Coeur d'Alene Tribe
Fisheries Program

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

2008 ANNUAL REPORT

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

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

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 both resilience and resistance 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 2008 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 the 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 2008. 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.

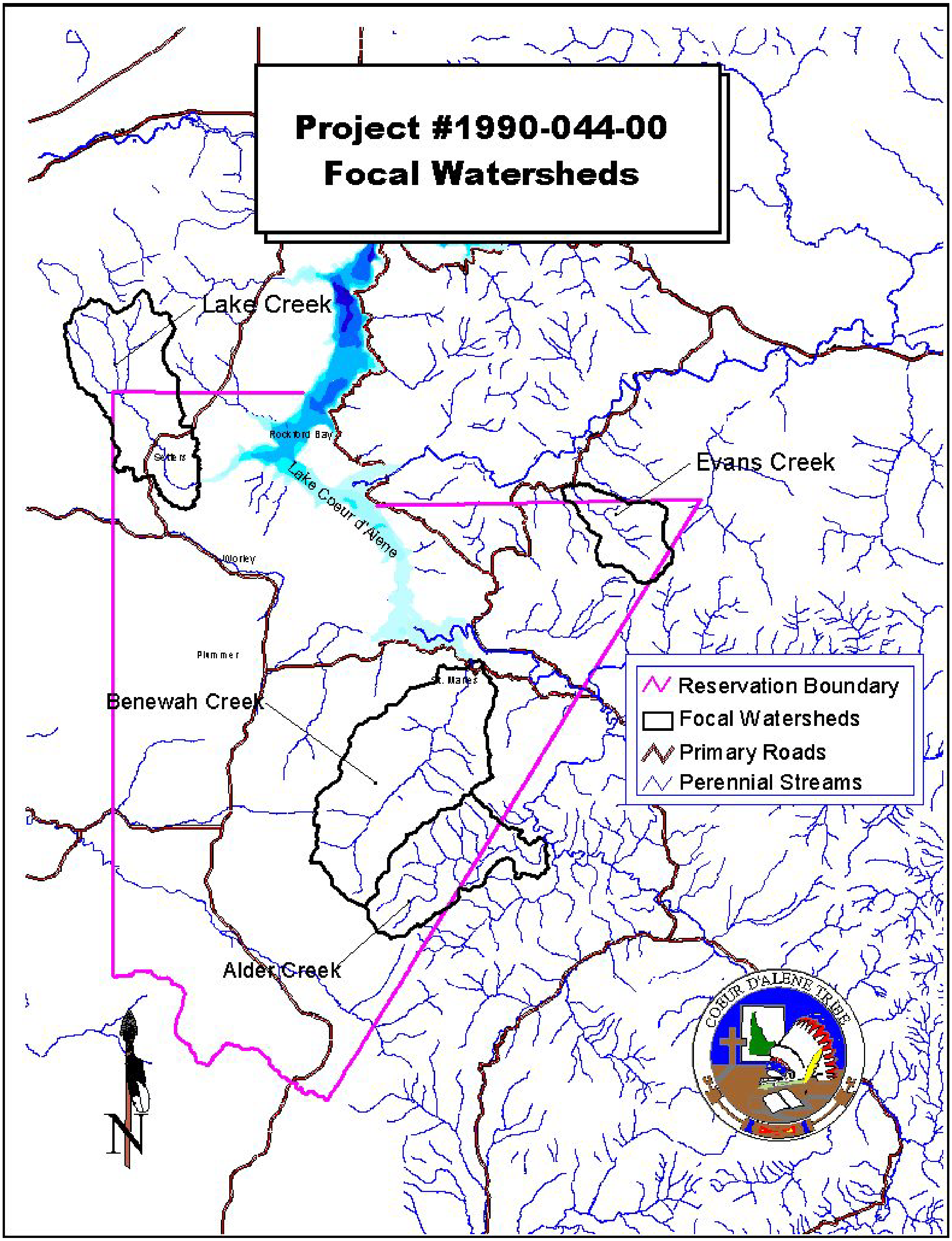

Figure 1. Locations of BPA Project 90-044-00 Focal Watersheds on the Coeur d'Alene Indian Reservation.

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 changes in status 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 number of juvenile outmigrants and returning adults in watersheds that support the adfluvial life-history. 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 to address factors limiting cutthroat trout recovery. We are monitoring the response of salmonids and physico-chemical habitat features to action implementation by measuring indicator variables in both treated and control reaches or watersheds. Effectiveness monitoring is currently being conducted in the upper Benewah watershed to evaluate responses to large-scale channel construction activities and non-native brook trout control.

From 2005 to 2007, 1874 m of mainstem channel habitat was reconstructed in the upper Benewah watershed upriver of 9-mile bridge to address dysfunctional stream processes and structure, including 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, 2008; Firehammer et al. 2009). 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. Channel reconstruction during these three years 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 and future woody debris recruitment. Monitoring the biological response to these enhancement actions included examining changes in trout abundances before and after habitat restoration in treated reaches relative to control reaches. Temperature responses were monitored by examining changes in the availability of thermal refugia in pool habitats before and after restoration. Physical responses to mainstem restoration were monitored by examining changes in large woody debris volume, substrate composition, canopy cover, and residual pool depth and volume.

A brook trout removal program was initiated in 2004 to suppress the numbers of brook trout found in mainstem 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 preventative re-colonization measures, such as passage barriers (Shepard et al. 2003), we use less intrusive 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 brook trout abundances estimated at our index sites in the upper Benewah watershed relative to those monitored at index sites in our control watershed, Alder Creek. In addition, we also track changes in maturation metrics in residual brook trout (e.g., fecundity-at-length, size-at-maturation) that would signal a reproductive compensatory response to our efforts.

The objectives of the monitoring and evaluation section with corresponding BPA Pisces scope of work 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 (Work Elements N,O,P,Q,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 (Work Elements R,U)

2) Collect and summarize longitudinal trends in water temperatures by deploying loggers within monitored watersheds (Work Elements W,Z)

3) Evaluate effectiveness of habitat restoration in the upper Benewah watershed
    a) Assess differences in trout abundance between restored treatment sites and unrestored control sites in mainstem reaches (Work Element U)
    b) Assess differences in thermal heterogeneity in pool habitats in treated mainstem reaches before and after restoration (Work Elements X,Z)
    c) Assess differences in physical habitat indicators measured at treatment and control sites (Work Elements V,Z)

4) Reduce the abundance and distribution of non-native brook trout in the upper Benewah watershed
    a) Remove brook trout from Benewah Creek (Work Element L)
    b) Test the effectiveness of the removal program by comparing brook trout abundances before program implementation to those evaluated in 2008 in both treated and control watersheds (Work Element M)

c) Examine compensatory responses in reproductive metrics in brook trout (Work Elements L,M)

3.2 Methods

3.2.1 Trend and status monitoring

3.2.1.1 Adfluvial cutthroat trout migration

Migration traps were installed in Lake Creek in November of 2007 and in Benewah Creek in February of 2008 to collect abundance and life-history information on adfluvial cutthroat trout. Resistance board weir (RBW) traps (Tobin 1994; Stewart 2002) were used in both watersheds to intercept adult cutthroat migrating upriver. This style of migrant trap has proven successful in capturing adult fish in past years during periods of heavy spring discharge. A modified fence-weir design was used in both watersheds as the downriver trap (DN) to capture post-spawn adults and outmigrating juveniles. The design incorporated pop-out panels that could be removed during periods of high flow to relieve pressure on the trap. Downriver traps were installed in the spring in both systems as early as possible under amenable discharge levels. The RBW trap on the Benewah Creek mainstem was installed at river kilometer (rkm) 14.5, with the DN trap located immediately upstream (Figure 3); the RBW trap on the Lake Creek mainstem was installed at rkm 6.0, with the DN trap located approximately 0.13 km upriver (Figure 5). In both watersheds, traps were positioned downriver of principal spawning tributaries and of most of the recently implemented and projected habitat restoration projects. 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 RBW 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. Logged detection data were downloaded several times a week to monitor both adult and juvenile fish passage throughout the migratory period. Only the system in Lake Creek was operational during 2008.

Lengths, weights, and scales were collected and condition factors (estimated as 10,000*Wt / TL^3) calculated from all captured adult cutthroat trout. Adults were also scanned for the presence of PIT-tags using a hand-held wand. In addition, all adults captured in the RBW traps received a hole punch in the upper lobe of the caudal fin. Recapture of these ‘tagged’ adults in the DN traps permitted an estimation of the abundance of upriver migrating adults, following Chapman’s (1951) modification of the Petersen index:

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

where:

N= the abundance estimate;

M= number of ‘tagged’ adults that received a caudal punch;

C= number of adults captured in the DN trap; and
R= number of adults captured in the DN trap that had been caudal punched.

The variance estimate of N was calculated as follows:

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

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 DN traps. In addition, at least 30% 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 DN 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):

$$U_h=\frac{(u_h)(M_h+1)}{m_h+1},$$

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:

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

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 considerably for tagged fish released above the DN 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). Tagged fish were considered available for recapture if during the trial period they were either trapped or detected by the PIT-tag antenna array (this assumes that all tagged fish that bypassed the trap without being recaptured were detected by the array). Because the PIT-tag system was not operational in Benewah creek, 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 Summer trout abundance 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 and September to quantify the abundance and distribution of salmonids 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. Typically, three passes were conducted at a site. However, at some sites catch was not adequately reduced with subsequent passes and consequently four passes were required to permit the calculation of an abundance estimate. At other sites, time constraints or habitat conditions only afforded two passes, though catch was adequately reduced to permit estimation of abundance.

Captured salmonids, including westslope cutthroat trout and brook trout *(Salvelinus fontinalis)* were identified, enumerated, and measured for total length (TL, mm). Weights (Wt, g) 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 keys derived from previously collected scale samples, cutthroat and brook trout respectively greater than 70 and 75 mm were considered to be at least one year of age. 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 initial electrofishing pass. For each site, 10 representative channel widths were collected to permit estimation of site area for fish density calculations.

Index site abundances were estimated 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 using the removal-depletion method (Zippen 1958; Seber and LeCren 1967). For sites in which only two passes were conducted, site estimates were calculated using the following equation (Armour et al. 1983):

$$N = \frac{U_1}{1-(U_2/U_1)},$$

where:

N = estimated population size;
U_1 = number of fish collected in the first pass; and
U_2 = number of fish collected in the second pass.

The standard error of the estimate ($se(N)$) was calculated as:

$$se(N) = \sqrt{\frac{M(1-M/N)}{A-\left[(2p)^2(U_2/U_1)\right]}},$$

where:

M = $U_1 + U_2$;
A = $(M/N)^2$; and
p = $1-\frac{U_2}{U_1}$.

Site estimates for three or four pass removals were calculated using the following equation (Armour et al. 1983):

$$N = \frac{M}{1-(1-p)^t},$$

where:

N = estimated population size;
M = sum of all removals ($U_1 + U_2 + \ldots.U_t$);
t = the number of removal occasions;
U_i = the number of fish in the i^{th} removal pass;
C = $(1)U_1 + (2)U_2 + (3)U_3 + \ldots..(t)U_t$;
R = $(C-M)/M$;
p = $(a_0)1 + (a_1)R + (a_2)R^2 + (a_3)R^3 + (a_4)R^4$; and
a_i = Polynomial coefficient from Table 8 (Armour et al. 1983).

The standard error of the estimate ($se(N)$) was calculated as:

$$se(N) = \sqrt{\frac{N(N-M)M}{M^2 - \frac{N(N-M)(tp)^2}{(1-p)}}}$$

The approximate 95% confidence interval for each abundance estimate was then calculated as follows:

$$95\%CI = N \pm 1.96 * se(N)$$

In some cases, few fish were captured during each subsequent pass but numbers were not adequately reduced to reliably generate an estimate. We determined that if the total fish captured was ten or less over three passes, the estimated abundance would be considered the total number of fish caught without an accompanying confidence interval.

To facilitate a simple short-term trend evaluation, site-specific slope estimates were generated from abundance data collected since 2004. Slopes were estimated for periods ranging from 2004 to 2008 (or from 2004 to 2007 if abundance data were not available in 2008) for each index site.

Slope estimates for each period were then assigned to one of seven qualitative categories to describe the strength of the directional trend: less than –5, between –5 and –2, between –2 and –0.5, between –0.5 and 0.5 (indicating stable abundance), between 0.5 and 2, between 2 and 5, and greater than 5. Given the low number of data points (i.e., 3 or 4) used in these computations, the intent of this procedure was not to generate statistically robust slope estimates. Rather, the intent was to produce an easily obtainable index that could be used to illustrate similarities and differences in trends among sites that could be displayed in figures or tables.

3.2.1.3 Longitudinal stream temperatures

Stream temperatures were continuously monitored every 15-20 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 both a forested and open meadow site in upper Benewah and Lake creek watersheds. Daily mean and maximum 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 the 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.

3.2.1.4 Physical habitat features

We used measurements of physical habitat attributes that have been collected since 2004 to examine the potential to detect regional trends in our watersheds and to determine achievement of target objectives. Analyses were conducted for both tributary and mainstem habitats given that we had repeated measurements at several mainstem sites in upper Benewah Creek and at several tributary sites in upper Lake Creek. Four attributes that have been linked to the quality of salmonid rearing habitat were used in the analyses: percent canopy cover, percent fines (< 2 mm) in riffles, mean residual pool depth, and large woody debris (LWD) availability. For LWD availability, the analysis examined both counts and volume (m^3) per 100 m (see section *3.2.2.1 Evaluating physico-chemical response to restoration* for details on how metrics were measured).

Regional trend detection for each habitat metric was examined by evaluating the number of sites required to enable detection of a synchronous trend with 80% probability over a specified period of time (i.e., power analysis). The analysis examined trend detection power for a set of 5 and 10 sites monitored annually over 5, 10, 15, and 20 years. In addition, the analysis examined trend detection power for a set of either 5 or 10 sites monitored every 4 years for periods lasting 12 and 20 years. For each site and metric, the analysis required an initial measurement and an estimate of annual variability, which were respectively calculated as the mean and standard deviation from annual empirical data. Because repeated annual measurements were available for only four sites in mainstem reaches of upper Benewah Creek, one site was used twice to create the set of 5 sites used in the analyses; this set of 5 sites was then duplicated to create the set of 10 sites. For the 7 available sites in tributary reaches of upper Lake Creek, 5 were representatively selected to create the set of 5 sites used in analyses; all 7 sites were used for the set of 10 with 3 of the sites used twice. Both increasing and decreasing trends that ranged from 1 to 10% were simulated in the power analysis. The online software package MONITOR 7.0 was used for all simulated power analyses (α was set at 0.05).

Precision analyses were also conducted to gauge the number of measurements (i.e., sample size) that would need to be collected to accurately assess the 'restoration distance' between the current state and the desired target objective for each habitat metric. These analyses were conducted separately for different channel types: unrestored tributary habitat, unrestored mainstem habitat, and restored mainstem habitat. Habitat sites in lower tributary reaches of Lake Creek provided empirical data for the tributary channel type, and habitat sites in restored and unrestored mainstem reaches of upper Benewah Creek provided data for the mainstem channel type. In addition, analyses were also conducted for quasi-reference reaches for both tributary and mainstem channel types; site 3 in upper Bozard Creek in the Lake Creek watershed and site 18 in the upper Benewah mainstem (upstream of 12-mile bridge) provided data for these two channel types, respectively.

For each habitat metric and channel type, sample size was calculated using the following equation:

$$n = \frac{4 * s^2}{d^2},$$

where:

n = sample size;
s = standard deviation; and
d = precision interval (i.e., half-width of a 95% confidence interval).

Standard deviation estimates were generated for each year from empirical data collected annually across all sample sites within each channel type. For all metrics other than those associated with LWD (which were calculated as count or volume per 100 m), all measurements within a site were used in calculations rather than mean values. For example, six measurements were typically collected to estimate percent canopy cover and two or three riffles were usually sampled to estimate percent fines within a given habitat site. In addition, the number of pools sampled within a site was dependent upon channel pattern and typically ranged from 5 to 20. A mean standard deviation was then computed from these annual estimates and used in the above formula to calculate sample size. For analyses of quasi-reference reaches, where only one site was available for each channel type, data from 2006 to 2008 were collectively used to generate standard deviation estimates for each metric. Precision intervals were selected to be small and were tailored to each habitat metric: intervals for percent canopy cover and fines included 5 and 10%; residual pool depth intervals included 0.1 and 0.2 m; LWD count intervals included 3 and 5 pieces per 100 m; and LWD volume intervals included 1 and 2 m^3 per 100 m.

3.2.2 Effectiveness monitoring – Response to restoration activities

We evaluated the response of habitat and trout populations to restoration measures by comparing metrics collected at treated and control sites before and after implementation of habitat enhancement activities. Physical attributes, which have been linked to the quality of trout habitat, were typically measured within established 152 m long sites, and included large woody debris volume, canopy cover, substrate composition, and pool depth and volume. Standardized electrofishing sites (i.e., 61 m) were typically encompassed by the habitat sites for the evaluation of relative changes in cutthroat trout abundance between restored and control reaches. Brief descriptions of treatment and control sites for evaluated restoration projects are provided in the following paragraphs (references are provided for a more detailed account of each of the restoration activities).

In 2007, large woody debris structures were introduced into lower Whitetail Creek to address deficiencies in habitat complexity and to increase channel stability (Firehammer et al. 2009). A 411 m long habitat site within the targeted area served as the treatment site, and a 152 m long habitat site along a reach of lower Windfall Creek, which had similar channel attributes, served as the control site. Cutthroat trout response was evaluated in both lower Whitetail and Windfall creeks at those index sites that have historically been used for trend monitoring.

In upper mainstem reaches of Benewah Creek, 2075 m of channel habitat have undergone restoration over the past four years (Chess et al. 2006; Vitale et al. 2007; Vitale et al. 2008; Firehammer et al. 2009). In 2004, two-hundred meters of mainstem habitat below the confluence of Windfall Creek was restored, in association with culvert replacement at the confluence, to improve fish habitat and re-establish fish passage. Stream restoration since 2004 has proceeded along contiguous mainstem reaches upstream of 9-mile bridge. The following treatment sites (i.e., habitat and shock sites) exist within each of the annually restored reaches:

- Site 16 (historical trend monitoring site for fish abundance) is located in the reach below the Windfall confluence that was restored in 2004;
- Site 15L is located in the reach restored in 2005, and was first sampled in 2006;
- Site 2006 is located in the reach restored in 2006, and was first sampled this year;
- Site 15 (historical trend monitoring site for fish abundance)is located in the reach restored in 2007; and
- Site 2008 is located in the reach that underwent restoration this year, and was first sampled this year.

Furthermore, control sites have also been established along mainstem reaches above and below the restored areas:

- Site 14 (historical trend monitoring site for fish abundance) is located below 9-mile bridge, and only serves as a control site for monitoring fish response;
- Site 16L is located in the unrestored mainstem reach (upstream of Gore Creek), and was first sampled in 2007;
- Site 17 (historical trend monitoring site for fish abundance) is located in the unrestored mainstem reach downstream of 12-mile bridge; and
- Site 18 is located in the quasi-reference reach upstream of 12-mile bridge, and was first sampled this year.

Sites have also been established in upper reaches of the Lake Creek watershed to track physical response to prior habitat enhancement projects and to provide pre-restoration data for reaches that are targeted for future enhancement activities. Prior riparian plantings, which included both conifers and deciduous trees and shrubs, have occurred at sites West Fork 2, Lake 11, and Lake 12. In addition, large wood was placed in the stream channel at Lake site 11 in 1999 to create pools, promote bank stablity, and increase sinuosity. Several other sites serve as both untreated and reference controls. Habitat metrics were collected at 10 of these sites in 2008.

3.2.2.1 Evaluating physico-chemical response to restoration

Stream typing
The classification of stream channel types followed guidelines presented by Rosgen (1996) and used data collected during the thalweg profile, cross section profile and sinuosity surveying efforts. The objective of classifying streams on the basis of channel morphology was to use discrete categories of stream types to develop consistent, reproducible descriptions of the stream reaches. These descriptions must provide a consistent frame of reference to document changes in the stream channels over time and to allow comparison between different streams. The dominant substrate type (i.e., slit/clay, sand, gravel, or cobble) was included as a modifier to the channel type. The numbering for this is 1 for bedrock, 2 for boulder, 3 for cobble, 4 for gravel, 5 for sand and 6 for silt and clay. The delineative criteria included entrenchment ratio, width-to-depth (W/D) ratio, sinuosity and slope.

Longitudinal thalweg profile
The first effort to be undertaken upon arrival at a monitoring site was to determine the location of the downstream end of the previously surveyed reach. Once this was found, the location was flagged with surveyors' ribbon. Bank pins were established on the banks of the channel above the high water mark at major changes in the channel planform. When the 500-foot mark was reached this marked the end of the reach. Profile surveys involved the determination of water depth, and water surface and channel bottom elevations along the thalweg of each 500-foot study reach using methods modified from Peck et al. (2001). Elevation measurements were made relative to a fixed benchmark, assigned an arbitrary elevation of 100.00 ft. All measurements were recorded as distances along the longitudinal profile. A sufficient number of measurements were taken to capture all changes in bed and water surface slope and habitat types along the reach. A SET 530R Sokkia Total Station was used to collect longitudinal profile data at most sites, in place of an autolevel, which had been used in previous surveys. Survey data was recorded on a Recon Pocket PC. After the survey was complete, data was downloaded into a text file and imported into Microsoft excel for analysis.

Cross section profiles
Cross section profiles were measured using a surveyor's level and rod at six locations along each studied reach. All but one of the sites had cross-sections that had been previously established in 2002 or 2003. All cross sections were monumented with permanent pins (rebar), stakes, lathe and flagging to allow for repeat surveying of the profiles in the future. In some cases, survey pins had to be reset because they had been moved or "lost". The benchmark established for the longitudinal profile was also used as the reference point for each of the six cross sections.

The cross section profiles were used to verify the bankfull depth and to calculate the bankfull cross sectional area, wetted perimeter, average and maximum depth and width-to-depth ratio. The flood-prone width, which is defined as the valley width at twice the maximum depth at bankfull, and entrenchment ratio, defined as the flood-prone width divided by the bankfull width, were determined by using floodplain cross-section information collected with the total station if it was collected. Survey data was input into the Reference Reach Spreadsheet.

Bed-form differencing
Identifying pool and riffle habitats is important in monitoring changes in bedform and fish habitat. Residual pool depth (RPD) is a particularly important habitat indicator because it can be accurately measured independent of discharge (Kershner et al 2004) and increasing RPD is

generally associated with increased salmonid biomass (Hogel 1993; Binns 1994). A macrohabitat identification technique called the Bed Form Differencing was applied to each of the longitudinal profiles collected to minimize the error in identifying pools and riffles due to acknowledged inconsistencies associated with field identification (Kershner et al 2004) and to facilitate comparisons across datasets (Arend 1999). This method was developed by O'Neill and Abrahams (1984) as a way to objectively identify bedforms in a survey reach.

Four types of bedforms are identified using this method: absolute maximums (riffles), absolute minimums (pools), local maximums, and local minimums. The tolerance value is determined by taking the standard deviation of all of the "differences" and multiplying it times a coefficient. If habitat units exceed this value they are classified as either a minimum or a maximum. If they do not exceed this value they are identified as not being a bedform. If a maximum is followed by a minimum then it is a absolute maximum (riffle). If a maximum is followed by another maximum, it is identified as a local maximum. If a minimum is followed by a maximum, it is defined as an absolute minimum (pool). A bed differencing program was developed in Microsoft Excel using Visual Basic. Residual pool depths were calculated by running a program that sorts the bedforms that are either absolute maximums or absolute minimums, then identifies the first "riffle" and starts calculating residual pools by subtracting the elevation of the absolute minimum from the adjacent downstream absolute maximum. The sample spacing is assumed to be equal to channel width though shorter spacing can be used. The resolution of our data is at a much tighter interval. As a result, we have modified our data in order to achieve spacing closer to bankfull width by running the program twice. After the first run is complete, the sign designation of each point is examined. If there is a series of more than two increasing or decreasing points, the intermediate points are deleted, then the program is ran again.

Pool volume

A reduced longitudinal survey was introduced in 2008 in order to collect detailed pool information at habitat survey sites. 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. 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 the minimum depth plus one foot. 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 was also collected. Pool forming mechanisms include boulder (B), meander (M), wood (W), and other (O). Types of pools include dammed pools (D), scour pools (S), and other types of pools (T). The aim with this methodology is to examine the quantity and quality of pool habitats that can be used low flow.

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. Pebble count data was input into the Reference Reach

Spreadsheets, which automatically graphed the distribution of particle sizes and calculated pertinent descriptive criteria such as percent by substrate class (size) and a particle size index (D value) for each habitat type for which data was collected.

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, and this is measured in percent. Canopy density can change drastically through the year if the canopy vegetation is deciduous. Canopy cover over the stream was determined at randomly selected locations throughout the survey reach. At each selected location, densiometer readings were taken one foot above the water surface at the following locations: 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 by multiplying the sum of the four readings by 1.5. If the result was between 30 and 65%, 1.0 % was subtracted; if the result is greater than 65, 2% was subtracted. The adjusted density readings were then averaged for the entire reach.

Large woody debris
The organic materials survey transect was walked along the thalweg starting at the downstream end of the reach. All woody debris that was greater than 4 inches in diameter at the small end was tallied and measured whether or not it crossed the line of the transect. This included material, other than living trees and shrubs, suspended above the water surface or partially located outside of the wetted stream width. Small and large end diameters (in) and lengths (ft) were recorded for each piece of LWD. If roots were attached, the large end diameter was measured immediately above the roots. Total volume and density of LWD within bankfull width was calculated for each habitat site.

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 impacting channel form within the site. Function categories included: accumulating sediment (AS), forcing a pool to form upstream or downstream (FP), providing in-stream cover (HC), providing bank stabilization (BS), or none of the above (N). More than one category could be assigned to individual wood pieces. Categories to describe the position of the indentified piece in relation to the stream included: elevation above the bankfull channel (1), one end within and the other end outside bankfull channel (2), completely within bankfull channel but exposed (3), or within bankfull channel but partially buried (4).

Thermal refugia
Thermal heterogeneity at fine-scale, riffle/pool sequences was assessed 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 changes in the availability of thermal refugia in upper mainstem reaches in Benewah Creek before and after restoration.

3.2.2.2 Evaluating cutthroat trout response to restoration

Cutthroat trout response was evaluated by comparing numbers of fish in restored sites before and after treatment application relative to that observed over similar time periods at control sites. An ANOVA was used to evaluate potential changes whereby site (i.e., treatment or control) and period (i.e., before and after treatment) served as factors in the model. A significant interaction between these two factors would indicate that the change in trout abundance between periods was not similar between restored and control sites, suggesting a response to the treatment. Cutthroat population response to restoration in the upper Benewah was also evaluated at a spatial scale larger than that defined by the sample site. First, abundances of age 1+ fish were aggregated over all sites sampled in tributaries upriver of 9-mile bridge in the Benewah Creek watershed for each year from 2002 to 2008. Similar annual aggregate abundances were also computed over all tributary sites sampled upriver of the confluence of Bozard and Lake creeks in the Lake Creek watershed, and over all sites sampled across the Evans Creek watershed. Aggregate abundance values for each watershed were then expressed as fish/100 m to permit standardization over years and across watersheds. Logarithmic ratios were then computed to create two time series that each compared Benewah aggregate values to those in the other watershed to evaluate whether abundances in upper Benewah were changing at a different rate than those in the Lake and Evans creek watersheds.

3.2.3 Effectiveness monitoring - Biological responses to brook trout removal in Benewah

In late summer and early fall, single-pass electrofishing was used to remove non-native brook trout from upper mainstem and tributary reaches in the Benewah watershed. Removal efforts started at the 9-mile bridge and proceeded upstream to the confluence of the West and South Forks, and then continued through the lower reaches of both forks. In addition, the RBW trap remained deployed from the end of spring trapping through the end of the removal efforts to prevent larger brook trout from ascending into upper reaches above 9-mile bridge. All index sites associated with the population surveys were sampled prior to brook trout removal.

Lengths were collected from all brook trout removed in the Benewah watershed. In addition, a subsample of fish were dissected to ascertain gender, reproductive maturity, gonad weight, and, in the case of females, fecundity. Weights and scale samples were also collected from dissected fish. A representative number of brook trout were also sacrificed from Alder Creek (which served as the control for the removal program) to obtain similar life-history data whereby compensatory reproductive responses could be compared between the two watersheds.

Changes in brook trout abundance due to the removal program were assessed by comparing mean index site abundances over the years 2002-2004 to those estimated during the 2008 surveys. Mean abundances from the pre-implementation period were used to minimize potential bias introduced by natural fluctuation in annual brook trout numbers. For Benewah Creek, only index sites impacted by the removal program were included in the analysis, which consisted of mainstem sites 15-17 and all index sites in Whitetail, Windfall, Schoolhouse, and South and West Fork creeks. Similarly, only index sites in Alder Creek in which brook trout have been consistently found were included in the comparative analysis. The non-parametric Wilcoxon rank sum test was used to assess statistically significant differences between time periods for both watersheds.

Logistic regression was used to assess annual differences in maturation probabilities over the period from 2004 to 2008 for both male and female brook trout (Johnson 1998). The model included maturation status as the dependent variable and length and year as independent variables. Analyses were also conducted separately by year for male brook trout to evaluate differences in maturation probabilities between watersheds. For 2008, maturation probability models, derived from dissected fish, were also used to permit the assignment of maturation status to fish that were removed but not dissected. However, in this case, the independent variable used in the logistic regression model was not fish length but the 10 mm length bin (i.e., 120, 130, 140, etc.) within which the length was included. Fish of unknown status were then grouped into these 10 mm length bins, and first classified as either male or female using the sex ratio derived from the first 102 fish that were randomly dissected (length distributions were relatively similar between these 52 females and 50 males; Kolmogorov-Smirnov non-parametric test, $p = 0.084$). The derived probability models were then used to calculate the percentage of males and females in each length class that would be expected to be mature.

Fecundity-at-length regressive relationships for female brook trout were also analyzed over the time period from 2004 to 2008 to evaluate potential compensatory responses to our removal program. Linear models were developed which included log-transformed fecundity as the dependent variable, and watershed and year as categorical variables and log-transformed length as a continuous covariate. All interaction terms were included in the full model given that detected interactions between watershed and year would indicate that the regressive relationship was not changing similarly over time between the two systems. Model selection procedures employed backward selection protocol in which insignificant terms (p=0.15) were incrementally removed from the full model until only significant terms remained. Because of the lack of large females captured in Alder Creek and in Benewah Creek during the early years when tributaries were primarily targeted, only fish that were smaller than 250 mm were included in the analysis to permit comparisons across years and between watersheds. We felt that the exclusion of larger fish would not bias our examination and interpretation of reproductive compensation, given that 97% of the brook trout removed from Benewah Creek in 2008 were less than 250 mm.

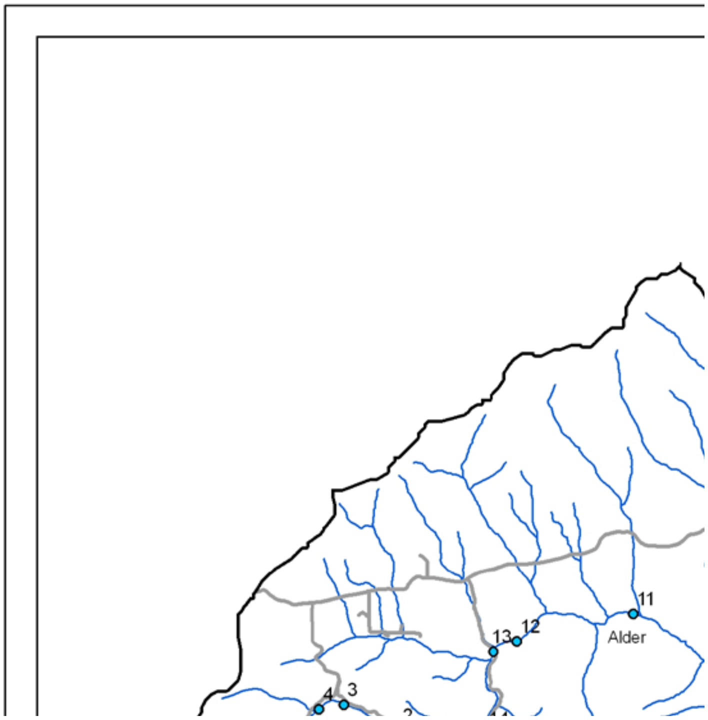

Figure 2. Map of Alder Creek depicting index sites sampled during salmonid population surveys in 2008.

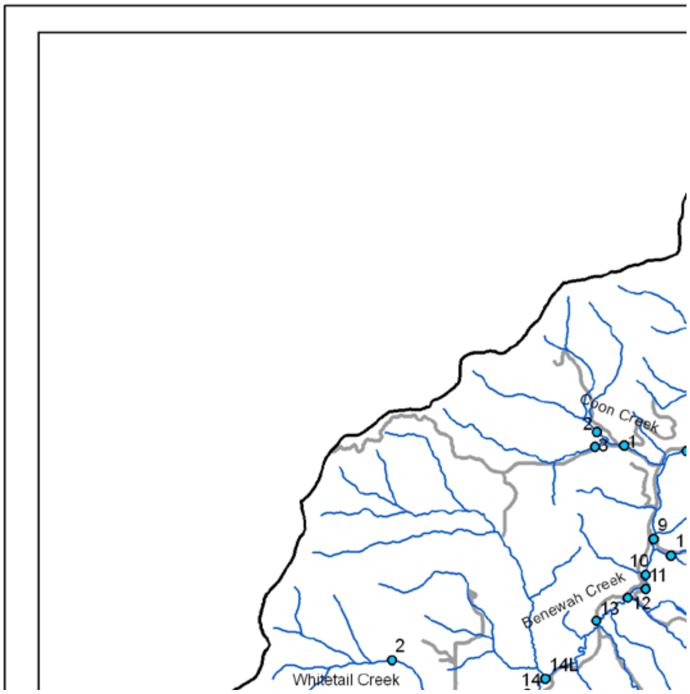

Figure 3. Map of Benewah Creek depicting index sites sampled during salmonid population and habitat surveys in 2008. The location of the traps and PIT-tag array is indicated by the star.

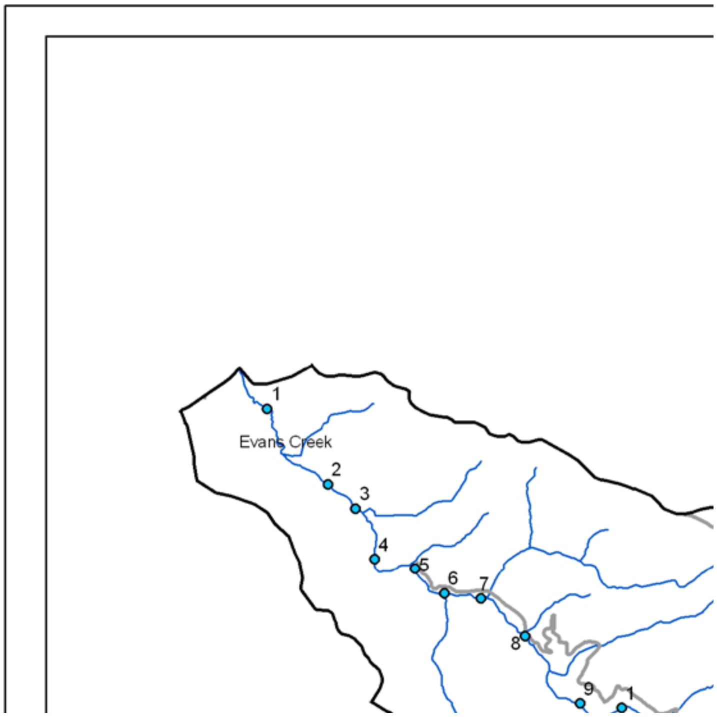

Figure 4. Map of Evans Creek depicting index sites sampled during salmonid population surveys in 2008.

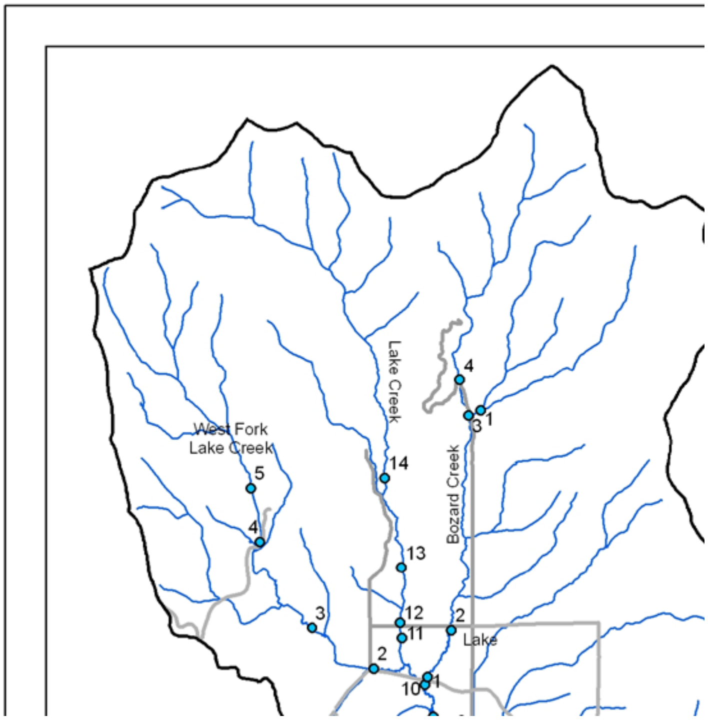

Figure 5. Map of Lake Creek depicting index sites sampled during salmonid population surveys in 2008. The location of the traps and PIT-tag array is indicated by the star.

3.3 Results

3.3.1 Trend and status monitoring – Biological indices

3.3.1.1 Lake Creek adult adfluvial cutthroat trout migration

The RBW trap was installed in the mainstem of Lake Creek on November 13 in 2007 and was removed on May 27 in 2008. From installation through the end of February (108 d), the trap was periodically checked for a total of 30 d and was considered fishing over 96% of the time. However, during the spring period from March 1 through May 27 (87 d), the trap was considered compromised 50% of the time over the 58 d that it was monitored (Figure 6). Trap panels were depressed below the water surface during several extended (i.e., $\geq$ 7 d) high water events on March 10-19, April 11-18, April 28-May 13, and May 16-22. The DN trap was installed in Lake Creek on April 7 and removed on July 17 in 2008 (100 d). However, over the 62 d that the trap was monitored, it was considered fishing only 59% of the time (Figure 6). Pop-out panels had to be removed during the aforementioned high water periods in mid-April and late May. In addition, the high water event in late April caused severe damage to many of the trap panels, resulting in a two week period of inoperation while the trap was being repaired.

A total of 38 adfluvial adult cutthroat trout was captured in the RBW trap (Table 1). Thirty-two (84%) of these 38 were females with a mean length and weight of 384 mm and 547 g, respectively. The other 6 were identified as males with a mean length and weight of 390 mm and 556 g, respectively. Adults were only captured over two 5 d periods from March 31 to April 4 and from April 23 to April 28 (Figure 7), a result not unexpected given the extended period of time in which the trap was compromised. Two additional adults, one male of 390 mm and one female of 404 mm, were captured by shocking the reach between the two traps on April 23 after the DN trap was installed. Given that it was difficult to determine the direction in which these fish were moving, they were released upriver of the DN trap. Twenty-seven of the captured adults received a caudal fin punch.

A total of 124 adfluvial adults was captured in the DN, over three times as many as that captured in the RBW (Table 1). Of these 124, 85 were identified as females (mean length of 380 mm; mean weight of 448 g) and 37 as males (mean length of 400 mm; mean weight of 524 g). The calculated sex ratio of females to males was much lower for those fish caught in the DN trap (2.3:1) than for fish captured by the RBW (5.3:1). Noticeably, both the mean weight ($t = -5.3$, $p < 0.001$) and the mean condition factor ($t = -8.6$, $p < 0.001$) was significantly lower for females captured in the DN than for females caught in the RBW, indicating that many of the outmigrating females likely spawned (Table 1).

Five of the 124 adults captured at the DN trap had a detectable caudal fin punch, yielding a spawner abundance estimate of 582 fish (±373). However, many of the post-spawn fish had frayed fins which precluded accurate mark recognition and likely negatively biased the number of recaptures. Most of the adults (64%) were captured after May 22, a result directly due to the inability to effectively operate the trap during the frequent high water events that occurred from late April to late May (Figure 7). Given that 27 adults, the most captured on any given day during the outmigration, were caught on May 15 when the trap was briefly operational between high discharge events, it was likely that a significant portion of post-spawn fish was not captured in 2008.

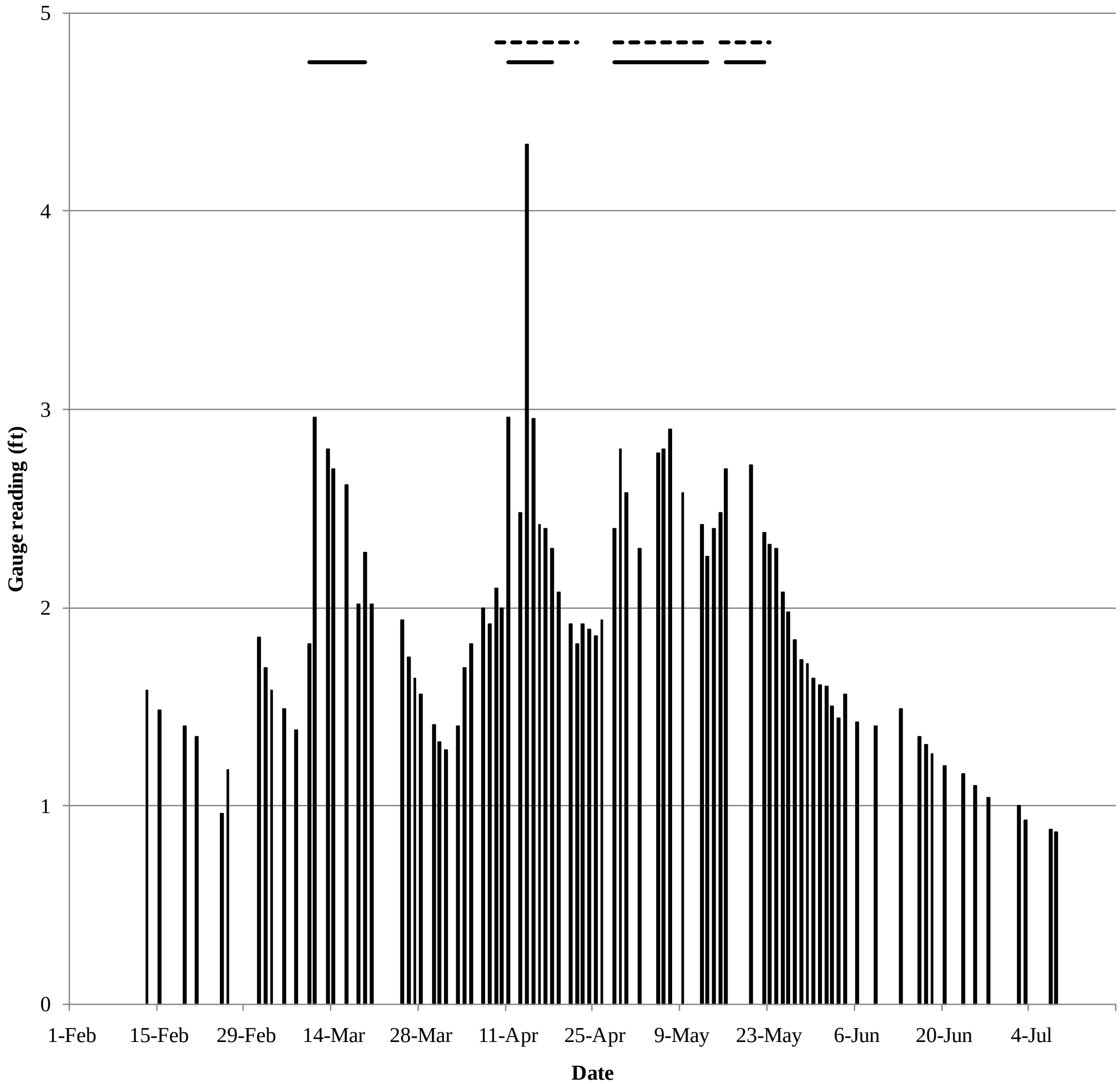

Figure 6. Gauge height readings (ft) collected at the old H95 bridge during the 2008 migratory period for adfluvial westslope cutthroat trout in Lake Creek. Solid and dashed bars at the top indicate periods when the RBW and DN traps were compromised, respectively.

Sixteen fish were detected by the Lake Creek PIT-tag array during migratory periods in 2008 (Table 2). Six, seven, and three of these 16 were tagged as juveniles in 2005, 2006, and 2007, respectively. Of the six fish that had been tagged in 2005, five were also detected either by the array or in traps during migratory periods in 2007. Initial detections for all but one of the detected fish occurred from March 25 to April 25 (Table 2), spanning a similar period over which adults were intercepted by the RBW (Figure 7). Generally, as supported by the abbreviated period of 1-6 d in which most fish were continuously detected after their first array detection, fish were either captured by the RBW or apparently ascended past the trap quickly (Table 2). Nine of the 16 fish (56%) were detected either in a trap or by the array during two different time periods, alluding to the detections occurring during both the upriver spawning migration and downriver post-spawn outmigration. The elapsed number of days between detection periods for these nine fish ranged from 18 to 60 d (Table 2).

Six of the 16 fish (five females and one male) that were detected by the PIT-tag array were also recaptured in either the RBW or DN traps in Lake Creek in 2008 (Table 3). Four of the six had been tagged as juveniles in 2006 and displayed two year growth increments that ranged from 187 to 234 mm. One of the six was a female that had been recaptured in the DN trap during outmigration periods of 2007 and 2008. As indicated by the insubstantial change in length of 15 mm from 2007 to 2008 for this fish (Table 3), annual body length increments likely decrease dramatically after maturation.

Table 1. Length, weight and condition factor means and standard deviations (SD) for adult adfluvial cutthroat trout captured during their upriver and downriver migrations in Lake and Benewah creeks in 2008.

		Total length (mm)			Weight (g)		Condition Factor	
Gender	**N**	**Range**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
				Lake Creek upriver trap				
Female	32	329-478	384	27.0	547	110.1	0.96	0.087
Male	6	316-446	390	48.4	556	177.3	0.91	0.075
				Lake Creek downriver trap [a]				
Female	85	308-482	380	25.9	448	81.1	0.82	0.074
Male	37	319-525	400	35.6	524	111.8	0.84	0.061
				Benewah Creek downriver trap				
Female	12	296-380	346	26.1	339	70.8	0.81	0.041
Male	3	318-440	370	63.0	457	236.6	0.85	0.036

[a] Two additional adults of unidentified sex captured in the DN trap

3.3.1.2 Lake Creek juvenile cutthroat trout migration

A total of 1953 juvenile cutthroat trout was captured by the DN trap in Lake Creek in 2008. Because of the inability to capture fish during high discharge periods in April and May, over 75% of the fish were captured after June 1 (Figure 8). Given that more than 50 fish were captured daily on several occasions before June when the trap was briefly operational, much of the early portion of the juvenile outmigration was likely missed. A noticeable difference in the size distribution between early and late outmigrating juveniles was detected with the mean size increasing from 130 mm in length during April and May to 140 mm or greater thereafter (Figure 9). However, given that we were unable to trap effectively throughout most of April and May, the size distribution of fish captured during that time may not accurately reflect that for all early outmigrants. In addition to those juveniles considered to be actively outmigrating to the lake, 48 other fish captured in the DN trap were classified as likely residents given their external markings. Mean total length of these 48 fish was 192 mm.

Table 2. Summary of 2008 detections for cutthroat trout PIT-tagged in previous years in Lake Creek. Fish were considered detected, either in the trap or by the array, during two periods if the absence of detections between the two periods lasted longer than 14 days. One or two asterisks next to the elapsed period indicate fish were captured at either the upriver or downriver trap, respectively.

