Coeur d'Alene Tribe
Fisheries Program

Implementation of Fisheries Enhancement Opportunities on the Coeur d’Alene Reservation

2010 ANNUAL REPORT

Coeur d'Alene Tribe Department of Natural Resources
Fisheries Program
401 Annie Antelope Road
Plummer, ID 83851-0408

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EXECUTIVE SUMMARY

The BPA project entitled “Implementation of Fisheries Enhancement Opportunities on the Coeur d’Alene Reservation” mitigates for lost fishery resources that are of cultural significance to the Coeur d’Alene Tribe. This project funds management actions, and research, monitoring, and evaluation (RME) activities associated with these actions, which are carried out by the Coeur d’Alene Tribe’s Fisheries Program to recover depressed populations of westslope cutthroat trout in the Coeur d’Alene basin. This report summarizes RME data collected during 2010 that describe the status and trends of cutthroat trout in target watersheds and the response of stream habitats and trout populations to implemented habitat restoration and non-native fish extraction measures. The report also describes the in-stream and riparian restoration actions that were implemented in 2010.

Research, monitoring, and evaluation summary

Abundance estimates of 113 and 72 adfluvial adult cutthroat trout were generated in 2010 for the Lake and Benewah creek watersheds, respectively. The Lake Creek abundance estimate, however, did not account for those fish that approached the trap but, because of trap avoidance behavior, were neither captured nor ascended upriver of the trap. When including these fish, an abundance estimate of 162 ascending adults was generated for Lake Creek. We intend to modify the trap used to capture upriver migrating spawning adults to minimize the apparent trap avoidance behavior.

Abundance estimates of 3858 and 394 adfluvial, outmigrating juvenile cutthroat trout were generated in 2010 for Lake and Benewah creeks, respectively. The lower estimate in upper Benewah Creek likely reflects the lower number of spawning adfluvial adults in Benewah than in Lake Creek in addition to the later trap deployment in Benewah Creek than in Lake Creek. Moreover, estimates in both systems were likely biased low in 2010 given that there was evidence that juveniles were moving downriver before traps could be deployed. Because accurate juvenile outmigrant abundance estimates are required to reliably track watershed-scale changes in in-stream productivity of our adfluvial cutthroat trout populations, modifications to sampling protocol and trapping techniques are being considered that will address this concern. Of the juvenile cutthroat trout that were captured in Lake and Benewah watersheds in 2010, 968 (28%) and 186 (63%) were respectively PIT-tagged to monitor their in-lake survival rates.

Detection data from juveniles that had been PIT-tagged in Lake Creek from 2005 to 2007 indicate that only 1.7% have returned to spawn as adults. This return rate is ten times lower than those that have been derived for adfluvial cutthroat trout in other systems. In addition, fish that have returned to spawn as adults in Lake Creek have generally been larger and outmigrated earlier when tagged as juveniles than those that have not been found to return. These results indicate that processes operating in Lake Coeur d’Alene are unduly impacting survival of juvenile cutthroat trout and that the strength of these processes may be dependent on attributes of juveniles at the time of outmigration. These findings also lend support to the need to further investigate whether predation is a predominant mechanism regulating survival rates. As such, we intend to initiate a two-year intensive study in 2011 to evaluate the impact of two non-native piscivores, northern pike and smallmouth bass, on cutthroat trout survival in Lake Coeur d’Alene.

We documented population level impacts from non-native rainbow trout in Benewah Creek in 2010 as 10% of the ascending adfluvial adults captured in the migrant trap were classified as potential hybrids based on external characteristics. Recent genetic analyses have confirmed the incidence of hybridization in adfluvial watersheds of the Coeur d'Alene Basin, and though levels of genetic introgression were generally low (i.e., less than 3%), there was evidence of some relatively recent hybridization with rainbow trout in the Benewah watershed. Much of this recent hybridization could be attributed to escapees from rainbow trout ponds that are located on private land in close proximity to stream reaches in the upper Benewah watershed. To minimize the impacts from pond escapees, we are proposing contacting landowners and offering the opportunity for sterile, triploid rainbow trout to be stocked in their ponds.

Results from electrofishing surveys conducted at index sites in 2010 across target watersheds revealed patterns of cutthroat trout abundance and distribution that were consistent with surveys conducted in previous years. Cutthroat trout of ages one and older were widespread and reported at moderate to high densities (mean of 29.4 fish/100 m) across mainstem reaches in Evans Creek. These results generally reflect the overall suitability of rearing habitat for cutthroat trout in Evans Creek. In contrast, similar aged cutthroat trout in Alder Creek were generally found at densities less than 10 fish/100 m, and were constrained to the lowermost reaches of the watershed. The observed spatial pattern of cutthroat distribution in Alder Creek is likely explained by their displacement by non-native brook trout, which can typically be found at high densities (e.g., 63.0 fish/100 m in 2010) in the upper watershed.

In adfluvial watersheds, electrofishing surveys indicated that densities of age one and older cutthroat trout were substantially greater in upper reaches of tributaries than in lower reaches of tributaries and in main-stem reaches. In Benewah Creek, mean densities of 30.5 fish/100 m were found across sites in upper reaches of four of the five primary rearing tributaries in the upper watershed, whereas densities in lower reaches of these tributaries were two to four times lower than those respectively recorded in upper reaches. Density indices were relatively moderate in lower main-stem reaches, averaging 17.1 fish/100 m, and generally low in upper main-stem reaches, averaging only 3.6 fish/100 m. In Lake Creek, a mean density index of 33.5 fish/100 m was calculated for age 1+ fish across sites in upper reaches of Bozard Creek and the West Fork tributary; density indices were less than 7.0 fish/100 m at sites in lower reaches of both tributaries. Density indices across lower main-stem sites were moderate, averaging 17.6 fish/100 m, and greater than those (mean, 4.4 fish/100 m) in upper main-stem reaches. The documented disparity in abundance among upper and lower reaches within both tributary and main-stem habitats is likely attributed to differences in the suitability of habitat among reaches, and is intended to be addressed by prospective habitat restoration measures.

Habitat surveys conducted in 2010 indicated that restored main-stem reaches in the upper Benewah watershed (i.e., those addressed during Phase I restoration) were approaching or maintaining performance benchmarks for those physical attributes that have been linked to the quality of salmonid rearing habitat. Notably, mean residual pool depths exceeded 1.0 m and mean percent fines within the active channel were less than 16% in these restored reaches. Large woody debris loadings were variable across sites (3.5 – 17.2 m^3/100 m), and likely reflect the dynamic nature of wood aggregation processes as large pieces, introduced by our restoration actions, become re-distributed during high flow events. Thermal refugia were also documented in deep pools along restored main-stem reaches as 25% of the monitored pools maintained pool bottom temperatures that were at least 3ºC lower than those measured in tail-outs.

We documented an overall lack of stability in beaver dam complexes in 2010 in that reach of the upper Benewah main-stem that is currently receiving treatment as part of Phase II restoration. Many of the measured natural dams (> 75%) were either not built with or not built upon stable materials (e.g., large woody debris) but instead were composed of small alder and grasses. Moreover, mean dam height decreased from the fall 2009 to the summer 2010 survey for those dams found in reaches that were laterally bounded by open meadows. Evidently, many of the dams were either blown out or lost structural materials over the winter and spring. Lack of stability precludes the ability for dam complexes to impound water during high discharge events and induce extended periods of overbank flooding to enhance floodplain connectivity and water retention. Furthermore, lack of stable impoundments limits the amount of seasonally-persistent deep, pool habitats that have been reported to be preferred by cutthroat trout. In comparison, dam height did not significantly change in a reach where a relatively intact riparian forest still exists and where relict large wood is found within the channel. In addition, large dams (e.g., > 2 ft), indicative of their relative persistence, were most often found in this reach. The apparent dam stability that was documented in this reach is intended to be emulated by our restoration approach for this portion of the upper Benewah main-stem. The engineered wood structures that are being introduced into the stream should serve to simulate the flow obstruction effects of natural wood jams and beaver dams, and confer a degree of stability to natural dams that will allow for more frequent and extensive floodplain connection and improve the trajectory for natural process recovery.

Overall, more than 8000 brook trout have been removed from the upper Benewah watershed since the inception of the suppression program in 2004, with 627 fish removed in 2010. Generally, the program has been effective at regulating numbers of brook trout at a manageable level. Though densities of age one and older brook trout in lower reaches of tributaries in the upper Benewah watershed averaged 22.6 fish/ 100m in 2010, upper reaches of these tributaries and upper main-stem reaches supported much lower mean densities of 3.0 fish/ 100m and 12.2 fish/ 100m, respectively. Furthermore, densities of brook trout in the upper Benewah watershed were overall three to five times lower than those calculated in upper portions of the Alder Creek watershed. Less brook trout were removed in 2010 than during the earlier years of the suppression program given that our current tactical approach targets a smaller contiguous segment of the upper Benewah main-stem upstream of the 12-mile bridge that has been considered to provide the most suitable spawning habitat for brook trout. Our tactics have also included the deployment of a temporary barrier upstream of 12-mile bridge to prevent brook trout that are residing in downstream main-stem reaches from ascending and accessing the seemingly more suitable spawning habitats upstream. Thus, our current approach aims to curb reproductive success rather than attempting to remove as many fish as possible. A comparison of the length distribution of brook trout removed from the reach upstream of 12-mile bridge in 2010 indicates that the percentage of age-0 fish (25%) was approximately half of that observed in derived length distributions from earlier years (e.g., 45 – 55%). Moreover, age-0 density indices in tributaries and in main-stem reaches in 2010 averaged only 2.5 and 5.6 fish/100 m, respectively, and were markedly lower than those calculated across sites in the upper Alder watershed (23.3 fish/100 m). Seemingly, our curbed removal efforts and re-focused tactics, which began in 2009, did not lead to substantial reproductive output, and in fact may have contributed to the lack of age-0 brook trout captured in 2010.

Restoration and enhancement activities

The second year of restoration actions were implemented in the upper mainstem of Benewah Creek, designated as Reach D2, to facilitate greater frequency of floodplain/stream interaction and to increase the diversity of both aquatic and terrestrial habitats. Actions included re-grading and partial excavation for 457 m of an existing relict channel to reconnect at its upstream junction with the main channel, just upstream of the confluence with Windfall Creek. Presumably, the connected side channel will provide improved opportunities for salmonid rearing particularly during the winter. A total of seven in-channel wood structures were constructed, which emulate flow obstruction effects of natural wood jams and beaver dams. This approach was based on observations that the most persistent, natural dams throughout the Benewah Creek stream corridor are built with mountain alder integrated with remnant in-channel large wood. The structures effectively create increased backwater effects during floods such that the valley floor becomes connected annually. Additionally, approximately 24 cubic meters of wood was added to the stream channel and near bank region within a 200 meter reach to aid beavers in dam construction and to increase wood loading to approximate a target volume of 6 m^3/100 m for mainstem and tributary habitats in the watershed. Furthermore, two natural beaver dams were reinforced with vertical uprights that were installed through the face of the dam. The approaches to channel wood additions and beaver dam augmentation were implemented as a natural analog alternative to large scale riffle construction that had been implemented previously in the upper mainstem of Benewah Creek.

A primary strategy being utilized for restoration of the valley bottom in Benewah Creek is the utilization of black cottonwood's unique life history characteristics to rapidly "flip" or change the current degraded riparian ecosystem into a diverse, self-sustaining riparian forest. A total of 27,957 herbaceous plugs and 6,494 woody trees and shrubs were planted in fall 2010 and spring 2011, treating 2.56 hectares of floodplain and off-channel wetlands and 900 m of streambank associated with side-channel habitat. Several existing wetland swales and groundwater fed wetlands were also planted to establish nursery areas for propagation of black cottonwood and willows and to provide forage and dam building materials for beaver. These wetlands have favorable hydrologic conditions for growing and propagation of black cottonwood and willows and these conditions have been further enhanced by more frequent overbank flows attributed to in-channel structures and obstructions that have been installed recently.

Channel and riparian enhancement measures to address severe channel incision and bank erosion were implemented in the West Fork of Lake Creek in 2010 as part of a strategy to create a new channel segment that is hydraulically connected with the adjacent floodplain. Channel construction involved the creation of 518 m of new channel length. Imported gravels and logs were used to create streambed and streambanks in the newly constructed channel. Logs were also placed on the new floodplain to provide roughness to prevent erosion. Riparian enhancement in 2010 followed the vegetation plan that was developed for the site and involved the planting of 14,663 herbaceous plugs along 1220 m of newly built streambank, and the planting of 3,670 deciduous trees along 0.4 ha of adjacent floodplain habitat to re-establish native vegetation.

1.0 PROJECT BACKGROUND

Historically, the Coeur d'Alene Indian Tribe depended on runs of anadromous salmon and steelhead along the Spokane River and Hangman Creek as well as resident and adfluvial forms of trout and char in Coeur d'Alene Lake for subsistence. Dams constructed in the early 1900s on the Spokane River in the City of Spokane and at Little Falls (further downstream) were the first dams that initially cut-off the anadromous fish runs from the Coeur d'Alene Tribe. These fisheries were further removed following the construction of Chief Joseph and Grand Coulee Dams on the Columbia River. Together, these actions forced the Tribe to rely solely on the resident fish resources of Coeur d'Alene Lake for their subsistence needs.

The Coeur d'Alene Tribe is estimated to have historically harvested around 42,000 westslope cutthroat trout (*Oncorhynchus clarki lewisi*) per year (Scholz et al. 1985). In 1967, Mallet (1969) reported that 3,329 cutthroat trout were harvested from the St. Joe River, and a catch of 887 was reported from Coeur d'Alene Lake. This catch is far less than the 42,000 fish per year the tribe harvested historically. Today, only limited opportunities exist to harvest cutthroat trout in the Coeur d'Alene Basin. It appears that a suite of factors have contributed to the decline of cutthroat trout stocks within Coeur d'Alene Lake and its tributaries (Mallet 1969; Scholz et al. 1985; Lillengreen et al. 1993). These factors included the construction of Post Falls Dam in 1906, major changes in land cover types, impacts from agricultural activities, and introduction of exotic fish species.

The decline in native cutthroat trout populations in the Coeur d'Alene basin has been a primary focus of study by the Coeur d'Alene Tribe's Fisheries and Water Resources programs since 1990. The overarching goals for recovery have been to restore the cutthroat trout populations to levels that allow for subsistence harvest, maintain genetic diversity, and increase the probability of persistence in the face of anthropogenic influences and prospective climate change. This included recovering the lacustrine-adfluvial life history form that was historically prevalent and had served to provide resiliency to the structure of cutthroat trout populations in the Coeur d'Alene basin. To this end, the Coeur d'Alene Tribe closed Lake Creek and Benewah Creek to fishing in 1993 to initiate recovery of westslope cutthroat trout to historical levels.

However, achieving sustainable cutthroat trout populations also required addressing biotic factors and habitat features in the basin that were limiting recovery. Early in the 1990s, BPA-funded surveys and inventories identified limiting factors in Tribal watersheds that would need to be remedied to restore westslope cutthroat trout populations. The limiting factors included: low-quality, low-complexity mainstem stream habitat and riparian zones; high stream temperatures in mainstem habitats; negative interactions with nonnative brook trout in tributaries; and potential survival bottlenecks in Coeur d'Alene Lake.

In 1994, the Northwest Power Planning Council adopted the recommendations set forth by the Coeur d'Alene Tribe to improve the Reservation fishery (NWPPC Program Measures 10.8B.20). These recommended actions included: 1) Implement habitat restoration and enhancement measures in Alder, Benewah, Evans, and Lake Creeks; 2) Purchase critical watershed areas for protection of fisheries habitat; 3) Conduct an educational/outreach program for the general public within the Coeur d'Alene Reservation to facilitate a "holistic" watershed protection process; 4) Develop an interim fishery for tribal and non-tribal members of the reservation through construction, operation and maintenance of five trout ponds; 5) Design, construct, operate and maintain a trout production facility; and 6) Implement a monitoring program to evaluate the effectiveness of the hatchery and habitat improvement projects. These activities provide partial mitigation for the extirpation of anadromous fish resources from usual and accustomed harvest areas and Reservation lands.

Since that time, much of the mitigation activities occurring within the Coeur d'Alene sub-basin have had a connection to the BPA project entitled "Implement of Fisheries Enhancement Opportunities on the Coeur d'Alene Reservation" (#1990-044-00), which is sponsored and implemented by the Coeur d'Alene Tribe Fisheries Program. Further, most of the aforementioned limiting factors are being addressed by this project either through habitat enhancement and restoration techniques, biological control, or with monitoring and evaluation that will provide data to refine future management decisions. This annual report summarizes previously unreported data collected during the 2010 calendar year to fulfill the contractual obligations for the BPA project. Even though the contract performance period for this project crosses fiscal and calendar years, the timing of data collection and analysis as well as implementation of restoration projects lends itself to this reporting schedule. The report is formatted into two primary sections:

- Monitoring and evaluation. This section comprises monitoring results for biological and physical indicators that describe the status and trends of trout populations and in-stream habitat features in our target watersheds. In addition, this section summarizes data that evaluate the effectiveness of implemented management actions in our watersheds, including recent channel restoration activities and a brook trout suppression program.
- Implementation of restoration and enhancement projects. This section comprises descriptions of the channel and riparian restoration projects that were implemented in 2010. Included in the action descriptions are summaries of the immediate effects that the restoration measures had on channel features.

To provide consistency between project objectives around which past reports have been structured and the work element format adopted by Pisces, relevant work elements and/or milestones found in our statement of work are referenced within each section.

2.0 STUDY AREA

The study area addressed by this report consists of the southern portion of Coeur d’Alene Lake and four watersheds – Alder, Benewah, Evans, and Lake - which feed the lake (Figure 1). These areas are part of the larger Coeur d'Alene sub-basin, which lies in three northern Idaho counties Shoshone, Kootenai and Benewah. The basin is approximately 9,946 square kilometers and extends from the Coeur d'Alene Lake upstream to the Bitterroot Divide along the Idaho-Montana border. Elevations range from 646 meters at the lake to over 2,130 meters along the divide. This area formed the heart of the Coeur d’Alene Tribe’s aboriginal territory, and a portion of the sub-basin lies within the current boundaries of the Coeur d’Alene Indian Reservation.

Coeur d'Alene Lake is the principle water body in the sub-basin. The lake is the second largest in Idaho and is located in the northern panhandle section of the state. The lake lies in a naturally dammed river valley with the outflow currently controlled by Post Falls Dam. The lake covers 129 square kilometers at full pool with a mean depth of 22 meters and a maximum depth of 63.7 meters.

The four watersheds currently targeted by the Tribe for restoration are located mostly on the Reservation (Figure 1), but cross boundaries of ownership and jurisdiction, and have a combined basin area of 34,853 hectares that include 529 kilometers of intermittent and perennial stream channels. The climate and hydrology of the target watersheds are similar in that they are influenced by the maritime air masses from the pacific coast, which are modified by continental air masses from Canada. Summers are mild and relatively dry, while fall, winter, and spring bring abundant moisture in the form of both rain and snow. A seasonal snowpack generally covers the landscape at elevations above 1,372 meters from late November to May. Snowpack between elevations of 915 and 1,372 meters falls within the “rain-on-snow zone” and may accumulate and deplete several times during a given winter due to mild storms (US Forest Service 1998). The precipitation that often accompanies these mild storms is added directly to the runoff, since the soils are either saturated or frozen, causing significant flooding.

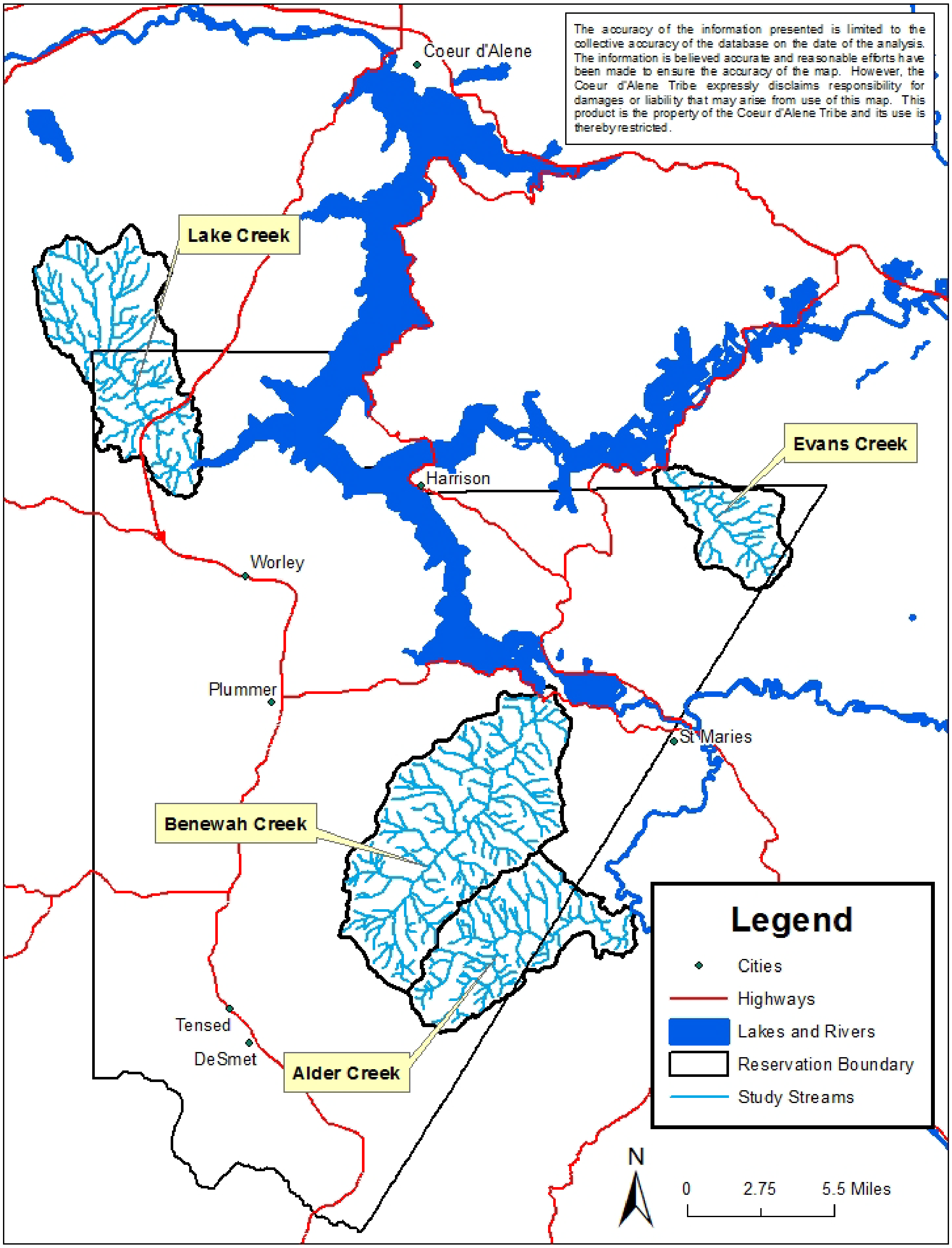

Figure 1. Locations of BPA Project 90-044-00 focal watersheds on reservation lands of the Coeur d'Alene Tribe.

3.0 MONITORING AND EVALUATION

3.1 Introduction

Salmonid populations and habitat features are monitored annually at index sites distributed across tributary and mainstem reaches to track trends within our four target watersheds (Vitale et al. 2002). Abundance trajectories for both native westslope cutthroat trout and non-native brook trout at index sites permits an examination of whether conditions are improving for either species and if improvements are operating at a local subbasin or a regional watershed scale. Further, the detection of declining trends may signal potential localized degradation or deficiencies in habitat conditions that need to be addressed. Trend monitoring also permits a description of temporal changes in spatial distributions to assess expansion and contraction rates of our salmonid populations to examine whether newly created suitable habitat is undergoing colonization. We not only assess relative changes in abundance at the reach scale, but also monitor overall trends at the watershed scale by tracking the number of juvenile outmigrants and returning adults in watersheds that support the adfluvial life-history. Monitoring of adfluvial production also entails assessing in-lake survival rates of both juvenile and adult cutthroat trout. In addition to our salmonid populations, we also track annual trends in temperatures given that high water temperatures during summer rearing periods have been considered to be a major factor limiting cutthroat trout production in our watersheds.

Effectiveness monitoring (Is the project achieving desired habitat and population benefits?) is also conducted in watersheds that are currently receiving treatments that address factors limiting cutthroat trout recovery. We are monitoring the response of salmonids and physico-chemical habitat features to implemented actions by measuring indicator variables in both treated and control reaches and watersheds. Effectiveness monitoring is currently being conducted in the upper Benewah watershed to evaluate responses to large-scale in-stream and riparian restoration actions and non-native brook trout suppression measures.

Since 2005, large-scale restoration actions have been directed toward approximately 5 km of contiguous main-stem habitat, and associated floodplain and riparian areas, in the upper Benewah watershed upriver of 9-mile bridge. These actions have been implemented to address dysfunctional stream processes, that have included channel incision, unstable streambanks and accelerated sedimentation, lack of habitat complexity, and elevated summer rearing temperatures from low stream canopy closure and reduced groundwater connection with adjacent floodplains (Vitale et al. 2007, Vitale et al. 2008; Firehammer et al. 2009, Firehammer et al. 2010). This main-channel reach was targeted because it had the potential to increase carrying capacity and production of juvenile cutthroat trout given its proximity and connectivity to important spawning tributaries.

Phase 1 restoration proceeded over the first four years, and consisted of the reconstruction of approximately 2500 m of channel habitats, which entailed reactivating meanders previously lost to channel avulsions; elevating riffle streambeds to promote overbank flooding and increase pool volume; adding large wood to in-stream habitats to provide cover, create pools, and aid in bank stabilization; and planting vegetation along channel margins and riparian zones for shade, bank stabilization, and future woody debris recruitment. Phase 2 restoration, which began in 2009, addresses approximately 2400 m of contiguous main-stem habitat upstream of that treated as part of Phase 1 restoration. Phase 2 restoration intends to use a more passive approach than that implemented during Phase 1, encouraging the establishment of persistent beaver dam complexes that will gradually aggrade the streambed over time and, via backwater effects, promote connectivity between the channel and adjacent floodplain habitats. In 2009, 439 m of stream channel were enhanced, and seven in-stream wood structures were installed to emulate the flow obstruction effects of natural dams and offer stability to existing dam complexes. Throughout both Phase 1 and 2 treated reaches, our monitoring program measures physical attributes linked to the quality of salmonid habitat at representative sites to evaluate whether restored conditions are being maintained or approaching desired conditions and benchmarks. In addition, supplemental effectiveness monitoring is conducted throughout the treated Phase 2 reach, whereby metrics associated with the stability of natural beaver dams (e.g., dam turnover) are monitored seasonally. Temperature responses are also monitored annually throughout this upper main-stem reach by examining changes in the availability and persistence of detectable thermal refugia in pool habitats before and after restoration.

A brook trout removal program was initiated in 2004 to suppress the numbers of brook trout found in main-stem and tributary habitats in the upper portion of the Benewah watershed. This control was deemed necessary because brook trout have been shown to negatively impact cutthroat trout when populations of the two species overlap (Griffith 1988; Adams et al. 2001; Peterson and Fausch 2003; Peterson et al. 2004a; Shepard 2004). However, unlike other brook trout removal projects that have focused on chemical eradication and subsequent measures to prevent re-colonization, such as passage barriers (Shepard et al. 2003), we have used less intensive methods (e.g., electrofishing, trapping) to annually control brook trout. Our approach was tempered by the desire to maintain connectivity with the lake to promote the migratory life-history variant of our cutthroat trout population and its concomitant high productivity potential. We felt that the benefits of unimpeded access and the expression of the cutthroat adfluvial life-history greatly outweighed the benefits of brook trout eradication in isolated tributaries (Peterson et al. 2008a). Further, eradication treatments have not always proven entirely successful, and, within our watershed, would require large-scale chemical treatments that may receive public opposition and an extensive trapping and hauling program to supply migratory adult cutthroat trout to the various isolated spawning tributaries. Monitoring the success of the removal program is conducted by examining changes in metrics of brook trout abundance in index reaches in the upper Benewah watershed.

The objectives of the monitoring and evaluation section with corresponding BPA Pisces scope of work (SOW) elements are as follows:

1) Assess temporal and spatial changes in cutthroat trout abundances and distribution
    a) Measure the productivity of the adfluvial life-history of cutthroat trout by analyzing data collected from migration traps and PIT tag systems installed in Lake and Benewah creek watersheds (SOW 2009 Elements L,M,N,O; SOW 2010 Elements S,T)
    b) Conduct electrofishing population surveys at index sites to assess relative changes in the distribution and abundance of salmonids in mainstem and tributary reaches within the four target watersheds (SOW 2010 Elements Q,S,U)

2) Collect and summarize longitudinal trends in water temperatures by deploying loggers within monitored watersheds (SOW 2010 Elements X,Z,AA)

3) Evaluate effectiveness of habitat restoration in the upper Benewah watershed

a) Assess differences in thermal heterogeneity in pool habitats in treated mainstem reaches before and after restoration (SOW 2010 Elements Y,Z,AA)
b) Assess differences in physical habitat indicators measured at treatment and control sites (SOW 2010 Elements V,Z,AA)
c) Assess seasonal changes in metrics measured at beaver dams in the upper Benewah watershed (SOW 2010 Elements W,Z,AB)

4) Reduce the abundance and distribution of non-native brook trout in the upper Benewah watershed
   a) Remove brook trout from Benewah Creek (SOW 2010 Element K)
   b) Test the effectiveness of the removal program by examining trends in brook trout abundances in the upper Benewah watershed (SOW 2010 Elements L,S)

3.2 Methods

3.2.1 Status and trend monitoring

3.2.1.1 Adfluvial cutthroat trout migration

Migration traps were installed in both Lake and Benewah creeks to collect abundance and life-history information on adfluvial cutthroat trout. A modified resistance board weir trap design (Tobin 1994; Stewart 2002) was used in both watersheds to intercept adult cutthroat migrating upriver (hereafter, referred to as UP traps). The modification entailed securing a cabled pulley system to the trap so that panels could be manually lowered or raised to maintain their height above the water surface, and was incorporated into the UP trap at Lake Creek in 2009 to address problems associated with periodic high-volume freshets depressing trap panels below the water surface. Given the increase in capture efficiency observed at the Lake Creek UP trap in 2009, similar modifications were introduced into the Benewah Creek UP trap in 2010 to improve trap performance. In both systems, UP traps were installed in late winter after ice out but early enough to attempt to capture the majority of the spawning run.

To capture post-spawn adults and outmigrating juveniles, a modified fence-weir design was used in both watersheds as the downriver trap (hereafter, referred to as the DOWN trap). The design incorporated pop-out panels that could be removed during periods of high flow to relieve pressure on the trap. DOWN traps were installed in the spring in both systems as early as possible under amenable discharge levels. In both watersheds, traps were positioned downriver of principal spawning tributaries and of most of the recently implemented and projected habitat restoration projects. The UP trap on the Benewah Creek mainstem was installed at river kilometer (rkm) 14.5, with the DOWN trap located immediately upstream (Figure 2); the UP trap on the Lake Creek mainstem was installed at rkm 6.0, with the DOWN trap located approximately 0.13 km upriver (Figure 3). Traps were checked and cleaned frequently during periods of operation, with checks occurring typically daily during high discharge and associated peak movement periods from March through early June to ensure proper trap performance and to assess migration timing and relative abundance.

PIT-tag arrays have been installed immediately downstream of the UP traps in both the Lake (~ 10 m downstream) and Benewah (~2 m downstream) systems. Detections by these arrays permit an evaluation of adult return rates from prior outmigrating cohorts and allow an in-season examination of trap performance. The Lake Creek array spans the entire stream channel and consists of three side-by-side 5x5 ft antennas; two side-by-side 10x4 ft antennas constitute the array in Benewah Creek and span the entire wetted width of the channel under most flows. The Lake and Benewah creek PIT-tag arrays were calibrated and started on February 26 and March 1, respectively. Logged detection data were downloaded several times a week to monitor both adult and juvenile fish passage throughout the migratory period. Benewah and Lake creek PIT-tag arrays were respectively shut down on July 12 and 16 because of lack of fish detections and the absence of fish captured in DOWN traps.

Total lengths (TL, mm), weights (Wt, g), and scales were collected and condition factors (estimated as $10{,}000^{*}Wt / TL^3$) calculated from all adult adfluvial cutthroat trout captured in both traps. Adults captured in traps were also scanned for the presence of PIT-tags using a hand-held wand. Adfluvial adults captured in the UP trap that did not scan and were not adipose-clipped received a hole puch along the outer margin of the left opercle. In addition, these adults received a PIT-tag that was inserted into the muscle tissue immediately posterior to the insertion of the right pelvic fin; tag insertion into the body cavity was not considered lest they would become expelled on the spawning grounds (Peterson et al. 2004a; Bateman et al. 2009). Adults that did scan at the UP trap (i.e., either tagged as juveniles and hence were adipose-clipped, or tagged as adults in previous years) received a hole punch along the outer margin of the right opercle. Tag retention for all groups of tagged adults was assessed during their recapture in the DOWN trap using the opercle punch as a double-tag. Opercle-punches also served as marks that would be used in recapture events at the DOWN trap to generate an estimate of the abundance of adults that migrated upriver of the UP trap. Adult abundance was estimated using Chapman's modification of the Petersen index:

$$N = \frac{(M+1)(C+1)}{(R+1)} - 1, \text{ (}\textit{Equation 1}\text{)}$$

where:

- N= the abundance estimate;
- M= number of adults that received a mark;
- C= number of adults captured in the DOWN trap; and
- R= number of adults captured in the DOWN trap that had been marked.

