



**HAL**  
open science

# Land use changes and raptor population trends: A twelve-year monitoring of two common species in agricultural landscapes of Western

Alain Butet, Y. Rantier, Benjamin Bergerot

## ► To cite this version:

Alain Butet, Y. Rantier, Benjamin Bergerot. Land use changes and raptor population trends: A twelve-year monitoring of two common species in agricultural landscapes of Western. *Global Ecology and Conservation*, 2022, 34, pp.e02027. 10.1016/j.gecco.2022.e02027 . hal-03556759

**HAL Id: hal-03556759**

**<https://hal.science/hal-03556759v1>**

Submitted on 30 May 2022

**HAL** is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.



Distributed under a Creative Commons Attribution 4.0 International License

Contents lists available at [ScienceDirect](https://www.sciencedirect.com)

# Global Ecology and Conservation

journal homepage: [www.elsevier.com/locate/gecco](http://www.elsevier.com/locate/gecco)

## Land use changes and raptor population trends: A twelve-year monitoring of two common species in agricultural landscapes of Western France

Alain Butet<sup>a,b,\*</sup>, Yann Rantier<sup>a,b</sup>, Benjamin Bergerot<sup>a,b</sup><sup>a</sup> UMR CNRS ECOBIO - Université de Rennes 1, Campus de Beaulieu, 35042 Rennes, France<sup>b</sup> LTSER, Site ZA Armorique, Rennes F-35000, France

### ARTICLE INFO

#### Keywords:

Raptor  
Long term survey  
Trends  
Landscape dynamics  
Ecological indicators

### ABSTRACT

There are serious concerns about the ecological degradation caused by modern agriculture and its impact on top-chains predators. But, scientists still disagree on how to assess which are the winners and losers of Anthropocene biodiversity changes.

In this field of research, many studies have been carried out on passerine birds but long-term ones, and especially on raptors, are still missing. Our study reports a twelve-year survey on two diurnal common raptors (the Buzzard, *Buteo buteo* and the Eurasian kestrel, *Falco tinnunculus*) facing recent land use changes in Western France farmland landscapes. Precise remote sensing data were analyzed each year, allowing us to precisely describe land use changes and stability along this time series.

Buzzards showed reduced abundances in intensified landscapes and responded negatively to wood habitat instability while kestrels were more abundant and stable when areas of meadows increased. According to life history traits of these two-raptor species, we may suppose that buzzards, as generalist predators, mainly suffer of nesting site decrease while kestrels suffer of food depletion and namely voles living in meadows.

No decreasing or increasing regional or local trends was observed for buzzards during these 12-years confirming that populations seem adjust their numbers to fit to local resources. Inversely, kestrels exhibit significant population collapses at both regional and local scales.

If limiting wood habitats fragmentation or disturbance would optimize buzzard densities, drastic management measures improving surfaces and quality of grassland habitats will be necessary to reverse the decline of the Eurasian kestrel, highlighted by this study.

### 1. Introduction

Nowadays, agricultural landscapes represent the dominant land use throughout Europe and since the 1950 s, a huge increase of production has arisen, resulting from significant changes in farming policies and improvements of farming practices (Turner and Meyer, 1994). The Common Agricultural Policy (European Union) has been the main driver of this dynamics (Potter, 1997). Major changes, from field to regional scales, have occurred in order to promote competitiveness and productivity. During this period, intensification of agriculture in Western Europe led to simplest, less heterogeneous and less diversified landscapes (Meeus et al., 1990;

\* Correspondence to: UMR CNRS ECOBIO - Université de Rennes 1, Campus de Beaulieu, 35042 Rennes, France.

E-mail address: [alain.butet@univ-rennes1.fr](mailto:alain.butet@univ-rennes1.fr) (A. Butet).

<https://doi.org/10.1016/j.gecco.2022.e02027>

Received 8 November 2021; Received in revised form 19 January 2022; Accepted 19 January 2022

Available online 21 January 2022

2351-9894/© 2022 The Authors. Published by Elsevier B.V. This is an open access article under the CC BY license

(<http://creativecommons.org/licenses/by/4.0/>).

Baessler and Klotz, 2006). These evolutions are known to be responsible of many detrimental effects on biodiversity (Benton et al., 2003).

Diversity of habitats and intensity of agricultural practices influence patterns of biodiversity. In many studies, benefits of landscape heterogeneity have often been underlined and widely recognized (Devictor and Jiguet, 2007; Fischer et al., 2011; Michel et al., 2007; Billeter et al., 2008; Chiron et al., 2014). The heterogeneity of agricultural landscapes is both determined by proportion and spatial configuration of the different land uses (crop vs. semi-natural covers). Main changes which occurred over the last decades included a widespread loss of uncultivated elements and a high conversion rate of grasslands to cropped areas (Burel and Baudry, 1995). Loss of natural grasslands, woodlot patches, field margins and ditches as well as fragmentation of hedgerow networks and enlargement of crop fields, created a complex array of factors that affect farmland wildlife diversity. Nevertheless, such land use changes have received first attention only in early 1980 s (Robinson and Sutherland, 2002).

Abundance of semi-natural habitats is associated to overall heterogeneity and connectivity of farming landscapes and have been pointed by numerous authors to be an essential factor of maintaining their biodiversity (Schweiger et al., 2005). Ecologists agree on the consensus of the decline of biodiversity in most of agricultural landscapes (Butchart et al., 2010) and a large part of European biodiversity depends on the management of farming habitats within these landscapes (Stoate et al., 2001). Therefore, managing heterogeneity of farming habitats for biodiversity appears crucial and more important for high-than low-intensified landscapes (Batáry et al., 2011). This also requires to better know responses and specific requirements of species facing such changes (Fahrig et al., 2011). In predominantly agricultural regions, biodiversity response to landscape changes of recent decades has been shown to be taxon specific (Burel et al., 2004).

In Europe, the period of intensification of farm management coincided with the decline of many specialist farmland bird species (Fuller et al., 1995; Chamberlain et al., 2000; Donald et al., 2001). Birds display many key ecological functions and are often observed as interesting ecological indicators (Sekercioglu, 2006; Whelan et al., 2008). The European Farmland Bird Indicator (EFBI), adopted by the European Union, is a structural and sustainable indicator considered as useful surrogate for trends of biodiversity in agricultural landscapes (Gregory et al., 2005). Similarly, the French Breeding Bird Survey, initiated in the 1980 s, furnishes temporal trends and provides biodiversity indicators for policy markers (Jiguet et al., 2012).

Global changes have strong effects on bird species which depend on their ecology and life history traits. To maintain sustainable populations of many species, better understanding the ways by which agricultural intensification affect bird populations abundance and distribution is still needed (Raymond et al., 2010). Direct or indirect effects are diverse including mainly food supply (Potts, 1986; Brickle et al., 2000; Cardador et al., 2014); nesting habitats and/or strategy (Wilson et al., 1997; Chamberlain and Crick, 1999; Bas et al., 2009) and migratory status (Vorisek et al., 2010; Pickett and Siriwardena, 2011). More precisely, the decline of raptor abundance has been often reported in areas devoted to intensified agrosystems (Leptich, 1994; Sergio et al., 2005). Being positioned at the top of food chains, raptor species have been, since many years, surveyed as indicators of ecosystem health (Newton, 1979; Donazar et al., 1997). Indeed, for raptors, numerous long-term studies reported marked numerical and functional responses to landscape changes and their effect on prey availability (Korpimäki and Norrdahl, 1991; Potapov, 1997; Salamolard et al., 2000).

France hosts a significant part of European raptor diversity and high fluctuations over the last decades have been observed (Millon and Bretagnolle, 2004). While populations were very low in early 70 s, a partial recovery of populations was observed up to early 90 s followed by a new decline until the beginning of 2000 s (Thiollay and Bretagnolle, 2004). Considering the most common species of farming systems, Millon and Bretagnolle (2004) reported that France hosted the greatest common buzzard (*Buteo buteo*) population of Western Europe and important populations of The Eurasian kestrel (*Falco tinnunculus*).

Based on these statements and in accordance with Boano and Toffoli (2002) who claimed that population monitoring of these two-raptor species is an important conservation task, we may consider fluctuation patterns of their populations as precious ecological indicators of the evolution of recent decades dynamics of farming landscapes. Raptors need low disturbed nesting sites and high profitable foraging areas (Cody, 1985). Most of species feed on small mammals but depend more or less on this resource according to their specialist vs. generalist characters. For example, rodents account more in the diet of the Eurasian kestrel than in the common buzzard (Gensbol, 1988). Buzzards are also known to forage mainly by questing preys perched on trees at hedges or wood edges (Wuczyński, 2005). Inversely kestrels are more often seen in active hunting over wide open spaces foraging small mammals living in sparse herbaceous vegetation (Costantini and Dell'Omo, 2020). Thus, we may suppose that development and/or quality of wood vs. grassland habitats will interfere with their foraging behavior and their presence in agricultural landscapes.

