

Where is the pulse to have the finger on? A retrospective analysis of two decades of Alpine Galliforms (Aves: Galliformes) census and game bag data in Italy

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Abstract Information on the abundance of the Italian populations of black grouse (*Lyrurus tetrix*), Alpine rock ptarmigan (*Lagopus muta helvetica*) and Alpine rock partridge (*Alectoris graeca saxatilis*) rely only on extrapolations of local data to the national scale, since there is no national standardized survey. Consequently, their status is virtually unknown. We performed a first-ever assessment of a medium-term (1996–2014) population trend of these species using and comparing post-breeding count and bag data at hunting district scale. These data were collected from various authorities in charge of wildlife management and allowed us to test the influence of hunting policies on the estimated trends. Rock partridge showed a stable trend with numbers fluctuating between years, while there was evidence of a severe decline for rock ptarmigan. No general conclusion could be drawn for the black grouse, as we detected lack of consistency of count and bag data. Counts were greatly overdispersed as a result of an uneven count effort among hunting districts. Adding the game management authority as model covariate resulted in more robust trend estimations, suggesting a significant effect of different policies that emerged also as similar hunting pressure across species within authorities. Hunting effort variation over the time was instead negligible. Species-specific game management bias is discussed. Our results highlight the need

for a survey scheme or guidelines to be applied uniformly at a national scale.

Keywords Alps · Grouse · Status · Italy · Hunting bags · Game management

Introduction

The assessment of a species conservation status in a given area requires a reliable estimate of its abundance in time (Silveira et al. 2003; Kozłowski 2008; Newton 2013). This is usually achieved through the application of a standardized direct or indirect count method, repeated over the years (see Bibby et al. 2000; Sutherland 2006), or the collection of other abundance indices, such as road kills (Gehrt et al. 2002; Seiler et al. 2004; George et al. 2011; González-Gallina et al. 2016) and game bags (Chanin and Jefferies 1978; Liberg et al. 2010; Rödel and Dekker 2012; Moleón et al. 2013). At a local scale, the assessment can be accomplished with good reliability, since it depends on a limited number of people and is usually tailored on the area and on the eco-ethology of the target population (Greenwood and Robinson 2006). At larger scales, such as regional or national, abundance assessment requires good supervision and coordination and is increasingly biased by differential skills and knowledge of the people involved, by possible under- or oversampling of areas and by the method that may not fit evenly (Sauer et al. 1994; Schmidt 2004; Reidy et al. 2011).

Monitoring of grouse populations has a long tradition in many parts of Europe and North America (Rogers 1963; Gates et al. 1968; Rajala 1974). Indeed, many grouse species are popular game birds and populations are locally surveyed in order to provide the basis for sustainable harvesting (Ellison et al. 1988; Aebischer 1991; Aanes et al. 2002). Anyway, local

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survey outcomes are often unsuitable when dealing with conservation issues at a large scale, for example to understand grouse population declines in many countries, making them one of the most threatened bird taxa on the Alps and in upland Fennoscandia (Lehikoinen et al. 2014; Chamberlain et al. 2016), possibly under increasing pressure by climate change (Selås et al. 2011; Revermann et al. 2012; Zurell et al. 2012; Imperio et al. 2013), forestry practices (Kurki et al. 2000; Ludwig et al. 2008; Pearce-Higgins et al. 2016), land abandonment (Anthelme et al. 2001; Ludwig et al. 2009; Maggini et al. 2014) or recreational activities (Patthey et al. 2008; Montadert 2013; Storch 2013).

Currently used grouse census methods are rather standardized and are usually performed in two different periods (Gilbert et al. 1998; Bibby et al. 2000): in spring lekking or displaying behaviour offer the chance to estimate male densities and winter survival (Bossert 1977; Cayford and Walker 1991; Favaron et al. 2006; Pearce-Higgins et al. 2007; Sadoti et al. 2016), while breeding success is evaluated through late summer counts, with (Moss and Oswald 1985; Bowker et al. 2007) or without (Helle et al. 2016) the help of pointing dogs. Nonetheless, the actual implementation of such methods can broadly change between and within states, as a consequence of different game management principles and policies (see for a European review, Putman et al. 2011), centralized or decentralized hunting authority, environmental characteristics (chiefly topography, Marty and Mossoll-Torres 2011; Bisi et al. 2017), bird population densities (Höglund and Stohr 1997) and hunting traditions. Variability lies in the number, size and density of sampling areas, timing of counts, number of repeats per season, fieldworkers' skill, use of playback devices and data recorded (sex, age, etc.). Only a few countries have a common standardized monitoring scheme that leads to comparable estimates of population sizes and trends (e.g. Hudson 1992; Lindén et al. 1996) and supranational monitoring schemes are yet uncommon (Lehikoinen et al. 2014).

