

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

Article

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- 10
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- 15

16 Abstract

- 17 Capsule: Distance sampling identified an increase in estimated population size of Common Buzzards
- 18 Buteo buteo in central southern England between 2011-16 of more than 50%. The rate of
- 19 population growth slowed in later years.
- 20 Aims: To assess the utility of a targeted distance sampling protocol to derive seasonal and annual
- 21 population estimates for Common Buzzards across an area of southern England.
- 22 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
- 24 2011and 2016 across a 2600km² area of central southern England.
- 25 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.
- 27 Discussion: A slowing of the growth in population size of Common Buzzards in central southern
- 28 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
- 30 general population trends for this species. Our data provide the first published estimates of the
- 31 Common Buzzard population in central southern England derived from direct empirical assessment.
- 32
- 33

35 Introduction

36 Identifying the population size of a species is a key step in developing and implementing an effective 37 conservation strategy (Soulé 1987, Frankham 1995). Unfortunately, determining population size can 38 be difficult and expensive. Difficulties can arise due to particular behaviours of the study species 39 (e.g. mobility, shyness, crypsis, nocturnality), scarcity or its occupancy of habitats where access or 40 movement is difficult etc. (Anderson et al. 2015). Also, the spatial scale required to produce 41 meaningful estimates influences survey effort, the level of sampling and cost. To improve efficiency 42 in data collection many broad-scale studies use multi-species surveys or monitoring programmes 43 utilising volunteer fieldworkers (see e.g. Riseley et al 2008, Jiguet 2009). There are consequences to 44 these approaches, however, and analyses must consider the effects of surveying for more than one 45 species at a time (e.g. reduced effort per species, heterogeneity in species detectability (Johnston et 46 al. 2014) and potential difficulties generating sufficient observations of rarer species (but see 47 Sanderlin et al. 2014)) and the variation in skills and intrinsic differences between volunteer 48 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). 49 50 51 Species abundance and density estimates are now often generated using a distance sampling 52 methodology. This technique uses the recorded distances of objects of interest to randomly-placed 53 survey routes or points to estimate animal density or abundance (Buckland et al. 2001). A key 54 assumption of this method is that all objects at zero distance (q(0)) are detected and that probability 55 of detection decreases with increasing distance from the route or point. Accurate measurement of 56 these distances is also assumed, however, it is often necessary to simplify survey methodologies 57 (e.g. by using a small number of distance bands rather than accurate measurement) to encourage 58 participation and adherence to protocols (Newson et al. 2008, Quesada et al. 2010). There is a 59 trade-off between the quality of the estimate and simplicity of the method (Rabe et al. 2002) and

60 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).

63

64 Estimates of population size for bird species in the UK tend to focus on breeding populations. These 65 estimates are usually derived from indices of relative abundance generated as part of the Breeding 66 Bird Survey (BBS) (e.g. Newson et al. 2008, Riseley et al. 2008, Musgrove et al. 2013). Although 67 useful for many species, the use of the breeding pair as the unit of interest is less appropriate for 68 certain species and will underestimate population size (Newson et al. 2008). This is particularly true 69 for many raptor species where individuals do not breed until into their third year (Davis & Davis 70 1992) and where a significant proportion of the adult population is not breeding in any one year 71 (Newton 1979, Hunt 1998, Kenward et al. 1999, 2001), as is the case with the Common Buzzard 72 (Buteo buteo, hereafter 'Buzzard'). Accurate estimation of population size is therefore necessary if 73 the aim of monitoring is to provide an objective assessment of population trends – particularly 74 where species may be increasing or decreasing. Using methodologies suitable for certain species 75 groups to produce population estimates may then provide a means of periodically validating or 76 calibrating indices that are applied more widely.

77

78 The Buzzard was lost from many parts of its range in Britain due to the combined effects of 79 widespread persecution in the 19th century, a crash in preferred prey populations (Rabbit, 80 Oryctolagus cuniculus) in the 1950s and the effects of organochlorine pesticides in the 1960s and 81 1970s (Sim et al. 2000). Until the 1980s Buzzards in Britain were confined to Scotland, Wales and 82 Western England. Since then, the enactment of improved wildlife conservation legislation (e.g. 83 banning of organochlorine pesticides and comprehensive legal protection) and increasing public 84 awareness have led to a significant increase in the species' population size and range. Most recent 85 assessments indicate that the species has now recolonised many of the areas of the UK from which it 86 had been lost (Clements 2002, Musgrove et al. 2013).

