Short Communication

Status and density of threatened Kori bustard in a woodland savanna

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Abstract:

Grassland habitats have disappeared or undergone substantial change worldwide and many grassland obligatory animal species have populations that are threatened and/or at risk of extinction. The Kori Bustard (*Ardeotis kori*) is the largest flying bird native to Africa and an open savanna specialist, but no research on the population ecology of Kori Bustard in Namibia has been published in the last 30 years. Using distance sampling from driven transects, we found no significant variation in density estimates among seasons (2016–2021) and we estimated 0.34 birds/km^2 (± 0.1) in the 2021–2022 calendar year, which projects a population of <200 (~196)

Kori Bustards for our 577 km^2 study area. A nationwide status survey and investigation of potential limiting factors, including survival analyses and connectivity between populations, will be important undertakings to better inform conservation strategies for this imperilled species in Namibia and in other range countries with data deficiency.

Keywords: *Ardeotis kori*, distance sampling, grassland, Namibia, Otididae, population estimation

Introduction:

The bustard family (Otididae) is known to be one of the most endangered avifauna families (Silva et al. 2022). Other than Northern Black Bustard (*Afrotis afraoides*; Shaw et al. 2021) and Karoo Bustard (*Eupodotis vigorsii*; Anderson et al. 2001), all bustards are listed as "vulnerable," "near threatened," or "critically endangered" (Anderson et al. 2001; Shaw et al. 2021; Ram et al. 2022; Uddin et al. 2021; Scott and Scott 2020; Kasambe and Gahale 2010). Habitat loss through conversion to crops and expansion of human settlements have altered foraging and safety refugia for many populations within the bustard family. Anthropocene-driven bush encroachment (Bernardino et al. 2018) and power-line-induced mortality (Uddin et al. 2021) also negatively impact some populations.

The Kori Bustard is the largest flying bird in southern and eastern Africa (Stevenson and Fanshawe 2002). Morphometry (Maloiy et al. 1987), the natural history of the species (Allan and Osborne, 2005), habitat use (McCollum et al. 2018), body size variability (Hallager 2003), courtship displays (Osborne and Osborne 1998), and participation in parental care (Hallager 2003, Mwangi 1998) have been addressed in literature reviews for the species. However, research on the population ecology of the Kori Bustard (*Ardeotis kori*) in Namibian farmlands is lacking, hindering projections of population growth, risk assessment, habitat suitability analysis, and conservation efforts by and large. Most Kori Bustard populations are data deficient and likely under severe threat (Senyetso et al. 2012), requiring the collection and analysis of baseline information that can assist with threat mitigation.

The Cheetah Conservation Fund (CCF) in Namibia has extensively monitored wildlife on Namibian farmlands for the last 30 years, mainly focusing on large carnivores and their prey base. From 2016 onwards, vehicle transects have been driven systematically on an annual basis across seasons, recording animal data, mainly focusing on estimating ungulate population trends and carrying capacity of large carnivores. Through this long-term collection of data, we also obtained observations of Kori Bustards in a distance sampling framework, to facilitate density estimation of this species in a woodland savanna. We surveyed the entire CCF property of 567 sq. km using 10 vehicle transects. Our study is the first in the last 30 years in Namibia to estimate population size (Osborne and Osborne 1998) of this near threatened species (IUCN 2016). Documenting the status of lesser-known species typical of grassland ecosystems is a conservation urgency, as grassland systems and component species are being affected by human activities on a global level (Jhala et al. 2021).

Methods:

Our study was conducted in a woodland savanna of north-central Namibia on property owned by the Cheetah Conservation Fund (577 km^2), subdivided into a wildlife reserve and mixed-use farms (wildlife and livestock; Fig. 1). The study area had three seasons (a) cold-dry (May–August), (b) hot-dry (September–December), and (c) hot-wet (January–April; Nghikembua et al. 2020; 2021). We used the habitat stratified conventional distance sampling method (Thomas et al. 2010) to estimate the density of Kori Bustard sighted along transects (Fig. 1) in 2021–2022. From July 2021–April 2022, we drove ten transects (19.2 ± 4.04 km) two times per season (2 replicates), resulting in a total of 54 transect runs for the year during early morning (around 6:00) and late afternoon (around 16:30). Our total annual sampling effort across the three seasons in 2016–2021 was 1,786.2 km.

