

Population status and trends of Ferruginous Pygmy-owls in northern Sonora Mexico: A 22 year perspective

Final Report

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ABSTRACT

Ferruginous Pygmy-Owls (*Glaucidium brasilianum*) are of major conservation concern in southern Arizona due to marked declines in distribution and abundance over the past century and threats driven by climate and landcover change. Populations of pygmy-owls in adjacent northern Mexico are important for recovery in the U.S. and the focus of extensive long-term monitoring and research efforts by University of Arizona staff that began in 2000. To assess the current status and trends of populations, I estimated abundance and territory occupancy across a vast region of northern Sonora, Mexico immediately south of Arizona in 2021, and evaluated spatiotemporal trends across as many as 22 years with data gathered between 2000 and 2021. Although evidence of systematic population declines between 2000 and 2014 was strong, large increases in abundance and occupancy in 2015-2016 eliminated statistical evidence of declining linear trends across this time period. In 2021, however, I found that high levels of abundance and occupancy in years 2015-2016 were not sustained, and documented marked contractions in distribution and abundance and estimates of these parameters that were low or very low compared to historical values for these populations. Annual estimates of territory occupancy in 2021 from a top-ranked generalized linear mixed effects model, for example, were the lowest observed (0.435) among all 16 annual estimates across the study, and 22% lower than the long-term average (0.560). Trend estimates for occupancy between 2001 and 2021 also provided suggestive evidence of a systematic linear decline across the broader study area but results depended on model assumptions. Abundance also declined markedly from levels last observed in 2015-2016, and was 6% lower than the long-term average. Such results indicate complex population dynamics and the importance of consistently monitoring populations of concern across time and space so that short-term changes in populations can be distinguished from long-term declines. In 2021, I also observed a broad range of new landscape disturbances linked to changes in land use and landcover within or immediately around many territory patches. These changes occurred within the last ~5 years, and impacted 22% of 93 patches I surveyed in 2021, with 29% of new disturbances at sites that had no prior impacts. Such disturbances included vegetation clearing linked to agriculture, grazing, and woodcutting that can eliminate or degrade pygmy-owl habitat and landscape connectivity. Historically low estimates of occupancy from 2021 together with observed impacts to habitat and anticipated increases in aridity across this region suggest high potential for future declines of pygmy-owls in this portion of the Sonoran Desert. Given the value and potential insights offered by long-term datasets, additional monitoring of historical sites that are part of this program and assessment of factors that are associated with population dynamics should be priorities for future efforts.

INTRODUCTION

The Ferruginous Pygmy-Owl (*Glaucidium brasilianum*; hereafter “pygmy-owl”) is an iconic predator of the Sonoran Desert that is threatened by habitat loss, fragmentation, and degradation linked to climate and landcover change (USFWS 2011, Flesch 2014, Flesch et al. 2015). In the late 1800s and early 1900s, pygmy-owls were described as locally common in mesic riparian areas in the Sonoran Desert of southern Arizona (Bendire 1888, Fisher 1893, Breninger 1898, Gilman 1909, Bent 1938), and also occurred in woodlands along dry washes and adjacent desert scrub (Brandt 1951, Phillips et al. 1964). By the mid-1900s, vegetation clearing for agriculture, water diversion, and other changes in land use and landcover in wooded bottomlands drove

widespread habitat loss, and associated declines in distribution and abundance (Johnson et al. 2003). As a result, the Arizona population of pygmy-owls was listed as endangered by the U.S. Fish and Wildlife Service (USFWS) in 1997, but then delisted for reasons unrelated to recovery in 2006 (USFWS 2011). Following over a decade of litigation, populations of pygmy-owls are now being considered for re-listing by USFWS and an ongoing species status assessment is underway. This report includes new information on the status of pygmy-owls in the northern Sonoran Desert that is relevant to this assessment.

Following listing and subsequent de-listing in Arizona, past monitoring suggests populations of pygmy-owls continued to decline, but few quantitative estimates of population trends or factors that influence them are available due to a lack of consistent and standardized monitoring. Recent information from Arizona indicates that population units in two of the three easternmost watershed regions in which pygmy-owls have recently occurred, in the southern Altar Valley and near Tucson, have declined to extirpation, whereas populations in the northern Altar and Avra valleys were stable or increasing through 2016 (Flesch et al. 2017, Flesch *in revision*). To my knowledge, more recent efforts from 2020 and 2021 indicate no recent evidence of occupancy near Tucson or in the southern Altar Valley (Arizona Game and Fish Department *pers. comm.*), suggesting populations in both of these watershed regions remain extirpated. Moreover, populations to the west in Organ Pipe Cactus National Monument seem to have also declined recently given there were no observations of pygmy-owls between 2016 and 2020 and just two non-breeding individuals observed in spring 2021 despite intensive survey effort by myself and by National Park Service biologists that surveyed virtually all historical sites across the monument (Flesch *unpublished data*, Flesch *in prep.*, N.P.S. *pers. comm.*). Finally, recent monitoring on Pima Country Conservation Lands in the northern Altar and adjacent Avra valleys suggest a largely stable population on these lands, but more effort is needed to assess these patterns given inferences are based on data from just two years (e.g., 2017, 2020; Flesch 2021). Extensive pygmy-owls surveys by the Arizona Game and Fish Department were funded by USFWS in years 2020 and 2021 in Arizona, and may provide quantitative inferences on the status and trends of populations. Regardless, more field effort and especially analyses of past survey data is required to better understand the recent status and trends of pygmy-owls in southern Arizona.