Tagging information			Dates of PIT-tag array detection				Initial period in which fish was detected by array			
Year	Length (mm)	Weight (g)	First detection	Last detection before absence	First detection after absence	Last detection	Elapsed days	Days detected	Elapsed days between detection periods	Detected in prior years
2005	215	85.0	19-Apr	.	.	19-Apr	1	1	.	2007
2005	200	65.5	22-Apr	.	.	26-Apr	5	4	.	2007 [a]
2005	147	25.8	15-Apr	15-Apr	5-May	5-May	1	1	20	2007 [b]
2005	173	48.8	1-May	.	.	1-May	1	1	.	2007
2005	164	42.5	17-Apr	.	.	17-Apr	1	1	60 **	.
2005	146	28.2	18-Apr	18-Apr	29-May	29-May	1	1	41 **	2007
2006	174	45.3	21-Apr	21-Apr	10-May	10-May	1	1	19	.
2006	177	45.6	15-Apr	.	.	15-Apr	1	1	.	.
2006	175	45.1	25-Apr	.	.	27-Apr	3	3	.	.
2006	134	20.4	16-Apr	.	.	16-Apr	1	1	29 **	.
2006	180	50.7	25-Mar	3-Apr	28-Apr	28-Apr	10	9	24 *	.
2006	170	41.6	2-Apr	.	.	11-Apr	10	3	26 **	.
2006	177	48.4	25-Apr	26-Apr	16-May	16-May	2	2	18 *	.
2007	108	10.3	20-Apr	20-Apr	16-Jun	16-Jun	1	1	57	.
2007	186	57.3	3-Apr	.	.	8-Apr	6	6	.	.
2007	120	15.4	16-Apr	.	.	16-Apr	1	1	.	.

[a] Detected in the DN trap in 2007, but not identified as an adfluvial adult; likely a resident fish.

[b] Detected in the RBW trap in 2007 with a total length of 339 mm.

Table 3. Summary data for adult adfluvial cutthroat trout PIT-tagged as juveniles in prior years and either recaptured at the resistant board weir (RBW) trap during their upriver migration or at the downriver (DN) trap during their outmigration in Lake Creek, 2008.

	Tagging information				Recapture information for 2008						Recapture information for 2007		
Sex	Year	Date	Length (mm)	Weight (g)	Location	Date	Length (mm)	Weight (g)	Years at large	Growth (mm)	Location	Length (mm)	Weight (g)
F	2006	21-Apr	180	50.7	RBW	4-Apr	367	515.9	2	187	.	.	.
F	2006	25-Apr	177	48.4	RBW	28-Apr	383	508.8	2	206	.	.	.
F	2005	29-Apr	146	28.2	DN	29-May	372	424.3	3	226	DN	357	372.4
M	2005	29-Apr	164	42.5	DN	16-Jun	416	609.4	3	252	.	.	.
F	2006	20-Apr	134	20.4	DN	15-May	368	420.1	2	234	.	.	.
F	2006	22-Apr	170	41.6	DN	28-Apr	382	491.8	2	212	.	.	.

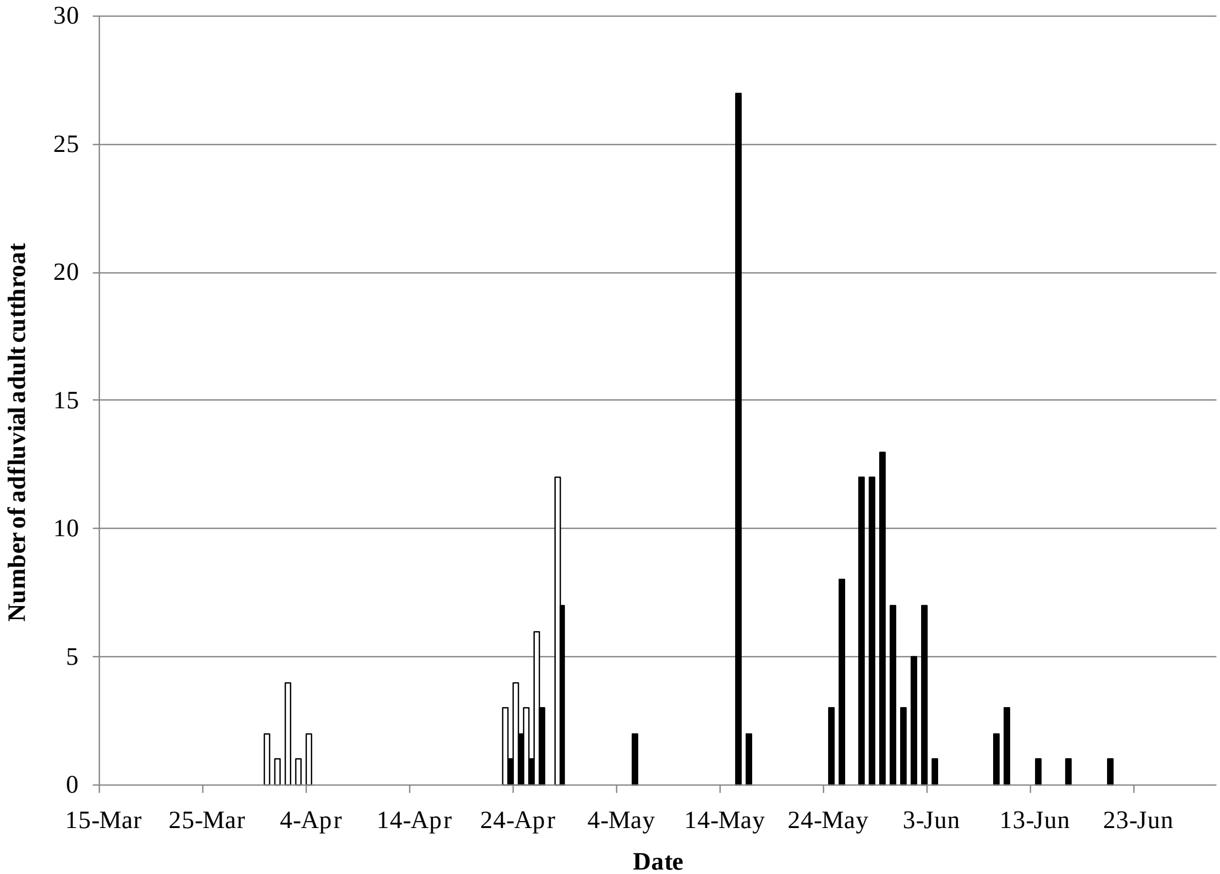

Figure 7. Timing of adult adfluvial cutthroat trout captured during their upriver (unshaded bars) and downriver (darkened bars) migrations in Lake Creek, 2008.

Of the 1953 adfluvial juveniles captured, 615 (31%) received PIT tags. Generally, fish were tagged representatively throughout most of their outmigration as supported by the similar pattern in the cumulative distribution curves for PIT-tagged juveniles and all captured juveniles (Figure 8). However, fish were no longer tagged after June 20th despite the capture of approximately 20% of the total after this date. The length distribution of PIT-tagged adfluvial juveniles was similar to that for all juveniles captured in the DN trap ($\chi^2 = 7.4$, $p = 0.28$; Table 4), with approximately 85% of both groups ranging between 101 and 160 mm. Eighteen of the 48 fish that were classified as resident cutthroat trout also received PIT tags.

Eight release trials were conducted from April 25 to June 17 in Lake Creek to generate an overall juvenile outmigrant abundance estimate of 6259 ±1783. However, this estimate should be considered unreliable given the difficulty in trapping fish during many of the trial periods in 2008. As expected, trap efficiency estimates grossly differed across the outmigration period, ranging from 0.06-0.83 (Table 5). Low trapping efficiencies, which contributed to wide confidence intervals for abundance estimates, typically coincided with periods during which the trap could not be effectively operated. For example, during the trial period from June 11 to June 17, only 12% of released fish were recaptured. Poor trapping performance likely resulted from a severe rainstorm on June 11 that washed a lot of debris downstream, clogging panel screens and forcing water over and around the trap for a couple days. During May 16-25 when pop-out panels had to be removed because of high flows, only 6% of trial fish were recaptured. Even though a trap efficiency estimate of 0.28 was generated from April 28 to May 14, a two week high discharge period during which trap panels were removed, this estimate was likely biased upwards given that all trial fish were captured by the trap the day following release and most likely before the trap was damaged the morning of the 29th. Indeed, only 10 untagged juvenile cutthroat were found in the livebox of the DN trap over the 14 d period from April 29 to May 12 before trap panels were reinstalled, suggesting that trapping efficiency was much lower than that estimated. Notably, our tagging procedures did not apparently affect tag recognition and consequently estimated trapping efficiencies given that all release trial fish held upstream of the trap for a 24 h period survived and retained their tags.

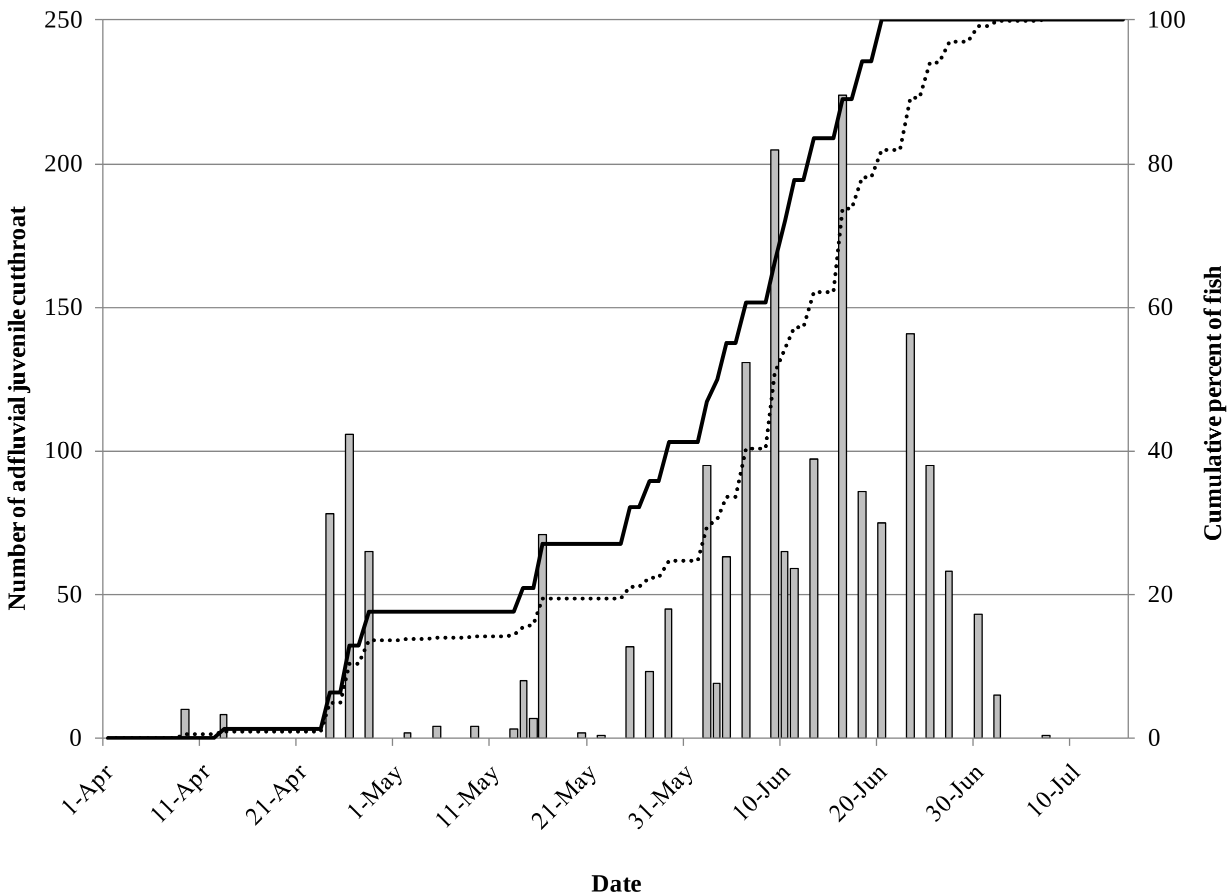

Figure 8. Timing of juvenile adfluvial cutthroat trout captured in the downriver trap during their outmigration in Lake Creek, 2008. Numbers of juveniles (gray bars) along with the cumulative distribution curves of all captured juveniles (dotted line) and PIT-tagged juveniles (solid line) are presented.

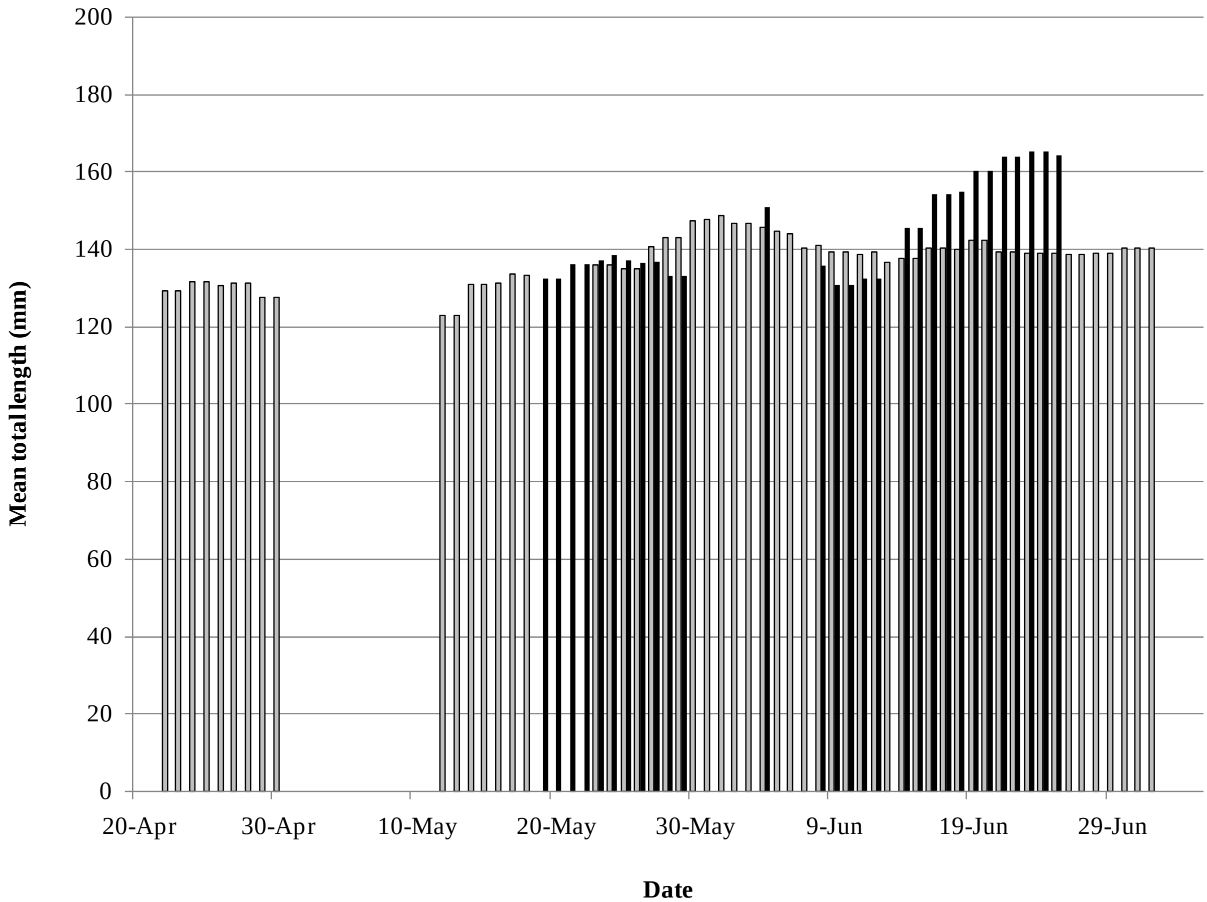

Figure 9. Five day daily moving averages of total length (mm) for adfluvial juvenile cutthroat trout captured in DN traps in Lake (gray bars) and Benewah (dark bars) creeks in 2008. For each day, a mean was calculated only if more than 20 fish were captured over the period that encompassed the 2 days before and after the given day.

Eight groups of PIT tagged juveniles that were not used in trap efficiency trials were released below the trap, but upriver of the PIT tag array, to evaluate potential behavioral differences between groups of fish released above and below the trap (Table 6). All but 4 of the 271 fish (98.5%) released downriver of the trap were detected by the array. Overall detection rates, either through a recapture event or an array detection, were also relatively high for fish released above the trap, exceeding 90% for all but the last release trial group. However, differences were observed when the number of elapsed days between release and detection were compared between groups of fish released above and below the trap (Table 6). For each group of fish released below the trap, more than 94% that were detected by the array were detected within 2 days. Conversely, a lower percentage of fish released above the trap were detected within 2 days, especially during periods in which amenable levels of discharge permitted efficient trap performance. For example, 88, 55 and 39% of recaptured fish were caught within 2 days after being released on April 25, May 25, and June 3, respectively. Similarly, only 7, 25, and 78% of the fish that escaped recapture but were detected by the array from these release groups were detected within 2 days. In addition, the mean number of elapsed days before detection was generally greater than 3.4 for recaptured fish from all but one of the release groups. Similarly, mean number of elapsed days was often large for fish released above the trap but only detected by the array, exceeding 10 d for two of the trial groups. Furthermore, there was great variability in the number of elapsed days before either trap or array detection for fish released above the trap, as illustrated by the large standard deviations (SD) calculated for many of the release trials (Table 6). Conversely, mean number of elapsed days for fish released below the trap was typically less than 1.1 with low SD values, indicating that most of the fish behaved similarly, moving past the array quickly after release.

Table 4. Number and relative percent of adfluvial juvenile cutthroat trout captured and PIT-tagged of different length groups in Lake and Benewah creeks, 2008.

	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	13	0.7	8	1.3	4	1.4	1	0.5
101-120	258	13.2	80	13.0	42	15.1	33	16.3
121-140	844	43.2	249	40.5	103	36.9	82	40.4
141-160	600	30.7	195	31.7	58	20.8	40	19.7
161-180	182	9.3	64	10.4	35	12.5	25	12.3
181-200	38	1.9	17	2.8	27	9.7	16	7.9
>200	18	0.9	2	0.3	10	3.6	6	3.0

3.3.1.3 Benewah Creek adult adfluvial cutthroat trout migration

In Benewah Creek, the RBW was installed on February 18 and was no longer monitored after July 7 because of the absence of fish in the trap's live box (140 d). The trap was considered fishing 88% of the time over the 49 d that it was monitored. During each of two high water periods from April 11-23 and April 25 – May 8, the trap was periodically checked and panels were observed depressed below the water surface possibly permitting fish to escape upriver (Figure 10). Because of the extended spring high water period, the DN trap in Benewah Creek was not installed until May 15. Over the two month period until it was removed on July 17 (62 d), the trap was considered fishing 94% of the time that it was monitored (26 d). A brief one-day rain event on the evening of June 11 damaged one of the panels, but the panel was fixed and replaced the following day.

Only one male adult adfluvial cutthroat trout was captured by the RBW on April 18 (total length of 335 mm and weight of 383 g). In addition, two other cutthroat trout with lengths of 228 and 232 mm were found in the live box of the RBW, one captured on May 27 and the other on July 7. Given their size and external markings, these two were classified as resident fish. Two brook trout (lengths of 281 and 297) were captured between June 30 and July 7.

Table 5. Abundance estimates for juvenile westslope cutthroat trout outmigrating in Lake and Benewah creeks, 2008. Tagged fish were released on the day denoted by the beginning of the trial period. For Lake Creek, tagged fish were considered available for recapture if they were detected either in the trap or by the array within the trial period. Number of available and recaptured tagged fish collectively included the current release trial fish and those from prior releases. For Benewah Creek, the number of available tagged fish was discounted by those captured during subsequent periods, and recaptured fish only included those from the current release trial.

Trial period	Fish captured	Tagged fish available for recapture	Tagged fish recaptured	Trap efficiency estimate	Abundance estimate	95% confidence interval
			Lake Creek			
Apr-25 - Apr-28	267	17	14	0.82	320	255 - 386
Apr-28 - May-14 [a]	33	69	19	0.28	116	63 - 168
May-14 - May-16	78	19	12	0.63	120	80 - 160
May-16 - May-25 [b]	35	33	2	0.06	397	10 - 783
May-25 - Jun-03	182	23	18	0.78	230	182 - 278
Jun-03 - Jun-11	523	59	49	0.83	628	554 - 701
Jun-11 - Jun-17 [c]	321	81	10	0.12	2393	1112 - 3674
Jun-17 - Jun-28	514	31	7	0.23	2056	884 - 3228
			Benewah Creek			
May-20 - May-21	21	8	3	0.43	47	14 - 81
May-21 - Jun-03	117	9	6	0.70	167	102 - 232
Jun-03 - Jun-11	39	18	3	0.21	185	34 - 337
Jun-11 - Jun-17 [d]	39	27	1	0.07	546	-64 - 1156
Jun-17 - Jun-26	63	12	7	0.61	102	58 - 146

[a] Trap severely damaged on April 29 and pop-out panels were removed until May 12; all recaptures occurred on April 29.

[b] Pop-out panels removed from May 16-23.

[c] Heavy rains on evening of June 11 compromised the trap over the next 3 d.

[d] Heavy rains on evening of June 11 damaged one of the trap panels; panel was fixed on June 13.

Fifteen adult adfluvial cutthroat trout were captured in the DN trap between May 16 and June 23, with 13 of these 15 captured during the first week after trap deployment on May 15. Given the immediate capture of adults after trap installation, a substantial portion of the outmigration could have been missed. Twelve of the 15 were females with a mean length and weight of 346 mm and 339 g, respectively (Table 1). Females were signficantly smaller in length (t = -3.9, p < 0.001) and weight (t = -4.2, p < 0.001) in Benewah Creek than in Lake Creek. The calculated mean condition factor of 0.81 for these females indicated that many of them had likely spawned. The other three adults were identified as males with a mean length and weight of 370 mm and 457 g, respectively (Table 1). Four additional adults, three females and one male, were captured on May 19 by shocking the reach between the RBW and the DN trap. Given the range of condition factors calculated for the females (0.69-0.84) and the timing of capture, these fish were likely engaging in a post-spawn outmigration and were consequently released below the DN trap.

Table 6. Summary statistics for detections of PIT-tagged juvenile adfluvial cutthroat trout released above and below the downriver trap in Lake Creek, 2008. Fish released below the trap and detected by the array on the day of release were given a value of 1 for number of elapsed days to permit comparisons with fish released above the trap given that the latter group are not evaluated for recapture until the day after release.

		Fish recaptured in the trap				Fish not recaptured but detected by array				
				Elapsed days before recapture				Elapsed days before recapture		
Release Date	Number released	Number (%)	Recaps within 2 d (%)	Mean	SD	Number (%)	Recaps within 2 d (%)	Mean	SD	Total detections either in trap or by array (%)
Fish released upriver of the trap										
25-Apr	31	16 (52)	88	7.1	16.6	14 (45)	7	10.5	14.3	30 (97)
28-Apr	69	20 (29)	95	3.6	11.6	45 (65)	76	5.7	12.5	65 (94)
14-May	22	13 (59)	92	4.4	8.6	7 (32)	86	3.3	4.8	20 (91)
16-May	38	3 (8)	0	8.3	7.5	32 (84)	91	2.4	5.1	35 (92)
25-May	32	22 (69)	55	6.5	6.8	8 (25)	25	13.4	9.4	30 (94)
3-Jun	56	46 (82)	39	3.4	2.3	9 (16)	78	2.7	4.3	55 (98)
11-Jun	71	7 (10)	71	3.4	2.7	64 (90)	97	1.1	0.5	71 (100)
17-Jun	35	4 (11)	75	1.5	1.0	22 (63)	86	1.4	1.0	26 (74)
Fish released downriver of the trap but upriver of the PIT-tag array										
27-May	25	.	.	.	.	25 (100)	100	1.0	0.0	.
29-May	35	.	.	.	.	33 (94)	94	1.7	3.3	.
4-Jun	34	.	.	.	.	34 (100)	100	1.1	0.2	.
6-Jun	35	.	.	.	.	35 (100)	100	1.0	0.0	.
9-Jun	35	.	.	.	.	35 (100)	100	1.0	0.0	.
13-Jun	36	.	.	.	.	36 (100)	97	1.1	0.5	.
18-Jun	36	.	.	.	.	36 (100)	97	1.1	0.7	.
20-Jun	35	.	.	.	.	33 (94)	100	1.0	0.2	.

3.3.1.4 Benewah Creek juvenile cutthroat trout migration

A total of 279 adfluvial juvenile cutthroat trout was captured in the DN trap from May 20 to June 27. Generally, captured fish were proportionately distributed across the outmigration period (Figure 11); however, given the late installation of the trap, it was difficult to ascertain when the outmigration began. A noticeable shift in the size distribution of juveniles occurred between the early and late portion of the captured outmigration. Mean length was approximately 130 mm from May to June 12, then increased thereafter to greater than 150 mm (Figure 9). Twenty-one of these 279, however, 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 length of these 21 fish was 188 mm. Thirteen other cutthroat trout captured in the DN trap were classified as resident fish based on their external markings. One of these 13 had a length of 232 mm, and the mean length for the other 12 was 180 mm. In addition, one other fish not considered to be adfluvial had markings suggestive of potential hybridization; total length for this fish was 257 mm.

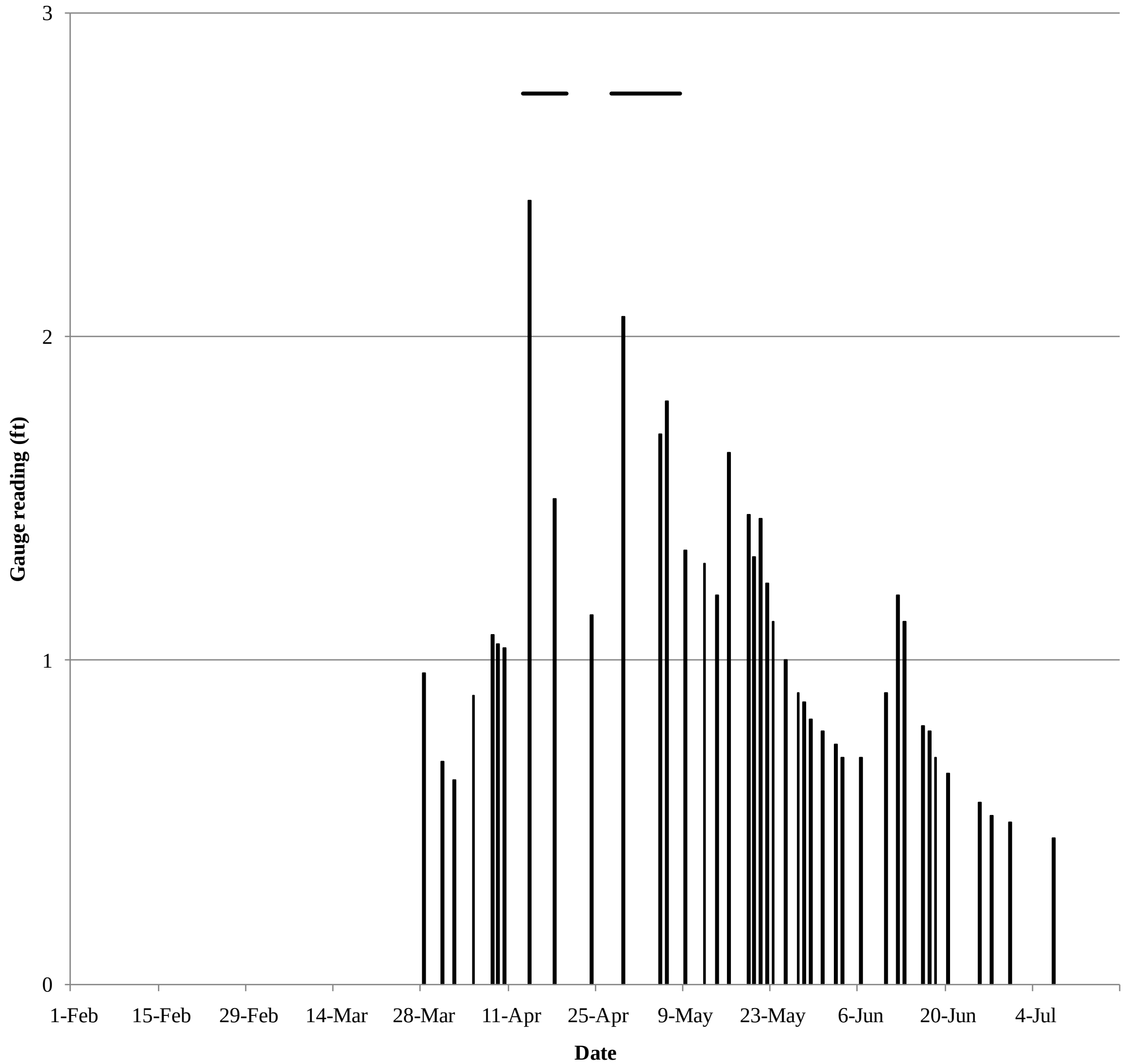

Figure 10. Gauge height readings (ft) collected at 9-mile bridge during the 2008 migratory period for adfluvial westslope cutthroat trout in Benewah Creek. Solid bars at the top indicate periods in which the RBW trap was compromised.

Of the 279 adfluvial juveniles captured, 203 (73%) received PIT tags. Fish were generally tagged representatively throughout the outmigration period as supported by the similar pattern in the cumulative distribution curves for PIT-tagged juveniles and all captured juveniles (Figure 11), though 10% of all juveniles were captured during late June after PIT-tagging ceased. In addition, the length distribution of PIT-tagged adfluvial juveniles was similar to that for all juveniles captured in the Benewah DN trap ($\chi^2 = 2.1$, $p = 0.91$; Table 4), with approximately 85-90% of both groups ranging between 101 and 180 mm. Eleven of the 14 fish that were classified as resident cutthroat trout also received PIT tags.

Five release trials were conducted from May 20 to June 17 in Benewah Creek to generate an overall abundance estimate of 1048 ±635 fish. Trap efficiency estimates differed across the outmigration period, ranging from a low of 0.07 from June 11 to June 17 to a high of 0.70 from May 21 to June 3 (Table 5). The low efficiency estimate was most likely due to a severe rainstorm on the evening of the 11th which blew out one of the trap panels and permitted unobstructed passage through the trap. Given the low estimated trapping efficiency during this trial period, there was a large degree of uncertainty generated in the period's abundance estimate (Table 5). Notably, our tagging procedures should not have unduly affected trap efficiency estimates given that all release trial fish held upstream of the trap for a 24 h period survived and retained their tags.

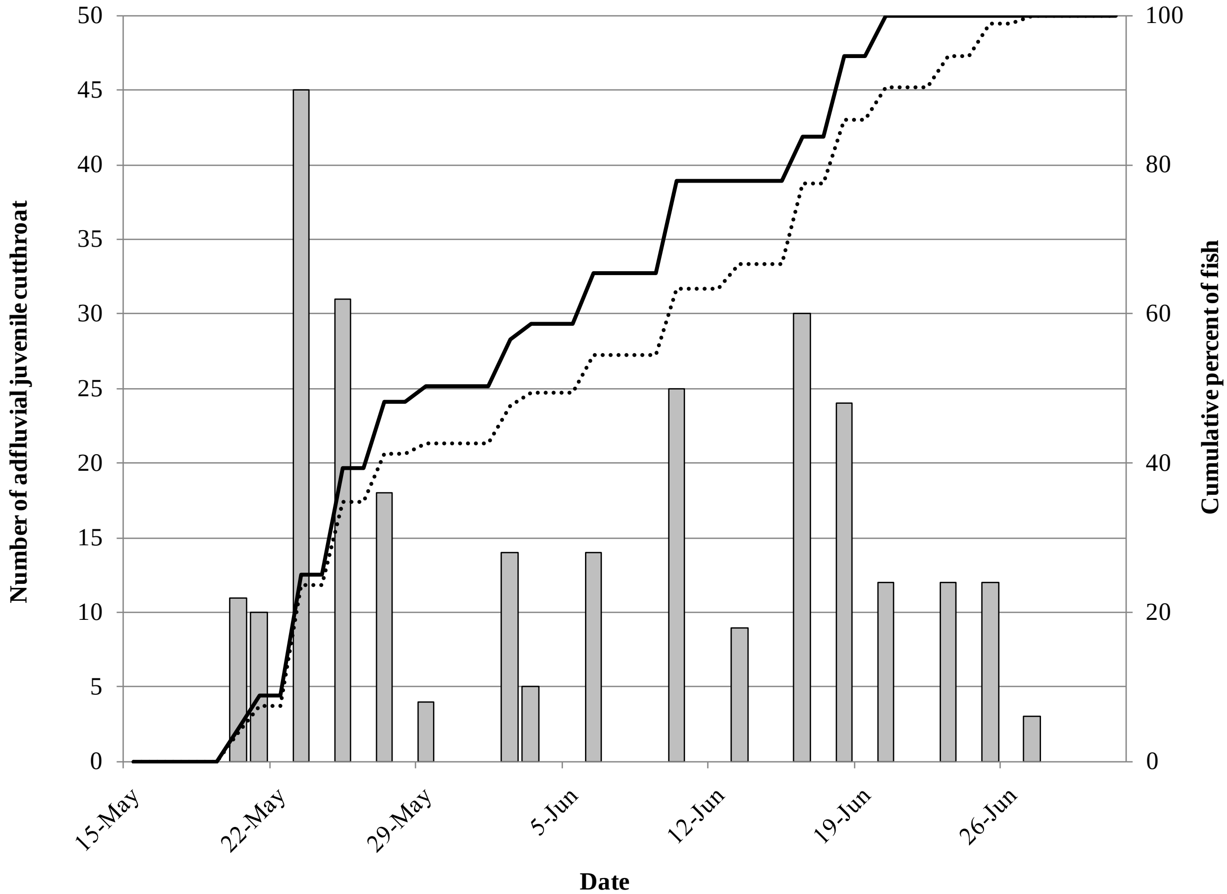

Figure 11. Timing of juvenile adfluvial cutthroat trout captured in the downriver trap during their outmigration in Benewah Creek, 2008. Numbers of juveniles (gray bars) along with the cumulative distribution curves of all captured juveniles (dotted line) and PIT-tagged juveniles (solid line) are presented.

3.3.1.5 Trout abundances at surveyed index sites

Time constraints during the monitoring season permitted only a subsample of index sites to be surveyed within each watershed. Fifteen, ten, twenty, and twenty-three sites were sampled in Alder, Evans, Lake, and Benewah watersheds, respectively. In Alder and Evans creeks, sites were subsampled to ensure adequate longitudinal spatial coverage in both mainstem and tributary habitats. In Lake and Benewah creeks, mainstem sites were subsampled to ensure a representative spatial distribution. However, all tributary index sites in these two systems were surveyed, except for those in Coon Creek in the Benewah watershed where lack of water precluded sampling. In addition, all mainstem sites in the upper Benewah watershed that served as either control or treatment sites for evaluating the effectiveness of restoration activities were sampled. 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 constrained to lower mainstem reaches with low overall abundances, a result not dissimilar to that documented in previous annual surveys (Table 7). Abundance estimates of cutthroat trout in 2008 were less than 3.0 at index sites upstream of site 5, with an absence of fish in surveys along the North Fork tributary and in the uppermost reaches of the Alder mainstem. This result is consistent with the general pattern observed over the last 4-6 years. Though abundances of age 1+ fish in the lower mainstem reach (i.e., sites 1-5) were generally greater than their respective six-year averages and suggested an increasing trend over the last four years, abundances were still low with only one estimate exceeding 10.0 fish.

Conversely, brook trout in the Alder Creek watershed were much more abundant than cutthroat trout (Table 8). Abundance estimates for age 1+ brook trout often exceeded 10.0 at many of the sites distributed in the upper reaches of the watershed, with abundances greater than 20.0 estimated for several of these sites. In addition, age-0 fish were often abundant at sites that had elevated abundances of age 1+ fish, often comprising 20-67% of the total abundance estimate. For many of the sites with high abundance estimates of age 1+ fish, values in 2008 were generally similar to their respective 6-year averages. However, decreasing or stable trends in abundance over the last 4 years were apparent over much of the upper watershed. Notably, low abundances for brook trout were estimated in lower mainstem reaches where cutthroat trout were predominantly captured.

In the Benewah watershed, cutthroat trout were captured in high numbers primarily in tributary reaches, a result consistent with that found in prior survey years (Table 9). Abundance estimates of age 1+ fish at all but one of the index sites in sampled tributaries exceeded 10.0, with abundances greater than 20.0 estimated at several of the sites in the two upper forks and at sites in both Bull and Windfall creeks. For many of the sites, age 1+ abundance comprised over 75% of the total abundance estimate; age-0 fish constituted a large percentage only at site 2 in Bull Creek 2 and site 1 in Whitetail Creek. Abundances of age 1+ fish at tributary sites in 2008 were generally greater than their respective 6-year averages, with relatively strong increasing trends displayed at most sites over the last four years. In contrast to the patterns displayed in the tributaries, abundances of age 1+ fish at mainstem sites were low, with estimated values typically less than 10.0 fish. Only at the two uppermost mainstem sites was there an increasing trend detected in age 1+ fish abundance.

Table 7. Abundances of cutthroat trout captured at survey sites in the Alder Creek watershed. Ordering of sites corresponds to relative longitudinal position in the watershed from downstream to upstream. Sites were sampled in 2008 if values for total number of captured fish of all ages are displayed. Abundance estimates without associated confidence intervals were obtained by summing total fish captured over all passes. Abundance trend indicators of '+', '++', and '+++' indicate an increasing slope of 0.5-2.0, 2.0-5.0, and >5.0, respectively; negative sign combinations are analogous for decreasing trends. For trends between -0.5 and 0.5, a 'o' was assigned. Trends were calculated from data collected since 2005.

		All ages		Age 1+ metrics				
Stream	Index site	Total captured 2008	2008 abundance estimate	Total captured 2008	2008 abundance estimate	95% CI for 2008	Mean abundance, 2002-2007 (n)	Trend indicator
Alder	1	0	0.0	0	0.0	.	0.0 (5)	o
Alder	2	8	8.0	8	8.0	.	1.6 (5)	+ +
Alder	3	3	3.0	2	2.0	2 - 2	2.6 (5)	o
Alder	4	18	18.0	18	18.0	18 - 18.1	9.6 (5)	+ +
Alder	5	9	9.0	6	6.0	6 - 6	4.4 (6)	+ +
Alder	6	.	.	.	.	.	5.0 (6)	+
Alder	7	3	3.0	3	3.0	3 - 3	3.2 (6)	o
Alder	8	.	.	.	.	.	7.1 (6)	+
Alder	9	2	2.0	2	2.0	2 - 2	4.2 (6)	-
Alder	10	.	.	.	.	.	2.0 (6)	o
Alder	11	1	1.0	1	1.0	1 - 1	2.4 (5)	o
Alder	12	.	.	.	.	.	0.7 (6)	o
Alder	13	2	2.2	2	2.2	2 - 3.6	0.2 (6)	+
Alder	14	.	.	.	.	.	0.8 (6)	+
Alder	15	0	0.0	0	0.0	.	0.0 (6)	o
Alder	16	.	.	.	.	.	0.2 (6)	+
Alder	17	0	0.0	0	0.0	.	0.4 (5)	o
North Fork	1	.	.	.	.	.	0.2 (6)	+
North Fork	2	0	0.0	0	0.0	.	0.3 (6)	o
North Fork	3	.	.	.	.	.	0.0 (6)	o
North Fork	4	0	0.0	0	0.0	.	0.0 (6)	o
North Fork	5	.	.	.	.	.	0.0 (6)	o
North Fork	6	0	0.0	0	0.0	.	0.0 (6)	o
North Fork	7	.	.	.	.	.	0.0 (6)	o
North Fork	8	0	0.0	0	0.0	.	0.0 (6)	o

Abundances of age 1+ brook trout in both mainstem and tributary reaches in the Benewah watershed were generally low, with estimated values not exceeding 5.0 at most index sites (Table 10). The interpretation of the abundance and distribution patterns was not appreciably changed when fish of all ages were included in the abundance estimates. Consistent with previous surveys, brook trout were not captured at lower mainstem sites in 2008. In addition, decreasing trends were observed at many of the index sites in the upper tributaries, especially along both the South and West Forks. Positive trends and elevated numbers of age 1+ brook trout were only observed at the index site in lower Schoolhouse Creek (where a reliable estimate could not be generated) and site 16 in the upper mainstem. However, the abundance estimate of 12.6 at site 16, a liberal estimate for age1+ fish given that lengths of fish were not recorded, is not any greater than the 6-year average (note that the trend indicator was only calculated over the period from 2005-2007).

Table 8. Abundances of brook trout captured at survey sites in the Alder Creek watershed. Ordering of sites corresponds to relative longitudinal position in the watershed from downstream to upstream. Sites were sampled in 2008 if values for total number of captured fish of all ages are displayed. Abundance estimates without associated confidence intervals were obtained by summing total fish captured over all passes. Abundance trend indicators of '+', '++', and '+++' indicate an increasing slope of 0.5-2.0, 2.0-5.0, and >5.0, respectively; negative sign combinations are analogous for decreasing trends. For trends between -0.5 and 0.5, a 'o' was assigned. Trends were calculated from data collected since 2005.

		All ages		Age 1+ metrics				
Stream	**Index site**	**Total captured 2008**	**2008 abundance estimate**	**Total captured 2008**	**2008 abundance estimate**	**95% CI for 2008**	**Mean abundance, 2002-2007 (n)**	**Trend indicator**
Alder	1	0	0.0	0	0.0	.	0.0 (5)	o
Alder	2	1	1.0	1	1.0	.	1.0 (5)	o
Alder	3	0	0.0	0	0.0	.	0.4 (5)	o
Alder	4	4	4.4	4	4.4	4 - 6.4	2.7 (5)	o
Alder	5	5	5.0	5	5.0	5 - 5.4	8.2 (6)	o
Alder	6	.	.	.	.	.	5.2 (6)	- - -
Alder	7	7	7.4	7	7.4	7 - 9.2	2.1 (6)	+
Alder	8	.	.	.	.	.	6.6 (5)	- - -
Alder	9	36	39.9	26	30.9	26 - 41.1	10.5 (6)	+ + +
Alder	10	.	.	.	.	.	15.0 (6)	- - -
Alder	11	15	15.1	12	12.1	12 - 13	21.6 (5)	- - -
Alder	12	.	.	.	.	.	26.0 (6)	- - -
Alder	13	79	85.2	57	58.9	57 - 62.7	46.2 (6)	- - -
Alder	14	.	.	.	.	.	50.9 (6)	+
Alder	15	73	79.2	23	28.5	23 - 40.7	29.9 (6)	o
Alder	16	.	.	.	.	.	19.0 (6)	+
Alder	17	27	30.5	8	8.0	.	34.6 (5)	- - -
North Fork	1	.	.	.	.	.	27.0 (6)	- -
North Fork	2	47	48.0	16	16.6	16 - 18.6	21.9 (6)	- - -
North Fork	3	.	.	.	.	.	13.9 (6)	- - -
North Fork	4	63	66.8	20	21.8	20 - 26.4	20.0 (6)	+ +
North Fork	5	.	.	.	.	.	25.9 (6)	- -
North Fork	6	28	29.2	22	23.0	22 - 25.9	23.2 (6)	o
North Fork	7	.	.	.	.	.	19.8 (6)	- - -
North Fork	8	11	11.7	5	5.2	5 - 6.5	11.2 (6)	- -

Similar to the Benewah Creek watershed, abundances of age 1+ cutthroat trout in Lake Creek were greatest in sampled tributaries in 2008 (i.e., Bozard and West Fork creeks, and Lake Creek upstream of site 10), but only in the uppermost reaches of the tributaries (Table 11). Abundance estimates of age 1+ fish exceeded 20.0 at Lake Creek 14, sites 4 and 5 in the West Fork subdrainage, and sites 3, 4, and the East Fork site in the Bozard subdrainage. In comparison, numbers of captured fish and abundance estimates were substantially lower at all other tributary index sites, with estimates of age 1+ fish not exceeding 7.0. In addition, trends were neither markedly increasing nor decreasing across those tributary sites that had elevated abundances, but were apparently stable over the last four years. Abundance estimates at mainstem sites were generally lower than those estimated in the upper tributaries, with most estimates of age 1+ fish less than 10.0. However, three sites along the lower mainstem reach had elevated estimates greater than 20.0 when captured cutthroat trout of all ages were included.

Table 9. Abundances of cutthroat trout captured at survey sites in the Benewah Creek watershed. Ordering of sites corresponds to relative longitudinal position in the watershed from downstream to upstream. Sites were sampled in 2008 if values for total number of captured fish of all ages are displayed. Abundance estimates without associated confidence intervals were obtained by summing total fish captured over all passes. Abundance trend indicators of '+', '++', and '+++' indicate an increasing slope of 0.5-2.0, 2.0-5.0, and >5.0, respectively; negative sign combinations are analogous for decreasing trends. For trends between -0.5 and 0.5, a 'o' was assigned. Trends were calculated from data collected since 2005.

		All ages		Age 1+ metrics				
Stream	**Index site**	**Total captured 2008**	**2008 abundance estimate**	**Total captured 2008**	**2008 abundance estimate**	**95% CI for 2008**	**Mean abundance, 2002-2007 (n)**	**Trend indicator**
Benewah	1	.	.	.	.	.	0.5 (6)	o
Benewah	2	4	4.0	4	4.0	4 - 4.5	2.2 (6)	o
Benewah	3	.	.	.	.	.	4.4 (6)	+
Benewah	4	5	5.0	5	5.0	5 - 5	6.3 (6)	-
Benewah	5	.	.	.	.	.	4.4 (6)	- -
Benewah	6	9	9.0	9	9.0	.	6.0 (6)	-
Benewah	7	.	.	.	.	.	7.8 (6)	- - -
Benewah	8	0	0.0	0	0.0	.	1.4 (6)	-
Benewah	9	.	.	.	.	.	1.5 (6)	-
Benewah	10	.	.	.	.	.	0.3 (6)	+
Benewah	11	.	.	.	.	.	1.0 (5)	+
Benewah	12	18	. [a]	17	. [a]	.	5.4 (5)	[b]
Benewah	13	.	.	.	.	.	3.7 (6)	o
Benewah	14L	.	.	.	.	.	. [c]	[b]
Benewah	14U	11	11.4	8	8.1	8 - 8.8	4.1 (5)	o
Benewah	14	3	3.1	2	2.2	2 - 3.6	1.5 (6)	o
Benewah	15	3	3.0	2	2.0	.	0.7 (6)	o
Benewah	16	8	8.0	8	8.0	.	4.2 (6)	+
Benewah	17	14	14.8	11	12.3	11 - 16.7	3.0 (5)	+ +
Coon	1	.	.	.	.	.	. [c]	[b]
Coon	2	.	.	.	.	.	. [c]	[b]
Coon	3	.	.	.	.	.	3.3 (4)	[b]
Bull	1	43	49.5	39	47.7	39 - 62.4	34.6 (5)	- -
Bull	2	35	38.4	17	18.8	17 - 23.8	9.1 (6)	+ +
Whitetail	1	44	. [a]	7	. [a]	.	4.0 (4)	[b]
Whitetail	2	12	12.1	12	12.1	12 - 12.8	9.0 (5)	+ + +
Windfall	1	14	16.0	14	16.0	14 - 21.7	12.7 (5)	+ +
Windfall	2	40	62.5	36	52.9	36 - 88.1	22.2 (6)	+ + +
Schoolhouse	1	15	16.0	12	13.5	12 - 18.6	4.0 (6)	+ +
Schoolhouse	2	7	7.0	5	5.0	5 - 5	7.0 (6)	+
South Fork	1	17	17.3	11	11.4	11 - 13	4.9 (6)	+ +
South Fork	2	23	25.4	23	25.4	23 - 30.9	15.8 (6)	+ +
South Fork	3	22	22.1	22	22.1	22 - 22.8	9.5 (5)	+ + +
West Fork	1	27	27.1	21	21.0	21 - 21.5	11.1 (6)	+ +
West Fork	2	17	17.0	16	16.0	16 - 16.1	9.1 (6)	+ + +

[a] Reliable abundance estimate could not be generated using the depletion model

[b] Insufficient data existed to compute trend value

[c] Insufficient data existed to compute mean value

Table 10. Abundances of brook trout captured at survey sites in the Benewah Creek watershed. Ordering of sites corresponds to relative longitudinal position in the watershed from downstream to upstream. Sites were sampled in 2008 if values for total number of captured fish of all ages are displayed. Abundance estimates without associated confidence intervals were obtained by summing total fish captured over all passes. Abundance trend indicators of '+', '++', and '+++' indicate an increasing slope of 0.5-2.0, 2.0-5.0, and >5.0, respectively; negative sign combinations are analogous for decreasing trends. For trends between -0.5 and 0.5, a 'o' was assigned. Trends were calculated from data collected since 2005.