The variance estimate of N was calculated as follows:

$$v(N) = \frac{(M+1)(C+1)(M-R)(C-R)}{(R+1)^2(R+2)}. \text{ (}\textit{Equation 2}\text{)}$$

An approximate 95% confidence interval was then calculated as $N \pm 1.96\sqrt{v(N)}$.

Lengths were collected from all outmigrating juvenile cutthroat trout captured in DOWN traps. In addition, at least 25% of the captured juveniles in each system received intra-peritoneal PIT tags following the Pacific States Marine Fish Commission PTAGIS guidelines. Weights and scales were collected from these tagged fish, and the adipose fin was clipped to identify its tagged status for recapture events. Attempts were made to representatively tag juvenile fish throughout the entire outmigration period, with subsamples of PIT-tagged juveniles used in trap efficiency trials to estimate outmigrant abundance. In addition, subsamples of PIT-tagged fish used in efficiency trials were held for a day in a PVC-framed net pen upriver of the DOWN trap before their release to permit estimates of post-implantation survival and tag retention rates. Outmigration estimates for each release trial period were derived from recaptured fish enumerated at the trap using the following equation (Carlson et al. 1998):

$$U_h = \frac{(u_h)(M_h+1)}{m_h+1}, \text{ (}\textit{Equation 3}\text{)}$$

where:

- U_h = outmigrant abundance, excluding recaptured fish, in trial period h;
- u_h = number of untagged fish in trial period h;
- M_h = number of tagged fish released in trial period h; and
- m_h = number of tagged fish recaptured in trial period h.

The variance estimate of U_h was calculated as follows:

$$v(U_h) = \frac{(M_h + 1)(u_h + m_h + 1)(M_h - m_h)(u_h)}{(m_h + 1)^2 (m_h + 2)}. \quad (\textit{Equation 4})$$

Total outmigration abundance (*U*) and variance (*v(U)*) were then calculated as the sum of the respective estimates over all trial periods. An approximate 95% confidence interval was then calculated as:

$$U \pm 1.96\sqrt{v(U)}.$$

Because observed rates of trap passage varied for tagged fish released above the DOWN trap, all marked fish did not have an equal probability of being caught during a release trial's recapture period. Because of this mark-recapture model violation, a modification of the stratified design used by Carlson et al. (1998) was used to estimate release trial abundances. During each trial period, only those tagged fish available for recapture were used in calculations rather than all tagged fish released during the trial period (i.e., M_h in the equation above). The number of tagged fish considered available for recapture during each trial period was calculated as the number of tagged fish released in that period discounted by those that were enumerated at the trap during subsequent release trial periods.

3.2.1.2 Salmonid stream surveys

The channel types delineated during prior pilot habitat surveys (Lillengreen et al. 1996) served as basic geomorphic units for selecting sample index sites for conducting fish population surveys. In these early surveys, stream reaches were stratified into relatively homogeneous types according to broad geomorphologic characteristics of stream morphology, such as channel slope and shape, channel patterns and channel materials, as defined by Rosgen (1994). Stream reaches were further stratified by basin area to ensure that both mainstem and tributary habitats were represented in the stratification scheme. Sample index sites within each reach stratum were randomly selected in proportion to the total reach length (Figure 2-5). The length of each index site was standardized to 61 meters to encompass at least 20 channel widths for most sites.

Sites were electrofished between July 20 and September 22 to describe the distribution of salmonids and provide an index of abundance during base flow conditions. Electrofishing was conducted using a Smith-Root Type VII pulsed-DC backpack electrofisher, and followed established guidelines and procedures to standardize capture efficiency (Reynolds 1983). Block nets were placed at the upstream and downstream boundaries of each site to prevent immigration and emigration during sampling. Only a single pass was conducted at each site.

Captured salmonids, including westslope cutthroat trout and brook trout *(Salvelinus fontinalis)* were identified, enumerated, and measured for total length. Weights and scales were collected from a subsample of 8-10 fish within each 10 mm length group for each species and watershed. Based on age-at-length keys derived from previously collected scale samples and length distributions derived from fish captured in 2010, cutthroat and brook trout respectively greater than 70 and 75 mm were considered to be at least one year of age. Indices of abundance were calculated for fish of all ages and for those considered at least one year of age (hereafter referred to as age 1+) separately for each salmonid species, and converted to fish/100 m of stream length to permit comparisons across sites. Other species, such as dace (*Rhinichthys* spp.), redside shiner (*Richardsonius balteatus*), longnose sucker (*Catostomus catostomus*), and sculpin (*Cottus* spp.), were considered incidental catch and were only counted during the electrofishing pass.

As an alternative experimental sampling technique, we also deployed fyke nets in deep pools in restored reaches of the Benewah mainstem to evaluate their effectiveness at capturing salmonids given that electrofishing has been found to be ineffective at sampling fish in these types of habitats. Fyke nets consisted of two rectangular frames in the front, each 6 by 3 ft in size, followed by four hoops, each 2.5 ft in diameter. A 50 ft long lead was attached to the first rectangular frame. Netting for both the lead and the net was 0.5 in. square. Nets were deployed in pools 4-6 ft in depth with the lead running the length of the pool and secured to the bank. Nets were strategically deployed so that fish could be captured moving from both upriver and downriver reaches. Fyke nets were deployed in four locations in restored reaches: (1) Approximately 1800 ft upstream of 9-mile bridge in a reach that was restored in 2005; (2) In a deep pool in a reach that was restored in 2006 at the confluence of Whitetail Creek; (3) Approximately 150 ft downstream of the confluence of Windfall Creek in a reach that was restored in 2004; and (4) In the deep pool created at the confluence of Windfall Creek. Nets were typically checked every 1-3 d and a total of 3 consecutive checks were conducted at each of the four locations. During each check, all fish captured were recorded and released, except for brook trout which were sacrificed. In addition, all fish released during the first two checks at a location received a caudal lobe clip to identify their recapture in subsequent checks. For each species, index of abundance was calculated as the number of fish captured divided by soak days.

3.2.1.3 Stream temperatures

Stream temperatures were continuously monitored every 30 minutes at fixed locations along mainstem reaches and in major tributaries of upper Benewah and Lake creek watersheds using HOBO Temp Pro (Onset Computer Corp.) digital temperature dataloggers (accurate to ±0.2 °C). In the upper mainstem of Benewah Creek, dataloggers were placed in main channel locations, in connected side-channels influenced by springbrooks, and in isolated springbrooks. Air temperatures were also recorded using HOBO H8 Pro Series loggers (Onset Computer Corp.) at a forested and open meadow site in both upper Benewah and Lake creek watersheds. Daily mean water temperatures, and the percent time in which logged temperatures exceeded 17°C were computed for each HOBO logger. The threshold value of 17°C was used because it has been considered to be a 95% upper limit for optimal cutthroat trout growth (Bear et al. 2007). Daily temperature metrics were used to calculate monthly mean values for July and August to permit comparisons within watersheds and across years.

3.2.2 Effectiveness monitoring – Response to habitat restoration in Benewah watershed

3.2.2.1 Monitoring and evaluation of physical habitat metrics

Riparian and in-channel physical attributes that have been linked to the quality of trout habitat were monitored at established 152 m index sites in treated and control reaches in the upper Benewah main-stem (Figure 6). One site was located in each of the four contiguous reaches (T-01 – T-04) that were treated as part of Phase I restoration from 2005 to 2008, and two sites (T-05 and T-06) were located in the reach that has been receiving Phase II treatment since 2009. Though bounded by Phase II reaches, site T-07, which is located at the Windfall confluence, had been treated in 2004 with channel reconstruction measures that resembled those implemented during Phase I restoration. Two sites (C-01 and C-02) were also located in a control, untreated reach upriver of the two phases of main-stem restoration. The methods used to measure the physical attributes, which included large woody debris, canopy cover, substrate composition, and pool habitat, are described in detail below.

Pool metrics in main-stem reaches

Pools were identified by first measuring the depth at the downstream control point. The maximum depth of the pool was calculated from measuring the depth at the deepest part of the pool. If the maximum depth minus the minimum depth was greater than one foot residual depth, the habitat unit was classified as a pool. Upstream and downstream pool boundaries were those locations at which residual pool depth equaled one foot (i.e., depths along the thalweg of a pool were greater than one foot of residual depth). For each pool, three stream widths were measured: 1) half-way between maximum depth and the downstream end of the pool, 2) the point of max depth, and 3) half-way between the maximum depth and the upstream end of the pool. Three depth measurements were taken where each channel width was measured. Channel widths only included the portion of the channel where the water depth was greater than one foot of residual depth. Pool lengths and stationing of each width location were collected so that a pool volume could be determined. In addition, information about the type of pool and the mechanism forming the pool were also collected. Pool forming mechanisms include boulder, channel hydraulics, wood, beaver dam, and artificial structure (e.g., culvert). Types of pools include plunge, dammed, scour, and backwater. The aim with this methodology was to examine the quantity and quality of pool habitats that were available at base-flow conditions.

Channel substrate

Wolman pebble counts (Wolman 1954) were completed at riffles and pool tailouts along the survey reach. At each of these points a measuring stick or finger was placed on the substrate and the one particle the tip touched was picked up and the size measured. Particle size was determined as the length of the "intermediate axis" of the particle; that is the middle dimension of its length, width and height. A total of 50 particles were counted across bankfull at each location, and a total of five riffle and two pool tailout locations distributed across the reach were sampled. Particles were noted whether they were sampled within or without the wetted channel width. Pebble count data were input into spreadsheets to graph the distribution of particle sizes and calculate pertinent descriptive criteria such as percent fines and a particle size index (D value) for each habitat type.

Canopy density

Vegetative canopy density (or shade) was determined using a conical spherical densiometer, as described by Platts et al. (1987). The densiometer determines relative canopy "closure" or canopy density, which is the amount of the sky that is blocked within the closure by vegetation. Canopy cover over the stream was determined at ten equidistant locations distributed throughout the survey reach. At each location, densiometer readings were taken one foot above the water surface at the following stations: once facing the left bank, once facing upstream at the middle of the channel, once facing downstream at the middle of the channel and once facing the right bank. Percent density was calculated collectively over these four readings for each of the ten locations with an overall mean calculated for the reach.

Large woody debris

Large woody debris (LWD) was surveyed throughout the entire 152 m reach. All LWD that was greater than 4 inches in diameter at the small end and 4 ft in length was counted. In addition to these criteria, LWD also had to be either partially located within bankfull or suspended across the channel above the water surface. Living trees and shrubs, however, did not qualify as LWD.

For all pieces, the mean diameter and length were estimated and tallied in appropriate size ranges. Size classes were 4-8, 8-12, 12-18, 18-24, and >24 inches for mean diameter. Size classes were 4-10, 10-15, 15-20, 20-25, and >25 feet for length. For every 5^{th} piece, the mean diameter and length were measured to calibrate the accuracy of the visual length and diameter estimates. Volume of each piece was calculated using the mid-point values of the length and diameter categories to which the piece was assigned. Total volume and density of LWD was calculated for each site and expressed per meter of stream length.

In addition to measuring the volume of LWD, data denoting the function and position of each identified piece were also collected to aid in describing how LWD was providing habitat and influencing channel form within the site. Function categories included: accumulating sediment, forcing a pool to form upstream or downstream, providing in-stream cover, providing bank stabilization, or none of the above. More than one category could be assigned to individual wood pieces. Categories to describe the position of the identified piece in relation to the stream included: elevated above the bankfull channel, one end within and the other end outside bankfull channel, completely within bankfull channel but exposed, or within bankfull channel but partially buried.

3.2.2.2 Monitoring and evaluation of thermal refugia

Thermal heterogeneity at fine-scale, riffle/pool sequences was assessed in upper Benewah main-stem reaches in mid-summer using a rapid-response digital thermistor probe (Cooper Instruments model TM99A-E, accurate to within ±0.1 °C). The thermistor probe was attached to a surveying rod, permitting simultaneous measurements of depth and temperature. While wading upstream, water temperature and depth (m) were recorded both at a riffle and at the deepest part of the associated pool upstream. The relationship between residual pool depth and the calculated riffle-pool temperature difference was examined to evaluate the availability of thermal refugia across restored reaches of the upper Benewah main-stem.

3.2.2.3 Monitoring and evaluation of beaver dam complexes

Beaver dams were surveyed during two different time periods along a 3.5 km reach of the upper Benewah main-stem that is currently receiving treatment as part of Phase 2 restoration implementation. The first survey occurred from late June to early July, and the latter survey occurred during early October. Various attributes that described dam morphology and in-stream habitat influenced by the dam were measured and recorded at each dam surveyed. Dam morphology attributes included dam type, which indexed the apparent stability, complexity, and derelict state of the dam; the materials used to build the dam; and the dam width and height (Table 1). The in-stream habitat influenced by the dam was considered to be that channel length that was backwatered by the dam (i.e., the length of channel upstream over which water surface elevation did not change). Attributes evaluated along the backwatered channel length included the inundated surface area, pool surface area, pool volume, and mean residual pool depth. Inundated surface area was calculated by multiplying the backwatered channel length by the average of five wetted channel widths measured at equidistant intervals along the channel length. Pools were identified and measured along the backwatered length using the criteria and protocol described above (see *Pool metrics in main-stem reaches*). Pool lengths and their respective measured widths and depths were used to calculate the collective pool surface area and volume, and the mean residual depth for pools associated with each dam. In 2010, in-stream habitat was only measured during the fall survey. Paired data collected at dam locations were used to evaluate seasonal changes in dam height from fall of 2009 to early summer of 2010 and from early summer of 2010 to fall of 2010. Seasonal means for dam height were analyzed separately for three different reaches: (1) the lowermost reach that is laterally bounded by an open meadow; (2) a middle reach with a proximate relatively intact riparian forest community and presence of large wood in the channel; and (3) an upper reach predominantly bounded by meadow and in which many of the engineered and reinforced LWD structures have been installed.

Table 1. List of categories that describe available dam types and dam-building materials. Active dams are considered those in which a presence of fresh material (e.g., green stems, recently placed mud) has been detected.

Attribute	Categories
Dam type	Acitve single dam with large wood
	Active dam complex composed of multiple dams utilizing large wood and/or mid-channel islands
	Active single dam without large wood
	Inactive single dam with large wood
	Inactive dam complex composed of multiple dams utilizing large wood and/or mid-channel island
	Inactive single dam without large wood
Dam materials	Key pieces (> 4 inches in diameter; length >= bankfull width)
	Other large wood (> 4 inches in diameter)
	Large wood with root wad
	Small wood (< 4 inches in diameter)
	Herbaceous plant material
	Mud
	Other

3.2.3 Effectiveness monitoring - Response to brook trout removal in Benewah watershed

In late summer and early fall, single-pass electrofishing was used to remove non-native brook trout from an index 2 km main-stem reach from the 12-mile bridge upstream to the confluence of the West and South Forks in the upper Benewah watershed. High densities of adult brook trout have consistently been found in this reach, and suitable spawning habitat is seemingly much more prevalent in this reach than in mainstem reaches downriver that are of lower gradient and dominated by beaver dam pools. Single-pass electrofishing was also used to remove non-native brook trout from lower reaches of Windfall, Schoolhouse, South Fork Benewah, and West Fork Benewah tributaries in the upper Benewah watershed. Removal efforts occurred before the spawning period for brook trout but after population surveys were completed in the upper Benewah watershed to prevent the removal activities from biasing index site abundance estimates.

A temporary trap was installed on the Benewah mainstem immediately upriver of 12-mile bridge to intercept ascending brook trout and hence prevent access to habitat upriver. The trap consisted of a downriver fixed weir that spanned most of the channel width but maintained a narrow opening along one bank to allow passage. Another fixed weir spanning the entire channel width and obstructing further upriver movement was installed approximately 25 m upriver. Periodically, the 25 m of enclosed stream length was shocked to remove any brook trout that had entered. In addition, a temporary fixed weir spanning the entire channel width was installed at the mouth of Windfall Creek to prevent access to habitat upriver in this tributary. Installation and removal dates were respectively August 25 and October 25 for the traps deployed on the main-stem and at the mouth of Windfall Creek. The UP trap at 9-mile bridge also remained deployed in 2010 from the end of spring trapping through the end of the removal efforts to prevent brook trout from ascending into the upper watershed.

Trends in brook trout abundance were examined using various indices to evaluate the population response to the suppression program. Changes in numbers of brook trout removed from the 2 km index main-stem reach were examined over the period from 2005 to 2010 given that this reach had been consistently sampled in all six years. Qualitative differences in the length distributions of brook trout removed from the 2 km mainstem index reach were also examined over years to evaluate whether the modified removal tactics that were first employed in 2009 (i.e., only removing fish from the 2 km index main-stem reach) had an impact on reproductive success.

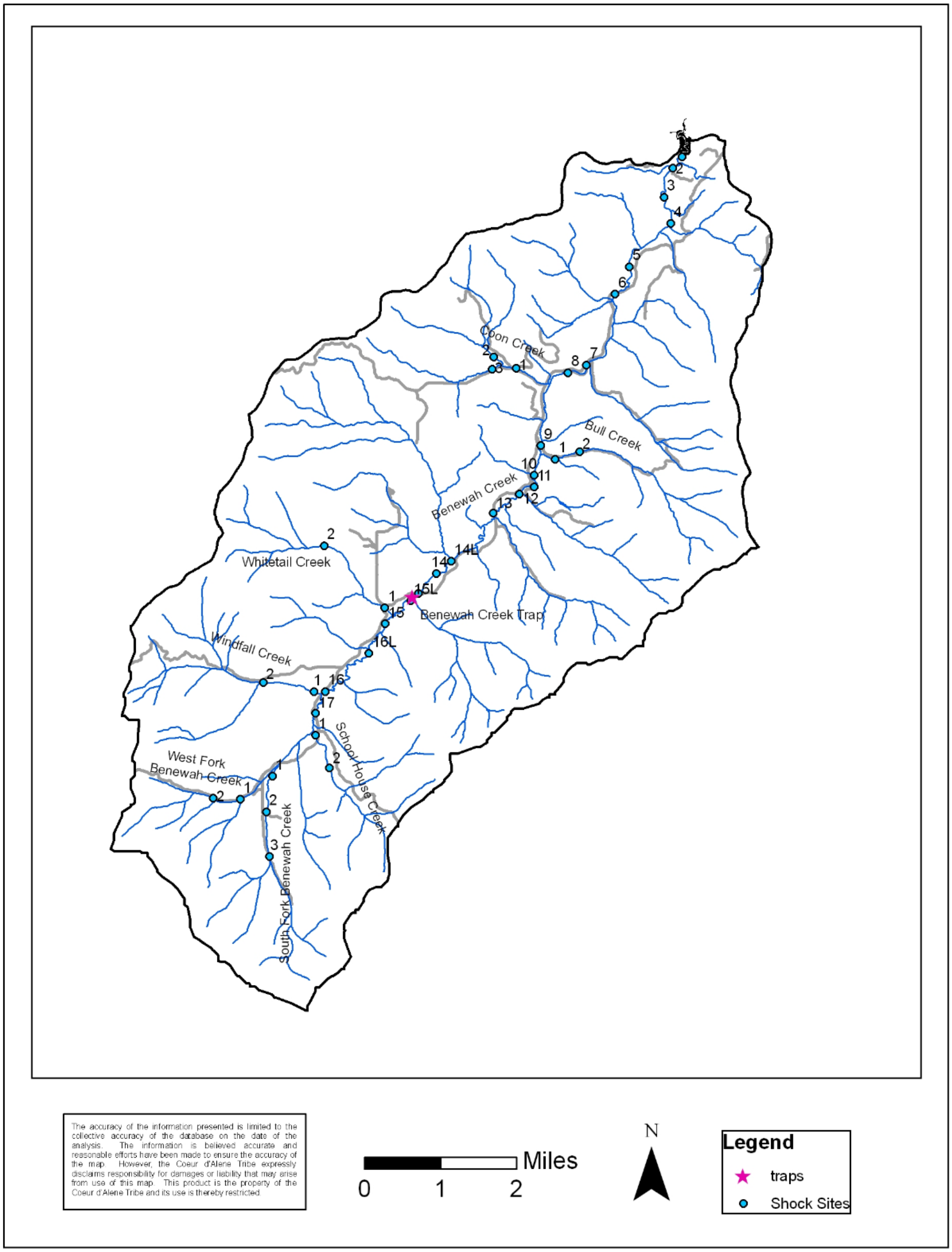

Figure 2. Map of Benewah Creek depicting index sites sampled during salmonid population surveys in 2010. The location of the traps and PIT-tag array is indicated by the star.

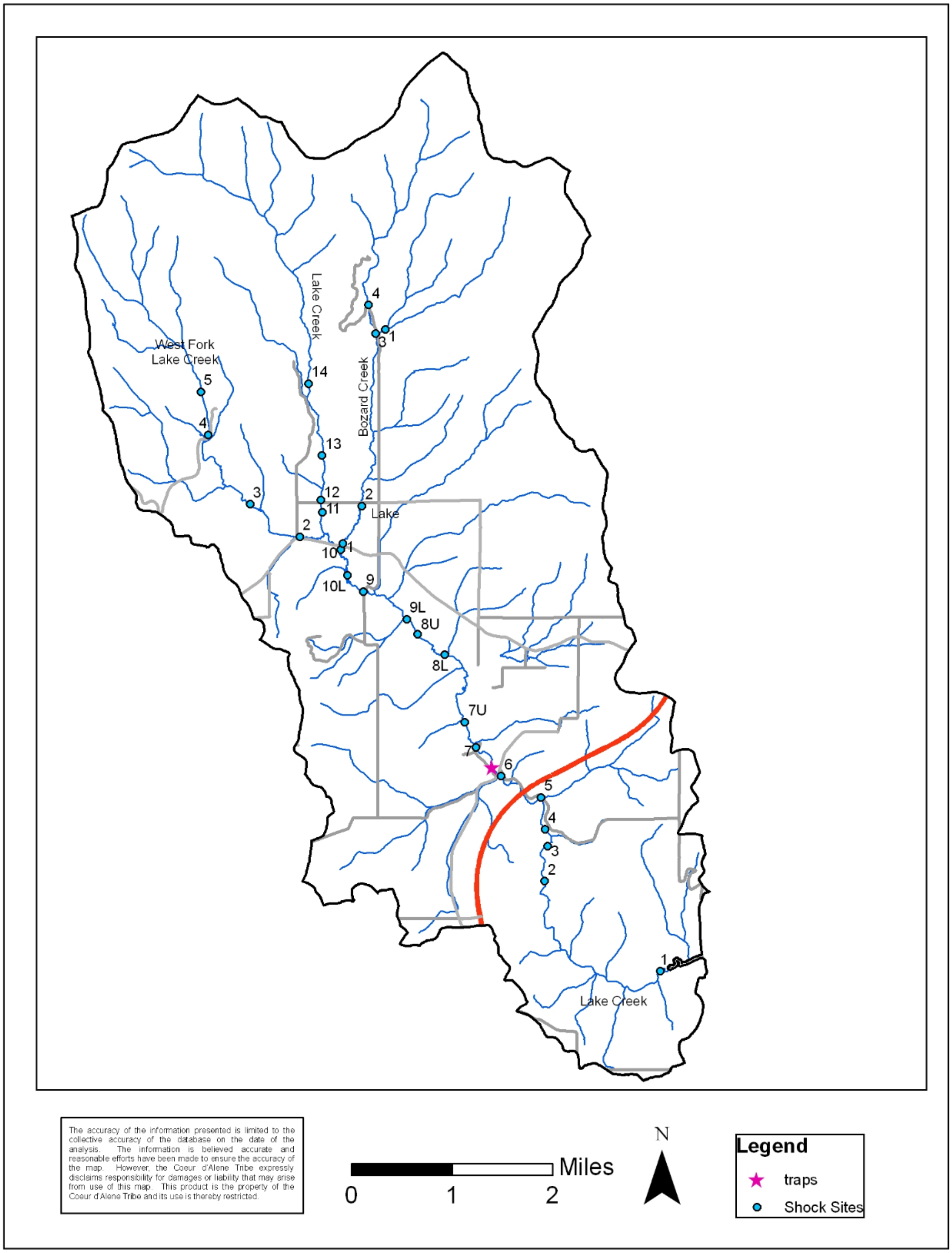

Figure 3. Map of Lake Creek depicting index sites sampled during salmonid population surveys in 2010. The location of the traps and PIT-tag array is indicated by the star.

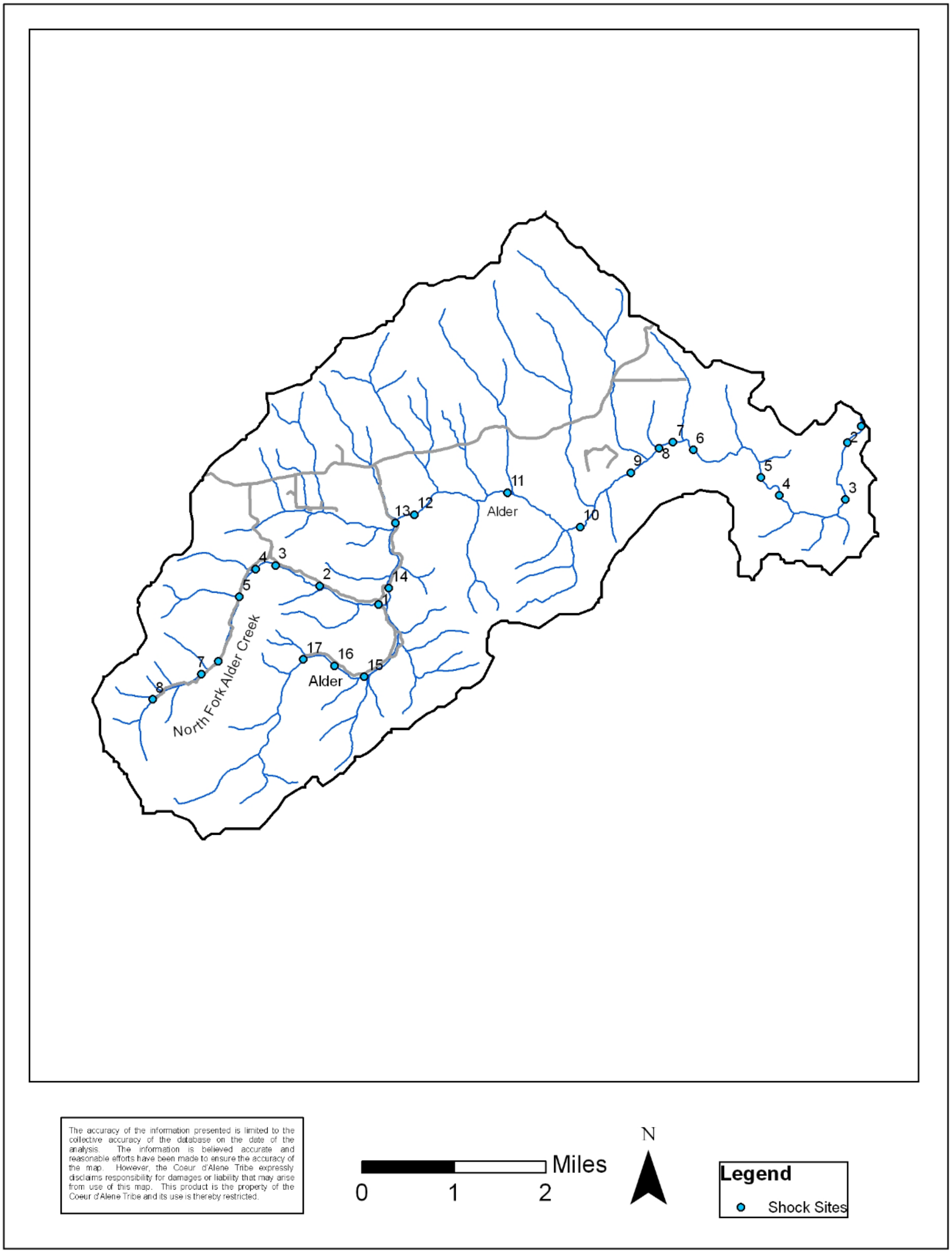

Figure 4. Map of Alder Creek depicting index sites sampled during salmonid population surveys in 2010.

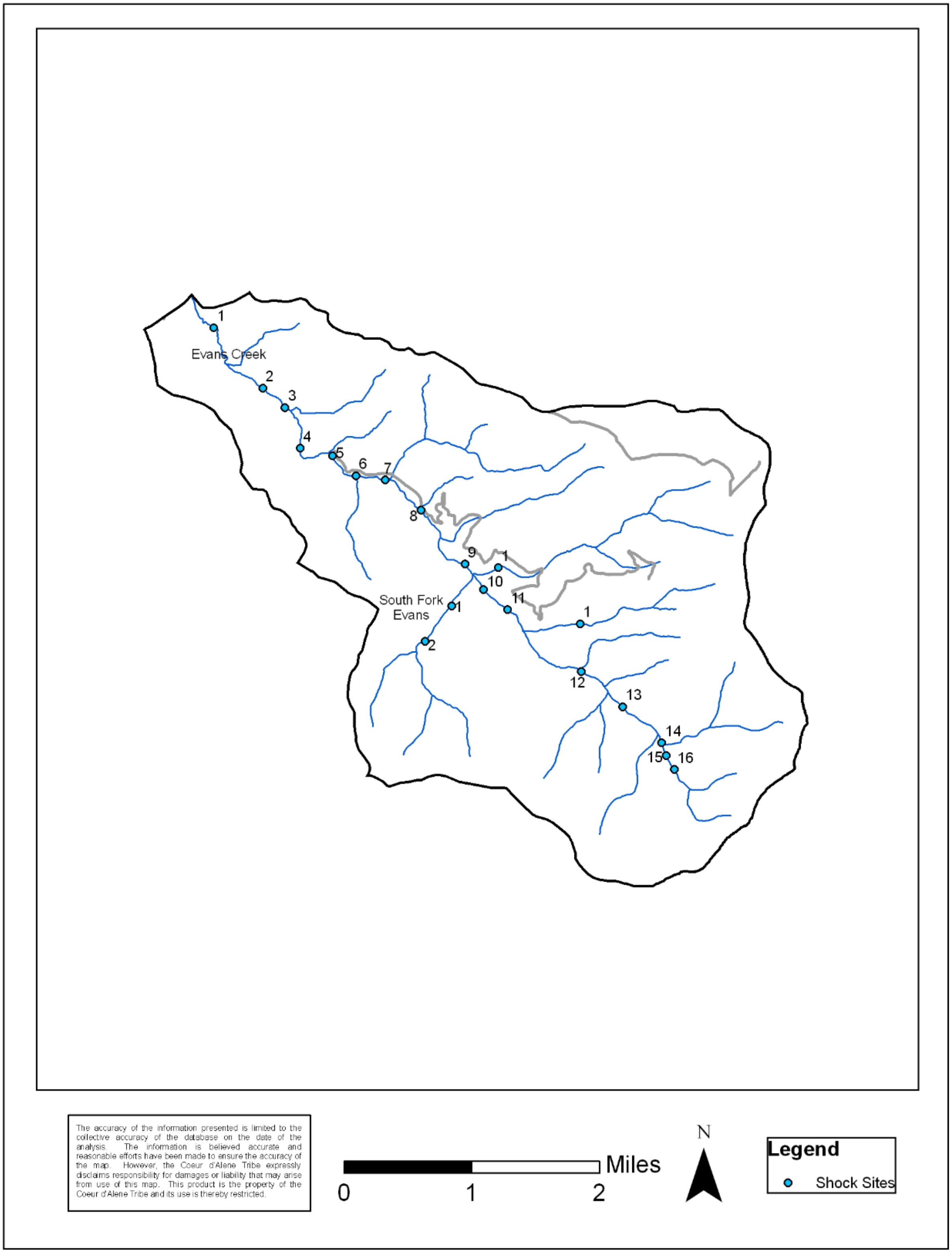

Figure 5. Map of Evans Creek depicting index sites sampled during salmonid population surveys in 2010.

