
AN UPDATED ASSESSMENT OF STATUS AND TREND IN THE DISTRIBUTION OF THE CASCADES FROG (*RANA CASCADEAE*) IN OREGON, USA

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Abstract.—Conservation efforts need reliable information concerning the status of a species and their trends to help identify which species are in most need of assistance. We completed a comparative evaluation of the occurrence of breeding for Cascades Frog (*Rana cascadae*), an amphibian that is being considered for federal protection under the U.S. Endangered Species Act. Specifically, in 2018–2019 we resurveyed 67 sites that were surveyed approximately 15 y prior and fit occupancy models to quantify the distribution of *R. cascadae* breeding in the Cascade Range, Oregon, USA. Furthermore, we conducted a simulation exercise to assess the power of sampling designs to detect declines in *R. cascadae* breeding at these sites. Our analysis of field data combined with our simulation results suggests that if there was a decline in the proportion of sites used for *R. cascadae* breeding in Oregon, it was likely a < 20% decline across our study period. Our results confirm that while *R. cascadae* detection probabilities are high, methods that allow the sampling process to be explicitly modeled are necessary to reliably track the status of the species. This study demonstrates the usefulness of investing in baseline information and data quality standards to increase capacity to make similar comparisons for other species in a timeframe that meet the needs of land managers and policy makers.

Key Words.—breeding site; Cascade Range; monitoring; montane; occupancy; wetlands

INTRODUCTION

Biodiversity conservation efforts have increased over the past roughly four decades in a unified effort to quantify and reduce species declines (reviewed in Olson 2006). It seems these efforts have not come too soon for amphibians, with numerous studies documenting their declines across the globe (Wake 1991; Houlahan et al. 2000; Stuart et al. 2004; Wake and Vredenburg 2008; Adams et al. 2013). Given the inherent complexity in ecological systems and amphibian life-history strategies, it is not surprising that amphibians face multiple threats, and their response to these threats is quite variable across species and locations (Grant et al. 2016, 2020). Thus, up-to-date species- and region-specific information on the status and trends of potentially at-risk amphibian species is needed to inform conservation efforts.

The Cascades Frog (*Rana cascadae*) is a pond-breeding amphibian that was historically widespread in the mountains of Washington, Oregon, and northern California, USA (Nussbaum et al. 1983; Pearl and

Adams 2005). The species has been extirpated from many historically occupied areas, particularly in California (Fellers and Drost 1993; Fellers et al. 2008) and Oregon (Pearl et al. 2009). Hypothesized threats to *R. cascadae* include increased exposure to introduced nonnative fishes (Welsh et al. 2006; Pope 2008; Larson et al. 2017), elevated *Batrachochytrium dendrobatidis* prevalence (Piovia-Scott et al. 2015; De León et al. 2017), unfavorable environmental shifts brought on by climate change (Belden et al. 2003; Thurman and Garcia 2017; Kissel et al. 2019), and other factors. Due to these concerns, the species is currently being considered for federal protection under the U.S. Endangered Species Act (U.S. Fish and Wildlife Service 2015); however, information concerning the status of *R. cascadae* breeding occurrence in California and Oregon is over a decade old. Thus, managers are being asked to evaluate whether *R. cascadae* qualifies for federal protection while having information that, although undoubtedly useful, may not reflect the current status of populations.

We had the opportunity to provide updated

information concerning the status and trend in the distribution of *R. cascadae* in Oregon. In particular, we conducted replicate detection/non-detection surveys at the identical sites surveyed by Pearl et al. (2009) to estimate the proportion of potential breeding sites that are currently being used by *R. cascadae* for breeding. Herein, we fit Occupancy Models to these older (2001–2004) and recent (2018–2019) survey data to evaluate change in the distribution of breeding by the species across these two time periods while accounting for imperfect detection. During this process, we also evaluated environmental factors hypothesized to be related to the distribution of *R. cascadae* breeding and our ability to detect *R. cascadae* young-of-year life stages. Furthermore, we developed simulation models based on estimates from our analysis to evaluate sampling designs and to help guide future *R. cascadae* monitoring efforts at Oregon sites.

MATERIALS AND METHODS

Study sites.—*Rana cascadae* is predominately found in the Cascade Range from just south of the USA-Canadian border to just south of Lassen Peak in northern California, USA. Exceptions include *R. cascadae* populations in the Olympic Mountains (Washington) and Trinity Alps (California). The Cascade Range extends north-south over 1,100 km and has several volcanic peaks > 3,000 m in elevation. Much of the land along the crest of the Cascade Range is managed as a part of the National Wilderness Preservation System. The sites we included in our field sampling were limited to the sites surveyed by Pearl et al. (2009) based on our study objectives. Briefly, Pearl et al. (2009) selected sites in Oregon after compiling historical records of *R. cascadae* breeding (i.e., eggs, larvae, and year 0 juvenile frogs) from museum collections, graduate theses, publications, knowledge of the authors based on field observations in prior years, and personal communication with herpetologists and agency biologists. Their effort resulted in 67 potential breeding sites within the Oregon range of the species (Fig. 1). As described by Pearl et al. (2009), this region can get heavy snow (> 3,000 mm/year) in November-April and has short dry summers in June-September. *Rana cascadae* generally breed in ponds and lakes at mid and upper elevations (approximately 600 to > 2,100 m) soon after snowmelt, and larvae usually metamorphose in 2–4 mo (Nussbaum et al. 1983). Most of the sites are within relatively closed forest of Douglas Fir (*Pseudotsuga menziesii*), Western Hemlock (*Tsuga heterophylla*), Mountain Hemlock (*T. mertensiana*), and Lodgepole Pine (*Pinus contorta*); however, some sites are in open subalpine forest dominated by Mountain Hemlock and Lodgepole Pine.

