Sacramento River Winter-run Chinook Salmon Redd Dewatering: a Note on Comparing Observed and Predicted

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Summary

Redd dewatering generally occurs after drops in flow, and consequently results in exposure and mortality of eggs and fry. The observed percentages of redds dewatered for Sacramento River Winter-run Chinook salmon (WRCS) between 2013 and 2022 ranged 0.0%–0.71% (Chelberg 2023). In comparison, the predicted percentages of redds dewatered, based on the method from USFWS (2006) ranged 0.1%–16.0%. A bias index of predicted to observed, scaled by the mean of observed across years, was estimated: 49.9 $\pm$ SD 42.3 with ACID Dam boards out and 53.6 $\pm$ 43.5 with boards in, thus showing a notable bias, but also large variation across years. In a linear model of observed to predicted percentages of redds dewatered across years, the inverse of the slope indicated that the predicted percentages were 60.6 (95% CI 32.3–501.0) times greater than the observed. While direct comparisons can lead to large biases, these can be invalid due to the difference in their definitions. It is important to note that predicted redd dewatering can include redds categorized as “dewatered” because of low water velocity (<0.87 ft/s), and not just low water depths depth (<0.52 ft), as part of WRCS habitat suitability criteria. We discuss other possible reasons for the bias in the predicted compared to observed percentages of redds dewatered.

Introduction

Redd dewatering generally occurs when flows drop during the period of egg and fry incubation. Redds constructed in the shallowest and slowest waters are at greatest risk, especially when large drops in flow rapidly decrease water depth and velocity. Predictions of redd dewatering are thus useful for evaluating alternatives of Shasta Reservoir operations and their effects on the salmon egg incubation stage.

A method to predict the percentage of redds dewatered by both water depth and velocity (USFWS 2006) is known to overpredict redd dewatering in comparison to observations of redds categorized as dewatered based primarily on water depth (Chelberg 2023). Thus, the predictions are generally used as a relative index when assessing the impacts of flow on salmon redd dewatering.

The main objectives of this Note are to provide a quantified estimate of the bias in the predicted percentages of redd dewatering (USFWS 2006) in comparison to field observations (Chelberg 2023), and share possible reasons for the bias. We focus on the federal- and state-listed endangered Sacramento River Winter-run Chinook salmon (*Oncorhynchus tshawytscha*).

Methods

Observed percentage of redds dewatered. Annual surveys of WRCS redds in the upper Sacramento River identified and tracked shallow-water redds across varying flow levels (Chelberg 2023). In the survey, the crew monitors "newly constructed Chinook salmon redds that are considered to be in jeopardy of being dewatered, and their survival impacted during later expected flow reductions." Generally, these redds were at depths shallower than 24 inches. If limited time was available for surveying and redd counts were high (e.g., for fall-run Chinook salmon), then redds in the shallowest waters were monitored. The redds were monitored on subsequent visits to determine status of individual redd dewatering. In the years 2013–2022, the number of redds monitored each year ranged 19–109.

The observed number of redds dewatered ($n_{Dewatered,y}$) and total number of redds ($N_{Redds,y}$) in cohort y were used in the current analysis to calculate the yearly, observed percentage of redds dewatered (ρ_y):

$$\rho_y = \frac{n_{Dewatered,y}}{N_{Redds,y}} \times 100 \qquad \text{Eq. 1}$$

This method computes the minimum percentage impacted because there may be one or more shallow-water redds that were not found during the survey, and redds may have been super-positioned over other redds.

We also compared the number of monitored redds (Chelberg 2023) with female escapement estimates (Killam 2023) to assess the consistency and representativeness of coverage in the redd dewatering surveys through the years.

Predicted percentage of redds dewatered. The predicted percentages were calculated using pre-determined percentages of redd dewatering due to water depth and velocity (tables in Appendix E, USFWS 2006), the river conditions experienced by each redd identified in the WRCS carcass survey (Killam 2023, CDFW 2024), and the duration of incubation periods from the SacPAS Egg-to-Fry Model tool (Anderson et al. 2022, CBR 2024). More specifically, the percentage of redds dewatered for cohort y was calculated as:

$$\hat{\rho}_y = \frac{\sum_{l=1,d=1}^{L,D}(R_{l,d} \times g_{l,d})}{\sum_{l=1,d=1}^{L,D}(R_{l,d})} \qquad \text{Eq. 2}$$

