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ARTICLE

Imperfect detection and misidentification affect inferences from data informing water operation decisions

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Abstract

Objective: Managers can modify river flow regimes using fish monitoring data to minimize impacts from water management infrastructure. For example, operation of the gate-controlled Delta Cross Channel (DCC) in California can negatively affect the endangered Sacramento River winter-run Chinook Salmon *Oncorhynchus tshawytscha*. Although guidelines have been developed for DCC operations by using real-time juvenile fish sampling count data, there is uncertainty about how environmental conditions influence fish occupancy and the extent to which those relationships are affected by sampling and identification error.

Methods: We evaluated the effect of environmental conditions, imperfect detection, and misidentification error on salmon occupancy by analyzing data using hierarchical multistate occupancy models. A total of 14,147 trawl tows and beach seine hauls were conducted on 1058 sampling days between October and December from 1996 to 2019. During these surveys, 2803 juvenile winter-run Chinook Salmon were identified, and approximately 29% of the sampling days had at least one winter-run juvenile detected.

Result: The probability of misidentifying an individual juvenile winter-run Chinook Salmon in the field was estimated to be 0.056 based on fish identification examinations and genetic sampling. Occupancy varied considerably and was related to flow characteristics, water clarity, weather, time of year, and whether occupancy was detected during the previous sampling day. However, these relationships and their significance changed considerably when accounting for imperfect detection and the probability of misidentifying individual juvenile salmon. Detection was <0.3 under average sampling conditions during a single sample and was influenced by flow, water clarity, site, and volume sampled.

Conclusion: Our modeling results indicate that DCC gate closure decisions could occur on fewer days when imperfect detection and misidentification error are not

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accounted for. These findings demonstrate the need to account for identification and detection error while using monitoring data to assess factors influencing fish occupancy and inform future management decisions.

KEYWORDS
Chinook Salmon, detection, error, identification, modeling, occupancy, water management

INTRODUCTION

The alteration of natural river flow regimes and routing caused by water management infrastructure (e.g., dams, diversions, levees, export facilities, etc.) is a major stressor causing freshwater fish imperilment in North America (Richter et al. 1997; Bunn and Arthington 2002; Jelks et al. 2008). Thus, water resource managers are increasingly being tasked with considering the impacts to fishes when making decisions concerning water operations and infrastructure development (Naiman et al. 2002; Cartwright et al. 2017). Balancing the human needs for freshwater while maintaining or restoring populations of fish and their habitats is becoming more challenging as water resources become scarcer and overutilized based on a changing climate and increased human demand (Poff et al. 2003; Arthington et al. 2006; Haddeland et al. 2014). A good example of this situation can be seen in the San Francisco Estuary (henceforth, "the Estuary") in the Central Valley of California (Figure 1).

The Estuary and its watershed constitute a vital socioeconomic and ecological resource. The Estuary provides urban drinking water for millions of residents, irrigation water to support a US$37 billion agricultural industry, and habitat for over 40 freshwater, estuarine, euryhaline, and anadromous fish species, including the endangered Sacramento River winter-run Chinook Salmon *Oncorhynchus tshawytscha* (Moyle 2002; Lund 2016). Currently, natural resource managers are tasked with achieving "co-equal goals" of ensuring a more reliable water supply for Californians while protecting and enhancing ecosystem attributes, including native fish (Delta Stewardship Council 2013; Lund 2016). Unfortunately, land use and water management during the past century have caused substantial environmental degradation (Nichols et al. 1986; Healey et al. 2008, 2016) and declines in native fish (Brown and Moyle 2005; Lindley et al. 2006; Sommer et al. 2007). Managers working in the Estuary and its watershed rely on fish count data to inform water operation decisions related to planning, assessments, and real-time surface water management (Kimmerer 2004; Brown et al. 2009; Brown and Bauer 2010; Reis et al. 2019). The validity of the fish count data that are used to inform these decisions can be affected by several sources of uncertainty, including imperfect and variable detection (Perry et al. 2016; Peterson and Barajas 2018) and species misidentification (Kirsch et al. 2018). The imperfect detection of a fish within a sample introduces false-negative error into the data, whereas the misidentification of a fish within a sample can introduce both false-positive and false-negative errors into the data. False-negative error occurs when a species is not recorded in a sample unit where it is present and can cause the underrepresentation of a species within a sample. Conversely, false-positive error occurs when a species is recorded in a sample unit where it is absent (i.e., confusing an alternative species as the target species), which can overrepresent a species within a sample. If both false-negative and false-positive errors exist in fish count data and are not accounted for, the overall effect of the error on inferences made using the count data is unclear and may misinform water operation decisions.

Impact statement

The San Francisco Estuary and its watershed provides urban drinking water for millions of residents, supports a US$37 billion agricultural industry, and serves as habitat for over 40 fish species, including the endangered winter-run Chinook Salmon. This study assessed the management of river diversion gates relative to protecting juvenile winter-run Chinook Salmon within the estuary using fish minitoring data while accounting for imperfect detection and misidentification. We found that imperfect detection and misidentification error altered our perception of the factors influencing juvenile winter-run Chinook Salmon occupancy and could affect water and fish management decisions. Our assessment approach and findings are applicable to any management decision in any system that is informed using fish count data and demonstrates the importance of accounting for all forms of sampling error when using count data to inform natural resource management decisions.

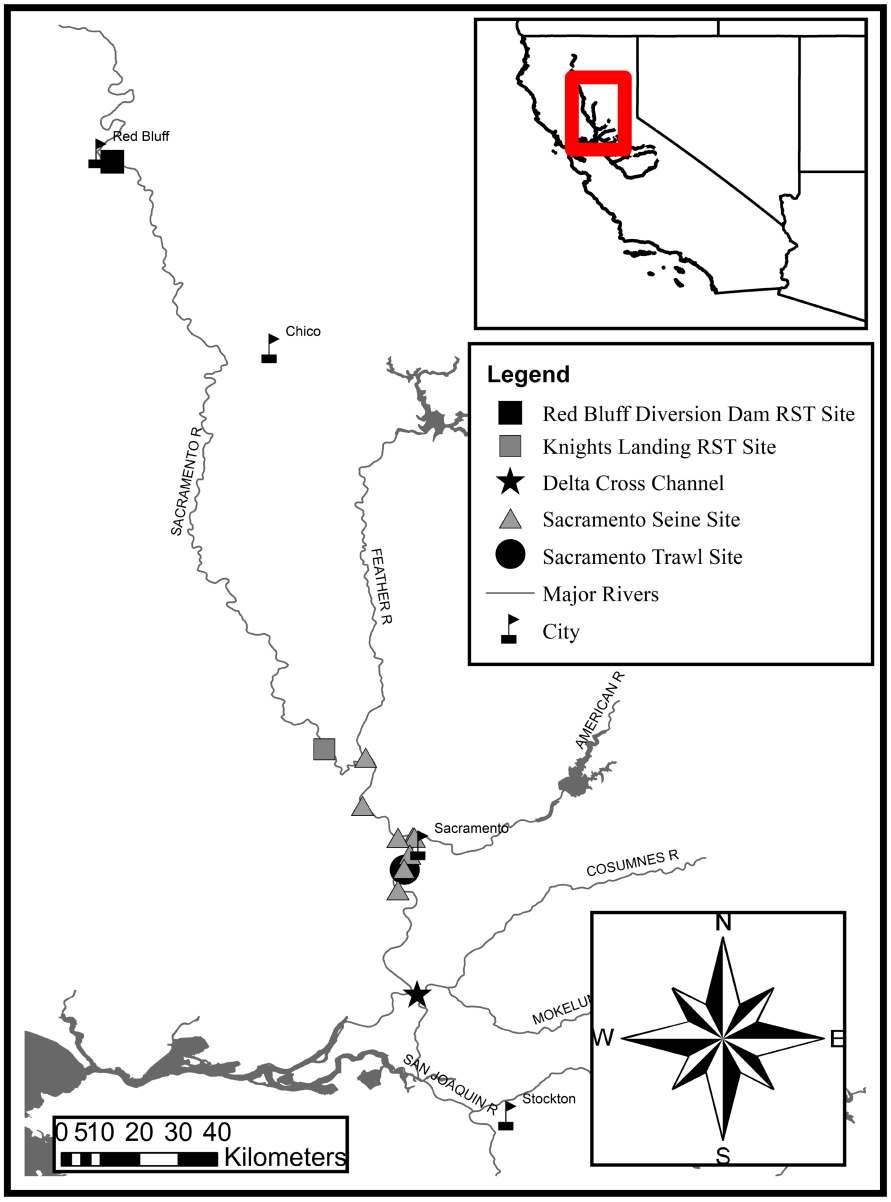

FIGURE 1 Map of the nine locations in or near the Sacramento River that were monitored by the U.S. Fish and Wildlife Service's Delta Juvenile Fish Monitoring Program to inform real-time Delta Cross Channel gate closure decisions during October–December from 1996 to 2019. The map also shows the locations of the Knights Landing rotary screw trap (RST) site, the Red Bluff Diversion Dam RST site, and the Delta Cross Channel referenced in this study.

