

Population modelling of North Scotland red kites in relation to the cumulative impacts of wildlife crime and wind farm mortality





Scottish Natural Heritage
Dualchas Nàdair na h-Alba

All of nature for all of Scotland
Nàdar air fad airson Alba air fad

COMMISSIONED REPORT

Commissioned Report No. 904

Population modelling of North Scotland red kites in relation to the cumulative impacts of wildlife crime and wind farm mortality

For further information on this report please contact:

Andrew Stevenson
Scottish Natural Heritage
Great Glen House
Leachkin Road
INVERNESS
IV3 8NW
Telephone: 01878 710381
E-mail: andrew.stevenson@snh.gov.uk

This report should be quoted as:

Sansom, A., Etheridge, B., Smart, J. & Roos, S. 2016. Population modelling of North Scotland red kites in relation to the cumulative impacts of wildlife crime and wind farm mortality. *Scottish Natural Heritage Commissioned Report No. 904*.

This report, or any part of it, should not be reproduced without the permission of Scottish Natural Heritage. This permission will not be withheld unreasonably. The views expressed by the author(s) of this report should not be taken as the views and policies of Scottish Natural Heritage.



COMMISSIONED REPORT

Summary

Population modelling of North Scotland red kites in relation to the cumulative impacts of wildlife crime and wind farm mortality

Commissioned Report No. 904

Project No: 15751

Contractor: RSPB Scotland

Year of publication: 2016

Keywords

Milvus milvus; Population Viability Analysis; PVA; poison; wind turbine; collision risk; raptor; re-introduction.

Background

The red kite (*Milvus milvus*) population in the North of Scotland has shown slow population growth compared with other re-introduced kite populations. Previous work, using data up to 2006, presented evidence that illegal killing affecting the North Scotland population was a major cause for this difference (Smart *et al.*, 2010). The North Scotland population has continued to grow slowly since 2006, and in early 2014 a major poisoning event affected the North Scotland red kites just prior to the breeding season. In total, 16 dead kites were recovered, of which 12 were confirmed to have died due to ingesting a poison. Given the concern over the slow growth of the population, this poisoning event has the potential for serious impact on population growth. In this report, we assess short and long-term effects of the poisoning event as required under the Environmental Liability (Scotland) Regulations 2009.

In continental Europe, red kites are often recorded as colliding with wind turbines, and in Germany the collisions frequency is a noted concern (Schaub, 2012). There are an increasing number of operational and proposed wind farms within the distribution of the North Scotland red kite population. Based on the concerns about the status of the population, it is important to understand what levels of additional mortality the population could withstand. This could inform decisions on wind farm proposals that have the potential to affect the North Scotland population. In this report, the effects of cumulative mortality from wind turbines have been modelled to examine what level of mortality could significantly constrain the population. Scenarios used include the current cumulative predicted annual mortality of two to three birds as well as higher rates of five and 10 birds.

Main findings

- The number of breeding pairs in the North Scotland red kite population has continued to grow very slowly and had around 64 pairs in 2014. This is significantly fewer pairs than the population in the Chilterns in England, which has over 1,000 breeding pairs, even though these populations were founded by re-introductions at the same time using the same number of released birds. The North Scotland population also has significantly lower growth rate than the three other spatially discrete Scottish red kite populations.

