Research Report
BC Journal of Ecosystems and Management

Response of bull trout (*Salvelinus confluentus*) to habitat reconnection through replacement of hanging culverts with bridges

J. Mark Shrimpton[1], Christopher J. Cena[2], and Adrian D. Clarke[3]

Abstract

We examined the effectiveness of road culvert replacement on providing access to fish habitat in two tributary streams of the Torpy River, in central British Columbia. For both study streams, culverts had been "hanging" at the downstream end, which created waterfalls to the streambed below. To facilitate fish passage upstream, culverts were replaced with steel bridges. In one of the streams, fish movement extended above the culvert. Benthic invertebrate and fish communities, and stream channel physical characteristics were assessed before and after replacing the culverts. Physical characteristics and the benthic invertebrate communities were similar for the two study streams. The primary species found in both streams was bull trout (*Salvelinus confluentus*), with rainbow trout (*Oncorhynchus mykiss*) comprising less than 3% of the total fish captured. Before culvert removal, fish were absent above the road in one of the streams. The presence of young-of-the-year fish 2 years after bridge construction, however, indicated that bull trout had colonized and spawned in the reconnected habitat within 1 year. The relative abundance of bull trout in the stream where access was not restricted by the culvert did not differ over the 4 years of the study. Conversely, abundance and year-class structure of bull trout improved in the stream where habitat above the road was reconnected. Our findings indicate that removal of hanging culverts to allow fish passage is an effective management approach to increase available fish habitat.

KEYWORDS: *bull trout, culvert, stream, Torpy River.*

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Introduction

The combination of natal homing behaviour among salmonids and anthropogenic impacts has led to declines in and losses of individual river populations of salmonids over much of the 20th century. Specifically, blockage of migratory routes by dams, destruction of spawning habitat, and over-harvesting have led to significant declines in numbers of returning fish, and the extirpation of some populations (Nehlsen *et al.* 1991; Waples 1994; Slaney *et al.* 1996). Recent management efforts for the enhancement of wild populations of fish have focussed on restoration and rehabilitation of natural waterways to achieve more natural watershed processes. The approach removes structures that limit access to habitat and creates new habitat using artificial structures. Monitoring restoration procedures is a key requirement for evaluating the effectiveness of stream habitat manipulation programs, and is vital for improving and adapting techniques used in stream restoration. Theoretically, the practice of restoring habitat to a natural state should succeed; however, little information is available regarding the effectiveness of this approach. In many cases, the engineering and construction of suitable structures has occurred without biological assessment of their effectiveness. An exception is the study by Lenhart (2003) that estimated the value of habitat "opened" by restoration projects in relation to access to spawning, rearing, and (or) foraging areas. Even this study, however, indicated the difficulty in assessing habitat value upstream of blockages in "a way that is both quantitative and meaningful" (Lenhart 2003:87). Recently, Rieman *et al.* (2001) and Wilson (2003) used mathematical models to assess recovery strategies for anadromous populations of salmon. Their models showed that the biggest improvement in salmonid populations can be realized by increasing access to upstream habitats.

Round culverts and corrugated metal pipes are commonly used where roads cross streams, but are likely to cause barriers to migration (Furniss *et al.* 1991; Langill and Zamora 2002). These structures may restrict upstream fish movement because of excessive water velocity, insufficient water depth, no resting pool below or above the culvert, or "hanging" at the downstream end which creates a waterfall with too high a jump (Furniss *et al.* 1991). Barriers to migration from culverts and corrugated pipes result in greater habitat loss and fragmentation than other crossing types, such as bridges (Harper and Quigley 2000). Historically, the reason for using corrugated metal pipes at road crossings has been financial. The cost of installing a 20 m single-span bridge can be up to 200 times the price of placing a 90 cm corrugated metal pipe (Gibson *et al.* 2005). The negative effect of culverts and poor construction practices has been well documented (see recent review by Gibson *et al.* 2005). The effectiveness of restoring access to fish habitat by replacing culverts with open-bottom structures that follow the natural contours of the streams must therefore be assessed.

Recent management efforts for the enhancement of wild populations of fish have focussed on restoration and rehabilitation of natural waterways to achieve more natural watershed processes.

It has been well documented that semelparous salmonids home to specific locations for spawning (Quinn 1993) and iteroparous salmonids show considerable site fidelity (Bahr and Shrimpton 2004). A crucial question is: would fish move into and use habitat that was inaccessible in the past? To answer this question, we sampled two streams for fish presence where habitat restoration projects had improved fish passage—one where no passage existed for approximately 30 years after road construction and another where access allowed some migration to upstream habitat.

