Effects of habitat improvement actions (HIAs) and reforestations on pheasants Phasianus colchicus in northern Italy

Authors: Nelli, Luca, Meriggi, Alberto, and Vidus-Rosin, Anna

Source: Wildlife Biology, 18(2) : 121-130

Published By: Nordic Board for Wildlife Research

URL: https://doi.org/10.2981/11-022

Wildl. Biol. 18: 121-130 (2012)
DOI: 10.2981/11-022

www.wildlifebiology.com

Original article

Effects of habitat improvement actions (HIAs) and reforestations on pheasants *Phasianus colchicus* in northern Italy

Luca Nelli, Alberto Meriggi & Anna Vidus-Rosin

Over the last 50 years, the modernisation and mechanisation of agricultural techniques caused important habitat alterations in agricultural ecosystems that lead to the decline of farmland wildlife populations throughout Europe. During 2008 and 2009, we investigated the effects of Habitat Improvement Actions (HIAs) and reforestations on populations of common pheasant *Phasianus colchicus* in order to evaluate the influence of both habitat management strategies on pheasant male density and distribution. We estimated the density of territorial males in four study areas located in the western Po Plain (northern Italy) through the method of crowing cock counts during the breeding season. We analysed the spatial distribution of territorial males in relation to the HIA and reforestation patches comparing points with crowing cocks with random ones. Moreover, we analysed the effects of habitat characteristics including HIAs and reforestations on male pheasant density by multiple regression analyses with theoretic information approach and multimodel inference. Our results showed that both HIAs and reforestations had an important role in determining male distribution. Moreover, HIAs and reforestations were good predictor variables of male pheasant density. We concluded that HIAs and reforestations provide common pheasant with suitable habitats and can represent proper solutions for enhancing habitat diversity and quality in intensive agro-ecosystems.

Keywords: agro-environmental measures, habitat improvement actions, habitat management, multi-model inference, Phasianus colchicus, pheasant, reforestation

Luca Nelli, Alberto Meriggi & Anna Vidus-Rosin, Department of Animal Biology, University of Pavia, via Ferrata 1 - 27100 Pavia, Italy - e-mail addresses: luca.nelli@unipv.it (Luca Nelli); meriggi@unipv.it (Alberto Meriggi); anna.vidus@unipv.it (Anna Vidus-Rosin)

Corresponding author: Luca Nelli

Received 16 March 2011, accepted 8 November 2011

Associate Editor: Hans Christian Pedersen

From the 1960s and onwards, agricultural ecosystems in Europe evolved into an overall reduction of habitat heterogeneity mainly due to the modernisation of agricultural techniques that led to important habitat changes. In particular, these alterations concerned: the increase of field size and monocultures, a reduction of grassy field margins, hedgerows and tree rows, a large employment of herbicides and pesticides and a general reduction of permanent cover (Stoate et al. 2001, Robinson & Sutherland 2002, Butler et al. 2010).

Furthermore, in less productive areas and in relation to common agricultural policy measures, the phenomenon of abandonment of agricultural lands has increased. This caused an increase in predation pressure because of habitat simplification and increased predator populations in those abandoned areas (Tapper et al. 1996). Moreover, the general reduction of resources (food and cover), especially in some critical periods of the year, probably intensified the effect of competition between sympatric species reducing density thresholds at which species can coexist. As a result, wildlife on farmland has declined throughout Europe (Tapper 1988, Tucker & Heat 1994, Potts 1997, Chamberlain et al. 2000, Green et al. 2005, Butler et al. 2007, 2010, Wretenberg et al. 2010).

In Italy, these effects have been amplified by exagerated hunting that caused the extinction and numerical fluctuations of populations of small game

species, resulting in a situation of constant threat (Merli & Meriggi 2000).

Agro-environmental measures of The European Commission (EC) gave rise to big expectations for a reversal of trend, but they could turn out negative if not readapted to small game requirements (Sotherton et al. 1992, 1994). In fact, diverging responses of different species make it difficult to optimise land management in a complex landscape towards many taxa simultaneously (Gottschalk et al. 2010).

Regarding small game species in Italy at the moment, most management actions are merely recurring restocking mainly with animals from game farms that are often expensive and lead to few benefits for long-term survival of populations (Dowell 1992, Meriggi & Mazzoni della Stella 2004, Meriggi et al. 2007). Hence, it is necessary to carry out specific projects on habitat improvements to increase Galliform and Lagomorph populations, with rising investments of provincial administrations, hunting districts, hunting estates, parks and natural reserves.

In intensively cultivated plain zones, habitat improvements for Galliforms are mainly aimed at increasing the availability of: 1) nesting cover, 2) shelter against predators and severe weather, 3) arthropod fauna as chick food and 4) food in winter. The possible actions already experienced with success are: 1) planting of high structural diversity hedgerows and tree rows, with spontaneous herbaceous belts, 2) small wood planting (1-5 ha), with autochthonous species and high structural diversity, 3) field margin conservation, 4) game crop sowing, 5) 'set-aside' management, 6) establishment of nesting, sheltering and feeding islands, 7) overwinter stubble maintenance and 8) regulation of agricultural processes on wildlife biological cycles (Sotherton et al. 1992, 1994, Moonen & Marshall 2001, Vickery et al. 2002, 2009, Wilson et al. 2005, Douglas et al. 2009).