- Breeding success (whether a nest produced at least one fledgling) differed significantly between the Scottish red kite populations. The North Scotland population had the highest proportion of successful nests, with circa 84% of nests producing at least one fledgling. Also, the number of fledglings produced per successful nest differed significantly across the Scottish red kite populations. Successful nests in North Scotland produced on average 2.23 (\pm 0.07) fledglings, which is the second highest breeding productivity in Scotland after the recently established population in Aberdeenshire. There was evidence suggesting that breeding productivity had declined significantly in recent years.
- Predictive models using updated estimates of breeding success and survival correctly replicated the observed number of breeding pairs in the North Scotland population as well as a reference population in the Chilterns in England not subjected to illegal killing.
- The poisoning event that affected the North Scotland red kite population in 2014 is likely to have a relatively minor effect on the number of breeding pairs in the next five years, but a more noticeable effect over the next 10 to 30 years. The loss of these birds is likely to result in a breeding population in 2044 of 513 (\pm 15) pairs rather than 550 (\pm 15) pairs if no poisoning event had occurred (i.e. 6.7% lower population size).
- A low level of mortality caused by collisions with wind turbines (i.e. current predicted cumulative risks of two to three birds killed per year) is predicted to have only a small effect on the population growth of red kites in North Scotland. However, when the number of fatalities increased to 10 per year, the stochastic population growth rate (r , where $r < 0$ indicates population decrease, $r = 0$ indicates population stability and $r > 0$ indicates population increase) is lowered from 0.101 (\pm SE 0.197) in the baseline scenario without wind turbine-related mortality to 0.074 (\pm SE 0.231). This reduction in growth rate will lead to a predicted population size which is 280 pairs lower (i.e. 270 \pm 12) than the baseline scenario (i.e. 550 \pm 15) in year 2044.
- The cumulative effects of ongoing illegal killing, the Ross-shire poisoning event and increased mortality due to collisions with wind turbines can, under circumstances when the wind turbine-related mortality exceeds five fatalities per year, severely lower the population growth rate of red kites in North Scotland. Together, when these unrelated sources of mortality affect five and 10 birds per year, the predicted population size in 2044 would be 340 (\pm 13) and 244 (\pm 13) pairs, respectively (i.e. 210 and 306 breeding pairs lower than in the baseline scenario, which excludes the mortality caused by the poisoning event and the wind turbine-related mortality). However, in comparison with the much stronger population growth in the Chilterns, it is clear that illegal killing is still the major factor limiting the population growth of red kites in North Scotland. There is no evidence that the rate of illegal killing has changed between the time periods 1989-2006 (i.e. the years used in the original paper by Smart *et al.*, 2010) and 2007-2014.
- This report has highlighted that the number of resightings of wing-tagged red kites have declined since 2002. The resighting rate has been kept at an adequate level up to and including 2012, but the resighting rate was too low in 2013 and 2014 to calculate survival estimates for these years. Therefore, if it is considered desirable to continue to monitor the ongoing impacts of illegal killing and other population-limiting factors on survival and population growth of the North Scotland red kite population, a concerted effort is needed to increase the resighting rate of wing-tagged individuals.

For further information on this project contact:

Andrew Stevenson, Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW.

Tel: 01878 710381 or andrew.stevenson@snh.gov.uk

For further information on the SNH Research & Technical Support Programme contact:

Knowledge & Information Unit, Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW.

Tel: 01463 725000 or research@snh.gov.uk

Table of Contents	Page
1. INTRODUCTION	1
1.1 Objectives	3
2. METHODS	4
2.1 Monitoring of red kites	4
2.2 Population trends of red kites in Scotland	4
2.3 Trends in breeding performance of red kites in Scotland	4
2.4 Quantifying causes of death	5
2.5 Updated estimation of survival rates	6
2.6 Population modelling	8
3. RESULTS	12
3.1 Population trends 1989-2014	12
3.2 Breeding performance 1989-2014	13
3.3 Rates of illegal killing between 1989 and 2014	16
3.4 Model validation 1989-2014	19
3.4.1 Updated survival estimates	19
3.4.2 Updated models of population growth	21
3.5 Predicting the future population growth	23
3.5.1 Short and long-term effects of the Ross-shire event	23
3.5.2 The effects of additional cumulative mortality caused by wind turbine collisions	24
3.5.3 The effects of the Ross-shire event and cumulative mortality caused by wind turbine collisions	24
4. DISCUSSION	28
4.1 Observed population growth and reproduction in North Scotland 1989-2014	28
4.2 Evaluation of population growth model accuracy	28
4.3 Illegal killing	28
4.4 The short- and long-term effects of the Ross-shire event on red kites	29
4.5 The effects of wind turbine mortality on red kites	29
4.6 Cumulative effects of illegal killing, wind turbines mortality and the Ross-shire event	30
4.7 Red kites and illegal killing	30
4.8 Technical aspects of the ecological modelling in this study	31
4.9 Suggestions of future work	32
5. CONCLUSIONS	33
6. REFERENCES	34
APPENDIX 1: DEFINITIONS OF CATEGORIES USED BY SCIENCE AND ADVICE FOR SCOTTISH AGRICULTURE (SASA)	37

Acknowledgements

We thank Jeremy Wilson, Elsie Ashworth and Alan Fielding for constructive comments on an earlier draft of this report. Much of this work builds on a previous paper by Jennifer Smart, Arjun Amar, Innes Sim, Brian Etheridge, Duncan Cameron, George Christie and Jeremy Wilson published in 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). We are grateful for their work that explains much of the background to this report. We would like to thank fieldworkers involved in the re-introduction of red kites to North Scotland (1989-1992) and the subsequent monitoring of the Scottish red kite populations. Finally, we thank Andrew Stevenson, the SNH project officer for this piece of work, who provided useful suggestions on an earlier draft of this report.

