



Combining trends from disparate monitoring programs to inform Red List assessments: The case of the Cape Vulture (*Gyps coprotheres*)

Christopher J.W. McClure^{a,*}, Brian W. Rolek^a, Ryno Kemp^b, Kerri Wolter^b

^a The Peregrine Fund, Boise, ID, USA

^b VulPro, Skeerpoort, South Africa

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ABSTRACT

Conservation decisions are often informed by Red List assessments, which are substantially influenced by estimated population trends. Population trend estimates used by Red List assessments should therefore be as quantitative, comprehensive, and transparent as possible. We combine counts of breeding pairs of the Cape Vulture (*Gyps coprotheres*) that occur at separate breeding colonies and span differing time periods. The methodological concept is simple: transform time series into interannual rates of change, put those rates into a year-by-series matrix, then average that matrix to estimate the interannual population wide rate of change. Our analysis uses state-space models and basic arithmetic to estimate interannual rates of change per time series. Analyses are performed under a Bayesian framework to ensure that uncertainty is propagated into a composite index that estimates the percent global population reduction over three generations. Our results indicate that the global Cape Vulture population declined by 25 % (95 % CRI = 0.5 %–44 %) from 1977 to 2019. Such a decline suggests the species should be listed as Near Threatened under Criterion A2, instead of Vulnerable, as the species is currently listed. However, we performed a sensitivity analysis that suggested the species might indeed be Vulnerable if unmonitored colonies are in decline. Although the analysis for each species and time series should be customized, we suggest that the general practice of averaging the interannual rates of change for all available time series could improve qualitative estimates of population reduction used in many Red List assessments.

1. Introduction

Conservationists often prioritize species for action based on risk of extinction, and a major determinant of extinction risk is population trend. The International Union for the Conservation of Nature (IUCN) therefore considers population trends during Red List assessments (Criterion A; IUCN, 2012). Given that the Red List is often used to determine conservation priorities (Rodrigues et al., 2006; Betts et al., 2020), it is important that Red List assessors have the best available information regarding population trends of focal species.

Composite indices of population trends can aid Red List assessments by formally and quantitatively combining multiple lines of evidence from disparate monitoring programs into a single estimate. Such indices allow for simplified inference where the single estimate is easier to interpret than multiple unrelated datasets. IUCN provides thorough guidance for estimating population reduction under Criterion A (IUCN Standards and Petitions Subcommittee, 2022)—even providing a spreadsheet for calculating a composite population index. Although this

guidance is certainly useful, it is not universally applicable.

Several studies have used different methods for calculating composite indices. For example, Sherley et al. (2020) used Bayesian state-space models to integrate regional time series and produce composite global population indices of shark population trends specifically for use in Red List assessments. Ogada et al. (2016) used medians across different population trends as composite indices for African vulture species, which influenced several changes in Red List status. There are thus various ways to produce composite indices useful for Red List assessments, and each method should be tailored to the data available per species.

Data collected across differing time periods present a particular challenge when calculating composite population indices. The Bayesian state-space models presented by Sherley et al. (2020) can account for some missing values, but monitoring programs often span different time periods. Time series with excessive missing values limit the utility of Sherley et al.'s (2020) models. IUCN methodology (IUCN Standards and Petitions Subcommittee, 2022) requires weighting subpopulations by

* Corresponding author.

E-mail address: cmclure@peregrinefund.org (C.J.W. McClure).

