TITLE: Evaluation of the removal of impassable barriers on anadromous salmon and steelhead in the Columbia River Basin

RUNNING HEAD: Evaluation of impassable barrier removal projects

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KEYWORDS: Culvert removal, anadromous, habitat restoration, fish passage

ACKNOWLEDGEMENTS: We thank Jonathon Drake, Todd Bennett, Dean Freundlich, Chris Repar for assistance with field work, Scott Blankenship for helpful comments on earlier versions of this manuscript, and Matt Campbell and Mike Ackerman with the Idaho Department of Fish and Game for genetic analysis. Funding was provided in part by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries.

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Article type : Article

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ABSTRACT

Despite the popularity of barrier removal as a habitat restoration technique, there are few studies that evaluate the biological effects of restored stream crossings. An extensive post-treatment study design was used to quantify fish populations (e.g. species, life stage, abundance) and habitat attributes (e.g. gradient, geomorphic channel units) at 32 culvert removal or replacement projects to determine their effectiveness in restoring habitat access for juvenile salmon, Oncorhynchus spp., and steelhead, O. mykiss (Walbaum), in the Columbia River Basin, USA. Anadromous fish [steelhead, Chinook salmon O. tshawytscha (Walbaum)] abundance, juvenile steelhead abundance, and habitat conditions were not significantly different between paired reaches (i.e. upstream and downstream of former barrier sites) suggesting these sites are no longer full barriers to movement. This suggests that barrier removal projects on small Columbia Basin streams provide adequate fish passage, increased habitat availability and increased juvenile anadromous fish abundance immediately upstream of former barriers.

KEYWORDS: anadromous, culvert removal, fish passage, habitat restoration, restoration effectiveness monitoring, Columbia River basin.

INTRODUCTION

The demand for natural resources (e.g. timber and minerals) and increases in transportation and energy infrastructure have resulted in an expansion of road development (Forman, 2003; Laurance et al., 2009; Laurance and Balmford, 2013) that often leads to increased installation of stream-crossing structures (i.e. culverts, low head irrigation dams), with implications on aquatic biota. Effects of culverts and dams at stream crossings on fish have long been reported (McClellan, 1970; Dryden and Jessop, 1974), and the negative impacts of these structures are becoming increasingly apparent (Warren and Pardew, 1998; see Hoffman and Dunham, 2007 for review; Burford et al., 2009; Mahlum et al., 2014). Poorly constructed or functioning culverts can impede movement and become barriers to migration for resident and anadromous fish species (Dynesius and Nilsson, 1994; Fullerton et al., 2010; Perkin and Gido, 2012). Barriers to movement can reduce habitat necessary for fish reproduction, foraging and rearing (Gibson et al., 2005; MacPherson et al., 2012). This loss of ecological connectivity can threaten biodiversity and reduce the overall capacity of a population to maintain trait diversity, which may lead to increased risk of local population extirpations (Wilcove et al., 1998). Movement and migration behaviours are key evolutionary traits that aid in the continued success of resident and anadromous fish populations. Restricting the expression of these traits can reduce survival and reproductive success, and undermine other local restoration actions (Morita and Yamamoto., 2002; Zitek and Schmutz, 2004; Wofford et al., 2005; Bouska and Paukert, 2010).

Installation of culverts is the most common road-building practice for crossing streams. Culverts act as under-road stream pathways whose construction often ignores natural habitat features necessary for fish movements. The U.S. Forest Service and Bureau of Land Management estimate that over 10,000 culverts exist in fish-bearing streams throughout Washington and Oregon as of 2005 (Gibson et al., 2005). Similarly, the Washington Department of Fish and Wildlife (WDFW) estimated that over 7,700 river kilometres (rkm) of historical salmon habitat are blocked by approximately 2,400 impassible culverts (Conroy, 1997). Therefore, removal of human-constructed barriers to fish migration and passage is a primary means of restoring salmon and other anadromous fish populations, and is a commonly used, cost-effective restoration technique (Bryant et al., 1999; Roni et al., 2002, 2008; Burdick and Hightower, 2006; Kiffney et al., 2009).

