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Population change of Red Kites *Milvus milvus* in central southern England between 2011 and 2016 derived from line transect surveys and multiple covariate distance sampling

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Abstract. Successful reintroduction programmes are usually defined by an increase in size and extent of the new population after a given period of time. Among studies of birds, these population estimates are often focused on the monitoring of nesting attempts and productivity. For many raptors, however, this approach can overlook a large number of non-breeding adults and immature birds leading to underestimation of population size and reproductive potential. A more thorough approach is to generate assessments of total population size. In this study we used a line transect survey methodology and multiple covariate distance sampling to assess the change in population size of a reintroduced raptor species, Red Kite *Milvus milvus*, across a 2600 km^2 area of central southern England. Surveys were performed in spring and autumn between 2011 and 2016 in an area 45 km to the south of the initial English reintroduction project which started 25 years previously. Survey routes avoided using roads where possible to counter the potential attraction that such landscape features may have (e.g. increased food availability, perches etc.). The use of roads was unavoidable in some instances; however, we found no evidence of Red Kite attraction to these landscape features when comparing distances of observations from stretches along roads with 5000 randomly-generated locations. Distance of detection was influenced by bird activity (greatest when the bird was on the ground or interacting with other birds) and extent of woodland but not by time of day, seasonality or when comparing between years. During the five years, estimated population size doubled from approximately 490 to 1100 individuals; a density of one Red Kite per 2.5–3.5 km^2. This suggests an increase in the breeding population in the study area from c. 95 to 174 pairs. During the study, rate of population growth was not uniform; rapid growth was recorded in years two and three followed by a slowing over the last two years of the study. While an overall increase in population and availability of suitable nesting habitat across south-eastern England indicates that there is still potential for expansion of the Red Kite breeding population, other factors are potentially limiting this growth.

Key words: raptor, multiple covariates distance sampling, line transects

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INTRODUCTION

Knowledge of the size of animal populations and their rates of change is essential for identifying conservation priorities, designing future population monitoring methods and ultimately determining effective management strategies (Soulé 1987, Simberloff 1988, Avery et al. 1994, IUCN 2004). This is true in particular for species that need or have needed conservation management in order to become re-established across their former ranges. In these scenarios, ongoing monitoring of the population of interest is a key component of conservation management, and such activities are entrenched in the policy guidelines of the IUCN SSC Conservation Translocation Specialist Group (formerly the Reintroduction Specialist Group) (IUCN/SSC 2013).

The Red Kite *Milvus milvus* was widespread across much of Britain until the 18th century but, like many predatory birds, it suffered an extended period of persecution and was extinct as a breeding species across England and Scotland by the start of the 19th century (Lovegrove et al. 1990). A series of reintroduction programmes instituted in the late 1980s across England and Scotland

(Carter et al. 2003, Smart et al. 2010) drastically improved the species' conservation status and have proven so successful that, over the course of 20 years, the Red Kite moved from being 'Red Listed' in 1996 (Gibbons et al. 1996) to having a 'Green' listing in 2015 (Eaton et al. 2015).

Monitoring of Red Kites continues; however, in common with many other raptor species in the UK, population estimates tend to be based upon the monitoring of nesting attempts (e.g. Newton et al. 1989, Underhill-Day 1998, Roberts et al. 1999). The most recent of these estimates suggest a UK breeding population of around 1600 Red Kite pairs (Musgrove et al. 2013). Whilst extremely valuable for determining productivity and generating data for inclusion in population models, monitoring nest attempts requires considerable investment of time and effort. In addition, monitoring focused on nesting attempts will also overlook the large number of non-breeding adults and immature birds present in raptor populations (Newton 1979, Newton et al. 1989) resulting in underestimation of the true population size. Although not contributing directly to current population growth, a large non-breeding and immature component of the population provides an important indication of the population growth potential. Knowledge of the both the total population size and the breeding population, therefore, provides a more rounded means of assessing potential future population change.

A more efficient means of estimating population density across large areas is through the use of surveys accounting for detectability such as distance sampling (Beavers & Ramsey 1998, Rosenstock et al. 2002). In the UK these methods form the basis of the Breeding Bird Survey (BBS, Harris et al. 2017) and are used to produce indices of population size and change over time for commoner bird species. This monitoring of annual changes in relative abundance provides a valuable indication of the population trajectory of a species over time and is often used to identify and focus conservation priorities (Gregory et al. 2002). Without any knowledge of population size however, it can be difficult to identify the true scale and significance of any population changes.

The aim of this study was to produce estimates of regional population size for a species of global conservation concern (BirdLife International 2017) currently becoming re-established across central southern England. Although recent BBS indices identify an increase in Red Kite numbers of 99% in England between 2011–2016 and highlight a continued increase (7%) across southeast England between 2016–2017 (www.bto.org/bbs), currently there are no published estimates of the size of the Red Kite population in southern England generated from empirical assessment. Since our study area lies 50–90km from the locations used in the original English reintroduction programme 25 years previously, this work is intended to provide further insight into the rate of growth and expansion of the Red Kite population as it recolonises the landscape. This will also permit further consideration of the potential for continued population growth in one of the few areas of its range where it is not declining. We also explore some of the considerations relevant to our approach and discuss the utility of the method for determining population sizes of Red Kites and other conspicuous diurnal raptor species.

