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Article in *Journal for Nature Conservation* · August 2018

DOI: 10.1016/j.jnc.2018.08.009

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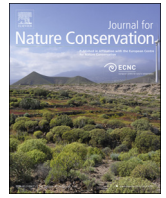
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Lesser kestrel diet and agricultural intensification in the Mediterranean: An unexpected win-win solution?

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ARTICLE INFO

Keywords:

Agricultural intensification
Farmland birds
Land use
Lesser kestrel diet

ABSTRACT

Farmland bird species have suffered dramatic declines in recent decades, especially in Mediterranean areas. The intensification of agricultural practices has led to reduced invertebrate prey, which represent the bulk of the diet of many farmland birds. In this study, we investigated the spatial and temporal variation in the diet of the lesser kestrel (*Falco naumanni*) during the breeding season, monitored over a five-year period between 2006 and 2013 in the Gela Plain (Sicily). Our aim was to understand whether, and to what extent, farming practices affected the reproductive outputs of this predominantly insectivore bird in order to find a profitable compromise between conservation of farmland birds and farming practices. During our study, lesser kestrel diet varied among farmland habitats, in terms of ingested biomass, prey items/pellet and diet breadth. This has reflected in the reproductive output because colonies characterised by a higher ingested biomass fledged more chicks than the others. The artichoke and grassland fields were found to provide the most beneficial dietary parameters. Unexpectedly, the intensive and high-profit artichoke farming might turn out to be suitable for lesser kestrels. Artichoke fields are chiefly used when abandoned after harvest, providing high prey availability and accessibility for kestrels during the brood raising stage of their breeding season. A mosaic of grassland and artichoke fields can thus be recommended for Mediterranean agricultural areas of the Natura2000 network, in which some intensive farming and lesser kestrels can coexist, if adequately framed in a friendly-to-wildlife agriculture policy. Such a potential optimal trade-off between avian population persistence and economic sustainability for farmers we have found, should be planned in alternative management of agro-ecosystems, enhancing the functioning of trophic chains. For our study area, we suggest at least farmers be: i) informed on the role of predators as biological agents for pest control; ii) granted to reduce the high level of chemicals currently used during cultivation in favour of organic farming; and iii) granted to maintain the artichoke fields until June, following completion of the harvest.

1. Introduction

Populations of several farmland bird species, representing a large proportion of European avifauna, have suffered dramatic declines in recent decades, especially in western and Mediterranean Europe (Butler, Boccaccio, Gregory, Vorisek, & Norris, 2010; Chamberlain, Fuller, Bunce, Duckworth, & Shrubbs, 2000; Donald, Green, & Heath, 2001; Sokos, Mamolos, Kalburtji, & Birtsas, 2013). The major cause of population decline is from the intensification of agricultural practices, such as heavy mechanisation, increased use of fertilisers and biocides, coupled with the abandonment of less productive and marginal lands (Benton, Vickery, & Wilson, 2003; Donald, Pisano, Rayment, & Pain, 2002; Fuller et al., 1995; Newton, 2004). Agricultural intensification has reduced the availability of refuges and reproduction areas of

invertebrate prey, which represent the bulk of the diet of most farmland birds (Benton, Bryant, Cole, & Crick, 2002; Boatman et al., 2004; Genghini, Gellini, & Gustin, 2006; Wilson, Morris, Arroyo, Clark, & Bradbury, 1999). Recent studies have shown that the use of pesticides, such as neonicotinoids, can cause both a disruption in the food web and a population decline in insectivorous birds (Easton & Goulson, 2013; Gibbons, Morrisey, & Mineau, 2015; Gill, Ramos-Rodríguez, & Raine, 2012; Hallmann, Foppen, van Turnhout, de Kroon, & Jongejans, 2014; Whitehorn, O'Connor, Wackers, & Goulson, 2012).

The quality of habitats surrounding a nest site may affect the distribution, reproduction, body mass and diet of many avian species in the same season (Bretagnolle et al., 2011; Geiger et al., 2014; Orłowski, Czarnecka, & Golawski, 2014), or even in subsequent years through carry-over effects (Harrison, Blount, Inger, Norris, & Bearhop, 2011;

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Norris, Marra, Kyser, Sherry, & Ratcliffe, 2004; Rioux Paquette, Pelletier, Garant, & Bélisle, 2014).

The Common Agricultural Policy (CAP) of the EU has accelerated the habitat degradation process and biodiversity loss, driven by the combination of intensification and abandonment (Berg, Wretenberg, Zmihorski, Hiron, & Pärt, 2015; Pe'er et al., 2014; Robinson & Sutherland, 2002). To halt biodiversity loss, the EU established the development of alternative environmentally friendly agricultural areas with reduced pesticide use (European Commission, 2006; Whittingham, 2007).

A comprehensive understanding of the association between land use and the diet of animal species living in a fast-changing and anthropogenic environment has been highly recommended, as this will provide important data to the EU for the support of bird conservation and management actions not only at the breeding grounds but also at the pre-migratory areas (Bijlsma, 2013; Kowalczyk, Chiaradia, Preston, & Reina, 2014; Sarà, Campobello, Zanca, & Massa, 2014). Thus, knowledge of the nutritional profile of food sources and of the temporal and spatial distribution in the diet of farmland species, have become crucial information to be recorded (Lihoreau et al., 2015; Raubenheimer, Simpson, & Mayntz, 2009), in order to understand whether, and to what extent, the intensification of farming practices has affected both the reproductive output and survival of insectivorous birds, and whether there are possible actions which are beneficial for both farmer profits and conservation of farmland birds.

Birds of prey represent a good indicator of ecosystem functioning due to their position at the top of food webs (Sergio et al., 2014), and species living in farmland landscapes promptly respond to agricultural intensification (see for instance Butet et al., 2010).

