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Accepted Version

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Journal:	*Bird Study/Ringing & Migration*
Manuscript ID	BS-2019-014.R2
Manuscript Type:	Original Article
Date Submitted by the Author:	12-Aug-2019
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Keywords:	multiple covariates distance sampling, raptor, Population

Population change of Common Buzzards *Buteo buteo* in central southern England between 2011 and 2016

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Population change of Buzzards

Keywords: multiple covariates distance sampling, raptor, population

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Abstract

Capsule: Distance sampling identified an increase in estimated population size of Common Buzzards *Buteo buteo* in central southern England between 2011-16 of more than 50% . The rate of population growth slowed in later years.

Aims: To assess the utility of a targeted distance sampling protocol to derive seasonal and annual population estimates for Common Buzzards across an area of southern England.

Methods: We used a line transect survey methodology and multiple covariate distance sampling to assess population density and abundance of Common Buzzards in spring and autumn between 2011and 2016 across a 2600km^2 area of central southern England.

Results: Estimated population size increased by more than 50%, from ca. 2900 to 4500 individuals, across the period in a trend similar to that shown by Breeding Bird Survey (BBS) data.

Discussion: A slowing of the growth in population size of Common Buzzards in central southern England suggests that the species may be approaching carrying capacity in this area. These results also suggest that currently employed broad scale survey methodologies adequately reflect the general population trends for this species. Our data provide the first published estimates of the Common Buzzard population in central southern England derived from direct empirical assessment.

Introduction

Identifying the population size of a species is a key step in developing and implementing an effective conservation strategy (Soulé 1987, Frankham 1995). Unfortunately, determining population size can be difficult and expensive. Difficulties can arise due to particular behaviours of the study species (e.g. mobility, shyness, crypsis, nocturnality), scarcity or its occupancy of habitats where access or movement is difficult etc. (Anderson *et al.* 2015). Also, the spatial scale required to produce meaningful estimates influences survey effort, the level of sampling and cost. To improve efficiency in data collection many broad-scale studies use multi-species surveys or monitoring programmes utilising volunteer fieldworkers (see e.g. Riseley *et al* 2008, Jiguet 2009). There are consequences to these approaches, however, and analyses must consider the effects of surveying for more than one species at a time (e.g. reduced effort per species, heterogeneity in species detectability (Johnston *et al*. 2014) and potential difficulties generating sufficient observations of rarer species (but see Sanderlin *et al*.2014)) and the variation in skills and intrinsic differences between volunteer observers (e.g. in hearing, visual acuity, level of concentration, stamina, image-processing, tiredness; e.g. Link *et al.* 1994, Peterjohn *et al.* 1995, Jiguet 2009, Eglington *et al.* 2010).

Species abundance and density estimates are now often generated using a distance sampling methodology. This technique uses the recorded distances of objects of interest to randomly-placed survey routes or points to estimate animal density or abundance (Buckland *et al.* 2001). A key assumption of this method is that all objects at zero distance (g(0)) are detected and that probability of detection decreases with increasing distance from the route or point. Accurate measurement of these distances is also assumed, however, it is often necessary to simplify survey methodologies (e.g. by using a small number of distance bands rather than accurate measurement) to encourage participation and adherence to protocols (Newson *et al.* 2008, Quesada *et al.* 2010). There is a trade-off between the quality of the estimate and simplicity of the method (Rabe *et al.* 2002) and simplification will result in detection functions that are less robust (Johnston *et al.* 2014), reduces estimated detectability (Stanbury & Gregory 2009) and decreases the precision of derived estimates (Stanbury & Gregory 2009, Ekblom 2010).

Estimates of population size for bird species in the UK tend to focus on breeding populations. These estimates are usually derived from indices of relative abundance generated as part of the Breeding Bird Survey (BBS) (e.g. Newson *et al.* 2008, Riseley *et al.* 2008, Musgrove *et al.* 2013). Although useful for many species, the use of the breeding pair as the unit of interest is less appropriate for certain species and will underestimate population size (Newson *et al.* 2008). This is particularly true for many raptor species where individuals do not breed until into their third year (Davis & Davis 1992) and where a significant proportion of the adult population is not breeding in any one year (Newton 1979, Hunt 1998, Kenward *et al.* 1999, 2001), as is the case with the Common Buzzard (*Buteo buteo*, hereafter 'Buzzard'). Accurate estimation of population size is therefore necessary if the aim of monitoring is to provide an objective assessment of population trends – particularly where species may be increasing or decreasing. Using methodologies suitable for certain species groups to produce population estimates may then provide a means of periodically validating or calibrating indices that are applied more widely.

The Buzzard was lost from many parts of its range in Britain due to the combined effects of widespread persecution in the 19th century, a crash in preferred prey populations (Rabbit, *Oryctolagus cuniculus*) in the 1950s and the effects of organochlorine pesticides in the 1960s and 1970s (Sim *et al*. 2000). Until the 1980s Buzzards in Britain were confined to Scotland, Wales and Western England. Since then, the enactment of improved wildlife conservation legislation (e.g. banning of organochlorine pesticides and comprehensive legal protection) and increasing public awareness have led to a significant increase in the species' population size and range. Most recent assessments indicate that the species has now recolonised many of the areas of the UK from which it had been lost (Clements 2002, Musgrove *et al.* 2013).