METHODS

Study area and fieldwork

Fieldwork was conducted between September 2011 and June 2016 across 2618 km^2 of central southern England in the counties of Hampshire, Wiltshire, West Sussex and Surrey (centred on 1°18′W and 51°13′N, Fig. 1). Land use across this area is predominantly mixed farming (49% of total land area is arable, 28% grassland) with scattered woodlands (17%) and some large urban areas (5%) — Andover, Basingstoke and Winchester are the only urban areas with populations exceeding 40000.

We used line transects and distance sampling to determine the size and density of the local Red Kite population. Transect routes followed a circuit based on a square pattern with sides 3 km in length. Even coverage was achieved by dividing the East and West sections into 24 smaller blocks and using a random number generator to identify a grid reference and start point for transects within each of these blocks. A 3 × 3 km square overlay was then used to determine an idealised transect route (ITR) at this location. To reduce bias, all randomized start locations were used in transect determination even when such routes resulted in only very short lengths of transect being within the study area. Each transect was walked by one of two fieldworkers.

To minimise difficulties in negotiating access across large areas of private land, transect routes utilised public rights of way (roads, footpaths, bridleways and byways) and open access land to

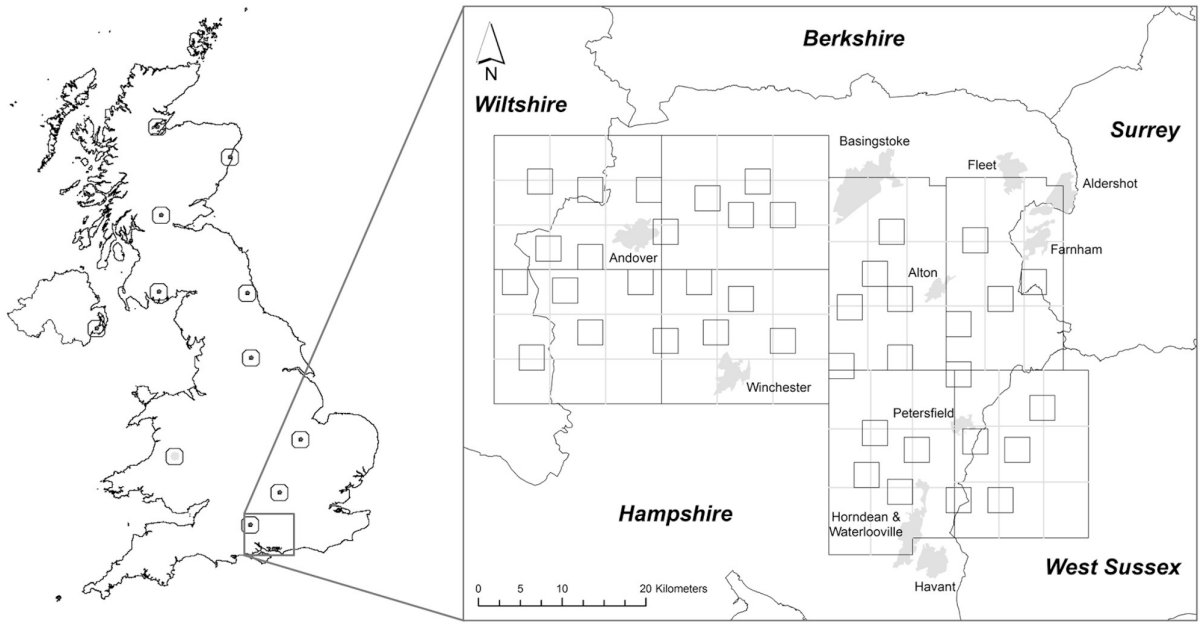

Fig. 1. Map of the UK showing the location of the study area in central southern England and the randomised locations of the 3km x 3km Idealised Transect Routes (ITRs) used in the first survey period in autumn 2011. Also shown are the 10 Red Kite reintroduction sites and the location of the relic population in Wales.

follow the ITR. The final transect route (FTR) minimized deviation from the ITR but emphasised utilisation of open access land and public rights of way rather than roads in an effort to reduce bias associated with following linear landscape features (Marques et al. 2010). In all cases the FTR differed in shape and length from the 12 km ITR square. To investigate the possible role of roads and roadside areas in attracting birds and the influence this may have had on density estimates, we compared the distribution of distances of 5000 randomly-generated points with that of our observations. Points were generated using the random number generator *runif()* function in R (R Core Team 2016) to produce pairs of latitude and longitude. These points were then plotted and their distances from the nearest road determined using the 'Near' function in ArcGIS. The distributions of these 'distance-to-roads' measurements were then compared with kite observations using two-sample Kolmogorov-Smirnov tests.

