

Population and future range modelling of reintroduced Scottish white-tailed eagles (*Haliaeetus albicilla*)





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COMMISSIONED REPORT

Commissioned Report No. 898

**Population and future range modelling of
reintroduced Scottish white-tailed eagles
(*Haliaeetus albicilla*)**

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COMMISSIONED REPORT

Summary

Population and future range modelling of reintroduced Scottish white-tailed eagles (*Haliaeetus albicilla*)

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Keywords

Re-introduction; sea eagle; persecution; collision risk; conflict; range expansion; raptor.

Background

Re-introductions are increasingly being used in conservation biology as a valuable tool in species recovery programmes. This technique was used to establish a population of white-tailed eagles (*Haliaeetus albicilla*) in Scotland, where the species went extinct in 1917. Three release phases have taken place, of which the first two (1975-1985 and 1993-1998) were on the west coast and the third (2007-2012) on the east coast of Scotland. All three phases have used birds sourced from Norway. In 2014, there were 98 territorial pairs of white-tailed eagles in Scotland, of which 90 were confirmed to have bred. For this report, a conservative approach was used, and only the 90 sites with confirmed nests were considered territories.

Overall, the re-introduction programme has been deemed a conservation success; however, the species may potentially come into conflict with land-use interests, including sheep farming, forestry and renewable energy. Therefore it is important to get a better understanding of how fast the population of white-tailed eagles will increase numerically and where the population will expand into. This is important for both mediating conflicts and for predicting how white-tailed eagles might be affected by other land-uses, such as forestry and renewable energy production (e.g. wind farms). There are also concerns that white-tailed eagles might be victims of persecution, but the impact of such illegal activity on the white-tailed eagle population is currently unknown.

The best available predictions of future population increase used data on breeding success and age-specific survival up to and including 2007. This report presents predicted population growth using estimates of breeding success and survival up to 2014. Additional mortality, potentially caused by illegal killing or collisions with wind turbines, was also incorporated in some modelled scenarios. Finally, models of predicted geographical range expansion in the next 25 years (i.e. up to and including the year 2040) are presented, based on habitat associations and nearest-neighbour distances.

Main findings

- The number of breeding white-tailed eagle pairs has continued to grow almost exponentially, and wild-bred eagles now greatly outnumber released eagles.

- There has been a continued increase in both the proportion of white-tailed eagle nests fledging young (i.e. "breeding success") and the number of chicks fledged per breeding attempt (i.e. "productivity") since the first breeding attempt in 1983. This is probably because the proportion of birds in the population with extensive breeding experience has increased, as breeding performance improves with experience. However, since 2006, the number of chicks fledged per breeding attempt appears to have remained relatively constant at an average of 0.67 chicks fledged per territorial pair.
- When the updated estimates of breeding success and age-specific survival were used in density-independent predictive models, the results suggest that the white-tailed eagle population could continue to grow to over 200 pairs by 2025 and almost 900 pairs by 2040, but obviously the long-term predictions are far less certain than the short-term.
- When modelling the impact of additional mortality, potentially caused by illegal killing and collisions with wind farms, population growth was reduced, but not to the extent of causing a population decline.
- Overall, the results presented here suggest a continued exponential population growth of white-tailed eagles in Scotland in the short-term. However, density-dependence in demographic rates would need to be considered in modelling population growth over the longer term.
- The associations between white-tailed eagle breeding sites and habitat, landscape and topographical variables were explored statistically by comparing the locations of real nest sites and random points in the landscape. The final model suggested that white-tailed eagles were positively associated with length of coastline, area of inland water and the area of forest (all within 1 km from the nest) and also with flat topography (within 3 km from the nest). This model explained 25% of the variation in occupancy status.
- Information on how far away from existing pairs new white-tailed eagle pairs settle was compiled, and it was found that most newly established territories were located between 6 and 10 km from the nearest other active white-tailed eagle nest. By integrating eagle-habitat associations and the nearest-neighbour distances for newly established pairs, future range expansion was modelled spatially. The results suggested that range expansion is most likely to occur mainly along the west coast of Scotland, the Great Glen and in central and eastern Scotland where birds from the east coast release start to breed.
- The geographical range expansion models presented in this report can inform stakeholders of likely white-tailed eagle range expansion in the next 10-25 years, but the exact settlement order is difficult to predict. In addition, occasional long-distance dispersal events could lead to the establishment of new pairs outside the range predicted here. In the long-term, these pairs might result in new population centres, which could expand the geographical range of white-tailed eagles even further.

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Sadly one of the authors, Richard Evans, passed away before this report was published. His passing will be a great loss to eagle conservation as he was an expert on white-tailed eagles and the reintroduction to Scotland having been associated with it for over 20 years.

1. INTRODUCTION

Re-introductions are increasingly being used in conservation biology as a valuable tool in species recovery programmes (e.g. Griffith *et al.*, 1989; Sarrazin & Barbault, 1996; Seddon *et al.*, 2007; IUCN/SSC, 2013). The method has been successfully implemented to aid the recovery and re-establishment of populations of bird species, in particular birds of prey, following local extinction due to historical persecution, habitat loss and pollution (Meretsky *et al.*, 2000; O'Toole *et al.*, 2002; Evans *et al.*, 2009; Smart *et al.* 2010).

Evidence suggests that the white-tailed eagle *Haliaeetus albicilla* was once widespread in Britain and Ireland, both in upland and lowland areas (Evans *et al.*, 2012). However, following a population decline and range contraction this population had been extirpated by 1918 after prolonged human persecution (Love, 2003). From 1975 onwards, releases of birds translocated from the wild in Norway in Norway led to the re-establishment of a breeding population of white-tailed eagles in western Scotland (Love & Ball, 1979; Love, 1983; Evans *et al.*, 2009). The re-introduction programme was originally established by the Nature Conservancy Council, the predecessor of Scottish Natural Heritage (SNH), and in 1980 the Royal Society for the Protection of Birds (RSPB) became involved in the project, which has operated as a partnership between the two organisations since.

There have been three release phases in Scotland, of which the first two took place on the west coast of Scotland. All phases have involved the release of first-year birds that were collected as chicks under licence from Norway and later reared in aviaries in Scotland until the birds reached an age when they would be able to cater for themselves (Love & Ball, 1979; Love, 1983). In phase one (1975-1985), 82 individuals were released on Rum and in phase two (1993-1998) 58 individuals were released (following suggestions in Green *et al.*, 1996) in Wester Ross. To increase geographic range of the species and minimise the risk of extinction due to stochastic effects, a third phase of releases took place on the east coast of Scotland between 2007 and 2012. The location was based on a growing body of evidence from continental Europe that the white-tailed eagle showed strong population growth in lowland and estuarine environments. This phase involved the release of 85 individuals in Fife. In total, 225 individuals were released in Scotland between 1975 and 2012.

The re-establishment of the white-tailed eagle in Scotland has been a conservation success story, with a high rate of population increase, especially in more recent years (Evans *et al.*, 2009), leading to a well established breeding population on the west coast of Scotland (Balmer *et al.*, 2013). The population increased from one to approximately 98 pairs between 1983 and 2014.

Although the secure re-establishment of a self-sustaining, wild-breeding population of white-tailed eagles in Scotland is a success in terms of nature conservation objectives, it has resulted in anxiety and opposition in some land-use sectors, notably farming, forestry, aquaculture and renewable energy. In particular, there has been a long-running concern in the sheep-farming community in the western Highlands and islands of Scotland that the species, or at least some individuals of the species, may predate large numbers of lambs, and that this may have an impact on farming livelihoods. Detailed, area-specific studies have indicated that significant impacts are relatively rare and extremely unlikely to be widespread (Marquiss *et al.*, 2004; Simms *et al.* 2010). Nonetheless, to mitigate the lamb losses that do occur and to encourage active landowner involvement in white-tailed eagle conservation, SNH has operated a voluntary management scheme that pays participating farmers and crofters living within 5 km of active white-tailed eagle nests to undertake various management measures.

Site-specific white-tailed eagle issues have also arisen in relation to aquaculture, forestry and renewable energy. All of these activities can cause disturbance to breeding eagles, and

wind turbines (and other above-ground electricity infrastructure such as cables) pose an additional risk of mortality through collision and/or electrocution. Consequently, the presence of eagles can be a significant constraint on consenting these activities (Kortland *et al.*, 2011). The extirpation of the species in Britain and Ireland demonstrates a vulnerability to persecution. Although white-tailed eagles largely breed outwith areas where illegal killing of raptors is currently prevalent (based on locations of recoveries of illegally killed birds), the risk of illegal killing seems likely to increase as the species' range expands. For all of these reasons, it would be helpful to have as robust an estimate as possible of the likely rates of population increase and range expansion of the species in Scotland. This would enable the identification of areas with a high likelihood of near-future occupation alongside estimates of future population size. This study models the potential change in the number of breeding pairs and the likely distribution of white-tailed eagles in Scotland over the next 25 years, using detailed long-term empirical data on the demography of the existing Scottish population as a starting point. Before this report, the most recent analyses of a similar kind used data up to and including 2007 (Evans *et al.*, 2009, 2010. Population modelling carried out for this report therefore takes into account how demographic rates (i.e. breeding performance and survival) have changed since 2007.

Currently there is no strong evidence of density-dependence in terms of the number of fledglings produced per territorial pair with known breeding outcome (Evans *et al.*, 2009), and for that reason density-dependence has not been modelled in detail. However, the report contains tests to explore whether breeding performance has declined with density in regions with the highest recorded density (i.e. on Skye and Mull).

Specifically, this study aimed at answering the following questions:

1. How has the white-tailed eagle population size changed since 2007?
2. What was the white-tailed eagle population composition, in terms of proportion of wild-bred and released individuals, as well as the age-structure, in 2013?
3. Have the probability of a successful breeding outcome (i.e. successful vs. failed) and the mean number of fledglings produced per territorial pair with known breeding outcome changed since 2007?
4. Have the age-specific and origin-specific survival rates changed since 2007?
5. What is the evidence for density-dependent breeding success and/or population growth nationally, and in the two sub-populations with the highest white-tailed eagle density (i.e. Mull and Skye)?
6. Assuming no limitation in suitable habitat and no density-dependent population regulation, what is the expected population size (i.e. number of territorial white-tailed eagle pairs) in the years 2025 and 2040?
7. How might density-dependence affect white-tailed eagle population growth given limitations, for example, in nest sites, habitat or prey availability (i.e. under scenarios with different carrying capacity)?
8. What would the effects of increased mortality, potentially caused by collisions with wind turbines and illegal killing, be on white-tailed eagle population growth and population size in the years 2025 and 2040?
9. What are the current associations between habitat and territorial white-tailed eagles?
10. How are nest locations distributed in relation to each other?
11. Given the habitat-eagle associations and current spatial distribution, in what geographical areas is range expansion most likely to occur with the increased population sizes predicted by modelled population growth?

2. METHODS

2.1 Field methods

Monitoring of white-tailed eagles has been conducted annually since 1975, with particular emphasis on detecting territorial pairs and determining the outcome of their breeding attempts. Between 1975 and 2013, all known territories were visited by professional fieldworkers and volunteers and searched for evidence of territorial birds and nest-building. Suitable habitat not previously occupied by breeding birds was also searched, with the objective of locating all occupied territories each year. The first year a breeding attempt occurred was in 1983, but this attempt failed. The first successful breeding attempt was recorded in 1985. The proportion of successful breeding attempts (i.e. where at least one chick fledged from a nest) and the number of chicks fledged per territorial pair have been recorded annually since breeding restarted and entered into a bespoke database. Active nests were closely monitored, with visits at roughly weekly intervals until young had fledged or the breeding attempt had failed. However, in more recent years, the monitoring effort of breeding attempts has been less intensive, but still managed to capture breeding outcome and the number of fledglings. Care was taken not to disturb the pair during sensitive periods, such as in the early phases of incubation (Hardey *et al.*, 2013). Fully-feathered large young aged 10 weeks or older were assumed to have fledged. As the population has grown, intensive monitoring of all nests by professional staff could not be maintained and survey work is shared between RSPB and experienced volunteers, many of them members of the Scottish Raptor Study Group. The annual survey covers a large proportion of the known geographical range, but the probability that a small number of newly established territories are missed may have increased in the last few years as the white-tailed eagle population has grown.

All released birds and as many wild-bred young as possible were marked with individually numbered BTO metal rings. Birds released during the earlier stages of phase one were also marked with colour rings, but due to high levels of ring loss, all birds released on the west coast of Scotland from 1982 onwards and those released in 2007 and between 2009 and 2012 on the east coast (i.e. release phases two and three) were marked with patagial wing tags. Because of changes to the licensing arrangements for 2008, birds released on the east coast in that year were colour-ringed only. Many wild-bred young were also fitted with wing tags until 2007. The wing tags were colour-coded by year, so that all birds in a cohort were tagged with the same basic colour. Contrasting alpha-numerical marks on each tag identified individual birds within each cohort. Colours were not re-used for at least five years, when birds of the previous cohort tagged with that colour would be in adult plumage. Some tag loss occurred, but the rates and ages at which tag loss occurred were such that it could be assumed that tagged young retained adequate marks for them to be assigned correctly to a cohort when settling on territory. It also meant that unmarked birds settling on territory could be assumed to have never been marked and so were wild-bred. Due to older birds losing wing tags, from 2008 onward wild-bred young were fitted with improved metal colour rings marked with unique alpha-numerical codes. Colour-rings used since 2008 were colour-coded by year until 2011, meaning that cohorts and individuals could still be identified for all potential breeding pairs in this study (only a few birds fledged up to 2009 are likely to have recruited to breed by 2013). Numbers of re-sightings of individuals fitted with colour-rings are lower than those fitted with wing tags, probably because colour-rings are more difficult to read in sufficient detail to allow identification of individuals than wing tags. This may be particularly true for non-breeding birds, which do not regularly visit the same site (i.e. a nest).

Throughout the re-introduction and re-establishment of white-tailed eagles in Scotland, records have been kept of re-sightings of marked individuals throughout the year, including non-breeding sub-adults. In addition, white-tailed eagles found dead have been individually identified by their metal rings, and these mortalities have been recorded throughout the

period 1975-2014. The database of re-sightings and dead recoveries has been used to estimate age and cohort-specific (i.e. released and wild-bred) survival rates (Evans *et al.*, 2009). Re-sighting records came from professional staff, volunteers and members of the public. Re-sighting effort outside the breeding season has gradually been reduced since 2011, which for the purpose of the current study will not affect estimated survival rates.

Through moult, white-tailed eagles change their plumage with age and generally do not attain full adult plumage until five or more years of age (Struwe-Juhl & Schmidt, 2003; Hardey *et al.*, 2013). Adult plumage can also vary, so that some individuals can be identified between years on plumage characteristics alone. Monitoring effort at territories was such that when an adult individual lost its tags it could be assumed to have remained on territory until it was positively identified as being replaced by a new adult (based on age, plumage or tags/rings of the replacement bird).

2.2 Statistical modelling

In general, data collected between 1975 and 2014 were used in all analyses. However, for some analyses, a shorter time period was used, as indicated in the following sections. In addition, for the majority of years, only data from the two initial release phases from the west coast population were used, because the first breeding attempt on the east coast of Scotland did not occur until 2013.

2.2.1 Number of breeding pairs

The number of breeding pairs of white-tailed eagles each year was based on data provided by professional staff and volunteers trained by RSPB. For almost all years, there was little ambiguity in determining the total number of territorial pairs. However, in the last few years, the proportion of territories monitored by volunteers has increased, and as a consequence, there were a small number of territories in 2014 where the volunteer could not confirm whether a seemingly settled pair had bred or not. Therefore, in 2014, the data suggested that there were 98 settled pairs, but for eight pairs the presence of a nest could not be confirmed. In this report, a conservative approach has been taken, and only the 90 pairs with a confirmed nest were used as the figure of the number of breeding pairs in 2014.

2.2.2 Population composition

Due to changes in monitoring in 2014, less information about the identity of white-tailed eagles was available for this year. This made it more difficult to determine the age and origin of birds breeding in the population in 2014. In general, there is very low annual turnover of birds at individual territories, with an average (\pm SE) of 1.41 ± 0.33 (range: 0-7) territories per year changing at least one individual. There were no indications that the turnover was higher between 2013 and 2014 than for previous years. However, the smaller number of submitted re-sightings in 2014 indicated that it would be better to use data up to and including 2013 when estimating the population composition in terms of age structure and origin (i.e. released or wild-bred) as well as age- and origin-specific survival rates.

2.2.3 Breeding success

In this report, the phrase "breeding performance" refers to the overall success of a pair. The breeding performance is comprised of "breeding outcome", which is the result of a breeding attempt (i.e. whether a nest was successful or failed), and "productivity", which is the number of fledglings produced per territorial pair where the outcome of the breeding attempt was known. The latter could take the value of zero, one, two and three fledglings produced per territorial pair. This is a slight difference compared with Evans *et al.* (2009), who reported the number of fledglings produced per territorial pair (i.e. including pairs where the final outcome was not known, so the minimum number of young produced was spread across all pairs).

Therefore, this report reports a slightly higher productivity than some figures in Evans *et al.* (2009). The number of fledglings produced per *successful* breeding attempt was used in the Population Viability Analysis (PVA; section 2.2.6). For these analyses, data from the years 1983 to 2014 were used.

To test whether breeding outcome (i.e. successful vs. failed) had changed since the start of the study, a binomial Generalised Linear Mixed Model (GLMM) with a logit error structure was fitted, with breeding outcome (i.e. 0 = failed breeding and 1 = successful breeding) as a response variable and year as an explanatory variable. Pair identity (i.e. the male/female parent combination) and territory were included as random effects to control for the effect of repeat sampling of the same pairs and territories. To examine if the mean number of fledglings produced per territorial pair had changed since the start of the study, a model with multinomial errors (i.e. specifying zero, one, two or three chicks) with a cumulative logit error structure was used with the same explanatory variable and random effects as for the binomial model described above.

The effects of breeding experience (number of years a pair had bred together) and origin of the pair (released, wild or mixed) on breeding outcome and the number of chicks fledged were studied by fitting a GLMM with binomial and cumulative logit error structures, respectively. Again, pair identity and territory were included as random factors to control for repeat sampling of these factors. Both breeding outcome and the number of fledglings produced by a pair were expected to be non-linear. Specifically, it was assumed that reproductive performance would increase with increased breeding experience, but could reach a maximum set by evolutionary and environmental constraints. Late in life, senescence might set in and breeding success might decrease (as shown in Evans *et al.*, 2009). For this reason, breeding experience was included both as a linear and quadratic term as explanatory variables.

Age at first breeding was estimated based on the subset of birds that were wing-tagged or colour-ringed when their first recorded breeding attempt was made.

2.2.4 Estimating survival rates

Following Evans *et al.* (2009), data between 1986/7 to 2013/4 were coded for all individually marked birds according to whether a bird was seen (1) or not seen (0) in a year defined as March to February, to correspond with the start of the breeding season. Birds were grouped by origin (released or wild-bred). For the released group, only phase two birds marked with wing tags were included, as phase one birds showed higher levels of tag loss, and thus had poorer resighting rates. For wild birds, only marked birds up to and including the 2007 cohort contributed to survival rate estimates, because wild-bred young from 2008 onwards were colour-ringed rather than wing-tagged, leading to lower resighting rates for pre-breeding individuals. (It is hoped that better long-term retention of colour rings as opposed to wing tags will eventually allow better estimates of adult survival rates, identified by Evans *et al.* (2009) as the main demographic driver of population change). The birds released on the east coast were not included in the survival analysis. Any marked birds that were known to have died before fledging were excluded, as was a single individual where there was confusion over its identity.

Survival was estimated using Program MARK (White & Burnham, 1999), which accounts for imperfect detection between years, that would cause survival be underestimated. All models were run through program R (R Development Core Team, 2011), using the RMark package (Laake, 2013), which allows models to be specified within R and run within MARK, reducing the likelihood of errors in model specification. As both continental and Scottish white-tailed eagles start to breed on average at five years of age, and both survival and site fidelity

increases until this point (Struwe-Juhl & Grünkorn, 2007; Evans *et al.*, 2009; Whitfield *et al.*, 2009a, 2009b) it was likely that survival and resighting rates would increase with age.

In order to determine the age structure that best described the age-related changes in survival and resighting rates, we tested and compared three possible age transition structures to model age-specific survival and resighting rates (Table 1). In the first age transition structure, models used five age transitions, with the transitions zero to one, one to two, two to three and three to four years as four separate classes and all transitions thereafter (i.e. birds older than four years) as another class (as in Evans *et al.*, 2009). The second age transition structure used six age transitions, with the transitions zero to one, one to two, two to three, three to four and four to five specified as separate classes and all transitions thereafter (i.e. birds older than 5 years) as another class (Table 1). The inclusion of the last age class was only possible because of an increased sample size of older birds since the study by Evans *et al.* (2009) was published. Finally, a third age transition structure used four transitions, with all age transitions zero to three as a single class, and age transitions three to four and four to five as separate classes, as well as all transitions thereafter (i.e. older than five years) as a final class (Table 1). The rationale behind this age transition structure is based on the results presented in Evans *et al.* (2009), which indicated that ages zero to three had similar survival and resighting rates.

All possible combinations of model parameters were run, and in order to determine whether survival rates differed between wild-bred and released birds, this included models with and without terms for age and origin (as additive terms and as interactions with age effects). A bootstrap goodness-of-fit test (GOF) was used to estimate variance inflation (\hat{c}), which was used to adjust the Akaike Information Criterion (AIC) before model selection to Quasi-Akaike Information Criterion adjusted for small sample sizes (QAICc). Models were sorted by QAICc (adjusted by \hat{c}), and the model with the best fit was determined, based on the differences in QAICc between the top-ranking model and other models (i.e. delta-QAICc or Δ -QAICc). Models with a Δ -QAICc of less than two units were selected on which to base estimates of survival (Burnham & Anderson 2002). Generally, there is broad agreement that a model with a Δ AIC (or any of the adjustments such as QAIC and QAICc) of greater than two units is regarded as having a better support than competing models.

Table 1. Age-transition structures used to compare which scenario that best described age-related changes in survival and resighting probability. Dark areas indicate separate age transitions included in each scenario. Contiguous dark areas indicate that the survival (and resighting) rate of birds in different age-transitions were pooled into one class. In each scenario, the same age-transition structure was used for both resighting and survival probability.

Scenario	Age-transition						No. of estimated parameters
	0-1	1-2	2-3	3-4	4-5	>5	
1							5
2							6
3							4

2.2.5 Evidence of density-dependent population limitation

The potential that breeding success and survival (and ultimately population growth) would decrease with increasing density of the white-tailed eagle population was investigated nationally and regionally. The two regions with highest breeding density of white-tailed eagles in Scotland in 2014 were the islands of Mull and Skye, with 21 and 18 territorial pairs, respectively (i.e. densities of 0.024 and 0.011 pairs per km²; Table 2). Although these densities might be regarded as high, there are other areas, notably in Norway, where the

density is much higher (Table 2). It was assumed that if density-dependent population regulation was already occurring in Scotland, it would be evident on Mull and Skye.