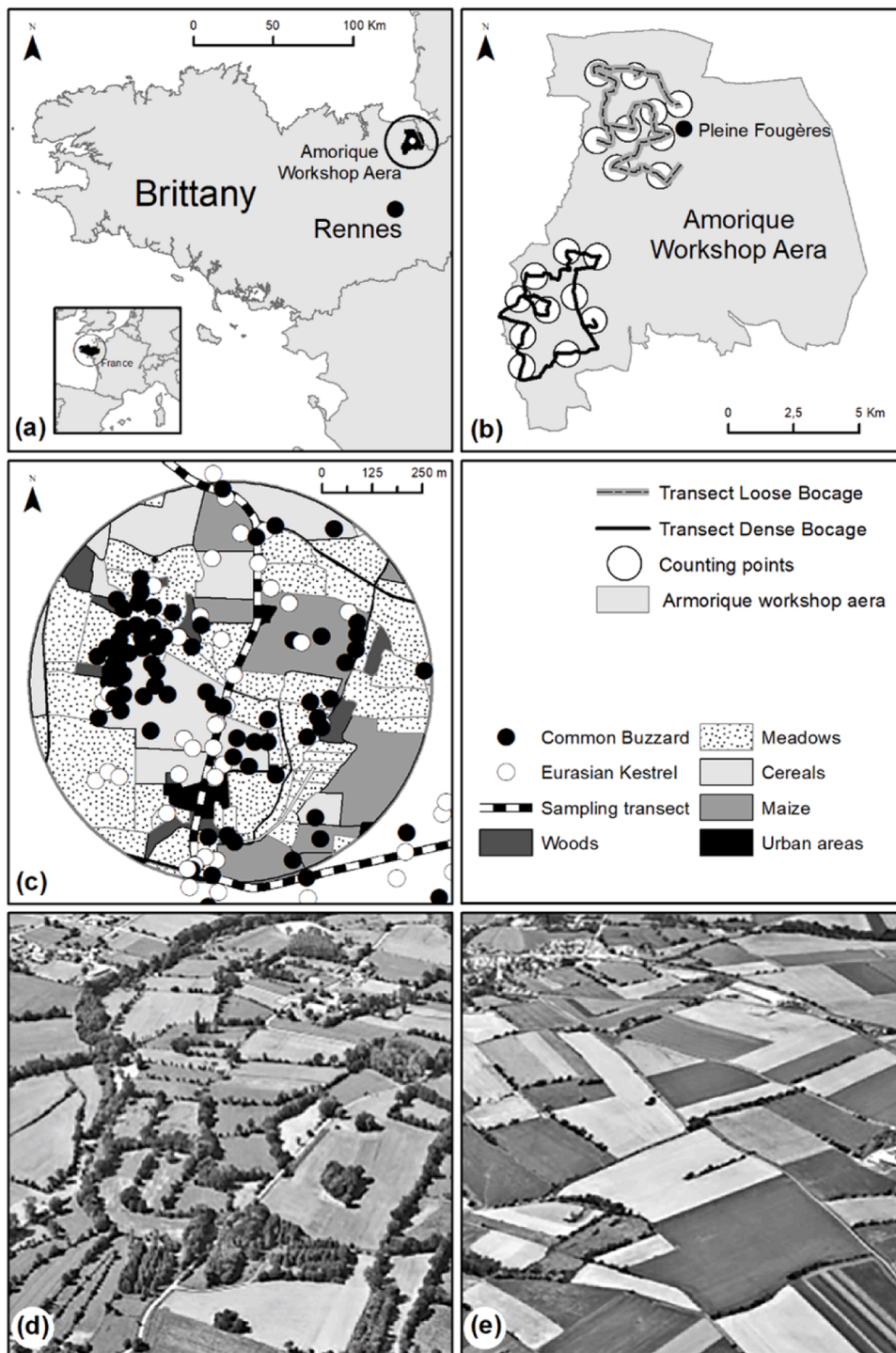
Based on a large time series compilation, a recent review reported a syndrome of consistent cycle amplitude dampening of vole populations throughout Europe (Cornulier et al., 2013). Factors responsible of such syndrome may have cascading impacts on food chains architecture but are still largely unknown and lot of controversies are still existing. For example, Hagemeyer and Blair (1997) postulated that buzzards would be less sensitive than kestrels to pesticides use. But later, Thiollay and Bretagnolle (2004) suggested that buzzards would be more sensitive to agricultural intensification than kestrels.

As suggested by many ecologists (Franklin, 1989; Lindenmayer et al., 2012), only long-term studies on dedicated agricultural areas could light such a subject. In that sense and based on a long-term study (i.e. 12 consecutive years), we collected abundance data on these two common diurnal raptors in contrasted agricultural landscape contexts of Western France whose land use dynamics varied over time and space since several decades. Aside annually raptor abundance data, annually remote sensing data were available and used to evaluate and describe precisely land use changes and stability along this time series.

The goals of this study were to test (1) if these two raptor species show positive, neutral or negative trends, observable both at regional and local scales, (2) if variations in abundance and stability of these two-raptor species can be related to dynamics of local landscape features (surfaces and stability of land uses). Finally (3), we will discuss the relevance of these results with regard to the threats that recent evolutions of agriculture in Europe induce on top-food-chain predators.

## 2. Study site

The study took place in Brittany (France) in the “Zone Atelier Armorique” (ZA, hereafter) (Fig. 1). The ZA is a research site area labeled by the French National Center for Scientific Research (CNRS) and which aims to conduct long term interdisciplinary research



**Fig. 1.** Study area: (a) Geographic localization of the Long-Term Ecological Research site “ZA Armorique”, Brittany, France – (b) Raptor transect with counting points in DB (dense bocage) and LB (loose bocage) sites. (c) 500 m radius window centered on a counting point with local land use and multi-years records of geolocated raptors. Overview of the landscape appearance of DB (d) and LB (e).

**Table 1**

Variability of the four dominant land uses illustrating the spatial heterogeneity at the level of the 19 landscape spatial replicates of DB and LB bocage sectors. Mean percent for each spatial replicate of DB and LB were calculated from land uses percent recorded in each of the 12 years of study.

	Dense Bocage										Loose Bocage												
	DB1	DB2	DB3	DB4	DB5	DB6	DB7	DB8	DB9	DB10	LB1	LB2	LB3	LB4	LB5	LB6	LB7	LB8	LB9	min	max	mean	SE
Mean % Wood areas (Wm)	2.2	6.9	8.5	18.3	6.2	3.8	8.3	2.8	4.1	21.8	0.4	5.0	9.7	0.4	2.8	0.4	0.5	2.5	2.3	0.4	21.8	5.6	5.9
Mean % Cereal areas (Cm)	14.4	11.5	14.1	15.4	8.9	13.6	19.2	13.3	16.9	25.8	36.1	18.7	13.1	42.9	30.8	24.8	42.7	32.4	34.6	8.9	42.9	22.6	10.9
Mean % Maize areas (Mam)	25.5	16.7	18.2	16.7	16.8	12.3	14.1	17.2	14.8	31.9	31.4	53.4	20.8	39.4	45.1	28.1	31.7	33.7	30.0	12.3	53.4	26.2	11.4
Mean % Meadow areas (Mm)	29.7	42.5	35.9	43.8	59.2	59.4	44.0	58.2	60.0	14.9	16.4	10.3	45.5	12.0	17.1	35.4	15.6	20.9	40.0	10.3	60.0	34.8	17.5

on anthroposystems in relation to societal issues (<https://osur.univ-rennes1.fr/za-armorique/>). The ZA is part of the European eLTER (European Long-Term Ecosystem Research) and international ILTER networks of long-term ecological research sites.

The ZA comprises three privileged sectors representing three different landscape entities: (1) a zone of agricultural landscape, (2) an alluvial plain and (3) the city of Rennes as urban context research site. Our study has been carried out in the agricultural zone which covers a total area of 13237 ha. and is dominated by a typical farming system of this region locally called “bocage”. The study area is located in the vicinity of the city Pleine-Fougères (60 km north of Rennes city, Brittany region, North-Western France), just South of the Mont-Saint-Michel Bay (48°33'26"N, 1°33'50"W) (Fig. 1a-b).

As typical landscape of Brittany, the bocage looks like a mosaic of semi-natural grasslands and crops, being bordered by some wood patches and hedgerows that play many ecological roles (Baudry et al., 2000). Due to intensification of agriculture, lot of areas of this farming system have evolved considerably since the second half of the 20th century. This result to more open landscapes, reduced hedgerow network length and semi-natural grasslands in favor of enlarged crop areas. Widespread crops are rotational grasslands, maize and cereals.

Our long-term study was conducted in two sectors of the agricultural area. The first one, in the south part, is a well preserved dense bocage (DB, hereafter) and the second one, in the north, is a more intensified loose bocage (LB, hereafter) (Fig. 1b). DB is a well conserved farming landscape where grasslands and fodder crops dominate and which is mainly oriented toward dairy production. LB, in the north part, is a more open and fragmented landscape where agriculture is mainly oriented toward mixed dairy cattle and some crop production (Fig. 1d-e).

In these bocage landscapes, the hedges bordering the fields are most often made up of pruned oaks with some clumps of pruned chestnut trees. Since 2005, due to the introduction of environmental protection conditions in the Common Agricultural Policy, most of the wetlands areas within the bocage are bordered by grass strips to restore water quality.

### 3. Material and methods

#### 3.1. Raptor abundance monitoring

The study focused on the Common buzzard, *Buteo buteo* (hereafter buzzard), and the Eurasian kestrel, *Falco tinnunculus* (hereafter kestrel). These two diurnal raptors are the most common raptors in agricultural landscape of Brittany.

The ideal habitats for buzzards are the forest edge or mosaics of afforestation within agricultural landscapes. Buzzards build their nest on a fork or branch of large trees, usually near the edge of woodlots (del Hoyo, 1994). Kestrels tolerate a wider range of open or semi-open partially forested areas. Location of the nest is variable (rocky ledges, buildings, isolated trees, wood edges...); but Kestrels hardly ever nest inside wooded areas. They rarely build nests and prefer to use abandoned corvid nests (del Hoyo, 1994).