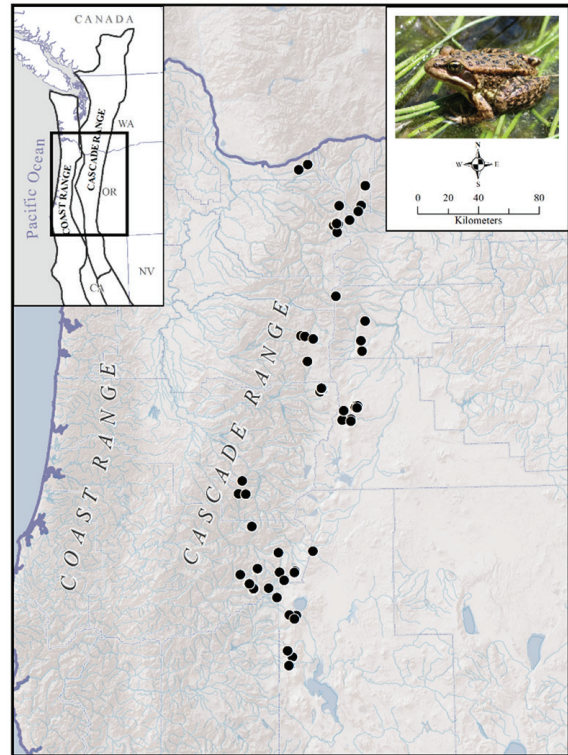


FIGURE 1. Sites surveyed for Cascades Frog (*Rana cascadae*) in 2001–2004 and 2018–2019, Cascade Range, Oregon, USA. An adult Cascades Frog is shown (Photographed by Brome McCreary). Map is in NAD 1983 UTM Zone 10N coordinate system. U.S. state abbreviations are CA = California, NV = Nevada, OR = Oregon, and WA = Washington.

Data collection.—In 2001–2004, Pearl et al. (2009) surveyed sites 1–4 times but they never surveyed a site > 1 time in a single year. In 2018–2019, we surveyed sites 1–2 times but we never surveyed a site in > 1 y. In this study, a body of water (i.e., pond, wetland, or lake) was considered a separate site if it was not connected seasonally by surface water and if it was not part of the same wet meadow or wetland complex. Notably, sampling protocols were consistent across years; however, field crews could not survey one and two sites (three unique sites) in 2001–2004 and 2018–2019, respectively, due to logistical constraints. Crews of 2–4 people conducted field surveys in summers of 2001–2004 and 2018–2019 between 0800 and 2000. In all years, field crews visually surveyed shoreline and littoral habitats at a site and used a dip-net handle to gently agitate vegetation and other cover objects (rocks, logs, etc.) to help flush amphibians. Field crews also conducted dip-net sweeps when there was habitat that may conceal amphibians from visual detection, such as dense vegetation, woody debris, etc. These efforts were not time constrained, and field crews aimed to survey the entire site when possible. Exceptions were a few surveys (< 20%) where small portions of the site were

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TABLE 1. Summary of explanatory variables considered in Occupancy Models to estimate site use probability (ψ) and detection probability (p) for Cascades Frog (*Rana cascadae*), Cascade Range, Oregon, USA. When a site was not surveyed within a given time period, we used the mean value for percentage aquatic vegetation cover and assumed no changes across time periods in classifications of fish presence, hydroperiod, and site origin. Easting and Northing are in NAD 1983 UTM Zone 10N coordinate system. The abbreviation SD = standard deviation.

Variable	Model parameter	Level	Summary
Aquatic vegetation cover (<i>Veg</i>)	ψ, p	Site \times Time period	Mean (SD): 42.9 % (34.62 %) Range: 0–100 %
Day of year (<i>DOY</i>)	p	Survey occasion	Mean (SD): 193.9 d (20.22 d) Range: 149–239 d
Easting (<i>East</i>)	ψ	Landscape	Mean (SD): 577016 m (26154.86 m) Range: 532075–615777 m
Elevation (<i>Elev</i>)	ψ	Landscape	Mean (SD): 1,529 m (323.58 m) Range: 755–2,022 m
Fish presence (<i>Fish</i>)	ψ, p	Site \times Time period	2001–2004: Absent: 44 sites Present: 23 sites 2018–2019: Absent: 45 sites Present: 22 sites
Hydroperiod (<i>Hydro</i>)	ψ	Site \times Time period	2001–2004: Seasonal: 19 sites Permanent: 48 sites 2018–2019: Seasonal: 23 sites Permanent: 44 sites
Northing (<i>North</i>)	ψ	Landscape	Mean (SD): 4859769 m (98960.46 m) Range: 4713503–5044414 m
Site origin (<i>Origin</i>)	ψ	Site \times Time period	2001–2004: Altered: 11 sites Natural: 56 sites 2018–2019: Altered: 21 sites Natural: 46 sites
Time Period (<i>Time</i>)	ψ	Time period	2001–2004: 67 sites 2018–2019: 67 sites
Year (<i>Yr</i>)	p	Survey occasion	2001: 40 sites 2002: 33 sites 2003: 24 sites 2004: 18 sites 2018: 59 sites 2019: 35 sites