Study Location

In the early 1970s, an access road for timber harvesting was constructed in the Torpy River watershed, British Columbia (53°44'N, 120°54W) (Figure 1). A portion of this road runs parallel to the mainstem of the Torpy River for approximately 35 km. Corrugated steel pipe culverts were installed to channel flow beneath the road at the majority of these stream crossings. An inventory of stream crossings completed for CANFOR in October and November 1998 identified culverts on 27 tributary streams. Twenty-three of these culverts were "hanging" at the downstream end creating waterfalls to the streambed below (Figure 2).

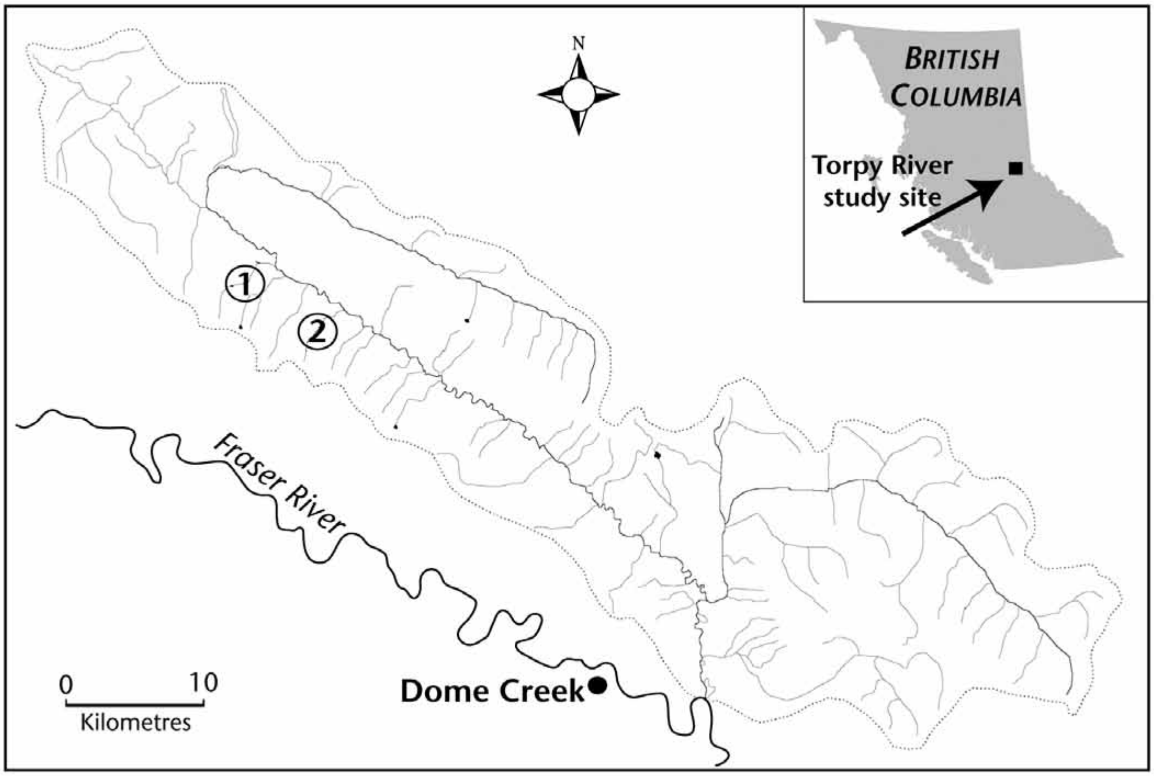

FIGURE 1. The upper Torpy River watershed showing the locations of the sampled streams where culverts were replaced by bridges. Location of stream at kilometre 19 is indicated by "1" and stream at kilometre 26 indicated by "2."

FIGURE 2. Culvert going under the lower Torpy road at stream 19 (July 1999).

FIGURE 3. Bridge on the lower Torpy road crossing stream 26 (July 2000).

A resource plan was implemented to remove culverts that impeded flow and restricted fish passage on the Lower Torpy Mainline. In late August 1999, culverts (1.8 m steel) were removed from two of the streams located at kilometre 19.4 (stream 19) and 26.6 (stream 26) and replaced with 15.7 m steel and concrete bridges. The drop from the culverts to the plunge pools before construction was 1.39 m and 1.15 m, respectively. Bridge construction was designed to restore the natural creek channel elevation upstream and downstream of each barrier (Figure 3).

