University of
Reading

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

Article

Accepted Version

the End User Agreement.

www.reading.ac.uk/centaur

CentAUR

Central Archive at the University of Reading

Reading's research outputs online

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

Journal:	*Bird Study/Ringing & Migration*
Manuscript ID	BS-2019-014.R2
Manuscript Type:	Original Article
Date Submitted by the Author:	12-Aug-2019
Complete List of Authors:	Stevens, Matt; Hawk Conservancy Trust, Conservation & Research Murn, Campbell; Hawk Conservancy Trust, Conservation & Research Hennessey, Richard; Hawk Conservancy Trust, Conservation & Research
Keywords:	multiple covariates distance sampling, raptor, Population

SCHOLARONE™
Manuscripts

1   Population change of Common Buzzards *Buteo buteo* in central southern

2   England between 2011 and 2016

3

4   Matthew Stevens*, Campbell Murn and Richard Hennessey

5

6   Conservation and Research Department, Hawk Conservancy Trust, Andover,

7   Hampshire, UK.

8

9   Population change of Buzzards

10

11   Keywords: multiple covariates distance sampling, raptor, population

12

13   *Corresponding author: matt@hawkconservancy.org

14

15

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 2600km² 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

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;

49   e.g. Link *et al.* 1994, Peterjohn *et al.* 1995, Jiguet 2009, Eglington *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

3

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).

4

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

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

5

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 *at* 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

6

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 ($5.8km^2$) from the total size

164   of the study area in all subsequent distance analyses.

7

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

9

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

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.

10

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, $t_{255.2}$ = -

11

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

13

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

14

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

15

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

16

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

17

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, A*ccipiter 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

475 **Acknowledgements**

479

480 **References**

481

482 Adams. L.W. & Geiss, A.D. 1983. Effects of roads on small mammals. *J Appl. Ecol.* 20: 403-415.

483

484 Anderson, A.S., Marques, T.A., Shoo, L.P. & Williams, S.E. 2015. Detectability in audio-visual surveys

485 of tropical rainforest birds: the influence of species, weather and habitat characteristics. *PLoS ONE*

486 10(6): e0128464. Doi:10.1371/journal.pone.0128464.

487

488 Balmer, D.E., Gillings, S., Caffrey, B.J., Swann, R.L., Downie, I.S. & Fuller, R.J. (eds) 2013. *Bird Atlas*

489 *2007-11: the breeding and wintering birds of Britain and Ireland*. BTO Books, Thetford.

490

491 Beckmann, C., & Shine, R. 2011. Toad's tongue for breakfast: exploitation of a novel prey type, the

492 invasive cane toad, by scavenging raptors in tropical Australia. *Biol. Invasions* 13(6): 1447-1455.

493

494 BTO. (n.d.) Retrieved from https://www.bto.org/volunteer-surveys/bbs/latest-results/population-

495 trends

496

497 Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. & Thomas, L. 2001.

498 *Introduction to distance sampling.* Oxford: Oxford University Press.

499

500 Buckland, S.T., Marsden, S.J. & Green, R.E. 2008. Estimating bird abundance: making methods work.

501 *Bird Conserv. Int.*18:S91-S108.

502

503 Burke, B.J., Finna, A., Flanagan, D.T., Fogarty, D.M., Foran, M., O'Sullivan, J.D., Smith, S.A., Linnell,

504 J.D. and McMahon, B.J., 2015. Reintroduction of White-tailed Eagles to the Republic of Ireland: A

505 case study of media coverage. *Irish Geography*, *47*(1):95-115.

506

507   Burnham, K., Anderson, D., & Laake, J. 1980. Estimation of density from line transect sampling of

508   biological populations. *Wildlife Monogr.* 72:7-202.

509

510   Burnham, K.P. 1981. Summarizing remarks: Environmental influences. *Studies Avian Biol-Ser.* 6:324-

511   325

512

513   Christensen, T.K., Lassen, P. & Elmeros, M. 2012. High exposure rates of anticoagulant rodenticides

514   in predatory bird species in intensively managed landscapes in Denmark. *Archives of Environmental*

515   *Contamination and Toxicology* 63:437-444.

516

517   Clements, R. 2002. The Common Buzzard in Britain: a new population estimate. *Brit. Birds* 95:377-

518   383.