The aim of our study was to verify if habitat improvement actions (HIAs) and reforestations could substitute the natural vegetation in intensively cultivated plain zones, which has been reduced to very small and isolated patches. For such verification, we chose the common pheasant *Phasianus colchicus* as an indicator of habitat quality for small game species because it has all the features that make it a good indicator (Grillas 1996). Moreover, it is demonstrated that habitat management aimed to increase pheasant populations can have positive effect on the local bird community, in particular passerine birds whose abundance and diversity is linked to habitat quality (Stoate 2002). We chose to monitor territorial pheasant male density and distribution because the presence and abundance of pheasant populations are linked to the presence of suitable vegetation for the settlement of territorial males (e.g. woodlots with dense undergrowth, tree rows and hedgerows with abundance of shrubs and shrubby areas). Hen density, on the other hand, appears to be more linked to suitable sites for nesting, because they choose to nest out of territories defended by males (Hill & Robertson 1988, Robertson et al. 1993, Robertson 1998, Leif 2005).

According to the kind of HIAs used, it is possible to obtain an increase of good areas for male territories or an increase of survival of young and adults from the end of summer to the next spring. In both cases, the consequence is an increase of pre-nesting densities. Consequently, our predictions were that the spatial distribution of pheasant males is different from a random one with an aggregation of territorial males around habitat improvements and reforestation patches, and that these variables positively affect male density.

Material and methods

Study areas

Our study was carried out between spring 2008 and summer 2009 in four protected study areas located in the western part of Po Plain (northern Italy; Fig. 1). These areas were oriented towards the production of wild pheasants and European hare *Lepus europaeus* with the aim of restocking surrounding hunting districts. Hunting was therefore forbidden and habitat improvement were usually carried out to increase density and productivity of wild populations of small game species. Basically, HIAs in our study areas constituted only small percentages of the land use during the study period ($\leq$ 2.4%; Table 1) and they were of two main types: 1) patches habitat improvement actions (PHIAs), consisting mainly of game crops *Sorghum* spp., uncropped wheat *Triticum* spp. fields and strips of maize *Zea mays*, and stubble maintenance, and 2) linear habitat improvement actions (LHIAs), consisting mainly of maintenance of grassy vegetation along field margins and along banks of irrigation canals. Reforestations funded by EC regulations 2079, 2080 and by Rural Development Plans of the Lombardy Region were

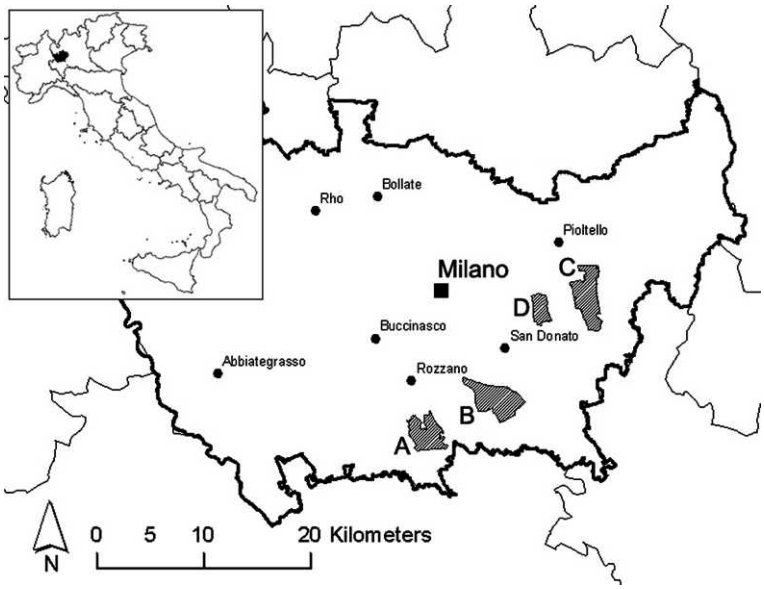

Figure 1. Location of our study areas (A-D).

only present in three study areas with percentages from 1.5 to 1.7% (see Table 1). Reforestations last 20 years and have a density of 330-800 plants/ha.

Two areas (A and B) had low level of natural vegetation, and they were dominated by rice *Oryza* spp. and maize monocultures. Areas C and D were characterised by large areas of natural vegetation, and the main crops were winter cereals (wheat and barley *Hordeum vulgare*), maize and hay fields (see Table 1).

The climate was continental-temperate with average annual rainfall between 700 and 1,000 mm concentrated in spring and autumn and a mean annual temperature of 12°C (minimum in January 2.2°C, maximum in July 25.2°C).

Table 1. Land use and size of our study areas during 2007-2009.

	Study areas			
Land use classes	A (803 ha)	B (1137 ha)	C (957 ha)	D (365 ha)
HIAs (%)	0.5	2.4	1.7	-
Reforestations (%)	1.7	1.5	-	1.5
Natural woods (%)	2.2	0.8	4.1	4.2
Poplar plantations (%)	5.0	0.8	-	-
Winter cereals (%)	2.9	8.4	39.9	34.7
Maize (%)	6.7	35.6	29.1	28.8
Rice (%)	62.8	38.6	-	-
Soybean (%)	-	2.4	13.0	3.8
Oilseed rape (%)	-	-	-	15.0
Hay fields (%)	0.2	0.9	2.0	3.7
Hedgerows and tree rows (%)	3.6	1.4	4.1	3.7
Basins (%)	2.4	0.4	0.1	-
Urban areas (%)	12.0	6.8	6.0	4.6

Pheasant counts

To estimate the density of male pheasants, we used the method of crowing cock counts from random points. The counts were conducted from 1 April to 1 June, in 2008 and 2009, between 7:00 and 11:00. We established a number of random points proportional to the size of each study area, with approximately 1 random point/70 ha. From each point, during 10 minutes, we recorded the number of crowing cocks, the exact time and UTM coordinates. We mapped each point and the position of each cock on aerial photographs at 1:5,000 scale, and then we measured the radial distance of each cock from the point. In total, we used 49 points, and we repeated the counts three times each year with an interval of 20 days between consecutive count sessions.

