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The UK great bustard *Otis tarda* reintroduction trial

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Abstract

Great bustards became extinct in the UK during the 19th century due to a combination of factors including hunting, egg collection and changes in agriculture. In 2003, a 10 year licence was granted to begin a trial to reintroduce the species back to the UK. Here we report on the first five years of the trial and assess the progress made towards establishing a founder population. From April 2004 to September 2009 a total of 102 great bustard chicks were imported from Russia and 86 released on Salisbury Plain. Monitoring showed that post-release survival was 18% in the first year following release, and that mortality of released bustards was mainly attributable to predation and collisions. Estimated adult survival was 74%, although the sample size was small. All known surviving great bustards are site faithful to the surroundings of the release site returning throughout the year. A lek area has been established where males have been observed displaying to females. The first nesting attempt was in 2007, and in 2009, two females aged 3 and 4 successfully nested, fledging one chick each. Models incorporating the new demographic estimates suggest that at the end of the 10-year trial period the project can expect to have between 8 and 26 adults as a founder population.

Keywords: great bustard, *Otis tarda*, reintroduction, translocation, monitoring, Britain, Russia, UK

1 **Introduction**

2 Reintroduction has become an accepted intervention in conservation (Seddon et al., 2007;
3 Armstrong & Seddon, 2008), and has been widely used for various organisms including birds
4 and mammals. Due to the financial costs and low success rate, rigorous assessment of feasibility
5 is essential prior to implementation as well as appropriately long post-release monitoring to
6 assess success (IUCN, 1998). However, when gaps in knowledge exist about the ecology of a
7 species in an area from where it was extirpated, it is often difficult to determine the ability of that
8 species to survive and persist once the original causes of extinction are removed. Consequently,
9 trial reintroduction provides an opportunity to fill in the gaps in understanding and to assess the
10 feasibility of a full scale reintroduction project (Osborne, 2005).

11

12 Although the aim of a reintroduction is to establish a free-living, self-sustaining population, the
13 progress of a reintroduction comprises a sequence of objectives, including the survival of
14 released individuals, breeding by released individuals in the wild and then subsequent growth
15 and persistence of the population (Seddon, 1999). Each of these stages must be assessed to
16 identify the appropriate methodology and limitations (Seddon et al., 2007; Sutherland et al.,
17 2010) and the importance of post-release monitoring has been increasingly emphasised in
18 assessing progress with reintroduction projects. In many cases, even today, monitoring is found
19 to be inadequate to appropriately assess success or failure (Fischer & Lindenmayer, 2000;
20 Armstrong & Seddon, 2008). In order to assess the feasibility of reintroducing the great bustard
21 *Otis tarda* to the UK, it was recognised that comprehensive post-release monitoring would be
22 needed to assess progress and to inform future strategic decisions.

23

24 The great bustard has a fragmented distribution extending across the middle latitudes from
25 Morocco to China (Morales & Martin, 2003). It is found on lowland grassland, steppe and arable
26 land and often displays a preference for low intensity agriculture and crops over natural
27 grasslands (Suarez-Seoane et al., 2002; Oparin et al., 2003; Morales et al., 2006; Magana et al.,
28 2010). It is a polygamous ground nesting bird and exhibits the largest sexual size dimorphisms of
29 mass (females 3.5-7.2kg, males 7-13kg) found in any bird species (Raihani et al., 2006; Székely
30 et al., 2007; Alonso et al., 2009). The great bustard was a common breeding bird across large
31 parts of Europe and Asia through the 18th century (Gewalt, 1959), but has experienced dramatic
32 declines and local extinctions across its range during the 20th century (Palacin, 2008). It is
33 currently classified as Globally Vulnerable (IUCN, 2010). The UK breeding population became
34 extinct *circa* 1830 (Morales & Martin, 2003) and although the true cause of extinction is
35 unknown (Collar, 1979) it was likely due to a combination of factors that included hunting, egg-
36 collection and changes in agricultural practice (Osborne, 2005). Due to the geographical isolation
37 of the UK from existing Central and Southern European populations and the high site fidelity of
38 great bustards (Martin et al., 2008), natural recolonisation is unlikely to occur, even if conditions
39 were suitable (Carter & Newbery, 2004). An attempt was made to re-establish the species in the
40 UK at Porton Down in Wiltshire, between 1970 and 1998, but this was unsuccessful because the
41 captive breeding approach failed to produce any chicks that survived to be released (Collar &
42 Goriup, 1980). Further captive-breeding attempts have so far been unsuccessful at producing
43 enough chicks to make a reintroduction attempt viable (Martin, 1996). Therefore, translocation
44 of captive-reared birds from a wild donor population is the only realistic option for a
45 reintroduction project. A donor population was identified in Russia where eggs from nests that
46 would otherwise be lost to cultivation could be rescued and birds hatched and reared in captivity.

47 Following a rigorous feasibility study based on the IUCN reintroduction guidelines (Osborne,
48 2002), it was concluded that there would be no detrimental effects to the donor population and
49 that the habitat and conditions on Salisbury Plain in Wiltshire could support this species.
50 However, gaps exist in our knowledge of great bustard ecology and its ability to persist within
51 the UK due to the long absence of the species. For that reason a 10 year licence allowing the
52 release of birds on a trial basis was issued in 2003 by the UK Government's Department for
53 Environment, Food and Rural Affairs (Defra). In accordance with the IUCN Guidelines on
54 Reintroductions, the licence application had a number of success indicators to help steer the
55 project through the initial stages of establishing a founder population.
56 Here we report on progress with this project, and evaluate it using the success indicators. We
57 then model population growth using revised parameter estimates based on data from the
58 reintroduced population and compare the outcomes with those predicted in the original feasibility
59 study. This review covers the period from 30 April 2004 (when the first eggs were collected) to
60 14 September 2009 (five years after the first release) by which time the first released birds were
61 approximately 60 months old.

