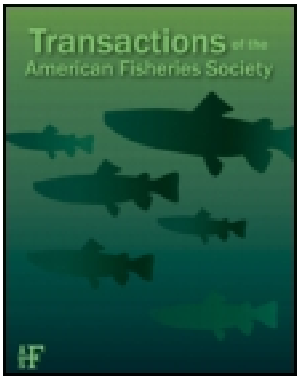

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ARTICLE

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

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Abstract

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

Native salmonids were once abundant and widespread across the western United States, but as natural landscapes were modified many populations declined dramatically to imperiled status (Nehlsen et al. 1991; Behnke 1992; Thurow et al. 1997). Declines are largely associated with mining activities, timber extraction, stream channelization, irrigation practices, dams, riparian grazing, overfishing, and the influence of introduced exotic species (e.g., Meehan 1991; Behnke 1992; Thurow et al. 1997). These anthropogenic activities often destroy and degrade aquatic habitats (Meehan 1991; Waters 1995), disrupt fish migrations (Rieman and McIntyre 1993; Thurow et al. 1997), and can alter environments in favor of exotic organisms (Bartholomew and Wilson 2002; Shepard 2004). As a result, many public and private organizations have developed strategies to improve recovery, management, and protection of native salmonids (e.g., Williams et al. 1997; MBTRT 2000; USFWS 2010).