unsafe to survey or under snow or where a historical record was very specific about a location within a very large site (i.e., a lobe of a reservoir). We considered *R. cascadae* breeding detected if young-of-year were encountered during surveys. Young-of-year were tadpoles or metamorphs (there are no known instances of overwintering in larval stage in our study area; pers. obs.) or young juveniles that have limb emergence scars or are < 30 mm snout-urostyle length.

In all years, field crews measured environmental conditions that we hypothesized would influence *R. cascadae* breeding occurrence and detectability (Table 1). Given the negative effect non-native fish can have on ranid frogs in mountains of the western USA generally (Knapp and Matthews 2000; Pilliod and Peterson 2001;

Vredenberg 2004), their potential to affect *R. cascadae* specifically (Welsh et al. 2006; Pope 2008; Joseph et al. 2011; Larson et al. 2017), and the prominence of introduced trout (*Oncorhynchus* spp. and *Salvelinus* spp.) in our study area (Bahls 1992), field crews noted if they encountered nonnative trout or other fish during field surveys. Because the sampling methods were not specifically designed to sample fish, we consulted with U.S. Forest Service and Oregon Department of Fish and Wildlife biologists with knowledge of study sites to better classify each site if we did not detect fish at a site known to be occupied by fish. Field crews visually assessed whether sites were natural or altered by human development (e.g., impoundments or excavations) so we could evaluate if altered sites had lower *R. cascadae*

breeding than natural sites. Anurans often use emergent, floating, and submergent aquatic vegetation for basking, foraging, and predator evasion. By extension, multiple studies have found that the amount of aquatic vegetation influences the occurrence and detectability of anurans (Gould et al. 2012; Hossack et al. 2015; Holgerson et al. 2019; Duarte et al. 2020). Thus, field crews visually estimated the percentage of each site that had aquatic vegetation cover. Hydroperiod can have a dramatic influence on the colonization and local persistence of anurans at a site (Zipkin et al. 2012; Davis et al. 2017; Duarte et al. 2020). We expected *R. cascadae* breeding would be less likely at sites that lacked permanent water. We classified the hydroperiod at each site (i.e., permanent or seasonal) based on the presence of hydrophytic vegetation, the depth of the water during site visits, and field observations of sites drying across years of sampling. Field crews recorded the day of year each survey took place so we could account for effects of *R. cascadae* development on our ability to detect the species. Lastly, we derived location-specific information such as elevation, northing, and easting to evaluate spatial gradients in *R. cascadae* breeding. The 2018–2019 field data can be accessed at Adams et al. (2019).

Occupancy analysis.—Species detection is considered perfect if field crews are guaranteed to detect a species at a site, given the species is present, without error. Herpetofauna encounter data both at the individual and species level are commonly biased by imperfect detection (reviewed in Mazerolle et al. 2007). Ignoring imperfect detection can lead to erroneous conclusions concerning the status and trends of species. For example, a decline in detection probabilities can lead to reduced encounter rates in the monitoring data, which can appear like a decline in occupancy or abundance when no actual decline has occurred (Anderson 2001; Thompson 2002; Duarte et al. 2017). Occupancy Models use replicate detection/non-detection survey data to correct monitoring data for imperfect detection by modeling the biological and observation process simultaneously to estimate parameters such as occupancy and detection probabilities (MacKenzie et al. 2002). Occupancy probability is the probability that a randomly selected sample unit within the larger study area (i.e., the collection of sample units) is occupied by a species, and detection probability is the probability of obtaining a positive detection for that species, given the sample unit is occupied by the species. Importantly, these probabilities can be related to explanatory variables using Logistic Regression. The flexibility of Occupancy Models has led to their widespread use to address various research and management questions in ecology (reviewed in Mackenzie et al. 2017). Still, a

fundamental assumption of Occupancy Models is that the occupancy state at a site does not change within a primary occasion (i.e., across replicate surveys within a sampling season). This assumption is likely violated for our study because sites were not surveyed multiple times within a year during the 2001–2004 surveys, and large fluctuations in amphibian populations across years are well documented (Pechmann et al. 1991; Semlitsch et al. 1996; Alford and Richards 1999; Trenham et al. 2003; Rowe et al. 2019). The closure assumption, however, can be relaxed if turnover in site occupancy can be assumed random (non-Markovian) and if we modify our interpretation of the parameters (reviewed in Mackenzie et al. 2017). Specifically, the occupancy parameter becomes the probability that the target species ever used the site within a primary occasion (hereafter, site use probability) and detection probability becomes the probability a species was detected given it was present at a site and available for detection during the sampling occasion.

