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Notes from the Editor

With this issue of the *Journal* we are making the transition to a new Editor-in-Chief. I have now retired, bringing to an end, my 32-plus year career with the Department of Fish and Wildlife. Looking back, it doesn't seem like it was that long ago that I started my career, my first day, at Grizzly Island Wildlife Area participating on a tule elk capture. That was a sign of good things to come. My career spanned three regions beginning in Santa Cruz as a unit wildlife manager. From there I was fortunate enough to move to Eureka to work on spotted owls and timber harvest review on the north coast, then to the central Sierra Nevada as a wildlife program manager. Then, finally, Headquarters as special advisor. My proudest accomplishment of course would be leading the 2015 update of the State Wildlife Action Plan. Admittedly the plan's a bit big, but for those who take the time to read it, they will learn it's a blueprint for implementing priority conservation in California. Literally just cut and paste the strategies, add the dates, location and a budget, and voila. It was the regional staffs that developed the strategies; I was privileged to be part of it. It was a remarkable career and along the way I met and worked with remarkable people like Chuck Graves, Bruce Elliott, Ken Moore, Gary Stacey, Mark Stopher, and Kevin Hunting. These awesome people were some of my supervisors who shared their knowledge and skills and helped guide my career to its successful conclusion. I'm leaving in hopes that the Department will continue on its path of reestablishing itself as a leader in science and conservation. There are tremendous people working here who will face an almost unsurmountable challenge ahead with climate change, invasive species and population growth. Species and habitats are changing and it's predictable that it's going to be difficult to keep up with the change once it gets going. This is foreseeable, which means it would be negligent to not take action now to prepare, e.g. SWAP. I'm not going far, in fact I be back although in the private non-profit sector following law school. I hope to see you again, and thanks for everything.

Armand Gonzales
Editor-in Chief

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Annual and seasonal variation, relative abundance, and effects of managed flows on timing of migration in Brown Trout (*Salmo trutta*) in the upper Trinity River

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We evaluated annual and seasonal patterns of relative abundance and timing of migration from historic trapping data in non-anadromous Brown Trout (*Salmo trutta*) inhabiting the upper Trinity River, California. Results of our analysis failed to support the hypothesis that the population of Brown Trout in the upper Trinity River has increased and continues to proliferate since 2000. Instead, we hypothesize that the peak in Brown Trout catch per unit effort (CPUE) in 2004, was not an indication of an increase in population size, but rather a secondary potamodromous behavioral response by Brown Trout already in the upper Trinity River system, in response to variation in managed flows and altered environmental conditions that ensued. We also tested the hypothesis of no significant difference in timing of migration in response to annually managed flow regimes. Managed hydrographs associated with the Trinity River Restoration Program (TRRP) and United States Bureau of Reclamation (USBR) were categorized into three flow types: 1) baseline Pre-ROD flows (1982-2002); 2) Record of Decision (ROD) flows (2005-2011, 2017); and 3) pulsed augmentation (Pulse) flows (2003, 2004, 2012-2016). Annual variation in CPUE showed cyclic fluctuations approximately every four to eight years and there was a significant positive relationship between CPUE and year (1982-2017). However, for the sampling period 2003 to 2017, the relationship between CPUE and year was significant and negative, indicating that Brown Trout have declined dramatically in relative abundance since peaking in 2004, especially after 2014. This sequence of dates coincides with establishment of the Trinity River Restoration Program in 2002, and subsequent Record of Decision “ROD flows” and periodic augmentation flows (“Pulse flows”) beginning in 2003. Additionally, our results failed to support the hypothesis of no significant difference in timing of migration among different flow types. Instead, annually managed flow regimes appear to have significantly affected timing of migration in Brown Trout. Deviation away from the baseline Pre-ROD flow pattern of seasonal migration occurred through reduction in counts of fish early to mid-season beginning in late May (Julian week 21), followed by an increase in counts late in the season (mid-December, Julian week 49); thus displacing the baseline Pre-ROD flow timing of migration to later in the season. Results of our analysis, together with a review of pertinent literature and available data do not support the suggestion that Brown Trout be actively removed from the upper Trinity River, because of increased population growth since 2000, competitive lifestyle, or negative impact to native juvenile anadromous salmonids, relative to other co-occurring adult piscivorous salmonids and fish-eating terrestrial vertebrates. We make several recommendations for future management actions to help resolve issues related to Brown Trout and other salmonids in the Trinity River.

Key words: annual and seasonal variation, Brown Trout, managed flow regimes, migration, piscivorous lifestyle, potamodromous, suppression of population, Trinity River

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Brown Trout (*Salmo trutta*) are a non-native species of salmonid found in the Trinity River, Klamath Basin of northwestern California. Although capable of developing an anadromous life history form in response to localized food limitation as a population expands (O'Neal and Stanford 2011), there are no definitive tagging studies to suggest that the current population of Brown Trout in the upper Trinity River is anadromous (M. Currier, California Department of Fish and Wildlife [CDFW] Reservoir Biologist, personal communication 2017). This species has coexisted with native anadromous salmonids in the Trinity River for over a century. Brown Trout are territorial, predatory, and potentially compete with co-occurring native anadromous salmonids for food, space, and cover (Glova and Field-Dodgson 1995, L'Abee-Lund et al. 2002). Large adult Brown Trout may predominate in areas of suitable habitat within the mainstem Trinity River. Preliminary analysis of count data suggested that the population of Brown Trout in the upper Trinity River has increased and continues to proliferate since 2000 (CDFW 2014, USBR 2014). This hypothesis, in conjunction with the view that Brown Trout adversely affect populations of juvenile Chinook Salmon (*Oncorhynchus tshawytscha*), steelhead (*Oncorhynchus mykiss*), Klamath River Lamprey (*Entosphenus similus*), and potentially impede recovery of listed Coho Salmon (*Oncorhynchus kisutch*) (NOAA Fisheries 2014), resulted in recommendations to specifically and systematically remove Brown Trout in the upper Trinity River (Alvarez 2017).

At issue is whether: 1) "continued proliferation" of Brown Trout undermines efforts to restore native anadromous fish in the upper Trinity River; 2) release of Brown Trout captured during salmonid monitoring is a breach of Tribal Trust Responsibilities (DOI 1993, TRFES 1999) constituting "take" of listed Coho Salmon; and 3) presence of Brown Trout significantly reduces commercial and sport fishing opportunities for native salmonids. However, the same piscivorous lifestyle is also true for resident steelhead, and Coho Salmon in other river systems (Ruggergone 1989, Ruggergone and Rogers 1992, McConnaughey 1999, TRFES 1999, Naman 2008, YTFP 2008). Moreover, numerous other aquatic and terrestrial piscivorous predators also inhabit the upper Trinity River (TRFES 1999). Further, comprehensive comparative studies that document: 1) competition among anadromous salmonids, and 2) the relative impact and importance of predation on juvenile salmonids by any of a suite of anadromous, aquatic, or terrestrial piscivorous taxa inhabiting the upper Trinity

River are lacking. As such, the long-term benefit to populations of juvenile salmonids, by systematically eliminating adult Brown Trout from the upper Trinity River, is lacking critical information and remains entirely unknown, as is the relative impact to the local economy in the context of both current and future opportunities for recreational angling.

Complicating this issue further is a lack of understanding of the potential effects of variable and intensely managed annual flow regimes, which characterizes the upper Trinity River, on the relative abundance estimates and timing of migration in several species of adult salmonids. For example, effects of seasonal variability in relative abundance of salmonid populations associated with annually managed flow regimes and restoration programs can be considerable (Platts and Nelson 1988, Holtby and Scrivener 1989, Bradford et al. 1997, Ham and Pearsons 2000, Bayley 2002, Hasler et al. 2014, Peterson et al. 2017). Such variability may severely constrain estimates of population size and trends, and interpretations of the effects of variable managed flow and temperature regimes on seasonal patterns of migration, local movements, habitat use, and rates of survival in resident non-anadromous and anadromous fish (Crisp 1993, Clark and Rose 1997, Cunjak et al. 1998).

Because Brown Trout in the Trinity River are non-anadromous and do not rely on ocean conditions for their life history requirements, their annual abundance and seasonal migratory responses to changes in flow patterns affected by managed flow regimes are independent of any oceanic influence, unlike anadromous species. As such, we view Brown Trout as an excellent "control" species for evaluating potential effects of managed hydrological variation within the upper Trinity River, compared to anadromous salmonids.

Our specific objectives were fourfold. First, we re-evaluate relative abundance, annual distribution, timing of seasonal migration, and potential impact of Klamath River Lamprey (*Entosphenus similus*) parasitism on Brown Trout, relative to other sympatric salmonids. Second, we test the hypothesis that the population of Brown Trout in the upper Trinity River has increased since 2000. Third, we test the hypothesis of no significant difference in pattern of timing of migration in relation to annually managed flow regimes (hydrographs). Forth, we use results of our analyses to address: 1) competition among sympatric salmonids inhabiting the upper Trinity River, 2) the potential impact to commercial and sport fishing opportunities, and 3) management recommendations advocating systematic removal of Brown Trout from the Trinity River because of its competitive and piscivorous lifestyle.

Background on history of introduction.—The United States Commission of Fish and Fisheries in the late 1800s imported both "Von Behr" trout from the Black Forest of Germany (stream type *S. trutta*), and "Loch Leven" trout from Scotland (lake type *S. trutta*). Von Behr trout eggs were brought to the New York State Hatchery at Cold Springs Harbor and the United States Fish Commission hatchery at Northville, Michigan in 1882. Loch Leven trout eggs were brought to Cold Springs Harbor Hatchery in 1884 (Dill and Cordone 1997). Although Brown Trout are frequently referred to as "German Brown Trout", Von Behr trout were eventually outcrossed with Loch Leven fish.

The US Fish Commission hatchery at Northville Michigan delivered Loch Leven, Von Behr, and hybrid Brown Trout eggs to Fort Gaston in Hoopa and Sisson Hatchery in Mt. Shasta, California (Adkins 2007). There were two introductions from these hatcheries into the Trinity River, one near the mouth at Fort Gaston and a separate effort closer to the headwaters in Stewart's Fork and the upper Trinity River near Lewiston, California (Adkins 2007). The U.S. Fish Commission conducted the first documented introduction of Brown Trout into the Trinity River in July 1883 (USCFF 1895). To promote recreational angling, 24,856 yearling Brown Trout were released into the tributaries in the lower Trinity River from fish reared at Fort Gaston in Hoopa Valley (Dill and Cordone 1997). Re-introductions (stocking) of Brown Trout to the Trinity River and tributaries occurred annually from 1911 to 1932, peaking at 180,000 Brown Trout stocked in 1925 (Wertz 1979). From 1964 to 1976, California Department of Fish and Game (CDFG) implemented a Brown Trout maintenance program at Trinity River Hatchery (TRH) and propagated Brown Trout from adult returns to TRH. Managers stocked Brown Trout from this maintenance program on a near annual basis at various locations in the Trinity River and below Lewiston Dam. There is a series of annual hatchery reports documenting TRH Brown Trout production and stocking from 1961 to 1968 by Murray (1968) and from 1970 to 1977 by Bedell (1977 and 1979), including references therein. We summarize data from these reports in Appendix I.

In 1969, CDFG released TRH-produced yearling Brown Trout into the lower Klamath River at the township of Klamath Glen. This practice ended in 1976, when 12,600 yearling Brown Trout were released into the Trinity River at TRH. In that same year, 29,500 two-year old Brown Trout (2nd brood year 1975 fish) were released into Trinity Lake (Bedell 1977). However, CDFG discontinued the Brown Trout maintenance program because of low returns, small size, and lack of development and retention of anadromous characteristics in the Trinity River population (Bedell 1979).

In 2001, CDFG began stocking reproductively viable Brown Trout into Trinity Lake but this practice stopped in 2008 (M. Currier, personal communication 2017). Also in 2008, CDFG marked (adipose fin clip) and released 64,750 Brown Trout into Trinity Lake to determine if a portion of these fish survive migration through the turbines at Trinity Dam, immigrate into Lewiston Lake, and escape into the upper Trinity River through Lewiston Dam. Such movement could potentially have provided a continuous source of Brown Trout into the Trinity River, particularly during periods of low water levels in Trinity Lake, in combination with pulsed augmentation flows into the Trinity River. However, although this management action potentially could have artificially augmented annual counts of Brown Trout at Junction City Weir (JCW) between 2001 and 2008, no marked Brown Trout have been recorded in the Trinity River (M. Currier, personal communication 2017).

At the terminal end of anadromy in the upper Trinity River at Lewiston Dam, only three "wild" Brown Trout (all unmarked and not weir-tagged) have been recorded captured in annual TRH adult Salmonid returns since 1978 (one each in 1998, 2005, and 2014). Moreover, information on Brown Trout in the Klamath River appears to be extremely uncommon. For example, historically (1997-2017) there have been no Brown Trout verified by creel censuses conducted by CDFW from the mouth of the Klamath River to Weitchpec (S. Borok, Environmental Scientist, CDFW personal communication 2017). Additionally, 1,618 trap-days resulted in only 39 Brown Trout counted from 1989 to 2017 at the Willow Creek Weir, lower Trinity River (M. Kier, CDFW Environmental Scientist, personal communication 2017).

Methods and Materials

Study area.—Trinity River is located in northwestern California and is the largest tributary of the Klamath River (Figure 1). Construction of Trinity and Lewiston dams occurred in the early 1960s. Trinity Dam creates Trinity Lake (NAD 83, Zone 10N, UTM 519,964.7 m east and 4,516,719.7 m north), storing up to 2.45 million acre-feet of water (USFWS and HVT 1999). Lewiston Lake, formed by Lewiston Dam, is located 11.8 km downstream of

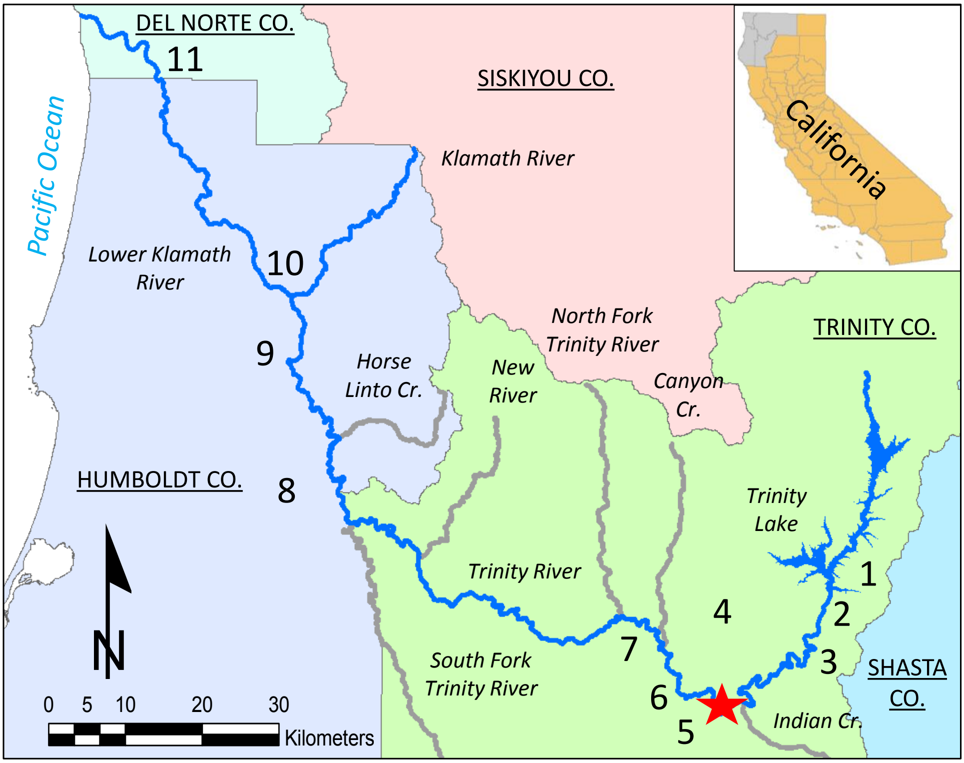

FIGURE 1.—Map of Lower Klamath River, Klamath River, Trinity River, and major tributaries of the upper Trinity River. Also included various other landmarks, including: 1) Trinity Dam, 2) Lewiston Lake, 3) Lewiston Dam and Trinity River Hatchery, 4) Weaverville, 5) Junction City Weir (red star), 6) Junction City, 7) Pear Tree Gulch, 8) Willow Cr., 9) Fort Gaston/Hoopa, 10) Weitchpec, and 11) Klamath Glen.