Land use mapping

During June of 2007-2009, we conducted direct surveys to map vegetative cover types in each study area. We took particular care of HIAs (both PHIAs and LHIAs) and reforestations. All cover types were digitalised using ArcView 3.3.

Data analysis

For each area and for both years, we estimated the density of territorial males through distance sam-

pling (Buckland et al. 2001). In order to investigate any association between the distribution of territorial males, HIAs and reforestations, we verified the null hypothesis that the observed distribution of males was not different from a random distribution. For this purpose, we created a number of random points equal to the number of observations. We measured the distance between each observation, the nearest HIA and reforestation lots. We used the same method for random points to obtain two distributions of distances. We compared the two distributions by Mann-Whitney and Kolmogorov-Smirnov tests. We used the former to verify differences between mean distance ranks of the two distributions and the latter to verify the difference between frequency distributions of distance classes of the two groups. We grouped distance classes in 200-m bands.

Furthermore, we performed logistic regression analysis (LogRA) between points with crowing cocks and an equal number of random points, using the distances from HIAs and from reforestations as predictor variables. In this way, we formulated a model that can be considered a resource selection probability function (RSPF) following a use vs availability design (Boyce et al. 2002, Keating & Cherry 2004). By the RSPF model, we tested the hypothesis that territorial males use the areas near HIAs and reforestation more than their availability. We tested the model performance by the percentage of correct classifications of original cases, Nagerkelke's R^2, and receiver operating characteristic (ROC) curve analysis.

In order to evaluate the effect of habitat variables and the role of HIAs and reforestations on pheasant density, we carried out multiple regression analysis (MRA) of pheasant density vs habitat variables (24 macro-habitat variables and six landscape metrics; see Appendix I). Density was calculated in 300-m radius buffer created on each point. For each year, we considered HIAs and reforestations of the former year. Moreover, we included the dummy variable 'year'. As dependent variable, we used the mean density of pheasants in each buffer weighted on the three repetitions.

First, we used 'stepwise forward' method to identify which habitat variables were the best predictors of male density. Then we identified all the possible subsets of uncorrelated ($P > 0.05$) predictor variables by calculating the correlation matrix (Pearson product moment coefficient) among habitat variables. For each subset, we performed MRA between pheasant density and habitat variables by the 'enter' method. We obtained a number of models that were ranked by the information-theoretic approach (Akaike 1973). We computed the corrected value of Akaike's information criterion (AIC_c). The corrected value was chosen because the ratio sample/parameter was small (Burnham & Anderson 2002). We selected the model with the lowest AIC_c as the best model, ranking the following ones by their differences from the lowest AIC_c (Δ_i). According to Burnham & Anderson (2002), models with $\Delta_i > 10$ have essentially no support and were omitted from further considerations. For the following analysis, we considered only models with $\Delta_i < 7$. Furthermore, we measured the relative importance of models by their Akaike's weights (w_i; Anderson et al. 2000, 2001). We followed the AIC statistical approach because it is a useful tool to compare different models performed on subsets of uncorrelated predictors. The main advantage of this statistical method is that it takes into account the likelihood and the number of parameters that occurred in each model. Therefore, it allows to compare all the models, as many as the uncorrelated subsets, and to select the ones that better explain the effect of habitat predictors on pheasant presence. Moreover, the AIC tool allowed us to conduct an explanatory analysis taking into account all possible predictor combinations on the probability of pheasant presence.

For each model, we calculated the variance inflation factor (VIF) in order to detect collinearity among predictor variables (Zuur et al. 2010). To validate the final models, we tested for deviation from normality of the residual distribution by a Shapiro-Wilk normality test and for residual autocorrelation by a Durbin-Watson test (Durbin & Watson 1950, Pires & Rodrigues 2007). To evaluate the importance of each individual variable within the best models, we calculated changes of R^2 and F for each entered predictor variable.

For all the analyses, we cumulated our data over the two study years and our four study areas.

Results

We detected 540 male pheasants in our four study areas during the two years. In 2008, by distance sampling we estimated a density ($\pm$ SD) of 6.6 ± 1.47 males/km^2 in area A, 10.2 ± 2.11 in area B, $17.9 \pm$

Table 2. Average distance (± SE) of crowing pheasant males (N = 540) and random points (N = 540) from HIAs and reforestations. Mann-Whitney U test is shown for the pooled study years and areas.

Mean distances (m)	HIAs	Reforestation
Crowing males	504 (28.6)	416 (25.6)
Random points	1073 (47.7)	832 (28.2)
U	20182.01	24011.52
P	< 0.001	< 0.001

2.77 in area C and 20.2 ± 6.68 in area D. In 2009, we estimated 22.5 ± 3.60 males/km^2 in area A, 10.0 ± 3.01 males/km^2 in area B, 13.3 ± 1.83 in area C and 27.0 ± 3.37 in area D.

Crowing males were significantly closer to both HIAs and reforestations than random points (Table 2). In particular, we found 50% of crowing males and only 25% of random points within 400 m from the nearest HIA (Fig. 2). Regarding reforestations, we found 62% of observations and only 21% of random points within 400 m from the nearest reforestation lot (Fig. 3).