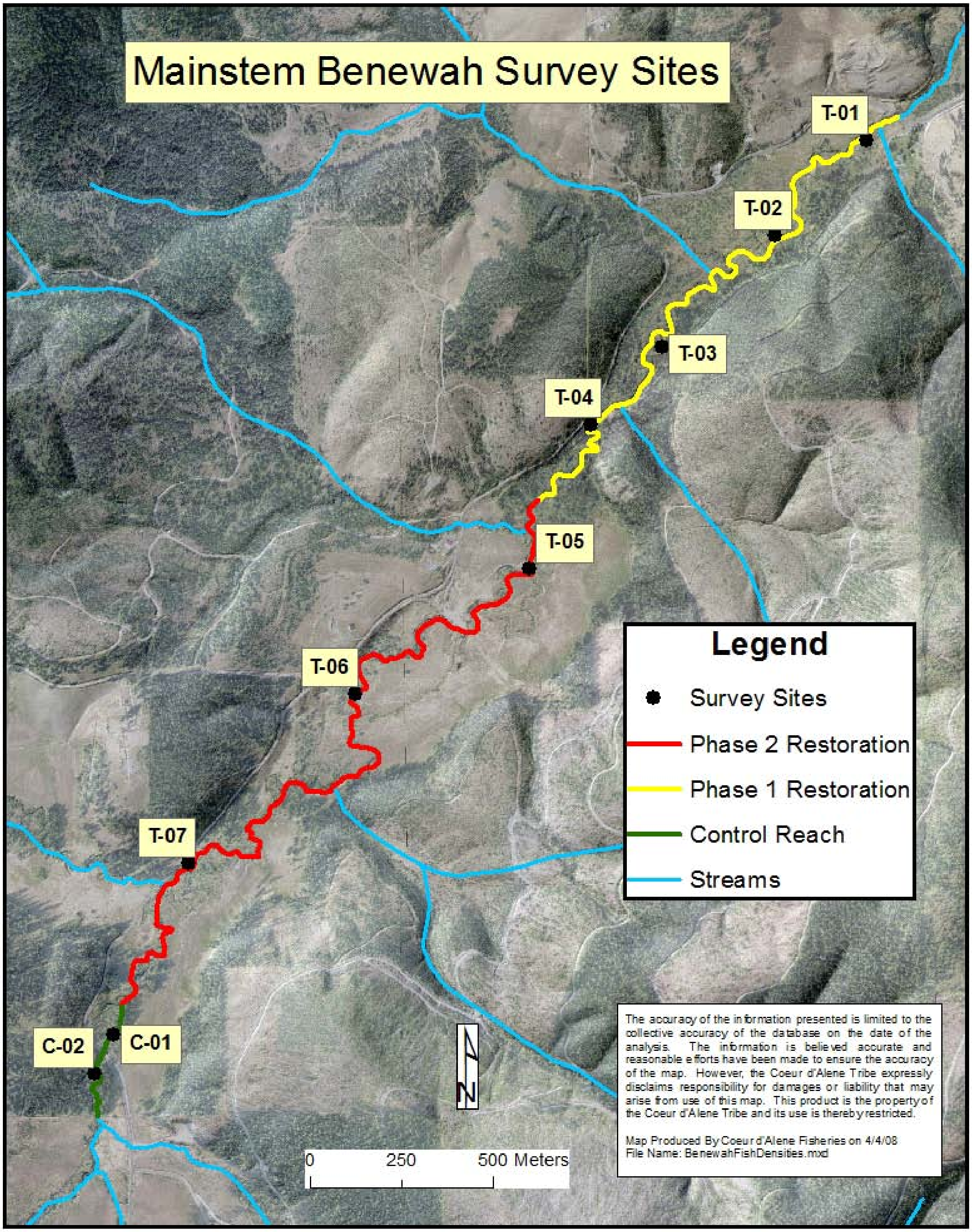

Figure 6. Location of habitat survey sites in the upper Benewah main-stem that have received treatment (T) as part of Phase 1 and 2 restoration, and that serve as control (C) sites for monitoring.

3.3 Results

3.3.1 Status and trend monitoring

3.3.1.1 Lake Creek adfluvial cutthroat trout migration

The UP trap was installed in the mainstem of Lake Creek on March 1 and was removed on May 10 in 2010, yielding an operable period of 71 d. During the operable period, the trap was checked a total of 49 d (69% of the days), and was considered fishing 100% of the time that it was monitored (Figure 7). The DOWN trap was installed in Lake Creek on April 16, and was removed on June 22 in 2010, yielding an operable period of 68 d. During this time, the trap was checked a total of 54 d (79% of the days), and was considered fishing approximately 91% of the time that it was monitored. The DOWN trap was compromised during several rain events in which water was found flowing over the trap panels or induced scouring underneath the trap panels. These high flow events occurred on April 30, from June 3-8, on June 17, and from June 20-22 (Figure 7).

A total of 106 adfluvial adult cutthroat trout was captured in the UP trap (Table 2). Seventy-nine of these were identified as females (75%) with a mean length and weight of 358 mm and 422 g, respectively. Twenty-four of the fish were identified as males with a mean length and weight of 370 mm and 473 g, respectively. Although fish were captured as early as March 17 and as late as April 29, 70 of the adults (66%) were captured during a 8 d period from April 14 to April 21 when discharge was relatively stable but mean daily water temperatures were increasing from 6 to 11^{o}C (Figure 7, Figure 8). Of the 106 adults captured at the trap, 104 received opercle punches with 84 of the punched fish receiving PIT-tags.

Twenty of the adults captured in the UP trap had been tagged in prior years, with eight fish tagged as adults in 2009 and 3, 1, 6, and 2 fish tagged as juveniles in successive years from 2005 to 2008, respectively (Table 3). All four of the fish tagged in 2005 and 2006 had also been detected as spawning adults in years prior to 2010, with one adult female from the 2005 outmigrating juvenile cohort detected in each spawning run from 2007 to 2009. Of those fish that had been tagged in 2007 and 2008, all except one was initially detected as an adult this year. As illustrated by an average annual length increment of 2.3 mm for those 8 repeat spawners that had been tagged as adults in 2009, somatic growth decreases markedly with age after maturation (Table 3). Incidentally, two of the adults captured in the UP trap were adipose-clipped, indicating that they had been tagged as juveniles, but did not scan.

Data collected by the interrogation antennae in Lake Creek indicated that all upriver adult migrants were not captured soon after they had approached the UP trap in 2010. Days of interrogation before trap capture ranged widely from 2 to 22 d for those 20 detected adults that had been tagged in prior years, with a mean detection period of 10.3 d (Table 3). Furthermore, an additional 11 presumed adults (i.e., tagged as adults in 2009 or as juveniles prior to 2009) that were interrogated by the array were never captured in 2010. The mean number of detected days between first and last interrogation dates for these 11 fish was 10.9 d; 6 of the fish were detected for more than 10 d (range, 11-30 d) before apparently moving back downriver (Table 3).

A total of 91 adfluvial adults was captured in the DOWN trap (Table 2). Of these 91, 67 (74%) were identified as females with a mean length of 359 mm and a mean weight of 388 g, and 22 as males with a mean length of 375 mm and a mean weight of 455 g. The mean condition factor was noticeably lower for females captured in the DOWN trap (0.82) than for females caught in the UP trap (0.90), indicating that many of the outmigrating females likely spawned (Table 2). Catch rates of outmigrating adults were relatively similar throughout the period from April 20 to May 13 in which most of the fish were captured; the 17 adults processed on May 10, though large relative to other days, was the result of a 3 d catch (Figure 8). Sixty-two of the PIT-tagged adults that had either been tagged or interrogated at the UP trap were captured in the DOWN trap. The mean elapsed period between detections for these fish was greater for males, 19.2 d, than for females, 11.4 d.

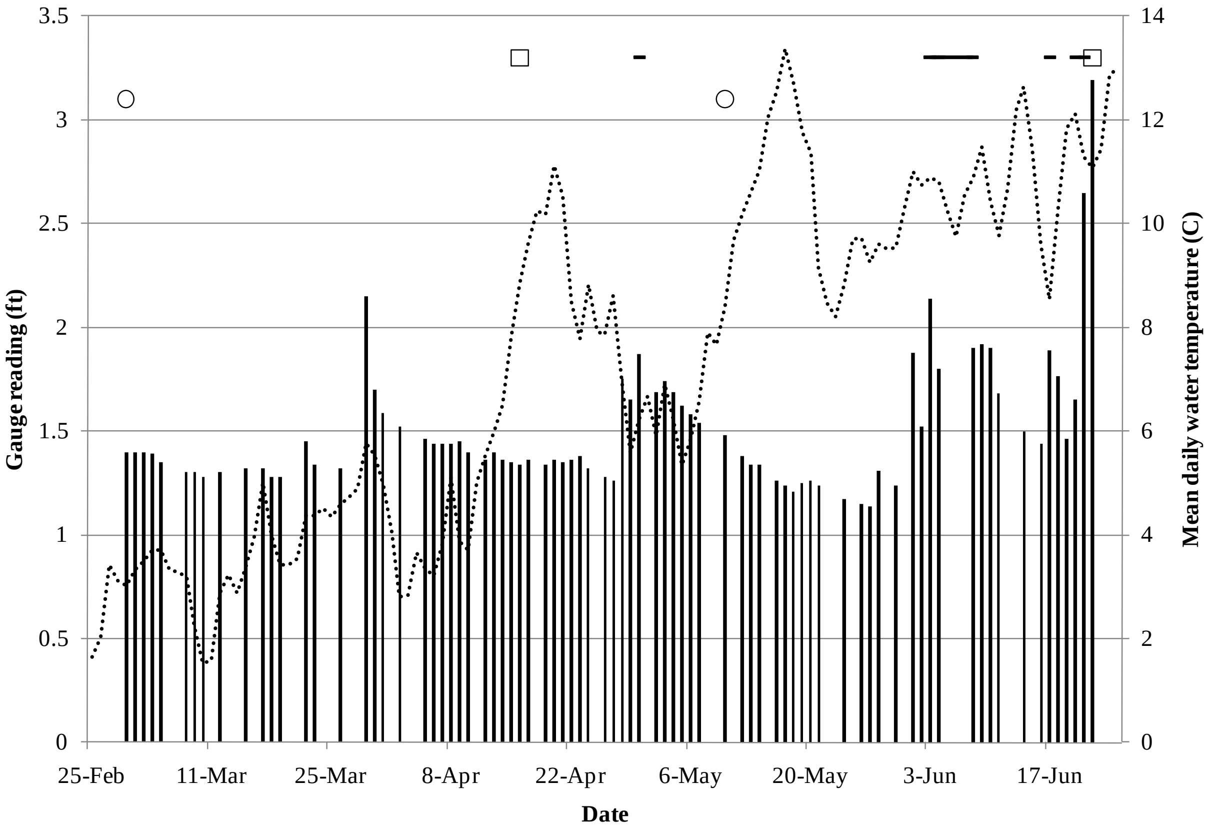

Figure 7. Gauge height readings (vertical bars) and mean daily water temperatures (dotted line) collected at the old H95 bridge during the 2010 migratory period in Lake Creek. Open circles and squares at the top represent installation and removal dates for the UP and DOWN traps, respectively Solid horizontal bars at the top, in line with their respective traps, indicate periods when traps were compromised.

Eighty-four of the 91 adults captured at the DOWN trap had a detectable opercle punch, yielding a spawner abundance estimate of 113 fish (95% confidence interval, 111 – 116). However, given the fact that a number of detected PIT-tagged fish had approached the UP trap but were not captured in 2010, the actual number of adults that ascended upriver *to* the UP trap was likely greater than that that ascended *beyond* the UP trap. Using the total number of antennae-interrogated fish as marks and the number of PIT-tagged fish captured in the UP trap as recaptures in a mark-recapture model, an estimate of 162 ascending adults was obtained (95% confidence interval, 126 – 198). In addition to 84 of the 104 opercle-punched PIT-tagged adults that were recaptured in the DOWN trap, two more of the 104 were interrogated by the antennae moving back downriver, yielding a minimum estimate of spawning ground survival of 82.7%.

Of the 67 adult fish captured in the DOWN trap that had been tagged this year at the UP trap, 20 did not scan which yielded an estimated percent tag loss of 29.9% for these fish. Notably, all 20 fish were females. Of the 10 adults that were adipose-clipped, marked at the UP trap, and recaptured in the DOWN trap, two did not scan indicating tag loss of 20% for this group of fish. The two fish that had shed their tags were females; all males were found to retain their tags. All seven of the fish that had been tagged as adults in 2009, marked at the UP trap, and recaptured in the DOWN trap, were females and were found to retain their tags.

Table 2. Length, weight, and condition factor means and standard deviations (SD) for adult adfluvial cutthroat trout with sex determined that were captured during their upriver and downriver migrations in Benewah and Lake creeks in 2010.

		Total length (mm)			Weight (g)		Condition Factor	
Gender	**N**	**Range**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
				Benewah Creek upriver				
Female	46	312 - 460	377.2	41.0	519.6	183.0	0.93	0.08
Male	25	287 - 455	374.1	39.9	466.4	138.7	0.87	0.08
				Benewah Creek downriver [a]				
Female	35	303 - 458	372.1	39.3	428.2	135.7	0.81	0.07
Male	17	333 - 455	376.6	29.5	462.3	87.9	0.86	0.08
				Lake Creek upriver [b]				
Female	79	275 - 416	357.8	32.7	422.3	108.6	0.90	0.05
Male	24	268 - 435	370.0	42.8	472.7	130.1	0.88	0.06
				Lake Creek downriver [c]				
Female	67	290 - 418	358.7	31.6	387.7	104.9	0.82	0.07
Male	22	272 - 426	375.0	39.3	455.0	118.5	0.84	0.06

[a] Also captured were two adults of undetermined sex

[b] Also captured were three adults of undetermined sex

[c] Also captured were two adults of undetermined sex

Over the time period from 2005 to 2007, 2272 juvenile cutthroat trout have been PIT-tagged during spring outmigration periods and were deemed alive upon release. Of these fish, only 38 (1.7%) have been uniquely detected either by the fixed antennae array or in the traps over the years 2006-2010, and deemed to be returning adults. Detected fish generally were larger and tagged earlier as juveniles than those PIT-tagged fish that have not been detected. For example, only approximately 17% of juvenile cutthroat trout had exceeded 160 mm in length at time of tagging. However, of those fish that have been detected, 82% were at least 160 mm when tagged (Figure 9). In a similar manner but not as dramatic, approximately 50% of all PIT-tagged fish had been tagged prior to May 4 (julian date of 124), whereas 80% of those that have been detected had been tagged before this date (Figure 10).

Eighteen of the 104 adults that were PIT-tagged in 2009 were either interrogated by the antennae-array or detected in traps in 2010. Given the tag retention estimate of 88.3% that was generated for this group of tagged adults in 2009 (Firehammer et al. 2011), 92 of the 104 fish were estimated to be available to be detected in subsequent years. As a result, twenty percent of the 2009 spawners (i.e., 18 of 92) were estimated to return as repeat consecutive year spawners in 2010. Of those 18 detected repeat spawners in which sex was able to be assigned, 15 of 17 (88%) were classified as females.

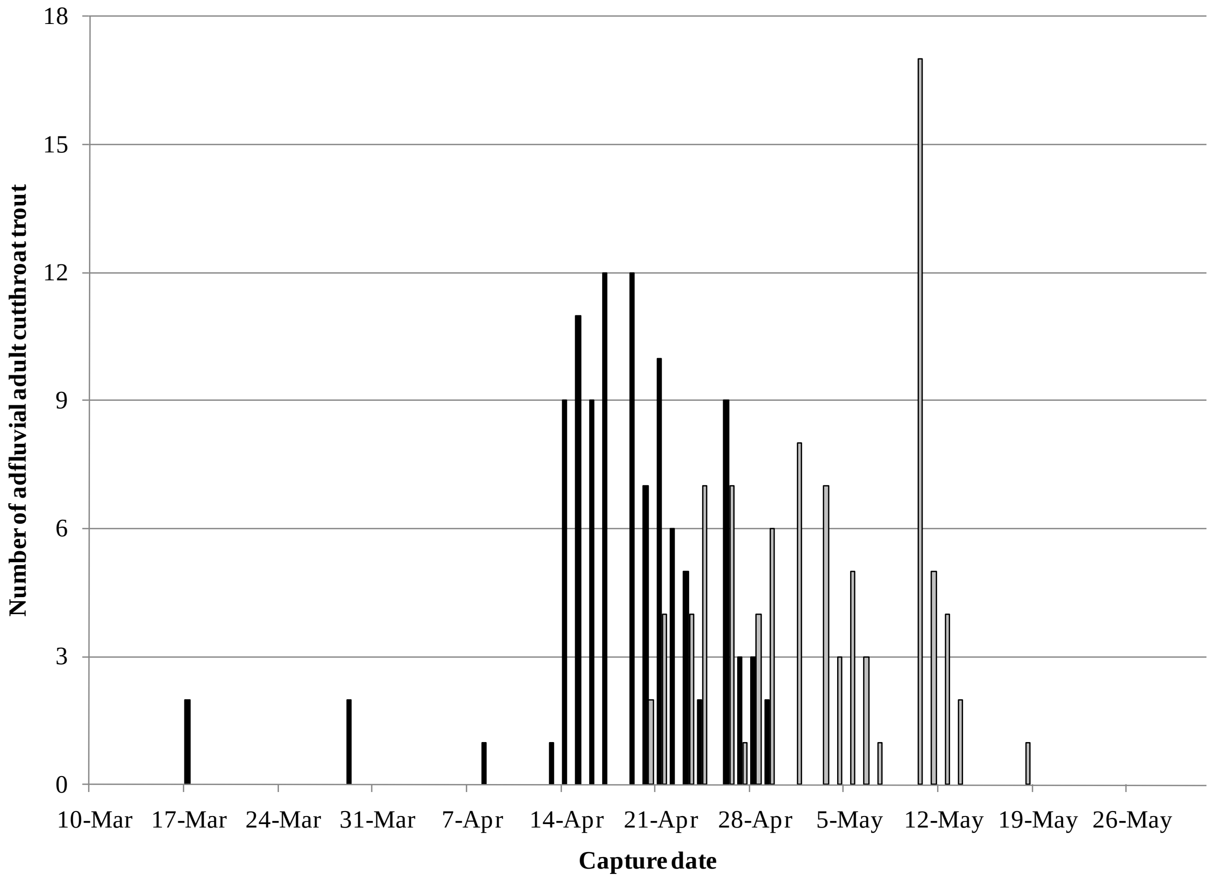

Figure 8. Timing of adult adfluvial cutthroat trout captured during their upriver (black bars) and downriver (gray bars) migrations in Lake Creek, 2010.

A total of 3501 juvenile adfluvial cutthroat trout was captured in the DOWN trap in Lake Creek in 2010. Juveniles were captured from April 20 through the month of May at variable rates, with approximately 90% of the fish captured before June and capture rates markedly decreasing thereafter (Figure 11). More than 100 juveniles were processed on 13 different days, with 5 of these capture events occurring from May 1 to May 6 during a 10 d period of high flows (Figure 7). In addition, more than 100 juveniles were captured soon after trap installation, which, in combination with a lack of a definable distribution of outmigration times, render it difficult to estimate the portion of the early part of the outmigration that was not sampled. Of the 3501 juveniles captured, 968 (28%) received PIT tags. Generally, fish were tagged representatively throughout their outmigration as supported by the similarity in the cumulative distribution curves for PIT-tagged juveniles and all captured juveniles (Figure 11). In addition, the length distribution of PIT-tagged adfluvial juveniles was similar to that for all juveniles captured in the DOWN trap (Table 4), with approximately 85% of both groups ranging between 121 and 180 mm. Seventy-nine other fish captured in the DOWN trap were classified as likely residents given their external markings. Mean total length of these fish was 199 mm. Thirty-one of the 79 purported resident cutthroat trout received PIT tags.

Table 3. Summary of detection data for juvenile (J) and adult (A) cutthroat trout tagged in previous years and either recaptured during upriver (UP) or downriver (DOWN) migrations or interrogated by the PIT-tag antennae array in Lake Creek in 2010. For those fish recaptured in a trap, 'Last day detected' and 'Days detected' indicates those values before trap capture.

Tagging information			2010 initial capture data				Length change (mm)	Juveniles detected from 2007-2009		PIT array detection data in 2010		
Year	Life stage	Total length (mm)	Trap	Date	Sex	Total length (mm)	since time of tagging	Years detected	Last year detected	First day detected	Last day detected	Days detected
2005	J	147	UP	8-Apr	F	396	249	2	2008	17-Mar	7-Apr	11
2005	J	146	UP	19-Apr	F	370	224	3	2009	15-Apr	18-Apr	4
2005	J	164	UP	26-Apr	M	428	264	1	2008	30-Mar	24-Apr	12
2006	J	177	UP	23-Apr	F	379	202	1	2008	6-Apr	21-Apr	13
2007	J	191	UP	29-Mar	F	387	196	.	.	28-Feb	27-Mar	17
2007	J	218	UP	21-Apr	F	366	148	.	.	10-Apr	20-Apr	11
2007	J	173	UP	22-Apr	F	366	193	1	2009	16-Apr	21-Apr	6
2007	J	186	UP	22-Apr	M	354	168	.	.	24-Mar	21-Apr	22
2007	J	171	UP	27-Apr	M	372	201	.	.	30-Mar	26-Apr	20
2007	J	173	UP	26-Apr	M	381	208	.	.	28-Mar	25-Apr	20
2008	J	143	.	.	.	.	.	.	.	16-Apr	28-Apr	13
2008	J	169	UP	17-Mar	F	341	172	.	.	2-Mar	16-Mar	10
2008	J	167	UP	24-Apr	F	318	151	.	.	4-Apr	23-Apr	15
2009	A	506	.	.	.	.	.	.	.	12-Apr	14-Apr	3
2009	A	378	.	.	.	.	.	.	.	29-Mar	2-May	26
2009	A	387	.	.	.	.	.	.	.	7-Apr	21-Apr	12
2009	A	384	.	.	.	.	.	.	.	12-Apr	12-Apr	1
2009	A	403	.	.	.	.	.	.	.	14-Apr	17-Apr	4
2009	A	357	.	.	.	.	.	.	.	8-Apr	20-Apr	11
2009	A	418	.	.	.	.	.	.	.	31-Mar	8-Apr	3
2009	A	367	.	.	.	.	.	.	.	31-Mar	20-Apr	11
2009	A	351	.	.	.	.	.	.	.	15-Apr	21-Apr	6
2009	A	399	.	.	.	.	.	.	.	31-Mar	6-May	30
2009	A	407	UP	15-Apr	.	407	0	.	.	1-Apr	14-Apr	11
2009	A	369	UP	28-Apr	F	371	2	.	.	26-Apr	27-Apr	2
2009	A	403	UP	15-Apr	F	397	-6	.	.	13-Apr	14-Apr	2
2009	A	388	UP	20-Apr	F	390	2	.	.	8-Apr	19-Apr	9
2009	A	390	UP	21-Apr	F	395	5	.	.	14-Apr	20-Apr	7
2009	A	355	UP	15-Apr	F	365	10	.	.	12-Apr	14-Apr	3
2009	A	364	UP	17-Apr	F	370	6	.	.	8-Apr	16-Apr	6
2009	A	363	UP	20-Apr	F	362	-1	.	.	15-Apr	19-Apr	5
2009	J	130	.	.	.	.	.	.	.	2-Apr	2-Apr	1
2009	J	144	.	.	.	.	.	.	.	8-Apr	8-Apr	1
2009	J	118	.	.	.	.	.	.	.	17-Apr	17-Apr	1
2009	J	160	.	.	.	.	.	.	.	20-Mar	20-Mar	1
2009	J	140	.	.	.	.	.	.	.	1-Apr	1-Apr	1
2009	J	137	DOWN	17-May	.	205	68	.	.	.	.	.
2009	J	136	DOWN	20-Apr	.	203	67	.	.	21-Mar	17-Apr	9
2009	J	134	DOWN	24-Apr [a]	.	.	.	.	.	.	.	.
2009	J	152	DOWN	20-Apr	.	185	33	.	.	.	.	.
2009	J	182	DOWN	14-Jun [a]	.	.	.	.	.	.	.	.
2009	J	114	DOWN	14-Jun [a]	.	.	.	.	.	17-Apr	17-Apr	1

[a] Length data not collected because fish was presumed to be a recapture from a 2010 release trial

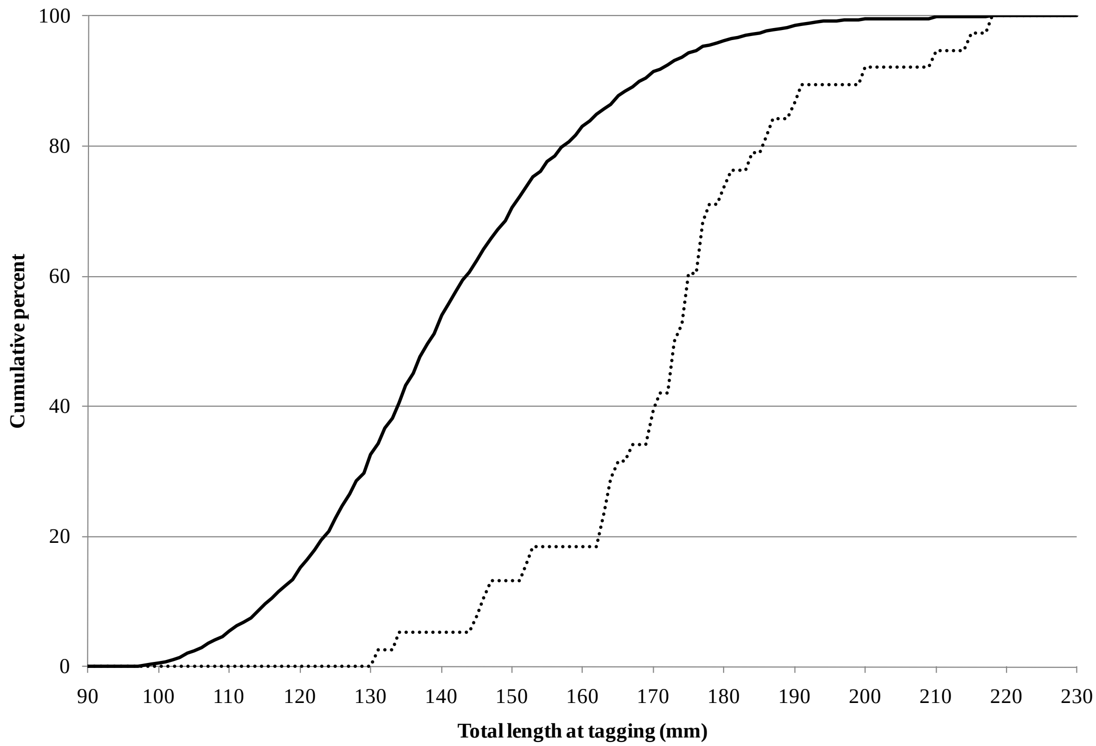

Figure 9. Cumulative distribution curves of length at tagging for all juvenile cutthroat trout tagged from 2005 to 2007 in Lake Creek (solid line) and for those fish from these cohorts uniquely detected as returning adults (dotted line).

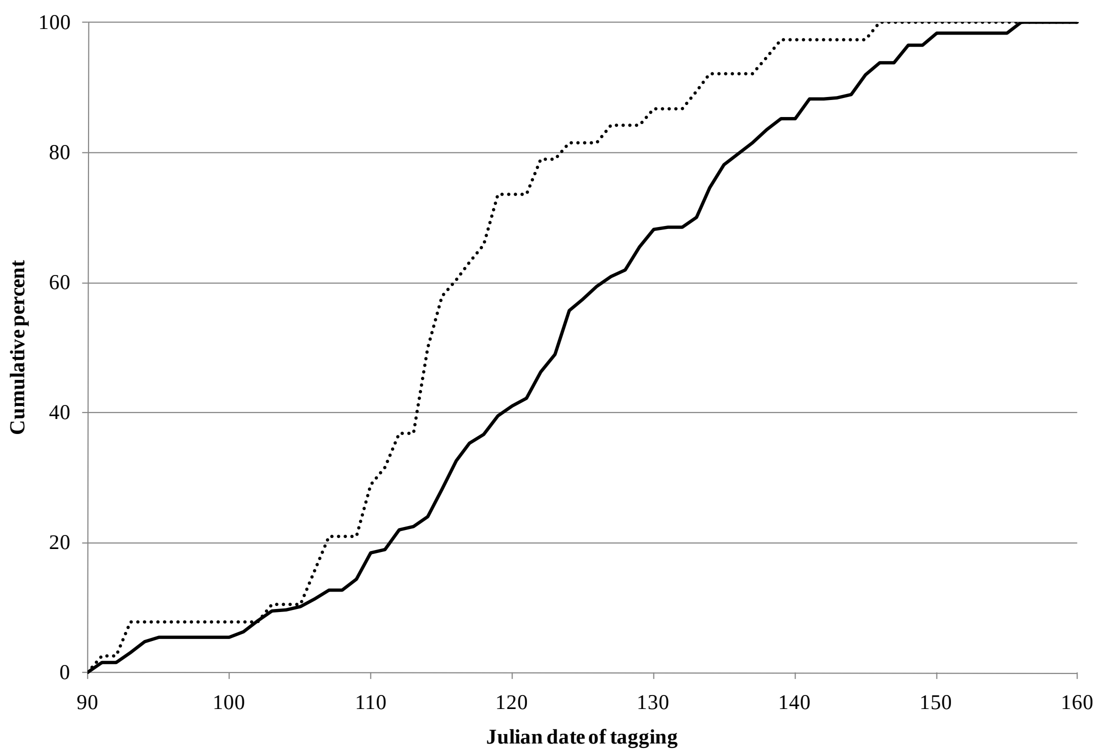

Figure 10. Cumulative distribution curves of date at tagging for all juvenile cutthroat trout tagged from 2005 to 2007 in Lake Creek (solid line) and for those fish from these cohorts uniquely detected as returning adults (dotted line).

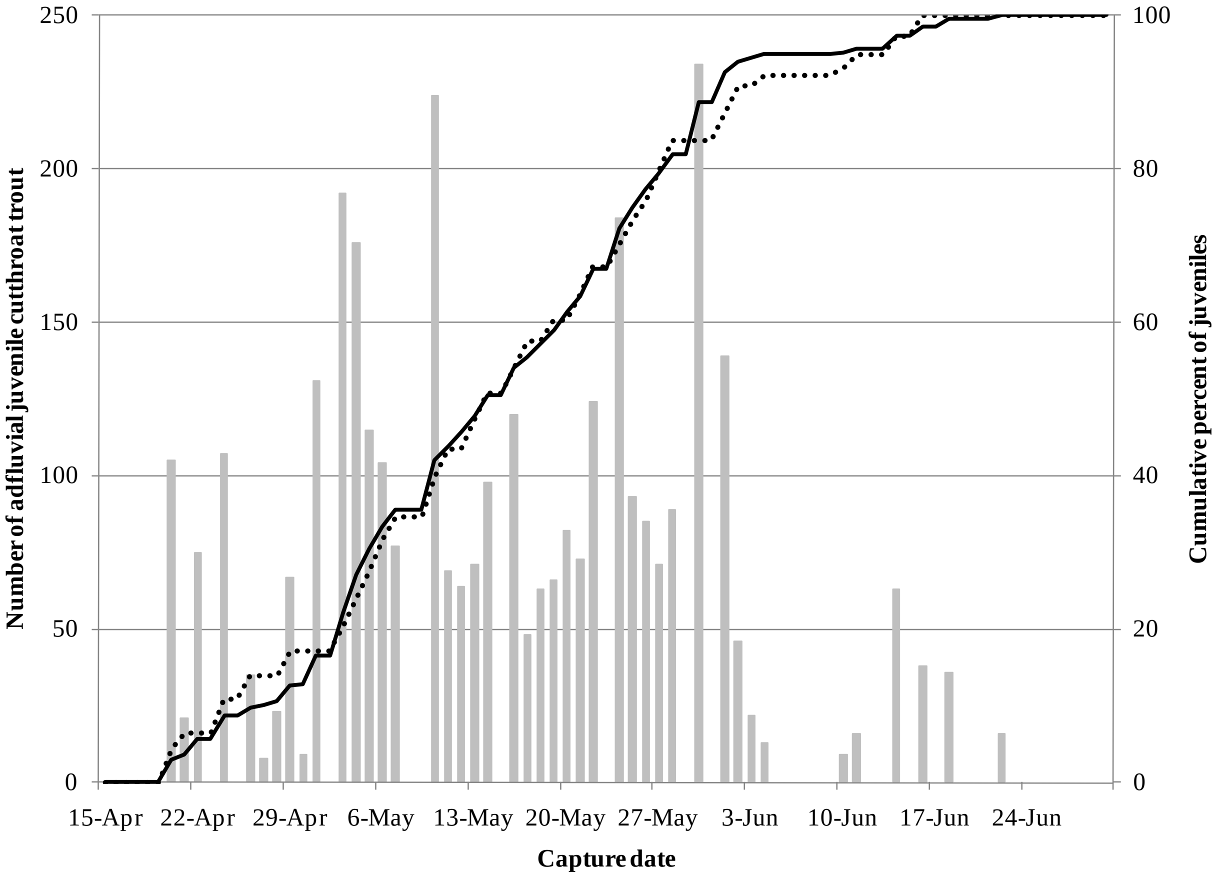

Figure 11. Timing of juvenile adfluvial cutthroat trout captured in the downriver trap during their outmigration in Lake Creek, 2010. Numbers of juveniles (gray bars) along with the cumulative distribution curves of all captured juveniles (solid line) and PIT-tagged juveniles (dotted line) are presented.

