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Is the wild rabbit (*Oryctolagus cuniculus*) a threatened species in spain? Sociological constraints in the conservation of species

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Abstract The Wild rabbit (*Oryctolagus cuniculus*) is an endemic species of the Iberian Peninsula and is essential for the conservation of endangered predators. Rabbits are also of high importance as a hunting species. From 1988, rabbits suffered the severe effects of rabbit hemorrhagic disease, which caused large declines in most populations. Despite this fact, the National Red Data Lists continued to classify rabbits as a "Least Concern" species. We used available hunting bag data from 1973 to 2002 to model national trends of rabbit abundance and to evaluate the conservation status according to the criteria of the National Red Data List and the World Conservation Union (IUCN). Generalized Additive Models were used as the statistical framework. The rabbit population of Spain suffered a large decline of about 71% between 1973 and 1993. This decline was 49% in the period 1980–1990. Based on both Spanish and World Conservation Union criteria, rabbits should be listed as 'Vulnerable', which demands a Conservation Plan Program. We suggest that the lack of concordance between the best available evidence and the conservation status of the species is a consequence of sociological constraints in conservation decisions. Rabbit conservation could face strong opposition from important socio-economic lobby groups (hunters and farmers). As such, governments and researchers may prefer to exclude rabbits from any status category requiring conservation action, despite the evidence of decline. We call for the urgent development of a nation-wide conservation program for rabbits which includes both socioeconomic constraints and the available biological data on population trends.

Keywords Decline · GAM · Haemorrhagic disease · Hunters · Population trends · Rabbit · Spain · Threatened species

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Introduction

The Wild Rabbit (*Oryctolagus cuniculus*) is an endemic species of the Iberian Peninsula (Thompson and King 1989; Monnerot et al. 1994), although it has expanded in range in Europe and has been introduced to many other parts of the world. The species is now considered a pest in many places and most management strategies have been directed towards population control (Drollette 1996). In its native range including Spain, however, the rabbit may be considered a key element of the ecosystem, and therefore of high conservation value. In fact, rabbits represent the bulk of the diet of many Iberian predators (Delibes and Hiraldo 1981), and are the key prey species of several critically endangered species, including the Iberian lynx (*Lynx pardinus*) and Iberian Imperial Eagle (*Aquila adalberti*) (Ferrer and Negro 2004).

Added to their functional importance in Spanish ecosystems, rabbits often arouse polarized social and economic interests. On the one hand, rabbits are traditionally one of the most important game species in Spain (REGHAB 2002) and therefore hunters have an interest in keeping their numbers high. On the other hand, the species causes significant agricultural damage and so farmers call for rabbit populations to be controlled or eradicated.

Although rabbits are historically numerous and widespread on the Iberian Peninsula, the introduction of myxomatosis into the wild in the 1950s caused a significant drop in their population (Muñoz 1960; Thompson and King 1989). In the following years rabbits attained some natural immunity to the disease, but in 1989 a new viral epizootic, rabbit hemorrhagic disease (RHD), reached Spain and rabbits suffered mortality close to 80% in most populations (Peiró and Seva 1991; Blanco and Villafuerte 1993; Villafuerte et al. 1995; Calvete et al. 2002). From the first appearance of RHD to 1993, the decline in the rabbit population was estimated to be around 55% based on a questionnaire survey answered by hunters (Blanco and Villafuerte 1993). Despite such evidence, the National Red Data List based on the Red Data Book listed the rabbit as a ''Least Concern'' species (Blanco and González 1992).

It now appears that the newest edition of the National Red Data List (currently in preparation) will again list the rabbit within the ''Least Concern'' category. At present the only available population decline estimate on a national scale comes from the survey conducted by Blanco and Villafuerte (1993), and updated information based on a rigorous monitoring program of rabbit populations is lacking.

Monitoring abundance trends of wildlife populations is an essential component of any management program (Geissler and Sauer 1990; Thomas and Martin 1996; Fewster et al. 2000), and should be used to evaluate the current status of the species for its potential inclusion on national or international red lists of endangered species. The lack of nation-wide survey schemes for common species is a problem in many wildlife monitoring programs (Barker and Sauer 1992; Thomas and Martin 1996).

National monitoring programs are usually based on indirect measures of abundance such as count estimates along fixed routes (Droege and Sauer 1990; Fewster et al. 2000). However, game species show an additional advantage because hunting bag data can be used as a proxy of abundance across time (e.g. Potts et al. 1984; Baines and Hudson 1995; Cattadori and Hudson 1999; Kitson 2004). The best-known example comes from fisheries, where management is primarily based on catch data (Hilborn and Walters 1992). One major concern, however, is that harvest records do not reflect true variations in species abundance (Gilpin 1973), although some studies indicate that correlations may be good (Royama 1992; Cattadori et al. 2003; Kitson 2004). Hunting statistics in Spain are available from 1973, and these may be the best data available for the determination of temporal dynamics and trends of game species across the country. Unfortunately, they have been misused by managers and researchers in Spain.