The primary objective of this study was to use a distance sampling methodology to produce local and regional population estimates of Buzzards in central Southern England. We also draw comparisons with population estimates derived using other methodologies and discuss the utility of our approach for determining population sizes of Buzzards and other conspicuous diurnal raptor species on a larger scale.

Methods

Study area and fieldwork

The study was conducted between September 2011 and June 2016 across two areas (designated 'East' and 'West') covering ca. 2600km^2 of central southern England in Hampshire, Wiltshire, Surrey and West Sussex (centred on 1^o 18'W and 51^o 13'N – see Fig. 1). Land use within the study area is primarily mixed farming (arable and grassland) with scattered small woodlands; although the extent of woodland is higher in the East (26687ha) than West (17634ha). The study area contains ten urban areas of which five have human populations exceeding 40000 (Nomis 2016).

We used a line transect combined with distance sampling methodology (Buckland *et al.* 2001, 2004) to determine the population size and density of Buzzards. Each transect was a circuit based on a square with each side measuring 3km. Even coverage of the study area 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. An idealised transect route (ITR) at this location was then identified using a 3km x 3km square overlay. Negotiating access across such a large area of private land was impractical and so transects made use of public rights of way and open access land, the ITR serving as a point of spatial reference to facilitate the identification of a circuit of appropriate length through the selected area. Transects followed the ITR as closely as possible. Where deviations were necessary, alternative routes prioritised open access land and other rights of way types before roads in an effort to reduce bias associated with following obvious linear landscape features (surfaced roads, field edges, fences and hedgerows; Ortega & Capen 2002, Marques *at* al. 2009). Edge effects were minimised by including all randomised start locations even when these resulted in the transect breaching the study area boundary. In these cases, only the lengths of transect within the study area were included in analyses.

Surveys were performed in ten transect periods, two each year, between Sep-Dec ('autumn') 2011 and Feb-May ('spring') 2016. Two seasonal transect periods were used to enable assessment of expected fluctuations in density associated with post-breeding abundance and overwinter mortality. Transects started between 08.30 and 10.00 from a randomised start point and took 3-7 hours to complete. The direction (anti- or clockwise) of travel was also randomised. Each transect was performed by one of two fieldworkers (MS or RH). Transects were walked only on days with no rain, good visibility and when wind strength was no greater than Beaufort force four.

All birds observed during the walked transects were recorded. When groups of birds were encountered, the number of individuals was noted. For each observation, observer location was recorded using Garmin 60 Csx GPS units and horizontal distance and bearing to each bird (or to the centre point of groups; Buckland *et al.* 2001) from the observer using Swarovski Laser Guide 8x30 or Nikon Forester 550 laser rangefinders and Silva compasses. Where a bird in flight had obviously been disturbed from a perch by the fieldworker just prior to detection, measurement of distance was taken from the fieldworker location to the original perch. Bird behaviour, situation (i.e. flying, perched or on the ground), habitat, time of observation and weather conditions were also recorded for each observation. To minimise double-counting, fieldworkers noted, where possible, the plumage morph of all birds (following Glutz von Blotzheim *et al.* (1971)), specific aspects of plumage (e.g. bright tail, prominent breast patches etc.) and location of obvious moult (in remiges or rectrices). Where there was still uncertainty regarding the status of an individual, the observer noted their confidence in the observation being new on a percentage scale where '0' indicated a certainty that the individual had already been recorded, and '100' where it had not. Bird movement and relative timings and location of previous observations were used to inform this assessment. This enabled later exclusion of observations from analyses, based on confidence. We adopted a conservative approach to inclusion of data, retaining only those where confidence exceeded 70%.

Where the ability of a fieldworker to detect birds was compromised by visibility from the transect route (e.g. obstruction by surrounding vegetation), the observer moved a short distance away from the transect to obtain a clearer view before returning and continuing along the route. Although the fieldworker followed a map of the transect route it is unlikely that they will have been standing exactly on the transect line (i.e. at $g(0)$ – zero distance from the line) at the time of making any observation. The perpendicular distance of the fieldworker from the transect route at the time of each observation was determined using the GPS locations viewed in GIS. These 'offsets' were then used to correct the calculated distances of the observations to the transect line through either addition or subtraction of the offset (depending on the relative positions of the observer and bird to the transect route). All GPS locations obtained using WGS-84 were transformed to British National Grid using the Ordnance Survey 'OSTN02' transformation in ArcGIS (ESRI 2011).

The restriction of transect routes to public rights of way and open access areas may have resulted in the violation of the distance sampling assumption that all areas have equal probability of being sampled. We determined the extent of entire study area unavailable for surveying using the 'Buffer' function in ArcGIS (ESRI 2011). In this we produced a survey strip corresponding to the maximum operational distance of the rangefinder (700m) on each side of all rights of way and open access areas and deducted the extent of study area not covered by these strips (5.8km^2) from the total size of the study area in all subsequent distance analyses.

To investigate the possible role of roads and roadside areas in attracting birds, we also 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 compared using two-sample Kolmogorov-Smirnov tests.

Density estimation

Population size and density estimates were derived using both the Conventional Distance Sampling (CDS) and Multiple Covariates Distance Sampling (MCDS) engines within program Distance 7.0 (Thomas *et al.* 2010). Distance analysis here follows the guidelines provided for that software and in the associated literature by Buckland *et al.* (2001, 2004).