We fit Single-season Occupancy Models to the *R. cascadae* survey data collected at all 67 unique sites using a Bayesian hierarchical framework with 2001–2004 and 2018–2019 being our two primary occasions. This approach does not link site use probability across time periods, but instead treats site use probability as a random process in each time period (MacKenzie et al. 2017). This matched our objective because we were primarily interested in estimating and comparing the distribution of *R. cascadae* breeding at our sites at two points in time and were less interested in estimating the underlying dynamic processes. Covariates considered on model parameters were based on our rationale outlined above. Specifically, we evaluated if site use probability was related to fish presence, percentage aquatic vegetation cover, hydroperiod, site origin (i.e., natural or altered by human development), northing, elevation, easting, and the interaction between northing and elevation. We evaluated if detection probability was related to fish presence, year of survey, percentage aquatic vegetation cover, day of year, and the interaction between fish presence and percentage aquatic vegetation cover. We compared models using the Watanabe-Akaike Information Criterion (WAIC; Watanabe 2013), which is similar to other information theoretic model selection procedures but is a fully Bayesian and valid approach for hierarchical models (Hobbs and Hooten 2015).

We developed models using a sequential-by-submodel strategy. In particular, we compared models for detection probability first while only allowing site use probability to vary by time period and selected the detection probability model with the lowest WAIC. Next, we developed models for site use probability. We followed the approach of Pearl et al. (2009) by developing and comparing all combinations of models

with site- and landscape-level covariates (see Table 1) separately to reduce the size of the model set. We had the *a priori* expectation that we would compare the top ranked models (i.e., lower WAIC values than our null model) from each model set. Because our primary objective was to estimate change, if any, in the distribution of *R. cascadae* breeding, all models we considered included a factor variable to differentiate the two time periods for site use probability. These analyses were completed using JAGS (Plummer 2003) within program R (R Development Core Team 2016) using the R2jags package (Su and Yajima 2009). Prior to fitting models, we standardized all continuous variables to have a mean of zero and standard deviation of one and assessed the variables for collinearity (all had $|r| < 7$). We used diffuse normal priors for all model intercepts and coefficient estimates ($\mu = 0$, $\tau = 0.368$). We specified model runs to have three independent chains, each consisting of 100,000 iterations following a burn-in of 50,000. We inspected models for convergence using the Brooks and Gelman diagnostic (all had $\hat{R} < 1.01$; Brooks and Gelman 1998) and visual inspection of trace and density plots of the posterior distributions.

We described model parameters by their mean, standard deviation (SD), and 95% equal-tail credible interval (95% CI). We also calculated odds ratios for model coefficients (Hosmer and Lemeshow 2000). We considered a covariate to be strongly influential if the posterior distribution of the coefficient estimate was $\geq 90\%$ on one side of zero (Fields et al. 2017). Lastly, we estimated the proportion of sites used for breeding in each time period as a derived parameter, relying strictly on covariates to predict site use probability for sites not surveyed in a given time period. Importantly, when a site was not surveyed within a given time period, we used the mean value for percentage aquatic vegetation cover and assumed no changes across time periods in classifications of fish presence, hydroperiod, and site origin.

Power to detect trends.—We used a simulation-based approach to evaluate our ability to detect a change in the proportion of sites used for *R. cascadae* breeding between two primary occasions and to help guide future *R. cascadae* monitoring efforts in Oregon. In all simulations we restricted the number of sites to 67 to match our collection of Oregon sites. Simulations began with a specified number of replicate surveys at each site per time period, an initial site use probability (ψ^1), and a percentage decline in site use probability ($\Delta\psi$). Site use probability in the second time period (ψ^2) was then calculated as $\psi^2 = \psi^1 - (\psi^1 \times \Delta\psi)$. Detection probability was randomly sampled in each iteration from a beta distribution using the mean and SD estimated from our analysis of field data (see Results). The true occupancy

state of each site in each time period was randomly generated in each iteration using a binomial distribution with a probability of success equal to the site use probability for that specific time period. Detection/non-detection data were then randomly generated in each iteration using a binomial distribution with the number of observations equal to the specified number of replicate surveys, the number of trials equal to the true occupancy state (i.e., 1 or 0), and the probability of success equal to the detection probability.