An overall juvenile outmigrant abundance estimate of 3858 ±117 was generated for Lake Creek in 2010 using the data from eleven release trials conducted from April 20 to June 20 (Table 5). The number of juveniles estimated to outmigrate between June 4 and 9, a time period when the trap was not fishing, was statistically interpolated based on daily capture rates before and after trap inoperability. In addition, 16 adfluvial juveniles captured after June 20 were not included in the outmigrant abundance estimate. Release trial periods typically lasted 4-7 d (mean, 5 d), with an average of approximately 52 tagged fish released in each trial. Mortality/retention trials were conducted in association with seven of the eleven release trials in which an average of 36 tagged fish were held overnight for each trial. All fish were found to retain their tags and survive the trial before their release. Estimated trap efficiencies were very high, exceeding 90% throughout most of the trial periods, until June when estimated efficiencies ranged from 66 to 81% (Table 5). Generally, other than the first couple of trial periods, released fish were recaptured in the trap during the trial period of their release, as evidenced by similar values for number of fish released and number of fish available for recapture (which is discounted by those captured in subsequent trial periods).

Table 4. Number and relative percent of adfluvial juvenile cutthroat trout captured and PIT-tagged of different length groups in Lake and Benewah creeks, 2010.

	Lake Creek				**Benewah Creek**			
Length	**All fish captured**		**Tagged fish**		**All fish captured**		**Tagged fish**	
group (mm)	**Number**	**Percent**	**Number**	**Percent**	**Number**	**Percent**	**Number**	**Percent**
81-100	7	0.2	3	0.3	3	1.0	2	1.1
101-120	219	6.3	43	4.4	57	19.5	37	19.9
121-140	1237	35.3	320	33.1	130	44.4	86	46.2
141-160	1258	35.9	341	35.2	83	28.3	49	26.3
161-180	472	13.5	141	14.6	17	5.8	10	5.4
181-200	211	6.0	82	8.5	2	0.7	2	1.1
>200	97	2.8	38	3.9	1	0.3	0	0.0

Table 5. Trial period abundance estimates and overall abundance estimates with associated 95% confidence intervals for adfluvial juvenile cutthroat trout outmigrating in Benewah and Lake creeks, 2010. For both systems, the number of tagged fish available for recapture within each trial period was discounted by those captured during subsequent trial periods..

Trial period	**Inclusive dates**	**Total fish captured**	**Tagged fish released**	**Tagged fish available for recapture**	**Tagged fish recaptured**	**Trap efficiency estimate**	**Abundance estimate**
			Benewah Creek				
1	Apr-27 - Apr-28	56	11	11	4	0.41	134
2	May-11 - May-19	64	9	8	6	0.78	82
3	May-19 - May-25	41	47	46	40	0.87	47
4	May-25 - Jun-01	91	25	25	22	0.88	103
5	Jun-01 - Jun-03	27	32	32	31	0.97	28
Overall							**394 ± 92**
			Lake Creek				
1	Apr-20 - Apr-26	343	63	57	55	0.97	355
2	Apr-26 - Apr-30	107	71	66	60	0.91	118
3	Apr-30 - May-04	499	32	31	28	0.91	551
4	May-04 - May-11	589	64	64	63	0.98	598
5	May-11 - May-17	401	85	81	74	0.91	438
6	May-17 - May-22	408	36	35	32	0.92	445
7	May-22 - May-28	522	67	67	62	0.93	563
8	May-28 - Jun-03	441	80	80	77	0.96	458
9	Jun-03 - Jun-04	13	35	35	23	0.66	20
10	Jun-09 - Jun-16	126	9	9	7	0.80	158
11	Jun-16 - Jun-20	36	27	26	21	0.81	44
Overall							**3858 ± 117** [a]

[a] Overall estimate includes number of fish statistically interpolated during June 4-9, but doesn't include 16 fish captured after June 20

The average size of captured adfluvial juveniles varied throughout the outmigration period, with total lengths of fish greater in April than in May and June (Figure 12). Seven day moving averages decreased from 174 to 152 mm in April, and averaged 145 mm thereafter. In total, after adjusting for trial-specific trap efficiencies, the estimated mean total length of the outmigrant juvenile cohort in Lake Creek was 148 mm in 2010 (Figure 13). Generally, the size of outmigrating juveniles was larger in 2010 than in 2009 and 2007 (Figure 14), two years in which DOWN traps also performed reasonably well to permit estimates of size distributions of outmigrant cohorts. For example, approximately 40% of outmigrating juveniles were greater than 145 mm in 2010, compared to only 26-28% greater than this size in 2007 and 2009.

Eleven fish that had been tagged as juveniles in 2009 were either detected in the DOWN trap or interrogated by the antennae array in Lake Creek in 2010 (Table 3). The five fish that were interrogated by the array were each detected on only one day, with most of the fish detected before installation of the DOWN trap. Of the six that were captured, data were not collected on three because they were inadvertently mistaken for tagged fish from 2010 trap efficiency trials, and consequently were likely of similar size to outmigrating juveniles. The other three ranged in size from 185 to 205 mm, but based on their external markings were not classified as resident fish. Seven of the eleven fish were tagged at lengths less than 141 mm, the estimated mean total length of outmigrating adfluvial juveniles in 2009, and 8 of the 11 were tagged in late May and early June (Firehammer et al. 2011).

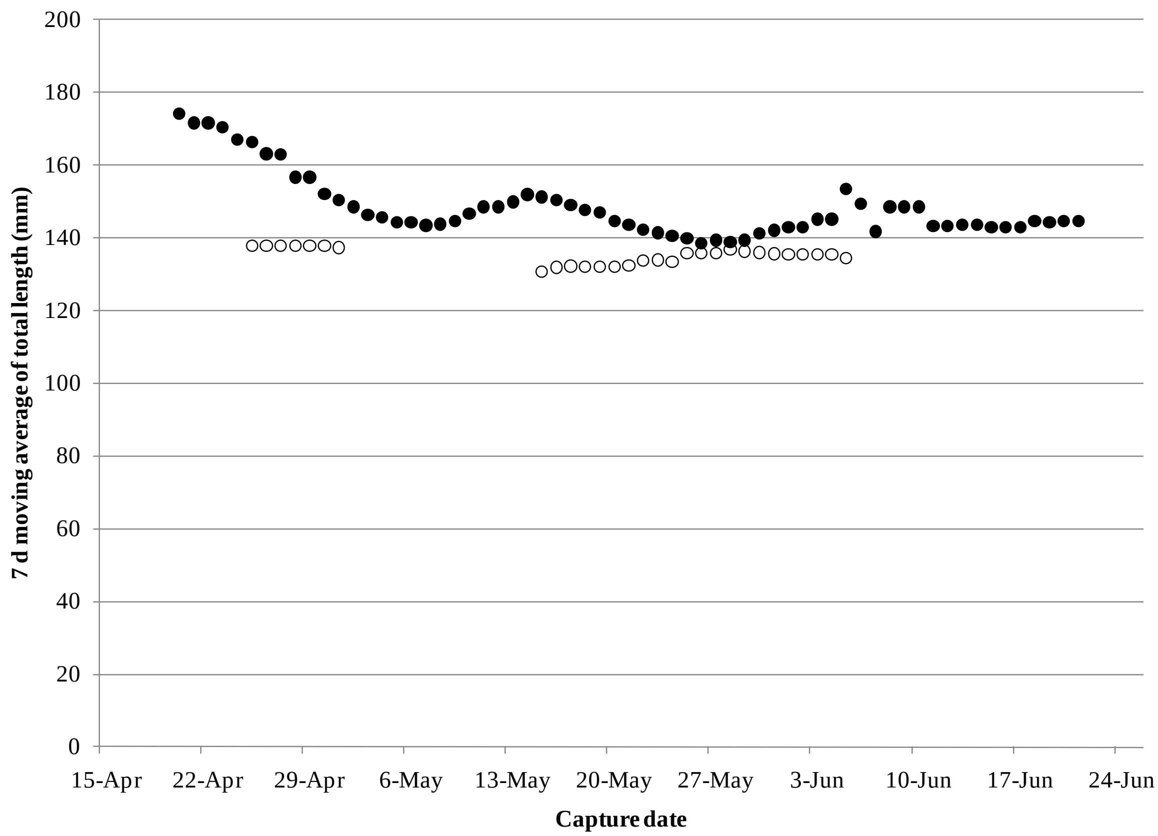

Figure 12. Seven-day moving averages of total length (mm) for adfluvial juvenile cutthroat trout captured in DOWN traps in Lake (filled circles) and Benewah (open circles) creeks in 2010. For each day, a mean was calculated only if more than 20 fish were captured over the period that encompassed the 3 days before and after the given day.

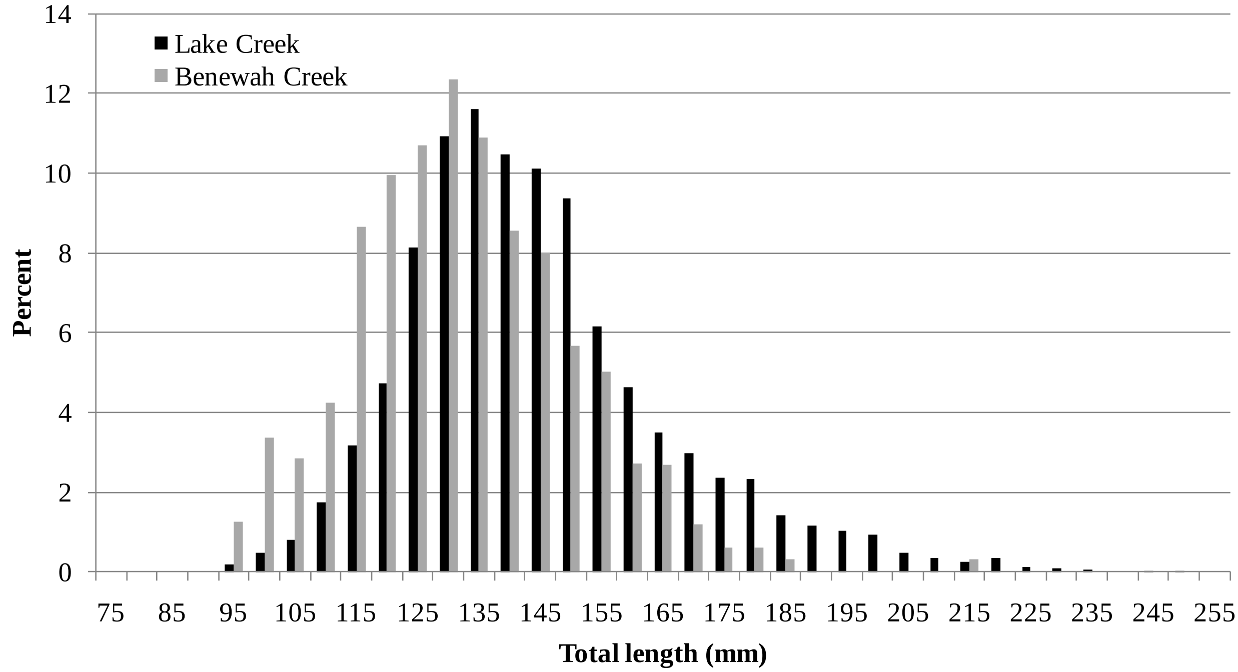

Figure 13. Relative length distribution of outmigrating adfluvial juvenile cutthroat trout in Lake and Benewah creeks, 2010. Numbers of fish captured were adjusted by trial period specific estimated capture efficiencies.

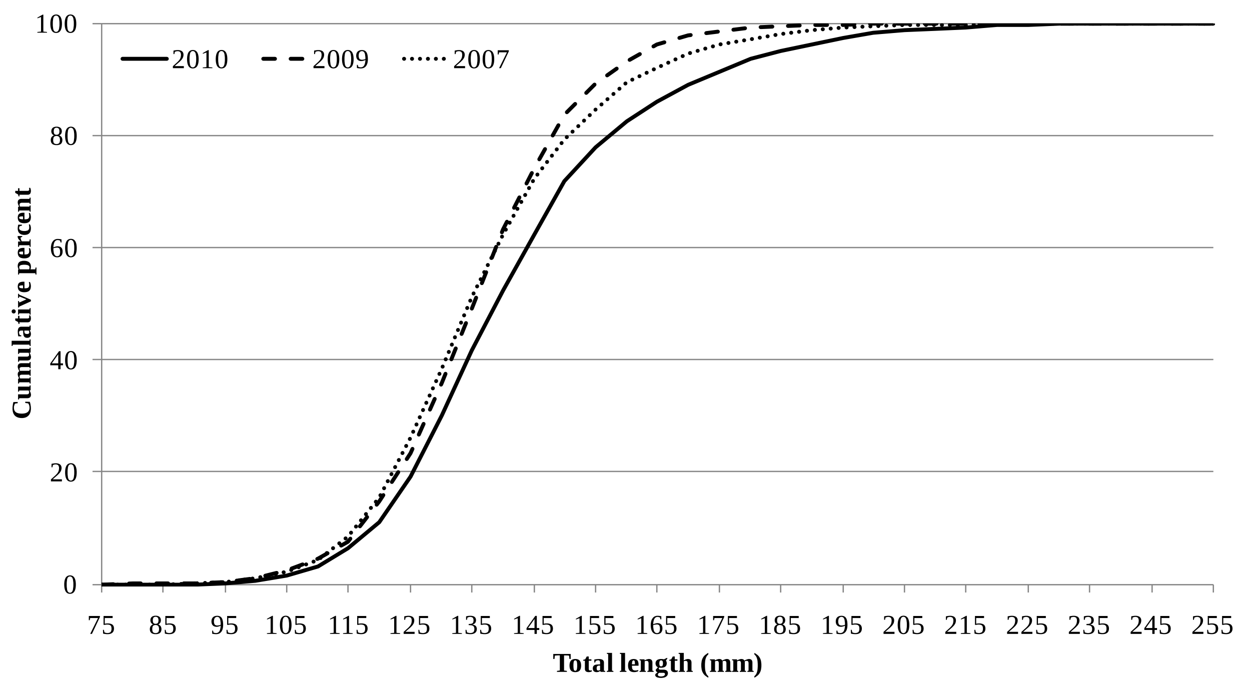

Figure 14. Cumulative total length distributions estimated for adfluvial juvenile outmigrant cohorts in Lake Creek in 2007, 2009, and 2010.

3.3.1.2 Benewah Creek adfluvial cutthroat trout migration

The UP trap was installed at 9-mile bridge in the mainstem of Benewah Creek on March 10 and was no longer monitored after July 6, because of the absence of fish in the trap's live box. During this time period, the trap was checked a total of 61 d, and was considered fishing 95% of the time that it was monitored. On only one occasion during a high flow period from April 29 to May 3 was the trap considered compromised (Figure 15). The DOWN trap was installed on April 22 and was removed on July 6, yielding an operable period of 76 d. During this time, the trap was checked a total of 41 d (54% of the time), and over the period in which it was monitored was considered effectively fishing approximately 79% of the time. The DOWN trap was compromised during several high flow events in which water was found flowing over the trap panels or induced scouring underneath the trap panels. These events rendered trap inoperability from April 28 to May 11, June 3-7, and June 22-27 (Figure 15).

A total of 71 adfluvial adult cutthroat trout was captured in the UP trap (Table 2). Forty-six of these were identified as females (65%) with a mean length and weight of 377 mm and 520 g, respectively. Twenty-five of the fish were identified as males with a mean length and weight of 374 mm and 466 g, respectively. Sixty-two of the 71 adults (87%) were captured from April 5 to April 21 when discharge was relatively stable but mean daily water temperatures were increasing from 3 to 8.4^{o}C (Figure 15; Figure 16). All 71 of the captured adults received opercle punches with 66 of these fish receiving PIT-tags. Two other cutthroat trout captured by the UP trap were classified as residents based on their external markings and size. These two fish ranged between 232 and 265 in total length.

Five of the cutthroat trout captured in the UP trap had been tagged as juveniles in 2008 (Table 6). Whereas three of the fish were captured rather quickly (≤ 3 d) after their initial interrogation by the antennae array, the other two fish were repeatedly detected 7 to 8 d over an 18 d period before they were captured. Four of the five fish were first detected as adults this year, were tagged at lengths ranging from 158 to 190 mm, and displayed two-year growth increments that ranged from 188 to 275 mm. The other of the five fish had been recaptured in 2009 in the DOWN trap at 229 mm and, at that time, was classified as a resident fish based on its external markings. This fish had only increased 147 mm in length since 2008, when it was tagged at 140 mm. Incidentally, all juvenile-tagged fish that were recaptured in the UP trap scanned in 2010. One additional fish tagged as a small 120 mm juvenile in 2008 was interrogated by the antennae array but not captured in traps in 2010 (Table 6). At this time, it is difficult to evaluate whether this fish was an adfluvial adult or a resident given that the fish was recaptured in 2009 in the DOWN trap, and at that time was apparently of such a size that it was mistaken for a juvenile release-trial fish in that year.

Fifty-four adult adfluvial cutthroat trout were captured in the DOWN trap in 2010 (Table 2). Of these 54, 35 were females (65%) with a mean length of 372 mm and a mean weight of 428 g, and 17 were males with a mean length of 377 mm and a mean weight of 462 g. The mean condition factor was appreciably lower for females captured in the DOWN trap (0.81) than for those captured in the UP trap (0.93), indicating that many of the outmigrating females likely spawned (Table 2). Thirty-five of the outmigrating adults (65%) were captured during April 26-28, soon after the DOWN trap had been installed (Figure 16). However, trap performance was seriously compromised from April 28 to May 11, and consequently, the distribution of capture rates during the spring does not likely reflect the distribution of outmigration times for post-spawn adults in 2010. Forty-five of the PIT-tagged adults that had either been tagged or detected at the UP trap were recaptured in the DOWN trap. The mean elapsed period between detections for these fish was greater for males, 23.3 d, than for females, 15.2 d.

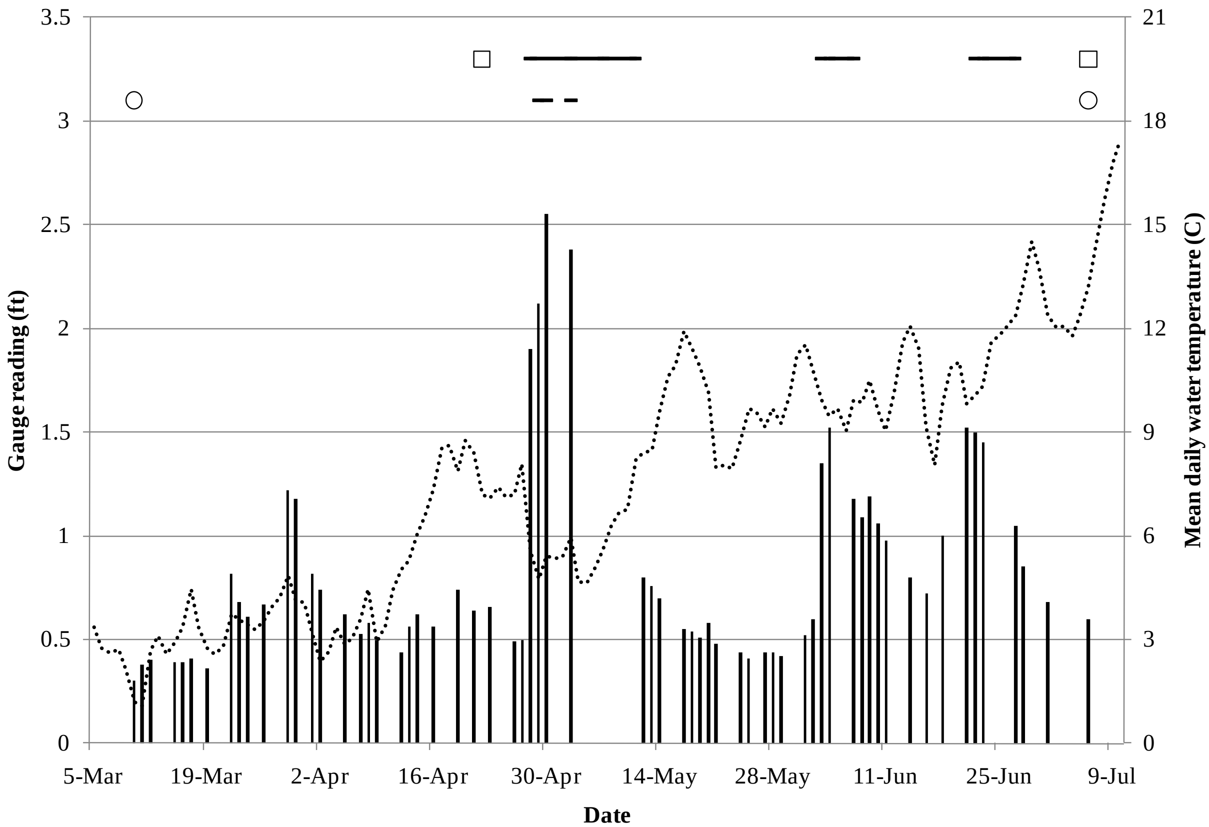

Figure 15. Gauge height readings (vertical bars) and mean daily water temperatures (dotted line) collected at 9-mile bridge during the 2010 migratory period in Benewah Creek. Open circles and squares at the top represent installation and removal dates for the UP and DOWN traps, respectively (Note that the UP was not removed but left intact during the summer to prevent ascension by large mature brook trout). Solid horizontal bars at the top, in line with their respective traps, indicate periods when traps were compromised.

Fifty-three of the 54 adults captured at the DOWN trap had a detectable opercle punch, yielding a precise spawner abundance estimate of 72 fish (95% confidence interval, 72 – 74). In addition to 53 of the 71 opercle-punched PIT-tagged adults that were recaptured in the DOWN trap, 10 more fish that were not recaptured were interrogated by the antennae array moving back downriver, yielding a minimum estimate of spawning ground survival of 88.7%. Of the 50 adults captured in the DOWN trap that had been tagged this year at the UP trap, 8 did not scan which yielded an estimated percent tag loss of 16% for these fish. All three of the adipose-clipped fish that were marked at the UP trap and recaptured in the DOWN trap scanned.

Eight of the adult fish that had been captured in traps in 2010 were identified as potential hybrids based on external features (e.g., faint slash on underside of jaw). All eight were classified as females, with four of the fish captured in both traps. Though the four that had been captured in the UP trap that were not later recaptured were not able to be re-evaluated for hybrid status, the four that were putative hybrids that were recaptured in the DOWN trap were not classified as hybrids when they were captured earlier in the UP trap. Notably, two of the eight were adipose-clipped, and had also been classified as suspect hybrids when they were tagged as large juveniles (185-190 mm) in 2008. These two fish also exhibited the largest two-year growth increments of 235 and 275 mm for those fish tagged in 2008 that were recaptured this year (Table 6).

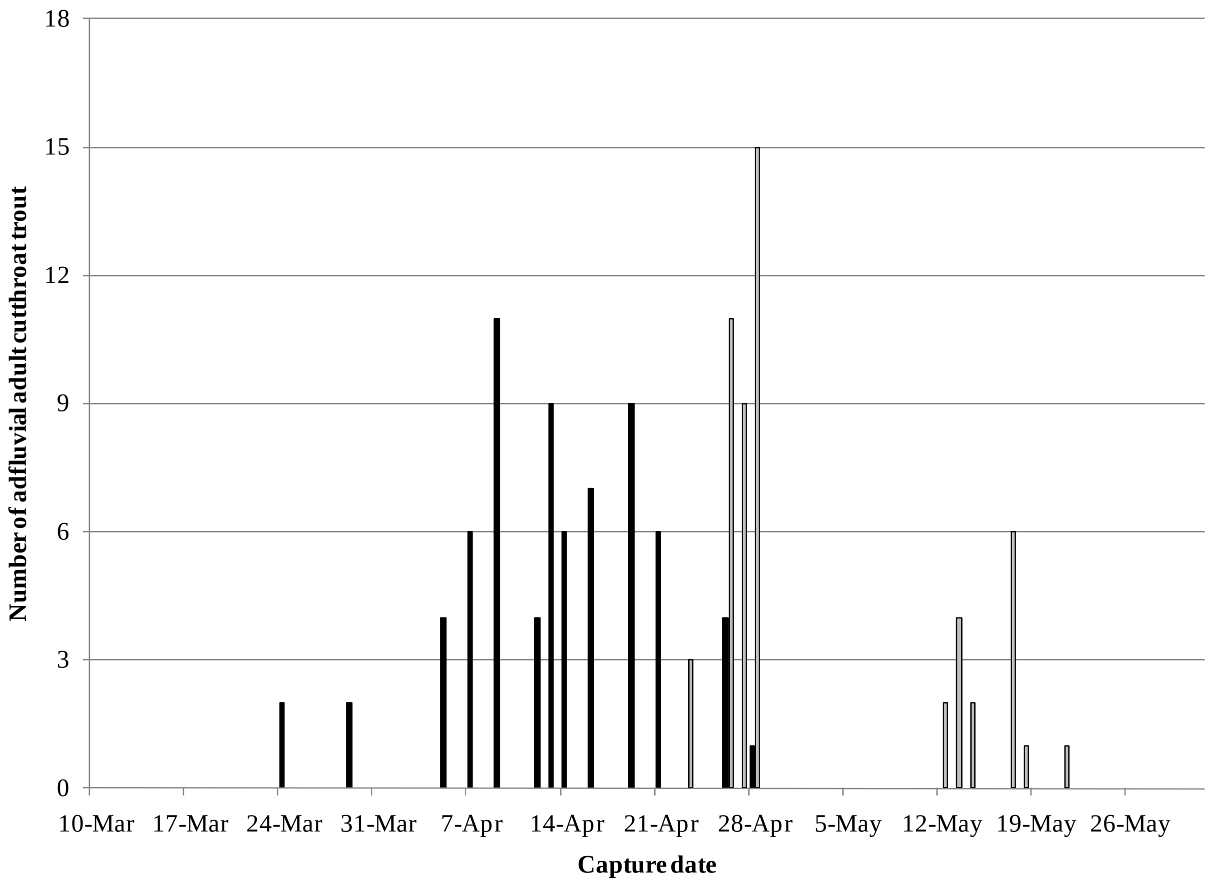

Figure 16. Timing of adult adfluvial cutthroat trout captured during their upriver (black bars) and downriver (gray bars) migrations in Benewah Creek, 2010.

A total of 293 juvenile adfluvial cutthroat trout was captured in the DOWN trap in Benewah Creek, with approximately 95% of the fish captured from April 27 to June 3 in 2010. More than 50 fish (19% of the total) were captured during April 27-28 (Figure 17), less than a week after trap installation and before the extended period of trap inoperability that was caused by a high flow event. The high capture rate soon after trap deployment suggests that a portion of the early part of the juvenile outmigration was not sampled. Of the 293 juveniles captured, 186 (63%) received PIT tags. Generally, fish were tagged representatively throughout the period in which they were captured as supported by the similarity in the cumulative distribution curves for PIT-tagged juveniles and all captured juveniles (Figure 17). In addition, the length distribution of PIT-tagged adfluvial juveniles was similar to that for all juveniles captured in the DOWN trap (Table 4), with approximately 92% of both groups ranging between 101 and 160 mm. Seven other cutthroat trout captured in the DOWN trap were classified as likely residents given their external markings, with 6 of the 7 receiving PIT-tags. Two of the seven with lengths of 168 and 169 mm were mature males and expressing milt.

Table 6. Detection data for juvenile cutthroat trout tagged in previous years and either recaptured in traps during upriver (UP) or downriver (DOWN) migrations or interrogated by the PIT-tag array in Benewah Creek in 2010. For those fish recaptured in a trap, 'Last day detected' and 'Days detected' indicate those values before trap capture.

Tagging information			2010 initial capture data				Length change (mm) since time of tagging	PIT array detection data in 2010			Footnoted detections in 2009
Year	Life stage	Total length (mm)	Trap	Date	Sex	Total length (mm)		First day detected	Last day detected	Days detected	
2008	J	120	.	.	.	.	.	27-Mar	4-Apr	5	a
2008	J	158	UP	28-Apr	F	346	188	27-Apr	27-Apr	1	.
2008	J	185	UP	14-Apr	F	460	275	26-Mar	13-Apr	7	.
2008	J	190	UP	7-Apr	F	425	235	4-Apr	4-Apr	1	.
2008	J	178	UP	14-Apr	M	393	215	26-Mar	13-Apr	8	.
2008	J	140	UP	29-Mar	M	287	147	26-Mar	27-Mar	2	b
2009	J	140	.	.	.	.	.	21-Apr	22-Jun	4	.
2009	J	133	.	.	.	.	.	13-Apr	13-Apr	1	.
2009	J	165	.	.	.	.	.	28-Apr	28-Apr	1	.
2009	J	150	.	.	.	.	.	18-Apr	18-Apr	1	.
2009	J	116	.	.	.	.	.	21-Mar	21-Mar	1	.
2009	J	107	.	.	.	.	.	29-Mar	29-Mar	1	.
2009	J	124	.	.	.	.	.	30-Mar	5-Apr	2	.
2009	J	154	DOWN	19-May	.	220	66	.	.	.	.

[a] Fish was captured in the downriver trap but because of its size was presumed a 2009 release trial fish
[b] Fish was captured in the downriver trap at 229 mm total length

An overall juvenile outmigrant abundance estimate of 394 ±92 fish was generated for Benewah Creek in 2010 using the data from five release trials conducted from April 27 to June 3, a value considerably less than that generated for the Lake Creek watershed (Table 5). This estimate excluded the 13 d period from April 29 to May 11 in which the trap was not fishing, and omitted the time period after June 3 in which only 14 juvenile fish (5% of the total) were captured. Release trial periods were variable ranging from 1 to 8 d in length; foreshortened trial periods were largely the result of trap failures from high discharge events. In addition, numbers of fish released during each period ranged widely from 9 to 47, with the release trial size predominantly influenced by capture rates and consequently the number of fish available to be tagged. Mortality/retention trials were conducted in association with three of the five release trials. All 57 fish that were held overnight retained their tags and survived the trial before their release. Other than the first trial period, estimated trap efficiencies were high, ranging from 78 to 97% (Table 5). Generally, released fish were recaptured in the trap during the trial period of their release, as evidenced by similar values for number of fish released and number of fish available for recapture (which is discounted by those captured in subsequent trial periods).

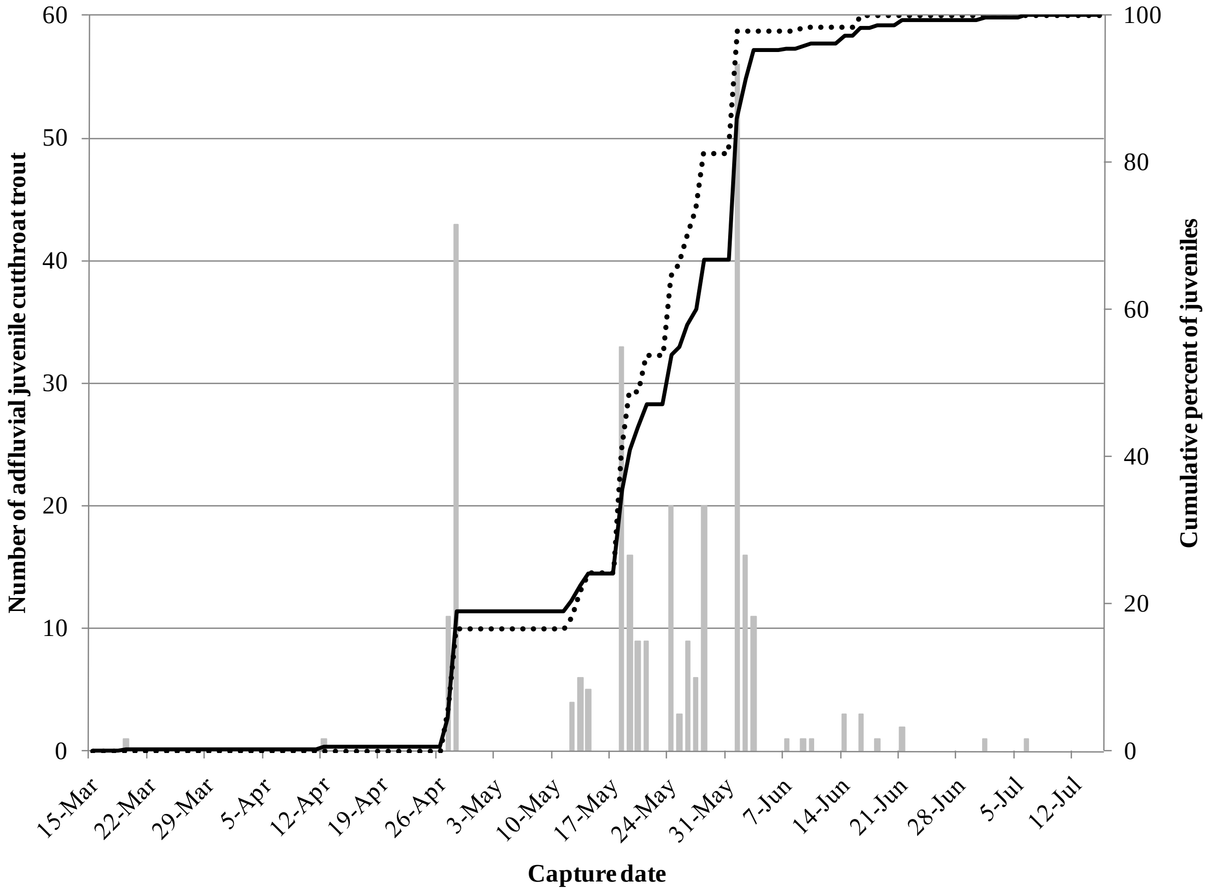

Figure 17. Timing of juvenile adfluvial cutthroat trout captured in the downriver trap during their outmigration in Benewah Creek, 2010. Numbers of juveniles (gray bars) along with the cumulative distribution curves of all captured juveniles (solid line) and PIT-tagged juveniles (dotted line) are presented.

