating effects of groundwater, stream improvements that renaturalize channels and reduce stream temperatures during the peak of summer may prove increasingly important given projections of climate warming, especially in low-elevation streams where native trout are most at risk (Meisner et al. 1988; Rieman et al. 2007; Williams et al. 2009).

Similar to restoration-induced cooling, the known potential of restoration to moderate the effects of whirling disease is currently limited by localized and short-term field studies (Hansen and Budy 2011; Thompson 2011). Salmonid whirling disease is a parasitic infection cause by the exotic myxosporean *Myxobolus cerebralis*, which is native to the Eurasian continent and arrived on the North American continent in the 1950s where it spread rapidly (Bartholomew and Reno 2002). Clinical signs of whirling disease include blacktail, radical whirling (tail chasing) behavior and skeletal deformities (MacConnell and Vincent 2002). Whirling disease has been associated with population declines of Rainbow Trout *Oncorhynchus mykiss* in certain Montana and Colorado rivers (Nehring and Walker 1996; Vincent 1996; Granath et al. 2007; McMahon et al. 2010). Whirling disease may be especially harmful in groundwater-dominated streams where several environmental factors are conducive to the proliferation of *M. cerebralis* and its obligate aquatic worm host *Tubifex tubifex* (Burckhardt and Hubert 2005; Neudecker et al. 2012; Pierce et al. 2012).

Habitat conditions favorable for the proliferation of *M. cerebralis* and *T. tubifex* generally include (1) high stream temperatures (MacConnell and Vincent 2002; Hansen and Budy 2011), (2) fine sediment (Krueger et al. 2006; Anlauf and Moffitt 2008; McGinnis and Kerans 2013), and (3) elevated nutrient concentrations (Kaeser et al. 2006), all of which can increase with the anthropogenic degradation of streams (Zendt and Bergersen 2000; Hansen and Budy 2011; McGinnis and Kerans 2013). As a result, the ability of restoration to offset whirling disease seems to require a reduction of one or more of these conditions, as well as an increasing recognition that mediating environmental conditions tied to whirling disease can vary greatly between basin-fed (Anlauf and Moffitt 2008; Hansen and Budy 2011; McGinnis and Kerans 2013) and groundwater-fed streams (Burckhardt and Hubert 2005; Neudecker et al. 2012; Pierce et al. 2012).

Myxobolus cerebralis has a complex, two-host life cycle and can affect most salmonids, which include trout, whitefish, and salmon (Bartholomew and Wilson 2002). Susceptibility to the pathogen depends on 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). Infectious conditions often vary by season (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 triactinomyxons (TAMs; El-Matbouli et al. 1999; De La Hoz Franco and Budy 2004; Kerans et al. 2005). However, recent studies show high *M. cerebralis* infection can be continuous across seasons and can occur at much lower temperatures (<5°C) with the moderating influence of groundwater inflows (Neudecker et al. 2012; Pierce et al. 2012). As an example of this elevated infections in groundwater environment, a prior study in the Blackfoot River basin found a majority of Mountain Whitefish *Prosopium williamsoni* were infected in early spring in the groundwater-induced upper Blackfoot River compared with no concurrent infection a river segment with very little groundwater influence (Pierce et al. 2012). Likewise, a second study found season-long higher infection rates in groundwater-dominated streams (spring creeks) versus receiving waters (Neudecker et al. 2012). High TAM release in groundwater-dominated streams (e.g., spring creeks) relates largely to stable stream temperatures (Neudecker et al. 2012; Pierce et al. 2012), especially where low channel gradients, lack of flushing flows, and high sediment loads are present (Hiner and Moffitt 2002; Hubert et al. 2002; Neudecker et al. 2012).

In the Blackfoot Basin of Western Montana, we monitored summer stream temperatures associated with the restoration of Kleinschmidt Creek over a 15-year before–after period (1998–2012), as well as the prevalence and severity of *M. cerebralis* infection during a 7-year before–after period (1998–2004). This study expands on a prior study that broadly describes restoration techniques in the Blackfoot basin, including changes in general channel morphology and increases in wild trout abundance in Kleinschmidt Creek (Pierce et al. 2013). The goal of this study is to examine the potential of restoration to alter water temperature in a groundwater-dominated stream and to clarify whether comprehensive stream restoration can mediate *M. cerebralis,* which is now present in many low-elevation streams of the Blackfoot basin (Pierce et al. 2009, 2012; Neudecker et al. 2012). Our specific study objectives were to (1) examine summer stream temperature changes after the full reconstruction of Kleinschmidt Creek, and (2) identify the prevalence and severity of *M. cerebralis* infection before and after stream restoration.

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 a low of 0.26 m^3/s during winter and spring to a high of about 0.42 m^3/s during midsummer months (Pierce and Podner 2006). Although Kleinschmidt Creek receives basin-fed runoff upstream of stream kilometer (skm) 3.2, approximately 90% of summer stream flows are generated by groundwater inflows from an alluvial aquifer, most of which surfaces between skm 1.6 and 3.2 (Pierce et al. 2002; Pierce and Podner 2006). To examine stream temperature reduction, we monitored stream temperatures in the North Fork of the Blackfoot River at U.S. Geological Survey (USGS) gauge station 12338300 and treated this as a control site in the study (Figure 1). Although the lower North Fork of the Blackfoot River is much larger and has a snow-fed and basin-fed hydrograph during the spring runoff, like Kleinschmidt Creek, the lower 12 skm of the North Fork of the Blackfoot River receives >80% of late summer (August–September) base flow (mean discharge = 5.8 m^3/s) from groundwater inflows (Montana Department of Natural Resource Conservation and USGS, unpublished data) and several small spring creeks entering the North Fork of the Blackfoot River between skm 8 and 10.

To reestablish natural features of a relatively deep and narrow channel (Table 1; Figure 2), the reconstruction of Kleinschmidt Creek was completed in the autumn of 2001

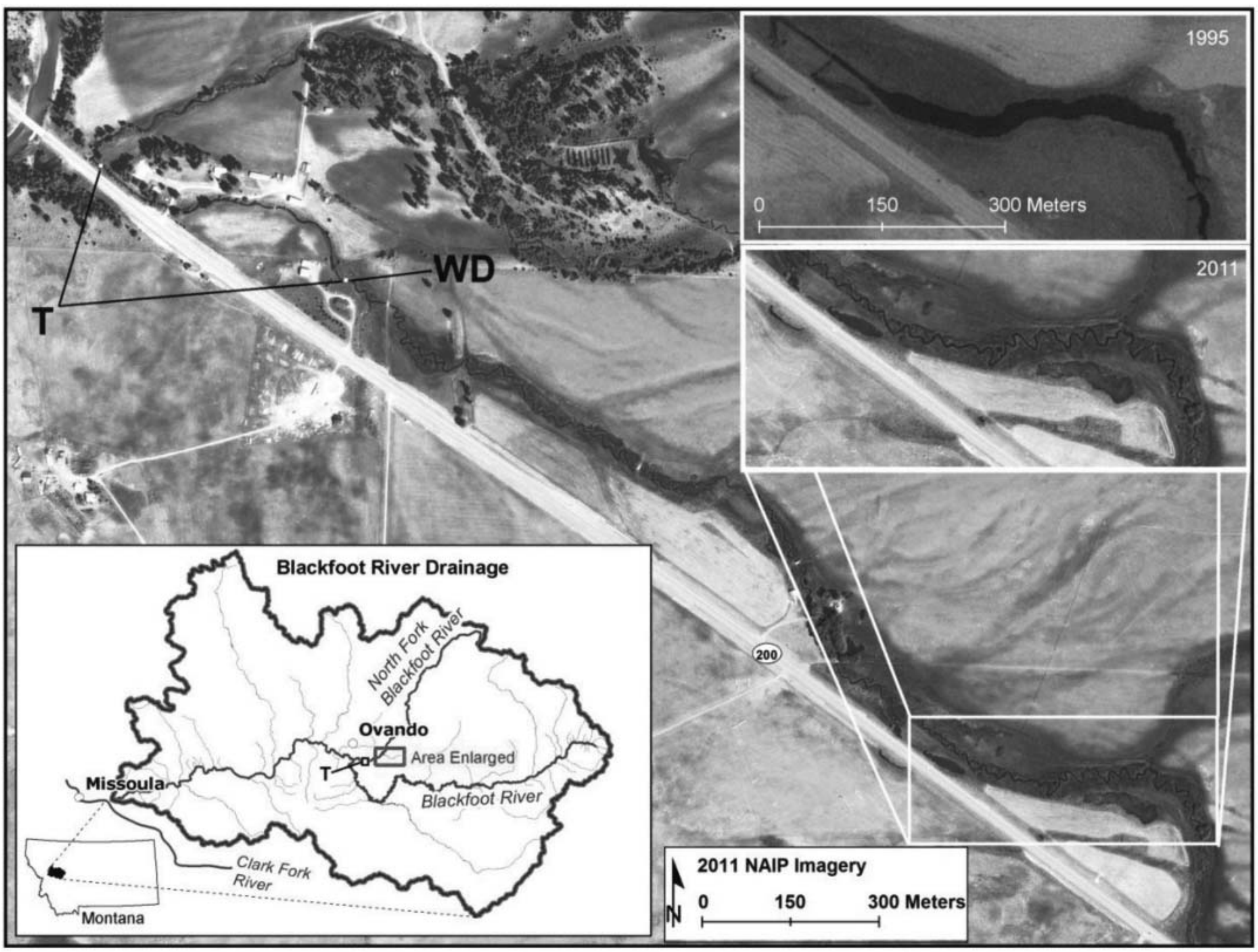

FIGURE 1. Map of the Blackfoot River drainage, showing the location of the stream temperature control site (T; at USGS station) on the North Fork of the Blackfoot River and the Kleinschmidt Creek project area (red box), which is expanded as aerial photographs, showing the stream temperature (T) and whirling disease (WD) monitoring sites. The enlarged aerials of the project area are before (1996) and after (2011) restoration aerials of the uppermost project area.

TABLE 1. Channel metrics before and after restoration modified from Pierce et al. 2013.

Period	Channel length (km)	Sinuosity	Mean wetted width (m)	Wetted surface area (f)	Number of pools/100 m	Mean maximum pool depth (m)	Number of woody stems/100 m	Percent fine sediment (<2 mm)
Before	2.5	1.1	12.6	3.2	0.6	0.7	0.03	30[a]
After	3.4	1.5	2.9	1.0	4.5	1.0	6.40	21

[a]Data from Neudecker et al. 2012.

(Pierce et al. 2013). Channel renaturalization increased channel length by 36% and reduced the wetted surface area of the channel by 69% by narrowing the channel. In addition, stream renaturalization recreated pools and riffles, increased habitat diversity, reduced instream sediment levels, fenced livestock from the riparian corridor, reestablished vegetation and secured instream flows.

FIGURE 2. Photographs of Kleinschmidt Creek before (2001; top panel), showing straightened and over-widened section of channel, and after (summer 2013; lower panel) showing the same location 11 years after channel restoration. [Figure available online in color.]

Kleinschmidt Creek supports a mixed community of salmonids, though Brown Trout *Salmo trutta* compose about 92% of the salmonid community (Pierce et al. 2013); other salmonids include, in order of decreasing density, Brook Trout *Salvelinus fontinalis*, native Westslope Cutthroat Trout *O. clarkii lewisi*, native Bull Trout *Salvelinus confluentus*, and Rainbow Trout. Prior to the restoration of Kleinschmidt Creek, the exotic parasite *M. cerebralis* was already present at high infection levels (Neudecker et al. 2012). Rainbow Trout are highly susceptible to whirling disease (MacConnell and Vincent 2002), whereas Brook Trout, Westslope Cutthroat Trout, and Bull Trout possess intermediate (i.e., partial) resistance to whirling disease (MacConnell and Vincent 2002). The nonnative Brown Trout is naturally much more resistant to the parasite, given their coevolution on the Eurasian continent (Bartholomew and Reno 2002).

METHODS

Stream temperature change.—To examine summer stream temperature change, we monitored stream temperatures near the mouth of Kleinschmidt Creek and at skm 4.2 on the lower North Fork of the Blackfoot River between 1 June and 1 October (Figure 2). Concurrent daily monitoring included 3 years prerestoration (1998, 1999, and 2001) and 5 years postrestoration (2002, 2004, 2010, 2012, and 2013) on both Kleinschmidt Creek and the North Fork of the Blackfoot River. For both Kleinschmidt Creek and the North Fork of the Blackfoot River site, we recorded temperatures continuously at 48–72 min intervals using Onset digital thermograph (Onset Computer Corporation, Pocasset, Massachusetts; accuracy = 0.2°C). During the monitoring periods, daily stream temperature statistics (maximum, mean and minimum) from Kleinschmidt Creek were paired with the same daily stream temperature statistics from the North Fork of the Blackfoot River control site. Concurrent daily air temperature statistics (maximum, mean, and minimum) were also extracted for each site before and after periods using TopoWx daily climatological surfaces (TopoWx 2014; J. W. Oyler, University of Montana, and colleagues, unpublished data), except for the year 2013, when climatological data were unavailable.

We used a before–after, control–impact (BACI) design to compare the mean, maximum and range of daily stream temperatures for the control (North Fork of the Blackfoot River) and treatment (Kleinschmidt Creek) sites and averaged each temperature metric over the 3 years prerestoration and 11-years postrestoration periods. We selected these stream temperature parameters because they are commonly used in both salmonid and whirling disease studies (e.g., Dunham et al. 2003; Kerans et al. 2005; Pierce et al. 2012). A single-factor ANOVA with before–after restoration periods as a fixed factor was used to test maximum and mean stream temperature change for Kleinschmidt Creek. A two-factor ANOVA with before–after restoration periods and site as fixed factors was used to test for differences in air temperatures, as well as stream temperatures between the control and treatment sites. All statistical analyses were conducted at $\alpha = 0.05$ in R version 2.15.0 software (R Development Core Team 2013).

Whirling disease testing in Kleinschmidt Creek.—Similar to previous whirling disease studies in the Blackfoot basin (Pierce et al. 2009, 2012; Neudecker et al. 2012), we monitored the prevalence (percent infected) and severity of *M. cerebralis* infection before and after restoration in lower Kleinschmidt Creek using sentinel cage exposures with hatchery Rainbow Trout fry (diploid age-0 cohorts) as surrogates for infection. Test fish ranged in age from 66 to 151 d posthatch, and total lengths ranged from 34 to 53 mm (Table 2). Five exposure trials were undertaken between late winter (March 15) and early summer (July 11) and spanned a 7-year period, which included 2 years prerestoration (1998–1999) and 3 years postrestoration (2002–2004). The March to July span of exposure trials in this study overlaps with hatching, emergence, and early rearing periods (i.e., periods of increased disease vulnerability) for fall-spawning Brown Trout, Brook Trout, Bull Trout, and Mountain Whitefish, as well as spring-spawning Rainbow Trout and Westslope Cutthroat Trout (Behnke 1992; Ryce et al. 2005; Neudecker et al. 2012; Pierce et al. 2009, 2012). To examine variation of infection among three trout species present in Kleinschmidt Creek, we also completed side-by-side postrestoration exposure trials of age-0 hatchery Brown Trout, Brook Trout, and Rainbow Trout in March 2002 (Table 2).

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 et al. 2004). Since 1999, this scale has categorically ranked the severity of infection into one of six qualitative groups: 0 = no infection, 1 = minimal, 2 = mild, 3 = moderate, 4 = high, and 5 = severe. Prior to 1999, grade-5 infections were not distinguished from grade 4 (E. Ryce, Montana Fish, Wildlife and Parks, personal communication). As in prior studies, the severity of infection for each exposure trial was considered high if >50% of exposed trout had histological scores of grades >3 severity (Pierce et al. 2009, 2012; Neudecker et al. 2012). At severity grades >3, *M. cerebralis* digests and destroys cartilage of young fish, which causes inflammation and lesions in the spine and cranium, resulting in skeletal damage as severity of infection increases. This leaves young fish crippled, weak, and unable to feed or evade predators, all of which ultimately elevate mortality (Hedrick et al. 1999; MacConnell and Vincent 2002; Ryce et al. 2004). According to Ryce et al. (2005), age-0 Rainbow Trout are most susceptible if exposed to *M. cerebralis* at <63 d posthatch and at <40 mm TL, after which time the effects of disease are reduced through increased resistance.

RESULTS

Air and Stream Temperature Change

To ensure that changes in stream temperatures before and after restoration were not driven by climatological differences between sites, we performed the same BACI two-factor

TABLE 2. Sentinel exposure results for three salmonid species in lower Kleinschmidt Creek before (1998-99) and after restoration (2002–2004).

Species	Exposure date	Mean daily water temperature (°C)	Number histologically examined	Individual histological scores						Group scores (%)		Age of fish (d) at exposure	Length of fish (mm) at exposure
				0	1	2	3	4	5	Infected	Grade > 3		
Rainbow Trout	Jul 1–11, 1998	12.5	48	5	3	7	13	20	[a]	90	69	75	37
Rainbow Trout	Jul 1–11, 1999	11.0	50	5	3	3	4	18	17	90	78	91	43
Rainbow Trout	Mar 15–25, 2002	5.8	48	0	0	0	3	17	28	100	100	71	45
Brown Trout	Mar 15–25, 2002	5.8	43	42	1	0	0	0	0	2	0	107	37
Brook Trout	Mar 15–25, 2002	5.8	50	2	1	3	4	18	22	96	88	115	38
Rainbow Trout	Apr 23–May 3, 2003	—	50	1	0	0	0	9	40	98	98	151	53
Rainbow Trout	Jun 20–30, 2003	9.0	49	0	0	0	0	3	46	100	100	111	44

[a]Not applicable.

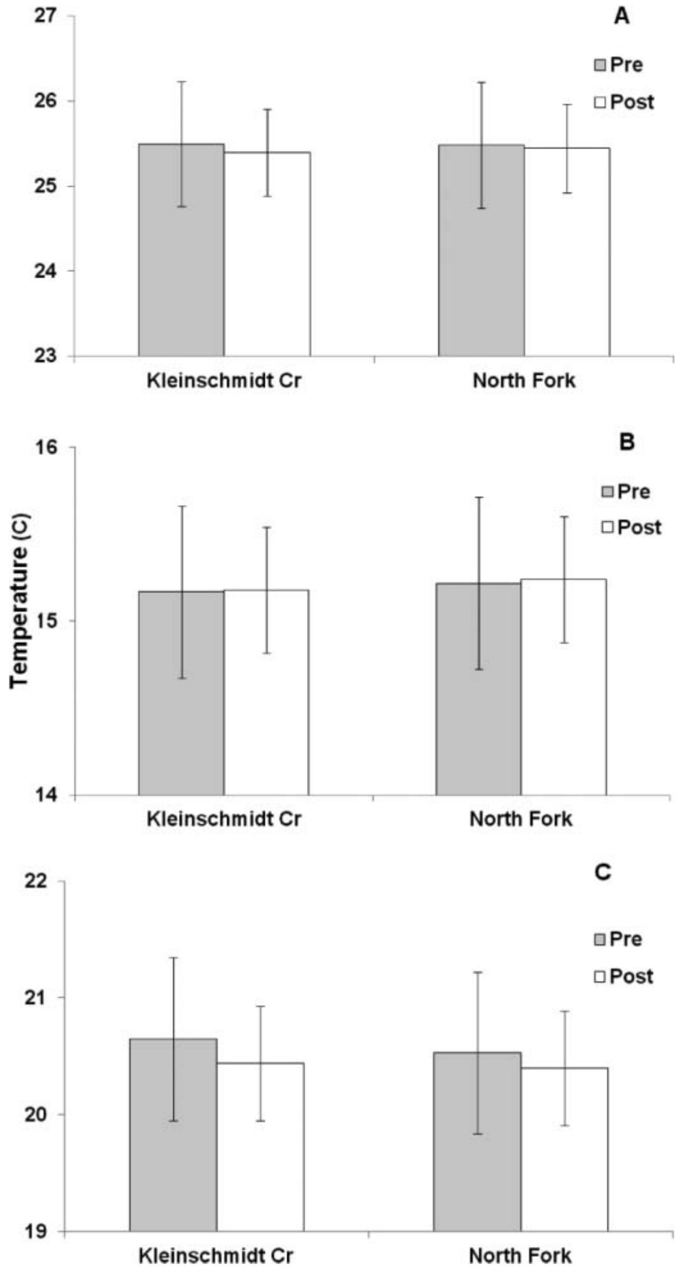

FIGURE 3. Prerestoration and postrestoration air temperatures for Kleinschmidt Creek (treatment site) and the North Fork of the Blackfoot River (control site): **(A)** average maximum daily temperatures, **(B)** mean daily temperatures, and **(C)** average daily range of temperatures.

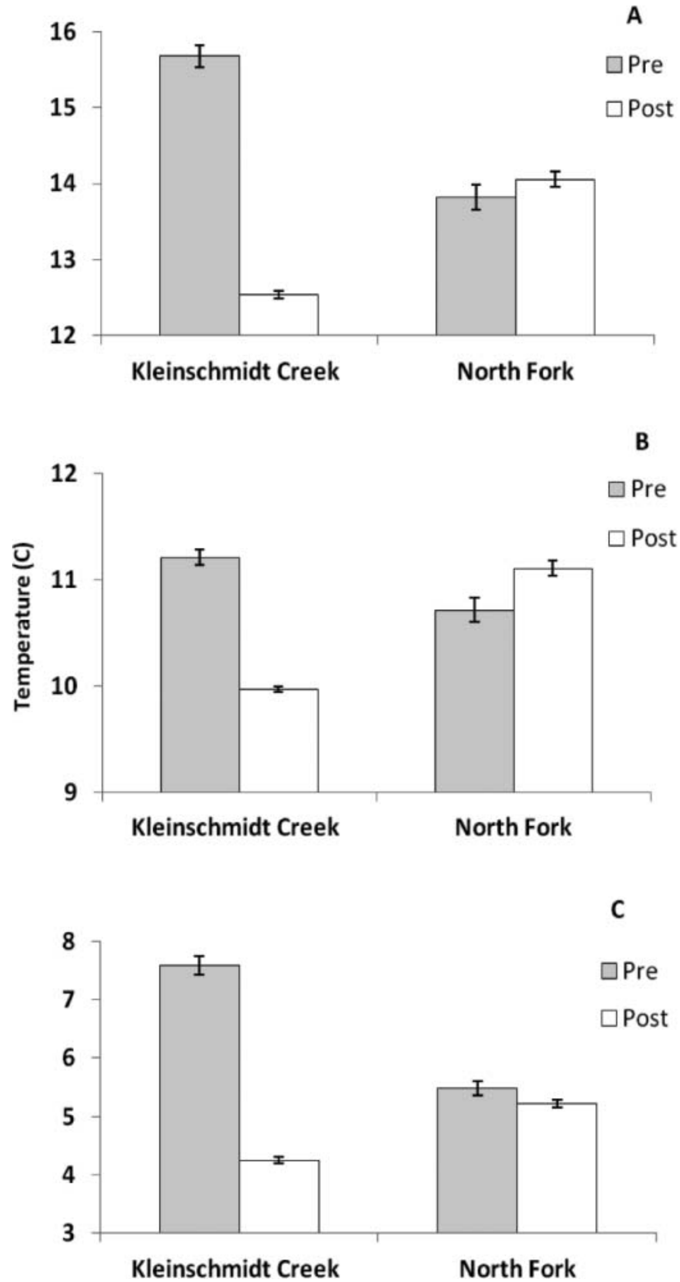

FIGURE 4. Prerestoration and postrestoration water temperatures for Kleinschmidt Creek (treatment site) and the North Fork of the Blackfoot River (control site): **(A)** average maximum daily temperatures, **(B)** mean daily temperatures, and **(C)** average daily range of temperatures.

ANOVA using TopoWx air temperature data as performed in the stream temperature analyses. We found no statistically significant before–after differences between control and treatment sites in average daily mean air temperatures ($F_{1,\ 1226} = 0.0015$, $P = 0.9691$, Figure 3A), average daily maximum air temperatures ($F_{1,\ 1226} = 0.0052$, $P = 0.9427$, Figure 3B), and the range in daily maximum and minimum air temperatures ($F_{1,\ 1226} = 0.0186$, $P = 0.8916$, Figure 3C).

Prior to restoration, daily maximum stream temperatures in Kleinschmidt Creek averaged 15.7°C in contrast to 13.8°C in the North Fork of the Blackfoot River (Figure 4A). During the 11-year postrestoration period (2002–2013), average daily maximum stream temperatures in Kleinschmidt Creek decreased to 12.5°C, significantly lower than the prerestoration (1998–2001) average of 15.7°C ($F_{1,\ 710} = 647.4$, $P < 0.0001$). More interestingly, postrestoration daily maximum stream temperatures in Kleinschmidt Creek were 1.5°C lower than temperatures at the North Fork of the Blackfoot River control site, where prerestoration averages were 1.9°C higher ($F_{1,\ 1420} = 220.9$, $P < 0.0001$; Figure 4A). This pattern of reduction was consistent for mean daily stream temperatures at the treatment site: prerestoration average = 11.2°C and postrestoration average = 10.0°C ($F_{1,\ 710} = 391.1$, $P < 0.0001$). In addition, average daily stream temperatures at the treatment site were 0.5°C higher (i.e., 11.2°C) than the control site prerestoration (i.e., 10.7°C) and 1.1°C lower postrestoration ($F_{1,\ 1420} = 125.1$, $P < 0.0001$; Figure 4B). Consistent with both trends in stream temperature reduction, the average range of daily stream temperatures (i.e., difference between daily maximum and minimum) in Kleinschmidt Creek declined 3.3°C from 7.6°C prerestoration to 4.3°C postrestoration. In comparison, differences at the North Fork of the Blackfoot River site were 8.3°C prerestoration and 8.6°C postrestoration, statistically different than those at the treatment site ($F_{1,\ 1420} = 243.7$, $P < 0.0001$; Figure 4C).

Sentinel Cage Exposures

Prior to restoration, 90% of the Rainbow Trout test fish were infected with *M. cerebralis*, of which most (69–78%) had high (grade, >3) severity of infection. Following restoration, the prevalence of infection increased to 93–100% of test fish, and the severity of high infection (grade, ≥3) increased to 96–100% during late winter to early summer exposures (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 Trout (100%) and Brook Trout (88%) versus minimal infection for Brown Trout (2% with grade 1). In addition to high infection, most Rainbow Trout examined exceeded the age (i.e., 63 d) and size (40 mm) at which resistance to whirling disease is conferred (Table 2; Ryce et al. 2005), which further demonstrates infection conditions during the March–July exposure period.

DISCUSSION

This study shows channel renaturalization can reduce summer stream temperatures in small low-elevation, groundwater-dominated streams. Despite full channel restoration and significant reductions in summer stream temperatures, 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, it appears unlikely that salmonids other than Brown Trout (due to their increased resistance to *M. cerebralis*) can reproduce and rear successfully in Kleinschmidt Creek due to infectious conditions that overlap with hatching and early rearing windows for spring and fall spawners. Though infectious conditions clearly favor resident brown trout, a reduction in stream temperature may favor age-1 and older native trout in Kleinschmidt Creek, as well as improve thermal habitat in receiving waters of the North Fork of the Blackfoot River, where infections are low (Pierce et al. 2009, 2012; Neudecker et al. 2012).

Channel Restoration and Stream Temperature

Spring creek restoration is important for wild trout because the potential for cool temperatures and stable flows can provide optimum conditions for spawning, rearing, and refugia (Decker-Hess 1985, 1987; Swanberg 1997; Pierce et al. 2014). Though the importance of groundwater influences to salmonid habitat is widely recognized (Brown and Mackay 1995; Baxter and Hauer 2000; Chu et al. 2008), few if any, published field studies examine the efficacy of restoration in buffering anthropogenic warming effects of stream temperatures in areas of strong groundwater inflows (Poole and Berman 2001; Ebersole et al. 2003). This form of temperature reduction is particularly important to coldwater salmonids and may be especially so for migratory native trout of western North America (e.g., Westslope Cutthroat Trout and Bull Trout) because migratory native trout often rely on a patchy network of cold, low-elevation streams for thermal refugia (Swanberg 1997; Rieman et al. 2007; Jones et al. 2014). The North Fork of the Blackfoot River near the Kleinschmidt Creek confluence provides a clear example of important low-elevation refugia. Here, fluvial Bull Trout from the Blackfoot River migrate, in some cases, long distances (>40 km) to summering areas in the lower North Fork of the Blackfoot River, where ambient summer water temperatures are about 5°C cooler than the Blackfoot River (Swanberg 1997; USFWS 2010; Pierce and Podner 2013).

Our study shows active restoration of groundwater-dominated streams can significantly reduce summer stream temperatures to levels suitable to native trout. In the case of Kleinschmidt Creek, a 69% reduction in wetted surface area of the channel preceded a 3.2°C reduction in the average maximum daily temperatures postrestoration, a 1.2°C reduction in mean daily stream temperatures, and a 3.3°C reduction in the range of daily temperatures (Figure 4). Following restoration, summer temperatures on Kleinschmidt Creek declined into the optimal thermal range of bull trout (i.e., maximum temperatures <13°C; Selong et al. 2001; Dunham et al. 2003) with maximum temperatures about 1.5°C colder than those in the North Fork of the Blackfoot River (Figure 4). Likewise, two additional spring creeks to the North Fork of the Blackfoot River have shown similar reductions in maximum temperatures (4–6°C) following full channel reconstruction (Pierce and Podner 2013). Considered together, restored spring creeks of the North Fork of the Blackfoot River show active restoration through reductions in wetted surface area, and revegetation can reduce summer temperatures in small, groundwater-dominated streams. Such buffering may ultimately prove important based on regional climate projections that point to continued loss of thermal habitat for coldwater salmonids, especially in low-elevation streams like the North Fork of the Blackfoot River (Rieman et al. 2007; Williams et al. 2009; Jones et al. 2014).

Groundwater and *M. cerebralis* Infection

The same groundwater-fed environments that often foster productive trout fisheries also make certain spring creeks more prone to the proliferation of *M. cerebralis* and the release of TAMs (MacConnell and Vincent 2002; Burckhardt and Hubert 2005; Neudecker et al. 2012). In exposure trials of surrogate rainbow trout, we found no restoration-induced moderation in the prevalence or severity of *M. cerebralis* infection, despite more natural channel morphology and reductions in summer stream temperatures. To the contrary, postrestoration exposures revealed higher (grade, $\geq$3) severity of infection in all Rainbow Trout trials than in prerestoration scores (Table 2). Increases in both the prevalence and severity of infection correspond in time with the *M. cerebralis* enzootic, which intensified in the Blackfoot basin between 1996 and 2005 (Pierce et al. 2009). Yet, our findings of high grades of severity are also consistent with recent studies showing season-long trends of high infection in groundwater-dominated streams (Pierce et al. 2012; Neudecker et al. 2012). Unlike the summer to autumn period of high TAM production in basin-fed streams and larger rivers (e.g., Gilbert and Granath 2001; Downing et al. 2002; Neudecker et al. 2012), high infections in groundwater environments relate to continuous TAM release under the influence of stable groundwater temperatures (Kerans and Zale 2002; Neudecker et al. 2012; Pierce et al. 2012).

Though this study emphasized summer temperature reduction, we also monitored winter stream temperatures prior to (December–March) and during the March 15 –25, 2002, sentinel exposures to explore groundwater–disease relationships. This monitoring identified an average temperature of 6.1°C from December through March and 5.8°C during the exposure period (Table 2). These values reveal relatively warm and stable winter temperatures compared with <1.0°C winter temperatures observed in nearby basin-fed streams during the same period (Pierce et al. 2004). These warmer temperatures are colder than the average temperatures (10–15°C) often associated with TAM release (El-Matbouli et al. 1999; Hansen and Budy 2011). Under these temperature conditions, all 48 Tainbow Trout in the 2002 exposure trial showed a high (grade, $\geq$3) severity of infection, and 28 showed grade-5 severity (Table 2). Similar to Neudecker et al. (2012), the March–July timing of high histological scores in our study overlapped temporally (winter and spring) with that of other Montana spring creeks where infectious conditions extend from autumn into spring. However, our study shows high infections can also extend from spring into summer in groundwater dominated streams. Unlike the summer–autumn peak of high infections for Montana Rivers at warmer water temperatures (i.e., 10–15°C; Gilbert and Granath 2001; Downing et al. 2002; Pierce et al. 2009), we found that high grades of infections in groundwater-dominated streams can occur regardless of season, which confirms high infections at much lower temperatures than in basin-influenced areas (e.g., Hansen and Budy 2011; Neudecker et al. 2012; Pierce et al. 2012). As described by Kerans et al. (2005), TAM release at lower temperatures relates the number of degree days that *T. tubifex* worms were exposed to *M. cerebralis* and that the development of *M. cerebralis* in worms is related to temperature accumulation. This may explain why infection in fish (and TAM production in general) is different in spring creeks where temperatures are high in winter.

Our study further suggests that certain groundwater-dominated streams, like Kleinschmidt Creek, may be predisposed to high infection rates. These conditions clearly contrast in both space and time to streams with more basin influences (Anlauf and Moffitt 2008; Hansen and Budy 2011; McGinnis and Kerans 2013), including the North Fork of the Blackfoot River (where sentinel exposures near the confluence of Kleinschmidt Creek consistently show low to no infection) and the main stem Blackfoot River (where high infections >3 in severity occur during summer; Pierce et al. 2009, 2012; Neudecker et al. 2012). For basin-fed streams, infectious conditions often occur in lower-elevation stream valleys with low gradients and fine sediments (De La Hoz Franco and Budy 2004; Anlauf and Moffitt 2008) and where temperatures (10–15°C) favor the seasonal release of TAMs (El-Matbouli et al. 1999; Hansen and Budy 2011). In addition to these natural stream features, human land uses that elevate temperature, sediment, and nutrient regimes (e.g., roads and heavy riparian grazing) have also been implicated in the proliferation of *M. cerebralis* by creating habitat favorable for *T. tubifex,* temperatures favorable to TAM release, or both (Zendt and Bergersen 2000; Anlauf and Moffitt 2008; Hansen and Budy 2011; McGinnis and Kerans 2013). Our study suggests the potential for restoration to mediate whirling disease may apply to streams with more basin influence. It also illustrates the potential for restoration to mitigate high summer stream temperatures or thermally unsuitable habitat, which could be especially important for basin-fed streams where temperatures are not buffered by groundwater influences and are significantly higher during the summer months.

In contrast with our results from a groundwater-dominated stream, Hansen and Budy (2011) showed disease reduction in a small basin-fed stream in a northern Utah watershed, where passive restoration (grazing exclusion) improved riparian condition, reduced total nitrogen and phosphorus levels, and reduced infection rates when mean daily summer stream temperatures fell below 10–15°C. These findings support an assertion that restoration potential varies between groundwater-fed and basin-fed streams. Implications of both studies are, however, limited by the short-term nature of the posttreatment data sets. In our study, exposure trials ended at 3 years postrestoration, which may not provide enough time for restoration-induced changes to alter tubificid lineages or otherwise mediate *M. cerebralis* through changes in benthic communities (Kerans et al. 2004; Beauchamp et al. 2005; Nehring et al. 2005). Long-term studies across hydro-physiological landscapes are needed to better explore the mechanisms of whirling disease reduction through restoration and stream temperature reduction.

With the exception of Brown Trout, most postrestoration exposure trials showed grades 4 and 5 severity (Table 2). Conversely, Brown Trout, a species with innate natural immunity, showed very low infection rates, as demonstrated by the side-by-side exposures in this study (Table 2). Following restoration, the abundance of brown trout have increased significantly in Kleinschmidt Creek (Pierce et al. 2012), whereas the presence of more susceptible species (Brook Trout, Rainbow Trout, Bull Trout, and Westslope Cutthroat Trout) remain incidental (Montana Fish, Wildlife and Parks, unpublished data). Though increases in resident Brown Trout can be attributed to habitat improvements, as well as disease resistance, our study suggests the incidental presence of other more susceptible salmonids may be the result of *M. cerebralis*.

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REFERENCES

Anlauf, K. J., and C. M. Moffitt. 2008. Models of stream habitat characteristics associated with tubificid populations in an intermountain watershed. Hydrobiologia 603:1579–5117.

Baldwin, T. J., E. R. Vincent. R. M. Sitlow, and D. Stanek. 2000. *Myxobolus cerebralis* in Rainbow Trout (*Oncorhynchus mykiss*) and Brown Trout (*Salmo trutta*) exposed under natural stream conditions. Journal of Veterinary Diagnostic Investigations 12:312–321.

Bartholomew, J. L., and P. W. Reno. 2002. The history and dissemination of whirling disease. Pages 3–24 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Bartholomew, J. L., and L. C. Wilson. 2002. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Baxter, C. V., and F. R. Hauer. 2000. Geomorphology, hyporheic exchange, and selection of spawing habitat by Bull Trout (*Salvelinus confluentus*). Canadian Journal of Fisheries and Aquatic Sciences 57:1470–1481.

Beauchamp, K. A., G. O. Kelly, R. B. Nehring, and R. P. Hedrick. 2005. The severity of whirling disease among wild trout corresponds to the differences in genetic composition of *Tubifex tubifex* populations in central Colorado. Journal of Parasitology 91:53–60.

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society Monograph 6, Bethesda, Maryland.

Brown, R. S., and W. C. Mackay. 1995. Spawning ecology of Cutthroat Trout (*Oncorhynchus clarki*) in the Ram River, Alberta. Canadian Journal of Fisheries and Aquatic Sciences 52:983–992.

Burckhardt, J. C., and W. A. Hubert. 2005. Evidence of relationships between *Tubifex* habitat and *Myxobolus cerebralis* across a mountain watershed. Intermountain Journal of Sciences 11:45–57.

Cassie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51:1389–1406.

Chu, C., N. E. Jones, N. E. Mandrak, A. R. Piggott, and C. K. Minns. 2008. The influence of air temperature, groundwater discharge, and climate change on the thermal diversity of stream fishes in southern Ontario watersheds. Canadian Journal of Fisheries and Aquatic Sciences 65:297–308.

De La Hoz Franco, E., and P. Budy 2004. Linking environmental heterogeneity to the distribution and prevalence of *Myxobolus cerebralis*: a comparison across sites in a northern Utah watershed. Transactions of the American Fisheries Society 133:1176–1189.

Decker-Hess, J. 1985. An inventory of Montana spring creeks. A report to the American Fisheries Society, Bethesda, Maryland by the Montana Fish, Wildlife and Parks, Helena.

Decker-Hess, J. 1987. Spring creeks—precious secrets. Montana Outdoors (May/June):2–5.

Downing, D. C., T. E. McMahon, K. L. Kerans, and E. R. Vincent. 2002. Relation of spawning and rearing of Rainbow Trout and susceptibility to *Myxobolus cerebralis* infection in the Madison River, Montana. Journal of Aquatic Animal Health 14:191–203.

Dunham, J., B. Rieman, and G. Chandler. 2003. Influences of temperature and environmental variables on the distribution of Bull Trout within streams at the southern margin of its range. North American Journal of Fisheries Management 23:894–904.

Ebersole, J. L., W. J. Liss, and C. A. Frissell. 2003. Thermal heterogeneity, stream channel morphology, and salmonid abundance in northeastern Oregon streams. Canadian Journal of Aquatic and Fisheries Sciences 60:1266–1280.

El-Matbouli, M. T., S. McDowell, D. B. Antonio, K. B. Andree, and R. P. Hedrick. 1999. Effect of water temperature on the development, release, and survival of triactinomyxon stage of *Myxobolus* *cerebralis* in its oligochaete host. International Journal for Parasitology 29:627–641.

Gilbert, M. A., and W. O. Granath Jr. 2001. Persistent infection of *Myxobolus cerebralis*, the causative agent of salmonid whirling disease in *Tubifex tubifex*. Journal of Parasitology 87:101–107.

Granath, W. O. Jr., M. A. Gilbert, E. J. Wyatt-Pescador, and E. R. Vincent. 2007. Epizootiology of *Myxobolus cerebralis*, the causative agent of salmonid whirling disease in the Rock Creek drainage of west-central Montana. Journal of Parasitology 93:104–119.

Hansen, E. S., and P. Budy. 2011. The potential of passive stream restoration to improve stream habitat and minimize the impact of fish disease: a short-term assessment. Journal of the North American Benthological Society 30: 573–588.

Hedrick, R. P., M. El-Matbouli, M. A. Adkinson, and E. MacConnell. 1999. Susceptibility of selected inland salmonids to experimentally induced infections with *Myxobolus cerebralis*, the causative agent of whirling disease. Journal of Aquatic Animal Health 11:330–376.

Hiner, M., and C. M. Moffitt. 2002. Modeling *Myxobolus cerebralis* infection in trout: associations with habitat variables. Pages 167–179 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Hubert, W. A., J. C. Burkhart, D. Zafft, K. Gelwicks, D. J. Money, and D. A. Hawk. 2002. The effects of landscape features on the distribution of *Myxobolus cerebralis* and occurrence of whirling disease among age-0 Mountain Whitefish in the Salt River drainage, Wyoming. Pages 181–193 *in* J. L. Bartholomew, and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Jones, L. A., C. C. Muhlfeld, L. A. Marshall, B. L. McGlynn, and J. L. Kershner. 2014. Estimating thermal regimes of Bull Trout and assessing the potential effect of climate warming on critical habitats. River Research and Applications 30:204–216.

Kaeser, A. J., C. Rasmussen, and W. E. Sharpe. 2006. An examination of environmental factors associated with *Myxobolus cerebralis* infections of wild trout in Pennsylvania. Journal of Aquatic Animal Health 18:90–100.

Kerans, B. L., C. Rasmussen, R. I. Stevens, A. E. L. Colwell, and J. R. Winston. 2004. Differential propagation of metazoan parasite *Myxobolus cerebralis* by *Limnidrilus hoffmeisteri*, *Ilyodilius templetoni*, and genetically distinct strains of *Tubifex tubifex*. Journal of Parasitology 90:1366–1373.

Kerans, B. L., R. I. Stevens, and J. C. Lemmon. 2005. Water temperature after a host-parasite interaction: *Tubifex tubifex* and *Myxobolus cerebralis*. Journal of Aquatic Animal Health 17:216–221.

Kerans, B. L., and A. V. Zale. 2002. The ecology of *Myxobolus cerebralis*. Pages 145–166 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Krueger, R. C., B. L. Kerans, E. R. Vincent, and C. Rasumussen. 2006. Risk of *Myxobolus cerebralis* infection to Rainbow Trout in the Madison River, Montana, USA. Ecological Applications 16:770–783.

MacConnell, E. and E. R. Vincent. 2002. The effects of *Myxobolus cerebralis* on the salmonid host. Pages 95–107 *in* J. L. Barholommew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Socieity, Symposium 29, Betesda, Maryland.

McGinnis, S., and B. Kerans. 2013. Land use and host community characteristics as predictors of disease risk. Landscape Ecology 28:29–44.

McMahon, T., J. Robison-Cox, J. Rotella, T. Horton, and B.Kerans. 2010. Trout populations response to whirling disease epizootics in Montana rivers. Pages 195–202 *in* R. F. Carline and C. LoSapio, editors. Wild Trout X: conserving wild trout. Wild Trout Symposium, Bozeman, Montana. Available: www.wildtroutsymposium.com. (April 2014).

Meehan, W. R., editor. 1991. Influences of forest and rangleand managment on salmonid fishes and their habitats. American Fiserhies Society, Special Publication 19, Bethesda, Maryland.

Meisner, J. D., J. S. Rosenfeld, and H. A. Regier. 1988. The role of groundwater in the impact of climate warming on stream salmonines. Fisheries 13(3):2–8.

Melinna, E., R. Dan Moore, S. G. Hinch, J. MacDonald, and G. Pearson. 2002. Stream temperature responses to clear-cut logging in British Columbia: the moderating influences of groundwater and headwater lakes. Canadian Journal of Fisheries and Aquatic Sciences 59:1886–1900.

Mote, P. W. 2006. Climate driven variability and trends in mountain snowpack in western North America. American Meteorological Society 19:6209–6220.

Nehring, R. B., K. J. Thompson, and M. Catanese. 2005. Decline in TAM production in Windy Gap Reservoir: is it related to a major shift in relative abundance of tubificid lineages? Pages 9–10 *in* Proceedings of the 11th annual whirling disease symposium: recipes for recovery. Whirling Disease Foundation, Bozeman, Montana.

Nehring, B., and P. G. Walker. 1996. Whirling disease in the wild: the new reality in the Intermountain West. Fisheries 21 (6):28–30.

Neudecker, R. A., T. McMahon, and E. R. Vincent. 2012. Spatial and temporal variation of whirling disease risk in Montana spring creeks. Journal of Aquatic Animal Health 24:201–212.

Pierce, R., R. Anderson, and C. Podner. 2004. The Blackfoot River restoration progress report for 2002 and 2003. Montana Fish, Wildlfie and Parks, Missoula.

Pierce, R., M. Davidson, and C. Podner. 2012. Spawning behavior of Mountain Whitefish and co-occurrence of *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 141:720–730.

Pierce, R., and C. Podner. 2006. The Big Blackfoot River fisheries restoration report for 2004 and 2005. Montana Fish, Wildlife and Parks, Missoula.

Pierce, R., and C. Podner. 2013. Fisheries investigations in the Big Blackfoot River basin, 2011–2012. Montana Fish, Wildlife and Parks, Missoula.

Pierce, R., C. Podner, and K. Carim. 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River Basin, Montana. Transactions of the American Fisheries Society 142:68–81.

Pierce, R., C. Podner, M. Davidson, and E. R. Vincent. 2009. Correlation of fluvial Rainbow Trout spawning life history with severity and infection of *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 138:251–263.

Pierce, R., C. Podner, and J. McFee. 2002. Blackfoot River fisheries inventory, monitoring and restoration report for 2001. Montana Fish, Wildlfie and Parks, Missoula.

Pierce, R., C. Podner, T. Wendt, R. Shields, and K. Carim. 2014. Westslope Cutthroat Trout movements through restored habitat in the Nevada Spring Creek complex, Blackfoot Basin, Montana. Transactions of the American Fisheries Society 143:230–239.

Poole, G. C., and C. H. Berman. 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management 27:787–802.

R Development Core Team. 2013. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Rieman. B. E., D. Isaak, S. Adams, D. Horan, D. Hagel, C. Luce, and D. Myers. 2007. Anticipated climate warming effects on Bull Trout habitats and populations across the interior Columbia River basin. Transactions of the Americian Fisheries Society 13:552–565.

Ryce, E. K. N., A. V. Zale, and E. MacConnell. 2004. Effects of fish age and parasite dose on the development of whirling disease in Rainbow Trout. Diseases of Aquatic Organisms 59:225–233.

Ryce, E. K. N., A. V. Zale, E. MacConnell, and M. Nelson. 2005. Effects of fish age versus size on the development of whirling disease in Rainbow Trout. Diseases of Aquatic Organisms 63:69–76.

Selong, J. H., T. E McMahon, A. V. Zale, and F. T. Barrows. 2001. Effect of temperature on growth and survival of Bull Trout with application of an improved method for determining thermal tolerances in fishes. Transactions of the American Fisheries Society 130:1026–1037.

Swanberg, T. R. 1997. Movements of and habitat use by fluvial Bull Trout in the Blackfoot River. Transactions of the American Fisheries Society 126:735–746.

Thompson, K. G. 2011. Evaluation of small-scale habitat manipulation to reduce the impact of the whirling disease parasite in streams. Aquatic Ecosystem Health and Management 14:305–317.

Thurow, R. F., D. C. Lee, and B. E. Rieman. 1997. Distribution and status of seven native salmonids in the interior Columbian River basin and parts of the Klamath and Great basin. North American Journal of Fisheries Management 17:1094–1110.

TopoWx. 2014. Montana climate office: University of Montana. Available: http://www.climate.umt.edu/products/meteorolgy/default.php. (July 2014).

USFWS (U.S. Fish and Wildlife Service). 2010. Endangered and threatened wildlife and plants; revised designation of critical habitat for Bull Trout in the coterminous United States; final rule. Federal Register 75:200(18 October 2010):63898–64070.

Vincent, E. R. 1996. Whirling disease and wild trout: the Montana experience. Fisheries 21(6):32–33.

Vincent, E. R. 2002. Relative susceptibility of various salmonids to whirling disease with emphasis on Rainbow and Cutthroat trout. Pages 109–116 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Williams, J. D., A. L. Haak, H. M. Neville, and W. T. Colyer. 2009. Potential consequences of climate change to persistence of Cutthroat Trout populations. North American Journal of Fisheries Management 29:533–548.

Zendt, J. S., and E. P. Bergersen. 2000. Distribution and abundance of the aquatic oligochaete host *Tubifex tubifex* for the salmonid whirling disease parasite *Myxobolus cerebralis* in the upper Colorado River basin. North American Journal of Fisheries Management 20:502–512.

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ARTICLE

Long-Term Increases in Trout Abundance following Channel Reconstruction, Instream Wood Placement, and Livestock Removal from a Spring Creek in the Blackfoot Basin, Montana

Ron Pierce* and Craig Podner
Montana Fish, Wildlife, and Parks, 3201 Spurgin Road, Missoula, Montana 59804, USA

Leslie Jones
U.S. Geological Survey, Northern Rocky Mountain Science Center Field Station, Glacier National Park, West Glacier, Montana 59936, USA; and Division of Biological Science, The University of Montana, 32 Campus Drive, Missoula, Montana 59812, USA

Abstract
To restore habitat for wild trout, Kleinschmidt Creek, a low-gradient, groundwater-dominated stream in the Blackfoot Basin, Montana, was reconstructed using natural channel design principles. Reconstruction increased stream sinuosity from a ratio of 1.1 to 1.6, decreased mean channel width from 14.5 to 2.8 m, and increased sediment transport capacity to reduce accumulations of fine instream sediment. To further improve trout habitat, coarse woody debris (CWD) was variably placed within the new channel and livestock were excluded to promote the vegetative recovery of the riparian area. To evaluate the response of wild trout (92% Brown Trout *Salmo trutta*) to channel restoration, the abundance (number of trout per linear meter) and biomass (g/linear m) of age 1+ trout were monitored for 15 years (1998–2012) in a reach with low density CWD (1.3 stems/100 m) and compared with regional (reference) trends. Posttreatment (2002–2012) trout numbers in the low-density CWD reach were also compared with those in a reach with high-density CWD (18.2 stems/100 m). Long-term trends for the reference reaches showed a significant negative trend in trout abundance and no significant trend for biomass. Long-term trends for the low-density CWD reach showed a significant positive trend in abundance, as well as a significant trend in biomass. Trout abundance and biomass increased over the posttreatment period in the low-density CWD reach. However, in the high density CWD reach, while posttreatment abundance increased significantly, there was no significant trend in biomass. These results demonstrated that channel restoration increased wild trout populations in a deep, narrow, vegetated stream and that instream wood provided primarily short-term benefits during the early phase of habitat recovery.

To offset human-induced degradation of river ecosystems, aquatic habitat restoration is expanding across North America. As restoration methods evolve, practitioners are applying natural restoration techniques more frequently to reestablish the ecological integrity and physical habitat necessary for the recovery of sensitive fisheries (Nagle 2007; Baldigo et al. 2008; Ernst et al. 2010; Pierce et al. 2013). Despite broad-scale increases in restoration, few projects document fisheries response to the use of natural channel design (NCD) as an active method to emulate the form and function of geomorphically stable natural streams (Rosgen 1996, 2007, 2011; Klein et al. 2007; Nagle 2007; Baldigo et al. 2008; Ernst et al. 2010). Likewise, few restoration studies investigate overlapping passive methods needed to mediate riparian damage caused by intensive land uses, such as heavy riparian grazing (Meehan 1991; Saunders and Fausch 2007; Pierce et al. 2013).

*Corresponding author at rpierce@mt.gov
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The paucity of published field studies limits the ability of restoration practitioners to develop and apply informed, effective, and more natural stream restoration techniques (Bernhardt et al. 2005; Baldigo et al. 2008, 2010; Roni et al. 2008).

For over two decades, restoration practitioners have increasingly focused projects on NCD and the recovery fluvial processes to increase channel stability, decrease erosion, restore natural flow, temperature, and sediment regimes, and revitalize sensitive fisheries (Nagle 2007; Baldigo et al. 2008, 2010; Pierce et al. 2013). Central to the NCD concept is the classification of natural rivers into stream types (Rosgen 1994). This classification provides a basis to emulate the dimension (e.g., valley bottom and stream cross-sectional features), pattern (e.g., meander features such as sinuosity), and profile (e.g., valley and channel slopes) of streams that are geomorphically stable (in equilibrium) with their stream valley (Rosgen 1994, 1996, 2007, 2011). Of the stream types described by Rosgen (1994), many are geologically controlled and thus resistant to human alterations. This study focuses on the restoration of a vegetatively controlled stream type that is considered more sensitive to anthropogenic disturbance such as heavy grazing than are streams controlled by geology.

Improper livestock grazing of certain riparian areas can degrade habitat for coldwater salmonids by increasing streambank erosion, reducing riparian vegetation, degrading streambanks, and lowering water tables, all of which cause streams to become wider, shallower, and warmer during summer (Meehan1991; Platts 1991; Pierce et al. 2014a). Though riparian impacts from heavy grazing are widely documented, the efficacy of most riparian grazing strategies to recover salmonid habitat remains uncertain with the exception of exclusion (Platts and Nelson 1985; Platts 1991; Roni et al. 2008; but see Myers and Swanson 1995). The impacts of heavy riparian grazing vary with timing, intensity, and frequency of grazing and type of animal (Platts 1991), as well as site conditions such as riparian plant composition, stream hydrology, and stream type (Hansen et al. 1988; Rosgen 1996; Bengeyfield and Svoboda 1998). Compared with armored channels having coarse (cobble, boulder, and bedrock) substrate, stream types that are controlled vegetatively with noncohesive alluvial soils (sand and gravel) are more sensitive to grazing disturbance (Myers and Swanson 1992; Rosgen 1996). This sensitivity can increase dramatically in areas of groundwater inflow where streambanks are wet during periods of grazing (C. B. Marlow and T. M. Pogacnik [paper presented at the North American Riparian Conference, 1985]).

Similar to the effects of heavy riparian grazing, the anthropogenic loss of instream wood can simplify and degrade salmonid habitat (e.g., Meehan 1991; Gregory et al. 2003; Roni et al. 2008) and ultimately reduce the overall ecological integrity of streams (Schmetterling and Pierce 1999; Bilby 2003; Rosenfeld and Huato 2003). The loss of instream wood is often the result of deforestation, excessive grazing, intentional forest clearing, road construction, and other streamside development pressures (Meehan 1991; Gregory et al. 2003; Jones et al. 2014). Conversely, input of coarse wood can improve the ecological integrity of streams by controlling gradient, increasing pools, and providing essential instream cover for fish (Roni et al. 2008; Whiteway et al. 2010; Jones et al. 2014) and diversifying habitat necessary for spawning and rearing of many salmonids (Schmetterling 2000; Gregory et al. 2003; Jones et al. 2014). Because of its high habitat value, instream wood has been used as a habitat improvement technique for decades (e.g., Tarzwell 1936; Hunt 1976; Binns 2003; Roni et al. 2008).

Despite the biological importance of instream wood, very few long-term (>5 years posttreatment) restoration studies have examined the response of salmonids to the placement of instream wood (Roni 2005; Roni et al. 2008). Exceptions include reach-scale treatments on confined (i.e., V-shaped valleys with narrow floodplains), stable channels and moderate (e.g., 2–4%) stream gradients (Rosgen 1996; Roper et al. 1998; Baldigo et al. 2008; White et al. 2011). The long-term effectiveness of wood placement in low-gradient (e.g., <2%) unconfined channels in alluvial valleys remains uncertain (Frissell and Nawa 1992; Schmetterling and Pierce 1999; Jones et al. 2014), particularly where meandering channel processes and rhizomatous meadow vegetation strongly influence channel morphology (Rosgen 1996).

In the Blackfoot River basin of western Montana, landowners, agencies, and private conservation groups have engaged in riverscape restoration actions for over 20 years (Aitken 1997; Pierce et al. 2013, 2014b). This study expands on a long-term monitoring study that showed restoration-induced reductions in water temperature in Kleinschmidt Creek (Pierce et al. 2014a). The purpose of this study was to describe changes to channel morphology following the conversion of an overwidened and degraded channel to a deep, narrow, meandering, and vegetated stream type, and to specifically examine the fisheries response to this conversion and the variable use of instream coarse woody debris (CWD) within the new channel.

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 approximately 3.4 river kilometers (rkm) within a terraced alluvial and morainal valley before entering the North Fork of the Blackfoot River at rkm 9.9 at an elevation of 1,268 m. The stream gains approximately 90% of its flow from groundwater inflows between rkm 1.6 and 3.2. Stream discharge ranges from 0.26 m^3/s during winter to about 0.42 m^3/s during summer (Pierce and Podner 2006) and the stream has a peak bankfull discharge of 0.71m^3/s (R. Shields, U.S. Geological Survey, retired, unpublished data). Streamside vegetation consists of wetland gramminoids (*Carex* spp., *Juncus* spp., *Phararis* spp.) and shrub

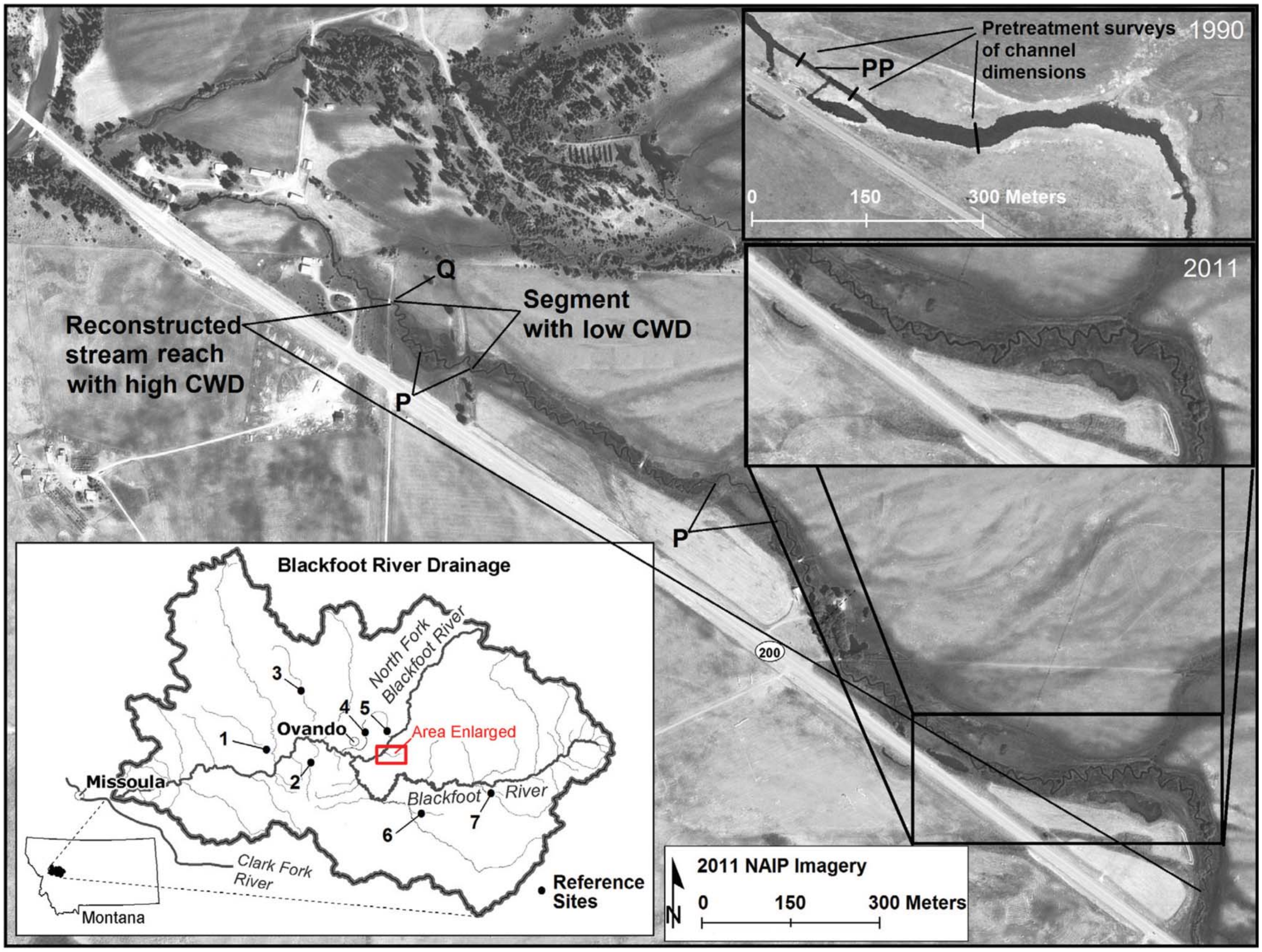

FIGURE 1. Kleinschmidt Creek and the 2001 restoration project area. High density CWD was placed throughout the reconstruction site with the exception of the stream segment with low density CWD. In addition, two fish population monitoring sites (P) and the bankfull flow measurement site (Q) are identified. The map inset (lower left panel) shows the Blackfoot River basin and the Kleinschmidt Creek study area in Montana, as well as seven reference reach fish population monitoring sites. The two upper right panel insets show pretreatment (1990; aerial photo) and posttreatment (2011; NAIP 2011) images of the upper reaches of the study area and the photopoint (PP) location. [Figure available online in color.]

(*Salix* spp., *Alnus* spp.) cover. Kleinschmidt Creek supports a mixed community of salmonids though Brown Trout *Salmo trutta* comprise >90% of the salmonid community. Other salmonids present in the order of decreasing abundance are Brook Trout *Salvelinus fontinalis*, native Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi*, native Bull Trout *S. confluentus*, and Rainbow Trout *O. mykiss*. The exotic parasite, *Mxyobolus cerebralis*, the cause of whirling disease in salmonids (Bartholomew and Wilson 2002), is also present in Kleinschmidt Creek where infection rates are high (Pierce et al. 2014a). Prior to restoration, Kleinschmidt Creek was subjected to heavy livestock grazing, accelerated streambank erosion and channelization resulting from highway construction, the installation of artificial grade controls that included rock dams and undersized culverts (Pierce 1991; Land and Water Consulting 1999; M. Marler, 1998 unpublished technical report to the U.S. Fish and Wildlife Service, on site assessment and summary of impacts of proposed stream restoration). A combination of these factors caused the channel to become wide, shallow, and straightened with elevated water temperature (Decker-Hess 1986; Pierce et al. 2014a) and high sediment loading (sand and silt), especially upstream from rock dams (Figure 2A). These conditions resulted in a corresponding reduction of instream habitat complexity, degraded wetlands, and the complete loss of woody riparian vegetation (Figure 2A). The pretreatment morphological values are shown in Table 1.

With the ultimate goal of restoring stream habitat for the recovery of wild trout, the restoration of the lower 0.64 rkm of Kleinschmidt Creek was completed in 1998. Then in 2001, restoration on the remaining upper 2.73 rkm of stream

A

B

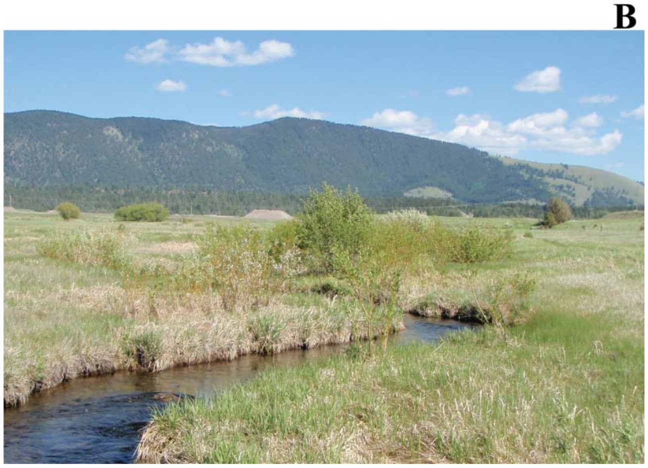

FIGURE 2. Kleinschmidt Creek (**A**) pretreatment, September 2001, and (**B**) posttreatment, June 2014. The upper photo (A) shows a straightened and over-widened section of channel with an example of a channel-altering rock dam that induced upstream deposition of fine sediment within the pretreatment channel. The lower photo (B) shows the restored stream at the same location. [Figure available online in color.]

(the emphasis of this study) was completed to reestablish natural form and function consistent with a vegetative-controlled, more sinuous, deep and narrow channel (Figures 1, 2B). With the use of the Rosgen (1996) stream classification, the 2001 project fit the stream to its valley, reconnected the channel with its original floodplain, removed artificial grade controls (rock dams and culverts), and converted a degraded, wide and shallow stream with fine substrate (i.e., Rosgen impaired C5 stream type; Figure 2A) to form a more natural deep and narrow meandering channel (i.e., Rosgen reference E4 stream type: Figure 2B; Rosgen 1996, 2007, 2011). In the absence of vegetative disturbance, the E4 stream type is a geomorphically stable, hydraulically efficient channel found within alluvial valleys. The stable E4 stream type is specifically characterized by low channel gradients (<2%), low width : depth (W:D) ratios (<12), high sinuosity (>1.5), gravel substrates, and pool–riffle bedforms. The degradation of an E4 stream type can convert a stable channel to an impaired C5 stream type through accelerated streambank erosion, channel widening, loss of sinuosity, and the instream accumulation of fine sediment (Rosgen 2007; e.g., Figure 2B). Delineative criteria for the E4 and C5 stream types are shown in Table 2.

During final channel restoration, the upper 2.73 rkm of stream was narrowed to a mean width of 3.0 m, pool–riffle bedforms were reestablished, and 183 coarse woody stems were anchored throughout the streambed and outer banks of most (100 of 108) pool–riffle sequences between rkm 0.95 and rkm 3.40 (Figure 1). This wood was specifically used to offset the anthropogenic loss of instream wood and to increase habitat complexity until riparian vegetation (including woody species) was reestablished. However, during final channel shaping, a 0.31-rkm segment (16 pool–riffle sequences; rkm 0.65–0.95) of new channel was left with a very minimal amount of CWD (n = 3 stems total) to explore the fisheries response as described below (Figure 1). Following channel restoration, riparian shrubs were planted, livestock were excluded from the riparian corridor to recover streamside vegetation, and a perpetual conservation easement was placed on a majority of the stream corridor to protect the long-term ecological integrity of wetlands and the riparian area and to prevent grazing-related damage to the new channel (MDT 2001). Although Kleinschmidt Creek had no recent history of dewatering, a large portion of the stream's discharge (0.25 m^3/s) was later dedicated to the maintenance of instream flows to prevent the possibility of future dewatering (MDNRC 2011).

METHODS

Stream channel morphology.—To identify pretreatment and posttreatment project-scale changes (rkm 0.64–3.37) in channel morphology, aerial photo–imagery measurements and field measurements with NCD-related techniques were used (Rosgen 2007, 2008, 2011). Channel lengths and sinuosity were measured from aerial photographs taken prior to restoration (1990) and from National Agricultural Imagery Program (NAIP) high-resolution (1 m) imagery taken 10 years after (2011) full channel reconstruction and livestock exclusion (Figure 1). Aerial photos were georectified and NAIP imagery was classified to contrast surface waters with riparian vegetation. ArcGIS (version 10.1) software was used to measure channel lengths (1:500 scale) along the centerline of the 1990 and 2011 channels. Stream sinuosity was calculated as a ratio of the total centerline channel length divided by the linear distance of the stream valley. In addition to aerial measurements, field surveys of stream slope (using a laser level) and bankfull discharge (measured at bankfull stage with a Marsh–McBirney model 2000 Flo-mate current meter) were measured at the lower portion of the restoration project in June 2011 (Figure 1). Pretreatment channel dimensions were surveyed in the upper treatment area by Land and Water Consulting (1999) (Figure 1). Posttreatment channel

TABLE 1. Summary of channel morphometrics for Kleinschmidt Creek before restoration (1990) and 10 years after restoration (2011) along with hydraulic relationships of the pretreatment (C5) and posttreatment (E4) stream types.

Variable	Pretreatment (C5)	Posttreatment (E4)
Channel length (km)	1.97	2.73
Sinuosity	1.1	1.6
Stream slope (S; m/m)	0.0058	0.0040
Valley slope (m/m)	0.0064	0.0064
Bankfull discharge (Q_{bkf}; m^3/s)	0.71	0.71
Mean bankfull width (W_{bkf}; m)	20.1	3.1
Mean bankfull depth (D_{bkf}; m)	0.13	0.35
Bankfull W:D ratio	150	8.8
Mean bankfull cross-sectional area (A_{bkf}; m^2)	2.6	1.1
Mean bankfull mean velocity ($\bar{u}_{bkf}$; m/s)	0.27	0.67
Bankfull shear stress (τ; N/m^2)	7.4	13.7
Particle entrainment size (mm)	11	21

dimensions were surveyed in 2013 at the low CWD reach (Figure 1). Associated morphological relationships representing the mean values were then calculated to compare the observed pretreatment (C5) and posttreatment (E4) vegetated stream types (Rosgen 2007, 2008, 2011; Table 1).

Hydraulic and sediment transport relationships were then calculated for the pre- and posttreatment channels. Bankfull mean velocity ($\bar{u}_{bkf}$) (m/s) was estimated using the flow continuity equation,

$$\bar{u}_{bkf} = Q_{bkf}/A_{bkf}, \tag{1}$$

where Q_{bkf} = bankfull discharge (m^3/s) and A_{bkf} = bankfull cross-sectional area (m^2). Pre- and posttreatment bankfull shear stress (τ) (N/m^2) was calculated using the relation,

$$\tau = \gamma RS, \tag{2}$$

where γ = specific weight of water (9.81 kN/m^3), R = hydraulic radius (substituted by mean bankfull depth), and S = stream slope. Pre- and posttreatment particle entrainment sizes (mm) were derived from the critical bankfull shear stress relation in Leopold et al. (1964):

$$\text{Particle diameter} = 77.966(\tau/47.88)^{1.042}. \tag{3}$$

To obtain specific reach-scale treatment data that may further affect trout populations, stream surveys were undertaken in each of the two 154-m-long fish population survey sites in 2013 in reaches with low and high density CWD using modified Rosgen level II survey methods (Rosgen 1996). For these surveys, stream surveys were conducted with a laser level and measuring rod to determine wetted widths and wetted depths of all pools (n = 8) and riffles (n = 8). In addition, bankfull W:D ratios at one representative riffle within each section were measured as was substrate composition using Wolman pebble counts (with a minimum of 100 measured pebbles) at each representative riffle. We also counted and measured all instream woody stems (>10 cm diameter and >1 m in length) anchored within each fish population monitoring site and measured the percent shrub cover overhanging the streambanks of each survey reach.

Fisheries data collection and organization.—Three years of pretreatment fish population data (1998–2000) were obtained as a baseline in the impaired (C5) stream reach with minimal wood (Figure 1). Following active channel work in 2001, annual fish population surveys continued over an 11-year posttreatment period (2002–2012) at rkm 0.80–0.95 in the (E4) reach with low CWD and at rkm 1.62–1.77 in the (E4) stream reach with high CWD. Each monitoring site was 154 m in length as described above. Fish populations were not surveyed in Kleinschmidt Creek in 2011 due to sampling difficulties related to high stream flow. Pretreatment and posttreatment fisheries data from the low-density CWD reach were compared with regional trends of abundance and biomass to analyze long-term (15 year) response trends. Reference fisheries data were compiled from seven small, alluvial, low-gradient (<2%) and

TABLE 2. Delineative criteria for the C5 and E4 stream types (Rosgen 1996).

Stream type	Entrenchment ratio	W:D ratio	Sinuosity	Channel slope	Predominate channel materials
C5	>2.2	>12	>1.2	<2%	Sand
E4	>2.2	<12	>1.5	<2%	Gravel

TABLE 3. Site conditions for the seven reference reaches in Kleinschmidt Creek. Stream identification (ID) relates to stream locations on Figure 1.

Stream ID	Stream name	Stream order	Elevation (m)	Stream slope (%)	Sinuosity	Bankfull riffle area (m^2)	*W*:*D* ratio	Substrate D50 (mm)	Stream type
1	Blanchard Creek	2	1,167	1.9	1.1	2.2	29	39.0	C4
2	Chamberlain Creek	2	1,306	1.0	1.2	1.8	18	35.0	C4
3	Cottonwood Creek	3	1,324	2.0	1.2	2.6	12	68.0	C4
4	Warren Creek	2	1,316	0.3	1.2	1.1	9	17.0	E4
5	Murphy Spring Creek	1	1,314	0.2	1.5	1.2	16	29.0	C4
6	Wasson Creek	1	1,325	1.2	1.5	0.5	10	12.0	E4
7	Grantier Spring Creek	1	1,379	0.3	1.6	0.8	15	12.0	C4

low-elevation reference reaches (Figure 1; Table 3). Reference reaches were defined as geomorphically and vegetatively stable (Rosgen 1996) with fish populations unaffected by direct human impacts (Pierce et al. 2013). Fish populations were surveyed in reference reaches every year from 1998 through 2012 ($n = 46$ surveys); each discreet reference reach averaged 8 years (range, 3–14 years) of fisheries survey data, and each monitoring year between 1998 and 2012 averaged three surveys (range, 1–6). Reference reaches are listed in Table 3.

All estimates of trout abundance and biomass in this study were derived from of age 1 and older trout. Estimates of abundance (number of trout per linear meter, hereafter trout/m) were conducted between August 7 and October 4 using backpack electrofishing units and depletion (two- and three-pass) estimator methods (Van Deventer and Platts 1985). Estimates of biomass (g/linear m, hereafter g/m) were calculated by multiplying population abundance by mean fish weight. Fish population estimates generated a mean capture probability of 0.75 (range, 0.50–0.94) for reference reaches and 0.61 (range, 0.48–1.0) for surveys in Kleinschmidt Creek. All fish population surveys began at a downstream pool–riffle break (e.g., riffle crest), proceeded upstream, and ended at an upstream pool–riffle break. Block nets were set at the upstream survey boundaries of the Kleinschmidt Creek survey sites in years with high water but not in low-water years. Once captured, all trout were sedated with tricaine methanesulfonate (MS-222; Argent Chemical Laboratories, Redmond, Washington) or clove oil. Individual trout were then identified by species, measured for TL (mm), and weighed (g) and then immediately placed in freshwater to regain their equilibrium before their release within the monitoring section. Because CPUE of age-0 trout did not differ between low-density and high-density CWD reaches in Kleinschmidt Creek during the monitoring period, age-0 trout were excluded from abundance and biomass analyses.

Analyses of trout response to restoration.—To analyze the fisheries response, ANCOVA was used to test for treatment effects between the low-density CWD, high-density CWD, and reference reaches for pre- and posttreatment periods. Linear regressions were used to test for significant trends (positive or negative slopes) in abundance and biomass before and after treatment and over the entire study period for low-density CWD, high-density CWD, and reference reaches. Increases in trout abundance and biomass were considered statistically significant if the slope of the trend line was significantly different from zero ($P < 0.05$). Prior to statistical analysis, all estimates of abundance were natural log transformed to meet assumptions of normality and homogeneity of variance. Before transformation, a value of 1 was added to each estimate to avoid generating a value of negative infinity when we attempted to transform values of zero. All analyses were performed using SAS version 9.4 statistical software (SAS Institute, Cary, North Carolina).

RESULTS

Stream Channel Morphology

The posttreatment channel was 39% longer, 85% narrower, and 169% deeper than the pretreatment channel, and riparian vegetation was reestablished (Figures 1, 2B; Table 1). With these changes in basic channel morphology, posttreatment hydraulic conditions of bankfull mean velocity, shear stress, and sediment entrainment all increased (Table 1).

Posttreatment habitat surveys of the two 154-m fish population survey sites (i.e., high and low CWD reaches) showed broad similarity between sites, including having low *W*:*D* ratios (range, 8.8–9.1) and a dominant gravel substrate (D50 range, 14–21) in riffles (Table 3), which further characterized both monitoring sites as the same E4 stream type (Rosgen 1996). Consistent with uniform features of this deep and narrow, meandering, vegetated stream type, habitat features showed comparable wetted widths and wetted depths for the eight pool–riffle sequences in both reaches (Table 4), although pools averaged 0.3 m deeper in the reach with low CWD. Concentrations of CWD were, however, 14 times higher in the reach with high density CWD compared with the reach with high density CWD, and shrub cover was 9% in the reach with high CWD and <1% in the low-density CWD reach.

TABLE 4. Summary of channel bedform features for low- and high-density CWD reaches in Kleinschmidt Creek including pool and riffle width and depth measurements, substrate, *W*:*D* ratios, and summaries of instream wood concentrations for the two fish population monitoring sites.

	Pools		Riffles					Instream CWD	
Reach	Mean (range) wetted depth (m)	Mean (range) wetted width (m)	Mean (range) wetted depth (m)	Mean (range) wetted width (m)	Substrate D50 (mm)	*W*:*D* ratio	Number of stems	Mean (range) diameter (cm)	Mean (range) length (m)
Low CWD	1.1 (0.8–1.2)	3.4 (3.2–3.8)	0.4 (0.3–0.5)	2.8 (2.4–3.6)	14	8.8	2	28 (25–30)	1.2 (1.0–1.5)
High CWD	0.8 (0.7–1.1)	3.5 (3.1–4.0)	0.3 (0.2–0.4)	3.1 (2.7–3.5)	21	9.1	28	25 (18–30)	2.3 (1.2–4.6)

Trout Response to Restoration

Composition of trout species for the two Kleinschmidt Creek fish population survey sections showed Brown Trout comprised 92% of the total catch followed by 7% Brook Trout, 0.5% Westslope Cutthroat Trout, 0.3% Bull Trout, and 0.2% Rainbow Trout (Figure 3). Pre- and posttreatment comparisons showed that pretreatment abundance was significantly higher in the reference reaches than in the low-density CWD reach ($F_{1,\ 10} = 14.64$, $P = 0.003$) and remained higher posttreatment ($F_{1,\ 44} = 15.39$, $P < 0.001$). Conversely, there was no significant difference is pretreatment biomass ($F_{1,\ 10} = 2.08$, $P = 0.18$) or posttreatment biomass ($F_{1,\ 44} = 0.61$, $P = 0.439$) between the low-density CWD and reference reaches. Further analysis showed a significant treatment effect between the low-density CWD and high-density CWD reaches for trout abundance ($F_{1,\ 18} = 7.73$, $P = 0.012$) and no significant difference for trout biomass ($F_{1,\ 18} = 4.11$, $P = 0.06$). Long-term trends for the reference reaches showed a significant negative trend in abundance ($F_{1,\ 43} = 4.75$, $P = 0.03$, slope $= -0.014$; Figure 4A) and no significant trend for biomass ($F_{1,\ 43} = 0.74$, $P = 0.395$, slope $= -0.691$; Figure 4B). Long-term trends for the low-density CWD reach showed a significant positive trend in abundance ($F_{1,\ 11} = 19.26$, $P = 0.001$, slope $= 0.025$) and a significant positive trend in biomass ($F_{1,\ 11} = 39.89$, $P < 0.001$, slope $= 3.75$).

During the 15-year study period, total trout abundance in the reach with low density CWD increased from a pretreatment average of 0.06 trout/m to a posttreatment average of 0.25 trout/m, compared with a 15-year average of 0.47 trout/m for the reference reaches (Figure 5A). Likewise, biomass in

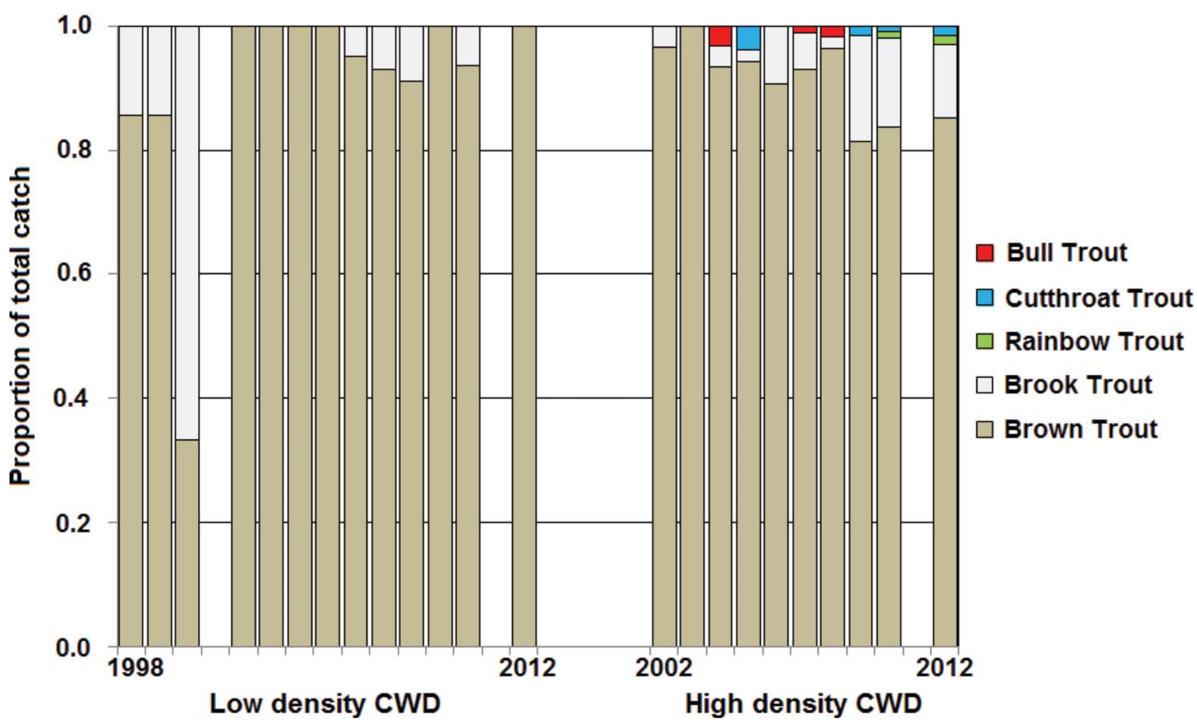

FIGURE 3. Species composition for age-1 and older trout for the low- and high-density CWD reaches. [Figure available online in color.]

