



Habitat selection and density of common pheasant (*Phasianus colchicus*) in Northern Italy: effects of land use cover and landscape configuration

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Abstract

Knowing the ecology of game species is important to define sustainable hunting pressure and to plan management actions aimed to maintain viable populations. Common pheasant (*Phasianus colchicus*) is one of the main gamebird species in Europe and North America, despite its native range extending from the Caucasus to Eastern China. This research aimed to define the environmental variables shaping the spatial distribution of male pheasants and to estimate their breeding density in an agroecosystem of northern Italy. During the breeding season, 2015, we carried out 372 point counts with unlimited distances, randomly placed following a stratified sampling survey design. The habitat requirements of the pheasant were evaluated following a presence vs. availability approach, using environmental variables related to land use cover and landscape configuration. We built generalized linear models with a binary distribution, selecting variables following an information-theoretic approach. Densities were estimated through both conventional and multiple-covariate distance sampling. We estimated a density of 1.45 males/km², with 4.26 males/km² in suitable areas and 0.91 males/km² in unsuitable ones. We found pheasants in areas with meadows and tree plantations, which were used to find food and refuges from predators and bad weather conditions. Similarly, woodlands have a positive effect on species occurrence, whereas arable lands were avoided, specifically maize and paddy fields. We found little evidence that landscape configuration affects pheasant occurrence. We found pheasants to be negatively affected by the length of edges between woodlands and arable lands, whereas edges between woodlands and grasslands seem to be beneficial for the species. These findings could help landscape and wildlife managers to plan habitat improvement actions useful to maintain self-sustaining populations of this species, by increasing cover of woodlands, meadows, and tree plantations.

Keywords Distance sampling · Edges · Galliformes · Gamebirds · Multi-model inference · Resource selection probability function

Introduction

Common pheasant (*Phasianus colchicus* Linnaeus, 1758) (hereafter, pheasant) is one of the main gamebird species in Europe, with 10–57 million birds released annually in the UK (Mason et al. 2020; Madden 2021). The introduction of pheasants into hunting grounds is a widespread practice also in other countries of Western Europe, such as in France (with 10–15 million pheasants released annually), though remains rare

in Fennoscandia (for instance, in Finland, 15,000 pheasants were released each year) (Arroyo and Beja 2002; Mustin et al. 2012). Its native range extends from Russian Far East, Korea, Eastern China, and Northern Vietnam, across Central Asia, west to the Caucasus, and western shores of the Caspian Sea. European native populations are found in Russia, and notably in Georgia, with small populations also present in Turkey, Greece and Armenia (BirdLife International 2021). However, it was introduced all over the world for hunting purposes, with viable and widespread populations mainly in Europe and North America (del Hoyo et al. 1994). In its native range, it is found mainly in the overgrown edges of rivers, hilly areas close to cultivations having small thickets, and flat arable lands. Introduced populations breed mostly in arable lands, including mosaics of cereal crops, grasslands, fallows, small woodlands,

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and hedgerows (Johnsgard 1986; del Hoyo et al. 1994). The European native population is estimated at 9,700–16,300 mature individuals with a stable trend, whereas the overall European population, including introduced populations, is estimated at 4,140,000–5,370,000 males, with an increasing trend (BirdLife International 2017, 2021). For this reason, the species was classified as least concern by IUCN (*International Union for Conservation of the Nature*) (BirdLife International 2021). Nonetheless, some populations are declining locally owing to habitat loss and over-hunting (BirdLife International 2016). In Italy, it is impossible to define its current distribution and to estimate the size and trend of the breeding population owing to the massive releases of birds for hunting because most stocks survive just a few months between shooting seasons. However, self-sustaining populations thrive in protected areas or in well-managed estates, where a few thousand territorial males can be estimated (Brichetti and Fracasso 2018). Furthermore, the conservation status of the Italian population is not evaluated by IUCN because it is considered an introduced species (Gustin et al. 2019). The aims of this research were (i) to define the environmental variables shaping the spatial distribution of male pheasants and (ii) to estimate its breeding density in an agroecosystem of northern Italy. We expected the species to be positively affected by meadows, hedgerows, woodlands, and tree plantations while avoiding a large amount of maize and paddy fields (Meriggi 1983, 1992; del Hoyo et al. 1994; Genovesi et al. 1999; Leif 2005; Nelli et al. 2012). This research is relevant because despite the pheasant being an important game species, there are few studies about its ecology in northern Italy (Meriggi 1983; Meriggi et al. 1996; Genovesi et al. 1999; Nelli et al. 2012). Moreover, knowing the environmental factors affecting the distribution and density of game species is very important to define sustainable hunting pressure and to plan management actions aimed to maintain viable populations (Sinclair et al. 2006; Sands et al. 2012; Robertson et al. 2017). Furthermore, increasing knowledge about gamebirds' ecology could be helpful to better understand the ecological effects occurring in areas where they are released and managed (Madden 2021). Additionally, few studies are aimed to investigate the ecology of common species, such as the pheasant (EBCC 2019); therefore, if the preservation of overall species richness patterns is considered a valid conservation goal, then attention must be given to common species as well as rare ones (Lennon et al. 2003).