Methods

Physical Measurements

Above and below the road, cross-sections of stream channels were measured at more than five intervals at locations influenced by the stream habitat improvements. Measurements were also conducted above and below areas affected by stream habitat alterations and the road. At each site, mean wetted width, depth at 25-cm intervals, and velocity at 60% of depth were measured. Gradient was measured at each site with a Total Station (Nikon Model D-50). Temperature loggers (Onset Instruments, Mass.) were placed approximately 30 m above the road for each stream. Temperature loggers were deployed in early July and removed by the end of August in 1999, 2000, and 2001; precision was ± 0.02°C.

Invertebrate Sampling

In each stream, macroinvertebrates were sampled in triplicate at a representative riffle above and below the road with a Surber sampler (mesh 250 μm) in July of 1999 and 2000. The 0.093 m^2 area within the sampler frame was disturbed and substrates were scrubbed for 5 minutes to dislodge attached invertebrates. Samples were preserved in 70% ethanol and subsequently identified to family. We calculated the family-level biotic index (FBI) according to Hilsenhoff (1988). A three-factor analysis of variance (ANOVA) was used to determine differences in FBI between the streams, above and below the road, and between years.

Fish Sampling

Single-pass electrofishing (Smith Root Model 12C, Vancouver, Wash.) was conducted on streams 19 and 26 in July and August of 1999 to assess presence or absence of fish above and below the road. We also set minnow traps, but they were not effective in capturing fish, limiting our analysis to electrofishing. Because of the negative effects of electrofishing on fish physiology, we chose to limit our effort on any given sample date to minimize stress. Instead, we sampled twice each season for the first 3 years to allow fish to recover fully from the impact of electrofishing. We started electrofishing approximately 250 m below the road and gradually worked upstream toward the road. Above the culvert, we continued to electrofish upstream until natural barriers were reached (210 m on stream 19 and 247 m on stream 26). After culvert removal and bridge installation in late August 1999, the same stream reaches were electrofished in July and August of 2000 and 2001, and in August 2002. Fish caught by electrofishing were identified to species and measured for fork length to 0.1 cm. For each stream, length of fish caught after culvert removal was compared to those caught before culvert removal, above and below the road, using Chi-squared tests. Because of the small number of fish caught and lack of all size classes, fish were grouped as larger or smaller than 10 cm.

Results

Physical Attributes

The overstorey within the riparian zones of both streams was dominated by balsam fir, black spruce, and poplar. Groundcover was composed of willow, horsetail, and devil's club. The bed material for both streams was primarily composed of cobble-size rocks (7.5–30 cm) with larger boulders. Other physical variables are given in Table 1. Flows were higher and channel width was greater in stream 26, while gradient was greater in stream 19. Temperatures were similar for the two streams.

TABLE 1. Physical and biological attributes for two tributaries of the Torpy River, British Columbia, where culverts were replaced by bridges to improve fish passage to habitat above road crossings. All values are presented as means and (where appropriate) ± 1 standard error.

	Stream 19	Stream 26
Water temperature (°C)[a]	8.73 ± 1.27	8.78 ± 1.42
Mean wetted width (m)[b]	3.5	4.3
Gradient %[c]		
Above road	5.81	4.51
Below road	7.35	4.95
Mean discharge (m^3/s)[d]	0.41 ± 0.05	0.48 ± 0.08
Substrate type	Cobble/boulder	Cobble/boulder
Family-level biotic index[e]		
1999	3.49 ± 0.05	3.05 ± 0.05
2000	3.16 ± 0.03	3.20 ± 0.12

[a] Average temperature from July 15 to August 15 for 1999, 2000, and 2001.

[b] Average of two measurements taken approximately 30 m above and below the road in mid-July.

[c] Gradient determined over 25 m.

[d] Average flow rates determined from measurements taken in mid-July of 1999, 2000, and 2001.

[e] Family-level biotic index for aquatic macroinvertebrates (Hilsenhoff 1988).