519

520   Dare, P.J. 1957. The post-myxomatosis diet of the buzzard. *Devon Birds* 10:2-6.

521

522   Davis, P.E. & Davis, J.E. 1992. Dispersal and age of first breeding of buzzards in central Wales. *Brit.*

523   *Birds* 85: 578-597.

524

525   Dean, W.R.J. & Milton, S.J. 2003. The importance of roads and road verges for raptors and crows in

526   the succulent and Nama-Karoo, South Africa. *Ostrich* 74(3-4):181-186.

527

528   Diefenbach, D.R., Brauning, D.W. & Mattice, J.A. 2003. Variability in grassland bird counts related to

529   observer differences and species detection rates. *Auk* 120(4):1168-1179.

530

22

531   Eglington, S.M., Davis, S.E., Joys, A.C., Chamberlain, D.E. & Noble, D.G. 2010. The effect of observer

532   experience on English breeding bird survey population trends. *Bird Study* 57:129-141

533

534   Ekblom, R. 2010. Evaluation of the analysis of distance sampling data: a simulation study. *Ornis*

535   *Svecica* 20:45-53

536

537   ESRI 2011. ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute.

538

539   Faanes, C.A. & Bystrak, D. 1981. The role of observer bias in the North American breeding bird

540   survey. *Studies Avian Biol-Ser.* 6:353-359.

541

542   Fewster, R.J.M, Southwell, C., Borchers, D.L., Buckland, S.T. & Pople, A.R. 2008. The influence of

543   animal mobility on the assumption of uniform distances in aerial line-transect surveys. *Wildlife. Res.*

544   35: 275-288

545

546   Forestry Commission. 2016. *National Forest Inventory Woodland England 2015*.

547

548   Forman, R.T.T. & Alexander, L.E. 1998. Roads and their major ecological effects. *Annu. Rev. Ecol.*

549   *Syst.* 29:207-231.

550

551   Frankham, R. 1995. Effective population size/adult population size ratios in wildlife: a review. *Genet.*

552   *Res. 66*(2), 95-107.

553

554   Glutz von Blotzheim, U., Bauer, K. & Bezzel, E. 1971. *Handbuch der Vögel Mitteleuropas*, Vol. 4. Aula

555   Verlag, Wiesbaden.

556

557   Harris, S.J., Risely, K., Massimino, D., Newson, S.E., Eaton, M.A., Musgrove, A.J., Noble, D.G., Procter,

558   D. & Baillie, S.R. 2014. *The Breeding Bird Survey 2013*. BTO Research Report 658. British Trust for

559   Ornithology, Thetford.

560

561   Harris, S.J., Massimino, D., Gillings, S., Eaton, M.A., Noble, D.G., Balmer, D.E., Procter, D. & Pearce-

562   Higgins, J.W. 2017. *The Breeding Bird Survey 2016*. BTO Research Report 700. British Trust for

563   Ornithology, Thetford.

564

565   Harris, S.J., Massimino, D., Eaton, M.A., Gillings, S., Noble, D.G., Balmer, D.E., Pearce-Higgins, J.W. &

566   Woodcock, P. 2019. *The Breeding Bird Survey 2018.* BTO Research Report 717. British Trust for

567   Ornithology, Thetford.

568

569   Hunt, W.G. 1998. Raptor floaters at Moffat's equilibrium. *Oikos* 82: 191-197.

570

571   Jiguet, F. 2009. Method learning caused a first-time observer effect in a newly started breeding bird

572   survey. *Bird Study* 56: 253-258.

573

574   Johnston, A., Newson, S.E., Risely, K. Musgrove, A.J., Massimino, D., Baillie, S.R. & Pearce-Higgings,

575   J.W. 2014. Species traits explain variation in detectability of UK birds. *Bird Study* 61: 340-350.

576

577   Kendall, C.J. 2014. The early bird gets the carcass: Temporal segregation and its effects on foraging

578   success in avian scavengers. *The Auk: Ornithological Advances* 131:12-19.

579

580   Kenward, R.E., Marcström, V. & Karlbom, M. 1999. Demographic estimates from radio-tagging:

581   models of age-specific survival and breeding in the goshawk. *J. Anim. Ecol.* 68: 1020-1033.

