INFORMATION REPORT

NUMBER 2015-02

FISH DIVISION

Oregon Department of Fish and Wildlife

MONITORING SALMONID OCCUPANCY USING CATARAFT ELECTROFISHING IN THE UPPER DESCHUTES RIVER

This program receives federal financial assistance in Sport Fish and/or Wildlife Restoration and prohibits discrimination on the basis of race, color, national origin, age, sex or disability. If you believe that you have been discriminated against as described above in any program, activity, or facility, or if you desire further information, please contact ADA Coordinator, Oregon Department of Fish and Wildlife, 3406 Cherry Avenue. NE, Salem, OR, 97303, 503-947-6000 or write Office for Human Resources, U.S. Fish and Wildlife Service, Department of the Interior, Washington, D.C. 20240.

This material will be furnished in alternate format for people with disabilities if needed. Please call (541) 757-4263 to request.

MONITORING SALMONID OCCUPANCY USING CATARAFT ELECTROFISHING IN THE UPPER DESCHUTES RIVER

Steven Starcevich[1]
Nancy Doran[2]
Ryan Carrasco[2]

Oregon Department of Fish & Wildlife
Native Fish Investigations Program[1]
Corvallis, Oregon

Deschutes Watershed District[2]
Bend, Oregon

Table of Contents

List of Tables

List of Figures

MONITORING SALMONID OCCUPANCY USING CATARAFT ELECTROFISHING IN THE UPPER DESCHUTES RIVER

STEVEN STARCEVICH[1], NANCY DORAN[2], AND RYAN CARRASCO[2]

Oregon Department of Fish and Wildlife
[1]Native Fish Investigations Program – 28655 Highway 34, Corvallis, Oregon 97333
[2]Deschutes Watershed District – 61374 Parrell Road, Bend OR, 97702

Abstract – The upper Deschutes River has experienced dramatic changes to its fish assemblage, flow regime, and the quality and connectivity of its habitat since large water storage dams were constructed and flow was altered for irrigation and other human needs. These changes, which extirpated native bull trout, have also led to concerns about the status of the native redband trout in the upper Deschutes River and highlighted the need for more information about this population to inform management. To determine current status of this redband trout population, we used cataraft electrofishing surveys to conduct an occupancy study in the upper Deschutes River. Our objectives were to 1) determine baseline distribution and relative abundance of salmonids, 2) evaluate the biotic and abiotic factors influencing redband trout distribution and abundance, and 3) develop a long-term monitoring protocol to track population response to management actions. In 2012 and 2013, 21 sites (200 m long) were sampled with repeated visits in two seasons, which were defined as the low-discharge water storage season (November 1 to March 30) and high-discharge irrigation season (April 1 to October 31). Redband trout, brown trout, and mountain whitefish were distributed throughout the entire study area. Mountain whitefish were the most abundant by several-fold. Nonnative brown trout were relatively more abundant than redband trout. Large redband trout (>250 mm total length) were relatively rare, averaging less than one fish per site, while large brown trout averaged about eight fish per site. Among redband trout, hatchery-stocked fish constituted a substantial proportion of the large size-class. The occupancy probability of large brown trout (ψ=0.85) was double that of large redband trout (ψ=0.42). The weak positive correlation between maximum site counts of these two species provides no evidence of competitive exclusion of redband trout. Low detection probability of the large size-class of both trout species and imprecise occupancy and relative abundance estimates suggest improvements are needed in the sampling protocol prior to its use as a long-term monitoring tool. We suggest continued development of the monitoring protocol by devising a test of the closure assumption in large river habitats, evaluation of other monitoring techniques that do not require cataraft electrofishing, and exploration of how the regulated flow regime affects habitat and the salmonid population.

The upper Deschutes River flows north toward the Columbia River through the high elevation, semi-arid plateau in central Oregon. The river is fed by large groundwater springs and was historically recognized for the extraordinary steadiness of its intra- and inter-annual flows (Gannett et al. 2003). During the past century, three large water storage dams were constructed in the upper basin: Crescent Lake Dam (1922), Crane Prairie Dam (1940), and Wickiup Dam (1949). The operation of these dams provides water for irrigation of agricultural lands throughout the Deschutes River basin. As a result of flow management at the dams, and irrigation withdrawals throughout the basin, the hydrology of the Deschutes River has been significantly altered and the upper Deschutes River is now characterized by extremely low flows during the winter (when water is stored in reservoirs) and extremely high flows during the

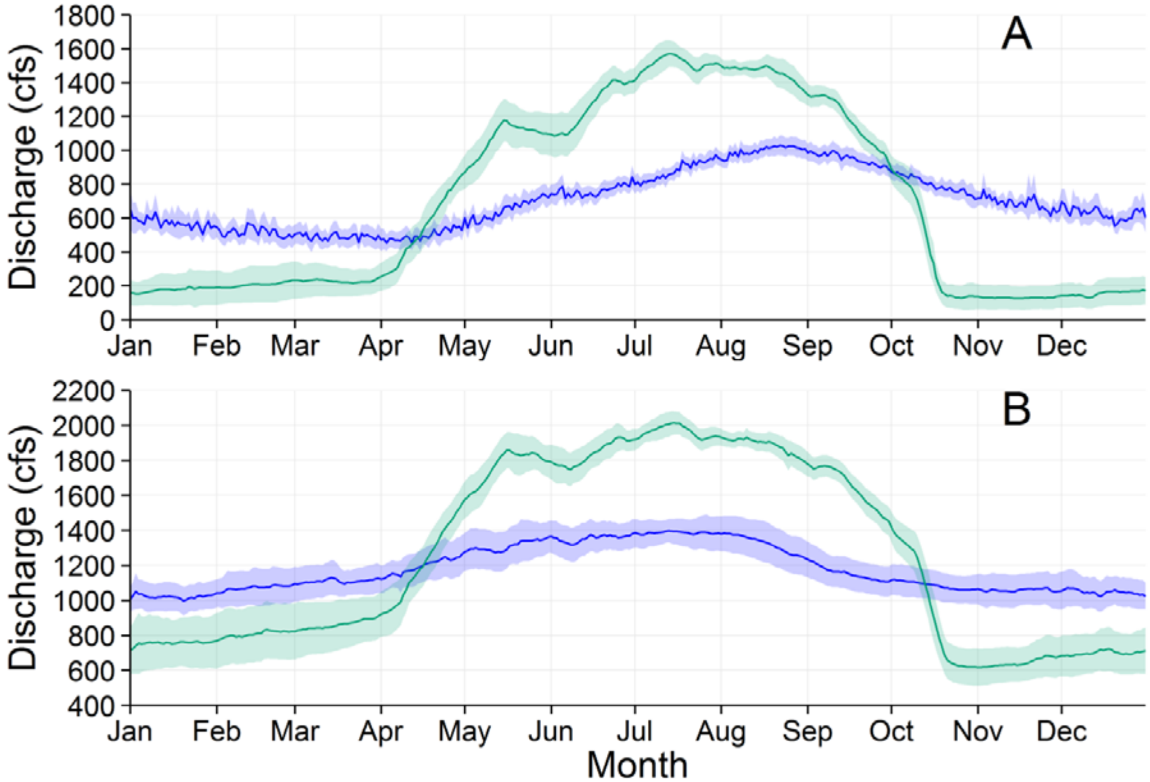

Figure 1. Hydrographs of the historical/unregulated (blue) and regulated (green) mean daily discharge and 95% confidence intervals (shaded ribbon) of the Deschutes River just below Wickiup Dam (A) and at Benham Falls (B). Unregulated flows below Wickiup Dam were summarized from gauged flows recorded prior to the construction of Wickiup and Crane Prairie dams (i.e., 1924-1940); those from Benham Falls, were a combination of historical flow data (1938-1940) and an estimate of unregulated mean daily flow (1983-2014) provided by the U.S. Bureau of Reclamation [BOR]. Regulated mean discharge was calculated for data from 1990 to 2014. All hydrograph data were obtained from the BOR Hydromet website (www.usbr.gov/pn/hydromet/arcread.html).

spring and summer months (when water is released from the reservoirs for use by irrigators) (Figure 1). Prior to dam construction and water management, natural winter flows were 400-700 cfs just downstream of where Wickiup Dam was constructed. Regulated mean daily flows during December and January since 1990 have averaged 143 cfs, with minimum flows dropping to as low as 3 cfs on individual days and under 50 cfs for several consecutive months (Figure 1). Natural summer mean daily flows averaged 900 cfs (range, 717-1030 cfs); while regulated mean daily flows since 1990 have increased almost 40%, averaging 1350 cfs (range, 925-1570 cfs).

This change in the upper Deschutes River hydrograph from steady discharge to one of managed extremes has altered the riverscape, water quality, and fish populations (NPCC 2004). The riparian zone has been significantly altered by loss of historically abundant riparian vegetation, including once extensive wet meadows and forested wetlands in low gradient reaches, caused by freezing and thawing of exposed river beds and riparian areas during low winter flows and scouring during high flows in spring and summer (NPCC 2004). The current flow regime hinders the re-establishment of vegetation along the river and thus prevents the primary natural means of channel stabilization (USDA 1996). Bank erosion has increased in many locations (Yake 2003). A photographic comparison of the Deschutes River from Benham

Falls up to Wickiup Dam suggests that, from 1943 to 1991, the river has widened 20% and the number of meander cutoffs has increased six-fold (USDA 1996). Large wood was removed in the early 1900s from the river segment between Benham Falls and Pringle Falls to aid in the transportation of saw logs to mills; this has exacerbated bank erosion and reduced fish habitat availability (USDA 1996). The controlled flow regime leads to water quality problems like turbidity from bank erosion during high flows and algal production in Wickiup Reservoir (USDA 1996) and severe ice formation during winter low flows (NPCC 2004). Some of the effects of the water management regime become less apparent as water from Fall River, Spring River, and the Little Deschutes River join the upper Deschutes River and the channel becomes constrained by volcanic geology (NPCC 2004).