The lesser kestrel, a small raptor species, experienced a marked decline in its western Palaearctic breeding range in the middle of the 20th century (Iñigo & Barov, 2011). The recovery of major Iberian populations through conservation actions has improved the conservation status of the species from 'vulnerable' to 'least concern' (IUCN, 2011). However, beyond the Iberian Peninsula their conservation status is fluctuating and requires further investigation, especially in southern and eastern Europe and particularly in Italy (Campobello, Lindstrom, Di Maggio, & Sarà, 2017; Morganti, Preatoni, & Sarà, 2017). In addition, the lesser kestrel is considered an ideal biological indicator for studying the effects of agricultural intensification on insectivorous predators (Bonat & Aparicio, 2008; Catry, Franco, & Moreira, 2014; Tella & Forero, 2000), as the effects on this species can be extended to several other farmland birds (Brickle, Harper, Aebischer & Cockayne, 2000; Butet et al., 2010; Delgado & Moreira, 2002). Previous studies have investigated the dietary composition of lesser kestrels under different conditions of anthropogenic disturbance and climate, however, the association with modern agricultural landscapes was not explicitly considered to be a potential determinant of population dynamics in the Mediterranean area (Choisy, Conteau, Lepley, Manceau, & Yau, 1999; Pérez-Granados, 2010 but see Lepley, Brun, Foucart, & Pilard, 2000 in France). Moreover, the feeding ecology of the lesser kestrel is virtually unknown outside the Iberian Peninsula.

Lowland areas of southern Europe include important Mediterranean habitats used by lesser kestrels during the breeding season (Di Maggio, Campobello, & Sarà, 2013). Our previous work (Di Maggio, Campobello, Tavecchia, & Sarà, 2016) demonstrated that lesser kestrel colonies located in non-irrigated cereal lands had a lower adult and juvenile survival probability and population growth rate compared to colonies located in grasslands and artichoke lands. Therefore, we hypothesised that the specific composition of foraging habitats during the breeding season may affect the intra-population variability in the diet of lesser kestrels. Such variability should be crucial for the seasonal reproductive success and ultimately will drive the demography of this species.

More specifically, we identified the need for additional investigation into the relationship between the dietary composition of the lesser

kestrel and level of agricultural intensification, in order to determine whether it is possible to find appropriate and realistic agricultural management actions which are able to fulfil both the habitat requirements for bird viability and farmer demands. Thus, the main aims of this study were to: (i) determine the diet of lesser kestrels during five breeding seasons (sampled across an eight-year period) and compare its composition in three farmland habitats, two of which characterised by a higher intensity of agricultural activity; (ii) understand the effect of diet on the breeding success of lesser kestrels; and (iii) provide new evidence, if any, for sustainable agro-ecosystems for both farmers and biodiversity.

2. Methods

2.1. Study species

The lesser kestrel is a small raptor that lives in pseudo-steppes of the western Palearctic (Iñigo & Barov, 2011) and spends the winter in West Africa (Limiñana, Romero, Mellone, & Urios, 2012). It is a facultative colonial species that usually breeds in association with jackdaws (*Corvus monedula*) and rock doves (*Columba livia*) (Campobello, Hare, & Sarà, 2015; Campobello, Sarà, & Hare, 2012). Several investigations showed lesser kestrel as a mainly insectivorous species, feeding mainly on beetles and grasshoppers with a marked change during the breeding season in terms of prey composition and prey richness (e.g. Rodríguez, Tapia, Kieny, & Bustamante, 2010). Recently, Catry, Catry, Alho, Franco, and Moreira, (2016) found a sex-specific differences in the diet of lesser kestrels limited to the courtship period. The lesser kestrel population in the Gela plain is the largest in Sicily and one of the most important in Italy with colony sizes ranging from 1 to 45 pairs (Di Maggio et al., 2014, 2016; Mascara & Sarà, 2006; Sarà, Campobello, & Zanca, 2012).

2.2. Study area and habitat types

The Gela plain (474 km²) in southern Sicily (Italy, 37° 07' N, 14° 19' E), is a Mediterranean pseudo-steppe formed by a mosaic of arable habitats, dominated by artichoke (*Cynara* spp.), wheat and leguminous cultivations (Triolo, Campobello, & Sarà, 2011). Agricultural intensification is increasingly changing the arable land use, from dry cereal and artichoke fields to irrigated crops (including new varieties of artichokes and vegetable growing) in the flat core area of the plain. In fact, several of our study colonies turned from being surrounded by non-irrigated crops to a more intensive land use (with pipelines and drip irrigation growing artichokes and vegetables, these latter replacing artichoke soon after the harvest) within our observation period (M. Sarà, personal observation). The rest of the area contains pastures and xeric vegetation, predominantly graminaceous plants and Mediterranean shrubs (Sarà et al., 2012). We identified the habitats, being the predominant crop types within a 3 km radius of each colony, because during the nesting period most foraging trips take place in this radius around colonies (Franco, Catry, Sutherland, & Palmeirim, 2004; Tella, Forero, Hiraldo, & Donázar, 1998; but see Hernández-Pliego, Rodríguez, & Bustamante, 2017). The annual land use of plots surrounding the colonies were obtained from digital maps at the scale of 1:10,000 of AGEA (www.sitr.regione.sicilia.it), historical photographs from Google maps and field validation records. Principal component analysis (PCA) was used to summarise the essential habitat characteristics of the agro-ecosystems (see Fig. A in supplementary materials). The optimum number of principal components were selected using a scree plot, which revealed two significant eigenvalues accounting for 96.89% of total variance, and three land-use variables which had the highest contribution to colony classification (see details in Di Maggio et al., 2016 and Fig. A in supplementary materials). The land uses were grouped into the following agro-ecosystems: dry grasslands and other semi-natural vegetation (hereafter, grassland, GRA) characterised by

low-intensity agricultural activities (like workers and machinery presence, etc.: see Sarà et al., 2012 for categorisation of impact); non-irrigated cereal crops on arable land with medium-intensity agricultural activities (hereafter, arable, ARA) and semi-permanently irrigated arable land with a high level of agricultural activities (hereafter, artichoke, ART; see Fig. A supplementary materials). Each colony, and the pellets collected from the colonies, were classified into one of these main categories for the purpose of further analysis (Soliveres et al., 2011). For each year of the study period (over a five-year period between 2006 and 2013), we re-classified each colony according to its prevalent land-use inside the 3-km buffer.