62

63 **Study area**

64 The release site is on Salisbury Plain, the largest continuous area of calcareous grassland in
65 North-west Europe, which is mainly contained within the county of Wiltshire. It has a low
66 density of settlements and roads, and land-use is split between low-intensity grazing, agriculture
67 and an extensive military training area (380 km²). The Salisbury Plain Site of Special Scientific
68 Interest and Special Protection Area cover 197 km² and is protected under domestic legislation
69 and the EU Birds Directive (Osborne 2005).

70

71 **Methods**

72 Donor population and egg collection

73 The donor population is located in the Saratov Oblast, Russian Federation (50°50' N, 46°12' E).

74 Eggs are rescued from nests that would otherwise be lost to cultivation and transported via a

75 portable incubation unit to a rearing station. The nesting period is prolonged with a gap of 4

76 weeks between the hatching of the first and the last clutches. All eggs that fail to hatch are

77 autopsied to check for infertility and other causes. After hatching all chicks are individually

78 marked with coloured leg rings. Chicks are reared in cohorts of similar age and fed using a

79 dehumanising suit and puppet to decrease the risk of imprinting on humans. Juvenile birds,

80 between the ages of 30 and 70 days, are transported to the UK in animal crates in compliance

81 with various national and international regulations, including CITES. Once in the UK, the birds

82 are kept in quarantine for one month during which time they are screened for avian influenza and

83 paramyxovirus. Chicks are sexed by body size from the sexual size dimorphism that becomes

84 apparent from about 60 days.

85

86 Release and monitoring

87 Release into the wild takes place each year between September and October depending on the

88 import date and weather conditions. The mean age at release is 100 days (range 58 -145), and

89 birds are released into an open-topped fenced area (hereafter release pen) designed to exclude red

90 foxes *Vulpes vulpes* and badgers *Meles meles*. The release pen was 3.5 hectares from 2004-2007

91 and then extended to 7 ha in 2008. The release pen is managed to contain a mosaic of arable

92 crops, including oil seed rape, alfalfa and natural grassland. The release technique used during

93 release has varied between years. In 2004, birds were kept in a netted enclosure within the

94 release pen to allow them to become familiar with the environment prior to release but this

95 method was discontinued due to two males being injured colliding with the pen during attempts

96 at flight (Osborne & Fraser, 2005). In subsequent years, all birds were released directly into the

97 release pen. Limited predator control against magpies *Pica pica*, carrion crows *Corvus corone*
98 (potential nest predators) and red foxes was carried out in the immediate area around the release-
99 pen. However, predator control does not occur in the adjacent military areas and there is
100 therefore the potential for rapid replacement of removed predators.

101
102 Released bustards were individually marked with numbered wing-tags, colour-coded to identify
103 release year. In addition, 55 individuals were fitted with back-pack, neck-collar or tail-mounted
104 radio-transmitters (Biotrack TW-3), and ten individuals were fitted with Argos/GPS enabled
105 Platform Transmitter Terminals (PTT) (PTT-100 105 gram LC4™ for males and PTT-100 40
106 gram LC4™ for females, Microwave Telemetry, Inc.) that provided daily information on
107 location.

108
109 The release site and its surroundings are intensively monitored by GBG staff. In addition, with
110 the aid of public relations work locally and nationally opportunistic sightings away from the
111 release site were reported by the public approximately once a week on average. Sightings of
112 birds reported by the public were only used in this analysis when individual identity was verified.
113 Mortalities were recorded via wing-tag, transmitter and carcass recoveries. Dates of mortality
114 were estimated and were usually thought to be accurate to within a few days of death. Post-
115 mortems were carried out when possible to establish cause of death, including screening for
116 disease. Released individuals that were injured after release and taken back into captivity were
117 recorded as ‘dead’ for the purposes of analysis with date of injury taken as the date of mortality.

118

119 Estimating reproductive and survival parameters

120 Breeding attempts were assessed through intensive monitoring of the release area during the
121 breeding period (March-June). Key signs indicative of breeding are displaying by males, signs of
122 pecking on a female's head and females exhibiting nesting behaviour (Morales & Martin, 2003).

123
124 Due to the small sample sizes, data for males and females were pooled and only age-specific
125 effects on survival were investigated. Pre-release survival is defined as the proportion of chicks
126 surviving from import to release. Post-release mortality was defined as the period from release to
127 12 months after release. Daily survivorship for the first year post-release was estimated using
128 Kaplan-Meier survival function. Individuals that were neither recovered nor resighted after 365
129 days were censored at their last resighting, whereas individuals that were known to survive for at
130 least 365 days were censored at 365 days.