The operation of the gate-controlled Delta Cross Channel (DCC) located in the Estuary has been directly informed by real-time fish monitoring count data since 1995 without any explicit consideration of imperfect detection or misidentification error. The DCC and its two radial gates were constructed by the U.S. Bureau of Reclamation in 1951 to divert freshwater from the Sacramento River into the interior Sacramento–San Joaquin River Delta (henceforth, "the Delta") to improve the water quality of diversions to the State Water Project and Central Valley Project export facilities (State Water Resources Control Board 1978). The opening of the DCC gates alters river and tidal flows in the Estuary and has been shown to incidentally entrain up to 80% of acoustically tagged emigrating juvenile Chinook Salmon into the interior Delta (Perry et al. 2010, 2012, 2015, 2018; Romine et al. 2013; Steel et al. 2013; Pope et al. 2021). Although entrained individuals may still complete their out-migration, studies have indicated that these individuals may experience considerably higher mortality rates based on exposure to water exportation, high temperatures, increased predation risk, and pollution (Kjelson and Brandes 1989; Brandes and McLain 2001; Newman and Brandes 2010; Buchanan et al. 2013). Consequently, juvenile Sacramento River winter-run Chinook Salmon (henceforth, "juvenile winter-run salmon") can be

negatively affected by DCC gate operations while these fish occur in the Delta (typically from October to June; Fisher 1992, 1994; Williams 2006; del Rosario et al. 2013).

Guidelines developed by the State Water Resources Control Board (State Water Resources Control Board 1995, 1999, 2006) and the National Oceanic and Atmospheric Administration's National Marine Fisheries Service (NMFS 1997, 2009, 2011, 2019) have generally mandated that the DCC gates be closed from December to May to protect, in part, juvenile winter-run salmon. Furthermore, the DCC gates may be temporarily closed during October–December, when real-time fish monitoring in the Sacramento River indicates that juvenile winter-run salmon are entering the Estuary near Sacramento, California. These gate closure decisions have been made by following an interagency salmon decision process that has been initiated when at least one of three salmon monitoring efforts in the Sacramento River independently detects three or more winter-run-sized (or larger) juvenile salmon during a day after the count data are standardized to 1 day of typical sampling effort (see NMFS 1997, 2009, 2011, 2019 for more information). Monitoring efforts that are used to initiate the decision process include continuous rotary screw trapping at Knights Landing, which has been implemented by the California Department of Fish and Wildlife (CDFW) from 2003 to present; and surface trawling at Sherwood Harbor and beach seining at eight fixed sites, which have been implemented by the U.S. Fish and Wildlife Service (USFWS) from 1996 to present (Figure 1). One day of typical sampling effort for the CDFW rotary screw trapping was defined as 24 h, whereas the typical daily sampling effort for the USFWS monitoring was defined as 10 separate tows for the surface trawling and eight sites sampled for the beach seining. This approach and these criteria correct for daily sampling effort but do not include adjustments for imperfect detection or misidentification of juvenile winter-run salmon as belonging to one of the other three Chinook Salmon runs in the Central Valley (i.e., fall, late-fall, and spring runs) using fish length at date of capture river criteria (known as LDC; see Fisher 1992; Greene 1992). In general, there is considerable uncertainty about how environmental conditions influence juvenile winter-run salmon occupancy and detection at the Sacramento River fish monitoring sites and about whether an understanding of those relationships could be affected by fish sampling methods and identification error. These observation errors could put juvenile winter-run salmon at risk if a lack of detection fails to trigger a DCC gate closure when winter-run individuals are present. Conversely, misidentification of other fish as juvenile winter-run salmon, triggering a DCC closure when winter-run individuals are absent, may inadvertently impair managers from diverting water into the interior Delta.

In this study, we assessed the effect of environmental conditions, false-negative (imperfect detection) error, and false-positive (misidentification) error on juvenile winter-run salmon occupancy near Sacramento using fish count data that have been used to inform real-time DCC gate closure decisions. Our objectives were to (1) estimate the occupancy of juvenile winter-run salmon by using count data collected after 1995 that have been used to inform DCC gate closure decisions; (2) assess the environmental factors that may influence the occupancy of juvenile winter-run salmon near Sacramento while ignoring and accounting for misidentification and imperfect detection; and (3) evaluate the effect of imperfect detection and misidentification on our perception of the factors influencing winter-run salmon occupancy and the extent to which the error can affect the initiation of the interagency salmon decision process.

METHODS

Fish monitoring data

We assessed the occupancy of juvenile winter-run salmon by using solely the USFWS Sacramento trawl and beach seine data. We obtained daily Sacramento trawl and beach seine data from the Delta Juvenile Fish Monitoring Program (DJFMP) conducted by the USFWS Lodi Fish and Wildlife Office. This data set represented juvenile winter-run salmon catch counts for all Sacramento trawl and beach seine samples collected during October–December from 1996 to 2019. In general, the Sacramento trawl and beach seine sites were sampled 3 days per week during this time period and were typically sampled on the same days each week. All sampling was conducted between sunrise and sunset. We did not use the CDFW rotary screw trap data for our assessment because the catch and sampling effort (e.g., volume of water sampled) data from that program were not consistently collected for discrete days prior to 2013 during October–December.

Sacramento trawling

The Sacramento trawl site is located at Sherwood Harbor (Figure 1) and represents a reach length of approximately 6.5 km. At this site, ten 20-min tows were conducted within the reach on each sampling day by using a Kodiak trawl net. All tows were conducted mid-channel

by two boats traveling upstream. The Kodiak trawl net was composed of five panels, each decreasing in mesh size (ranging from 5.1 to 0.6 cm stretch) from the mouth of the net toward a live-box at the cod end. The live-box (36 cm wide × 36 cm tall × 49 cm long) was composed of 0.18-cm-thick aluminum that was perforated with 0.46-cm-diameter holes. A float line and lead line enabled the net to remain at the top few meters of the water column while sampling. The fully extended mouth size of the Kodiak trawl net was 1.96 × 7.62 m, and the net was fished approximately 31 m behind the boats. In general, the Kodiak trawl net was towed at speeds between 0.4 and 0.7 m/s. The measure of the distance traveled during each tow was recorded using a calibrated mechanical flowmeter (General Oceanics Model 2030) deployed alongside one of the boats. We estimated the volume of water sampled for each tow by multiplying the distance traveled by the mean net mouth area during sampling (12.54 m^2; from Barnard et al. 2015).

Sacramento beach seining

Sacramento beach seine sampling was conducted at eight fixed sites within or adjacent to the Sacramento River (Figure 1) using a 15.2- × 1.3-m beach seine net containing 3-mm delta square mesh, a 1.2-m^3 bag in the center of the net, and a continuous lead bottom line and foam floats along the top line attached to 1.8-m-tall wood poles on each side. In general, one beach seine haul was conducted at each site during each sampling day. Beach seines were deployed along the shoreline by two crew members within unobstructed habitats, including boat ramps, mud banks, and sandy beaches. The beach seines were generally deployed starting from the downstream portion of each site to limit disturbance. The beach seine was taken perpendicular from the shoreline to a depth of 1.2 m or an obstacle with a maximum distance of 15 m, was deployed parallel to the shoreline, and was then pulled toward the shoreline. Obstacles were defined as structure that could compromise crew safety or gear efficiency, including steep banks or holes, fast water current, submerged aquatic vegetation, or large woody debris. During sampling, the width and length of the beach seine haul were measured to the nearest 0.5 m using a standard measuring tape. In addition, the maximum depth of the beach seine haul was recorded by measuring the depth to the nearest 0.1 m using the wood poles of the fully deployed beach seine net prior to pulling it to the shore. If the depths of the beach seine varied between measurements, the maximum seine depth was obtained by averaging the two depth measurements. We estimated the volume of water sampled for each beach seine haul by multiplying the haul length, width, and maximum seine depth divided by 2, assuming a constant slope (following Dekar et al. 2013).

Fish processing and identification

All fish collected during each trawl and beach seine haul were identified to species using field keys derived through the compilation of field notes, Miller and Lea (1972), Wang (1986), and Moyle (2002). Prior to release at the site of capture, fish were measured to the nearest 1 mm fork length (FL). The run identity of all unmarked juvenile Chinook Salmon was determined using the LDC method (Fisher 1992; Greene 1992). Fish that could not be accurately identified or properly measured in the field were preserved and brought back to the DJFMP laboratory. Preserved fish were later identified to species or run by more experienced staff. Juvenile Chinook Salmon with missing (clipped) adipose fins were considered marked, and their run was identified by removing and reading the coded wire tag, which provided information about the run, the hatchery of origin, and the date and location of release for each individual.

