

**POPULATION AND COMMUNITY DYNAMICS OF LIZARDS AND RODENTS IN
ORGAN PIPE CACTUS NATIONAL MONUMENT 1989-2006
AND IMPLICATIONS FOR FUTURE MONITORING**

Final Report

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On the cover: Side-blotched lizard (*Uta stansburiana*) by Thomas C. Brennan, Arizona State University (left) and desert pocket mouse (*Chaetodipus penicillatus*) by Cecil Schwalbe, U.S. Geological Survey, Sonoran Desert Research Station (right); all rights reserved.

EXECUTIVE SUMMARY

I evaluated long-term monitoring data for lizards and rodents in Organ Pipe Cactus National Monument (ORPI), described changes in the structure and composition of populations and communities over time, and provided recommendations for improving future efforts. For lizards, 4,743 surveys totaling 701 km of effort were completed over 18 years (1989-2006) and 14,217 individuals of 14 species were encountered. For rodents, 432 two-occasion trapping events were completed over 16 years (1991-2006) and 16,835 individuals of 10 species were captured. Abundance varied markedly across time for most species, yet I observed relatively few systematic temporal trends. Relative abundance of western whiptail (*Aspidoscelis tigris*) and zebra-tailed lizard (*Callisaurus draconoides*) declined by average rates of 0.9 to 1.5% per year (SE = 0.4-0.6%), whereas relative abundance of side-blotched lizard (*Uta stansburiana*) increased by an average of $2.5 \pm 0.5\%$ per year (\pm SE; $P \leq 0.035$) over 18 years. In contrast, no rodent species declined systematically across time and relative abundance (no./100 trap nights) of Bailey's pocket mouse (*Chaetodipus baileyi*), Arizona pocket mouse (*Perognathus amplus*), and rock pocket mouse (*Chaetodipus intermedius*) all increased by average rates of 4.1 to 9.7% per year (SE = 1.3-2.4; $P \leq 0.009$), with nearly identical trend estimates for abundance (no./grid). Temporal variation in abundance was especially prominent in populations of rodents and some trends were strongly curvilinear and characterized by high abundance during the early 1990s, late 1990s, and mid 2000s. Mass and age and reproductive structure varied across time in some populations and factors that drove these trends seemed complex. Species richness did not vary systematically during the study suggesting that all species present in ORPI when monitoring began were still present ≥ 16 years later. Composition of both rodent and lizard communities varied systematically across time and these changes were characterized by increasing abundance of side-blotched lizard and heteromyid rodents relative to other species.

I largely validated field methods used to sample lizard and rodent populations in ORPI and described procedures to improve sampling methods, process data, and complete trend and power analyses during future efforts. Time-of-day, season, and air temperature had important effects on encounter rates of lizards. Given these effects, maximum counts among repeated measurements across a range of variation in these factors seemed highly effective to index abundance. In comparison, estimates of density based on distance-sampling methods were much less efficient and yielded more variable estimates that had lower statistical power to detect trends likely because key assumptions of distance sampling were violated. For most species of rodents, temporal and lunar factors had little effect on capture rates such that modifying protocols or considering these factors as covariates during estimation may be unnecessary.

I found moderately-high levels of statistical power to detect small annual changes in abundance of common species of rodents and lizards during future years. Power was much higher for common species of lizards however, than for common species of rodents because inter-annual variation in abundance of rodents was typically much higher and because this variation did not contribute to trends. For rodents, power varied little when abundance (no./grid) was substituted for relative abundance (no./100 trap nights) as the parameter, yet for lizards, power was much higher for relative abundance (no./100 m) than for abundance (no./ha) due to higher levels of unexplained variation around trend estimates in abundance. Depending on resources available to complete monitoring during future years, reducing effort from surveys every year to every other year may be an efficient alternative for lizards but much less so for rodents because power was much lower overall. Abundance was too low to generate precise estimates of abundance for several species with use of the existing design. For these rarer species, additional sampling effort is required and sampling designs should be based on patterns of species occurrence. Because optimizing monitoring often depends on resources that are available for sampling, I outlined a range of potential options for future implementation.

Topography, soil texture, and hydrologic factors had significant effects on abundance of lizards and rodents, yet temporal trends in abundance often varied with these factors only for rodents. For six of eight species of rodents that I considered, either the magnitude or direction of temporal trends in abundance varied among topographic or hydrologic features ($P \leq 0.043$ for time \times factor interactions) in ways that suggested variation in habitat quality drove trends. Although information on the influence of disturbance on vertebrate populations is important for understanding and mitigating the effects of human migration and associated law-enforcement activities on biotic resources in OPRI, estimates of these effects based on the existing design were largely inconclusive. To assess the influence of disturbance on small vertebrates, new design elements should be incorporated.

Quantity of precipitation affected both lizard and rodent abundance in ways that suggested rainfall is an important driver of population and community dynamics in ORPI. Cool-season rainfall was typically more influential to populations of rodents and warm-season rainfall was typically more influential to populations of lizards, and as such, some species of rodents increased markedly after winters when the El Niño Southern Oscillation was active. Despite these trends, populations of rodents responded much more variably to precipitation than lizards both seasonally and with respect to time lags. Importantly, lizard species that were most affected by variation in precipitation tended to decline in abundance across time whereas rodent species that were least affected by precipitation tended to increase in abundance across this same time period. These trends combined with declines in cool-season and annual rainfall in ORPI during the study suggest communities of small vertebrates may already be responding to regional climate change. Models that estimate the long-term effects of precipitation on abundance of small vertebrates will be useful for predicting the influence of climate change across a range of potential climate scenarios, especially if precipitation continues to decline.

INTRODUCTION

Ecological monitoring is built on a foundation of repeatedly measuring resources over time so that the presence, magnitude, and direction of trends can be detected in sufficient time to make informed management decisions (Thompson et al. 1998, Yoccoz et al. 2001, Pollock et al 2002). For agencies charged with protecting and preserving natural resources, such as the National Park Service (NPS Organic Act, 1916), monitoring provides a critical feedback mechanism for guiding management and for assessing and revising the status of natural resources over time. At Organ Pipe Cactus National Monument (ORPI) in southern Arizona, an Ecological Monitoring Program (EMP) was initiated in the 1980s with help from the Cooperative Parks Studies Unit (CPSU) and associated scientists, including those at the University of Arizona. In the early and mid-1990s, several new sites and components were added to the EMP and oversight by an advisory committee of managers and researchers began. Throughout this process, researchers and ORPI staff were tasked with presenting relevant information from the program to both guide management and to allow periodic reviews and modifications. Most recently, these efforts culminated in a comprehensive report on trends in vegetation, wildlife, and other environmental factors (e.g., weather and air and water quality) (ORPI 2006). Currently, the EMP is in the midst of a thorough evaluation to determine its relevance, efficiency, and application to management. Because the EMP is likely the longest standing monitoring program in the National Park System (A. Hubbard, Sonoran Desert Network, pers. comm.) these data have great potential to guide management and monitoring. Furthermore, they may also provide valuable information to educators, policy makers, scientists, resource managers, and non-governmental organizations that are interested in the status and trends of natural resources on public lands.

Periodic reviews are an essential component of long-term monitoring efforts because they ensure that methods are appropriate and that objectives are both relevant and being efficiently met. Most importantly, frequent reviews of monitoring data enable managers to identify and respond to current threats so that appropriate actions can be taken. In recent years, ORPI and other areas along the international boundary have experienced marked increases in human migration and illegal smuggling from the south and in law enforcement associated with these activities. These activities have made monitoring more difficult and costly and pose potential threats to biological and cultural resources. Because the EMP was not explicitly designed to address these new and developing stressors, review and modification of the program are especially relevant.

Because appropriate management actions often depend on detecting changes in relevant resource conditions, ability to detect temporal trends in these conditions is often the most important attribute of a monitoring program. In ORPI, a partial internal review of the EMP was accomplished between 1997 and 1999 by Petterson (1998) who recommended modified protocols and objectives to better meet NPS standards. For example, it was recommended that lizard monitoring be designed to detect a 15% annual change in populations of western whiptail (*Aspidoscelis tigris*) over a ten-year period ($\alpha = 0.10$, $\beta = 0.20$). With support from the Desert Southwest Cooperative Ecosystem Studies Unit, a panel of scientists met to review the EMP in March 2006. Early in the workshop, it was evident that thorough analyses were needed before the program could be properly evaluated. During follow-up discussions, the panel determined that beginning the review process with the rodent and lizard data was most appropriate.

Data analyses require large investments of time and although recent efforts have been informative (ORPI 2006), information from the EMP has not been thoroughly analyzed since 1998 (Rosen 2000). Because existing data span a broad temporal frame during which weather conditions have varied markedly and sites vary widely across topographic, vegetation, and hydrologic gradients, they have great potential to elucidate population and community dynamics of vertebrates and the factors that drive these dynamics. Further, evaluation and enhancement of existing protocols have applications

throughout the park system, to inventory and monitoring networks throughout NPS, and to other entities that are now developing monitoring programs in the Sonoran Desert and elsewhere.

OBJECTIVES

The following general objectives were proposed:

1. Analyze lizard and rodent datasets and estimate trends in population and community parameters.
2. Use estimates of population trends and variance to assess power to detect trends in the future.
3. Recommend changes to protocols and sampling designs so that they more efficiently address relevant management and ecological issues.
4. Outline procedures for periodic data analyses.

METHODS

Site Selection—Study sites were selected non-randomly within ORPI. Seven sites were initially selected because they had rain monitoring dating back to the early 1960s (Alamo Canyon, Arch Canyon, Armenta Enclosure, Dos Lomitas, Dripping Springs, Growler Valley, and Quitobaquito). In successive years, additional sites that were representative of the park at large were added to capture variation in soil texture, rainfall, and vegetation; sites were not selected in proportion to the coverage of these features within the park. At each site, transects and grids were positioned so that they were representative of the surrounding landscape and typically included a mixture of vegetation, soil, and topographic features (P. Holm, ORPI, pers. comm.).

Diurnal Lizard Surveys—Detailed descriptions of field methods used to sample lizard populations in ORPI are available elsewhere (ORPI 1995, Rosen and Lowe 1996) and are summarized here to provide sufficient context to evaluate results and recommendations. Over 18 years (1989-2006), ORPI staff completed visual-encounter surveys (Crump and Scott 1994) at 7-19 sites per year and at 1-4 transects per site that ranged from 100-300 m in length and were surveyed during both spring and summer. Sampling times were selected to coincide with periods of peak activity of diurnal lizards both seasonally and with time-of-day. Surveys in spring were between 5 April and 23 June and surveys in summer were between 19 July and 7 October. Because lizard activity varies with temperature and temporal factors, ORPI staff surveyed each transect 3-8 times per day to ensure surveys coincided with periods of peak above-ground activity of each species. All surveys were completed during sunny and fairly calm days. A small subset of surveys were completed two times per season during some years ($n = 20$).

A single observer surveyed each transect by walking the center line and recording the time of each lizard detection, distance to the initial point of detection out to a maximum distance of 7.5 m, and the age class (adult, sub-adult, juvenile) when known. Animals were initially detected either aurally or visually and nearly all aural detections were confirmed visually. In most situations, observers began surveys by walking in a westward direction (away from the rising sun) to augment visual detection probabilities. A small number of highly experienced observers completed all surveys.

Nocturnal Rodent Trapping—Detailed descriptions of methods used to sample small mammal communities in ORPI are available elsewhere (Petryszyn 1995, ORPI 2006) and are summarized here to provide sufficient context to evaluate results and recommendations. Over 16 years (1991-2006), ORPI staff used mark-recapture techniques to sample nocturnal rodents at 7-18 sites per year and at 1-2 trap grids per site during late spring and summer. Sampling times were selected to

coincide with periods of peak rodent activity, which for most species in the Sonoran Desert is from June to September (Petryszyn 1982). Trapping began between 15 June and 10 July and ended between 15 July and 23 August across all years. Sherman traps were opened at dusk and animals were processed the following dawn while temperatures remained low. No trapping occurred during rain and surveys avoided trapping during full moon when possible (C. Conner, ORPI, pers. comm.).

Number of trapping occasions and the size and arrangement of trapping grids varied somewhat across time. The primary design included 7 x 7 trap grids with 15-m trap spacing across a 90 x 90 m area (8,100 m²). In 1998 and 1999, a total of 7 sampling events at 4 grids employed 12 x 12 trap grids with 10-m trap spacing across a 110 x 110 m area (12,100 m²). During each year, grids were sampled for two consecutive nights; from 1997 to 1999 some grids were trapped for four consecutive nights. In this report, I only considered the first two nights of trapping. To determine animals that had been re-captured on successive trapping occasions, ORPI staff used a felt-tipped ink pen to mark the ventral surface of each animal. Between 1991 and 1996, animals captured during the first occasion were given a generic temporary mark that allowed identification of recaptures but not specific individuals. From 1997 to 2006, new captures were given a unique temporary mark that allowed identification of individuals. For each animal captured, ORPI staff recorded age class (adult, sub-adult, juvenile) and sex, when known, the trap station, and estimated mass to the nearest tenth of a gram. Reproductive condition of each animal was recorded in four categories (non-reproductive, lactating, pregnant, testes descended) from 1997 onward.

Environmental, Spatial, Temporal, Weather, and Lunar Factors—I considered a range of factors when evaluating patterns of abundance across time and space, which provided information on 1) the influence of potential covariates on abundance, 2) need to adjust estimates for these effects, 3) needed modifications to survey protocols, and 4) the degree to which temporal trends varied with environmental factors. During lizard surveys, observers recorded the date, time, and air temperature at the start and end points of each survey. During mammal trapping, observers recorded the start and end date of each trapping event. Because moon phase can influence activity of nocturnal rodents (Price et al. 1984, Travers et al. 1988), I calculated the number of days between each sampling event and the new moon (range = 0-15). To describe environmental factors at each transect and trap grid, sites were classified according to topography (rocky slope, bajada, or valley floor), soil texture (fine or coarse), and hydrologic regime (upland or xeroriparian). Although vegetation associations were also classified, these data were not used because too many classifications were present for meaningful comparisons. As an index of disturbance by humans, six sites immediately along the international border (Quitobaquito, Aguajita, Vulture, Burn, Dos Lomitas, and Salsola) were differentiated from 13 interior sites that were presumably less disturbed (ORPI 2006). To describe annual and seasonal variation in rainfall, ORPI staff established 17 weather stations that were an average of 0.4 km (range 0.1 to 6.4) from sites where vertebrates were monitored and precipitation was tallied at each site during each month; data from the same weather stations were used to estimate rainfall at two sites because they were in close proximity (1.2-2.8 km) and had similar elevation and topography.

ANALYSES

Lizard Abundance—I used two methods to estimate abundance. For four relatively common species with an average of ≥50 encounters per year, I calculated detectability-corrected estimates of density (no./ha) for each species at each site during each year with use of distance-sampling methods (Buckland et al. 2001) and program DISTANCE (Thomas et al 2005). For six less common species with an average of 2-21 encounters per year, I calculated relative abundance by summing the maximum number of individuals that were detected during surveys among repeated measurements of each transect during each year and then divided by effort (m). I used the maximum number of detections because it indexed abundance at the time-of-day, season, and temperature when

detection probability was highest (Rosen 2000). I estimated abundance at each site so that I could adjust annual park-wide estimates for site effects, which was necessary because several new sites were added after early years of the study and not all sites were sampled during successive years. I did not estimate abundance of four remaining species because data were sparse. When estimating density and relative abundance, I assumed that the proportion of animals that were above ground and available for sampling did not vary systematically among years, an assumption that seemed reasonable based on results from mark-recapture sampling (P. Rosen, pers. comm.). Estimates of density are therefore not absolute because the actual proportion of animals that were available for sampling was unknown.

To estimate density, I partitioned data by year and stratified by site because program DISTANCE does not consider multiple strata. To assess the overall shape of detection functions, I pooled distance data for each species among years and considered models with half-normal and hazard rate key functions and cosine, polynomial, and Hermite expansion terms as candidates. For each species, I selected a single model from among these candidates with use of Information-Theoretic model selection procedures and Akaike's information criterion; I then used Goodness-of-fit tests to confirm model fit (Buckland et al. 2001). I assessed the influence of four potential covariates (air temperature, season, day-of-year, and observer) on the width of detection functions with use Multiple Covariates Distance Sampling procedures; because these analyses indicated an effect of both air temperature and season for each species, I included them as covariates in all analyses. When estimating density during each year, I used the same key function and expansion term for all models for a given species to reduce potential model bias. I considered a single detection function for each year because data were too sparse to model separate functions at each site. To smooth data, I binned distances by visually inspecting histograms of the percent observations in each 1-m interval, which remedied problems associated with evasive movements or limited observations in some intervals. Because animals were not recorded at distances >7.5 m, and therefore no outlying observations were present, I did not truncate data. I pooled data for each species across all repeated surveys on each transect because using data only from surveys during which maximum numbers were detected would have reduced sample sizes by as much as 80%. In using distance-sampling methods, I assumed that 1) all animals were detected with certainty on the transect line, 2) animals were detected in their initial locations, and 3) distances were measured accurately.

Rodent Abundance—I used two methods to estimate rodent abundance. To estimate relative abundance, I divided the total number of unique individuals captured on each grid during both trapping occasions by effort during each year. To calculate effort, I assigned sprung traps a value of 0.5 trap intervals (Nelson and Clark 1973), which is an effective means of adjusting effort for trap saturation and misfires (Beauvais and Buskirk 1999); 1,122 misfires occurred during the study. I calculated relative abundance with the following equation:

$$\hat{N}_r = \frac{\text{Total unique animals captured}}{(\text{traps} \times 2) - (\text{sprung traps} \times 0.5)} \times 100$$

Because estimates of relative abundance can be biased by variation in detectability (Lancia et al. 1994), I considered survey-specific values of temporal and lunar factors as covariates when they described variation ($P \leq 0.10$) in relative abundance. To describe moon phase, I divided the lunar cycle into three periods based on the quantity of moon light that was visible during trapping: dark (-3 to +5 days from new moon), bright (± 4 days from full moon), and intermediate (all other times). To describe temporal factors, I calculated Julian date of each trapping event. I also adjusted estimates of relative abundance during each year for site effects because the study design was unbalanced and several new sites were added or removed during later years.

To estimate abundance, I used the Lincoln-Petersen (L-P) estimator (Williams et al. 2002), which is appropriate when there are two sampling occasions and a relatively short time interval between occasions. In using the L-P estimator, I assumed 1) closure between trapping occasions (e.g., no immigration or emigration), 2) marks were neither lost nor overlooked, and 3) animals were equally likely to be captured during each trapping occasion. Because over the duration of the study trapping resulted in mortality of 30 animals during the first trap occasion, I removed these animals from analyses to satisfy the closure assumption and added these individuals to estimates of population size following calculations. To adjust estimates of population size on 12 x 12 trap grids so that they were comparable to estimates from 7 x 7 trap grids, I considered the ratio of grid area including a 15 m buffer which is equal to the average diameter of home ranges occupied by nocturnal rodents that occur in ORPI (Petryszyn 1995). This resulted in a correction factor (0.735) that adjusted estimates on larger grids. I did not adjust for differences in trap density between grid designs because too few 12 x 12 grids ($n = 4$) were available to estimate an effect. I calculated the L-P estimator for each species and species group (heteromyids and murids) on each grid during each sampling event and reported least square means that were adjusted for temporal, lunar, and site effects.

Rodent Mass, Reproductive Status, and Age Structure—To estimate mass of individuals of each species at each site during each year, I averaged measurements among individuals for which measurements were obtained. I did not consider biomass as a parameter (ORPI 2006) because it is a function of both abundance and mass, each of which I analyzed separately. To estimate the proportion of rodents that were reproductive at each site during each year, I differentiated non-reproductive individuals of each species from those that were lactating, pregnant, or had descended testes. To estimate the proportion of rodents and lizards that were adults, I differentiated them from sub-adults and juveniles.

Species Richness—To estimate species richness, I used three methods. I calculated observed species richness at each site and across all sites by summing the number of species that were observed during both seasons during each year. Because some species that are present during sampling are rarely detected, estimates of observed species richness are often negatively biased. Therefore, I used probabilistic methods to estimate species richness (\hat{R}) of lizards and rodents within ORPI during each year. Species' detection probabilities vary with both abundance and behavioral attributes, and as such, are heterogeneous among species. Therefore, I used model M_h from capture-recapture methods and the jackknife estimator (Burnham and Overton 1979), which allows detection probability to vary among species and is both robust and often the most appropriate model in these situations (Boulinier et al 1998). I estimated species richness with program SPECRICH2 (available at <http://www.mbr-pwrc.usgs.gov/software/>); inputs included the number of species detected at each site during each year and the frequencies or number of species detected at exactly 1, 2, 3... i sites each year.

Trend Estimation—To assess the presence, magnitude, and direction of trends in populations and communities over time, I used generalized linear mixed models to regress a range of different parameters against time (Proc GLIMMIX in SAS). Because sites were measured repeatedly across time, I considered site as the subject of these analyses and as a random effect and treated year as a fixed effect. To adjust for autocorrelation among sites sampled in successive years, I used a first-order autoregressive covariance structure. To assess nonlinear trends, I fit quadratic terms; I did not consider cubic terms because models with such terms generally failed to converge. I log transformed some response variables to improve normality. When assessing trends for rodents, I considered survey-specific values of temporal and lunar factors as covariates when they described variation ($P \leq 0.10$) in abundance. I did not consider covariates that influenced detections of lizards (e.g., air temperature, season, and time-of-day) because I used maximum counts among repeated measurements across a range of variation in these factors. I modeled population (abundance, relative abundance, rodent mass) and community (richness) parameters that had continuous

distributions with use of the identity (least squares) link function in SAS. When rodent mass was the response I also considered age as a covariate.

To assess temporal trends in age structure and reproductive status, I computed the proportion of observations that were adults, sub-adults, and juveniles and that were reproductive or non-reproductive for each species during each year. To estimate temporal trends in age structure and reproductive status, I used generalized linear mixed models to regress each parameter against year with use of the modeling procedure described above. Because these response variables were binomials (e.g., ratio of adults or reproductive individuals to all individuals observed), I used events/trials syntax in SAS and a logit link function, which is equivalent to logistic regression for binomial counts. For one species of lizard (western whiptail), I also calculated density of adults only and assessed trends across time using the modeling procedure described above; time constraints precluded estimating density and abundance by age class for other species.

To describe trends in the composition of lizards and rodent communities over time, I used principal components analysis to ordinate year- and site-specific estimates of 1) relative abundance (max no./100 m) of the nine most common species of lizards and 2) abundance (no./plot) of the eight most common species of rodents. To assess trends in community composition across time, I used generalized linear mixed models to regress principal components (PC) that had an eigenvalues >1.0 against year with use of the modeling procedure described above. To interpret these trends, I calculated Pearson correlation coefficients between values of each PC and estimates of abundance at each site and year for each species. For rodents, I also estimated trends in abundance of heteromyids and murids separately because these families have different natural histories (Hoffmeister 1986) and therefore high potential to exhibit varying trends across time.

Relationships between Population Parameters—To assess relationships between population parameters, I calculated Pearson correlation coefficients between annual estimates of relative abundance (max. no./100 m) and density (no./ha) for each species of lizard and between relative abundance (no./100 trap nights) and density (no./grid) of each species of rodent. To facilitate additional comparisons between density and relative abundance for less common species of lizards, I estimated density of three additional species that had low sample sizes (e.g., 8-21 encounters per year) with use of program DISTANCE by stratifying by year and fitting site as a random effect.

Power Analyses—To evaluate power to detect linear trends across time and provide guidance for future monitoring, I assessed relationships among power, sample effort, effect size, and the duration of sampling. To estimate variance, I calculated the root mean squared error (RMSE) around linear trends across time, which measures variation between the trend line and the sample data used to fit the line. I considered a time period of 25 years, sampling frequencies of every year, every other year, and every third year, a Type-I error rate (α) of 0.05, and a range of effect sizes varying from a 1 to 20% change per year depending on the levels of power that I estimated for each species. Because estimates of rodent abundance were not log transformed, I divided mean abundance (or relative abundance) of each species among years by RMSE to produce a quotient that was proportional to power, then estimated power for a representative group of species across the entire range of this quotient. To compute power, I used programming code for SAS that was developed by R. Steidl (Appendix A); power analyses did not consider precision of within-year estimates of parameters.

Survey Efficiency—Survey methods for lizards were based on assumptions that detection probability and encounter rates varied widely with both temporal (time-of-day, season) and weather-related factors (air temperature, wind speed, cloud cover) and declined to levels at distances >7.5 m to made recording these individuals inefficient. To ensure high detection probability and encounter rates, ORPI staff repeatedly surveyed transects across a range of times and during two seasons, procedures that greatly increased the time required for sampling. To provide justification and

appropriate modifications to survey methods, I assessed the influence of time-of-day and temperature on encounter rates (no./100 m) during each survey and the influence of season on relative abundance (max no./100 m) at each site among years. To assess seasonal effects, I compared relative abundance of each species between spring and summer with use of paired t -tests. To assess times of day when encounter rates were highest, I combined all detection data for each species and plotted the proportion of encounters within 30-minute periods across time and scaled encounters by quantity of effort within each period; to assess uncertainty of these estimates I calculated a binomial standard error for each proportion. Because variation in encounter rates across the day were likely a function of temperature, I plotted encounter rates versus temperature and used linear regression to describe these patterns. To assess if 7.5 m was an appropriate maximum detection distance, I used detection functions calculated by program DISTANCE to estimate detection probability at 7.5 m. For species for which there were too few encounters to model a detection function, I assessed the proportion of detections within each 1-m interval across all years.

Survey methods for rodents were based on the assumptions that capture rates and detection probability did not vary markedly with lunar and temporal (day-of-year) factors. To guide future efforts, I assessed the effects of moon phase and day-of-year on estimates of abundance. Because bias of abundance estimates based on the L-P estimator decreases as both initial marking effort and especially as the proportion of animals that are recaptured increase (Williams et al. 2002), knowledge of periods of peak activity can improve the precision of estimates.

Influence of Environmental Factors on Abundance and Trends in Abundance—To assess the influence of environmental factors on abundance, I compared estimates of relative abundance on each transect (for lizards) and grid (for rodents) for all years combined across topographic, soil, and hydrologic classifications. I did not consider site- and year-specific estimates of lizard density because 1) environmental factors often varied markedly among transects within a site, 2) sample sizes were rarely sufficient to calculate transect-specific estimates of density, and 3) relative abundance and density were highly correlated. I used generalized linear mixed models to compare abundance among environmental classifications and considered transects or grids as the subject of analyses; I fit year as a nominal factor (random effect) to adjust for differences among years. For rodents, I also considered survey-specific values of temporal and lunar factors as covariates when they described variation ($P \leq 0.10$) in relative abundance. I considered each environmental factor independently because soil texture and topography were correlated.

To assess if trends in abundance across time varied with topographic, soil, or hydrologic factors, I used the mixed modeling procedure described above and fit time \times factor interactions. Because for most species abundance typically oscillated over time rather than systematically increasing or decreasing, I fit year as a nominal covariate to adjust for differences among years and thereby isolate variation that could be attributable to environmental factors. For species that did exhibit systematic temporal trends, I fit year as a continuous factor. To illustrate significant time \times factor interactions ($P \leq 0.05$), I removed variation in relative abundance that was attributable to site, year, and covariates, and plotted residual abundance across time for each level of each factor.

To assess the effect of rainfall on abundance and species richness, I used generalized linear mixed models (Proc GLIMMIX in SAS) and quantified annual and seasonal rainfall in a variety of ways. As response variables for lizards, I used estimates of density and relative abundance at each site during each year and for rodents I used estimates of abundance and relative abundance at each grid during each year. To account for repeated measurements across time, I used the same modeling procedure as described above and considered site as the subject of analyses and as a random effect. To assess the strength of association between each parameter and rainfall, I considered rainfall during the warm season (April–September), cool season (October–March), and annually (sum of two periods) both with and without lag times of one and two years. Cool-season rainfall

without a lag time was that which fell immediately prior to the survey season, warm-season rainfall without a lag time fell during the survey period; and annual rainfall was the sum of these two periods.

Influence of Environmental Disturbance—I compared transect- and site-specific estimates of relative abundance for common species of lizards between disturbed (border) and undisturbed (interior) areas using the mixed modeling procedure described above (see *Trend Estimation* above). To adjust for other relevant factors, I first considered environmental covariates that explained variation ($P \leq 0.10$) in abundance then assessed whether any remaining variation was attributable to disturbance. To assess if trends in relative abundance across time depended on disturbance, I fit time \times disturbance interactions. Because difference in relative abundance between border and interior sites could be attributable to other factors, these comparisons seemed somewhat inconclusive, and were therefore not repeated for rodents.

RESULTS

Diurnal Lizards

Effort and Detections—ORPI staff completed 4,743 surveys totaling 701 km of effort over 18 years. (Table 1). During each year and season, total length of all transects combined averaged 3.8 ± 0.2 km (\pm SE) and ranged from 1.95-4.85 km overall. During each year, total survey effort averaged 20.1 ± 1.3 km in spring (1.28 ± 0.04 km per site) and 18.8 ± 1.2 km in summer (1.24 ± 0.04 km per site; Table 2) with an average of 26 ± 2 transects (min. = 14, max = 32) and 15.5 ± 1 sites (min. = 7, max. 19) surveyed each season. Number of transects, surveys, and total effort per site varied little between

Table 1: Annual and seasonal effort during surveys of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Totals include repeated measurements.

Year	Sites		Lines (no.)		Distance (km)		Surveys (no.)		Total effort (km)	
	Spring	Summer	Spring	Summer	Spring	Summer	Spring	Summer	Spring	Summer
1989	10	10	17	14	2.35	2.20	70	67	9.7	10.9
1990	9	10	14	17	2.10	2.35	73	90	10.9	12.3
1991	11	7	18	14	2.55	1.95	124	70	18.2	9.5
1992	7	7	14	14	1.95	1.95	75	85	10.3	12.9
1993	15	15	24	24	3.55	3.45	148	119	22.2	17.0
1994	15	16	23	26	3.35	3.65	114	140	16.6	20.9
1995	18	18	30	29	4.25	4.15	174	160	25.6	24.2
1996	18	18	30	30	4.25	4.25	207	183	29.5	26.4
1997	18	18	30	30	4.25	4.25	169	152	24.4	21.6
1998	18	18	30	30	4.25	4.25	154	143	22.0	20.5
1999	18	18	30	30	4.25	4.25	153	151	22.5	22.0
2000	18	18	30	30	4.25	4.25	148	147	21.0	20.9
2001	18	19	30	32	4.25	4.85	151	152	21.8	23.1
2002	19	19	32	32	4.85	4.85	146	144	22.7	21.6
2003	19	19	32	32	4.85	4.85	161	139	25.0	21.4
2004	19	19	32	32	4.85	4.85	138	140	21.1	21.4
2005	16	13	27	23	4.25	3.45	116	92	18.5	14.1
2006	19	15	32	26	4.85	4.15	138	110	21.1	17.9
Totals	285	277	475	465	69.25	67.95	2459	2284	362.7	338.1
Mean	16	15	26	26	3.85	3.78	137	127	20.1	18.8
SE	1	1	2	2	0.2	0.2	9	8	1.3	1.2

Table 2: Annual and seasonal effort on a per site basis during surveys of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006.

Year	Lines per site (no.)				Surveys per site (no.)				Total effort per site (km)			
	Spring		Summer		Spring		Summer		Spring		Summer	
	mean	SE	mean	SE	mean	SE	mean	SE	mean	SE	mean	SE
1989	1.70	0.30	1.40	0.16	7.00	1.21	6.70	0.84	0.97	0.11	1.09	0.19
1990	1.56	0.24	1.70	0.30	8.11	1.49	9.00	1.51	1.21	0.15	1.23	0.14
1991	1.64	0.28	2.00	0.38	11.27	2.23	10.00	1.85	1.65	0.32	1.36	0.12
1992	2.00	0.38	2.00	0.38	10.71	2.00	12.14	1.50	1.47	0.14	1.84	0.33
1993	1.60	0.21	1.60	0.21	9.87	1.14	7.93	0.89	1.48	0.12	1.13	0.06
1994	1.53	0.22	1.63	0.20	7.60	0.97	8.75	0.72	1.10	0.10	1.31	0.13
1995	1.67	0.18	1.61	0.18	9.67	0.76	8.89	0.73	1.42	0.11	1.34	0.14
1996	1.67	0.18	1.67	0.18	11.50	1.23	10.17	1.05	1.64	0.14	1.46	0.12
1997	1.67	0.18	1.67	0.18	9.39	0.84	8.44	0.75	1.36	0.08	1.20	0.05
1998	1.67	0.18	1.67	0.18	8.56	0.86	7.94	0.65	1.22	0.08	1.14	0.06
1999	1.67	0.18	1.67	0.18	8.50	0.76	8.39	0.74	1.25	0.11	1.22	0.10
2000	1.67	0.18	1.67	0.18	8.22	0.72	8.17	0.93	1.17	0.06	1.16	0.09
2001	1.67	0.18	1.68	0.17	8.39	0.74	8.00	0.74	1.21	0.07	1.21	0.11
2002	1.68	0.17	1.68	0.17	7.68	0.68	7.58	0.72	1.19	0.12	1.14	0.09
2003	1.68	0.17	1.68	0.17	8.47	0.64	7.32	0.60	1.31	0.12	1.12	0.10
2004	1.68	0.17	1.68	0.17	7.26	0.60	7.37	0.61	1.11	0.10	1.12	0.09
2005	1.69	0.20	1.77	0.23	7.25	0.66	7.08	0.71	1.16	0.11	1.08	0.06
2006	1.68	0.17	1.73	0.21	7.26	0.61	7.33	0.69	1.11	0.10	1.19	0.11
All years	1.67	0.02	1.69	0.03	8.71	0.33	8.40	0.31	1.28	0.04	1.24	0.04

seasons (Table 2). Mean survey timing during spring was 17 May \pm 0.6 days and during summer was 2 September \pm 0.9 days. Across all years combined, transects were surveyed an average of 4.9 ± 0.1 times per season between the hours of 6:30 AM and 12:44 PM with 96% of transects surveyed between 7:00 and 11:00 AM. Charles Conner completed 75.2% of surveys followed by Ami Pate (17.8%), and Phil Rosen (6.5%); 0.5% of surveys were completed by two other observers.

Effort varied with time-of-day and 96% of surveys were between 7:00 and 11:00 AM (Fig. 1A). Most effort was as times when temperatures were between 25 and 35° C and a higher proportion of effort during summer was during relatively warmer periods compared to that during spring (Fig 1B). Temperature during surveys increased linearly with increasing time-of-day and although temperatures at the same time were higher during summer, temperature increased with time at the same rate during both seasons (Fig 1C).

A total of 14,217 encounters of 14 species were obtained during the study. Total encounters for each species over all years ranged from 4 for desert horned lizard (*Phrynosoma platyrhinos*) to 5,980 for western whiptail. Mean, minimum, and maximum number of encounters per year varied markedly among species and only six species were detected during all years (Table 3). Nine species (64%) averaged ≤ 10.4 encounters per year versus only four species (29%) that averaged ≥ 80.5 . All species of diurnal species of lizard known to occur in ORPI were detected during the study except the chuckwalla (*Sauromalus ater*), which typically occurs in steep rocky uplands that were not represented along transects.

Throughout this report, I used common names for lizards because they are largely standardized and widely known and Latin names for rodents because they are more widely recognized and because common names often vary.