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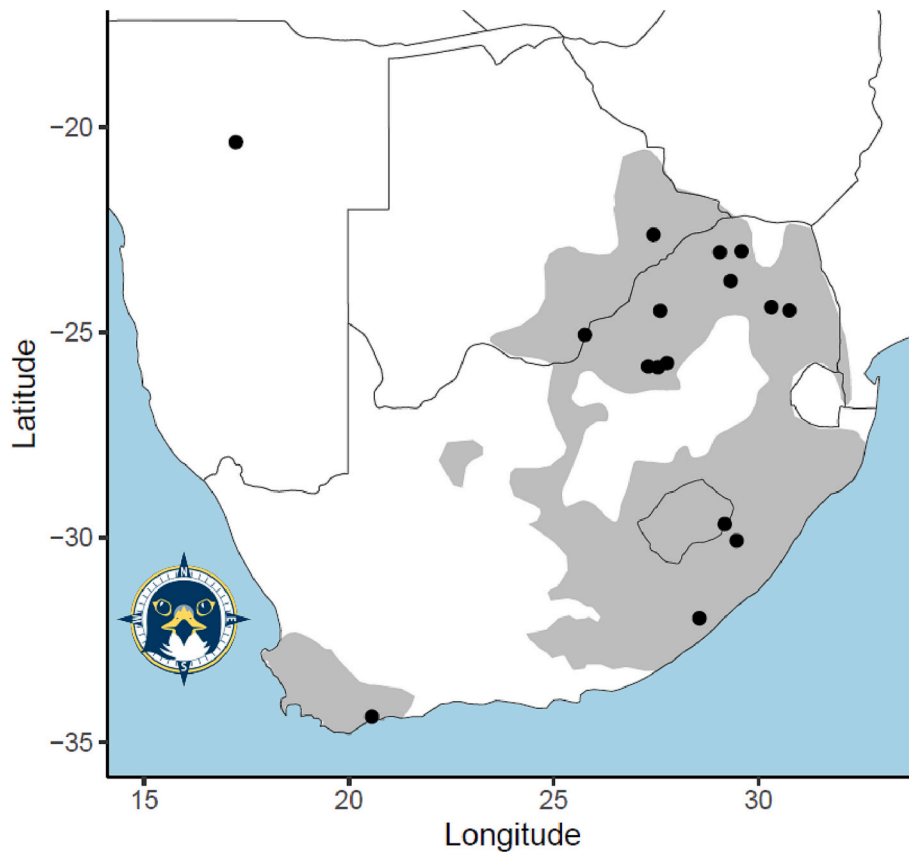


Fig. 1. Map of the breeding range of the Cape Vulture (grey shading) attained from BirdLife International. Points depict locations of colonies analyzed in the current study. The most northwestern point is outside of the current breeding range because it is now extirpated.

their abundances three generations ago, thus recently colonized sites would receive no weight. Here, we present an analysis that combines timeseries collected haphazardly. Our methodology is conceptually simple, we reduce each timeseries to a yearly rate of change and then calculate a weighted average across time series of those yearly changes. We weight each time series by the estimated abundance during a given year, instead of abundance three generations ago. The result is a composite index that estimates the percent population change over the course of three generations.

The Cape Vulture (*Gyps coprotheres*) is a species of conservation

concern, yet its proper Red List status is subject of debate (BirdLife International, 2022). Ogada et al. (2016) included three trend estimates in their composite index of Cape Vultures and calculated a yearly rate of change of -5.1% (lower quartile = -5.8% ; upper quartile = -4.1%), which corresponds to an 89 % decline over three generations (one generation = 13.9 years; BirdLife International, 2022). Such a decline would justify an Endangered listing under Criterion A1 (IUCN, 2012). However, Ogada et al. (2016) did not include any data collected since 2003, when the population appears to have begun increasing (Benson and McClure, 2020). The species is currently listed as Vulnerable (A2acde+3cde+4acde; C2a(ii)). However, the suspected -30% to -49% trend estimate over three generations in the Red List assessment is qualitative and was determined by informally assessing several time series collected at different places over different time periods (BirdLife International, 2022). The Cape Vulture therefore presents an opportunity to clarify the Red List status of a species of conservation concern, while demonstrating that a single composite index can be calculated using disparate, sparse, and partially overlapping time series.

2. Methods

2.1. Data collection

The Cape Vulture is a cliff nesting species that is often monitored by counting breeding pairs at colonies situated on large cliff faces. We obtained time series of counts of breeding pairs (i.e. occupied nests) of Cape Vultures from several sources that monitored the species at colonies across much of its breeding range in southern Africa (Fig. 1). In total, we examined time series from 16 Cape Vulture Colonies (Figs. 1 and 2). Hirschauer et al. (2021) examined counts at seven colonies from 2010 to 2019. Wolter et al. (2007) monitored three colonies during 2006