Limiting surveying to certain routes may violate a major assumption of distance sampling, i.e. that all areas have equal probability of being sampled (Buckland et al. 2001). We identified the extent of the study area that was unavailable for survey by determining the area that would remain un-surveyed if all possible transect routes (i.e. all rights of way and open access areas) were to be taken. Assuming the survey extended 700 m either side of the survey line (i.e. the maximum horizontal detection distance for a laser rangefinder), the extent of the study area not covered by this survey strip was calculated using the 'Buffer' function in ArcGIS (ESRI 2011) applied to all public rights of way and open access areas. The 5.8 km^2 (0.22%) of the entire study area not covered by this was deducted from the total size of area in all analyses. Although this does not resolve the issue of non-random placement of transects, it does ensure that all areas included within the study area have the theoretical possibility of being surveyed.

Surveys were performed in autumn (September–December) and spring (February–June). These timings were selected to enable the assessment of variation in density estimates resulting from expected peaks and troughs in population size (i.e. following post-breeding abundance and post-winter hardships respectively). To reduce the impact of nesting behaviours on detection during the spring period, surveys between February and April were prioritised and data from later surveys included only where sample sizes were considered insufficient. Transects were walked between 08.30 and 18.30 on days without rain, where wind speed was Beaufort Scale 4 or less and in conditions of good visibility.

All kites observed during the walked transects were recorded and the exact horizontal distance to each bird from the observer was determined using Swarovski 8 × 30 or Nikon Forester 550 laser rangefinders and its bearing, with a Silva compass. Red Kites can be gregarious at certain times of the year and in certain feeding situations which may lead to a bias in detectability — groups of birds are more readily observed, and often at a greater distance, than single individuals. To account for this, when encountered in groups, all birds were recorded as individuals but with size of the group and observation distance to the centre point of the group also recorded to enable analysis as both individuals and groups (clusters). Observations of Red Kites made within 90 minutes of sunset between October and March were excluded from analyses since these may have been birds moving to, or gathering at, a pre-roost (Carter & Grice 2000). This behaviour significantly alters the distribution of birds across the landscape in a way that could not be adequately accounted for during analysis. For each observation, bird behaviour, situation (i.e. whether flying, perched or on the ground), habitat, time of observation and weather conditions were also recorded.

Where vegetation surrounding the transect route restricted visibility, and thus the ability to detect birds, the observer moved away from the transect route to obtain a clearer view. These deviations from the transect route and the location of the observers at the time of each observation were recorded using Garmin 60Csx GPS units. Perpendicular distances of the observer from the transect route ('offsets') were determined using the GPS locations in ArcGIS software. Perpendicular distance of the bird to the transect was then amended appropriately through addition or subtraction of the offset.

Modelling population density

Estimates of population size and density were derived using the Multiple Covariate Distance Analysis (MCDS) engine within program Distance 7.0 (Thomas et al. 2010) and the analysis here follows the guidelines provided for that software and in the associated literature by Buckland et al. (2001).

Eight covariates (Table 1) were included in the modelling process based on *a priori* consideration of their likely influence on detection distances. These effects were presumed to be either biological (MAS, SSN, YR, TP — annual, seasonal and diurnal changes in activity, behaviour and, therefore, detectability of raptors — e.g. Bunn et al. 1995, Vergara 2010, Martínez-Miranzo et al. 2016; BLD — raptors may show a positive or negative association with urbanisation — e.g. Palomino & Carrascal 2007, Orros & Fellowes 2015), methodological (e.g. OBS — differences in observer capability — e.g. Eglington et al. 2010) or a combination of both (e.g. WD — behavioural and habitat preferences by an individual bird and its consequent reduced detection in woodland). Exploratory analyses were performed on each covariate to gain additional insight into the likely scale of its effect on detection distance. Welch's t-tests or ANOVA were used in testing factor covariates depending on the number of levels. The extent of correlation of non-factor covariates with distance was tested using Pearson's r. Strongly correlated covariates (SSN, TP, YR) were never included in the same model.

Table 1. Covariates used during modelling of population size and density of Red Kites in central southern England.

Covariate	Code	Description	Levels
Building	BLD	Area of built land within 200m radius of observer location	Non-factor
Location	LOC	Location of bird at time of observation	Flying
			On the ground
			Interacting
			Perched (above ground level)
Minutes after sunrise	MAS	Time of observation, in minutes after sunrise	Non-factor
Observer	OBS	Observer performing survey	MCS, RH
Season	SSN	Season of the survey	Autumn
			Spring
Transect period	TP	Seasonal set of surveys	10 levels
			Autumn 2011–2015
			Spring 2012–2016
Woodland	WD	Area of woodland within 200m radius of observer location	Non-factor
Year	YR	Calendar year of the survey	2011–2016

Model selection was performed according to the information theoretic approaches outlined by Burnham & Anderson (2002). Candidate models were developed according to a forward stepwise addition and selection of covariates as recommended by Thomas et al. (2010). Each covariate was assessed individually and those producing the best model fit were retained and used as a base for the selection of additional covariates. Support for individual models was assessed using evidence ratios (derived from Akaike weights) alongside a general consideration of parsimony. Because data were analysed as exact distances rather than combined into distance categories, cosine-weighted Cramér-von Mises and Kolmogorov-Smirnov tests were preferred to chi-squared goodness-of-fit tests in assessing model fit.