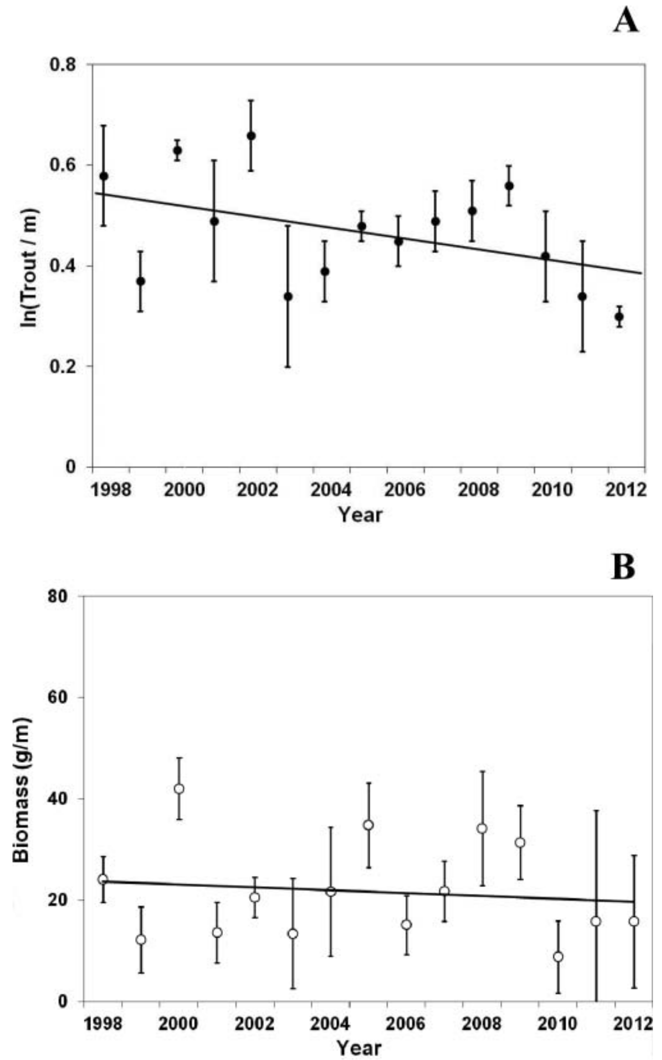

FIGURE 4. Comparisons by year of (**A**) abundance and (**B**) biomass and across-year comparisons of slopes for seven reference reaches.

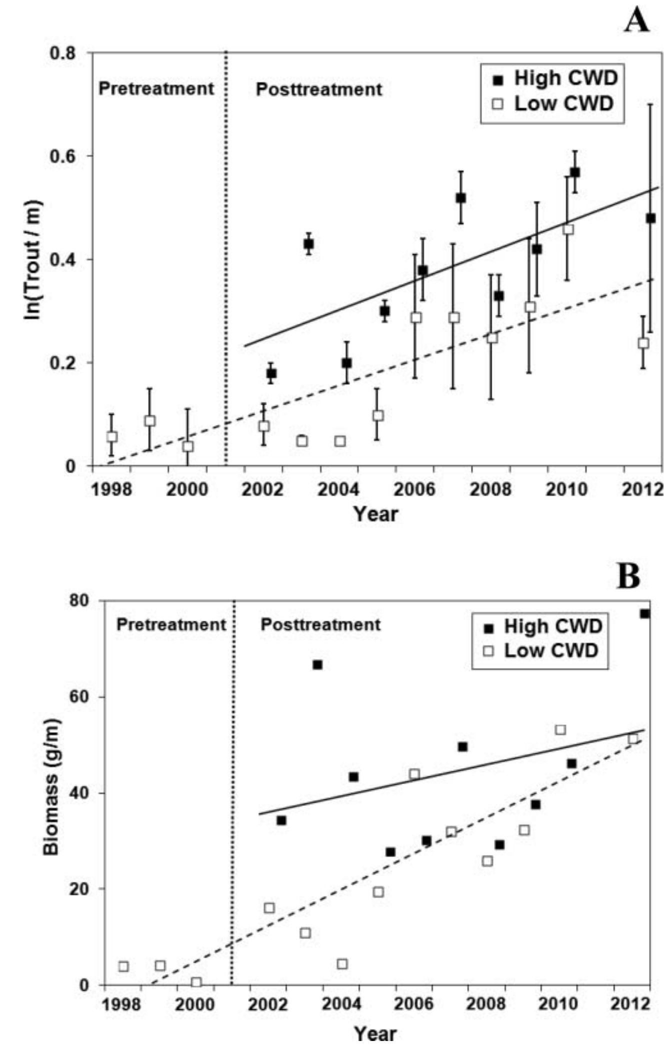

FIGURE 5. Estimates of (**A**) total trout abundance and (**B**) total trout biomass for the low- and high-density CWD reaches of Kleinschmidt Creek. The solid black trend lines in both panels show the positive increase in the posttreatment high-density CWD reach, and the dashed lines show the positive population increase in the low-density CWD reach over the entire monitoring period.

the low-density CWD reach increased from a pretreatment average of 3.1 g/m to a posttreatment average of 29.1 g/m, compared with 44.2 g/m in the reach with high CWD and the long-term reference reach average of 21.7 g/m. No significant trends in abundance or biomass were found posttreatment for the reference reaches (abundance: $F_{1,\ 9} = 1.84$, $P = 0.21$; biomass: $F_{1,\ 9} = 0.18$, $P = 0.68$). Whereas in the low-density CWD reach, abundance and biomass increased significantly over the posttreatment period (abundance: $F_{1,\ 8} = 11.97$, $P = 0.009$; biomass: $F_{1,\ 8} = 18.0$, $P = 0.003$; Figure 5A, B). In the high-density CWD reach, posttreatment abundance increased significantly ($F_{1,\ 8} = 7.45$, $P = 0.03$; Figure 5A) and biomass had no significant trend ($F_{1,\ 8} = 0.91$, $P = 0.37$; Figure 5B). Posttreatment rates of increase in trout abundance and biomass were found to be highest in the low-density reach (abundance: slope = 0.03; biomass: slope = 4.34).

DISCUSSION

This study had several limitations: (1) the inability to separate trout response from the effects of channel reconstruction, CWD, and livestock exclusion in the low-density CWD reach, (2) pretreatment data limitations, especially the lack of fisheries data in the high-density CWD reach, (3) the possible influence of prior restoration on trout response, and (4) the high severity of whirling disease for salmonids other than Brown Trout (Pierce et al. 2014a). Despite these limitations, this study presents one of the few case studies that describe the long-term response of wild trout to comprehensive stream restoration. This study specifically clarifies biological responses to the conversion of an overwidened and degraded stream to a deep, narrow, more natural channel, as well as changes of fluvial processes as a function of channel shape. This study further provides qualified support that certain low-gradient, meandering, meadow streams (i.e., E4 stream type) may require the addition of a minimal instream woody structure as a long-term (>5 years) wild trout habitat improvement technique.

Stream Channel Morphology

To reestablish natural form and function of Kleinschmidt Creek, restoration began with the removal of instream dams, full channel reconstruction, and the conversion of an overwidened and heavily degraded (C5) stream type to a more natural, vegetated, deep, narrow (E4) stream type. As described by Rosgen (1996), the reconstructed E4 stream type represents the evolutionary end point (i.e., reference condition) for geomorphically stable, self-maintaining channels in gently sloping alluvial valleys where densely rooted vegetation is firmly established. This conversion to a deep and narrow vegetated channel not only created a shift in stream type (Table 1), but also increased transport capacity of fine sediment (Table 1). Because this conversion reduces the potential for aggradation, it helps maintain the stable form and function of gravel-bed stream that has stable vegetated streambanks and a low *W*:*D* ratio. This increased transport capacity may be especially important in low-gradient, groundwater-dominated streams where high flow (i.e., flushing) events are infrequent. Groundwater-dominated streams are thereby more prone to accumulations of fine sediment from anthropogenic erosion and other forms of channel degradation than are basin-fed streams. Without this conversion to a more hydraulically efficient vegetated stream type, the placement of instream CWD alone would do little to offset basic limiting factors (e.g., low pool frequency, low bank cover, loss of spawning areas) or otherwise mediate sediment and temperature impacts associated with the pretreatment channel (Pierce et al. 2014a).

Though instream wood is widely used to improve salmonid habitat, the use of CWD may be especially effective for habitat improvement in low-gradient (<2%) alluvial stream types with high *W*:*D* ratios (>12; i.e., morphologically stable C stream types: Rosgen 1996), as well as confined stream types with moderate gradient (>2%) and step-pool morphology (Rosgen 1996; Binns 2003; Gregory et al. 2003; White et al. 2011). In our study area, the placement of CWD was used to improve and diversify trout habitat and provide interim bank cover during the period of vegetative recovery. Though placement of instream wood can clearly increase carrying capacity for trout 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 for habitat improvement varies widely depending on geomorphic conditions (e.g., stream types: Rosgen 1996; Schmetterling and Pierce 1999; White et al. 2011); although, few fisheries studies explicitly link wood to geomorphic condition stream types (e.g., stream types: Rosgen 1996; Schmetterling and Pierce 1999; Baldigo et al. 2008). Indeed, we are unaware of any prior published long-term (>5 years) study that links the response of wild trout to the use of instream wood in reconstructed, low-gradient, meandering meadow streams with strong groundwater influence and the prevailing influence of rhizomatous riparian vegetation.

Following channel restoration and grazing changes, perpetual conservation easements were used to ensure long-term vegetative stability and habitat protection given the high sensitivity of the E4 stream type to vegetative disturbance (Rosgen 1996; Figure 2B). For Kleinschmidt Creek, this sensitivity relates to spring seeps, wet stream banks, and high summer flows that significantly elevate the potential for mechanical (e.g., hoof-shear) streambank damage during the summer livestock grazing season. Because of these site conditions, grazing changes and easement protection were considered essential to reestablish a fully vegetated, more natural, self-maintaining stream with sustainable fish habitat.

Trout Response to Restoration

During the study, total trout abundance in the reach with low density CWD increased compared to the reference reaches. Likewise, biomass in the low density CWD reach increased. Elevated biomass, especially in the high-density CWD reach, speaks to the productive nature of spring creeks in general (Decker-Hess 1986, 1987) and the specific presence of large (>500 g) resident Brown Trout in the case of Kleinschmidt Creek. Ironically, the high severity of whirling disease infection in Kleinschmidt Creek may actually favor elevated biomass by favoring resident Brown Trout (Pierce et al. 2014a), a species with natural resistance to *M. cerebralis* (Bartholomew and Wilson 2002), over other salmonids in Kleinschmidt Creek with less resistance and/or greater migratory behavior (Pierce et al. 2014a).

During the initial 4-year recovery period, monitoring identified notable differences in population size between the two fish monitoring sites. Though a lack of pretreatment data limited the strength of between-site comparisons, posttreatment differences were large and biologically revealing. Large differences initially included an 11-times higher abundance and six-times higher biomass in the high-density CWD treatment reach 2 years after restoration was completed (Figure 4A, B). After a 4-year recovery period, differences decreased with estimates of abundance to 1.6 times (range, 1.3–2.2) higher in the high-density CWD reach and only 1.1 times (range, 0.7–1.5) greater biomass in the high-density CWD reach (Figure 4A). These short-term differences indicate movement into and aggregation within pools with complex woody structure (e.g., Hunt 1976; Gowan and Fausch 1996; Dolloff and Warren 2003), which may be especially relevant to Brown Trout given their innate preference for low-gradient streams with undercut streambanks, abundant cover, and dim light (Wesche 1980; Bachman 1984; Larscheid and Hubert 1992). More incremental long-term (≥5year) changes may reflect compensatory mechanisms driven by density dependence, as indicated by a delayed response in the low-density CWD reach. After this delay, annual estimates of abundance and biomass followed similar trajectories at both monitoring sites (Figure 4A, B).

Though long-term (≥5 year) population trajectories were ultimately similar (Figure 4A, B), we continued to identify biological differences between the two fish population monitoring sites when considering the full monitoring data set. As one example, mean trout abundance was 63% higher (i.e., 0.57 trout/m) in the high-density CWD reach at >5 years posttreatment compared with that (0.36 trout/m) in the low-density CWD reach. In comparison, biomass was only 11% higher (i.e., 45 g/m) in the high-density CWD reach compared with the low-density CWD reach (40 g/m). These differences reflect an increasing percentage of smaller fish in the section with high CWD density (e.g., mean TL = 168 mm, mean weight = 79 g) versus larger fish in the section with low CWD density (e.g., mean TL = 195 mm, mean weight = 122 g). Likewise, the posttreatment CPUE of age-0 trout averaged 26% higher in the high CWD reach although these differences were not significant. Interestingly, all age-1 and older Rainbow Trout, Westslope Cutthroat Trout, and Bull Trout sampled in this study ($n = 10$) were captured in the high-density CWD reach where physical habitat was more diverse, although the proportion of these species within the trout assemblage was very small (Figure 3). Though other factors (e.g., variation in pool depth, shrub cover, or forage availability) could explain some of these differences, such differences were expected and formed the basis of decisions to add instream wood over most of the project area.

Rosgen (1996) suggested the placement of wood within the E stream type provides limited habitat benefit given the low *W*:*D* ratios of the stable channel form, lateral habitat forming processes, and increased vegetative bank cover typical of deeper and narrower (E) channels. However, *W*:*D* ratios are highly variable (range, 2–12) under the Rosgen geomorphic stream classification for the E stream type (Rosgen 1996). Our stream surveys found *W*:*D* ratios within the upper range (i.e., 8.8–9.1) of E stream type classification at the two fish population monitoring sites. Though long-term trout response trends in the low-density CWD reach suggest that wood is a minor habitat feature in the E stream type (Rosgen 1996), higher trout abundance, more diverse population structure, and more species diversity indicate that the addition of wood could be an effective fish habitat improvement. Based on these results, the use of instream wood should be considered when developing restoration goals or species targets, especially at the higher range of *W*:*D* ratio values for the E stream types.

After 11 years of posttreatment fisheries monitoring, estimates of abundance and biomass showed no clear indication that wild trout populations have reached equilibrium at either treatment site (Figure 4A, B). This indicates long-term recovery periods (>10 year) may be required in fully reconstructed streams where recovery of biotic communities, including maturation of 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. In addition to the essential role of vegetation in habitat creation and maintenance for the E stream type, trout population recovery rates in vegetated streams further relate to food-web pathways, including increased aquatic and terrestrial prey (e.g, macroinvertebrates) used by trout. In the case of Kleinschmidt Creek, terrestrial prey may be especially relevant because terrestrial prey can significantly increase following reductions in grazing pressure and corresponding increases in riparian vegetation (Saunders and Fausch 2007).

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REFERENCES

Aitken, G. 1997. Restoration of trout waters in the West: Blackfoot River of Montana. Pages 402–424 *in* J. E. Williams, C. A. Wood, and M. P. Dombeck editors. Watershed restoration: principles and practices. American Fisheries Society, Bethesda, Maryland.

Bachman, R. A. 1984. Foraging behavior of free-ranging wild trout and hatchery Brown Trout in a stream. Transactions of the American Fisheries Society 113:1–32.

Baldigo B. P., A. G. Ernst, D. R. Warren, and S. J. Miller. 2010. Variable responses of fish assemblages, habitat, and stability to natural-channel-design restoration in Catskill Mountain Streams. Transactions of the American Fisheries Society 139:449–467.

Baldigo, B. P., D. R. Warren, A. G. Ernst, and C. I. Mulvhill. 2008. Response of fish population to natural channel design restoration in stream of the Catskill Mountains, New York. North American Journal of Fisheries Management 28:954–969.

Bartholomew, J. L., and L. C. Wilson, editors. 2002. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Bengeyfield, P., and D. Svoboda. 1998. Determining allowable use levels for livestock movement in riparian areas. Rangeland Management and Water Resources, American Water Resources Association 5:243–257.

Bernhardt, E. S., M. A. Palmer, J. D. Allen, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G. M. Kondolf, P. S. Lake, R. Lave, J. L. Meyer, T. K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Synthesizing United States river restoration efforts. Science 308:636–637.

Bilby, R. E. 2003. Decomposition and nutrient dynamics of wood in streams and rivers. Pages 135–147 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Binns, N. A. 2003. Effectiveness of habitat manipulation for wild salmonids in Wyoming streams. North American Journal of Fisheries Managment 24:911–921.

Decker-Hess, J. 1986. An inventory of spring creeks in Montana. Montana State Library, Report to the American Fisheries Society, Helena.

Decker-Hess, J. 1987. Spring creeks-precious secrets. Montana Outdoors (May/June):2–5.

Dolloff, C. A., and M. L. Warren Jr. 2003. Fish relationships with large wood in small streams. Pages 179–194 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Ernst, A. G., B. P. Baldigo, C. I. Mulvhill, and M. Vian. 2010. Effects of natural-channel-design restoration on habitat quality in Catskill Mountain streams, New York. Transactions of the American Fisheries Society 139:468–482.

Frissell, C. A., and R. K. Nawa. 1992. Incident and causes of physical failure of artificial habitat structures in streams of western Oregon and Washington. North American Journal of Fisheries Management 12:182–197.

Gowan, C., and K. D. Fausch. 1996. Long-term demographic responses of trout populations to habitat manipulation in six Colorado streams. Ecological Applications 6:931–946.

Gregory, S., K. Boyer, and A. Gurnell, editors. 2003. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Hansen, P. L., S. W. Chadde, and R. D. Pfister. 1988. Riparian dominance types of Montana. University of Montana, Montana Riparian Association, Miscellaneous Publication 49, Missoula.

Hunt, R. L. 1976. A long-term evaluation of trout habitat development and its relation to improving managment-related research. Transactions of the American Fisheries Society 105:361–364.

Jones, K. K., K. Anlauf-Dunn, P. S. Jacobsen, M. Strickland, L. Tennant, and S. E. Tippery. 2014. Effectiveness of upstream wood treatments to restore stream complexity and winter rearing habitat for juvenile Coho Salmon. Transactions of the American Fisheries Society 143:334–345.

Klein, L. R., R. Stephen, R. Clayton, J. R. Alldredge, and P. Goodwin. 2007. Long-term monitoring and evaluation of the lower Red River meadow restoration project, Idaho, U.S.A. Restoration Ecology 15:223–239.

Land and Water Consulting. 1999. Kleinschmidt Creek channel restoration, final design. Report to Montana Fish, Wildife and Parks, Missoula.

Larscheid, J. G., and W. A. Hubert. 1992. Factors influencing the size structure of Brook Trout and Brown Trout in southwestern Wyoming mountain streams. North American Journal of Fisheries Management. 12:109–117.

Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. Fluvial processes in geomorphology. Freeman, San Francisco.

Meehan, W. R., editor. 1991. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fiserhies Society, Special Publication 19, Bethesda, Maryland.

MDNRC (Montana Department of Natural Resource Conservation). 2011. Change authorization for water right #76F-98201. MDNRC, Helena.

MDT (Montana Department of Transportion). 2001. Deed of conservation easement #15024. Powell County Court House, Book 105, Deer Lodge, Montana.

Myers, T. J., and S. Swanson. 1992. Variation of stream stablity with stream type and livestock bank damage in northern Nevada. Water Resources Bulletin 28:743–754.

Myers, T. J., and S. Swanson. 1995. Impact of deferred rotation grazing stream charcteristics in central Nevada: a case study. North American Journal of Fisheries Management 15:428–439.

Nagle, G. 2007. Invited commentary for hydrological process evaluation 'natural channel design' stream projects. Hydrological Processes 21: 2539–2545.

NAIP (National Agricultural Imagery Program). 2011. USDA-FSA-APFO NAIP county mosaic. Available: http://datagateway.nrcs.usda.gov/Catalog/ProductDescription/NAIP.html. (December 2014).

Pierce, R. 1991. A stream habitat and fisheries analysis for six tributaries to the Big Blackfoot River. Montana Fish, Wildlife and Parks, Missoula.

Pierce, R., and C. Podner. 2006. The Big Blackfoot River fisheries restoration report for 2004 and 2005. Montana Fish, Wildlife and Parks, Missoula.

Pierce, R., C. Podner, and K. Carim. 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River Basin, Montana. Transactions of the American Fisheries Society 142:68–81.

Pierce, R., C. Podner, L. Marczak, and L. Jones. 2014a. Instream habitat restoration and stream temperature reduction in a whirling disease positive spring creek in the Blackfoot Basin, Montana. Transactions of the American Fisheries Society 143:1188–1198.

Pierce, R., C. Podner, T. Wendt, K. R. Shields, and K. Carim. 2014b. Westslope Cutthroat Trout movements through restored habitat and coanda diversion in the Nevada Spring Creek complex, Blackfoot Basin Montana. Transactions of the American Fisheries Society 143:230–239.

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

Platts, W. S., and R. L. Nelson. 1985 Impacts of rest-rotation grazing on stream banks in forested watersheds in Idaho. North American Journal of Fisheries Management 5:547–556.

Roni, P., editor. 2005. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Roni, P., K. Hansen, and T. Beechie. 2008. Global review of the physical and biological effectiveness of stream rehabilitation techniques. North American Journal of Fisheries Management 28:856–890.

Roper, B. B., D. Konnoff, D. Heller, and K. Wieman. 1998. Durability of Pacific Northwest instream structures following floods. North American Journal of Fisheries Managment 18:686–693.

Rosenfeld, J. S., and L. Huato. 2003. Relationship between large woody debris characteristics and pool formation in small coastal British Columbia streams. North American Journal of Fisheries Management 23:928–938.

Rosgen, D. L. 1994. A classification of natural rivers. Catena 22:169–199.

Rosgen, D. L. 1996. Applied fluvial geomorphology. Wildland Hydrology, Pagosa Springs, Colorado.

Rosgen, D. L. 2007. Rosgen geomorphic channel design. Pages 11-1–11-76 *in* Part 654 stream restoration design national engineering handbook (electronic). U.S. Department of Agriculture, Natural Resources Conservation Service, Washington, D.C. Available: http://directives.sc.egov.usda.gov/OpenNonWebContext.aspx?content=17771.wba. (December 2014).

Rosgen, D. L. 2011. Natural channel design: fundamental concepts, assumptions, and methods. Geophysical Monograph 194:69–93.

Rosgen, D. L, H. L. Silvey, and D. Frantilla. 2008. River stability field guide. Wildland Hydrology, Fort Collins, Colorado.

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

Schmetterling, D. A. 2000. Redd characteristics of fluvial Westslope Cutthroat Trout in four tributaries of the Blackfoot River, Montana. North American Journal of Fisheries Management 20:776–783.

Schmetterling, D. A., and R. W. Pierce. 1999. Success of instream habitat structures after a 50-year flood in Gold Creek, Montana. Restoration Ecology 7:369–375.

Tarzwell, C. M. 1936. Experimental evidence on the value of trout stream improvement in Michigan. Transaction of the American Fisheries Society 66:177–187.

Van Deventer, J. S., and W. S. Platts. 1985. A computer software system for entering, managing, and analyzing fish capture data from streams. U.S. Forest Service Research Note INT 352.

Wesche, T. A. 1980. The WRRI trout cover rating method, development and applications. University of Wyoming, Water Resource Research Institute, Water Resources Series 78, Laramie.

White, S. L., C. Gowan, K. D. Fausch, J. G. Harris, and W. C. Saunders. 2011. Response of trout populations in five Colorado streams two decades after manipulation. Canadian Journal of Fisheries and Aquatice Science 68:2057–2063.

Whiteway, S. A., P. M. Biron, A. Zimmermann, O. Venter, and J. W. A. Grant. 2010. Do instream restoration structures enhance salmonid abundance: a meta-analysis. Canadian Journal of Fisheries and Aquatic Sciences 67: 831–841.

Channel form, spawning riffle quality and macroinvertebrate assemblages in small restored spring creeks of western Montana

Draft Report by

Ron Pierce and Craig Podner,

Montana Fish, Wildlife and Parks, Missoula, Montana

and

Sean Sullivan,

Rhithron Associates, Missoula, Montana

Abstract

Excessive siltation from anthropogenic impacts is one of the most widespread causes of lotic ecosystem degradation in the United States. Spring creeks, located on the bottomlands of many river valleys in western North America, possess high ecological value due largely of their potential to support wild trout populations. Yet spring creeks are highly prone to degradation from sedimentation associated with land and water use practices. This study explores relationships of channel form with fine sediment, trout spawning habitat, and macroinvertebrate communities in four actively restored (reconstructed, revegetated with > 10 years rest from livestock grazing) spring creeks (i.e., treatment group) and four unrestored (impacted by land use, including riparian livestock grazing) spring creeks (i.e., control group) in western Montana, USA. Riffles in restored reaches had lower width-to-depth ratios (10.2±1.8 versus 19.2±4.6), higher velocities (0.71±0.18 versus 0.39±0.09m/s) and lower percentage of fine sediment <6.3mm (25.9±6.6 versus 41.4±6.2) than riffles in control streams. However, macroinvertebrate richness was higher in control streams (41.2±6.8) than the treatment streams (32.2+8.3) due primarily to the increased presence of sediment-tolerant Chironomidae taxa. Primary trout forage as expressed by percent relative abundance of Ephemeroptera, Plecoptera and Trichoptera (EPT) was higher in treatment (64.8±15.1) than control (46.0±5.5) streams due largely to an increased presence of Trichoptera. This study found that restored streams with lower width/depth ratios had higher velocity, larger substrate, higher quality spawning sites, and more trout forage as measured by the percent EPT.

Introduction

Alluvial spring creeks, located in the valley bottoms of many western river valleys, often provide stable flow regimes with high quality spawning and rearing habitat for wild trout populations (Decker-Hess 1987, 1989). Because of their location and prevailing influence of groundwater inflows, they can also provide critical coldwater refugia during the summer and protection from anchor ice formation during winter (Pierce et al. 2014a, 2014b, 2015). However, spring creeks are highly prone to anthropogenic disturbance (e.g., channel alterations, excessive riparian grazing), which can widen channels, increase water temperature, and instream sediment levels, all of which can degrade aquatic habitat and diminish wild trout fisheries (Decker-Hess 1987; Pierce et al. 2015). This sensitivity to disturbance, in part, reflects the location of spring creeks on agricultural bottomlands and site conditions where upwelling areas (spring seeps) tend to saturate stream banks, which are more easily damaged from human-induced disturbance such as intensive livestock grazing (Marlow and Pogacnik 1985; Pierce et al. 2015).

Compared to steeper, higher energy streams, spring creeks, once disturbed, are also slow to recover their natural channel form (e. g., sinuosity, width and depth) because stable flow regimes inhibit channel scour, which reduces the ability to mobilize and deposit instream sediment along channel/floodplain margins. This low energy process inhibits encroachment of streambank vegetation and the timely recovery and maintenance of deeper, narrower and more sinuous channels.

Because of their slow recovery rates and high ecological values, anthropogenically disturbed spring creeks are often the focus of active restoration. Active restoration that increases sinuosity, narrows and deepens spring creek channels can reduce water temperatures, increase sediment transport capacity, coarsen the substrate and greatly improve salmonid habitat especially when essential pool and riffle sequences, along with instream cover and riparian vegetation are also reestablished (Pierce et al. 2014a, 2014b, 2015). However, other aspects of spring creek restoration ecology, such as the post-restoration influence of instream sediment on riffle spawning site quality and invertebrate production are rarely evaluated and poorly understood.

Riffles not only provide essential spawning areas for salmonids, but also essential habitat for the production of macroinvertebrates such as Ephemeroptera, Plecoptera, Trichoptera (EPT) and many other taxa. Though macroinvertebrates provide critical ecological function, (e.g., prey for a myriad of fish and wildlife species), they are also common bioindicators of anthropogenic inputs such as fine inorganic sediment (Angradi 1999; Relyea et al. 2012), hydrologic conditions, and riparian integrity (Allan et al. 1997). Biotic indices of aquatic health have been developed using macroinvertebrates for streams the Northern Rocky Mountain ecoregion (Relyea et al. 2012), as well as streams with the mountains and foothills regions of Montana (Bollman 1998). However, the relevance of biotic indices to spring creeks has not been explored.

This study expands on three prior spring creek restoration studies (Pierce et al. 2014a, 2014b, 2015) by exploring basic relationships of channel form with riffle substrate and macroinvertebrates of four restored (treatment) and four unrestored (control) spring creeks in Western Montana. All streams in this study posses the same basic site conditions and site potential (*see* study area). However compared to control streams, treatment stream were converted from over-grazed or otherwise altered conditions to more natural form. These conversions reestablished more natural pool and riffle sequences with native alluvium, and relied on native vegetation and livestock removal to help recover natural channel stability and to maintain deeper and narrower channels.

Our objectives were to: 1) characterize stream channel features and riffle sediment composition in restored and unrestored streams; 2) examine relationships between sediment levels and trout spawning habitat quality in restored and unrestored streams: and 3) assess riffle associations with sediment-sensitive invertebrates including those preferred as forage items by trout. Our broader aim is to better understand the ecological effects of spring creek restoration for conservation of wild trout populations and critical habitats.

Study Area

The eight small spring creeks in this study are all located on valley floors of three western Montana River valleys (Figure 1). Five spring creeks are located in the Blackfoot River valley, two in the Madison River valley, and one in the Flint Creek valley. These spring creeks all drain alluvial aquifers and support spawning for wild trout populations, including nonnative brown trout (*Salmo trutta*), nonnative rainbow trout (*Oncorhynchus mykiss*), nonnative brook trout (*Salvelinus fontinalis*) and native westslope cutthroat trout (*O. clarkii lewisi*), a Montana species of concern (MNHP 2015). These spring creeks all have a variable history of human disturbance from irrigation, intensive streamside grazing, road encroachment, channelization and drained wetlands. To restore more natural channel form, the four treatment spring creeks in this study were actively converted (i. e., reconstructed) from wide, shallow (e.g., Rosgen C-type) or straightened channels to deep, narrow more sinuous (e.g., Rosgen E-type) channels. Three of the treatment streams in the Blackfoot Basin were reconstructed within the existing channel to deeper, narrower channels (width depth ratios of 19.7-150; Pierce et al. 2013; 2015). The channel of O'Dell Creek on was relocated from a channelized ditch at the toe of the valley wall and reconstructed as new channel within an alluvial terrace. All treatment streams were reconstructed in native alluvium and livestock were removed to recover native riparian vegetation and provide natural channel stability and habitat value. Each reconstructed stream had a minimum 10 years (mean=16 years) of passive vegetative recovery (i.e., livestock exclusion) thereafter (Table 1). The four control streams were unrestored, relatively wide and shallow, and each of these had recent history of channel disturbance from riparian grazing, although two of these (Spring Creek to O'Dell and Spring Creek to Cottonwood Creek) are currently in a state of passive recovery by excluding livestock grazing. All streams were selected for this study because they all 1) fell within a narrow range of low channel gradients (i. e., <1 %), 2) possessed similar physical features necessary for salmonid spawning (i. e., gravel-formed riffles), and 3) the same site potential involving deep, narrow and sinuous channel (i. e., E stream type, Rosgen Classification 1996).

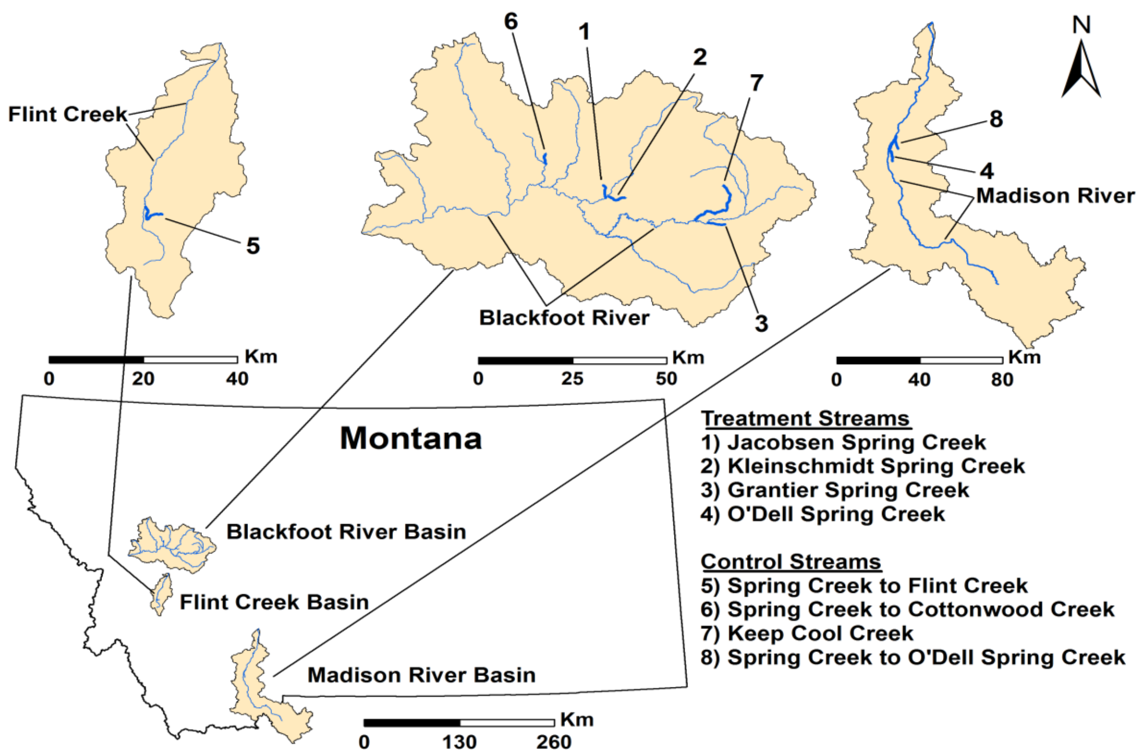


Figure 1. Map of the Montana showing three western Montana river basins and eight spring creeks study sites including four treatment (numbered 1-4) and four control (numbered 5-8) streams. Study streams relate to site conditions in Table 1, instream sediment conditions on Table 2 and macroinvertebrate taxa on Table 3.

Methods

Field Procedures

All eight spring creeks were surveyed for physical and biological characteristics in early spring between 26 March and 20 April 2015. Prior to data collections, we identified a representative reach of each stream and established three floodplain valley cross-sectional transects within each reach. Cross-sections spanned the floodplain and were set perpendicular to the stream at riffle crests with measuring tapes.

To prevent any channel disturbance that could potentially bias invertebrate collections, we started stream surveys by first collecting macroinvertebrates. Invertebrate sampling involved disturbing (kicking) substrates for 15 seconds across a one meter^2 area of each riffle crest and capturing invertebrates in a Wildco 46x20cm rectangular 500 micron net set immediately downstream of the disturbed area. The three replicate invertebrate samples were pooled into a composite sample and immediately preserved in 95% ethanol at streamside.

Sediment samples were then taken from each riffle transect using a McNeil core sampler (McNeil and Ahnell 1954). When selecting sites for coring, we avoided redds if present. Each McNeil core was then pushed approximately 10cm into the riffle substrate and immediately adjacent (within 0.5 m) of each macroinvertebrate sampling site. Following the extraction of substrate, water depth was measured within the core sampler to calculate water volume. A one liter water sample was then immediately taken from the core sampler and placed within an Imhoff cone to estimate the amount of suspended sediment within the core as described by Shepard et al. (1984).

Elevational and distance measurements were taken for each floodplain transect with a laser level. Measurements included floodplain width, bankfull widths and bankfull depths (Rosgen 1996). Elevations were surveyed at the upper- and lower-most riffle crest and distance between the two points was measured (with measuring tapes) along the bankfull line to determine percent channel gradient. Mean stream velocity (m/s) were measured at one riffle transect per stream using a Marsh-McBirney flow meter where cross-section conditions were conducive to laminar flow (Gallagher and Stevenson 1999). We visually estimated percent vegetation cover (i.e., bare ground, grasses and forbs, shrub coverage) to gauge vegetative channel stability within each floodplain transect using modified methods of Stevenson and Mills (1999). Lastly, photo points were taken of each stream reach.

Data summary and analyses

Channel relationships and sediment samples - Prior to analyses of channel form and sediment, we converted bankfull width and mean bankfull depth to width/depth ratios to normalize width and depth relationships (Rosgen 1996). We then averaged the three width/depth ratios to generate a composite ratio for each stream (Table 1). Stream sinuosity was measured for each stream reach from 2014 National Agricultural Imagery Program high-resolution (1m) imagery using ArcMap (version 10.1) GIS.

Sediment core samples were oven-dried and shaken through sieves containing 75, 50, 25, 12.5, 6.3, 4.75 2.36, 0.85, 0.075 mm mesh screens. Substrates retained within each sieve were weighed to the nearest gram using an A&D SK-10KWP digital scale. Extremely fine sediment (<0.075mm) and the sediment calculation within the Imhoff cone and core sampler were added to the dry weight of 0.075 sieve (Shepard et al. 1984). The three substrate samples were then pooled by size class into a composite sample for each study stream. For each of the nine particle size classes in the composite sample, riffle substrate composition was calculated as a percentage of dry weight. To summarize (and visualize) differences for the nine sediment size classes, cumulative percent curves were plotted for the total weight of all substrates by sieve size class for treatment and control groups.

Treatment streams (Map ID)	Year restored	Floodplain width (m)	Bankfull width (m)	Bankfull depth (m)	W/D ratio	Sinuosity	Channel slope (%)	Velocity (m/s)	Grasses (%)	Shrubs (%)	Bare ground (%)
Jacobsen Spring Creek (1)	2005	14.8	3.2	0.27	11.8	1.4	0.35	0.61	89	9	2
Kleinschmidt Creek (2)	2001	26.8	3.6	0.30	11.6	1.6	0.39	0.61	97	3	0
Grantier Spring Creek (3)	1991	16.2	3.9	0.30	12.4	2.3	0.32	0.57	93	5	2
O'Dell Spring Creek (4)	1998	7.1	1.9	0.40	5.2	1.3	0.46	1.05	92	8	0
	mean	16.2	3.2	0.32	10.3	1.6	0.38	0.71	93	6	1
	median	15.5	3.4	0.30	11.7	1.5	0.37	0.61	92	7	1
Control streams (map ID)											
Spring Creek to Flint Creek (5)	na	10.6	5.3	0.23	27.5	1.5	0.55	0.23	98	0	2
Spring Creek to Cottonwood Creek (6)	na	13.3	4.3	0.27	16.4	1.6	0.48	0.43	75	25	0
Keep Cool Creek (7)	na	8.2	5.2	0.30	18.4	1.4	0.23	0.51	42	0	58
Spring Creek to O'Dell (8)	na	12.9	2.9	0.23	14.6	1.2	0.48	0.42	94	0	6
	mean	11.3	4.4	0.26	19.2	1.4	0.43	0.39	77	6	16
	median	11.8	4.8	0.25	17.4	1.45	0.48	0.42	84	0	4

Table 1. Study streams: Includes summaries of floodplain and channel cross-sections, along with sinuosity, channel slope, mean velocity and riparian vegetative cover for four fully restored spring creeks and four unrestored spring creeks. Numbers adjacent to the stream name refer site locations on Figure 1.

To relate substrates to salmonid spawning site quality, sediment from each composite stream sample was summarized into two size groups of fine sediment: 1) <0.85mm (silt and fine sand), and 2) <6.3mm (silt to fine gravel), both of which are common to evaluations of trout embryo survival and fry emergence (Weaver et al. 1993, Waters 1995, Kondolf et al. 2008). For each composite sediment sample, we also calculated a geometric mean and fredle index score to further identify spawning site quality (Lotspeich and Everest 1981) relative to published salmonid egg survival relationships (Waters 1995, Kondolf et al. 2008). As described by Lotspeich and Everest (1981), the fredle index is a multi-metric measure of central tendency (geometric mean) and dispersion (the 25th and 75th percentile), which categorically ranks spawning substrates on a scale from 0-11. The fredle Index can estimate embryo survival rates of 0-20% with a score of 1, as compared to survival rates of 80-90% for a score of 10 (Waters 1995). Like the two measures of fine sediment (<0.85mm and <6.3mm), the geometric mean and fredle index are widely accepted techniques for evaluations of spawning site substrates (Shirazi and Seim 1981; Waters 1995, Kondolf et al. 2008).

Enumeration, identification and summary of macroinvertebrates - Macroinvertebrate samples were homogenized in the laboratory and subsamples were extracted containing a minimum of 500 organisms using Caton sub-sampling devices (Caton 1991), divided into 30 grids, each approximately 6 cm by 6 cm. Organisms were individually examined and taxonomic determinations were made to the lowest possible taxonomic level, usually genus and species.

To broadly identify macroinvertebrate communities, mean taxa diversity and percent relative abundance by non-insect and insect orders were calculated for treatment and control groups (Figures 3 and 4). Eight metrics of specific macroinvertebrate taxa were also calculated for all individual streams: 1) total taxa richness, 2) EPT richness, 3) EPT percent, 4) Trichoptera richness, 5) clinger richness (Clingers are defined here as organisms with behavioral or morphological adaptations to assist in the attachment to surfaces in stream riffles (Merritt et al. 2008), 6) sediment-tolerant richness, 7) number Chironomidae taxa, and 8) percent Chironomidae. These groups were selected because they broadly describe macroinvertebrate community composition (categories 1-3), sediment-intolerant groups (categories 5-6) as well as sediment-tolerant groups (categories 6-8) (Lenat et al. 1981; Relyea et al. 2012). Macroinvertebrate indices of biotic integrity using the Montana Mountain Valleys and Foothill

Prairies (MVFP) Bioassessment index (Bollman 1998) and the Fine Sediment Biotic Index (FSBI) for the Northern Rockies ecoregion (Relyea et al. 2012) were calculated for each stream.

Statistical methods - We used a paired watershed study design to examine differences between treatment and control stream. To validate physical habitat differences between the treatment and control streams, we used paired Mann-Whitney U tests for the following habitat and sediment metrics: sinuosity, width to depth ratio, stream velocity, percent fine sediment <0.85mm, percent fine sediment <6.3mm, geometric mean, fredle index scores and stream gradient. A non-parametric test was chosen due to these data failing to meet the assumptions necessary for parametric statistical tests, most notably unequal variance and small sample sizes. Therefore, central tendency values for the treatment groups are reported as median values, unless otherwise stated. Spearman Rank correlations were used to describe the relationships among these variables and macroinvertebrate metrics (see above). Due to the small sample sizes and the exploratory nature of this study, differences were considered significant at the alpha level of 0.10. All statistical analyses were performed using the computer programming language R (R Development Core Team 2009).

Treatment streams	Percent substrate by size class (mm)									Fredle index calculations					Percent Fine sediment	
	<0.075	0.85	2.36	4.75	6.3	12.5	25	50	75	Geometric mean (mm)	D25 (mm)	D75 (mm)	Sorting Coeff.	Fredle Index	<0.84mm	<6.35mm
Jacobsen Spring Creek	10.4	3.9	2.7	1.4	5.5	33.1	37.3	5.9	0.0	13.7	12.9	37.8	1.7	8.0	10.4	18.4
Kleinschmidt Creek	7.5	5.9	5.9	2.6	8.5	10.3	11.9	17.9	26.6	18.6	6.4	77.7	3.5	5.3	7.5	24.8
Grantier Spring Creek	13.9	9.2	9.5	5.1	15.8	20.7	15.7	10.0	0.0	7.7	2.8	26.6	3.1	2.5	13.9	37.7
O'Dell Spring Creek	12.0	4.8	3.9	2.2	9.7	18.5	38.6	10.3	0.0	13.1	7.6	41.1	2.3	5.7	12.0	22.9
mean	11.0	5.9	5.6	2.8	9.9	20.7	25.9	11.0	6.7	13.3	7.4	45.8	2.7	5.4	11.0	26.0
median	11.2	5.4	4.9	2.4	9.1	19.6	26.5	10.2	0.0	13.4	7.0	39.5	2.7	5.5	11.2	23.9
Composite	10.9	6.8	5.6	2.9	10.0	20.3	24.9	11.2	7.2	12.6	5.6	44.1	2.8	4.5	11.4	24.0
Control streams																
Spring Creek to Flint Creek	19.5	10.2	7.3	3.2	9.9	1.7	21.0	3.9	8.1	7.4	1.7	35.1	4.6	1.6	19.5	40.2
Spring Cr to Cottonwood Cr	10.7	9.0	9.9	4.7	15.3	18.6	15.3	5.7	10.8	9.8	3.6	36.7	3.2	3.1	10.7	34.3
Keep Cool Creek	15.4	11.3	8.3	3.7	14.2	16.6	24.5	6.0	0.0	7.4	2.1	31.1	3.8	1.9	15.4	38.7
Spring Creek to O'Dell	34.2	7.5	7.4	3.4	12.4	19.5	15.7	0.0	0.0	3.8	0.6	19.2	5.5	0.7	34.2	52.5
mean	20.0	9.5	8.2	3.8	13.0	14.1	19.1	3.9	4.7	7.1	2.0	30.5	4.3	1.8	20.0	41.4
median	17.5	9.6	7.9	3.6	13.3	17.6	18.4	4.8	4.1	7.4	1.9	33.1	4.2	1.8	17.5	39.5
Composite	19.5	9.5	8.3	3.8	13	17.9	19	4.1	5.4	6.9	1.8	29.9	4.1	1.7	19.2	40.7

Table 2. Riffle substrate by percent of dry weight, fredle indices and related metrics and two primary measures of fine sediment associated with spawning success.

Results

Channel form and riffle sediment - All composite floodplain and channel transect measurements for individual streams are shown in Table 1. Floodplain widths ranged from 6.4 to 31.1m, surveyed reaches ranged in length from 33.2 to 153.0m between upper and lower riffle crests and bankfull widths ranged from 1.7 – 8.4m within the floodplain cross-sections. Survey data showed the treatment group had lower width/depth ratios (11.7 versus 17.4; P=0.02857) and higher velocities (0.61 versus 0.42 m/s; P=0.0294). Sinuosity (1.5 versus 1.4; P=0.662) and percent channel slope (0.37% versus 0.48%; P=0.3094) were insignificantly different between treatment and groups.

Analyses of fine sediment found insignificantly lower levels of fine sediment in the <0.85mm size class for the treatment group versus the control group (Table 2; 11.2% and 17.5%, respectively; P=0.114), and significantly lower levels of <6.3mm size sediment in the treatment group versus the control group (23.9% and 39.5% respectively; P=0.057). Treatment streams had higher geometric mean than the control streams (13.4 mm and 7.4 mm, respectively; P=0.059). Fredle index scores in the treatment group (median=5.5, range=2.5-8.0) were significantly different (P=0.0.059) than the scores of the control group (median=1.8, range=0.7-3.1) (Table 2). A composite cumulative percent curve of all McNeil core sampled substrates by size class showing coarser sediment in the treatment group compared versus the control groups is shown in Figure 2.

Treatment streams	Taxa richness	EPT richness	EPT percent	Sediment tolerant richness	Tricoptera richness	Clinger richness	% chironomidae	# Chironomidae taxa	FSBI	MVFP index (%)
Jacobsen Spring Creek	47	15	66.4	0	7	17	10.2	14	80	89.9
Kleinschmidt Creek	26	8	42.7	1	4	9	49.3	12	35	50
Grantier Spring Creek	24	8	88.8	1	4	8	4.2	8	20	44.4
O'Dell Spring Creek	32	11	61.2	2	6	16	2.9	9	70	55.6
mean	32.2	10.5	64.8	1.0	5.2	12.5	16.6	10.8	51.2	60.0
median	29.0	9.5	63.8	1.0	5.0	12.5	7.2	10.5	52.5	52.8
Control streams										
Spring Creek to Flint Creek	41	7	37.7	3	2	7	43	18	35	38.9
Spring Creek to Cottonwood Creek	52	14	52.3	5	5	16	32.8	21	80	83.3
Keep Cool Creek	41	13	43.2	1	6	14	46.8	14	55	77.8
Spring Creek to O'Dell	31	7	51	5	3	7	4	9	25	38.9
mean	41.2	10.2	46.0	4.0	4.0	11.0	31.6	15.5	48.8	59.7
median	41.0	10.0	47.1	4.0	4.0	10.5	37.9	16.0	45.0	58.4

Table 3. Macroinvertebrate sampling results for eight taxa groups and indices of habitat integrity (MVFP) and sediment (FSBI).

Macroinvertebrates - A broad comparison of taxa diversity and relative abundance by non-insect and insect orders for treatment and control groups are shown in Figures 3 and 4, respectively. Individual stream summaries of the eight specific invertebrate groups (taxa richness, EPT richness and percent, number sediment tolerant taxa, Trichoptera richness, clinger richness, number and percent chironomids) along with the MVFT and FSBI index scores are shown in Table 3.

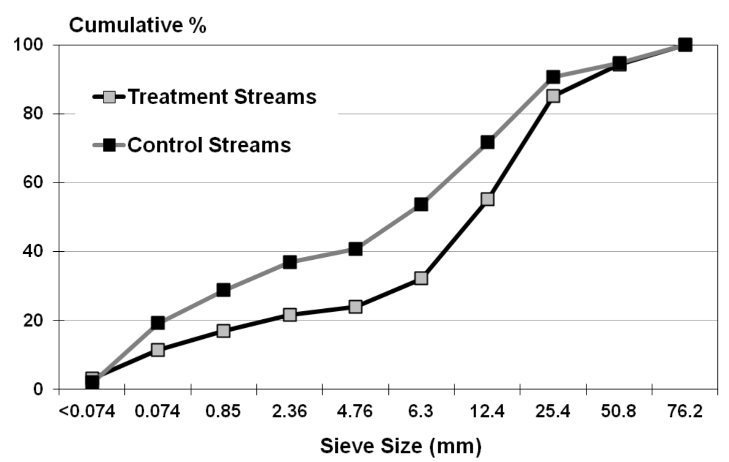


Figure 2. Cumulative percent curve for all substrates (total dry weight by sieve class) by treatment and control streams.

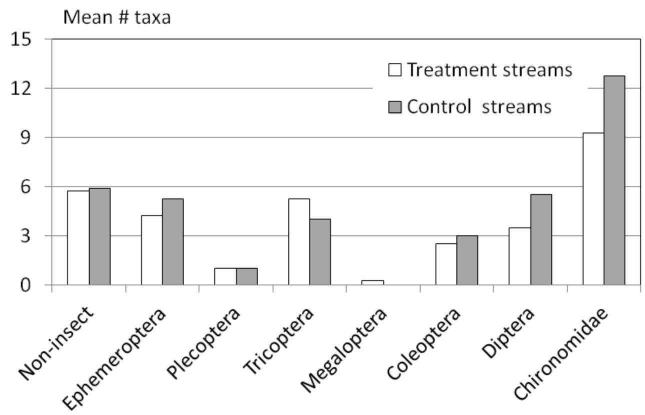
No significant differences were found between treatment groups for all invertebrate metrics (Table 4). However, there was a strong positive correlation between the number of sediment tolerant taxa and the relative percent of sediment <0.85mm (rho=0.901; P=0.002) and a negative correlation between the relative percent of <6.3mm sediment and the number of Clinger taxa (rho=-0.650; P=0.080). In addition, two invertebrate metrics (EPT richness and Clinger Richness) were positively correlated with the relative percent composition of shrubs at each site (rho=0.612; P=0.10) and (rho=0.621; P=0.099), respectively. The Fine Sediment Biotic Index (FSBI) showed a moderately strong negative correlation with percent fine sediment < 6.83mm (rho=-0.622; P=0.097), suggesting that although the FSBI model was not calibrated for this large of sediment size fraction the model is responding in the predictive fashion for increased sediment. The MVFP index showed no significant relationships to the measured sediment variables.

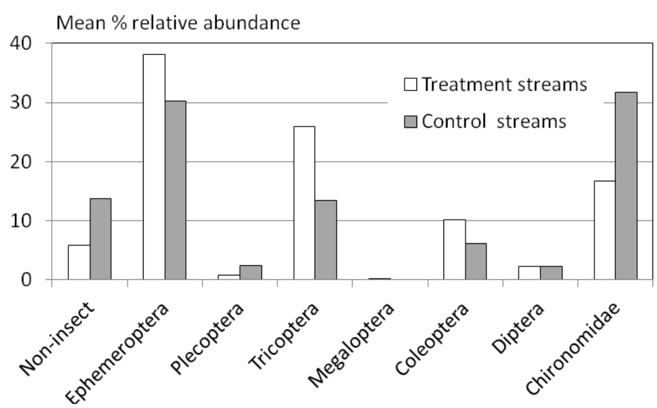
Discussion

This study had several limitations including small sample size and pretreatment data limited to width depth ratios for three restored streams in the Blackfoot River Basin. With these limitations in mind, this study focused on relationships of channel form with fine sediment, trout spawning habitat, and macroinvertebrate communities for eight small spring creeks, all with similar site potential. Riffles in restored streams had lower width-to-depth ratios, higher mean velocities, larger substrate and lower percentage of fine sediment and higher quality of spawning riffles than disturbed streams. Though macroinvertebrate richness was higher in control streams, not surprisingly, increases were due to the increased presence of sediment-tolerant Chironomidae and diptera taxa. Primary trout forage as expressed by percent relative abundance of EPT was higher in treatment streams due largely to an increased presence of Trichoptera.

<u>Physical assessment</u>

Channel form – Prior to restoration, treatment streams were wider, shallower had lower sinuosity, or otherwise altered compared to control streams. After restoration all treatment streams were converted to low (<12) width/depth ratios, high sinuosity (>1.4) and floodplain connection (Rosgen 1996; Table 1). As described by Rosgen (1996), deep and narrow streams are hydraulically efficient for sediment transport and represent the evolutionary end-point for low-gradient (<2%), vegetatively-controlled channels. Compared to restored streams, impacted streams in the control group were wider, shallower as primarily expressed by high (>14) width/depth ratios, lower in sinuosity and had more bare ground. As with the control group, agricultural activities, including livestock damage to stream banks (e. g., hoof shear, bank sloughing), and other channel-altering activities have been implicated in the widening and degradation of many spring creeks in Blackfoot Valley (Pierce et al. 2013, 2014a, 2015) and across western Montana (Decker-Hess 1987, 1989). Similar to the control group, the three treatment streams reconstructed within preexisting channels had high pre-treatment width/depth ratios (i.e., 19.7-150; Pierce et al. 2013; 2015). Anthropogenic channel widening (e. g., livestock damage) can introduce sediment, reduce sediment transport capacity and thereby induce sediment deposition (Pritchard et al. 1993; Rosen 1996; George et al. 2002; Pierce et al. 2015). Conversely, the conversion of over-widened streams to deep, narrow streams can increase sediment transport capacity, coarsen the substrate, reduce summer water temperatures and increase trout abundance and return stream to a much more





Figures 3 and 4. Mean taxa richness of non insect and insect orders for four treatment and four control streams (top), and mean percent relative abundance of non insect and insect orders for treatment and control streams (bottom). These graphs also separated chironomidae taxa from other Diptera taxa.

natural state (Rosgen 1996; Pierce et al. 2014a, 2014b, 2015). Consistent with this conversion, the treatment group in this study had lower width/depth ratios, higher mean velocity, coarser sediment and higher quality of spawning sites than control streams.

	Sediment group						Biotic indices		Invertebrate taxa group							
	Geometric Mean	<0.85mm	<6.35m	Fredle index	D25	D75	FSBI	MVFP	Clinger Richness	EPT Percent	EPT Richness	Sediment Tolerant Percent	Sediment Tolerant Richness	Trichoptera Richness	Taxa Richness	Chironomidae Percent
Fine (<0.85mm)	-0.797**															
Coarse (<6.35m)	-0.882***	0.848***														
Fredle	-0.383	-0.033	0.176													
D25	-0.365	-0.05	0.183	0.987***												
D75	0.055	-0.203	-0.34	0.521	0.406											
FSBI	0.358	-0.505	-0.622*	-0.155	-0.247	0.455										
MVFP	-0.443	-0.056	0.342	0.508	0.546	0.026	-0.173									
Clinger Richness	0.364	-0.549	-0.651*	0.053	-0.038	0.548	0.961***	-0.202								
EPT Percent	-0.04	-0.17	-0.189	0.631*	0.689*	0.28	-0.052	0.296	0.11							
EPT Richness	0.243	-0.503	-0.53	0.024	-0.04	0.303	0.912***	-0.125	0.944***	0.095						
Sediment Tolerant Percent	-0.467	0.831**	0.443	-0.15	-0.177	0.023	-0.243	-0.35	-0.274	0.071	-0.332					
Sediment Tolerant Richness	-0.615	0.901***	0.709**	-0.268	-0.305	-0.129	-0.33	-0.059	-0.459	-0.307	-0.491	0.831**				
Trichoptera Richness	-0.622*	0.168	0.573	0.694*	0.711**	0.069	-0.433	0.833**	-0.381	0.143	-0.306	-0.277	0			
Taxa Richness	-0.089	-0.158	-0.142	-0.303	-0.363	0.013	0.763**	0.122	0.622*	-0.304	0.741**	-0.19	-0.04	-0.133		
Chironomidae Percent	0.23	-0.298	0.006	-0.373	-0.363	-0.442	-0.012	0.045	-0.104	-0.750**	0.007	-0.657*	-0.267	0.143	0.248	
Chironomidae Richness	0.126	-0.397	-0.211	-0.366	-0.399	-0.139	0.639*	0.235	0.482	-0.461	0.603	-0.538	-0.238	-0.02	0.891***	0.603

*=P<0.10 **=P<0.05 ***= P<0.01

Table 4. Results of tables showing significant (P<0.10) correlations between sediment values, indices of habitat integrity (MVFP) sediment (FSBI), and 8 taxa groups of macroinvertebrates

Sediment and spawning conditions – All measures of riffle sediment in this study reveal high spawning site quality in treatment streams versus control streams. Because no single metric fully describes relationships of sediment and spawning success (Young et al. 1991; Waters 1995; Kondolf et. al. 2008), we used multiple measures of riffle substrate composition to evaluate spawning site quality. Among measures of fine sediment, substrates <0.85mm are not only indicators of land use intensity (Beschta 1982), but also among the most damaging to spawning success because these silt and fine sand particles can reduce permeability and limit oxygen exchange within the redd (Gard 2002). When these fine sediments (<0.85mm) in the redd approaches 20% of the substrate composition, developing embryos can asphyxiate resulting in high embryo mortality (Reiser and White 1988). Despite slightly higher channel slopes in control streams (median=0.37% versus 0.48%), this study found a proportionally higher level of fine sediment <0.85mm in control (median=17.5%) versus treatment streams (median=11.2%), though these relationships were statistically insignificant. Though 17.5% fine sediment <0.85mm may place the control group at a level approaching significant egg mortality (i. e., 20%; Reiser and White 1988), redd construction is a "cleansing" process that reduces fine sediment within the actual egg pocket and thereby mediates sediment-related mortality to eggs and alevins (Young et al. 1989). Because we avoided redds in our sediment sampling, actual levels of fine sediment are likely higher in our sediment samples, and mortality relationships likely lower, than those within actual redds.

Sediment <6.3mm may permit water to flow through the redd, but limit fry emergence through an overhead substrate (Shepard et al. 1984; Weaver and Fraley 1993; Kondolf 2000). Sediment in the <6.3mm size class was higher the control group with a median of 39.5% versus 23.9% for the treatment group. Using these values specific to westslope cutthroat trout, predicted survival to emergence would approximate 20% and 40%, respectively (Weaver and Fraley 1993). In the case of Grantier Spring Creek, a restored westslope cutthroat spawning stream (Pierce et al. 2012), the composite width/depth ratio was relatively high (12.4) and emergence success was calculated as relatively low (23%) based on a fine sediment <6.3mm level of 38% using the Weaver and Fraley (2012) equation. In the example of Grantier Creek, where spring creek restoration would be considered a native trout recovery action, active restoration that further reduces width/depth ratios to increase sediment transport capacity would likely improve spawning site quality and spawning success.

Unlike measures of fine sediment, the geometric mean and fredle index scores both consider the entire particle distribution, and thus provides a general measure of permeability, porosity and size of spawning substrates (Bjornn and Reiser 1991; Waters 1995). For our control group, the geometric grand mean of 7.1mm and associated fredle index of 1.8 both approximate 25% egg survival for salmonids (Waters 1995). This compares to the geometric grand mean of 13.3mm and fredle index of 5.4 for the treatment group (Table 2) and egg survival rates of approximately 50% (Waters 1995). The low quality of spawning site in the control group clearly indicate restoration could improve spawning success by increasing sediment transport capacity. This may especially apply when considered for streams with recruitment limitations brought on by sediment-induced spawning limitations (Reiser and White 1988; Young et al. 1991: Weaver and Fraley 1993). An example of this is the side-by-side comparison of two streams in the O'Dell spring creek drainage. Here, the restored stream had a width/depth ratio of 5.2 versus 14.6 in the control, mean velocity of 1.05 versus 0.42 m/s and fredle index scores of 5.7 versus 0.7 in the control.

Macroinvertebrates - With the exception of sediment-sensitive taxa, most macroinvertebrate groups were not easily explained by measures of fine sediment. This relates to the ecological complexity of lotic systems (Lancaster and Belyea 2006), whereby variable velocity, temperature, stream size, organic detritus, food-web pathways, as well as other substrate relationships all influence invertebrate community composition (Hynes 1970; Mihuc et al. 1996; Stagliano 2005).

Although statistically insignificant, taxa richness was noticeably higher in the control group than the treatment group. These results contrast with studies that report elevated anthropogenic sediment leads to reductions in invertebrate populations (Waters 1995). Not surprisingly, higher mean taxa richness in the control streams corresponds with an increased richness of sediment-tolerant taxa including Chironomidae and other diptera taxa. These relationships suggest anthropogenic alteration may actually increase species diversity in certain spring creeks, or other sediment enriched environments, as predicted by the Intermediate Disturbance Hypothesis (Townsend et al. 1997). In prior studies, sediment enrichment has been shown to decrease species diversity in streams (Hynes 1970; Debrey and Lockwood 1990; Gard 2002). However these studies make very little, if any, distinction between spring creeks and snowmelt-dominated streams.

EPT taxa richness was similar for both treatment and control streams; however, percent EPT was higher in treatment (63.8%) versus control (47.1%) streams though these differences were statistically insignificant. The observed higher percent EPT for the treatment group relate to higher numbers of *Baetis tricaudatus* and *Ephemerella excrucians* mayflies and higher numbers of *Micrasema* and *Hydroptila* caddis flies. *Baetis tricaudatus* is a ubiquitous mayfly with little indicator value; whereas, *Ephemerella excrucians* is a shredder and clinger, indicative of riparian health, functional hydrology and stony substrates (Merritt et al. 2008 and Hubbard and Peters, 1978). Of the two caddisflies, *Hydoptilla* are sediment-tolerant; whereas, *Micrasema* are clingers and indicative of clean, stony substrate. Interestingly, *Micrasema* were abundant in the restored section of O'Dell Creek (n=177) where fine sediment (<6.3mm) comprised 23% of the substrate, but absent from the adjacent control stream where fine sediment (<6.3mm) comprised 52% of the substrate. As shown in other studies (Waters 1995; Cummings and Lauff 1969), higher prevalence of EPT in treatment streams were consistent with increased interstitial space as a function of larger particle size. The higher percent of EPT as indicated in this study is also important because EPT are considered among the most productive, preferred and available aquatic prey items for salmonids (Waters 1995). Conversely, the percent Chironomidae in the control group (median = 37.9%) was over five times greater that of the treatment group (median = 7.2%). Prior studies recognize Chironomidae taxa as sediment-tolerant (burrowers) and their richness and abundances often relate to elevated levels of anthropogenic sediment (Waters 1995; Relyea et al. 2012).

In addition to invertebrate relationships with instream conditions, we also observed positive relationships between relative percent composition of shrubs with EPT richness and with clinger richness. These relationships suggest the stability and vigor of riparian vegetation can help favor or provide indicator value to certain invertebrate communities associated with low-sediment and stony substrates, hydrologic conditions and riparian integrity (Allan et al. 1997).

Though sediment-sensitive assemblages were better represented in treatment streams, biotic indices of aquatic health revealed no difference between the treatment and control groups, and suggest only moderate aquatic health for both groups. Other studies have found similar “non-relationships” between sediment and whole macroinvertebrate community indices (Relyea et al. 2012, Stagliano 2006). These biotic indices were developed from steeper, higher elevation basin-fed streams with run-off hydrology, higher gradients, more variable stream velocities and stony substrates, all of which tend to greatly favor EPT taxa (Bollman 1998). Whereas, spring creeks are generally low-gradient, fine-grained, and groundwater-fed and possess their own distinct community composition (Stagliano 2005). In this study, Jacobsen Spring Creek had the highest MVFP scores and the lowest levels of fine sediment (<6.3mm); it originates from multiple spring creeks and flows within a forested canopy. Whereas, Grantier Spring Creek had the lowest MVFP score and the highest percent fine sediment (<6.35mm) of the treatment streams; it originates from springs in glacial potholes and flows primarily though sedge and shrub-dominated wetland vegetation. Though both streams were restored to emulate their natural site potential, moderate MVFP scores for fully restored spring creek point to the need for more relevant and refined biotic indices for spring creeks. The FSBI was created using data from a variety of habitats, slopes and stream sizes / channel morphology. Though the FSBI did correlate well to composition of sediment in this study, more sampling and experimentation is warranted to fully understand macroinvertebrate communities and assess biotic indices in novel spring creek environments.

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Figure 5. Photos of all streams surveyed in this study. The top four streams are treatment streams and bottom four are control streams.

Literature cited

Allan, J. D., D. L. Erickson and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. *Freshwater Biology* 37:149-162.

Angradi, T. R. 1999. Fine sediment and macroinvertebrate assemblages in Appalachian streams: A field experiment with biomonitoring applications. *Journal of the North American Benthological Society*. 18(1): 49-66

Barbour, M.T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency, Washington, D.C.

Baxter, C. V. and F. R. Hauer. 2000. Geomorphology, hyporheic exchange, and selection of spawning habitat by bull trout (*Salvelinus confluentus*). *Canadian Journal of Fisheries and Aquatic Sciences* 57:1470-1481.

Beschta, R. L. 1982. Comment on "Stream system evaluation with emphasis on spawning habitat for salmonids" by M. A Shirazi and W. K. Seim. Water Resources Research 18:1292-1295

Bollman, W. 1998. Improving stream Bioassessment methods for the Montana valleys and foothill prairies ecoregion. Master's Thesis (MS). University of Montana. Missoula, Montana.

Cairns, J., Jr. and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. Chapter 2 *in* Rosenberg, D. M. and V. H. Resh, editors. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York.

Cummins,K.W. and G.H. Lauff.1969. The influence of substrate particle size on the microdistribution of stream macrobenthos. Hydrobiologia.34:2:145-181

Debrey, L. D. and J. A. Lockwood. 1990. Effects of sediment and flow regime on the aquatic insects of a high mountain stream. Regulated Rivers: *Research & Management* 5(3): 241-250

Decker-Hess, J. 1985. An inventory of Montana spring creeks. A report to the American Fisheries Society, Bethesda, Maryland. Montana Fish, Wildlife and Parks, Helena, Montana.

Decker-Hess, J. 1987. Spring creeks - precious secrets. *Montana Outdoors*, May/June: 2-5.

Fore, L. S., J. R. Karr and R. W. Wisseman. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. *Journal of the North American Benthological Society* 15(2): 212-231.

Gard, M. F. 2002. Effects of Sediment Loads on the Fish and Invertebrates of a Sierra Nevada River, California. *Journal of Aquatic Ecosystem Stress and Recovery* 9(4): 227-238

Hynes, H. B. N. 1970. The Ecology of Running Waters. Liverpool University Press.

Hughes, R. A., Byrnes, J. E. Kimbro, D. L. Stachowicz, J. J. 2007. Reciprocal relationships and potential feedbacks between biodiversity and disturbance. *Ecology Letters* 10 (9): 849–64.

Hubbard, M.D. and W.L. Peters. 1978. Environmental requirements and pollution tolerance of Ephemeroptera. EPA-600/4/78-061. 460 pages.

Karr, J. R. and E. W. Chu. 1999. Restoring Life in Running Waters: Better Biological Monitoring. Island Press. Washington D.C.

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

Kondolf, G. M., S. C. Williams, T. C. Horner, and D. Milan. 2008. Assessing the physical quality of spawning habitat, pages 249-274, *in* D. A. Sear and P. Devries, editors. Salmonid spawning habitat in rivers: physical controls, biological responses and approaches to remediation. American Fisheries Society, Symposium 65, Bethesda, Maryland.

Lancaster, J. and L. R. Belyea. 2006. Defining the limits to local density: alternate views of abundance environment relationships. *Freshwater Biology* 51:783–796.

Lenat, D. R., D. L. Penrose, and K. W. Eagleson. 1981. Variable effects of sediment addition on stream benthos. *Hydrobiologia* 79:187-194.

Lotspeich, F. B. and F. H. Everest. 1981. A new method of reporting and interpreting textural composition of spawning gravel. USDA, Forest Service, Research paper PNW-369.

Marlow and Pogacnik. 1985. Paper presented at the North American Riparian Conference. The University of Arizona, Tucson, April 15-19, 1985.

Merritt, R.W., K.W. Cummins, and M.B. Berg.2008. Ecology and distribution of aquatic invertebrates, pages 105-122,*in* Merritt, R.W., K.W. Cummins, and M.B. Berg, editors. An introduction to the Aquatic Insects of North America. Kendal-Hunt publishing, Dubuque, Iowa

McNeil, W. J., and W. H. Ahnell. 1964. Success of pink salmon spawning relative to sized of spawning bed materials. U.S. Fish and Wildlife Service Special Scientific Report Fisheries 469.

Mihuc, T.M., G. W.. Minshall, and J. R. Mihuc. 1996 Species-environmental relationships among filter-feeding caddisflies (Trichoptera: Hydopsychidae) in Rocky Mountain Streams. *The Great Basin Naturalist* 56:287-293.

MNHP 2015. Montana Natural Heritage Program and Montana Fish, Wildlife and Parks. MT Animal Species of Concern Report. [web application] http://mtnhp.org/SpeciesOfConcern/ Retrieved 2/5/2015

Neudecker, R. A., T. McMahon and E. R. Vincent. 2012. Spatial and temporal variation of whirling disease risk in Montana spring creeks. *Journal of Aquatic Animal Health* 24:201-212.

Pierce, R., C. Podner and K. Carim. 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River Basin, Montana**.** *Transactions of the American Fisheries Society* 142:68-81.

Pierce, R., C. Podner, T. Wendt, R. Shields and K. Carim. 2014a. Westslope cutthroat trout movements through restored habitat in the Nevada Spring Creek complex, Blackfoot Basin, Montana. *Transactions of the American Fisheries Society* 143:230-239.

Pierce, R., C. Podner, L. Marczak and L. Jones. 2014b. Instream habitat restoration and stream temperature reduction in a whirling disease positive spring creek in the Blackfoot River Basin, Montana. *Transactions of the American Fisheries Society* 143:1188–1198.

Pierce, R., C. Podner and L. Jones. 2015. Long-term increases in trout abundance following channel reconstruction, instream wood placement, and livestock removal from a spring creek in the Blackfoot Basin, Montana. *Transactions of the American Fisheries Society* 144:184-195.

Pritchard, D., F. Barret, H. Berg, W. Hagenbuck, R. Krapf, R. Leinard, S. Leonard, M. Manning, C. Noble, D. Tippy. 1993. Riparian area management: A user guide to assessing proper functioning condition. Technical Reference 1737-9. USDI Bureau of Land Management Service Center. Denver, Colorado. USA. 109 pp.

R Development Core Team. 2009. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna. ISBN 3-900051-07-0. Available at http://CRAN.R-project.org.

Reiser, D. W., and R. G. White. 1988. Effects of two sediment size-classes on survival of steelhead and chinook salmon eggs. *North American Journal of Fisheries Management* 8:432-437.

Relyea, C. D., G. W. Minshall, and R. J. Danehy. 2012. Development and validation of an aquatic fine sediment biotic index. *Environmental Management* 49(1):242-252.

Rosgen, D. 1996. Applied River Morphology. Wildland Hydrology, Pagosa Springs Colorado.

Shepard, B., S. Leathe, T. Weaver and M. Enk. 1984. Monitoring levels of fine sediments within tributaries to Flathead Lake and impacts of fine sediment to bull trout recruitment. Pages 146-156 *in* F. Richardson and R.H. Hamre, editors. Wild Trout III. Federation of Fly Fishers and Trout Unlimited, Virginia.

Shirazi, M. A., and W. K. Seim 1981. Stream system evaluation with emphasis on spawning habitat for salmonids. *Water Resources Research* 17:592-594.

Stagliano, D. M. 2005. Aquatic community classification and ecosystem diversity in Montana's Missouri River watershed. Prepared for Bureau of Land Management. Montana Natural Heritage Program, Helena, Montana.

Stagliano, D. M. 2006. Aquatic macroinvertebrate survey and assessment of Smith Creek, Park County, MT. Report to the USFS Northern Region Livingston Ranger District. Montana Natural Heritage Program, Helena, Montana.

Stribling, J. B., S. R. Moulton II, and G. T. Lester. 2003. Determining the quality of taxonomic data. *Journal of the North American Benthological Society*. 22:4:621-631.

Townsend, C. R., M. R. Scarsbrook, and S. Doledec. 1997. The intermediate disturbance hypothesis, refugia, and biodiversity in streams. Limnology and Oceanography 42(5):938-949.

Waters, T. F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society Monograph 7.

Weaver, T. M. and J. J. Fraley. 1993. A method to measure emergence success of westslope cutthroat trout from varying substrate compositions in a natural stream channel. *North American Journal of Fisheries Management* 13:817-822.

Wood, P. J. and P. D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. *Environmental Management* 21(2): 203-217

Young, M. K., W. A. Hubert and T. A. Wesche. 1991. Selection of measures of substrate composition to estimate survival to emergence of salmonids and to detect changes in stream substrates. *North American Journal of Fisheries Management* 11:339-346.

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ARTICLE

Does Whirling Disease Mediate Hybridization between a Native and Nonnative Trout?

Kellie Carim* **and Lisa Eby**
Wildlife Biology Program, University of Montana, 32 Campus Drive, Missoula, Montana 59812, USA

Ron Pierce
Montana Fish, Wildlife, and Parks, 3201 Spurgin Road, Missoula, Montana 59804, USA

Abstract
The spread of nonnative species over the last century has profoundly altered freshwater ecosystems, resulting in novel species assemblages. Interactions between nonnative species may alter their impacts on native species, yet few studies have addressed multispecies interactions. The spread of whirling disease, caused by the nonnative parasite *Myxobolus cerebralis*, has generated declines in wild trout populations across western North America. Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* in the northern Rocky Mountains are threatened by hybridization with introduced Rainbow Trout *O. mykiss*. Rainbow Trout are more susceptible to whirling disease than Cutthroat Trout and may be more vulnerable due to differences in spawning location. We hypothesized that the presence of whirling disease in a stream would (1) reduce levels of introgressive hybridization at the site scale and (2) limit the size of the hybrid zone at the whole-stream scale. We measured levels of introgression and the spatial extent of hybridization between Rainbow Trout and Westslope Cutthroat Trout in four disease-positive streams and six disease-negative streams within the Blackfoot River basin of Montana. In addition to disease status, we considered habitat quality, stream slope, distance from the confluence, temperature, and elevation. Whirling disease presence was not associated with either the level of introgression at a site or the size of the hybrid zone. Temperature, elevation, and stream slope were all influential in determining levels of introgression at the site scale. Stream slope was the most influential factor determining the size of the hybrid zone, as longer, steeper streams contained smaller hybrid zones. Stream slope is a driver of many habitat characteristics that may provide refuge from invasive species in the coming decades. Although the multispecies interactions examined in this study did not alter the impacts of invasion on native species, community assemblages will continue to change with the spread of nonnative species, requiring continued assessment to determine their impacts on native species.

Freshwater ecosystems are highly imperiled, exhibiting the greatest number of threatened and endangered species as well as the highest rates of species extinction worldwide (Pimm and Raven 1995; Ricciardi and Rasmussen 1999; Burkhead 2012). Anthropogenic degradation of habitat has caused fragmentation of aquatic populations, loss of critical habitat, and subsequent loss of biodiversity (Dudgeon et al. 2006). In addition, both climate change and human activities are facilitating the spread of nonnative species (including but not limited to protozoans, plants, and animals) across freshwater ecosystems at alarming rates (Walther et al. 2002; Strayer and Dudgeon 2010). This spread of nonnative species creates novel species assemblages in which nonnative species interact with native species as well as other nonnatives. Novel interactions between multiple nonnatives may have varied effects on the viability of native species across the landscape. Nonnatives may negatively impact one another through competition or predation (Simberloff and Von Holle 1999; Braks et al. 2004) or through commensal or mutualistic interactions that increase the spread and intensity of their individual impacts (Ricciardi

*Corresponding author: kellie.carim@gmail.com
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2001). In some cases, the presence of multiple nonnatives may amplify the negative impacts on native species (Ross et al. 2004; Johnson et al. 2009). Furthermore, the occurrence and impact of nonnative species may differ across the landscape due to natural variation in abiotic conditions that favor certain species over others. Nonetheless, interactions between nonnative species are explored less frequently than the negative impacts of nonnative species on the native community (Simberloff and Von Holle 1999). We need to better understand how interactions between multiple nonnative species affect the native species community, how landscape factors may alter interactions between native and nonnative species, and how such interactions influence our conservation strategies (Lindenmayer et al. 2008; Hobbs et al. 2009).

The persistence of native Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* is threatened by loss of habitat from human activities and by hybridization with nonnative Rainbow Trout *O. mykiss* (Shepard et al. 2005). Studies have shown that the proportion of Rainbow Trout alleles present in a population sample (i.e., introgression) varies with distance from the source of Rainbow Trout alleles and is altered by tributary characteristics (e.g., elevation, flow regime, and temperature) and human disturbances (e.g., stocking of Rainbow Trout, logging, agricultural practices, and grazing practices; Hitt et al. 2003; Weigel et al. 2003; Heath et al. 2009; Muhlfeld et al. 2009b; Rasmussen et al. 2010; Kovach et al. 2011). However, research has not explored whether the presence of additional nonnative species may alter the landscape-level gradients associated with hybridization both within and among watersheds.

The unintentional spread of parasites has impacted wildlife populations globally, and differences between native and nonnative species in terms of their vulnerability to disease may be a mechanism influencing the spread of nonnative species (Moyle and Light 1996; Peterson and Fausch 2003). For example, Fausch (2007) hypothesized that whirling disease (WD) has limited the invasion of Rainbow Trout in the United Kingdom. The causative agent of WD is the myxosporean parasite *Myxobolus cerebralis*, which is endemic to Eastern Europe. Human-facilitated transport of infected fish after World War II contributed to the global spread of the parasite, causing outbreaks that have decimated wild fish populations across multiple continents (Bartholomew and Reno 2002). *Myxobolus cerebralis* requires two hosts to complete its life cycle: oligochaete worms *Tubifex* spp. and salmonid fishes (Hedrick and El-Matbouli 2002). Young fish with substantial skeletal cartilage (<9 weeks posthatch) are the most susceptible to infection (Ryce et al. 2005). Infection can lead to substantial cartilage destruction, resulting in whirled swimming patterns, skeletal deformities, reduced growth rates, and death (MacConnell and Vincent 2002).

Salmonid species in the genus *Oncorhynchus* appear to be among the most susceptible to WD, but susceptibility varies depending on the species. Vincent (2002) found that Rainbow Trout suffered higher infection rates and severity than various Cutthroat Trout subspecies when exposed to WD in a laboratory setting. In many wild populations, Rainbow Trout may also be more vulnerable than Westslope Cutthroat Trout due to differences in their spawning location (Pierce et al. 2009). The rate of *M. cerebralis* infection decreases predictably in an upstream direction, presumably due to the reduction in habitat (i.e., slow-moving water with fine sediment) for the oligochaete hosts (De la Hoz and Budy 2004; Hallett and Bartholomew 2008), and Westslope Cutthroat Trout spawn higher in tributaries than Rainbow Trout (Pierce et al. 2007, 2009; Muhlfeld et al. 2009a; Buehrens et al. 2013). Thus, in addition to lower susceptibility, Westslope Cutthroat Trout likely experience a lower level of exposure to *M. cerebralis* than Rainbow Trout.

Research has yet to explore the susceptibility of Rainbow Trout × Westslope Cutthroat Trout hybrids to WD, but hybridization between other salmonid species has been examined. Wagner et al. (2002) found that F_1 hybrids of moderately susceptible Brook Trout *Salvelinus fontinalis* and mildly susceptible Lake Trout *Salvelinus namaycush* showed intermediate susceptibility compared with parental strains. Therefore, Rainbow Trout × Westslope Cutthroat Trout hybrids may be more susceptible to WD than pure Westslope Cutthroat Trout due to their Rainbow Trout ancestry. Hybrids may also be more vulnerable than Westslope Cutthroat Trout due to differences in spawning habitat and rearing of hybrids in warmer, lower-elevation areas (Muhlfeld et al. 2009a). If the hybrid offspring of Rainbow Trout and Westslope Cutthroat Trout are more susceptible to WD, then we would expect the presence of WD to reduce the survival of Rainbow Trout and hybrids and to subsequently alter the spatial patterns of introgression between the two species.

Our research objective was to determine whether WD is associated with introgressive hybridization between Westslope Cutthroat Trout and Rainbow Trout in streams of the Blackfoot River basin, west-central Montana. We focused on the following questions: (1) does WD interact with physical and environmental variables (e.g., elevation, temperature, stream slope, distance from the source of Rainbow Trout alleles, and habitat quality) to alter introgressive hybridization between Rainbow Trout and Westslope Cutthroat Trout at the site scale, and (2) how are these variables (landscape characteristics, habitat quality, and WD) associated with the spatial extent of introgression within a stream?

Overall, we expected introgression to decline with increases in elevation, distance from the confluence, and stream slope and to be lower in areas with higher habitat quality and cooler temperatures. We also expected the presence of WD in a stream to interact with landscape characteristics and habitat quality variables by increasing the effects of these variables on the level of introgression and the spatial extent of hybridization in disease-positive streams.

METHODS

Study Area

The Blackfoot River is a free-flowing, fifth-order tributary of the upper Columbia River and drains a 5,998-km^2 watershed through 3,038 km of perennial streams. The river lies in west-central Montana and flows westward 212 river kilometers from the Continental Divide to its confluence with the Clark Fork River at Bonner, Montana. Beginning in 1902, Rainbow Trout were heavily stocked throughout Montana's streams and rivers, including the Blackfoot River basin. Detailed records documenting the location and volume of stocking events throughout Montana were not well kept; however, the stocking of all trout in streams and rivers ceased in 1974 to encourage wild fish production (Zachheim 2006).

Our study focused on 10 tributaries located in the lower half of the Blackfoot River basin (Figure 1) where nonnative Rainbow Trout are present and express both resident and fluvial life histories (Pierce et al. 2009). Native Westslope Cutthroat Trout are present basinwide but are most prevalent in streams of the middle to upper elevations, such as the upper reaches of tributaries to the main stem, and throughout the upper basin (Pierce et al. 2008). Despite intensive stocking of Rainbow Trout throughout the Blackfoot River watershed through the early 1970s, hybridization between Rainbow Trout and Westslope Cutthroat Trout has been detected most commonly in the lower watershed but rarely in the upper basin (Pierce et al. 2005, 2008). Other salmonid species present in the basin include native Bull Trout *Salvelinus confluentus* and Mountain Whitefish *Prosopium williamsoni* as well as nonnative Brook Trout and Brown Trout *Salmo trutta*. Whirling disease was detected in the Blackfoot River basin during initial testing in 1998 (Pierce and Podner 2006), 4 years after Montana's first documented outbreak in the Madison River (Vincent 1996).

