Changes in capture rates and body size among vertebrate species occupying an insular urban habitat reserve

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Abstract
Long-term ecological monitoring provides valuable and objective scientific information to inform management and decision-making. In this article, we analyze 22 years of herpetofauna monitoring data from the Point Loma Ecological Conservation Area (PLECA), an insular urban reserve near San Diego, CA. Our analysis showed that counts of individuals for one of the four most common terrestrial vertebrates declined, whereas counts for other common species increased or remained stable. Two species exhibited declines in adult body length, whereas biomass pooled over the five most common species increased over time and was associated with higher wet season precipitation. Although the habitat and vegetation at PLECA have remained protected and intact, we suspect that changes in arthropod communities may be driving changes in the abundance, growth, and development of insectivorous lizards. This study underscores the value of long-term monitoring for establishing quantitative baselines to assess biological changes that would otherwise go undetected.

Keywords
capture rates, habitat fragmentation, herpetofauna conservation, lizard, long-term monitoring, management, Mediterranean ecosystem, snake, urban ecology

1 | INTRODUCTION

Long-term ecological monitoring has demonstrable value for providing objective scientific information to inform management and decision-making. In recognition of this value, and to provide National Park managers with information on the status and trends of key natural resources, the National Park Service (NPS) Natural Resource Inventory and Monitoring Program was created in 1998 (Fancy, Gross, & Carter, 2009; National Park Service, 2015). Subsequently, more than 270 National Parks were grouped into 32 inventory and monitoring networks based on geography and shared natural resource characteristics (Fancy et al., 2009). Among those 32 networks is...
the Mediterranean Coast Network (MEDN) that includes three parks: Cabrillo National Monument, Channel Islands National Park, and Santa Monica Mountains National Recreation Area.

A reptile and amphibian monitoring program is one of many MEDN monitoring programs, and is of high interest due to population declines associated with anthropogenic (e.g., urban development and exotic species) and environmental stressors (e.g., temperature extremes, drought, ultraviolet radiation; Blaustein & Wake, 1990, Fisher, Suarez, & Case, 2002, Delaney, Riley, & Fisher, 2010, Sinervo et al., 2010, Fisher et al., 2018, Mitrovich, Diffendorfer, Brehme, & Fisher, 2018). Of particular concern is reptile and amphibian persistence at Cabrillo National Monument, and more generally at the Point Loma Ecological Conservation Area (PLECA) of which Cabrillo is part (Figure 1). This is because PLECA is an isolated reserve, or “habitat island,” that is surrounded on three sides by water and is bounded to the north by urban development. Thus, as can happen on habitat islands (Fernández-Juricic & Jokimäki, 2001), herpetofauna on PLECA sharing common ancestry with “mainland” species may now be on an independent evolutionary trajectory due to limited or no gene flow. Of additional concern is evidence that seven species of herpetofauna historically known to occur on PLECA are now extirpated (Fisher, 2004; Fisher et al., 2002).

**FIGURE 1** Point Loma Peninsula, located west-southwest of San Diego, CA, and the boundaries of the Point Loma Ecological Conservation Area (PLECA), Cabrillo National Monument (CABR), and U.S. Navy property. Green circles show the locations of the 16 active herpetofauna trapping arrays; red circles show the locations of the two permanently closed arrays.
Herpetofauna monitoring on PLECA has occurred since 1995 and continues to the present, and over that period several reports have summarized important findings (Atkinson, Fisher, Rochester, & Brown, 2003; Brown & Fisher, 2002; Busteed et al., 2006; Delaney, Riley, Lee, et al., 2010; Fisher & Case, 2000; Yang & Fisher, 2003). These studies have established critical baselines for the presence and numbers of herpetofauna, institutionalized best practices for field methodology and data management, and documented the local extirpation of several species historically known to occur at this site. However, to date no comprehensive modeling of trap data has been undertaken to compare capture rates within and between species, and there has been no formal method established for estimating the probability of detection for rare species. Herein we address these gaps by analyzing the 1995–2016 monitoring data to complete the following objectives:

1. Determine how species-specific capture rates vary over a calendar year.
2. Evaluate whether trends in species-specific counts and body length, and herpetofaunal biomass, are evident.
3. Determine whether variation in temperature or precipitation affect species-specific counts and body length, and herpetofaunal biomass.
4. Estimate the probability of detecting a rare species during monitoring, conditional on the species being present during monitoring.

2  |  METHODS

2.1  |  Study area

PLECA was established as a cooperative land management partnership for conservation of the native undisturbed habitat remaining on Point Loma Peninsula. The reserve is a 6-km long mosaic of development and sensitive natural areas that lies west-southwest of San Diego, CA (Figure 1; Atkinson et al., 2003, Cooperative Conservation America, 2017). Many of the peninsula's natural communities remain intact because of limited development and limited public access to the adjacent U. S. Navy facility.

The Mediterranean climate at PLECA consists of warm dry summers and cool wet winters. Typical annual rainfall ranges from 8.6 to 49.3 cm, with an average of 24.1 cm, and the mean annual temperature is 17.8°C (National Park Service, 2016a, 2016b). Habitats on PLECA are dominated by maritime succulent scrub, coastal sage scrub, and chaparral (Atkinson et al., 2003).

2.2  |  Herpetofauna sampling

Herpetofauna were sampled using 18 pitfall trapping arrays encompassing the range of local habitats (Figure 1). Prior to array placement, PLECA was stratified into eight distinct habitat types and arrays were allocated among habitats in approximate proportion to the total area of each habitat (Busteed et al., 2006). Placement of arrays within habitats was nonrandom and was based on feasibility, access, visibility, and habitat sensitivity (Atkinson et al., 2003). Despite the nonrandom placement within vegetation types, PLECA was small enough for array placement in nearly all locations where it was physically feasible to do so. Nevertheless, because of constraints on sample site selection, the inferences made herein pertain only to the sites sampled and may not be representative of PLECA as a whole.

Trapping arrays were constructed using a three-armed configuration of drift fences with a combination of pitfall and funnel traps (Figure 2), as described in Fisher et al. (2008) and Fisher and Rochester (2012). Pitfall traps were constructed using 19-L (i.e., 5-gal) white plastic buckets buried such that the rim of the bucket was flush with the ground. Food and shelter for trapped animals were placed in each bucket (e.g., polyester or cotton batting, PVC hide tubes, and mealworms). Funnel traps
were cylindrical and were constructed using hardware cloth, as described in Fisher et al. (2008).

### 2.3 Monitoring design

Trapping arrays on PLECA have been in operation since 1995. From 1995 to 2000, there were fewer sampling sessions per year than post 2000, but each sampling session was conducted for a larger number of consecutive nights (Table S1). In contrast, from 2001 to 2016, an emphasis was placed on more sampling sessions per year but arrays were operated for only four consecutive nights. This change was implemented to streamline logistics and accommodate anticipated statistical analyses (Atkinson et al., 2003). In some years, weather or staff shortages prevented the operation of arrays or prevented sampling for all nights. Currently there are only 16 arrays in operation due to the presence of a fragile, listed plant species requiring permanent closure of two arrays (Figure 1). In addition, individual traps within an array were occasionally closed during a sampling session to protect animals from Argentine ants (*Linepithema humile*), which can fatally injure trapped individuals. Typically, the distribution of sampling effort among months within a calendar year (expressed as array-nights; where one array operated for one night = one array-night) varied considerably.

