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Counting bears in the Iranian Caucasus: Remarkable mismatch between scientifically-sound population estimates and perceptions

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ABSTRACT

Lack of reliable information on the status of species often leads managers to exclusively rely on experiential knowledge, opinions or perceptions, usually derived from personnel associated with natural resource management agencies. Yet, the accuracy of these sources of information remains largely untested. We approached this challenge, which is particularly common for wildlife monitoring programs in developing countries, using a population of Asian brown bears (*Ursus arctos*) in the Iranian Caucasus as case study. We conducted a non-invasive, genetic, spatial capture-recapture (SCR) study to estimate bear density across a core protected area, Arasbaran Biosphere Reserve, and compared our estimate of bear abundance with rangers' perceptions as collated through interviews. The perceived abundance of bears by local rangers was between 3 and 5 times higher than our SCR estimate of 40 bears (2.5–97.5% Bayesian Credible Intervals = 27–70; density: 4.88 bears/100 km^2). Our results suggest that basing management of the local bear population on perceptions of population status may result in overestimating the likelihood of population persistence. Our findings offer a scientific baseline for an evidence-based conservation policy for brown bears in Iran, and the broader Caucasus Ecoregion. The majority of threatened terrestrial megafauna occur in developing countries, where collecting and analyzing demographic data remain challenging. Delayed conservation responses due to the lack of, or erroneous knowledge of population status of such imperiled species may have serious consequences.

1. Introduction

Reliable information on the status of wildlife populations is essential to inform decision-making processes, assess the degree of compliance with planned conservation and management goals, or avoid undesirable outcomes from the implementation of interventions (Nichols and Williams, 2006; McCarthy and Possingham, 2007; Jones et al., 2013). Lack of accurate estimates of population parameters such as abundance, or worse, use of biased information in decision-making may mislead the prioritization of conservation and management actions (Katzner et al., 2011; Gopalaswamy et al., 2015; Bischof et al., 2016). Understanding confounding factors influencing conservation practitioners and wildlife managers' judgments about imperiled species, such as those related with the status of populations or the expected impact of interventions, is required to improve current management and conservation practices (Popescu et al., 2016; Heeren et al., 2017).

Insufficient financial resources and logistical constraints imposed by harsh climates or inaccessibility, often prevent conducting reliable population estimates, particularly in developing countries (Danielsen et al., 2009; Brook et al., 2014). Under this situation, wildlife managers may base their decisions on lower-cost, proxy-based, approaches, usually derived from experiential knowledge, opinions or perceptions of the status of target species (Sutherland et al., 2004; Fazey et al., 2006; Jones et al., 2013; Bennett, 2016). Several studies have pointed out the utility of employing trained local people in monitoring and evaluation of conservation programs (e.g., Steinmetz et al., 2006; Danielsen et al., 2009; Kindberg et al., 2009). However, the reliability of this source of information to form the solid basis for evidence-based practices remains doubtful (Sutherland et al., 2004; Adams and Sandbrook, 2013). Lack of calibration and validation of population estimates may result in wrong decisions and inappropriate allocations of limited resources (Katzner et al., 2011; Gopalaswamy et al., 2015).

An important controversy surrounding large carnivore conservation emerges in relation to the accuracy of the available information on the

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status of these species, particularly when different stakeholder groups proffer disparate information (Kendall et al., 2009; Chapron et al., 2014; Ripple et al., 2016). Contentious debates about the population estimates of the brown bear (*Ursus arctos*) is such an example. Although effective conservation and management planning for bear populations require robust estimates of population parameters (Bischof et al., 2016; Morehouse and Boyce, 2016), failure to collect reliable data and use of biased approaches dictated by non-scientific incentives (e.g., public interest or trophy hunting) pose a central problem in supporting ecologically-meaningful actions (Kendall et al., 2009; Artelle et al., 2013; Popescu et al., 2016). Further, the conservation status and allocation of monitoring efforts for bear populations are contrasting across the species' global range (McLellan et al., 2017).

In Asia, hunting pressure to obtain bear body parts and conflict-related persecution, coupled with the anthropogenic habitat loss, have resulted in a drastic decline of bear populations (Nawaz, 2007; Lortkipanidze, 2010; Latham et al., 2012; McLellan et al., 2017). Particularly in Southern and Western Asia, reliable data on bear status is extremely limited and many bear populations were extirpated well before any information on their status was available (Bellemain et al., 2007; Garshelis and McLellan, 2011; McLellan et al., 2017). In Iran, brown bears often persist in habitat patches within human-dominated landscapes, and anecdotal sources of information suggest that bear populations are under decline (Gutleb and Ziaie, 1999; Yusefi et al., 2015). Nevertheless, scientifically-sound population estimates of Iranian bears are still lacking, and the available data is based on either experiential knowledge (Gutleb and Ziaie, 1999; Gutleb et al., 2002) or opportunistic visual counts (Farhadinia and Valizadegan, 2015; Parchizadeh, 2017).