In addition to the lack of efficient survey methods and adequate data for the analysis of population trends, several problems are related to the selection of the analytical procedures to determine the magnitude and the direction of the trend, two essential pieces of data from the perspective of monitoring. As a consequence, a diverse and complex range of methods has been employed (reviewed in Thomas and Martin 1996; Fewster et al. 2000). The recent use of generalized additive models (GAMs) has shown flexibility and robustness when analyzing abundance trends of bird species over different source data and other monitoring features (Fewster et al. 2000).

In this paper, we describe for the first time the population trends of rabbits in Spain during the period 1973–2002. We used available hunting records as the source of abundance data across years and GAMs as the statistical framework to analyze trends. This information was used to identify the true conservation status of wild rabbit populations in Spain according to IUCN Red List categories guidelines and principles (IUCN 2001). We also discuss the sociological context of rabbit conservation in Spain and its potential solutions.

Methods

Hunting statistics

We obtained hunting bag data from the Ministry of Agriculture and Fisheries of Spain. This information is free and at the time of this study was available from 1973 to 2002. Data are based on questionnaires that all owners of hunting lands are required to complete each year. Ministry of Agriculture obtained a summary value of the bag record in each province by summing the data from each hunting land in each province for each year. The number of rabbits hunted is recorded for all Spanish provinces (n = 50). Provinces that had records for less than 20% of years were excluded from the study. We did not include hunting records from the Balearic or Canary islands because our aim was to check population trends in the area where rabbits are endemic and not in those areas where it may be considered an invasive species (Thompson and King 1989). Our final data set was drawn from 45 provinces from across the country. We used a relatively long time series for the analysis, starting in 1973, approximately 20 years after the first myxomatosis outbreak (Muñoz 1960). The time series ended in 2002, 14 years after the first detection of RHD in Spain (Argüello et al. 1988). Because we used bag records for each year in the series as the data points in the analyses, our study was based on data for 30 years.

Hunting statistics may be considered a good description of rabbit populations in each province because hunting lands cover 70–75% of Spain, with similar figures in all provinces (REGHAB 2002). Because the number of rabbits hunted may be partially due to hunting effort, this factor needs to be controlled in the analysis of population numbers (Bostford et al. 1983; Cattadori et al. 2003) by using the number of hunting licenses issued as a measure of hunting effort in each province and in each year.

In all provinces and across the study period, rabbit hunting was regulated based on annual quotas proposed by hunters each year on the basis of the hunting bag of the preceding year. We thus considered that hunted rabbits were a suitable proxy of rabbit abundance in the hunting lands of Spain. In some cases, hunters imposed limitations in the form of restricted quotas, but these limitations were not related to rabbit abundance (Angulo and Villafuerte 2003). This is important because harvesting theory predicts that when density is low hunters tend to reduce quotas, which can lead to underestimations of population size. In contrast, when density is high hunters tend to increase quotas and so true numbers could be overestimated (Gilpin 1973; Cattadori et al. 2003). Both factors could influence trend estimates, but the lack of correlation between rabbit abundance and restricted quotas in Spain may minimize this source of bias.

Analyzing population trends by fitting generalized additive models (GAMs)

Most analyses of population trends have been conducted using route regression, Poisson regression or Mountford methods (e.g., Geissler and Sauer 1990). However, these methods cannot be used to extract genuine changes or trends from noisy data and they are ill suited to the investigation of nonlinear population trends (Siriwardena et al. 1998; Fewster et al. 2000). These problems can be overcome by using smoothing algorithms applied to abundance estimates (Fewster et al. 2000). Several of these smoothing techniques have been applied to bird census data both in North America and the United Kingdom (James et al. 1996; Siriwardena et al. 1998). GAMs extended these procedures, allowing both the incorporation of smoothing algorithms and a comprehensive modeling framework (Fewster et al. 2000). GAMs are an extension of generalized linear models (see Hastie and Tibshirani 1990) that allow any shape in the curve of the abundance estimates, through a range of non-parametric models (Fewster et al. 2000), and are thus a very flexible way to model how a response variable (e.g., population abundance) changes in response to several predictors (e.g., year and site). In a GAM, the level of smoothing (i.e., roughness) of the curves is defined in the model by the degrees of freedom (Hastie and Tibshirani 1990; Fewster et al. 2000). Our model took the following form:

$$\text{abundance index} = \text{site effect} + \text{smooth (year)}$$

For a comprehensive explanation of the procedures used to calculate the abundance index of the counts and the methods of GAM fitting, see Fewster et al. (2000). The site predictor may be of little biological interest but it has been incorporated into the model because abundance is likely to differ between sites. In our study the main predictor is year, which is represented as a smooth curve with predetermined degrees of freedom. We fitted curves based on splines (Hastie and Tibshirani 1990; Fewster et al. 2000), which were chosen to satisfy a penalized least square criterion, optimizing fit but penalizing roughness (Fewster et al. 2000). We used the recommendations of Fewster et al. (2000) and in our modeling framework we set the degrees of freedom to be 0.3 times the number of years in the time series, in our case 10.