Environmental data

We obtained readily available data representing water quality and other environmental characteristics that were hypothesized to influence juvenile winter-run salmon occupancy and detection near Sacramento (Table 1; Brandes and McLain 2001; Sommer et al. 2005; del Rosario et al. 2013; Acierto et al. 2014; Hassrick et al. 2022). Water temperature was measured by DJFMP staff to the nearest 0.1°C approximately 0.5 m below the water surface by using a nonmercury thermometer, a YSI Model 85 meter, or a YSI PRO 2030 meter during or immediately before each Sacramento trawl tow and beach seine haul. Water clarity was measured to the nearest centimeter using a Secchi disc at the Sacramento trawl site after each tow. We estimated mean daily water temperature and water clarity by averaging the measurements collected among all trawl tows and seine hauls during the same day. We also determined whether the mean daily temperature during a sampling day was at or below a temperature threshold of 13.5°C, which is used as a fish migration indicator within the Sacramento River (NMFS 2009, 2011, 2019).

In addition to data collected by the DJFMP while sampling, mean daily Sacramento River flow data collected at Wilkins Slough near Grimes, California (U.S. Geological Survey [USGS] gauge 11390500), were obtained from the USGS National Water Information System (USGS 2016).

TABLE 1 Mean, standard deviation (SD), and range (minimum [min], maximum [max]) of environmental data collected with or for juvenile Sacramento River winter-run Chinook Salmon monitoring samples within the Sacramento River during October–December from 1996 to 2019.

Predictor variable	Mean	SD	Min	Max
Sample volume (m^3)	5118.341	5004.952	2.7	315,148.6
Seine method used for sample[a]	0.373	0.484	0	1
Mean daily water temperature (°C)	12.901	3.227	5.029	21.169
Mean daily water temperature below 13.5°C[a]	0.558	0.497	0	1
Mean water clarity (m)	1.042	0.473	0.0832	2.316
Day of year (days from October 1)	47.364	26.574	1	92
Mean daily Sacramento River flow at Wilkins Slough (m^3/s)	218.723	147.588	96.136	914.861
Mean daily Sacramento River flow at Wilkins Slough above 212.4 m^3/s[a]	0.258	0.438	0	1
Pulse flow (percent change in mean daily river flow from the previous 5 days)	0.031	0.249	−0.516	2.271
Pulse flow with a 4-day lag (percent change in mean daily river flow from the previous 5 days with lag)	0.022	0.224	−0.448	2.098
Daily Red Bluff Diversion Dam rotary screw trap catch with flow adjustments (catch per 1233.5 m^3)	1.263	1.323	0	7.629
Sacramento River stage at the Fremont Weir exceeding 10.2 m[a]	0.023	0.149	0	1
7-day precipitation amount (mm)	0.580	0.888	0	5.86
7-day precipitation events (days)	1.738	1.796	0	7
Observed occupancy state during previous sample day[a]	0.283	0.450	0	1
Observed abundant state during previous sample day[a]	0.157	0.364	0	1

[a]Represents a binary indicator variable.

Using these data, we determined whether the mean daily flow during a sampling day exceeded a flow threshold of 212.4 m^3/s, which is used as another fish migration indicator within the Sacramento River (NMFS 2009, 2011, 2019). Further, we calculated the percent change in mean daily flow during a sampling day (henceforth, referred to as "pulse flow") by dividing the mean daily flow by the average of mean daily flows during five consecutive days prior to the sampling day with and without a 4-day lag. To account for juvenile winter-run salmon out-migrating through the Yolo Bypass (the primary floodplain of the Sacramento River) during flooding events instead of out-migrating through the Sacramento River, where fish were monitored (del Rosario et al. 2013), we obtained hourly river stage data from the Sacramento River gauge at Fremont Weir (available from the California Data Exchange Center; California Department of Water Resources 2021) and recorded whether the sample day had a river stage exceeding 10.2 m—the stage at which water crests over the weir and enters the Yolo Bypass (Acierto et al. 2014). Daily precipitation data collected at the Red Bluff Municipal Airport in Red Bluff, California (station USW00024216) were obtained from the National Weather Service's Climate Data Online System. The number of days with a precipitation event (defined as a day with precipitation >0.25 mm) and the total amount of precipitation during the 7 days prior to a sample day were calculated.

We calculated the day of year for each sample day (measured as days from October 1) given that the distribution of out-migrating juvenile Chinook Salmon varies over time (Brandes and McLain 2001; del Rosario et al. 2013). To account for juvenile winter-run salmon beginning their migration from their spawning and rearing habitat below the Shasta Reservoir to the Estuary, we obtained daily rotary screw trap catch-per-unit-effort data (representing juvenile winter-run salmon catch per 1233.5 m^3 of sampling) collected at the Red Bluff Diversion Dam from the USFWS Red Bluff Fish and Wildlife Office (see Johnson et al. 2017). The dam is approximately 263 km upstream of the Sacramento monitoring sites. For each sample day, we determined the maximum daily catch at the Red Bluff Diversion Dam between 30 and 45 days prior to each Sacramento trawl tow and seine haul when there was evidence of a flow pulse or between 50 and 65 days prior to each Sacramento trawl tow and seine haul when there was no evidence of a flow pulse to account for varying travel migration times. We defined a flow pulse as a 30% increase in the mean daily flow relative to the average mean daily flows during five consecutive days prior to the sampling day. The migration travel time ranges between the Red

Bluff Diversion Dam and the Sacramento monitoring sites under the different flow scenarios were approximated using the SacPAS Fish Model version 2.7.4 (Beer et al. 2017); the Delta Survival, Travel Time, and Routing Simulation Model (Perry et al. 2018; Hance et al. 2022); and the findings from Hassrick et al. (2022).

Misidentification data

Notably, the field identification of juvenile Chinook Salmon in the Estuary and its watershed is imperfect (Kirsch et al. 2018). Further, genetic studies have demonstrated that the accuracy in differentiating juvenile winter-run salmon among the other three runs of Chinook Salmon co-occurring in the Estuary by using the LDC method is imperfect and varies in both space and time (Harvey 2011; del Rosario et al. 2013; Pyper et al. 2013; Harvey et al. 2014; Johnson et al. 2017). As a result, we gathered available species and run identity data collected between 2012 and 2019 to determine the probability of falsely identifying an individual juvenile winter-run salmon. We obtained species identification accuracy data from the DJFMP, which represented a total of 199 observations of juvenile Chinook Salmon made by trained full-time DJFMP observers during 13 fish identification exams conducted under controlled conditions at the USFWS Lodi Fish and Wildlife Office (Table 2). The fish identification exams were conducted following the methodology described by Kirsch et al. (2018). Any juvenile Chinook Salmon observation that was made incorrectly during an exam was considered a false-positive species identification.

We obtained juvenile Chinook Salmon run identification accuracy data from the DJFMP and CDFW, which represented a total of 275 juvenile winter-run salmon observations collected at the Sacramento trawl and beach seine sites or the Knights Landing rotary screw trap site during October–December (Table 3). For these observations, juvenile Chinook Salmon were identified in the field as juvenile winter-run salmon by using the LDC method, their tissue was collected in the field via a nonlethal fin clip, and their LDC identification was assessed using genetic run assignments. To determine the genetic run assignment and generate an estimated genetic run assignment probability (i.e., probability of correct genetic assignment) for each of the juvenile Chinook Salmon, individual genotypes were compared with published baselines (Clemento et al. 2014; Meek et al. 2016) using the ONCOR program (Kalinowski et al. 2008). A 0.80 assignment probability was used as the minimum cutoff to assign individuals to a Chinook Salmon run (Daly et al. 2012; Clemento et al. 2014; Meek et al. 2016). For 22 of the observations (M. Meek and S. Hugentobler, unpublished data), the collected tissues were genotyped using single-nucleotide polymorphism genetic loci following Meek et al. (2016), and the genotypes were assigned to run using the ONCOR program and the reference baseline from Meek et al. (2016). For the other observations ($n = 253$), the collected tissues were genotyped using single-nucleotide polymorphism genetic loci following Clemento et al. (2011, 2014), and the genotypes were assigned to run using the ONCOR program and the reference baseline from Clemento et al. (2014). Any juvenile winter-run salmon observation made in the field using the LDC method that did not have a matching genetic run assignment was considered a false-positive run identification.

We calculated the probability of misidentifying an individual juvenile winter-run salmon (M) as

$$M = 1 - \left\{\left[1 - (\mathrm{SM}/\mathrm{Ns})\right]\left[1 - (\mathrm{RM}/\mathrm{Ng})\right]\right\},$$

where SM is the number of false-positive species identifications recorded during the fish identification exams, Ns is the total number of juvenile Chinook Salmon identifications made during the fish identification exams, RM is the number of false-positive run identifications made using the LDC method in the field, and Ng is the total number of juvenile

TABLE 2 False-positive species error probabilities using juvenile Chinook Salmon identification accuracy data collected by the U.S. Fish and Wildlife Service's Delta Juvenile Fish Monitoring Program (DJFMP) during observer identification exams from 2012 to 2019.

Data set (relevant citation)	Time period (years)	Number of fish identification exams	Number of Chinook Salmon identifications	False-positive species error count	False-positive species error probability
DJFMP Fish Identification Program (Kirsch et al. 2018)	2012–2014	5	83	2	0.025
DJFMP Fish Identification Program (A. Goodman, unpublished data)	2017–2019	8	116	4	0.034
All	All	13	199	6	0.030

TABLE 3 False-positive run identification error probabilities for juvenile Sacramento River winter-run Chinook Salmon based on genetic run assignment and fork length at date of capture river criteria identification data collected by the U.S. Fish and Wildlife Service's Delta Juvenile Fish Monitoring Program (DJFMP) and the California Department of Fish and Wildlife (CDFW) at the Sacramento trawl and beach seine sites and the Knights Landing rotary screw trap site during October–December from 2012 to 2019.