Hunting records represented historically the first proxy data for population trend assessment for game or harvested species, since Elton and Nicholson's pioneering work on Canadian lynx pelt-sale records (Elton and Nicholson 1942), and are still used to analyse long-term grouse population dynamics and trends (Barnes 1987; Cattadori et al. 2003; Aebischer and Baines 2008; Ranta et al. 2008). In a few countries, bag records proved to complete well or even substitute census data (Liberg et al. 2010; Soininen et al. 2016). Bag records' reliability is questioned however as variation in hunting effort can introduce significant bias (Ranta et al. 2008; Imperio et al. 2010; Willebrand et al. 2011) and therefore trends need to be corrected in order to provide meaningful results. However, if shooting plans are not used as means of providing sustainable harvesting, hunting effort can be easily

accounted with the number of hunters or licences issued per year (Imperio et al. 2010; Soininen et al. 2016), information which can be obtained by privately owned hunting reserves or centralized public hunting authorities. Further, bias may derive from weather conditions, which may limit hunters in case of persistent unsuitable weather, or from stocks dropping below critical densities, at which hunters' seek is no longer fruitful and so they usually stop (Newton 2013).

In Italy, only black grouse (*Lyrurus tetrix* L.) and Alpine rock ptarmigan (*Lagopus muta helvetica* Thienemann) are game species among the four grouse species that live in the Alps, as both capercaillie (*Tetrao urogallus* L.) and hazel grouse (*Tetrastes bonasia* L.) are protected since the early 1990s. Alpine rock partridge (*Alectoris graeca saxatilis* Bechstein) is a game bird grouped together with grouse into the so-called *tipica alpina* (typical Alpine) hunting form, due to taxonomic and eco-ethological affinity. Thus, these three species are often dealt with jointly in hunting policies.

Despite three out of five Alpine Galliform species being listed as Vulnerable on the Italian Red list of birds (Alpine rock ptarmigan, capercaillie and Alpine rock partridge, Peronace et al. 2012) there is no national survey on Galliforms in Italy. Thus, hunting districts' bird counts actually provide the longest and most widespread survey of game birds on the Italian Alps, but application of methods can markedly differ between hunting districts and provinces, since according to current regulations, hunting is mostly under regional and provincial authority. As stated before in a broader sense, also in the particular case of the Italian breeding populations of Alpine Galliform species, there are no reliable estimates (BirdLife International 2004; Brichetti and Fracasso 2004; Peronace et al. 2012; BirdLife International 2015a). Indeed, known population sizes are outdated and mostly derive from extrapolations of local data to the national scale (Brichetti and Fracasso 2004; Franzetti and Toso 2009). Short- and long-term population trends are based on expert knowledge and do not rely on field data (Nardelli et al. 2015). As a consequence of this situation, Italy is the only Western European country that lacks an evaluation of black grouse population trend both at short- and long-terms (BirdLife International 2015b). In addition, rock ptarmigan and rock partridge trend estimates are considered of poor quality (BirdLife International 2015c; BirdLife International 2015d). Similarly, Italy does not have a national archive of Galliform bags (Franzetti and Toso 2009), because hunting records at best are collected at the regional or provincial scale.

In this study, we assessed the status in Italy of black grouse (hereinafter BG), Alpine rock ptarmigan (PTA) and Alpine rock partridge (PAR). We further tested (i) whether count and bag data reflect similar trends, (ii) the influence of game management policies and (iii) whether hunting effort significantly changed over the studied period.

Materials and methods

Study area

According to the Italian legislation on wildlife protection (L. 157/92) that also regulates hunting, the Alpine territory is divided into hunting districts that manage hunting activities under provincial or regional supervision and control, since wildlife in Italy is state-owned and hunting is managed through a licence system. Law 157/92 defines the Alpine territory as “the area where Alpine fauna lives” and it is shared between five regions (Valle d’Aosta, Piedmont, Lombardy, Veneto and Friuli-Venezia Giulia) and two autonomous provinces (Trento and Bolzano/Bozen). The game management districts’ extent is uneven among regions, mostly for historical reasons: the eastern regions (Veneto, Trento, Bolzano and Friuli-Venezia Giulia) still adhere to a hunting regulation form originating under the Austro-Hungarian Empire dominion in the nineteenth century, where hunting was regulated at the municipality level, and therefore hunting districts in the Eastern Alps are almost municipality sized (about 37 km², ISTAT 2013), whereas in the western regions hunting districts are at subprovincial size (in the order of magnitude of about 500 km²). For the purpose of our study, we grouped hunting districts data (counts and bags) of the eastern regions, in order to have an even dataset in terms of values range per surface area; if present (Trento, Friuli-Venezia Giulia), we used other game management entities that already group hunting districts. In the case of Belluno Province of Veneto region, we grouped hunting districts’ data to geographical coherent areas that matched the average size of the other districts. For this reason hereinafter, we are using a more general “game management unit” (GMU) concept to identify the target areas. Overall, 72 GMUs have been analysed, with an average extent of 468 (±285) km² (Fig. 1). Similarly, we associated each GMU to an “authority” representing the province or the region according to the administrative level in charge of game management, in order to evaluate the effect of different policies.