131
132 To estimate annual age-specific survival probabilities, we used mark-resight data from all 89
133 released individuals and recoveries of dead birds. Date of marking was considered the day of the
134 bustard's release, and intervals were set to one year from the date of first release on 15
135 September 2004 until 14 September 2009. Data used in the annual survival analysis were from
136 resightings at the release site from September to December to meet the assumptions that
137 resightings are obtained during a predefined sampling interval and within a defined sampling
138 area. Dead recoveries from throughout the year were considered. Probabilities were estimated
139 with program MARK (White & Burnham, 1999), using a Burnham Joint Live and Dead
140 Encounters model that controls for study area effects (fidelity versus dispersal) that may
141 influence detectability and probability of recovering dead individuals. The model was
142 parameterised to explore differences between the year following release (juvenile 0-1 year) and
143 adulthood (>1 years). The candidate models were ranked using Corrected Akaike's Information
144 Criterion (AICc) and the top performing model chosen. The logit link function was used for all
145 the models.

146

147 Success criteria

148 Success indicators (Table 1) were proposed in the original feasibility study to assess short-term
149 progress and to identify limitations. They were based on the best data available at the time from
150 comparable great bustard rear-and-release projects in Hungary and Germany, and from wild
151 populations in Spain (Osborne, 2002).

152

153 Population modelling

154 We developed two demographic models to estimate the size of the founder population at time t
155 (N_t) at the end of the trial period, and used these models to explore various scenarios. Firstly, we
156 used the deterministic model (Eq. 1) to compare predicted population estimates before the trial
157 (Osborne, 2002) with new estimates based on data presented in this paper. In the original model
158 (Model 1a), the number of eggs collected, fertility rate and hatch rate were used to estimate the
159 number of birds required to be imported into the UK. These three parameters were replaced by
160 the mean number of chicks imported (I), a product of the three components. Mortality was
161 applied instantaneously to reflect the number of birds surviving by May each year where S_a is
162 adult survival and S_{pre} and S_{post} are pre and post-release survival respectively. We used the mean
163 for each parameter estimate and did not incorporate measures of variation. Two scenarios were
164 modelled; (Model 1b) using the actual mean number of chicks translocated each year, and
165 (Model 1c) the desired number of chicks imported each year (40). These were compared with the
166 pre-release survival model of Osborne (2002) and with the actual population growth.

167

168 **Eq.1** $N_{t+1} = N_t \times S_a + I \times S_{pre} \times S_{post}$

169

170 Secondly, we incorporated demographic stochasticity in age-specific survival probabilities. To
171 account for uncertainty with survival estimates, 1,000 iterations of the population model were
172 simulated. Each iteration was randomly assigned a survival probability for each age class from a

173 survival probability beta distribution using the mean survival probability and its variance. Using
174 three post-release survival rates (13%, 18%, 23%) and four importation rates (10, 20, 30, 40),
175 twelve scenarios were investigated. Simulations were performed in R (R Development Core
176 Team 2008). Neither modelling approach incorporated breeding for two reasons. First, our aim
177 was to understand how many captive-reared individuals will be established through release only;
178 and secondly, there are still few data available on reproductive parameters of released birds or
179 survival of wild-reared bustards.

180

181 **Results**

182 **Number of released bustards**

183 The number of eggs rescued varied between years influencing the number of imported and
184 released bustards (Table. 2). Ultimately, 102 chicks were imported into the UK (at an average of
185 approximately 20 birds per year). Pre-release survival from import to release (typically one
186 month) was high at 88.0% (SE \pm 6.3 %). A total of 86 birds were released, of which 45 were
187 females (52.3%), 33 were males (38.3%), and 8 were unsexed (9.3%) (all birds of unknown sex
188 died or were not re-sighted before any obvious sexual dimorphism was apparent). Of the birds
189 released, 69.7% had known fates (7 alive, 46 dead and 7 disabled) whereas 30.3% had unknown
190 fates by September 2009.

191

192 **Survival and causes of mortality**

193 Mortality was high for the first 150 days after release (Fig. 1a) with only 26.9% (95 % CI 17.7 -
194 41.1) of individuals surviving through this period. Subsequently survival from 150 to 365 days
195 remained constant with no deaths reported, although, two individuals were not seen after 305

196 days. There was no significant difference between the survival rate of males and females (Cox
197 regression: $n = 78$, z -value = 0.224, p -value = 0.82, Fig. 1b). Annual survival from the top-
198 ranked model in the mark-recapture analysis (Table 3) for one year after release was estimated as
199 18.2 % (95 % CI 10.8 - 28.9). Annual survival for birds greater than one year was estimated at
200 74.6 % (95 % CI 51.4 - 89.2). There was a high probability of resighting newly released
201 individuals at the release site (82.2%, 95% CI 50.2 - 95.4), and all surviving individuals greater
202 than 1 year old returned to the release site. Using the joint live and dead encounters model the
203 estimated probability of recovering dead individuals was 67.4% (95% CI. 55.7 -77.2).

204
205 Pre-release survival through transportation and quarantine was high but some individuals were
206 injured as a result of collisions with the quarantine pens during attempts to fly. Post-release,
207 there was no evidence in recovered carcasses of starvation or malnourishment. In many cases, it
208 was difficult to determine the cause of death because carcasses had been scavenged. There were
209 two isolated incidences of infection in 2005 with one instance of yersiniosis, probably from a
210 wild rodent, and one unidentified infection. Predation accounted for death in most cases where
211 the cause of death could be confirmed. Red foxes are thought to be responsible for the majority
212 of predation, although the European badger may also have been responsible. Collision, with
213 power-lines and agricultural fences, was the second most important cause of mortality identified
214 (Fig. 2).