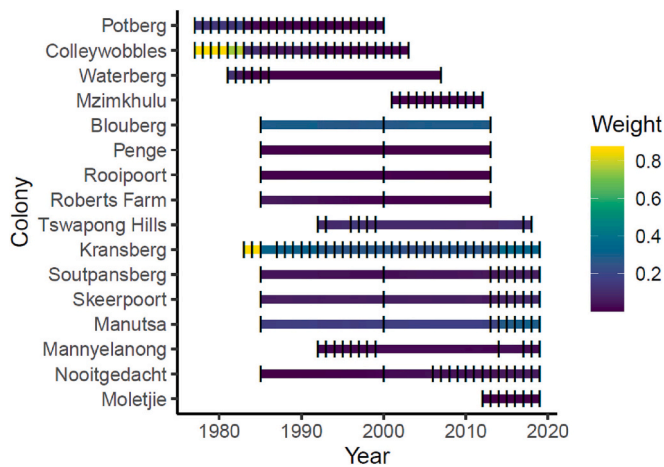


Fig. 2. The years when each time series of Cape Vulture surveys were conducted and their associated weights when determining the population-wide average growth rate (r). Vertical lines depict survey years.

and Whittington-Jones et al. (2011) monitored those same colonies 2007–2009. Benson and McClure (2020) examined counts at the Kransberg colony over 35 years. Benson (2015) conducted surveys at 9 colonies during 1985, 2000, and 2013. Vernon (1999) surveyed Colleywobblers from 1977 to 1997 and then again from 1994 to 2003 (Vernon, 2003). Borello and Borello (2002) counted pairs of Cape Vultures at Mannyananong from 1992 to 1999. Cape Vultures at the Tswapong Hills were surveyed by two studies several years apart (Borello and Borello, 2002; Goikantswemang et al., 2021). The isolated colony at Potberg has been sporadically surveyed since the 1950's but we used data from 1977 to 2000 (Boshoff and Currie, 1981; Robertson, 1984; Scott, 1997; Shaw and Scott, 2003). Counts from 1982 to 1986 of occupied nests at the now extinct colony of Waterberg, Namibia were reported by Brown and Cooper (1987). Brown (1985) reported a count of 62 free-flying Cape Vultures at Waterberg in 1981. We converted this count of vultures to an estimate of occupied nests using the eq. $(62 * 0.74) / 2 = 22.94$ (Piper, 1994). Bamford et al. (2007) indicated that there was a single breeding pair present at Waterberg in 2007. Finally, we tried to source time series from the Living Planet Index's global database of vertebrate counts (www.livingplanetindex.org). However, the only time series from this database for which we could verify methodology was from Schabo et al. (2017) who monitored Cape Vultures at the Mzimkhulu colony from 2001 to 2012. Data are uploaded as Appendix 1.

2.2. Analysis

We assume an exponential growth model for all time series (Sherley et al., 2020) where $\mu_{t+1} = \mu_t + r_t$. μ_t represents the log of abundance in year_t and r_t is the normally distributed change in logged abundance from year_t to year_{t+1} (Appendix 2). We recognize that other patterns of decline, including linear and accelerating, are possible (IUCN Standards and Petitions Subcommittee, 2022), but feel that the exponential pattern is most plausible. As noted above, the basic premise is to transform all the time series into inter-annual estimates of change (i.e. r values), put them into a year-by-series matrix (the 'r matrix'), then use that matrix to estimate interannual population-wide r values. From there, a composite population index can be calculated by scaling yearly abundances (McClure et al., 2023).

The r matrix can simply be averaged per year if the time series have equal weight. However, in general, larger colonies should receive more weight because proportional changes in larger colonies result in greater absolute changes in population size. We therefore weighted each r value by the proportion of breeding pairs at a given colony during that year. This weight calculation required an estimate of number of breeding pairs for the years between surveys. For each colony, we therefore used the equation $\text{abundance}_t = \exp(\text{cumulative sum}(r_t)) * \text{initial count}$ to estimate interannual numbers of breeding pairs across entire time series from the year of the initial count to the year of the final count. Colonies

therefore only influenced the r matrix during the years between and including their initial and final surveys.