Materials and methods

Study area

This study was carried out in the western Po Plain (north-western Italy), on a surface of about 2900 km² (45° 11' N 9° 05' E) (Fig. 1). The Ticino River crosses the study area from

north to south, flowing into the Po River that runs from west to east. The Sesia River and the Lambro River flow along the western and the eastern boundaries of the study area, respectively. The landscape is characterized by cultivated areas, especially paddy fields (39.4%), other annual crops (mainly maize, soybean, oil-seed rape, winter wheat, and alfalfa) (29.1%), and tree plantations (6.8%). Broad-leaved forests and urban areas represent 4.9% and 10.3% of the total area, respectively (ERSAF 2014). Continuous forests (composed mainly of *Quercus robur*, *Carpinus betulus*, *Salix* sp., *Poplar* sp., and the invasive *Robinia pseudoacacia*) are located along the Ticino River and in the southern part of the study area, near the Apennine slopes, whereas remnants of broad-leaved forest fragments are scattered in the cultivated area (95% of which are smaller than 10 ha) (ERSAF 2012). Tree plantations are mainly constituted by monospecific plantations of hybrid poplar species, but also of reforestations and short rotation coppices (SRCs). Due to its particular geographical location, this area is difficult to relate to other European lowlands. Even though the Po Plain presents a subcontinental climate, it is separated from the rest of continental Europe by the Alps, which could act as a zoogeographical barrier (Bianco 1990; Hermansen et al. 2011). Furthermore, even if it is attributable to the Mediterranean Basin, the climate, the land use, and the vegetation in the Po Plain differ from the rest of the Mediterranean area (Mikusiński and Angelstam 1997; Capotorti et al. 2012).

Survey design and data collection

To obtain a representative sample in this large and heterogeneous study area, a stratified sampling design was planned (Krebs 1999; Sutherland 2006). According to this approach, the area was firstly divided into landscape units (LUs), i.e., homogeneous portions of the area in terms of habitat or ecological characteristics likely to induce systematic variations in population density (Sutherland 2006). To this aim, a 2 × 2 km grid of sampling cells was superimposed on the study area and each cell was assigned to a LU, based on its characteristics. More details on LU definition are reported in Appendix S1 (Supplementary Materials, SM). Then, data collection was conducted in 62 sampling cells randomly selected (approximately 10% of the study area). The number of sampling cells selected for each LU was proportional to the LU extent, in such a way that all the landscape characteristics of the study area were represented and each stratum was investigated with an effort proportional to its surface (Krebs 1999). Within each selected sampling cell, six point counts were carried out (Bibby et al. 2000; Sutherland 2006) during the breeding season 2015, between May and early June, because in this period, there is the peak of the call and the breeding activity (Cramp and Simmons 1980; Brichetti and Fracasso 2004). Data on species occurrence