215
216 **Breeding**
217 The first great bustard clutch was laid in 2007 by a 2 year old female. She also nested in 2008,
218 although, clutches in both years proved to be infertile. In 2009, two females nested (a 3 year old

219 female and the 4 year old female that had nested the previous two years) with one clutch
220 hatching one chick and the other hatching two chicks. Both females successfully fledged one
221 chick.

222

223 Success indicators

224 Four of the nine indicators were met (Table 1). In addition, three of the targets were potentially
225 met given the large range of uncertainty in parameter estimates due to the small sample sizes.
226 The following three targets were below adequate. First, the average number of birds imported
227 has been half of the licence import quota of forty birds each year. Second, post-release survival
228 was lower than the expected level, and third, adult survival failed to meet the adequate target for
229 either sex although the confidence limits are large.

230

231 Population Modelling

232 Based on the parameters in Table 4, model 1a estimated 108 individuals after 10 years using 400
233 imported bustards (Fig. 3). However, it was apparent after the first year of release that this would
234 be an overly optimistic prediction (Osborne & Fraser 2005). Model 1b predicts 12 individuals
235 will be recruited into the breeding population from 200 imported birds during the trial period
236 (Fig. 3), and is consistent with actual population growth (Fig. 3). Importing the originally
237 estimated 40 birds per year would double the predicted founder population to 24 individuals.

238 Model 2 shows that there is uncertainty in how the population might develop due to large
239 variation in the estimates of post-release and adult survival (Fig. 4). At the current rates of 20
240 birds imported per year and 18% post-release survival, the founder population has a 95 %
241 probability of reaching 8 to 26 individuals at an equal sex ratio. Improving the post-release

242 survival rate to 23 % and importing 40 chicks each year resulted in a predicted founding
243 population of 32 to 56 individuals. The predictions are on average higher than estimated in
244 Model 1. However, currently the population growth is at the lower end of the outcomes predicted
245 by these models. The models suggest that if importation rates are maintained at less than twenty
246 birds then the founder population after 10 years would be between 1 and 11 individuals.

247

248 **Discussion**

249 The first 5 years of the reintroduction trial have demonstrated that great bustards can be hatched
250 in captivity from wild collected eggs, and that juveniles can be translocated from Russia and
251 successfully released into the wild in the UK. Further, it has shown that released birds can
252 survive in the wild over long periods and are generally faithful to the release area. In recent
253 years, released birds have reached maturity and have reproduced successfully on Salisbury Plain.

254

255 The progress of the project has been assessed through intensive monitoring of individuals that
256 were systematically marked from the beginning of the trial. Although post-release mortality has
257 been high, individuals from every year of release have survived to adulthood, confirming that the
258 habitat around Salisbury Plain can support individual great bustards. The reintroduced great
259 bustards have not exhibited the long distance migratory behaviour known from the donor
260 population in Russia (Watzke, 2007b), with the exception of three individuals that flew to France
261 in 2005 shortly after their release. Ultimately two of these individuals died and one remains
262 unaccounted for. All known surviving bustards have returned to the release site throughout the
263 year but they are also known, based on reports of wing-tagged birds and data from birds fitted
264 with satellite tags, to have explored more widely in south-west England (unpublished data).

265

266 The presence of conspecifics has been shown to influence the natal dispersal of great bustards in
267 wild populations as well as the use of lek sites by males, with juvenile birds using the presence
268 of other great bustards as indicators of habitat quality (Alonso et al., 2004; Martin et al., 2008).
269 However, even in the absence of these cues, males in the UK have been observed displaying and
270 returning to the release site to lek. This has management implications in relation to the potential
271 benefits of establishing new release sites and leks in south-west England to start creating a meta-
272 population structure.

273

274 Whilst released females have been able to locate suitable nesting habitat and have successfully
275 reared chicks suggesting that arthropod abundance/biomass is sufficient in the breeding area. The
276 number of breeding attempts is so far too small to be able to draw any empirical conclusions
277 about nesting success or productivity rates. The level of breeding success was expected to be low
278 at the beginning on the project as first time breeders tend to have a lower success rate than more
279 mature and experienced birds (Ena et al., 1987; Morales et al., 2002; Watzke, 2007a; Martinez,
280 2008). As the population grows and the age structure develops, we would expect breeding
281 success rates to improve as more females will be available to breed and older females have a
282 higher probability of successfully rearing chicks (Morales et al., 2002). The social stimulation
283 created by recruitment of breeding individuals will reduce potential Allee effects associated with
284 small populations.

285

286 The low numbers of birds released together with low post-release survival rates have clearly
287 limited the short-term success of the project. In extant populations, juvenile survival is estimated