Where the number of observations in a survey period failed to meet the threshold of 60 observations recommended for analysis by Buckland et al. (2001), a global detection function was calculated for the pooled dataset using only those covariates providing the best fit of the model. Estimates of population size and density for all relevant time periods were then obtained through post-stratification based on the covariate values of the birds observed in each stratum (Marques et al. 2007).

Half-normal and hazard-rate models with all combinations of cosine, simple polynomial or hermite polynomial expansion series were fitted to the data. The number of adjustment terms was limited to two for all models to improve chances of convergence and reduce the likelihood of non-monotonicity and unrealistic model fit (Buckland et al. 2001). Data were right-truncated at 450 m to reduce the effect of data having an extended tail. Models featuring the use of adjustment terms were scaled by the truncation distance. Variance was estimated empirically for the separate survey periods and using the non-parametric bootstrap resampling procedure with 400 resamples for pooled estimates. This was necessary in order to counter the lack of independence of density estimates for the pooled data (Buckland et al. 2001).

All statistical analyses, other than distance sampling, were performed using R version 3.3.1 (R Core Team 2016).

RESULTS

299 transects were performed during the ten survey periods of the study (Table 2). Coverage was reasonably even across East and West sections within each survey period. Standard deviation of total length of transect walked in each region was < 8.3km for seven of the ten survey periods. Increased variability (max s.d. = 12.9 km) in the three remaining periods related to adverse weather conditions preventing survey completion on some days. Average transect length (15.13 km) was 25% longer than the ITR.

In total, 962 Red Kites were seen in 773 observations during the 4525 km of walked surveys. Of these, 128 observations were of more than one individual and birds in groups accounted for 34% (330) of all Red Kites seen.

There was no evidence of violation of the three main assumptions of distance sampling i.e. absolute detection of animals on the transect route, heaping of distances or responsive movement of target animals prior to detection (Buckland et al. 2001). Median detection distance across all observations was 181 m and the maximum, 795 m. Truncation of the data resulted in removal of 7.1% of data. The majority of these were for survey periods having adequate sample sizes, however, for two periods these were reduced to 38 and 48 observations (Table 2).

Table 2. Number of observations of Red Kites made during walked surveys in central southern England in autumn and spring between 2011 and 2016.

Season Year	Number of transects	Σ Transect lengths (km)	Observations	Observations after Truncation (450m)	Encounter rate (birds/km)
Autumn 2011	40	590.577	68	64	0.108
Spring 2012	35	531.249	70	62	0.117
Autumn 2012	26	379.15	41	38	0.100
Spring 2013	26	394.962	57	48	0.122
Autumn 2013	39	575.122	119	107	0.186
Spring 2014	35	523.74	142	130	0.248
Autumn 2014	25	372.912	187	135	0.362
Spring 2015	22	356.667	69	64	0.179
Autumn 2015	25	399.373	81	79	0.198
Spring 2016	26	401.205	128	124	0.309
Total	299	4524.957	962	859	

There was no effect of cluster size on detection distance (ANOVA, $F_{1,771} = 0.013$, p = 0.91) although mean detection distance decreased for clusters of more than five birds. For ease of analysis, all subsequent distance analyses were performed on the observations recorded as individuals.

Despite more than half (2470 km — 55%) of the transects being walked along roads, fewer than one quarter (234 of 974) of all observations were of Red Kites within 100 m of any road and 8% within 100 m of the same road from which the observation was made. The lack of effect of attraction to roads on observation distances was also demonstrated when comparing this distribution with that of 5000 randomly-generated locations. Red Kites were seen significantly further from roads than the random points (two-sample Kolmogorov-Smirnov, D = 0.163, p < 0.001; Red Kite median — 238 m, random median — 158 m). 36% (1796) of the random locations were within 100 m of a road.

Exploratory analysis of covariates revealed a difference in mean detection distances by fieldworker ($t_{96.6} = 3.36$, p = 0.001). The location of a bird at the time of observation had an effect on the distance at which it was detected (ANOVA $F_{3,978} = 11.28$, p <0.001) with individuals standing on the ground or interacting with other birds seen at greater distances than birds in other situations (Tukey test, p < 0.05). There was no difference in distance at which flying birds were seen compared with birds engaged in interaction behaviours (p = 0.58) since the latter mostly occurred when in flight. Area of woodland at the point of observation was negatively correlated with observation distance ($r_{654} = -0.08$. p = 0.04). No significant relationship was detected between any of the remaining covariates and detection distance (TP — $F_{1,973} = 1.55$, p = 0.21; YR — $F_{1,973} = 0.77$, p = 0.38; SSN — $t_{953.2} = 1.71$, p = 0.09; MAS — $r_{654} = 0.05$. p = 0.21; BLD — $r_{654} = -0.06$, p = 0.14). Despite this, the potential for interactions between covariates to lead to significant effects meant that they were still used during model testing.