Biological Attributes

For benthic invertebrates, five families of Ephemeroptera were dominant and consisted of more than 75% of the total number of animals. The most common and abundant families were Heptageniidae (49% in stream 19) and Baetidae (38% in stream 26). The other orders represented in both streams were Plecoptera (principally 6% Chloroperlidae, 1.7% Leuctridae, and 1.4% Capniidae) and Diptera (principally 5.4% Chironomidae). The order Tricoptera represented 2.7% of the invertebrates sampled. We found little difference in abundance between samples taken above or below the road. The FBI did not differ significantly before and after culvert removal ($F = 0.453$; $p = 0.52$), between streams ($F = 3.08$; $p = 0.10$), or above and below the road ($F = 0.93$; $p = 0.35$) (Table 1).

The principal species of fish captured in both streams was bull trout (*Salvelinus confluentus*). A few rainbow trout (*Oncorhynchus mykiss*) were caught (less than 3% of the total catch), but not on every sample date. No other species of fish were captured in either stream. In all years, the number of fish captured was higher in August than July. In July, fish captured did not represent multiple age classes. Consequently, in our final year of sampling in 2002, we only sampled in August. Data in this research report are presented for August as it covers the greater time frame.

Before culvert removal, fish were present throughout stream 26, but in stream 19 fish were only found below the road (Figure 4). Multiple size classes of bull trout were captured above and below the road each year for stream 26. During our sampling efforts for 2000, size

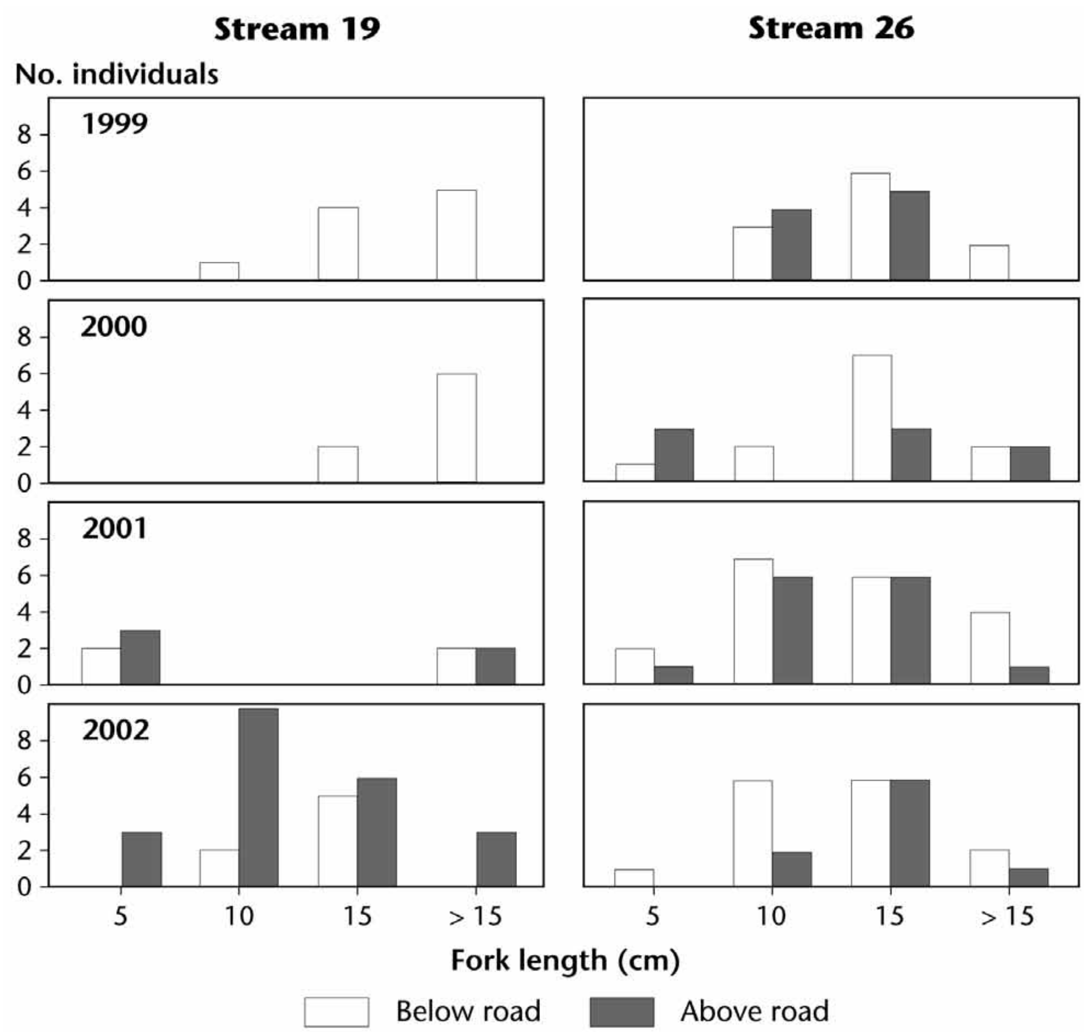

FIGURE 4. Number of different-sized bull trout captured in two tributary streams of the Torpy River, British Columbia. Data are presented for fish sampled in August of 1999, 2000, 2001, and 2002.

