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Assessing different management scenarios to reverse the declining trend of a relict capercaillie population: A modelling approach within an adaptive management framework

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ABSTRACT

Capercaillie populations are declining in most of its distribution area and are especially vulnerable in southern Europe. There, the causes of decline have not been properly identified and conservation decisions are too frequently based on poor evidence. We analysed the trend of a relict capercaillie population in the Spanish Pyrenees on the basis of bird densities during the period 1989–2010, and we developed the first modelling exercise for the dynamics of a capercaillie population in southern Europe. We also explored management actions commonly used to enhance endangered prey species thought to be affected by hyperpredation: the release of captive-bred females, in varying numbers, the removal of predators and the combination of both actions, using available information from past captive-breeding experiments and from an ongoing experiment involving the removal of terrestrial mesopredators. The population was found to be declining at an annual rate of 4%. Restraining the unknown adult survival rate according to values reported in the literature, our modelling approach showed that recruitment (productivity + fledgling survival), rather than adult survival, was the demographic parameter to be improved as a result of management and, thus, most likely to increase the population growth rate. The removal of terrestrial mesocarnivores may lead to the stabilisation of the capercaillie population (lambda 0.99 ± 0.06), through the improvement of productivity, although this result should be considered as preliminary. However, the most effective management strategy was the combination of predator removal together with the release of 15, 30 or 45 adult females per year. These strategies should be viewed as urgency measures and should be implemented in combination with other long-term measures, such as the reduction of forest density, which may influence predation rate and food availability, and the control of the numbers of wild ungulates, which could be subsidising mesopredators in the area. We also present the adaptive management framework which will allow the results obtained from our current modelling to be updated in the near future, what will reduce the uncertainty regarding the best conservation strategies to apply in this and other similar populations.

1. Introduction

The southern European peninsulas housed many boreal species, which were isolated there after the retreat of the ice sheet that originated during the last glacial period of the Pleistocene Epoch (Hewitt, 1999). These marginal populations are known to be prone to extinction and genetically impoverished, as they tend to occur in less favourable habitats and at lower and more variable densities (Vucetich and Waite, 2003). Predation may compromise the persistence of such isolated populations, in the case of prey species (Macdonald et al., 1999).

Although many studies have demonstrated that predators may only have a considerable impact on those prey populations that live

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under unfavourable conditions (Genovart et al., 2010), many authors have stated that predators can, at least, limit prey populations living under a wider range of situations (Connolly, 1978; Sinclair et al., 1998). For example, hyperpredation occurs whenever there is an enhanced predation impact on an alternative prey species, due to an increase in predator numbers caused by either a rapid increase or a sudden drop in the abundance of its main prey (Moleón et al., 2008). Unintended supplementary feeding or subsidisation of predators by human activities (i.e. landfill sites, hunting and fishing remains) may play the same role in increasing the abundance of their main prey, leading to a numerical response by predators that can drastically affect alternative prey populations, especially when those are in low numbers (Gompper and Vanak, 2008).

To alleviate the effect of predation on vulnerable prey, many possible management strategies may be proposed. The restoration of the original top predator community (Gompper and Vanak, 2008), the elimination of subsidisation sources for predators (Bino et al., 2010; Tablado et al., 2010) or the improvement of shelter availability within prey habitat (Evans, 2004; Lombardi et al., 2007), have been widely invoked in conservation programmes. Although these strategies are more advisable because they aim at the long-term, continued increase or recovery of the target species, often other shorter-term strategies are carried out, sometimes because it is necessary to "buy" a few years before addressing the root of the problem (Igual et al., 2009), but often also because they are more visible to the donating public (Ludwig et al., 2001). Two of the most widespread examples of these strategies are predator control and the translocation or release of captive-bred individuals. However, both strategies may be controversial, due to ethical and practical problems (e.g. Beck, 1995; Reynolds and Tapper, 1996; Smith et al., 2010).

The positive effects of predator control tend not to be maintained over time, because density-dependent buffering processes tend to counter-balance the effect of removals (see e.g. Bosch et al., 2000; Smith et al., 2010). In addition, most studies have evaluated the effect of predator removal in terms of improved breeding success. However, whether the associated increase in productivity actually represents a relevant effect on the growth rate of the prey population is something that has seldom been assessed (Fletcher et al., 2010; Lavers et al., 2010). In relation to the reintroduction of captive-reared wildlife, recently this technique has become more popular (Seddon et al., 2007). However, the success of these programmes may be low when the causes of the original decline are not reversed at the time of the release or because captive-reared animals have not had previous exposure to local predators or food sources in the wild (Snyder et al., 1996; Wolf et al., 1998; Fischer and Lindenmayer, 2000).