The best-supported models featured half normal key functions with no adjustment terms (Table 3). Of the correlated temporal covariates, models incorporating the transect period outperformed those including either year or season. Of the three top models (ΔAIC < 2), Akaike weights and evidence ratios indicated all to have very similar levels of support. Following the principle of parsimony, we selected model '2' over models '1' and '3' for subsequent generation of density estimates since it contained fewest parameters.

During the five year period of the study, estimated population size increased from 492 to 1111 kites (a mean yearly change of approximately 128%: 95% CI — 103.6, 159.9) (Table 4, Fig. 2). Overall, there was a general increase in population size between all periods (when comparing between breeding season), with the notable exception of 2015. Here, estimates were around 50% of those of preceding or subsequent periods.

DISCUSSION

Population estimates

Our results provide an insight into the status of the reintroduced Red Kite population in central southern England. From a first nesting record in Hampshire since the mid-19th century (Holloway 1996) in 1996, and building on the success,

Table 3. Model-selection results for population and density estimates of Red Kites. Models are sorted by difference in Akaike's Information Criteria (ΔAIC) between the candidate model and model with lowest AIC. All preferred models incorporated the half normal key function; GOF — Goodness of fit test, K-S — Kolmogorov-Smirnov, CvM — Cramér von Mises with cosine weighting; w_i — Akaike weight; Evidence Ratio — ratio of current against best supported model i.e. w_1/w_j; Covariates – see Table 1. * indicates the global model.

Model No.	Covariates	AIC	ΔAIC	No. parameters	GOF K-S p	GOF CvM p	w_i	Evidence Ratio
1	TP + LOC + MAS + OBS + WD	10259.75	0.00	15	0.150	0.200	0.2365	
2	TP + LOC + OBS + WD	10259.21	0.04	14	0.184	0.200	0.2318	1.0
3	TP + BLD + LOC + MAS + OBS + WD*	10259.52	0.35	16	0.158	0.200	0.1985	1.2
4	TP + BLD + LOC + MAS + OBS	10260.98	1.81	15	0.168	0.300	0.0957	2.5
5	TP + OBS + WD	10261.44	2.27	12	0.141	0.200	0.0760	3.1
6	TP + MAS + OBS + WD	10262.12	2.95	13	0.144	0.300	0.0541	4.4
7	TP + OBS	10262.19	3.02	11	0.100	0.200	0.0522	4.5
8	TP + MAS + OBS	10263.22	4.05	12	0.146	0.300	0.0312	7.6

Table 4. Density estimates of Red Kites in south central England by year (autumn and subsequent spring) and season. CI — 95% confidence intervals empirically determined, [a] — CIs derived from 2.5% & 97.5% percentiles of the bootstrap estimate, CV — coefficient of variation.

Period	Density (Individuals/km^2)	CI[a]	No individuals	CI	CV
Year (autumn–spring)					
2011–2012	0.188	0.12–0.30	492	305–793	24.31
2012–2013	0.186	0.11–0.31	487	295–804	25.43
2013–2014	0.361	0.26–0.50	945	683–1307	16.39
2014–2015	0.456	0.29–0.72	1195	760–1880	22.81
2015–2016	0.424	0.29–0.63	1111	750–1647	19.8
Season					
Autumn	0.306	0.22–0.42	800	585–1093	15.93
Spring	0.324	0.26–0.41	849	679–1063	11.40
Pooled					
2011–2016	0.315	0.27–0.38	824	680–998	9.75

expansion and dispersal of birds from local reintroduction programmes (Carter & Grice 2002, Murn et al. 2008), we estimate the current population of Red Kites across the study area to be between 1000–1500 individuals, at a density of approximately one kite per 2.5–3.5 km^2.

The rapid growth and expansion of the population studied here is markedly different to that of the remnant UK population of Red Kites in Wales. There, a combination of high levels of natal philopatry and sub-optimal habitat has resulted in a slow rate of population growth (Newton et al. 1994). In contrast, the expansion of the population in central southern England has been comparatively rapid — taking just five years for birds from the initial release site to begin nesting 45 km to the south, in the study area (Betton & Jacobs 2009). Data from ring recoveries, resightings and radio-tagging show that first-year birds from these areas regularly disperse > 50 km from the nest site (Evans et al. 2002, Betton & Jacobs 2009).