Stream Selection

For the last two decades, Montana Fish, Wildlife, and Parks (MFWP) has conducted sentinel cage exposures with hatchery

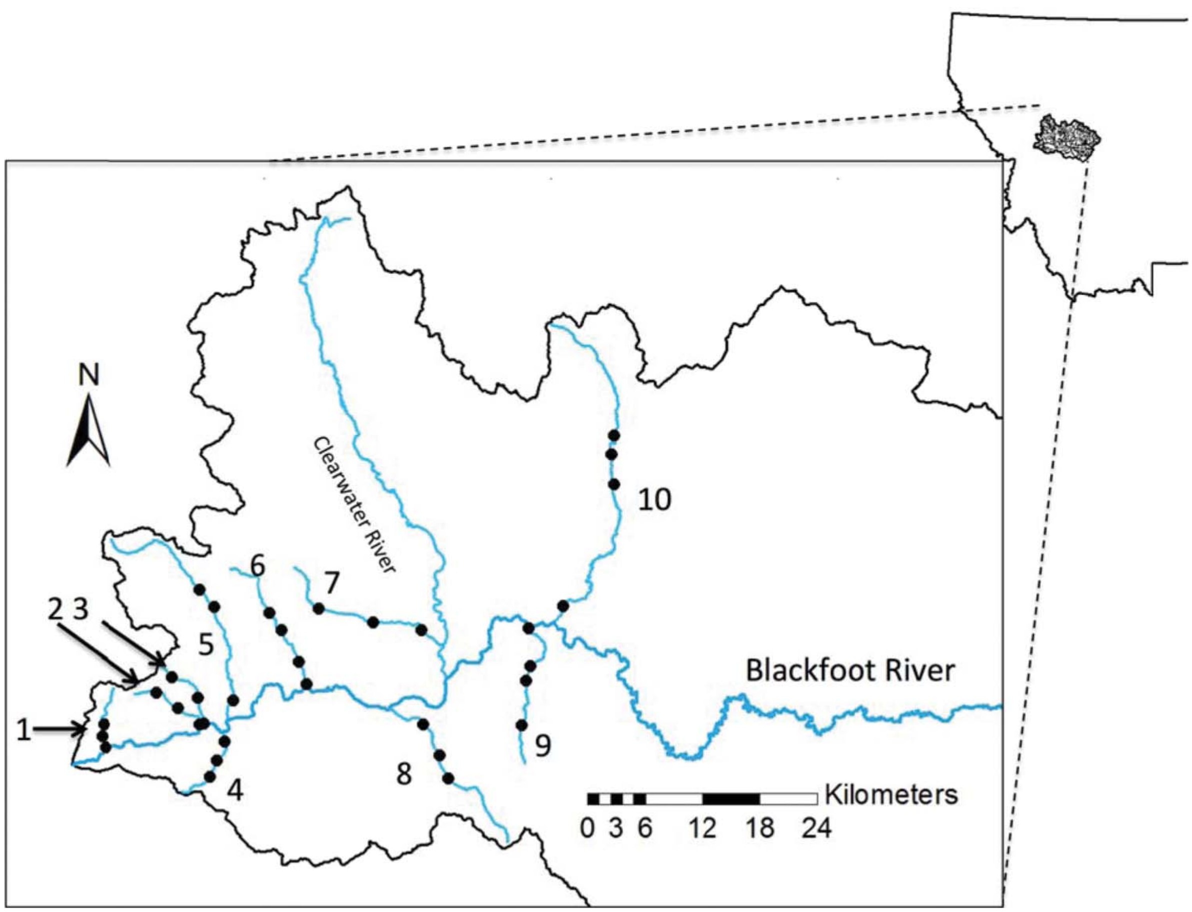

FIGURE 1. Sampling locations used to evaluate introgression between Rainbow Trout and Westslope Cutthroat Trout in the Blackfoot River basin of west-central Montana. Points indicate sampling locations; numbers correspond to the map codes defined in Table 1. The lowermost site in each stream was also the sentinel cage exposure site for whirling disease assessment.

Rainbow Trout to monitor the presence and severity of WD in streams throughout the Blackfoot River basin. Exposures in the present study followed the methods of Pierce et al. (2009); we included all basin-fed streams in the watershed that (1) were sites of known hybridization between Rainbow Trout and Westslope Cutthroat Trout, (2) were repeatedly monitored for WD within 4.5 km of the confluence (median distance = 0.7 km), and (3) were monitored for WD at least once between 2004 and 2008 (see Supplementary Table S.1 in the online version of this article; Pierce et al. 2001, 2008; Pierce and Podner 2006).

In our study, we assumed that a stream was disease-negative (i.e., WD was not present) if no infection was detected in sentinel-caged fish for all tests conducted in that stream. We categorized streams as disease-positive if at least 70% of caged fish had an infection severity greater than 3 on the MacConnell–Baldwin rating scale and if the mean infection severity for all exposed fish was higher than 3. This level of disease severity is considered high enough to influence fish survival and to have population-level effects based on laboratory experiments and case studies (Vincent 2002; Granath et al. 2007). For example, Granath et al. (2007) found that declines in wild Rainbow Trout were associated with increasing infection severity (>2.5) of trout held in sentinel cages throughout the drainage. Six disease-negative streams and four disease-positive streams met the criteria for inclusion in our study (i.e., streams with known hybridization between *Oncorhynchus* species and where repeated tests documented either no detection of WD or the presence of high-severity infection).

Site-Scale Data Collection

Within each stream, we sampled three to four locations between 2009 and 2011 to determine the level of introgression between Rainbow Trout and Westslope Cutthroat Trout (Figure 1; Table S.1). Two sites were sampled again in 2013 to increase the sample sizes. The lowermost sampling site in each stream corresponded to the location of sentinel cage exposures for that stream. Sites were spaced roughly 1.3–16.2 km apart (median = 3 km) in order to describe the longitudinal pattern of introgressive hybridization. The uppermost sampling site targeted areas where we expected to find little to no introgression between Rainbow Trout and Westslope Cutthroat Trout (i.e., <5% Rainbow Trout alleles within a sample of fish) based on phenotypic indicators and initial analyses of our genetic samples collected in 2009. If we did not achieve our goal during the first sampling visit, we returned to the watershed to sample a site further upstream. We used 5% introgression as a threshold for defining the end of the hybrid zone. This threshold allows for the occurrence of natural polymorphisms, which may otherwise alter the detection of nonhybridized populations in these systems (Allendorf et al. 2012), yet it is more conservative than the 10% threshold outlined for consideration as a conservation population under the Memorandum of Understanding and Conservation Agreement for Westslope Cutthroat Trout (Allendorf et al. 2001; MTFWP 2007).

At each site, we collected all *Oncorhynchus* species present by using a backpack electrofishing unit until (1) we obtained a sample size of 25 individuals, (2) sampling time exceeded 2.5 h, or (3) the sample reach exceeded 550 m. For each fish, we measured total length (mm), removed a tissue sample, and placed the sampled tissue in 95% ethanol for genetic analysis. For sites that were sampled in multiple years, we compared the genotypes of sampled individuals to ensure that the same individual was not represented more than once in our data set.

To assess habitat quality and other tributary characteristics known to influence hybridization at the site scale, we recorded elevation, distance from the confluence (Stream_km), stream slope, temperature, and bank stability at each site (Figure 2). Elevation and Stream_km were measured in ArcMap (ESRI 2010) using U.S. Geological Survey digital elevation map layers and the Montana Streams layer maintained by MFWP (//nris.mt.gov/gis/). We calculated stream slope as the change in elevation from the confluence to the site divided by Stream_km. We obtained mean August temperatures for each site from the NorWeST Stream Temp interactive map (www.sciencebase.gov/flexviewer/NorWeST/).

We measured bank stability at each sampling site to quantify measures of habitat quality, as grazing is the primary anthropogenic riparian disturbance in this watershed. Previous studies have found a positive association between introgression with Rainbow Trout alleles and either logging activity or road density (Heath et al. 2009; Muhlfeld et al. 2009b). Authors of those studies attributed the pattern to stream alterations resulting from human activities and infrastructure, including increases in fine-sediment deposition and changes in hydrologic regimes. Although logging and the presence of roads are common disturbances throughout the Blackfoot River basin, agriculture and grazing practices are the most widespread disturbances in the watershed, often causing riparian degradation and loss of bank stability (Pierce et al. 2013). To assess bank stability and animal impacts, we used rating systems for vegetation cover, bank stabilization by rock, and animal damage (wild or domestic) as outlined by Stevenson and Mills (1999) and summed the ratings across the three categories to obtain a single variable for bank stability at a site.

Genetic Analysis

To ensure that our data were representative of the population at a given site, we analyzed all fish between 70 and 230 mm TL. We did not sample fish smaller than 70 mm because these individuals are typically young of the year and it is difficult to obtain a sufficient tissue sample without lethal effects. Based on the expert opinion of local biologists, we did not analyze fish larger than 230 mm to minimize the

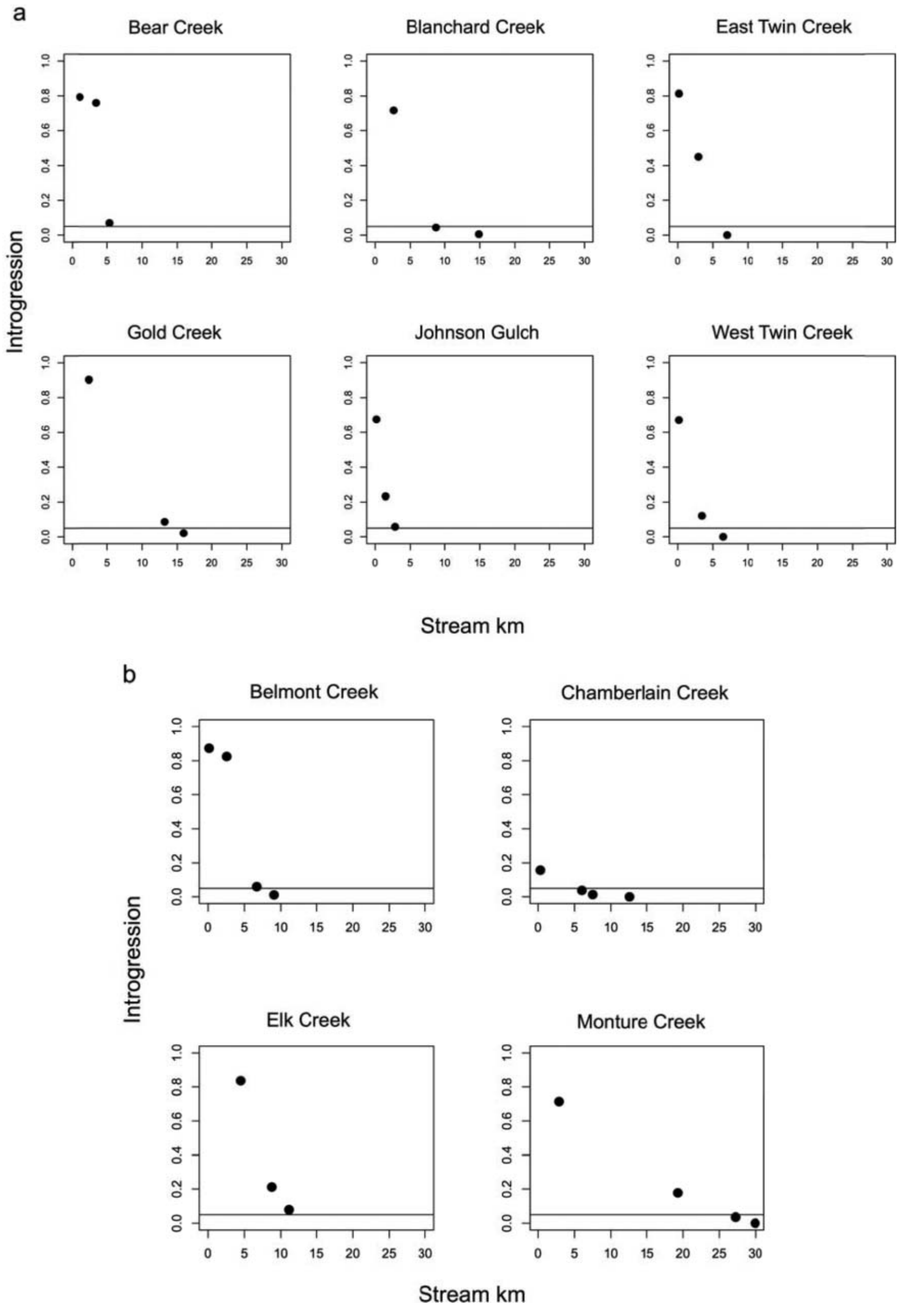

FIGURE 2. Box-and-whisker plots showing the range of site-scale variables between whirling disease-negative and disease-positive streams sampled in the Blackfoot River basin: **(a)** slope, **(b)** elevation (m), **(c)** temperature (°C), **(d)** distance upstream from the confluence (Stream_km), **(e)** fine-sediment deposition (1 = >75% of surface covered by fines; 2 = 50–75% coverage; 3 = 25–50% coverage; 4 = 5–25% coverage; 5 = 0–5% coverage), and **(f)** bank stability ranking (as summed across rankings of animal damage, bank stabilization by rock, and vegetation cover; 3 = low stability and 12 = high stability) across all sites. Whiskers represent 1.5 times the interquartile range.

likelihood of including migratory fish that were using the site as summer habitat.

For all samples, DNA was extracted following the Gentra Isolation Kit protocol. All samples were amplified in 10-μL reactions and were analyzed using three different PCR profiles with the QIAGEN Multiplex PCR Kit (QIAGEN, Valencia, California) in accordance with the manufacturer's instructions. To determine levels of introgression, we analyzed two panels that included a total of 10 diagnostic markers (each denoted by an asterisk below). The first panel consisted of seven insertion/deletion loci and one microsatellite locus: *Occ34**, *Occ35**, *Occ36**, *Occ37**, *Occ38**, *Occ42, Om55** (Ostberg and Rodriguez 2004), and *Ssa408** (Cairney et al. 2000). The second panel consisted of six microsatellite loci: *Omm1037-1, Omm1037-2, Omm1050** (Rexroad et al. 2002), *Omy0004** (Holm and Brusgaar 1999), *Omy1001** (Spies et al. 2005), and *Oki10* (Smith et al. 1998). We used an ABI 3130XL Genetic Analyzer (Applied Biosystems, [ABI], Foster City, California) to visualize PCR products. The ABI GS600LIZ ladder was used to determine allele sizes, and chromatogram output was viewed and analyzed using ABI GeneMapper version 3.7. Using results from the diagnostic alleles only, we quantified introgression at a given site as the proportion of Rainbow Trout alleles in a sample:

$$\Pr(\text{RBT}) = (\text{RBTalleles})/(2 \cdot L \cdot N),$$

where "RBT alleles" is the number of Rainbow Trout alleles detected in a sample from a given site, L is the number of diagnostic loci examined, and N is the total number of fish analyzed from that site (Kanda et al. 2002). With the 10 diagnostic markers listed above, a sample size of 25 fish yields a 99.3% probability of detecting as little as 1% Rainbow Trout admixture in the sample.

All genetic analyses were conducted at the Conservation Genetics Laboratory, University of Montana, Missoula.

Stream-Scale Data Collection

To examine which variables best predicted the size of the hybrid zone (i.e., the distance from the confluence at which introgression declined to 5%), we first estimated the Stream_km where introgression would equal 5% by fitting a linear regression (introgression versus Stream_km) between the two sites where introgression was closest to 5%. When possible, we interpolated between two adjacent sampling sites that tested above and below this threshold. In a given stream, if we were unable to obtain a sample with introgression below 5%, we extrapolated and used the two adjacent sites with introgression levels that were closest to 5%. We did not fit a logistic or linear regression using all data points for a given stream because this would have provided a model that best fit all points rather than more accurately pinpointing the location where population-level introgression reached 5%.

To obtain a slope measure that was independent of the response variable (i.e., distance from the upper end of the hybrid zone to the confluence) and that represented the stream scale, we used the slope of the entire stream as a variable for predicting the size of the hybrid zone. We calculated whole-stream slope as the change in elevation over the distance from the headwaters of the main stem to the confluence using data layers in ArcMap as described above (ESRI 2010). We did not include elevation in this analysis because the change in elevation throughout the hybrid zone was incorporated into our slope parameter, and we had no prediction for how elevation at the end of the hybrid zone would influence the overall size of the zone.

To obtain a measure of temperature corresponding to the hybrid zone in each stream, we used the average of temperatures at the confluence and at the upper limit of the hybrid zone (Temp_zone), as obtained from the NorWeST Stream Temp interactive map for each stream. To quantify bank stability within the hybrid zone, we averaged the scores for this variable across all sampling sites within the hybrid zone to obtain a single estimate of bank stability (Bank_zone).

Statistical Analyses

Which variables are associated with introgression at a site?—To evaluate whether WD presence influenced the level of introgression at a given site, we standardized variables and used a generalized linear mixed regression model with a logit-link function. The WD, temperature (Temp), elevation (Elev), Stream_km, slope, and bank stability (Bank) variables were included as fixed effects. We evaluated a one-way interaction between WD and all other fixed effects because we hypothesized that the presence of WD would alter the influence of these variables on introgression. Because there were multiple sites within a stream, "Stream" was included as a random effect in every model.

We calculated the variance inflation factor (VIF) for each fixed effect to assess multicollinearity. The VIF quantifies the degree to which the variance increases as a result of multicollinearity with other variables in an ordinary least-squares regression model. For example, a VIF of 10 for a single variable would mean that the variance of the parameter estimate is 10 times larger than it would be if that variable was completely uncorrelated with all others in the model (Montgomery et al. 2012). This assessment of multicollinearity allows a model to include variables that might be correlated but that have differing relationships with the response variable. If the VIF was high (>5) for a given combination of variables, we created several global models so that variables with a high degree of multicollinearity could be included in separate models.

Elevation was used to predict temperature in the NorWeST Stream Temp models. As a result, these two variables were inherently confounded, so we created separate global models to avoid including these variables together in the same model. Preliminary analysis of models that included both WD and Slope or both Temp and Stream_km revealed that these two sets of variables were significantly correlated. Specifically, results from models that included these combinations of variables suggested associations between these variables and introgression that were not observed in the raw data, thus indicating that multicollinearity between predictor variables was affecting the model results (Montgomery et al. 2012). A Welch's t-test revealed that slope was significantly shallower in streams where WD was present ($P < 0.001$; Figure 2a). We also found that Temp was significantly correlated with Stream_km ($r = -0.51$, $P < 0.01$).

To account for these issues, we evaluated four independent global model structures:

Model structure A: Introgression $\sim$ WD $\times$ (Elev + Stream_km + Bank) + (1|Stream),

Model structure B: Introgression $\sim$ WD $\times$ (Temp + Bank) + (1|Stream),

Model structure C: Introgression $\sim$ Slope + Elev + Stream_km + Bank + (1|Stream),

and

Model structure D: Introgression $\sim$ Slope + Temp + Bank + (1|Stream),

where the $\times$-symbol denotes an interaction between WD and all parameters shown within the parentheses; and Stream is a random effect. We analyzed all possible subsets of the fixed effects in each global model structure down to univariate model structures. The VIFs for variables in each global model were less than 2.3.

Model selection was based on Akaike's information criterion (AIC; Burnham and Anderson 2002) and error around parameter estimates. The top model from a given subset of models was the one with the lowest AIC value that also had significant parameter estimates ($\alpha = 0.05$) for all interaction terms as well as for any base variables not included in the interaction terms (Arnold 2010). Error structure was calculated by using the same methods for all four model sets. As a result, we were able to select the best overall model as the model with the lowest AIC value (Burnham and Anderson 2002).

What influences the spatial extent of introgression within a stream?—To evaluate which variables best predicted the size of the hybrid zone in a stream, we standardized the variables and performed a multiple linear regression of estimated hybrid zone size on slope, Temp_zone, and Bank_zone. To account for multicollinearity of variables, we assessed VIFs as described above. The VIFs for Temp_zone, whole-stream slope, and Bank_zone exceeded 5 for model structures that included any combination of these three parameters. Therefore, we created three global models that included each of the three parameters independently. Our global model structures for predicting the size of the hybrid zone were as follows:

Model structure Zone_A: Hybrid Zone Size $\sim$ WD + Slope,

Model structure Zone_B: Hybrid Zone Size $\sim$ WD + Temp_zone,

and

Model structure Zone_C: Hybrid Zone Size $\sim$ WD + Bank_zone.

We compared all possible subsets of these global models. In each case, the top model was the model with the lowest value of AIC corrected for small sample size (AIC_c) and in which parameter estimates for all variables were significantly different from zero ($\alpha = 0.05$). The overall best model describing hybrid zone size from these three global model structures was the top model that explained the highest proportion of variance as indicated by the R^2 value.

All statistical analyses were conducted in R (R Development Core Team 2012), and the following R packages were used: lme4, Hmisc, HH, and MASS (Venables and Ripley 2002; Bates et al. 2014; Harrell 2014; Heiberger 2014).

RESULTS

Quantification of Introgressive Hybridization and the Size of the Hybrid Zone

We obtained a minimum of 25 tissue samples at all sites except for five sites in three different streams (West Twin, Monture, and Gold creeks); we were unable to achieve a sample size of 25 fish at those sites due to low densities of Westslope Cutthroat Trout (0.03–0.09 fish/m; Table S.1). At four of the five sites, our sample sizes ranged from 20 to 24 individuals, but in one case (West Twin Creek site WT3), we obtained only 13 individuals over three sampling years. However, all of the fish captured at that site were nonhybridized. Based on the 10 diagnostic markers, we still had a 92.6% probability of detecting as little as 1% population admixture (Kanda et al. 2002) with these 13 samples, so the WT3 site was included in the analyses as a nonhybridized site.

To estimate the size of the hybrid zone, we generally interpolated between the highest-elevation sites in each stream (Table 1). The exceptions were Johnson Gulch, Bear Creek, and Elk Creek, where we did not obtain a sample with population-level introgression less than 5% (Figure 3). In Elk Creek, we detected 8.6% admixture at the highest-elevation site (EK3), but Westslope Cutthroat Trout were not present at the next site upstream. The highest-elevation site sampled in Bear Creek (BR3) had 7.7% admixture, but we only obtained two fish at the next site upstream of BR3. In Johnson Gulch, the uppermost site (JG2) had 6.3% admixture, and we were not

TABLE 1. Estimated distance from the confluence (Stream_km), elevation, and change in elevation from the confluence (Delta elevation) marking the upper limit of the Westslope Cutthroat Trout × Rainbow Trout hybrid zone in each stream sampled within the Blackfoot River basin. Map codes correspond to the stream numbers shown in Figure 1. Slope refers to the whole-stream slope from the headwaters to the confluence for each stream. Bank stability (Bank_zone; 3 = low stability, 12 = high stability) was averaged across all sites within the hybrid zone of the stream; average temperature of the hybrid zone (Temp_zone) was calculated as the mean of temperatures measured at the confluence and at the upper limit of the hybrid zone. Total stream length (km) is the total length from the confluence to the headwaters.

Map code	Stream	Stream_ km	Elevation (m)	Delta elevation (m)	Slope	Bank_ zone	Temp_ zone (°C)	Temp range	Total stream length (km)
				Disease-negative streams					
1	Johnson Gulch	2.89	1,177	171	0.14	11.33	10.2	9.8–10.7	7.7
2	West Twin Creek	5.22	1,424	388	0.112	11.5	10.5	8.8–12.2	8.9
3	East Twin Creek	6.58	1,429	391	0.084	10.75	10.7	9.7–11.7	8.9
4	Bear Creek	5.4	1,350	311	0.079	9.33	11.3	10–12.6	9.1
5	Gold Creek	14.7	1,344	299	0.036	10.0	11.9	10.7–13.2	29.2
6	Blanchard Creek	8.7	1,433	261	0.028	9.25	13.7	12.5–15	20.7
				Disease-positive streams					
7	Belmont Creek	7.18	1,330	263	0.046	8.5	12.0	11.4–12.5	16.5
8	Elk Creek	11.63	1,275	158	0.028	10.0	13.9	12.5–15.4	21.8
9	Chamberlain Creek	5.44	1,292	105	0.039	7.5	12.8	11.7–13.9	16.9
10	Monture Creek	26.42	1,341	140	0.023	9.0	11.6	9.5–13.7	46.0

able to access higher sites. For these three streams, we estimated hybrid zone size by extrapolating from the two highest-elevation sampling sites.

Across all sites, individuals that tested positive for Rainbow Trout alleles appeared to be backcross hybrids. No F_1 hybrid individuals (i.e., fish with 50% admixture that were heterozygous for Rainbow Trout and Westslope Cutthroat Trout alleles at each diagnostic locus) and no pure Rainbow Trout were observed in this study.

Which Site-Scale Variables Are Associated with Introgression?

In each evaluation of global model structures (A–D) for explaining the level of introgression, the top model reported had the lowest AIC value and met our criteria for significant parameter estimates (outlined in Methods); in each case, no other models had AIC values that were within 2 points of the top model's AIC value (Table 2).

Results of top model from evaluation of global model A indicated that lower levels of introgression at a particular site were associated with increasing Stream_km, higher elevation, and higher-quality habitat as indicated by greater bank stability (Table 3). The presence of WD increased the effects of elevation and bank stability but decreased the effect of Stream_km. Although the association between introgression and Stream_km was still negative in the presence of WD, the effect of Stream_km was reduced.

Results of the top model from evaluation of global model B suggested that introgression at a site declined with greater bank stability and increased with higher temperatures. In the presence of WD, the relationship between temperature and introgression was reduced.

Results of the top model from evaluation of global model C indicated that introgression declined with increases in stream slope, elevation, Stream_km, and bank stability. Similarly, the top model based on the evaluation of global model D indicated that introgression was lower at sites with steeper slopes, higher bank stability, and cooler temperatures.

All four top models from the evaluations of model structures A–D were consistent in their results, indicating lower introgression at sites that were cooler, were further from the confluence, were at a higher elevation, and had higher bank stability. The top model from the global model C evaluation had the lowest AIC value of all the top models, so it was considered the overall best model for explaining the level of introgression at a site.

What Influences the Spatial Extent of Introgression within a Stream?

For the Zone_A model structure, the top model for predicting the size of the hybrid zone contained only stream slope (Table 4) and explained nearly 40% of the variation observed in the response variable for this data set. No other Zone_A models had AIC_c values that were within 2 points of the top model's value. For the Zone_B and Zone_C model structures, two models' AIC_c values were within 2 points of the top models' AIC_c values. In each case, the model that contained WD alone had the lowest AIC_c value. Surprisingly, the presence of

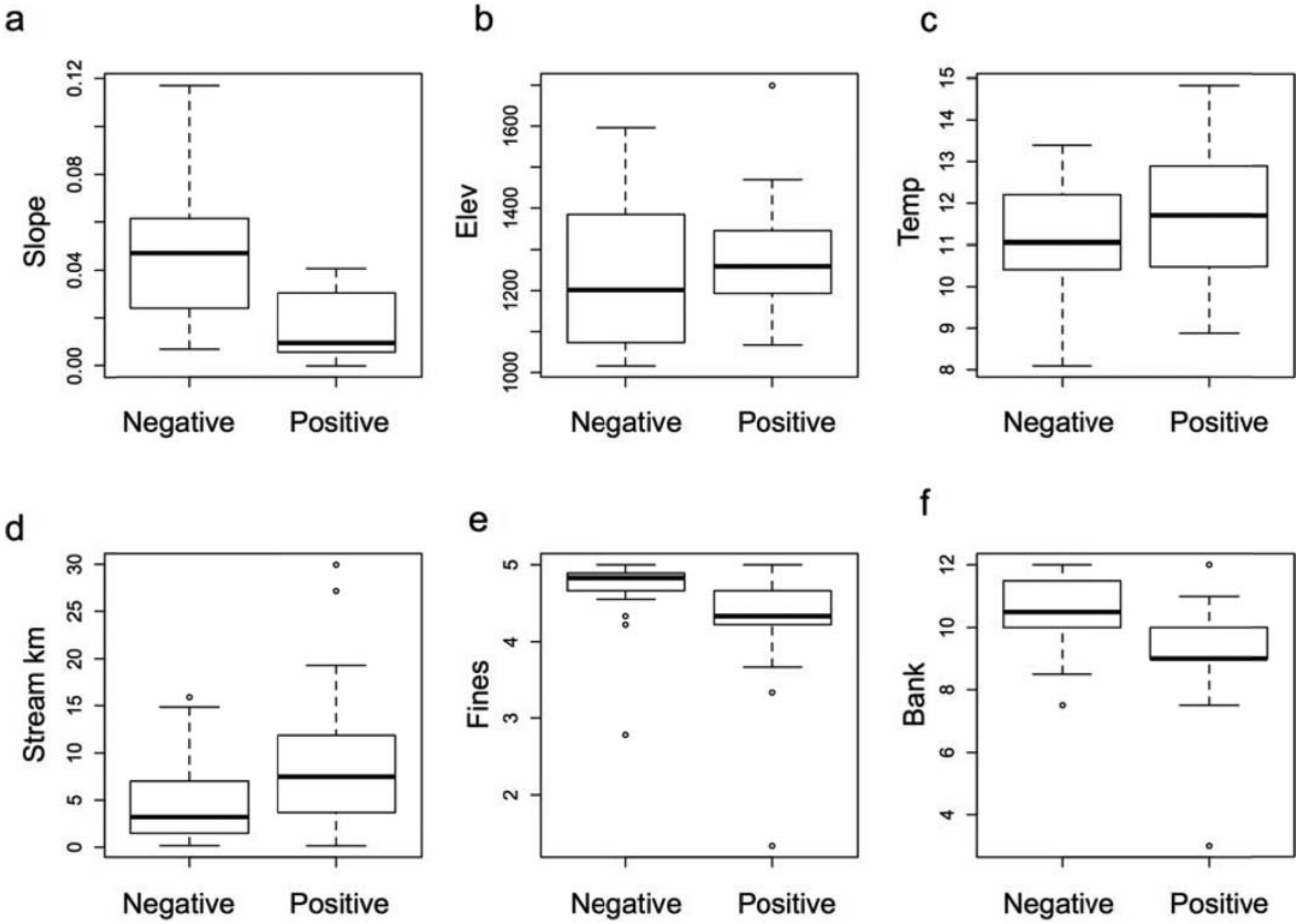

FIGURE 3. Level of introgression (proportion of Rainbow Trout alleles in a sample) versus distance upstream from the confluence (Stream_km) for all Blackfoot River basin sites sampled in **(a)** disease-negative streams and **(b)** disease-positive streams. The horizontal line represents 5% introgression of Westslope Cutthroat Trout with Rainbow Trout.

WD was positively associated with hybrid zone size, although the association was not significant. The second-best model (i.e., within 2 AIC_c points of the top model) for the Zone_B structure displayed a positive association between Temp_zone and hybrid zone size, whereas the second-best model for Zone_C had a negative association between Bank_zone and hybrid zone size. Across all three global model structures, the top model from the Zone_A evaluation explained the greatest proportion of variance in the size of the hybrid zone and was the only model in which parameter estimates were significantly different from zero. As a result, no other model was considered for selection as the overall best-fit model for predicting hybrid zone size.

DISCUSSION

In our study, WD was not included in the best overall model for examining associations with hybrid zone size at the whole-stream scale. In analyzing introgression at the site scale, the effects of distance from the confluence (Stream_km) and temperature were reduced in the presence of WD, contrary to our expectations. There are several possible reasons why we did not observe patterns that supported our hypotheses. First, studies on WD susceptibility have only been performed on fish of hatchery origin. It is possible that (1) the susceptibility of wild Rainbow Trout and hybrids does not differ from that of pure, wild Westslope Cutthroat Trout; or (2) the difference is so weak that there is not an observable effect on the overall levels of introgressive hybridization between the two species. Miller and Vincent (2008) found evidence that a wild strain of Rainbow Trout from Harrison Lake, Montana, underwent rapid natural selection for resistance to WD. It is also possible that the wild populations in this study experienced a similar process of selection after the disease initially emerged in the Blackfoot River basin. The presence of WD may have had an impact on hybridization between Rainbow Trout and Westslope Cutthroat Trout soon after *M. cerebralis* was introduced, but evolution of resistance to WD and

TABLE 2. Top models predicting the level of introgression (proportion of Rainbow Trout alleles) at a site for each of the four global model structures (each global model was examined independently). Models with structures A and B included interactions of fixed effects with whirling disease presence (WD) but excluded stream slope (Bank = bank stability); models with structures C and D included stream slope and excluded WD. Models with structures A and C included elevation (Elev) and distance from the confluence (Stream_km); models with structures B and D included temperature (Temp). The number of parameters (k) includes the intercept and the random factor (Stream). Uninformative parameters are variables with parameter estimates that were not significantly different from zero at $\alpha = 0.05$ The top model with structure C had the lowest Akaike's information criterion (AIC) value and was selected as the overall top model for explaining the level of introgression at a site.

Model structure	Description	k	Log-likelihood	AIC	Uninformative parameters
A	WD × (Stream_km + Elev + Bank) + (1|Stream)	9	−292.6	603.1	WD
B	WD × (Temp) + Bank + (1|Stream)	6	−1,100.0	2,212.1	WD
C	Slope + Elev + Stream_km + Bank + (1|Stream)	6	−291.1	594.1	None
D	Slope + Temp + Bank + (1|Stream)	5	−1,220.5	2,451.0	None

subsequent recovery of Rainbow Trout and hybrid populations prior to our study could have obscured any apparent evidence of this interaction.

Stream slope repeatedly emerged as a strong predictor of hybridization between Rainbow Trout and Westslope Cutthroat Trout: introgression was lower at higher-gradient sites, and hybrid zones were smaller in shorter, more steeply sloping streams. In the Blackfoot River basin, landscape-level estimates of valley slope are correlated with stream slope at the site scale and serve as a good predictor of both fine-sediment loads and WD severity at a site (Pierce et al. 2008). Specifically, less-steep sites had higher disease severity in sentinel cage studies, presumably due to the higher loads of fine sediment, which provide habitat for the disease's alternative host, *T. tubifex*. In addition, less-steep, disease-positive streams registered some of highest instances of disease severity in sentinel cage studies conducted throughout the Blackfoot River basin. Monture Creek had the lowest stream slope and the largest hybrid zone in our data set. At our lowest-elevation sampling site in Monture Creek (2.9 km upstream from the confluence), we observed introgressive hybridization that was close to 80%. Over 90% of sentinel-caged fish at that location had mean grade infections exceeding 3 (MacConnell–Baldwin rating scale) in 2005, 2006, 2007, and 2009 (Pierce et al. 2008, 2012). If WD was impacting wild Rainbow Trout and hybrid populations in a manner that reduced introgressive hybridization with Westslope Cutthroat Trout, we would expect a stream like Monture Creek to have a much smaller hybrid zone and lower levels of introgressive hybridization at sites known to induce high-severity infection. These data

TABLE 3. Details of top models (Table 2) for predicting the level of introgression (proportion of Rainbow Trout alleles) at a site, including parameter estimates, SEs, and P-values for the fixed-effect variables and variance estimates for the random effect (Stream). Variables are defined in Table 2.

	Top model A			Top model B			Top model C			Top model D		
Effect	Estimate	SE	P-value	Estimate	SE	P-value	Estimate	SE	P-value	Estimate	SE	P-value
Fixed effects												
Intercept	−2.13	0.35	<0.0001	−1.49	0.54	<0.01	−2.00	0.31	<0.0001	−1.58	0.73	<0.05
WD	−0.10	0.36	0.79	−1.04	0.55	0.06						
Slope							−0.35	0.05	<0.0001	−0.32	0.04	<0.001
Elev	−2.39	0.09	<0.0001				−2.52	0.08	<0.0001			
Stream_km	−1.04	0.11	<0.0001				−0.73	0.07	<0.0001			
Temp				2.48	0.05	<0.0001				2.45	0.05	<0.001
Bank	−0.45	0.04	<0.0001	−0.53	0.04		−0.52	0.03	<0.0001	−0.40	0.04	<0.001
WD × Elev	−0.40	0.09	<0.0001									
WD × Stream_km	0.41	0.10	<0.0001									
WD × Temp				−0.75	0.04	<0.0001						
WD × Bank	−0.12	0.03619	<0.01									
Variance of random effects												
Stream	1.17			2.87			0.98			4.51		

TABLE 4. Models from three independent global model structures (Zone_A, Zone_B, and Zone_C) for predicting the size of the Westslope Cutthroat Trout × Rainbow Trout hybrid zone in a stream; only models with AIC_c values that were within 2 points of the top model's AIC_c value are presented here (AIC_c = Akaike's information criterion corrected for small sample size; WD = whirling disease presence; Temp_zone = average of temperatures at the confluence and at the upper limit of the hybrid zone; Bank_zone = bank stability averaged across all sites within the hybrid zone). Model structure, the number of parameters (k), proportion of variation explained by the model (multiple R^2), the parameter estimate with SE, the P-value of the parameter estimate, and the negative log-likelihood of the model are presented. The difference in AIC_c between Zone_B models 1 and 2 was 1.21, and the difference in AIC_c between Zone_C models 1 and 2 was 1.04 Zone_A model 1 was chosen as the best overall model for predicting the size of the hybrid zone in a stream because it was the only model with parameter estimates that significantly differed from zero.

Model	Description	k	Multiple R^2	Estimate	SE	P-value	Negative log-likelihood
Zone_A1	Slope	2	0.39	−4.35	1.93	0.05	−30.63
Zone_B1; Zone_C1	WD	2	0.16	2.80	2.26	0.26	−32.23
Zone_B2	Temp_zone	2	0.05	1.53	2.41	0.54	−32.84
Zone_C2	Bank_zone	2	0.06	−1.77	2.39	0.48	−32.75

highlight stream slope as a comprehensive variable influencing *T. tubifex* habitat and distribution and thus the presence of WD.

Slope may also influence habitat characteristics associated with the current distribution and spawning success of Rainbow Trout and Westslope Cutthroat Trout. The association of stream slope with introgression at the site scale may be explained by differences in life history between Rainbow Trout and Westslope Cutthroat Trout. Multiple studies comparing habitat and occupancy of Rainbow Trout, Cutthroat Trout, and hybrids have found that Rainbow Trout and hybrids occupy lower-gradient sections of stream in areas where Rainbow Trout have been introduced as well as in areas where the two species are naturally sympatric (Hitt et al. 2003; Weigel et al. 2003; Buehrens et al. 2013). Muhlfeld et al. (2014) found that the expansion of hybridization from 1978 to 2008 in the upper Flathead River basin of northwestern Montana was strongly correlated with decreases in May precipitation. Rainbow Trout and hybrids tend to spawn earlier in the spring as runoff associated with snowmelt increases and peaks, whereas Westslope Cutthroat Trout spawn later in the spring as high flows subside (Muhlfeld et al. 2009a; Corsi et al. 2013). Muhlfeld et al. (2014) attributed the expansion of hybridization in the Flathead River in part to lower spring runoff, which would result in reduced scouring of redds and disturbance of newly emerged juveniles. In this context, we would expect streams with steeper slopes to have faster, more turbulent flows during spring spates. Redds and juveniles in these more steeply sloping streams would likely experience more disturbance from spring flow events, and Rainbow Trout and hybrids would be particularly susceptible to these disturbances due to their timing of spawning and emergence. A similar association between stream slope and salmonid community composition has also been observed in mountain streams of the Pacific Northwest. Montgomery et al. (1999) found that on a basin scale, steeper streams favored salmonid species whose spawn timing resulted in egg incubation and juvenile emergence periods that were offset from the most severe flood events. Although fine-scale habitat characteristics certainly play a role, our results support landscape-level geomorphology as a factor determining salmonid community composition across both native and nonnative species.

As expected, introgression was lower at higher-elevation sites in this study. Elevation generally displayed a negative correlation with introgressive hybridization between our two focal species, as has been observed in other studies (Hitt et al. 2003; Bennett et al. 2010; Rasmussen et al. 2010; Yau and Taylor 2013). For example, in a study of the upper Oldman River (Alberta, Canada), Rasmussen et al. (2010) reported that the proportion of Rainbow Trout alleles present in a population decreased exponentially with increases in site elevation; introgression greater than 5% was only observed at 1 out of 16 sites with elevations of 1,471 m or higher (median introgression = 1%; maximum elevation = 1,722 m). Hitt et al. (2003) found a similar transition to nonhybridized Westslope Cutthroat Trout at roughly 1,450 m in the upper Flathead River basin. Among a total of 12 sites at elevations above 1,300 m, we observed only one site where introgression was greater than 5% (median introgression for those 12 sites = 1.4%; maximum elevation = 1,699 m). Results of these studies suggest that an elevational threshold exists for the persistence of Rainbow Trout and hybrids. Mechanistically, however, the associations are likely the result of changes in habitat and climate that follow an elevational gradient.

In previous studies, temperature was negatively associated with both the occurrence of hybridization and the degree of introgression at the site scale (Muhlfeld et al. 2009b; Yau and Taylor 2013). We identified similar associations in our data set, but it should be noted that temperature was not present in either the top model for explaining the level of introgression at a site or the top model for explaining the overall size of the hybrid zone. Elevation and distance from the confluence were significant predictors determining the level of introgression at a site, and temperature was associated with both of those predictor variables. Temperature did not emerge as a significant predictor of site-scale introgression or hybrid zone size at the whole-stream scale, suggesting that generalized summertime temperature metrics alone (e.g., mean August temperature from the NorWeST Stream Temp models) may not represent the key limiting climatic conditions that affect hybridization at a whole-stream scale. For example, Fausch et al. (2001) found that success of Rainbow Trout invasions in Colorado, the southern Appalachians, and Japan were strongly influenced by flow regime. Bennett et al. (2010) reported that tributaries to the upper Kootenay River (British Columbia) that were located in warmer and drier biogeoclimatic zones were associated with higher levels of introgression between Westslope Cutthroat Trout and introduced Rainbow Trout. In a study on physiological performance, Rasmussen et al. (2012) suggested that the metabolic needs of individuals with Rainbow Trout ancestry (both pure and introgressed) are not met in less-productive, high-elevation habitat, thereby allowing Westslope Cutthroat Trout to dominate those areas. These studies provide evidence that broader climatic variables incorporating aspects of temperature, precipitation, and flow regime serve as better predictors of hybridization between Rainbow Trout and Westslope Cutthroat Trout than temperature alone.

Distance from the confluence is associated with temperature and elevation, but it may also address variation in introgression associated with propagule pressure that is not represented in measures of temperature or elevation. Bennett et al. (2010) determined that introgression between Rainbow Trout and Westslope Cutthroat Trout at sites in the upper Kootenay River was strongly influenced by propagule pressure—a variable they defined as a combination of historical stocking intensity and distance from the stocking locations. Although the entire main stem of the Blackfoot River was heavily stocked with Rainbow Trout in the 20th century (1902–1974; Zachheim 2006), detailed stocking records are not available for this watershed (R. Pierce, personal observation). Consistent with other studies, we found that introgression decreased with increasing distance from the confluence, as main river sections are currently considered the putative source of Rainbow Trout alleles in the Blackfoot River basin and other river basins (Hitt et al. 2003; Weigel et al. 2003; Muhlfeld et al. 2009b; Rasmussen et al. 2010; Kovach et al. 2011).

Similar to the findings of Muhlfeld et al. (2009b), we observed that sites with higher habitat quality generally had lower levels of introgression. In our study streams, introgression tended to increase with disturbances that erode streambanks and increase rates of sedimentation, such as hoof shearing, lack of vigorous riparian vegetation, and bank stabilization by rocks. Such disturbances also tend to increase stream temperatures (which may favor Rainbow Trout and hybrids), an association that was observed in this data set as well as in other studies of the Blackfoot River basin (Pierce et al. 2013). An additional mechanism for this trend could be related to early development: embryos of Rainbow Trout and hybrids may have a higher tolerance for fine sediment than embryos of Westslope Cutthroat Trout. Sowden and Power (1985) did not find a negative association between nonnative Rainbow Trout survival and fine sediments (<2 mm in diameter) in a tributary to Lake Erie in Ontario, Canada. Conversely, fry emergence success declined significantly in redds with proportion of fine sediment less than 6.5 mm for Westslope Cutthroat Trout (Weaver and Fraley 1993) or less than 4 mm for Bonneville Cutthroat Trout *O. clarkii utah* (Budy et al. 2012). In short, habitat alterations resulting in an increased proportion of smaller substrate and fine sediment may inhibit the spawning success of Westslope Cutthroat Trout. However, more-direct studies of the effects of fine sediment and preferred spawning gravels for Rainbow Trout and hybrids are needed to better address this hypothesis.

In our study, hybridization between native Westslope Cutthroat Trout and invasive Rainbow Trout was not influenced by multispecies interactions that included an introduced parasite. Researchers have predicted that climate change will warm stream temperatures, resulting in reduced habitat for native trout and increased habitat for nonnative trout throughout the Rocky Mountains (Williams et al. 2009; Wenger et al. 2011). Similarly, human activities and climate change are expected to cause further expansion of wildlife diseases and to alter host–pathogen interactions (Daszak et al. 2001; Fuller et al. 2012; Gallana et al. 2013).

Cutthroat Trout inhabit some of the highest-gradient streams of all salmonids and often occupy reaches where no other fish are present (Bozek and Hubert 1992; Paul and Post 2001; Quist and Hubert 2004; Rasmussen et al. 2010; D'Angelo and Muhlfeld 2013). Geomorphic characteristics (e.g., stream slope) may limit species expansion in certain types of stream, such as high-gradient, high-elevation tributaries. Biologists should incorporate geomorphic variables in addition to variables like temperature and precipitation when outlining their expectations for community composition and native species conservation in the coming decades. Once hybridization has occurred, habitat restoration efforts alone cannot remove Rainbow Trout alleles from a population. Additionally, restoration cannot change the broad-scale geomorphic characteristics of habitat, such as stream slope. However, results from this study may help to prioritize areas where restoration aimed at maintaining temperatures, bank stability, and hydrologic regimes that favor Westslope Cutthroat Trout could reduce the likelihood of Rainbow Trout invasion.

As community assemblages continue to change, we must continually evaluate the effects of biotic interactions across communities and across landscapes. Interactions between various nonnative species could either control or facilitate their invasions. Knowledge of how nonnative species interact with each other and with native species in the communities and habitats they invade will assist managers in developing and prioritizing conservation action strategies for the long-term protection of native species in the wild.

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REFERENCES

Allendorf, F., R. F. Leary, P. Spruell, and J. K. Wenburg. 2001. Problems with hybrids: setting conservation guidelines. Trends in Ecology and Evolution 16:613–622.

Allendorf, F. W., G. Luikart, and S. N. Aitken. 2012. Conservation and the genetics of populations. Wiley, Chichester, UK.

Arnold, T. W. 2010. Uninformative parameters and model selection using Akaike's information criteria. Journal of Wildlife Management 74:1175–1178.

Bartholomew, J. L., and P. W. Reno. 2002. The history and dissemination of whirling disease. Pages 3–24 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Bates, D., M. Maechler, B. Bolker, and S. Walker. 2014. lme4: linear mixed-effects models using Eigen and S4. R package version 1.1-7. Available: http://CRAN.R-project.org/package=lme4. (February 2015).

Bennett, S. N., J. R. Olson, J. L. Kershner, and P. Corbett. 2010. Propagule pressure and stream characteristics influence introgression: Cutthroat and Rainbow trout in British Columbia. Ecological Applications 20:263–277.

Bozek, M. A., and W. A. Hubert. 1992. Segregation of resident trout in streams as predicted by three habitat dimensions. Canadian Journal of Zoology 70:886–890.

Braks, M. A. H., N. A. Honório, L. P. Lounibos, R. Lourenço-De-Oliveira, and S. A. Juliano. 2004. Interspecific competition between two invasive species of container mosquitoes, *Aedes aegypti* and *Aedes albopictus* (Diptera: Culicidae), in Brazil. Annals of the Entomological Society of America 97:130–139.

Budy, P., S. Wood, and B. Roper. 2012. A study of the spawning ecology and early life history survival of Bonneville Cutthroat Trout. North American Journal of Fisheries Management 32:436–449.

Buehrens, T. W., J. Glasgow, C. O. Ostberg, and T. P. Quinn. 2013. Spatial segregation of spawning habitat limits hybridization between sympatric native steelhead and Coastal Cutthroat Trout. Transactions of the American Fisheries Society 142:221–233.

Burkhead, N. M. 2012. Extinction rates in North American freshwater fishes, 1900-2010. Bioscience 62:798–808.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer-Verlag, New York.

Cairney, M., J. B. Taggart, and B. Hoyheim. 2000. Characterization of microsatellite and minisatellite loci in Atlantic Salmon (*Salmo salar*) and cross species amplification in other salmonids. Molecular Ecology 9:2175–2178.

Corsi, M. P., L. A. Eby, and C. A. Barfoot. 2013. Hybridization with Rainbow Trout alters life history traits of native Westslope Cutthroat Trout. Canadian Journal of Fisheries and Aquatic Sciences 70:895–904.

D'Angelo, V. S., and C. C. Muhlfeld. 2013. Factors influencing the distribution of native Bull Trout and Westslope Cutthroat Trout in streams of western Glacier National Park, Montana. Northwest Science 87:1–11.

Daszak, P., A. A. Cunningham, and A. D. Hyatt. 2001. Anthropogenic environmental change and the emergence of infectious diseases in wildlife. Acta Tropica 78:103–116.

De la Hoz, E., and P. Budy. 2004. Linking environmental heterogeneity to the distribution of prevalence of *Myxobolus cerebralis*: a comparison across sites in a northern Utah watershed. Transactions of the American Fisheries Society 133:1176–1189.

Dudgeon, D., A. H. Arthington, M. O. Gessner, Z.-I. Kawabata, D. J. Knowler, C. Lévêque, R. J. Naiman, A.-H. Prieur-Richard, D. Soto, M. L. J. Stiassny, and C. A. Sullivan. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81:163–182.

ESRI (Environmental Systems Research Institute). 2010. ArcGIS desktop: release 10. ESRI, Redlands, California.

Fausch, K. D. 2007. Introduction, establishment and effects of non-native salmonids: considering the risk of Rainbow Trout invasion in the United Kingdom. Journal of Fish Biology 71:1–32.

Fausch, K. D., Y. Taniguchi, S. Nakano, G. D. Grossman, and C. R. Townsend. 2001. Flood disturbance regimes influence Rainbow Trout invasion success among five holarctic regions. Ecological Applications 11:1438–1455.

Fuller, T., S. Bensch, I. Müller, J. Novembre, J. Pérez-Tris, R. E. Ricklefs, T. B. Smith, and J Waldenström. 2012. The ecology of emerging infectious diseases in migratory birds: an assessment of the role of climate change and priorities for future research. Ecohealth 9:80–88.

Gallana, M., M. P. Ryser-Degiorgis, T. Wahli, and H. Segner. 2013. Climate change and infectious diseases of wildlife: altered interactions between pathogens, vectors and hosts. Current Zoology 59:427–437.

Granath, W. O., M. A. Gilbert, E. J. Wyatt-Pescador, and E. R. Vincent. 2007. Epizootiology of *Myxobolus cerebralis*, the causative agent of salmonid whirling disease, in the Rock Creek drainage of west-central Montana. Journal of Parasitology 93:104–119.

Hallett, S. L., and J. L. Bartholomew. 2008. Effects of water flow on the infection dynamics of *Myxobolus cerebralis*. Parasitology 135:371–384.

Harrell, F. E. 2014. Hmisc: Harrell miscellaneous. R package version 3.14-4. Available: http://CRAN.R-project.org/package=Hmisc. (February 2015).

Heath, D., C. M. Bettles, and D. Roff. 2009. Environmental factors associated with reproductive barrier breakdown in sympatric trout populations on Vancouver Island. Evolutionary Applications 3:77–90.

Hedrick, R. P., and M. El-Matbouli. 2002. Recent advances with taxonomy, life cycle, and development of Myxobolus crebralis in the fish and oligochaete host. Pages 45–54 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: review and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Heiberger, R. M. 2014. HH: statistical analysis and data display: Heiberger and Holland. R package version 3.0-4. Available: http://CRAN.R-project.org/package=HH. (February 2015).

Hitt, N. P., C. A. Frissell, C. C. Muhlfeld, and F. W. Allendorf. 2003. Spread of hybridization between native Westslope Cutthroat Trout, *Oncorhynchus clarki lewisi*, and nonnative Rainbow Trout, *Oncorhynchus mykiss*. Canadian Journal of Fisheries and Aquatic Science 60:1440–1451.

Hobbs, R. J., E. Higgs, and J. A. Harris. 2009. Novel ecosystems: implications for conservation and restoration. Trends in Ecology and Evolution 24:599–605.

Holm, L. E., and K. Brusgaar. 1999. Two polymorphic dinucleotide repeats in Rainbow Trout (*Oncorhynchus mykiss*). Animal Genetics 30:162–163.

Johnson, P. J., J. Olden, C. Solomon, and M. J. Zanden. 2009. Interactions among invaders: community and ecosystem effects of multiple invasive species in an experimental aquatic system. Oecologia 159:161–170.

Kanda, N., R. F. Leary, and F. W. Allendorf. 2002. Evidence of introgressive hybridization between Bull Trout and Brook Trout. Transactions of the American Fisheries Society 131:772–782.

Kovach, R. P., L. A. Eby, and M. P. Corsi. 2011. Hybridization between Yellowstone Cutthroat Trout and Rainbow Trout in the upper Snake River basin, Wyoming. North American Journal of Fisheries Management 31:1077–1087.

Lindenmayer, D. B., J. Fischer, A. Felton, M. Crane, D. Michael, C. Macgregor, R. Montague-Drake, A. Manning, and R. J. Hobbs. 2008. Novel ecosystems resulting from landscape transformation create dilemmas for modern conservation practice. Conservation Letters 1:129–135.

MacConnell, E., and E. R. Vincent. 2002. The effects of Myxobolus cerebralis on the salmonid host. Pages 95–108 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Miller, M. P., and E. R. Vincent. 2008. Rapid natural selection for resistance to an introduced parasite of Rainbow Trout. Evolutionary Applications 1:336–341.

Montgomery, D. C., E. A. Peck, and G. G. Vining. 2012. Introduction to linear regression analysis, 5th edition. Wiley, Hoboken, New Jersey.

Montgomery, D. R., E. M. Beamer, G. R. Pess, and T. P. Quinn. 1999. Channel type and salmonid spawning distribution and abundance. Canadian Journal of Fisheries and Aquatic Sciences 56:377–387.

Moyle, P. B., and T. Light. 1996. Biological invasions of fresh water: empirical rules and assembly theory. Biological Conservation 78:149–161.

MTFWP (Montana Fish, Wildlife, and Parks). 2007. Memorandum of understanding and agreement for Westslope Cutthroat Trout and Yellowstone Cutthroat Trout in Montana. MTFWP, Helena.

Muhlfeld, C. C., R. P. Kovach, L. A. Jones, R. Al-Chokhachy, M. C. Boyer, R. F. Leary, W. H. Lowe, G. Luikart, and F. W. Allendorf. 2014. Invasive hybridization in a threatened species is accelerated by climate change. Nature Climate Change 4:620–624.

Muhlfeld, C. C., T. E. MacMahon, D. Belcer, and J. L. Kershner. 2009a. Spatial and temporal spawning dynamics of native Westslope Cutthroat Trout, *Oncorhynchus clarkii lewisi*, introduced Rainbow Trout, *Oncorhynchus mykiss*, and their hybrids. Canadian Journal of Aquatic Science 66:1153–1168.

Muhlfeld, C. C., T. E. McMahon, M. C. Boyer, and R. E. Gresswell. 2009b. Local habitat, watershed, and biotic factors influencing the spread of hybridization between native Westslope Cutthroat Trout and introduced Rainbow Trout. Transactions of the American Fisheries Society 138:1036–1051.

Ostberg, C. O., and R. J. Rodriguez. 2004. Bi-parentally inherited species-specific markers identify hybridization between Rainbow Trout and Cutthroat Trout subspecies. Molecular Ecology 4:26–29.

Paul, A. J., and J. R. Post. 2001. Spatial distribution of native and nonnative salmonids in streams of the eastern slopes of the Canadian Rocky Mountains. Transactions of the American Fisheries Society 130:417–430.

Peterson, D. P., and K. D. Fausch. 2003. Testing population-level mechanisms of invasion by a mobile vertebrate: a simple conceptual framework for salmonid streams. Biological Invasions 5:239–259.

Pierce, R., R. Aasheim, and C. Podner. 2005. An integrated stream restoration and native fish conservation strategy for the Blackfoot River basin. Montana Fish, Wildlife, and Parks, Missoula.

Pierce, R. W., R. B. Aasheim, and C. S. Ponder. 2007. Fluvial Westslope Cutthroat movements and restoration relationships in the upper Blackfoot basin, Montana. Intermountain Journal of Sciences 13:72–85.

Pierce, R., M. Davidson, and C. Podner. 2012. Spawning behavior of Mountain Whitefish and co-occurrence of *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 141:720–730.

Pierce, R., and C. Podner. 2006. The Big Blackfoot River fisheries restoration report for 2004 and 2005. Montana Fish, Wildlife, and Parks, Missoula.

Pierce, R., C. Podner, and K. Carim. 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 142:68–81.

Pierce, R., C. Podner, M. Davidson, and E. R. Vincent. 2009. Correlation of fluvial Rainbow Trout spawning life history with severity of infection by *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 138:251–263.

Pierce, R. W., C. Podner, M. Davidson, L. Knotek, and J. Thabes. 2008. The Big Blackfoot River fisheries and restoration investigations for 2006 and 2007. Montana Fish, Wildlife, and Parks, Missoula.

Pierce, R., C. Podner, and J. McFee. 2001. Blackfoot River fisheries inventory monitoring and restoration report for 2001. Montana Fish, Wildlife, and Parks, Missoula.

Pimm, S. L., and P. Raven. 2000. Biodiversity: extinction by numbers. Nature 403:843–845.

Quist, M. C., and W. A. Hubert. 2004. Bioinvasive species and the preservation of Cutthroat Trout in the western United States: ecological, social, and economic issues. Environmental Science and Policy 7:303–313.

R Development Core Team. 2012. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Rasmussen, J. B., M. D. Robinson, and D. D. Heath. 2010. Ecological consequences of hybridization between native Westslope Cutthroat (*Oncorhynchus clarkii lewisi*) and introduced Rainbow (*Oncorhynchus mykiss*) Trout: effects on life history and habitat use. Canadian Journal of Aquatic Science 67:357–370.

Rasmussen, J. B., M. D. Robinson, A. Hontela, and D. D. Heath. 2012. Metabolic traits of Westslope Cutthroat Trout, introduced Rainbow Trout and their hybrids in an ecotonal hybrid zone along an elevation gradient. Biological Journal of the Linnean Society 105:56–72.

Rexroad, E., R. L. Coleman, A. L. Gustafson, W. K. Hershberger, and J. Killefer. 2002. Development of Rainbow Trout microsatellite markers from repeat enriched libraries. Marine Biotechnology 4:12–16.

Ricciardi, A. 2001. Facilitative interactions among aquatic invaders: is an "invasional meltdown" occurring in the Great Lakes? Canadian Journal of Fisheries and Aquatic Sciences 58:2513–2525.

Ricciardi, A., and J. B. Rasmussen. 1999. Extinction rates of North American freshwater fauna. Conservation Biology 13:1220–1222.

Ross, D. J., C. R. Johnson, C. L. Hewitt, and G. M. Ruiz. 2004. Interaction and impacts of two introduced species on a soft-sediment marine assemblage in southeast Tasmania. Marine Biology 144:747–756.

Ryce, E. K. N., A. V. Zale, E. MacConell, and M. Nelson. 2005. Effects of fish age versus size on the development of whirling disease in Rainbow Trout. Diseases of Aquatic Organisms 63:69–76.

Shepard, B. B., B. E. May, and W. Urie. 2005. Status and conservation of Westslope Cutthroat within the western United States. North American Journal of Fisheries Management 25:1426–1440.

Simberloff, D., and B. Von Holle. 1999. Positive interactions of nonindigenous species: invasional meltdown? Biological Invasions 1:21–32.

Smith, C. T., B. F. Koop, and J. R. Nelson. 1998. Isolation and characterization of Coho Salmon (*Oncorhynchus kisutch*) microsatellites and their use in other salmonids. Molecular Ecology 7:1614–1616.

Sowden, T. K., and G. Power. 1985. Prediction of Rainbow Trout embryo survival in relation to groundwater seepage and particle size of spawning substrate. Transactions of the American Fisheries Society 114:804–812.

Spies, B., D. J. Brasier, P. T. L. O'Reilly, T. R. Seamons, and P. Bentzen. 2005. Development and characterization of novel tetra-, tri-, and dinucleotide markers in Rainbow Trout (*Oncorhynchus mykiss*). Molecular Ecology Notes 5:278–281.

Stevenson, N. J., and E. K. Mills. 1999. Streambank and shoreline condition. Pages 115–124 *in* M. B. Bain and N. K. Stevenson, editors. Aquatic habitat assessment: common methods. American Fisheries Society, Bethesda, Maryland.

Strayer, D. L., and D. Dudgeon. 2010. Freshwater biodiversity conservation: recent progress and future challenges. Journal of the North American Benthological Society 29:344–358.

Venables, W. N., and B. D. Ripley. 2002. Modern applied statistics with S. Springer, New York.

Vincent, E. R. 1996. Whirling disease and wild trout: the Montana experience. Fisheries 21(6):32–33.

Vincent, E. R. 2002. Relative susceptibility of various salmonids to whirling disease with emphasis on Rainbow and Cutthroat trout. Pages 109–115 *in* J. L. Bartholomew and J. C. Wilson, editors. Whilring disease: review and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Wagner, E., R. Arndt, M. Brough, C. Wilson, and G. Nelson. 2002. Survival, performance, and resistance to *Myxobolus cerebralis* infection of Lake Trout × Brook Trout hybrids. North American Journal of Fisheries Management 22:760–769.

Walther, G. R., E. Post, P. Convey, A. Menzel, C. Parmesan, T. J. Beebee, J. M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein. 2002. Ecological responses to recent climate change. Nature 416:389–395.

Weaver, T. M., and J. J. Fraley. 1993. A method to measure emergence success of Westslope Cutthroat Trout fry from varying substrate compositions in a natural stream channel. North American Journal of Fisheries Management 13:817–822.

Weigel, D. E., J. T. Peterson, and P. Spruell. 2003. Introgressive hybridization between native Cutthroat Trout and introduced Rainbow Trout. Ecological Applications 13:38–50.

Wenger, S. J., D. J. Isaak, C. H. Luce, H. M. Neville, K. D. Fausch, J. B. Dunham, D. C. Dauwalter, M. K. Young, M. M. Elsner, B. E. Rieman, A. F. Hamlet, and J. E. Williams. 2011. Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. Proceedings of the National Academy of Sciences of the USA 108:14175–14180.

Williams, J. E., A. L. Haak, H. Neville, and W. T. Colyer. 2009. Potential consequences of climate change to persistence of Cutthroat Trout populations. North American Journal of Fisheries Management 29:533–548.

Yau, M. M., and E. B. Taylor. 2013. Environmental and anthropogenic correlates of hybridization between Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*) and introduced Rainbow Trout (*O. mykiss*). Conservation Genetics 14:885–900.

Zachheim, H. 2006. A history of Montana Fish, Wildlife, and Parks Fisheries Division, 1901–2005. Montana Fish, Wildlife, and Parks, Helena

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ARTICLE

Multiscale Prediction of Whirling Disease Risk in the Blackfoot River Basin, Montana: a Useful Consideration for Restoration Prioritization?

L. A. Eby*
Wildlife Biology Program, College of Forestry and Conservation, University of Montana, 32 Campus Drive, Missoula, Montana 59812, USA

R. Pierce
Montana Fish, Wildlife and Parks, 3201 Spurgin Road, Missoula, Montana 59804, USA

M. Sparks[1] and K. Carim
Wildlife Biology Program, College of Forestry and Conservation, University of Montana, 32 Campus Drive, Missoula, Montana 59812, USA

C. Podner
Montana Fish, Wildlife and Parks, 3201 Spurgin Road, Missoula, Montana 59804, USA

Abstract

Habitat restoration for inland trout (family Salmonidae) is common across western North America, but planners rarely consider disease risk when prioritizing restoration sites. Whirling disease is a parasitic infection caused by the invasive myxosporean parasite *Myxobolus cerebralis* and has been implicated in declines of wild trout populations across western North America. For planners to consider disease, disease risk needs to be predictable across the landscape and influence restoration outcomes. We collated the history of whirling disease infection severity scores on the MacConnell–Baldwin scale from sentinel cage studies for hatchery Rainbow Trout *Oncorhynchus mykiss* in the Blackfoot River basin from 1998 to 2009. At these same sites, we performed reach-scale geomorphic assessments, derived landscape variables from GIS data layers, and assembled fish composition data. We examined relationships between the severity of infection and several landscape-scale and reach-scale variables for 13 basin-fed streams in the Blackfoot River basin of west-central Montana using classification and regression tree analyses. In our data set, valley slope and forest cover were the best predictors of fine sediment. Both spring creeks and gently sloping alluvial basin-fed tributaries to the Blackfoot River basin with higher proportions of fine sediment (particle size < 0.85 mm) were associated with a high severity (≥ grade 3) of infection. Additionally, we explored differences in trout species composition (i.e., susceptible versus resistant species) before and after the whirling disease enzootic using seven basin-fed streams and two spring creeks. We did not detect trout community shifts from susceptible to disease-resistant salmonids in basin-fed disease-positive streams. However, spring creeks showed a negative trend in disease-susceptible salmonids after the whirling disease enzootic. Disease risk appears to be predictable across the landscape and may limit possible restoration outcomes by influencing species composition in spring creeks.

*Corresponding author: lisa.eby@umontana.edu
[1]Present address: University of Alaska Fairbanks, Post Office Box 75700, Fairbanks, Alaska 99775, USA.
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The prioritization of habitat restoration efforts for inland wild trout (family Salmonidae) is often based on the distribution of focal species (i.e., species of special concern), cooperation of landowners or land availability (Sudduth et al. 2007), feasibility, costs, benefits, and available funding (Aitken 1997; Roni et al. 2002; Beechie et al. 2008; Pierce et al. 2008). When prioritization frameworks are developed, they rarely consider disease risk. But if the presence of disease constrains restoration outcomes, it would be useful to consider disease when prioritizing and planning restoration activities. Whether disease should be integrated into restoration planning depends on whether we can predict the risk of disease across landscapes and whether the presence of disease influences restoration outcomes.

Whirling disease is a parasitic infection caused by the invasive myxosporean parasite *Myxobolus cerebralis* (Hoffman 1990; Bartholomew and Wilson 2002), which coevolved on the Eurasian continent with Brown Trout *Salmo trutta* (Bartholomew and Reno 2002). *Myxobolus cerebralis* was introduced on the East Coast of North America in the 1950s, then spread rapidly westward (Bartholomew and Reno 2002). The parasite is now widely established across western North America; however, the prevalence and severity of whirling disease infection is highly variable across river systems (e.g., Hiner and Moffitt 2001; De la Hoz and Budy 2004; Neudecker et al. 2012). As part of its life cycle, *M. cerebralis* requires an oligochaete worm (such as the sludge worm *Tubifex tubifex*) to develop triactinomyxon (TAM) actinospores, which are released into the water typically when stream temperatures are between 10°C and 15°C (El-Matbouli et al. 1999; De la Hoz and Budy 2004; Kerans et al. 2005). Young salmonids (<9 weeks posthatch) are most susceptible to infection by TAM actinospores (MacConnell and Vincent 2002). If exposed to TAMs at this susceptible age, severe infection can occur (Ryce et al. 2004). Clinical signs of whirling disease may include discoloration of the tail, whirling behavior, and skeletal deformities (MacConnell and Vincent 2002).

Whirling disease has been implicated in the decline of several wild trout populations in western North America (Nehring and Walker 1996; Vincent 1996; Nehring 2006), including various rivers in western Montana (MacConnell and Vincent 2002; Granath et al. 2007; McMahon et al. 2010). These declines have been implicated in shifting trout communities dominated by susceptible species (e.g., Rainbow Trout *Oncorhynchus mykiss*) to ones dominated by resistant species, such as Brown Trout (Granath and Vincent 2010). Thus, restoring trout habitat where whirling disease is present may create an ecological sink for susceptible trout, while favoring resistant species (Granath et al. 2007; McMahon et al. 2010). Unfortunately, forecasting the impacts of whirling disease on local salmonid communities is especially difficult given the complex nature of the multihost life cycle, the environmental preferences of both the hosts and the parasite (Hedrick et al. 1999; Kerans and Zale 2002), and the lack of long-term field studies examining fish populations in streams with and without whirling disease (Karr et al. 2005; Hansen and Budy 2011).

Within and among watersheds, the prevalence (percent infected) and severity of whirling disease infection depends upon the abundance and presence of susceptible fish hosts (MacConnell and Vincent 2002), the strains of *T. tubifex* present and overall oligochaete community composition (Beauchamp et al. 2005; Nehring et al. 2013, 2014), and the physical environment (e.g., Anlauf and Moffitt 2008; Neudecker et al. 2012). Environmental conditions, such as substrate composition, stream temperature, and velocity, can influence the abundance of *T. tubifex*, the production of TAMs, and the susceptibility of different salmonid hosts to infection (e.g., Allen and Bergersen 2002; Kerans et al. 2005; Hallett et al. 2009). Other studies highlight the specificity of this parasite to particular strains of *T. tubifex* and the potential role of resistant strains serving as a filter and reducing the number of spores that complete their life cycle (Beauchamp et al. 2005; Nehring et al. 2013, 2014), as well as the potential for the development of resistance in certain fish populations (Baerwald et al. 2008; Miller and Vincent 2008).

Given these complexities, predicting high-quality habitat for *T. tubifex* may be the most practical approach for determining areas of highest risk for *M. cerebralis* infection (e.g., Allen and Bergersen 2002; Schisler et al. 2006; Anlauf and Moffitt 2010). The relationship between the occurrence of *T. tubifex* and fine substrate is well established (Lazim and Learner 1987; Kaeser and Sharpe 2006; Anlauf and Moffitt 2008), and *T. tubifex* distribution is positively associated with organic matter and nutrients (Sauter and Gude 1996; Arndt et al. 2002). Previous studies found that incorporating variables from both the reach scale (e.g., amount of slow habitat such as pools) and landscape scale (e.g., watershed size and land cover) were the best approach to predicting fine sediment but highlighted that these predictors of fine substrate needed to be validated across multiple drainages to generalize broader trends (Anlauf and Moffitt 2010). At the reach scale, characteristics such as stream slope, depth, channel sinuosity, bank stability, flow regime, and riparian livestock damage can also impact local substrate composition. While, at the landscape scale, habitat formation and substrate characteristics are influenced by variation in basin hydrology linked to climate, valley slope, lithology, and properties of the soils (e.g., Frissell et al. 1986; Poff and Ward 1989), as well as anthropogenic changes to land cover (i.e., reduced forest cover [Allan et al. 1997]).

To evaluate our ability to predict the risk of whirling disease across a river basin, we must validate whether landscape-scale and reach-scale variables can predict fine sediment (e.g., Anlauf and Moffitt 2008). To understand the potential for restoration to alter the spatial patterns of whirling disease presence, we must also understand whether disease severity is associated with fine sediment and the significant physical predictors of fine sediment. In addition, we need a better understanding of whether whirling disease can influence community composition and, therefore, limit restoration outcomes. In the Blackfoot River basin, Montana, sites were periodically monitored for whirling disease prevalence and severity (Table 1; Pierce et al. 2009, 2012; McMahon et al. 2010; Neudecker et al. 2012) between 1998 and 2009, allowing us to investigate whirling disease risk across the basin. Our study addresses three questions: First, how well do landscape-scale characteristics (valley slope, sinuosity, stream order, and percent forest cover) and reach-scale characteristics (bank-full width and depth, channel slope, and entrenchment ratio) predict fine sediment? Second, do the same variables that predict fine sediment also predict whirling disease infection severity? Finally, does the presence of whirling disease result in a community dominated by more resistant species thus limiting recovery of more susceptible species? Answering these questions will help us understand how the presence of whirling disease may alter the range of possible restoration goals that are achievable at a given site.

METHODS

Study Area

The Blackfoot River, a fifth-order tributary (Strahler 1957) of the upper Columbia River, lies in west-central Montana and flows west 211 km from the Continental Divide to its confluence with the Clark Fork River in Bonner, Montana (Montana DNRC 1984). The geography of the watershed is a physically diverse, glacial landscape with alpine and subalpine mountains at the upper elevations, montane forests at the middle elevations, and semiarid glacial pothole and outwash topography on the valley floor. Larger tributaries of the Blackfoot River, located in the mid to upper basin, typically begin in glacial valleys, flow through steep headwaters, and then transition to meandering streams in broad valleys with gentle relief on the floor of the Blackfoot Valley. Conversely, smaller tributaries in the lower Blackfoot River basin flow through confined steeper channels before directly entering the lower Blackfoot River (Alt and Hyndman 1986). The Blackfoot River contains diverse self-sustaining wild trout populations, many of which have migratory behavior and reproduce in tributaries (Swanberg 1997; Schmetterling 2001; Pierce et al. 2009). Many of the Blackfoot River basin tributaries have been targets for a variety of restoration activities, including all the tributaries used in this study (Pierce et al. 2008, 2013). Typical restoration activities include a mix of improved fish passage, reduction of entrainment, active channel restoration, grazing changes, removal of streamside feedlots, and increased instream flows in each tributary. Even though all of the tributaries in this study have received some efforts towards habitat improvement, restoration activities occurred upstream of the whirling disease monitoring sites.

Native salmonids of the Blackfoot River basin include Westslope Cutthroat Trout *O. clarkii lewisi*, a Montana Species of Special Concern (Shepard et al. 2005), native Bull Trout *Salvelinus confluentus*, a char designated as threatened under the Endangered Species Act (USFWS 2010), and Mountain Whitefish *Prosopium williamsoni*, a species common to the Blackfoot River (Pierce et al. 2012). Nonnative trout include Rainbow Trout, Brook Trout *S. fontinalis*, and Brown Trout (Pierce et al. 2012). Other native nongame fishes are present in the main stem but those occasionally captured in the tributaries in low numbers include Slimy Sculpin *Cottus cognatus* and Longnose Dace *Rhinichthys cataractae*.

Based on laboratory exposures, Rainbow Trout, Brook Trout, Westslope Cutthroat Trout, Bull Trout, and Mountain Whitefish possess high or intermediate susceptibility to whirling disease (MacConnell and Vincent 2002), whereas nonnative Brown Trout are the only fish naturally more resistant to the parasite due to their coevolution with *M. cerebralis* in Eurasia (Bartholomew and Reno 2002). Considering the spatial and temporal overlap of young, small, vulnerable fish and the production of TAM actinospores can help link susceptibility and exposure to predict vulnerability and highlight where whirling disease may be most likely to cause population-level impacts. In basin-fed streams, the emergence of Rainbow Trout and Cutthroat Trout fry overlaps with the peak of TAM production in the early to midsummer (Vincent 2000; Downing et al. 2002; Pierce et al. 2009). Fall-spawning susceptible fishes (Brook Trout and Bull Trout) in basin-fed streams have lower exposure to *M. cerebralis* (MacConnell and Vincent 2002) due to hatching periods that do not overlap with the seasonal peak in TAM production (Pierce et al. 2009; Neudecker et al. 2012). In spring creeks TAM production begins in late fall and lasts longer, resulting in exposure of young Brook Trout spawned in low to middle elevation sites (Neudecker et al. 2012). Overall, Bull Trout make up a small component of the catches in the streams included in this dataset and they typically spawn higher in the watershed (higher slopes, bigger substrate, and cooler temperatures), where the presence of whirling disease is less frequently observed.

Whirling Disease Exposures

Sentinel cage exposures of hatchery Rainbow Trout (50 age-0 diploid cohorts) were used to determine disease prevalence and severity within each of the 17 study streams (Figure 1). Streams had between two and nine exposure events over the 12-year study period. As described in prior studies (Pierce et al. 2009, 2012; Neudecker et al. 2012), cages were placed in flowing water and exposures were completed in July within 9 weeks posthatch to coincide with fry emergence in Blackfoot River tributaries, periods of susceptibility (Ryce et al. 2005), and the known seasonal peak of TAM production within many rivers in western Montana (Vincent 2000). Spring-fed systems have a more protracted period of peak TAM production from late fall through spring, which results

TABLE 1. Histological scores indicating the severity of infection summarized as both the mean score from sentinel cages and the percent of exposed individuals scoring above a 3 for severity of infection on the MacConnell–Baldwin scale (0 = nondetect; 5 = severe); scores are presented as follows: mean cage score; percent with >3 infection severity. Stream ID and name relate to the location on the study area map (Figure 1). When no data was available for a year, it is indicated with "nd." The Presence column indicates whether the stream was considered disease positive (1) or negative (0) for the analyses. Fourteen streams were basin-fed streams and two streams (Rock and Kleinschmidt creeks) were spring creeks. Thirteen streams were included in the physical assessments (indicated with a P) and nine streams were used in fish assessment (indicated with an F).

		Year												
Stream ID and name	Assessment	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	Presence
1. Johnson Creek	P	nd	nd	nd	nd	nd	nd	nd	0.0; 0	0.0; 0	nd	nd	nd	0
2. West Twin Creek	P	nd	nd	nd	nd	nd	nd	nd	0.0; 0	0.0; 0	0.0; 0	nd	nd	0
3. East Twin Creek	P	nd	nd	nd	nd	nd	nd	nd	0.0; 0	0.0; 0	nd	nd	nd	0
4. Bear Creek	P, F	nd	nd	nd	nd	nd	nd	0.0; 0	nd	0.0; 0	nd	nd	nd	0
5. Gold Creek	P, F	nd	0.1; 0	0.0; 0	nd	0.0; 0	nd	0.0; 0	0.0; 0	0.0; 0	0.0; 0	nd	0.0; 0	0
6. Belmont Creek	P, F	nd	nd	0.0; 0	nd	0.2; 0	0.4; 4	1.5; 27	2.5; 49	0.3; 2	3.4; 76	2.9; 63	4.4; 89	1
7. Elk Creek	P, F	nd	0.0; 0	0.0; 0	nd	0.0; 0	2.8; 64	4.3; 100	4.8; 98	nd	nd	nd	nd	1
8. Blanchard Creek	P	nd	nd	nd	nd	nd	nd	nd	0.0; 0	nd	nd	nd	nd	0
9. Cottonwood Creek	P	3.7; 94	4.5; 98	nd	nd	4.5; 96	nd	nd	3.8; 100	4.0; 81	4.3; 96	nd	nd	1
10. Chamberlain Creek	P, F	0.2; 0	2.7; 64	nd	nd	2.6; 63	nd	4.3; 98	3.8; 78	nd	1.9; 44	nd	nd	1
11. Monture Creek	P	0.0; 0	0.0; 0	nd	nd	3.2; 68	nd	nd	4.8; 97	4.6; 95	4.3; 91	2.8; 64	4.6; 95	1
12. Rock Creek (spring)[a]	F	nd	0.0; 0	2.3; 47	3.9; 77	nd	3.4; 82	nd	nd	nd	nd	nd	nd	1
13. Kleinschmidt Creek (spring)[a]	F	nd	3.6; 78	4.5; 86	3.8; 80	nd	4.9; 98	4.7; 93	nd	nd	nd	nd	nd	1
14. Arrastra Creek	P, F	nd	nd	nd	nd	nd	0.3; 4	1.2; 12	0.0; 0	0.1; 0	nd	nd	nd	0
15. Poorman Creek	F	nd	nd	nd	nd	nd	nd	0.8; 12	nd	nd	4.7; 95	nd	nd	1
16. Landers Fork	P	nd	nd	nd	nd	nd	nd	0.1; 0	0.0; 0	0.0; 0	0.0; 0	nd	nd	0

[a]Spring creek sentinel cage exposures were completed in April. All basin-fed stream exposures were completed in July.

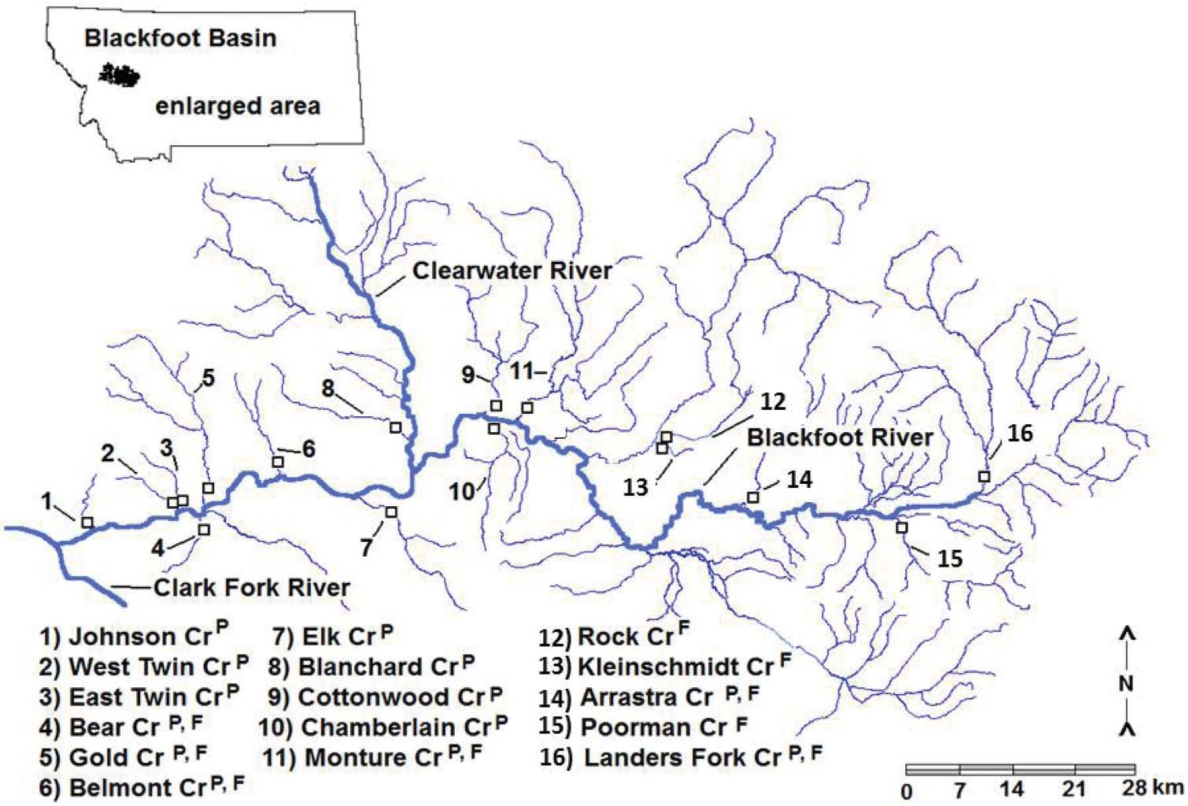

FIGURE 1. Blackfoot River basin (inset in Montana map) and 16 study streams with sentinel cage exposure data for whirling disease ranking (square map symbol on drainage map). Stream ID and stream name relate to histological scores in Table 1. Physical assessments (P) were completed on 13 streams, and 9 sites had fish community trend analyses (F). *Myxobolus cerebralis* is present (indicated by the bold line) throughout the main stem of the Clearwater River, as well as throughout the Blackfoot River from its confluence with the Clark Fork River to upstream of Landers Fork; Cr = Creek. [Figure available online in color.]

in parasite exposure for fall-spawning species (i.e., Brook Trout) during the most susceptible early life stage (Neudecker et al. 2012). In spring creeks, sentinel cage exposures occurred in April. All cages were placed in known areas of spawning and rearing for wild trout. Following field exposures and a holding period to allow the infection to develop, fish were sacrificed and their heads were histologically examined and scored using the MacConnell–Baldwin rating scale (Hedrick et al. 1999; Baldwin et al. 2000; Ryce et al. 2004), which categorically ranked the severity of infection into six qualitative groups: (0) no infection, (1) minimal, (2) mild, (3) moderate, (4) high, and (5) severe. In infections with severity scores >3, *M. cerebralis* digest and destroy cartilage of susceptible young fish, causing inflammation and lesions in the spine and cranium and skeletal damage, which can ultimately elevate mortality (Hedrick et al. 1999; MacConnell and Vincent 2002; Ryce et al. 2004). Based on these impacts to survival, we categorized streams as those with (positive) and without (negative) expected disease population impacts. Specifically, streams in our study were considered disease negative if the average histological score for the sentinel cage exposures was <1.5 for the severity of infection in any year tested and disease positive if the average histological score for the exposure group was >3 severity and the majority of the fish scored grade >3 severity (Table 1).