Our goal was not to record nesting sites or to evaluate nesting density or breeding success of the two species but how frequentation of the studied area evolved over this 12-year period, regardless of wintering or breeding status of individuals. Indeed, data collection during monitoring surveys might include both nesting and non-nesting individuals during spring and summer as well as sedentary, nomadic and migratory individuals during winter. We considered long term fluctuations in abundance of all these individuals as an indicator of changes in carrying capacities of the different landscape contexts.

We used the census method developed and recommended by Leptich (1994). Activity monitoring of raptors were made by counting individuals along 2 transects of around 20 km, one in DB and one in LB (Fig. 1b). Transects were based on the road network and selected to obtain a good prospection of the two-bocage sites. Short stops (i.e. counting points) were also distributed regularly along these 2 transects (Fig. 1b-c). Both road course and counting points were chosen to obtain field of view as less disrupted as possible by physical environments and we assumed a maximal detection distance of 500 m using pairs of binoculars (Fornasari et al., 1992). This resulted in prospected areas along these two transects of 1760 ha in DB and 1687 ha in LB respectively.

Raptor census sessions along transects were done at a constant speed of 30 km.h<sup>-1</sup> with stops of 5 min at the different counting points. Counting points along transects were chosen to take into account, as much as possible, internal heterogeneity specific to each sector DB and LB.

Counting points along transects were over 1 km apart from each other to ensure independent raptor counting and to avoid overlapping of environmental data descriptors. Due to landscape configuration and visibility, 10 and 9 counting-points stops were respectively done each year in DB and LB during the different bird census session (Fig. 1b). Considering the maximal distance of view of 500 m, areas prospected at each counting point were of 78,5 ha. These counting points represented 19 landscape replicates of spatial land use heterogeneity of the bocage landscape on which we based and assessed relationships between land uses dynamics and abundance trends of raptors. This land use variability at the level of these 19 spatial replicates is summarized in Table 1.

Minimum and maximum values of the main four land cover show the significant local variability of the landscape contexts considered at the level of the 19 spatial replicates.

All along the two transects (at and between counting points), individuals perched or being in stationary flights were counted. During a same session of raptor census survey, fast moving birds were excluded to avoid multiple counting. On the other hand, from one session to the following ones, we could obviously identify the same individuals since they could not be recognized individually. This was not a problem insofar as our objective was not to estimate absolute densities but to record numerical variations of individual contact probabilities throughout the year and their long-term evolution trends.

A similar bias exists concerning the detection probabilities of the two species which display different sizes and behaviors. Again, regardless of these differences in detection probabilities, we consider that our standardized method was adapted to our expectations, namely to reveal variations of raptor activity within our study area that we consider representative of long-term trends populations of

these two-raptor species at this regional scale.

They were identified and geolocated using a laser rangefinder (Trimble True Pulse 360B) which was linked to a DGPS receiver (GNSS Trimble GeoXH 6500). This allowed us to make remote measurements which were recorded in decimetric version via the TerraSync data entry software. Then, data were all post-processed in differential mode using the GPS Pathfinder Office processing software and finally loaded into the ESRI Geographic Information System software (ArcGIS Desktop – ArcMap V.10.8). The precision of these geolocations allowed us to check that all the individuals identified were well located within the 500-meter buffer bordering the transects, but especially within the 78.5 ha counting points areas that are used further to evaluate relationships with the environmental descriptors. We made a total of 72 survey census sessions (6 per year) from January 2005 to November 2016.

Raptor population fluctuation trends were first analyzed at the regional scale (whole ZA = DB+LB or DB and LB taken separately) by including all individuals observed at and between counting points of each bocage site. Secondly, a detailed analysis at intra-site scale was performed by considering only individuals observed at counting points of DB ( $n = 10$ ) and LB sites ( $n = 9$ ).

We then studied how landscape features can influence raptor abundances and stability. To do this, only individuals observed inside counting points buffers ( $n = 19$ ) were considered to facilitate calculation of environmental variables, insofar, settled individuals in these buffer zones are supposed to respond to habitat conditions that prevail at this local scale. Raptor abundances (Buz-ab and Kes-ab) were expressed each year as the mean record of each raptor species inside the 500 m radius circle windows centered on the counting points. Between-year raptor population stability was expressed by the relative standard deviation of mean values (Buz-rsd and Kes-rsd). Both abundance and stability population data were used as intra DB- and LB-site replicates for statistical evaluation of the relations with local landscape features.

### 3.2. Land use characteristics and dynamics

To evaluate spatio-temporal variations of land use within DB and LB sites, we used vertical aerial photographs (orthophotoplan provided by IGN, National Institute of Geographic and Forest Information) as well as vertical aerial or perspective shots obtained via microflight flights ("air-papillon" service provider). At first years, photographs were delivered only as color photo prints that have subsequently been completed by georeferenced digital format. The digitization was carried out with the Geographic Information System software from ESRI company (ArcGIS Desktop, ArcMap V.10.8). With the same software, the photo-interpretation method was used for the identification of land use. Each year (from 2005 to 2016), we extracted land use information of the 19 counting points of DB and LB bocage sites by using 500 m buffer windows centered on these counting points (Fig. 1c).

We considered six main land use items that can be easily recognize and considered as most important for the settlement of raptors. Wooded habitats are essential as nesting habitats, fallows and grasslands habitats are essentials as prey reserves, cereals crops, maize crops and built areas may be considered as unfavorable foraging and nesting habitats. For each year ( $n = 12$ ) and each counting point ( $n = 10$  and  $n = 9$  for DB and LB respectively), land use descriptors were expressed as mean cover ( $m = \text{mean}$ ) and stability ( $rsd = \text{relative standard deviation of mean land use values}$ ). We thus obtained 12 landscape descriptors (mean cover and stability) labelled as follow: Wood (Wm; Wrsd) – Cereals (Cm; Crsd) – Maize (Mam; Marsd) - Fallows (Fm; Frsd) - Meadows (Mm; Mrsd) – Built areas (Bm; Brsd). All these 12 descriptors were considered as explanatory variables of abundance variations and stability of raptors (dependent variables).

### 3.3. Data analysis and statistics

Raptor abundance fluctuation patterns were observed at different scales: intra-site (counting points), inter-site (DB and LB) and whole ZA study area (DB+LB).

We performed ANOVA to test if the raptor abundance between sites (DB and LB) varied significantly or not. Before concluding about the test, we checked that the residuals of the model follow a normal distribution using Shapiro-Wilk test of normality.

To detect significant monotonic trends in our time series, we performed non-parametric Mann-Kendall tests ("trend" package, Mann, 1945, Hipel and McLeod, 2005). The strength of the trend is provided by the "tau" value and its associated *p-value*. When Mann-Kendall tests were significant, the slope of the trend was determined with the Sen's slope.

Generalized Linear Models were then used to examine the relations between all descriptors of raptor (abundance and stability) and the environmental characteristics of the counting points of the two bocage sites (DB and LB). To avoid excessive correlation between explanatory environmental variables, we selected a limited set of variables considering a correlation threshold of 0.7 (Dormann et al., 2013) based on a correlation matrix between all environmental variables (see Online Resource 1). Before running models considering a Gaussian distribution, we performed Shapiro-Wilk tests of normality to check the distribution for abundance and stability data.

According to raptor life history trait assumptions, we built 4 models. In previous results (Butet et al., 2010), we already pointed the negative effect of cultivations on these two-raptor species and their positive, but differentiated, relations to woodland and grassland habitats. Buzzards are known to depend strongly on the amount and quality of wooded habitat for nesting and hunt mostly, on the lookout, perched along wooded edges bordering rather grasslands than crops. Inversely, kestrels may nest in woodland of more varied quality but are more dependent on grasslands as main foraging habitat.

Therefore, after having considered the correlation between variables, we modeled abundance and temporal stability of buzzards (Buz-ab and Buz-rsd respectively) with the mean woodland area (Wm), the mean meadow area (Mm), the woodland temporal stability (Wrsd), the cereal temporal stability (Crsd), the maize temporal stability (Marsd), and their interactions as explanatory variables. Second, we modeled abundance and temporal stability of kestrels (Kes-ab and Kes-rsd respectively) with the mean meadow area (Mm), the cereal temporal stability (Crsd), the maize temporal stability (Marsd), the mean woodland area (Wm), the woodland temporal

stability (Wrsd) and their interactions as explanatory variables.

Dredging and model averaging procedures were performed using the “MuMin” package (Barton, 2016). Models were ranked by AICc, the Akaike information criterion corrected for small sample sizes (Burnham and Anderson, 2002), and averaged estimates of the variables were calculated over a model selection with a cumulated weight of AICc of 0.95. The effect of an explanatory variable was considered significant when its estimate (i.e. the slope of the relation) was different from zero (i.e. when its 95% confidence interval excluded zero). The quality of the models was evaluated using  $r^2$  values calculated for fixed effects for the global model (“MuMin” package, Barton, 2016).

## 4. Results

### 4.1. Raptor abundance trends

During the 72-bird census performed over the 12 successive years, a total of 1824 individuals were contacted in the whole study area. Buzzards were globally more abundant (1197 contacts) than kestrels (627 contacts).

We also observed marked seasonal fluctuations each year, buzzard and kestrel being always more abundant outside breeding season (see online resource 2). This can be easily interpreted considering that during breeding sessions, individuals were less detectable because of restricted activity and high development of vegetation. Moreover, as already pointed in method section and following assumptions of Clements (2003), we may also obviously suppose that sessions performed outside breeding season included both wintering sedentary and semi-migratory or nomadic individuals. Such seasonal fluctuations were neglected and only between-year fluctuations make sense and were considered in our study.