First, to study if the number of chicks produced per breeding attempt had declined over time, whilst controlling for the effects of breeding experience and origin, the linear and quadratic terms of the variable "number of years since establishment" (i.e. of breeding pairs on each island) were included as explanatory variables in a multinomial GLMM. Second, a more direct test was carried out, in which a similar GLMM was fitted, with the linear and quadratic terms of the variable "number of pairs" (i.e. on each island separately) replaced the linear and quadratic terms of "number years since establishment". The models of reproductive performance, described above, were re-fitted with the data from Mull and Skye separately. Finally, the number of territorial pairs on Mull and Skye was plotted from 1982 until 2014 to visually inspect whether the population growth had slowed down with time since establishment and with increased density.

Table 2. The density of white-tailed eagles in various "high density" locations across the species geographical range. The years of the studies and evidence of density-dependent population limitation are also indicated.

Location	Year	Evidence of density-dependence	Density (pairs/km ²)	Reference
Danube Delta, Romania	2009-2011	No	0.005	(Sandor <i>et al.</i> 2015)
Schleswig-Holstein, Germany	1947-2008	No	0.014	(Krüger <i>et al.</i> 2010, 2012)
Smøla, Norway	2009	No	0.180	(Dahl <i>et al.</i> 2012)
Mull, Scotland	2014	No	0.024	This study
Skye, Scotland	2014	No	0.011	This study

The potential effects of limited resources (i.e. prey availability, suitable foraging habitat and suitable nest sites) on national population growth were also investigated by setting a carrying capacity of the entire Scottish population in Vortex. These models assumed that if the population size exceeded the carrying capacity, additional mortality was imposed equally across all age and sex classes in order to reduce the population back to this upper limit (Lacy *et al.*, 2005).

Carrying capacity of the population was set at three arbitrary levels: 2,000, 3,000 and 4,000 individuals. Models were then run using the demographic rates as described above with no additional mortality (i.e. "harvest" was set to 0 in Vortex). The result of each level of carrying capacity was then plotted to visualise the effect it had on the estimated number of breeding pairs in the population between 2014 and 2040.

2.2.6 Predicting future population growth

The Population Viability Analysis (PVA) software Vortex (Lacy *et al.*, 2005) was used to model population growth covering the period over which there were observed estimates of the number of territorial pairs (1975-2014). Population models were also extended into the future, over 10 (2025) and 25 (2040) years. Observed population growth was compared with hypothetical population trajectories based on a number of assumptions. First, modelling was conducted so that released and wild-bred individuals, with their specific demographic characteristics, were recognised but with free mixing of individuals between the two groups. Second, the models assumed age-specific survival (\pm SE), which combined the demographic rates of both wild and released populations. The number of released birds in each year was included, adjusted for first year mortality, from all three phases of release (the two west

coast releases and the single east coast release) as supplementary individuals. The initial model used 999 iterations, assuming no density-dependent limitation of carrying capacity (K) and no harvest (e.g. illegal killing). Modelling incorporated demographic stochasticity in reproductive rates by incorporating the mean (\pm SD; data from 1975-2014) percentage of territorial pairs laying an egg in each year (identified via direct nest visits, but also from behaviour of the parental birds). This is a change in approach from the study by Evans *et al.* (2009) which assumed that 100% of the pairs were breeding. In addition, the mean percentage of territorial breeding pairs fledging at least one young (data from 1983-2014) was incorporated into the models. Finally, 10-year means (\pm SD) of the number of fledglings per successful breeding attempt were used. Population size was the predicted number of adult territorial pairs, based on the mean minimum number of males or female (whichever sex had the lowest predicted number of individuals) in each year across all iterations.

2.2.7 Exploring the effects of additional mortality on the number of breeding pairs nationally

The population modelling was expanded by running scenarios in Vortex aimed at reflecting additional mortality caused by collisions with wind turbines and illegal killing. In terms of the modelling set-up and the effect on population growth, the cause of death does not matter (i.e. a dead bird will not contribute to future population growth, regardless of how it died). Therefore, this report presents scenarios with varying numbers of white-tailed eagles removed annually from 2015 onwards (i.e. using both the established west coast population and the emerging population originating from the east of Scotland release phase). This mortality is additional to the background mortality estimated for the Scottish west coast population of white-tailed eagles. The background mortality includes natural mortality, e.g. from disease and starvation, but also undetected illegal killing and collisions with man-made objects.

The models presented here used scenarios in which between two and 14 birds were removed annually. The rationale for these figures comes from various sources. For example, in some areas, white-tailed eagles have been reported to be killed in collisions with wind turbines. One study in a Norwegian area with overlap between high densities of breeding white-tailed eagles and a wind farm found that, on average, 7.8 individuals died annually in turbine collisions, with 53% of the birds being adults (May *et al.*, 2010; May *et al.*, 2013). Similarly, between 2002 and 2015, 108 white-tailed eagles have been reported killed in collisions with wind turbines in Germany¹. However, only one white-tailed eagle has been reported killed in a collision with a wind turbine in Scotland. Similarly, relatively few white-tailed eagles have been confirmed to be persecuted in Scotland, potentially because their current breeding range has limited overlap with the driven grouse moor areas that have been associated with high historical levels of illegal killing of raptors (Etheridge *et al.*, 1997; Green & Etheridge, 1999; Whitfield *et al.*, 2003, 2004b; Smart *et al.*, 2010). For golden eagles, which occupy some regions with high historical levels of illegal killing, the annual number of persecuted individuals is higher. For example, using data on golden eagle illegal killing events and turnover of individuals in territories, Whitfield *et al.* (2004a) estimated that approximately 3-5% of the adult golden eagles were illegally killed annually (i.e. between 13 and 21 adult golden eagles, based on an estimated 420 pairs in 2003). Thus, the modelled figures for additional mortality of white-tailed eagles are within the range of illegal killing levels for another eagle species in Scotland. With no information on age and sex-biased collision and rates of illegal killing in white-tailed eagles, mortalities were evenly split between males and females and between two-year old and adult eagles in the models presented in this study. The choice of two-year old birds was based on the fact that the most extensive movements of white-tailed eagles occurred in the first two years after fledging

¹ Tobias Dürr maintains the European Collision reporting spreadsheet (available at <http://www.lugv.brandenburg.de/cms/detail.php/bb1.c.312579.de>) and it shows the number of fatalities of many bird species across Europe. Numbers reported here were correct on 02/09/2015.

(Whitfield *et al.*, 2009b). Thus, the wide-ranging movements could put one to two year old white-tailed eagles at higher risk of exposure to collisions and illegal killing than birds aged three to five years.

Finally, a set of modelled scenarios included the effect of removing a number of birds in direct proportion to the national population size of white-tailed eagles in Scotland (i.e. 0.5%, 1.0% and 2.0% of the population). The rationale behind this scenario is that progressively more birds might be at risk of colliding with wind turbines and falling victims of illegal killing when the population size increases. However, as the future relative importance of mortality caused by wind farms and illegal killing is unknown, e.g. the location of consented onshore wind farms and changes in levels of illegal killing, it was impossible to predict the temporal and spatial variation in these mortality factors.

2.2.8 Defining associations between habitat and breeding white-tailed eagles

Data processing

White-tailed eagle territories often contain more than one nest site (Hardey *et al.*, 2013). Thus, all white-tailed eagle nest locations between 1983 and 2014 ($N_{\text{nests}}=335$) were mapped within territories ($N_{\text{territories}}=104$). To assess if the area surrounding the used nest locations differed significantly from the available habitat in the surrounding landscape, an equivalent number of random points were generated ($N_{\text{random}}=335$). This was done in a spatially stratified manner in relation to both the regions where white-tailed eagles had nested between 1983 and 2014 (Fig. 1), and the clustering of breeding locations within territories.

The regions followed the same boundaries as in Evans *et al.* (2010), with the addition of an "Eastern & northern" region to cover areas used by white-tailed eagles more recently in Scotland (Fig. 1). Using ArcGIS 10.2 (ESRI, 2011), each region was then populated with the same number of random points as the number of historical territories (i.e. territories currently in use or used in the past; $N = 104$), representing random territory centres. These were set to be at least 1 km from each other, which is the minimum distance so far recorded in Scotland between occupied nests in different territories within years. The area within each random territory centre that could be used to allocate random nest sites was defined by a buffer with a 9.3 km radius, representing the 95th percentile of all within-territory distances between real nest locations across years. Each set of regional random territories was then populated, at random, with the same number of random nest locations that occurred in that region. The minimum distance between random nest locations within territories was set to 100 m (based on the observed minimum distance the mapping resolution allowed). Each random nest location was assigned an individual identity and a random territory identity. During this process, a total of 14 random territories were assigned to be unoccupied. This represents the number of real white-tailed eagle territories that were unoccupied in 2014. This meant that 90 random territories, containing the 335 random nest locations across the four regions in Scotland were created (Fig. 1).

Each real and random nest location was buffered with circles of 1, 2 and 3 km radii centred on the nest location. The area of different land covers were then calculated for each buffer, using the British Land Cover Map 2000 (LCM, 2000; Centre for Ecology and Hydrology, 2000) and the national forest inventory 2013² as underlying land cover maps. We were only interested in land cover characteristics that previous research (e.g. Radović & Mikusca, 2009; Evans *et al.*, 2010; Krone *et al.*, 2013; Sandor *et al.*, 2015) had identified as being significantly associated with white-tailed eagle occupancy rates, that is the area of, and the distance to, sea, freshwater and woodland. These variables were all positively associated with white-tailed occupancy, but negative variables would also have been included if there was evidence that white-tailed eagles avoided any habitat feature. In addition, distance to

² © Crown copyright and database right 2013 Ordnance Survey [100021242]

the coast, the nearest large (>1 km²) inland water body and the nearest woodland were calculated. Finally, the length of coastline within each buffer was calculated. The topography, specifically the mean and coefficient of variation (CV) of altitude, was also calculated within each buffer, using the OS 50 m terrain map.

Regions

- Western and Small Isles
- Skye and Rosshire
- Mull and south west
- Eastern and northern

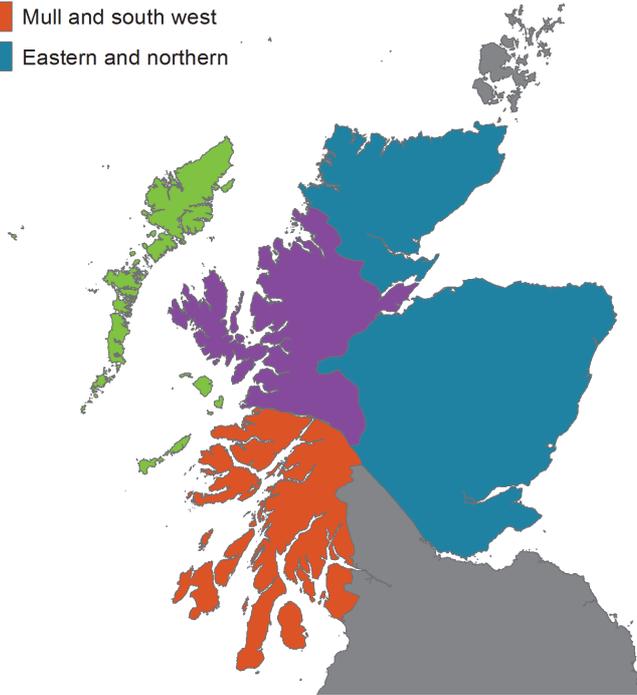


Figure 1. Map of regions across Scotland in which white-tailed eagles have nested between 1983 and 2014. These regions were used to generate the same number of randomly located points (i.e. random nest sites) within randomly located territories as the number of real nest sites and territories in each region. The number of real and random nest sites (and territories) by region was: Western and Small Isles: 96 (30), Skye & Ross-shire: 114 (32), Mull and south west: 117 (36) and Eastern and Northern: 8 (6).

Data analysis

Assessments of associations between white-tailed eagles and habitat were done using binomial Generalised Linear Models³ (GLMs). To minimise the risk of pseudo-replication, the mean value of each explanatory variable was calculated for each real and random territory (i.e. using the real and random nest sites within each territory). Thus, "territory" and not "nest site" was the replicate in these analyses. Specifically, binomial GLMs with a logit error structure were fitted with territory type (i.e. "real"=1 and "random"=0) as the response variables and the mean per territory of land-cover types, distances (coastline length, to

³ Data from white-tailed eagle nests within the same territory were likely to be highly similar in terms of habitat and topographical characteristics. Thus, when testing for associations between habitat and nest sites, it is recommended to include territory identity as a random factor, which means that GLMMs should be used (Zuur *et al.*, 2009. *Mixed effects models in ecology with R*. Springer, New York.). However, binomial GLMMs with territory identity as a random factor left too little variation to be explained by the explanatory variables, causing non-convergence. Therefore, GLMs (i.e. without territory identity as a random factor) and mean values for the explanatory variables for each real and random territory had to be used.

forestry, large freshwater bodies and nearest active nest) and topographical features as explanatory variables were fitted to explore the strength of association of white-tailed eagle nests with certain land-cover and landscape characteristics. Separate models were fitted for the 1, 2 and 3 km scales. The mean distances to the sea, nearest large freshwater body and woodland were obviously not scale-dependent, so the territories had the same values for these variables regardless of scale.

All scale-specific models were checked for collinearity by calculating the Variance Inflation Factors (VIF) of all covariates in each model. If none of these were above five, it was assumed there was no strong collinearity among variables (Zuur *et al.*, 2009). If a variable had a VIF above five, the GLM was re-fitted without this variable and new VIFs were calculated until all remaining variables had a VIF below five. Models were then simplified via backward selection, removing non-significant variables stepwise and testing for changes in significance at each stage. Variables were removed until further eliminations significantly reduced the model fit. Once models of significant land-use, distances and topographical variables had been obtained, the AICs of the final models at each spatial scale were compared to determine which spatial scale best described occupancy by white-tailed eagles (Burnham & Anderson, 2002). The scale with the lowest AIC (i.e. best model fit) was carried forward to be used as a baseline model predicting occupancy by white-tailed eagles. Any additional variables that were significant at other spatial scales were then added to the baseline model. Backward selection was then re-run on this mix of spatial scales to produce a final predicative multi-scale model of white-tailed eagle occupancy.

Assessment of model performance

In order to assess the effectiveness of the multi-scale model in predicting white-tailed eagle presence and absence in individual 1 km² squares, predicted values from the model were used in a Receiver Operating Characteristics (ROC) analysis (see Guénette & Villard (2004) for a worked example), using package pROC in R (Robin *et al.*, 2011). The resulting Area Under the Curve (AUC; \pm 95% confidence intervals) was used as an estimate of model performance. A value of 1.0 would represent a perfect model (i.e. perfect discrimination of presence and absence) and a value of 0.5 would indicate no significant difference in discrimination between the two events.

Applicability of white-tailed eagle habitat associations outside the core breeding range

The vast majority of white-tailed eagle nest location data used in this study came from pairs nesting on the west coast of Scotland (the "core breeding area"). The observed habitat associations from the core breeding area may not be the same elsewhere in Scotland (e.g. in areas that the white-tailed eagles are likely to expand into in the next 10-25 years; see Results). Indeed, associations between white-tailed eagles and habitats from other parts in Europe suggest that the species shows a very plastic set of habitat preferences (Radović & Mikusca, 2009; Krüger *et al.*, 2010; May *et al.*, 2013; Sandor *et al.*, 2015). Thus, to determine how applicable the final eagle-habitat association model might be to other parts of Scotland, it was important to assess whether the range of values of the different habitat variables from 1 km squares within the core breeding area were similar to values of the habitat variables from 1 km squares outside the core breeding area.

This was done by calculating the fifth and 95th percentile for each of the four habitat variables in the final eagle-habitat association model in the core breeding area (CV in altitude, area of forest cover, area of inland water and length of coastline). Outside the core breeding area, the values of the four habitat variables were assessed separately for each 1 km square. Squares were assigned a value of 1 if a habitat variable had a value outside this 90% range of the values in the core breeding area and a value of 0 if it had a value within this range. For each 1 km square outside the core breeding area, an index of how many variables had values outside the 90% range, and for which the final eagle-habitat association

model would have limited ability to predict white-tailed eagle presence, was calculated using the following formula:

$$a = \sum_{i=1}^4 x$$

Equation 1.

where a is the total number of variables with values outside the 90% range observed in the core breeding area and x is the binomial value (0 or 1) of whether the i^{th} variable (of four habitat/topographical variables) had a value outside the 90% of values observed in the core breeding area.

For example, this means that where $a = 0$, all variables in the final model had values that fell within the 90% range of values in the west coast core breeding area. For $a = 1$, one variable fell outside the 90% range of values in the west coast core breeding area. For $a = 4$, all four variables fell outside the 90% range of values observed in the west coast core breeding area, and the final model presented in this report would be entirely non-applicable to that 1km square.

2.2.9 *New territories in relation to distance to existing nest locations*

Using ArcGIS, for the period 1994-2014, the distance between the nest location of each new breeding pair and the nearest active nest of an established pair of white-tailed eagles was calculated. A frequency distribution was then plotted with all distances between new nests and their nearest established neighbour. The probability of different distance bands being occupied, based on proximity to an established nest, could be calculated from the frequency in each band divided by the total number of new pairs ($N_{\text{new territories}}=92$).

2.2.10 *Predicting future white-tailed eagle distributions*

The results of the habitat association models suggested that white-tailed eagles were mainly linked with land-cover features measured within a circle with a radius of 1 km, but also with the coefficient of variation (CV) in altitude within 3 km of the nest (see Results). In addition, the frequency distribution of distances between newly established territories and nearest other active white-tailed eagle nest suggested that new nests were not established at random in relation to other active nests. Thus, by combining these results and land-cover and topographical maps, the probability of each 1x1 km square in the Ordnance Survey (OS) grid being occupied in the future as follows:

$$\text{Probability of occupancy} = \text{Land-cover Probability} * \text{Distance Probability} \quad \textbf{Equation 2.}$$

Thus, the relevant variables (see Table 10) were calculated for each 1x1 km square in the OS grid in Scotland and northern England. The land-cover variables inland water and woodland cover were taken from the summary LCM 2007 data (Morton *et al.*, 2011). The length of coastline within each 1x1 km square was calculated by intersecting the 1 km OS grid with the coastline of the UK in ArcGIS. Distances to nearest large freshwater body and woodland were calculated in ArcGIS using the LCM 2007 data. In addition, the coefficient of variation of altitude was calculated within a radius of 3 km of the centroid of each 1x1 km OS square. Altitude data were derived from OS 50 m terrain raster data, and the mean and standard deviation (SD) was calculated in ArcGIS. The coefficient of variation for each buffer was derived by the formula:

$$\text{CV} = \text{Standard Deviation} / \text{Mean} \quad \textbf{Equation 3.}$$

Using these habitat values for each 1x1 km square in a new data set, the "predict" function in R was used to generate predicted values of habitat probability. This used the model coefficients from the final model of habitat preferences and predicted on the scale of the response (i.e. a probability from 0 to 1.0). These predicted values were used to produce a map of habitat suitability, as well as feeding into the model of predicted range expansion. Given that the predictive power of the land-cover association model was low (i.e. the land-cover model explained only 25% of variance in the data), probability of occupancy in each 1x1 km OS grid square based on land-cover values were converted into probability bands rather than using exact values. This was done by grouping probabilities at three resolutions bands of 20% and 50% bins. For example, using 20% resolution bands, all squares with a land-cover probability >0.80 would be assigned a value of 1.0; all squares with a land-cover probability between 0.61 and 0.80 would be assigned a value of 0.80 and so on. Equivalent grouping at the 50% resolution band meant that all squares with a land-cover probability between 0.51 and 1.0 would be assigned a value of 1.0, and all other squares would be assigned a value of 0.50 (and in practice never selected in the models). This produced two different scenarios of range expansion, effectively varying the strength of habitat associations.

The probability of a 1x1 km square being occupied in relation to the distance to nearest active white-tailed eagle nest was calculated by measuring the distance from every 1x1 km square to the nearest occupied 1x1 km square and then assigning each square to a distance band with an associated probability of being occupied, based on the probability distance function (as described above in section 2.2.9).

To predict the areas that white-tailed eagle pairs would be more likely to move into, the predicted population size based on the Vortex models was used to populate 1x1 km squares year-by-year from 2014 until 2040. In each year, the number of "new" pairs (n) added to the previous year's population were assigned to the top ranking n 1x1 km grid squares with the highest occupancy probability (see Equation 2). Because habitat and distance probabilities were in bands, equal ranking between squares occurred. Therefore, before squares were ranked, they were assigned a random integer number and they were sorted by the probability of being occupied and the random number. New random integers were generated during each iteration of the model ($N_{\text{iterations}}=100$), so that each square received a different integer in every year and iteration of the model. This meant that in each year of a full 25-year sequence, squares were sorted slightly differently, which produced variation in which squares that ended up being occupied. In each year, distance probabilities were recalculated based on new and existing nest locations from the previous year, whereas habitat probabilities remained constant. Once a square had been occupied in one year it was assumed to remain occupied throughout the iteration, and its probability of being occupied in subsequent years by another pair was set to zero (i.e. a 1x1 km square could only ever be occupied by a single pair during the 25 year sequence).

As the model was iterated 100 times, the probability of a square being occupied was calculated as the number of times it was occupied across all iterations / 100. This was done for each of the two classifications of habitat preference (i.e. 100 runs for each of the 20%, and 50% resolution bands).

2.2.11 Assessing accuracy of breeding range expansion predictions

As there is no independent dataset from Scotland of white-tailed eagle nest locations, it was impossible to build a predictive model using one dataset (the training dataset) and validate with another (the testing dataset). However, to validate the accuracy of the breeding range predictions presented in this report, the data were partitioned over time into two time periods; "early" (1983-2012) and "late" (2013-2014). Thus, all new territories established in 2013 and 2014 fell in the latter category.

By re-running the white-tailed eagle habitat associations for the "early" dataset only, the resulting parameter estimates were used to create a habitat suitability map for white-tailed eagle for the whole of Scotland and northern England (i.e. using the same approach as described in section 2.2.8). These values were placed into 50% habitat preference bands, where >50% indicated preferred habitat and <50% indicated areas of less favoured habitat (i.e. habitat not to be used). The preference bands were then combined with a probability distance function, based on distance between new and established nests between 1983 and 2012, to predict where new nests would occur in 2013 and 2014, using the same process as the predicted range expansion described above (i.e. sections 2.2.9 and 2.2.10).

To assess the validity of the predictions from this exercise, the predicted nest site locations in 2013 and 2014 were intersected ("matched in space") with real (new) nest locations in 2013 and 2014. The probabilities that the predicted nest locations would intersect with a real nest location using all new nests across Scotland as well as using a restricted west coast sub-set of new nests were calculated. The probability of intersection was assessed using various search distances from predicted to real new nests: 0 km (i.e. a "perfect match"), 0.5 km, 1 km, 2 km and 3 km. Thus, for the 3 km search distance, a predicted nest site was considered correctly predicting a real nest site if the distance was 3 km or less between the two sites.

The effectiveness of these models was assessed in a step-wise manner using two very similar approaches. In the first approach, all predicted 1 km squares predicted to be occupied at least once were used. In the second approach, 1 km squares predicted to be occupied two or more times were used.

The effectiveness of the full model of white-tailed eagle habitat association combined with distance probability from established nests was then compared with three other models: i) models using only distance from established nest, ii) models using only habitat associations within the core west coast breeding range (i.e. within 70 km of established west coast nests, which is the maximum recorded inter-nest distance for this core breeding range up to 2012), and iii) a null model where 1 km squares could be occupied at random within 70 km of established nests (i.e. without regards to habitat and distance to established nests).