The salmonids indigenous to the upper Deschutes River are redband trout (*Oncorhynchus mykiss gairdnerii*), bull trout (*Salvelinus confluentus*), and mountain whitefish (*Prosopium williamsoni*) (Fies et al. 1996). Historical information on the relative abundance of these fishes is mainly limited to anecdotes. These anecdotes and the historical presence of high-quality habitat suggest that these species were highly abundant. For example, in a Bend Bulletin article published in 1907, over 3,000 trout from the upper Deschutes River were caught by four anglers over four days for the annual July 4th barbecue in Bend. The Bend Bulletin reported in 1903 that large numbers of bull trout migrating in July and August toward upstream spawning areas were harvested at the base of Pringle Falls, which acted like a natural fish trap (Fies et al. 1996). Large bull trout runs were reported at Pringle Falls as late as 1923 (Fies et al. 1996). The upper Deschutes River basin became an important recreational trout fishery for residents and visitors alike and an important contributor to the local economy (Fies et al. 1996)

Local biologists now believe human-caused changes in the upper Deschutes River have dramatically altered trout populations (Fies et al. 1996, NPCC 2004). These changes and their effects include the following: construction of dams that block fish movement, migration, and access to spawning grounds (Fies et al. 1996) and restrict sediment and organic matter transport needed for spawning gravels and habitat complexity (Ligon et al. 1995); water management that reduces food and habitat availability and causes direct fish mortality by stranding fish in dewatered side channels and by instream ice formation (Fies et al. 1996, NPCC 2004); hatchery stocking of rainbow trout (*O. mykiss*) that may compete or hybridize with native redband trout; and the introduction of brown trout (*Salmo trutta*), which can have a competitive advantage over other trout species (Fausch and White 1981; Shirvell and Dungey 1983; Wang and White 1994; McHugh and Budy 2005) and may be favored by the water management regime in the upper Deschutes River (NPCC 2004). These changes led to the extirpation of bull trout in the 1950s and a perceived decline in redband trout abundance, especially in the reach between Benham Falls and Wickiup Dam (Fies et al. 1996), where effects of the managed flows are at their most extreme. Boat electrofishing surveys in this river section in the 1960s and 1990s suggested mountain whitefish were highly abundant, there were relatively large numbers of brown trout, and there was a small redband trout population comprised mainly of hatchery fish (Fies et al. 1996).

The decline in redband trout abundance has led to management concern about the status of this population and has highlighted the need for data to accurately assess status and monitor the population response to water management, restoration, and other management actions. To address these needs, the Deschutes Watershed District of the Oregon Department of Fish and Wildlife [ODFW], in collaboration with the Upper Deschutes Watershed Council [UDWC] and Mitigation and Enhancement Board [M & E] have conducted a field study designed to obtain baseline information on the current status of native and non-native salmonids and begin development of a monitoring protocol that will enable managers to monitor fish populations and guide research and management activities in the upper Deschutes River. The specific objectives were the following:

1) Determine current distribution and relative abundance of three size classes of redband trout, mountain whitefish, and brown trout.

2) Evaluate the relationship between trout detection/occupancy and discharge, temperature, non-native fish presence, and sampling timing.

3) Develop a long-term monitoring protocol by testing sampling designs and timing for feasibility and effectiveness.

Study area

The Deschutes River headwaters emanate from the eastern slope of the Cascade Mountains, in an area that receives on average 254 cm of precipitation each year, mostly as snow. The high elevation forest community of western hemlock (*Tsuga heterophylla*), and alpine and subalpine plant species transitions in the mid-elevations to forest community dominated by lodgepole (*Pinus contorta*) and Ponderosa (*P. ponderosa*) pines. Around the city of Bend, the river enters the Cascade rain shadow and the semi-arid continental climate of the high desert plateau, which is characterized by the sagebrush steppe plant community (NPCC 2004). Riparian vegetation is dominated by Ponderosa pine and lodgepole pine, willow thickets, and sedge meadows (Fies et al. 1996). We defined the upper Deschutes River as the 100 km between Wickiup Dam (river km [RK] 365) and the city of Bend (RK 266). Construction of Wickiup Dam was finished in 1949 and provided no means of upstream fish passage (Figure 2). There are three major tributaries that enter the Deschutes River within this reach: Fall River (annual mean daily discharge, 150 cfs; RK 330), Little Deschutes River (385 cfs; RK 311), and Spring River (150 cfs, RK 306). Maximum water temperatures range from 10-18˚ C in summer and 0-7˚ C in winter.

We identified two study segments. One river segment, between Wickiup Dam and the Little Deschutes River confluence, is sinuous and low gradient, except at Pringle Falls (RK 349), which may be a fish passage barrier at low flows. The second segment, from the Little Deschutes River to Bend, the river runs downstream through basalt formations starting at Benham Falls, which results in a series of falls and cascades. In this section, the river splits into two channels around Lava Island, one of which is dewatered when flows are reduced for water storage at Wickiup Dam. Benham Falls (RK 291), Dillon Falls (RK 286) and Lava Island Falls (RK 281) may be barriers to upstream movement by fish during certain flows. Within the upper Deschutes River, we identified river sections that were accessible to the electrofishing cataraft and for which access was granted by private landowners. This resulted in a discontinuous study area, from which 21 study sites were randomly selected using ArcGIS (Figure 2).

Methods

Fish sampling – Fish were captured using a 14-foot cataraft equipped with a Smith-Root 2.5 GPP Electrofisher with 32-inch array droppers. We sampled with one rower and two netters at the bow of the raft. The electrofishing unit was set for direct current (DC) with a pulse rate of 120 pulses/s and 60% power. Sample sites were 200 m in length. Up to five sampling transects, consisting of longitudinal downstream passes of the cataraft electrofisher through the site, were conducted. The number of transects at each site depended on the wetted width (i.e., wider sites were sampled with more transects) and the feasibility of getting the cataraft back to the upstream end of the site for subsequent transects. Captured fish were held in a live well until the final transect for each site was completed. Fish were identified to species and measured for total length [mm TL]. Fish ≥150 mm TL were weighed to the nearest gram. All fish were then released at the end of the site. In 2012, the number of transects ranged from 1 to 5. We sampled for fish at 11 sites during the

Figure 2. Sample sites (yellow circles) and major falls (red circles) and city limits (grey outline) on the upper Deschutes River.

irrigation season (September 3-25) and 19 sites during the water storage season. (October 29 – December 13). The mean electrofishing time per transect was 160 s during the irrigation season and 235 s during the water storage season. In 2013, we sampled fish at 21 sites during the irrigation season (August 5 ¬– October 3) and at 19 sites during the water storage season (16 October–7 November).

Data analysis – To obtain baseline data on salmonid distribution, relative abundance, and the effectiveness of the capture method, we used a single-season occupancy model (MacKenzie et al. 2006) to estimate occupancy (ψ = "psi"), and detectability (p) over two study years (2012-2013) for three size classes of redband trout, brown trout, and mountain whitefish. The three size classes were <151, 151-250, >250 mm TL. The upper limit of the smallest size class was chosen based on a natural break at 150 mm TL in the length frequency histograms of all the species (Figure 3). The largest size class was operationally defined as large, catchable fish of prime interest to anglers and special management concern. Each species was modeled separately. Season and year were used as factors to evaluate differences in occupancy and detection in different time periods.

We plotted the distribution and relative abundance of the three species and evaluated their relationships in three ways. First, maximum counts by species and size class were graphed by site. Maximum count during any single visit in the first season was considered a measure of relative abundance because it corrected for differences in the number of visits among the sample sites during the sampling season. Second, we used the Royle N-mixture model for repeated counts (Royle 2004) to estimate mean site abundance (λ = "lambda") and detectability (p) across sample sites during the first season. Differences among the species were considered significant when the 95% confidence intervals of the estimates did not overlap. This model assumes demographic closure of sites during the sample season, the distribution of animals across sample sites follows the Poisson distribution, and the detection probability at a site represents a binomial trial of the true number of animals at that site (Royle 2004). Since the closure assumption was likely not met, these Royle abundance estimates are used here to compare relative abundance among the species for an average site within the study area. Because of our interest in catchable fish, count data for the largest size class (i.e., >250 mm) of each salmonid species was analyzed separately using the computer software Program PRESENCE 6.4. Third, we used linear regression to evaluate the relationship between relative site abundance of redband trout, mountain whitefish, and brown trout. We included data from two size classes in this analysis: >150 and >250 mm TL.

We selected covariates for occupancy modeling a priori. For detectability, the covariates included year, season, size class, segment (i.e., river was divided into upper and lower segments at the Little Deschutes River), site discharge, site water temperature, and total visit seconds (i.e., electrofishing time during all sampling transects for each visit to a site). Only single-covariate models were evaluated because of a relatively small sample size. Prior to analysis, all continuous covariates were standardized into z-scores. The occupancy covariates evaluated were year, season, size class, and segment. It was assumed that site occupancy by any species and size class did not change during the study. We used the best fitting detection model as the baseline for modeling occupancy.