The average size of captured adfluvial juveniles did not vary much throughout the sampled outmigration period in Benewah Creek, with seven-day moving averages ranging from 131 to 138 mm, values which approached those calculated for the latter part of the Lake Creek juvenile outmigration (Figure 12). Notably, 15 juvenile fish in Benewah Creek had exterior markings (e.g., faint red slash, dense spotting pattern on anterior portion of flank) that resembled those of a cutthroat trout hybridized with a rainbow trout; mean total length of these 15 fish was 150 mm. In total, after adjusting for trial-specific trap efficiencies, the estimated mean total length of the outmigrant juvenile cohort in Benewah Creek was 135 mm in 2010 (Figure 13). Overall, captured juvenile cutthroat trout were generally smaller in length in Benewah Creek than in Lake Creek in 2010 (Figure 12, Figure 13). For example, the percentage of outmigrating juveniles greater than 150 mm in total length in Lake Creek (38%) was twice that in Benewah Creek (19%).

Eight cutthroat trout that had been tagged as juveniles in 2009 were detected in 2010. Seven of these eight were only interrogated by the antennae array, with six of these fish briefly detected (i.e., ≤ 2 d) over abbreviated time periods ranging from late March to late April (Table 6). Mean length at tagging for these seven fish was 133.6 mm (range, 107 – 165). One of the eight was recaptured in the DOWN trap on May 19 and, though 220 mm in length, was not classified as a resident fish (Table 6). In addition, seven of these eight fish were tagged in late May and early June.

Other salmonids were also captured in migrant traps in Benewah Creek in 2010. Six brook trout were captured in the UP trap, with a mean length of 218 mm (range, 191-233 mm) computed for these fish. In addition, sixteen brook trout were intercepted by the DOWN trap, with a calculated mean length of 164 mm (range, 112-211 mm) for these fish. Four fish with markings that resembled rainbow trout were captured in the DOWN trap, with total lengths ranging between 306 and 334 mm (mean, 324 mm).

3.3.1.3 Salmonid stream surveys

Twenty five, thirty-two, twenty-three, and twenty index sites were sampled in 2010 using single pass electrofishing methodology in Alder, Benewah, Lake, and Evans creek watersheds, respectively. Five of the 20 index sites in Evans Creek were omitted from analysis because of technical deficiencies associated with sampling. In addition, four sites in the upper Benewah mainstem were sampled using trap nets in 2010. Cutthroat trout were found in all four watersheds, and brook trout were captured only in Alder and Benewah creeks.

In Alder Creek, the distribution of cutthroat trout was generally constrained to lower main-stem reaches with low overall densities throughout the watershed (Table 7), a result consistent with that documented in previous annual surveys. At the lowermost eight index sites within the watershed, the mean index density of age 1+ cutthroat trout was 7.9 fish/100 m, with fish detected at all but one of the sites. In comparison, age 1+ cutthroat trout were only captured at 5 of the 17 sites in upper reaches of the watershed (upstream of Alder 8), with index densities of less than 5 fish/100 m at these five sites. Age-0 cutthroat were not captured at any of the index sites in 2010 (Table 7).

Brook trout in the Alder Creek watershed displayed distribution patterns that were converse of those exhibited by cutthroat trout, and generally were much more abundant (Table 7). Though modest numbers of brook trout were captured at a couple of the lowermost eight index sites, first-pass indices of brook trout abundance were much greater in upper reaches of the watershed. The mean index density of age 1+ brook trout at the 17 sites in the upper watershed was 63 fish/100 m, with four of the sites yielding estimates greater than 100 fish/100 m (Table 7). In addition, age-0 fish were often abundant at those 17 sites, comprising a mean of 28% of the brook trout captured and yielding a mean density index of 23.3 fish/100 m.

In the Benewah watershed, results from the 2010 survey were consistent with those reported in previous years, with substantially greater numbers of cutthroat trout found in tributary than in main-stem reaches (Table 8). Mean density indices of 46.0 and 30.5 fish/100 m were calculated respectively for age 1+ fish at two sites in Bull Creek and at five sites in upper reaches of Whitetail, Windfall, West Fork, and South Fork creeks. Density indices of age 1+ fish at lowermost sites within each of the latter four tributaries, however, were two to four times lower than those respectively recorded in upper reaches. Relative numbers of age-0 cutthroat trout captured at tributary index sites varied greatly in 2010, comprising 35-83% of captured fish (mean, 57%) at 6 of the 15 sites, though not captured at six other sites (Table 8). Age-0 densities were greatest at the Bull Creek sites (21.3 and 57.4 fish/100 m) and at the lowermost Coon Creek site (32.8 fish/100 m). Compared to tributary reaches, density indices of age 1+ cutthroat trout were relatively moderate in lower main-stem reaches of the Benewah watershed, averaging 17.1

fish/100 m at sites 1-7 (range, 4.9-44.3 fish/100 m), and generally low in main-stem reaches further upstream, averaging only 3.6 fish/100 m (Table 8).

Brook trout distribution patterns in the Benewah watershed were converse of those displayed by cutthroat trout in 2010. Age 1+ brook trout were relatively lacking and outnumbered by cutthroat trout in upper reaches of West Fork, Whitetail, Windfall, Schoolhouse, and South Fork tributaries, with a mean density index of only 3.0 fish/100 m calculated across sites (Table 8). In comparison, density indices of age 1+ brook trout were generally greater than cutthroat trout in downstream tributary reaches, with a mean value of 22.6 fish/100 m (range, 13.1 – 36.1 fish/100 m) calculated across lowermost index sites of the latter four tributaries. Moreover, in upper main-stem reaches where age 1+ cutthroat were comparatively scarce, age 1+ brook trout were captured at moderate numbers, with a mean density index of 12.2 fish/100 m calculated across sites 14L to 18 (Table 8). Though age-0 brook trout were also relatively prevalent in these upper main-stem reaches, comprising an average of 30% of brook trout captured, the mean density was only 5.6 fish/100 m (Table 8). Furthermore, age-0 brook trout were infrequently captured in tributary reaches in the upper watershed, with a mean density of 2.5 fish/100 m calculated across all index sites. Consistent with surveys conducted in previous years, brook trout were almost absent in main-stem and tributary reaches in the lower portion of the watershed.

Table 7. Single pass density index (fish/100m) for cutthroat trout and brook trout of all ages and those of age 1 and older sampled by electrofishing at mainstem (M) and tributary (T) index sites in the Alder Creek watershed in 2010. Ordering of index sites corresponds to relative longitudinal position within either mainstem or tributary habitat from downstream to upstream.

		Cutthroat trout density index (fish/100 m)		**Brook trout density index (fish/100 m)**	
Index site	**Channel type**	**All ages**	**Age 1 and older**	**All ages**	**Age 1 and older**
Alder 1	M	0	0	0	0
Alder 2	M	1.6	1.6	3.3	3.3
Alder 3	M	9.8	9.8	0	0
Alder 4	M	13.1	13.1	21.3	21.3
Alder 5	M	8.2	8.2	11.5	11.5
Alder 6	M	8.2	8.2	4.9	4.9
Alder 7	M	1.3	1.3	1.3	1.3
Alder 8	M	21.0	21.0	18.4	17.1
Alder 9	M	0	0	23.0	21.3
Alder 10	M	3.3	3.3	67.3	49.2
Alder 11	M	0	0	68.0	32.0
Alder 12	M	0	0	23.0	19.7
Alder 13	M	4.9	4.9	144.4	139.4
Alder 14	M	3.3	3.3	72.2	60.7
Alder 15	M	0	0	77.1	72.2
Alder 16	M	3.3	3.3	98.4	49.2
Alder 17	M	0	0	93.5	37.7
North Fork 1	T	0	0	182.1	142.7
North Fork 2	T	3.3	3.3	78.7	55.8
North Fork 3	T	0	0	101.7	47.6
North Fork 4	T	0	0	160.8	139.4
North Fork 5	T	0	0	141.1	100.1
North Fork 6	T	0	0	75.5	62.3
North Fork 7	T	0	0	32.8	27.9
North Fork 8	T	0	0	23.0	9.8

Table 8. Single pass density index (fish/100m) for cutthroat trout and brook trout of all ages and those of age 1 and older sampled by electrofishing at mainstem (M) and tributary (T) index sites in the Benewah Creek watershed in 2010. Ordering of index sites corresponds to relative longitudinal position within either mainstem or tributary habitat from downstream to upstream.

		Cutthroat trout density index (fish/100 m)		Brook trout density index (fish/100 m)	
Index site	**Channel type**	**All ages**	**Age 1 and older**	**All ages**	**Age 1 and older**
Benewah 1	M	4.9	4.9	0	0
Benewah 2	M	13.1	13.1	0	0
Benewah 3	M	16.4	16.4	0	0
Benewah 4	M	16.4	16.4	0	0
Benewah 5	M	11.5	9.8	0	0
Benewah 6	M	14.8	14.8	0	0
Benewah 7	M	45.9	44.3	1.6	1.6
Benewah 8	M	1.6	1.6	0	0
Benewah 9	M	14.8	4.9	0	0
Benewah 13	M	4.9	1.6	1.6	1.6
Benewah 14L	M	6.6	3.3	19.7	9.8
Benewah 14	M	4.9	4.9	14.8	9.8
Benewah 14U	M	4.9	1.6	9.8	9.8
Benewah 16L	M	0	0	31.2	21.3
Benewah 2010	M	4.9	4.9	14.8	6.6
Benewah 17	M	11.5	3.3	18.0	14.8
Benewah 18	M	11.5	9.8	16.4	13.1
Coon 1	T	39.4	6.6	0	0
Coon 3	T	0	0	0	0
Bull 1	T	60.7	39.4	1.6	1.6
Bull 2	T	109.9	52.5	0	0
Whitetail 1	T	6.6	6.6	23.0	23.0
Whitetail 2	T	31.2	31.2	0	0
Windfall 1	T	11.5	11.5	18.0	18.0
Windfall 2	T	47.6	44.3	11.5	11.5
Schoolhouse 1	T	16.4	16.4	42.7	36.1
Schoolhouse 2	T	24.6	9.8	8.2	4.9
West Fork 1	T	29.5	14.8	19.7	6.6
West Fork 2	T	34.4	27.9	0	0
South Fork 1	T	18.0	6.6	13.1	13.1
South Fork 2	T	24.6	24.6	4.9	1.6
South Fork 3	T	27.9	24.6	1.6	0

Abundance of age 1+ cutthroat trout in Lake Creek was generally greater in Bozard and West Fork tributaries than in main-stem habitats in 2010, but only in the uppermost reaches of these tributaries, a pattern consistent with previous years (Table 9). A mean density index of 33.5 fish/100 m (range, 24.6 – 37.7 fish/100 m) was calculated for age 1+ fish across the three uppermost sites in Bozard Creek and the two uppermost sites in the West Fork tributary. Moderate densities of 18.6 and 11.5 fish/100 m were also respectively calculated for tributary index sites 12 and 14 in upper Lake Creek. In comparison, age 1+ densities were substantially lower at index sites in lower reaches of tributaries, with an average density of 4.1 fish/100 m. In addition, the mean percent of the total catch constituted by age-1+ fish in tributaries was 95% (range, 70 – 100), indicating the lack of age-0 fish captured at tributary sites in 2010 (Table 9).

Density indices of age 1+ fish in upper main-stem sites were similar to those calculated for lower tributary reaches, with values ranging from 1.6 to 8.2 fish/100 m (mean, 4.4 fish/100 m) at sites 8 through 10. However, density indices across main-stem sites further downriver (i.e., sites 1-7) were generally greater than those upstream, with a mean value of 17.6 fish/100 m calculated (Table 9). Furthermore, age-0 cutthroat trout were prevalent in these lower main-stem reaches, comprising an average of 51% (range, 30-78) of the cutthroat captured at the lowermost six main-stem sites.

In the Evans Creek watershed, age 1+ cutthroat trout were found at moderate to high densities and distributed across all main-stem sites in 2010 (Table 10). Age 1+ density indices averaged 29.4 fish/100 m (range, 14.8-54.1 fish/100 m) in mainstem reaches. In comparison, tributary site indices were the lowest recorded in the Evans Creek watershed in 2010, averaging only 9.4 age 1+ fish/100 m (range, 0 – 14.8 fish/100 m) at the four sites sampled. Moreover, age-0 cutthroat trout were rarely captured at index sites, with age 1+ fish constituting a mean percent of 94 of the total trout captured in 2010 (Table 10).

Table 9. Single pass density index (fish/100m) for cutthroat trout and brook trout of all ages and those of age 1 and older sampled by electrofishing at mainstem (M) and tributary (T) index sites in the Lake Creek watershed in 2010. Ordering of index sites corresponds to relative longitudinal position within either mainstem or tributary habitat from downstream to upstream.

		Cutthroat trout density index (fish/100 m)		Brook trout density index (fish/100 m)	
Index site	**Channel type**	**All ages**	**Age 1 and older**	**All ages**	**Age 1 and older**
Lake 1	M	41.0	24.6	0	0
Lake 2	M	32.8	19.7	0	0
Lake 3	M	29.5	6.6	0	0
Lake 4	M	44.3	31.2	0	0
Lake 5	M	26.2	8.2	0	0
Lake 6	M	23.0	11.5	0	0
Lake 7	M	21.3	21.3	0	0
Lake 8	M	4.9	1.6	0	0
Lake 9	M	3.3	3.3	0	0
Lake 10	M	8.2	8.2	0	0
Bozard 1	T	1.6	1.6	0	0
Bozard 2	T	3.3	3.3	0	0
Bozard 3	T	37.7	36.1	0	0
Bozard 4	T	24.6	24.6	0	0
East Fork Bozard 1	T	42.7	37.7	0	0
Lake 11	T	8.2	8.2	0	0
Lake 12	T	18.6	18.6	0	0
Lake 14	T	16.4	11.5	0	0
West Fork 1	T	6.6	6.6	0	0
West Fork 2	T	0	0	0	0
West Fork 3	T	4.9	4.9	0	0
West Fork 4	T	36.1	36.1	0	0
West Fork 5	T	36.1	32.8	0	0

Table 10. Single pass density index (fish/100m) for cutthroat trout and brook trout of all ages and those of age 1 and older sampled by electrofishing at mainstem (M) and tributary (T) index sites in the Evans Creek watershed in 2010. Ordering of index sites corresponds to relative longitudinal position within either mainstem or tributary habitat from downstream to upstream.

		Cutthroat trout density index (fish/100 m)		Brook trout density index (fish/100 m)	
Index site	**Channel type**	**All ages**	**Age 1 and older**	**All ages**	**Age 1 and older**
Evans 3	M	60.7	54.1	0	0
Evans 6	M	42.7	42.7	0	0
Evans 7	M	14.8	14.8	0	0
Evans 8	M	21.3	19.7	0	0
Evans 9	M	19.7	19.7	0	0
Evans 11	M	24.6	24.6	0	0
Evans 12	M	37.7	36.1	0	0
Evans 13	M	45.9	39.4	0	0
Evans 14	M	29.5	29.5	0	0
Evans 15	M	29.5	27.9	0	0
Evans 16	M	19.7	14.8	0	0
South Fork 1	T	9.8	9.8	0	0
South Fork 2	T	16.4	14.8	0	0
East Fork 1	T	14.8	13.1	0	0
Rainbow Fork 1	T	0	0	0	0

Fyke nets were deployed at four sites in restored pool habitats in the upper main-stem of Benewah Creek in 2010 from August 24 to September 8. A total of 98 fish were captured at these sites, with catostomids and salmonids accounting for 68% of the catch (redside shiner, bullheads, dace, and sculpins comprised the remainder of the captured fish). Whereas catostomids were the most frequently captured group of fish at sites 1 (mean CPUE of 2.3 fish/d) and 2 (mean CPUE of 7.0 fish/d), they were not captured at sites 3 and 4 located further upriver near the confluence of Windfall Creek (Table 11). Conversely, though brook trout were captured at all four locations, they were much more prevalent in the catch at the two main-stem sites near Windfall Creek (mean CPUE of 2.8 fish/d) than at the two lower sites (mean CPUE of 0.6 fish/d; Table 11). Only one cutthroat trout was captured across all four locations, in the pool at site 3 located downriver of the Windfall confluence. Two other large trout (mean total length of 326 mm) were captured at site 2, but based on their external markings were apparently either rainbow trout or hybrids. In total, of the 26 salmonids captured, approximately 90% were brook trout. Generally, numbers of fish captured during a net set did not proportionately increase with the number of soak days at sites sampled in 2010 (Table 11).

3.3.1.4 Stream temperatures

In the upper Benewah watershed, ambient summer stream temperatures generally increased downstream over the 6.4 km section of the main-stem from the mouth of Schoolhouse Creek to 9-mile bridge in 2010, though the longitudinal temperature change was more gradual in upper than in lower reaches. The mean of daily mean temperatures recorded by data loggers over the months of July and August increased 1.7ºC from 13.1 to 14.8ºC downriver across the uppermost 3.2 km reach (Figure 18). In comparison, stream temperatures increased 2.4 ºC from 14.8 to 17.2 ºC along the lowermost 3.2 km reach. Compared with mean ambient stream temperatures, the percentage of time logged water temperatures exceeded 17ºC during July and August in 2010 increased much more dramatically across the lower than the upper half of the 6.4 km monitored main-stem reach. Along the uppermost 3.2 km, daily stream temperatures exceeded 17ºC less than 17% of the time, with percentages increasing downstream by only 12% from 2.4 to 16.5%, whereas percent exceedance increased by over 40% along the lowermost 3.2 km from 16.5 to 57.4% (Figure 19). Consistent with previous years, ambient stream temperatures were cooler in tributaries than in main-stem reaches in the upper Benewah watershed in 2010. Mean temperatures computed over July and August in lower reaches of Whitetail, Windfall, and Schoolhouse creeks ranged between 12.3 and 12.7 ºC, values that were lower than those in main-stem reaches. Moreover, water temperatures rarely exceeded 17ºC in monitored lower reaches of tributaries during the summer of 2010, with the percent time in which loggers recorded values greater than this threshold ranging between 0.3 and 1.3%.

Ambient stream summer temperatures along the 6.4 km section of the upper Benewah mainstem were relatively cool in 2010 compared to previous years. Mean water temperatures across the monitored main-stem reach computed over July and August in 2010 were comparable to those calculated in 2008 but were on average 1.8 and 1.0 ºC lower than those calculated in 2007 and 2009, respectively (Figure 18). In addition, the percent time in which main-stem stream temperatures in July and August exceeded 17ºC was on average 22% less in 2010 than in 2007 (Figure 19). Percent exceedances were also less in 2010 than in 2009, but differences were more pronounced in the lowermost kilometer than in reaches further upriver.

Table 11. Number and catch-per-unit-effort (CPUE, fish/d) for catostomids and salmonids captured by fyke nets deployed at four sites in restored upper Benewah mainstem habitats in 2010. Soak days indicate the elapsed period before a net was checked.

		Catostomids			Cutthroat trout			Rainbow trout / hybrid			Brook trout		
Set date	Soak days	Number	CPUE (fish/d)	Mean length (mm)	Number	CPUE (fish/d)	Mean length (mm)	Number	CPUE (fish/d)	Mean length (mm)	Number	CPUE (fish/d)	Mean length (mm)
Site 1 - pool habitat at the confluence of Whitetail Creek in a reach that was restored in 2006													
24-Aug	2	5	2.5	178	0	0	.	0	0	.	1	0.5	314
26-Aug	1	3	3.0	183	0	0	.	0	0	.	1	1.0	154
27-Aug	3	4	1.3	146	0	0	.	0	0	.	0	0	.
Site 2 - pool habitat in a reach that was restored in 2005													
27-Aug	3	9	3.0	175	0	0	.	0	0	.	0	0	.
30-Aug	1	11	11.0	174	0	0	.	0	0	.	1	1.0	156
31-Aug	1	7	7.0	168	0	0	.	2	2.0	326	1	1.0	233
Site 3 - pool habitat downriver of the confluence of Windfall Creek in a reach that was restored in 2004													
30-Aug	1	0	0	.	0	0	.	0	0	.	2	2.0	162
31-Aug	1	0	0	.	1	1.0	199	0	0	.	9	9.0	168
1-Sep	1	0	0	.	0	0	.	0	0	.	0	0	.
Site 4 - pool habitat at the confluence of Windfall Creek													
2-Sep	5	0	0	.	0	0	.	0	0	.	3	0.6	204
7-Sep	1	0	0	.	0	0	.	0	0	.	5	5.0	178
8-Sep	1	0	0	.	0	0	.	0	0	.	0	0	.

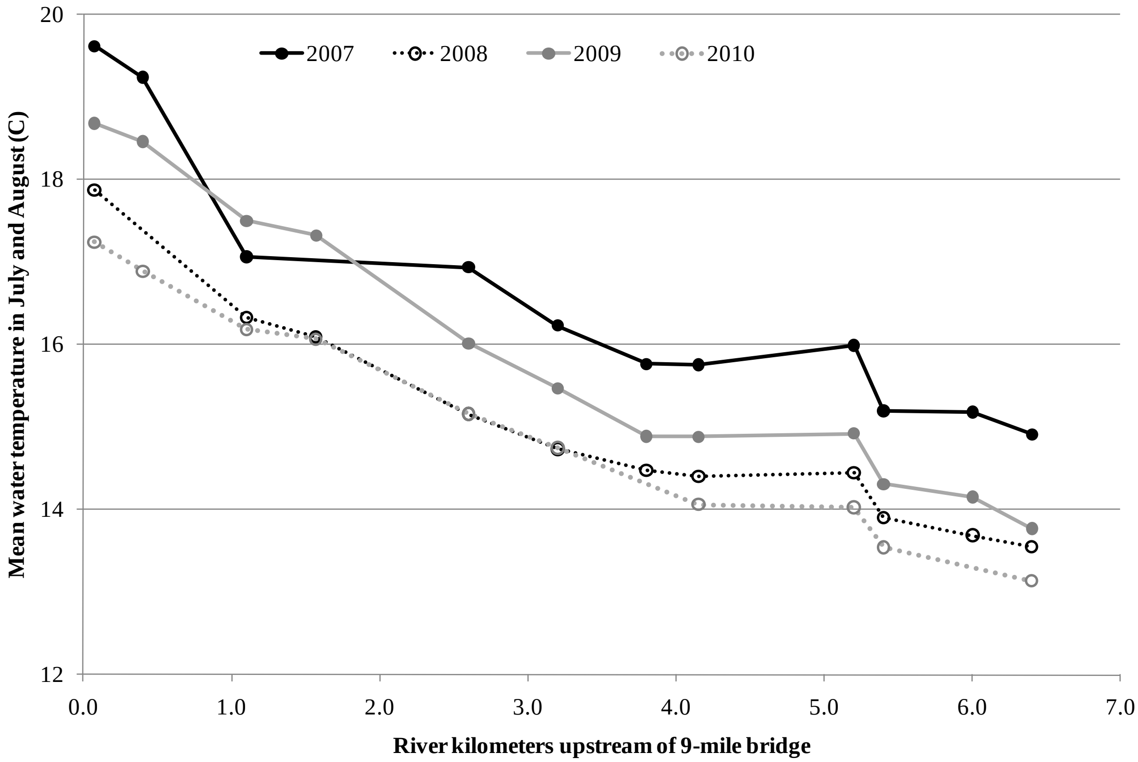

Figure 18. Longitudinal change in the mean stream temperatures calculated over July and August across main-stem Benewah reaches upstream of 9-mile bridge, 2007-2010.

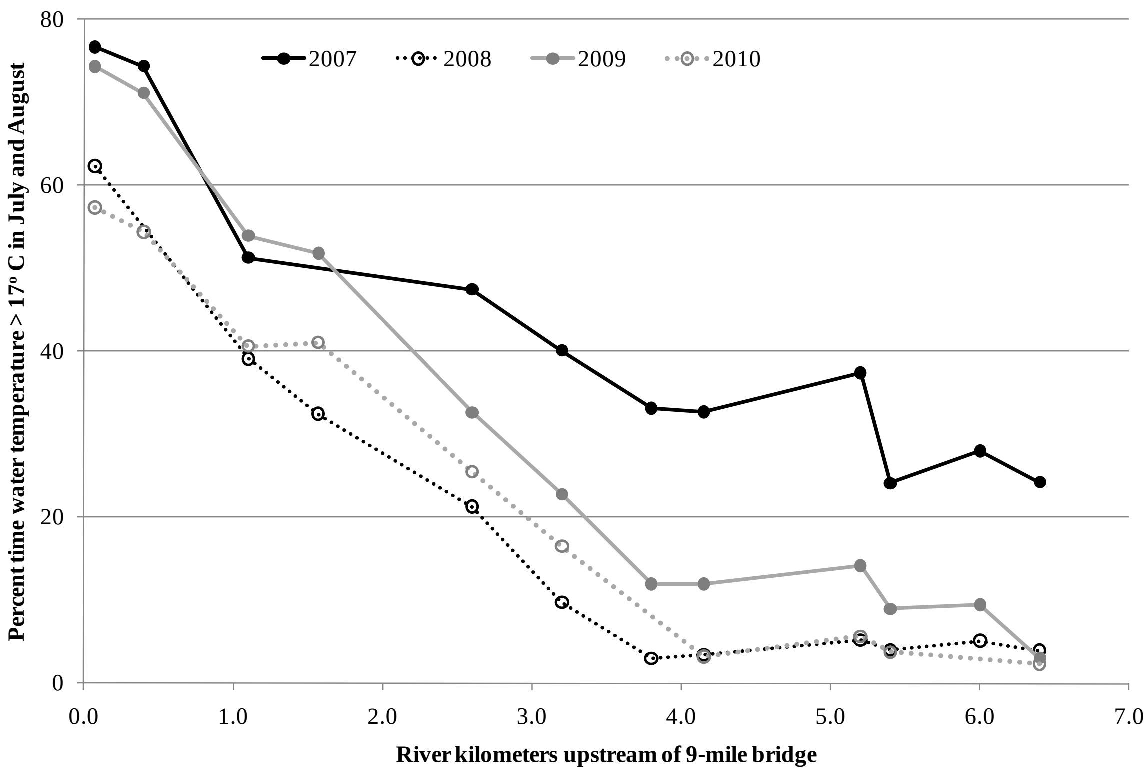

Figure 19. Longitudinal change in the percent time temperatures exceeded 17°C over July and August across main-stem Benewah reaches upstream of 9-mile bridge, 2007-2010.

In the upper Lake Creek watershed, ambient stream temperatures were generally cool throughout most of the monitored reaches during the summer of 2010 (Table 12). Mean stream temperatures calculated over July and August ranged from 13.8 to 14.4ºC for loggers located in reaches proximate to the confluence of the three upper forks. Loggers located further upstream in the Bozard sub-drainage had calculated means during these two months that ranged from 12.2 to 12.6ºC. The percentage of time recorded temperatures exceeded 17ºC was also generally low across the upper Lake Creek watershed during the summer of 2010 (Table 12). Percent exceedances during July and August were between 7.5 and 13.6 for the group of loggers positioned near the confluence of the three forks. Moreover, stream temperatures virtually never exceeded 17ºC during summer months in the upper Bozard sub-drainage. In comparison, stream temperatures in main-stem reaches further downriver were warmer than those recorded in the upper watershed. The mean stream temperature near the old H95 bridge (in close proximity to the location of the migrant traps) calculated over July and August was 16.1ºC, with recorded values during these two months exceeding 17ºC approximately 40% of the time.

Similar to the results documented in the upper Benewah watershed, stream temperatures in the upper Lake Creek watershed in 2010 were comparable to those recorded in 2008 and cooler than those documented in 2007 and 2009 (Table 12). Mean temperatures calculated from loggers (except that deployed in the West Fork of Lake Creek) over the months of July and August were approximately 1.5 and 1.0ºC lower in 2010 than in 2007 and 2009, respectively. Though the percent time summer water temperatures exceeded 17ºC was also generally lower in 2010 than in 2007 and 2009, differences were most prominent when comparing 2010 with 2007. For example, percent exceedances were on average 20% less in 2010 than in 2007 when comparing the data logged in lower Bozard Creek (e.g., near the confluence with Lake Creek main-stem) and in main-stem reaches further downriver.

3.3.2 Effectiveness monitoring – Response to habitat restoration in Benewah watershed

3.3.2.1 Monitoring and evaluation of thermal refugia

Temperature measurements collected along the bottom of pool habitats and their associated tail-outs on August 16 and 18 of 2010 revealed the availability of thermal refugia in the 2.5 km reach of the upper main-stem of Benewah Creek that was restored from 2005 to 2008 (Figure 20). Of the 63 pools surveyed throughout this reach, 24% exhibited temperature differentials (i.e., temperature difference between pool bottom and pool tail-out) that were at least 3ºC. Deep pools typically exhibited much greater temperature differentials than shallow pools ($R^2 = 0.199$; $p < 0.001$). For example, of the 13 pools that were at least 1.5 m in residual pool depth, 8 (62%) displayed temperature differentials that were greater than 3ºC. In comparison, of the 37 pools with residual depths between 1 and 1.5 m, only 6 (16%) displayed differentials greater than 3ºC.

Differences in the magnitude of the temperature differentials among the four restored reaches were also found during the surveys in 2010 (Figure 20). Mean temperature differentials were greater for pools surveyed in the 2006 (2.48ºC) and 2007 (2.19ºC) restored reaches than those surveyed in the 2005 (1.56ºC) and 2008 (1.54ºC) restored reaches. However, given that the calculated temperature difference between pool bottom and pool tail-out is likely dependent on the time of day in which the pool is surveyed (e.g., pool tail-outs may warm as the day progresses and consequently yield greater temperature differentials), and the fact that certain reaches were surveyed later in the day than others, temperatures along pool bottoms may be a more appropriate metric to use when drawing reach comparisons. Thirty percent of the pools in each of the 2006 and 2007 restored reaches had bottom temperatures that were less than 15.6ºC (i.e., 60ºF), whereas only 20% of the pools in the 2008 reach and none of the pools in the 2005 reach displayed bottom temperatures less than 15.6ºC.

Table 12. Comparison of summary statistics among 2007, 2008, 2009, and 2010 water years over July and August for water temperatures recorded by data loggers located in reaches of the upper mainstem of Lake Creek and of proximate tributaries. Logger locations are listed in order of relative longitudinal position in the watershed from lowermost to uppermost. 17^{o}C was considered the upper 95% confidence interval limit for westslope cutthroat trout optimal growth (Bear et al. 2007).

Logger location	2007	2008	2009	2010
	Mean water temperature (C)			
Lake Creek mainstem, near old H95 bridge	17.7	16.1	17.2	16.1
Lake Creek mainstem, downstream of Bozard Creek confluence	15.8	14.4	15.3	14.4
Bozard Creek, upstream of Lake Creek confluence	15.6	13.9	14.8	13.8
West Fork Lake Creek, upstream of Lake Creek confluence	14.0	14.6	14.8	14.2
Upper Lake Creek, upstream of West Fork confluence	15.1	14.8	15.1	.
Bozard Creek, downstream of East Fork Bozard confluence	13.7	12.4	13.3	12.3
East Fork Bozard, upstream of Bozard Creek confluence	13.6	12.2	13.2	12.2
Bozard Creek, upstream of East Fork Bozard confluence	13.9	12.6	13.4	12.6
	Percent time > 17 ^{o} C			
Lake Creek mainstem, near old H95 bridge	55.0	37.6	52.1	40.6
Lake Creek mainstem, downstream of Bozard Creek confluence	34.2	6.3	15.9	13.6
Bozard Creek, upstream of Lake Creek confluence	31.0	5.4	14.4	7.5
West Fork Lake Creek, upstream of Lake Creek confluence	20.6	6.2	13.1	11.5
Upper Lake Creek, upstream of West Fork confluence	24.3	8.2	10.6	.
Bozard Creek, downstream of East Fork Bozard confluence	4.4	0.2	1.2	0.1
East Fork Bozard, upstream of Bozard Creek confluence	2.9	0.0	0.6	0.0
Bozard Creek, upstream of East Fork Bozard confluence	7.4	0.8	2.5	0.0

3.3.2.2 Monitoring and evaluation of beaver dam complexes

Beaver dam monitoring conducted along the upper Benewah mainstem in 2010 demonstrated evidence of dam-building activity occurring between the summer and fall survey periods. During the initial survey, only 6 of the 38 (16%) documented intact natural beaver dams were considered to be active. In comparison, 33 of the 39 (85%) intact beaver dams identified during the fall survey were considered to be active as inferred by the presence of recently placed materials. However, unlike the surveys conducted in 2009, significant increases in dam height were not detected in 2010 for those dams measured in both survey periods for any of the three monitored reaches (Figure 21). The lack of a detectable increase in height for seasonally paired measurements in the lowermost reach (i.e., reach 1) may have been due to the turnover of dams in that reach. For example, six of the intact dams documented in the summer, all of which were built with unstable materials and had a mean dam height of 0.42 ft, were either no longer present or were not structurally intact during the fall survey and thus were evaluated as a decrease in dam height over the survey periods. Of the remaining dams in this lowermost reach, dam height actually increased over the survey periods by 0.4 ft on average, so that mean dam height in the fall was greater than that calculated in the summer (Figure 22). The lack of a detectable increase in the upper reach (i.e., reach 3), on the other hand, may have been due to the disturbance imposed by our in-stream restoration activities. Eight dam locations in this reach were either reinforced with the placement of LWD or had stable engineered structures installed. These activities may have either disturbed the structural integrity of nearby natural dams or discouraged beaver activity in this reach.