Count and bag data

Mandatory counts were first introduced in 1992 with the above cited Law 157/92 on wildlife protection. For this reason, standard monitoring was uncommon before 1993, when the national law was enforced at the regional level. Depending on the region, Galliform counts are run by hunters and/or by game wardens and are usually performed twice a year, using standard techniques applied in most countries. Spring pre-breeding census (April–May) aims at evaluating densities and winter survival by counting males displaying in leks for BG (Léonard 1989; Baines and Hudson 1995; Hancock et al. 1999) or singing in their territories for PAR and PTA from

points or along transects, with (Bernard-Laurent and Laurent 1984; Bernard-Laurent 1994; Nopp-Mayr and Zohmann 2007; Mossoll-Torres and Marty 2011) or without (Bossert 1977) the help of playback calls. Reproductive success is evaluated in post-breeding summer counts (August–September) with pointing dogs that help locate broods (Zbinden 1987; Ellison et al. 1988; Léonard 1992; Novoa et al. 2008).

Yearly, a shooting plan that establishes bag limits is set for each hunting district and it is usually based on the estimated breeding success of each species, derived by counts. Bag limits vary between different administrations. The plan is set by the local hunting authority, but it can derive from hunting district boards’ proposal. Grouse and partridge hunting is popular in the Western and Central Alps (e.g. Piedmont and Lombardy), but it is a marginal hunting form in the Eastern Alps (Veneto, Friuli-Venezia Giulia).

Only a limited number of authorities issued guidelines establishing minimum requirements for planning counts in terms of period, extent of sampling areas, methods and criteria for setting up the hunting plan. Thus, count data can markedly differ across authorities or even among hunting districts within the same authority. Only few hunting districts or authorities have a georeferenced database for storing data, or use at least a GIS for sampling areas definition or data archival.

Overall, counts, bag sizes and shooting plans were available as GMUs’ annual grand totals. Data were collected from 72 GMUs, covering the whole Italian Alps with the exception of the Bolzano/Bozen Autonomous Province. Game bags and bird counts were used to calculate population trends, while shooting plan data were used as a proxy variable to evaluate hunting effort. Post-breeding counts were chosen as state variable since only a few districts regularly perform pre-breeding counts, which are more commonly used to estimate population trends. Data series lengths are uneven by species and variables, as well as by GMU (Table S1), due to different archival policies and harvest suspensions or bannings.

Data analysis

Trend analysis Population trends of the target species were estimated using TRIM (Pannekoek and van Strien 2001), a software used in many bird monitoring schemes (e.g. PECBMS 2009; Géhin and Montadert 2016; Helle et al. 2016); and for each species, two separate trend estimations were carried out using game bags and post-breeding count data series. Due to unequal time series lengths, we limited our dataset from 1996 to 2014. Considering the three studied species and the two putative state variables, we calculated 18 trend estimates using TRIM as follows: a first model (linear model) was calculated with a linear trend function and default options, without accounting for covariates and assuming a random effect of time (Fig. 2). In a subsequent TRIM run,

we added data origin (i.e. authority) as a covariate (covariate model) to test for game management fixed effect. A final run used a refined model considering a fixed effect of time, i.e. checking for significant nonlinearities in the trends (nonlinear model). We were unable in the latter case to use data origin as a covariate due to missing data for some authorities. Spatial variation of trends can depend also on environmental factors, such as climate and habitat, acting at larger scale, but grouping authorities at the regional level resulted in weaker models or nonsignificant regional covariate (for details, see Table S3). Trends are calculated on the estimated log-transformed annual population totals. For all three models, TRIM was configured to account for serial correlation and overdispersion and to use 2001 index as base index (2001 index = 1.00). Model robustness was evaluated through comparison of AIC and overdispersion values provided by the software. An autocorrelation test was performed on bag and count indices to check for population cycles, using *Acf* function in *forecast* R package (Hyndman and Khandakar 2008).

Proxy comparison We tested the consistency of bags and counts building for each species a linear regression model with count and bag TRIM indices as dependent variable, year as continuous independent variable and proxy type (bag or count) as categorical independent variable, adding the year by proxy type interaction to assess for significant differences in slopes. For each species and proxy, we used indices from the models with the lowest AIC values. Bag values of Trento Province were removed from this analysis since for this authority count data were unavailable. Since the regressions were performed on time series data, we used a GLS (generalized least squares) method to account for the autocorrelation structure of residuals. We further tested whether counts and bags were correlated by performing a pairwise Pearson's correlation on index series that was previously differenced to meet the assumptions of independence and stationarity by removing autocorrelation and trends. Data analyses were performed using R software, version 3.3.2, *gls*, *cor* and *diff* functions from *nlme* and *base* packages (Pinheiro et al. 2016; R Core Team 2016).