We calculated yearly r values for each colony differently based on the frequency and pattern of data collection. See Appendix 2 for details on each individual colony. Generally, we implemented state-space models (Kéry and Schaub, 2012) when there was enough data across a time series. When there were few observations (≤ 4) in a time series—e.g. Benson (2015)—we calculated the rate of change between observations divided by the number of years between observations (r). This procedure assumed a constant rate of change between observations. To analyze data reported by Benson (2015), we therefore calculated r separately for each colony, then added the process variance from Kransberg because it was the best monitored colony. We followed a similar process for Waterberg. For time series at colonies except Kransberg that were surveyed by Benson (2015) and later by Hirschauer et al. (2021), we performed the above procedure for the years reported by Benson (2015), then analyzed the remaining years monitored by Hirschauer et al. (2021) using a state-space model. We analyzed data from counts at Kransberg using an integrated state-space model that used data from Benson and McClure (2020) and Hirschauer et al. (2021). All other time series were analyzed using individual state-space models.

We conducted time series analysis, weight estimation, and r matrix calculations within the JAGS (Just Another Gibbs Sampler; Kellner, 2016) environment using R (R Core Team, 2021). This ensured that all uncertainty in yearly r values calculated when analyzing time series was propagated both into the composite estimates of r and changes in abundance. We implemented all models in a single run using three chains, 10,000 iterations with a burn in of 1000, and a thinning rate of 10. We calculated the Gelman–Rubin statistic (\hat{R} ; Gelman and Rubin, 1992) and determined convergence of chains when parameters had an $\hat{R} < 1.1$. We also visually assessed trace plots of parameter chains to check for convergence.

2.3. Post processing

Our main goal was to estimate the percent decline over three generations, which corresponds for Cape Vultures during this study as from 1977 to 2019. We thus aimed to estimate an average r and the change in abundance scaled to the initial year of data collection (i.e. composite population index). We used the equation: $\text{abundance}_t = \exp(\text{cumulative sum}(r_t))$, to calculate the composite population index. Thus, each year's composite index is the proportion of the first year of surveys during that year. Each time series therefore begins at an index value of 1 with no uncertainty because relative abundance during the first year is certainly 100 % of itself.

The posterior distribution of the rate of decline over three generations can be used to calculate the probability that the population declined by certain percentages. For example, the probability Cape Vultures have declined since 1977 can be calculated as the proportion of MCMC draws > 0 . Red List criterion A2 stipulates that certain thresholds of population growth rate determine threatened status (IUCN, 2012). Following Sherley et al. (2020) we calculated the proportion of the posterior distribution that fell within the thresholds of each category (Table 1) and considered the most likely status as that with thresholds containing the greatest proportion of the posterior distribution.

2.4. Sensitivity

Most of the monitoring of this species has occurred at the largest colonies. However, there are many smaller unmonitored colonies in Lesotho, and in South Africa within Eastern Cape province and KwaZulu-Natal (Piper, 2004a; Boshoff et al., 2009). We endeavored to test the sensitivity of our results to unmonitored and possibly declining colonies within these areas. Piper (2004a) estimated that the global population of Cape Vultures totaled > 3000 pairs, of which > 30 % were

Table 1

The proportion of posterior distribution for percent decline of global Cape Vulture population 1977–2019 that falls within the threshold for Criterion A2 for each IUCN Red List category.

Red List Category	Threshold of Decline	Proportion of Posterior
Least Concern	<19%	0.267
Near Threatened	$\geq 20\%$	0.376
Vulnerable	$\geq 30\%$	0.354
Endangered	$\geq 50\%$	0.003
Critically Endangered	$\geq 80\%$	0