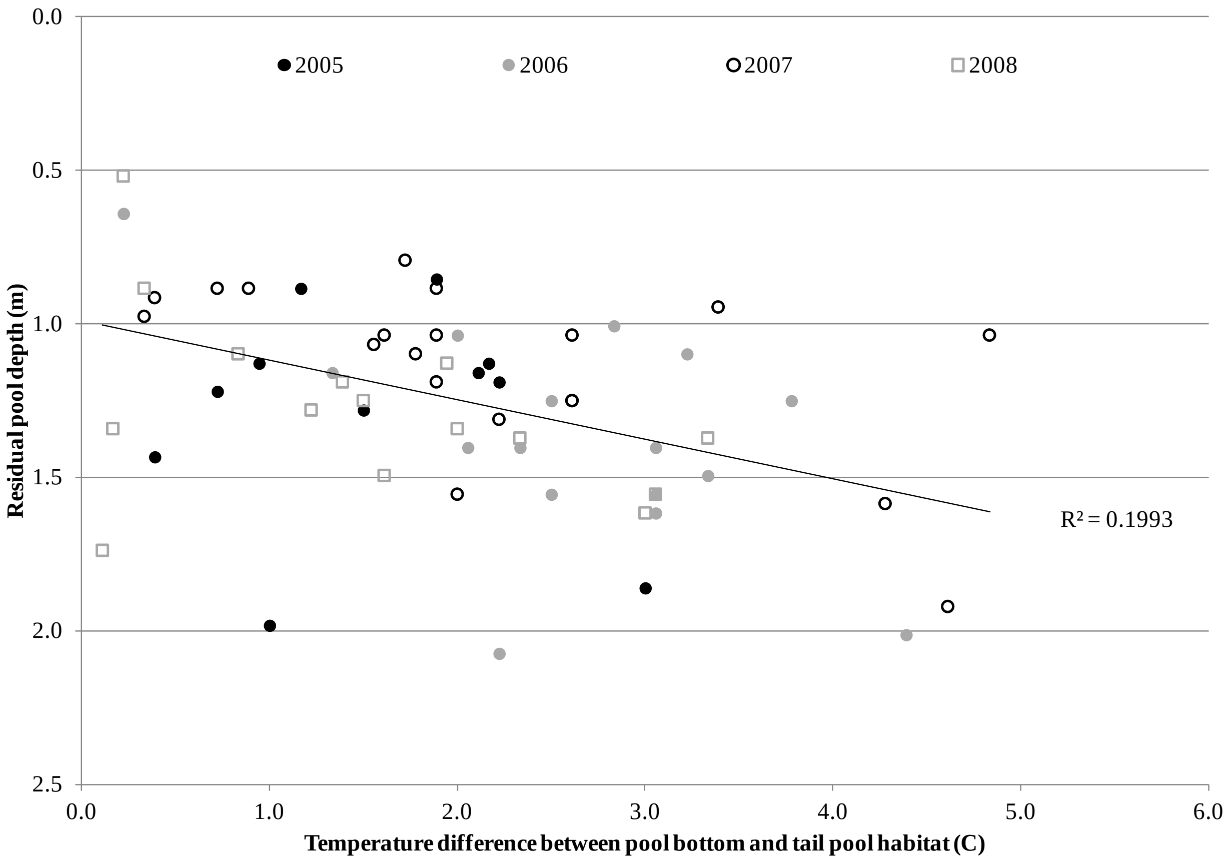

Figure 20. Temperature differences between pool bottoms and associated tail pool habitats for pools of various residual depth that were surveyed in 2010 along the four contiguous main-stem reaches that were restored from 2005 to 2008 in the upper Benewah watershed. Symbols differentiate pools by the year in which the reach was restored. The trend line for all measured pools is also displayed.

Paired seasonal dam height data collected in the fall of 2009 and the summer of 2010 permitted an assessment of differences in the apparent degree of dam complex stability among the three reaches. In the lower and upper reaches that are laterally bounded by meadows, mean dam height respectively decreased by 0.93 and 0.64 ft (Figure 21). Evidently, many of the dams were either blown out or lost structural materials over the winter and spring. However, dam height did not significantly change in the middle reach (i.e., reach 2) where a relatively intact riparian forest still exists (Figure 21). In addition, mean dam height in this reach was over twice that recorded in the other reaches during the summer 2010 survey (Figure 22), with most of the largest dams (e.g., > 2 ft) that were measured in both survey periods in 2010 documented in this reach. Because of the availability of large, stable dams in this reach, extensive dam building may not have been required to provide the habitat desired by beavers and may have explained the lack of increase in dam height in this reach from summer to fall in 2010 (Figure 21).

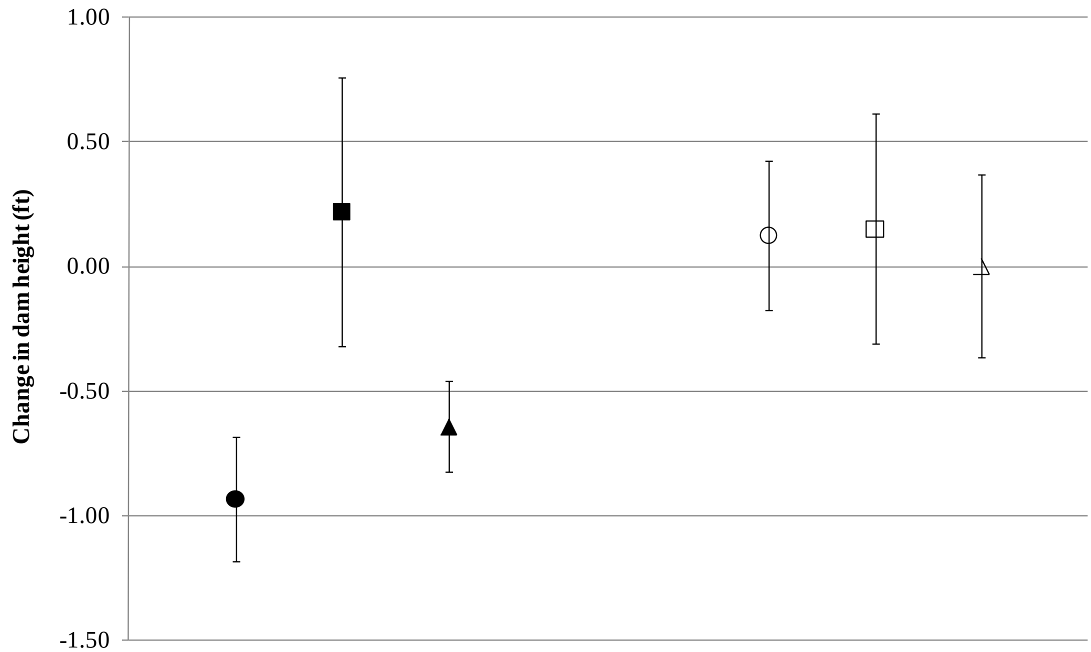

Figure 21. Mean change in dam height with associated 95% confidence intervals from the fall 2009 to the summer 2010 survey (filled symbols), and from the summer 2010 to the fall 2010 survey (open symbols). Circle, squares, and triangles represent reaches 1, 2, and 3, respectively.

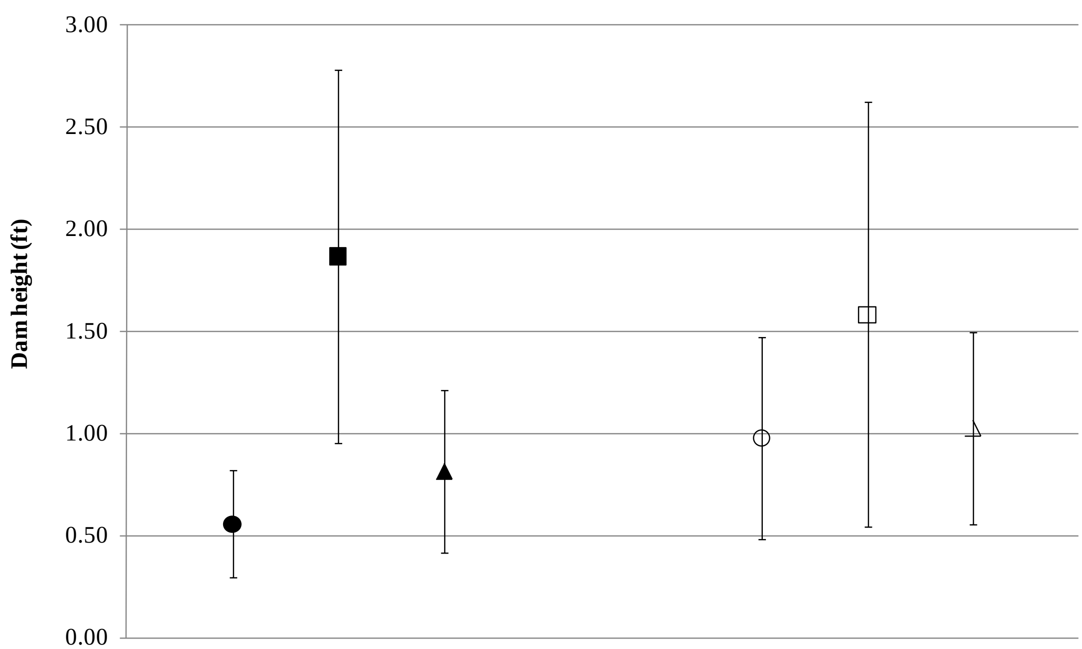

Figure 22. Mean height (± one standard deviation) for beaver dams documented during the summer (filled symbols) and fall (open symbols) surveys in 2010. Circle, squares, and triangles represent reaches 1, 2, and 3, respectively.

3.3.2.3 Monitoring and evaluation of physical habitat metrics

Percent canopy cover at representative sites along the main-stem reach that was restored from 2005 to 2008 (i.e., T-01 – T-04) ranged from 15 to 34% in 2010 (Table 13). Generally, percent canopy cover along the Phase I reach was positively correlated with the number of elapsed years since individual segments were treated (e.g., canopy cover at T-01, which was restored in 2005, was greater than that at T-02, which was restored in 2006). Mean canopy cover values at sites located within treated reaches of Phase II restoration (i.e., T-05 and T-06) ranged from 46 to 56%, and were both comparable to those computed in the control reach upstream and greater than those recorded in the downstream Phase 1 reach (Table 13). These results collectively reflect the higher degree of disturbance imposed upon the riparian area by Phase I (e.g., removal of vegetation during channel modifications and re-alignment) than by Phase II restoration actions.

Generally, the mean percent fines calculated across bankfull in riffles were relatively high across sites located in both Phase 1 and Phase 2 restored reaches in 2010, with values ranging between 18 and 44% (mean, 33%; Table 13). Values in the control reach were also comparatively high, ranging from 54 to 60%. These high values reflect the geology of the upper Benewah watershed and the fine-grained, erodible soils that constitute the banks and streambed outside the active channel. Indeed, the mean percent fines in the active channel (i.e., wetted channel width) was substantially less than that calculated across bankfull for each of the survey sites, with calculated values at or below our performance standards (15% fines; Table 13). Regarding linkages to suitable fish habitat (e.g., spawning gravels, invertebrate production), percent fines in the active, wetted channel rather than bankfull channel may be a more appropriate metric to track over time. The lowest mean percent fine values across bankfull and within the wetted, active channel were those computed at sites in the Phase 1 restoration reach where rock was imported to construct the elevated riffles (Table 13).

Large woody debris (LWD) metrics computed for sites located in the Phase I reach varied widely in 2010 (Table 13). Counts and volumes were much greater across the uppermost (T-04) and lowermost (T-01) sites (26.4 – 28.9 pieces/100 m and 12.4 – 17.2 m^3/100 m, respectively) than across the two intervening sites (5.9 – 9.2 pieces/100 m and 3.5 – 6.1 m^3/100 m, respectively). The differences in the LWD metrics among sites could be attributed to the dynamic nature of this reach, especially during the initial years subsequent to restoration, as LWD introduced by our restoration actions becomes mobilized and displaced during high discharge events. Mean LWD counts (3.3 pieces/100 m) and volumes (0.3 m^3/100 m) at the two sites, T-05 and T-06, encompassed by Phase II restoration were markedly lower than those at the Phase I restoration sites (Table 13). These values reflect the paucity of relict LWD throughout the Phase II reach, in addition to less wood introduced during Phase II than during Phase I actions. Mean values for LWD metrics in the control reach (14.1 pieces/100 m and 4.3 m^3/100 m) were intermediate of those calculated for the Phase I and Phase II restoration reaches (Table 13).

Pool habitat exceeded 50% of the surveyed stream length for all but one of the main-stem sites located in Phase I and II restoration reaches in 2010 (Table 13). At site T-06, within which an engineered logjam structure and beaver dams were located, approximately 95% of the surveyed stream length was impounded. Pool habitat was not as extensive within the two sites in the control reach upstream where only 21 – 47% of the stream length was classified as pools (Table 13). However, the lack of classified pool habitat in this control reach may be partly due to the criterion of one foot of residual pool depth that has been used to delineate main-stem pool habitat. Mean residual pool depth for sites located in the restoration reaches ranged between 0.8 and 1.3 m, with the deepest pools located in the Phase I restoration reach (Table 13). This was not unexpected given that Phase I restoration actions created deep, meander bend pools. Notably, mean residual pool depth in restored main-stem reaches, other than site T-05 where dam turnover was prevalent (see *3.2.2.3 Monitoring beaver dam complexes in upper Benewah main-stem reaches*), was maintained at our performance standard of one meter. Mean residual pool depths in the control reach (0.5 – 0.6 m) were approximately half that computed for the restoration reaches (Table 13).

Table 13. Physical habitat attributes measured at main-stem index sites in the upper Benewah watershed in 2010. Sites T-01 – T-04 were located in the reach that was addressed by Phase I channel reconstruction actions from 2005 to 2008, and sites T-05 – T-07 were located in the reach that has received Phase II restoration actions since 2009. Sites C-01 and C-02 were located in a control reach upstream. For each site, mean percent canopy cover was calculated from 10 equidistant channel transects and mean percent fines was calculated from 5 riffle transects, unless otherwise noted. Large woody debris and pool habitat were assessed throughout the entire site length.

		Mean percent fines in riffles		Large woody debris metrics		Pool habitat metrics		
Site	**Mean percent canopy cover**	**Bankfull**	**Wetted**	**Count (#/100 m)**	**Volume (m^3/100 m)**	**Percent pools**	**Mean residual pool depth (m)**	**Pool volume (m^3/100 m)**
T-01	34	34	16	26.4	17.2	54	1.3	81.4
T-02	23	28	4	5.9	3.5	58	1.1	122.7
T-03	15	18	3	9.2	6.1	35	1.3	71.0
T-04	30	29	6	28.9	12.4	61	1.3	227.4
T-05	46	38	7	1.3	0.1	55	0.8	21.9
T-06	56	43 [a]	17 [a]	5.2	0.5	94	1.0	93.3
T-07	19	44	12	17.7	10.7	66	1.0	78.5
C-01	57	60	11	9.8	2.5	47	0.6	13.2
C-02	42	54	12	18.4	6.2	21	0.5	5.4

[a] Because of the lack of riffle habitat, only two transects were established

3.3.3 Effectiveness monitoring – Response to brook trout removal in Benewah watershed

In 2010, 627 brook trout were removed from the upper Benewah watershed during removal efforts (Figure 23). Of these 627, 291 were captured by shocking the 2.0 km reach of contiguous main-stem habitat upstream of the 12-mile bridge to the confluence of the two forks. Another 249 brook trout were removed from lower reaches of five tributaries in the upper watershed. A total of 7 days was expended on these removal efforts within the timeframe of August 25 – September 8. An additional 68 brook trout were removed from the enclosure upstream of 12-mile bridge over the course of eight different sampling occasions from August 30 to October 25. Given the minimal time expended shocking the enclosed reach (5-10 min/sampling occasion), much less effort was spent in 2010 than in those years when main-stem reaches downriver of the 12-mile bridge were shocked (i.e., 2006-2008). However, the number of ascending brook trout captured by the enclosure in 2010 was substantially less than the numbers of brook trout removed annually from main-stem reaches below the 12-mile bridge (514 – 1192) in those years. Lastly, 19 brook trout were intercepted by the weir at 9-mile bridge and removed from the live box in 2010. Mean total length for these 19 fish was 271 mm. In comparison, the mean total length for those brook trout removed in upper main-stem habitats during shocking efforts in 2010 was only 142 mm.

The total number of brook trout removed in 2010 was much less than that removed during each of the first four years of the suppression program when more main-stem habitat was shocked (Figure 23). Although this was the direct result of our reduced effort in these two years, the 291 brook trout that were removed from the 2.0 km index main-stem reach upstream of 12-mile bridge, a reach that has been regularly sampled since 2005, was the lowest value recorded over the last six years, and markedly lower than the numbers removed during the initial suppression years of 2005 (962) and 2006 (904). In addition, a noticeable difference in the length distribution of brook trout removed from the index reach upstream of 12-mile bridge was observed when comparing data collected from the 2010 efforts to that collected in 2005, the first year in which the index reach was shocked, and in 2009, the first year in which the enclosure upstream of 12-mile was deployed (Figure 24). Approximately 45 and 55% of the brook trout removed from the index reach were considered to be young-of-the-year (total lengths <= 80 mm) in 2005 and 2009, respectively. In comparison, only 25% of the brook trout removed in 2010 were considered to be young-of-the-year.

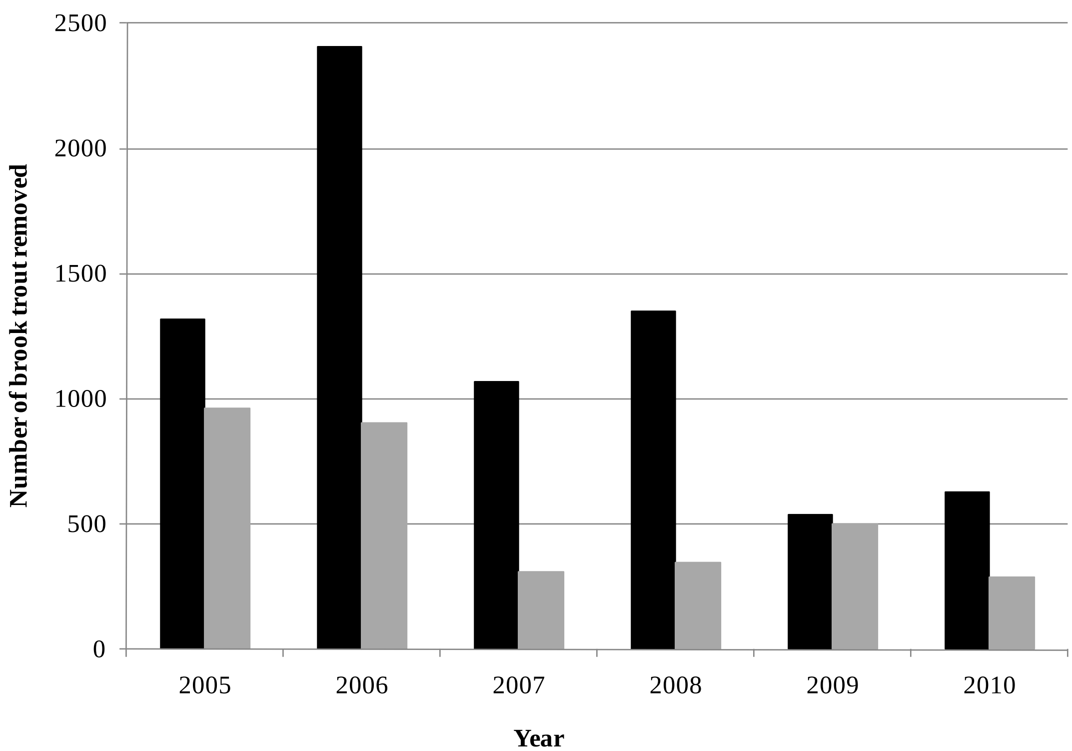

Figure 23. Total number of brook trout removed (darkened bars) and those removed from an index main-stem reach upstream of 12-mile bridge (gray bars) in the upper Benewah watershed from 2005 to 2010. Curtailment of main-stem removals downriver of 12-mile bridge began in 2009.

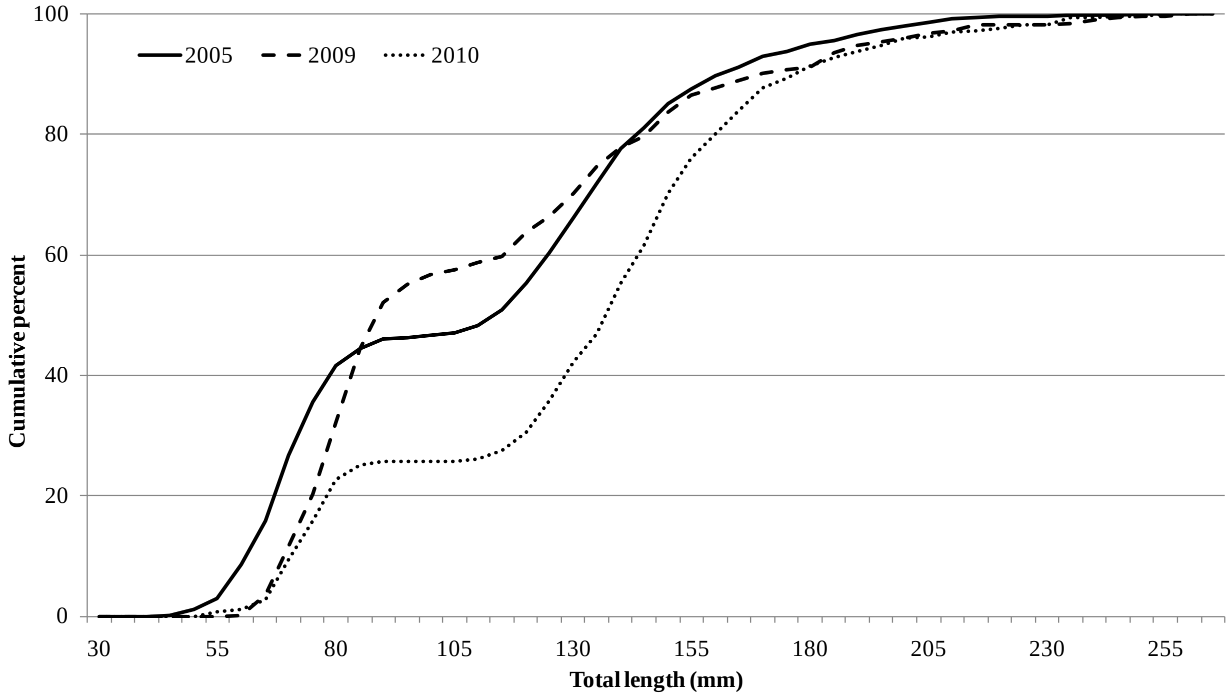

Figure 24. Cumulative length distributions for brook trout removed from the 2.0 km index main-stem reach upstream of 12-mile bridge in the upper Benewah watershed in 2005, 2009, and 2010. The threshold size for young-of-the-year brook trout was considered to be approximately 80 mm.

3.4 Discussion

3.4.1 Status and trend monitoring

3.4.1.1 Adfluvial cutthroat trout migration

It is imperative that we reliably track temporal changes in adult spawners given that one of the primary objectives of our recovery efforts is to augment the number of returning adult cutthroat to our adfluvial watersheds. With the advent of the modifications made to the UP trap in Lake Creek in 2009 and to the UP trap in Benewah Creek in 2010 to improve trap performance, we have been able to intercept and mark (e.g., opercle punch) a large percentage of the available fish to provide precise estimates of 113 and 72 spawning adults in 2010, respectively. In previous years, periodic spring freshets had been observed to repeatedly depress the trap panels below the water surface during spring migratory periods permitting ascending adults to pass. Our modifications have enabled trap panels to remain elevated during high spring flows and have allowed the traps to effectively fish under a much wider range of conditions than before. In addition, during previous spring periods we had noticed insufficient current velocities at the entrance to the live box raceway at the UP trap in Benewah Creek, which likely had not been providing the necessary cues to attract fish, and could have been contributing to our inability to capture upriver migrants. We addressed this deficiency in 2010 and redirected the flow upriver of the UP trap to augment that delivered through the raceway to provide a more prominent velocity cue. The capture of 71 adults in 2010 relative to the lack of captured fish in previous years (e.g., one fish) in the Benewah UP trap attests to the effectiveness of our improvements. We are also in the process of building a floating weir for installation at the mouth of Benewah Creek that has the capability of trapping both ascending and descending spawners. This trap is expected to be operable in the spring of 2012, and our marking protocol will be employed to obtain spawner abundance estimates for the entire Benewah watershed. The ability to annually obtain accurate estimates of adult abundance should permit a reliable assessment of the status and trends of adfluvial cutthroat trout spawners in both watersheds.

Our estimate of 113 spawning adults in Lake Creek in 2010 was lower than the estimate of 175 adults generated in 2009 and much lower than the range of post-spawn fish, 233 to 257, that had been captured over the years 2005-2007, a range that was likely biased low given that it did not account for mortality on the spawning grounds. The decrease in estimated spawners in both 2010 and 2009 may reflect a true decline in the Lake Creek subpopulation of mature adfluvial adults present in Lake Coeur d'Alene. However, we should not exclude the possibility that our improved trap performance could have adversely influenced the number of upriver migrants captured. We did document trap avoidance behavior, as many of the captured adults apparently were reluctant to ascend into the live box as evidenced by the days in which they were repeatedly interrogated by the fixed PIT-tag array before capture. In addition, a number of the detected adults were never captured in the UP trap in Lake Creek nor later detected indicating that they had not ascended upriver of the trap. In fact, we had estimated a total of 162 adults had approached the Lake Creek UP trap, which was 43% more than our estimate of those adults that ascended upriver of the trap. Even in Benewah Creek, we noticed that a couple of fish lingered downriver of the UP trap (e.g., elapsed periods of 18 d) before they were captured. If there is a tendency for some fish to engage in trap avoidance behavior, then the improved ability to impede upriver movement as a result of our UP trap improvements may be discouraging some fish from entering the entrance to the livebox raceway. Whatever the reason for the low numbers of adults estimated to have ascended past the Lake Creek UP trap in 2010, it is imperative that we continue to annually monitor the behavior of spawners with our interrogation arrays as they approach our traps to evaluate if the low numbers are a real declining trend or the result of our trap modifications.

Adult spawner estimates, in combination with juvenile outmigration estimates and associated age structure information, should permit the derivation of outmigrant per spawner ratios, metrics that would allow tracking of watershed-wide trajectories in juvenile production in addition to aiding in the assessment of in-stream population response to our restoration actions (Bradford et al. 2005). Though results from our watersheds in 2010 indicate the potential for attaining accurate spawner abundances, we do not yet have the capability of obtaining accurate juvenile outmigrant abundance estimates. When our DN traps were considered fishing, we were able to obtain rather precise outmigrant estimates, with trap efficiencies typically exceeding 80%. However, the inability to deploy DN traps during the ascending limb of the hydrograph likely contributed to the omission of those juveniles that outmigrated early. The findings of juveniles captured soon after trap deployment and high juvenile catch rates during peak discharge periods (e.g., Lake Creek), allude to the possibility that fish may have been moving downriver as flows increased prior to trap deployment. The interrogation of PIT-tagged juveniles in late March and early April in both systems during a period of increasing discharge supports this supposition. Furthermore, though amenable levels of discharge in 2010 permitted effective trapping throughout much of the time when traps were deployed, most notably in Lake Creek, several high discharge events seriously compromised trap performance. For example, trapping operations were suspended in Benewah Creek over a two-week period in late April and early May during a period of high flows. Given that a large number of juveniles may have been outmigrating at this time, this period of trap inoperability may have explained in large part the outmigrant abundance estimate in Benewah Creek (394) that was approximately a tenth of that in Lake Creek (3858). As such, our juvenile outmigrant abundance estimates in both systems are undoubtedly biased low in most years.

The biases inherent in our juvenile outmigrant abundance estimates, however, may have the potential to be redressed by using PIT-tagged juveniles in mark-recapture models. Juveniles that have been tagged during late summer and early fall electrofishing surveys in tributary habitats and passively detected by fixed antennas the following spring would serve as marked fish in models. The recapture of a percentage of these ‘marked’ individuals, along with other unmarked fish, in DN traps during operable periods would then enable the calculation of the total number of outmigrating juveniles, and thus obviate the need to effectively capture fish throughout the spring outmigration period. Such a change in protocol is contingent upon the restructuring of sampling procedures in tributary habitats during our summer and fall population surveys and the ability to reformulate PIT-tag methodology, which will be given further consideration in the near future.

Alternatively, there may be a need to re-evaluate the techniques we employ to capture juveniles to ensure that the full range of traits expressed in the outmigrating cohort is reflected in those individuals that are tagged. Though juveniles in both watersheds were tagged representatively throughout *capture* periods in 2010, it may be necessary to capture and tag those early outmigrants to further our understanding of the apparent observed relationship between timing at outmigration and the probability of return to first spawn that has been illustrated in Lake Creek. Furthermore, as evidenced by the juvenile data collected in Lake Creek in 2010, a relationship may exist between the time (early) and size (large) at which juvenile cutthroat trout initiate their downstream movement to the lake, a relationship that has been described for other migratory salmonids (Irvine and Ward 1989; Bohlin et al. 1993). However, this relationship cannot be rigorously assessed without handling the early component of the outmigration. Trapping modifications would require those that permit structures to be installed under a wide variety of flow conditions, especially at high discharge during early spring when juveniles may be first cued to outmigrate.

The data collected during trapping periods in Lake Creek suggest that outmigrating juvenile cutthroat trout were generally larger in 2010 than in 2007 and 2009. However, these length differences could be attributed to annual differences in the timing of trap deployment relative to peak discharge events. In 2010, the DN trap was installed before many of the peak discharge events, whereas in the earlier years traps were installed after much of the high flow periods had passed. If larger cutthroat trout tend to outmigrate earlier on peak discharge events, then many of the larger individuals may have already moved downriver before traps were deployed in 2007 and 2009. In a similar manner, though the mean length of captured juveniles in Benewah Creek was comparable to that in Lake Creek during trapping periods in late May and June in 2010, the overall size distribution of outmigrating juveniles in Benewah Creek was generally smaller than that in Lake Creek,. However, the DN trap in Benewah Creek was not fully operational until May 12 after the greatest peak discharge period, and consequently many of the earlier migrants, putatively larger, may have already migrated downriver. These comparisons demonstrate the need to capture fish throughout the entire outmigration to better understand the full suite of characteristics expressed by adfluvial juvenile cutthroat trout in both watersheds.

There is also a need to better understand the reason for the apparent lack of motivation to outmigrate that was exhibited by several of the juveniles that were PIT-tagged in both watersheds in 2009 and briefly interrogated during the spring of 2010. Evidently, these fish resided in the stream for another year before moving downriver. A large percentage of these fish (80%) had been tagged in late May and June of 2009 during the tail end of the spring hydrograph at low levels of discharge. Notably, similar phenomena were observed for juveniles tagged in 2008 in the Benewah watershed that were interrogated during 2009 outmigration periods. At this time, it is unclear as to whether juveniles captured late in the season are actively moving out of the system or are just inadvertently intercepted by the trap during localized early-summer foraging movements. On the other hand, the DN trap may be disrupting the behavior of outmigrants during latter spring periods as discharge declines. At low flows, the DN traps tend to create a slack water environment upstream, and consequently, may not provide the appropriate velocities that juveniles require to cue continued downriver movement. Similar delayed movements have been noted for juvenile salmonids outmigrating through impounded reaches of larger river systems (Venditti et al. 2000). In support of this conjecture, we noticed a marked decrease in DN trap efficiencies during June trial periods in 2009 as discharge decreased. In fact, of those juvenile fish tagged in late May and June of 2009 that were interrogated this year, 87% were release trial fish that were not recaptured in 2009. Based on trap inspection in 2009, the lack of recaptures for these tagged fish was not likely due to gaps in the trap, but possibly the result of dilatory behavior exhibited by these fish after their release either due to avoidance behavior or to difficulties in re-negotiating the trap. Whatever the reason, it is imperative that we continue to monitor this behavior to better evaluate whether our traps are adversely impacting the motivation to outmigrate.