		All ages		Age 1+ metrics				
Stream	**Index site**	**Total captured 2008**	**2008 abundance estimate**	**Total captured 2008**	**2008 abundance estimate**	**95% CI for 2008**	**Mean abundance, 2002-2007 (n)**	**Trend indicator**
Benewah	1	.	.	.	.	.	0.0 (6)	o
Benewah	2	0	0.0	0	0.0	.	0.2 (6)	o
Benewah	3	.	.	.	.	.	0.5 (6)	o
Benewah	4	0	0.0	0	0.0	.	0.0 (6)	o
Benewah	5	.	.	.	.	.	0.0 (6)	o
Benewah	6	0	0.0	0	0.0	.	0.0 (6)	o
Benewah	7	.	.	.	.	.	0.0 (6)	o
Benewah	8	0	0.0	0	0.0	.	0.0 (6)	o
Benewah	9	.	.	.	.	.	0.0 (6)	o
Benewah	10	.	.	.	.	.	0.0 (6)	o
Benewah	11	.	.	.	.	.	0.2 (5)	o
Benewah	12	0	0.0	0	0.0	.	0.0 (5)	o
Benewah	13	.	.	.	.	.	0.0 (6)	o
Benewah	14L	.	.	.	.	.	2.6 (5)	- -
Benewah	14U	4	4.0	2	2.0	.	4.3 (5)	- -
Benewah	14	1	1.0	0	0.0	.	1.5 (6)	-
Benewah	15	3	3.0	2	2.0	.	0.6 (5)	o
Benewah	16	12 [a]	12.6	.	.	.	12.4 (6)	+ + +
Benewah	17	11	. [b]	3	3.0	.	8.6 (4)	[c]
Coon	1	.	.	.	.	.	. [d]	[c]
Coon	2	.	.	.	.	.	. [d]	[c]
Coon	3	.	.	.	.	.	0.3 (4)	[c]
Bull	1	1	1.0	1	1.0	1 - 1	0.7 (6)	o
Bull	2	0	0.0	0	0.0	.	0.0 (6)	o
Whitetail	1	1	1.0	1	1.0	1 - 1	3.0 (4)	- -
Whitetail	2	0	0.0	0	0.0	.	0.4 (5)	-
Windfall	1	3	3.0	3	3.0	3 - 3	5.8 (5)	o
Windfall	2	4	4.0	2	2.0	2 - 2	2.6 (6)	+
Schoolhouse	1	21	. [b]	14	. [b]	.	7.8 (6)	+ +
Schoolhouse	2	0	0.0	0	0.0	.	1.2 (6)	o
South Fork	1	8	8.3	5	5.0	5 - 5.4	6.8 (5)	-
South Fork	2	1	1.0	0	0.0	.	2.2 (6)	-
South Fork	3	7	8.0	2	2.0	2 - 2	2.8 (5)	-
West Fork	1	1	1.0	1	1.0	.	15.9 (6)	- - -
West Fork	2	6	6.2	6	6.2	6 - 7.1	7.2 (6)	+

[a] Fish lengths not recorded

[b] Reliable abundance estimate could not be generated using the depletion model

[c] Insufficient data existed to compute trend value

[d] Insufficient data existed to compute mean value

Table 11. Abundances of cutthroat trout captured at survey sites in the Lake Creek watershed. Ordering of sites corresponds to relative longitudinal position in the watershed from downstream to upstream. Sites were sampled in 2008 if values for total number of captured fish of all ages are displayed. Abundance estimates without associated confidence intervals were obtained by summing total fish captured over all passes. Abundance trend indicators of '+', '++', and '+++' indicate an increasing slope of 0.5-2.0, 2.0-5.0, and >5.0, respectively; negative sign combinations are analogous for decreasing trends. For trends between -0.5 and 0.5, a 'o' was assigned. Trends were calculated from data collected since 2005.

		All ages		Age 1+ metrics				
Stream	**Index site**	**Total captured 2008**	**2008 abundance estimate**	**Total captured 2008**	**2008 abundance estimate**	**95% CI for 2008**	**Mean abundance, 2002-2007 (n)**	**Trend indicator**
Lake	1	24	30.3	8	9.6	8 - 15.6	11.0 (5)	o
Lake	2	.	.	.	.	.	10.4 (6)	- - -
Lake	3	20	22.9	7	7.8	7 - 11.3	10.9 (6)	- - -
Lake	4	.	.	.	.	.	10.5 (5)	- -
Lake	5	17	22.1	12	16.0	12 - 28.3	5.7 (5)	+
Lake	6	.	.	.	.	.	14.6 (6)	- - -
Lake	7	10	10.0	9	9.0	.	9.7 (6)	- -
Lake	7U	11	11.0	7	7.0	.	6.8 (5)	+
Lake	8L	.	.	.	.	.	5.2 (5)	+ +
Lake	8U	.	.	.	.	.	3.4 (5)	o
Lake	9L	.	.	.	.	.	4.8 (4)	[a]
Lake	9	5	5.2	3	3.1	3 - 3.8	3.9 (5)	-
Lake	10L	.	.	.	.	.	5.9 (5)	+
Lake	10	11	11.0	10	10.0	.	7.3 (6)	+ +
Lake	11	3	3.0	3	3.0	.	2.9 (6)	o
Lake	12 [b]	0	.	0	.	.	2.6 (6)	+
Lake	13	.	.	.	.	.	0.0 (4)	[a]
Lake	14	25	25.4	20	20.2	20 - 21.3	. [c]	[a]
Bozard	1	10	10.0	7	7.0	.	6.3 (6)	+
Bozard	2	6	6.5	6	6.5	6 - 9	6.8 (6)	-
Bozard	3	22	24.0	22	24.0	22 - 28.8	30.6 (6)	o
Bozard	4	26	27.3	23	24.3	23 - 27.7	28.8 (5)	-
East Fork Bozard	1	44	45.0	38	38.8	38 - 41.1	45.9 (6)	o
West Fork	1	1 [d]	1.0	.	.	.	1.7 (6)	-
West Fork	2 [b]	1	.	1	.	.	7.5 (5)	-
West Fork	3	1	1.0	1	1.0	1 - 1	1.0 (4)	[a]
West Fork	4	25	28.3	22	24.0	22 - 28.8	17.2 (6)	o
West Fork	5	26	32.3	20	21.8	20 - 26.4	24.9 (5)	o

[a] Insufficient data existed to compute trend value

[b] Only one pass conducted

[c] Insufficient data existed to compute mean value

[d] Fish length not recorded

In the Evans Creek watershed, cutthroat trout were found to be proportionately distributed across all sampled index sites other than the lowermost mainstem site and the site in the Rainbow Fork (Table 12). Abundance estimates of age 1+ fish at these sites exceeded 16.0, and at three of the eight sites numbers of captured age 1+ fish exceeded 40.0 (estimates at sites 3 and 6 were conservative given that numbers were not depleted substantially over subsequent passes to generate a reliable estimate). Moreover, age 1+ fish constituted 92-100% of the total abundance estimates for all these sites. In addition, strong, increasing trends over the last four years were apparent at most of the sites across the watershed.

Table 12. Abundances of cutthroat trout captured at survey sites in the Evans Creek watershed. Ordering of sites corresponds to relative longitudinal position in the watershed from downstream to upstream. Sites were sampled in 2008 if values for total number of captured fish of all ages are displayed. Abundance estimates without associated confidence intervals were obtained by summing total fish captured over all passes. Abundance trend indicators of '+', '++', and '+++' indicate an increasing slope of 0.5-2.0, 2.0-5.0, and >5.0, respectively; negative sign combinations are analogous for decreasing trends. For trends between -0.5 and 0.5, a 'o' was assigned. Trends were calculated from data collected since 2005.

		All ages		Age 1+ metrics				
Stream	**Index site**	**Total captured 2008**	**2008 abundance estimate**	**Total captured 2008**	**2008 abundance estimate**	**95% CI for 2008**	**Mean abundance, 2002-2007 (n)**	**Trend indicator**
Evans	1	3	3.0	0	0.0	.	2.2 (6)	-
Evans	2	.	.	.	.	.	13.0 (5)	[a]
Evans	3	42	. [b]	41	. [b]	.	23.7 (5)	+ + +
Evans	4	.	.	.	.	.	13.4 (6)	+ +
Evans	5	.	.	.	.	.	14.3 (6)	+ + +
Evans	6	55	. [b]	54	. [b]	.	32.6 (6)	+ + +
Evans	7	21	. [b]	21	. [b]	.	13.6 (6)	+ + +
Evans	8	.	.	.	.	.	15.3 (5)	+ + +
Evans	9	18	24.9	17	22.8	17 - 38.1	17.2 (5)	+ + +
Evans	10	.	.	.	.	.	12.6 (6)	+ + +
Evans	11	16	16.3	16	16.3	16 - 17.8	11.9 (5)	+ +
Evans	12	.	.	.	.	.	29.9 (6)	+ + +
Evans	13	44	47.9	42	45.2	42 - 51	21.8 (6)	+ + +
Evans	14	.	.	.	.	.	20.6 (6)	+ + +
Evans	15	19	19.6	19	19.6	19 - 21.8	10.2 (6)	+ +
Evans	16	.	.	.	.	.	0.8 (5)	o
South Fork	1	.	.	.	.	.	7.8 (6)	+ +
South Fork	2	.	.	.	.	.	8.7 (6)	+ +
East Fork	1	16	16.0	16	16.0	16 - 16.1	21.0 (6)	+ +
Rainbow Fork	1	0	0.0	0	0.0	.	8.7 (6)	-

[a] Insufficient data existed to compute trend value

[b] Reliable abundance estimate could not be generated using the depletion model

3.3.2 Trend and status monitoring – Stream temperatures

3.3.2.1 Benewah Creek temperatures

Ambient summer stream temperatures generally increased downstream over the 6.4 km section of the Benewah mainstem from the mouth of Schoolhouse Creek to 9-mile bridge in 2008, though the longitudinal temperature change was more gradual in upper than in lower reaches (Table 13). For example, monthly means of daily mean temperatures recorded by data loggers only increased 1.3ºC from 13.8 to 15.1ºC in July and 1.1ºC from 13.3 to 14.4ºC in August over the uppermost 3.2 km reach. In comparison, temperatures increased 3.1ºC from 15.1 to 18.21ºC in July and 3.2ºC from 14.4 to 17.6ºC in August along the lowermost 3.2 km reach. Similarly, monthly means of maximum daily temperatures during the two months increased only 0.7-0.8ºC along the upper reach, but 4.0-4.1ºC over the lower reach. Furthermore, monthly means of daily maximum temperatures were at least 17ºC for all loggers positioned downstream but not upstream of the logger located 3.2.km upstream of 9-mile bridge.

Similar to the trends observed for mean and maximum temperatures, the percentage of time logged water temperatures exceeded 17ºC during July and August in 2008 were higher and increased more rapidly along the lower half than along the upper half of the 6.4 km reach of the upper Benewah mainstem (Table 13). Along the uppermost 3.2 km, daily stream temperatures exceeded 17ºC less than 11% of the time during July and August with percentages increasing downstream by only 4.0-7.7%. Conversely, percentages in the lowermost 3.2 km increased downstream over 1.6 km increments by 27% and then 30% in July, and by 18% and then 29% in August. Ambient stream temperatures exceeded 17ºC during July and August over 50% of the time at the logger positioned immediately upstream of 9-mile bridge.

Summer stream temperatures in the upper Benewah mainstem, however, were cooler in 2008 than in 2007 (Table 14). July averages of daily mean and maximum temperatures, calculated for loggers located along the 6.4 km section upriver of 9-mile bridge, were typically 1.9-3.9ºC less in 2008 than in 2007. In addition, the percentage of time logged water temperatures exceeded 17ºC in 2008 was typically 40-50% lower than that recorded in 2007 for loggers located along this upper mainstem stretch.

Temperature indices were generally cooler in lower reaches of monitored tributaries than in mainstem reaches in the upper Benewah watershed in 2008 (Table 13). In July and August, monthly means of daily mean temperatures respectively ranged from 12.2 to 13.9ºC and from 11.3 to 12.3ºC, and monthly means of daily maximums respectively ranged from 13.5 to 15.5ºC and from 11.7 to 13.8ºC. In addition, water temperatures rarely exceeded 17ºC in monitored tributaries during the summer of 2008. Recorded temperatures were greater than 17ºC less than 5% of the time and only in July in lower Whitetail and Windfall creeks.

In addition to the tributaries in the upper mainstem of the Benewah watershed, various springbrooks also displayed temperature signatures during summer months in 2008 that were much cooler than those recorded in adjacent mainstem habitats (Table 13). At monitored connected springbrooks located 1.3, 2.5, and 4.2 km upstream of 9-mile bridge, monthly averages of mean daily July temperatures were respectively 2.6, 5.1, and 4.5ºC less than those in proximate (i.e., less than 0.2 km) main channel reaches. Similarly, monthly means of daily July maximums were respectively 4.2, 7.0, and 5.0 less than those in nearby main channel reaches. August results were similar though the discrepancies between the springbrooks and the adjacent main channel habitats were not as large. Notably, the lowest monthly averages for both temperature metrics were recorded in the monitored isolated springbrook that was located in the unconstrained valley reach, with values never exceeding 10.0 in July and August. During the summer of 2008, temperatures at connected and isolated springbrooks never exceeded 17ºC.

3.3.2.2 Lake Creek temperatures

Ambient stream temperatures were generally cool throughout most of the upper Lake Creek watershed during the summer of 2008 (Table 15). Monthly means of mean daily temperatures in July and August ranged from 13.8 to 15.4ºC for loggers located in reaches proximate to the confluence of the three upper forks. Loggers located further upstream in the Bozard subdrainage had calculated monthly means during these two months that ranged from 12.1 to 12.6ºC. Similar patterns emerged for the calculated monthly averages of daily maximum values. Averages for loggers located near the confluence of the three forks ranged from 15.0 to 16.6ºC, whereas averages for the group of loggers positioned further up the Bozard subdrainage ranged from 13.4 to 14.4ºC. However, summer temperatures recorded in the reach of the mainstem near the old H95 bridge (in close proximity to the location of the migrant traps) were much warmer than those recorded upstream. Monthly averages of daily maximum temperatures were approximately 2.5ºC higher than the highest values calculated for upriver loggers during July and August.

The percentage of time recorded temperatures exceeded 17ºC was also generally low across the upper Lake Creek watershed during the summer of 2008 (Table 15). Temperatures were greater than 17ºC in July and August less than 10 and 2% of the time for groups of loggers positioned near the confluence of the three forks and in the upper Bozard subdrainage, respectively. In comparison, temperatures exceeded 17ºC between 32.5 and 42.7% of the time during these two months in the lower mainstem reach.

Similar to the results documented in the upper Benewah watershed, temperatures in the upper Lake Creek watershed were much cooler in 2008 than in 2007 (Table 16). July averages of daily mean temperatures in monitored reaches were 1.2-3.2ºC lower in 2008 than in 2007. Similarly, July averages of daily maximum temperatures were 2.5-4.6ºC lower in 2008 than in 2007 for all but one of the loggers. In addition, the percent of time in which July water temperatures exceeded 17ºC was much higher in 2007 than in 2008. For example, percentages were less than 10 in 2008, but ranged from 41 to 59 in 2007, for those loggers located near the confluence of the three upper forks.

Table 13. Summary statistics for July and August water temperatures recorded by data loggers located in the upper Benewah watershed in 2008. Rkm refers to the number of river kilometers above 9-mile bridge; loggers placed in tributaries were located < 0.1 km from their confluence with the Benewah mainstem, and in this case, Rkm refers to the relative position of the tributary mouth to 9-mile bridge. 17°C was considered the upper 95% confidence interval limit for optimal growth for westslope cutthroat trout (Bear et al. 2007).

Stream	**Site**	**Rkm**	**Mean of daily means**	**Mean of daily maximums**	**Percent time > 17°C**
		July temperatures			
Benewah	Main channel	0.1	18.2	20.8	68.1
Benewah	Main channel	0.4	18.0	20.2	68.9
Benewah	Main channel	1.1	16.8	19.5	45.8
Benewah	Main channel	1.6	16.4	18.9	37.5
Benewah	Main channel	2.6	15.5	17.8	23.5
Benewah	Main channel	3.2	15.1	16.8	10.5
Benewah	Main channel	3.8	14.8	16.2	4.5
Benewah	Main channel	4.2	14.8	16.1	3.3
Benewah	Main channel	5.2	14.7	16.3	4.8
Benewah	Main channel	5.4	14.2	16.4	4.7
Benewah	Main channel	6.0	13.9	16.2	3.5
Benewah	Main channel	6.4	13.8	16.1	2.8
Benewah	Springbrook	1.3	14.2	15.3	0.0
Benewah	Springbrook	2.5	10.4	10.8	0.0
Benewah [a]	Springbrook	3.8	8.2	8.3	0.0
Benewah	Springbrook	4.2	10.3	11.1	0.0
Whitetail	Tributary	1.1	12.6	14.2	4.8
Windfall	Tributary	5.3	13.9	15.5	2.3
Schoolhouse	Tributary	6.4	13.1	14.5	0.0
Unnamed tributary	Tributary	7.1	12.2	13.5	0.0
		August temperatures			
Benewah	Main channel	0.1	17.6	20.3	56.6
Benewah [b]	Main channel	0.4	-	-	-
Benewah	Main channel	1.1	15.9	18.1	32.4
Benewah	Main channel	1.6	15.8	18.4	27.5
Benewah	Main channel	2.6	14.8	17.0	19.2
Benewah	Main channel	3.2	14.4	16.2	9.1
Benewah	Main channel	3.8	14.1	15.0	1.5
Benewah	Main channel	4.2	14.0	15.3	3.6
Benewah	Main channel	5.2	14.2	15.5	5.6
Benewah	Main channel	5.4	13.6	15.3	3.4
Benewah	Main channel	6.0	13.5	15.6	6.8
Benewah	Main channel	6.4	13.3	15.4	5.1
Benewah	Springbrook	1.3	13.4	14.5	0.0
Benewah	Springbrook	2.5	11.2	11.5	0.0
Benewah [a]	Springbrook	3.8	9.0	9.2	0.0
Benewah	Springbrook	4.2	11.2	11.9	0.0
Whitetail	Tributary	1.1	11.3	11.7	0.0
Windfall	Tributary	5.3	12.3	13.3	0.0
Schoolhouse	Tributary	6.4	12.3	13.8	0.0
Unnamed tributary	Tributary	7.1	12.0	13.2	0.0

[a] Springbrook was isolated from the main channel

[b] Data could not be retrieved from the logger

Table 14. Comparison of summary statistics between 2007 and 2008 for July temperatures recorded by data loggers in upper Benewah mainstem reaches. Rkm refers to the number of river kilometers above 9-mile bridge. 17°C was considered the upper 95% confidence interval limit for optimal growth for westslope cutthroat trout (Bear et al. 2007).

	Mean of daily means		Mean of daily maximums		Percent time > 17°C	
Rkm	**2007**	**2008**	**2007**	**2008**	**2007**	**2008**
0.1	21.4	18.2	24.7	20.8	95.2	68.1
1.1	18.8	16.8	20.5	19.5	86.6	45.8
1.6	18.5	16.4	20.8	18.9	78.9	37.5
2.6	18.6	15.5	21.5	17.8	71.5	23.5
3.2	17.8	15.1	20.3	16.8	64.5	10.5
4.2	17.4	14.8	18.8	16.1	62.3	3.3
5.4	16.7	14.2	18.6	16.4	47.2	4.7
6.4	16.4	13.8	19.1	16.1	42.5	2.8

Table 15. Summary statistics for July and August water temperatures recorded by data loggers located in reaches of the upper mainstem of Lake Creek and of proximate tributaries in 2008. Logger locations are listed in order of relative longitudinal position in the watershed from lowermost to uppermost. 17°C was considered the upper 95% confidence interval limit for optimal growth for westslope cutthroat trout (Bear et al. 2007).

Logger location	**Mean of daily means**	**Mean of daily maximums**	**Percent time > 17°C**
		July temperatures	
Lake Creek mainstem, near old H95 bridge	16.5	19.3	42.7
Lake Creek mainstem, downstream of Bozard Creek confluence	14.8	16.4	6.1
Bozard Creek, upstream of Lake Creek confluence	14.1	15.8	2.2
West Fork Lake Creek, upstream of Lake Creek confluence	15.2	16.6	9.2
Upper Lake Creek, upstream of West Fork confluence	15.4	16.4	9.5
Bozard Creek, downstream of East Fork Bozard confluence	12.3	13.7	0.0
East Fork Bozard, upstream of Bozard Creek confluence	12.1	13.4	0.0
Bozard Creek, upstream of East Fork Bozard confluence	12.6	14.3	0.0
		August temperatures	
Lake Creek mainstem, near old H95 bridge	15.8	18.3	32.5
Lake Creek mainstem, downstream of Bozard Creek confluence	14.1	15.5	6.6
Bozard Creek, upstream of Lake Creek confluence	13.8	16.0	8.6
West Fork Lake Creek, upstream of Lake Creek confluence	13.9	15.0	3.3
Upper Lake Creek, upstream of West Fork confluence	14.3	16.0	6.9
Bozard Creek, downstream of East Fork Bozard confluence	12.4	13.8	0.4
East Fork Bozard, upstream of Bozard Creek confluence	12.3	13.5	0.0
Bozard Creek, upstream of East Fork Bozard confluence	12.6	14.4	1.6

Table 16. Comparison of summary statistics between 2007 and 2008 for July 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°C was considered the upper 95% confidence interval limit for optimal growth for westslope cutthroat trout (Bear et al. 2007).

	Mean of daily means		Mean of daily maximums		Percent time > 17°C	
Logger location	**2007**	**2008**	**2007**	**2008**	**2007**	**2008**
Lake Creek mainstem, near old H95 bridge	19.7	16.5	22.8	19.3	81.6	42.7
Lake Creek mainstem, downstream of Bozard Creek confluence	17.3	14.8	19.5	16.4	58.6	6.1
Bozard Creek, upstream of Lake Creek confluence	17.1	14.1	20.4	15.8	50.3	2.2
West Fork Lake Creek, upstream of Lake Creek confluence	16.5	15.2	17.9	16.6	41.2	9.2
Upper Lake Creek, upstream of West Fork confluence	17.0	15.4	19.5	16.4	48.6	9.5
Bozard Creek, downstream of East Fork Bozard confluence	14.6	12.3	16.4	13.7	8.9	0.0
East Fork Bozard, upstream of Bozard Creek confluence	14.5	12.1	16.0	13.4	5.8	0.0
Bozard Creek, upstream of East Fork Bozard confluence	14.8	12.6	16.9	14.3	14.8	0.0

3.3.3 Trend and status monitoring – Physical habitat attributes

3.3.3.1 Power analysis for regional trends in habitat attributes

Generally, for all habitat metrics and for reaches in both upper mainstem Benewah and Lake Creek tributaries , the power to detect directional trends increased markedly when the duration of the monitoring surveys exceeded 10 years or when the number of monitored sites was doubled from 5 to 10 (Figure 12-16). In addition, for most habitat metrics, generated power curves for sets of sites monitored annually over 10 years were similar to those generated for equivalent sets of sites monitored every 4 years over a period of 12 years. Results for each habitat metric are examined more fully in the following paragraphs in which the likelihood of detecting subtle directional trends of 2-4% with high probability (i.e., 80% power) over a time horizon of ten years was evaluated. A 2-4% annual change over this timeframe amounted to a proportional positive change of 25-50% or negative change of 15-35% in a metric's value. Ten years was considered to be of sufficient length within the context of an adaptive management paradigm for a reasonable evaluation of the progressive success of implemented habitat enhancement actions or of changing habitat conditions in our watersheds.

Percent canopy cover

For five monitored sites, the power to detect subtle positive and negative trends of 4% in percent canopy cover exceeded 80% only when sites were monitored for 15 and 20 years, respectively (Figure 12). When the number of sites was doubled, however, positive subtle trends for both Benewah mainstem and Lake Creek tributary reaches were capable of being detected with high probability in 10 years; negative subtle trends had similar detection probabilities over the same timeframe in tributary but not in mainstem reaches. Similar results were obtained when 10 representative sites were monitored every four years over a period of 12 years (Figure 12). High detection probabilities for subtle positive trends were attainable for both channel types, and negative subtle trends were detectable for tributary reaches. The lack of power to detect subtle negative trends in Benewah mainstem habitats was likely due to relatively low empirical estimates of initial canopy cover (30-55%) and relatively high levels of estimated annual variability (standard deviation, 15-20). When power curves for a set of 10 representative sites monitored every 4 years were simulated with annual variability estimates of 10, small directional trends of 3-4% were capable of being detected with a high likelihood in 12 years, and approached that for prior simulations with empirical estimates of annual variability and a similar cyclical monitoring design that was surveyed over 20 years (Figure 12).

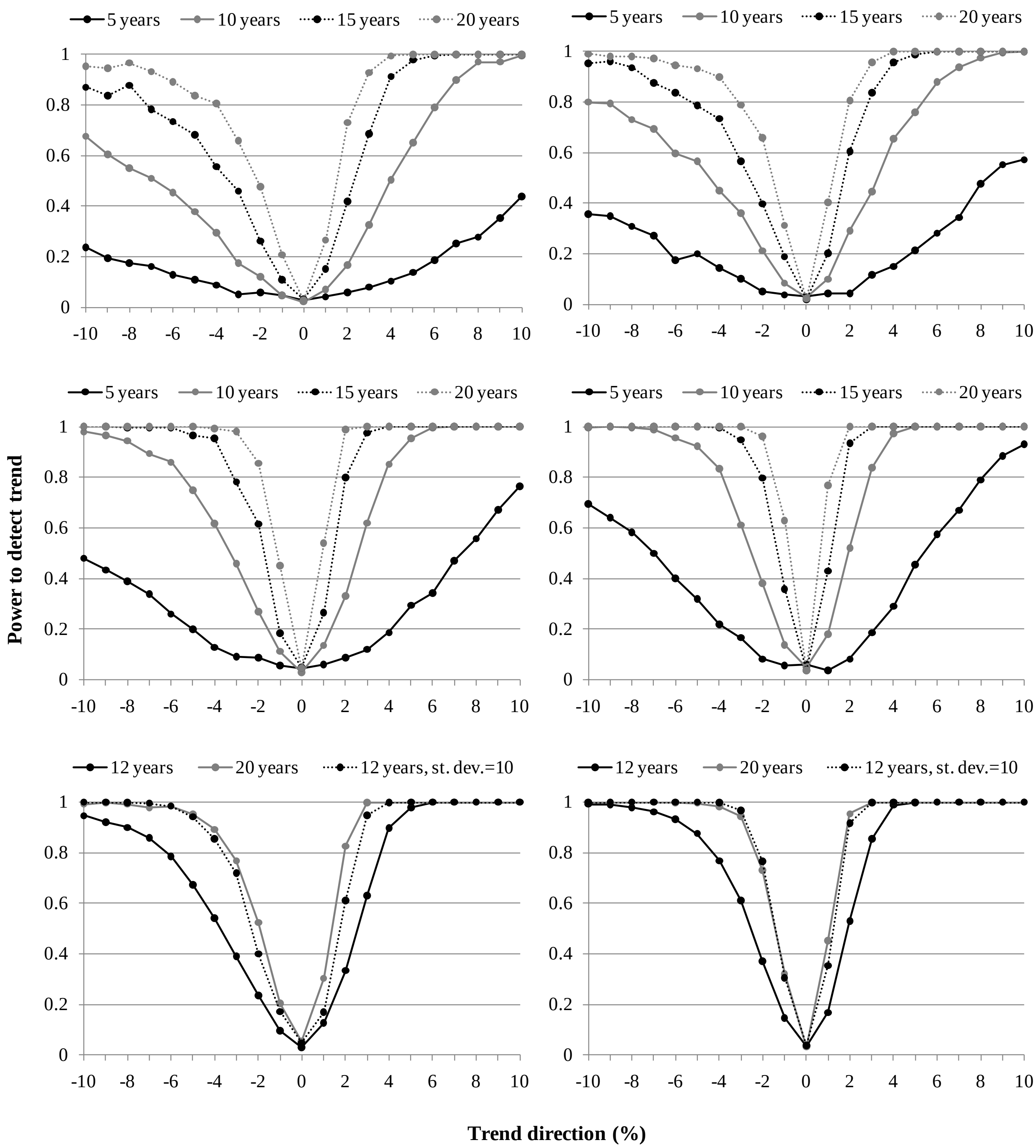

Figure 12. Power analysis to detect increasing and decreasing trends of varying strength (%) in percent canopy cover over various survey durations for sites in mainstem reaches of upper Benewah Creek (left panels) and for sites in tributary reaches of upper Lake Creek (right panels). Upper and middle panels respectively display results for 5 and 10 sites monitored annually, and lower panels display results for 10 sites monitored every 4 years. The standard deviation (st. dev.) was adjusted for two simulations (lower panels) to represent expected annual variability as percent canopy cover approached desired levels.

Percent fines in riffles

Positive subtle trends in percent fines were attainable only after 20 years of monitoring when 5 sites were included in the monitoring design; the power to detect negative trends of any strength never exceeded 80% (Figure 13). When 10 sites were included in the monitoring design, subtle increasing and decreasing trends of 3% were capable of being detected with high likelihood for Lake Creek tributary sites after 10 years. However, for a set of 10 sites in the Benewah mainstem, detection of positive and negative trends of 3% were only attainable after 15 and 20 years of monitoring, respectively. The power to detect subtle trends was preserved for Lake Creek tributary reaches when a set of 10 sites was monitored every 4 years over a period of 12 years (Figure 13). For a similar monitoring design for Benewah mainstem sites, the ability to detect subtle positive trends of less than 5% with high likelihood would be attainable after 12 years; however, the power to detect negative trends of any strength with 80% probability was not possible within this timeframe.

Similar to the canopy cover results, the inability to detect negative trends in percent fines for mainstem Benewah reaches was likely due to relatively low estimates for initial values and relatively high estimates of annual variability. For the four available sample sites in the upper mainstem of Benewah, two had mean estimates of percent fines that were less than 15% (restored sites) and the other two sites had annual variability estimates greater than 10 (unrestored sites). In comparison, mean percent fine estimates ranged between 60 and 85% with site-specific annual variability estimates less than 10 for two-thirds of the available sites in tributary reaches of upper Lake Creek. When initial percent fine estimates were adjusted upwards to values ranging between 25 and 35% and annual variability estimates were set at a value of 10, directional trends of 4-5% could be detected with high probability after 12 years (Figure 13). Similar results were observed when empirical initial percent fine estimates were retained but annual variability estimates were further reduced to values of 5.

Mean residual pool depth

Subtle directional trends of 3-4% in mean residual pool depth were detectable with high probability in both Benewah mainstem and Lake Creek tributary reaches when 5 sites were monitored over a period of 10 years (Figure 14). Although a substantial increase in power was attained when survey duration was increased from 5 to 10 years for both channel types, not much power was gained when the monitoring duration was increased beyond 10 years. High probabilities to detect subtle directional trends were preserved in both channel types when the set of 5 sites was monitored every 4 years over a period of 12 years (Figure 14).

Large woody debris (LWD) availability

For upper mainstem Benewah reaches, subtle positive and negative trends of 3% for LWD counts (# / 100 m) were detectable with high probability when 5 representative sites were monitored over a period of 15 and 20 years, respectively (Figure 15). When the set of representative sites was increased to 10, similar subtle directional trends could be readily detected after only 10 years of monitoring. In addition, high detection probabilities for subtle trends were retained when the set of 10 sites was monitored every 4 years over a period of 12 years (Figure 15).

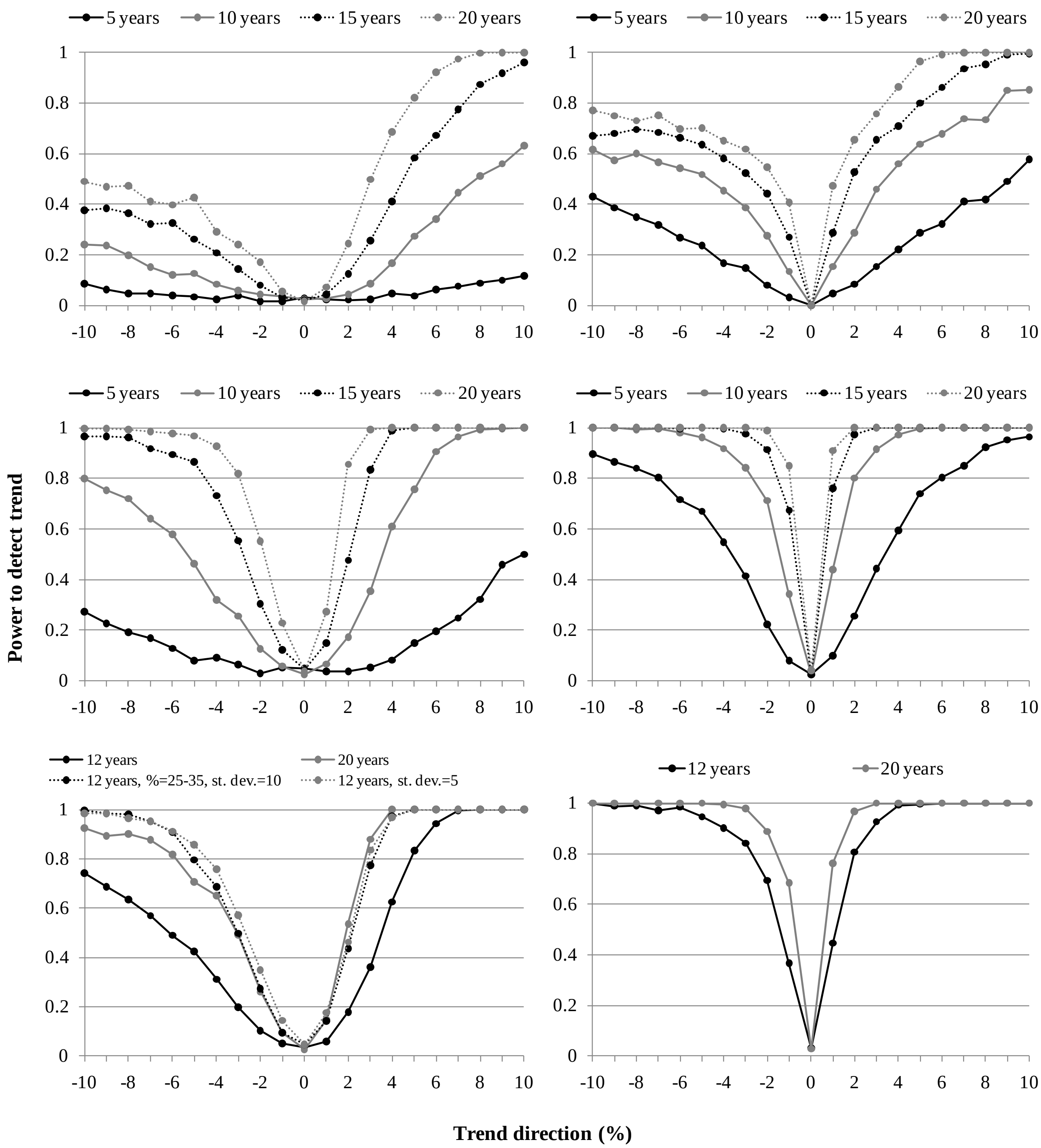

Figure 13. Power analysis to detect increasing and decreasing trends of varying strength (%) in percent fines in riffles over various survey durations for sites in mainstem reaches of upper Benewah Creek (left panels) and for sites in tributary reaches of upper Lake Creek (right panels). Upper and middle panels respectively display results for 5 and 10 sites monitored annually, and lower panels display results for 10 sites monitored every 4 years. Initial values and standard deviation (st. dev.) estimates were adjusted for two simulations (lower left panel) to represent expected values for unrestored mainstem reaches and to simulate annual variability as the percent fines in riffles approached desired levels.

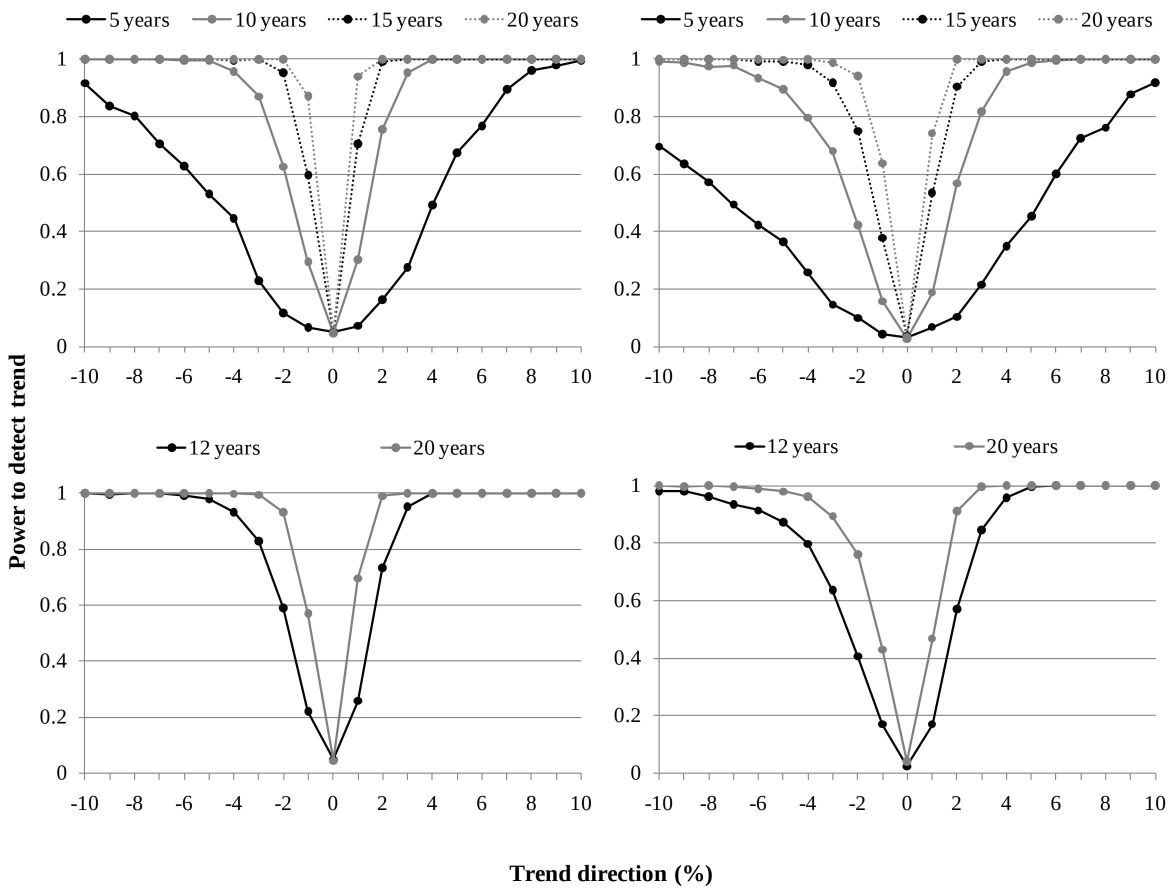

Figure 14. Power analysis to detect increasing and decreasing trends of varying strength (%) in mean residual pool depth (m) over various survey durations for sites in mainstem reaches of upper Benewah Creek (left panels) and for sites in tributary reaches of upper Lake Creek (right panels). Upper and lower panels respectively display results for 5 sites monitored annually and every 4 years, respectively.

Compared with LWD counts, subtle directional trends in LWD volume (m^3 / 100 m) could not be detected with high probability, even after 20 years, in upper mainstem reaches of Benewah creek when only 5 sites were monitored (Figure 15). Increasing the number of monitored sites to 10 improved detection power, as positive subtle trends of 4% could be detected with 80% probability in 10 years and negative subtle trends could be detected with similar power in 15 years. Similar results were obtained when the set of 10 sites was cyclically monitored every 4 years (Figure 15). Positive trends of 4% could be detected with high probability in 12 years; negative trends of similar strength could be detectable only after 12 years had elapsed.

For tributary reaches in upper Lake Creek, subtle positive trends of 4% for LWD counts (# / 100 m) were detectable with high probability only when 5 representative sites were monitored over a period of 15 to 20 years; subtle negative trends could not be reliably detected over the survey durations that were examined (Figure 16). Increasing the set of sites to 10 permitted subtle positive and negative trends to be detected with high probability in approximately 10 and 15 years, respectively. When the set of 10 representative sites was cyclically surveyed every four years, similar subtle positive and negative trends could be detected in 12 and 20 years, respectively (Figure 16).

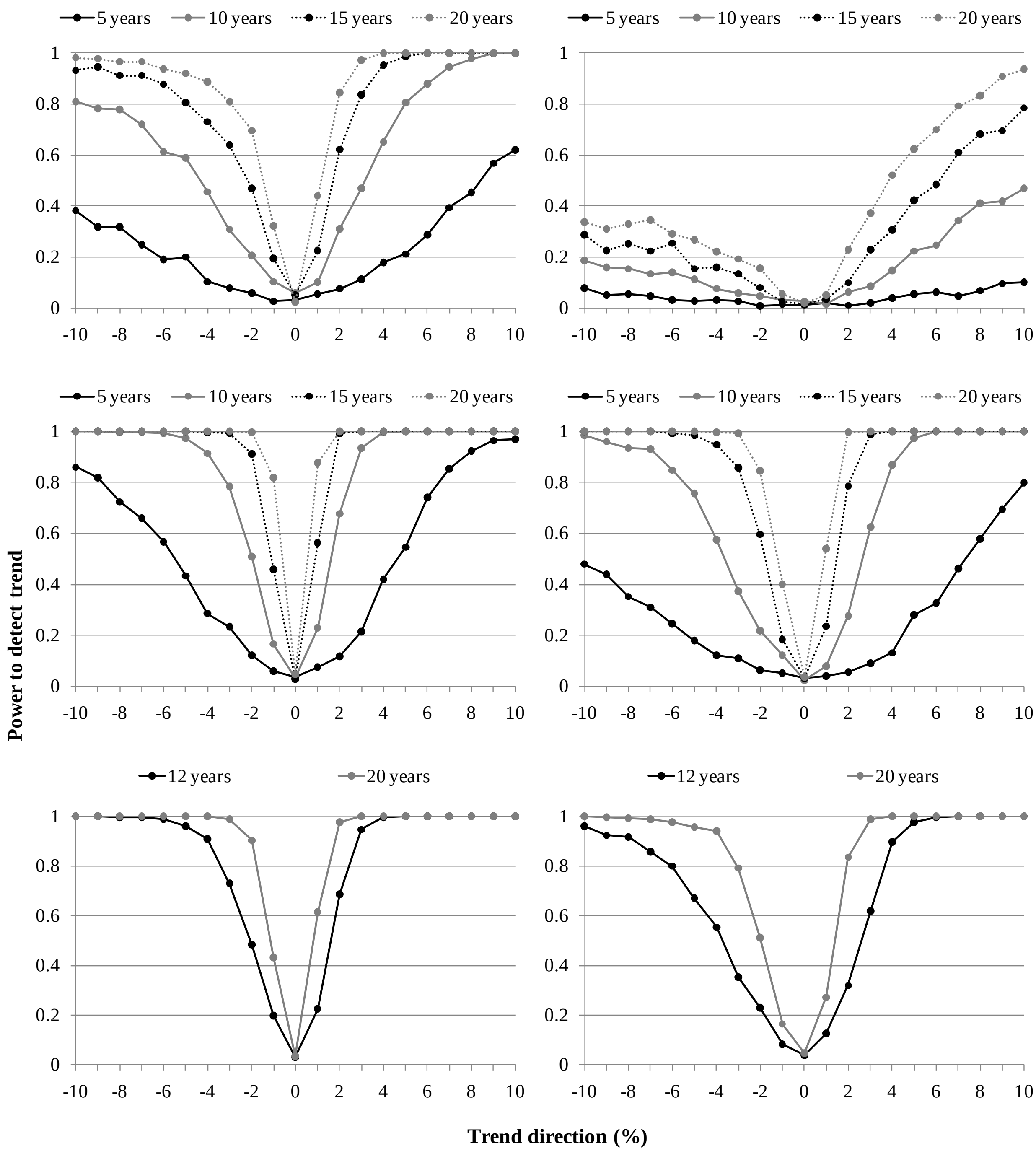

Figure 15. Power analysis to detect increasing and decreasing trends of varying strength (%) in counts of LWD (# / 100 m; left panels) and LWD volume (m^3 / 100 m; right panels) over various survey durations for sites in mainstem reaches of upper Benewah Creek. Upper and middle panels respectively display results for 5 and 10 sites monitored annually, and lower panels display results for 10 sites monitored every 4 years.

The inability to detect subtle negative trends over relatively short durations was likely due to a combination of low LWD counts and relatively large estimates of annual variability (i.e., high coefficients of variation) for some of the sites that were used in the model simulations. When these sites were omitted from the power analysis, subtle trends of less than 3% could be reliably detected in 12 years using a monitoring design of 10 sites cyclically monitored every 4 years (Figure 16). Trends in LWD volume in Lake Creek could not be detected under any monitoring scheme because of the magnitude of variability in empirical volume estimates among tributary sites.

3.3.3.2. Precision analysis for assessing the regional status of habitat attributes

Canopy cover measurements that were collected in any given year varied greatly across sites within designated reach types (e.g., tributaries, unrestored mainstem habitats) resulting in rather large required sample sizes to obtain desired levels of precision (Table 17). For all reach types examined, approximately 15 to 25 and 60 to 95 canopy cover measurements would need to be collected to obtain ±10 and ±5% levels of precision around a mean estimate, respectively. However, if measurements per site were increased from 6 to 10, then only 6 to 10 sites would need to be sampled to obtain the desired precision level of ±5%. For quasi-reference reach types, which had lower estimates of variability, sample sizes were much lower, with only 10 to 35 samples required to obtain a ±5% level of precision.

The estimated level of variability in the mean percent of fines computed across sampled riffles was much greater for sites sampled in lower tributary reaches in Lake Creek than in mainstem reaches of the Benewah system (Table 18). Consequently, approximately 30 and 110 samples would need to be collected in Lake Creek tributaries to obtain ±10 and ±5% precision levels around a mean estimate, respectively. In comparison, only 13 to 35 measurements would be required in mainstem reaches of the Benewah mainstem to obtain a precision level of ±5%. Incidentally, though the low standard deviation estimate for restored reaches of the Benewah mainstem may have been biased because of the low number of samples collected in any given year, a similar estimate of variability (standard deviation, 10.7) was obtained when riffle measurements from all sites sampled from 2006 to 2008 were included in the calculation. If three riffles are randomly sampled within a site, then approximately 10 sites would need to be sampled to obtain ±5 and ±10% precision in reaches of Benewah Creek mainstem and Lake Creek tributaries, respectively. Required sample sizes to obtain ±5% precision were relatively small (i.e., 7 to 10) for quasi-reference reaches for both channel types (Table 18); however, results should be interpreted with caution because of the lack of data used to generate the low standard deviation estimates.

In Lake Creek tributary reaches, approximately 15 pools would need to be sampled to obtain a precision of ±0.1 m around an estimate of mean residual pool depth (Table 19). As expected, pool depths were more variable in mainstem than in tributary reaches, and thus required a sample size of 22 to 35 pools to yield a precision of ±0.1 m. Given that many of the monitored 152 m long habitat sites in our watersheds contain approximately 10 pools, a rather precise estimate of mean residual pool depth could be obtained from a survey conducted across 3 to 5 sites.

Large woody debris (LWD) counts were more variable among sites in Benewah mainstem reaches than among sites in lower reaches of Lake Creek tributaries (Table 20). Consequently, 8 to 13 sites would need to be sampled to obtain a precision of ± 5 pieces, and 23 to 36 sites would need to be sampled to obtain a precision of ± 3 pieces in mainstem Benewah reaches.