288 at 29.9% for the first year (Martin et al., 2007). Our results are comparable although the high
289 mortality phase takes place post-release, when birds are 3-8 months old, rather than during the
290 first 3 months as recorded by Martin et al. (2007). Low post-release survival is common in
291 reintroductions across various taxa (Teixeira et al., 2007). This is potentially the result of
292 captive-reared individuals lacking the appropriate behavioural responses to survive in the wild
293 (Griffin & Blumstein, 2000). Maternally learned skills are likely to be important to juvenile great
294 bustard survival because in the wild they stay with the mother for greater than six months
295 (Martin et al., 2008). A number of studies have shown that wild-reared individuals in general
296 have higher survival rates than captive-reared conspecifics (Griffith et al., 1989; Wolf et al.,
297 1996). The Brandenburg (Germany) great bustard release project has experienced similar and
298 variable post-release survival estimated between 15 % and 40% of released individuals from
299 release to the following spring (Eisenberg, 2008). Fox predation had been the main cause of
300 mortality in this project, which was mitigated to some extent by the use of predator-free fenced
301 areas. Recently however, white-tailed eagles *Haliaeetus albicilla* are reported to have caused
302 substantial mortality in juvenile birds (Eisenberg, 2008). Houbara bustard *Chlamydotis undulata*
303 release projects have had similar difficulties with fox predation (Combreau & Smith, 1998). It
304 has been demonstrated in a wide range of taxa that captive reared animals lack essential skills
305 such as predator recognition (Griffin & Blumstein, 2000). Predator-awareness training with a
306 live predator has been shown to improve post-release survival in houbara bustards (van Heezik et
307 al., 1999), although, there are few empirical studies that confirm its effectiveness in other species
308 of birds. In great bustards, the effort to elicit the correct flight response to a predator can lead to
309 injuries when the birds are in confinement (D. Waters, pers. comm.) Collisions have also been an
310 important cause of mortality post-release which may be due in part to behavioural naivety or

311 perhaps to a reduction in feather quality due to time spent in captivity and the handling necessary
312 during transport and veterinary checks. However, collisions are known to be an important cause
313 of mortality in wild great bustard populations (Janss & Ferrer, 2000) and in other species of
314 bustards in the wild; kori bustard *Ardeotis kori* (Martin & Shaw, 2010), Denham's bustard *Neotis*
315 *denhami* (Shaw et al., 2010) and in little bustard *Tetrax tetrax* (Silva et al., 2010).

316

317 In the wild, adult survival is thought to be high, being estimated at 92% in Iberian populations
318 (Martin et al., 2007). The reintroduced UK population has a similarly high rate although the
319 estimate is based on only a very small sample. Estimates and the resulting reliability of model
320 predictions should improve in future when more data become available. On current estimates we
321 predict that the growth of the founder population through continued supplementation will be
322 slower than originally thought (Osborne, 2002). This is primarily due to the small numbers
323 recruiting into the population which is a product of low import numbers and post-release
324 survival. Assuming these parameters do not improve, a longer period of time will be required to
325 establish a founder population sufficiently large to have a high chance of persisting in the wild in
326 the long term. Whilst the population is still small it remains prone to stochastic events such as a
327 period of high mortality due to extreme weather and a reduction in the potential benefits of social
328 stimuli from con-specifics.

329

330 Small population size also increases the challenge of assessing the project quantitatively rather
331 than qualitatively, making it harder to plan for the future (Seddon et al., 2007; Armstrong &
332 Seddon, 2008). There is limited value in measuring progress of a reintroduced population using
333 data from existing wild populations, as there will be considerable variation in factors such as

334 anthropogenic disturbance, infrastructure, predator types, climate, attitudes to conservation, and
335 availability/access to other sub-populations in different areas. The values for demographic
336 parameters required to ensure a self-sustaining population are thus likely to be different in
337 southern England from those of other populations. For this reason, modelling is the appropriate
338 approach to investigate the rates of recruitment needed for sustainable growth and to set success
339 indicators accordingly (Seddon, 1999; Seddon et al., 2007; Armstrong & Seddon, 2008). The
340 reliability of these models should improve as more data become available in future from
341 monitoring of the expanding population.

342 Reintroduction projects involving red kite *Milvus milvus*, white-tailed eagle and, to a lesser
343 extent, osprey *Pandion haliaetus* in Britain have already achieved considerable success (Green et
344 al., 1996; Evans et al., 2009). Although, Cade (2000) pointed out the success rates tend to be
345 higher in birds of prey than in other birds. The great bustard is one of a number of bird
346 reintroductions that are currently in progress in Britain, with other projects involving the
347 corncrake *Crex crex*, ciril bunting *Emberiza cirilus* and common crane *Grus grus*. The results of
348 these projects may have considerable influence on the extent to which this sometimes
349 contentious approach to conservation is utilised in future.

350

351

352 **Recommendations**

353 Although this experimental reintroduction has shown some encouraging signs of success, it is
354 still at an early stage and further work will be required in order to establish a self-sustaining
355 population. The numbers of birds released annually are, on average, about half of that planned.
356 While this may not undermine the project in the long term, it does increase the likely timescale
357 for success. Accordingly, it is recommended that priority should be given to increasing the
358 number of birds released each year. Continued supplementation will provide a valuable buffer

359 against stochastic effects that could result in the population being reduced to dangerously low
360 levels or even extinction. Current rates of post-release survival are limiting the growth of the
361 population. Experimentally investigating ways to improve survival may offset low import
362 numbers and will be of potential benefit to global bustard conservation as a whole. The
363 importance of accumulating improved demographic data for the newly established population for
364 modelling purposes cannot be overstated. Long-term post-release monitoring will be essential to
365 inform a strategy for taking this reintroduction project forward, through improving our
366 understanding of habitat use, breeding productivity, survival of wild-reared chicks and likely
367 rates of population growth.