We used Akaike information criterion model selection procedures with a correction factor for low sample size [AICc] to select the models of best fit. Models were ranked by AICc values and evaluated using the ΔAIC (i.e., the difference in AICc values between a given model and the highest ranked model) and Akaike weight (wi), which is a relative measure of the weight of evidence for a model given the data (Burnham and Anderson 2002). The best fitting model had the lowest AICc and the greatest weight. We conducted the analysis using Program MARK.

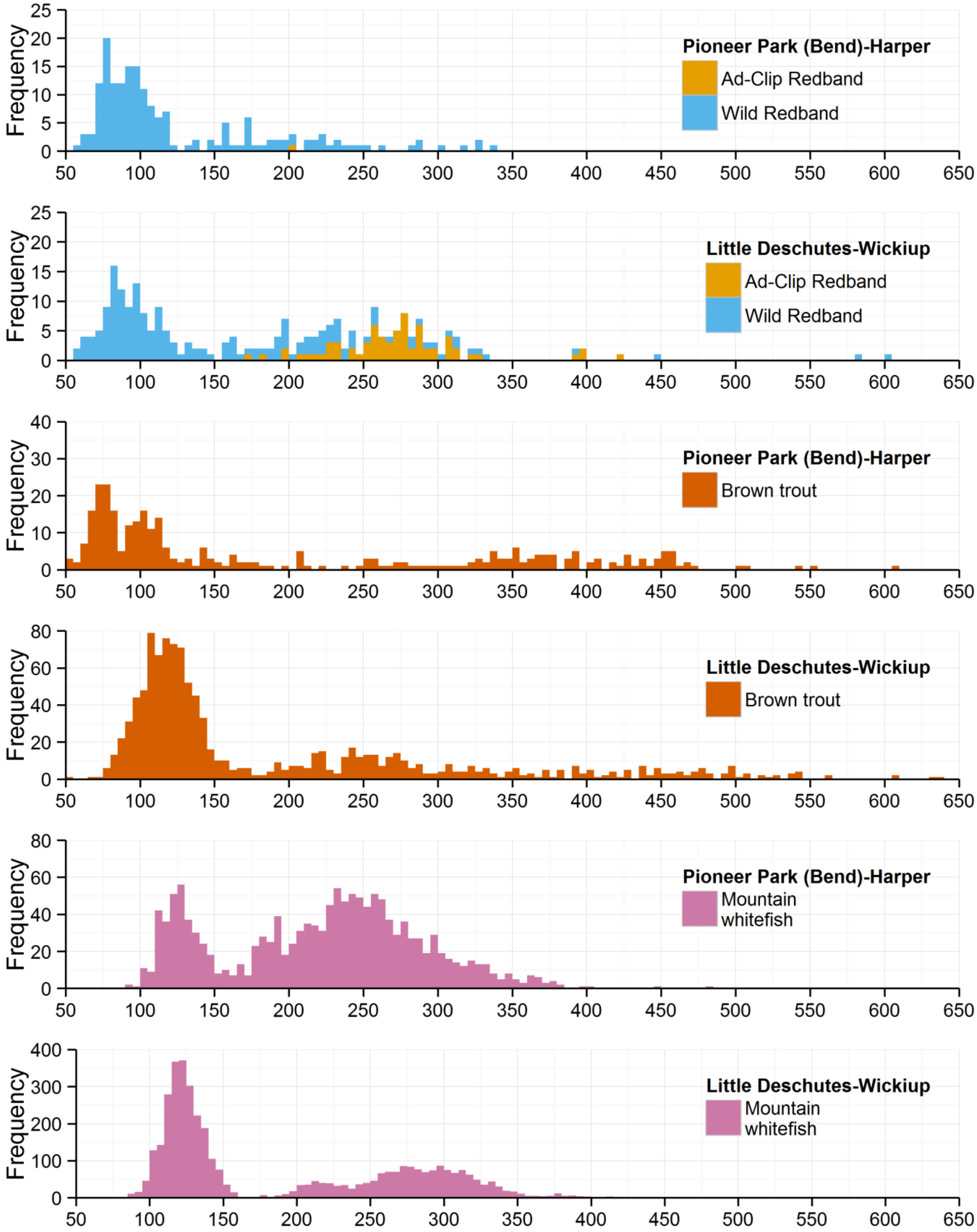

Figure 3. Frequency distributions of total length (mm) for all salmonids captured during sampling of 21 sites on the upper Deschutes River, from the city of Bend to Wickiup Dam, in 2012 and 2013. The adipose fin of hatchery-raised redband trout is clipped ("ad-clip") prior to stocking in the river.

Flow and temperature monitoring – Discharge data was obtained from the U.S. Bureau of Reclamation Hydromet system for two locations on the upper Deschutes River: just downstream (RK 364) of Wickiup Dam and just upstream (RK 293) of Benham Falls. Temperature loggers were deployed on November 16, 2012, and were located in six locations in the upper Deschutes River: downstream of Wickiup Dam (RK 364), downstream and upstream of Fall River (RM 329.1 and 329.3), at Harper Bridge (RM 310), and downstream and upstream of Spring River (RM 306.4 and 306.7). We used Vemco Minilog II-T temperature loggers to maintain consistency with partners and the Oregon Department of Environmental Quality [ODEQ]. Before deployment, the temperature loggers were calibrated following ODEQ guidelines. Each logger was secured to the bank with a cable extending 2-15 m into the water to accommodate high and low flows. Loggers were checked every other month and downloaded onto a Vemco field reader.

Results

Fish assemblage – Mountain whitefish was the dominant species captured during electrofishing surveys in both study years (Table 1). Nonnative brown trout comprised 17% of the catch and wild redband trout (i.e., no adipose fin clip) made up 3% of the catch in 2012 and 5.3% in 2013. Other fishes captured were sculpins (*Cottus spp*), three-spined stickleback (*Gasterosteus aculeatus*), and hatchery redband trout, as well as the nonnatives brown bullhead (*Ameiurus nebulosis*), kokanee (*O. nerka*), and tui chub (*Gila bicolor*).

Distribution and relative abundance – Redband trout, mountain whitefish, and brown trout were distributed throughout the entire upper Deschutes River sampling area as at least one individual from each species was captured at every sample site over the two-year study period (Figure 4). In individual years (2012/2013), redband trout were captured at 19 and 20 of the 21 sites, mountain whitefish were captured at all sites, and brown trout were captured at all sites except one in 2012.

Mountain whitefish were in high relative abundance throughout the sample area, brown trout were in moderate relative abundance in all but the downstream end of the sample area, and redband trout were in low relative abundance throughout most of the sample area (Figure 4). We observed peaks in the relative abundance of small redband trout downstream of the Sunriver site and large redband trout were the least abundant of all species and size classes and were observed at the fewest sites. Across the sample sites, mean site abundance for the largest size class of brown trout and mountain whitefish was significantly higher than that of redband trout (Table 2).

There was slight support for a positive relationship between the relative abundance of redband trout (>150 mm TL) and that of brown trout (>150 mm TL) in the study area (Table 3, Figure 5). However, the effect was small (i.e., slope of relationship = 0.14; P-value = 0.065) and less than 8% of the variation among sites was explained by the regression. This weak relationship did not hold when only the largest size class (>250 mm) was evaluated. The relative abundances of other salmonids were not related.

Occupancy and detection modeling – The best-fitting detection model included the size-class factor, suggesting that detection of the species varied by size class (Table 4). There was little support for other covariates influencing detection. The best-fitting occupancy model contained the largest size-class as a covariate and only slight support for season, segment, or year (Table 5). The best linear model suggested that redband trout in the largest size class (>250 mm TL) had significantly lower occupancy of the study area than the other size classes (Table 6). Assuming perfect detection of species at individual sites in the study area, the naïve estimates of occupancy for redband trout size classes were 0.60, 0.60, and 0.21 (Table 7). Simply put, we captured **s**mall and medium sized redband trout at 60% and large redband trout at 21% of the sample sites. However, the probability of detection for the redband trout varied by size-class from 0.51 for small fish to 0.28 for large fish (Table 7). When factoring in this imperfect detection, modeled occupancy was substantially greater than the naïve estimates (Table 7). Put another way, the model estimated there was an 81% probability that small and medium sized redband trout (i.e., <250 mm) and a 42% probability that large redband trout occupied a given site within our study area (Table 7). The best fitting linear model for brown trout was composed of the large size-class for estimating detectability and no covariates for occupancy (Table 7). For whitefish, the model was composed of season for detectability and size-class for occupancy (Table 7). Modeled occupancy probabilities for brown trout were similar to redband trout; except for the largest size class, in which large brown trout (ψ=0.85) were twice as likely to occupy a site than large redband trout (ψ=0.42). All size classes of whitefish were highly likely to occupy individual sites in the study area (Table 7).

Table 1. Electrofishing counts of all fish species captured (and as a percentage of the total catch) in the upper Deschutes River in 2012 and 2013.

	2012		Length (mm)				2013		Length (mm)			
Species	N	%	Mean	SD	Min	Max	N	%	Mean	SD	Min	Max
Mountain whitefish	2313	63.3	205	76	79	483	3072	69.4	191	84	86	483
Brown trout	634	17.4	199	119	53	635	777	17.6	180	121	50	630
Three-spined stickleback	251	6.9	45	28	19	440	42	0.9	42	10	19	60
Sculpin spp	228	6.2	56	17	28	97	183	4.1	62	20	26	107
Redband trout (Wild)	124	3.4	137	69	56	327	236	5.3	138	83	59	602
Redband trout (hatchery)	26	0.7	261	59	170	420	45	1.0	275	36	220	395
Brown bullhead	28	0.8	200	80	52	325	30	0.7	177	34	71	256
Kokanee	28	0.8	105	18	72	154	16	0.4	155	97	44	360
Tui chub	22	0.6	86	46	40	191	26	0.6	98	48	41	186

Table 2. Mean site abundance (λ -lambda) and detection probability (p), estimated using the Royle repeated count model, for fish greater than 250 mm in the upper Deschutes River.