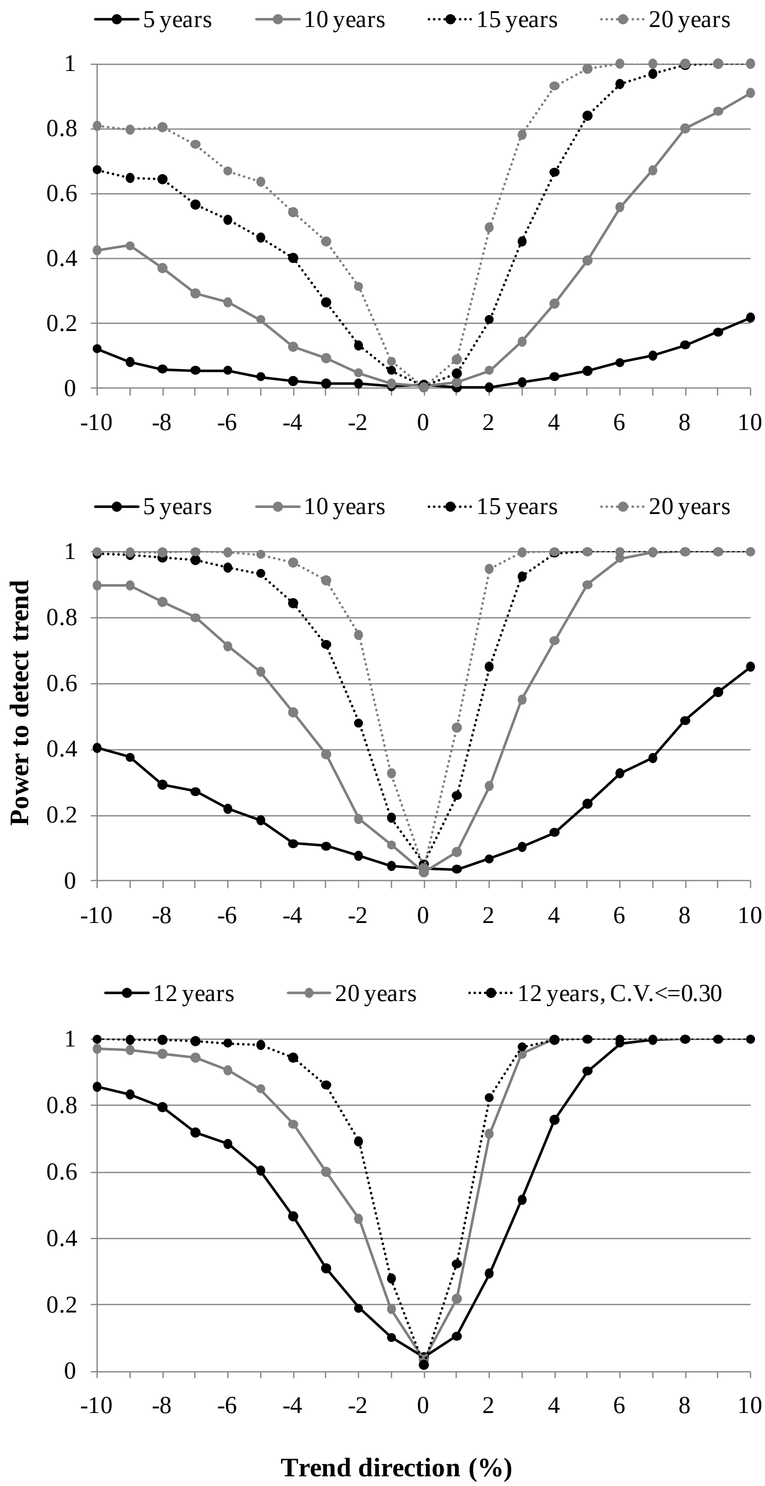

Figure 16. Power analysis to detect increasing and decreasing trends of varying strength (%) in counts of LWD (# / 100 m) over various survey durations for sites in tributary reaches of upper Lake Creek. Upper and middle panels respectively display results for 5 and 10 sites monitored annually, and lower panel displays results for 10 sites monitored every 4 years. For one simulation (lower panel), only those sites with coefficient of variation (C.V.) estimates less than 0.30 were used to create the set of 10 representative sites.

In comparison, only 6 and 16 sites would need to be sampled in tributary reaches of Lake Creek to obtain similar low and high precision levels. Precision analyses for the LWD volume metric were more variable among reach types than those for the LWD count metric (Table 20). Whereas 44 and 56 sites would need to be respectively sampled in restored Benewah mainstem and Lake Creek tributary reaches to obtain a highly precise volumetric estimate (respectively higher sample sizes than those required for LWD counts), only two sites in unrestored mainstem Benewah reaches would require sampling. The low sample size for this channel type was due to the lack of LWD volume, and consequent low levels of among-site variability, in unrestored mainstem reaches. Precision results for LWD metrics should be interpreted with caution given that sample sizes for generating estimates of variability among sites were small (i.e., only one LWD metric measurement could be collected per site).

Table 17. Summary of annual estimates of mean percent canopy cover and variability (standard deviation, S.D.), and required sample sizes to obtain precision levels of 5 and 10% for lower tributary reaches in upper Lake Creek and mainstem reaches in upper Benewah Creek, 2004-2008. Summaries for quasi-reference reaches in which data from 2006 to 2008 were aggregated are also provided for comparison.

		Overall mean estimates from annual measurements, 2004-2008				**Samples required for specified precision**	
System	**Reach location**	**Sites sampled**	**Samples collected**	**Canopy cover (%)**	**S.D. estimate**	**5% precision**	**10% precision**
		Targeted reaches for ongoing or projected habitat enhancement projects					
Lake	Tributary lower reach	6	34	54	23.8	91	23
Benewah	Upper mainstem restored	2	11	27	19.6	62	15
Benewah	Upper mainstem unrestored	2	12	56	24.4	95	24
		Quasi-reference reaches					
Lake	Tributary upper reach	2	12	91	7.9	10	2
Benewah	Upper mainstem reference	1	6	32	14.8	35	9

Table 18. Summary of annual estimates of mean percent fines in riffles and variability (standard deviation, S.D.), and required sample sizes to obtain precision levels of 5 and 10% for lower tributary reaches in upper Lake Creek and mainstem reaches in upper Benewah Creek, 2004-2008. Summaries for quasi-reference reaches in which data from 2006 to 2008 were aggregated are also provided for comparison.

		Overall mean estimates from annual measurements, 2004-2008				**Samples required for specified precision**	
System	**Reach location**	**Sites sampled**	**Samples collected**	**Percent fines**	**S.D. estimate**	**5% precision**	**10% precision**
		Targeted reaches for ongoing or projected habitat enhancement projects					
Lake	Tributary lower reach	5	9	67	26.4	111	28
Benewah	Upper mainstem restored	2	4	11	9.0	13	3
Benewah	Upper mainstem unrestored	2	4	38	14.9	36	9
		Quasi-reference reaches					
Lake	Tributary upper reach	2	3	9	8.1	10	3
Benewah	Upper mainstem reference	1	2	10	6.5	7	2

Table 19. Summary of annual estimates of mean residual pool depth and variability (standard deviation, S.D.), and required sample sizes to obtain precision levels of 0.1 and 0.2 m for lower tributary reaches in upper Lake Creek and mainstem reaches in upper Benewah Creek, 2004-2008. Summaries for quasi-reference reaches in which data from 2006 to 2008 were aggregated are also provided for comparison.

		Overall mean estimates from annual measurements, 2004-2008				Samples required for specified precision	
System	**Reach location**	**Sites sampled**	**Samples collected**	**Residual pool depth (m)**	**S.D. estimate**	**0.1 m**	**0.2 m**
		Targeted reaches for ongoing or projected habitat enhancement projects					
Lake	Tributary lower reach	6	74	0.45	0.194	15	4
Benewah	Upper mainstem restored	2	9	0.83	0.295	35	9
Benewah	Upper mainstem unrestored	2	21	0.51	0.236	22	6
		Quasi-reference reaches					
Lake	Tributary upper reach	1	11	0.21	0.077	2	1
Benewah	Upper mainstem reference	1	8	0.47	0.093	3	1

Table 20. Summary of annual estimates of mean counts (# / 100 m) and volume (m^3 / 100 m) of large woody debris (LWD) and variability (standard deviation, S.D.), and required sample sizes to obtain specified low and high precision levels for lower tributary reaches in upper Lake Creek and mainstem reaches in upper Benewah Creek, 2004-2008. For LWD counts, high and low precision levels were selected as 3 and 5 pieces / 100 m, respectively. For LWD volume, high and low precision levels were selected as 1 and 2 m^3 / 100 m.

		Overall mean estimates from annual measurements, 2004-2008			Samples required for specified precision	
System	**Reach location**	**Sites sampled**	**LWD metric (value / 100 m)**	**S.D. estimate**	**High precision**	**Low precision**
		LWD counts - high and low precision estimates of 3 and 5 (pieces / 100 m), respectively				
Lake	Tributary lower reach	6	11.6	5.91	16	6
Benewah	Upper mainstem restored	2	26.2	8.97	36	13
Benewah	Upper mainstem unrestored	3	15.0	7.21	23	8
		LWD volume - high and low precision estimates of 1 and 2 (m^3 / 100 m), respectively				
Lake	Tributary lower reach	6	2.28	3.75	56	14
Benewah	Upper mainstem restored	2	14.73	3.32	44	11
Benewah	Upper mainstem unrestored	3	1.99	0.70	2.0	0.5

3.3.4 Effectiveness monitoring – Response to habitat restoration activities

3.3.4.1 Cutthroat trout response to habitat restoration in the Benewah watershed

The estimated abundance of cutthroat trout of all ages at the lower Whitetail treatment site increased over five times from a mean value of 7.4 fish before restoration, calculated over the period from 2003 to 2007, to 44 fish one year after restoration (Table 21). Furthermore, given that a reliable depletion estimate could not be generated from the total number of captured fish in 2008, the estimate of 44 should be considered conservative. In comparison, the 2008 abundance estimate of 16.0 at the control site in lower Windfall Creek was only approximately 1.5 times greater than the mean value of 10.8 fish (calculated over the period from 2003-2007), a much smaller change than that observed at the treated site in Whitetail Creek. On the other hand, changes in age 1+ fish abundances were not appreciably different between the two sites. The 2008 estimate of 7.0 at the Whitetail site increased 1.7 times over the mean estimate of 4.0. Similarly, the estimate of 16 fish at the Windfall site was 1.6 times greater than its mean estimate of 10.0.

Generally, cutthroat trout abundances have not increased in mainstem reaches of the upper Benewah watershed that have undergone restoration over the period from 2005-2007 (Table 21). Although site 15, which is located in the reach that received treatment in 2007, is the only site that has data collected before and after implementation of habitat enhancement activities, the lack of fish captured at sites in the 2005 and 2006 restored reaches (i.e., ≤ 1 fish) over the last two years further supports the absence of a detectable positive response. On the other hand, abundances of cutthroat trout at site 16 (located in the reach that received treatment in 2004) are much higher over the past three years than over the period from 2002 to 2005 (Table 21). However, when comparing results at site 16 with those obtained at site 17, a control mainstem site further upstream, a significant difference in abundance trends between these two sites could not be detected for fish of all ages (ANOVA; $p = 0.10$ (site), $p = 0.14$ (interaction between site and time period)) and for age 1+ fish (ANOVA; $p = 0.39$ (site), $p = 0.55$ (interaction between site and time period)). Furthermore, similar abundances of cutthroat trout have also been recorded over the past two years at site 16L, a control site downstream of site 16 that is located in a degraded, unrestored reach of the Benewah mainstem. Notably, abundances of cutthroat trout at site 14, a control site in an unrestored mainstem reach downstream of the restored habitats, have not increased over time and have consistently been extremely low.

Though positive responses in treated mainstem habitats relative to control reaches were not detected, aggregate abundances of age 1+ cutthroat trout have increased more in tributaries in the upper Benewah watershed than in Lake Creek tributaries and habitats in the Evans Creek watershed in recent years (Figure 17). From 2005 to 2008, the period over which restoration has proceeded in the Benewah watershed, the logarithmic ratio of Benewah to Lake creek abundance has been monotonically increasing, indicating that cutthroat trout abundance is increasing at a more rapid rate in tributaries in upper Benewah Creek than in upper Lake Creek. Although a similar extended increasing trend over the last 4 years was not detected in the Benewah to Evans creek ratio, a substantial increase in the ratio was observed between 2007 and 2008. In addition, these recent increasing trends in both ratios have occurred over a period during which the aggregate abundance across upper Benewah tributary sites has been markedly increasing (Figure 17).

Table 21. Temporal trends in cutthroat trout abundance estimates at treatment and control sites for habitat restoration projects implemented in the upper Benewah watershed. Estimates are provided for fish of all ages and those classified as age 1+ (> 70 mm). For the mainstem restoration, sites 15L, 2006, 15, and 2008 underwent channel reconstruction in subsequent years from 2005-2008, respectively; reconstruction occurred at site 16 in 2004 (sampling occurred at treatment sites before annual reconstruction activities).

				Cutthroat trout abundance estimate						
Stream	**Site**	**Treatment category**	**Age class**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**
Large woody debris additions in lower Whitetail Creek, 2007										
Whitetail	1	Treatment	All ages	.	11.6	4.0	.	4.0	10.0	44.0 [a]
Windfall	1	Control	All ages	21.3	7.2	.	4.5	18.8	12.5	16.0
Whitetail	1	Treatment	Age 1+	.	5.0	2.0	.	2.0	7.0	7.0 [a]
Windfall	1	Control	Age 1+	21.3	7.2	.	4.5	18.8	11.6	16.0
Benewah mainstem restoration, 2004-2007										
Benewah	14	Control	All ages	0.0	2.0	4.5	2.0	4.0	0.0	3.1
Benewah	15L	Treatment	All ages	.	.	.	.	.	0.0	0.0
Benewah	2006	Treatment	All ages	.	.	.	.	.	.	1.0
Benewah	15	Treatment	All ages	0.0	4.0	2.0	0.0	1.0	0.0	3.0
Benewah	2008	Treatment	All ages	.	.	.	.	.	.	2.0
Benewah	16L	Control	All ages	.	.	.	.	.	7.1	27.9
Benewah	16	Treatment	All ages	1.0	3.0	1.0	1.0	13.3	15.0	8.0
Benewah	17	Control	All ages	1.0	3.0	2.0	1.0	.	17.5	14.8
Benewah	18	Control	All ages	.	.	.	.	.	.	13.0
Benewah	14	Control	Age 1+	0.0	2.0	1.0	2.0	4.0	0.0	2.2
Benewah	15L	Treatment	Age 1+	.	.	.	.	.	0.0	0.0
Benewah	2006	Treatment	Age 1+	.	.	.	.	.	.	1.0
Benewah	15	Treatment	Age 1+	0.0	3.0	0.0	0.0	1.0	0.0	2.0
Benewah	2008	Treatment	Age 1+	.	.	.	.	.	.	0.0
Benewah	16L	Control	Age 1+	.	.	.	.	.	7.1	6.0
Benewah	16	Treatment	Age 1+	1.0	2.0	1.0	1.0	12.1	8.1	8.0
Benewah	17	Control	Age 1+	0.0	3.0	2.0	1.0	.	9.2	12.3
Benewah	18	Control	Age 1+	.	.	.	.	.	.	9.0

[a] Total number of captured fish are displayed because a reliable depletion estimate could not be generated

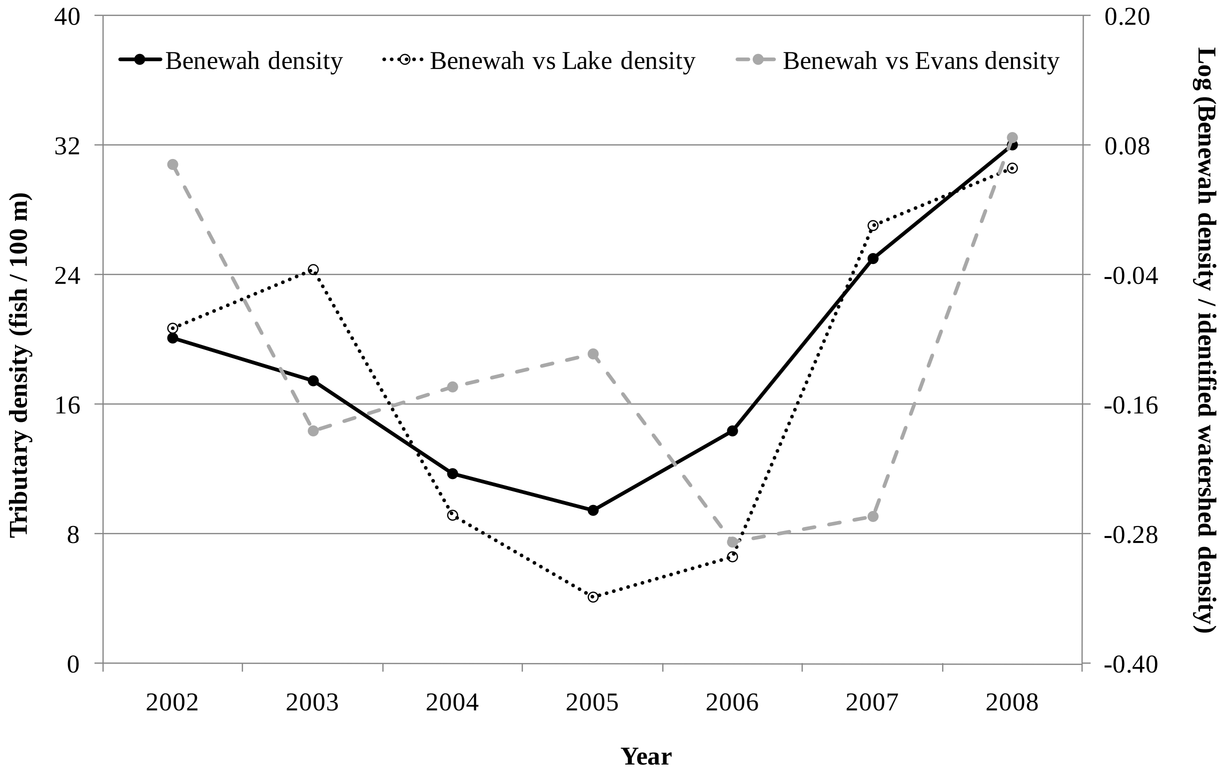

Figure 17. Aggregate densities (fish / 100 m) of age 1+ cutthroat trout summed across survey sites in tributaries of upper Benewah Creek, and compared to similar aggregate densities summed across tributary survey sites in upper Lake Creek and across all surveys sites in Evans Creek, 2002-2008. Comparisons were drawn between watersheds by computing the logarithmic ratio of annual aggregate densities in both watersheds.

3.3.4.2 Thermal responses to habitat restoration in the Benewah watershed

Temperature measurements collected from pool habitats and their associated downstream riffles revealed thermal refugia in restored reaches of the upper mainstem of Benewah Creek that were not captured by the temperature logger data. Before channel reconstruction occurred along the reach that underwent restoration in 2005 and 2006 (i.e., 9-mile bridge upstream to the Whitetail Creek confluence), stream temperatures measured along pool bottoms were generally less than 0.5ºC cooler than those measured at downstream riffles (Figure 18). In addition, most of the pre-restored pool habitats were less than one meter deep. However, after reconstruction in which pool habitats were deepened, pool temperatures measured in August of 2008 were frequently between 2 and 6ºC cooler than those in downstream riffles when residual pool depths were at least a meter (Figure 18). The riffle-pool temperature discrepancy was not as large for the survey conducted earlier in July, though the mean daily temperature (recorded by the logger upstream of 9-mile bridge) was 2.2ºC cooler during the July survey than the August survey.

Similarly, pre-restoration residual pool depths were typically less than one meter in the mainstem reach that underwent channel reconstruction in 2007, with most of the measured temperature differentials less than 0.5ºC (Figure 18). After reconstruction in which pools were deepened, temperature differentials ranged between 2.5 and 6.5°C for the survey conducted in mid-August of 2008. Again, riffle-pool temperature differences were not as great during the survey conducted earlier in July when mean water temperatures were cooler.

3.3.4.3 Evaluation of habitat response to Benewah restoration, 2008

Differences in habitat attributes between restored and un-restored sites in both mainstem and tributary reaches in the upper Benewah Creek watershed were detected in 2008 (Table 22). Un-restored mainstem sites had a higher percentage of percent fines in riffle habitats than either restored or reference sites. Percent fines for un-restored mainstem sites ranged from 15 to 51% as compared to restored sites which ranged from 11 to 15%. The mainstem reference site had percent fines of 10%. The difference between un-restored and restored sites for tributary habitats was more pronounced with percent fine estimates of 92% and 21%, respectively (Table 22). The performance objective for substrate composition is less than 15% fines in riffle/run habitats. All mainstem restored sites and the reference site met this objective in 2008; neither of the two tributary sites met the objective.

Canopy cover data collected in 2008 show that un-restored sites have equivalent or higher percent canopy cover values than restored sites (Table 22). Canopy cover densities ranged from 46 to 80% for un-restored mainstem sites. Restored sites had canopy cover values of 11% and 19%. The reference site, Benewah 18, had a canopy density of 32% which was higher than the restored sites but lower than all un-restored sites. Windfall 1 and Whitetail 1 had similar canopy cover densities of 71 and 73% (Table 22). The target for riparian canopy density for 2nd and 3rd order tributaries is 75%. Both tributary sites were within 5% of this value. For mainstem sites, percent canopy cover exceeded 75% for only the 2010 un-restored site.

Though the number of pieces of large woody debris (LWD) was similar between restored and un-restored mainstem reaches in 2008, LWD volume was substantially greater in restored than unrestored reaches (Table 22). The large woody debris count for un-restored sites ranged from 6 to 30 pieces of wood with LWD loadings ranging from 0.55 to 2.76 m^3/100 m. In comparison, restored sites had similar counts of 10 and 22, but LWD loadings of 6.84 and 9 m^3/100 m, values at least two times higher than those recorded in un-restored sites. Generally, pieces of wood at un-restored sites were considerably smaller than those found at the two restoration sites. For example, the 2008 site had the highest wood count at 30 but only had a loading value of 2.76 m^3/100 m. The reference site had the highest wood loading of 10.51 m^3/ 100 m. For tributary sites, LWD counts and loadings in the restored site were over 10 and 50 times greater than those values recorded in the un-restored site, respectively (Table 22). Target wood loadings of 9 m^3/ 100m were met in the 2007 mainstem restoration site, the restored Whitetail 1 site, and the reference mainstem site. The 2006 restoration site and all un-restored sites did not meet our target objective in 2008.

Residual pool metrics were generally larger for restored than un-restored mainstem sites, though only one un-restored site was available for comparison in 2008 (Table 22). The mean and maximum pool depths for the un-restored site were 0.61 and 0.82 m, respectively. In comparison, mean and maximum pool depths in restored mainstem sites ranged from 0.86 to 1.31 and 1.49 to 1.34, respectively. In addition, the pool volume at restored sites, which ranged between 177.8 and 195.9 m^3, was substantially greater than the value of 27.8 m^3 that was calculated for the un-restored site. Though the number of pools was less in restored than in un-restored site, this was an artifact of the channel modification measures that were implemented in restored reaches. Residual pool metrics measured at the reference mainstem site, however, were all lower than those measured at both restored and un-restored mainstem sites (Table 22). Mean and maximum pool depths and pool volume were 0.47 m, 0.62 m, and 3.2 m^3, respectively. Our provisional objective for mean residual pool depth in mainstem habitats is depths of at least 1 m, which was met by only one restored site in 2008. Notably, the reference site did not meet the provisional criterion for pool depth. In contrast to that observed in mainstem habitats, mean and maximum residual pool depth values were higher in the un-restored tributary site (0.63 and 0.85 m, respectively) than in the restored tributary site (0.46 and 0.59 m, respectively).

3.3.4.4 Evaluation of habitat responses to Benewah mainstem restoration, 2005 - 2008

Physical habitat data collected in Benewah mainstem habitats in 2008 generally reflect that which has been documented over the course of mainstem channel reconstruction in the upper watershed since 2005. Over the last four years, 2523 m of mainstem channel have been restored, marking the end of Phase I for this project. Overall changes in physical response variables due to these modifications have been summarized using a combination of before/after and treatment/control comparisons from monitored sites (Table 23). Restoration activities have increased channel length by 506 m, resulting in an overall 25% increase in sinuosity from 1.28 to 1.68. Slope deceased by 58% from 0.0048 pre-construction to 0.002. Mean residual pool depth increased significantly (p <0.001) from 0.57 m pre-construction to 1.18 m and mean low-flow thalweg depth also increased significantly from 0.38 m pre-construction to 0.52 m (p<0.0167). Pool volume increased by over 500% from 28 m^3/100 m to a mean value of 187 m^3/100 m. Also, the percent of slow, deep water pool habitats (mean=49.2%), which provide most of the summer and over-winter rearing opportunities for cutthroat trout, increased by 24%, putting these restored mainstem reaches well within the range of values of 48 to 53% described for PACFISH/INFISH long-term monitoring sites with similar channel geometry (Henderson et al. 2005). Generally, instream large wood volume increased 143% from 3.15 m^3/100 m pre-construction to 7.67 m^3/100 m (the increase in wood frequency and volume was even greater for some constructed channel segments).

Given the scope of restoration in the Benewah watershed, project effectiveness was also evaluated in the context of ecological functions that consider a broad range of values that extend beyond the boundaries of the low flow channel. Several of these additional measures of project effectiveness are described below and have provided evidence for improvements in ecological function and value that support multiple species and address processes operating at larger spatial scales.

Streambank erosion, a significant issue that we have been addressing through our restoration and enhancement activities, has likely been substantially reduced in the upper Benewah mainstem. Changes in stream bank erosion rates were estimated for treated and untreated control sites along the mainstem of upper Benewah Creek using the BANCS model (Rosgen 2006). The control site was characterized by unstable stream banks with accelerated erosion rates and increased sediment yield to the channel; 30% of stream banks showed active erosion with estimated erosion rates of 0.7 metric tons/year/m (Table 23). Restoration efforts have significantly improved stream bank conditions to reduce erosion potential as indicated by a positive response in the rooting character (e.g., root density and depth) of stream bank vegetation, and a reduction of 50% in the bank height ratio. In addition, active bank erosion was evident at only 10% of stream banks two years post-restoration, a reduction of 65% compared with an untreated control reach (Table 23). We estimate that erosion rates have been reduced by 73% with a reduction in total sediment yield of greater than 1,294.6 metric tons/yr for the 2,523 m of treated channel.

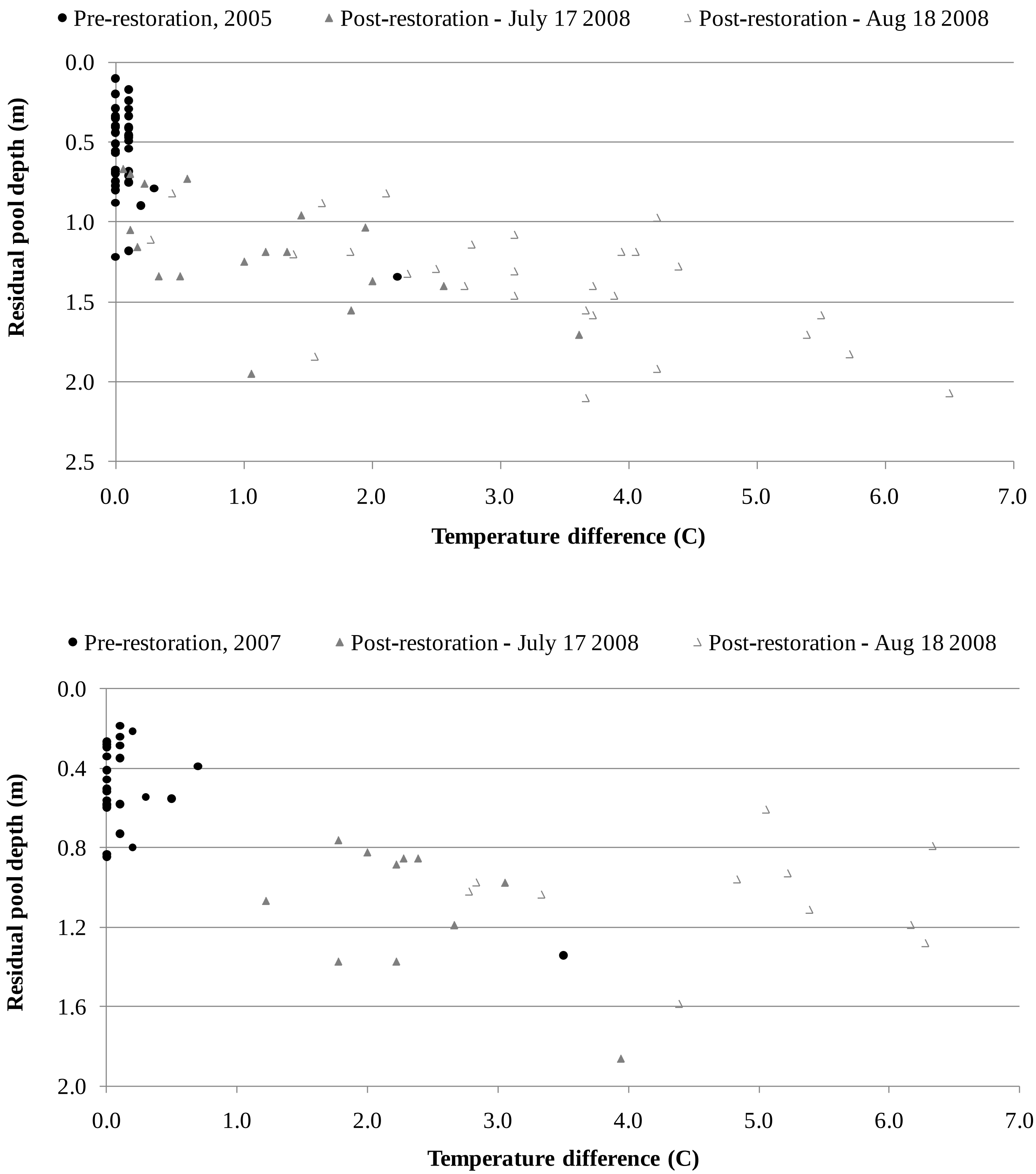

Figure 18. The relationship between temperature difference and residual pool depth for surveys conducted before and after restoration actions along upper mainstem Benewah reaches that were restored in 2005 and 2006 (upper panel) and in 2007 (lower panel). Temperature difference was calculated as the temperature measured along the pool bottom minus the temperature measured in the associated downstream riffle. Surveys in 2008 were conducted on July 17 and August 18 (mean daily temperatures at 9-mile bridge were 18.6 and 20.8, respectively).

Table 22. Habitat indicator variables measured at survey sites in the upper Benewah Creek watershed, 2008. Metric values are displayed for unrestored and restored sites in both mainstem and tributary reaches. Site lengths were 152 m for all sites except Whitetail 1 which was 396 m in length.

	Mainstem reaches							Tributary reaches	
	Unrestored sites				Restored sites		Reference site	Unrestored site	Restored site
Physical habitat category Metric	2008	16L	2010	17	2006	2007	18	Windfall 1	Whitetail 1
Morphology									
Bankfull width (m)	.	8.0	.	6.1	.	.	.	5.5	3.5
Bankfull wetted perimeter (m)	.	8.9	.	11.9	.	.	.	6.2	4.3
Bankfull mean depth (m)	.	0.57	.	0.65	.	.	.	0.45	0.34
Cross sectional area (m^2)	.	4.46	.	3.95	.	.	.	2.48	1.18
Riffle w/d ratio	.	26.3	.	11.4	.	.	.	9.2	17.1
Substrate composition									
Less than 2 mm (%)	51	15	33	46	15	11	10	92	21
Canopy cover									
Density (%)	56	47	80	46	19	11	32	71	73
Large woody debris									
Total count	30	6	13	18	10	22	23	10	132
Volume (m^3)	4.21	1.22	0.84	2.76	10.43	13.71	16.03	0.44	44.06
Loading (m^3/100 m)	2.76	0.80	0.55	1.81	6.84	9.00	10.51	0.29	11.10
Residual pools									
Mean depth (m)	.	.	0.61	.	0.86	1.31	0.47	0.63	0.46
Minimum depth (m)	.	.	0.32	.	0.49	1.28	0.34	0.36	0.30
Maximum depth (m)	.	.	0.82	.	1.49	1.34	0.62	0.85	0.59
Number of pools	.	.	8	.	4	2	8	9	10
Residual pool volume (m^3)	.	.	27.8	.	195.9	177.8	3.2	14.5	.

Table 23. Summary of change for selected response variables following four years of restoration in Benewah Creek.

Response variable	**Before**	**After**	**% Change**	**Objective**
Sinuosity [a]	1.28	1.68	+25	>1.50
Slope [a]	0.0048	0.002	-58	<0.003
Entrenchment ratio [a]	2	12.3	+515	>12
Belt width (m) [a]	41.7	71.0	+70	>60
Bank height ratio [b]	2.05	1.02	-50	1.0
Percent eroding banks [b]	30.0	10.5	-65	<10%
Erosion rate (metric tons/yr/m) [b]	0.70	0.19	-73	<0.10
Residual pool depth (m) [a]	0.57	1.18	+107	>1.0
Residual pool volume (m^3/100 m) [b]	28	187	+567	NA
Percent pools [b]	39.6	49.2	+24	48-53
Mean low flow thalwag depth (m) [c]	0.38	0.52	+37	NA
Large wood (# pieces/100 m) [c]	13.8	14.5	+5	>15
Large wood volume (m^3/100 m) [c]	3.15	7.67	+143	>9.0
Mean depth to groundwater (m) [b]	1.32	0.62	-47	<1.2

[a] values are derived from before/after comparisons of treatments at the scale of the project reach
[b] values are derived from comparisons of discrete, monitored treatment and control sites
[c] values are derived from before/after comparisons at discrete, monitored treatment sites

Given that one of the project's goals in the upper Benewah mainstem was to improve connectivity, and consequently groundwater exchange, between the main channel and the adjacent floodplain, it was imperative to better understand and track the response of groundwater dynamics to our restoration activities. As a result, a total of 35 piezometers were installed in early July of 2008, with 17 of these installations consisting of clustered, near-channel wells located in treated, untreated and reference reaches. The remaining 18 wells were randomly scattered across the valley bottom in the portion of the mainstem that is scheduled for treatment over the next three years. Monitoring of these wells over base flow conditions during the summer of 2008 indicated that groundwater levels in a restored reach were higher than those measured in an un-restored reach. Mean depth to groundwater, measured at near-channel wells (n=10), was 50% lower in restored reaches (mean = 0.62 m) compared with untreated reaches (mean = 1.32 m; Table 23).

By increasing shallow groundwater levels and consequent water availability in floodplain habitats for native wetland species, restoration activities are expected to increase survival rates for planted vegetation as well as for naturally recruited propagules. In treated Benewah mainstem reaches, the extent of wetland habitats has increased by an estimated 48% as former upland habitats, falling within the belt width of the constructed channel, have been directly converted to wetlands, and as uplands, further removed from the channel, have been incorporated within the larger floodprone area of the valley bottom. Wetland function has also been improved over a broad range of indices, as measured by the qualitative and semi-quantitative methods identified in the Idaho Interim Functional Assessment for Riverine Wetlands (Jankovsky-Jones 1999). The results of this assessment suggested a 49% increase in functional capacity (FCU) over time, attributable to the increase in hydrologic interaction between the restored channel and it's floodplain, with the greatest improvements evident in maintaining detrital biomass, dissipation of energy, sediment and nutrient retention/removal, and dynamic and long term surface water storage.

Vegetation responses to mainstem restoration in the Benewah watershed were also empirically monitored over the past several years. Survival rates, monitored at fixed radius plots (n = 31) over a period of three years, were estimated to be 83% for all woody plant species combined. Growth rates were evaluated at established greenline transects, that were located parallel to newly constructed stream segments, and at fixed plot transects, that were placed perpendicular to the restored channel and generally located in areas that were most disturbed during construction. Both percent ground cover and canopy density were measured along greenline and fixed plot transects to evaluate vegetation response (Winward 2000; Bonham 1989; Daubenmire 1959).

Growth rate responses were variable and dependent upon the relative location in which they were monitored. A significant positive linear trend in mean percent ground cover was detected along greenline transects that were monitored at 8, 20 and 32 month post-treatment intervals (repeated measures ANOVA, $F = 13.01$, $p < 0.001$). Mean change in ground cover percent was 15.7 over both the 8 to 20 and 20 to 32 month post-restoration periods (Figure 19). For those greenline transects that were monitored at 20, 32, and 44 month intervals, an increase in mean percent ground cover was also detected (repeated measures ANOVA, $F = 3.03$, $p = 0.09$). However, the change was only detectable between the second and third monitoring periods, where mean percent ground cover increased by 21% (Figure 19). Significant trends in canopy cover were not detected for either of the two sets of greenline transects. A significant positive linear trend in mean percent ground cover was also detected at floodplain vegetation plots that were monitored at 8, 20 and 32 month post-treatment intervals (repeated measures ANOVA, $F = 17.48$, $p < 0.001$). Mean change in cover percent was 16.6 and 10.2 over the 8 to 20 and 20 to 32 month post-restoration periods, respectively (Figure 20). Although a negative linear trend in mean percent canopy cover was detected at these floodplain vegetation plots over similar time periods (repeated measures ANOVA, $F = 3.33$, $p = 0.061$), mean percent coverage was less than 2.2 in all three periods. Trends in either ground or canopy cover were not detected at those floodplain vegetation plots that were monitored at 20, 32, and 44 month post-restoration intervals (Figure 20).

3.3.4.5 Evaluation of habitat response in Lake Creek, 2008

Differences in substrate composition between restored and un-restored habitat sites in the Lake Creek watershed were observed in 2008. Un-restored sites had a higher percentage of percent fines in riffle habitats than the reference site (Table 24). Percent fines for un-restored tributary sites ranged from 44% to 89%. In comparison, the reference site had 17% fines, a percentage 2-4 times lower than that recorded in the un-restored sites. The performance objective of less than 15% fines in riffle/run habitats was not met at the un-restored sites, but was approached at the reference site.

The reference site had equivalent or higher overall canopy cover values than most of the other sites (Table 24). The reference site had the highest canopy density of 95.8%, whereas total canopy cover densities ranged from 48-77% at all un-restored sites other than the Bozard 2 site. However, reed canary grass accounted for much of the shading at these sites and consequently impacted canopy cover readings. When grass cover was removed from the canopy calculations, canopy densities for the un-restored sites ranged from 6 to 60% (Table 24). The restored site had a total canopy cover density of 57.1%, a value that was equivalent or lower than that recorded at all un-restored sites (Table 24). The relatively low estimated canopy density at Lake 11 was likely due to the impact of prior construction activities that removed vegetation, and the channel width at this site. Bankfull width of the channel at Lake 11 was 7.6 m while the widths at un-restored sites ranged from 3.2 to 4.9 m. With cover due to grass included, two un-restored sites and the reference site met our target objective of 75%for riparian canopy density for 2nd and 3rd order tributaries.

Large woody debris volume in the upper Lake Creek watershed was found to be higher in the restored Lake 11 site than in un-restored sites in 2008 (Table 24). The large woody debris count for un-restored sites ranged from 7 to 28 pieces of wood with wood densities ranging from 0.14 to 2.79 m^3/100 m. The restored site had a wood density of 7.08 m^3/100 m, at least two times higher than the highest wood densities found for un-restored sites. Bozard site 1, an un-restored site, had the highest wood count at 28 but only had a wood density of 2.79 m^3 /100 m, indicating that the pieces of wood at this site were considerably smaller than those found at the restoration site. The reference site had a low wood density of only 0.14 m^3/ 100 m (Table 24). This site is in a tree stand of mostly non-coniferous trees which provide shade but do not provide for large wood recruitment. Target wood densities of 9 m^3/ 100m were not met at any tributary site in Lake Creek in 2008.

The maximum and mean pool depths were greater for both the un-restored and restored sites than for the reference site (Table 24). Mean pool depth for un-restored sites ranged from 0.41 to 0.58 m, and was 0.49 m at the restored site. In comparison, a mean residual depth of only 0.37 m was recorded at the reference site. Similarly, maximum depths for un-restored and restored sites ranged from 0.67 to 0.93, whereas the maximum pool depth at the reference site was 0.37 m. However, only one pool in the Bozard 3 reference site had greater than one foot of residual depth (Table 24). On the other hand, the number of pools that were greater than one foot of residual depth in both restored and un-restored sites ranged from 4 to15. Relative to the reference reach in upper Bozard, the other sites were located in tributary reaches that were impacted by beaver dams, which in turn caused backwater effects that created deeper pools. Pool volumes were relatively similar for all sites for which the metric was calculated, with values ranging between 2.9 to 5.1 m^3. Given that mean residual pool depth for all sites was substantially less than our performance standard of 1 m that was established for mainstem reaches, it was apparent that this criterion would not be appropriate for capturing changes in pool depth in our smaller upper tributary reaches.

3.3.4.6 Comparison of cross-sectional data collected at habitat sites, 2003 – 2008

Habitat site cross-sections were compared between 2003 and 2008 to examine changes in channel form over time (see Appendix A for graphical illustrations for all sites). The cross section graphs display small changes to the bed and banks at monumented cross section locations. They show changes over time with regard to vertical and lateral stability. Several factors may bias this data, including inconsistency in how the tape is stretched along the channel, the size of substrate lining the bottom of the channel, and whether head pins on some of the cross-sections were disturbed or dad to be re-established. The following figures highlight some of the sites where moderate changes to channel bed and banks occurred due to erosion or deposition.

From 2003 to 2008, one of the pools at site 3 in the West Fork of Lake Creek (i.e., cross-section 6; Figure 21) experienced bank erosion between 2003 and 2008 on the left bank looking downstream, resulting in an area of 2.2 m^2 of streambank that was lost and coinciding with a local adjustment in the bankfull dimensions. In comparison, the channel dimensions of cross-secton 4 at site 12 in upper Lake Creek displayed minimal changes over the 5-year period (Figure 21). The dominant vegetation at Lake 12 is reed canary grass, which is currently protecting the channel from erosion and keeping the banks stable at this cross-section.

As another example, comparative survey data revealed that an estimated 1.67 m^2 of sediment was deposited at cross-section 3 at Lake site 10 over the 5 year time period from 2003 to 2008 (Figure 22). Similar depositional processes were illustrated by the comparative data collected at cross-section 6 at site 11 in upper Lake Creek. In 2003, a pool existed at cross-section 6 that had two channels divided by a 0.91 m island. By 2008, the deepest point of the cross-section had deepened by only 0.18 m, while the secondary channel had filled in with 1.07 m^2 of sediment (Figure 22).

In 2003, cross-section 3 in site 9 in the Lake Creek mainstem was a riffle that, during low flow, had two channels divided by a 0.76 m wide island. This island was eroded away by 2008 (Figure 23). In addition, both banks experienced erosion. In some sites, both erosion and deposition on the same bank took place over the 5-year period, as exemplified by cross-section 2 at site 2 in Bozard Creek (Figure 23).

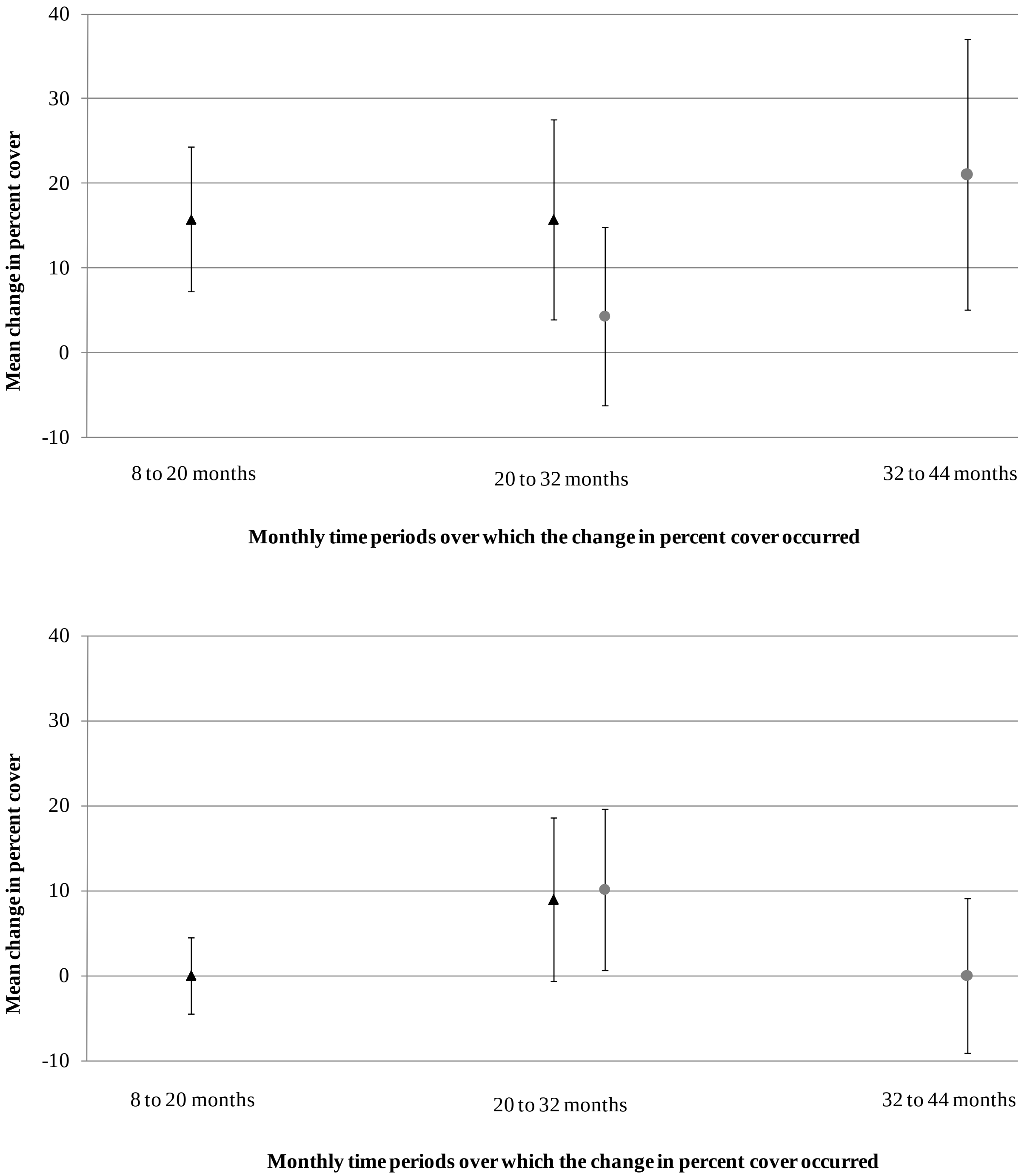

Figure 19. Mean change in percent ground cover (upper panel) and canopy cover (lower panel) and associated 95% confidence intervals calculated over three consecutive post-restoration time periods for greenline transects located in a restored reach of the upper Benewah watershed. Two transects (black triangles) were monitored at 8, 20, and 32 month intervals, and two other transects (gray circles) were monitored at 20, 32, and 44 month intervals.

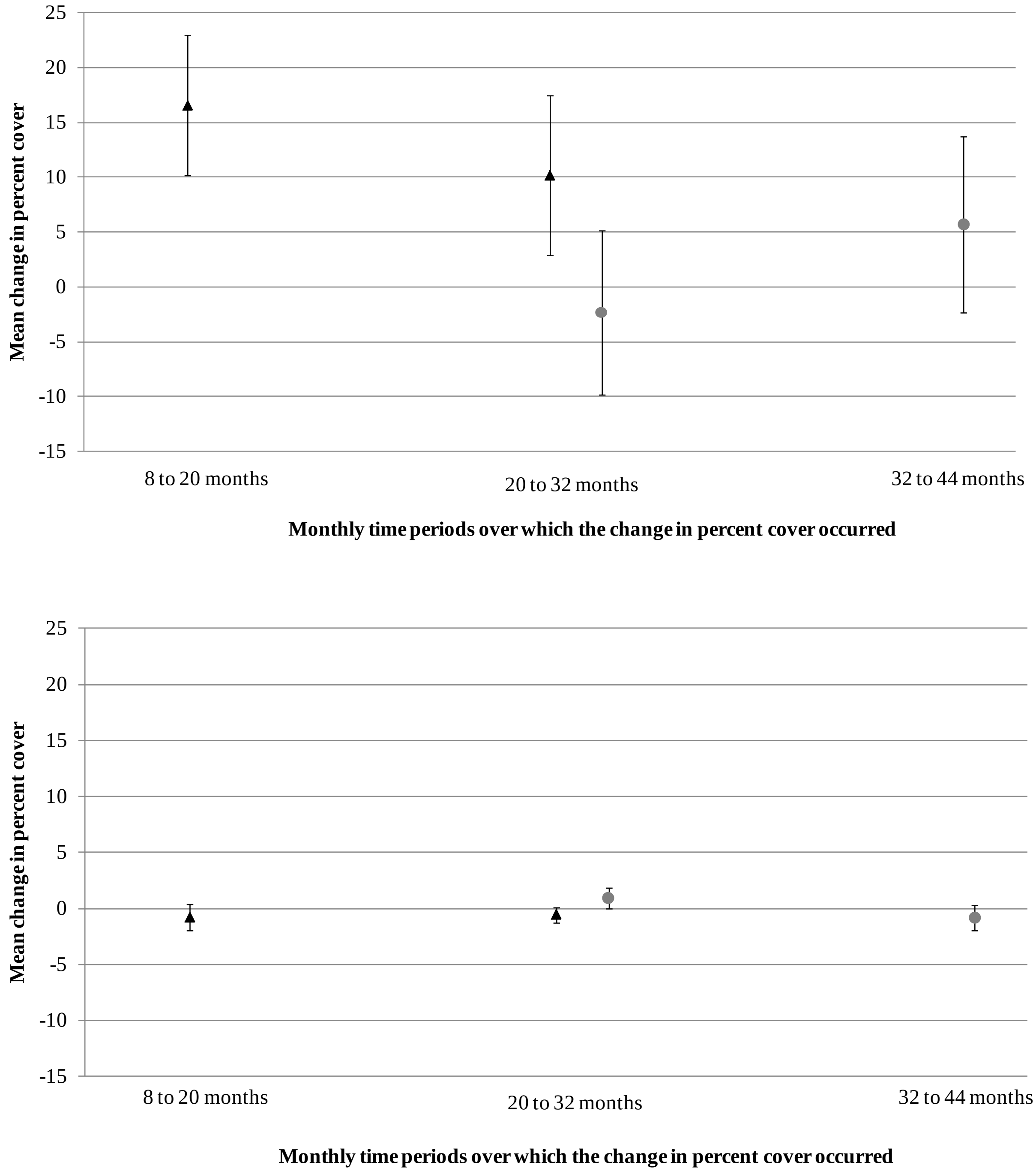

Figure 20. Mean change in percent ground cover (upper panel) and canopy cover (lower panel) and associated 95% confidence intervals calculated over three consecutive post-restoration time periods for vegetation plots located along eight floodplain transects in a restored reach of the upper Benewah watershed. Four transects (black triangles) were monitored at 8, 20, and 32 month intervals, and four other transects (gray circles) were monitored at 20, 32, and 44 month intervals.