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

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

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

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

STUDY AREA

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

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

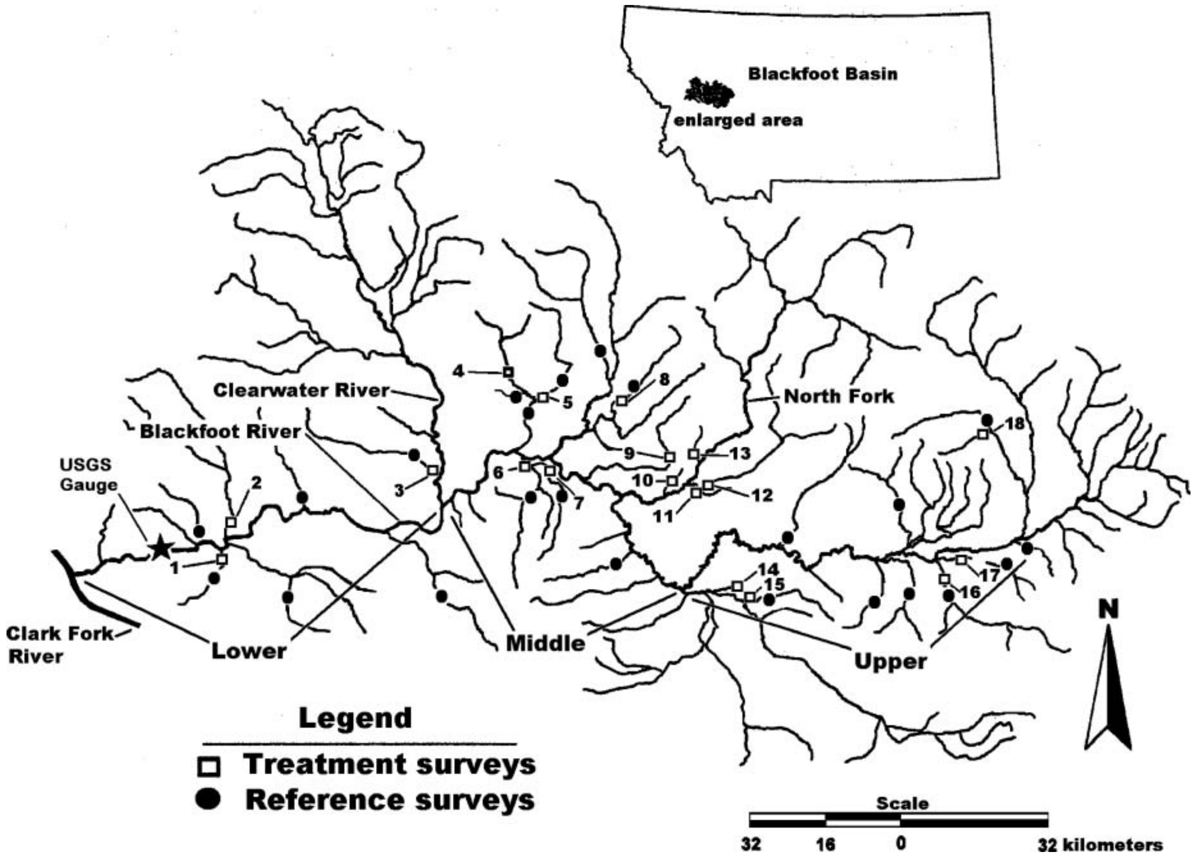

FIGURE 1. The Blackfoot River basin in western Montana showing treatment and reference sites and demarcations of the lower, middle, and upper Blackfoot River basin. Treatment sites (open squares with site numbers from 1 to 18) relate to restoration sites and techniques in Table 1 and fisheries response on Figure 2.

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

Natural channel restoration techniques were employed in the most degraded streams to return them to geomorphically stable and natural states that are capable of maintaining habitat-forming processes. These methods incorporated bankfull theory (Dunne and Leopold 1978) and the core principles of natural channel design (Rosgen 1994, 1996, 2007), and relied on geomorphic indicators of the bankfull channel, measured reference reaches, and design validation using empirically derived regional curves of channel geometry for western Montana streams (Lawlor 2002). In the Blackfoot River basin, these methods further incorporated the placement of instream habitat features suited to the geomorphic potential (Rosgen 1996; Schmetterling and Pierce 1999), vegetative setting (Manning et al. 1989; Hansen et al. 1995), and local fisheries resource of the site (MBTRT 2000; Brown et al. 2001; Pierce et al. 2005).

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

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

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

METHODS

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

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

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

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

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

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

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

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

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

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

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

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

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

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

RESULTS

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

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

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

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

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

Spring, and Poorman creeks) displayed significant increases in native trout abundance. Statistically significant declines in both native and nonnative trout groups were observed at only two sites. Native trout declined significantly in Gold Creek concurrent with an increasing trend in nonnative trout. Conversely, nonnative trout decreased in Grantier Spring Creek concurrent with a significant increase in native trout. While increases in native trout were not significant in Cottonwood or Chamberlain creeks, data from both streams show native trout abundance increased quickly and remained elevated for several years following treatment (Figure 2). As shown in these examples, linear regressions can mask short-term nonlinear responses and reduce statistical rigor compared with best-fit regression models (e.g., Akaike information criterion models, Akaike 1974; Burnham and Anderson 2002).

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

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

	Total trout			Native trout			Nonnative trout		
Stream name	Slope	*P*-value	r^2	Slope	*P*-value	r^2	Slope	*P*-value	r^2
Bear Creek	0.024	0.026	0.40	–0.003	0.249	0.13	0.025	0.023	0.42
Gold Creek	0.020	0.138	0.29	–0.008	0.004	0.73	0.026	0.060	0.42
Blanchard Creek	–0.016	0.293	0.14	0.001	0.180	0.21	–0.017	0.273	0.15
Cottonwood Creek	0.010	0.353	0.07	0.010	0.322	0.08	0.0005	0.787	<0.01
Shanley Creek	0.005	0.668	0.07	0.001	0.604	0.10	0.004	0.713	0.05
Chamberlain Creek	0.005	0.535	0.07	0.007	0.393	0.06	–0.004	0.043	0.32
Pearson Creek	–0.011	0.361	0.08	–0.010	0.394	0.07	–0.002	0.185	0.17
McCabe Creek	0.023	0.224	0.34	0.024	0.135	0.47	NA	NA	NA
Warren Creek	0.020	0.242	0.32	0.0004	0.944	<0.01	0.019	0.259	0.3
Jacobsen Spring Creek	0.016	0.043	0.68	–0.001	0.158	0.43	0.017	0.038	0.70
Kleinschmidt Creek	0.033	0.006	0.63	0.001	0.213	0.19	0.033	0.007	0.62
Rock Creek	0.015	0.008	0.60	0.001	0.551	0.05	0.015	0.011	0.57
Murphy Spring Creek	0.026	<0.001	0.99	0.021	<0.001	0.95	0.003	0.319	0.46
Nevada Spring Creek	0.057	0.002	0.78	0.060	0.002	0.73	–0.003	0.864	0.00
Wasson Creek	0.023	0.350	0.15	0.025	0.298	0.18	–0.002	0.858	0.01
Poorman Creek	0.029	0.030	0.64	0.011	0.045	0.59	0.020	0.064	0.53
Grantier Spring Creek	–0.016	0.215	0.62	0.013	0.020	0.96	–0.028	0.071	0.86
Snowbank Creek	0.076	0.135	0.47	0.076	0.135	0.47	NA	NA	NA