2.3. Pellet collection, prey identification and dietary variables

We collected a total of 927 pellets from 23 colonies (39 ± 0.60 pellets per colony, mean \pm SE) over five breeding seasons (2006, 2007, 2010, 2012 and 2013) sampled over an 8-year period in the Gela plain. Unfortunately, we could not sample all 23 colonies every year; as a consequence, we inspected a mean of 3.66 ± 0.65 colonies per habitat/year, collecting an annual mean of 11.9 ± 1.17 nests among the total sample of colonies. As main land uses changed across the time of study around some colonies, they were reclassified accordingly.

Each colony was inspected at least three times during the breeding season, at the time of site occupation and egg-laying (mid-April to early May), incubation and hatching (mid-May to early June) and fledging (mid-June to mid-July). This schedule allowed the collection of sufficient data while simultaneously minimising disturbance at the reproductive sites (Di Maggio et al., 2013). At each visit, the old prey remains and pellets were removed from the nests. We then collected fresh pellets in the proximity of, or inside, the nest sites during successive visits (Catry et al., 2016; Rodríguez et al., 2010). We did not collect the pellets from specific nests but we chose randomly from the active nests available during each visit. After collection in the field, each pellet was stored in a plastic bag and kept at room temperature (20 °C) until laboratory analysis.

For each pellet, we assigned large identifiable remains to different taxa using taxonomic guides, whereas small insect fragments were identified at the species level by comparison with samples from entomological collections (Rizzo & Massa, 1995), and with collections of dissected samples of species commonly found in the study area (Massa, 2011). Pellets were dry-dissected in the laboratory under magnification using a stereomicroscope (M5, Wild, Heerbrugg - Switzerland). Every mandible was paired with its partner and each head and leg fragment was isolated, making it was possible to count the total number of specimens in every pellet. Most prey were identified at the species level, but in the case of heavily fragmented remains, we applied a conservative criterion and assigned the fragments at the family or genus level. As we were not interested in specific prey-predator relationships, but rather in inter-habitat prey availability, we then grouped all prey into 11 large taxonomic categories (hereafter, prey-type) for statistical analyses. They represented three vertebrates (rodents, reptiles and birds) and eight insect taxa: Coleoptera Scarabaeidae (beetles), Coleoptera Carabidae (ground beetles), other Coleoptera, Orthoptera Tettigoniidae (bush crickets), Orthoptera Acrididae (grasshoppers), Orthoptera Grillidae (crickets), other Orthoptera, and other Insecta (i.e. Hymenoptera, Lepidoptera and Mantodea).

We used three dependent variables: i) ingested biomass per pellet, ii) number of individual prey items consumed per pellet (hereafter prey items/pellet), and iii) diet breadth index, in order to describe the lesser kestrel food niche and investigate whether there was an effect of the reproductive stage and/or habitat. Ingested biomass was calculated by adding the weights of each prey item in a single pellet (Sumasgutner, Nemeth, Tebb, Krenn, & Gamauf, 2014). Prey biomasses were calculated from body weight data recorded in the study area (Massa, 2011) and from literature (Rodríguez et al., 2010; Sumasgutner et al., 2014). The prey items/pellet were calculated from the number of same-side

anatomical remains found for each taxonomic category within a given pellet, thus, each pellet was treated as an independent unit (Bonafant & Aparicio, 2008; Sumasgutner et al., 2014). The diet breadth index is an estimate of diet specialisation and is calculated according to Levin's Index (Levin, 1968) as $B = 1/\sum p_i^2$, where p_i is the proportion in the diet represented by the prey-type i for each pellet. We used Levin's Index in this study because we were interested in expressing the specialisation of diet with respect to the most abundant prey items (Magurran, 2003).

2.4. Breeding parameters

Colony descriptions and study protocols for lesser kestrels in the study area have previously been described in Sarà et al. (2012), Di Maggio et al. (2014) and Di Maggio et al. (2016). Shortly, we defined a breeding colony as a man-made building, often abandoned and ruined rural houses, where at least one pair of kestrels performed some reproductive behaviour, such as a male delivering prey to a female, copulation or the inspection of nest chambers, egg laying and brood raising. For each colony sampled for pellets the occupied nests were inspected to determine the: i) mean number of eggs, and ii) mean number of fledglings.

2.5. Statistical analysis

We first used generalised linear mixed models (GLMMs; McCullagh & Searle, 2000) with normal (for ingested biomass and diet breadth) and Poisson distributions (for prey items/pellet analysis) to study the effect of habitat (3 levels: ARA, ART and GRA), reproductive stages (two levels: egg incubation and chick rearing) and their first-order interactions (habitat*reproductive stages) on: i) prey items/pellet, ii) diet breadth and on iii) ingested biomass.

The colony identity and the study years (2006, 2007, 2010, 2012 and 2013) were included as random factors. The large sample size and use of a single analysis (Underwood, 1997), however reduced the risk of type-2 errors otherwise caused by the multiple testing of first-order interactions (Zuur, Hilbe, & Ieno, 2013). Tukey's HSD (Honest Significant Difference) post-hoc tests were performed after the GLMM analyses in order to examine differences in diet breadth, ingested biomass and prey items/pellet for the different reproductive stages and habitats. The Akaike information criteria (AIC; Burnham & Anderson, 2002) was used to rank the models according to the information-theoretic approach (Symonds & Moussalli, 2011). The model with the lowest AIC value was considered to be the best compromise between model deviance and model complexity. Models that differed by less than 2 AIC points from the best one, were considered to provide equal support to the data (Burnham & Anderson, 2002; Grueber, Laws, Nakagawa, & Jamieson, 2011), however the AIC weight (AICw) informed about the relative importance of predictors (Symonds & Moussalli, 2011).