3. RESULTS

3.1 National and regional population growth

After a slow start following the re-introduction, the Scottish (i.e. including both the west and east coast) population of white-tailed eagles has grown rapidly in recent years. 98 pairs were recorded in 2014, of which 90 were confirmed as having a nest (some of the remaining eight pairs might have had a nest that was not found; Fig. 2). In the two regions with highest density of white-tailed eagles, Mull and Skye, with 21 and 18 territorial pairs in 2014, respectively (i.e. densities of 0.024 and 0.011 pairs per km²; Table 1 and Fig. 3), the populations have also grown rapidly. On Mull, the increase between 2010 and 2014 was most notable, when the population grew from 10 to 21 pairs (Figs 2 and 3).

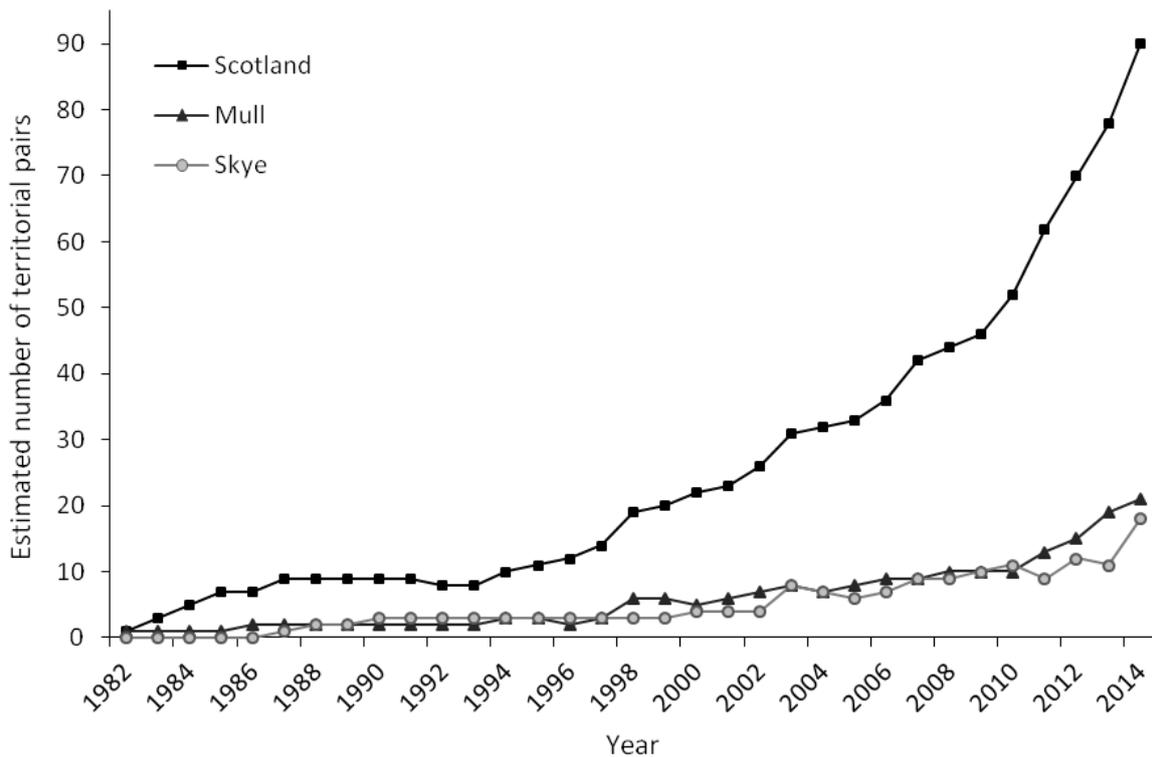


Figure 2. The number of confirmed breeding pairs of white-tailed eagles in Scotland (black line, squares), on Mull (black line, triangles) and on Skye (grey lines, circles line) between 1982 and 2014.

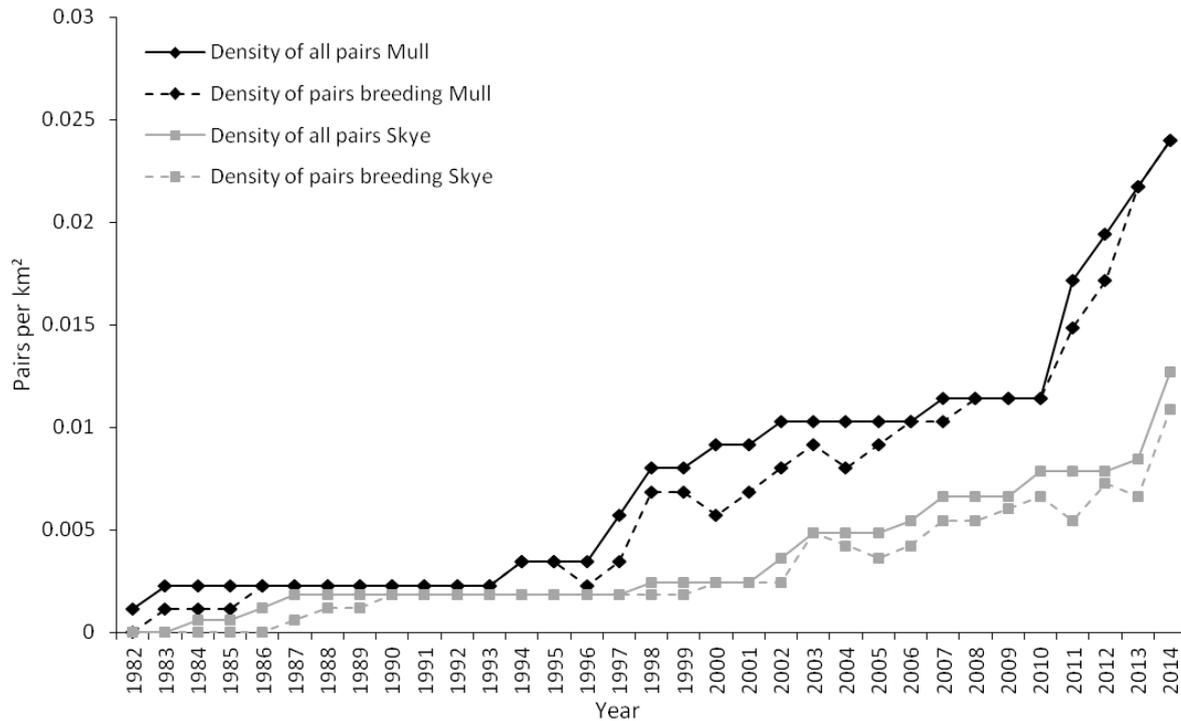


Figure 3. The density of confirmed breeding pairs (dashed lines) and all territorial pairs (confirmed breeding as well as newly established non-breeding pairs; solid lines) of white-tailed eagles on Mull and Skye between 1982 and 2014.

3.2 Population composition

Most breeding individuals in 2013 were between six and 10 years old. The youngest and oldest breeding birds in 2013 were five and 38 years of age, respectively. However, breeding individuals over the age of 30 were uncommon in the population (Fig. 4).

Between 1983 and 1993, all individuals in the breeding population consisted of birds from the first release phase (Fig. 5a). From 1995 onwards, the proportion of wild-bred birds within the breeding population increased rapidly (Fig. 5b). By the mid-2000s, over 50% of the population were wild-bred and, by 2013, wild-bred birds by far outnumbered the released birds, making up 89.6% of the breeding population. In fact, in 2013 when the origins of 156 breeding individuals were known, only nine originated from the west coast releases and four from the east coast releases. The number of breeding individuals originating from the east coast releases is likely to increase in the short term, as these birds reach maturity and are recruited into the breeding population (cf. Fig. 5a). Still, wild bred-birds will continue to form the majority of the Scottish population.

Estimated age at first breeding of wild-bred birds (i.e. an update from Evans *et al.*, 2009), showed that on average wild birds bred at five years of age, with males and females showing very similar ages of first breeding (4.9 and 5.0 respectively; Table 3). There was a weak tendency for released males to breed earlier than released females (Table 3).

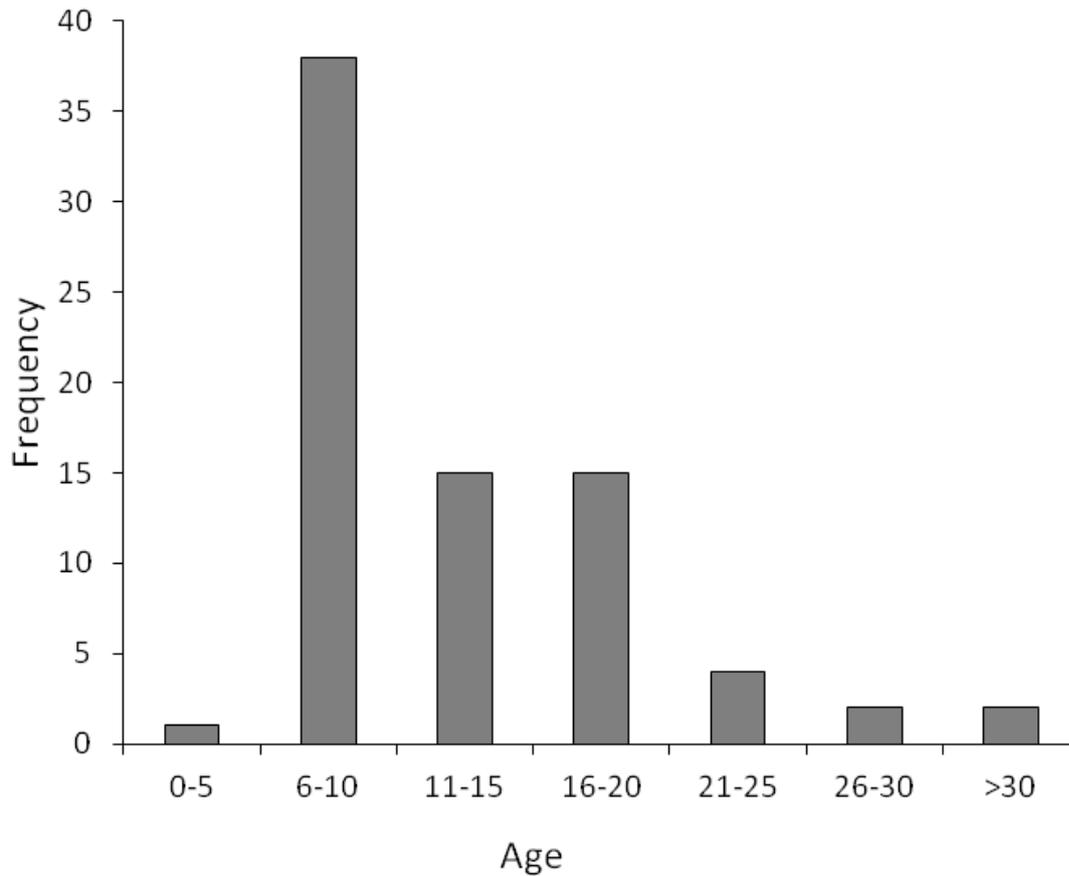


Figure 4. The age distribution of territorial white-tailed eagles in the Scottish breeding population in 2013.

3.3 National and regional breeding success

3.3.1 National trends in breeding success

Nationally and over time, there has been a significant increase in both the proportion of nests fledging at least one young (GLMM, $F_{1, 860} = 19.50$, $p < 0.0001$) and the number of chicks fledged per breeding attempt (GLMM, $F_{1, 858} = 15.52$, $p < 0.0001$; Fig. 6). There was no evidence that the proportion of successful nests had declined over time, which would have been expected if density-dependent reproduction occurred (i.e. effect of the quadratic effect of year: $p = 0.60$). However, since 2006, the number of chicks fledged per breeding attempt appears to have remained relatively constant at an average of 0.673 chicks fledged per territorial pair, although reproduction was relatively low in 2014 (Fig. 6). The results suggest that this increase over time is due to the increased proportion of birds in the population with more breeding experience (Fig. 7). There was no evidence that origin of pairs (released, wild or mixed) had a significant effect on the number of fledglings produced ($p = 0.149$; Table 4). Similarly, there was no significant effect of latitude, longitude and the interaction between latitude and longitude on the number of fledglings produced (Table 4).

a)

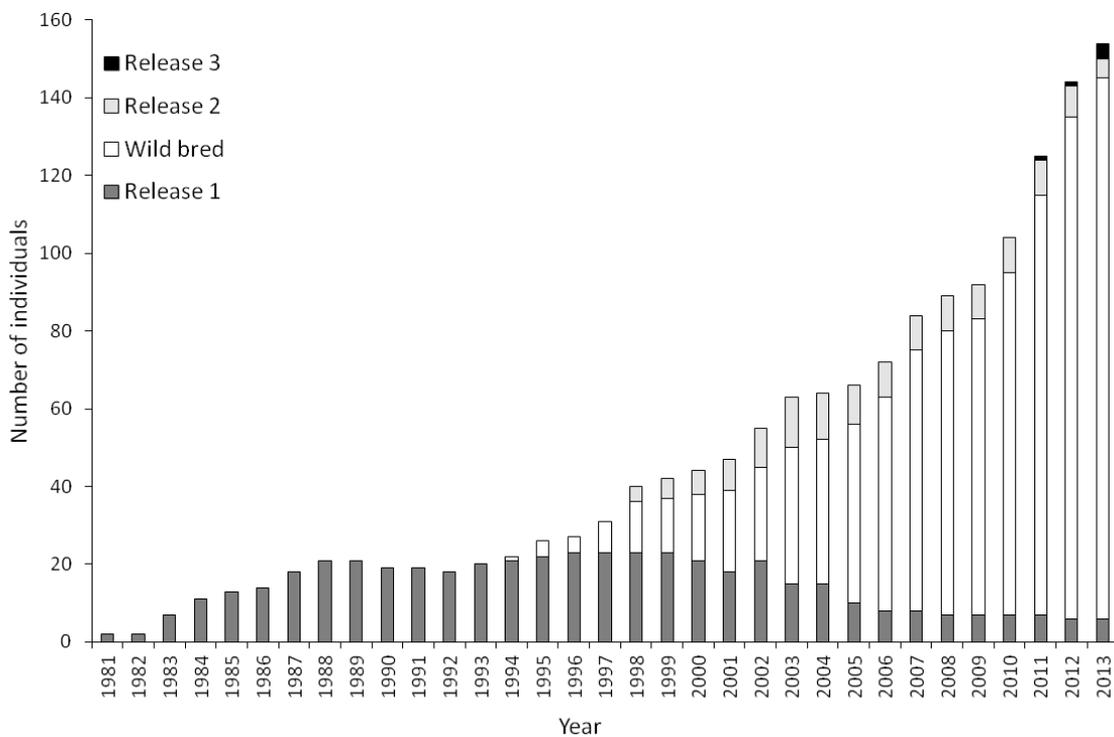
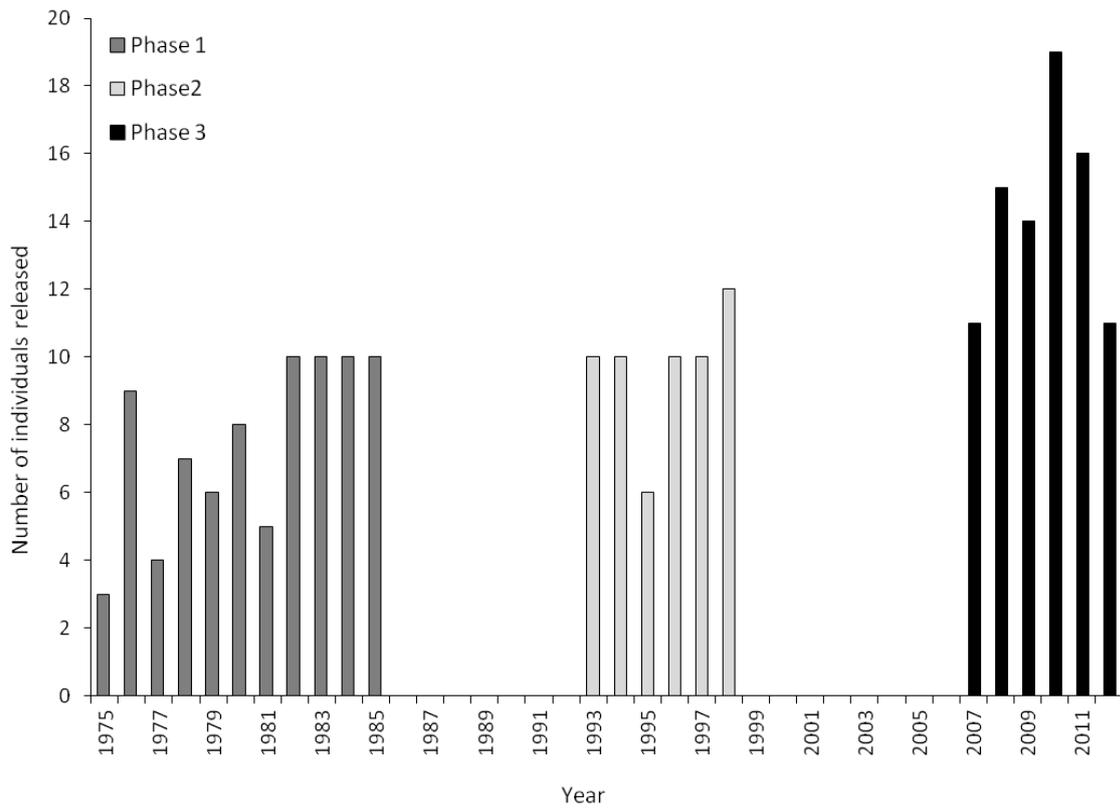


Figure 5. In a), the number of white-tailed eagles released per year between 1975 and 2012. Release phases one and two occurred on the west coast of Scotland, whereas the third phase involved releases of birds on the east coast of Scotland. In b), the number and origin of individual white-tailed eagles recorded on territories in Scotland between 1981 and 2013.

Table 3. Mean (\pm SE) of age at first breeding of wild-bred and released white-tailed eagles in Scotland. The sample size (N) is shown in brackets for each sub-group. The figures from released birds are from Evans et al. (2009). The figures for wild-bred birds have been updated based on breeding attempts up to and including 2013.

Origin	Males	Females	All
Released phase 1	5.2 \pm 0.7 (5)	6.4 \pm 1.2 (7)	5.9 \pm 0.7 (12)
Released phase 2	4.7 \pm 0.5 (6)	5.1 \pm 0.3 (10)	5.0 \pm 0.3 (16)
Wild-bred	4.9 \pm 0.2 (37)	5.0 \pm 0.2 (36)	5.0 \pm 0.1 (73)
All	4.9 \pm 0.1 (48)	5.2 \pm 0.5 (53)	5.1 \pm 0.2 (101)

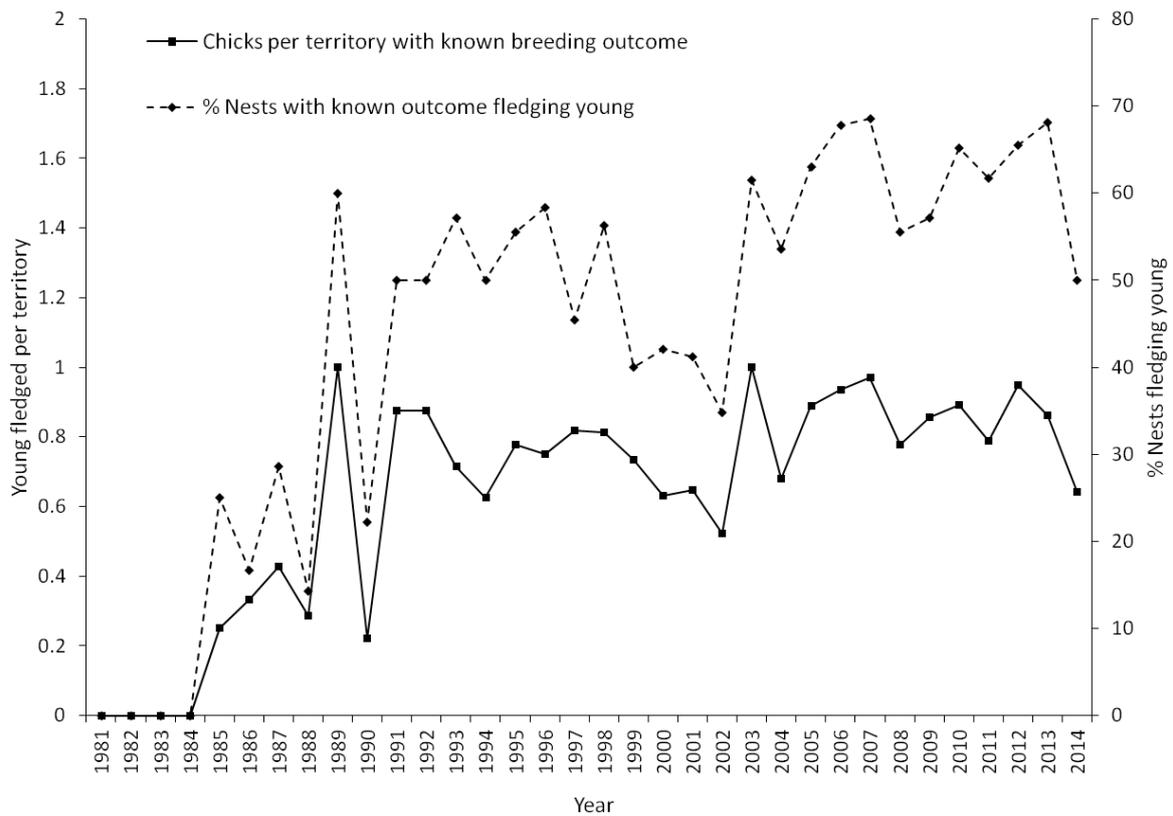


Figure 6. Mean number of white-tailed eagle chicks fledged per territorial pair with known breeding outcome (solid line, left y-axis) and mean percentage white-tailed eagle nests fledging at least one chick (dashed line, right y-axis) in Scotland between 1981 and 2014.

Table 4. Model of the effects of latitude, longitude, the interaction between latitude and longitude, origin of birds in a pair (released, wild and mixed) and breeding experience of the pair (linear and quadratic terms) on the number of chicks fledged per territorial pair with known breeding outcome between 1983 and 2014.

