Does small-game management benefit steppe birds of conservation concern? A field study in central Spain

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Short title: Game management and conservation

Abstract

In Europe, hunting and game management are centuries-old activities that have had profound effects on the landscapes and the biodiversity they support. Game management is a potentially important tool to enhance game populations, and may also affect other wildlife. We investigated the relationship between red-legged partridge (*Alectoris rufa*) game management and steppe-bird abundance in central Spain. We surveyed 54 red-legged partridge hunting estates with varying game-management intensity in spring and/or summer in 2006 and 2008-2010. Information about game management was gathered through questionnaires to game managers. Birds were counted from fixed points and the amount of different land uses in each point was visually estimated. Our results show that the abundance of granivorous species (sandgrouse) increased significantly with the density of feeders, whereas non-granivorous species were more abundant in estates with more intensive fox control, although the latter relationship was weaker. Land-use variables, specifically agricultural lands, appeared in the models of all the groups considered. Therefore, there is an option for designing optimal management for red-legged partridge estates that potentially benefits non-target species: an effective combination of habitat management, adequate use of supplementary food and water, and possibly selective legal predator control when necessary and effective for the target game species, although the latter needs further investigation.

Keywords: bustards, game management, predator control, red-legged partridge, sandgrouse, small game, steppe birds, supplementary feeding.

Introduction

In Europe, sport hunting and especially small-game hunting is an important social and cultural tradition in many agrarian areas (Willebrand, 2009; Fischer *et al.*, 2013). Management to improve the abundance or access to game has been carried out for centuries and has had profound effects on landscapes and their associated biodiversity (Duckworth *et al.*, 2003; Pohja-Mykra, Vuorisalo & Mykra, 2005). Game managers currently apply a variety of different tools to maintain or increase harvest including predator control (Reynolds & Tapper, 1996), habitat management (Bro *et al.*, 2004), provision of food and water (Stoate & Szczur, 2001), and release of farm-bred animals (Laikre *et al.*, 2010). The use of these practices has become increasingly common among small-game managers (Martin, 2011), not only because many small-game species populations have suffered strong declines within their ranges (Bro *et al.*, 2000; De Leo *et al.*, 2004; Blanco-Aguiar *et al.*, 2012), but also because there is greater interest in extracting economic revenue from this activity (Martin, 2011).

Game-management activities are aimed to affect game species, but they may also affect other species inhabiting the same area, in a positive (Oldfield *et al.*, 2003) or negative way (Villafuerte, Viñuela & Blanco, 1998). An argument typically used in defence of hunting is that it can contribute to the conservation of species or habitats that are beneficial to biodiversity (Oldfield *et al.*, 2003). For example, in UK, areas managed for hunting red grouse (*Lagopus lagopus scoticus*) maintained more moorland habitat (Robertson, Park & Barton, 2001) and areas managed for shooting pheasants maintained more lowland woodlands than unmanaged areas (Duckworth *et al.*, 2003). In addition, there are some game-management tools used with the aim of allowing high densities of game species, such as the provision of supplementary food and water, which may also benefit some non-game species (Draycott *et al.*, 2005; Gaudioso Lacasa *et al.*, 2010). In

contrast, other game-management tools have been considered detrimental for conservation in certain circumstances. For example, predator control is frequently carried out as part of game management in some parts of Europe because hunters believe that predators limit game numbers (Valkama *et al.*, 2005; Holt *et al.*, 2008; Park *et al.*, 2008), but this practice causes conflicts with other stakeholders (Bro, Arroyo & Migot, 2006; Holt *et al.*, 2008; Thirgood & Redpath, 2008). Although some authors found that predator control might be beneficial for some non-target species (Suárez *et al.*, 1993; Fletcher *et al.*, 2010), illegal control of some predators of conservation concern still takes place, and this has at times led to their subsequent elimination from large areas (Etheridge, Summers & Green, 1997; Villafuerte *et al.*, 1998). Finally, where game management is intensive, practices such as restocking of game populations with farm-bred animals, or introducing new, alien species for hunting frequently occur. Such practices have been suggested to have a negative effect on biodiversity, e.g., through the introduction of new pathogens or the release of hybrids that could represent a threat for the long term survival of native species (Villanúa *et al.*, 2007; Sanchez-Donoso *et al.*, 2012).

