Research, Monitoring, and Evaluation of Emerging Issues and Measures to Recover the Snake River Fall Chinook Salmon ESU

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Abstract

The portion of the Snake River fall Chinook Salmon *Oncorhynchus tshawytscha* ESU that spawns upstream of Lower Granite Dam transitioned from low to high abundance during 1992–2016 in association with U.S. Endangered Species Act recovery efforts and other federally mandated actions. This annual report focuses on (1) numeric and habitat use responses by natural- and hatchery-origin spawners, (2) phenotypic and numeric responses by natural-origin juveniles, and (3) predator responses in the Snake River upper and lower reaches as abundance of adult and juvenile fall Chinook Salmon increased. Spawners have located and used most of the available spawning habitat and that habitat is gradually approaching redd capacity. Timing of spawning and fry emergence has been relatively stable; whereas the timing of parr dispersal from riverine rearing habitat into Lower Granite Reservoir has become earlier as apparent abundance of juveniles has increased. Growth rate (g/d) and dispersal size of parr also declined as apparent abundance of juveniles increased. Passage timing of smolts from the two Snake River reaches has become earlier and downstream movement rate faster as estimated abundance of fall Chinook Salmon smolts in Lower Granite Reservoir has increased. In 2016, we described estimated the consumption rate and loss of subyearlings by Smallmouth Bass before, during, and after four hatchery releases. Before releases, Smallmouth Bass consumption rates of subyearling was low (0–0.36 fish/bass/d), but the day after the releases consumption rates reached as high as 1.6 fish/bass/d. Bass consumption in the upper portion of Hells Canyon was high for about 1–2 d before returning to pre-release levels, but in the lower river consumption rates were reduced but took longer to return to pre-release levels. We estimated that most of the subyearlings consumed by bass were of hatchery origin. Smallmouth Bass predation on subyearlings is intense following a hatchery release, but the predation pressure is relatively short-lived as subyearlings quickly disperse downstream. This information will allow us to better estimate subyearling loss to predation from our past efforts at time intervals less than 2 weeks. These findings coupled with stock-recruitment analyses presented in this report provide evidence for density-dependence in the Snake River reaches and in Lower Granite Reservoir that was influenced by the expansion of the recovery program. The long-term goal is to use the information covered here in a comprehensive modeling effort to conduct action effectiveness and uncertainty research and to inform Fish Population, Hydrosystem, Harvest, Hatchery, and Predation and Invasive Species Management RM&E.

Introduction

The ISAB (2015) wrote "Understanding density dependence—the relationship between population density and population growth rate—is important for effective implementation of the Columbia River Basin Fish and Wildlife Program, biological opinions, recovery plans, and tribal programs. Information on how density dependence limits fish population growth and habitat carrying capacity is vital for setting appropriate biological goals to aid in population recovery, sustain fisheries, and maintain a resilient ecosystem. Habitat restoration and population recovery actions can be planned and implemented more effectively by understanding mechanisms that cause density dependence in particular cases, such as limited food supply, limited rearing or spawning habitat, or altered predator-prey interactions."

Management efforts have been implemented in response to listing under the Endangered Species Act (ESA; NMFS 1992) to increase the size of the population and survival of Snake River basin fall Chinook Salmon *Oncorhynchus tshawytscha* (e.g., reduced harvest, Peters et al. 2001; stable minimum spawning flows, Groves and Chandler 1999; summer flow augmentation, Connor et al. 2003b; predator control, Beamesderfer et al. 1996; increased hatchery production and supplementation; improved dam passage structures, Rainey et al. 2006; summer spill operations, CBR 2015). To track changes in attributes of the natural-origin population as abundance increased, Connor et al. (2013) divided the years 1991–2011 into periods of low and high abundance. To track changes in the attributes of spawning, rearing, emigration, and predation in this report, we added the years 2012 through 2016 to the period of record. The low and high abundance periods for adults were set at 1991–1998 and 1999–2016. The low and high abundance periods for juveniles were set at 1992–1999 and 2000–2016. Estimated escapement of natural- and hatchery-origin spawners upstream of Lower Granite Dam (hereafter, total

escapement) increased markedly between the two abundance periods reaching a post-ESA listing high of 52,989 in 2014 and 52,338 in the most recent year 2016 (Table 1).

To assist with the monitoring of recovery measures, staff of project 199102900 have collected and analyzed data on adult and juvenile fall Chinook Salmon collected along the lower Snake River upper and lower reaches to the tailrace of Lower Granite Dam (Figure 1) since brood year 1991 (fry emergence year 1992). That project functions in the long-term as a research project by publishing papers to help to answer uncertainty and action effectiveness questions, while reporting interim information on status and trends as the data are collected. Predation by nonnative fishes is one factor that has been implicated in the decline of juvenile salmonids *Oncorhynchus* spp. in the Pacific Northwest, but it has been scantly studied in the case of Snake River fall Chinook Salmon. The only evaluation of predation on subyearling Snake River fall Chinook Salmon in the Snake River upper and lower reaches was conducted by Nelle (1999). Within the upper reach, Nelle (1999) reported that subyearlings only made up 1.9% and 0.8% of Smallmouth Bass diets by weight in 1996 and 1997, respectively. That study was conducted during the low abundance period soon after the Snake River fall Chinook Salmon ESU was listed under the ESA in 1992. Thus, low abundance of fall Chinook Salmon could explain why Smallmouth Bass consumption rates were relatively low compared to those from studies conducted in the Columbia and Yakima rivers where salmon abundance was higher (Tabor et al. 1993; Fritts and Pearsons 2004).

To date, our predation work has focused on estimating the loss of subyearling fall Chinook Salmon to Smallmouth Bass predation in Hells Canyon. Our methodology relies on estimating Smallmouth Bass consumption rate of subyearlings and then expanding that rate by bass abundance in a reach over a specific sampling interval, which was two weeks in our past

studies. Thus an assumption of this method is that consumption rate does not vary over this time interval. However, the assumption is violated when large hatchery releases occur and bass consumption rate changes within the two-week interval. Without knowing how the consumption rate changes when a hatchery release occurs, estimates of subyearling loss can be either over or under estimated. In 2016, we attempted to better understand changes in bass consumption rates before, during, and after hatchery releases so as to correct our results when such events occur. An additional objective was to determine the number of hatchery fish consumed in our study area during each release.

TABLE 1. Estimates of escapement of natural- and hatchery-origin adult (≥ 53 cm FL) fall Chinook Salmon from the Snake River basin ESU, 1991–2016. Reference the following for the results: Busack (1991), Cooney (1991), LaVoy (1992, 1993, 1994, 1995), LaVoy and Mendel (1996), Mendel and LaVoy (1997), Mendel (1998, 1999, 2000), Young et al. (2012, unpublished). The mean (± SD) escapements for the abundance period estimates (Low, 1991–1998; High, 1999–2016) are also given.

Year	Natural	Hatchery	Total
1991	318	253	571
1992	549	111	660
1993	742	195	937
1994	406	186	592
1995	350	267	617
1996	639	260	899
1997	797	195	992
1998	306	610	916
1999	905	890	1,795
2000	1,148	1,410	2,558
2001	5,163	4,382	9,545
2002	2,116	7,231	9,347
2003	3,455	8,974	12,429
2004	2,637	9,773	12,410
2005	4,584	5,340	9,924
2006	3,984	2,501	6,485
2007	2,816	5,538	8,354
2008	2,995	8,930	11,925
2009	4,273	16,412	20,685
2010	7,347	32,417	39,764
2011	8,072	15,508	23,580
2012	11,315	19,048	30,363
2013	20,425	30,813	51,238
2014	13,141	39,848	52,989
2015	15,422	36,916	52,338
2016	8,762	23,110	31,872
Low	513 ± 183	260 ± 141	773 ± 167
High	6,587 ± 5,206	14,947 ± 12,182	21,533 ± 16,743

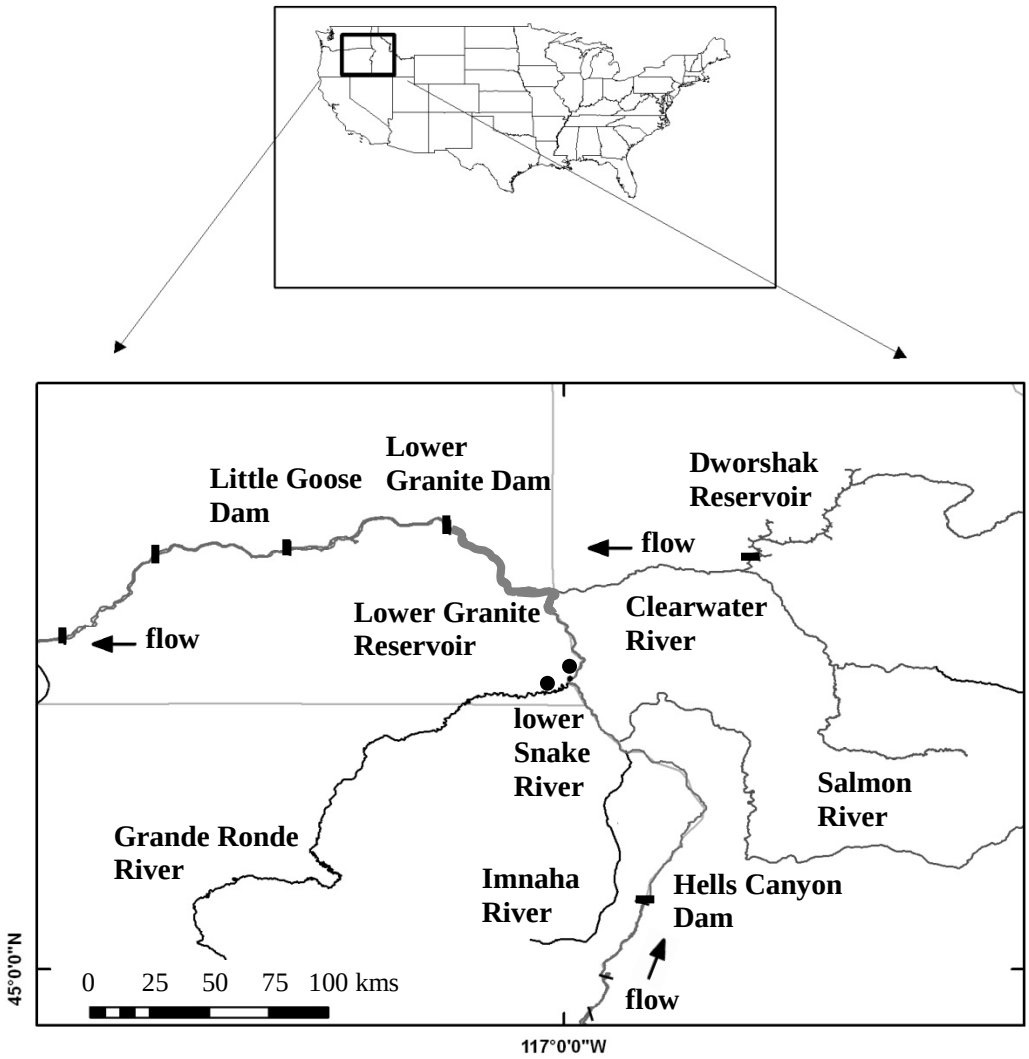

FIGURE 1. The Snake River basin including the free-flowing but regulated Snake River upper (Hells Canyon Dam to Salmon River mouth) and lower (Salmon River mouth to the upper end Lower Granite Reservoir) reaches where natural- and hatchery-origin adults spawned during 1991–2016 and natural-origin fall Chinook Salmon subyearlings were captured seined and PIT tagged while rearing during 1992–2016. Lower Granite Reservoir and Lower Granite Dam are the first of the impoundments and dams encountered by the fish after they had become smolts during early seaward migration.

The goal of this annual report is to describe how the status and trends in the data collected on fall Chinook Salmon in the Snake River upper (Hells Canyon Dam to the Salmon River mouth) and lower (Salmon River mouth to upper end of Lower Granite Reservoir; Figure 1) reaches provide evidence for density-dependent interactions including an update on the response to predation. The objectives of this report are to summarize information collected on attributes of (1) spawning in riverine habitat, (2) natural-origin juveniles rearing in riverine habitat, (3) natural-origin juveniles emigrating through Lower Granite Reservoir, and (4) seasonal variation in Smallmouth Bass diets and consumption of subyearling Chinook Salmon during rearing in riverine habitat.

Methods

Attributes of Spawning in Riverine Habitat (Protocol ID 2058, Published)

Aerial surveys were scheduled at 7-d to 14-d intervals starting in late October and ending in early December. The number of flights made varied by year. Redds were counted from a helicopter flown about 200 m above the river, which allows observing 100% of the river bottom at depths approximately < 3 m (i.e., shallow water). Beginning in 2015, aerial surveys were replaced with surveys conducted by staff of the Idaho Power Company by use of a small unmanned aircraft system (sUAS). For the sUAS survey samples, 35 individual sites were flown four times (every other week) during the spawning season, beginning the week of 10/26, and ending during the week of 12/07. The total number of new redds was compiled for each of the 35 sample sites, for each survey, and that provided a total number of redds present at each site that were incorporated into a sampling proportional to size model to estimate (95% C.I.) total

redd counts for each Snake River reach. The efficacy of the sUAS surveys is described by Groves et al. 2016). Problems associated with sampling error in 2015 were resolved in 2016.

Potential deep water spawning locations (>3 m) were searched for redds as described in Protocol ID 2058 and by Groves et al. (2013). We tabulated the redd counts by year and reach. We calculated inter-annual mean redd counts by abundance period. To evaluate the evidence for density-dependence during spawning, we plotted the annual numbers of shallow redds counted (or estimated after 2014), number of shallow sites used (restricted to years prior to 2015), deep redds counted, and deep sites used against the corresponding brood year estimates of total escapement (Table 1). Plots were made for the reaches jointly. To assess the plausibility and strength of density-dependence during spawning, we assumed a log-normal error structure and fit three types of linear models. First, in the following manner we fit a simple linear regression model so that recruitment (*R*; e.g., aerial redd counts) increased proportionately and indefinitely (no density-dependence) with increases in total escapement (*S*):

(1) $$R_{t+1} = \theta_0 + \theta_1 S_t \, .$$

Secondly, we fit the Beverton-Holt model:

(2) $$R_{t+1} = \frac{\alpha S_t}{1 + (\beta S_t)} \, ,$$

whereby α measures productivity when the number of spawning adults is near zero (i.e., density-independence), and β measures the strength of density-dependence as the number spawning adults increases. The Beverton-Holt model reaches an asymptote in the number recruits produced at α/β.

Lastly, we fit the Ricker production model expressed as:

(3) $$R_{t+1} = \rho S_t e^{-\delta S_t},$$

where ρ is similar to α in the Beverton-Holt and measures density-independent productivity (proportional to fecundity) and δ measures the strength of density-dependence. We used Akaike's Information Criterion (AIC) to pick the model with the best fit.