Logistic regression analysis between points with crowing males and random points showed a negative effect of the distance from both HIAs and reforestations on pheasant presence. The logistic model explained 32.1% variance of the response variable and correctly classified 76.1% of original cases, 79.0% of crowing male points and 73.2% of random

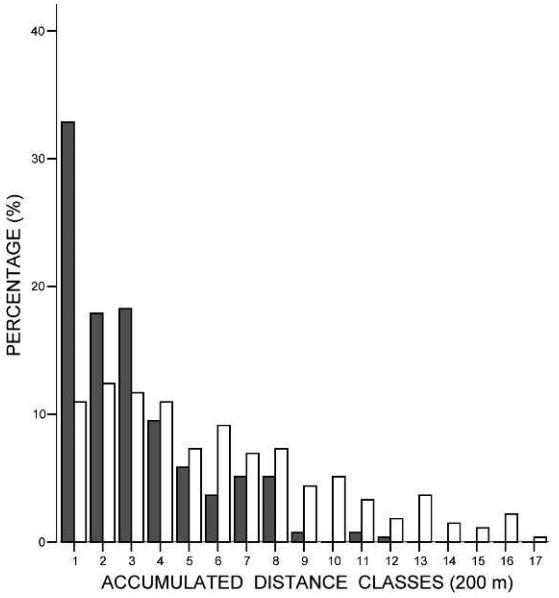

Figure 2. Distribution of crowing pheasant males (■) and random points (□) in relation to HIAs. Kolmogorov-Smirnov test for the pooled study years and areas: Z = 3.97, P < 0.001.

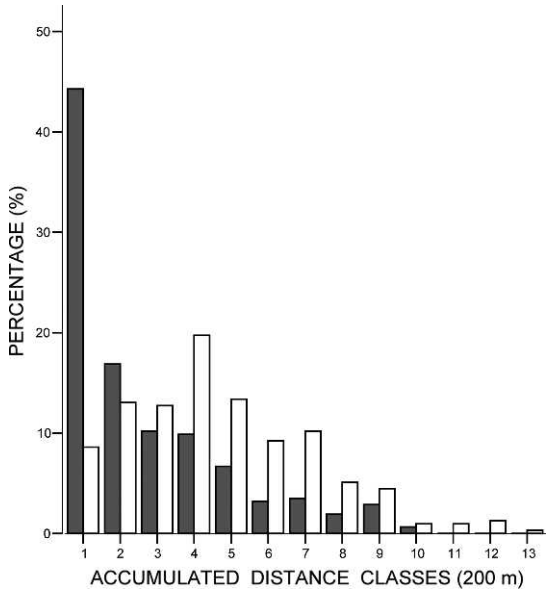

Figure 3. Distribution of crowing pheasant males (■) and random points (□) in relation to reforestations. Kolmogorov-Smirnov test for the pooled study years and areas: Z = 5.03, P < 0.001.

ones (Table 3). The area under the ROC curve was significantly greater than that of a model that randomly classifies the cases (Fig. 4).

The stepwise multiple regression model of male pheasant density vs habitat variables explained 52.6% of the total variance of the dependent variable. Five variables entered the model with positive effect on male pheasant density, and the most important once were reforestations, PHIAs and year (Table 4). Furthermore, reforestations and PHIAs were the first two habitat variables that contributed significantly to R^2 and F change (Reforestations: $R^2 = 0.246$, $F = 31.347$, $P < 0.001$, PHIAs: $R^2 = 0.162$, $F = 25.950$, $P < 0.001$).

The correlation matrix among habitat variables provided 137 subsets of uncorrelated predictors from which 93 significant regression models resulted.

Table 3. Results of logistic regression analysis between points with crowing pheasant males (N = 138) and random ones (N = 138). Data were pooled for study years and areas.

Variables	B	SE	P
Distance from HIAs	-0.001	0.0003	0.002
Distance from reforestations	-0.002	0.0004	< 0.0001
(Constant)	1.651	0.252	< 0.0001
-2 Loglikelihood	306.60		
Nagelkerke's R^2	0.321		
Model χ^2	76.02		< 0.0001

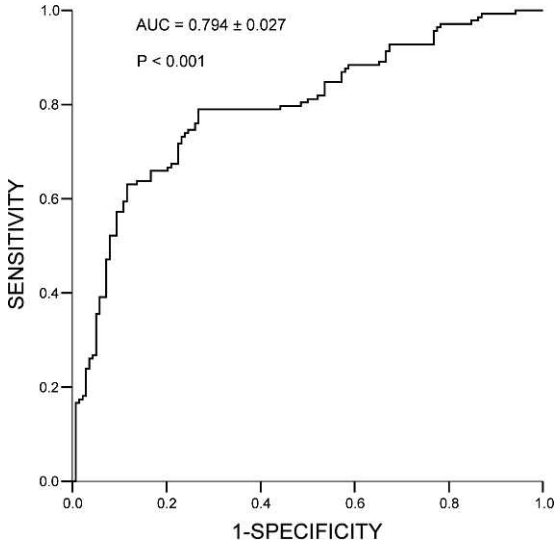

Figure 4. Receiver operating characteristic (ROC) curve of the logistic model between used and random points. AUC = area under curve.