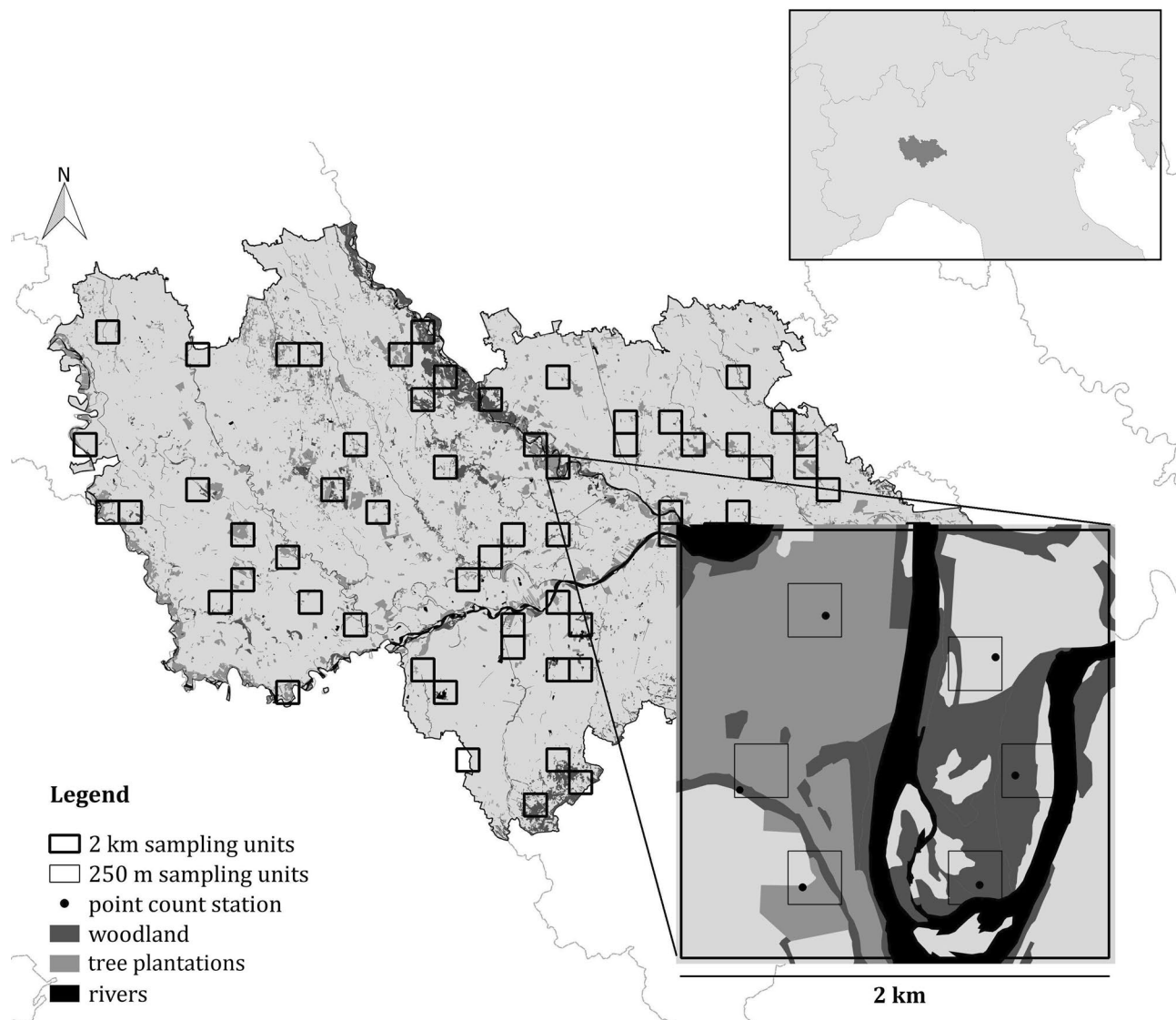


Fig. 1 The survey design used in the study. Both the 2-km and the 250-m sampling units are shown, as well as the point count stations. The location of the study area in northern Italy is shown in the insert

were therefore collected in 372 point counts. The selection of the six point counts inside the 2-km cell was carried out by a multi-level sampling (Sutherland 2006), subdividing the 2 km cells into cells of 250×250 m and randomly selecting six of them (Fig. 1). By using this second grid, it was possible to obtain more accurate estimates. Each point, placed randomly inside the cells, was surveyed once from dawn to 10:30, and the count lasted for 10 min (Chamberlain and Rolando 2014); overall, data collection took place for 32 effective days. During the fieldwork, we measured the exact distance from the observer to the birds with a laser rangefinder (Leica Rangemaster 900; Leica, Solms, Germany). When we did not see a calling bird (23% of cases),

we mapped its approximate position on aerial photographs (1:5000 scale) based on the likely attenuation and direction of its vocalization. We then measured the distance from observer to the position of each calling bird using the software QGIS v.3.14.16 “Pi.”

Environmental variables

The habitat requirements of the pheasant were evaluated following a presence vs. availability approach (Manly et al. 2002; Boyce et al. 2002). The environmental variables (Table 1) were measured at the home range scale (Manly et al. 2002; Morrison et al. 2006) at presence sites and

Table 1 Environmental variables used to investigate the habitat selection of the common pheasant in northern Italy (values are measured in the 359-m grid). Details on landscape metrics are provided in SM Table S1. In bold are the variables retained after their reduction and used in the final GLM

Environmental variables	Mean \pm SD	min–max
<i>Land use cover</i>		
Urban areas (%)	9.6 \pm 0.075	0.0–100.0
Cereal crops (e.g., winter wheat, barley) (%)	5.8 \pm 0.056	0.0–100.0
Maize (%)	11.0 \pm 0.077	0.0–100.0
Paddy fields (%)	32.7 \pm 0.130	0.0–100.0
Fodder crops (e.g., alfalfa, clover) (%)	4.5 \pm 0.050	0.0–100.0
Industrial crops and legumes (e.g., rape, soybean) (%)	4.6 \pm 0.052	0.0–100.0
Meadows (%)	2.6 \pm 0.035	0.0–100.0
Permanent crops (e.g., vineyards, orchards) (%)	2.4 \pm 0.041	0.0–100.0
Poplar plantations (%)	6.0 \pm 0.061	0.0–100.0
Other tree plantations (i.e., reforestation, SRCs) (%)	0.6 \pm 0.019	0.0–100.0
Woodlands (%)	1.0 \pm 0.026	0.0–100.0
Shrublands (%)	1.2 \pm 0.022	0.0–100.0
Fallows (%)	1.4 \pm 0.021	0.0–100.0
Rivers and water bodies (%)	1.7 \pm 0.033	0.0–100.0
<i>Landscape configuration</i>		
Hedgerows density (m/ha)	12.93 \pm 0.081	0.0–294.53
Habitat heterogeneity (Shannon index)	1.29 \pm 0.516	0.0–2.605
Aggregation index (%)	68.6 \pm 15.931	0.0–100.0
Mean patch size (ha)	6.11 \pm 5.16	1.0–64.0
Edge density (m/ha)	69.77 \pm 29.455	0.0–153.06
Mean fractal dimension	1.02 \pm 0.011	1.0–1.113
Number of patches	10.78 \pm 5.432	1–36
Mean perimeter-area ratio	0.03 \pm 0.004	0.005–0.040
Mean shape index	1.17 \pm 0.082	1.0–1.95
Edge woodlands/arable lands (m)	74.9 \pm 176.19	0.0–1819.5
Edge woodlands/grasslands (m)	32.6 \pm 112.9	0.0–1658.7
Edge shrublands/arable lands (m)	16.6 \pm 65.9	0.0–871.5
Edge shrublands/grasslands (m)	6.3 \pm 35.3	0.0–735.7