368

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380 experiences of rearing and releasing great bustards from Körös-Maros National Park (Hungary),
381 and Torsten Langgemach from the Brandenburg Project (Germany). Initial monitoring work

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385

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505

506

507 **Biographical sketches**

508 ROBERT JOHN BURNSIDE is interested in the use of reintroduction as a conservation tool and
509 the design of monitoring techniques for assessing success. IAN CARTER has worked as an
510 ornithologist with Natural England (and predecessors) for over 20 years and has a particular
511 interest in bird reintroductions. ALASDAIR DAWES has worked in conservation for over 10
512 years, with a particular interest in birds. DAVID WATERS is the Director of the GBG, an
513 organisation he founded in 1998. LEIGH LOCK works for the RSPB on various species
514 recovery projects across the UK. PAUL GORIUP has worked with great bustards for 30 years.
515 TAMAS SZEKELY is interested in biodiversity conservation and specialises in wader breeding
516 systems.

517 **Table 1.** Targets and success indicators for the first five years of the great bustard
 518 reintroduction and estimates of demographic rates achieved. Success indicators are derived
 519 from release projects in Hungary and Germany, and from wild birds in Spain (Osborne,
 520 2002).

Measure	Adequate	Excellent	Reintroduction project 2004-2009 mean ± Standard Error (n)	Target met?
1. Hatching success of				YES
artificially incubated	54%	75%	66.4 (232)	
eggs				
2. Number of chicks moved				NO
from Russia each year	30	40	20.4 ± 5.9 (5)	
3. Pre-release survival				YES
(males)	53%	75%	} 88.0 ± 6.3 % (102)*	
4. Pre-release survival				YES
(females)	45%	75%		
5. Post-release survival to				NO
end of year one (males)	25%	28%	} 18.2 ± 4.6% (86)* 95% CI 10.8% - 28.9%	

6. Post-release survival to end of year one (females)	38%	42%		NO
7. Post-release survival from year one per annum (males)	78%	87%	} 74.6 ± 10% (10)* 95%CI 51.4% -	NO
8. Post-release survival from year one per annum (females)	83%	92%		89.2%
9. Year in which first evidence of breeding is recorded	5	4	4	YES

521

522 * Male & female data are pooled

523

524 **Table 2.** The number of eggs collected and hatched in Russia and chicks transported and
525 released in the UK great bustard reintroduction trial from 2004 to 2008.

526

Year	2004	2005	2006	2007	2008	Total
Number of eggs collected	61	61	25	32	53	232
Number of eggs hatched	48	47	14	6	39	154
Number of chicks transported to UK	28	38	9	6	21	102
Number of chicks released in the UK	22	32	9	6	17	86

527

528

529 **Table 3.** Summary of model selection from annual survival of great bustards from the UK
 530 reintroduction trial as calculated from a Burnham Live and Dead Encounters. Age specific
 531 mortality (S_i), probability of resighting a live individual (p_i), probability of recovering a dead
 532 individual (r_i) and probability an individual will remain in the sampling area (F_i) were
 533 considered. Age structure (i) was defined as 1, where the estimate is constant across age
 534 groups and, 2 for two age groups split into first year and adult (2-5 years). All models were
 535 fitted with a logit link function and ranked according Corrected Akaike Information Criteria
 536 (AICc).

537

Model	AICc	Delta	AICc	Model	Number of	Deviance
		AICc	Weights	Likelihood	Parameters	
S₂ p₂ r₁ F₁	209.89	0	0.475	1	4	34.79
S₂ p₂ r₂ F₂	211.09	1.20	0.260	0.54	5	33.80
S₂ p₂ r₁ F₂	211.78	1.89	0.184	0.38	5	34.49
S₂ p₀ r₂ F₂	213.73	3.83	0.069	0.14	5	36.44
S₂ p₂ r₂ F₂	217.94	8.05	0.008	0.017	8	33.80
S₂ p₀ r₁ F₁	224.84	14.9	0.00027	0.0006	7	43.03
S₀ p₂ r₁ F₁	227.89	18.00	0.00006	0.0001	3	54.95

538

539

540 **Table 4.** Model parameters for the reintroduced great bustard population

541

Parameter	Parameters of model (Osborne 2002)	Achieved parameters (2004 – 2009)
Eggs collected each year	75	46.4*
Fertile Eggs	75 %	84.3 %*
Eggs Hatch	72 %	66 %*
Number of chicks imported	40.5	20.4
Conservative survival until release (male)	53 %	88 %
Conservative survival until release (female)	43 %	88 %
Post-release survival to end of year one (females)	88 %	18.2 %
Post-release survival to end of year one (males)	88 %	18.2 %
Survival > 1yr old female	87 %	74.6 %

Survival > 1yr old

92 %

74.6 %

male

542 *Parameters were not used in the model

543

544 **Figure legends**

545 **Figure 1.** Estimated survival of reintroduced great bustards. a) solid line represents all 86
546 individuals released between September 2004 - September 2009, and dashed lines are 95%
547 confidence intervals. b) Survivorship separated by sex for 78 individuals (45 females, 33
548 males). Cross-hair ticks indicate censored individuals that were not recovered or resighted
549 after the indicated date during the 365 days.

550

551 **Figure 2.** Fates of captive-reared great bustard juveniles reintroduced to the UK between
552 2004 and 2009. Sample sizes are on the right hand axis.

553

554 **Figure 3.** Predicted population size of UK great bustard reintroduced population through
555 rear-and-release and assuming there is no breeding occurring; 1a- (+) original projected
556 growth before the start of the project (Osborne 2002); 1b- (●) revised model importing 20
557 chicks a year; 1c- (○) revised model importing 40 chicks a year; and (▲) actual population
558 growth from 2004.