Buzzards were significantly less abundant in LB than in DB (ANOVA, Fvalue=54.84, Df=1,  $p < 0.001$ ; Fig. 2) while kestrels did not present significant difference between LB and DB (ANOVA, Fvalue=0.21, Df=1,  $p = 0.65$ ; Fig. 2).

Analyses by aggregating data by year at the regional scale (whole ZA; Fig. 2) indicated that only kestrel showed a significant decline trend ( $\text{tau} = -0.58$ ,  $p = 0.01$ ) whereas no significant trend was detected for buzzard ( $\text{tau} = 0.08$ ,  $p = 0.78$ ). Similar results were observed when considering raw data of the two raptor abundances that include all individual contacts on the whole ZA, during the 72 census record sessions (see online resource 3). No significant trend was detected for buzzard (Mann-Kendall test,  $\text{tau} = -0.04$ ,  $p = 0.64$ ) whereas significant decline was detected for kestrels (Mann-Kendall test,  $\text{tau} = -0.29$ ,  $p < 0.01$ ).

An analysis at intra-site scale (counting points) showed that buzzards displayed significant trends (Mann-Kendall tests with  $p$ -values  $< 0.05$ ) only in 4/19 counting points (three positive and one negative, Fig. 3), while kestrels showed significant negative trends in 10/19 sites (Fig. 4).

### 4.2. Land use characteristics and raptor abundance and stability

We examined, using generalized linear models, the relations between descriptors of birds (abundance and stability) and all the environmental characteristics of the 19 counting points of the two bocage sites (DB and LB). Except for buzzard stability (Buz-rsd), averaged estimates of the variables in each model showed that only one predictor was found significant for each of the four models (Table 2). Buzzard abundance (Buz-ab) was negatively related to the woodland temporal stability (Wrsd). Kestrel abundance (Kes-ab) and temporal stability (Kes-rsd) were respectively positively and negatively related to the mean meadow area (Mm).

## 5. Discussion

How land use affects abundance of organisms is a question increasingly associated with landscape ecology (Flather and Sauer, 1996). Since the 1980 s, ecologists agree to recognize significant declines in farmland biodiversity all over Europe (McKinney and Lockwood, 1999; Sala et al., 2000; Butler et al., 2007). These declines did not spare birds and similar concordant patterns have been reported for numerous European farmland bird species (Chamberlain et al., 2000; Donald et al., 2001; Wretenberg et al., 2006).

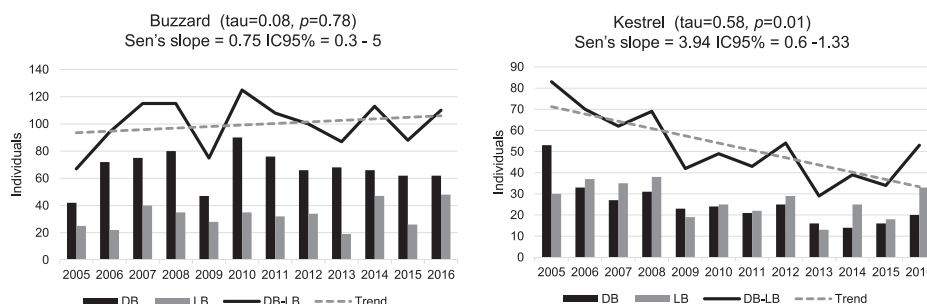
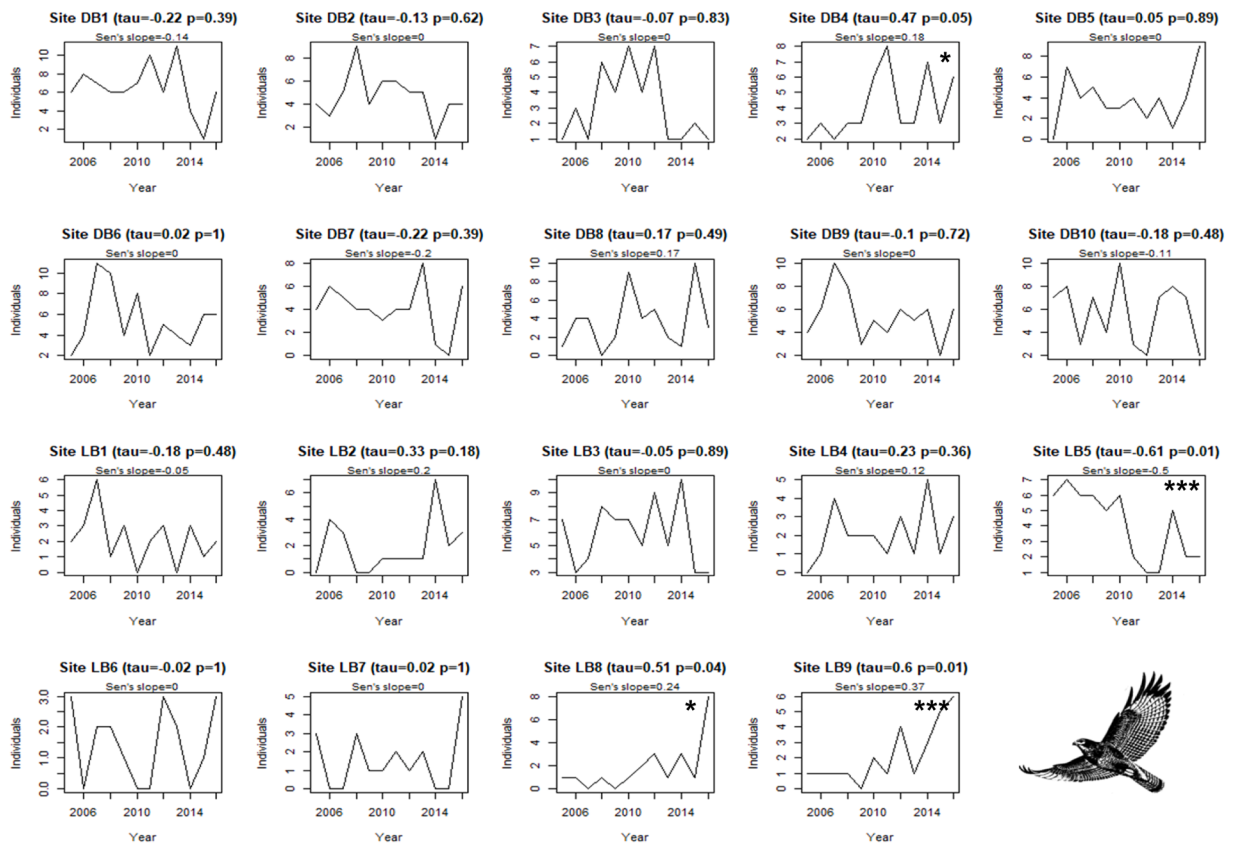


Fig. 2. Population abundance trends of Common Buzzard and Eurasian Kestrel over the period 2005 – 2016. Inter-annual variations (black lines) with a modelled linear curve (dotted grey lines) are plotted for each species. Year abundance values for DB and LB are detailed each year with colored histograms.





**Fig. 3.** 12-year population abundance trends of buzzards observed at counting point scales ( $n = 10$  for DB;  $n = 9$  for LB). For each intra-site replicate, Tau value,  $P$ -value and Sen's slope are mentioned. (\*): Significant trends.

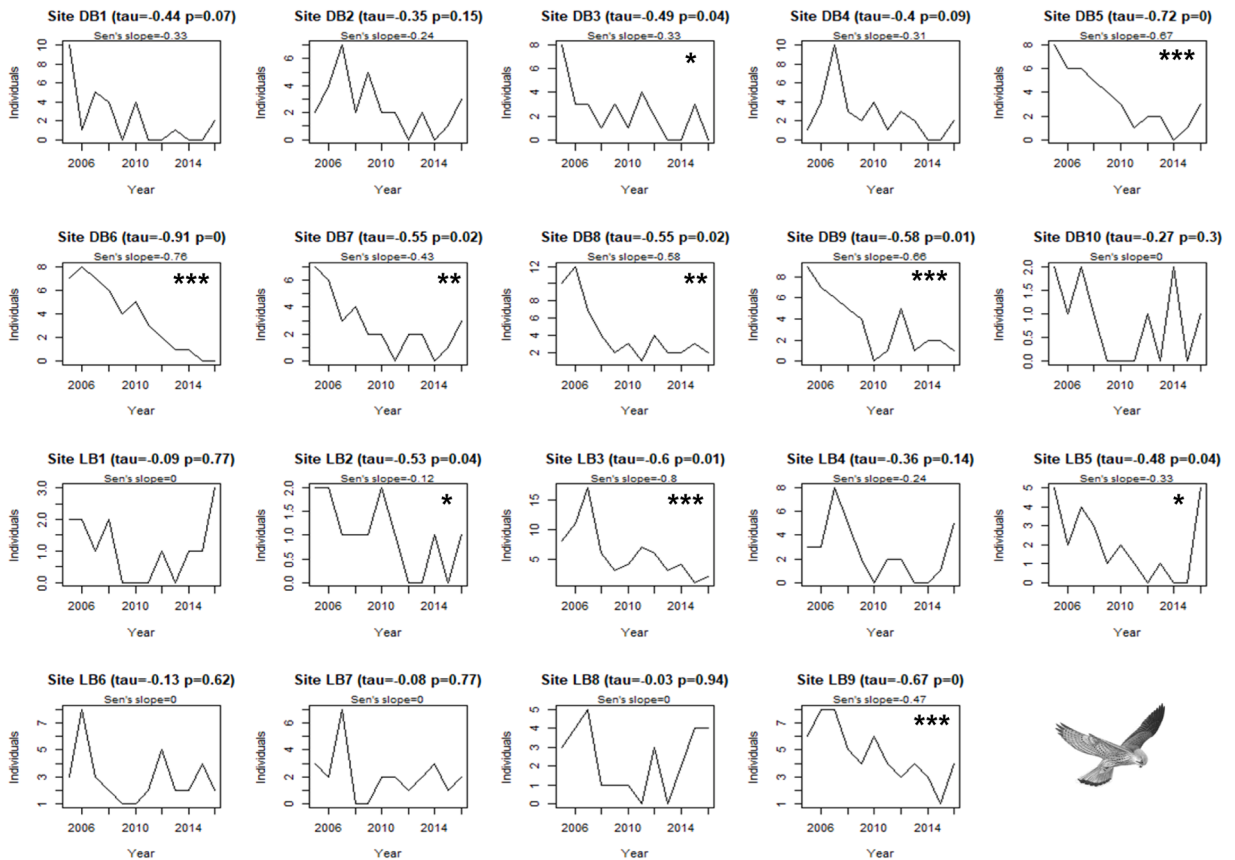
Nevertheless, a long-term trend in the North American Breeding Bird Survey, revealed a balance in which 49% of the species were increasing and 51% were decreasing (Schipper et al., 2016).