FIGURE 5. Mature male and young-of-the-year bull trout captured in stream 19 above the road in August 2001. Young-of-the-year bull trout can be seen just above the anal fin of the older fish (white arrow).

class and distribution of bull trout in stream 26 did not differ from the previous year ($p = 0.78$ below road; $p = 0.56$ above road). For stream 19 sizes of fish below the road did not differ after culvert removal ($p = 0.72$).

In 2001 and 2002, we captured bull trout above and below the road in both streams. For stream 26, size classes did not differ significantly ($p = 0.26$ above, 2001; $p = 0.11$ above, 2002; $p = 0.13$ below, 2002), except for below in 2001 ($p < 0.05$). For stream 19 in 2001, there were significant differences in sizes of fish caught above ($p < 0.05$) and below ($p < 0.001$) the road; bull trout above the road comprised two age classes—young-of-the-year and mature fish greater than 25 cm (Figure 5). In 2002, multiple size classes of fish were caught in stream 19, which differed significantly from before culvert removal ($p < 0.001$ above; $p < 0.05$ below). Our estimates of abundance based on catch per unit effort (CPUE) showed differences between the streams and over time. Catch per unit effort was lower in stream 19 than 26 in the first 2 years, but little difference was evident between the two streams 3 years following creation of access to upstream habitat (Figure 6).

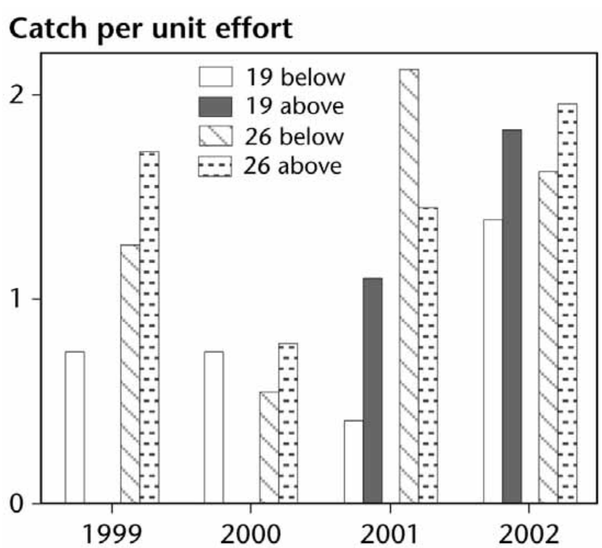

FIGURE 6. Catch per unit effort (number of individuals caught per minute of electrofishing) for the two study streams 1 year before and 3 years following culvert removal. Data are presented for August of each year.

Discussion

Little difference was evident in physical attributes above and below the road for both streams (Table 1). Physical habitat and invertebrate abundance suggested suitable habitat for fish. The calculated values of FBI also indicated that the water quality within these streams was excellent (Hilsenhoff 1988). Between streams, the notable difference was the absence of fish above the road in stream 19 before culvert replacement. The drop of water from the hanging culvert on stream 19 was only 0.24 m greater than stream 26, but fish passage above the road was prevented. The size of the plunge pool below the road on stream 19 was also smaller, which may have limited the ability of fish to move upstream (Furniss *et al.* 1991); however, we cannot rule out that at times of high flow the culverts may have allowed passage. The lack of fish caught above the road and the presence of young-of-the-year bull trout after culvert removal suggest that the culvert on stream 19 blocked all passage of fish above the road. We do not know whether fish passage was possible in the years immediately following culvert placement, but before bridge construction access to fish habitat above the road had been lost in this stream.