				95% CI				95% CI	
Species	Year	λ	SE	Lower	Upper	p	SE	Lower	Upper
Redband trout (wild only)	2012	0.5	0.3	0.1	1.8	0.29	0.19	0.06	0.72
	2013	0.5	0.6	0.0	4.8	0.17	0.20	0.01	0.77
Redband trout (wild + ad clip)	2012	1.1	0.4	0.5	2.4	0.34	0.12	0.15	0.59
	2013	0.7	0.2	0.4	1.4	0.43	0.12	0.22	0.66
Brown trout	2012	7.9	1.8	5.1	12.2	0.29	0.06	0.19	0.43
	2013	8.3	4.8	2.7	25.5	0.17	0.10	0.05	0.44
Mountain whitefish	2012	36.3	6.2	26.0	50.7	0.27	0.05	0.19	0.37
	2013	23.7	2.3	19.6	28.7	0.46	0.04	0.38	0.54

Table 3. Linear regressions (y ~ x) of maximum count during a single sampling visit for each study site between two salmonid species. Two size classes were evaluated. Data were from the first sampling season in 2012 and 2013.

	Intercept			Slope				
Regression	Estimate	SE	P-value	Estimate	SE	P-value	R^2	P-value
>250 mm FL								
Redband trout ~ Brown trout	0.52	0.42	0.22	0.10	0.08	0.19	0.04	0.19
Redband trout ~ Whitefish	1.01	0.51	0.05	0.00	0.03	0.89	0.00	0.89
Brown trout ~ Whitefish	2.93	0.99	<0.01	0.08	0.05	0.17	0.05	0.17
>150 mm FL								
Redband trout ~ Brown trout	1.51	0..56	0.01	0.14	0.08	0.07	0.08	0.07
Redband trout ~ Whitefish	3.03	0.69	<0.01	-0.03	0.02	0.25	0.03	0.25
Brown trout ~ Whitefish	4.95	1.40	<0.01	0.03	0.05	0.52	0.01	0.52

Discharge and temperature – The annual hydrographs of the upper Deschutes River in 2012 and 2013 resembled the regulated flow pattern of extreme high flows in spring and summer and extreme low flows in the fall, but the winter flow more closely matched the historical norm (Figure 6). During the spring and summer regulated high-flow period, discharge fluctuated dramatically during each month, with intra-month fluctuations increasing and decreasing by up to 600 cfs. This also differs dramatically from the steady historical flows.

Maximum water temperatures in the upper Deschutes River peaked in July and August, ranging from 13.3-17.1˚ C near Wickiup Dam, 13.8-15.6˚ C just upstream of the Fall River confluence, 11.1-14.8˚ C just downstream of Fall River, 15.7-17.6˚ C upstream of the Spring River confluence, 15.7-17.9˚ C downstream of the Spring River confluence, and 15.9-18.1˚ C at Harper Bridge (Figure 7). Minimum water temperatures occurred in December and January, ranging from 0-7˚ C. During this period, cold water temperatures were slightly ameliorated downstream of the tributaries Fall River and Spring River.

Discussion

The objectives of this project were to determine current status of salmonids by size class, evaluate some of the biotic and abiotic factors influencing fish populations, and develop a long-term monitoring protocol. Prior experience using CMR (capture-mark-recapture) methods in the middle Deschutes River reach known as Foley Waters suggested that this approach would be labor intensive and highly imprecise (Jacobsen and Jacobs 2010, Carrasco et al. 2012). Given this, we have evaluated the utility of a temporal revisit occupancy design to determine the spatial distribution of salmonids and measure relative abundance. From 2012-2013, salmonid distribution surveys were conducted by cataraft electrofishing in the upper Deschutes River. Analysis of this dataset suggests that the sampling methodology has utility for estimating the distribution and relative abundance of the three salmonid species. However, the sampling to date has highlighted some deficiencies that could potentially be addressed by modifying the sampling approach.

Distribution and abundance – Redband trout, brown trout, and mountain whitefish were distributed throughout the upper Deschutes River. The occupancy probability and “Royle repeated count” abundance estimates suggest that large redband trout (>250 mm TL) are significantly less abundant and are less extensively distributed than large brown trout and mountain whitefish. This is a management concern because redband trout were thought to

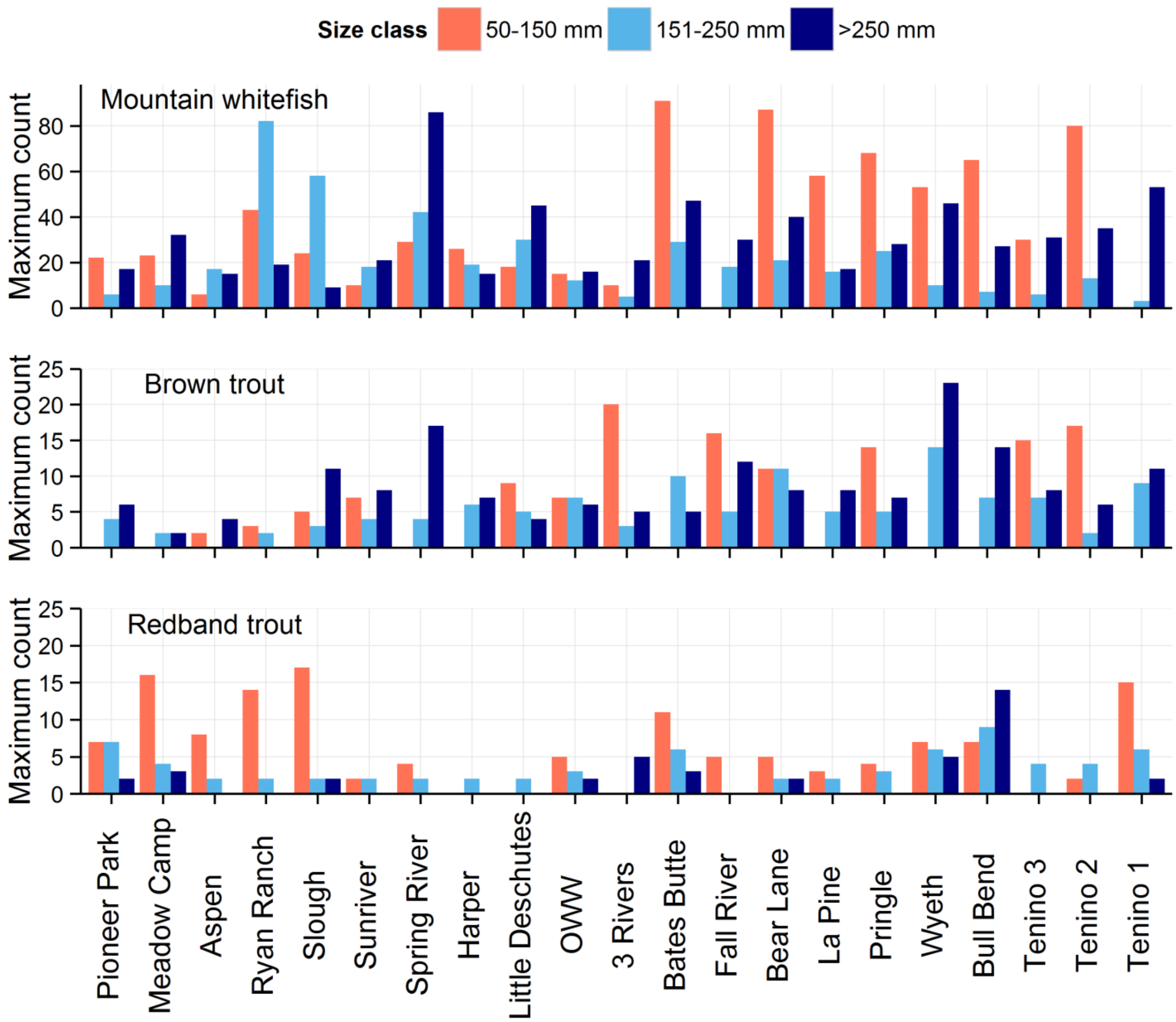

Figure 4. The relative abundance using the maximum count of salmonids captured during cataraft electrofishing of 21 sample sites in 2012 and 2013, from Pioneer Park in Bend to Tenino 1 near Wickiup Dam, in the upper Deschutes River. Each 200-300 m site was sampled from 1 to 6 times during each sampling year. Maximum count, correcting for this range in sampling effort among the sites, represents the greatest number of each species captured during an individual visit to site. Maximum counts in 2012 and 2013 were summed for each sample site by size class and for each salmonid species. Redband trout barplot includes wild and hatchery fish.

be abundant historically and provided an attractive recreational fishery in the upper Deschutes River. It is not clear what factors may be limiting the distribution and abundance of large redband trout in the upper Deschutes River relative to other species and size classes. There have been many anthropogenic changes to the upper Deschutes River over the last century. As a result, there may be many potential confounding factors that limit salmonid abundance. We consider several of these factors that may be limiting redband trout abundance, including interspecific competition and the effects of discharge and temperature.