While driving the transects, the driver maintained a low speed (approximately 10 km/h), while 2–4 observers searched for potential prey species of Cheetah (*Acinonyx jubatus*) and Leopard (*Panthera pardus*), including ungulates and large birds. We recorded the number of conspecific individual animals observed at each sighting. Whenever possible, age and sex of the animals were also recorded, but these data can have potential observer bias, and were therefore not included in our analysis. After spotting an animal, we recorded the perpendicular distance to it from the transect using a Bushnell rangefinder (Bushnell golf tour V5 patriot, Kansas, United States). Transects were separated by 2-5 km to maintain independence of observations.

We used the conventional distance sampling approach (CDS) in Program DISTANCE (Version 7.4; Thomas et al. 2010) to estimate the annual density of Kori Bustard for the most recent calendar year (2021–2022) based on data pooled across the three seasons. In addition, we assessed seasonal variation in Kori Bustard density based on long-term transect dataset (2016–2021) wherein we pooled the data of these multiple years into the corresponding seasons.

We fit four detection functions (1) half-normal; (2) uniform; (3) hazard rate; and (4) negative exponential with cosine, simple polynomial, and hermite polynomial series expansion. We fit these expansions to examine and explain the fitted graph between the detection function and perpendicular distance and we truncated the outliers to improve goodness of fit. We truncated distance beyond 150 m to improve the goodness of fit of the best model. We used Akaike's Information Criterion (AIC) values to select the best model, and Kolmogorov–Smirnov statistics to examine each model's goodness of fit (Buckland et al. 2004). We estimated the population of Kori Bustard by multiplying bustard density by the total area of the wildlife reserve and farms.

Results:

We recorded 33 sightings of Kori Bustard with an encounter rate of 0.02 birds/km in the 2021–2022 calendar year, which yielded a density of 0.34 birds/km^2 (± 0.1 SE; [95% CI: 0.18–0.63]). Our data from north-central Namibia elicits a population projection of 196 Kori Bustards on Cheetah Conservation Fund property during 2021–2022. The uniform cosine model was selected as the base model based on the AIC value (AIC = 291.04) with a detection probability of 0.21 and effective strip width of 40.97 m (Supplementary Figure S1). The mean cluster size of observed bustards was 1.13 +/- 0.64E-01 birds.

We recorded 53, 101, and 58 independent observations of Kori Bustard in cold-dry, hot-dry, and hot-wet seasons respectively during 2016-2021. We found no statistically significant seasonal variation in density estimates, although the mean density in the hot-dry season (0.52 ± 0.17) appeared slightly higher than in the hot-wet (0.44 ± 0.14) and cold-dry seasons (0.44 ± 0.12; Fig. 2). Details on mean cluster size, seasonal detection probability, and the goodness of fit of the best models are reported in Tables 1 & 2.

Discussion:

The largest global populations of Kori Bustard reside in Namibia and Botswana (Allan 2005; Herremans 1998). Although the total population size of the Kori bustard in its native resident habitat (Namibia, Botswana, South Africa) is unknown, the population trend is decreasing (Birdlife International, 2023). In the rest of Africa, estimates range between 5,000-10,000 individuals with an area of occupancy of 721,000 km^2| (Osborne & Osborne, 1998). To our knowledge, only one study has addressed the ecology and conservation management of Kori Bustard in Namibia (Osborne and Osborne 1998). Due to lack of observations, we were unable to project the growth rate of the Kori Bustard population in our study area to investigate the trend for this population. The detection probability of Kori Bustard was higher during the cold-dry and hot-dry seasons (0.34 and 0.33, respectively) compared to the hot-wet season (0.26), likely due to the growth of graminoid and of forb foliage in the savanna landscape. Still, the detection probability had a contrasting effect on density estimates. This might be explained by observer bias and observer-favored sampling sites with variable vegetation type, road network, accessibility, and under-reporting of more difficult-to-reach areas. Instead of driven transects as used in our study, random transects distributed across the area that are sampled by walking might provide more accurate density estimates (SNZP 2014), allow nest monitoring (Mmassy et al. 2018), computing of sex ratio, and inferring movement (Mmassy et al. 2019). However, walked transects are substantially more labor intensive than driven transects, take longer to execute, and are unsuited for areas with dangerous wildlife. Still, in the absence of marked birds in our study area, we cannot conclude the underlying reason for any potential seasonal variation of density estimates. A recent study in the Serengeti ecosystem highlighted variation in the seasonality and sex-specific movement attributes of Kori Bustards (Mmassy et al. 2019). Breeding females confined their movements to a small area to participate in parental duties, whereas males used large open grasslands to elicit courtship displays to conspicuous females (Mmassy et al. 2018). Previously it was reported that Kori Bustards are nomadic in Namibia and resident in countries in other parts of their range, such as Zimbabwe and Botswana (Shaw 2013). In our study area, permanent and predictable water was available year-round at natural and artificial waterpoints, which might have precluded the need for the local Kori Bustard population to migrate seasonally to access water.

Kori Bustard populations are thought to be declining throughout the species' entire range, and in 2016 IUCN denoted the species as "Near-threatened." To our knowledge, reliable population estimates are not available for any country within Kori Bustard range. Nonetheless, widespread population declines are conceivable and could be related to hunting and persecution (Astley-

Maberly 1937; Porter 1949; Herremans 1998), bush encroachment driven by overgrazing (Collar 1996; Nghikembua et al. 2021; Senyatso 2011), and infrastructural development, especially powerline mortalities (Collar et al. 2017).

Due to data deficiency and challenges to their conservation as exemplified above, the status and viability of Kori Bustards across Namibia and throughout their range are largely unknown. Localized populations likely persist in some areas, such as open grasslands protected as wildlife reserves, and remote areas with minimal human footprint. We suggest that investigating the distribution and density of component populations, as well as connectivity between them should be prioritized for conservation research. Information on survival of various age classes, disease risk, and direct anthropogenic impacts such as through hunting, powerline mortality, and roadkills also need to be evaluated.

Conflict of interest:

None

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Data availability statement:

The data associated with this article is available in the Zenodo online repository (10.5281/zenodo.7903107).

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Table 1: A single global detection function was modeled for Kori bustard in 2021–2022 from the entire study area. Density was estimated in Program DISTANCE. SE is standard error, the best models are UN = Uniform, HN = Half-normal, and HR = Hazard rate. Adjustments: Cos = Cosine and Pol = Simple Polynomial, and ESW= Effective Strip Width

Kori bustard	Observations	Model	Adjustment	Density (SE)	AIC value	P-value	Mean cluster size	ESW (m)
Year								
2021-2022	33	UN	Cos	0.34 (0.1)	291.04	0.21	1.13	40.97

Table 2: A single global detection function was modeled for Kori bustard by pooling multiple-year data (2016–2021) of corresponding seasons from the entire study area. At the same time, the density across pooled years for each season was subsequently estimated in Program DISTANCE. SE is standard error, the best models, are UN = Uniform, HN = Half-normal, and HR = Hazard rate. Adjustments: Cos = Cosine and Pol = Simple Polynomial, and ESW = Effective Strip Width (m).

Kori bustard	Observations	Model	Adjustment	Density (SE)	AIC value	P-value	Mean cluster size	ESW (m)
Season								
Cold-dry	53	UN	Cos	0.44 (0.12)	533.28	0.34	1.4	68.11
Hot-dry	101	HZ	Pol	0.52 (0.17)	1035.24	0.33	1.24	93.25
Hot-wet	58	HN	Cos	0.44 (0.14)	569.45	0.26	1.37	69.21

Figure 1: (Left) The current geographic distribution of the Kori bustard (IUCN, 2022). The red star indicates the study area location. (Right) Spatial distribution of transects on Cheetah Conservation Fund property. Individual transects are illustrated with distinct colors. Arrows indicate the route of the vehicle transect.