### 2.4 Data collection

Pitfall and funnel traps were opened at the beginning of the sampling session and checked daily, usually in the morning so that animals could be processed and released before temperatures reached levels causing mortality. For captured individuals we recorded the array number, trap number, date, species, sex, age (i.e., adult or juvenile), weight (to the nearest 0.5 g), snout-vent length (mm), and whether it was a new capture or a recapture. In addition, the four most common lizard species were toe clipped in a manner allowing identification of unique individuals (Fisher et al., 2008). All species sampled during the study are listed in Table 1, along with shorthand versions of their common names that we use in the text hereafter. Slender salamanders, snakes, and legless lizards were not marked because they occurred in such low numbers within sampling sessions that distinguishing among new captures and recaptures was not problematic (length and weight were used to disqualify individuals as recaptures within sampling periods). All individuals were released at the point of capture after processing.

**Table 1** The total number of adults and juveniles captured in trapping arrays on PLECA from 1995 through 2016. Throughout the manuscript species are referred to using a shortened version of their common name (bolded)

| Species                     | Common name            | Adults | Juveniles | Total |
|-----------------------------|------------------------|--------|-----------|-------|
| **Amphibians**              |                        |        |           |       |
| Batrachoseps major          | Garden slender salamander | 143    | 23        | 166   |
| **Lizards**                 |                        |        |           |       |
| A. stebbinsi                | Legless lizard         | 7      | 1         | 8     |
| Aspidoscelis hyperythra     | Orange-throated whiptail | 813    | 271       | 1,084 |
| E. multicarinata            | San Diego alligator lizard | 555    | 27        | 582   |
| Sceloporus occidentalis     | Great Basin fence lizard | 1,317  | 804       | 2,121 |
| Uta stansburiana            | California side-blotched lizard | 972    | 869       | 1,841 |
| **Snakes**                  |                        |        |           |       |
| Crotalus oreganus           | Western rattlesnake    | 31     | 9         | 40    |
| Diadophis punctatus         | Western ringneck snake | 9      | 2         | 11    |
| Hypsiglena ochrorhynchus    | Night snake            | 3      | 3         | 6     |
| Lampropeltis californiae    | California kingsnake   | 7      | 19        | 26    |
| Masticophis lateralis       | California striped racer | 76     | 19        | 95    |
| Pituophis catenifer         | Gopher snake           | 14     | 5         | 19    |
| **Total**                   |                        | 3,947  | 2,052     | 5,999 |

Abbreviation: PLEC, Point Loma Ecological Conservation.
2.5 | Statistical analyses

Because amphibians, snakes, and one lizard species (*Anniella stebbinsi*) were not individually marked, and because there were few recaptures of marked lizards within and among sampling sessions, we were not able to estimate abundance using mark-recapture models (e.g., Huggins, 1991). Consequently, we instead used counts adjusted by number of array-nights as an index of abundance. Because counts are a function of both abundance and detection probabilities they may be a biased index of abundance, though if detection probabilities remain relatively constant over time and space counts likely reflect relative abundance.

We determined how species-specific capture rates varied over a calendar year (Objective 1) by summing count and array-night data for the *i*th month, then pooling these monthly data over the years 1995–2016. We then computed monthly capture rates (captures in month *i*/array-nights in month *i*) and plotted these for the more abundant species.

For Objectives 2 and 3, we pooled counts of individuals captured during each year and used this as our response variable. Pooling within calendar years was necessary because of inconsistencies in the number of sessions among years (Table S1), inconsistencies in the timing of sessions among years (Table S2), and small or zero counts for many species within sampling sessions. Similarly, we pooled body mass measurements within each year for slender salamanders, whiptails, alligator lizards, fence lizards, and side-blotched lizards to evaluate overall trends in herpetofaunal biomass (denoted BPUE, where BPUE = total biomass/total array-nights).

We evaluated trends and the effects of temperature and precipitation on counts, body length, and BPUE using generalized linear models (GLIMs; Agresti, 1990). For count data, we assumed annual counts were Poisson distributed and used a log link function. To account for variable sampling effort (i.e., array-nights, Table S1), we used log(array-nights) as an offset. Thus, for count data, we had: $\log(E(y)) = \beta_0 + \beta_1 X_1 + \ldots + \beta_k X_k + \log(\text{array } \chi \text{ nights})$. For body length and BPUE, we assumed errors ($\epsilon$) were normally distributed and used an identity link function, such that: $E(y) = \beta_0 + \beta_1 X_1 + \ldots + \beta_k X_k + \epsilon$. The predictor variables we considered (i.e., the $X_i$), and the candidate model set we evaluated, are presented in Table 2. We note that other predictors we considered, such as a drought index and predictors based on minimum temperatures, were so highly correlated with the variables in Table 2 (correlation $>0.90$) that we excluded them from the analyses to avoid multicollinearity issues. The precipitation and temperature data we used to construct predictor variables for the GLIM analyses were obtained from the San Diego Lindbergh Field weather station located within 5 km (Climate Analyzer, 2018). We assessed trends in precipitation and maximum temperatures over the entire 22-year study using a GLIM assuming normal errors and using an identity link function. For all GLIM analyses, we used SAS Proc Genmod (SAS Institute Inc, 2016) to estimate model coefficients; 95% confidence intervals (CIs) on coefficients were obtained from the profile likelihood function using the likelihood ratio CI option in SAS (i.e., lrci).

For counts, body length, and BPUE, we estimated parameters for each model in the candidate set (Table 2), then used AIC<sub>c</sub> to rank models from those best supported by the data to those least supported by the data (BURNHAM & ANDERSON, 2002). We considered models with a $\Delta$AIC<sub>c</sub> $\leq$ 2 (hereafter referred to as the “top models”) to be competitive with the model having the lowest AIC<sub>c</sub> value (hereafter referred to as the “best model”), where $\Delta$AIC<sub>c</sub> is equal to AIC<sub>c</sub> for the model under consideration minus AIC<sub>c</sub> for the best model. We present only the top models in our results (the best model also belongs to the set of top models). We assessed the degree of support for the top models by computing model weights (w; BURNHAM & ANDERSON, 2002). For the best model, we used the 95% CI on parameter estimates to assess which variables were good predictors for the response variable. Specifically, if the 95% CI did not overlap zero, we

| TABLE 2 | The set of candidate models we used to evaluate the effects of time, temperature, and precipitation on annual counts and body size of herpetofauna. The predictor variables we considered are defined as follows: year = calendar year of sampling; precip11_3 = average monthly wet season precipitation, data are from November of year *(t-1)* through March of year *t*; tmax11_3 = average monthly wet season maximum temperature, data are from November of year *(t-1)* through March of year *t*; and tmax4_10 = average monthly dry season maximum temperature, data are from April of year *t* through October of year *t*. All listed models included an intercept term |
|---|---|
| **Candidate models** | |
| precip11_3 + tmax11_3 + precip11_3 × tmax11_3 null | |
| precip11_3 + tmax11_3 | |
| precip11_3 | |
| tmax11_3 | |
| year + precip11_3 | |
| year + tmax11_3 | |
| year + tmax4_10 | |
| year | |
| tmax4_10 | |
| null (i.e., intercept only) | |
considered the associated variable to be a good predictor. In the results where we show the percent change in counts or body length over the course of the study, these were computed using the model-based expected values over the years 1995–2016 after setting the second covariate in the model and array-nights (the offset for sampling effort, for counts only) equal to their average values.