As indicators of agricultural habitat degradation, raptors are less studied than passerine birds despite they are considered as sensitive organisms of landscape environmental quality (Newton, 1998). Lot of controversies still exist on their population health and, while their distributions are well known, their dynamics are based on rough estimates and scarcity of old data makes it difficult to assess long-term trends (Thiollay and Bretagnolle, 2004). In Europe, lowest population levels for raptors seem to have occurred during the 1960 s but protection laws and abandonment of some pesticides have allowed populations to gradually recover since the 1970 s (Thiollay and Bretagnolle, 2004). In France, the FIR-UNAO survey (*Fonds d'Intervention pour les Rapaces - Union Nationale des Associations Ornithologiques*) which was the first large-scale population estimate study, dates from the 1980 s (Thiollay and Terrasse, 1984) and was completed 20 years later (Thiollay and Bretagnolle, 2004). Because France hosts two-thirds of Western Europe raptor species and the greatest density of buzzards and kestrels, our investigation was timely justified to precise recent effect of land use changes on abundance evolution trends of these two-common raptors. Moreover, our study attempted to highlight the landscape factors underlying these trends. To reach this goal, we performed annual monitoring of land use changes and mainly crop rotations at the regional scale of the whole study area.

During this study, 1824 individuals were recorded and we observed almost twice more buzzards than kestrels. We can't exclude that such difference may partly reflect an easier detection of buzzard than kestrel. But, as mentioned in methodology, our goal was not to compare precisely densities of each species but to evaluate how their detection probabilities evolved along time and to what extent we can link them to landscape dynamics.

Another bias of our study is that our monitoring method did not allow us to estimate accurate abundances of breeding or wintering populations each year, but rather indices of activity or site frequentation by the two species. We could have applied the distance sampling method (Buckland et al., 1993) especially since we had made the effort to geolocate the individuals detected by a procedure using a laser rangefinder connected to a DGPS receiver with post-processing of data for the GIS.

The implementation of distance sampling applications (Thomas et al., 2010) nevertheless shows biases (Prieto Gonzalez et al., 2017) and requires rigorous conditions of use. Firstly, individuals close to the transect are all detected, which was almost the case during our monitoring sessions. Second, individuals must not move before their detection, a condition that was globally possible to admit as negligible. Thirdly, distances are measured with accuracy. This condition was easily met for individuals perched but sometimes impossible for individuals on the ground or escaping away at our arrival. Fourthly, a minimum of 60 individuals are



**Fig. 4.** 12-year population abundance trends of kestrels at counting point scales (n = 10 for DB; n = 9 for LB). For each intra-site replicate, Tau value, P-value and Sen's slope are mentioned. (\*): Significant trends.

**Table 2**

Results of GLM testing the influence of a set of uncorrelated environmental variables on the raptor population descriptors in the 19 counting points of DB and LB sites. For the sake of simplicity, only significant variables from the model selections (estimates with a 95% confidence interval) are provided in the table.

Bird descriptor	Environmental variable	Estimate (mean and 95% CI)	r <sup>2</sup>
Buzzard abundance (Buz-ab)	Wrsd	-43.58[- 81.83 - 13.36]	0.40
Buzzard stability (Buz-rsd)	-	-	-
Kestrel abundance (Kes-ab)	Mm	0.67 [0.35 / 1.02]	0.52
Kestrel stability (Kes-rsd)	Mm	-0.01 [- 0.01 / - 0.005]	0.63

required during a monitoring session to obtain reliable abundance estimates. During our monitoring sessions, the number of contacts per transect varied from 5 to 30 individuals which was well below the minimum required.

It is mainly this last condition that led us not to convert our activity data into abundance data using this method. Moreover, our monitoring took into account the activity of breeders as well as migratory or sedentary winterers, which show very heterogeneous activity behavior and detection probabilities over time. We then considered that these temporal variations of the activity levels of the two species during the 12 years of study were a reliable proxy of the levels of frequentation of our study site by these two species and thus, whether their populations were significantly declining or not over time.

Whatever, this difference in abundance between the two species confirms previous knowledges that buzzard is the most abundant raptor in France whereas kestrels, although being better adapted to a larger range of environments, are less abundant in half of European countries and several regions of France (Millon and Bretagnolle, 2004).

We are aware that our results confirm trends already felt and mentioned in various databases (BirdLife International, 2021; IUCN, 2021). Nevertheless, the need to conserve and to plan new management of semi-natural habitats such as woodlands or meadows is most often mentioned there as hypothesis. Our research over 12 years of study is based on a landscape ecology approach in which an in-depth analysis of land uses dynamics is carried out integrating evolution of surfaces and their temporal variability. Moreover, we considered both breeding and wintering periods to evaluate changes in carrying capacities of these landscapes for raptors. Therefore,

we believe that our study provides a solid demonstration of these species / landscape interactions on which future conservation actions can be based.

During these 12 successive years, our results did not show any significant trends for buzzards at the regional scale. This corroborates conclusions of [Thiollay and Bretagnolle \(2004\)](#) who consider that buzzards enjoy a good status of its population and seem to have recovered its numbers thanks to full protection since the 1970s

Nevertheless, abundances of buzzards were always lower in the more intensified landscape (LB). It indicates that this raptor is strongly sensitive to intensification of agriculture as reported by [Sergio et al. \(2005\)](#). This global and stable dynamics observed during this decade was fully validated at local scales where no significant trends were most often observed and with only few localities displaying significant growth or declines. Moreover, the few increasing trends were either seen in dense or loose bocage indicating that these evolutions of abundance occurred regardless of the intensification level of the landscape.

If bocage landscapes are considered as rather optimal habitat for this raptor ([Nore, 1999](#)), high densities have often been reported for forest landscapes or even within forest core ([Suetens, 1989](#)). We may suppose that buzzard can maintain large populations in agricultural landscapes as long as the afforestation rate is high but are sensitive to disturbance affecting wooded habitats probably for their nesting activity. Intensification of bocage landscape always results in fragmentation of hedgerow networks and of small woodlot habitats ([Meeus et al., 1990](#)). This may explain why difference of abundances between dense and loose bocage were well differentiated for this raptor species. In a previous study carried out over three successive years ([Butet et al., 2010](#)) we already pointed that buzzard respond negatively to the decrease of wooded habitats which seems to be linked to the depletion of the small mammal species living in this habitat. In the present study, we did not have tracking data on prey availability but, the only landscape factor influencing significantly buzzard abundances, were the stability of wood habitats while analyzes do not validate a role of wooded surfaces.

As buzzard feed partly on small mammals, we could have expected a significant positive effect of meadows that our analyzes did not bring out. This may reflect that buzzard have a lot of alternative preys which make them generalist predators ([Graham et al., 1995](#)). We may suppose that disturbances of nesting habitat may more affect abundances than food depletion, confirming statements of [Nore \(1999\)](#) reporting that development of agriculture and mainly cereals spread is a real threat for buzzard but that population sizes remain stable by adapting mainly to the availability of nesting sites. Such findings are still recently relevant. For example, [Gryz and Krauze-Gryz \(2019\)](#) reported that buzzards adapted well to both positive or negative changes affecting forest landscapes of central Poland. They concluded that despite being forced to switch to alternative preys, buzzard populations grew significantly underlying its great adaptability.

Kestrels were less contacted than buzzards corroborating the general assumption that kestrel undergo negative status in Europe ([Tucker and Heath, 1994](#)) and is a species that need to be closely monitored in France ([Rocamora and Yeatman-Berthelot, 1999](#)). As pointed in [Thiollay and Bretagnolle \(2004\)](#), long-term studies of population were largely missing for this species in France and a great uncertainty still existed about trends of population dynamics. Contrary to buzzard, our results showed clear significant declining trends of kestrel abundances all over the study period. Inversely, no marked difference of abundance levels was observed according to intensification of agriculture. This overall decreasing pattern, observed at regional scale, was also confirmed at the local intra-site scales and observed both in the dense or the loose bocage. Moreover, when local trends were not significant, population trajectories seem to be most often negative, never positive. These results are rather unexpected as according to [Village \(1990\)](#), kestrels can maintain sustainable populations in rather more artificial anthropogenic landscapes than bocages. For example, kestrels are observed in suburban habitats or human-made environments like industrial fallows and herbaceous habitats associated to transport infrastructures. Although kestrels are probably adapted to a wider range of habitats than buzzards, our results provide new data validating that they seem no more adapted to agricultural practices currently developed in Western France "bocages" and exhibit a current trend of populations decline in the context of agricultural landscapes.

