**Factors influencing the accuracy and precision of aerial redd counts used to estimate Chinook salmon (*Oncorhynchus tshawytscha*) abundance**

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**Abstract = 175 words max; 1st cut--more substantive result needed**

Redd counts are commonly used to monitor trends in salmon populations when adult escapements are unknown. Despite widespread use, few studies have investigated the accuracy of aerial redd counts or the factors that decrease precision and introduce bias. We examined probabilities of aerially detecting Chinook salmon redds by addressing four objectives to: 1) census redds and develop an unbiased estimate; 2) apply the census as a baseline for estimating redd count accuracy; 3) evaluate the effectiveness of a mark-resight approach for measuring count accuracy; and 4) evaluate the influence of environmental and redd-level characteristics on redd detectability. We censused redds in 30 stream reaches across six major drainages and compared independent aerial counts to redd census data. Errors were measurable and count bias was predictable. Most aerial counts contained errors that varied annually, by site, and increased at larger redd densities. Aerial counts tended to be negatively biased and errors of commission were rare. Redd contrast and age, riparian canopy, and shading were important predictor variables. Our results may be applied to improve aerial redd counts.

Keywords: Aerial redd counts, Accuracy, Precision, Bias, Chinook salmon, Abundance

**Introduction = 500 words max**

Abundance is the state variable of interest in most population–level ecological research (Nichols and MacKenzie 2004) and is also **essential for calculating vital statistics (Ricker 1975). Direct abundance observations are important for identifying independent populations, represent a key viability parameter, and a primary delisting criterion (McElhany et al. 2000). Consequently, it is critical to understand the accuracy and precision of abundance estimates (Chasco et al. 2014).**

Columbia River Basin (CRB) Chinook salmon (*Oncorhynchus tshawytscha*) distribution and abundance have declined as a result of major anthropogenic changes (Nehlsen et al. 1991; NRC 1996; Thurow et al. 2000). As an integral part of efforts to restore anadromous salmonids (Lee and Grant 1995; NMFS 2000a), the NWPPC (1994) has emphasized the need for long-term monitoring of salmon stocks. Simultaneously, a variety of analytical efforts support evidence-based restoration decisions (e.g., McCann et al. 2019) and require reliable abundance estimates for long-term monitoring and assessments of management actions. The usefulness of any population survey method depends upon obtaining unbiased, or nearly unbiased, and precise estimates in a cost-efficient, logistically feasible manner (Thompson et al. 1998). Accurate abundance estimates are available for a fraction of Pacific salmon populations (McElhany et al. 2000) because estimates are often unavailable or infeasible. Consequently, relative rather than direct indices are often used to assess populations.

Since the 1940s, CRB biologists have employed redd (nest) counts as a surrogate for monitoring spring/summer Chinook salmon (Hassemer 1993; Emlen 1995). Despite widespread use of redd counts to calculate measures of salmon population performance, the accuracy of aerial redd counts and the factors that affect count precision and bias have rarely been evaluated. Researchers have compared single “peak” redd counts to multiple pass counts (Kucera and Orme 2007), compared counts between multiple observers or crews (Gallagher and Gallagher 2005), or compared counts derived by different methods (e.g., visual surveys vs. aerial photography; Visser et al. 2002). In each approach, however, actual redd abundances and the proportion of redds detected remained unknown.

The relationship between a raw or unadjusted redd count and actual redd abundance is the detection probability (Engeman 2005). Many biologists assume uncorrected redd counts represent a *constant proportion* of the true numbers of redds or the *true number* of redds without error. With no correction for omitted or mis-identified (committed) redds, the bias is assumed to be constant across space and time. That assumption is untenable given the variety of factors affecting redd detection. Failure to account for detection errors may cause errors of inference (Thompson et al. 1998), particularly when detection probabilities vary with habitat types, observers, or other factors (Anderson 2001, 2003). Relying on a raw, unadjusted count may lead to misleading conclusions about population trends, spatial distribution, and habitat associations.

We initiated research to understand accuracy and precision of aerial Chinook salmon redd counts. If redd detection probabilities are measurable, differences in detection can be accounted for. We established four objectives to: 1) census redds and develop an unbiased estimate of the true number of redds; 2) apply the redd census as a baseline for estimating aerial count accuracy; 3) evaluate the effectiveness of a mark-resight approach; and 4) evaluate the influence of environmental and redd-level characteristics on redd detectability. Our results have important implications for improving Chinook salmon aerial redd surveys.