In adjacent northern Sonora, Mexico, pygmy-owls are more common, occupy similar environments, and systematic monitoring efforts that began in 2000 provide strong evidence that drought and extreme temperatures are associated with marked declines in abundance (Flesch 2003, Flesch and Steidl 2006, Flesch 2014). Between 2000 and 2011, for example, abundance in four randomly-selected watershed regions in northern Sonora declined by an estimated 19-27%, with 75% of temporal variation in abundance explained by precipitation and temperature (Flesch 2014). Additionally, abundance was lower and varied more across time in areas with higher land-use intensity, but abundance was higher and less variable in areas that supported more potential nest cavities and riparian vegetation, suggesting these factors are important drivers of population dynamics. Such patterns have alarming implications for population persistence and recovery, and hence, data on current status and threats are critical for conservation and management. Auspiciously, more recent monitoring in 2015 and 2016 in northern Sonora indicated substantial increases in abundance and occupancy in these years that largely eliminated statistical evidence of past declines (Flesch et al. 2017). Nonetheless, information on whether these levels of

abundance persist today was unavailable, but is addressed in detail in this report. Moreover, pygmy-owls surveys by the Arizona Game and Fish Department were funded by USFWS in year 2021 in Sonora, but the results of this work, how it was designed and implemented, and whether it was linked to any baseline data, remain unknown, but could provide additional insights on the status and trends of populations in the future.

With support from Tucson Audubon Society and Defenders of Wildlife in 2020, I extended monitoring efforts for pygmy-owls in northern Sonora across one additional year so that data now span a period of 22 years. Although planned field efforts in spring 2020 in Mexico were cancelled due to the global pandemic, results of surveys in spring 2021 are summarized here. Complex access and limited funding precluded surveys of all historical territories in northern Sonora, but I surveyed all past abundance transects and a large sample of territories in April and May 2021. Limited funding also precluded assessments of how recent changes in weather, habitat, and landscape disturbance influenced population attributes, but I noted the presence and type of any new changes in land use and landcover observed at and around each historical territory patch to help evaluate current threats to habitat. Although monitoring data for pygmy-owls had not been gathered since 2016 in northern Sonora, field work in 2021 was timely given an ongoing status assessment of populations and threats across the northern portion of the pygmy-owl's range. Here I summarize information on population trends and spatiotemporal variation in abundance and territory occupancy of pygmy-owls at the same sites between 2000 and 2021 in northern Sonora, and describe the presence and type of recent changes in land use and landcover.

METHODS

Study Area and System—I considered populations of pygmy-owls in an approximately 20,000 km² region of northern Sonora within approximately 125 km of Arizona (Figure 1). In Sonora, I considered 11 watershed regions between the upper Río San Miguel watershed near Cucurpe west to the upper Río Sonoyta watershed near Sonoyta (Figure 1). In these arid environments, pygmy-owls are generalist predators and non-migratory residents in woodlands associated with saguaro cacti (*Carnegiea gigantea*) that provide nest cavities. The study region included both major vegetation communities occupied by pygmy-owls in the northern Sonoran Desert: the Arizona Upland subdivision of the Sonoran Desert and semi-desert grassland (Brown 1982). Arizona Uplands are dominated by woodlands and scrub of short leguminous trees such as mesquite (*Prosopis velutina*) and saguaros. Semi-desert grasslands are dominated by open mesquite woodlands and savannah, bunchgrasses, sub-shrubs, and saguaros are often uncommon. Riparian areas in both communities are dominated by mesquite woodlands. Annual precipitation in the region is bimodal and dominated by a summer monsoon in late June-Sept and winter storms that are most intense during the warm phase of the El Niño Southern Oscillation. Summers are hot with maximum temperatures >40°C and winters are cool with minimum temperatures near 0°C. Throughout their range, pygmy-owls are diurnal and crepuscular generalists that in our region prey largely on lizards during the warm season. In the Sonoran Desert, pygmy-owls establish breeding territories in January-March, and typically lay eggs in April and brood in May-June.