Efforts to eliminate barriers to fish migration and restore longitudinal connectivity by replacing or removing culverts have been a focus of stream restoration actions over the last two decades, resulting in the reconnection of large amounts of stream habitat (NOAA 2016). Along the West Coast, nearly 15,000 km of fish habitat have been made accessible through removal of barriers since 2000 (NOAA 2016). Approximately 3,500 culverts that were barriers to fish passage were replaced in Washington State alone, resulting in 5,990 km of newly accessible migratory fish habitat. In the Columbia River basin, the Northwest Power and Conservation Council (NWPCC) Fish and Wildlife (F&W) Program implemented by the Bonneville Power Administration (BPA) and its partners have opened nearly 4,000 km of habitat that was previously blocked to fish passage (Governor's Salmon Recovery Office, 2008). Surprisingly, a relatively small number of biological evaluations of culvert replacements (retrofits, removals, or replaced with other stream crossing structures) have been published despite decades of recommendations from scientists for improved monitoring and evaluation of barrier removal and other restoration activities (Tarzwell, 1937; Reeves and Roelofs, 1982; Roni et al., 2002, 2013; Rumps et al., 2007). For the purpose of this manuscript an impassable barrier culvert that is replaced with a passable culvert is considered removal of the fish migration barrier.

Through the NWPCC F&W Program, more than 100 impassable (i.e. complete or full) barrier removal projects have been funded since 2000. To evaluate the success of these barrier removal projects, 32 interior Columbia Basin full barrier removal sites with a range of 1–14 years post project completion were surveyed. The main objective was to determine the effectiveness of full barrier removal to restore habitat connectivity. It was hypothesised that (i) post-barrier-removal treatment and reference reaches would not differ in juvenile salmon or steelhead abundance, (ii) a positive relationship exists between a) the abundance of juvenile salmon upstream of the barrier relative to downstream of the barrier, and b) time since barrier removal, and (iii) the abundance of juvenile salmon and steelhead trout upstream relative to downstream of the barrier is related to habitat quality upstream of the barrier.

METHODS

An extensive post-treatment (EPT) design comparing the reach immediately upstream (treatment) and downstream (reference) of the barrier were used to assess the effectiveness of full barrier removals (culvert replacement or removal) implemented prior to 2013 in providing passage to anadromous fish (Hicks et al., 1991). Assessment of full barriers to fish passage do not require collection of pre-project data, as anadromous fish were not found above full barriers prior to removal. The EPT design samples paired treatment and reference reaches after (> 1 year) the habitat improvement has occurred (post-treatment). Because the EPT design samples paired reaches, a site comprises a treatment and reference reach. Juvenile salmonids can rapidly recolonise areas upstream of former barriers in one or two years (Anderson and Quinn, 2007;

Pess et al., 2014; Anderson et al., 2015). Based on office screening and site visits of an initial list of more than 100 potential full-barrier projects completed since 2000, 43 potential sites were identified for sampling. To qualify for inclusion in the programme, suitable treatment and reference reaches immediately above and below the former impassable barrier were required. Thus, many sites were excluded because paired reaches did not have similar (within ~10%) channel gradient, channel width, valley confinement (valley width), flow regime, land use, or riparian cover or barrier removal coincided with other restorative actions (e.g. in-stream large wood placement). Of those 43 potential sites, two sites in 2014 and nine sites in 2015 were not sampled due to fire, landowners denying access, or lack of water (dry channel). For the remaining 32 sites, 18 were sampled in 2014 and 14 were sampled during 2015 (Figure 1). Treatment and reference reach lengths were ten times the bankfull width or a minimum length of 50 m. Of the 32 sites, barriers were completely removed at 12 of the sites. The remaining 20 culverts were constructed of corrugated steel (12 bottomless pipe arch and eight countersunk squash) (Table 1).