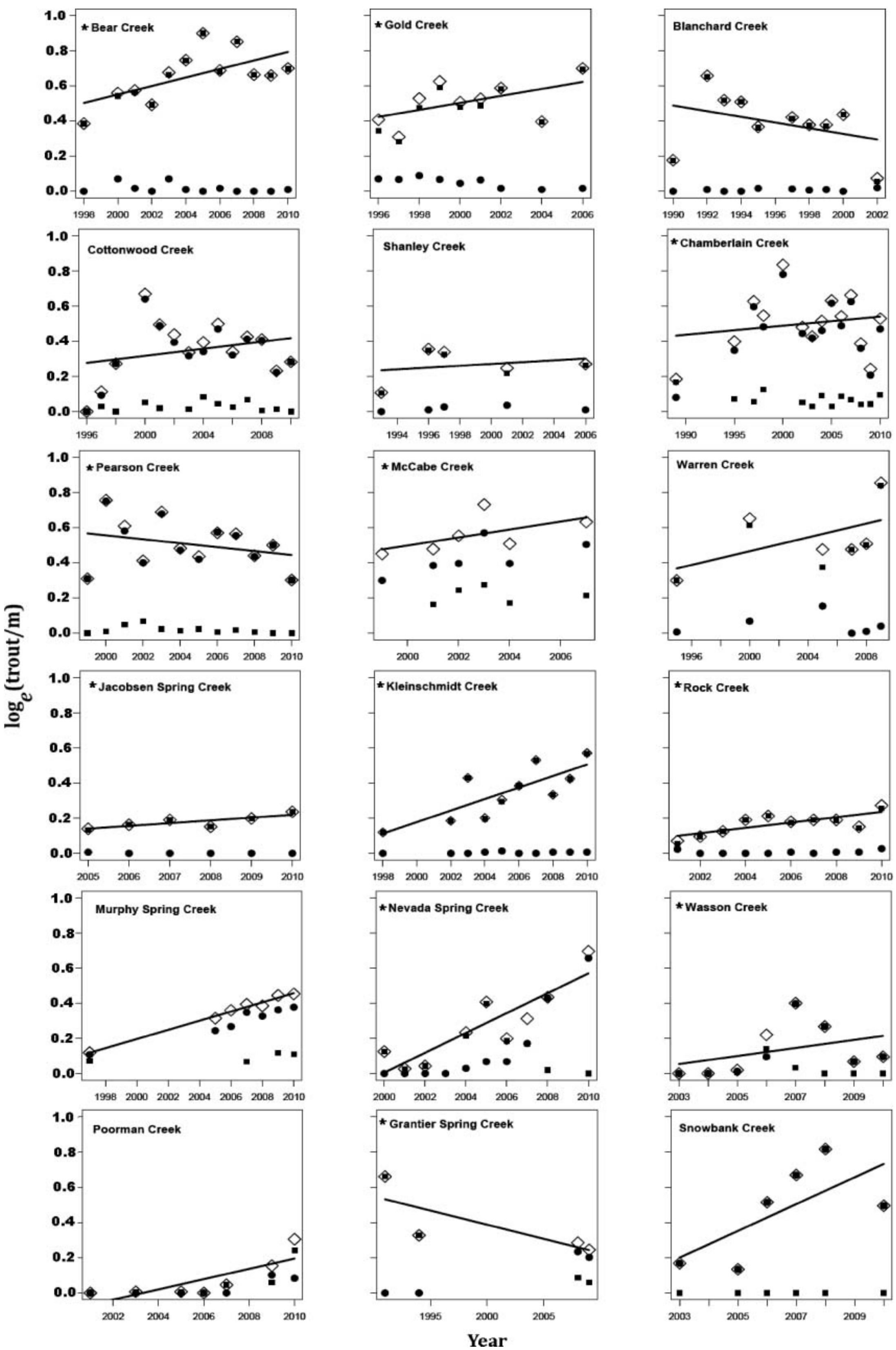

FIGURE 2. Wild trout response for 18 individual treatment streams, 1989–2010. Diamonds represent estimates of total trout abundance, circles represent estimates of native trout abundance, and squares represent estimates of nonnative trout abundance. Black line represents the linear trend line for total trout abundance. The first year on the *x*-axis denotes the pretreatment year. An asterisk (*) denotes a stream with active instream habitat treatments.