In contrast to count data used in other monitoring programs (e.g., Geissler and Sauer 1990; Fewster et al. 2000), our model was based on a response variable in the form of an index (bag record/hunting effort). The data were log-transformed and then fitted to a GAM with Gaussian errors and identity link (Hastie and Tibshirani 1990). To evaluate whether population changes observed in the data were statistically significant, we calculated confidence intervals for abundance index estimates from the GAM curves by bootstrapping (Fewster et al. 2000). We performed 399 replicates with replacement of the data for each year and fitted a GAM curve for each replicate, thus obtaining a series of abundance index replicates which were used to obtain 95% confidence intervals following the methods of Buckland (1984) and Fewster et al. (2000).

Goodness of fit testing of our model was performed using plots of residuals and standard errors. We used the partial deviance residuals which measured the difference between the log-likelihood of the fitted model and the saturated model (Hastie and Tibshirani 1990). We considered a model as satisfactory when the residuals were distributed evenly above and below the fitted curve and the standard errors were small (Fewster et al. 2000).

We also calculated significant turning points in the time series using the methods of Siriwardena et al. (1998) and Fewster et al. (2000). We computed second derivatives (rates of change of the rates of change) of our abundance index estimates and their 95% confidence intervals from the 399 bootstrapped replicates. We calculated second derivatives from the 6th differences in equation 11 from Fewster et al. (2000). We also used a value of $r = 1$ (window size) in Equations 9, 10 and 11 from Fewster et al. (2000). Those years that did not contain a zero may be considered as years with a significant curvature in the trend. These turning points are an objective measure of significant decline or increase years and they may also be used to divide the time series into parts with similar trends (Siriwardena et al. 1998; Fewster et al. 2000). Because these turning points are associated with changes in the rate of change (e.g. changes in population growth) rather absolute values of abundance index, it is possible to detect significant decline although absolute values of abundance index did not reflect a clear population reduction.

For all analyses we used S-Plus code from Dr. R. Fewster with some modifications to accommodate Gaussian errors instead of Poisson errors. Analyses were run in S-Plus 2000 for Windows (Mathsoft Inc. 1999).

Assignation of a threatened category

In Spain, the conservation of species is based on the criteria established in the national list of threatened species (hereafter CNEADI, the Spanish acronym). This list only recognizes two threatened categories: Endangered (EN) and Vulnerable (VU). We used our data to evaluate rabbit status according to the criteria of the CNEADI.

We also evaluated rabbit status for the three threatened categories established by the IUCN: Critically Endangered (CR), Endangered (EN) and Vulnerable (VU) (IUCN 2001). Each level of threat (CR, EN or VU) is defined by a range of quantitative criteria, and a taxon that meets any one of the criteria for a threat level is assigned to that level.

Results

After controlling for the number of hunting licenses, the number of rabbits hunted ranged between 0 and 97.68 (mean ± SE: 5.13±0.23 considering data from all provinces and years). The highest number of rabbits hunted (averaged over all years) was found in Toledo, a province of central Spain (mean ± SE: 23.81±4.09) and the second highest was found in Cádiz, in southern Spain (mean ± SE: 22.26±1.43) (Fig.1). The two lowest mean abundances were found in two northern provinces: Vizcaya (mean ± SE: 0.0020 ± 0.0009) and Guipúzcoa (mean ± SE: 0.017 ± 0.003). Figure 1 shows the pattern of rabbit abundance in Spain over the entire 30-year period considered. Overall, the highest number of rabbits hunted were found in central and southern Spain (Fig. 1).

Figure 2 shows the GAM trends for the abundance index with six different degrees of freedom: 3, 6, 10, 12, 16 and 20. Visual inspection of the five curves reveals that our 'a priori' selected condition of 10 degrees of freedom was indeed the best choice. The curve generated using 10 degrees of freedom revealed new features of the trends compared to 3 and 6 degrees of freedom, without being too rough. Values above 10 degrees of freedom generated rougher patterns, and as a consequence they lacked the smooth features that help to characterize true trends.

Diagnostic plots (Fig. 3) showed good fits of all GAMs with well distributed residuals around the index curve and very narrow confidence intervals. For brevity, here we only show the partial deviance residuals for the overall data.