Several studies have addressed the effect of hunting and small-game management on biodiversity, both on target (Tapper, Potts & Brockless, 1996; Aebischer & Ewald, 2004; Delibes-Mateos, Ferreras & Villafuerte, 2008; Casas & Viñuela, 2010), and on non-target species (Parish & Sotherton, 2004; Draycott, Hoodless & Sage, 2008; White *et al.*, 2008; Fletcher *et al.*, 2010). However, the latter have frequently focused on one management tool only [e.g., predator control (Suárez *et al.*, 1993; Smith *et al.*, 2010), game crops (Sage *et al.*, 2005), supplementary feeding and water (Gaudioso Lacasa *et al.*, 2010)], or simply compare managed to non-managed areas (Stoate, Borralho & Araújo, 2000; Beja *et al.*, 2009; Caro *et al.*, 2014a), which

makes it difficult to identify the relative effect of individual management tools on biodiversity when applied simultaneously. It is important to evaluate which of the management techniques employed to promote small-game species have a stronger effect on non-target fauna as this, along with an evaluation of their efficacy to increase game species populations, may be used to identify those management systems that combine the most efficient and beneficial (or least detrimental) techniques for wildlife in general. This could help us to shift the debate about whether hunting is beneficial for biodiversity conservation to recommending how to best use game management for the promotion of biodiversity.

In Spain, small-game species in farmland areas include birds such as red-legged partridges (*Alectoris rufa*) or quails (*Coturnix coturnix*), and mammals such as rabbits (*Oryctolagus cuniculus*) and hares (*Lepus sp.*). Small-game hunting in central Spain is very important both socially and economically, being practised in more than 85% of the whole territory (Ríos-Saldaña, 2010). Management for small game is also very widespread there, occurring on more than 90% of the hunting estates (Ríos-Saldaña, 2010). Moreover, central Spain is rich in species of conservation concern (Martí & del Moral, 2003). Thus, this region constitutes an ideal location to evaluate the relationship between game-management practices and the conservation of biodiversity.

We investigate the relationship between different game-management activities and the abundance of steppe birds of conservation concern in small-game hunting areas in central Spain. We focused on the main game-management practices employed to improve red-legged partridge populations, the main small-game species in our study area (Díaz-Fernández, Viñuela & Arroyo, 2012). We selected steppe birds as the study species because: i) they are important from a conservation point of view (Council Directive 2009/147/EC on the Conservation of Wild Birds); and ii) steppe birds share

ecological needs with partridges [i.e. ground-nesting species inhabiting open landscapes with morphological, physiological and behavioural resemblances (De Juana, 2005)]. Finally, we discuss how game-management practices could help to promote biodiversity conservation.

Methods

Study area and game-management data

We surveyed 54 hunting estates managed for partridge hunting in central Spain (Fig. 1). Estate size ranged from 2 to 280 km^2 (mean ± SD = 36.79 ± 54.70). Land was privately managed for hunting purposes and surveys were carried out with the approval of the person responsible for the game activity within the estate.

Hunting estates vary largely in the intensity of implementation of management practices (Arroyo *et al.*, 2012). Information about game management was gathered through semi-structured interviews to game managers, who voluntarily participated in the study. Interviews were mostly carried out before or immediately after the bird surveys (permission to carry out surveys was usually granted during the interview or while arranging that meeting). Therefore, game-management measures as reported reflect the management performed when birds were surveyed. For each estate, we obtained data on estate size, number of gamekeepers (converted to full-time equivalents such that two part-time gamekeepers were considered one full-time gamekeeper), number of red foxes killed annually the year prior to the interview, number of feeders (devices with grain or commercial feed for consumption by partridges, always refilled during spring and summer, and sometimes also in winter), and number of artificial water ponds. The latter were of two types: small and large. Small water points (less than

500 l) are water tanks maintaining a constant water level in an external small dish. Large water points are shallow artificial ponds containing more than 500 l of water and covering up to more than 100 m^2. Variables were expressed per estate surface area (Table S1 in Appendix S1) to express variation in game-management intensity. Additionally, there are differences related to the economic regime to which each estate adheres (*Hunting regime* in Table S1). These regimes are: i) non-commercial hunting estates (where the stated aim was to provide recreational hunting for hunting groups or societies, not economic profit; n = 12); ii) commercial estates (where the stated aim was to obtain economic benefit from the hunting rights; n = 36); and iii) intensive estates, a specific type of commercial hunting estate characterized by the legal release of farm-reared partridges throughout the hunting season, higher frequency of driven-shooting days, higher revenues, and management practices that are more intensive than in other estate types (n = 6). More details in Díaz-Fernández *et al.* (2012) and Arroyo *et al.* (2012).