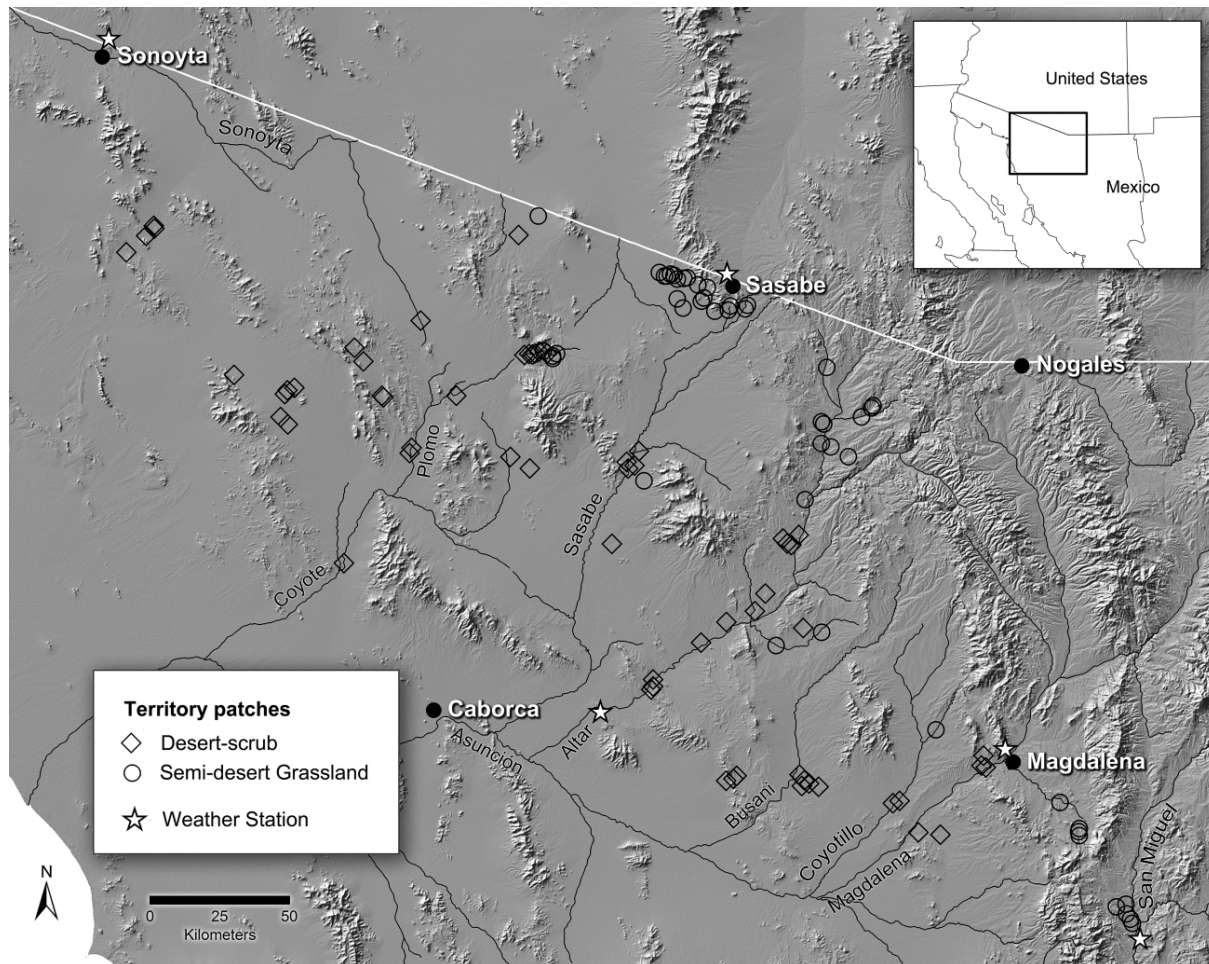


Figure 1. Study area in northern Sonora, Mexico showing distribution of territory patches used by Ferruginous Pygmy-Owls. Territory patches were located in two major vegetation communities across 11 watershed regions: San Miguel, upper Magdalena, Magdalena-Coyotillo, Busani, upper Altar, lower Altar, upper Sasabe, lower Sasabe, upper Plomo, lower Plomo, and Sonoyta). Territory patches were 50 ha in area and are not shown to scale. Fewer watershed regions were considered for abundance monitoring and included upper Altar, upper Sasabe, lower Sasabe, and upper Plomo. Figure from Flesch et al. (2015).

Design and Survey Methods—Design and survey methods used during this effort have been extensively peer reviewed and used to develop a broad range of inferences published in scholarly journals (see Literature Cited). I estimated abundance in Sonora by repeatedly surveying the same transects across time. In spring 2000, I surveyed 71 transects that were selected at random in northern Sonora (see Flesch and Steidl 2006). After these initial surveys, I selected 18 transects in landscapes that were occupied by pygmy-owls at random and surveyed each transect in spring for 15 of the next 16 years (all years except 2012) and again in 2021. All transects were within 75 km of the U.S.-Mexico border and placed along drainage channels. To survey transects, I placed 5-10 calling stations spaced 400 m apart along transects and broadcast recorded, territorial vocalizations of pygmy-owls to elicit responses from owls. This method combined with a minimum of eight minutes of survey effort at each station, the arrangement of stations, and timing of surveys yields nearly perfect detection probability of territorial males (Flesch and Steidl 2007). To minimize chances of double-counting individual owls that often

move toward broadcasts, station spacing was increased to 550-600 m after initial detection of each male. For each owl detected, I recorded the time, distance and direction to the initial point of detection, and the sex based on vocalization type. To estimate the number of pygmy-owls along each transect, I used distance, timing, and direction of responses to differentiate among multiple individuals that did not respond simultaneously. In some cases, I remained at stations for longer than eight minutes to estimate number of respondents and returned to stations for follow-up efforts to confirm the number of estimated individuals. As an index of abundance, I calculated the number of territorial males along each transect in each year. All transects were surveyed between April and early June from 1 hour before to 3 hours after sunrise. All 18 transects combined totaled 54 km in length (mean = 3.0 km, range = 2.3-3.9 km) and were located between 740 and 1,035 m elevation.

To assess territory occupancy, I delineated individual territory patches based on patterns of space use by owls, which I estimated with repeated surveys and nest searches (see Flesch et al. 2015). The basic units of inference were individual territory patches that could each be occupied by single territorial individuals or breeding pairs. To delineate territories, I surveyed transects near random and non-random points in spring of 2000-2004, and searched for nests along occupied transects until I located the nests of most individuals. From 2001 to 2011, 2013 to 2016, and in 2021, I surveyed areas within 300 m of most nests (or occupied areas if nests were not found initially) known from prior years by broadcasting recorded, territorial vocalizations of pygmy-owls in the manner described above, and through 2010 searched for nests exhaustively at nearly all occupied sites. New territories I discovered were added to the study mainly before 2004. To delineate territories, I plotted nest coordinates across time, identified clusters of use in space, and placed 399-m radius circles (50 ha) around the average coordinates of each cluster, which is similar in area to a breeding territory (Flesch et al. 2015). This approach allowed easy identification of breeding territories because the spatial arrangement of potential nest cavities was clumped, owls used the same general areas over time, and owl abundance peaked in early years (Flesch 2014) when presumably most habitat was occupied.