Trinity Dam (river kilometer [rkm] 180; UTM 517,489.4 m east and 4,508,408.4 m north), which serves as a re-regulating reservoir for flow to the Trinity River and diversion to the Sacramento River Basin, comprising the Trinity River Division of the Central Valley Project. Lewiston Dam is the uppermost limit of anadromous fisheries on the Trinity River. From Lewiston Dam, the Trinity River flows for approximately 180 kilometers before joining the Klamath River at the township of Weitchpec, California (UTM 440,575.2 m east and 4,559,590.2 m north). The Klamath River flows for an additional 70 rkm before entering the Pacific Ocean. The upper Trinity River is the stretch from the confluence of the North Fork Trinity River to 63.1 km upstream to Lewiston Dam. Trinity River Hatchery (TRH) is located immediately below Lewiston Dam.

Weir sampling.—Data presented herein derive from JCW, which is a Bertoni (Alaskan) style fish-tagging weir located 43.7 km downstream of Lewiston Dam. CDFW has operated JCW on an annual basis since 1978 and in cooperation with the Hoopa Valley Tribe since 1996. JCW functions to mark spring-run Chinook Salmon as part of an annual single mark-recapture estimate for the upper Trinity River above the weir. JCW also traps Coho Salmon, steelhead, Brown Trout, and Klamath Smallscale Sucker (*C. rimiculus*), but these species are considered "by-catch" by CDFW, as the primary target species was spring-run Chinook Salmon. Although annual sampling of Brown Trout began in 1982, lack of funding and administration mandate prevented data collection in 1983, 1984, 1992, and 1995. Prior to 1996, installation of JCW occurred when spring flows receded in June or July, and the weir was "fished" through December depending upon flow conditions. However, in 1996 a decision was made to truncate annual trapping efforts at the end of September, a procedure that continues to today. There are no trapping efficiency estimates for Brown Trout. Spring-run Chinook Salmon efficiency estimates at JCW vary from 26.6% of the annual run-size in 1992 (n =5,329) to 0.5% in 2012 (n = 35,326). Long-term average trapping efficiency for spring-run Chinook Salmon at JCW was 7.9% of the annual run size estimate. Same-season marked Brown Trout are not common, indicating that multiple captures of individual fish are rare.

Operation of JCW is a passive process, in which the weir is "fished" five days per week (Sunday evening - Friday afternoon). Trap-days start one half hour before sunset and end mid-day the following day, in order to exploit crepuscular behavior of the target species (spring-run Chinook Salmon), and capture both dusk and dawn migrating fish. JCW is open to both boat traffic and passage of migrating fish on a daily basis from mid-day to early evening, and on weekends. Limitations on scheduling are a function of safety, funding, and staffing. The term "fished" refers to blocking river to passage of adult fish except at a small opening at a pair of fyke panels spaced 11.4 cm apart inside a trap box, where the gap is located. The trap box consists of a cage immediately upstream of the weir, with a "V"-shaped opening (fyke) with wide end facing downstream that narrows towards the upstream interior of the trap box, where the gap is located. Upstream migrating fish swim through an 11.4 cm funneled gap in the fyke panels into the trap box, trapping adult fish. Staff check the trap box twice daily, once in the morning and again in the afternoon each trap-day, before opening the 4.9 m wide panel to recreational boat navigation. Unimpeded passage of fish occurs after the second trap check, and on weekends. Beginning in 2005, captured Brown Trout were measured, tagged (serial numbered T-bar [Floy tag]), and all salmonids evaluated for condition (i.e., evidence of predator wounds, gill net scars, and wounds by Klamath River Lamprey, etc.).

Study design.—The Trinity River Restoration Program (TRRP 2018), created by the Record of Decision (ROD) outlined a plan for restoration of the upper 63.1 km (mainstem) of the Trinity River and its fish and wildlife populations (TRFES 1999). The Trinity River Mainstem Fishery Restoration Environmental Impact Statement/Report was the document upon which the ROD was based (USDI 2000). TRRP restoration strategy included: 1) flow management through manipulation of the annual hydrograph, 2) mechanical channel rehabilitation, 3) sediment management, 4) watershed restoration, 5) infrastructure improvements, 6) adaptive environmental assessment and monitoring, and 7) environmental compliance and mitigation. Timing, extent, and volume of restoration flows appear in Appendix II. Information on the intended benefit of each ROD and Pulse flow hydrograph varies on an annual basis depending upon water availability and the particular restoration objective at the time of implementation (TRRP 2018).

To test the hypothesis of no significant difference in the annual pattern of timing of migration associated with managed hydrographs, we designated three flow year-types (henceforth called flow types): 1) baseline Pre-ROD flow (1982-2002), 2) ROD flow (2005-2011, 2017), and 3) Pulse flow (2003, 2004, 2012-2016). Pulsed augmentation flows were designed to cue migration of Chinook Salmon out of the Lower Klamath River to prevent risk of infection due to the ciliate parasite *Ichthyophthirius multifiliis*. Prior to 2003, there were no annually managed ROD or Pulse flows. Additionally, we note that each Pulse flow event was accompanied by a single ROD flow hydrograph (ROD plus Pulse flows), beginning in 2003. Thus, for each Pulse flow, effects of each pulsed augmentation are not completely separable or independent from effects of its companion ROD flow.

Since 2001, total restoration releases have included flows for: 1) restoration flows, 2) Tribal Ceremonial Boat Dance flows, and 3) pulsed augmentation flows. Ceremonial Tribal Boat Dance flows occur in odd years just prior to any pulsed flow augmentation. However, because they only amount to 0.6% of the total release into the Trinity River (TRRP 2018), we did not include them in our analysis, even though pulse flows occasionally tier off the trailing ends of ceremonial flows. Shapes of the ascending limbs of the hydrographs were mostly rapid (19/22) with few years in which there were benches (7/22), all of which were associated with managed flows. In contrast, shapes of the descending limbs of the hydrographs were all gradual with numerous benches associated with virtually all managed flows (14/22). Benches in hydrographs included stabilization of water release for approximately one or more days. There were two double peaked ROD flows (2016 and 2017). All Pulse flows had rapid ascending hydrographs and at least one bench. Similarly, all descending limbs were rapid with at least one bench. Spring-summer base flows historically equate to 13 m^3/s.

ROD flows occurred annually from late April to August. Conjoining Pulse flows occurred from August to September. Actual magnitude and duration of ROD and Pulse flows varied in hydrologic characteristics, cubic meters per second (m^3/s), shape of the hydrograph, and duration of the hydrograph depending upon the specific management intent. Average duration of ROD flows was about 89.8 days (range 62.0-112.0 days) from mid-April to early August, and averaged 221.9 m^3/s (range 124.9-328.6 m^3/s) of flow at the top end of the hydrograph. Average duration of Pulse flows was about 28.3 days (range 11.0-40.0 days) from mid-August to late September, and averaged 61.1 m^3/s (range 35.3-97.0 m^3/s) of flow at the top end of the hydrograph. For the same general monthly period, average duration of baseline Pre-ROD flows was about 52.4 days (range 28.0-81.0 days) from late April to late July, and averaged 119.6 m^3/s (range 62.3-192.3 m^3/s) of flow at the top of the hydrograph. Water summary data and a typical flow release diagram (hydrograph) tiered to water-year type are available at the TRRP website (TRRP 2018). We obtained digital and printed hydrographic data from the US Bureau of Reclamation (USBR) Lewiston Water Quality Gauge (LWS) in the upper Trinity River (rkm 178.2 at Lewiston Dam) downloaded from the California Department of Water Resources, California Data Exchange Center (DWR 2017).

Statistical analysis.—We used catch per unit effort (CPUE) in units of adult fish trapped (caught) per trap-day (effort) to estimate relative annual abundance and evaluate "population" trends over time. Brown Trout were considered adults if they were at least 32 centimeters in fork length (one-year-old fish). Although CPUE is not a measure of true abundance, it is an established indicator of relative abundance (Bonar et al. 2009). Estimates of CPUE derive from by-catch data collected at JCW for Brown Trout (1982-2017). A test of the hypothesis that the annual distribution of CPUE was derived from a normally distributed population was rejected (Shapiro-Wilk test (W) = 0.88, P <0.01, n = 32; McDonald 2014). Because annual estimates of CPUE were skewed significantly to the right, they were ranked, visually inspected by use of normalized (0.0, 1.0) quantile-quantile (Q-Q) plots (R Core Team 2013), and found to be normally distributed (W = 0.96, P = 0.23, n = 32). Thus, all subsequent statistical analyses of count data used non-parametric methods (McDonald 2014). Because of small annual sample size, we used the Spearman rank correlation (r_s) to assess evidence of trends in parasitism (wounding) by Klamath River Lamprey.

We analyzed trends in seasonal data by use of Julian weeks (JW), defined as one of seven consecutive-day-sets of 52 weekly periods in a calendar year, beginning 01 January of each year. This procedure allowed inter-annual comparisons of identical weekly periods. The extra day in leap years was included in the ninth week. Wilcoxon signed-rank test, computed from an approximate normal variate (Z) using non-zero data, evaluated the hypothesis that the median difference between pairs of JW was zero among different flows (Hasler et al. 2014). To determine if timing of seasonal migration in ROD and Pulse flows deviated from the baseline Pre-ROD flow, we calculated a Percent Deviation Index (PDI) from total trap counts:

PDI ROD flow = %ROD flow count – %Pre-ROD flow count

PDI Pulse flow = %Pulse flow count – %Pre-ROD flow count

To evaluate the specific timing of migration, we tested the hypothesis that counts of Brown Trout captured during individual JW were not significantly different between Pre-ROD, ROD, or Pulse flow types (years 1982-2017, JW21-JW49). We attempted to standardize sampling effort by including in our analysis only those pairwise comparisons that had a sample size ≥ 5 for each flow type. Pairwise comparisons of non-zero counts using JW as attributes were then evaluated using the nonparametric Dwass-Steel-Chritchlow-Fligner (*DSCF*) test (Critchlow and Fligner 1991).

We used Robust Regression (ROBREG) analysis to test the hypothesis that the population of Brown Trout in the upper Trinity River has increased and continues to proliferate since 2000 (SYSTAT 2009 and references therein). We conducted all regressions on ranked counts, used the Least Trimmed Squares (LTS) method and FAST-LTS algorithm and the weighted median to compute estimates of regression coefficients in determining adequacy of the model to generate a robust regression estimator (Rousseeuw and Leroy 1987, Huber and Ronchetti 2009). This method uses ranks of residuals instead of observed residuals, has few distributional assumptions, and is useful in detecting and deleting outliers in both the Y-space and X-space prior to performing ordinary least-squares regression on outlier-free data. Robust regression statistics and plots included the least-squares regression correlation coefficient (R) on outlier-free data, adjusted and robust coefficients of determination (R^2) to assess adequacy of the model (Rousseeuw and Van Driessen 2000, Maronna et al. 2006), and 95% confidence intervals surrounding the regression line. We accepted statistical significance at $P \leq 0.05$ (McDonald 2014).

Results

Annual variation in trap counts.—The relationship between: 1) total days the JCW was in place and 2) total days the weir was fished was positive and highly significant (R = 0.98, Robust R^2 = 0.98, Adj. R^2 = 0.96, F = 752.0, P <0.01, $d.f._{1,30}$; Figure 2A). Whereas, the relationship between year and these two variables was both negative and significant from 1982 to 2017 (Figure 2B and 2C). Average days of operation for this period was 72.5 trap days (range = 15 [2012] - 139 [1991], with the largest number of trap days associated with sampling from 1982 to 1994 (average = 104.3 trap days). In contrast, after truncating sampling at the end of September in 1995, the relationship between year and number of days JCW was fished, although negative, was not significant. Thus, except for 2005 when JCW was fished through October, sampling effort was relatively consistent from 1996 to 2017

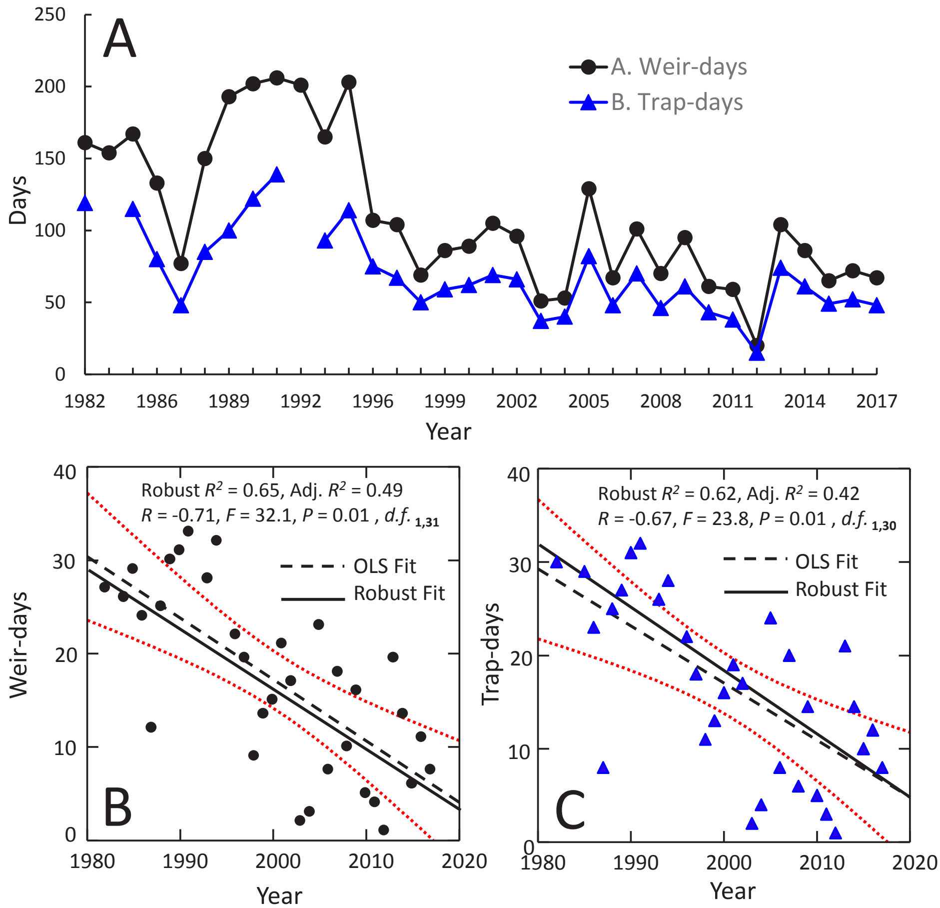

FIGURE 2.—A) Annual variation in total days Junction City Weir was in placed and total days the weir was fished. B) Relationship between total days the weir was in place versus total days weir was fished. OLS Fit = Ordinary Least Squares regression for outlier free data, Robust Fit = Least Trimmed Squares regression, and 95% confidence intervals on regression line.

(average = 55.1 trap days). Beginning in 2003, however, weir operations were temporarily and routinely halted in ROD and Pulse flow years until flows in those years subsided sufficiently to reinstate JCW (average = 50.9 trap days).

Annual variation in CPUE for Brown Trout exhibited cyclic fluctuations approximately every four to six years (Figure 3A, Table 1). These fluctuations were relatively muted between 1982 and 2002, but CPUE increased beginning in 2003, peaked in 2004, and was followed by a sharp decline through 2017. Regression analysis showed a significant and positive relationship between CPUE and year for the sampling period 1982 to 2017 (Figure 3B). However, for the sampling period 2003 to 2017, the relationship between year and CPUE was significantly negative (Figure 3C), indicating that Brown Trout have declined dramatically in relative abundance since 2003.