Inter- and intra-specific interactions – High relative abundance of brown trout did not result in competitive exclusion of redband trout. Instead, there was a weak positive relationship among these species. In other words, when

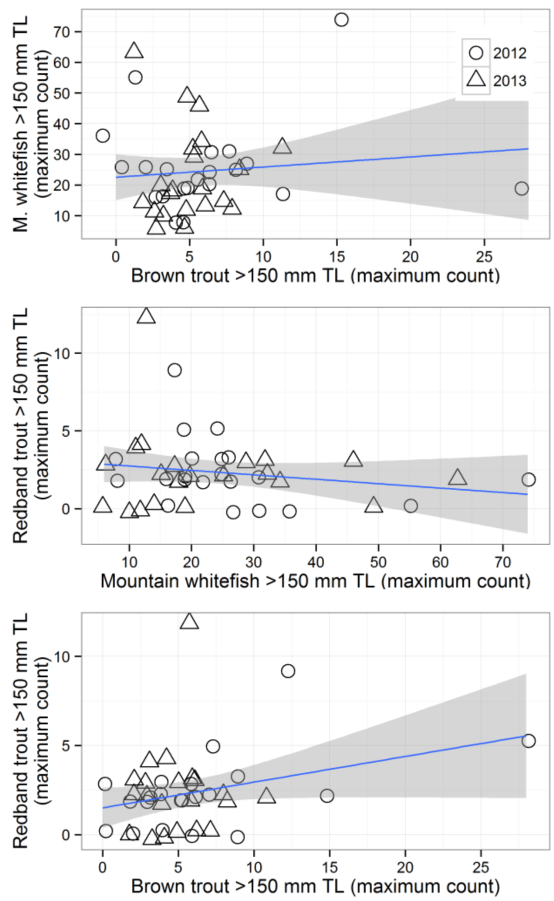

Figure 5. Linear regressions of the maximum count of each species (fish >150 mm FL) during any single visit to 21 sites in the first sampling season in the upper Deschutes River in 2012 and 2013. Each symbol represents an individual site and the symbols are jittered to show stacked points.

brown trout were present at a site, it was slightly more likely that redband trout were also present at that site. Higher relative abundance of browntrout was expected for a number of reasons. Brown trout have high overlap in food and habitat use with other trout species (Gatz et al. 1987; McHugh et al. 2006) and they are more aggressive at similar sizes, even when smaller than other trout (Fausch and White 1981; Shirvell and Dungey 1983; Wang and White 1994; McHugh and Budy 2005). When in sympatry with brown trout, other trout species shift to less preferred habitat (Gatz et al. 1987; Wang and White 1994), change dietary habits (McHugh and Budy 2006), and display worse performance, such as smaller prey size, slower growth, and worse condition (McHugh and Budy 2005). This information comes from studies conducted in experimental channels or small streams (i.e., 3rd and 4th order). In the Wood River, a higher-order tributary of Upper Klamath Lake, large populations of redband trout and brown trout have coexisted for decades (ODFW, unpublished data). This occurs, in part, through habitat partitioning, with fluvial brown trout foraging mainly in the Wood River and adfluvial redband trout foraging mainly in the lakes (ODFW, unpublished data). Brown trout and redband trout in the upper Deschutes River, where this is no lacustrine habitat available, likely share a similar fluvial life history and substantial overlap in their niches. If so, negative interactions with brown trout may be one of the factors limiting growth and relative abundance of redband trout in the upper Deschutes River.

Mountain whitefish and redband trout occur in sympatry in this region (Whitman 2002, WPN 2002) in high abundance, historically (e.g., Gray 1986)' and currently in some watersheds (e.g., Odell Creek: Gray 1986; unpublished data, ODFW, 2014). Mountain whitefish removal programs have been conducted in the Deschutes River basin (e.g., Odell Creek, Fies et al. 1996) and in other areas of the western United States (see Meyer et al. 2009) because of the perception that they limited trout production through competition for food and habitat. However, this assertion has not been substantiated and the few studies that address the question of competition between these two salmonid have been inconclusive (IDFG 2007). These studies found that mountain whitefish and rainbow trout, when in sympatry, can have substantial dietary overlap, but whitefish tended to feed more on larval insects inhabiting the stream bottom while trout tended to feed more on terrestrial insects on the surface and invertebrates in the drift (e.g., McHugh 1940, Fuller 1981, Pontius and Parker 1973). One feeding ecology study in the Kootenai River found that rainbow trout and mountain whitefish did not compete for resources; with whitefish more selective and keying in on chironomids from the benthos and drift, while rainbow trout were more opportunistic column and surface feeders (Dos Santos 1985).

The extensive distribution of redband trout in the upper Deschutes River suggests that this species is able to successfully compete or partition resources to some degree while in sympatry with brown trout and mountain whitefish. However, the lower relative abundance of redband trout suggests competition for food and habitat near the benthos with highly abundant whitefish and competition in the water column and at the surface with an aggressive generalist like brown trout may be narrowing the availability and access to resources, thereby shrinking the niche of the redband trout in the upper Deschutes River. There is currently little research exploring interspecific competition and predation among all three species in sympatry, therefore there is little understanding if, or to what degree, this factor is limiting redband trout abundance and growth. Other factors, besides interspecific interactions, may help explain the low relative abundance of redband trout.

Table 4. Detection modeling results for redband trout in the upper Deschutes River based on three visits, during two seasons, and over two years (2012-2013) at 21 sites sampled by boat electrofishing. Covariates modeled included size class, year, segment, season, and electro-fishing seconds. Continuous covariates were standardized as z-scores prior to modeling.

Model	AICc	ΔAIC	w_i	Model Likelihood	Parameters	Deviance
All size classes	616.0	0.0	0.69	1.00	4	607.8
Large size class	617.6	1.6	0.31	0.44	3	611.5
No covariates	643.0	27.0	0.00	0.00	2	639.0
p2=p3	643.7	27.7	0.00	0.00	3	637.6
Year	644.0	28.0	0.00	0.00	2	637.9

Table 5. Occupancy modeling results for redband trout in the upper Deschutes River based on three visits, during two seasons, at 21 sites sampled by boat electrofishing. Covariates modeled included size class, year, segment, and season. As "All size classes" provided the best model for detection, these indicator variables were used in all occupancy models.

Model	AICc	ΔAIC	w_i	Model Likelihood	Parameters	Deviance
Large size class	615.3	0.0	0.32	1.00	5	605.0
No covariates	616.0	0.7	0.23	0.72	4	607.8
All size classes	617.4	2.1	0.11	0.35	6	605.0
Season	617.6	2.3	0.10	0.32	5	607.3
Segment	618.0	2.7	0.08	0.26	5	607.7
Year	618.1	2.7	0.08	0.26	5	607.8

Table 6. The best linear model results for detection (p) and occupancy (ψ) of redband trout in the upper Deschutes River study area. Betas represent the slope of the linear relationship of individual covariates and are considered significant if their confidence interval does not overlap zero.

			95% CI	
Parameter	Beta	SE	Lower	Upper
p.intercept	0.035	0.207	-0.370	0.441
p.medium size class	-0.526	0.266	-1.046	-0.005
p.large size class	-0.974	0.519	-1.991	0.043
ψ.intercept	1.434	0.413	0.624	2.244
ψ.large size class	-1.741	0.695	-3.103	-0.378

Table 7. Detectability and occupancy estimates with 95% confidence intervals (CI) for three salmonid species in the upper reach (Pioneer Park in Bend to Wickiup Dam) of the Deschutes River. Detection and occupancy were estimated using the best model for each species; covariates with the most support differed among species. Cataraft electrofishing was used in three visits to 21 sites in each of two seasons in both 2012 and 2013.

	Detectability (p)					Occupancy (ψ)					
Species	Covariate	Modeled	SE	CI:Lower	CI:Upper	Covariates	Naïve	Modeled	SE	CI:Lower	CI:Upper
Redband	50-150	0.51	0.05	0.40	0.62	50-150	0.60	0.81	0.06	0.68	0.93
	151-250	0.38	0.06	0.26	0.50	151-250	0.60	0.81	0.06	0.68	0.94
	>250	0.28	0.10	0.09	0.47	>250	0.21	0.42	0.14	0.15	0.69
Brown	50-150	0.77	0.03	0.71	0.83	No covariates	0.80	0.85	0.04	0.77	0.93
	151-250	0.77	0.03	0.71	0.83						
	>250	0.48	0.06	0.37	0.59						
Whitefish	Season 1	0.91	0.02	0.88	0.94	50-150	0.87	0.90	0.03	0.84	0.96
	Season 2	0.68	0.04	0.60	0.76	151-250	0.97	1.00	na	na	na
						>250	0.88	0.90	0.03	0.84	0.96

The majority of the large redband trout captured in the upper Deschutes River were adipose fin-clipped hatchery fish. Each year 7,500 hatchery rainbow trout of “catchable” size (>200 mm) are stocked in the upper segment of this study reach. The stocking appears to be accomplishing its intended effect of providing fish for a put-and-take recreational fishing opportunity, but there could also be unintended effects. For example, these large size-class hatchery fish may reduce the native population size through intra-specific competition and predation. Genetic studies, such as in the upper Snake River basin in Idaho (Kozfkay et al. 2011) and the Metolius River in Oregon (Currens et al. 1997, Williams et al. 1997) have demonstrated that hybridization can occur when coastal rainbow trout (*O. m. irideus*) are stocked in areas where native inland redband trout occur. Native populations are thought to have better fitness and greater long term probability of persistence than hatchery-raised fish because they have adapted to local environmental conditions (Allendorf and Leary 1986), especially when nonnative strains are more susceptible to certain diseases (Currens et al. 1997). The myxosporan parasite Ceratomyxa shasta is lethal to susceptible salmonids, but salmonids indigenous to areas where the parasite naturally occurs have resistance (Bartholomew et al. 1989, 1992). When native resistant redband trout and susceptible hatchery stocks interbred in the Metolius River, the hybrids exhibited an intermediate susceptibility to *C. shasta* (Currens et al. 1997). ODFW ended the stocking program on the Metolius River in 1996 and redband trout redd census counts increased dramatically in subsequent years (Riehle and Dachtler 2011). Current stocking is unlikely to be increasing susceptibility of native fish to *C. shasta* because the ”Crane Prairie” stock used since 2006 in the upper Deschutes River was established in the hatchery using wild fish from the upper Deschutes River and is thought to be resistant to *C. shasta*. In the decades prior to this, several different coastal rainbow hatchery strains were stocked in the upper Deschutes River and in the reservoirs upstream of Wickiup Dam (Matala et al. 2007). A genetic analysis showed introgression to be highly restricted in the reservoirs (Matala et al. 2007), which suggests the same is likely true between hatchery and native fish in the upper Deschutes River.