Results from the PIT-tagging efforts that have been implemented since 2005 in Lake Creek suggest that only 1.7% of outmigrating juveniles return to spawn as adults. Although empirical estimates of in-lake survival rates for adfluvial cutthroat trout are scarce, several studies have provided values with which comparisons may be drawn. Annual survival rates of 49% were estimated in Lake Koocanusa for cutthroat trout from reservoir entry as juveniles to first time spawning two years later which equate to approximately a 25% return rate (Huston et al. 1984). Gresswell et al. (1994) estimated a 16-25% return rate for adfluvial juvenile Yellowstone cutthroat trout emigrating from Arnica Creek in the Yellowstone Lake system in the early 1950's. Compared with these studies, our juvenile-to-spawner return estimates are substantially lower. These comparisons underscore the importance of understanding the reason for these extremely low return rates given that in-lake juvenile survival has been considered a key vital rate in determining overall population trajectories for adfluvial cutthroat trout (Stapp and Hayward 2002).

The low apparent survival rates may be an artifact of sampling procedures, including compromised detection probability, and either tag loss or tag-related mortality. With regards to detection probability, however, it is highly likely that even without capture a tagged adult returning to spawn will be detected by our interrogation systems. The fixed PIT-tag antennas span the entire channel width in Lake Creek and interrogate the wetted channel in Benewah creek under most flows, and are positioned immediately downriver of the RBW traps in both systems where the likelihood of detection would be great as upriver migrants linger in the vicinity of the detection field as they attempt to negotiate the trap. Furthermore, tagged juveniles released as test batches upriver of the antenna array in Lake Creek demonstrate detection rates of 95-100% (Firehammer et al. 2010). High rates of immediate tag loss or unintended handling mortality also likely do not sufficiently explain the absence of detected fish. Over the past five years in which mortality/retention trials have been conducted, all fish have been alive upon release and found to retain their tags. Although we have captured adults with a clipped adipose fin that have not scanned, indicating that tags have been shed, tag loss may have occurred during a prior spawning event. Indeed, we documented tag loss in 2010 between capture events for several of the adult females that had been tagged as outmigrating juveniles. Tag loss in cutthroat trout on the spawning grounds has been reported in other systems as well (Bateman et al. 2009).

Alternatively, the low juvenile-to-adult survival rates may be real and attributed to processes operating in Lake Coeur d'Alene. Although these seemingly limiting processes are not well understood, the juvenile attributes of those fish that have been detected as adults may yield insight into some of the mechanisms. Juveniles that have survived to return to spawn in Lake Creek generally migrated earlier in the spring, but more importantly, were larger when tagged than those that have not returned. Though data are limited, juveniles that have returned to spawn in the Benewah watershed were also relatively large when tagged (158-190 mm). Size at outmigration may reflect the energetically-mediated capacity to survive, especially if size at tagging is positively related to the size or condition of the individual at onset of the overwintering period. In addition, large size at outmigration may confer benefits to the individual by decreasing its vulnerability to predation either through enhanced swimming performance or the capability to escape gape-limited predators. Although the processes that are apparently limiting survival and giving rise to the observed discrepancies in timing and size at outmigration are largely unknown, it is imperative to better understand whether predation is a predominant mechanism regulating survival rates in Lake Coeur d'Alene and potentially inhibiting recovery of cutthroat trout.

As such, we intend to initiate a study in 2011 to evaluate the impact of two non-native piscivores, northern pike and smallmouth bass, on cutthroat trout survival in Lake Coeur d'Alene. Cutthroat trout had been found to be a major dietary item for northern pike in earlier studies conducted on Lake Coeur d'Alene (Rich 1992), and smallmouth bass, a documented salmonid predator, have apparently increased in numbers in the last ten years according to lake-wide surveys (Maiolie et al. 2010). Moreover, the seasonal habitat preferences of both northern pike and smallmouth bass coupled with the migratory behavior of adfluvial cutthroat trout suggest a high potential for spatial and temporal overlap among species. Northern pike use shallow, vegetated habitats throughout the year and are especially common in those habitats during spring when they are spawning (Casselman and Lewis 1996). Smallmouth bass are also particularly common in shallow-water habitats during the spring (Brown and Bozek 2010). In Lake Coeur d'Alene, these shallow-water habitats are located in the bays into which our monitored streams enter, and through which cutthroat trout during spring periods move. Consequently, the study will incorporate two field seasons in which both Windy Bay and the southern end of the lake, into which Lake and Benewah creeks respectively enter, will be frequently sampled during spring migratory periods when predation on cutthroat trout by both species may be high and occur over relatively short time periods. In addition, sampling will be conducted less frequently but more extensively across the lake during non-migratory periods (e.g., summer through late fall). Demographic (e.g., age structure, growth, seasonal abundance) and dietary data will be collected from both predators during these repeated sampling events and incorporated into bio-energetic models to estimate the consumption of cutthroat trout by both species. Information gained from this study will support the development of alternative actions that may be considered for implementation to manage the fish assemblage in Lake Coeur d'Alene.

In-lake processes may not only be impacting juveniles but could also be influencing survival rates of post-spawn adults. Because of the lack of returning adults PIT-tagged as juveniles, the supplemental PIT-tagging of adults, which was first conducted in Lake Creek in 2009 and in Benewah Creek in 2010, will increase the sample size and allow us to more accurately evaluate post-spawn survival rates and return frequency. Given our estimates of spawning ground survival (e.g., 83-89%), much of the estimated post-spawn return rates for adults should be attributed to processes in the lake. Though only one year has elapsed since adults were tagged in Lake Creek in 2009, approximately 20% have already returned as consecutive year spawners. After a couple more years in which alternate year spawners have had a chance to return, we will be able to better evaluate adult return rates for this group of tagged adults. Furthermore, several more years of data from consecutive, tagged adult groups from both Lake and Benewah creeks should provide insight into whether in-lake processes are impacting adult survival and whether the strength of these processes differs between watersheds.

Accurate estimates of adult return rates require accounting for tag loss, which as evidenced by the results from our double-tagging protocol can be considerable in the short-term. Compared to the 12-16% tag shed rates that were estimated in Lake Creek in 2009 and in Benewah Creek in 2010, we estimated that 30% of adults shed their tags in Lake Creek in 2010. The reason for the disparity in tag loss rates is unknown. For Lake Creek adults tagged in 2010, it was possible that the tag, rather than left embedded in the muscle tissue, was inserted too deeply during implantation and pushed into the body cavity. Given that we have documented females, which were tagged in the body cavity as juveniles, shedding their tags and the fact that all of the tagged adults that were found to shed their tags in Lake Creek in 2010 were females, this explanation is highly probable. Though it will be difficult to evaluate whether tags are lost after adults outmigrate to the lake, all of the adult females that had been tagged in 2009 and were interrogated at the Lake Creek UP trap this year retained their tags between capture events in 2010. This suggests that long-term tag retention may be high if tags are not shed during the initial spawning event after implantation. Furthermore, these results highlight the importance of estimating short-term tag loss rates for each group of tagged adults rather than applying a single estimate over years or across watersheds.

In addition to the potential impacts non-native predators may be having on cutthroat trout survival during lake residence, we have also documented population level impacts from non-native rainbow trout in stream habitats of Benewah Creek. In 2010, we classified 10% of the adults captured in the UP trap in Benewah Creek as potential hybrids based on external characteristics. Furthermore, two of these fish had been tagged as outmigrating juveniles and at that time had also been classified as hybrids. Given that a recent genetic analysis has corroborated our ability to correctly classify highly hybridized juveniles (Corsi et al. 2010), it is likely that these adults were actually hybridized fish. The analyses conducted by Corsi et al. (2010) has confirmed the incidence of hybridization in adfluvial watersheds of the Coeur d'Alene Basin, though levels of genetic introgression were generally low (i.e., less than 3%) and much of the hybridization had been considered to be the result of infrequent episodic events. However, in the Benewah watershed there was evidence of some relatively recent hybridization with rainbow trout. Much of this recent hybridization could be attributed to escapees from stocked, rainbow trout ponds that are located on private land in close proximity to stream reaches in the upper Benewah watershed. Indeed, we captured four fish in the Benewah DN migrant trap in 2010 that appeared to be mature non-native rainbow trout. Because of the reported overlap in reproductive behavior (e.g., timing and location of spawning) between non-native strains of rainbow trout and westslope cutthroat trout (Muhlfeld et al. 2009a), there is a high probability for interbreeding to occur in the Benewah watershed. To minimize the impacts from pond escapees, we are proposing contacting these landowners and offering the opportunity for sterile, triploid rainbow trout to be stocked in their ponds.

At this time, we do not have a protocol for culling hybridized adult fish from our watersheds, though the development of procedures are duly needed given the apparent impacts hybridization can have on the fitness of westslope cutthroat trout populations (Muhlfeld et al. 2009b). Moreover, in the Benewah watershed, juvenile fish that have been identified as hybrids in traps were generally some of the larger outmigrants captured, and, though based on a small sample size, have displayed some of the largest two-year juvenile-to-adult growth increments we have documented to date for recaptured tagged fish. These characteristics may translate to elevated competitiveness during stream residence and increased reproductive investments, respectively. Despite the desire to remove putative adult hybrids, apparently field crews had difficulty in consistently classifying an adult migrant as a hybrid in the Benewah system in 2010. For example, the four fish that were assigned a hybridized status that were recaptured in the DN trap were not similarly classified earlier when they were captured in the UP trap. We do not want to inadvertently remove highly fecund fish if there are uncertainties in status assignments. As a result, a rigorous classification protocol, corroborated by genetic assignment, will need to be developed to train field personnel in correctly identifying highly hybridized fish. Moreover, more information is needed regarding the prevalence of hybridization in different parts of the watershed. Corsi et al. (2010) identified higher levels of genetic introgression in lower reaches of the Benewah watershed (e.g., Bull Creek, lower main-stem) than in upper reaches (e.g., South

Fork of Benewah, Windfall Creek). Trapping and PIT-tagging putative adult hybrids, in combination with genetic clips, at the mouth of Benewah Creek, and interrogating these fish at fixed PIT-tag antennae stationed in the main-stem and in tributaries throughout the Benewah watershed will aid in evaluating whether there are certain locations to which hybridized adults commonly return to spawn.

3.4.1.2 Salmonid stream surveys

Population surveys conducted at index sites during the summer and fall of 2010 permitted an assessment of cutthroat trout abundance at a much finer spatial scale than that attainable using our migrant trap data. Consistent with surveys conducted in previous years, indices of cutthroat trout density in our adfluvial watersheds were predominantly greater in tributary than in main-stem habitats. Furthermore, in both Lake and Benewah creeks, cutthroat trout were often found at greater numbers in upper than in lower reaches of tributaries. Sub-optimal rearing conditions could be contributing to the low numbers of fish in these lower tributary reaches. Prioritizing these reaches for prospective habitat improvements should increase the productive potential of tributaries, and in the case where tributary mouths are in close proximity to one another, improve connectivity and promote a more robust meta-population structure in upper portions of both watersheds.

Findings from our stream population surveys also indicated moderate but greater densities of cutthroat trout in lower than in upper mainstem reaches of both watersheds. The reasons for the spatial disparity in main-stem densities within watersheds remain unclear. Juvenile cutthroat trout from tributaries in the upper portions of these watersheds may be engaging in stepwise migratory behavior during periods of stream residence in which they gradually move downstream to larger-sized rearing habitats (Zydlewski et al. 2009). This would imply an avoidance of upper main-stem habitats and a preference for lower reaches, and would suggest that these lower reaches provide more suitable rearing habitat than those upriver. Alternatively, the moderate numbers of cutthroat trout in these lower main-stem reaches may reflect the prevalence of reproduction in the lower portions of these watersheds. In the Benewah watershed, densities of age one and older cutthroat across Bull Creek index sites were some of the highest observed. Moreover, both Bull Creek and Coon Creek supported the highest densities of age-0 fish. Forced emigration either through density-dependent mechanisms or de-watering of tributary reaches (e.g., Coon Creek) may be giving rise to the densities observed in lower main-stem reaches of the Benewah watershed. In Lake Creek, age-0 fish were also most abundant in lower main-stem reaches which may be indicative of the presence of spawning in and emigration from intermittent tributaries in the lower part of this watershed.

In Evans and Alder creek watersheds, which support prevailing resident cutthroat trout populations, spatial distributions were vastly different between systems, but were similar to those documented during previous surveys within each system. Consistent with past years, cutthroat trout in Alder Creek were only found in lower reaches, and at low densities, and have been seemingly displaced from upper reaches of the watershed, where they were virtually absent but brook trout were numerous. In comparison, cutthroat trout in Evans Creek were found to be evenly distributed at moderate densities across main-stem reaches of the entire watershed, a pattern that has been repeatedly observed in our annual population surveys.

In 2010, we changed our sampling protocol and replaced the multi-pass depletion procedures with single pass efforts to track trends in indices of cutthroat trout abundance in our watersheds.

The mark-recapture study that was conducted in our watersheds in 2009 provided evidence that a single-pass estimator of abundance would have the capability of tracking true abundance (Firehammer et al. 2011). Furthermore, results from trend analyses that have been conducted using our cutthroat trout index site data indicated that first pass catch data provided similar interpretations of temporal trends in watershed-wide abundance as did our multi-pass depletion estimates (Firehammer et al. 2011). Others have also found single-pass indices to perform well in predicting abundances for salmonid populations in small-streams (Strange et al. 1989; Jones and Stockwell 1995; Kruse et al. 1998; Mitro and Zale 2000; Bateman et al. 2005). We intend to continue to conduct only single passes in our summer and fall population surveys, and use these first-pass catch data in periodic trend analyses every three to four years to re-evaluate temporal patterns of cutthroat trout abundance in our watersheds.

3.4.1.3 Stream temperatures

The ambient stream temperatures recorded in Lake and Benewah watersheds in 2010 still support the suitability of tributaries over main-stem reaches as cutthroat trout rearing habitats during mid-summer periods. Summer water temperatures remained below 17^{o}C, a value above which is considered sub-optimal for cutthroat trout growth (Bear et al. 2007), more than 98% of the time in upper reaches of Lake Creek tributaries and in tributaries in the upper Benewah Creek watershed. In contrast, temperatures exceeded this threshold more than 40% of the time in main-stem reaches proximate to our trap site in Lake Creek and in main-stem reaches that were restored from 2005 to 2008 in Benewah Creek. Given the consistently higher densities of cutthroat trout observed in tributary than in upper main-stem habitats, the mid-summer differences in rearing temperatures between tributary and main-stem reaches likely explain in part the distributional patterns of cutthroat trout observed in both watersheds (Dunham et al. 1999; Paul and Post 2001; Sloat et al. 2001; de la Hoz Franco and Budy 2005).

However, in the Benewah watershed, the main-stem meadow reach that is currently receiving Phase 2 restoration treatments (i.e., 3.2-6.0 km upriver of 9-mile bridge) afforded more suitable ambient stream temperatures than reaches downriver that had been restored during Phase 1. Temperatures were relatively consistent throughout the Phase 2 reach, and remained below the 17^{o}C benchmark at least 80%, and often more than 90%, of the time during the summer. Observed differences in temperature signatures between these two main-stem reaches may in part be explained by differences in available canopy cover. An enclosed canopy of hawthorne and alder is regularly present along the Phase 2 reach, whereas much of the Phase 1 reach is still relatively exposed as a result of the channel re-construction activities that removed a substantial portion of the riparian vegetation. Years will be required before the post-construction streamside and riparian plantings will ameliorate the conditions introduced by the channel disturbances.

Alternatively, the observed differences in Benewah main-stem stream temperatures may also be explained by the greater influence of groundwater inputs in the upper than in the lower reach. In past years, monitored springbrooks in the upper watershed have consistently displayed temperature signatures during summer months that were much cooler than those recorded in adjacent main-stem habitats. In addition, data from groundwater wells that have been installed within floodplain habitats of the unconstrained Phase 2 reach indicated that transmission of groundwater from off-channel sources to the main channel generally occurs along the interface between the gravel/cobble and silt/clay layers located 4-6 feet below the surface. Apparently, this reach of the main-stem is closer to these off-channel groundwater sources and/or receives substantially cooler groundwater inputs than downstream reaches that were addressed during Phase 1 restoration.

Ambient main-stem stream temperatures recorded in 2010 in both Benewah and Lake creek watersheds were similar to those recorded in 2008, somewhat cooler than those recorded in 2009, and considerably cooler than those in 2007. These results indicate that a proper evaluation of whether our habitat enhancement activities are moderating thermal regimes will require accounting for all those drivers that may influence the thermal regime in any given year. For example, mean air temperatures were overall much cooler during summer months in 2010 than in 2007. Consequently, models that examine the influence of channel restoration actions (e.g., increasing riparian canopy cover or improving water retention and groundwater recharge in floodplain habitats) on base-flow stream temperatures will require other covariates, such as air temperature and descriptive indices of the annual flow regime (e.g., snow water equivalents), to clarify linkages. Cooler summer stream temperatures over the last three years could have also provided more favorable growing environments for cutthroat trout. An analysis that examines age specific growth rates of cutthroat trout that have been captured during our sampling efforts over the last 4-5 years is currently being completed in part to address this question.

3.4.2 Effectiveness monitoring – Response to habitat restoration in Benewah watershed

3.4.2.1 Monitoring and evaluation of beaver dam complexes

From 2005 to 2008, stream restoration activities in the upper Benewah watershed involved major channel alterations by building new meander bends and filling in existing sections of incised channel, elevating the streambed in other incised reaches to create new riffles with the addition of imported rock, and adding a sizeable amount of large woody debris to stabilize banks and provide in-stream cover (Phase 1 restoration). As a result, sinuosity has increased and channel entrenchment reduced such that bankfull floods can access the floodplain. Channel modifications also significantly increased large wood loadings and created pools that are generally longer and deeper than those that previously existed before restoration. However, given the intensive physical manipulation and channel hardening involved in this restoration phase, an alternative design concept was desired that involved less extensive channel modification and that would work with and reflect natural channel forming processes, hydrology and geomorphic conditions.

Beginning in 2009 and continuing in 2010, the restoration approach for the upper Benewah main-stem has shifted from an active to a more passive approach which relies on beavers to aggrade the stream channel over time and promote the exchange of water between the channel and the adjacent floodplain. There has been widespread recognition that beaver dams play a vital role in maintaining and diversifying stream and riparian habitat (Naiman et al. 1988; Pollock et al. 1994, 2003; Gurnell 1998; Collen and Gibson 2001), and increasingly, beaver are being viewed as valuable partners in restoration initiatives in the Pacific Northwest and elsewhere (Pollock et al. 2007; Bondi 2009; Walker et al. 2010; Finnigan and Marshall 1997). For one, beaver dam complexes have the potential to attenuate peak flows during high discharge periods and retard erosive current velocities. For example, Beedle (1991) estimated that five large beaver ponds in series could reduce peak flows of a 2-year flood event by 14% and peak flows of a 50-year event by 4%. Furthermore, the slackened current velocities upstream of beaver dams facilitate the deposition of sediment and organic material transported from upstream reaches, and contribute to bed aggradation. Over decadal time scales, sediment accumulation rates of 0.25 to 6.5 cm/year have been reported upstream of beaver dams (Scheffer 1938; Devito and Dillon 1993; Butler and Malanson 1995). In some cases, rates of sediment aggradation, especially during initial filling periods can be substantial. For example, Pollock et al. (2007) documented vertical aggradation rates as high as 0.47 m/year over the first several years in an entrenched tributary in the John Day River basin in eastern Oregon.

Beaver dam complexes also have the potential to drive hydrologic processes in low-gradient, adjacent riparian areas by impounding water during high discharge events and promoting extended periods of overbank flooding. For example, Westbrook et al. (2006) found that beaver dams located in an unconstrained fourth-order reach of the Colorado River substantially enhanced the depth, extent, and duration of floodplain inundation during peak flow periods. In addition, the authors reported elevated water tables in areas both adjacent to and downriver of the beaver ponds during high flow periods and even during base flow periods as beaver impoundments curbed the recession of water tables during the summer months. Dam complexes also have the potential to promote the seasonal persistence of pool habitats because of their ability to retain impounded water for extended periods of time. Research on the effects of wood dams in small, third-order streams suggests that they can retain water at least 50% longer than streams where such dams are absent (Ehrman and Lamberti 1992). Given the much lower permeability of beaver dams compared to large wood jams, it is reasonable to expect beaver dams to retain water for much longer periods of time. The maintenance of beaver impoundments throughout both mid-summer and winter periods will provide the deep pool habitats that have been reported to be preferred by cutthroat trout in small stream systems (Heggenes et al. 1991; Rosenfeld et al. 2000; Harper and Farag 2004; Lindstrom and Hubert 2004).

The documentation of numerous beaver dams during both the 2009 and 2010 field seasons demonstrated a significant influence of beaver in the upper mainstem of Benewah Creek, and justifies the use of beavers as partners in our stream restoration efforts. In addition, geological evidence from test pits in the upper mainstem of Benewah Creek suggests that historical engagement of flood flows on the valley floor was most likely in response to both the blockage effects of large wood falling into the channel and aggregating smaller wood, and the influence of beaver dams (DeVries and Fetherston 2008). However, the degree to which beaver-aided restoration efforts can move habitats toward a re-expression of natural habitat capacity and quality is predicated on the stability of dams and their ability to function during flows that are generally greater than the 5-10% exceedance level (~84 cfs). It is not until the depths associated with these higher flows are reached that the Shields relation (Leopold et al. 1964) predicts initial entrainment of the bed surface in this reach. Accordingly, it is at these flows that the channel blockages and flow constrictions associated with stable, persistent beaver dams then begin to more effectively sort the mobilized sediments, storing finer particles and progressively enhancing floodplain connectivity via backwater effects mediated by channel aggradation. The beaver dam surveys that were conducted in 2010 provided evidence that dam stability and persistence existed in portions of the upper Benewah main-stem, and that stable dams were often composed of large pieces of wood or were found in reaches with high LWD loadings (e.g., reach 2). Moreover, there was also a clear association of these stable, persistent dams with an adjacent, intact riparian forest and a historical continuity of recruitment of coarse wood to the channel.

Overall, however, a lack of stability exists in dam complexes in the upper Benewah main-stem as many of the documented dams (> 75%) are either not built with or not built upon stable materials (e.g., large woody debris) but instead are composed of small alder and grasses. Dams composed of such small, unstable materials tend to be either eliminated or lose their structural integrity during wintertime rain-on-snow events (e.g., shearing forces of mobilized ice) or high flow events during spring freshets. Indeed, high rates of dam turnover have been documented in our surveys. Moreover, of those surveyed dams that were not found to be blown out, many lost structural building materials and decreased in dam height over the winter and spring. Because dam re-building efforts have been found to generally commence in late summer and early fall in upper main-stem reaches, the loss of dam height and deep impounded areas limits the amount of deep pool habitat and associated thermal refugia (e.g., Firehammer et al. 2009) available to cutthroat trout during the warmest mid-summer periods. Consequently, not only is the role of natural dams in flood attenuation and sediment storage, and the long-term benefits arising from gradual streambed aggradation and floodplain connectivity uncertain, but the short-term benefits in providing suitable summer rearing habitats for cutthroat trout may also not be realized under the current unstable state of beaver dams in the upper main-stem.

To aid beavers in their channel forming processes, a series of in-stream structures have been constructed that serve to emulate flow obstruction effects of natural wood jams and beaver dams. The augmentation of stable wood structures in this reach are intended to support a short-term goal of conferring stability to natural dams and to improve the trajectory for natural process recovery. These actions should allow for more frequent and extensive floodplain connection during annual floods, and is a natural analog alternative to large scale construction of elevated riffles. Seven of thirteen structures have been constructed at cross-sections previously occupied by less stable natural dams. The remaining structures were built in relict channels that will be either reactivated or available as side-channel habitats in the future, with the effects being measurable at that time. In addition, several natural dams have been reinforced, and, in other locations, large wood has been placed to supply materials for dam construction. Implementation planned for the 2011 field season will replace and/or reinforce approximately 7 additional natural dams spaced relatively evenly throughout the larger 3.5 km reach.

Tracking the interactions of natural dams and engineered structures and making additional observations regarding the temporal stability of these dam complexes may provide some valuable insights for restoration approaches that can be applied to other similarly degraded systems. Because this new restoration approach partially relies on beavers to facilitate channel-forming processes, channel changes (e.g., bed aggradation) will likely occur more gradually over time, and certainly not as abruptly as was measured in response to Phase 1 actions implemented downstream. In addition, some refinement of the survey protocols may make future monitoring more efficient in evaluating the effectiveness of this approach. Making detailed measurements within the backwatered channel over the last two field seasons identified important relationships between dam characteristics (e.g., height) and dependent variables such as inundated area and residual pool depth. Though describing the changes in these metrics following restoration is useful for purposes of implementation and effectiveness monitoring, data collection is time-intensive. Moreover, these relationships are only likely to change if natural dams are constructed differently in the future or if significant accumulation of sediment over time results in measureable changes in channel dimensions. While these types of changes are anticipated, the frequency of measuring these variables could be reduced without losing much information. On the other hand, observations that better describe the timing of natural dam construction and failure have been overlooked. During the 2011 field season, we intend to devise a rapid, reach-scale assessment of dam condition that will be conducted multiple times throughout the summer and fall at critical timeframes to provide greater temporal resolution of dam-building dynamics as they relate to our restoration measures.

3.4.2.2 Monitoring and evaluation of thermal refugia

Thermal refugia were documented in 2010 in that reach of the upper Benewah main-stem that underwent channel restoration from 2005 to 2008, though temperature heterogeneity in these restored pools was not as great as that observed during similar surveys conducted in previous years (Firehammer et al. 2009; Firehammer et al 2010). The observed differences among survey years may be attributed to pool tail-out temperatures that were generally lower in 2010 than in previous years, in part because of overall cooler summer air temperatures, which in turn constrained the development of a strong thermal difference between tail-out and pool bottom temperatures. Because of the apparent influence of annual weather patterns on the strength of temperature heterogeneity in stratified, restored pool habitats, future monitoring efforts should supplement our periodic, spatially-continuous surveys with temporally-continuous temperature records in spatially-distinct, restored pools. Deploying temperature loggers at the bottom of several, representative pools in each of the restored reaches should allow our monitoring program to track changes in these bottom temperatures as main-stem pools undergo natural progression. In addition, continuous temperature records in both riffles (supplied by our current logger configuration) and pool bottoms will permit the detection of when thermal stratification develops during the summer and if stratification processes differ among restored reaches.

The magnitude of thermal refugia (i.e., mid-summer pool bottom temperatures) documented during the 2010 survey was found to differ among the four reaches, with pool bottom temperatures generally cooler in those reaches restored in 2006 and 2007 than those restored in 2005 and 2008. Though deep pools tended to display more thermal stratification, and often cooler bottom temperatures, than shallow pools, mean residual pool depths are not appreciably different among the four reaches. Rather, these results may be due to inherent variability in groundwater pathways and dynamics that influence stream temperatures differently across main-stem reaches that have been restored from 2005 to 2008. More importantly, given that cold-water patch frequency and area have been considered important indices that explain salmonid occurrence and abundance in small stream systems (Torgersen et al. 1999; Ebersole et al. 2001, 2003), our data illustrate the potential for erroneous conclusions to be drawn regarding the positive impacts from stream restoration if only select reaches are surveyed for the presence of thermal refugia.

3.4.2.3 Monitoring and evaluation of cutthroat trout response

Despite the mosaic of thermal refugia and the complex habitat (e.g., deep pools and LWD additions) that has been created in restored reaches of the upper Benewah mainstem, we have yet to see evidence of a significant response by cutthroat trout. In 2010, as in previous years, cutthroat trout were virtually absent in summer surveys conducted in restored main-stem reaches. Various explanations have been proffered for the apparent lack of utilization of these restored habitats, which have been described in detail in previous annual reports (Firehammer et al. 2009, 2010). Briefly, these include, but are not limited to, the following: (1) a sufficient degree of isolation between core rearing tributaries and restored habitats, mediated by distance or other barriers (e.g., temperature), that inhibit dispersal (Bond and Lake 2003; Pretty et al. 2003); (2) insufficient tributary densities to induce density-dependent emigration responses (Johnson et al. 2005; Shrank and Rahel 2006); (3) a lag in positive fish response because of the repeated, acute artificial disturbances imposed by channel reconstruction on ecological and hydrological stream properties over the period from 2005 to 2008; and (4) the persistence of limiting factors in reaches adjacent to those restored (Moerke and Lamberti 2003; Cowx and Van Zyll de Jong 2004). We realize that because we are not only amending local deficiencies in habitat complexity but also addressing impaired processes that operate at larger spatial scales, the re-establishment of natural processes will occur gradually, and as such, detection of positive responses by cutthroat trout may require a longer timeframe. As we progressively address contiguous reaches in the upper Benewah mainstem with Phase 2 implementation, we expect to continue to increase the extent of favorable rearing habitats that are conducive for cutthroat trout colonization and growth.

Another explanation for the absence of cutthroat trout in restored reaches in past survey efforts has been attributed to the shortcomings and ineffectiveness of our electroshocking techniques in capturing fish when employed in deep, pool habitats (Firehammer et al. 2011). In 2010, we elected to experiment with fyke nets as an alternative method to sample the salmonid assemblage in restored main-stem pools in upper Benewah Creek. Overall, numbers of salmonids captured with fyke nets were low, with more fish captured in restored pools near the mouth of Windfall Creek than in pools located in Phase 1 restored reaches. Furthermore, most of the captured salmonids were brook trout. Though catch rates were low, the spatial distribution and species composition of captured salmonids in these main-stem reaches generally reflect that demonstrated by our electroshocking surveys that have been conducted in recent years (Firehammer et al. 2011). Despite the equivocal evidence of their feasibility as an effective sampling method, we still intend to further experiment with fyke nets in these restored reaches in subsequent sampling years. Moreover, because we found that longer set durations did not likewise increase the number of fish captured, and likely biased CPUE estimates, the sampling protocol would likely consist of replicate single-day sets conducted in randomly selected pools throughout each of the restored reaches.

Though a direct numerical response to restoration has not been observed in mainstem reaches from our summer surveys, deepened main-stem reaches may be providing suitable overwintering habitat that was available only in a limited capacity before restoration. Both juvenile and adult cutthroat trout have been found to prefer deep pools as winter refuge habitat in small stream systems (Jakober et al. 1998; Brown and Mackay 1995; Harper and Farag 2004; Lindstrom and Hubert 2004). In addition, cutthroat trout have been found to respond positively to improvements to winter refuge habitat. Solazzi et al. (2000) found cutthroat trout abundance to increase, presumably owing to higher overwinter survival rates, following the creation of winter habitat for salmonids in coastal Oregon streams. In addition, Roni and Quinn (2001) found higher densities of cutthroat trout at sites with experimental large woody debris additions than at control sites, but only during winter and not summer sampling. Evaluating the winter distribution of cutthroat trout in upper Benewah mainstem habitats may reveal benefits of our channel construction activities that have not been realized from our summer surveys. In order to perform such an evaluation, cutthroat trout will need to be PIT-tagged in tributaries in the upper watershed during summer and fall electrofishing surveys, and their movements monitored throughout the fall and winter using strategic placement of fixed, PIT-tag interrogation stations in main-stem habitats. We intend to use half-duplex technology for this monitoring effort, and currently we are in the process of experimenting with optimizing the performance of half-duplex antennae using dimensions that will accommodate the size of our stream reaches.

3.4.3 Effectiveness monitoring – Response to brook trout removal in Benewah watershed

Brook trout abundance and expansion in the Benewah watershed are apparently being regulated at a modest level, especially when their distributional patterns and densities in this watershed are compared with those in Alder Creek, a neighboring drainage. For example, brook trout in Benewah Creek were only abundant within a spatially-distinct, core location in the upper watershed which included main-stem reaches upriver of 9-mile bridge and lower reaches of tributaries that are proximate to this main-stem segment. In comparison, brook trout in the Alder Creek watershed were more widely and continuously distributed across sampled reaches. In addition, densities of age one and older brook trout within the core location in the upper Benewah watershed were three to five times lower than those calculated across reaches in the upper Alder watershed. The differences in brook trout abundance between Benewah and Alder creeks that were found in 2010 may be partly the result of our suppression program. Over the last seven years, we have removed approximately 8000 fish from the upper Benewah watershed, and in some instances have documented a substantial decline in densities in those tributaries where removal efforts have been concentrated (Firehammer et al. 2011). However, at the watershed scale, densities of brook trout have consistently been greater in Alder Creek than in Benewah Creek, even prior to the onset of our suppression program.

Alternatively, the observed differences in abundance patterns may be attributed to biological or physical differences between watersheds that govern colonization mechanisms and probabilities of establishment. Differences in apparent vulnerabilities of proximate systems have been reported by others that have examined brook trout invasions in the west (Adams et al. 2002; Dunham et al. 2002; Shepard 2004; Benjamin et al. 2007). For one, the invasion process may still be in its incipient stage in the Benewah watershed. Populations of cutthroat trout and brook trout overlap in the upper Benewah watershed, whereas the spatial distribution of these two salmonids are almost wholly disjunct in Alder Creek, suggesting a longer history of co-occurrence and eventual displacement of cutthroat trout by brook trout in the Alder watershed (Dunham et al. 2002). However, given the proximity of these watersheds to each other, brook trout expansions should have proceeded at similar rates if colonizing migrants arrived from common downriver sources (Peterson and Fausch 2003). As another possible explanation, the productive adfluvial life-history strategy that is prevalent in the Benewah but not the Alder watershed may confer an advantage to cutthroat trout in the former that permits a greater biotic resistance to invasion (Griffith 1988).