Data set (relevant citation)	Time period (years)	Observation location(s)	Number of juvenile winter-run identifications	False-positive run identification error count	False-positive run identification error probability
DJFMP and Michigan State University Research (M. Meek and S. Hugentobler, unpublished data)	2012–2019	DJFMP Sacramento trawl and beach seine sites	22	0	0.000
DJFMP Central Valley Salmonid Coordinated Genetic Monitoring (J. A. Israel and R. W. Perry, unpublished data)	2018–2019	DJFMP Sacramento trawl site	15	1	0.067
CDFW Central Valley Salmonid Coordinated Genetic Monitoring (Israel and J. Julienne, unpublished data)	2017–2019	CDFW Knights Landing rotary screw trap site	238	6	0.025
All	2012–2019	All	275	7	0.025

winter-run salmon identifications made using the LDC method in the field.

Occupancy modeling

We used a modification of multistate occupancy models (MacKenzie et al. 2009) to assess the effect of environmental characteristics on daily juvenile winter-run salmon occupancy at Sacramento trawl and beach seine sites using data from 1996 to 2019. The modified multistate occupancy models provided a framework to relate occupancy at three states (absent, occupied, or occupied and abundant) to environmental characteristics while accounting for spatial dependency, false-negative (imperfect detection) error, and false-positive (misidentification) error. The primary assumption of the occupancy estimator given our data is that the juvenile winter-run salmon occupancy state among the sampling sites cannot change between individual Sacramento trawl tows and beach seine hauls conducted within a day (MacKenzie et al. 2002, 2009). Our multistate occupancy models estimated the following parameters:

$\psi^1_{i,k}$ = occupancy probability for juvenile winter-run salmon during a sample day among all monitoring sites; and
$\psi^2_{i,k}$ = probability that juvenile winter-run salmon were abundant during a sample day given that at least one monitoring site was occupied that day,

where i indexes the sample days and k indexes the sample year (defined as October 1–December 31). The probability that at least one monitoring site was occupied by an abundant number of juvenile winter-run salmon is $\psi^1 \times \psi^2$. We defined the abundant state as two or more individuals in a single trawl tow or seine haul because the sample days when this abundant state was observed were highly correlated ($r^2 = 0.92$) with the days when the two monitoring efforts

independently met the salmon decision process criteria (i.e., detected three or more winter-run-sized [or larger] juvenile salmon during a day after the catch count data were standardized to 1 day of typical sampling effort).

To account for annual dependency within the data from, in part, interannual variation in juvenile winter-run salmon production (Johnson et al. 2017), we estimated the occupancy states as logit-linear functions of covariates, including random intercepts (Royle and Dorazio 2008) as

$$\text{Logit}\left(\psi^1_{i,k}\right) = \alpha_0 + \alpha_{0,k} + \alpha_1 W_{1,i} + \ \ldots \ + \alpha_r W_{r,i},$$

$$\text{Logit}\left(\psi^2_{i,k} \,|\, \text{true state} = \text{occupied}\right) = \gamma_0 + \gamma_{0,k} + \gamma_1 X_{1,i} + \ \ldots \ + \gamma_r X_{r,i},$$

where W represents an occupancy state predictor variable, α_0 and γ_0 represent fixed intercepts, $\alpha_{0,k}$ and $\gamma_{0,k}$ represent randomly varying intercepts that varied among sample years, α represents the effect of W on the occupancy state, X represents an abundant state predictor variable, and γ represents the effect of X on the abundant state. Our first modeling objective was to estimate the occupancy of juvenile winter-run salmon during sample days using environmental covariates that were hypothesized to influence occupancy near Sacramento during October–December. Specifically, we modeled ψ^1 using the following predictor variables: amount of 7-day precipitation and number of 7-day precipitation events in Red Bluff, mean daily water temperature and clarity at the sampling sites, mean daily Sacramento River flow at Wilkins Slough, pulse flow, upstream daily rotary screw trap catch at the Red Bluff Diversion Dam, day of year, and a binary indicator representing whether the occupancy state (ψ^1) was observed during the previous sample day (unobserved = 0; observed = 1). In addition, we incorporated several interaction covariates on ψ^1, including interactions between precipitation amounts and number of events, water clarity and pulse flow, and water clarity and temperature. Within occupied sites, we hypothesized that juvenile winter-run salmon would be abundant (ψ^2) if (1) there was low water clarity, (2) particular flow and temperature thresholds were reached or exceeded, (3) Sacramento River water was not being diverted into the Yolo Bypass floodplain, and (4) mean daily flow was declining 4 days prior to the sampling day. Thus, we modeled ψ^2 using the following predictor variables: water clarity, pulse flow with a 4-day lag, mean daily Sacramento River flow above 212.4 m^3/s (coded as a binary indicator: flow at or below 212.4 m^3/s = 0; flow above 212.4 m^3/s = 1), mean daily temperature below 13.5°C (coded as a binary indicator: temperature at or above 13.5°C = 0; temperature below 13.5°C = 1), Sacramento River stage at the Fremont Weir exceeding 10.2 m (coded as a binary indicator: river stage at or below 10.2 m = 0; river stage above 10.2 m = 1), day of year, day of year^2 (to allow for nonlinearity in the effect of day of year), and the abundant state (ψ^2) being observed during the previous sample day (coded as a binary indicator: unobserved = 0; observed = 1). In addition, we included several interaction covariates on ψ^2, including interactions between (1) the flow threshold being met or exceeded and the water temperature threshold being met or exceeded, (2) water clarity and pulse flow with a 4-day lag, (3) water clarity and the temperature threshold being met or exceeded, and (4) pulse flow with a 4-day lag and Sacramento River stage at the Fremont Weir exceeding 10.2 m.

Our objective was to assess the effects of environmental characteristics on daily juvenile winter-run Chinook Salmon occupancy and to determine whether those effects were influenced by false-negative (imperfect detection) error and false-positive (misidentification) error. Therefore, we developed a total of three multistate occupancy models, including (1) a naïve model that did not account for imperfect detection or misidentification, (2) a detection model that accounted for imperfect detection but did not account for misidentification, and (3) a detection and misidentification model that accounted for both imperfect detection and misidentification. In general, we used the naïve model to represent our best understanding of the factors influencing juvenile winter-run salmon occupancy data that were used to inform past DCC gate closure decisions.

In addition to estimating the occupancy states ($\psi^1_{i,k}$ and $\psi^2_{i,k}$), the models that accounted for imperfect detection estimated the following parameters:

$p^1_{i,j,h}$ = probability that a juvenile winter-run salmon was detected during a sample day given that the true occupancy state was present but not abundant;
$p^2_{i,j,h}$ = probability that a juvenile winter-run salmon was detected during a sample day given that the true occupancy state was present and abundant; and
$\delta_{i,j,h}$ = probability that evidence of the abundant state was detected during a sample day given that the true occupancy state was present and abundant and the species was detected,

where h indexes the sites and j indexes the replicate samples (individual trawl tows and seine hauls) collected during sample day i. The probability of detecting the abundant state is $p^2 \times \delta$. The conditional detection probabilities used in the detection model were derived following Nichols et al. (2007) and are presented in Table 4.

To account for misidentification error in the model, we incorporated the probability of misclassifying the observed occupancy states into the conditional detection probabilities following Shea et al. (2013). We calculated the probability of misclassifying the occupancy states

TABLE 4 Conditional detection probability matrix for the true occupancy states given the observed state for the naïve model, the detection model, and the detection and misidentification model, where p^1 is the probability that a juvenile Sacramento River winter-run Chinook Salmon is detected in a sample unit given that the true occupancy state is present but not abundant; p^2 is the probability that a juvenile winter-run Chinook Salmon is detected in a sample unit given that the true occupancy state is present and abundant; δ is the probability that evidence of the abundant state is detected in a sample unit given that the true occupancy state is present and abundant; M is the probability of misidentifying an individual juvenile winter-run Chinook Salmon; and Nf is the number of individuals collected in a replicate sample that could be misidentified as juvenile winter-run Chinook Salmon.