87	
88	The primary objective of this study was to use a distance sampling methodology to produce local
89	and regional population estimates of Buzzards in central Southern England. We also draw
90	comparisons with population estimates derived using other methodologies and discuss the utility of
91	our approach for determining population sizes of Buzzards and other conspicuous diurnal raptor
92	species on a larger scale.
93	
94	
95	Methods
96	Study area and fieldwork
97	The study was conducted between September 2011 and June 2016 across two areas (designated
98	'East' and 'West') covering ca. 2600km ² of central southern England in Hampshire, Wiltshire, Surrey
99	and West Sussex (centred on 1° 18'W and 51° 13'N – see Fig. 1). Land use within the study area is
100	primarily mixed farming (arable and grassland) with scattered small woodlands; although the extent
101	of woodland is higher in the East (26687ha) than West (17634ha). The study area contains ten urban
102	areas of which five have human populations exceeding 40000 (Nomis 2016).
103	
104	We used a line transect combined with distance sampling methodology (Buckland et al. 2001, 2004)
105	to determine the population size and density of Buzzards. Each transect was a circuit based on a
106	square with each side measuring 3km. Even coverage of the study area was achieved by dividing the
107	East and West sections into 24 smaller blocks and using a random number generator to identify a
108	grid reference and start point for transects within each of these blocks. An idealised transect route
109	(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
- 111 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.

119

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.

127

128 All birds observed during the walked transects were recorded. When groups of birds were 129 encountered, the number of individuals was noted. For each observation, observer location was 130 recorded using Garmin 60 Csx GPS units and horizontal distance and bearing to each bird (or to the 131 centre point of groups; Buckland et al. 2001) from the observer using Swarovski Laser Guide 8x30 or 132 Nikon Forester 550 laser rangefinders and Silva compasses. Where a bird in flight had obviously 133 been disturbed from a perch by the fieldworker just prior to detection, measurement of distance 134 was taken from the fieldworker location to the original perch. Bird behaviour, situation (i.e. flying, 135 perched or on the ground), habitat, time of observation and weather conditions were also recorded 136 for each observation. To minimise double-counting, fieldworkers noted, where possible, the 137 plumage morph of all birds (following Glutz von Blotzheim et al. (1971)), specific aspects of plumage 138 (e.g. bright tail, prominent breast patches etc.) and location of obvious moult (in remiges or

Bird Study/Ringing & Migration

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%.

145

146 Where the ability of a fieldworker to detect birds was compromised by visibility from the transect 147 route (e.g. obstruction by surrounding vegetation), the observer moved a short distance away from 148 the transect to obtain a clearer view before returning and continuing along the route. Although the 149 fieldworker followed a map of the transect route it is unlikely that they will have been standing 150 exactly on the transect line (i.e. at g(0) – zero distance from the line) at the time of making any 151 observation. The perpendicular distance of the fieldworker from the transect route at the time of 152 each observation was determined using the GPS locations viewed in GIS. These 'offsets' were then 153 used to correct the calculated distances of the observations to the transect line through either 154 addition or subtraction of the offset (depending on the relative positions of the observer and bird to 155 the transect route). All GPS locations obtained using WGS-84 were transformed to British National 156 Grid using the Ordnance Survey 'OSTN02' transformation in ArcGIS (ESRI 2011).

157

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 (5.8km²) from the total size of the study area in all subsequent distance analyses.

166 To investigate the possible role of roads and roadside areas in attracting birds, we also compared the 167 distribution of distances of 5000 randomly-generated points with that of our observations. Points 168 were generated using the random number generator runif() function in R (R Core Team 2016) to 169 produce pairs of latitude and longitude. These points were then plotted and their distances from 170 the nearest road determined using the Near function in ArcGIS. The distributions of these 'distance-171 to-roads' measurements were compared using two-sample Kolmogorov-Smirnov tests. 172 173 174 **Density estimation** 175 Population size and density estimates were derived using both the Conventional Distance Sampling 176 (CDS) and Multiple Covariates Distance Sampling (MCDS) engines within program Distance 7.0 177 (Thomas et al. 2010). Distance analysis here follows the guidelines provided for that software and in 178 the associated literature by Buckland et al. (2001, 2004). 179

180 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 181 182 (Burnham 1981, Thompson 2002, Diefenbach et al. 2003). A two-level factor covariate (OBS) was 183 included to account for the likely variation in ability of fieldworkers to detect birds. Area of 184 woodland at the point of observation (WDS) is also likely to impact detection distance due to an 185 inverse relationship with range of view (i.e. the maximum range of vision). Values for this covariate 186 were determined from the CEH Land Cover Map 2007 (Morton et al. 2014) by measuring the area of 187 woodland within a 250m radius of the point of observation using ArcGIS 10.2 (ESRI 2011). STRATA 188 was included as a covariate in order to account for potential differences in topography or habitat 189 quality between the two sections of the study area, since this may result in differential detection 190 distances. Bird activity and behaviour, and thus detectability, will vary throughout the day (e.g.