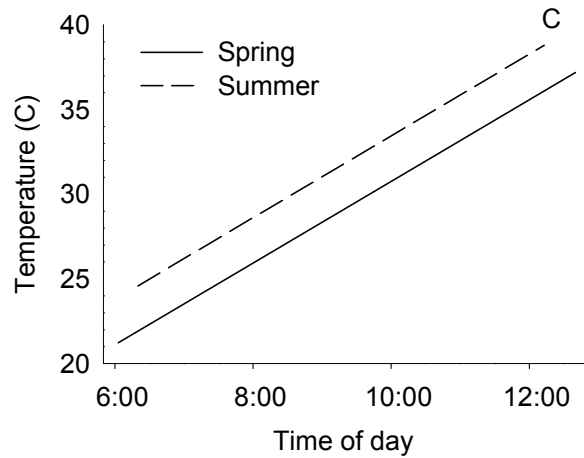
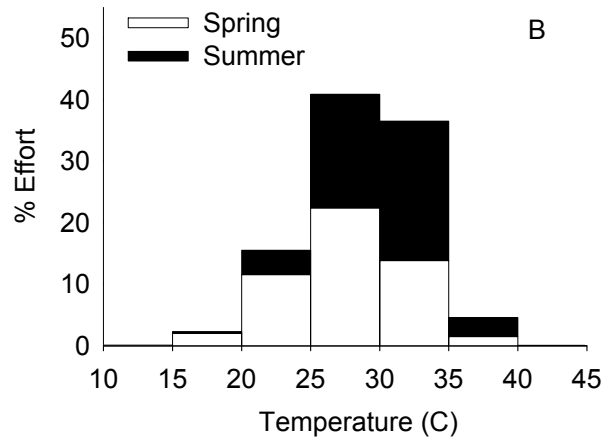
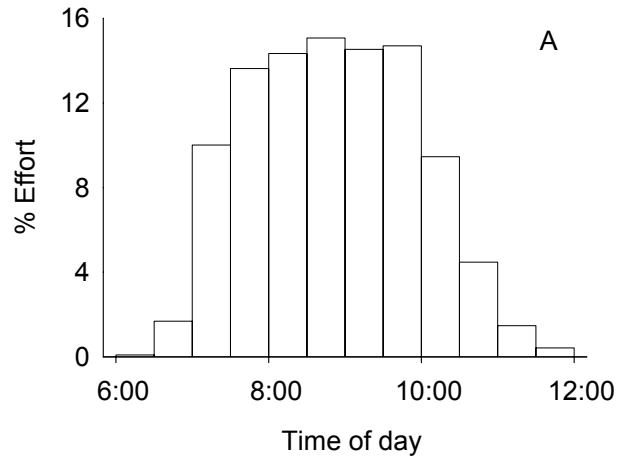


Figure 1: Effort, temperature, and time-of-day during surveys of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Effort is for all surveys including repeated measurements during the same day along transects. Time-of-day was not partitioned seasonally because effort varied little between seasons.

Table 3: Species and number of encounters of diurnal lizards detected during surveys in Organ Pipe Cactus National Monument 1989-2006. Common names of lizards were used in the text because they are largely standardized and known.

Family	Scientific name	Common Name	Encounters	Encounters per year			
				Min.	Max.	Mean	SE
Iguanidae	<i>Dipsosaurus dorsalis</i>	desert iguana	64	0	11	3.6	0.77
Crotaphytidae	<i>Crotaphytus nebrius</i>	Sonoran collared lizard	64	0	9	3.6	0.59
	<i>Gambelia wislizenii</i>	longnose leopard lizard	27	0	4	1.5	0.35
Phrynosomatidae	<i>Callisaurus draconoides</i>	zebratail lizard	1449	29	191	80.5	10.1
	<i>Sceloporus magister</i>	desert spiny lizard	370	8	48	20.6	2.6
	<i>Sceloporus clarkii</i>	Clark's spiny lizard	140	0	19	7.8	1.5
	<i>Uta stansburiana</i>	side-blotched lizard	3727	51	366	207.1	22.3
	<i>Urosaurus graciosus</i>	brush lizard	35	0	10	1.9	0.7
	<i>Urosaurus ornatus</i>	tree lizard	2149	36	229	119.4	14.7
	<i>Phrynosoma platyrhinos</i>	desert horned lizard	4	0	2	0.2	0.15
	<i>Phrynosoma solare</i>	regal horned lizard	12	0	5	0.7	0.29
	Teiidae	<i>Aspidoscelis burti</i>	red-backed whiptail	188	1	37	10.4
<i>Aspidoscelis tigris</i>		western whiptail	5980	163	583	332.2	28.5
Helodermatidae	<i>Heloderma suspectum</i>	Gila monster	8	0	2	0.4	0.15

Temporal Trends.—Abundance varied markedly across time for all species of lizards (Table 4; Figs. 2-4). Densities of zebra-tailed lizard (*Callisaurus draconoides*) and western whiptail peaked during middle years of the study and declined thereafter ($P \leq 0.0001$ for quadratic terms) yet there was no evidence of any systematic declines across all years (Table 4). For both species, density was highest from 1991 to 1993 and from 1997 to 2000 and increased somewhat in 2004 after reaching a decadal low in 2003. Trends for tree lizard (*Urosaurus ornatus*) were similar until 2004, after which density increased somewhat in later years resulting in little evidence of either linear or curvilinear trends in density (Table 4). Importantly, when relative abundance was considered the parameter (Fig. 3), populations of western whiptail and zebra-tail lizard declined systematically across the study by rates of 0.9-1.5% per year (Table 4), and there was stronger evidence that relative abundance was highest during middle years of the study and declined thereafter (for test of quadratic terms; Table 4). Both density and relative abundance of side-blotched lizard (*Uta stansburiana*) increased systematically; density increased by an average of 0.053 ± 0.010 log individuals per ha per year ($P < 0.0001$); there was also evidence that increases in abundance stabilized after 1997 ($P < 0.0001$ for quadratic term; Table 4, Figs. 2-3). When density of adult western whiptails was considered the parameter and other age classes were excluded (Fig. 2), there was less variation across time and less evidence of any linear or curvilinear temporal trends (Table 4).

I found no evidence of linear ($P \leq 0.21$) or curvilinear ($P \leq 0.21$) trends in relative abundance of desert spiny lizard (*Sceloporus magister*), Clark's spiny lizard (*Sceloporus clarkii*), red-backed whiptail (*Aspidoscelis burti*), brush lizard (*Urosaurus graciosus*), desert iguana (*Dipsosaurus dorsalis*), or Sonoran collared lizard (*Crotaphytus nebrius*) (Table 4, Fig. 4). However, patterns of relative abundance for many of these species, especially desert spiny lizard, were similar to that for zebra-tailed lizard, western whiptail, and tree lizard because abundance generally peaked during the same years (e.g., 1992 and 1998). Data were too sparse to estimate abundance of longnose leopard lizard (*Gambelia wislizenii*), desert horned lizard (*Phrynosoma platyrhinos*), regal horned lizard (*P. solare*), and Gila monster (*Heloderma suspectum*).

Table 4: Trend estimates for populations of diurnal lizards and for species richness in Organ Pipe Cactus National Monument 1989-2006. Results are based on generalized linear mixed models for density (log no./ha) and relative abundance (log max. no./100 m); site was considered the subject and as a random effect.

Response Species	Linear Trend					Quadratic Trend			
	Est.	SE	<i>t</i>	<i>P</i>	RMSE	Est.	SE	<i>t</i>	<i>P</i>
Density									
zebratail lizard	-0.010	0.011	0.97	0.33	0.671	-0.0074	0.0019	3.90	0.0001
side-blotched lizard	0.053	0.010	5.14	<0.0001	0.656	-0.010	0.0017	6.13	<0.0001
tree lizard	0.013	0.010	1.30	0.19	0.640	-0.0012	0.0020	0.58	0.56
western whiptail	-0.010	0.011	0.88	0.38	0.671	-0.0077	0.0019	4.01	<0.0001
western whiptail – adults only	0.0062	0.0091	0.68	0.50	0.612	-0.0027	0.0018	1.54	0.12
Relative Abundance									
zebratail lizard	-0.0093	0.0044	2.12	0.035	0.296	-0.0019	0.0009	2.18	0.030
side-blotched lizard	0.025	0.0049	5.06	<0.0001	0.331	-0.0038	0.0009	4.29	<0.0001
tree lizard	0.0017	0.0056	0.30	0.76	0.357	-0.0005	0.0011	0.44	0.66
western whiptail	-0.015	0.0058	2.58	0.010	0.352	-0.0038	0.0010	3.82	0.0002
desert iguana	-0.0026	0.0020	1.27	0.21	0.149	-0.0003	0.0004	0.76	0.45
Sonoran collared lizard	-0.0003	0.0011	0.31	0.76	0.073	-0.0001	0.0002	0.26	0.80
desert spiny lizard	-0.0004	0.0030	0.14	0.89	0.213	-0.0008	0.0006	1.26	0.21
Clark's spiny lizard	0.0004	0.041	0.28	0.78	0.081	0.0000	0.0003	0.02	0.98
brush lizard	-0.0008	0.0014	0.60	0.55	0.114	0.0000	0.0003	0.31	0.76
red-backed whiptail	-0.0013	0.0019	0.67	0.51	0.115	0.0001	0.0004	0.13	0.90
Species Richness - scale									
Observed - site	0.0093	0.011	0.85	0.40	0.855	-0.0065	0.0022	2.99	0.0031
Observed - study area	0.13	0.043	3.02	0.0082	0.956	-0.010	0.0093	1.09	0.29
Estimated - study area	0.020	0.11	0.18	0.86	2.34	-0.022	0.023	0.96	0.35

Age structure varied across time in populations of some species. Although abundance of tree lizards did not vary across time (Table 4), the proportion of individuals that were adults declined systematically ($t_{252} = 4.55$, $P < 0.0001$; Fig. 5). In contrast, for side-blotched lizard, a species that increased in abundance across time, the proportion of individuals that were adults also increased systematically ($t_{411} = 5.50$, $P < 0.0001$; Fig. 5) and there was a similar but much weaker trend for western whiptail ($t_{428} = 2.13$, $P = 0.034$). I observed no temporal trends in age structure in populations of zebra-tailed lizard ($t_{245} = 0.94$, $P = 0.35$).

Trends in species richness depended on scale and estimation methods. At a site-specific scale, trends in observed species richness (Fig. 6) were similar to those for abundance; richness averaged higher during middle years of the study ($P = 0.0031$ for quadratic term), peaked in 1991-1992, 1996-2000, and again in 2004, and richness did not decline systematically across time. Mean richness among sites during each year was highly correlated with relative abundance of all species combined ($r = 0.71$, $P = 0.001$). In contrast, although observed species richness measured at the scale of the study area increased by 0.13 ± 0.04 species per year, there was no temporal trend in estimated species richness (Table 4).

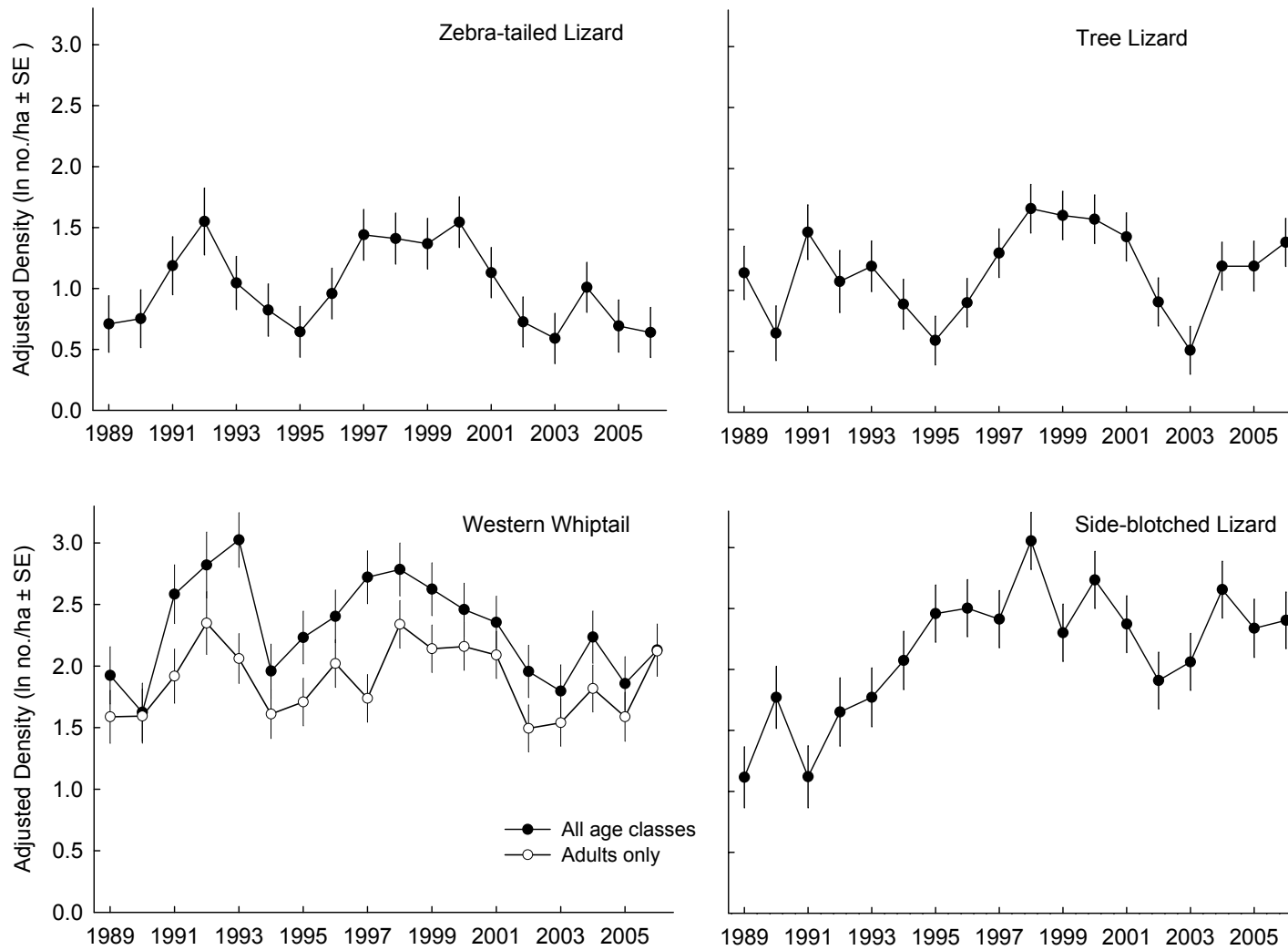


Figure 2: Trends in density (log no./ha) across time for the four most abundant species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Adjusted densities are least square means (± 1 SE) adjusted for site effects to account for the addition of several new sites in the mid 1990's. Density was calculated using distance-sampling methods with season and temperature considered as covariates. Density is plotted on the same scale for each species to facilitate comparisons among species. Density of adults was calculated for western whiptail only.

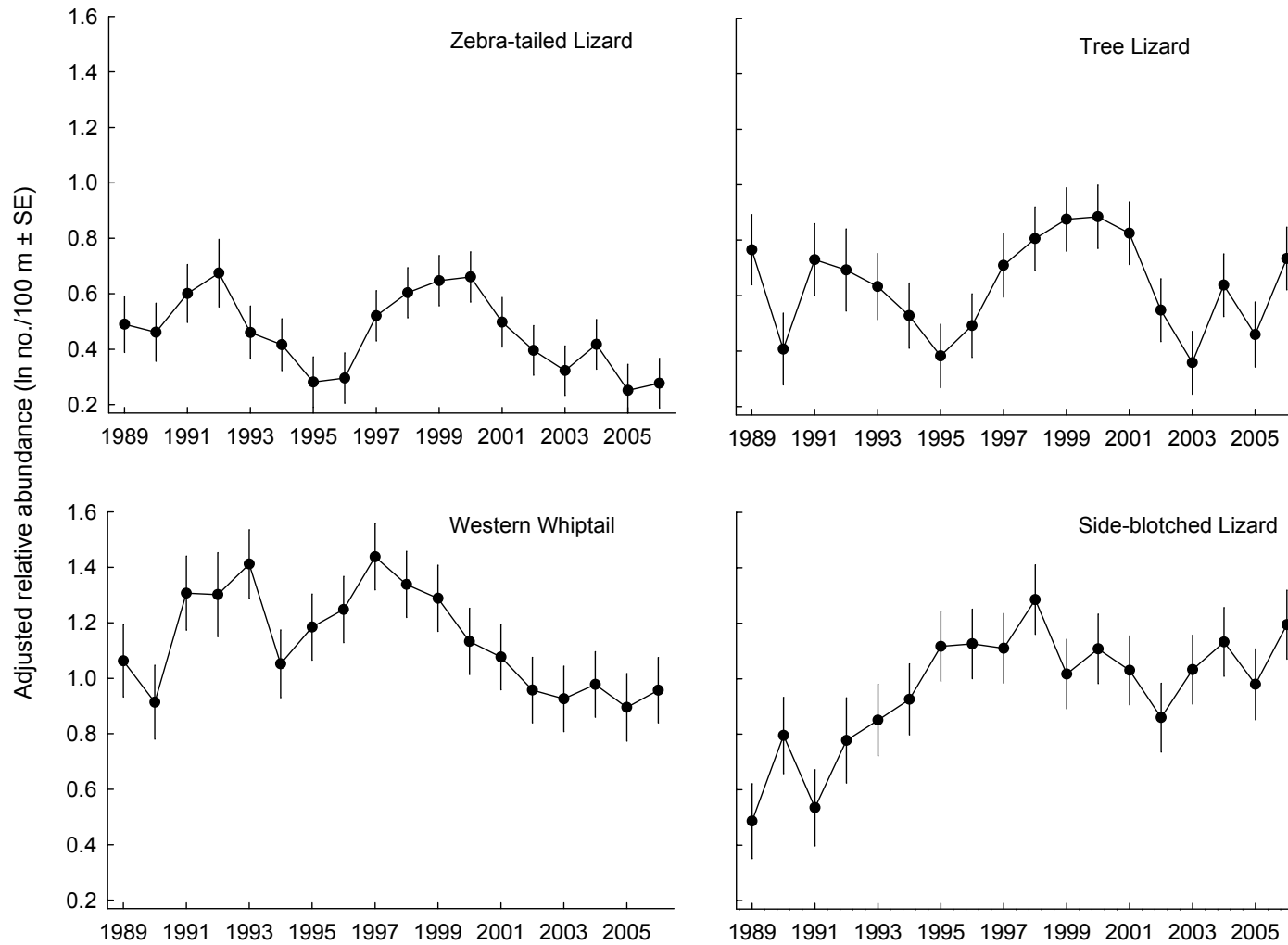


Figure 3: Trends in relative abundance (log max. no./100 m) across time for the four most abundant species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Adjusted relative abundances are least square means (± 1 SE) that were adjusted for site effects so as to account for the addition of several new sites in the mid 1990's. Relative abundance is plotted on the same scale for each species to facilitate comparisons among species.

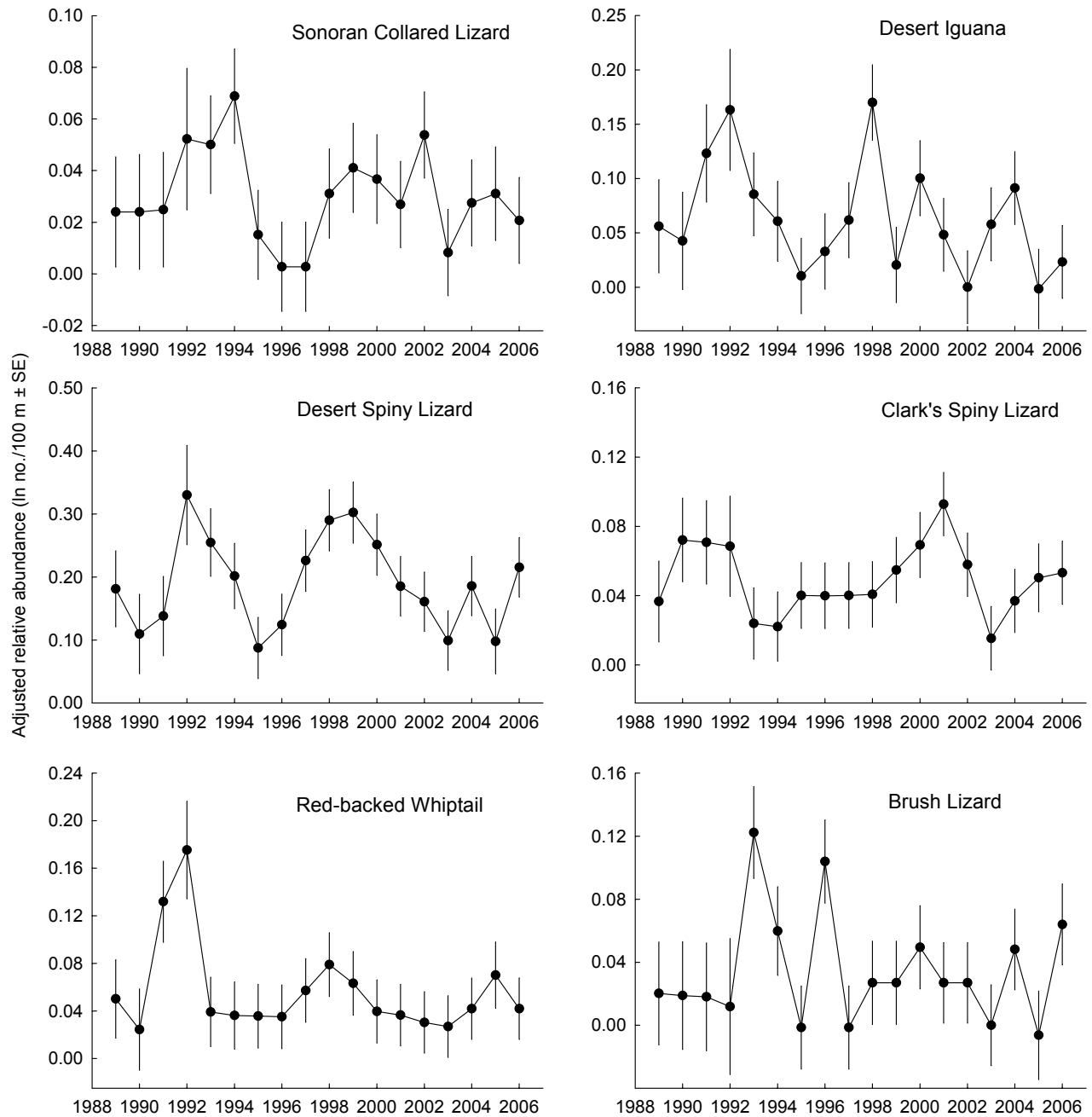


Figure 4: Trends in relative abundance (log max. no./100 m) across time for six uncommon species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Estimates are least square means (± 1 SE) that were adjusted for site effects to account for the addition of several new sites in the mid 1990's. Relative abundance is plotted on a different scale for each species because it varied widely among species.

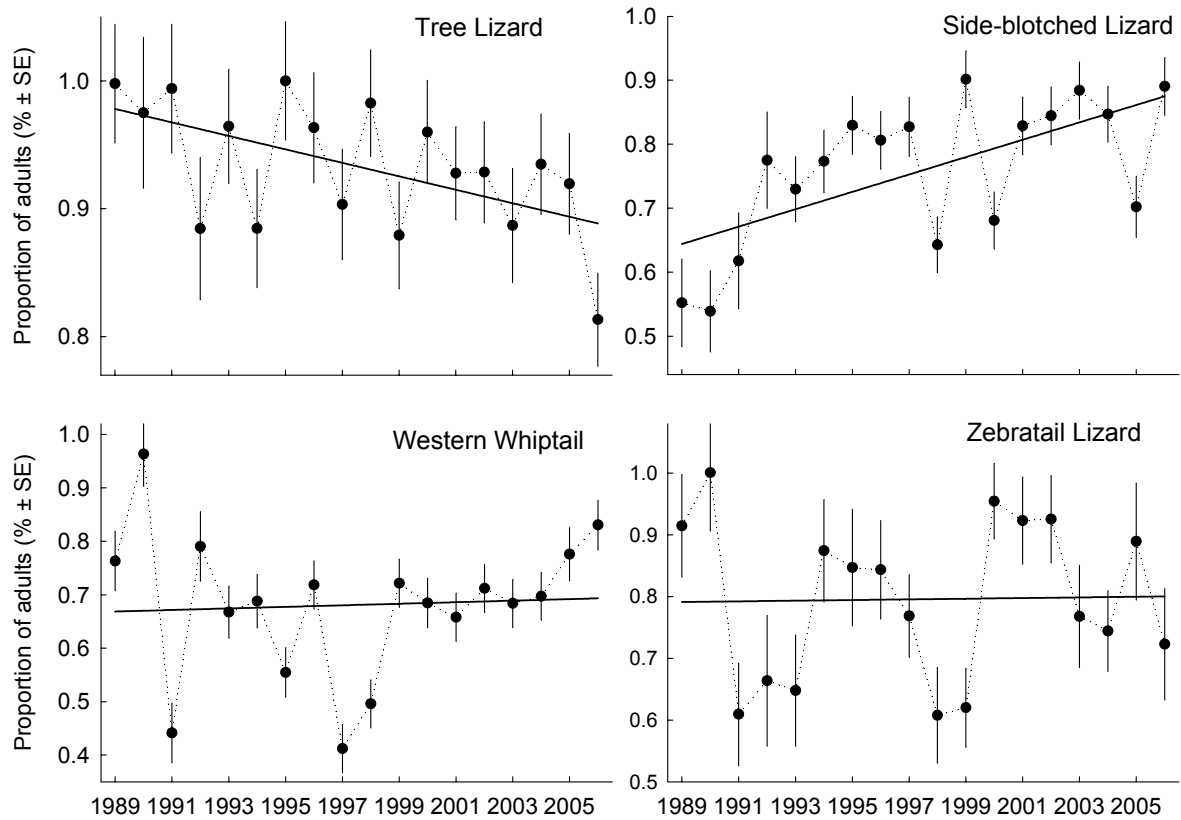


Figure 5: Trends in age structure of the four most abundant species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Estimates are mean proportions of individuals that were adults among sites during each year (± 1 SE) and are adjusted for site effects.

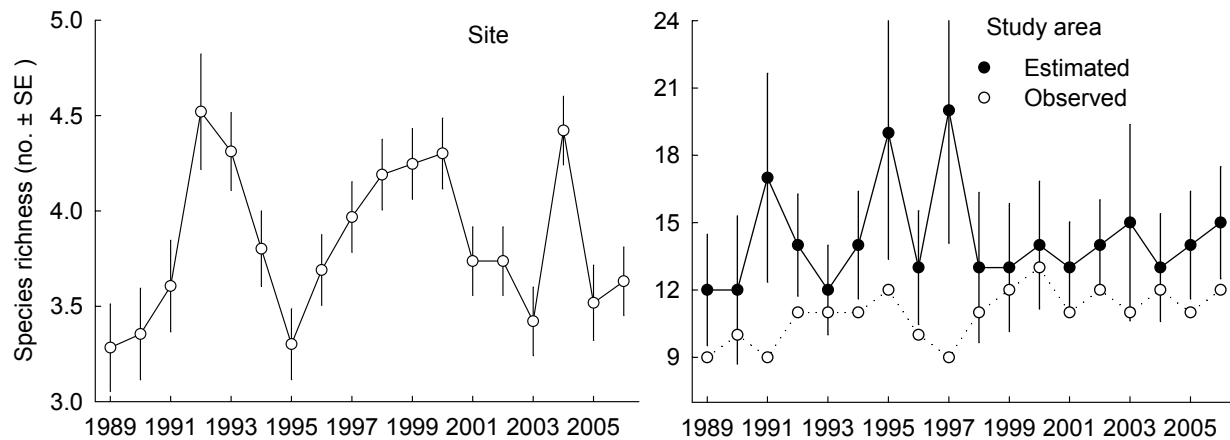


Figure 6: Temporal trends in species richness of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Observed richness at the scale of sites is equal to the average (± 1 SE) number of species detected during each year among sites. Estimated richness was calculated for the entire study area using model M_h and the jackknife estimator. Observed species richness at the scale of the study area was equal to the raw number of species detected each year.

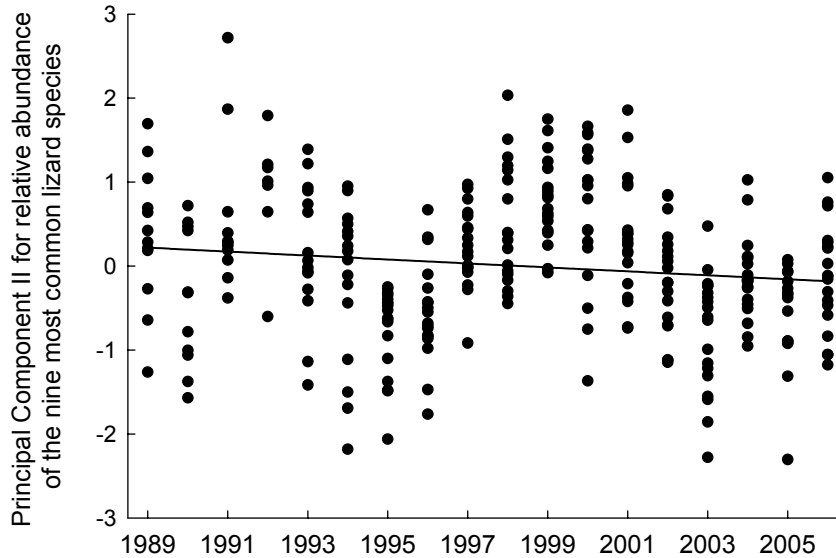


Figure 7: Temporal trends in community composition of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Values are based on the second principal component, which explained 18% of total variation in estimates of relative abundance of the nine most common species of lizards at each site and each year. Loadings were positive for all species except side-blotched lizard and desert iguana. Trend line ($P = 0.019$) is based on a generalized linear mixed model adjusted for site effects.

Composition of the lizard community varied across time. The first principal component (PC1) explained 33% of variation in site- and year-specific estimates of relative abundance of the nine most common species of lizards and PC2 explained 18%. Correlations between PC2 and relative abundance indicated that PC2 was negatively correlated with relative abundance of side-blotched lizard ($r = -0.30$, $P < 0.0001$) but not with desert iguana ($r = -0.03$, $P = 0.69$) and was positively correlated with relative abundance of all remaining species (mean $r = 0.40$, $P \leq 0.0074$) except Clark's spiny lizard (0.08 , $P = 0.15$); a tenth species (brush lizard) was not correlated with PC2 ($r = 0.00$). PC2 declined systematically across time ($t_{271} = 2.37$, $P = 0.019$; Fig. 7) suggesting that side-blotched lizard comprised an increasing proportion of the community relative to other species over time. Correlations for PC1 were ambiguous with respect to community composition and are not reported.

Relationships between Population Parameters—Species-specific estimates of density during each year were highly correlated with estimates of relative abundance during the same years ($P \leq 0.0002$; Table 5). Correlation coefficients ranged from as low as 0.76 for Clark's spiny lizard to as high as 0.97 for side-blotched lizard. Magnitude of correlation averaged $13 \pm 4\%$ higher ($t_6 = 3.39$, $P = 0.019$) for the four species for which sample sizes were sufficient to calculate site- and year-specific estimates of density compared to three species for which density was calculated across all sites during each year (Table 5).

Correlations in annual estimates of relative abundance varied widely among species (Table 6). Side-blotched lizard and Clark's spiny lizard were the only species for which estimates were not correlated with those of any other species at the $P \leq 0.10$ level. These exceptions aside, relative abundance of desert spiny lizard was highly correlated with that of all other species ($n = 6$), and levels of association were similar but somewhat lower for desert iguana (five other species), and

Table 5: Pearson correlation coefficients between annual estimates of density (log no./ha) and relative abundance (log max. no./100 m) for seven species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Strata and covariates are those used while estimating density with use of distance-sampling methods.

Species	Density Estimation		<i>r</i>	<i>P</i>
	Strata	Covariates		
zebratail lizard	Site and year	Season, temperature	0.86	<0.0001
side-blotched lizard	Site and year	Season, temperature	0.97	<0.0001
tree lizard	Site and year	Season, temperature	0.92	<0.0001
western whiptail	Site and year	Season, temperature	0.92	<0.0001
desert spiny lizard	Year	Site, season, temperature	0.85	<0.0001
Clark's spiny lizard	Year	Site, season, temperature	0.76	0.0002
red-backed whiptail	Year	Site, season, temperature	0.79	0.0001

red-backed whiptail (four other species; $P \leq 0.10$ for all comparisons). Relative abundance of tree lizard, zebra-tail lizard, and western whiptail were all highly correlated (Table 6) as suggested by high levels of correspondence among trends in relative abundance across time (Fig. 3)

Annual estimates of density were correlated with annual estimates of age structure, yet patterns varied widely among species. As density of western whiptails declined (e.g., 1998-2003), the proportion of the population comprised of adults increased relative to sub-adults and juveniles ($r = -0.53$, $P = 0.022$; Fig. 8). Correspondingly, during periods when density increased (e.g., 1994-1998) the opposite pattern occurred. Although this pattern was somewhat apparent for zebra-tailed lizards between 1991 and 1999 ($r = -0.27$, $P = 0.28$) it was not apparent for tree lizards ($r = -0.16$, $P = 0.53$), likely because only very small proportions of these populations were comprised of younger individuals (Fig. 9). In contrast to associations for western whiptail, as density of side-blotched lizards increased from 1989 to 1997, the proportion of the population comprised of adults increased relative to sub-adults and juveniles ($r = 0.57$, $P = 0.013$; Fig. 8).

Table 6: Pearson correlation coefficients for annual estimates of relative abundance (max. no./100 m) for all possible pairs of nine species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006.