Five covariates (Table 1) were included in the MCDS modelling process on the basis that each was assumed *a priori* to influence the ability to detect birds through a biological or methodological effect (Burnham 1981, Thompson 2002, Diefenbach *et al.* 2003). A two-level factor covariate (OBS) was included to account for the likely variation in ability of fieldworkers to detect birds. Area of woodland at the point of observation (WDS) is also likely to impact detection distance due to an inverse relationship with range of view (i.e. the maximum range of vision). Values for this covariate were determined from the CEH Land Cover Map 2007 (Morton *et al.* 2014) by measuring the area of woodland within a 250m radius of the point of observation using ArcGIS 10.2 (ESRI 2011). STRATA was included as a covariate in order to account for potential differences in topography or habitat quality between the two sections of the study area, since this may result in differential detection distances. Bird activity and behaviour, and thus detectability, will vary throughout the day (e.g.

Kendall 2014, Öberg *et al.* 2015). Here, TIME was defined as the number of minutes after sunrise for each observation. Lastly, the situation of the bird, i.e. whether on the ground, perched or in flight, was included as a factorial covariate, LOC. The inclusion of flying birds in distance sampling can present a number of problems, primarily due to violation of the assumption of uniform distances through responsive movement, and double-counting (see Fewster *et al.* 2008, Anderson *et al.* 2015). Where this occurs, estimates will tend to be overestimated (Buckland *et al.* 2001). Although exclusion of flying birds from analyses is possible, this approach is best used for species in which only a small proportion of the population will be in flight at any one time (Buckland *et* al. 2008). This is clearly not the case for many soaring raptor species and exclusion of such data was not appropriate. Instead, we adopted a 'look-ahead' approach to improve the likelihood of birds being recorded before they responded to the presence of the fieldworker (Burnham *et al.* 1980, Anderson *et al.* 2015).

Relationships between covariates and the ability to detect birds were explored prior to modelling, although failure to detect any effect here did not prevent inclusion in model assessments. Factor covariates were tested against perpendicular distance using either Welch's t-tests or ANOVA. Exploration of the potential relationship between continuous, non-factor covariates and distance was performed using Pearson's r and regression.

Models with uniform, half-normal and hazard-rate key functions were fitted to the data. Automatic addition of adjustment terms was enabled for analyses using CDS. For the MCDS engine, however, this was restricted to a maximum of two cosine, simple polynomial or hermite polynomial adjustment terms. Model fit was assessed with reference to cosine-weighted Cramér-von Mises and Kolmogorov-Smirnov tests. Data were truncated at 550m to remove a lengthy tail and all models incorporating adjustment terms were scaled by the truncation distance.

Overloading of the MCDS engine with covariates is more likely to result in failure of the algorithm to converge (Thomas *et al.* 2010). To counter this, we follow the guidelines of Thomas *et al.* (2010) who advocate the forward stepwise addition of individual covariates, retaining those which contribute to reducing Akaike's Information Criteria (AIC). AIC was used to select between models (Burnham & Anderson 2002).

All statistical analyses, other than distance sampling, were performed using R version 3.3.1.

Results

4490km of surveys were completed during the 10 transect periods (Table 2). Coverage was higher in the eastern section of the study area with 2295km of surveys walked on 151 transects compared with 2194km on 145 transects in the western section. The average duration of each transect was 371 minutes (365 in Spring vs 377 in Autumn).

4274 observations of 5174 individuals were made during the study. Birds were seen in groups of up to 32 individuals, however, 85% of observations were of single birds (mean group size = 1.2 ± 0.75). 63% of observations were of birds in flight (cf. 37% perched or on the ground). Of these, 62% were birds which were soaring, hovering or interacting with other species, rather than in obvious directional transit movements.

There was no difference between the distributions of number of observations and the number of transects walked (and therefore, transect length) for each season (autumn X^2 (4) = 3.08, p = 0.55; spring (X^2 (4) = 1.75, p = 0.78) indicating that more observations were made when more transects were walked. Significantly more individuals were seen during spring surveys than in autumn (X^2 (3) = 160.25, p <0.001) despite the total length of surveys undertaken being similar.

Histograms of the distribution of perpendicular distances indicated detection on and close to the transect line remained at or near 100% in all survey periods. Median detection distance across all data was 178m. Truncation of data above 550m resulted in the loss of 2.8% of observations (99 observations of 143 individuals) but left more than 330 observations per period; comfortably above the threshold of 60-70 generally recommended for modelling using Distance (Buckland *et al.* 2001). Sufficient data were available to enable the modelling of separate detection functions, and the inclusion of different covariates, for each period.

55% of the total length of transects was walked along roads. 28% of all observations involved Buzzards within 100m of any road and only 11% were of birds within 100m of the same road as that from which the observation was made. There was no indication of a bias in observation of birds near to roads when comparing the distribution of distances with that of 5000 random locations (two-sample Kolmogorov-Smirnov, D- <0.001, p = 0.99; Buzzard median – 264m, Random median – 158m). 36% of the random locations were within 100m of a road compared with 21% of Buzzard locations

Exploratory analyses of covariates

The distance at which birds were detected reduced as the extent of woodland at the point of observation increased. This effect was negative across the entire dataset (r = -0.166, t_{4157} = -10.8, p < 0.001) and in all survey periods (p < 0.002) except spring 2013 (r = -0.079, t_{363} = -1.51, p = 0.13).