Table 24. Habitat indicator variables measured at tributary habitat survey sites in the upper Lake Creek watershed in 2008. The Lake 11 restored site was treated with large woody debris placement in 1999.

	Unrestored sites					Reference site	Restored site
Physical habitat category Metric	Lake 12	WF Lake 2	WF Lake 3	Bozard 1	Bozard 2	Bozard 3	Lake 11
Morphology							
Bankfull width (m)	4.9	3.8	3.2	4.9	3.4	3.8	7.6
Bankfull wetted perimeter (m)	6.6	4.9	3.9	6.1	4.3	4.2	8.5
Bankfull mean depth (m)	0.67	0.49	0.45	0.79	0.62	0.33	0.37
Cross sectional area (m^2)	3.23	1.84	1.43	3.96	2.10	1.28	2.90
Riffle w/d ratio	16.5	8.0	12.0	6.9	7.1	11.9	22.1
Substrate composition							
Less than 2 mm (%)	89	68	77	44	60	17	.
Canopy cover							
Overall density (%)	70	77	53	48	95	96	57
Tree and shrub density (%) [a]	6	60	30	27	48	.	.
Large woody debris							
Total count	7	7	7	28	10	8	26
Volume (m^3)	0.22	0.58	0.77	4.26	0.91	0.21	10.79
Loading (m^3/100 m)	0.14	0.38	0.51	2.79	0.59	0.14	7.08
Residual pools							
Mean depth (m)	0.56	0.58	0.41	0.51	0.44	0.37	0.49
Minimum depth (m)	0.34	0.37	0.30	0.36	0.36	0.37	0.37
Maximum depth (m)	0.88	0.93	0.67	0.74	0.64	0.37	0.67
Number of pools	15	9	8	4	11	1	6
Residual pool volume (m^3)	.	4.9	.	4.9	2.9	.	5.1

[a] Metric was not differentiated from overall canopy density for Bozard 3 or Lake 11

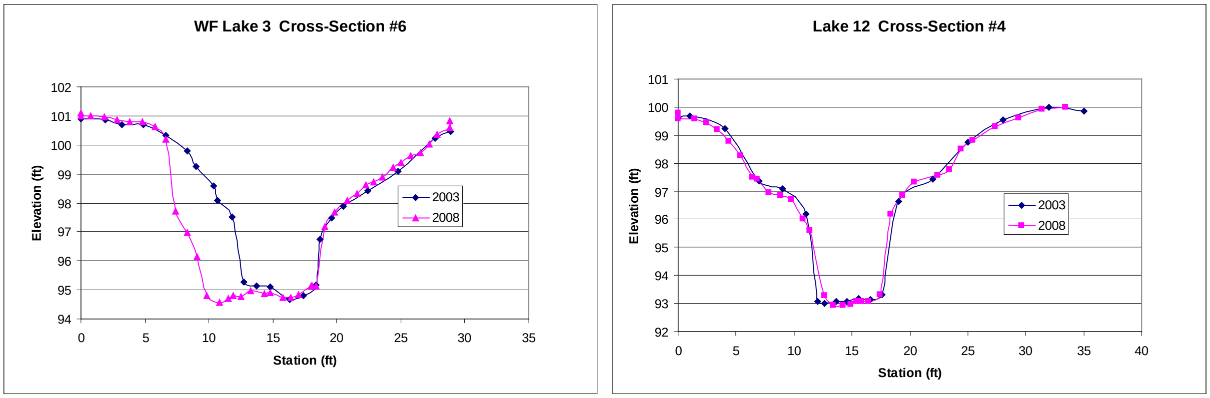

Figure 21. Comparisons of cross-sections at site 3 in WF Lake Creek site (left panel) and site 12 in upper Lake Creek (right panel) that were surveyed in both 2003 and 2008.

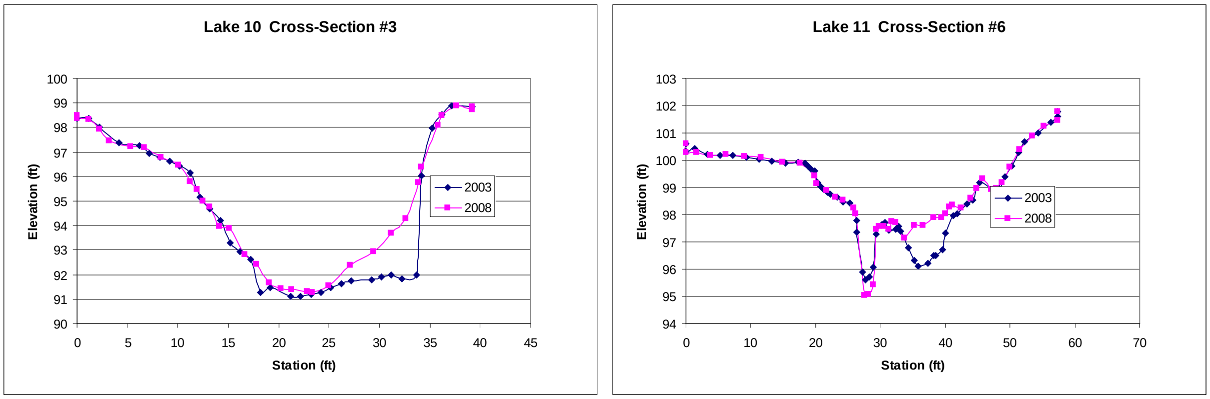

Figure 22. Comparisons of cross-sections at site 10 (left panel) and site 11 (right panel) in upper Lake Creek that were surveyed in both 2003 and 2008.

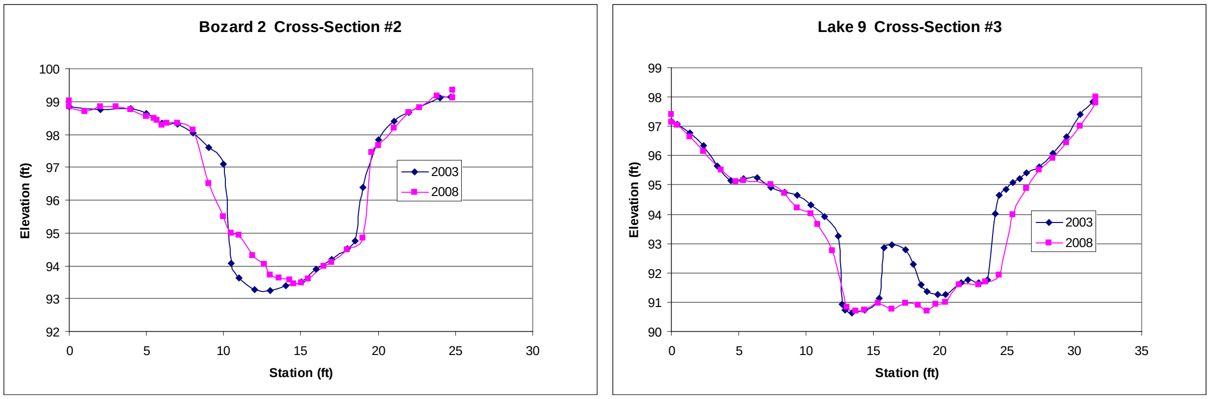

Figure 23. Comparisons of cross-sections at site 2 in Bozard Creek (left panel) and site 9 in the Lake Creek mainstem (right panel) that were surveyed in both 2003 and 2008.

3.3.5 Effectiveness monitoring – Response to brook trout removal in Benewah Creek

A total of 1351 brook trout were captured in the upper Benewah Creek watershed during removal efforts that occurred over a period of 15 d from August 12 to October 1 in 2008 (Table 25). Of these 1351, 829 (61%) were removed from approximately 5.2 km of contiguous mainstem habitat that extended upstream from 9-mile bridge to the 12-mile bridge. An additional 384 (29%) fish were removed from a 2.0 km mainstem reach upriver of 12-mile bridge to the confluence of the two Benewah forks. The other 138 (10%) fish were removed from approximately 1.5 km of lower tributary habitat in WF Benewah and SF Benewah. Brook trout densities (fish / km) were highest in the upper mainstem reach and lowest in tributary reaches. A greater percentage of fish collected in mainstem reaches than in tributary reaches were larger than 150 mm in length (Figure 24). Of the total number of fish greater than 150 mm in length, 95% were captured in mainstem reaches. In addition, a greater percentage of large mature adults were captured below than above the RBW at 9-mile bridge. Of the 52 brook trout captured below the RBW, 21 (40%) and 14 (27%) were at least 200 and 250 mm, respectively. In comparison, of those fish captured in reaches above the RBW, only 147 (11%) and 29 (2%) were at least 200 and 250 mm, respectively.

One hundred and ninety of the brook trout captured in the Benewah watershed were evaluated for maturation status. Eighty-two of the 190 were classified as males, with 53 of these identified as mature. Gonadal weights were collected from 37 of the 53 mature fish. The remaining 105 were females with 86 of these identified as mature. Ovarian weights were collected from 76 of the fish, and fecundity was either measured directly or estimated for 42 of these 76. For each of 19 females, egg counts were conducted on both the subsampled ovary (22-40% of ovarian weight) and the entire ovary to assess the relationship between predicted and observed fecundity (Figure 25). The preciseness of the derived relationship (slope not significantly different from one; R^2 = 0.9885) supported using the percent weight of the ovarian subsample to expand fecundity estimates for those fish in which all eggs were not counted. In Alder Creek, 46 males and 38 females were sacrificed to obtain comparable maturation data. Of the 46 males, 26 were identified as mature, with gonadal weights collected from 12 of the 26. Eighteen of the 38 females were mature, with fecundity and ovarian weights collected from 9 of the 18; many of the other sacrificed mature females in Alder Creek had already spawned.

Maturation schedules differed among years from 2004 to 2008 for brook trout of both sexes captured in the upper Benewah watershed (Figure 26). Females matured at a smaller size in 2006 than in 2008 (p = 0.009), but at a larger size in 2007 than in 2008 (p = 0.002). However, significant difference in length-specific maturation probabilities were not detected between 2008 and the first two years. For males, brook trout were predicted to mature at a smaller size in 2004 than in 2008 (p < 0.001), but a significant difference in maturation probability was not detected between 2007 and 2008 (results for 2005 and 2006 could not be confidently evaluated given the lack of immature males sampled in these two years). Notably, males sampled in Benewah tributary habitats were predicted to mature at a smaller size than those collected in mainstem reaches when only 2008 data were analyzed (p = 0.062). In addition, when maturation probabilities for male brook trout were examined separately by year to assess watershed differences, no differences were detected between Alder and Benewah creeks in 2004, but in 2007 and 2008, males were predicted to mature at a significantly smaller size in Alder than in Benewah Creek (p = 0.001). Predictions from the sex-specific logistic regression models indicate that of the 1351 brook trout removed from upper Benewah Creek in 2008, 250 (19%) were mature females and 198 (15%) were mature males (Table 25).

Female brook trout were significantly more fecund in 2008 given their body size than similar sized fish assessed in 2004, 2006, and 2007 (Year effect: $F = 10.07$, $p < 0.001$; Tukey pairwise comparisons, 2008 > 2004 = 2006 = 2007). However, the detected annual differences among the fecundity-at-length relationships were similar between the two systems as supported by the lack of significant system-by-year interaction terms (System-Year interaction, $F = 0.726$, $p = 0.537$; System-Year-Length interaction, $F = 1.635$, $p = 0.182$; Figure 27). In other words, even though size-specific fecundities were greater in 2008 than in the other three years for Benewah Creek fish, a similar upward shift in size-specific fecundity was detected in Alder Creek fish.

More than 7000 brook trout have been removed from the upper Benewah watershed since the incipience of the suppression program (Table 25). Numbers of fish removed substantially increased over the first three years of effort, in large part due to the progressive targeting of additional mainstem habitat. In addition, larger fish were found more frequently in mainstem than in tributary habitats. Twelve to eighteen percent of removed fish were greater than 150 mm during the years in which tributaries were primarily targeted, whereas percentages as high as 40 have been recorded as more of the mainstem habitat has been electrofished in recent years. The trends in the percent of mature adults removed, from 12-14% to 26-29%, also reflected this spatial re-distribution of effort from 2004 to 2006. However, numbers of fish captured and the estimated percent of mature adults have decreased over the last two years of removal efforts (Table 25).

Overall, abundances of age 1+ brook trout (fish >75 mm) estimated at survey index sites in the upper Benewah watershed were not significantly lower in 2008 than before program initiation in 2004 (Wilcoxon rank sum test, $p = 0.814$). Abundances had increased at some sites, such as site 16 in the mainstem and at the lower Schoolhouse Creek site, whereas at other sites, such as those in the West Fork, abundances were considerably reduced (Table 26). Abundances at many of the other sites in upper Benewah remained relatively low. Conversely, brook trout abundances compared over a similar time period in the upper Alder Creek watershed have significantly increased (Wilcoxon rank sum test, $p = 0.011$; Table 26). Furthermore, brook trout abundances in Alder Creek in 2008 were greater than mean pre-program values for 6 of the 8 index sites that had mean abundances greater than 15 during the pre-program period.

Table 25. Summary of stream length sampled and brook trout removed from two mainstem reaches and tributary habitats in the upper Benewah watershed, 2004-2008. Probability of maturation models, derived separately for each year and sex, were used to assign maturation status to fish that were not assessed.

	9-mile bridge to 12-mile bridge			12-mile bridge to confluence of south and west forks			Tributaries					Mature fish removed (%)	
Year	Survey length (km)	Fish removed	Fish/ km	Survey length (km)	Fish removed	Fish/ km	Survey length (km)	Fish removed	Fish/ km	Total fish removed	Percent fish > 150 mm	Females	Males
2004	.	.	.	0.5	61	122.0	3.7	605 [a]	163.5	666	12	81 (0.12)	95 (0.14)
2005	0.8	193	241.3	2.0	962	481.0	3.7	233	63.0	1388	18	319 (0.23)	207 (0.15)
2006	3.4	1192	350.6	2.0	904	452.0	3.7	421 [b]	113.8	2517	36	736 (0.29)	659 (0.26)
2007	6.0	514	85.7	2.0	311	155.5	3.7	260 [c]	70.3	1085	40	181 (0.17)	141 (0.13)
2008	5.2	829	159.4	2.0	384	192.0	1.5	138	92.0	1351	31	250 (0.19)	198 (0.15)

[a] All but 5 of the fish were removed from South and West Benewah forks

[b] All but 33 of the fish were removed from South and West Benewah forks

[c] All but 28 of the fish were removed from South and West Benewah forks

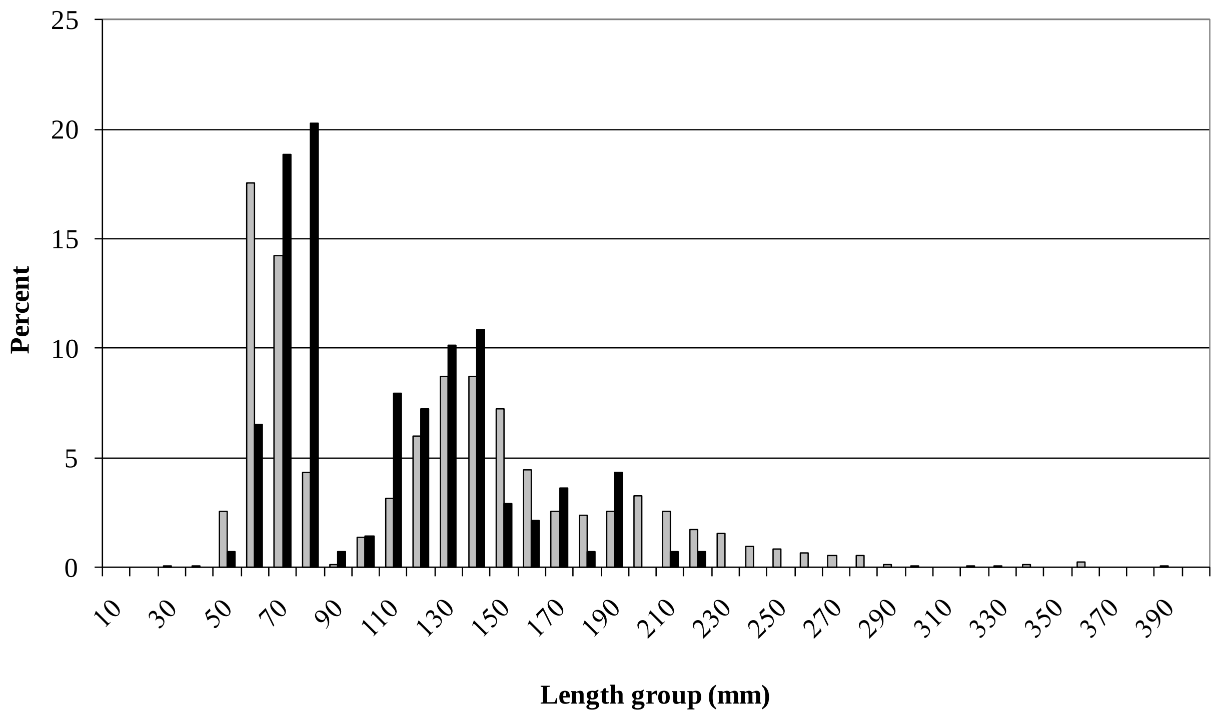

Figure 24. Relative length distributions of captured brook trout, calculated separately for fish removed from mainstem (grey bars) and tributary (dark bars) reaches in the upper Benewah watershed in 2008.

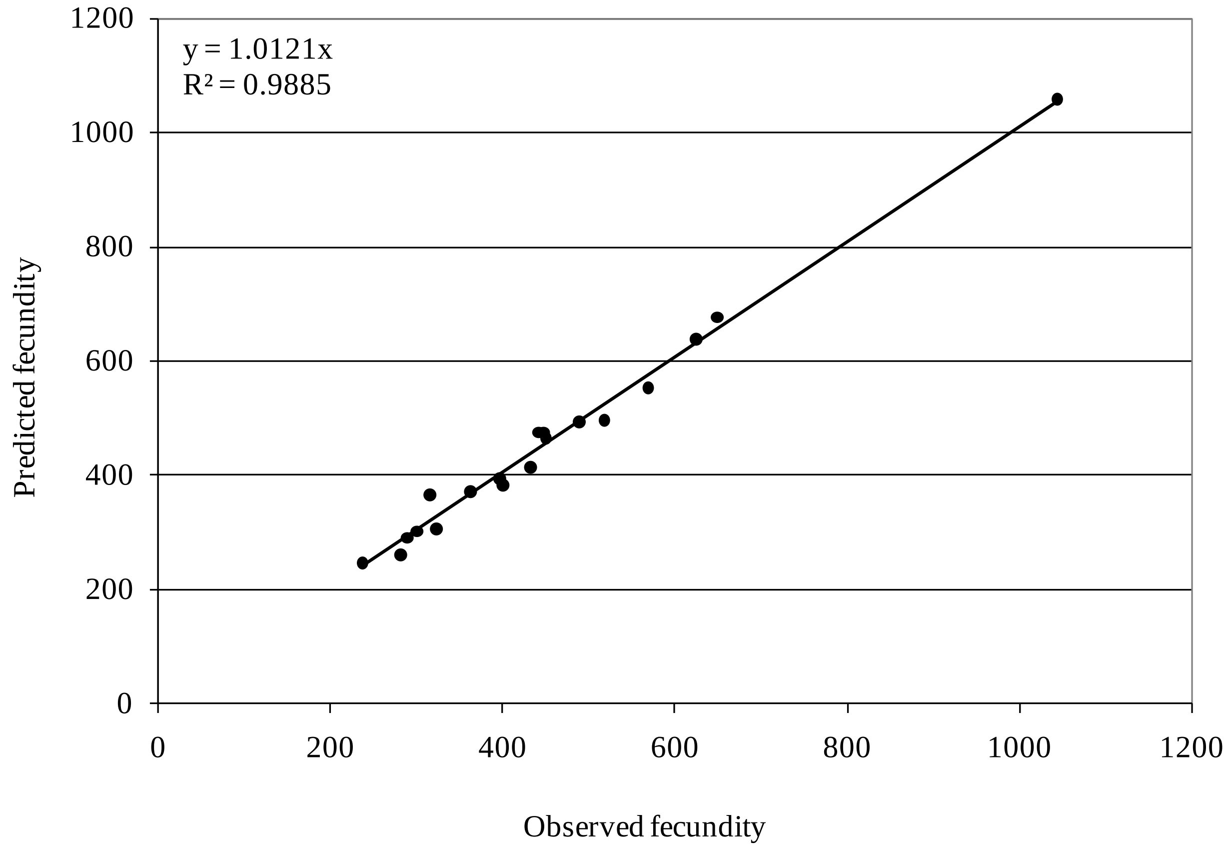

Figure 25. Relationship between predicted and observed fecundity for 19 brook trout captured in the upper Benewah watershed in 2008. Predicted fecundity was estimated by expanding the egg counts in the ovarian subsample by the percent weight of the subsample. The slope of the derived equation was not significantly different from one (intercept was set to 0).

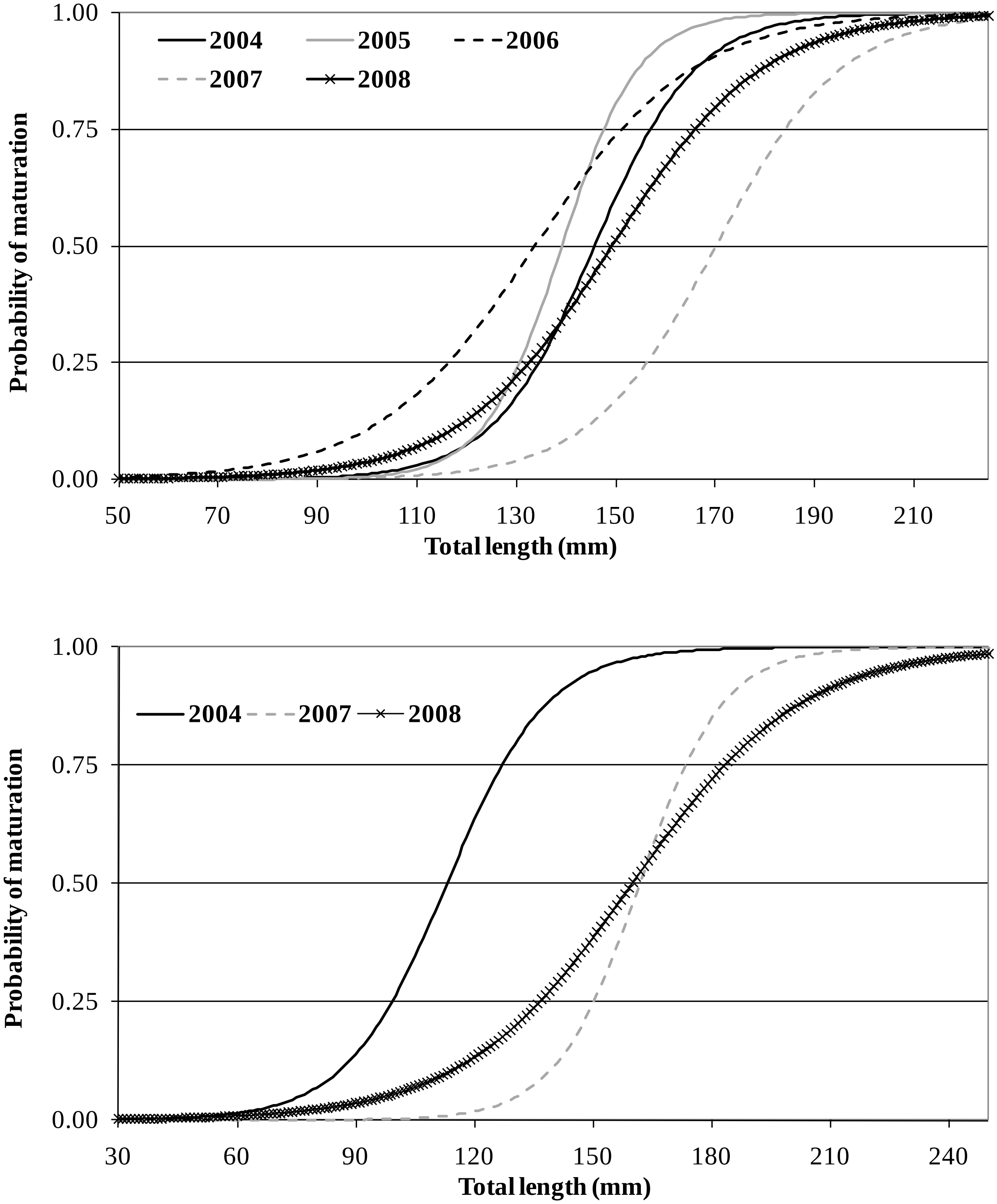

Figure 26. Probability of maturation curves estimated from logistic regression equations derived separately for female (upper panel) and male (lower panel) brook trout captured in the upper Benewah watershed, 2004 to 2008.

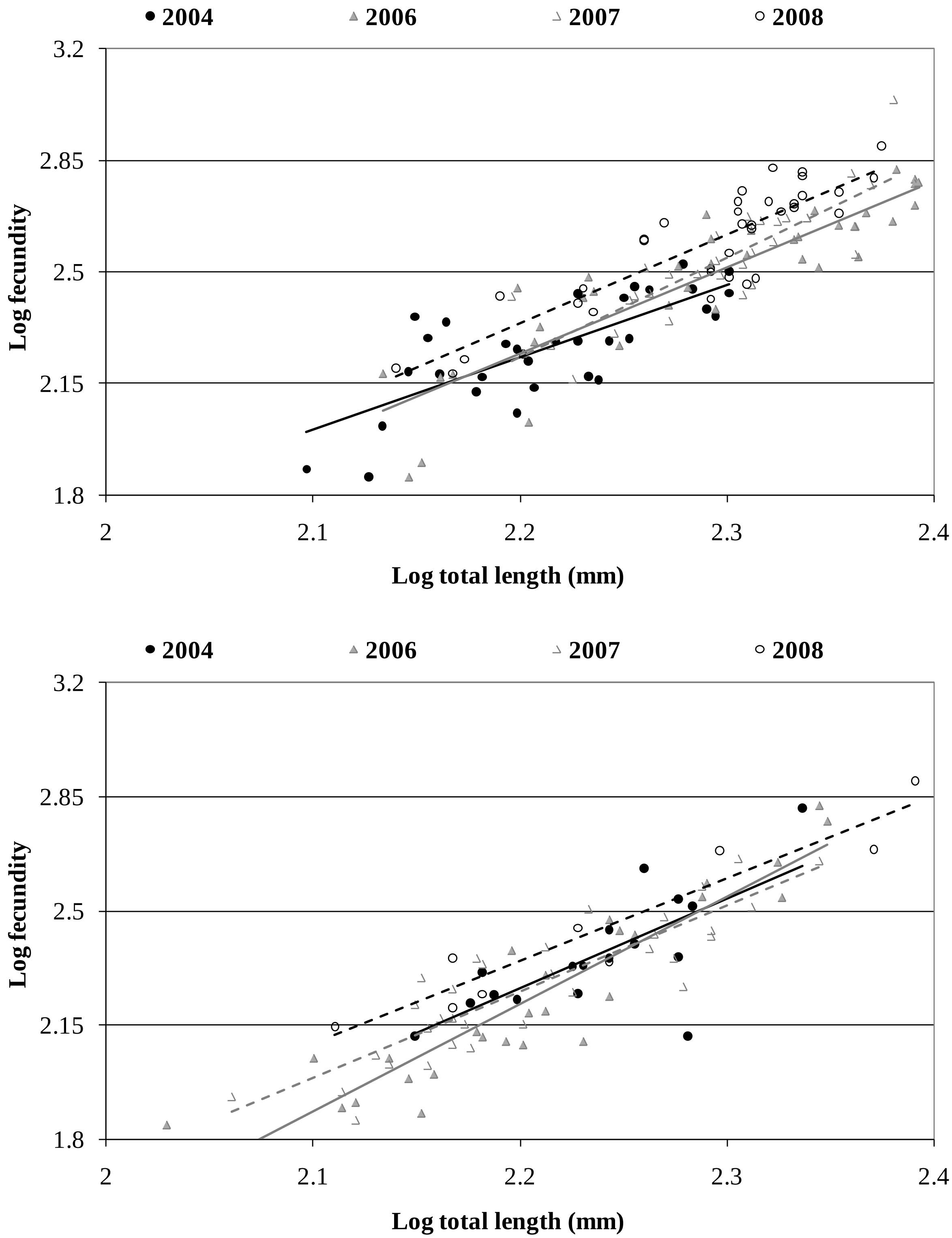

Figure 27. Relationship of fecundity to total length (log transformed) for brook trout removed from the upper Benewah watershed (upper panel) and sacrificed from the Alder Creek watershed (lower panel) in 2004 and from 2006 to 2008. Only those fish less than 250 mm total length were used in regression models to ensure consistency among years and between watersheds.

Table 26. Abundance of brook trout (larger than 75 mm) estimated at survey index sites before the initiation of the brook trout suppression program in 2004 and after four consecutive years of removal efforts. Brook trout control was only implemented in the upper Benewah Creek watershed, whereas the upper Alder Creek watershed served as the control. Bold values denote greater values in 2008 than during 2002-2004.

		Brook trout abundance	
Stream	**Site**	**Mean, 2002-2004**	**2008**
Upper Benewah Creek watershed			
Benewah mainstem	15	0.67	**2.00**
Benewah mainstem	16	2.67	**12.59**
Benewah mainstem	17	8.07	3.00
Whitetail creek	1	1.00	1.00
Whitetail creek	2	0.00	0.00
Windfall creek	1	0.00	**3.00**
Windfall creek	2	0.00	**2.00**
Schoolhouse creek	1	5.16	**16.08**
Schoolhouse creek	2	0.33	0.00
South Fork Benewah	1	5.10	5.03
South Fork Benewah	2	1.67	0.00
South Fork Benewah	3	0.50	**2.00**
West Fork Benewah	1	18.32	1.00
West Fork Benewah	2	12.67	6.15
Upper Alder Creek Watershed			
Alder mainstem	11	16.08	12.13
Alder mainstem	12	22.97	**24.00**
Alder mainstem	13	44.25	**58.93**
Alder mainstem	14	47.02	**51.28**
Alder mainstem	15	14.58	**28.51**
Alder mainstem	16	11.89	**29.11**
Alder mainstem	17	20.42	8.00
North Fork Alder	1	16.24	**34.43**
North Fork Alder	2	12.88	**16.57**
North Fork Alder	3	5.40	**21.01**
North Fork Alder	4	16.49	**21.78**
North Fork Alder	5	14.48	**36.02**
North Fork Alder	6	18.14	**23.03**
North Fork Alder	7	13.77	**17.02**
North Fork Alder	8	6.00	5.22

3.4 Discussion

3.4.1 Status and trend monitoring – Biological indices

3.4.1.1 Index site cutthroat trout abundance

In both Benewah and Lake Creeks, elevated numbers of westslope cutthroat trout (WCT) were observed across many of the sites sampled in tributary reaches in 2008. However, in tributaries of the Lake Creek watershed, most of the cutthroat trout were constrained to upper reaches, with estimated abundances similar to those documented over the last four to five years. Given the current level of adfluvial spawners returning to the upper Lake Creek watershed, the relatively stable juvenile densities in upper tributary reaches may suggest that a carrying capacity has been reached. Further, the sub-optimal conditions present in lower reaches of these upper tributaries (e.g., low LWD loadings and high percent fines in riffle habitats) may provide limited opportunities for successful downstream expansion. These spatial trends should prove useful in prioritizing areas for prospective habitat improvements. Targeting the lower reaches of these tributaries for restoration should not only increase the spatial distribution and connectivity of WCT (e.g., promote a more robust meta-population structure) but also improve the carrying capacity in the upper portion of Lake Creek.

In comparison to Lake Creek, estimated abundances across most of the sampled sites in the upper Benewah watershed were typically greater than the 5-6 year average and displayed strong positive trends over the last four years. Given that these trends were displayed at sites where high densities have frequently been documented during our annual surveys, the results suggest a genuine overall increase in juvenile densities along these tributary reaches rather than a redistribution from more populated to less populated sites. The short-term positive trends in cutthroat trout abundances noted in upper Benewah Creek may have been attributed to regionally favorable environmental conditions that increased either spawning success or early life-stage survival rates. Elevated densities were not only found across tributaries within the Benewah Creek watershed, but also at sample locations in Evans Creek, a spatially-distinct subbasin of the Coeur d'Alene system. Because most of the captured fish in our summer surveys comprised immature juveniles of age 2 or younger (based on length at age keys (Vitale et al. 2003)), relatively strong year classes across watersheds within the last two to three years could have given rise to the trajectories observed. Concordant population abundances, indicative of regional climatic influence, have commonly been reported in regional networks of small salmonid streams (Platts and Nelson 1988; Gowan and Fausch 1996). Trends in juvenile rearing densities of cutthroat trout measured in other Northern Idaho sub-basins may aid in elucidating whether favorable in-stream conditions existed across the region in the recent past. On the other hand, the results noted in our surveys may have been due to a recent increase in in-lake survival rates that translated into a larger number of returning adfluvial spawners and greater reproductive output. However, a favorable lacustrine environment would not explain the trends noted for the prevailing resident population in Evans Creek. In addition, the increasing trends observed in tributaries of the upper Benewah watershed cannot be adequately explained by an increase in spawning adults given that only 9 to 12 adults have been captured annually in the DN trap from 2004 to 2007.

Alternatively, the elevated numbers of WCT recorded at sites across tributaries in the upper portion of the Benewah Creek watershed may have been a response to actions recently implemented to address factors limiting population recovery. Aggregate densities of cutthroat trout summed across sites in the upper Benewah watershed were found to increase at a more aggressive rate than those summed across sites in other monitored watersheds, most notably those in tributaries of upper Lake Creek. In the Benewah watershed, extensive channel reconstruction to increase habitat complexity, improve floodplain connectivity, and reduce mainstem summer rearing temperatures has occurred in upper mainstem reaches over the last four years. In addition, the Fisheries Program has been actively engaged in a brook trout suppression program in upper mainstem reaches of Benewah Creek and associated tributaries since 2004. In comparison, Lake and Evans Creek have received minimal habitat intervention in recent years. Consequently, the current trajectories observed may suggest a positive response to the collective influence of habitat restoration and non-native fish removal. As we collect additional years of data, the continued use of Lake and Evans Creek as controls will allow us to further evaluate whether this apparent trend is a response to the corrective actions implemented in the Benewah watershed.

Though index site data allow an evaluation of relative reach-specific changes over time, they do not permit a reliable examination of trends in absolute abundances at the reach scale. The small percentage of available habitat that is sampled during our population surveys and the high variability in estimated densities typically observed among sample sites both contribute to a high level of uncertainty when expanding estimates over larger spatial scales (Firehammer et al. 2009). In addition, various authors have cautioned against the use of depletion-removal estimates as unbiased measures of fish abundance. Removal estimators have been found to overestimate capture probability during subsequent passes and consequently underestimate population size for salmonids in small stream systems (Riley and Fausch 1992; Peterson et al. 2004b). The decrease in capture efficiency with each subsequent pass has also been found to be influenced by habitat features such as pool area and the presence of large woody debris (Rodgers et al. 1992; Rosenberger and Dunham 2005), rendering it difficult to account for the negative bias consistently across all sample sites. The multipass technique can also be time-consuming, especially in our watersheds where the relatively large amount of fine sediment leads to much time expended between passes waiting for water clarity to improve. Because of all these biases and shortcomings, an index of abundance may be preferable to an absolute estimate, providing that it tracks true abundance over time.

Given these concerns and the desire to increase sampling efficiency, we are considering using a single-pass estimator as an index of salmonid abundance in our watersheds. Others have 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). In 2009, we intend to conduct a small-scale study within the purview of our annual sampling regime to more accurately examine the predictive abilities of a single-pass index to track absolute abundance. Approximately twenty-five index sites will be selected across varying levels of habitat complexity, such as pool area and LWD volume, and mean salmonid density (estimated from past surveys) in our watersheds. For each site, fish captured by a single pass electrofishing effort will be marked by a fin-clip and released within the blocked-off site. The next day the site will be re-visited and sampled again using our multipass removal protocol, and numbers of marked fish captured during each subsequent pass will be recorded. The relationship between marked fish captured during the first pass and known marked fish for each site will then be evaluated across all sites using regressive techniques, and will allow us to assess how habitat complexity and density affects the precision of this relationship. Results from this study are intended to support the utility of using a first pass index as an abundance indicator for monitoring salmonid trends in our watersheds. Single-pass efforts will permit additional sites to be incorporated into our annual sampling protocols, and will thus enable us to expand our efforts across a greater percentage of our watersheds to better understand distributional changes in salmonid populations.

Further, although observed trends in cutthroat trout densities from index site surveys may imply a positive response to implemented actions, they do not permit a rigorous evaluation of the effectiveness of recovery measures. For example, in the Benewah watershed, positive trends were observed primarily at sites in upper mainstem and tributary reaches that have not been directly modified by channel reconstruction. In addition, index site data alone is insufficient when attempting to separate the potentially confounding influences of habitat restoration and brook trout suppression on population response. Complementary analyses that were used to assess the effectiveness of both large-scale actions implemented in Benewah Creek are discussed more fully in following sections. In addition, other metrics of watershed-scale productivity, such as outmigrants per spawner, may be more useful in tracking overall fish response to changes in stream rearing environments over time (Bradford et al. 2005). Currently, the Fisheries Program is engaged in a trapping and tagging program to monitor changes in the number of outmigrating juveniles and returning adfluvial spawners in both Lake and Benewah creek watersheds. This is discussed more fully in the following section.

3.4.1.2 Adfluvial cutthroat trout migration

One of the primary objectives of our recovery efforts is to augment the number of returning adult cutthroat to our adfluvial watersheds. Thus, it is imperative that we reliably track temporal changes in adult spawners, not only to monitor the achievement of this objective, but also to derive outmigrant per spawner indices that will permit an evaluation of the response of in-stream production to our restoration actions. However, this requires a rather precise estimate of spawning adults, which currently cannot be attained by using adult counts at upriver traps because of the inconsistency in trapping efficiency that can occur due to flow regime variability across years. Although trap efficiency was markedly improved when the conventional, fixed-weir design was replaced with the RBW design, high levels of spring discharge can depress RBW panels below the water surface permitting fish to pass. For example, in Benewah Creek where the RBW design was implemented for the first time in 2008, only one adult WCT was captured. Though adult upriver trap counts have been consistently lower in Benewah than in Lake Creek in the recent past, high spring flows, which were found to temporarily depress trap panels in 2008, likely contributed in part to the absence of captured WCT in the Benewah trap. In addition, high levels of discharge compromised RBW trap efficiency in Lake Creek repeatedly during the spring, and most likely contributed to the lower number of fish captured in 2008 (< 40 fish) than in recent years (> 100 fish in 2 of the last 3 years). Panels were observed submerged below the water during time periods in which PIT-tagged adults were briefly detected by the array, likely representing peak periods of prompt upriver movement for many of the adults. Downriver adult trap counts in Lake Creek corroborated the inadequate performance of the upriver trap. Similar to previous years, more than twice as many adfluvial adult WCT were captured in the downriver than our upriver trap in 2008. To improve upon performance, prospective modifications to the RBW design are considered for implementation in Lake Creek in 2009. Trap panels will be able to be manually lowered or raised, using a cabled pulley system, to maintain their height above the water surface.

The observed discrepancy in adult counts between traps in Lake Creek, however, may in part be attributed to a portion of the run exhibiting an early migratory behavior, ascending upriver in late fall or winter before RBW trap installation. Extensive fall and winter seasonal movements to deep pools that provide suitable overwintering habitat have been documented for cutthroat trout (Jakober et al. 1988; Brown 1999; Brown and Mackay 1995; Lindstrom and Hubert 2004). Protracted upriver spawning migrations have also been reported for sea-run coastal cutthroat trout, with some populations exhibiting bimodal peaks in timing separated by at least two months (Johnson et al. 1999). Early ascension by pre-spawning adults has also been described for other spring-spawning salmonids (Mayer et al. 2006). Given that the ratio of males to females was higher for those fish captured in the DN than in the RBW trap in Lake Creek in 2008, either males were more able to navigate the RBW under high discharge levels than females, or sex-specific differences in pre-spawning migratory behavior were occurring. Though it is unclear which mechanism was operating in 2008, the prevailing evidence does not support a substantial early migratory component. For example, given that the RBW was installed in mid-November in 2007 and presumably did not incur any losses to its structural integrity throughout the winter, we should have expected to capture at least one adult before the spring if a considerable portion of the run migrated early. In addition, PIT-tagged adults were initially detected by the Lake Creek array predominantly during time periods in which upriver migrating adult fish were intercepted by the RBW trap. If initial detections had occurred within the time period in which post-spawn migrants were captured in the downriver trap, this could have denoted PIT-tagged adults had ascended early before the array was operational, and would suggest the potential for other early migrants. Additional tag detections in future years should permit a better assessment of whether early migratory behavior by returning adults occurs in our watersheds.

Given the unreliability of trap counts to permit estimates of adult returns, an attempt was made to estimate spawning abundance using recaptures of adult fish in downriver traps (DN) that were tagged in RBW traps. However, the mark that was used, a hole-punch placed in the upper lobe of the caudal fin, was not completely recognizable in those adults captured in the DN due to fin abrasion on spawning grounds. Accordingly, the low number of identifiable recaptured fish, generated a highly imprecise and biased estimate of spawning abundance for Lake Creek in 2008. Such an imprecise estimate will not permit the detection of trends in adult returns in our adfluvial watersheds. In 2009, we plan to double-tag adults captured in the RBW traps, using more semi-permanent marks than fin punches. First, the outer margin of the right opercle will be hole-punched for all adults. Given the cold water temperatures and the relatively short period of time spent by fish on the spawning grounds (Firehammer et al. 2009), the punch should not rapidly heal over and thus remain recognizable until recapture in DN traps. In addition, for all adults that were not PIT-tagged as juveniles, a PIT-tag will be inserted into the muscle tissue 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). The recapture of opercle-punched fish will permit both a calculation of spawner abundance and an assessment of short term PIT-tag retention for post-spawn fish. In turn, PIT-tags will permit an estimate of long-term post-spawn survival and return frequency, two metrics which currently cannot be accurately estimated given the low number of PIT-tagged juveniles that have been detected in our watersheds.

Periods of high discharge, some of which were damaging to trap structure, also frequently compromised downriver trap (DN) performance in 2008, especially in the Lake Creek watershed. Numbers of post-spawn adults and juveniles captured in the Lake Creek DN trap were consequently lower relative to that recorded over the last couple of years. However, periods of inefficient trap performance should not prevent relatively precise spawner estimates from being obtained in future high water years under the new marking protocol, given that a sufficient number of post-spawn adults are able to be captured in the DN trap (e.g. 124 were able to be captured in the high water year of 2008). Conversely, high water years may have a serious impact on the calculation of reliable outmigration estimates for juvenile WCT. In Lake Creek in 2008, the inability to capture both tagged and untagged outmigrating juveniles during periods when either trap panels were damaged, or removed to prevent damage, not only generated imprecise trap efficiency estimates during release trials, but also yielded negatively-biased estimates of abundance. In order to generate more precise estimates of juvenile outmigrant abundance, the performance of the DN trap needs to be improved so that it can capture fish throughout the spring in all but the most extreme water years. Currently, the configuration of the Lake Creek DN trap funnels the channel's flow through a relatively small surface area, which increases current velocities in the immediate vicinity upriver of the trap at high flows, and increases the probability that the trap screens become clogged with debris (e.g., Alder catkins), which consequently impounds water behind and then over and around panels. In 2009, we plan to add more panels to lengthen both arms that extend from the apex of the trap upstream to the stream banks. This will not only re-distribute flow over a larger screened surface area, but should also attenuate some of the focused pressure that is currently unduly placed on the panels.

Juvenile outmigrant estimates were also found to be influenced by the behavior of fish that were used in release trials to estimate trap efficiencies. Similar to those results obtained in Lake Creek in 2007, comparative analyses of juvenile downriver movements in Lake Creek in 2008 indicated that fish released above the DN trap during moderate levels of flow were more likely to linger than those released below the trap. Either the release-trial fish were engaging in trap-avoidance behavior or they had difficulties in negotiating the trap. Under low to moderate levels of stream discharge, the DN trap tends to create a slack-water environment immediately upriver and consequently may not present appropriate velocities to cue downriver movement. Similar delayed movements have been noted for juvenile salmonids outmigrating through impounded reaches of larger river systems (Venditti et al. 2000). Because differences in rates of trap passage among outmigrating juveniles were observed, the assumption of equal probability of recapture was likely violated. In each trial, the number of marked fish that were available for recapture was probably less than the number of marked fish released. Inflating the number of available marked fish positively biases estimates of outmigrant abundance in stratified mark-recapture analyses, especially if trap efficiencies change markedly over trial periods. To remedy this potential bias, we were able to use downriver array detections of juveniles that had evaded the trap to provide estimates of the numbers of marked fish available for recapture. This adjustment presumed all fish that had bypassed the trap were detected by the antenna array, a presumption that is not unfounded given that 99% of fish released below the trap in 2008 were detected. This model modification not only should increase the accuracy of outmigrant abundance estimates but also their precision under most spring flow regimes.

In addition to obtaining outmigration estimates, PIT-tagging has also been conducted in our watersheds to better understand the processes affecting survival rates of adfluvial fish during early lake residence. To ensure that the survival rates of the entire cohort is reflected by those observed in tagged juveniles, it is necessary to capture the full range of expressed traits of the juvenile outmigrant run. In both watersheds, juveniles of all lengths classes were tagged representatively throughout the entire outmigration period. Consequently, if size at outmigration or at timing of lake entry significantly influences the likelihood of survival to adulthood, survival estimates for this cohort should not be unduly biased.

The size of outmigrating juveniles was generally larger for those cutthroat captured in the DN trap in Benewah Creek than for those captured in Lake Creek. However, many of the large fish in the Benewah DN trap were captured in mid to late June at the end of the outmigration period, so it is unclear whether these fish were adfluvial migrants actively moving out of Benewah Creek or were resident fish that were engaging in short-distance early-summer feeding excursions that were inadvertently captured by the trap. In addition, many of the large fish captured in the Benewah DN trap had external markings indicative of potential hybridization with rainbow trout (*O. mykiss*). Though a study conducted approximately 10 years ago indicated minimal hybridization between cutthroat and rainbow trout in tribal watersheds of the Coeur d'Alene basin (Spruell et al. 1999), we are planning on conducting an additional genetic study in 2009 to further evaluate the extent of potential genetic introgression, in combination with a more fine-scale evaluation of population structure, in adfluvial watersheds across the Coeur d'Alene basin.