Effects of in-river parasitism by Klamath River Lamprey.—Combined data for all species analyzed herein, showed that the largest number of annual observations of adult fish

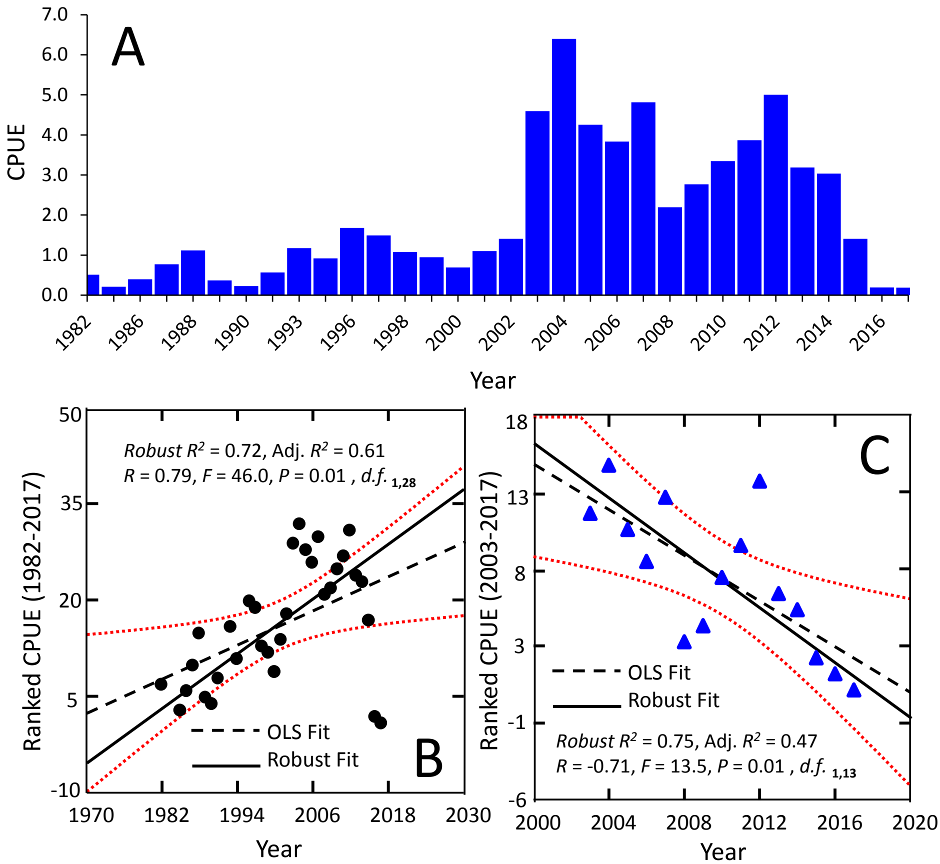

FIGURE 3.—Annual variation in catch per unit effort (CPUE) of total counts of Brown Trout (n = 3,596) from the Junction City Weir for years: A) 1984 to 2017 and B) 2003 to 2017, which included all ROD and Pulse flows over the last 15 years. OLS Fit = Ordinary Least Squares regression for outlier free data, Robust Fit = Least Trimmed Squares regression, and 95% confidence intervals on regression line.

trapped at JCW with fresh circular Klamath River Lamprey wounds (2.0 - 3.0 cm diameter) on their lateral surfaces occurred in 2015, 2016, and 2017 (Table 2). The percentage of all species of live adult fish with fresh lamprey wounds was significant and positively correlated with year (r_s = 0.67, P <0.05, n = 8). This apparent increased trend of visible lamprey wounds on adult fish was significant for Chinook Salmon (r_s = 0.71, P <0.05, n = 8) and Brown Trout (r_s = 0.69, P <0.05, n = 8), but not for steelhead (r_s = 0.61, P > 0.05, n = 8). For Brown Trout and Chinook Salmon the largest percentage of adult fish with lamprey wounds occurred in 2015, 2016, and 2017, and for adult steelhead the largest percentages occurred in 2012, 2015, and 2016. Additionally, we note that adult Coho Salmon generally occur in the upper Trinity River no earlier than late September after wier operations cease for the season, which precludes observations of lamprey wounds for this taxon.

Further, although the percentage of non-weir tagged mortalities that drifted downriver and impinged upon the panels of weir for all adult species combined showed no significantly correlated with year (r_s = -0.14, P >0.05, n = 8); this relationship was significant and positive

Table 1.—Annual and Julian week sample data for Brown Trout from 1982 to 2017; and a summary of non-weir tagged mortalities that washed onto the weir from up river. Data for 1983, 1984, 1992, and 1995 are missing (na = no data) because of lack of funding for Junction City Weir; including total days the weir was in place (Weir-days) and total days the weir was fished (Trap-days).

Year	Weir-days	Trap-days	Count	Non-weir tagged mortality	Julian week	Count
1982	161	119	61	na	21	3
1984	154	na	na	na	22	8
1985	167	115	24	na	23	19
1986	133	80	32	na	24	104
1987	77	48	37	na	25	249
1988	150	85	95	na	26	280
1989	193	100	37	na	27	303
1990	202	122	28	na	28	360
1991	206	139	79	na	29	265
1992	201	na	na	na	30	480
1993	165	93	109	na	31	481
1994	203	114	105	na	32	368
1996	107	75	126	0	33	135
1997	104	67	100	0	34	71
1998	69	50	54	0	35	22
1999	86	59	56	0	36	36
2000	89	62	43	0	37	35
2001	105	69	76	0	38	63
2002	96	66	93	0	39	118
2003	51	37	170	0	40	62
2004	53	40	256	0	41	29
2005	129	82	349	4	42	41
2006	67	48	184	1	43	30
2007	101	70	337	na	44	9
2008	70	46	101	0	45	15
2009	95	61	169	0	46	4
2010	61	43	144	0	47	1
2011	59	38	147	0	48	1
2012	20	15	75	0	49	4
2013	104	74	236	5		
2014	86	61	185	9		
2015	65	49	69	48		
2016	72	52	10	4		
2017	67	48	9	1		

for Brown Trout (r_s = -0.88, P <0.01, n = 8). However, the historically elevated counts of non-weir tagged mortalities in adult Brown Trout observed in 2015 were coincident with a tagging study initiated in 2013 (Table 1, CDFW 2014, USBR 2014), and included three individual radio tagged and one anchor-style tagged fish (02 July 2015). As such, the relationship between the percentage of non-weir tagged mortality and wounding by lamprey was significant and positive for Brown Trout (r_s = 0.76, P <0.05, n = 8), but not for any other species ($r_s \leq$ -0.50, P >0.05, n = 8).

Seasonal variation in trap counts.—Relative abundance of Brown Trout fluctuated weekly, beginning in late May (JW21), and continued through mid-December (JW49, Figure 4). Brown Trout occurred most frequently in the upper Trinity River from late June (JW25) through mid-August (JW33), with a primary peak in late July (JW30) and early August (JW31), declining abruptly through early September (JW35), with very few fish lingering in the area through mid-December (JW49). The relationship between percent seasonal trap counts of Brown Trout and JW exhibited a significant negative trend (Figure 4), with the percent count decreasing (negative trend) over the total season but increased in the early part of the season, declining late in the season.

Deviation in timing of migration from baseline flow type.—Total counts for each flow type were: 1) baseline Pre-ROD flow = 1,155; 2) ROD flow = 1,001; and 3) Pulse flow = 1,440 from 1982 to 2017 (n = 3,596; Figure 5A and B). A positive or negative PDI (Y-axis) signaled deviation from the baseline Pre-ROD flow pattern in timing of migration, by addition or subtraction of fish along the X-axis (JW) in ROD and Pulse flows (Figure 5C and D). Deviation away from the baseline occurred through: 1) reduction in counts of fish at the ascending limb, and 2) addition of fish along the declining central segment and trailing end of migration. A Wilcoxon signed-ranks test showed a significant overall difference between baseline Pre-ROD and ROD flows (Z = 2.0, P = 0.05, n = 29), ROD and Pulse flows (Z = 2.5, P = 0.01, n = 29), but not baseline Pre-ROD and Pulse flows (Z = 0.37, P = 0.71, n = 29).

Further, of 29 JW sampled, 37.9% (n = 11) had sample sizes $\geq$5 for each flow type (Table 3). Of these, eight showed significant differences among flow groups. For example, 87.5% differed significantly between baseline Pre-ROD and ROD flows; 62.5% differed significantly between baseline Pre-ROD and Pulse flows; but there were no significant differences between ROD and Pulse flows. Total counts of Brown Trout that encompassed all deviations away from the baseline Pre-ROD flow pattern of migration (positive plus negative counts), ranged from 488 fish (ROD flows) to 775 fish (Pulse flows; Table 4). Hence, the combined influence of both ROD and Pulse flow hydrographs post-2003 affected 1,263 Brown Trout relative to the baseline Pre-ROD flow pattern in timing of migration. The relationship between total counts of ROD and Pulse flow-affected fish was both significant and positive (Figure 6). Thus, as the count difference of ROD flows to baseline flows increases so does the count difference of Pulse flows to baseline flows.

Additionally, of all Brown Trout affected by ROD and Pulse flows, 59.1% encompassed JW28 through JW32. As indicated in Appendix II, implementation of ROD flows occurred from early April (JW17) through early August (JW32), whereas Pulse flows occurred from mid-August (JW35) to late September (JW39). Thus, in Brown Trout, alteration in the baseline Pre-ROD flow pattern of migration appeared to be most affected by the descending limbs of ROD flows, especially in wet years with hydrographs that have relatively long descending limbs.

TABLE 2.—Percent frequency of fish with visible Klamath River Lamprey wounds and number of mortalities counted for Brown Trout and anadromous steelhead and Chinook Salmon, co-occurring in the Trinity River, and trapped at Junction City Weir.

Year	All species (%)			Chinook Salmon (%)			Steelhead (%)			Brown Trout (%)		
	n	Wounds	Mortality	*n*	Wounds	Mortality	*n*	Wounds	Mortality	*n*	Wounds	Mortality
2010	387	14.2	2.1	222	10.8	3.6	21	0.0	0.0	144	21.5	0.0
2011	449	4.9	5.6	247	2.8	9.7	55	0.0	1.8	147	10.2	0.0
2012	274	11.7	1.8	189	4.8	2.6	10	20.0	0.0	75	28.0	0.0
2013	1155	8.0	1.1	835	4.4	1.0	84	3.6	0.0	236	22.0	2.1
2014	1246	9.0	1.6	1028	7.1	1.1	33	9.1	0.0	185	19.5	4.9
2015	468	35.8	13.9	343	28.9	5.0	56	25.0	0.0	69	78.3	69.6
2016	227	42.9	3.1	154	50.6	1.9	63	19.0	0.0	10	70.0	40.0
2017	269	41.3	1.5	208	48.6	1.4	52	9.6	0.0	9	55.6	11.0

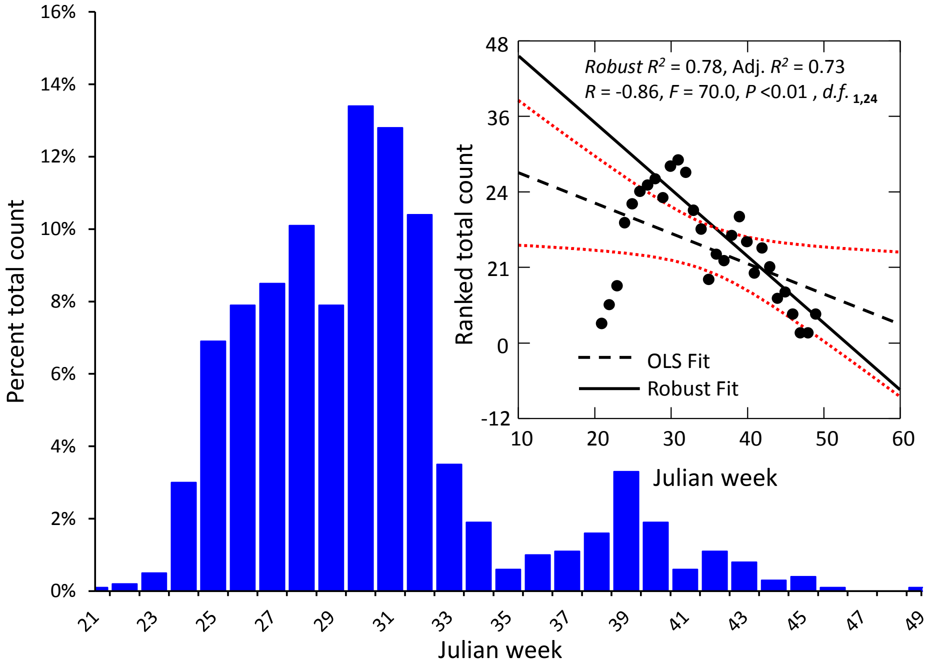

FIGURE 4.—Seasonal variation in percent total counts (*n* = 3,596) by Julian week (JW) of Brown Trout from Junction City Weir (years 1982-2017 and JW21-JW49). OLS Fit = Ordinary Least Squares regression for outlier free data, Robust Fit = Least Trimmed Squares regression, and 95% confidence intervals on regression line.

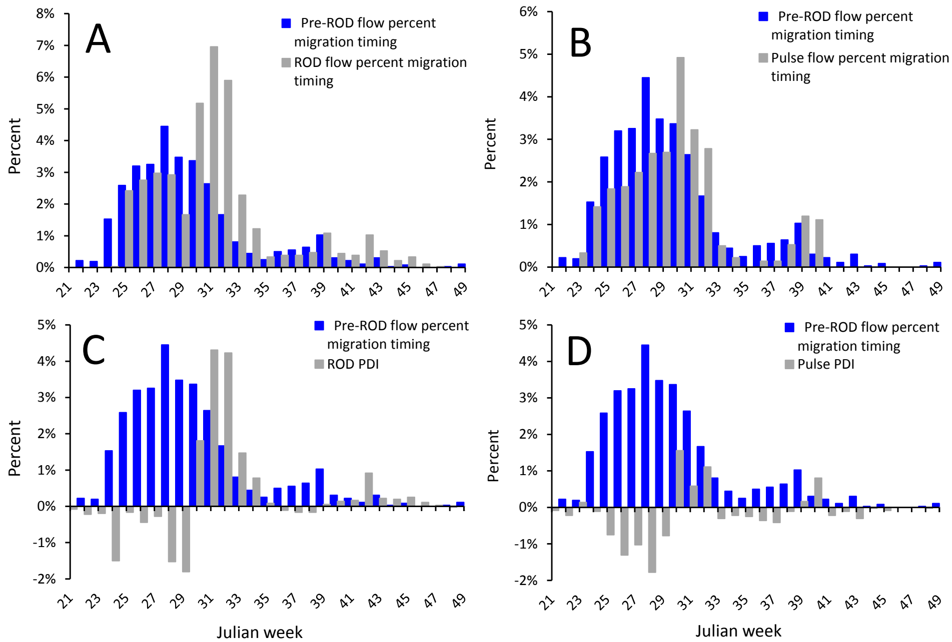

Figure 5.—Percent total count by Julian week (JW) of Brown Trout for baseline Pre-ROD flow (*n* = 1,155) in relation to: A) ROD flow (*n* = 1,001) and B) Pulse flow (*n* = 1,440; years 1982-2017 and JW21-JW49). Percent total count by JW of Brown Trout baseline Pre-ROD flow relative to the Percent Deviation Index (PDI) for: C) ROD and D) Pulse flows.

Discussion

Annual and seasonal variation in estimates of relative abundance.—Although the overall pattern of annual variation in Brown Trout CPUE showed a significant increase in relative abundance from 1982 to 2017, we show that counts of Brown Trout have decreased significantly from 2003 to 2017. These results deviate dramatically from the hypothesis that the population of Brown Trout in the Trinity River has increased in number and continues to proliferate since 2000. This sequence of dates coincides with establishment of the Trinity River Restoration Program in 2002, and subsequent "ROD flows" in combination with periodic Pulse Flows beginning in 2003. As such, we do not interpret any increased "trend" as a reflection of an increase in relative abundance (population size) of Brown Trout in the upper Trinity River in recent times. Not only did the increase in Brown Trout actually start in 2003, but the magnitude of change in CPUE from 1.4 (2002) to 6.4 (2004) is explained more parsimoniously as an extreme migratory response by Brown Trout already in the Trinity River system coincident with managed flow regimes initiated in 2003 by the USBR and TRRP. It is not possible to attribute an increase in "size" of the Brown Trout population based on reproductive output, relative to the baseline Pre-ROD Flow trap count, given the timeline and extent of sampling that occurred between 2002 and 2004. This means that the peak in Brown Trout CPUE at JCW beginning in 2003 was not an indication of an increase in population size, but rather an indication of a secondary behavioral response to managed flows and the altered environmental conditions that ensued.

Table 3.—Dwass-Steel-Chritchlow-Fligner (*DSCF*) statistical tests for pairwise comparisons of ranked non-zero total counts of Brown Trout by Julian week. Only those non-zero pairwise comparisons that had a sample size ≥5 for each flow type were included in our analysis; and only comparisons that were significant ($P \leq 0.05$) were included in the table.