191 Kendall 2014, Öberg et al. 2015). Here, TIME was defined as the number of minutes after sunrise for 192 each observation. Lastly, the situation of the bird, i.e. whether on the ground, perched or in flight, 193 was included as a factorial covariate, LOC. The inclusion of flying birds in distance sampling can 194 present a number of problems, primarily due to violation of the assumption of uniform distances 195 through responsive movement, and double-counting (see Fewster et al. 2008, Anderson et al. 2015). 196 Where this occurs, estimates will tend to be overestimated (Buckland et al. 2001). Although 197 exclusion of flying birds from analyses is possible, this approach is best used for species in which only 198 a small proportion of the population will be in flight at any one time (Buckland et al. 2008). This is 199 clearly not the case for many soaring raptor species and exclusion of such data was not appropriate. 200 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. 201 202 2015). 203 204 Relationships between covariates and the ability to detect birds were explored prior to modelling, 205 although failure to detect any effect here did not prevent inclusion in model assessments. Factor 206 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 207 208 was performed using Pearson's r and regression.

209

210 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

214 Kolmogorov-Smirnov tests. Data were truncated at 550m to remove a lengthy tail and all models

215 incorporating adjustment terms were scaled by the truncation distance.

217	Overloading of the MCDS engine with covariates is more likely to result in failure of the algorithm to
218	converge (Thomas et al. 2010). To counter this, we follow the guidelines of Thomas et al. (2010)
219	who advocate the forward stepwise addition of individual covariates, retaining those which
220	contribute to reducing Akaike's Information Criteria (AIC). AIC was used to select between models
221	(Burnham & Anderson 2002).
222	
223	All statistical analyses, other than distance sampling, were performed using R version 3.3.1.
224	
225	
226	Results
227	4490km of surveys were completed during the 10 transect periods (Table 2). Coverage was higher in
228	the eastern section of the study area with 2295km of surveys walked on 151 transects compared
229	with 2194km on 145 transects in the western section. The average duration of each transect was
230	371 minutes (365 in Spring vs 377 in Autumn).
231	
232	4274 observations of 5174 individuals were made during the study. Birds were seen in groups of up
233	to 32 individuals, however, 85% of observations were of single birds (mean group size = 1.2 ± 0.75).
234	63% of observations were of birds in flight (cf. 37% perched or on the ground). Of these, 62% were
235	birds which were soaring, hovering or interacting with other species, rather than in obvious
236	directional transit movements.
237	
238	There was no difference between the distributions of number of observations and the number of
239	transects walked (and therefore, transect length) for each season (autumn X^2 (4) = 3.08, p = 0.55;
240	spring (X^2 (4) = 1.75, p = 0.78) indicating that more observations were made when more transects
241	were walked. Significantly more individuals were seen during spring surveys than in autumn (X^2 (3) =
242	160.25, $p < 0.001$) despite the total length of surveys undertaken being similar.

243	
244	Histograms of the distribution of perpendicular distances indicated detection on and close to the
245	transect line remained at or near 100% in all survey periods. Median detection distance across all
246	data was 178m. Truncation of data above 550m resulted in the loss of 2.8% of observations (99
247	observations of 143 individuals) but left more than 330 observations per period; comfortably above
248	the threshold of 60-70 generally recommended for modelling using Distance (Buckland et al. 2001).
249	Sufficient data were available to enable the modelling of separate detection functions, and the
250	inclusion of different covariates, for each period.
251	
252	55% of the total length of transects was walked along roads. 28% of all observations involved
253	Buzzards within 100m of any road and only 11% were of birds within 100m of the same road as that
254	from which the observation was made. There was no indication of a bias in observation of birds
255	near to roads when comparing the distribution of distances with that of 5000 random locations
256	(two-sample Kolmogorov-Smirnov, D- <0.001, p = 0.99; Buzzard median – 264m, Random median –
257	158m). 36% of the random locations were within 100m of a road compared with 21% of Buzzard
258	locations
259	
260	
261	Exploratory analyses of covariates
262	The distance at which birds were detected reduced as the extent of woodland at the point of
263	observation increased. This effect was negative across the entire dataset (r = -0.166, t_{4157} = -10.8, p
264	< 0.001) and in all survey periods (p < 0.002) except spring 2013 (r = -0.079, t_{363} = -1.51, p = 0.13).
265	
266	None of the remaining covariates showed any consistent relationship with detection distance. Mean
267	detection distances were similar between both sections of the study area and varied significantly
268	only in 2012 (spring, $t_{474.13}$ = -3.5, p <0.001, mean East – 178.9m, West – 207.7m; autumn, $t_{255.2}$ = -