To account for potential differences in abundance above and below barrier sites that may be related to habitat quality rather than movement restriction habitat data for both treatment and reference reaches were collected and quantified. A modified thalweg profile and habitat survey methods from Mossop and Bradford (2006) were used to classify habitat type (pool, riffle, glide), slope, substrate type (organics, silt, sand, gravel, cobble, or boulder), and residual pool depth (Lisle, 1987). Wolman pebble counts were used to characterise median particle size in treatment and reference reaches (Wolman, 1954). From data collected in longitudinal profiles, the average stream channel gradient and the proportion of pools (area and length) in both the reference and treatment reaches were calculated (Table 2). The 16th, 50th, and 84th percentiles of the streambed particle size were identified from data collected in pebble counts . To determine whether a project met fish passage criteria, an assessment of the completed project was conducted, including both qualitative and quantitative measures (e.g. channel slope, crossing length, high water depth) following WDFW barrier survey protocols (Recreation and Conservation Office, 2013; Washington Department of Fish and Wildlife, 2009; Table 1). According to the WDFW barrier survey protocols, if the water surface drop is greater than or equal to 1 m, 0% passability is assumed at the culvert. If a culvert is less than 18.3 m long and the slope is greater than 4%, then 0% passability is assumed. If a culvert is greater than 18.3 m long and the slope is greater than 2% then 0% passability is assumed (see WDFW 2009 for details).

Three-pass backpack electric fishing was used to quantify the number of juvenile fish in each treatment and reference reach during the summer low flow period (July - September) at all but two sites where the flow was too deep and wide to use backpack electric fishing methods. Block nets were placed at the upstream and downstream ends of each reach to prevent fish from moving in or out of study reaches during electrofishing. Captured fish were anesthetised, identified to species, measured and placed in a recovery bucket in the stream (after all backpack electric fishing was completed) for at least 15 min before fish were released into the same reach where they were captured. Fish abundance in each reach was estimated for each species using a multiple removal estimator (Carle and Strub, 1978) (Table 3). Because they were too deep and wide to effectively electrofish and therefore accurately quantify juvenile salmonids with multiple-removal electrofishing, snorkel surveys were used to enumerate fish at two sites in 2015 (Taneum and Reecer sites). Divers entered the downstream end of a reach and slowly moved upstream, stopping to occasionally report the numbers and sizes of all fish species (Roni and Fayram, 2000). Fish density was calculated by dividing multiple removal estimates or snorkel counts of fish abundance by total wetted area of a reach.

The response and analysis for an EPT design is based on the difference in fish abundance or habitat metrics between paired treatment and reference reaches. Paired t-tests were used to compare the difference in means between the treatment and reference reaches on untransformed data, as the data were approximately normally distributed. Because Chinook and coho salmon were rarely encountered at most sites (four and one respectively), analysis was done on steelhead and all anadromous fish combined. Resident cutthroat and brook trout were common at sites and true resident fish were likely above and below the impassable barrier. Therefore, resident fish abundance was reported to demonstrate that other fish were present and to demonstrate the quality of habitat, although resident fish were not included in the analysis to determine success of the barrier removal to passing fish due to the likelihood that resident fish were present upstream of the impassable barrier before barrier removal. Pearson correlation analyses were used to determine whether there was a significant correlation between differences in fish numbers in treatment and reference reaches and project age, or differences between treatment and reference reaches in gradient and percent pool. A significance level of $\alpha = 0.05$ was used for all statistical tests.

RESULTS

All barrier removal sites that had culverts (20 of 32) appeared to be completely passable based on the slope and drop, and other fish passage design criteria (WDFW 2009) (Table 1). For the remaining 12 barrier removal sites the culvert or former barrier was completely removed and replaced with a natural stream channel. Steelhead, O. mykiss, cutthroat trout, O. clarki (Richardson), Chinook salmon, coho salmon, brook trout, Salvelinus fontinalis (Mitchill), bull trout, S. confluentus (Suckley), sculpin (Cottus spp.), suckers (Catostomus spp.) and dace (Rhinichthys spp.) were encountered. Steelhead were present in 20 of 32 sites sampled. No fish were detected at Parachute Creek (Lochsa River Basin) or Little Camas Creek (Wenatchee River Basin), and data from these creeks were excluded from the analysis (Table 3). The most common salmonid species captured were O. mykiss (N = 766), and cutthroat trout (N = 713). There was no significant difference in fish density between treatment and reference reaches for either O. mykiss (t-statistic: -0.79, df = 21, p = 0.44) or all anadromous salmonids combined (t-statistic: -5.45, df = 21, p = 0.19; Table 3).