Second, we used a generalised linear model (GLM) to determine the effect of diet breadth, ingested biomass and prey items/pellet on: i) the mean number of eggs per colony; and ii) the mean number of fledglings per colony. In this case, we employed a normal error distribution because the mean number of eggs per colony (Kolmogorov-Smirnov normality test, $P = 0.105$) and mean number of fledglings per colony (Kolmogorov-Smirnov normality test, $P = 0.124$) were normally distributed. Since diet is likely to vary with land use and reproductive stage, any colony sampled within the year (a different reproductive stage in the same colony) and across years (same colony with different land use) was considered independent. The collinearity among model predictors (diet breadth, ingested biomass and prey items/pellet) calculated with the Variance inflation factors (VIF) was ~ 1 , meaning a lack of correlation among the predictor variables (Zuur et al., 2013).

All the values were reported as mean \pm SE. The analyses were conducted using R version 3.0.2 (R Development Core Team, 2011)

Table 1

Number and frequency of prey items in the ingested biomass (g) found in the 927 lesser kestrel pellets analysed over the study period in the Gela plain (2006–2013). Totals and subtotals of the main prey categories have been reported in bold.

Taxon	N	N %	Ingested biomass (g)	Ingested Biomass %
Rodentia	176	3.90	3,013	22.64
Reptilia	157	3.48	4,781	36.0
Aves	81	1.79	1,885	14.16
Sub-total Vertebrate	414	9.03	9,679	72.80
Scarabaeidae	961	21.28	690	5.18
Carabidae	296	6.55	149	1.12
Other Coleoptera	489	10.83	224	1.68
Sub-total Coleoptera	1,746	38.65	1,063	7.98
Tettigoniidae	663	14.68	1192	8.95
Acrididae	581	12.86	325	2.44
Gryllidae	309	6.84	82	0.62
Other Orthoptera	330	7.31	496	3.73
Sub-total Orthoptera	1,883	41.69	2,095	15.74
Other Insecta	474	10.49	474	3.60
Total	4517		13,311	

with the R package lme4 (Bates, Maechler, & Bolker, 2013) and STATISTICA 10.0 (Statsoft Inc., 2001).

3. Results

We analysed 4517 prey items in 927 pellets collected over five years (185 ± 52.4 pellets per year). The core of the lesser kestrel diet (80%) was equally divided between Coleoptera (38.6%) and Orthoptera (41.7%) in terms of prey items percentage (Table 1). Vertebrates accounted for only 9% of prey items, but represented 72.8% of ingested biomass, being represented especially by reptiles and rodents (36% and 22.6%, respectively; Table 1). Orthoptera ranked second to vertebrates in terms of ingested biomass (15.7%), followed by Coleoptera (8%). Among the Coleoptera, *Pentodon bidens* (Scarabaeidae) was the most frequently consumed prey (53%), whereas *Grylloderes brunneri* (Grillidae) (16%) and *Aiolopus strepens* (Acrididae) (16%) were the main prey among the Orthoptera. Large Acrididae were most consumed in artichoke fields whereas medium/small size Tettigoniidae and Gryllidae were most consumed in grasslands and arable lands, respectively (Table B).

3.1. Effect of habitat and reproductive stages on diet

During the study years, diet breadth and the prey items/pellet were best explained by the interaction between the habitat type and the reproductive stage (Table 2). The larger diet breadth was found in cereal crops of arable land (ARA, hereafter arable) during the eggs incubation (2.56 ± 0.08 ; $F_{1,2} = 3.25$, $P < 0.001$, $N = 927$; Fig. 1A), followed by that in grassland during the eggs incubation, although not statistically significant (2.34 ± 0.10 , $F_{1,2} = 3.25$, $P = 0.08$, $N = 927$; Fig. 1A). Artichoke fields, both during eggs incubation and chicks rearing showed the lowest diet breadth (1.88 ± 0.08 , $F_{1,2} = 3.25$, $P < 0.001$, $N = 927$; Fig. 1A).

Likewise, the highest prey items/pellet were found in grassland during the chicks rearing (6.16 ± 0.37 ; $F_{1,2} = 3.60$, $P < 0.001$, $N = 927$), followed by values found in arable during both eggs incubation and chicks rearing (5.20 ± 0.40 , $F_{1,2} = 3.25$, $P = 0.02$, $N = 927$; Fig. 1B). Artichoke fields, during eggs incubation (but not during chicks rearing) showed the lowest number of prey items/pellet (2.91 ± 0.37 , $F_{1,2} = 3.59$, $P < 0.001$, $N = 927$; Fig. 1B).

Two models were ranked as giving the best support to the analysis of ingested biomass across the reproductive season (ΔAIC values < 2.0 ;

Table 2

GLMMs testing whether habitat and reproductive stages (RS) are important predictors of: i) ingested biomass, ii) prey items/pellet and iii) diet breadth ($N = 927$). Colony identity and study year are included as random factors. The best models are presented in bold and are compared to the baseline levels of Arable as Habitat type, and Egg incubation as Reproductive stage.