269 2.06, p = 0.04, mean East – 179.2m, West – 214.2m). The situation of birds (i.e. whether on the 270 ground, perched or in flight) had no significant influence on detection distance (ANOVA $F_{2,4156} = 2.42$, 271 p = 0.09, Tukey test, p > 0.15). Observer effects on detection distances were identified in one of the 272 four survey periods where data were collected by more than one fieldworker (spring 2013, $t_{139} =$ 273 3.67, p < 0.001, mean MS – 208m, RH – 148m). Although timing of an observation had a bearing on 274 detection distance in two periods (autumn 2012, $F_{1,402} = 4.91$, p = 0.027; and spring 2015, $F_{1,307} =$ 275 5.99, p = 0.015), there was no significant effect during the other eight periods.

276

277

278 Model fitting

279 MCDS models having reasonable fit (i.e. with Cramér von Mises and Kolmogorov-Smirnov tests p >280 0.3) were developed for all periods (Table 3) except periods 5-7. Although statistics assessing model 281 fit for these periods produced p > 0.1, their detection functions and quantile-quantile plots indicated 282 that more birds than expected were observed close to the transect route. As model fit was 283 reasonable in these periods, we still present the outputs from these but emphasise their being on 284 the margins of acceptability. CDS models were preferred in period 7 (p > 0.5), however, the model 285 with lowest AIC (Half-normal + three cosine adjustments) showed signs of over-fitting and issues in 286 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 287 288 appeared in the majority of preferred models for each survey period.

289

290 **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

294	rate of increase across the four-and-a-half years of the study was 12.5% but this slowed in successive
295	years (from 15% to 1% in autumn and 43% to -1% in spring; Figure 2).
296	
297	More birds were seen in spring surveys (1.31km ⁻¹) than preceding autumn periods (1.0 bird km ⁻¹)
298	even though total lengths of transects walked were shorter in spring for all periods except autumn
299	2012-spring 2013. Estimated density was also consistently higher for surveys performed during the
300	spring (Table 4; means: spring – 1.59 autumn – 1.44).
301	
302	
303	Discussion
304	
305	Population density estimates
306	We used a distance sampling-based methodology to estimate the population density of a
307	conspicuous diurnal raptor species within an area of central southern England. These estimates
308	indicate that the Buzzard population increased by more than 50% over the course of the study
309	(Figure 2). In contrast, Buzzard populations in the adjacent SW region have shown a comparatively
310	modest rate of increase since 1995 (+13% - Harris <i>et al</i> . 2017). Differences in the rate of population
311	change between these two regions may be a function of there being a higher number of available
312	potential territories in regions neighbouring the SW population and the consequent dispersal of
313	individuals from higher to lower density areas (Walls & Kenward 1998).
314	
315	The reduced rate of population growth during the last three survey periods mirrors estimates
316	derived from BBS data for the SE region (<u>www.bto.org/bbs</u>) which, although showing an overall
317	increase of 1104% since 1995, indicate a slowing of population growth to a point of a slight decline (-
318	2%) between 2016-17. The reasons for this are unclear, especially since rates of breeding success for
319	Buzzards have increased across the UK during these years (Woodward et al. 2018) – although

320 regional differences will be masked in these national estimates. Nevertheless, a number of factors 321 may be operating to limit population growth, including: the ongoing impacts of viral haemorrhagic 322 disease (VHD) on UK rabbit populations (Harris et al. 2019), the abundance of which has been shown 323 to influence breeding productivity and population increase in Buzzards in the UK (Graham et al. 1995; 324 Swann & Etheridge 1995); the continued impacts of secondary poisoning by rodenticides (e.g. 325 Christensen et al. 2012) and ingestion of lead in spent ammunition (Pain et al. 2009); and a potential 326 increase in persecution in response to the perceived predation pressures on game bird populations 327 from increasing Buzzard abundance.