Model	Observed occupancy	True occupancy state		
		Absent	**Present**	**Present and abundant**
Naïve	Absent	1	0	0
	Present	0	1	0
	Present and abundant	0	0	1
Detection	Absent	1	$(1-p^1)$	$(1-p^2)$
	Present	0	p^1	$p^2(1-\delta)$
	Present and abundant	0	0	$p^2\delta$
Detection and misidentification	Absent	$(1-M)^{Nf}$	$(1-p^1)(1-M)^{Nf}$	$(1-p^2)(1-M)^{Nf}$
	Present	$M\left[(1-M)^{Nf-1}\right]$	$p^1(1-M)^{Nf}+(1-p^1)\left[M(1-M)^{Nf-1}\right]$	$p^2(1-\delta)(1-M)^{Nf}+(1-p^2)\left[M(1-M)^{Nf-1}\right]$
	Present and abundant	$1-\left[(1-M)^{Nf}+M(1-M)^{Nf-1}\right]$	$p^1\left[1-(1-M)^{Nf}\right]+(1-p^1)\left(1-\left\{\left[(1-M)^{Nf}\right]+M(1-M)^{Nf-1}\right\}\right)$	$p^2\delta+p^2(1-\delta)\left[1-(1-M)^{Nf}\right]+(1-p^2)\left[1-\left((1-M)^{Nf}+\left\{M\left[(1-M)^{Nf-1}\right]\right\}\right)\right]$

using the probability of misidentifying an individual juvenile winter-run salmon (M) as a function of the number of individuals collected in a replicate sample that were identified as juvenile winter-run salmon in the field (Nf). Because the observed occupancy states can be the result of more than one misclassification and detection scenario, the conditional detection probability for each true and observed occupancy state combination represented whether (1) the observed occupancy state was detected or not and (2) individuals (none, one, or at least two) were misidentified (Table 4). For example, assuming that the true occupancy state is present, the present occupancy state can be observed by either (1) detecting the present state and not misidentifying any individuals or (2) not detecting the present state and misidentifying one individual (Table 4).

There was likely spatial dependency among the replicate samples because the Sacramento trawl tows and beach seine hauls were conducted at fixed sites over time. As a result, we estimated each of the detection probabilities as logit-linear functions of covariates including random intercepts as

$$\eta_{i,j,h} = \beta_0 + \beta_{0,h} + \beta_1 Y_{1,i,j} + \ \ldots \ + \beta_r Y_{r,i,j},$$

where η is the log-odds of the response (p^1, p^2, and δ), Y represents a predictor variable, β_0 represents a fixed intercept, $\beta_{0,h}$ represents a randomly varying intercept that varied among sites, and β represents the effect of Y on the response variable. Fish detection varies based on sampling method and environmental conditions (Perry et al. 2016; Peterson and Barajas 2018; Duarte and Peterson 2021; Mahardja et al. 2021). Thus, we modeled p^1, p^2, and δ using the following predictor variables: sample volume, daily mean water clarity, daily mean Sacramento River flow at Wilkins Slough, beach seine sample method (coded as a binary indicator variable, with trawl as the baseline category), and interactions between the beach seine sample method and flow and sample volume. We also modeled p^2 as equal to p^1 but included an abundant state covariate (coded as a binary indicator variable: unobserved = 0; observed = 1) to account for the effect of a great number of individuals on detection (Bayley and Peterson 2001; Royle and Nichols 2003).

Prior to model fitting, we estimated Pearson's correlation coefficients between all pairs of fixed-effect covariates to ensure that the occupancy models would not be affected by multicollinearity (Dormann et al. 2013). No covariates used were considered strongly correlated ($|r^2| > 0.70$; Dormann et al. 2013). In addition, all continuous covariate data were standardized with a mean of zero and a standard deviation of 1 to facilitate model fitting. Thus, the baseline condition used for the modeling represented average continuous covariate values (see Table 1) and the binary indicator covariate condition not being observed (e.g., river flow at or below 212.4 m^3/s; water temperature at or above 13.5°C). To accommodate model structures and to ensure that the modeling results were comparable, all three occupancy models were fitted using Markov chain–Monte Carlo in JAGS (Plummer 2003), implemented in the R package jagsUI (Kellner 2015). Models were fitted using three chains running 250,000 iterations with a burn-in of 50,000 iterations. We assessed and ensured the convergence of each model using the Gelman and Rubin diagnostic test (convergence diagnostic $\hat{R}$; Gelman and Rubin 1992). Convergence was assumed when $\hat{R}$ was <1.05 (Gelman and Rubin 1992). Noninformative (i.e., diffuse) priors were used for each parameter (Gelman et al. 2008; Kéry and Royle 2008). Random effects were assumed to be normally distributed with a mean of zero and a random-effect-specific variance (Royle and Dorazio 2008). The annual dependency within the occupancy data and the spatial dependency within the detection data were confirmed by assessing residual plots of the models that did and did not contain the random-effect intercepts.

We determined the effect of imperfect detection and misidentification by comparing the parameter estimates for the two occupancy states among the naïve model, the detection model, and the detection and misidentification model. The precision of fixed-effect parameter estimates was assessed by calculating 95% credible intervals (Congdon 2001). Credible intervals that contained zero either represented a weak effect or were too imprecise to draw conclusions about the strength of the relationship. Odds ratios (ORs) were used to interpret relationships for each fixed-effect parameter estimate (Hosmer and Lemeshow 2000). To evaluate the extent to which imperfect detection and misidentification error can affect the initiation of the interagency salmon decision process that is used to make DCC gate closure decisions, we used the three occupancy models and estimated the probability that at least one site was occupied by an abundant number of juvenile winter-run salmon during sample days based on the environmental data. Thereafter, we arbitrarily determined the number of sampling days in each month and year for which the abundant occupancy estimate's upper 95% credible interval exceeded a probability of 0.90.

RESULTS

In total, 14,147 Sacramento trawl tows and beach seine hauls were conducted during 1058 sampling days between October and December from 1996 to 2019. Sacramento trawl tows represented approximately 63% ($n = 8865$) of the samples collected. A total of 689 sampling days

had both Sacramento trawl tows and beach seine hauls conducted. During the sampling, 2803 juvenile winter-run salmon were collected and identified using the LDC method. The FL of these fish averaged 68 mm (ranging from 35 to 109 mm). Approximately 29% ($n = 308$) of the sampling days had at least one juvenile winter-run salmon detected, and 173 sampling days had the abundant occupancy state of juvenile winter-run salmon detected in at least one trawl tow or beach seine haul. Of the 135 sample days on which only the occupancy state was detected, approximately 78% of those days ($n = 106$) had only one replicate sample detect the occupancy state. In general, the sampling days represented a wide range of environmental conditions, including river flow, water temperature, and clarity (Table 1).

Misidentification

A total of six individuals were falsely identified as a juvenile Chinook Salmon among the 199 observations of juvenile Chinook Salmon made by trained full-time DJFMP observers during controlled exams (Table 2). As a result, we calculated the false-positive species error probability at 0.03. Similarly, the false-positive run identification error probability for identifying juvenile winter-run salmon using the LDC method was calculated as approximately 0.03 based on seven juvenile winter-run salmon being misidentified among the 275 observations assessed for accuracy using genetic run assignments (Table 3). Using these false-positive error probabilities, we calculated the probability of misidentifying an individual juvenile winter-run salmon in the field as 0.056, and this probability was used in the conditional detection probability matrix for the detection and misidentification occupancy model (Table 4).

Detection

The detection of the juvenile winter-run salmon occupancy states was estimated by both the detection model and the detection and misidentification model (Table 4). Both models indicated that the probability of detecting the occupancy states was below 1.0, and the detection estimates for the occupancy states varied considerably between the models (Table 5). For example, the average detection probability of the present occupancy state was <0.01 and 0.26 while conducting an individual trawl tow at the Sacramento trawl site under baseline conditions based on the detection model and the detection and misidentification model, respectively. Similarly, the average detection probability of the abundant occupancy state at the occupied Sacramento trawl site was 0.20 and 0.39 while conducting an individual trawl tow under baseline conditions based on the detection model and the detection and misidentification model, respectively. The probability of detecting the present occupancy state was significantly negatively related to clearer water and higher Sacramento River flow in both models. Water clarity had the largest effect size. The ORs suggested that detection of the present occupancy state was, on average, 3.58 and 2.03 times less likely for every 0.47-m increase in water clarity based on the detection model and the detection and misidentification model, respectively. The probability of detecting the abundant occupancy state at an occupied site was only related to water clarity in the detection model (Table 5). The ORs suggested that the detection of the abundant occupancy state was, on average, 1.94 times less likely for every 0.47-m increase in water clarity based on the detection model. The random-effect estimates from both models for each occupancy state indicated that considerable variation remained among sampling sites after accounting for surveying method (trawl versus seine), sample volume, and environmental characteristics (Table 5), thus suggesting considerable spatial dependency in the count data.

Occupancy

The occupancy states of juvenile winter-run salmon were estimated by all three occupancy models using the same predictor variables, and the estimates for the occupancy states varied considerably between the models (Table 5). For example, the average present occupancy state probability was 0.22, 0.90, and <0.01 under baseline conditions based on the naïve model, the detection model, and the detection and misidentification model, respectively. The average probability of the abundant occupancy state was 0.06, 0.41, and <0.01 under baseline conditions based on the naïve model, the detection model, and the detection and misidentification model, respectively. Using observed environmental data, the estimated number of sampling days that had a site occupied by an abundant number of juvenile winter-run salmon was lowest using the naïve model and highest using the detection model (Table 6).