Rank	Model	Estimate	SE	AIC	ΔAIC	AICw
<i>Ingested biomass</i>						
1	Habitat			7852.86	0.00	0.59
	Habitat Artichoke	4.37	1.65			
	Habitat Grassland	1.81	1.86			
2	Habitat + Reproductive Stage			7854.62	1.76	0.25
	Habitat Artichoke	4.44	1.66			
	Habitat Grassland	2.01	1.91			
	RS: Chicks rearing	-0.67	1.36			
3	Habitat * Reproductive Stage			7856.11	3.25	0.11
	Habitat Artichoke	4.67	2.07			
	Habitat Grassland	4.33	2.49			
	RS: Chicks rearing	0.67	2.13			
	Habitat Artichoke * Chicks rearing	-0.23	2.79			
	Habitat Grassland * Chicks rearing	-4.23	2.99			
4	Reproductive Stage			7857.7	4.85	0.05
	RS: Chicks rearing	-0.39	1.32			
<i>Prey items/pellet</i>						
1	Habitat * Reproductive Stage			5166.50	0.00	0.78
	Habitat Artichoke	0.58	0.52			
	Habitat Grassland	2.16	0.68			
	RS: Chicks rearing	1.12	0.52			
	Habitat Artichoke * Chicks rearing	0.53	0.66			
	Habitat Grassland * Chicks rearing	-1.40	0.73			
2	Habitat + Reproductive Stage			5170.5	4.00	0.11
	Habitat Artichoke	0.75	0.43			
	Habitat Grassland	1.40	0.56			
	RS: Chicks rearing	0.90	0.32			
3	Reproductive Stage			5170.53	4.01	0.10
	RS: Chicks rearing	1.03	0.32			
4	Habitat			5176.23	9.71	0.00
	Habitat Artichoke	0.84	0.43			
	Habitat Grassland	1.66	0.55			
<i>Diet Breadth</i>						
1	Habitat * Reproductive Stage			2589.00	0.00	0.93
	Habitat Artichoke	-0.13	0.14			
	Habitat Grassland	0.33	0.20			
	RS: Chicks rearing	0.11	0.13			
	Habitat Artichoke * Chicks rearing	0.04	0.17			
	Habitat Grassland * Chicks rearing	-0.54	0.19			
2	Reproductive Stage			2595.40	6.40	0.04
	RS: Chicks rearing	-0.04	0.08			
3	Habitat			2596.00	7.01	0.03
	Habitat Artichoke	-0.14	0.12			
	Habitat Grassland	0.03	0.18			
4	Habitat + Reproductive Stage			2597.72	8.72	0.01
	Habitat Artichoke	-0.14	0.12			
	Habitat Grassland	0.05	0.18			
	RS: Chicks rearing	-0.04	0.08			

Table 2). The first model in Table 2 revealed a significant effect of the habitat type, so that the highest ingested biomass was recorded in artichoke habitat ($F_{1,2} = 3.02$, $P = 0.008$, $N = 927$). In more detail, post-hoc test revealed, over the study period, a significant difference between grassland and artichoke (Fisher LSD test: $F_{1,2} = 3.78$, $P = 0.04$, $N = 927$) and between artichoke and arable (Fisher LSD test: $F_{1,2} = 3.78$, $P < 0.001$, $N = 927$), whereas no significant difference was between grassland and arable (Fisher LSD test: $F_{1,2} = 3.78$, $P = 0.20$, $N = 927$). The mean of ingested biomass across years was

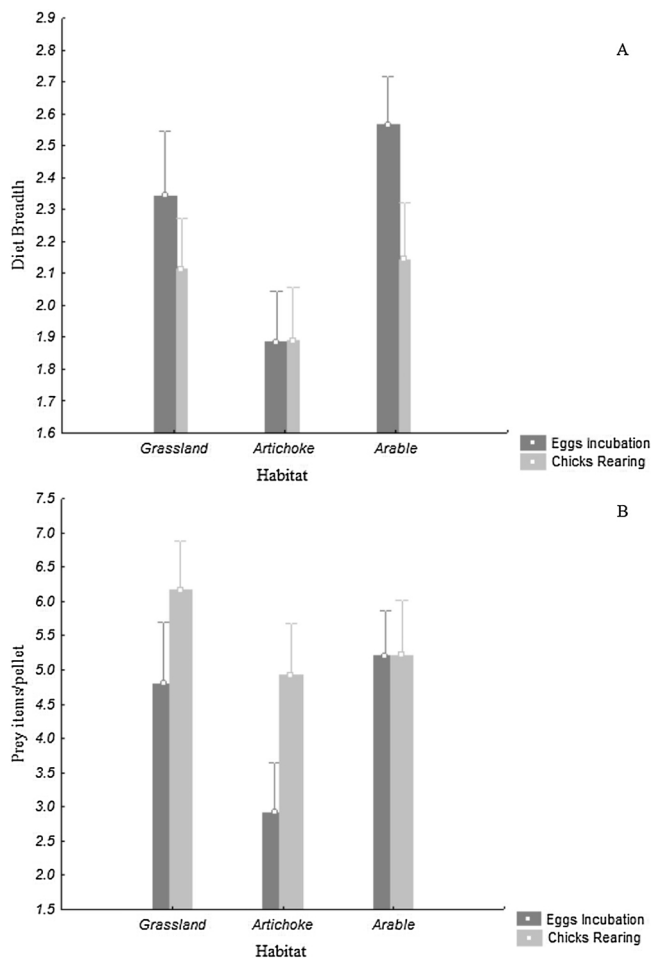


Fig. 1. Diet Breadth (A) and prey items/pellet (B) as a function of the interactive effect of habitat and reproductive stage (N = 927 pellets).