**Materials and methods**

Study area

The Middle Fork Salmon River (MFSR), one of eight original National Wild and Scenic Rivers designated in 1968 (NWSRS 2016), drains about 7,330 km2 of a remote area of central Idaho and, for most of its length, flows through the Frank Church River of No Return Wilderness. From its origin at the confluence of Bear Valley and Marsh Creeks, the MFSR flows north-northwest for 171 km through the Salmon River Mountains and joins the Salmon River 92 km downstream from Salmon, Idaho and 1,144 km from the Pacific Ocean. From 1930-1980, the US Forest Service (1998) managed most of the region in “Primitive Area” status. In 1980, the Central Idaho Wilderness Act established 906,136 hectares of wilderness that remains the largest contiguous wilderness in the lower 48 states and the largest in the National Forest system (US Forest Service 1998). Minshall et al. (1981), Thurow (2000), Servheen et al. (2001), and Thurow et al. (2020) provide more detailed study area descriptions.

Native fishes include seven salmonid taxa: bull trout, westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), redband trout (*O. mykiss* ssp.), mountain whitefish (*Prosopium williamsoni*), summer steelhead (*O. mykiss*), and spring/summer Chinook salmon (Thurow 1985). Adult spring Chinook salmon cross Bonneville Dam (the first dam encountered) predominately from March to May and summers from June to July (Burner 1951, Mathews and Waples 1991). Chinook salmon represent a continuum of forms falling along a temporal cline related to incubation and rearing temperatures; their population structure in the CRB reflects the genetic composition of founding sources within the region, shaped by the environment (principally temperature), and resulting in a life history evolutionary strategy to maximize fitness under the conditions delineated (Brannon et al. 2004). Within the MFSR, spring-run fish are predominant and summer-run fish occur sympatrically in several drainages with spring-runs spawning earliest at higher elevations compared to summer-runs that spawn later at lower elevations (Thurow et al. 2020).

Chinook salmon in the MFSR are designated as the Middle Fork Salmon River Major Population Group (MPG) within the Snake River spring/summer Chinook salmon Ecologically Significant Unit (ESU). Concern for the persistence of the Snake River Chinook salmon ESU culminated in a final rule (Office of the Federal Register 57[April 22, 1992]:23458) that listed Snake River spring/summer Chinook salmon as “Threatened” under the Endangered Species Act of 1973 (ESA). Despite abundant, high quality natal (spawning and rearing) habitat; absence of hatchery fish; and low ocean harvest rates verified by tag returns (Schaller et al. 2014); MFSR Chinook salmon and steelhead remain at risk of extirpation; primarily as a result of outside-basin factors in the Columbia and Snake River migration corridors, estuary, and ocean (Thurow et al. 2020). Anadromous fish destined for the MFSR navigate eight dams (four in the Columbia River and four in the lower Snake River) as smolts and ascend those same dams as adults returning to spawn. Adverse effects of these dams and impoundments on salmon and steelhead survival are well documented (Raymond 1979; Schaller et al. 1999; Petrosky et al. 2001). Eight Chinook salmon populations within the Middle Fork Salmon River MPG are non-viable and at high risk of extinction (NMFS 2017).

About 800 km of tributaries and the mainstem MFSR are accessible to Chinook salmon (Mallet 1974; Thurow 1985) and connectivity among its populations is high (Fullerton et al. 2016). Ten major tributaries and the mainstem MFSR currently support Chinook salmon spawning (Thurow et al. 2020).

**Redd Census**

Redd counts have many benefits for estimating Chinook salmon abundance. Counts are less intrusive than methods that require capture and handling of fish. Chinook salmon redds tend to be large (4-5 m2) (King and Thurow 1991), highly visible, and persist for weeks to months. Redd counts can provide an index of effective population size (Meffe 1986) and, coupled with age data, counts can be used to calculate population growth rates and examine metapopulation dynamics (Isaak and Thurow 2006). In addition, other key information may be obtained by collecting tissues from salmon carcasses encountered during redd surveys. Fin ray sections provide spawner ages (Copeland et al. 2007) and DNA may be extracted from tissues (Neville et al. 2006).

We applied intensive (every 4-5 days) surveys and censused redds to develop an unbiased estimate of the “true” number of redds in a series of study reaches. Study reaches were selected non-randomly and were selected to encompass a range of morphological channel types (confined, steep gradient reaches; unconfined, low gradient reaches; and intermediate reaches), canopy covers (open, dense, and intermediate), and redd densities. To assist collaboration and increase efficiency, many study reaches were selected to coincide with previously established redd index reaches (Thurow et al. 2020) annually monitored by IDFG, the Nez Perce and Shoshone-Bannock tribes, and the USDA-Forest Service. We established 30 study reaches across six major spawning tributaries and the upper mainstem MFSR (Figure 1-see PPT slide 1). Our sample size was robust because our study reaches included a large percentage (26-46%) of the total redds annually observed with the MFSR (Table 1).