Julian week	Flow group(*i*)	*n*	Flow group(*j*)	*n*	*DSCF* statistic	*P*-value
29	Pre-ROD	15	ROD	5	11.8	0.00
	Pre-ROD	15	Pulse	5	12.6	0.00
30	Pre-ROD	15	ROD	7	6.2	0.00
	Pre-ROD	15	Pulse	6	7.2	0.00
31	Pre-ROD	13	Pulse	9	6.6	0.00
	Pre-ROD	13	Pulse	6	8.3	0.00
32	Pre-ROD	12	Pulse	6	9.1	0.00
	Pre-ROD	12	Pulse	6	12	0.00
33	Pre-ROD	8	Pulse	5	3.3	0.05
34	Pre-ROD	7	ROD	5	5.8	0.00
35	Pre-ROD	9	ROD	7	5.9	0.01
39	Pre-ROD	11	ROD	8	8.1	0.00

Although we focused specifically on the potential effects on a river system subjected to highly managed flow regimes and geomorphological restoration of the mainstem, other covariates besides hydrology and geomorphology affect annual and seasonal patterns of relative abundance and timing of migration in salmonids. For example, factors responsible for decreasing stocks of anadromous salmonids in both Trinity and Klamath rivers reference recent ocean conditions and drought (Dettinger and Cayan 2014, Diffenbaugh et al. 2015, and Mann and Gleick 2015). Since 2001, 38.9% of regional water-years had "dry" or "critically dry" designations, including two periods of three consecutive dry water-years (2007-2009 and 2013-2015.

Moreover, CPUE estimates of Brown Trout relative abundance in 2015, 2016, and 2017 are consistent with estimates of abundance observed pre-2003. This decrease post-2003 also likely reflects in part, the historically unprecedented level of non-weir tagged Brown Trout mortalities observed in 2015 that drifted downriver and impinged upon weir panels beginning in 2015 relative to any other previous year (see Table 1). In our analysis, a potential complicating factor in determining population trends of Brown Trout included documentation of non-weir tagged mortalities, which may be associated with in-river wounding by Klamath River Lamprey that parasitize adult salmonids. Brown Trout in the upper Trinity River spend their entire life cycle in the river, which likely subjects them to a higher risk of in-river Klamath River Lamprey parasitism compared to other sympatric salmonids. Although wounds from Klamath River Lamprey parasitism may contribute to a decline in Brown Trout abundance in the upper Trinity River (Alvarez 2017), we found no evidence to suggest a strong relationship between wounding by Klamath River Lamprey and high

Table 4.—Total counts (positive and negative) and total cumulative counts (positive plus negative) by Julian week of Brown Trout affected by ROD and Pulse flows, relative to the pattern in timing of migration representative of the baseline Pre-ROD flow (1982-2017 and JW21-JW49).

				1st Release group			2nd Release group		
Brood year	Adults	Females spawned	Eggs taken	Date	Number released	Location	Date	Released	Location
1963	34	22	58,100	23-Apr-65	28,500 (UM)	Trinity River Hatchery			
1964	145	64	163,600	18-Apr-66	28,900 (AD)	Trinity River Hatchery			
1965	100	39	109,900	1-Nov-67	35,200 (LV)	Hawkin's Bar			
1966	152	62	75,600	1-Oct-68	41,514 (UM)	Hawkin's Bar			
1967	231	62	158,700	na	na	na			
1968	170	32	76,644	Oct-70/Nov-70	18,967 (RV, LV)	China Slide (LV)	Nov-70	10841 (RV)	Klamath Glen
1969	70	26	53,000	29-Mar-71	23,370 (UM)	Lime Point	Nov-72	2321 (UM)	Trinity River Hatchery
1970	23	2	na	na	na	Trinity River Hatchery			
1971	7	34	140793	17-Apr-73	30,220 (UM)	Trinity River Hatchery			
1972	111	43	73,000	8-Aug-74	23,030 (UM)	Old Lewiston Bridge			
1973	39	26	55,227	14-Apr-75	17,820 (UM)	Old Weir Site			
1974	32	7	13,200	23-Apr-76	73,60 (UM)	Trinity River Hatchery			
1975	24	9	17,272	1-Apr-77	12,600 (AD)	Trinity River Hatchery			
1976	49	28	52,000	5-May-77	29,500 (UM)	Trinity Lake			

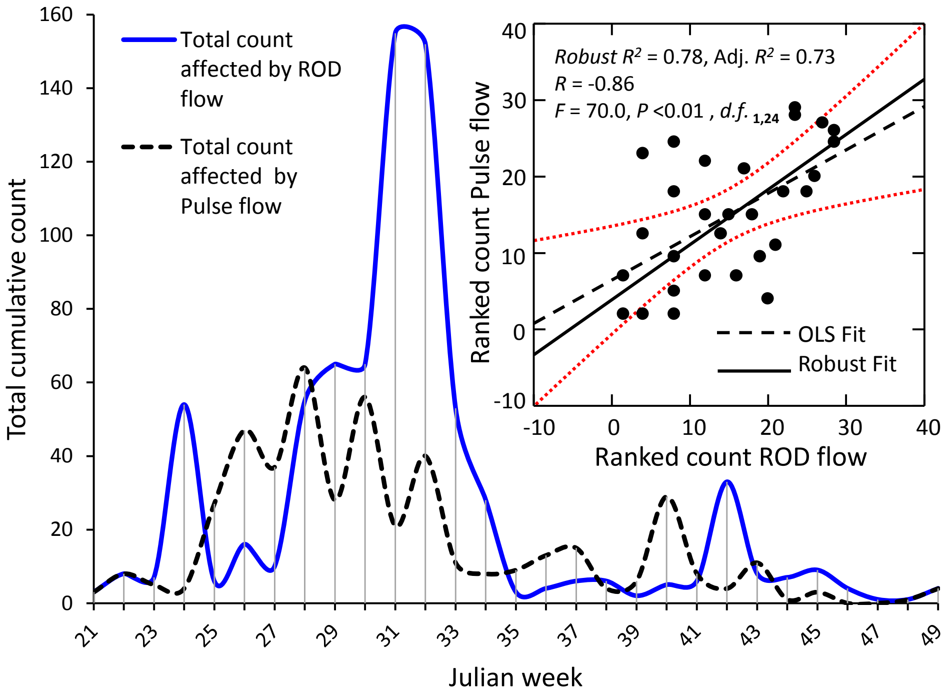

FIGURE 6.—Total cumulative count (positive plus negative) and Robust Regression of Brown Trout affected by both ROD and Pulse flows relative to baseline Pre-ROD flow (years 1982-2017 and JW21-JW49). OLS Fit = Ordinary Least Squares regression for outlier free data, Robust Fit = Least Trimmed Squares regression, and 95% confidence intervals on regression line.

levels of mortality in Brown Trout, relative to co-occurring anadromous salmonids. Instead, our data showed that although all three sympatric species of salmonids may exhibit elevated levels of lamprey wounds, mortalities in these species did not increase proportionately. Without additional information demonstrating a significant relationship between wounding by Klamath River Lamprey and subsequent mortality, we are unable to determine if parasitism by Klamath River Lamprey is a major factor contributing to fluctuations in populations of non-anadromous or anadromous species of fish in the upper Trinity River.

Potamodromous migration in Brown Trout.—Anadromy is a life history strategy in which adult fish migrate from saltwater to an upstream body of flowing fresh water (river or stream) to spawn (Moyle 2004). In contrast, a potamodromous life history refers to fish whose migrations occur wholly within fresh water (Maki-Petays et al. 1997). There have been no comprehensive tagging studies of Brown Trout in the Trinity River to suggest that Brown Trout are not anadromous. However, our analysis suggests that Brown Trout in the upper Trinity River are best described as a potamodromous population, born in upstream freshwater habitats, migrating downstream as juveniles (but still in freshwater), and growing into adults before migrating back upstream to spawn. Meyers et al. (1992) found that seasonal movements in Brown Trout may range from 7.2 to 20.1 km during spring and fall, but were relatively sedentary at other times (Burrell et al. 2000). Rapidly fluctuating flow conditions are critical factors with which spawning Brown Trout below peaking hydroelectric dams must contend. Indeed, Heggenes et al. (2007) found that Brown Trout appeared to move more when high flows continued for longer durations.

From 2012 to 2017, there were five consecutive ROD and Pulse flows in the upper Trinity River. Other studies have hypothesized that varying water flow may induce longer movements when Brown Trout are predisposed to move (e.g. spawning movements). Ovidio et al. (1998) and Young et al. (2010) reported that varying flow, in conjunction with varying temperature, triggered movements to the spawning areas for Brown Trout. Both Clapp and Clark (1990) and Brown et al. (2001) found a correlation between water flow and longer movements in relatively large Brown Trout. In contrast, Bunt et al. (1999) reported no effects of pulsed flows on Brown Trout movements within their study site; however, pulsed flows were regular on a diurnal basis, which may have allowed fish to adapt or acclimate behaviorally to recurrent pulsed flow augmentation.

Our hypothesis that Brown Trout populations are responding behaviorally to managed flow regimes is consistent with the recent suggestion that the magnitude and duration of flows are more important than quality of additional pulsed cold water intended to stimulate fish to move for prevention of disease (Strange 2010, USBR 2016). For example, we show that timing of migration in the baseline Pre-ROD Flow of Brown Trout from 1994 to 2017 has changed in response to both ROD and Pulse flows, both separately and in combination, since 2003. If variation in Brown Trout CPUE is a behavioral response to ROD and Pulse flows, this likely implies that these flows enable Brown Trout to occupy downriver habitats for a longer period relative to baseline conditions. Potentially well beyond the duration that juvenile salmon and steelhead out-migrate. Salmon fry typically emerge from the gravel around mid-February and out-migrate from March through June.

Additionally, ROD flows in conjunction with Pulse flows may facilitate prolonged opportunities for feeding on out-migrating juvenile salmonids. This condition would constitute “prey switching” by Brown Trout as a function of frequency-dependent predation associated with release of approximately 3- to 5-million fingerlings annually by the TRH (Larry Glenn, CDFW TRH Manager, personal communication 2016). Further, anecdotal information from local anglers suggest that Brown Trout follow spawning Klamath Smallscale Suckers to feed on sucker roe during the early summer in the upper Trinity River. Empirical evidence does suggest that Klamath Smallscale Suckers do spawn in the early summer in the upper Trinity River as exemplified by capture of gravid female suckers on June 30, 2009 (JW26) at JCW.

Seasonal variation in migration in relation to flow type.—Seasonal variation in trap counts of Brown Trout showed significant differences in the timing of migration between baseline Pre-ROD, ROD, and Pulse flows. There also was a significant difference between ROD and Pulse flows, suggesting that pulsed augmentation flows may represent an important additional and independent factor affecting the timing of migration, relative to a ROD Flow hydrograph. Deviation away from the baseline Pre-ROD migration pattern occurred through reduction in counts of fish early to mid-season and an increase in counts late in the season, which displaced the actual timing of migration in post-2003 flows to later in the season. That both ROD and Pulse flows have altered the timing of migration, relative to the baseline Pre-ROD condition, fails to support the hypothesis of no significant difference in the timing of migration of Brown Trout in relation to annually managed flow regimes.

Peterson et al. (2017) used a variety of environmental attributes to assess the relative influence of managed pulse flows to explain the magnitude of daily counts and proportions of fall-run Chinook Salmon observed at a weir on the Stanislaus River, California. They concluded that, although managed pulse flows resulted in immediate increases in daily passages, the measured response was brief, representing only a small portion of the total run relative to a stronger response between migratory activity and discharge levels. As relates to the upper Trinity River, we interpret these observations to be more reflective of the effects of implementing annual ROD flow hydrographs as opposed to short-term pulsed flow augmentations. The effects of managed flow on the timing of adult migration clearly needs further investigation in relation to the potential measured impacts of flow management, as well as other physical and biological covariates, prior to implementing any actions that actively suppress adult Brown Trout in the upper Trinity River.

Although we show that Brown Trout responded behaviorally on an annual and seasonal basis to flow augmentation, we lack reproductive data for Brown Trout to test an additional hypothesis that managed flow regimes likely affect multiple brood-year cycles post-2003 if ROD and Pulse flows continue. Flow-related impacts to multiple brood-year cycles likely have even greater implications for co-occurring anadromous species of salmonids inhabiting the Trinity River, particularly those that overlap in the pattern of run-timing, most notably spring-run and fall-run Chinook Salmon. Currently these issues have not been part of the long-term effects analysis to protect adult anadromous salmon in the Lower Klamath River, even though flows designed to facilitate such protection originate in the upper Trinity River (USBR 2016). As of 25 July 2016, there was no plan to address these issues for any salmonid in the upper Trinity River or as part of any proposed environmental impact assessment (Mary Paasch, USBR, personal communication 25 July 2016).

How viable is the competition scenario?—Brown Trout predation on native populations of salmonids and use of suitable habitat within the upper Trinity River, has resulted in criticism that there is significant competition between Brown Trout and native anadromous salmonids for limited food, space, and cover (McHugh and Budy 2006, Naman 2008, Waters 1983, Wang and White 1994, Alvarez 2017). However, documenting interspecific competition in nature is equivocal at best and only potentially possible where the combined demand for a resource is in excess of the supply (Larson and Moore 1985, Fausch 1988, Lohr and West 1992, Brewer 1994, Meffe and Carroll 1994, Blanchet et al. 2007). Additionally, documenting competition is particularly problematic in a large riverine system continuously subjected to variation in hydrology, temperature of water, and in-river restoration associated with floodplain reconstruction.

Several studies show that adult steelhead and Coho Salmon consume hatchery and naturally produced salmonid fry or smolts (Ruggergone 1989, Ruggergone and Rogers 1992, McConnaughey 1999, Pearsons and Fritts 1999, Naman and Sharpe 2012). Naman (2008) stated that release of large numbers of hatchery steelhead from the TRH could result in substantial counts of Chinook Salmon and Coho Salmon fry being consumed even with relatively low predation rates (i.e., 25,000 fry per day equating to approximately 9.0% of all Chinook Salmon and Coho Salmon fry produced). Studies in other river systems concluded that Brown Trout were superior competitors to sympatric Brook Trout (*Salvelinus fontinalis,* Fausch and White 1981, Blanchet et al. 2007, Korsu et al. 2010). Whereas, several investigations suggest that co-occurring piscivorous species were a superior pairwise competitor relative to Brown Trout (Fausch and White 1986, Magoulick and Wilzbach 1998, Strange and Habera 1998). Additionally, McKenna et al. (2013) found evidence for a decline of Brook Trout in the presence of Brown Trout across many watersheds. Yet a model of the relationship between Brook Trout and Brown Trout abundance explained less than 1% of the variation documented; and ordination showed extensive overlap in habitat used by these two taxa, with only small components of the “hypervolume“ (multidimensional space) being distinctive (McKenna et al. 2013).

The relative importance of competition and predation also changes with life stage and seasonal availability of different prey items (Jonsson et al. 1999, L’abee-Lund et al. 2002, Browne and Rasmussen 2009). In controlled laboratory stream experiments, Coho Salmon dominated Brook Trout and Brown Trout of equal size, and Brook Trout dominated equal-size Brown Trout. However, when released from competition, subordinate species shifted to positions that were more resource profitable (Fausch and White 1986). Further, laboratory growth rates of Coho Salmon equaled rates measured in tributaries, whereas both Brook Trout and Brown Trout grew more slowly in the laboratory than in the field as a result of intraspecific competition due to lack of cover affording visual isolation (Fausch and White 1986). These results suggest that larger size and competitive superiority of Coho Salmon give them an advantage over juvenile Brook Trout and Brown Trout in tributaries when resources are limiting.

Based on recent dietary and bioenergetics analyses, Alvarez (2017) concluded that predation by Brown Trout poses a potential impediment to recovery of native salmonids in the Trinity River. However, the comparative impact of predation by co-occurring anadromous salmonids, as well as terrestrial *piscivorous* predators, on juvenile salmonids in the Trinity River is unknown relative to Brown Trout. Without comparative and simultaneous equal sampling effort, co-occurring species of adult salmonids, individually or in combination, could be a far bigger problem that Brown Trout, in relation to the overall impacts to survival of juvenile fish. Therefore, we maintain that, without such comparative information on both aquatic and terrestrial fish predators, it is premature to advocate or implement any comprehensive management strategy that would systematically remove Brown Trout from the Trinity River.