None of the remaining covariates showed any consistent relationship with detection distance. Mean detection distances were similar between both sections of the study area and varied significantly only in 2012 (spring, $t_{474.13}$ = -3.5, p <0.001, mean East – 178.9m, West – 207.7m; autumn, $t_{255.2}$ = -

2.06, p = 0.04, mean East – 179.2m, West – 214.2m). The situation of birds (i.e. whether on the ground, perched or in flight) had no significant influence on detection distance (ANOVA $F_{2,4156}$ = 2.42, p =0.09, Tukey test, p > 0.15). Observer effects on detection distances were identified in one of the four survey periods where data were collected by more than one fieldworker (spring 2013, t_{139} = 3.67, p <0.001, mean MS – 208m, RH – 148m). Although timing of an observation had a bearing on detection distance in two periods (autumn 2012, $F_{1,402}$ = 4.91, p = 0.027; and spring 2015, $F_{1,307}$ = 5.99, p = 0.015), there was no significant effect during the other eight periods.

Model fitting

MCDS models having reasonable fit (i.e. with Cramér von Mises and Kolmogorov-Smirnov tests p > 0.3) were developed for all periods (Table 3) except periods 5-7. Although statistics assessing model fit for these periods produced p> 0.1, their detection functions and quantile-quantile plots indicated that more birds than expected were observed close to the transect route. As model fit was reasonable in these periods, we still present the outputs from these but emphasise their being on the margins of acceptability. CDS models were preferred in period 7 (p >0.5), however, the model with lowest AIC (Half-normal + three cosine adjustments) showed signs of over-fitting and issues in maintaining monotonicity. A model using CDS with a Uniform key function is preferred for this period. Among the MCDS models, TIME, WDS/WDD and STRATA had the greatest effect on AIC, and appeared in the majority of preferred models for each survey period.

Population estimates

Population size and density estimates increased throughout the course of study and were 0.6 birds km^{-2} higher by 2016 than at the start of the study. Our analyses suggest an increase in estimated population size of 56%, from 2883 individuals in 2011 to 4485 in 2016 (Table 4). The average annual rate of increase across the four-and-a-half years of the study was 12.5% but this slowed in successive years (from 15% to 1% in autumn and 43% to -1% in spring; Figure 2).

More birds were seen in spring surveys (1.31km^{-1}) than preceding autumn periods (1.0 bird km^{-1}) even though total lengths of transects walked were shorter in spring for all periods except autumn 2012-spring 2013. Estimated density was also consistently higher for surveys performed during the spring (Table 4; means: spring – 1.59 autumn – 1.44).

Discussion

Population density estimates

We used a distance sampling-based methodology to estimate the population density of a conspicuous diurnal raptor species within an area of central southern England. These estimates indicate that the Buzzard population increased by more than 50% over the course of the study (Figure 2). In contrast, Buzzard populations in the adjacent SW region have shown a comparatively modest rate of increase since 1995 (+13% - Harris *et al*. 2017). Differences in the rate of population change between these two regions may be a function of there being a higher number of available potential territories in regions neighbouring the SW population and the consequent dispersal of individuals from higher to lower density areas (Walls & Kenward 1998).

The reduced rate of population growth during the last three survey periods mirrors estimates derived from BBS data for the SE region (www.bto.org/bbs) which, although showing an overall increase of 1104% since 1995, indicate a slowing of population growth to a point of a slight decline (-2%) between 2016-17. The reasons for this are unclear, especially since rates of breeding success for Buzzards have increased across the UK during these years (Woodward *et al.* 2018) – although regional differences will be masked in these national estimates. Nevertheless, a number of factors may be operating to limit population growth, including: the ongoing impacts of viral haemorrhagic disease (VHD) on UK rabbit populations (Harris *et al.* 2019), the abundance of which has been shown to influence breeding productivity and population increase in Buzzards in the UK (Graham *et al.*1995; Swann & Etheridge 1995); the continued impacts of secondary poisoning by rodenticides (e.g. Christensen *et al*. 2012) and ingestion of lead in spent ammunition (Pain *et al.* 2009); and a potential increase in persecution in response to the perceived predation pressures on game bird populations from increasing Buzzard abundance.

Despite the observed declines between 2016-17 in this study and BBS, continued population growth in areas of the SE region which lie to the north and east of the study area still appears likely since they will have been recolonised later and will be further from reaching carrying capacity; a situation highlighted by Harris *et al.* (2014). In addition, now that the scale of human-induced population constraints appears to have substantially reduced, carrying capacities are likely to have increased and be governed mostly by the availability and suitability of food and breeding habitat. In southern England, there is likely to be a proportionally greater extent of suitable breeding habitat in the SE region compared to the historical strongholds in the SW since the landscape is more heavily wooded (Forestry Commission 2016). As a result, continued population growth in this region is likely for the foreseeable future.