Comparison among models showed that only four models predicted male pheasant density well ($\Delta AIC_c \leq 10$ and $w \geq 0.001$; Table 5). The model that explained the variance of male pheasant density best was the one that included both PHIAs and LHIAs, traditional poplar plantations and field mean shape index (MSI). The second ranked model ($\Delta AIC_c = 0.3$) substantially supported the former. It included PHIAs, LHIAs and field mean patch size (MPS). The other two models we considered showed a $\Delta AIC_c > 5$, so they had lower predictive power. The VIF values showed no collinearity among predictors in all the models (see Table 5).

The Durbin-Watson test showed inconclusive results for models 1, 2 and 3 and no autocorrelation of residuals for the fourth and stepwise model (Savin & White 1977). Only the latter model showed a normal distribution of residuals (Shapiro-Wilk test: $W = 0.990$, $P = 0.676$; Durbin-Watson statistic = 1.895).

Discussion

Our results showed that PHIAs and LHIAs had an important role in determining the spatial distribution of male pheasants during the breeding season. In our study areas, territorial cocks were very close to HIAs and reforestations. In the logistic regression model, the distances from HIAs and reforestations both showed a negative effect on male presence, which confirms the former analysis.

From multiple regression analyses on the subsets of uncorrelated variables, HIAs entered in the first two models with the most predictive power. Actually, the residual normality test showed a considerable limitation to the models that we selected through the AIC method. Thus, we should only consider the model obtained with the stepwise method as valid. In that model, the importance of HIAs and reforestations for male density clearly emerged with positive effects. The model even showed a positive influence of the variable 'year' on pheasant density. The importance of year can be ascribed to climatic conditions more or less suitable to pheasant populations or can be due to the protraction of these actions for following years (Meriggi et al. 2007). In our analysis, we considered the land use of the former year, pointing out that the effects of HIAs and

Table 4. Results of stepwise multiple regression analysis of male pheasant density vs habitat variables. Data were pooled for study years and areas. VIF = variance inflation factor.

	Unstandardised coefficients		Standardised coefficients			
Variables	B	SE	β	t	P	VIF
PHIAs	1.268	0.237	0.386	5.358	0.000	1.061
Reforestations	0.553	0.077	0.524	7.170	0.000	1.095
Year	3.924	0.937	0.296	4.186	0.000	1.024
Winter cereals	0.065	0.020	0.238	3.283	0.001	1.073
Tree rows	5.681	2.527	0.164	2.248	0.027	1.086
(Constant)	0.221	0.972	-	0.228	0.820	
AIC_c	331.400					
Adjusted R_2	0.526					
SE of estimate	4.580					
F	22.560					
P	< 0.0001					

Table 5. AIC_c of the multiple regression models of male pheasant density. Data were pooled for study years and areas. Only models with $w > 0.001$ and $\Delta AIC_c < 10$ are shown. AIC_c = corrected Akaike's information criterion, ΔAIC_c = difference in AIC between a given model and the best model, w = Akaike's weights, VIF = variance inflation factor, LHIAs = linear habitat improvement actions, MSI = field mean shape index, PHIAs = patches habitat improvement actions and MPS = field mean patch size.

Model	Variables (VIF)	w	R^2	Max log-likelihood	K	AIC_c	ΔAIC_c
1	LHIAs (1.022), MSI (1.023), poplars (1.011), PHIAs (1.026)	0.511	0.111	-91.699	6	196.3	0.0
2	LHIAs (1.031), MPS (1.023), PHIAs (1.024)	0.443	0.125	-92.979	5	196.6	0.3
3	Winter cereals (1.062), basins (1.038), hedgerows (1.039), poplars (1.011), PHIAs (1.036)	0.037	0.106	-93.157	7	201.6	5.3
4	Hedgerows (1.025), hay fields (1.028), oilseed rapes (1.022), natural woods (1.007)	0.009	0.125	-95.757	6	204.4	8.1

reforestations are more easily noticeable on mid- and long-term intervals, and that the prompt effects in terms of density and productivity increase are limited.

In our study, because of the scarce percentage of HIA, we could not discriminate between the different kinds of HIAs (e.g. uncropped areas and stubble maintenance). We only distinguished actions with patch and linear features (hedgerow and tree row maintenance) and reforestations. The two HIAs probably had different ecological values for pheasants, since HIAs are rarely enacted for more than one year, while reforestations have a 20-year duration.

Several studies on the demography and habitat selection by pheasants in the Po Plain (northern Italy) underlined that male territories are preferentially placed along natural or spontaneous vegetation strips (wood edges, fences, hedgerows and tree rows) and in areas characterised by high habitat diversity (Meriggi et al. 1996). Nevertheless, high densities can be reached only in agricultural habitats with minimum 15-20% of cultivated fields that ensure adequate resources all year round (Biadi & Mayot 1990).

Surprisingly, in our case we did not find any relationships between pheasant density and the presence and size of natural woods or hedgerows that should be the most selected habitats in male territories (Lachlan & Bray 1976, Hill & Robertson 1988, Robertson et al. 1993). This is probably due to the low presence of woods and hedgerows that are concentrated in a few lots within our study areas, so pheasants had to find substitutive habitats for territory placement (Hill & Robertson 1988). These habitats were represented by habitat improvements and reforestations that, on the contrary, were more scattered.