Economic impact of the Trinity River Brown Trout sport fishery.—Flow regimes managed annually in combination with massive programs of habitat restoration in the upper Trinity River have contributed to a substantial recreational fishery for Brown Trout, particularly among fly anglers. This industry brings intrinsic value and economic stimulus to the local community. The Trinity River Brown Trout fishery is unique in that, unlike fisheries in other regions of California, and on the West Coast, the Trinity River offers an opportunity to catch both trophy steelhead and Brown Trout. Commercial sport fishing guides operate under special recreation permits issued by the US Bureau of Land Management, which issue 100 guide permits for the Trinity River on a first-come, first-serve basis. Commercial fishing guides charge upward of $450 per day to fish steelhead and Brown Trout on the upper Trinity River. They typically book clients 4 days per week for 15 weeks (October-January), yielding an estimated $27,000 generated per guide annually (Bill Dickens, former President of the Trinity River Guides Association, personal communication 2017). If half of the commercial fly-fishing guides book clients at this rate, the conservative estimated income generated from commercial guide fees is approximately $1,350,000 annually, which does not include revenues benefitting local hotels, restaurants, businesses, and the community as a whole through tax-generated revenues. In theory, any financial loss from a managed fishery that seeks to remove Brown Trout could potentially benefit by economic opportunities derived from increased numbers of native salmonids. However, the validity of this premise remains untested in practice for the upper Trinity River.

Management recommendations.—Several factors are important in determining whether programmatic removal of Brown Trout from the Trinity River is necessary and has the potential to be successful. First, is the consideration of whether removal of Brown Trout is required for enhancing populations of target species. Fetherman et al. (2015) found that Brown Trout removal did not dramatically affect survival or emigration from the study site of sympatric salmonids. Second, it is important to consider whether removal will be successful after one removal effort, or are multiple removal efforts needed to overcome biotic resistance. A single removal of 66% of the Brown Trout population in the Au Sable River in Michigan did not result in population or size at age increases within sympatric Brook Trout populations (Shetter and Alexander 1970). Third, focus should be placed on whether environmental resistance factors, such as temperature, flow, and abiotic resources, may prevent successful removal (Moyle and Light 1996). Williams et al. (2009) showed that lower flows resulted in higher summer water temperatures and lower dissolved oxygen levels; both variables directly affect salmonid survival (Hicks et al. 1991, Fetherman et al. 2015). Fourth, the logistical and economic constraints of conducting large-scale removals in large river systems are substantially unattractive as a viable management option.

For example, although nearly $4.4 million was spent to remove 1.5 million nonnative predatory fish from the Colorado River, 86% of published reports (as of 2005) suggested that native species do not benefit from removal efforts (Mueller 2005). Fetherman et al. (2015) also found that Brown Trout removal had only a short-term positive benefit on Rainbow Trout (*O. mykiss*). However, the overall benefit of removal was equivocal, which led these authors to conclude that removal of adult Brown Trout was not a viable management option to pursue in future conservation efforts of Rainbow Trout, and certainly not in perpetuity. As such, resource managers and policy makers must weigh the logistical constraints, economic costs, and achievable measures of success associated with removal efforts against benefit(s) of the action. This is the only approach by which a resource agency responsible for the stewardship of fish and wildlife, can reasonably determine whether removal of adult Brown Trout from a large hydrologically influenced, temperature and water variable, and habitat managed riverine system is a viable long-term management option, pursuant to future species, conservation, and economic needs.

Facilitating completion of the adaptive management loop is often disconnected from reality by the politics of resource management (Murphy and Weiland 2014). Attributes of effective and comprehensive species-focused management for the upper Trinity River must rely upon implementation of the best available science, which includes relevant aspects of species life history requirements (TRFES 1999, Sullivan et al. 2006). CDFW does not currently have a management policy that mandates systematic removal of Brown Trout captured through any sampling effort or caught by anglers. We believe that advocating destruction of captured Brown Trout and development of recommendations for suppression to population levels that do not "significantly" impede the Endangered Species Act (ESA), Tribal-trust species recovery goals and objectives (Alvarez 2017), or programmatic restoration efforts within the upper Trinity River are premature, and the possible outcomes of such actions are likely not knowable. Our view is particularly relevant given the lack of: 1) information on comparative predatory impacts of other fish and terrestrial species on juvenile salmonid survival, 2) a thorough economic analysis of the long-term consequences of any proposed management actions, and 3) an analysis of the relative impact to the local commercial sport fishing industry focused on the upper Trinity River.

Instead, we maintain that progressive actions derive from *a priori* assessment of the: 1) comparative impacts of managed flow regimes on timing of migration in adult Brown Trout, as well as anadromous salmonids (Peterson et al. 2017); 2) population size and age structure of sympatric co-occurring salmonids; and, 3) metric-driven prey-base and dietary requirements of co-occurring riverine salmonid communities. We also suggest that studies be implemented that focus on age-specific habitat use within both aquatic and terrestrial piscivorous communities inhabiting the upper Trinity River, in conjunction with experimental and in-river studies focused on specific species; including non-salmonid fish taxa. Such actions would allow a better understanding of the potential for managed flows in facilitating conservation in all connected co-varying segments of this highly regulated river system (Hasler et al. 2014, Peterson et al. 2017), which constitute essential and integral elements of any coordinated science-based adaptive management program.

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Appendix I.—History of annual Trinity River Hatchery spawning and juvenile release dates for Brown Trout in the upper Trinity River; na = no data, blank = not applicable, UM = unmarked release, AD = adipose fin clipped release, LV = left ventral fin clipped release, RV= right ventral fin clipped release.

				1st Release group			2nd Release group		
Brood year	Adults	Females spawned	Eggs taken	Date	Number released	Location	Date	Released	Location
1963	34	22	58,100	23-Apr-65	28,500 (UM)	Trinity River Hatchery			
1964	145	64	163,600	18-Apr-66	28,900 (AD)	Trinity River Hatchery			
1965	100	39	109,900	1-Nov-67	35,200 (LV)	Hawkin's Bar			
1966	152	62	75,600	1-Oct-68	41,514 (UM)	Hawkin's Bar			
1967	231	62	158,700	na	na	na			
1968	170	32	76,644	Oct-70/Nov-70	18,967 (RV, LV)	China Slide (LV)	Nov-70	10841 (RV)	Klamath Glen
1969	70	26	53,000	29-Mar-71	23,370 (UM)	Lime Point	Nov-72	2321 (UM)	Trinity River Hatchery
1970	23	2	na	na	na	Trinity River Hatchery			
1971	7	34	140793	17-Apr-73	30,220 (UM)	Trinity River Hatchery			
1972	111	43	73,000	8-Aug-74	23,030 (UM)	Old Lewiston Bridge			
1973	39	26	55,227	14-Apr-75	17,820 (UM)	Old Weir Site			
1974	32	7	13,200	23-Apr-76	73,60 (UM)	Trinity River Hatchery			
1975	24	9	17,272	1-Apr-77	12,600 (AD)	Trinity River Hatchery			
1976	49	28	52,000	5-May-77	29,500 (UM)	Trinity Lake			

Appendix II.—Annual attributes of ascending and descending limbs of hydrographs that characterized baseline Pre-ROD, ROD, and Pulse flows for the years (1995-2000 and 2002-2017). Rate of flow measured in cubic meters per second (m^3/s) and flow release in hector meters. For each hydrograph, a bench indicated a temporary holding steady of flow release volume and flattening of the hydrograph. Rapidness indicated a steep and immediate increase or decrease in rate of flow, relative to a more prolonged or gradual increase or decrease in rate of flow. Abbreviations: na = no data, R = rapid, G = gradual, B = number of benches, and 2P = double peak. Digital data to verify online printed hydrographs for 1994 and 2001 were not available through California Department of Water Resources, California Data Exchange Center website for the time-period encompassing all 3-flow types.

					Ascending and descending limbs of hydrograph								Pulse flow ascending-descending limbs of hydrograph					
Year	Water Year	Flow	Low release magnitude (m^3/s)	Shape ascending	Peak release magnitude (m^3/s)	Restoration release (hector m)	Shape descending	Low release magnitude (m^3/s)	Date and duration flow to base-flow	Total days	Low (m^3/s)	Shape ascending	Peak release magnitude (m^3/s)	Pulse restoration release (hector m)	Shape descending	Low release magnitude (m^3/s)	Date and duration flow to base-flow	Total days
1995	na	Pre-ROD	14	R	131	na	G	27	25 Apr-22 May	28								
1996	na	Pre-ROD	9	R	144	na	G, 1B	14	10 May-9 Jun	31								
1997	na	Pre-ROD	10	R	62	na	G, 3B	13	2 May-2 Jul	62								
1998	na	Pre-ROD	47	R	192	na	G	13	24 May-27 Jul	65								
1999	na	Pre-ROD	15	G	71	na	G	13	8 May-18 Jul	72								
2000	na	Pre-ROD	9	R	66	na	G	13	8 May-27 Jul	81								
2002	normal	Pre-ROD	9	R	171	59,540	G	13	27 Apr-25 Jun	28								
2003	wet	Pulse	9	R	74	55,272	G, 2B	12	29 Apr-22 Jul	85	13	R	51	4,194	G	12	23 Aug-18 Sep	27
2004	wet	Pulse	9	R	176	80,300	G, 4B	12	4 May-22 Jul	80	16	R	48	4,465	G	14	21 Aug-14 Sep	25
2005	wet	ROD	8	R, 2 B	197	79,880	G, 1B	13	27 Apr-22 Jul	87								
2006	extra wet	ROD	8	G, 5B	286	99,900	G, 2B	13	16 Apr-22 Jul	98								
2007	dry	ROD	8	R	135	55,963	G	13	25 Apr-25 Jun	62								
2008	dry	ROD	9	R, 1B	183	80,016	G, 3B	20	22 Apr-15 Jul	85								
2009	dry	ROD	8	R	125	54,952	G, 4B	12	24 Apr-6 Jul	74								
2010	wet	ROD	9	R	194	81,003	G, 3B	12	22 Apr-2 Aug	102								
2011	wet	ROD	7	R, 2B	329	89,033	G, 2B	13	26 Apr-1 Aug	98								
2012	normal	Pulse	9	R, 2B	172	79,819	G, 4B	13	4 Apr-26 Jul	114	13	R, 1B	39	4,811	R, 1B	13	12 Aug-20 Sep	40
2013	dry	Pulse	8	R, 1B	125	55,741	G, 2B	13	13 Apr-25 Jun	74	13	R	74	2,294	R, 1B	13	24 Aug-20 Sep	28
2014	critically dry	Pulse	9	R, 1B	97	45,701	G, 1B	13	21 Apr-26 Jun	67	12	R, 1B	97	7,993	R, 1B	13	15-25 Sep	11
2015	dry	Pulse	9	R	241	55,593	G, 3B	13	21 Apr-1 Jul	72	19	R, 1B	83	5,908	R, 1B	13	20 Aug-21 Sep	31
2016	wet	Pulse	9	R, 2P	283	87,429	G	13	20 Apr-2 Aug	105	14	R, 1B	35	4,835	R, 2B	13	24 Aug-28 Sep	36
2017	extra wet	ROD	9	R, 2P	326	101,536	G	13	22 Apr-11 Aug	112								

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Biodiversity of amphibians and reptiles at the Camp Cady Wildlife Area, Mojave Desert, California and comparisons with other desert locations

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We examined the biodiversity of amphibian and reptile species living in and near constructed ponds in the riparian area at the Camp Cady Wildlife Area (CCWA) in the Mojave Desert of San Bernardino County, California, based on field work from 1998-1999, 2016-2017, review of the literature, and searches for museum specimens using VertNet.org. A total of 11 species (201 captures), including two frogs and toads (one non-native frog), one turtle, three snakes, and five lizards were captured at terrestrial drift fences with pitfall traps encircling two ponds (0.5 hectares total) on the property in 1999. Four additional species (one frog, one lizard, and two snakes) were previously reported in 1978 from a ranch 1.6 km southwest from CCWA for a total of 15 species in the local area. The southwestern pond turtle (*Actinemys pallida*), was commonly observed at CCWA from 1998 to 1999 and documented as a breeding population. However, the species was extirpated at CCWA sometime after 2014 when the last individuals were photographed, and none have been detected since then despite significant efforts to do so. Biodiversity of amphibians and reptiles at CCWA is relatively low compared with sites elsewhere in the

Mojave Desert with more elevational diversity. The 14 native species documented at CCWA accounts for approximately 21% of the native reptile and amphibian species reported by Stewart (1994) for the entire Mojave Desert, including peripheral species. Our smaller sample likely represents a group of easily detected species and is biased toward those found in or near water, especially amphibians. However, the relative proportion of amphibians vs. reptiles that inhabited CCWA in the last 40 years is not significantly different from the recently compiled proportions at five military installations in the California deserts. The herpetofauna inhabiting CCWA is notable for including riparian obligates like the western toad (*Anaxyrus boreas*), Northern Baja California treefrog (*Pseudacris h. hypochondriaca*), and *A. pallida* that are otherwise absent from large portions of the Mojave Desert. Other species are typical of those that are expected in the low-elevation creosote scrub-dominated ecosystem in the area.

Key words: amphibians, California, Camp Cady Wildlife Area, herpetofauna, Mojave Desert, Mojave River, reptiles

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Species richness is a fundamental metric of ecosystems and has long been proposed as a criterion for directing conservation efforts (Myers et al. 2000, Gotelli and Colwell 2001). There is little doubt that biotic inventories aimed at documenting species composition are essential for effective management of natural resources. Biodiversity surveys can provide valuable baseline data on species occurrences and distribution over varying spatial and temporal scales (Gibbons et al. 1997, Hillebrand et al. 2017, Schmeller et al. 2017) that can reveal the effects of sampling bias as well as the effects of natural or anthropogenic stressors on community structure.

Large areas of the Mojave Desert in California have been heavily impacted by human activities for centuries (Lovich and Bainbridge 1999), especially in the western portion where urbanization continues (Hunter et al. 2003). In particular, the Mojave River corridor has been heavily impacted by anthropogenic changes over time resulting in significant changes from pre-colonization times (Lines 1996, Webb et al. 2001, Laity 2003). Increases in municipal and agricultural water use since the 1980s have eliminated or dramatically reduced surface water availability, especially along the middle and lower reaches of the Mojave River (Izbicki and Michel 2004, Todd Engineers 2013). Climate simulations for the near future (2021-2040) project continuing declines in surface water availability across the southwest USA including reduced soil moisture and runoff in California (Seager et al. 2012). De-watering the Mojave River presents challenges for wildlife conservation, especially for riparian obligates and associates that have few other water sources in this otherwise arid landscape.

The Camp Cady Wildlife Area (CCWA) is a state-managed facility along the lower reach of the Mojave River that supports a variety of wildlife species, including several that are dependent on riparian habitat. Only one previous survey of amphibians and reptiles (herpetofauna) was conducted in the vicinity of the Camp Cady Wildlife Area (CCWA), but that was 40 years ago (Brown 1978). In contrast, numerous studies have been published discussing various aspects of the hydrology, geology, and vegetation along the river (e.g., Pluhar et al. 1991, Lines and Bilhorn 1996, Webb et al. 2001, Todd Engineers 2013). This study provides an assessment of the biodiversity of amphibians and reptiles at CCWA with emphasis on those found in or near water. A wide variety of species were expected, especially riparian obligates, due to the presence of permanent water at this otherwise arid site. In addition, we compare and contrast the herpetofauna of Camp Cady with data from other parts of the California deserts, including several military installations that have recently assembled checklists.