We simulated scenarios with initial site use probability ranging from 0.55 to 0.85 in increments of 0.10; percentage decline in site use probability ranging from 5% to 30% in increments of 5%; and number of replicate surveys at each site per time period ranging from two to six in increments of one. During each iteration we fit an Occupancy Model to the simulated survey data using a Bayesian hierarchical framework, where detection probability was modeled as a constant and we included a factor variable to differentiate the two time periods for site use probability. We fit models, again, using JAGS (Plummer 2003) within program R (R Development Core Team 2016) using the R2jags package (Su and Yajima 2009). Preliminary simulations demonstrated that model runs specified to have three independent chains, each consisting of 75,000 iterations following a burn-in of 50,000, were sufficient to achieve convergence. We considered the model to have detected a decline in site use probability if the posterior distribution of the estimate for the model coefficient representing the second time period was $\geq 90\%$ below zero (Fields et al. 2017). We ran all scenarios for 500 iterations, and we calculated the proportion of iterations that detected a decline in site use for each scenario.

RESULTS

In 2001–2004, Pearl et al. (2009) surveyed 18 to 40 sites each year. During this primary occasion, they surveyed each of the 67 unique sites 0 ($n = 1$), 1 ($n = 32$), 2 ($n = 19$), or 3 ($n = 15$) times, and detected *R. cascadae* breeding in at least one survey at 44 sites (66.7% of 66 sites). We surveyed 41 and 24 sites in 2018 and 2019, respectively. During this primary occasion, we surveyed each of the 67 unique sites 0 ($n = 2$), 1 ($n = 36$), or 2 ($n = 29$) times, and we detected *R. cascadae* breeding during at least one survey at 37 sites (41.5% of 65 sites).

We compared 20 models for detection probability (Table 2). Although WAIC scores for the top three models were similar, the most supported model out of these was also the most parsimonious model. It only included the individual effect of fish presence on detection probability. Thus, we based our inferences for detection probability on this model. The probability *R. cascadae* breeding was detected given it was present at a

TABLE 2. Model selection results for the occupancy analysis to estimate detection probability (p) and site use probability (ψ) for breeding Cascades Frog (*Rana cascadae*), Cascade Range, Oregon, USA. Models were developed using a sequential-by-submodel strategy by evaluating p first, followed by ψ . Models were ranked using Watanabe-Akaike Information Criterion (WAIC). Variables in the models are defined in Table 1.

Detection probability (p)		Site use probability (ψ)	
Model	WAIC	Model	WAIC
ψ (Time), p (Fish)	178.43	ψ (Time), p (Fish)	178.43
ψ (Time), p (Fish+Veg)	179.52	ψ (Time+East), p (Fish)	179.23
ψ (Time), p (DOY+Fish)	179.73	ψ (Time+North), p (Fish)	180.19
ψ (Time), p (Fish+Veg+Fish×Veg)	180.65	ψ (Time+Elev), p (Fish)	180.27
ψ (Time), p (DOY+Fish+Veg)	180.81	ψ (Time+Elev+North), p (Fish)	180.59
ψ (Time), p (DOY)	181.62	ψ (Time+Veg), p (Fish)	180.89
ψ (Time), p (.)	181.75	ψ (Time+Origin), p (Fish)	181.07
ψ (Time), p (DOY+Fish+Veg+Fish×Veg)	182.03	ψ (Time+Hydro), p (Fish)	181.07
ψ (Time), p (DOY+Veg)	182.74	ψ (Time+East+North), p (Fish)	181.46
ψ (Time), p (Veg)	182.76	ψ (Time+East+Elev), p (Fish)	181.70
ψ (Time), p (Fish+Veg+Yr)	190.02	ψ (Time+East+Elev+North), p (Fish)	182.11
ψ (Time), p (Fish+Yr)	190.18	ψ (Time+Origin+Veg), p (Fish)	183.47
ψ (Time), p (DOY+Fish+Yr)	191.55	ψ (Time+Hydro+Veg), p (Fish)	183.77
ψ (Time), p (Veg+Yr)	191.59	ψ (Time+Hydro+Origin), p (Fish)	183.78
ψ (Time), p (DOY+Fish+Veg+Yr)	191.69	ψ (Time+Fish), p (Fish)	185.46
ψ (Time), p (Fish+Veg+Fish×Veg+Yr)	191.81	ψ (Time+Hydro+Origin+Veg), p (Fish)	186.99
ψ (Time), p (Yr)	192.18	ψ (Time+Fish+Hydro), p (Fish)	187.68
ψ (Time), p (DOY+Veg+Yr)	192.78	ψ (Time+Fish+Origin), p (Fish)	187.90
ψ (Time), p (DOY+Fish+Veg+Fish×Veg+Yr)	193.07	ψ (Time+Fish+Veg), p (Fish)	188.24
ψ (Time), p (DOY+Yr)	193.14	ψ (Time+Elev+North+Elev×North), p (Fish)	188.60
		ψ (Time+Fish+Hydro+Origin), p (Fish)	190.45
		ψ (Time+Fish+Origin+Veg), p (Fish)	190.82
		ψ (Time+East+Elev+North+Elev×North), p (Fish)	191.44
		ψ (Time+Fish+Hydro+Veg), p (Fish)	191.51
		ψ (Time+Fish+Hydro+Origin+Veg), p (Fish)	194.44

site and available for detection during a survey was 0.81 (SD = 0.05, 95% CI = 0.70–0.90) when fish were absent at a site. The odds of detecting *R. cascadae* breeding given it was present at a site and available for detection was 2.05 (95% CI = 0.80–5.18) times lower when fish were present at a site, with the probability at 0.68 (SD = 0.08, 95% CI = 0.52–0.83; Table 3).