Physical Variables as Predictors of Fine Sediment and Whirling Disease Severity

Tributary selection.—Landscape-scale and reach-scale variables, as well as disease presence and severity in trout in sentinel cage studies, were collected for 13 basin-fed tributaries to the Blackfoot River (Figure 1; Table 1). Reach-scale field assessments occurred once at each of the whirling disease monitoring reaches near known spawning areas for *Oncorhynchus* spp. (Rainbow Trout, Westslope Cutthroat Trout, or hybrids). All sites in this dataset had the potential for direct invasion by *M. cerebralis* because they are connected to the Blackfoot River in areas where fish infected with *M. cerebralis* occur (Pierce et al. 2009, 2012).

Physical assessments.—At each reach, we examined the substrate for the amount of fine sediment (defined as a particle size <0.85 mm) by extracting a McNeil core sample from six separate riffles using modified methods first described by McNeil and Ahnell (1964). For this assessment, the hollow cone of a McNeil core sampler was pushed 10 cm into the streambed. Substrate was then extracted, dried, and sieved following standardized methods (Shepard et al. 1984). The turbid water within the sampler was measured for fine-sediment content utilizing an Imhoff cone as described in Shepard and Graham (1982) and Shepard et al. (1984). The estimated dry weight of the sediment within the Imhoff cone was added to the weight of material <0.85 mm. We calculated the percent of the sample that was <0.85 mm of particle size to quantify fine sediment (clay, silt, and fine sands) for each sample.

In addition to fine sediment, a related suite of physical stream assessments were conducted at both the landscape and reach scales to obtain our predictor variables. For landscape variables, we used 1:24,000 scale topographic maps and aerial photos in ArcView GIS version 3.3 (http://nris.mt.gov/) to calculate valley slope, sinuosity, stream order, and percent forest cover (Table 2). For reach-scale variables, we performed geomorphic surveys across varying reach lengths (295–3,270 m) to ensure a sampling reach of at least 30 bank-full widths in each stream using methods described by Rosgen (1996). These surveys included bank-full width and depth, and width/depth ratios at riffles, as well as percent channel slope and an entrenchment ratio for a reach (Table 2). We also performed a visual categorical assessment of streambank stability and animal damage as described in Stevenson and Mills (1999).

Analyses.—To identify how well landscape-scale and reach-scale characteristics predict fine sediment, we first

TABLE 2. Description of landscape-scale and reach-scale data used in analyses to predict fine sediment (<0.85 mm) and whirling disease presences in the hierarchical analyses across 13 basin-fed streams in the Blackfoot River basin. Landscape variables were summarized for the watershed, typically upstream of sites, and reaches were 30 bank-full widths.

Habitat variable	Description of measure	Data source
	Landscape variables	
Valley slope	Average stream slope calculated upstream of the sampled reach	1:24,000 digitized stream layer and U.S. Geological Survey (USGS) topographic map
Sinuosity	Average sinuosity calculated upstream of the sampled reach	1:24,000 digitized aerial photos and USGS topographic map
Percent forest	Area classified as forest, typically includes Douglas fir *Pseudotsuga menziesii* and lodgepole pine *Pinus contorta*	1:24,000 USGS topographic maps and quad aerial photos
Stream order	Headwaters are first order and the confluence of two streams of order *n* forms a stream of order *n+1*	1:24,000 digital stream layer from USGS topographic maps
	Reach variables	
Percent reach slope	Longitudinal profile	Measured in field (Rosgen 1996)
Bank-full width/depth ratio	Ratio of bank-full width and bank-full depth measured at a riffle	Measured in field (Rosgen 1996)
Bank-full depth	Average depth of the thalweg in a representative riffle along the reach	Measured in field (Rosgen 1996)
Entrenchment ratio	A measure of floodplain connectivity and vertical containment	Measured in field (Rosgen 1996)
Bank stability (rock and vegetative cover)	Reach visually classified between 5 (very stable) indicating > 90% vegetative cover or > 65% large boulders to a rank of < 1 (no or low stability) evidenced by no or low vegetative cover and banks composed of gravel and fines with no cover from large boulders or other features that would provide resistance to erosion	Measured in field (Stevenson and Mills 1999)
Animal damage	Reach visually classified into one of four categories ranging from undamaged (4) to excessive damage (1) with 76–100% of the reach length impacted as evidenced by erosion	Measured in field (Stevenson and Mills 1999)

examined Pearson's correlation coefficients of the variables to ensure significantly correlated variables were not in the same analyses. We constructed scatterplots of each potential predictor variable versus fine sediment (<0.85 mm) to look for outliers and nonlinearities. We used classification and regression trees (Venables and Ripley 1997) to examine whether the fine sediment differed in response to any of the landscape-scale predictor variables. Classification and regression trees partition a dataset (categorical or continuous data) by recursively partitioning the data into subsets using either continuous or categorical dependent variables (Breiman et al. 1984). Because of the small data set, we set the required minimum node size to three but reduced the number of potential splits of the dataset ("pruned the tree") and set the required minimum deviance explained to 0.05 to prevent overfitting the data. We used this same analytical approach to examine reach-scale stream features that best predict fine sediment. Reach-scale predictors included the bank-full width and depth, channel slope, and entrenchment ratio, as well as rankings for animal damage and streambank stability. We combined the significant predictors from both the landscape-scale and reach-scale analyses to examine whether combining predictors across scale improved our results for predictions of fine sediment.

After establishing which landscape-scale and reach-scale variables predicted fine sediment, we then examined whether fine sediment was associated with the presence of whirling disease at a site. First we conducted a *t*-test to compare differences in fine sediment at sites with disease absence and disease presence. Then we used the same analytical approach as above (classification and regression trees) but limited predictor variables to those relevant for predicting fine sediment at the landscape and reach scales. To better illustrate the relationship between whirling disease infection severity and landscape and reach variables, we plotted the significant predictor variable or variables resulting from the classification and regression tree analysis with the most recent histological results from sentinel cage results for each site reported in Table 1.

Effects of Whirling Disease on Fish Species Composition

To examine the possible whirling-disease-related shifts in susceptible species, we compared trout community composition before and after disease detection in disease-positive and disease-negative streams (as defined above). Even though other species are present in the Blackfoot River basin, salmonid fishes dominate the catch at these tributary sites.

Tributary selection.—To examine the potential changes to species composition, we selected disease-positive tributaries with at least three fish population monitoring sites that had two or more years of fish data collection before the detection of whirling disease and several years of fish monitoring after whirling disease exceeded a histological score of 3.0. We limited our fish dataset to monitoring sites in the lower reaches of tributaries (0.32–6.44 km upstream from the mouth) to maintain proximity with sentinel cage study sites and to avoid confounding trends associated with longitudinal changes in trout community composition. The fish data were averaged across all three sites for each year. This would ensure a robust estimate of the community composed of susceptible species before and after substantial disease impacts in streams that became disease positive. We used the sentinel cage field exposures to estimate the year in which whirling disease was above our disease-negative threshold. For the two streams (Cottonwood and Kleinschmidt creeks) that exceeded our threshold at first testing, we used 1994, the year that whirling disease was detected in the state. For our disease-negative tributaries, we selected tributaries with a similar fish data structure in time and space that also had sentinel cage data indicating low to no exposure to whirling disease. Ultimately, there were seven basin-fed tributaries (four were disease positive) and two spring-fed tributaries (both disease positive) that met these criteria for our analyses.

Fish population data collection.—To determine the relative abundance of wild trout, we performed a single-pass survey using a backpack electrofishing unit during base flow in the summers between 1989 and 2010. Pierce et al. (2013) demonstrated that single-pass estimates were linearly related to population estimates for the same watershed with the same sampling procedures. All fish were identified to species, counted, and measured (total length in millimeters). Due to sampling inefficiencies for age-0 trout, we removed age-0 trout for our analyses (using length-frequency histograms) and used ≥ age-1 fish in our analyses.

Analyses.—To define our high and low susceptible-species groups, we considered susceptibility among species (MacConnell and Vincent 2002) and the seasonality of high parasite exposure in basin-fed streams versus spring-fed streams. For basin-fed streams, all spring spawners (*Oncorhynchus* spp.) were combined to examine trends in the abundance of highly susceptible fish because of both the susceptibility of *Oncorhynchus* spp. to whirling disease (Vincent 2002) and the overlap in emergence of fry during the height of TAM production in the early to midsummer (Vincent 2000; Downing et al. 2002; Pierce et al. 2009). Conversely, fall-spawning fish in basin-fed streams are less vulnerable to *M. cerebralis* (MacConnell and Vincent 2002) because fry emergence occurs at periods that do not overlap with the seasonal peak in TAM production (Neudecker et al. 2012). In the two spring creeks, we included Brook Trout with *Oncorhynchus* spp. into a category of susceptible species because of their susceptibility in laboratory studies (Vincent 2000) and the overlap of young fish with TAM production. Because of the species differences in exposure to TAM basin-fed and spring-fed streams, we analyzed trends in community composition for these stream types separately.

To examine whether whirling disease influenced the proportion of the community composed of susceptible species, we examined the proportion of the total catch composed of susceptible species for each sampling event. We standardized the proportion of total catch composed of susceptible species within each tributary by transforming them to *z*-scores and fit a linear mixed model ($Z_{\text{susceptible}} \sim$ Time, |Stream) to examine trends across time. The fixed variable "Time" refers to the monitoring year (as opposed to calendar year), with the first year of monitoring as year 0. Stream was included in the mixed model as a random variable (similar to blocking by stream). We examined trends in basin-fed disease-positive and disease-negative streams, as well as spring creek disease-positive streams, separately. We ensured our analytical assumptions were met and examined residuals for trends. Statistical analyses were conducted in R Statistical Software (R Development Core Team 2012).

RESULTS

Physical Variables as Predictors of Fine Sediment

The percent of fine sediment (particle size <0.85 mm) measured in Elk Creek was 2.5 times that of any other stream in our study. Therefore, we examined the associations of landscape-scale and reach-scale variables with and without the inclusion of Elk Creek to ensure that this site did not have undue influence on our results.

Among the landscape variables, Pearson's correlation coefficients of all variables predicting fine sediment were less than 0.6 and not significant. As a result, all landscape-scale variables were included in this analysis. Valley slope was the primary explanatory variable predicting fine sediment with and without Elk Creek included in the analyses. With Elk Creek in the analyses, valley slope (breaks <0.8 and <1.75) was the only variable in the model, with a residual deviance of 21.65. Without Elk Creek in the analyses, valley slope (<1.75) and percent forest cover (<86.85%) remained in the final model, with a final residual mean deviance of 2.04. In this final model, the majority of variance in fine sediment was explained by valley slope. Within more gentle-sloping valleys, less forest cover was also associated with higher fine sediment in the substrate.

There were no significant correlations among the reach-scale variables and all Pearson's correlation coefficients were less than 0.65. As a result, all reach-scale variables were included in this analysis. With Elk Creek included in the analyses, regression tree results indicated slope (<0.00775) and depth (<1.37 m) were important explanatory reach-scale variables of fine sediment, with a residual mean deviance of 29.05. Lower-gradient channels and sites with greater bank-full depth had more fine sediment. With the removal of Elk Creek, bank-full depth was the best explanatory variable, with an initial break at <1.50 m and then <1.02 m, and a model residual mean deviance of 3.11. Not surprisingly, deeper sites, again, had more fine sediment.

To combine information across landscape and reach scales, we examined correlations between all landscape-scale and

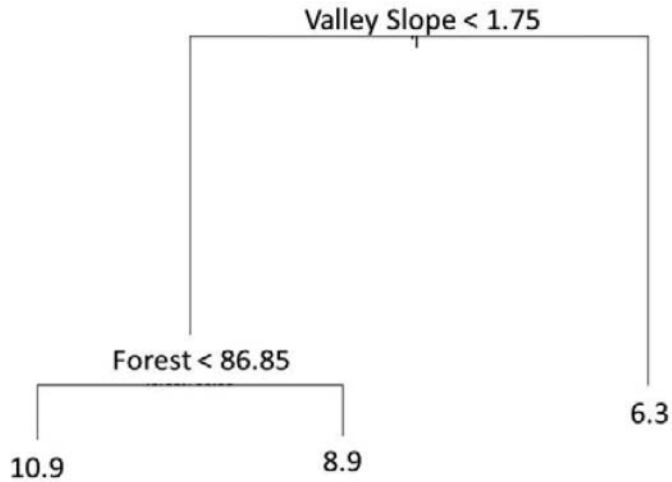

FIGURE 2. Results for the predictions of the percentage of fine sediment in cores (<0.85 mm) from the regression tree analysis of landscape-scale and reach-scale variables, for which branch length indicates the amount of variance explained by the split and end nodes are the predicted values of fine sediment. A final model with only landscape-scale predictors had the lowest residual mean deviance (2.04) of all models examined, and valley slope explained the most variance, with steeper watersheds having less fine sediment. Secondarily within the gentle valley slope grouping, those sites with more percent forest cover had less fine sediment.

reach-scale variables and found only one significant correlation—between valley slope and channel slope (0.88; $P < 0.01$). As valley slope was a key landscape predictor, we retained the landscape valley slope variable in the multiscale analysis. The resultant regression tree model only contained landscape variables (valley slope and forest cover; Figure 2), as including the reach-scale variable of depth did not improve our model.

Physical Variables as Predictors of Whirling Disease Severity

As expected, we found that fine-sediment levels were higher in disease-positive than disease-negative streams (Figure 3A; $t = -2.01$, $P = 0.03$, $n = 13$; one tailed). Classification and regression tree results considering the landscape-scale and reach-scale variables selected above found that valley slope (<0.8) and forest cover (<91.3%) were the best predictors of whirling disease and resulted in 1 misclassification out of 12 streams. Similar to the results examining variance in fine sediment, the landscape-scale variables had a better fit than the reach-scale predictors when combined. We plotted this relationship to better illustrate the association between valley slope and whirling disease (Figure 3B).

Effect of Whirling Disease on Species Composition

We did not observe any trends in trout community composition over time that were associated with whirling disease

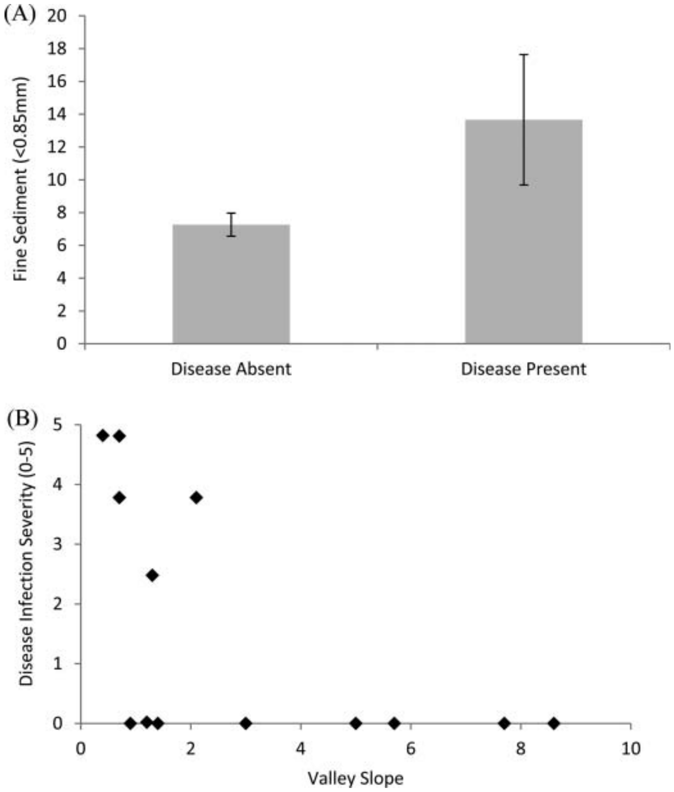

FIGURE 3. Results for basin-fed tributary sites, showing (**A**) the average percentage of fine sediment (error bars are 95% CIs) in disease-positive sites (sites with a severity of infection > 3) and disease-negative sites (infection severity < 2) for basin-fed tributarties and (**B**) the most recent average histological scores on the MacConnell–Baldwin rating scale from sentinel cages (from Table 1) for basin-fed tributary sites versus valley slope.

presence in our basin-fed streams (Figures 4, 5). The slope of the trend line was not significant in either basin-fed disease-negative streams ($Z_{\text{susceptible}} = 0.014 \times \text{Time} - 0.19$; 95% CI = −0.041 to +0.07) or disease-positive streams ($Z_{\text{susceptible}} = -0.025 \times \text{Time} + 0.32$; 95% CI = −0.08 to +0.03; Figures 4, 5). For the two spring creeks, we detected a negative trend over time in susceptible species ($Z_{\text{susceptible}} = -0.096 \times \text{Time} + 1.27$; 95% CI = −0.154 to −0.038; Figure 6).

DISCUSSION

Our study showed a higher whirling disease risk for gently sloping alluvial valleys and spring creeks in the Blackfoot River basin. For basin-fed tributaries, the landscape variables of valley slope and forest cover were the overall best indicators of fine sediment (which is *T. tubifex* potential habitat), and valley slope was the best predictor of the presence of whirling disease. Streams with higher valley slopes had significantly lower levels of fine sediment and were categorized as disease negative (Figure 3). Spring creeks were not only predisposed to whirling disease, but the disease appeared to be influencing species composition—potentially constraining possible restoration outcomes—in these systems. Given the association of whirling disease risk with stream characteristics and the potential impacts on community composition in spring creeks, managers may want to consider whirling disease risk when prioritizing restoration sites across the landscape or setting restoration goals in spring creeks.

Predicting Risk (Fine Sediment and Severity of Infection)

Our analysis of landscape variables found high fine sediment and high infection severity in broad alluvial valleys with gentle, down-valley gradients (e.g., valley type VIII in Rosgen 1996). Here, alluvial floodplains are the most predominant landforms, which typically produce a high fine-sediment supply. Soils are developed over alluvium; thus, meandering streams in alluvial valleys are susceptible to naturally high levels of bank erosion and fine-sediment input. In the upper Blackfoot River basin, broad stream valleys are often utilized for intensive grazing and other land uses (e.g., farming, timber harvest, road construction) that commonly increase instream sediment levels and elevate water temperatures. By contrast, the steeper streams of the lower Blackfoot River basin support lower in-channel sediment levels, lower stream temperatures (R. Pierce, unpublished data), and thus a lower risk of whirling disease.

Similar to Anlauf and Moffitt (2010), our analyses of both landscape-scale and reach-scale features found that natural geomorphic variables and anthropogenic impacts can influence the proportion of fine sediment at a site. Anlauf and Moffitt (2010) found that the amount of slow habitat (pools, backwaters) versus fast habitat (riffles, runs) and riparian land cover type (conifer cover or agriculture) predicted differences in fine sediment at the reach scale. Most of the variation in our data was explained by geomorphology (valley slope), but anthropogenic degradation can create and enhance *T. tubifex* habitat (Waters 1995; Zendt and Bergersen 2000; McGinnis and Kerans 2013), playing a larger role in substrate composition and whirling disease than illustrated by our study. All sites in our data set are impacted to some degree by forest management practices, grazing, or agriculture. However, with the exception of Elk Creek, the riparian areas of the streams in this data set were not severely impacted by heavy grazing. Elk Creek, a disease-positive stream, had 2.5 times more fine sediment than any other site, the most gentle valley slope, the highest sinuosity, the most animal damage, and the lowest stream bank stability. That said, previous literature and our study demonstrated that some streams may be naturally at higher risk of disease because of their geomorphology. The relative role of natural versus anthropogenic drivers of whirling disease is context dependent. Even though we have focused on physical factors, certainly other factors, such as oligochaete community composition (Nehring et al. 2013, 2014), could further explain the variation in whirling disease infection severity among sites with gentle valley slopes.

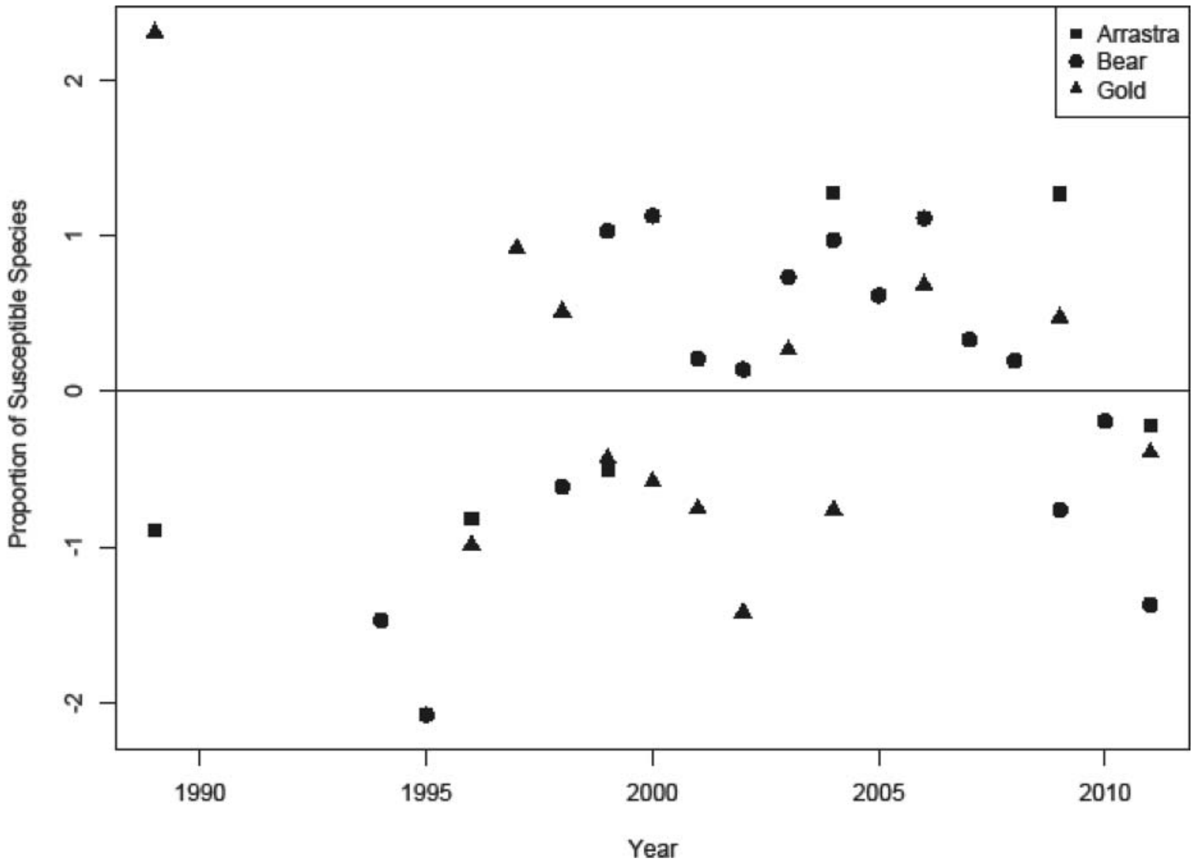

FIGURE 4. Plot of the *z*-transformed percent of the community made up of species susceptible to whirling disease for each year in disease-negative streams (Arrastra, Bear, and Gold creeks). The slope of the trend line was not significant in these basin-fed disease-negative streams ($Z_{susceptible} = 0.014 \times$ Time – 0.19; 95% CI = −0.04 to +0.07).

Though high instream sediment may predispose certain basin-fed streams to whirling disease, water temperatures between 10°C and 15°C may likewise facilitate whirling disease infection by promoting the production of TAMs (El-Matbouli et al. 1999; De la Hoz and Budy 2004; Kerans et al. 2005). Hansen and Budy (2011) showed a short-term reduction in the prevalence of *M. cerebralis* infection in a small stream in a northern Utah watershed where passive restoration (via grazing exclusion) reduced summer stream temperatures below 10°C. This suggests that the potential for restoration to reduce whirling disease risk may be possible when linked with substrate and temperature.

In the Blackfoot River basin, we have not seen indications that the stream habitat restoration efforts reduced the average histological scores for whirling disease. Several basin-fed tributaries with high severity of infection (Belmont, Cottonwood, Chamberlain, Elk, and Monture creeks) have undergone substantial habitat restoration efforts, including instream channel restoration, riparian vegetation improvement, increased stream flows, and removal of streamside feedlots, during this time period. These restoration actions were designed to improve fish habitat and reestablish movement corridors for migratory native trout, which typically spawn and rear upstream of the sentinel cage sites, and were not designed to reduce whirling disease prevalence or severity. The warmer and less variable seasonal temperature profiles paired with the higher sediment loads in spring creeks influence whirling disease dynamics and result in a higher risk compared with basin-fed streams (Kerans et al. 2005; Neudecker et al. 2012; Pierce et al. 2014a). Kleinschmidt Creek had extensive restoration (channel reconstruction and grazing exclusion), which resulted in a decrease in daily average summer stream temperature from 11.2°C to 10.0°C; however, there were no reductions in the severity of *M. cerebralis* infection at the reach. Infection severity remained high (≥3; Pierce et al. 2014a), thus the natural characteristics of spring creeks may make them more susceptible to whirling disease regardless of typical habitat restoration efforts.

Restoration Outcomes (Community Composition)

For basin-fed streams, there were no apparent changes in community composition (susceptible versus disease-resistant species) before and after the whirling disease epizootic (Figures 4, 5). Larger river sections of western Montana, including the main-stem Blackfoot River, have documented

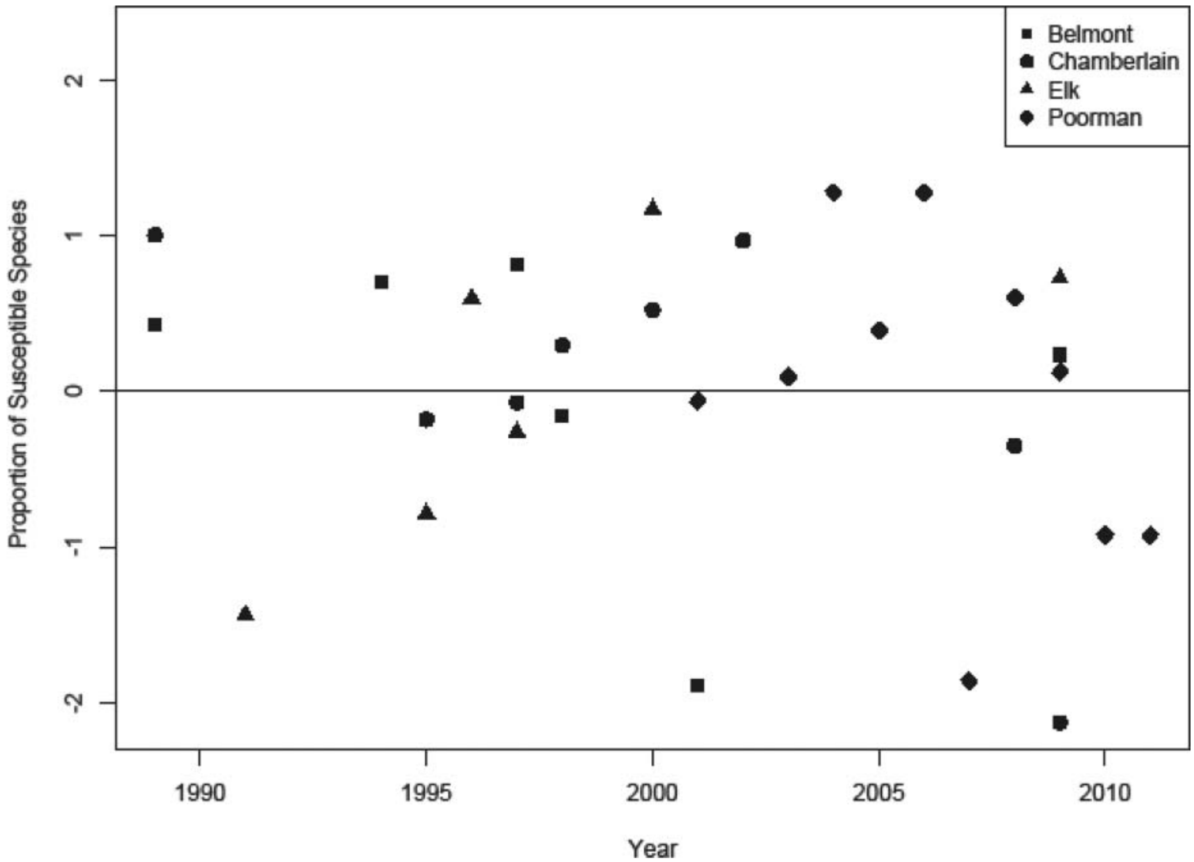

FIGURE 5. Plot of the z-transformed percent of the community made up of species that are susceptible to whirling disease for each year in disease-positive streams (Belmont, Chamberlain, Elk, and Poorman creeks). The slope of the trend line was not significant in these basin-fed disease-positive streams ($Z_{\text{susceptible}} = -0.025 \times \text{Time} + 0.32$; 95% CI = -0.08 to $+0.03$).

that susceptible species (specifically juvenile Rainbow Trout) declined in the presence of whirling disease (Vincent 1996; Granath et al. 2007; McMahon et al. 2010). The effects of whirling disease may be most apparent in river communities because of their high susceptibility and high levels of exposure, whereas the tributary assemblages may be buffered by contributions from upstream spawning areas where exposure may be lower (Pierce, unpublished data).

Similar to the findings in our two spring creeks in this study (Figure 6, Supplementary Figure S.1 found in the online version of this article), the shift to a community dominated by Brown Trout that was associated with exposure to whirling disease has been observed in other studies. In Kleinschmidt Creek, Brown Trout abundance increased in the presence of whirling disease following full channel restoration in 2001 (Pierce et al. 2015), supporting these study results. Similarly in Rock Creek near Missoula, Montana (a tributary to the Clark Fork River and a different Rock Creek than in this data set), the trout community also shifted dramatically from about 90% Rainbow Trout prior to whirling disease to primarily Brown Trout following the whirling disease epizootic (McMahon et al. 2010).

Restoration Prioritization

If managers are prioritizing restoration to support and augment susceptible salmonid populations in the presence of whirling disease, then physical features of the broader landscape, as well as life histories of target salmonids, should be considered. Within the heterogeneity of the Blackfoot River basin, salmonid distributions vary with longitudinal gradients. For example, Brown Trout and Rainbow Trout occupy the Blackfoot River and lower tributary system where *M. cerebralis* is present (Pierce et al. 2009, 2014b). Westslope Cutthroat Trout are prevalent across tributaries of the Blackfoot River basin from the headwaters to the rivers, with migratory life histories connecting these habitats (Schmetterling 2001; Pierce et al. 2014b). McMahon et al. (2010) found evidence of Rainbow Trout declines in the Blackfoot River but no indication of disease-related Brown Trout increases (McMahon et al. 2010). Additionally, long-term monitoring in the main-stem Blackfoot River (1989–2014) has shown a positive trend in Westslope Cutthroat Trout abundance and an increasing proportion of Westslope Cutthroat Trout within the trout community of the Blackfoot River (Pierce and Podner 2013).

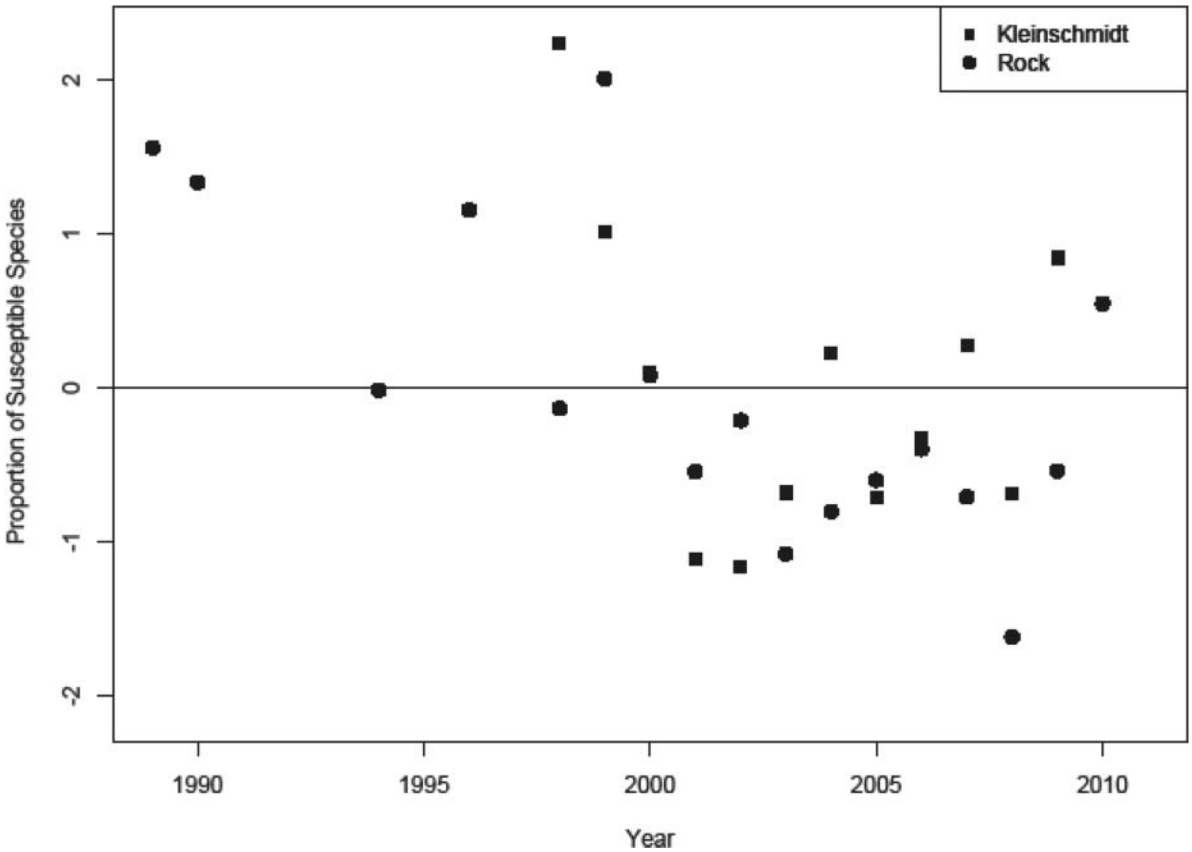

FIGURE 6. Plot of the *z*-transformed percent of the community made up of species susceptible to whirling disease for each year in disease-positive spring creeks (Kleinschmidt and Rock creeks). For these two spring creeks, we detected a negative trend over time in susceptible species ($Z_{\text{susceptible}} = -0.096 \times \text{Time} + 1.27$; 95% CI = −0.154 to −0.038).

Certainly, habitat connectivity between disease-free headwaters and disease-prone streams on the valley floor could help maintain susceptible trout throughout the basin. Although our spring creek sample size was especially small, our results suggest that spring creeks may be ecological sinks for susceptible species and may promote Brown Trout on the landscape. This may present a special challenge for decision makers given the potential of nonnative Brown Trout to increase predation or competition with more susceptible species of fisheries and conservation value (e.g., Rainbow Trout or Westslope Cutthroat Trout; McHugh et al. 2008). As novel parasites and diseases move across the landscape, the consideration of disease in prioritizing restoration plans is likely to become more critical.

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REFERENCES

Aitken, G. 1997. Restoration of trout waters in the West: Blackfoot River of Montana. Pages 402–424 *in* J. E. Williams, C. A. Wood, and M. P. Dombeck, editors. Watershed restoration: principles and practices. American Fisheries Society, Bethesda, Maryland.

Allan, D., D. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37:149–161.

Allen, M. B., and E. P. Bergersen. 2002. Factors influencing the distribution of *Myxobolus cerebralis*, the causative agent of whirling disease, in the Cache la Poudre River, Colorado. Diseases of Aquatic Organisms 49:51–60.

Alt, D., and D. W. Hyndman. 1986. Roadside geology of Montana. Mountain Press Publishing, Missoula, Montana.

Anlauf, K. J., and C. M. Moffitt. 2008. Models of stream habitat characteristics associated with tubificid populations in an intermountain watershed. Hydrobiologia 603:147–158.

Anlauf, K. J., and C. M. Moffitt. 2010. Modelling of landscape variables at multiple extents to predict fine sediments and suitable habitat for *Tubifex tubifex* in a stream system. Freshwater Biology 55:794–805.

Arndt, R. E., E. J. Wagner, Q. Cannon, and M. Smith. 2002. Triactinomyxon production as related to rearing substrate and diel light cycle. Pages 87–91 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Baerwald, M. R., A. B. Welsh, R. P. Hedrick, and B. May. 2008. Discovery of genes implicated in whirling disease infection and resistance in Rainbow

Trout using genome-wide expression profiling. BMC Genomics [online serial] 9:37.

Baldwin, T. J., E. R. Vincent, R. M. Silflow, and D. Stanek. 2000. *Myxobolus cerebralis* infection in Rainbow Trout (*Oncorhynchus mykiss*) and Brown Trout (*Salmo trutta*) exposed under natural stream conditions. Journal of Veterinary Diagnostic Investigations 12:312–321.

Bartholomew, J. L., and P. W. Reno. 2002. The history and dissemination of whirling disease. Pages 3–24 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Bartholomew, J. L., and L. C. Wilson, editors. 2002. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Beauchamp, K. A., G. O. Kelley, R. B. Nehring, and R. P. Hedrick. 2005. The severity of whirling disease among wild trout corresponds to the differences in genetic composition of *Tubifex tubifex* populations in central Colorado. Journal of Parasitology 91:53–60.

Beechie, T., G. Pess, P. Roni, and G. Giannico. 2008. Setting river restoration priorities: a review of approaches and a general protocol for identifying and prioritizing actions. North American Journal of Fisheries Management 28:891–905.

Breiman, L., J. H. Friedman, R. A. Olshen, and C. J. Stone. 1984. Classification and regression trees. Chapman and Hall, New York.

De la Hoz, E., and P. Budy. 2004. Linking environmental heterogeneity to the distribution of prevalence of *Myxobolus cerebralis*: a comparison across sites in a northern Utah watershed. Transactions of the American Fisheries Society 133:1176–1189.

Downing, D. C., T. E. McMahon, B. L. Kerans, and E. R. Vincent. 2002. Relation of spawning and rearing life history of Rainbow Trout and susceptibility to *Myxobolus cerebralis* infection in the Madison River, Montana. Journal of Aquatic Animal Health 14:191–203.

El-Matbouli, M., T. S. McDowell, D. B. Antonio, K. B. Andree, and R. P. Hedrick. 1999. Effect of water temperature on the development, release, and survival of triactinomyxon stage of *Myxobolus cerebralis* in its oligochaete host. International Journal for Parasitology 29:627–641.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199–214.

Granath, W. O. Jr., M. A. Gilbert, E. J. Wyatt-Pescador, and E. R. Vincent. 2007. Epizootiology of *Myxobolus cerebralis*, the causative agent of salmonid whirling disease in the Rock Creek drainage of west-central Montana. Journal of Parasitology 93:104–119.

Granath, W. O. Jr., and E. R. Vincent. 2010. Epizootiology of *Myxobolus cerebralis*, the causative agent of salmonid whirling disease in the Rock Creek drainage of west-central Montana: 2004–2008. Journal of Parasitology 96:252–257.

Hallett, S. L., H. V. Lorz, S. D. Atkinson, C. Rasmussen, L. Xue, and J. L. Bartholomew. 2009. Propagation of the myxozoan parasite *Myxobolus cerebralis* by different geographic and genetic populations of *Tubifex tubifex*: an Oregon perspective. Journal of Invertebrate Pathology 102:57–68.

Hansen, E. S., and P. Budy. 2011. The potential of passive stream restoration to improve stream habitat and minimize the impact of fish disease: a short-term assessment. Journal of the North American Benthological Society 30:573–588.

Hedrick, R. P., T. S. McDowell, K. Mukkatira, M. P. Georgiadis, and E. MacConnell. 1999. Susceptibility of selected inland salmonids to experimentally induced infections with *Myxobolus cerebralis*, the causative agent of whirling disease. Journal of Aquatic Animal Health 11:330–339.

Hiner, M., and C. M. Moffitt. 2001. Variation in *Myxobolus cerebralis* infections in field-exposed Cutthroat and Rainbow trout in Idaho. Journal of Aquatic Animal Health 13:124–132.

Hoffman, G. L. 1990. *Myxobolus cerebralis*, a worldwide cause of salmonid whirling disease. Journal of Aquatic Animal Health 2:30–37.

Kaeser, A. J., and W. E. Sharpe. 2006. Patterns of distribution and abundance of *Tubifex tubifex* and other aquatic oligochaetes in *Myxobolus cerebralis* enzootic areas in Pennsylvania. Journal of Aquatic Animal Health 18:64–78.

Karr, J. R., S. B. Adams, and L. S. Fore. 2005. Recommendations for data management, synthesis of knowledge, and future research under the Whirling Disease Initiative. Montana Water Center, Final Report to Whirling Disease Intiative Advisory Panel, Bozeman.

Kerans, B. L., R. I. Stevens, and J. C. Lemmon. 2005. Water temperature after a host-parasite interaction: *Tubifex tubifex* and *Myxobolus cerebralis*. Journal of Aquatic Animal Health 17:216–221.

Kerans, B. L., and A. V. Zale. 2002. The ecology of *Myxobolus cerebralis*. Pages 145–166 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Lazim, M. N., and M. A. Learner. 1987. The influence of sediment composition and leaf litter on the distribution of tubificid worms (Oligochaeta). Oecologia 72:131–136.

MacConnell, E., and E. R. Vincent. 2002. Review: the effects of *Myxobolus cerebralis* on the salmonid host. Pages 95–108 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

McGinnis, S., and B. L. Kerans. 2013. Land use and host community characteristics as predictors of disease risk. Landscape Ecology 28:29–44.

McHugh, P., P. Budy, G. Thiede, and E. VanDyke. 2008. Trophic relationships of nonnative Brown Trout, *Salmo trutta*, and native Bonneville Cutthroat Trout, *Oncorhynchus clarkii utah*, in a northern Utah, USA river. Environmental Biology of Fishes 81:63–75.

McMahon, T., J. Robison-Cox, J. Rotella, T. Horton, and B. Kerans. 2010. Trout population responses to whirling disease epizootics in Montana rivers. Pages 195–202 *in* R. F. Carline and C. LoSapio, editors. Wild Trout X: conserving wild trout. Wild Trout Symposium, Bozeman, Montana.

McNeil, W. J., and W. H. Ahnell. 1964. Success of Pink Salmon spawning relative to size of spawning bed material. U.S. Fish and Wildlife Service Special Scientific Report Fisheries 469.

Miller, M. P., and E. R. Vincent. 2008. Rapid natural selection for resistance to an introduced parasite of Rainbow Trout. Evolutionary Applications 1:336–341.

Montana DNRC (Department of Natural Resources and Conservation). 1984. River mile index for the Columbia River basin. Montana DNRC, Water Resources Division, Helena.

Nehring, R. B. 2006. Colorado's cold water fisheries: whirling disease case histories and insights for risk management. Colorado Division of Wildlife Special Technical Report 79.

Nehring, R. B., B. Hancock, M. Catanese, M. E. T. Stinson, and D. Winkelman. 2013. Reduced *Myxobolus cerebralis* actinospore production in a Colorado reservoir may be linked to changes in *Tubifex tubifex* population structure. Journal of Aquatic Animal Health 25:205–220.

Nehring, R. B., P. M. Lukacs, D. V. Baxa, M. E. T. Stinson, L. Chiaramonte, S. K. Wise, and A. Horton. 2014. Susceptibility to *Myxobolus cerebralis* among *Tubifex tubifex* populations from ten major drainage basins in Colorado where Cutthroat Trout are endemic. Journal of Aquatic Animal Health 26:19–32.

Nehring, R. B., and P. G. Walker. 1996. Whirling disease in the wild: the new reality in the intermountain west. Fisheries 21(6):28–30.

Neudecker, R. A., T. E. McMahon, and E. R. Vincent. 2012. Spatial and temporal variation of whirling disease risk in Montana spring creeks and rivers. Journal of Aquatic Animal Health 24:201–212.

Pierce, R., M. Davidson, and C. Podner. 2012. Spawning behavior of Mountain Whitefish and co-occurrence of *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 141:720–730.

Pierce, R., and C. Podner. 2013. Fisheries investigations in the Big Blackfoot River basin, 2011–2012. Montana Fish, Wildlife, and Parks, Missoula.

Pierce, R., C. Podner, and K. Carim. 2013. Response of wild trout to stream restoration over two decades in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 142:68–81.

Pierce, R., C. Podner, M. Davidson, L. Knotek, and J. Thabes. 2008. Big Blackfoot River fisheries and restoration investigations for 2006 and 2007. Montana Fish Wildlife and Parks, Missoula, Montana.

Pierce, R., C. Podner, M. Davidson, and E. R. Vincent. 2009. Correlation of fluvial Rainbow Trout spawning life history with severity of infection by *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 138:251–263.

Pierce, R., C. Podner, and L. Jones. 2015. Long-term increases in trout abundance following channel reconstruction, instream wood placement, and livestock removal from a spring creek in the Blackfoot Basin, Montana. Transactions of the American Fisheries Society 144:184–195.

Pierce, R., C. Podner, L Marczak, and L. Jones. 2014a. Instream habitat restoration and stream temperature reduction in a whirling disease positive spring creek in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 143:1188–1198.

Piece, R., C. Podner, T. Wendt, R. Shields, and K. Carim. 2014b. Westslope Cutthroat Trout movements through restored habitat in the Nevada spring creek complex, Blackfoot Basin, Montana. Transactions of the American Fisheries Society 143:230–239.

Poff, N. L., and J. V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46:1805–1818.

R Development Core Team. 2012. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

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

Rosgen, D. 1996. Applied fluvial geomorphology. Wildlands Hydrology, Pagosa Springs, Colorado.

Ryce, E. K. N., A. V. Zale, and E. MacConnell. 2004. Effects of fish age and parasite dose on the development if whirling disease in Rainbow Trout. Diseases of Aquatic Organisms 59:225–233.

Ryce, E. K. N., A. V. Zale, E. MacConnell, and M. Nelson. 2005. Effects of fish age versus size on the development of whirling disease in Rainbow Trout. Diseases of Aquatic Organisms 63:69–76.

Sauter, G., and H. Gude. 1996. Influence of grain size on the distribution of tubificid oligochaete species. Hydrobiologia 333:97–101.

Schisler, G. J., K. A. Myklebust, and R. P. Hedrick. 2006. Inheritance of *Myxobolus cerebralis* resistance among F1-generation crosses of whirling disease resistant and susceptible Rainbow Trout strains. Journal of Aquatic Animal Health 18:109–115.

Schmetterling, D. A. 2001. Seasonal movements of fluvial Westslope Cutthroat Trout in the Blackfoot River drainage, Montana. North American Journal of Fisheries Management 21:507–520.

Shepard, B. B., and P. J. Graham. 1982. Monitoring spawning gravel bed material used by Bull Trout on the glacier view district Flathead National Forest. Montana Fish Wildlife and Parks, Completion Report, Kalispell.

Shepard, B., S. Leathe, T. Weaver, and M. Enk. 1984. Monitoring levels of fine sediments within tributaries to Flathead Lake and impacts of fine sediment to Bull Trout recruitment. Pages 146–156 *in* F. Richardson and R. H. Hamre, editors. Wild trout III: proceedings of the symposium. Wild Trout Symposium, Bozeman, Montana.

Shepard, B. B., B. E. May, and W. Urie. 2005. Status and conservation of Westslope Cutthroat Trout within the western United States. North American Journal of Fisheries Management 25:1426–1440.

Strahler, A. N. 1957. Quantitative analysis of watershed geomorphology. Transactions of the American Geophysical Union 38:913–920.

Stevenson, N. J., and E. K. Mills. 1999. Streambank and shoreline condition. Pages 115–124 *in* M. B. Bain, and N. K. Stevenson, editors. Aquatic habitat assessment: common methods. American Fisheries Society, Bethesda, Maryland.

Sudduth, E. B., J. L. Meyer, and E. S. Bernhardt. 2007. Stream restoration practices in the southeastern United States. Restoration Ecology 15:573–583.

Swanberg, T. R. 1997. Movements of and habitat use by fluvial Bull Trout in the Blackfoot River, Montana. Transactions of the American Fisheries Society 126:735–746.

USFWS (U.S. Fish and Wildlife Service). 2010. Endangered and threatened wildlife and plants; revised designation of critical habitat for Bull Trout in the contiminous United States; final rule. Federal Register 75:200(18 October 2010):63898–64070.

Venables, W. N., and B. D. Ripley. 1997. Modern applied statistics with S-PLUS, 2nd edition. Springer, New York.

Vincent, E. R. 1996. Whirling disease and wild trout: the Montana experience. Fisheries 21(6):32–33.

Vincent, E. R. 2000. Whirling disease report 1997–98. Montana Fish Wildlife and Parks, Project 3860, Helena.

Vincent, E. R. 2002. Relative susceptibility of various salmonids to whirling disease with emphasis on Rainbow and Cutthroat trout. Pages 109–115 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Waters, T. F. 1995. Sediment in streams: sources, biological effects and controls. American Fisheries Society, Monograph 7, Bethesda, Maryland.

Zendt, J. S., and E. P. Bergersen. 2000. Distribution and abundance of the aquatic oligochaete host *Tubifex tubifex* for the salmonid whirling disease parasite *Myxobolus cerebralis* in the upper Colorado River basin. North American Journal of Fisheries Management 20:502–512.

GENETIC ASSIGNMENT OF BULL TROUT IN CLEARWATER BASIN LAKES AND BLACKFOOT RIVER TO NATAL TRIBUTARIES

W.L. Knotek, W. Schreck, R. Pierce, and C. Podner Montana Fish, Wildlife & Parks
R. Leary, S. Amish, S. Painter, and A. Lodmell University of Montana Conservation Genetics Laboratory
Shane Hendrickson, Lolo National Forest

Introduction

Migratory bull trout (*Salvelinus confluentus*) occupy interconnected rivers and lakes along with several headwater tributaries of the Blackfoot River Basin. Within the basin, bull trout are generally present in low abundance because of long-term population declines and the limited number of spawning and rearing tributaries currently occupied. Determining the tributary of origin for individuals captured in the Blackfoot River and Clearwater Lakes is useful for (1) evaluating habitat use and the spatial scale of bull trout movements, (2) monitoring the relative levels of recruitment from various tributary populations, and (3) for identifying bull trout recovery actions associated with movement corridors.

Genetic assignment can trace individual bull trout to their natal tributary population based on the probability of occurrence for selected genetic markers. *Assignments* are based on comparisons of diagnostic alleles from an individual fish's genome with the presence and frequency of these alleles in the tributary populations where the fish may have originated. If genetic markers (allele combinations) of plausible tributary populations are distinct, there is a high probability that an individual from one population can be accurately matched or assigned to the population where it originated. If tributary populations are not genetically distinct (typically due to more frequent exchange of spawners), the 'confidence' or probability of correct assignment is decreased.

Using this genetic assignment application, we collected genetic samples from juvenile bull trout in all Blackfoot River Basin tributary streams where viable spawning population spawning had been identified. These include adfluvial stocks in the Clearwater River Basin tributaries (Morrell Creek, Deer Creek, Marshall Creek, East Fork Clearwater River and West Fork Clearwater River), fluvial stocks outside of the Clearwater River drainage (Copper Creek and its tributary Snowbank Creek, the North Fork Blackfoot River, Monture Creek and its tributary Dunham Creek), as well as two streams supporting small populations of resident bull trout (Cottonwood Creek and Poorman Creek) (Figure 1). After the genetic composition of these samples was analyzed to establish a genetic baseline for each tributary population, genetic assignment models were constructed based on observed allele frequencies. We then collected additional samples from bull trout captured at three sites in the mainstem Blackfoot River and four lakes on the mainstem Clearwater River and attempted to assign fluvial and adfluvial fish to their tributary of origin.

Study Area and Methods

Collection of Tributary Baseline Samples - Juvenile bull trout were collected using a backpack electrofishing unit in 12 tributaries where core populations had previously been identified (Figure 1, Table 1). Whenever possible, genetic samples (fin clips) were taken from juvenile trout at multiple locations within upper reaches of rearing habitats, including nearby connected tributaries. Target "baseline" sample sizes were 30 bull trout from each tributary population. However, some streams (Poorman and Cottonwood Creeks) with small populations produced <30 individuals. In addition, since bull trout x brook trout (*Selvelinus fontinalis*) hybrids (n=3) were present in certain samples (Table 1), which were removed from the analyses once identified. All fin clips collected in the field (lakes, river and tributaries) were taken from either the caudal or anal fin, immediately preserved at streamside in 95% non-denatured ethanol. These samples were then submitted to the University of Montana Conservation Genetics Laboratory (Genetics Lab) for analysis.

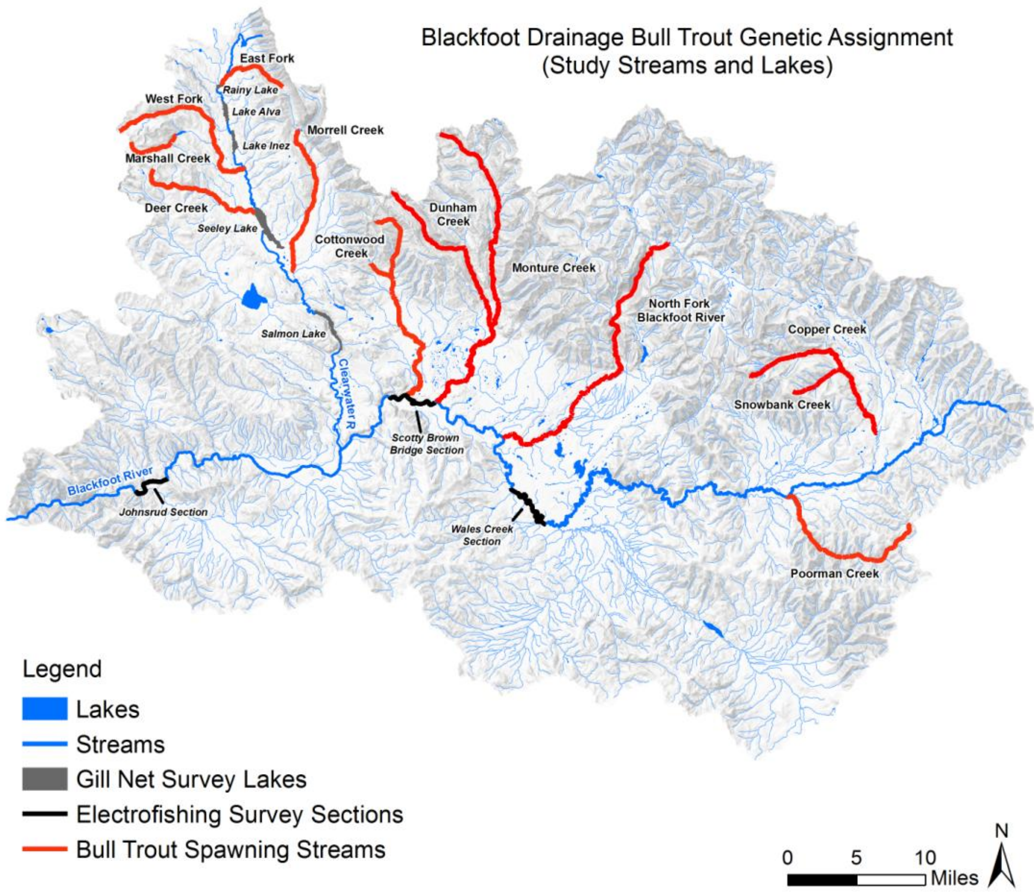

Figure 1. Location map: Tributaries (highlighted in red) in the Blackfoot Basin where bull trout populations were sampled for genetic assignment analyses. The map also shows three mainstem Blackfoot River locations where fluvial bull trout were collected, plus five Clearwater River Lakes where adfluvial bull trout were collected.

Collections from the Blackfoot River and Clearwater Lakes - In addition to juvenile bull trout in natal tributaries, non hybrid adult and sub-adult bull trout were sampled in the mainstem Blackfoot River at three long-term monitoring sites (Wales Creek (n=8), Scotty Brown Bridge (n=34) and Johnsrud (n=58)) using a drift boat electrofishing unit during the 2012 and 2014 population monitoring at these sites (Figure 1). Likewise, non hybrid subadult and adult bull trout were sampled in four main stem Clearwater lakes (Salmon Lake (n=10), Seeley Lake (n=33), Lake Inez (n=39), Marshal Lake (n=2), Lake Alva (n=24), Rainy Lake (n=14)) using floating and sinking experimental gill nets (2007-2015).

Similar to the tributaries, all bull trout x brook trout hybrids from the Blackfoot River (n=4) and Clearwater Lakes (n=3) were removed from the study once identified.

Laboratory Methods and Data Analysis - Bull trout genetic samples were processed and analyzed at the University of Montana Conservation Genetics Laboratory. Using the polymerase chain reaction (PCR), each fish's genotype was determined at 16 microsatellite loci. Of these, seven distinguish bull trout, *Salvelinus confluentus*, from brook trout, *Salvelinus fontinalis*. The latter loci are commonly termed diagnostic loci because the alleles (length of the DNA fragment copied during PCR) detected at them can be used to determine whether an individual was a non-hybridized bull or brook trout or was of hybrid origin between these fishes. A non-hybridized bull trout would possess only bull trout alleles (homozygous) at all the diagnostic loci. Likewise a non-hybridized brook trout would be homozygous for alleles characteristic of brook trout at all diagnostic loci. A first generation hybrid (F_1) would possess alleles characteristic of both bull and brook trout (heterozygous) at all diagnostic loci. Later generation hybrids would be homozygous at some diagnostic loci and heterozygous at the other diagnostic loci.

TRIBUTARY	BULL TROUT (n)	BULL x EBT HYBRIDS (n)
Clearwater River tributaries		
East Fork Clearwater R.	31	
West Fork Clearwater R.	30	
Marshall Creek	24	7
Deer Creek	28	2
Morrell Creek	33	
Blackfoot River tributaries		
Copper Creek	25	
Snowbank Creek	25	
Poorman Creek	27	
North Fork Blackfoot	49	
Monture Creek	37	
Dunham Creek	13	
Cottonwood Creek	13	1

Table 1. Bull trout (and hybrid bull trout x brook trout) sample sizes collected from Blackfoot River Basin tributaries for genetic assignment baseline. Hybrid individuals were removed from the analysis.

Considering just the tributaries, after removing hybrids from the data, we used the log likelihood G test of Goudet et al. (1996) in GENEPOP version 4.0 (Rousset 2008) to determine if there was evidence of allele frequency differences among bull trout collected from the same stream at different locations, between streams from the same drainage, and among the tributary samples. Since multiple comparisons were performed at all levels of analysis, we accounted for the possibility that a significant difference may simply represent a chance departure from homogeneity using Rice's (1989) correction for multiple comparisons (modified level of significance). When significant differences existed at the modified level between samples at one or more loci, we interpreted this to indicate that significant genetic differences existed between them and they were generally kept separate for subsequent analysis.

We estimated the amount of genetic divergence among bull trout from the various tributary samples using the proportion of the total genetic variation detected between two samples due to allele frequency differences between them (F_{ST}) using the procedure of Weir and Cockerham (1984) in GENEPOP version 4.0. In this and subsequent analyses, unless noted otherwise, we also included five previous samples from what are believed to be the major spawning tributaries for migratory bull trout in the Clearwater River drainage (Leary et al. 2012; West Fork Clearwater River #3485, Marshall Creek #4386, Deer Creek #4387, Morrell Creek #4388, and East Fork Clearwater River #4389). We also estimated the amount of genetic variation in the tributary samples using average expected heterozygosity (H_e) calculated in

GENALEX 6 (Peakall and Smouse 2006) and allelic richness (A_R) using the program HP-Rare of Kalinowski (2005). We also used two procedures to determine how well individuals could be placed to their sample of origin. First, we used the assignment test of Rannala and Mountain (1997) available in GENECLASS2 (Piry et al. 2004). We determined the likelihood of assignment to a tributary using an individual's assignment score. This is the highest probability of assignment to a tributary divided by the sum of the probability of assignment to all tributaries. Thus, if an individual was assigned only to a single tributary it would have an assignment score of 100 to that tributary. Values less than 100 would indicate that the individual had a probability of greater than zero of being assigned to two or more tributaries. The lower the score, the less one is sure of the correct assignment. Next, we used the program STRUCTURE (Pritchard et al. 2000, 2007) and set the number of groups (K) to 12 and 2. The former is equal to the number of tributary samples and we expected the latter to contrast the Blackfoot and Clearwater River drainages.

For the individuals collected from the Blackfoot River and lakes in the Clearwater River drainage, we attempted to determine their tributary of origin by treating them as unknowns and the tributary samples as knowns in the assignment test of Rannala and Mountain (1997). Again, we determined the likelihood of assignment to a tributary using an individual's assignment score. In the analyses, we included individuals that had previously been collected from Lake Inez and Lake Alva.

Results and Discussion

F_{ST}, Assignment, and STRUCTURE Results **-** Considering just the tributaries to the Blackfoot and Clearwater rivers, there tended to be more divergence between tributaries from the two drainages than between tributaries within the drainages indicating substantial divergence between bull trout in the different drainages (Table 2). With a few exceptions (e. g., Copper and Snowbank Creek in the Blackfoot River drainage and the West Fork Clearwater River, Marshall and Deer Creek in the Clearwater River drainage), there was also moderate to large levels of genetic divergence between tributaries within each drainage (Table 2). Overall, there appeared to be more divergence among the Clearwater (global F_{ST}=0.1583) than the Blackfoot (global F_{ST}=0.1091) tributaries. The results from the assignment test are highly concordant with the F_{ST} estimates. In general, individuals tended to assign back to the tributary from which they were collected with a score of greater than 99 and no individuals from the Blackfoot River drainage were mis-assigned to the Clearwater River drainage and vice versa, suggesting very limited, if any, gene flow between the drainages.

Although we detected evidence of spatial genetic differences among streams of the Clearwater drainage, these differences were generally smaller than those detected among samples from different streams (Figure 2). Thus, the results suggest that bull trout in the Clearwater River drainage broadly form three genetic groups: (1) West Fork Clearwater River/Marshall Creek/Deer Creek, (2) Morrell Creek, and (3) East Fork Clearwater River. At the stream level, the assignment test generally placed individuals back to the stream from which they were sampled greater than 85% of the time. The exception being the West Fork Clearwater River in which only 80% of the individuals sampled from this stream were assigned back to it.

The results obtained from the STRUCTURE analyses were fairly similar to those obtained from F_{ST} and the assignment test. With K=12, all the Blackfoot tributary samples except Copper and Snowbank Creek were identified as distinct groups (Figure 2). Interestingly, only Morrell Creek was identified as a distinct group in the Clearwater drainage. All the other samples were placed in the same group. This is probably, at least partially, a consequence of the greater amounts of genetic diversity detected within the Blackfoot than the Clearwater tributaries with the exception of Morrell Creek. With K=2, the analysis identified the Blackfoot and Clearwater tributaries as constituting distinct groups again indicating more divergence between tributaries from the different drainages than between those within each drainage.

	Sample and F_{ST}										
Sample	Copper	Snow	NF Cotton	NF Black	Monture	Dunham	Poorman	WF Clear	Marshall	Deer	Morrell
Snow	0.0109										
NF Cotton	0.1608	0.1644									
NF Black	0.0679	0.0651	0.1518								
Monture	0.0845	0.0892	0.1443	0.0722							
Dunham	0.1066	0.1114	0.1891	0.1056	0.0724						
Poorman	0.1101	0.1131	0.2453	0.1592	0.1465	0.1524					
WF Clear	0.2485	0.2448	0.2763	0.2363	0.2555	0.3139	0.3737				
Marshall	0.2460	0.2392	0.2772	0.2242	0.2481	0.3148	0.3695	0.0502			
Deer	0.2985	0.2999	0.3200	0.2709	0.2785	0.3481	0.4092	0.0813	0.1174		
Morrell	0.1694	0.1641	0.2088	0.1594	0.1629	0.2153	0.2738	0.1268	0.1160	0.1553	
EF Clear	0.3399	0.3419	0.3416	0.3151	0.3283	0.3961	0.4410	0.1898	0.2315	0.2483	0.2158

Table 2. Estimates of F_{ST} between samples of bull trout from the Blackfoot River tributaries (Copper Creek, Snowbank Creek, North Fork Cottonwood, North Fork Blackfoot River, Monture Creek, Dunham Creek and Poorman Creek) and Clearwater River tributaries (West Fork Clearwater River, Marshall Creek, Deer Creek, Morrell Creek and the East Fork Clear Water River).

Assignment of the Blackfoot River bull trout to tributaries - The majority of individuals from the Johnsrud and Scotty Brown Bridge sections of the Blackfoot River were assigned to the North Fork Blackfoot River with a score greater than 99 (Figure 3). Monture Creek accounted for the next highest proportion of the fish collected from the Johnsrud section followed and Snowbank Creek. These fish assigned to these creeks again with generally with a score greater than 99. A fairly substantial proportion of the fish collected from the Scotty Brown section also assigned to Monture Creek with a high degree of certainty (score greater than 99) (Figure 3).

The Wales section of the Blackfoot River mainly had individuals assigning to the Copper Creek drainage but, a fairly substantial proportion (0.375) were assigned to the North Fork Blackfoot River. The majority (0.750) of the fish were assigned to a tributary with a score greater than 99. The exceptions involved two fish that assigned to the Copper Creek drainage (score=94.014 and 85.608).

No bull trout collected from the Blackfoot River were assigned to having originated from the Cottonwood drainage, Dunham Creek and Poorman Creek. This suggests that these tributaries may largely contain resident bull trout. Of the drainages definitely appearing to contain fluvial bull trout, the North Fork Blackfoot River appears to be by far the most important in providing fish to the Blackfoot River (74.7%). Its importance, however, tends to decrease in the upriver direction (Johnsrud 87.7%, Scotty Brown Bridge 61.8%, Wales 37.5%). The Copper Creek drainage appears to be overall the least important (5.1%) and apparently contributes fish primarily to the upper reaches of the Blackfoot River. However one bull trout sampled in the Johnsrud Section assigned to Snowbank Creek, a distance of >100 river miles.

These results clearly highlight the importance of past screening of fish from diversions on Dunham Creek, the North Fork Blackfoot River, Snowbank Creek and the mainstem Blackfoot River. For the first time, this study also provides compelling evidence of bull trout movements from Monture Creek and the North Fork Blackfoot River into Salmon Lake (assignment score >99). This suggests improved large scale connectivity and the importance of improving habitat connectivity between the Blackfoot River and Salmon Lake.

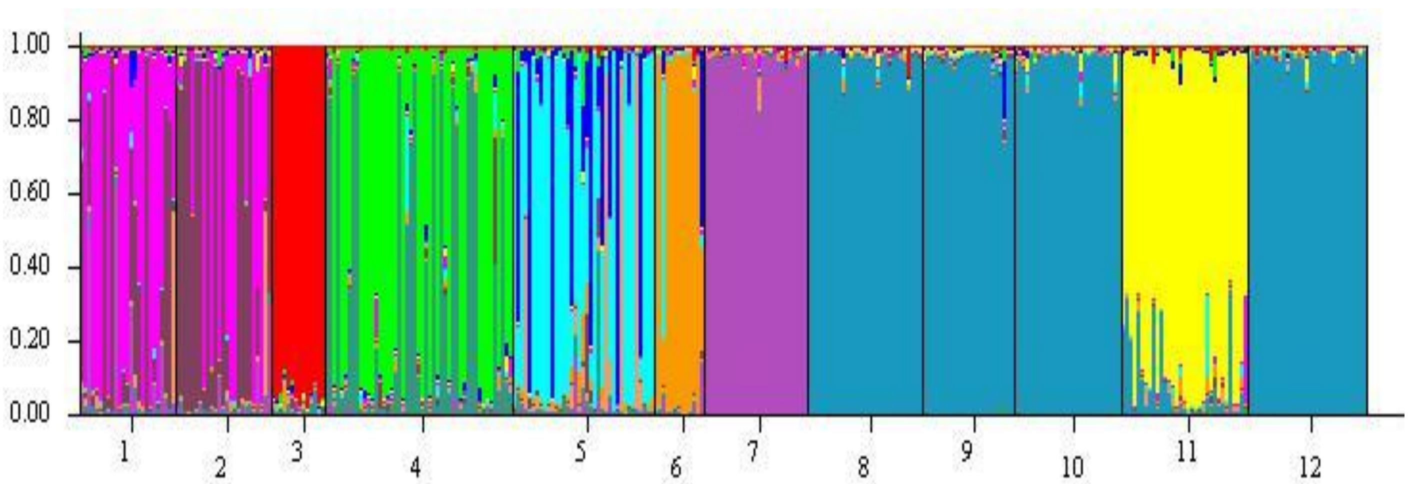

Figure 2. Results of STRUCTURE with K=12 using samples from tributaries to the Blackfoot River and Clearwater River. 1=Copper Creek. 2=Snowbank Creek. 3=North Fork Cottonwood Creek. 4=North Fork Blackfoot River. 5=Monture Creek. 6=Dunham Creek. 7=Poorman Creek. 8=West Fork Clearwater River. 9=Marshall Creek. 10=Deer Creek. 11=Morrell Creek. 12=East Fork Clearwater River

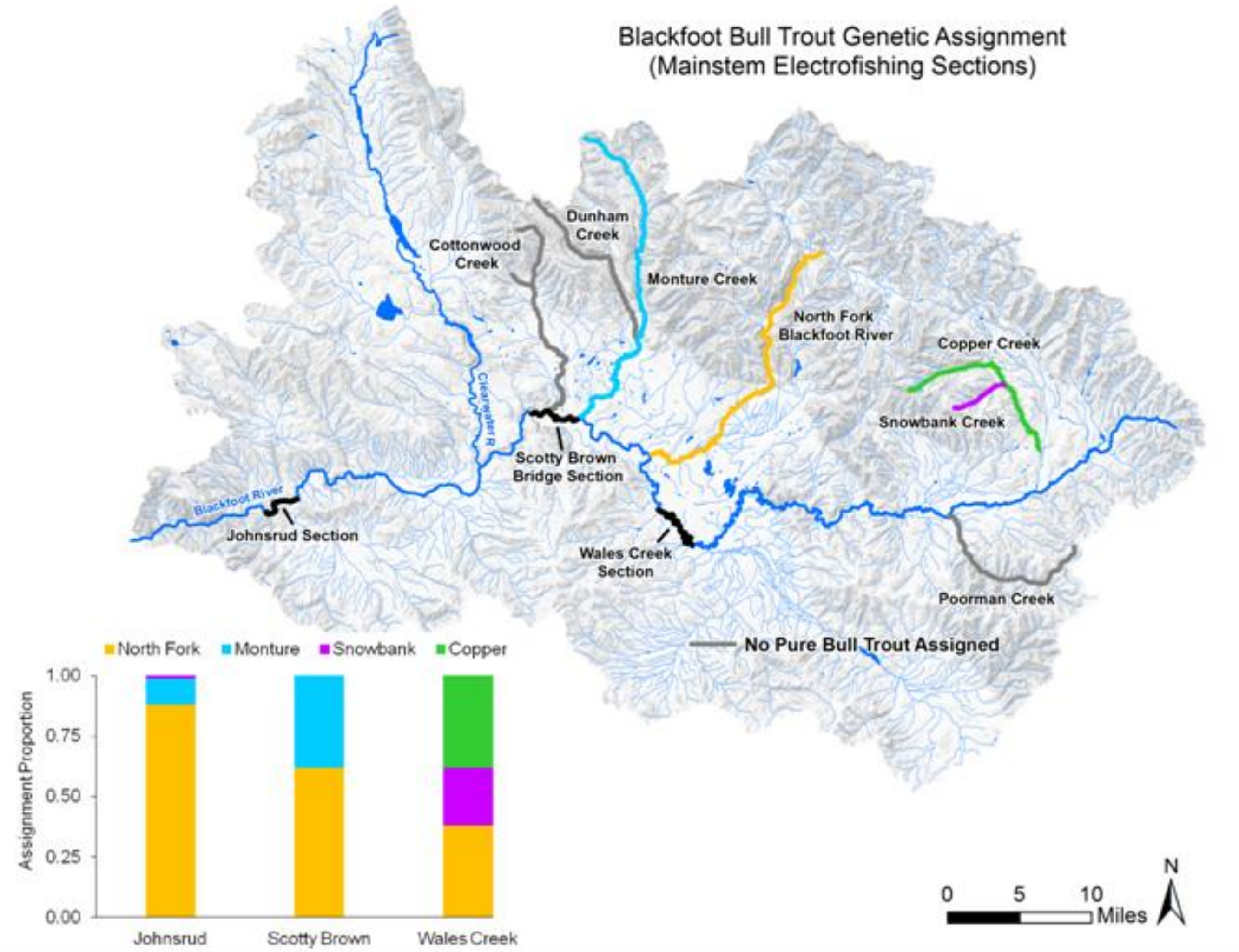

Figure 3. Map depicting genetic assignment results for adult and sub-adult bull trout sampled at three locations on the Blackfoot River. Colored shading on the Bars corresponds with the color of tributaries where individual bull trout were assigned and presumably originated.

Assignment of Individuals from Clearwater Lakes to Tributaries - Tributary assignments for 122 bull trout captured in the five Clearwater lakes are displayed in Table 3. Results highlight the importance of the West Fork Clearwater River and Morrell Creek as primary sources of recruitment for main stem lakes, with smaller contributions from other tributaries. The geographic distribution of assignments indicates that most fish in lakes originate in adjacent tributaries. However, results also confirm that certain individuals move considerable distances among lakes and river sections.

Bull trout originating in the West Fork Clearwater River were most common in the mainstem lake system and had the widest geographic distribution. Fish from this stream were detected in all lakes except Salmon Lake. In contrast, bull trout from the East Fork Clearwater River and Morrell Creek were also well represented but indicate a narrower distribution. The limited number and limited geographic distribution of bull trout from Deer Creek and Marshall Creek likely reflect the low abundance of these populations and lack of genetic divergence relative to neighboring streams.

CAPTURE LOCATION (n)	PROBABLE TRIBUTARY POPULATION OF ORIGIN					
	Morrell Cr.	Deer Cr.	Marshall Cr.	West Fork	East Fork	Other
Salmon Lake (10)	8	0	0	0	0	2*
Seeley Lake (33)	20	6	1	6	0	0
Lake Inez (39)	0	2	5	32	0	1
Marshall Lake (2)	0	0	1	1	0	0
Lake Alva (24)	0	1	0	14	9	0
Rainy Lake (14)	0	0	0	1	13	0

* Two bull trout captured in Salmon Lake assigned with > 99.9% probability of accuracy to other Blackfoot Basin tributaries: Monture Creek and North Fork Blackfoot

Table 3. Genetic assignment results for 122 bull trout captured in Clearwater Basin lakes and assigned to probable tributaries of origin.

Salmon Lake was somewhat unique. As expected, the majority of bull trout (80%) assigned to Morrell Creek, which enters the Clearwater River upstream of the lake. Mostly interestingly, two individuals from Salmon Lake were assigned to tributaries outside of the Clearwater drainage (Monture and North Fork Blackfoot River). A majority of fish sampled in Seeley Lake assigned to Morrell Creek, followed by the Marshall Creek, Deer Creek and Marshall Creek. Most bull trout in Lake Inez originated in the West Fork Clearwater River or Marshall Creek. Marshall Lake had a small samples size (n=2). Genetic assignment indicates these fish originated in the Upper West Fork Clearwater River. Interestingly, several bull trout captured in Lake Inez and Seeley Lake assigned to Marshall Creek, indicating that fish pass through or temporarily occupy Lake Marshall on their migrations to and from the mainstem lake and river system. Rainy Lake, located immediately downstream of the East Fork Clearwater River, supports genetically distinct bull trout, a majority of which assigned to the East Fork with one fish assigned to the West Fork Clearwater River. This fish provides evidence of upstream movement at Rainy dam (located downstream of Rainy Lake), which was retrofitted in 2011 to provide selective passage for adult salmonids, while precluding upstream movement of northern pike and other introduced fish not found in Rainy Lake.

Conclusions

Genetic assignment is a useful tool in identifying source populations for migratory bull trout, especially when tributary populations are genetically distinct. In this study, we were able to identify the tributary sources for bull trout at three sites on the Blackfoot River and in five lakes in the Clearwater Basin as well as the relative importance of the various tributaries for recruitment to these lake and river waterbodies. In addition, genetic assignment confirmed larger scale movements of bull trout among rivers, tributaries and interconnected lakes, as well as between major basins in the Blackfoot Watershed. From a restoration and recovery perspective, these results show the importance of habitat connectivity within corridors connecting natal streams and rivers and lakes in this study and need to continue to eliminate fish losses at unscreened irrigation ditches located within movement corridors.

Acknowledgments

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References

Goudet, J., M. Raymond, T. deMeeus, and F. Rousset. 1996. Testing differentiation in diploid populations. Genetics 144:1933-1940.

Kalinowski, S. T. 2005. HP-Rare: a computer program for performing rarefaction on measures of allelic diversity. Molecular Ecology Notes 5:187-189.

Nei, M. 1975. Molecular population genetics and evolution. Elsevier-North Holland, New York.

Paetkau, D., R. Slade, M. Burden, and A. Estoup. 2004. Genetic assignment methods for the direct, real-time estimation of migration rate: a simulation based exploration of accuracy and power. Molecular Ecology 13:55-65.

Peakall, R., and P. E. Smouse. 2006. GENALEX 6: genetic analysis in excel. Population genetic software for teaching and research. Molecular Ecology Notes 6:288-295.

Piry, S., A. Alapetite, J.-M. Cornuet, D. Paetkau, L. Baudouin, and A. Estoup. 2004. GENECLASS2: a software for genetic assignment and first-generation migrant detection. Journal of Heredity 95: 536-539.

Pritchard, J. K., M. Stephens, and P. Donnelly. 2000. Inference of population structure using multilocus genotype data. Genetics 155:945-959.

Pritchard, J. K., W. Wen, and D. Faulsh. 2007. Documentation for *structure* software: version 2.2. http://pritch.bsd.uchicago.edu/software.

Rannala, B., and J. L. Mountain. 1997. Detecting immigration using multilocus genotypes. Proceedings National Academy of Sciences 94:9197-9221.

Rice, W. R. 1989. Analyzing tables of statistical tests. Evolution 43:223-225.

Rousset, F. 2008. GENEPOP'007: a complete re-implementation of the GENEPOP software for Windows and Linux. Molecular Ecology Resources 8:103-106.

Weir, B. S., and C. C. Cockerham. 1984. Estimating F-statistics for the analysis of population structure. Evolution 38:1358-1370.

APPENDICES

Appendix A: Summary of catch and size statistics for Blackfoot tributaries, 2013-2015.

Appendix B: Summary of two-pass estimates for tributaries, 2013-2015.

Appendix C: Mark and recapture estimates, 2013-2015.

Appendix D: Summary of water chemistry readings, 2013-2015.

Appendix E: Summary of water temperatures, 2013-2015.

Appendix F: Genetic test results for westslope cutthroat trout, 2013-2015.

Appendix G: Restoration streams and tables of activities.

Appendix H: Potential restoration projects.

Appendix I: Table of restoration streams and cooperators.

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2013-2015.