Dams – The influence of dams on fish populations and riverine ecosystems has been widely documented (see Bednarek 2001). The upper Deschutes River hydrograph has been dramatically altered by the construction of storage dams and water management mainly for

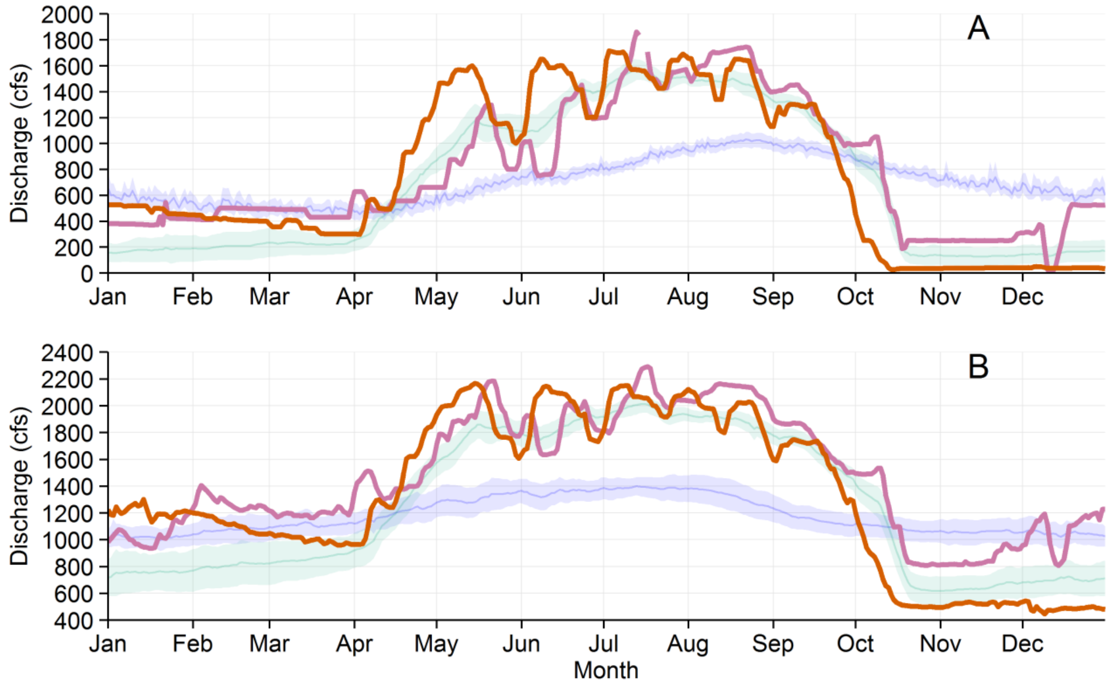

Figure 6. Daily mean discharge of the Deschutes River just downstream of Wickiup Dam (Panel A) and at Benham Falls (Panel B) for 2012 (pink line) 2013 (orange). Historical (blue line) and regulated (green) flows and their 95% confidence intervals are shown in the background.

irrigation and other human activities (Golden and Alyward 2006). In the upper Deschutes River, Wickiup Dam directly blocks upstream migratory access to historical spawning areas and degrades downstream habitat by restricting natural wood and sediment transport needed for formation of spawning habitat and channel complexity. The loss of access to spawning areas and a reduction in spawning gravels has likely reduced migratory fish abundance in the river downstream of the dam.

Discharge and water quality – Current water management of the upper Deschutes River, which used to be known for its remarkably steady flows (Gannet et al. 2003), produces relatively extreme low flows from November through April and extreme high flows from June through September. Redband trout and brown trout have different life cycle timing and may be affected differentially by this managed flow regime. In the upper Deschutes River, redband trout begin spawning when water temperatures reach 6-7˚ C in April and spawn through June (NPCC 2004); although spawning has been observed in the lower Deschutes River from March to August (Zimmerman and Reeves 1999). Under the current flow management regime, redband trout spawning begins as river flows are at first increasing rapidly and then are relatively unstable throughout the spring and summer spawning period. In the upper Deschutes River just downstream of Wickiup Dam in 2012 (see Figure 6), April flows increased from 300 cfs to 1500 cfs, May flows dropped from 1600 cfs to 1000 cfs, and June flows fluctuated from 1100 to 1700 and back to 1200 cfs. Similar flow fluctuations occurred in 2013. This rapid ramp up of flow also increases turbidity and sedimentation downstream of Wickiup Dam (NPCC 2004). Rainbow trout eggs take 4-8 weeks to hatch at water temperatures between 8-11˚ C; once hatched, alevins emerge from the redd several weeks later (Quinn 2005). During their development in the redd, eggs and alevins would be exposed to several large flow fluctuations and increased sedimentation. Depending on where and when in the flow fluctuations spawners constructed a redd, eggs and alevins developing in redds may be harmed by changing interstitial flow dynamics related to an increase in sedimentation or change in water depth, or killed through dewatering.

Shortly after redband trout fry are estimated to emerge from redds, flows at Wickiup Dam are rapidly lowered. The rapid transition from extreme high flows in September to extreme low flows by November has indeed caused direct mortality by stranding fish in at least one rapidly dewatered side channel. In 2013, near Benham Falls, September high flows of 1900 cfs were reduced to 800 cfs by mid-October (which was a moderate transition relative to 2012 and the historical average). During this transition, a private citizen noticed hundreds of stranded and dead fish in the Lava Island side channel of the Deschutes River, which was dewatered as flows were reduced at Wickiup Dam. She notified ODFW and other volunteers, who helped salvage surviving fish. In 2014, an organized salvage operation was conducted by ODFW staff and volunteers at the mile-long side channel. Almost 7000 fish were rescued, 72% of which were juvenile redband trout (i.e., <150 mm long) and less than 1% were brown trout (ODFW, High Desert Region, unpublished data, 2014). The dewatering of this channel and the attendant fishkill presumably has occurred on a regular basis for several decades prior to the salvage operation in 2013. This example suggests that this rapid transition to extreme low flows increases risks of fish stranding in isolated or dewatered lateral and side channel habitats, reduces availability of lateral habitats used for juvenile rearing, and may reduce fry dispersal.

Brown trout spawn in October and November, generally when flows are at their lowest, and fry emerge in March (Fies et al. 1996), just as managed flows start rapidly increasing. It is not clear how the altered flow regime impacts this brown trout life cycle, but it is possible that the current flow regime favors brown trout over redband trout. For example, in 2012 and 2013, autumn low flows appeared to be more stable than spring and summer high flows. Although low flows may reduce available spawning habitat, more stable flows may also reduce the risk of harm to redds. It is also possible that brown trout fry have access to relatively more rearing habitat throughout the spring and summer high flows and experience relatively greater dispersal. Furthermore, brown trout fry are larger and possess greater swimming ability than redband trout fry during the rapid transition to low flows and, as a result, may have better survival during this period. Clearly, more work needs to be done to understand the magnitude of the stranding problem, how much lateral habitat is lost during low flow periods, and if the current managed flow regime more adversely affects the habitat and recruitment of certain species or life-stages.

Alteration of the natural flow regime and diverting water for irrigation are likely influencing stream temperature in the upper Deschutes River and this in turn could be affecting fish populations. The main concern is that regulated low flow during the winter water storage period exacerbates cold water temperatures and allows for ice formation (Fies et al. 1996, NPCC 2004). Icy conditions may cause extended periods of stressful conditions, direct mortality of trout, and force trout to crowd into deeper pools thereby increasing their vulnerability to predation (Fies et al. 1996). The prevalence in winter of severely cold temperatures and instream ice , and their impact on trout populations, have not been quantified in this basin.