For estimating the probability of detecting a rare species during monitoring (Objective 4), we assumed that the insularity of PLECA meant that, with the exception of common species that can persist in urban habitats (i.e., slender salamander, alligator lizard, fence lizard), there would be no immigration of individuals into PLECA. Thus, for example, if we did not detect at least one individual of species “A” during monitoring in 1995 or 1996 but did detect one during monitoring in 1997, then under our assumption we can conclude species “A” was present in 1995 and 1996 but it went undetected. In contrast, if we do not detect species “A” during monitoring from 2013 to 2016 (i.e., in the last 4 years of data, there are trailing zeroes), we cannot conclude it was present because we do not know if the species actually went locally extinct. Under this reasoning, we computed the species-specific conditional on presence binomial detection probability (\( p \)) by omitting years with trailing zeroes from the computation. However, because this estimate will be positively biased for species with trailing zeroes if the species actually was present during those years, we also computed the binomial detection probability (\( p' \)) that included those zeroes. For species without trailing zeroes \( p = p' \). For species with trailing zeroes we used \( p \) and \( p' \) in the formula \((1-p')^t\) to bracket the probability of monitoring for \( t \) years and not detecting the species, even though it was present (McArdle, 1990). We note that neither \( p \) nor \( p' \) alone can be used to estimate the probability a species is present (or not present) given it is not detected (Wintle, Walshe, Parris, & McCarthy, 2012).

3 | RESULTS

3.1 | Overall patterns

Over the 22 years of sampling, there were no general trends in wet season precipitation (\( \beta_{year} = -0.115, 95\% \text{ CI } -0.377, 0.147 \)) or average annual monthly precipitation from November of year (\( t-1 \)) through October of year \( t \) (\( \beta_{year} = -0.096, 95\% \text{ CI } -0.380, 0.187 \)), but maximum temperatures increased during both the wet (\( \beta_{year} = 0.015, 95\% \text{ CI } 0.044, 0.249 \)) and dry seasons (\( \beta_{year} = 0.126, 95\% \text{ CI } 0.012, 0.240 \)). Using model-based expected values over the years 1995–2016, we found wet season maximum temperatures increased 3.1°C (4.7%) and dry season maximum temperatures increased 2.7°C (3.7%).

Between 1995 and 2016, we captured 5,999 individuals representing 12 species: 1 amphibian, 5 lizards, and 6 snakes (Table 1). These 12 species are among the 19 species historically known to occur on PLECA (Fisher, 2004). Ninety-eight percent of all captures were from the four most common lizards: whiptail lizards, alligator lizards, fence lizards, and side-blotched lizards. The seven species identified as potentially extirpated from the peninsula (Fisher, 2004) remained undetected through the additional 12 years of survey effort since the publication of those findings in 2004.

3.2 | Long-term trends in counts

For the 12 species captured, there were adequate data to model adult counts for six species. The top models for five of the six species had moderate to high support based on the model weights (\( w \)) and had CIs that did not overlap zero (Table S3). The exception was the striped racer, where multiple top models had small weights and the best model was the null (i.e., intercept only). Because the non-null models for the striped racer were within two AICc units of the null model but had only one additional parameter, they are what Arnold (2010) called “uninformative parameters.” Such parameters do not explain enough variation to justify inclusion in the model and should not be interpreted as having any ecological effect (Arnold, 2010). Parameter estimates for the best models showed that over the 22 years of sampling whiptail counts declined (−27.9%) but counts increased for alligator lizards (61.1%), fence lizards (231.0%), and side-blotched lizards (112.5%; Tables S3 and S4). Wet season precipitation was important for four of the five species, but the direction of the effect was mixed. For slender salamander, there was a synergistic interaction between wet season precipitation (precip11_3) and wet season maximum temperatures (tmax11_3), which exhibited an increase in the expected count as precip11_3 or tmax11_3 increased, and the increase was especially rapid when wet season precipitation exceeded 38 cm (range 5–46 cm) and wet season maximum temperatures exceeded 21°C (range 17–22°C).

For juveniles, there were adequate data to model counts for six species. For juvenile slender salamanders, whiptails, and side-blotched lizards, the large model weights indicated very strong support for the best models whereas model support was moderate for the remaining species. Nevertheless, for juveniles of all species, the parameters for the best models had CIs that did not overlap zero (Table S5). Parameter estimates for the best
models did not show a consistent pattern among species (Table S6). However, like the respective adults, juvenile whiptail counts declined (−57.0%) and juvenile side-blotched lizard counts increased (88.9%). For juvenile slender salamanders, dry season maximum temperatures affected counts, whereas for all remaining species wet season precipitation or maximum temperatures affected counts. For juvenile fence lizards, there was a synergistic interaction between wet season precipitation (precip11_3) and wet season maximum temperatures (tmax11_3) that, except at the lowest levels of precip11_3 and tmax11_3, exhibited an increase in the expected count as precip11_3 or tmax11_3 increased, and the increase was especially rapid when wet season precipitation exceeded 30 cm (range 5–46 cm) and wet season maximum temperatures exceeded 20°C (range 17–22°C).

Patterns of adult and juvenile capture rates through time are summarized in Table 3.

### 3.3 | Long-term trends in body size

For adult males and females, there were adequate data to model body length for the four most common lizards, though whiptail and alligator lizard sex could not be determined reliably so we pooled these data. For all species by sex combinations, the best model had parameters with 95% CIs that did not overlap zero. For alligator lizards and male fence lizards, the top model had relatively large model weights, whereas for the remaining species by sex combinations there was only moderate support for the best model (Table S7). Our analysis showed that over the 1995–2016 sampling period, the change in body length was −7.5 mm (−6.1%) for alligator lizards (sexes pooled), −5.9 mm (−9.3%) for female fence lizards, and −4.5 mm (−7.0%) for male fence lizards. For whiptails, male fence lizards, and side-blotched lizards wet season precipitation was in the top model and in all cases the parameter was positive (Table S8), indicating body length was directly proportional to wet season precipitation. Maximum temperatures appeared in the top model for three of the species by sex combinations, but the signs on the parameter estimates were mixed (Table S8). Adult body length patterns through time are summarized in Table 3.

Our analysis of BPUE (Table S9) indicated that herpetofaunal biomass increased over time ($\beta_{\text{year}} = .073$, 95% CI 0.029, 0.117) and was positively associated with wet season precipitation ($\beta_{\text{precip11}_3} = .125$, 95% CI 0.052, 0.197).

### 3.4 | Probability of detecting rare species

For each year of monitoring, we tabulated the years that at least one individual of each species was detected. For five species (whiptail, alligator lizard, striped racer, fence lizard, and side-blotched lizard), at least one individual was detected every year. For the remaining seven species, the years of detection are shown in Table 4. We refer to these seven species as rare because, with the exception of slender salamander, fewer than 41 individuals were detected over the 22 years of monitoring.