subsequently compared with those of an equal number of availability sites which were randomly selected in the study area without any constraints (Manly et al. 2002; Boyce et al. 2002). We measured variables related both to land use cover ($N=14$) and to landscape configuration ($N=13$); formulas and descriptions of landscape metrics used (McGarigal and Marks 1994) are provided in SM Table S2. In addition to these metrics, we measured also the edges (in meters) between woodlands and both arable lands and grasslands, as well as between shrublands and both arable lands and grasslands. To take into account the spatial ecology of the species (Brennan et al. 2002; Manly et al. 2002), presence/availability sites corresponded to cells the extent of which was defined based on the home range size of the species during the breeding season, as suggested by the available literature. Specifically, the home range of the species is equal to 2.1–52.7 ha (for details and references, see SM Table S2); hence, considering an average of about 18.4 ha, a grid was generated with square cells of 359 m on side.

Habitat suitability

We used generalized linear models (GLM) with a binary distribution (link function = logit) (Rushton et al. 2004; Keating and Cherry 2004) to relate the presence of male pheasants to the environmental variables. The presence of the species was 1 when at least one male was detected in the 359 m grid cells, or 0 otherwise. To model the spatial distribution of pheasant, first, we selected only the variables with a remarkable effect on its occurrence (therefore with some evidence of importance), with a pairwise comparison of the second-order Akaike information criterion (AIC_c ; Akaike 1973) of two simple GLM: one with the intercept only and the other with each variable (Burnham et al. 2011; Chiantante et al. 2021). When the AIC_c value of the GLM with the variable was lesser than the one with the intercept only (the null model), with a difference of at least two ($\Delta AIC_c \leq 2$), that variable was retained (Burnham and Anderson 2002). Once the number of variables was reduced, we ran a priori

sets of GLM built with all combinations of the retained environmental variables; these models were built using not correlated variables ($|r| < 0.70$). Then, for each model, the AIC_c was calculated and the models with the lowest AIC_c ($\Delta AIC_c \leq 2$) were selected as the best (Burnham and Anderson 2002). These models were averaged and the importance of each variable (Σw_i) in the set was calculated (Burnham and Anderson 2002). For this analysis, all the variables considered were standardized by normalization; that is, each variable had a mean of zero and a standard deviation of one (Quinn and Keough 2002; Zuur et al. 2007). The model's ability to distinguish between occupied and available sites was tested through the area under the curve (AUC) of the receiver operating characteristic plot (ROC curve) (Pearce and Ferrier 2000; Fawcett 2006). Moreover, we tested the residuals' spatial autocorrelation by the Moran I test (Zuur et al. 2007; Bivand et al. 2008). We used the variance inflation factor (VIF) with a threshold of 3 to exclude variable collinearity (Fox and Monette 1992; Zuur et al. 2010) and the explained deviance D^2 was used as a measure of the variance explained by the model (Zuur et al. 2007). Finally, we predicted the probability of occurrence of the pheasant in the study area. Precisely, using a grid covering the whole area and with a spatial resolution of 359 m, first, we measured for each square of the grid the environmental variables included in the model. Then, after their standardization (as we did to build the models), we reclassified these values using the coefficients estimated by the average model. In this way, we obtained the probability of occurrence of the species for each square of the grid, allowing us to calculate the average (\pm SD) probability of occurrence of the pheasant in the study area. Data used to measure the environmental variables were obtained from the regional land use map *DUSAF 5.0* (ERSAF 2017) and the regional forest map "*Carta dei tipi forestali reali della Lombardia*" (ERSAF 2012) and processed by the software Quantum GIS v.3.14.16 "Pi." All the analyses were performed using the statistical software R v.3.3.2 (R Core Team 2019) and the packages *MuMIn* (Tillé and Matei 2016), *car* (Fox and Weisberg 2011), *landscapemetrics* (Hesselbarth 2021), *spdep* (Bivand et al. 2015), and *ROCR* (Sing et al. 2007).