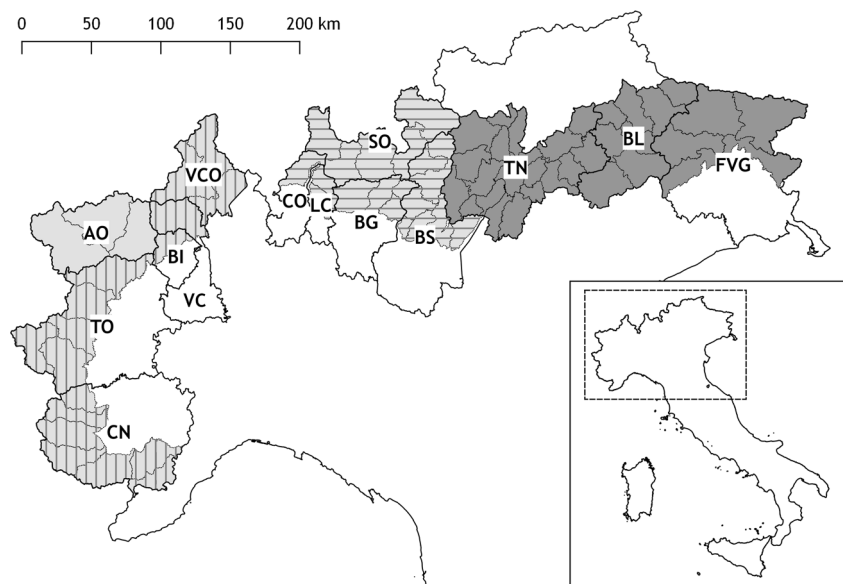
Hunting effort

The utilization of bag data as proxy for population abundance requires the evaluation of hunting effort, which is usually calculated as the number of hunters or hunting licences issued per year (e.g. Imperio et al. 2010; Soininen et al. 2016). In Italy, the number of hunters has decreased in the last decades (ISTAT 2009), but these information are not available at the local scale, so we were unable to relate bag data to the number of hunters or hunting days. The policy of shooting plans adopted in Italy can partly overcome the bias introduced by the

variation of the number of hunters over time, since the annual yield in each district is limited at the top regardless of the hunting effort, but it may not control if hunting effort decreases, which is the most likely scenario. Assuming that shooting plans were proportional to summer counts (i.e. population abundance), we used the annual yield to shooting plan ratio in each GMU as proxy for hunting effort. The ratio can occasionally exceed 1 if early closure of the hunting season is delayed despite the shooting plan is about to be completed. We assessed linear trends in the hunting effort by regressing the ratio against years for each GMU and then we calculated the mean of the slopes of all significant ($p < 0.05$) regressions. Autocorrelation of the error at lag 1 was tested with a Durbin-Watson test for autocorrelation of disturbances, using *dwtest* function in *lmtest* R package (Zeileis and Hothorn 2002). Hunting units showing significant autocorrelation were modelled with a GLS linear regression method. Since, in some GMUs, hunting had been opened only for a few years, regression was applied only to a time series spanning for at least 10 years. Hunting effort on BG in the Valle d'Aosta region was assessed at a regional level because, according to the local hunting regulations, the plan is set for the whole region and not subdivided among hunting districts. Regressions were performed using the R software, version 3.3.2, *lm* and *gls* functions from *stats* and *nlme* packages (Pinheiro et al. 2016; R Core Team 2016).

Hunting pressure Hunting pressure is one of the main features of game management. In Italy, hunting pressure is regulated by setting of shooting plans, but since criteria to set plans are at best established at the regional scale, we tested for hunting pressure differences across management authorities. Within authorities, we also compared hunting pressure across species in order to show possible common patterns. We elaborated a hunting pressure index (HPI) based on the catch-per-unit area (CPUA) of a species' suitable habitat. For each species and GMU, we estimated the extent of suitable habitat by reclassifying the Corine Land Cover map (Büttner et al. 2014) according to a species' ecology. Then, we calculated the log-transformed ratio of the annual bag values and the estimated size of the suitable habitat. An annual HPI for each species and GMU was defined as the difference between bags to habitat ratio and the species' GMU median value. A MANOVA with Wilks' statistic was performed to test for the common effect of the hunting authority on the three species using *manova* function from *stats* R package (R Core Team 2016), after testing the assumption of normal distribution of residuals with Shapiro-Wilk's test. The MANOVA was run using only full combinations of annual HPI for the three species ($n = 345$). Univariate ANOVA was also performed to check for the effect of authority on single species.

Fig. 1 Location of the 72 Game Management Units (GMUs) whose data have been analysed. Eastern regions, whose hunting district data were grouped due to a different zoning, are shown in *dark grey*. Solid outlines represent game management authorities. Regions that do not correspond to authorities in our study are shown with *filling lines* (horizontal lines Lombardy; vertical lines Piedmont). CN Cuneo. TO Turin. AO Aosta. BI Biella. VC Vercelli. VCO Verbano-Cusio-Ossola. CO Como. LC Lecco. SO Sondrio. BG Bergamo. BS Brescia. TN Trento. BL Belluno. FVG Friuli-Venezia Giulia. *Inset* Location of the study area with respect to Italy



Results

Trend analysis and proxy comparison

The proportion of missing values imputed by TRIM for both game bags and count data ranged from 11.5 to 36.1% depending on the species and proxy variable, with the lowest proportion in BG bag data and the highest in PAR bag data (Table 1).

Considering the period 1996–2014, all game bag-based models estimated a negative trend for the three species, while only PTA showed a significant negative trend according to count data (Table 1). Bags estimated for BG a mean annual population reduction equal to 2.1–2.4% depending on the

model, whereas according to counts, BG populations remained essentially stable ($-0.5/+0.1\%$; Wald test on significance of slope in the linear model, $p = 0.449$).