Reforestations, when sufficiently mature, show features that make them similar to natural woods. Pheasants utilise the marginal belts that supply them with food and refuges from predators and severe weather conditions more, due to the development of herbaceous and bushy cover. In our study areas, reforestations are ≥ 5 years old, and they consisted of rapid growth species such as wild cherry *Prunus avium*, ashes *Acer campestre*, white poplars *Populus alba* and alders *Alnus glutinosa* with interspersed bushes of whitethorns *Crataegus monogyna*, blackthorns *Prunus spinosa,* cornels *Cornus mas* and dog roses *Rosa canina*. Moreover, they allowed males to reach the most exposed places for territorial display. Sage & Tucker (1998) showed that nesting bird density, for pheasants in particular, was significantly lower in bigger patches of short-rotation forestry and pointed out the positive effect of ecotone on nesting birds. Previously, it was shown that patches with smaller size and characterised by long and irregular edges are more suitable habitats for pheasants, compared to bigger and more regular ones (Baxter et al. 1996). In our study, we found a negative effect of both patch size and shape index, suggesting that pheasants in our study areas selected small patches with regular edges. This was probably because reforestation patches in our study areas in general are small and consequently have a high shape index value.

Territorial male density represents a good index of habitat quality for pheasants. It is known that habitat suitability for pheasants is associated with natural bush and shrub vegetation available in the agricultural landscape (Lachlan & Bray 1976, Hill & Robertson 1988, Robertson et al. 1993, Robertson 1998, Scott et al. 1999, Leif 2005). In addition, reproductive success and survival of young are greatly associated with habitat quality. Monocultures with the consequent use of herbicides and insecticides reduce the availability of nesting sites and of food resources for the young by reducing the invertebrate abundance and diversity and increase

the effects of predation and mortality of chicks (Warner et al. 1984, 1999, Meriggi et al. 1996, Riley & Schultz 2001, Brittain et al. 2010).

If we consider territorial pheasant males as indicators of habitat quality and heterogeneity, we can attribute an important role to HIAs and reforestations in increasing habitat diversity of intensively cultivated agro-ecosystems. These actions could increase the specific composition of spontaneous vegetation and improve the availability of complex habitats in the simplified agro-ecosystems. Our results provide support to other studies that highlighted the contribution of habitat improvements in extending the natural habitat, acting as ecological corridors between isolated areas of natural vegetation with a positive impact on biodiversity in agricultural zones (Beamont 1993, Britt et al. 1995, Graham et al. 1996, Britt 2003, Sage et al. 2006).

Within reforestations, it is necessary to mechanically control weed development at the beginning of the plantation establishment, but once the plantation is mature, the growth of the herbaceous stratum is allowed. Herbaceous cover supports invertebrate populations, leading to an increase of small mammals and birds (Christian et al. 1997, Giordano & Meriggi 2009). In the same way, HIAs grant a good soil cover that is very important especially in the case of uncropped areas and stubble maintenance during fall and winter season, when the other habitats are usually bare (Douglas et al. 2009, Vickery et al. 2009).

In Italy, HIAs aimed at wildlife are still looked upon with certain scepticism by some farmers that consider wildlife and hunting activities as a potential source of damage, by some administrators that are in charge of the wildlife management but also by hunters that prefer to buy reared animals for restocking (Genghini et al. 2002). However, if correctly applied, habitat improvements can effectively enhance the density of small game populations with a positive spin-off on farmland wildlife and biodiversity.

Acknowledgements - all the HIAs were funded by the hunting district 'A.T.C. Milano1 est' and Province of Milan. We are grateful to all the administrative and technical staff for their support and assistance. We also thank all the farmers who accepted to collaborate with us in performing this research.

References

Akaike, H. 1973: Information theory and an extension of the maximum likelihood principle. - In: Petrove, B.N. & Csaki, F. (Eds.); Proceedings of the 2nd International Symposium on Information Theory. Akademiai Kiado, Budapest, Hungary, pp. 267-281.

Anderson, D.R., Burnham, K.P. & Thompson W.L. 2000: Null hypothesis testing: problems, prevalence, and an alternative. - Journal of Wildlife Management 64: 912-923.

Anderson, D.R., Link, W.A., Johnson, D.H. & Burnham, K.P. 2001: Suggestions for presenting the results of data analysis. - Journal of Wildlife Management 65: 373-378.

Baxter, D.A., Sage, R.B. & Hall, D.O. 1996: A methodology for assessing game bird use of short rotation coppice. - Biomass and Bioenergy 10: 301-306.

Beamont, N. 1993: Short Rotation Coppice: a Literature Review. - Environmental Resources Ltd. For the Department of Trade and Industry, ETSU B/W5/00216/REP, 32 pp.

Biadi, F. & Mayot, P. 1990: Les Faisans. Collection faune sauvage. - Haitier, Paris, France, 212 pp. (In French).

Boyce, M.S., Vernier, P.R., Nielsen, S.E. & Schmiegelow, F.K.A. 2002: Evaluating resource selection functions. - Ecological Modelling 157: 281-300.

Britt, C. 2003: Methodologies for ecological monitoring in bioenergy crops. A review and recommendations. - ADAS contract reports for the Department for Environment, Food and Rural Affairs, London, UK, 61 pp.

Britt, C., Buckland, M. & Health, M. 1995: Arable Energy Coppice. A review of published R&D and discussion of the potential for widespread production on surplus agricultural land in UK. - ADAS Report to the Ministry of Agriculture, Fisheries and Food, Oxford, UK, 170 pp.

Brittain, C.A., Vighi, M., Bommarco, R., Settele, J. & Potts, S.G. 2010: Impacts of a pesticide on pollinator species richness at different spatial scales. - Basic and Applied Ecology 11: 106-115.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. & Thomas, L. 2001: Introduction to Distance Sampling. Estimating abundance of biological populations. - Oxford University Press, New York, New York, USA, 432 pp.