where $R_{l,d}$ is the number of redds created at location l, on day-of-year d, belonging to a particular cohort $C_{l,d}$ of redds (i.e., all redds with the same l and d values) , and $g_{l,d}$ is a pre-determined percentage of redds dewatered due to water depth and velocity from the tables in Appendix E of USFWS (2006). The value for g associated with $R_{l,d}$ is determined by looking up the relevant value based on: 1) the species (e.g., WRCS tables on p. 63-66; USFWS 2006]); 2) Anderson Cottonwood Irrigation District (ACID) Dam configuration (i.e., boards out [table on p. 63-64] or boards in [table on p. 65-66]), 3) the flow (Keswick Dam, KWK; USGS) associated with when and where the redd was created (table columns "Spawning Flow"); and 4) the minimum flow experienced by cohort $C_{l,d}$ during the incubation period from day of spawning to day of fry emergence from the redd (table rows "Dewatering Flow"). Values of initial and minimum flow were rounded downward to levels stipulated in the redd dewatering tables (USFWS 2006).

We determined the incubation period associated with redds $R_{l,d}$ using the SacPAS Egg-to-Fry Model tool (Anderson et al. 2022, CBR 2024) as the time to accumulate 1850 degree-Fahrenheit days (or 1028 degree-Celsius days; ATUs), a threshold used in the monitoring program (Chelberg 2023). The R script for this analysis is available in Appendix 1. Daily temperatures associated with each $R_{l,d}$ were determined using data from monitoring locations KWK (Sacramento R, Keswick, WQ; USBR; https://www.cbr.washington.edu/sacramento/data/query_river_graph.html), CCR (Sacramento R abv Clear Ck; USBR), and BSF (Sacramento R at Balls Ferry Bridge; USBR). Daily temperatures between KWK and CCR were determined by distance-weighted linear interpolation between these two monitoring sites. Downstream of CCR, redd temperatures were extrapolated using the KWK to CCR gradient for redd locations closer to CCR than BSF, and redds closer to BSF than CCR were determined with the distance-weighted linear interpolation between CCR and BSF.

To quantify any bias in the predicted vs observed percentages of redds dewatered, we looked at a ratio index, scaled by the mean of the observed across years:

$$\text{Bias index: } \frac{\hat{\rho}_y - \rho_y}{\frac{1}{N}\sum_y^Y \rho_y} \qquad \text{Eq. 3}$$

Thus, if the predicted percentage was the same as the observed percentage, the bias ratio index would be 0. If the predicted was double the observed, and the observed was close to the mean of observed across years, the bias ratio index would be close to 1. An index greater than 0 would represent predicted percentages greater than observed percentages, and an index less than 0 would represent predicted percentages smaller than observed percentages values. Scaling by the mean of the observed percentages across years prevents having 0 as a denominator and generates smoothing of the scalar by historical observations. Still, caution in interpretation is needed because the bias ratio index is not an absolute measure of bias, but one scaled by the mean of the observations.

Results

The estimates of female escapement (Killam 2023) and the number of redds monitored in the redd dewatering survey (Chelberg 2023) were correlated (*r* = 0.77; Figure 1). Thus, the comparison of observed percentages of redds dewatered across years for the population was reasonable.

In each year, the predicted percentage of redds dewatered due to water depth and velocity ($\hat{\rho}_y$) was greater than the observed percentage of redds dewatered due to depth (ρ_y) and the bias index was large in some years (Table 1). The lowest bias indices were in 2015 and 2022. The highest bias index was in 2013 with values greater than 100. The mean bias index across years was 49.9 $\pm$ SD 42.3 with ACID Dam boards out and 53.6 $\pm$ 43.5 with boards in, which altogether represents notable biases but also large variation across years. Furthermore, in a linear model between the observed and predicted percentages of redds dewatered (ρ_y= 0 + 0.0165$\hat{\rho}_y$; with SD 0.0074 for the slope; r^2 = 0.3538), the slope is quite shallow and different from a 1:1 line (Figure 2). With these results, the predicted percentages of redds dewatered with boards in were 60.6 (95% CI 32.3–501.0) times greater than the observed.