Steppe-bird surveys and land-use variables

Field surveys were carried out in spring and/or summer in 2006 and 2008-2010. Data were recorded using point-count methods (Bibby, Burgess & Hill, 1992): observers drove along tracks distributed throughout the whole of the estate or, when it was too large (n = 2 estates with more than 200 km^2), through a third part of the estate stratifying by habitat (where tracks were selected on aerial photographs so that the surveyed area had the same proportion of open and close habitats as the whole of the estate). Every 700-750 m observers stopped, and the steppe birds (Table 1) observed (using binoculars) during 10 minutes were recorded. The number of points assessed at each estate was 59 ± 57 (range 4-420), depending on estate surface area. Observations

took place in the early morning (sunrise to three hours later) and in the evenings (three hours before sunset), avoiding the hottest central hours of the day when bird activity is the lowest (Bibby *et al.*, 1992). We calculated the total number of individuals observed (hereafter called 'abundance').

We also sampled the availability of different land uses in each estate to control the effect of habitat on steppe-bird abundance. We visually estimated the cover of each land use in a circular buffer of 250-300 m radius around each observation point. Each land-use type (Table 2) within that distance was noted as a percentage and we then obtained the average of all observation points as an estimate of land-use availability in the hunting estate.

More details about the bird and land-use surveys are included in Appendix S2 in Supporting Information.

Statistical analyses

We observed five different steppe-bird species in the study area (Table 1): black-bellied sandgrouse (*Pterocles orientalis*), pin-tailed sandgrouse (*Pterocles alchata*), little bustard (*Tetrax tetrax*), great bustard (*Otis tarda*) and Eurasian thick-knee (*Burhinus oedicnemus*). We obtained two different response variables related to steppe-bird abundance in each hunting estate: abundance of granivorous steppe birds (as the sum of observations of both sandgrouse species), and abundance of non-granivorous steppe birds (the sum of observations of the other three species, Table 1). We did this classification because the two groups of species differ in their diet (Herranz & Suárez, 1999; Lane *et al.*, 1999; Green, Tyler & Bowden, 2000; Jiguet, 2002) and may respond differently to grain supply; therefore our prediction was that feeders (providing grain for

partridges) would have a stronger effect on granivorous than on non-granivorous species.

We reduced the five land-use categories into two orthogonal factors using a Principal Component Analysis [PCA; e.g. (Caro *et al.*, 2014a)]. Multicollinearity between these components of land use and game-management variables was assessed using Spearman's correlation coefficients (rho) and the variance inflation factor (VIF).

Statistical analyses were performed in R 3.0.3 (R Core Team, 2014) through the interface of RStudio (RStudio, 2012). Given that some of the hunting estates were surveyed twice (in spring and summer, see Appendix S2), we had repeated measures of the independent variables in those estates, sometimes in different years. For this reason, we performed generalized linear mixed models (GLMMs) and included year nested in hunting estate as a random variable. We included season as a fixed variable. Our abundance response variables are count data and fitted a Poisson distribution, therefore we used a GLMM with a Poisson error distribution, and used the number of surveyed points in each estate ($\log_{10}$ transformed) as an offset. To control the potential effect of the different survey effort (especially in very large estates), we used the log-number of observation points divided by the area of the estate as a weighting term in the analyses, thus giving more weight to the observations from estates that were sampled more thoroughly. We assessed whether models were affected by overdispersion, accepting dispersion parameter levels below 1.5 (Zuur *et al.*, 2009). Models were calculated with the function *glmer* [library *lme4* (Bates, Maechler & Bolker, 2013)].