Materials and Methods

Study area.—The Camp Cady Wildlife Area (34° 56.187’ N, 116° 36.650’ W) is an approximately 770-ha area (Figure 1) in the Mojave Desert managed by the California Department of Fish and Wildlife for bird watching, wildlife viewing, hiking, and hunting (birds and rabbits). The site is located 32 km east of Barstow, in San Bernardino County, California. Elevation ranges from 512 to 536 m. Using the Westmap (https://cefa.dri.edu/Westmap/) pixel function to calculate average climate conditions from yearly averages for the period 1999-2017, summers are hot and dry with an average high temperature of 40.5°C,

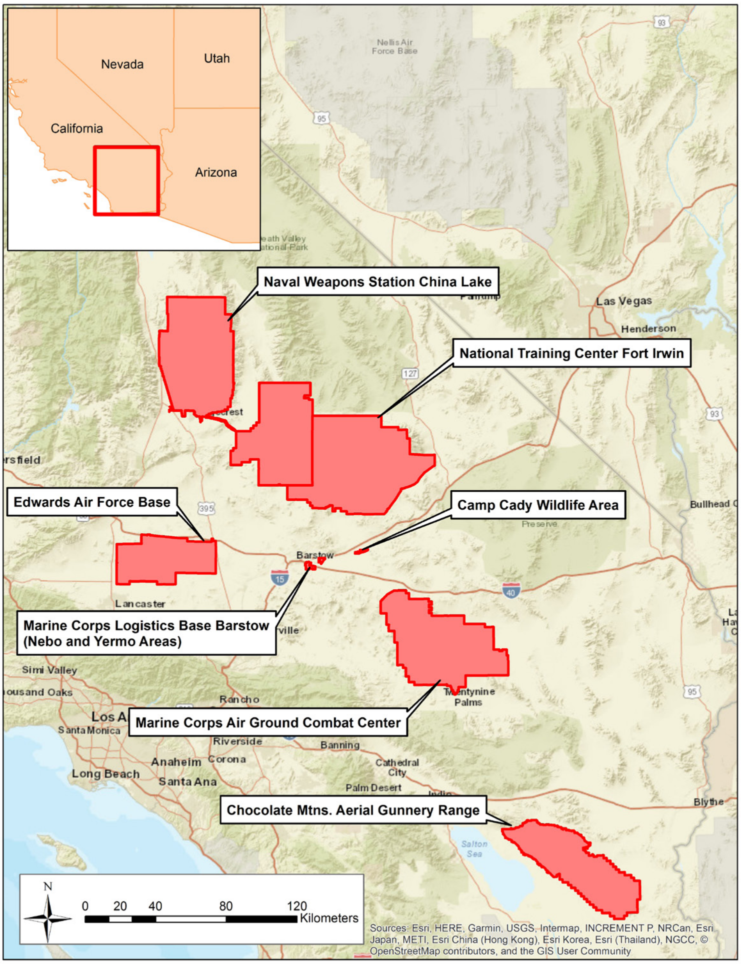

Figure 1.— Map showing the location and boundaries of Camp Cady Wildlife Area and the Department of Defense military installations for which data on amphibian and reptile diversity are available.

winters are cool with average low temperatures of 2.2˚C, and average annual precipitation is 8.1 cm. Plant species in the riparian area include: mesquite (*Prosopis* spp.), tamarisk (*Tamarix ramosissima*), willow (*Salix* spp.), cottonwood (*Populus fremontii*), saltgrass (*Distichlis spicata*), saltbush (*Atriplex* spp.), and cattails (*Typha domingensis*).

Three artificial ponds, less than 2 m in depth, are maintained at CCWA through groundwater pumping (Figure 2). The ponds are adjacent to the Mojave River and its floodplain (Lovich and Meyer 2002). The ponds were constructed between 1983 and 1984 to maintain an assurance population of endangered Mojave tui chubs (*Siphateles bicolor mohavensis*). "Bud's Pond," the central pond (hereafter Pond 2), is approximately 0.25 ha and surrounded primarily by cattails. The easternmost pond (Pond 3), also approximately 0.25 ha, is surrounded by cattails and honey mesquite and has a small island in the center. Pond 1, the westernmost pond, was originally created as a plastic-lined fire suppression pond (approximately 0.1 ha). It was not sampled in this study due to its small size and lack of natural vegetation or habitat. The Mojave River in this area once included dense stands of riparian vegetation (Todd Engineers 2013) and a small lake (the latter shown on U.S. Geological Survey topographic maps as recently as 1993) but neither the river nor the lake held water during the time of our sampling.

From May 1998 through October 1999 and April 2016 through July 2017, we set baited hoop traps (Gibbons 1988) in the riparian area at CCWA to sample for southwestern pond turtles (*Actinemys pallida*) as detailed in Lovich and Meyer (2002). From 12 May 1999 to 12 July 1999, we installed drift fences and pitfall traps (Gibbons and Semlitsch 1981) on berms around the perimeters of Ponds 2 and 3 to intercept nesting female *A. pallida*, but we recorded any amphibians or reptiles observed or captured incidentally. We installed pitfall traps (19-L plastic buckets) on both sides of the drift fence (Lovich and Meyer 2002) and checked them periodically during daylight hours. After identifying species, we released all

Figure 2.— Satellite image of the Camp Cady Wildlife Area study site, San Bernardino County, California, USA. Photo modified from ©2018 Google Earth Pro.

animals on the opposite side of the fence from where they were captured. Daytime water temperatures averaged 19° C during field sampling in 1999. Rainfall was only observed on one day during the study period while we were present (2 June 1999).

We obtained other records of species found in the immediate vicinity of Camp Cady from Brown (1978) and VertNet.org (Table 1), the latter for museum specimens. Brown (1978) conducted his three-month study (21 April-9 July 1978) on private property, 1.6 km southwest of our study site, along the south floodplain of the Mojave River. He used eight 19-L pitfall bucket traps along the dry edges of the river, along with hand captures and nooses, to sample the terrestrial herpetofauna. He used seines and dip nets in areas with water. The pitfall traps were checked every two weeks. We compiled records for the greater California desert region from other literature sources as cited below. We obtained records for Department of Defense military installations in the California desert from Peterson et al. (2017; *in press*) and the associated database they compiled. Common and scientific names of herpetofauna follow Crother et al. (2017).

We compared the relative proportion of amphibian vs. reptile species at CCWA to those same proportions for five military installations in the California deserts (Figure 1). We hypothesized that species diversity at CCWA would be disproportionately weighted toward amphibians due to the concentration of our sampling near ponds along the Mojave River channel. We tested proportions across sites with a contingency table analysis. We also analyzed comparable data on reptile species richness and abundance from ongoing research at the California Desert Studies Center (Zzyzx, near Baker, California) spanning 27 years (June 1991-May 1993, January 2000-December 2001, January 2008-June 2018), as described, in part, by Wallace (2003, Jason Wallace, CSU, Fullerton, unpublished data). That study used an array of 129 19-L bucket pitfall traps without drift fences on a creosote bush (*Larrea tridentata*)-dominated alluvial fan (288-358 m ASL) in the Soda Mountains approximately 50 km northeast of CCWA. We used the Shannon Index where

$$H' = \sum_{i=1}^{R} p_i \log p_i$$

and an Equitability Index where

$H'/H_{max} = H'$/natural log of the total number of species observed

to compare data collected at Camp Cady in 1998-1999 with those of Wallace (2003, Jason Wallace, CSU, Fullerton, unpublished data). Equitability Indices range from 0 to 1 with higher values reflecting more similar numbers of individuals among species. Although there are vastly different timescales involved between our study and that of Wallace, his are the only comprehensive reptile data available in the region for comparisons of species richness and evenness. Our conclusions on these comparisons are tempered accordingly by not conducting statistical tests of differences between indices.

Results

We captured 201 amphibians and reptiles in our drift fence in 1999 (Table 1). No *A. pallida* were captured or observed in the period from 2016 to 2017 despite significant effort. Since the only species marked for individual recognition during our studies was *A.*

TABLE 1.—List of all species reported in this study, an earlier study by Brown (1978), and museum specimens recorded over time in and near the Camp Cady Wildlife Area. MVZ = Museum of Vertebrate Zoology, UCMP = University of California Museum of Paleontology.

Scientific name	Location	Captures	Capture Method	Year	Months Found	Collector	Comments
Anaxyrus boreas (western toad)	Camp Cady	98	Drift Fence	1999	May-July	this study	Brown reported as "*Bufo boreas*"
	Camp Cady	5	Can trap	1978	April-July	Brown, 1978	Museum specimen MVZ Amphibian and reptile specimens #172757
Actinemys pallida (southwestern pond turtle)	Camp Cady	27	Drift Fence	1999	May-July	this study	Brown reported sightings by other people at Camp Cady
"Emydidae" (assumed *Actinemys pallida*)	Camp Cady	1	Visual	1965	N/A	Daily & Hutchison	Carapace fragments, Museum specimen # UCMP V 74679
Aspidoscelis tigris (tiger whiptail)	Camp Cady	14	Drift Fence	1999	May-July	this study	
	Camp Cady	1	Can trap	1978	April-July	Brown, 1978	Brown reported as "*Cnemidophorus tigris*"
Coluber flagellum (coachwhip)	Camp Cady	1	Drift Fence	1999	June	this study	
	Camp Cady	2	Can trap	1978	April-July	Brown,1978	Brown reported as "*Masticophis flagellum*"
Crotalus cerastes (sidewinder)	Camp Cady	1	Can trap	1978	April-July	Brown, 1978	
Hypsiglena chlorophaea (desert nightsnake)	Camp Cady	1	Drift fence	1999	June	this study	
Lampropeltis californiae (California kingsnake)	Camp Cady	4	Drift Fence	1999	May-June	this study	Brown reported as "*Lampropeltis getulus californiae*"
Lithobates catesbeianus (American bullfrog)	Camp Cady	16	Drift Fence	1999	June-July	this study	Brown reported as "*Rana catesbeiana*"
	Camp Cady	18	Can trap	1978	April-July	Brown, L.T. Findley	Museum specimen MVZ Amphibian and reptile specimens #172783
Pituophis catenifer deserticola (Great Basin gophersnake)	Camp Cady	-	-	1978	–	Brown, 1978	Brown reported sightings by other people at Camp Cady
Pseudacris h. hypochondriaca (Northern Baja California treefrog)	Camp Cady	15	Can trap	1978	April-July	Brown, 1978	Brown reported as "*Hyla regilla*"
	Camp Cady	10	Hand	1978	May	Brown, L.T. Findley	Lot of 10 tadpoles, Museum specimen MVZ Amphibian and reptile specimens #172766
Sceloporus occidentalis (western fence lizard)	Camp Cady	28	Can trap	1978	April-July	Brown, 1978	Museum Specimen MVZ Amphibian and reptile specimens #172897
	Camp Cady	10	Drift Fence	1999	June	this study	
	Camp Cady	2	Hand	1978	May	Brown, L.T. Findley	Museum specimen MVZ Amphibian and reptile specimens # 172895 & 172896
Sceloporus magister (desert spiny lizard)	Camp Cady	12	Can trap	1978	April-July	Brown, 1978	Museum specimen MVZ Amphibian and reptile specimens #172869
Urosaurus graciosus (long-tailed brush lizard)	Camp Cady	1	Drift Fence	1999	June	this study	
Uta stansburiana (common side-blotched lizard)	Camp Cady	17	Drift Fence	1999	May-July	this study	
	Camp Cady	4	Can trap	1978	April-July	Brown, 1978	
Xantusia vigilis (desert night lizard)	Camp Cady	9	Drift Fence	1999	May-June	this study	

pallida, it is unknown how many individuals of other species were captured. Captures and recaptures included 114 frogs and toads, 27 turtles, 6 snakes, and 54 lizards for a total of 87 reptile and 114 amphibian captures (including 3 lizards captured but not identified to species by field technicians). At Pond 3 there were 112 amphibian captures and 63 reptile captures, while Pond 2 only had 2 amphibian and 20 reptile captures (four reptiles caught were not associated with any pond). Eleven species were identified from captures in 1998 and 1999 (Figure 3), including two species of frogs and toads, one turtle, three snakes, and five lizard species. Four additional species [Northern Baja California treefrog (*Pseudacris h. hypochondriaca*), desert spiny lizard (*Sceloporus magister*), Great Basin gopher snake (*Pituophis catenifer deserticola*), and sidewinder (*Crotalus cerastes*)] were reported by Brown (1978) bringing the total herpetofauna in the CCWA area to 15 species. Brief species accounts of the recorded herpetofauna are given below with records of museum specimens, if available.

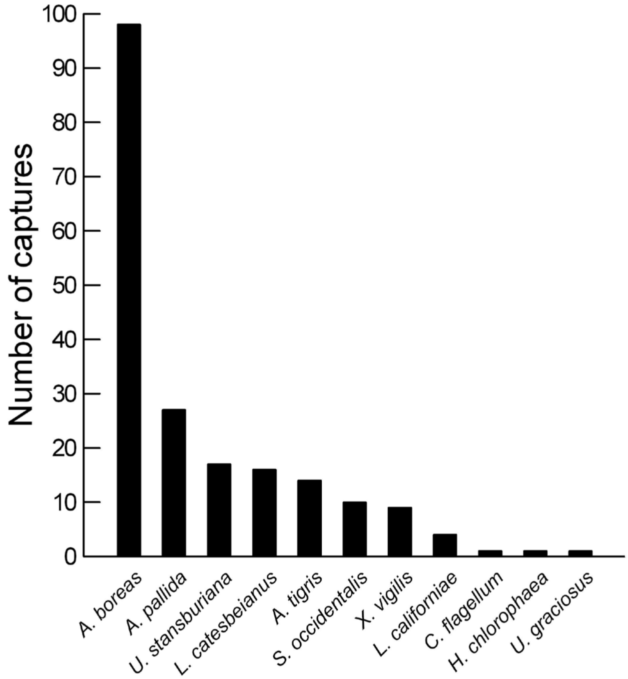

Figure 3.—Number of captures of amphibians and reptiles by species from 1998 to 1999 at the Camp Cady Wildlife Area (not including 3 unidentified lizards).

In the ongoing study of Wallace (2003, Jason Wallace, CSU, Fullerton, unpublished data) at Zzyzx, California, 3,027 captures of 20 reptile species were recorded as of June 2018 (Table 2). The Shannon Index was 1.50, H_{max} was 3.00, and the Equitability Index was 0.50. By comparison, CCWA data from 1998 to 1999 had a Shannon Index of 1.67, H_{max} was 2.40, and the Equitability Index was 0.70. Since Wallace's data do not include amphibians, we removed the two amphibian species from the same CCWA data resulting in a Shannon Index of 1.12, an H_{max} of 2.20, and an Equitability Index of 0.51. The data of Brown (1978) including amphibians (Table 1) had a Shannon Index of 1.73, an H_{max} of 2.20, and an Equitability Index of 0.79.

Table 2.—List of all reptile species reported in the ongoing study of Wallace (2003, Jason Wallace, CSU, Fullerton, unpublished data) at Zzyzx, California. Data collected monthly June 1991-May 1993, January 2000-December 2001, and January 2008-June 2018.

SCIENTIFIC NAME	COMMON NAME	CAPTURES
Uta stansburiana	common side-blotched lizard	1,529
Aspidoscelis tigris	tiger whiptail	809
Urosaurus graciosus	long-tailed brush lizard	187
Chionactis occipitalis	western shovel-nosed snake	117
Coleonyx variegatus	western banded gecko	108
Callisaurus draconoides	zebra-tailed lizard	92
Dipsosaurus dorsalis	desert iguana	68
Phrynosoma (Doliosaurus) platyrhinos	desert horned lizard	54
Phyllorhynchus decurtatus	spotted leaf-nosed snake	29
Rena humilis	western threadsnake	8
Xantusia vigilis	desert night lizard	5
Gambelia wislizenii	long-nosed leopard lizard	4
Hypsiglena chlorophaea	desert nightsnake	4
Arizona elegans	glossy snake	3
Sauromalus ater	common chuckwalla	3
Crotalus cerastes	sidewinder	2
Crotaphytus bicinctores	Great Basin collared lizard	2
Uma scoparia	Mojave fringe-toed lizard	1
Coluber flagellum	coachwhip	1
Pituophis catenifer deserticola	Great Basin gophersnake	1

The relative proportion of native amphibian vs. reptile species at CCWA and five military installations in the California deserts (Naval Air Weapons Station China Lake, Edwards Air Force Base, National Training Center [Ft. Irwin], Logistics Base Barstow, and Chocolate Mountains Aerial Gunnery Range) listed in Tables 3 and 4 were not statistically different when tested with contingency table analysis ($X^2 = 4.66$, $df = 5$, $P = 0.46$).

Frogs and toads.—American bullfrog (*Lithobates catesbeianus*). 16 captures. This species is native to eastern and central North America but is introduced all over the world (Behler and King 1979, Kupferberg 1997, Stebbins 2003). Kupferberg (1997) stated this

Table 3.—List of amphibians and reptiles at six military bases in the Mojave and Sonoran deserts of California for comparison with data on amphibian and reptile diversity at Camp Cady Wildlife Area. Data are from Petersen et al. (2017, in press) as tabulated in the associated database that they provided to us. Species occurrence is listed as X = confirmed on site or P = potentially on site but unconfirmed. Although their compilation included various subspecies we condensed the table to include only species-level taxonomy.