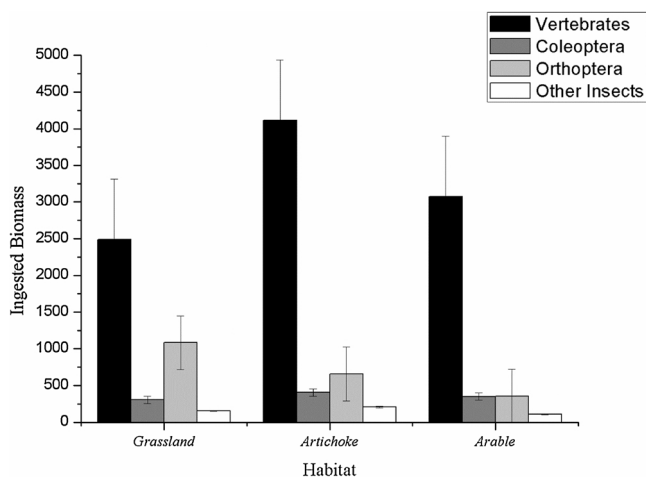


Fig. 2. Total ingested biomass (g ± SE) of the main prey groups as function of habitat type (N = 927 pellets).

similar in both artichoke land (20.64 ± 1.27 g) and grassland (18.29 ± 1.06 g), and was significantly lower in arable colonies (16.13 ± 1.05 g). Nonetheless, the 49% of large Orthoptera Acrididae and especially the 43% of total vertebrate prey biomasses, were preyed in artichoke, thus explaining the inverse relationship between the large ingested biomass and the low number of prey/pellet in such a habitat (Fig. 2, Table B).

Table 3

GLMs testing whether ingested biomass, number of prey items and diet breadth are important predictors of: i) number of eggs (N = 42 samples in 19 colonies), and ii) number of fledglings (N = 36 samples in 21 colonies). The best models are presented in bold.

Rank	Model	Estimate	SE	AIC	ΔAIC	AICw
<i>Number of Eggs</i>						
1	N prey items	-0.0007	0.0009	46.12	0.00	0.23
2	Ingested biomass	-0.0002	0.0002	46.17	0.05	0.23
3	Diet Breadth	-0.0008	0.0020	46.62	0.50	0.09
4	Diet Breadth + N prey items			47.35	1.23	0.18
	Diet Breadth	0.0043	0.0051			
	N prey items	-0.0025	0.0023			
5	Diet Breadth + Ingested biomass			47.90	1.78	0.13
	Diet Breadth	0.0020	0.0039			
	Ingested biomass	-0.0004	0.0005			
6	Ingested biomass + N prey items			48.12	2.00	0.10
	N prey items	-0.0007	0.0035			
	Ingested biomass	0.0000	0.0009			
7	Ingested biomass + Diet Breadth + N prey items			49.27	3.15	0.05
	Ingested biomass	0.0003	0.0009			
	Diet Breadth	0.0048	0.0054			
	N prey items	-0.0036	0.0047			
<i>Number of Fledglings</i>						
1	Ingested biomass	0.0013	0.0005	113.01	0.00	0.36
2	N prey items	0.0036	0.0018	114.74	1.74	0.15
3	Diet Breadth + Ingested biomass			114.82	1.81	0.14
	Diet Breadth	0.0033	0.0079			
	Ingested biomass	0.0012	0.0007			
4	Ingested biomass + N prey items			114.96	1.95	0.14
	N prey items	-0.0008	0.0039			
	Ingested biomass	0.0016	0.0012			
5	Diet Breadth	0.0111	0.0068	116.06	3.05	0.08
6	Ingested biomass + Diet Breadth + N prey items			116.07	3.06	0.08
	Ingested biomass	0.0022	0.0014			
	Diet Breadth	0.0110	0.0123			
	N prey items	-0.0049	0.0060			
7	Diet Breadth + N prey items			116.72	3.72	0.06
	Diet Breadth	0.0015	0.0110			
	Number of prey items	0.0033	0.0030			

In addition to habitat type, the second model of Table 2 revealed the global effect of reproductive stage, although the post-hoc test did not distinguish significantly between eggs incubation and chicks rearing (Fisher LSD test: $F_{1,2} = 0.22$, $P = 0.36$, $N = 927$).

3.2. Effects of diet on lesser kestrel breeding success

In total, for the breeding success analysis, we investigated 23 lesser kestrel colonies, where we collected 78 independent samples of pellets according to the main land use surrounding the colony and to the reproductive stage. In more detail, in the 3 different habitats, we got 42 samples during incubation in 19 colonies, and 36 during nestling period in 21 colonies, with an average of 11.0 ± 0.59 colony/year (range: 7–18) across the study period.

Five out of seven models from GLM analysis on the number of eggs differed by less than 2 AIC points from the best one, thus providing equal support to our data (Table 3). However, when we compared our best model with the null model (i.e. in which all of the terms, but the intercept, are excluded) we found that the null model better explains the number of eggs. Regards to the number of fledglings, four out of seven models differed by less than 2 AIC points from the best one, and were considered to provide equal support to our data. Specifically, the

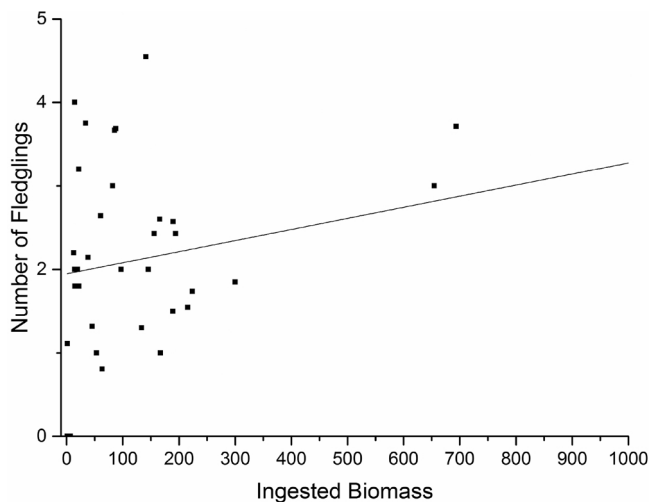


Fig. 3. Effect of the ingested biomass (g) on the mean number of lesser kestrel fledglings ($r = 0.45$; $P = 0.02$; $N = 36$ samples in 21 colonies).

first model showed a noteworthy effect ($AIC_w = 0.36$) of the ingested biomass on the number of fledglings (Table 3). Also in this case, we compared our best model with the null one and we found the latter had no important effect on the number of fledglings. As predicted by the first model, colonies with higher ingested biomass fledged more young kestrels (Fig. 3).