Considering all regions together, rabbit populations in Spain suffered a major decline of about 71% when the years with the highest and lowest abundance indexes were compared (1973 vs. 1993) (Fig. 4). The series shows three significant negative turning points detected by the values of second derivatives of abundance index

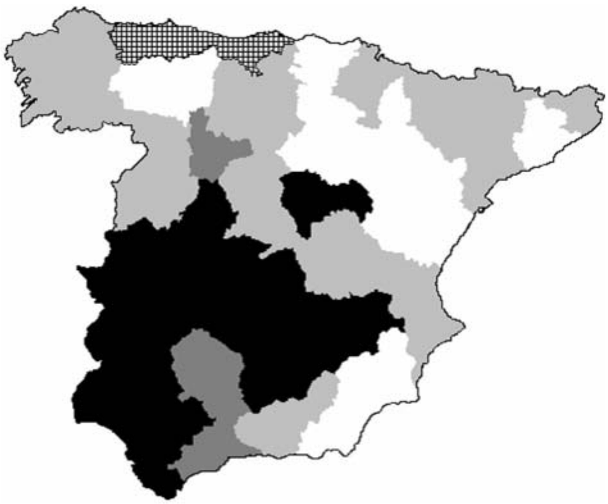

Fig. 1 Map of rabbit abundance in Spain based on the distribution of mean values of hunting bags (controlled by the number of licenses issued) from 1973 to 2002. Colored areas represent different values according to quartile distribution. Black and white areas represent the highest and lowest values, respectively (first and last quartiles of the distribution), while grey areas represent medium values of abundance (higher according to intensity)

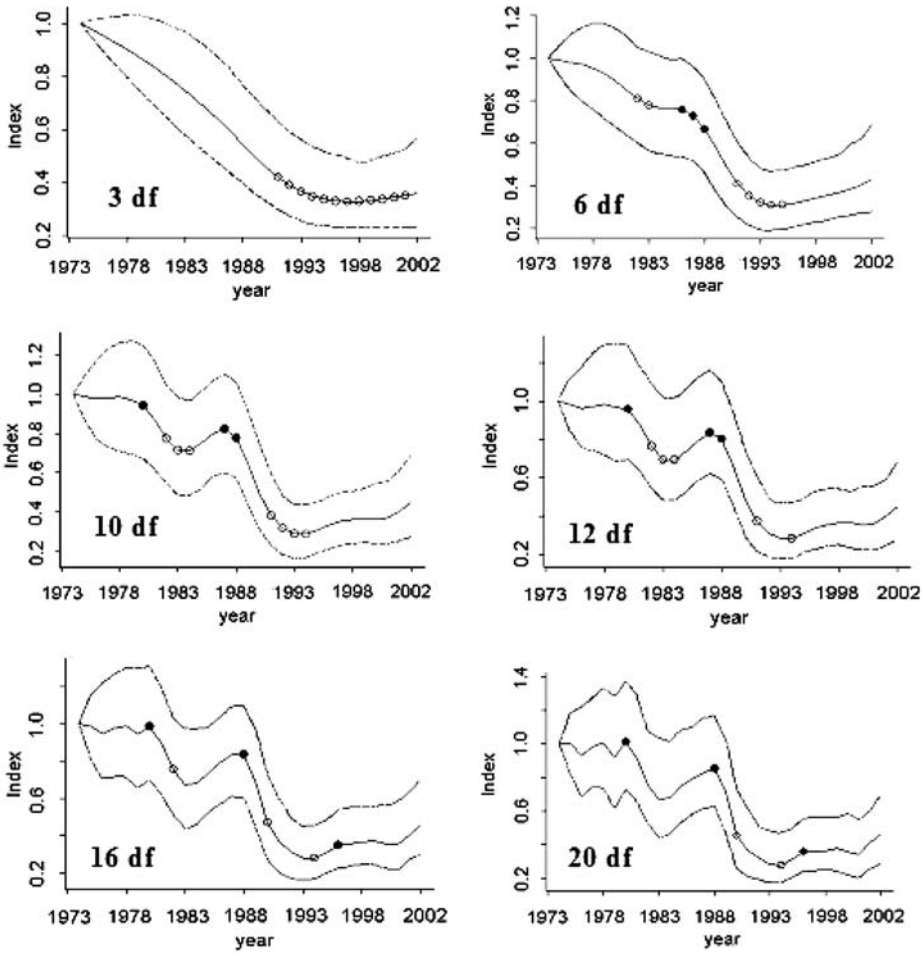

Fig. 2 Abundance index GAMs with different degrees of freedom fitted to hunting bag records from 1973 to 2002 in Spain

(Fig. 4, Table 1); the first was in 1979–1980, when a 13% decline between years was observed, but was followed by a significant positive turning point in 1982. During the period 1982–1987, and previous to the second significant negative downturn, rabbit populations increased. During 1987–1989 rabbit populations suffered a second crash, reaching minimum levels in 1993. The decrease between 1987 and 1993 totaled 48%, a very significant reduction in only 6 years, and a similar decrease was observed for several 10-year periods of the series. For example, a 49% decline was observed during the period 1980–1990 (a 10 year period very close to the publication of first Red Data Book in Spain). It is interesting to note that although a significant downturn was observed in 1987–1988 (a 13% decline from the previous year), the largest decline for any two years when absolute values of the abundance index were inspected was observed between 1988–1989, with a 15.5% reduction.