Species	Naval Weapons Station China Lake	Edwards Air Force Base	National Training Center Ft. Irwin	Marine Corps Air Ground Combat Center	Marine Corps Logistics Base Barstow	Chocolate Mtns. Aerial Gunnery Range
Actinemys pallida (western pond turtle)		X			P	
Anaxyrus boreas (western toad)	X	X		X	X	
Anaxyrus punctatus (red-spotted toad)	X	X		X	P	P
Anaxyrus woodhousii (Woodhouse's toad)						P
Arizona elegans (glossy snake)	X	X	X	X	P	P
Aspidoscelis tigris (tiger whiptail)	X	X	X	X	X	X
Batrachoseps robustus (Kern Plateau salamander)	P					
Callisaurus draconoides (zebra-tailed lizard)	X	X	X	X	X	X
Chelydra serpentina (snapping turtle)		X				
Chilomeniscus stramineus (variable sandsnake)		X				
Chionactis occipitalis (western shovel-nosed snake)	X	X	X	X	P	P
Chrysemys picta (painted turtle)		X				
Coleonyx variegatus (western banded gecko)	X	X	X	X	X	P
Coluber flagellum (coachwhip)	X	X	X	X	X	P
Coluber taeniatus (striped whipsnake)	X					
Crotalus atrox (western diamond-backed rattlesnake)						X
Crotalus cerastes (sidewinder)	X	X	X	X	X	X
Crotalus pyrrhus (southwestern speckled rattlesnake)		P	X	X	X	P
Crotalus oreganus (western rattlesnake)	X					
Crotalus scutulatus (Mohave rattlesnake)	X	X	X	X	X	
Crotalus stephensi (Panamint rattlesnake)	X					
Crotaphytus bicinctores (Great Basin collared lizard)	X	X	X	X	X	P
Diadophis punctatus (ring-necked snake)	X					
Dipsosaurus dorsalis (desert iguana)	X	X	X	X	X	X
Elgaria panamintina (Panamint alligator lizard)	X					
Gambelia wislizenii (long-nosed leopard lizard)	X	X	X	X	X	X
Gopherus agassizii (Mohave desert tortoise)	X	X	X	X	X	X
Heloderma suspectum (Gila monster)						P
Hemidactylus turcicus (Mediterranean gecko)		P			P	P
Hypsiglena chlorophaea (desert nightsnake)	X	X	X	P	X	P
Lampropeltis californiae (California kingsnake)	X	X	X	X	P	P
Lichanura orcutti (rosy boa)	X	X				
Lichanura trivirgata (three-lined boa)			X			P
Lithobates berlandieri (Rio Grande leopard frog)						P
Lithobates catesbeianus (American bullfrog)	X	X		P	X	
Phrynosoma (Doliosaurus) platyrhinos (desert horned lizard)	X	X	X	X	X	X
Phyllorhynchus decurtatus (spotted leaf-nosed snake)	X	X	X	X	X	P
Pituophis catenifer (gopher snake)	X	X	X	X	X	
Plestiodon gilberti (Gilbert's skink)	X	P				
Pseudacris hypochondriaca (Baja California treefrog)		X			X	
Pseudacris sierra (Sierran treefrog)	X					
Rena humilis (western threadsnake)	P	P	X	P	P	P
Rhinocheilus lecontei (long-nosed snake)	X	X	X	X	X	P
Salvadora hexalepis (western patch-nosed snake)	X	X	X	X	X	P
Sauromalus ater (common chuckwalla)	X	X	X	X	P	X
Sceloporus graciosus (common sagebrush lizard)	X		P			
Sceloporus magister (desert spiny lizard)			X			
Sceloporus occidentalis (western fence lizard)	X	X	X	X	X	P
Sceloporus uniformis (yellow-backed spiny lizard)	X	X	P	P	P	
Sonora semiannulata (western groundsnake)	X	X	X			
Tantilla hobartsmithi (Smith's black-headed snake)	P			X	P	P
Thamnophis hammondii (two-striped gartersnake)		X				
Trachemys scripta (pond slider)		X				
Trimorphodon lambda (Sonoran lyresnake)		X				
Trimorphodon lyrophanes (California lyresnake)	X		P			
Uma scoparia (Mohave fringe-toed lizard)	P	P	X	X	P	P
Urosaurus graciosus (long-tailed brush lizard)	P	X	X	X	X	
Urosaurus ornatus (ornate tree lizard)			P	X	X	P
Uta stansburiana (common side-blotched lizard)	X	X	X			
Xantusia vigilis (desert night lizard)	X	X	X	X	X	X
Xenopus laevis (African clawed frog)		X		X	X	

Table 4.—Number of amphibian and reptile species reported from various localities within the California deserts. Non-native species numbers are in parenthesis. Camp Cady Wildlife Area (CCWA) totals include our study (1998-1999, 2016-2017), Brown (1978) records, and museum specimens. The data for Brown (1978) include only those species reported near CCWA by him at that time. Department of Defense (DoD) (Petersen et al. 2017) records include confirmed and potential species for the military installations listed. Non-natives were not included in the Stebbins (1995) report. Wallace data includes Wallace (2003) and Jason Wallace, CSU, unpublished data, collected at California Desert Studies Center (Zzyzx, near Baker California). Neither Wallace nor Stebbins included amphibians in their study.

	Number of Species		
	Amphibian	**Reptile**	**Total**
CCWA	2(1)	12	14(1)
Brown (1978)	2(1)	9	11(1)
DoD-Naval Air Weapons Station China Lake	4(1)	38	42(1)
DoD-Edwards Air Force Base	3(2)	35(4)	38(6)
DoD-National Training Center Ft. Irwin	0	34	34
DoD-Marine Corps Air Ground Combat Center	3(1)	29	32(1)
DoD-Logistics Base Barstow	4(1)	30(1)	34(2)
DoD-Chocolate Mtns. Aerial Gunnery Range	2(1)	28(1)	30(2)
Stebbins (1995)a	–	24	24
Stebbins (1995)b	–	56	56
Stewart (1994)c	2(1)	35	37(1)
Wallace-California Desert Studies Center	–	20	20

[a] Only includes species from Pisgah Lava Flow approximately 31 km southeast of CCWA

[b] Entire California desert including Mojave, Sonoran, and Great Basin deserts

[c] Not including peripheral species (occurs in the habitat peripheral to the Mojave Desert)

invasive species was first introduced into California in 1896, and Brown (1978) stated that bullfrogs were introduced to the Mojave River at CCWA in 1927 and 1969. Bullfrogs are a known predator of several different frog, fish, turtle, bird, and snake species (Jancowski and Orchard 2013, Rockney 2015), causing declines in some native species (Stebbins 2003, Pearl et al. 2004). Brown (1978) observed 18 specimens 1.6-4.82 km west/southwest of Camp Cady.

L. catesbeianus museum specimens: Museum of Vertebrate Zoology 172783

Northern Baja California treefrog (*Pseudacris h. hypochondriaca*). No captures in 1999; however, Brown (1978) had 15 captures 1.6 km southwest of Camp Cady Ranch. Formerly recognized as *P. regilla*, this widespread species group was recently split into several taxa ranging from Canada to Baja California, Mexico (Recuero et al 2006). *P. h. hypochondriaca* is considered by some to be the most abundant and ubiquitous amphibian

in western North America (Recuero et al 2006). The species is found in widely scattered oases and other water sources in the Mojave Desert, as well in more mesic coastal locations.
P. h. hypochondriaca museum specimens: Museum of Vertebrate Zoology 172766

Western toad (*Anaxyrus boreas*). 98 captures adults and juveniles (no tadpoles were observed). Distributed from Alaska to Baja California, Montana, and Colorado, with wide elevational range from sea level to 3.0 km ASL (Stebbins 2003). *A. boreas* can endure both hot and freezing temperatures through the use of burrows (Mullally 1952). There is one previous capture record of the species (five specimens) 1.6 km southwest of CCWA (Brown 1978). The taxonomy of this species complex is still undergoing changes in western USA deserts (Gordon et al. 2017).
A. boreas museum specimens: Museum of Vertebrate Zoology 172757

Turtles.—Southwestern pond turtle (*Actinemys pallida*). We intercepted *A. pallida* 27 times at drift fences. Lovich and Meyer (2002) estimated a population size of 25 (± SE 3.28) turtles in 1999. *A. pallida* inhabits aquatic and sometimes surrounding terrestrial habitats at different times of the year (Bury 2012). It is the only extant, native, freshwater turtle in California (Lovich and Beaman 2008) where it is a species of special concern (Thompson et al. 2016). Records show a historical and prehistorical presence of *Actinemys* at various points along the Mojave River (Jefferson 1987, Ernst and Lovich 2009), including CCWA. The species became extirpated at CCWA after April or May of 2014 based on observations by the preserve manager and substantiated by our inability to trap or observe turtles from 2016 to 2017. The species was not included in the list for CCWA produced by Brown (1978): however, he noted that *A. pallida* was sighted in the area during that time by others.
A. pallida museum specimens: University of California Museum of Paleontology 74679 (carapace fragments).

Lizards.—Common side-blotched lizard (*Uta stansburiana*). 17 captures. Distributed throughout western North America, *U. stansburiana* prefers arid or semi-arid habitats (Wilson 1991, Stebbins 2003). Another local record of four captures was noted 1.6 km southwest of Camp Cady (Brown 1978).
U. stansburiana museum specimens: none

Desert night lizard (*Xantusia vigilis*). Nine captures. This small, nocturnal, secretive species lives under rocks and fallen trees primarily in California but extending into surrounding desert states and Mexico (Stebbins 2003). *X. vigilis* has somewhat specialized habitat needs that frequently include fallen yucca (*Yucca schidigera*) or Joshua tree (*Yucca brevifolia*) limbs and trunks (Marlow et al. 1988, Stebbins 2003). However, at CCWA they used dead cottonwood logs since yucca and Joshua tree are absent from the site.
X. vigilis museum specimens: none

Desert spiny lizard (*Sceloporus magister*). No captures in 1999; however, Brown (1978) lists 12 captures 1.6 km southwest of Camp Cady. Widely distributed throughout the deserts of the western United States and Mexico, this species inhabits both terrestrial and arboreal microhabitats (Parker and Pianka 1973, Leaché and Mulcahy 2007).
S. magister museum specimens: Museum of Vertebrate Zoology 172869

Long-tailed brush lizard (*Urosaurus graciosus*). One capture. This mostly arboreal species is distributed in the western United States but is common in California (including the Mojave Desert) and Arizona (Vitt and Ohmart 1975, Stebbins 2003). *U. graciosus* spends very little time terrestrially, preferring tree limbs, which may explain why only one was captured, despite their abundance in some locations (Vitt et al. 1978). It may have fallen

from one of the many mesquite or tamarisk trees fringing the margins of Pond 3.
U. graciosus museum specimens: none

Tiger whiptail (*Aspidoscelis tigris*). 14 captures. *A. tigris* prefers desert habitats with sparse plants and plenty of open space in the western United States (Pianka 1970, Stebbins 2003). One specimen is known from 1.6 km southwest of Camp Cady (Brown 1978).
A. tigris museum specimens: none

Western fence lizard (*Sceloporus occidentalis*). 10 captures. *S. occidentalis* prefers arid, open spaces with plenty of perch sites as a terrestrial and arboreal species in the western United States (Adolph 1990, Stebbins 2003). Twenty-eight specimens were previously reported 1.6-4.82 km southwest of Camp Cady (Brown 1978).
S. occidentalis museum specimens: Museum of Vertebrate Zoology 172895, 172896, 172897

Snakes.—California kingsnake (*Lampropeltis californiae*). Four captures. This wide-ranging snake is distributed throughout the western United States and is common in California; therefore, our capture of *L. californiae* was not unexpected (Stebbins 2003, Pyron and Burbrink 2009).
L. californiae museum specimens: none

Coachwhip (*Coluber flagellum*). One capture. *C. flagellum* ranges widely throughout the southern United States and northern Mexico (Stebbins 2003). They prefer xeric, sandy environments, use rodent holes for shelter from the elements, and sometimes occur around water where prey are abundant (Palermo et. al 1988). Two captures were documented 1.6 km southwest of CCWA (Brown 1978).
C. flagellum museum specimens: none

Great Basin gophersnake (*Pituophis catenifer deserticola*). No captures in 1999; however, Brown (1978) reported sightings in the Camp Cady area. This species is widespread across the western United States and slightly into Canada, occupying a wide variety of habitats (Stebbins 2003). *P. catenifer*, while mostly terrestrial, are excellent climbers (Eichholz and Koenig 1992) and good swimmers (Rodríguez-Robles 2003).
P. c. deserticola museum specimens: none

Desert nightsnake (*Hypsiglena chlorophaea*). One capture. This small, nocturnal snake is rarely seen, yet widely distributed, and inhabits most of the western United States (Stebbins 2003). *Uta stansburiana* is a common prey item for this snake (Stebbins 2003).
H. chlorophaea museum specimens: none

Sidewinder (*Crotalus cerastes*). No captures in 1999; however, Brown (1978) had one capture 1.6 km southwest of Camp Cady Ranch. *C. cerastes* is one of the most commonly seen snakes in the Mojave and Sonoran deserts, preferring sandy habitats (Secor 1994, Persons and Nowak 2007). This species was expected but not encountered during our study.
C. cerastes museum specimens: none

Discussion

We captured 11 different species of herpetofauna, consisting of 201 captures at CCWA in 1999, using drift fences with pitfall traps. Four additional species were reported from near CCWA (Brown 1978) bringing the total documented herpetofauna in the local area to 15 species. Amphibian drift fence captures were greater than that of reptiles at CCWA (114 amphibians and 87 reptiles), even though there was greater species diversity in reptiles (nine species) than amphibians (two species). Brown (1978) reported three species of amphibians and nine species of reptiles 1.6 km southwest of our study site, several of which we did not encounter in our later surveys (including *C. cerastes*, *P. h. hypochondriaca*, *S. magister,* and *P. c. deserticola*). In total, 3 species of amphibians and 12 reptile species have been reported at CCWA. Stebbins (1995) reported 10 lizard species, 13 snake species, and the desert tortoise (*Gopherus agassizii*) at the Pisgah Lava flow (approximately 31 km southeast of CCWA), while (Stewart 1994) reported 3 anuran species, 1 turtle species, 16 lizard species, and 18 snake species with another 28 peripheral species in the entirety of the Mojave Desert (Table 4). Stebbins (1995) did not include amphibians in his survey. More recently, Mittermeier et al. (2002) tallied 14 amphibian species and 45 reptile species in the entirety of the Mojave Desert. We observed far fewer Mojave reptile species than Stebbins (1995), Stewart (1994), or Mittermeier et al. (2002) due to the short duration of our study, limited capture techniques, and the small area we sampled. More species are doubtless present in the greater 770-ha CCWA, away from the ponds where our sampling was concentrated.

Amphibians dominated our captures with the largest number being 98 for the native species *Anaxyrus boreas*. In contrast, we only captured 16 of the invasive species *Lithobates catesbeianus*. This is surprising due to the hardy nature and wide-ranging habitat of the latter. *L. catesbeianus* is a gape-limited predator, consuming a wide variety of prey items (Cook and Currylow 2014). It is possible that the large size, semi-terrestrial niche, and toxic skin secretions of *A. boreas* provide some protection from predation by the mostly aquatic *L. catesbeianus* (Olson 1989, Benard and Fordyce 2003, Pearl et al. 2004). Our data contrast with those of Brown (1978), who observed a reverse proportional abundance of these species: 18 *L. catesbeianus* and 5 *A. boreas*, 1.6 km southwest of Camp Cady (Brown 1978). None of the frogs and toads we captured were tadpoles, whereas Brown (1978) captured both adults and tadpoles.

Short-term studies like ours and Brown (1978) had relatively high Shannon Indices compared to the long-term study of Wallace (2003, Jason Wallace, CSU, Fullerton, unpublished data). The latter detected more rare species and larger numbers of common species, resulting in a lower Equitability Index, compared to the short-term studies with fewer species and a more even distribution of numbers of individuals per species. Predictably, our 1998-1999 CCWA data excluding amphibians resulted in a lower Shannon Index than Wallace (2003, Jason Wallace, CSU, Fullerton, unpublished data), but the Equitability Indices were almost identical. The ratio of native amphibians to reptiles at CCWA was 16.6%; however, that ratio was not statistically different from the amphibian/reptile ratio (0-10.5%) for five other military installations in the California deserts.