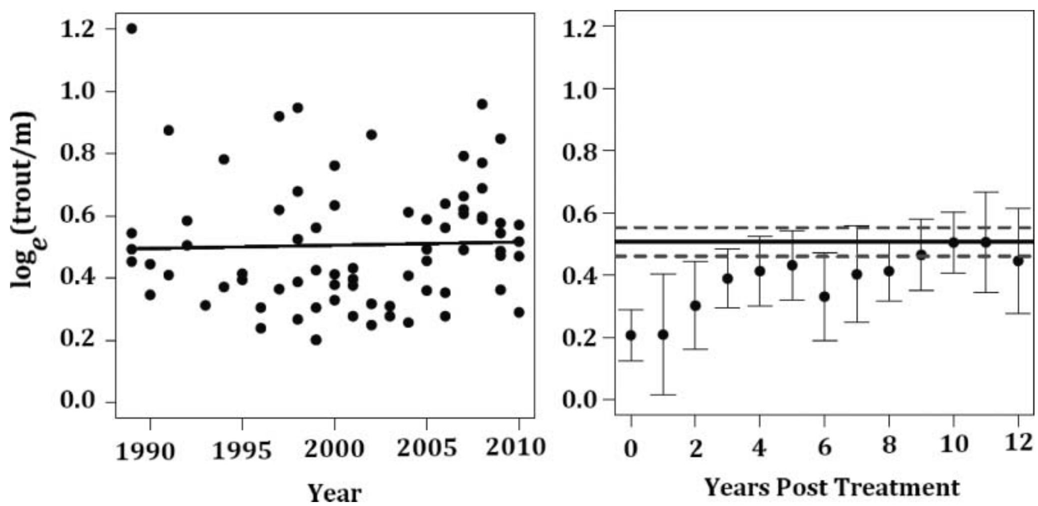

FIGURE 3. (**a**) Estimates of total trout abundance at reference sites by calendar year. Linear regression analysis indicates a long-term stable trend with a slope not significantly different from zero during the study period (slope = 0.001, $P = 0.78$). (**b**) Average total trout abundance by years posttreatment. The solid black line represents the grand average of total trout abundance for all monitoring observations in reference sites (0.65 trout/m). Gray dashed lines represent the 95% confidence interval around the reference average (0.61–0.69 trout/m). Note: this grand average incorporates both year-to-year and stream-to-stream variation in the reference data set. Grantier Spring Creek is not included in these data.

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

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

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

Subbasin Scale Trout Response

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

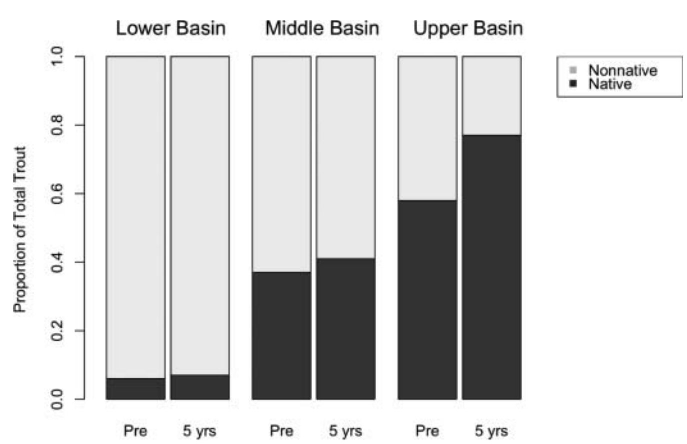

FIGURE 4. Average proportion of native and nonnative trout in restored sites at pretreatment (pre) and 5 years posttreatment across the three subbasins of the Blackfoot River watershed.