Land-use and Landcover Change—In 2016 with help of USGS, I estimated the aerial cover of landscape structures linked to anthropogenic land uses and landcover (see Flesch et al. 2017). This included areas used for agriculture, housing and urban development, roadways, and other vegetation clearings that create soil and vegetation disturbance. To estimate the location and size of these areas, I digitized them in Google Earth during each successive year that new structures appeared by evaluating all available imagery. I considered structures within 1 km of the center of each territory patch to quantify land-use and landcover change both within estimated owl territories and adjacent landscapes. To represent land use and landcover in a given year, I considered structures that appeared between May of the prior year through April of the current year to match the approximate phenology leading up to the breeding season. Limited support for field work and analyses in 2021 precluded updated quantitative measures of new landscape structures linked to anthropogenic land uses and landcover. In 2021, however, I noted the presence and type of any new structures observed during site surveys that had occurred 5-6 years after the last surveys, and summarized these data to evaluate changes in land use and landcover, which could impact the quantity and quality of habitat for pygmy-owls. Because not all areas of all territories were observed, these estimates are likely biased low.

Analyses—To evaluate spatiotemporal dynamics in abundance, I used linear mixed-effects models. I fit three models to represent different potential patterns of population dynamics and used model selection to evaluate support among models. Each of these three models assessed a different hypothesis regarding population dynamics. The first included a single fixed effect for year fit across data for all four watershed regions combined to represent a single trend for the entire population. The second included a year by watershed region interaction as fixed effects to assess regional variation in trends. The third was an intercepts only model with no trend parameter to represent random or other more complex variation in dynamics. To assess spatial variation in observation error, I considered models that estimated observation variances for each region, which was supported based on model selection. To adjust for temporal autocorrelation, I fit first-order autoregressive correlation structures. All models included a random intercept for transect identify because each of the 18 transects were repeatedly measured across time. As a response variable, I log transformed estimates of the number of territorial males along each transect. Model selection was based on Akaike information criterion adjusted for small sample sizes (AIC_c) and models within 2 AIC_c points considered competitive for all analyses (Burnham and Anderson 2002).

To evaluate spatiotemporal trends in territory occupancy across the broader study area, I fit generalized linear mixed-effects models. These models fit occupancy as a binary response variable (occupied or unobserved) with a logit link function, and different fixed effects to represent the same three potential patterns of population dynamics assessed for abundance. This included models with: 1) a single fixed effect for year across data for territories across all 11 watershed regions combined to represent a single trend for the entire population, 2) a model that considered variation in trends among watershed regions with year, watershed region, and time by region interactions fit as fixed effects, and 3) an intercepts only model with no trend parameter to represent random or other more complex variation in dynamics. When modeling occupancy, I included a fixed autoregressive term for occupancy state in the prior time step to account for Markovian dependencies inherent in occupancy data. This factor adjusted for the tendency of territories having the same occupancy status as was observed in the prior time step, which is especially relevant in this system given site fidelity and fairly long life span of adult male pygmy-owls. Because territories must be occupied to be discovered and included in the study, I also censored data from the initial year each territory was discovered. Because there were gaps in monitoring with no data gathered in 2012 and 2017-2020, 17% of observations had gaps greater one year from the prior time step. For comparison, I also report results from models fit without the autoregressive term for occupancy state in the prior time step. To adjust for correlations in repeated measurements of the same territories across time and of territories embedded in the same landscapes, I considered two potential forms of the random effects: 1) random intercept for territory identity, and 2) random intercepts for territory and landscape identities. I based landscapes identities ($n = 39$) on the proximity of territories in space, and assigned territories located within approximately 5 km to the same landscapes, and used model selection procedures and AIC_c to compare models. Annual estimates were based on least square means adjusted for all fixed and random effects. To evaluate models and validate fit, I plotted scaled residuals against fitted values and assessed patterns in the mean and variance of those values and presence of outliers with large influence. Additionally, I plotted histograms of residuals and q-q plots to visually confirm normality, and confirmed estimates of random effects variances were greater than zero. When modeling occupancy, I assumed perfect detection probability based on evidence

from experimental trials (Flesch and Steidl 2007). Models for abundance were fit with nlme library in R and those for occupancy fit with the lme4 library (Bates *et al.*, 2015; R Core Team, 2020).

RESULTS

Effort and Observations—I surveyed abundance transects ($n = 18$) 306 times across 17 years that spanned a period of 22 years from 2000 to 2021. I observed a total of 607 estimated territorial males during these surveys and an average of 2.0 ± 0.1 males per individual transect (\pm SE) survey (range = 0-8 males). With regard to territory monitoring, I surveyed an average of 90 ± 5 territory patches each year over 16 years (range = 31-108) between 2001 and 2021 for a total of 1,440 occupancy surveys across time. Effort was lowest in 2001 (31 patches) because few patches had been identified during the initial year of study, but increased markedly each year to 2004 after which 98-108 patches were surveyed per year except in 2011 and 2016 (77) and 2021, when I surveyed 93 patches. Raw estimates of observed territory occupancy averaged $59.1 \pm 2.5\%$ among years and ranged from a maximum of 81.6% in 2001 to a minimum of 45.3% in 2009. In 2021, observed occupancy was 50.5% or the fourth lowest value observed since 2001.