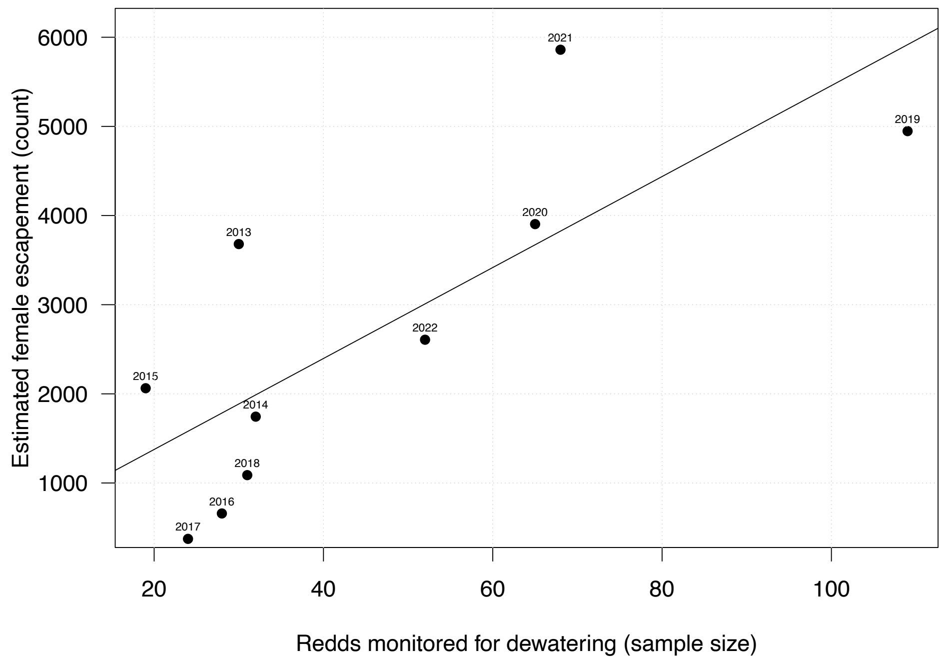

Figure 1. Linear relationship between the estimated female escapement (Killam 2023) and the sample size of redds monitored for dewatering (Chelberg 2023) in each year.

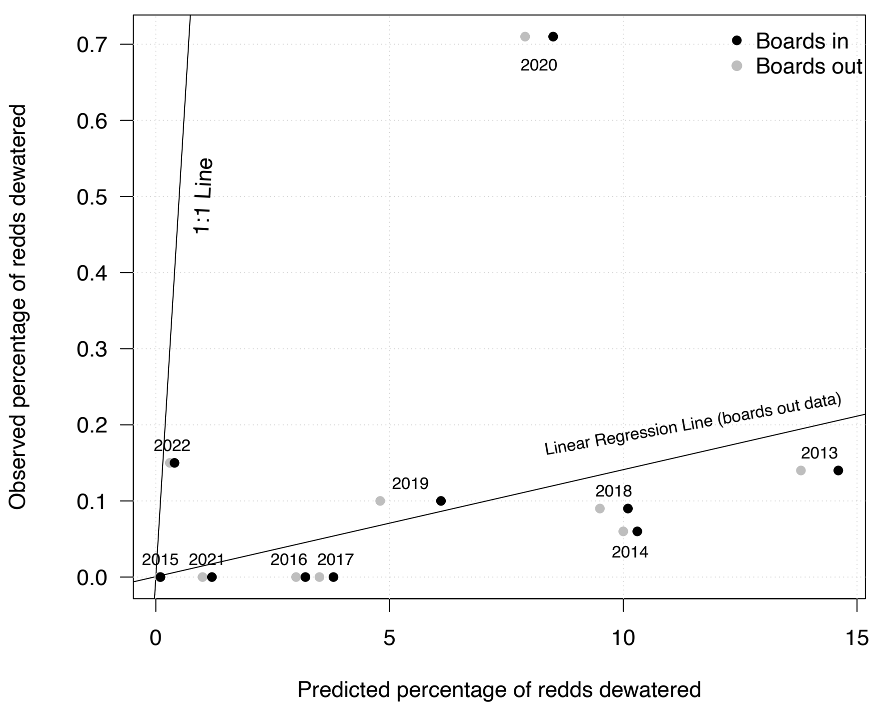

Figure 2. Observed percentages of redds dewatered due primarily to water depth (Chelberg 2023) compared to predicted percentages of redds dewatered due to water depth and velocity, using methods from USFWS (2006) and the SacPAS Egg-to-Fry Model tool (CBR 2024).

Table 1. Observed redd dewatering data (Chelberg 2023) and the percentage of redds dewatered, due primarily to water depth, calculated from Eq. 1. Predicted percentage of redds dewatered, due to water depth and velocity, calculated from Eq. 2, using the method from USFWS (2006) with inputs of redd counts based on the number of total females (Killam 2024), and other input parameters of redd distributions, incubation period (associated with 1028 degree-Celsius days of ATUs) and flows into the SacPAS Egg-to-Fry Model tool (CBR 2024). The bias index is calculated from Eq. 3.

