

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

Article

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

3

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8

9 Population change of Buzzards

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14

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
23 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,
26 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
29 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

34

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 al.*
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;
49 e.g. Link *et al.* 1994, Peterjohn *et al.* 1995, Jiguet 2009, Eglinton *et al.* 2010).

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 ($g(0)$) 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

61 estimated detectability (Stanbury & Gregory 2009) and decreases the precision of derived estimates
62 (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. 2600km² 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
110 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
112 identification of a circuit of appropriate length through the selected area. Transects followed the ITR

113 as closely as possible. Where deviations were necessary, alternative routes prioritised open access
114 land and other rights of way types before roads in an effort to reduce bias associated with following
115 obvious linear landscape features (surfaced roads, field edges, fences and hedgerows; Ortega &
116 Capen 2002, Marques *et al.* 2009). Edge effects were minimised by including all randomised start
117 locations even when these resulted in the transect breaching the study area boundary. In these
118 cases, only the lengths of transect within the study area were included in analyses.

119

120 Surveys were performed in ten transect periods, two each year, between Sep-Dec ('autumn') 2011
121 and Feb-May ('spring') 2016. Two seasonal transect periods were used to enable assessment of
122 expected fluctuations in density associated with post-breeding abundance and overwinter mortality.
123 Transects started between 08.30 and 10.00 from a randomised start point and took 3-7 hours to
124 complete. The direction (anti- or clockwise) of travel was also randomised. Each transect was
125 performed by one of two fieldworkers (MS or RH). Transects were walked only on days with no rain,
126 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

139 rectrices). Where there was still uncertainty regarding the status of an individual, the observer
140 noted their confidence in the observation being new on a percentage scale where '0' indicated a
141 certainty that the individual had already been recorded, and '100' where it had not. Bird movement
142 and relative timings and location of previous observations were used to inform this assessment. This
143 enabled later exclusion of observations from analyses, based on confidence. We adopted a
144 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

158 The restriction of transect routes to public rights of way and open access areas may have resulted in
159 the violation of the distance sampling assumption that all areas have equal probability of being
160 sampled. We determined the extent of entire study area unavailable for surveying using the 'Buffer'
161 function in ArcGIS (ESRI 2011). In this we produced a survey strip corresponding to the maximum
162 operational distance of the rangefinder (700m) on each side of all rights of way and open access
163 areas and deducted the extent of study area not covered by these strips (5.8km²) from the total size
164 of the study area in all subsequent distance analyses.

165

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
181 assumed *a priori* to influence the ability to detect birds through a biological or methodological effect
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
201 before they responded to the presence of the fieldworker (Burnham *et al.* 1980, Anderson *et al.*
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.
207 Exploration of the potential relationship between continuous, non-factor covariates and distance
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
211 addition of adjustment terms was enabled for analyses using CDS. For the MCDS engine, however,
212 this was restricted to a maximum of two cosine, simple polynomial or hermite polynomial
213 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.

216

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 $\chi^2(4) = 3.08, p = 0.55$;
240 spring $\chi^2(4) = 1.75, p = 0.78$) indicating that more observations were made when more transects
241 were walked. Significantly more individuals were seen during spring surveys than in autumn ($\chi^2(3) =$
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
287 period. Among the MCDS models, TIME, WDS/WDD and STRATA had the greatest effect on AIC, and
288 appeared in the majority of preferred models for each survey period.

289

290 **Population estimates**

291 Population size and density estimates increased throughout the course of study and were 0.6 birds
292 km^{-2} higher by 2016 than at the start of the study. Our analyses suggest an increase in estimated
293 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 *et al.* 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 (www.bto.org/bbs) 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
330 in areas of the SE region which lie to the north and east of the study area still appears likely since
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

340 Atlas data (Balmer *et al.* 2013) show the Buzzard to be uniformly distributed across all 10x10km
341 squares of SE England and from more than 85% of all 2km x 2km tetrads covered by atlas fieldwork
342 (2007-2012) in the SE region. Assuming that habitat quality and availability within our study area is
343 representative of that throughout the remainder of the SE region, then our density estimates
344 indicate a population size of 27500-32500 individuals in SE England. Translation of this figure into an
345 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-
352 Wales, and substantially lower than the densities (78 pairs 100km⁻²) recorded in ideal wooded
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
361 the first few months after fledging (Davis & Davis 1992, Walls *et al.* 1999) and that most do not
362 disperse more than 50km from the natal site in their first year (Walls & Kenward 1998). This is
363 particularly the case in landscapes with a significant arable component (Walls *et al.* 1999) where
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
367 impacts of overwinter and courtship mortality (Tubbs 1974, Simpson 1993) would theoretically
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

372 been offset by dispersal (Walls & Kenward 1995) and high rates of mortality for Buzzards in the four
373 months after fledging (Kenward *et al.* 2001).