Abundance Dynamics—The intercepts only model that did not include a linear trend parameter was the top-ranked model (Table 1). In comparison, there was limited support ($\Delta AIC_c = 1.70$) for a model that included a linear trend across time, and as in past assessments, no support for a model with regionally varying trends (Table 1). A linear trend estimate for annual change in abundance across time was estimated as a $0.26 \pm 0.40\%$ decline per year and was neither statistically or biologically significant. Regardless, abundance varied widely across time and was high initially in 2000 (least square mean = 3.0 males/transect on average), declined steadily to 2008 (1.2 males/transect), increased in 2009 and 2010 (2.0-2.1 males/transect), declined again from 2011-2014, then increased markedly to near initial levels in 2015-2016 (2.7 males/transect; Figure 2). In 2021, abundance declined markedly from levels last observed in 2015-2016 and was 6% lower than the long-term average estimate of 2.0 males/transect (Figure 2).

Territory Occupancy Dynamics—Across a much broader area of 11 watershed regions, occupancy dynamics were fairly similar to those for abundance, but results depended somewhat on model assumptions. The intercepts only model that did not include a linear trend parameter was the top-ranked model but only when an autoregressive parameter for occupancy state in the prior time step was considered (Table 1). In comparison, there was also some support ($\Delta AIC_c = 0.65$) for a model that included a linear trend across time, but in contrast to past assessments, no support for a model with regionally varying trends (Table 1). After adjusting for occupancy state in the prior time step, a trend estimate for change in the odds of occupancy was estimated as a $1.4 \pm 1.2\%$ decline per year, but was not statistically significant ($P = 0.24$). Occupancy state in the prior time step had major impacts on observed occupancy in the current year and reduced AIC_c by ~60 points; odds of occupancy were 3.1 ± 1.2 times higher when a territory was occupied in the prior time step. In general, including occupancy in the prior time step as a covariate adjusted annual estimates of occupancy probability down but mainly just during the first three years of study, which is when most new territories were added to the study. When occupancy in the prior time step was not included, evidence of a decline in occupancy increased

Table 1: Rankings and descriptions of models of spatiotemporal variation in abundance and territory occupancy of Ferruginous Pygmy-Owls in northern Sonora, Mexico between 2000 and 2021. Abundance models consider data from 18 transects in four watershed regions gathered between 2000 and 2021, and are based on linear mixed effects models with abundance (log no. of territorial males) fit as the response variable and transect identity fit as a random intercept. Occupancy models consider data from 112 territories across 30 landscapes in 11 watershed regions gathered between 2001 and 2021 and are based on generalized linear mixed effects models with occupancy (occupied or unoccupied) fit as the response variable and territory and landscape identity fit as random intercepts. Each set of models consider different fixed effects including year, watershed region, and their interaction fit as fixed effects, and a null model with intercepts only. One set of occupancy models includes an autoregressive parameter for occupancy state in the prior time step to adjust for Markovian dependencies inherent in occupancy data.

Parameter				
Model {terms}	K	AIC _c	Δ AIC _c	w_i
Abundance				
Null model {intercepts only}	7	244.90	0.00	0.70
One population with equal trend {Year}	8	246.61	1.70	0.30
Regional variation in trends {Year + Region + Region \times Year}	14	256.46	11.56	0.00
Occupancy with autoregressive parameter				
Null model {intercepts only}	4	1700.91	0.00	0.58
One population with equal trend {Year}	5	1701.56	0.65	0.42
Regional variation in trends {Year + Region + Region \times Year}	25	1713.29	12.38	0.00
Occupancy without autoregressive parameter				
One population with equal trend {Year}	4	1765.24	0.00	0.52
Regional variation in trends {Year + Region + Region \times Year}	24	1766.18	0.94	0.32
Null model {intercepts only}	3	1767.61	2.37	0.16

greatly (Table 1), and a trend estimate for change in the odds of occupancy was estimated as a $2.4 \pm 1.2\%$ decline per year, which was statistically significant ($P = 0.035$). Annual occupancy probabilities across all territories varied widely across time and were high initially in 2001, generally declined to 2009, increased in 2010, were much lower from 2011-2014, then increased markedly to near initial levels in 2015 and 2016 (Figure 3). In 2021, occupancy probability declined markedly from levels last observed in 2015-2016 (Figure 3). After adjusting for the effects of occupancy state in the prior time step, annual estimates of occupancy probability in 2021 were the lowest observed (e.g., 0.435 ± 0.063) across all 16 years of study, and 22% lower than the long-term average estimate of 0.560. When unadjusted for occupancy state in the prior time step, annual estimates of occupancy probability in 2021 were the third lowest observed (e.g., 0.490 ± 0.066) across all 16 years of study, and 18% lower than the long-term average estimate of 0.597. On the raw observed scale, annual occupancy probability in 2021 (0.505) was 15% lower than the long-term average estimate of 0.591. Interestingly, estimates of occupancy were only moderately correlated with estimates of abundance ($r = 0.50$, $P = 0.05$), likely because the study area in which I measured occupancy was much larger than that for abundance.