Table 1. Percentage of Chinook salmon redds annually observed within study reaches in the Middle Fork Salmon River basin, 2001-2005.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | Number of study reaches | Aerial redd count within study reaches | Total, annual redd count in the basin | % redds within study reaches |
| 2001 | 8 | 555 | 1789 | 31% |
| 2002 | 16 | 789 | 1730 | 46% |
| 2003 | 11 | 764 | 2271 | 34% |
| 2004 | 14 | 289 | 832 | 35% |
| 2005 | 9 | 121 | 458 | 26% |
| Total |  | 2522 | 7080 | Mean= 36% |

Prior to the onset of Chinook salmon redd construction in late July, we used aerial photos and onsite visits to develop physically detailed maps of each study reach. To facilitate data recording, we divided longer reaches into 400 m subsections and flagged boundaries. Reach lengths ranged from 0.3 km to 8.1 km.

Trained observers (redd monitors) were accompanied by experienced redd counters and began surveying reaches prior to the initiation of spawning. Monitors installed continuously recording thermographs at the upstream and downstream boundaries of each study reach. During their initial survey, monitors recorded all “false redds”, defined as features that might be mis-identified as newly constructed Chinook salmon redds. False redds included still-visible Chinook salmon redds constructed the previous year, steelhead redds constructed earlier that spring, and redd-like features that could be misidentified as redds but were caused by hydrologic scour, beaver or other animal activity, and wading anglers. After the initial survey, monitors walked the stream banks adjacent to each study reach on a 4-5 day interval, searched for newly completed redds, recorded locations with a GPS (Global Positioning System), and flagged redds. Monitors took precautions to avoid disturbing ESA listed salmon and GPS locations were obtained in the least obtrusive manner possible. Redd counts were conducted between 0930 and 1700 h to maximize direct sunlight, monitors wore polarized sunglasses and recorded completed redds by 400 m subsections.

Redds were typically found in locations that contained sorted gravels and suitable water depths and velocities in unconfined and meandering valley segments, pool tails (transitions between pools and riffles), confluences of tributaries or debris flows, and sites adjacent to instream roughness elements such as wood and boulders. Multiple criteria were applied to verify completed redds, including: 1) presence of live adult salmon or carcasses near a suspected redd; 2) an elliptical area of disturbed gravel of sufficient size and oriented parallel with the current; 3) a 3-dimensional streambed morphology with a visible pit and tailspill (Burner 1951); and 4) in a location where natural scour and deposition of substrate were unlikely to produce redd-like morphologies. Redd dimensions are proportional to female length (Crisp and Carling 1989), and MFSR Chinook salmon were of similar length to those measured by King and Thurow (2000) who reported a disturbed redd area of nearly 5 m2.

Surveys continued through the spawning period until all redds were completed. Depending on reach elevations, spawning was completed between late August and mid-September. In addition to completed redds and false redds, monitors also encountered smaller redds of non-target fall spawning bull trout, test redds which were typically small and oval without a completed redd morphology, and in-progress redds with adult salmon still actively spawning. In-progress redds were marked and inspected on subsequent surveys and most were ultimately completed. Decades of observations in the Salmon River basin, Idaho suggested that female Chinook salmon typically remained on newly constructed redds for a minimum of 3-5 days. We assumed that a census completed on a 4-5 day frequency would detect all newly constructed redds. All recorded redds were integrated to produce a cumulative redd map for each study reach. Maps served as baselines for comparison with subsequent independent aerial redd counts.

Simultaneous to RMRS redd monitor surveys, IDFG biologists completed an average of 3 multiple-pass surveys and georeferenced redds in sections of Marsh Creek and its tributaries Beaver, Cape Horn, and Knapp creeks (Thurow et al. 2020). We merged the two data sets to increase the number of reaches with redd census baseline data.