Atlas data (Balmer *et al.* 2013) show the Buzzard to be uniformly distributed across all 10x10km squares of SE England and from more than 85% of all 2km x 2km tetrads covered by atlas fieldwork (2007-2012) in the SE region. Assuming that habitat quality and availability within our study area is representative of that throughout the remainder of the SE region, then our density estimates indicate a population size of 27500-32500 individuals in SE England. Translation of this figure into an estimate of the breeding population is difficult, since a significant proportion of Buzzards will not make a nesting attempt each year, either due to immaturity, lack of status and inability to find a mate or hold a territory (Davis & Davis 1992, Kenward *et al*. 2000). Using the estimate suggested by Kenward *et al.* (2000) of only one in four individuals breeding each year, results in an estimate of between 3440-4125 pairs in SE England. This represents a breeding density of 18-22 pairs per 100km^2, similar to that found by Sim *et al.* (2001) in one of their West Midlands study areas. This is still lower than the 41 pairs 100km^{-2} recorded by Newton *et al.* (1982) across a large area in mid-Wales, and substantially lower than the densities (78 pairs 100km^{-2}) recorded in ideal wooded habitat in central Europe (Melde 1956, Thiollay 1967). Since the coarse regional population estimate presented here is an extrapolation from our derived estimates, any variation in the quality of those landscape characteristics representing suitability for Buzzards (e.g. food and prey density, disturbance, persecution, habitat structure and mosaic etc.) will influence its validity.

Alongside the estimates of overall abundance within the study area, our study provides an interesting comparison of the apparent abundance of Buzzards between autumn and spring periods. Several studies have determined that juvenile Buzzards tend to remain within their natal territory for the first few months after fledging (Davis & Davis 1992, Walls *et al.* 1999) and that most do not disperse more than 50km from the natal site in their first year (Walls & Kenward 1998). This is particularly the case in landscapes with a significant arable component (Walls *et al.* 1999) where Buzzards often exploit the easy foraging for invertebrates provided by ploughed fields (Dare 1957). As a result, there is unlikely to have been any significant loss of first year birds from the study area in the autumn, and in fact we expected higher densities for surveys in this period. The potential impacts of overwinter and courtship mortality (Tubbs 1974, Simpson 1993) would theoretically compound this expected difference in seasonal abundance. That this is not the case may reflect more on seasonal variation in bird behaviour, and its influence on detectability, than demographics. Increased time spent soaring and in display behaviours in spring resulted in improved detectability during spring surveys. The supplementation of the autumn population by juveniles will also have been offset by dispersal (Walls & Kenward 1995) and high rates of mortality for Buzzards in the four months after fledging (Kenward *et al.* 2001).

Methodological assessment

We encountered few obvious methodological issues with the study. Poor model fit using the MCDS engine for the autumn 2014 was most likely the result of higher than expected numbers of birds recorded between 275-325m in this survey period. This problem was not identified in other periods suggesting that it is unlikely to represent any significant issue with survey design. Similarly, the issue of poor precision was limited to one survey period and stems from reduced coverage; the level of effort being lowest in this period (Table 2).

The covariates most frequently included within preferred models (WDS, STRATA and TIME) indicate that woodland cover was the most important factor affecting Buzzard detectability. Increasing density of woodland reduces the view of surrounding habitats leading to birds generally being detected at shorter distances than in more open habitats. This effect is also likely to account for the inclusion of STRATA in many preferred models since a greater proportion of the landscape area was woodland (and, therefore, a higher proportion of surveys performed through woodlands) in the eastern section of the study area. Lastly, the inclusion of TIME is likely to relate to the behaviour of birds at differing times of the day e.g. birds perched during cooler periods (during morning) and soaring in warmer periods (from late morning onwards). The level and type of activity of birds will have an obvious impact on detection distance. Daily variation in temperature and weather conditions will make this a complicated relationship which is unlikely to be detected by these analyses.

Transects running through dense woodland may result in undetected responsive movements of birds which may, in turn, lead to incorrect distance measurement or incomplete detection at $g(0)$. In such habitats Buzzards were almost always heard to call prior to, or immediately after, taking flight when disturbed by a fieldworker. Use of such cues to identify original locations for measurement should have reduced the number of undetected responsive movements along transects performed in these habitats.

The use of public rights of way and roads for this study will have resulted in some sections of transect necessarily following linear landscape features such as hedgerows, fences and runs of power lines and poles. These features can influence the distribution of raptors such as Buzzards through their impact on the abundance of preferred prey items (e.g. Adams & Geiss 1983, Meunier *et al.* 2000) or carrion (Lambertucci *et al.* 2009, Lees *et al.* 2013), the ways in which they can improve hunting efficiency (e.g. Beckmann & Shine 2011) or how they permit the adoption of less energy-demanding hunting strategies (Meunier *et al.* 2000). Failure to place transects randomly across a study area (e.g. by following linear landscape features) can lead to biases arising from the association and preferences for certain habitats or landscape features. This will effectively remove the validity of extrapolating sample statistics to the population of interest (Buckland *et al.* 2001). Despite this, and the potential effects listed above, we found no evidence for the attraction of Buzzards to roads in our data. Whether the inclusion of roadside transect data has a significant influence on the derived density estimates is open to question.

Although we adopted a number of strategies to reduce double counting, the duration of each transect (mean - 371 minutes) means that there was ample opportunity for birds to move across the study area. This is likely to have resulted in the double-counting of a small number of individuals and possible positive bias to our estimates. Similarly, the inclusion of flying birds may also have affected our results. Buckland *et al.* (2001) suggest that independent movement of birds can be accommodated they must, 'on average', be moving at less than half the speed of the observer if they are not to introduce a positive bias to the results. 24% of the observations here were of birds engaged in purposeful, directional flight. Since neither the destination of flying birds nor their duration of flight was recorded here, it is not possible to determine whether the average speed of these individuals was less than half that of the observer. Whether the inclusion of these observations has resulted in a significant positive bias to estimates is open to question. Nevertheless, inclusion of some assessment of the nature and distances of flight behaviour in future surveys would enable greater discrimination of data and exploration of impacts on derived estimates.