Relatively few adult cutthroat trout that were PIT-tagged as juveniles in previous outmigrations were uniquely detected (i.e., not detected in previous years) in 2008. For example, only one additional fish from the Lake Creek outmigration cohort of 2005 was detected in 2008, increasing the total number of unique detections over the last three years from this tagged group to 14. Assuming high detection probabilities at the PIT-tag array, this equates to a return rate of only approximately 2% (14 of 688 tagged juveniles). Return rates from Lake Creek outmigrant classes of 2006 and 2007 are beginning to reveal similar results, given that only 10 fish from these two cohorts were detected in 2008. To date, none of the juveniles that were tagged in the Benewah system in 2007 (first year of tagging) have yet to be detected; however, as demonstrated in the Lake Creek system, many of the fish first return after two years of lake residence. Several more years of PIT-tag detections should provide a better assessment of return rates in our monitored adfluvial watersheds and of any potential differences in in-lake survival rates between Benewah and Lake Creek fish. More importantly, as more tagged fish are detected in subsequent years, we should be able to start examining probable linkages between factors such as juvenile growth and timing of outmigration and the likelihood for survival.

Several more years of adult return data may also aid in evaluating whether the differences in the size of returning adults between watersheds that was detected in 2008 is genuine or was just an artifact of the small sample size in Benewah Creek. A finding of consistently smaller sized adults in Benewah than in Lake Creek may indicate that there are less repeat spawners (i.e., a lack of older fish) present in Benewah Creek which in turn could suggest that post-spawn survival rates are lower for these fish. However, caution should be exercised when interpreting the age of an adult spawner from its size at capture. Our PIT-tag data suggest that whereas growth rates are relatively high during early lake residence before maturation, somatic growth may considerably decrease after the initial spawn. Additional information regarding the age, growth, and return rates of adfluvial fish in our watersheds should provide insight into potential mechanisms, such as predation, that may be impacting WCT populations in Coeur d'Alene Lake and whether the strength of these mechanisms differs depending on where they are operating within the lake.

3.4.2 Status and trend monitoring - Habitat metrics

3.4.2.1 Longitudinal water temperatures

The ambient stream temperatures recorded in Lake and Benewah watersheds still support the suitability of tributaries over mainstem reaches as cutthroat trout rearing habitats during mid-summer periods. In 2008, it was not uncommon for temperature in some of the monitored mainstem reaches in both watersheds to exceed those considered optimal for growth (e.g., 17^{o}C; Bear et al. 2007) more than 25% of the time during periods in which juvenile trout may be redistributing from natal tributary habitats to summer rearing habitats. In comparison, tributary temperatures remained below this optimal growth benchmark value over 90% of the time. Given the consistently higher densities of cutthroat trout observed in tributary than in mainstem habitats, the mid-summer differences in rearing temperatures between tributary and mainstem reaches likely explain in part the distributional patterns of cutthroat trout observed in our watersheds (Dunham et al. 1999; Paul and Post 2001; Sloat et al. 2001; de la Hoz Franco and Budy 2005).

However, not all mainstem reaches presented sub-optimal rearing temperatures for cutthroat trout in 2008. For example, in the Benewah watershed, summer temperatures in upper portions of monitored mainstem reaches remained below the benchmark value of 17^{o}C approximately 95% of the time, whereas downstream reaches more proximate to 9-mile bridge (located at river mile 8.9 upstream from Coeur d'Alene Lake) exceeded this value over 50% of the time. Much of this difference may be explained by the presence of nearby, off-channel groundwater sources located along the unrestored areas of the unconstrained, broad alluvial mainstem reach in the upper Benewah watershed. Various springbrooks along these mainstem reaches have consistently been monitored over the last several years, and have displayed temperature signatures during summer months that were much cooler than those recorded in adjacent mainstem habitats. In addition, data from the peizometers that were installed within floodplain habitats of the broad alluvial reach of the upper mainstem 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, the upper portion of the mainstem (~ 3 km above 9-mile bridge) is closer to these off-channel groundwater sources and/or receives significantly more cool groundwater inputs than downstream reaches that have already been restored. Progressive channel restoration along contiguous reaches within the upper mainstem valley segment that improves water retention capability and restores floodplain connectivity should increase the accessibility of these cold-water sources and promote hyporheic dynamics that moderate main channel summer temperatures. In turn, this should increase the availability of optimal rearing habitats for WCT and provide favorable corridors that promote tributary connectivity.

Continued monitoring of ambient mainstem stream temperatures in our watersheds should provide insight as to whether our habitat enhancement activities are moderating thermal regimes and increasing the distribution and amount of preferable rearing habitats for cutthroat trout. However, a proper evaluation will require accounting for all those drivers that may influence the thermal regime in any given year. For example, the unusually large snowpack that accumulated in our watersheds over the winter of 2007-2008 in combination with a prolonged spring runoff moderated stream temperatures during the summer of 2008 relative to that recorded during the previous summer. Temperature models that examine the influence of channel restoration actions on floodplain connectivity and groundwater input will thus require other inputs, such as canopy cover or descriptive indices of the annual flow regime, to clarify the linkages.

3.4.2.2 Physical habitat metrics

The heuristic exercise of the power and precision analyses informed our monitoring program of the network of sites that would be required to detect regional trends in habitat condition and to determine whether target objectives are being met within our watersheds. Generally, for most habitat metrics that were examined (i.e., all those except residual pool depth), it would require approximately 20 years of sampling to confidently detect subtle regional trends if only five sites were monitored. We felt that this timeframe was too long within an adaptive management paradigm to evaluate whether our restoration actions were having the desired effect or if habitat conditions in monitored sub-basins were changing unexpectedly over time. Increasing the number of monitored sites to approximately 10, however, would permit trend detection in half the time, and allow us to more promptly re-examine potential limitations to our implemented measures and respond with additional corrective actions. In addition, the trend analyses suggested that a rotating panel design could be readily incorporated into our watershed monitoring matrix without losing the power to detect regional changes in habitat conditions. Furthermore, a panel design would permit finer spatial resolution within monitored sub-basins because a greater number of sites across watersheds could be visited over time.

For many of the analyzed metrics, the inability to confidently detect subtle regional trends in a reasonable timeframe was the result of rather large empirical estimates of annual variability calculated at many of our monitored habitat sites. However, we expect this source of within-site variability to decrease over time, especially in those reaches that are restored, as our corrective actions gradually improve habitat conditions (e.g., percent fines in riffles, percent canopy cover) to those levels established by our target objectives. The relatively low levels of metric variability calculated from our reference tributary and mainstem sites (see section *3.3.3.2. Precision analysis for assessing the regional status of habitat attributes*) corroborate this assumption. In addition, increasing the number of samples collected at each site in any given year should provide a more accurate estimate of a site's mean metric value, which should in turn increase the precision of this estimate over time. For example, currently only 6 canopy cover measurements and 2 to 3 riffles are sampled for each habitat site. Increasing the number of samples for canopy cover to 10 and riffles to 5 should provide more precise mean estimates at a site without incurring an inordinate amount of additional sampling effort. However, increasing sampling effort within a site is not an option for improved monitoring of LWD availability given that a single value (e.g., number of LWD pieces or LWD volume per 100 m) is computed at each site. Rather, selecting a LWD metric that can be measured more rapidly, such as the number of pieces of a specified size class, may be a more viable option to improve our sampling efficiency under a more intensive (i.e., additional sites) monitoring design. In addition, choosing a LWD metric that can be measured more easily (i.e., counts rather than volume) may remove some of the subjectivity and measurement error that can inflate estimates of annual variability.

The precision analyses allowed us to assess the number of samples that would be required to confidently evaluate the 'restoration distance' between a metric's current value and the desired target objective in a specified region of our monitored watersheds. Similar to the trend analyses, results suggested that additional measurements would be required to obtain a relatively precise estimate for all but the residual pool depth metric. For example, approximately 90 canopy cover measurements and 30 riffles would need to be sampled in any given year to effectively evaluate the current status of regional habitat conditions. However, under a regional monitoring matrix of 10 sites that includes the aforementioned number of samples per site for each metric (see paragraph above), the required number of samples for a desired precision level could be readily attained. Furthermore, as with the trend analysis, as habitat conditions approach those that are desired in our sub-basins, the expected level of variability among regional measurements should decrease, thus improving the precision around each metric's estimate. For example, the current patchiness of canopy cover within recently restored mainstem reaches of the upper Benewah mainstem yields relatively high estimates of variability among collected measurements across our established index sites. As riparian conditions improve over time across this restored reach, we should expect to see not only higher percent canopy cover measurements but less variability among these measurements.

3.4.3 Effectiveness monitoring – Response of indicators to habitat restoration

3.4.3.1 Thermal response to restoration in the Benewah watershed

In mainstem reaches of the upper Benewah watershed that underwent large-scale channel restoration from 2005 to 2007, thermal heterogeneity was found to be more prevalent, during time frames critical for salmonid rearing, after than before implementation of these measures. The presence of cool-water refugia in these restored reaches were often detected at depths greater than 1 m and were apparently created by the concomitant deepening and lengthening of pool habitats during the process of streambed elevation in designated riffles. The creation of these refugia should increase the availability of suitable rearing habitat for cutthroat trout in mainstem habitats of the Benewah watershed. Cold-water patch frequency and area have been considered important indices that explain salmonid occurrence and abundance in other small stream systems (Torgersen et al. 1999; Ebersole et al. 2001, 2003). Our data also suggests that the detection of these refugia may only be apparent during periods of elevated ambient stream temperatures. Therefore, in order to measure the thermal response to restorative actions that increase pool depth, it is essential that monitoring efforts are conducted during appropriate time periods.

3.4.3.2 Habitat response to restoration in the Benewah watershed

Generally, those habitat metrics that have been linked to the suitability of salmonid rearing habitat displayed a positive response to the restorative actions that have been implemented in the upper Benewah watershed over the last four years. For example, the mean percent fines measured in riffle habitats in restored reaches was lower than that measured in unrestored habitats and approached that found in our reference site. Similarly, LWD loadings were substantially higher in restored mainstem and tributary reaches than untreated areas and reflected those measured at the reference site. Residual pool metrics were also appreciably greater in restored than unrestored mainstem sites, with the greatest difference observed for pool volume. Because thermal refugia were often detected in large pools in restored mainstem habitats, pool volume, a metric that was first measured in 2008, should continue to be monitored in mainstem reaches of the upper Benewah watershed to accurately assess the amount of suitable habitat that is augmented due to our channel reconstruction activities.

Measured differences between restored and unrestored habitats was likely the direct result of our reconstruction activities. Along restored reaches, large substrate was imported into designated riffle habitats, large woody debris was introduced to both stabilize banks and create structure in pool habitats, and deep pools were created both through the re-meandering of lost channel length and the concomitant elevating of riffle habitats. Periodic monitoring of both treated and control reaches in the upper Benewah watershed will permit the detection of potential changes in this initial 'restoration effect' as habitats are no longer acutely, artificially disturbed by our reconstruction activities and as ecological processes are allowed to equilibrate over time.

Indices that indirectly influence those metrics linked to suitable salmonid habitat have also shown a positive response to our restoration actions in the Benewah mainstem. For example, modeled bank erosion rates were 75% less in restored than in restored reaches, with a total estimated reduction in sediment yield of over 1295 metric tons / yr. This reduction is deemed substantial given that the erosion assessments that have been completed to date indicate that streambank erosion is likely the most significant source of sediment in the watershed. For example, a recent assessment of road derived sediment using the WARSEM model estimated a delivery of 273 metric tons/year from 41.1 km of native surface roads to spawning streams in the Benewah Creek watershed (Duck Creek Associates 2009). By comparison, 6.4 km of incised mainstem habitats may yield up to 4505 metric tons/year from streambank erosion. Given the discrepancy in sediment yield between these two sources, the predicted reduction in sediment yield due to our restoration actions should be reflected in decreased levels of percent fines in our riffle habitats over time.

Another surrogate index of in-stream habitat suitability that displayed a positive response to mainstem restoration was groundwater level. The monitoring of installed piezometers in upper mainstem reaches of the Benewah watershed indicated that groundwater levels were 50% higher in restored than in unrestored habitats. Although there was not sufficient replication to provide a statistical evaluation, these data suggested real benefits from restoration by raising shallow groundwater levels and increasing the water available to native wetland plant species. Continued monitoring of these wells will allow us to track the response of shallow groundwater dynamics to the progressive restoration of contiguous mainstem habitats. Improving groundwater recharge should not only increase the extent of wetland habitats, but the resulting expected increase in connectivity between floodplain and mainchannel habitats should improve rearing temperatures for cutthroat trout which will be able to be monitored using our current configuration of temperature loggers. In addition, the findings supported by data collected from these piezometers informed decisions regarding the new restoration design for reach D2 as outlined in section *4.5 Project B_9.7: Restoration Design for Instream/Channel Construction*.

Though many of the monitored metrics displayed a positive response to restoration in the upper Benewah watershed, we did not detect a response in percent riparian cover in treated mainstem sites relative to that measured in control sites. The lower canopy cover percentages measured at restored sites was predominantly due to the removal of established vegetation during construction activities that modified and re-aligned the channel. In comparison, un-restored sites still had an established canopy adjacent to the creek that consisted of predominantly alder and hawthorn. Given the width of the restored mainstem reaches, longer periods of time will be required before newly planted trees and shrubs become established and attain the height required to provide consistent shade across the channel.

Because the establishment of deep, dense rooted native plant communities is essential to the long-term stability of restored sites, we were especially interested in how hydric and mesic vegetation community patterns evolve as a result of both sudden and gradual changes to reach hydrology. For this reason, more intensive monitoring of vegetation responses was conducted at fixed plots and transects in newly-constructed stream segments over a 3-4 year time span.

Results from this monitoring were intended to inform management regarding the expectations for wetland recovery potential as the project evolves. Survival rates measured at the fixed radius plots were rather high (83%), and based on these results, suggested that the widespread establishment of herbaceous and woody plant species was very promising. Growth rates for newly planted vegetation, however, exhibited variable results and was most likely driven by changes in seasonal water-table depths and soil conditions along elevation gradients and possibly distance from the channel. To illustrate, ground cover data from the greenline transects and fixed plots located within 5 meters of the channel were consistently greater than those for plots further from the channel, though the median values for all data combined generally did not come close to meeting the success criteria (80% ground cover by 20-months) until as late as 44-months post-restoration. Rapid vegetation responses (i.e., growth) are arguably most critical in areas adjacent to the channel as they are subjected to greater sheer forces, and early establishment provides a disproportionate measure of stability to the site. On the other hand, measured canopy cover values, as expected, were generally low along all monitored transects and did not display any significant temporal trends. This variable is expected to have a longer response time (4-6 years) and the measured canopy values largely reflected what was left undisturbed following construction.

The vegetation data reflect the variability of growing conditions within the larger restored area and illustrate some of the challenges in establishing clearly definable success criteria for evaluating wetland recovery. We realize that we would have to measure additional variables (e.g., water table depth, redox potential etc.) to better understand the interaction of water-table and vegetation responses to restoration. Some possible sources of observed variation include:

- Soil compaction, as an artifact of construction, likely affected the observed growth responses, while also affecting rates of infiltration and groundwater recovery. Forty-two percent of sample plots at floodplain transects generally corresponded to areas of incised channel that were filled during construction, whereas there was less disturbance and compaction elsewhere.
- Deposition of sediments across the floodplain during several over bank flows post-construction may have had physiological effects on vegetation growth as well as on germination ability (Dittmar and Neely 1999). Also, deposited sediment at measured plots likely resulted in observer bias and underestimation of ground cover by obscuring rooted vegetation.
- A subset of sample plots from greenline transects as well as floodplain transects were representative of drier site conditions located either on low terraces or further removed from the channel. As expected, growth rates and vegetation responses were slower in these locations.
- Imported channel fill, if lacking a viable seed bank, may have influenced rates of recovery and vegetation response.

3.4.3.3 Habitat monitoring in the Lake Creek watershed

Habitat sampling conducted in tributaries of the upper Lake Creek watershed provided information that both informed the effectiveness of our monitoring program and increased our understanding of deficiencies in these subbasins. First, the sampling that was conducted at sites in the West Fork Lake Creek and Bozard Creek will provide an additional year of pre-restoration data which will permit a more robust before-after control-impact (BACI) analysis of restoration activities that are planned for implementation in the West Fork (see section 4.8 Project 8.2/0.7:

Restoration Design for *Hnmulshench* Project, WF Lake Creek). Second, conditions in these tributaries illustrated the potential for bias to be introduced in some of the monitored habitat metrics. For example, at some sites, reed canary grass accounted for a substantial percentage of the estimated canopy cover. However, the inclusion of this streamside non-native component in our monitoring scheme may mask any detectable trends in percent canopy cover in those reaches that receive riparian plantings. As such, more accurate trend monitoring will be attained by omitting reed canary grass when measuring the canopy cover metric in the field. As another example, the criterion that has been established to identify pools in restored mainstem habitats may not be applicable in small, upper tributary reaches that are not heavily influenced by beaver dams. For example, only one pool deeper than one foot of residual pool depth was identified in site 3 in Bozard Creek. In order to more precisely monitor changes in residual pool depth at sites that receive restorative actions in these subbasins, it may be necessary to redefine pool habitat for small, tributary reaches.

Habitat sampling in Lake Creek tributaries in 2008 also revealed a widespread degraded state for a couple of the metrics that were examined. For example, the values for percent fines that were estimated at many of the sites in lower tributary reaches were exceptionally high (> 50%). Reaches in this part of the Lake Creek watershed are heavily influenced by beaver dams that impede movement of sediment through the system. As another example, a paucity of large woody debris was apparent across all sites except Lake 11, which was addressed in the late 90's with wood additions. A lack of large wood was evident even at Bozard 3, a quasi-reference site which met or approached desired levels for percent canopy cover and percent fines in riffles. Although this site is in a forested area that provides adequate shade, the tree stand consists of mostly non-coniferous species which do not provide for large wood recruitment. Baseline data, such as that recorded for LWD availability, will aid in selecting those enhancement actions that should have the most potential to improve habitat condition in these tributary reaches.

3.4.3.4 Response of cutthroat trout to restoration

Despite the mosaic of thermal refugia and the complex habitat (e.g., deep pools and LWD additions) created in the restored Benewah mainstem, cutthroat trout were rarely captured at sites located in those reaches that have undergone large-scale channel restoration over the past 3 years. The reason for the apparent lack of utilization of newly created rearing habitats may be attributed to one or more of several factors. First, core spawning and rearing areas may have been sufficiently separated from restored mainstem reaches so that extensive dispersal by juvenile cutthroat trout was inhibited. The degree of isolation between restored stream segments and colonizing source populations, either by distance or the presence of physico-chemical barriers (e.g., temperature), has been considered to be an important factor influencing the probability that fish populations will positively respond to restoration measures (Bond and Lake 2003; Pretty et al. 2003). Source populations in many of the upstream tributaries (e.g., Windfall Creek, Schoolhouse Creek, and South and West Forks of Benewah Creek) were approximately 4 km from restored reaches, and fish would have had to traverse extensive warm riffles (e.g., temperatures in excess of 17^{o}C), most notably those along the 2.5 km of restored habitat, to colonize mainstem habitats downriver. Second, densities of cutthroat may have been insufficient in lower reaches of upper mainstem tributaries to induce density-dependent emigration responses. Juvenile fish do not need the territorial space required by larger adults (Grant and Kramer 1990), and at the densities observed in our survey, available capacity in tributaries may have been adequate. Similarly, Johnson et al. (2005) suggested that low rearing densities likely contributed to the lack of colonization by salmonid fry of newly-created habitats in their study.

Shrank and Rahel (2006) also found smaller cutthroat to remain in tributary habitats and to display minimal displacement to downriver reaches during the summer if suitable foraging habitat was readily available nearby. Simultaneous monitoring of tributary and restored reaches over time will allow us to assess relationships between seeding densities, tributary habitat saturation, and expansion to restored reaches.

However, there may be some evidence that juvenile fish are beginning to redistribute downriver from upriver tributary sources. Over the past three years, the numbers of fish were substantially higher at sites located in those Benewah mainstem reaches that are upriver of the restored area (sites 16L, 16, and 17). Given that aggregate tributary densities have been steadily increasing since 2005, this may allude to the operation of density-dependent mechanisms that induce movement out of those tributaries proximate to the upper mainstem reach. Moreover, ambient stream temperatures were much lower in the unrestored mainstem reach than in downriver restored reaches, which may have provided less inhospitable conditions for foraging movements. This finding suggests that close proximity to tributary sources and favorable temperatures for dispersion may improve detection of a positive population response to habitat restoration.

Though a numerical response in mainstem reaches to restoration has not been observed, tributary trends over the past three years may suggest an indirect response to mainstem habitat improvements. Despite the apparent lack of utilization of restored habitat by cutthroat trout during the summer, the deepened mainstem reaches may have provided suitable overwintering habitat that was previously available only in limited amounts. Both juvenile and adult cutthroat trout have been found to prefer deep pools as winter refuge habitat in small stream systems (Jakober et al. 1988; 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 were not realized from summer surveys.

The realignment of ecological processes in Benewah mainstem habitats with those of naturally functioning stream-riparian ecosystems may require a longer timeframe than other instream enhancement projects to detect a positive response by cutthroat trout. Salmonds have exhibited localized, rapid increases in abundance to placement of habitat-forming in-stream structures (e.g. large woody debris, log weirs, and channel deflectors) as noted in our small-scale projects (e.g., lower Whitetail Creek) and in other studies (Roni et al 2002, 2008). However, the large-scale measures that have been implemented in the Benewah mainstem are likely much more intrusive than the formerly reviewed in-stream structural additions that have been found to elicit positive responses. Consequently, more time may be needed for ecological and hydrological properties to adjust to the repeated, acute artificial disturbances that were imposed by our annual channel reconstruction activities.

Further, we are not only amending local deficiencies in habitat complexity (e.g., additions of LWD as in-stream cover), but also addressing impaired processes that operate at larger spatial scales. Because of the scale at which we are rehabilitating degraded habitat, it is recognized that the reestablishment of natural processes will occur gradually, both from a biological and a logistical perspective. For example, as planted vegetation along channel margins and in adjacent floodplain habitats advance toward their desired state, riparian shade should help ameliorate main-stem temperatures. Moreover, additional prospective actions that promote water retention and augment groundwater recharge are targeted for main channel habitats upstream of the restored reach where habitat conditions are still suboptimal (see section *4.5 Project B_9.7: Restoration Design for Instream/Channel Construction*). Notably, our results are not unlike those reported for other large-scale re-meandering projects in which authors speculated that the lack of fish response was due to the persistence of limiting factors in reaches adjacent to those restored (Moerke and Lamberti 2003; Cowx and Van Zyll de Jong 2004). As we progressively address contiguous reaches in the upper Benewah mainstem, we expect to observe an improved thermal regime that is more conducive for cutthroat trout colonization and growth.

3.4.4 Effectiveness monitoring – Nonnative brook trout control

The total number of brook trout, in addition to the percent comprised by adults, removed over the past two years has been considerably lower than that recorded in 2006 when mainstem habitats were first heavily emphasized. However, we were unable to detect an appreciable reduction in densities from our survey data since the initiation of the removal program in 2004. The lack of a measurable reduction was in part explained by the differences in trends observed among tributaries in the upper portion of the Benewah watershed. Whereas numbers of fish declined substantially across sites in the West Fork of Benewah, estimated abundances displayed increasing trends in Schoolhouse and Windfall creeks. Densities at most of the other sites remained relatively low.

The differences in trends observed across tributaries may be attributed to one or more of several factors including probabilities of establishment, changes in colonization patterns, and varying degrees of effort applied in previous removal activities. First, the location of Schoolhouse and Windfall creeks in the upper part of the watershed may in part explain the positive trends observed in both tributaries. The mouths of both creeks are located along the mainstem reach where removal densities have consistently been the highest, thus increasing the probability for mobile individuals to colonize these tributary reaches. Others have noted the importance of both proximity and connectivity to source localities in determining probabilities of brook trout establishment (Benjamin et al. 2007). Brook trout expansion into Windfall Creek, however, was likely inhibited until 2004 when culvert replacement and channel reconstruction virtually eliminated this barrier. Thus, local sub-populations, colonized by the more mobile individuals, may not have yet had the opportunity to become firmly established (Peterson and Fausch 2003). This may explain the lesser increase in density observed in Windfall than in lower Schoolhouse Creek. As a result of the recent re-connectedness of Windfall Creek with the mainstem, this tributary should continue to be monitored in the future to assess rates of brook trout expansion into this newly accessible habitat.

Prevailing differences in habitat attributes among tributary reaches may have also played an explanatory role by yielding dissimilar suitabilities in rearing environments for juvenile brook trout. For example, low densities were sustained in Whitetail Creek despite its close proximity to upper mainstem reaches. However, summer temperatures in Whitetail Creek are typically lower than those recorded in Schoolhouse and Windfall creeks. Temperature has been considered to be a major factor in limiting the competitive advantage of brook trout over cutthroat trout in areas where sympatric populations occur (De Staso III and Rahel 1994; Adams 1999; Dunham et al. 2002).

Alternatively, the differences observed among tributaries may have been due to the focus of removal efforts during the first couple of years. Initially, before it was discovered that many of the larger adults were residing in upper mainstem habitats, efforts were concentrated in tributaries, most notably the South and West Forks. Given that the most marked decrease in abundance was demonstrated in the lower West Fork, the unequal distribution of past sampling efforts may partly explain the results from our survey data.

Although an overall reduction in brook trout in the upper Benewah watershed was not significantly detected, the comparison with a neighboring watershed, Alder Creek, over the same time period did reveal that we were effective in regulating abundance at a low level. Brook trout densities at index sites in the upper Alder Creek watershed were significantly greater in 2008 than before the suppression program in the Benewah watershed began. Abundances at the most populated index sites, where we would expect density-dependent compensatory mechanisms to predominate, even displayed substantial numerical increases. These findings suggest that regional conditions were favorable for growth and survival. Because these two watersheds presumably share common environmental drivers that govern recruitment rates, we should have expected similar responses in Benewah creek. Apparently, our efforts have been successful at suppressing a compensatory numerical response by brook trout and maintaining abundances at a manageable level.

Watershed comparisons also permit insight into whether removal efforts could be curtailed in future years. Even before the control program commenced in 2004, densities of brook trout in Alder Creek were consistently higher than those documented in Benewah Creek. In addition, whereas distributions of cutthroat and brook trout are almost entirely disjunct in Alder Creek suggesting probable displacement by the latter (Dunham et al. 2002), distributions of both species overlap in Benewah Creek. Differences between these two watersheds could be explained by an invasion process that is still in its incipient stage in Benewah, though given the proximity of these watersheds to each other, expansions should have proceeded at similar rates if colonizing migrants arrived from common downriver sources (Peterson and Fausch 2003).

Alternatively, habitat conditions that are more conducive to brook trout establishment may be more prevalent in Alder than in Benewah. For example, 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). In addition, brook trout have been found to exhibit a competitive advantage over cutthroat trout at warmer temperatures (De Staso III and Rahel 1994; Adams 1999; Dunham et al. 2002). Both the availability of preferred micro-habitats and recent thermal regimes may have been more suitable for a successful brook trout invasion in Alder than in the Benewah watershed. Further, the productive adfluvial life-history strategy that is prevalent in the Benewah watershed may confer an advantage to cutthroat trout that permits a greater biotic resistance to invasion (Griffith 1988).

Whatever the reason, the upper Benewah watershed appears less vulnerable to invasion and establishment by brook trout than the neighboring Alder Creek watershed. 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; Benjamin et al. 2007). In particular, systems that have been degraded from their natural condition (e.g., loss of riparian vegetation) may be more vulnerable to invasion by brook trout than those that have been relatively undisturbed (Shepard 2004). Implementing an abridged brook trout removal program in the Benewah watershed in conjunction with improving rearing conditions for cutthroat trout through our restoration actions should provide a better opportunity for cutthroat trout recovery and persistence in the presence of low brook trout abundances.

Results from the last five years were used to inform measures that can be implemented by our suppression program to reduce the effort applied annually but still effectively control brook trout. For example, our removal data have shown that brook trout, especially the larger mature adults, were found more often in mainstem habitats with proportionately greater pool area than in tributaries, a finding comparable to other studies that have examined brook trout habitat preferences (Chisholm et al. 1987; Cunjak 1996; Lindstrom and Hubert 2004; Benjamin et al 2007). Given that brook trout were captured during periods prior to spawning, these deep pools may be serving as holding habitats for adults before upriver movement to spawning grounds. Furthermore, our habitat data indicate that suitable spawning substrate for brook trout is seemingly much more prevalent in mainstem reaches upriver of 12-mile bridge than in those reaches downriver that are of lower gradient and dominated by beaver dam pools. A cursory continuous survey of much of the upper mainstem habitat in the fall of 2008 corroborated this supposition. Not only were large brook trout observed congregated over riffles much more often in reaches upriver than downriver of 12-mile bridge, but brook trout redds were also positively identified upriver of 12-mile bridge. Apparently, this reach may serve as a primary spawning location for brook trout.

Drawing on these results and the finding that high densities of adults are consistently found in the 2.0 km reach upstream of 12-mile bridge, we intend to reduce removal efforts in 2009 by concentrating sampling only along the mainstem reach that extends from 12-mile bridge to the confluence of South and West Forks. Foregoing shocking efforts in the tributaries, where the ratio of cutthroat to brook trout is relatively high, will minimize undesired impacts to cutthroat trout. In addition, we will set up a temporary trap above 12-mile bridge, consisting of two fixed weirs that enclose approximately 50 ft of pool habitat, to intercept spawning adults. The downriver fixed weir will have a narrow opening that will allow upriver migrants to enter, but the upriver fixed weir will span the entire channel to obstruct further upstream movement. Periodically, this 50 ft long reach will be shocked and the captured brook trout removed. This alternative approach is expected to minimize crew effort because of the ease with which the small pool can be shocked compared with the inordinate amount of time that has been annually allocated to shocking the deeper, pool habitats from 9-mile bridge to 12-mile bridge. In addition, similar to this year, we intend to keep the RBW trap deployed at 9-mile bridge from spring until fall to prevent brook trout from ascending into the upper watershed. Given that two large adults were captured in the RBW in early summer of 2008, and that the size distribution of brook trout captured by electrofishing in reaches immediately below the trap was greater than that upriver of the trap, the prevention of large adults from reaching spawning grounds in the upper mainstem should aid in suppressing production. Over time, if these methods prove successful, than we may be able to reduce the frequency under 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 compensatory resilience of brook trout in the Benewah watershed (Meyer et al. 2006).

Our suppression program also entailed monitoring changes in maturation metrics in brook trout to detect potential compensatory reproductive responses to our removal efforts owing to a release from conspecific competition. Specifically, we were interested in whether residual brook trout expressed changes in the average size at maturation or in the fecundity-at-length relationship. Thus far, our removal program apparently has not induced compensatory responses in the brook trout population. Female brook trout sampled from the Benewah watershed were not more likely to mature at a given length in 2008 than in 2004. Though Benewah males were predicted to mature at a smaller length in 2004 than in 2008, this result could have been an artifact of the predominant sampling of tributaries in 2004 and of mainstem habitats in 2008. If mechanisms that govern growth rates and movements differ between tributary and mainstem habitats, then average lengths at maturation may differ between males sampled in these two habitats. Specifically for brook trout, low growth rates have been associated with earlier age at maturation regardless of size (Hutchings 2004). Thus, processes that limit growth rates in upper Benewah tributaries could have given rise to the differences detected in maturation probabilities for male brook trout. The finding that brook trout sampled in tributary reaches of the Benewah watershed in 2008 were more likely to mature at smaller sizes than those sampled in mainstem reaches bears this out. In further support, males were predicted to mature at a smaller size in Alder than in Benewah Creek in 2008, but no difference in maturation probabilities was detected in 2004. Though we did detect an increase in fecundity in 2008 compared to previous years for brook trout from the upper Benewah watershed, fecundities for Alder Creek females displayed a similar relationship over time, suggesting similar mechanisms may have been operating in both watersheds. Though continued monitoring would better inform the potential for long-term compensatory responses, it appears that the maintenance of low brook trout densities in the upper Benewah watershed through periodic removals should not increase individual reproductive investment (e.g., increased fecundity at a given length) nor induce an earlier maturation schedule (inferred from length at maturation) that would shorten generation times. More importantly, our results illustrate the advantage of using a control watershed when evaluating the effectiveness or potential undesired impacts of a non-native removal program.

4.0 RESTORATION AND ENHANCEMENT ACTIVITIES

Implementation of restoration and enhancement activities occurred is several watersheds, including Benewah, Lake, and Evans creeks during 2008. Significant effort was also directed toward preparation of restoration designs for large projects in both the Benewah and Lake creek watersheds. All activities completed during the contract period June 1, 2008 through May 31, 2009 are summarized in Table 27 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_5.2/0.2 is located in the Lake Creek watershed 0.2 miles up on a tributary that has its confluence with the mainstem 5.2 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 27. Summary of restoration/enhancement activities and associated metrics completed for BPA Project #199004400.

Project Description			Project Chronology				
Project ID	**Activity**	**Treatments (Metrics)**	**Pre-2005**	**2005**	**2006**	**2007**	**2008**
B_6.5 (p. 97)	O&M for Instream Habitat	Channel construction (695 m); Riparian planting (4.3 ha, 1390 m of streambank); Riparian fencing	Permitting and construction of original project completed in 2001				Completed design and NEPA; completed O&M work to stabilize 175 m of stream bank and floodplain
B_8.9 (p. 101)	Stream Channel Construction	Constructed 2524 m of channel (Increased channel length by 506 m)	Completed baseline HEP; channel assessment and development of restoration prescriptions (2002)	Channel design finalized; NEPA completed; constructed lower 518 m of channel on the property	Constructed 594 m of channel	Constructed 762 m of channel	Constructed 650 m of channel
B_8.9 (p. 104)	Plant Vegetation	Streambank stabilization (8.14 ha, 5200 m of streambank		Planted 15,850 herbaceous plugs, 4,100 deciduous trees (1.82 ha of floodplain, 1036 m of stream bank)	Planted 26,387 herbaceous plugs and 7,450 deciduous trees (2.32 hectares of floodplain, 1340 m of streambank)	Planted 18,471 herbaceous plugs and 6,369 deciduous trees (2.2 hectares of floodplain, 1524 m of streambank)	Planted 18,470 herbaceous plugs and 9,745 deciduous trees (1.8 hectares of floodplain, 1300 m of streambank)
B_10.4 (p. 106)	Plant Vegetation	Riparian enhancement (48.16 ha; 3,689 m of streambank)	Planted 31,068 conifers and 5,663 deciduous trees (41.4 ha of floodplain, 1,810 m of streambank)	Planted 8,000 conifers (4.9 ha of floodplain, 1,879 meters of stream bank)	Planted 10,000 conifers (restocked 10.6 ha)		Planted 2,100 conifers (1.86 ha of floodplain)

Project Description			Project Chronology				
Project ID	**Activity**	**Treatments (Metrics)**	**Pre-2005**	**2005**	**2006**	**2007**	**2008**
B_9.7 (p. 108)	Stream Channel Construction						Developed restoration design for 2.4 km of mainstem habitats (Reach D-2)
E_0.1 (p. 113)	Plant Vegetation	Riparian enhancement (2.4 ha; 396 m of streambank)	Developed landowner contract and cost-share agreement (1999); planted 4,200 trees and shrubs (1999-2001)				Planted 1,550 conifers and 1,100 deciduous trees/shrubs (2.4 hectares of floodplain, 396 m of streambank)
L_8.5 (p. 115)	Plant Vegetation	Riparian enhancement (1.8 ha; 480 m of streambank)	Signed landowner agreement (1998); site prepped and planted 1150 trees and shrubs (1998-99)				Planted 750 conifers and 1300 deciduous trees/shubs (1.1 hectares of floodplain, 450 m of streambank)
L_8.2/0.7 (p. 117)	Stream Channel Construction						Developed restoration design for 1.2 km of tributary habitats in West Fork Lake Creek.

4.1 Project B_6.5: O&M for Instream Habitat

Project Location:

Watershed: Benewah	Legal: T45N, R3W, S4, SW ¼
Sub Basin (River Mile): RM 6.5-6.9	Lat: 47.271856N Long: 116.728735W

Site Characteristics:

Slope/Valley gradient: 1%	Aspect: NE	Elevations: 804 m
Valley/Channel type: B2/C3	Proximity to water: In channel	

Other: *Project treats 175 meters of stream channel and associated floodplain within an existing project site that encompasses 695 meters*

Problem Description: A splash dam and flume were constructed on this site between the years 1915 and 1920 to convey logs through the Benewah valley downstream to Benewah Lake and the St. Joe River. Following the dismantling of the splash dam, sometime in the 1930's, the creek was straightened and the natural floodplains cleared and drained to develop cropland and pastures adjacent to the creek. Straightening increased the channel gradient, which in turn, increased the channel's ability to convey bed material and subsequently caused the channel to degrade. This deeper, incised channel was vertically separated from its floodplain and unable to sub-irrigate the riparian vegetation it once depended upon for stability. Recent grazing pressure intensified the bank stability problem by reducing plant density and diversity. Streambanks were extremely unstable and instream habitats have little value as summer rearing for cutthroat trout. Results of the pretreatment channel survey help illustrate these problems: sinuosity = 1.06; flood prone width at twice maximum bankfull depth (d_{maxbf}) = 24.4 m; Entrenchment ratio = 1.92; bankfull width = 12.7 m; mean bankfull depth(dbf) = 0.54 m, and; $width_{bf}/depth_{bf}$ ratio = 23.2. Most of these values fall well outside the range of median values more typical of undisturbed reference reaches.

Previous work on this site involved implementation of a stream channel design which converted the existing degraded channel from an F4 to C4 stream type by increasing the meander width ratio and sinuosity, lowering the bankfull width/depth ratio, and reducing the channel entrenchment ratio. A new meandering channel, which added nearly 152 m of channel length, was constructed and portions of the existing unstable channel and floodplain was filled and regraded. The new channel was built just large enough to convey the bankfull discharge within its banks. The controlling riffle elevations were set at a consistent gradient and the bank heights at all the riffles and bends were built so that the banks would overtop simultaneously during flood events. During construction, ten riffles, 4 j-hook structures, and more than 40 pieces of large wood were placed to enhance streambank stability and instream habitat diversity. Additional implementation work conducted from 2000-2001 included riparian planting and construction of an exclusion fence to help manage riparian grazing. The original project treated 695 linear meters of stream channel and 4.3 hectares of associated floodplain.

Since construction of the initial design was completed in 2001, erosion of floodplain surfaces at one meander and bank erosion along 61 m of streambank have compromised channel stability at the lower end of the site and reduced the habitat potential for native fishes. These ongoing issues necessitated additional design work to identify appropriate treatments to ensure that the original design objectives were met and to protect the investments in the project. The existing conditions hydraulic model indicated that the channel and floodplain between stations 8+00 to 8+41 was narrower than upstream and downstream sections. This narrow reach appears to have caused elevated flood water upstream of the constriction to seek a floodplain flow path to the relatively wide and shallow section downstream of the constriction. When this type of situation occurs naturally in streams – by a fallen tree for instance – a localized increase in stream power occurs, causing the channel to deform its boundaries until the channel capacity is restored. This dynamic process may continue through several floods until the stream power becomes balanced with the erosion resistance provided by the stream boundaries. In this narrow section of Benewah Creek, the floodplain has been eroded by recent flooding as the stream began to naturally adjust to increase its capacity. Although significant floodplain erosion has occurred, the hydraulic model indicates that a constriction still exists, so it is expected that additional floodplain erosion will occur, possibly leading to a channel cut-off.

<u>Description of Treatment:</u> The proposed stream channel, banks, and floodplain remedies were composed of logs, imported riffle stone mix, and native gravels. The design included building and reconfiguring existing riffles and pools, and installing logs. Hydraulic modeling indicated that the final design does not significantly decrease erosive stream power, but rather the design proposes to create channel and floodplain surfaces that will better resist erosion when subjected to that stream power. Site 1 included roughly the upper one-half of the project reach. The site exhibited mild bank erosion and severe floodplain erosion (Figure 28). The remedies for Site 1 included constructing logjams adjacent to the channel to resist lateral channel migration, and increasing erosion resistance on the floodplain surfaces by installing logs and gravel armor below grade. Site 2 is the downstream portion of the project reach. The Site 2 treatment was a large logjam to stop the progression of lateral channel migration.

Approximately 47 m^3 of large logs were be used to build streambanks and create floodplain surfaces that will not readily deform. Emulating natural logjams, the logs were constructed into complex matrixes configured to resist bank erosion while creating fish habitat (Figure 29). Since existing floodplain surfaces remain vulnerable to continued degradation during overbank flooding, the at-risk areas at Site 1 were treated with logs installed to create temporary floodplain roughness. Over time the floodplain logs will decay and be replaced by vegetation that will take over the role of resisting erosion. Logs were secured by burial, bracing to existing trees and other logs and vertical snags, and by cable and earth anchor. Native streambed gravels were collected from pool excavations and from gravel bars. These materials were used to fill void space within constructed logjams and to provide an armor layer on the constructed floodplain surfaces. Native stream gravels were also used to augment materials used for riffle construction, and to create fish habitat in pool tail-outs. Riffles within the designed reach were reconfigured and/or augmented with approximately 15 m^3 of imported stone. The design riffle stone mix was composed of rocks large enough to prevent significant riffle deformation caused by 25-yr and lesser magnitude floods, yet small enough to have minimal pore space, which can be filled with native gravel, sand, and silt to prevent subsurface flow. In areas where excavation occurred within topsoil, the topsoil and sod was salvaged and stockpiled for use in finish grading the constructed logjams and floodplain surfaces. Disturbed areas were planted with a mix of seven species of woody trees and shrubs and 10 species of herbaceous sedges (*Carex sp. and Scirpus sp.*) and rushes (*Juncus sp.*), and then hydroseeded at an application rate of 45 kg/hectare.

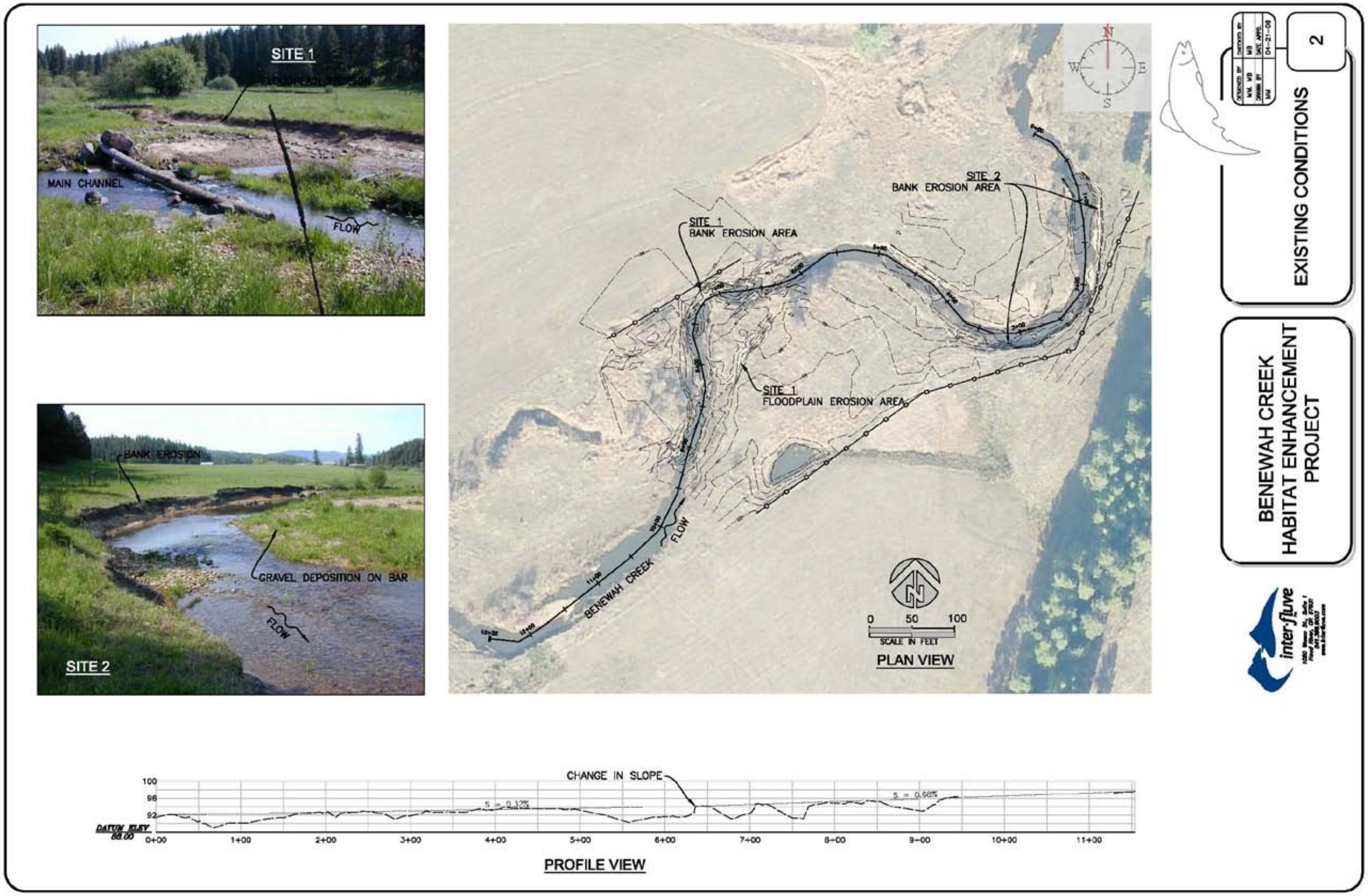

Figure 28. Photos and plan view of existing impaired conditions at project site B_6.5 prior to treatment.

Figure 29. Photo log showing the construction sequence of the streambank log jam at Site 2. Logs were placed in a complex matrix adjacent to the eroded terrace to create a low bench at the bankfull elevation (A). Vertical snags and horizontal logs were buried in the terrace to anchor and stabilize the structure (B). Native stream gravels excavated from pools were used to fill the void spaces in the constructed logjam (C). Salvaged topsoil, sod and woody plants were used to cover the final structure (D).

<u>Project Timeline:</u> Project implementation required a site inspection by a certified archaeologist and subsequent clearance by the Tribal Cultural Officer and the SHPO, as well as a wetland delineation and USACOE 404 permit. NEPA clearances were received by August 2000. Phase I channel reconstruction was completed October 2000 and the remaining Phase II channel work was finished by July 2001. Plantings were completed in fall 2000 and also in 2001. Riparian fencing was completed September 2001. The maintenance work conducted in 2008 was completed under USACOE permit NWW No. 001201070, issued August 13, 2008.

<u>Project Goals & Objectives:</u> Restore the channel and floodplain to a naturally appearing and functioning geometry consistent with a C4 channel type using native materials. Create a stable creek and riparian environment that will naturally develop into optimal fish habitat. Restore proper bedload balance within the reach and minimize the flood potential for adjacent cropland.

<u>Relationship to Scope of Work:</u> This project fulfills the Program commitments for WE H in the 2008 Scope of Work and Budget Request (Contract #37842) for the contract period June 1, 2008 through May 31, 2009.

4.2 Project B_8.9: Instream/Channel Construction

Project Location:

Watershed: Benewah	Legal: T45N, R3W, S18 NE ¼ NE ¼
Sub Basin (River Mile): RM 8.9	Lat: 47.249851N Long: 116.762181W

Site Characteristics:

Slope/Valley gradient: <1%	Aspect: N	Elevations: 822 m
Valley/Channel type: B2/C4	Proximity to water: In channel	

Other: *Project restores channel planform, grade and profile to what is believed to be within the range of historic conditions for 650 meters of stream.*

Problem Description: The Benewah valley between river miles 8.9 and 11.9 can be broken into three general reaches that relate to the level of sinuosity and the degree of channel incision that has taken place. The lower 2.3 km and upper 0.8 km have experienced more avulsions and channel straightening than the middle 2.1 km. The valley slope is 0.007 throughout, however sinuosity in the lower and upper reaches is 1.38 and 1.3, respectively, compared to 1.8 in the middle reach. Downstream avulsions and head cutting have moved upstream through the lower reach where this project is located, causing it to be become incised and substantially reducing the access to its old floodplain. Hydraulic analysis of representative channel cross-sections show the overall level of incision is approximately equivalent to the capacity of a 5-year return interval peak flow event with some areas exhibiting incision that approaches the 10-year peak flow.