In summer, the upper Deschutes River does not meet federal water quality standards and is currently on Oregon's "303(d)" list for several pollutants, including water temperature. The upper Deschutes River currently does not meet the ODEQ temperature standards (7-day average maximum temperature) for bull trout rearing (12˚ C) or core coldwater habitat (16˚ C) and rearing/migration habitat (18˚ C) for salmon and trout (ODEQ 2004). Even so, during the summer irrigation season, water is released at Wickiup Dam from the cold layer of the reservoir and stream temperatures are thought to be suitable for brown and redband trout (Fies et al. 1996). In summers with severe drawdowns, warm water

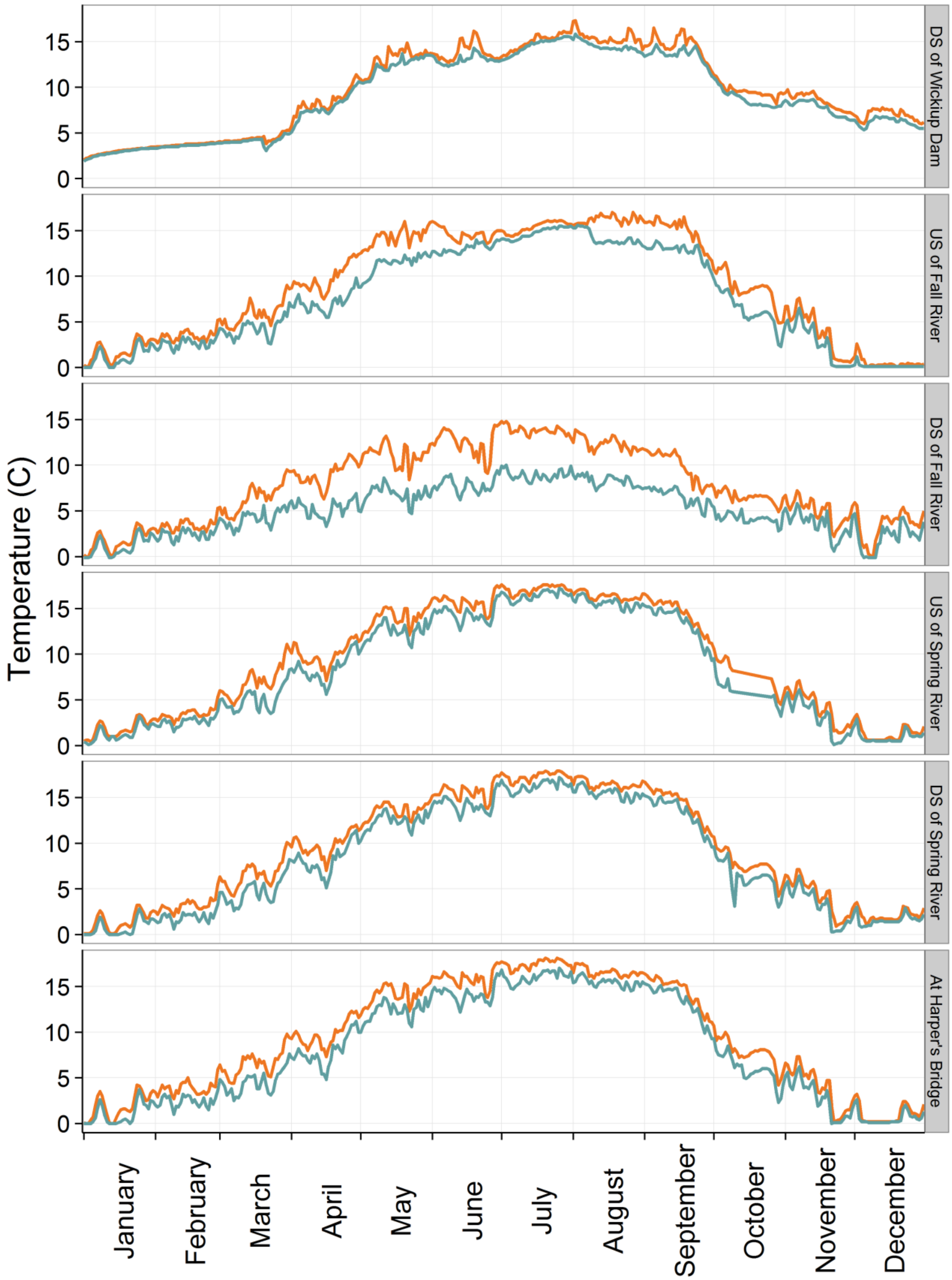

Figure 7. Daily maximum (orange) and minimum (blue) stream temperature for 2013 at six locations in the upper Deschutes River. U.S. BOR data were used for the site downstream of Wickiup Dam. The ODFW thermograph upstream of Spring River was exposed to air October 10-25 for which data are extrapolated.

Table 8. Important thermal tolerance characteristics of brown trout (*S. trutta*) and rainbow/redband trout (*O. mykiss*) (after Jonsson and Jonsson 2009)

Species	Lower critical range (˚C)	Upper critical range (˚C)	Upper incipient lethal (˚C)	Optimum for growth (˚C)
S. trutta	0-4	20-30	24.7	13-17
O. mykiss	0-9	19-30	26.2	15-19

from shallow upstream reservoirs may adversely affect trout in the upper Deschutes River (Fies et al. 1996, Fitzpatrick et al. 2006). Data from our thermograph sites suggest that the upper

Deschutes River from Harper Bridge upstream to Wickiup Dam met the standards in 2013 for trout rearing and migration. How the thermal regime in summer influences the trout populations in the upper Deschutes River has not been evaluated. It would likely be a complex evaluation because even though brown trout and redband trout have similar thermal tolerance characteristics (Table 8), they have different life cycle timing (as described above) and as a result the thermal regime in summer may affect each species differently.

Monitoring protocol evaluation – This occupancy monitoring protocol was effective at determining the baseline distribution and relative abundance of salmonids in the upper Deschutes River. The extensive distribution of redband trout suggests that this species is not presently at-risk of extinction in this study area; however, the low relative abundance of the redband trout is a management concern. Understanding how the river can be managed to improve the relative abundance of redband trout, and trout populations overall, is a complex endeavor with many interacting factors (e.g., flow regime, habitat quality and connectivity, interspecific competition, and water quality) likely influencing trout populations. Understanding the influence of these factors and how to prioritize restoration actions may be best explored using a model designed to simulate salmonid population response to flow and riverine habitat alteration (e.g., Railsback et al. 2009).

Cataraft electrofishing in large river habitats had many deficiencies. The method was time-consuming for a single crew of three people. This led to a small sample size and unequal effort among the sample sites, which decreases precision of estimates and representativeness of the study. Abundance estimates of large redband trout and brown trout in this study and in past studies using boat electrofishing were imprecise, in part because capture probability was very low. Low capture probability may result from fish avoiding the boat, fleeing the electric field, being shocked but not captured, or emigrating from the site between revisits. The validity of abundance and occupancy estimates in occupancy and CMR sampling designs are dependent on the assumption of site closure to mortality, emigration, and immigration. The demographic closure assumption has been studied using backpack electrofishers in small wadeable streams in which closure can be actively attempted using blocknets (Peterson et al. 2005). In large river settings, demographic closure of a sample site is difficult to attain. Since these sample sites were too wide or water velocity and discharge too high for active closure of a site, an assumption was made that one may sample over a short time period to ensure no loss or gain of individuals at a site during the study (Pine et al. 2003). In this study, the first season was conducted over a 1-2 month period and the degree to which the closure assumption was violated was not known. We are not aware of studies that test this assumption in the context of boat electrofishing in larger river habitats. If boat electrofishing is to be used either for CMR or occupancy estimates, this assumption must be tested and the degree to which it is met should be better understood.

There was also concern that boat electrofishing in rivers was biased against the capture of smaller fish (<150 mm TL). In a companion study in the higher gradient middle Deschutes River, boat electrofishing was generally relegated to one transect down the main flow line. As a result, sampling was minimal in the juvenile rearing habitats (e.g., river margins, secondary channels) and few fish less than 100 mm TL were captured (Carrasco and Moberly 2014). Although small juvenile fish were captured in the upper Deschutes River at most sites, the results were likely biased by the restriction of cataraft electrofishing to main channels and deeper areas. To better understand the distribution of early rearing juvenile trout and, in turn, the distribution of spawning in the upper Deschutes, an alternative to boat electrofishing should be explored.

Future research – In 2014, we designed the sampling in the upper Deschutes River to use a model that incorporates the probability of temporary emigration from a sampling site (Gwinn et al. 2011). Analysis of this dataset will occur in 2015 and may improve our understanding of the degree to which this protocol meets the demographic closure assumption. For field sampling in 2015, we suggest testing capture and estimation methods that eliminate the need for active demographic closure at sample sites. Focusing sampling on juvenile rearing areas (e.g., margins, side channels, off-channel habitats) and sampling without replacement (i.e., spatial replicates within a site) would avoid the closure requirement and also provide information on spawning and early rearing habitat and distribution. Emerging techniques in genetic assessment and monitoring have been used to estimate effective population size and effective number of breeders of trout, salmon, and other animals (Waples and Do 2010; Allendorf et al. 2013), often using the young of the year cohort for the analysis (e.g., Whiteley et al 2011). Genetic assessment redband trout in the middle and upper Deschutes River would provide needed information on population structure in regard to natural and artificial barriers to movement and introgression with hatchery trout.

Acknowledgements

We thank the Mitigation and Enhancement Board for providing project funding and support. Thanks to Beth Bailey (ODFW Native Fish Investigations Program) for assistance with data management; Michael Harrington (ODFW Klamath Watershed District Fisheries Biologist) and Terry Shrader (ODFW Pelton-Round Butte Mitigation Program) for technical support; Shaun Clements (ODFW Native Fish Investigations Program) for helpful reviews of previous drafts; and Kristen Childress, Matt Colver, and Jessica Ruthardt for all their hard work in the field.