328

329 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 330 331 they will have been recolonised later and will be further from reaching carrying capacity; a situation 332 highlighted by Harris et al. (2014). In addition, now that the scale of human-induced population 333 constraints appears to have substantially reduced, carrying capacities are likely to have increased 334 and be governed mostly by the availability and suitability of food and breeding habitat. In southern 335 England, there is likely to be a proportionally greater extent of suitable breeding habitat in the SE 336 region compared to the historical strongholds in the SW since the landscape is more heavily wooded 337 (Forestry Commission 2016). As a result, continued population growth in this region is likely for the 338 foreseeable future.

339

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

346 make a nesting attempt each year, either due to immaturity, lack of status and inability to find a 347 mate or hold a territory (Davis & Davis 1992, Kenward et al. 2000). Using the estimate suggested by 348 Kenward et al. (2000) of only one in four individuals breeding each year, results in an estimate of 349 between 3440-4125 pairs in SE England. This represents a breeding density of 18-22 pairs per 350 100km², similar to that found by Sim et al. (2001) in one of their West Midlands study areas. This is 351 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 352 353 habitat in central Europe (Melde 1956, Thiollay 1967). Since the coarse regional population estimate 354 presented here is an extrapolation from our derived estimates, any variation in the quality of those 355 landscape characteristics representing suitability for Buzzards (e.g. food and prey density, 356 disturbance, persecution, habitat structure and mosaic etc.) will influence its validity. 357 358 Alongside the estimates of overall abundance within the study area, our study provides an 359 interesting comparison of the apparent abundance of Buzzards between autumn and spring periods. 360 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 361 362 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 363 364 Buzzards often exploit the easy foraging for invertebrates provided by ploughed fields (Dare 1957). 365 As a result, there is unlikely to have been any significant loss of first year birds from the study area in 366 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 367 368 compound this expected difference in seasonal abundance. That this is not the case may reflect 369 more on seasonal variation in bird behaviour, and its influence on detectability, than demographics. 370 Increased time spent soaring and in display behaviours in spring resulted in improved detectability 371 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).

374

375 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).

382

383 The covariates most frequently included within preferred models (WDS, STRATA and TIME) indicate 384 that woodland cover was the most important factor affecting Buzzard detectability. Increasing 385 density of woodland reduces the view of surrounding habitats leading to birds generally being 386 detected at shorter distances than in more open habitats. This effect is also likely to account for the 387 inclusion of STRATA in many preferred models since a greater proportion of the landscape area was 388 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 389 390 birds at differing times of the day e.g. birds perched during cooler periods (during morning) and 391 soaring in warmer periods (from late morning onwards). The level and type of activity of birds will 392 have an obvious impact on detection distance. Daily variation in temperature and weather 393 conditions will make this a complicated relationship which is unlikely to be detected by these 394 analyses.

395

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.

402

403 The use of public rights of way and roads for this study will have resulted in some sections of 404 transect necessarily following linear landscape features such as hedgerows, fences and runs of 405 power lines and poles. These features can influence the distribution of raptors such as Buzzards 406 through their impact on the abundance of preferred prey items (e.g. Adams & Geiss 1983, Meunier 407 et al. 2000) or carrion (Lambertucci et al. 2009, Lees et al. 2013), the ways in which they can improve 408 hunting efficiency (e.g. Beckmann & Shine 2011) or how they permit the adoption of less energy-409 demanding hunting strategies (Meunier et al. 2000). Failure to place transects randomly across a 410 study area (e.g. by following linear landscape features) can lead to biases arising from the 411 association and preferences for certain habitats or landscape features. This will effectively remove 412 the validity of extrapolating sample statistics to the population of interest (Buckland et al. 2001). 413 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 414 415 influence on the derived density estimates is open to question.

416

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.

431

432 The population trends derived here closely follow those obtained for the same period by BBS. This 433 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 434 435 effect on results. This may not be the case for density estimates though, since the use of a small 436 number of distance bands (in BBS), rather than accurate distance measurement, has been shown to 437 over-estimate density (Quesada et al. 2010). The extent of any difference cannot be assessed here 438 since there are no published BBS-derived population estimates for this species in this region for the 439 period covered by our study.