Other species were notable for their absence in our survey. The red-spotted toad (*Anaxyrus punctatus*) has a wide California desert distribution, a need for permanent water sources (due to limited migration patterns), and an affinity for dried, ephemeral stream beds (Weintraub 1974, Stewart 1994). However, unexpectedly, we found none in this study. Brown (1978) also noted the unexpected absence of *A. punctatus* in the Camp Cady area. *A. punctatus* may not have colonized the CCWA ponds, due to their relatively recent creation, lack of rocky substrate, and great distance from other permanent water sources (Bradford et al. 2003). CCWA combines a xeric environment with permanent lentic water sources to create a rare desert oasis for amphibians, allowing consistent resources to meet the demands of reproduction and hydration for continued survival that would otherwise be impossible (Mayhew 1995).

Of the 11 species of reptiles reported from CCWA, including four snakes, all are common to this area, yet only a few specimens of each species were caught, in contrast to the large number of lizards captured (perhaps because we sampled in the summer, during the hottest part of the year, instead of during the spring). Since there were six different species of lizards (52 captures), comprising most of the reptiles, it is surprising that more snake species were not represented since lizards (and some amphibians) are a prey source to many snakes, including those captured in this study (Arnold 1972, Ferguson et al. 1982, Rodríguez-Robles et al. 1999). *Crotalus cerastes* and other congeners have been found in Camp Cady and surrounding areas (Brown 1978; Reynolds 2004; Persons and Nowak 2007; Petersen et al. 2017, *in press*; Jason Wallace, CSU, Fullerton, unpublished data) but were not captured or observed in our study. Large snakes in particular are difficult to capture in buckets of the size we used (Gibbons and Semlitsch 1981).

Only one species of turtle was caught at the site. *A. pallida* is a species known to occur along the Mojave River with previous historical and prehistorical records of occurrence (Lovich and Meyer 2002). Camp Cady provides adequate habitat for *A. pallida*; however, they may have been adversely affected by predators, including *L. catesbeianus* (Rockney 2015), and decreasing water supplies that diminish suitable habitat in the river channel (Spinks et al. 2003). The reason for their disappearance at CCWA after 2014 is unknown, but predation of hatchlings by bullfrogs and predation on juveniles and adults from a host of mammalian carnivores (Ernst and Lovich 2009; Vander Haegen et al. 2009) are possible explanations, especially for such a small, potentially vulnerable population. The disappearance of *A. pallida* represents a 6% decrease in biodiversity of the herpetofauna reported in this paper over the last 40 years. However, if we used drift fences and pitfall traps during the 2016-2017 trap sessions, we may have detected additional changes in the herpetofauna.

Site-specific inventories for amphibians and reptiles in the California desert are often scattered and fragmentary; however, systematic efforts to census biodiversity on Department of Defense installations recently became available (Petersen et al. 2017, 2018; Table 3). Due to increased urbanization and a changing climate, reptiles and amphibians are forced to perish or adjust, whether it is by location, elevation, lifestyle, prey selection, or through evolutionary adaptation (Gibbons et al. 2000, Urban et al. 2014). Since this study was largely conducted in 1999, anthropogenic influences and persistent drought (Griffin and Anchukaitis 2014, Mann and Gleick 2015) have had continued impacts on the Mojave River and the CCWA. Future monitoring of amphibians and reptiles at CCWA are needed to evaluate continued changes in biodiversity.

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Geographic range and biology of Spinyeye Rockfish (S*ebastes spinorbis* Chen, 1975), an endemic species to the Gulf of California, Mexico

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The genus *Sebastes* is composed of 110 species that inhabit marine waters from the intertidal zone to depths greater than 1000 m (Haldorson and Love 1991, Love et al. 2002, Hyde and Vetter 2007). Members of this genus (99 species) are distributed throughout the North Pacific with at least four in the North Atlantic and two in the southern hemisphere (Hyde and Vetter 2007). Seven species of the genus *Sebastes*, including the Spinyeye Rockfish (*Sebastes spinorbis*), are found in the Gulf of California and six of these are endemic to the Gulf (Love et al. 2002, Hyde and Vetter 2007, Palacios-Salgado et al. 2012). The Spinyeye Rockfish differs from other members of the genus by the presence of a spine on the lateral surface of the corner of the eye and/or the orbital border of the first and second suborbital (Chen 1975). Adults are pink to red with some orange shading; they have six pale dorsal spots and can reach 344 mm total length (TL) (Love et al. 2002, Robertson and Allen 2015). Little information is available about the bathymetric and geographic distribution of this species. Its previously known geographic range was reported associated with rocky bottoms in the vicinity of Bahía de Los Angeles and Ángel de la Guarda Island in the central region of the Gulf of California, primarily on the continental external shelf at depths from 130–200 m (Castro-Aguirre and Balart 1996, Castro-Aguirre et al. 2005). Nevertheless, Acevedo-Cervantes et al. (2009) have reported this species recently at depths of 200–540 m.

At present, Spinyeye Rockfish is not a fishing target, and it is not considered a species of conservation concern. Little information is available on its biology (Castro-Aguirre et al. 2005) because it inhabits deep waters and is absent in commercial and recreational fisheries (Castro-Aguirre and Balart 1996, Love et al. 2002). Therefore, this research provides basic information on latitudinal and bathymetric distribution, sizes and reproductive biology of Spinyeye Rockfish in the Gulf of California. The study documents an expansion of the current range to the central Gulf region where it had not been historically recorded. Finally, the presence of Spinyeye Rockfish has now been documented over soft bottoms, contrasting with previous studies reporting that it was only associated with rocky bottoms.

Material And Methods

Specimens were obtained from three research cruises on board the R/V BIP XII, during September 2004 and February and May 2005. The primary purpose of these cruises was shrimp species exploration in waters greater than 80 m deep. Fishing trawl tows were performed on soft bottoms from Puerto Peñasco, Sonora (31°N) to Topolobampo, Sinaloa (25°N) at depths from 90–540 m in the eastern coast (Figure 1). The fishing equipment used was a bottom polyethylene trawl 38 m long, 68 m mouth perimeter, 38 m headline, 2.54 cm mesh size and operated astern.

After one hour of effective trawling (fishing haul time average, 2.0 knots average velocity), a sample (20 kg) was taken from the total catch at each station and frozen on

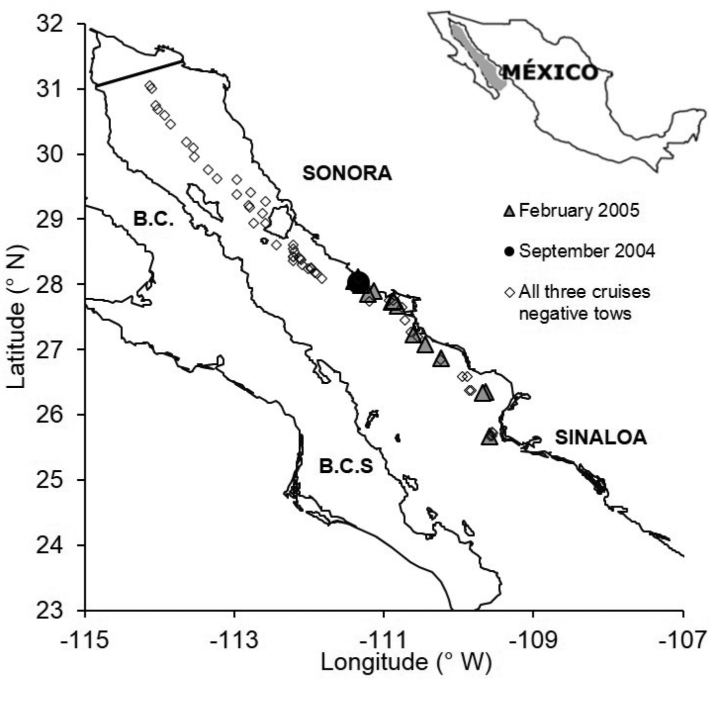

Figure 1.—Geographical location of *Sebastes spinorbis* collected in the Gulf of California in September 2004 and February 2005. All cruises negative tows are also shown.

board until it was processed at CIBNOR Fisheries Laboratory at Guaymas, Sonora. The depth and vessel position of each trawl haul was recorded with an echo sounder and GPS. Water temperature and dissolved oxygen in the water column were measured with CTD-SD204 at every station.

In the laboratory, the *Sebastes* specimens were separated and identified following Chen (1975) and Love et al. (2002). Each specimen was measured for standard length (SL) (mm) using a conventional ichthyometer to 1 mm precision and total weight (g) with a conventional Ohaus balance to 0.1 g precision. The length/weight relationship for Spinyeye Rockfish was calculated using the equation $W = a\ L^{*b}$. In addition, the gonad maturity stage from females and males was recorded visually according to Nikolsky (1963) morph-chromatic scale, which is based on colour and gonad texture, as well as its space in the abdominal cavity. The specimens in stages I–II were considered immature (gonads flaccid, transparent and less than one half of the space in the abdominal cavity); those in stages III–V were considered mature (gonads coloured and more than one half of the space in the abdominal cavity). According to the depth in which specimens of Spinyeye Rockfish were caught, the relationship between depth and fish size was explored.

Results

A total of 392 Spinyeye Rockfish were collected during the three cruises (71 trawl tows). Eighteen individuals were caught in September 2004, 374 in February 2005, and none in May 2005. Capture depth of Spinyeye Rockfish rangedfrom 95 to 482 m (Figure 1).

The observed sizes for Spinyeye Rockfish ranged from 68 to 268 mm with two modal frequencies at 130 and 200 mm SL (Figure 2). No difference by sex in sizes (Figure 2) and evident relationship between size and capture depth were observed. The length/weight relationship for the sampled size interval showed a coefficient of allometry *b* of 2.9787 (Limits: lower 2.92-3.03 upper, 95% confidentiality) and *a* value of 0.00002.

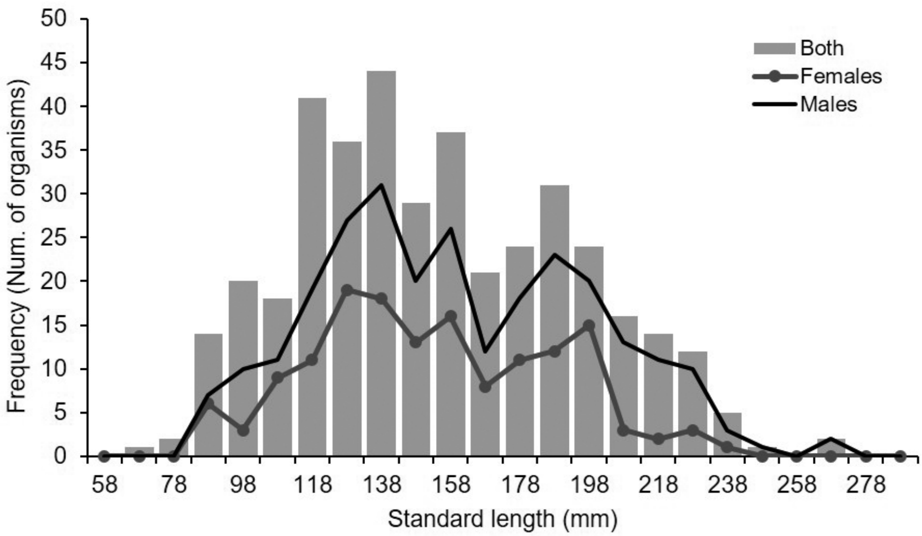

Figure 2.—Size frequency by sex of *Sebastes spinorbis* caught in the Gulf of California in September 2004 and February 2005.

Of the 392 fish analysed on September 2004 and February 2005, 40% were females, 31% males (M/F ratio 1:1.35) and 29% were undetermined. Five gonadal maturity stages were recorded in females, of which 33% were in process to maturity (stages I and II) and 67% sexually mature (stages III to V). Four gonadal maturity stages were recorded in males, 57% were in process to maturity (stage II) and 43% sexually mature (stages III to V).

The temperature and dissolved oxygen profiles in the water column of one station where Spinyeye Rockfish was collected (February 2005- 400 m deep) showed a rapid fall starting from 70 m in depth, reaching temperatures of less than 10°C and levels of almost anoxia at 400 m (Figure 3).

Discussion

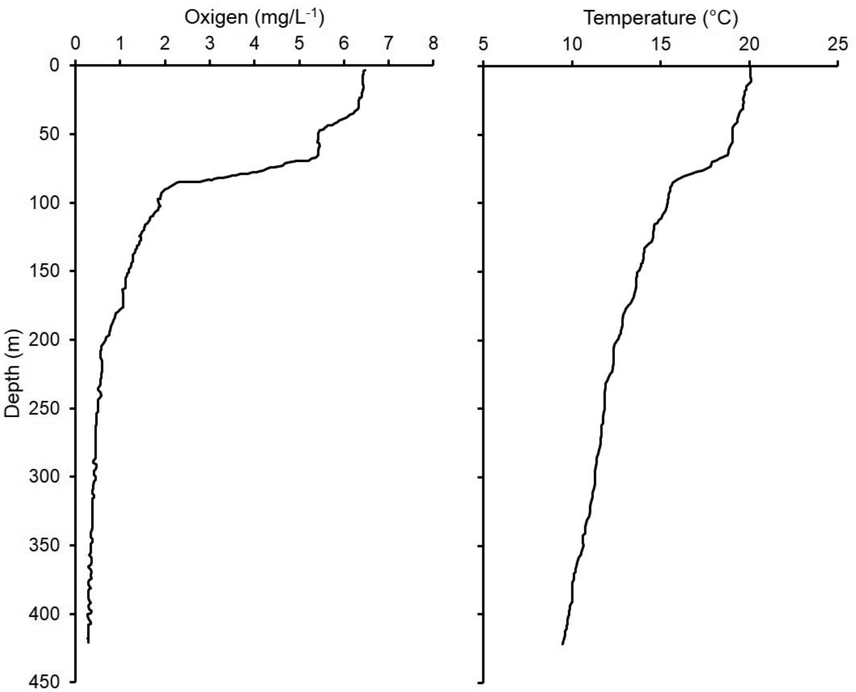

Figure 3.—Vertical distribution of the temperature and oxygen concentration, recorded during a positive tow in the Gulf of California.

This work shows an expansion of the documented distribution for Spinyeye Rockfish of approximately 3° in latitude (500 km) from Bahía de Los Ángeles and Isla Ángel de la Guarda in the Gulf of California southward to Topolobampo, Sinaloa (Figure 1). This report also confirms the depth record established for the species by Acevedo-Cervantes et al. (2009) of 480 m in contrast to Love et al. (2002), Castro-Aguirre et al. (2005) and Robertson and Allen (2015) who documented the presence of this species to a depth of 200 m. In this study Spinyeye Rockfish were caught on soft bottoms, contrasting also with the reports that it had been associated only with rocky bottoms.

The wide bathymetric distribution of this species suggests substantial environmental plasticity and perhaps the ability to tolerate low temperatures and anoxic conditions in the Gulf of California (Figure 3). These extreme conditions in the central Gulf of California had already been reported by Hendrickx and Serrano (2010) and Acevedo-Cervantes et al. (2009). The influence of these extreme conditions, especially on Caridea crustaceans and fishes, causes a drastic reduction in species number and abundance in the oxygen minimum zone (OMZ). This zone, which started with oxygen values less than 0.5 mg L^{-1}, was located close to 250 m in the central Gulf of California; however, the species develops in shallower water off Sinaloa (Acevedo-Cervantes et al. 2009, Hendrickx and Serrano 2010, Serrano 2012). The Spinyeye Rockfish is a species that uses the prevalent environmental conditions in the OMZ successfully, both over rocky and soft bottoms within the Gulf of California.

With respect to length of Spinyeye Rockfish, they ranged from 68 to 268 mm SL, 5 mm less than the maximum recorded by Love et al. (2002) of 273 mm SL (344 mm TL). Its maximum size recorded suggests it is not a big species, compared to other members of the genus distributed off coastal California and which are attractive to fishing (Love et al. 1990).

The bathymetric distribution of Spinyeye Rockfish is apparently not related to size, contrary to size segregation with depth in several species of this genus observed by Love et al. (1990). The length-weight relationship showed a value of *b* close to isometry (2.978), which agreed with similar results for 19 species of the *Sebastes* genus obtained by Love et al. (1990) on the California coast.

Based on the gonadal maturity stages of the females caught, 67% were sexually mature (stages III to V). It may indicate that spawning for this species occurs from February to August in the Gulf of California.

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Back.—Brown Trout (*Salmo trutta*). Photo by Eric Engbretson, USFWS

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