Regarding relations with landscape change, our results indicated clearly that kestrel were more abundant and stable when meadow areas were high. This strong dependence on meadows has been already pointed in previous studies ([Village, 1982](#); [Butet et al., 2010](#)). In Italy, [Costantini et al. \(2014\)](#) observed that kestrels breeding in areas of intensive agriculture delayed egg-laying and had offspring in poorer condition compared to those breeding in areas of temporary or permanent grasslands. Kestrels feed on a reduced range of preys varying according to habitats (for example individuals living in urban contexts). In farming systems, contrary to buzzards, numerous studies reported that they are highly dependent on vole availability ([Korpimäki and Norrdahl, 1991](#); [Viitala et al., 1995](#)). That is why there are commonly labelled as specialist predators displaying active hunting over wide open grassland spaces. Such dependence on voles has been observed for other vole-specialist raptors like Montagu's harrier ([Butet and Leroux, 1993, 2001](#); [Salamolard et al., 2000](#)) and most studies agree on the fact that specialized species of farmland habitats generally show steeper decrease than generalist ones ([Reif et al., 2008](#)). A fairly recent synthesis ([Cornulier et al., 2013](#)) revealed a wide dampening of vole populations throughout Europe which real causes are still unknown but for which effects of climatic and land use changes are highly suspected. Permanent declines of kestrels observed in our study may reflect such resource dampening associated to detrimental development of crops against permanent natural meadow areas.

Of course, it is always difficult to distangle direct effect (quality, fragmentation, loss of habitats) or indirect effect (increase of pesticides, trophic depletion) that land use changes cause on raptor populations. For example, some authors have pointed contaminations of raptors in some anthropized landscapes ([Weinburg and Shore, 2004](#); [Battaglia et al., 2005](#)) but whose relationship with the intensification or modification of landscapes is not clearly demonstrated.

Likewise, along with our analysis of landscape dynamics, monitoring the availability of prey in woods and meadows would certainly have enabled us to better understand the underlying mechanisms responsible of these two raptor population trends. We may suppose such prey depletion to be of major importance for kestrels which is known as specialist predator of grassland voles. This appear more marginal for buzzards which are more generalist predators but may more suffer from wooded habitat disturbances for nesting.

Finally, future data on dispersal rate and migration would be also useful to understand the mechanisms at work in relation to these land use changes.

## 6. Conclusions

Our long-term study covering 12 successive years of monitoring raptor abundances and land use changes provides new information on the recent evolution of populations of common diurnal raptors in the bocage landscapes of Western France.

We confirm the good health of buzzard populations which do not show any sign of pronounced declines. This stability seems to be related to the generalist character of this species, which numerically adapts to both food and nesting-habitat resources. However, we highlighted that the fragmentation of wooded habitats induced by intensification of these landscapes is able to significantly weaken population abundances. So, the buzzard is a good indicator of the quality of the wooded frame within agricultural landscapes. Maintaining significant areas of wooded habitats within these agricultural landscapes will support optimal and sustainable population levels of this raptor.

The common kestrel has shown lower abundance levels consistent with its status as the second most common raptor species in France. Compared to buzzard, the less contrasting responses observed to intensification of agriculture reflect its ability to adapt to various types of landscapes. Nevertheless, the major result of this study lies in the demonstration of a continuous decline observed in most of agricultural landscape contexts. We highlighted that this collapse of abundances was closely related to the decline of permanent grasslands, the main habitat for voles, which are basic preys of this specialist raptor. Kestrel appears to be a good ecological indicator of the abandonment of mixed farming-breeding practices that have drastically induced decrease of the amount and quality of meadow areas. Only the maintenance of meadows or permanent short crops such as alfalfa highly favorable to voles would be likely to reverse this decline.

Of course, recent evolutions of agricultural practices can act in different ways on nesting and wintering success of raptors. Beyond our demonstration of long-term correlated effects of land use changes and raptors population trends, it would be important to carry out additional investigations to better elucidate the mechanisms involved (predation, dispersal, breeding success, food depletion, ecotoxicology, etc.).

## Funding information

This study has been first supported by a grant (#Pnbc20) from the ANR-ACI ECCO research program (INSU-CNRS) and from the ZA Armorique (a Long Term Ecological Research site). Data collection was continued up to 2016, meeting the long-term survey objective of the ZA Armorique.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Acknowledgements

We thank students (Caroline Sourisseau, Yves Hoareau, Vincent Comor) who contributed to first collections and data treatments. This paper is a contribution of the research team UMR CNRS 6553 ECOBIO member of the OSU Rennes "Earth Sciences and Astronomy Observatory".

## Author contribution statement

AB managed the research program and designed the methodology. AB and YR collected field data about land use and raptor populations parameters. BB carried out statistical analyses. AB was main editor of the manuscript but all authors made substantial contributions to the interpretations of results and manuscript formatting.

## Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.gecco.2022.e02027](https://doi.org/10.1016/j.gecco.2022.e02027).

## References

- Baessler, C., Klotz, S., 2006. Effects of changes in agricultural land-use on landscape structure and arable weed vegetation over the last 50 years. *Agric. Ecosyst. Environ.* 115, 43–50. <https://doi.org/10.1016/j.agee.2005.12.007>.
- Barton, K., 2016. MuMIn: multi-model inference. R package version 1.15.6. (<https://CRAN.R-project.org/package=MuMIn>).