582

583   Kenward, R.E., Walls, S.S., Hodder, K.H., Pakhala, M., Freeman, S.N. & Simpson, V.R. 2000. The

584   prevalence of non-breeders in raptor populations: evidence from rings, radio-tags and transect

585   surveys. *Oikos* 91:271-279.

586

587   Kenward, R.E., Walls, S.S. & Hodder, K.H. 2001. Life path analysis: scaling indicates priming effects of

588   social and habitat factors on dispersal distances. *J. Anim. Ecol.* 70:1-13.

589

590   Knight, R.L. & Kawashima, J.Y. 1993. Responses of Raven and Red-tailed Hawk populations to linear

591   right-of-ways. *J. Wildlife Manage.* 57(2):266-271.

592

593   Lambertucci, S.A., Speziale, K.L., Rogers, T.E. & Morales, J.M. 2009. How do roads affect the habitat

594   use of an assemblage of scavenging raptors? *Biodivers. Conserv.* 18(8):2063-2074.

595

596   Lees, A.C., Newton, I. & Balmford, A. 2012. Pheasants, buzzards, and trophic cascades. *Conserv. Lett.*

597   00: 1-4. https://doi.org/10.1111/j.1755-263X.2012.00301.x

598

599   Link, W.A., Barker, R.J. & Sauer, J.R. 1994. Within-site variability in surveys of wildlife populations.

600   *Ecology* 75: 1097-1108.

601

602   Marques, T.A., Buckland, S.T., Borchers, D.L., Tosh, D. & McDonald, R.A. 2010. Point transect

603   sampling along linear features. *Biometrics* 66:1247-1255.

604

605   Melde, M. 1956. *Die Mausebussard.* Neue Brehm Bucherei, Leipzig.

606

607   Meunier, F.D., Verheyden,C., & Jouventin, P. 2000. Use of roadsides by diurnal raptors in agricultural

608   landscapes. *Biol. Conserv.* 92:291-298.

609

610   Moleón, M., Sánchez-Zapata, J.A., Gil-Sánchez, J.M., Barea-Azcón, J.M., Ballesteros-Duperón, E. and

611   Virgós, E., 2011. Laying the foundations for a human-predator conflict solution: assessing the impact

612   of Bonelli's eagle on rabbits and partridges. *PLoS One*, *6*(7), p.e22851.

613

614   Morelli, F. Beim, M., Jerzak, L., Jones, D. & Tryjanowski, P. 2014. Can roads, railways and related

615   structures have positive effects on birds? – A review. *Transport. Res. D-Tr. E* 30:21-31

616

617   Morton, R.D., Rowland, C.S., Wood, C.M., Meek, L., Marston, C.G. & Smith, G.M. 2014. Land Cover

618   Map 2007 (vector, GB) v1.2. NERC Environmental Information Data Centre.

619

620   Musgrove, A., Aebischer, N., Eaton, M., Hearn, R., Newson, S., Noble, D., Parsons, M., Risely, K. &

621   Stroud, D. 2013. Population estimates of birds in Great Britain and the United Kingdom. *Brit. Birds*

622   106, 64-100.

623

624   Newson, S.E., Evans, K.L., Noble, D.G., Greenwood, J.J.D. & Gaston, K.J. Use of distance sampling to

625   improve estimates of national population sizes for common and widespread breeding birds in the

626   UK. *J. Appl. Ecol.* 45:1330-1338.

627

628   Newton, I. Davis, P.E., & Davis, J.E. 1982. Ravens and buzzards in relation to sheep farming and

629   forestry in Wales. *J. Appl. Ecol.* 19: 681-706.

630

631   Nomis – Official Labour Market Statistics. 2016. *KS101EW (Usual resident population)*. Available

632   from: http://www.nomisweb.co.uk/census/2011/ks101ew [Accessed August 2018].

633

26

634   Öberg, M., Arlt, D., Pärt, Laugen, A.T., Eggers, S & Low, M. 2015. Rainfall during parental care

635   reduces reproductive survival components of fitness in a passerine bird. *Ecol. Evol.* 5:345-356.