Both bags and counts showed a negative trend for PTA ($-3.9/-5.4\%$). PAR populations are characterized by fluctuations over the considered time span both in bag- and count-based models that ultimately resulted in a negligible negative trend according to bags ($-0.9/-1.2\%$), while counts estimated a stable trend ($-0.4/+0.2\%$; Wald test on slope parameter, $p = 0.14$). A major feature that emerged was a population peak in 2006–2007 and two troughs in 2001 and 2011, resulting both in bag- and count-based models. A significant negative autocorrelation was found at lag 6 (-0.607 , CI 95% ± 0.449) in

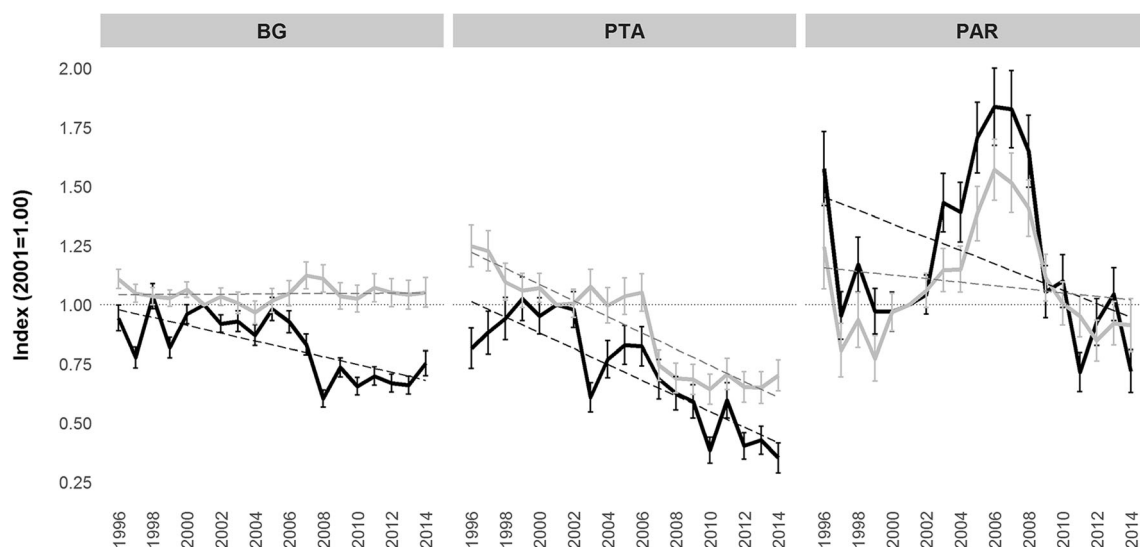


Fig. 2 Population indices of the three target species (BG = black grouse; PTA = rock ptarmigan; PAR = rock partridge), selected from the models with the lowest AIC values (BG and PTA counts = covariate; the

remainders = time effect). Grey solid lines represent count indices; black solid lines represent bag indices. GLS regressions fitted through population indices are shown with dashed lines

bag time effect model and at lag 5 and 6 (-0.527 and -0.518 , CI 95% ± 0.449) in count time effect model. Conversely, no significant autocorrelation was found in BG and PTA.

In bag-based models, the lowest AIC values were provided by the time effect models for all species. Conversely, covariate models were selected in BG and PTA count-based models, while still the time effect model performed better in PAR counts. Nevertheless, covariate models performed better than the linear models and the effect of the authority covariate is highly significant (Wald test, $p < 0.01$) for all species and proxy variable. Count data were highly overdispersed relative to bags, with overdispersion values far beyond the suggested threshold for TRIM use (Pannekoek and van Strien 2001). Bag-based models exceed the 3.0 threshold only for PAR, while count-based models overdispersion values was up to sixfold.

Bag and count slopes were significantly different as evidenced by the GLS regression interaction term for BG (0.017 , $p = 0.018$), but not for PAR and PTA (respectively, $p = 0.515$ and $p = 0.906$). Conversely, pairwise Pearson's correlation on differenced indices emerged for PAR ($r = 0.77$, $p < 0.001$), but not for BG and PTA (respectively, BG, $r = -0.05$, $p = 0.839$; PTA, $r = 0.03$, $p = 0.890$). We concluded that slope consistency between bag- and count-based PTA models may be spurious, since a significant correlation would be expected if the two variables described annual abundance likewise. Albeit GLS regression accounts for autocorrelation, it must be noted however that fitting a linear model through the peaks and troughs of PAR population resulted in high standard errors. This constrained the power of the model and the capability to detect different trends between proxies.

Hunting effort

We detected a limited variation in time of the bag to plan ratio for all three species. Regression slope coefficients for BG, PAR and PTA were significant ($p < 0.05$), respectively, in 19% ($n = 69$), 28% ($n = 44$) and 8.5% ($n = 35$) of the GMUs. In addition, mean significant coefficients were small (BG, 0.007 ± 0.020 , $n = 8$; PAR, 0.027 ± 0.026 , $n = 12$; PTA, -0.009 ± 0.039 , $n = 3$), indicating that hunting effort variation could be considered negligible on the estimated bag-based trends. Nevertheless, positive average slope values emerging for BG and PAR are not consistent with the hypothesis that hunting pressure would have likely decreased in time following the downward trend of the number of hunters.