The number of significant fixed-effect predictor variables for estimating the present and abundant occupancy states varied among the occupancy models, with the naïve model having the highest number of significant predictor variables ($n = 13$) and the detection and misidentification model having the lowest ($n = 5$; Table 5). Sacramento River mean flow and pulse flow were the only significant covariates for predicting the present occupancy state among all three models (Table 5; Figure 2). For every 148-m^3/s increase in mean daily Sacramento River flow, the

TABLE 5 Parameter estimates, 95% confidence intervals (CIs), and fixed-effect odds ratios (ORs) based on the three models (naïve model, detection model, and detection and misidentification model) used to predict the occupancy of juvenile Sacramento River winter-run Chinook Salmon at nine sites near Sacramento, California, that were monitored by the U.S. Fish and Wildlife Service's Delta Juvenile Fish Monitoring Program to inform real-time Delta Cross Channel gate closure decisions during October–December from 1996 to 2019.

	Naïve model				Detection model				Detection and misidentification model			
Submodel (in bold) and parameter	**Logit-scale estimate**	**95% CI (lower)**	**95% CI (upper)**	**OR**	**Logit-scale estimate**	**95% CI (lower)**	**95% CI (upper)**	**OR**	**Logit-scale estimate**	**95% CI (lower)**	**95% CI (upper)**	**OR**
Occupancy state (ψ^1)												
Intercept	−1.314	−1.772	−0.860	0.269	2.248	0.478	4.388	9.469	−6.457	−8.180	−4.752	0.002
Precipitation amount	0.220	−0.147	0.590	1.246	0.699	−1.226	2.584	2.012	0.496	−0.342	1.360	1.642
Precipitation events	0.479	0.194	0.765	1.614	0.795	−0.518	2.731	2.214	0.112	−0.686	0.927	1.119
Precipitation amount × precipitation events	−0.198	−0.396	0.006	0.820	0.437	−0.865	2.455	1.548	−0.130	−0.650	0.390	0.878
Temperature	0.178	−0.334	0.695	1.195	0.258	−1.689	2.146	1.294	1.261	−0.481	3.044	3.529
Flow	0.485	0.154	0.832	1.624	2.822	1.085	4.972	16.810	1.106	0.158	2.274	3.022
Change in flow	−0.734	−1.167	−0.330	0.480	−1.765	−3.180	−0.098	0.171	−1.153	−2.481	−0.010	0.316
Water clarity	−0.714	−1.069	−0.369	0.490	0.350	−1.363	2.609	1.419	−3.474	−5.046	−1.978	0.031
Water clarity × pulse flow	−0.560	−0.919	−0.227	0.571	−0.976	−2.052	−0.013	0.377	−0.622	−1.531	0.175	0.537
Water clarity × temperature	−0.373	−0.678	−0.077	0.689	−0.972	−2.393	0.738	0.378	−1.132	−2.595	0.234	0.322
Red Bluff rotary screw trap catch	0.138	−0.078	0.352	1.148	0.365	−1.175	2.586	1.441	0.707	0.100	1.352	2.028
Day of year	0.438	−0.108	0.993	1.550	0.587	−1.469	2.614	1.799	1.079	−0.458	2.710	2.942
Occupancy state on previous sample day	0.905	0.500	1.305	2.472	1.534	−0.193	3.727	4.637	0.657	−0.594	1.905	1.929
Random effect intercept: year	0.650				9.066				8.328			
Abundant state (ψ^2 | true state = occupied)												
Intercept	−0.965	−1.846	−0.125	0.381	−0.187	−1.891	1.656	0.829	1.618	−1.281	4.531	5.043
Temperature below 13.5°C	0.588	−0.469	1.656	1.800	−0.481	−1.923	0.937	0.618	1.009	−2.008	4.049	2.743
Flow above 7500 m^3/s	−0.163	−1.541	1.184	0.850	1.373	−0.428	3.297	3.947	0.793	−2.213	3.827	2.210
Flow above 7500 m^3/s × temperature below 13.5°C	0.840	−0.686	2.392	2.316	0.900	−1.103	2.832	2.460	0.634	−2.431	3.723	1.885
Pulse flow with 4-day lag	−0.682	−1.327	−0.070	0.506	−0.313	−1.388	1.203	0.731	0.537	−1.889	3.115	1.711
Water clarity	−0.988	−2.024	−0.043	0.372	0.461	−2.102	2.725	1.586	−1.514	−4.462	1.372	0.220
Water clarity × pulse flow with 4-day lag	−0.367	−1.003	0.242	0.693	−0.921	−2.015	0.406	0.398	−0.598	−2.949	1.608	0.550
Water clarity × temperature below 13.5°C	0.401	−0.637	1.502	1.493	−0.608	−2.554	1.328	0.544	−1.256	−4.203	1.659	0.285

(Continues)

TABLE 5 (Continued)

Submodel (in bold) and parameter	Naïve model				Detection model				Detection and misidentification model			
	Logit-scale estimate	95% CI (lower)	95% CI (upper)	OR	Logit-scale estimate	95% CI (lower)	95% CI (upper)	OR	Logit-scale estimate	95% CI (lower)	95% CI (upper)	OR
Day of year	0.213	−0.530	0.999	1.237	0.798	−0.434	2.170	2.221	0.816	−2.183	3.834	2.261
Day of year × day of year	−0.557	−1.059	−0.088	0.573	−1.128	−1.940	−0.274	0.324	1.083	−1.694	3.992	2.954
Abundant state on previous sample day	0.799	0.161	1.435	2.223	1.084	0.086	2.095	2.956	0.840	−2.045	3.770	2.316
River overtopping Fremont Weir	−0.153	−1.506	1.269	0.858	0.905	−1.247	3.460	2.472	0.088	−3.081	3.273	1.092
River overtopping Fremont Weir × pulse flow with 4-day lag	0.188	−0.748	1.368	1.207	0.165	−1.666	2.503	1.179	0.221	−2.660	3.260	1.247
Random effect intercept: year	0.318				6.361				8.214			
Detection of occupancy state (p^1 | true state = present, p^2 | true state = abundant)												
Intercept					−3.632	−5.470	−1.465	0.026	−0.610	−3.067	2.038	0.543
Abundant state observed					2.194	1.830	2.595	8.971	−0.614	−3.113	1.949	0.541
Seine method					0.049	−2.614	2.650	1.050	0.299	−2.184	2.729	1.349
Volume					0.215	−0.118	0.653	1.240	0.271	−0.277	0.803	1.311
Seine method × volume					0.145	−2.434	2.588	1.156	−0.164	−2.742	2.447	0.849
Water clarity					−1.275	−1.483	−1.062	0.279	−0.707	−1.209	−0.189	0.493
Flow					−0.137	−0.246	−0.029	0.872	−0.275	−0.457	−0.090	0.760
Seine method × flow					0.085	−0.035	0.206	1.089	0.267	0.052	0.490	1.306
Random effect intercept: site					2.396				1.754			
Correct classification of abundant state (δ | true state = abundant and species detected)												
Intercept					−1.113	−2.327	0.378	0.329	−0.357	−1.803	1.205	0.700
Seine method					0.628	−1.875	3.070	1.874	0.271	−2.252	2.747	1.311
Volume					0.162	−0.480	0.814	1.176	−0.404	−1.228	0.388	0.668
Seine method × volume					−0.398	−2.825	2.007	0.672	−0.249	−2.676	2.199	0.780
Water clarity					−0.663	−0.972	−0.355	0.515	0.183	−0.561	0.968	1.201
Flow					−0.159	−0.341	0.020	0.853	0.092	−0.187	0.388	1.096
Seine method × flow					0.115	−0.102	0.332	1.122	−0.082	−0.434	0.265	0.921
Random effect intercept: site					0.534				0.598			

TABLE 6 The number of sample days by month and year when abundant occupancy ($\psi^1 \times \psi^2$) of juvenile Sacramento River winter-run Chinook Salmon exceeded a 0.90 probability using the upper 95% credible limit parameter estimates in the naïve model, the detection model, and the detection and misidentification model.