Density estimation

The density of male pheasants was estimated through the distance sampling method, particularly the conventional distance sampling (CDS) (Buckland et al. 1993). After a visual inspection of distances distribution, we truncated the 10% of the greatest distances as suggested by Buckland et al. (1993) and transformed the distance data into equal intervals of 60 m. As suggested by Buckland et al. (1993) and

Thomas et al. (2010), we tested the following combinations of key functions and series adjustments: (1) uniform key with cosine adjustments, (2) half-normal key with cosine adjustments, (3) half-normal key with Hermite polynomial adjustments, and (4) hazard-rate key with simple polynomial adjustments. Anyway, the probability of detecting a bird depends not only on distance but also on many other factors, such as habitat, weather, observer, and bird behavior (Buckland et al. 1993), a circumstance that could exist, at least in part, in this research due to the variability of our data. Therefore, ignoring all these other factors, besides distances, could cause some bias in the estimate (Beavers and Ramsey 1998; Bas et al. 2008; Anderson et al. 2015). For this reason, besides CDS, we used also multiple-covariate distance sampling (MCDS) (Marques et al. 2007), an extension of CDS that allow modelling the detection probability as a function of variables other than distance. Therefore, a graphical exploratory analysis was run to assess if the habitat around points might bias the estimate of the density. We considered the cover of open areas (arable lands, meadows, and fallows), shrublands, and arboreal vegetation (woodlands, poplar plantations, other tree plantations) in a 300-m circular plot around the sampling points (Chiatante et al. 2020, 2021). The results of this analysis (ESM Figure S2) showed that both the cover of open areas ($r_s = 0.223$, $P = 0.003$) and arboreal vegetation ($r_s = -0.255$, $P < 0.001$) could bias our estimate because the detection distance changed with them; the correlation with shrublands was not significant ($r_s = -0.072$, $P = 0.337$). Therefore, we used the cover of open areas and arboreal vegetation as covariates. To estimate density with MCDS, only half-normal and hazard-rate keys are allowed (Marques et al. 2007). We built the detection functions using the habitat suitability as strata, i.e., suitable vs. unsuitable areas. To this purpose, we reclassified the map of probability of occurrence (see section "[Habitat suitability models](#)") into a binary map of suitable/unsuitable areas using as threshold the average probability in presence sites (equal to 0.803) (Liu et al. 2005). Once the models were computed, ran both with CDS and MCDS, we used Akaike information criterion (AIC) to select the best model (Buckland et al. 1993; Thomas et al. 2010). Considering that data were binned, the goodness-of-fit of the models was assessed by χ^2 tests (Buckland et al. 1993; Thomas et al. 2010). Finally, the average probability of detection was estimated and the effective detection radius (EDR) was defined. For each estimate, both the coefficient of variation (CV) and the 95% confidence intervals (CI) were calculated. The analyses were performed using the statistical software R v.3.6.1 (R Core Team 2019) and the package *Distance* (Miller et al. 2019).

Results

Habitat suitability

We detected 183 male pheasants in 100 sampling points, corresponding to 139 cells (42.2%). The pairwise comparison suggested that 10 environmental variables affected in a remarkable way the species occurrence (SM Tables S3 and S4). These almost encompassed exclusively land use variables, but also two variables of landscape configuration. Indeed, we found pheasants in areas with meadows, tree plantations (including poplar plantations), and woodlands, in the proximity of water bodies and with long edges between woodlands and grasslands (SM Figure S3). Contrarily, they avoided urban areas, arable lands (especially maize and paddy fields), and areas with long edges between woodlands and arable lands (SM Figure S3).

Nine models best explained the occurrence of the pheasant (SM Table S5). The average GLM showed that four variables affected in a marked way the pheasant occurrence (Table 2, Fig. 2). In particular, meadows and poplar plantations increased its probability of occurrence, while urban areas and edges between woodlands and arable lands decreased it. Contrarily, the length of edges between woodlands and grasslands had a positive effect on the species distribution. The VIF revealed no collinearity among variables (Table 2) and the model residuals did not show any spatial correlation (Moran test, $I=1.051$, $P=0.668$). ROC analysis showed an excellent ability of the model to distinguish between occupied and random sites (AUC = 0.945, $P < 0.001$) and the explained deviance D^2 was equal to 55.8%. The average probability of occurrence estimated in the study area by the model was equal to 0.482 ± 0.273 (SD) (min. 0.00, max. 1.00) (SM Figure S4).

Density estimation

We collected 179 observations of 183 pheasants. On average, 1.02 pheasants per point were detected (SE = 0.018, min = 1, max = 4). The best detection probability function was the hazard-rate with simple polynomial adjustments with open areas and arboreal vegetation as covariate (Table 3; Fig. 3). The goodness-of-fit of the model was good ($\chi^2 = 0.622$, df = 2, $P = 0.733$). This model gave an EDR of 273 m and the average probability of detection was estimated to be 0.422 ± 0.050 SE. The estimated density in unsuitable areas was 0.91 males/km² (SE = 0.204, LCI 95% = 0.60, UCI 95% = 1.42, CV = 22.2%), whereas in suitable areas was 4.26 males/km² (SE = 0.854, LCI 95% = 2.88, UCI 95% = 6.30, CV = 20.1%). In general, the density of pheasant estimated in the study area was 1.45 males/km² (SE = 0.240, LCI 95% = 1.06, UCI 95% = 2.01, CV = 16.5%).