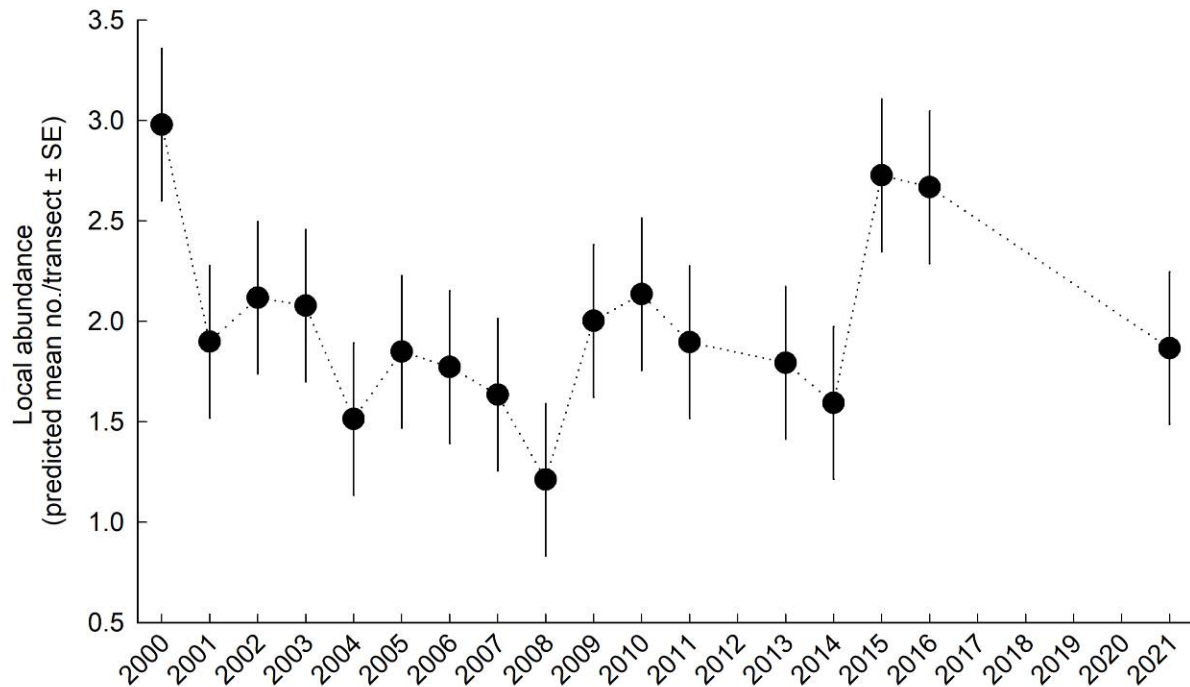


Figure 2: Trends in abundance of Ferruginous Pygmy-Owls along 18 transects (54 km) in northern Sonora, Mexico, 2000-2021. Estimates are mean number of males per transect and are least square means from a linear mixed effects model of log abundance that were back-transformed.

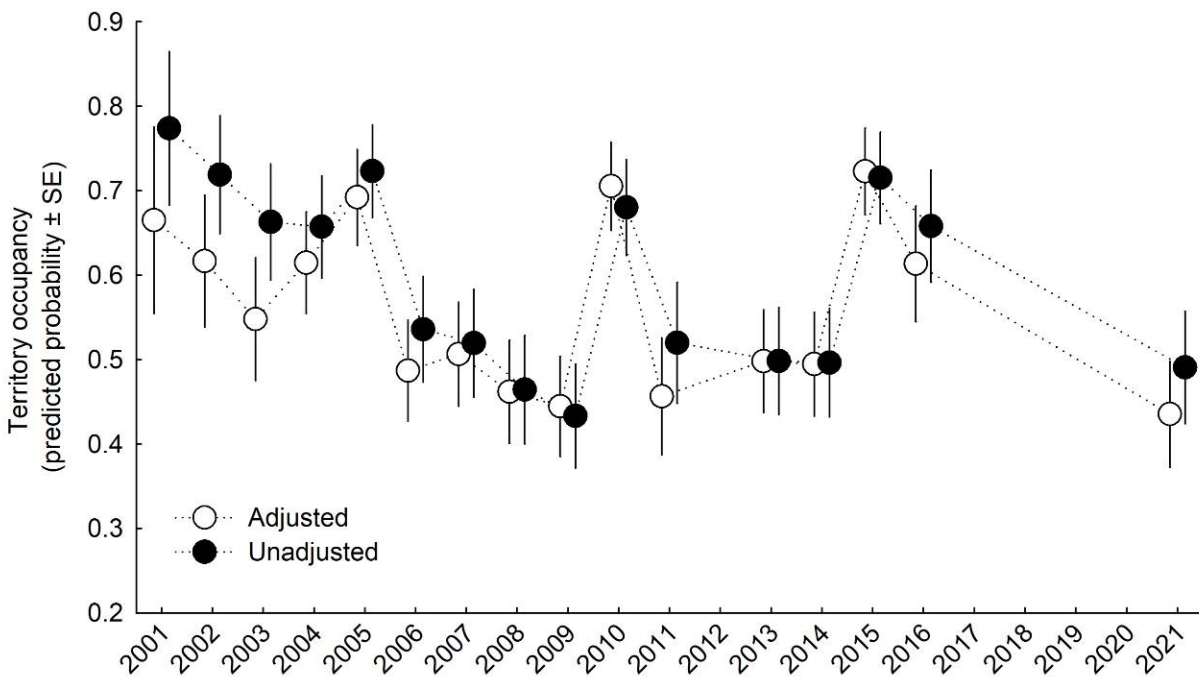


Figure 3: Trends in territory occupancy of Ferruginous Pygmy-Owls in northern Sonora, Mexico, 2001-2021. Estimates are least-square means from a generalized linear mixed effects model that fit year as a nominal factor. Between 31 and 108 territory patches were surveyed each year. Adjusted estimates consider the effects of a first-order autoregressive term for occupancy state in the prior time step.