The conditional on presence detection probabilities ($p$) are presented in Table 4. For legless lizards, ringneck snakes, and kingsnakes, there were trailing zeroes and $p$ was .44, .44, and .67, respectively, whereas under the assumption, these species were present during all years the conditional detection probability $p'$ was .36, .36, and .64, respectively. Using these values, we estimated that the probability of monitoring for $t$ years and not detecting the species even though it was present was between 0.098 and 0.168 for legless lizards and ringneck snakes, and was between 0.333 and 0.364 for kingsnakes.

### 3.5 | Capture rate patterns over a calendar year

We had adequate data to assess capture rate patterns over a calendar year (Objective 1) for the adults of six species

| Species               | Adult counts by year | Juvenile counts by year | Adult body length by year |
|-----------------------|----------------------|-------------------------|---------------------------|
| Slender salamander    | NA                   | Increased               | NA                        |
| Whiptail              | Decreased            | Decreased               | NA                        |
| Alligator lizard      | Increased            | NA                      | Decreased                 |
| Fence lizard          | Increased            | NA                      | Decreased                 |
| Side-blotched lizard  | Increased            | Increased               | NA                        |

Abbreviation: NA, not applicable.

**Table 3** Overall patterns of temporal change over 22 years of monitoring for species with a year effect in the best AICc model. The adult counts column summarizes Tables S3 and S4, the juvenile counts column summarizes Tables S5 and S6, and the adult body length column summarizes Tables S7 and S8.
and the juveniles of four of those species (Figure S1). For adult slender salamanders, whiptails, striped racers, side-blotched lizards, and fence lizards, plots were generally unimodal, whereas for alligator lizards, the plot appeared to have both a dominant and minor mode. The dominant mode for adults generally occurred during the spring or early summer. For juvenile whiptails and fence lizards, capture rates appeared bimodal with the dominant mode shifted later in time relative to the dominant adult mode (Figure S1). Data for juvenile alligator lizards were too sparse to exhibit strong patterns.

### Discussion

Our analysis of 22-years of herpetological monitoring data at PLECA provided several insights regarding the herpetological community occupying this insular conservation area. We identified temporal and climate (i.e., temperature and precipitation) variables associated with herpetofauna counts and body size at PLECA, gained insights into species capture rate patterns over a calendar year, and estimated conditional detection probabilities for some of the rarer species inhabiting PLECA. Below we discuss these patterns in greater detail.

#### 4.1 Impacts of temperature and precipitation

We found strong patterns of year-to-year variation in counts and body length associated with temperature and precipitation (Tables S3–S8). Given the Mediterranean climate, our a priori expectation was that herpetofaunal reproduction and growth, and thus abundance and average size, would increase in wetter and warmer years. This is because vegetation growth and arthropod abundance associated with coastal sage scrub increase rapidly following winter rains (Gray & Schlesinger, 1981; Longcore, 1999), and in arid climates arthropod abundance and

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**Table 4** For species considered rare (see text), the years that at least one individual of that species was detected (1) or not detected (0) on PLECA, and the estimated annual probability of detection conditional on presence (p)

| Year | Legless lizard | Slender salamander | Rattlesnake | Ringneck snake | Night snake | Kingsnake | Gopher snake |
|------|----------------|--------------------|-------------|----------------|-------------|-----------|-------------|
| 1995 | 1              | 0                  | 1           | 0              | 0           | 0         | 0           |
| 1996 | 0              | 1                  | 0           | 0              | 0           | 0         | 0           |
| 1997 | 0              | 1                  | 1           | 1              | 0           | 0         | 1           |
| 1998 | 1              | 1                  | 1           | 1              | 0           | 1         | 1           |
| 1999 | 0              | 1                  | 0           | 0              | 0           | 1         | 1           |
| 2000 | 0              | 1                  | 1           | 1              | 0           | 0         | 1           |
| 2001 | 1              | 1                  | 1           | 1              | 0           | 0         | 1           |
| 2002 | 0              | 1                  | 1           | 0              | 0           | 0         | 0           |
| 2003 | 0              | 1                  | 1           | 1              | 0           | 1         | 1           |
| 2004 | 0              | 1                  | 1           | 0              | 0           | 1         | 0           |
| 2005 | 0              | 1                  | 1           | 0              | 0           | 1         | 1           |
| 2006 | 1              | 0                  | 1           | 0              | 0           | 1         | 1           |
| 2007 | 0              | 1                  | 1           | 1              | 1           | 1         | 0           |
| 2008 | 1              | 1                  | 1           | 0              | 0           | 1         | 1           |
| 2009 | 1              | 1                  | 1           | 1              | 0           | 1         | 1           |
| 2010 | 0              | 1                  | 1           | 1              | 1           | 1         | 0           |
| 2011 | 1              | 1                  | 1           | 0              | 1           | 1         | 1           |
| 2012 | 1              | 1                  | 1           | 1              | 0           | 1         | 0           |
| 2013 | 0              | 1                  | 1           | 0              | 0           | 1         | 0           |
| 2014 | 0              | 1                  | 1           | 0              | 1           | 1         | 1           |
| 2015 | 0              | 1                  | 1           | 0              | 1           | 1         | 1           |
| 2016 | 0              | 1                  | 1           | 0              | 1           | 0         | 1           |

*p* .44 .91 .91 .44 .27 .67 .64

Abbreviation: PLEC, Point Loma Ecological Conservation.
lizard growth and reproduction increase with high levels of precipitation (Ballinger, 1977; Dunham, 1978, 1980; Parker & Pianka, 1975).

We observed higher counts of adult slender salamanders during warmer and wetter years, likely because slender salamanders tend to move on the surface exclusively at night when it is wet and are not active on cold or dry nights (Cunningham, 1960). Thus, the positive association between counts and temperature and precipitation likely reflects increased detectability rather than increased abundance.

Temperature and precipitation effects on lizard counts and body length were less clear-cut. We found wetter winters were associated with larger body lengths of adult whiptail lizards, male fence lizards, and adult side-blotched lizards, and higher counts of adult and juvenile alligator lizards, adult fence lizards, and juvenile side-blotched lizards. Conversely, counts of adult side-blotched lizards were higher following drier winters. Adult fence lizards are only slightly larger than adult side-blotched lizards and their life history and ecological niche are similar, so perhaps interference competition caused an inverse relationship between side-blotched lizard and fence lizard counts (e.g., Dunham, 1980). Alternatively, the lack of fire within the reserve may be creating closed canopy scrublands that are better for fence lizards while simultaneously reducing the open patches preferred by side-blotched lizards, which respond positively to disturbances creating a more open canopy (Rochester et al., 2010).

Although whiptail lizard counts were not associated with precipitation, counts were greater following warmer winters. Members of its genus (Aspidoscelis) are specialized for living in very dry desert regions and exhibit one of the highest active body temperatures of all lizard genera (Brattstrom, 1965; Schall, 1977). Thus, whiptails living in our comparatively temperate Mediterranean community may grow and reproduce more readily in warmer years.