From 1993 to 2002 rabbit populations showed a slow but continuous recovery, although in 2002 abundance was still 55% lower than in 1973. Although the major

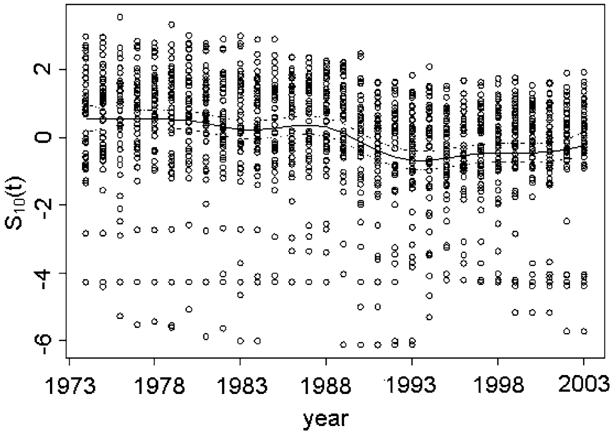

Fig. 3 Residual partial plot for the year effect curve $S_{10}(t)$ from a GAM with 10 degrees of freedom fitted to hunting bag records controlled by the number of licenses issued. Shown are 2 SE bands (dashed line), partial deviance residuals (black dots) and a rug plot. The figure shows the balanced distribution of the residuals around the index curve and very narrow confidence intervals

declines were mainly observed in the late 1980s, we noted that the decline was constant from 1973, with a 22.2% decline observed up to 1987.

We next fitted different GAMs to different regions with the aim of testing the generality of the decline pattern. We selected four regions: Galicia (NW Spain), Castilla-La Mancha and Madrid (Central Spain), Valencia and Murcia (E Spain) and Andalucia (S Spain). The populations of north-west Spain suffered declines similar to those observed in the overall pattern, although less pronounced, and we also observed a pronounced decline in the last 5 years (Fig. 5a). In these populations, the relatively large confidence intervals probably explain the non-significant nature of the downturns observed in the years 1979, 1987 and 1997, when detected declines for

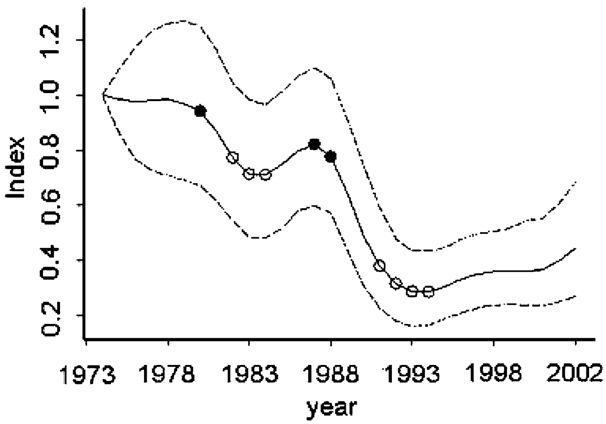

Fig. 4 Index curve showing rabbit population trends in Spain during the period 1973–2002. The solid line shows the abundance index from a GAM with 10 degrees of freedom. Dashed lines represent upper and lower 95% bootstrapped confidence intervals. Full circles correspond to significant negative turning points and empty circles represent significant positive turning points based on second derivatives of the index curve

Table 1 Significant upturns (U) and downturns (D) of rabbit populations in the study period based on the second derivatives of abundance index (rate of change of the rate of change of the absolute abundance index)

Region	1978	1980	1982	1984	1988–1989	1990	1993	1996	1998–1999
All Spain		D	U	U	D	D	U		
Galicia (N Spain)			U					D	
Central Spain	D		U		D	D	U		D
Valencia + Murcia (E Spain)				U	D		D		
Andalucia (S Spain)							U	D	

absolute values of the abundance index were around 12–15% between consecutive years (Fig. 5a).

In eastern Spain, the pattern was different to the overall trend described. We observed a significant downturn in 1978, but the population showed a significant upturn in 1983 (Fig. 5b, Table 2). In the late 1980s, a significant downturn based on second derivatives of the index was observed, but this fact was not associated to a significant decline in the absolute values of the abundance index (Fig. 5b). Although not significant based on turning points (but this may reflect a very large confidence interval, see Fig. 5b), populations in eastern Spain suffered a strong decline in the absolute value of abundance index from 1998 to 2002 (Fig. 5b).