To evaluate the time of spawning, if redds were counted during a given aerial survey we assigned a spawning date to that survey by subtracting seven days from the flight date. For example, if the first flight was made on 10/21 and 10 redds were counted, the first spawning date would be 10/14. We calculated the first, peak, and last spawning dates as just described and calculated the percentages of the total redd count made on those dates. We calculated inter-annual means for the first, peak, and last spawning dates and for the percentages on those dates by abundance period. Analyses on time of spawning analyses were restricted to the years 1992–2014 because the flight schedules for the sUAS surveys that began in 2015 were shifted to slightly later dates and the resulting count data were not directly comparable to the data collected in prior years.

Attributes of Natural-Origin Juveniles Rearing in Riverine Habitat (Protocol ID 2057, Published)

Attributes evaluated during rearing in riverine habitat included apparent abundance of natural-origin subyearlings, timing of fry (< 46-mm FL) presence, and timing of parr (> 45-mm FL; after Connor et al. 2002) dispersal from riverine habitat, parr dispersal size, and parr growth. To collect data on those attributes, we used a beach seine at 11–15 permanent stations located along 142 contiguous kms of the two riverine reaches studied. Large portions of the hatchery

smolts released into the river were released without an external mark or fin clip (i.e., unmarked). After hatchery smolts were released upstream of a given seining station, the origin (i.e., natural or hatchery) of each collected unmarked fish was classified based on morphology (overall accuracy 98.7%; Tiffan and Connor 2011). During 1992–2007, we implanted each natural-origin parr ≥ 60 mm long collected during seining in riverine habitat with an 11.5-mm passive integrated transponder (PIT) tag (Prentice et al. 1990a). During 2008–2016, in addition to tagging fish longer than 59 mm with 11.5-mm tags, we also tagged natural-origin parr that were 50–59 mm long with 8.5-mm tags. Natural-origin parr were given 15 min to recover from tagging in an aerated 19-L bucket of river water before release at their capture site.

Mean daily CPUE (natural-origin subyearlings per seine haul) values for each sampling station were adjusted for the presence of natural-origin, spring/summer Chinook Salmon and averaged across sampling stations and weeks within a year to calculate mean annual CPUE (± SE) by reach and for reaches combined as an indices of apparent abundance in riverine rearing habitat, where *N* was the number of station visits. We also calculated mean inter-annual CPUE by abundance period (low abundance period, 1992–1999; high abundance period, 2000–2016; after Connor et al. 2013). To evaluate the evidence for density-dependence during rearing in riverine habitat, mean annual CPUE for the combined reaches was plotted against the corresponding brood year estimates of total escapement (Table 1) and curves were fit to the data as described for redd counts.

The median, minimum, and maximum day of year (January 1 = 1) of fry and parr presence were tabulated by reach and year. We calculated inter-annual means from the annual median dates of presence for each abundance period under the premise that dispersal timing into the reservoir became earlier as the median dates of parr presence became earlier. We calculated

annual mean and abundance period wet weights (0.1 g) of all natural-origin parr captured in the riverine habitat reasoning that decreases in weight reflected decreases in size at dispersal (and vice versa). We calculated absolute growth rates (g/d) of individual PIT-tagged, natural-origin parr recaptured by beach seine as (WT2 – WT1) /(Day2 – Day1) and used those growth rates to calculate annual mean growth rates for fish recaptured within each year, and inter-annual mean growth rates for each abundance period.

Attributes of Natural-Origin Juveniles Emigrating through Lower Granite Reservoir (Protocol ID 2057,Published)

The basin-wide population of subyearling smolts consists of the aggregate of natural- and hatchery-origin fish produced or released upstream of Lower Granite Reservoir. We modified Method ID 3999 (Published) to estimate the daily number of natural-origin, fall Chinook Salmon subyearling smolts that passed Lower Granite Dam each year. We summed the daily estimates made for March through October to estimate seasonal passage abundance. To evaluate the evidence for density-dependence during early seaward migration in the reservoir, we plotted estimated annual passage abundance at the dam against the corresponding brood year estimates of total escapement (Table 1) and curves were fit to the data as described for redd counts..

We also used Method ID 3999 (Published) to estimate daily passage abundance of natural-origin subyearlings at Lower Granite Dam that had been PIT tagged in the Snake River reaches. The annual median passage dates for each reach were plotted against year, and the inter-annual means of those medians were calculated by abundance period. Downstream movement rate was calculated for individual, natural-origin subyearlings that had been PIT tagged while rearing in riverine reaches of the Snake River as the elapsed days between release

and detection at a point downstream divided by the channel distance in river kms traversed between release and subsequent detection at Lower Granite Dam. Annual means, medians, minimums, and maximums were calculated by reach and abundance period using the downstream movement rates of individual fish.

Variation in Smallmouth Bass Diets and Consumption of Subyearling Chinook Salmon during Rearing in Riverine Habitat (Protocol ID 299, Published)

We examined the daily consumption and diet of Smallmouth Bass before and during four hatchery releases of subyearling fall Chinook Salmon within Hells Canyon, an unimpounded section of the Snake River (Figure 2). These releases occur annually within our study area. The first two hatchery releases we studied in 2016 were made at Hells Canyon Dam (rkm 398.5) during which 1,041,185 fish were released over 2 days (May 16 and 18; Table 2). A third release of 398,086 subyearlings was made on May 20 from the Pittsburg Landing acclimation facility (rkm 345.5). To evaluate these three releases, we first divided our study area into two reaches: one downstream of Pittsburg Landing from rkm 328.1 to 345.5, and one upstream from rkm 345.5 to 365.3. We began sampling in both reaches on May 15, the day before the first release at Hells Canyon Dam. Sampling continued in each reach until subyearlings were no longer observed (estimated in the field) in the diets of bass. The fourth hatchery release we studied was made on June 10 at the Captain John acclimation facility (rkm 262.8) during which 198,983 fish were released. We sampled the river from the acclimation facility downstream to the head of Lower Granite Reservoir at Asotin, WA (rkm 234.2). Sampling began the morning before the release and continued until the presence of subyearlings in bass stomachs were similar to prerelease levels.

Smallmouth Bass ≥150mm TL were collected by angling to evaluate consumption for the Hells Canyon Dam and Pittsburg Landing releases and by boat electrofishing to evaluate the Captain John release. Different gears were used to be consistent with the methods we used during our previous consumption studies. All sampling sites were randomly selected according to shoreline habitat type that was provided as a GIS habitat layer by the Idaho Power Company. The following shoreline habitat types were sampled: pool, riffle, glide, bar, and fan. We did not sample "rapids" because they represented a relatively small (<5%) amount of shoreline and were difficult to sample. Sites that were angled were sampled for ~20 min by 2 people fishing from a boat. Sites that were electrofished were sampled along 80- to 800-m transects that typically encompassed an entire specific habitat type. One netter collected fish.

Consumption

We calculated the consumption rate *C* (number of Chinook Salmon/Smallmouth Bass/day) in a series of steps similar to Fritts and Pearsons (2004). First, we identified ingested fish from bass stomachs using diagnostic bones (i.e., dentary, cleithrum, opercle; Parrish et al. 2006) and estimated their original FL and weight at ingestion using various regressions (Hansel et al. 1988; Vigg et al. 1991; Parrish et al. 2006). These weights were summed with weights of other diet items (if present) to derive a meal weight (*MW*) for each individual bass (Vigg et al. 1991). We accounted for a 21.3% weight loss associated with preservation in 90% ethanol for all diet items used in the calculation (Shields and Carlson 1996). Next, we input *MW* into an evacuation rate model of Smallmouth Bass digestion of salmonids developed by Rogers and Burley (1991) and modified by Fritts and Pearsons (2004) that predicts time (in hours) to 90% evacuation (ET_{90}):

$$ET_{90} = (24.542)(MW^{0.29}e^{-0.15T}W^{-0.23})(24),$$

where W is bass weight (g; measured or estimated), and T is temperature. Finally, we calculated C for each individual bass using the equation presented by Ward et al. (1995):

$$C = n(24/ET_{90}),$$

where n is the number of Chinook Salmon found in the bass stomach. Mean C was calculated for each reach per day from all bass examined including those with empty stomachs.

We estimated total Chinook Salmon loss to predation by multiplying daily estimates of C to abundance estimates from our previous predation research. Mean abundance during May of 2013 and 2014 equated to 1,089 Smallmouth Bass/rkm in the upper portion of our study area and mean June abundances equated to 896/rkm in the lower portion of our study area (Connor et al. 2015). Applying this to our study area equated to 21,562 bass in our reach above Pittsburg Landing, 18,949 bass in our reach below Pittsburg Landing, and 25,626 in our reach below Captain John.

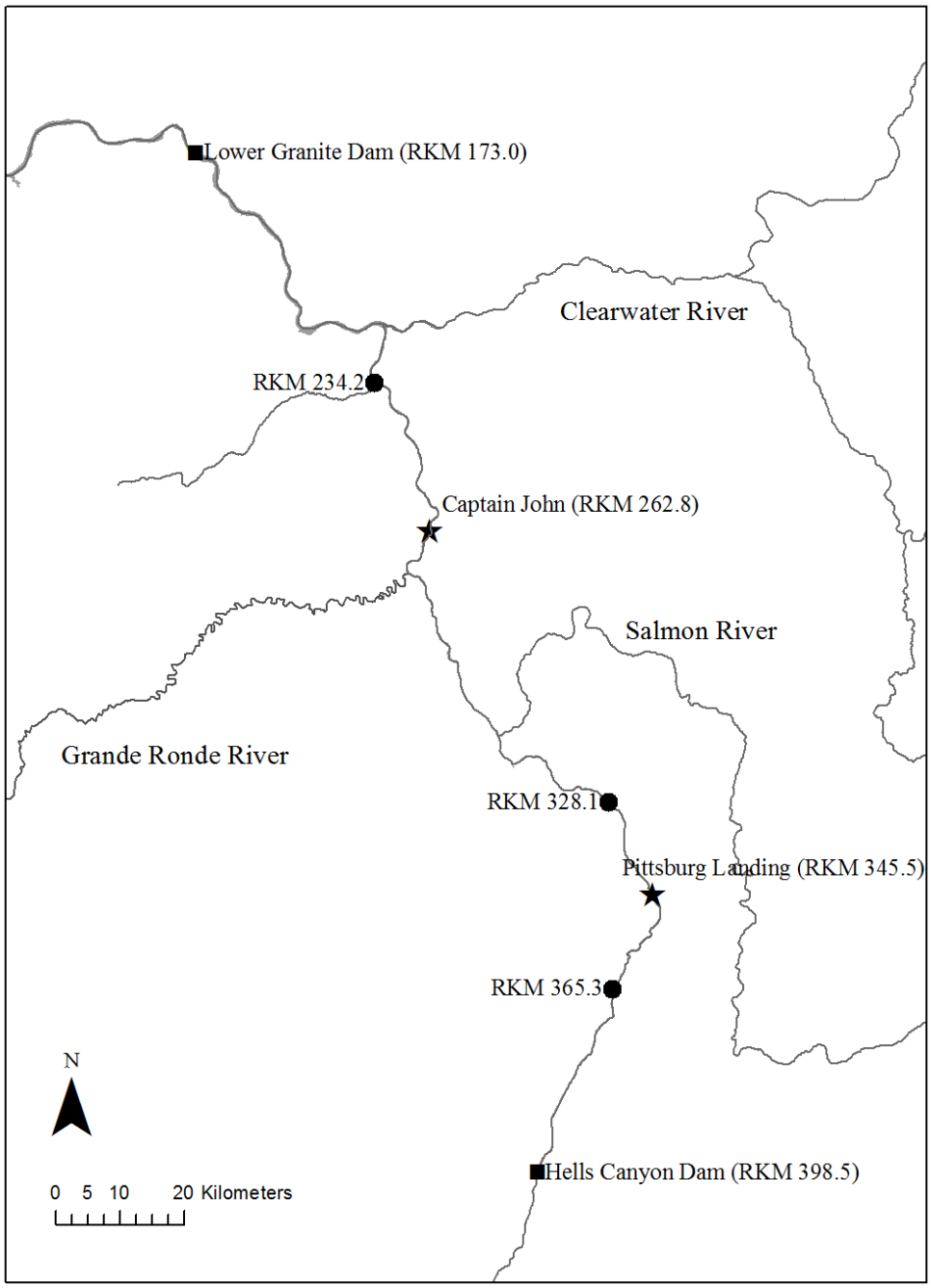

FIGURE 2. Map of Hells Canyon showing the upper (rkm 328.1–365.3) and lower (rkm 234.2–262.8) portions of our study area that were sampled in 2016.

TABLE 2. Summary of hatchery releases of fall Chinook Salmon in the Snake River during 2016.

Date	Location	Release agency	Rearing location	Number released	Number PIT tagged	Number coded-wire tagged
			Releases studied			
May 16-18	Hells Canyon Dam	Idaho Power Company	Irrigon	1,041,185	2,998	247,407
May 20	Pittsburg Landing	Nez Perce Tribe	Lyons Ferry	398,086	26,052	199,287
			Other releases in study area			
May 25	Captain John	Nez Perce Tribe	Lyons Ferry	509,235	25,973	200,348
May 31	Grande Ronde River	Idaho Power Company	Irrigon	429,889	3,000	200,075
June 10	Captain John	Nez Perce Tribe	Lyons Ferry	198,983	2,000	197,857

Results

Attributes of Spawning in Riverine Habitat

The total number of redds in the Snake River upper reach (estimated shallow and counted deep combined) was 1,381 in 2016 (Table 3), which ranked sixth for the 26-year period of record for that reach. The total number of redds in the Snake River lower reach was 593 in 2016 (Table 3), which ranked eleventh for the 26-year period of record for that reach. The grand total redd count for both reaches combined in 2016 was 1,974 (Table 3), which ranked eighth for the 26-year period of record for the free-flowing Snake River.

Mean total redd counts were similar between reaches during the low abundance period, whereas the mean total redd count was 1.6 times higher in the upper reach compared to the lower reach during the high abundance period (Table 3). As such, there was a large change in the percentage of the total redds counted between reaches and abundance periods with the upper reach supporting the majority of redds during the high abundance period (Table 3).

The Beverton-Holt (AIC = 351.8) provided more information about the shape of the relation between total escapement and aerial redd counts for the reaches combined compared to Ricker (AIC = 355.1) and linear (AIC = 368.9) models. The Beverton-Holt model provided the best fit (AIC = 178.6) to the data set composed of total escapement and the number of shallow spawning sites used (Linear, AIC = 229.5; Ricker, AIC = 201.9; Figure 4).

The fits to the total escapement and deepwater redd count data were similar between the Beverton-Holt (AIC = 329.8) and Ricker (AIC = 330.1) models, but better than the fit of the linear (AIC = 344.6) model (Figure 5). The Beverton-Holt model (AIC = 180.2) provided the best fit to data collected on the total escapement and the number of deepwater sites used (Linear, AIC = 206.6; Ricker, AIC = 184.3; Figure 6).