### 4.2 Temporal patterns in counts and body length

With the exception of whiptails, counts for the most common species were stable or increasing over our sampling period even with the decreases in years with unfavorable climatic conditions. Likewise, BPUE for the most common species increased over the sampling period (Table S9). However, adult body length showed a long-term decline in two species—a pattern that could indicate deterioration of habitat if growth and adult body length are primarily driven by resource abundance (i.e., Sparkman et al., 2018). Taken together, these patterns may indicate a shift in the ecological community favoring smaller, less active species (i.e., not whiptails), along with accelerated life histories (i.e., achieving sexual maturity more quickly at smaller body sizes and producing more offspring over a shorter lifespan). We note that because BPUE increased over the study period, the declines in whiptail counts and declines in adult length of alligator and fence lizards appear to be more than offset by the increased counts of alligator lizards, fence lizards, and side-blotched lizards.

The long-term decrease in counts of adult and juvenile whiptails we observed, first hinted at by Atkinson et al. (2003), was not entirely unexpected. Orange-throated whiptails (Aspidoscelis hypelrythra) are active insectivores that forage widely in open habitats but avoid crossing larger paved roads (Brehme, Tracey, McLenaghan, & Fisher, 2013). Although home ranges are only ~600 m² (Bostic, 1965), individuals typically move 400–500 m in a given day in the active season (Karasov & Anderson, 1984). As teiid lizards, they have much higher activity rate and energy requirements than similarly sized phrynosomatid lizards (Karasov & Anderson, 1984), such as fence lizards and side-blotched lizards, and so smaller habitat fragments like PLECA may lack sufficient resources to maintain large populations. It is also possible that top-down factors differentially affect whiptail mortality—for example, their increased movement and activity could make them more vulnerable to introduced predators, such as feral cats. Hence, whiptails could serve as an indicator species for habitat condition in small fragments in southern California (Brehme et al., 2013), and we recommend that count and body length data from other monitoring sites be used for comparable analyses.

In contrast, side-blotched lizards, fence lizards, and alligator lizards (which exhibited increased counts across years) are much less active than whiptails, leading to lower energy costs and resource needs (Karasov & Anderson, 1984; Nagy, 1982). Following this logic, alligator lizards would be predicted to do especially well in small fragments; field metabolic rates for Elgaria multicarinata are among the lowest reported for lizards (Kingsbury, 1995). Indeed, of all the lizards that persist in urban fragments in southern California, alligator lizards are among the most common species across sites (they were one of two species found in all sites analyzed by Case and Fisher (2001)) and persist even in heavily urbanized settings such as backyards and college campuses (Spear, Pauly, & Kaiser, 2017).

Even though the counts for four of the five most common species at PLECA were stable or increasing, the insular nature of PLECA has impacted these populations...
in other ways. Genetic analysis of fence lizards and whiptail lizards (and mice in the genus *Peromyscus*) found a decrease in genetic diversity within PLECA (Lion, Rice, & Clark, 2018; Luckau, 2015), and similar patterns have been found for these species in other urbanized habitats (Delaney, Riley, & Fisher, 2010).

The decrease in adult body length of alligator lizards and fence lizards over the sampling period was unexpected. Because we found a positive association of adult length with temperature and precipitation, we suspect the observed long-term decrease may be due to declining growth rates associated with a degraded environment. Studies on the energetics and growth of lizards support this idea. Individuals with high availability of food grow quickly and attain larger sizes sooner, as well as larger sizes at time of maturity and reproduction (Adolph & Porter, 1996; Ballinger, 1977; Dunham, 1978; Iraeta, Monasteria, Salvador, & Diaz, 2006). Although reptiles arguably experience indeterminate growth, body growth slows drastically following sexual maturity, and smaller average sizes of adults are associated with limited resources (Andrews, 1982; Iraeta et al., 2006; Litzgus & Brooks, 1998; Sparkman et al., 2018; Thornill, 1982).

### 4.3 Capture rate patterns over a calendar year

For the more common species, we assessed seasonal capture rates qualitatively by plotting monthly capture rates over a calendar year (Figure S1). As is typical for reptiles and amphibians in seasonal habitats, each species showed clear peaks in capture rates at certain times of year that were likely a function of both abundance and increased activity, the latter of which would tend to drive up capture probabilities. Slender salamanders, which readily desiccate when on the surface and live largely subterranean lives, were captured almost exclusively during the November–March wet season, whereas lizards and snakes were generally captured more in late spring and early summer—after the cooler winter months but prior to the onset of the hot, dry autumn period. Juvenile lizards were captured most in late fall due to recruitment of hatchlings incubated as eggs over the summer; all four lizard species are oviparous with courtship and mating generally taking place in spring and early summer, oviposition soon thereafter, and hatching in the late summer to fall (Dunham & Miles, 1985; Parker & Pianka, 1975; Stebbins, 2003). Lizard growth rates are variable and dependent on environmental conditions and food availability (Andrews, 1982; Dunham, 1978; Sinervo & Adolph, 1994), so capture of juveniles outside the typical fall hatching period likely reflects individuals that were hatched the previous year but have not yet attained adult body size. Additionally, side-blotched lizards and fence lizards can lay more than one clutch of eggs per year (Dunham & Miles, 1985), which is likely why these species showed broader seasonal capture rates of juveniles (Figure S1).

### 4.4 Probability of detecting rare species

By definition, rare species in this study were caught infrequently, thus data beyond detected/not detected is limited. Nevertheless, we were able to use detection data to calculate conditional detection probabilities (Table 4) and estimate the probability a species was present but undetected. For example, we estimated the probability that ringneck snakes and legless lizards were present but went undetected from 2013 through 2016 was between 0.098 and 0.168, which suggests more intensive surveys for these species might be warranted to rule out local extirpation. Continued herpetofaunal surveys at PLECA will allow us to further refine conditional detection probability estimates.

### 4.5 Persistence in an insular reserve

It is possible that a relatively diverse herptile community could be maintained indefinitely at PLECA, despite its insular nature. A nearby offshore oceanic island, Islas Coronados Sur, has been separated from the mainland for at least 10,000 years and is smaller in area (~200 ha) than the protected habitat at the tip of PLECA (~300 ha). Islas Coronados Sur contains apparently stable populations of herpetofauna including slender salamanders, legless lizards, side-blotched lizards, skinks, tiger whiptail lizards, alligator lizards, night snakes, and rattlesnakes (Grismer, 2002). Thus, in principle, the herptile community at PLECA could adapt over time and persist as an isolated community. However, the edge to interior ratio of PLECA is more than three times that of Islas Coronados Sur, and PLECA is subject to factors like human disturbance, road mortality, light pollution, and invasive species that are absent on Islas Coronados Sur. Management and mitigation of these factors would be necessary to facilitate stabilization of the herptofaunal community at its new “relaxed” species richness (Fisher, 2004).

### 4.6 Isolation and body size

It is notable that many of the reptile species endemic to Islas Coronados Sur are smaller in body size than their
mainland counterparts, including the rattlesnake (Klauber, 1949), whiptail lizard (Walker, 1981), and alligator lizard (Fitch, 1934), which supports our speculation regarding resource availability and body length (Sparkman et al., 2018). If we are in fact correct, then this is concerning for the continued stability of the herpetofauna at PLECA because it suggests a general decline in resource availability over the 22 years of monitoring.