Discussion

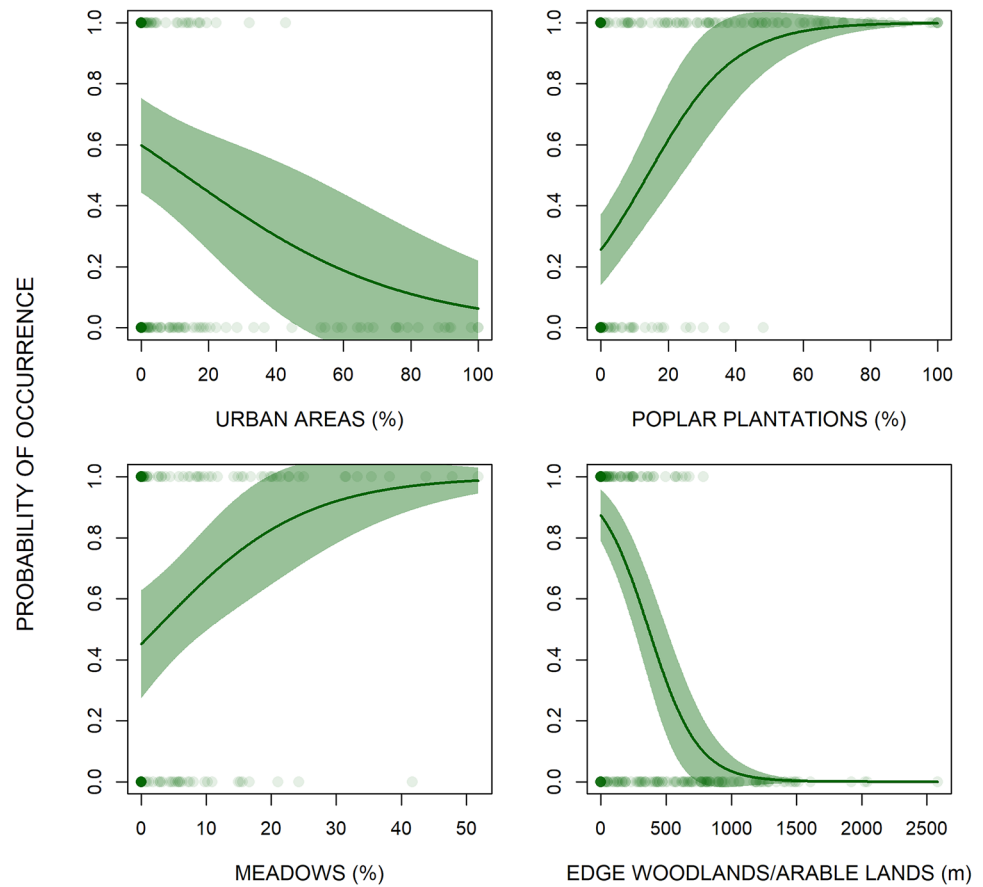
This research was aimed to explore the relationships between the occurrence of male pheasants during the breeding season and environmental variables, particularly land use cover and landscape configuration. As expected, we found them in areas with meadows and poplar plantations. Pheasants typically select meadows, grasslands, and other herbaceous areas (Wasilewski 1986; Leptich 1992; Clark et al. 1999; Smith et al. 1999; Holá et al. 2015). Nonetheless, in some cases, pheasants avoid grasslands during the breeding season (Genovesi et al. 1999; Smith et al. 1999) likely to avoid predators (Smith et al. 1999). This happens especially with tall dense vegetation cover, having significantly more predated individuals than the short or sparse

Table 2 The average GLM explaining the occurrence of the common pheasant in northern Italy

Environmental variables	β	SE	LCI 95%	UCI 95%	Σw_i	VIF
Intercept	0.133	0.326	-	-	-	-
Urban areas	-0.652	0.282	-1.208	-0.096	1.00	1.027
Meadows	0.768	0.298	0.181	1.355	1.00	1.281
Poplar plantations	2.004	0.477	1.065	2.944	1.00	1.054
Other tree plantations	0.442	0.330	-0.207	1.091	0.61	1.057
Woodlands	0.318	0.219	-0.113	0.748	0.55	1.174
Rivers and water bodies	0.285	0.202	-0.113	0.683	0.56	1.052
Edges woodlands/arable lands	-2.469	0.438	-3.331	-1.606	1.00	1.355
Edges woodlands/grasslands	1.087	0.617	-0.127	2.301	0.94	1.146

β standardized coefficient for model predictors, SE unconditional standard error, LCI 95% 95% lower confidence interval, UCI 95% 95% upper confidence interval, Σw_i relative importance, VIF variance inflation factor

Fig. 2 The response curves of the most important environmental variables ($\Sigma w_i = 1$) selected in the average GLM built to investigate the occurrence of the common pheasant in northern Italy



cover (Smith et al. 1999). Regarding poplar plantations, they are commonly used during the breeding season in northern Italy (Meriggi et al. 1996). Likewise, other tree plantations, such as reforestations and short rotation coppices, have a positive effect on the species occurrence. This is not surprising, because the selection for this land use type is well known, both in the native range (Ashoori et al. 2018) and in Europe (Baxter et al. 1996; Nelli et al. 2012; Chiatante et al. 2019). Particularly, Baxter et al. (1996) found the species more often in willow coppices than in poplar coppices. Still, Nelli et al. (2012) found a negative relationship between crowing males and distance from reforestations; similarly, densities increased with a higher cover of reforestations.