The population trends derived here closely follow those obtained for the same period by BBS. This suggests that the potential issues often associated with broad-scale, multi-species, volunteer surveys (e.g. the dilution of effort between target species and differential abilities of volunteers) have little effect on results. This may not be the case for density estimates though, since the use of a small number of distance bands (in BBS), rather than accurate distance measurement, has been shown to over-estimate density (Quesada *et al*. 2010). The extent of any difference cannot be assessed here since there are no published BBS-derived population estimates for this species in this region for the period covered by our study.

The methods employed here provide a reasonably straightforward means of assessing the absolute population size of an abundant, conspicuous, raptor species across the UK landscape. However, this methodology is unlikely to be suitable for more secretive (e.g. Sparrowhawk, A*ccipiter nisus*) or scarcer species. The methods used here are applicable across most landscape types and could provide a useful means of population monitoring stratified by habitat and area. The broader application of such methods is perhaps limited by the cost of equipment (laser rangefinder and gps); however, rapid technological advances and falling costs are likely to remove such obstacles in the near future. Individual variation in skill levels, abilities to detect birds in the landscape and the need to train individuals in survey methodology may also pose certain problems; however, these are challenges faced by all survey protocols.

The recovery of raptor populations is often accompanied by concerns relating to potential impacts on conservation (e.g. of prey species or competitors; e.g. Moleón *et al.* 2011), sociology (e.g. Burke *et al.* 2015) or economy (e.g. of game populations; e.g. Parrot 2015). Indeed, the recovery of Buzzard populations has been followed by increasing pressure for population control measures to protect game stocks (Lees *et al.* 2012). Although Buzzards are protected under UK law (Wildlife & Countryside Act 1981), provision exists to issue licences to kill individuals to prevent agricultural damage (including 'damage to livestock'). Licences are issued only after careful consideration of a number of factors, including local abundance. Without accurate population data, such assessments will be affected by subjective perceptions of abundance. Producing estimates of actual population size for this species is therefore timely, and will prove useful in assisting decision-makers in assessing the potential impacts of any licensed action.

Conclusion

Our results show how the population size of a previously persecuted species of raptor in central southern England has increased by more than 50% over a five year period, and how the previously high rate of population growth appears to be stalling. The next phase of this study will focus on producing density estimates using this methodology across a larger area. This approach will enable comparison of the predicted population estimates for the SE region produced here with those utilising fieldwork undertaken across all parts of the region and a direct comparison with BBS estimates. Further assessment of the utility of this method and the viability of using volunteers to derive estimates across a broader geographical scale will also be possible.

Acknowledgements

This work was funded by the visitors and supporters of the Hawk Conservancy Trust. The John Spedan Lewis Foundation assisted with funding for the purchase of essential equipment. Two anonymous reviewers provided helpful comments on earlier drafts of the manuscript.

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Tables

Table 1. Covariates used in modelling distance sampling estimates of Common Buzzards *Buteo buteo* in central southern England.

Covariate	Description	Levels
OBS	Fieldworker	Factor - MS or RH
LOC	Situation of the bird	Factor - Ground, Perched, Flying
STRATA	Section of study area	Factor - East or West
TIME	Minutes after sunrise	Continuous
WDS	Area (m^2) of woodland within 250m radius of observer location	Continuous

Table 2. Survey effort and numbers of observations made during surveys of Common Buzzards *Buteo buteo* in central southern England.

Transect Period	Season Year	Number of transects	Σ Transect lengths (km)	Mean transect duration (min)	Observations	Observations after Truncation (550m)
1	Autumn 2011	40	594.7	320	489	475
2	Spring 2012	35	531.2	343	611	599
3	Autumn 2012	26	379.2	345	359	336
4	Spring 2013	25	382.2	339	471	445
5	Autumn 2013	39	570.3	388	590	565
6	Spring 2014	35	523.7	404	830	814
7	Autumn 2014	25	373.4	363	409	396
8	Spring 2015	22	361.6	414	437	431
9	Autumn 2015	25	393.3	411	469	467
10	Spring 2016	24	379.9	384	509	502
TOTAL		299	4489.5	3711	5174	5030

Table 3. Details of model selection statistics for distance sampling estimates of the Common Buzzard *Buteo buteo* population in central southern England. Preferred models are indicated by bold type. (Period – survey period number, season and year; Engine – analysis engine; Key – key function, U-Uniform, HN – Half normal, HZ – Hazard rate; Adjustment Term – series expansion type (number of terms), Cos – Cosine, HP – Hermite polynomial, SP – Simple polynomial; Covariates – included in the model; Parameters – total number of parameters in the model; ΔAIC – difference in Akaike Information Criterion between model with best fit and the current model; CvM(p) - cosine-weighted Cramér-von Mises Goodness of fit test value and (P value))