There are several interacting factors that could be driving a decline in resources, but perhaps the most critical are isolation and fragmentation from urbanization of the San Diego region, which has isolated PLECA to such a degree it is effectively a habitat island with no realistic possibility of reconnection with other habitat fragments. Population density often decreases in small fragments (Connor, Courtney, & Yoder, 2000), and experimental studies have demonstrated that arthropod abundance is negatively impacted by fragmentation and isolation (Bender, Contreras, & Fahrig, 1998; Bolger, Saurez, Crooks, Morrison, & Case, 2000; Didham, Hammond, Lawton, Eggleton, & Stork, 1998). Because all the lizards at PLECA are generalist insectivores, we would expect ongoing declines in terrestrial arthropod abundance to lead to declines in body length like those we detected over the 22-year sampling effort.

Because our data are observational, they should be taken only as a guide to develop hypotheses concerning changes in body size. The mechanisms underlying variation in growth and size are complex and may involve interactions between factors like food availability (Dunham, 1978), thermal regime (Sinervo & Adolph, 1994), and predation risk (Mangel & Stamps, 2001). Experimental manipulations will be necessary to determine why adult body sizes of some species are decreasing in this increasingly insular population.

4.7 Impacts of temperature and precipitation

It is possible that changing weather patterns may be compounding the impacts of fragmentation. Whereas we found no trends in precipitation during our study, wet and dry season maximum temperatures increased over the course of the study. This is consistent with more detailed climatological analyses for southern California that show a clear trend of recent warming, as well as increased precipitation variability (Cayan, Maurer, Dettinger, Tyree, & Hayhoe, 2008; Kelly & Goulden, 2008; Vicuna & Dracup, 2007).

Animals living in completely isolated fragments like PLECA have little opportunity to migrate, and therefore must adapt to fragmentation effects, changing climate, and other impacts through natural selection, or become locally extinct. It is unclear whether enough genetic diversity remains for adaptation to occur, as recent genetic analyses show that populations at PLECA have lower genetic diversity than at other sites in San Diego County (Lion et al., 2018; Luckau, 2015). Similar results were found in other southern California species in coastal habitat patches (Barr et al., 2015; Delaney, Riley, & Fisher, 2010; Ruell et al., 2012; Vandergast et al., 2009).

4.8 Summary and future directions

The ongoing monitoring of herpetofauna at PLECA has yielded a valuable dataset and we believe it is important that monitoring continues. Earlier analyses of the data found seven of 19 species historically present at PLECA were absent (Atkinson et al., 2003; Fisher, 2004), and we found those species continue to remain undetected. Our analysis also indicated that ringneck snakes and legless lizards have not been detected in recent years, and additional years of monitoring are required to determine if they have been locally extirpated. If these species have become locally extirpated in recent years, that could be an indication that the local herpetofaunal community has not stabilized, and could continue to decline without more active management interventions. An additional benefit of continued monitoring is the early detection of invasions by other herpetofaunal species (e.g., released pets) that might threaten the integrity of the reserve (Difffendorfer, Rochester, Fisher, & Brown, 2005; Fisher, DelPinto, & Fisher, 2020; Palmer & Fisher, 2010; Winkleman & Backlin, 2016). At least one of these species the brown anole (Anolis sagrei) has been shown to potentially displace fence lizards in southern California when established (Fisher et al., 2020).

One snake species, four lizards, and one amphibian remain relatively abundant at PLECA. Of these, the snake is too rare for quantitative analysis, one of the lizards is in decline, and two of the remaining lizard species are not growing as large. Additional work is needed to determine the taxonomic breadth of these trends at PLECA, and which of a host of potential ecological threats (spread of invasive species, changing weather patterns, light pollution, declines in arthropod abundance, loss of genetic diversity, increased human usage, etc.) are contributing to local loss of biodiversity (Fisher et al., 2002; Menke, Holway, Fisher, & Jetz, 2009; Mitrovich et al., 2018; Perry, Buchanan, Fisher, Salmon, & Wise, 2008; Perry & Fisher, 2006).

Incorporating our findings into the suite of cost-effective management objectives established previously for
this reserve would complement ongoing conservation efforts in the following ways:

1 Our analyses could be used to establish quantitative decision triggers for specific conservation actions at PLECA, and those triggers could be evaluated annually during continued herpetofauna monitoring. Specific actions and trigger points would be decided in consultation with managers at both Cabrillo National Monument and the Point Loma Ecological Reserve in order to reflect the resources and regulations constraining conservation efforts.

2 Analyses of other taxonomic groups being monitored at PLECA could be used to complement our analysis of the herpetofauna. Doing this would allow managers to gain a broader view of the extent to which local biodiversity may be in decline and may help determine underlying causal mechanisms.

3 Additional efforts undertaken to periodically survey arthropod diversity and abundance would be useful for directing research efforts and may help establish a potential causal link between resource availability for insectivores and insectivore abundance and growth rate. Baseline surveys of arthropod diversity were conducted ~15 years ago, so updated surveys could also assess changes in the arthropod community.

4 Our study found that current monitoring techniques do not result in sufficient recapture data that would be necessary for more sophisticated estimates of key demographic parameters, such as population size and survivorship. To address this shortcoming, it would be useful to supplement pitfall trap monitoring with genomic monitoring. Tissue sampling and database management is already built into the monitoring plan, and with support for genomic analyses these samples could be used to establish effective population sizes, genetic diversity, and local adaptation. Combined with ongoing ecological monitoring, genomic analyses are likely to become a cornerstone of future conservation efforts.

ACKNOWLEDGMENTS
The authors thank L. Lee for providing data and background materials on the project, and S. Hathaway and S. Weber for help finding and managing the volunteer crews when monitoring began in 1995. The authors also thank the NPS “Volunteers in Parks” who dedicated their time and effort assisting with the fieldwork associated with this project; and the U.S. Navy for their partnership on this study and allowing the NPS to conduct work on military property. B. Halstead and a second reviewer provided excellent comments that helped us improve the manuscript. This is Contribution Number 743 of the U.S. Geological Survey Amphibian Research and Monitoring Initiative (ARMI). Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

CONFLICT OF INTEREST
The authors declare no potential conflict of interest.

AUTHOR CONTRIBUTIONS
Thomas R. Stanley took the lead on data analyses and contributed to preparation of the original draft manuscript. Rulon W. Clark contributed to data analyses and preparation of the original draft manuscript. Robert N. Fisher was instrumental in conceiving and implementing the project, developing methodology, and contributing to project administration, funding acquisition, and supervision. Carlton J. Rochester contributed to project implementation, data curation, project administration, software development, data analyses, and data visualization. Stephanie A. Root contributed to data curation, project implementation, and data visualization. Keith J. Lombardo provided project administration. Stacey D. Ostermann-Kelm contributed to project conceptualization, funding acquisition, and project administration. All authors contributed to writing, reviewing, and editing the manuscript.

DATA AVAILABILITY STATEMENT
Data will be made available to all interested researchers upon request.

ETHICS STATEMENT
The National Park Service Institutional Animal Care and Use Committee (IACUC) does not require a formal protocol for terrestrial herpetological monitoring activities. Nevertheless, this study was carried out in strict accordance with the ethical recommendations outlined in the American Society of Ichthyologists and Herpetologists Guidelines for the use of Live Amphibians and Reptiles in Field and Laboratory Research, 2004.