Because of the nature of the sUAS surveys, spawn timing statistics could not be calculated in 2015 using the approach applied to previous years. It can be said that spawning in the Snake River in 2015 began around the second week of October, peaked in late October to early November, and was complete by the middle of December. Based on the data collected during the 1991–2014 standardized aerial surveys, there was a 7-d difference between the mean first dates of spawning of the two abundance periods, that was partly caused by starting the aerial surveys early during several years of the high abundance period (Table 4). On average, however, relatively low proportions of the total number of redds counted within each abundance period

were counted on the first date of spawning (Table 4). There was no large difference between the mean peak and last dates of spawning between abundance periods (Table 4). Thus, difference in time of spawning between abundance periods was not a large factor for changes in the attributes of juveniles described later in this report.

TABLE 3. Redd count data collected during aerial surveys (< 3 m deep; 1991–2016), sUAS surveys (< 3 m deep; 2015 and 2016) and deepwater searches (> 3 m deep; 1993–2016) conducted along the Snake River upper and lower reaches, 1991–2016. The mean (± SD) total counts and mean (± SE) percentages by reach are also given by abundance period (Low, 1991–1998; High, 1999–2016).

	Upper reach			Lower reach				Percent by reach	
Year	Aerial	Deep	Total	Aerial	Deep	Total	Grand total	Upper	Lower
1991	27		27	24	5	29	56	48.2	51.8
1992	16		16	31	0	31	47	34.0	66.0
1993	14	5	19	46	62	108	127	15.0	85.0
1994	29	6	35	22	8	30	65	53.8	46.2
1995	28	5	33	13	19	32	65	50.8	49.2
1996	49	7	56	22	26	48	104	53.8	46.2
1997	20	4	24	29	5	34	58	41.4	58.6
1998	109	28	137	26	22	48	185	74.1	25.9
1999	225	67	292	48	33	81	373	78.3	21.7
2000	186	42	228	74[a]	49	123	351	65.0	35.0
2001	301	87	388	234	86	320	708	54.8	45.2
2002	533	114	647	345	120	465	1,112	58.2	41.8
2003	675	165	840	455	229	684	1,524	55.1	44.9
2004	685	279	964	533	210	743	1,707	56.5	43.5
2005	662	203	865	380	195	575	1,440	60.1	39.9
2006	452	147	599	244	181	425	1,024	58.5	41.5
2007	482	241	723	232	162	394	1,117	64.7	35.3
2008	761	368	1,129	472	218	690	1,819	62.1	37.9
2009	948	379	1,327	563	205	768	2,095	63.3	36.7
2010	1,110	641	1,751	840	375	1,215	2,966	59.0	41.0
2011	874	521	1,395	1,075	344	1,419	2,814	49.6	50.4
2012	1,100[b]	274	1,374	594[c]	142	736	2,110	65.1	34.9
2013	1,338[d]	332	1,670	1,209[e]	264	1,473	3,143	53.1	46.9
2014	1,949[f]	583	2,532	1,029[g]	259	1,288	3,820	66.3	33.7
2015	1,738[h]	589	2,327	591[i]	237	828	3,155	73.8	26.2
2016	1,043[j]	338	1,381	445[k]	148	593	1,974	70.0	30.0
Low			43 ± 37	45 ± 25			88 ± 45	46.4 ± 16.1	53.6 ± 16.1
High			1,135 ± 634	712 ± 399			1,847 ± 982	61.9 ± 1.7	38.1 ± 1.7

[a]Groves et al. (2013) did not report 5 redds counted at rkm 238.1; [b]1,016/0.924 based on observed helicopter counts divided by counts made with a UAS;[c]396/0.667; [d]1,220/0.912; [e]851/0.704;[f]1,129/0.579;[g]837/0.813;[h]± 670 redds;[i]± 222 redds;[j]± 96redds; [k]± 138redds.

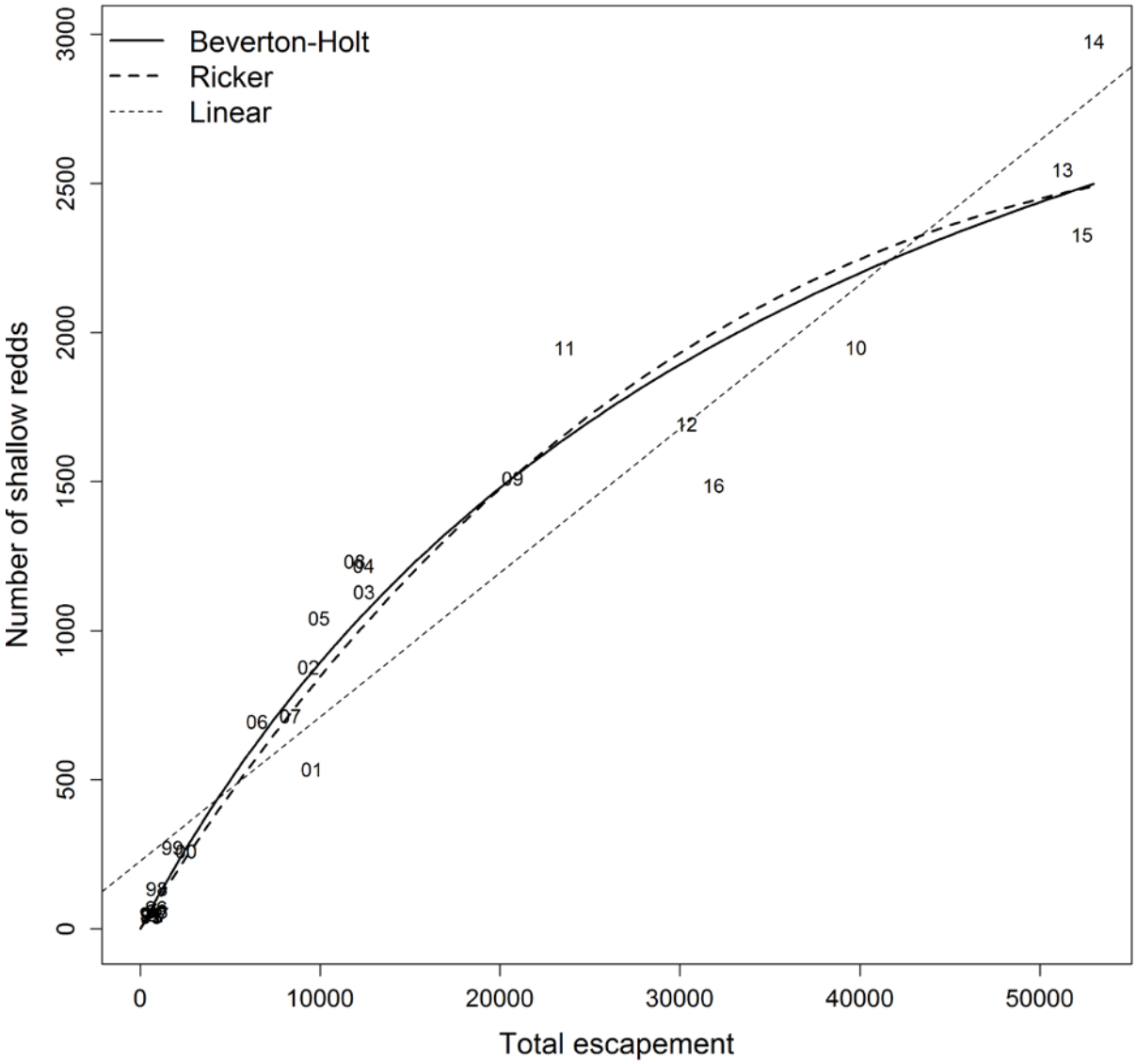

FIGURE 3. The relation between estimated annual total escapement (Table 1) and the annual number of redds counted (1992–2010) counted and adjusted for inaccuracy (2012–2014), and estimated (2015 and 2016) based on manned helicopter (1992–2014) and unmanned aircraft flights over shallow (< 3 m deep) spawning sites along the Snake River upper and lower reaches combined (Table 3), 1991–2016. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt and Ricker models (of similar fit) were: alpha = 0.11293±0.01244 and beta = 0.000026±0.000006; and alpha = 0.09704±0.00815 and beta = 0.000014±0.0000021.

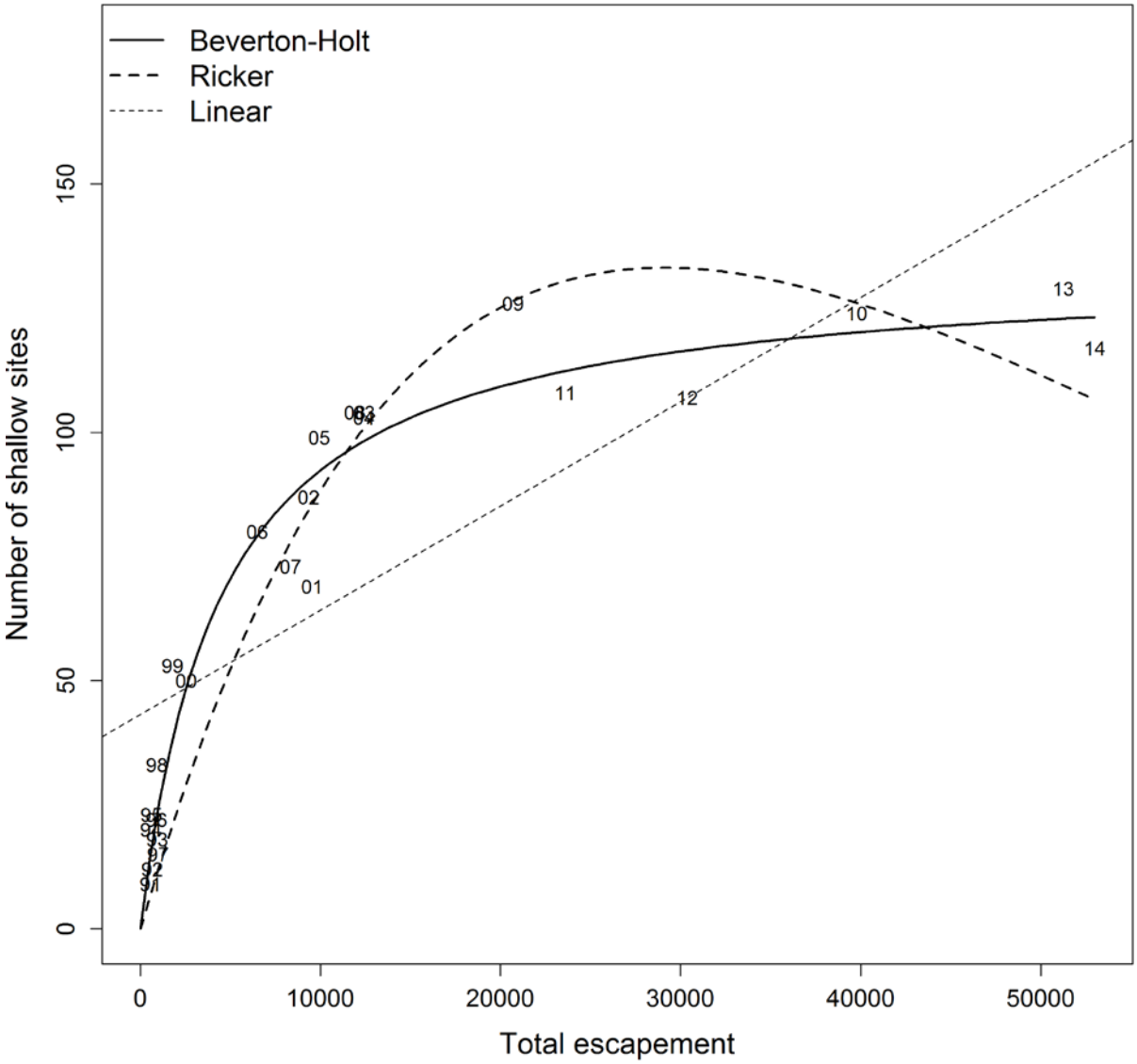

FIGURE 4. The relation between estimated annual total escapement (Table 1) and the annual number of shallow (< 3 m deep) sites at which redds were counted from the air in the Snake River upper and lower reaches combined, 1991–2014. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt model (the best fit) were alpha = 0.029912±0.0036 and beta = 0.000224±0.00003.

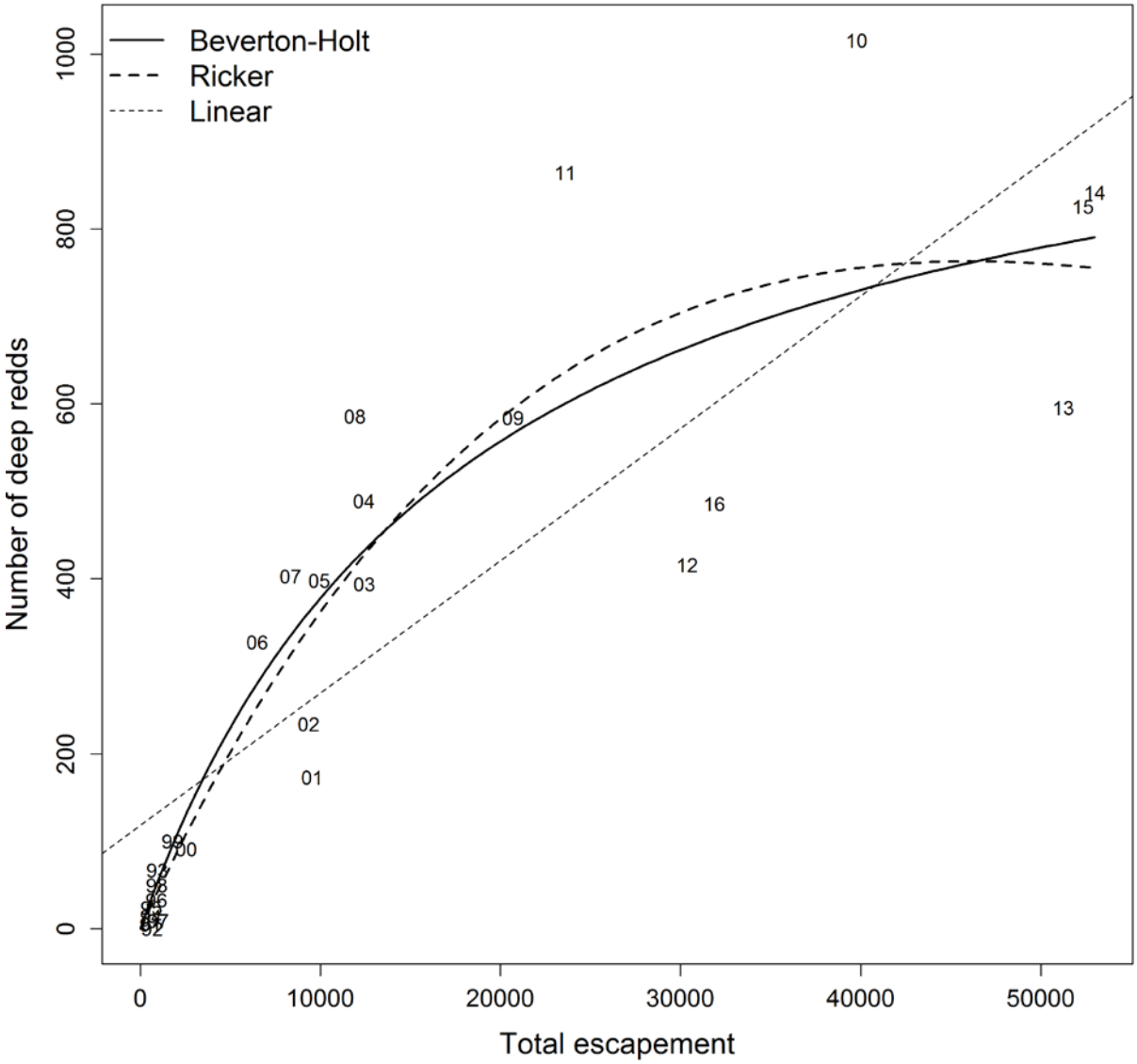

FIGURE 5. The relation between estimated annual total escapement (Table 1) and the annual number of redds counted at sites > 3 m deep with underwater video along the Snake River upper and lower reaches combined (Table 3), 1991–2016. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt and Ricker models (of similar fit) were: alpha = 0.058803±0.0134256 and beta = 0.0000555±0.0000203; and alpha = 0.0450296±0.00589and beta = 0.000022±0.000004.