Year	Redd dewatering survey data and estimates (From Table 1 in Chelberg 2023)				Data input and output estimates, using SacPAS Egg-to-Fry model (CBR 2024)			Bias index	
	Estimated female escapement (count)	Total redds monitored (count)	Dewatered redds observed (count)	ρ_y Minimum % redds dewatered out of estimated female escapement	Redd counts data input to SacPAS, based on number of total females in-river (Killam 2024)	$\hat{\rho}_y$ Predicted % redds dewatered using USFWS (2006) method: ACID Dam boards out	$\hat{\rho}_y$ Predicted % redds dewatered using USFWS (2006) method: ACID Dam boards in	ACID Dam boards out	ACID Dam boards in
2013	3680	30	5	0.14 %	3680	15.3%	16.0%	121.3	126.9
2014	1744	32	1	0.06 %	1744	11.6%	11.8%	92.3	93.9
2015	2063	19	0	0.00 %	2063	0.1%	0.2%	0.8	1.6
2016	658	28	0	0.00 %	658	3.9%	4.1%	31.2	32.8
2017	373	24	0	0.00 %	373	4.2%	4.5%	33.6	36.0
2018	1088	31	1	0.09 %	1088	11.5%	12.0%	91.3	95.3
2019	4947	109	5	0.10 %	4947	5.8%	7.3%	45.6	57.6
2020	3904	65	28	0.71 %	4023	9.7%	10.3%	71.9	76.7
2021	5860	68	2	0.00 %	6199	1.1%	1.3%	8.8	10.4
2022	2607	52	4	0.15 %	2663	0.4%	0.7%	2.0	4.4
2013-2022 Mean	2692.4	45.8	4.6	0.13 %	2743.8	6.4%	6.8%	49.9	53.6
2013-2022 SD	1854.6	28.0	8.5	0.21 %	1929.0	5.4%	5.5%	42.3	43.5

Discussion

Between 2013 and 2022, observed percentages of redds dewatered due primarily to water depth (Chelberg 2023) were only a small proportion of the predicted percentages due to water depth and velocity, based on the USFWS (2006) method, likely because these were not equivalent metrics of dewatered redds. There was also notable variation across years, to the extent that we could not determine a correction factor for the bias with a reasonable amount of certainty. Thus, if the predicted percentages of redds dewatered is used as a relative index, we caution its use due to the potential for any one annual prediction to differ greatly from the overall historical pattern. For example, the year 2020 exhibited the highest observed percentage of redds dewatered at 0.71% but had the fourth highest predicted percentage out of the 10 years investigated. The years predicted to have the highest percentages of redds dewatered were 2013 and 2014, which respectively had low and average observed percentages. Differences in the predicted and observed percentages may be due to differing criteria. Observed redd dewatering is based on water depth and some general conditions surrounding the redd, while predicted redd dewatering is based on water depth and velocity. Differences between predicted and observed percentages may also be due to: changes in where most, if not all, WRCS have been spawning in recent years, conceivably because of water temperature management; changes in operations over the years; river and spawning conditions; variation in escapement numbers; changing relationships among all these potential factors; assumptions in the methods of USFWS (2006); and assumptions in the methods applied to estimate egg and fry incubation periods.

There are differences in the definitions of “dewatered” redds for what is observed in the field and what is predicted. “Redds with tail spills in less than 30 inches of water, during initial measurements, were considered at risk of dewatering at future flow reductions. … As redds were remeasured, they were classified based on five stages of dewatering, ranging from fully submerged to totally dry. Dewatered redds were assigned a depth of negative inches to show how far exposed the gravel is from the top of the water column using a stadia rod and level. Redds were also designated dewatered if they are in isolated pools or in areas where there is no flow to oxygenate eggs” (Chelberg 2023). In contrast, the predicted percentages of redd dewatering are based on water depth and velocity. “We assumed that there would be reduced survival of eggs or pre-emergent fry, and thus spawning habitat would be lost, if the tailspill was exposed or if velocities dropped to the point where there was insufficient intragravel flow through the redd” (USFWS 2006). Thus, the higher percentages of predicted redd dewatering, based on criteria of depths below 0.52 ft or velocity below a threshold of 0.87 ft/s (see Table 9 in USFWS 2006), are not directly comparable to observations in the redd dewatering monitoring program (Chelberg 2023). Field observations do not include redds in deep water that would be considered “dewatered” when below the velocity threshold. Furthermore, assumptions underlying linear relationships, rather than non-linear relationships, are also possible reasons for differences between predicted and observed percentages of redd dewatering (for more details, see Appendix 2).