4. Discussion

Reduced availability of food during the breeding season is one of the main factors driving the decline of farmland birds (Boatman et al., 2004); however, the association between dietary composition of these bird species and modern agricultural landscapes is still poorly known (Clarke, Combridge, & Middleton, 2003; Robinson, 2004). In this study, we considered the lesser kestrel, an acknowledged representative of farmland species in Mediterranean agro-ecosystems, and we showed that its diet, analysed over five years, varied among farmland habitats and during the reproductive season.

The population decline observed in farmland specialists has been attributed to policy-driven agricultural changes (Donald et al., 2001; Donald, Sanderson, Burfield, & Van Bommel, 2006). Since its appearance in 1960s, the CAP has promoted the conversion of natural habitats and grassland to large-scale cultivated fields, on which pesticides and mechanised harvesters are used (Donald et al., 2002; Sanderson, Donald, & Burfield, 2005). This has determined a drastic reduction of traditional mixed farming based on rotation of crops, and of grassland habitats which had been the most suitable habitats for pseudo-steppe birds (Pe'er et al., 2014; Wolff et al., 2005). In order to reverse the biodiversity loss, the EU endorsed the reformed CAP for 2014–2020 with three principal measures: establish Ecological Focus Areas (EFAs) in farmed areas, maintain existing grassland, and grow a minimum of three different crops on any farm with > 3 ha of arable land (Pe'er et al., 2014). Despite the subsidies provided to farmers by the reformed CAP to maintain their traditional low-income farming systems, the scale of agricultural conversions is still increasing throughout the EU, and the decline of farmland biodiversity continues too (EEA, 2015). Economic costs of the reformed-CAP have increased over time, without any evident benefit for farmland birds, and as consequence the new EU agricultural policy has been flagged like a 'conservation trap' (Torres-Orozco, Arroyo, Pomarol, & Santangeli, 2016). Until now, the EU agricultural reform has failed to find a conservation management scenario applicable at large scale without generating social conflicts, in order to be beneficial for both farmers and farmland birds (Pe'er et al., 2014; Torres-Orozco et al., 2016).

In our study area, the habitat type, i.e. the land use surrounding the colony, has a strong effect on all the diet parameters of the lesser kestrel.

Traditional agri-ecosystems are an incontrovertible source of insects for farmland species (Rodríguez & Bustamante, 2008; Pokluda, Hauck, & Cizek, 2011; Littlewood, Stewart, & Woodcock, 2012). Grassland areas are peculiar components of traditional agricultural landscapes, which are already known to be particularly important for kestrel species (Costantini, Dell'Omo, La Fata, & Casagrande, 2014; Franco et al., 2004) and pseudo-steppe wildlife in general (Moreira et al., 2005; Zamora, Verdù, & Galante, 2007). Therefore, grassland areas are currently managed with measures, generally included in agri-environmental schemes (AESs) to provide more suitable habitats to insectivorous birds (Moreira et al., 2005). Not surprisingly, in grasslands of our study area, we have observed the more abundant diet (in terms of number of prey items) and the highest predation both in terms of frequency and biomass on all Orthopteran families, when compared to cereal in arable lands and artichokes. The number of prey items and the occurrence of Orthopteran families were also higher during chicks rearing than during eggs incubation; this could be the reason why we found an important contribution of the prey items/pellet in determining the number of fledglings. Contrariwise, arable land was the worse agro-ecosystem for lesser kestrels (e.g. lower ingested biomass, larger diet breadth). This is in agreement with results found both by García et al. (2006) and us (Di Maggio et al., 2016). Nonetheless, this habitat can be a relatively good foraging ground for lesser kestrels, as prey abundance peaks during cereal harvesting, giving a large but temporary contribute to diet (Catry et al., 2014). Yet, arable land suitability for lesser kestrel may change depending from vegetation height and cover (i.e. varieties of cereal crops used), and the quantity and longitude of margins (Catry et al., 2014; Rodríguez & Bustamante, 2008; Rodríguez, Tapia, Ribeiro, & Bustamante, 2013). Therefore, local conditions, we may have not considered, likely cause the low suitability of arable lands in our sample area. For instance, the borders among cereal plots are practically absent due to the wrong practice of stubble-fires after harvest, that have been destroying year after year margins and associated prey (Triolo et al., 2011).

Habitat type has proved to have a strong effect also on diet breadth during both incubation and rearing stages. In particular, individuals foraging on artichoke habitats had a higher level of diet specialization, as the small diet breadth index values show. Whether difference in diet specialization depend on intrinsic parent traits (e.g. age, experience see Forero et al., 2002; Ludynia et al., 2013) or on habitat remunerability (see Rodríguez, Johst, & Bustamante, 2006) is still an open question. Nonetheless, although benefits derived from specialism depend on multiple factors, diet specialization can influence many population parameters such as survival probability and breeding success, because individuals specializing on few prey will be more efficient than generalists (MacArthur & Pianka, 1966). This occurs amongst raptor species, in which individuals with specialized diets have commonly higher reproductive outputs than those with general diets (e.g. López-López, Verdejo, & Barba, 2009; Resano-Mayor et al., 2014).

Artichoke cultivation is a typical crop within the study area that represents an unusual foraging habitat for lesser kestrels in southern Europe (Catry et al., 2014; García et al., 2006). Although it is considered to be an intensive (see Lo Giudice, Mbohwa, Clasadonte, & Ingraio, 2014) and high-profit crop, the local temporal dynamics of cultivation could make artichoke fields suitable for lesser kestrels. Habitually they were sprayed only in winter and abandoned after the harvest, i.e. from late April – early May, so the human-induced stressors (chemical inputs, harvest, etc.) finish at the beginning of lesser kestrel reproduction, and secondary flowering plants and stubbles in harvested fields attract rich insect and small vertebrate communities, in turn increasing prey availability especially during chicks rearing. As a matter of fact, artichoke fields provide high prey availability, as witnessed by the larger number of prey items trapped there than in arable lands or

grasslands (Di Maggio et al., unpublished results), which are also easily accessible to lesser kestrels (Di Maggio et al., 2016). The low number of prey items per pellets found in artichoke fields should be thus generated from the choice of predator on the most remunerable prey among the large availability offered in such a habitat.