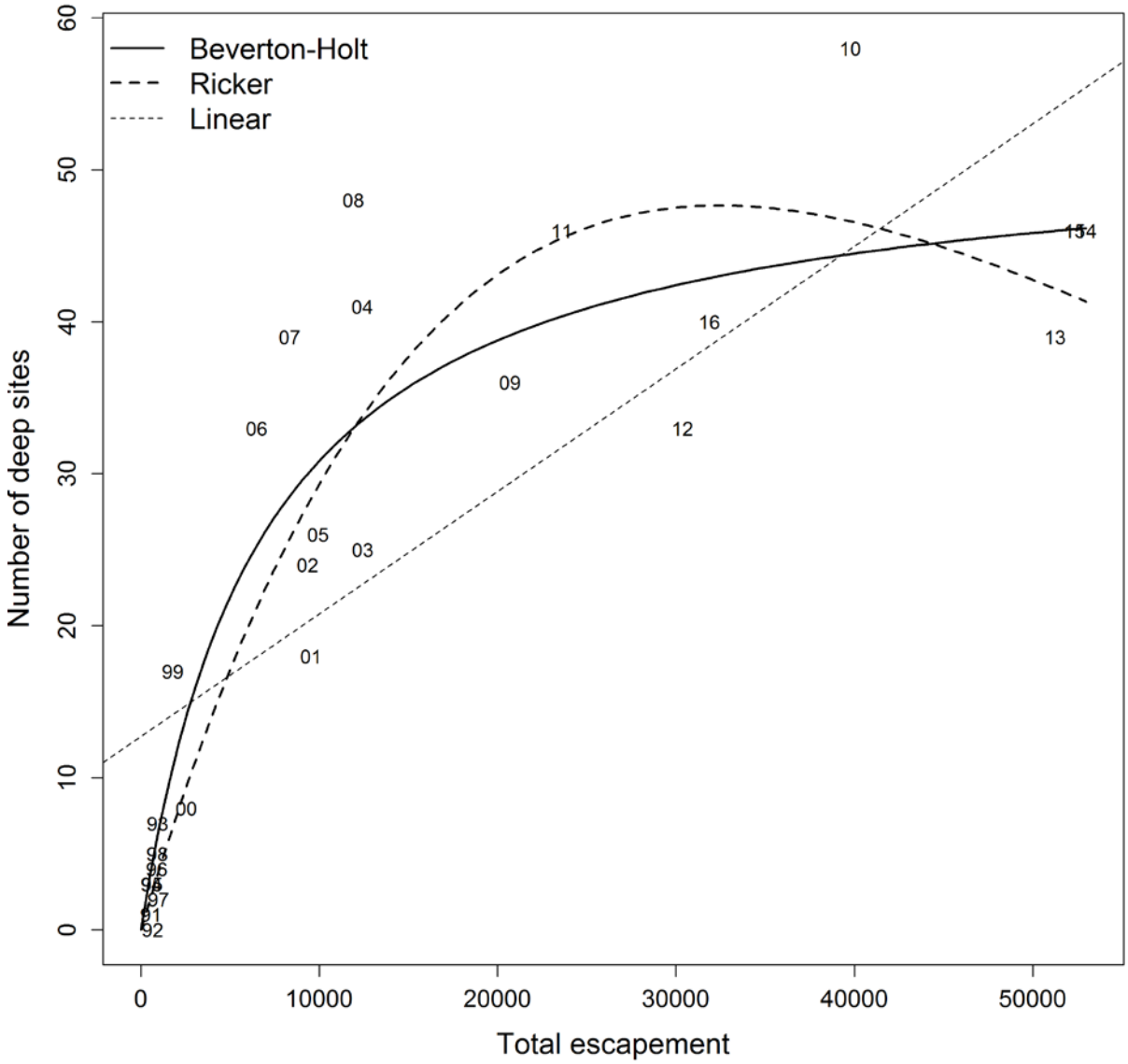

FIGURE 6. The relation between estimated annual total escapement (Table 1) and the annual number of deep (> 3 m deep) sites at which redds were counted with underwater video in the Snake River upper and lower reaches combined, 1991–2016. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt model (the best fit) were alpha = 0.00755±0.0018 and beta = 0.000145±0.000046.

TABLE 4. Information from the annual aerial surveys made to count redds in shallow water (< 3 m) along the Snake River upper and lower reaches including the number of flights made, the spawning date represented by the first flight (flight date minus 7 d; Earliest), and the first, peak, and last spawning dates (flight dates minus 7 d) observed including the cumulative percentages (%) of eventual total redd counts made on those dates. The means (± SD) of the annual abundance period (Low, 1991–1998; High, 1999–2014) first, peak and last dates redds were counted are also given. sUAS survey data collected in 2015 and 2016 were not comparable to early years with regards to evaluating time of spawning.

	Flights		First		Peak		Last	
Year	*N*	Earliest	Date	%	Date	%	Date	%
1991	9	10/07	10/21	2.2	11/11	26.2	12/02	4.8
1992	8	10/09	10/29	8.5	11/16	38.3	12/05	6.4
1993	8	10/18	10/18	1.7	10/25	30.0	12/06	3.3
1994	8	10/17	10/17	2.0	10/31	35.3	11/28	2.0
1995	7	10/16	10/16	9.8	10/30	43.9	11/20	2.4
1996[a]	7	10/14	10/14	2.8	10/21	36.6		
1997	8	10/13	10/13	2.0	10/20	32.7	11/24	2.0
1998	8	10/12	10/19	20.7	10/26	31.9	11/23	3.7
1999	9	10/04	10/11	1.8	11/04	43.6	11/30	0.4
2000	9	10/02	10/02	0.4	10/23	31.4	11/21	2.0
2001	10	10/02	10/02	1.7	11/06	39.8	12/03	1.5
2002	7	10/14	10/14	3.4	10/28	33.1	11/25	0.3
2003	7	10/13	10/13	0.4	11/03	37.4	12/01	0.5
2004	8	10/11	10/11	0.1	11/01	40.5	11/29	1.1
2005	9	10/11	10/11	0.8	10/31	32.5	12/05	0.1
2006	6	10/16	10/16	5.0	10/23	39.3	11/27	1.3
2007	8	10/08	10/08	0.1	10/30	39.5	11/26	6.0
2008	8	10/13	10/13	1.9	10/27	46.0	12/01	0.5
2009	8	10/12	10/12	1.2	10/26	46.4	11/30	0.9
2010	4	10/18	10/18	3.2	11/02	51.5	11/28	6.9
2011	5	10/17	10/17	1.7	10/31	47.5	11/28	6.7
2012	4	10/15	10/15	2.4	10/29	68.5	11/26	1.8
2013	4	10/14	10/14	5.0	10/28	75.6	11/27	0.8
2014	4	10/13	10/13	1.9	10/27	53.8	11/24	2.4
Low		10/14±4	10/19±5	6.2±6.3	10/31±9	34.1±5.6	11/28±6	3.1±1.5
High		10/12±5	10/12±4	1.9±1.5	10/30±4	45.4±11.9	11/28±3	2.2±2.2

[a]Counts were prohibited by turbidity during the last three flights.

Attributes of Natural-Origin Juveniles Rearing in Riverine Habitat

Mean (± 95% C.L.) annual CPUE for natural-origin Snake River fall Chinook Salmon subyearlings along the Snake River upper reach during 2016 was 37.1 ± 21.5 fish per seine haul (Table 5), which ranked seventh for the 22-year period of record for the upper reach. Mean (± 95% C.L.) annual CPUE for natural-origin Snake River fall Chinook Salmon subyearlings along the Snake River lower reach during 2016 was 19.9 ± 4.8 fish per seine haul (Table 5), which ranked ninth for the 25-year period of record for the lower reach. Mean (± 95% C.L.) annual CPUE for natural-origin Snake River fall Chinook Salmon subyearlings along the Snake River upper and lower reaches combined during 2015 was 25.3 ± 7.5 fish per seine haul (Table 5), which ranked sixth for the 25-year period of record. Mean inter-annual CPUE was lower during the low abundance period compared to the high abundance period for the upper reach, lower reach, and the combined reaches (Table 5).

The Ricker (AIC = 193.6) and Beverton-Holt (AIC =195.5) models provided similar amounts of information about the shape of the relation between estimated total escapement and mean annual CPUE calculated jointly between Snake River reaches, but more information about that relation than was provided by the linear model (AIC = 207.7; Figure 7).

TABLE 5. Mean (± 95% C.I.) CPUE (fish per seine haul) for natural-origin fall Chinook Salmon subyearlings along the Snake River upper and lower reaches, 1992–2016. The start and end dates for beach seining, the total number of station visits (*N*), and the grand means (± SE) of the abundance period (Low, 1992–1999; High, 2000–2016) annual means are also given.

	Upper reach				Lower reach				Combined			
Year	*N*	Start	End	CPUE	*N*	Start	End	CPUE	*N*	Start	End	CPUE
1992					173	04/01	06/11	3.5±0.9	173	04/01	06/11	3.5±0.9
1993					247	04/06	07/20	1.5±0.3	247	04/06	07/20	1.5±0.3
1994					249	04/06	07/13	6.0±1.8	249	04/06	07/13	6.0±1.8
1995	70	04/07	06/29	8.2±2.7	199	04/05	07/06	3.3±0.8	269	04/05	07/06	4.6±0.9
1996	54	04/18	07/11	0.8±0.3	145	04/16	07/17	2.2±0.5	199	04/16	07/17	1.8±0.4
1997	71	04/24	07/03	0.6±0.3	167	04/22	07/16	2.5±0.7	238	04/22	07/16	1.9±0.5
1998	73	04/15	07/06	4.7±1.8	149	04/14	07/08	5.2±1.3	222	04/14	07/08	5.0±1.1
1999	81	04/08	07/09	8.8±3.4	171	04/06	07/15	4.2±0.9	252	04/06	07/15	5.7±1.2
2000	41	04/06	06/29	31.4±24.2	98	04/04	07/06	17.5±5.0	139	04/04	07/06	21.6±7.8
2001	49	04/06	06/21	11.3±8.2	140	04/04	07/03	11.1±3.0	189	04/04	07/03	11.2±3.1
2002	56	04/04	07/11	29.6±16.6	160	04/02	07/17	7.8±1.8	216	04/02	07/17	13.5±4.6
2003	52	03/27	06/26	40.1±16.6	146	03/25	07/02	19.7±4.8	198	03/25	07/02	25.1±5.7
2004	55	03/25	06/24	90.9±49.2	150	03/23	06/30	41.8±11.3	205	03/23	06/30	54.9±15.6
2005	60	03/31	06/30	88.7±45.9	199	03/29	07/12	39.7±8.6	259	03/29	07/12	51.0±12.6
2006	78	03/31	07/06	7.0±3.7	216	03/29	07/11	5.8±1.2	294	03/29	07/11	6.1±1.3
2007	68	03/29	07/05	65.2±38.5	144	03/27	07/03	23.1±5.4	212	03/27	07/05	36.6±13.0
2008	92	03/27	07/17	34.7±14.4	169	03/25	07/16	10.2±2.2	261	03/25	07/17	18.8±5.4
2009	79	03/26	06/25	25.6±13.7	155	03/23	07/07	14.5±5.1	234	03/23	07/07	18.3±5.7
2010	86	03/25	07/22	20.6±8.0	187	03/23	07/27	26.7±5.3	273	03/23	07/27	24.8±4.4
2011	79	03/31	07/14	20.4±8.4	161	03/29	07/20	13.8±3.7	240	03/29	07/20	16.0±3.7
2012	85	03/28	07/11	49.0±22.4	178	03/27	08/01	21.2±4.5	263	03/27	08/01	30.2±7.9
2013	68	03/28	06/27	65.0±46.2	145	03/26	07/03	28.8±6.5	213	03/26	07/03	40.4±15.3
2014	60	03/27	06/19	24.9±13.0	133	03/25	07/01	20.5±4.4	193	03/25	07/01	21.9±5.0
2015	54	03/26	06/11	19.5±9.7	125	03/24	06/24	27.5±5.1	179	03/24	06/24	25.1±5.0

TABLE 5. (Extended)

	Upper reach				Lower reach				Combined			
Year	*N*	Start	End	CPUE	*N*	Start	End	CPUE	*N*	Start	End	CPUE
2016	65	03/24	06/16	37.1±21.5	141	03/22	06/28	19.9±4.8	206	03/22	06/28	25.3±7.5
Low				4.6 ±1.7				3.6 ±0.5				3.8±0.6
High				38.9±6.3				20.6±2.5				25.9±3.3