References

Allendorf, F.W., and R.F. Leary. 1986. Heterozygosity and fitness in natural populations of animals. Conservation biology: the science of scarcity and diversity. M.E. Soule, editor. Sinauer Associates Inc., 584 p. 57-76

Allendorf, F.W., G. Luikart, and S.N. Aitken. 2013. Conservation and the genetics of populations. In 2nd edition. Wiley-Blackwell, Chichester.

Bednarek, A. T. 2001. Undamming rivers: a review of the ecological impacts of dam removal. Environmental Management 27:803–814.

Bartholomew, J.L., J.S. Rohovee, and J.L. Fryer. 1989. *Ceratomyxa shasta*, a myxosporean parasite of salmonids. U.S. Fish and Wildlife Service Fish Disease Leaflet 80.

Bartholomew, J.L., J.L. Fryer, and J.S. Rohovee. 1992. *Ceratomyxa shasta* infections of salmonid fish. Proceedings of the OJI international symposium on salmonid disease. Hokkaido University Press. Sapporo, Japan.

Budy, P., T. G. P. Thiede, P. McHugh, E. S. Hansen, W. J, and J. Wood. 2008. Exploring the relative influence of biotic interactions and environmental conditions on the abundance and distribution of exotic brown trout (*Salmo trutta*) in a high mountain stream. Ecology of Freshwater Fish 17:554–566.

Carrasco, R. and E. Moberly. 2014. 2013 Middle Deschutes fisheries monitoring report: Fish distribution and abundance in the Middle Deschutes River. Oregon Department of Fish and Wildlife, Bend, Oregon.

Currens, K. P., A. R. Hemmingsen, and R. A. French. 1997. Introgression and susceptibility to disease in a wild population of rainbow trout. North American Journal of Fisheries Management 17:1065–1078.

DosSantos, J. M. 1985. Comparative food habits and habitat selection of mountain whitefish and rainbow trout in the Kootenai River, Montana. Master's thesis. Montana State University, Bozeman.

Fausch, K. D., and R. J. White. 1981. Competition between brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) for positions in a Michigan stream. Canadian Journal of Fisheries and Aquatic Sciences 38:1220–1227.

Fies, T., J. Fortune, B. Lewis, M. Manion, and S. Marx. 1996. Upper Deschutes River subbasin fish management plan. Bend, Oregon.

Fitzpatrick, K., K. Gorman, and B. Aylward. 2006. Reservoir management: DWA final report. Deschutes Water Alliance Issue Paper. Bend, Oregon.

Fuller, R. K. 1981. Habitat utilization, invertebrate consumption, and movement by salmonid fishes under fluctuating flow conditions in the Big Lost River, Idaho. Master's thesis. Idaho State University, Pocatello.

Gannett, M. W., M. Manga, and K. E. Lite Jr. 2003. Groundwater hydrology of the upper Deschutes Basin and its influence on streamflow. Water Resources and Application 7:31–70.

Gatz, A. J., M. J. Sale, and J. M. Loar. 1987. International association for ecology habitat shifts in rainbow trout: competitive influences of brown trout. Oecologia 74:7–19.

Golden, B., and B. Aylward. 2006. Instream flow in the Deschutes Basin: monitoring, status and restoration needs: DWA final report. Bend, Oregon.

Gray, E. 1986. Roughing it on the Little Deschutes River 1934-1944: A history of a sawmill camp and its people. K & M Printing and Lithographing, Inc. Eugene, Oregon.

Gwinn, D. C., P. Brown, J. C. Tetzlaff, and M. S. Allen. 2011. Evaluating mark - recapture sampling designs for fish in an open riverine system. Marine and Freshwater Research 62:835.

IDFG 2007. Mountain whitefish conservation and management plan for the Big Lost River drainage, Idaho. Idaho Department of Fish and Game.

Jacobsen, R.L. and S. Jacobs. 2010. Evaluation of raft electrofishing for fish sampling in the Middle Deschutes River. Oregon Department of Fish and Wildlife, Native Fish Investigations Project, Corvallis, Oregon.

Jonsson, B., and N. Jonsson. 2009. A review of the likely effects of climate change on anadromous Atlantic salmon *Salmo salar* and brown trout *Salmo trutta*, with particular reference to water temperature and flow. Journal of Fish Biology 75:2381–447.

Kozfkay, C. C., M. R. Campbell, K. A. Meyer, and D. J. Schill. 2011. Influences of habitat and hybridization on the genetic structure of redband trout in the upper Snake River Basin, Idaho. Transactions of the American Fisheries Society 140:282–295.

Ligon, F. K., W. E. Dietrich, and W. J. Trush. 1995. Ecological effects downstream of dams: a geomorphic perspective. BioScience 45:183–192.

MacKenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2006. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Elsevier Inc., Burlington MA.

Matala, A. P., T. G. Wise, S. Marx, and W. R. Ardren. 2007. Evaluating population genetic structure and introgression between native redband trout (*Oncorhynchus mykiss gairdneri*) and out-of-basin hatchery-origin rainbow trout (*O. m. irideus*) in Crane Prairie Reservoir, Oregon. U.S. Fish and Wildlife Service Report. Longview, WA.

McHugh, J. L. 1940. Food of the Rocky Mountain whitefish *Prosopium williamsoni*. Journal of the Fisheries Research Board of Canada 5:131–137.

McHugh, P., and P. Budy. 2005. An experimental evaluation of competitive and thermal effects on brown trout (*Salmo trutta*) and Bonneville cutthroat trout (*Oncorhynchus clarkii utah*) performance along an altitudinal gradient. Canadian Journal of Fisheries and Aquatic Sciences 62:2784–2795.

McHugh, P., and P. Budy. 2006. Experimental effects of nonnative brown trout on the individual- and population-level performance of native Bonneville cutthroat trout. Transactions of the American Fisheries Society 135:1441–1455.

Meyer, K. A., F. S. Elle, and J. A. Lamansky. 2009. Environmental factors related to the distribution, abundance, and life history characteristics of mountain whitefish in Idaho. North American Journal of Fisheries Management 29:753–767. DOI:10.1577/M08-015.1

NPCC. 2004. Deschutes Subbasin Plan. Northwest Power and Conservation Council, Portland, Oregon.

ODEQ. 2004. Oregon's 2004 Water Quality Assessment. Water Quality Division. Portland, Oregon: Oregon Department of Environmental Quality.

Peterson, J. T., N. P. Banish, and R. F. Thurow. 2005. Are block nets necessary?: Movement of stream-dwelling salmonids in response to three common survey methods. North American Journal of Fisheries Management 25:732–743.

Pine, W.E., K.H. Pollock , J.E. Hightower , T. J. Kwak, and J.A. Rice. 2003. A review of tagging methods for estimating fish population size and components of mortality. Fisheries 28:10-23. DOI: 10.1577/1548-8446(2003)28 [10:AROTMF]2.0.CO;2

Pontius, R. W., and M. Parker. 1973. Food habits of the mountain whitefish, *Prosopium williamsoni (Girard).* Transactions of the American Fisheries Society 102:764–773.

Quinn, T.P. 2005. The behavior and ecology of Pacific salmon and trout. University of Washington Press, Seattle, Washington. 320 pp.

Railsback, S.F., B.C Harvey, S.K. Jackson, K. Stephen, and R.H. Lamberson. 2009. InSTREAM: the individual-based stream trout research and environmental assessment model. Gen. Tech. Rep. PSW-GTR-218. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 254 p.

Riehle, M., and N. Dachtler. 2011. Redband trout (*Oncorhynchus mykiss*) spawning surveys in the upper Metolius River, Oregon, 2009-2010. Item 11b in Pelton Rounde Butte 2011 Fisheries Workshop DVD.

Royle, J. A. 2004. N-mixture models for estimating population size from spatially replicated counts. Biometrics 60:108–15.

Shirvell, C. S., and R. G. Dungey. 1983. Microhabitats chosen by brown trout for feeding and spawning in rivers. Transaction of the American Fisheries Society 112:355– 367.

USDA 1996. Upper Deschutes wild and scenic river: record of decision and environmental impact statement. Portland, Oregon.

Wang, L., and R. J. White. 1994. Competition between wild brown trout and hatchery greenback cutthroat trout of largely wild parentage. North American Journal of Fisheries Management 14:475–487.

Waples, R.S., and Do, C. 2010. Linkage disequilibrium estimates of contemporary N e using highly variable genetic markers: a largely untapped resource for applied conservation and evolution. Evolutionary Applications 3: 244–262. DOI: 10.1111/j.1752-4571.2009.00104.x.

Whiteley, A.R., Coombs, J. a., Hudy, M., Robinson, Z., Nislow, K.H., and Letcher, B.H. 2012. Sampling strategies for estimating brook trout effective population size. Conservation Genetics 13: 625–637. doi: 10.1007/s10592-011-0313-y.

Whitman, T. 2002. Crooked River watershed assessment. Prineville, Oregon.

Williams, R. N., R. F. Leary, and K. P. Currens. 1997. Localized genetic effects of a long-term hatchery stocking program on resident rainbow trout in the Metolius River, Oregon. North American Journal of Fisheries Management 14:1079–1093.

WPN 2002. Little Deschutes River Subbasin assessment. Watershed Professionals Network, Boise, Idaho.

Yake, K. E. 2003. Upper Deschutes Subbasin assessment. Bend, Oregon.

Zimmerman, C.E. and G.H. Reeves. 2000. Population structure of sympatric anadromous and nonanadromous *Oncorhynchus mykiss*: evidence from spawning surveys and otolith microchemistry. Canadian Journal of Fisheries and Aquatic Sciences 57:2152-2162.

OREGON

Fish & Wildlife