Hunting pressure

MANOVA performed on the HPI was highly significant ($F_{8,336} = 19.314$, $p < 0.001$, Wilks' lambda = 0.322), evidencing that the hunting pressure is a distinctive feature of the authority regardless of the species. Only a limited number of

combinations year $\times$ GMU $\times$ species were available for the MANOVA, since hunting of all three species during the considered period occurred only in some GMUs. Nevertheless, similar hunting pressure emerges also in authorities that allow only BG and PAR hunting, as shown in Fig. 3. Authorities with the highest hunting pressure are all located in the Lombardy region, which is the only one that lacks management guidelines (see Table S2). Conversely, low hunting pressure is present in regions that defined guidelines on shooting plans or with a regional management authority that acts as a control on hunting.

Discussion

Choosing the best proxy data is crucial in the evaluation of the status of a species (Joseph et al. 2006), especially in poor data systems (Popescu et al. 2016). Data on the Italian populations of BG, PTA and PAR fit this definition, as already highlighted by the latest national reports on the application of the Birds Directive (Nardelli et al. 2015) and on conservation status of game species (Franzetti and Toso 2009). Our study addressed this issue by performing a medium-term evaluation of population trends of these species considering two different data sources and evaluating the effect of hunting policies on data.

In managed wildlife populations, both count and bag statistics are influenced by game management policies (see Apollonio et al. 2010). Galliform hunting in the Italian Alps makes no exception from the results of our study, as we detected a significant effect of the game management authority on both count and bag data in all species. Extrapolations of local data to national scale as a tool to assess the species' conservation status are therefore misleading if data authority source is not accounted. Covariate count-based models were selected over time effect models for BG and PTA. For these species, accounting for game management by means of a fixed effect of the authority resulted in more robust models than accounting for a fixed effect of time. Conversely, for bag-based models, time effect models were selected over covariate models for all species, suggesting that game management may have less influence. Overdispersion of counts is remarkably high and cannot be explained only by means of different population densities, since all species are evenly distributed across the Italian Alps, especially BG (Brichetti and Fracasso 2004). Thus, overdispersion of counts is likely to result from an uneven count effort among GMUs (e.g. number and size of sampled areas), which is ultimately derived from different game managements. On the other side, the decreasing number of hunters that is occurring at the national scale seems not to plausibly affect the robustness of bags as state variable for population abundance.

However, jumping to the conclusion that bags should be chosen over counts to estimate trends of Italian Galliform

Table 1 Model comparison and additive annual growth rates (on log-transformed annual population indices) as calculated by TRIM software

Species	Proxy	<i>n</i> sites	<i>n</i> observations	% missing/total	Model	Overdispersion	AIC	Wald test	<i>p</i> value	Trend
BG	Bags	72	1368	11.5	Linear	2.598	897.0	73.7	***	−0.023
					Covariate	2.339	595.2	80.6	***	−0.024
					Time effect	2.385	499.9	152.5	***	−0.021
	Counts	52	988	31.7	Linear	17.327	9523.9	0.6		−0.002
					Covariate	15.868	8579.6	38.3	***	0.001
					Time effect	16.945	9571.5	28.4	*	−0.005
PAR	Bags	60	1140	36.1	Linear	4.283	1867.7	6.2	*	−0.009
					Covariate	3.901	1568.5	34.4	***	−0.012
					Time effect	2.964	631.8	244.2	***	−0.012
	Counts	50	950	32	Linear	22.102	15,396.5	0.4		−0.004
					Covariate	20.189	13,707.5	34.6	***	0.002
					Time effect	16.764	9215.8	109.5	***	0.002
PTA	Bags	39	741	35.4	Linear	2.230	287.0	71.4	***	−0.049
					Covariate	2.095	204.9	24.9	**	−0.050
					Time effect	1.996	44.1	71.8	***	−0.053
	Counts	30	570	35.1	Linear	10.044	3604.3	71.9	***	−0.054
					Covariate	7.315	2253.3	80.3	***	−0.039
					Time effect	10.103	3466.0	30.3	*	−0.051

Wald test values refer alternatively to significance of slope parameter (linear model), authority covariate (covariate model) and deviations from linearity (time effect model). Overdispersion is calculated with respect to a Poisson data distribution

* ≤ 0.05 ; ** ≤ 0.01 ; *** ≤ 0.001 ; significance in terms of *p* value of Wald tests

populations could be misleading since species-specific considerations should be made. As a general rule, a strong relationship between post-breeding counts and game bags of the subsequent hunting season is expected, even when shooting plans are set, since the breeding success evaluated through post-breeding counts is the parameter most commonly used to limit annual yields to a sustainable rate. From the results of our study, this seems to apply well only to PAR, while both BG and PTA show specific inconsistencies.

BG was characterized by a significant difference between the trends estimated by bags and counts, with a strong divergence in 2003–2008. In this period, bags markedly declined even though counts remained stable. We speculate that this lack of consistency is likely to result from changes in game management policies that occurred during that period. In fact, some local administrations issued guidelines that established parameters to set shooting plans given the results of summer counts (i.e. Piedmont region and Trento Autonomous Province, both in 2004), therefore limiting subjectivity that used to lead to higher harvests. This hypothesis may explain why hunting effort seemingly increased in some GMUs. The underlying assumption that shooting plans are proportional to population abundance estimated through summer counts may not be effective throughout all the time span considered. Given the number of hunters (i.e. the true hunting effort), the bag to plan ratio may have increased as a consequence of a progressive reduction of shooting plans with respect to

summer counts. In other terms, game management policies changed from less to more conservative, accordingly to an increased control on game management policies resulting from the Natura 2000 network and EU Birds Directive enforcement. This likely happened also to PAR, as we found an increase in the bag to plan ratio in 12 out of 44 analysed GMUs, despite the trend difference between bags and counts resulted in no significance due to limited power of the model with highly autocorrelated errors.