Effect	DF	F	P
Latitude	1	1.03	0.329
Longitude	1	0.96	0.312
Latitude*Longitude	1	1.03	0.313
Origin	2	1.95	0.149
Years of experience as a pair	1	25.61	<0.0001
Years of experience as a pair ²	1	15.13	0.0001

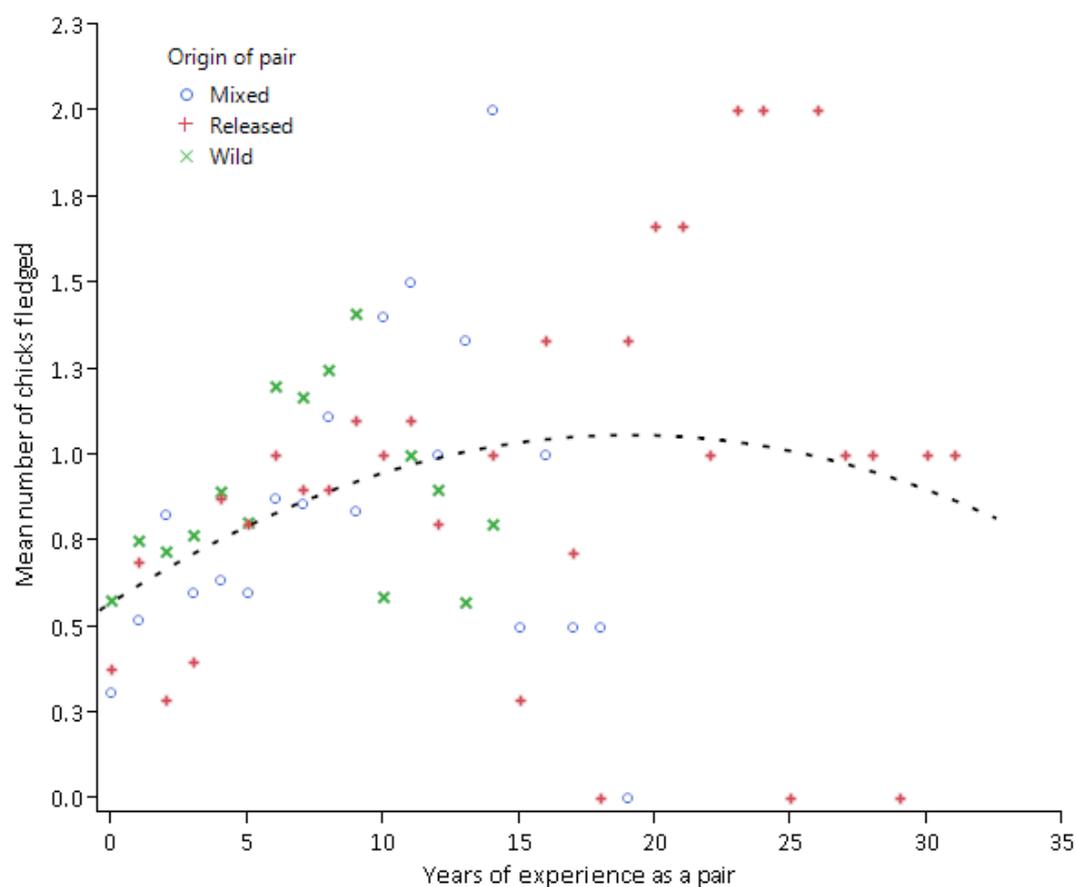


Figure 7. The effect of origin (released, wild-bred and mixed) of breeding pairs of white-tailed eagles on the mean number of young fledged per pair, for all breeding attempts recorded between 1983 and 2014. Regardless of origin of parental birds, the fledgling rate increases with increased breeding experience. The dashed line is a quadratic curve fitted to all data to aid the visual interpretation.

3.3.2 Regional trends in breeding success and evidence of density-dependent productivity

There was no significant increase in the number of white-tailed eagle fledglings produced per territorial pair with known breeding outcome over time since the eagles started to breed on the islands of Mull and Skye ($p > 0.056$; Fig. 8). In addition, there was no evidence that the nest success had declined in recent years on either of the islands, as there was no significant quadratic effect of years since start of breeding ($p \geq 0.150$).

On both Mull and Skye, the number of fledglings produced per territorial pair with known breeding outcome was not significantly associated with the total number of pairs on the island, indicating that breeding performance was not density-dependent (Table 6, Fig. 9). However, the number of fledglings produced was positively related to the number of years of breeding experience the pair had (Table 6). The number of fledglings produced was marginally significantly related to the origin of the parental birds on Mull, but not on Skye (Table 6). This was most likely related to the fact that in the early years (i.e. 1985-1995), only released birds bred on Mull, and in several of those years, the productivity was high. The effect of origin is not significant when excluding the variable "Years of experience as a pair".

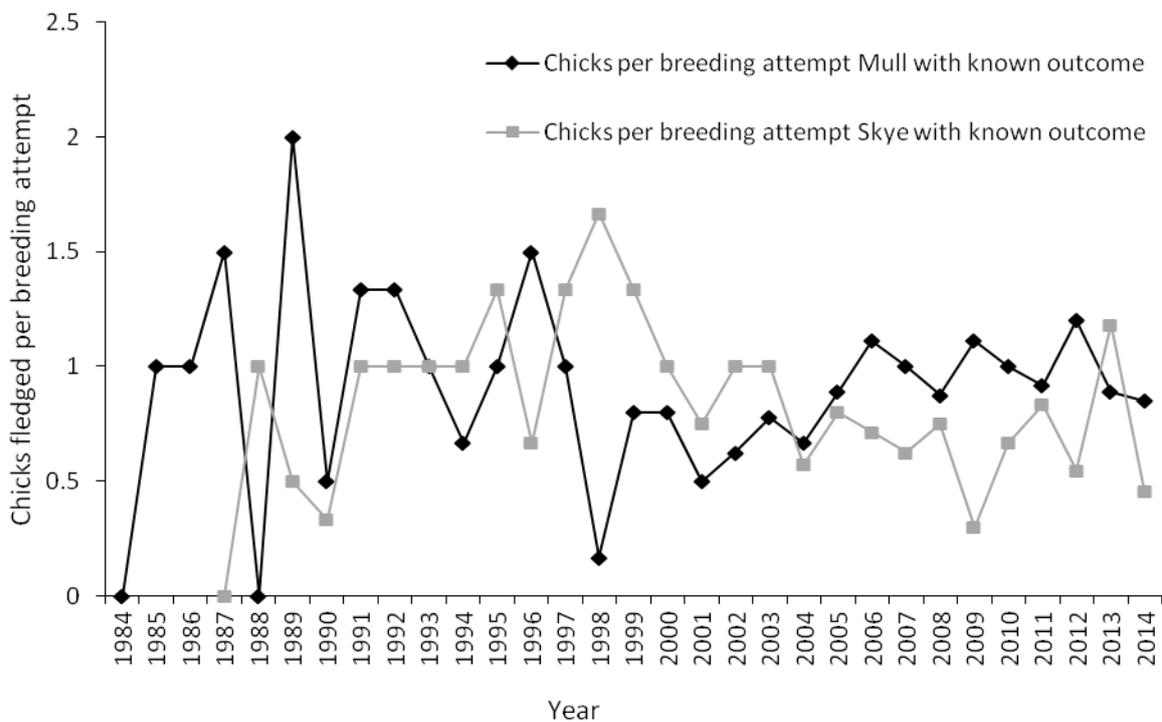


Figure 8. The number of white-tailed eagle fledglings produced per territorial pair with known breeding outcome on Mull and Skye between 1984 and 2014.

Table 6. Models of the effects total number of territorial pairs as a measure of population density (linear and quadratic terms), origin of parental birds (released, wild-bred or mixed) and years of experience as a pair (linear and quadratic terms) on the number of white-tailed eagle fledglings per breeding attempt on a) Mull between 1983 and 2014 and b) Skye between 1987 and 2014.

Effect	DF	F	P
a) Mull			
Number of pairs	1	2.57	0.110
Number of pairs ²	1	2.66	0.120
Origin	2	3.11	0.047
Years of experience as a pair	1	14.36	0.0002
Years of experience as a pair ²	1	14.45	0.0002
b) Skye			
Number of pairs	1	0.10	0.754
Number of pairs ²	1	0.03	0.871
Origin	2	2.25	0.123
Years of experience as a pair	1	5.29	0.023
Years of experience as a pair ²	1	2.33	0.129

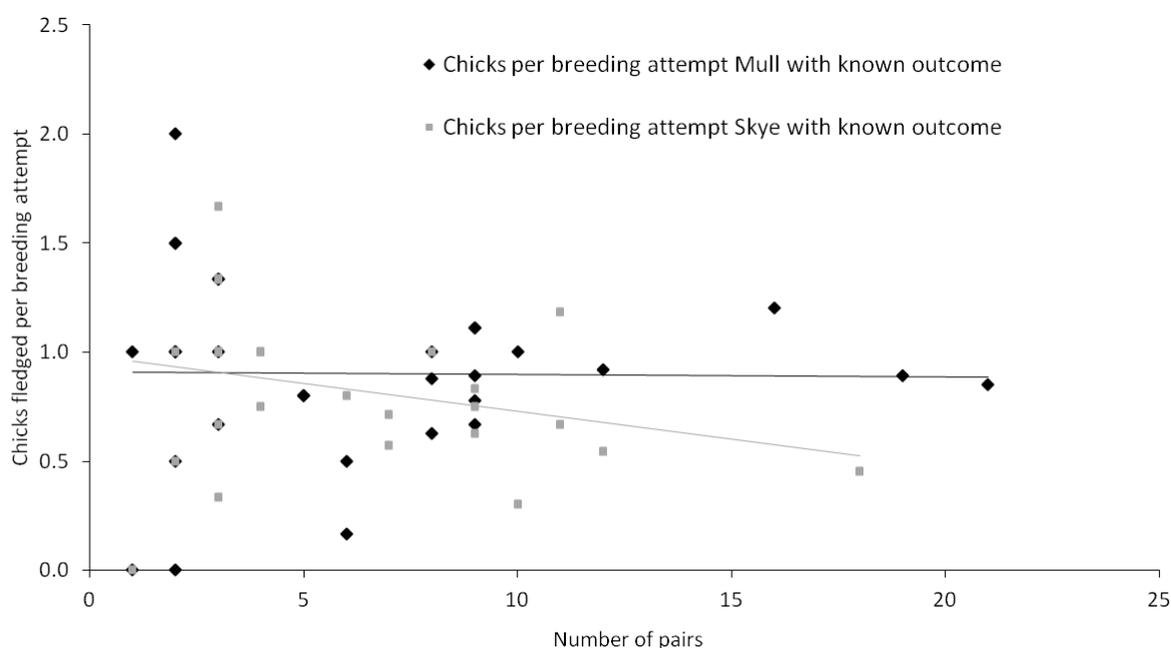


Figure 9. The number of fledglings produced per territorial white-tailed eagle pair with known breeding outcome on Mull (black diamonds) and Skye (grey squares) between 1984 and 2014 in relation to the number of pairs on the islands. The straight lines (black=Mull, grey=Skye) depict the best fit from region-specific linear regressions and are included to aid the visual interpretation of the graph.

3.4 Annual survival rates

Capture-mark-recapture modelling of encounter histories of birds released in the second phase on the west coast between 1993 and 1998 and wild-bred birds between 1983 and 2007 showed a very low over-dispersion, with a bootstrap Goodness-Of-Fit (GOF) \hat{c} of 1.15. After adjusting for this, the results suggested that models using four age transitions classes for survival (zero to three; three to four; four to five; and five years and older) and six age transitions classes for probability of resighting (zero to one; one to two; two to three; three to four; four to five; and five years and older) had a superior fit to the data than models using other age groupings. In fact, the top three of the five best-fitting models contained this combination of age structures (Table 7). The best-fitting model had a QAICc weight of 0.40 (i.e. there was a 40% support for this model compared to other models), compared with the second best model with a QAICc weight of only 0.14. The second best model used an age structure that occurred in only a small number of the 10 best-fitting models and had a Δ QAIC of >2 (Table 7). Therefore, only the best-fitting model was used to estimate survival by age and origin.

In general, the estimated annual survival rates (\pm SE) of white-tailed eagles were high, varying from $75.7 \pm 6.0\%$ for released birds aged four years to $96.1 \pm 0.8\%$ for wild-bred birds older than five years. For all ages, survival rates were higher for wild-bred compared to release birds (Table 8).

Table 7. The 10 models (of a total of 81 tested) with best fit to the data regarding survival and re-sighting probability of white-tailed eagles in Scotland between 1983 and 2013. "Phi" is the probability of survival and "p" the probability of re-sighting. Age transition classes tested were: AGEBIN4, with the transitions 0-3, 3-4, 4-5 >5 years; AGEBIN5, with the transitions 0-1, 1-2, 2-3, 3-4, >4 years, and AGEBIN6, with the transitions 0-1, 1-2, 2-3, 3-4, 4-5 and >5 year. The age transition AGEBIN5 was not among the 10 highest ranking models. All models with best fit to the data contained "origin" (i.e. released or wild-bred) in both the "Phi" and "p" terms. The different models are ordered based on their QAICc-values.

Model	QAICc	Δ QAICc	QAICc weight	No. of parameters
Phi(AGEBIN4 + origin) p(AGEBIN6 + origin)	1588.971	0	0.404	12
Phi(AGEBIN4 + origin) p(AGEBIN4 + origin)	1591.135	2.165	0.137	10
Phi(AGEBIN4 * origin) p(AGEBIN6 + origin)	1592.166	3.195	0.082	15
Phi(AGEBIN6 + origin) p(AGEBIN6 + origin)	1592.224	3.254	0.079	14
Phi(AGEBIN6 * origin) p(AGEBIN6 + origin)	1592.883	3.912	0.057	19
Phi(AGEBIN4 + origin) p(AGEBIN6 * origin)	1593.029	4.058	0.053	17
Phi(AGEBIN4 * origin) p(AGEBIN4 + origin)	1594.083	5.112	0.031	13
Phi(AGEBIN6 * origin) p(AGEBIN4 + origin)	1594.099	5.128	0.031	17
Phi(AGEBIN6 + origin) p(AGEBIN4 + origin)	1594.755	5.784	0.022	12
Phi(AGEBIN4 + origin) p(AGEBIN6 + origin)	1588.971	6.017	0.020	13

Table 8. Survival estimates by age class for released and wild-bred white-tailed eagles in Scotland based on the model with the lowest QAICc (i.e. with best fit to the data) in Table 7.

Age class	Origin	
	Released	Wild
0-3	0.785 ± 0.032	0.874 ± 0.016
3-4	0.809 ± 0.062	0.890 ± 0.037
4-5	0.757 ± 0.060	0.855 ± 0.036
5+	0.928 ± 0.016	0.961 ± 0.008

3.5 Predicting future population growth rate

The predicted population growth rate of white-tailed eagles, which were based on values for the mean (\pm SD) number of fledglings produced per successful pair (Table 9) and survival (Table 8), showed a good fit to the observed population growth of white-tailed eagles between 1975 and 2014 (Fig. 10). After 2007, the predictive model used in this study seems to match the observed population sizes better than the model presented by Evans *et al.* (2009; cf. Fig. 10). This suggests that the model presented here captures the recent changes in demographic processes well (e.g. the increasing proportion of wild-bred individuals with increasingly more breeding experience and higher survival than released birds; cf. Table 8), and it provides support that predictions into the future might be broadly accurate. For 2015-2025, the model suggested an almost exponential growth rate (Fig. 10).

The observed mean annual growth rate in the Scottish west coast population over the *previous* 10 years has been 9.7% (range: 3%-16%). This suggests that, if population growth continues at its current rate, by 2025 the population may increase to approximately 221 pairs. The Vortex model predicted a mean annual growth rate over the *next* 10 years of 8.6% (range: 8.0%-9.2%), which also would result in approximately 221 pairs.

When predicting the population growth over the next 25 years (i.e. up to and including 2040), the Vortex model estimated a population of 889 pairs by 2040 (Fig. 11). If the observed annual growth rate for the *previous* 10 years remains at its current mean rate of around 9.7% per year, the population might reach approximately 1,005 pairs by 2040. This assumes an absence of density-dependent population regulation (i.e. that survival and/or breeding success will not change according to breeding density).

It proved difficult to predict future regional population growth, as the data were too sparse to estimate regional survival rates. However, Appendix 2 of this report contains an example of how the extent of suitable habitat in different regions might give an insight into potential carrying capacity in each region.

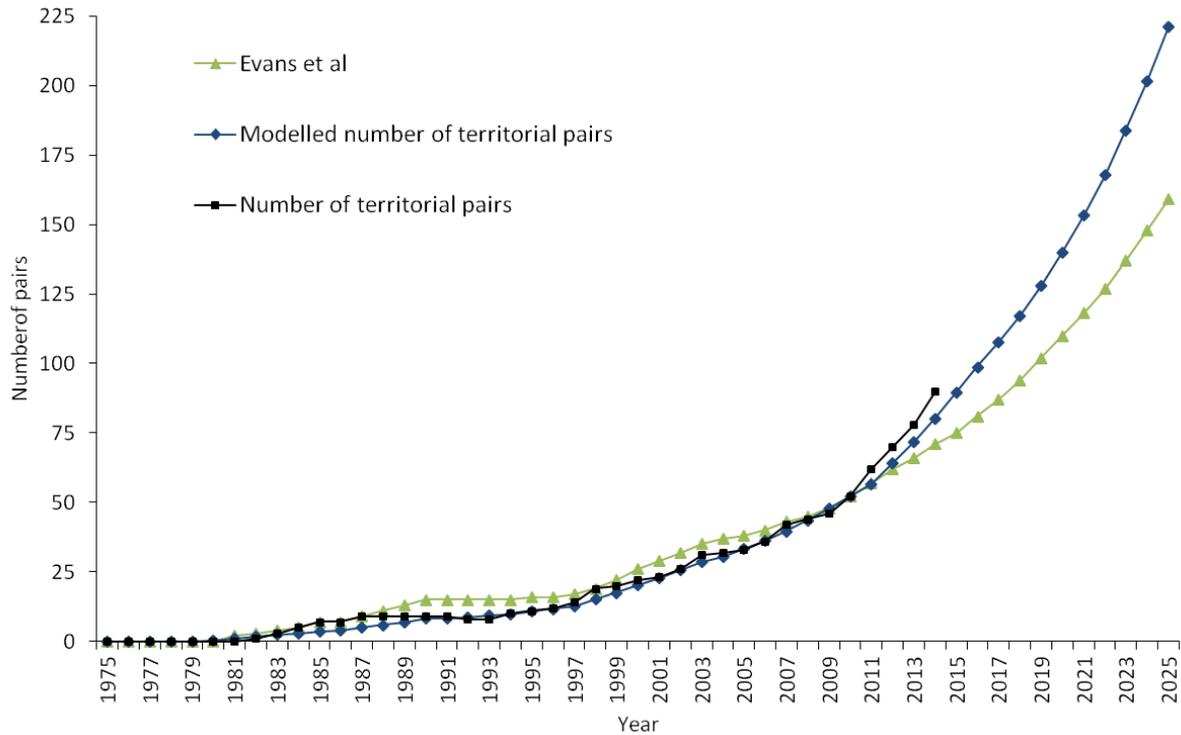


Figure 10. Estimated population growth of white-tailed eagles in Scotland until 2025 under two different modelled scenarios, as well as the observed number of territorial pairs in Scotland (black line, squares). The previously best predictive model of future population growth (Evans et al. (2009); green line, triangles) provided a good fit to the observed population growth until approximately year 2010. However, a model that used up-to-date demographic rates from released and wild-bred birds, the overall proportion of territorial pairs that bred (mean \pm SD from 1983-2014), the mean proportion of successful nests and the mean (\pm SD) number of chicks per successful attempt shows a better fit to the observed population trend (dark blue line, diamonds). The better fit of the updated model compared with the model by Evans et al. (2009) is due to an improved way of capturing the mortality of released first-year birds in the models.

Table 9. Mean (\pm SD) number of fledglings produced per successful breeding attempt used to model population growth in Vortex. Wild and released birds were assumed to have the same success rates based on the results of analysis of factors affecting number of chicks fledged (Table 4).

Time period	Mean \pm SD of chicks fledged per successful breeding attempt
1975-1984	0.000 \pm 0.000
1985-1994	1.506 \pm 0.359
1995-2004	1.524 \pm 0.191
2005-2014	1.382 \pm 0.080

3.6 Effects of additional mortality

The effects of increased (i.e. additive) annual mortality caused by potential collisions with wind turbines and increased illegal killing predicted a reduction in population growth (Fig. 12). For example, the scenario including the deaths of an additional 14 birds per annum predicted a mean (\pm SE) population size in 2040 that would be around 489 (\pm 10) pairs, which is 400 pairs lower than the modelled population with no extra mortality (Fig. 12). However, an annual loss of only two white-tailed eagles would result in around 794 (\pm 10) breeding pairs.

Perhaps a more realistic modelling approach is to assume that the annual mortality is related to the overall population size. Three scenarios were modelled using this approach, assuming that 0.5%, 1% and 2% of the total population size were killed annually, with mortality equally spread between the different age classes and between the two sexes. In none of these scenarios did the estimated number of breeding pairs decline (Fig. 13). However, the modelled density-dependent cumulative mortality lowered the estimated population size in 2040 to between 537 and 777 pairs of breeding white-tailed eagles (Fig. 13).

Thus, despite potentially limiting the overall population size, the modelled additive mortality levels would not cause a population decline or extinction (across the whole population) and would only reduce the rate at which population growth occurs.

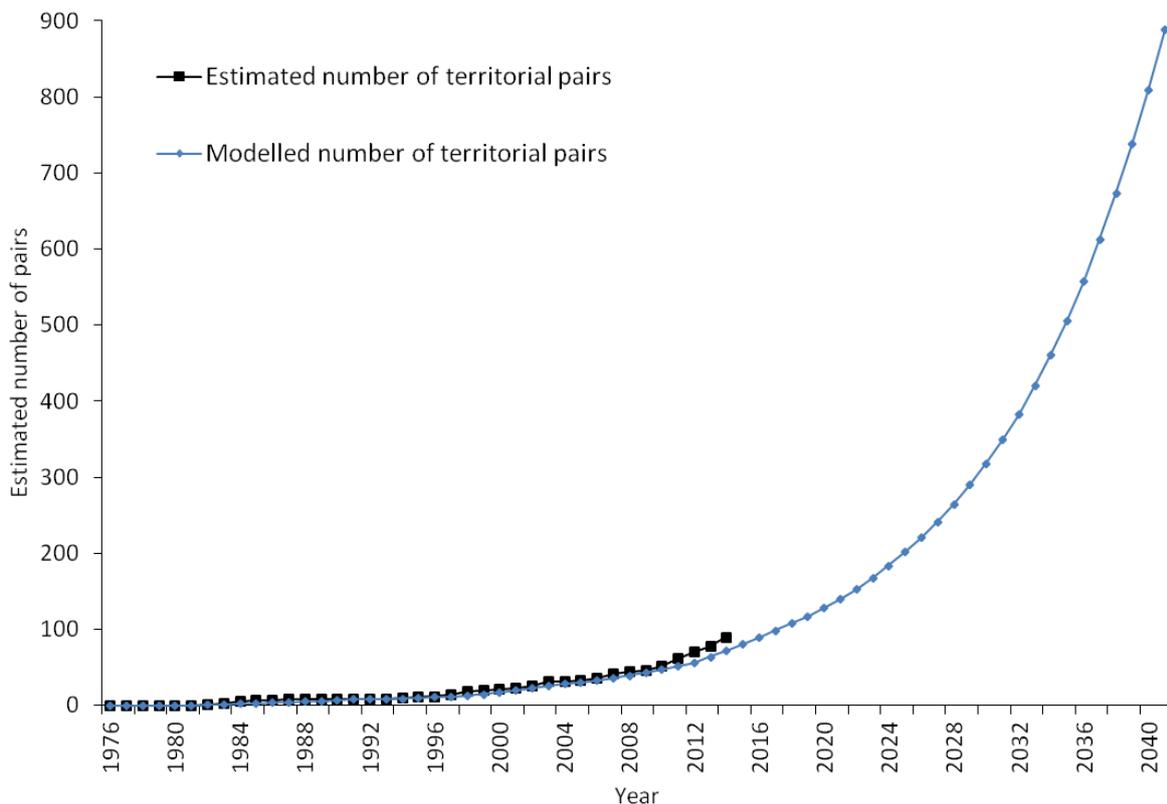


Figure 11. The observed number of territorial pairs (black squares, black line) between 1976 and 2014 as well as the modelled predicted population growth of white-tailed eagles in Scotland until year 2040 (blue dots, blue line) under the updated population growth model, using the mean proportion of territorial adults breeding and the number of chicks fledged per successful attempt.