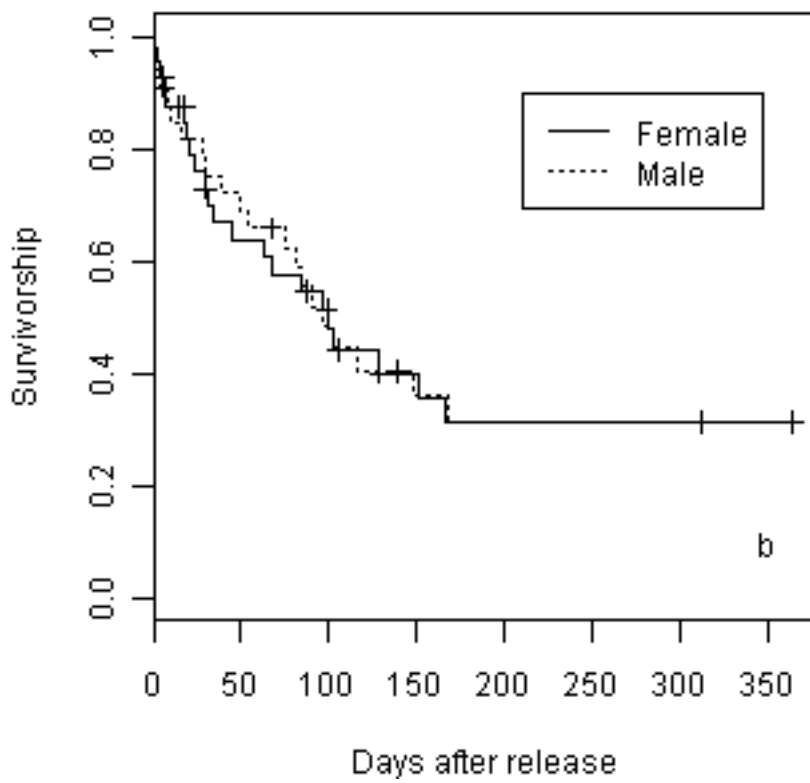
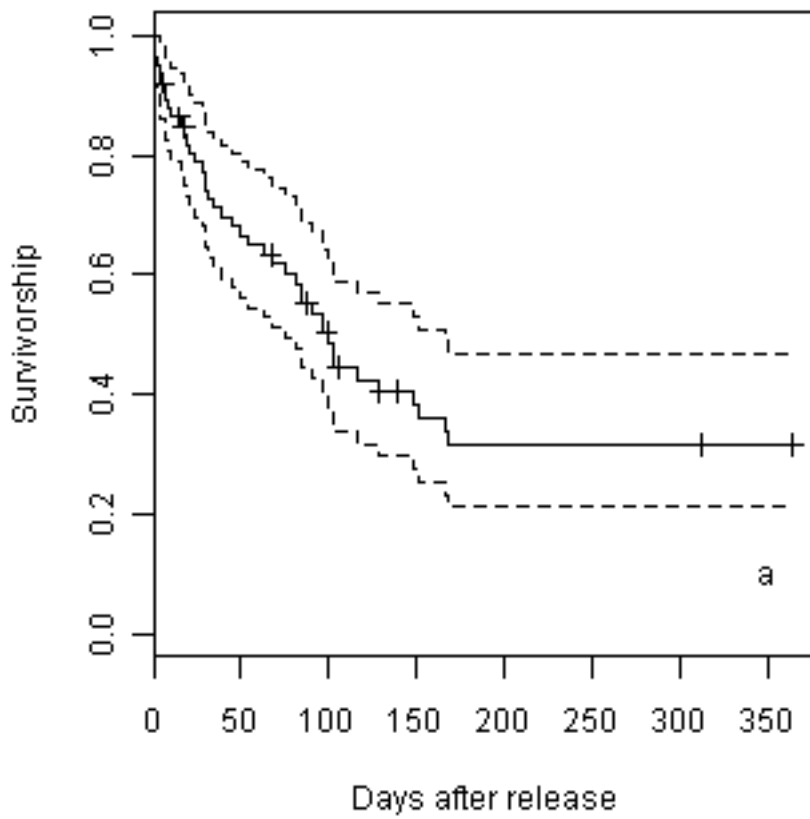
559

560 **Figure 4.** Estimated founder population sizes for great bustards reintroduced to the UK at the
561 end of the 10 year trial period using 12 scenarios of post-release survival and number of birds
562 imported to the UK (see Methods for details). Lines are two standard deviations above and
563 below the estimated mean after 1,000 runs and represent the 95 % probability of attaining a
564 population this size.