	Sonoran collared lizard	desert spiny lizard	desert iguana	Clark's spiny lizard	red-backed whiptail	tree lizard	side-blotched lizard	western whiptail	zebra-tailed lizard
Sonoran collared lizard		0.47 ^b	0.16	-0.02	0.20	0.18	-0.25	-0.01	0.32
desert spiny lizard	0.47 ^b		0.54 ^b	0.00	0.46 ^c	0.75 ^a	0.10	0.57 ^b	0.75 ^a
desert iguana	0.16	0.54 ^b		-0.01	0.60 ^a	0.42 ^c	-0.12	0.50 ^b	0.69 ^a
Clark's spiny lizard	-0.02	0.00	-0.01		0.31	0.35	-0.19	-0.14	0.31
red-backed whiptail	0.20	0.46 ^c	0.60 ^a	0.31		0.32	-0.34	0.41 ^c	0.57 ^b
tree lizard	0.18	0.75 ^a	0.42 ^c	0.35	0.32		0.01	0.43 ^c	0.73 ^a
side-blotched lizard	-0.25	0.10	-0.12	-0.19	-0.34	0.01		0.02	-0.25
western whiptail	-0.01	0.57 ^b	0.50 ^b	-0.14	0.41 ^c	0.43 ^c	0.02		0.55 ^b
zebra-tailed lizard	0.32	0.75 ^a	0.69 ^a	0.31	0.57 ^b	0.73 ^a	-0.25	0.55 ^b	

^a $P \leq 0.01$

^b $P \leq 0.05$

^c $P \leq 0.10$

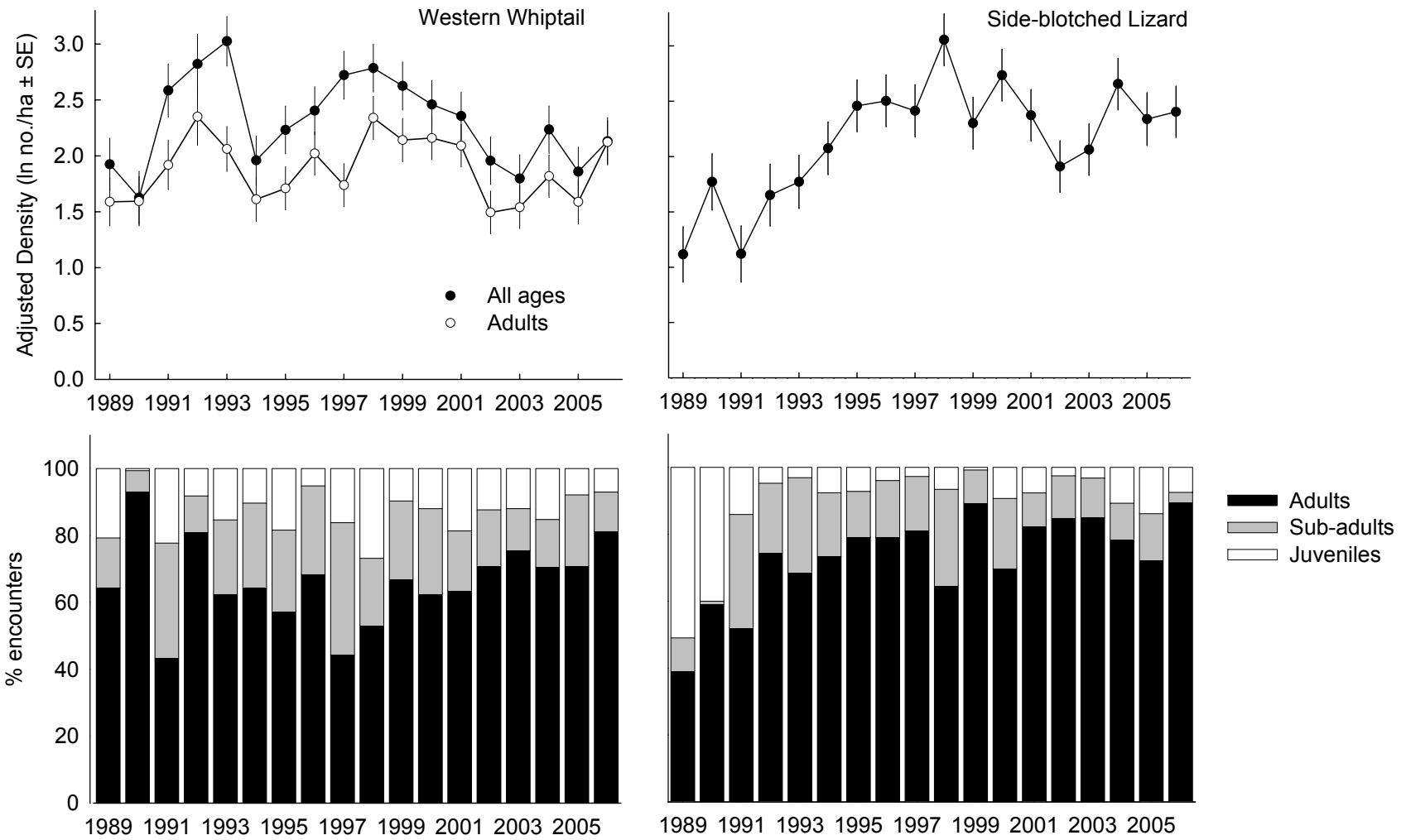


Figure 8: Annual estimates of age structure across time for the two most abundant species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Estimates of density (log no./ha) from Fig. 2 are also provided for comparison.

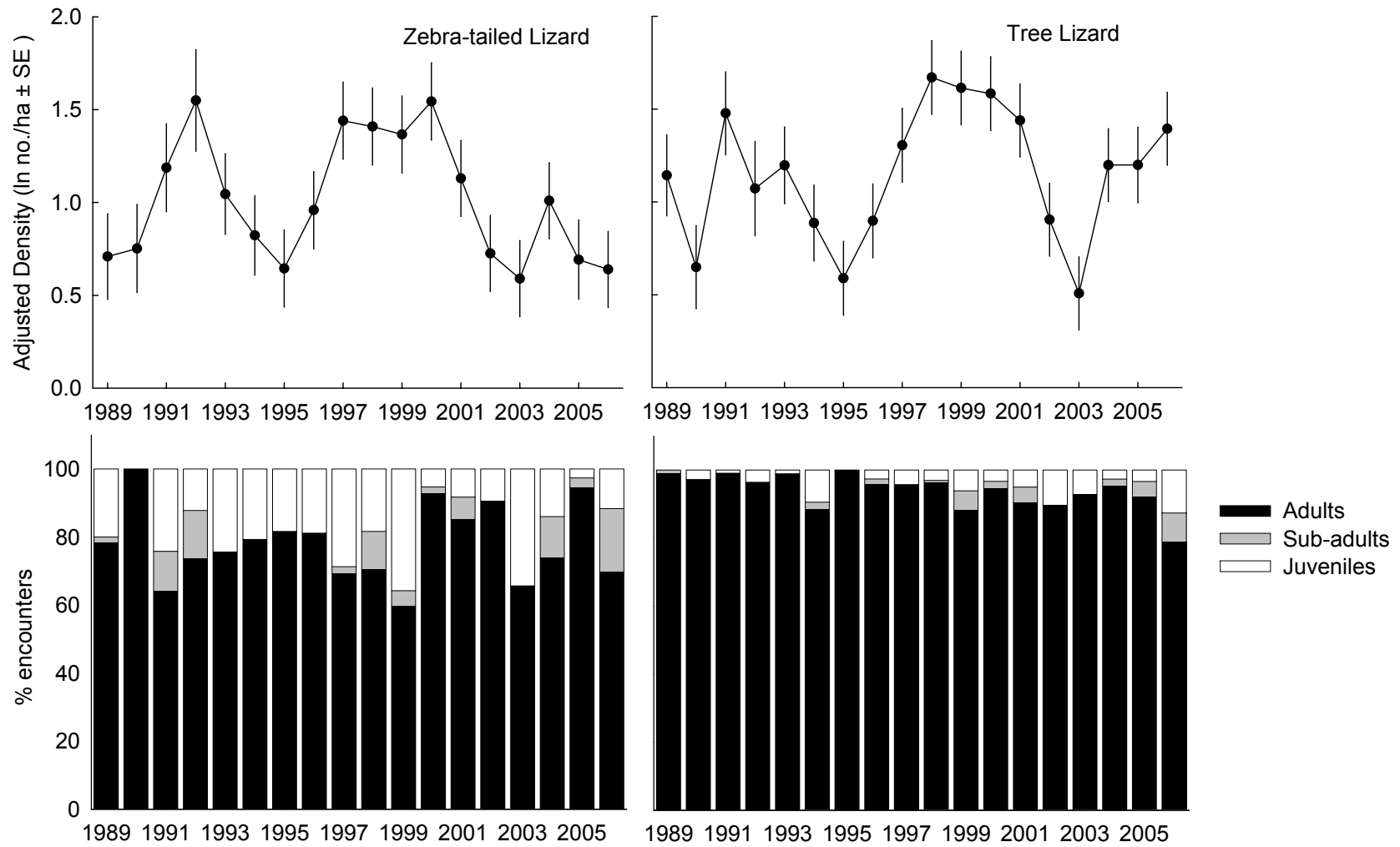


Figure 9: Annual estimates of age structure across time for the third and fourth most abundant species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Estimates of density (log no./ha) from Fig. 2 are also provided for comparison.

Power Analyses—Power to detect trends in relative abundance were much higher than those for density because residual error around trend lines in density (RMSE) averaged approximately three times greater than that for relative abundance (Table 4). This pattern was consistent regardless of species due to fairly large degrees of separation between species-specific values of RMSE around trends in density (range = 0.640-0.671) versus those for relative abundance (range = 0.073-0.357) (Table 4). Power to detect small annual changes (1-2%) in relative abundance of western whiptails (RMSE = 0.352) and three other relatively abundant species that had similar values of RMSE around trends (0.296-0.357), were low even after sampling every year for 25 years (Fig. 10). For western whiptail however, sampling each year for another 25 years would enable detection of a 3% annual change in relative abundance with power $\geq 80\%$ ($\alpha = 0.05$); importantly, a $\geq 4\%$ annual change in relative abundance with power $\geq 80\%$ could be detected after reducing survey effort to once every other year (Fig. 10). In comparison, power to detect trends in relative abundance for six less common species (RMSE = 0.073-0.213) was high (see Fig. 11 based on RMSE of 0.213). As a general rule, power to detect trends across time increased as the degree of inter-annual variation in the parameter of interest decreased relative to the overall magnitude of the trend being estimated.

Despite limited power to detect trends in density, power to detect trends for adults alone was higher than that for trends for all age classes combined (compare Fig. 12 with 13 for western whiptail). Regardless, power to detect trends in density of adult western whiptails did not reach 80% until effect sizes were $\geq 5\%$, even after sampling every year for the next 25 years. Power to detect trends in estimated species richness was high even with relatively low levels of sample effort (Fig. 14). For example, after reducing effort to surveys every other year, power to detect small annual changes in estimated species richness (e.g., 2%) exceeded 80% after sampling for approximately 24 years; a 2% annual change in estimated species richness is equaled to a change of only 0.3 species per year.

Survey Efficiency—Relative abundance of five of the 11 most frequently detected species were greater in summer than in spring ($P \leq 0.061$) whereas only one species, the tree lizard, had higher relative abundance in spring (Table 7); relative abundance did not vary between seasons for five remaining species. Differences between seasons for desert iguana, desert spiny lizard, and brush

Table 7: Relative abundance (max. no./100 m) of 11 species of diurnal lizards during both spring and summer in Organ Pipe Cactus National Monument 1989-2006. Test statistics and *P*-values were calculated with paired *t*-tests comparing 274 year- and site-specific estimates.

Species	Spring		Summer		Difference			
	mean	SE	mean	SE	% ^a	SE	<i>t</i>	<i>P</i>
desert iguana	0.030	0.007	0.062	0.011	107	41	2.42	0.016
Sonoran collared lizard	0.023	0.006	0.018	0.005	28	160	0.68	0.50
longnose leopard lizard	0.014	0.005	0.027	0.007	93	69	1.53	0.13
zebratail lizard	0.56	0.047	0.54	0.042	3	376	0.26	0.80
desert spiny lizard	0.15	0.017	0.207	0.021	40	46	2.22	0.027
Clark's spiny lizard	0.039	0.009	0.032	0.009	22	171	0.62	0.54
side-blotched lizard	1.02	0.069	2.26	0.11	122	11	9.12	<0.0001
brush lizard	0.010	0.005	0.026	0.007	160	51	1.88	0.061
tree lizard	1.04	0.069	0.78	0.058	33	32	3.14	0.0019
red-backed whiptail	0.052	0.016	0.049	0.017	6	800	0.13	0.90
western whiptail	1.74	0.079	2.45	0.11	41	18	5.43	<0.0001
observed species richness	3.3	0.06	3.6	0.07	9	28	3.60	0.0004

^a Indicates percent increase during season where encounter rate was greatest; SE are % units.

Ln(Relative Abundance) Western Whiptail

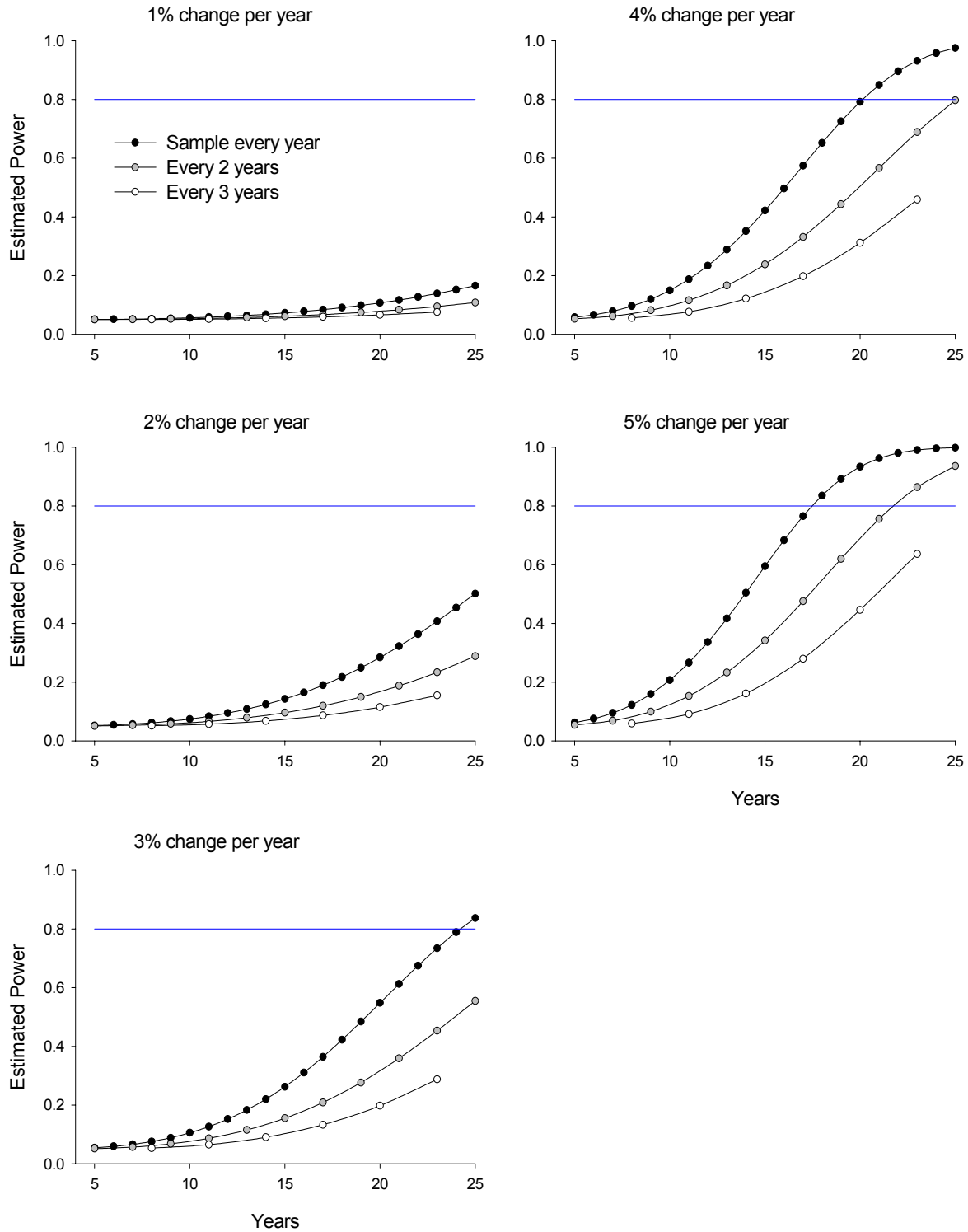


Figure 10: Power to detect a linear change of 1 to 5% per year in relative abundance of all age classes of western whiptails combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Ln(Relative abundance) Desert Spiny Lizard - all age classes

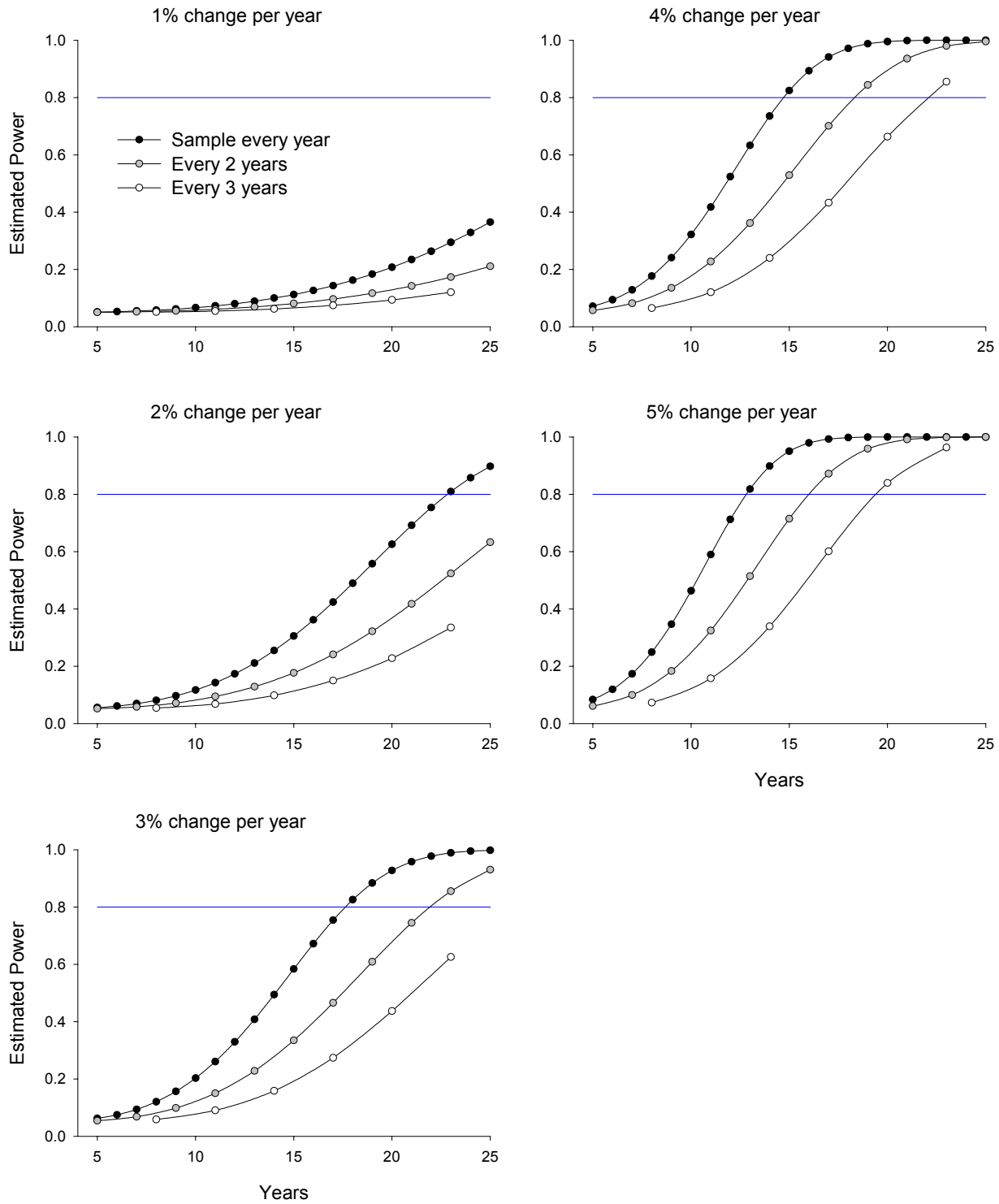


Figure 11: Power to detect a linear change of 1 to 5% per year in relative abundance of all age classes of desert spiny lizards combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Ln(Density) Western Whiptail - adults only

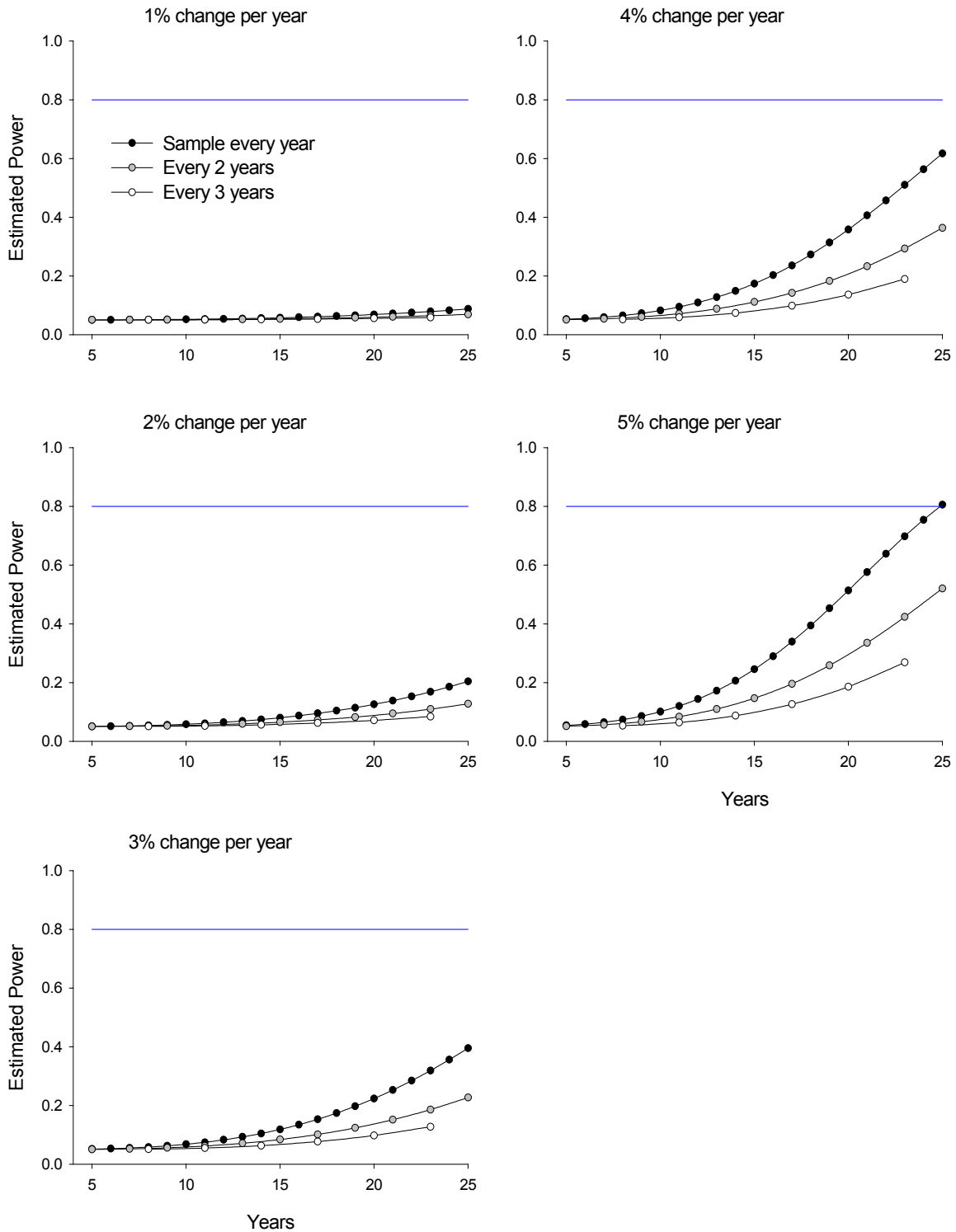


Figure 12: Power to detect a linear change of 1 to 5% per year in density of adult western whiptails based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Ln(Density) Western Whiptail - all age classes

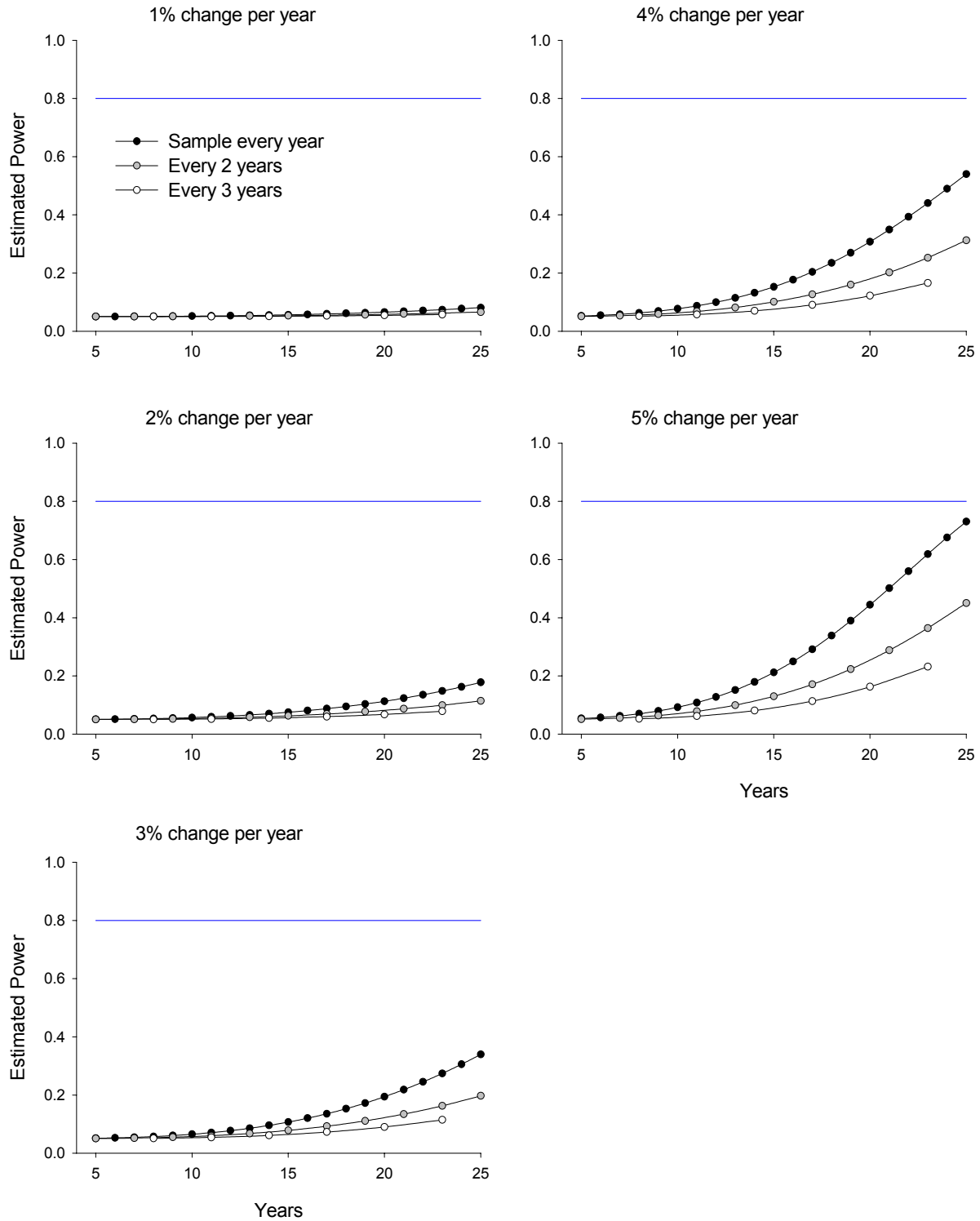


Figure 13: Power to detect a linear change of 1 to 5% per year in density of all age classes of western whiptails combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Estimated Species Richness

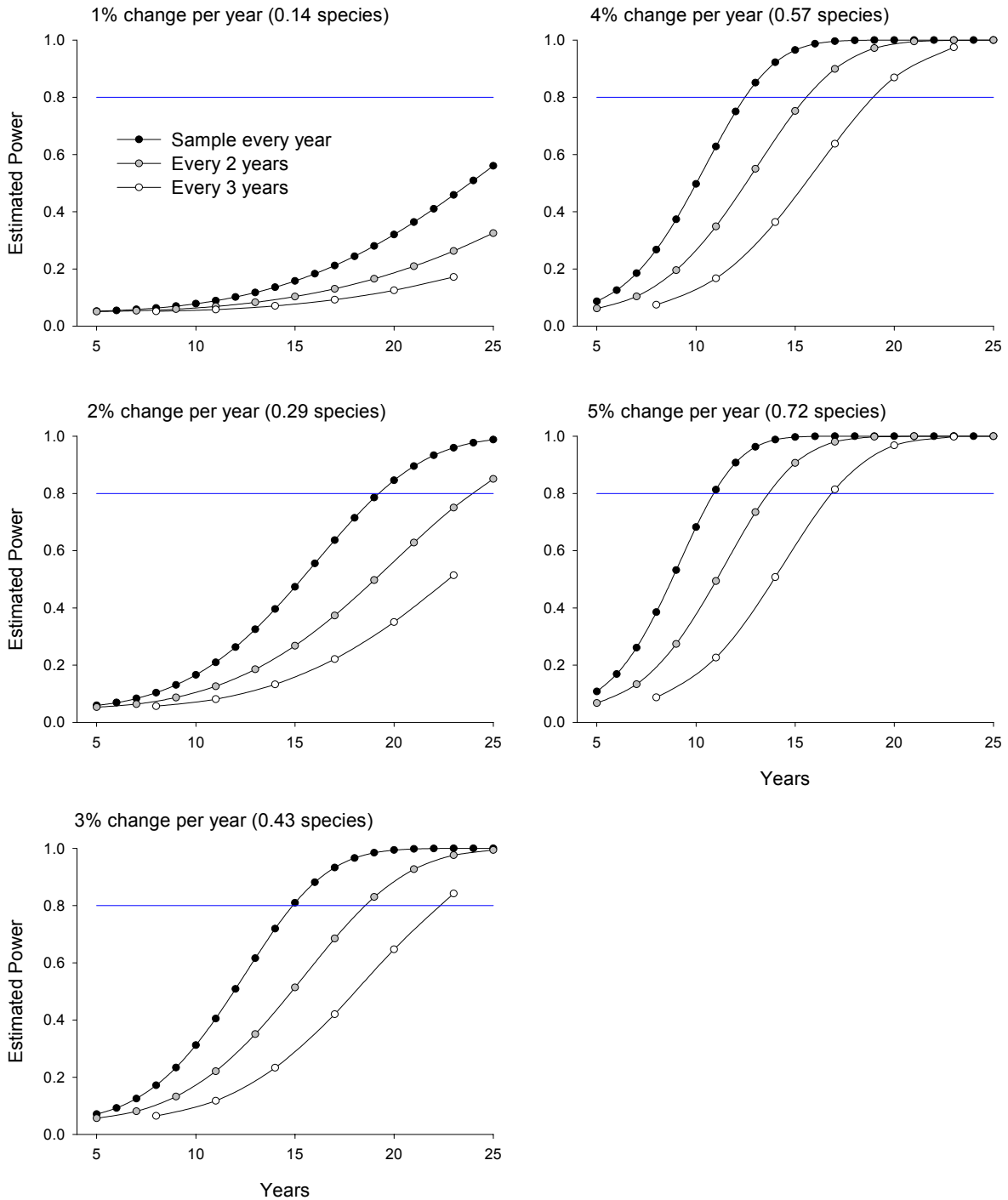


Figure 14: Power to detect a linear change of 1 to 5% per year in estimated species richness based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

lizard were attributable to much larger proportions of juveniles and sub-adults detected during summer whereas the proportion of adults detected in each season were similar (Table 8). In contrast seasonal differences in relative abundance of side-blotched lizard and tree lizard were clearly attributable to differences in the number of adults (Table 8).

Encounter rates of each species varied with time-of-day in three general ways. For side-blotched lizard, temporal variation in encounter rates suggested that surveys earlier in the day were most efficient (Fig. 15). In contrast, for tree lizard, desert spiny lizard, and western whiptail, mid-morning surveys were most efficient. Despite these general trends, encounter rates of western whiptail increased rapidly between 6:00 and 8:30 AM whereas for tree lizard encounter rates varied less during earlier hours of morning. For desert iguana, Clark’s spiny lizard, red-backed whiptail, and zebra-tailed lizard, encounter rates increased with time-of-day such that survey efficiency may be enhanced by surveying later in the day, especially for desert iguana.

Relationships between encounter rates and temperature varied among species (Fig. 16). Because temperature increased with time-of-day (Fig. 1), time-of-day and temperature had similar effects on encounter rates. Encounter rates of western whiptail and zebra-tailed lizard peaked near 30°C (Fig. 16) and declined thereafter ($t_{4623} = 7.14$, $P < 0.0001$; for quadratic term). In contrast, for zebra-tailed lizard, there was more evidence that encounter rates increased across the entire range of temperature ($t_{4624} = 5.67$, $P < 0.0001$; for linear trend) rather than stabilizing or declining beyond an upper threshold ($t_{4623} = 2.78$, $P = 0.0055$; for quadratic term). Encounter rates of tree lizard were highest at low to moderately-low temperatures and decreased rapidly thereafter (Fig. 16). In contrast, encounter rates for desert spiny lizard were highest at moderately-low temperatures and decreased slowly thereafter. Encounter rates of side-blotched lizard were greatest near 25°C, yet because prevalence of these relatively low temperatures in summer were much less common (Fig. 1), encounter rates declined by 61% more with each degree of temperature during summer than during spring ($t_{4621} = 7.45$, $P < 0.0001$, for temperature \times season interaction).

Table 8: Proportion of encounters that were adults and non-adults (sub-adults and juveniles) during spring and summer for 11 species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006.

Species	Total encounters	Spring		Summer	
		Adults (%)	Non-adults (%)	Adults (%)	Non-adults (%)
desert iguana	64	32.8	4.7	26.6	35.9
Sonoran collared lizard	64	48.4	7.8	42.2	1.6
longnose leopard lizard	27	33.3	0.0	40.7	25.9
zebratail lizard	1449	40.5	15.2	34.9	8.6
desert spiny lizard	370	31.4	8.9	30.0	28.4
Clark's spiny lizard	140	36.4	12.9	42.9	6.4
side-blotched lizard	3727	25.0	5.2	51.5	17.8
brush lizard	35	42.9	0.0	45.7	11.4
tree lizard	2149	55.4	2.7	37.6	4.2
red-backed whiptail	188	44.7	10.1	37.8	7.4
western whiptail	5980	28.0	13.2	35.4	23.0

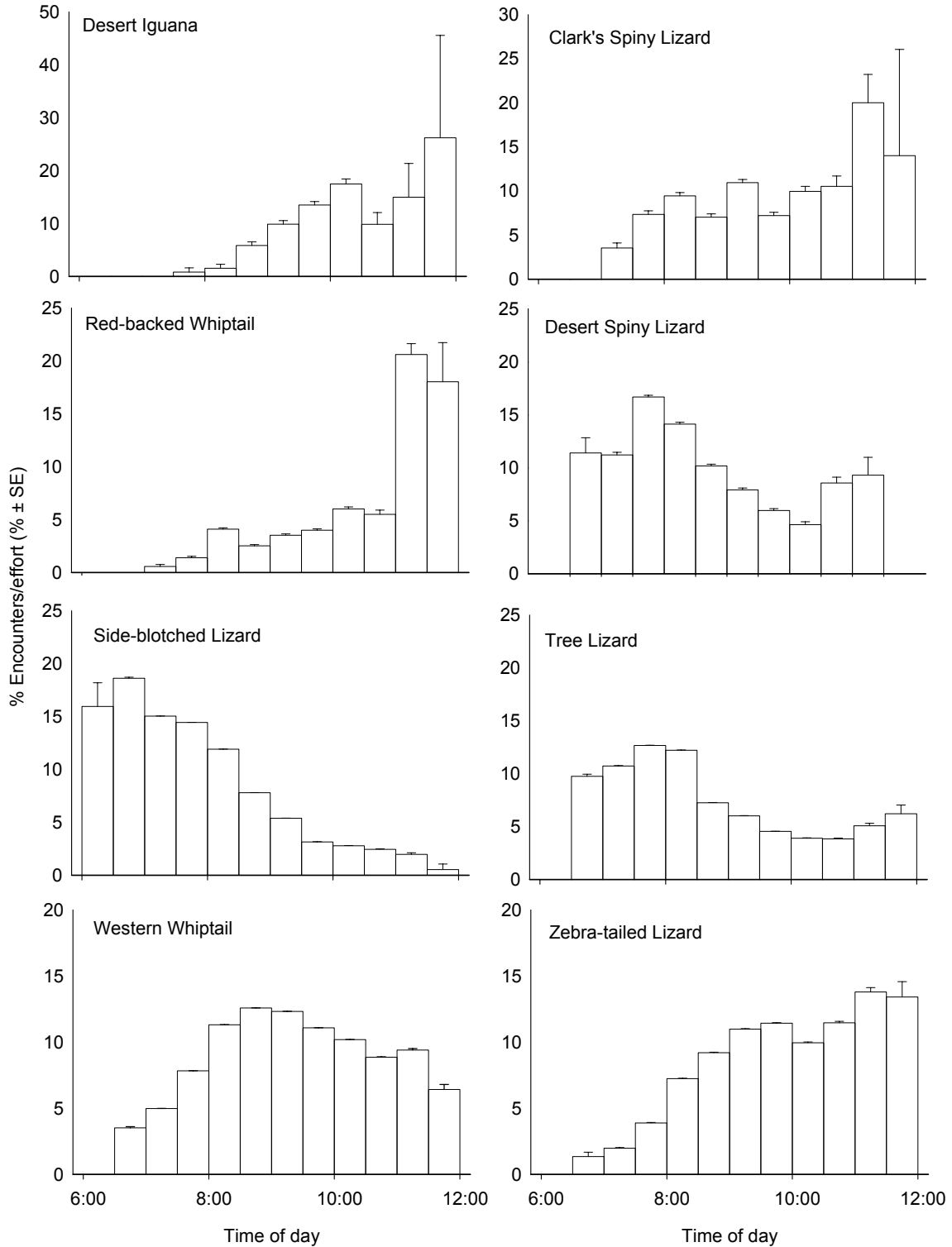


Figure 15: Proportion of encounters of eight species of diurnal lizards across time-of-day in Organ Pipe Cactus National Monument 1989-2006. Proportions are within each 30-min. period were scaled by the amount of effort within each period. Error bars equal one binomial standard error for each proportion and are generally large late in the day when less effort occurred.