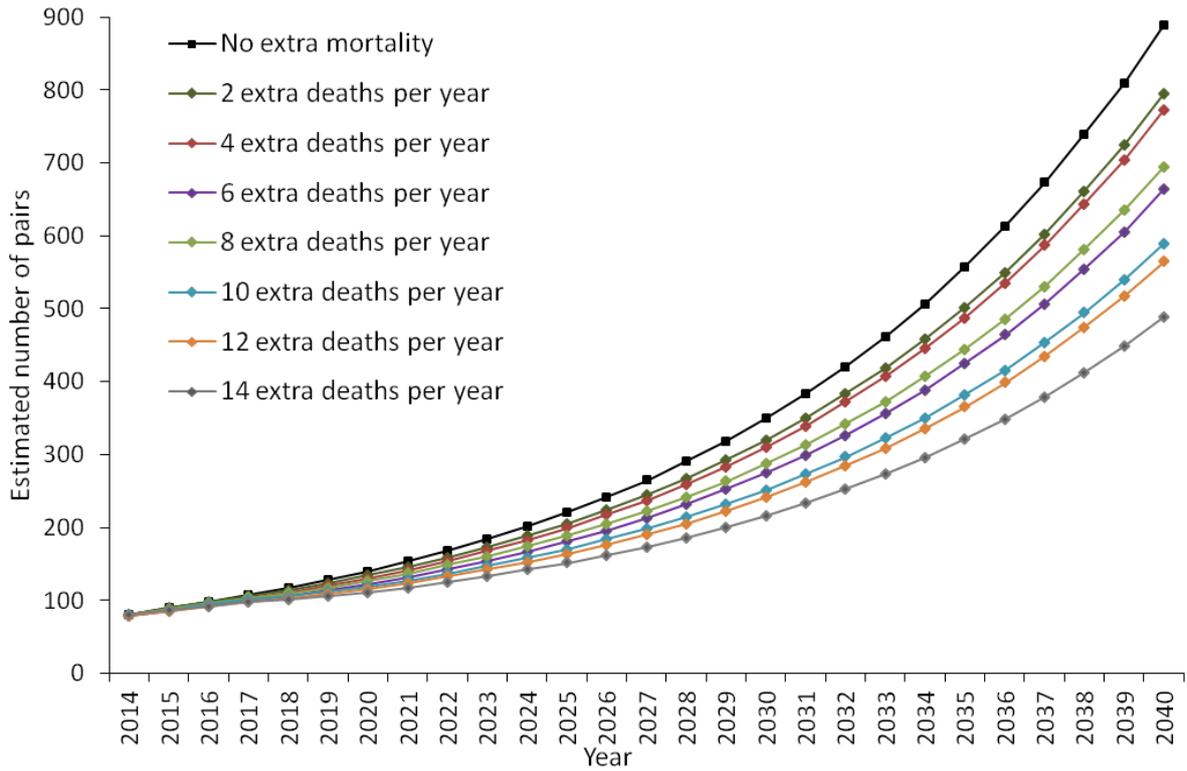


Figure 12. Predicted increase in the number of breeding pairs of white-tailed eagles in Scotland from year 2014 to 2040 from a model without additive mortality (potentially caused by collisions with wind turbines and illegal killing; black line, squares). Seven other scenarios using the same underlying demographic rates but with additive mortality affecting between two and 14 white-tailed eagles annually from 2015 onwards are depicted by coloured lines and solid diamonds). The impact of the additional mortality suggests that the predicted population growth will be lower with increased mortality rates, but that extinction is not likely to occur.

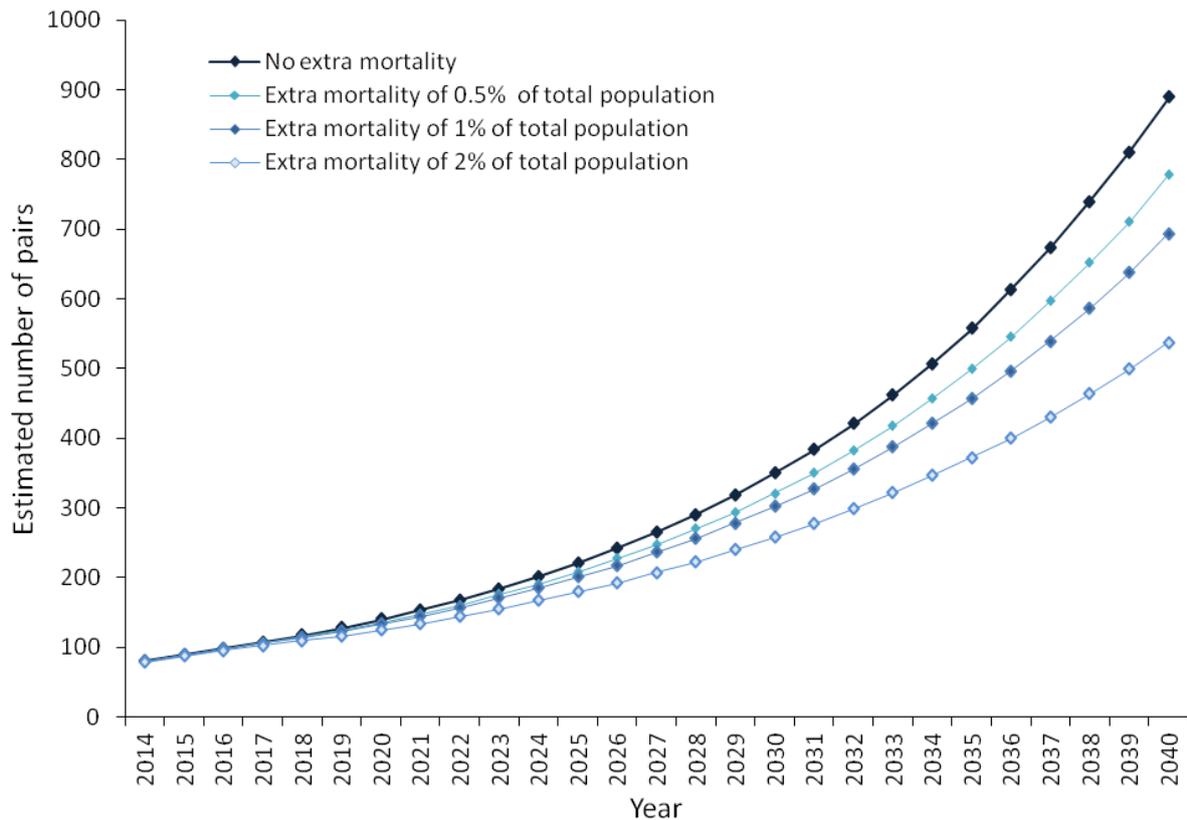


Figure 13. Predicted increase in the number of pairs of white-tailed eagles in Scotland from year 2014 to 2040 from a model without additive mortality (potentially caused by collisions with wind turbines and illegal killing; black line, black diamonds). Three other scenarios using the same underlying demographic rates are depicted. For these scenarios, additive mortality affects 0.5% (light blue line, solid light blue diamonds), 1% (dark blue line, solid dark blue diamonds) and 2% of the total population (light blue line, open diamonds) from 2015 onwards. The impact of the additional mortality suggests that the predicted population size will be lower with increased mortality rates, but that extinction is not likely to occur.

3.7 Effects of limited carrying capacity

Setting the carrying capacity to an arbitrary level of 4,000 individuals had very little effect on the total number of breeding pairs, over the time-scale considered (Fig. 14). By 2040, a total of 816 white-tailed eagle pairs were predicted by this model, compared with 889 when no restriction was placed on carrying capacity. When the carrying capacity was set to 3,000 individuals, the white-tailed eagle population growth began to slow after year 2034 (relative to the model with no restrictions), and by 2040, the decreasing population growth rate meant the population was predicted to be 692 pairs. Under lowest carrying capacity of 2,000 individuals, the population growth began to slow after 2029, and by 2040, the population was predicted to be 494, by which time population growth had begun to level off (tend towards zero; Fig. 14). For an examination of possible regional carrying capacities, see Appendix 2.

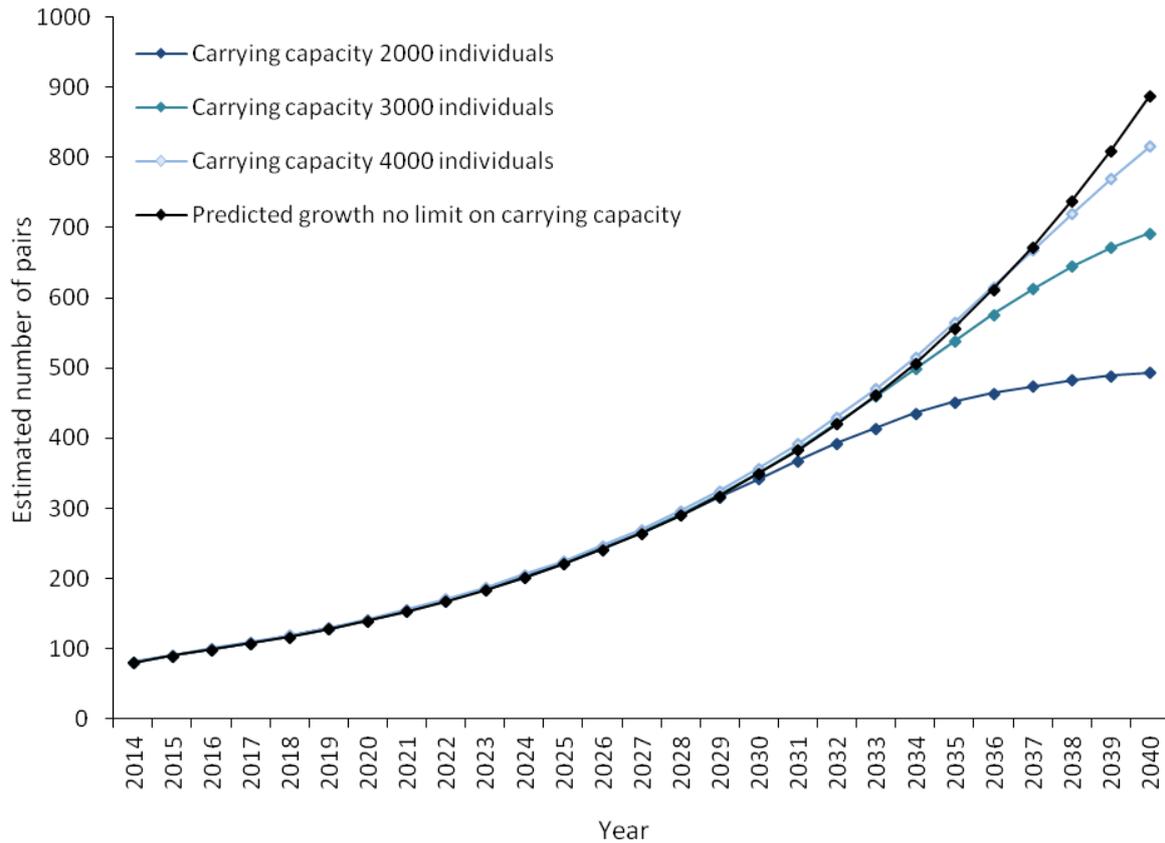


Figure 14. Predicted population growth of white-tailed eagles in Scotland from 2014 to 2040 for a model without any limit on carrying capacity (black line, solid diamonds), compared to population growth using the same demographic rates but setting the carrying capacity at various levels (blue lines). Carrying capacity was specified in terms of the total number of individuals (including all age classes); outputs are shown in terms of the predicted number of breeding pairs.

3.8 Habitat associations

After the model selection exercise, the final model of associations between white-tailed eagle breeding sites and habitats at the 1 km scale suggested a significant preference for greater cover of inland water and forest, and for a greater length of coast. At this scale, eagle territories were also closer to forest but further from inland water than random territories (Table 9a). At the 2 km scale, only coast length, distances to forest and inland water were found to be significantly different from random locations (Table 9b). At the 3 km scale, real territories contained more coast length, were closer to forest and, unlike at smaller scales, were significantly more likely to occur in areas of lower and less variable altitude (i.e. lower and flatter ground), relative to random territories (Table 9c). The model at the 1 km scale showed the best fit to the data, with an AIC of 210.83, with the 2 and 3 km models having a $\Delta AICs \geq 34$ (Table 9).

To explore whether a model of white-tailed eagle presence using a combination of significant variables across the spatial scales (i.e. 1, 2 and 3 km) would improve the fit to the data, significant variables from the 2 and 3 km models (Table 9b-c) were added to the 1 km model. Thereafter, a backward elimination procedure took place, after which four variables remained significantly associated with white-tailed eagle presence (Table 10). This multi-scale model suggested that white-tailed eagles were significantly positively associated with the area of inland water and forest, as well as the length of coastline at the 1 km scale

(Table 10). In addition, white-tailed eagles were significantly negatively associated with the coefficient of variation in altitude (i.e. less variation in altitude) within 3 km of the nest (Table 10). This model had an AIC of 211.42, which is within two AIC units from the 1 km model (Table 9a), and this model explained 24.8% of the variation in the breeding occupancy of white-tailed eagles (i.e. the percentage deviance explained was 24.8%). Based on the fact that the multi-scale model and the 1 km model had a similar good fit to the data (i.e. $\Delta AIC < 2$ units) and that the multi-scale model included a variable describing topography, which Evans *et al.* (2010) found was a good discriminator between white-tailed eagle and golden eagle territories, it was deemed that the multi-scale model was biologically more relevant than the 1 km scale model.

Table 9. Habitat, landscape and topographical variables found to be significantly different between real and random White-tailed Eagle territories at the 1, 2 and 3 km, after backward model selection of a binomial GLM.

Variable	Estimate	SE	d.f.	χ^2	P
a) 1 km buffers					
Intercept	0.309	0.181			
Inland water	0.526	0.203	1	8.77	<0.003
Coast length	1.783	0.310	1	53.75	<0.001
Forest cover	0.711	0.207	1	13.17	<0.001
Distance to forest	-0.455	0.243	1	4.37	0.037
Distance to inland water	0.400	0.209	1	3.90	0.048
AIC = 210.83					
b) 2 km buffers					
Intercept	0.232				
Coast length	0.982	0.249	1	22.29	<0.001
Distance to Forest	-0.451	0.185	1	6.77	0.009
Distance to inland water	0.384	0.176	1	5.07	0.024
AIC= 245.39					
c) 3 km buffers					
Intercept	0.188				
Coast length	0.764	0.277	1	9.747	0.002
Distance to forest	-0.332	0.177	1	4.015	0.045
Coefficient of variation of Altitude	-0.492	0.220	1	5.309	0.021
Mean Altitude	-0.611	0.220	1	8.345	0.004
AIC= 248.37					

3.8.1 Assessment of model performance

The ROC plot analyses of the final multi-scale habitat model and the resulting area under the curve (AUC) showed good performance (AUC = 0.829, 95% C.I. 0.780-0.888). This suggested that the multi-scale model was highly effective at predicting the occurrence of breeding white-tailed eagles in relation to habitat characteristics.

Table 10. Habitat, landscape and topographical variables found to be significantly different between real and random white-tailed eagle territories at multiple spatial scales within a single model. The model explains 24.8% of the variation in white-tailed eagle occupancy.

Variable	Estimate	SE	d.f.	χ^2	P
Intercept	0.341				
Inland water 1km	0.576	0.217	1	9.38	0.002
Coast length 1km	2.048	0.346	1	58.97	<0.001
Forest cover 1km	0.861	0.198	1	22.77	<0.001
Coefficient of variation of Altitude 3km	-0.447	0.227	1	4.15	0.042

AIC= 211.42

3.8.2 Applicability of white-tailed eagle habitat associations outside the core breeding range

The range of values of the habitat variables from the core breeding area in the final model presented here (i.e. Table 10) were similar to the values of 1 km squares located outside the core breeding area in Scotland and northern England (Fig. 15). No 1 km squares had values of all four habitat variables that all fell outside the 90% range of values observed in the core breeding area. The variable that most commonly had values outside the range of values observed within the core breeding area was the CV of altitude within 3 km. Interestingly, the values outside the 90% range indicated that these areas were flatter than what was observed within the core breeding area (e.g. large parts of the Cairngorm Plateau and many parts of Scotland south of the Central Belt). In addition, in some places (e.g. parts of the Kintyre peninsula, in Galloway and Kielder Forest), the amount of forestry cover was higher than observed within the core breeding area. Finally, in a few places (e.g. around large lochs and lakes such as Loch Lomond and Kielder Reservoir), the amount of inland (fresh water) was higher than in the core breeding area. However, in general, this meant that the final model presented here (Table 10) is applicable to a large part of Scotland and northern England (Fig. 15).

With the caveat that the final model of associations between white-tailed eagles and habitats presented here is not fully applicable to all parts of Scotland and northern England (e.g. Fig. 15), these habitat associations can be visualised in a map. This "habitat suitability map" suggests that many parts of Scotland and northern England contain a combination of habitat and topography that is suitable for white-tailed eagles (Fig. 16)

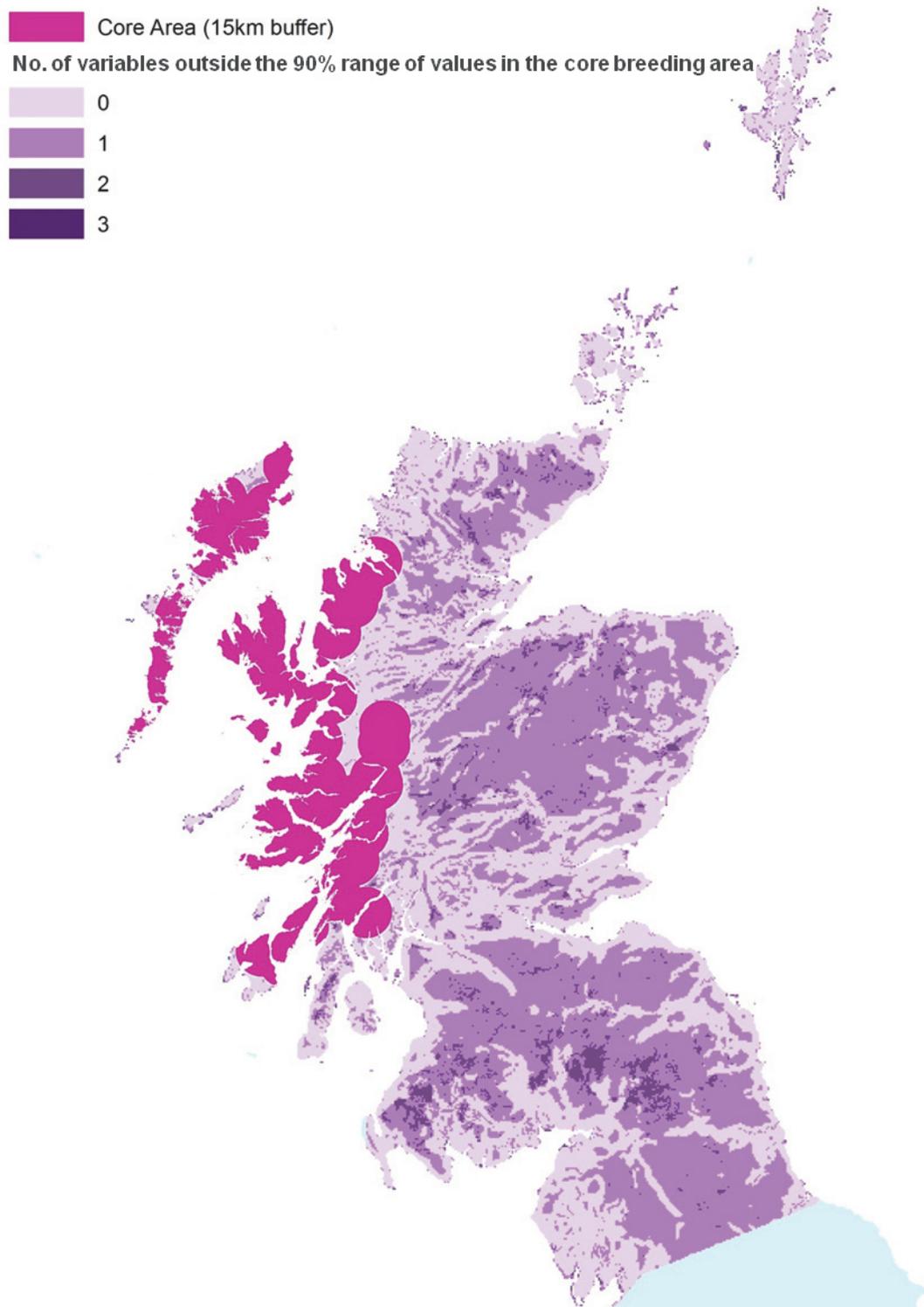


Figure 15. Map showing the areas of the core breeding area of white-tailed eagles on the west coast of Scotland from which the majority of nest site data came from. The map also shows individual 1 km squares with the number of variables with values outside the 90% range of values observed in the core breeding area. The applicability of the final white-tailed eagle–habitat association model presented in this report (i.e. Table 10) decreases in areas where one, two and three variables have values outside the range of values observed in the core breeding area.

Predicted Habitat Suitability

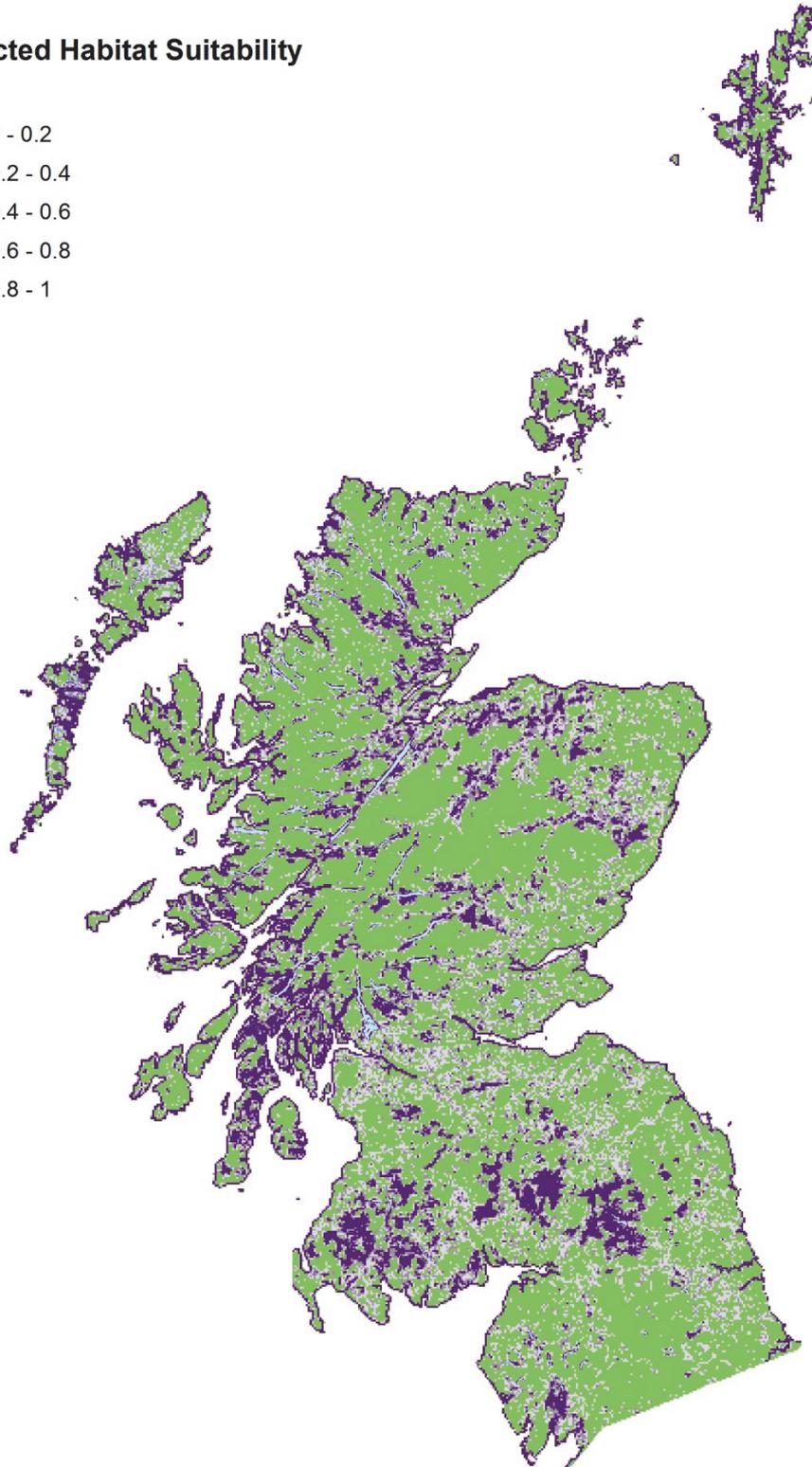
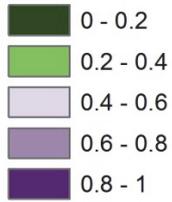


Figure 16. Probability of a 1x1 km square being occupied by a breeding pair of white-tailed eagles, based on habitat preferences determined from the model in Table 10. The habitat suitability increases with darker colouring. This map should be viewed with the caveats explained in the text and in Figure 15.