Burnham, K.P. & Anderson, D.R. 2002: Model selection and multimodel inference: a practical information-theoretic approach. 2nd edition. - Springer-Verlag, New York, New York, USA, 488 pp.

Butler, S.J., Boccaccio, L., Gregory, R.D., Vorisek, P. & Norris, K. 2010: Quantifying the impact of land-use change to European farmland bird populatios. - Agriculture, Ecosystems and Environment 137: 348-357.

Butler, S.J., Vickery, J.A. & Norris, K. 2007: Farmland biodiversity and the footprint of agriculture. - Science 315: 381-384.

Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C. & Shrubb, M. 2000: Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. - Journal of Applied Ecology 37: 771-788.

Christian, D.P., Collins, P.T., Hanovski, J.M. & Niemi, G.J. 1997: Bird and small mammal use of short-rotation hybrid

poplar plantations. - Journal of Wildlife Management 61: 171-182.

Douglas, D.J.T., Vickery, J.A. & Benton, T.G. 2009: Improving the value of field margins as foraging habitat for farmland birds. - Journal of Applied Ecology 46: 353-362.

Dowell, S.D. 1992: Problems and pitfalls of gamebird reintroduction and restocking: an overview. - Gibier Faune Sauvage 9: 773-780.

Durbin, J. & Watson, G.S. 1950: Testing for Serial Correlation in Least Squares Regression: I. - Biometrika 37: 409-428.

Genghini, M., Spalatro, F. & Gellini, S. 2002: Farmers' attitudes toward the carrying out of wildlife habitat improvement actions (WHIA) in intensive agricultural areas of Northern Italy. - Zeitschrift für Jagdwissenschaft 48: 309-319.

Giordano, M. & Meriggi, A. 2009: Use by small mammals of short-rotation plantations in relation to their structure and isolation. - Hystrix, the Italian Journal of Mammology 20: 127-135.

Gottschalk, T.K., Dittrich, R., Diekötter, T., Sheridan, P., Wolters, V. & Ekschmitt, K. 2010: Modelling land-use sustainability using farmland birds as indicators. - Ecological Indicators 10: 15-23.

Graham, R.L., Downing, M. & Walsh, M.E. 1996: A framework to assess regional environmental impacts of dedicated energy crop production. - Environmental Management 20: 475-485.

Green, R.E., Cornell, S.J., Scharlemann, J.P.W. & Balmford, A. 2005: Farming and the fate of wild nature. - Science 307: 550-555.

Grillas, P. 1996: Identification of indicators. - In: Tomas Vives, P. (Ed.); Monitoring Mediterranean Wetlands: A Methodological Guide. Med Wet Publication, Wetlands International, Slimbridge, UK and Instituto da Conservacao da Naturez, Lisbona, Portugal, pp. 35-60.

Hill, D.A. & Robertson, P. 1988: The pheasant: ecology, management and conservation. - Blackwell Scientific Publication, Oxford, UK, 281 pp.

Keating, K.A. & Cherry, S. 2004: Use and interpretation of logistic regression in habitat-selection studies. - Journal of Wildlife Management 68: 774-789.

Lachlan, C. & Bray, R.P. 1976: Habitat selection by cock pheasant in spring. - Journal of Applied Ecology 13: 691-704.

Leif, A.P. 2005: Spatial Ecology and Habitat Selection of Breeding Male Pheasants. - Wildlife Society Bulletin 33: 130-141.

Meriggi, A., Brangi, A., Pella, F., Zaccaroni, M., Vidus Rosin, A., Gandini, S., Greco, C., Dessi Fulgheri, F. & Biagioli, O. 2007: Miglioramenti ambientali finalizzati al potenziamento delle popolazioni di Galliformi e Lepre (*Lepus europaeus,* Pallas). - Quaderni MIPAF, Modena, Italy, pp. 191-220. (In Italian).

Meriggi, A. & Mazzoni della Stella, R. 2004: Dynamics of a reintroduced population of red-legged partridges *Alectoris rufa* in central Italy. - Wildlife Biology 10(1): 1-19.

Meriggi, A., Pandini, W. & Cesaris, C. 1996: Demography of the pheasant in relation to habitat characteristics in northern Italy. - Journal of Wildlife Research 1: 15-23.

Merli, E. & Meriggi, A. 2000: L'utilizzo delle analisi di vitalità delle popolazioni nella gestione della fauna selvatica. - Ecology, Ethology and Evolution (Suppl. 1) 12: 1-10. (In Italian).

Moonen, A.C. & Marshall, E.J.P. 2001: The influence of sown margin strips, management and boundary structure on herbaceous field margin vegetation in two neighbouring farms in southern England. - Agriculture, Ecosystems and Environment 86: 187-202.

Pires, A.N. & Rodrigues, I.M. 2007: Multiple linear regression with some correlated errors: classical and robust methods. - Statistics in medicine 26: 2901-2918.

Potts, G.R. 1997: Cereal farming, pesticides and grey partridges. - In: Pain, D.J. & Pienkowski, M.W. (Eds.); Farming and birds in Europe: The Common Agricultural Policy and its Implications for Bird Conservation. Academic Press, San Diego, California, USA, pp. 150-177.

Riley, T.Z. & Schulz, J.H. 2001: Predation and ring-necked pheasant population dynamics. - Wildlife Society Bulletin 29: 33-38.