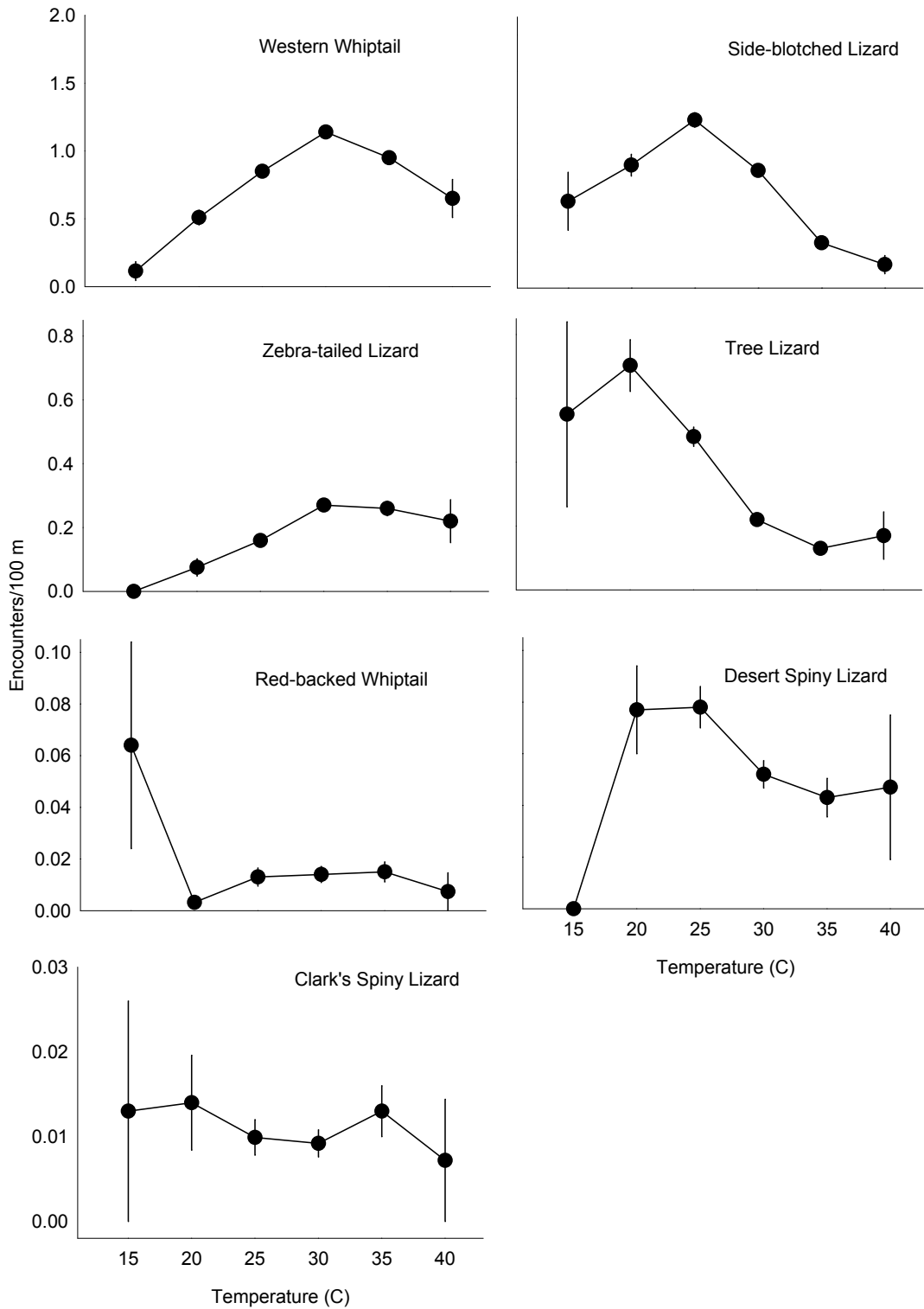


Figure 16: Relationships between encounter rates (no./100 m) and air temperature for seven species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. Point estimates equal the mean encounter rate across all years combined at temperatures ± 2.5 degrees. Error bars equal one standard error and are typically large at temperatures at which few individuals were encountered.

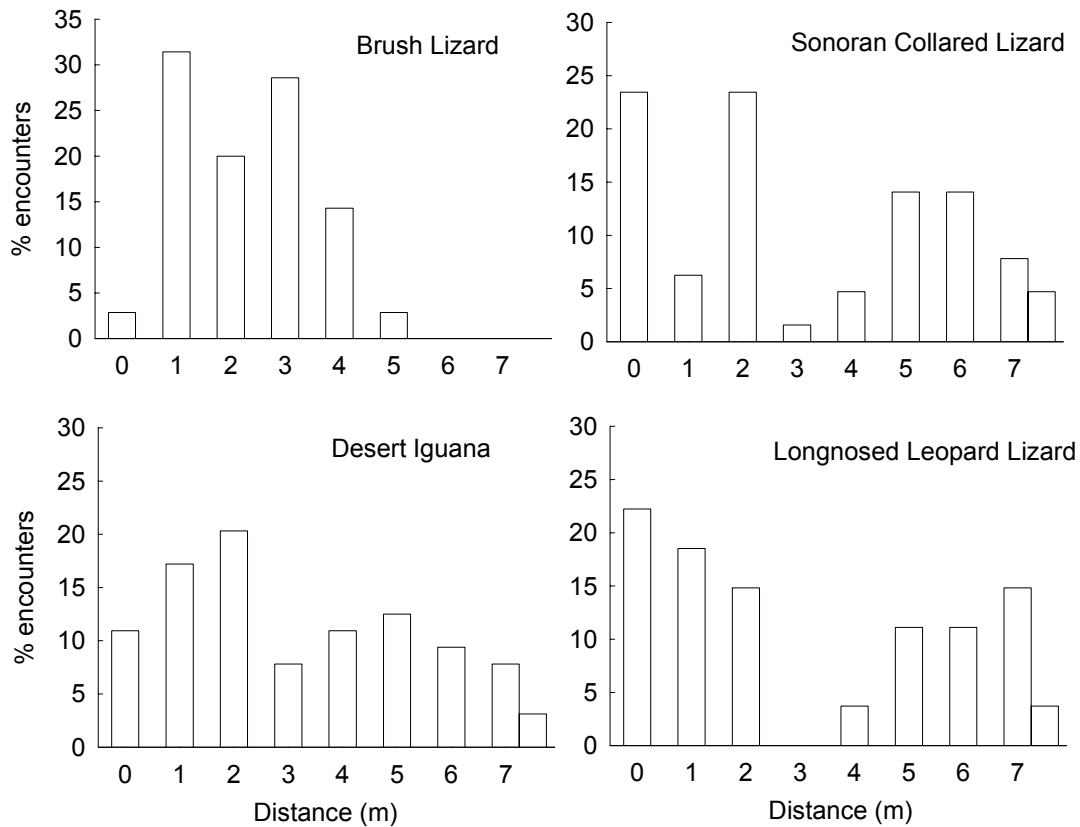


Figure 17: Proportion of encounters within 1-m intervals from transect center lines for four rare species of diurnal lizards in Organ Pipe Cactus National Monument 1989-2006. One small-bodied species (brush lizard) was not detected at distances ≥ 6 m whereas a relatively large proportion of encounters of three large-bodied species were at distances > 5 m.

Detection probability of seven species for which sample sizes were sufficient to estimate a detection function averaged $14.5 \pm 5.4\%$ at 7.5 m from transect centerlines. When encounters of all species were pooled to estimate a detection function, detection probability averaged 9.4% at 7.5 m from the transect centerline. Detection probability at 7.5 m ranged from 1.6% for red-backed whiptail to 42.0% for Clark's spiny lizard. Large-bodied species such as Clark's and desert spiny lizard (24.4%) had higher detection probabilities at 7.5 m compared to small-bodied species such as side-blotched lizard (3.4%), zebra-tailed lizard (7.0%) and western whiptail (10.5%). Although detection functions were not estimable for seven less common species, histograms of detection distances for four of these species with 27-64 total encounters, indicated that 11-19% of detections were at distances ≥ 7 m for three large-bodied species and that no detections occurred at distances > 5 m for a single small-bodied species (brush lizard) (Fig. 17).

Table 9: Relative abundance (max. no./100 m) of six species of diurnal lizards among topographic formations in Organ Pipe Cactus National Monument 1989-2006.

	Rocky Slope		Bajada		Valley Floor		<i>P</i>
	mean	SE	mean	SE	mean	SE	
zebratail lizard	0.48	0.40	0.84	0.27	0.92	0.25	0.62
desert spiny lizard	0.076	0.17	0.37	0.11	0.35	0.10	0.31
side-blotched lizard	2.50	0.81	2.33	0.56	1.99	0.52	0.83
tree lizard	0.63	0.53	1.07	0.35	1.35	0.33	0.51
red-backed whiptail	0.45	0.10	0.00	0.00	0.00	0.00	0.0003
western whiptail	1.32	0.52	2.61	0.34	3.52	0.31	0.0011

Influence of Environmental Factors on Abundance and Trends in Abundance—Relative abundance often varied with topographic, soil, and hydrologic factors. Relative abundance of western whiptail was 2.7 times greater on the valley floor than on rocky slopes whereas red-backed whiptail occurred only on rocky slopes and not on bajadas or the valley floor (Table 9). Relative abundance did not vary among topographic formations for other species ($P \geq 0.31$). Variation in relative abundance between course- and fine-grain soils was similar to that among topographic formations. As such western whiptail was most abundant on fine soils and red-backed whiptail occurred only in areas with coarse soils (Table 10). Relative abundance of desert spiny lizard was almost 3 times greater in areas with fine versus coarse soils. Relative abundance of zebra-tailed lizard, desert spiny lizard, tree lizard, and western whiptail were all greater in xeroriparian areas than in uplands (Table 10), yet relative abundance of side-blotched lizard and red-backed whiptail did not vary.

Although relative abundance of western whiptail varied with topographic, soil, and hydrologic factors, the magnitude and direction of trends in relative abundance across time did not depend on any of these factors ($t_{400} = 1.54$, $P \geq 0.22$; for time \times factor interactions). Similarly, neither did trends for side-blotched lizard, desert spiny lizard, or zebra-tailed lizard ($t_{400} \leq 1.46$, $P \geq 0.23$). For tree lizards, however, relative abundance declined on bajadas yet was relatively stable on valley bottoms across the same time period ($t_{400} = 2.48$, $P = 0.041$; for time \times factor interaction; Fig. 18), and there was also suggestive evidence that relative abundance declined in uplands versus xeroriparian areas ($t_{400} = 1.51$, $P = 0.13$).

Table 10: Relative abundance (max. no./100 m) of six species of diurnal lizards among hydrologic and soil types in Organ Pipe Cactus National Monument 1989-2006.

	Hydrologic Regime					Soil Texture				
	Xeroriparian		Upland		<i>P</i>	Course		Fine		<i>P</i>
	mean	SE	mean	SE		mean	SE	mean	SE	
zebratail lizard	1.18	0.23	0.53	0.21	0.030	0.55	0.27	0.96	0.21	0.21
desert spiny lizard	0.54	0.087	0.13	0.078	0.0006	0.14	0.11	0.40	0.081	0.061
side-blotched lizard	2.01	0.52	2.37	0.46	0.58	2.16	0.57	2.24	0.44	0.90
tree lizard	2.08	0.23	0.35	0.21	<0.0001	0.82	0.36	1.29	0.27	0.30
red-backed whiptail	0.083	0.071	0.087	0.081	0.98	0.23	0.081	0.000	0.000	0.028
western whiptail	3.58	0.32	2.20	0.28	0.0012	1.87	0.36	3.33	0.27	0.0012

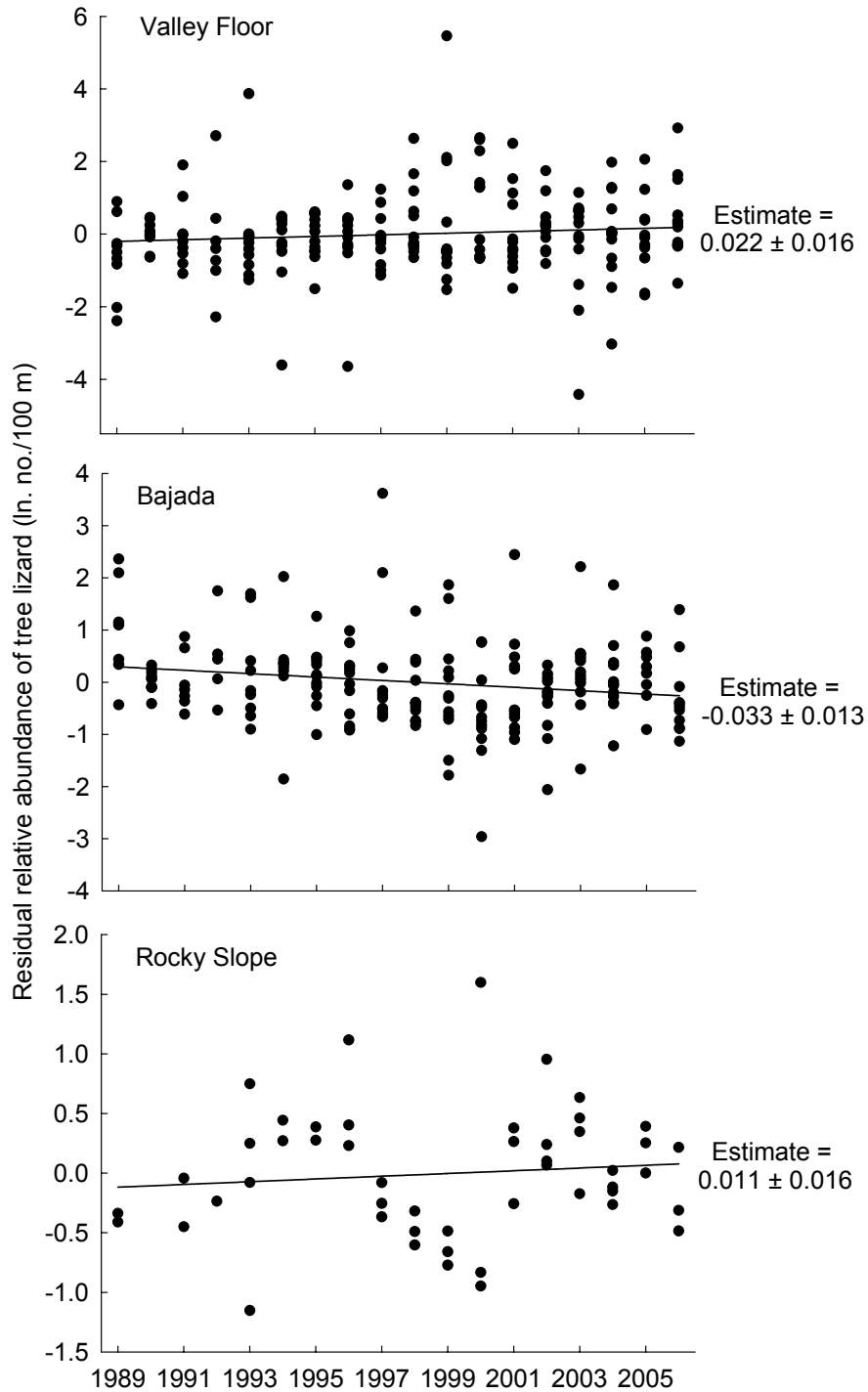


Figure 18: Variation in temporal trends in relative abundance (log max. no./100 m) of tree lizards among three topographic formations in Organ Pipe Cactus National Monument 1989-2006. Residual relative abundance is variation remaining after site (transect) and year were fit as nominal factors. Estimates equal mean annual rate of annual change in relative abundance (± 1 SE).

Influence of Rainfall on Abundance—Relationships between lizard abundance and quantity of rainfall suggested that rainfall was an important driver of population and community dynamics during the study (Fig. 19). Densities of zebra-tailed lizard, tree lizard, and western whiptail increased most with quantity of rainfall during the warm season at a lag time of one year (Table 11). In comparison there was much less evidence that densities varied with seasonal or annual rainfall during the same year (Table 12) or with a lag time of two years (Table 13). For these three species, density increased by as much as $9.1 \pm 1.4\%$ with each additional inch of rainfall that fell the prior warm season (Table 11, Fig. 19). In contrast, there was some evidence ($P = 0.024$) that density of side-blotched lizard increased as annual rainfall increased with no lag time and much stronger evidence ($P = 0.006$) that density declined with annual rainfall at a lag time of one year (Table 11). There was much less evidence that densities varied with cool-season rainfall (Tables 11-13). Similarly, there was little evidence that relative abundance of five less common species varied with annual or seasonal rainfall regardless of lag time (Tables 11-13). Species richness measured on a site-specific scale also increased most strongly with warm-season rainfall at a lag time of one year.

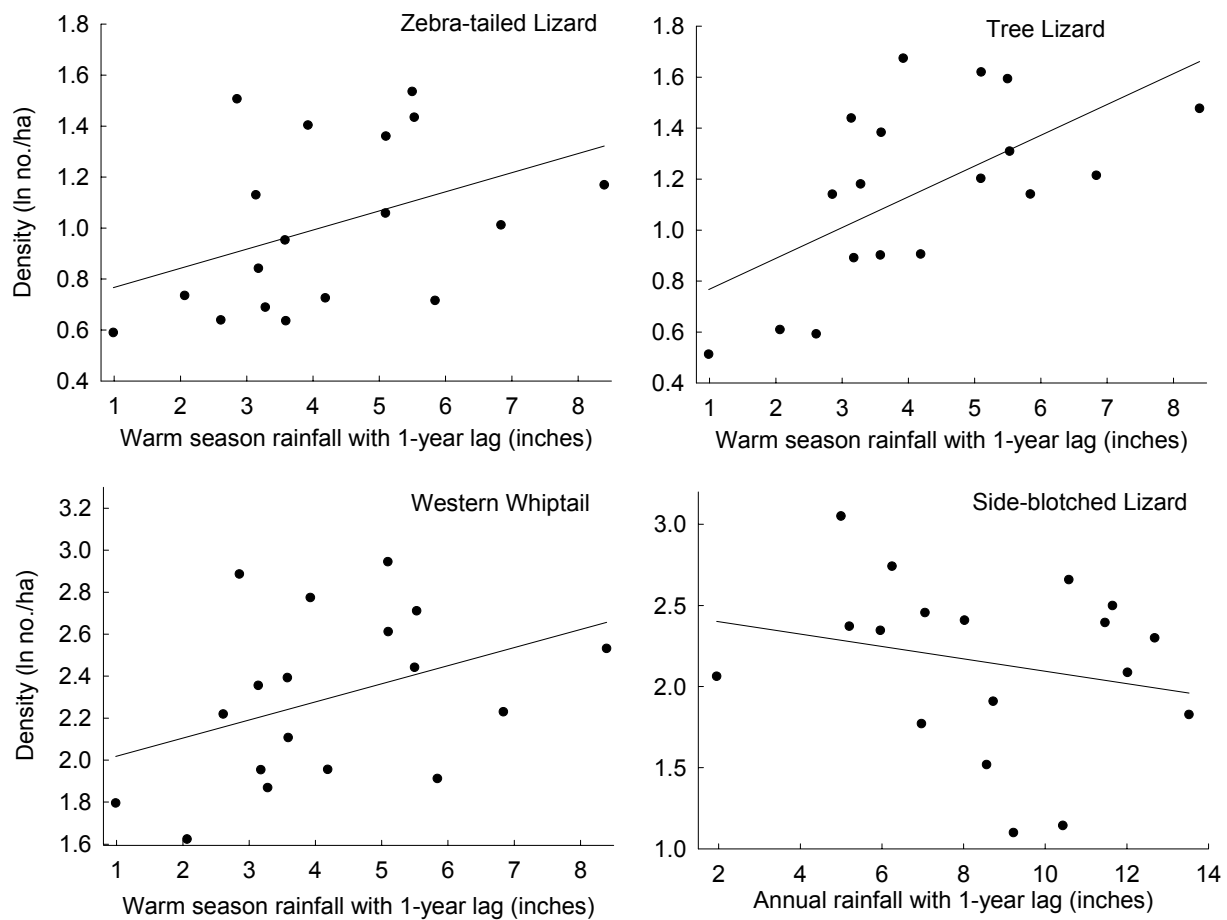


Figure 19: Relationships between density (log no./ha) of the four most abundant species of diurnal lizards and quantity of warm-season (April–September) or annual rainfall measured at a lag time of one year in Organ Pipe Cactus National Monument 1989-2006. Rainfall was measured each month at each site across all years. Plotted values are means for each year at sites where animals were sampled. Because not all sites were visited each year means are adjusted for site effects

Table 11: Associations between diurnal lizards and quantity of rainfall measured at a lag time of one year in Organ Pipe Cactus National Monument 1989-2006. Results are based on generalized linear mixed model for density (log no./ha), relative abundance (log max. no./100 m), and species richness (no./site) in which site was considered the subject and as a random effect and rainfall was repeatedly measured at each site during each month across time.

Response	Warm Season			Cool Season			Annual			
	Species	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>
Density										
zebratail lizard	0.056	0.016	<0.001	-0.024	0.013	0.055	0.006	0.010	0.57	
side-blotched lizard	-0.025	0.016	0.11	-0.028	0.012	0.023	-0.026	0.010	0.006	
tree lizard	0.091	0.014	<0.001	-0.005	0.012	0.65	0.031	0.010	<0.001	
western whiptail	0.067	0.014	<0.001	-0.004	0.012	0.73	0.023	0.009	0.011	
Relative Abundance										
desert iguana	0.0034	0.0041	0.40	-0.0062	0.0030	0.041	-0.0029	0.0025	0.24	
Sonoran collared lizard	-0.0005	0.0019	0.80	0.0018	0.0014	0.20	0.0010	0.0011	0.39	
desert spiny lizard	0.0089	0.0058	0.12	0.0034	0.0043	0.43	0.0054	0.0035	0.12	
Clark's spiny lizard	0.0015	0.0018	0.42	-0.0006	0.0014	0.66	0.0002	0.0011	0.87	
red-backed whiptail	0.0030	0.0026	0.26	-0.0022	0.0021	0.29	-0.0003	0.0016	0.88	
Species Richness										
Observed - site	0.10	0.024	<0.001	0.0065	0.018	0.71	0.040	0.014	0.006	

Table 12: Associations between diurnal lizards and quantity of rainfall measured with no lag time in Organ Pipe Cactus National Monument 1989-2006. Results are based on generalized linear mixed model for density (log no./ha), relative abundance (log max. no./100 m), and species richness (no./site) in which site was considered the subject and as a random effect and rainfall was repeatedly measured at each site during each month across time.

Response	Warm Season			Cool Season			Annual			
	Species	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>
Density										
zebratail lizard	-0.008	0.017	0.63	-0.009	0.012	0.50	-0.008	0.010	0.42	
side-blotched lizard	0.020	0.016	0.20	0.023	0.012	0.055	0.021	0.010	0.024	
tree lizard	-0.033	0.015	0.03	0.000	0.012	0.95	-0.012	0.009	0.20	
western whiptail	-0.015	0.015	0.30	0.027	0.011	0.018	0.011	0.009	0.21	
Relative Abundance										
desert iguana	0.0054	0.0041	0.19	0.0056	0.0030	0.066	0.0055	0.0024	0.025	
Sonoran collared lizard	-0.0033	0.0019	0.086	0.0002	0.0014	0.86	-0.0010	0.0011	0.38	
desert spiny lizard	-0.0010	0.0058	0.87	-0.0028	0.0043	0.51	-0.0021	0.0034	0.53	
Clark's spiny lizard	-0.0017	0.0018	0.36	-0.0010	0.0014	0.49	-0.0012	0.0011	0.28	
red-backed whiptail	-0.0045	0.0027	0.091	0.0043	0.0020	0.032	0.0011	0.0016	0.49	
Species Richness										
Observed - site	-0.0079	0.024	0.75	0.010	0.018	0.58	0.0093	0.014	0.52	

Table 13: Associations between diurnal lizards and quantity of rainfall measured at a lag time of two years in Organ Pipe Cactus National Monument 1989-2006. Results are based on generalized linear mixed model for density (log no./ha), relative abundance (log max. no./100 m), and species richness (no./site) in which site was considered the subject and as a random effect and rainfall was repeatedly measured at each site during each month across time.

Response	Warm Season			Cool Season			Annual			
	Species	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>
Density										
zebratail lizard	-0.002	0.017	0.90	0.007	0.013	0.60	0.003	0.01	0.74	
side-blotched lizard	0.016	0.016	0.34	0.003	0.013	0.83	0.007	0.01	0.46	
tree lizard	-0.014	0.016	0.36	-0.031	0.012	0.015	-0.022	0.01	0.019	
western whiptail	-0.034	0.015	0.022	-0.013	0.012	0.31	-0.02	0.009	0.025	
Relative Abundance										
desert iguana	0.0050	0.0042	0.24	-0.0006	0.0032	0.84	0.0013	0.0025	0.59	
Sonoran collared lizard	0.0015	0.0020	0.46	-0.0007	0.0015	0.64	0.0001	0.0012	0.93	
desert spiny lizard	0.0009	0.0060	0.88	-0.0045	0.0045	0.32	-0.0025	0.0035	0.48	
Clark's spiny lizard	0.0029	0.0019	0.12	-0.0023	0.0015	0.14	-0.0002	0.0011	0.87	
red-backed whiptail	0.0057	0.0027	0.038	-0.0023	0.0022	0.30	0.0008	0.0016	0.64	
Species Richness										
Observed - site	-0.0079	0.024	0.75	0.010	0.018	0.58	0.0093	0.014	0.52	

Influence of Environmental Disturbance—Relative abundance along transects located near the international border did not vary from that along transects in the interior of the park (Table 14). Relative abundance of zebra-tailed lizard, desert spiny lizard, and tree lizard in the interior were all similar to estimates near the border after adjusting for other relevant environmental factors, and precision of these estimates suggested that very large sample sizes would be required to detect differences for these species. Notably, side-blotched lizard and especially western whiptail were more abundant along transects near the border, which may suggest that these species responded positively to disturbance. Temporal trends in relative abundance did not vary between border and interior transects (Table 14).

Table 14: Mean relative abundance (max. no./100 m) of five species of diurnal lizards at six sites along the international border and 13 sites in the interior of Organ Pipe Cactus National Monument 1989-2006. Means are adjusted for the influence of environmental factors that explained variation ($P \leq 0.10$) in relative abundance. Sites along the border were presumed to have higher levels of disturbance due to human and vehicle traffic that are related to human migration and smuggling. Disturbance by time interactions tested whether the slope of trends in relative abundance across time depended on whether sites were along the border or in the interior.

	Location					Location x time	
	Border		Interior		<i>P</i>	<i>t</i>	<i>P</i>
	mean	SE	mean	SE			
zebratail lizard	0.72	0.32	0.90	0.19	0.62	1.26	0.21
desert spiny lizard	0.31	0.12	0.34	0.076	0.85	0.49	0.62
side-blotched lizard	3.25	0.67	1.80	0.41	0.064	0.70	0.48
tree lizard	1.15	0.33	1.24	0.21	0.82	0.25	0.80
western whiptail	3.41	0.39	2.19	0.23	0.0050	1.17	0.24

Nocturnal Rodents

Effort and Detections—Over 16 years, ORPI staff sampled up to 34 trap grids per year and a total of 432 two-sample occasion trapping events over all years combined (Table 15). On average, 14.3 ± 0.8 (\pm SE) sites and 27.0 ± 1.5 trap grids (range = 14-34) were sampled each year and the vast majority of sites ($90 \pm 1\%$) contained two trap grids. Effort was relatively consistent during each year with the exception of substantially lower effort during the first two years of study. Only seven, two-sample occasion trapping events used a 12 x 12 trap design (in 1998 and 1999). On average trapping began 26 June and ended on 2 August (SE = 0.6 days) across all years combined.

ORPI staff captured 16,835 individuals and obtained 20,715 total captures of 10 species during the study (Table 16). Additionally, two non-target species, *Spermophilus tereticaudus* (round-tailed ground squirrel) and *Ammospermophilus harrisi* (Harris' antelope squirrel) were also trapped. Three species of nocturnal rodents that have been detected in ORPI in the past were not detected during the study; *Peromyscus merriami* (mesquite mouse; Petryszyn and Russ 1996), *Perognathus longimembris* (little pocket mouse; Petryszyn and Cockrum 1990), and *Dipodomys deserti* (desert kangaroo rat; Hoffmeister 1986). *Thomomys bottae* (Botta's pocket gopher) also occur in ORPI yet are rarely captured by Sherman traps. Total encounters for each species over all years ranged from 16 for *Sigmodon arizonae* (Arizona cotton rat) to 6,788 for *Chaetodipus penicillatus* (desert pocket mouse; Table 16). Mean, minimum, and maximum numbers of encounters among years varied markedly among species. Only four species (40%) averaged >100 encounters per year and three species *Onychomys torridus* (southern grasshopper mouse), *Dipodomys spectabilis* (banner-tail kangaroo rat), and *S. arizonae*, averaged <6 encounters per year.

Table 15: Effort and timing of nocturnal rodents trapping across time in Organ Pipe Cactus National Monument 1991-2006.

Year	Start date	End date	Sites	Grids	No. of sites with 1 grid	No. of sites with 2 grids	No. 7x7 grids ^a	No. 12x12 grids ^b
1991	2-Jul	26-Jul	7	14	0	7	14	0
1992	23-Jun	15-Jul	7	14	0	7	14	0
1993	23-Jun	23-Jul	12	23	1	11	23	0
1994	22-Jun	22-Jul	13	25	1	12	25	0
1995	27-Jun	30-Jul	18	34	2	16	34	0
1996	21-Jun	7-Aug	18	34	2	16	34	0
1997	17-Jun	2-Aug	15	28	2	13	28	0
1998	16-Jun	1-Aug	14	27	1	13	23	4
1999	15-Jun	12-Aug	14	27	1	13	24	3
2000	4-Jul	3-Aug	16	30	2	14	30	0
2001	1-Jul	3-Aug	16	30	2	14	30	0
2002	3-Jul	29-Jul	15	28	2	13	28	0
2003	10-Jul	9-Aug	15	28	2	13	28	0
2004	6-Jul	12-Aug	16	30	2	14	30	0
2005	29-Jun	14-Aug	16	30	2	14	30	0
2006	28-Jun	23-Aug	16	30	2	14	30	0
Totals			228	432	24	204	425	7
Mean	26-Jun	2-Aug	14.3	27.0	1.5	12.8	26.6	0.4
SE	0.6	0.6	0.8	1.5	0.2	0.6	1.5	0.3

^a traps with 15 m spacing

^b traps with 10 m spacing

Table 16: Species and number of captures of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. Individuals were not permanently marked and only discernable within years. Latin names of rodents were used in the text.

Family	Species	Common name	Individuals	Individuals per year			
				Min.	Max.	Mean	SE
Heteromyidae	<i>Chaetodipus baileyi</i>	Bailey's pocket mouse	1949	10	302	121.8	21.5
	<i>Chaetodipus intermedius</i>	rock pocket mouse	646	1	102	40.4	7.0
	<i>Chaetodipus penicillatus</i>	desert pocket mouse	6788	265	549	424.3	23.2
	<i>Perognathus amplus</i>	Arizona pocket mouse	1814	29	136	79.1	10.0
	<i>Dipodomys merriami</i>	Merriam's kangaroo rat	3812	84	567	238.3	33.0
	<i>Dipodomys spectabilis</i>	banner-tail kangaroo rat	21	0	4	1.8	0.3
Muridae	<i>Neotoma albigula</i>	white-throated woodrat	1265	44	194	113.4	12.8
	<i>Onychomys torridus</i>	grasshopper mouse	84	1	15	5.6	1.0
	<i>Peromyscus eremicus</i>	cactus mouse	440	11	65	27.5	3.6
	<i>Sigmodon arizonae</i>	Arizona cotton rat	16	1	8	3.2	1.2

Temporal Trends—Abundance varied markedly across time for all eight species for which abundance was estimable (Figs. 20-21), yet importantly, there was no evidence of any systematic declines in abundance (Table 17). Both abundance and relative abundance of *Chaetodipus baileyi* (Bailey's pocket mouse) and *Perognathus amplus* (Arizona pocket mouse) increased across time (Table 17); relative abundance of *C. baileyi* increased by an average of $9.7 \pm 2.4\%$ per year whereas relative abundance of *P. amplus* increased by $4.1 \pm 1.3\%$ per year. Although there was also evidence that abundance and relative abundance of *P. amplus* reached asymptotes in later years of the study, systematic linear trends in both parameters explained more variation overall (Table 17). Relative abundance of *Chaetodipus intermedius* (rock pocket mouse) also increased by an average of $5.6 \pm 2.2\%$ per year, yet there was much weaker evidence ($P = 0.13$) of a similar trend ($5.1 \pm 3.3\%$ per year) in abundance due to higher levels of inter-annual variation among estimates across the study. Temporal trends in abundance and relative abundance of *C. penicillatus*, *Dipodomys merriami* (Merriam's kangaroo rat), *Neotoma albigula* (white-throated woodrat), and *Peromyscus eremicus* (cactus mouse) were moderately to strongly curvilinear ($P \leq 0.055$). For these species, abundance was high during early and late years and low during middle years of the study (Figs. 20-21). Abundance of *O. torridus* was somewhat higher during middle years than during early or late years ($P = 0.088$).

Age and reproductive structure varied across time in some populations of rodents. The proportion of individuals that were adults declined systematically across time in populations of *N. albigula* ($t_{211} = 3.31$, $P = 0.0011$; Fig. 22) and increased in populations of *P. amplus* ($t_{253} = 2.28$, $P = 0.024$). In contrast, no temporal trends in age structure occurred in populations of *C. baileyi*, *C. penicillatus*, and *D. merriami* ($t \leq 1.42$, $P \geq 0.16$). Interestingly, the proportion of individuals that were reproductively active (e.g., lactating, pregnant, or testes descended) declined systematically across time (1997-2006) in populations of *C. penicillatus*, *C. baileyi*, *D. merriami*, and *P. amplus* ($t \geq 2.25$, $P \leq 0.026$; Fig. 23) and the magnitude of change was greatest in populations of *P. amplus*, a species that increased in abundance across the same period (Table 17). Reproductive structure in populations of *N. albigula* did not vary across time ($t_{133} = 0.25$, $P = 0.80$). Small sample sizes precluded estimation of changes in age and reproductive structure for five additional species.