Average wetted width (t-statistic: -0.98, df = 31, p = 0.34), proportion of pools (t-statistic: -0.03, df = 31, p = 0.98) and gradient (t-statistic: 1.04, df = 31, p = 0.30) did not differ between treatment and reference reaches (Table 2). The average slope within culverts was 3.33%, and the average gradient for reference and treatment reaches was 4.35% and 4.60%, respectively (Tables 1 and 2). Substrate characteristics (D50) did not differ between reference and treatment sites (t-statistic: -0.08, df = 31, p = 0.94; Table 2). Pearson correlation analysis yielded little evidence of a linear relationship between differences in density (O. mykiss, all anadromous fish combined) above and below former barriers and project age, difference in gradient, or percent pool habitat (p > 0.30).

DISCUSSION

Results suggest that removal of full barriers has allowed anadromous salmonids to access suitable habitat above former man-made barriers that were impassible to anadromous fish passage. Given that these sites were full barriers one would expect anadromous fish below the barrier but not above, thus abundance between reaches below and above the barrier would be different. Little difference was found in fish numbers above and below former barriers indicating the removal of the barrier has resulted in conditions that do not block movement or migration of anadromous fish. The habitat and culvert surveys indicated similar habitat quality and quantity immediately above and below former barriers. This is consistent with previous studies, which found that salmonid fishes quickly recolonise habitat once a barrier to migration was removed (Roni et al., 2008; Anderson, 2011; Pess et al., 2014; Erkinaro et al., 2017). In addition, despite some barrier removal projects being more than 10 years old, the current fish and culvert surveys suggest that most sites are passable to anadromous fish based on regional fish passage criteria (WDFW, 2009) (Table 1 & 3).

The absence of anadromous salmonids at some of the sites sampled suggests that some sites get intermittent or no use by these species. One of the key criteria in the site selection was that these sites were used by steelhead and Chinook salmon. However, juvenile Chinook salmon were encountered at only four sites. It is possible that Chinook salmon only intermittently use the sites or that escapement levels were low, although detailed escapement or redd survey data for most study streams was not available. Chinook salmon typically spawn and rear in larger streams and rivers (> 5 m bankfull width), although juveniles may move into tributaries to rear for one to two years before migrating to sea (Quinn, 2005). Given the small size of the streams sampled, it is more likely they are rarely used by Chinook salmon for spawning or rearing, or Chinook are only found well downstream of the sites. Moreover, seven sites exhibited zero anadromous fish and zero fish were observed at two sites (Little Camas and Parachute), suggesting that some of these sites might not be used or only intermittently used by Chinook salmon and steelhead. It is also possible that some of the steelhead captured during the surveys were resident rainbow trout. Tissue samples from O. mykiss collected above and below barriers from 19 of the sites suggested little hybridization with cutthroat trout. Moreover, sibship analysis indicated that juvenile steelhead which were members of the same families (siblings) were found above and below former impassible culverts at 15 of 19 sites (Roni et al., 2014). This suggests upstream and downstream movement of juvenile or adult steelhead or resident O. mykiss through formerly impassible culverts providing further evidence that these former barriers are now passable.