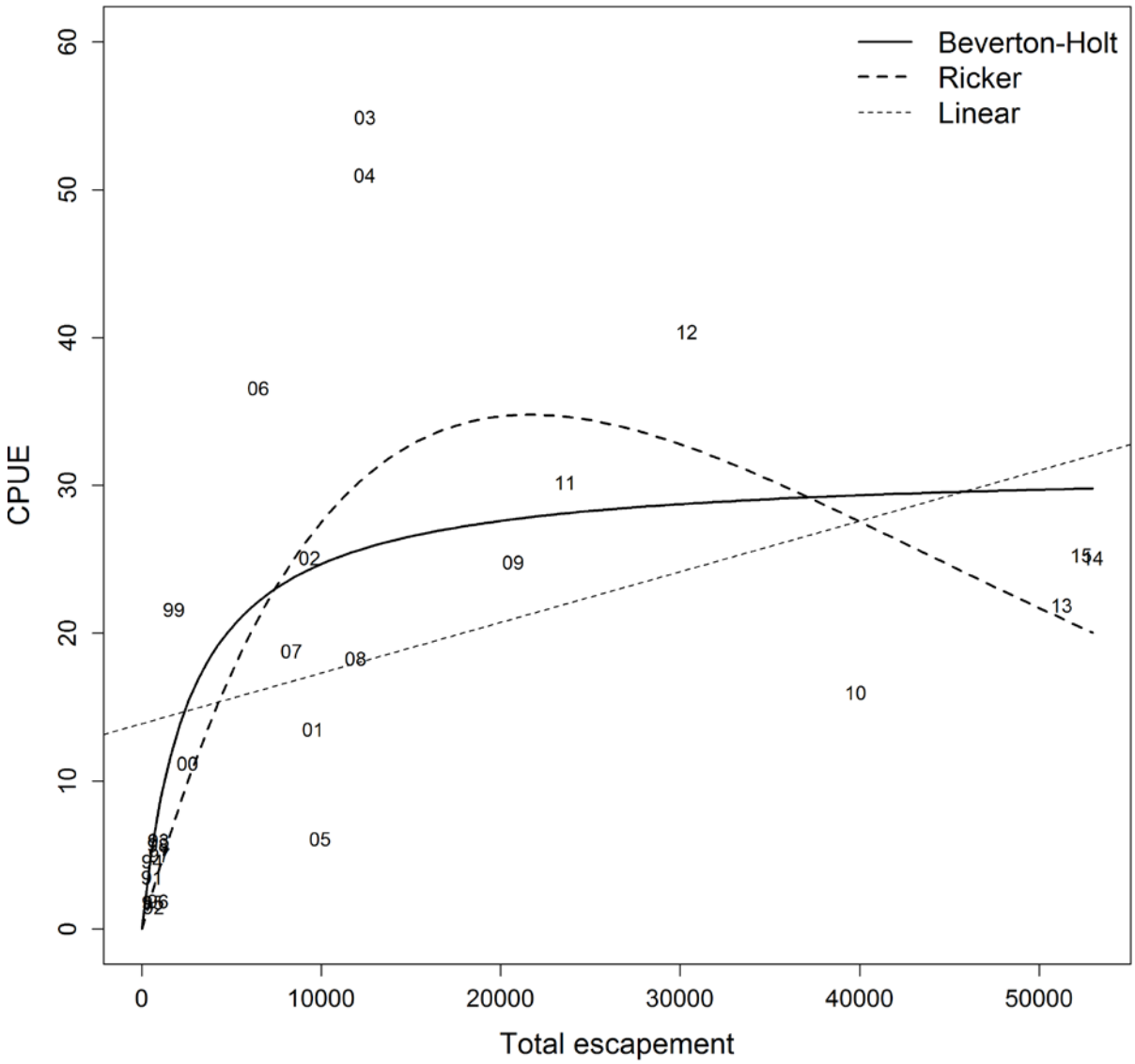

FIGURE 7. The relation between estimated annual total escapement (Table 1; 1991–2016) and annual mean CPUE of natural-origin Fall Chinook Salmon subyearlings along the Snake River upper and lower reaches combined (Table 5; 1992–2016). The numbers are the last two digits of the brood year (e.g., 15 = CPUE for juveniles in 2016 produced by spawning in 2015). The parameters (± SE) for the Ricker model (the best fit) were alpha = 0.011673±0.006642 and beta = 0.000373± 0.000252.

Of the life stages evaluated, timing of fry presence has been the most stable across the years 1992–2016 with emergence timing generally being earlier in the relatively warmer Snake River upper reach than in the relatively cooler Snake River lower reach in most years (Table 6). The median date of parr presence along the shorelines is an indicator of when the number of parr rearing in riverine habitat became less than the number of parr that had begun downstream dispersal into Lower Granite Reservoir. The median date of parr presence along both Snake River reaches generally became earlier as density along the shorelines increased as can be seen by examining the means of the median dates of parr presence for the periods of low (1992–1999) and high (2000–2016) abundance (Table 7).

Grand mean (± SE) wet weight of natural-origin parr in the Snake River upper reach declined from 4.7 ± 0.4 g during the low abundance period to 2.4 ±0.2 g during the high abundance period, and a similar decline in wet weight of parr was observed in the Snake River lower reach (Table 8). The decline in wet weight of parr was accompanied by a decline in growth in weight in both reaches, but especially in the upper reach (Table 8) where rearing habitat availability and connectivity are relatively low and apparent abundance high (i.e., CPUE in Table 5).

TABLE 6. Timing of natural-origin fry presence (Sunday's day of year; January 1 = 1) along the Snake River upper and lower reaches, 1992–2016. The means (± SE) of the annual abundance period (Low, 1992–1999; High, 2000–2016) medians are also given.

	Upper reach				Lower reach			
Year	*N*	Median	Min	Max	*N*	Median	Min	Max
1992					356	117	89	145
1993					199	136	94	171
1994					441	135	93	156
1995	117	113	92	141	257	120	92	155
1996	14	119	105	126	268	126	105	175
1997	1	110	110	110	114	124	110	180
1998	101	109	102	130	322	116	102	165
1999	97	122	94	143	278	122	94	178
2000	683	100	93	135	415	100	93	156
2001	552	119	91	140	1,268	126	91	154
2002	2,289	111	90	153	965	125	90	167
2003	962	103	82	145	1,726	110	82	173
2004	6,123	109	81	144	4,952	123	81	158
2005	5,462	107	86	135	3,786	107	86	156
2006	75	106	85	141	588	134	85	162
2007	4,311	112	84	154	1,771	119	84	154
2008	1,628	118	90	146	1,231	118	83	167
2009	811	109	81	137	1,017	116	81	165
2010	1,572	115	80	157	4,393	115	80	171
2011	1,778	128	86	163	1,408	128	86	191
2012	3,782	120	85	169	2,207	127	85	162
2013	7,874	111	83	146	4,033	118	83	167
2014	1,174	107	82	159	2,755	117	82	166
2015[a]	1,349	102	81	144	5,678	102	81	158
2016	866	101	80	129	2,164	108	80	178
Low		115 ± 3				125 ± 3		
High		110 ± 2				118 ± 2		

[a]During week 4 of the seining season a new seine was used. That seine was found to have a mesh size of 1/4 inch instead of the standard 3/16 inch. Catch was adjusted by dividing the week 4 catch by 0.75. The seine used after before and after week 4 had 3/16 inch mesh.

TABLE 7. Timing of natural-origin parr presence (Sunday's day of year; January 1 = 1) along the Snake River upper and lower reaches, 1992–2016. The means (± SE) of the annual abundance period (Low, 1992–1999; High, 2000–2016) medians are also given.

	Upper reach				Lower reach			
Year	*N*	Median	Min	Max	*N*	Median	Min	Max
1992					1,765	138	89	159
1993					2,166	157	101	199
1994					4,348	149	93	191
1995	985	148	99	169	1,408	155	92	183
1996	118	133	105	168	756	147	105	196
1997	119	145	110	166	938	159	110	194
1998	1,078	137	102	186	2,512	151	102	186
1999	1,493	143	101	178	1,647	157	94	192
2000	1,064	114	93	163	1,578	135	93	177
2001	794	123	91	161	3,076	140	91	175
2002	3,013	125	97	181	3,620	146	90	188
2003	4,523	124	82	173	6,821	131	82	180
2004	6,310	123	88	172	11,225	137	88	179
2005	8,119	121	86	170	16,803	135	86	184
2006	1,344	134	85	176	2,658	134	99	176
2007	7,226	119	91	182	7,500	133	84	182
2008	6,610	139	97	195	3,357	146	83	195
2009	3,876	130	88	165	4,706	137	95	186
2010	2,502	129	87	199	9,193	143	87	199
2011	2,237	135	93	184	2,759	149	86	191
2012	4,769	134	106	176	5,791	148	99	190
2013	4,503	125	104	174	7,171	139	83	174
2014	2,484	131	82	159	3,785	138	89	180
2015[a]	2,221	109	81	151	7,050	137	81	172
4,210	115	94	157		4,980	122	87	178
Low		141 ± 3				152 ± 2		
High		125 ± 2				138 ± 2		

[a]During week 4 of the seining season a new seine was used. That seine was found to have a mesh size of 1/4 inch instead of the standard 3/16 inch. Catch was adjusted by dividing the week 4 catch by 0.75. The seine used after before and after week 4 had 3/16 inch mesh.

TABLE 8. Mean (± SD) wet weights (0.1 g) and absolute growth rates (0.01 g/d) of natural-origin parr rearing along the Snake River upper and lower reaches, 1992–2016. The means (± SE) of the annual abundance period (Low, 1992–1999; High, 2000–2016) grand means are also given.

	Upper reach				Lower reach			
Year	*N*	Weight	*N*	Growth	*N*	Weight	*N*	Growth
1992					1,128	4.2±2.5	36	0.17±0.11
1993					1,901	4.2±3.1	161	0.13±0.13
1994					3,712	3.8±2.8	238	0.19±0.17
1995	606	4.4±3.2	29	0.28±0.13	888	4.0±3.3	35	0.21±0.11
1996	112	3.8±2.5	17	0.26±0.13	714	4.2±3.4	49	0.20±0.14
1997	114	6.1±2.7	20	0.34±0.10	922	4.7±3.1	78	0.20±0.11
1998	981	4.7±3.0	89	0.25±0.11	2,145	4.0±2.8	86	0.18±0.09
1999	1,489	4.3±3.0	169	0.30±0.13	1,642	3.7±2.8	92	0.24±0.12
2000	932	4.2±3.9	61	0.38±0.11	1,553	3.7±3.1	45	0.20±0.10
2001	724	2.0±1.3	11	0.22±0.03	2,981	2.6±1.8	120	0.18±0.07
2002	3,005	2.1±1.6	169	0.24±0.09	3,620	3.1±2.3	185	0.17±0.07
2003	4,480	2.5±1.5	359	0.18±0.10	6,821	2.5±1.5	186	0.14±0.06
2004	4,031	2.2±1.4	150	0.18±0.10	10,373	2.6±1.7	422	0.17±0.06
2005	7,739	2.1±1.2	218	0.20±0.08	15,995	2.4±1.6	353	0.18±0.07
2006	1,138	3.4±1.8	46	0.22±0.12	2,648	2.7±1.9	52	0.20±0.10
2007	7,223	1.7±1.0	128	0.21±0.08	7,318	2.9±1.9	425	0.19±0.07
2008	5,570	3.2±2.3	770	0.17±0.10	3,272	2.9±2.1	168	0.15±0.08
2009	3,348	2.2±1.0	121	0.09±0.03	4,478	2.5±1.6	265	0.10±0.07
2010	1,754	2.3±1.6	184	0.11±0.07	8,080	2.4±1.7	759	0.09±0.07
2011	1,544	2.1±1.6	139	0.15±0.09	2,454	2.5±1.7	149	0.13±0.08
2012	3,772	1.9±1.1	133	0.12±0.05	4,973	2.9±1.9	512	0.14±0.07
2013	2,092	2.0±1.1	149	0.12±0.10	5,534	2.7±1.9	436	0.11±0.06
2014	1,253	2.4±1.6	96	0.23±0.10	2,669	2.5±1.8	165	0.12±0.07
2015	1,354	1.7±0.7	27	0.09±0.04	4,618	2.6±1.8	291	0.11±0.06
2016	2,217	1.7±0.8	102	0.09±0.05[a]	3,614	2.3±1.6	225	0.09±0.05
Low		4.7±0.4		0.29±0.02		4.1±0.1		0.19±0.01
High		2.4±0.2		0.18±0.02		2.7±0.1		0.15±0.01

[a]Does not include fish< 50-mm FL experimentally tagged with 8-mm tags.

Attributes of Natural-Origin Juveniles Emigrating through Lower Granite Reservoir

The estimate of passage abundance for the basin-wide population of natural-origin, subyearling smolts at Lower Granite Dam in 2015 should be viewed with caution (i.e., it may be an overestimate; Table 9) because the low flow levels and high spill percentages at Lower Granite Dam in 2015 were not represented in the process of modeling collection probability at the dam. As such, the estimates are extrapolations and are not considered to be reliable at this time. Estimated passage abundance in 2016, when environmental conditions were in the range used for modeling fitting, is given in (Table 9).

The Beverton-Holt model (AIC = 697.4) provided more information about the shape of the relation between total escapement and estimated passage abundance of the basin-wide population of subyearling smolts compared to the Ricker (AIC = 700.6) and linear (AIC = 703.1) models (Figure 8).

Passage of PIT-tagged natural-origin subyearling smolts at Lower Granite Dam was the second earliest on record for fish from both the Snake River upper and lower reaches (Table 10). The median (± SE) date of passage of PIT-tagged, natural-origin subyearling smolts from the Snake River upper reach was later during the low abundance period compared to the high abundance period (Table 10; 8-d difference). The same pattern was observed for smolts that had been PIT tagged as parr in the lower reach, but the difference was greater than was observed for smolts from the upper reach (Table 10; a 13-d difference). In addition to the change in dispersal timing of parr from riverine habitat observed between abundance periods (Table 7), changes in downstream movement rate also contributed to the general shift in passage timing at Lower

Granite Dam between abundance periods. Smolts that had been PIT-tagged as parr rearing along the Snake River upper reach moved downstream slower on average during the low abundance period compared to the high abundance period (Table 11; 1.5-km/d difference). A similar shift in rate of downstream movement was observed for smolts that had been PIT tagged as parr in the lower reach, but the difference was not as great as observed for smolts from the upper reach (Table 11; a 0.6-km/d difference).

TABLE 9. Estimated (*N*^; 95% lower C.L.; 95% upper C.L.) passage abundance of natural-origin, subyearling fall Chinook Salmon smolts from the basin-wide population at Lower Granite Dam, 1992–2016. The means (± SE) of the annual estimates for the abundance periods (Low, 1992–1999; High, 2000–2016) are also given. The method is under development. Passage abundance was not estimated from November of year *t* through the third week of March year *t + 1* when the Smolt Monitoring Program was not in operation and juvenile fish bypass system was dewatered.