		Number of days the abundant state was predicted		
Year	**Month**	**Naïve model**	**Detection model**	**Detection and misidentification model**
1996	Oct	0	0	0
	Nov	1	11	4
	Dec	15	16	15
1997	Oct	0	0	0
	Nov	5	6	10
	Dec	13	18	13
1998	Oct	0	0	0
	Nov	1	1	1
	Dec	12	18	11
1999	Oct	0	0	0
	Nov	0	6	0
	Dec	0	23	0
2002	Oct	0	12	0
	Nov	0	12	3
	Dec	6	14	7
2003	Oct	0	12	0
	Nov	0	13	3
	Dec	15	19	18
2004	Oct	0	7	1
	Nov	0	13	4
	Dec	8	16	8
2005	Oct	0	16	0
	Nov	4	13	7
	Dec	11	12	11
2006	Oct	0	13	0
	Nov	0	13	5
	Dec	7	13	7
2007	Oct	0	6	0
	Nov	0	8	0
	Dec	0	3	2
2008	Oct	0	9	0
	Nov	0	7	2
	Dec	0	10	0
2009	Oct	0	13	0
	Nov	0	13	0
	Dec	2	13	1
2010	Oct	0	13	1
	Nov	0	20	0
	Dec	12	14	12

(Continues)

TABLE 6 (Continued)

Year	Month	Number of days the abundant state was predicted		
		Naïve model	Detection model	Detection and misidentification model
2011	Oct	0	15	0
	Nov	0	15	1
	Dec	0	1	0
2012	Oct	0	16	0
	Nov	2	15	5
	Dec	16	17	17
2013	Oct	0	6	0
	Nov	0	13	0
	Dec	0	4	0
2014	Oct	0	16	0
	Nov	1	13	0
	Dec	11	15	11
2015	Oct	0	11	0
	Nov	0	13	0
	Dec	2	18	3
2016	Oct	0	14	0
	Nov	4	14	3
	Dec	11	14	6
2017	Oct	0	8	0
	Nov	0	15	0
	Dec	0	29	0
2018	Oct	0	16	0
	Nov	1	13	2
	Dec	13	24	13
2019	Oct	0	15	0
	Nov	0	13	0
	Dec	14	24	18
All	Oct	0	218	2
	Nov	19	260	50
	Dec	168	335	225

present occupancy state was 1.6, 16.8, and 3.0 times more likely according to the naïve model, the detection model, and the detection and misidentification model, respectively (Figure 2). The present occupancy state was 2.1, 5.8, and 3.2 times more likely for every 25% decrease in mean daily river flow relative to the previous 5 days based on the naïve model, the detection model, and the detection and misidentification model, respectively (Figure 2). However, decreases in water clarity reduced these pulse flow effects according to the interaction parameters in the naïve model and the detection model (Table 5; Figure 2). For every 0.47-m increase in water clarity, the present occupancy was 2.0 times less likely based on only the naïve model and 32.3 times less likely based on the detection and misidentification model. Finally, the present occupancy state was positively related to upstream daily rotary screw trap catch at the Red Bluff Diversion Dam for only the detection and misidentification model, which indicated that the present occupancy state was 2.0 times more likely for every 1.3-unit increase in juvenile salmon catch per 1233.5 m^3 of sampling at the Red Bluff Diversion Dam (Table 5).

The abundant occupancy state at an occupied site was positively related to lower water clarity and a decrease in mean daily river flow relative to the previous 5 days based only on the detection model (Table 5). Both the naïve model and the detection model demonstrated that the abundant occupancy state was between 2.2 and 3.0 times more likely to occur when the abundant occupancy state

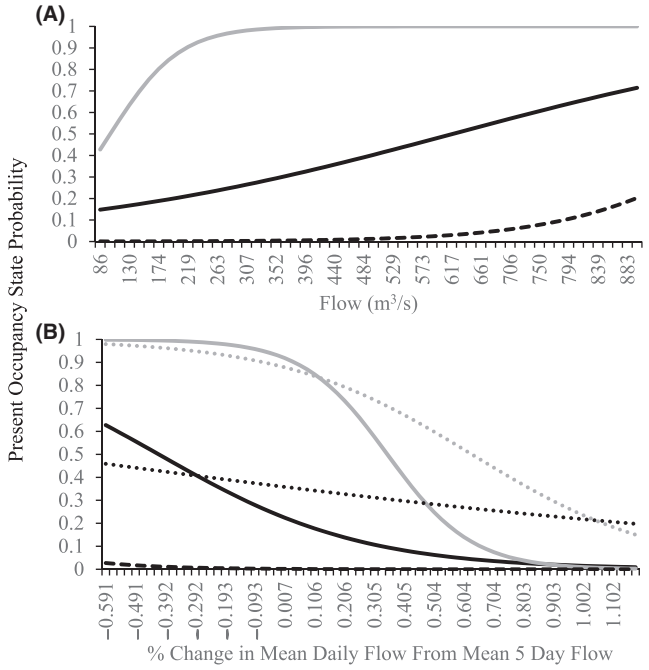

FIGURE 2 Estimated effect of (A) mean daily Sacramento River flow at Wilkins Slough and (B) change in mean daily Sacramento River flow (pulse flow) at Wilkins Slough relative to the previous 5 days on the probability that a Sacramento monitoring site was occupied by juvenile Sacramento River winter-run Chinook Salmon during a sampling day under baseline conditions using the naïve model (solid black line), the detection model (solid gray line), or the detection and misidentification model (dashed black line). Dotted lines in panel B represent the environmental condition in which water clarity decreased by 0.47 m and other environmental variables remained under the baseline condition using the naïve model (dotted black line) or the detection model (dotted gray line).

was observed during the previous sampling day. However, the detection and misidentification model did not have any significant fixed effects for predicting the abundant occupancy state at an occupied site (Table 5). The year random-effect estimates from the models for each occupancy state indicated that considerable variation remained among years after accounting for environmental characteristics, and the amount of variation was highest within the detection model and the detection and misidentification model (Table 5).

DISCUSSION

The protection of early life stage individuals is critical to conserving the Sacramento River winter-run Chinook Salmon (NMFS 2014; Crozier et al. 2021; Meyers 2021). The effectiveness of water management actions that are intended to protect juvenile winter-run salmon is dependent on an unbiased and precise understanding of the distribution of individuals and which factors influence their distribution over space and time (Peterson and Duarte 2020; Wohner et al. 2022). Despite the existing Sacramento River winter-run Chinook Salmon monitoring network, which is aimed at directly or indirectly informing water operation and fish management decisions, there is uncertainty in the fish count data that inform these decisions (Johnson et al. 2017; Wohner et al. 2022). Here, we developed hierarchical multistate occupancy models (following Nichols et al. 2007 and Shea et al. 2013) and assessed the effect of environmental factors, imperfect and variable detection, and misidentification on juvenile winter-run salmon count data collected at monitoring sites from October through December to inform real-time DCC gate operation decisions. Overall, we found that river flow characteristics and water clarity were the most important environmental factors affecting juvenile winter-run salmon occupancy. Further, our modeling approach demonstrated that detection and species misidentification errors influenced our understanding of the relationships between environmental factors and juvenile winter-run salmon occupancy and could affect how DCC gate operation decisions would be informed using the juvenile winter-run salmon count data.

Previous research has demonstrated that occupancy estimates can be substantially biased when false-negative and false-positive error rates exceed 20% and 5%, respectively (Tyre et al. 2003; Royle and Link 2006). As expected, we found that juvenile winter-run salmon identification accuracy in the field using the LDC method was imperfect, with a false-positive error rate of 5.6%. In general, juvenile Chinook Salmon are fairly distinct morphologically and are a species of conservation concern (Moyle 2002); thus, it is understandable that trained full-time observers rarely misidentified another fish species as a juvenile Chinook Salmon (Kirsch et al. 2018). Although we anticipated that the false-positive run identification error probability using the LDC method would be higher than 3%, this result is consistent with previous genetic validation studies (del Rosario et al. 2013; Pyper et al. 2013; Harvey et al. 2014). The primary assumption of the LDC method is that the FL distributions of juvenile Chinook Salmon in the Central Valley do not overlap among the different runs during a day based on runs spawning at different times and juveniles having identical growth rates (Fisher 1992). Juvenile winter-run salmon identification accuracy using the LDC method likely decreases over time during the juvenile out-migration period because of the known inter- and intra-annual variability in spawn timing and growth rates (Yoshiyama et al. 1998; Williams 2006; del Rosario et al. 2013; Harvey et al. 2014). This could result in false-positive run identification error probabilities as high as 33–86% (Pyper et al. 2013; Harvey et al. 2014;

Johnson et al. 2017). Our relatively low false-positive run identification error probability was likely the result of assessing accuracy during the beginning of the juvenile outmigration period (October–December), when there would likely be few juveniles from other runs at a similar size (Williams 2006; del Rosario et al. 2013). Notably, this potential temporal trend in false-positive errors could be incorporated into our modeling framework in future efforts.

We also found that the detection of juvenile winter-run salmon was imperfect. Our detection probability submodels estimated that false-negative error was, on average, >70% during an individual trawl tow under the baseline condition. Assuming that 10 trawl tows were conducted during a sample day under baseline conditions, the false-negative error rates for detecting the occupancy state would be 95% and 5% under the baseline condition based on the detection model and the detection and misidentification model, respectively. The lower false-negative error rate estimated by the detection and misidentification model was likely based on the model considering some of the field observations of juvenile winter-run salmon as false identifications. Since the vast majority (78%) of the sampling days had the occupancy state observed in just one replicate sample, we surmise that the detection and misidentification model estimated much fewer samples as truly occupied (i.e., it treated many of these single observations as false-positive detections), which would result in a higher detection rate, particularly under the baseline condition. Further, we found that the detection of juvenile winter-run salmon occupancy was spatially dependent among sampling locations and negatively related to clearer water and higher mean river flow, which is consistent with other fish studies conducted in the Estuary (Perry et al. 2016; Peterson and Barajas 2018; Duarte and Peterson 2021; Mahardja et al. 2021). Therefore, our study demonstrates that the false-negative and false-positive error rates in the juvenile winter-run salmon count data used to inform DCC gate operation decisions are capable of biasing occupancy estimates and thereby distorting our understanding of the factors influencing juvenile winter-run salmon distribution.