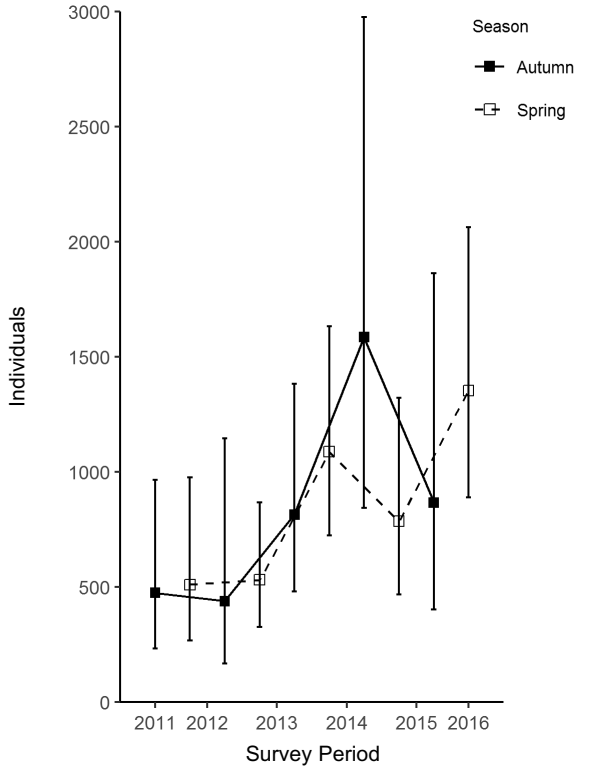

Fig. 2. Estimated population size of Red Kites in central southern England during each survey period. Confidence intervals are derived from 2.5% and 97.5% percentiles of the bootstrap estimate.

The observation of higher density estimates for autumn survey periods (when comparing within-years) is to be expected since the autumn population will also include that year's juveniles. This effect was not seen in 2012, most probably because of reduced coverage during the autumn survey. Higher estimates were expected in autumn than spring when comparing between breeding periods (i.e. autumn v subsequent spring) due to the effects of first-year and over-winter mortality. In fact, spring estimates are higher. This may be the result of dispersal and ranging of young birds from surrounding areas. Although from a very small sample, data on the movements of juvenile birds radio-tagged in southern England (Betton & Jacobs 2009) revealed that individuals spent the majority of their first six months in the vicinity of the nest site before dispersing farther afield in mid-late winter. Prior to becoming a regular breeder within the study area in 2003, more kites were recorded in this area in spring (March–June) than in autumn each year (K. Betton, pers. comm.). Given the timings of survey periods, wide-ranging movements of birds in spring may result in more birds being recorded across a wider

area than in autumn, thereby increasing the estimates of density in these periods. If this does reflect the situation within the study area in spring then estimates of breeding population should be based on analyses for the autumn survey periods in order to reduce the confounding effects of transient birds.

Age-structured population models of kites in Scotland have assumed that the majority of birds delay nesting until into their second or even third years (Smart et al. 2010), although successful nesting has been recorded by some individuals in their first summer (Evans et al. 1998, Betton & Jacobs 2009). With this in mind, and assuming a conservative estimate of 40% of all individuals being engaged in breeding in southern England (see Betton 2015), our estimates equate to an increase in the study area breeding population from approximately 95 pairs in 2011, to 174 pairs in 2015. Given that recent estimates suggest a breeding population in excess of 800 pairs in the vicinity of the original release site in the Chiltern Hills (Betton 2015), it seems possible that central southern England alone (in an area of ca. 9000 km^2) may hold 1000 pairs of breeding Red Kites. Assuming that populations have continued to increase in other areas of the country, then this would put the UK Red Kite population considerably in excess of the 1600 pairs estimated in 2013 (Musgrove et al. 2013).

Wootton et al. (2002) identified reduced productivity amongst reintroduced kites nesting in areas with the highest nesting densities and suggested that density-dependent processes may result in a slowing of the intrinsic rate of population growth. Although this may be the case, extension of the range of the species into central southern England over the intervening period, as indicated by our data and that of Betton & Jacobs (2009), suggests that the necessity of finding a nesting territory outweighs the natal philopatry of the species. Further exploration of the movements of ringed or radio-tagged birds would help identify whether this is the case. The species' preference for mixed open farmland for foraging and abundant woodlands for nesting (Minns & Gilbert 2001) is mirrored by the landscape across south and south-eastern England. Combined with an abundance of game shoots across this area and the plentiful supply of carrion this produces (both through road casualties and wounded birds), this environment appears to present an ideal opportunity for continued population growth.

In terms of population growth, whilst the aim of this study was not to perform a detailed investigation into the reasons behind the change in population size of Red Kites, an assessment of the impacts of climate variability, changing land use and the seasonal and annual variability of food resources on this would be an interesting next step. This approach will be possible once a longer time series of data becomes available and could provide interesting insights into how the population of a reintroduced species responds to environmental variability.

Methodological assessment

The number of individuals available for detection at the edge of a species' range will be lower than that in its heartland since fewer areas will be occupied (Brown 1984, Sagarin et al. 2006). As a result, surveys undertaken in these edge areas will inevitably yield fewer data since encounter rates will be low, or even zero, during some transects. This unpredictability of detection will influence the accuracy and precision of such surveys. This situation was encountered in two of the survey periods (autumn 2012 and spring 2013); sample sizes being smaller than the threshold usually considered appropriate for distance sampling (Buckland et al. 2001). However, detection probabilities, encounter rates and subsequent density estimates for these periods were neither unusually high nor low when compared with the preceding or subsequent periods. In fact, the period for which there was a marked difference in estimate was that in which most observations were made (autumn 2014). Total transect length in this period was shorter than for many others (Table 2). It seems likely that this, in combination with transects being performed in areas of highest densities, resulted in inflation of the population estimate for the study area.