1. INTRODUCTION

There are many examples of how humans have tried to re-introduce species that have been extirpated regionally and nationally. Most early attempts to re-introduce species were unsuccessful, probably due to a combination of low number of released individuals and failure to eliminate the threats that caused the original population to go extinct before re-introduction began (Burgman *et al.*, 1993; Meffe & Carroll, 1994; Godfray & Crawley, 1998). The International Union for Conservation of Nature (IUCN) and the Species Survival Commission (SSC) has developed guidelines for re-introductions, and the most important recommendation is that the factors that caused the original population to go extinct should have been removed, or at least reduced to such low levels that they will not jeopardise the success of the re-introduction programme (IUCN, 1998; IUCN/SSC, 2013). In addition, the guidelines recommend that thorough monitoring is implemented, so that the outcome of the re-introduction is recorded.

Historically, many European countries encouraged the killing of predators, including birds of prey, to protect livestock and to minimise perceived competition over game species. However, the strengthening of environmental laws, both within the EU and within individual countries, means that almost all large mammalian carnivores and all birds of prey are now legally protected across the EU. Only in a small number of EU member states is controlled and licensed hunting of certain large carnivore species still legal. The improvements of the legal framework and the wish to re-instate viable populations of carnivores have led to several attempts to re-introduce predators across Europe (Breitenmoser, 1998; Ritchie *et al.*, 2012). As a response to the reduced levels of illegal killing, large predators have expanded their geographical range throughout Europe (and North America) through natural dispersal (Deinet *et al.*, 2013). However, the modern protective legislation is not always supported by cooperation from people living in the areas to which carnivores are re-introduced or colonise naturally, and this often leads to persecution of the predators (Liberg *et al.*, 2012). Especially in rural areas, large carnivores are still regarded as unrestrained killers of wildlife and livestock, with a potential to damage the rural lifestyle and economy (Breitenmoser, 1998; Liberg *et al.*, 2012).

The red kite (*Milvus milvus*), a medium-sized raptor breeding in Europe and rarely in north Africa, was once widespread across the UK, but human persecution caused the extirpation of the species in Scotland and England by the end of the 19th century (Evans *et al.*, 1999; Wotton *et al.*, 2002). A small population remained in Wales, but due to continued illegal killing, inbreeding depression and potentially poor land productivity, the Welsh population grew slowly (Newton *et al.*, 1994). This slow recovery combined with the species' philopatric habits (i.e. rarely dispersing from their natal area (Newton *et al.* (1994)), at least in Wales, prompted conservation organisations to start planning a re-introduction programme across the UK. The first kites were released on the Black Isle, near Inverness, in North Scotland and in the Chilterns in England in 1989. Juvenile red kites from Sweden, Spain and Wales were released during five and six years in North Scotland and Chilterns, respectively, with 93 individuals being released in each location (Evans *et al.*, 1999). The first successful breeding attempts were recorded in 1992 in both populations, and an ambitious annual monitoring programme started with the aim to document the number of pairs and breeding success (Evans *et al.*, 1999). In addition, another seven re-introduced populations have been established in Scotland and England, as well as three in Ireland.

Overall, the re-introduction programme has been an overwhelming success, with most of the populations growing strongly. This population growth is remarkable, especially when contrasted with population declines in continental Europe due to habitat degradation and increased mortality from secondary rodenticide poisoning (Coeurdassier *et al.*, 2012, 2014). Currently, the UK holds about 8% of the global population of red kites. In Scotland, there are four discrete populations (North Scotland, Aberdeenshire, Central Scotland and Dumfries &

Galloway; Fig. 1), albeit with some dispersal movements recorded between them. These populations have been re-introduced at different times, and they have all been successful in terms of achieving population growth and becoming self-sustaining (Etheridge *et al.*, 2013; Challis *et al.*, 2014). However, it is clear that population growth differed between the populations, with strong population growth in Dumfries & Galloway, and moderate population growth in Central Scotland (Smart *et al.*, 2010). For the red kite population in North Scotland, the population growth has been slow since the re-introduction began in 1989, especially when compared with the other Scottish populations (Smart *et al.*, 2010). The releases had just begun in Aberdeenshire when the original paper was published.