Year	*N*^	95% Lower C.L.	95% Upper C.L.
1992	13,672	12,236	21,002
1993	15,222	14,593	19,387
1994	15,895	15,165	22,403
1995	82,797	81,367	109,757
1996	36,752	35,725	48,805
1997	298,553	274,731	461,608
1998	130,765	124,893	192,413
1999	303,808	280,765	438,467
2000	585,424	471,100	1,172,425
2001	446,497	406,453	633,339
2002	255,237	216,226	405,064
2003	683,169	663,222	873,430
2004	1,177,956	1,118,431	1,514,057
2005	558,317	481,085	910,945
2006	268,364	257,713	394,865
2007	197,907	120,629	370,693
2008	429,650	398,996	644,548
2009	406,498	385,639	552,105
2010	849,839	809,427	1,235,190
2011	423,060	419,057	554,740
2012	584,172	559,331	825,038
2013	654,498	638,684	942,771
2014	486,115	448,715	709,970
2015	1,357,482	1,217,318	2,240,308
2016	530,458	248,436	2,662,701
Low	112,183 ± 43,656		
High	582,038 ± 74,422		

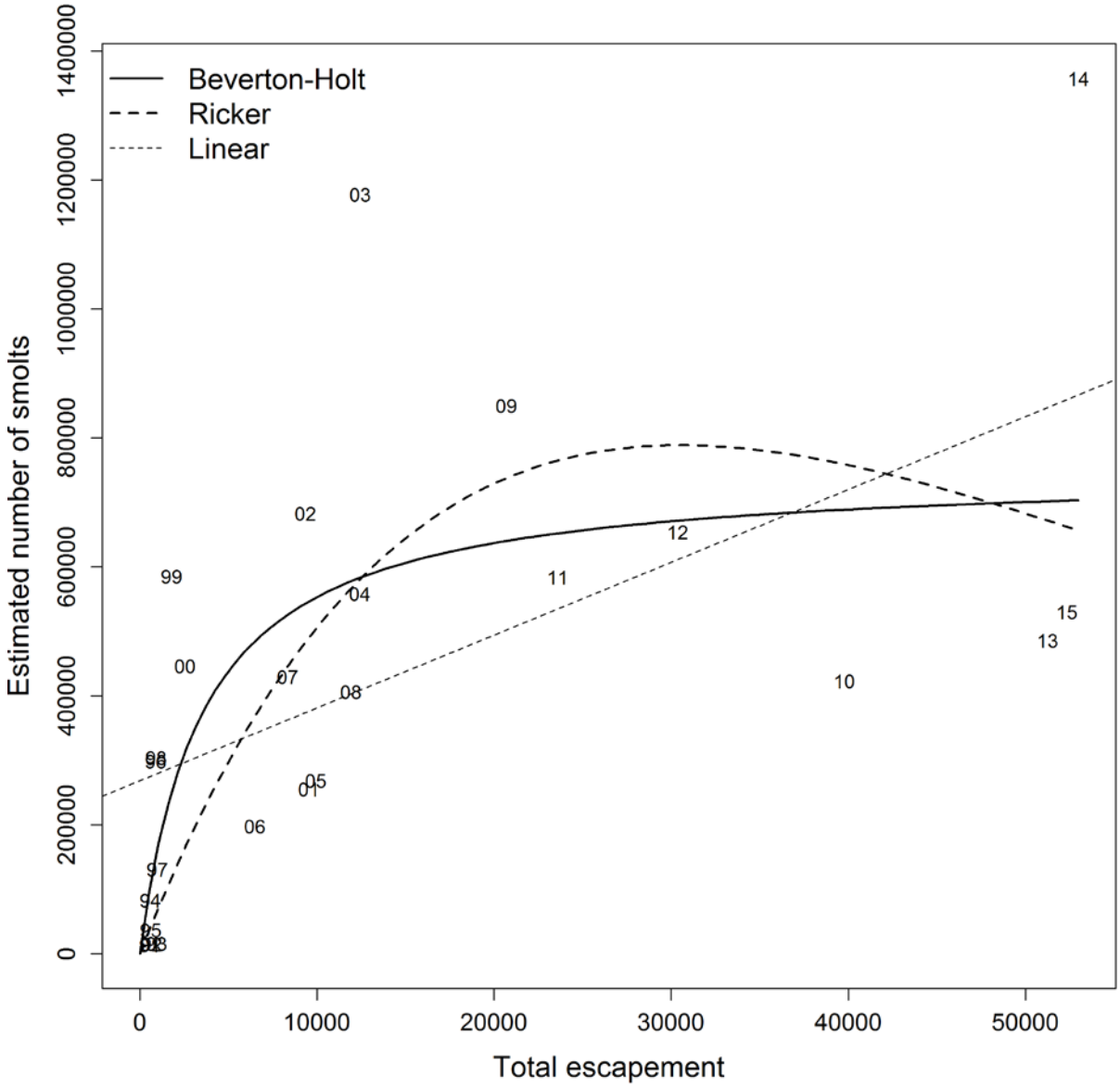

FIGURE 8. The relation between estimated annual total escapement (Table 1) and the estimated annual passage abundance of natural-origin, subyearling smolts from the basin-wide population at Lower Granite Dam (Table 9), 1992–2016. Passage abundance was not estimated from November of year *t* through the third week of March year *t + 1*. The numbers are the last two digits of the brood year (e.g., 15 = smolt abundance in 2016 produced by spawning in 2015). The parameters (± SE) for the Beverton-Holt and Ricker models (of similar fit) were: alpha = 1210.38±119.23and beta = 0.00020±0.0002; and alpha = 70.21±15.3869 and beta 0.000033±0.000007.

TABLE 10. Median dates of passage at Lower Granite Dam for natural-origin, subyearling smolts that were PIT tagged along the Snake River upper and lower reaches, 1992–2016. Estimated annual passage abundance (*N*^) of the PIT-tagged fish, and the means (± SE) of the abundance period medians (Low, 1992–1999; High, 2000–2016) are also given.

	Upper reach		Lower reach	
Year	*N*^	Median	*N*^	Median
1992			85	06/22
1993			483	07/22
1994			450	07/17
1995	401	07/18	490	08/02
1996	47	06/29	256	07/19
1997	55	06/20	192	07/16
1998	349	07/07	739	07/11
1999	712	06/26	556	07/27
2000	203	06/19	532	07/02
2001	20	07/12	374	07/07
2002	310	07/01	1,156	07/04
2003	642	06/24	1,694	06/27
2004	337	06/24	2,907	06/24
2005	1,059	06/20	2,938	06/24
2006	335	06/15	411	06/27
2007	238	06/18	1,243	06/27
2008	1,382	07/01	957	07/17
2009	405	06/22	1,023	07/03
2010	263	07/02	2,014	07/12
2011	402	06/17	589	07/12
2012	674	06/25	1,555	07/09
2013	179	07/01	903	07/04
2014	249	07/02	971	07/15
2015	83	05/28	2,473	06/23
2016	283[a]	06/12	1,994	06/16
Low	07/07 (4 d)		07/17 (4 d)	
High	06/24 (2 d)		07/03 (2 d)	

[a]Does not include fish experimentally tagged with 8-mm tags.

TABLE 11. Median, minimum, and maximum rates of downstream movement (km/d) for natural-origin parr that were PIT tagged along the Snake River upper and lower reaches and subsequently detected at Lower Granite Dam as subyearling smolts, 1992–2016. The means (± SE) of the abundance period medians (Low, 1992–1999; High, 2000–2016) are also given.

	Upper reach				Lower reach			
Year	*N*	Median	Min	Max	*N*	Median	Min	Max
1992					39	3.3	1.1	18.8
1993					233	1.5	0.7	19.7
1994					194	1.5	0.4	13.8
1995	203	3.4	1.1	44.8	239	1.5	0.4	20.3
1996	19	3.7	2.1	43.8	126	1.5	0.4	19.3
1997	24	5.3	1.7	70.5	99	2.5	0.5	21.6
1998	173	3.4	1.6	20.1	380	2.0	0.4	28.3
1999	326	4.7	1.0	35.0	280	2.1	0.4	19.5
2000	72	5.3	2.6	19.9	257	1.9	0.4	13.6
2001	10	3.6	1.3	4.5	185	1.8	0.5	15.0
2002	95	4.6	2.5	25.0	395	2.3	0.8	19.3
2003	304	5.3	2.3	43.8	814	2.5	0.5	29.5
2004	186	5.0	2.9	21.9	1,597	2.6	0.7	21.3
2005	314	6.0	2.9	26.3	672	3.4	1.0	19.7
2006	96	8.0	3.5	58.3	103	3.0	0.6	45.0
2007	37	5.3	3.8	13.5	177	2.8	0.4	15.6
2008	359	5.3	1.2	87.5	211	2.1	0.5	57.0
2009	145	4.9	2.2	29.8	319	2.5	0.4	42.5
2010	72	6.7	2.8	61.7	475	2.9	0.3	58.0
2011	167	7.6	2.4	64.3	231	3.4	0.5	59.0
2012	163	5.0	2.2	59.7	303	2.6	0.3	42.5
2013	33	4.1	2.6	29.2	143	2.3	0.4	21.8
2014	61	5.4	2.1	46.3	198	2.2	0.5	27.3
2015	17	5.0	3.1	20.6	296	2.8	1.1	18.0
2016	45[a]	8.1	2.7	25.6	379	2.6	1.1	42.5
Low		4.1 ± 0.4				2.0 ± 0.2		
High		5.6 ± 0.3				2.6 ± 0.1		

[a]Does not include fish experimentally tagged with 8-mm tags.

Seasonal Variation in Smallmouth Bass Diets and Consumption of Subyearling Chinook Salmon during Rearing in Riverine Habitat

The consumption rate (*C*) of Chinook Salmon by Smallmouth Bass increased following all four hatchery releases. Before the first release at Hells Canyon Dam, daily estimates of *C* ranged from 0 to 0.052 fish/bass/d, but increased to 0.338 fish/bass/d by May 17 (the day after the release) in our study section above Pittsburg Landing (Figure 9; actual). Pre-release consumption below Pittsburg Landing ranged from 0.012–0.355 fish/bass/d. No sampling was conducted on May 18 but we assume that *C* decreased to a pre-release level following a similar pattern to the release on May 18 (Figure 9; hypothetical). Following the second release of hatchery fish on May 18, *C* increased to 0.585 fish/bass/d above Pittsburg Landing and to 0.653 fish/bass/d below by May 19. On May 20, *C* decreased to near pre-release levels above Pittsburg Landing and remained low on May 21 as well. However, below Pittsburg Landing, *C* increased to a high of 1.604 fish/bass/d on the day of the release (May 20) at Pittsburg Landing. Consumption rate decreased slightly to 1.322 fish/bass/d on May 21 before dropping below pre-release levels on May 22.

We used the above information to estimate the loss of Chinook Salmon to Smallmouth Bass in the upper portion of our study area. We conclude that hatchery releases from Hells Canyon Dam produce a sharp, 1-d increase in Smallmouth Bass consumption while releases from Pittsburg Landing produce a 2-d increase in consumption. We assume that consumption rates in the reach below Pittsburg Landing followed a similar temporal pattern as in the reach upstream resulting in returns to pre-release levels on May 18 following the dam release on May 16 (Figure 9; hypothetical). Using this assumption, we estimate 110,177 subyearlings were consumed during 2016 in the upper portion of our Hells Canyon study area of which ~98,000

were hatchery origin based on sizes of natural and hatchery origin subyearlings consumed (Figure 10). We estimate that 38,669 hatchery subyearlings released at Hells Canyon Dam were consumed by Smallmouth Bass both above and below Pittsburg Landing. This equates to a consumption of 1,039 subyearlings/rkm in our study area and 3.65% of the total release. By individual release, we estimate 13,684 (368 subyearlings/rkm) subyearlings were consumed during the May 16 release and 24,986 (672 subyearlings/rkm) subyearlings were consumed during the May 18 release. For the Pittsburg release, we estimate 59,123 (14.85% of the release) subyearlings were consumed, which equates to 3,398 subyearlings/rkm.

Smallmouth Bass consumption rate was much lower during the June release of subyearlings from the Captain John acclimation facility. Consumption increased from a pre-release rate of 0.030 fish/bass/d on June 10 to a peak of 0.183 fish/bass/d on June 11and declined gradually thereafter to pre-release levels of 0.022 fish/bass/d by June 15 (Figure 11). We estimate 13,728 Chinook Salmon were consumed by Smallmouth Bass between Captain John and the head of Lower Granite Reservoir. Assuming bass mainly consumed hatchery fish following the release from the Captain John, we estimate 12,405 hatchery subyearlings (0.06% of the release) were lost to predation in our lower study area.

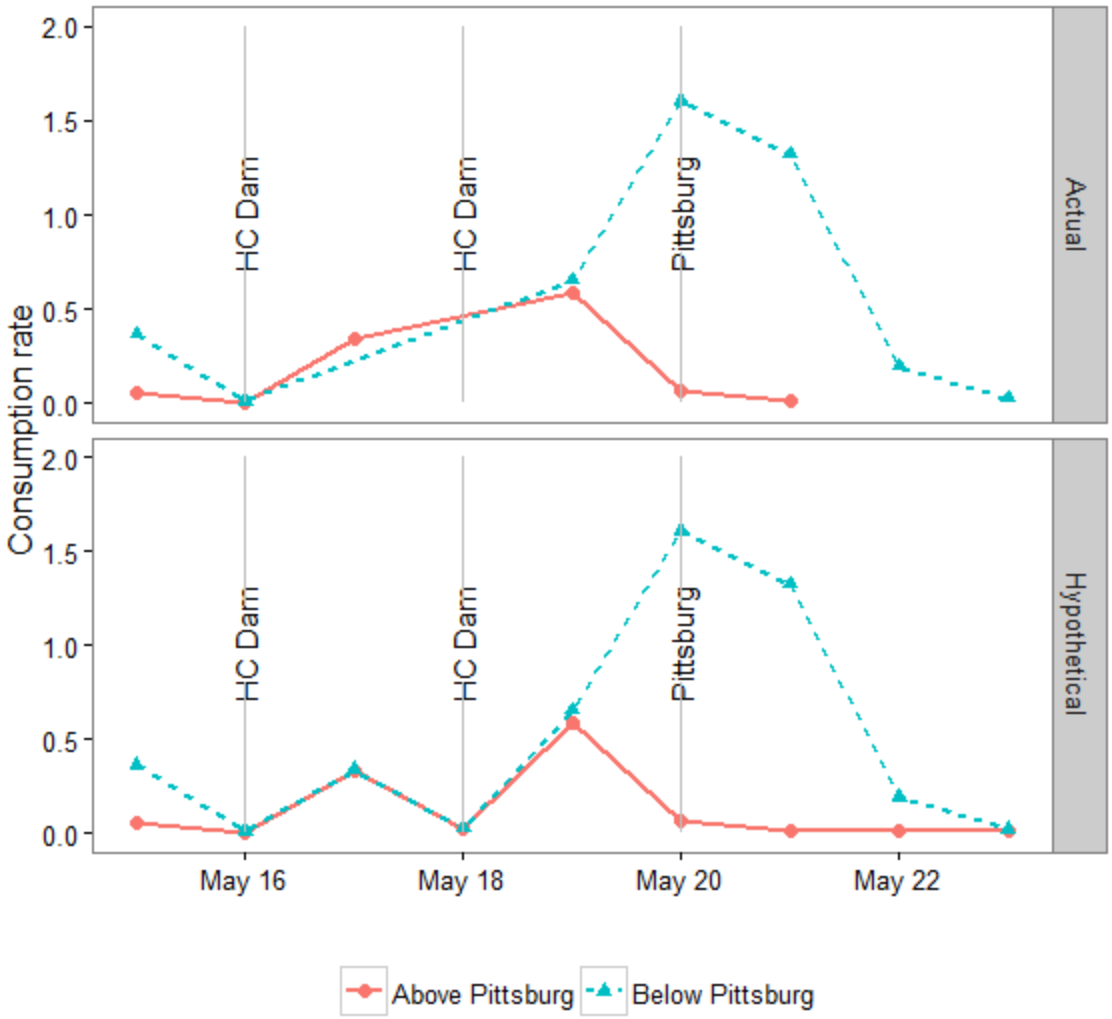

FIGURE 9. Actual (top panel) and hypothetical (bottom panel) daily consumption rate (fish/bass/day) of subyearling Chinook Salmon by Smallmouth Bass above and below Pittsburg Landing in the upper portion of the Snake River. Vertical bars represent dates of hatchery releases of subyearlings at Hells Canyon Dam and Pittsburg Landing. Hypothetical consumption is that which we expected if we had sampled every day following hatchery releases.