We were interested in knowing if game-management activities influenced non-target species abundance, whilst controlling for habitat effects. Therefore, we performed a general model which included season, the first two axes of the land-use PCA and game-management variables (Tables 2 and S1). All game-management variables

considered (and thus all possible combinations of them) could theoretically influence steppe-bird abundance. For example, predator control could benefit steppe birds as they are ground-nesting (Fletcher *et al.*, 2010). Additionally, food or water provided for partridges could also benefit steppe birds, as occurs with other bird species (Gaudioso Lacasa *et al.*, 2010). The number of gamekeepers could be related to steppe-bird abundance in two different ways: positively, as more gamekeepers could imply more feeders or predator control but, at the same time, more keepers could disturb steppe birds (Sastre *et al.*, 2009), expecting a negative relationship. Finally, we also included type of hunting regime, because we wanted to know if the economic interests of the estate, and the related intensity of game management and releases (Arroyo *et al.*, 2012), had an effect on the abundance of each group considered.

We performed all possible combinations of the independent variables with the function *dredge* [library *MuMIn* (Bartón, 2012)], as we wanted to specifically evaluate the relative importance of different game-management variables when considered together with habitat related ones. We then calculated an averaged model with all the models that presented a difference of AIC_c (ΔAIC_c) < 2 (Burnham & Anderson, 2002) to obtain the relative importance of each variable in the selected models. We performed the analyses with the estates that have information for all game-management variables, thus our final database included 66 records from 47 estates.

Results

The PCA with land-use variables produced two orthogonal axes which together accounted for more than 85% of the variance (Table 2). The first axis mainly reflects a gradient from hunting estates with more agricultural land to estates with lower proportion (lower PC1 values indicate higher proportion of agricultural lands). The

second axis contrasts encroached Mediterranean scrubland *versus* open and managed oak woodland (‘dehesas’).

The number of small water points and the number of feeders in each estate were highly correlated (rho = 0.85), as they are usually placed together. For this reason we only used artificial feeders and ponds (large water points) in the analyses (Table S1). No other pair of our land use (PCA axes) or game-management variables presented a rho correlation value higher than 0.5, and our models were not affected by multicollinearity between the variables (maximum VIF value was 1.22). Additionally, none of our abundance models were affected by overdispersion.

Best general models for the different response variables considered ($\Delta AIC_c < 2$) are shown in Table S2 in Appendix S1. Table 3 summarizes model-averaged coefficients of those models. Both steppe-bird groups showed a positive correlation with some game-management activities, in addition of being affected by land use. The abundance of granivorous species increased significantly with the density of feeders. Non-granivorous steppe birds were more abundant in estates with more intensive fox control, although this relationship was not significant when considering the 95% confidence interval of the parameter estimate, as the lower limit of interval was just under zero. Abundance of both groups was positively and significantly influenced by the availability of agricultural lands while non-granivorous species seemed to benefit also from the presence of Mediterranean shrubs, although the latter relationship was weaker (Table 3).

Discussion

One objective of the European Union Treaty is the promotion of game management as a form of sustainable development (Brainerd, 2007). In Europe, millions of hectares of land are currently managed for hunting, and therefore it is essential to identify which game-management practices can positively contribute to the conservation of wildlife and their habitats. In this study, we found a positive correlation between the provision of supplementary feeding and the abundance of granivorous steppe birds (sandgrouse). Given that in our study area feeders are placed together with small water points, it is not possible to entirely disentangle the relative benefit of food or water for these steppe birds. However, since the effect of feeders was only apparent for granivorous species, it is likely that the provision of food itself is beneficial. This management activity has also been found to benefit partridge productivity in the same area (Díaz-Fernández *et al.*, 2013), which would suggest that food or water are potentially limiting resources for these farmland species. Stoate & Szczur (2001) suggested that supplementary feeding for game species increased breeding densities and winter survival of omnivorous and granivorous passerines in some farms of Great Britain. In northwest Spain, Gaudioso Lacasa *et al.* (2010) observed that non-game bird species also used water troughs designed for game species, suggesting that the provision of water itself may also be beneficial. In any case, the way in which food or water supplementation benefit steppe birds in Mediterranean farmland regions, where water could be a limiting factor during summer, are still unclear, and require further investigation.