- Bas, Y., Renard, M., Jiguet, F., 2009. Nesting strategy predicts farmland bird response to agricultural intensity. *Agric. Ecosyst. Environ.* 134, 143–147. <https://doi.org/10.1016/j.agee.2009.06.006>.
- Batáry, P., Fischer, J., Báldi, A., Crist, T.O., Tschamtké, T., 2011. Does habitat heterogeneity increase farmland biodiversity? *Front. Ecol. Environ.* 9, 152–153. <https://doi.org/10.1890/11.WB.006>.
- Battaglia, A., Ghidini, S., Campanini, G., Spiggiari, R., 2005. Heavy metal contamination in little owl (*Athene noctua*) and common buzzard (*Buteo buteo*) from northern Italy. *Ecotoxicol. Environ. Saf.* 60 (1), 61–66.
- Baudry, J., Bunce, R.G.H., Burel, F., 2000. Hedgerows: an international perspective on their origin, function and management. *J. Environ. Manag.* 60, 7–22. <https://doi.org/10.1006/jema.2000.0358>.
- Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? *Trends Ecol. Evol.* 18, 182–188. [https://doi.org/10.1016/S0169-5347\(03\)00011-9](https://doi.org/10.1016/S0169-5347(03)00011-9).
- Billetter, R., Liira, J., Bailey, D., Bugter, R., Arens, P., Augenstein, I., Aviron, S., Baudry, J., Bukacek, R., Burel, F., Cerny, M., De Blust, G., De Cock, R., Diekötter, T., Dietz, H., Dirksen, J., Dormann, C., Durka, W., Frenzel, M., Hamersky, R., Hendrickx, F., Herzog, F., Klotz, S., Koolstra, B., Lausch, A., Le Coeur, D., Maelfait, J.P., Opdam, P., Roubalova, M., Schermann, A., Schermann, N., Schmidt, T., Schweiger, O., Smulders, M.J.M., Speelmanns, M., Simova, P., Verboom, J., van Wingerden, W.K.R.E., Zobel, M., Edwards, P.J., 2008. Indicators for biodiversity in agricultural landscapes: a pan-European study. *J. Appl. Ecol.* 45, 141–150. <https://doi.org/10.1111/j.1365-2664.2007.01393.x>.
- BirdLife International, 2021. IUCN Red List for birds. Downloaded from (<http://www.birdlife.org>).
- Boano, G., Toffoli, R., 2002. A line transect survey of wintering raptors in the western Pô Plain of northern Italy. *J. Raptor Res.* 36, 128–135.
- Brickle, N.W., Harper, D.G.C., Aebischer, N.J., Cockayne, S.H., 2000. Effects of agricultural intensification on the breeding success of corn buntings *Miliaria calandra*. *J. Appl. Ecol.* 37, 742–755. <https://doi.org/10.1046/j.1365-2664.2000.00542.x>.
- Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., 1993. *Distance Sampling: Estimating Abundance of Biological Populations*. Chapman and Hall, London.
- Burel, F., Baudry, J., 1995. Species biodiversity in changing agricultural landscapes: a case study in the Pays d'Auge, France. *Agric. Ecosyst. Environ.* 55, 193–200. [https://doi.org/10.1016/0167-8809\(95\)00614-X](https://doi.org/10.1016/0167-8809(95)00614-X).
- Burel, F., Butet, A., Delettre, Y.R., Millán de la Peña, N., 2004. Differential response of selected taxa to landscape context and agricultural intensification. *Landsc. Urban Plan.* 67, 195–204. [https://doi.org/10.1016/S0169-2046\(03\)00039-2](https://doi.org/10.1016/S0169-2046(03)00039-2).
- Burnham, K.P., Anderson, D.R., 2002. *Model Selection and Multimodel Inference: A Practical Information-theoretic Approach*. Springer-Verlag, New York.
- Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann Jr., P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G., V.M. Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J.-F., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Morcillo, M.H. n, Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vié, J.C., Watson, R., 2010. Global biodiversity: indicators of recent declines. *Science* 328, 1164–1168. <https://doi.org/10.1126/science.1187512>.
- Butet, A., Leroux, A.B.A., 1993. Effect of prey on a predator's breeding success. A 7-year study on common vole (*Microtus arvalis*) and Montagu's harrier (*Circus pygargus*) in a west France marsh. *Acta Oecol.* 14, 857–865.
- Butet, A., Leroux, A.B.A., 2001. Effects of agriculture development on vole dynamics and conservation of Montagu's harrier in western French wetlands. *Biol. Cons.* 100, 289–295. [https://doi.org/10.1016/S0006-3207\(01\)00033-7](https://doi.org/10.1016/S0006-3207(01)00033-7).
- Butet, A., Michel, N., Rantier, Y., Comor, V., Hubert-Moy, L., Nabucet, J., Delettre, Y., 2010. Responses of Common Buzzard (*Buteo buteo*) and Eurasian Kestrel (*Falco tinnunculus*) to land use changes in agricultural landscapes of Western France. *Agric. Ecosyst. Environ.* 138, 152–159. <https://doi.org/10.1016/j.agee.2010.04.011>.
- Butler, S.J., Vickery, J.A., Norris, K., 2007. Farmland biodiversity and the footprint of agriculture. *Science* 315. <https://doi.org/10.1126/science.1136607>.
- Cardador, L., De Cáceres, M., Bota, G., Giralt, D., Casas, F., Arroyo, B., Mougeot, F., Cantero-Martínez, C., Moncunill, J., Butler, S.J., Brotons, L., 2014. A resource-based modelling framework to assess habitat suitability for steppe birds in semiarid Mediterranean agricultural systems. *PLoS One* 9, e92790. <https://doi.org/10.1371/journal.pone.0092790>.
- Chamberlain, D.E., Crick, H.Q.P., 1999. Population declines and reproductive performance in skylarks *Alauda arvensis* in different regions and habitats of the United Kingdom. *Ibis* 38–51. <https://doi.org/10.1111/j.1474-919X.1999.tb04261.x>.
- Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C., Shrubbs, M., 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *J. Appl. Ecol.* 37, 771–788. <https://doi.org/10.1046/j.1365-2664.2000.00548.x>.
- Chiron, F., Chargé, R., Julliard, R., Jiguet, F., Muratet, A., 2014. Pesticide doses, landscape structure and their relative effects on farmland birds. *Agric. Ecosyst. Environ.* 185, 153–160. <https://doi.org/10.1016/j.agee.2013.12.013>.
- Clements, R., 2003. The common buzzard in Britain: a new population estimate. *Br. Birds* 95, 377–383.
- Cody, M.L., 1985. *Habitat Selection in Birds*. Academic Press Inc, San Diego.
- Cornulier, T., Yoccoz, N.G., Bretagnolle, V., Brommer, J.E., Butet, A., Ecke, F., Elston, D.A., Framstad, E., Henttonen, H., Hornfeldt, B., Huitu, O., Imholt, C., Ims, R.A., Jacob, J., Jedrzejewska, B., Millon, A., Petty, S.J., Pietiäinen, H., Tkadlec, E., Zub, K., Lambin, X., 2013. Europe-wide dampening of population cycles in keystone herbivores. *Science* 340 (6128), 63–66. <https://doi.org/10.1126/science.1228992>.
- Costantini, D., Dell'omo, G., 2020. *The Kestrel: Ecology, Behaviour and Conservation of an Open-land Predator*. Cambridge University Press, Cambridge, UK.
- Costantini, D., Dell'omo, G., La Fata, I., Casagrande, S., 2014. Reproductive performance of Eurasian Kestrel *Falco tinnunculus* in an agricultural landscape with a mosaic of land uses. *Ibis* 156, 768–776. <https://doi.org/10.1111/ibi.12181>.
- del Hoyo, J., Elliott, A., Sargatal, J., 1994. *Handbook of the Birds of the World, vol. 2: New World Vultures to Guineafowl*. Lynx Edicions, Barcelona, Spain.
- Devictor, V., Jiguet, F., 2007. Community richness and stability in agricultural landscapes: the importance of surrounding habitats. *Agric. Ecosyst. Environ.* 120, 179–184. <https://doi.org/10.1016/j.agee.2006.08.013>.
- Donald, P.F., Green, R.E., Heath, M.F., 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. *Proc. R. Soc. B* 268, 25–29. <https://doi.org/10.1098/rspb.2000.1325>.
- Donazar, J.A., Naveso, M.A., Tella, J.L., Campio, D., 1997. Extensive grazing and raptors in Spain. In: Pain, D., Pienkowski, M.W. (Eds.), *Farming and Birds in Europe*. Academic Press, London, pp. 117–149.
- Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., García Marquéz, J.R., Gruber, B., Lafourcade, B., Leitão, P.J., Münkemüller, T., McClean, C., Osborne, P.E., Reineking, B., Schröder, B., Skidmore, A.K., Zurell, D., Lautenbach, S., 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography* 36, 27–46. <https://doi.org/10.1111/j.1600-0587.2012.07348.x>.
- Fahrig, L., Baudry, J., Brotons, L., Burel, F.G., Crist, T.O., Fuller, R.J., Sirami, C., Siritwardena, G.M., Martin, J.-L., 2011. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. *Ecol. Lett.* 14, 101–112. <https://doi.org/10.1111/j.1461-0248.2010.01559.x>.
- Fischer, C., Flohre, A., Clement, L.W., Batáry, P., Weisser, W.W., Tschamtké, T., Thies, C., 2011. Mixed effects of landscape structure and farming practice on bird diversity. *Agric. Ecosyst. Environ.* 141, 119–125. <https://doi.org/10.1016/j.agee.2011.02.021>.
- Flather, C.H., Sauer, J.R., 1996. Using landscape ecology to test hypotheses about large-scale abundance patterns in migratory birds. *Ecology* 77, 28–35. <https://doi.org/10.2307/2265651>.
- Fornasari, L., Bottoni, L., Massa, R., Fasola, M., Bricchetti, P., Vigorita, V., 1992. Atlante degli uccelli svernanti in Lombardia. Regione Lombardia, Università degli Studi di Milano: 378 pp.
- Franklin, J.F., 1989. Importance and justification of long-term studies in ecology. In: Likens, G.E. (Ed.), *Long-term studies in ecology*. Springer, New York, pp. 3–19.
- Fuller, R.J., Gregory, R.D., Gibbons, D.W., Marchant, J.H., Baillie, S.R., Carter, N., 1995. Population declines and range contractions among lowland farmland birds in Britain. *Cons. Biol* 9, 1425–1441. (<https://www.jstor.org/stable/2387188>).
- Gensbol, B., 1988. Guide des rapaces diurnes, Europe, Afrique du Nord, Proche-Orient. Delachaux et Niestlé.
- Graham, I.M., Redpath, S.M., Thirgood, S.J., 1995. The diet and breeding density of common buzzards *Buteo buteo* in relation to indices of prey abundance. *Bird. Study* 42, 165–173. <https://doi.org/10.1080/00063659509477162>.