Indeed, these kinds of tree plantations, when mature, show characteristics that make them similar to natural woodlands, having a high density of stems near ground level, a characteristic that seems to enhance pheasant survival (Leif 2005). Therefore, pheasants can use tree plantations to find food and refuges from predators and bad weather conditions (Baxter et al. 1996; Leif 2005; Nelli et al. 2012). Similarly, woodlands have a positive effect on the species occurrence, although this relationship was found only for univariate analyses. However, this result is in accordance with previous research. In general, this species avoids large forests but a small amount of woody vegetation in the landscape, as in our study area, is beneficial for this species (Hill and Robertson

Table 3 Distance sampling models computed to estimate the density of common pheasants in northern Italy. We showed the function (key + series adjustment), the model used, the AIC and its Δ , and the average estimated detection probability (P_a)

Function	Model	AIC	Δ AIC	P_a (mean \pm SE)
MCDS hazard-rate simple polynomial	~ OPEN + ARB	605.32	0.00	0.422 \pm 0.050
MCDS half-normal cosine	~ OPEN + ARB	605.36	0.04	0.333 \pm 0.033
MCDS half-normal Hermite polynomial	~ OPEN + ARB	605.36	0.04	0.333 \pm 0.033
CDS uniform cosine	~ 1	606.95	1.63	0.359 \pm 0.021
CDS half-normal cosine	~ 1	607.11	1.80	0.343 \pm 0.033
CDS half-normal Hermite polynomial	~ 1	607.11	1.80	0.343 \pm 0.033
CDS hazard-rate simple polynomial	~ 1	609.04	3.72	0.410 \pm 0.052

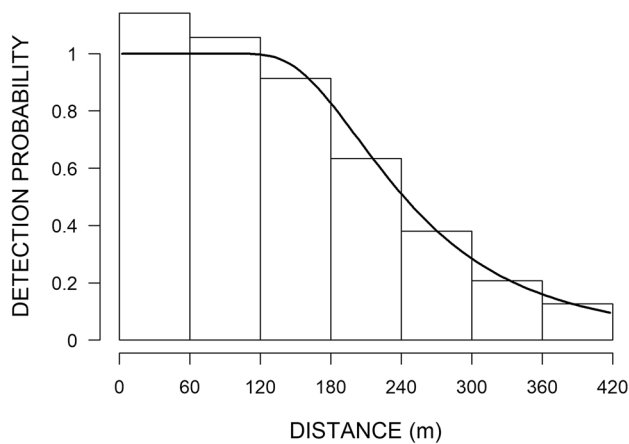


Fig. 3 Histograms of the detection function calculated to estimate the density of the common pheasant in northern Italy. On the y-axis, the detection distance in meters, on the x-axis the detection probability (from 0 to 1)

1988; del Hoyo et al. 1994; Ronnenberg et al. 2016; Keller et al. 2020). Indeed, the cover provided by woody habitats reduces home range sizes and enhances home range suitability for breeding male pheasants (Leif 2005). In addition, woody habitats provide shelter and refuge from hunters and predators throughout the year (Hill and Robertson 1988; Leif 2005; Holá et al. 2015). Nonetheless, in North America, generally, woody vegetation does not affect the occurrence and abundance of pheasants (Riley et al. 1998; Schmitz and Clark 1999). Besides, we did not find any relation with hedgerow density or shrubland cover. This result is a little unexpected because hedgerows and shelterbelts are usually essential for the species occurrence (Hill and Robertson 1988; Gabbert et al. 1999; Leif 2005). Similarly, also Nelli et al. (2012) did not find any relation with hedgerows, probably due to the low presence of these landscape features in their study area, which could also be true in our case, considering that on average, there are only 13 m of hedgerows per hectare. Univariate analyses showed that in the study area, arable lands are avoided, specifically maize and paddy fields. This is in accordance with the fact that maize and paddy fields are not suitable habitats during the breeding season (Meriggi et al. 1996). Indeed, in North America, conversion of barley to paddy fields has been the strongest driver of habitat loss for the species likely because they are flooded for much of the growing season (Coates et al. 2017). On the other hand, in Germany, with approximately 15–35% of maize cover, the effect is moderately positive, whereas, at the highest percentages, maize has a negative effect on the species densities (Ronnenberg et al. 2016). In general, cultivated areas were positively selected by pheasants (del Hoyo et al. 1994; Keller et al. 2020). In North America, pheasants are largely limited to irrigated croplands (Johnsgard 1986), selecting wheat and alfalfa during the breeding