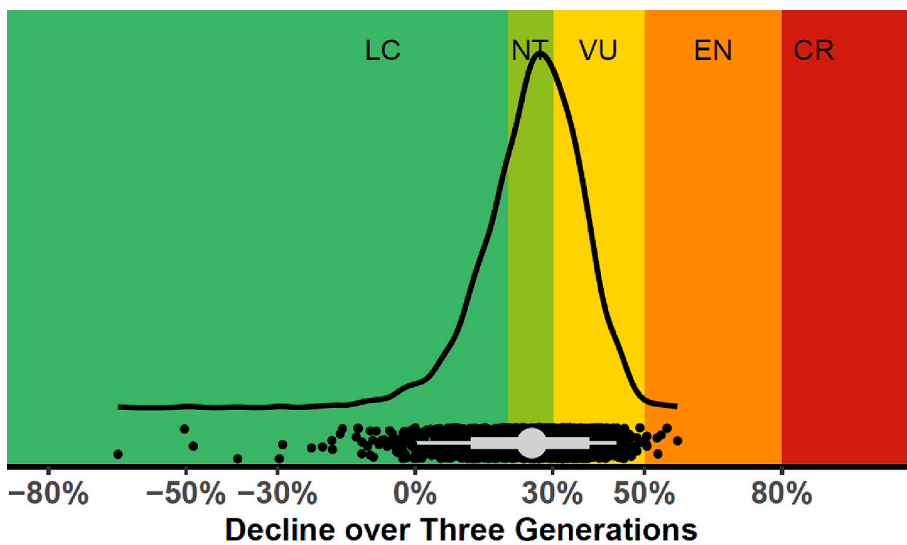


Fig. 3. Raincloud plot depicting the posterior distribution of the decline of Cape Vultures over three generations from 1977 to 2019. The grey point depicts the median of the distribution, the thin grey line depicts the 95% CRI and the thick grey line depicts the 80 % CRI. Colors depict the International Union for the Conservation of Nature (IUCN) Red List categories of Least Concern (LC; green; <20 %), Near-threatened (NT; yellow-green; ≥20 %) Vulnerable (VU; yellow; ≥30 %), Endangered (EN; orange; ≥50 %) and Critically Endangered (CR; red; ≥80 %) as determined by Criterion A2. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