The Iranian protected areas (see UNEP-WCMC, 2017) are primarily designed for, and much of the conservation efforts by wildlife managers have been centered around, the conservation of wild ungulates; not only in response to widespread poaching (mainly for meat; Ashayeri and Newing, 2012), but also because the abundance of wild ungulates is generally perceived as an indicator of local managers' enforcement effectiveness (Tourani et al., 2014). However, lack of information about the abundance and population trends of large carnivores, such as the case of brown bears, preclude a proper evaluation of the conservation status of this guild, and the mitigation of human-carnivore conflicts (Chapron et al., 2014; Ripple et al., 2014). Because of the widespread human-bear conflict, mainly related to bear damages to crops and livestock, and occasional attacks on humans (Gutleb and Ziaie, 1999; Qashqaei et al., 2014; Yusefi et al., 2015), it is important that future conservation plans for bears in Iran be based upon a realistic knowledge of the status of local bear populations.

Globally, expert and non-expert experiential knowledge have been used in large carnivore monitoring programs, particularly in large populations across regional scales (e.g., Steinmetz et al., 2006; Chapron et al., 2014). Using an Iranian brown bear population and local perceptions about its abundance as case study, we therefore asked the general question: how reliable is the experiential knowledge about the status of large carnivore populations for making sound management decisions? To address this issue, we examined the reliability of local perceptions as the only available source of information that is commonly used as decision-making shortcuts (Bennett, 2016; Heeren et al., 2017), to evaluate whether such heuristics can be used to support management decisions for brown bears in Iran. To do this, we conducted a noninvasive, genetic, spatial capture-recapture (SCR) study to estimate bear density across a core protected area, and compared our estimates of bear abundance with local rangers' guesstimates collated through interview surveys.

SCR models provide spatially-referenced estimates of density and abundance by linking individual encounter history data with space, and predict a latent variable representing the location and number of individuals' activity centers (Efford, 2004; Royle and Young, 2008; Royle et al., 2014). The collection of activity centers can be thought as the realization of a statistical point process describing a biological pattern. A standard model is the half-normal encounter model, where the probability of encounter depends on the distance between the detector location and the individuals' activity centers (Royle et al., 2017). Additionally, the SCR framework can support flexible sampling (i.e., trap) arrangements, and incorporate both individual- and station-level covariates (Sollmann et al., 2013; Efford and Fewster, 2013; Royle et al., 2014; Sun et al., 2014). Therefore, the noninvasive, genetic, SCR approach would be ideal for obtaining reliable estimates of density and abundance for small bear populations. We used our results to expand the current knowledge of Asian brown bear populations, and evaluate how the use of unverified perceptions may influence the interpretation of priorities for conservation and management of such imperiled species.

2. Material and methods

2.1. Study area

Arasbaran Biosphere Reserve (ABR) spreads across approximately 807 km^2 of the Caucasus Ecoregion in northwestern Iran (38° 40′ to 39° 08′ N, 46° 39′ to 47° 02′ E; Fig. 1). ABR is geographically dominated by mountainous and semi-arid steppe habitats with elevations ranging from 256 to 2896 m (Sarhangzadeh and Makhdoum, 2002). Subalpine meadows, grasslands and agricultural lands are intermixed with relatively large patches of temperate mixed broad-leaved forests (Fig. 1) typical from the Caucasus-Hyrcanian biome (Sagheb-Talebi et al., 2014). Aras River marks northern boundaries of ABR with Republic of Azerbaijan (Fig. 1), and several smaller streams draining from this transboundary river flow into the study area. Climate is temperate Mediterranean, and annual precipitation and mean annual temperature vary from 316 to 686 mm and from 5 to 14 °C, respectively (Sagheb-Talebi et al., 2014).

Human activity within ABR is substantial. At least 66 inhabited villages and thousands of nomads occur inside the reserve (human density: > 18.0 people/km^2; www.amar.org.ir). Local people are mostly agro-pastoralists who graze the entire ABR, with the exception of the study area's ca. 90-km^2 core zones (Fig. 1) that collectively were upgraded to a legal status of National Park in 2012. Sheep (*Ovis aries*), cattle (*Bos taurus*), and goat (*Capra hircus*) are stocked at remarkable densities within ABR (> 104.4 livestock units/km^2; Sarhangzadeh and Makhdoum, 2002). Seven permanent and two seasonal ranger stations guarded ABR during this study (Fig. 1). Each ranger station was responsible for a patrol section within the study area. Although exact geographical boundaries did not exist for patrol sections, each section was defined according to geographical features, nearby human population areas, road access, and location of wild goat's (*C. aegagrus*) core habitats (Fig. 1).

2.2. Noninvasive genetic sampling

In 2012, we collected bear feces between July 3 and September 17 (10 weeks) in ABR, within the period of hyperphagia, and just after the peak of infanticides reported in bear populations from similar temperate regions (Steyaert et al., 2012). This design reduced potential violations of population closure assumptions. We divided ABR into eight sampling areas based on the existing patrol sections at the time of this study (Fig. 1). We followed a single-sampling occasion approach (Bellemain et al., 2007; Puechmaille and Petit, 2007; Royle et al., 2014), surveying each section once. Sampling was opportunistic focusing on potential bear habitats identified by rangers and local villagers (Moqanaki et al., 2013). Surveys in each section lasted 3–5 days depending to the availability of potential bear habitats, accessibility, and logistical constraints, and all survey routes were recorded with a GPS unit. We did not sample the lowland sections along the Aras River, as they were dominated by towns and human infrastructures (Fig. 1).