WRCS now spawn in the Upper Sacramento River, further upstream than where redds were observed in the USFWS (2006) study (D. Killam, CDFW, *pers. comm.*). Few, if any, WRCS spawn below the Clear Creek (CCR) gage where some of the sites were observed in the USFWS (2006) study. Gravel augmentation (Gorman and Marine 2010, USACE 2020) in the river near Keswick Dam has made the upper river, above ACID Dam, a spawning hotspot in deep water. For these reasons, redd dewatering may be occurring at lower percentages in recent years.

There may also be other factors that salmon are cueing to that we did not include in our analysis (e.g., population density). In 2020, the estimated female escapement was among the highest in recent years, and there was a high percentage of redd dewatering. There may have also been interactive effects between relatively high population density and low flow conditions in 2020. In addition to depth and water velocity, other potentially important physical characteristics that could be more explicitly studied include hyporheic flow, groundwater, and changes in sediment inputs with changes in runoff events and flows (Geist and Dauble 1998, Zimmermann and LaPointe 2005).

Overall, it is known that "[at] higher releases, more shallow spawning habitat is available than would be at lower releases, allowing salmon to spawn in areas that are at high risk of dewatering when releases drop. Ideally a lower sustained flow would be implemented prior to peak spawning periods to deter salmon from spawning in shallow areas. By doing this, if flows need to be lowered for water conservation purposes during incubation and emergence periods, then there is less impact on populations, as salmon will spawn in shallow areas during higher flows, when more shallow spawning habitat is available" (Chelberg 2023). This is seemingly straightforward but is an important consideration in light of tradeoffs in decisions related to higher releases. Furthermore, as part of the consideration to lower flows in shallow areas prior to peak spawning, a decrease in available spawning habitat would be counter to the efforts in recent years related to gravel augmentation and spawning pads (J. Chelberg, PSMFC, *pers. comm.*).

Near-real-time monitoring is in many ways more useable than the current prediction method that we assessed here. The prediction method, based on depth and velocity, is biased and variable in comparison to field observations that primarily uses water depth criteria. "Close communication between the regulatory agencies and the USBR over the past five years have led to the development of artificial flow regimes that have decreased winter-run redd dewatering" (Chelberg 2023). Still, pre-season planning is needed and improvements to methods that predict redd dewatering, particularly models that incorporate underlying mechanisms, are warranted. To the best of our knowledge, there are such works in progress, including the use of LiDAR data. Any continuation of these efforts could help refine predictions of redd dewatering with updated data and non-linear modeling (Appendix 2; Pearce and Ferrier 2000, Gard 2023).

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APPENDICES

Appendix 1

```
# R code to obtain dewater percentage using USFWS 2006 method.
# Generate dewatering for historical WRCS redd distributions,
# using SacPAS Egg-to-Fry Model (www.cbr.washington.edu/sacramento/fishmodel/),
# and collate.
#
# Available March 19, 2024

library(tidyverse)
years <- 2013:2022
DWfracOut <- DWfracIn <- redds <- rep(NA,length=length(years))
base <- "https://cbr.washington.edu/sac-bin/fishmodel/getandplottemp.pl?dewater=onkwk"
base <- paste0(base,"&dirUseId=thisDir&redds=dbcarcass&tempsource=dbtemp&atus=1028")
for( i in 1:length(years)){
         cat(years[i])
         this <- paste0("&reddyear=",years[i],"&tempyear=",years[i])
         out <- read_csv(paste0(base,this,"&raw=13"),show_col_types = FALSE)
         DWfracOut[i]   <- as.numeric(out[14,2])
         redds[i]       <- as.numeric(out[2,2])
         out <- read_csv(paste0(base,this,"&raw=13&dewatertype=boards"),show_col_types
       = FALSE)
         DWfracIn[i]   <- as.numeric(out[14,2])
}
dewaterUSFWS <- cbind.data.frame(years,redds,DWfracOut,DWfracIn)
print(dewaterUSFWS)
```