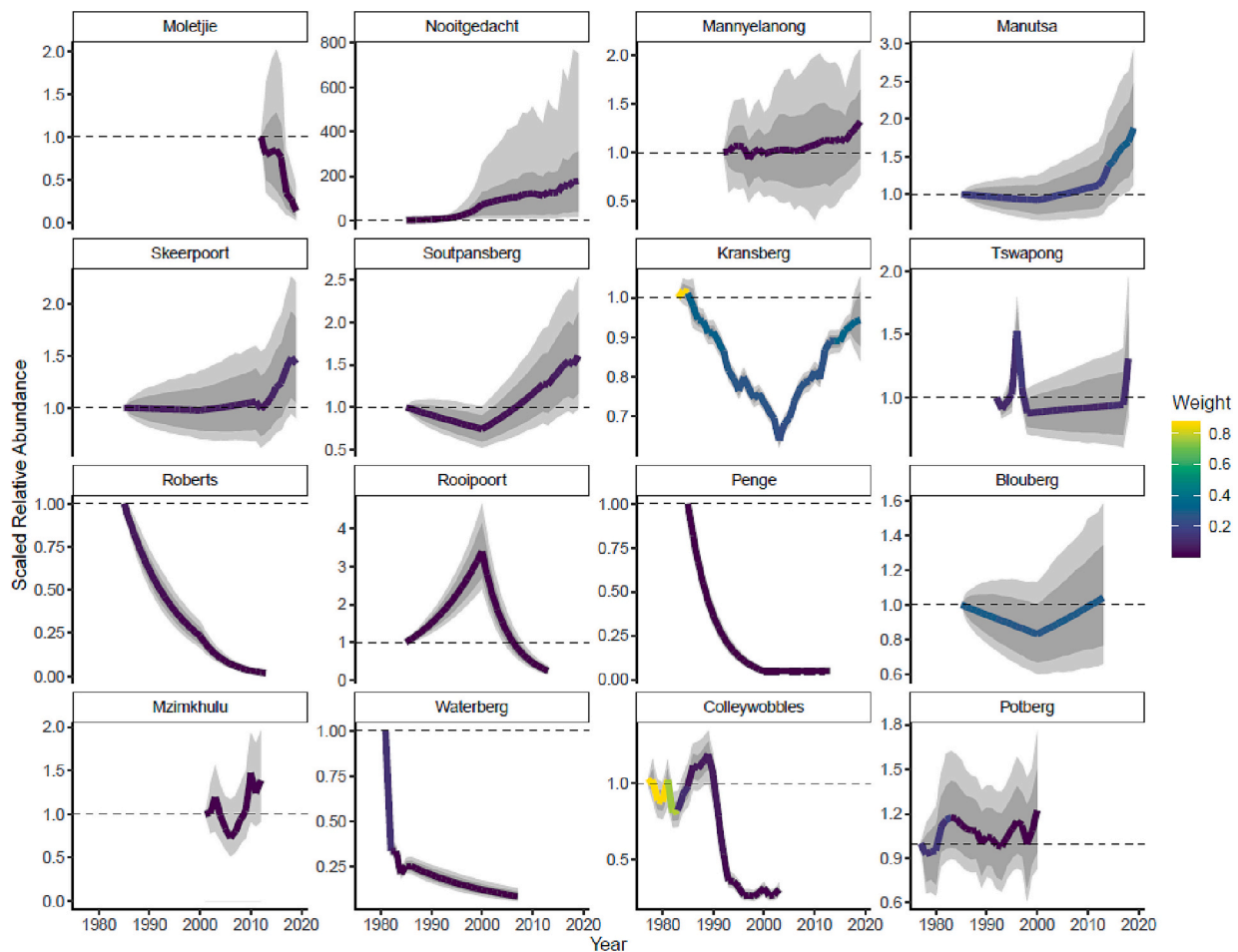


Fig. 4. Relative abundance of each time series scaled to the first year of data collection.

in Lesotho, the Eastern Cape, and KwaZulu-Natal. We therefore created a time series (hereafter the ‘sensitivity time series’) that assumed there were 1000 unmonitored pairs as of 2004 that were declining at the rate that the monitored population was observed to be declining until 2003 ($r = -0.019$). This time series thus tested the potential ramifications of the unmonitored pairs continuing the decline observed in the monitored population, despite the population increases observed post-2003. We

considered this time series a test of the sensitivity of our analysis to the worst-case scenario where pre-2003 declines continued unnoticed. This scenario is consistent with IUCN methodology of estimating a ‘remainder’ of the monitored population (IUCN Standards and Petitions Subcommittee, 2022).

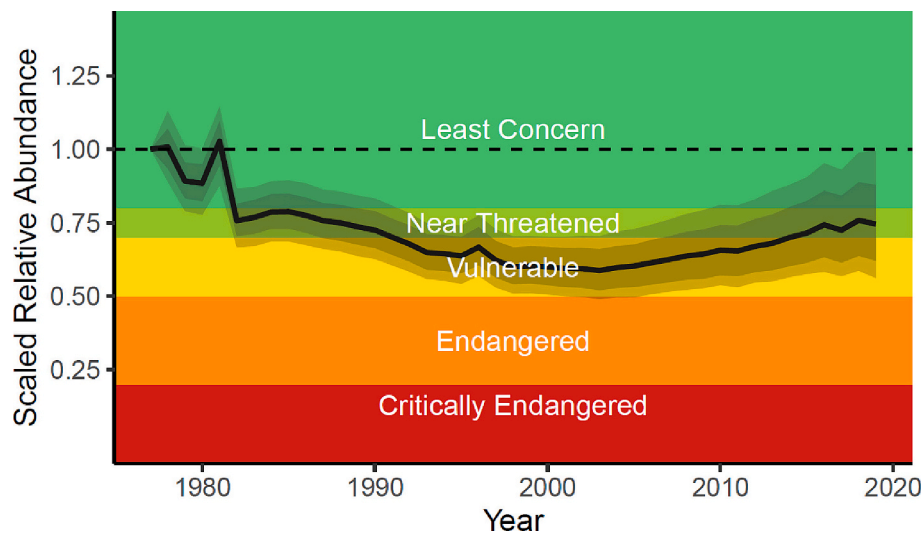


Fig. 5. Composite yearly scaled relative abundance depicted over the Red List thresholds for Criterion A2. Dashed horizontal line at one indicates stability.

3. Results

Some time series (Fig. 4) show great amounts of growth or decline (e.g. Penge and Rooipoort), but counts at these colonies were low and thus these time series did not receive much weight. Kransberg was the most influential time series because it generally had many pairs counted and was the best-monitored colony, resulting in little uncertainty.