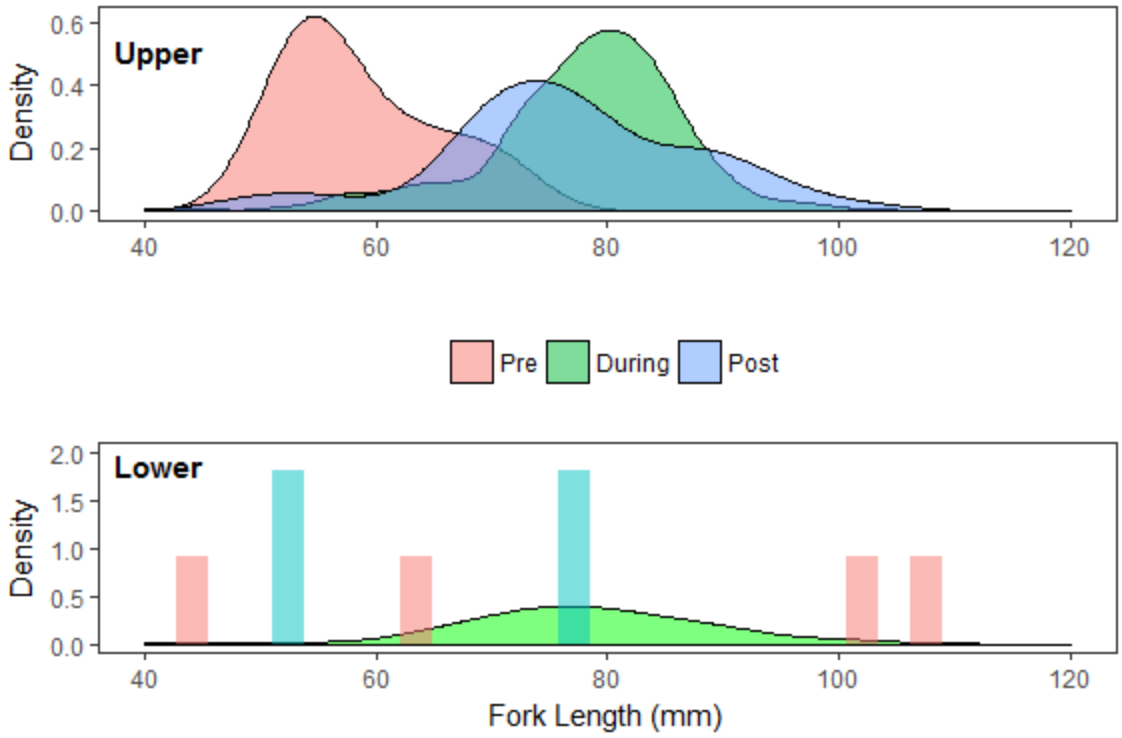

FIGURE 10. Size distribution of subyearling Chinook salmon consumed by smallmouth bass before (Pre), during, and after (Post) hatchery releases of subyearlings in the upper and lower reaches of the Snake River.

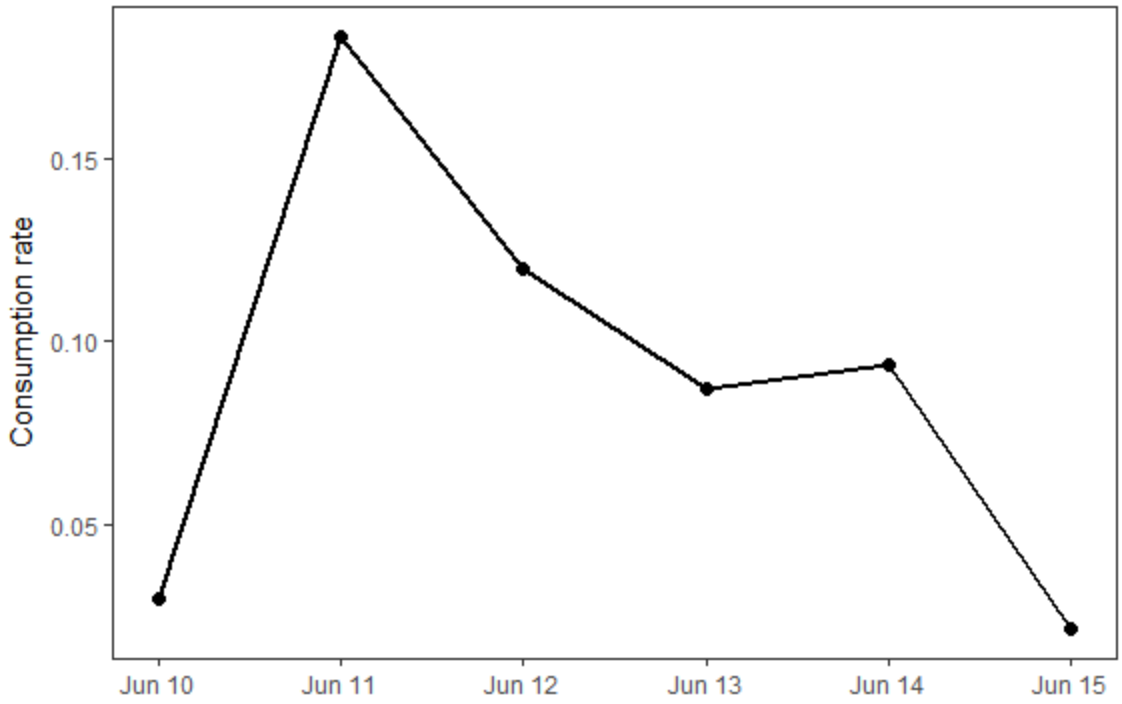

FIGURE 11. Smallmouth bass consumption rate (fish/bass/day) of subyearling Chinook Salmon following a hatchery release at Captain John acclimation facility in the lower portion of the Snake River, 2016.

Discussion/Conclusion

Attributes of Spawning in Riverine Habitat

The functional growth relations that we modeled, between total escapement and the number of shallow and deep sites used, were the first of several lines of evidence for density dependence in this report. The functional growth relation between total escapement and the numbers of shallow and deep redds counted provided growing but relatively weak evidence for density dependence as clear maximums were not attained. Together, the four curve fitting exercises for attributes of spawning showed that most of the suitable spawning habitat has been used by spawners, but the habitat has not reached redd capacity. In support of that conclusion, the maximum redd count observed in the Snake River reaches combined was 3,155 or 1,287 below the estimated redd capacity of 4,442 for the river (Groves et al. 2013). We are questioning the accuracy of our redd counts in recent years, however, because the large numbers of redds are difficult to count from the air especially when redds are superimposed. Superimposition occurs when a new redd is constructed on top of one or more previously constructed redds. In addition to being a source of counting error, that process can dislodge eggs in the previously constructed redds and be the underlying mechanism leading to the Beverton-Holt and Ricker-type recruitment we reported earlier.

In cooperation with staff of the Idaho Power Company who pioneered the use of Unmanned Aerial Systems (sUAS) for counting redds, we worked with statisticians from the University of Idaho to design and implement a study to improve monitoring of superimposition and accuracy of our redd counts. That effort came to fruition in 2015. For the present, superimposition is likely common in the Snake River reaches at the spawner escapement levels observed in recent years. In 2014, video taken from a sUAS was used to determine that some

superimposition occurred at 43% of the sites used along the Snake River reaches, and that heavy superimposition occurred at 28% of the sites used (P. Groves, Idaho Power Company, unpublished data). Continued collection of redd count data will aid in monitoring and managing the recovery of the Snake River fall Chinook Salmon ESU.

Attributes of Natural-Origin Juveniles Rearing in Riverine Habitat and Emigrating through Lower Granite Reservoir

We found that recruitment of natural-origin fall Chinook Salmon subyearlings assessed based on CPUE declined at high total escapements (i.e., overcompensation or Ricker-recruitment). The three most plausible causes of that decline are redd superimposition, egg predation, and juvenile predation. As mentioned, superimposition has only been assessed in 2014. Additional analyses of video footage collected by UAS in earlier years as wells 2015 would be informative. White Sturgeon *Acipenser transmontanus* have been observed along the Snake River reaches hovering over redds in both underwater and sUAS video footage (P. Groves, Idaho Power Company, unpublished data). That footage provides circumstantial evidence for egg predation. Ongoing research by staff of project 199102900 (described later in this report) has also provided evidence that predation on natural-origin fall Chinook Salmon subyearlings by Smallmouth Bass has increased as total escapement of fall Chinook Salmon spawners has increased. Evidence for density dependent mechanisms, however, is not solely limited to the spawning life stage or predator/prey interactions.

Intra-specific density-dependent mechanisms during rearing in riverine habitat are also evident in this report. Growth rates of natural-origin parr in riverine rearing habitat declined

slightly as parr abundance increased. Timing of downstream dispersal into the reservoir was also earlier and dispersal size decreased between abundance periods.

Our results on natural-origin subyearling smolts in Lower Granite Reservoir provided ample evidence for density dependence noting that we are questioning our 2015 estimate of annual passage abundance of natural-origin subyearling smolts. In addition to finding evidence for Beverton-Holt and Ricker-type recruitment, smolt growth declined between abundance periods in association with density-dependent changes in size at dispersal from riverine habitat, migrational behavior in the reservoir, and increased concentrations of subyearling smolts in the reservoir as the hatchery program expanded (Connor et al. 2013). In turn, time of passage through the reservoir of natural-origin smolts became earlier and smolt size decreased. During the low abundance period, natural-origin smolts also remained in the reservoir later into the summer, which allowed them to experience warmer temperatures and grow to larger sizes. However, during the high abundance period, natural-origin smolts spent more time migrating and less time lingering and feeding resulting in passage through the reservoir before they could benefit from warmer water temperatures that favored growth.

Attributes of Seasonal Variation in Smallmouth Bass Diets and Consumption of Subyearling Chinook Salmon during Rearing in Riverine Habitat

Not unexpectedly, Smallmouth Bass responded to releases of hatchery juvenile Chinook Salmon by preying heavily on them which was reflected in their consumption rates. Responses were much more pronounced in the upper portion of our study area where consumption increased to 1.60 subyearlings/bass/day. Nelle (1999) also found a similar predation response in the late 1990s when he evaluated Smallmouth Bass consumption following a release of hatchery

subyearlings at Pittsburg Landing. He estimated a consumption rate of 1.143 subyearlings/bass/day for 150–249-mm bass following a release during July of 1997. Our estimates of consumption in upper Hells Canyon were probably conservative as angled bass often regurgitated numerous juvenile salmon as they were being reeled in and we could not account for this loss.

The predation response by Smallmouth Bass to a release of hatchery subyearlings is relatively short in Hells Canyon. We found that consumption rates typically returned to pre-release levels within a day or two, which may be due to the relatively short residence time of subyearlings in the river. Following release, hatchery fish tend to disperse downstream rapidly in the unimpounded Snake River in Hells Canyon (USFWS, unpublished), which probably shortens their exposure to predators. Thus, predation risk and loss is very high for a short time, but then quickly drops. However, natural-origin subyearlings rearing along shorelines would be exposed to predation for a longer time. Sampling logistics prevented us from sampling every day following each release which is why we speculated on the predation response on days following some hatchery releases (Figure 9). The best support for the assumption we made is shown by the response to the hatchery fish released at Hells Canyon Dam on May 18. Those fish would have travelled downstream to reach our study area by May 19 during with predation was likely high. On May 20, we observed low consumption rates suggesting that most hatchery fish had moved downstream. At a minimum, most consumption rates return to pre-release levels within 2 d of a hatchery release, but using a return to pre-release levels after 1 d results in more conservative loss estimates. Sampling at 1-d intervals in the future should help refine the pattern in consumption rate.

The pattern in Smallmouth Bass consumption rates was different in the lower portion of Hells Canyon. Below Captain John, bass consumption rates increased rapidly following the release of hatchery fish but never approached the rates observed in the upper canyon. Consumption rates also took longer to return to pre-release levels. There may be a number of reasons for these observations. First, other prey such as Sand Rollers *Percopsis transmontana*, which are absent in the upper canyon, may have reduced bass consumption of subyearlings. Second, there are fewer bass in the river below Captain John than in the upper canyon and this may have resulted in lower overall consumption rates. Finally, lingering hatchery fish from upstream releases that moved through the lower river may have contributed to the longer duration of elevated bass consumption rates.

Our consumption and loss estimates were probably influenced by spatial factors. The much higher consumption and loss estimates from the Pittsburg Landing release was most likely influenced by the study area being located immediately downstream of the release site. In contrast, the study reach above Pittsburg Landing was located 33.2 km downstream of Hells Canyon Dam. This spatial difference allowed fish to disperse more as they moved downstream from the dam which likely resulted in lower densities compared to fish released at Pittsburg Landing. We hypothesize that bass consumption rates decrease with increasing distance from release locations as the pulse of released subyearlings spreads out and as their numbers decrease from mortality. We also hypothesize that as migration slows further downriver that bass consumption rates remain elevated longer as shown by our results from the lower river. Since bass densities are similar between our two upper study reaches, and daily release numbers were likely similar, then there is some evidence to support our first hypothesis.

Qualitative examination of length frequency distributions suggest that most of the salmonids consumed in the upper portion of our study area 1–2 days after the release were hatchery subyearlings. This is reasonable since hatchery fish are probably more naïve of predators than natural-origin fish. We were unable to determine origin of consumed subyearlings from the June release in the lower river due to the fact that hatchery fish were already in the system before the release. However, we suspect that bass consumed mainly hatchery subyearlings following the release at Captain John.

Adaptive Management & Lessons Learned

Multistage life cycle models provide a powerful framework for understating how each life stage of a population contributes to population growth rate (Moussalli and Hilborn 1986; Brooks and Powers 2007). When used for simulation, multistage models allow the relative effects of density dependence at different life stages to be explored in the context of management actions including Fish Population RM&E, Hydrosystem RM&E, Harvest RM&E, Hatchery RM&E, Predation and Invasive Species Management RM&E. For example, by using a multistage model, Greene and Beechie (2004) found that the importance of habitat restoration to population recovery of Chinook Salmon depended on the mechanisms of density dependence affecting particular life stages.