Period	Engine	Key	Adjustment Term	Covariates	Parameters	ΔAIC	CvM (*p*)
(1)AUT 2011	CDS	HN	Cos(1)	-	2	11.52	-
		HZ	HP(3)	-	5	11.72	-
	MCDS	**HN**	**0**	**STRATA TIME WDS**	**4**	**0.00**	**0.277 (0.1)**
		HN	SP(1)	TIME WDS	6	1.74	0.139 (0.3)
		HN	Cos(1)	TIME WDS	4	1.81	0.121 (0.4)
		HN	Cos(1)	WDS	3	3.11	0.123 (0.3)
(2)SPR2012	CDS	HN	0	-	1	1538.46	0.011 (1.0)
		HZ	SP(1)	-	3	1541.85	0.012 (1.0)
	MCDS	**HN**	**0**	**STRATA TIME WDS**	**4**	**0.00**	**0.038 (0.9)**
		HN	Cos(1)	STRATA TIME WDS	5	1.75	0.021 (1.0)
		HN	0	TIME WDS	3	8.99	0.036 (0.9)
		HN	0	TIME	2	26.07	0.015 (1.0)
(3)AUT2012	CDS	HN	0	-	1	13.27	0.117 (0.4)
		HZ	0	-	2	17.45	0.124 (0.3)
	MCDS	HN	0	STRATA TIME WDS	6	0.00	0.109 (0.4)
		HN	**0**	**WDS**	**4**	**0.77**	**0.115 (0.4)**
		HN	0	TIME	2	13.88	0.114 (0.4)
		HN	0	LOC	3	15.95	0.114(0.4)
(4)SPR2013	CDS	HN	Cos(1)	-	2	18.38	0.039 (0.9)
		HZ	Cos(1)	-	3	19.24	0.038 (0.9)
	MCDS	**HN**	**Cos(1)**	**OBS TIME**	**4**	**0.00**	**0.032 (0.9)**
		HN	SP(1)	OBS TIME	4	0.11	0.070 (0.6)
		HN	Cos(1)	OBS TIME WDS	5	1.82	0.031 (0.9)
		HN	Cos(1)	TIME	3	8.10	0.030 (0.9)
(5)AUT2013	CDS	HN	0	-	1	46.97	0.286 (0.1)
		U	HP(1)	-	1	49.05	0.647(0.01)
	MCDS	HN	0	OBS STRATA TIME WDS	5	0.00	0.252(0.1)
		HN	**0**	**OBS TIME WDS**	**4**	**0.59**	**0.235(0.15)**

		HN	0	TIME WDS	3	1.20	0.233(0.15)
(6)SPR2014	CDS	HN	0	-	1	12.45	0.106 (0.4)
		HZ	Cos(1)	-	3	14.42	0.076 (0.6)
	MCDS	**HN**	**SP(1)**	**STRATA WDS**	**4**	**0.00**	**0.300 (0.1)**
		HN	SP(1)	STRATA TIME WDS	5	1.87	0.270 (0.1)
		HN	SP(1)	WDS	3	3.42	0.191(0.15)
		HN	0	WDS	2	3.94	0.084 (0.5)
(7)AUT2014	CDS	HN	Cos(3)	-	4	0.00	0.094 (0.5)
		U	**Cos(5)**	**-**	**5**	**0.94**	**0.074 (0.6)**
		HN	Cos(2)	-	3	11.91	0.199(0.15)
	MCDS	HN	Cos(1)	WDS	3	9.02	0.34 (0.05)
		HN	Cos(1)	WDD	5	10.99	0.314 (0.1)
(8)SPR2015	CDS	HN	Cos(2)	-	3	110.17	0.022 (1.0)
		HZ	SP(2)	-	4	111.16	0.020 (1.0)
	MCDS	**HZ**	**0**	**LOC TIME WDS**	**7**	**0.00**	**0.069 (0.6)**
		HN	Cos(2)	TIME WDS	5	1.21	0.022 (1.0)
		HZ	SP(1)	WDS	6	6.94	0.032 (0.9)
		HZ	SP(1)	TIME	4	9.63	0.054 (0.7)
(9)AUT2015	CDS	HN	SP(1)	-	2	10.71	0.113 (0.4)
		HZ	HP(1)	-	3	12.92	0.101 (0.4)
	MCDS	**HN**	**SP(1)**	**WDS**	**3**	**0.00**	**0.093 (0.4)**
		HN	SP(1)	STRATA WDS	4	0.82	0.091 (0.5)
		HN	SP(1)	TIME WDS	4	1.56	0.094 (0.5)
		HN	0	WDS	2	1.59	0.093 (0.5)
(10)SPR2016	CDS	HN	SP(1)	-	2	5.02	0.151 (0.3)
		HZ	SP(1)	-	3	5.26	0.063 (0.6)
	MCDS	**HN**	**0**	**WDS**	**4**	**0.00**	**0.087 (0.5)**
		HN	0	TIME WDS	5	1.37	0.079 (0.5)
		HN	0	STRAT WDS	5	1.78	0.088 (0.5)
		HN	0	LOC TIME WDS	7	3.97	0.077 (0.6)

Table 4. Estimates of density and population size of Common Buzzards *Buteo buteo* with Lower (LCL) and Upper (UCL) 95% confidence intervals. %CV - coefficient of variation, df - degrees of freedom.