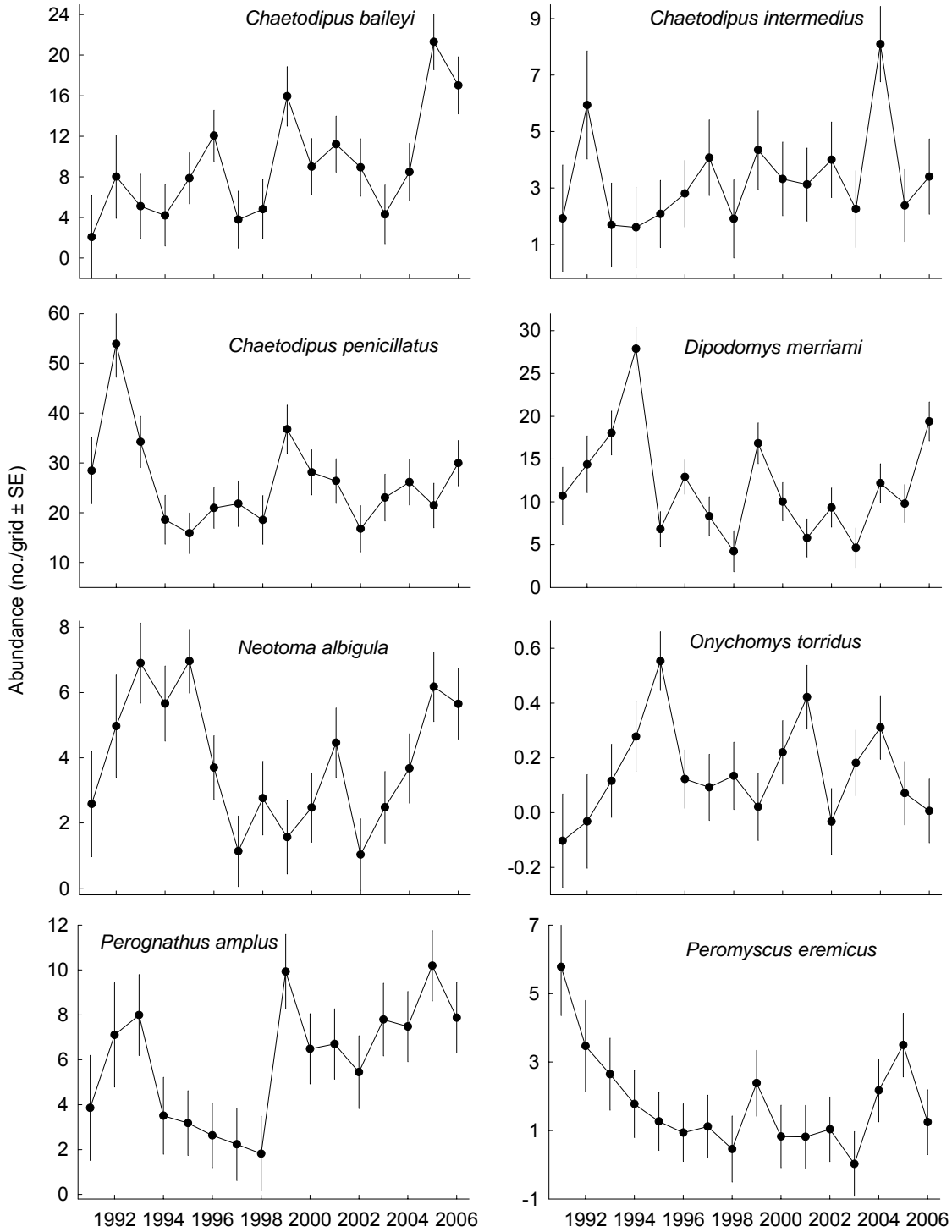


Figure 20: Variation in abundance (no./grid) across time for the eight most common species of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. Abundance is adjusted for site effects to account for the addition of several new sites in the mid 1990's. Abundance was calculated with use of the Lincoln-Peterson estimator. Standard errors are based on site-adjusted means among years.

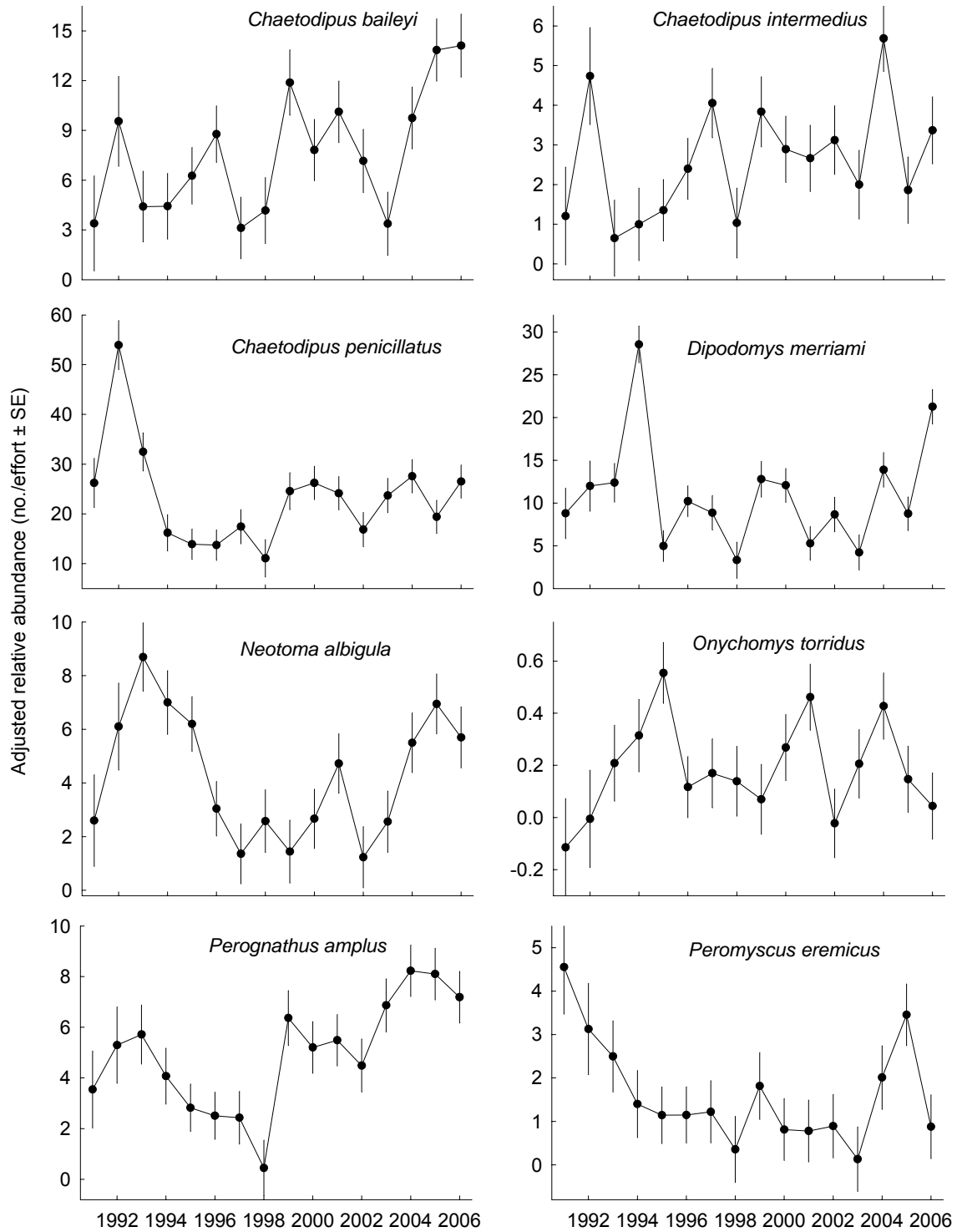


Figure 21: Variation in relative abundance (no./100 trap nights) across time for the eight most common species of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. Relative abundance was adjusted for effects of site, moon phase, and time-of-year. Effort was calculated by assigning all sprung traps a value of 0.5 trap intervals.

Table 17: Trend estimates for populations of nocturnal rodents and for species richness in Organ Pipe Cactus National Monument 1991-2006. Results are based on generalized linear mixed models for abundance (no./grid) and relative abundance (no./100 trap nights) in which moon phase, day-of-year, and day-of-year² were considered as covariates when they described variation ($P \leq 0.10$) in parameters; site was considered the subject and a random effect. Estimates equal the annual change in parameters and can be converted to percent change by dividing parameter values by (see Table 19).

Response	Linear Trend					Quadratic Trend				
	Species	Est.	SE	<i>t</i>	<i>P</i>	RMSE	Est.	SE	<i>t</i>	<i>P</i>
Relative Abundance										
<i>Chaetodipus baileyi</i>	0.60	0.150	3.92	0.0001	8.87	0.051	0.035	1.45	0.14	
<i>Chaetodipus intermedius</i>	0.12	0.047	2.63	0.0089	5.49	-0.0045	0.012	0.38	0.70	
<i>Chaetodipus penicillatus</i>	-0.045	0.260	0.17	0.87	18.86	0.21	0.0600	3.47	0.0006	
<i>Perognathus amplus</i>	0.320	0.074	4.33	<0.0001	5.26	0.049	0.017	2.84	0.0047	
<i>Dipodomys merriami</i>	0.150	0.19	0.76	0.45	11.53	0.14	0.040	3.44	0.0006	
<i>Neotoma albigula</i>	-0.019	0.10	0.19	0.84	5.63	0.064	0.021	2.99	0.0029	
<i>Onychomys torridus</i>	-0.001	0.0077	0.08	0.94	0.74	-0.0027	0.0019	1.43	0.15	
<i>Peromyscus eremicus</i>	-0.065	0.058	1.12	0.27	3.48	0.0017	0.00087	1.92	0.055	
Heteromyidae	1.38	0.47	2.91	0.0038	27.58	0.42	0.098	4.31	<0.0001	
Muridae	-0.095	0.13	0.74	0.46	7.28	0.10	0.027	3.77	0.0002	
Abundance										
<i>Chaetodipus baileyi</i>	0.78	0.20	3.92	0.0001	13.85	0.048	0.046	1.05	0.300	
<i>Chaetodipus intermedius</i>	0.12	0.079	1.50	0.13	7.72	0.00036	0.018	0.02	0.98	
<i>Chaetodipus penicillatus</i>	-0.21	0.33	0.64	0.52	23.64	0.15	0.073	2.03	0.043	
<i>Perognathus amplus</i>	0.34	0.11	3.20	0.0015	8.02	0.044	0.026	1.73	0.085	
<i>Dipodomys merriami</i>	-0.12	0.19	0.61	0.54	12.34	0.14	0.042	3.27	0.0012	
<i>Neotoma albigula</i>	-0.034	0.087	0.39	0.70	5.36	0.055	0.019	2.84	0.0047	
<i>Onychomys torridus</i>	-0.0033	0.0071	0.46	0.65	0.68	-0.003	0.0017	1.71	0.088	
<i>Peromyscus eremicus</i>	-0.097	0.079	1.21	0.23	4.34	0.0430	0.017	2.48	0.013	
Heteromyidae	1.280	0.49	2.61	0.0095	33.57	0.3900	0.11	3.56	0.0004	
Muridae	-0.13	0.11	1.15	0.25	7.16	0.0970	0.025	3.90	0.0001	
Species Richness - scale										
Observed - site	0.022	0.014	1.58	0.11	0.90	0.00071	0.0035	0.21	0.84	
Observed - study area	0.029	0.028	1.05	0.31	0.51	-0.005	0.007	0.65	0.52	
Estimated - study area	0.010	0.052	0.20	0.85	0.96	0.014	0.013	1.10	0.29	

Average mass of individuals within most populations of rodents varied little across time. Mass of individuals in populations of *D. merriami*, however, increased systematically even after age structure was considered a covariate ($t_{325} = 2.26$, $P = 0.035$; Fig. 24). In contrast, mass of individuals in populations *C. intermedius* decreased across time, a trend that was influenced strongly by very high estimates in 1992 and 1993 ($t_{94} = 2.56$, $P = 0.012$). Mass did not vary systematically in populations of *C. penicillatus*, *C. baileyi*, *N. albigula*, or *P. amplus* ($t \leq 0.96$, $P \geq 0.34$) yet there were curvilinear trends in populations of *C. penicillatus* and *P. amplus* ($t \geq 2.06$, $P \leq 0.037$; Fig. 24).

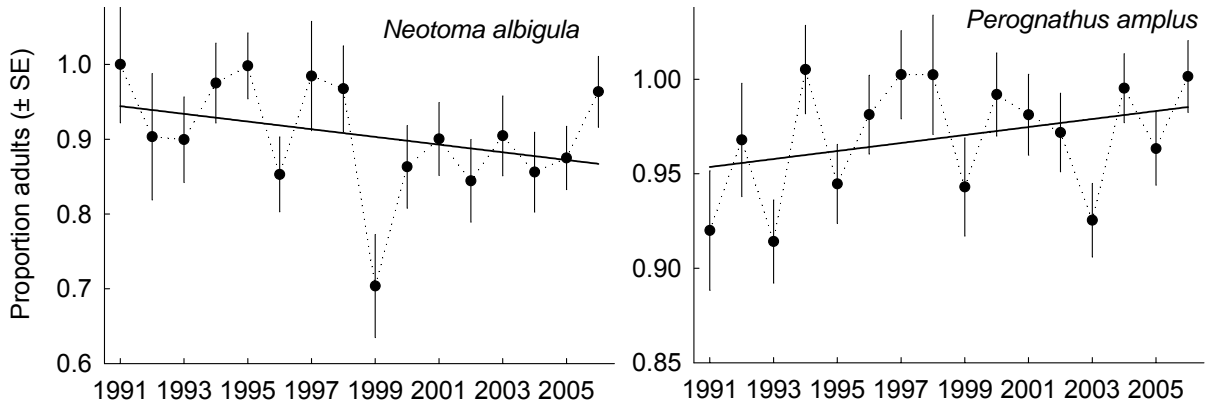


Figure 22: Trends in age structure in populations of two species of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. Estimates are mean proportions of individuals that were adults among sites during each year (± 1 SE) and are adjusted for site effects.

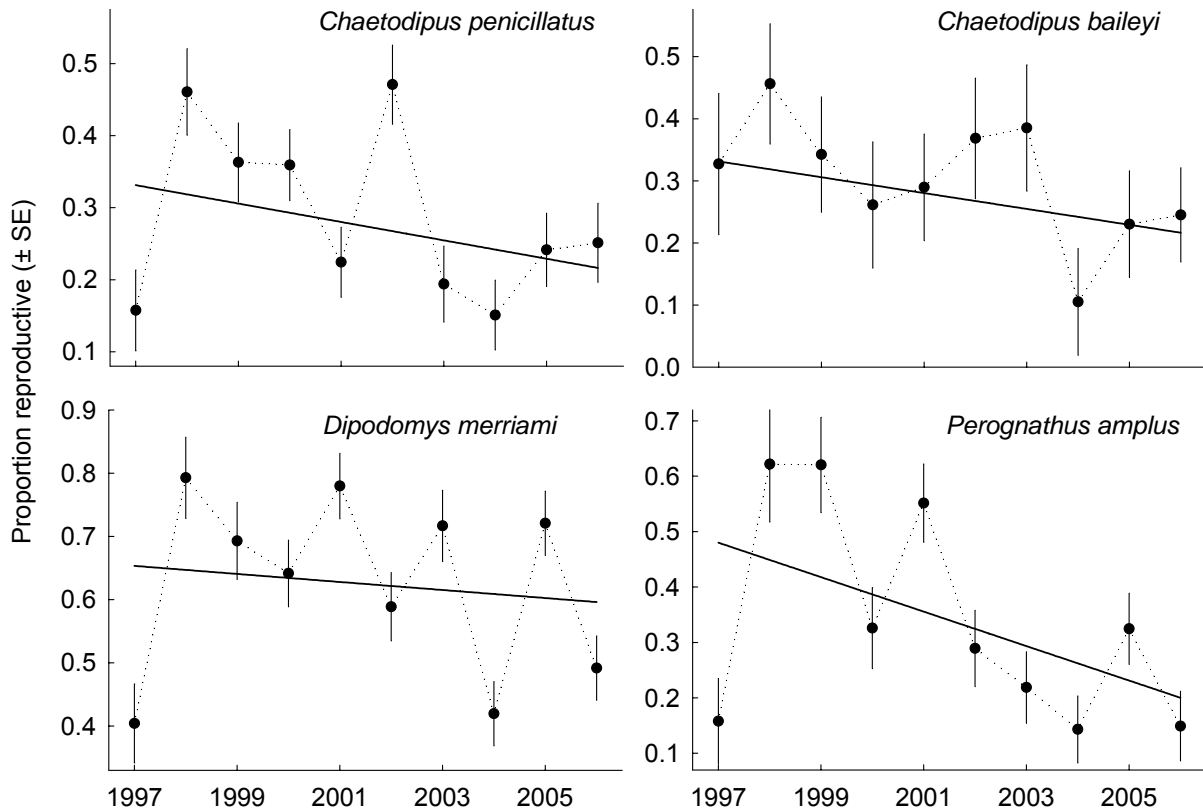


Figure 23: Trends in reproductive structure in populations of four species of nocturnal rodents in Organ Pipe Cactus National Monument 1997-2006. Estimates are mean proportions of individuals that were reproductive among sites during each year (± 1 SE) and are adjusted for site effects.

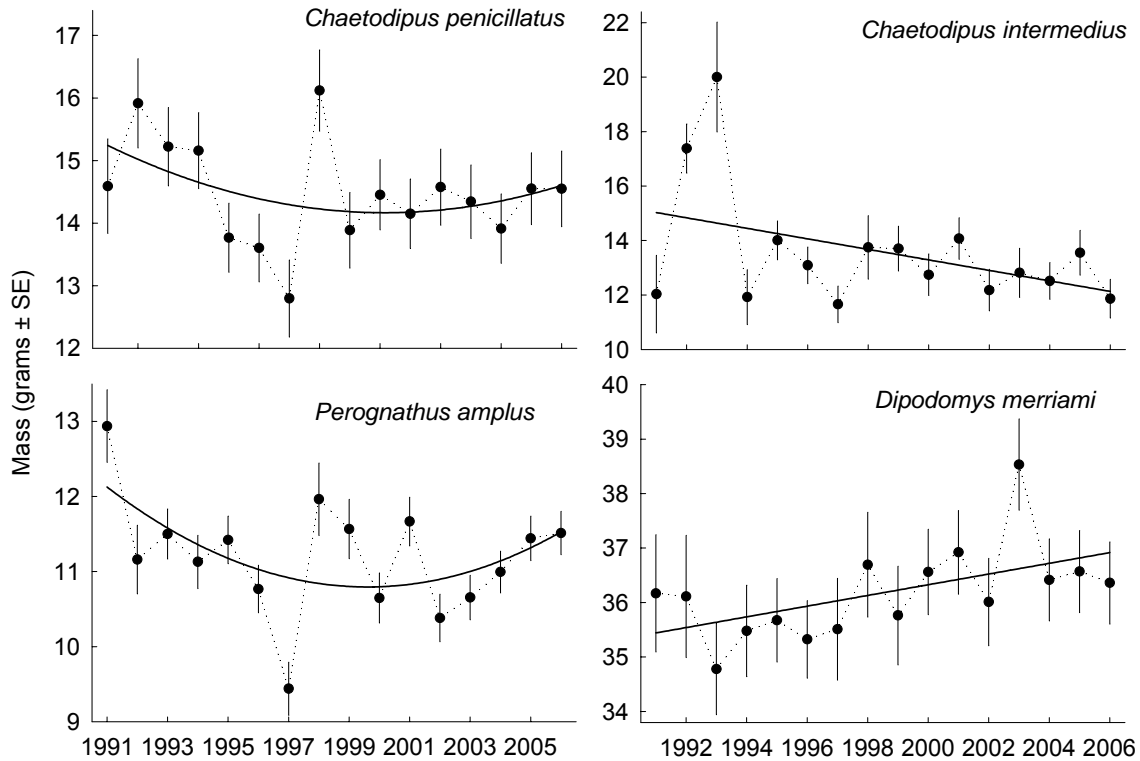


Figure 24: Trends in mass of individuals in populations of four species of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. Estimates are means among all individuals at each site during each year (± 1 SE) and are adjusted for the effects of site and age structure.

Estimated and observed species richness did not vary systematically across time (Table 17). Overall, however richness was low and varied from only 8-12 species among years and as such the jackknife estimator performed poorly. Estimated and observed species richness were highest in 1998 and in 2005. Observed species richness at the scale of study sites, however, increased somewhat by $3.7 \pm 2.3\%$ over the duration of the study ($P = 0.11$; Fig. 25), a trend that corresponded with that for increasing abundance and relative abundance ($t_{413} \geq 5.41$, $P < 0.0001$) of all species combined across time.

Composition of the rodent community varied across time. The first three principal components explained 28%, 14%, and 13% of variation, respectively, in estimates of abundance of the eight most common species of rodents across space and time. PC2 was negatively correlated with abundance of *P. amplus* (-0.49), *C. baileyi*, (-0.41), *D. merriami* (-0.14; $P \leq 0.0030$ for all correlations), and to a lesser extent *C. intermedius* (-0.09, $P = 0.069$), and abundance in populations of three of these species systematically increased across the study. Conversely, PC2 was positively correlated with abundance of *C. penicillatus* ($r = 0.65$), *O. torridus* ($r = 0.46$), and *P. eremicus* ($r = 0.20$; $P < 0.0001$), and populations of these species had relatively high abundance during early to middle years of the study and declined thereafter (Figs. 20-21). PC2 declined systematically across time ($t_{413} = 4.90$, $P < 0.0001$; Fig. 26) indicating changes in community composition that were characterized by relatively higher abundance of *P. amplus*, *C. baileyi*, *D. merriami*, and *C. intermedius* and relatively lower abundance of *C. penicillatus*, *O. torridus*, and *P. eremicus*. Correlations for PC1 and PC3 were ambiguous with respect to community composition and are not reported.

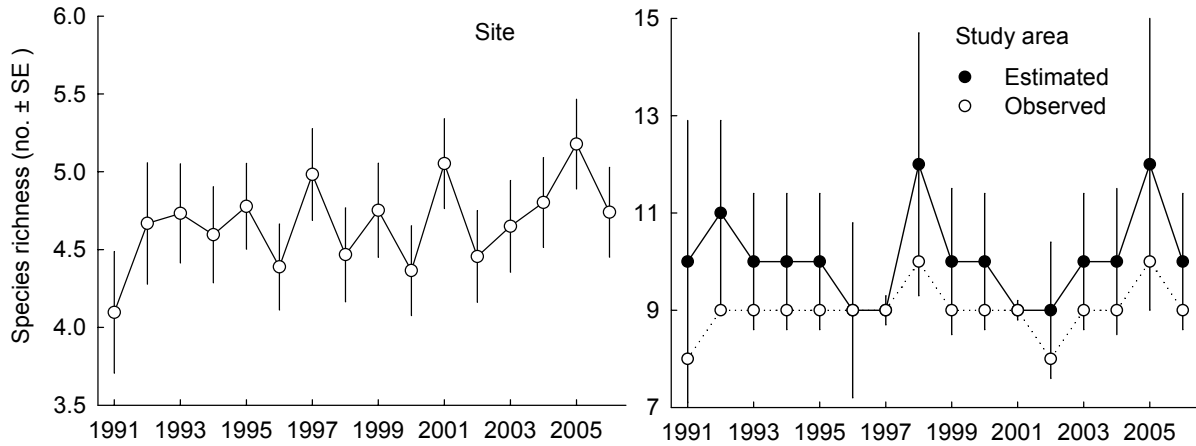


Figure 25: Temporal trends in species richness of nocturnal rodents in Organ Pipe National Cactus Monument 1991-2006. Observed richness at the scale of sites is equal to the average (\pm 1 SE) number of species detected during each year. Estimated richness was calculated for the entire study area using model M_h and the jackknife estimator. Observed species richness at the scale of the study area was equal to the raw number of species detected each year.

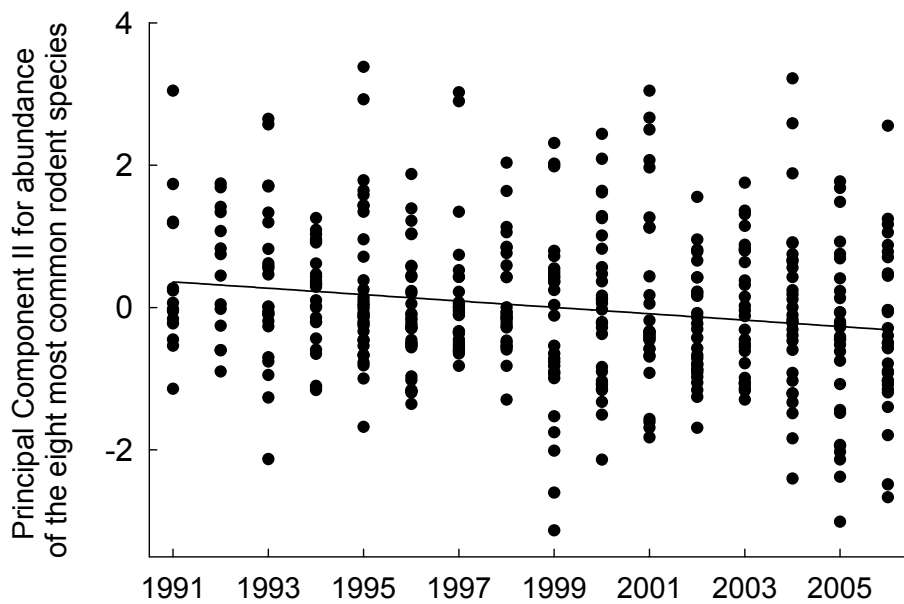


Figure 26: Temporal trends in community composition of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. Values are based on the second principal component that explained 14% of total variation in estimates of abundance of the eight most common species. Loadings were negative for four species, three of which increased in abundance across time. Trend line ($P < 0.0001$) is based on a generalized linear mixed model adjusted for site effects.

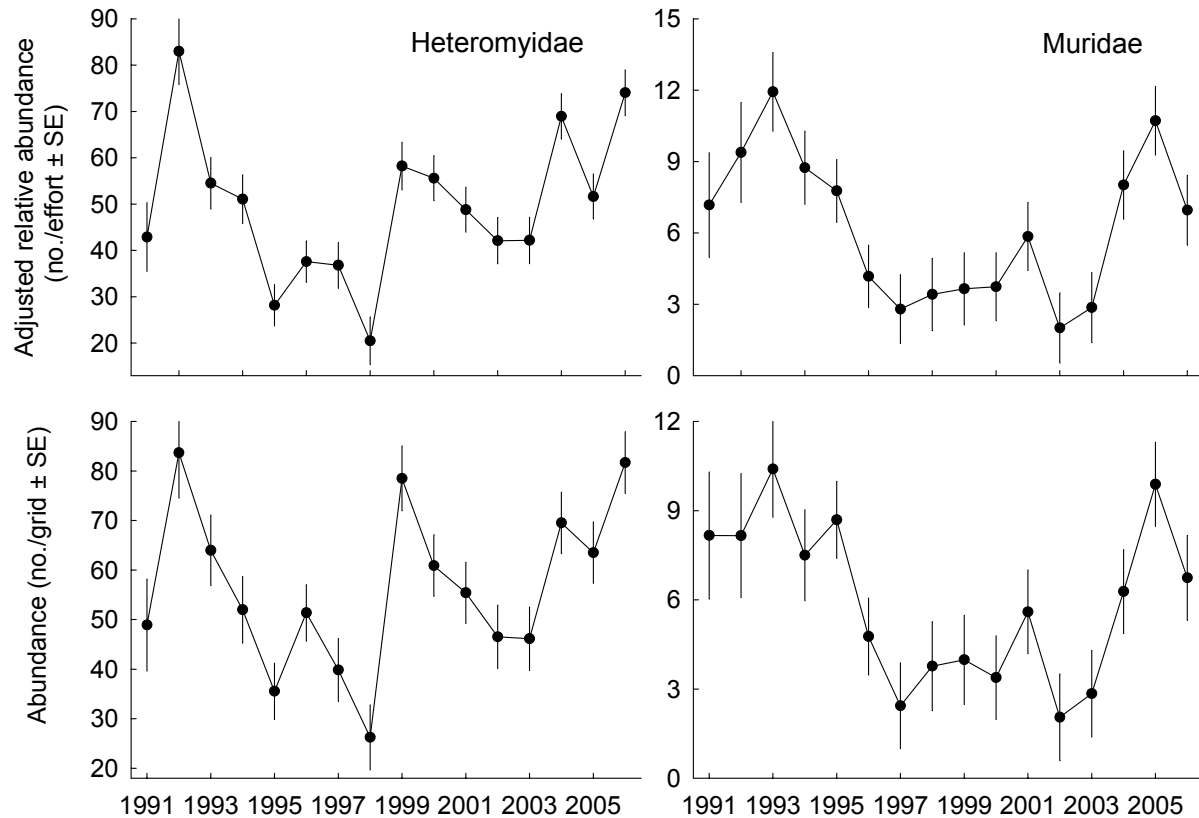


Figure 27: Variation in abundance (no./grid) and relative abundance (no./100 trap nights) of two families of nocturnal rodents across time in Organ Pipe Cactus National Monument 1991-2006. Effort was calculated by assigning sprung traps a value of 0.5 trap intervals and abundance was calculated with use of the Lincoln-Peterson estimator. Standard errors are based on site-adjusted means among years.

Abundance of individuals in each family of rodents varied across time. Both abundance and relative abundance of heteromyids increased systematically across time yet there was stronger evidence of curvilinear trends characterized by high abundance during both early and especially late years of the study (Table 17, Fig. 27). Although there were no linear trends for murids across time, curvilinear trends were somewhat similar to those for heteromyids; abundance was high during early and late years of the study yet was relatively low during middle years, especially compared to that for heteromyids (Fig. 27). Despite some similarities in trends across time between families, site- and year-specific estimates of abundance of heteromyids and murids exhibited only a weak negative correlation ($r = -0.14$, $P = 0.0032$).

Relationships between Population Parameters—Site- and year-specific estimates of abundance were highly correlated with estimates of relative abundance measured at the same scales for the same species ($P < 0.0001$). Correlation coefficients ranged from as low as 0.86 for *C. intermedius* to as high as 0.97 for *O. torridus* and averaged 0.91 ± 0.01 among all eight species. Distributions of estimates of relative abundance for each of the five most common species however, had lower kurtosis and therefore shorter tails than those for estimates of abundance.

Table 18: Pearson correlation coefficients for annual estimates of abundance (no./grid) between all possible pairs of 10 species of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006.

	<i>Chaetodipus baileyi</i>	<i>Chaetodipus intermedius</i>	<i>Chaetodipus penicillatus</i>	<i>Dipodomys merriami</i>	<i>Dipodomys spectabilis</i>	<i>Neotoma albigula</i>	<i>Onychomys torridus</i>	<i>Perognathus amplus</i>	<i>Peromyscus eremicus</i>	<i>Sigmodon arizonae</i>
<i>Chaetodipus baileyi</i>		0.24 ^a	-0.21 ^a	-0.26 ^a	-0.06	0.11 ^b	-0.13 ^a	-0.16 ^a	0.13 ^a	-0.04
<i>Chaetodipus intermedius</i>	0.24 ^a		-0.11 ^b	-0.21 ^a	-0.04	0.24 ^a	-0.08 ^c	-0.16 ^a	0.24 ^a	-0.03
<i>Chaetodipus penicillatus</i>	-0.21 ^a	-0.11 ^b		0.06	0.02	-0.13 ^a	0.06	-0.05	-0.12 ^a	0.09 ^c
<i>Dipodomys merriami</i>	-0.26 ^a	-0.21 ^a	0.06		0.15 ^a	-0.29 ^a	0.12 ^a	0.28 ^a	-0.24 ^a	0.00
<i>Dipodomys spectabilis</i>	-0.06	-0.04	0.02	0.15 ^a		-0.02	-0.02	0.08 ^c	-0.05	-0.02
<i>Neotoma albigula</i>	0.11 ^b	0.24 ^a	-0.13 ^a	-0.29 ^a	-0.02		-0.07	-0.15 ^a	0.48 ^a	-0.05
<i>Onychomys torridus</i>	-0.13 ^a	-0.08 ^c	0.06	0.12 ^b	-0.02	-0.07		-0.02	-0.07	0.05
<i>Perognathus amplus</i>	-0.16 ^a	-0.16 ^a	-0.05	0.28 ^a	0.08 ^c	-0.15 ^a	-0.02		-0.19 ^a	-0.06
<i>Peromyscus eremicus</i>	0.13 ^a	0.24 ^a	-0.12 ^a	-0.24 ^a	-0.05	0.48 ^a	-0.07	-0.19 ^a		-0.02
<i>Sigmodon arizonae</i>	-0.04	-0.03	0.09 ^c	0.00	-0.02	-0.05	0.05	-0.06	-0.02	

^a $P \leq 0.01$

^b $P \leq 0.05$

^c $P \leq 0.10$

Correlations in annual estimates of abundance between all possible pairs of species varied widely (Table 18). Overall however, there were fairly high levels of correlation between 49% of all possible pairs of species ($P < 0.05$). Positive correlations in abundance were highest between the murids *N. albigula* and *P. eremicus* ($r = 0.48$), and between the heteromyids *D. merriami* and *P. amplus* (0.28), and *C. intermedius* and *C. baileyi* (0.24) (Table 18). Negative correlations in abundance were highest between murids and heteromyids (e.g., *N. albigula* the *D. merriami*, $r = -0.29$; *P. eremicus* and *D. merriami*, $r = -0.24$), and between the heteromyids *D. merriami* and *C. baileyi* (-0.26). Abundance of *O. torridus* and *D. spectabilis*, species that were rare in ORPI, exhibited the lowest levels of correlation with other species of rodents.

Because annual estimates of abundance for some pairs of species covaried, trends in abundance across time were similar for many of these same species. Trends in abundance over time for example, were similar for *C. intermedius* and *C. baileyi* and for *P. amplus* and *D. merriami* (Fig. 28). Further, variation in abundance over time was also similar for the two most abundant species of murids (*N. albigula*, *P. eremicus*), whereas that for the insectivorous *O. torridus* varied (Fig. 28)

In contrast to patterns in many populations of lizards, annual estimates of rodent abundance were not correlated with annual estimates of age structure ($P > 0.05$). Further, annual estimates of mass were not correlated with estimates of abundance during the same year or at a lag time of one year ($P > 0.05$). The proportion of individuals that were reproductively active increased as the average mass of individuals increased in populations of *P. amplus* ($r = 0.63$, $P = 0.052$) and *C. penicillatus* ($r = 0.66$, $P = 0.037$) but not in other populations ($P \geq 0.14$). I found no other significant correlations between other population parameters in other species ($P > 0.05$) except for obvious correlations between age structure and mass.

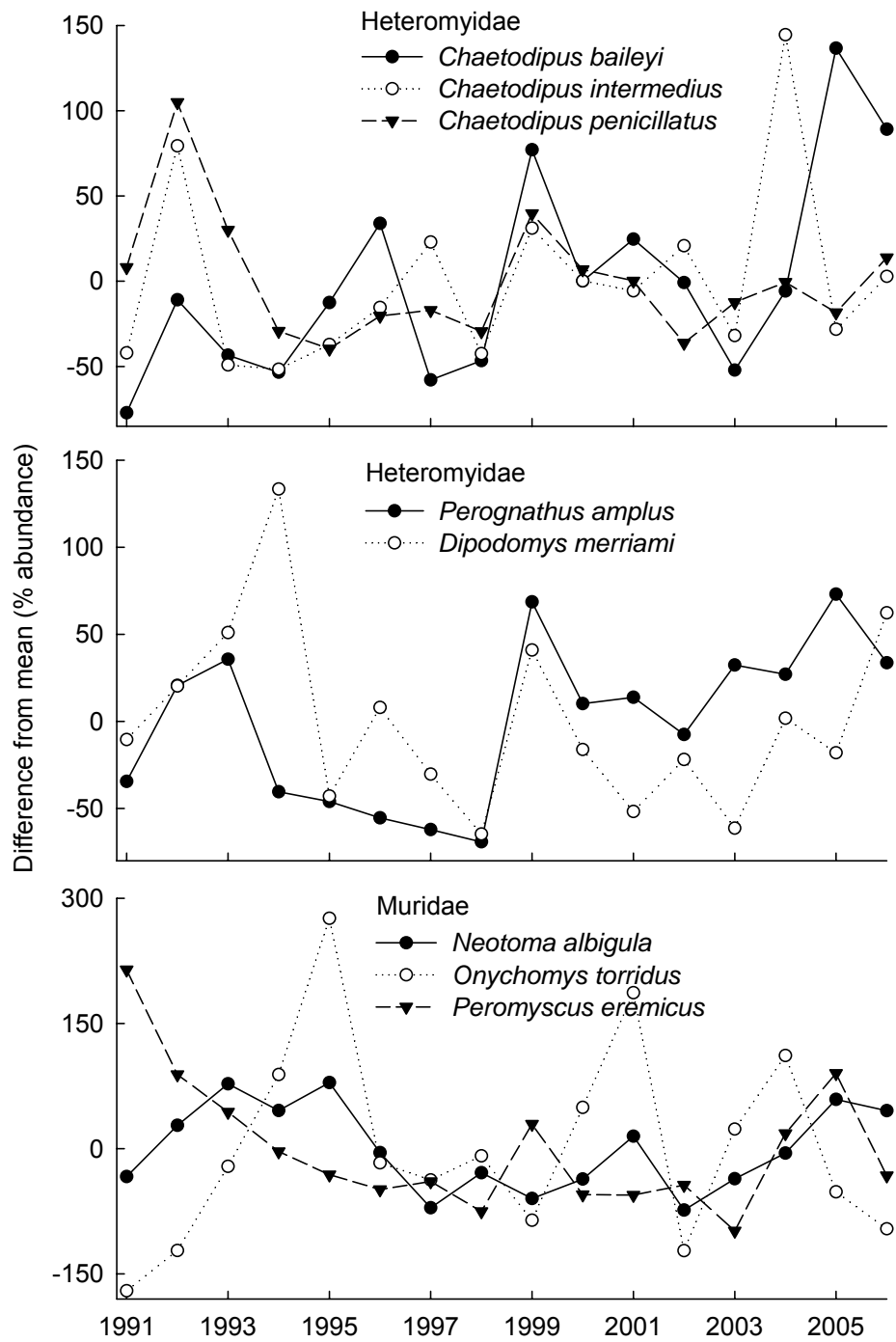


Figure 28: Changes in abundance (no./grid) across time for three groups of related species that exhibited somewhat similar temporal variation in Organ Pipe Cactus National Monument 1991-2006. Plotted values were standardized by dividing annual estimates for each species by the grand mean for each species.