Figure 4: Examples of landcover change and landscape disturbance at and around territories of Ferruginous Pygmy-Owls I monitoring in northern Sonora, Mexico, 2001-2021. Clockwise from top left show clearing and plowing of desert-scrub, presumably to encourage grass growth for grazing, remains of a charcoal operation that involves clearing and burn pits, buffelgrass invasion on hillside among succulents, and clearing and “pitting” for pipeline installation along Mexico Route 2.

Land-use and Landcover Change—In 2021, I observed new changes in land use or landcover within or immediately around 20 of the 93 (21.5%) territory patches I surveyed. These changes occurred sometime since the last survey of territory patches in 2015 or 2016 that averaged 5.2 ± 0.1 years. Territories that experienced new changes in land use and landcover since 2015 or 2016 were not always those that had the greatest levels of prior disturbance ($P \geq 0.39$, t -tests), indicating a range of less disturbed sites experienced new disturbances. Nine of the 20 territories that experienced new changes in land use or landcover, for example, had no prior disturbances when last surveyed across a period of more than a decade of surveys. Moreover, 31 of 93 territories we surveyed in 2021 had no prior disturbances, and 9 of these experienced new changes in land use or landcover. Most new changes in land use or landcover were characterized by general vegetation clearing ($n = 11$ territories or 58%), with lesser frequencies of new more intensive grazing activities and grazing infrastructure (16%), mining (11%), woodcutting (11%), or water diversion (5%). More specifically, new vegetation clearings I observed were linked to a variety of land uses including agriculture ($n = 2$ territories), clearing for unknown reasons ($n =$

2), buffelgrass planting ($n = 1$), or pipeline ($n = 1$), check-dam type water catchment ($n = 1$), or intensive woodcutting ($n = 1$). Examples of some of the types of landcover change I observed in 2021 at and around territories are shown in Figure 4.

DISCUSSION

Population monitoring is an essential component of wildlife conservation and management, especially for species of conservation concern such the Ferruginous Pygmy-Owl. This report summarizes efforts to monitor populations of Ferruginous Pygmy-Owls in northern Sonora, Mexico that began in 2000 and continued in 2021 following a 4-year gap in field work from 2017 to 2020. Although past monitoring through 2014 indicated marked systematic declining trends in abundance and occupancy that conform generally to findings after years 2005 and 2011 (Flesch and Steidl 2006, Flesch 2014, 2015), results after the 2015-2016 breeding seasons found little evidence of a declining linear trend across a vast area of northern Sonora, Mexico between 2000 and 2016 (Flesch et al. 2017). This more auspicious pattern was driven by large increases in abundance and territory occupancy in 2015 and 2016, which in a statistical sense largely eliminated evidence of a declining linear trend model over the entire time period. Results summarized here that include data from the 2021 breeding season, however, show marked declines of similar magnitude in both abundance and occupancy probability in year 2021 compared to estimates from 2015 and 2016, and were generally low in a historical sense for these populations, especially for occupancy. For example, estimates of territory occupancy based on a model with the highest observed support, were the lowest observed across all 16 years of study (e.g., 0.435) and 22% lower than the long-term average. Moreover, trend estimates for occupancy provided some suggestive evidence of declines, but this model did not account for autocorrelation in successive occupancy estimates across time, which is an important property of these time-series data. Abundance also declined markedly from levels last observed in 2015-2016 and was 6% lower than the long-term average. Such results indicate complex population dynamics and the importance of consistently monitoring populations of conservation concern across time and space so that short-term changes in populations can be distinguished from systematic long-term declines. Additional survey work commissioned by USFWS in Sonora 2021, which may or may not be available in the future, may shed additional light on the status of pygmy-owl populations in northern Sonora, but lacking any baseline for comparison is unlikely to provide any inferences on population trends.

Despite limited evidence of statistically significant declining linear trends in abundance or occupancy across the study region and time period considered here, such declines are likely to develop in the future. This is because populations are expected to respond to increases in habitat loss and degradation, including those documented here, and because aridity is predicted to increase across much of the region (Cook et al. 2015, Pascale et al. 2017, Williams et al. 2020). These stressors both individually and in combination can have negative impacts on populations (Flesch 2014, Flesch et al. 2015, 2017). Additional monitoring and research are needed to assess whether these anticipated declines materialize, and to apply the more than two decades of monitoring from northern Mexico into an analytical framework to assess the drivers of population dynamics.