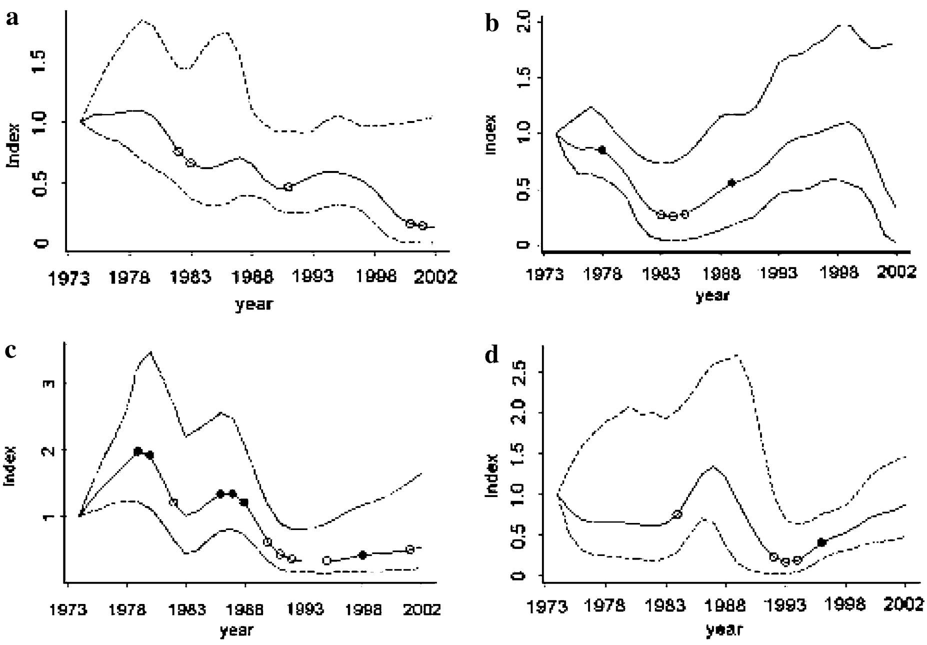

Fig. 5 Index curves (solid lines) for rabbit population trends in different areas of Spain from 1973 to 2002: Galicia, NW Spain (**a**), Levante, E Spain (**b**), central Spain (**c**) and Andalucía, South Spain (**d**). In all cases index curves were based on a GAM with 10 degrees of freedom. Dashed lines represent upper and lower 95% bootstrapped confidence intervals. Full circles correspond to significant negative turning points and empty dots represent significant positive turning points in second derivatives

Table 2 Threatened categories in the Spanish list of threatened species of 1990 (CNEA, Spanish acronym) and the type of program to be performed in each category

Threatened categories CNEA 1990	Government obligation
Endangered	Recovery program
Sensitive to habitat perturbation	Habitat conservation program
Vulnerable	Conservation program
Special interest	Management program

In central Spain, the pattern was very similar to the overall pattern, although the first decline appeared slightly earlier (Fig. 5c). A decline of 82.56% was observed between the highest abundance in 1978 and the lowest in 1994, among the most marked declines across the country during the study period. We observed three significant downturns, in the years 1978, 1987 and 1995, whereas significant upturns were recorded in 1981, 1990–1992 and 2001 (Fig. 5c, Table 2). In contrast to the north-west and eastern regions, central populations showed some signs of recovery in the last 7 or 8 years, similar to the overall pattern.

In the south of Spain, rabbit populations appeared stable during the late 1970s (Fig. 5d). A significant upturn was detected in 1984–1985 (Fig. 5d, Table 2), when populations increased by 16–18%. A non-significant downturn occurred between 1987 and 1992 (Fig. 5d, Table 2), when populations crashed by 72%. In this calculation, the large confidence intervals likely precluded the detection of this decrease as significant in the analysis of second derivatives, a problem that was outlined in a similar study by Fewster et al. (2000). A significant upturn was recorded in 1995 and populations increased after this point, as in the case of central Spain.

In summary, rabbit population trends were similar country-wide, although the declines occurred at different moments depending on the region. Thus, some areas showed a large decline during the late 1980s and others showed declines in the last 5–6 years of the time series. Different regions also differed in the existence of previous declines before RHD outbreaks in the late 1980s.

Discussion

Hunting bags and GAM use in the analysis of population trends of game species

The study of population trends is a key element of the management of populations and a critical aspect of modern conservation biology (Geissler and Sauer 1990; Fewster et al. 2000). Several wild species are harvested or hunted by humans and records of these hunting activities potentially can be used when monitoring populations and conducting other ecological or environmental studies. An exemplary case is the use of fish catches in the analysis, forecast and management of fisheries (Hilborn and Walters 1992). Hunting bags also offer this opportunity, and they have been used for a variety of taxa to model population fluctuations, population trends and predator-prey dynamics (e.g., Cattadori and Hudson 1999; Kitson 2004).

Nevertheless, it is possible that hunting bag records are not good indicators of true population numbers or trends, and a direct validation has been called for. In some cases a good correlation has been shown between hunting bag data and true population density or abundance (Royama 1992; Cattadori et al. 2003; Kitson 2004). Although a direct comparison of hunting bags and density of rabbits was not available, this is no reason to think that bags do not represent true numbers, or at least represent reliable population trends. Rabbit hunting method did not substantially change during the period of study (hunting with dogs and by individual hunters on foot), in contrast to hunting method in relation to other forms of small game such as the red-legged partridge (REGHAB 2002). We thus believe that the relationship between hunting bags and rabbit density did not differ between years. An interesting 'ad hoc' result that validated this source of data was the good correlation between significant population declines (turning points) and the appearance of RHD in multiple areas in Spain. RHD first appeared in some regions in 1988 (Argüello et al. 1988) and in the following five years most areas of Spain suffered the effects of this new disease (Blanco and Villafuerte 1993; Villafuerte et al. 1994; Cooke 2002). The significant turning points detected in our study were mainly located in the period 1987–1994 when declines of 50–90% were estimated in most areas, similar to data reported in local areas where the disease was studied (Peiró and Seva 1991; Blanco and Villafuerte 1993; Villafuerte et al. 1994).