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REFERENCES
Adolph, S. C., & Porter, W. P. (1996). Growth, seasonality, and lizard life histories: Age and size at maturity. *Oikos*, **77**, 267–278. https://doi.org/10.2307/3546065
Agresti, A. (1990). *Categorical data analysis*. New York, NY: John Wiley & Sons.
Andrews, R. M. (1982). Patterns of growth in reptiles. In C. Gans & F. H. Pough (Eds.), *Biology of the reptilia* (vol 13, pp. 273–320). New York: Academic Press.
Arnold, T. W. (2010). Uninformative parameters and model selection using Akaïke’s information criterion. *Journal of Wildlife Management, 74*, 1175–1178. https://doi.org/10.2193/2009-367

Atkinson, A. J., Fisher, R. N., Rochester, C. J., & Brown, C. W. (2003). Sampling design optimization and establishment of baselines for herpetofauna arrays at the Point Loma Ecological Reserve. Report. USGS Western Ecological Research Center. 39.

Ballinger, R. E. (1977). Reproductive strategies: Food availability as a source of proximal variation in a lizard. *Ecology, 58*, 628–635. https://doi.org/10.2307/1939102

Barr, K. R., Kus, B. A., Preston, K. L., Howell, S., Perkins, E., & Vandergast, A. G. (2015). Habitat fragmentation in coastal southern California disrupts genetic connectivity in the cactus wren (*Campylorhynchus brunneicapillus*). *Molecular Ecology, 24*, 2349–2363.

Bender, D. J., Contreras, T. A., & Fahrig, L. (1998). Habitat loss and population decline: A meta-analysis of the patch size effect. *Ecology, 79*, 517–533. https://doi.org/10.1890/0012-9658(1998)079[0517:HLAPDA]2.0.CO;2

Blaustein, A. R., & Wake, D. B. (1990). Declining amphibian populations: A global phenomenon? *Trends in Ecology and Evolution, 5*(7), 203–204. https://doi.org/10.1016/0169-5347(90)90129-2

Bolger, D. T., Saurez, A. V., Crooks, K. R., Morrison, S. A., & Case, T. J. (2000). Arthropods in urban habitat fragments in southern California: Area, age, and edge effects. *Ecological Applications, 10*, 1230–1248.

Bostic, D. L. (1965). Home range of the teiid lizard, *Cnemidophorus hyperythrus beldingi*. *The Southwest Naturalist, 10*, 278–281. https://doi.org/10.2307/3669303

Brattstrom, B. H. (1965). Body temperatures of reptiles. *American Midland Naturalist, 73*, 376. https://doi.org/10.2307/2423461

Brehme, C. S., Tracey, J. A., McLenaghan, L. R., & Fisher, R. N. (2013). Permeability of roads to movement of scrubland lizards and small mammals. *Conservation Biology, 27*, 710–720. https://doi.org/10.1111/cobi.12081

Brown, C. W., & Fisher, R. N. (2002). Inventory and management needs study of point Loma Herpetofauna (reptiles and amphibians) with comments on mammals and invertebrates, 2001. Report. US Geological Survey Technical Report. 33.

Burnham, K. P., & Anderson, D. R. (2002). *Model selection and multimodel inference: A practical information-theoretic approach* (2nd ed.). New York, NY: Springer-Verlag.

Busteed, G., Cameron, J. L., Robertson, M., Riley, S. P. D., Lee, L., Compton, A., ... Duffield, T. (2006). Monitoring of Terrestrial Reptiles & Amphibians in the Mediterranean Coast Network: Santa Monica Mountains & Cabrillo National Monument, Version 1.0. Natural Resources Technical Report NPS/MEDN/NRTR–2006/005. National Park Service, Thousand Oaks, CA.

Case, T. J., & Fisher, R. N. (2001). Measuring and predicting species presence: Coastal sage scrub case study. In C. T. Hunsaker, M. F. Goodchild, M. A. Friedl, & T. J. Case (Eds.), *Spatial uncertainty in ecology* (pp. 1–25). New York, NY: Springer.

Cayan, D. R., Maurer, E. P., Dettinger, M. D., Tyree, M., & Hayhoe, K. (2008). Climate change scenarios for the California region. *Climatic Change, 87*(Suppl 1), 21–42. https://doi.org/10.1007/s10584-007-9377-6

Climate Analyzer. (2018). San Diego Lindbergh Field. Retrieved from http://www.climateanalyzer.org/medn/ghcn/san_diego_lindbergh/

Connor, E. F., Courtney, A. C., & Yoder, J. M. (2000). Individuals-area relationships: The relationship between animal population density and area. *Ecology, 81*, 734–748. https://doi.org/10.1890/0012-9658(2000)081[0734:1ARTRB]2.0.CO;2

Cooperative Conservation America. (2017). Point Loma Ecological Reserve. Retrieved from http://www.cooperativeconservation-america.org/viewproject.aspx?id=409.

Cunningham, J. D. (1960). Aspects of the ecology of the slender salamander, *Batrachoseps pacificus*, in Southern California. *Ecology, 41*, 88–99. https://doi.org/10.2307/1931942

Delaney, K. S., Riley, S. P., & Fisher, R. N. (2010). A rapid, strong, and convergent genetic response to urban habitat fragmentation in four divergent and widespread vertebrates. *PLoS One, 5*, e12767. https://doi.org/10.1371/journal.pone.0012767

Delaney, K. S., Riley, S. P. D., Lee, L., Pister, B., Fitting, H., & Ostermann-Kelm, S. (2010). Monitoring terrestrial reptiles and amphibians in the Mediterranean Coast network, Santa Monica Mountains National Recreation Area, Cabrillo National Monument and Channel Islands National Park. Natural Resource Data Series NPS/MEDN/NRDS–2011/135. National Park Service, Fort Collins, CO.

Didham, R. K., Hammond, P. M., Lawton, J. H., Eggleton, P., & Stork, N. E. (1998). Beetle species responses to tropical forest fragmentation. *Ecological Monographs, 68*, 295–323. https://doi.org/10.1890/0012–9658(1998)068[0295:BSRTTF]2.0.CO;2

Diffendorfer, J. E., Rochester, C. J., Fisher, R. N., & Brown, T. K. (2005). Movement and space use by coastal rosy boas (*Lichanura trivirgata roseofusca*) in coastal southern California. *Journal of Herpetology, 39*, 24–36.

Dunham, A. E. (1978). Food availability as a proximate factor influencing individual growth rates in the iguand lizard *Sceloporus merriami*. *Ecology, 59*, 770–778. https://doi.org/10.2307/1938781

Dunham, A. E. (1980). An experimental study of interspecific competition between the iguand lizards *Sceloporus merriami* and *Urosaurus ornatus*. *Ecological Monographs, 50*, 309–330. https://doi.org/10.2307/2937254

Dunham, A. E., & Miles, D. B. (1985). Patterns of covariation in life history traits of squamate reptiles: The effects of size and phylogeny reconsidered. *American Naturalist, 126*, 231–257. https://doi.org/10.1086/284411

Fancy, S. G., Gross, J. E., & Carter, S. L. (2009). Monitoring the condition of natural resources in US National Parks. *Environmental Monitoring and Assessment, 151*, 161–174. https://doi.org/10.1007/s10661–008–0257–y

Fernández-Juricic, E., & Jokimäki, J. (2001). A habitat island approach to conserving birds in urban landscapes: Case studies from southern and northern Europe. *Biodiversity and Conservation, 10*, 2023–2043. https://doi.org/10.1023/A:1013133038987

Fisher, R. N. (2004). Life on the island: Animals. In R. Houk (Ed.), *Understanding the life of point Loma* (pp. 112–131). San Diego, CA: Cabrillo National Monument Foundation.
Lion, K. A., Rice, S. E., & Clark, R. W. (2018). Genetic patterns in Sauria species in Southern California. Journal of Mammalogy, 99, 923–934.