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+
Ashby Creek	2	13N,16W,26A	5-Aug-14	300	CT	10	9	0	3.9 - 7.3	5.1	0.0	3.0
					EB	2	1	0	4.5 - 6.9	5.7	0.0	0.3
					Spotted frog	observed						
	2.7	13N,16W,26D	31-Jul-13	489	CT	32	29	0	3.9 - 9.1	6	0.0	5.9
			5-Aug-14	489	CT	65	49	1	1.9 - 9.1	5.6	0.2	9.8
					EB	6	4	0	3.9 - 7.7	5.4	0.0	0.8
					Spotted frog	common						
Bear Creek (lower river trib)	1.1	13N,16W,18B; 13N,16W,7C	4-Aug-14	393	CT	1	1	0	8.1	8.1	0.0	0.3
					RB	58	45	0	3.0 - 8.4	4.6	0.0	11.5
					LL	22	18	1	3.9 - 8.5	5.4	0.3	4.3
					EB	7	4	1	2.6 - 9.8	5.2	0.3	0.8
					Sculpins	abundant						
Beartrap Cr	0.2	15N,6W,27B	14-Aug-13	403	EB	1	1	0	6.6	6.6	0.0	0.2
Beaver Creek	0.2	14N,9W,22B	22-Jul-15	603	LL	39	39	2	2.4 -14.6	6.8	0.3	6.1
					Sculpins	abundant						
	1.4	14N,9W,15B	22-Jul-15	450	LL	17	17	0	4.5 - 14.3	8.8	0.0	3.8
					EB	3	3	0	5.8 - 8.2	7.0	0.0	0.7
					Sculpins	common						
	5.4	15N,9W,32A	22-Jul-15	360	CT	51	51	0	2.1 - 7.3	4.4	0.0	14.2
					EB	4	4	0	3.7 - 5.5	4.5	0.0	1.1
					Sculpins	common						
Belmont Creek	0.1	14N,16W,24C	16-Jul-15	471	RB	14	14	3	1.1 - 7.8	4.0	0.6	2.3
					LL	4	4	2	2.0 - 7.0	4.0	0.4	0.4
					MWF	5	5	5	3.0 - 3.2	3.0	1.1	0.0
					Sculpins	present						
	0.3	14N,16W,24C	14-Jul-15	342	RB	33	33	3	1.2 - 9.6	4.2	0.9	8.8
					LL	16	16	14	1.8 - 6.7	2.7	4.1	0.6
					Sculpins	present						
	0.6	14N,16W,24B	16-Jul-15	410	RB	36	36	0	3.0 - 10.8	4.8	0.0	8.8
					LL	8	8	4	1.9 - 8.9	4.5	1.0	1.0
					Sculpins	common						
	1.2	14N,16W,14D	15-Jul-15	312	RB	35	35	8	1.1 - 8.4	3.5	2.6	8.7
					LL	17	17	13	1.6 - 8.4	3.0	4.2	1.3
					Sculpins	present						
	1.5	14N,16W,14A	15-Jul-15	405	RB	26	26	2	1.3 - 8.9	4.3	0.5	5.9
					LL	3	3	0	4.1 - 6.2	4.8	0.0	0.7
					Sculpins	present						
	7.4	15N,16W,20A	15-Jul-15	372	DV	2	2	0	4.3 - 6.3	5.3	0.0	0.5
					CT	19	19	0	3.0 - 8.5	5.3	0.0	5.1

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2013-2015 (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+
Blackfoot River	130.5	15N,6W,20A	14-Aug-13	751	CT	1	1	0	6.3	6.3	0.0	0.1
(above Pass Cr)					EB	18	18	7	1.8 - 8.2	4.5	0.9	1.5
					Spotted frog	present						
(above Shave Cr)	131.8	15N,6W,21D	14-Aug-13	536	EB	12	12	6	2.2 - 7.5	4.0	1.1	1.1
					Spotted frogs	present						
Braziel Creek	0.2	12N,10W,10D	28-Aug-13	300	CT	79	64	3	1.6 - 6.3	4.2	1.0	20.3
			18-Aug-14	300	CT	56	49	41	1.6 - 8.2	2.8	13.7	2.7
					EB	2	0	0	4.1 - 4.2	4.1	0.0	0.0
					Sculpins	26	20	7	1.1 - 5.0	3.2	2.3	4.3
			9-Sep-15	300	CT	47	41	1	2.8 - 6.1	4.2	0.3	13.3
					Sculpins	common	Spotted frog	observed				
Chamberlain Creek	0.1	15N,13W,32A	9-Sep-14	358	CT	66	51	8	1.6 - 8.4	4.6	2.2	12.0
					LL	3	3	0	5.7 - 7.5	6.7	0.0	0.8
					Sculpins	common	Spotted frogs	present				
			1-Oct-15	358	CT	41	22	7	2.1 - 8.0	3.8	2.0	4.2
					LL	6	4	1	2.8 - 9.8	6.1	0.3	0.8
					Sculpins	common	RSS & LNS	common				
Cottonwood Creek	1	15N,13W,29B	27-Jul-15	546	CT	2	2	0	9.3 - 9.7	9.4	0.0	0.4
					LL	21	21	16	2.2 - 14	4.3	2.9	0.9
					EB	1	1	0	6.3	6.3	0.0	0.2
					ONC	5	5	5	1.3 - 1.9	1.7	0.9	0.0
					Sculpins	abundant	MWF	present				
	3.3	15N,13W,17D	27-Jul-15	473	LL	34	34	11	2.1 - 16.5	4.8	2.3	4.9
					EB	5	5	2	2.8 - 7.1	5.1	0.4	0.6
					Sculpins	abundant						
	4.7	15N,13W,8D	6-Aug-15	416	LL	22	22	8	2.2 - 12.2	5.3	1.9	3.4
					EB	6	6	0	5.4 - 9.1	6.6	0.0	1.4
					Sculpins	common						
	7.5	15N,13W,5C	28-Jul-15	480	LL	16	16	2	3.7 - 11.8	7.6	0.4	2.9
					EB	89	89	25	1.9 - 8	5.1	5.2	13.3
					Sculpins	common						
	12.0	16N,14W,24D	24-Sep-14	515	CT	32	25	2	1.6 - 10.6	5.1	0.4	4.5
					EB	4	3	0	4.1 - 6.4	5.7	0.0	0.6
					EB x DV	3	2	0	6.7 - 7.5	7.1	0.0	0.4
					Sculpins	common						
			30-Sep-15	515	CT	57	43	0	3.3 - 9.5	5.9	0.0	8.3
					EB	12	8	2	2.6 - 8.5	5.2	0.4	1.2
					EB x DV	4	3	0	8.3 - 8.6	8.4	0.0	0.6
					Sculpins	common						
Bull trout genetics assignment study	12.5	16N,14W,24A	15-Jul-13	440	DV	3	3	3	2.7 - 3.6	3.3	0.7	0.0
					CT	28	28	16	2.0 - 10.4	4.3	3.6	2.7
Cottonwood Creek, North Fork	0.2	16N,14W,13B	16-Sep-13	2150	DV	12	12	1	4.0 - 9.4	7.2	0.0	0.5
Bull trout genetics assignment study												

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2013-2015 (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+
Dunham Creek	2.3	16N,12W,19B	18-Jul-13	600	DV	3	3	1	2.0 - 8.0	5.1	0.2	0.3
Bull trout genetics assignment study					CT	18	18	1	2.3 - 14.5	7.0	0.2	2.8
					EB	14	14	11	1.7 - 7.9	3.0	1.8	0.5
Gold Creek	0.1	13N,16W,6B	20-Jul-15	501	RB	8	8	0	3.3 - 13	5.7	0.0	1.6
					CT	1	1	0	9.5	9.5	0.0	0.2
					LL	3	3	2	2.3 - 14.7	6.5	0.4	0.2
					Sculpins	present	LND	present				
	1.9	14N,16W,30D	20-Jul-15	400	RB*	33	23	2	1.4 -10.3	5.1	0.5	5.3
					CT	1	1	0	6.9	6.9	0.0	0.3
					LL	9	7	1	2.4 -10.4	6.5	0.3	1.5
					Sculpins	common						
	5.7	14N,16W,7C	21-Jul-15	300	CT	4	4	0	4.1 - 6.0	5.0	0.0	1.3
					RB	18	18	0	3.4 - 8.4	4.5	0.0	6.0
					EB	2	2	1	2.8 - 5.5	4.1	0.3	0.3
					LL	44	44	22	1.9 - 10.4	4.3	7.3	7.3
					ONC	4	4	4	1.3 - 1.8	1.5	1.3	0.0
					Sculpins	common						
	9	15N,17W,25C	21-Jul-15	342	CT	29	29	1	1.1 - 6.3	4.4	0.3	8.2
					RB	2	2	0	5.1 - 6.1	5.6	0.0	0.6
					LL	4	4	0	5.0 - 8.0	6.6	0.0	1.2
					EB	31	31	0	1.7 - 6.2	4.0	0.0	9.1
					No sculpins	observed						
Gold Creek ,West Fork	0.1	14N,17W,1D	6-Aug-15	361	CT	7	7	0	4.3 - 5.8	4.9	0.0	1.9
					RB	37	37	0	3.8 - 7.9	5.2	0.0	10.2
					LL	14	14	0	4.3 - 8.6	6.5	0.0	3.9
					EB	60	60	25	2.3 - 6.2	4.4	6.9	9.7
					ONC	19	19	19	1.5 - 2.4	2.0	5.3	0.0
					Spotted frog	present						
Grantier Spring Creek	1.0	14N,9W,25A	12-Aug-14	521	CT	15	11	4	1.2 - 18.2	9.5	0.8	1.3
					LL	26	24	22	2.3 - 15.2	3.8	4.2	0.4
					EB	10	10	9	2.1 - 6.8	2.9	1.7	0.2
					Sculpins	abundant	Spotted frogs	present				
Keep Cool Creek	1.8	14N,9W,14C	30-Jul-14	870	LL	8	8	0	4.4 - 9.7	6.9	0.0	0.9
					CT	1	1	0	9.4	9.4	0.0	0.1
					EB	1	1	0	8.7	8.7	0.0	0.1
					Sculpins	present						
	3.3	14N,9W,14A	8-Jul-14	620	LL	3	3	0	4.6 - 13.4	8.0	0.0	0.5
	4.3	14N,9W,13B	8-Jul-14	360	Sculpins	3	3	0	2.4 - 3.7	3.1	0.0	0.8
	5.5	14N,8W,18C	30-Jul-14	560	LL	1	1	0	10.2	10.2	0.0	0.2
					EB	3	3	0	6.8 - 8	7.6	0.0	0.5
					Sculpins	5	5	0	3.2 - 4.8	3.8	0.0	0.9
	7.7	14N,8W,16B	8-Jul-14	288	CT*	2	2	0	5.0 - 5.2	5.1	0.0	0.7
					Sculpins	12	12	1	1.2 - 3.0	2.4	0.3	3.8
	8.9	14N,8W,9B	8-Jul-14	285	CT	8	8	0	4.5 - 10.3	6.3	0.0	2.8
					EB	3	3	0	5.0 - 7.5	6.6	0.0	1.1

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2013-2015 (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+
Klondike Creek	0.1	15N,9W,29C	7-Jul-14	390	CT	17	16	1	1.7 - 5.6	3.9	0.3	3.8
						Tailed frog	larva	abundant				
Lincoln Spring Creek	3.8	14N,9W,13D	29-Aug-13	385	LL	24	19	3	2.1 - 14.4	8.9	0.8	4.2
					EB	10	7	2	1.9 - 8.7	5.2	0.5	1.3
					Sculpins	common						
			6-Aug-14	385	LL	15	11	0	4.3 - 13	8.3	0.0	2.9
					EB	2	2	0	5.4 - 8.1	6.8	0.0	0.5
					Sculpins	common						
			18-Aug-15	385	LL	34	26	12	2.2 - 15.9	5.1	3.1	3.6
					EB	8	5	1	3.4 - 8.3	6.6	0.3	1.0
					Sculpins	common						
Murphy's Spring Creek	0.6	15N,11W,21B	10-Sep-13	348	DV	3	1	0	4.7 - 8.2	6.0	0.0	0.3
Bull trout genetics assignment study					CT	58	45	15	1.6 - 7.8	3.8	4.3	8.6
					EB	3	3	3	1.8 - 2.4	2.2	0.9	0.0
					Sculpins	common	Tailed frog	present				
			2-Sep-14	348	DV	2	1		4.7 - 5.2	5.0	0.0	0.3
					CT	31	22	5	1.2 - 8.1	4.2	1.4	4.9
					EB	1	0	0	3.9	3.9	0.0	0.0
					Sculpins	common						
			31-Aug-15	348	CT	44	40	22	1.5 -7.0	3.2	6.3	5.2
					EB	3	3	2	2.4 - 4.3	3.0	0.6	0.3
					Sculpins	common						
Nevada Creek	5.0 - 6.3	13N,11W,9C	21-Sep-13	6500	CT	110	64	0	5.5 - 15	10.3	0.0	1.0
					DV	2	2	0	15.4 - 16.2	15.8	0.0	0.0
					LL	62	43	0	8.5 - 19.4	12.4	0.0	0.7
					MWF & LSS	common						
			28-Sep-15	6500	CT	57	39	0	6.5 - 16.2	11.5	0.0	0.6
					DV	1	1	0	15	15.0	0.0	0.0
					RB	1	1	0	14.8	14.8	0.0	0.0
					LL	86	63	0	4.1 - 19.2	10.4	0.0	1.0
					MWF & LSS	common						
	29.0	12N,10W,11C	17-Sep-13	3440	CT	64	64	0	6.3 - 15.2	10.5	0.0	1.9
					RB	248	248	0	3.9 - 16.7	8.3	0.0	7.2
					LL	23	23	0	15 - 21.9	17.8	0.0	0.7
					EB	12	12	0	6.5 - 11.5	7.8	0.0	0.3
					Sculpins,RSS	MWF ,LSS	& LNS	observed				
			15-Sep-15	3440	CT	55	39	0	5.2 - 16.9	10.9	0.0	1.1
					RB	153	95	7	3.2 - 18	9.0	0.2	2.6
					LL	18	14	0	8.9 - 22.8	15.3	0.0	0.4
					Sculpins,RSS	MWF ,LSS	& LNS	observed				
Nevada Spring Creek	0.1	13N,11W,9D	28-Sep-15	353	CT	8	8	0	5.6 - 9.1	7.6	0.0	2.3
					LL	3	3	0	4.4 - 4.9	4.7	0.0	0.8
					LNS	1	1	0	6.5	6.5	0.0	0.3
					MWF	1	1	0	4.4	4.4	0.0	0.3
					RSS	1	1	1	3.4	3.4	0.3	0.0
Devils Dip trib to Nevada Spring Creek	0.1	13N,11W,11A	9-Sep-15	225	No fish found	Spotted frog	observed					

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2013-2015 (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+
Pearson Creek	0.5	15N,13W,33D	9-Sep-14	300	CT	41	33	0	4.7 - 8.4	6.1	0.0	11.0
					EB	1	1	0	4.5	4.5	0.0	0.3
					LNS	1	1	0	4.5	4.5	0.0	0.3
			16-Sep-15	300	CT	16	15	0	3.8 - 6.4	4.9	0.0	5.0
					Sculpins	present						
	1.1	14N,13W,3B	16-Sep-15	370	CT	55	44	0	2.8 - 7.0	3.9	0.0	11.9
Poorman Creek	1.3	14N,9W,36A	26-Aug-13	510	CT	6	5	2	1.8 - 6.8	4.0	0.4	0.6
					LL	46	25	23	2.1 - 4.3	2.5	4.5	0.4
					EB	1	1	0	6.1	6.1	0.0	0.2
					Sculpins	common						
			17-Aug-15	510	CT	15	11	8	1.4 - 4.1	2.2	1.6	0.6
					LL	146	115	106	1.7 - 4.8	2.6	20.8	1.8
					Sculpins	common						
	1.5	14N,9W,36A	26-Aug-13	270	CT	12	8	3	1.8 - 11.5	5.8	1.1	1.9
					LL	44	25	11	2.0 - 14.7	6.7	4.1	5.2
					Sculpins	present						
			17-Aug-15	270	CT	10	10	4	1.5 - 4.6	3.0	1.5	2.2
					LL	39	31	14	2.0 - 14.8	5.6	5.2	6.3
					Sculpins	present						
Bull trout genetics assignment study	8.4	13N,8W,23B	16-Jul-13	300	CT	26	26	0	3.0 - 9.1	5.4	0.0	8.7
					DV	5	5	0	4.1 - 5.3	4.6	0.0	1.7
					Sculpins	present						
Bull trout genetics assignment study	9.9	13N,8W,24A	16-Jul-13	340	DV	22	22	5	1.7 - 20.3	5.1	1.5	5.0
		13N,7W,19B			CT	31	31	0	3.9 - 9.3	5.9	0.0	9.1
					EB	1	1	0	6.1	6.1	0.0	0.3
					EBxDV	1	1	0	8.1	8.1	0.0	0.3
					Sculpins	common						
Poorman Creek, South Fork	0.6	13N,7W,19A	16-Jul-13	510	CT	12	12	5	1.8 - 5.4	3.3	1.0	1.4
					EB	1	1	0	4.3	4.3	0.0	0.2
Sauerkraut Creek	2.9	13N,9W,5D	14-Aug-13	228	CT	22	22	18	1.2 - 4.8	2.0	7.9	1.8
					EB	1	1	0	8.2	8.2	0.0	0.4
					Sculpins	present	Spotted frogs	present				
			13-Aug-14	297	CT	39	35	5	1.2 - 7.4	4.1	1.7	10.1
					EB	5	4	0	4.3 - 4.8	4.6	0.0	1.3
					Spotted frogs	common						
	3.2	13N,9W,8A	14-Aug-13	303	CT	52	45	2	1.3 - 6.5	3.5	0.7	14.2
					EB	2	2	0	5.4 - 5.9	5.6	0.0	0.7
					Sculpins	common						
			13-Aug-14	273	CT	19	17	1	1.3 - 7.2	4.6	0.4	5.9
					EB	4	3	0	5.2 - 7.8	6.8	0.0	1.1
					Sculpins	common						
			24-Aug-15	303	CT	62	51	2	1.3 - 6.2	3.9	0.7	16.2
					EB	3	3	2	2.1 - 7.2	3.9	0.7	0.3
					CT YOY	present	Sculpins	common				

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2013-2015 (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+
Shanley Creek	0.2	15N,13W,9B	5-Aug-15	360	CT	5	4	0	6.3 - 8.0	6.9	0.0	1.1
					LL	18	14	5	2.4 - 9.1	5.5	1.4	2.5
					RB	1	1	0	6.8	6.8	0.0	0.3
					EB	72	50	24	2.0 - 8.2	4.5	6.7	7.2
					Sculpins	common						
	1.6	15N,13W,3B	5-Aug-15	498	CT	38	34	0	3.0 - 8.1	4.6	0.0	6.8
					EB	26	22	14	1.9 - 8.0	3.6	2.8	1.6
					Sculpins	common						
Snowbank Creek	0.4	15N,8W,9A	27-Aug-13	450	DV	38	24	17	2.4 - 9.0	3.4	3.8	1.6
					CT	32	24	4	1.3 - 12.4	4.3	0.9	4.4
					Sculpins	present	Tailed frogs	observed				
			19-Aug-14	450	DV	21	15	5	1.9 -16.3	4.8	1.1	2.2
					CT	15	12	0	2.6 - 6.7	4.0	0.0	2.7
					Sculpins	present	Tailed frogs	common				
			8-Sep-15	450	DV	6	5	3	2.7 - 5.8	3.7	0.7	0.4
					CT	3	2	2	1.6 - 2.2	1.9	0.4	0.0
					Sculpins	present	Western toads	observed	Small toads	common		
Stonewall Creek	0.1	14N,9W,14C	29-Jul-14	405	CT	1	1	0	11	11.0	0.0	0.2
					LL	2	2	0	5.4 - 9.4	7.4	0.0	0.5
					EB	1	1	1	3.2	3.2	0.2	0.0
					Sculpins	common						
	0.65	14N,9W,14B	29-Jul-14	400	CT	4	4	0	5.2 - 10.2	7.9	0.0	1.0
					LL	1	1	0	5.2	5.2	0.0	0.3
					EB	3	3	2	2.4 - 7.4	4.1	0.5	0.3
					Sculpins	common						
	3	14N,9W,2B	29-Jul-14	300	CT	4	4	0	3.3 - 4.0	3.6	0.0	1.3
					LL	1	1	0	7.1	7.1	0.0	0.3
					EB	5	5	2	2.2 - 5.2	3.9	0.7	1.0
					Sculpins	common						
Patterson-Yonderian irrigation ditch at Stonewall Creek stream mile 4.3	0.1	15N,9W,34A	29-Jul-14	375	CT	8	8	0	3.6 - 6.1	4.4	0.0	2.1
					Traied frogs	observed						
	4.7	15N,9W,34A	28-Jul-14	282	CT	12	12	2	2.1 - 6.8	4.0	0.7	3.5
					Tailed	frogs	common					
	5.2	15N,9W,27C	28-Jul-14	456	CT	17	17	0	3.2 - 8.6	5.0	0.0	3.7
					Tailed frogs	common	Western toads	observed				
	5.7	15N,9W,27C	28-Jul-14	399	CT	8	8	0	3.7 - 6.3	4.3	0.0	2.0
					Tailed frogs	common						

Appendix A: Catch and size statistics for tributaries to the Blackfoot River, 2013-2015 (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+
Sucker Creek	1.6	14N,8W,7B	9-Jul-14	260	No fish	captured or	observed					
	2.6	14N,8W,6D	9-Jul-14	282	CT	4	4	0	3.6 - 7.3	5.6	0.0	1.4
					EB	3	3	0	5.1 - 5.2	5.2	0.0	1.1
					Sculpins	5	5	0	2.0 - 4.0	3.0	0.0	1.8
	3.8	15N,8W,32C	9-Jul-14	228	CT	5	5	0	4.0 - 5.7	5.0	0.0	2.2
Theodore Creek	0.1	15N,9W,33C	7-Jul-14	318	CT	24	23	1	2.0 - 6.0	3.8	0.3	6.9
Downstream of USFS road 4106 culvert					EB	7	7	4	2.4 - 6.8	4.2	1.3	0.9
Upstream of USFS road 4106 culvert	0.15	15N,9W,33C	7-Jul-14	396	CT	18	18	0	2.0 - 5.4	3.6	0.0	4.5
					EB	3	3	3	2.2 - 3.0	2.7	0.8	0.0
					Tailed frog	larva	abundant					
Un-named spring creek entering North Fork at mile 3.0	0.2	14N,12W,11C	31-Jul-13	225	LL	1	1	1	2.7	2.7	0.4	0.0
					Sculpins	present						
Yukon Creek	0.1	15N,9W,29C	7-Jul-14	249	CT	21	21	0	3.2 - 6.3	4.2	0.0	8.4
					Tailed frog	larva	present					

***** Sample may include rainbow trout / cutthroat trout hybrids
****** Sample may include bull trout / brook trout hybrids
******* Sample maybe include Yellowstone cutthroat hybrids
CT = Cutthroat trout
DV = Bull trout (Dolly Varden)
LL = Brown trout (Loch Leven)
RB = Rainbow trout
EB = Eastern brook trout
MWF = Mountain 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, 2013-2015.

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
Ashby Creek	2	13N,16W,26A	5-Aug-14	300	CT	Age 1+	9	1		0.89	10.1 ± 0.9	3.4 ± 0.3
					EB	Age 1+	1	1				
					All	Age 1+	10	2		0.80	12.5 ± 2.1	4.2 ± 0.7
	2.7	13N,16W,26D	31-Jul-13	489	CT	Age 1+	29	3		0.90	32.4 ± 1.4	6.6 ± 0.3
			5-Aug-14	489	CT	YOY	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
						Age 1+	48	16		0.67	72 ± 11.8	14.7 ± 2.4
					EB	YOY	0	1				
						Age 1+	4	1		0.75	5.3 ± 1.9	1.1 ± 0.4
					All	YOY	1	1				
						Age 1+	52	17		0.67	77.3 ± 11.7	15.8 ± 2.4
Bear Creek	1.1	13N,16W,18B	4-Aug-14	393	RB	Age 1+	45	13		0.71	63.3 ± 8.5	16.1 ± 2.2
lower river tributary					CT	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					LL	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	17	4		0.76	22.2 ± 3.6	5.7 ± 0.9
					EB	YOY	1	2				
						Age 1+	3	1		0.67	4.5 ± 2.9	1.1 ± 0.7
					All	YOY	2	2				
						Age 1+	66	18		0.73	90.8 ± 9.3	23.1 ± 2.4
Braziel Creek	0.2	12N,10W,10D	28-Aug-13	300	CT	YOY	3	1		0.67	4.5 ± 2.9	1.5 ± 1.0
						Age 1+	61	14		0.77	79.2 ± 6.6	26.4 ± 2.2
			18-Aug-14	300	CT	YOY	41	6		0.85	48 ± 2.7	16 ± 0.9
						Age 1+	8	1		0.88	9.1 ± 1.0	3.0 ± 0.3
					EB	Age 1+	0	2				
					Sculpins	YOY	7	0		1.00	7.0 ± 0.0	2.3 ± 0.0
						Age 1+	13	6		0.54	24.1 ± 13.6	8.0 ± 4.5
			9-Sep-15	300	CT	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	40	6		0.85	47.1 ± 2.8	15.7 ± 0.9
Chamberlain Creek	0.1	15N,13W,32A	9-Sep-14	358	CT	YOY	8	0		1.00	8.0 ± 0.0	2.2 ± 0.0
						Age 1+	43	15		0.65	66 ± 12.3	18.4 ± 3.4
						All CT	51	15		0.71	72.3 ± 9.4	20.2 ± 2.6
					LL	Age 1+	3	0		1.00	3.0 ± 0.0	0.8 ± 0.0
					All	YOY	8	0		1.00	8.0 ± 0.0	2.2 ± 0.0
						Age 1+	59	15		0.75	79.1 ± 7.7	22.1 ± 2.2
			1-Oct-15	358	CT	YOY	7	2	1	0.71	10 ± 0.0	2.8 ± 0.0
						Age 1+	15	12	4	0.51	33.9 ± 5.1	9.5 ± 2.7
						All CT	22	14	5	0.55	44.1 ± 5.2	12.3 ± 2.7
					LL	YOY	1	1				
						Age 1+	3	1		0.67	4.5 ± 2.9	1.3 ± 0.8
					All	YOY	8	3	1	0.71	12 ± 0.0	3.4 ± 0.0
						Age 1+	18	13	4	0.54	37.6 ± 4.6	10.5 ± 2.5

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2013-2015 (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	24-Sep-14	515	CT	YOY	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
						Age 1+	23	7		0.70	33.1 ± 6.8	6.4 ± 1.3
					EB	Age 1+	3	1		0.67	4.5 ± 2.9	0.9 ± 0.6
					EB x DV	Age 1+	2	1		0.50	4.0 ± 6.8	0.8 ± 1.3
					All	YOY	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
						Age 1+	28	9		0.68	41.3 ± 8.3	8.0 ± 1.6
			30-Sep-15	515	CT	Age 1+	43	14		0.67	63.8 ± 10.6	12.4 ± 2.1
					EB	YOY	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
						Age 1+	6	4		0.33	18 ± 37.2	3.5 ± 7.2
					EB x DV	Age 1+	3	1		0.67	4.5 ± 2.9	0.9 ± 0.6
					All	YOY	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
						Age 1+	52	19		0.63	82 ± 15	16 ± 3.0
Gold Creek	1.9	14N,16W,30D	20-Jul-15	400	RB*	YOY	2	0		1.00	2.0 ± 0.0	0.5 ± 0.0
						Age 1+	21	10		0.52	40.1 ± 18.9	10 ± 4.7
					CT	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					LL	YOY	1	1				
						Age 1+	6	1		0.83	7.2 ± 1.2	1.8 ± 0.3
					All	YOY	3	1		0.67	4.5 ± 2.9	1.1 ± 0.7
						Age 1+	28	11		0.61	46.1 ± 13	11.5 ± 3.3
Grantier Spring Creek	1.0	14N, 9W, 25A	12-Aug-14	521	CT	YOY	4	0		1.00	4.0 ± 0.0	0.8 ± 0.0
						Age 1+	7	4		0.43	16.3 ± 20.2	3.1 ± 3.9
					LL	YOY	22	2		0.91	24.2 ± 1.1	4.6 ± 0.2
						Age 1+	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
					EB	YOY	9	0		1.00	9.0 ± 0.0	1.7 ± 0.0
						Age 1+	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
					All	YOY	35	2		0.94	37.1 ± 0.8	7.1 ± 0.1
						Age 1+	10	4		0.60	16.7 ± 8.1	3.2 ± 1.6
Lincoln Spring Creek	3.8	14N,9W,13D	29-Aug-13	385	LL	YOY	3	1		0.67	4.5 ± 2.9	1.2 ± 0.8
						Age 1+	16	4		0.75	21.3 ± 3.9	5.5 ± 1.0
					EB	YOY	2	2				
						Age 1+	5	1		0.80	6.3 ± 1.5	1.6 ± 0.4
					All	YOY	5	3		0.40	12.5 ± 20.8	3.2 ± 5.4
						Age 1+	21	5		0.76	27.6 ± 4.1	7.2 ± 1.1
			6-Aug-14	385	LL	Age 1+	11	4		0.64	17.3 ± 6.8	4.5 ± 1.8
					EB	Age 1+	2	0		1.00	2.0 ± 0.0	0.5 ± 0.0
					ALL	Age 1+	13	4		0.69	18.8 ± 5.2	4.9 ± 1.3
			18-Aug-15	385	LL	YOY	12	6		0.50	24 ± 16.6	6.2 ± 4.3
						Age 1+	14	2		0.86	16.3 ± 1.5	4.2 ± 0.4
					EB	YOY	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
						Age 1+	4	3		0.25	16 ± 62.2	4.2 ± 16.2
					All	YOY	13	6		0.54	24.1 ± 13.6	6.3 ± 3.5
						Age 1+	18	5		0.72	24.9 ± 5.0	6.5 ± 1.3

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2013-2015 (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
Murphy's Spring Creek	0.6	15N,11W,21B	10-Sep-13	348	DV	Age 1+	1	2				
					CT	YOY	15	6		0.60	25 ± 10	7.2 ± 2.9
						Age 1+	30	7		0.77	39.1 ± 4.7	11.2 ± 1.4
					EB	Age 1+	3	0		1.00	3.0 ± 0.0	0.9 ± 0.0
					All	YOY	18	6		0.67	27 ± 7.2	7.8 ± 2.1
						Age 1+	31	9		0.71	43.7 ± 7.1	12.6 ± 2.1
			2-Sep-14	348	DV	Age 1+	1	1				
					CT	YOY	5	1		0.80	6.3 ± 1.5	1.8 ± 0.4
						Age 1+	17	8		0.53	32.1 ± 16.5	9.2 ± 4.7
					EB	Age 1+	0	1				
					All	YOY	5	1		0.80	6.3 ± 1.5	1.8 ± 0.4
						Age 1+	18	9		0.50	36 ± 20.4	10.3 ± 5.9
			31-Aug-15	348	CT	YOY	22	0		1.00	22 ± 0.0	6.3 ± 0.0
						Age 1+	18	4		0.78	23.1 ± 3.4	6.7 ± 1.0
					EB	YOY	2	0		1.00	2.0 ± 0.0	0.6 ± 0.0
						Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					All	YOY	24	0		1.00	24 ± 0.0	6.9 ± 0.0
						Age 1+	19	4		0.79	24.1 ± 3.2	6.9 ± 0.9
Pearson Creek	0.5	15N,13W,33D	9-Sep-14	300	CT	Age 1+	33	8		0.76	43.6 ± 5.3	14.5 ± 1.8
					EB	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					LNS	Age 1+	1	0		1.00		
					All trout	Age 1+	34	8		0.76	44.5 ± 5.1	14.8 ± 1.7
			16-Sep-15	300	CT	Age 1+	15	1		0.93	16.1 ± 0.6	5.4 ± 0.2
	1.1	14N,13W,3B	16-Sep-15	370	CT	Age 1+	44	11		0.75	58.7 ± 6.5	15.9 ± 1.7
Poorman Creek	1.3	14N,9W,36A	26-Aug-13	510	CT	YOY	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
						Age 1+	3	1		0.67	4.5 ± 2.9	0.9 ± 0.6
					LL	YOY	23	21		0.09	265 ± 1570	52 ± 308
						Age 1+	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
					EB	Age 1+	1	0		1.00	1.0 ± 0.0	0.2 ± 0.0
					All	YOY	25	21		0.16	156.3 ± 436.2	30.6 ± 85.5
						Age 1+	6	1		0.83	7.2 ± 1.2	1.4 ± 0.2
						Total	31	22		0.29	106.8 ± 120.1	20.9 ± 23.6
			17-Aug-15	510	CT	YOY	8	3		0.63	12.8 ± 6.2	2.5 ± 1.2
						Age 1+	3	1		0.67	4.5 ± 2.9	0.9 ± 0.6
					LL	YOY	106	31		0.71	149.8 ± 13.4	29.4 ± 2.6
						Age 1+	9	0		1.00	9.0 ± 0.0	1.8 ± 0.0
					All	YOY	114	34		0.70	162.5 ± 14.4	31.9 ± 2.8
						Age 1+	12	1		0.92	13.1 ± 0.7	2.6 ± 0.1
						Total	126	35		0.72	174.5 ± 13.2	34.2 ± 2.6
	1.5	14N,9W,36A	26-Aug-13	270	CT	YOY	3	0		1.00	3.0 ± 0.0	1.1 ± 0.0
						Age 1+	5	3	1	0.40	9.0 ± 0.0	3.3 ± 0.0
					LL	YOY	11	7		0.36	30.3 ± 40	11.2 ± 14.8
						Age 1+	14	10	2	0.29	26.7 ± 2.0	9.9 ± 1.2
					All	YOY	14	7		0.50	28 ± 18	10.4 ± 6.7
						Age 1+	19	13	3	0.32	36.6 ± 3.3	13.6 ± 2.0
						Total	33	20	3	0.39	57.9 ± 3.4	21.4 ± 2.1

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2013-2015 (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
Poorman Creek (cont'd)	1.5	14N,9W,36A	17-Aug-15	270	CT	YOY	4	0		1.00	4.0 ± 0.0	1.5 ± 0.0
						Age 1+	6	0		1.00	6.0 ± 0.0	2.2 ± 0.0
					LL	YOY	14	2		0.86	16.3 ± 1.5	6.0 ± 0.6
						Age 1+	17	6		0.65	26.3 ± 7.9	9.7 ± 2.9
					All	YOY	18	2		0.89	20.3 ± 1.2	7.5 ± 0.5
						Age 1+	23	6		0.74	31.1 ± 5.0	11.5 ± 1.9
						Total	41	8		0.80	50.9 ± 4.1	18.9 ± 1.5
Sauerkraut Creek	2.9	13N,9W,5D	14-Aug-13	228	CT	YOY	18	0		1.00	18 ± 0.0	7.9 ± 0.0
						Age 1+	4	0		1.00	4.0 ± 0.0	1.8 ± 0.0
					EB	Age 1+	1	0		1.00	1.0 ± 0.0	0.4 ± 0.0
					All	YOY	18	0		1.00	18 ± 0.0	7.9 ± 0.0
						Age 1+	5	0		1.00	5.0 ± 0.0	2.2 ± 0.0
			13-Aug-14	297	CT	YOY	5	0		1.00	5.0 ± 0.0	1.7 ± 0.0
						Age 1+	30	4		0.87	34.6 ± 2.0	11.7 ± 0.7
					EB	Age 1+	4	1		0.75	5.3 ± 1.9	1.8 ± 0.7
					All	YOY	5	0		1.00	5.0 ± 0.0	1.7 ± 0.0
						Age 1+	34	5		0.85	39.9 ± 2.5	13.4 ± 0.8
	3.2	13N,9W,8A	14-Aug-13	303	CT	YOY	2	0		1.00	2.0 ± 0.0	0.7 ± 0.0
						Age 1+	43	7		0.84	51.4 ± 3.2	17 ± 1.1
					EB	Age 1+	2	0		1.00	2.0 ± 0.0	0.7 ± 0.0
					All	YOY	2	0		1.00	2.0 ± 0.0	0.7 ± 0.0
						Age 1+	45	7		0.84	53.3 ± 3.1	17.6 ± 1.0
			13-Aug-14	273	CT	YOY	1	0		1.00	1.0 ± 0.0	0.4 ± 0.0
						Age 1+	16	2		0.88	18.3 ± 1.4	6.7 ± 0.5
					EB	Age 1+	3	1		0.67	4.5 ± 2.9	1.6 ± 1.1
					All	YOY	1	0		1.00	1.0 ± 0.0	0.4 ± 0.0
						Age 1+	19	3		0.84	22.6 ± 2.0	8.3 ± 0.7
			24-Aug-15	303	CT	YOY	2	0		1.00	2.0 ± 0.0	0.7 ± 0.0
						Age 1+	49	11		0.78	63.2 ± 5.7	20.9 ± 1.9
					EB	YOY	2	0		1.00	2.0 ± 0.0	0.7 ± 0.0
						Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					All	YOY	4	0		1.00	4.0 ± 0.0	1.3 ± 0.0
						Age 1+	50	11		0.78	64.1 ± 5.5	21.2 ± 1.8

Appendix B: Two-pass depletion estimates for tributaries to the Blackfoot River, 2013-2015 (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
Shanley Creek	0.2	15N, 13W, 9B	5-Aug-15	360	CT	Age 1+	4	1		0.75	5.3 ± 1.9	1.5 ± 0.5
					RB	Age 1+	1	0		1.00	1.0 ± 0.0	0.3 ± 0.0
					LL	YOY	5	2		0.60	8.3 ± 5.8	2.3 ± 1.6
						Age 1+	9	2		0.78	11.6 ± 2.4	3.2 ± 0.7
					EB	YOY	24	9		0.63	38.4 ± 10.8	10.7 ± 3.0
						Age 1+	26	13		0.50	52 ± 24.5	14.4 ± 6.8
					All	YOY	29	11		0.62	46.7 ± 12.2	13 ± 3.4
						Age 1+	40	16		0.60	66.7 ± 16.3	18.5 ± 4.5
	1.6	15N, 13W, 3B	5-Aug-15	498	CT	Age 1+	34	4		0.88	38.5 ± 1.8	7.7 ± 0.4
					EB	YOY	14	3		0.79	17.8 ± 2.8	3.6 ± 0.6
						Age 1+	8	1		0.88	9.1 ± 1.0	1.8 ± 0.2
					All	YOY	14	3		0.79	17.8 ± 2.8	3.6 ± 0.6
						Age 1+	42	5		0.88	47.7 ± 2.1	9.6 ± 0.4
Snowbank Creek	0.4	15N,8W,9A	27-Aug-13	450	DV	YOY	17	11		0.35	48.2 ± 53.9	10.7 ± 12
						Age 1+	7	3		0.57	12.3 ± 8.1	2.7 ± 1.8
					CT	YOY	4	2		0.50	8.0 ± 9.6	1.8 ± 2.1
						Age 1+	20	6		0.70	28.6 ± 6.1	6.3 ± 1.4
					All trout	YOY	21	13		0.38	55.1 ± 49	12.3 ± 10.8
						Age 1+	27	9		0.67	41 ± 8.8	9.0 ± 2.0
			19-Aug-14	450	DV	YOY	5	3		0.40	12.5 ± 20.8	2.8 ± 4.6
						Age 1+	10	3		0.70	14.3 ± 4.3	3.2 ± 1.0
					CT	Age 1+	12	3		0.75	16.0 ± 3.4	3.6 ± 0.7
					All trout	YOY	5	3		0.40	12.5 ± 20.8	2.8 ± 4.6
						Age 1+	22	6		0.73	30.3 ± 5.3	6.7 ± 1.2
			8-Sep-15	450	DV	YOY	3	1		0.67	4.5 ± 2.9	1.0 ± 0.7
						Age 1+	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0
					CT	YOY	2	1		0.50	4.0 ± 6.8	0.9 ± 1.5
					All trout	YOY	5	2		0.60	8.3 ± 5.8	1.9 ± 1.3
						Age 1+	2	0		1.00	2.0 ± 0.0	0.4 ± 0.0

* 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, 2013 - 2015.

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	19-May-14	17680	RB	≥ 6	198	216	32	0.15	1307.6 ± 370	688.40	74 ± 21	38.9	34.4
				LL	≥ 6	34	29	10	0.34	94.5 ± 35.6	124.30	5.3 ± 2.0	7.0	31.9
				CT	≥ 6	62	65	9	0.14	415 ± 208	269.00	23.5 ± 11.7	15.2	37.0
				DV	≥ 6	14	8	1	0.13					
				All trout	≥ 6	308	318	52	0.16	1859 ± 412.3	1313.10	105.1 ± 23.3	74.3	34.7
Blackfoot River, Scotty Brown Bridge	43.9	20-May-14	20064	RB	≥ 6	124	102	25	0.25	494.2 ± 144	497.00	24.6 ± 7.2	24.8	35.1
				LL	≥ 6	39	45	10	0.22	166.3 ± 70.3	244.00	8.3 ± 3.5	12.2	34.9
				CT	≥ 6	91	72	9	0.13	671 ± 348.1	619.10	33.4 ± 17.4	30.9	36.2
				DV	≥ 6	16	13	4	0.31	47 ± 25.7	152.70	2.3 ± 1.3	7.6	32.2
				All trout	≥ 6	270	232	48	0.21	1288 ± 287.3	1517.00	64.2 ± 14.3	75.6	35.3
Blackfoot River Wales Creek Section	63	21-May-14	31635	RB	≥ 6	8	10	1	0.10					
				LL	≥ 6	73	63	21	0.33	214.3 ± 59.8	262.40	6.8 ± 1.9	8.3	32.2
				CT	≥ 6	22	13	3	0.23	79.5 ± 54.2	49.40	2.5 ± 1.7	1.6	33.7
				All trout	≥ 6	104	89	25	0.28	362.5 ± 100.3	398.80	11.5 ± 3.2	12.6	32.6
	63.6	9-Jun-08	23760	MWF	≥ 8	261	252	22	0.09	2881 ± 1050	2100	121 ± 44	88	38
		19-May-10		MWF	≥ 8	348	346	36	0.10	3272 ± 930	2177	138 ± 39	92	37
		17-May-12		MWF	≥ 8	657	551	134	0.24	2690 ± 350	1483	113 ± 15	62	35
		21-May-14		MWF	≥ 8	617	478	114	0.24	2573 ± 368	1258	108 ± 16	53	31
Blackfoot River Canyon Section	95.3	22-Sep-14	5422	CT	≥ 6	13	11	4	0.36	32.6 ± 16.5	32.40	6.0 ± 3.0	6.0	34.0
				LL	≥ 6	21	23	3	0.13	131 ± 95.5	114.80	24.2 ± 17.6	21.2	33.3
				All trout	≥ 6	34	34	7	0.21	152.1 ± 77.2	139.10	28.1 ± 14.2	25.7	33.5
	95.3	20-Sep-06	5422	MWF	≥ 8	177	121	24	0.20	868 ± 276	654	160 ± 51	121	35
		24-Sep-09		MWF	≥ 8	109	92	16	0.17	601 ± 231	525	111 ± 43	97	37
		29-Sep-11		MWF	≥ 8	177	54	11	0.20	815 ± 379	745	150 ± 70	137	35
		22-Sep-14		MWF	≥ 8	60	64	4	0.06	792 ± 584	565	146 ± 108	104	32
Nevada Creek upstream of H2-O	5.0 - 6.3	27-Sep-13	6500	CT	≥4.0	64	71	25	0.35	179 ± 42	83.5	28 ± 6.5	12.8	38.7
				DV	≥4.0	2	0	0						
				LL	≥4.0	43	34	15	0.44	95.3 ± 27	75.9	15 ± 4.1	11.7	38.8
				All trout	≥4.0	109	105	40	0.38	283 ± 53.3	167.6	43.6 ± 8.2	25.8	38.6
		28-Sep-15	6500	CT	≥4.0	39	39	21	0.54	71.7 ± 13.4	41.5	11.03 ± 2.1	6.4	34.9
				DV	≥4.0	1	0	0						
				LL	≥4.0	63	53	30	0.57	110.5 ± 18.1	58.7	17 ± 2.8	9.0	36.3
				RB	≥4.0	1	1	1	1.00					
				All trout	≥4.0	104	93	52	0.56	185.2 ± 23.1	103.0	28.5 ± 3.6	15.7	35.6
Nevada Creek Stit project	29	8-Sep-14	3440	CT	≥4.0	26	29	13	0.45	56.9 ± 14.8	36.1	16.5 ± 4.3	10.5	37.3
				RB	≥4.0	64	48	29	0.60	105.2 ± 17.1	74.6	30.6 ± 5.0	21.7	38.1
				LL	≥4.0	14	20	7	0.35	38.4 ± 13.8	67.6	11.2 ± 4.0	19.7	38.2
				All trout	≥4.0	104	97	49	0.51	204.8 ± 28.6	179.3	59.5 ± 8.3	52.1	37.9
		15-Sep-15	3440	CT	≥4.0	39	26	10	0.38	97.2 ± 36.4	57.8	28.3 ± 11	16.8	35.5
				RB	≥4.0	88	85	27	0.32	272.4 ± 67.6	124.5	79.2 ± 19.7	36.2	36.5
				LL	≥4.0	14	10	6	0.60	23 ± 7.2	39.2	6.6 ± 2.1	11.4	43.7
				All trout	≥4.0	141	121	43	0.36	393 ± 76.4	234.4	114.2 ± 22.2	68.1	36.8

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: Summary of water chemistry readings for 2013.

Stream name	Date	River Mile	pH	Conductivity (uS)	TDS (ppm)	Temp (°F)	Lat	Long	TRS
Ashby Creek	31-Jul-13	2.7	8.9	399	283	55	N46.85028	W113.58539	13N,16W,26D
Blackfoot River above Pass Creek	14-Aug-13	130.5	7.78	303	215	63	N47.04251	W112.38390	15N,6W,20A
Blackfoot River above Shave Creek	14-Aug-13	131.8	8.5	461	326	70	N47.03933	W112.37019	15N,6W,21D
Braziel Creek	28-Aug-13	0.2	8.9	142	101	60	N46.80900	W112.84003	12N,10W,10D
Lincoln Spring Creek	29-Aug-13	3.8	8.1	327	232	54	N46.96301	W112.67539	14N,9W,13D
Murphy Spring Creek	10-Sep-13	0.6	8.6	189	89	52	N47.04339	W113.00664	15N,11W,21B
Poorman Creek	26-Aug-13	1.3	9.1	257	184	58	N46.92926	W112.67264	14N,9W,36A
Poorman Creek	26-Aug-13	1.5	9.1	258	183	55	N46.92707	W112.67097	14N,9W,36A
Poorman Creek	16-Jul-13	9.9	8.8	283	201	51	N46.87288	W112.54892	13N,8W,24A
Poorman Creek (South Fork)	16-Jul-13	0.6	8.6	295	209	50	N46.87187	W112.52802	13N,7W,20B
Sauerkraut Creek	14-Aug-13	2.9	9.1	94	66	52	N46.90465	W112.75515	13N,9W,5D
Sauerkraut Creek	14-Aug-13	3.2	8.3	95	67	51	N46.90013	W112.75790	13N,9W,8A
Snowbank Creek	27-Aug-13	0.4	8.9	183	86	53	N47.07219	W112.61725	15N,8W,9A
Un-named Spring Creek to N F Blackfoot River at mile 3.0	31-Jul-13	0.2	8.8	256	182	67	N46.98116	W113.08167	14N,12W,11C

Summary of water chemistry readings for 2014.

Stream name	Date	River Mile	pH	Conductivity (uS)	TDS (ppm)	Temp (°F)	Lat	Long	TRS
Ashby Creek	5-Aug-14	2	8.8	389	276	55	N46.85638	W113.57493	13N,16W,26A
Ashby Creek	5-Aug-14	2.7	8.8	402	285	55	N46.85028	W113.58540	13N,16W,26D
Bear Creek	4-Aug-14	1.1	8.8	116	82	59	N46.89804	W113.68071	13N,16W,18B
Braziel Creek	18-Aug-14	0.2	8.6	131	93	56	N46.80819	W112.83969	12N,10W,10D
Cottonwood Creek	24-Sep-14	12	8.7	181	129	48	N47.12143	W113.30469	16N,14W,24D
Grantier Spring Creek	12-Aug-14	1	8.4	305	217	51	N46.94054	W112.67441	14N,9W,25A
Keep Cool Creek	30-Jul-14	1.8	8.1	268	190	51	N46.96206	W112.70662	14N,9W,14C
Keep Cool Creek	8-Jul-14	3.3	8.2	307	218	51	N46.96946	W112.69367	14N,9W,14A
Keep Cool Creek	8-Jul-14	4.3	8.1	325	230	52	N46.97018	W112.68021	14N,9W,13B
Keep Cool Creek	30-Jul-14	5.5	8.2	277	197	52	N46.96468	W112.66357	14N,8W,18C
Keep Cool Creek	8-Jul-14	7.7	8.5	234	166	53	N46.97188	W112.62686	14N,8W,16B
Keep Cool Creek	8-Jul-14	8.9	8.4	232	166	52	N46.98730	W112.62083	14N,8W,9B
Klondike Creek	7-Jul-14	0.1	9.1	56	39	48	N47.01562	W112.75661	15N,9W,29C
Lincoln Spring Creek	6-Aug-14	3.8	8.2	331	234	53	N46.98352	W112.67420	14N,9W,13D
Murphy Spring Creek	2-Sep-14	0.6	8.7	179	127	47	N47.04339	W113.00664	15N,11W,21B
Nevada Creek (Stit's project)	8-Sep-14	29	9.6	217	154	60	N46.80296	W112.81900	12N,10W,11C
North Fork Blackfoot River	27-Aug-14	4	8.8	269	190	56	N46.979344	W113.099466	14N,12W.10D
Sauerkraut Creek	13-Aug-14	2.9	8.6	84	59	60	N46.90482	W112.75529	13N,9W,5D
Sauerkraut Creek	13-Aug-14	3.2	8.4	84	60	55	N46.90017	W112.75787	13N,9W,8A
Snowbank Creek	19-Aug-14	0.4	8.7	170	120	53	N47.07219	W112.61725	15N,8W,9A
Stonewall Creek	29-Jul-14	0.1	7.9	219	160	52	N46.96140	W112.70415	14N,9W,14C
Stonewall Creek	29-Jul-14	0.65	8.3	233	166	53	N46.96872	W112.70193	14N,9W,14B
Stonewall Creek	29-Jul-14	3	8.2	231	163	53	N46.98134	W112.70444	14N,9W,2B
Stonewall Creek	28-Jul-14	4.7	8.2	84	59	53	N47.01182	W112.71848	15N,9W,34A
Stonewall Creek	28-Jul-14	5.7	8.2	83	59		N47.02321	W112.72295	15N,9W,27C
Sucker Creek	9-Jul-14	1.6	8	374	265	50	N46.98129	W112.66113	14N,8W,7B
Sucker Creek	9-Jul-14	2.6	8.3	356	253	52	N46.98910	W112.64864	14N,8W,6D
Sucker Creek	9-Jul-14	3.8	8.3	363	257	44	N47.00255	W112.64140	15N,8W,32C
Theodore Creek	7-Jul-14	0.15	8.1	40	28	49	N47.00800	W112.74594	15N,9W,33C
Yukon Creek	7-Jul-14	0.1	8.2	48	44	48	N47.01921	W112.76720	15N,9W,29C

Summary of water chemistry readings for 2015.

Stream name	Date	River Mile	pH	Conductivity (uS)	TDS (ppm)	Temp (^{o}F)	Lat	Long	TRS
Beaver Creek	22-Jul-15	0.2	8.9	247	124	56	N46.95470	W112.72324	14N,9W,22B
Beaver Creek	22-Jul-15	1.4	8.6	229	133	61	N46.96624	W112.72500	14N,9W,15B
Beaver Creek	22-Jul-15	5.4	8.9	146	73	55	N47.01232	W112.75526	15N,9W,32A
Belmont Creek	16-Jul-15	0.1	9.1	292	146	52	N46.95434	W113.57043	14N,16W,24C
Belmont Creek	14-Jul-15	0.3	9	288	144	54	N46.95686	W113.57127	14N,16W,24B
Belmont Creek	16-Jul-15	0.6	9.1	288	145	55	N46.96332	W113.57437	14N,16W,24B
Belmont Creek	15-Jul-15	1.2	9.1	295	147	53	N46.96833	W113.57629	14N,16W,14D
Belmont Creek	15-Jul-15	1.5	9.1	291	146	52	N46.97546	W113.58207	14N,16W,14A
Belmont Creek	15-Jul-15	7.4	8.9	282	141	47	N47.03749	W113.63885	15N,16W,20D
Braziel Creek	9-Sep-15	0.2	8.5	130	92	48	N46.80819	W112.83969	12N,10W,10D
Chamberlain Creek	1-Oct-15	0.1	8.4	151	75	49	N47.01407	W113.26826	15N,13W,32A
Cottonwood Creek	27-Jul-15	1	9.2	255	127	53	N47.03044	W113.27301	15N,13W,29B
Cottonwood Creek	27-Jul-15	3.3	9.2	246	123	51	N47.05053	W113.27157	15N,13W,17D
Cottonwood Creek	6-Aug-15	4.7	8.8	251	126	49	N47.06260	W113.26529	15N,13W,8D
Cottonwood Creek	28-Jul-15	7.5	8.7	244	123	48	N47.08307	W113.27251	15N,13W,5C
Cottonwood Creek	30-Sep-15	12	8.9	181	90	44	N47.12143	W113.30469	16N,14W,24D
Gold Creek	20-Jul-15	1.9	9.1	240	120	53	N46.93935	W113.66874	14N,16W,30D
Gold Creek	21-Jul-15	5.7	9.1	238	119	62	N46.98347	W113.67787	14N,16W,7C
Gold Creek	21-Jul-15	9	8.9	240	120	49	N47.02389	W113.70021	15N,17W,25C
Gold Creek (West Forrk)	6-Aug-15	0.1	8.9	174	87	58	N46.99434	W113.68931	14N,17W,1D
Lincoln Spring Creek	18-Aug-15	3.8	8.8	335	168	50	N46.96352	W112.67421	14N,9W,13D
Murphy Spring Creek	31-Aug-15	0.6	8.8	178	86	51	N47.04339	W113.00664	15N,11W,21B
Nevada Creek	28-Sep-15	6.3	8.9	380	193	48	N46.89553	W112.9986	13N,11W,9C
Nevada Spring Creek	28-Sep-15	0.1	8.7	378	190	46	N46.89566	W112.99908	13N,11W,9C
Pearson Creek	16-Sep-15	1.1	8.9	85	40	47	N47.00516	W113.23421	14N,13W,3B
Poorman Creek	17-Aug-15	1.3	9.2	268	136	49	N46.92808	W112.67198	14N,9W,36A
Poorman Creek	17-Aug-15	1.5	9.2	270	134	49	N46.92707	W112.67097	14N,9W,36A
Sauerkraut Creek	24-Aug-15	3.2	8.8	103	52	48	N46.90017	W112.75787	13N,9W,8A
Shanley Creek	5-Aug-15	0.2	9.1	195	139	51	N47.07777	W113.25694	15N,13W,9B
Shanley Creek	6-Aug-15	1.4	8.5	197	135	47	N47.08821	W113.23683	15N,13W,3B
Shanley Creek	5-Aug-15	1.6	8.6	193	93	49	N47.08750	W113.23327	15N,13W,3B
Snowbank Creek	8-Sep-15	0.4	8.8	174	124	48	N47.07219	W112.61725	15N,8W,9A

Appendix E: Temperature sensor locations in the Blackfoot drainage, 2013.

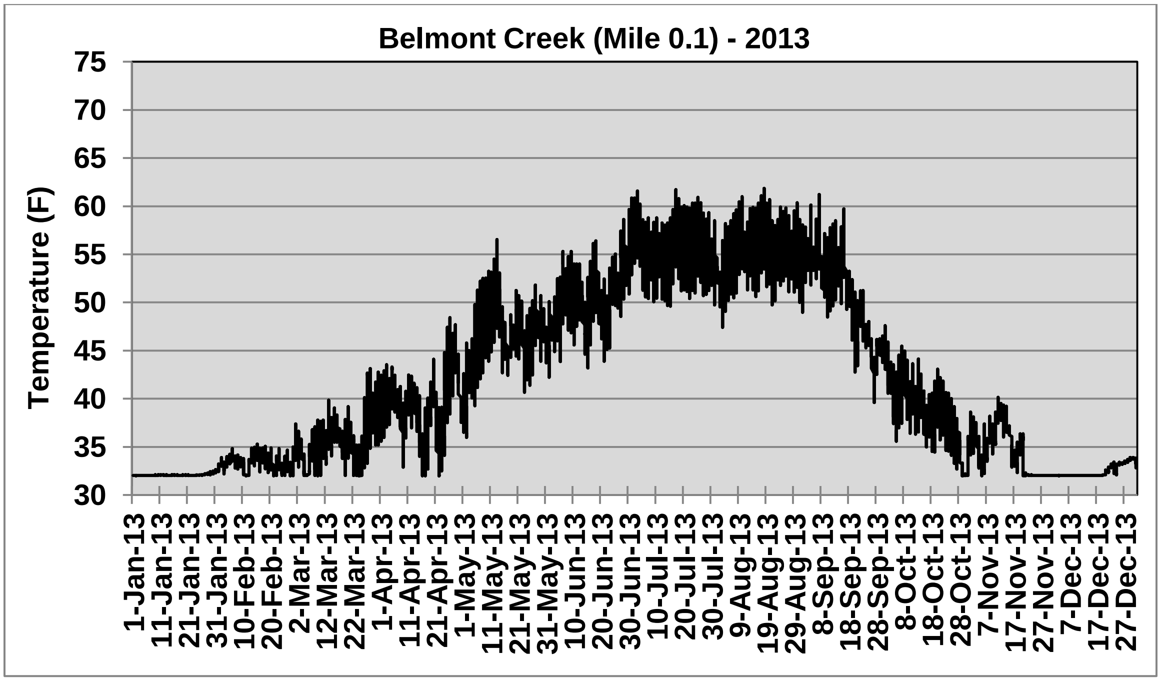
Stream Name	Location (stream mile)	Legal Description	Duration	Sensor Type	Recording Interval
Belmont Creek @ mouth	0.1	14N,16W,24C	1/1/13-12/31/13	Tidbit	50mins
Blackfoot River above Belmont Creek	21.8	14N,16W,24C	1/1/13-12/31/13	Tidbit	50mins
Blackfoot River @ Cuttoff Rd Bridge	72.2	14N,11W,32D	1/1/13-12/31/13	Tidbit	50mins
Blackfoot River @ Dalton Mtn Rd Bridge	104.5	14N,9W,28B	1/1/13-12/31/13	Tidbit	50mins
Blackfoot River @ Raymond Bridge	60	14N,12W,28D	1/1/13-12/31/13	Tidbit	50mins
Blackfoot River @ Scotty Brown Bridge	46.1	15N,13W,33A	1/1/13-12/31/13	Tidbit	50mins
Blackfoot River @ USGS Gage Station	7.9	13N,17W,9B	1/1/13-12/31/13	Tidbit	50mins
Copper Creek @ Sucker Creek Rd Bridge	1.1	15N,8W,25C	1/1/13-12/31/13	Tidbit	50mins
Cottonwood Creek @ Hwy 200	1	15N,13W,29B	1/1/13-12/31/13	Tidbit	50mins
Ender's Spring Creek	0.1	14N,11W,6B	6/26/13-12/31/13	Hobo	72mins
Gold Creek	1.6	14N,16W,30C	1/1/13-12/31/13	Tidbit	50mins
Jacobsen's Spring Creek	0.1	14N,12W,1C	6/26/13-12/31/13	Hobo	72mins
Kleinschmidt Creek	0.3	14N,11W,6A	6/26/13-12/31/13	Tidbit	50mins
Monture Creek @ FAS	1.8	15N,13W,22D	1/1/13-12/31/13	Tidbit	50mins
Nevada Creek above Nevada Spring Creek	6.3	13N,11W,9C	6/26/13-10/22/13	Tidbit	50mins
Nevada Creek below Nevada Spring Creek	5	13N,11W,8D	6/26/13-12/31/13	Tidbit	50mins
Nevada Spring Creek @ mouth	0.1	13N,11W,9C	6/26/13-12/31/13	Tidbit	50mins
North Fork Blackfoot River	2.6	14N,12W,10D	1/1/13-12/31/13	Tidbit	50mins
Un-named Spring Creek to NF Blackfoot River at mile 3.0	0.1	14N,12W,11C	7/31/13-10/8/13	Hobo	72mins
Un-named Spring Creek to NF Blackfoot River at mile 3.7	0.1	14N,12W,11A	7/31/13-10/8/13	Hobo	72mins
Wasson Creek @ mouth	0.1	13N,11W,11D	6/26/13-12/31/13	Tidbit	50mins

Temperature sensor locations in the Blackfoot drainage, 2014.

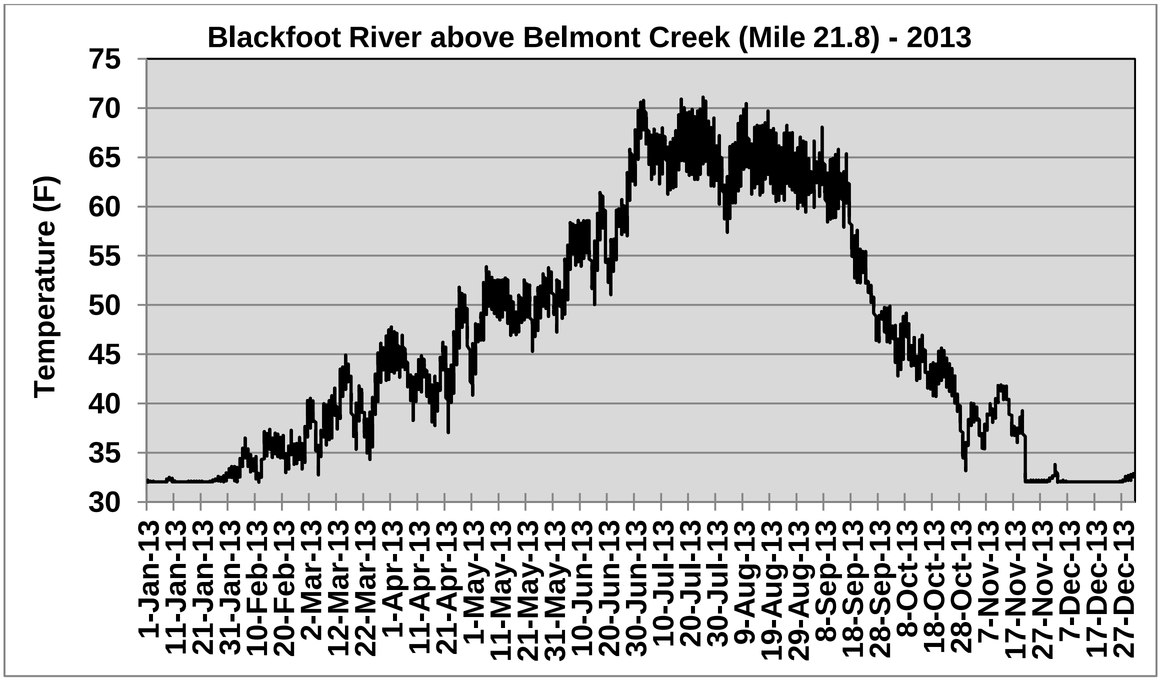
Stream Name	Location (stream mile)	Legal Description	Duration	Sensor Type	Recording Interval
Anaconda Creek	0.3	15N,6W,27A	7/9/14-10/1/14	Hobo	72mins
Beartrap Creek	0.2	15N,6W,27B	7/9/14-10/1/14	Hobo	72mins
Belmont Creek @ mouth	0.1	14N,16W,24C	1/1/14-12/31/14	Tidbit	50mins
Blackfoot River above Belmont Creek	21.8	14N,16W,24C	1/1/14-12/31/14	Tidbit	50mins
Blackfoot River @ Cuttoff Rd Bridge	72.2	14N,11W,32D	1/1/14-12/31/14	Tidbit	50mins
Blackfoot River @ Dalton Mtn Rd Bridge	104.5	14N,9W,28B	1/1/14-12/31/14	Tidbit	50mins
Blackfoot River @ Raymond Bridge	60	14N,12W,28D	1/1/14-12/31/14	Tidbit	50mins
Blackfoot River @ Scotty Brown Bridge	46.1	15N,13W,33A	1/1/14-12/31/14	Tidbit	50mins
Blackfoot River above Shave Creek	131.8	15N,6W,21C	7/9/14-10/1/14	Hobo	72mins
Blackfoot River @ USGS Gage Station	7.9	13N,17W,9B	1/1/14-12/31/14	Tidbit	50mins
Copper Creek @ Sucker Creek Rd Bridge	1.1	15N,8W,25C	1/1/14-12/31/14	Tidbit	50mins
Cottonwood Creek @ Hwy 200	1	15N,13W,29B	1/1/14-12/31/14	Tidbit	50mins
East Fork Warren Creek	1.6	15N,11W,19B	7/1/14-10/6/14	Hobo	72mins
Ender's Spring Creek	0.1	14N,11W,6B	1/1/14-10/1/14	Hobo	72mins
Gold Creek	1.6	14N,16W,30C	1/1/14-12/31/14	Tidbit	50mins
Hoyt Creek	1.2	15N,12W,19C	7/1/14-10/1/14	Hobo	72mins
Hoyt Creek	4.3	15N,12W,28C	7/1/14-10/1/14	Hobo	72mins
Jacobsen's Spring Creek	0.1	14N,12W,1C	1/1/14-10/1/14	Hobo	72mins
Kleinschmidt Creek	0.1	14N,11W,6A	1/1/14-12/31/14	Tidbit	50mins
Monture Creek @ FAS	1.8	15N,13W,22D	1/1/14-12/31/14	Tidbit	50mins
Murphy Irrigation Ditch	0.1	15N,11W,19B	7/1/14-10/1/14	Hobo	72mins
Nevada Creek above Nevada Spring Creek	6.3	13N,11W,9C	4/9/14-12/31/14	Tidbit	50mins
Nevada Creek below Nevada Spring Creek	5	13N,11W,8D	1/1/14-12/31/14	Tidbit	50mins
Nevada Spring Creek @ mouth	0.1	13N,11W,9C	1/1/14-12/31/14	Tidbit	50mins
North Fork Blackfoot River	2.6	14N,12W,10D	1/1/14-12/31/14	Tidbit	50mins
Wasson Creek @ mouth	0.1	13N,11W,11D	1/1/14-12/31/14	Tidbit	50mins

Appendix E: Temperature sensor locations in the Blackfoot drainage, 2015.

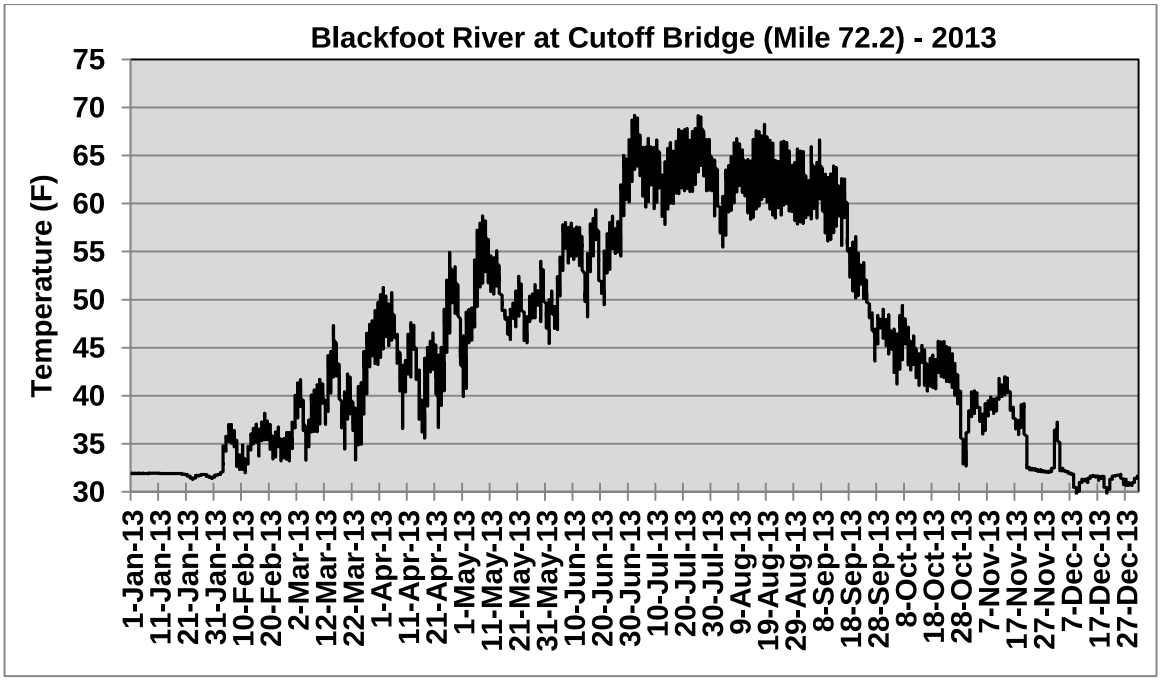
Stream Name	Location (stream mile)	Legal Description	Duration	Sensor Type	Recording Interval
Belmont Creek @ mouth	0.1	14N,16W,24C	1/1/15-10/7/15	Tidbit	50mins
Blackfoot River above Belmont Creek	21.8	14N,16W,24C	1/1/15-10/7/15	Tidbit	50mins
Blackfoot River @ Cuttoff Rd Bridge	72.2	14N,11W,32D	1/1/15-10/7/15	Tidbit	50mins
Blackfoot River @ Dalton Mtn Rd Bridge	104.5	14N,9W,28B	1/1/15-10/7/15	Tidbit	50mins
Blackfoot River @ Raymond Bridge	60	14N,12W,28D	1/1/15-10/7/15	Tidbit	50mins
Blackfoot River @ Scotty Brown Bridge	46.1	15N,13W,33A	1/1/15-10/7/15	Tidbit	50mins
Blackfoot River @ USGS Gage Station	7.9	13N,17W,9B	1/1/15-10/7/15	Tidbit	50mins
Copper Creek @ Sucker Creek Rd Bridge	1.1	15N,8W,25C	1/1/15-10/7/15	Tidbit	50mins
Cottonwood Creek @ Hwy 200	1	15N,13W,29B	1/1/15-10/7/15	Tidbit	50mins
Gold Creek	1.6	14N,16W,30C	1/1/15-10/7/15	Tidbit	50mins
Kleinschmidt Creek	0.1	14N,11W,6A	1/1/15-10/7/15	Tidbit	50mins
Monture Creek @ FAS	1.8	15N,13W,22D	1/1/15-10/7/15	Tidbit	50mins
Nevada Creek above Nevada Spring Creek	6.3	13N,11W,9C	1/1/15-9/23/15	Tidbit	50mins
Nevada Creek below Nevada Spring Creek	5	13N,11W,8D	1/1/15-9/23/15	Tidbit	50mins
Nevada Spring Creek @ mouth	0.1	13N,11W,9C	1/1/15-9/24/15	Tidbit	50mins
North Fork Blackfoot River	2.6	14N,12W,10D	1/1/15-10/7/15	Tidbit	50mins
Warren Creek @lower bridge	1.1	15N,12W,31C	6/22/15-10/7/15	Hobo	50mins
Warren Creek @ middle bridge	2.1	15N,12W,31A	6/22/15-10/7/15	Hobo	50mins
Wetland outlet into Warren Creek	1.8	15N,12W,31A	6/22/15-10/7/15	Hobo	50mins
Wasson Creek @ mouth	0.1	13N,11W,11D	1/1/15-9/23/15	Tidbit	50mins



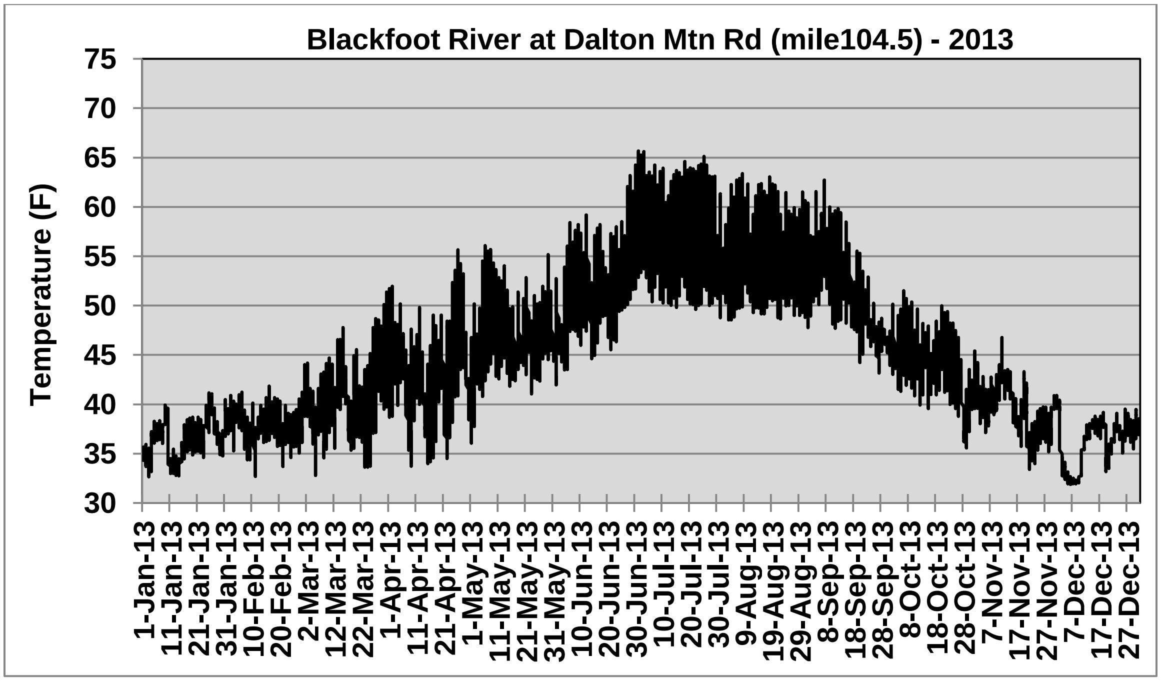
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	32.6	32	32.1	0.1	0.0
February	35.3	32	33.2	0.8	0.6
March	43.1	32	35.1	2.4	5.6
April	48.4	32	39.2	3.3	10.9
May	56.5	36	46.6	3.5	12.5
June	59.7	42.2	50.4	3.2	10.1
July	61.7	49.6	55.2	2.8	7.6
August	61.8	47.4	54.7	2.8	8.1
September	61.2	39.6	50.6	4.5	20.1
October	47.6	32	38.4	3.4	11.2
November	40.2	32	34.5	2.3	5.4
December	33.9	32	32.5	0.6	0.4



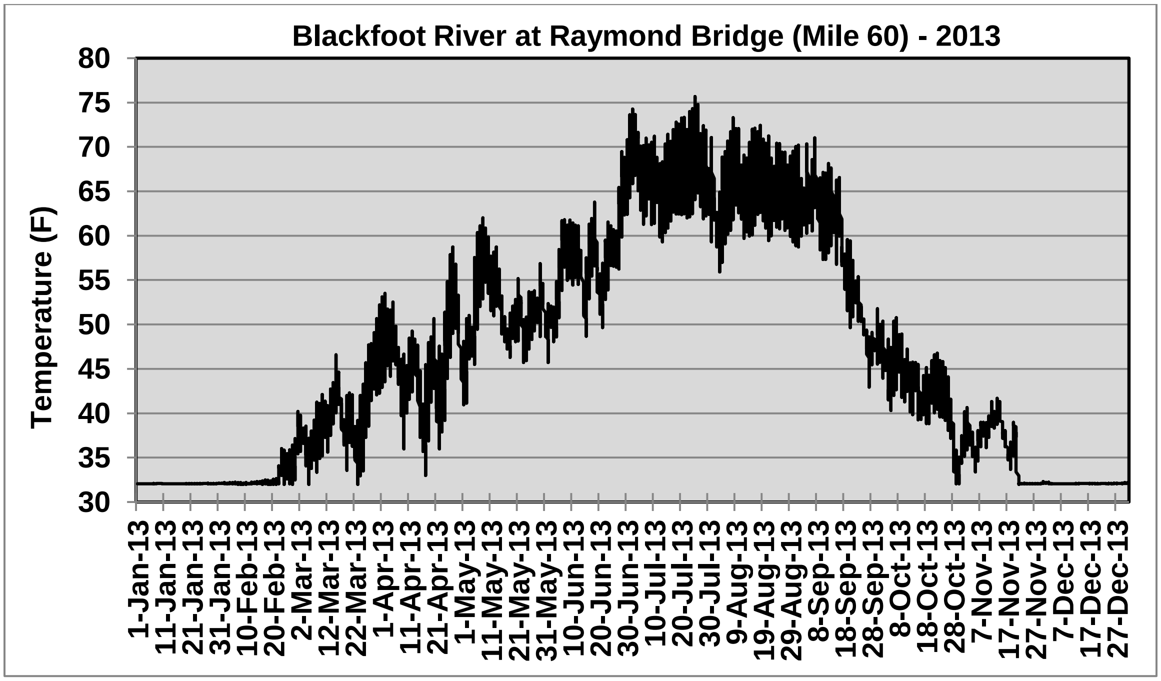
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	33.4	32.0	32.1	0.2	0.0
February	37.7	32.0	34.5	1.3	1.7
March	47.5	32.7	40.0	3.0	9.2
April	51.8	37.0	43.9	3.0	9.0
May	53.9	40.9	49.6	2.3	5.5
June	67.8	47.2	56.4	4.0	16.1
July	71.1	60.3	66.3	2.2	4.9
August	70.5	57.4	64.1	2.4	6.0
September	68.1	46.3	57.7	5.8	33.2
October	49.9	33.2	43.3	3.4	11.6
November	41.9	32.0	36.5	3.4	11.2
December	33.8	32.0	32.2	0.3	0.1



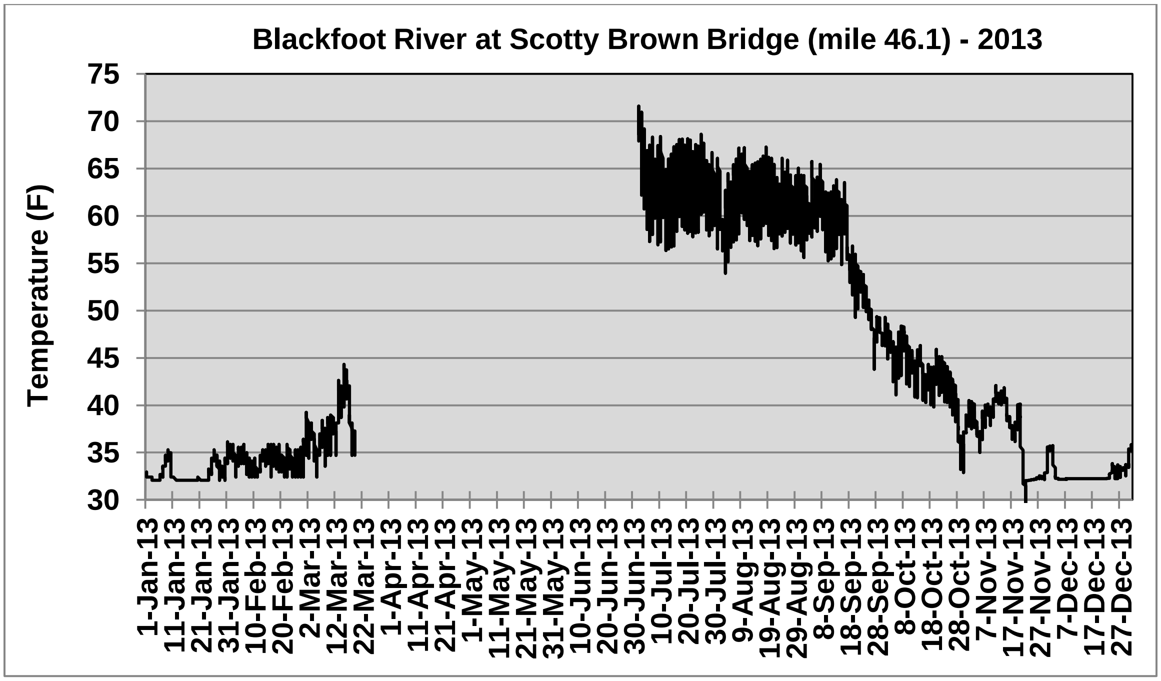
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	31.9	31.3	31.7	0.2	0.02
February	38.2	31.7	37.8	1.5	2.3
March	49.7	33.3	40.2	3.4	11.8
April	54.9	35.6	45	4	16.2
May	58.7	40	50	3.3	11.1
June	66.6	45.4	54.9	4.1	16.6
July	69.2	57.9	64	2.4	5.7
August	68.2	55.5	62.3	2.6	6.6
September	66.6	43.6	56.1	5.8	33.8
October	49.4	32.7	43	3.3	11
November	42	32	36.7	3.3	10.7
December	37.2	29.6	31.6	1.3	1.6



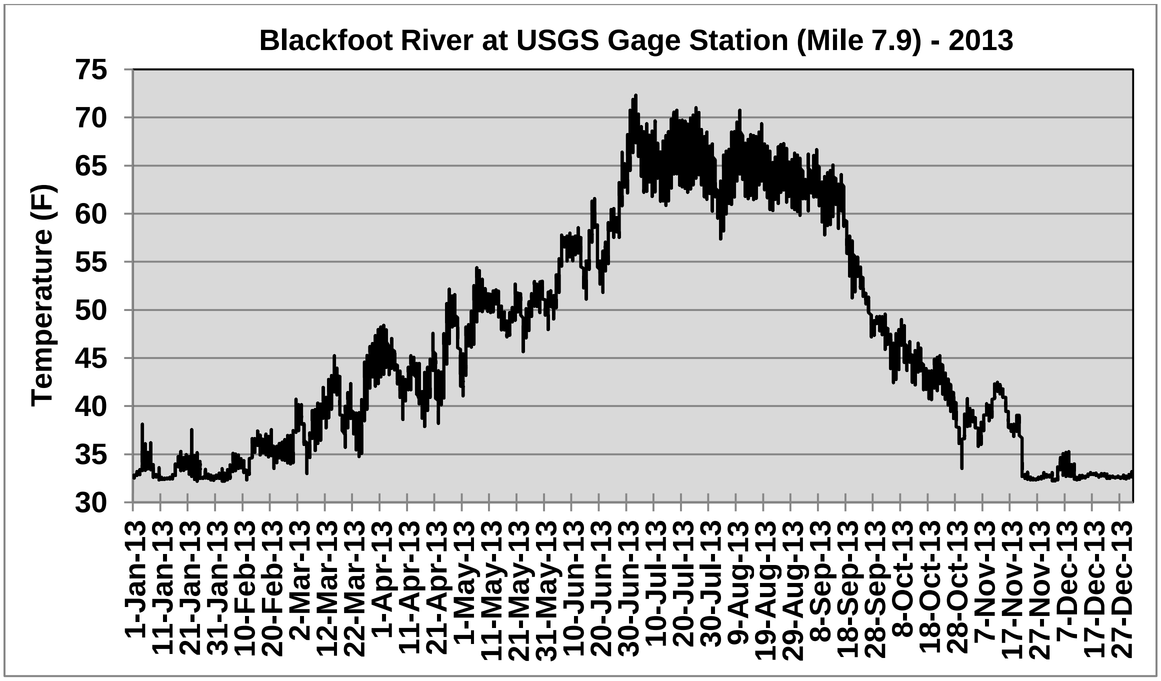
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	41.1	32.6	36.4	1.9	3.8
February	41.8	32.7	37.7	1.7	2.8
March	51.4	32.8	40.4	3.5	12.1
April	55.6	33.7	43	4.4	19.3
May	56.1	36.1	46.9	3.6	13.2
June	64.2	42	51.7	4.4	19.4
July	66	49	57	4.4	19.2
August	63.3	48.5	55	3.9	15.2
September	62.7	43.2	51.3	4.2	17.6
October	51.5	35.6	43.8	3.1	9.5
November	46.8	33.4	39.5	2.4	5.8
December	41	31.9	36.1	2.4	5.9