We developed and compared site use probability models at the site- and landscape-level separately with the *a priori* expectation that we would compare the top ranked models from each model set. Our null model that treated site use probability as a function of time period, however, was the most supported model in both model sets. Therefore, we combined the model sets, which resulted in 25 models considered for site use probability (Table 2). Similar to before, the top four models had similar WAIC values, but the most supported model was

also the most parsimonious model with only time period as a covariate. Thus, we based our inferences for site

TABLE 3. Posterior means, standard deviations (SD), 95% equal-tail credible intervals (CI), and the proportion of the posterior distribution on the same side of zero as the mean (f) for parameters of the best-approximating Occupancy Model based on Watanabe-Akaike Information Criterion (WAIC) for breeding Cascades Frog (*Rana cascadae*), Cascade Range, Oregon, USA.

Parameter	Mean	SD	Lower CI	Upper CI	f
Detection probability (p)					
Intercept	1.502	0.355	0.832	2.225	1.000
Fish: Present	-0.717	0.474	-1.644	0.219	0.935
Site use probability (ψ)					
Intercept	1.078	0.348	0.452	1.821	1.000
Time: 2018–2019	-0.356	0.453	-1.247	0.535	0.788

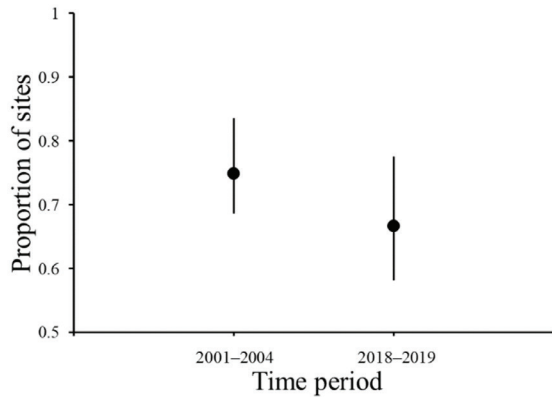


FIGURE 2. The predicted proportion of sites used for breeding (mean \pm 95% equal-tail credible interval) by Cascades Frog (*Rana cascadae*) in 2001–2004 and 2018–2019, Cascade Range, Oregon, USA.

use probability on this model. Site use probability was estimated to be lower in 2018–2019 (Fig. 2), with the odds of *R. cascadae* using a site for breeding being 1.43 (95% CI = 0.59–3.48) times lower in 2018–2019 (Table 3). The effect of time period was not strong, however, with only 78.8% of the posterior distribution of the estimated model coefficient being less than zero.

As expected, our simulations indicated that the ability to detect a decline in the occupancy parameter increased as the percentage decline and initial occupancy parameters increased (Fig. 3). Notably, a higher initial occupancy parameter translated into an increased

change in occupancy probability across primary occasions for a given percentage decline due to how change was calculated in the simulations. Our ability to detect a decline was relatively invariant to increasing the number of replicate surveys at a site, which is directly related to the high probability of detection when fish were absent during our surveys. It should also be noted that simulations suggesting fewer surveys increased our power to detect a decline in some instances is directly related to our high estimates of detection probability and the stochastic nature of our simulations. In other words, this pattern is related to sampling variability among iterations and it should not be interpreted that fewer replicate surveys are better.

DISCUSSION

We provide updated information concerning *R. cascadae* breeding occurrence in Oregon by undertaking a comparative analysis of older (2001–2004) and recent (2018–2019) survey data. Despite the negative trend in estimated site use and in raw detection/non-detection data, there was considerable uncertainty in the estimates. Thus, there is insufficient evidence to conclude that an actual decline in site use over the last 15 y has occurred.

At a fundamental level, the ability to detect an increasing or decreasing trend in site use is a tradeoff between signal and noise: the biological signal must be greater than the noise in the data. Like many others, we approached this problem by explicitly modeling