440

441 The methods employed here provide a reasonably straightforward means of assessing the absolute 442 population size of an abundant, conspicuous, raptor species across the UK landscape. However, this 443 methodology is unlikely to be suitable for more secretive (e.g. Sparrowhawk, Accipiter nisus) or 444 scarcer species. The methods used here are applicable across most landscape types and could 445 provide a useful means of population monitoring stratified by habitat and area. The broader 446 application of such methods is perhaps limited by the cost of equipment (laser rangefinder and gps); 447 however, rapid technological advances and falling costs are likely to remove such obstacles in the 448 near future. Individual variation in skill levels, abilities to detect birds in the landscape and the need 449 to train individuals in survey methodology may also pose certain problems; however, these are 450 challenges faced by all survey protocols.

451

452 The recovery of raptor populations is often accompanied by concerns relating to potential impacts 453 on conservation (e.g. of prey species or competitors; e.g. Moleón et al. 2011), sociology (e.g. Burke 454 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 455 456 protect game stocks (Lees et al. 2012). Although Buzzards are protected under UK law (Wildlife & 457 Countryside Act 1981), provision exists to issue licences to kill individuals to prevent agricultural 458 damage (including 'damage to livestock'). Licences are issued only after careful consideration of a 459 number of factors, including local abundance. Without accurate population data, such assessments 460 will be affected by subjective perceptions of abundance. Producing estimates of actual population 461 size for this species is therefore timely, and will prove useful in assisting decision-makers in assessing 462 the potential impacts of any licensed action. SELIC

463

464

465 Conclusion

466 Our results show how the population size of a previously persecuted species of raptor in central 467 southern England has increased by more than 50% over a five year period, and how the previously 468 high rate of population growth appears to be stalling. The next phase of this study will focus on 469 producing density estimates using this methodology across a larger area. This approach will enable 470 comparison of the predicted population estimates for the SE region produced here with those 471 utilising fieldwork undertaken across all parts of the region and a direct comparison with BBS 472 estimates. Further assessment of the utility of this method and the viability of using volunteers to 473 derive estimates across a broader geographical scale will also be possible.

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to per period

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Review

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724 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, UUniform, 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

740 Cramér-von Mises Goodness of fit test value and (P value))

Period	Engine	Кеу	Adjustment Term	Covariates	Parameters	ΔΑΙϹ	CvM (p)
(1)AUT 2011	CDS	HN	Cos(1)	-	2	11.52	-
		ΗZ	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)
		ΗN	Cos(1)	WDS	3	3.11	0.123 (0.3)
(2)SPR2012	CDS	HN	0	-	1	1538.46	0.011 (1.0)
		ΗZ	SP(1)	-	3	1541.85	0.012 (1.0)
	MCDS	HN	0	STRATA TIME WDS	4	0.00	0.038 (0.9)
		ΗN	Cos(1)	STRATA TIME WDS	5	1.75	0.021 (1.0)
		ΗN	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)
		ΗZ	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)
		ΗZ	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	ΗN	0	OBS STRATA TIME	5	0.00	0.252(0.1)
		HN	0	OBS TIME WDS	4	0.59	0.235(0.15)

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		HN	0	TIME WDS	3	1.20	0.233(0.15)
(6)SPR2014	CDS	HN	0	-	1	12.45	0.106 (0.4)
		ΗZ	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)
		ΗN	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	ΗZ	0	LOC TIME WDS	7	0.00	0.069 (0.6)
		HN	Cos(2)	TIME WDS	5	1.21	0.022 (1.0)
		ΗZ	SP(1)	WDS	6	6.94	0.032 (0.9)
		ΗZ	SP(1)	TIME	4	9.63	0.054 (0.7)
(9)AUT2015	CDS	HN	SP(1)	9	2	10.71	0.113 (0.4)
		ΗZ	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)
		ΗN	0	WDS	2	1.59	0.093 (0.5)
(10)SPR2016	CDS	HN	SP(1)	_	2	5.02	0.151 (0.3)
		ΗZ	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.

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Year	Period	LCL – Density – UCL	LCL – No. individuals - UCL	% CV	df			
		(Individuals km ⁻²)						
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 748

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- 750 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. 751
- 752 Shading represents urban areas.

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- 755 Figure 2. Estimates of Common Buzzard Buteo buteo population size within the study area in central
- Jetwe. south England for each survey period between autumn 2011 and spring 2016 (± 95% confidence 756

757 intervals).

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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 (\pm 95% confidence intervals).

81x185mm (96 x 96 DPI)