Litzgus, J. D., & Brooks, R. J. (1998). Growth in a cold environment: Body size and sexual maturity in a northern population of spotted turtles, Clemmys guttata. Canadian Journal of Zoology, 76, 773–782.

Longcore, T. R. (1999). Terrestrial arthropods as indicators of restoration success in coastal sage scrub. (Dissertation). University of California Los Angeles. Los Angeles, CA.

Luckau, T. (2015). Comparative conservation genetics of two sympatric lizard species across multiple landscapes in San Diego County. (Thesis). San Diego State University. San Diego, CA.

Mangel, M., & Stamps, J. (2001). Trade-offs between growth and mortality and the maintenance of individual variation in growth. Evolutionary Ecology Research, 3, 583–593.

McAdie, B. H. (1990). When are rare species not there? Oikos, 57, 276–277.

Menke, S. B., Holway, D. A., Fisher, R. N., & Jetz, W. (2009). Characterizing and predicting species distributions across environments and scales: Argentine ant occurrences in the eye of the beholder. Global Ecology and Biogeography, 18, 50–63.

Mitrovich, M. J., Diffendorfer, J. E., Brehme, C. S., & Fisher, R. N. (2018). Effects of urbanization and habitat composition on site occupancy of two snakes using regional monitoring data from southern California. Global Ecology and Conservation, 15, e00427. https://doi.org/10.1016/j.gecco.2018.e00427

Nagy, K. A. (1982). Energy requirements of free living iguanid lizards. In G. M. Burghardt & A. S. Rand (Eds.), Iguanas of the world behavior, ecology, and evolution (pp. 49–59). Park Ridge, NJ: Noyes Publications.

National Park Service. (2015). Inventory and Monitoring Program Brief. Retrieved from https://science.nature.nps.gov/im/assets/docs/IM_Program_Brief.pdf.

National Park Service. (2016a). Cabrillo - Climate. Retrieved from https://www.nps.gov/cabr/learn/nature/climate.htm.

National Park Service. (2016b). Inventory & Monitoring (I&M). Retrieved from https://science.nature.nps.gov/im/units/medn/.

Palmer, D. D., & Fisher, R. N. (2010). Geographic distribution: Ramphotyphlops braminus. Herpetological Review, 41, 518.

Parker, W. S., & Pianka, E. R. (1975). Comparative ecology of populations of the lizard Uta stansburiana. Copeia, 1975, 615–632.

Perry, G., Buchanan, B. W., Fisher, R. N., Salmon, M., & Wise, S. E. (2008). Effects of artificial night lighting on amphibians and reptiles in urban environments. Herpetological Conservation, 3, 239–256.

Perry, G., & Fisher, R. N. (2006). Night lights and reptiles: Observed and potential effects. In C. Rich & T. Longcore (Eds.), Ecological consequences of artificial night lighting (pp. 169–191). Washington, DC: Island Press.

Rocha, C. J., Brehme, C. S., Clark, D. R., Stokes, D., Hathaway, S. A., & Fisher, R. N. (2010). Reptile and amphibian responses to large-scale wildfires in southern California. Journal of Herpetology, 44, 333–351.

Ruell, E. W., Riley, S. P., Douglas, M. R., Antolin, M. F., Pollinger, J. R., Tracey, J. A., … Crooks, K. R. (2012). Urban habitat fragmentation and genetic population structure of bobcats in coastal southern California. American Midland Naturalist, 168, 265–280.

SAS Institute Inc. (2016). Base SAS® 9.4 procedures guide: Statistical procedures (6th ed.). Cary, NC: SAS Institute.

Schall, J. J. (1977). Thermal ecology of five sympatric species of Cnemidophorus (Sauria: Teiidae). Herpetologica, 33, 261–272.

Sinervo, B., & Adolph, S. C. (1994). Growth plasticity and thermal opportunity in Sceloporus lizards. Ecology, 75, 776–790.

Sinervo, B., Mendez-de-la-Cruz, F., Miles, D. B., Heulin, B., Bastiaans, E., Cruz, M. V. S., … Sites, J. W., Jr. (2010). Erosion of lizard diversity by climate change and altered thermal niches. Science, 328, 894–899.
Sparkman, A. M., Clark, A. D., Brummett, L. J., Chism, K. R., Combrink, L. L., Kabey, N. M., & Schwartz, T. S. (2018). Convergence in reduced body size, head size, and blood glucose in three island reptiles. *Ecology and Evolution*, 8, 6169–6182. https://doi.org/10.1002/ece3.4171

Spear, D. M., Pauly, G. B., & Kaiser, K. (2017). Citizen science as a tool for augmenting museum collection data from urban areas. *Frontiers in Ecology and Evolution*, 5, 86. https://doi.org/10.3389/fevo.2017.00086

Stebbins, R. C. (2003). *A field guide to Western reptiles and amphibians* (3rd ed), Boston, MA: Houghton Mifflin Company.

Thornill, G. M. (1982). Comparative reproduction of the turtle, *Chrysemys scripta elegans*, in heated and natural lakes. *Journal of Herpetology*, 16, 347.

Vandergast, A. G., Lewallen, E. A., Deas, J., Bohonak, A. J., Weissman, D. B., & Fisher, R. N. (2009). Loss of genetic connectivity and diversity in urban microreserves in a southern California endemic Jerusalem cricket (*Orthoptera: Stenopelmatidae: Stenopelmatus n. sp*). *Journal of Insect Conservation*, 13, 329–345.

Vicuna, S., & Dracup, J. A. (2007). The evolution of climate change impact studies on hydrology and water resources in California. *Climatic Change*, 82, 327–350.

Walker, J. M. (1981). A new subspecies of *Cnemidophorus tigris* from South Coronado Island, Mexico. *Journal of Herpetology*, 15, 193–197.

Winkleman, R. S., & Backlin, A. R. (2016). *Aspidoscelis flagellicauda/sonorae* complex (spotted whiptail). *Herpetological Review*, 47, 256–257.

Wintle, B. A., Walshe, T. V., Parris, K. M., & McCarthy, M. A. (2012). Designing occupancy surveys and interpreting non-detection when observations are imperfect. *Diversity and Distributions*, 18, 417–424.

Yang, B. S., & Fisher, R. N. (2003). Surveys for rare reptiles at Cabrillo National Monument on the Point Loma Peninsula in San Diego, California. Technical Report. US Geological Survey, 15.

**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of this article.

**How to cite this article:** Stanley TR, Clark RW, Fisher RN, et al. Changes in capture rates and body size among vertebrate species occupying an insular urban habitat reserve. *Conservation Science and Practice*. 2020;2:e245. https://doi.org/10.1111/csp2.245