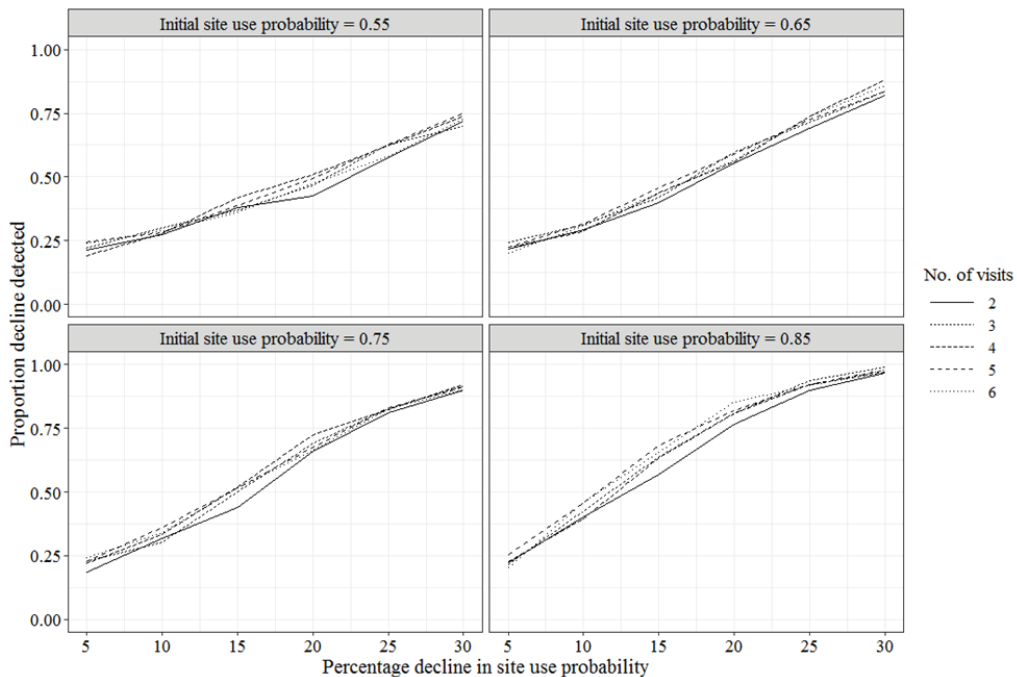


FIGURE 3. The proportion of simulations that detected a decline in site use probability (y-axis) for scenarios with different percentage decline (x-axis), initial site use probability, and number of replicate surveys within a primary occasion.

the observation and biological process separately using Occupancy Models. The accuracy of estimates derived from Occupancy Models is related to how well we were able to meet model assumptions and, of course, the data themselves. We are confident that in our case the only likely assumption violation was population closure within a season, which is why we modified the interpretation of the estimated parameters. We do not consider estimating site use probabilities, rather than occupancy probabilities, to be a weakness in our approach given the objective was to provide contemporary estimates of where *R. cascadae* currently breed relative to 2001–2004 conditions. Still, it is worth reiterating that 2001–2004 surveys at sites spanned 1–3 y, whereas our recent surveys restricted replicate surveys at a site within a single year. Thus, 2001–2004 surveys had a greater opportunity to detect *R. cascadae* breeding if turnover was high across years. Nevertheless, detection probability estimates indicate that with only two surveys at a site we had a high probability of detecting *R. cascadae* breeding given breeding occurred at a site and young-of-year were available for detection (0.89, SD = 0.05, 95% CI = 0.77–0.97 when fish were present and 0.96, SD = 0.02, 95% CI = 0.91–0.99 when fish were absent). Furthermore, a high population turnover rate would translate into lower and more variable detection probabilities because of the effect of availability. There was no evidence that detection probabilities varied by year. Thus, we believe any noise in the detection/non-detection data related to the sampling design was largely accounted for with our analytical approach.

Again, the accuracy of Occupancy Models is also related to the data themselves. In general, accuracy improves as the number of sites, number of replicate surveys at a site, and detection probability increase. In our case the number of sites was restricted to sites with historical records of *R. cascadae* breeding and the number of replicate surveys was limited due to logistical constraints. Therefore, any improvement in the accuracy of our estimates needed to come from higher detection probabilities, although our estimated detection probabilities were relatively high. Thus, we evaluated the power of our study to detect a decline in the occupancy parameter using a simulation-based approach. Based on our simulations and the estimated site use probability of approximately 0.75 for 2001–2004, it seems our ability to detect a decline was relatively high if the decline in the occupancy parameter was $\geq 20\%$ (Fig. 3). Thus, the results from our occupancy analysis and simulations suggest that if there was a decline in the number of sites used for *R. cascadae* breeding in Oregon across our study period it was likely a $< 20\%$ decline (i.e., a $< 0.75 \times 0.20 = 0.15$ change in the proportion of sites used for breeding). Although this may be considered a substantial decline, our simulation demonstrates the potential limits

of our data to quantify change and highlights the value in developing sampling designs while considering the ever-increasing need to understand species status and trends when initiating field studies.

It is important to consider how the site selection process and analytical approach might have influenced our findings. Sites were originally selected if there was documented *R. cascadae* breeding at some point prior to 2001. Although these records were quite variable in age, most were within 10 y of the initial 2001 surveys (Pearl et al. 2009). This could mean the site selection process predisposed our results to detect a decline, rather than an increase, in the proportion of sites *R. cascadae* used for breeding. The oldest documented individual *R. cascadae* in our study area, however, was measured at 5 y of age (Briggs and Storm 1970) and site use in 2001–2004 was not at capacity (i.e., the proportion of sites used for breeding was < 1). Thus, *R. cascadae* populations had ample opportunity to undergo both local extinction and colonization at sites multiple times between time periods that could have just as likely resulted in an increase in the distribution of *R. cascadae* breeding. Still, our analytical approach evaluated if there was a decline in the proportion of sites used for breeding and does not examine changes in the number of *R. cascadae* individuals within populations. Unfortunately, methods to estimate abundance that strictly rely on count data (i.e., N-mixture models) are extremely sensitive to assumption violations and unmodeled heterogeneity in the count data (Barker et al. 2017; Duarte et al. 2018; Link et al. 2018). Thus, we opted to use Occupancy Models, and declines in population abundances, if present, would not be reflected in our results concerning site use if ≥ 1 individual attempted to breed at a site.