Appendix 2

A brief background on study design and observations used in the redd dewatering method (USFWS 2006) can help clarify assumptions and possible improvements of the method. Starting in 1995, the USFWS conducted a commendable series of studies to produce models for predicting characteristics of spawning sites of Sacramento River Chinook salmon (USFWS 1999), habitat suitability criteria (USFWS 2003), and redd dewatering and juvenile stranding (USFWS 2006). Aerial redd survey data, collected in 1989-1994 by Frank Fisher (California Department of Fish and Game) across six river segments, were used to determine which spawning mesohabitat units were most heavily used and thereby chosen for further study (USFWS 1999). Eight study sites were selected in Segments 6 through 4, after the USFWS conducted a reconnaissance in 1997 of mesohabitat units to determine feasibility of study sites, given riverbank and floodplain characteristics and with considerations on staff availability. Thus, spawning sites used for the redd dewatering analysis were in Lower Lake Redding, Upper Lake Redding, Salt Creek, Bridge Riffle, Posse Grounds, Above Hawes Hole, Powerline Riffle and Price Riffles (USFWS 1999). A total of 34 transects were placed in areas heavily used by spawning Chinook salmon. Given that only areas heavily used by spawners were included, results from these surveys may have differed if areas minimally (and not) used by spawners were included.

For WRCS, historically, 80% spawned in Segment 5 (Bridge Riffle–Above Hawes Hole), and ranges of 2%-9% spawned in Segment 6 (Salt Creek–Lower Lake Redding) and Segment 4 (Battle Creek to Cow Creek) through Segment 2 (Deer Creek to Red Bluff Diversion Dam) (1989-1994 survey data by

Frank Fisher, as referenced in USFWS 2003). Data on 227 WRCS redds in Segments 6–4 were then collected in 1996 (only deep areas) and 1998–2001 (shallow and deep areas). Sacramento River flows at KWK from mid-April through end of sampling ranged about 11,000–30,000 cfs in 1998, 8,500–14,000 cfs in 1999, 8,000–15,000 cfs in 2000 and 6,000–14,500 cfs in 2001. Even though it is not exactly known what the flows were at the time of redd site selection, it is noticeable that flow ranges in recent years have been wider than at the time of these studies and will likely widen further in range, in the future, due to climate change.

WRCS used depths of 1.2–15.6 ft, average water column velocity of 0.87–8.48 ft/s, and substrate types that were small gravel to medium cobble (0.1–1 inches to 4–6 inches; Table 2 in USFWS 2003). Depth habitat utilization and habitat suitability criteria (HSC) were determined under conditions from decades ago that are likely different from current conditions and warrant new observations. Furthermore, assumptions of lower limits may be too conservative: “We assumed that there would be insufficient intragravel flow through the redd if the spawning velocity was less than the lowest velocity at which we found a… winter-run chinook salmon redd in the Sacramento River... The lowest velocities we found in measurements of… winter-run chinook salmon [was]… 0.87 ft/s (U.S. Fish and Wildlife Service 2003)” (USFWS 2006). Assuming a lower value for the velocity threshold could yield higher estimates of habitat availability and lower the predicted percentage of redds dewatered. However, using this velocity threshold could be prudent and conservative, and thus worthwhile using.

In looking at the patterns of use and availability of habitat for redds over depths 3.5–15.5 ft (see Figure 19 in USFWS 2003): the proportional use of habitat declined as depth increased, while the proportional availability of habitat increased with depth from 3.5 ft to 8.5 ft, and then declined with depth to 15.5 ft. To determine a “standardized use/availability ratio”, an analysis was run with: 1) linear models to obtain predicted estimates of use as well as predicted estimates of availability, using the data up to 12.5 ft in depth; 2) determination of a use/availability ratio using the predicted estimates of use and availability; 3) scaling the use/availability ratio to a maximum of 1.0 (i.e., the “standardized use/availability ratio”); and 4) a linear model between the “standardized use/availability ratio” and depth. Given the data that was available and used in the linear model with the “standardized use/availability ratio”, caution in the use of its predictions is warranted given the non-linear patterns in the data. Additional data to assess any extrapolations and re-analyzing the data with non-linear models may help improve the predictions and with greater certainty.

Calibrated hydraulic decks were created to simulate depths and velocities at each of the 34 transects, and available habitat was simulated for each transect using input files of HSC and summarized as weighted usable area (WUA; sq ft) as a function of flow (cfs) (USFWS 2003, USFWS 2006). If the “standardized use/availability ratio” linear model plays a major role in determination of HSC and WUA, then improvements at this step of the analysis may help improve the predictions in percentage of redds dewatered.