Power Analyses—Power to detect trends in relative abundance across time averaged somewhat higher than that for abundance, yet overall, differences were much lower than that for lizards. Power to detect a 10% annual change in relative abundance of *C. penicillatus* after sampling each year for 25 years (power = 0.99) averaged just 10% higher than that for abundance given the same effect size and frequency and duration of sampling (compare Fig. 29 with 30). These patterns were attributable to somewhat lower levels of residual error around trend lines in relative abundance as compared to abundance (Table 19). Power to detect trends across time were highest for *C. penicillatus*, the most common species in ORPI, and only somewhat lower for *P. amplus* and *D. merriami* that were much less abundant. In comparison to these species, power to detect trends across time were lower for *N. albigula* and *C. baileyi* and much lower for *P. eremicus*, and *O. torridus* (Figs. 31-35). Despite these patterns, power to detect trends in population sizes of rodents were substantially lower than that for populations of most lizards (compare Figs. 10-14 with Figs. 29-35). This was likely because estimates of rodent abundance tended to oscillate more over time and occasionally exhibited very large fluctuations from year to year.

Even after sampling every year for the next 25 years, the smallest annual change in relative abundance that could be detected with power $\geq 80\%$ was approximately 7% for *C. penicillatus*, the species for which estimated power was highest. In comparison, sampling each year for the next 25 years was sufficient to detect only an approximate 12% annual change in relative abundance with power $\geq 80\%$ in populations of *N. albigula* (Fig. 32) and was not sufficient to detect even a 10% annual change in relative abundance (with power $\geq 80\%$) in populations of *C. baileyi*. For *O. torridus*, the rarest of the eight most abundant species of rodents in ORPI, sampling each year for the next 25 years was not sufficient to detect even a 20% annual change in relative abundance (Fig. 35).

Reducing survey effort from every year to every other year reduced power to detect trends after 25 years of sampling by approximately 20-30% (Figs. 29-35). Because overall, power was relatively low, this decrease substantially reduced prospects for detecting relevant population changes for all but the most abundant species. For example, after reducing survey effort by half over 25 years, a 10% annual change in relative abundance of *C. penicillatus* was still detectable with power of 80%, yet this same trend at the same level of power was not detectable for *D. merriami* or other species. Reducing effort from surveys every other year to every third year also reduced power yet at a lower rate than after reducing effort from surveys every year to every other year.

Table 19: Mean annual estimates of abundance (no./grid) and relative abundance (no./100 trap nights) of the eight most common species of nocturnal rodents and estimates of RMSE (root mean squared error) for linear trends in these parameters across time in Organ Pipe Cactus National Monument 1991-2006. RMSE quantifies unexplained variation around trends lines. Differences between ratios are proportional to the relative difference in estimated power for each parameter.

	Relative Abundance			Abundance			% difference in ratios
	Mean	RMSE of trend	Ratio mean:RMSE	Mean	RMSE of trend	Ratio mean:RMSE	
<i>Chaetodipus baileyi</i>	6.2	8.87	0.70	8.0	13.85	0.58	17.46
<i>Chaetodipus intermedius</i>	2.1	5.49	0.39	2.4	7.72	0.31	21.57
<i>Chaetodipus penicillatus</i>	23.0	18.86	1.22	26.0	23.64	1.10	9.95
<i>Perognathus amplus</i>	5.7	5.26	1.07	6.5	8.02	0.82	24.11
<i>Dipodomys merriami</i>	12.3	11.53	1.07	12.8	12.34	1.04	2.31
<i>Neotoma albigula</i>	4.1	5.63	0.73	4.0	5.36	0.74	-0.55
<i>Onychomys torridus</i>	0.28	0.74	0.37	0.23	0.68	0.34	9.49
<i>Peromyscus eremicus</i>	1.5	3.48	0.42	1.6	4.34	0.36	14.52

Abundance *Chaetodipus penicillatus* - All age classes

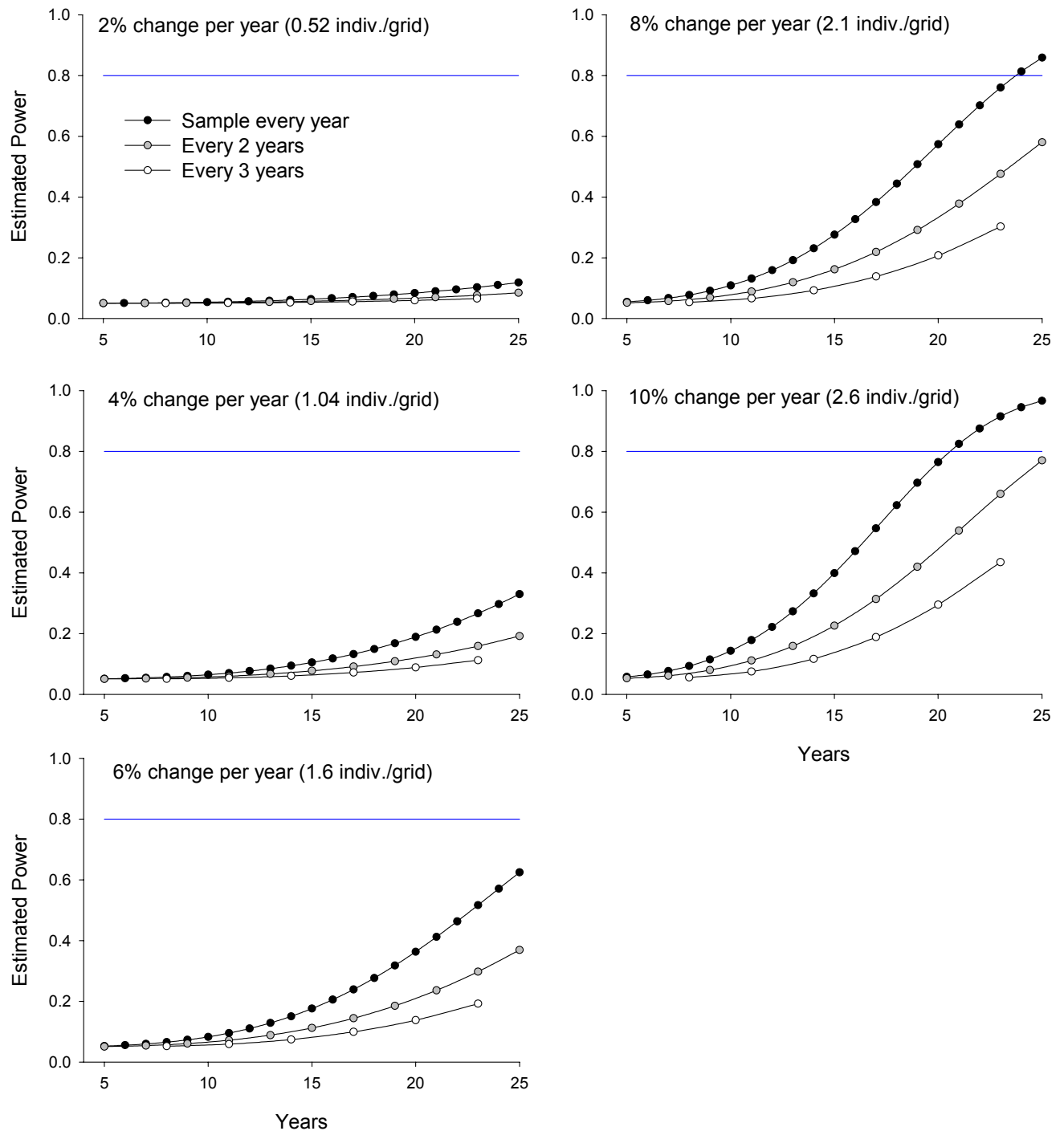


Figure 29: Power to detect a linear change of 2 to 10% per year in abundance of all age classes of *Chaetodipus penicillatus* combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Relative Abundance *Chaetodipus penicillatus* - All age classes

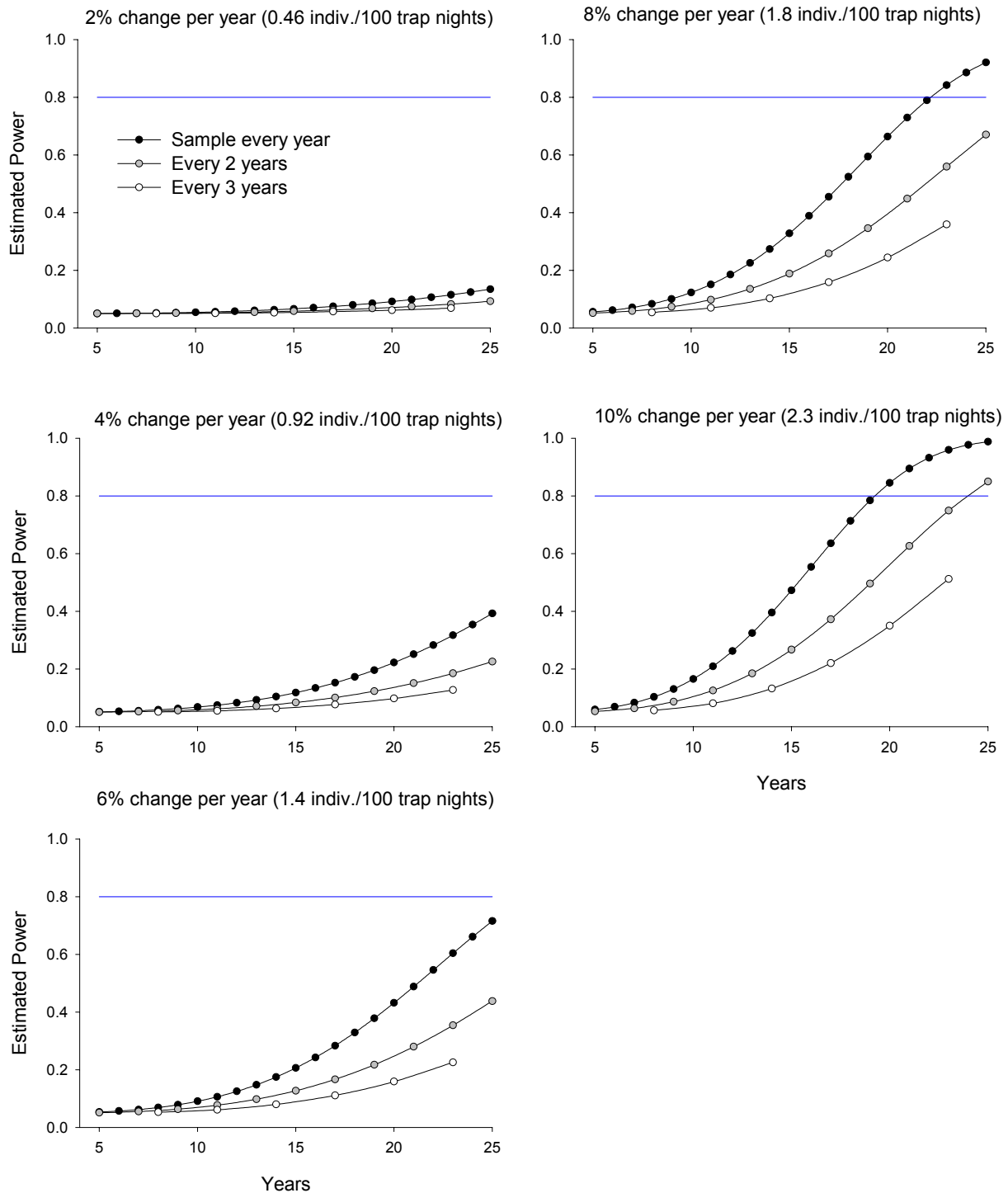


Figure 30: Power to detect a linear change of 2 to 10% per year in relative abundance of all age classes of *Chaetodipus pennicillatus* combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Relative Abundance *Dipodomys merriami* - All age classes

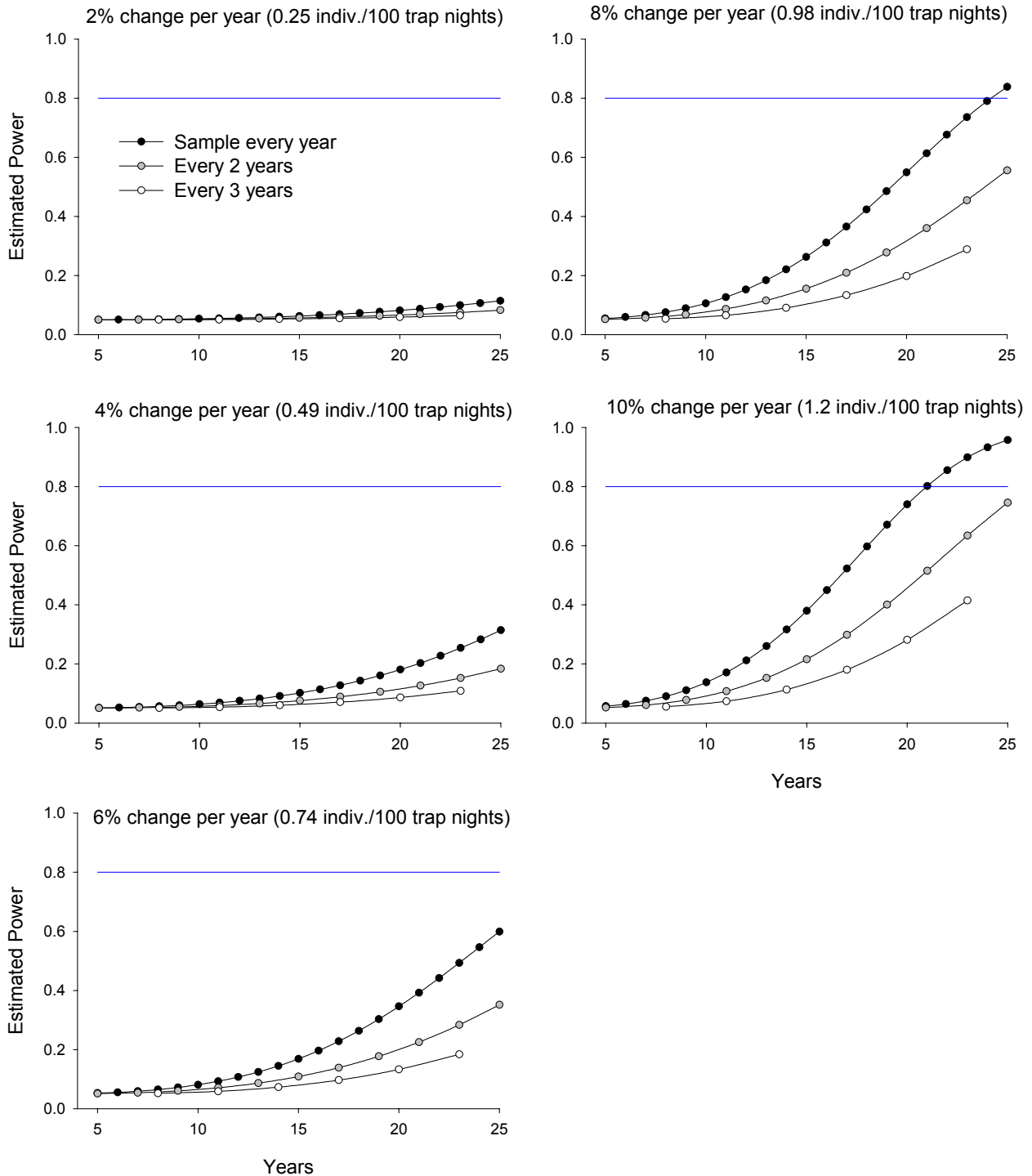


Figure 31: Power to detect a linear change of 2 to 10% per year in relative abundance of all age classes of *Dipodomys merriami* combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Relative Abundance *Neotoma albigula* - All age classes

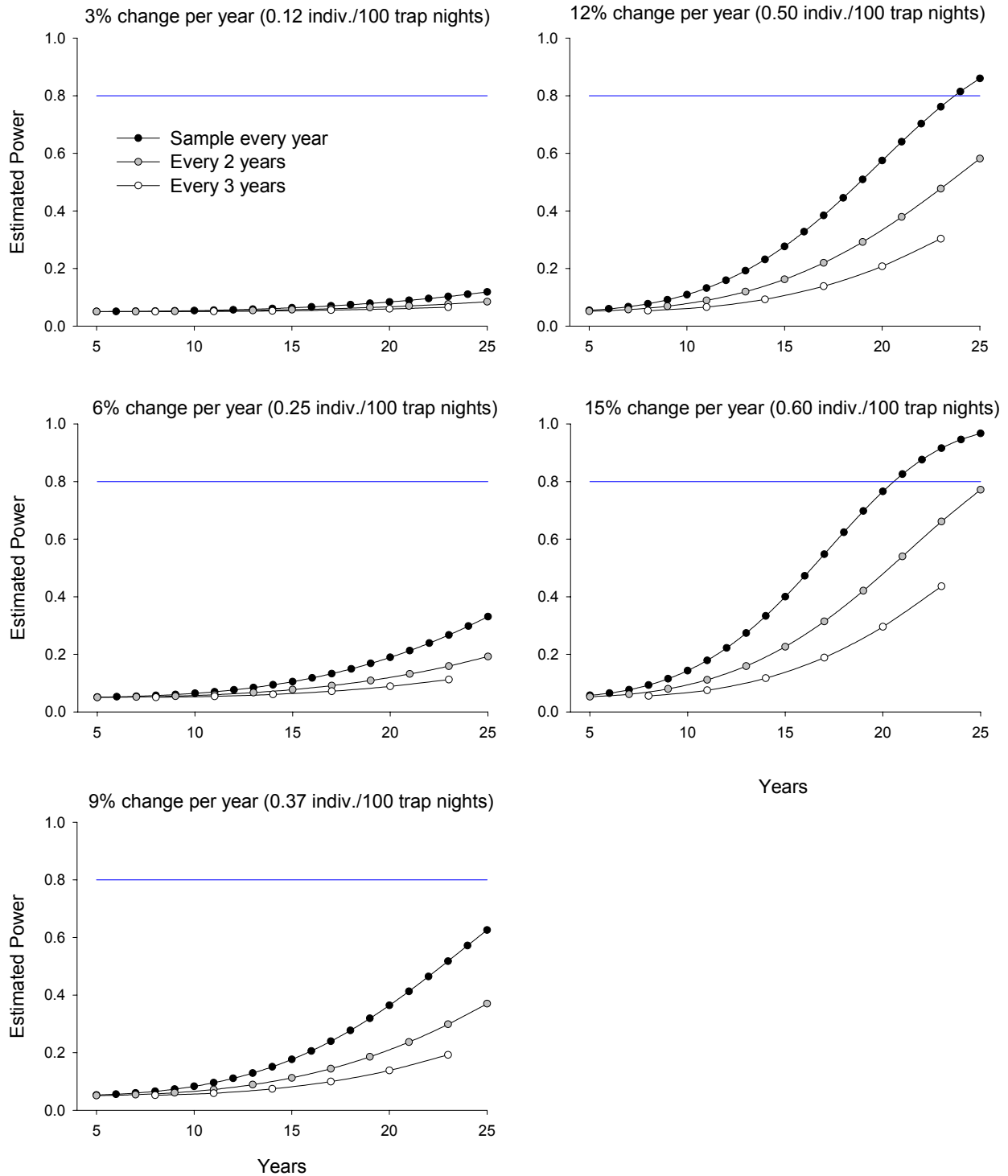


Figure 32: Power to detect a linear change of 3 to 15% per year in relative abundance of all age classes of *Neotoma albigula* combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Relative Abundance *Chaetodipus baileyi* - All age classes

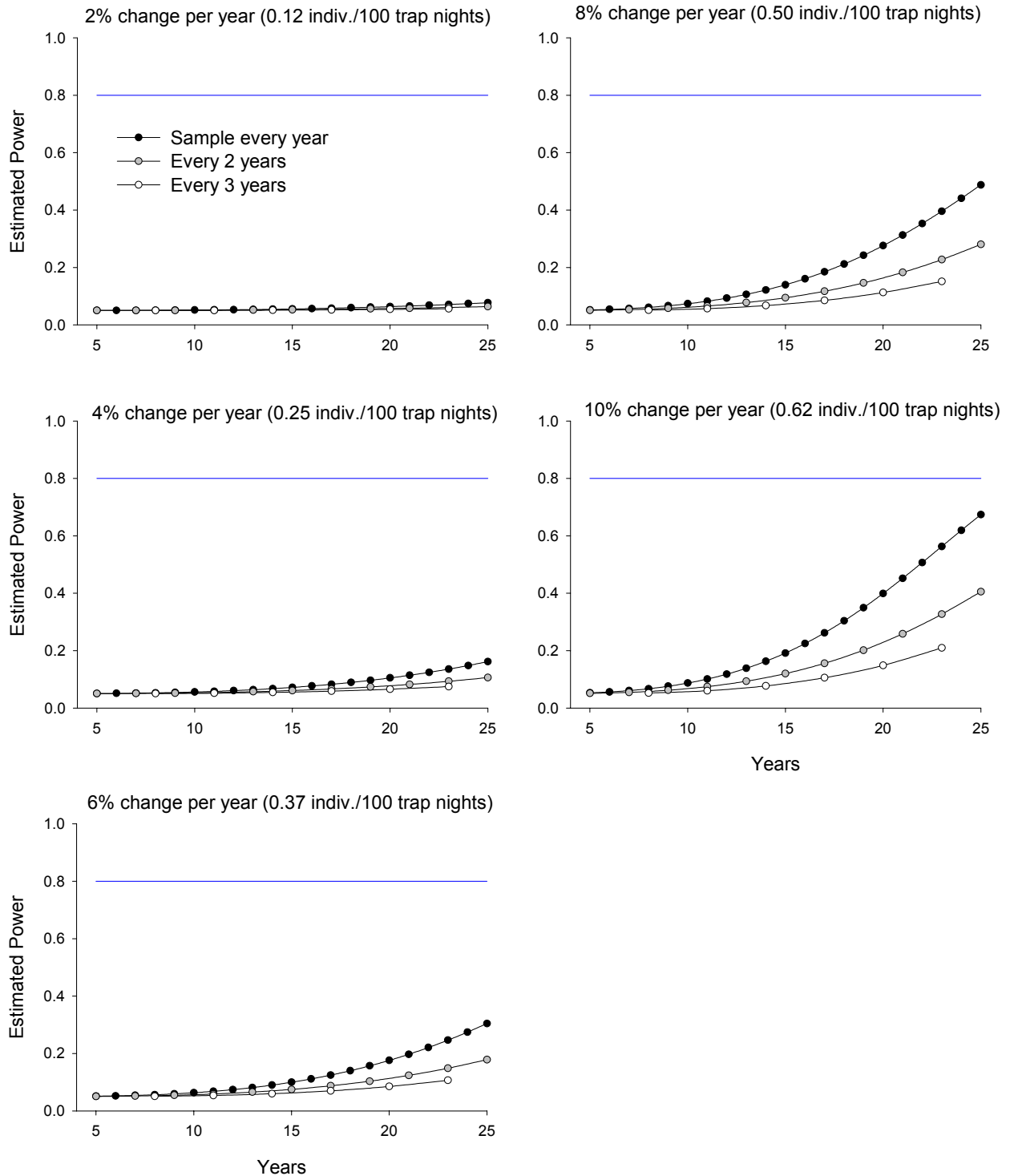


Figure 33: Power to detect a linear change of 2 to 10% per year in relative abundance of all age classes of *Chaetodipus baileyi* combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Relative Abundance *Peromyscus eremicus* - All age classes

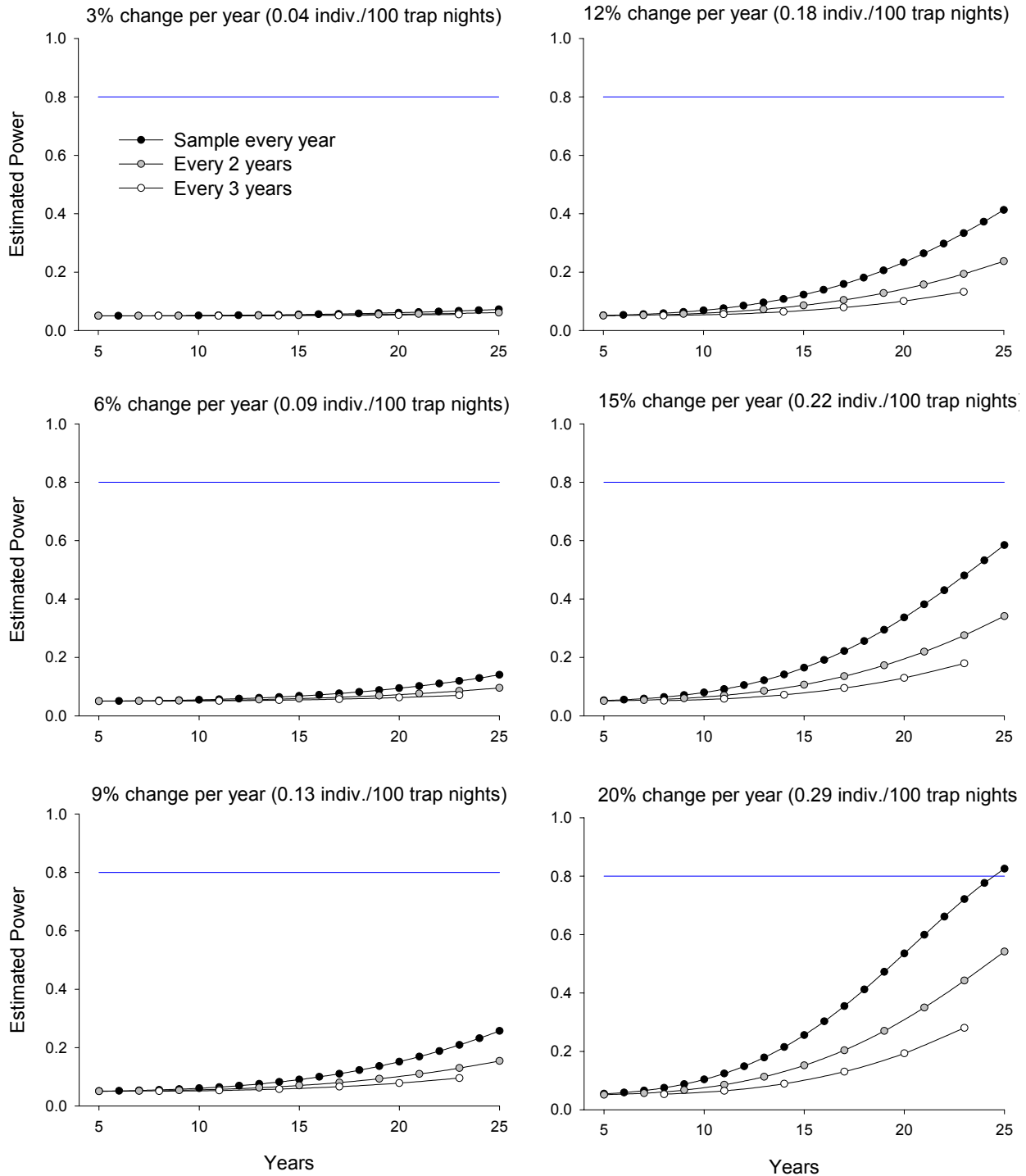


Figure 34: Power to detect a linear change of 3 to 20% per year in relative abundance of all age classes of *Peromyscus eremicus* combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Relative Abundance *Onychomys torridus* - All age classes

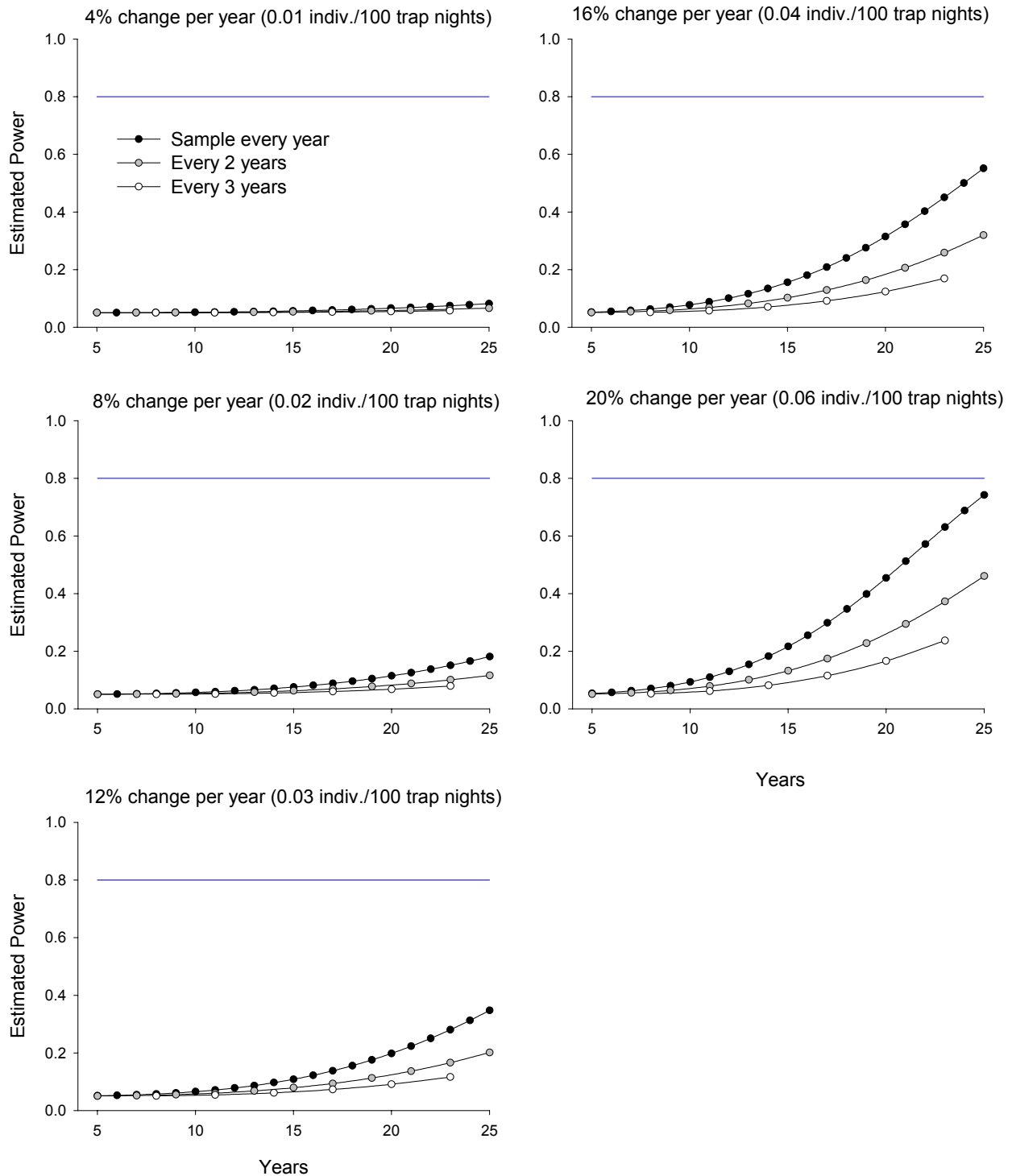


Figure 35: Power to detect a linear change of 4 to 20% per year in relative abundance of all age classes of *Onychomys torridus* combined based on surveys each year, every other year, and every third year for 25 years. Type-I error rate (α) was set at 0.05.

Table 20: Influence of moon phase on abundance (no./grid) and relative abundance (no./100 trap nights) of the eight most common species of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. The lunar cycle was divided into three periods based on the quantity of light that was visible during trapping: dark (-3 to +5 days from new moon), bright (± 4 days from full moon), and intermediate (all other times). *P*-values are based on a generalized linear mixed model.

Response Species	Moon Phase						F	<i>P</i>
	Bright		Intermediate		Dark			
	mean	SE	mean	SE	mean	SE		
Relative Abundance								
<i>Chaetodipus baileyi</i>	8.2	1.1	6.3	1.0	8.5	1.0	2.77	0.078
<i>Chaetodipus intermedius</i>	2.3	0.50	2.3	0.40	2.5	0.40	0.68	0.51
<i>Chaetodipus penicillatus</i>	25.1	1.8	20.9	1.7	24.6	1.7	2.32	0.12
<i>Perognathus amplus</i>	4.0	0.56	5.2	0.51	5.6	0.51	2.93	0.068
<i>Dipodomys merriami</i>	11.5	1.1	12.5	1.0	9.0	1.0	4.36	0.022
<i>Neotoma albigula</i>	3.8	0.63	4.0	0.59	4.9	0.59	1.62	0.21
<i>Onychomys torridus</i>	0.11	0.069	0.19	0.061	0.24	0.062	1.13	0.34
<i>Peromyscus eremicus</i>	1.4	0.40	1.7	0.37	1.7	0.37	0.26	0.77
Abundance								
<i>Chaetodipus baileyi</i>	8.9	1.5	7.4	1.4	10.6	1.4	1.74	0.19
<i>Chaetodipus intermedius</i>	3.2	0.7	3.84	0.62	2.9	0.63	0.56	0.56
<i>Chaetodipus penicillatus</i>	27.1	2.5	22.6	2.2	29.2	2.2	2.78	0.077
<i>Perognathus amplus</i>	4.6	0.9	6.4	0.80	6.4	0.80	1.86	0.17
<i>Dipodomys merriami</i>	12.7	1.2	13.4	1.1	9.7	1.1	3.88	0.031
<i>Neotoma albigula</i>	3.4	0.6	3.8	0.56	4.3	0.55	0.97	0.39
<i>Onychomys torridus</i>	0.10	0.063	0.17	0.056	0.16	0.057	0.40	0.67
<i>Peromyscus eremicus</i>	1.8	0.53	1.7	0.50	2.0	0.50	0.12	0.89

Lunar and Temporal Factors—Capture rates did not vary markedly with lunar or temporal factors for most species of rodents (Tables 20-21). With respect to lunar factors, capture rates varied markedly among bright, intermediate, and dark moon phases only for *D. merriami*, which seemed most trappable during intermediate and bright phases of the moon (Table 20). In contrast, trapability of *P. amplus* seemed higher during dark and intermediate phases of the moon than during bright phases. With respect to temporal factors, the number of individual *D. merriami* that were trapped per 100 trap nights increased by $1.5 \pm 0.4\%$ per day across the survey season and there was suggestive evidence of a similar trend for *C. baileyi* (Table 21; $P = 0.11$). Somewhat more *P. eremicus* were trapped early and late than during the middle portion of the survey season, yet there were few curvilinear trends in trapability across the survey season for other species (Table 20).