565

566

567 Fig. 1



568

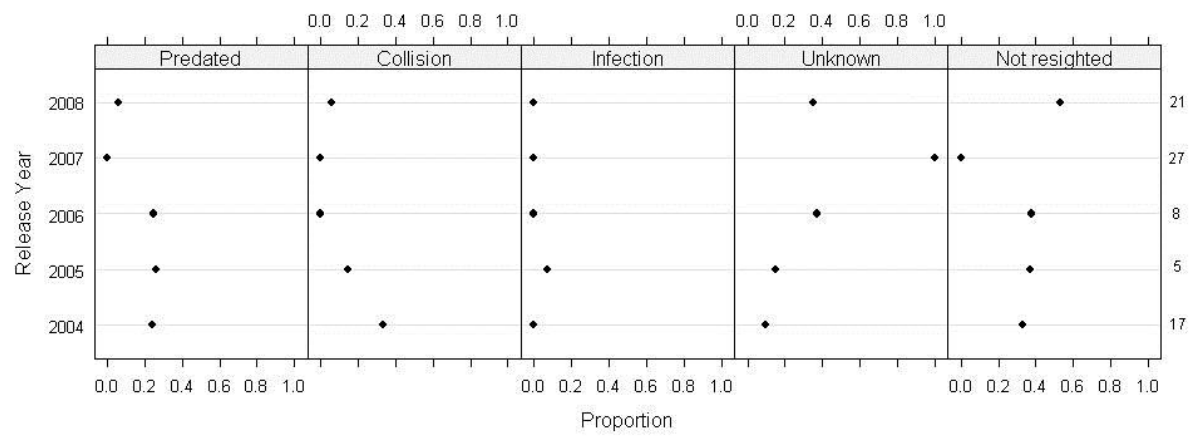
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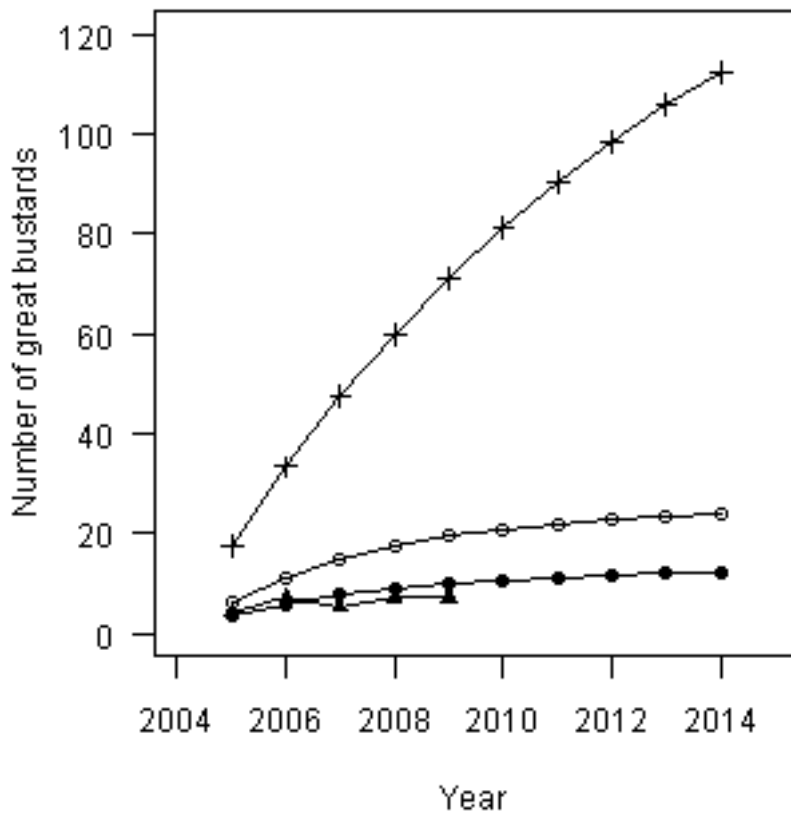
572 Fig 2.

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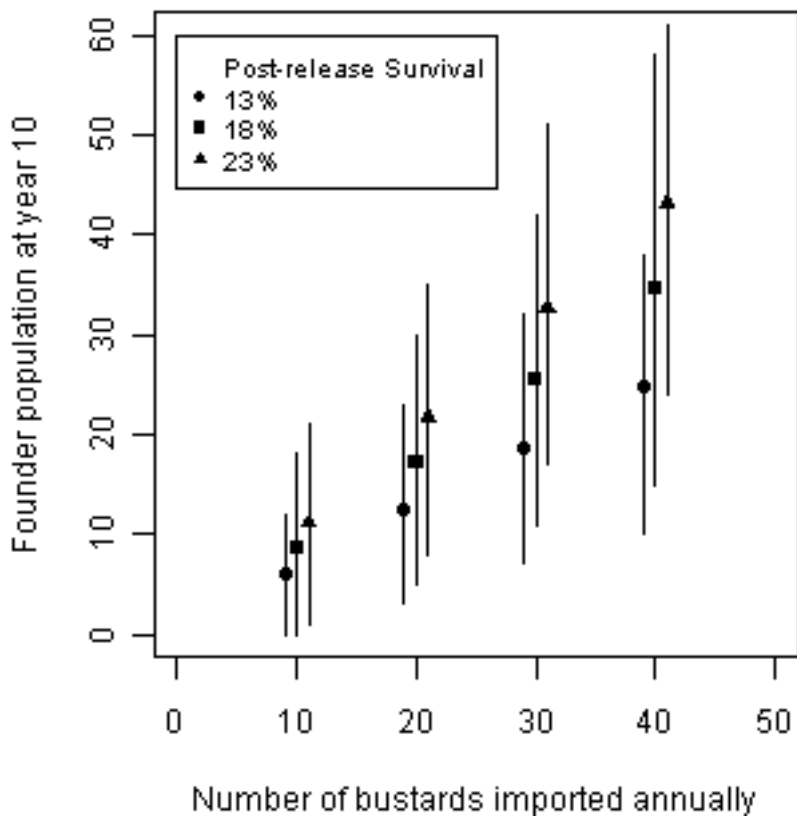
575 Fig. 3



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578 Fig. 4



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