**Covariates**

Existing literature (Bonneau and LaBar 1997; Maxell 1999; Dunham et al. 2001; Gallagher and Gallagher 2005; Muhlfeld et al. 2006; Murdock et al. 2019) and our own experience (Thurow and McGrath 2010) led us to evaluate three classes of variables that may affect the probability of aerially detecting salmonid redds: 1) variables associated with observers, such as training, experience, eyesight, interest, and fatigue level; 2) variables associated with the environment, such as substrate size and color; water depth, clarity, and turbidity; periphyton growth; presence of bed features that mimic redds (e.g., natural hydrological scour), and sun angle and shading; and 3) redd-level variables, such as redd age, size, morphology, color, contrast with surrounding substrate, density, overlap with other redds, and spatial arrangement within a reach. Counting errors resulting from these variables may obscure important population trends (Beland 1996) and models that account for variation in detection rates will improve estimates of redd abundance.

***Observer variables***

To avoid introducing interobserver bias (Bonneau and LaBar 1997; Dunham et al. 2001), all aerial surveys were conducted by the same observer (R. Thurow, RMRS). The aerial observer had 10 years of experience aerially and annually counting Chinook salmon redds in the same MFSR streams and nearly 30 years of experience completing ground-based salmonid redd surveys. As a result, we assumed observer training, experience, interest, eyesight, and fatigue level did not vary during the aerial surveys.

***Reach-scale environmental variables*** Covariates in a Table 2- here or in the modeling section?

After spawning was completed in each reach, we measured a series of environmental characteristics with the potential to affect redd count accuracy. We also retrieved the thermographs that had been installed when reaches were established. Thermographs recorded hourly water temperatures from the initial reach survey until spawning was completed. A two-person team measured habitat variables that described reach physical characteristics. Reach-scale variables including wetted width, stream depth, channel slope, large wood abundance, riparian characteristics, and the number of channels. We measured variables along transects perpendicular to the flow and spaced at 200 m intervals. Coordinates of each transect were recorded using a GPS. Reach measurements began at the downstream end and progress upstream. To minimize interobserver variability, the same person on each team measured variables while the other person recorded data. As during redd counts, crews took care to avoid disturbing redds.

Variables were measured as follows. Wetted width was measured with a tape run perpendicular to the direction of stream flow. When multiple channels were encountered, we recorded the number of wetted channels intersected by each transect that were wide enough (2 m) to support Chinook salmon spawning and measured all other variables in those channels. Stream depth was measured along each transect to the nearest cm at ¼, ½, and ¾ the stream width. Slope was measured as the change in elevation along each study reach, divided by the reach length. We counted pieces of wood at least 10 cm in diameter, 1 m long, and lying within the wetted width of the reach. Three methods characterized riparian characteristics: a) we classified the dominant riparian flora that shaded the stream as tree, shrub, or grass; b) we used a clinometer to measure the angle (in degrees) to the top of riparian vegetation on each side of the reach and used this angle in combination with stream width to calculate the height of riparian vegetation, and c) we used a densiometer to measure riparian density at four locations (upstream, downstream, right bank, left bank) from the mid-point of each transect. Finally, we used a conductivity meter as a surrogate for measuring water clarity at upper and lower boundaries of reaches. Conductivity is an easily measured and important indicator of stream health; increased runoff and more stream ions elevate conductivity (Morgan et al. 2012).

We also applied variables including current year peak flow, prior year aerial redd count, and aerial survey conditions. Current year peak flow was obtained from the Middle Fork Salmon River stream gauge near Middle Fork Lodge from November 1 to August 1 in same year as the September aerial survey. The prior year aerial count was applied for each reach. When aerial survey conditions were poor (e.g., high winds or stream turbidity caused by storms), they were noted by the aerial observer.

***Redd-scale variables***

Each team also measured a series of redd-scale variables that have the potential to affect redd count accuracy. Redd-scale variables included redd age, redd contrast, adjacent water depth, overlap with adjacent redds, distance to nearest bank, stream width, adjacent riparian vegetation, and percent cover. Variables were measured as follows: Redd age was derived by counting the days between the first date a completed redd was recorded (during the census) to the date of the independent aerial survey. We estimated the average age for all redds in each reach. The percent of redds in the sun was derived with a GIS shade model that applied aspect, topography, riparian vegetation height, and sun position at time of the aerial survey to predict whether redds were shaded or in the sun. Within each reach, we calculated the percent of redds in the sun at the time of the aerial survey. Redd contrast was visually estimated by comparing the contrast of the gravels inside the redd to undisturbed substrate outside the redd. Monitors rank the amount of contrast from 1 to 5, with 5 being the largest contrast and 1 the least contrast. Adjacent water depth was measured as the maximum water depth over undisturbed gravels adjacent to the redd pit. Shortest bank depth was measured as the shortest distance (cm) between the nearest bank and the edge of the redd pit. Stream width was measured as the wetted stream width adjacent to the middle of the redd and perpendicular to the direction of stream flow. Riparian vegetation adjacent to redds was characterized by three methods. First, we inspected the dominant riparian flora (which shaded the stream) on the banks nearest the redd and classified it as tree, shrub, or shrub. Next, we used a clinometer to measure the angle to the top of riparian vegetation on the opposite side of stream. Last, we used a densiometer to measure riparian density at four locations (upstream, downstream, right bank, left bank) adjacent to each redd. Percent cover was estimated as the proportion of the redd surface area that was obscured from sight (to the nearest 5%) when viewed directly from above. Cover elements that obscured redds included large woody debris, surface turbulence, overhanging terrestrial vegetation, aquatic macrophytes, and undercut banks. Percent overlap was visually estimated (to the nearest 5%) as the proportion of the redd that overlapped with an adjacent redd. Average nearest neighbor distance per reach was derived by averaging the distance between redds and the redds closest to them.