There were little to no differences in physical habitat above and below former barriers at most sites (Table 2). This is not unexpected as one of the criteria for selecting a site was that there could not be dramatic differences (e.g. greater than ~10%) in gradient, flow and other channel characteristics that might limit upstream fish movement. The age or time since restoration was completed is a common factor believed to affect the success of the project (successful passage in this case) (Roni, 2002; Kail et al., 2015). This is particularly the case for road crossings such as culverts that have a fixed lifespan and can become impassable with time if the channel aggrades or degrades (Wilhere et al., 2016). Sites sampled ranged in age from 1 to 14 years after project completion (Table 2), but no correlation was found between the difference in fish numbers above and below barriers and the age of the project. While 14 years is a considerable length of time, it is still early in the life of structures such as culverts and different results may have been found had older structures been sampled. Sites with culverts appeared to be meeting WDFW fish passage criteria. This is likely because all of these sites were bottomless culverts with natural stream bottom, which have been shown to have higher rate of compliance with WDFW fish passage criteria than other types of culverts (Price et al., 2010).

With thousands of impassible culverts and other man-made barriers on streams in western North America (Conroy, 1997; Gibson et al., 2005; Price et al., 2010), prioritising projects that will have the biggest benefit is important for wise use of resources. In the Columbia Basin, barrier removal projects that will allow access to and increase habitat for threatened and endangered Chinook salmon and steelhead are given the highest priority for funding. The results from 32 barrier removal projects implemented to benefit Chinook salmon and steelhead suggest that many of these sites are infrequently used by Chinook salmon. This suggests that more detailed surveys of fish use prior to funding and implementation of projects are needed to help prioritise projects. That is not to say that only fish use should be considered when prioritizing culverts and other man-made barriers for removal or replacement. Culverts and other barriers can impede movement of aquatic invertebrates, sediment, large wood, organic material and nutrients (Ward and Stanford 1983, Essington and Carpenter, 2000, Mueller et al., 2011). These factors, coupled with information on fish use and the quality and quantity of newly available habitat, should be considered when prioritising culverts and other man-made barrier projects for removal.

Despite the popularity of barrier removal as a habitat restoration technique, studies evaluating the biological effects of restored stream crossings are rare (Roni et al., 2008). While all barrier removal projects in the Columbia River Basin were not evaluated, the sites examined are representative of the few hundred barrier removal projects funded since 2000 in the Columbia Basin as part of the BPA’s Fish and Wildlife Program. Thus, the results indicate that barrier removal projects in the Columbia Basin provide adequate fish passage and result in increased habitat availability and anadromous fish population abundance immediately upstream of former barriers. Moreover, this provides evidence that addressing impassable barriers to salmon and steelhead migration can result in range expansion and increased available habitat for anadromous fishes and resident fishes. Some sites exhibited greater abundances of fish below the barrier than above the barrier (e.g. Badger Site 1), which may be due to the stream crossing, but could also be due to upstream colonisation rates. Kiffney et al. (2009) suggested that three factors influence rapid and successful natural colonisation after a barrier removal: 1) sufficient source population below the barrier; 2) high-quality habitat above the barrier; and 3) relatively low densities of resident fish upstream of the barrier. Taking these factors into account and understanding constraints on local biotic production can help prioritise barriers for removal and help determine whether they will be successful in the short and long term (Pess et al., 2014).

ACKNOWLEDGMENTS

We thank Jonathon Drake, Todd Bennett, Dean Freundlich, Chris Repar for assistance with field work, Scott Blankenship for helpful comments on earlier versions of this manuscript, and Matt Campbell and Mike Ackerman with the Idaho Department of Fish and Game for genetic analysis. Funding was provided in part by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries.

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Table 1. Culvert characteristics including shape, length, span (width of culvert), water depth within the culvert, slope through the culvert crossing, and road fill and width based on culvert surveys using WDFW culvert survey protocol. At sites that are not listed, the barrier or culvert was completely removed; thus, no culvert survey was possible. SQSH = squash, ARCH = bottomless pipe arch. Span indicates length of culvert or stream crossing. Drop or culvert height from water surface to bottom of downstream end of culvert is also recorded but was zero at all sites.