3.9 Distribution of nearest neighbours

New nests in new territories were most likely to be located between 5 and 15 km from the nearest established neighbour (Fig. 16). The probability of a new nest being between 5 and 10 km and between 10 and 15 km from the nearest neighbour was 0.25 and 0.23, respectively (Fig. 17). Very short (<1 km) and long (>100 km) nearest-neighbour distances, had the lowest probability of being occupied (0.022; Fig. 17).

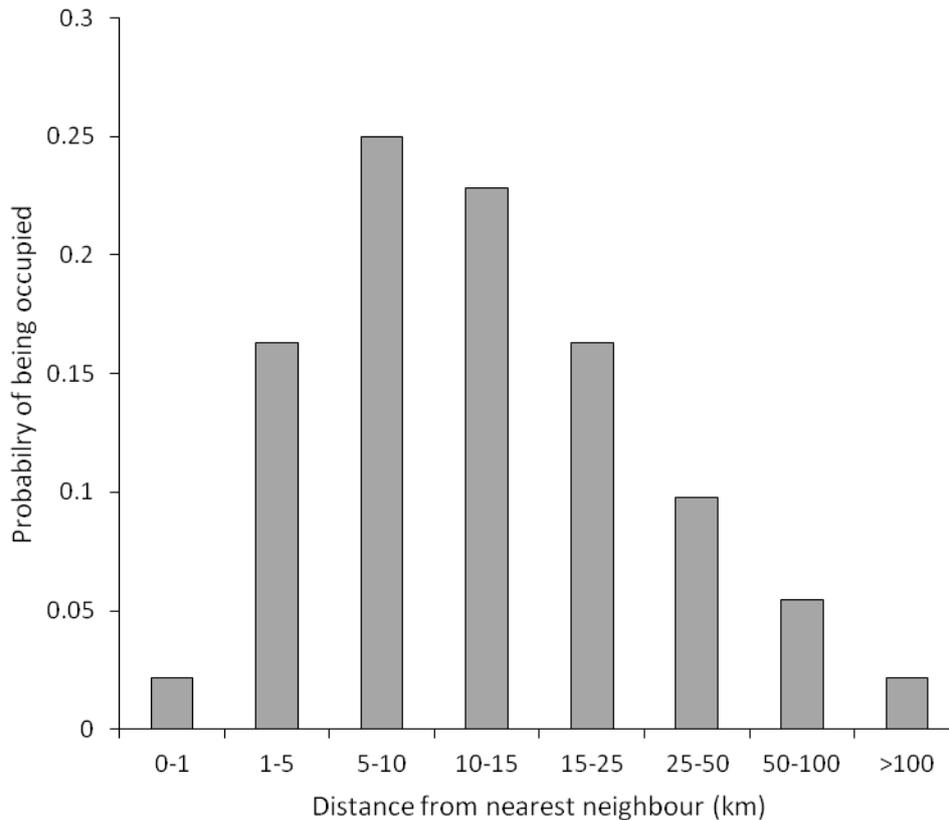


Figure 17. Probability of occupancy by a white-tailed eagle pair in relation to the distance from the nearest neighbour for "new" territory (nest) locations.

3.10 Predicted range expansion

Overall habitat suitability across Scotland for white-tailed eagles (Fig. 16) combined with probability of occupancy based on nearest neighbour distance (Fig. 17), were used to produce maps of predicted white-tailed eagle population range each year between 2020 and 2040. As the majority of the white-tailed eagles occur in the core breeding area in the west of Scotland, this area was used as a starting point for the modelled range expansion. In addition, as white-tailed eagles from the east coast releases have just reached breeding age, it is difficult to predict exactly where new pairs are likely to establish new territories in east and central Scotland.

For brevity, only the years 2025 and 2040 are shown here (Figs 18-19). By varying the strength of the habitat selection using habitat probability bands of 20% and 50% (see Methods), two scenarios were visualised. When the habitat probability bands were narrow (i.e. 20% bands; Fig. 18), the population range expansion became spatially restricted, compared with more relaxed habitat selection (50% bands; Fig. 19). This was particularly visible in year 2040 and most evident for the inner parts of the Isle of Lewis, and many inland areas of mainland Scotland (e.g. north and east of Loch Lomond and around the south part

of Loch Ness). The exceptions were the areas northeast of Loch Ness, the Kintyre peninsula and Arran, where the 20% probability band model suggested a slightly more widespread range expansion (cf. Figs 18 and 19). Overall, these model predictions suggest that if habitat preferences are very strong (with areas required to contain a high proportion of forest cover and large lengths of coastline), the future range might be more spatially restricted. Given that the habitat model on its own explained just 25% of the deviance in the data, a more relaxed habitat selection scenario (i.e. the 50% probability bands) is more likely to capture the uncertainty of where future range expansion will take place.

A general result from these models was that the probability of an individual 1 km square being predicted to be occupied was relatively low (typically below 0.5) in all models. This probably reflects the presence of a relatively large amount of moderately to highly suitable habitat within the optimal distance from other established nest locations. This meant that for each iteration of the model there was a higher likelihood that a different set of candidate squares would be predicted to be occupied. In the early years of range expansion (five to 10 years), this was especially true, with fewer pairs being spread across the suitable habitat areas available in each run (e.g. Fig. 18). However, all models suggested that areas of the Outer Hebrides and mid-Argyll had the highest likelihood of being most densely occupied in the future. There was also some predicted range expansion along the far north-west coast of Scotland (Fig. 18-19).

Tentatively, range expansion models that included the few established breeding pairs of white-tailed eagles in the east, central and far north of Scotland were also run. In these models, it became clear that predictions of future range expansion were very sensitive to where the first pioneering pairs settle in these parts of Scotland (Fig. 20). The pioneering pairs might act as nuclei for range expansion, and the predicted expansion in these areas is likely to be highly influenced by whether the birds remain on their current territories and where new territories are established. Given that the east coast population consists of mainly non-breeding individuals that are likely to start breeding in the next few years (at unknown locations), the output of these models are fraught with uncertainties. A cautious approach was therefore taken, in which only the predicted short-term range expansion for the year 2025 was visualised (Fig. 20). It would be prudent to wait to produce long-term predictions for these areas until most of the surviving white-tailed eagles from the east coast release have started to breed and re-run models at that point.

3.10.1 Assessing accuracy of breeding range expansion predictions

The nest site data from the core breeding area along the west coast of Scotland was partitioned into two time periods; "early" (1983-2012) and "late" (2013-2014). This made it possible to test how well the models presented in this report could predict where new nests in 2013 and 2014 would be located. The assessment compared how effective "the full model" of habitat associations and distance to nearest established nest was in predicting the location of new real nests with models containing only habitat associations and only nest distance. Finally, a "null model", in which 1 km squares could be occupied at random within 70 km of established nests without regards to nest distance and habitat associations, was also included in the comparisons (see section 2.2.11 for details).

The results of the comparisons suggested that the models' effectiveness in predicting a real nest site location increasing with increased search distance (i.e. a more relaxed approach for identifying an occupied site; Fig. 21).

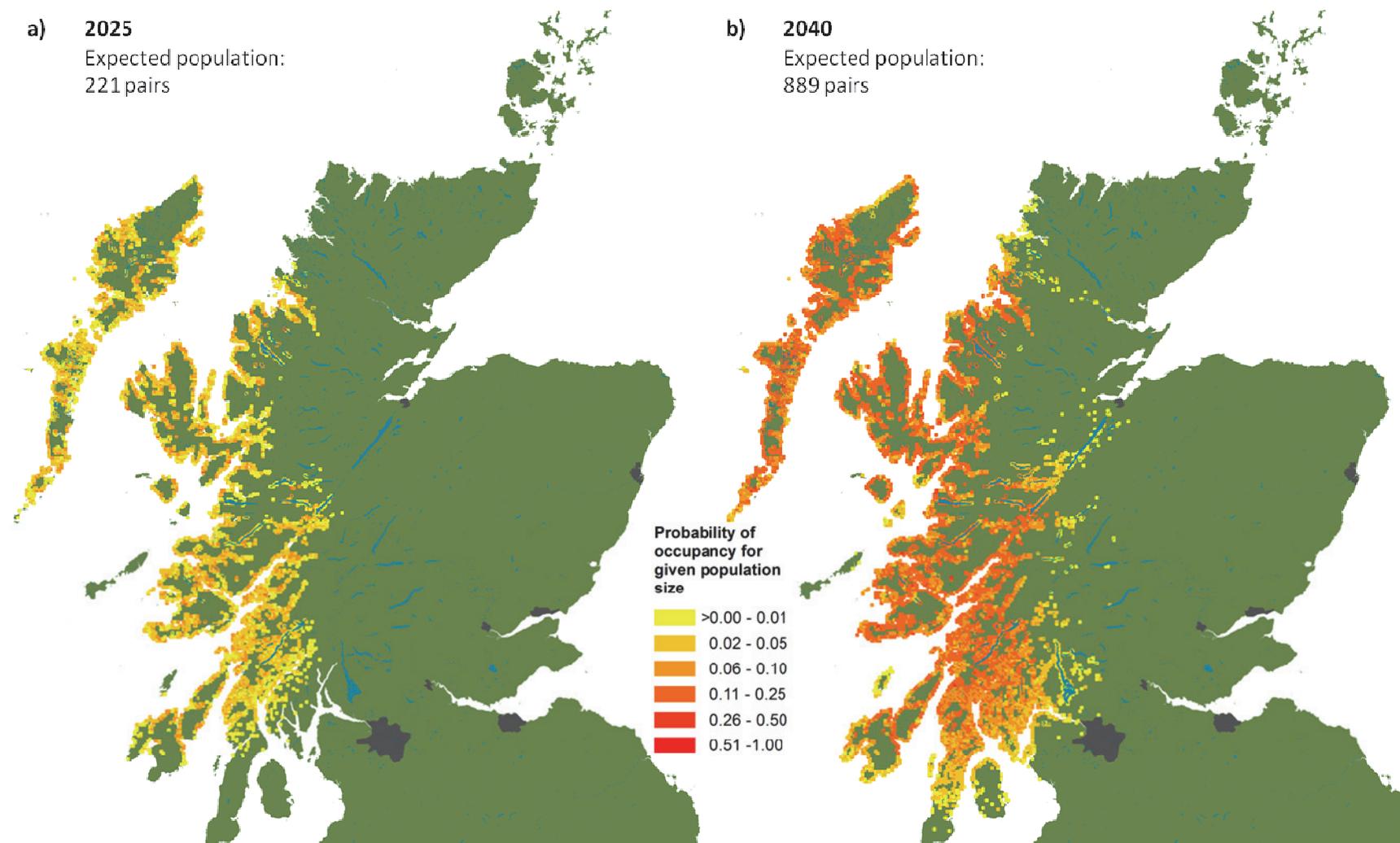


Figure 18. Probability of 1x1 km squares being occupied by a breeding pair of white-tailed eagles, based on the habitat suitability (Fig. 16) divided into 20% probability bands, distance from nearest neighbour (Fig. 17) and predicted population size in 2025 and 2040, as determined from 100 iterations of the range expansion model. Dark grey areas are cities and blue areas are water bodies (based on the LCM2000).

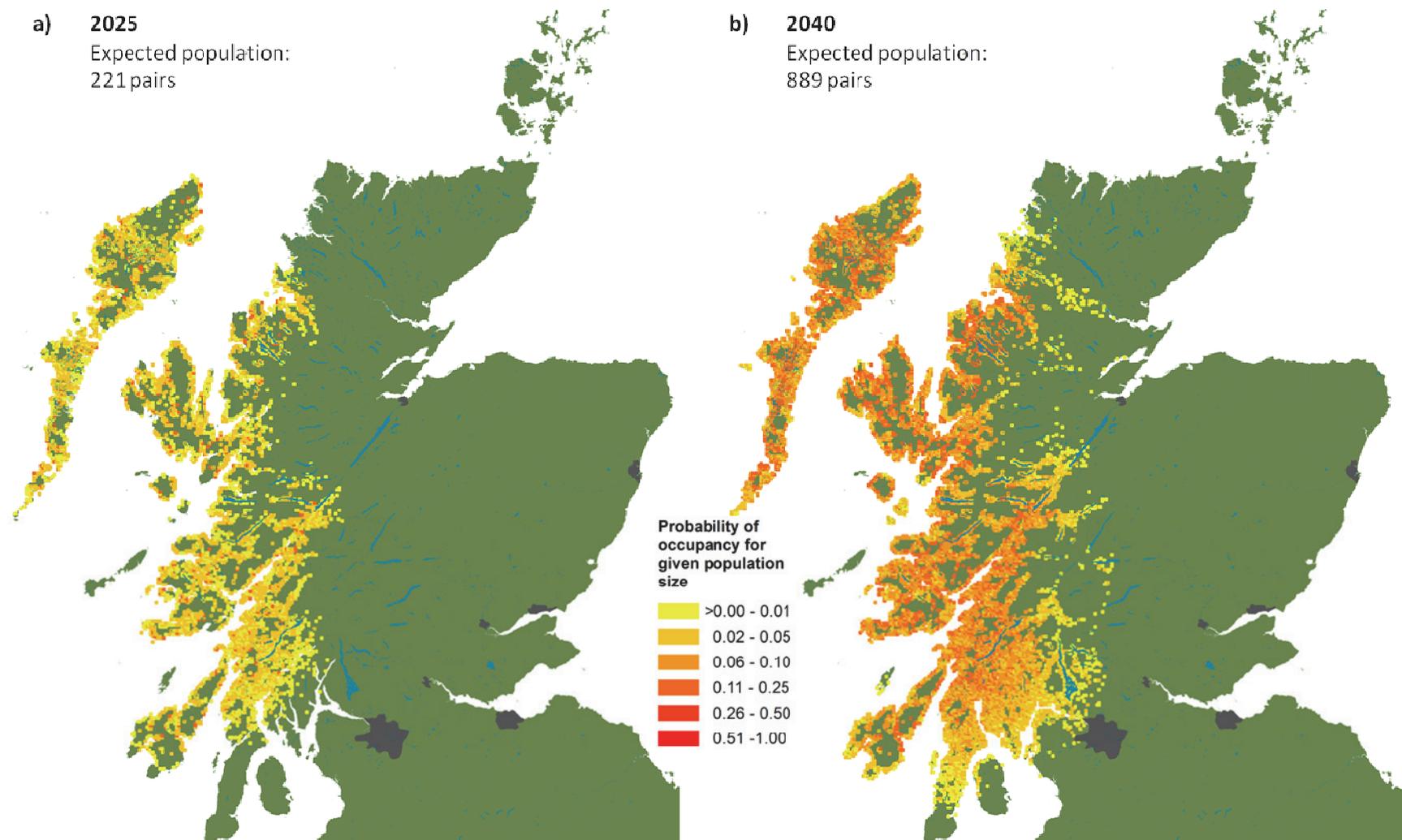


Figure 19. Probability of 1x1 km squares being occupied by a breeding pair of white-tailed eagles, based on the habitat suitability (Fig. 16) divided into 50% probability bands, distance from nearest neighbour (Fig. 17) and predicted population size in 2025 and 2040, as determined from 100 iterations of the range expansion model. Dark grey areas are cities and blue areas are water bodies (based on the LCM2000).

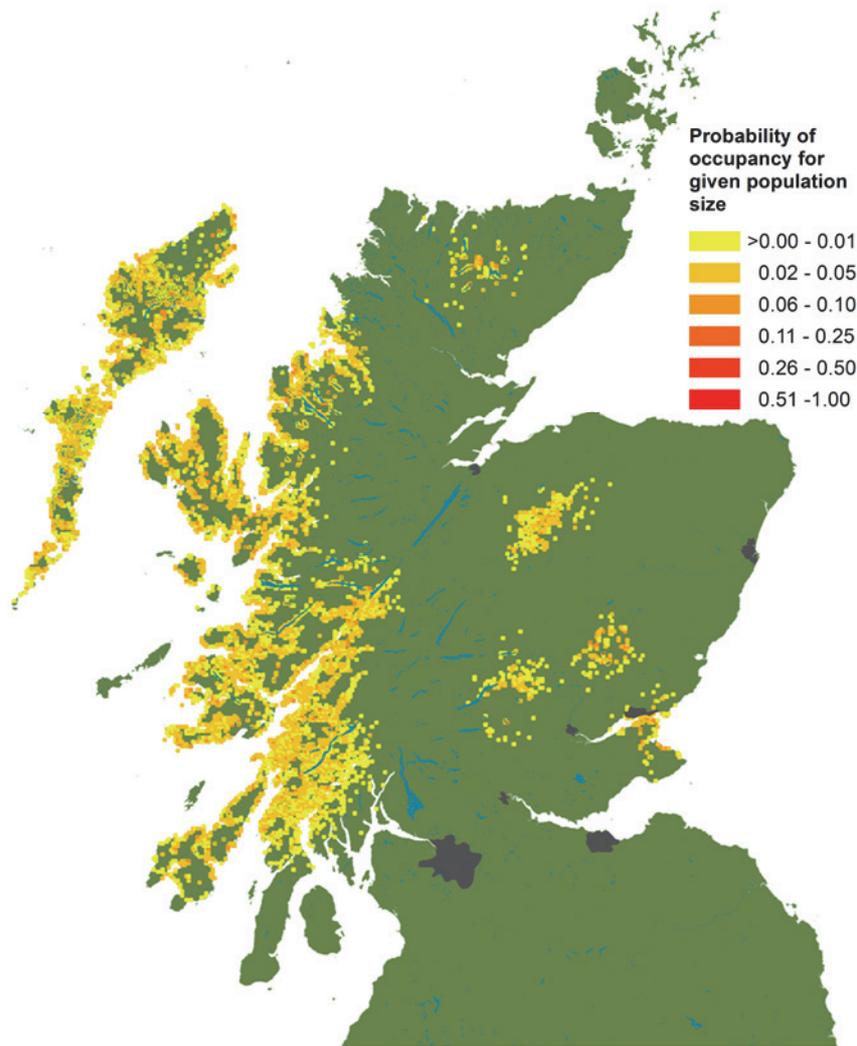


Figure 20. Probability of 1x1 km squares being occupied by a breeding pair of white-tailed eagles, based on the habitat suitability (Fig. 16) divided into 50% probability bands, distance from nearest neighbour (Fig. 17) and predicted population size in 2025, as determined from 100 iterations of the range expansion model. Dark grey areas are cities and blue areas are water bodies (based on the LCM2000). This map includes both west and east coast pairs. The strong effect of already established pairs on the east coast is evident, and it is therefore important to bear in mind that new pairs establishing territories in other parts of north, east and central Scotland might affect the range expansion in ways that are difficult to predict.

Not surprisingly, the models were also better at predicting the location of new nests in the core breeding area (i.e. the solid lines in Fig. 21) than predicting all nests across Scotland (i.e. the dashed lines in Fig. 21). In addition, in order for the model to effectively predict new nest locations, both habitat suitability and distance to established nests should be accounted for (Fig. 21). However, this was not apparent when using *all* sites that the models predicted to be occupied to intersect the real nest site locations (Fig. 21a). In fact, at the widest search distance (3 km), the null model performed as well as the full model (Fig. 21a). However, when only using sites that the models predicted to be occupied two or more times, the effectiveness of the full model in predicting the location of real nests was evident (Fig. 21b). Models containing only habitat associations or only distance to established nests were consistently less accurate in predicting where new nest sites would appear. This was particularly true for scenarios when using only sites that the models predicted to be occupied two or more times (Fig. 21b).