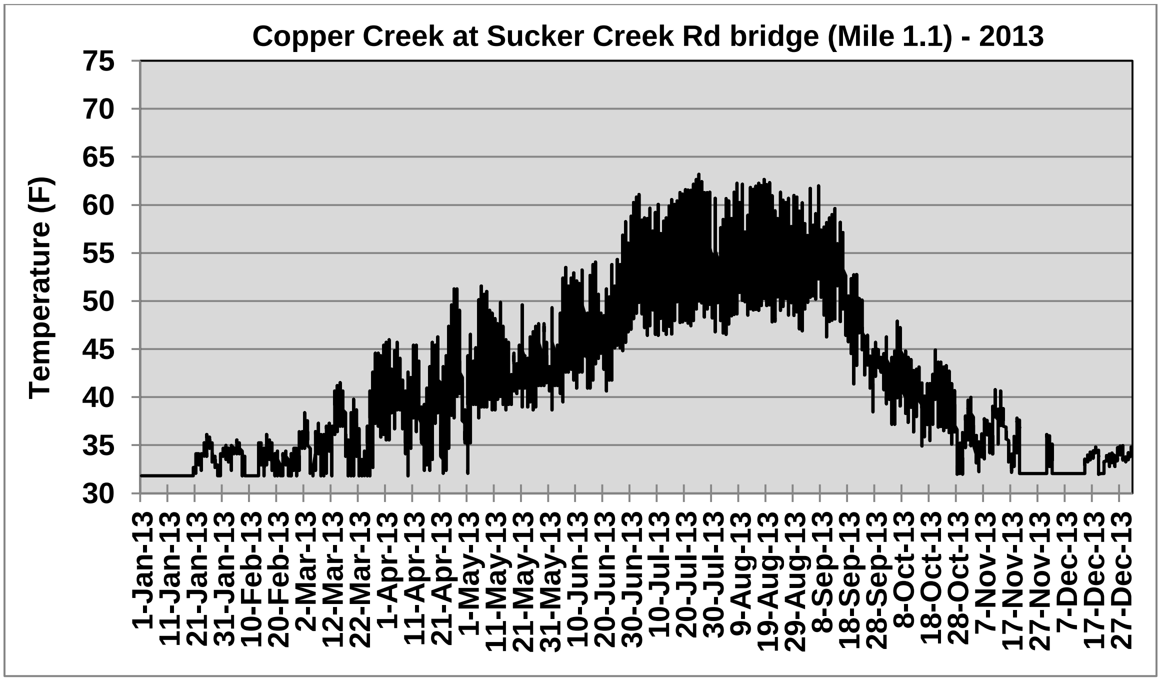
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	32.1	32.0	32.0	0.0	0.0
February	37.1	32.0	32.5	1.0	1.0
March	52.2	32.0	39.3	3.9	15.1
April	58.7	33.0	45.5	4.8	23.0
May	62.0	41.0	51.5	4.0	15.6
June	70.8	45.7	57.0	4.8	22.8
July	75.7	59.3	67.1	3.6	13.0
August	73.3	56.0	65.1	3.6	12.8
September	71.0	43.0	58.0	6.9	47.6
October	50.7	32.0	42.5	4.1	16.4
November	41.7	32.0	35.7	2.9	8.7
December	32.2	32.0	32.1	0.0	0.0



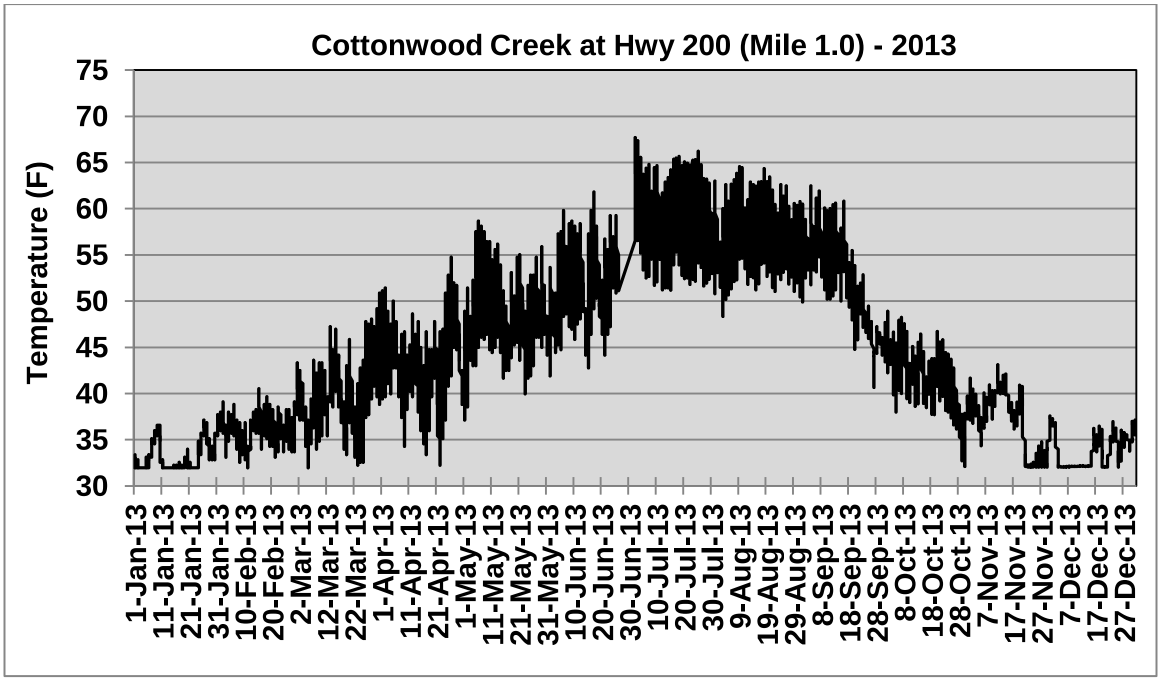
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	36.1	32.1	32.7	0.9	0.9
February	36.4	32.4	34	1	1.1
March	44.3	32.4	37.6	2.5	6.5
April	ND	ND	ND	ND	ND
May	ND	ND	ND	ND	ND
June	ND	ND	ND	ND	ND
July	71.6	56.3	63.1	3.4	11.5
August	67.2	54	61.4	3	8.4
September	65.7	43.8	56	5.4	28.9
October	49.3	32.9	43	3.3	11
November	42.1	29.5	36.7	3.3	11.2
December	35.8	32.2	32.8	1	1.02



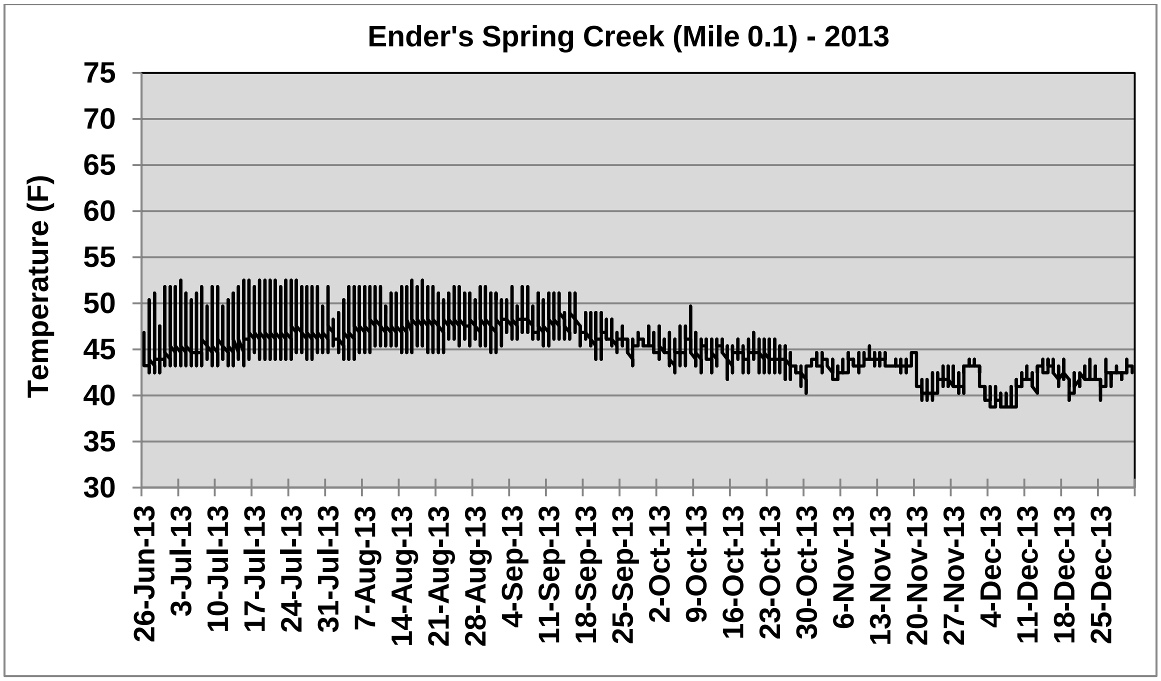
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	38.1	32.2	33.2	0.9	0.7
February	37.6	32.2	34.7	1.4	2.0
March	48.0	33.0	39.8	3.0	8.8
April	52.2	37.9	44.1	3.0	8.8
May	54.4	41.1	49.7	2.3	5.1
June	68.2	48.0	56.6	3.9	14.8
July	72.3	60.3	66.2	2.6	6.8
August	70.8	57.4	64.4	2.4	5.8
September	66.6	47.2	57.9	5.6	31.5
October	49.6	33.5	43.5	3.3	10.7
November	42.5	32.3	37.0	3.3	11.0
December	35.2	32.2	32.9	0.5	0.3



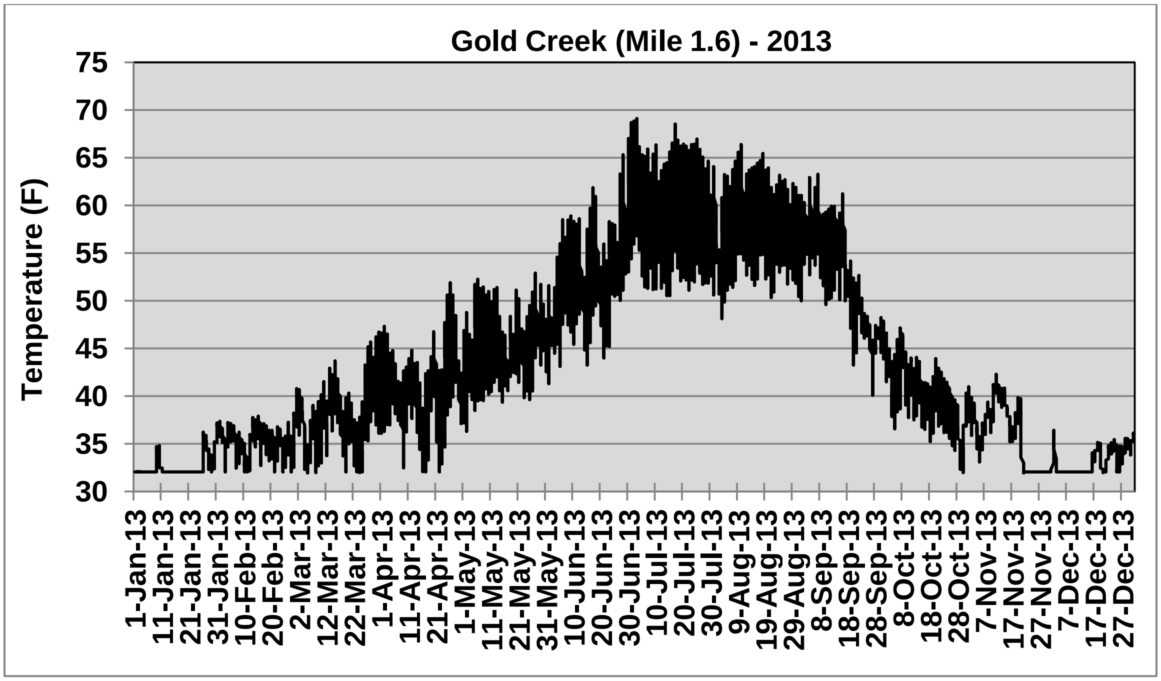
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	36.1	31.8	32.4	1.1	1.2
February	36.4	31.8	33.2	1.3	1.6
March	45.4	31.8	35.8	3.2	10.0
April	51.3	31.8	39.5	4.0	15.8
May	51.6	32.1	42.9	3.2	10.2
June	58.8	38.7	47.2	4.1	17.1
July	63.2	46.4	54.1	4.4	19.2
August	62.7	46.5	54.4	4.2	17.5
September	62.0	38.5	50.0	5.2	27.2
October	47.9	32.0	39.6	3.3	10.6
November	40.8	32.0	34.7	2.4	5.9
December	36.0	32.0	33.1	1.0	1.0



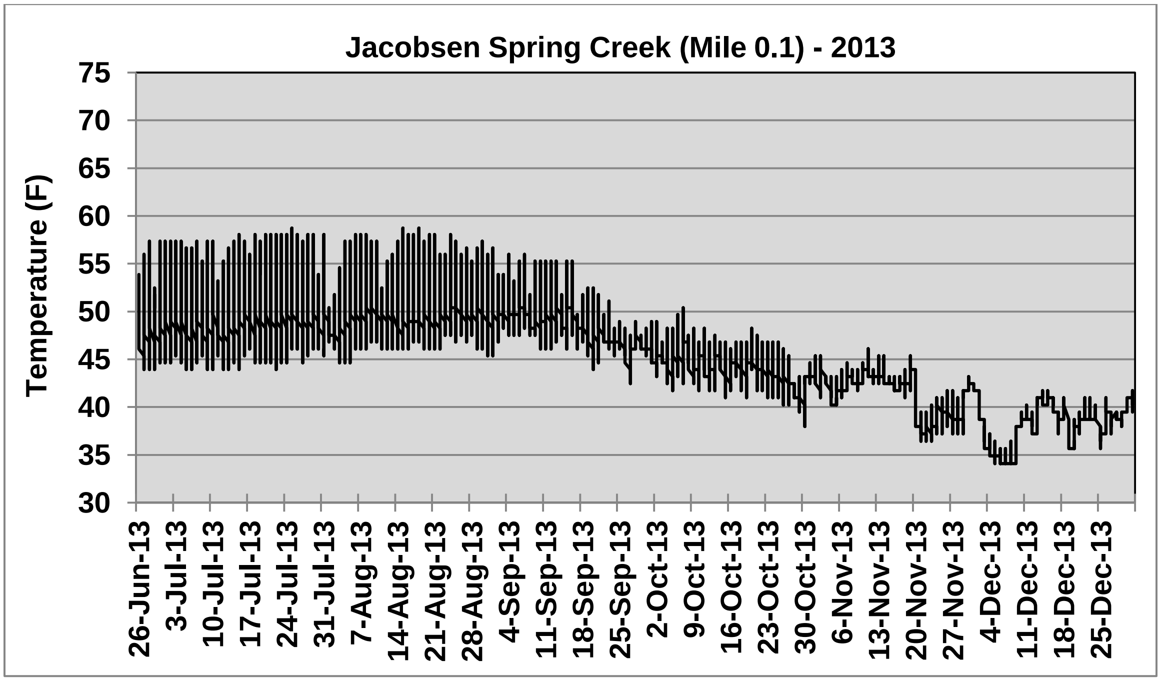
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.7	32.0	33.1	1.6	2.5
February	40.5	32.0	36.0	1.6	2.5
March	50.9	32.0	39.8	3.8	14.2
April	54.8	32.3	43.2	4.1	17.1
May	58.7	37.1	48.4	4.1	17.1
June	61.8	41.9	51.5	4.2	17.9
July	67.7	50.8	58.7	4.1	16.9
August	64.5	48.4	56.9	3.5	12.0
September	62.5	40.7	52.2	4.8	22.7
October	48.9	32.1	41.6	3.2	10.6
November	43.1	32	36.9	3.2	10.2
December	37.3	32	33.7	1.7	3



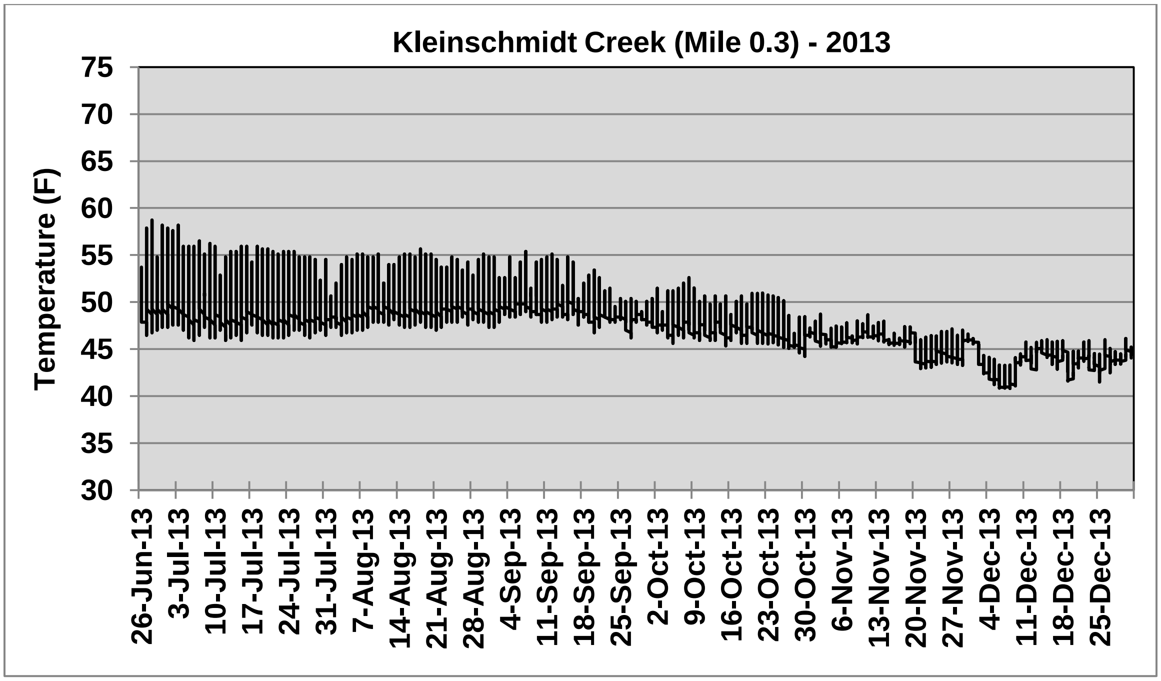
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	51.8	42.5	45.6	2.7	7.5
July	52.5	43.2	47.2	3.0	8.8
August	52.5	43.9	48.0	2.3	5.4
September	51.8	43.2	47.2	1.8	3.3
October	49.7	40.2	44.3	1.4	2.0
November	45.4	39.5	42.8	1.3	1.6
December	43.9	38.7	41.7	1.4	2.0



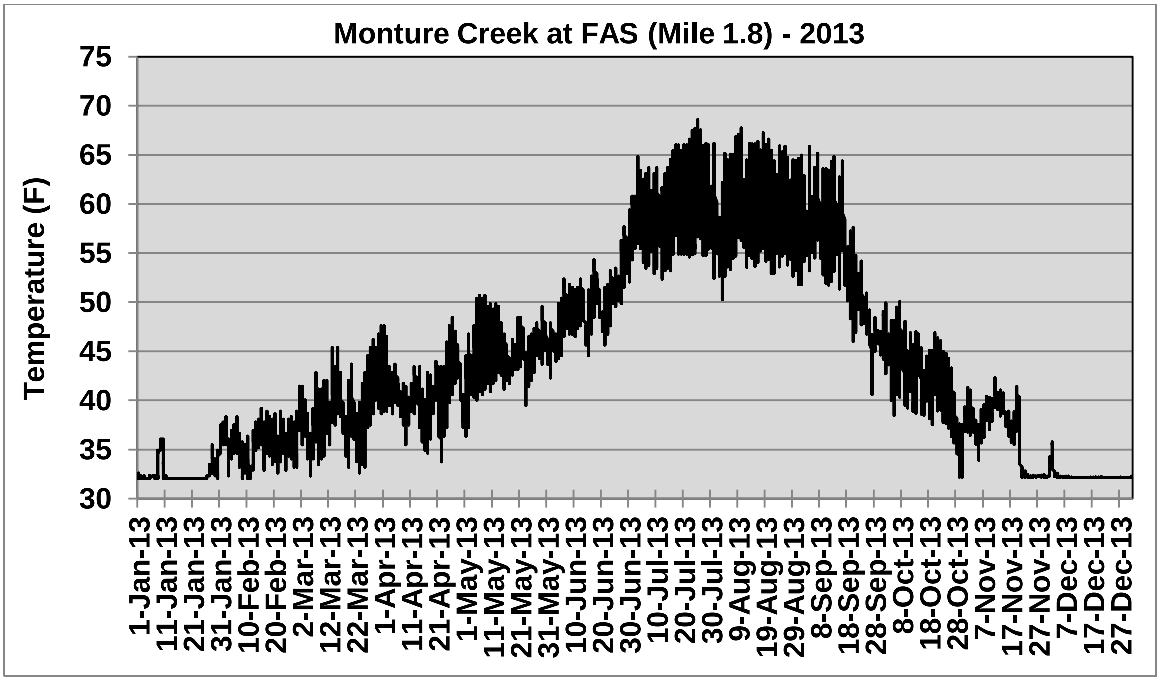
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.2	32.0	32.5	1.1	1.2
February	38.2	32.0	35.0	1.5	2.3
March	46.7	31.9	37.6	3.2	10.3
April	51.9	32.0	40.5	3.8	14.6
May	52.9	36.3	44.8	3.4	11.6
June	67.0	41.3	52.3	4.8	23.4
July	69.1	50.5	59.4	4.8	23.2
August	66.4	48.1	57.8	4.0	16.0
September	63.3	40.1	52.6	5.1	25.9
October	47.9	32.0	40.3	3.3	10.8
November	42.3	31.9	36.0	3.2	10.3
December	36.4	32.0	33.1	1.3	1.7



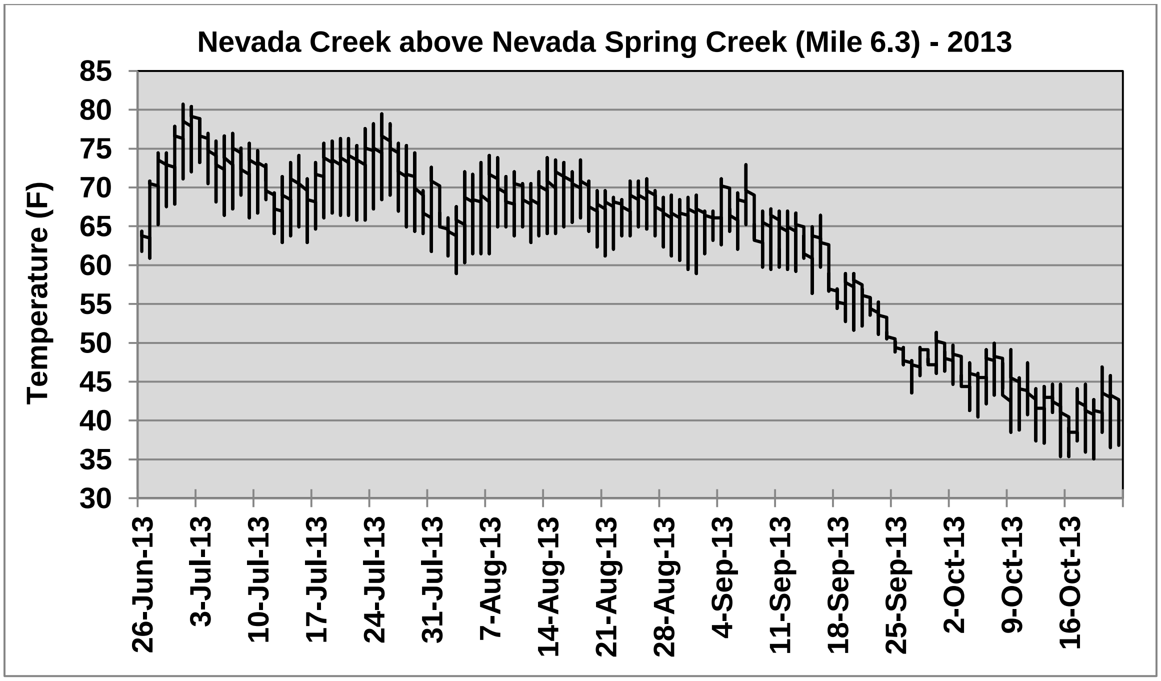
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	57.4	43.9	49	4.1	17
July	58.7	43.9	50.0	4.3	18.2
August	58.7	44.7	50.4	3.6	12.8
September	56.7	42.5	48.8	2.7	7.5
October	50.4	38	44.1	2.2	4.7
November	46.1	36.4	41.6	2.2	4.9
December	42.5	34.1	38.5	2.2	4.7



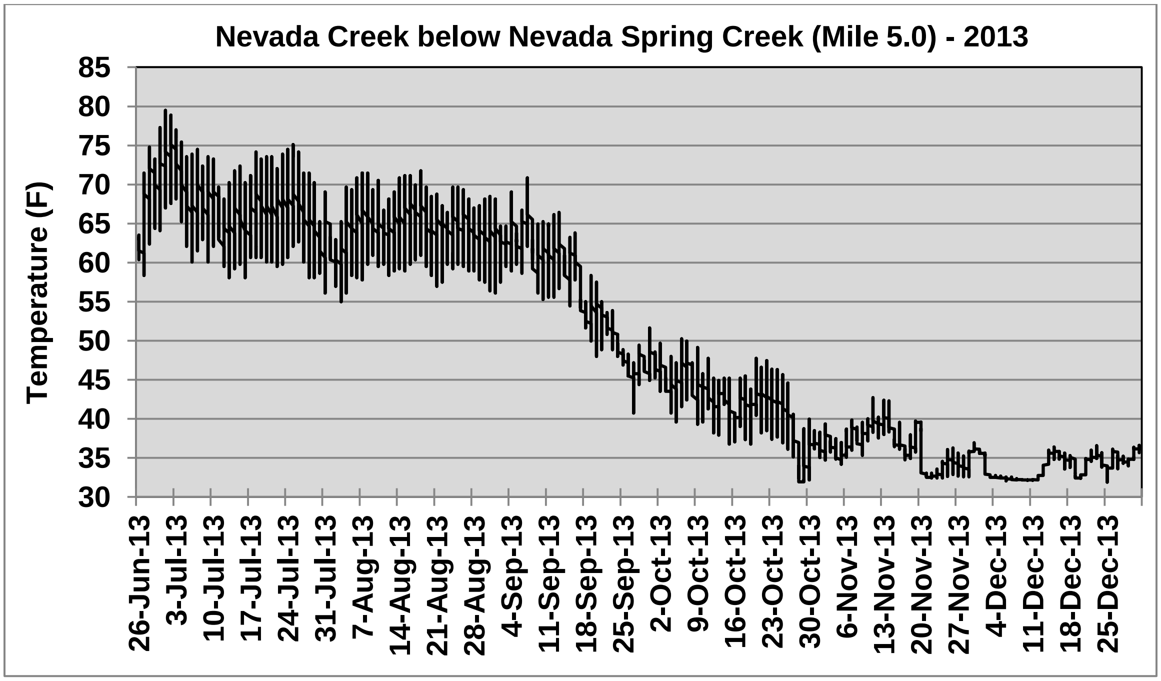
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	58.7	46.5	51.1	3.6	12.6
July	58.2	46	50.2	3.1	9.8
August	55.7	46.5	50.2	2.4	5.8
September	55.4	46.2	50	1.9	3.6
October	52.6	44.2	47.5	1.6	2.6
November	48.7	42.9	45.8	1.2	1.4
December	46.1	40.8	43.8	1.3	1.6



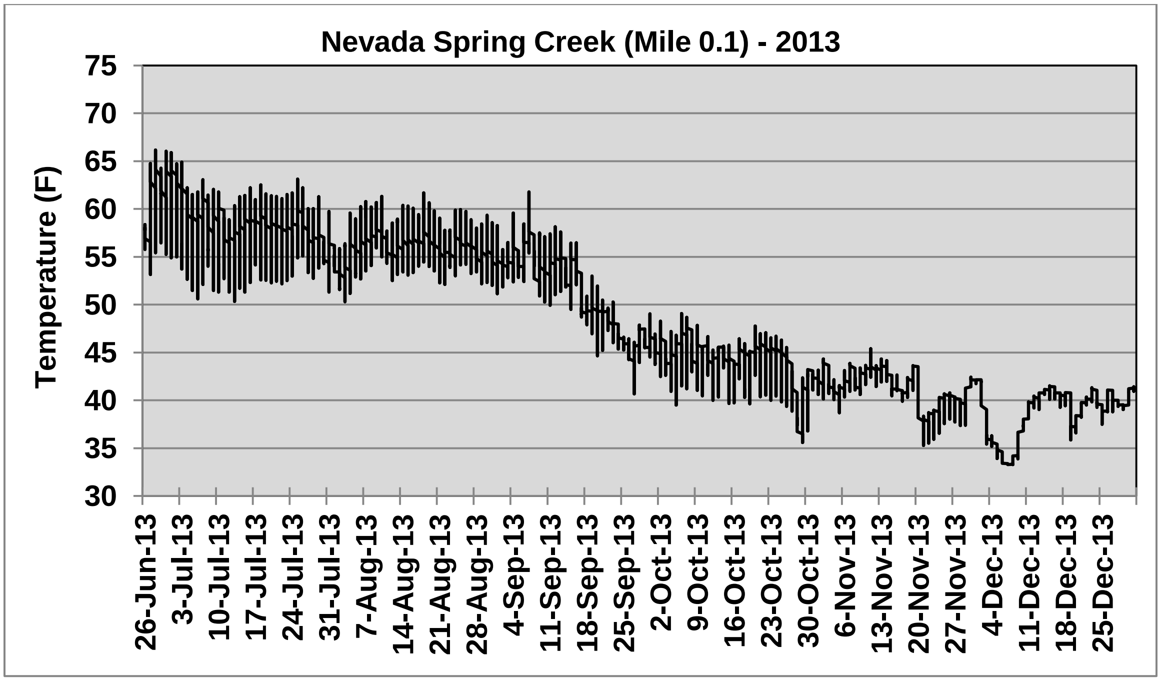
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.5	32.0	32.6	1.1	1.2
February	39.2	32.0	35.4	1.7	2.8
March	47.6	32.3	38.7	3.2	10.1
April	48.4	33.8	40.8	2.7	7.5
May	50.7	36.4	44.7	2.7	7.1
June	59.4	42.3	49.6	3.1	9.5
July	68.6	52.4	59.6	3.7	13.4
August	67.7	50.3	59.3	4.0	15.7
September	65.8	40.6	54.1	5.6	31.8
October	50.0	32.2	42.0	3.7	13.5
November	42.3	32.1	36.3	3.1	9.7
December	35.8	32.1	32.3	0.5	0.3



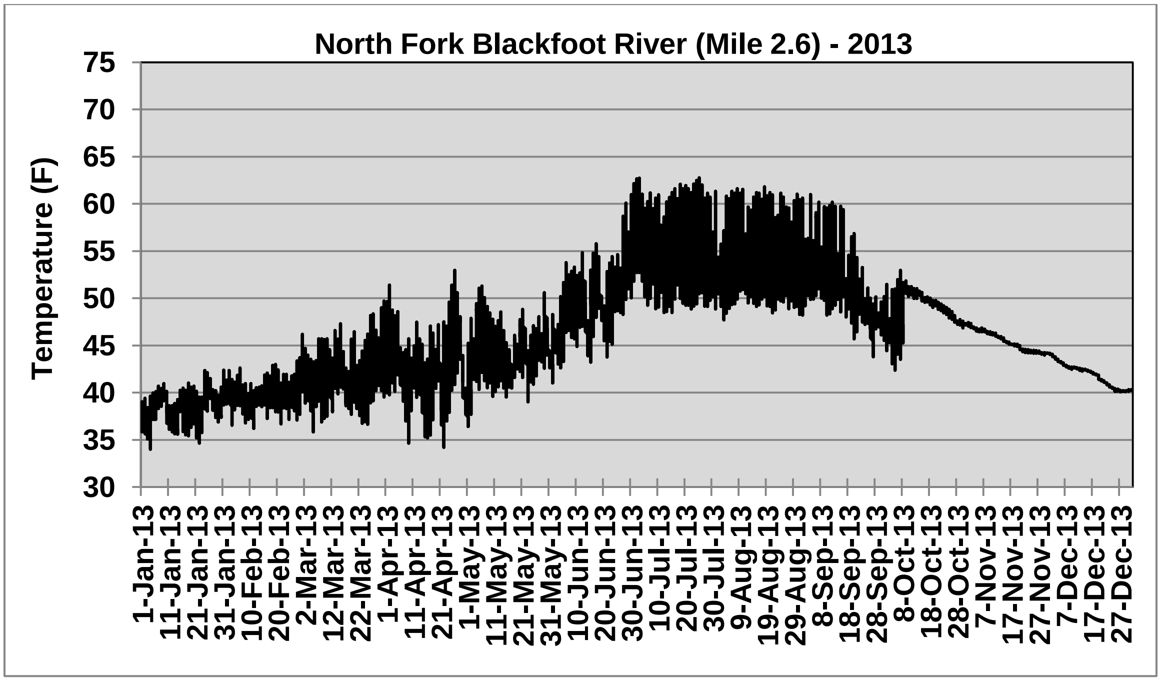
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	78	61	69	4.6	20.7
July	80.7	61.8	71.1	4	16.2
August	74.1	59	67	3.2	10.3
September	73	43.6	59	7.1	50.4
October	50	35.1	43	3.6	12.9



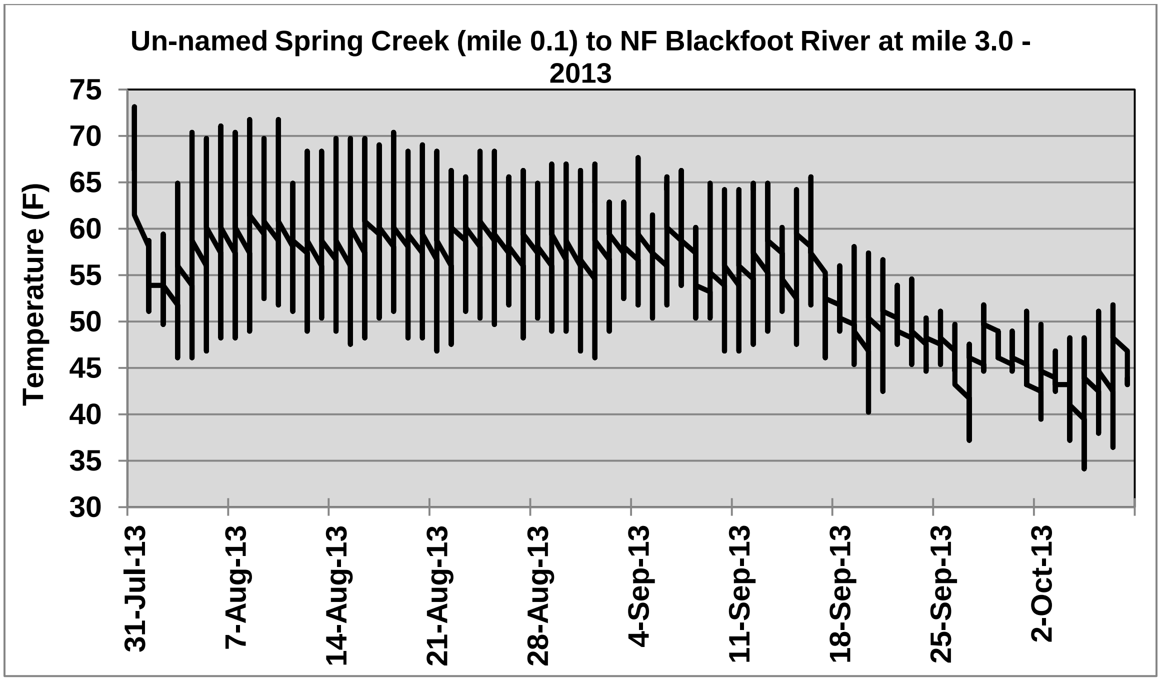
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	77.3	58.4	67.6	4.9	23.6
July	80	56.1	67	4.9	23.6
August	72	55	63.7	3.8	14.4
September	71	41	56.1	6.7	44.9
October	50.3	31.9	41.8	4	16
November	42.7	32.4	36.3	2.4	5.6
December	36.6	31.9	34	1.4	2



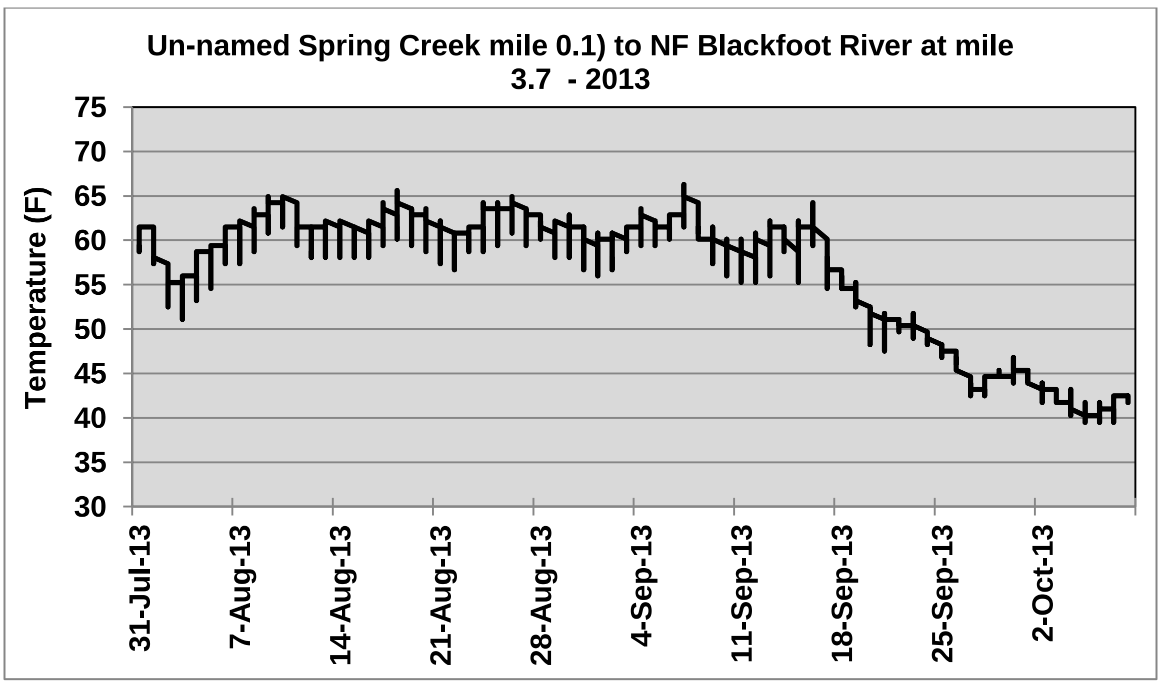
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	66.2	53.2	60.2	3.7	13.5
July	65.9	50.4	57.5	3.3	10.6
August	61.7	50.3	56.0	2.3	5.3
September	61.8	40.7	51.2	4.2	17.6
October	49.1	35.6	43.4	2.6	6.9
November	45.4	35.3	40.8	2.1	4.2
December	42.1	33.3	38.6	2.5	6.4



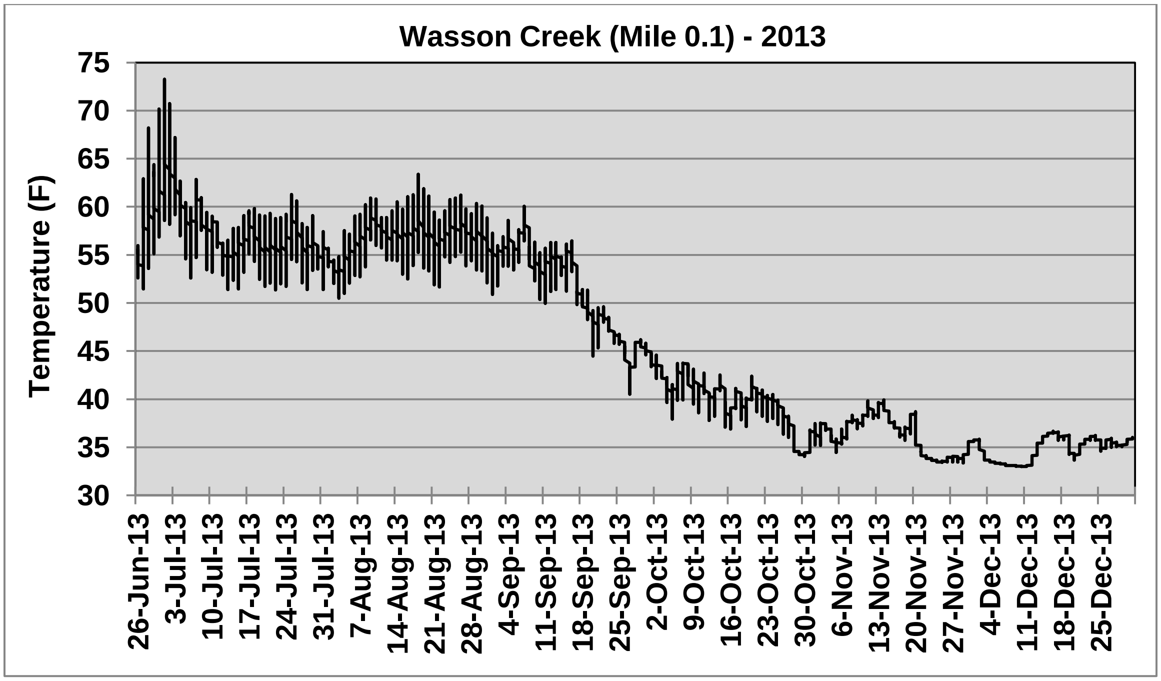
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	42.4	34.0	38.1	1.6	2.7
February	43.7	36.2	39.4	1.4	2.0
March	49.7	35.8	41.2	2.6	6.8
April	52.9	34.2	42.2	3.5	12.4
May	51.3	36.4	44.1	2.6	6.9
June	61.0	41.0	49.4	3.6	13.3
July	62.7	48.5	55.0	3.9	15.2
August	61.8	47.7	53.9	3.7	13.4
September	61.0	43.8	51.2	3.6	12.7
October	52.9	42.4	48.6	1.9	3.8
November	47.4	44	45.4	1.01	1.02
December	44.2	40.1	41.9	1.2	1.5



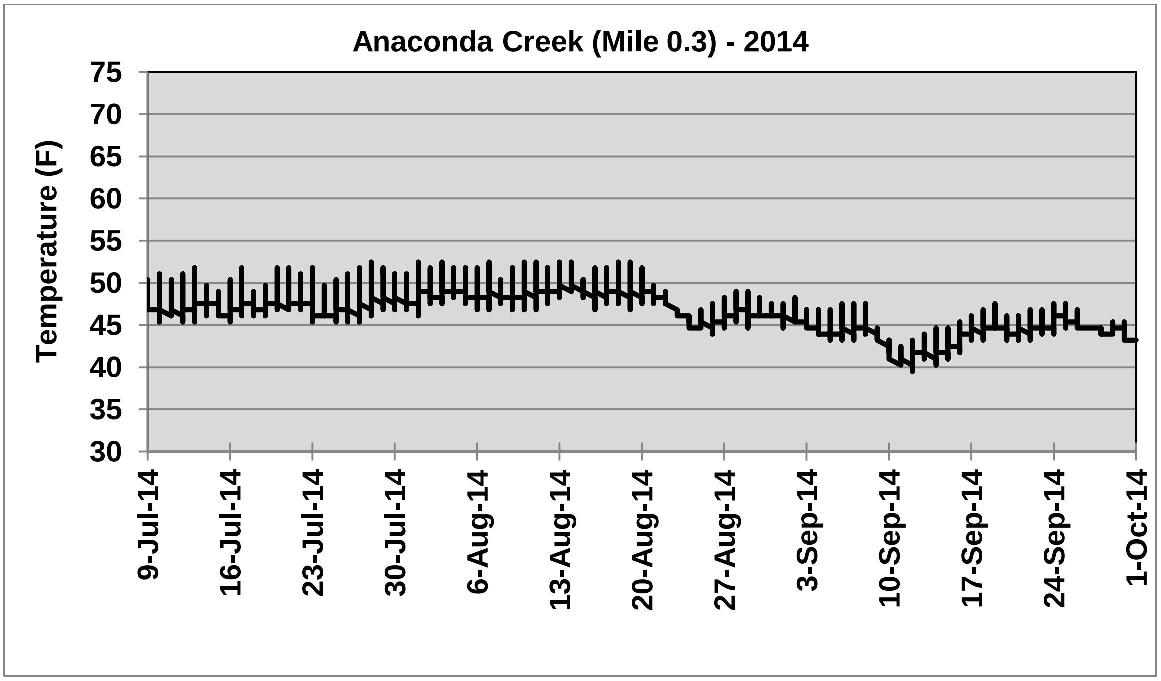
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	73.2	61.5	68.3	3.9	14.9
August	72	46.1	58	6.7	46
September	68	37.2	53	6.2	39
October	52	34.1	44	4.2	17.4



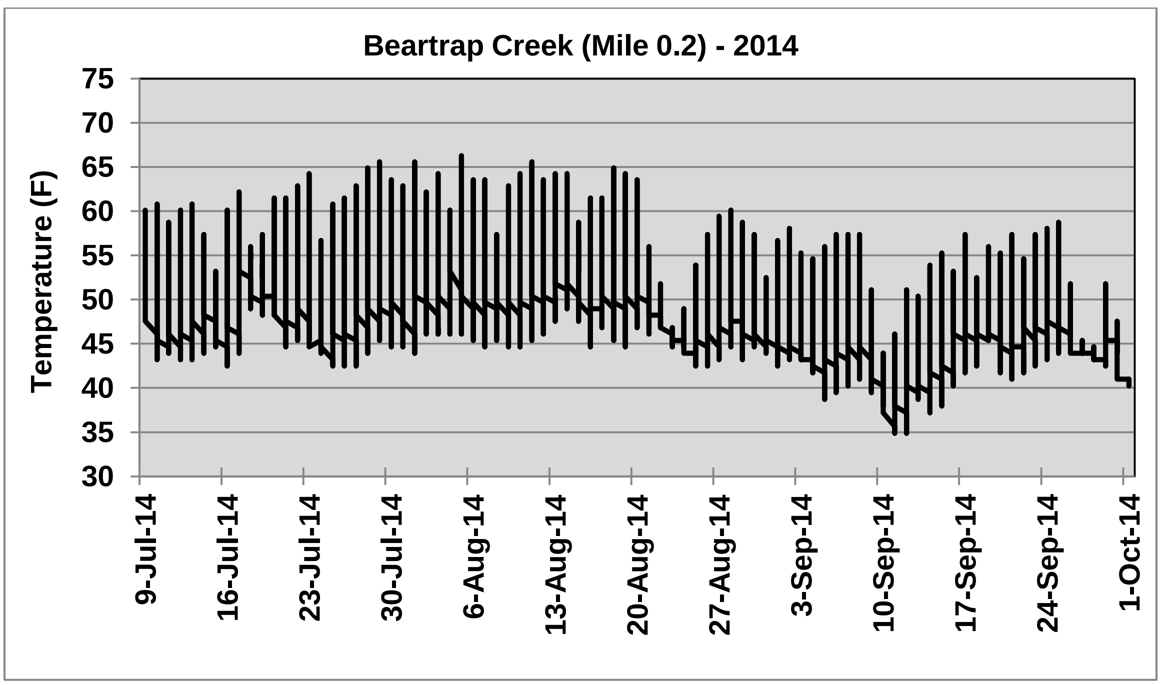
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	61.5	59	60.3	1	0.92
August	66	51.1	60.2	2.7	7.1
September	66.3	42.5	55	6.3	40.2
October	45.4	39.5	42	1.5	2.1



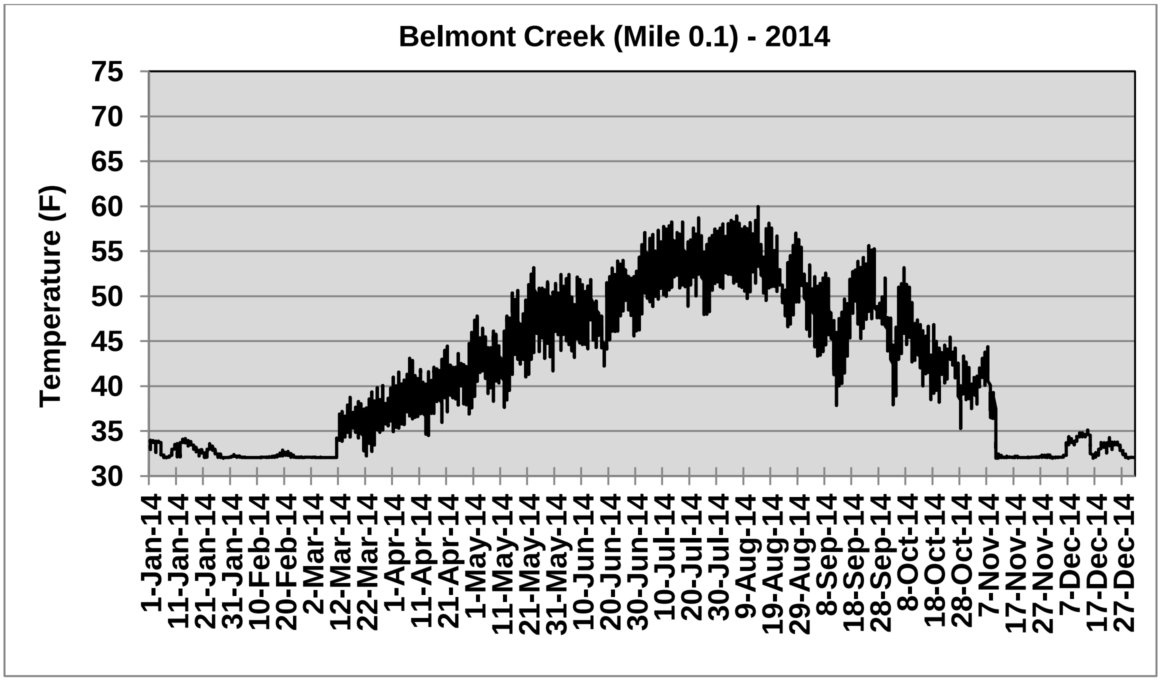
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	70.2	51.5	58.9	4.6	20.9
July	73.3	51.4	57.0	3.5	12.3
August	63.4	50.5	56.5	2.5	6.2
September	60.0	40.5	51.1	4.4	19.4
October	44.9	34.1	39.7	2.6	6.7
November	39.9	33.4	36.1	1.9	3.6
December	36.6	33	34.8	1.2	1.4



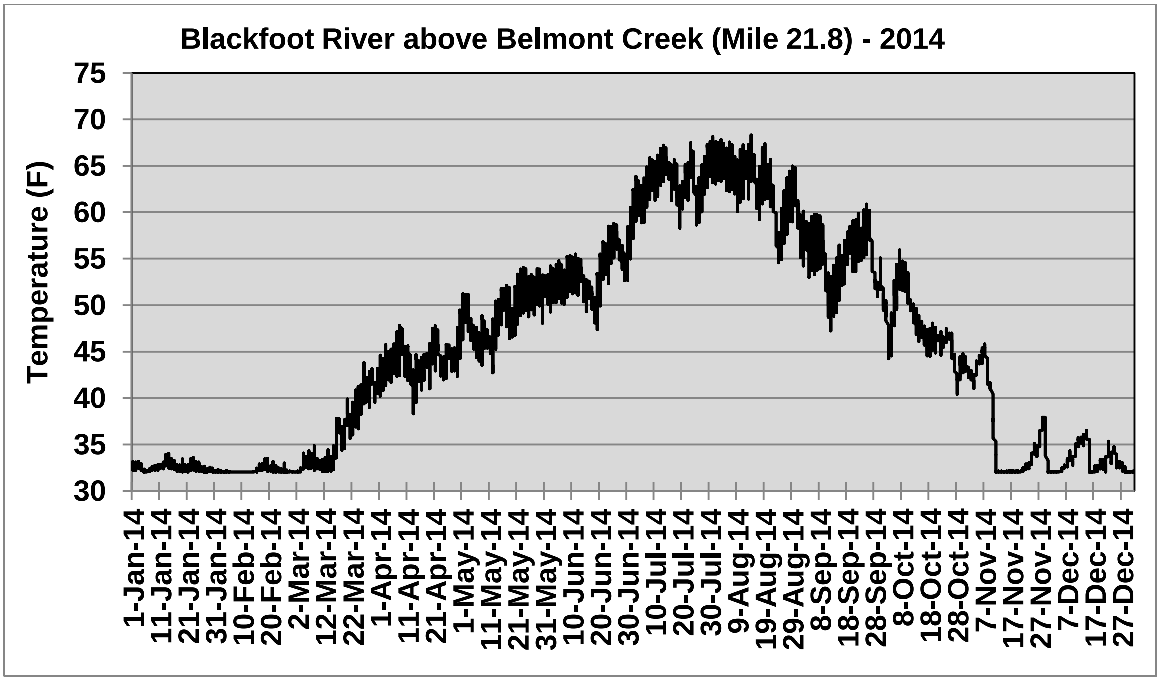
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	52.5	45.4	48	1.7	2.9
August	52.5	43.9	48.4	2	3.8
September	48.3	39.5	44.3	1.7	3



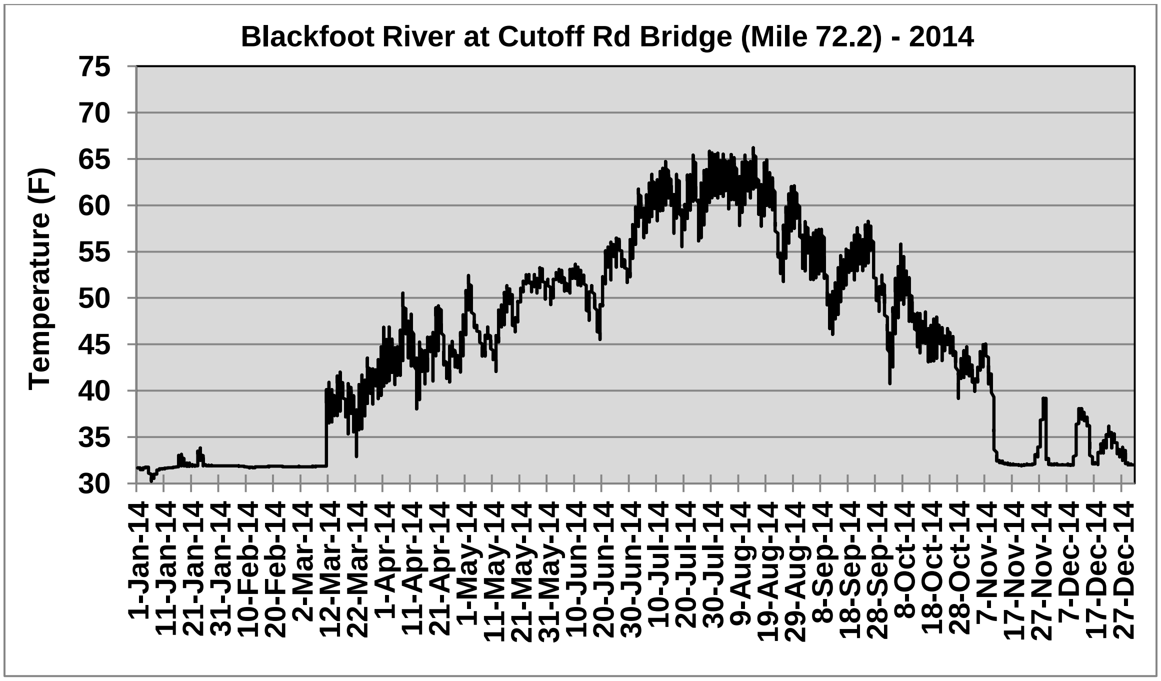
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	65.6	42.5	50.5	5.8	34
August	66.3	42.5	50.9	5.8	33.4
September	58.7	34.9	45.5	5.1	25.6



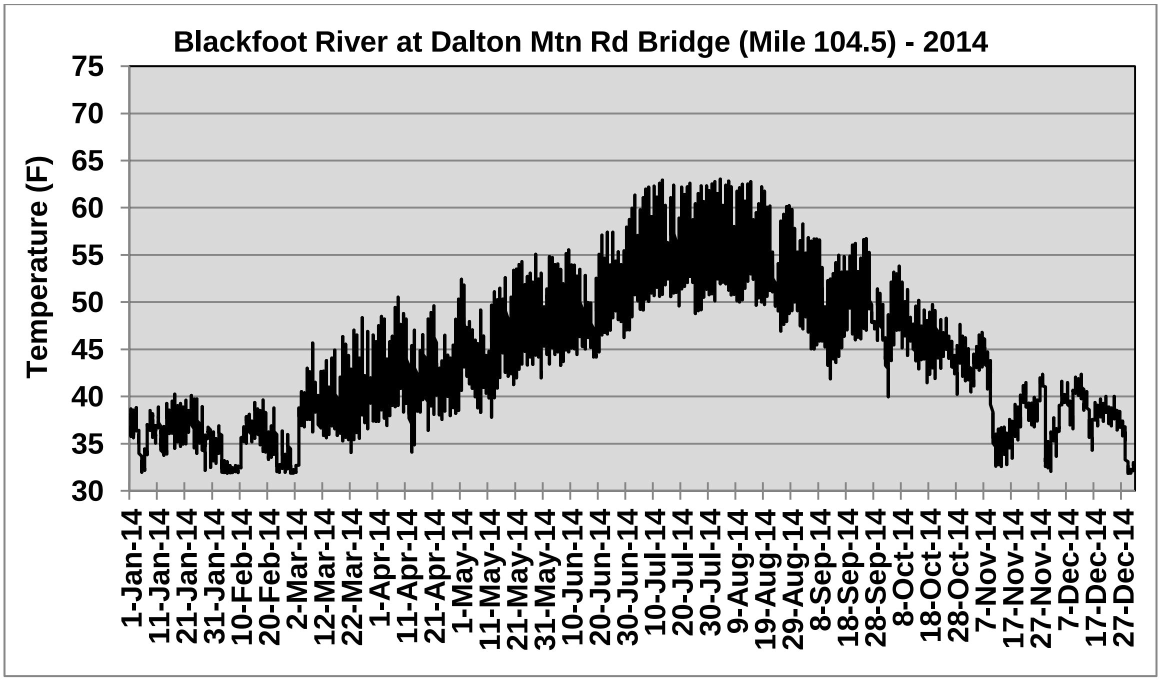
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	34.2	32	32.9	0.7	0.5
February	32.9	32	32.2	0.2	0
March	40.1	32	34.5	2.2	4.8
April	46	34.5	39.2	2.2	4.9
May	53.2	37.7	45.1	3.4	11.6
June	54	42.2	48.6	2.7	7
July	58.7	46.3	53.5	2.3	5.5
August	60	46.6	53.2	2.5	6.5
September	55.6	37.8	47.9	3.2	10.1
October	53.2	35.3	43.2	3.1	9.5
November	44.4	32	34.6	3.9	14.9
December	35.1	32	33.1	0.9	0.8



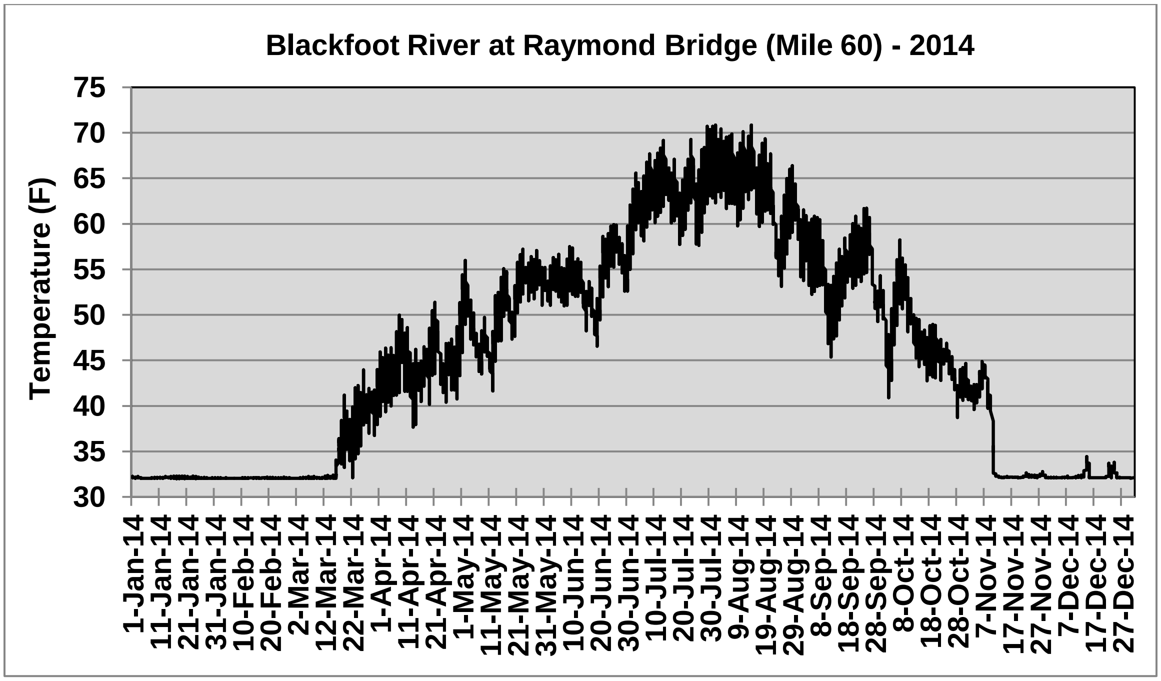
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	34	32	32.5	0.4	0.1
February	33.5	32	32.2	0.3	0.07
March	43.8	32	35.9	3.6	13
April	49.5	38.3	43.9	1.8	3.1
May	54	42.7	49	2.6	6.6
June	58.8	47.4	53.2	2.5	6
July	68.1	55	63.2	2.4	5.8
August	68.3	54.6	62.8	2.9	8.7
September	60.9	47.2	55.1	2.7	7.5
October	56	40.4	47.5	3.2	10.1
November	45.8	32	36.3	4.8	23.1
December	37	32	33.2	1.2	1.5



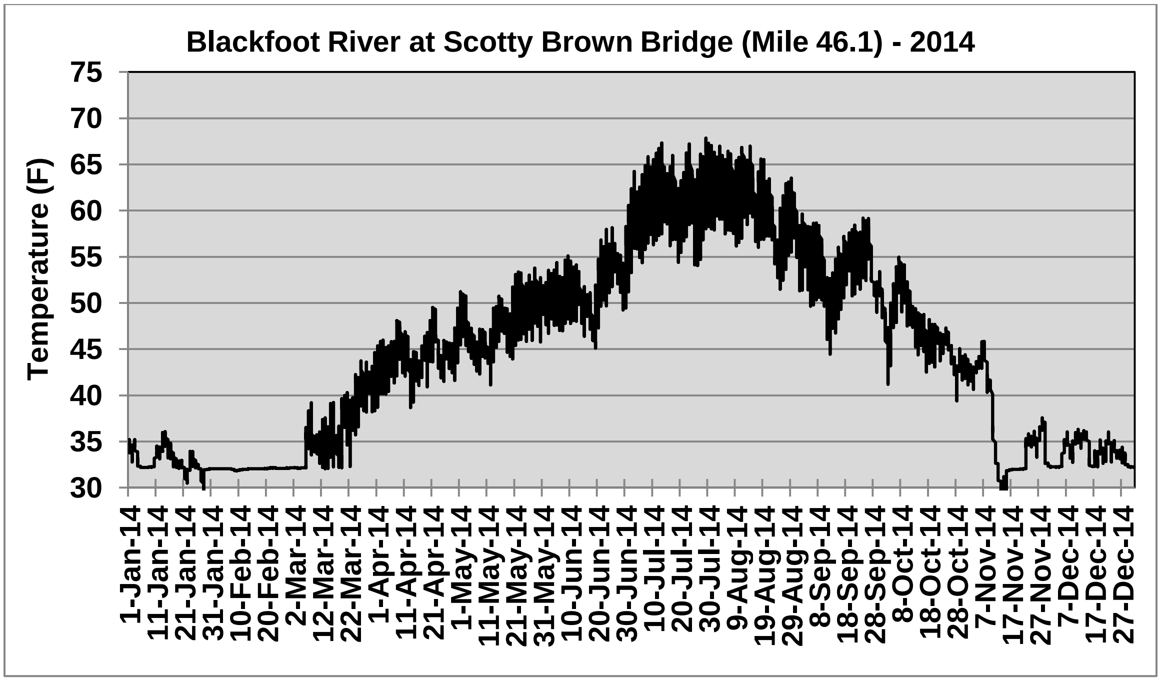
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	33.8	30.2	31.8	0.5	0.3
February	31.9	31.7	31.8	0.1	0.0
March	44.7	31.8	36.8	3.9	15.2
April	50.5	38.0	44.1	2.2	4.6
May	53.4	42.1	48.7	2.8	7.9
June	56.5	45.5	52.0	2.2	5.0
July	65.8	54.3	60.5	2.3	5.4
August	66.2	51.8	60.8	3.1	9.7
September	58.4	46.1	53.5	2.7	7.3
October	55.8	39.2	46.5	3.1	9.7
November	45.0	31.9	35.8	4.6	21.5
December	38.1	32.0	33.5	1.8	3.2



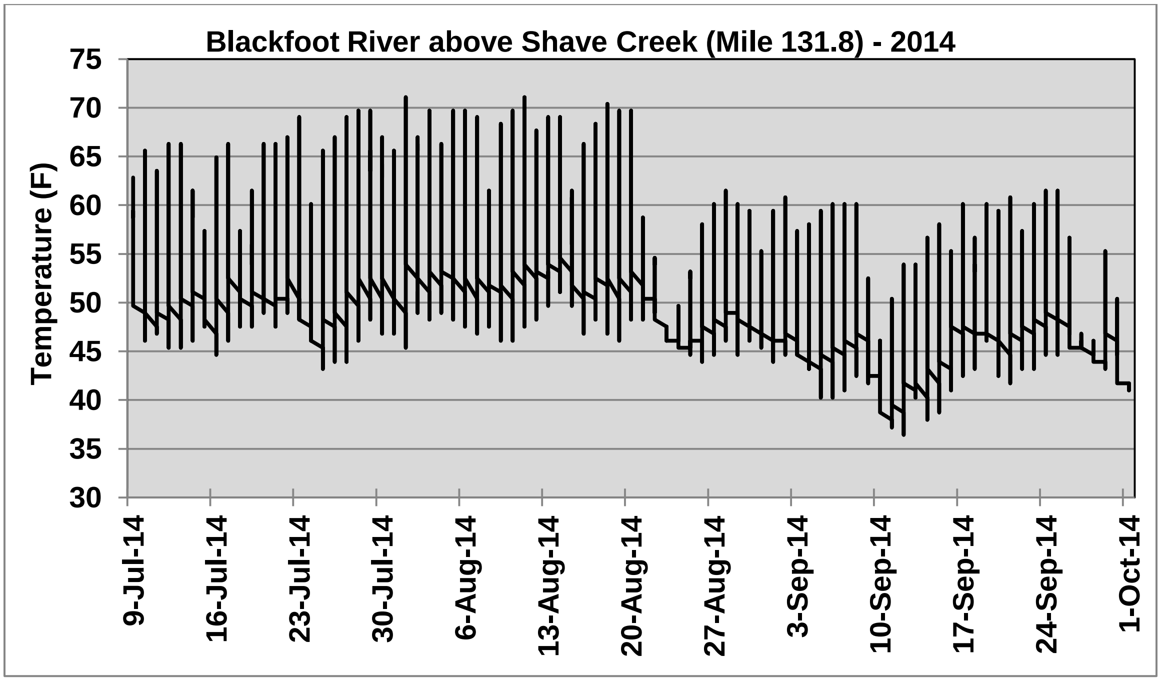
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	40.3	32.0	36.2	1.7	3.0
February	39.6	32.0	34.5	2.1	4.5
March	48.4	32.0	39.0	3.2	10.4
April	50.5	34.1	42.1	3.2	10.5
May	55.1	37.8	46.1	3.8	14.1
June	57.9	43.3	49.5	3.4	11.5
July	62.9	47.1	55.4	3.8	14.3
August	63.1	46.9	54.9	3.8	14.8
September	58.3	41.9	50.0	3.3	11.1
October	53.8	40.0	46.1	2.6	6.6
November	46.8	32.4	39.0	3.8	14.3
December	42.3	31.9	37.6	2.5	6.5



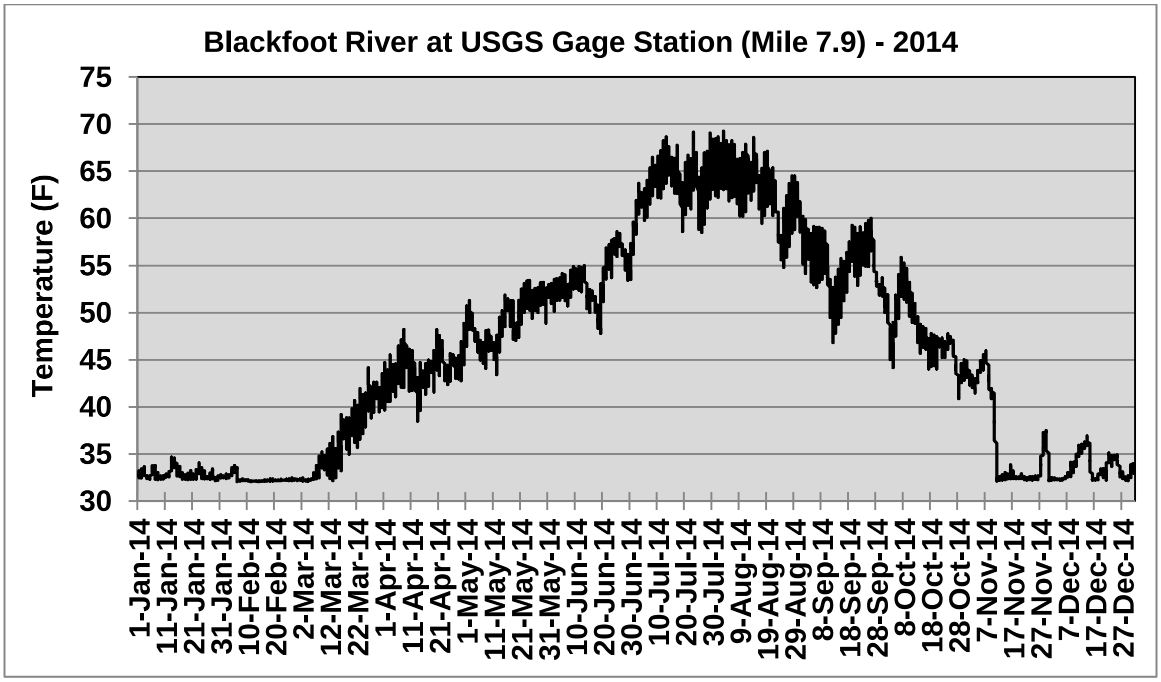
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	32.3	32.0	32.1	0.1	0.0
February	32.2	32.0	32.1	0.0	0.0
March	44.0	32.0	35.0	3.5	12.5
April	51.4	37.7	44.2	2.6	7.0
May	57.2	41.7	50.6	3.6	13.0
June	59.9	46.6	54.0	2.8	7.7
July	70.7	55.0	63.4	3.0	9.1
August	70.9	53.1	63.6	3.8	14.4
September	61.7	45.4	55.0	3.4	11.7
October	58.2	38.7	46.9	3.8	14.5
November	44.9	32.1	35.2	4.5	19.9
December	34.4	32.0	32.3	0.4	0.2



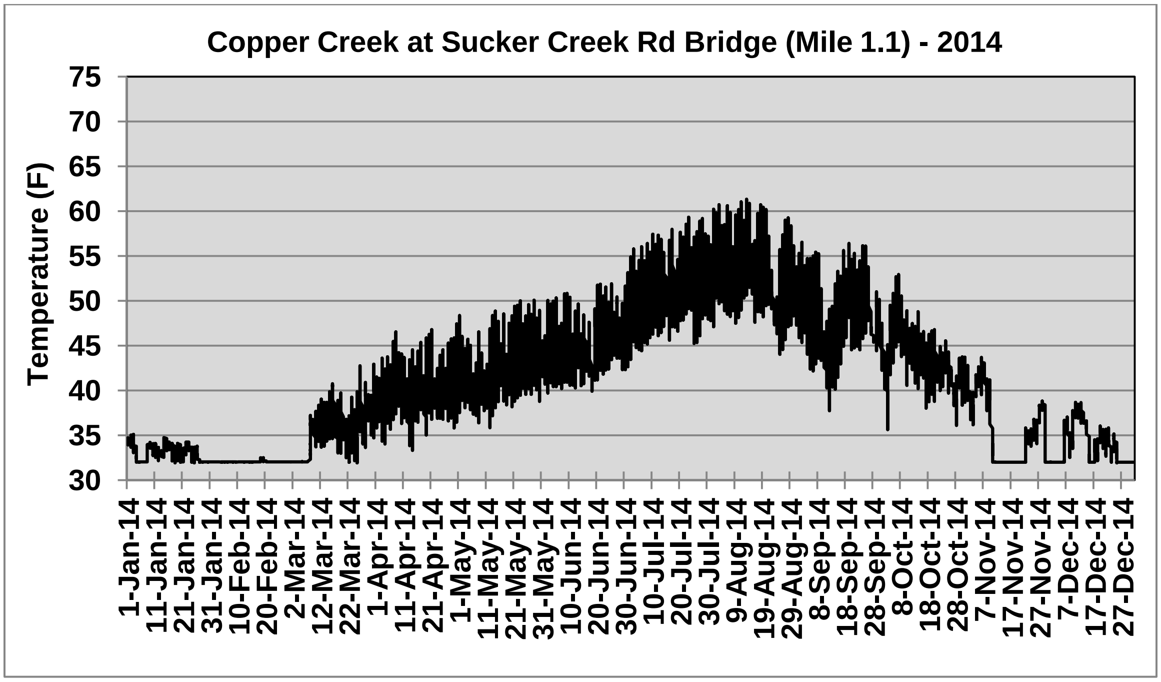
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	36.1	29.3	32.8	1.2	1.4
February	32.2	32.0	32.0	0.1	0.0
March	44.6	32.0	36.2	3.4	11.7
April	49.5	38.7	44.0	2.1	4.2
May	53.8	41.1	47.6	2.7	7.2
June	58.3	45.1	51.3	2.8	7.0
July	67.8	51.2	60.5	3.5	12.1
August	67.0	51.5	60.5	3.5	12.0
September	59.6	44.5	53.8	3.1	9.5
October	55.0	39.4	47.0	3.2	10.1
November	45.9	25.9	36.0	4.9	24.1
December	36.3	32.2	33.7	1.2	1.5



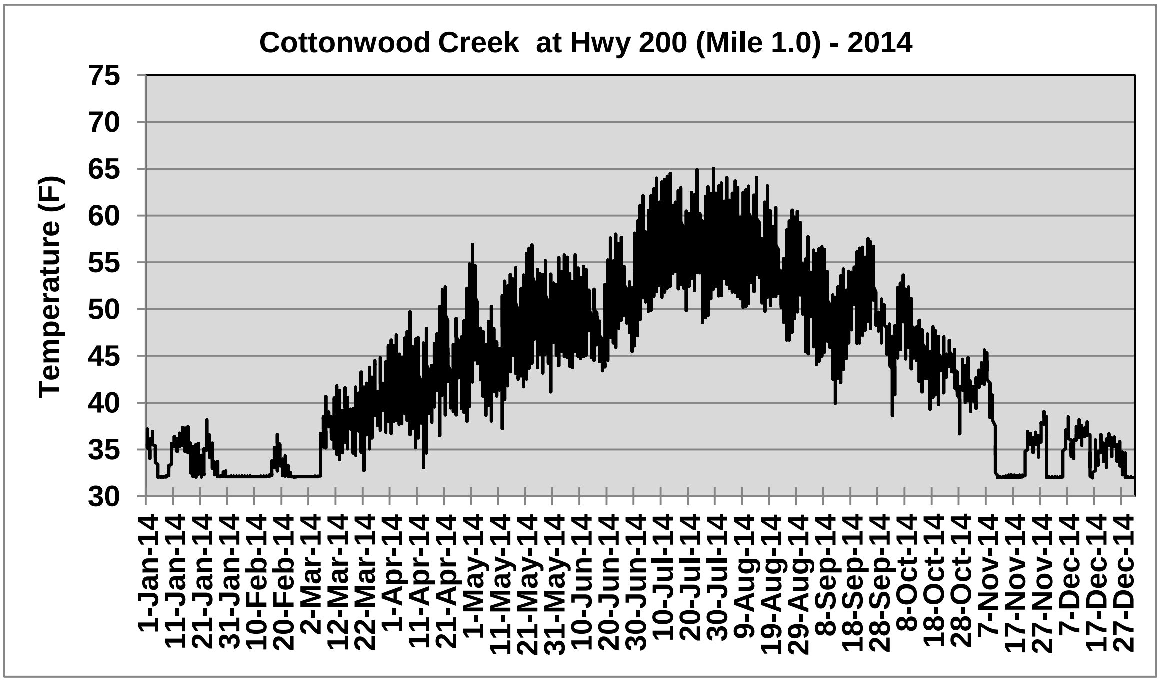
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	69.7	43.2	53.7	6.8	46.7
August	71.1	43.9	53.8	7.1	49.7
September	61.5	36.4	47.6	5.8	33.1



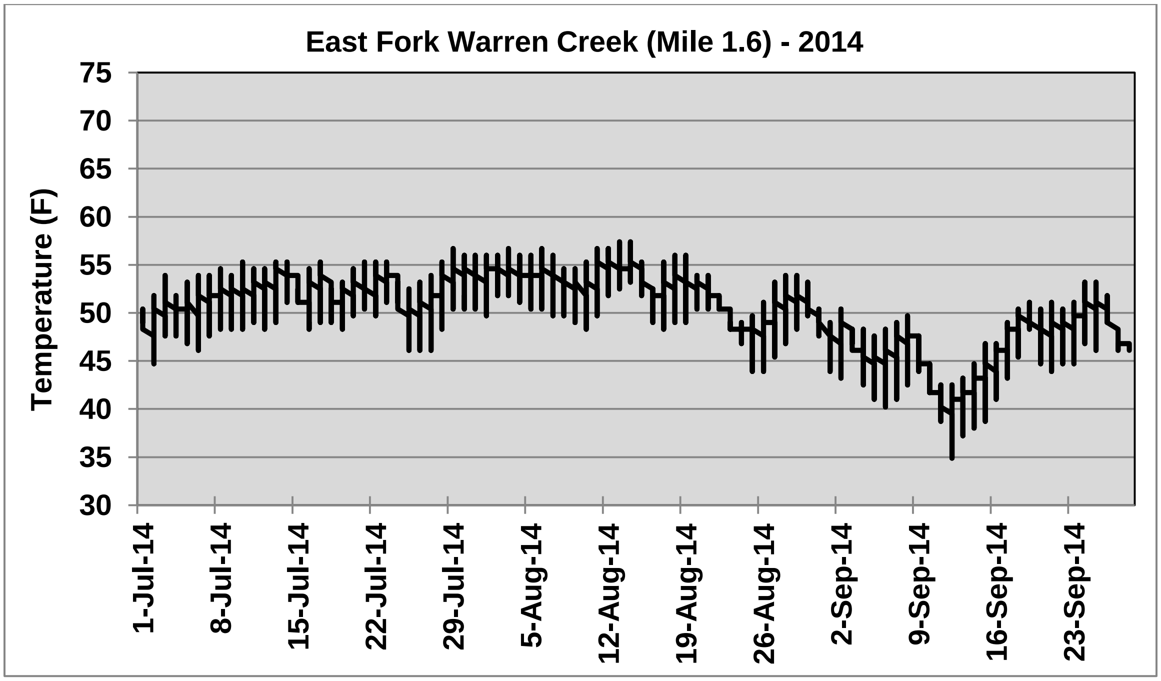
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	34.7	32.1	32.8	0.5	0.3
February	33.8	32.0	32.3	0.3	0.1
March	44.1	32.1	36.4	3.5	12.3
April	48.9	38.5	43.9	1.8	3.2
May	53.4	43.4	49.1	2.4	5.7
June	58.6	47.8	53.6	2.3	5.4
July	69.2	56.2	63.5	2.5	6.4
August	69.3	54.8	63.1	3.0	9.3
September	60.2	46.8	55.3	2.8	7.6
October	55.9	40.8	47.8	3.1	9.9
November	45.9	32.1	36.3	4.9	24
December	36.9	32.1	33.4	1.2	1.5



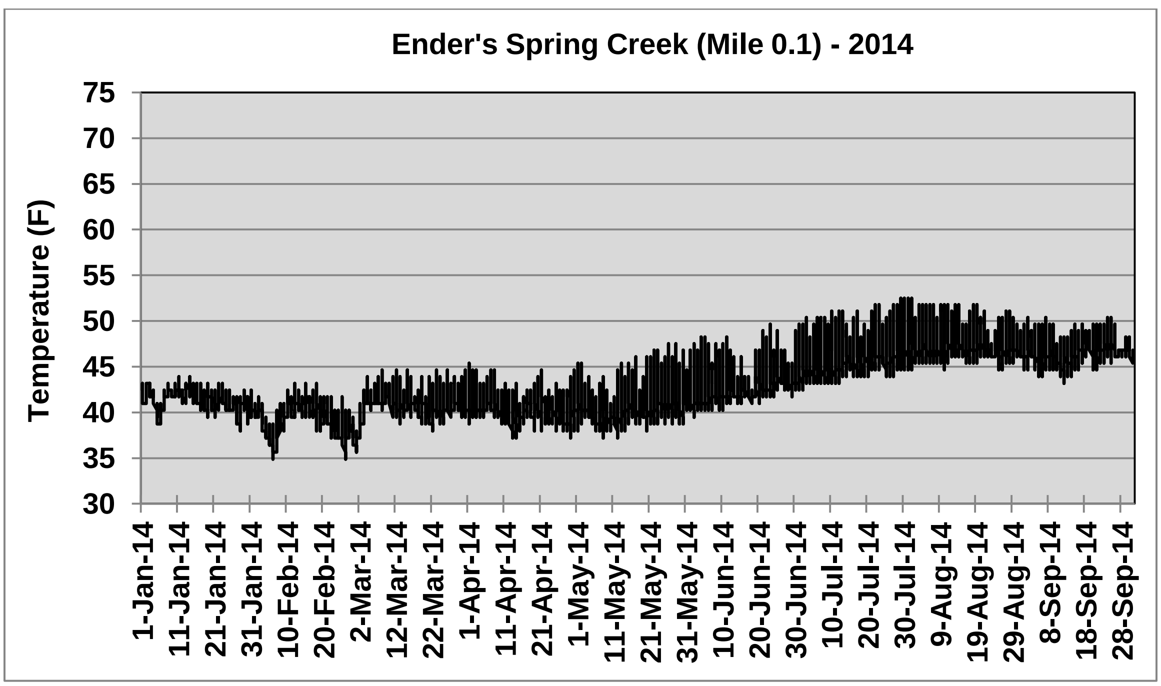
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	35.1	31.9	32.9	0.9	0.7
February	32.5	32.0	32.1	0.1	0.0
March	42.9	31.9	35.3	2.6	6.9
April	47.4	33.3	39.8	2.9	8.4
May	50.1	35.9	42.5	3.3	10.7
June	51.9	39.7	45.0	3.0	9.2
July	59.3	42.7	51.3	3.8	14.2
August	61.3	44.0	52.9	4.0	16.3
September	56.6	37.8	48.2	3.9	15.4
October	52.9	35.6	43.4	3.2	9.9
November	43.7	32	35.4	3.7	13.6
December	38.7	32	33.8	2	4.1