Predator control is frequently used by hunters to increase the numbers of game species in central Spain, being foxes the most common legally controlled predator (Delibes-Mateos *et al.*, 2013). Its use is controversial, since conservationists usually argue that it detrimentally affects some predators of conservation concern (Thirgood & Redpath, 2008), while some authors have shown that predator control may benefit some

species of conservation concern [see a review in Smith *et al.* (2010)]. Regarding steppe birds, predator control has contributed towards the stabilization of bustard populations in central Europe (Faragó, Giczi & Wurm, 2001). In our study, fox control was included in the best model explaining the abundance of non-granivorous steppe-bird species in central Spain (Table S2) and its relative importance in the averaged model was high (Table 3). The relationship between fox control and abundance was apparently positive (although the lower limit of the 95% confidence interval for the parameter estimate of this variable was just under zero). If confirmed in further studies, the positive correlation between abundance of non-granivorous steppe birds and predator control could indicate higher mortality or breeding failure in areas without fox control, or that birds disperse from areas with higher fox densities (Suárez *et al.*, 1993). Nevertheless, the lack of significance of predator control in our results indicates that we have to be cautious regarding the potential positive effects of fox control on non-granivorous species. Interestingly, the relationship of predator control with granivorous species was less important than for non-granivorous species. A possible explanation of this result is that the number of feeders is more important for granivorous species than any other game-management variable considered in the analyses, so the contribution of predator control to the model (once including feeders) is very low, or that this group of species is less responsive behaviourally or demographically to the presence of predators. In any case, even if there is potential for predator control to have benefits for some vulnerable bird populations (Smith *et al.*, 2010), this requires a whole-ecosystem view to avoid unforeseen negative effects. For example, removal of some predators can result in changes in intraguild interactions and outbreaks of other predators, leading in turn to increase predation on prey (Ritchie & Johnson, 2009). This may be particularly important in predator-rich communities like those observed in central Spain (Virgós &

Travaini, 2005). Therefore, if fox densities are high, fox control using legal and selective methods might be a complementary conservation strategy to help recover certain steppe-bird populations in central Spain, but additional research is needed to confirm this fact and to evaluate how fox control may affect predator communities (Virgós & Travaini, 2005).

Density of gamekeepers appeared in the model explaining abundance of non-granivorous species, but the coefficient confidence interval included zero and its relative importance was very low. Keepers can be theoretically related to steppe-bird abundance both positively, implying for instance more positive management not considered in our analyses, or negatively, as birds might be more disturbed with higher human presence (Sastre *et al.*, 2009); thus, the overall effect of this variable may be diluted if both aspects happen simultaneously.

Over the past decades, the release of farm-bred partridges for shooting has exponentially increased in central Spain (Caro *et al.*, 2014b). The spread of diseases and parasites by farm-reared partridges has not only affected wild partridge populations (Díaz-Sánchez *et al.*, 2012), but also steppe birds (Villanúa *et al.*, 2007). In this sense, a lower steppe-bird abundance could be expected in intensive estates, where there are more gamekeepers that can cause disturbance and massive partridge releases take place (Arroyo *et al.*, 2012). However, this was not observed in our study. A possible explanation is that these estates are also associated with more intensive supplementary feeding and fox control (Arroyo *et al.*, 2012), both of which may be, in themselves, beneficial to steppe birds. In any case, further research is needed to assess the relationship between the release of farm-reared partridges for shooting and steppe birds.

Finally, land-use variables were very important in explaining the estimated abundance of all groups considered. In general, steppe birds were more common in

hunting estates with more surface of agricultural lands, which was expected because these are their preferred habitats in the western Palaearctic (Suárez, Naveso & De Juana, 1997). These findings suggest that habitat conservation and management in hunting estates could be very important for improving the biodiversity value of these areas (Robertson *et al.*, 2001; Duckworth *et al.*, 2003), especially if applied in intensive agricultural lands (Bretagnolle *et al.*, 2011). For example, the maintenance of nesting and foraging habitats in Portugal for red-legged partridges seems to have favoured populations of the corn bunting (*Miliaria calandra*) (Stoate *et al.*, 2000), which share habitat requirements with game birds. In agricultural areas within central Spain, habitat management actions also seem to be favourable for game and non-game species (Delibes-Mateos *et al.*, 2008; Buenestado *et al.*, 2009). In these areas, actions aimed at increasing the proportion of natural vegetation within the farmland matrix (such as Mediterranean scrub or the maintenance of grassy field edges) is likely to be beneficial for both partridges (Buenestado *et al.*, 2009; Casas & Viñuela, 2010) and overall farmland biodiversity (Reino *et al.*, 2009; Vickery, Feber & Fuller, 2009). Indeed, a positive (albeit not significant) relationship has been found between the PCA factor positively related to Mediterranean shrubs and the abundance of non-granivorous steppe birds (Tables 2 and 3). As land-use variables were the most relevant predictors explaining abundance of both studied steppe-bird groups (Table 3), changes in landscape characteristics, habitat availability and agricultural practices could be key elements for the conservation of these species (White *et al.*, 2008).