The incised channel is characterized by unstable stream banks with accelerated erosion rates and increased sediment yield to the channel. The most recent estimates of stream bank erosion were made using the BANCS model (Rosgen 2006), which combines quantitative measures of stream bank characteristics with derived values of near-bank sheer stress to generate estimates of average annual erosion rates. In measured reaches erosion rates were estimated at 0.16±0.07 tons/yr/ft with an estimated sediment yield of 156.1 tons/yr. When these results are extrapolated to the larger reach located between river miles 8.9 and 11.9, total annual sediment yield from streambanks is estimated at 1,689.6±739.2 tons/yr. 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 groundwater tables have been lowered along with the streambed, as many of the wetland areas are only marginal in size and a band of xeric vegetation of variable width is located along the channel margin throughout the incised reach. Based on analysis of observational data, including current vegetation patterning, wetland delineations, and historic soils data from 1904, it is estimated that lowering of the water table related to channel incision has reduced wetlands habitats by up to 40% compared with historic conditions. 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, reduced infiltration of water from adjacent wetlands, and elevated water temperatures are all factors that limit the productivity of these reaches.

Description of Treatment: The initial work to develop a restoration design began with development of the relationship between the runoff characteristics of the watershed and stable hydraulic geometry for the stream channel. Subsequently, the HEC-RAS hydraulic model was used to estimate hydraulic conditions and simulate water surface elevations, flow regimes, velocities and shear stress for the design channel. A substrate specification was developed to withstand some vertical movement during the 10-year return interval discharge but not over sized to the point of complete immobility. Implementation of the restoration design involves filling the stream channel to historical elevations and utilizing historical alignments where possible. The designed planform creates channel grade and profiles within the range of historical channel conditions, based on topographic and field analysis. Historical conditions will be met by lifting the incised channel by filling the channel with imported rock at intervals along its length that correspond to areas that would naturally be riffles. Pools between these riffles will remain unnaturally deep until existing basin sediment loads slowly fill them. In areas that have laterally expanded following entrenchment, new banks and floodplain will be created. 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.

A total of 650 m of channel was constructed in 2008, increasing the extent of restoration at the site to 2,523 m of channel since 2005. Twelve riffles were constructed using a total of 1653 cubic meters of imported gravel. A little more than 83 m of the existing incised channel was "plugged" with approximately 1609 cubic meters of imported fill to create new floodplain habitats. Channel length was increased by 60 m (10%) through the constructed reach. A total of 141 cubic meters of large wood, the equivalent of 15 truckloads of 10 m long logs, was placed in the channel and on the floodplain to provide cover, increase habitat complexity, and increase roughness and stability. Within the channel, four wood structures were constructed to form stable bankfull benches adjacent to actively eroding terraces located on the outer edges of meander bends. Additional inchannel wood was placed in random configurations to allow for sorting and redistribution by high flows. Much of the wood placed outside the bankfull channel was buried below grade and no anchors or cable were used as in past years. Completion of construction activities this season marks the end of Phase I for this project.

<u>Project Timeline</u>: A 30% stream channel design, appropriate for fit in the field construction, was completed for the lower 2,621 m of channel in January 2005. A wetland delineation and function assessment were completed for the same area in May 2005. All NEPA analysis and permitting requirements, including CWA certification, 404 and 401 authorizations, NPDES permits and the supplemental analysis for the BPA Watershed Management Program EIS, were completed for the project in 2005. Clean Water Act permits were reauthorized in 2007 for the continuation of channel construction through the dates of planned completion in 2008. Design work for phase II of the project, covering the 3,050 m upstream of completed construction was initiated in 2008 and is described below (See *4.5 Project B_9.7: Restoration Design for Instream/Channel Construction*).

<u>Project Goals & Objectives</u>: Implement 2,621 m of stream channel construction as part of a larger project to restore historic wetland habitats and hydraulic connections with the valley bottom for 5.1 km of stream over a 10-year timeframe. Restore stable channel configurations to treatment areas and increase the frequency and duration of over bank flooding equal to the 1.5-year return interval. Increase coldwater refuge by improving dynamic and long-term surface and

ground water storage. Provide for a measurable increase in abundance and distribution of westslope cutthroat trout in treatment areas.

Relationship to Scope of Work: This project fulfills the Program commitments for WE G in the 2008 Scope of Work and Budget Request (Contract #37842) for the contract period June 1, 2008 through May 31, 2009.

4.3 Project B_8.9: Riparian/Planting

Project Location:

Watershed: Benewah	Legal: T45N, R3W, S18 NE ¼ NE ¼
Sub Basin (River Mile): RM 8.9	Lat: 47.249851N Long: 116.762181W

Site Characteristics:

Slope/gradient: <1%	Aspect: N	Elevations: 822 m
Valley/Channel type: B2/C4	Proximity to water: Floodplain	

Other: *Project specifically treats the 1,300 meters of streambanks and 1.8 hectares of associated floodplain disturbed during stream channel construction in 2008 (see project description above).*

Problem Description: Restoration of the mainstem Benewah Creek is underway to restore stable channel pattern and geometry at the presumed historic elevation of the channel in the valley bottom. Approximately 2,524 m of channel have been constructed since 2005. Implementation of the design will result in 7.2 ha of direct disturbance from construction, development of temporary access, and site dewatering during construction. These areas 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 much of the project area as a result of the processes of channel incision and channel enlargement that have occurred over a period of approximately 80 years. 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. Although the geomorphic location of these wetlands is clearly riverine floodplain, the dominant water source in some areas has probably transitioned over time to seasonally perched groundwater and/or direct precipitation owing to the disconnection between the creek and its current floodplain. A band of xeric vegetation of variable width is located along the channel margin throughout the incised reach. Based on analysis of observational data, including current vegetation patterning, wetland delineations, and historic soils data from 1904, it is estimated that lowering of the water table related to channel incision has reduced wetlands habitats by up to 40% compared with historic conditions.

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 at and at a control site in the watershed. The plan is documented in the Benewah Creek Restoration Design (InterFluve, Inc. 2005) 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 18,470 herbaceous plugs and 6,369 woody trees and shrubs were planted in fall 2008 along 1,300 meters of streambanks and 1.8 hectares of associated floodplain that was disturbed or created during construction. In addition, all floodplain surfaces, access roads and the bypass trench used in dewatering the construction site, were hand seeded and mulched with herbaceous grasses applied at a rate of 48 kg/ha. In the spring of 2009, 3,376 live willow poles were planted to complete the vegetation treatments on the site. In the time since stream channel restoration was initiated in 2005, a total of 79,178 herbaceous plants and approximately 35,000 woody trees and shrubs have been planted, treating 8.14 ha of floodplain and 5,200 m of streambank.

Project Timeline: Annual plantings will be completed in the fall and the spring immediately following stream channel construction. 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) increase stream shading; 2) provide a long-term source of large woody debris for natural recruitment; 3) promote streambank and floodplain stabilization; 4) increase riparian species diversity and cover; and 5) enhance stream buffer capacity. 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 I in the 2008 Scope of Work and Budget Request (Contract #37842) for the contract period June 1, 2008 through May 31, 2009.

4.4 Project B_10.4: Riparian/Planting

Project Location:

Watershed: Benewah	Legal: T45N, R4W, S13, SE¼
Sub Basin (River Mile): RM 10.4	Lat: 47.239439N Long: 116.773531W

Site Characteristics:

Slope/gradient: <1%	Aspect: N	Elevations: 834 m
Valley/Channel type: B2/C4	Proximity to water: Floodplain	
Other: *Project treats 1.86 hectares of floodplain.*		

Problem Description: The Benewah valley has a history of anthropogenic disturbance by logging and agricultural activities that date to the early twentieth century. Logging removed many of the coniferous trees in the valley bottom between 1915-1930. Splash dams and flumes were developed in the creek to facilitate the movement of harvested logs to down valley mill sites. The combination of direct land clearing adjacent to the creek and the construction and operation of splash dams had a direct affect on channel form and function with negative implications for the productivity of habitats for juvenile rearing. In the most recent past, dating from approximately the 1940's through 2000, the property was managed for grazing and/or hay production, which has precluded the regeneration and establishment of a diverse native riparian plant community along much of the 3.2 miles of streams associated with this property.

Current riparian function is degraded as evidenced by low stream canopy closure, little overhanging vegetation, and low volumes of instream large woody debris. The wood that is present in the channel is mostly comprised of small pieces that generally do not function to shape channel morphology or maintain habitat diversity. Also, the existing riparian community offers little potential for providing recruitment of large wood in the future. Currently, discharges greater than the 5-year return interval flood begin to exit the existing channel in a non-uniform manner. As a result several avulsion channels have developed in portions of the floodplain as a direct result of low roughness and lack of root mass in floodplain soils. Active avulsions have the potential to cut-off remaining channel length and lead to abandonment of relatively high quality habitat.

This stream reach is located in a portion of the watershed that historically provided important summer rearing habitat for westslope cutthroat. Mainstem reaches of the property were likely utilized as over-winter habitat as well.

Description of Treatment: Riparian plantings have been undertaken to re-establish forest plant communities adjacent to the stream channel and provide long-term roughness across the valley bottom. Restoring a forested valley bottom will improve structural habitat conditions in the coming decades and is fundamental to the long-term restoration and enhancement of this site. A total of 2,100 coniferous seedlings were installed in 2009, treating an area of approximately 1.86 hectares of floodplain. Plantings consisted of Engelmann spruce (*Picea engelmannii*), lodgepole pine (*Pinus contorta*) and ponderosa pine (*P. ponderosa*).

Project Timeline: Conceptual restoration prescriptions were developed for this project site following completion of a detailed stream channel assessment in October 2002. The prescriptions were outlined in a report entitled, Benewah Creek Assessment and Restoration Prescriptions (Inter-Fluve, Inc. 2002) and reforestation of floodplain habitats has been occurring since 2002. Addition, design work for 10,000 ft of the Benewah was completed and outlined in a report entitled "Benewah Creek reach D2 restoration project basis of conceptual design report". (DeVries, P. and K. Fetherston, 2008). This plan outlines additional prescriptions related to reforestation to be completed in 2009-2011.

Project Goals & Objectives: Goals for this project include 1) increase stream shading; 2) provide a long-term source of large woody debris for natural recruitment; 3) promote streambank and floodplain stabilization; 4) increase riparian species diversity and cover; and 5) enhance stream buffer capacity. Establish mixed coniferous/deciduous forest vegetation types on floodplain surfaces at a minimum stocking density of 197 trees/hectare and provide for significant increases in canopy density and overhanging vegetation over the next 20 years.

Relationship to Scope of Work: This project fulfills the Program commitments for WE J in the 2008 Scope of Work and Budget Request (Contract #37842) for the contract period June 1, 2008 through May 31, 2009.

4.5 Project B_9.7: Restoration Design for Instream/Channel Construction

Project Location:

Watershed: Benewah	Legal: T45N, R4W, S13 NE ¼ SE ¼
Sub Basin (River Mile): RM 9.7	Lat: 47.241292N Long: 116.771454W

Site Characteristics:

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

Other: *Develop a restoration design to treat up to 2735 meters of stream and associated floodplain habitats.*

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. 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 0.16±0.07 tons/yr/ft. When these results are extrapolated to the larger reach located between river miles 8.9 and 11.9, total annual sediment yield from streambanks is estimated at 1,689.6±739.2 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 Design: The entrenched nature of the mainstem Benewah Creek channel was inferred during early stages of planning to have been the result of land use practices including logging, clearing, and grazing activities, leading to incision from historic dimensions and characterized by a shallower bankfull depth and increased channel instability. Such channel changes were linked to degradation of stream habitat and reduced connectivity between the channel and floodplain. The determination was based in large part on the observation that floodplain inundation was restricted to flows higher than about the 5-year return interval event, whereas a more typical alluvial channel would have been expected to overflow onto the floodplain at around a 1.5- to 2-year event level. This determination formed the basis for restoration designs for the lower 2.5 km ("D1" reach), in which the base level control provided by the stream bottom was raised approximately 0.9 m by importing approximately 8410 cubic meters of gravel and constructing large, raised riffles (See *4.2 Project B_8.9: Instream/Channel Construction*).. The channel was relocated at multiple locations to reoccupy abandoned meander scrolls, the old channel filled, and large wood placed on the floodplain to help resist channel migration while native plant communities were reestablished. Construction through this reach was completed in 2008.

Given the intensive physical manipulation and channel hardening involved in the D1 reach, we set out to conceive an alternative design concept for the upper 1.7 miles ("D2" reach) that involved less extensive channel modification and that would work with and reflect natural channel forming processes, hydrology and geomorphic conditions. The key concept proposed was to construct a new channel with bankfull width, cross-sectional area, and meandering planform that reflect prevailing hydrology, slope, riparian potential, floodplain stratigraphy, upstream sediment supply, and in-channel sediment transport characteristics, rather than retrofit the existing channel and floodplain by increasing grade control elevations and surface placement of wood, respectively.

During field work conducted in 2008 to develop the design, twelve test pits were dug in the valley floor to ascertain the depth to historic alluvial gravel deposits that should be present. The pits were dug at four different locations spanning the D2 reach, with a central pit dug in the thalweg of a relict channel at each location and two pits dug on either side of the channel in the floodplain proper. The test pit excavation results did not support the original project design premise however, forcing us to reconsider the design concept. Analysis of the floodplain stratigraphy indicated the absence of alluvial gravel deposits at elevations that would have been consistent with a geologically recent incision hypothesis. Instead, the test pit excavations consistently showed a cobble-gravel layer at elevations comparable to that found in the current channel thalweg at cross-valley riffle locations (Figure 30). The cobble-gravel layer was overlain by a silty clay loam strata of 15-25 cm in seven of the ten test pits. The upper 1.2-1.8 m of each soil profile above the cobble-gravel or silt clay loam layer was a relatively uniform silt loam. Given the location at the eastern edge of the Palouse region of eastern Washington and northwestern Idaho, the silt loam soil appears to have been formed primarily through aeolian deposited loess and episodic volcanic ash. No geologically recent fluvial deposits (silty sand, gravels) were found in any of the test pits above the gravel silty clay loam that were indicative of fluvial deposition. In addition, the soil had a relatively low permeability, such that precipitation and overbank flow would be slow to filter down to the groundwater table, and there were numerous small, isolated wetland depressions distributed across the valley floor. Thus, historic forest vegetation was maintained by periodic overbank flows, with densest vegetation occurring in swales and isolated wetland depressions.

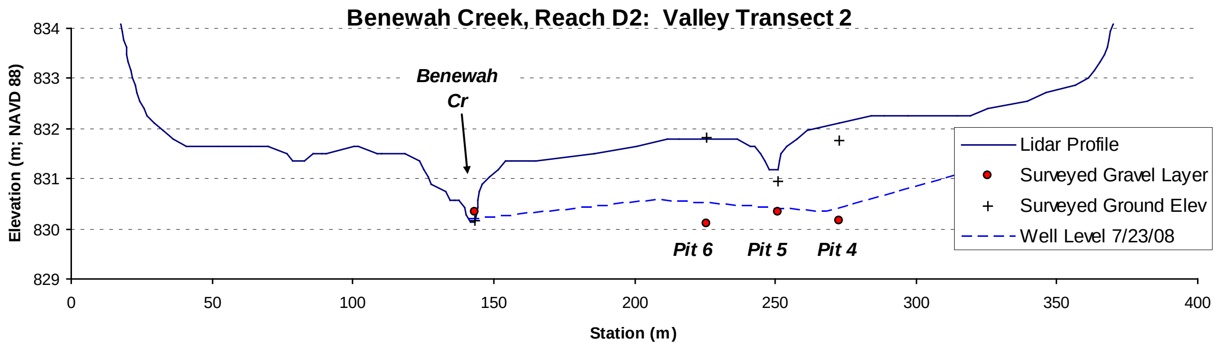

Figure 30. Representative example of cross valley topographic profiles and elevations of the gravel sub-layer found in test pit excavations. Each profile crosses the Benewah Creek valley at the test pit locations. Flow is into the page. Approximate groundwater elevations are indicated for July 23, 2008.

The collective evidence from the test pits, accompanying elevation survey data and available geologic maps indicated to us that the valley floor is not a typical broad alluvial floodplain. In its early stages (pre-Holocene), the channel likely was alluvial in nature and migrated over the valley floor more freely, as indicated by the presence of gravel at comparable levels in all test pits. However, as geologic time progressed, the floodplain rose through aerial deposition and a riparian forest existed that stabilized the stream banks, resulting in a width:depth (channel aspect) ratio likely narrower than would be expected from a more classic alluvial channel. Charcoal fragments and dead roots found in all test pits indicated woody vegetation occurred historically throughout the valley bottom, with fire as a primary natural forest disturbance type in the basin. Following removal of the valley forest, and 70 years of cattle grazing, the stable stream banks, created by root cohesion provided by the riparian forest, have unraveled as a result of widespread bank failure throughout the course of Benewah Creek. The persistence of the present channel and relict channels in aerial photographs dating back to 1933 indicates that channel migration occurred slowly in the form of meandering. The topographic and test pit results collectively suggest that overbank flows do not appear to have resulted in significant filling of relict channels. That, plus topographic evidence that relict channels were limited in extent across the valley floor, provided evidence that the primary mode of channel migration was through episodic avulsion.

We also noted the presence of low profile beaver dams in less heavily managed sections of the project reach. The dams were mostly constructed at riffle crest locations, with a deeper pool upstream. It became clear that overbank flooding was most likely to have been effected historically because of beaver dams, and we thus modified our design accordingly. Our observations of flooding processes indicated that the relatively low height of the dams was likely a result of periodic shearing by ice sheets that typically break up when the stream floods during snowmelt runoff. In addition, rapid filling overnight of the test pits and a cooler temperature in the reach than downstream indicated a strong influence of groundwater on water quality in the D2 reach. We accordingly developed a modular log crib dam design that could be constructed at any location. The dam was designed to emulate the backwater effect of beaver dams at high flow by constricting (or, choking) the flow. The dam was designed to also maintain a low flow water profile comparable to existing beaver dams that would not decrease the head difference between groundwater and stream channel during the critical low flow summer months and provide summer habitat conditions that most likely replicate pre-settlement conditions.

Several new design elements for the D2 reach based on these findings will be implemented over the next 3 years, including:

- Creation of 1,500 ft. of new channel near the downstream end of the reach following the course of an existing swale, with narrow aspect ratio comparable to that observed in more heavily vegetated segments where the banks do not appear to have been significantly eroded. The existing, bypassed channel segment will be filled and vegetated at the upstream end as a high flow swale, and left as a channel backwater connected at the downstream end. Construction will result in 646 ft of added channel length and will lead to a locally reduced stream gradient, from 0.45% to 0.24%.
- Re-grading of an existing swale to create a high flow swale with native vegetation. The wetland swale will be used as a nursery area for propagation of black cottonwood and willow whips and live stakes for riparian zone restoration throughout the Benewah Creek valley.
- Re-grading and partial excavation of 1,452 ft. of relict channel to create a flowing side channel, while preserving existing grade controls in the main channel below Windfall Creek that were constructed to maintain fish passage through culvert under Benewah Creek Road.
- Construction of wood in-channel structures emulating flow obstruction effects of natural wood jams and beaver dams and placement of large wood throughout the stream corridor to aid beavers in dam construction. This project element allows for more frequent and extensive floodplain connection during annual floods, and is a natural analog alternative to large scale riffle construction that maintains connectivity with cooler groundwater during summer months.
- Reestablish a patchwork of native vegetation communities on the valley floor to lay the foundation for a compositionally and structurally diverse forest ecosystem to develop over the next 25-50 years. Plantings will consist of more than 148,000 plants representing 26 species planted over 25 acres, with an emphasis on rapid establishment of cottonwood and aspen stands that can be utilized by beaver once they are established.

<u>Project Timeline</u>: Coordination with the landowners in the area began in May 2008. A field survey of the site, including a wetland delineation, was completed in October 2008. Two design alternatives were developed and the design approach was selected in January 2009. The site design was finalized in May 2009. Restoration work is to be completed over three years starting in August 2009 and ending in October 2011.

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 E in the 2008 Scope of Work and Budget Request (Contract #37842) for the contract period June 1, 2008 through May 31, 2009.

4.6 Project E_0.1: O&M for Riparian Planting

Project Location:

Watershed: Evans Creek	Legal: T47N, R2W, S3, NW ¼
Sub Basin (River Mile): RM 0.1	Lat: 47.456725N Long: 116.5785083W

Site Characteristics:

Slope/Valley gradient: <1%	Aspect: N	Elevations: 658 m
Valley/Channel type: B2/C6	Proximity to water: Floodplain	

Other: *Project provides O&M to replace lost plants installed adjacent to 396 m of stream channel and associated floodplain.*

Problem Description: This project, started in 1999, involved establishing a riparian forest buffer encompassing approximately 2.4 hectares and 396 m of stream channel. Conifers and deciduous trees and shrubs were planted and interspersed with existing vegetation to achieve a stocking rate of 988 trees/hectare. Planted species included ponderosa pine (*Pinus ponderosa*), western white pine (*P. monticola*), western larch (*Larix occidentalis*), engelmann spruce (*Picea engelmannii*), western red cedar (*Thuja plicata*), grand fir (*Abies grandis*), thinleaf alder (*Alnus incana*), water birch (*Betula occidentalis*), vine maple (*Acer circinatum*), mountain ash (), scouler willow (*Salix scouleriana*), drummond willow (*S. drummondiana*), quaking aspen (*Populus tremuloides*), and red osier dogwood (*Cornus stolonifera*). Initial planting efforts began in 1999. A total of 1,650 live cuttings of willow and dogwood were installed on the lower stream banks during the first year. The remaining buffer areas were planted with 1,500 trees and shrubs during 2000. A year-end survival estimate indicated 83% survival of willow cuttings, 39% survival of dogwood cuttings and 75% survival for conifers. An additional 1,050 trees were planted in 2001 to replace lost trees in floodplain habitats and fill in all remaining unplanted areas.

Over a longer time period, several factors have reduced the overall survival of plantings on the site and warrant additional effort to reestablish native riparian plant communities on both streambanks and floodplain surfaces. These factors include browsing by animals, flood frequency and duration, and competition with established non-native vegetation. Prolonged flooding has occurred on site and water occasionally lingers into the early growing season due to backwater effects from Medicine Lake. This has particularly affected conifer survival on the lower floodplain elevations. Non native reed canary grass is well established on most streambanks and competes with planted vegetation and can limit both growth and survival. Many trees on the left side of the creek also died due to a fire. Western larch was most affected by browsing, while western red cedar also had high mortality due to the lack of shade. The best conifer growth has been on the east side of the creek in areas that appeared to be shaded in the afternoon.

Description of Treatment: Additional trees and shrubs were planted at the site in 2009. A total of 1.7 hectares were treated in order to fill in areas where survival has been poor. Four hundred containerized alder and 600 live willow poles were planted in areas adjacent to the creek. Live willow poles were planted so that they have a root: shoot ratio (below ground: above ground) of no less than 1.5. On the floodplain, 1,550 conifers were planted at the site, including a mix of lodgepole pine, englemann spruce and ponderosa pine, as well as 100 live black cottonwood (*Populus trichocarpa*) poles.

Project Timeline: All NEPA analysis and permitting requirements were completed for the project in 2008. Plantings were installed in May 2009.

Project Goals & Objectives: Goals for this project include 1) increase stream shading; 2) provide a long-term source of large woody debris for natural recruitment; 3) promote bank stabilization; 4) increase riparian species diversity and cover; and 5) enhance stream buffer capacity.

Relationship to Scope of Work: This work was conducted to fulfill the Program commitments for WE J in the 2009 Scope of Work and Budget Request (Contract #37842) for the contract period June 1, 2008 - May 31, 2009.

4.7 Project L_8.5: O&M for Riparian Planting

Project Location:

Watershed: Lake Creek	Legal: T48N, R6W, S1, SW ¼
Sub Basin (River Mile): RM 8.5	Lat: 47.52796N Long: 117.03564W

Site Characteristics:

Slope/Valley gradient: <1%	Aspect: N	Elevations: 780 m
Valley/Channel type: C4/E5	Proximity to water: Floodplain	

Other: *Project provides O&M to replace lost plants installed adjacent to 450 m of stream channel and associated floodplain.*

Problem Description: Reed canary grass has become well established within the riparian areas of this site to the exclusion of virtually all other species. Although reed canary grass is providing good stabilization of the stream banks, it is severely inhibiting establishment and/or regeneration of native, woody plant species that provide a more complete suite of riparian functions. The project site is located in an area where significant thermal loading can occur during base flow conditions and the canary grass community lacks the ability to adequately shade the stream channel. Revegetation with native, woody plant species is necessary to aid in reestablishing proper riparian function.

Previous plantings were completed on the site in 1998 and 1999. A power auger attached to an excavator was used to prepare the planting site by removing reed canary grass stems and rhizomes in 0.9 m diameter plots spaced approximately 3 m apart throughout the site. A combination of bareroot and containerized stock was shovel planted in the scalped areas. Species included western white pine (*Pinus monticola*), thinleaf alder (*Alnus incana*), quaking aspen (*Populus tremuloides*), and red osier dogwood (*Cornus stolonifera*). Several species of live willow poles (*S. lasiandra, S. scouleriana, and S. rigida v. mackenzieana*; 1.8 m tall, 7.6 cm diameter) were installed on the lower stream banks.

The survival of these plantings has been low due to browsing from animals, including voles and beaver, and competition from reed canary grass that has recolonized much of the site. In 2008-09, a beaver dam was established 30 m downstream of the site. This beaver killed 5 large (>30 cm dbh) willows that had survived from initial plantings.

Description of Treatment: Additional trees and shrubs were planted at the site in 2009, treating a total of 1.1 hectares of the 1.8 hectares initially planted. One-thousand live willow poles (3 *Salix sp.*) were planted along the stream banks. Willow poles were planted so that they have a root: shoot ratio (below ground: above ground) of no less than 1.5. In floodplain habitats, 300 black cottonwood (*Populus trichocarpa*) poles were planted, along with 750 conifer seedlings, including a mix of lodgepole pine, englemann spruce, ponderosa pine, and Western white pine.

Project Timeline: All NEPA analysis and permitting requirements were completed for the project in 2008. The project was completed in May 2009.

Project Goals & Objectives: Goals for this project include 1) increase stream shading; 2) provide a long-term source of large woody debris for natural recruitment; 3) promote bank stabilization; 4) increase riparian species diversity and cover; and 5) enhance stream buffer capacity. Target stream canopy density provided by woody species is 75% for the site.

Relationship to Scope of Work: This work was conducted to fulfill the Program commitments for WE J in the 2009 Scope of Work and Budget Request (Contract #37842) for the contract period June 1, 2008 - May 31, 2009.

4.8 Project 8.2/0.7: Restoration Design for *Hnmulshench* Project, WF Lake Creek

Project Location:

Watershed: Lake Creek	Legal: 24N 45E S36 E ½ of SE 1/4
Sub Basin (River Mile): West Fork 8.2/0.7	Lat: 47.526627N Long: 117.048639W

Site Characteristics:

Slope/Valley gradient: 0.6%	Aspect: N	Elevations: 826 m
Valley/Channel type: C4/E5	Proximity to water: In channel	

Other: *The project will improve conditions for fish and wildlife by creating in-stream habitat, reducing bank erosion, increasing the quantity and quality of wetlands on the site and improving water quality.*

Problem Description: The lower reaches of the West Fork Lake Creek (WFLC) 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 a history of active channel incision, accelerated fine sediment input, elevated stream temperatures, reduced instream cover and lack of large woody debris. Fish population data has been collected for the watershed since 1996. This section of the WFLC had an average westslope cutthroat trout density from 2002-2008 of 1.1 fish/100 sq. m while densities further upstream were greater than 20 fish/100 sq. m.

This stream rehabilitation project encompasses 800 m of West Fork Lake Creek and 300 m of an unnamed tributary. Both streams exhibit many of the classic signs of impairment attributed to channel ditching and straightening, which occurred sometime prior to 1937 (Figure 31 and Figure 32). The West Fork Lake Creek (WFLC) is deeply entrenched as a result of incision of the streambed as a series of headcuts migrated upstream through the reach. Historic headcuts have already moved upstream through the project site, and three additional active headcuts were identified within the reach. These active headcuts suggest that the incision trend is expected to continue as the headcuts progress upstream. There is exposed bedrock 300-ft upstream of the site preventing further incision above that point. An 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 an ongoing process of channel adjustment since initial incision occurred in both WFLC and the tributary. Direct sediment contribution from bank erosion on WFLC was estimated to be 8.9 tons/year upstream of a stream crossing and 31.11 tons/year downstream of a stream crossing. Streambank vegetation consists of mountain alder and non-native reed canary grass. The historic floodplain, where hay is produced, is perched and rarely accessed by flood flows. The upstream adjacent property is owned by Washington Department of Natural Resources and is managed for timber. The downstream property is managed as agriculture land.

Although 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 - occurring 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. The channel within the project reach is currently in different stages of adjustment leading to more stable conditions. In some channel segments the new inset floodplain width approaches 40 feet, while in other segments the width is less than 15 feet. The channel 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.

Figure 31. Photo log of streambank conditions and erosion at the WF Lake Creek project site. Evidence of severe bank erosion and channel incision resulting from the historical realignment of the natural channel prior to 1937 is present throughout the site (A). Examples of recent streambank failure and channel enlargement are found in the upper portion of the site (B). Upward migration of headcuts and subsequent erosion affects a seasonal tributary to the mainstem (C).

<u>Description of Design</u>:

Site Hydrology

Hydrologic and hydraulic parameters for the site were developed to aid in the development of the design. To estimate the hydrologic runoff characteristics of the project reach, data sets from stream gages in Lake Creek and Hayden Creek were used as well as regression equations developed for the area (Table 28). The Lake Creek gage was selected as the best suited for estimating project hydrology because the gage is within the same watershed as the project reach, it is relatively consistent with nearby Hayden Creek so regional response to precipitation is fairly consistent, and results fall approximately in the middle of the range of predicted flows for the three methods.

Regional regression equations for channel geometry, and direct measurements of existing WFLC bankfull indicators were used as sources to determine an appropriate bankfull dimension (Castro and Jackson 2001). This analysis produced results consistent with local gage data. In the field, it was difficult but possible in some areas to estimate bankfull geometries based on vegetative indicators and gravel bar forms. Measurements of analog bankfull channel indicators showed that top widths ranging 8 to 10 feet are common natural widths exhibited in WFLC. The best analog channel segment was near the upstream fence line at the DNR property. This analog section was analyzed by the Manning Equation to determine the discharge that would be conveyed by the measured channel shape and slope. The resulting flowrate was calculated to be 41 cfs, which is just slightly greater than the bankfull discharge estimations based on Lake Creek and Hayden Creek hydrology (36 cfs).

Figure 32. Historic aerial photo of the project reach taken in 1937. Stream channels were straightened and ditched prior to this time, although remnants of the natural stream channel are still visible south of the existing alignment and adjacent to a nearby hillside. Agriculture was well established as the land use for the area by this time.

Design Approach
Two conceptual design alternatives were developed with the objective of creating a stable channel and floodplain configuration, using self-sustaining natural materials, while improving riparian and in-channel fish habitat. The first alternative proposed would fast-forward the natural processes by building an appropriate floodplain width at the lower inset level. The other alternative would essentially reverse time to put the creek back up onto the valley floor as it was before incising. After discussing options with the landowner, the second alternative was chosen to the design approach.

Table 28. Peak flow discharges for the West Fork of Lake Creek.

Return Interval Flood Discharge	Est. BF	2-Year	5-Year	10-Year	25-Year	50-Year
Based on Hayden Creek Gage	40 cfs	71 cfs	126 cfs	159 cfs	193 cfs	217 cfs
Based on Lake Creek Gage	**41 cfs**	**56 cfs**	**97 cfs**	**136 cfs**	**183 cfs**	**217 cfs**
USGS Regression Equations	N/A	40 cfs	N/A	99 cfs	136 cfs	165 cfs

(Basin Area = 4.79 square miles)

The final design is displayed in Figure 33. Two thousand feet of existing incised West Fork Lake Creek channel will be completely filled and flows will be diverted into the new channel that is 3,025 ft long, increasing the stream length by more than 50%. 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 West Fork 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. Nine acres of wetlands will be created through this project (0.82 acres will be filled).

The following is the construction sequence for the project:

Phase 1A - Floodplain Grading: Create new floodplain along the southwest side of the valley. Silt fences shall be installed prior to any ground disturbing activities. Temporary stockpiles of topsoil and general fill will be created.

Phase 1B - New Channel Grading: New channel grading will generate fill while creating a new create channel excavated into the new floodplain surface to channel subgrade depth. New channel habitat to be constructed over channel subgrade by using imported gravels and logs to create streambed and streambanks. Logs will also be placed on the new floodplain to provide erosion protection and will be anchored or buried. A berm will be created to separate riparian area from crop land. Fill is to be placed in temporary stockpile areas. Stockpiles shall be stabilized by surface roughening, seeding, and mulching. The berms around the existing irrigation ponds would be decreased in height and width so that the excess material could be used as channel fill for Phase 2B.All disturbed ground will be revegetated with native plants.

Phase 1B – Culvert Installation: A new culvert will be installed where the new channel intersects the existing road.

Phase 1C - Grade Control: Grade control will be constructed of large rock designed to be relatively immobile for up to the 50-year flood.

Phase 1D - Diversion structure: A temporary diversion structure will divert low flow through the constructed new channel during the growing season. During the wet season, the diversion will be opened so that all flows will pass to the existing ditched channel. The diversion structure will keep flow out of the existing channel during phase 2 construction (filling existing channel). The diversion structure will be removed after phase 2 stabilization has been completed.

Phase 2A -Upstream Channel Improvements: Logs will be added to the existing channel upstream of the newly constructed channel to increase channel cover. Water will be pumped around this work section after fish rescue has been completed.

Phase 2B - Decommission Existing Channel: Portions of the existing WF Lake Creek channel will be filled with soil salvaged from previous excavations and stored in temporary stockpiles. Silt fences will be installed at the ends of the new stream channel. Water will be permanently diverted into the newly constructed channel. Disturbed areas will be seeded and mulched.

Phase 2C -Repair Incised Tributary: The seasonal channel will be filled with gravel. Disturbed areas will be seeded and mulched.

<u>Project Timeline</u>: Coordination with the landowners in the area began in May 2008. A field survey of the site, including a wetland delineation, was completed in October 2008. Two design alternatives were developed and the design approach was selected in January 2009. The site design was finalized in May 2009. Restoration work is to be completed over three years starting in August 2009 and ending in October 2011.

<u>Project Goals & Objectives:</u> Goals for this project include 1) create wetland habitats and increase 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.

<u>Relationship to Scope of Work</u>: This work was conducted to fulfill the Program commitments for WE F in the 2008 Scope of Work and Budget Request for the contract period June 1, 2008 - May 31, 2009.

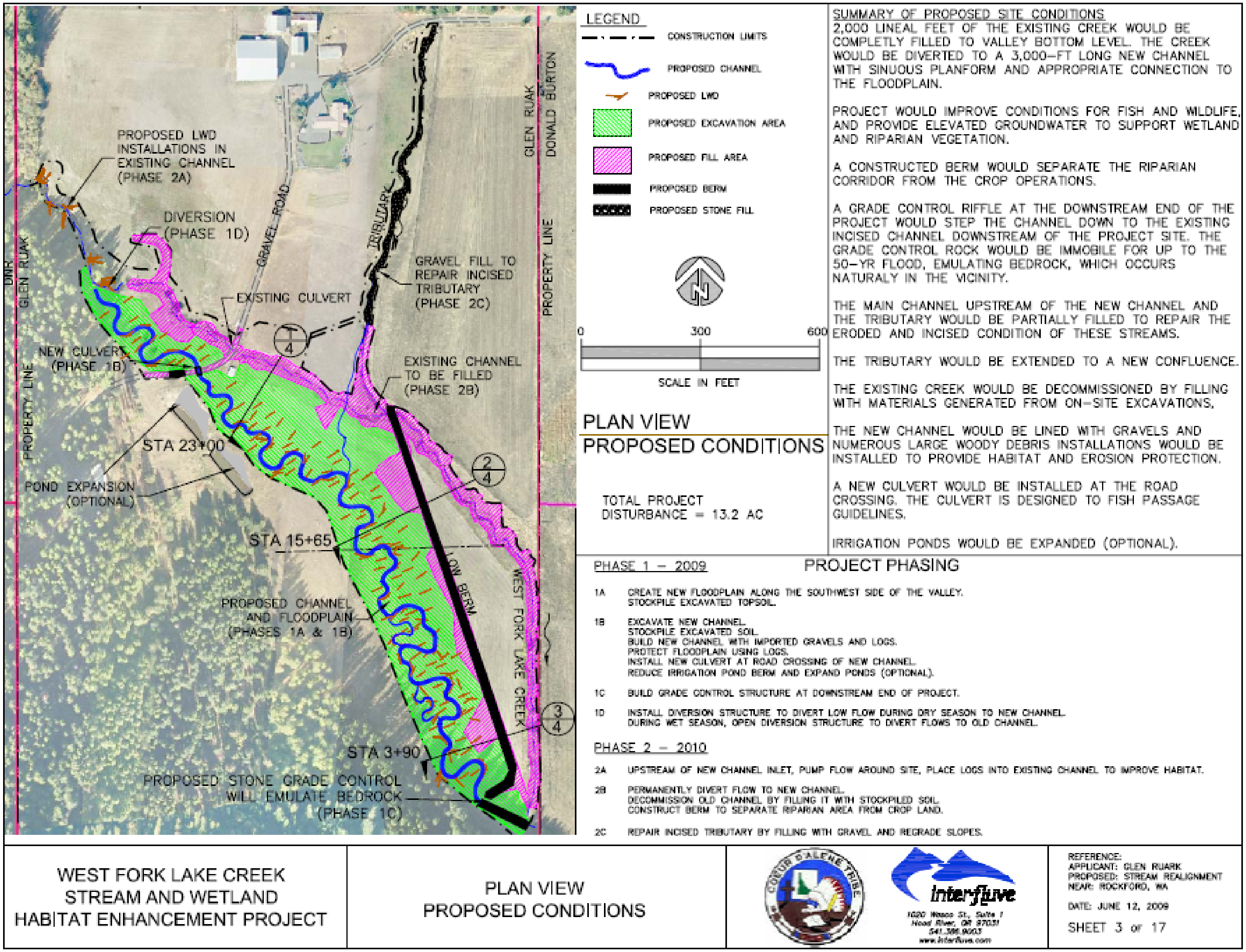

Figure 33. Design approach for the West Fork of Lake Creek project.

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APPENDIX A – HABITAT SITE CROSS-SECTION COMPARISONS

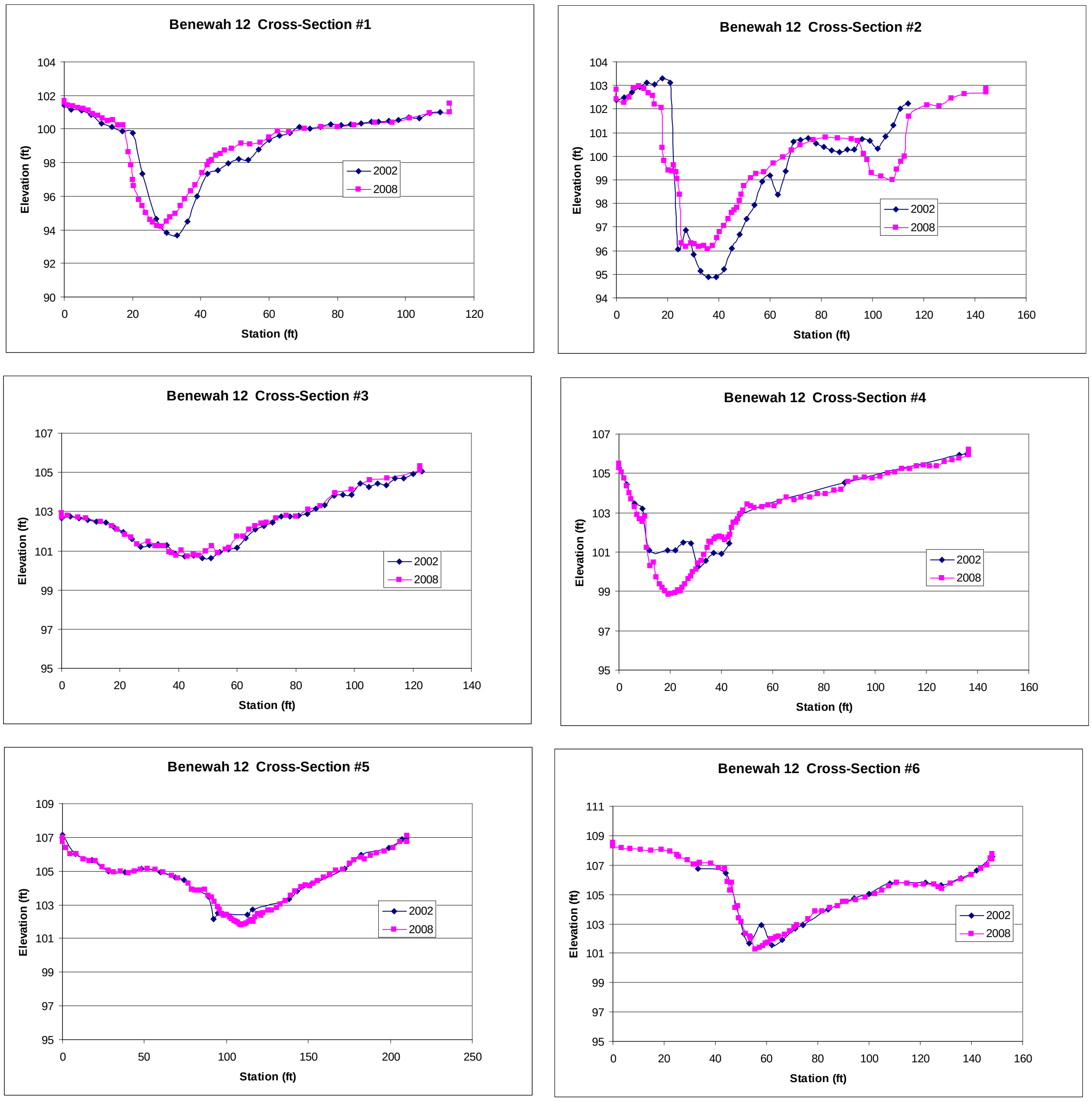


Figure A-1. Cross-section comparison for site Benewah 12 for 2002 and 2008.

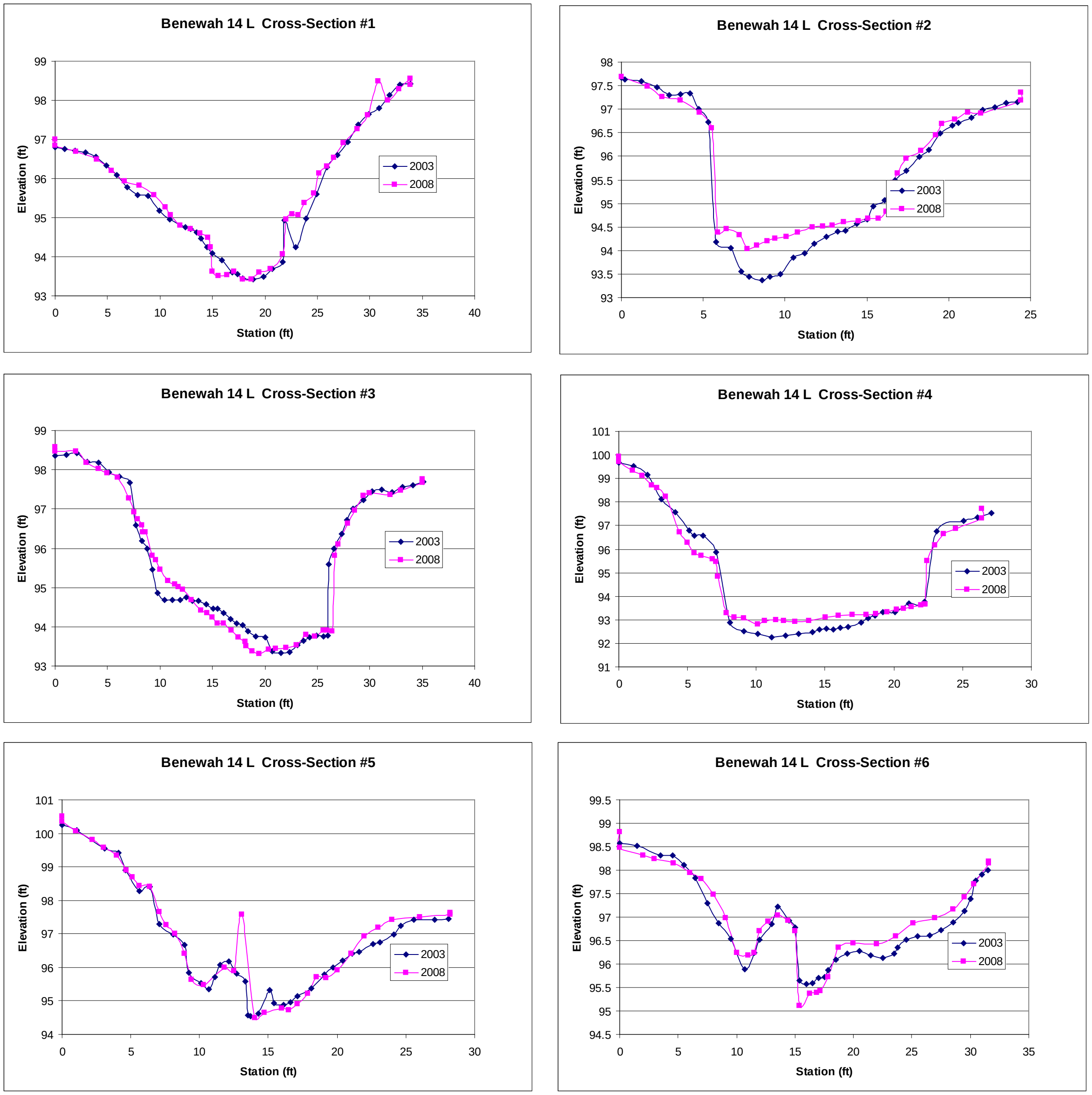
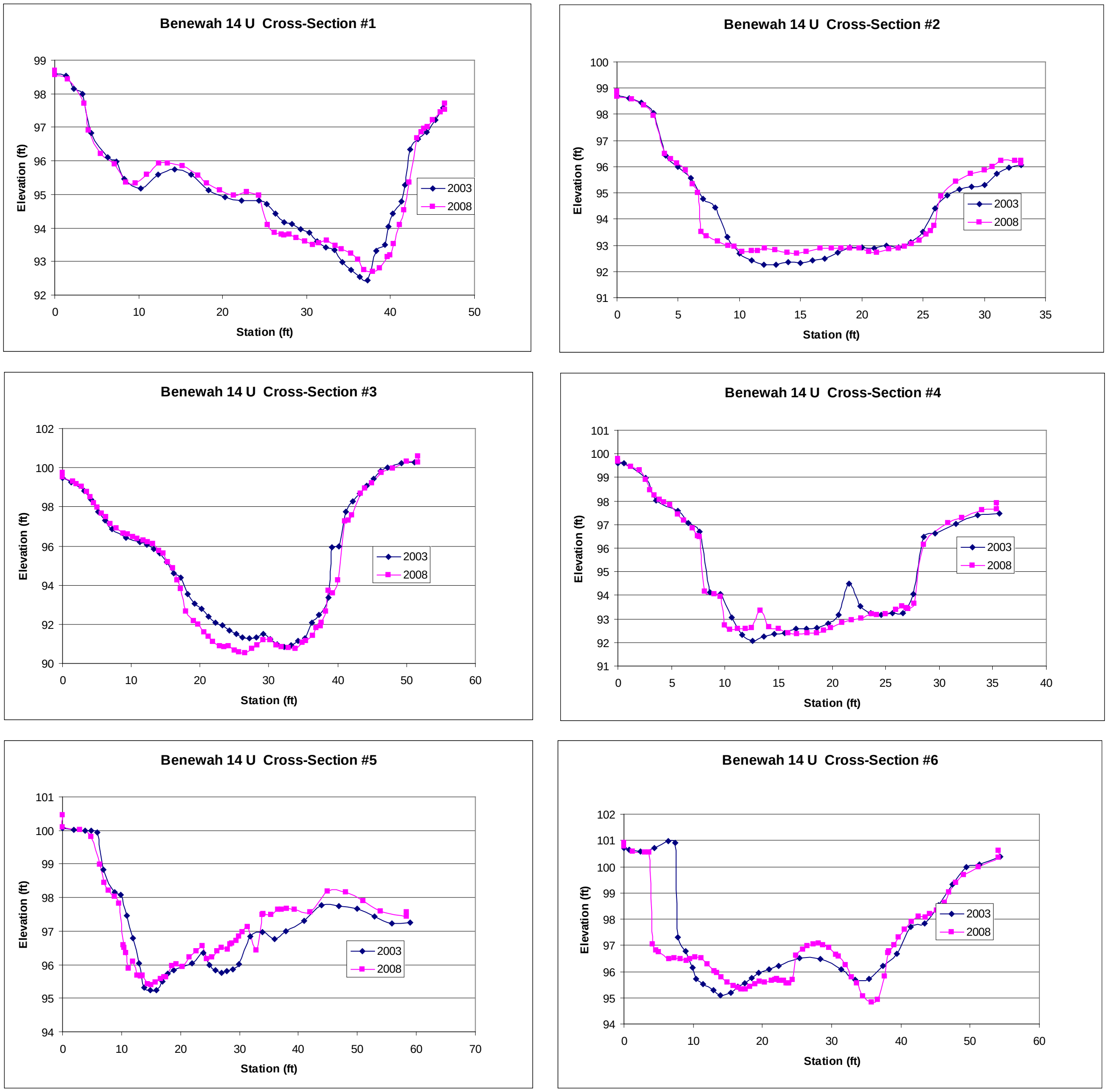


Figure A-2. Cross-section comparison for site Benewah 14L for 2003 and 2008.