Site	Shape	Length (m)	Span (m)	Depth (m)	Slope (%)	Road fill (m)	Road width (m)
Badger Lower	SQSH	18.8	3.4	0.10	3.1	1.9	8.1
Cabin	ARCH	43.9	4.5	0.13	5.9	10.3	9.6
Camp	SQSH	14.5	4.2	0.12	3.6	1.0	7.6
Corral	SQSH	12.1	2.6	0.10	2.4	1.5	9.0
Doe	SQSH	20.1	3.4	0.20	4.5	1.9	8.3
Granite	SQSH	8.3	4.4	0.22	7.5	0.9	4.9
Hepner	ARCH	14.8	4.1	0.22	1.8	0.8	7.3
Jack	ARCH	23.6	6.3	0.09	1.4	1.6	11.6
Mare	ARCH	15.6	3.9	0.18	3.0	2.2	7.0
Mertin	ARCH	18.2	4.0	0.15	4.9	1.9	9.7

Mule	ARCH	19.6	4.5	0.15	1.7	4.1	7.5
Parachute	SQSH	20.9	2.6	0.10	4.6	3.1	6.8
Wendover Lower	SQSH	17.2	3.0	0.11	1.3	1.5	9.0
Wendover Upper	SQSH	14.7	2.2	0.04	2.3	1.7	7.0
Ireland Gulch	ARCH	7.6	3.6	0.11	9.5	1.2	4.1
Little Camas	ARCH	9.7	3.6	0.05	0.8	0.9	4.6
Orr Creek	ARCH	12.3	4.1	0.26	6.7	1.1	4.9
Sand 1	ARCH	9.4	3.9	0.10	2.6	1.3	4.5
Sand 2	ARCH	10.2	3.3	0.10	2.4	0.9	4.8
Tillicum	ARCH	18.6	5.3	0.13	4.8	2.7	6.3

Table 2. Habitat metrics for each site. Average wetted width, average slope, percent pool and substrate shown separately for reference (R) and treatment (T) reaches. cfs = cubic feet per second.

Site	Year barrier removed	Flow (cfs)	Wetted width (m)		Slope (%)		D50 (mm)		Pools (%)	
			R	T	R	T	R	T	R	T
Antoine	2013	0.58	1.72	1.90	8.92	10.68	57.5	56	8.63	3.6
Badger Lower	2002	3.23	3.56	3.34	2.77	1.82	113	76.5	4.7	24.4
Badger Upper	2005	3.66	2.37	2.62	3.34	2.65	68.5	47.5	0	14.86
Cabin	2006	2.05	3.87	2.94	3.98	7.77	75	102.5	9.66	25.57
Camp	2006	4.49	3.55	3.23	3.03	4.35	24	17	11.36	15.61
Corral	2004	1.45	2.49	2.17	2.70	4.03	22.5	19	24.27	15.26
Doe	2008	4.18	3.95	4.59	3.49	4.45	60.5	48	9.62	11.92
Dead Cow Gulch	2007	2.27	1.48	1.30	7.07	4.67	55.5	63	3.62	7.06
Granite	2006	0.75	1.25	1.32	10.54	8.97	7	10	0	3.77
Hepner	2008	2.62	4.04	3.70	0.71	2.36	11	48.5	20.72	0
Indian N.F.T.	2009	2.25	2.13	2.31	2.98	2.33	31	22	27.69	49.53
Jack	2009	0.72	3.16	2.36	3.87	4.36	51.5	50.5	40.69	19.14
Mare	2007	0.69	2.32	2.50	2.05	1.47	22	15	5.01	10.61
Mertin	2008	1.35	2.76	1.96	3.68	2.27	3.5	3	4.66	11.34
Mule	2008	0.97	2.65	3.38	1.78	1.94	52	46.5	5.55	0
Parachute	2005	0.78	2.51	1.80	7.39	8.23	60.5	91.5	9.61	5.44
Wendover L.	2000	1.38	2.45	3.30	4.71	5.44	91.5	115.5	18.89	9.46