Maximizing hunting outputs and conservation of farmland birds

Our study was not experimental, and thus it is possible that observed effects of management are confounded by other (non-evaluated) environmental variables.

Additionally, the survey methods used may be associated with certain biases (see Appendix S2). However, our analyses control for potential biases, and our results suggest that both the conservation of farmland habitats and certain game-management activities (i.e., density of feeders and possibly fox control) might be particularly relevant in our study area for the conservation of steppe birds. Interestingly, our results show that not all of the game-management tools affect in the same way steppe-bird abundance, a result also found for wild red-legged partridge summer density (Díaz-Fernández *et al.*, 2013). Thus, there is a potential for improving management in order to optimize both red-legged partridge hunting and also benefit steppe birds of conservation concern: an effective combination of habitat management, an adequate use of supplementary food and water, and possibly a selective legal predator control when necessary and effective for the target game species (supported by further investigations). This could mean a sustainable natural resource management approach promoting both nature conservation and socio-economic benefits to rural communities.

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Table 1. Species considered in the analysis. Total: total number of individuals observed in the surveys. Mean: Mean number of individuals in estates with presence of the species. SD: standard deviation.

Species	**Total**	**Mean**	**SD**
Granivorous steppe birds			
Black-bellied sandgrouse *Pterocles orientalis*	122	7.18	6.00
Pin-tailed sandgrouse *Pterocles alchata*	963	26.75	28.43
Non-granivorous steppe birds			
Little bustard *Tetrax tetrax*	900	28.13	48.28
Great bustard *Otis tarda*	575	19.83	29.42
Eurasian thick–knee *Burhinus oedicnemus*	434	9.86	9.28

Table 2. Summary results of the Principal Component Analysis of land use variables.

Variables	**Code**	**PC1**	**PC2**
% Agricultural land (mainly arable land with herbaceous crops or fallows, either ploughed or with vegetation, sometimes also small vineyards or olive groves)	Agri	-0.811	-0.054
% 'Dehesa': herbaceous crops with oaks	Dehesa	0.446	-0.70
% Mediterranean shrubs	Shrub	0.377	0.712
% Grasslands	Grass	-0.027	0.010
% Woodland	Wood	0.003	0.003
Cumulative explained variance		56.5%	85.4%

Table 3. Model-averaged coefficients for all variables included in the models with $\Delta AIC_c < 2$.

	Ab. granivorous steppe birds					**Ab. non-granivorous steppe birds**				
Variable	**β**	**SE**	**RVI**	**-CI**	**+CI**	**β**	**SE**	**RVI**	**-CI**	**+CI**
Intercept	-0.008	0.305				0.205	0.323			
PC1	-0.033	0.008	1	-0.049	-0.018	-0.048	0.011	1	-0.071	-0.025
PC2						0.030	0.019	0.73	-0.009	0.069
Feeder	0.023	0.01	1	0.004	0.043					
Ponds	0.227	0.181	0.32	-0.135	0.590					
Fox	0.036	0.052	0.19	-0.067	0.140	0.074	0.038	0.75	-0.003	0.150
Keeper						1.074	1.334	0.13	-1.596	3.744

Ab: Abundance; β: coefficients; SE: standard errors; RVI: relative variable importance. –CI and +CI: Confidence limits for coefficient estimates at the 95% confidence interval (CI). Variable codes as in Table 2 and Table S1. In bold, game management variables.

Figure 1. Municipalities (light grey) where hunting estates were located and their situation in mainland Spain (top left).