Interestingly, although the greatest decline was observed during the period 1988–1989, a large crash was also observed in 1987–1988. These data may indicate that RHD virus was introduced to Spain a year before it was first recorded. The lack of suitable disease monitoring programs in Spain may have prevented the detection of the disease in the first epizootics. In France and central Europe, the first outbreaks were recorded in previous years (Cooke 2002) and it is therefore possible that this could also have been the case in Spain as suggested by our results.

In addition to demonstrating the value of using hunting bag data as a proxy for population numbers, we also highlighted the usefulness of GAMs for robust and flexible descriptions of population trends (see also Fewster et al.2000; Siriwardena et al. 2000). The use of GAMs may enhance the scientific study of species status and conservation by including all possible functional relationships between population numbers and time.

Rabbit population decline and conservation

The wild rabbit is native to the Iberian Peninsula (Thompson and King 1989; Monnerot et al. 1994) and is an important prey item for many endangered predators, emphasizing the importance of its conservation. The rabbit is also a very important species for small game hunting in Spain (REGHAB 2002). Despite these facts, little effort has been made to conserve this species in Spain, where most action plans related to rabbit conservation have been developed within conservation programs for endemic endangered predators (MIMAM 1999, 2001; but see Calvete and Estrada 2004; Calvete et al. 2005).

Until the proclamation of the Law of Nature Conservation in 1989, which included the establishment of the National Threatened Species List (CNEA), Spain had no nationally organized concept of conservation. In 1990, the CNEA introduced three categories for classifying threat levels to native species that obliged the government to establish a particular conservation program for a species based on its threat category (Table 2). In 1992 the first CNEA was published and wild rabbit was listed as a "Least Concern" (LC) species (based on the Red Data Book for Spanish Mammals, Blanco and González 1992). Listing of taxa in this category implies that there is no evidence that the survival of the species is threatened. Paradoxically, the best available evidence at that moment indicated that rabbit population crashes exceeding 80% had occurred in several regions of Spain (Peiró and Seva 1991). One year later, a technical report based on questionnaires was published suggesting that rabbit populations had declined by around 55% over a 5-year period and that this decline coincided with the first appearance of RHD in 1988 (Blanco and Villafuerte 1993).

At an international scale, the IUCN Red List System was established in 1994, which specified quantitative criteria for how taxa should be listed and conservation efforts initiated (IUCN 2001). Under these guidelines the IUCN stated that any taxon fitting at least one of five criteria should be included in one of the threatened categories (Critically Endangered, Endangered, Vulnerable; IUCN 2001). Moreover, the IUCN stated that the best available evidence about a species have to be used in order to list the species. In 2003, when the new version of the CNEA (CNEADI, in preparation) was proposed, only the studies of Peiró and Seva (1991) and Blanco and Villafuerte (1993) were available to evaluate rabbit status. Nevertheless, Criteria A of the CNEADI and the IUCN Red List System (population decline) require data from 50- and 10-year periods, respectively. Thus, although the available data were alarming, they were not sufficient to qualify rabbits for any category other than ''Data Deficient'' in the IUCN Red List system. Listing of taxa in this category indicates that more information is required and acknowledges the possibility that future research will show that a threatened classification is appropriate. Surprisingly, the rabbit has again been proposed as a LC species in the CNEADI. A taxon is LC when it has been evaluated against the criteria and does not qualify for any of the threatened categories (IUCN 2001).

The IUCN states that it is important to make positive use of whatever data are available. In many cases great care should be exercised in choosing between 'Data Deficient' and a threatened status (IUCN 2001). Our study is the first scientific attempt to quantify rabbit population trends in Spain over the last 30 years, and the data are clear and relevant: wild rabbits have suffered a major national decline of around 70%, with values of up to 90% observed in some areas. Indeed, such a strong decline may be considered as catastrophic (Reed et al. 2003), which greatly increases the probability of extinction of these populations (Lande 1993; Reed et al. 2003). Following the criteria of the IUCN Red List System, our analysis indicates a population size reduction of 42–49% over most 10-year periods within the 30 year-period of the present study. Thus, our result proves that the rabbit meets Criteria A4 and should be listed as Vulnerable. According to the CNEADI requirement that data cover a period of more than 50 years, our study period is too short; however, if experts were in agreement about rabbit status, the species could also be listed as VU and as such, the Spanish government would be obliged to establish a conservation program (Table 2). Nevertheless, it is important to emphasize that a taxon may require conservation action even if it is not officially listed as threatened.