636

637   Ortega, Y.K. & Capen, D.E. 2002. Roads as edges: Effects on birds in forested landscapes. *Forest Sci.*

638   48:381-390.

639   Pain, D.J., Fisher, I.J. & Thomas, V.G. 2009. A global update of Lead poisoning in terrestrial birds from

640   ammunition sources, In Watson, R.T., Fuller, M., Pokras, M. & Hunt, W.G. (eds) *Ingestion of Lead*

641   *from Spent Ammunition: Implications for Wildlife and Humans.* The Peregrine Fund, Boise, Idaho,

642   USA.

643

644   Parrott, D., 2015. Impacts and management of common buzzards Buteo buteo at pheasant

645   *Phasianus colchicus* release pens in the UK: a review. *European Journal of Wildlife Research*, 61:181-

646   197.

647

648   Peterjohn, B.G., Sauer, J.R., & Robbins, C.S. 1995 Population trends from the North American

649   breeding bird survey. In Martin, T.E. & Finch, D.M. (eds) *Ecology and Management of Neotropical*

650   *Migratory Birds*: 3-39. Oxford University Press, New York.

651

652   Pollock, K.H., Nichols, J.D., Simons, T.R., Farnsworth, G.L., Bailey, L.L, & Sauer, J.R. 2002. Large scale

653   wildlife monitoring studies: statistical methods for design and analysis. *Environmetrics* 13:105-119.

654

655   Quesada, J. Guallar, S., Pérez-Ruiz, N.J., Estrada, J. & Herrando, S. 2010. Observer error associated

656   with band allocation is negligible in large scale bird monitoring schemes, but how precise is the use

657   of bands at all? *Ardeola* 57:23-32.

658

659   R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for

660   Statistical Computing, Vienna, Austria. URL https://www/R-project.org/.

661

662   Rabe, M. J., Devos, S. S. & Rosenstock Jr, J.C. 2002. Review of big-game survey methods used by

663   wildlife agencies of the western United States. *Wildlife Soc. B*. 30:46–53.

664

665   Ralph, C.J. & Scott, J.M. Eds. 1981. Estimating numbers of terrestrial birds. *Studies Avian Biol.-Ser* 6.

666

667   Ramsey, F.L. & Scott, J.M. 1981. Tests of hearing ability. *Studies Avian Biol.-Ser* 6:342-345.

668

669   Riseley, K., Noble, D.G., & Baillie, S.R. 2008. *The breeding bird survey 2007.* BTO Research Report

670   508, Thetford, UK.

671

672   Robbins, C.S., Bystrak, D. & Geissler, P.H. 1986. *The Breeding Bird Survey: Its first fifteen years, 1965-

673   1979.* U.S. Fish. Wildl. Serv., Resour. Publ. 157.

674

675   Sanderlin, J.S., Block, W.M. & Ganey, J.L. 2014. Optimizing study design for multi-species avian

676   monitoring programmes. *J.Appl. Ecol* 51:860-870*.*

677

678   Sauer, J.S., Peterjohn, B.G. & Link, W.A. 1994. Observer differences in the North American Breeding

679   Bird Survey. *Auk* 111:50-62.

680

681   Sim, I.M.W., Campbell, L., Pain, D.J. & Wilson, J.D. 2000. Correlates of the population increase of

682   Common Buzzards *Buteo buteo* in the West Midlands between 1983 and 1996. *Bird Study* 47:154-

683   164.

684

685   Sim, I.M.W., Cross, A.V., Lamacraft, D.L. & Pain, D.J. 2001. Correlates of Common Buzzard Buteo

686   buteo density and breeding success in the West Midlands. *Bird Study* 48:317-329.

687

688   Simpson, V.R. 1993. Causes of mortality and pathological conditions observed in Common Buzzards

689   (*Buteo buteo*) in Cornwall, England. – In: Dorrestein, G. and van den Bergh (eds), *Proc. 1993 Eur.*

690   *Conf. Avian Med. Surgery.* Association of Avian Veterinarians, Utrecht, pp. 423-440.