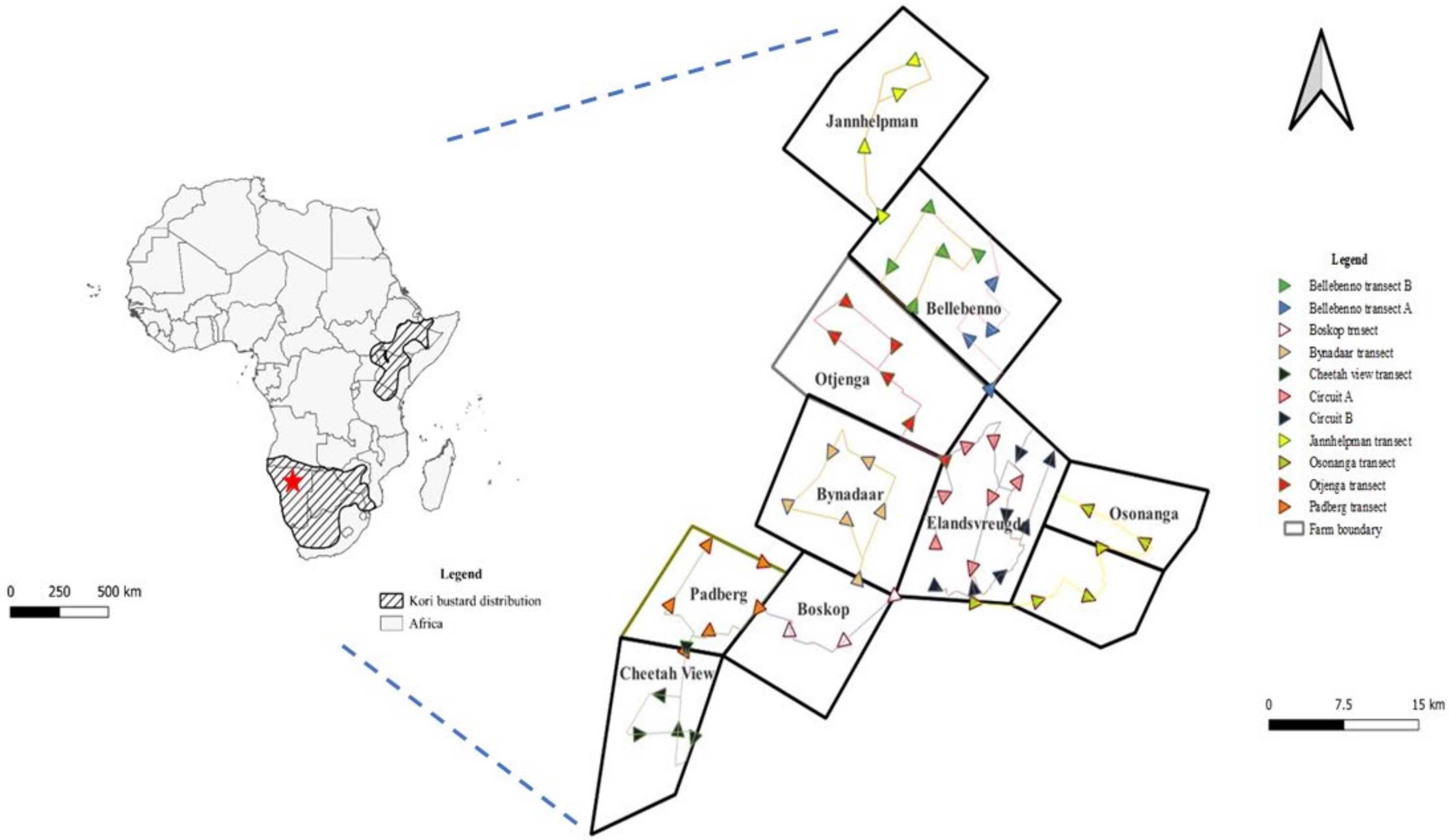

Figure 2: Annual and seasonal density estimates (number/km^2) of Kori bustard during 2016-2021; error bars are 95% confidence intervals. Photo credit: Kathan Bandyopadhyay.