The lack of a nationwide conservation program for wild rabbit in Spain is probably one of the most paradoxical conservation issues in Europe, and possibly the world. Similar to the declines observed in Spain, the appearance of RHD in Australia and New Zealand in 1995 caused large population crashes of 40–95% (Neave 1999; Cooke 2002). In contrast to the Spanish situation, researchers, managers and conservationists in those countries all agreed upon the usefulness of national monitoring programs and research on rabbits. This attitude promoted governments to perform such monitoring not only for studying population trends, but also for the detection and study of RHD outbreaks (Neave 1999; Cooke 2002; Henzell et al. 2002). Furthermore, it must be noted that, in contrast to Spain, the rabbit is considered a major pest in the southern hemisphere, where it has been introduced and has caused significant environmental damage. The large discrepancy between the Spanish and Australian attitudes to the researching and monitoring of rabbits highlights how similar situations can elicit very different conservation responses.

The major declines observed in Spain give rise to a general question: why is the wild rabbit not a protected species in Spain? The case of rabbits in Spain is a good example of the constraints faced by conservationists in most countries, where in addition to demographic baseline data, conservation practices are dictated mainly by socio-economic constraints including political trade-offs, 'lobby' pressures and traditional management (Kellert 1985; Pullin and Knight 2001). Since the IUCN World Conservation Strategy (1980), the management of species has shifted from a strict 'conservationist' view to one where people's perceptions and attitudes play a key role in conservation decisions (Bandara and Tisdell 2003). Within this new paradigm, even small but important stakeholder groups may exert a strong influence on conservation decisions (Kellert 1985).

Despite the decrease in the economic value of rabbit hunting in the last decades (perhaps as a consequence of rabbit decline), rabbits continue to be socially important. Rabbit hunting is mainly carried out by municipal and local associations of hunters who do not have enough economic resources to access other small game hunting practices, such as red-legged partridge hunting (Bernabeu 2000). Therefore, rabbit is the only hunting species for a large proportion of hunters. Moreover, many of these hunters consider rabbit hunting a traditional activity with large emotional value (REGHAB 2002). The hunter collective represents a very important social force in Spain; for instance, 80% of the country is covered by hunting lands (REGHAB 2002) and around 1.3 million people are linked to game activities, with most of them linked to rabbit hunting (REGHAB 2002; Angulo and Villafuerte 2003).

It is clear that politicians are influenced by hunter demands or opinions about wildlife conservation or management. Although in the last decades the conservation movement has strengthened in Spain, its political influence remains low, especially in rural areas (Kellert 1985; Bandara and Tisdell 2003). It is also clear that any conservation measures that may reduce traditional hunting activities, especially actions related to the conservation of game species, will face strong opposition from hunters and politicians, and so managers or conservationists may be reluctant to implement such measures.

Furthermore, the Spanish Hunters Federation and other hunter associations often make economic contributions to researchers who study rabbits and other game species. Such contributions are good for conservation biology, but could be questionable in instances where the interests of multiple lobby groups are in opposition. Unfortunately, we feel that most decisions regarding rabbit conservation in Spain could be tainted by such negative feedback between research, management and funding sources.

The sociological scenario for rabbit conservation in Spain is therefore very complex, with politicians, hunters, conservationists and other social forces playing distinct roles. Similar social or political interference in conservation plans has been observed elsewhere. Perhaps the best-known example in recent decades has been the restocking of wolf populations in Yellowstone National Park, USA (Wilson 1997; Bangs et al. 1998). Conservationists and managers sought the recovery of the park community structure prior to the depletion of the wolf population by man. In contrast, landowners, mainly devoted to cattle raising, were very reluctant and opposed to attempts to bolster the population of such a predator. After much heated debate, politicians and managers agreed to re-establish wolves based on ecological principles, but assuming a certain social cost (Wilson 1997). This example shows how conservation priorities may overcome social opposition when biological reasoning prevails. In some circumstances biological reasoning may not be as useful, and a strong opposition may impede implementation of conservation actions or reduce the efficacy of whatever action is taken. For example, the restocking of a species such as the wolf in areas with high cattle interests may lead to landowners using poison, negatively affecting both wolves (the target of the management action) and other species of conservation concern (e.g., other carnivores, raptors, etc.). We thus advocate a balance between social and biological reasoning.

In conclusion, we call for the urgent development of a national conservation plan for rabbits in Spain, independent of the Iberian lynx and Imperial eagle recovery programs. We want to emphasize that the conservation of rabbits cannot be impeded for sociological reasons or to accommodate any particular lobby group. This conservation plan does not need to be restricted to the protection of rabbits, which may elicit strong opposition from hunters and farmers; rather, it should be based on an adaptive approach (MacNab 1983; Parma et al. 1998). Within this framework, management actions should be based on the particular circumstances of each region, including social opposition in areas where rabbits cause significant crop damage. We also wish to emphasize the need to make positive use of whatever data are available (Pullin and Knight 2001), and advocate the use of scientific methods and standard scientific procedures when making conservation decisions, especially when classifying species into conservation categories.

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