In the case of PTA, population TRIM model slopes of bag and count series were similar and not statistically different as evidenced by the GLS regression, but the result emerging from the Pearson's correlation suggests that proxy consistency is spurious. Thus, population abundance reduction that occurred in the considered time span shaped the trends independently. This consideration has a great relevance from a management point of view, since it implies that the two proxies may be used either to assess the species' national status only in case of a persistent sharp trend, but minor changes or fluctuations might not be detected likewise. In other terms, population reduction likely represents the main driver of the observed PTA negative trend both for counts and bags, but other factors may affect the estimates. For example, PTA hunting is heavily affected by weather conditions occurring during the 2-month hunting season. In fact, early snowfalls may limit access to hunting grounds causing a mismatch between count and bag data. This is the case of 2003, when persistent snow cover at

the end of October was reported in many locations below 1800 m a.s.l. (ARPA Lombardia 2003) and it was recorded in the bag series by a trough in the trend. For this trough, there is no basis to speculate on a possible increased mortality in autumn caused by early snowfalls, since weather effects on population dynamics have been reported in many grouse species mostly in spring and summer (Scherini et al. 2003; Novoa et al. 2008; Furrer et al. 2016; García-González et al. 2016). Effects of game management on trends also emerged as a sharp decline in counts and, less evidently, in bag series in 2007. In this year, the Italian government enacted a law (DM 17/10/2007) on minimum conservation criteria in Natura 2000 sites, that transposed the EU “Habitat” 92/43/EEC and “Birds” 2009/147/EC Directives. According to this law, PTA hunting in Natura 2000 sites was forbidden, resulting in a massive reduction of available area open to PTA hunting, given that the 60% of the Italian breeding population is estimated to be located within SPAs (Nardelli et al. 2015). As a result, many sample areas in hunting grounds ceased to be surveyed, since only the estimated breeding success in hunting grounds is used as reference data for setting the shooting plans.

Italian PAR population shows an overall stable trend in the considered period according to both the count and bag data and appears to be characterized by a major 10-year cycle as evidenced by the autocorrelation. Population cycles are typical of grouse species, mostly in boreal regions (Ranta et al. 1995; Kvasnes et al. 2009; Lampila et al. 2011), but also PAR populations are known to exhibit cycles (Cattadori et al. 1999), despite the observed lag is larger than the 4–7 year

lag reported. Larger lags may be derived from a harmonic effect of multiple lags, as hypothesised by Cattadori et al. (1999). Interestingly, the same population peak has been recorded also in Swiss populations that is based on pre-breeding counts (Sattler et al. 2016). However, further analyses with a longer series are needed to confirm cyclic behaviour of PAR populations, since our 19-year time series fits only one full cycle. Among the three target species, PAR is the only species with consistent count and bag trends. We speculate that this feature derives from peculiar characteristics of PAR hunting in Italy. Firstly, bags are hardly biased by weather conditions during the hunting season, as in PTA hunting, since PAR avoids slopes with persistent snow cover and it is then pushed to lower altitudes by snowfalls, where it is reached by hunters without difficulty. Secondly, PAR hunting is practiced by fewer hunters compared to BG hunting, which is the most popular species among *tipica alpina* hunters. This leads to a high-quality standard of post-breeding counts, because they are often run by few skilled hunters and dogs over the years, with high knowledge of the eco-ethology of PAR and their hunting grounds, thus reducing the bias introduced by the observers.

Albeit managed by regulations, the role of hunting as a threat to Galliform conservation is debated (Aebischer 1991; Ellison 1991; Sandercock et al. 2011). We showed that hunting pressure is remarkably different on the Italian Alps. As already mentioned, this denotes that Galliform hunting is not evenly important within local traditions of the western and eastern Alps, but the highest pressures are found in the absence of regional control on game management, such as Lombardy, in which management used to be entirely