Influence of Environmental Factors on Abundance and Trends in Abundance—Relative abundance of virtually all species of rodents varied among topographic and soil types, but not among hydrologic types (Tables 22-23). Relative abundance of *C. intermedius* was much greater on rocky slopes than on bajadas and was rarely detected on the valley floor (Table 22). Similarly *C. baileyi* and the murids *N. albigula* and *P. eremicus* were also most abundant on rocky slopes, somewhat less abundant on bajadas, and occurred at low abundance on the valley floor ($P < 0.0001$). In contrast, *D. merriami* and *P. amplus* were most abundant on the valley floor and declined in abundance in formations that were higher up in the landscape, especially *D. merriami*, which did not occur on

rocky slopes. Although *C. penicillatus* and *O. torridus* also seemed to increase in abundance from low to high topographic formations, relative abundance varied more with soil type (Table 23). Relative abundance of *C. penicillatus* was >5 times higher in xeroriparian areas than in uplands and there was suggestive evidence of a similar pattern for *O. torridus* and a contrasting pattern for *P. amplus* (Table 23). Relative abundance did not vary among hydrologic types for any other species (Table 23).

Table 21: Influence of time-of-year on abundance (no./grid) and relative abundance (no./100 trap nights) of the eight most common species of nocturnal rodents in Organ Pipe Cactus National Monument 1991-2006. P-values are based on a generalized linear mixed model; parameter estimates are not shown for nonlinear effects.

Response	Day-of-year					
	Linear effect				Quadratic effect	
	Est.	SE	t	P	t	P
Species						
Relative Abundance						
<i>Chaetodipus baileyi</i>	0.069	0.042	1.62	0.11	1.14	0.26
<i>Chaetodipus intermedius</i>	0.010	0.022	0.44	0.66	0.53	0.60
<i>Chaetodipus penicillatus</i>	0.13	0.088	1.46	0.15	1.05	0.30
<i>Perognathus amplus</i>	0.00014	0.027	0.02	0.99	0.020	0.99
<i>Dipodomys merriami</i>	0.19	0.05	3.71	<0.001	3.67	0.0003
<i>Neotoma albigula</i>	0.024	0.026	0.94	0.35	0.92	0.36
<i>Onychomys torridus</i>	-0.0018	0.0034	0.27	0.61	0.26	0.61
<i>Peromyscus eremicus</i>	0.0093	0.017	0.53	0.59	1.85	0.066
Abundance						
<i>Chaetodipus baileyi</i>	0.15	0.071	2.08	0.039	0.26	0.79
<i>Chaetodipus intermedius</i>	0.032	0.035	0.93	0.35	1.69	0.092
<i>Chaetodipus penicillatus</i>	0.16	0.12	1.38	0.17	1.79	0.074
<i>Perognathus amplus</i>	-0.0012	0.041	0.03	0.98	0.26	0.80
<i>Dipodomys merriami</i>	0.19	0.057	3.35	<0.001	0.68	0.50
<i>Neotoma albigula</i>	0.0011	0.027	0.04	0.97	1.10	0.27
<i>Onychomys torridus</i>	-0.0017	0.0031	0.54	0.59	0.41	0.68
<i>Peromyscus eremicus</i>	0.019	0.02	0.92	0.36	1.00	0.32

Table 22: Relative abundance (no./100 trap nights) of eight species of nocturnal rodents among topographic formations in Organ Pipe Cactus National Monument 1991-2006. P-values are based on a generalized linear mixed model.

	Rocky Slope		Bajada		Valley Floor		P
	mean	SE	mean	SE	mean	SE	
<i>Chaetodipus baileyi</i>	16.3	2.7	13.0	3.5	0.58	1.9	<0.0001
<i>Chaetodipus intermedius</i>	7.7	1.1	1.3	1.4	0.08	0.74	<0.0001
<i>Chaetodipus penicillatus</i>	14.7	6.4	21.7	8.2	29.0	4.6	0.13
<i>Perognathus amplus</i>	1.0	1.7	5.7	2.2	7.2	1.2	0.0065
<i>Dipodomys merriami</i>	0.0	0.0	7.0	2.6	18.7	1.9	<0.0001
<i>Neotoma albigula</i>	9.1	1.5	4.9	1.9	1.4	1.0	<0.0001
<i>Onychomys torridus</i>	0.0	0.0	0.22	0.17	0.37	0.091	0.062
<i>Peromyscus eremicus</i>	4.4	0.82	1.2	1.1	0.21	0.57	<0.0001

Table 23: Relative abundance (no./100 trap nights) of eight species of nocturnal rodents between hydrologic and soil types in Organ Pipe Cactus National Monument 1991-2006. *P*-values are based on a generalized linear mixed model.

	Hydrologic regime							Soil texture				
	Xeroriparian		Mixed		Upland		<i>P</i>	Course		Fine		<i>P</i>
	mean	SE	mean	SE	mean	SE		mean	SE	mean	SE	
<i>Chaetodipus baileyi</i>	4.1	4.3	5.7	2.8	9.0	3.1	0.58	15.1	2.2	0.58	1.9	<0.0001
<i>Chaetodipus intermedius</i>	0.50	1.8	2.4	1.2	3.1	1.3	0.50	5.4	1.0	0.09	0.85	<0.0001
<i>Chaetodipus penicillatus</i>	44.4	5.8	27.8	4.1	8.7	4.5	<0.0001	17.2	5.2	29.0	4.6	0.057
<i>Perognathus amplus</i>	1.5	2.1	5.9	1.4	6.5	1.6	0.12	2.7	1.4	7.2	1.2	0.010
<i>Dipodomys merriami</i>	8.6	4.2	12.2	2.9	13.2	3.2	0.62	2.6	2.1	18.6	2.0	<0.0001
<i>Neotoma albigula</i>	4.8	2.2	3.6	1.4	3.9	1.6	0.90	7.5	1.3	1.4	1.1	<0.0001
<i>Onychomys torridus</i>	0.43	0.16	0.31	0.11	0.071	0.12	0.13	0.079	0.11	0.37	0.091	0.036
<i>Peromyscus eremicus</i>	1.9	1.2	0.80	0.8	1.4	0.9	0.72	3.2	0.74	0.21	0.61	0.0008

Both the magnitude and direction of temporal trends in relative abundance varied markedly across space (Table 24-25). Among hydrologic types for example, relative abundance of *C. penicillatus* declined precipitously in uplands, where overall abundance was low, whereas in xeroriparian and in areas with both upland and xeroriparian environments, relative abundance increased (Table 24, Fig. 36). In contrast, relative abundance of *N. albigula* and *P. eremicus*, which were similar among hydrologic classifications, declined over time in both xeroriparian and upland environments and increased in abundance in areas with both upland and xeroriparian environments. Temporal trends in relative abundance did not vary among hydrologic types for five other species that were considered ($t_{408} \leq 1.67$, $P \geq 0.19$; for time \times factor interactions).

Trends in relative abundance across time also varied markedly among topographic formations and there was evidence of time \times formation interactions for five of the eight species that were considered (Table 25). Relative abundance of *C. intermedius*, *C. baileyi*, and *P. amplus* increased at a greater rate on topographic formations on which they were most abundant (Table 22 and 25); for *C. intermedius* and *C. baileyi* abundance increased most on rocky slopes and declined in valley bottoms, whereas for *P. amplus* the opposite pattern occurred (Fig. 37). In contrast, relative abundance of *N. albigula* and *P. eremicus* declined on rocky slopes where they were most abundant overall and increased in valley bottoms where they were rarest (Table 22 and 25, Fig. 37). Temporal trends in relative abundance of *D. merriami* and *O. torridus* may not have varied with topography ($t_{408} \leq 0.88$, $P \geq 0.42$; for time \times factor interactions) because these species did not occur on rocky slopes.

Table 24: Variation in annual rates of change (\pm SE) in relative abundance (no./100 trap nights) of three species of nocturnal rodents among hydrologic types in Organ Pipe Cactus National Monument 1991-2006. *P*-values are for hydrologic type \times time interactions and are based on generalized linear mixed model. Five other common species for which factor \times interactions were not significant ($P \geq 0.19$) are not shown.

Species	Xeroriparian		Mixed		Upland		Factor \times Time	
	Est.	SE	Est.	SE	Est.	SE	F	<i>P</i>
<i>Chaetodipus penicillatus</i>	0.073	0.48	0.63	0.29	-0.81	0.18	6.24	0.0021
<i>Neotoma albigula</i>	-0.25	0.11	0.25	0.064	-0.18	0.10	5.60	0.0041
<i>Peromyscus eremicus</i>	-0.065	0.077	0.062	0.052	-0.034	0.052	5.01	0.0071

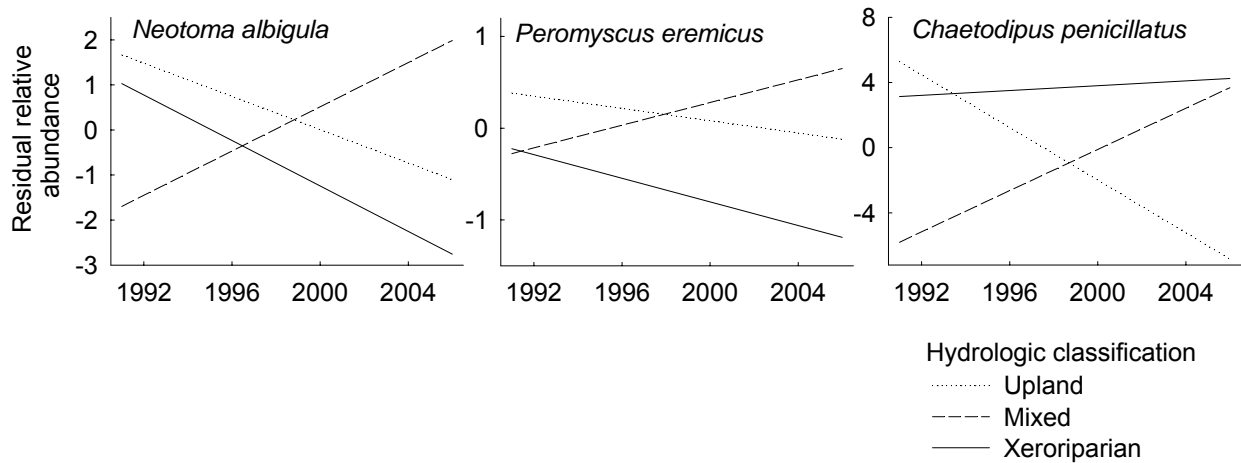


Figure 36: Variation in temporal trends in relative abundance (no./100 trap nights) of three species of nocturnal rodents among hydrologic types in Organ Pipe Cactus National Monument 1991-2006. Residual relative abundance represents variation remaining after site, year, and relevant covariates (e.g., moon phase) were considered.

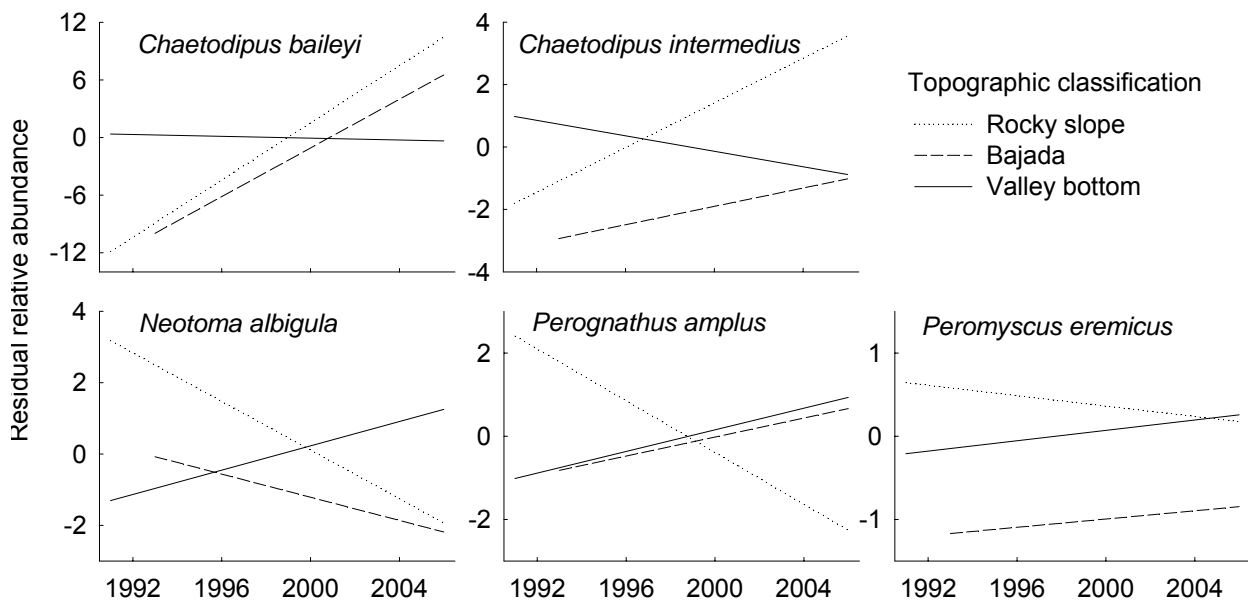


Figure 37: Variation in temporal trends in relative abundance (no./100 trap nights) of five species of nocturnal rodents among topographic formations in Organ Pipe Cactus National Monument 1991-2006. Residual relative abundance represents variation remaining after site, year, and relevant covariates (e.g., moon phase) were fit as nominal or continuous factors.

Table 25: Variation in annual rates of change (\pm SE) in relative abundance (no./100 trap nights) of five species of nocturnal rodents among topographic formations in Organ Pipe Cactus National Monument 1991-2006. *P*-values are for formation x time interactions and are based on generalized linear mixed models. Three other common species for which factor x interactions were not significant ($P \geq 0.42$) are not shown.

Species	Rocky Slope		Bajada		Valley Floor		Factor x Time	
	Est.	SE	Est.	SE	Est.	SE	F	<i>P</i>
<i>Chaetodipus baileyi</i>	1.5	0.26	1.3	0.43	-0.047	0.025	19.25	<0.0001
<i>Chaetodipus intermedius</i>	0.36	0.21	0.15	0.19	-0.12	0.0083	7.88	0.0004
<i>Perognathus amplus</i>	-0.31	0.039	0.11	0.15	0.13	0.083	3.40	0.034
<i>Neotoma albigula</i>	-0.34	0.16	-0.16	0.15	0.17	0.039	10.62	<0.0001
<i>Peromyscus eremicus</i>	-0.031	0.12	0.025	0.093	0.031	0.015	3.16	0.043

Influence of Rainfall on Abundance—Abundance of several species of rodents varied with quantity of rainfall during one or more seasons (Fig. 38), yet in contrast to patterns for lizards (Fig. 19), the effects of rainfall on populations of rodents were more variable and complex. Abundance of *D. merriami* increased sharply with increasing rainfall during the cool season at a lag time of one year ($P < 0.0001$) and there was a similar yet much weaker relationship between abundance of *C. baileyi* and rainfall during this same time period (Table 26; Fig. 38). Similarly, abundance of *N. albigula* also increased sharply with increasing cool-season rainfall yet it was for the period immediately prior to rodent sampling rather than at a lag time of one year. In contrast, abundance of *C. intermedius* declined somewhat as cool-season rainfall increased with no lag time ($P = 0.096$).

Table 26: Associations between nocturnal rodents and quantity of rainfall measured at a lag time of one year in Organ Pipe Cactus National Monument 1991-2006. Results are based on generalized linear mixed models for abundance (no./grid), relative abundance (no./100 trap nights), and species richness (no./site) in which site was considered the subject and as a random effect and rainfall was repeatedly measured each month at each site.

Response	Warm Season			Cool Season			Annual		
	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>	Est.	SE	<i>P</i>
Relative Abundance									
<i>Chaetodipus baileyi</i>	-0.020	0.20	0.92	0.47	0.15	0.0015	0.27	0.11	0.017
<i>Chaetodipus intermedius</i>	0.23	0.11	0.044	-0.076	0.076	0.31	0.018	0.064	0.77
<i>Chaetodipus penicillatus</i>	0.18	0.46	0.69	0.29	0.33	0.36	0.25	0.26	0.33
<i>Perognathus amplus</i>	0.19	0.13	0.15	0.057	0.095	0.54	0.099	0.076	0.19
<i>Dipodomys merriami</i>	0.70	0.27	0.011	1.34	0.19	<0.0001	1.06	0.15	<0.0001
<i>Neotoma albigula</i>	-0.054	0.12	0.66	0.097	0.089	0.27	0.039	0.069	0.57
<i>Peromyscus eremicus</i>	0.23	0.08	0.004	0.029	0.058	0.62	0.092	0.046	0.045
Abundance									
<i>Chaetodipus baileyi</i>	-0.28	0.36	0.43	0.48	0.25	0.059	0.21	0.20	0.29
<i>Chaetodipus intermedius</i>	0.17	0.17	0.31	-0.080	0.11	0.48	-0.002	0.096	0.98
<i>Chaetodipus penicillatus</i>	0.73	0.59	0.46	0.55	0.41	0.18	0.50	0.33	0.13
<i>Perognathus amplus</i>	0.11	0.20	0.57	0.033	0.14	0.82	0.058	0.12	0.62
<i>Dipodomys merriami</i>	0.76	0.30	0.012	1.56	0.20	<0.0001	1.23	0.16	<0.0001
<i>Neotoma albigula</i>	-0.19	0.13	0.14	0.098	0.091	0.28	-0.007	0.072	0.92
<i>Peromyscus eremicus</i>	0.30	0.091	0.0013	0.058	0.068	0.39	0.13	0.052	0.013
Species Richness									
Site-specific scale	-0.019	0.029	0.52	-0.018	0.02	0.37	-0.018	0.016	0.27

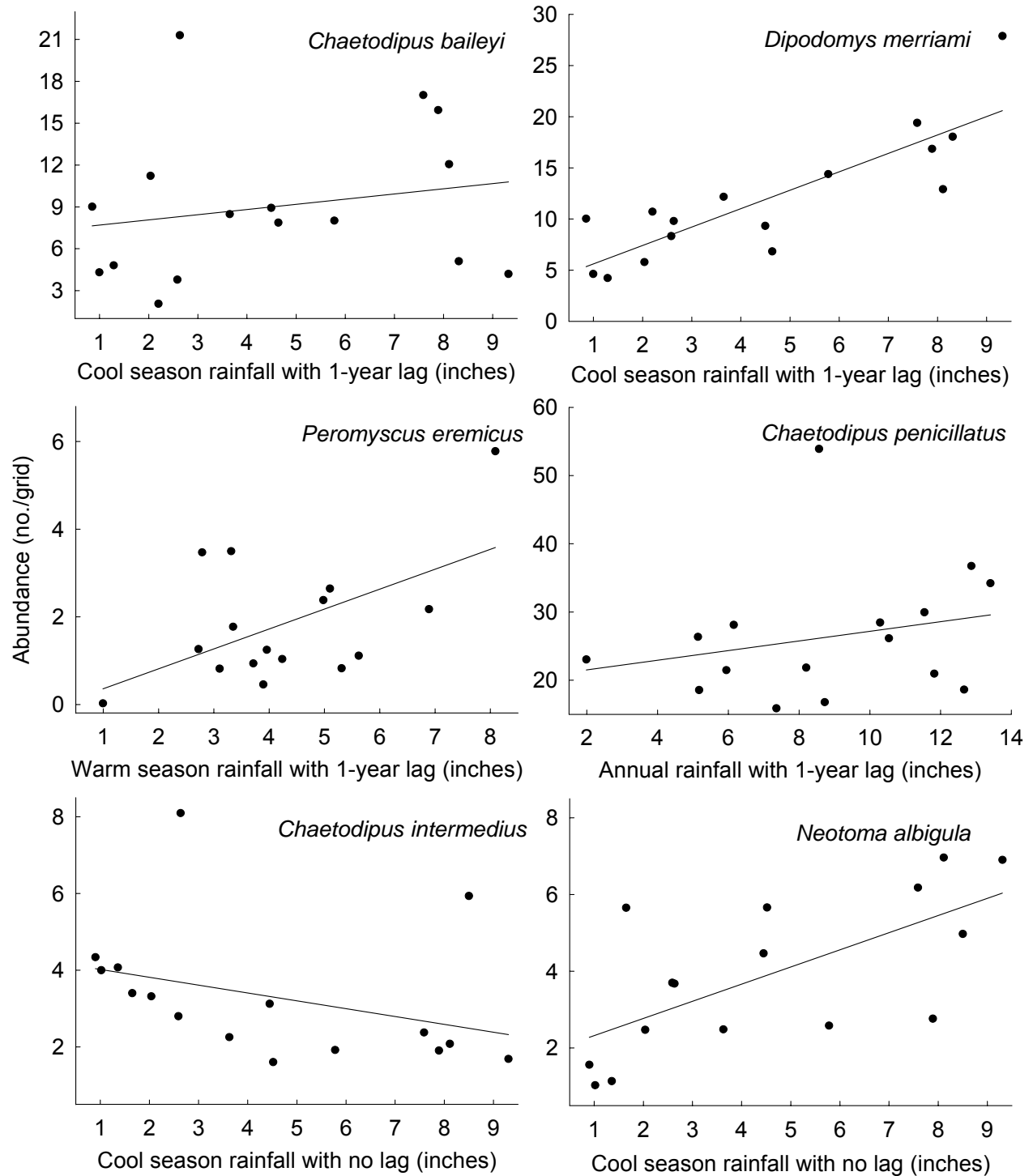


Figure 38: Relationship between abundance (no./grid) of six common species of nocturnal rodents and quantity of seasonal or annual rainfall in Organ Pipe Cactus National Monument 1991-2006. Rainfall was measured each month at each site across all years. Plotted values of rainfall are means for each year among sites at which rodents were sampled during the year. All means are adjusted for site effects.

Table 27: Associations between nocturnal rodents and quantity of rainfall measured with no lag time in Organ Pipe Cactus National Monument 1991-2006. Results are based on generalized linear mixed models for abundance (no./grid), relative abundance (no./100 trap nights), and species richness (no./site) in which site was considered the subject and as a random effect and rainfall was repeatedly measured each month at each site.

Response Species	Warm Season			Cool Season			Annual		
	Est.	SE	P	Est.	SE	P	Est.	SE	P
Relative Abundance									
<i>Chaetodipus baileyi</i>	-0.17	0.22	0.43	-0.16	0.14	0.27	-0.14	0.11	0.20
<i>Chaetodipus intermedius</i>	0.0017	0.12	0.99	-0.26	0.075	0.0007	-0.18	0.064	0.0047
<i>Chaetodipus penicillatus</i>	-0.017	0.50	0.97	0.030	0.32	0.93	0.015	0.26	0.95
<i>Perognathus amplus</i>	-0.11	0.14	0.45	-0.097	0.093	0.30	-0.09	0.074	0.23
<i>Dipodomys merriami</i>	-0.80	0.29	0.0063	-1.11	0.19	<0.0001	-0.86	0.15	<0.0001
<i>Neotoma albigula</i>	-0.14	0.13	0.29	0.33	0.084	<0.0001	0.16	0.066	0.013
<i>Peromyscus eremicus</i>	-0.17	0.087	0.051	0.073	0.056	0.20	-0.0005	0.045	0.99
Abundance									
<i>Chaetodipus baileyi</i>	-0.056	0.38	0.88	-0.14	0.25	0.57	-0.10	0.20	0.60
<i>Chaetodipus intermedius</i>	0.005	0.19	0.98	-0.10	0.11	0.096	-0.13	0.96	0.17
<i>Chaetodipus penicillatus</i>	0.50	0.64	0.44	-0.26	0.40	0.53	-0.038	0.33	0.91
<i>Perognathus amplus</i>	-0.0065	0.22	0.97	-0.037	0.14	0.79	-0.026	0.11	0.81
<i>Dipodomys merriami</i>	-0.42	0.32	0.19	-0.80	0.21	0.0001	-0.59	0.16	0.0004
<i>Neotoma albigula</i>	-0.079	0.14	0.56	0.37	0.086	<0.0001	0.21	0.069	0.0027
<i>Peromyscus eremicus</i>	-0.23	0.098	0.019	0.031	0.065	0.63	-0.044	0.050	0.38
Species Richness									
Site-specific scale	-0.0005	0.032	0.99	0.012	0.020	0.56	0.0075	0.016	0.64

Table 28: Associations between nocturnal rodents and quantity of rainfall measured at a lag time of two years in Organ Pipe Cactus National Monument 1991-2006. Results are based on generalized linear mixed models for abundance (no./grid), relative abundance (no./100 trap nights), and species richness (no./site) in which site was considered the subject and as a random effect and rainfall was repeatedly measured each month at each site.

Response Species	Warm Season			Cool Season			Annual		
	Est.	SE	P	Est.	SE	P	Est.	SE	P
Relative Abundance									
<i>Chaetodipus baileyi</i>	0.37	0.19	0.056	-0.68	0.15	<0.0001	-0.25	0.11	0.029
<i>Chaetodipus intermedius</i>	-0.23	0.11	0.038	-0.21	0.075	0.0065	-0.21	0.061	0.0006
<i>Chaetodipus penicillatus</i>	0.73	0.43	0.094	-0.80	0.33	0.016	-0.23	0.26	0.38
<i>Perognathus amplus</i>	0.018	0.13	0.89	-0.26	0.096	0.0062	-0.14	0.075	0.054
<i>Dipodomys merriami</i>	-0.22	0.26	0.40	-0.034	0.20	0.87	-0.098	0.16	0.53
<i>Neotoma albigula</i>	0.075	0.11	0.51	0.077	0.092	0.41	0.070	0.069	0.31
<i>Peromyscus eremicus</i>	0.0028	0.076	0.97	-0.039	0.060	0.52	-0.021	0.046	0.64
Abundance									
<i>Chaetodipus baileyi</i>	0.77	0.34	0.023	-0.82	0.25	0.0012	-0.23	0.20	0.25
<i>Chaetodipus intermedius</i>	-0.36	0.16	0.027	-0.22	0.11	0.052	-0.27	0.090	0.0041
<i>Chaetodipus penicillatus</i>	0.81	0.56	0.15	-0.87	0.42	0.036	-0.27	0.33	0.42
<i>Perognathus amplus</i>	0.22	0.19	0.26	-0.43	0.14	0.0026	-0.18	0.11	0.11
<i>Dipodomys merriami</i>	-0.30	0.29	0.30	-0.16	0.22	0.47	-0.16	0.17	0.34
<i>Neotoma albigula</i>	0.17	0.12	0.16	0.16	0.090	0.083	0.15	0.07	0.032
<i>Peromyscus eremicus</i>	-0.02	0.09	0.82	-0.04	0.070	0.53	-0.030	0.05	0.54
Species Richness									
Site-specific scale	0.0073	0.028	0.79	0.004	0.021	0.84	0.0051	0.016	0.76

and there was much stronger evidence of this relationship when relative abundance was considered the parameter ($P < 0.001$) due to two outlying observations (Fig. 38). *P. eremicus* was the only species of rodent for which variation in abundance was most strongly associated with rainfall during the warm season; abundance of *P. eremicus* increased as warm-season rainfall increased at a lag time of one year, a relationship that was strongly influenced by two outlying observations at both low and high levels of each factor (Fig. 38). In contrast, there was much weaker evidence that abundance of *P. eremicus* decreased with increasing warm-season rainfall with no lag time (Table 27). Abundance of *C. penicillatus* increased somewhat as annual rainfall increased with a lag time of one year and this relationship was not significantly influenced by an outlying observation at which abundance was high yet rainfall was only moderate (year 1992; Fig. 38). Associations between abundance of *P. amplus* and rainfall were weaker ($P \geq 0.15$) than those for any of the six other species considered. Importantly, both seasonal and annual rainfall declined in ORPI over the duration of the study (Fig. 39).

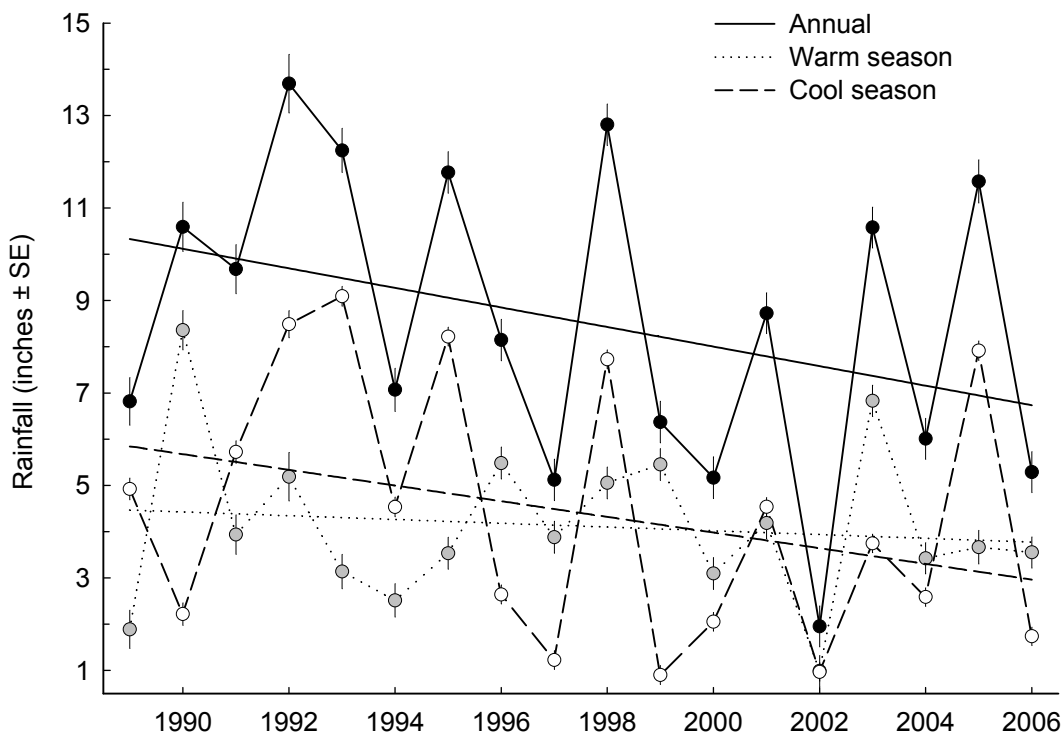


Figure 39: Variation in seasonal and annual rainfall across time in Organ Pipe Cactus National Monument 1989-2006. Rainfall was measured each month at each site across all years. Plotted values are means (\pm SE) for each year among sites at which vertebrates were sampled during that year. El Niño events occurred during the winters of 1992-93, 1999, and 2004. Extreme drought occurred in 2002. Straight lines are based on linear regression.

DISCUSSION

Data obtained by Organ Pipe Cactus National Monument (ORPI) while implementing the Ecological Monitoring Program (EMP) has provided detailed insights into the status and trends of populations and communities of rodents and lizards across time and space. Based on a detailed review, my overall finding is that current protocols and levels of sampling are adequate to detect changes in relevant population and community parameters over time. Further, in many cases, the existing program is capable of assessing the influence of environmental factors such as rainfall and soil type on the distribution and abundance of lizards and rodents across time and space. When coupled with additional data on biotic (e.g., predation, vegetation), abiotic (e.g., temperature), and other (e.g., disturbance) factors, and the recommendations that I provide, future assessments of EMP data are likely to be even more elucidative and useful.

As for virtually all monitoring programs, improvements to protocol efficiency, sampling design, and analyses are always possible. Improving the EMP, however, is not simply a function of fine tuning existing protocols or reallocating effort so that uncommon species or species with more localized distributions can be effectively monitored. Greater potential for improving the EMP likely depends on modifying the existing design to allow the flexibility needed to assess and monitor the influence of new and developing environmental stressors that were not present or foreseen when the program was first developed. For example, ORPI staff now suspects that influxes of human and vehicular traffic from neighboring Mexico combined with associated law-enforcement activities are having negative impacts on wildlife and habitat resources in and around the monument. Despite these expectations, detailed information on the effects of these stressors on vertebrate populations along the border is lacking, in part, because I failed to detect them with use of current protocols. To both assess and guide an effective management response to these potential stressors, ORPI staff must employ new design elements. Ideally, these elements should be incorporated into the existing program so that the value of data collected during past years can be optimized for future applications.

Another gathering threat to vertebrate populations in ORPI and the surrounding landscape is climate change. Given the detail and scale with which ORPI has monitored variation in weather parameters (ORPI 2006), and relationships that I observed between abundance of several species and rainfall, the existing design is capable of tracking the effects of rainfall on populations of common species over time and space. By including additional factors such as intraspecific interactions and predation into these models (e.g., Rosen 2000), the potential effects of climate change can be estimated across a range of future climate scenarios. Herein, I describe results of a detailed evaluation of EMP data and discuss recommendations for improving and modifying the program in the future.

Temporal Trends

Abundance of several populations of diurnal lizards (Figs. 2-5) and nocturnal rodents (Figs. 20-21) varied across time and these changes contributed to systematic trends in community composition in both taxa (Figs. 9 and 26). Despite trends in both population and community parameters, when estimates of species richness were adjusted for variation in detection probability, richness did not vary systematically across time (Figs. 6 and 25). Therefore, despite considerable changes in the status of some populations over time, all species that were present in ORPI when monitoring began are likely still present almost two decades later, with few possible exceptions. Species that may no longer occur or that may exhibit periodic colonization and extinction dynamics in ORPI, include those that were rarely detected (e.g., desert horned lizard) and those that have not been detected in ORPI since the EMP began (e.g., *Peromyscus merriami*, *Dipodomys deserti*).

Relative abundance of zebra-tail lizard and western whiptail declined while relative abundance of side-blotched lizard increased. Despite trends for these common species, I detected no systematic linear trends in abundance of less common or rare species of lizards. Annual estimates of relative abundance of zebra-tail lizard and western whiptail however, were highly correlated with estimates of some rare species (e.g., desert spiny lizard, red-backed whiptail, and desert iguana) and with tree lizard (Table 6). Such correlations suggest that declines in some populations of rare species may have gone undetected because precise estimates of abundance were unavailable and that declines in populations of tree lizard may become evident in the future. In contrast, there was no evidence of any systematic declines in abundance in rodent populations, and abundance of *Chaetodipus baileyi*, *Chaetodipus intermedius*, and *Perognathus amplus* all increased by as much as 10% per year across 16 years. Furthermore, had abundance not been particularly high during early years of the study, abundance of *Chaetodipus penicillatus* and perhaps *Peromyscus eremicus* may also have increased (Figs. 20-21).

Whereas systematic linear trends in abundance were apparent for some species, abundance tended to fluctuate widely across time for most species, especially for rodents. These patterns contributed to strong evidence of non-linear trends and often reduced power to detect linear trends. For most species, abundance tended to peak in the early 1990s, in the late 1990s, and again in the mid 2000s (2004 and 2005) (Figs. 2-4, 20-21). For many species of lizards however, these peaks in abundance had relatively similar magnitude in the early and mid 1990s whereas for rodents, peaks in the early 1990s tended to be much greater than those in the late 1990s. Furthermore, peaks in abundance during the mid 2000s were relatively much lower for lizards than for rodents and smaller species of rodents (*Chaetodipus* and *Perognathus*) typically reached much higher levels of abundance in 2004 and 2005 than larger species (Fig. 28). Interestingly, most of these trends seemed to correspond to the differential effects of rainfall on populations (see *Rainfall* below).