- Gregory, R.D., van Strien, A., Vorisek, P., Meyling, A.W.G., Noble, D.G., Foppen, R.P.B., Gibbons, D.W., 2005. Developing indicators for European birds. *Philos. Trans. R. Soc. B* 360, 269–288. <https://doi.org/10.1098/rstb.2004.1602>.
- Gryz, J., Krauze-Gryz, D., 2019. The common buzzard *Buteo buteo* population in a changing environment, central Poland as a case study. *Diversity* 11 (3), 35. <https://doi.org/10.3390/d11030035>.
- Hagemeijer, W.J.M., Blair, M.J., 1997. *The EBCC Atlas of European Breeding Birds*. T & AD Poyser, London.
- Hipel, K.W., McLeod, A.I., 2005. Time Series Modelling of Water Resources and Environmental Systems. <https://doi.org/10.1002/iroh.19950800107>.
- IUCN (2021). The IUCN Red List of Threatened Species. Version 2021–2. (<https://www.iucnredlist.org>).
- Jiguet, F., Devictor, V., Julliard, R., Couvet, D., 2012. French citizens monitoring ordinary birds provide tools for conservation and ecological sciences. *Acta Oecol.* 44, 58–66. <https://doi.org/10.1016/j.actao.2011.05.003>.
- Korpimäki, E., Norrdahl, K., 1991. Numerical and functional responses of kestrels short-eared owls and long-eared owls to vole densities. *Ecology* 72, 814–826. <https://doi.org/10.2307/1940584>.
- Leptich, D.J., 1994. Agricultural development and its influence on raptors in southern Idaho. *Northwest Sci.* 68, 167–171.
- Lindenmayer, D.B., Likens, G.E., Andersen, A., Bowman, D., Bull, C.M., Burns, E., Dickman, C.R., Hoffmann, A.A., Keith, D.A., Liddell, M.J., Lowe, A.J., Metcalfe, D.J., Phinn, S.R., Russell-Smith, J., Thurgate, N., Wardle, G.M., 2012. Value of long-term ecological studies. *Austral Ecol.* 37, 745–757. <https://doi.org/10.1111/j.1442-9993.2011.02351.x>.
- Mann, H.B., 1945. Nonparametric tests against trend. *Econometrica* 13, 245–259. <https://doi.org/10.2307/1907187>.
- McKinney, M.L., Lockwood, J.L., 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. *Trends Ecol. Evol.* 14, 450–453. [https://doi.org/10.1016/S0169-5347\(99\)01679-1](https://doi.org/10.1016/S0169-5347(99)01679-1).
- Meeus, J.H.A., Wijermans, M.P., Vroom, M.J., 1990. Agricultural landscapes in Europe and their transformation. *Landsc. Urban Plan.* 18, 289–352. [https://doi.org/10.1016/0169-2046\(90\)90016-U](https://doi.org/10.1016/0169-2046(90)90016-U).
- Michel, N., Burel, F., Legendre, P., Butet, A., 2007. Role of habitat and landscape in structuring small mammal assemblages in hedgerow networks of contrasted farming landscapes in Brittany, France. *Landsc. Ecol.* 22, 1241–1253. <https://doi.org/10.1007/s10980-007-9103-9>.
- Millon, A., Bretagnolle, V., 2004. Les populations nicheuses de rapaces en France: analyse des résultats de l'enquête rapaces 2000. In: Thiollay, J.M., Bretagnolle, V. (Eds.), *Rapaces Nicheuses de France*. Delachaux & Niestlé, Paris, pp. 129–140.
- Newton, I., 1979. *Population Ecology of Raptors*. T & AD Poyser, London.
- Newton, I., 1998. *Population Limitation in Birds*. Academic Press, London.
- Nore, T., 1999. Vingt ans de marquage des nichées de Buse variable (*Buteo, buteo*) dans le centre de la France. *Alauda* 57, 307–318.
- Pickett, S.R.A., Sirivardena, G.M., 2011. The relationship between multi-scale habitat heterogeneity and farmland bird abundance. *Ecography* 34, 955–969. <https://doi.org/10.1111/j.1600-0587.2011.06608.x>.
- Potapov, E.R., 1997. What determines the population density and reproductive success of rough-legged buzzards in the Siberian tundra? *Oikos* 78, 362–376. <https://doi.org/10.2307/3546304>.
- Potter, C., 1997. Europe's changing farmed landscapes. In: Pain, D., Pienkowski, M.W. (Eds.), *Bird Conservation and Farming Policy in the European Union*. Academic Press, London.
- Potts, G.R., 1986. *The Partridge: Pesticides, Predation and Conservation*. Collins, London.
- Prieto Gonzalez, R., Thomas, L., Marques, T.A., 2017. Estimation bias under model selection for distance sampling detection functions. *Environ. Ecol. Stat.* 24, 399–414. <https://doi.org/10.1007/s10651-017-0376-0>.
- Raymond, C.M., Fazey, I., Reed, M.S., Stringer, L.C., Robinson, G.M., Evelyn, A.C., 2010. Integrating local and scientific knowledge for environmental management. *J. Environ. Manag.* 91, 1766–1777. <https://doi.org/10.1016/j.jenvman.2010.03.023>.
- Reif, J., Vorříšek, P., Náštný, K., Bejček, V., Petr, J., 2008. Agricultural intensification and farmland birds: new insights from a central European country. *Ibis* 150, 596–605. <https://doi.org/10.1111/j.1474-919X.2008.00829.x>.
- Robinson, R.A., Sutherland, W.J., 2002. Post war changes in arable farming and biodiversity in Great Britain. *J. Appl. Ecol.* 39, 157–176. <https://doi.org/10.1046/j.1365-2664.2002.00695.x>.
- Rocamora, G., Yeatman-Berthelot, D., 1999. *Oiseaux menacés et à surveiller en France. Liste Rouge et Priorités*. Société d'études ornithologiques de France et LPO, Paris.
- Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., LeRoy Poff, N., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. *Science* 287 (5459), 1770–1774. <https://doi.org/10.1126/science.287.5459.1770>.
- Salamolard, M., Butet, A., Leroux, A., Bretagnolle, V., 2000. Responses of an avian predator to variations in prey density at a temperate latitude. *Ecology* 81, 2428–2441. [https://doi.org/10.1890/0012-9658\(2000\)081\[2428:ROAAPT\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2000)081[2428:ROAAPT]2.0.CO;2).
- Schipper, A.M., Belmaker, J., Dantas de Miranda, M., Navarro, L.M., Böhning-Gaese, K., Costello, M.J., Dornelas, M., Foppen, R., Hortal, J., Huijbregts, M.A.J., Martín-López, B., Pettorelli, N., Queiroz, C., Rossberg, A.G., Santini, L., Schiffrers, K., Steinmann, Z.J.N., Visconti, P., Rondinini, C., Pereira, H.M., 2016. Contrasting changes in the abundance and diversity of North American bird assemblages from 1971 to 2010. *Glob. Change Biol.* 22, 3948–3959. <https://doi.org/10.1111/gcb.13292>.
- Schweiger, O., Maelfait, J.P., Van Wingerden, W., Hendricks, F., Billeter, R., Speelmans, M., Augenstein, I., Aukema, B., Aviron, S., Bailey, D.W., Bukacsek, R., Burel, F., Diekotter, T., Dirksen, J., Frenzel, M., Herzog, F., Liira, J., Roubalova, M., Bugter, R., 2005. Quantifying the impact of environmental factors on arthropod communities in agricultural landscapes across organizational levels and spatial scales. *J. Appl. Ecol.* 42, 1129–1139. <https://doi.org/10.1111/j.1365-2664.2005.01085.x>.
- Sekercioglu, C.H., 2006. Increasing awareness of avian ecological function. *Trends Ecol. Evol.* 21, 464–471. <https://doi.org/10.1016/j.tree.2006.05.007>.
- Sergio, F., Scandolara, C., Marchesi, L., Pedrini, P., Penteriani, V., 2005. Effect of agroforestry and landscape changes on common buzzards (*Buteo buteo*) in the Alps. *Anim. Conserv.* 7, 17–25. <https://doi.org/10.1017/S1367943004001623>.
- Stoate, C., Boatman, N.D., Borralho, R.J., Rio Carvalho, C., de Snoo, G.R., Eden, P., 2001. Ecological impacts of arable intensification in Europe. *J. Environ. Manag.* 63, 337–365. <https://doi.org/10.1006/jema.2001.0473>.
- Suetens, W., 1989. *Les Rapaces d'Europe*. Perron Editions, Liège.
- Thiollay, J.M., Bretagnolle, V. (Eds.), 2004. *Rapaces nicheuses de France. Distribution, effectifs et conservation*. Delachaux & Niestlé, Paris.
- Thiollay, J.M., Terrasse, J.F., 1984. Estimation des effectifs de rapaces nicheurs diurnes et non rupestres en France 1979–1982. Fonds d'intervention pour les rapaces et union nationale des associations ornithologiques, Paris.
- Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R.B., Marques, T.A., Burnham, K.P., 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. *J. Appl. Ecol.* 47, 5–14. <https://doi.org/10.1111/j.1365-2664.2009.01737.x>.
- Tucker, G.M., Heath, M.F., 1994. *Birds in Europe: their conservation status*. *Birdlife Conservation Series n°3*. Birdlife International, Cambridge.
- Turner II, B.L., Meyer, W., 1994. Global land-use and land-cover change: an overview. In: Meyer, W.B., Turner II, B.L. (Eds.), *Changes in Land Use and Land Cover: A Global Perspective*. Cambridge University Press, Cambridge, UK, pp. 3–10.
- Viitala, J., Korpimäki, E., Palokangas, P., Koivula, M., 1995. Attraction of kestrels to vole scent marks visible in ultraviolet light. *Nature* 373, 425–427. <https://doi.org/10.1038/373425a0>.
- Village, A., 1982. The home range and density of Kestrels in relation to vole abundance. *J. Anim. Ecol.* 51, 413–428. <https://doi.org/10.2307/3974>.
- Village, A., 1990. *The Kestrel*. T & A.D. Poyser, London.
- Vorříšek, P., Jiguet, F., van Strien, A., Skorpilova, J., Klvanova, A., Gregory, R.D., 2010. Trends in abundance and biomass of widespread European farmland birds: how much have we lost? BOU Proceedings - Lowland Farmland Birds III.
- Weinburg, C.L., Shore, R.F., 2004. Factors influencing liver PCB concentrations in sparrowhawks (*Accipiter nisus*), kestrels (*Falco tinnunculus*) and herons (*Ardea cinerea*) in Britain. *Environ. Pollut.* 132, 41–50.

- Whelan, C.J., Wenny, D.G., Marquis, R.J., 2008. Ecosystem services provided by birds. *Ann. N.Y. Acad. Sci.* 1134, 25–60. <https://doi.org/10.1196/annals.1439.003>.
- Wilson, J.D., Evans, J., Browne, S.J., King, J.R., 1997. Territory distribution and breeding success of skylarks *Alauda arvensis* on organic and intensive farmland in southern England. *J. Appl. Ecol.* 34, 1462–1478. <https://doi.org/10.2307/2405262>.
- Wretenberg, J., Lindström, Å., Svensson, S., Thierfelder, T., Pärt, T., 2006. Population trends of farmland birds in Sweden and England: similar trends but different patterns of agricultural intensification. *J. Appl. Ecol.* 43, 1110–1120. <https://doi.org/10.1111/j.1365-2664.2006.01216.x>.
- Wuczyński, A., 2005. Habitat use and hunting behaviour of Common Buzzards *Buteo buteo* wintering in south-western Poland. *Acta Ornithol.* 40, 147–154. <https://doi.org/10.3161/068.040.0210>.