Although beyond the scope of this report, understanding factors that explain spatiotemporal variation in abundance and territory occupancy is fundamental for guiding management and conservation, and for evaluating why population parameters were generally low in 2021. Recent studies of population dynamics (e.g., Flesch 2014, Flesch et al. 2017) and reproductive output (e.g., Flesch et al. 2015) of pygmy-owls provide strong evidence that precipitation and temperature are major drivers of carrying capacity and population processes in this system. This is because in arid environments, precipitation can drive rapid increases in plant biomass, seed production, and insect abundance, and create resource pulses that directly bolster food availability for small vertebrates and subsequently for predators such as pygmy-owls (Lima et al. 2002, Holmgren et al. 2006). Predator populations such as pygmy-owls often respond indirectly to these resources pulses at somewhat longer lag times of two or more years (Jaksic et al. 1992, Dennis and Otten 2000, Letnic et al. 2005, Flesch 2014). Moreover, hot temperatures can have indirect effects on populations of pygmy-owls by affecting activity or abundance of food resources such as lizards, or direct physiological effects such as heat stress. Marked increases in pygmy-owl populations I observed in 2015 and 2016, for example, were likely due to relatively high levels of precipitation during the 2014 monsoon seasons and in fall 2014 when several tropical storms of Pacific origin made landfall in the region. Much lower estimates in 2021 are likely due, in part, to average precipitation in late 2019 through mid-2020 and to extreme drought in late 2020 and early 2021. These recent droughts may continue to negatively impact pygmy-owl populations into the 2022 breeding season and beyond, especially given lagged effects observed during prior work (Flesch 2014). More broadly and of long-term concern, observed negative effects of high temperature and low precipitation have alarming implications for pygmy-owl in the Sonoran Desert given future forecasts associated with climate change. This is because drought is predicted to dominate future climates of southwestern North America (Seager et al. 2007, Cook et al. 2015, Pascale et al. 2017, Williams et al. 2020), which could drive prolonged ecological crunches that augment extinction risk for pygmy-owls, especially in landscapes where habitat loss and degradation are high (Maron et al. 2015, Flesch 2014, Flesch et al. 2017).

In 2021, I observed a broad range of new landscape disturbances linked to changes in land use and landcover within or immediately around territory patches, which had occurred within the last approximately five years. These new disturbances affected a relatively large number of territory patches I considered during 2021 (e.g., 22% of 93 patches) despite the relatively short time period, and fact that much of the study area includes vast open space with very low human population densities. Such impacts were not limited only to areas that had histories of prior disturbance, with only 29% of new impacts at sites that had evidence of past disturbances. Many of these impacts included vegetation clearing linked to agriculture, grazing, and woodcutting. Discussions with local landowners indicated that the ongoing drought has affected traditional livelihoods and incomes in the region, and driven a need for new sources of income. Hence, evidence of new more extensive woodcutting and charcoal production, new grazing infrastructure, and efforts to clear native vegetation, which is thought to increase grass production, seem to be a result of climate change. Such impacts are likely to have compounding negative effects on habitat and pygmy-owl populations given a range of past observations. For example, past work in this system has found that occupancy declines markedly as territories and the landscapes surrounding them became increasingly dominated by anthropogenic land uses and landscape disturbance (Flesch et al. 2017, Flesch *in revision*). Such patterns are not surprising

given well-known impacts of land-use and landcover change on the quantity, quality, and connectivity of habitats, which simultaneously influence both abundance and movement of potential colonists, reduce colonization rates, and promote edge effects and other stressors that augment extinction risk (Hanski and Gaggiotti 2004, Fischer and Lindenmayer 2007, Lindenmayer and Fischer 2013). Moreover, such impacts can directly interact with other stressors such as climate change. For example, occupancy of pygmy-owl territories imbedded in increasingly disturbed landscapes declined at greater rates with warming winter air temperatures, which had little effect in intact landscapes suggesting they buffer the impacts of climate warming on populations (Flesch et al. 2017, Flesch *in revision*). Due likely to a combination of these stressors, populations of pygmy-owls in regions that have experienced the greatest increases in anthropogenic disturbance in Arizona have declined to extinction (Flesch et al. 2017). Such patterns mirror past population declines that occurred across a much larger region of southern Arizona over the past century (Johnson et al. 2003, USFWS 2011), during which the northern edge of the range of pygmy-owls contracted south by approximately 150 km. Monitoring changes in land use and landcover within pygmy-owl habitat in this region is important for understanding likely future changes to pygmy-owl populations and recovery potential. Future work should quantify these changes using satellite imagery and integrate these data into models of population dynamics.

Population trends of pygmy-owls in northern Sonora, Mexico have important management and recovery implications in Arizona where pygmy-owls are of major conservation concern. Natural or facilitated dispersal of juveniles and adult pygmy-owls from Mexico was identified among the preferred alternatives for recovery compared to captive propagation (USFWS 2003, pg. 123). This later technique was considered feasible only after all other techniques to maintain or improve populations had failed (USFWS 2003, pg. 123), but inexplicably continues to be a major focus of state and federal agencies despite little or no evidence of success and high cost. If populations of pygmy-owls in northern Sonora decline, recovery strategies that rely on individuals from Mexico may be jeopardized. Efforts to conserve, enhance, and create habitat for pygmy-owls will likely have much higher potential for success. This is especially the case when efforts are focused in areas with nearby populations within dispersal range, or when combined with facilitated dispersal of wild birds. Such efforts and others can be guided by broad sets of detailed inferences on habitat quality, population dynamics, habitat selection, dispersal, and movement gathered by University of Arizona researches for nearly two decades (see Literature Cited). To this end, promoting abundances of potential nest cavities, structural complexity and cover of riparian woodlands, reducing deleterious changes in land use and landcover, and promoting landscape connectivity through high levels of woody vegetation cover and large unfragmented woodlands will enhance reproductive performance, movement, and population growth by pygmy-owls (Flesch 2014, 2017, Flesch et al. 2010, 2015, Flesch and Steidl 2010).

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