Culvert removal and bridge construction took place during the last week in August 1999. In stream 19, we did not detect the presence of any fish above the road in our sampling efforts on July 16 and August 15, 2000; in stream 26, the numbers and sizes of fish caught were similar to the previous year (Figure 4). In 2001, bull trout captured from stream 19 were smaller than 5 cm (young-of-the-year) indicating that bull trout must have been present above the road in 2000 and that spawning occurred. The flow of stream 19 directly below the bridge likely created a barrier to upstream movement of small fish as the stream channel narrows, gradient increases, and the velocities measured were greater than 1.5 m/s. It is unlikely that any young-of-the-year bull trout could have migrated through this reach. Bull trout in the interior of British Columbia spawn in mid- to late September (Bahr and Shrimpton 2004), and were likely present in these tributaries after our sampling dates. We found a mature male bull trout (fork length 25.8 cm; Figure 5) running at the end of August 2001. It appears likely, therefore, that mature fish exploited habitat the previous year after we had sampled the stream. This indicates that bull trout exploited the habitat only 1 year after passage was provided to the area above the road.

Utilization of reconnected habitat by salmonids occurs between 1 to 5 years (Roni *et al.* 2002). Our finding that spawners exploited newly accessible habitat within 1 year indicates that bull trout will move into new reaches to spawn. We do not know, however, whether bull trout strayed from other streams or had historically spawned in sites between the mainstem river and the road in stream 19.

The restoration of access to habitat above the road in stream 19 where bull trout spawned had a positive effect on bull trout abundance within this stream. Although we captured fish in 1999 below the road in stream 19, the numbers were lower than those observed for stream 26 in 1999. Following culvert replacement, bull trout abundance showed an increase both above and below the road in the subsequent years (Figure 6). In contrast, stream 26 CPUE for bull trout was similar in all 4 years and improving fish passage apparently had little effect.

Estimates of fish species richness that use single-pass electrofishing increase significantly with a decrease in stream width (Meador *et al.* 2003), although single-pass electrofishing may underestimate abundance (Mitro and Zale 2000). As our study examined small, first-order streams (Table 1), we believe that a single-pass sampling design repeated with temporal separation was appropriate for detecting presence or absence, which was the main goal of our study. Fish were not detected above the road in stream 19 on six separate occasions: two conducted before our study, two conducted in 1999 before culvert removal, and two conducted in 2000 after bridge installation. Additionally, our sampling design should also have provided a relative index of abundance. Peterson *et al.* (2004) showed that capture efficiency was low for the first pass (28%) and decreased considerably with successive passes, suggesting that fish responded to the electrofishing procedures. Consequently, the first pass through a reach should provide the best opportunity to collect fish while they are still naive to sampling (Mesa and Schreck 1989). This may provide a better estimate of relative abundance than a depletion approach, especially if some species are more likely to develop avoidance behaviours than others (Edwards *et al.* 2003).

Our work empirically supports models that demonstrate the biggest improvement in salmonid populations can be realized by increasing access to upstream habitats.

Management Implications

The increased fish abundance in stream 19, and particularly the recruitment of young-of-the-year bull trout to the population, indicates that the culvert removal was effective in increasing fish numbers. Our work empirically supports the models developed by Rieman *et al.* (2001) and Wilson (2003) that demonstrate the biggest improvement in salmonid populations can be realized by increasing access to upstream habitats. Available spawner habitat has also been linked to higher estimates of effective population size (N_e), a measurement that reflects genetic drift (Shrimpton and Heath 2003). Rate of loss in genetic diversity depends on N_e rather than actual number of animals in a population (Kalinowski and Waples 2002). The correlation between spawning habitat and N_e argues for the importance of maintaining or creating additional suitable habitat for spawning within river systems. Reconnecting spawning habitat may therefore benefit the population of bull trout in the Torpy River system not only by enhancing numbers but by potentially reducing loss of genetic diversity. Our findings, however, do not mean that all projects to restore or enhance access to habitat will be successful. Further monitoring of restoration projects is recommended to ensure that projects are biologically relevant.

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Test Your Knowledge . . .

Response of bull trout (**Salvelinus confluentus*****) to habitat reconnection through replacement of hanging culverts with bridges***

How well can you recall some of the main messages in the preceding Research Report?
Test your knowledge by answering the following questions. Answers are at the bottom of the page.

1. Spawning bull trout typically home to areas in tributary streams for what?
   A) Sites with appropriate intergravel flow
   B) Sites with appropriate intergravel temperature
   C) Sites with appropriate instream cover
   D) All of the above

2. Bull trout utilize newly accessible habitat within:
   A) 6 months
   B) 1 year
   C) 2 years

3. Iteroparous species of fish, such as bull trout, show spawning site fidelity.
   A) True
   B) False

Answers

1. D 2. B 3. A, but there is often a high rate of straying