We also applied redd density and total salmon escapement variables. Redd density was estimated by dividing the total redds observed during the census by the reach length. The total adult Chinook salmon escapement returning to the Middle Fork Salmon River in the aerial survey year was obtained from the annual, cumulative redd count database compiled by RMRS (Thurow et al. 2020).

**Analytical approach** \*\*To be drafted by Claire and Kevin

We applied a multi-step approach to evaluate factors influencing the bias and precision of aerial Chinook salmon redd counts. First, we applied the redd census as the baseline for estimating aerial redd count accuracy. Next, we modeled the reach- and redd- scale variables to examine their effects on redd sightability. We also assessed inter- and intra- year sources of variation in redd counts and compared evaluated the effectiveness of a modified two-sample, Lincoln-Petersen mark-resight estimator for obtaining unbiased and precise redd abundance estimates.

Notes to elaborate on:

-Net error: the ratio of observed redds to true redds, modeled using *log*(net error)

-Omission rate: proportion of redds available to be counted that were missed by the observer

-Commission rate: proportion of redds counted by an observer that were not redds

-Covariates were Normalized (Z scores)

-Model sets were developed to investigate covariates related to reach-scale environmental variables and redd-scale variables as fixed effects

-Some of the covariates we attempted to measure in the field were ultimately not included in the analysis because large numbers of records were missing (e.g., turbulence, conductivity) or improperly collected (e.g., riparian height).

-Reach and year covariates were modeled as random effects

-Models fit using *glmer* and *lmer* functions in R

-Models evaluated via *AICc*

\*Also, Claire has built lots of tables and appendices that present the data. It might be useful to include some of those in “Supplemental materials” as allowed by CanJ via an online link?

**Results** \*\*To be drafted by Claire and Kevin

We found that aerial redd counting errors were measurable and count bias was predictable.

**Study Reaches**

Thirteen of the original study reaches were subsequently omitted from our analysis because of incomplete data. Omitted sites included Marsh 2, Capehorn Cr (Marsh 4), Knapp Cr (Marsh 5), Elk 1, Sulphur 1 and 3, Big 4, 6, 7, 8, and 9, Loon 4, and Mainstem MFSR 1. As a result of some surveyors not following data collection protocols, we also omitted data collected in certain reaches in individual years (e.g., 2001 for certain surveyors). The final dataset included redd census data collected within 17 study reaches ranging from 1.6 to 8.1 km total length. These reaches were sampled for one year (two reaches), two years (three reaches), three years (one reach), four years (ten reaches), and five years (two reaches). Monitors completed a total of 59 individual redd census surveys.

**Temporal and spatial variation**

Most aerial redd counts contained errors that varied temporally by year and spatially by study reach. Aerial counts tended to be negatively biased, commission errors were rare, and errors of omission were most prevalent. Errors of omission varied by year and increased with increasing redd abundances (Figure 2- see PPT slide 2). Net error also varied across years (Figure 3) and study reaches (Figure 4).

**Model outputs**

Important predictor variables included redd density, contrast, and age; riparian canopy; sunlight on redds; distance between redds; and redd overlap. \*Possible Figures 5,6,7 in PPT PLUS Kevin built lots of other Figures that could be included here or in Supplemental materials?

*Subheadings?*

**Discussion**

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**Tables**

Table 1. Percentage of Chinook salmon redds annually observed within study reaches in the Middle Fork Salmon River basin, 2001-2005.

Table 2. Reach- and redd-scale covariates.

**Figures** -see the PPT: 21\_Redd Validation\_possible Figures

Figure 1. Redd census study reaches within the Middle Fork Salmon River basin, Idaho.