The restriction of our surveys to public rights of way and roads might be expected to have had a significant influence on density estimates. This is of particular interest given that kites and other raptor species often utilise roads and verges to exploit the higher densities of carrion (Ortega & Casado 1991, Lees et al. 2013) and small mammal prey (Adams & Geiss 1983, Meunier et al. 1999, 2000) in these areas. Despite 55% of total transect length being walked along roads, < 30% of kite observations were made there. Of these, < 25% were of birds within 100 m of roads. A similar proportion (18%) seen from rights of way (other than roads) were recorded within 100 m of the right of

way from which the observation was made. This suggests that the use of these routes for surveying had little impact on the overall distribution of observations and, therefore, the estimates produced.

Average duration for completion of each transect was 371 mins equating to a walking speed of 41 m/min. During this time there is considerable scope for birds to move. The inclusion of flying birds in distance sampling can result in the violation of the assumption of uniform distances (see Fewster et al. 2008, Peak 2011, Anderson et al. 2015). In studies where only a small proportion of the target population is in flight at any one time flying birds are often excluded. For larger raptor species such as Red Kite, which spends a considerable amount of time in flight (Viñuela 1998), exclusion of such birds will remove a significant proportion of the population from surveys (in this study > 80% of records were of flying birds). Buckland et al. (2001) suggest that independent movement of target animals is acceptable provided that 'on average' they move at less than half the speed of the observer. Although soaring kites and those engaged in interaction behaviours may be considered to fall within this category (since they tended to stay in the same general area and were, therefore, moving at less than half the speed of the observer) the inclusion of birds in purposeful and directional flight (33% of all observations) is more problematic and may have led to a positive bias in our results. Whilst many of these birds may have been engaged in short-distance movements (flying to a perch etc.), the probable/actual destination of such birds was not recorded here. Determination of the nature and distances of flight behaviour in future surveys would enable greater discrimination of data and exploration of impacts on derived estimates.

Our methods provide an efficient means of assessing the density and abundance of larger bird species (such as Red Kite) which tend to be more obvious in the landscape. Although Red Kites tend to be very visible when present, their sparse distribution means that longer transect routes will increase the likelihood of encountering individuals within a suitable distance (< 500 m) compared with surveys (such as BBS) which utilise much shorter survey units. Further efficiency is achieved through a reduction in lost survey time as a result of travelling between many shorter transects. These advantages and efficiencies will be reduced should the Red Kite population continue on its current trajectory. Nevertheless, sufficient replicates (> 20 transects) are still required in order to adequately determine variance of encounter rates and produce appropriate confidence intervals. Accurate measurement of distance is one of the key assumptions of distance sampling (see Buckland et al. 2001). If violated, this can lead to a negative bias on density estimates that cannot be overcome by increasing sample size (Chen 1998). The use of rangefinders to produce accurate distance measurements also enables more thorough quantitative assessment of the goodness of fit of the model than is possible when using estimation and distance bands (Buckland et al. 2001, Newson et al. 2005). However, this means that greater effort and expense is required to train and equip fieldworkers to perform the surveys accurately and effectively.

Although the rate of population growth slowed during the course of this study, there appear to be few natural factors identifiable in the short term which would lead to a significant reduction in the rate of expansion of the reintroduced Red Kite population across southern England. Despite improved conservation legislation and public awareness, it is likely that non-natural mortality arising from anthropogenic sources such as secondary poisoning by second-generation anticoagulant rodenticides (SGARs) (e.g. Murn & Hunt 2011, Hughes et al. 2013, Walker et al. 2018), ingestion of lead (e.g. Miller et al. 2002, Fisher et al. 2006, Meyer et al. 2016) and ongoing persecution (Holmes et al. 2000, RSPB 2009, 2018, Smart et al. 2010) are likely to be the most significant constraint on population size of this, and other, raptor species in the UK.

Despite these threats, the expansion in size and range of the UK Red Kite population presents a positive outlook for the species' long term persistence. This is significant, particularly when viewed in the context of considerably lower productivity and reduced Red Kite population sizes across Europe (Knott et al. 2009), and leaves few doubts as to the value of undertaking similar, carefully-considered reintroduction programmes.

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STRESZCZENIE

[Zmiany wielkości populacji kani rudej w południowej Anglii w latach 2011–2016]

Sukces programu reintrodukcji jest zazwyczaj określany na podstawie wzrostu liczebności i zasięgu nowej populacji po upływie określonego czasu. W przypadku badań nad ptakami, takie szacunki liczebności populacji często bazują na monitorowaniu podejmowanych prób gniazdowania oraz sukcesu lęgowego w reintrodukowanej populacji. W przypadku wielu ptaków drapieżnych takie podejście może jednak pomijać dużą liczbę ptaków nielęgowych, co prowadzi do niedoszacowania wielkości populacji i jej potencjału rozrodczego.