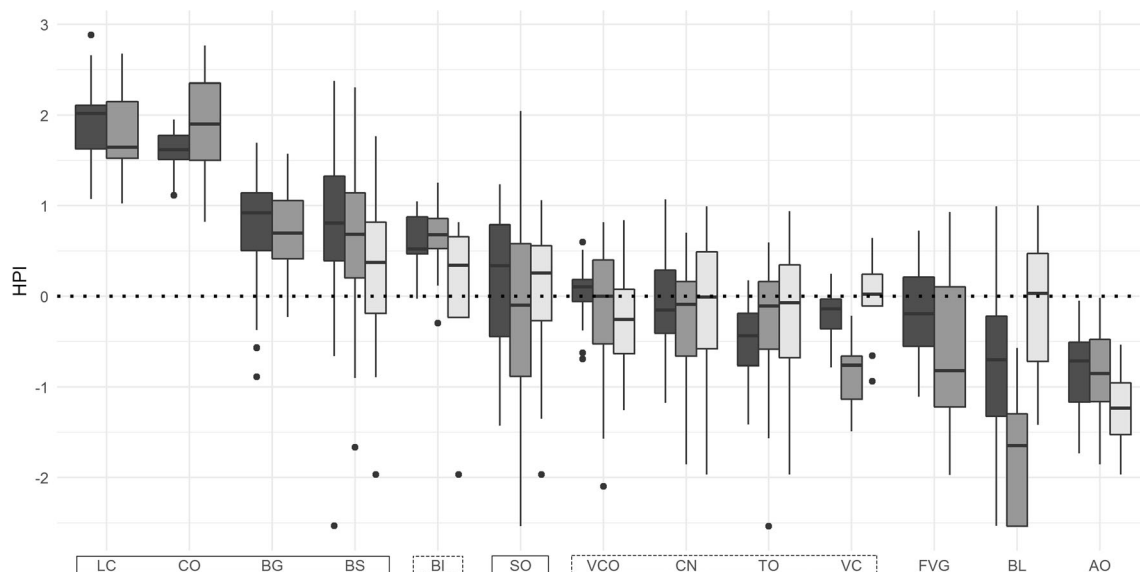


Fig. 3 Comparison of hunting pressure across authorities and species (dark grey BG; intermediate grey PAR; light grey PTA). HPI (hunting pressure index) is calculated as the log-transformed ratio of the annual bag size of each GMU and estimated extent of suitable habitat for each

species and GMU, minus the species' median value. The plot is sorted by authorities' median values. The highest values are found in the Lombardy (solid outline) and Piedmont (dashed outlines) regions

delegated to the provincial scale until 2015. We are unable to evaluate whether higher pressures are locally affecting species' conservation, since more robust data on spring abundance and breeding success are required. Nonetheless, this lack of common guidelines at a larger scale leaves populations at the risk of overexploitation.

Conclusions

Both bag and count data of Italian populations of Alpine Galliforms are subject to species-specific flaws and careful considerations should be taken while assessing the conservation status of these species at a national scale, in particular BG and PTA. For BG, no general conclusions on the current trend can be drawn. Given that Alpine Galliforms are all listed in the Annex I and II of the Birds Directive, we highlight the need for a national survey scheme and national game management guidelines applied uniformly across the Italian Alps. However, since 2015, there is an ongoing process of transferring of the hunting authority from provinces previously in charge to regions, which have already started drafting guidelines. This could be considered a first step towards a national standardization. Here, below, we summarized some potential improvements to current enforced game management.

- **Data gathering.** Count and bag data should be archived in a national georeferenced database, on the model of ungulate management, for which Italy has already defined a protocol on data flow from local administrations to national institutions. This would improve the evaluation of the conservation status and the hunting pressure at the national scale, quickly providing information to researchers and policymakers, in compliance with reporting requirements of the European Union. Notable examples of national hunting statistics and wildlife data databases exist in Fennoscandian (e.g. Finland, Helle et al. 2016; Norway, www.ssb.no) as well as in Alpine countries (France, www.observatoire-galliformes-montagne.com; Switzerland, www.vogelwarte.ch).
- **Counts.** Sampling areas should be standardized across administrations in terms of number and average size. The extent of each area should be large enough to include both winter/spring and summer/autumn ranges, according to the species eco-ethology. In a perspective of resource optimisation, we suggest the choice of few large areas instead of several small areas. Within each sampling area, spring and summer count parcels should be defined. Spring counts should become mandatory and annual shooting plans should be set taking into account not only the breeding success but also population trends deriving from spring counts. Sampling methods (point counts, line transects) should be standardized whenever possible; the

choice should take into account local environmental conditions and human resources available. We recommend the application of repeated counts, at least in spring, in order to minimize bias from weather conditions and to allow the utilization of population modelling techniques that include species' detectability (e.g. N-mixture models, Kéry and Schaub 2011). Collected data variables should be standardized, with specification of sex and age when possible. Derived parameters such as breeding success or population density should be calculated from defined criteria.

- **Hunting statistics and plans.** Shooting plans should be set with standardized criteria, taking into account, as already mentioned, population trends, population densities and breeding success. Similarly to counts, we recommend the collection of sex and age information of harvested birds in order to estimate sex ratio and to compare juvenile recruitment from summer counts and game bags. Finally, information of number of hunting licences and hunting days in each hunting district should be collected at the national scale to allow hunting effort estimation as hunting person-days.

In our opinion, application of these “good practices” will overcome the problem of different population trend estimation from different proxies, leading to a robust monitoring of conservation statuses both at the national and local scales in compliance with the Birds' Directive demands.

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Compliance with ethical standards

Ethical statement The hunting of black grouse, rock ptarmigan and rock partridge in Italy complies with the national law on wildlife protection, Legge 11 febbraio 1992, n. 157 “Norme per la protezione della fauna selvatica omeoterma e per il prelievo venatorio”, and with local, regional or provincial regulations. No animals were shot for the specific purposes of the study.

Conflict of interest The authors declare that they have no conflict of interest.

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