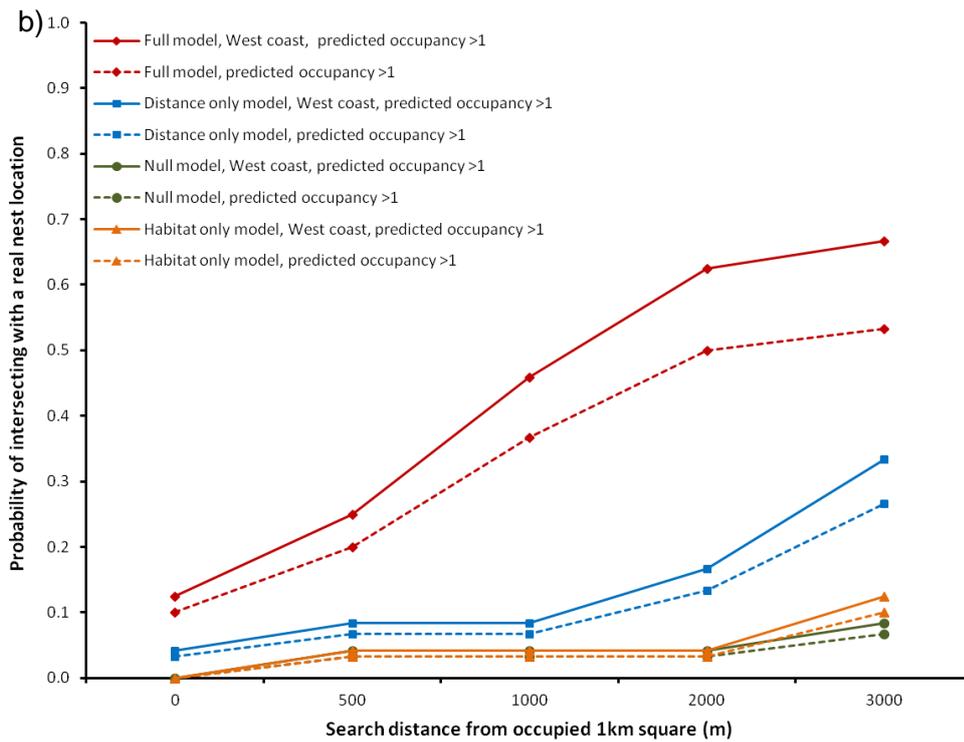
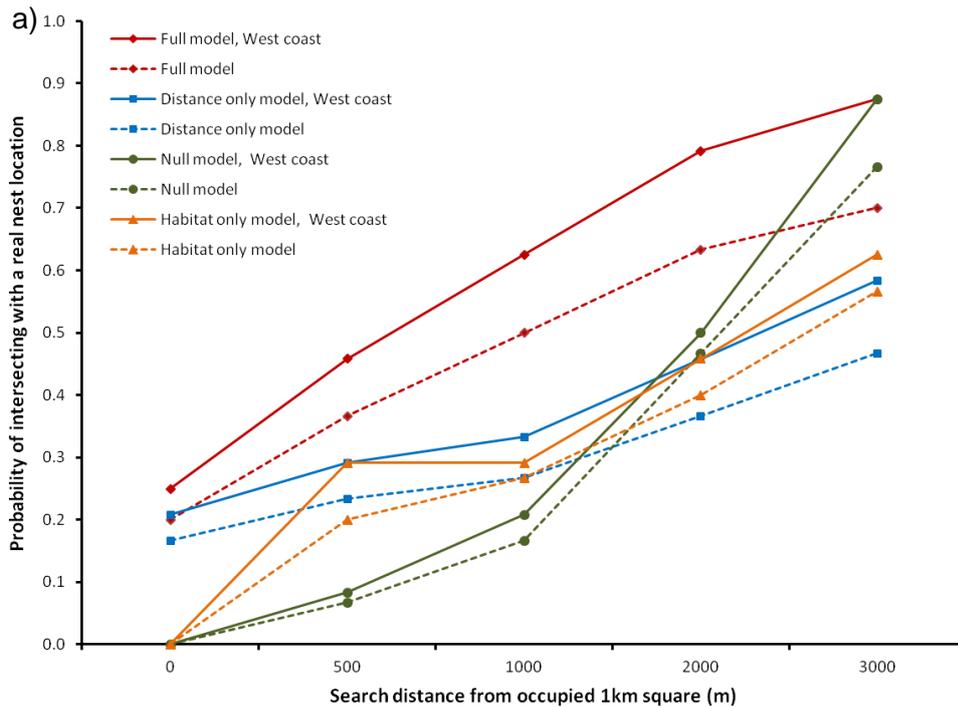


Figure 21. The probability that predicted white-tailed eagle nest sites were located within search distances of 0 to 3000 m from real nests in 2013 and 2014. Different scenarios are shown, with model complexity varying from simple "null models" with a complete random location of predicted nests to "full models" that located nests by taking habitat associations and distance to established nest into account. Separate models for the west coast core breeding area (solid lines) and all nests across Scotland (dashed lines) are shown. In a), models using any predicted nest site location to intersect real nest sites are shown, whereas in b), only nest site locations predicted more than once are used to intersect the real nest sites are shown.

4. DISCUSSION

4.1 Population growth and density-dependent population regulation

Previous best estimates of predicted population growth (i.e. Evans *et al.*, 2009) correctly predicted the almost exponential population growth up to approximately year 2002 (Fig. 10). However, the increasing proportion of wild-bred individuals (Fig. 5), with higher survival than released birds (Table 8), and experienced breeders (i.e. high breeding success; Fig. 7) in the current Scottish population, means that the previous models slightly underestimated population growth since 2010. This report has updated the estimates of breeding success and age-specific survival rates, and by using these updated estimates as well as the gradual increase in the proportion of wild-bred individuals in population models, the results more accurately predict the recent past (i.e. 2003-2014). However, the updated model shows a tendency to underestimate the population growth since 2011 (Fig. 10), but this error is comparatively small. With the population now comprising almost entirely wild-bred birds (cf. Fig. 5), it is possible that demographic parameters reflecting wild-bred birds will now be better at predicting population growth further into the future. For example, after 2040, most released birds, even from the most recent releases, will no longer be alive. In summary, the results presented here suggest a sustained strong numerical increase of the breeding population size of white-tailed eagles in Scotland.

If population growth continues at the predicted or currently observed rate, the white-tailed eagle population is expected to have between 889 and 1,005 territorial pairs by 2040. However, crucially, most of models presented here did not specify any density-dependence in either breeding success or survival, and they assumed an infinite amount of available habitat and nest sites (i.e. no carrying capacity was set). As most examples of density-dependent population regulation in UK raptors come from small-bodied species (e.g. sparrowhawks *Accipiter nisus* and kestrel *Falco tinnunculus*; Newton, 1986; Village, 1990), there are no previous data from the UK on how potential density-dependence might regulate white-tailed eagles. For most large raptors in the UK, such as goshawks (*Accipiter gentilis*), golden eagle (*Aquila chrysaetos*) and white-tailed eagles, population sizes have been regulated by other factors. For example, illegal killing, deforestation (particularly for goshawks) and direct and indirect poisoning from organo-chlorine pesticides have been identified as causes of population change in these species (Rutz *et al.*, 2006; Newton, 2013). As there have been systematic increases in the population size of these species following the introduction of measures to limit the effects of these factors, and populations have not yet fully recovered, it has been difficult to establish the strength of the density-dependence and whether it would act primarily on mortality, reproduction or both.

Most newly founded populations exhibit a logistic population growth rate with three phases: establishment, expansion and saturation. The expansion phase often involves an exponential growth trajectory but in the saturation phase the population growth rate levels off, indicating that the population is limited by an extrinsic factor, such as food or suitable nest sites. The almost exponential growth rate of the Scottish white-tailed eagle population suggests that the population is currently in the expansion phase. The results presented here suggest that there is no indication of a decrease in either breeding success or age-specific survival in Scotland, suggesting that the Scottish white-tailed eagle population is not regulated by density-dependent factors yet. In addition, using the area with highest density of breeding white-tailed eagle pairs, the islands of Mull and Skye, there was no evidence that the reproduction had decreased with time (Fig. 8) or when the density of white-tailed eagles increased to current levels (e.g. 0.024 and 0.011 pairs per 1 km², respectively; Fig. 9).

However, studies in continental Europe suggest that top predators such as golden eagles and goshawk might be regulated by density-dependent breeding success. In Italy, the proportion of golden eagles that laid eggs and fledged at least one chick declined with

increasing density (Fasce *et al.*, 2011). As it was mainly in the early stages of the breeding cycle nest failures occurred, Fasce *et al.* (2011) suggested that it was either food competition or interference from floating non-breeding birds or neighbouring pairs that reduced the breeding productivity. For goshawks, Rutz *et al.* (2006) reviewed the evidence for density-dependent population limitation, and found several examples in which both access to suitable habitat (woodland) and prey availability could limit goshawk populations in a density-dependent fashion.

When density-dependent population regulation starts to occur in the Scottish white-tailed eagle population, it is unlikely that the strong population growth will continue. Instead, it is more plausible that population growth will decrease and the population size level off at a certain density. Ecological processes likely to affect density-dependence and carrying capacity might be intra-specific competition for habitat and resources, such as food and nest sites. In areas with high density of white-tailed eagles, for example in Norway, Germany and increasingly on Mull and Lewis in Scotland, a number of white-tailed eagles have been found dead with injuries that suggest that they have been killed by other white-tailed eagles or golden eagles, and there are indications from Norway that this mortality is particularly high among sub-adults and non-territorial adults (Whitfield *et al.*, 2008; Krone *et al.*, 2013; D. Sexton, pers. comm.; T. Nygård, pers. comm.).

In Chesapeake Bay in the USA, bald eagles (*Haliaeetus leucocephalus*) increased the amount of time guarding their nests from other bald eagles when the population density increased, thereby reducing the time they were able to allocate for foraging (Turrin & Watts, 2015). This could reduce the breeding success both for individual pairs and for the population as a whole. Similar processes might occur in white-tailed eagle populations in Europe but, to our knowledge, few studies exploring behavioural responses to increased intra-specific densities have taken place yet (but see Krone *et al.*, 2013). Interestingly, in comparison with the survival estimates for the age transition three to four years presented in Evans *et al.* (2009; i.e. 0.922 ± 0.058 and 0.951 ± 0.038 for released and wild-bred birds, respectively), the equivalent updated survival rates presented here for the same population (i.e. the Scottish west coast population) are lower for both released and wild-bred birds (cf. Table 8). The survival rates for the age transition four to five years are also lower (0.757 ± 0.060 for released birds and 0.855 ± 0.036 for wild-bred birds), but Evans *et al.* (2009) did not include this age-transition in their study, so straightforward comparisons are difficult. The indication that survival rates in the years immediately before the white-tailed eagles settle in a territory for their first breeding attempt have declined over time, could suggest that there is now increased competition for breeding territories leading to increased mortality. If this is the case, it suggests that density-dependent survival may have already started to act regionally in Scotland, but has not yet manifested itself in reduced population growth rate.

There is also some suggestion that interactions with golden eagles can be fierce, even fatal for either species (Whitfield *et al.*, 2008). As the diet of the two eagle species does not overlap significantly during the breeding season (Whitfield *et al.*, 2013) and there is no evidence that the two species compete for a limited number of nest sites (Evans *et al.*, 2010), the underlying factors for the few witnessed fierce interactions are unclear. In the longer term, it is possible that the distribution and density of both white-tailed and golden eagles might change, but most of the evidence to date indicates that the two species are able to coexist. The direct competition is probably reduced by partitioning of habitat within the landscape, largely by altitude and variation in altitude (golden eagles seem to prefer high variation and white-tailed eagle prefer landscapes with low variation in altitude; Evans *et al.*, 2010). This study (e.g. Fig. 16) combined with historical records of the distribution of both white-tailed and golden eagles in Britain and Ireland (e.g. Evans *et al.*, 2012) suggests that there is a super-abundance of unoccupied suitable habitat, especially at low altitudes. However, there is a probability that competition for food, nest sites and space might increase when the distribution and density of the two eagle species increase. The big current

knowledge gap is related to fine-scale foraging habitat preferences in areas where the two species occur side by side, and to examine whether this might result in density-dependent population regulation.

With only limited suggestions at present of, at most, a weak density-dependent regulation of white-tailed eagles only in one or two areas, and no evidence of density-dependent population growth nationally in Scotland, the results presented here come with the caveat that density-dependence might occur more widely in the future, and that such regulation will reduce the presented population growth estimates. It is also likely that the maximum density will vary regionally across Scotland, mainly based on food abundance and availability of nest sites. This report tentatively gives illustrative examples of how density-dependence national population growth curves might look like when the carrying capacity was set to between 2,000 and 4,000 birds (Fig. 14), but currently the data cannot show whether density-dependence will act on breeding productivity, survival or both. Once the carrying capacity has been reached, either before or after the end of the time period of this report, it is likely that the population will stabilise around that density, unless there are changes in regulating factors, such as food abundance, nest site availability and density of competitors (see also Appendix 2 for an exploration of potential carrying capacities in different Scottish regions).

4.2 Breeding success

White-tailed eagle breeding outcome (i.e. the proportion of nests producing at least one fledgling) and the mean number of fledglings produced per breeding attempt have shown a continued statistically significant increase since the first breeding attempt in Scotland in 1983 (Fig. 6). This is partly because an increasing number and proportion of birds in the population have gained breeding experience, which is linked to high breeding success (Fig. 7). However, there is also some evidence that breeding success has now stabilised nationally and regionally (e.g. Figs 6 and 8). It is possible that the ratio of experienced to inexperienced breeders in the population has become more stable, and that ecological limitations (e.g. food abundance) might prevent further increases in the mean number of fledglings produced per territory.

4.3 Additional mortality

Additional mortality from illegal killing (based on historical levels observed in Scotland; RSPB, 2013; RSPB Scotland, 2013) and collisions with wind farms at levels observed in Norway (May *et al.*, 2010), respectively, appear to have the potential to reduce the total number of breeding pairs and to slow population growth rates. However, additional mortality at the rates chosen for our scenarios does not stop population growth, even at relatively high levels of additional mortality (14 individuals per year; Fig. 12). This suggests that cumulative impacts of various types of mortality factors, at the levels tested in this study, are not likely to cause a population decline, at least in the next 25 years (i.e. up to year 2040; Figs 12-13). However, it is once again important to note that most population growth models presented here and the predicted population sizes did not include any density-dependent effects or habitat limitations, which may become important in limiting population growth in the future.

It is possible that increased mortality caused by collisions at wind farms or increased levels of illegal killing could have a large influence on where new pioneering pairs establish themselves; consequently, these pressures could restrict population growth at a local or regional scale, even if they are unlikely to do so nationally (e.g. Whitfield *et al.*, 2004b, Smart *et al.*, 2010). This could slow the recovery of the former breeding range in Scotland, especially when the white-tailed eagle population expands into areas that historically have been associated with high levels of illegal killing of raptors (Green & Etheridge, 1999, Whitfield *et al.*, 2003). One of the key objectives in the third phase of the white-tailed eagle release programme in east Scotland was to increase the speed of geographical range

recovery, and the first breeding pairs from this release phase have indeed increased the breeding distribution of the species.

4.4 Geographical range expansion

The breeding habitat associations of white-tailed eagles suggested that they select nest sites close to water (i.e. coastal or freshwater), woodlands and in flatter topography (regardless of altitude). This accords with studies from continental Europe and previous work in Scotland (Evans *et al.*, 2010). For example, both Radović & Mikusca (2009) and Sandor *et al.* (2015) showed that white-tailed eagles nested close to large alluvial wetlands but away from human settlements in Croatia and Romania, respectively. Krone *et al.* (2013) used radio-tracking to examine home range characteristics of white-tailed eagles in northern Germany. Their results suggested that white-tailed eagles tended to occupy home ranges characterised by shallow and fish-rich freshwater lakes with undisturbed nesting areas in surrounding forests.

This and other studies show the wide range of wetland and coastal landscapes that white-tailed eagles will utilise across their distribution (e.g. eastern European floodplains, Norwegian fiords, German lakes and Baltic islands). The common factor is the presence of water, regardless of whether it is fresh water, brackish or salt water). Clearly, white-tailed eagles breed successfully in widely different landscapes, which suggest that also in Scotland, the expansion will not be restricted to coastal areas only.

As the Scottish population so far has been mainly restricted to the west coast of Scotland, the habitat associations presented in this report may not fully capture the plasticity of the species' breeding habitat selection at a landscape scale. Thus, as the white-tailed eagles from the third release on the east coast of Scotland start to occupy breeding territories in central and eastern Scotland, it would be informative to update the eagle-habitat associations to improve the predictions of where new pairs might settle, and to improve our limited knowledge of eagles' selection of foraging habitat patches within the landscape, particularly in relation to nest site location. It should also be noted that white-tailed eagles can breed in locations close to human activity (e.g. fish farms, near popular walking and canoeing routes and near houses). Just like other avian top predators for which illegal killing levels have dropped regionally, habituation to local conditions with human presence can make previously unoccupied areas suitable. For example, goshawks have started to nest in inner-city environments in northern Germany and peregrines often nest in industrial or highly populated environments (Rutz *et al.*, 2006; Banks *et al.*, 2010).

The model of associations between white-tailed eagle breeding sites and habitat, landscape and topographical variables presented in this report explained around 25% of the variation between real nest sites and non-occupied (random) points. Although the explanatory power of this model seems modest, the ROC analysis showed that our habitat suitability model was good at predicting the presence of breeding white-tailed eagles. The model had an Area Under the Curve (AUC) of 0.829 (95% C.I. of 0.780-0.888), which is well above the 0.700 that Swets (1988) identified as a threshold value for a model to be considered useful. Therefore, the habitat suitability map (Fig. 16) will give a good indication of where white-tailed eagles will start to breed in the future, but the exact order of settlement is more difficult to predict. However, it is important to remember that the habitat suitability map was built on the range of values observed in the core breeding area along the west coast of Scotland. There are areas across Scotland that have values that are outside the range of values observed within the core breeding area, notably regarding the topography (e.g. Fig. 15). Thus, the habitat suitability map (Fig. 16) should be viewed in conjunction with the map showing areas where the habitat suitability model might be less applicable (i.e. Fig. 15).

Nonetheless, by using the habitat associations (i.e. Figs 15-16) in conjunction with information on where new territories have been established in relation to active conspecific

nests in Scotland (Fig. 17), a gradual geographical range expansion is predicted. This starts along the west coast with subsequent filling up of suitable habitats along the north-west and north coast of Scotland, as well as the Outer Hebrides, the Great Glen and other inland areas. Depending on how strictly the eagle-habitat associations were applied in the models, the rate of range expansion varies, being quicker when habitat associations were less strict (cf. Figs 18-19). It was also clear that the first pioneering pairs settling in an area might act as nuclei for subsequent local and regional population growth (e.g. Fig. 20). Specifically, there is low likelihood of any one new site being located at a very long distance from its nearest neighbour, but a much higher likelihood of a subsequent new site being close to the original long-distance one. With more young white-tailed eagles in the west now and with birds from the east coast releases likely to establish new territories in the next few years, the likelihood of occasional unexpected jumps in range has increased. One such example is the pair of white-tailed eagles that set up a territory on the island of Hoy in Orkney in 2015. The models presented here did not predict such a re-colonisation of Orkney in the near future, however the lack of wing tags and colour rings on these birds makes it difficult to identify the origin of these birds.

The assessment of the effectiveness the range expansion model suggested that when using the relatively relaxed definition of a "site" (e.g. within 2-3 km of a predicted nest site), the model can predict reasonably well where new white-tailed eagle territories will be established. At the scale of a country the size of Scotland, distances of 2-3 km are small. Therefore, the full range expansion model presented here, in which both associations between white-tailed eagle nest locations and habitat as well as distances between new and established nests are taken into account, can predict the general areas where white-tailed eagles are likely to occupy within the next years and decades (e.g. Figs 18-20). However, it is important to bear in mind that the predictions are most accurate in the short-term and especially along the west coast of Scotland.

There was no density-dependence included in either the model of population growth or range expansion so, in the models presented here, new pairs could settle without restriction on density except for 1 km squares that were already occupied, which were treated as not available. Studies from elsewhere in Europe suggest that white-tailed eagles in extreme cases can nest as close as around 350 m from each other, but the mean distance is normally several kilometres (Radović & Mikusca, 2009). It is likely that under density-dependent scenarios, new pairs are unable to fill the gaps between existing territories. This might result in delayed age of first breeding (i.e. whilst waiting for a vacancy in an existing territory), or more likely a faster range expansion (i.e. individuals have to disperse further to find vacant suitable habitat). In the latter scenario, the pioneer pairs might act as "seeds" for new territory clusters in previously unoccupied areas.

A factor that might affect the settlement order, the natal and breeding dispersal distances, and ultimately the geographical range expansion of white-tailed eagles, is forest management. The Scottish Government has a target of increasing woodland cover from 17% to 25% in this century, and this will in time create new nesting sites for white-tailed eagles. However, in the short to medium term, many forest plantations will be clear-felled, restocked, and also restructured. The restructuring will include changes such as replanting with a higher proportion of the woodland with deciduous trees compared with previous plantations, which mainly consisted of non-native Sitka spruce, *Picea sitchensis*, retention of some trees and forest blocks (i.e. "eternity trees") as well as creation of wider margins around water features, boggy and rocky areas, which will create more open space within the forest. Taken together, the restructuring could make some forest blocks more suitable for white-tailed eagles in the long term. Apart from direct disturbance from forestry operations (e.g. Kortland *et al.*, 2011, Santangeli *et al.*, 2013), indirect disturbance from roads created to facilitate forestry operations and the subsequent higher number of recreational visitors to the area might cause redistribution of existing pairs and potentially make some areas

unsuitable for white-tailed eagles. The effects of such large-scale changes in forestry policy on white-tailed eagle population dynamics requires study, but is beyond the scope of this report.

4.5 Effect of increasing numbers and spatial range of white-tailed eagles on other species

The increased size and geographical range of the white-tailed eagle population in Scotland may affect other species of birds, either directly through the predation of individuals or indirectly via predator avoidance behaviour of prey species. In addition, intra-guild predation and competition between white-tailed eagles and other raptors may affect distribution of these raptor species, at least locally.

Of the prey species, it is likely that the increased number of white-tailed eagles will increase the predation rates of some seabird species and perhaps also of breeding and wintering geese. For example, in northern Norway, white-tailed eagles have been suggested to have contributed to the demise of some gannet colonies (Pettex *et al.*, 2015). In a review of impacts of eagles of the genus *Haliaeetus* on seabirds, Hipfner *et al.* (2012) found that many colonies and sub-colonies of a wide variety of seabirds in the northern hemisphere have been negatively affected by recovering eagle populations. The effects of these eagle populations range from direct effects (e.g. predation of adults and chicks) and indirect effects (e.g. by inducing anti-predation behaviour that minimises time to forage and facilitating nest predators, mainly gulls and corvids, to take eggs and young seabirds). White-tailed eagle predation may also have contributed towards the male-biased sex ratio of eiders (*Somateria mollissima*) in the Gulf of Finland by predominantly depredating incubating female eiders (Lehikoinen *et al.*, 2008). On the other hand, white-tailed eagles have reduced dispersal distance in the non-native American mink (*Neovison vison*) in the Finnish Baltic archipelago by direct predation of mink swimming between the islands as well as induced anti-predator behaviour of mink, such as avoidance of long-distance swimming between islands (Salo *et al.*, 2008). This may have positive effects on the local seabird community, which suffer from heavy predation by American mink (Nordström & Korpimäki, 2004).

Similarly, it is likely that at least locally some populations of breeding and wintering geese may be subjected to higher predation rates by white-tailed eagles. However, few quantitative studies of this have been done, and it is therefore unknown how such predation will affect population trends of wintering and breeding geese numbers in Scotland.

As discussed above, it is likely that competition for nest sites, food or just general territorial space between white-tailed and golden eagles might increase in intensity when the white-tailed eagle numbers increase. The slightly larger size of white-tailed eagles suggests that they will win the majority of territorial fights. On the other hand, the higher flight manoeuvrability of golden eagles suggests that they would be stronger competitors in aerial fights. There is very little evidence to suggest which of the species would be most negatively affected. The Scottish golden eagle population might be negatively affected also by small reductions in breeding productivity, so it would be of interest to monitor the effects of an expanding white-tailed eagle population on golden eagles in the future. The national golden eagle survey carried out in 2015 would be an ideal starting point for such analyses.

In addition, it would be interesting to study how the expanding population of white-tailed eagles affect other raptor species of conservation concern. In particular, the hen harrier (*Circus cyaneus*) breeds in high densities on some Scottish islands (e.g. Orkney, Uist, Mull and Skye), and it is possible that increased abundance of white-tailed eagles might have negative impact on hen harriers on these islands. In an international perspective, it is very rare that the geographical breeding range of hen harriers and white-tailed eagles overlap, so

it would be interesting to monitor the effects of white-tailed eagles on hen harriers in Scotland.