Although some temporal trends corresponded across taxa, trends were much more variable in populations of rodents. Among the three general peaks in abundance that occurred across time, abundance of small species of rodents (*Chaetodipus* and *Perognathus*) tended to increase first and over a much shorter duration whereas abundance of the largest species (*N. albigula*) increased more slowly, peaked later, and had lower rates of relative change (Fig. 28); patterns for the intermediate-sized *D. merriami* were roughly intermediate between small and large species of rodents. Interestingly, declines in abundance after these peaks were approximately equally rapid for most species in the early to mid 1990s, which Rosen (2000) attributed to predation. Since Rosen's comprehensive description of population and community dynamics of rodents between 1988 and 1998, these patterns have largely repeated themselves. Overall three cycles of abundance have occurred, although patterns were most pronounced in the early to mid 1990s. In contrast, peaks in abundance in the late 1990s were relatively lower especially for *C. baileyi* and *C. intermedius*, populations that may have been depressed due a simultaneous increase in abundance of the competitively dominant *D. merriami* with which abundances were negatively correlated.

When temporal trends in abundance of adult western whiptails were considered separately, there was less variation over time than when all age classes were considered together (Fig. 2). This pattern is likely consistent across many species because once adults are recruited into the population they likely have more stable dynamics than that of sub-adults and juveniles. Similar patterns were also suggested by variation in age structure across time (Figs. 7-8) and have implications for monitoring long-term changes in populations. Future reviews should explore age-specific estimates of population trends for additional species.

Several other population parameters in addition to abundance varied systematically across time and year-specific estimates of some of these parameters were associated with abundance. When changes in age structure and abundance of lizards were compared over time, patterns suggested the ecological processes that may have driven these trends. As abundance of western whiptail and,

to a lesser extent, zebra-tail lizard declined, the proportion of populations comprised of adults increased relative to sub-adults and juveniles (Figs. 7-8), a pattern that suggested poor reproductive performance contributed to declines (Rosen and Lowe 1996). Conversely, during periods when abundance increased, so did the proportion of juveniles and sub-adults suggesting that high reproductive performance contributed to increases. Interestingly, these patterns contrasted those for side-blotched lizard potentially due to differences in life-history characteristics (e.g., they are winter-spring not summer breeders) or to the influence of predation or competition (Rosen 2000). Side-blotched lizards are depredated by larger species of lizards such as western whiptails and by snakes that may take relatively more sub-adults and juveniles, especially during periods of population increase. Similar to patterns for side-blotched lizard, as abundance of *P. amplus* increased so did the proportion of the population comprised of adults. This pattern may also have been related to life-history characteristics because like side-blotched lizard, *P. amplus* typically breeds in late winter, which is earlier than many other species in ORPI (Hoffmeister 1986). For other species of rodents (except *N. albigula*; Fig. 22), there were no systematic temporal trends in age structure and annual estimates of age structure were not correlated with abundance. Specific mechanisms that drive population and community dynamics of lizards and rodents in ORPI are complex and may require experimental approaches to tease apart the confounding effects of competing mechanisms (Rosen 2000).

The proportion of individuals that were reproductive declined across time in populations of four species of rodents (Fig. 23) and these trends were not attributable to variation in age structure. Interestingly, two species for which the proportion of reproductively active individuals declined most (*P. amplus* and *C. baileyi*) increased in abundance across the same time period (Fig. 20). This suggests that these populations may have begun breeding earlier in the year and that fewer individuals were reproductively active when trapped or that fewer individuals bred due to higher levels of intraspecific competition that occurred at high population densities. A simultaneous increase in the proportion of adults in populations of *P. amplus* may not suggest either of these scenarios, whereas a simultaneous increase in mass of individuals (Fig. 24) suggests that intraspecific competition was not factor.

Composition of both rodent and lizard communities has changed over the past two decades in ORPI (Figs. 9 and 26). For lizards, these changes were characterized by an increasing proportion of side-blotched lizards in the community compared to virtually all other species except perhaps desert iguana and brush lizard. For rodents, these changes were characterized by an increasing proportion of *P. amplus*, *C. baileyi*, *D. merriami*, and *C. intermedius* in the community and a decreasing proportion of *C. penicillatus*, *O. torridus*, and *P. eremicus*. Further, when temporal trends of each rodent family were assessed individually, heteromyids increased in abundance across time whereas abundance of murids did not vary systematically (Fig. 27). These results suggest that physical or biotic components of the environment are shifting in directions that are better suited for some species and less suited for others. Interestingly, because these changes seem largely driven by rainfall, they may foreshadow future or developing responses of vertebrate communities to climate change (see *Rainfall* below). Despite considerable changes in abundance and community composition however, I found little variation in estimated species richness across time.

Parameter Estimation and Choice

Diurnal Lizards—I focused portions of this effort on obtaining accurate estimates of population parameters, applying these estimates to trend estimation, and assessing the most appropriate parameters, sampling procedures, and computation methods to monitor changes in populations and communities over time. To assess changes in populations, I considered both relative abundance and density as parameters. To assess changes in communities, I considered species richness as well as synthetic variables based on ordination techniques (PCA) that quantified changes in groups

of related populations simultaneously. To calculate abundance (density), I used distance-sampling procedures that corrected estimates for variation in detectability and that also adjusted for the effects of covariates that are relevant during the detection process (e.g., season, temperature, observers). To calculate relative abundance, I used a method developed by Rosen and Lowe (1995) that considered the maximum number of individuals detected among repeated surveys. Because each of these repeated measurements were made across gradients in time-of-day, season (spring and summer), and temperature, maximum values were assumed to index abundance when detectability was highest. Importantly, species-specific estimates of relative abundance were highly correlated with those for density; when sample sizes were sufficient to calculate estimates at site- and year-specific scales, these correlations were nearly perfect (e.g., $r = 0.86-0.97$) and remained high ($r = 0.76-0.85$) when only year-specific estimates of density were possible.

A important assumption of distance sampling is that all animals are detected with certainty on the transect line ($g(0) = 1$; Buckland et al. 2001). Although I explicitly adjusted for the influence of season and temperature on the width of detection functions, this critical assumption would have been violated if some varying proportion of a population was typically underground at a given time of day. Strong relationships between encounter rates and temperature, time-of-day, and season (Figs. 15-16, Tables 7-8) suggested that some species may not have been available for sampling at certain points along these gradients and that these patterns varied in a complex way both within and among species. In situations where animals are not detected with certainty on the transect line, adjustments can correct estimates (Buckland et al 2001), yet these adjustments would be complex if the proportion of animals that were available for sampling varied widely across a range of factors both within and among species. Given the effects of time and temperature on lizard activity, I suspect that much higher levels of variation around trend estimates in density resulted from pooling data across repeated surveys during which the proportion of animals that were available for sampling varied. If a sizable proportion of animals were underground at certain times, and this proportion varied among repeated surveys, even covariate adjustments would not correct estimates. In the future, distance data could be partitioned to satisfy needed assumptions, yet these strategies would need to vary widely among species, would require additional time and effort to complete, and necessitate validation and additional assumptions. Although using distance data at only those times and temperatures when above-ground activity peaked could improve estimates, it would reduce sample sizes by as much as 80% for some species and may therefore require additional survey effort even for common species. Given that assumptions of distance sampling were likely not met and that estimates of density were highly correlated with estimates of relative abundance, estimating relative abundance with use of Rosen and Lowe's (1995) method should be adequate for long-term monitoring.

As statistical methods for population estimation increase in complexity (Williams et al. 2002), it is important for analysts to explicitly consider tradeoffs between ease of interpretation, computation and sampling time required for estimation, data requirements, assumptions, and whether a parameter is sufficient to reach a desired objective. With respect to distance-sampling methods and abundance, knowledge of the proportion of animals that are available for sampling is needed to estimate absolute population size. In the case of monitoring, absolute estimates of abundance may not be required unless the actual number of individuals lost or gained within a population must be known as could be the case for populations are endangered or threatened species. Because the EMP does not require this level of detail, indices of abundance based on explicit and justifiable assumptions should be adequate for monitoring long-term population trends. In using indices to monitor population trends, however, it is important that the ratio of the index to actual population size does vary systemically across time because this variation will confound trend estimation. Because the method of Rosen and Lowe (1995) considers maximum values across time-of-day and between seasons, and surveys within narrow belt transects in which peak above-ground detectability is not likely to vary systematically across time, these indices seem adequate for monitoring. In future years, application of mark-recapture or radio-telemetry techniques to quantify the proportion of

populations that are available for sampling across the survey period, and more importantly, whether and how this proportion changes, can provide important information.

Nocturnal Rodents—As for lizards, I considered two metrics of rodent abundance based on different estimation techniques. I estimated relative abundance by calculating the number of individuals captured per unit trap effort and adjusted trap effort for sprung traps (captures and misfires) to reduce bias associated with trap saturation and misfires (Beauvais and Buskirk 1999). As a second method, I used the Lincoln-Petersen estimator, which considers the number of individual animals that are captured and the proportion of animals that are recaptured to compute detectability-corrected estimates of abundance. When comparing each technique, I found very high levels of correlation between site- and year-specific estimates of each parameter suggesting that they provided fairly equivalent metrics for monitoring. Further, the relative magnitude of trend estimates for each parameter for a given species were virtually identical suggesting that adjusting for detectability did not appreciably change my findings or conclusions. Abundance distributions however, tended to have longer tails and higher inter-annual variation, factors that were likely responsible for greater quantities of residual error around trends lines in abundance than in relative abundance. Trapping for >2 occasions should improve the precision of abundance estimates by providing more precise estimates of detectability, but the added effort may not be needed for general monitoring applications. Should additional resources be secured, estimating abundance with data from the four-occasion trap effort that occurred on 24 grids between 1997 and 1999 should proceed so that the efficacy of adding this effort can be evaluated.

Another option for potentially improving the precision of abundance estimates is also available and has the added benefit of not requiring any additional field effort. A. Litt (2007) devised a method to improve the precision of abundance estimates in situations where capture-recapture data are sparse. This method involves aggregating data into groups that have similar detection probabilities and using these groups during estimation. Furthermore, Rosen (2000) applied a similar method when estimating rodent abundance in OPRI that involved aggregating sampling grids by habitat type, which he found increased precision. Although evaluation of Litt's technique was based on data from a five-occasion trap design that included vastly more information on detectability, her aggregation technique could also improve precision for designs that employ only two trapping occasions.

Species Richness—There was little variation in species richness over time. In some cases when computing richness, the jackknife estimator performed poorly and resulted in large standard errors and poor Goodness-of-fit, which was likely due to the low number of lizard and rodent species that are present in ORPI. In the future, precision of these estimates could be improved by considering a wider range of candidate models (e.g., Boulinier et al. 1998). At the scale of individual sites, however, I found that changes in observed species richness across time were similar to changes in relative abundance. These patterns likely resulted because variation in species' detection probabilities is largely a function of abundance. Importantly, for the purpose of guiding management in ORPI, species richness may not be a sufficiently sensitive parameter to monitor. This is because declines in richness at park-wide scales indicate that one or more species have in fact gone extinct and that management and conservation actions in response to these changes may be too late. Given the ease with which richness can be estimated however, and high levels of statistical power for detecting small temporal trends in richness, use of this parameter seems valuable. Future efforts should consider using probabilistic techniques to estimate species richness at site-specific scales because changes in richness at these smaller scales may suggest future extinctions before they occur at larger scales. Because changes in species diversity are not as easily interpretable with respect to management implications, I did not estimate diversity as has been done in the past (ORPI 2006).

Survey Methodology

Diurnal Lizards—Methods for efficiently surveying and monitoring herpetofauna communities present a range of challenges to biologists and managers. Some of these challenges are due to the effects of temperature and temporal factors on activity patterns (Figs. 15-16, Tables 7-8) that result in variation in detectability during surveys. Importantly, the influence of temperature and temporal factors on encounter rates varied markedly among species indicating that detectability likely peaked at different points along each gradient for each species. Due to these patterns, surveys across a wide range of times and temperatures, as has been completed in ORPI in the past, is required to effectively index abundance and other parameters in lizard communities.

Although I found existing survey methods to be largely adequate, a variety of issues and recommendations should be considered. Encounter rates were too low to estimate and monitor abundance of several species of lizards that occur at low abundances or that have localized distributions in ORPI. As such, methods that augment detections of these species that are independent of effort may facilitate abundance estimation and monitoring in the future. Importantly, I found that detectability at 7.5 m from transect lines was exceptionally high for species with large body sizes such as spiny lizards. Further, I also found that a large proportion of encounters of other large-bodied species (e.g., desert iguana) that were too rare to estimate a detection function were at distances ≥ 7 m. These data strongly suggest that counting all individuals of large-bodied species regardless of distance from the transect line will augment sample sizes considerably. In contrast, detectability of small-bodied species tended to decline markedly at distances > 7 m suggesting that recording these species at greater distances is not efficient, especially if it distracts surveyors from observing animals near transect lines.

Relationships between encounter rates and temperature and time-of-day suggest that survey times overlapped periods of peak activity of most species. These data also suggest that surveys could begin at 6:30 AM and still overlap periods that are optimal for surveys (Fig. 15). In spring, surveys could perhaps begin even later (e.g., 7:00 AM). Extending surveys times later into the afternoon should augment encounters of desert iguana and perhaps zebra-tailed lizard, red-backed whiptail, and Clark's spiny lizard.

Nocturnal Rodents—In contrast to patterns for lizards, few factors appreciably influenced encounter rates of rodents. These results were especially notable for moon phase which is known to influence activity levels in some populations of rodents (Price et al. 1984, Travers et al. 1988), yet had little apparent effect on most populations in ORPI (Table 20). Similarly, although temporal factors have also been cited as influencing activity levels of rodents during summer, day-of-year had a strong effect for only one species of rodent (*D. merriami*) and the direction of this effect based on 16 years of study varied from that which had been observed previously in the Sonoran Desert based on 9 different years of study (Petryszyn 1982). Given that few potential covariates of rodent activity had appreciable effects on the number of animals that were trapped, modifying the timing of sampling does not seem necessary. Where needed however, these covariates can be considered during parameter estimation to improve the reliability of trend estimates.

Statistical Power and Implications for Future Efforts

Statistical power that is sufficient to detect relevant changes in resource conditions is an essential component of a monitoring program. This is because without sufficient power, a great deal of time and effort may be expended with little or no opportunity to ever detect a trend. As such, understanding trade-offs between effort, power, sampling frequency and duration, and effect size (e.g., the magnitude of change in population size) will allow managers to select the most appropriate balance among these factors to meet a desired objective across a range of resources levels that are

available for monitoring. Overall, my findings indicate low to moderately-high levels of statistical power to monitor long-term population trends of lizards and rodents in ORPI (Figs. 10-14 and 29-35, Table 19). For relatively common species, however, my findings largely validate the capability of the EMP to monitor population trends. When comparing between taxa, there were much higher levels of power to detect trends in lizard populations than for rodents at equal sampling duration and effect size (compare Figs. 10-14 with 29-35). Further, although species-specific estimates of power to detect trends in relative abundance of rodents averaged only slightly higher than that for abundance, power to detect trends in relative abundance of lizards were much higher than those for density. These patterns resulted from greater quantities of residual error around trend lines in density than around trend lines in relative abundance. For rodents, power was likely lower overall because 1) annual estimates of abundance tended to vary more across time, 2) estimates occasionally exhibited very large changes in abundance from year to year, and 3) this variation typically did not contribute to trends across time. Power to detect trends in lizard density may have been much lower than for relative abundance because important assumptions of distance sampling were not met (see *Parameter Estimation and Choice* above). Interestingly, power to detect trends in abundance was higher for adults than for all age classes combined (Fig. 10). This is likely because adults have more stable population dynamics than that for other age classes, which reduces variation around trend lines and contributes to more deterministic changes in population size.

Although precise recommendations for optimizing future EMP efforts depend on the level of resources that are available to implement monitoring, results of power analyses can enable these evaluations as data on resource levels become available. With respect to reducing effort from surveys every year to every other year, this seems like an efficient alternative for lizards because overall power was relatively high and because surveys every other year are still capable of detecting an approximate 5% annual change in relative abundance of the four most common species of lizards over the next 20 years (with power of 80%). With respect to rodents however, reducing effort to every other year would greatly reduce the potential of detecting trends over time. For example, the smallest change in populations that could be detected with surveys every year over 25 years (and power equal to 80%) was a 7% annual change in relative abundance of *C. penicillatus*; when effort was reduced to surveys every other year, only a 10% annual change could be detected after 24 years (Fig. 29) and power to detect changes in populations of other species were also lower. Because rodents have very dynamic population cycles in ORPI, sampling every year will aid evaluations of the causes and implications of these trends.

Influence of Environmental Factors

Soil, Topographic, and Hydrologic Factors—Topography, soil texture, and hydrologic conditions had strong effects on abundance of lizard and rodent populations in ORPI, patterns that were not unexpected given known patterns of natural history and ecology (Lowe 1964, Hoffmeister 1986, Stebbins 2003). For lizards, the direction and magnitude of temporal trends in abundance varied with environmental factors for only one species (tree lizard; Fig. 18), whereas for rodents, trends varied widely across topographic or hydrological features for six species (Figs. 36-37). Notably, for three species of rodents, abundance did not vary systematically across time until the effect of time was partitioned across space. For three other species of rodents that increased in abundance across time, abundance was either stable or declined across the same period on one topographic formation. For tree lizard, trends in abundance were negative on bajadas, stable or perhaps increasing on valley bottoms, and did not vary on rocky slopes where they were much less common (Table 9). Because soil moisture and tree sizes often decline as one travels up the bajada, quality of vegetative resources for tree lizards may also decline along this gradient, especially given drought conditions that have prevailed throughout ORPI for much of the past decade. In contrast to suggestions that valley bottoms act as refugia for western whiptails during times of drought (Rosen and Lowe 1996), there was no evidence that populations declined at different rates among

topographic formations. Trends in abundance of western whiptail and other species may in fact be more stable in certain vegetation rather than topographic formations because areas with dense vegetation, such as mesquite and other xeroriparian woodlands, likely provide more food resources and cover, especially during times of drought. For rodents, all four species of heteromyids that exhibited varying population trends across space increased in abundance on the same topographic or hydrological features on which they were most common, and typically declined in areas where they were least common. These results suggest that variation in habitat quality may have driven these trends. In contrast, for both species of murids that had varying trends in abundance across space, abundance increased on the valley floor where overall abundance averaged lower and either decreased or remained stable elsewhere.

Rainfall—Precipitation had marked effects on abundance of both lizard and rodent populations (Figs. 19 and 38) suggesting that it is an important driver of population and community dynamics in ORPI. Interestingly however, populations of lizards and rodents responded differently to rainfall and responses were more uniform for lizards than for rodents. Abundance of lizard species typically varied most with rainfall during the warm season one year prior to surveys and there was much less evidence that cool-season rainfall was influential (Tables 11-13, Fig. 18). Importantly however, not all populations of lizards increased with summer rainfall. Abundance of side-blotched lizard declined with increasing annual rainfall at a lag time of one year and increased somewhat with annual and cool-season rainfall with no lag time, with similar patterns for desert iguana. These relationships are important for several reasons. First, large increases in winter rains that occur during years when the El Niño Southern Oscillation is active may not influence population dynamics for most species of lizards independent of summer rain. These patterns were first noted by Rosen (2000) who described population declines of lizards in the mid 1990s despite high levels of winter rainfall that were produced by a prolonged El Niño between 1992 and 1993. Following the mid 1990s however, above-average summer rains contributed to steady increases in lizard abundance until summer rains diminished during a three-year period after 1999; a short but wet summer in 2003 also influenced abundance the following year but these effects were short-lived for most species. Virtually all species of lizards (except side-blotched lizard and desert iguana) responded to a severe 100-year drought in 2002 with marked declines in abundance in 2003. Importantly, two of the three species that were most influenced by rainfall (western whiptail and zebra-tailed lizard) tended to decline in abundance across time.

Effects of rainfall on populations of rodents were more complex and varied somewhat from those that had been described in the past. For heteromyids, cool-season rainfall was most influential at either a lag time of one year or with no lag time. For murids, patterns were more variable and abundance of *P. eremicus* increased with warm-season rainfall with a lag time of one year whereas abundance of *N. albigula* increased with cool-season rainfall that fell immediately prior to trapping. Rainfall affected abundance of *D. merriami* more than any other species and the effect was strongest with cool-season rainfall at a lag time of one year and much weaker for warm-season rainfall, which is contrary to the findings of Rosen (2000). Cool-season rainfall also had very strong effects on abundance of *N. albigula*, and annual rainfall had a somewhat weaker effect, which is also contrary to the findings of Rosen (2000). *P. amplus* was the only species that exhibited no apparent response to rainfall and rainfall had relatively weak effects on relative abundance of *C. baileyi* and *C. intermedius*. Notably, these three species of heteromyids that largely did not respond to rainfall were the only species of rodents that increased in abundance over time in ORPI. Given that cool-season rainfall was typically much more influential to rodents, declines in winter rains that are associated with La Niña could have a major impact on population dynamics of rodents independent of summer rain, especially should winter rains continue to decline (Fig. 39).

Associations between vertebrate abundance and rainfall (Figs. 19 and 38) together with changes in seasonal and annual rainfall over time (Fig. 39) largely explained population cycles of lizards and rodents in ORPI. Peaks in lizard abundance in the early and mid 1990s had relatively similar

magnitude yet were sustained over a longer time period in the late 1990s. These patterns corresponded to high levels of summer rains during these same periods and monsoon rains that were above average during 3 of 4 years in the late 1990s. For rodents however, peaks in abundance during the early 1990s tended to be much greater than those in the late 1990s because of a prolonged El Niño between 1992 and 1993 and a relatively shorter El Niño in 1998-99 that was followed by a period of very lower winter rainfall. Peaks in abundance during the mid 2000s may have been much lower for lizards than for rodents because a relatively wet monsoon in 2003 followed the 100-year drought of 2002 and winter rains peaked during an El Niño in 2005 before which rodents had two years of relatively moderate winter rains to recover. Despite these general trends, estimating the effects of predation and interspecific competition on abundance may be necessary to fully understand how rainfall affects population and community dynamics of small vertebrates (Rosen 2000). Models that include these factors can be used to estimate the effects of rainfall on communities across a range of potential climate scenarios.

Disturbance—Disturbance associated with increases in human activity near the international border could be a significant stressor to biological resources in ORPI. Despite expectations that disturbance is negatively affecting populations of vertebrates, documentation of these effects were largely lacking, in part because I observed few differences between sites along the border and in the interior and because temporal trends in relative abundance did vary between border and interior sites (Table 14). Interestingly however, side-blotched lizard and western whiptail were more abundant at sites along the border even after adjusting for relevant covariates, which suggests disturbance may have positive effects for these species. Inferences regarding disturbance however, based on the current design are somewhat inconclusive because 1) group membership alone (e.g., border vs. interior) may not be closely associated with differences in disturbance, 2) other factors that were not considered (e.g., vegetation) are likely relevant, and 3) abundance may be a misleading indicator of disturbance. This later possibility could apply if abundance is not associated with habitat quality, demographic performance, or if significant lag times are present. Other methods for evaluating the influence of disturbance in ORPI are discussed below.

Effort – Past, Present, and Future

Determining an appropriate level of sampling effort is an important consideration for monitoring and often represents a tradeoff between budget constraints, inference, and precision. These considerations often involve whether to allocate more effort across space or at the same locations across time. For monitoring, it is often recommended that effort be replicated across space especially when variation among sites is high and sufficiently precise parameter estimates can be obtained without intensive repeated sampling at each site. Monitoring lizard populations in ORPI, however, requires a relatively moderate level of repeated sampling at the same sites because detectability varies markedly across time and temperature, constraints that will reduce resources for replicating effort across space. Although rodents were sampled on only two consecutive nights at each site during each year, additional season-specific estimates of abundance and reproductive status may be needed, especially should activity levels and breeding periods vary systematically across time.

In ORPI, sampling effort varied widely among years in terms of the number of sites that were visited yet effort was relatively consistent among sites that were selected for sampling during a given year. To adjust annual park-wide estimates for site effects that resulted from this unbalanced design, I considered site as the subject of analyses and as a covariate to adjust annual park-wide estimates. Although this technique greatly reduced degrees of freedom, it likely reduces bias of annual estimates that are due to the addition or subtraction of sites where some species tend to be more or less abundant on average. Further, this method also allows effort to be added or subtracted during future years without biasing estimates.

Current levels of effort are more than adequate to monitor the four most abundant species of lizards in ORPI, species that accounted for 94% of encounters over all 18 years. However, even if individuals of large-bodied species are counted at unlimited distances and at maximum levels of past effort (e.g., 19 sites per year), encounters may be too few to generate high-precision estimates of abundance that are required for reliable trend estimation. Although power to detect trends for many of these uncommon species was relatively high, this finding may be somewhat misleading given the methods used to estimate power. Because encounter rates for uncommon species were low, precision of these estimates were likely also low as indicated by wider standard errors relative to common species (compare Figs. 3 and 4). Error associated with these estimates during each year is not captured by RMSE around trend lines, and therefore estimates of power for uncommon species are biased high. To reliably monitor abundance of uncommon species, additional effort will be required to increase the precision of estimates. Allocation of this effort may also need to be more spatially focused for species such as red-backed whiptail and Clark's spiny lizard that have more localized distributions. For these species, sample sizes that are required to reach a desired level of power can be modeled for each species using estimates of initial abundance and sampling variation gathered over the course of this effort. For rodents, adding effort to increase the precision of abundance estimates may not significantly improve power to detect temporal trends over time. This is because inter-annual variation in abundance in many populations of rodents is much higher, especially for species with relatively small-body size for which abundance tended to oscillate widely over time.

Depending on whether anticipated support for this program is increasing or decreasing, I recommend two general strategies for optimizing the EMP. Should resources allow additional effort, I recommend identifying uncommon species that have relatively high priorities for monitoring and conservation, identifying locations in ORPI where they occur, and then placing additional transects at random at these sites. Because some species occupy only small portions of ORPI (e.g., brush lizard and red-backed whiptail), more focused site selection is needed to efficiently increase encounter rates of these species. In contrast, for species that occur at low densities and are widely distributed (e.g., longnosed leopard lizard) surveys that are more spatially focused may not yield higher encounter rates and therefore more random sampling may be a reasonable yet less efficient alternative.

If budgets require reducing effort, a number of potential scenarios are possible. First, encounter rates are high enough for the most abundant species of lizards and rodents to reduce effort somewhat and still monitor changes in populations over time at similar effect sizes to those I considered during power analyses. Reducing effort for lizards could involve sampling fewer sites each year or eliminating surveys during either spring or summer. Comparisons between seasons indicated that encounter rates were generally higher in summer, a pattern that was attributable mainly to large influxes of juveniles and sub-adults that had not yet been recruited into the breeding population. Further, power for detecting trends in adults only was higher than for that of all age classes combined. Given that losses of breeding adults over time has more serious long-term consequences for population persistence, reducing effort during summer surveys and maintaining or expanding spring surveys over as wide an area as possible may be advantageous. To the contrary, knowledge of younger age classes is valuable for projecting future abundances and providing insights into reproductive performance during a given year. Another viable option could be to reduce surveys to every other or every third year, which would reduce power but still allow detection of somewhat small annual changes in abundance of some lizard populations (e.g., >3%) but much less so for rodent populations. Although this strategy would facilitate long-term monitoring, it would also reduce information for assessing factors that drive population and community dynamics over time. Another option that could reduce effort and bolster potential to monitor less common species would be to reduce effort during one season and reallocate it to sampling more sites during the other season. Effort during summer seems needed because adult side-blotched lizards, an extremely important species to monitor in ORPI, were encountered twice as frequently in summer.

Future Directions

In addition to potential modifications discussed above, ORPI staff should consider adding design elements to better facilitate assessing the influence of disturbance. This can be achieved either by incorporating new design elements into the existing program or through a separate research effort. If disturbance is in fact affecting populations of small vertebrates in ORPI, these effects are likely a function of soil compaction, changes in the structure or composition of vegetation, or perhaps changes in animal activity due to the direct actions of humans. Within the context of the existing program, vegetation and soil attributes can be easily measured each season along transects at existing sites and compared to baseline conditions over time. Further, additional sites can be added across a gradient of actual or anticipated disturbance and other relevant environmental attributes such as topography, vegetation, or other factors that influence parameters could be measured. The most reliable means of detecting and understanding the effects of disturbance on lizard and rodent populations in ORPI is through experimental approaches. This strategy however, necessitates that experimental treatments mimic the actual type, intensity, and scale of disturbance.

Sampling a full suite of environmental factors along transects in conjunction with vertebrate surveys will be valuable for understanding the drivers of population and community dynamics in ORPI, especially should unforeseen environmental stressors arise. Although monitoring typically focuses on simply estimating changes in populations over time, generating effective management responses to these changes may depend on knowledge of factors that are driving trends. As such, monitoring programs need to be sufficiently flexible to address stressors that were not anticipated when monitoring was first initiated. To this end, a mechanistic understanding of how and why monitoring parameters interact and vary across a broad suite of inter-related environmental parameters is the most valuable context in which to monitor.

CONCLUSIONS

My assessment of EMP data indicate that, 1) protocols are adequate to estimate population and community parameters of several species of rodents and lizards, 2) current levels of sampling are adequate to detect trends in these parameters over time for common species, 3) abundance and distribution of these species varies across ORPI with topographic, hydrologic, and soil conditions, 4) estimating relative abundance of lizards is likely more efficient than estimating density with use of distance sampling methods, 5) abundance varied widely across time for most species yet declined systematically for only two species or lizards while increasing for one species of lizard and three species of rodents, 6) composition of both rodent and lizard communities have changed systematically over time, 7) despite these changes there was little evidence that new species have either colonized or become extinct in ORPI over the last 18 years, 8) abundance of lizards typically varied with quantity of warm-season rainfall at a lag time of one year with one very notable exception, 9) abundance of rodents also typically varied with rainfall yet the effect varied widely between seasons and lag times, 10) declines in rainfall are likely driving changes in populations and communities of small vertebrates over time, 11) a wide range of options exist for improving and adapting the EMP while retaining the basic foundation of the program, and 12) selection of the best options for modifying and improving the EMP depend on resources that are available to implement the program in future years and on specific management questions and interests. Further, I also outlined methods, strategies, and options for analyzing and presenting data obtained by the EMP.

The EMP has important applications both within and well beyond ORPI. Within ORPI, my findings suggest that changes in populations and communities over time may be related to declines in rainfall and that monitoring these trends is important to understand the effects of climate change on vertebrates. Given the long temporal frame, broad range of environments, and the number of

parameters and taxa that are considered, application of EMP data to other park units should aid them in developing monitoring plans. In the Sonora Desert, trend data from ORPI can also help elucidate the drivers of population and community changes at larger scales. Recent evidence for example, indicates that populations of cactus ferruginous pygmy-owls (*Glaucidium brasilianum cactorum*), which depend on diurnal lizards for food, have declined by an estimated 36% between 2000 and 2008 in adjacent northern Sonora, Mexico (Flesch 2008). When trends in abundance of pygmy-owls were compared to those for lizards in ORPI at a lag time of one year, and with variation in annual rainfall at a lag time of two years, the direction, magnitude, and percent annual change of trends in owl abundance were nearly identical to that for lizards and rainfall (Flesch 2008). At larger international and global scales, long-term datasets such as that obtained by ORPI are very rare, yet extremely valuable for monitoring global change at a range of scales. To this end, data from the EMP could be used with other datasets in a larger meta-analysis to assess the level of coherence in trends among regions (e.g., Parmesan and Yohe 2003). The value of a well-designed monitoring program increases exponentially with the amount of time it has existed. Because the EMP may be the longest-standing vertebrate monitoring program anywhere in the National Park System (A. Hubbard, Sonoran Desert Network, pers. comm.), continuation of this program in at least some capacity is important.

The EMP is a good example of a monitoring program in a dynamic time. Scientists in both the physical (Seager et al. 2007, Seidel et al. 2007) and biological (Parmesan and Yohe 2003, Root et al. 2003, Parmesan 2006) sciences are now documenting the direct and indirect effects of climate change on ecosystems nearly worldwide. Should these changes influence the structure and composition of plant and animal communities in ORPI and in the surrounding region, this program will be fundamental to both detecting and responding to these changes. ORPI is situated near an important transition zone where the western edge of the Arizona Upland subdivision of the Sonoran Desert abuts the lower and more arid Lower Colorado River Valley subdivision. Therefore, the region surrounding ORPI is likely at or near the upper or lower tolerance limits for many species of plants and animals. Given this location, biological resources in ORPI may experience more pronounced and rapid changes to warming or to variation in the quantity or periodicity of rainfall. Trends that I observed in ORPI over the last decade are consistent with some of the predictions of climate change and suggest these changes may already be occurring. Future monitoring is necessary to document the full nature and consequences of these trends. Given the foundation, history, and data already acquired by the EMP, ORPI and their cooperators are in a unique position to document and predict the influence of climate change and other environmental stressors on wildlife, to formulate a response to these changes, and to provide an example to other entities with similar goals.

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Appendix A: SAS programming code used to evaluate relationships between statistical power, sample effort, effect size, and the duration of sampling required to detect linear trends in monitoring parameters across time. Input values denote the parameter and species of interest, maximum number of years considered, the observed mean of the parameter among years, the root mean squared error around the trend line, and the Type-I error rate (α). Programming code was provided by Dr. Robert J. Steidl.

```

/* Investigates power for prescribed change per year in a parameter.
   ObsMean and RMSE from regressions based on observed values.
   Slope range is calculated as a % of mean value (e.g., 1-5%) */

%let Param = ln(abund); /* Parameter being investigated */
%let Species = whiptail; /* Species of interest */
%let MaxYears = 25; /* Max number of years to consider */
%let ObsMean = 48.5; /* Mean value - ignored for RELATIVE changes based
on log (counts) */
%let RMSE = 0.352; /* Estimated Root MSE from appropriate regression */
%let alpha = 0.05; /* Alpha level */

Data LRPower;

   * Choice of ABSOLUTE or RELATIVE changes made with one of the two following
lines ;
   * Asterisk out the one you're not interested in ;

   /* ABSOLUTE */

   * l_slope = &ObsMean*0.01; h_slope = &ObsMean*0.05; i_slope =
&ObsMean*0.01;

   /* RELATIVE */

   l_slope = 0.01; h_slope = 0.05; i_slope = 0.01;

l_int=1; h_int=3; i_int=1;
l_yrs=5; h_yrs=&MaxYears; i_yrs=1;

array Yr {&MaxYears} Yr1-Yr&MaxYears; /* sampling years */

Do slope = l_slope to h_slope by i_slope;
  Do int = l_int to h_int by i_int;
    Do yrs = l_yrs to h_yrs by int;

      T = (CEIL(yrs/int));

      Do j = 1 to T;
        if j=1 then yr(j) = 1;
        else yr(j) = yr(j-1) + int;
      end;

      SS_t = CSS(of yr1-yr&MaxYears);
      df = T -2;

      * Calculate Noncentrality Parameter and t critical values;

```

```

NonCen = slope / (&RMSE/sqrt(SS_t));
tCrit_up = tInv(1-&alpha/2,df);
tCrit_lo = -tCrit_up;

* Calculate Power and its confidence interval;

Power = 1 - Probt(tCrit_up,df,ABS(NonCen)) +
Probt(tCrit_lo,df,ABS(NonCen));
output;
Do j = 1 to &MaxYears;
    yr(j) = .;
end;
end;
end;
end;

drop l_slope h_slope i_slope l_int h_int i_int l_yrs h_yrs i_yrs
j yr1-yr&MaxYears;

Label yrs = "Years"
int = "Years Between surveys"
slope = "Hypothesized Annual Change in &Param";

run;

Options nocenter nodate nonumber;

Proc sort; by slope int yrs;

Proc Print NoObs;
Var Slope T yrs int SS_t Noncen Power ;
Title "Linear Trends for &Param for &Species ";
run;

proc gplot;
symbol interpol=join value=dot;
plot power*yrs=int/haxis=0 to &MaxYears by 5 vaxis=0 to 1 by .1
vref=0.80 lvref=3 hminor=0 vminor=0;
by slope;
run;

quit;

```