691

692   Soulé, M. E. (Ed.). 1987. *Viable populations for conservation*. Cambridge university press.

693

694   Stanbury, A. & Gregory, R. (2009) Exploring the effects of truncated, pooled and sexed data in

695   distance sampling estimation of breeding bird abundance. *Bird Study* 56:298-309

696

697   Stephens, P.A., Pettorelli, N., Barlow, J., Whittingham, M.J, & Cadotte, M.W. 2015. Management by

698   proxy? The use of indices in applied ecology. *J. Appl. Ecol.* 52:1-6

699

700   Thiollay, J.M. 1967. Ecologie d'une population de rapaces diurnes en Lorraine. *Terre Vie* 114: 116-

701   183.

702

703   Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, R.B.,

704   Marques, T.A. & Burnham, K.P. 2010. Distance software: design and analysis of distance sampling

705   surveys for estimating population size. *J. Appl. Ecol.* 47: 5-14.

706

707   Thompson, W.L. 2002. Towards reliable bird surveys: Accounting for individuals present but not

708   detected. *Auk* 119:18-25.

709

710   Tubbs, C.R. 1971. *The Buzzard.* David and Charles.

711

712   Walls, S.S. & Kenward, R.E. 1998. Movements of radio-tagged Buzzards *Buteo buteo* in early life. *Ibis*

713   140:561-568.

714

715   Walls, S.S., Mañosa, S., Fuller, R.M., Hodder, K.H. & Kenward, R.E. 1999. Is early dispersal enterprise

716   or exile? Evidence from radio tagged buzzards. *J. Avian Biol.* 30:407-415.

717

718   Wildlife & Countryside Act 1981, Ch 69, *s*.1 & *s*.16

719

720   Woodward, I.D., Massimino, D., Hammond, M.J., Harris, S.J., Leech, D.I., Noble, D.G., Walker, R.H.,

721   Barimore, C., Dadam, D., Eglington, S.M., Marchant, J.H., Sullivan, M.J.P., Baillie, S.R. & Robinson,

722   R.A. (2018) *BirdTrends 2018: trends in numbers, breeding success and survival for UK breeding birds.*

723   Research Report 708. BTO, Thetford. www.bto.org/birdtrends

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

727

728

31

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

32

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; ΔAIC – 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	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)

741

742

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.

745

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

746

747

35

748   **Legends to Figures**

749

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.

753

754

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 (± 95% confidence

757   intervals).

758

759

760

761

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.
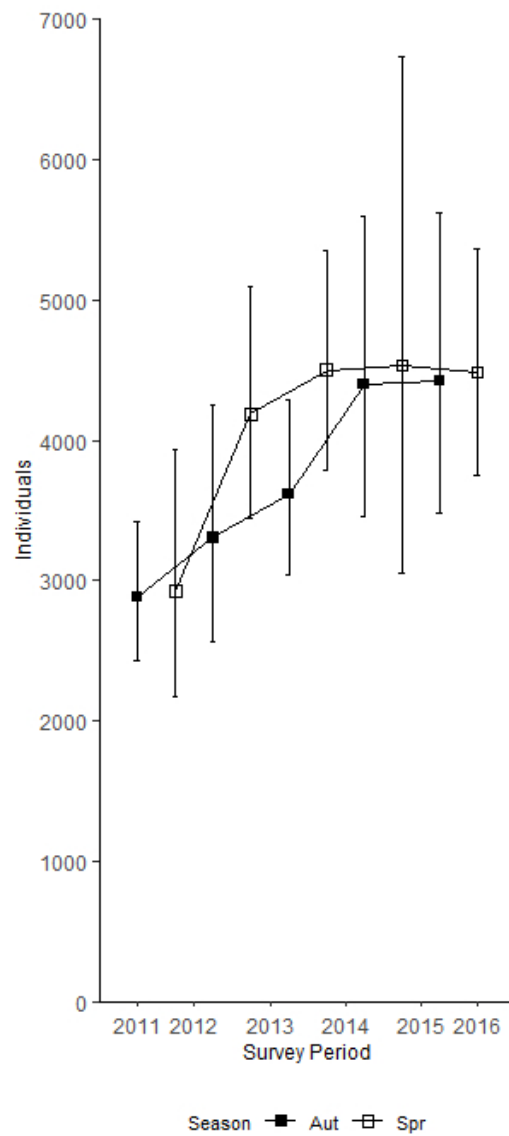
164x83mm (96 x 96 DPI)

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)