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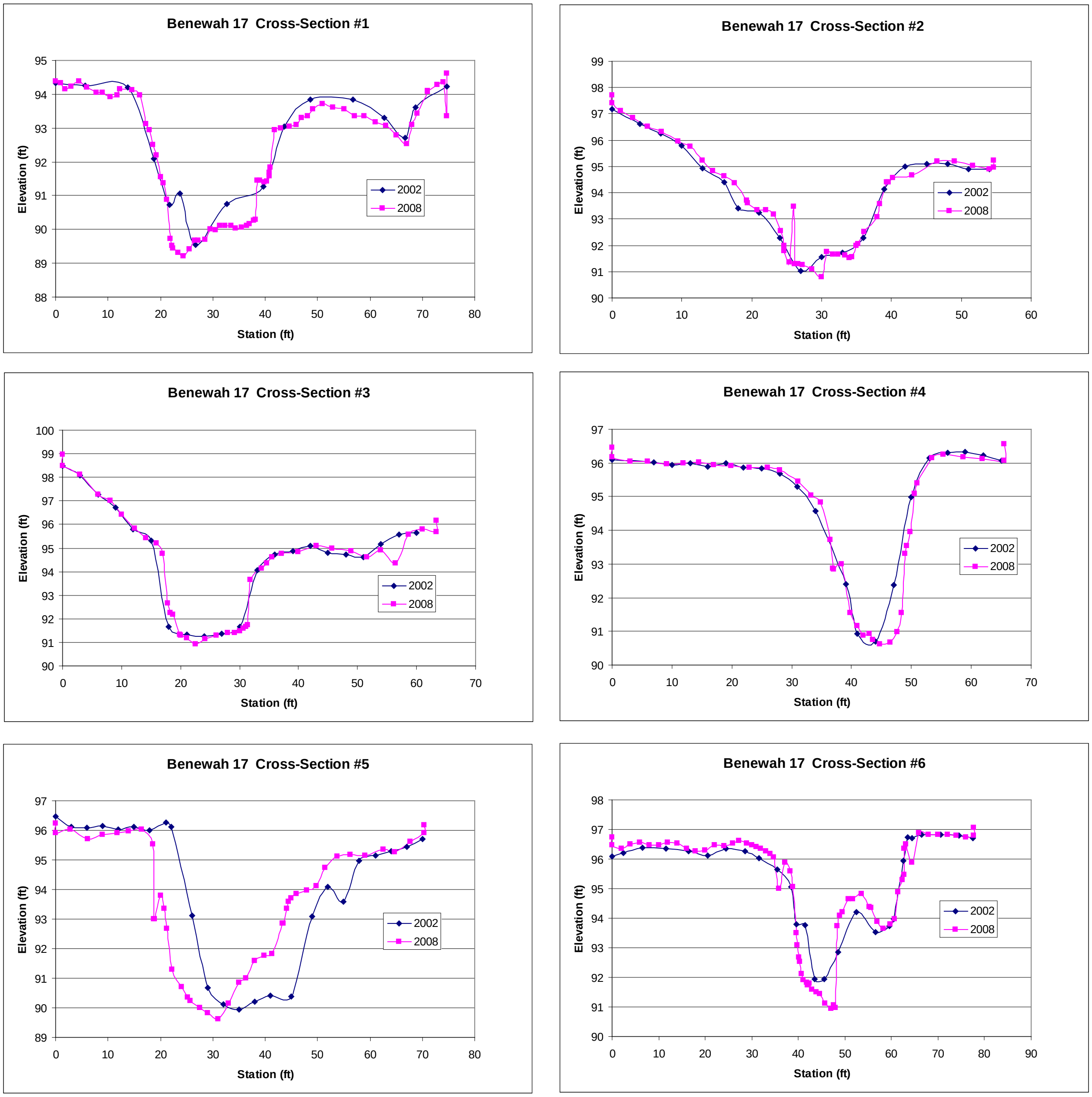

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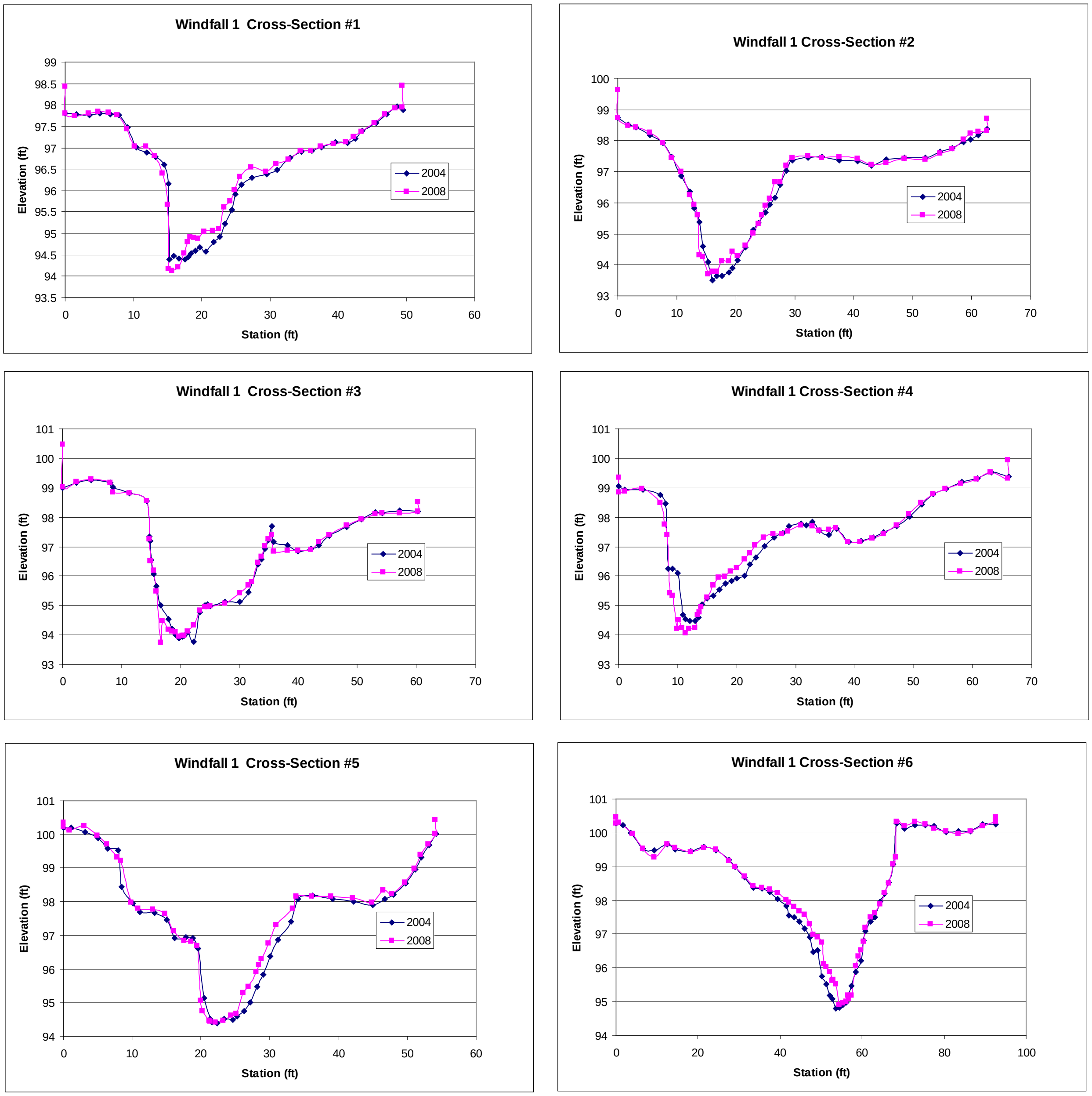


Figure A-5. Cross-section comparison for site Windfall 1 for 2004 and 2008.

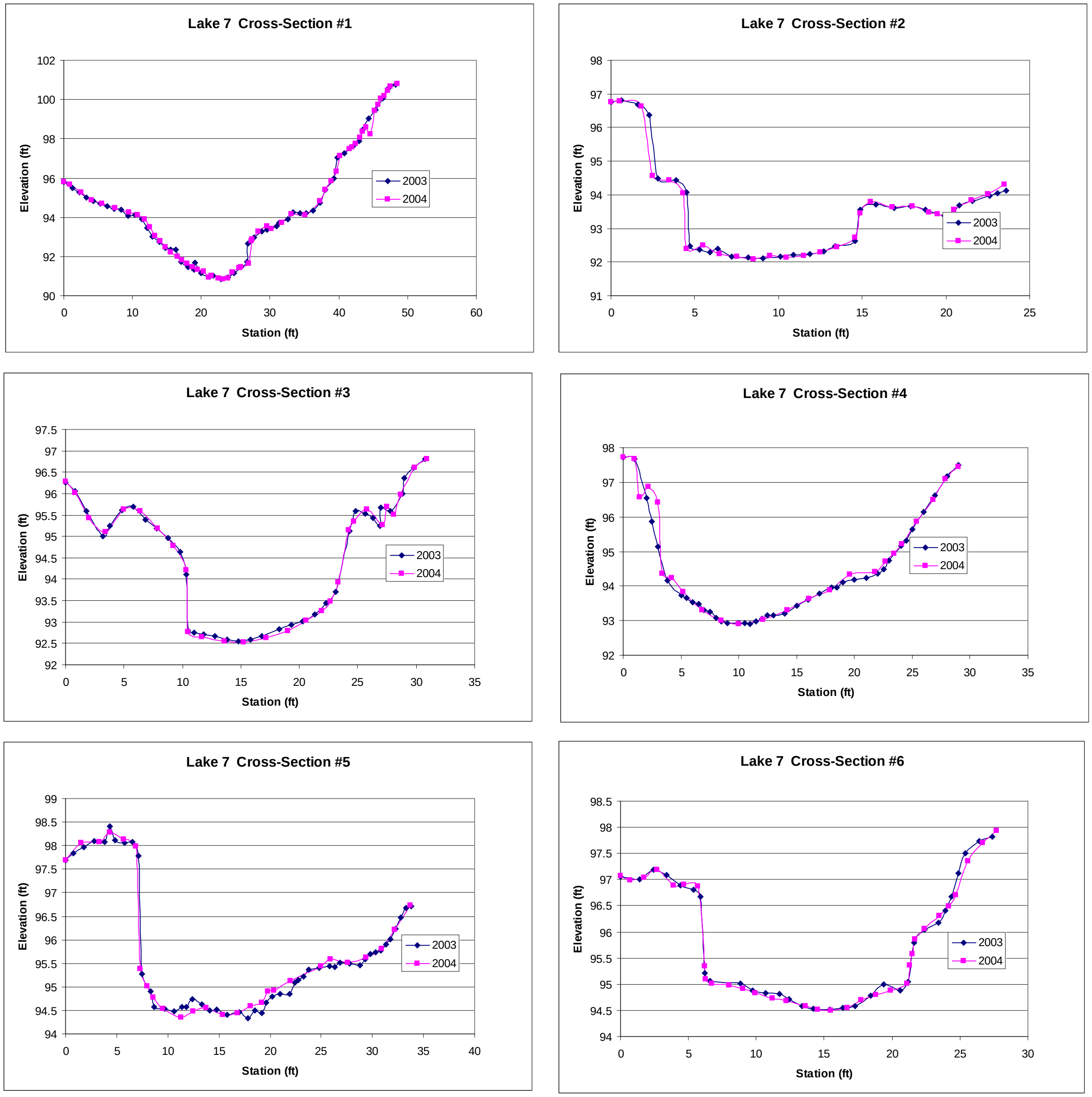


Figure A-6. Cross-section comparison for site Lake 7for 2003 and 2008.

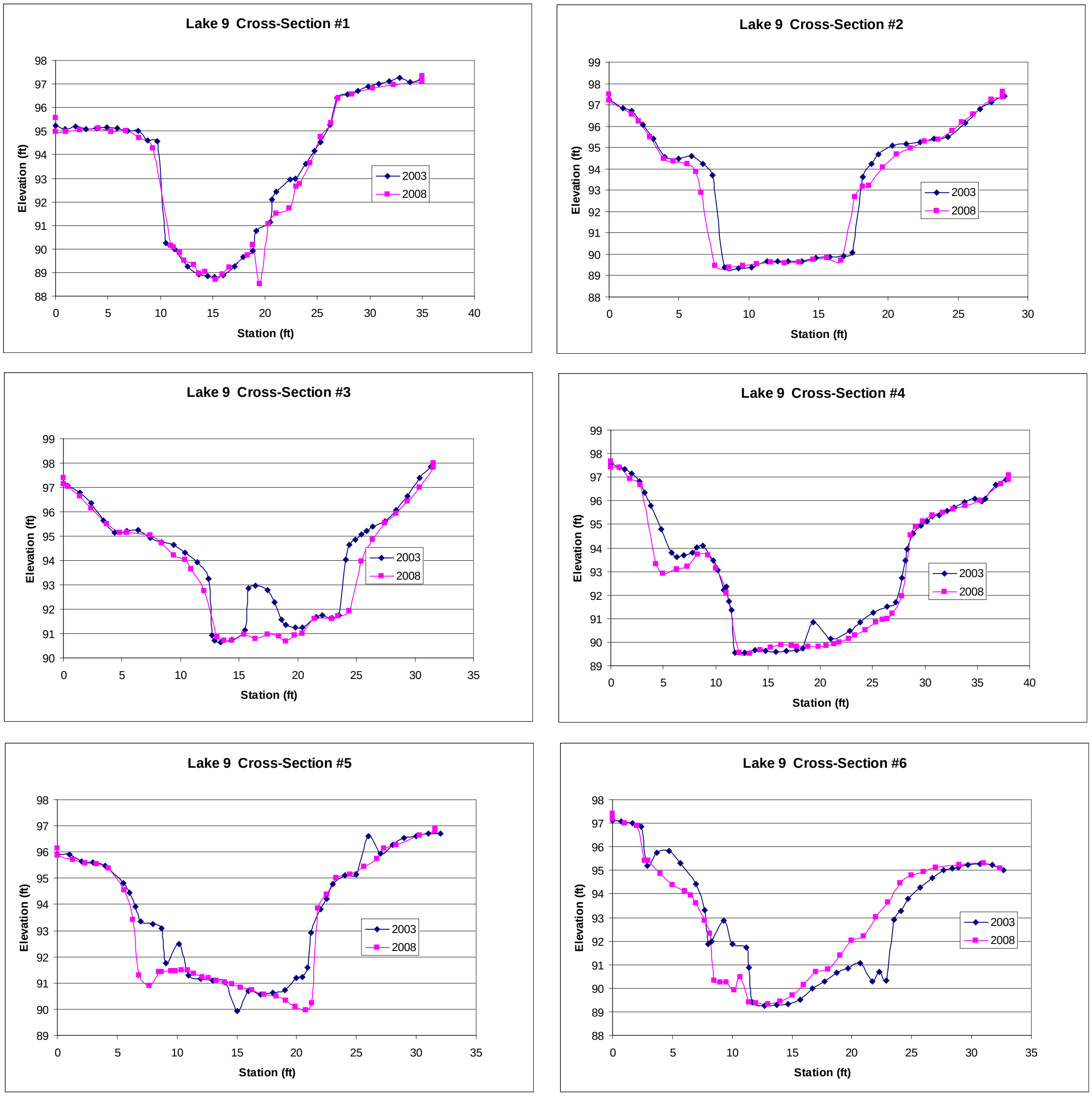
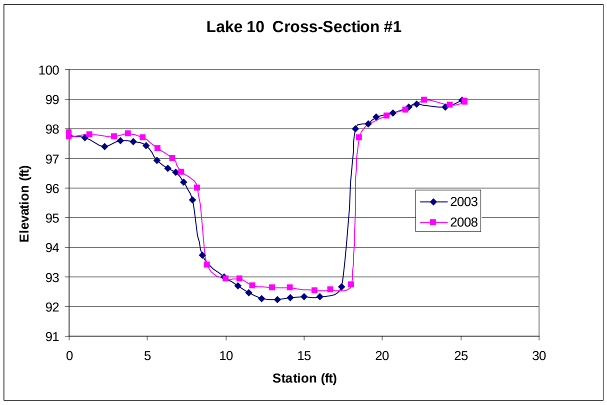
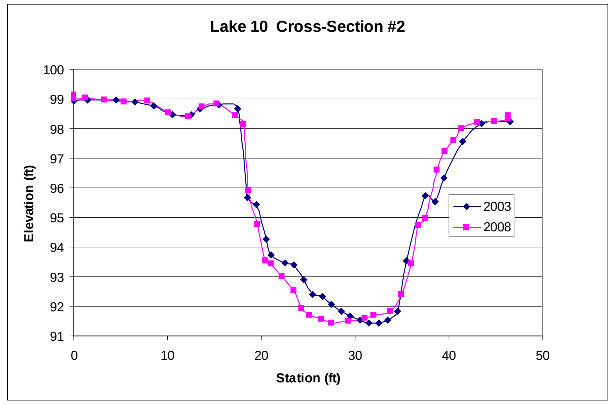
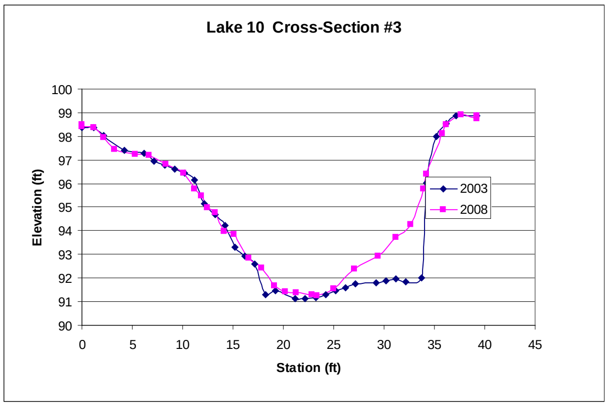
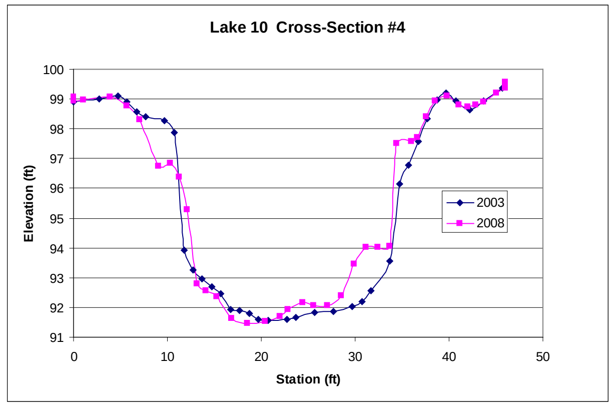


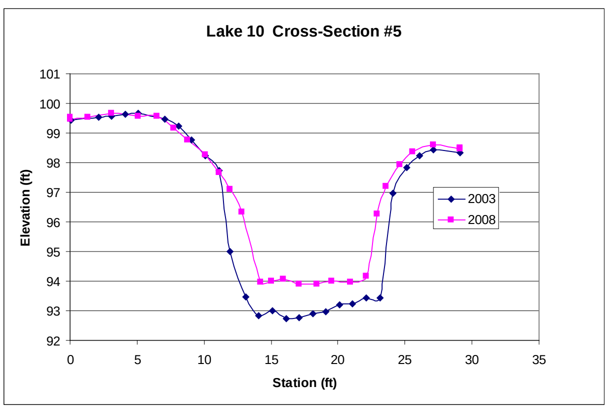
Figure A-7. Cross-section comparison for site Lake 9 for 2003 and 2008.











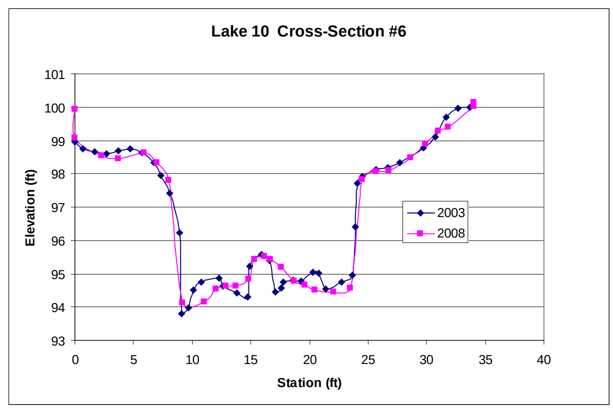


Figure A-8. Cross-section comparison for site Lake 10 for 2003 and 2008.

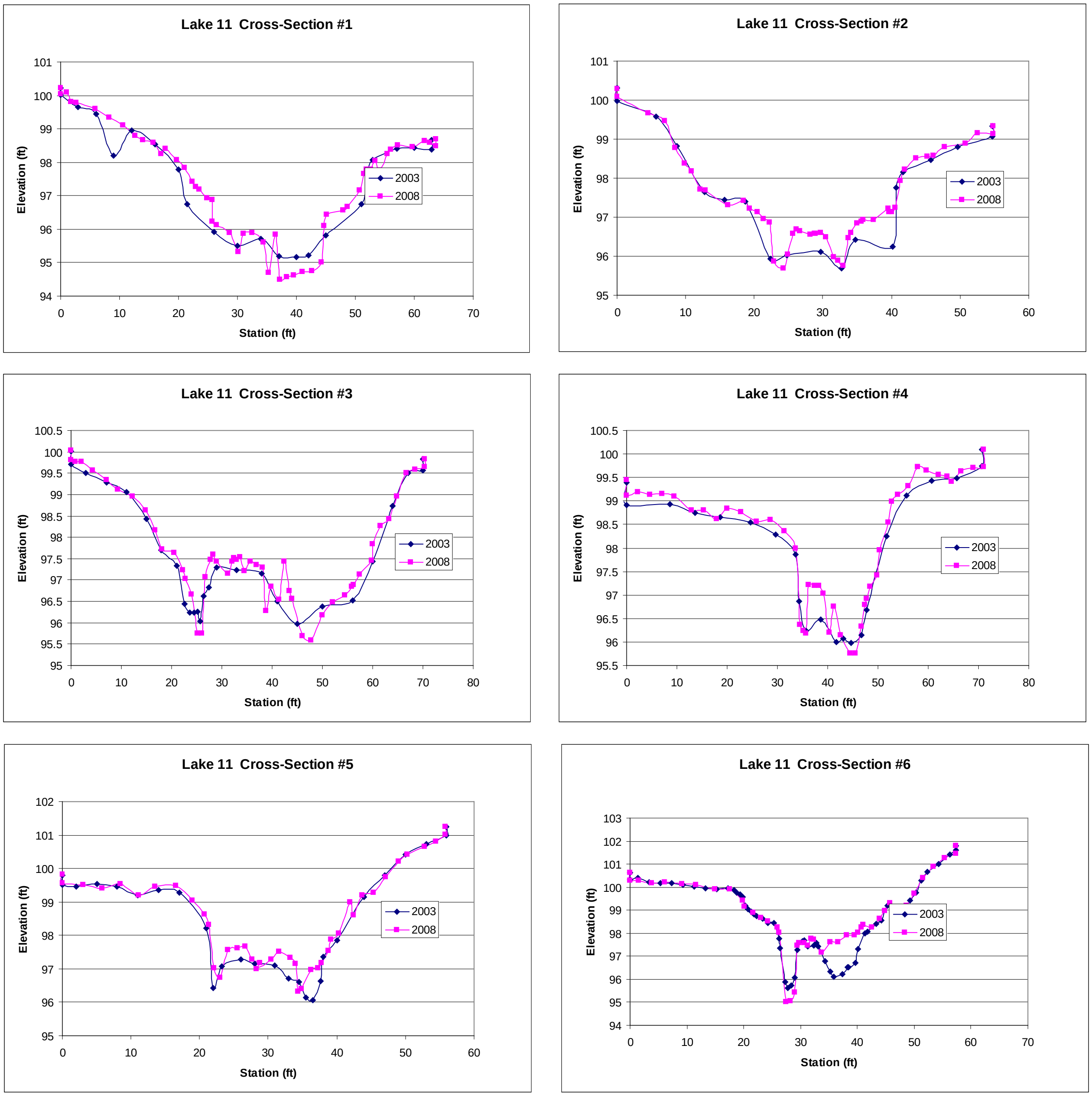


Figure A-9. Cross-section comparison for site Lake 11 for 2003 and 2008.

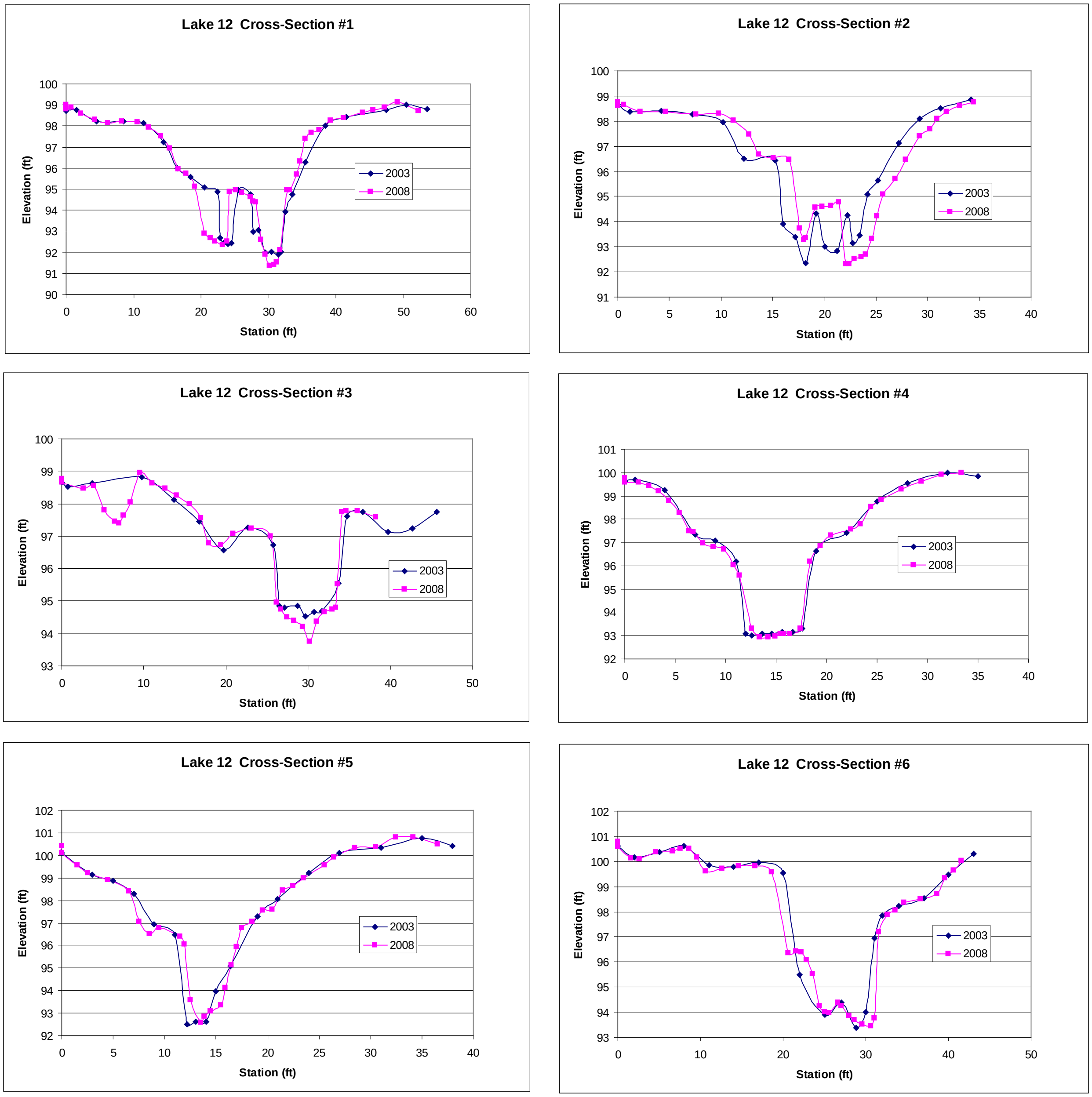


Figure A-10. Cross-section comparison for site Lake 12 for 2003 and 2008.

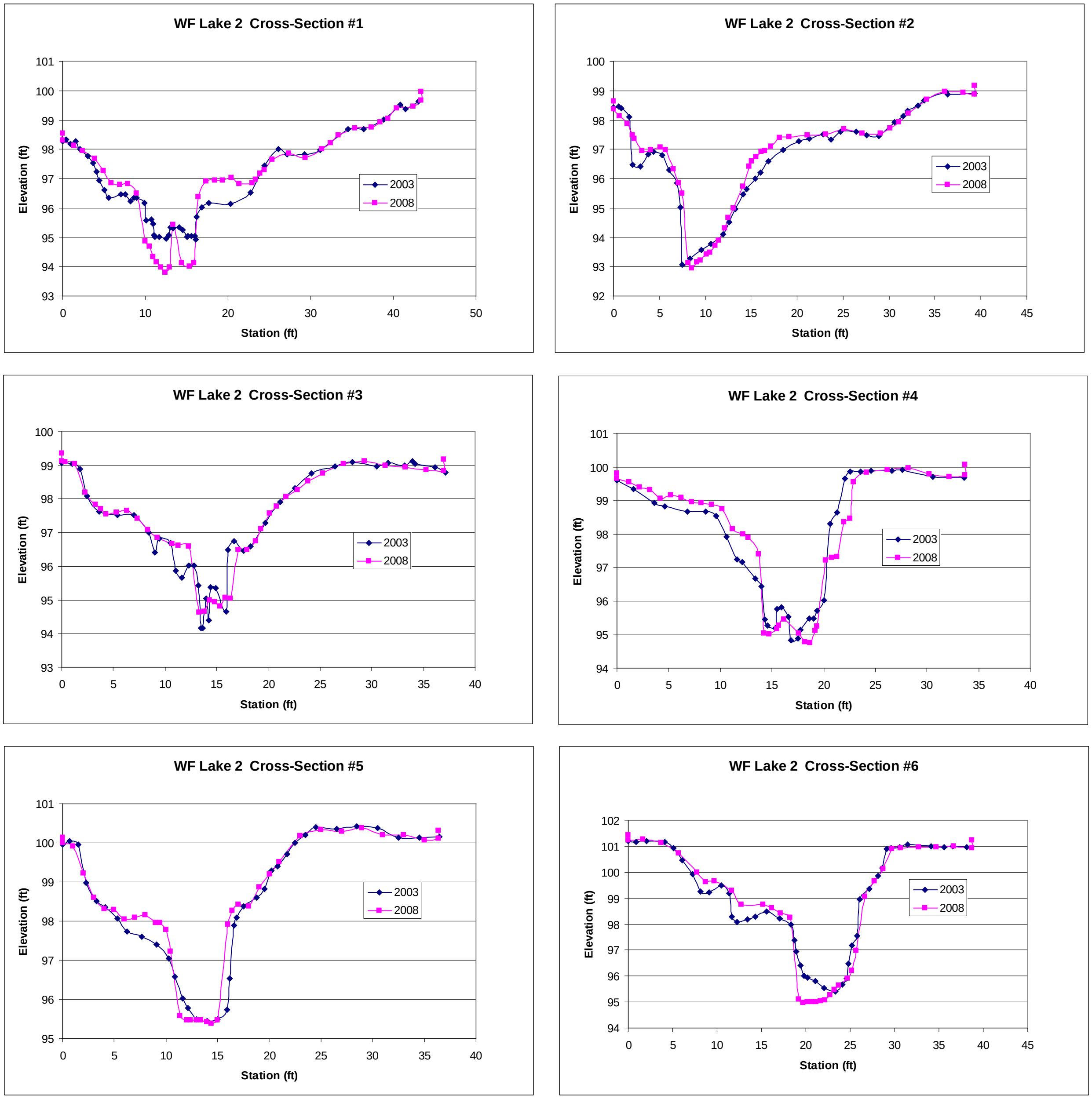


Figure A-11. Cross-section comparison for site West Fork 2 for 2003 and 2008.

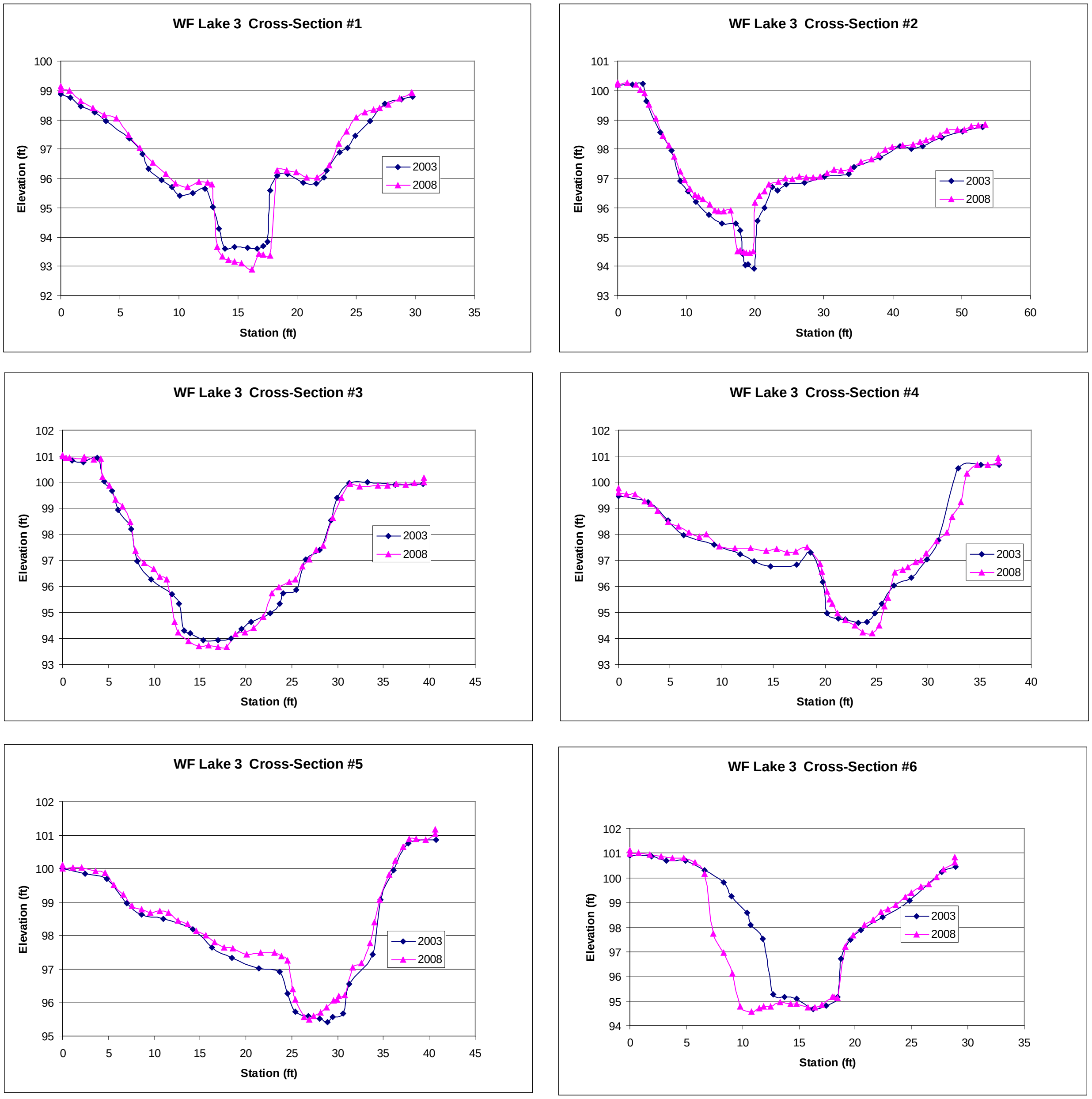


Figure A-12. Cross-section comparison for site West Fork 3 for 2003 and 2008.

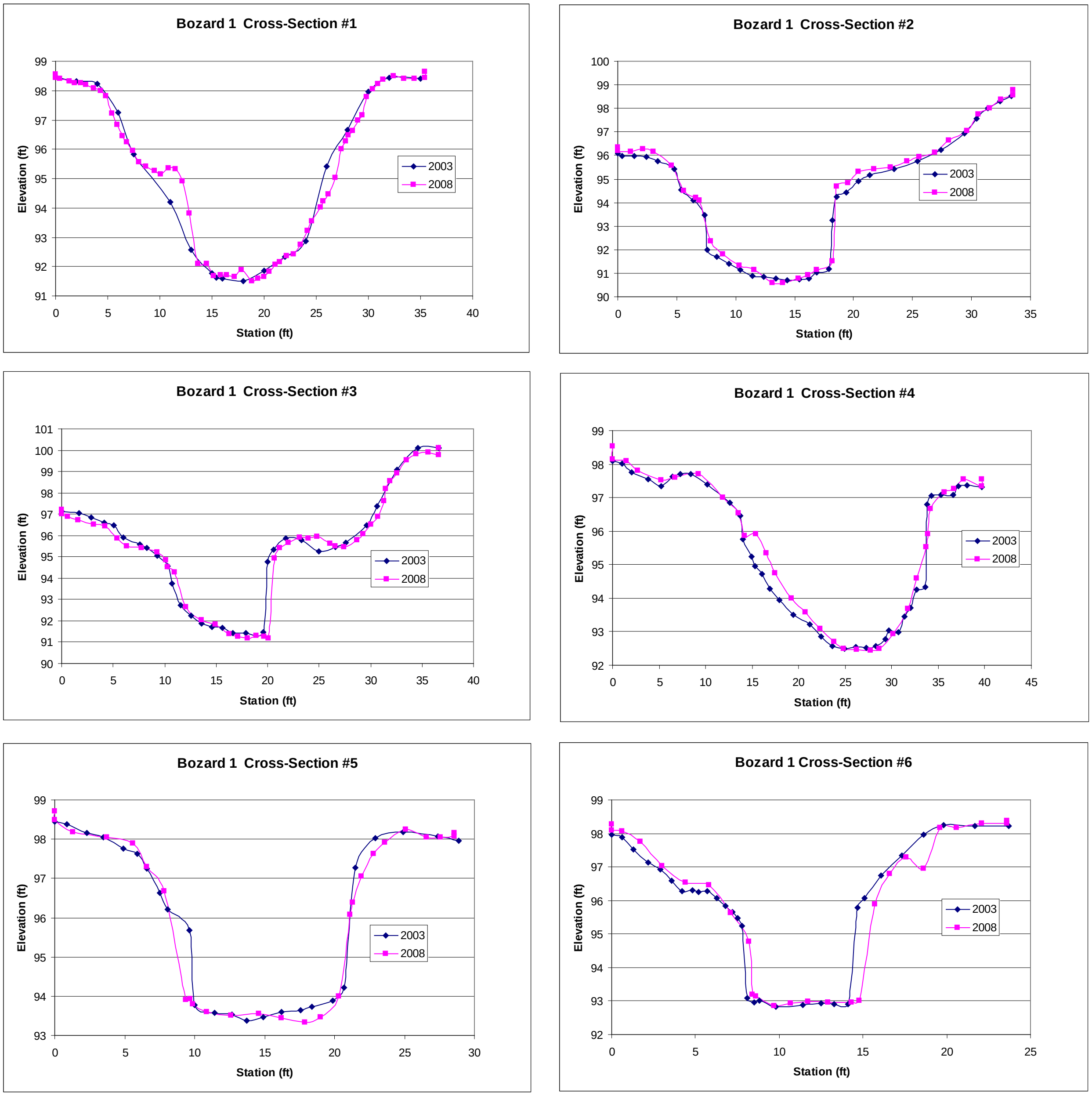


Figure A-13. Cross-section comparison for site Bozard 1for 2003 and 2008.

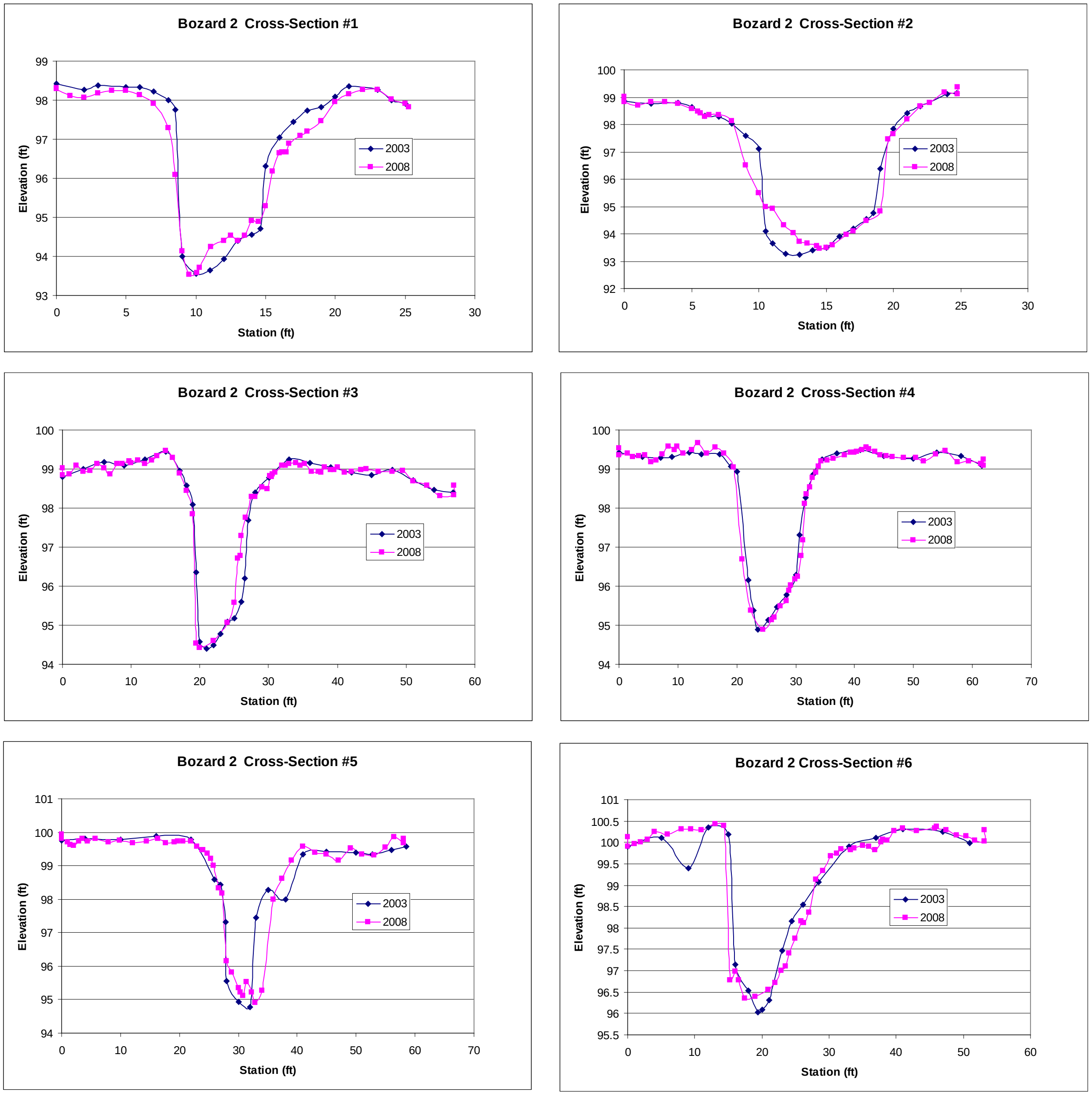


Figure A-14. Cross-section comparison for site Bozard 2for 2003 and 2008.

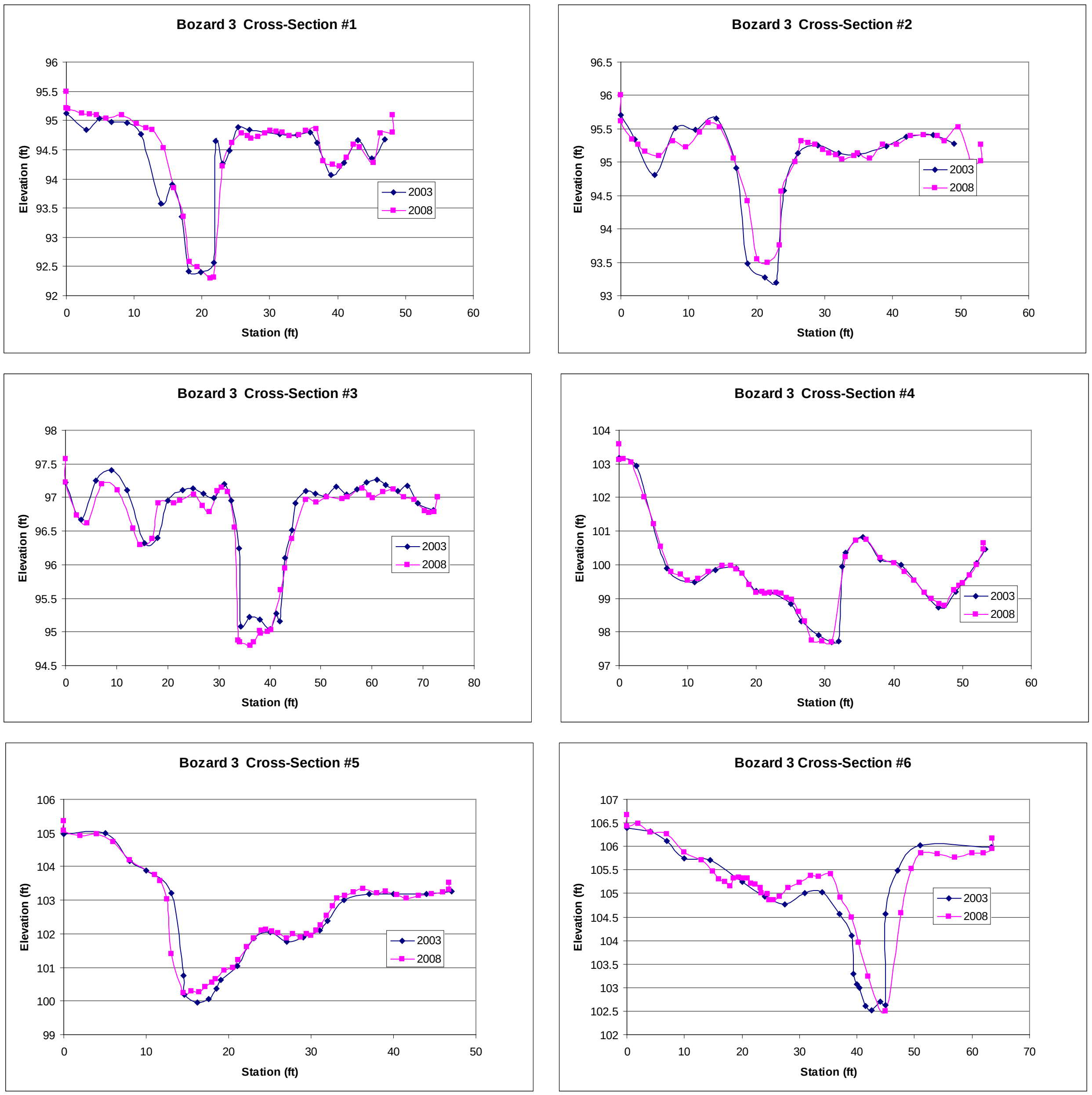


Figure A-15. Cross-section comparison for site Bozard 3 for 2003 and 2008.