Robertson, P.A. 1998: Habitat selection and local abundance of breeding pheasants (*Phasianus colchicus*) in Utah. - Gibier Faune Sauvage, Game and Wildlife 15: 433-446.

Robertson, P.A., Woodburn, M.I.A., Neutel, W. & Bealey, C.E. 1993: Effects of land use on breeding pheasant density. - Journal of Applied Ecology 30: 465-477.

Robinson, R.A. & Sutherland, W.J. 2002: Post-war changes in arable farming and biodiversity in Great Britain. - Journal of Applied Ecology 39: 157-176.

Sage, R., Cunningham, M. & Boatman, N. 2006: Birds in willow short-rotation coppice compared to other arable crops in central England and a review of bird census data from energy crops in the UK. - Ibis 148: 184-197.

Sage, R. & Tucker, K. 1998: Integrated crop management of SRC plantations to maximise crop value, wildlife benefits and other added value opportunities. - Contract Report by the Game Conservancy Trust for the Department of Trade and Industry. Report: ETSU B/W2/00400/REP. ETSU, Harwell, Oxford, UK, 362 pp.

Savin, N.E. & White, K.J. 1977: The Durbin-Watson Test for Serial Correlation with Extreme Sample Sizes or Many Regressors. - Econometrica 45: 1989-1996.

Scott, A.S., Stewart, N.J. & Gates, J.T. 1999: Home Ranges, Habitat Selection and Mortality of Ring-Necked Pheasants (*Phasianus colchicus*) in North-Central Maryland. - American Midland Naturalist 141: 185-197.

Sotherton, N.W., Boatman, N.D., Manosa, S. & Robertson, P.A. 1994: Management of set-aside for game and wildlife. - Aspects of Applied Biology 40: 497-503.

Sotherton, N.W., Boatman, N.D. & Robertson, P.A. 1992: The use of set-aside land for game conservation. - In:

Clarke, J. (Ed.); BCPC Monograph no. 50, BCPC Publications, Bracknell, UK, pp. 223-229.

Stoate, C. 2002: Multifunctional use of a natural resource on farmland: wild pheasant (*Phasianus colchicus*) management and the conservation of farmland passerines. - Biodiversity and Conservation 11: 561-573.

Stoate, C., Boatman, N.D., Borralho, R.J., Carvalho, C.R., de Snoo, G.R. & Eden, P. 2001: Ecological impacts of arable intensification in Europe. - Journal of Environmental Management 63: 337-365.

Tapper, S.C. 1988: Population changes in gamebirds. - In: Hudson, P.J. & Rands, M.R.W. (Eds.); Ecology and Management of Gamebirds. BSP Professional Books, Oxford, UK, pp. 18-47.

Tapper, S.C., Potts, G.R. & Brockless, M.H. 1996: The effect of an experimental reduction in predation pressure on the breeding success and population density of grey partridges *Perdix perdix*. - Journal of Applied Ecology 33: 965-978.

Tucker, G.M. & Heath, M.F. 1994: Birds in Europe: their conservation status. - Birdlife Conservation Series no. 3. Birdlife International, Cambridge, UK, 600 pp.

Vickery, J.A., Carter, N. & Fuller, R.J. 2002: The potential value of managed cereal field margins as foraging habitats for farmland birds in the UK. - Agriculture, Ecosystems & Environment 89: 41-52.

Vickery, J.A., Feber, R.E. & Fuller, R.J. 2009: Arable field margins for biodiversity conservation: A review of food resource provison for farmland birds. - Agriculture, Ecosystems and Environment 133: 1-13.

Warner, R.E., Etter, L.S., Joselyn, G.B. & Ellis, J.A. 1984: Declining survival of ring-necked pheasant chicks in Illinois agricultural ecosystems. - Journal of Wildlife Management 48: 82-88.

Warner, R.E., Mankin, P.C., David, L.M. & Etter, L.S. 1999: Declining survival of ring-necked pheasant chicks in Illinois during the late 1900s. - Journal of Wildlife Management 63: 705-710.

Wilson, J.D., Whittingham, M.J. & Bradbury, R.B. 2005: The management of crop structure: a general approach to reversing the impacts of agricultural intensification on birds? - Ibis 147: 453-463.

Wretenberg, J., Pärt, T. & Berg, A. 2010: Changes in local species richness of farmland birds in relation to land-use changes and landscape structure. - Biological Conservation 43: 375-381.

Zuur, A.F., Ieno, E.N. & Elphick, C.S. 2010: A protocol for data exploration to avoid common statistical problems. - Methods in Ecology & Evolution 1: 3-14.

Appendix I. Variables used in the multiple regression analyses of pheasant density vs habitat variables with units of measurement (UOM). LHIA = linear habitat improvement action and PHIA = patches habitat improvement action.

Variables	UOM
Asphalt roads	%
Basins	%
Cartway	%
Corn	%
Dirt roads	%
Farms	%
Grassy field margins	%
Hay fields	%
Hedgerows	%
Irrigation canals	%
LHIA	%
Natural woods	%
Oilseed rape	%
PHIA	%
Poplars	%
Reforestations	%
Rice	%
Soybean	%
Tree rows	%
Uncultivated land	%
Urban	%
Winter cereals	%
Edge density (ED)	m/ha
Mean patch edge (MPE)	m
Mean patch size (MPS)	ha
Mean perimeter-area ratio (MPAR)	m/ha
Mean shape index (MSI)	-
Shannon diversity index (H')	-
Year	Dummy variable (0;1)