DISCUSSION

Blackfoot River Restoration: a Riverscape Conservation Strategy

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

Restoration Techniques and Reach-Scale Response

In our study, average total trout abundance for 17 sites increased rapidly, approached reference conditions about 3 years posttreatment, and remained elevated near reference conditions (Figure 3b). The initial rapid increase in total trout abundance can be attributed to several projects involving instream flow enhancement (e.g., Blanchard, Cottonwood, and Snowbank creeks) or enhanced fish passage, entrainment reduction, or both (Figure 2; Table 2). As intended, these projects encouraged short-term redistribution of fish older than age 1 into treatment reaches as described elsewhere (Gowan and Fausch 1996; Roni and Quinn 2001). Yet, examination of all survey data from treatments sites also revealed increased production of age-0 trout as well as diverse community-level responses in the posttreatment environments, depending on the treatment and its location within the basin. As one example, increased downstream recruitment of juvenile Rainbow Trout from an upstream population was the target of the treatment in Blanchard Creek, the lowermost instream flow project in this study (Pierce et al. 1997). Here, Rainbow Trout of age >1 showed a rapid and sustained increase. Likewise, estimates of age-0 Rainbow Trout abundance also increased from an average of 0.17 trout/m (range, 0.03–0.38) during the first 3 years of monitoring (1989–1992) to a 7-year average of 0.94 trout/m (range, 0.46–1.57) between 1992 and 2002 (R. Pierce, unpublished data). In addition, five native fishes (Westslope Cutthroat Trout, Mountain Whitefish *Prosopium williamsoni*, Northern Pikeminnow *Ptychocheilus oregonensis*, Longnose Dace *Rhinichthys cataractae*, Largescale Sucker *Catostomus macrocheilus*, and sculpin *Cottus* sp.) were present in posttreatment surveys but were not detected in pretreatment surveys (Pierce et al. 1997). As a second example, the Snowbank Creek instream flow project (i.e., the uppermost treatment) was intended to foster a community response by restoring flows and habitat connectivity with a downstream tributary. Here, monitoring showed a sharp initial increase in Westslope Cutthroat Trout of age >1 along with the upstream expansion of Bull Trout into the project area, which included successful spawning (i.e., redds and age-0 fish present) within 3 years of treatment (Pierce et al. 2011; U.S. Forest Service, unpublished data;).

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

In addition to irrigation-related treatments, we implemented natural channel design techniques along with riparian grazing changes on the most treatment sites (Table 1). Compared with habitat enhancement techniques that rely heavily on structures (Roni 2005; Roni et al. 2008; Stewart et al. 2009), natural channel design integrates the geomorphic, hydrologic, and vegetative setting of the site and its valley in a manner that emulates natural (e.g., reference) channel conditions (Rosgen 1996; Baldigo et al. 2008; this study). Natural channel design methods are more natural and resilient than traditional methods (Schmetterling and Pierce 1999; Baldigo et al. 2008, 2010; Whiteway et al. 2010), yet few fisheries studies have documented the efficacy of this approach. However, one study (Baldigo et al. 2008) demonstrated that both community biomass and species richness increased following natural channel design treatments over short-term (i.e., <5 years) monitoring periods. Consistent with those findings, 9 of 11 active treatments in our study showed positive trends in total trout abundance over a 6–21-year monitoring period (Figure 2). For certain sites requiring full reconstruction (e.g., Bear, Kleinschmidt, and Nevada Spring creeks), estimates of total trout abundance showed continuous linear increases 10–12 years posttreatment (Figure 2). With the exception of Gold Creek, most (8 of 9) active treatments with positive trends also required multiple techniques (Table 1). These incremental long-term increases contrast with the rapid increases observed in instream flow projects (this study), as well as with other studies that suggest about 5 years is required for the full effects of habitat manipulation alone to be realized (Hunt 1976; Whiteway et al. 2010).

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

Subbasin Trends and the Response of Native Trout

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

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

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

Because regulations governing the harvest of trout have remained consistent for native and nonnative trout in small streams trout since 1990, it appears unlikely direct angling pressure strongly influenced reach-scale trends in this study. This appears evident given a common pattern in which trout abundance increases soon after habitat treatments (Figure 3b), which developed 8 years after angling regulation changes were enacted. Most treatment (and reference) reaches in this study are, in fact, located on small, brushy streams that provide limited access and support very little angling pressure (MFWP 2011).

Monitoring and Adaptive Management

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

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

Conclusions

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

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

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