Multistage models may also be used as an analytical framework to explicitly estimate demographic parameters of a population model. This approach has an advantage over single-stage stock-recruitment models by allowing population growth rates to be partitioned among life stages rather than aggregated over an entire life cycle. Such partitioning allows for estimating 1) stage-specific estimates of density dependence, and 2) stage specific effects of environmental

factors or management actions. Zabel et al. (2005) estimated parameters of a multistage model used in the context of a population viability analysis spring/summer Chinook Salmon in the Snake River, but such an approach has yet to be applied to fall Chinook Salmon in the Snake River basin.

Typically, data informing estimates of abundance at particular "check points" in the life cycle dictates the complexity of multistage models that can be fit to data. For fall Chinook Salmon, we will start with a two-stage model that encompasses: 1) upstream passage of spawners at Lower Granite Dam to the subsequent the downstream passage of their progeny at the Dam, and 2) downstream passage of juveniles at Lower Granite Dam to their subsequent return from the ocean and passage at the Dam 3–5 years later. This approach partitions the life cycle of fall Chinook Salmon both spatially and temporally, which will allow us to fit and compare alternative models with covariates specific to each stage. We are building such a model while participating in the AMIP process where ideas on model development are shared among the members. Information is also presented at AFS meetings and regional forums to disseminate it and receive useful feedback from peers.

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Appendices

A.1: Data Links

Redd count data: Available:http://www.fpc.org/
Seining data : Available: http://www.streamnet.org/
PIT-tag data: Available:http://www.ptagis.org/

Other data is backed up on site and remotely. Inquires can be sent to william_connor@fws.gov or ktiffan@usgs.gov.

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A.3: Additional Research

Detectability of 8-mm, 9-mm, and 12-mm PIT Tags implanted in juvenile Chinook Salmnon at Lower Granite Dam

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Introduction

The ability to represent a population of migratory juvenile fish with Passive Integrated Transponder (PIT) tags becomes difficult when the minimum tagging size is larger than the average size at which fish begin to move downstream. The current minimum recommended size for tagging juvenile salmonids with 12-mm PIT tags in the Columbia River basin is ≥65 mm FL. From 1992 to1999, the mean FL of subyearling Chinook Salmon from the Snake River upstream of Lower Granite Dam was 65 mm ± 1 mm (95% CI) at initial downstream dispersal, but subsequently decreased to 55 ± 1 mm from 2000 to 2011 during the same dispersal period (Connor et al. 2013). Tags that are smaller (e.g., 8 and 9 mm) than the commonly used 12-mm PIT tag are now currently available. Tiffan et al. (2015) found that 40–49-mm subyearling Chinook salmon could be implanted with 8-mm and 9-mm tags without largely affecting growth and survival in the laboratory, and recommended experimental use of the smaller tags for subyearlings in the field. The use of smaller PIT tags (i.e., 8 and 9 mm) would allow researchers to more fully represent the juvenile Snake River fall Chinook Salmon population that undergoes size-related dispersal, as well as reducing tag burden-related effects (Tiffan et al. 2015).

Many PIT-tagged juvenile salmonids in the Snake and Columbia river basins are detected at mainstem dams to provide various population, behavioral, and survival metrics. The detectability of 8-mm tags at mainstem dams has not been evaluated in juvenile salmonids, so it is unknown whether reliable detection information can be obtained from fish implanted with these tags. In 2016, we conducted a simple test of detectability of 8-mm, 9-mm, and 12-mm PIT tags at Lower Granite Dam. The objective of the test was to determine the detection efficiency of the different tags implanted in subyearling fall Chinook salmon and released within the fish bypass system.

Methods

We evaluated the detectability of four different PIT tags: 8-mm, Biomark; 8-mm, Oregon RFID; 9-mm, Biomark; and 12-mm, Biomark (Table 1). On June 7, Smolt Monitoring Program personnel at Lower Granite Dam collected approximately 300 untagged subyearling Chinook salmon and held them in four 50-gallon containers supplied with river water. The next day, we tagged fish systematically by implanting the first fish with an 8-mm Biomark tag, the second fish received an 8-mm Oregon RFID tag, the third fish received a 9-mm tag, and the fourth fish received a 12-mm tag. This cycle was then repeated until 75 fish were tagged with each tag type. PIT tagging and anesthetization followed the recommendations of Prentice et al. (1990a, 1990b) and the PIT-tag Steering Committee (PTSC 2014) with some exceptions stated below.

Tagging proceeded by first anesthetizing fish in 18.9 L of oxygenated water that contained 5 mL of MS-222 in solution at a concentration of 100 mg/L, 0.5 g $NaHC0_3$ as a buffer, and 2.5 mL of Polyaqua as a replacement "slime" mucoprotein coating (Connor et al. 1998). A fish was then randomly selected and weighed to the nearest 0.1 g, measured (FL) to the nearest mm, and tagged using an injector. Our tagging apparatus was a 10-cc syringe body affixed with 12-gauge needle for use with 9- and 12-mm tags or a 14-gauge needle for use with 8-mm tags. Each syringe body had a slot cut into the side that was approximately 10 × 3 mm that allowed a thumb screw to be inserted into a plastic cylinder contained within the syringe that had an attached push rod that extended into the barrel of the needle. PIT tags were inserted into the beveled end of the needle and then inserted into the fish by moving the thumb screw toward the end of the needle. Sedated fish were placed head first and dorsal surface down into the corner of a wet, notched sponge leaving only a slight portion of the ventral and posterior portion of the body exposed. The syringe was then held parallel to the fish and a slight downward pressure was applied to the needle to make a small incision (large enough only to accommodate the insertion of the needle bevel) off the ventral midline. The tag was then inserted into the fish's peritoneal cavity while the hands and forearms of the tagging person were braced for stability (Tiffan et al. 2015). All tagging information, including the presence of coded-wire tags and additional marks, were recorded using P3 software and uploaded to the PIT Tag Information System (PTAGIS 2016).

After tagging, fish were allowed to recover for 15 min in 18.9 L of oxygenated water with 2.5 mL of Polyaqua solution and then transferred to one of four 75-gallon replicate circular tanks supplied with river water. Each tank contained equal numbers of fish tagged with each tag size and type. Tagged fish were held for an additional 20–24 h to check for shed tags and mortalities. On the day of release, groups of 20 fish were transferred to aerated 5-gallon buckets and released into the upwell area of the bypass system upstream of the wet separator. Ten fish were released every 5 min to minimize the chance of tag detection "collisions" and to maximize individual detections. This resulted in approximately 30 releases of 10 fish each over two and half hours. The release location exposed fish to detection by up to six PIT tag antennas located throughout the bypass system depending on the route they took (Figure 1). The first route would

be across the separator to the adult return where fish could be detected by two antennas (F1 and F2; Figure 1). A second route would be through the separator gate antennas (either A1 and A2 or B1 and B2) and through three additional antennas (51, 52, and 53) to the sample tank. Fish taking this route could be detected by a maximum of five antennas. The final route would be through the separator gate antennas (either A1 and A2 or B1 and B2) and then through four additional antennas (C1, C2, 91, and 92) enroute to exiting to the river (Figure 1). Fish taking this route could be detected by a maximum of six antennas.

Table 1. Specifications of PIT tags implanted in subyearling fall Chinook Salmon at Lower Granite Dam in 2016.

Tag size (mm)	Model	Manufacturer	Weight (mg)	Length (mm)	Diameter (mm)
8	684293-003	Oregon RFID, Portland, OR	30	8.24	1.40
8	MiniHPT8	Biomark, Boise, ID	30	8.43	1.40
9	TXP148511B	Biomark, Boise, ID	66	8.91	2.10
12	TXP1411SST	Digital Angel, South St. Paul, MN	104	12.37	2.02

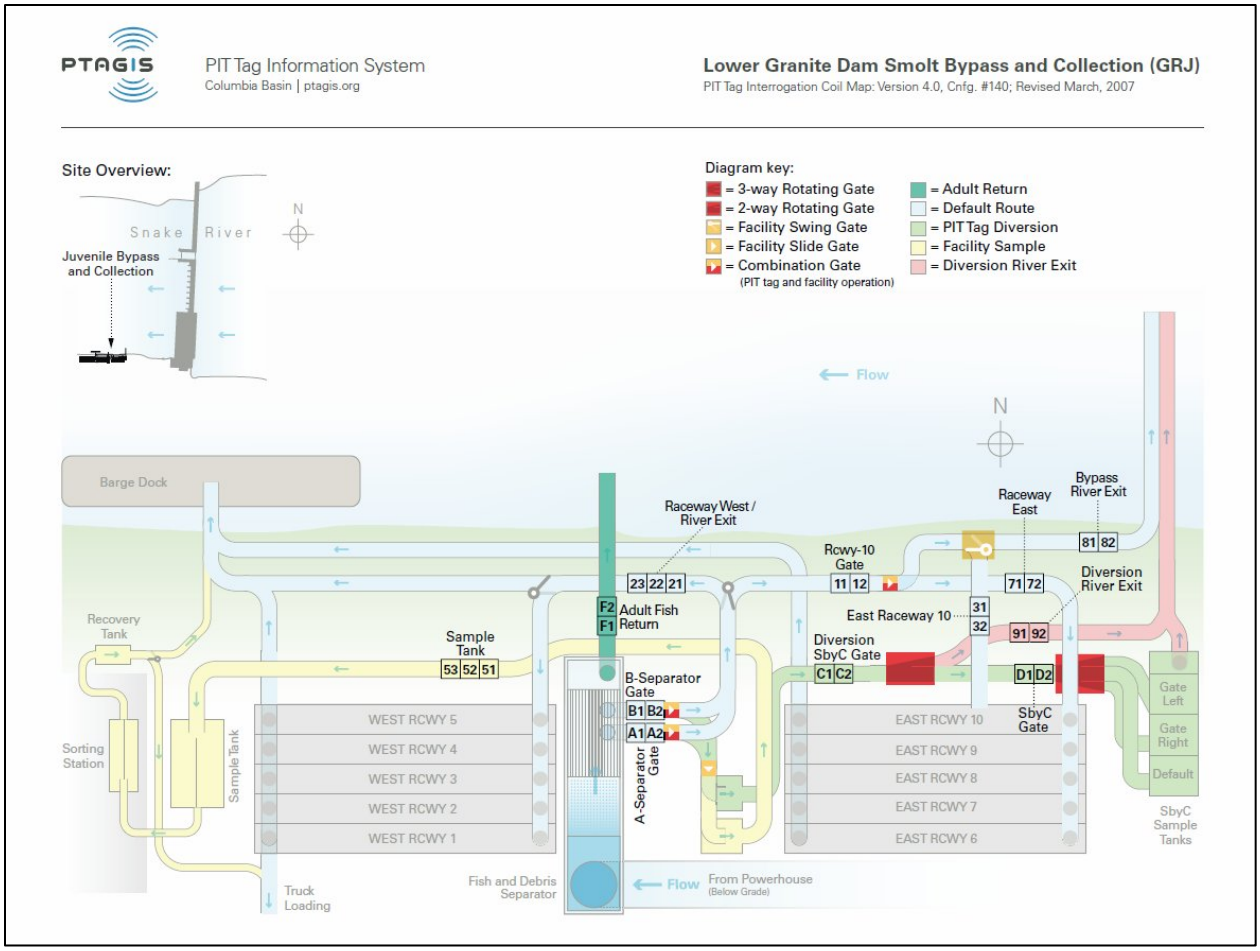

Figure 1. Map of PIT-tag interrogation system at Lower Granite Dam. Fish were released upstream of the upwell (the blue circle).

Results

We PIT tagged 304 fish, of which nine fish died after 24 h. No tags were expelled during the 24-h post-tagging holding period. Tagged fish lengths ranged from 57 to 128 mm and weights ranged from 1.4 to 22.4 g (Table 2). We released 295 fish into the bypass system for subsequent detection which comprised 74 fish tagged with 8-mm Biomark tags, 72 fish tagged with 8-mm Oregon RFID tags, 75 fish tagged with 9-mm Biomark tags, and 74 fish tagged with 12-mm Biomark tags. Detection rate of all tag sizes combined was 99.7%. Only one tagged fish was not detected, and it was tagged with an 8-mm Biomark tag.

From 98.6 to 100% (depending on tag type) of tagged fish were detected on at least two antennas (Table 3). The fish detected on only two antennas comprised five individuals that passed into the adult return, thus bypassing further detection opportunities. Four fish were detected passing into the sample tank; two of which were detected on four antennas and two of which were detected on all five antennas. Between 76.4 and 98.7% of the remaining 285 fish were detected on three, or more, of the possible six antennas (Table 3; Figure 1). In general, the 8-mm Oregon RFID tags were detected at slightly lower rates than the Biomark tags (Table 3).

Discussion

Total detection rate of combined tag groups was high (99.7%) with only one tag (an 8-mm Biomark tag) not being detected in this efficiency test. In addition, relatively high percentages (>76) of tagged fish were detected on all six PIT tag antennas. Even the few fish that passed through the adult return were detected on both antennas located there. These results are encouraging because they suggest that subyearlings implanted with 8-mm PIT tags should have detection efficiencies at Lower Granite Dam that are comparable to 9-mm and 12-mm tags. This also means that field use of 8-mm PIT tags should be practical when subsequent detection at Lower Granite Dam is of interest.

The mortality rate of post-tagged fish was relatively high at 3% (9 of 304 fish) and was likely due to stress related from collection, longer holding periods, tagger experience, and relatively high water temperatures (~17°C). All fish that died were >84 mm and 5 g. Tagging-related mortality was not too concerning for this study because we were primarily interested in using surviving fish as “vehicles” to carry tags through the bypass system. Mortality of much smaller fish tagged in the field would be of greater concern but has not been evaluated. However, 8-mm PIT tags may have advantages over larger tags. During field tagging in 2016, field personnel noted that the size of the incision associated with 14-gauge needles used to implant 8-mm tags was smaller than that made by 12-gauge needles used to implant 9-mm and 12-mm tags. Furthermore, we observed that fish recaptured in the field that were tagged with 8-mm tags (using 14-gauge needles) had healed more quickly than fish tagged with larger tags and needles. This might facilitate tag retention, reduce recovery time, and possibly reduce tagging mortality in the field.

Table 2. Mean±SD lengths and weights of subyearling fall Chinook salmon tagged at Lower Granite Dam to evaluate detection efficiency of different PIT tags.

Tag size (mm) and brand	FL (mm)	FL range (mm)	WT (g)	WT (g) range
8 Biomark	105±7.6	80-128	9.8±7.4	5.2-19.0
8 Oregon RFID	103±11.3	57-126	10.8±9.3	1.4-18.2
9 Biomark	104±8.7	79-122	10.8±2.5	4.4-16.1
12 Biomark	104±9.5	62-127	11.2±10.3	1.8-22.4

Table 3. Percent of PIT-tag groups detected on unique antennas in the bypass system at Lower Granite Dam.

Tag size (mm) and brand	≥2 antennas	≥3 antennas	≥4 antennas	≥5 antennas	6 antennas
8 Biomark	98.6	98.6	98.6	97.3	86.5
8 Oregon RFID	100.0	97.2	95.8	90.3	76.4
9 Biomark	100.0	98.7	98.7	92.0	77.3
12 Biomark	100.0	98.6	98.6	97.3	95.9

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