In addition, habitat conditions that are more conducive to brook trout establishment may be more prevalent in Alder Creek than in Benewah Creek. The spatial distribution of brook trout and their habitat preferences have commonly been associated with low gradient reaches with deep, low velocity habitats (e.g., beaver ponds) that serve both as summer rearing and overwintering habitat (Chisholm et al. 1987; Cunjak 1996; Lindstrom and Hubert 2004; Benjamin et al 2007). Recent habitat surveys conducted across our watersheds have indicated that pool habitat is approximately three times as great in Alder Creek than in Benewah Creek (Miller et al. 2008). Moreover, these surveys found that 33% of the pool habitat documented in Alder Creek was formed by dams, whereas only 3% of the pool habitat in Benewah Creek was dammed. These results underscore the importance of maintaining the suppression program in Benewah Creek given that our current restoration approach in the upper watershed, which promotes the stability of beaver dam complexes and augments pool habitat, will likely increase the suitability of habitat for brook trout.

Because of the potential for our restoration actions to improve brook trout rearing habitats in upper main-stem reaches of the Benewah watershed, it is imperative that we offset these unintended benefits and create recruitment bottlenecks at other vital life stages. As a result, the current suppression approach aims to curb reproductive success rather than attempting to remove as many fish as possible. In the past, an inordinate amount of time was being annually allocated to shocking the deep, pool habitats that had been created by our restoration actions in that reach of the upper main-stem from 9-mile bridge to 12-mile bridge. Capture probabilities can be less than 30% in these deep habitats (Firehammer et al. 2011), and as a result we may have been only capturing a minority of the brook trout residing in these main-stem reaches. Furthermore, a substantial portion of this main-stem stretch is dominated by low-gradient depositional beaver dam complexes, which, though likely serving as suitable rearing habitats, may not provide suitable spawning substrates.

As of 2009, we modified our suppression tactics and currently are concentrating our removal efforts in that reach of the main-stem upstream of 12-mile bridge where shocking has proven to be more effective because of the lack of large, deep pools (i.e., > 3 ft), and which has been considered more suitable for spawning because of more preferable substrates than reaches downriver. Moreover, this reach has consistently supported the greatest adult brook trout densities over the course of the suppression program, and may be serving as a source of mobile, reproductive individuals for the colonization and establishment of localized sub-populations in proximate tributaries (Benjamin et al. 2007). Our tactics have also included the deployment of a temporary barrier upstream of 12-mile bridge to prevent brook trout that are residing in downstream main-stem reaches from ascending and accessing the seemingly more suitable spawning habitats upstream. The interception and capture of around seventy brook trout in 2010 in the enclosed area upstream of 12-mile bridge that is bounded by the barrier attest to the presence of such upstream fall spawning movements by brook trout in this part of the watershed.

The examination of numerical and length distribution data for brook trout removed from the main-stem index reach upstream of 12-mile bridge should permit an evaluation of the effectiveness of our new approach in inhibiting brook trout reproduction. In 2010, numbers of brook trout removed from this index reach were comparable to those enumerated over the previous three years. More importantly, the percent of fish removed from the index reach that comprised young-of-the-year in 2010 was half that observed in prior years. Moreover, indices of young-of-the-year density in 2010 at main-stem (5.6 fish/100 m) and tributary (2.5 fish/100 m) index sites in the upper Benewah watershed were low, and markedly less than that calculated across sites in the upper Alder watershed (23.3 fish/100 m). Seemingly, our curbed removal efforts, which began in 2009, did not lead to substantial reproductive output, and our re-focused tactics in fact may have contributed to the lack of age-0 brook trout captured in 2010. Additional years of monitoring data should permit a better evaluation of whether these results were indeed genuine or just an artifact of natural variability.

The current approach has also substantially reduced the amount of time annually invested in our suppression program. Prior to 2009, approximately three weeks of crew effort were expended in fall removal activities, whereas only 5 to 7 days have been dedicated in each of the last two years. Moreover, over time, if our current approach proves to be successful in maintaining brook trout abundance at a low, manageable level, then we may be able to further curtail efforts and reduce the frequency at which we conduct our suppression measures. Several years of consecutive removals followed by a couple years of suspended implementation may minimize the costs of the program but still provide benefits to our cutthroat trout population (Peterson et al. 2008b). In addition, refraining from removing fish over a year or two will allow us to examine the extent of compensatory resilience in the Benewah watershed that has been documented for invasive brook trout in other systems (Meyer et al. 2006).

4.0 RESTORATION AND ENHANCEMENT ACTIVITIES

4.1 Introduction to Project Summaries

Implementation of restoration and enhancement activities occurred in Benewah and Lake creeks during 2010, with most of the projects related to large scale channel restoration efforts in both watersheds. All activities completed during the contract period June 1, 2010 through May 31, 2011 are summarized in *Table 14* followed by a more detailed site characterization and summary of activities for individual treatments. In several locations, multiple treatments have been implemented to meet the objectives for larger sites. These treatments are grouped under the same project ID heading so that the interrelationship of activities is more apparent.

A brief explanation of the project ID that is used in the summary table and in the detailed descriptions is warranted here. The project ID is an alphanumeric code that corresponds to the location of individual treatments in relation to the river-mile of the drainage network for the watersheds of interest. The first digit of the code signifies the watershed that the treatment is located in, using the first letter in the watershed name (e.g., B=Benewah Creek, E=Evans Creek, etc.). The series of numbers that follow correspond to the river-mile location (in miles and 10^{ths}) at the downstream end of treatment sites. River mile is tabulated in an upstream direction from mouth to headwaters and treatments that are located in tributary systems have river mile designations separated by a forward slash (/). For example, the downstream end of project L_8.2/0.7 is located in the Lake Creek watershed 0.7 miles up on a tributary that has its confluence with the mainstem 8.2 miles from the mouth. This nomenclature is intended to indicate the spatial relationship of treatments to the mainstem and tributary aquatic habitats having significance to the target species. Furthermore, it readily conveys information about the relationship of multiple treatments by indicating the distance to common points in the drainage network.

Table 14. Summary of restoration/enhancement activities and associated metrics completed for BPA Project #199004400.

Project Description			**Project Chronology**			
Project ID	**Activity**	**Treatments (Metrics)**	**Pre-2007**	**2008**	**2009**	**2010**
B_9.7 (page 73)	Stream Channel Construction	Channel construction (1,267 m); habitat enhancement (416 m)		Developed restoration design for 2.4 km of mainstem habitats (Reach D-2)	Constructed/enhanced 810 m of stream channel; installed 7 instream wood structures (156 m)	Regraded and activated 457 m of side-channel habitats; installed 7 instream wood structures (260 m)
B_9.7	Plant Vegetation	Streambank stabilization (0.78 ha, 969 m of streambank			Planted 14,904 herbaceous plugs and 6,950 deciduous trees (0.78 ha of floodplain, 969 m of streambank)	
B_9.7 (page 77)	Plant Vegetation	Riparian enhancement (54.03 ha; 5,331 m of streambank)	Planted 49,068 conifers (46.3 ha of floodplain, 3689 meters of stream bank)	Planted 2,100 conifers (1.86 ha of floodplain)	Planted 10,058 herbaceous plugs, 4,634 deciduous trees, 3,800 conifers (3.31 ha of floodplain, 742 m of streambank)	Planted 27,957 herbaceous plugs, 6,494 deciduous trees, 50 conifers (2.56 ha of floodplain, 900 m of streambank)
L_8.2/0.7 (page 80)	Stream Channel Construction	Channel construction (624 m, 2.56 ha of floodplain wetlands)		Developed restoration design for 1.2 km of tributary habitats in WF Lake Creek.	Signed landowner contract. Constructed 106 m of new channel, created 2 ha of new floodplain, instsalled 8 instream structures	Constructed 518 m of new channel, created 0.56 ha of new floodplain, installed 33 instream structures
L_8.2/0.7 (page 83)	Plant Vegetation	Riparian enhancement (0.4 ha; 1,431 m of streambank)			Planted 800 conifers, 300 herbaceous plugs, 450 deciduous trees (212 m of streambank)	Planted 14,663 herbaceous plugs, 3,670 deciduous trees (0.4 ha floodplain, 1219 m of streambank)

4.1.1 Project B_9.7: Instream/Channel Construction for the ʼEłumish Project

Project Location:

Watershed: Benewah	Legal: T45N, R4W, S13 NE ¼ SE ¼
Sub Basin (River Kilometer): 15.6 rkm	Lat: 47.241292N Long: 116.771454W

Site Characteristics:

Slope/Valley gradient: 0.7%	Aspect: N	Elevations: 830 m
Valley/Channel type: B2/C4 E4	Proximity to water: In channel	

Other: *Project implements second year actions identified in the Reach D2 restoration design, including: regrading and activation of 457 m of side-channel habitat (element D2-3); and construction of seven in-channel wood structures affecting 260 m of mainstem habitat (element D2-4).*

Problem Description: Historically, the Benewah Creek valley was a mosaic of open stands of conifers, wet meadows and stream corridor riparian forest (Mikkelsen and Vitale 2006). Forest composition and structure was maintained by frequent fires. A compositionally diverse, coniferous dominated forest was likely distributed along complex gradients of elevation, aspect and site water balance. Historically, frequent engagement of flood flows on the valley floor was most likely in response to both (i) blockage effects of large wood pieces falling into the channel and aggregating smaller wood, and (ii) beaver dams, with local gravel and fine sediment accumulations upstream. Whenever the channel did avulse in response to blockages, it likely did so through rapid down-cutting through the easily eroded loess layer, reaching a base gravel layer in the valley relatively quickly and then remaining at the grade defined by that layer. Following a more recent history of intensive logging, forest clearing, beaver trapping, and grazing, the hydraulic influence of local beaver dam/sediment accumulation was reduced or removed. The stream banks were more susceptible to unraveling and channel widening, leading to the state seen at some locations where a new, lower elevation alluvial floodplain appears to have established between the upper bank surfaces defined by the valley floor. Hydraulic analysis of representative channel cross-sections show the overall level of channel incision/containment is approximately equivalent to the capacity of a 5-year return interval peak flow event with some areas exhibiting a capacity that approaches the 10-year peak flow.

The significantly reduced access of flood flows to the former floodplain and broader valley bottom has affected wetland habitats on a large scale and accelerated streambank erosion. Several avulsion channels and to a lesser extent, remnant historical channels have left portions of the valley bottom with some wetland habitat, however, it appears that shallow groundwater tables have been lowered and recharge of wetlands by overbank flows has been greatly reduced. Many of the remaining wetland areas are only marginal in size and a band of xeric vegetation of variable width is located along the channel margin throughout the project reach. The most recent estimates of stream bank erosion indicate that erosion rates approach 476±208 metric tons/yr/km. When extrapolated to the larger reach located between river kilometer 14.3 and 19.1, total annual sediment yield from streambanks ranges from 1286-3283 metric tons/yr.

This stream reach is located in a portion of the watershed that historically provided important summer and winter rearing habitats for westslope cutthroat trout. Existing conditions currently support low densities of cutthroat trout (<2 fish/100 sq. m). Lack of habitat diversity, localized loss of low gradient channel segments, reduced infiltration of water from adjacent wetlands, and elevated water temperatures are all factors that limit the productivity of these reaches.

Description of Treatment: Several design elements for the D2 reach were implemented during this second year of construction to address the findings and specific needs identified in the problem assessment:

Element D2-3. Regrading and partial excavation for 457 m of an existing relict channel was completed with the channel reconnected at its upstream junction with the main channel, just upstream of the confluence with Windfall Creek. Currently, water is backed up at this location because of rock grade controls that were constructed to inundate the culvert and establish fish passage into Windfall Creek in 2004. This local grade control appears to have resulted in local elevation of the water table such that the relict channel now has standing water in it during summer months. The work involved constructing an inlet control structure of wood and rock that meters water into the side channel, while the majority of flood flows continue to be conveyed by the main channel. Two flow choke structures (D2-4I and D2-4J) were constructed in the side channel (see detailed description below for element D2-4). The channel was graded to flow at the summer low flow level, and sufficiently deep to preclude near term establishment of reed canary grass on the bottom until the riparian corridor is re-established. Presumably, the connected side channel will provide improved opportunities for salmonid rearing particularly during the winter.

Element D2-4. A total of seven in-channel wood structures were constructed, which emulate flow obstruction effects of natural wood jams and beaver dams. One of these structures utilized a passive approach by placing 2-4 large logs in the channel, with key pieces anchored in the bed and banks, to provide a key framework that beavers could use in dam construction and which serves as a natural analog that approximates historical, wood recruitment processes. This approach was based on observations that the most persistent, existing dams throughout the Benewah Creek stream corridor are built with mountain alder integrated with remnant in-channel large wood. MacCracken and Lebovitz (2005) found that this technique can work when the channel is unconfined with a wide floodplain, there are no logjams nearby, and when deep pools and banks suitable for beaver dens are nearby. Individual logs are placed across the channel bottom at riffle crest locations, and wedged between small boles driven vertically into the substrate. Fresh black cottonwood and aspen saplings may also be placed along the stream banks above the log structures to encourage beavers to finish the dam construction (Muller-Schwarze and Sun 2003).

The remaining six structures were engineered “flow choke structures” in which the concept was to create increased backwater effects during floods such that the valley floor would become connected annually. The structures also affect approximately 260 m of mainstem habitats by increasing residual pool depth and volume at base flow conditions. The concept involves two types of flow and thus upstream water surface elevation controls (Figure 25):

1. Weir flow over a horizontal cross-log, with sufficient depth to permit passage of floating debris at the bankfull level (2 structures built with this configuration); and

2. A combination of weir flow over a horizontal cross-log as well as orifice flow under the log, with both lateral and vertical constriction throttling down the flow past the structure (2 structures built with this configuration).

To implement the design concept, construction involved:

1. Placement of a horizontal cross-log that acts as a control weir at flood flows. The bottom elevation of the orifice was designed to emulate general low flow control elevations formed by numerous beaver dams present in the reach, where median depths were 0.36 m at the riffle crest and 0.97 m below the floodplain; these served as natural process-based design criteria for situating the orifice control elevation and the depth of impounded gravel upstream. An additional horizontal log was buried beneath the weir at a depth that exceeded the estimated scour depth for each site.
2. A series of horizontal cross-logs protruding from each stream bank that project a blocked area in the downstream direction leaving a central orifice area for lower flows to pass through.
3. A pad of rock placed at the downstream end of the structure as a scour countermeasure, to protect the integrity of the structure.
4. A deposit of finer gravel, sized to be comparable to stones occurring naturally in the river banks and bed, placed on the bed of the upstream side of the structure to facilitate smoother streamlines and potentially provided trout spawning habitat.
5. Laid back stream banks within the upstream and downstream footprints of the structure to prevent saturated bank collapse, avulsion, and loss of structure integrity. A maximum graded slope of 1.5H:1V was specified here as an initial approximation to reduce the amount of excavation on either side of the structure while maintaining a saturated slope stability safety factor above 3. The laid back banks were re-vegetated with herbaceous plants.

Figure 25. Engineered "flow choke structures" constructed in Benewah Creek illustrating two variations of flow type and surface elevation controls, including 1) weir flow over a horizontal cross-log (left), and 2) a combination of weir flow and orifice flow under the horizontal log (right). These structures were built in the active mainstem channel of Benewah Creek in locations where natural beaver dams had been previously surveyed.

Additionally, as part of this design element approximately 24 cubic meters of wood (40 20-33 ft. long logs) was added to the stream channel and near bank region within a 200 meter reach to aid beavers in dam construction and to increase wood loading to approximate a target volume of 6 m^3/100 m for mainstem and tributary habitats in the watershed. Furthermore, two natural beaver dams were reinforced with vertical uprights that were installed through the face of the dam at 3-5 ft. intervals using an excavator. The premise is that these "reinforced" natural dams should be more persistent during high flows and facilitate channel/floodplain connectivity over a longer, contiguous reach.

The three approaches to channel wood additions and beaver dam augmentation that were implemented as part of this design element allows for more frequent and extensive floodplain connection during annual floods, seeks to increase the stability of natural dam complexes, and is a natural analog alternative to large scale riffle construction that helps maintains connectivity with cooler groundwater during summer months.

Project Timeline: Coordination with the landowners in the area began in May 2008. A field survey of the site, including wetland delineation, was completed in October 2008. Two design alternatives were developed initially and the preferred site design was finalized in May 2009. The initial restoration work was completed from June through August 2009. Implementation is planned through the summer of 2012 to complete all of the design elements that were identified.

Project Goals & Objectives: Goals for this project include 1) create wetland habitats and increase the hydraulic connections with the valley bottom; 2) reduce bank erosion 3) provide a long-term source of large woody debris for natural recruitment; and 4) provide measurable increase in abundance and distribution of westslope cutthroat trout.

Relationship to Scope of Work: This project fulfills the Program commitments for WE F in the 2010 Scope of Work and Budget Request (Contract #49932) for the contract period June 1, 2010 through May 31, 2011.

4.1.2 Project B_9.7: Riparian/Planting

Project Location:

Watershed: Benewah	Legal: T45N, R4W, S13 NE ¼ SE ¼
Sub Basin (River Kilometer): 15.6 rkm	Lat: 47.241292N Long: 116.771454W

Site Characteristics:

Slope/Valley gradient: 0.7%	Aspect: N	Elevations: 830 m
Valley/Channel type: B2/C4 E4	Proximity to water: Floodplain	

Other: *Project treats 2.56 hectares of floodplain and off-channel wetlands and 900 m of streambank associated with side-channel habitat.*

Problem Description: Historically, the Benewah Creek valley was a mosaic of open stands of conifers, wet meadows and stream corridor riparian forest (Mikkelesen and Vitale 2006). Forest composition and structure was maintained by frequent fires. A compositionally diverse, coniferous dominated forest was likely distributed along complex gradients of elevation, aspect and site water balance. Tree species likely included: ponderosa pine, western white pine, western larch, Douglas fir, lodgepole pine, grand fir, western red cedar, Engelmann spruce, aspen and black cottonwood. Historic land use since European contact, including valley-wide forest removal, beaver trapping, in-channel large wood removal, construction of splash dams, timber mill operations, pasture grass management and 70+ years of extensive cattle grazing, has resulted in a radically altered valley ecosystem with eroding stream banks and a plant community dominated by invasive forbs, grasses and woody species unpalatable to cattle. Given the extreme perturbation of stream channel and forest structure and processes, the goal of the ecological restoration of the riparian forest and wetland ecosystem is to steer the system toward recovery using both ecological engineering and restoration forestry.

Description of Treatment: A primary strategy being utilized for the Benewah Creek restoration is the utilization of black cottonwood's unique life history characteristics to rapidly "flip" or change the current degraded riparian ecosystem into a diverse self-sustaining riparian forest. Although black cottonwood's regenerative strategy (seedling establishment on bare alluvial substrates and branch fragment vegetative propagules) likely resulted in it historically playing a non-dominant role in the riparian forest, its life history characteristics make it ideal for rapidly establishing a complex riparian forest. Establishment of a cottonwood forest along the Benewah Creek floodplain and stream banks will provide exceptional hydrologic, biogeochemical and plant and animal habitat functional lift within 5-10 years as well as control the trajectory of ecosystem development over next 100+ years.

Hydrologically, dense plantings of cottonwood will supply local beaver populations with ample dam building materials resulting in local backwater flooding of adjacent wetlands. These hydrologically restored areas will support a diverse emergent, scrub-shrub and forested wetland plant community. Additionally, other hydrologic functions will be enhanced (per Jankovsky-Jones 1999) including: dynamic water storage; energy dissipation; and long-term surface water storage. Enhanced biogeochemical functions (also per Jankovsky-Jones 1999) will include the ability of the wetland to contribute to local or regional water quality by the removal of imported nutrients, contaminants, and other elements or compounds. Given the active use of private lands for cattle and horse pasture, enhanced beaver dam construction will significantly support wetland sediment and nutrient retention and removal functioning.

An established cottonwood forest will rapidly enhance plant community functions through the maintenance of a characteristic native plant community in terms of species composition and physical characteristics of living plant biomass, and of detrital biomass in terms of the production, accumulation and dispersal of dead plant biomass of all sizes (Jankovsky-Jones 1999). The planting restoration design calls for establishing a matrix of floodplain cottonwood interplanted with understory cedar and Engelmann spruce. Cottonwood will establish a closed canopy within about 5 years and act as nursery cover for establishing understory conifers. Cottonwood break-up will occur at about 60-90 years, relinquishing understory conifers to a dominant canopy position. This technique has been used successfully with cottonwood and western red cedar in trials in British Columbia (Peterson et al. 1996). The establishment of an interior forest micro-climate following canopy closure will support the development of native understory riparian plant community.

The cottonwood forest will provide significant enhancement of fish and wildlife habitat throughout the Benewah Creek valley as well as the riparian ecosystem. Specifically, the new riparian forest will provide for maintenance of habitat interspersion and connectivity, reflecting the capacity of a wetland to permit aquatic organisms to enter and leave the wetland via permanent or ephemeral surface channels, overbank flow, or unconfined hyporheic grave aquifers, and access of terrestrial or aerial organisms to contiguous areas of food and cover (Jankovsky-Jones 1999). The forest will support enhanced fish habitat through stream shading, allochthonous input of fine, coarse and organic carbon to the aquatic ecosystem, and input of large wood structures in the stream. Vertical and horizontal forest structural elements will maintain bird and mammal habitat throughout the riparian corridor. Cottonwood will also provide dead snags for cavity nesting birds and mammals within about 50 years.

A total of 27,957 herbaceous plugs and 2,560 woody trees and shrubs were planted in fall 2010 along nearly 900 meters of streambanks and 1.16 hectares of associated floodplain that was disturbed during construction. In addition, all floodplain surfaces and the temporary roads used to access the site were hand seeded and mulched with herbaceous grasses applied at a rate of 48 kg/ha. In the spring of 2011 an additional 200 live willow poles were planted to complete the vegetation treatments on these sites. Plant species included eleven species of woody trees and shrubs, ten species of herbaceous sedges (*Carex sp. and Scirpus sp.*) and rushes (*Juncus sp.*), and six species of herbaceous grasses. Several existing wetland swales and groundwater fed wetlands covering approximately 1.4 hectares were also planted to establish nursery areas for propagation of black cottonwood and willows and to provide forage and dam building materials for beaver. In much of these areas, invasive reed canarygrass (*Phalaris arundiacea*) that had become established was mechanically scraped from planting areas prior to treatment. These wetlands have favorable hydrologic conditions for growing and propagation of black cottonwood and willows and these conditions have been further enhanced by more frequent overbank flows attributed to in-channel structures and obstructions that have been installed recently (See *4.1.1 Project B_9.7: Instream/Channel Construction, Element D2-4*). Approximately 1,260 containerized aspen, cottonwood and willow (sp.) saplings were planted in fall 2010, and an additional 2,474 cottonwood and willow cuttings were planted in the same areas in spring 2011. Survival to date appears to be quite good (Figure 26).

Figure 26. New growth arising from cottonwood and willow (sp.) cuttings planted adjacent to side-channel habitat that was reactivated in the summer of 2010.

Project Timeline: Two design alternatives were developed initially and the preferred site design and vegetation plan was finalized in May 2009. Annual plantings will be completed in the fall and the spring of each year between 2009-2012. Annual and periodic inspections will be completed to evaluate survival and growth and determine if restocking of planting sites is warranted.

Project Goals & Objectives: Reestablish a patchwork of native vegetation communities on approximately 25 acres of the valley floor to lay the foundation for a compositionally and structurally diverse forest ecosystem to develop over the next 25-50 years. Achieve minimum stocking densities of 197 trees/hectare and provide for significant increases in canopy density and overhanging vegetation over a 20 year timeframe.

Relationship to Scope of Work: This project fulfills the Program commitments for WE G and H in the 2010 Scope of Work and Budget Request (Contract #49932) for the contract period June 1, 2010 through May 31, 2011.

4.1.3 Project Lake 8.2/0.7: Instream/Channel Construction for the *Hnmulshench* Project

Project Location:

Watershed: Lake Creek	Legal: 24N, 45E, S36 E ½ of SE ¼
Sub Basin (River Kilometer): 13.1/1.1	Lat: 47.526627 N Long: 117.048639 W

Site Characteristics:

Slope/gradient: 0.6%	Aspect: N	Elevations: 792 m
Valley/Channel type: C4/C5	Proximity to water: In-channel and floodplain	

Other: *Project implements second year actions identified in the Hnmulshench restoration design, including: construction of 518 m of new channel to final grade (an additional 152 m partially constructed);construction of 33 in-channel wood structures within the new channel; and re-grading of a field to create 0.56 ha of new floodplain.*

Problem Description: The lower reaches of the West Fork contain an important stream corridor linking the headwaters to the mainstem of Lake Creek. Currently, there is limited production potential for cutthroat trout within the reach due to channel incision, fine sediment, increased stream temperatures, lack of cover, and lack of large woody debris. Fish population data has been collected for the watershed since 1996. This section of the West Fork of Lake Creek had an average westslope cutthroat trout density from 2002-2008 of 1.1 fish/100 square m while fish densities further upstream were greater than 20 fish/100 square m.

This stream rehabilitation project includes about 805 m of WF Lake Creek and 305 m of an unnamed tributary. Both streams exhibit many of the classic signs of impairment caused by channel ditching and straightening. WF Lake Creek (WFLC) is deeply entrenched as a result of incision of the streambed as a series of head-cuts migrated upstream through the reach. Historic head-cuts have already moved upstream through the project, and three additional head-cuts were identified within the reach. These existing headcuts imply that the incision trend is expected to continue as the head-cuts progress upstream. There is exposed bedrock 91 m upstream of the site preventing further incision above that point. The unnamed seasonal tributary intersects WFLC at approximately mid way up the project reach. This tributary channel is also deeply incised and two head-cuts were observed. Bank erosion and bankslope failures have been ongoing since initial incision occurred in both WFLC and the tributary. Several bank erosion sites were observed and streambanks will likely continue to fail. Bank erosion rates on WFLC were estimated to be 8.07 metric tons/year upstream of a stream crossing and 28.24 metric tons/year downstream of a stream crossing. Streambank vegetation is generally reed canary grass and Mountain Alder. The historic floodplain, where hay is produced, is perched and rarely accessed by flooding. There are 1.1 hectares of wetlands on the property.

Although these erosion processes negatively influence short-term sediment loading, vegetation establishment, and aesthetic, they are the natural processes by which an incised stream can eventually recover over the long term. Through erosion and sediment transport processes (of the streambed initially, and then streambanks and terraces) over several decades the channel will gradually create a new inset floodplain and riparian habitat at the lower level, terraced several feet below the existing valley bottom. Currently, the channel at the project reach is underway in this recovery process, but at different stages of development through the reach. In some channel segments the new inset floodplain width approaches 12 m while in other segments, width is less than 4.5 m. It is expected to continue to erode downward and laterally until a new floodplain forms that has enough width to allow floods to spread out and when vegetation can become established, to resist the rapid erosion processes that are currently underway.

Description of Design: The design developed for this project calls for filling 610 m of the existing incised West Fork Lake Creek channel and diverting flows into a newly constructed, 922 m long channel that is well connected with the valley bottom to allow dissipation of flood flows over a broad floodplain. Upstream of the newly constructed channel, imported wood will be placed in the existing channel to create habitat. A seasonal stream will be partially filled to repair the degradation that has occurred and will be extended to the newly built WF Lake Creek stream channel. Native plants will be planted in riparian and adjacent upland areas. Large wood material will be used throughout the project to increase lateral roughness where needed, create banks and maintain planform until hydric plant communities become fully established. Construction will increase the stream length by more than 50 percent and 3.64 ha of wetlands will be created through this project (0.33 ha will be filled).

The following construction phases were the focus of restoration work in summer-fall 2010:

Phase 1A- Floodplain Grading: New floodplain was created along the southwest side of the valley by the access road for 0.56 ha of the project. Temporary stockpiles of topsoil and general fill were created. These areas were seeded and mulched in November 2010. The berms around the existing irrigation ponds were reshaped and decreased in height by 0.9 m. Excess material was stockpiled to be used as channel fill for Phase 2B. All constructed and disturbed ground was revegetated with native plants.

Phase 1B- New Channel Grading (Figure 27). New channel grading involved creating a new channel excavated into the new floodplain surface to channel subgrade depth. The subgrade was 5.5 m wide downstream of the new confluence with the seasonal tributary and 5.2 m wide upstream of the new confluence. Bankfull width for riffles was 3.7 m downstream and 3.4 m feet upstream of the confluence. New channel habitat was constructed over the channel subgrade by using imported gravels and logs to create streambed and streambanks. Rock was placed in the channel combined with logs to form riffles and pools. Logs were placed on the new floodplain to provide erosion protection and will be anchored or buried. Fill was placed in temporary stockpile areas. In 2010, a total of 305 m of channel subgrade was excavated. A total of 518 m of channel was constructed to final grade and 152 m of channel was partially completed. Sections of floodplain in this area were re-graded after the channel work was complete.

Project Goals & Objectives:
Goals for this project include 1) create wetland habitats and hydraulic connections with the valley bottom; 2) reduce bank erosion 3) provide a long-term source of large woody debris for natural recruitment; and 4) provide measurable increase in abundance of westslope cutthroat trout.

Figure 27. Channel construction for the WF Lake Creek Hnmulshench project proceeded in two stages: excavation to subgrade (left), then refilling the channel with rock and wood to achieve the final design dimensions (right). The bedrock outcroppings seen in the foreground were incorporated into the new channel.

<u>Project Timeline</u>: The site design was finalized in May 2009. All NEPA work was completed by August 2009. Construction occurred between August-October 2009 and July-August 2010. Restoration work is to be completed over three years ending in October 2011.

<u>Relationship to Scope of Work</u>: This project fulfills the Program commitments for WE I in the 2010 Scope of Work and Budget Request (Contract #49932) for the contract period June 1, 2010 through May 31, 2011.

4.1.4 Project Lake 8.2/0.7: Riparian/Planting

Project Location:

Watershed: Lake Creek	Legal: 24N, 45E, S36 E ½ of SE ¼
Sub Basin (River Kilometer): 13.1/1.1	Lat: 47.526627 N Long: 117.048639 W

Site Characteristics:

Slope/gradient: 0.6	Aspect: N	Elevations: 2600
Valley/Channel type: C4/C5	Proximity to water: In-stream and adjacent floodplain	

Other: *Project specifically treats 3.2 ha of ground disturbed during stream channel construction in 2010. It includes planting 0.4 ha of new floodplain adjacent to the stream channel with permanent vegetation.*

Problem Description: Restoration of the West Fork of Lake Creek is underway to restore stable channel pattern and geometry by creating 944 m of new stream channel in the historic valley. In 2010, 3.2 ha of ground were disturbed through construction activities. This area will require rapid establishment of woody and herbaceous species to support the short- and long-term stability of the site.

Current wetland function is degraded in the entrenched West Fork of Lake Creek channel as a result of the processes of channel incision that has occurred since before the 1930's. Based on local site conditions and conditions in reference wetlands in other nearby watersheds, it is evident that both groundwater and periodic overbank flooding once provided much of the hydrology to maintain wetlands in the project area. A band of xeric vegetation of variable width is located along the channel margin throughout the incised reach. A series of springs that historically connected to the historic channel are now feeding an irrigation pond.

Description of Treatment: A vegetation plan was developed for the site based on inventories of native wetland plant species conducted during wetland delineations and functional assessments on the project site and at a control site in the watershed. Planting activities are described in the West Fork Lake Creek Restoration Planting Plan and in the Stormwater Pollution Prevention Plan (SWPPP) for construction activities. The plan identifies a mix of 27 native species to be planted on the site, delineates planting areas based on key environmental gradients, and provides material specifications and planting densities. Plant species include seven species of woody trees and shrubs, 10 species of herbaceous sedges (*Carex sp. and Scirpus sp.*) and rushes (*Juncus sp.*), and 10 species of herbaceous grasses.

A total of 14,663 herbaceous plugs and 3,670 woody plants were planted along 1219 m of newly built stream bank and 0.4 ha of floodplain created in 2009. In addition, all disturbed areas and temporary stockpiles were hand seeded and mulched with herbaceous grasses applied at a rate of 48 kg/ha to provide for site stabilization consistent with the SWPPP.

Project Timeline: The site design was finalized in May 2009. All NEPA work was completed by August 2009. Construction for 2010 occurred between July and August. Woody plants and herbaceous plugs were planted in September 2010. Seeding and mulching occurred in October 2010. Restoration work is to be completed over three years ending in October 2011. Annual and periodic inspections will be completed to evaluate survival and growth and determine if restocking of planting sites is warranted.

Project Goals & Objectives: Goals for this project include 1) create wetland habitats and hydraulic connections with the valley bottom; 2) reduce bank erosion 3) provide a long-term source of large woody debris for natural recruitment; and 4) provide measurable increase in abundance of westslope cutthroat trout. Success criteria include: establish at least 80% herbaceous cover by native species at the end of 2 years following site disturbance.

Relationship to Scope of Work: This project fulfills the Program commitments for WE J in the 2010 Scope of Work and Budget Request (Contract #49932) for the contract period June 1, 2010 through May 31, 2011.

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