Wendover U.	2002	0.37	1.89	1.93	5.15	6.69	62	70	8.98	10.33
Butler	2014	< 0.1	2.10	1.56	3.37	3.78	114	70.5	19	17
Cahail	2010	< 0.1	1.76	2.25	2.69	3.35	19	39	61	53
Indian W.S.	2012	0.17	1.76	2.59	6.51	3.87	69	68	25	10
Ireland Gulch	2014	1.47	1.85	2.04	8.25	8.40	55	54	42	10
Jenkins	2013	< 0.1	1.73	1.72	2.57	1.67	15	19	44	46
Little Camas	2004	0.17	1.57	1.41	4.29	5.26	21.5	26	7	19
North Road	2009	2.02	3.71	3.25	3.08	2.19	113	66	27	5
Orr	2012	5.14	2.32	2.48	12.62	13.48	86	68.5	24	13
Reecer	2009	28.9	6.88	6.56	2.02	1.57	3	14	0	0
Sand 1	2004	0.54	2.27	2.37	3.80	4.70	107.5	40	0	13
Sand 2	2004	0.54	2.43	2.11	1.61	1.99	32	154	28	26
Taenum	2007	9.21	8.09	7.05	1.26	0.79	60.5	57	41	28
Tillicum	2013	2.71	2.60	2.69	4.11	5.08	52	43	0	44
Whitney	2011	2.75	2.76	2.55	4.99	6.57	64	73	22	46

Table 3. Juvenile salmonid abundance by species for treatment (T) and reference (R) reaches estimated from three-pass electric fishing. The combined column is the total count of the anadromous salmonids.

	Steelhead		Coho		Chinook		Combined		Cutthroat		Brook Trout	
Site	R	T	R	T	R	T	R	T	R	T	R	T
Antoine	7	6	0	0	0	0	7	6	0	0	29	27
Badger Lower	17	4	0	0	0	0	17	4	7	14	0	0
Badger Upper	0	0	0	0	0	0	0	0	95	44	0	0
Cabin	0	0	0	0	0	0	0	0	24	20	0	0
Camp	1	0	0	0	0	0	1	0	13	26	0	0
Corral	8	5	0	0	0	0	8	5	18	36	0	0
Doe	0	0	0	0	0	0	0	0	29	22	0	0
Dead Cow Gulch	21	23	0	0	17	18	38	41	0	0	0	0
Granite	6	6	0	0	0	0	6	6	0	0	0	0
Hepner	1	1	0	0	0	0	1	1	25	44	2	0
Indian	14	22	0	0	0	0	14	22	1	1	3	3
Jack	39	24	0	0	0	0	39	24	0	0	23	9
Mare	0	0	0	0	0	0	0	0	19	60	0	0
Mertin	0	0	0	0	0	0	0	0	14	22	0	0

Mule	0	0	0	0	0	0	0	0	53	42	0	0
Parachute	0	0	0	0	0	0	0	0	0	0	0	0
Wendover Lower	14	5	0	0	1	0	15	5	6	19	0	0
Wendover Upper	0	0	0	0	0	0	0	0	25	34	0	0
Butler	12	14	0	0	0	0	12	14	0	0	0	0
Cahail	27	56	0	0	0	0	27	56	0	0	44	49
Indian 2015	4	5	0	0	40	11	44	16	0	0	0	0
Ireland Gulch	3	3	0	0	0	0	3	3	0	0	0	0
Jenkins	7	3	0	0	0	0	7	3	0	0	0	0
Little Camas	0	0	0	0	0	0	0	0	0	0	0	0
North Road	68	46	209	157	17	15	294	218	0	0	1	0
Orr	0	0	0	0	0	0	0	0	0	0	10	14
Reecer	0	1	0	0	0	0	0	1	0	0	0	0
Sand 1	17	22	0	0	0	0	17	22	0	0	0	0
Sand 2	29	19	0	0	0	0	29	19	0	0	0	0
Taenum	22	28	0	0	0	0	22	28	0	0	0	0
Tillicum	42	22	0	0	0	0	42	22	0	0	0	0
Whitney	44	48	0	0	0	0	44	48	0	0	0	0

FIGURE LEGENDS

Figure 1. Location of full-barrier removal projects sampled in 2014 and 2015.