Kania ruda była szeroko rozpowszechniona w Wielkiej Brytanii do XVIII w., ale w związku z działalnością człowieka jej populacja lęgowa zanikła w Anglii i Szkocji do końca tego wieku. Pod koniec lat 80-tych XIX w. w Anglii i Szkocji rozpoczęto programy reintroducji tego gatunku, które zakończyły się sukcesem. Jednak monitorowanie liczebności populacji kani rudej jest trudne i nie uwzględnia frakcji ptaków niedojrzałych płciowo i nielęgowych. Celem badań było oszacowanie liczebności kań rudych na terenie południowej Anglii (na powierzchni 2600 km^2) w odległości 50–90 km od miejsc, gdzie 25 lat wcześniej rozpoczęto program reintrodukcji tego gatunku (Fig. 1). Obszar ten w większości pokrywają tereny rolnicze (49% powierzchni zajmują uprawy, a 28% – łąki).

W badaniach wykorzystano metodykę badań transektowych z zastosowaniem szacowania odległości obserwowanych ptaków od transektu. W ten sposób możliwe było oszacowanie liczebności, jak i zagęszczenia ptaków. Na początkowym etapie wyznaczono transekty w kształcie kwadratu o długości boku 3 km (Fig. 1). W celu zminimalizowania problemów z dostępem do terenów prywatnych, rzeczywiste trasy transektów poprowadzono przez tereny publiczne. Transekty prowadzono w taki sposób, aby (na ile to było możliwe) unikać dróg, gdyż potencjalnie mogą one przyciągać znajdujące się w okolicy ptaki w związku ze zwiększoną w takich miejscach dostępnością pożywienia oraz miejsc wykorzystywanych jako czatownie. Ponadto, w przypadku, gdy możliwość obserwacji terenu była utrudniona (np. poprzez zadrzewienia), obserwator zmieniał ustaloną trasę transektu tak, aby uzyskać odpowiednią widoczność. W związku z powyższym we wszystkich przypadkach rzeczywiste trasy transektów odbiegały od początkowo wyznaczonych. Kontrole transektów prowadzono jesienią (wrzesień–grudzień) oraz wiosną (luty-czerwiec). Taki wybór terminów kontroli umożliwił oszacowanie najwyższej i najniższej wielkości populacji, gdyż badania obejmowały okres polęgowy, z licznymi młodymi ptakami, które wykluły się w danym sezonie, oraz okres po zimie, podczas której śmiertelność ptaków jest wyższa.

W analizach statystycznych uwzględniono czynniki zarówno związane z charakterystyką badanego transektu, jak i samych kontroli, które potencjalnie mogły wpływać na obserwacje ptaków. Po uwagę wzięto także odległość obserwowanych ptaków od transektu, oraz ich zachowanie podczas obserwacji, gdyż wpływa to na możliwość ich wykrycia. W modelowaniu detekcji ptaków uwzględniono łącznie 8 zmiennych (Tab. 1).

Łącznie w latach 2011–2016 skontrolowano 299 transektów, obserwując 962 osobniki kani rudej (Tab. 2). 34% wszystkich ptaków obserwowanych było w grupach liczących co najmniej dwa osobniki. Największa odległość, z jakiej zaobserwowano ptaki to 795 m. Analizując czynniki wpływające na możliwość zaobserwowania ptaków nie stwierdzono, aby drogi istotnie przyciągały te ptaki. Pora dnia, sezon (jesień/wiosna) czy rok prowadzenia badań nie miały związku z odległością, z jakiej zauważano ptaki. Na odległość, z jakiej ptaki były wykrywane miały wpływ: aktywność ptaków — były one obserwowane najdalej od transektu wtedy, gdy znajdowały się na ziemi lub wchodziły w interakcję z innymi ptakami, oraz udział zadrzewień — im wyższy, tym bliżej transektu obserwowano ptaki (Tab. 3).

W ciągu pięciu lat (2011–2016) szacowana wielkość populacji podwoiła się z około 490 do 1100 osobników (Tab. 4, Fig. 2). Szacunkowe zagęszczenie kań na badanym terenie wyniosło jeden osobnik na 2,5–3,5 km2. Na podstawie danych literaturowych można zakładać, że ok. 40% wszystkich osobników w populacji kani rudej przystępuje do lęgów. Uzyskane wyniki sugerują więc wzrost populacji lęgowej na badanym obszarze z ok. 95 do 174 par. W trakcie badań tempo wzrostu populacji nie było jednolite. Szybki wzrost zanotowano w początkowych latach badań, a następnie nastąpiło pewne spowolnienie (szczególnie wyraźny spadek liczebności odnotowano w 2015 r., Fig. 2). Autorzy sugerują, że ogólny wzrost populacji oraz dostępność odpowiednich siedlisk lęgowych w południowowschodniej Anglii mogą wskazywać, że nadal istnieje potencjał do ekspansji populacji lęgowej kani rudej, jednak inne czynniki (zatrucia i prześladowanie przez człowieka) potencjalnie mogą ograniczać wzrost liczebności populacji .