While our analysis did not provide conclusive evidence of a decline in the distribution of *R. cascadae* breeding, both this study and the previous assessment (Pearl et al. 2009) confirmed the effects of fish on detection probability of *R. cascadae* young-of-year life stages. There are several biological mechanisms that may be related to this pattern. First, predatory fish may influence the behavior of palatable prey in ways that make prey (in our case *R. cascadae*) less detectable (e.g., increased use of cover and reduced movement activity; Hartman et al. 2013). Second, fish can reduce abundance of palatable amphibians like *R. cascadae* via predation, disease, or reduced somatic growth and survival based on activity changes (Vredenberg 2004; Pope 2008). Reduced abundance would likely result in lower detection probabilities given the detection probability of a species is directly related to population abundance and the capture probability of individuals within a population (Bayley and Peterson 2001; Royle and Nichols 2003). Third, fish in our study were mainly trout historically introduced for angling, and these

sites receive more human use than fishless habitats (pers. obs.). Factors associated with recreational use of sites by humans may affect both the abundance and detectability of montane amphibians (Dunham et al. 2004). Last, sites that support trout often have aquatic habitats with increased water depth and cooler water temperatures. These environmental conditions might provide suboptimal habitat for *R. cascadae* for reasons other than a direct fish effect on abundance. Regardless of the biological mechanism, this finding suggests using field and analytical methods that account for the effects of fish on detection probability are warranted when monitoring amphibians in mountain landscapes, and it should be noted that Co-occurrence Occupancy Models (MacKenzie et al. 2004) that estimate co-occurrence probabilities while accounting for imperfect detection for both species should be considered in similar studies, if possible.

Both this study and the previous assessment (Pearl et al. 2009) failed to find strong support for environmental predictors of *R. cascadae* site use probability. This could be related to at least two factors. First, a fundamental assumption of Occupancy Models is that the occupancy state does not change within a primary occasion. Thus, covariate values used when fitting models to the occupancy parameter must accurately capture the conditions at a site without varying across replicate surveys within a primary occasion. This is complicated for the 2001–2004 survey data because habitat conditions such as hydroperiod and percentage aquatic vegetation cover likely varied across years. Although we attempted to capture these covariates well within the primary occasion, our use of the average measurement across years coupled with the natural annual variability in these habitat conditions may have limited our ability to find a strong relationship between these covariates and the occupancy parameter. Second, it could also be due to us not considering other, perhaps more influential, environmental factors on the occupancy parameter. For example, weather stations high in two large basins in our study area (i.e., the Klamath and Deschutes River Basins) recorded snow-water equivalent (SWE) values that were 25–75% of the 30-y median in four of the 6 y prior to our second primary period (2013, 2014, 2015, 2018), with the minimum SWE in the period recorded in 2015 (<https://www.nrcs.usda.gov/Internet/WCIS/basinCharts/POR/WTEQ/OR/>). We attempted to capture changes in water availability by classifying the hydroperiod of sites in each time period. Furthermore, we included time period as a covariate on the occupancy parameter, which should capture large-scale changes in environmental conditions across time. Still, our use of data collected 15 y apart limited the ability to examine annual effects of drought on *R. cascadae* breeding site use. Importantly, many aspects of *R. cascadae* adult

longevity and philopatry to breeding sites are not well known. These processes likely affect continuity of site use for breeding and could be affected by drought conditions. Thus, further work concerning the influence of snowpack on *R. cascadae* is needed as snowpack continues to decrease in the region (Mote et al. 2005).

As biodiversity conservation efforts continue to work toward identifying and minimizing threats to amphibians across the globe, reliable information concerning the status and trends of species within regions is the first step to identify species that are in most need of assistance and where assistance is most needed. This often requires older, large-scale survey data to make comparisons, but such data are not often available for species of concern generally and for amphibians specifically. Furthermore, this information must be timely to help inform policy and management decisions that often need to be made within relatively short timeframes. Our comparative evaluation of the distribution of *R. cascadae* breeding in Oregon was possible because of the extensive effort Pearl et al. (2009) undertook to compile a list of sites with historical *R. cascadae* breeding, their use of a sampling design that allowed their survey data to be corrected for imperfect detection, and our ability to replicate their field efforts approximately 15 y later to make direct comparisons within a single analysis. Our study confirms that the ability to correct monitoring data (both past and present) for imperfect detection is needed to be confident that trends in the data (or lack thereof) are not an artifact of the sampling process, especially since detection probability often varies with changing habitat conditions. We hope this study demonstrates the usefulness of investing in such baseline information and data quality standards to increase the capacity to make similar comparisons for other species, as needed, in a timeframe that meets the needs of land managers and policy makers.

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Herpetological Conservation and Biology



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