Year	Period	LCL – Density – UCL (Individuals km^{-2})	LCL – No. individuals - UCL	% CV	df
2011	Autumn	0.936 - 1.111 - 1.319	2428 - 2883 - 3423	8.62	65.4
2012	Spring	0.836 - 1.126 - 1.517	2169 - 2922 - 3936	14.81	39.2
	Autumn	0.990 - 1.274 - 1.639	2568 - 3305 - 4254	12.48	35.31
2013	Spring	1.327 - 1.614 - 1.963	3444 - 4187 - 5093	9.66	34.58
	Autumn	1.172 - 1.393 - 1.654	3043 - 3613 - 4292	8.58	50.10
2014	Spring	1.458 - 1.734 - 2.064	3782 - 4500 - 5354	8.66	47.94
	Autumn	1.333 - 1.695 - 2.156	3458 - 4399 - 5595	11.86	32.84
2015	Spring	1.176 - 1.746 - 2.593	3051 - 4531 - 6729	19.70	36.58
	Autumn	1.342 - 1.705 - 2.164	3483 - 4423 - 5616	11.77	32.45
2016	Spring	1.445 - 1.728 - 2.068	3749 - 4485 - 5365	8.82	32.15

Legends to Figures

Figure 1. Study area in central southern England showing randomised locations of the Idealised Transect Routes (ITRs) for the first survey of Common Buzzards *Buteo buteo* in autumn 2011. Shading represents urban areas.

Figure 2. Estimates of Common Buzzard *Buteo buteo* population size within the study area in central south England for each survey period between autumn 2011 and spring 2016 (± 95% confidence intervals).

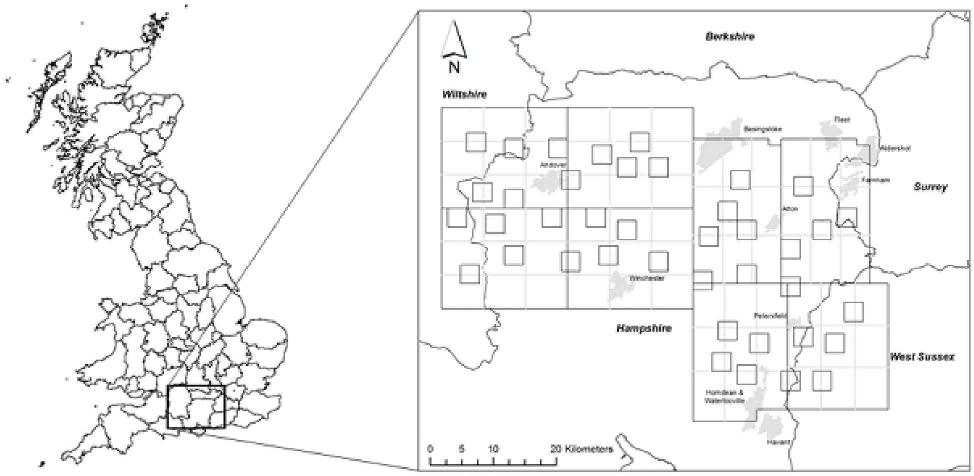

Figure 1. Study area in central southern England showing randomised locations of the Idealised Transect Routes (ITRs) for the first survey of Common Buzzards Buteo buteo in autumn 2011. Shading represents urban areas.

164x83mm (96 x 96 DPI)

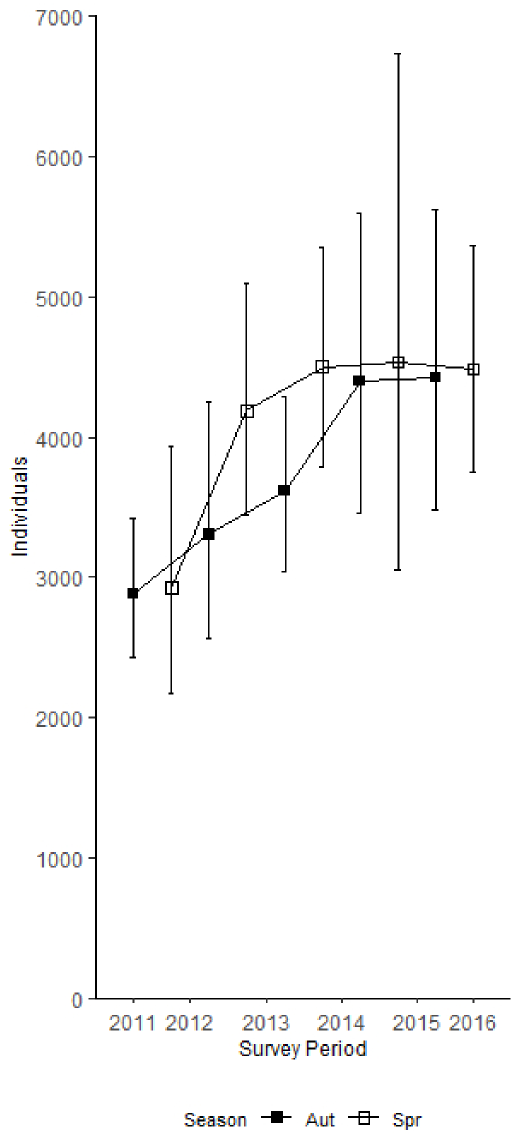

Figure 2. Estimates of Common Buzzard Buteo buteo population size within the study area in central south England for each survey period between autumn 2011 and spring 2016 (± 95% confidence intervals).

81x185mm (96 x 96 DPI)