Numerous environmental factors have been hypothesized to influence the distribution of juvenile winter-run salmon in the Estuary. Previous studies have demonstrated relationships between juvenile salmon occupancy or abundance and flow characteristics, water clarity and temperature, weather, upstream juvenile production and distribution, and time (Brandes and McLain 2001; Sommer et al. 2005; del Rosario et al. 2013; Acierto et al. 2014; Perry et al. 2016; Hassrick et al. 2022). However, few studies (e.g., del Rosario et al. 2013; Hassrick et al. 2022) have directly evaluated the influence of these factors on juvenile winter-run salmon occupancy, and no studies have accounted for sampling uncertainties, including imperfect detection and misidentification during their evaluation. In general, we found that when imperfect detection and species misidentification were unaccounted for, juvenile winter-run salmon occupancy was most influenced by flow characteristics, water clarity, weather, and time of year. However, we found that these relationships and their significance changed considerably when we accounted for imperfect detection and species misidentification error. In particular, our results showed a poorer understanding of the factors influencing juvenile winter-run salmon occupancy when more sampling error was incorporated into the modeling. This is evidenced by the decreasing number of significant fixed-effect predictor variables for estimating the occupancy states among the three models (i.e., naïve model: 13 significant parameters; detection model: 6 significant parameters; detection and misidentification model: 5 significant parameters). In addition, the increased amount of variation contained in the year random effects within the models that accounted for imperfect detection and species misidentification suggests a decreased understanding of the factors affecting occupancy within a year (Royle and Dorazio 2008).

We surmise that juvenile winter-run salmon occupancy is likely influenced by a complex, hierarchically structured suite of many tapering effects in which there are some large and important effects, followed by smaller and less important effects (Frissell et al. 1986; Tonn 1990; Poff 1997). As such, our ability to detect smaller effects may have been diminished as we incorporated more sampling error (and thereby statistical noise) into the modeling while keeping the sample size constant (Burnham and Anderson 2002, 2004). Alternatively, the relationships between juvenile winter-run salmon occupancy and the environmental factors that were significant in the model ignoring sampling error and nonsignificant in the other models may also represent false or untrue relationships caused by sampling bias (Peterson and Paukert 2009). Nonetheless, we believe that the environmental factors that were significant in all or most of the models represent the important and influential factors affecting juvenile winter-run salmon occupancy given our data set.

We observed that river flow characteristics, water clarity, and their interactions were the most important environmental factors affecting juvenile winter-run salmon occupancy. Although the estimated effects varied among our models, juvenile winter-run salmon occupancy was positively related to higher mean daily river flow and decreases in river flow immediately after a pulse in flow. These results are consistent with other studies, which found that juvenile winter-run salmon occupancy, abundance, or survival during migration to and through the Estuary was positively associated with higher river

flows (Brandes and McLain 2001; del Rosario et al. 2013; Hassrick et al. 2022). Past studies had difficulty in quantifying the effect and verifying the importance of water clarity independent of pulses in river flow (e.g., del Rosario et al. 2013; Hassrick et al. 2022). In general, pulses in river flow during the beginning of the wet season (November–May; Singer and Dunne 2004) are commonly associated with decreases in water clarity (Schoellhamer et al. 2016). Nevertheless, we found that juvenile winter-run salmon occupancy was positively related to decreases in water clarity apart from river flow. Furthermore, we observed that decreases in water clarity muted the effect of flow pulses on occupancy. As such, we surmise that juvenile winter-run salmon occupancy in our study was likely not the result of merely passive downstream displacement (see Brandes and McLain 2001) but rather the result of juveniles occupying locations that possessed more favorable habitat conditions (Morrice et al. 2020; Hassrick et al. 2022). In general, decreases in water clarity can improve juvenile salmon survival by providing refugia from predators as well as supplying foraging opportunities (Gregory and Levings 1998; McElroy et al. 2018). Thus, from October to December, fewer juvenile salmon may opt to migrate during a pulse in river flow if they already occur in a location with favorable habitat conditions, such as lower water clarity. Although additional research on these relationships should be conducted, we believe that natural resource managers trying to balance competing demands for water should recognize that managed flow pulses during October–December may be less effective in influencing downstream occupancy or movement of juvenile winter-run salmon when ambient water clarity is low.

Because the estimated probabilities of the occupancy states varied considerably among our models, we expected and confirmed that there would be considerable effects of the sampling error on how the data could inform DCC gate operation decisions. The interagency salmon decision framework has been initiated when at least one of the salmon monitoring efforts in the Sacramento River independently detects three or more winter-run-sized (or larger) juvenile salmon during a day after the catch count data were standardized to 1 day of typical sampling effort (NMFS 2009, 2011, 2019). For our analysis, we used the abundant occupancy state (two or more juvenile winter-run salmon observed during a sample day) as a proxy for the interagency salmon decision framework initiation criteria, which were highly correlated based on the observed data. Assuming that both detection and species identification were imperfect (represented by the detection and misidentification model), our modeling results indicated that the interagency salmon decision process would be initiated on (1) relatively fewer days when imperfect detection and misidentification error are not accounted for (represented by the naïve model) and (2) relatively more days when only imperfect detection error is accounted for (represented by the detection model). Further, these discrepancies were highest during October and November, when fewer juvenile winter-run salmon were captured during the monitoring. These findings demonstrate that if DCC gate operation decisions are informed by using raw or uncalibrated fish count data (i.e., ignoring imperfect detection and species misidentification error), then juvenile winter-run salmon may have fewer protections. In contrast, if DCC gate operation decisions are informed by using fish count data calibrated based on a detection model (i.e., ignoring species misidentification error), then the DCC would close more often and prevent movement of juvenile winter-run salmon into the interior Delta. In general, the October–December time period on which this study focused can be characterized by less river flow magnitude and variability and higher juvenile winter-run salmon identification accuracy using the LDC method relative to the January–May time period (Williams 2006; Johnson et al. 2017). Therefore, we speculate that the discrepancies between our occupancy modeling results would likely increase if we evaluated juvenile winter-run salmon occupancy at the DCC monitoring sites later in the juvenile outmigration period. Although our results do not represent how historical DCC gate closure decisions were informed or influenced by the salmon count data based on, in part, our analysis of a proxy for the interagency salmon decision framework initiation criteria, our study demonstrates the utility of accounting for both imperfect detection and species misidentification error prior to informing water operation decisions using fish count data.

CONCLUSIONS

Fishery managers have traditionally accounted for data uncertainties by taking a precautionary risk management approach when informing or making critical natural resource management decisions (Hilborn et al. 2001; van Asselt and Vos 2006). However, precautionary fishery management approaches may not be feasible when tasked with conserving endangered species occurring in systems that are managed, in part, to achieve other competing objectives. Thus, critical steps to optimize decision making and best serve all management objectives include evaluating and reducing data uncertainties that could impede our ability to achieve the best management outcomes (Gregory and Long 2009; Conroy and Peterson 2013; Peterson and

Duarte 2020). To better inform contentious fish management and water operation decisions in the Estuary and elsewhere, studies are needed that assess the effects of and account for sampling uncertainties when evaluating the distribution and abundance of fishes and the risk or benefit of particular management actions (Rosenberg and Restrepo 1994). Recent statistical advancements (e.g., Shea et al. 2013; Clement et al. 2022), including our modeling framework, allow natural resource managers to better evaluate the effect of and account for common uncertainties associated with fish count data.

Based on our findings, imperfect detection and species misidentification error altered our perception of the factors influencing juvenile winter-run salmon occupancy, and could affect DCC gate operation decisions and other decisions in the Estuary and its watershed. Thus, fish and water management in the Estuary may benefit from allocating more resources to ensure the accurate identification of juvenile winter-run salmon in the field through a combination of additional observer trainings (Kirsch et al. 2018) and confirming the LDC method run identifications by using genetic run assignment techniques (Harvey et al. 2014; Johnson et al. 2017). Natural resource managers may need to rely on modeling tools to properly calibrate fish count data and reduce bias when informing real-time water operation decisions, as it is not currently possible to generate rapid and accurate genetic run assignments for Chinook Salmon in the Central Valley during sampling days (yet see DeFilippo et al. 2020; Deeg et al. 2022). Further, the development of predictive modeling tools would allow real-time water operation decisions to be informed by empirical fish count data on days when monitoring is not conducted while accounting for known uncertainties, which would likely improve management effectiveness, flexibility, and responsiveness (Håkanson 2004; Conroy and Peterson 2013; Welch et al. 2019; Wohner et al. 2022). Future studies should consider expanding on our analysis by developing predictive occupancy models that account for imperfect detection and misidentification error to directly inform ongoing water operation planning within the Estuary (see Notice of Intent 2022).

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CONFLICT OF INTEREST STATEMENT

There is no conflict of interest declared in this article.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available on the Environmental Data Initiative Data Portalor can be obtained from the U.S. Fish and Wildlife Service's Delta Juvenile Fish Monitoring Program.

ETHICS STATEMENT

There are no ethical guidelines that were applicable to this study.

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