5. CONCLUSIONS

In conclusion, Scottish white-tailed eagles are breeding more successfully now relative to the early years of the re-introduction programme, probably because the proportion of eagles with extensive breeding experience has increased over the years. The results presented in this report suggest that the number of breeding pairs is predicted to continue increasing rapidly in the next 25 years. Even if reproduction and survival became regulated in a density-dependent way, the observed and predicted population growth suggests that the risk of extinction is extremely low, and that the reintroduction programme has been a conservation success. The models of geographical range expansion suggest that the west coast of Scotland will remain the stronghold for the species in the foreseeable future. A gradual expansion along the west and north coast of Scotland, the Great Glen and in areas where pioneer pairs from the east coast release will establish themselves is also likely, but the models presented in this report should be treated cautiously and can only serve as an indication of areas where eagles might settle to breed in the next 10 to 25 years. The sequence of settlement of these areas is difficult to predict, and would be strongly influenced by any long-distance jumps in range, which the models have necessarily treated as individually having a low likelihood.

6. REFERENCES

- Balmer, D. E., Gillings, S., Caffrey, B. J., Swann, R. L., Downie, I. S. & Fuller, R. J. 2013. *Bird Atlas 2007-2011: the breeding and wintering birds of Britain and Ireland*. BTO Books, Thetford, UK.
- Banks, A. N., Crick, H. Q. P., Coombes, R., Benn, S., Ratcliffe, D. A. & Humphreys, E. M. 2010. The breeding status of Peregrine Falcons *Falco peregrinus* in the UK and Isle of Man in 2002. *Bird Study*, 57, 421–436.
- Burnham, K. P. & Anderson, D. R. 2002. *Model Selection and Multi-Model Inference: A Practical Information-Theoretic Approach*. Springer.
- Centre for Ecology and Hydrology. 2000. *LCM 2000*. Centre for Ecology and Hydrology.
- Dahl, E. L., Bevanger, K., Nygard, T., Roskaft, E. & Stokke, B. G. 2012. Reduced breeding success in white-tailed eagles at Smøla windfarm, western Norway, is caused by mortality and displacement. *Biological Conservation*, 145, 79-85.
- ESRI. 2011. *ArcGIS Desktop. Environmental Systems Research Institute*, 10.1, Redlands, CA, USA.
- Etheridge, B., Summers, R. W. & Green, R. E. 1997. The effects of illegal killing and destruction of nests by humans on the population dynamics of the hen harrier *Circus cyaneus* in Scotland. *Journal of Applied Ecology*, 34, 1081-1105.
- Evans, R. J., O'Toole, L. & Whitfield, D. P. 2012. The history of eagles in Britain and Ireland: an ecological review of placename and documentary evidence from the last 1500 years. *Bird Study*, 59, 335-349.
- Evans, R. J., Pearce-Higgins, J., Whitfield, D. P., Grant, J. R., MacLennan, A. & Reid, R. 2010. Comparative nest habitat characteristics of sympatric White-tailed *Haliaeetus albicilla* and Golden Eagles *Aquila chrysaetos* in western Scotland. *Bird Study*, 57, 473-482.
- Evans, R. J., Wilson, J. D., Amar, A., Douse, A., MacLennan, A., Ratcliffe, N. & Whitfield, D. P. 2009. Growth and demography of a re-introduced population of White-tailed Eagles *Haliaeetus albicilla*. *Ibis*, 151, 244-254.
- Fasce, P., Fasce, L., Villers, A., Bergese, F. & Bretagnolle, V. 2011. Long-term breeding demography and density dependence in an increasing population of Golden Eagles *Aquila chrysaetos*. *Ibis*, 153, 581-591.
- Green, R. E. & Etheridge, B. 1999. Breeding success of the hen harrier *Circus cyaneus* in relation to the distribution of grouse moors and the red fox *Vulpes vulpes*. *Journal of Applied Ecology*, 36, 472-483.
- Green, R. E., Pienkowski, M. W. & Love, J. A. 1996. Long-term viability of the re-introduced population of the white-tailed eagle *Haliaeetus albicilla* in Scotland. *Journal of Applied Ecology*, 33, 357-368.
- Griffith, B., Scott, J. M., Carpenter, J. W. & Reed, C. 1989. Translocation as a species conservation tool: status and strategy. *Science*, 245, 477-480.
- Guénette, J.-S. & Villard, M.-A. 2004. Do empirical thresholds truly reflect species tolerance to habitat alteration? *Ecological Bulletins*, 51, 163–171.

Hardey, J., Crick, H. Q. P., Wernham, C. V., Riley, H. T., Etheridge, B. & Thompson, D. B. A. 2013. *Raptors: a field guide for surveys and monitoring*. Scottish Natural Heritage, Inverness, UK.

Hipfner, M. J., Blight, L. K., Lowe, R. W., Wilhelm, S. I., Robertson, G. J., Barrett, R., Anker-Nilssen, T. & Good, T. P. 2012. Unintended consequences: how the recovery of sea eagle *Haliaeetus* spp. populations in the northern hemisphere is affecting seabirds. *Marine Ornithology*, 40, 39-52.

IUCN/SSC. 2013. *Guidelines for reintroductions and other conservation translocations*. 57 pp. IUCN Species Survival Commission, Gland, Switzerland.

Kortland, K., Evans, R., Douse, A. & Patterson, G. 2011. Managing forests for white-tailed eagles. 10pp. Practice Note, Forestry Commission Scotland.

Krone, O., Nadjafzadeh, M. & Berger, A. 2013. White-tailed Sea Eagles (*Haliaeetus albicilla*) defend small home ranges in north-east Germany throughout the year. *Journal of Ornithology*, 154, 827-835.

Krüger, O., Grünkorn, T. & Struwe-Juhl, B. 2010. The return of the white-tailed eagle (*Haliaeetus albicilla*) to northern Germany: Modelling the past to predict the future. *Biological Conservation*, 143, 710-721.

Krüger, O., Chakarov, N., Nielsen, J. T., Looft, V., Grünkorn, T., Struwe-Juhl, B. & Møller, A. P. 2012. Population regulation by habitat heterogeneity or individual adjustment? *Journal of Animal Ecology*, 81, 330-340.

Laake, J. L. 2013. *RMark: An R interface for analysis of capture-recapture data with MARK*. Alaska Fisheries Science Center, NOAA, National Marine Fisheries Service, Seattle, WA, USA.

Lacy, R. C., Borbat, M. & Pollak, J. P. 2005. *Vortex: A stochastic simulation of the extinction process*. Chicago Zoological Society, 9.50, Brookfield, IL.

Lehikoinen, A., Christensen, T. K., Öst, M., Kilpi, M., Saurola, P. & Vattulainen, A. 2008. Large-scale change in the sex ratio of a declining eider *Somateria mollissima* population. *Wildlife Biology*, 14, 288-301.

Love, J. A. 1983. *The Return of the Sea Eagle*. Cambridge University Press, Cambridge, UK.

Love, J. A. 2003. A history of the White-tailed Sea Eagle (*Haliaeetus albicilla*) in Scotland. In: Marquiss, M., Madders, M., Irvine, J. and Carss, D. N. (eds.), *The impact of white-tailed eagles on sheep farming on Mull*. 48 pp. Centre for Ecology and Hydrology.

Love, J. A. & Ball, M. E. 1979. White-tailed sea eagle *Haliaeetus albicilla* reintroduction to the Isle of Rhum, Scotland, 1975–1977. *Biological Conservation*, 16, 23-30.

Marquiss, M., Madders, M., Irvine, J. & Carss, D. N. 2004. *The impact of White-tailed Eagles on sheep farming on Mull*. 48 pp. Centre for Ecology and Hydrology.

May, R., Nygård, T., Dahl, E. L. & Bevanger, K. 2013. Habitat utilization in White-tailed Eagles (*Haliaeetus albicilla*) and the displacement impact of the Smøla wind-power plant. *Wildlife Society Bulletin*, 37, 75-83.

- May, R., Hoel, P. L., Langston, R., Dahl, E. L., Bevanger, K., Reitan, O., Nygård, T., Pedersen, H. C., Røskaft, E. & Stokke, B. G. 2010. *Collision risk in white-tailed eagles: Modelling collision risk using vantage point observations in Smøla wind-power plant*. 25 pp. Trondheim, Norway.
- Meretsky, V. J., Snyder, N. F., Beissinger, S. R., Clendenen, D. A. & Wiley, J. W. 2000. Demography of the California Condor: implications for reestablishment. *Conservation Biology*, 14, 957-967.
- Morton, D., Rowland, C., Wood, C., Meek, L., Marston, C., Smith, G., Wadsworth, R. & Simpson, I. C. 2011. *Final Report for LCM2007 - the new UK Land Cover Map* 115 pp. CS Technical Report No 11/7, Centre for Ecology and Hydrology.
- Newton, I. 1986. *The Sparrowhawk*. T & A D Poyser, London, UK.
- Newton, I. 2013. *Bird Populations*. HarperCollins Publishers, London, UK.
- Nordström, M. & Korpimäki, E. 2004. Effects of island isolation and feral mink removal on bird communities on small islands in the Baltic Sea. *Journal of Animal Ecology*, 73, 424-433.
- O'Toole, L., Fielding, A. H. & Haworth, P. F. 2002. Re-introduction of the golden eagle into the Republic of Ireland. *Biological Conservation*, 103, 303-312.
- Pettex, E., Barrett, R. T., Lorentsen, S.-H., Bonadonna, F., Pichegru, L., Pons, J.-B. & Grémillet, D. 2015. Contrasting population trends at seabirds colonies: is food limitation a factor in Norway? *Journal of Ornithology*, 156, 397-406.
- R Development Core Team. 2011. *R: A language and environment for statistical computing*. 2.13.1, <http://www.R-project.org>, Vienna. Austria.
- Radović, A. & Mikusca, T. 2009. Population size, distribution and habitat selection of the white-tailed eagle *Haliaeetus albicilla* in the alluvial wetlands of Croatia. *Biologica*, 64, 156-164.
- Robin, X., Turck, N., Hainard, A., Tiberti, N., Lisacek, F., Sanchez, J.-C. & Müller, M. 2011. pROC: an open-source package for R and S+ to analyze and compare ROC curves. *BMC Bioinformatics*, 12, 77.
- RSPB. 2013. *Birdcrime - offences against wild bird legislation in 2012*. 71 pp. RSPB, Sandy, Bedfordshire, UK.
- RSPB Scotland. 2013. *The illegal killing of birds of prey in Scotland 2012*. 45 pp. RSPB Scotland, Edinburgh, UK.
- Rutz, C., Bijlsma, R. G., Marquiss, M. & Kenward, R. E. 2006. Population limitation in the Northern Goshawk in Europe: A review with case studies. *Studies in Avian Biology*, 31, 158-197.
- Salo, P., Nordström, M., Thomson, R. L. & Korpimäki, E. 2008. Risk induced by a native top predator reduces alien mink movements. *Journal of Animal Ecology*, 77, 1092-1098.
- Sandor, A. D., Alexe, V., Marinov, M., Dorosencu, A., Domsa, C. & Kiss, B. J. 2015. Nest-site selection, breeding success, and diet of white-tailed eagles (*Haliaeetus albicilla*) in the Danube Delta, Romania. *Turkish Journal of Zoology*, 39, 300-307.

- Santangeli, A., Hogmander, J. & Laaksonen, T. 2013. Returning white-tailed eagles breed successfully in landscapes under intensive forestry regimes as in protected areas. *Animal Conservation*, 16, 500-508.
- Sarrazin, F. & Barbault, R. 1996. Reintroduction: challenges and lessons for basic ecology. *Trends in Ecology & Evolution*, 11, 474-478.
- Seddon, P. J., Armstrong, D. P. & Maloney, R. F. 2007. Developing the science of reintroduction biology. *Conservation Biology*, 21, 303-312.
- Simms, I. C., Ormston, C. M., Somerwill, K. E., Cairns, C. L., Tobin, F. R., Judge, J. & Tomlinson, A. 2010. *A pilot study into sea eagle predation on lambs in the Gairloch area - Final Report*. Commissioned Report No.370., 41 pp. Scottish Natural Heritage, Inverness, UK.
- Smart, J., Amar, A., Sim, I. M. W., Etheridge, B., Cameron, D., Christie, G. & Wilson, J. D. 2010. Illegal killing slows population recovery of a re-introduced raptor of high conservation concern - The red kite *Milvus milvus*. *Biological Conservation*, 143, 1278-1286.
- Struwe-Juhl, B. & Schmidt, R. 2003. Flight feather moult of White-tailed Eagle (*Haliaeetus albicilla*) in Schleswig-Holstein, Germany. *Journal Fur Ornithologie*, 144, 418-437.
- Struwe-Juhl, B. & Grünkorn, T. 2007. Results of colour-ringing White-tailed Sea Eagles *Haliaeetus albicilla* in Schleswig-Holstein: site fidelity, movements, dispersal, age of first breeding, age structure and breeding of siblings]. *In German with English summary. Vogelwelt*, 128, 117-129.
- Swets, J. A. 1988. Measuring the accuracy of diagnostic systems. *Science*, 240, 1285-1293.
- Turrin, C. & Watts, B. D. 2015. Nest guarding in Chesapeake Bay Bald Eagles. *Journal of Raptor Research*, 49, 18-28.
- Village, A. 1990. *The Kestrel*. T & AD Poyser Ltd.
- White, G. C. & Burnham, K. P. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study*, 46, 120-139.
- Whitfield, D. P., Fielding, A. H., McLeod, D. R. A. & Haworth, P. F. 2004a. Modelling the effects of persecution on the population dynamics of golden eagles in Scotland. *Biological Conservation*, 119, 319-333.
- Whitfield, D. P., Fielding, A. H., McLeod, D. R. A. & Haworth, P. F. 2004b. The effects of persecution on age of breeding and territory occupation in golden eagles in Scotland. *Biological Conservation*, 118, 249-259.
- Whitfield, D. P., Fielding, A. H., McLeod, D. R. A. & Haworth, P. F. 2008. *A conservation framework for golden eagles: implications for their conservation and management in Scotland*. 163 pp. Scottish Natural Heritage, Edinburgh.
- Whitfield, D. P., McLeod, D. R. A., Watson, J., Fielding, A. H. & Haworth, P. F. 2003. The association of grouse moor in Scotland with the illegal use of poisons to control predators. *Biological Conservation*, 114, 157-163.

Whitfield, D. P., Marquiss, M., Reid, R., Grant, J., Tingay, R. & Evans, R. J. 2013. Breeding season diets of sympatric White-tailed Eagles and Golden Eagles in Scotland: no evidence for competitive effects. *Bird Study*, 60, 67-76.

Whitfield, D. P., Douse, A., Evans, R. J., Grant, J., Love, J., McLeod, D. R. A., Reid, R. & Wilson, J. D. 2009a. Natal and breeding dispersal in a reintroduced population of White-tailed Eagles *Haliaeetus albicilla*. *Bird Study*, 56, 177-186.

Whitfield, D. P., Duffy, K., McLeod, D. R. A., Evans, R. J., MacLennan, A. M., Reid, R., Sexton, D., Wilson, J. D. & Douse, A. 2009b. Juvenile dispersal of White-tailed Eagles in Western Scotland. *Journal of Raptor Research*, 43, 110-120.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A. & Smith, G. M. 2009. *Mixed effects models in ecology with R*. Springer, New York.

APPENDIX 1: PARAMETER ESTIMATES USED IN THE VORTEX PVA

Table A1. Assumptions and estimates used in the Vortex Population Viability Analyses.

Parameter	Values used and assumption
Breeding system	Long-term monogamy
Age of first production of offspring	5 years (see Table 3)
Maximum age of reproduction	30 years (used by Evans <i>et al.</i> 2009)
Maximum number of young fledged	3
Sex ratio at birth	50:50
% Females (and males) breeding	82 ± 12.26 (assumed to be the percentage of females hatching at least one egg and thereafter producing at least one fledgling).
Fledglings per brood	Based on 10 year averages (Table 9)
Annual mean (± SE) mortality rate	From Table 8 by origin (males and females assumed identical)
Initial population size	0
Supplementation	Numbers released in each year of each of the three release phases adjusted to account to 1 st year mortality of released individuals.
Harvesting	Between 2 and 14 birds per year, or proportionally to national population size (i.e. 0.5%, 1.0% and 2.0% of the national population; see legends in Figs 12 and 13).

APPENDIX 2: PREDICTING THE REGIONAL CARRYING CAPACITY

Background

The underlying constraint of predicting regional population growth was that the wing-tagging data were too sparse to estimate regional survival rates. In addition, the dispersal rates between the different regions are unknown, and it would have been a very time-consuming and costly exercise to collate such dispersal rates. Instead, the approach adopted incorporated the amount of suitable habitat in each of five regions (Mull, Skye, the Western Isles, Wester Ross and Argyll without Mull) and for the north of Great Britain. This habitat-related restriction might affect the future maximum regional population size (i.e. regional carrying capacity). How fast the population will grow to this estimated carrying capacity is not known, but it is nevertheless hoped that this Appendix will be able to inform stakeholders about potential future regional population sizes. It should be noted that initially in this project, the future geographical range expansion was impossible to predict. Therefore, it was important to include the northern part of England, just in case the models presented here suggested a quick expansion south from the core areas in Scotland. As the results suggested that the range expansion will not reach England in the time modelled here (i.e. up to and including 2040), the results below are for Scotland only.

Methods

The habitat suitability map was used to define how many 1 km² squares of suitable white-tailed eagle habitat were available in each region. The suitability index varies from 0.0 (i.e. 1 km² squares with no predicted suitable habitat) to 1.0 (i.e. 1 km² squares with the highest predicted habitat suitability), but in practice white-tailed eagles nest in a range of circumstances that the habitat suitability map may not be able to identify. For example, food abundance *per se* has not been taken into account when producing the habitat suitability map, so at sites surrounded by habitat predicted to be of low suitability, high and predictable food abundance may compensate for the otherwise poor habitat so that the site is regularly occupied by white-tailed eagles. Thus, to acknowledge this uncertainty, the area of available suitable habitat was varied by using different thresholds of what constitutes suitable habitat, from 0.5 to 0.9. This meant that by increasing the threshold, the amount of suitable habitat decreases.

Just as in the main report, it was assumed that only one pair of breeding white-tailed eagles can occur in each 1 km² square. It was also assumed that the density of breeding white-tailed eagles would only increase to levels slightly higher than the current highest breeding density recorded in Scotland, which occurs around Loch Frisa on Mull. There, in an area of 312 km², eight pairs nested in 2014 (i.e. an approximate density of 0.026 pairs per km² in all available habitat). However, not all habitats within the 312 km² were deemed suitable in our previous models (see Fig. 15 in the main report). Thus, within this area, the amount of habitat that is deemed suitable for white-tailed eagles is between 140 and 261 km², depending on how strict a habitat suitability threshold is being used (Table A2-1). It can therefore be assumed that the breeding density of white-tailed eagles (pairs per km²) will change according to the area of suitable habitat, with increasing densities with a higher habitat suitability threshold (Table A2-1).

To predict the range of possible maximum number of breeding white-tailed eagle pairs in the north of Great Britain and in each Scottish region, the amount of suitable habitat in each region at each habitat suitability threshold was multiplied by the density of breeding white-tailed eagles on Mull at the corresponding level of habitat suitability threshold (i.e. the values from Table A2-1).

Table A2-1. The density of breeding white-tailed eagles in an area of 312 km² around Loch Frisa on Mull in 2014. The breeding density in suitable habitat increases with increasing habitat suitability threshold.

Variable	Habitat suitability threshold			
	0.50	0.75	0.85	0.90
Amount of habitat km ²	261	200	165	140
Density (breeding pairs per km ² of suitable habitat)	0.031	0.040	0.048	0.057

Results

In Scotland and northern England, there were a total of 33,972 and 13,699 1 km² squares with suitable habitat at the 0.5 and the 0.9 habitat suitability thresholds, respectively. In the five regions Mull, Skye, the Western Isles, Wester Ross and Argyll (excluding Mull), the number of squares with suitable habitat varied between 368 and 2,074 depending on the threshold level set to specify suitability (Table A2-2).

Table A2-2. The number of 1 km² squares with suitable habitat for white-tailed eagles in five regions of Scotland under various threshold levels of what constitutes suitable habitat (i.e. increasing from 0.50 to 0.90).

Region	Number of 1 km ² squares with suitable habitat			
	0.50	0.75	0.85	0.90
Habitat suitability threshold				
Mull	642	493	423	368
Skye	920	694	604	528
Western Isles	2,074	1,606	1,459	1,372
Wester Ross	1,496	1,010	812	685
Argyll (without Mull)	4,062	3,119	2,592	2,149

Multiplying the number of km² squares with suitable habitat at each threshold level with the corresponding densities of breeding white-tailed eagles around Loch Frisa on Mull (i.e. the values in Table A2-1), the results suggested that across the Scottish regions, the predicted maximum number of breeding pairs would vary between around 28 (Skye) to 125 (Argyll excluding Mull; Table A2-3). As Mull was used as a baseline scenario, the predictions for Mull were close to the observed current population size.

Across Scotland, the predicted maximum number of pairs ranged from 783 (at the 0.9 level) to 1,041 (at the 0.5 level). This estimate is surprisingly close to the estimate included in the main report of between 889 and 1,005 pairs in 2040.

Table A2-3. The predicted maximum number of breeding white-tailed eagle pairs in different regions in Scotland rounded to nearest integer. The predicted maximum number of pairs is based on current highest density of white-tailed eagles in suitable habitat in an area of 312 km² around Loch Frisa on Mull. The predicted numbers presented here will increase if the number of pairs increases within the area around Loch Frisa.

Region	Predicted maximum number of breeding white-tailed			
	0.50	0.75	0.85	0.90
Habitat suitability threshold				
Mull	20	20	21	21
Skye	28	28	29	30
Western Isles	64	64	70	78
Wester Ross	46	40	39	39
Argyll (without Mull)	126	125	125	123

Discussion

The factors and processes influencing breeding density and density-dependent survival and reproduction of white-tailed eagles in Scotland are poorly known, and it is possible that not all 1 km² squares with suitable habitat will be occupied. For example, the density at which pairs can nest within the regions might be affected by food availability and intra-specific territorial behaviour, but there are no data available from Scotland that specify levels at which habitat, prey or competition with neighbouring pairs might become a limiting factor. Therefore, it might not be realistic to expect ongoing exponential growth at the regional scale. The approach used here does not make assumptions of immigration, emigration and density-dependent population regulation. However, before the carrying capacity is reached in many of the regions, it is likely that a geographical range expansion will take place (see main report).

It should be noted that the density has continued to increase on Mull in the last years, and the population growth is not showing signs of slowing down. Thus, the estimate of "highest density recorded in Scotland" around Loch Frisa is the only currently available information about what densities are possible for breeding white-tailed eagle in Scotland. Higher densities have been recorded elsewhere, e.g. in Norway. If the number of pairs continues to increase on Mull, this will have consequences for the estimates of carrying capacity regionally and nationally in Scotland. No prediction regarding the maximum number of white-tailed eagle pairs can be made for Mull, because Mull was used as a baseline scenario for this approach, and it is not known whether the densities recorded in continental Europe could occur here. When the increase of the breeding population of white-tailed eagle on Mull (or in another area) slows down, it would give a better indication of the future maximum density, density-dependent population regulation and carrying capacity across Scottish regions. Until this happens, the densities on Mull provide the best available information on potential future densities under differing areas of suitable habitat.

Overall, the authors of this report recommend that the regional predictions of regional carrying capacity should come with the strong caveat that they are based on the current highest density observed in Scotland, i.e. around Loch Frisa on Mull. The predictions about regional carrying capacity presented in this Appendix are likely to change if the densities of breeding white-tailed eagles increase even further in the Loch Frisa area.

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