374

375 ***Methodological assessment***

376 We encountered few obvious methodological issues with the study. Poor model fit using the MCDS
377 engine for the autumn 2014 was most likely the result of higher than expected numbers of birds
378 recorded between 275-325m in this survey period. This problem was not identified in other periods
379 suggesting that it is unlikely to represent any significant issue with survey design. Similarly, the issue
380 of poor precision was limited to one survey period and stems from reduced coverage; the level of
381 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
389 eastern section of the study area. Lastly, the inclusion of TIME is likely to relate to the behaviour of
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

396 Transects running through dense woodland may result in undetected responsive movements of
397 birds which may, in turn, lead to incorrect distance measurement or incomplete detection at $g(0)$. In

398 such habitats Buzzards were almost always heard to call prior to, or immediately after, taking flight
399 when disturbed by a fieldworker. Use of such cues to identify original locations for measurement
400 should have reduced the number of undetected responsive movements along transects performed
401 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
414 Buzzards to roads in our data. Whether the inclusion of roadside transect data has a significant
415 influence on the derived density estimates is open to question.

416

417 Although we adopted a number of strategies to reduce double counting, the duration of each
418 transect (mean - 371 minutes) means that there was ample opportunity for birds to move across the
419 study area. This is likely to have resulted in the double-counting of a small number of individuals
420 and possible positive bias to our estimates. Similarly, the inclusion of flying birds may also have
421 affected our results. Buckland *et al.* (2001) suggest that independent movement of birds can be
422 accommodated they must, 'on average', be moving at less than half the speed of the observer if they
423 are not to introduce a positive bias to the results. 24% of the observations here were of birds

424 engaged in purposeful, directional flight. Since neither the destination of flying birds nor their
425 duration of flight was recorded here, it is not possible to determine whether the average speed of
426 these individuals was less than half that of the observer. Whether the inclusion of these
427 observations has resulted in a significant positive bias to estimates is open to question.
428 Nevertheless, inclusion of some assessment of the nature and distances of flight behaviour in future
429 surveys would enable greater discrimination of data and exploration of impacts on derived
430 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
434 (e.g. the dilution of effort between target species and differential abilities of volunteers) have little
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
455 Buzzard populations has been followed by increasing pressure for population control measures to
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.

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.

474

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479

For Peer Review

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724 **Tables**

725 Table 1. Covariates used in modelling distance sampling estimates of Common Buzzards *Buteo buteo*
 726 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

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729 Table 2. Survey effort and numbers of observations made during surveys of Common Buzzards
 730 *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

731

732

733 Table 3. Details of model selection statistics for distance sampling estimates of the Common Buzzard
 734 *Buteo buteo* population in central southern England. Preferred models are indicated by bold type.
 735 (Period – survey period number, season and year; Engine – analysis engine; Key – key function, U-
 736 Uniform, HN – Half normal, HZ – Hazard rate; Adjustment Term – series expansion type (number of
 737 terms), Cos – Cosine, HP – Hermite polynomial, SP – Simple polynomial; Covariates – included in the
 738 model; Parameters – total number of parameters in the model; Δ AIC – difference in Akaike
 739 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	Key	Adjustment Term	Covariates	Parameters	Δ AIC	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)

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

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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

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748 **Legends to Figures**

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750 Figure 1. Study area in central southern England showing randomised locations of the Idealised

751 Transect Routes (ITRs) for the first survey of Common Buzzards *Buteo buteo* in autumn 2011.

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

756 south England for each survey period between autumn 2011 and spring 2016 (\pm 95% confidence

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)

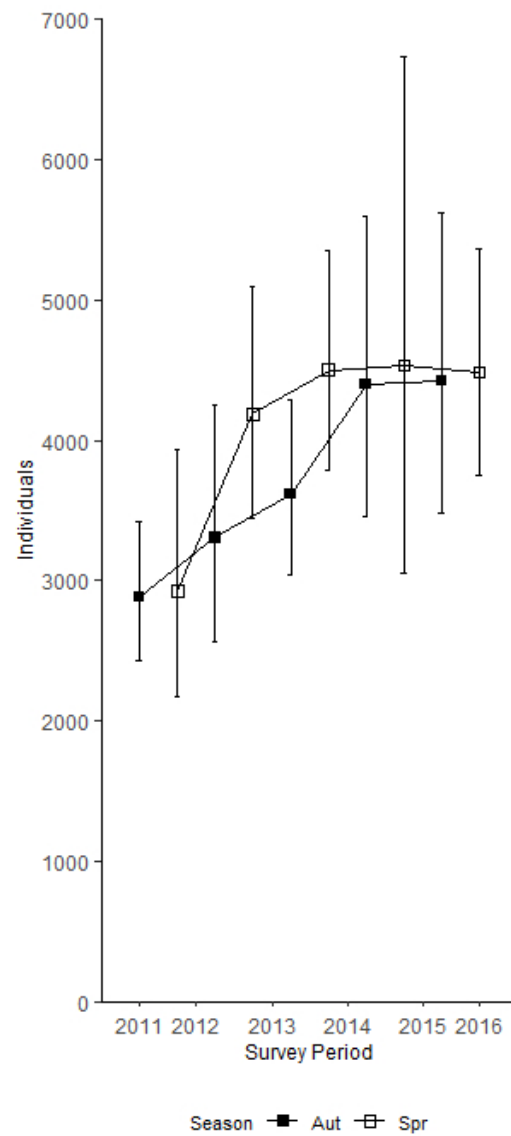


Figure 2. Estimates of Common Buzzard *Buteo buteo* population size within the study area in central south England for each survey period between autumn 2011 and spring 2016 (\pm 95% confidence intervals).

81x185mm (96 x 96 DPI)