season (Whiteside and Guthery 1983) but avoiding maize and soybean (Clark et al. 1999; Leif 2005). Selection for irrigated crops might be due to the higher availability of arthropods in moist soils with respect to dry ones (Hart et al. 2009). In addition, irrigated croplands require ditches and canals that could provide both arthropods and safe places, especially if grassed. Nonetheless, intensive agriculture and changes in agricultural practices are detrimental for pheasants (Ronnenberg et al. 2016; Coates et al. 2017). For instance, monocultures and the consequent use of herbicides and insecticides reduce both the availability of refugia and food resources, increasing the effects of predation and mortality (Rands 1986; Meriggi et al. 1996; Riley and Schulz 2001; Coates et al. 2017). The analyses showed also a positive effect of rivers and water bodies, which is common for the species both in native and introduced ranges (Cramp and Simmons 1980; Johnsgard 1986; Holá et al. 2015). Indeed, in Russia, it occurs in shrublands and thickets in river valleys and desert-adapted subspecies are largely limited to riverine areas or other areas of available fresh water (Johnsgard 1986). Similarly, in the western Palearctic, the pheasants occur in broad river valleys. Likewise, wetlands are very important habitat throughout the year (Gatti et al. 1989; Leptich 1992; Smith et al. 1999; Ronnenberg et al. 2016). Avoidance of urban areas is also in accordance with general habits of the species. For instance, road density has a strong negative effect on the species, probably because the increase mortality rate of individuals is caused by intensive road traffic (Holá et al. 2015; Madden and Perkins 2017).

We found little evidence that landscape configuration affects pheasant occurrence, which was shown to be important in other studies. For instance, Clark et al. (1999) found that areas with small patches of grasslands clustered have a moderate probability to be selected. On the other hand, it was shown that patches with smaller sizes and characterized by long and irregular edges are more suitable for pheasants, compared to bigger and more regular ones (Baxter et al. 1996; Nelli et al. 2012). Moreover, habitat heterogeneity was shown to be important, as in Germany, where municipalities with a low crop diversity host fewer pheasants than more diverse areas (Ronnenberg et al. 2016). The only effect we found was related to edges, which are generally beneficial for the pheasant (Clark et al. 1999; Genovesi et al. 1999). Particularly, we found pheasants to be negatively affected by the length of edges between woodlands and arable lands, which agrees with the general avoidance of arable lands we found. Generally, in croplands, pheasants spent most of their time in spontaneous vegetation growing along drainage ditches, field edges, roadsides, and shelterbelts (Warner and Joselyn 1986; Wasilewski 1986; Meriggi et al. 1996; Genovesi et al. 1999). Contrarily, we found a positive effect of edges between woodlands and grasslands. This could be due to the higher suitability of meadows with respect to arable lands we observed, related

also to lesser disturbances common to cultivated areas (e.g., due to spraying, weed control). Anyhow, along edges, vulnerability to predation is high (Schmitz and Clark 1999; Kuehl and Clark 2002); therefore, one may ask why pheasants do not avoid edges that increase mortality risk, perhaps because breeding activity around edges is a stronger factor in habitat selection that compete with predator avoidance (Schmitz and Clark 1999) and the selection of edges between woodlands and grasslands is a better compromise with respect to use edges between woodlands and arable lands because of their higher suitability.

The second aim of this study was to estimate the pheasant density. Our results showed a density of 1.45 males/km², with 4.26 males/km² in suitable areas and 0.91 males/km² in unsuitable ones. These values are below other densities estimated in northern Italy. Nelli et al. (2012) estimated densities of 6.6–27.0 males/km² in four protected areas near Milan. However, in these areas, habitat improvements (e.g., game crops, stubble maintenance, and maintenance of linear grassy vegetation) were carried out to increase the density and productivity of wild pheasants. Other Italian studies estimated densities between 2.8 and 255 pheasants/km², but these values are likely biased by releases of birds for hunting (Brichetti and Fracasso 2018). Our results differed from other estimates likely because of the method of data analysis in as much these estimates usually were calculated in relation to fixed-radius plots, which assume the perfect detectability. Indeed, many studies have found differences between indices of relative abundances calculated within fixed-radius plots and distance sampling estimates (Norvell et al. 2003; Buckland et al. 2008; Gottschalk and Huettmann 2011). All of them stressed the more convincing estimates of distance sampling, which gives precise unbiased results with respect to a study design based on a relative abundance approach, in as much it takes into account the variable detectability of individuals (Buckland et al. 1993). However, relative abundances reflect species responses to ecological gradients and can be easily employed to investigate trends, therefore are very useful in some circumstances (Hutto and Young 2003; Johnson 2008).

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Data availability The datasets generated during and/or analyzed during the current study are available from the corresponding author on a reasonable request.

Declarations

Ethics approval This research was conducted with ethical approval from the University of Pavia (Department of Earth and Environmental Sciences). Bird surveys were conducted with permission from local landowners where necessary. Data collection did not involve sampling procedure, and experimental manipulation of birds and the fieldwork was conducted under the Law of the Republic of Italy on the Protection of Wildlife (February 25, 1992).

Conflict of interest The authors declare no competing interests.

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