The strong habitat effects on species' diet may explain why both adult and juvenile lesser kestrels living in colonies surrounded by grassland and artichoke fields have a higher survival probability than colonies in arable fields (Di Maggio et al., 2016). In the Gela plain, other pseudo-steppe birds than lesser kestrels, such as the European stone curlew (*Burhinus oedicanus*), showed a marked preference for the short vegetation cover (stubbles) of harvested artichoke fields (Triolo et al., 2011).

Alternative crops have been suggested to be capable for sustaining high levels of biodiversity (Ursúa, Serrano, & Tella, 2005). In Spain, a conservation plan aimed at restoring little bustard (*Tetrax tetrax*) populations was based on an AES in alfalfa (*Medicago sativa*) crop areas. Such a conservation plan well increased population productivity, which was mainly associated with bustard nesting in AES fields (Bretagnolle et al., 2011). Alfalfa was also highly selected by lesser kestrels after harvest, and was considered to be a useful alternative when economic pressures made irrigation unavoidable in dry-lands (Ursúa et al., 2005).

4.1. Recommendations for the management of farmland habitats

Agricultural management generally aims to produce large quantities of food in the shortest time with the lowest economic costs. These short-term goals often clash with conservation and management of biodiversity and other long-term ecosystem services (EEA, 2015). Moreover, a short-term focus on maximising productivity can endanger the agro-ecosystems, in such a way that they could inevitably collapse (Foley et al., 2005). A very dangerous aspect of this maximised productivity is the increase in the level of pesticides (e.g. neonicotinoids) used in agricultural practices that in turn negatively affect invertebrate communities and consequently result in food deprivation and negative population trends for 15 insectivorous farmland species (Hallmann et al., 2014).

In our study area, artichoke fields appeared to play a key role in the intensive agricultural landscape. After the 2010, instead to be left abandoned after the harvest as before, and although opposed by the Management Plan of the Site of Community Importance and Special Areas of Conservation, ITA050001, which includes the Gela plain (<http://natura2000.eea.europa.eu/Natura2000/SDF.aspx?site=ITA050001#6>), we assisted to the increasing replacement of the majority of artichoke fields soon after their harvest with the implantation of irrigated summer crops (e.g. tomato, eggplant and melon) heavily sprayed with pesticides (M. Sarà, personal observation). This could be detrimental to lesser kestrels, especially when they are raising nestlings, due to both the reduction of available prey still present in harvested plots, and the use of pesticides in the new growing fields. This latter was particularly dangerous, as observed by the large number of poisoned females found dead inside their nests (Di Maggio et al., 2016). Thus, the maintenance of harvested artichoke fields is a specific recommendation for the Gela plain that could be extended, in case of lesser kestrel presence, to other Mediterranean agricultural areas with this distinctive cultivation.

We believe that artichoke fields can be transformed in a habitat with relatively little impact from management operations, and potentially have a larger associated biodiversity than other annual intensive crops (Di Maggio et al., unpublished results). A considerable reduction in chemical use during artichoke cultivation in autumn and winter (Lo Giudice et al., 2014), and a corresponding increase in organic farming of vegetable summer crops, plus specific actions aimed to increase the current suitability of the arable lands (e.g. halting the stubble-fires and margin restorations) would be absolutely necessary to change the agricultural management of the Gela plain. These latter actions would

reduce the depletion of arthropods and other small vertebrates, thus hindering the decline of species' ingested biomass and other diet parameters.

Therefore, if properly adjusted to environmentally friendly agriculture, a mosaic of land uses centred on artichoke fields could preserve the functioning of trophic chains in intensive agro-ecosystems. Locally, AESs and other subsidies, such as the measure 4.1 of the Regional Rural Development Regulation provided to transform the artichokes in organic cultivation, could fully achieve this target. This management, would allow the maintenance of both rich lesser kestrel populations and high-profitability crops for farmers.

Additionally, the change from chemically-based to organic cultivation, would provide a further fundamental ecosystem service to farmers, as predators are one of the biological agents for pest control (Paz et al., 2013). Thus, large populations of lesser kestrels and of other species in the guild (e.g. owls, common kestrel, stone curlew, etc.) foraging in the whole area could provide a natural rodenticide and insecticide, and this information should be disseminated among farmers and stakeholders.

Alfalfa in Spain (Ursúa et al., 2005) and artichoke in South Sicily are among the few demonstrated cases of alternative cultivation that potentially offer a win-win scenario, as they provide an optimal trade-off between biodiversity and economic sustainability for farmers.

In conclusion, we investigated a typical Mediterranean agro-ecosystem in which agricultural intensification is causing progressive habitat deterioration with time. The discovery of a profitable crop potentially benefitting our model species suggested new strategies for effective application of alternative agri-environmental practices that could halt the decrease in farmland biodiversity. Our observations in the Gela agricultural plain promoted the first steps of a process directed to create a multifunctional agricultural landscape locally capable of supporting trade-offs between financial returns from crops, landscape quality, and nature conservation.

Acknowledgements

We thank L. Zanca, J. Tysseire, R. Mascara and S. Triolo for field assistance and B. Massa, C. Cusimano and M. Coco for pellet collection and identification of prey remains. We also thank D.E. Chamberlain for his useful comments on the manuscript and S. Bondi for field and GIS assistance. Funding was provided by the Italian Ministry of Education, University and Research (PRIN 2010-2011, 20108 TZKHC). This study complies with the current Italian laws regulating scientific research on animals. Language was revised by Proof-Reading-Service.com.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:<https://doi.org/10.1016/j.jnc.2018.08.009>.

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