In this paper, we analyse the trend of a relict population of a ground-nesting prey species, the capercaillie (*Tetrao urogallus*) in the Pyrenees, a species that is classified as "vulnerable" under Spanish law. We also examine different management scenarios for enhancing the population trend, using a population modelling approach, in an attempt to deal with the uncertainty surrounding its conservation. We use an integrative procedure that combines field-monitoring data from over 20 years and simulations, which accounts for environmental stochasticity. Specifically, we explore the effects of the release of captive-bred birds and of the removal of terrestrial mesopredators on a capercaillie population, since it is suspected that predation may be aggravating the situation of this isolated population, due to multiple and complex anthropogenic changes in a high-mountain environment. We have used available information from past experiences with captive breeding and from an ongoing terrestrial mesopredator removal experiment involving this population. We also present the current adaptive management framework for the conservation of this capercaillie population which should allow the update of our results in the near future.

2. Materials and methods

2.1. Study population and monitoring data

The capercaillie (*T. urogallus*) is the largest member of the grouse family in Europe and has been proposed as an 'umbrella' species (Suter et al., 2002). Two of the twelve subspecies described for the species (*T. u. cantabricus* and *T. u. aquitanus*) are present in the Iberian Peninsula. Both populations are located in the southernmost edge of their world distribution, and are geographically isolated both from one another and from the other European populations. More than the 80% of the known leks of the Pyrenean capercaillie are located in Catalonia, in the northeast of the Iberian Peninsula (Robles et al., 2006). The estimated number of males at leks has decreased by 31% in Catalonia since the early 1990s (op. cit.). The study population is located in the Pallars Sobirà region, which accounts for 35% of the known leks in Catalonia. Most of the study area is included in the Alt Pirineu Natural Park. The species mainly inhabits subalpine forests, located between 1500 and 2000 m a.s.l. They are dominated by mountain pine (*Pinus uncinata*), usually with a bilberry (*Vaccinium myrtillus*) and alpenrose (*Rhododendron ferrugineum*) shrub cover.

Empirical data consisted of "route censuses" (Rajala, 1974; Leclercq, 1987) from the summer monitoring programme, which has been carried out by the Catalan environmental agency since 1988. Censuses consists of a line transect of 8–18 people arranged at 10-m intervals, walking simultaneously through forest patches. At the time when the census is carried out, chicks are well grown and most of them are able to fly, so hereafter we will call them "fledglings". The number of flushed males, females and fledglings is recorded, as well as the area covered by each census. Undetermined birds are omitted from demographic parameter calculations. Sampling effort has exceeded 100 people per year (calculated by adding up the number of people who has participated in each census each year), since 1988.

2.2. Empirical estimation of population trend

The density of the adult birds (birds 100 ha^{-1}) was estimated by dividing the number of adult males and females by the surface covered in each census. The annual density was calculated as the mean value of all the densities of the censuses carried out each year. This is the first occasion in which the available time series of summer counts has been compiled to estimate the trend of the Spanish Pyrenean population. Nevertheless, the use of this type of density data to estimate population trends is common in other countries (see Leclercq, 1987). The total surface covered every year may affect the validity of the density estimates obtained, leading to non-representative estimates in those years in which the sampling effort was low due to logistic limitations. Preliminary linear regressions of density estimates with the surface covered each year determined that the minimum surface sampled to obtain density estimates irrespective of the surveying effort was 450 ha. Therefore, we used population densities from 1989 to 2010, filtering out years with a surveying effort of less than 450 ha, in order to estimate the population growth rate. Thus, we obtained a database that spanned 17 years. The finite annual population growth rate or deterministic growth rate (λ) was calculated considering N_t (the population size at time t) to be asymptotically proportional to $N_0\lambda^t$, and hence estimated λ is $\lambda = (N_t/N_0)^{1/t}$. Finally, in order to analyse the population trend, we performed a linear regression of population density and time so as to obtain the 95% confidence interval for the slope of the regression line. We tested the null hypothesis of population stability by checking whether the 95% confidence interval of the slope bracketed the value zero or not.