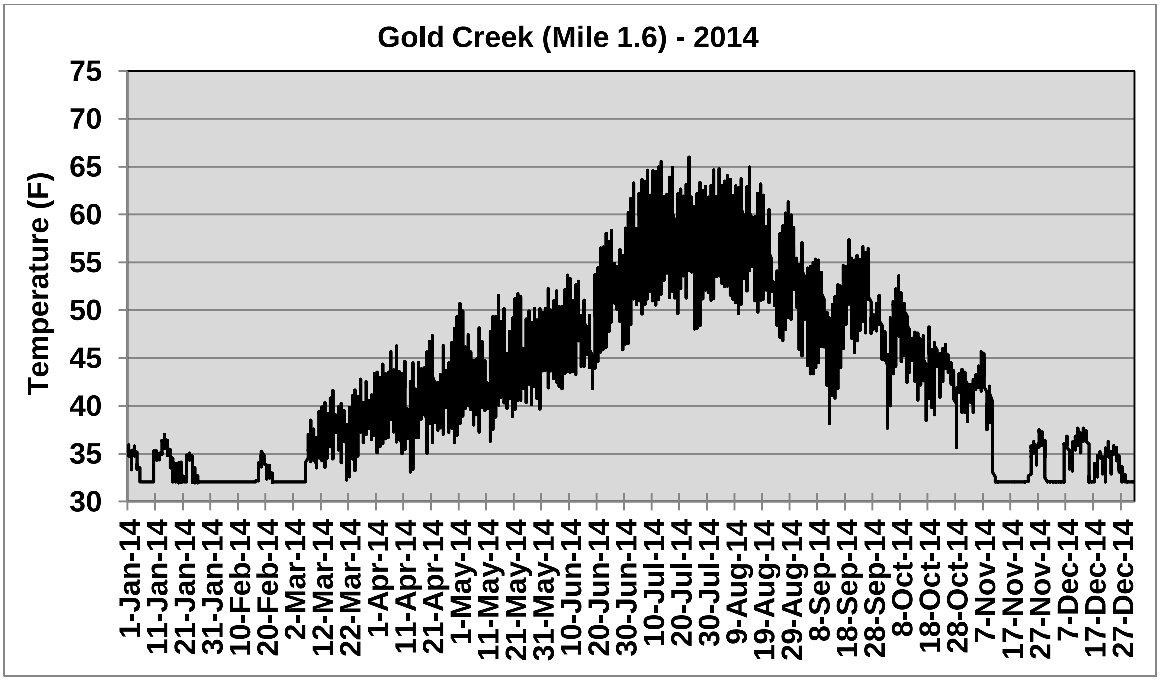
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	38.2	32.0	34.0	1.7	2.8
February	36.6	32.0	32.4	0.8	0.6
March	46.1	32.1	37.4	3.4	11.2
April	54.8	33.1	42.6	3.8	14.3
May	56.9	37.2	47.2	4.2	17.3
June	58.1	43.4	49.7	3.5	12.4
July	65.0	47.2	56.6	3.9	15.5
August	64.1	46.7	55.6	3.8	14.7
September	57.7	39.9	50.2	3.4	11.9
October	53.6	36.7	45.0	3.1	9.4
November	45.6	32	36.3	4.2	17.8
December	38.5	32	34.4	1.9	3.6



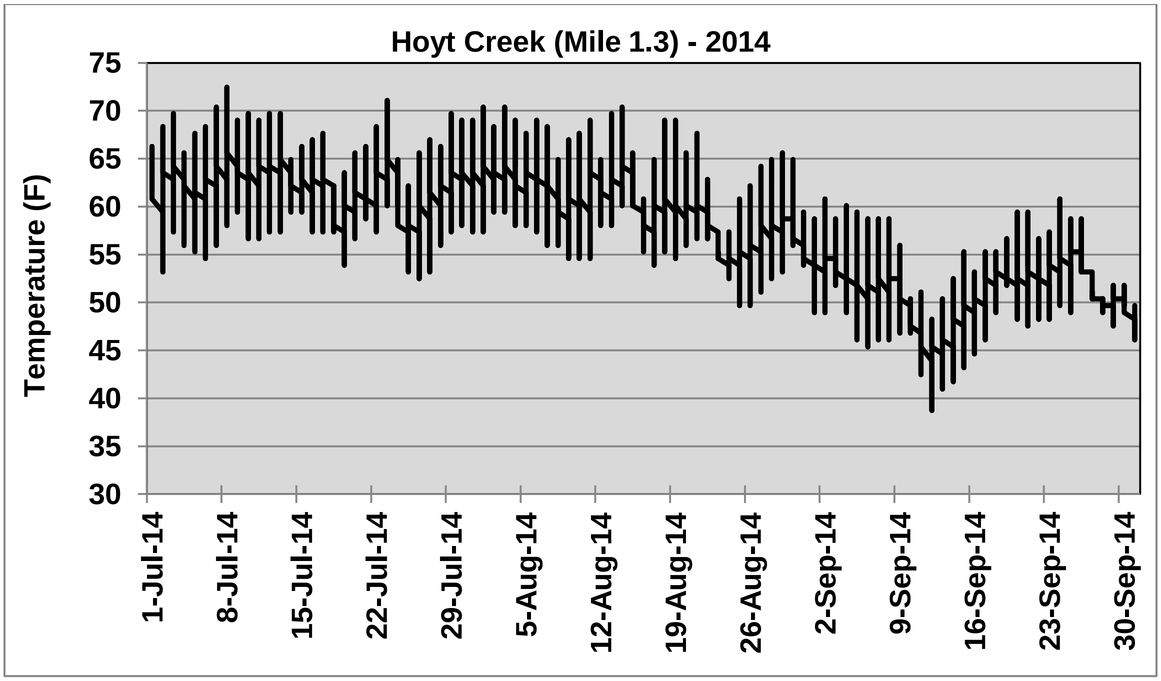
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	56.7	44.7	51.3	2.4	5.8
August	57.4	43.9	51.7	2.7	7.5
September	53.2	34.9	45.6	3.6	12.6



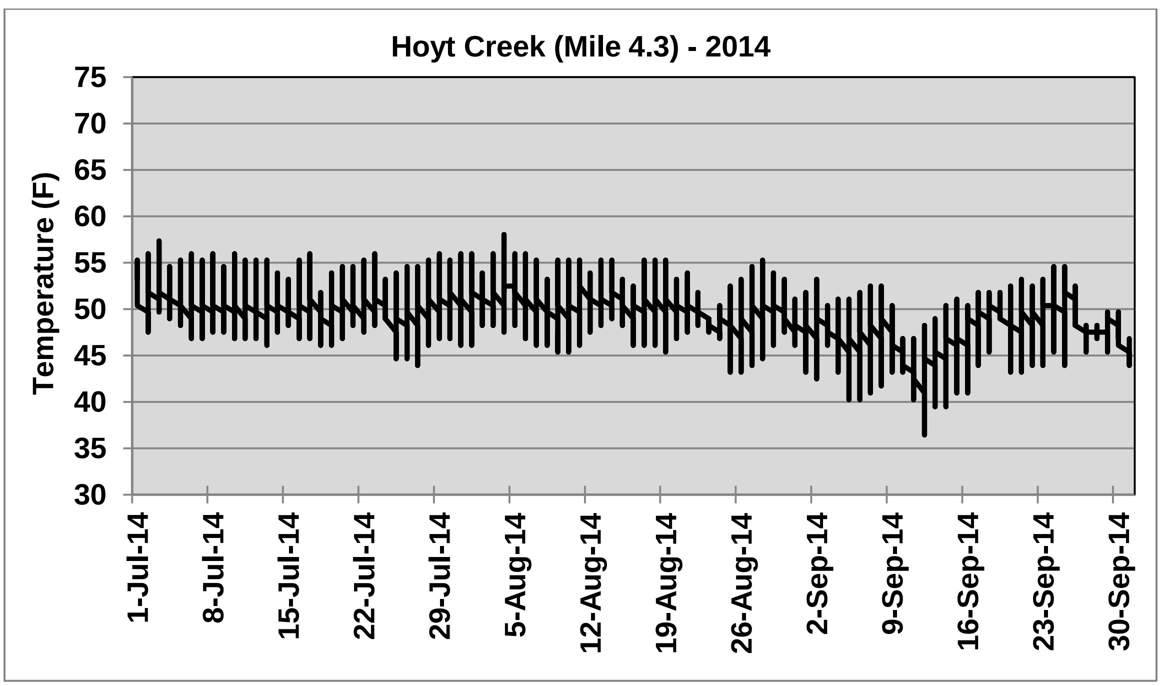
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	43.9	38	41.4	1.2	1.5
February	43.2	34.9	39.5	1.7	3
March	44.7	35.7	40.9	1.6	2.7
April	45.4	37.2	40.6	1.7	2.8
May	47.5	37.2	41.1	2.4	5.6
June	49.7	39.5	43.3	2.2	4.8
July	52.5	42.5	46.5	2.5	6.2
August	52.5	44.7	47.6	2.0	3.8
September	50.4	43.2	46.8	1.6	2.6



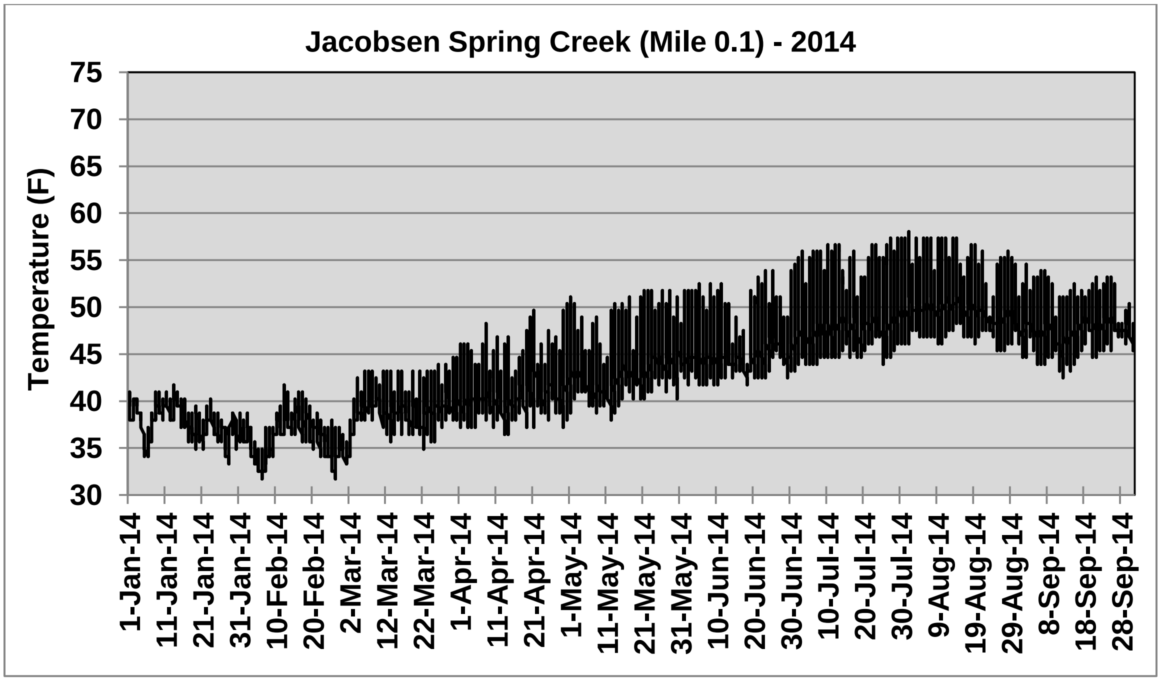
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.0	31.9	33.3	1.5	2.1
February	35.2	32.0	32.3	0.7	0.4
March	43.4	32.0	36.4	2.9	8.6
April	49.3	33.1	40.2	3.1	9.6
May	51.7	36.3	43.7	3.4	11.8
June	58.6	41.8	48.8	3.8	14.7
July	66.0	46.5	57.0	4.4	19.6
August	65.0	46.8	56.0	4.2	17.3
September	57.4	38.1	49.9	3.7	13.9
October	53.6	35.6	44.7	3.2	10.0
November	45.7	32	35.8	4.3	18.2
December	37.7	32	33.9	1.8	3.2



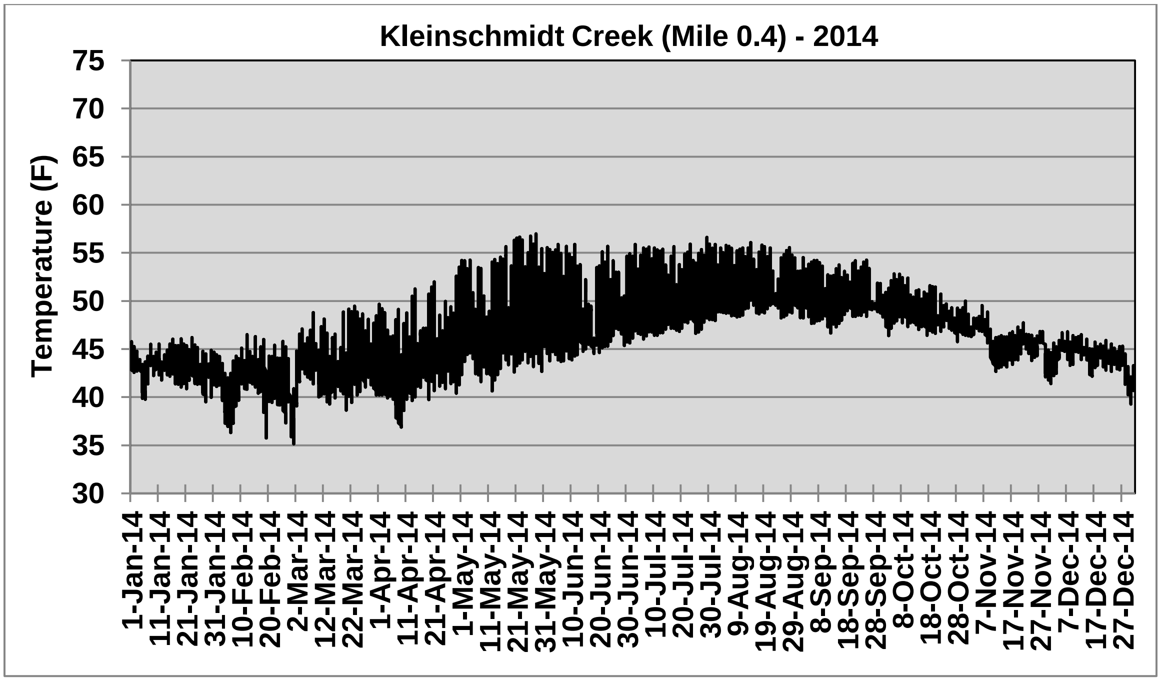
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	72.5	52.5	62	4.2	17.3
August	70.4	49.7	60.1	4.5	20.4
September	60.8	38.7	51.3	4.2	17.5



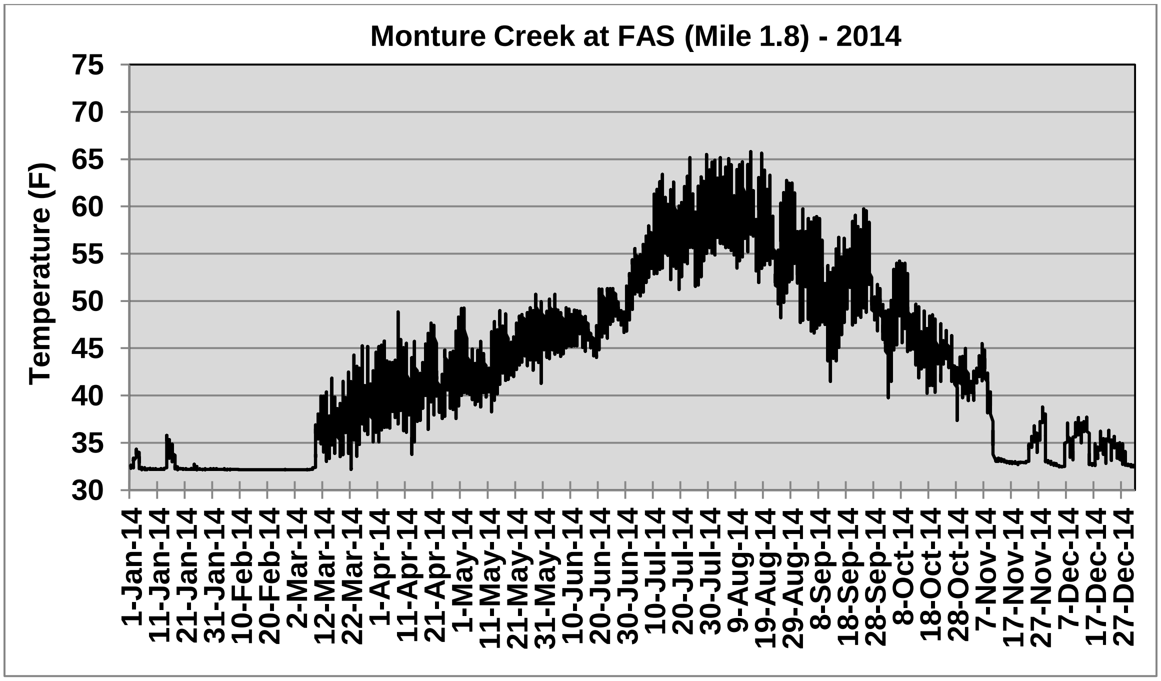
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	57.4	43.9	50.7	3	9
August	58	43.2	50.2	3	9.2
September	54.6	36.4	47.1	3.5	12.5



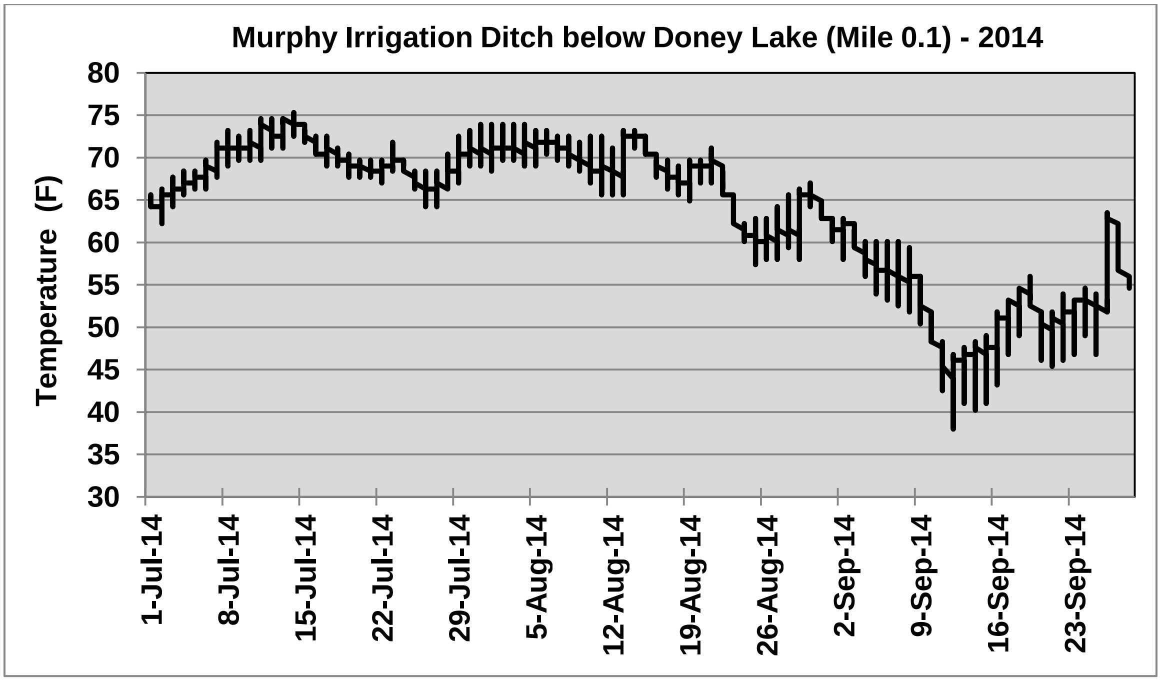
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	41.7	33.3	37.9	1.7	3
February	41.7	31.7	36.3	2	4.1
March	44.7	33.3	39.2	2.3	5.4
April	50.4	36.4	41.3	2.9	8.1
May	51.8	38	44.1	3.3	11.1
June	53.9	41.7	46.1	3	9.2
July	57.4	43.2	49.3	3.8	14.1
August	58	45.4	50.3	3.1	9.5
September	54.6	42.5	48	2.5	6.2
October	48.3	45.4	46.3	0.8	0.7



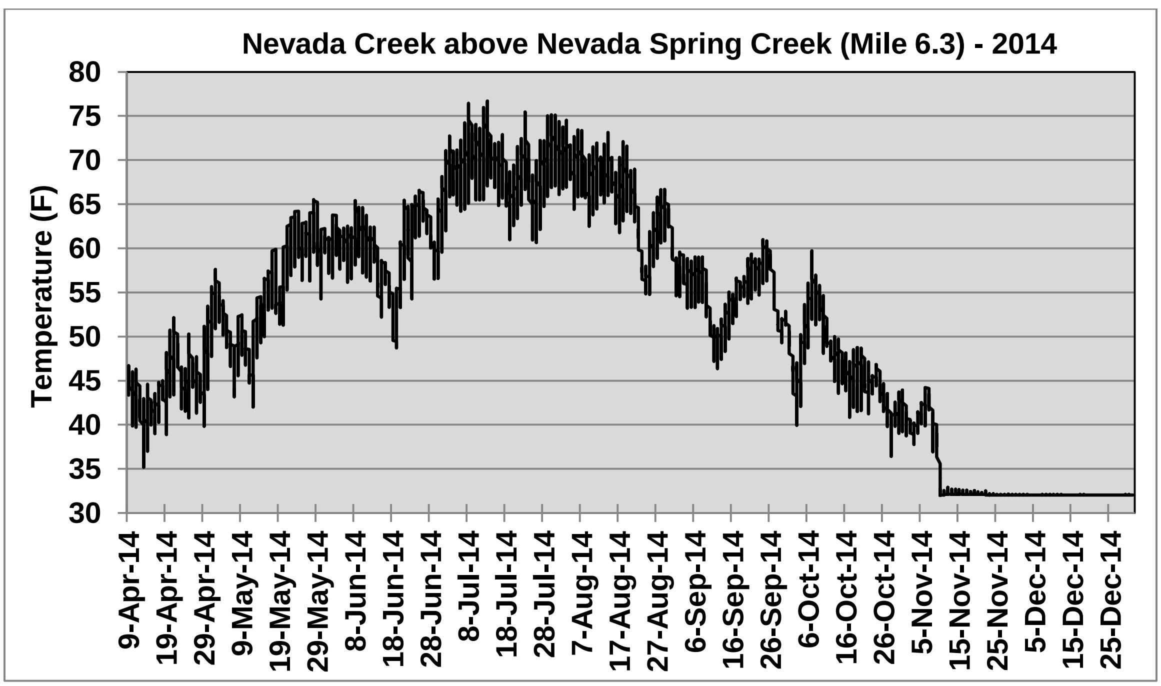
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	46.2	39.5	42.9	1.3	1.7
February	46.5	35.7	41.4	2.2	4.7
March	49.5	35.2	42.9	2.4	5.9
April	53.5	36.9	43.7	3.1	9.4
May	57.0	40.6	47.4	3.8	14.1
June	55.9	43.7	48.2	3.1	9.4
July	56.6	45.6	50.1	2.7	7.5
August	56.1	48.0	51.2	2.1	4.2
September	54.5	46.7	50.1	1.7	3.0
October	52.8	45.8	48.5	1.4	1.9
November	49.5	41.8	45.5	1.5	2.4
December	46.8	39.3	44	1.3	1.8



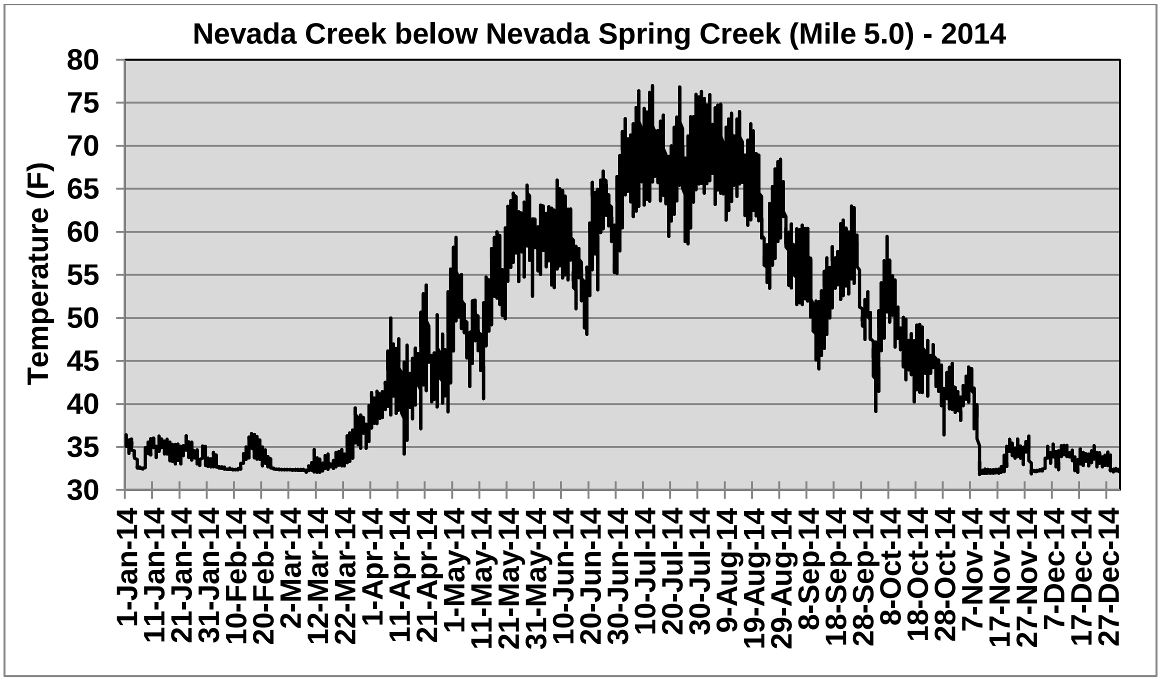
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	35.8	32.1	32.5	0.7	0.5
February	32.2	32.1	32.2	0.0	0.0
March	45.3	32.1	36.2	3.4	11.3
April	48.9	33.8	41.1	2.9	8.4
May	50.7	38.3	44.1	2.7	7.1
June	51.6	43.9	47.4	1.9	3.4
July	65.5	48.0	56.5	3.7	13.4
August	65.8	48.2	57.9	3.7	13.6
September	59.7	41.5	51.8	3.7	13.5
October	54.2	37.4	45.5	3.3	11.0
November	45.5	32.7	36.3	3.9	14.8
December	37.7	32	34.2	1.5	2.2



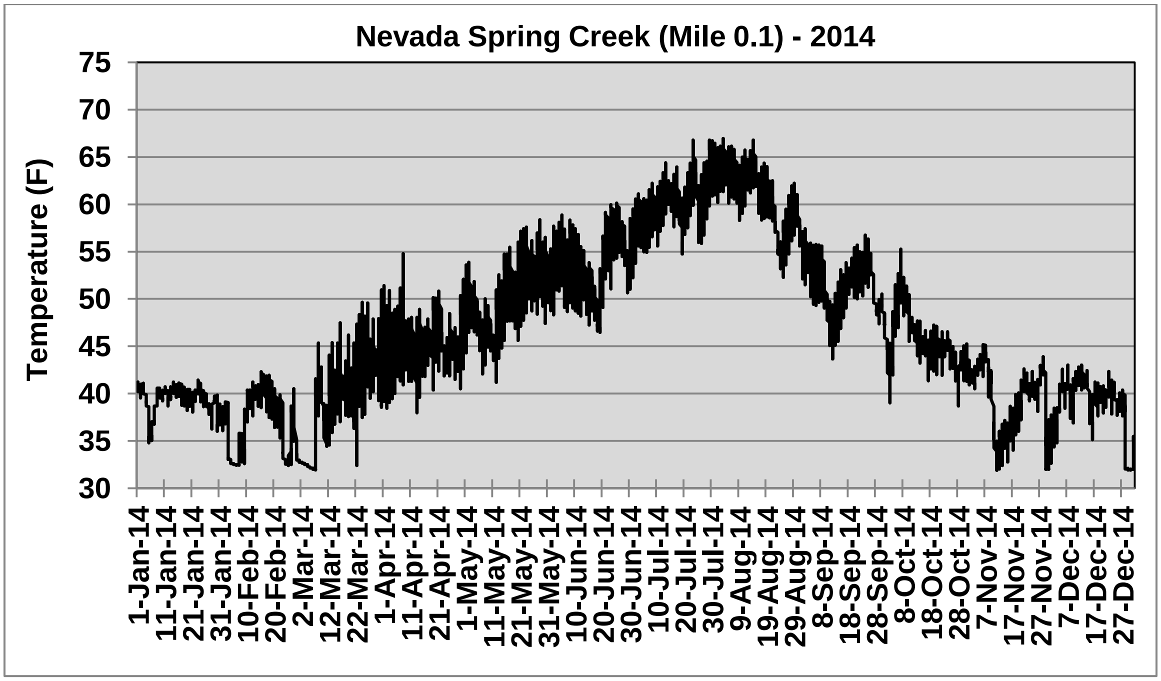
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
July	75.3	62.2	69.4	2.7	7
August	73.9	57.4	67.6	4.2	17.4
September	63.5	38	52.2	5.8	33.5



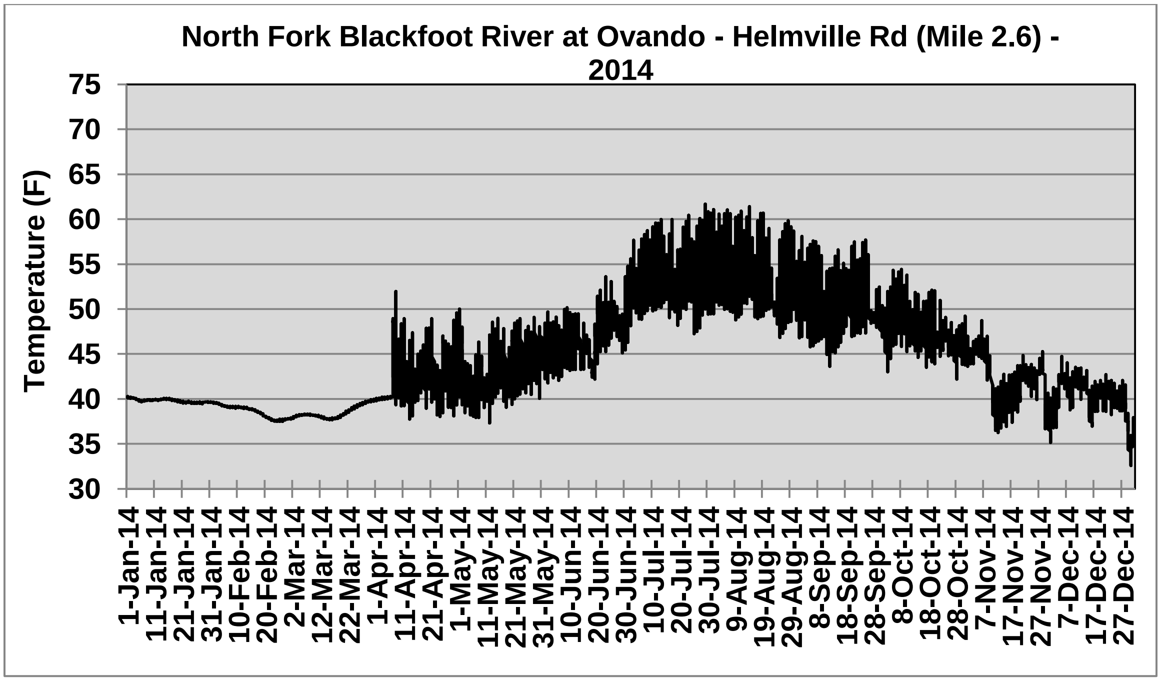
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
April	53.4	35.2	44.0	3.3	11.0
May	65.5	42.0	54.4	5.4	29.3
June	66.6	48.7	59.5	3.8	14.2
July	76.7	59.6	68.5	3.5	12.0
August	74.5	54.8	65.8	4.5	20.1
September	61.0	46.3	54.7	3.2	10.5
October	59.7	36.4	46.4	4.4	19.3
November	44.2	32.0	34.6	3.9	15.1
December	32.1	32.0	32.0	0.0	0.0



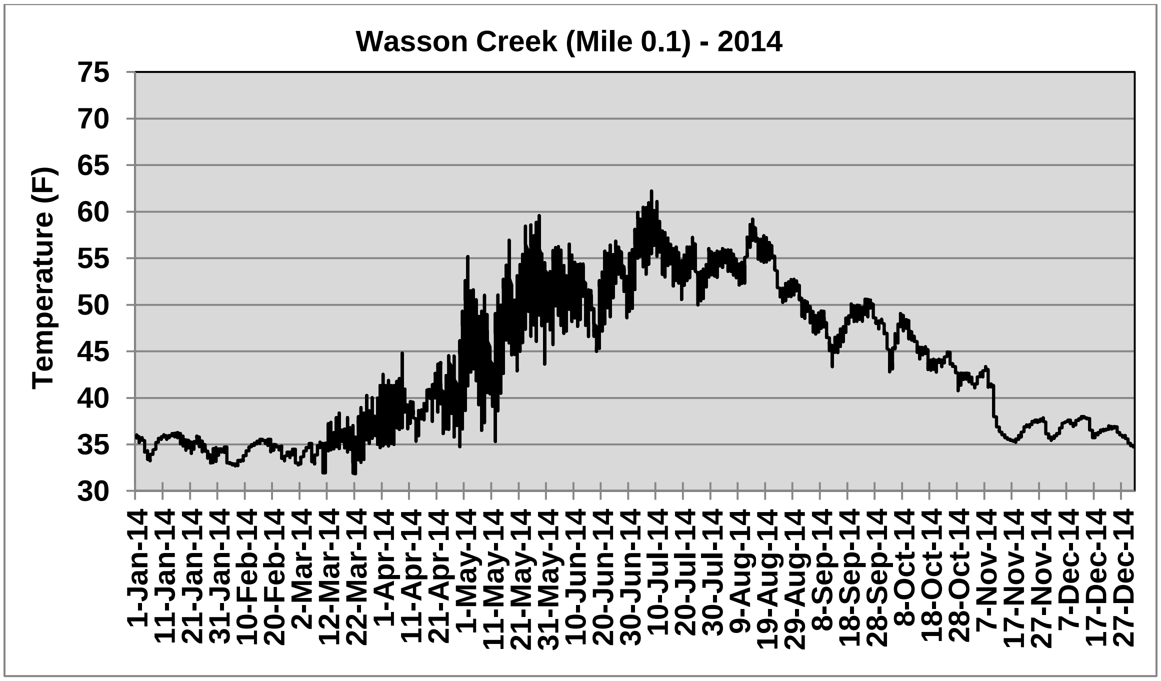
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	36.4	32.4	34.3	1.0	1.0
February	36.6	32.3	33.2	1.1	1.2
March	39.8	32.0	33.7	1.8	3.3
April	55.7	34.2	42.9	3.8	14.1
May	65.4	40.6	54.0	5.5	30.7
June	67.1	48.1	58.9	4.1	16.5
July	77.0	57.8	67.9	4.0	16.3
August	75.9	53.4	65.6	5.0	25.2
September	63.0	44.1	54.2	3.9	15.3
October	59.5	36.4	46.1	4.1	17.1
November	44.3	31.8	35.5	3.8	14.1
December	35.3	32	33.4	0.8	0.7



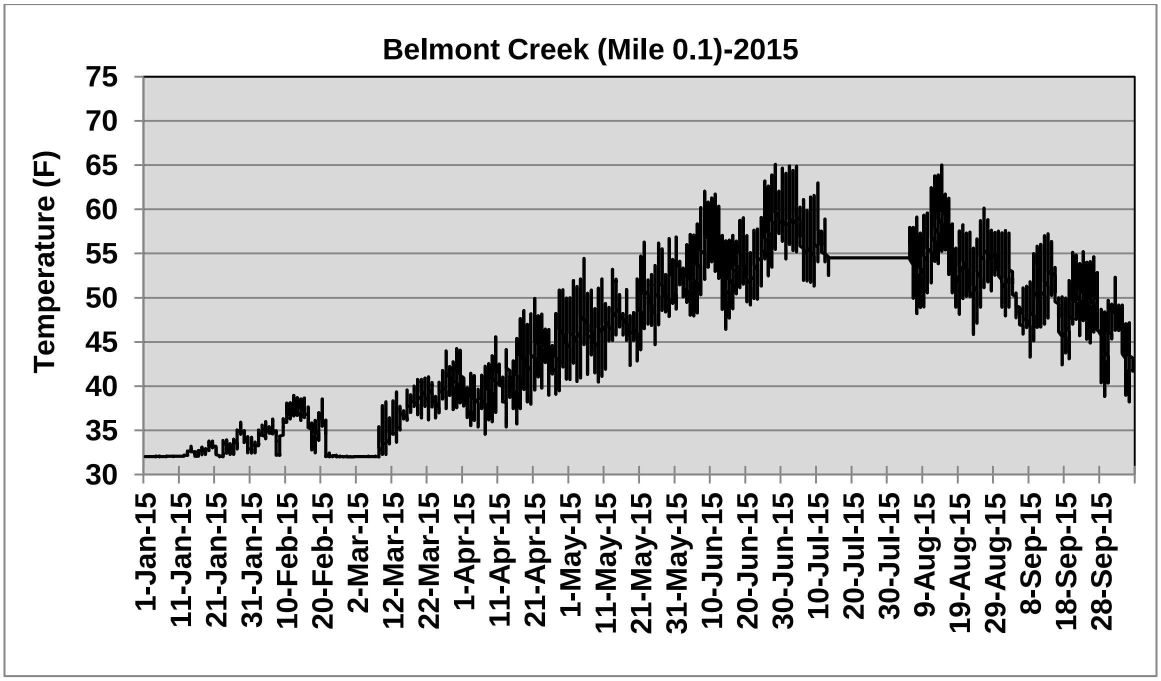
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	41.4	34.8	39.2	1.4	2.0
February	42.3	32.4	36.6	3.1	9.5
March	51.0	31.9	39.3	4.8	22.7
April	54.8	38.0	44.7	3.0	8.8
May	58.4	41.2	49.8	3.6	13.2
June	60.1	46.5	53.4	3.4	11.3
July	66.8	52.2	60.0	2.7	7.5
August	66.9	52.3	60.9	3.3	10.9
September	57.4	43.7	51.5	2.8	7.7
October	55.3	38.7	45.5	2.8	7.7
November	45.2	32	39.3	3.5	12.1
December	43	32	38.8	2.9	8.1



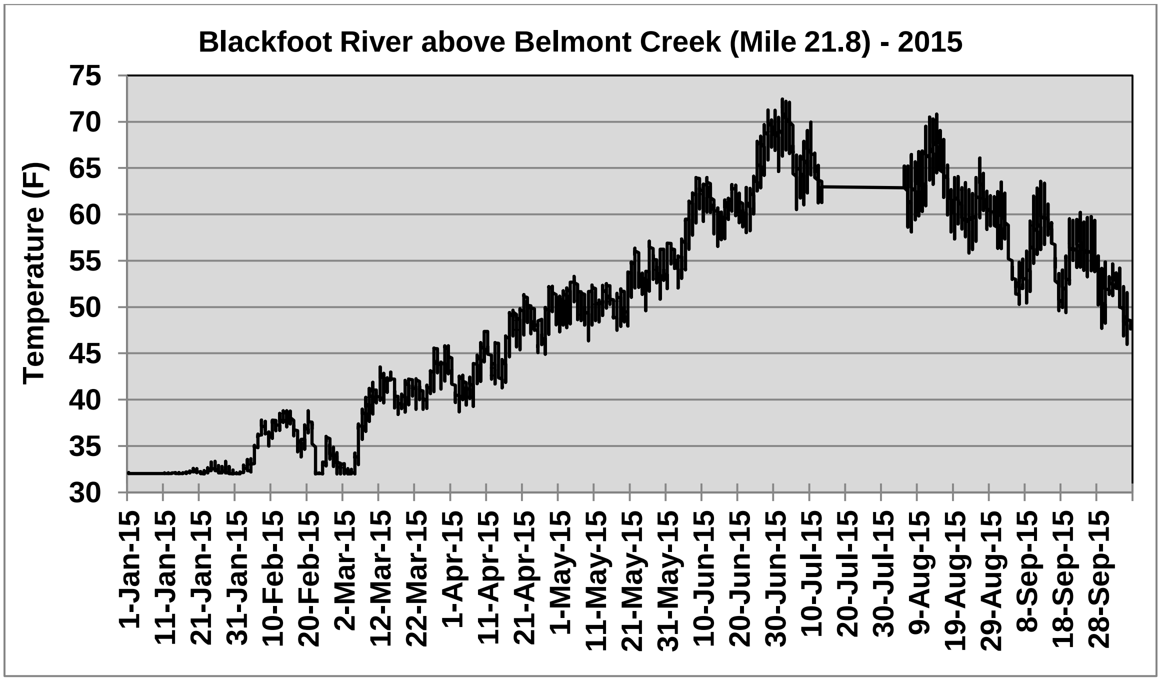
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	40.3	39.5	39.8	0.2	0.0
February	39.6	37.5	38.6	0.7	0.5
March	40.0	37.7	38.5	0.7	0.4
April	51.9	37.8	42.0	2.7	7.5
May	50.0	37.3	43.1	2.8	7.7
June	53.6	41.8	46.7	2.5	6.1
July	61.7	46.3	53.4	3.3	10.8
August	61.4	46.8	53.4	3.5	12.1
September	58.1	43.6	50.4	3.1	9.8
October	54.4	42.2	47.3	2.3	5.4
November	48.7	36.3	42	2.8	8
December	44.7	32.6	40	2.2	4.6



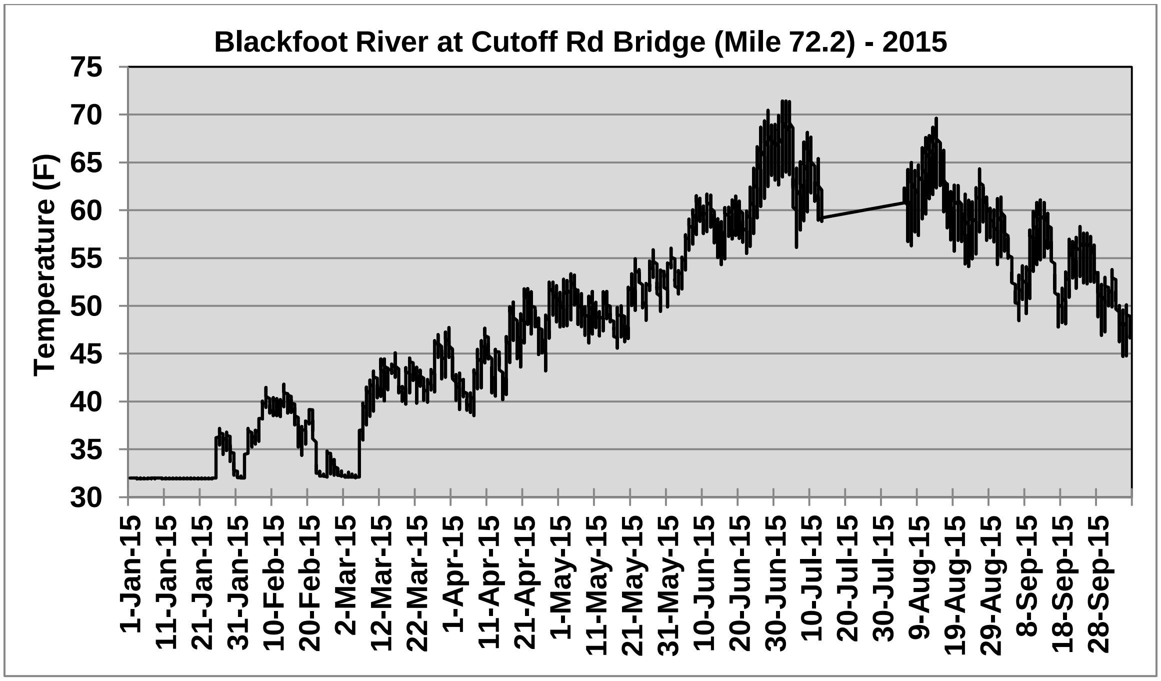
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	36.3	33.0	35.0	0.8	0.7
February	35.5	32.7	34.2	0.8	0.7
March	41.3	31.8	35.2	1.7	2.9
April	49.3	34.8	39.2	2.5	6.0
May	59.6	35.3	48.0	5.0	25.4
June	56.8	45.0	51.6	2.9	8.2
July	62.2	49.6	55.1	2.4	5.7
August	59.2	50.3	54.3	2.1	4.3
September	50.7	43.4	48.2	1.5	2.3
October	49.1	40.8	44.8	1.9	3.5
November	43.4	35.2	38.3	2.6	7
December	38	34.7	36.6	0.8	0.7



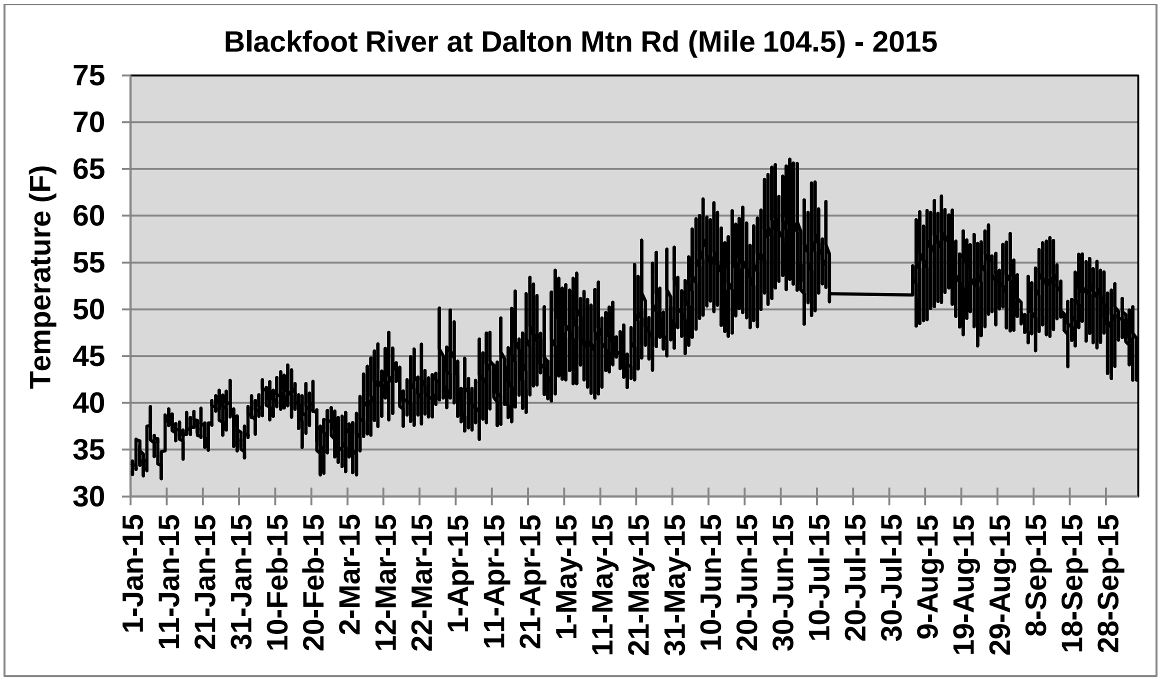
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	35.9	32.0	32.7	0.8	0.7
February	38.9	32.0	34.6	2.0	4.0
March	44.2	32.0	36.5	3.4	11.5
April	50.9	34.6	41.4	3.4	11.6
May	56.9	40.5	48.1	3.4	11.3
June	65.1	46.4	54.7	3.7	13.5
July	64.9	51.3	57.2	3.1	9.9
August	65.0	45.9	54.5	3.6	12.9
September	57.6	38.8	49.2	3.6	12.9
October	52.3	38.2	45.6	3.3	10.7



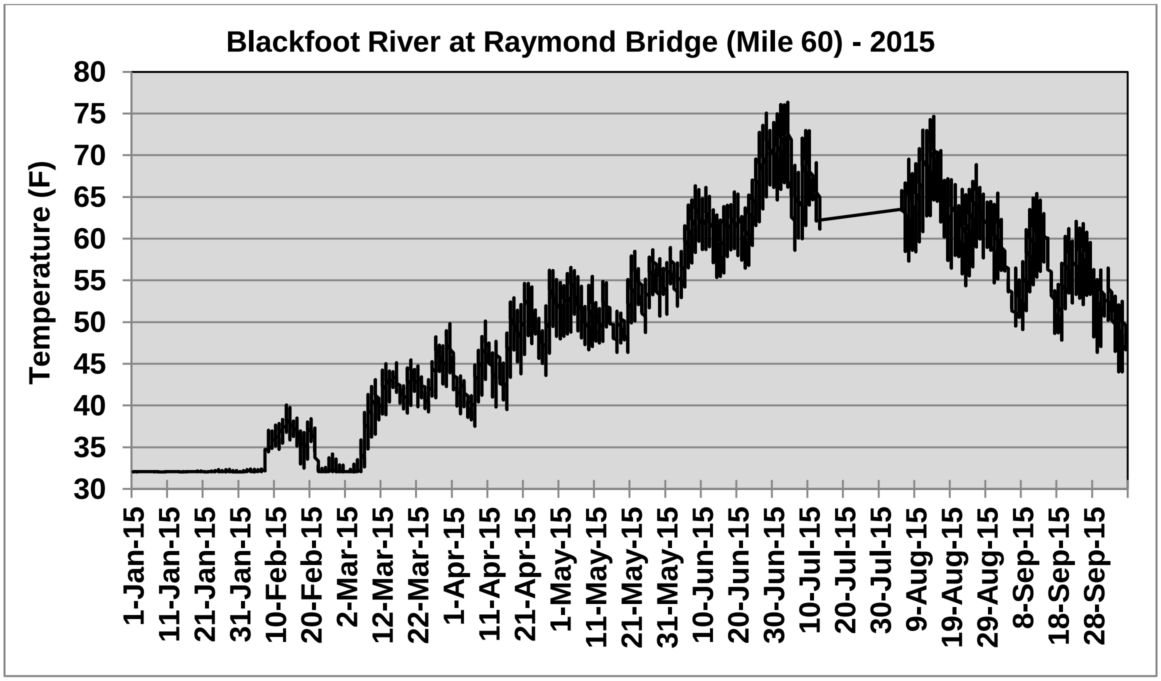
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	33.3	32.0	32.2	0.2	0.0
February	38.8	32.0	35.2	2.1	4.5
March	45.8	32.0	39.5	3.7	13.9
April	52.2	38.7	45.5	3.4	11.6
May	57.1	46.3	51.4	2.2	4.8
June	71.3	52.1	61.2	4.2	17.3
July	72.4	60.5	66.2	3.0	8.7
August	70.8	55.8	62.3	3.2	10.2
September	63.6	47.7	55.3	3.3	10.8
October	54.6	46.0	50.9	2.2	5.0



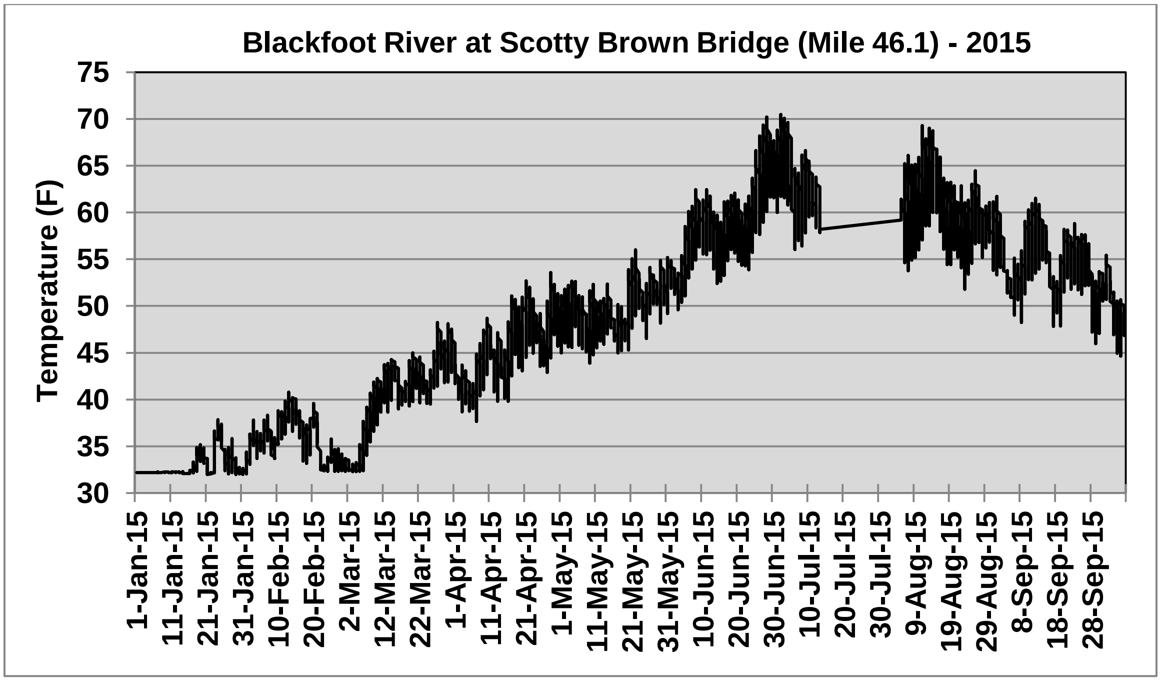
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.2	31.9	32.5	1.3	1.7
February	41.8	32.0	36.5	2.9	8.6
March	47.7	32.0	40.4	4.3	18.7
April	52.5	38.5	45.2	3.6	12.7
May	55.9	45.6	50.3	2.2	4.9
June	70.5	51.2	59.2	4.1	16.5
July	71.4	56.1	64.0	3.6	12.6
August	69.6	54.1	60.8	3.3	10.6
September	61.4	46.9	54.2	3.2	10.5
October	53.8	44.7	49.4	2.2	5.1



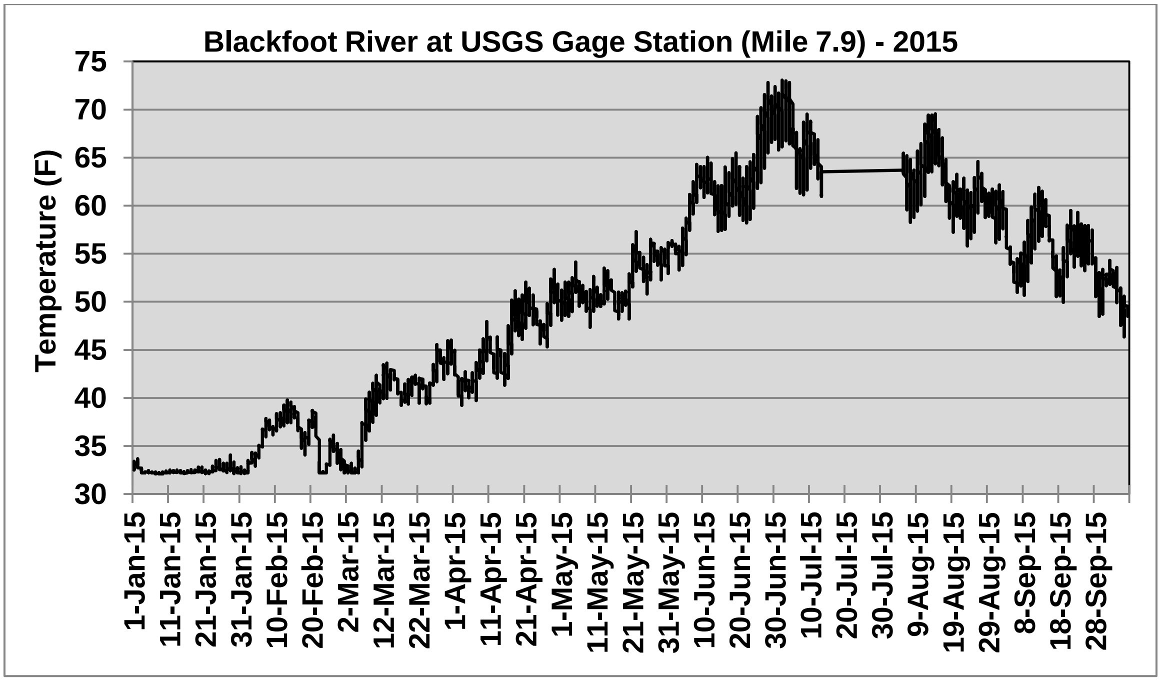
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	42.4	31.9	36.8	2.1	4.6
February	44.0	32.3	38.6	2.4	6.0
March	50.1	32.3	40.7	3.4	11.2
April	54.2	36.1	43.4	4.0	15.9
May	57.4	40.5	47.4	3.5	12.2
June	65.4	45.3	54.3	4.4	19.4
July	66.0	48.4	56.4	4.4	19.0
August	62.1	46.1	53.7	3.6	12.6
September	58.1	42.6	50.3	3.1	9.5
October	51.1	42.4	47.3	2.3	5.2



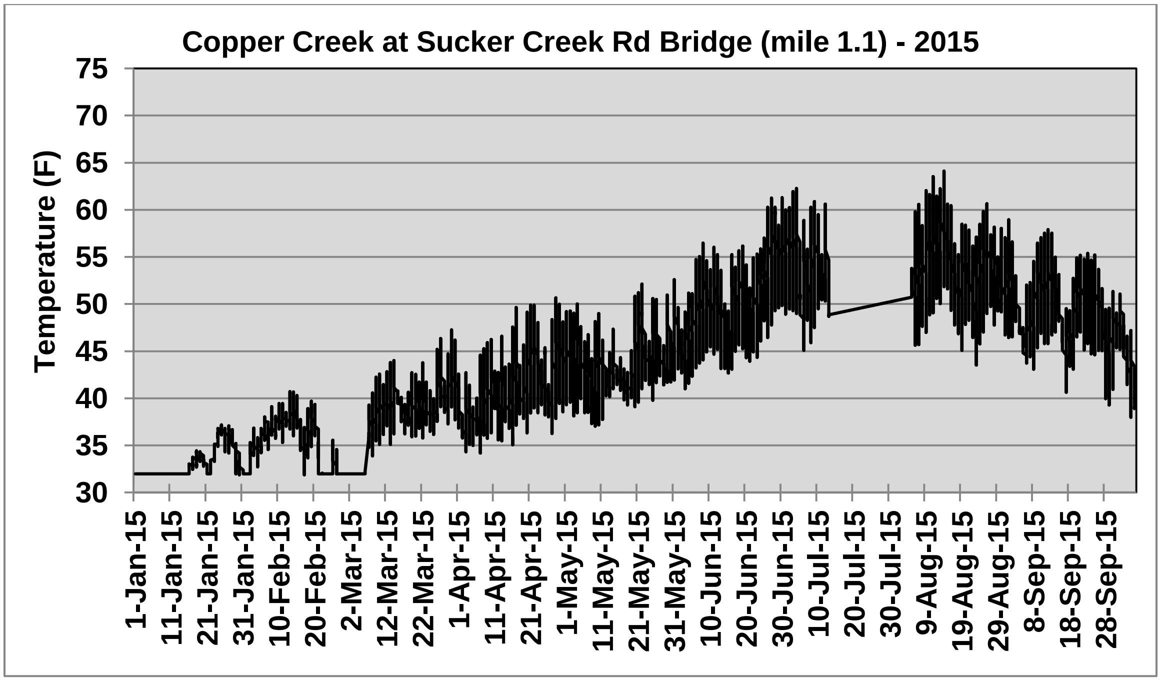
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	32.3	32.0	32.1	0.0	0.0
February	40.1	32.0	34.3	2.3	5.1
March	49.8	32.0	40.2	4.6	21.5
April	56.2	37.5	46.1	4.4	19.3
May	58.7	46.4	52.0	2.8	8.1
June	75.0	51.9	61.7	4.8	22.8
July	76.3	58.6	67.3	4.2	17.9
August	74.6	54.3	63.4	4.2	17.4
September	65.4	46.4	55.8	4.0	16.0
October	56.5	44.0	50.4	2.9	8.6



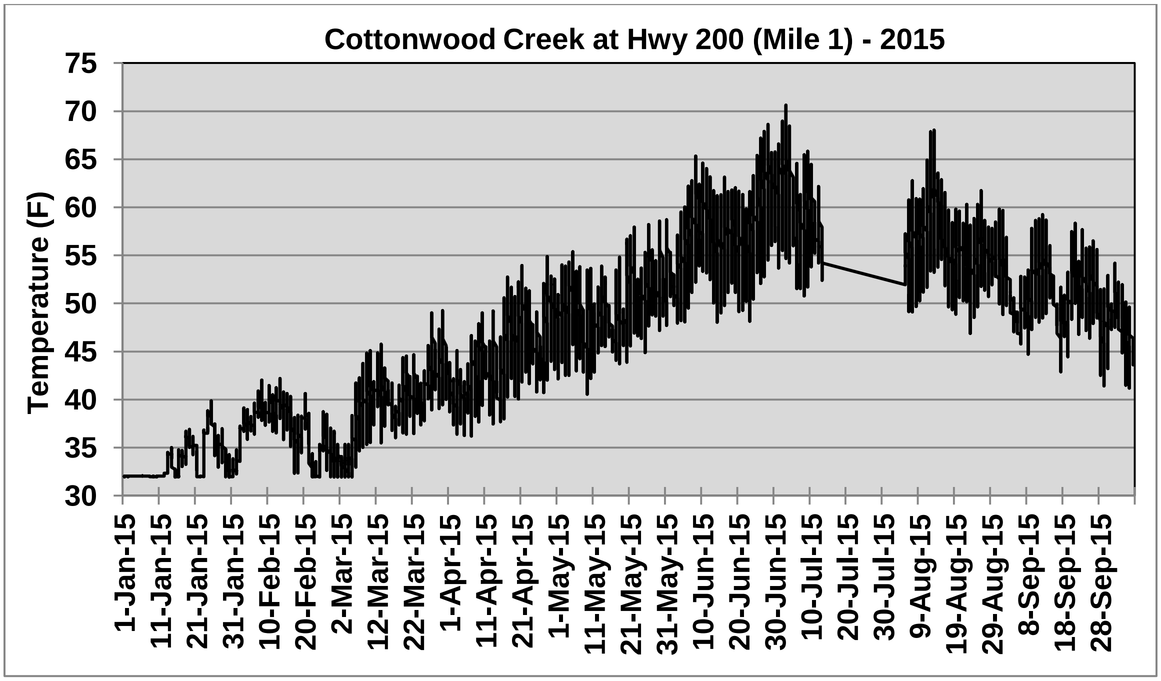
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.8	32.0	32.9	1.3	1.6
February	40.8	32.0	35.6	2.2	4.8
March	48.2	32.3	40.1	4.1	16.9
April	53.6	37.7	44.9	3.5	12.5
May	56.0	43.9	49.4	2.5	6.2
June	70.2	49.6	58.3	4.3	18.9
July	70.5	56.0	62.8	3.4	11.9
August	69.3	51.8	60.3	3.6	12.9
September	61.7	46.0	54.3	3.2	10.4
October	55.4	44.6	50.3	2.5	6.4



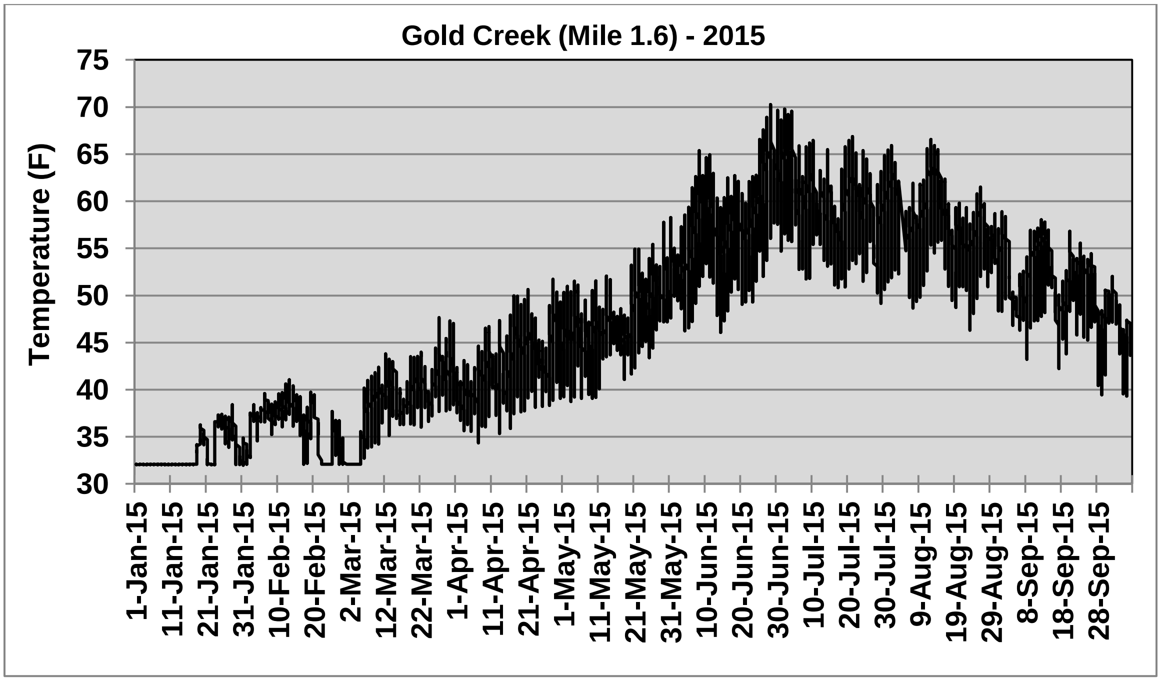
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	34.1	32.1	32.4	0.3	0.1
February	39.8	32.1	35.6	2.2	4.6
March	46.0	32.2	39.7	3.8	14.4
April	53.4	39.2	45.7	3.4	11.3
May	57.3	47.3	51.8	2.1	4.6
June	72.8	53.3	61.7	4.2	17.9
July	73.0	60.9	66.7	3.0	9.1
August	69.6	55.8	62.1	2.9	8.6
September	62.1	48.5	55.6	2.9	8.5
October	54.3	46.3	51.2	1.8	3.4



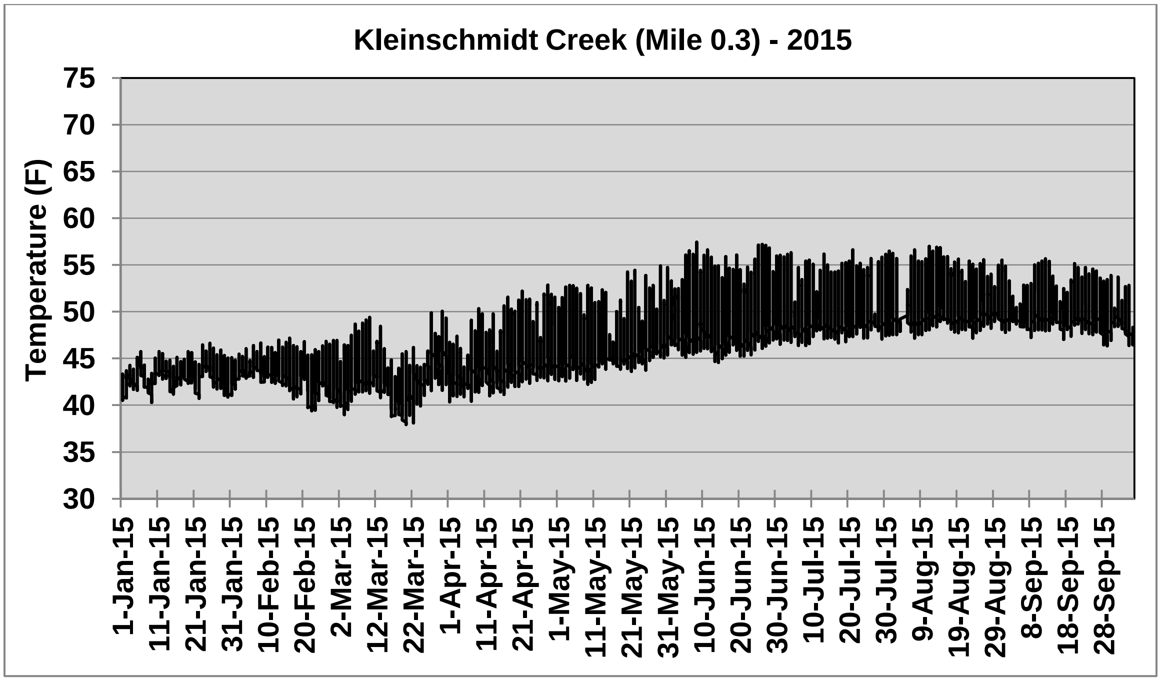
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.2	31.9	32.9	1.5	2.2
February	40.7	31.9	35.1	2.5	6.3
March	47.2	32.0	37.8	3.6	13.2
April	50.7	34.2	40.9	3.8	14.1
May	52.6	37.0	43.6	3.2	10.4
June	61.3	41.0	49.8	4.5	20.5
July	62.3	45.1	53.7	4.2	17.9
August	64.1	43.5	53.3	4.3	18.8
September	58.9	39.3	49.0	3.9	15.3
October	51.1	38.0	45.2	3.1	9.8



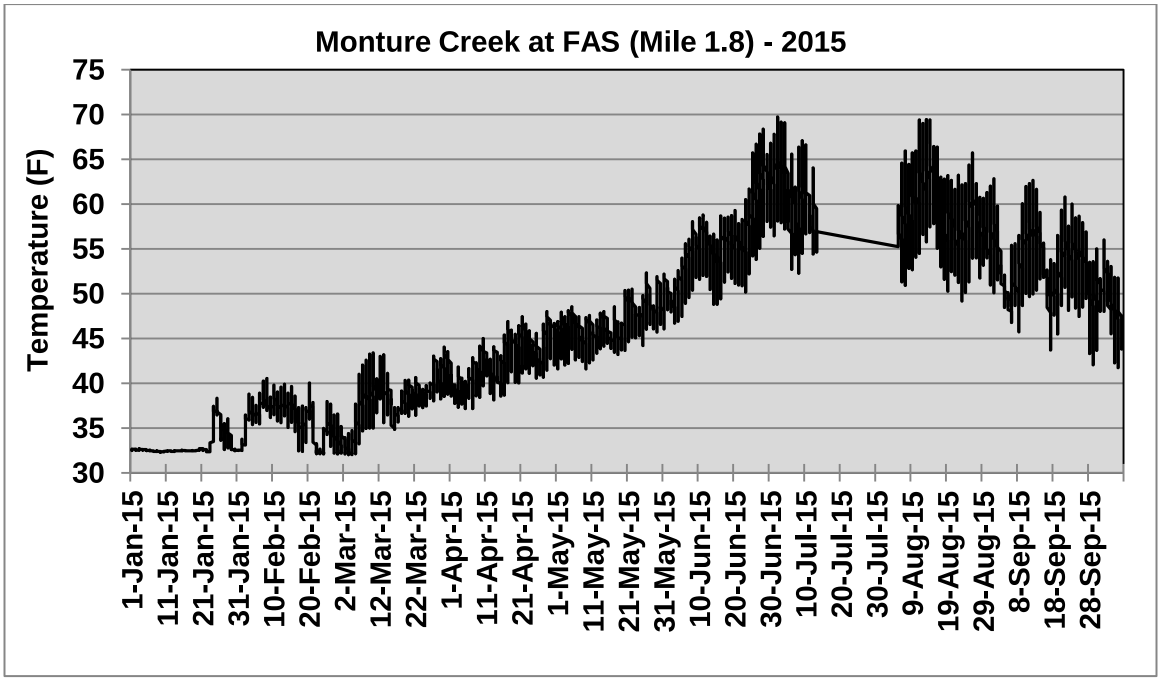
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	39.9	31.9	33.5	1.9	3.5
February	42.2	31.9	36.7	2.6	6.7
March	49.2	31.9	39.4	3.9	14.9
April	54.9	36.2	44.0	4.1	17.1
May	58.7	40.6	49.3	3.7	13.7
June	68.6	48.0	56.8	4.7	21.8
July	70.6	50.8	59.1	4.6	20.8
August	68.0	46.9	55.6	3.9	15.3
September	59.7	41.4	50.8	3.6	12.9
October	54.2	41.2	47.8	3.0	8.7



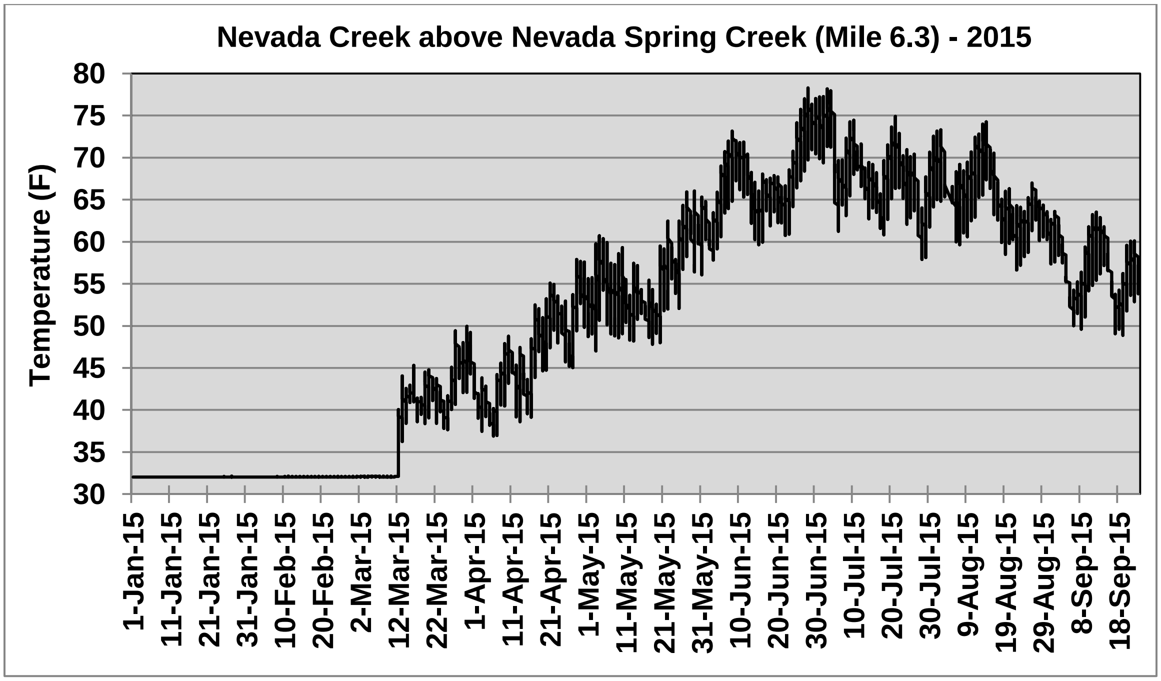
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	38.4	32.0	33.1	1.7	2.9
February	41.0	32.1	36.0	2.5	6.0
March	47.6	32.1	38.3	3.7	13.9
April	51.7	34.4	41.9	3.8	14.4
May	58.3	38.7	47.1	4.0	16.0
June	70.2	46.1	56.7	5.3	28.5
July	69.8	49.2	59.0	4.6	21.1
August	66.6	46.3	56.5	4.2	17.6
September	58.9	39.5	50.6	3.8	14.3
October	52.0	39.3	46.9	3.0	8.8



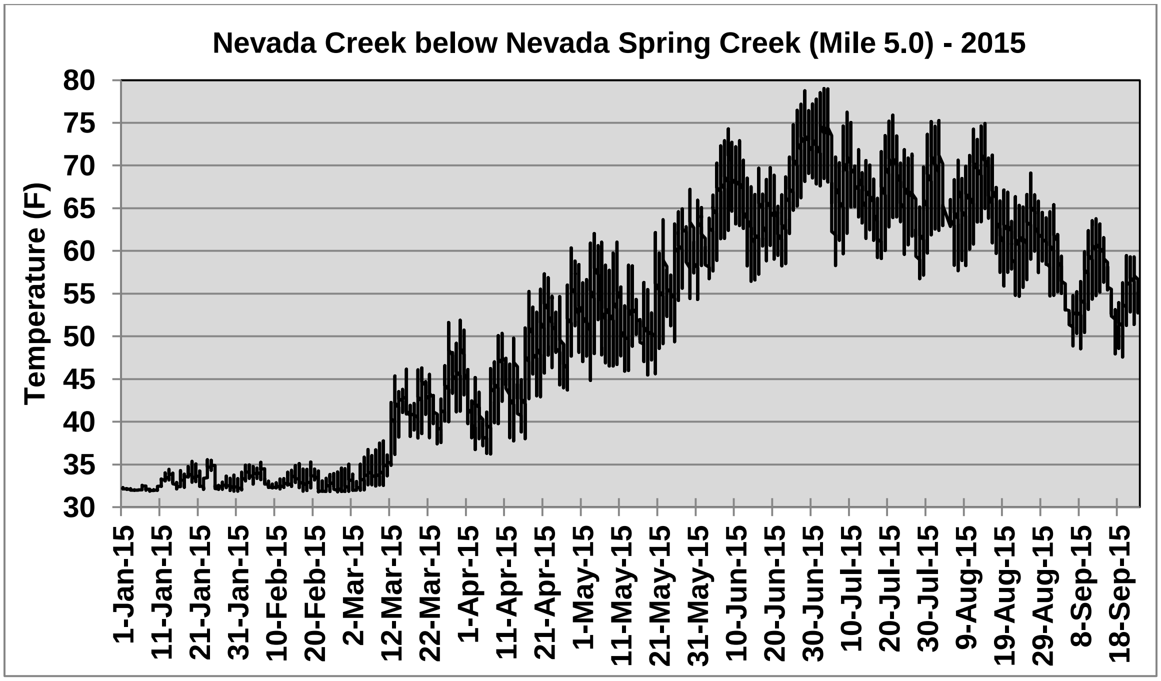
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	46.6	40.3	43.1	1.3	1.7
February	47.2	39.4	43.3	1.7	2.8
March	50.0	37.9	42.9	2.5	6.2
April	52.9	40.3	44.8	2.9	8.2
May	54.9	42.2	46.9	2.9	8.3
June	57.4	44.6	49.4	3.3	10.9
July	56.6	46.4	50.2	2.8	7.7
August	57.0	47.2	50.8	2.5	6.3
September	55.6	46.3	50.0	2.1	4.4
October	53.7	46.4	49.2	1.7	2.8



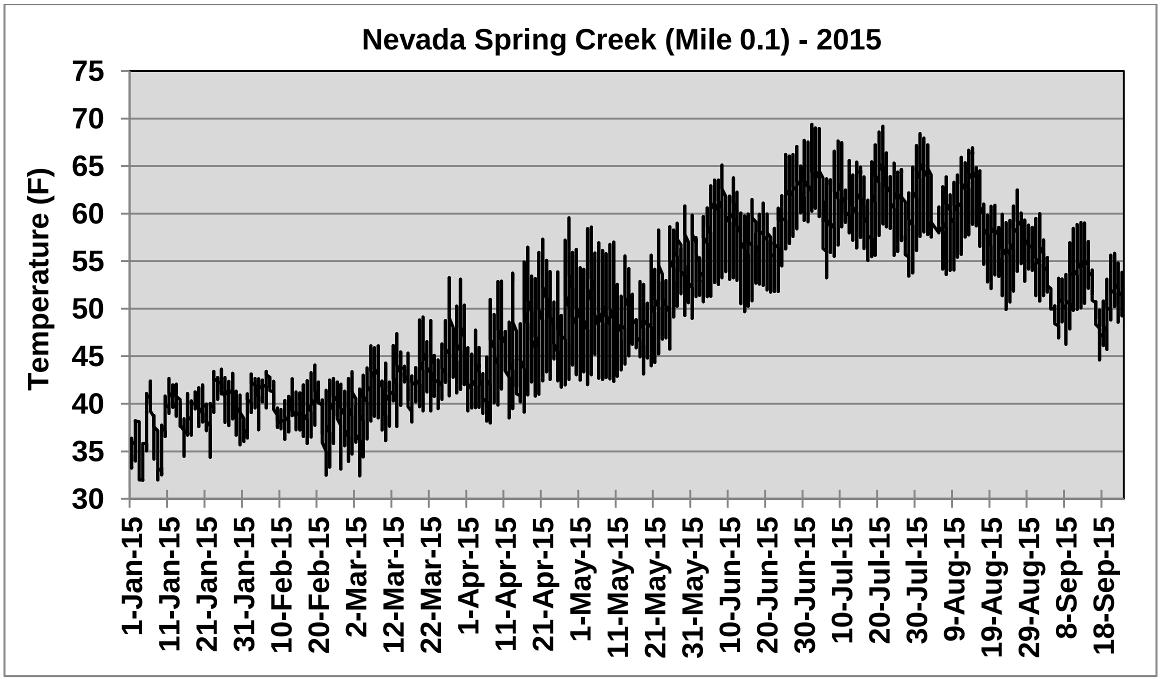
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	38.3	32.3	33.0	1.2	1.4
February	40.5	32.1	35.9	2.1	4.5
March	44.0	32.0	37.8	2.9	8.1
April	48.0	37.2	42.0	2.5	6.2
May	52.3	41.6	46.1	2.2	4.8
June	68.4	46.7	55.0	4.7	21.7
July	69.7	52.3	60.5	4.1	16.9
August	69.4	49.2	58.5	4.5	20.1
September	62.8	42.1	52.8	4.2	17.4
October	56.0	41.7	49.0	3.3	10.6