The composite index indicates that the Cape Vulture population declined until around 2003 at which point the population began a slight increase (Fig. 5). The average r value from 1977 to 2003 was -0.019 (95 % CRI = -0.026 to -0.012) and from 2004 to 2019 it was 0.014 (95 % CRI = 0.003 – 0.026). In 2003, the population reached its lowest point at 58 % (95 % CRI = 50 %–71 %) of 1977 levels. Comparing the final year of the study (2019) to the first (1977) reveals a decline of 25 % (95 % CRI = 0.5 %–44 %; 80 % CRI = 12 %–0.38 %) over three generations (Fig. 3). There is a 97.6 % probability that the Cape Vulture population declined from 1977 to 2019. Our analysis of observed data suggests that the Cape Vulture is most likely Near Threatened under Criterion A2, given stipulations by Sherley et al. (2020). Indeed, a plurality (37.6 %) of the posterior distribution for percent decline fell within the thresholds for Near Threatened (Table 1), as did the median (Figs. 3 and 5).

After addition of the sensitivity time series, the composite index indicated a decline of 34 % (95 % CRI = 14 %–52 %; 80 % CRI = 23 %–45 %) since 1977 (Figs. S1, S2; Table S1). Under this hypothetical scenario, 68 % of the posterior distribution was within the thresholds for Vulnerable (Table S1). Our sensitivity analysis therefore suggests that the species potentially qualifies as Vulnerable under Criterion A2.

4. Discussion

Our analysis combined 16 timeseries spanning different time periods and sampled at different intervals into a single, easily interpretable index of population change. We specifically improve the global population trend estimate for the Cape Vulture. The current Red List assessment for this species incorporates a population trend that was qualitatively and informally estimated (BirdLife International, 2022). We combine all available information into an estimate that results in a different classification than the current one.

The Cape Vulture is currently listed as Vulnerable under Criteria A2, A3, A4, and C2 (A2acde+3cde+4acde; C2a(ii), BirdLife International, 2022). Our analysis directly addresses the listing under A2 and indirectly suggests that the species should also be reconsidered under the other criteria. Criterion C2 assumes that the species is undergoing a continuing decline, and the other criteria assume the species will

experience future declines. Our analysis suggests that the species has not declined since roughly 2003 (Fig. 5; also see Benson and McClure, 2020). Expectations of future declines are contrary to the observation that counts were increasing as of 2019. According to all evidence we could find, and the resulting composite index, the species seems to be Near Threatened.

However, our sensitivity analysis suggests the current listing of Vulnerable for the Cape Vulture is possible. Monitored colonies were declining before 2003 and our sensitivity analysis effectively asked the question “what if 1000 unmonitored pairs continued declining at the observed pre-2003 rate?” Results from this sensitivity analysis suggest that these potentially declining and unmonitored pairs could result in an uplisting of the species from Near Threatened to Vulnerable.

The range of the Cape Vulture has contracted from historical limits. The species is no longer a breeding resident in Zimbabwe or Namibia. Most of this contraction seems to have occurred before 1977, and thus associated population declines are not relevant to our modeling effort or Red List assessments. Five colonies in Namibia were extinct sometime before 1985 and most of the decline at the Waterberg colony occurred before 1977 (Brown, 1985). We are aware of a single extinct breeding site in Zimbabwe (Hirschauer et al., 2021), and Piper (1994) suggested this site no longer contained breeding pairs by the early 1980's. Although they no longer contain resident breeding Cape Vultures, Zimbabwe and Namibia remain important foraging areas (Kane et al., 2016; Hirschauer et al., 2017).

Reasons for the observed decline and subsequent rise of Cape Vulture counts from 1977 to 2019 remain enigmatic (Benson and McClure, 2020). Benson and McClure (2020) speculated that changes in grazing practices led to more available carrion, which perhaps caused an increase in Cape Vulture counts post-2003. Another hypothesis is that substantial conservation actions for the Cape Vulture over recent decades including: the establishment of vulture restaurants (Schabo et al., 2017); release of captive bred and rehabilitated birds (Jobson et al., 2021); and outreach and education efforts (Hirschauer et al., 2022) have been effective. With many Accipitrid vultures in decline, researchers should determine the cause of the post-2003 increase in Cape Vultures and apply the lesson to management of other vulture species.

We feel our quantitative population trend is an improvement over the current estimate for several reasons. First, we consider more time series than the current Red List assessment. Some of the time series we analyze are decades old and therefore difficult to informally incorporate into a 42-year trend. Second, many of the timeseries considered by the current Red List assessment are themselves informal estimates based on expert opinion (Barnes, 2000; McKean and Botha, 2007; Allan, 2015),

whereas we only consider empirical time series. Future developments might incorporate trends estimated using expert opinion either through Bayesian priors or simply inserting them into the r-matrix.

A third advantage of our analysis is that our assumptions are explicit whereas informal assessments often make unstated assumptions. We assume that it is best to use all information available to estimate the population trend during a given time period. For example, even though there were only two timeseries available during the first few years of the study, we assert that it is better to use these timeseries for inference than disregard them and assume or ignore trends over those years. We also assume that the timeseries monitoring a greater number of individuals are more representative of global population trends than timeseries of fewer individuals and should receive greater weight. Such an assumption generally follows IUCN guidance for quantitative estimation of global population reductions (IUCN Standards and Petitions Subcommittee, 2022), but our methodology allows for the incorporation of colonized sites. Similar weighting could be applied in situations where some timeseries are not as reliable as others, e.g. nest box occupancy versus actual counts (McClure et al., 2017; McClure, 2023). Or, more uncertainty could be added to these time series by increasing their assigned variance. Other types of time series such as aerial or road counts might also be considered for inclusion in future composite indices (McClure et al., 2023).

Our methodology forces practitioners to consider each timeseries formally and quantitatively regarding its relation to the global population trend. We are not suggesting that our exact statistical methodology be used. Every species and time series should be considered separately with customized analysis. Indeed, improvements in survey design would allow for direct incorporation of detection probabilities, which would improve inference but necessitate different statistical analyses. Regardless of the study design or analysis, we suggest that the practice of averaging the interannual rates of change for all available time series would make estimates of population reduction under Criterion A2 more quantitative, comprehensive, interpretable, and transparent.

The sites we analyzed were chosen for monitoring because they are colonies and therefore contain particularly large numbers of individuals compared to surrounding landscapes. Piper (2004b) suggested that the 15 largest Cape Vulture colonies be monitored annually because they collectively contain >80 % of the global population. The dataset we gathered therefore superficially risks regression to the mean whereby sites are chosen for monitoring because they are at random peaks of abundance and then subsequently regress to a mean of lower abundance (Buckland and Johnston, 2017; Fournier et al., 2019; Didham et al., 2020). Regression to the mean can produce ‘false alarms’ where stable populations seem to be declining. However, McClure and Rolek (2023) specifically examined this potential pitfall when monitoring long-lived colonial raptor species such as the Cape Vulture. Their simulation study found that colony monitoring does not risk regression to the mean when the monitored sites are truly of consistently great abundance. Further, even if present, regression to the mean could not explain the observed pattern of decline and subsequent rise in abundance that our results reveal (Fig. 5). Inference from our composite index should therefore be free of artifacts of site selection bias.

Composite population trends require the collation of multiple datasets, which itself might be a difficult task. Indeed, both Ogada et al. (2016) and the current Red List assessment for the Cape Vulture (BirdLife International, 2022) considered substantially fewer timeseries than we did. An important tool for synthesizing the literature is a scoping review (McClure et al., 2022), where all studies of a species are cataloged and categorized in a single searchable database. Such a formal literature review helped us discover several timeseries we examined. Scoping reviews for other species would aid in Red List assessments because they would allow assessors to easily find the studies examining population trends, demography, threats, and conservation actions (McClure et al., 2022).

We appreciate IUCN methodology and the substantial work of Red

List assessors. Indeed, BirdLife International, the Red List Authority for all birds, faces the difficult task of keeping current the assessments of >10,000 bird species. We are therefore not proposing to increase the workloads of Red List assessors, but instead suggest that outside entities can help improve the efficacy of Red List assessments by conducting scoping reviews and producing composite population indices. Conservationists must become aware of population losses with enough time to implement countermeasures (Martin et al., 2012; Lindenmayer et al., 2013; Woinarski et al., 2017), and the Red List is an important warning system. We believe that composite population indices will help Red List assessments become more efficient and more accurate.

CRediT authorship contribution statement

Christopher J.W. McClure: Conceptualization, Writing – original draft, Visualization, Formal analysis, Data curation. **Brian W. Rolek:** Writing – review & editing. **Ryno Kemp:** Writing – review & editing. **Kerri Wolter:** Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data are in Appendix 1

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Appendices. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.biocon.2023.110175>.

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