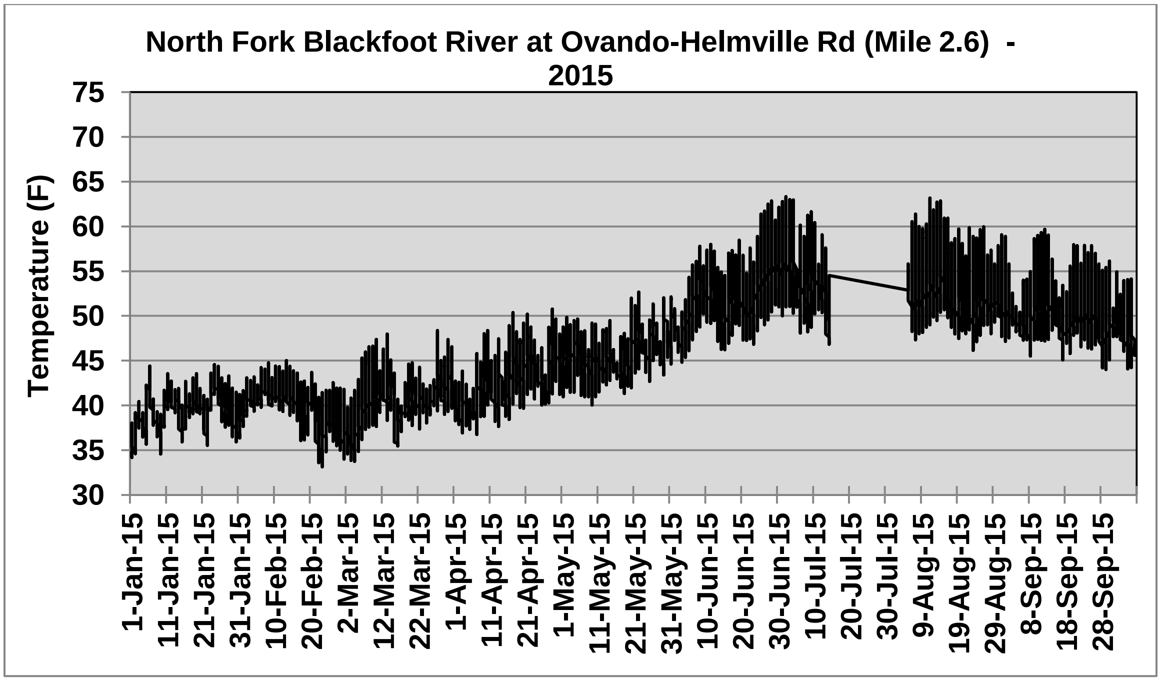
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	32.1	32.0	32.0	0.0	0.0
February	32.1	32.0	32.1	0.0	0.0
March	50.0	32.0	38.4	5.4	28.9
April	57.9	36.9	46.2	5.1	26.0
May	66.0	47.0	55.2	4.4	19.4
June	78.3	57.8	66.8	4.4	19.8
July	78.2	57.9	67.9	4.1	17.1
August	74.3	56.6	64.7	3.9	15.0
September	63.6	48.9	56.2	3.6	12.8



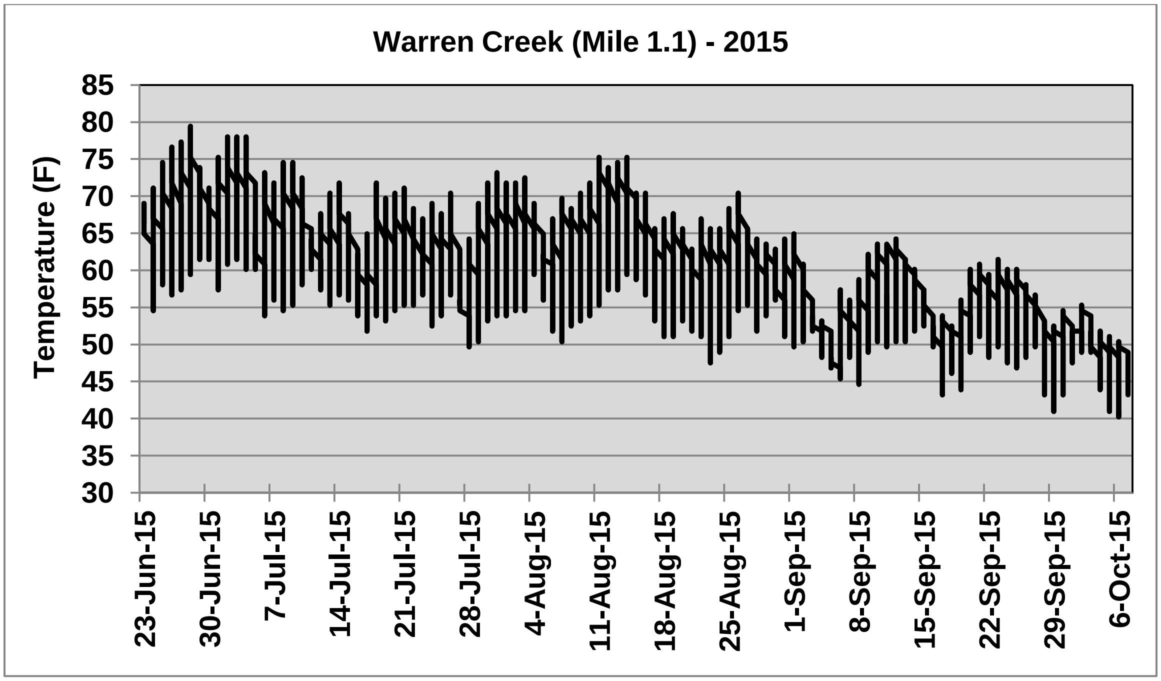
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	35.5	31.9	32.8	0.9	0.7
February	35.3	31.8	33.1	0.9	0.7
March	51.9	31.9	39.3	5.0	24.8
April	60.3	36.2	46.1	5.5	30.4
May	67.2	44.9	54.3	4.9	24.5
June	78.7	56.5	65.7	5.0	24.7
July	79.0	56.8	67.1	4.8	23.0
August	75.2	54.7	63.9	4.6	20.7
September	65.4	47.6	55.4	3.8	14.3



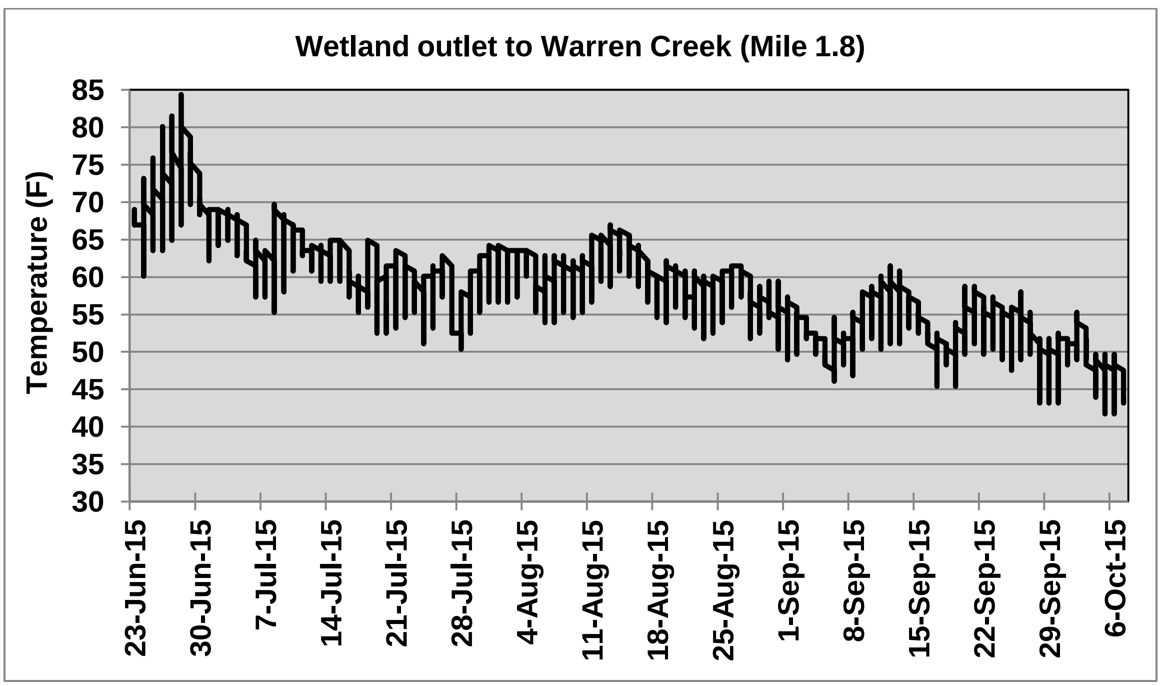
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	43.6	31.9	38.3	2.9	8.4
February	44.1	32.5	39.3	2.4	5.9
March	53.3	32.4	42.2	3.6	13.2
April	59.6	38.0	45.9	4.8	22.9
May	60.8	42.0	50.1	4.4	19.1
June	67.7	49.7	57.7	4.0	15.6
July	69.4	53.3	61.1	3.6	12.9
August	67.9	49.9	58.5	3.8	14.1
September	60.0	44.6	51.8	3.1	9.8



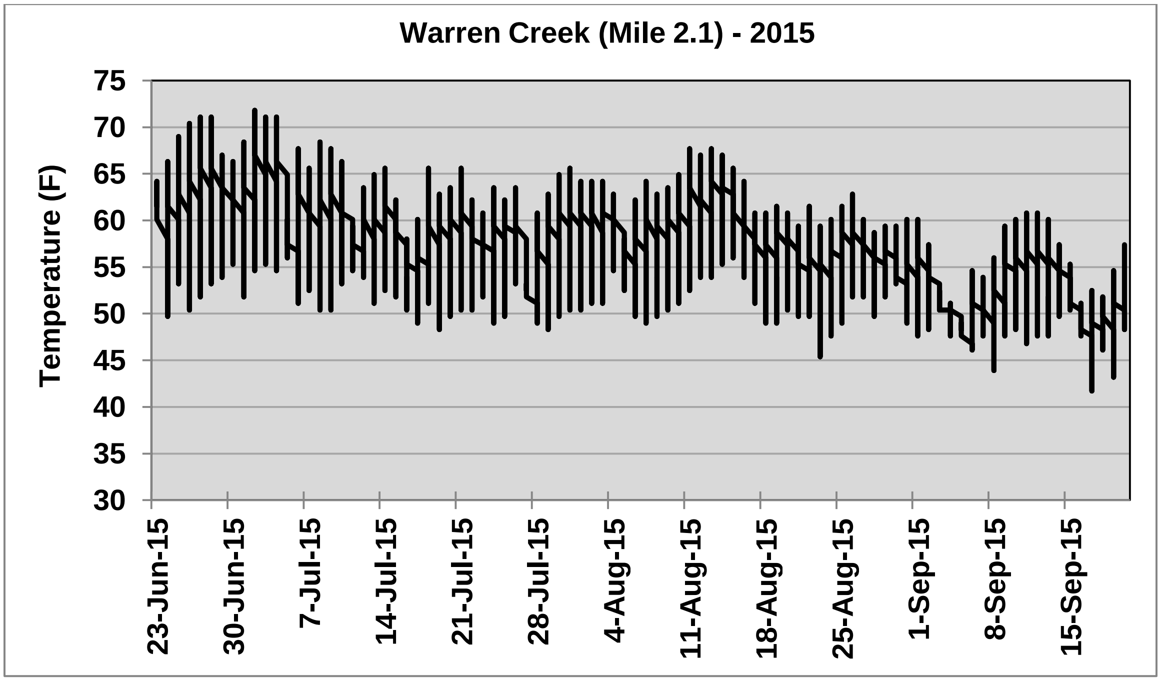
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	44.5	34.2	39.4	2.2	4.7
February	45.0	33.1	39.9	2.4	5.7
March	48.4	33.7	40.4	2.9	8.5
April	50.8	36.7	42.7	3.1	9.8
May	52.6	40.0	45.6	2.6	6.6
June	62.8	44.8	52.1	3.8	14.6
July	63.3	46.8	54.2	4.0	15.9
August	63.2	46.2	53.1	4.0	15.6
September	59.7	44.0	50.5	3.4	11.4
October	54.9	44.1	48.9	2.7	7.5



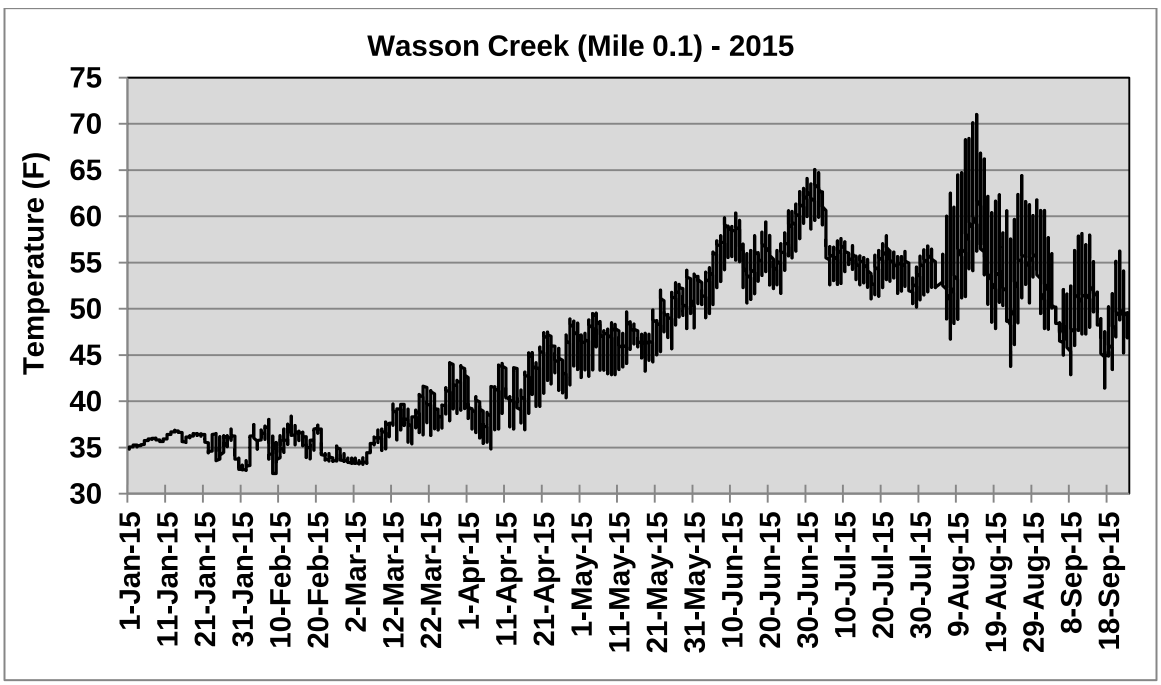
Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	79.4	54.6	66.5	6.1	37.8
July	78.0	49.7	62.4	6.2	38.5
August	75.2	47.5	60.6	6.1	37.0
September	64.9	41.0	52.7	4.8	23.2
October	55.3	40.2	48.2	3.6	13.0



Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	84.4	60.1	71.2	5.4	28.8
July	69.7	50.4	60.4	4.2	17.6
August	67.0	50.4	58.9	3.3	11.1
September	61.5	43.2	52.2	3.6	12.6
October	55.3	41.7	48.1	3.2	10.4



Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
June	71.1	49.7	60.5	5.8	33.3
July	71.8	48.3	57.7	5.2	27.3
August	67.7	45.4	56.5	4.6	21.1
September	60.8	41.7	51.2	4.0	15.6



Month	Max Temp	Min Temp	Avg Temp	StDev	Variance
January	37.0	32.6	35.6	1.0	0.9
February	38.4	32.2	35.2	1.4	1.9
March	44.2	33.2	37.5	2.6	7.0
April	48.9	34.9	41.5	3.2	10.5
May	54.2	42.6	47.4	2.5	6.3
June	64.1	49.0	55.8	3.1	9.5
July	65.1	50.2	55.2	3.0	9.0
August	71.0	43.8	55.7	4.8	23.5
September	60.6	41.4	49.8	3.5	12.2

Appendix F: Westslope cutthroat trout genetic sampling sites and results, 2013-2015

University of Montana Conservation Genetics Laboratory
Division of Biological Sciences, University of Montana, Missoula, Montana 59812
Phone (406) 243-6749 or 6725; Fax (406) 243-4184

Methods and Data Analysis

We developed a 'chip' specifically for analysis of westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) populations. This chip allows us to simultaneously genotype up to 95 single nucleotide polymorphic loci (SNPs) in 91 trout using a Fluidigm EP1 Genotyping System. Each SNP locus has only two states (alleles). Thus, considering hybridization among rainbow (*O. mykiss*), westslope cutthroat, and Yellowstone cutthroat trout (*O. c. bouvieri*) a single locus can only distinguish one of the taxa from the other two. In order to address hybridization issues among these fishes, therefore, each chip contained 19 loci that differentiate rainbow from westslope cutthroat and Yellowstone cutthroat trout (rainbow markers), 20 loci that distinguish westslope cutthroat from rainbow and Yellowstone cutthroat trout (westslope markers), and 20 loci that distinguish Yellowstone cutthroat from westslope cutthroat and rainbow trout (Yellowstone markers, Table 1). We verified the diagnostic property of each marker by analyzing them in reference samples that had previously been determined to be non-hybridized westslope cutthroat, Yellowstone cutthroat, or rainbow trout by analysis of allozymes, paired interspersed nuclear elements (PINEs), a combination of insertion/deletion (indel loci) events and microsatellite loci, or two or all of these techniques (Table 2).

If a sample possessed alleles characteristic of only westslope cutthroat trout at all westslope markers and had no alleles characteristic of rainbow trout at the rainbow markers or Yellowstone cutthroat trout at the Yellowstone markers, then it was considered to contain non-hybridized westslope cutthroat trout. Evidence for potential hybridization between rainbow and westslope cutthroat trout was generally considered to be present when three criteria were met. First, the sample had to contain alleles characteristic of rainbow trout at, at least, some of the rainbow markers. Next, at least some of the westslope markers also had to be genetically variable (polymorphic). Finally, no Yellowstone cutthroat trout alleles were detected at the Yellowstone markers. In this situation, the alleles at the rainbow markers shared between westslope cutthroat and Yellowstone cutthroat trout can confidently be assigned to having originated from westslope cutthroat trout and the alleles shared between rainbow and Yellowstone cutthroat trout at the westslope markers can confidently be assigned to having originated from rainbow trout. Thus, in terms of hybridization between westslope cutthroat and rainbow trout the data set contains information from 39 diagnostic loci. Likewise, when evidence of hybridization was detected only between westslope and Yellowstone cutthroat trout (no rainbow alleles at rainbow markers, at least some westslope markers polymorphic, and Yellowstone cutthroat trout alleles present at, at least, some Yellowstone markers) the data set contains information from 40 diagnostic loci. When all three sets of markers were polymorphic, this generally indicates hybridization among all three taxa. In this situation, the rainbow markers (19) provide information about rainbow trout hybridization and the Yellowstone markers (20) provide information about Yellowstone cutthroat trout hybridization.

An important aspect of SNPs is that they demonstrate a codominant mode of inheritance. That is, all genotypes are readily distinguishable from each other. Thus, at marker loci the genotype of individuals in a sample can directly be determined. From these data, the proportion of alleles from different taxa in the population sampled can be directly estimated at each marker locus analyzed. These values averaged over all marker loci yields an estimate of the proportion of alleles in the population that can be attributed to one or more taxa (proportion of admixture). In samples showing evidence of hybridization among all three taxa, we estimated the amount of rainbow trout admixture using only the 19 rainbow markers and the amount of Yellowstone cutthroat trout admixture using only the 20 Yellowstone markers. The amount of westslope cutthroat trout admixture was then estimated by subtracting the sum of the former two values from one. We used this procedure so the estimates would sum to one. Because of sampling error, it is unlikely that all three estimates from the marker loci would sum to one.

When evidence of hybridization is detected, the next issue to address is whether or not the sample appears to have come from a hybrid swarm. That is, a random mating population in which the alleles of the hybridizing taxa are randomly distributed among individuals such that essentially all of them are of hybrid origin.

A common, but not absolute, attribute of hybrid swarms is that allele frequencies at marker loci are similar among them because their presence can all be traced to a common origin or origins. Thus, one criterion we used for the assessment of whether or not a sample appeared to have come from a hybrid swarm was whether or not the allele frequencies among diagnostic loci reasonably conformed to homogeneity using contingency table chi-square analysis.

In order to determine whether or not alleles at the marker loci were randomly distributed among the fish in a sample showing evidence of hybridization, we calculated a hybrid index for each fish in the sample. The hybrid index for an individual was calculated as follows. At each marker locus, an allele characteristic of the native taxon was given a value of zero and an allele characteristic of the non-native taxon a value of one. Thus, at a single diagnostic locus the hybrid index for an individual could have a value of zero (only native alleles present, homozygous), one (both native and non-native alleles present, heterozygous), or two (only non-native alleles present, homozygous). These values summed over all diagnostic loci analyzed yields an individual's hybrid index. Considering westslope cutthroat and rainbow trout, therefore, non-hybridized westslope cutthroat trout would have a hybrid index of zero, non-hybridized rainbow trout a hybrid index of 78, F_1 (first generation) hybrids a hybrid index of 39, and post F_1 hybrids could have values ranging from zero to 78. The distribution of hybrid indices among the fish in a sample was statistically compared to the expected random binomial distribution based on the proportion of admixture estimated from the allele frequencies at the diagnostic loci. If the allele frequencies appeared to be statistically homogeneous among the marker loci and the observed distribution of hybrid indices reasonably conformed to the expected random distribution, then the sample was considered to have come from a hybrid swarm.

In old or hybrid swarms with small effective population size, allele frequencies at marker loci can randomly diverge from homogeneity over time because of genetic drift. In this case, however, the observed distribution of hybrid indices is still expected to reasonably conform to the expected random distribution. Thus, if the allele frequencies were statistically heterogeneous among the marker loci in a sample but, the observed distribution of hybrid indices reasonably conformed to the expected random distribution the sample was also considered to have come from a hybrid swarm.

The strongest evidence that a sample showing evidence of hybridization did not come from a hybrid swarm is failure of the observed distribution of hybrid indices to reasonably conform to the expected random distribution. The most likely reasons for this are that the population has only recently become hybridized or the sample contains individuals from two or more populations with different amounts of admixture. At times, previous samples and the distribution of genotypes at marker loci and the observed distribution of hybrid indices can provide insight into which of the latter two factors appears mainly responsible for the nonrandom distribution of the alleles from the hybridizing taxa among individuals in the sample. At other times, the distribution of genotypes at marker loci and the observed distribution of hybrid indices may provide little or no insight into the cause of the nonrandom distribution of alleles among individuals. The latter situation is expected to be fairly common as the two factors usually responsible for the nonrandom distribution of alleles are not necessarily mutually exclusive. Regardless of the cause, when alleles at the marker loci do not appear to be randomly distributed among individuals in a sample, estimating the amount of admixture has little if any biological meaning and, therefore, is generally not reported.

Failure to detect evidence of hybridization in a sample does not necessarily mean the population is non-hybridized because there is always the possibility that we would not detect evidence of hybridization because of sampling error. When no evidence of hybridization was detected in a sample, we assessed the likelihood the population is non-hybridized by determining the chances of not detecting as little as a 0.5 percent genetic contribution of a non-native taxon to a hybrid swarm. This is simply 0.995^{2NX} where N is the number of fish in the sample and X is the number of marker loci analyzed.

The chip also contained 34 loci that are generally polymorphic within westslope cutthroat trout populations. Information from these loci can be used to address issues concerning the relative amount of genetic variation within and divergence among westslope cutthroat trout populations.

Finally, the chip contained two mitochondrial DNA (mtDNA) loci that differentiate cutthroat and rainbow trout. Data from these loci were used only if an individual appeared to be an F_1 hybrid. Because mtDNA is inherited only from females (maternal inheritance), in this situation we can determine the taxon of the female, and by default the taxon of the male, that produced the hybrid.

When two or more samples were collected from the same area of a water body in different years or different reaches of a stream in the same year, we used the log likelihood G test of Goudet et al. (1996) in GENEPOP version 4.2 (Rousset 2008) to test for genetic differences among the samples. In instances where multiple loci were compared among samples and some demonstrated significant differences, significance was determined using Rice’s (1989) method for correcting for multiple comparisons (modified level of significance). When no differences were detected at the modified level, any observed differences were considered to most likely represent chance departures from homogeneity and the samples were combined for further analysis. When evidence of genetic differences was detected between samples they were kept separate for analysis and the relative amount of divergence between them was estimated as F_{ST} using the method of Weir and Cockerham (1984) available in GENEPOP version 4.2.

In samples containing 10 or more individuals appearing to have come from non-hybridized westslope cutthroat trout populations, we compared the observed to the expected random mating genotypic proportions (Hardy-Weinberg proportions) at the polymorphic loci using the Markov Chain method of Guo and Thompson (1992) available in GENEPOP version 4.2. A deficit of observed heterozygotes can arise in a sample if it contains individuals from two or more genetically divergent populations or is experiencing a fair to high amount of inbreeding. Conversely, a population produced from a very small number of parents may show an excess of heterozygotes compared to expected random mating proportions (Pudovkin et al. 1996, 2010; Luikart and Cornuet 1999). Since multiple comparisons were performed in most cases, significance was again determined at the modified level. In cases showing a significant departure from expected Hardy- Weinberg genotypic proportions because of a tendency for there to be an excess of heterozygotes, we used the program ML-RELATE of Kalinowski et al. (2006) to estimate the degree of relationship among the fish in the sample as this could possibly provide some insight into the cause for the deviations.

Results and Discussion

Sample #	Water Name/Location/ Collection Date/ Collector	a N	b #Markers	c Taxa ID	d Power	e %WCT	f # Fish
4478	Gleason Creek above and below culvert T12N R8W S22B 46.78407-78152 112.59690-59241 Mile 0.10-0.15 10/3/2012 Ron Pierce	50	R19W20Y20	WCT X RBT WCT X RBT		W99.8 X R0.2	48 2
4700	Indian Meadows T15N R8W S12B 47.0704 112.56097 Mile 0.4 Ron Pierce	24	R19W19Y19	WCT?	R99Y99		
4753	Sucker Creek 46.98910-00255 112.64864-64140 Mile 2.6 and 3.8 7/9/2014 Ron Pierce	9	R19W20Y20	WCT	R99Y99		
4754	Klondike Creek 47.01562-01645 112.75661-75603 Mile 0.1 and 0.15 7/7/2014 Ron Pierce	10	R19W19Y20	WCT	R99Y99		
4755	Keep Cool Creek 46.97188-98730 112.62686-62083 Mile 7.7 and 8.9 7/8/2014 Ron Pierce	11	R19W20Y20	WCT WCT X RBT	R99Y99		10 1
4756	Yukon Creek 47.02795-01921 112.77384-76720 Mile 1.0 and 0.1 7/29/14 and 7/7/14 Ron Pierce	9	R19W20Y20	WCT	R99Y99		
4757	Theodore Creek 47.00721-00954 112.74654-74516 Mile 0.1 and 0.2 7/7/14 and 7/30/14 Ron Pierce	10	R17W20Y20	WCT?			

4758	Bighorn Creek 47.19756-17573 112.64398-64992 Mile 2.0-3.6 8/11-13/14 Ron Pierce	29	R19W20Y20	WCT X RBT	W99.3 R0.7
4762	East Fork North Fork Blackfoot River 47.18352-154462 112.86468-76804 9/4/13 & 8/6/13 Ron Pierce	20	R19W20Y20	RBT X YCT X WCT	
4763	Blondie Creek 47.15471 112.74631 Mile 0.2 8/5/2013 Ron Pierce	10	R19W20Y20	RBT X YCT X WCT	
4764	Cooney Creek 47.25836 112.81499 Mile 0.2-0.4 9/5/2013 Ron Pierce	8	R19W20Y20	RBT X YCT	
4765	East Fork Meadow Creek 47.1183065 112.79995 Mile 0.8 8/7/2013 Ron Pierce	9	R19W20Y20	RBT X YCT X WCT	R88.3Y10.8W0.9
4766	Lost Pony Creek 47.1752811 112.79342 Mile 0.8 8/8/2013 Ron Pierce	10	R19W20Y20	RBT X YCT X WCT	R88.2Y9.3W2.5
4767	Spaulding Creek 47.1759062 112.8201 Mile 0.1 8/7/2013 Ron Pierce	2	R19W20Y20	RBT X YCT	
4770	Camp Creek 47.18376 112.86497 Mile 0.1 9/4/2013 Ron Pierce	10	R19W20Y20	RBT X YCT	R93.2Y6.8

[a]Number of fish successfully analyzed. If combined with a previous sample, the number in parentheses indicates the combined sample size.
[b]Number of diagnostic loci analyzed for the taxon (R=rainbow trout *Oncorhynchus mykiss,* W=westslope cutthroat trout *O. clarkii lewisi,* Y=Yellowstone cutthroat trout *O. c. bouvieri*).
[c]Taxa: WCT = westslope cutthroat trout; RBT = rainbow trout; YCT = Yellowstone cutthroat trout . Only one taxon code is listed if the sample was considered to contain only individuals from it. However, we cannot definitely rule out the possibility that some or all of the individuals are hybrids. We may not have detected any evidence of hybridization at the loci analyzed because of sampling error (see *d*). Taxa separated by "x" indicate hybridization between them was detected.
[d]Power: the number corresponds to the percent chance we have to detect 0.5% introgression in a hybrid swarm (a random mating population in which taxa markers are randomly distributed among individuals such that essentially all of them in the population are of hybrid origin) given the number of individuals and diagnostic markers analyzed. For example, with 12 individuals we have better than a 95 % chance to detect as little as a 0.5% rainbow (39 diagnostic loci) or Yellowstone cutthroat trout (40 diagnostic loci) genetic contribution to a hybrid swarm that once was a non-hybridized westslope cutthroat trout population Not reported when hybridization is detected. Taxa as in b.
[e]Indicates the genetic contribution of the hybridizing taxa (amount of admixture) denoted as in b. This number is usually reported only if the sample appears to have come from a hybrid swarm.
[f]Indicates the number of individuals with genetic characteristics corresponding to the taxa ID code column when the sample contains individuals from two or more genetically distinct groups.

Gleason Creek above and below culvert 4478

Samples were collected from above and below a road culvert in Gleason Creek. Between the samples, 63 loci were polymorphic. The allele frequencies significantly differed between the samples at three of these loci. At the modified level of significance, however, these differences were not significant suggesting their most likely represented chance departures from homogeneity. Since there was no conclusive evidence of genetic differences between the samples, they were combined for further analysis.

In the sample from Gleason Creek, alleles characteristic of rainbow trout were detected at 16 of the rainbow and 15 of the westslope markers that were analyzed. No alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers. Although the allele frequencies were statistically homogeneous (X^2_{38}=25.497; P>0.90) among the rainbow and westslope markers, the rainbow trout alleles were not randomly distributed (X^2_3=52.833; P<0.001) among the fish in the sample. The nonrandom distribution, however, appeared to mainly be due to the presence of two fish one with a hybrid index of 20 and the other 21 (Figure 1). When these fish were removed from the data, the rainbow trout alleles appeared to be randomly distributed (X^2_1=1.441; P>0.10) among the remaining fish. This sample, therefore, appears to have contained a mixture of trout from a hybrid swarm between westslope cutthroat and rainbow trout with a predominant westslope cutthroat trout (0.998) genetic component and two hybrids with a much higher amount of admixture. Both of the latter fish were collected below the culvert.

When the two individuals with unusually high hybrid indices are eliminated from the data, there was some indication that the observed genotypic distributions significantly deviated from expected random mating proportions. Out of 34 meaningful comparisons to Hardy-Weinberg proportions, four were statistically significant. These differences remained significant at the modified level with three involving an excess of heterozygotes and one a deficit. Thus, considering the significant differences there did not appear to be a trend for there to be either an excess or deficit of heterozygotes. This was also apparent when all loci were considered as 17 had an excess and 17 a deficit of heterozygotes. Since there was no apparent tendency for there to be either an excess or deficit of heterozygotes, it is unclear biologically what the significant departures from expected random mating genotypic proportions in the sample indicate.

Indian Meadows 4700

No alleles characteristic of rainbow trout were detected at the rainbow markers analyzed in the sample. Among the westslope markers, only alleles characteristic of westslope cutthroat trout were detected in the sample except at *OclWD_P53_307Kal*. At this locus, a single copy of the allele usually characteristic of rainbow and Yellowstone cutthroat trout was detected. Likewise, no alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers analyzed except *OclYGD106457_Garza* were a single copy of the allele usually characteristic of Yellowstone cutthroat trout was detected. The latter allele we strongly feel represents westslope cutthroat trout genetic variation as it has been detected in many other populations that otherwise appear to be non-hybridized westslope cutthroat trout (Table 3). We are not so sure about the variation detected at *OclWD_P53_307Kal* as this locus has only been observed to be variable in one other population (#4658, Crawford Creek, frequency=0.442) that otherwise appears to be non-hybridized westslope cutthroat trout. With this uncertainty we suggest the fish in Indian Meadows be considered non-hybridized westslope cutthroat trout but, conservatively suggest that they note be used for broodstock purposes.

In the sample, 22 loci allowed a meaningful comparison of observed to expected Hardy-Weinberg genotypic proportions. At three of these loci, the observed genotypic proportions significantly differed from the expected random mating distribution. These differences, however, were not significant at the modified level. Thus, there was no conclusive evidence that the observed genotypic proportions in the sample significantly deviated from expected random mating proportions and that the sample contained fish from more than a single random mating population.

Sucker Creek 4753

Fish were collected from stream mile 2.6 and 3.8 in Sucker Creek. Between the samples, 30 loci were polymorphic. The allele frequencies were statistically heterogeneous between the samples at four of these loci. These differences, however, were not significant at the modified level indicating they most likely represented chance departures from homogeneity rather than evidence of genetic differences between the samples. Thus, the samples were combined for subsequent analysis.

In the sample from Sucker Creek, no alleles characteristic of rainbow trout were detected at the rainbow markers, only alleles characteristic of westslope cutthroat trout were detected at the westslope markers, and no alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers. With the 702 rainbow and 720 Yellowstone cutthroat trout diagnostic alleles analyzed, we had better than a 97 percent chance of detecting as little as a 0.5% rainbow or Yellowstone cutthroat trout genetic contribution to a hybrid swarm that once was a non-hybridized westslope cutthroat trout population. This sample, therefore, very likely contained non-hybridized westslope cutthroat trout.

Klondike Creek 4754

Samples were collected from stream mile 0.10 and 0.15 in Klondike Creek. Between the samples, 32 loci were polymorphic. The allele frequencies were statistically heterogeneous between the samples at two of these loci. These differences, however, were not significant at the modified level indicating they most likely represented chance departures from homogeneity rather than evidence of genetic differences between the samples. The samples, therefore, were combined for further analysis.

No alleles characteristic of rainbow trout were detected at the rainbow markers, only alleles characteristic of westslope cutthroat trout were detected at the westslope markers except *OmyWD_RAD_55391_Hoh*, and no alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers in the sample from Klondike Creek. At *OmyWD_RAD_55391_Hoh*, five alleles usually characteristic of either rainbow or Yellowstone cutthroat trout were detected. We believe the variation detected at *OmyWD_RAD_55391_Hoh* more likely represents westslope cutthroat trout genetic variation rather than evidence of hybridization because this locus has been found to be polymorphic in other samples that otherwise appear to be non-hybridized westslope cutthroat trout (Table 3). With the 760 rainbow and 780 Yellowstone cutthroat trout diagnostic alleles analyzed, we had about a 98 percent chance of detecting as little as a 0.5% rainbow or Yellowstone cutthroat trout genetic contribution to a hybrid swarm that once was a non-hybridized westslope cutthroat trout population. Thus, the fish sampled from Klondike Creek were very likely non-hybridized westslope cutthroat trout.

There were 31 loci in the sample that allowed a meaningful comparison of observed to expected Hardy-Weinberg genotypic proportions. At five of these loci, the observed genotypic proportions significantly differed from expected random mating proportions. These differences, however, were not significant at the modified level. Thus, they most likely represented chance departures from conformity to expected random mating genotypic proportions. There was no compelling evidence, therefore, that this sample contained individuals from more than one essentially random mating population.

Keep Cool Creek 4755

Fish were collected from stream mile 7.7 and 8.9 in Sucker Creek. Between the samples, 34 loci were polymorphic. The allele frequencies were statistically homogeneous between the samples at all of these loci. Thus, there was no evidence of genetic differences between the samples and they were combined for subsequent analysis.

Alleles characteristic of rainbow trout were detected at one of the rainbow markers, two of the westslope markers were polymorphic, and no alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers in the sample from Keep Kool Creek. The rainbow trout alleles were all detected in one fish collected from stream mile 8.9. This fish, therefore, was almost certainly of hybrid origin between westslope cutthroat and rainbow trout. The other fish in the sample appeared to be non-hybridized westslope cutthroat trout.

Among the westslope cutthroat trout in the sample, 26 loci allowed a meaningful comparison of observed to expected Hardy-Weinberg genotypic proportions. The observed genotypic proportions statistically conformed to expected random mating proportions at all of these loci. Thus, there was no evidence that this sample contained individuals from more than one essentially random mating population.

Yukon Creek 4756

Samples were collected from stream mile 0.1 and 1.0 in Yukon Creek. Between the samples, 31 loci were polymorphic. The allele frequencies were statistically heterogeneous between the samples at three of these loci. These differences, however, were not significant at the modified level indicating they most likely represented chance departures from homogeneity rather than evidence of genetic differences between the samples. The samples, therefore, were combined for further analysis.

In the sample from Yukon Creek, no alleles characteristic of rainbow trout were detected at the rainbow markers, only alleles characteristic of westslope cutthroat trout were detected at the westslope markers, and no alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers. With the 702 rainbow and 720 Yellowstone cutthroat trout diagnostic alleles analyzed, we had better than a 97 percent chance of detecting as little as a 0.5% rainbow or Yellowstone cutthroat trout genetic contribution to a hybrid swarm that once was a non-hybridized westslope cutthroat trout population. This sample, therefore, very likely contained non-hybridized westslope cutthroat trout.

Theodore Creek 4757

Fish were collected from stream mile 0.1 and 0.2 in Theodore Creek. Between the samples, 33 loci were polymorphic. The allele frequencies were statistically homogeneous between the samples at all of these loci. Thus, there was no evidence of genetic differences between the samples and they were combined for subsequent analysis.

Alleles characteristic of rainbow trout were detected at two of the rainbow markers, only alleles characteristic of westslope cutthroat trout were detected at the westslope markers, and no alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers in the sample from Theodore Creek. We are not certain if the single "rainbow trout" allele detected at *OmyRD_RAD_77157_Hoh* and *OmyRD_ RAD_20663_Hoh* represents evidence of hybridization with rainbow trout or simply westslope cutthroat trout genetic variation as each locus was heterozygous in a different individual. With this uncertainty, we suggest that the fish from Theodore Creek be considered to be non-hybridized westslope cutthroat trout. The trout, however, should not be used for broodstock or transfer purposes until their genetic status is better determined.

There were 30 loci in the sample that allowed a meaningful comparison of observed to expected Hardy-Weinberg genotypic proportions. At one of these loci, the observed genotypic proportions significantly differed from expected random mating proportions. This difference, however, was not significant at the modified level. Thus, it most likely represented a chance departure from conformity to expected random mating genotypic proportions. There was no compelling evidence, therefore, that this sample contained individuals from more than one essentially random mating population.

Bighorn Creek 4758

Samples were collected from stream mile 2.0, 2.4, and 3.6 in Bighorn Creek. Among the samples, 28 loci were polymorphic. The allele frequencies were statistically heterogeneous among the samples at four of these loci. These differences, however, were not significant at the modified level indicating they most likely represented chance departures from homogeneity rather than evidence of genetic differences between the samples. The samples, therefore, were combined for further analysis.

Alleles characteristic of rainbow trout were detected at one of the rainbow markers, two of the westslope markers were polymorphic, and no alleles characteristic of Yellowstone cutthroat trout were detected at the Yellowstone markers in the sample from Bighorn Creek. This sample, therefore, probably contains evidence of hybridization between westslope cutthroat and rainbow trout. The proportion of rainbow trout alleles in the sample (0.007) was too small to allow for any further analysis of the extent of apparent hybridization except to note that the alleles were detected in multiple individuals and none had more than two.

The above results differ from those obtained from a previous PINE analysis (#1349, col. 7/14/98, T17N R8W S32, N=25) of trout collected from Bighorn Creek. These data suggested the fish were non-hybridized westslope cutthroat trout. With the 300 rainbow trout diagnostic PINE alleles analyzed, however, we had about a 12 percent chance of not detecting a 0.007 rainbow trout genetic contribution to a hybrid swarm with westslope cutthroat trout. Thus, we can not conclude with any certainty that the genetic characteristics of the trout in Bighorn Creek have changed between 1998 and 2014.

East Fork North Fork Blackfoot River 4762

Fish were collected from stream mile 1.9 and 9.0 in the East Fork of the North Fork Blackfoot River. Between the samples, 38 loci were polymorphic. The allele frequencies were statistically heterogeneous between the samples at two of these loci. These differences, however, were not significant at the modified level indicating they most likely represented chance departures from homogeneity rather than evidence of genetic differences between the samples. The samples, therefore, were combined for further analysis.

Alleles characteristic of rainbow trout were detected at all of the rainbow markers, 18 of the westslope markers were polymorphic, and alleles characteristic of Yellowstone cutthroat trout were detected at all of the Yellowstone markers in the sample from the East Fork of the North Fork Blackfoot River. This sample, therefore, provided conclusive evidence of hybridization among westslope cutthroat, rainbow, and Yellowstone cutthroat trout. Although the Yellowstone cutthroat (X^2_{19}=22.115, P>0.10) and westslope cutthroat trout (X^2_{19}=11.725, P>0.50) allele frequencies were statistically homogeneous among the respective markers, the Yellowstone cutthroat (X^2_{8}=41.184, P<0.001, Figure 1) and westslope cutthroat trout (X^2_{5}=46.967, P<0.001, Figure 2) alleles did not appear to be randomly distributed among the fish in the sample. The non-random distribution of the Yellowstone cutthroat trout alleles was due to two fish with a hybrid index of ten or 11. When these two individuals are removed from the data, the Yellowstone cutthroat trout alleles appear to be randomly distributed (X^2_{5}=5.599, P>0.50) among the remaining fish. The non-random distribution of the westslope cutthroat trout alleles was due to three fish with a hybrid index of five, six, or 11. When these fish are removed from the data, the westslope cutthroat trout alleles appear to be randomly distributed (X^2_3=7.237, P>0.0.05) among the remaining fish. Considering both the westslope and Yellowstone markers, only one fish in the sample showed no evidence of hybridization. The fish in the East Fork of the North Fork Blackfoot River, therefore, should simply be considered to be hybrids among rainbow, Yellowstone cutthroat, and westslope cutthroat trout with a major rainbow (about 87 percent) and minor Yellowstone (about nine percent) and westslope cutthroat trout (about four percent) genetic contribution.

The East Fork of the North Fork Blackfoot River was previously sampled twice. Allozyme (#1203, col. 8/1/96, T16N R10W S1, N=9) and indel/microsatellite (#3360, col. 7/11/06, T16N R9W S7, N=5) analyses also indicated the fish to be hybrids among westslope cutthroat, rainbow, and Yellowstone cutthroat trout with a predominant rainbow trout genetic component.

Blondie Creek 4763

In the sample from Blondie Creek, alleles characteristic of rainbow trout were detected at all of the rainbow markers, alleles characteristic of westslope cutthroat trout were detected at only four of the westslope markers, and alleles characteristic of Yellowstone cutthroat trout were detected at all of the Yellowstone markers. This sample, therefore, provided conclusive evidence of hybridization among westslope cutthroat, rainbow, and Yellowstone cutthroat trout. The Yellowstone cutthroat (X^2_{19}=31.561, P<0.05) and rainbow trout (X^2_{18}=65.332, P<0.001) allele frequencies were statistically heterogeneous among the respective markers and the alleles were clearly not randomly distributed among the fish in the sample (Figures 3 and 4). All of the fish in the sample, however, were definitely of hybrid origin among all three taxa. The trout in Blondie Creek, therefore, should simply be considered to be hybrids among rainbow, Yellowstone cutthroat, and westslope cutthroat trout with a major rainbow (about 50 percent) and Yellowstone cutthroat (about 48 percent) and minor westslope cutthroat trout (about two percent) genetic contribution.

Cooney Creek 4764

Alleles characteristic of rainbow trout were detected at all of the rainbow markers, no alleles characteristic of westslope cutthroat trout were detected at the westslope markers, and alleles characteristic of Yellowstone cutthroat trout were detected at 13 of the Yellowstone markers in the sample from Cooney Creek. Thus, this sample provided good evidence of hybridization between rainbow and Yellowstone cutthroat trout. Although the Yellowstone cutthroat and rainbow trout allele frequencies were statistically heterogeneous (X^2_{38}=71.423, P<0.001) among the markers all fish in the sample were definitely of hybrid origin (Figure 5). The fish in Cooney Creek, therefore, should simply be considered to be hybrids between rainbow and Yellowstone cutthroat trout with a major rainbow trout (about 88 percent)) genetic contribution.

The above results are in stark contrast to those obtained from an indel/microsatellite analysis of a single trout collected from Cooney Creek (#3510, col. 7/12/07, 47.258 112.81). This fish was believed to be a rainbow trout but, the results suggested it was a non-hybridized westslope cutthroat trout. If such fish still persist in Cooney Creek, the recent results suggest that they are relatively uncommon.

East Fork Meadow Creek 4765

In the sample from East Fork Meadow Creek, alleles characteristic of rainbow trout were detected at all of the rainbow markers, alleles characteristic of westslope cutthroat trout were detected at only two of the westslope markers, and alleles characteristic of Yellowstone cutthroat trout were detected at 18 of the Yellowstone markers. This sample, therefore, provided evidence of hybridization among westslope cutthroat, rainbow, and Yellowstone cutthroat trout. Ignoring the very small (0.009) westslope cutthroat trout genetic component, the Yellowstone cutthroat trout allele frequencies were statistically homogeneous (X^2_{38}=51.216, P>0.05) among the rainbow and Yellowstone markers and all the fish in the sample were definitely of hybrid origin (Figure 6). East Fork Meadow Creek, therefore, should be considered to contain hybrids among rainbow, Yellowstone cutthroat, and westslope cutthroat trout with a substantial rainbow (about 88 percent) and minor Yellowstone (about 11 percent) and westslope cutthroat trout genetic contribution.

The above results are highly concordant with those obtained from a previous indel/microsatellite analysis of a sample of trout collected from East Fork Meadow Creek (#3858, col. 9/7/08, 47.11820 112. 80013, N=4). This analysis also indicated the creek contained hybrids among rainbow, Yellowstone cutthroat, and westslope cutthroat trout with a substantial rainbow and minor Yellowstone and westslope cutthroat trout genetic component.

Lost Pony Creek 4766

Alleles characteristic of rainbow trout were detected at all of the rainbow markers, alleles characteristic of westslope cutthroat trout were detected at seven of the westslope markers, and alleles characteristic of Yellowstone cutthroat trout were detected at ten of the Yellowstone markers in the sample from Lost Pony Creek. This sample, therefore, provided good evidence of hybridization among rainbow, Yellowstone cutthroat, and westslope cutthroat trout. Disregarding the small (0.025) westslope cutthroat trout genetic component, the Yellowstone cutthroat trout allele frequencies were statistically heterogeneous (X^2_{38}=89.392, P<0.001) among the rainbow and Yellowstone markers. All fish in the sample, however, were definitely of hybrid origin (Figure 7). Thus, Lost Pony Creek should simply be considered to contain hybrids among rainbow, Yellowstone cutthroat, and westslope cutthroat trout with a substantial rainbow (about 88 percent) and minor Yellowstone (about 9.5 percent) and westslope cutthroat trout genetic contribution.

The above results are highly concordant with those obtained from a previous indel/microsatellite analysis of a sample of trout collected from Lost Pony Creek (#3363, col. 7/11/06, T16N R10W S1 47.173 112. 796, N=5). This analysis also indicated the creek contained hybrids among rainbow, Yellowstone cutthroat, and westslope cutthroat trout with a substantial rainbow and minor Yellowstone and westslope cutthroat trout genetic component.

Spaulding Creek 4767

Only two trout were collected from Spaulding Creek. Both fish were definitely of hybrid origin between rainbow and Yellowstone cutthroat trout with hybrid indices calculated using only the Yellowstone cutthroat trout alleles at the rainbow and Yellowstone markers of four and five.

Camp Creek 4770

Alleles characteristic of rainbow trout were detected at all of the rainbow markers, no alleles characteristic of westslope cutthroat trout were detected at the westslope markers, and alleles characteristic of Yellowstone cutthroat trout were detected at six of the Yellowstone markers in the sample from Camp Creek. This sample, therefore, provided conclusive evidence of hybridization between rainbow and Yellowstone cutthroat trout. Although the Yellowstone cutthroat trout allele frequencies were statistically heterogeneous (X^2_{38}=148.388, P<0.001) among the rainbow and Yellowstone markers, the Yellowstone cutthroat trout alleles appeared to be randomly distributed (X^2_{10}=6.338, P>0.50) among the fish in the sample. Furthermore, all of the fish in the sample were definitely of hybrid origin (Figure 8). Camp Creek, therefore, appears to contain a hybrid swarm between rainbow and Yellowstone cutthroat trout with a predominant (0.932) rainbow trout genetic component.

Literature Cited

Amish, S. J., P. A. Hohenlohe, S. Painter, R. F. Leary, C. Muhlfeld, F. W. Allendorf, and G. Luikart. 2012. RAD sequencing yields a high success rate for westslope cutthroat and rainbow trout species-diagnostic SNP assays. Molecular Ecology Resources 12:653-660.

Campbell, N. R., S. J. Amish, V. L. Pritchard, K. S. McKelvey, M. K. Young, M. K. Schwartz, J. C. Garza, G. Luikart, and S. R. Narum. 2012. Development and evaluation of 200 novel SNP assays for population genetic studies of westslope cutthroat trout and genetic identification of related taxa. Molecular Ecology Resources 12:942-949.

Finger, J. A., M. R. Stephens, N. W. Clipperton, and B. May. 2009. Six diagnostic single nucleotide polymorphism markers for detecting introgression between cutthroat and rainbow trouts. Molecular Ecology Resources 9:759-763.

Goudet, J., M. Raymond, T. deMeeus, and F. Rousset. 1996. Testing differentiation in diploid populations. Genetics 144:1933-1940.

Guo, S. W. and E. A. Thompson. 1992. Performing the exact test of Hardy-Weinberg proportions for multiple alleles. Biometrics 48:361-372.

Harwood, A. S., and R. B. Phillips. 2011. A suite of twelve single nucleotide polymorphism markers for detecting introgression between cutthroat and rainbow trout. Molecular Ecology Resources 11:382-385.

Kalinowski, S. T., B. J. Novak, D. P. Drinan, R. deM Jennings, and N. V. Vu. 2011. Diagnostic single nucleotide polymorphisms identifying westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), Yellowstone cutthroat trout (*Oncorhynchus clarkii bouvieri*) and rainbow trout (*Oncorhynchus mykiss*). Molecular Ecology Resources 11:389-393.

Kalinowski, S. T., A. P. Wagner, and M. L. Taper. 2006. ML-RELATE: a computer program for maximum likelihood estimation of relatedness and relationship. Molecular Ecology Notes 6:576-579.

Luikart, G. and J.-M. Cornuet. 1999. Estimating the effective number of breeders from heterozygote excess in progeny. Genetics 151:1211-1216.

Pudovkin, A. I., O. L. Zhdanova, and D. Hedgecock. 1996. On the potential for estimating the effective number of breeders from heterozygote excess in progeny. Genetics 144:383-387.

Pudovkin, A. I., O. L. Zhdanova, and D. Hedgecock. 2010. Sampling properties of the heterozygote-excess estimator of the effective number of breeders. Conservation Genetics 11:759-771.

Rice,W.R. 1989. Analyzing tables of statistical test. Evolution 43:223-225.

Rousset, F. 2008. GENEPOP'007: a complete re-implementation of the GENEPOP software for Windows and Linux. Molecular Ecology Resources 8:103-106.

Weir, B. S., and C. C. Cockerham. 1984. Estimating F-statistics for the analysis of population structure. Evolution 28:1358-1370.

Appendix G: Restoration Streams and Table of Activities.

Blackfoot River Basin

Stream Name	Fish passage improvement	Prevent Irrigation ditch losses	Spawning habitat protection	Channel restoration	Fish habitat improvement	Riparian vegetation improvement	Improve instream flows	Improve wetlands	Improve range/ riparian habitat	Improve irrigation	Conservation easements or other protection	Remove streamside feedlots
Alice Creek												
Anaconda Creek												
Arkansas Creek											X	
Arrastra Creek	X										X	
Ashby Creek	X	X		X	X	X	X	X	X	X	X	
Ashby Creek, East Fork											X	
Bartlett Creek												
Basin Spring Creek				X	X	X	X	X	X	X	X	X
Bear Creek (Blackfoot trib.at R.M. 12.2)	X			X	X	X	X		X	X	X	X
Bear Creek (Blackfoot trib.at R.M. 37.5)											X	
Bear Creek, East Fork (Blackfoot trib.at R.M. 37.5)											X	
Bear Creek (North Fork drainage)												
Bear Gulch											X	
Beaver Creek	X	X						X		X	X	
Beartrap Creek				x	x	x						
Belmont Creek	X		X						X		X	
Black Bear Creek												
Blackfoot River (mouth to Clearwater River)							X				X	
Blackfoot River (Clearwater River to N.F.)							X		X		X	
Blackfoot River (N.F. to Nevada Creek)								X	X		X	
Blackfoot River (Nevada Cr. to Arrastra Cr.)								X	X	X	X	X
Blackfoot River (Arrastra Cr. to Lincoln, MT)											X	
Blackfoot River (Lincoln, MT to Headwaters)											X	
Braziel Creek	X	X		X	X	X	X		X	X		X
Buffalo Gulch												
Burnt Bridge Creek											X	
California Gulch												
Camas Creek						X			X		X	
Chamberlain Creek	X	X	X	X	X	X	X	X	X	X	X	X
Chamberlain Creek, East Fork	X		X			X					X	
Chamberlain Creek, West Fork	X					X			X		X	
Chicken Creek											X	

Appendix G: Restoration Streams and Table of Activities (cont'd).

Blackfoot River Basin (cont'd)

Stream Name	Fish passage improvement	Prevent Irrigation ditch losses	Spawning habitat protection	Channel restoration	Fish habitat improvement	Riparian vegetation improvement	Improve instream flows	Improve wetlands	Improve range/ riparian habitat	Improve irrigation	Conservation easements or other protection*	Remove streamside feedlots
Chimney Creek (Douglas Cr tributary)												
Chimney Creek (Nevada Cr tributary)												
Clear Creek											X	
Clear Creek trib to Lincoln Gulch												
Coopers Lake trib												
Copper Creek												
Cottonwood Creek (Blackfoot trib.at R.M. 43)	X	X		X		X	X	X	X	X	X	X
Cottonwood Creek, North Fork (trib.at mile 13.4)											X	
Cottonwood Creek (Nevada Cr tributary)	X			X		X			X	X		X
Coyote Creek											X	
Devils Dip	X			X	X			X				
Dick Creek	X	X		X	X	X	X	X	X	X	X	X
Douglas Creek	X					X					X	
Dry Creek						X			X		X	
Dunham Creek		X		X	X	X				X	X	
East Twin Creek	X										X	
Elk Creek				X	X			X	X			X
Enders Spring Creek	X			X	X				X			
Finn Creek												
Fish Creek	X	X								X	X	
Frazier Creek	X	X		X							X	
Frazier Creek, North Fork											X	
Gallagher Creek												
Game Creek											X	
Gleason Creek	X											
Gold Creek					X						X	
Gold Creek, West Fork												
Grantier Spring Creek	X		X	X	X	X		X	X		X	
Halfway Creek												
Hogum Creek												
Hoyt Creek	X			X	X	X	X	X	X	X	X	
Humbug Creek											X	
Humbug Creek, South Fork											X	
Humbug Creek, Midddle Fork to South Fork											X	
Humbug Creek, West Fork to South Fork											X	

Appendix G: Restoration Streams and Table of Activities (cont'd).

Blackfoot River Basin (cont'd)

Stream Name	Fish passage improvement	Prevent Irrigation ditch losses	Spawning habitat protection	Channel restoration	Fish habitat improvement	Riparian vegetation improvement	Improve instream flows	Improve wetlands	Improve range/ riparian habitat	Improve irrigation	Conservation easements or other protection	Remove streamside feedlots
Indian Creek												
Indian Creek, East Fork												
Indian Creek, West Fork												
Indian Meadows Creek												
Jacobsen Spring Creek	X			X	X	X	X	X	X	X	X	
Jacobsen Spring Creek, East Fork	X			X	X	X		X	X		X	
Jefferson Creek												
Johnson Creek	X										X	
Keep Cool Creek	X					X					X	
Kleinschmidt Creek	X		X	X	X	X		X	X		X	
Klondike Creek	X										X	
Landers Fork												
Lincoln Gulch												
Lincoln Spring Creek	X	X	X	X	X	X			X		X	
Little Fish Creek						X			X		X	
Little Moose Creek												
Little Shanley Creek	X										X	
Liverpool Creek	X	X					X			X	X	
Lodgepole Creek												
McCabe Creek	X	X		X	X	X	X		X	X	X	
McDermott Creek												
McElwain Creek						X			X			
Mike Horse Creek												
Mitchell Creek												
Monture Creek			X	X	X	X	X	X	X	X	X	
Moose Creek	X										X	
Murphys Spring Creek	X	X				X	X		X	X	X	
Murray Creek												
Nevada Creek (lower)	X			X	X	X	X	X	X	X	X	
Nevada Creek (upper)				X							X	
Nevada Spring Creek	X		X	X	X	X	X	X	X	X	X	X
Nora Creek												

Appendix G: Restoration Streams and Table of Activities (cont'd).

Blackfoot River Basin (cont'd)

Stream Name	Fish passage improvement	Prevent Irrigation ditch losses	Spawning habitat protection	Channel restoration	Fish habitat improvement	Riparian vegetation improvement	Improve instream flows	Improve wetlands	Improve range/ riparian habitat	Improve irrigation	Conservation easements or other protection	Remove streamside feedlots
North Fork Blackfoot River	X	X		X	X	X	X		X	X	X	
North Fork Blackfoot River trib at mile 3.1											X	
Park Creek												
Pass Creek												
Paymaster Creek												
Pearson Creek	X	X	X	X	X	X	X	X	X	X	X	X
Poorman Creek	X	X			X	X	X		X	X	X	
Poorman Creek, South Fork				X	X	X						
Rock Creek	X	X		X	X	X	X	X	X	X	X	X
Salmon Creek	X	X	X	X	X	X	X	X	X	X	X	X
Sauerkraut Creek	X	X		X	X	X	X		X	X	X	
Sauerkraut Creek trib at mile 0.9											X	
Sauerkraut Creek trib at mile 1.2											X	
Seven Up Pete Creek												
Shanley Creek	X	X				X	X		X	X	X	
Shave Creek												
Sheep Creek												
Shingle Mill Creek												
Smith Creek												
Snowbank Creek	X	X					X					
Spring Creek (Cottonwood Cr trib. at R.M 43) (sec. 24)	X											
Stonewall Creek		X									X	
Strickland Creek												
Sturgeon Creek												
Sturgeon Creek, spring creek trib												
Sucker Creek												
Tamarack Creek												
Theodore Creek	X											
Union Creek											X	
Wales Creek									X			
Wales Spring Creek												
Ward Creek												
Warm Springs Creek												

Appendix G: Restoration Streams and Table of Activities (cont'd).

Blackfoot River Basin (cont'd)

Stream Name	Fish passage improvement	Prevent Irrigation ditch losses	Spawning habitat protection	Channel restoration	Fish habitat improvement	Riparian vegetation improvement	Improve instream flows	Improve wetlands	Improve range/ riparian habitat	Improve irrigation	Conservation easements or other protection	Remove streamside feedlots
Warren Creek	X		X	X	X	X	X	X	X	X	X	X
Warren Creek (Doney lake trib.)												
Washington Creek												
Washoe Creek											X	
Wasson Creek	X	X	X	X	X	X	X	X	X	X	X	X
West Twin Creek	X											
Willow Creek (above Lincoln, MT)												
Willow Creek (below Lincoln, MT)	X											
Wilson Creek												
Yourname Creek												
Yukon Creek	X											

Clearwater River Basin

Stream Name	Fish passage improvement	Prevent Irrigation ditch losses	Spawning habitat protection	Channel restoration	Fish habitat improvement	Riparian vegetation improvement	Improve instream flows	Improve wetlands	Improve range/ riparian habitat	Improve irrigation	Conservation easements or other protection	Remove streamside feedlots
Auggie Creek	X											
Benedict Creek	X											
Bertha Creek												
Blanchard Creek	X					X			X	X		
Blanchard Creek, North Fork												
Blind Canyon Creek												
Boles Creek												
Buck Creek	X											
Camp Creek												
Clearwater River Section 1	X										X	
Clearwater River Section 2	X					X						
Clearwater River Section 3	X			X		X					X	
Clearwater River Section 4	X											
Clearwater River Section 5	X											
Clearwater River, East Fork	X											
Clearwater River, West Fork			X			X						
Cold Brook Creek												
Colt Creek	X					X						
Deer Creek	X											
Drew Creek												
Fawn Creek												
Findell Creek												

Appendix G: Restoration Streams and Table of Activities (cont'd).

Clearwater River Basin (cont'd)

Stream Name	Fish passage improvement	Prevent Irrigation ditch losses	Spawning habitat protection	Channel restoration	Fish habitat improvement	Riparian vegetation improvement	Improve instream flows	Improve wetlands	Improve range/ riparian habitat	Improve irrigation	Conservation easements or other protection	Remove streamside feedlots
Finley Creek	X											
First Creek												
Grouse Creek	X											
Horn Creek												
Inez Creek												
Lost Horse Creek												
Lost Prairie Creek												
Marshall Creek	X		X									
Morrell Creek	X	X	X			X	X			X		
Mountain Creek		X										
Murphy Creek												
Owl Creek												
Placid Creek												
Placid Creek, North Fork												
Rice Creek		X										
Richmond Creek												
Sawyer Creek												
Second Creek												
Seeley Creek												
Sheep Creek												
Slippery John Creek	X											
Swamp Creek												
Trail Creek	X	X				X	X			X		
Uhler Creek												
Vaughn Creek												

* includes recent land purchases from TNC or transfers of pivate lands (Plum Creek Timber Company) to public ownerhip

Streams approaching final restoration phases

Appendix H: Table of Potential Restoration Projects

Blackfoot River Basin

Stream Name	Road Crossings	Irrigation Impacts	Channel Alterations	Lacks Complexity	Riparian Vegetation	Instream Flow	Road Drainage	Feedlots, Grazing	Recreation Impacts	Whirling Disease	Mining	Residential
Alice Creek				X	X				X			
Anaconda Creek											X	
Arkansas Creek							X	X				
Arrastra Creek	X						X		X	X		
Ashby Creek	X	X	X	X	X	X	X	X				
Ashby Creek, East Fork												
Bartlett Creek					X				X			
Basin Spring Creek												
Bear Creek (Blackfoot trib. at R.M. 12.2)					X		X					
Bear Creek (Blackfoot trib. at R.M. 37.5)	X											
Bear Creek, East Fork (Blackfoot trib. at R.M. 37.5)												
Bear Creek (North Fork drainage)		X				X						
Bear Gulch	X	X	X	X	X	X	X	X				
Beartrap Creek			X	X	X						X	
Beaver Creek		X			X	X		X		X		
Belmont Creek							X			X		
Black Bear Creek	X					X		X				
Blackfoot River (mouth to Clearwater River)			X	X			X	X	X	X		
Blackfoot River (Clearwater River to N.F)			X		X			X	X	X		
Blackfoot River (N.F. to Nevada Creek)						X		X		X		
Blackfoot River (Nevada Cr. to Arrastra Cr.)		X		X	X	X		X		X		
Blackfoot River (Arrastra Cr. to Lincoln, MT)		X	X	X	X	X		X	X	X		X
Blackfoot River (Lincoln, MT to Headwaters)		X	X	X		X			X	X	X	X
Braziel Creek	X		X	X	X	X	X	X				
Buffalo Gulch	X			X	X			X			X	
Burnt Bridge Creek	X	X	X		X	X	X					
California Gulch	X			X	X			X				
Camas Creek			X	X				X				
Chamberlain Creek												
Chamberlain Creek, East Fork												
Chamberlain Creek, West Fork												
Chicken Creek	X		X	X	X			X				

Appendix H: Table of Potential Restoration Projects (cont'd).

Blackfoot River Basin (cont'd)

Stream Name	Road Crossings	Irrigation Impacts	Channel Alterations	Lacks Complexity	Riparian Vegetation	Instream Flow	Road Drainage	Feedlots, Grazing	Recreation Impacts	Whirling Disease	Mining	Residential
Chimney Creek (Douglas Cr tributary)		X	X	X	X			X				
Chimney Creek (Nevada Cr tributary)	X	X	X			X		X				
Clear Creek	X		X		X			X				X
Clear Creek trib to Lincoln Gulch												
Coopers Lake trib												
Copper Creek	X											
Cottonwood Creek (Blackfoot trib. at R.M. 43)	X		X	X	X	X	X	X		X		
Cottonwood Creek, North Fork (Blackfoot trib. at R.M. 43)												
Cottonwood Creek (Nevada Cr tributary)	X	X	X	X	X	X		X				X
Coyote Creek												
Devils Dip	X		X	X	X							
Dick Creek	X	X	X	X	X	X		X				
Douglas Creek		X	X	X	X	X	X	X				
Dry Creek				X	X			X				
Dunham Creek					X	X						
East Twin Creek												
Elk Creek	X	X	X	X	X	X	X	X		X	X	
Enders Spring Creek				X	X							
Finn Creek		X		X		X		X				
Fish Creek		X	X									
Frazier Creek	X	X	X	X	X	X	X	X				
Frazier Creek, North fork		X	X		X	X		X				
Gallagher Creek	X							X				
Game Creek	X							X				
Gleason Creek	X										X	
Gold Creek									X	X		
Gold Creek, West Fork												
Grantier Spring Creek												
Halfway Creek				X	X			X				
Hogum Creek	X				X			X				
Hoyt Creek		X		X	X	X		X		X		
Humbug Creek		X	X		X	X		X			X	
Humbug Creek, South Fork												
Humbug Creek, Middle Fork to South Fork												
Humbug Creek, West Fork to South Fork												

Appendix H: Table of Potential Restoration Projects (cont'd).

Blackfoot River Basin (cont'd)

Stream Name	Road Crossings	Irrigation Impacts	Channel Alterations	Lacks Complexity	Riparian Vegetation	Instream Flow	Road Drainage	Feedlots, Grazing	Recreation Impacts	Whirling Disease	Mining	Residential
Indian Creek				X								
Indian Creek, East fork												
Indian Creek, West Fork												
Indian Meadows Creek	X			X	X							
Jacobsen Spring Creek										X		
Jacobsen Spring Creek, East Fork												
Jefferson Creek	X		X	X		X					X	
Johnson Creek												
Keep Cool Creek	X	X	X		X	X		X			X	
Kleinschmidt Creek								X		X		
Klondike Creek												
Landers Fork			X	X	X	X			X			
Lincoln Gulch	X		X	X	X			X			X	X
Lincoln Spring Creek	X	X	X	X	X			X		X		X
Little Fish Creek	X				X		X	X				
Little Moose Creek												
Little Shanley Creek												
Liverpool Creek												
Lodgepole Creek												
McCabe Creek	X				X							
McDermott Creek												
McElwain Creek	X	X			X	X	X	X				
Mike Horse Creek	X		X	X	X		X				X	
Mitchell Creek	X			X				X				
Monture Creek	X		X	X	X			X	X	X		X
Moose Creek											X	
Murphys Spring Creek												
Murray Creek	X	X		X	X	X	X	X				
Nevada Creek (lower)		X	X	X	X	X		X				
Nevada Creek (upper)	X	X			X		X	X			X	
Nevada Spring Creek										X		
Nora Creek												

Appendix H: Table of Potential Restoration Projects (cont'd).

Blackfoot River Basin (cont'd)

Stream Name	Road Crossings	Irrigation Impacts	Channel Alterations	Lacks Complexity	Riparian Vegetation	Instream Flow	Road Drainage	Feedlots, Grazing	Recreation Impacts	Whirling Disease	Mining	Residential
North Fork Blackfoot River									X	X		
North Fork Blackfoot River trib at mile 3.1								X				
Park Creek	X	X	X		X	X	X					
Pass Creek	X		X	X			X					
Paymaster Creek											X	
Pearson Creek	X				X							
Poorman Creek	X		X	X	X		X	X			X	
Poorman Creek, South Fork												
Rock Creek	X	X		X	X	X		X		X		X
Salmon Creek		X				X						
Sauerkraut Creek			X	X	X		X				X	
Sauerkraut Creek trib at mile 0.9		X	X		X	X		X				
Sauerkraut Creek trib at mile 1.2		X	X		X	X		X				
Seven up Pete Creek	X								X		X	
Shanley Creek		X		X	X	X	X	X		X		
Shave Creek	X										X	
Sheep Creek						X		X				
Shingle Mill Creek		X						X				
Smith Creek					X			X				
Snowbank Creek												
Spring Creek (Cottonwood Cr tributary)		X	X		X	X						
Stonewall Creek	X		X								X	
Strickland Creek				X	X			X				
Sturgeon Creek			X		X	X		X				
Sturgeon Creek, spring creek trib												
Sucker Creek	X	X	X	X	X	X		X				
Tamarack Creek	X	X	X	X	X	X	X					X
Theodore Creek												
Union Creek	X	X		X	X	X		X				
Wales Creek		X	X		X	X		X				
Wales Spring Creek			X		X			X				
Ward Creek	X	X	X	X	X	X		X				
Warm Springs Creek	X	X				X	X					

Appendix H: Table of Potential Restoration Projects (cont'd).

Stream Name	Road Crossings	Irrigation Impacts	Channel Alterations	Lacks Complexity	Riparian Vegetation	Instream Flow	Road Drainage	Feedlots, Grazing	Recreation Impacts	Whirling Disease	Mining	Residential
Blackfoot River Basin (cont'd)												
Warren Creek	X	X	X	X	X	X		X		X		
Warren Creek (Doney Lake trib.)												
Washington Creek	X	X	X	X	X			X			X	
Washoe Creek				X				X				
Wasson Creek					X	X		X				
West Twin Creek												
Willow Creek (above Lincoln)					X			X				
Willow Creek (below Lincoln)	X	X	X	X	X	X	X	X			X	X
Wilson Creek	X	X			X	X	X	X				
Yourname Creek		X	X	X	X	X		X				
Yukon Creek												
Clearwater River Basin												
Auggie Creek	X					X	X					
Benedict Creek	X				X							
Bertha Creek												
Blanchard Creek	X	X	X	X	X	X	X	X				
Blanchard Creek, North Fork												
Blind Canyon Creek							X					
Boles Creek					X		X					
Buck Creek	X				X	X						
Camp Creek	X				X	X	X					
Clearwater River Section 1		X	X		X	X			X			X
Clearwater River Section 2			X		X	X			X			X
Clearwater River Section 3					X	X		X				
Clearwater River Section 4			X			X			X			
Clearwater RiverSection 5	X											
Clearwater River, East Fork							X					
Clearwater River, West Fork					X	X	X					
Cold Brook Creek												
Colt Creek	X		X		X	X	X					
Deer Creek	X				X		X					
Drew Creek	X	X	X	X	X		X					X

Appendix H: Table of Potential Restoration Projects (cont'd).

Clearwater River Basin (cont'd)

Stream Name	Road Crossings	Irrigation Impacts	Channel Alterations	Lacks Complexity	Riparian Vegetation	Instream Flow	Road Drainage	Feedlots, Grazing	Recreation Impacts	Whirling Disease	Mining	Residential
Fawn Creek	X					X	X					
Findell Creek	X				X		X					
Finley Creek	X				X		X					
First Creek	X				X		X					
Grouse Creek	X						X					
Horn Creek						X	X					
Inez Creek	X				X		X					
Lost Horse Creek	X			X	X		X					
Lost Prairie Creek						X	X					
Marshall Creek					X		X					
Morrell Creek		X	X	X	X	X	X		X			X
Mountain Creek	X	X			X	X	X					X
Murphy Creek	X				X		X					
Owl Creek				X	X	X	X		X			
Placid Creek	X			X			X					
Placid Creek, North Fork	X	X					X					
Rice Creek		X					X					
Richmond Creek	X				X		X					
Sawyer Creek	X						X					
Second Creek							X					
Seeley Creek		X					X					X
Sheep Creek	X					X	X					
Slippery John Creek	X					X						
Swamp Creek	X		X	X	X	X	X					X
Trail Creek	X	X	X	X	X		X					X
Uhler Creek	X				X	X	X					
Vaughn Creek							X					

Appendix I : Table of Restoration Streams and Cooperators.

Blackfoot River Basin

Stream Name	State					Federal					Private					
	FWP	MDT	NPCD	DEQ	DNRC	USFWS	BLM	NRCS	BOR	USFS	BC	TU	PL	CF	NFWF	NWE
Alice Creek																
Anaconda Creek																
Arkansas Creek																
Arrastra Creek	X			X			X		X		X	X				X
Ashby Creek	X			X	X	X	X	X			X	X	X	X		X
Ashby Creek, East Fork																
Bartlett Creek																
Basin Spring Creek	X					X						X	X			
Bear Creek (Blackfoot trib. at R.M. 12.2)	X				X	X		X				X	X			X
Bear Creek (Blackfoot trib.at R.M. 37.5)																
Bear Creek, East Fork (Blackfoot trib.at R.M. 37.5)																
Bear Creek (North Fork drainage)																
Beartrap Creek	X			X						X						
Bear Gulch																
Beaver Creek						X						X	X			
Belmont Creek	X						X					X				
Black Bear Creek																
Blackfoot River (mouth to Clearwater River)	X		X				X					X	X	X		
Blackfoot River (Clearwater River to N.F.)	X					X						X	X			
Blackfoot River (N.F. to Nevada Creek)													X			
Blackfoot River (Nevada Cr. to Arrastra Cr.)	X					X						X	X			
Blackfoot River (Arrastra Cr. to Lincoln, MT)																
Blackfoot River (Lincoln, MT to Headwaters)																
Braziel Creek	X			X		X					X	X	X	X		
Buffalo Gulch																
Burnt Bridge Creek																
California Gulch																
Camas Creek					X	X						X				
Chamberlain Creek	X					X	X					X	X		X	
Chamberlain Creek, East Fork	X					X	X									
Chamberlain Creek, West Fork							X									
Chicken Creek																

Appendix I : Table of Restoration Streams and Cooperators.

Blackfoot River Basin (cont'd)

Stream Name	State					Federal					Private					
	FWP	MDT	NPCD	DEQ	DNRC	USFWS	BLM	NRCS	BOR	USFS	BC	TU	PL	CF	NFWF	NWE
Chimney Creek (Douglas Cr tributary)																
Chimney Creek (Nevada Cr tributary)																
Clear Creek																
Clear Creek trib to Lincoln Gulch																
Coopers Lake trib																
Copper Creek										X						
Cottonwood Creek (Blackfoot trib.at R.M. 43)	X			X	X	X	X		X	X	X	X	X	X		X
Cottonwood Creek, North Fork (Blackfoot trib.at R.M. 43)																
Cottonwood Creek (Nevada Cr tributary)				X		X										
Coyote Creek																
Devils Dip	X					X						X				
Dick Creek	X					X		X		X	X	X	X	X	X	
Douglas Creek	X		X			X		X	X		X	X	X			
Dry Creek	X					X		X				X		X		
Dunham Creek	X					X				X	X	X	X	X		X
East Twin Creek	X	X														
Elk Creek	X					X	X		X		X	X	X		X	X
Enders Spring Creek	X					X						X		X		
Finn Creek																
Fish Creek						X		X			X	X	X	X		
Frazier Creek	X					X						X	X	X		
Frazier Creek, North Fork																
Gallagher Creek																
Game Creek																
Gleason Creek																
Gold Creek	X				X	X	X					X	X			X
Gold Creek, West Fork																
Grantier Spring Creek	X					X						X	X			
Halfway Creek																
Hogum Creek																
Hoyt Creek	X			X		X		X	X		X	X	X	X		
Humbug Creek																
Humbug Creek, South Fork																
Humbug Creek, Middle Fork to South Fork																
Humbug Creek, West Fork to South Fork																

Appendix I : Table of Restoration Streams and Cooperators.

Blackfoot River Basin (cont'd)

Stream Name	State					Federal					Private					
	FWP	MDT	NPCD	DEQ	DNRC	USFWS	BLM	NRCS	BOR	USFS	BC	TU	PL	CF	NFWF	NWE
Indian Creek																
Indian Creek, East Fork																
Indian Creek, West Fork																
Indian Meadows Creek																
Jacobsen Spring Creek	X					X		X	X		X	X	X	X		
Jacobsen Spring Creek, East Fork	X					X						X				
Jefferson Creek																
Johnson Creek	X					X				X		X	X			X
Keep Cool Creek	X		X			X		X	X		X	X	X	X		
Kleinschmidt Creek	X	X	X	X		X		X			X	X	X	X	X	X
Klondike Creek										X		X				
Landers Fork																
Lincoln Gulch																
Lincoln Spring Creek	X					X		X	X			X	X	X		
Little Fish Creek																
Little Moose Creek																
Little Shanley Creek										X						
Liverpool Creek	X				X	X										
Lodgepole Creek																
McCabe Creek	X					X		X		X	X	X	X			X
McDermott Creek																
McElwain Creek	X					X						X	X			
Mike Horse Creek																
Mitchell Creek																
Monture Creek	X					X		X		X		X	X		X	
Moose Creek										X		X		X		
Murphys Spring Creek	X				X	X		X			X	X	X			
Murray Creek																
Nevada Creek (lower)	X		X		X	X		X				X	X	X		
Nevada Creek (upper)	X		X			X		X			X		X		X	
Nevada Spring Creek	X		X			X		X	X		X	X	X	X		X
Nora Creek																

Appendix I : Table of Restoration Streams and Cooperators.

Blackfoot River Basin (cont'd)

Stream Name	State					Federal					Private					
	FWP	MDT	NPCD	DEQ	DNRC	USFWS	BLM	NRCS	BOR	USFS	BC	TU	PL	CF	NFWF	NWE
North Fork Blackfoot River	X				X	X					X	X	X	X	X	X
North Fork Blackfoot River trib at mile 3.1																
Park Creek	X				X											
Pass Creek																
Paymaster Creek																
Pearson Creek	X		X			X	X	X			X	X	X			X
Poorman Creek	X	X		X	X	X		X		X	X	X	X	X		X
Poorman Creek, South Fork	X									X		X				
Rock Creek	X	X	X	X	X	X		X			X	X	X	X	X	X
Salmon Creek	X					X		X				X	X	X	X	
Sauerkraut Creek	X					X		X		X		X	X	X	X	
Sauerkraut Creek trib at mile 0.9																
Sauerkraut Creek trib at mile 1.2																
Seven Up Pete Creek																
Shanley Creek	X					X		X		X		X	X			
Shave Creek																
Sheep Creek																
Shingle Mill Creek																
Smith Creek																
Snowbank Creek	X									X		X		X		
Spring Creek (Cottonwood Cr tributary)(sec. 24)										X						
Stonewall Creek	X					X				X		X	X			
Strickland Creek																
Sturgeon Creek																
Sturgeon Creek, spring creek trib																
Sucker Creek	X									X		X				
Tamarack Creek																
Theodore Creek	X									X		X				
Union Creek																
Wales Creek	X										X		X			X
Wales Spring Creek																
Ward Creek			X	X		X					X	X	X	X		
Warm Springs Creek																

Appendix I : Table of Restoration Streams and Cooperators.

Blackfoot River Basin (cont'd)

Stream Name	State					Federal					Private					
	FWP	MDT	NPCD	DEQ	DNRC	USFWS	BLM	NRCS	BOR	USFS	BC	TU	PL	CF	NFWF	NWE
Warren Creek	X		X	X	X	X		X	X		X	X	X	X		
Warren Creek (Doney lake trib.)																
Washington Creek																
Washoe Creek																
Wasson Creek	X		X	X		X		X	X		X	X	X			X
West Twin Creek	X	X														
Willow Creek (above Lincoln, MT)																
Willow Creek (below Lincoln, MT)	X					X						X	X			
Wilson Creek																
Yourname Creek																
Yukon Creek																
Clearwater River Basin																
Auggie Creek																
Benedict Creek										X						
Bertha Creek																
Blanchard Creek	X				X	X	X					X	X			
Blanchard Creek, North Fork																
Blind Canyon Creek																
Boles Creek																
Buck Creek										X						
Camp Creek																
Clearwater River Section 1	X				X											
Clearwater River Section 2					X											
Clearwater River Section 3	X												X			
Clearwater River Section 4	X									X						
Clearwater River Section 5	X									X						
Clearwater River, East Fork										X						
Clearwater River, West Fork	X									X						
Cold Brook Creek																
Colt Creek										X						
Deer Creek	X															
Drew Creek										X						
Fawn Creek																
Findell Creek																

Appendix I : Table of Restoration Streams and Cooperators.

Clearwater River Basin (cont'd)

Stream Name	State					Federal					Private					
	FWP	MDT	NPCD	DEQ	DNRC	USFWS	BLM	NRCS	BOR	USFS	BC	TU	PL	CF	NFWF	NWE
Finley Creek										X						
First Creek																
Grouse Creek										X						
Horn Creek																
Inez Creek																
Lost Horse Creek																
Lost Prairie Creek																
Marshall Creek	X															
Morrell Creek	X					X				X		X	X			X
Mountain Creek																
Murphy Creek																
Owl Creek																
Placid Creek																
Placid Creek, North Fork																
Rice Creek																
Richmond Creek																
Sawyer Creek																
Second Creek																
Seeley Creek																
Sheep Creek																
Slippery John Creek										X						
Swamp Creek																
Trail Creek	X					X				X		X	X			
Uhler Creek																
Vaughn Creek																

FWP - Montana Fish, Wildlife and Parks
MDT - Montana Department of Transportation
NPCD - North Powell Conservation District
DEQ - Dept.of Environment Quality
DNRC - Dept. of Natural Resources and Conservation
USFWS - U.S.Fish and Wildlife Service
BLM - Bureau of Land Management
USFS - U.S. Forest Service
BC - Blackfoot Challenge
TU - Trout Unlimited
PL - Private Landowners
CF - Chutney Foundation
NFWF - National Fish and Wildlife Foundation
NWE - Northwestern Energy
PCT - Plum Creek Timber Company