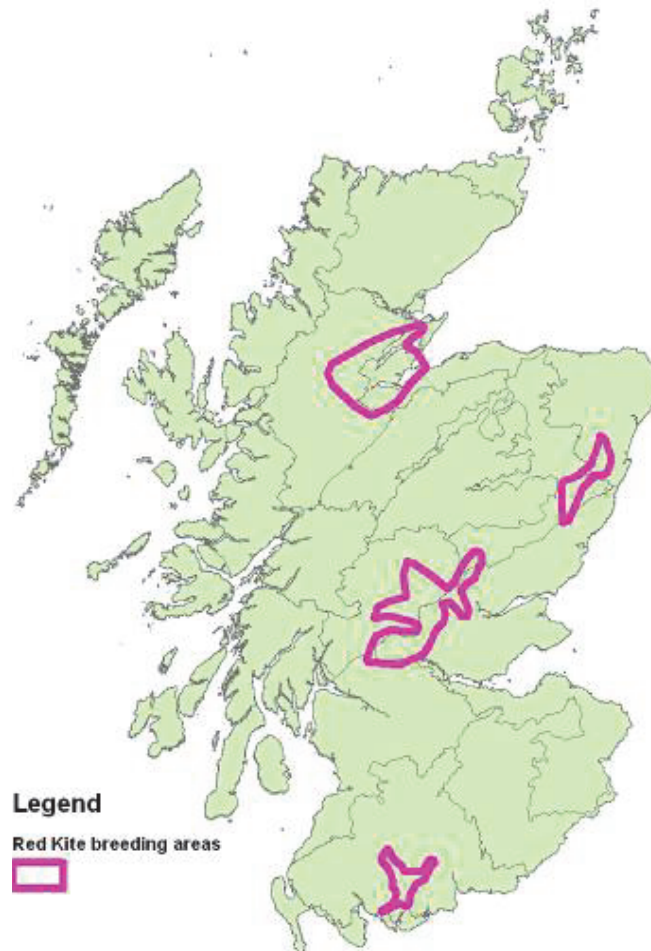


Figure 1. The approximate core breeding areas of the red kite populations in Scotland.

Smart *et al.* (2010) demonstrated that the reason for the slow population growth in North Scotland was illegal killing. By following red kites marked with wing-tags and radio-transmitters, they showed that the low survival of sub-adult (second-year) birds was a key factor in explaining the slow population growth. The North Scotland population had significantly lower second-year survival than the red kite populations in Central Scotland and Dumfries & Galloway. Sub-adult red kites search wide areas to find suitable habitat for territory establishment. Smart *et al.* (2010) suggested that these movements increased the probability of encountering unprotected poisoned baits. In total, 41 (40%) of 103 recovered dead kites between 1989 and 2006 were illegally killed, mainly due to poisoning (Smart *et al.*, 2010). As a scavenging bird, red kites are likely to be victims of badly placed poisoned baits, regardless of whether the perpetrator was aiming to kill red kites or any other predator.

In 2014, 16 red kites and six buzzards were found dead in a small area near the core breeding area for the North Scotland red kite population. This event became known as the "Ross-shire event", and will be referred to in this way hereafter. The subsequent post-mortem analyses of the carcasses ruled out shooting, and toxicology tests carried out by Science and Advice for Scottish Agriculture (SASA) revealed that at least 12 red kites had died as a result of consuming "a banned substance". The other red kites had been scavenged and could not be analysed. As the North Scotland red kite population had already been shown to be under pressure from ongoing illegal killing, there were fears that the Ross-shire event would have both short- and long-term implications for the viability of this population. It should be noted that the Ross-shire event occurred away from moors managed for grouse shooting and was atypical in that so many red kites died, the event was reported swiftly and so many carcasses could be collected.

Under the Environmental Liability (Scotland) Regulations 2009, the UK Government, via Defra, has a duty to report incidents such as the Ross-shire one to the EU, and to make an assessment of the level of impact on the population. The best way to assess the short and long-term consequences of the Ross-shire event is to model the population trajectory using up-to-date figures on population size, survival and reproduction. This can provide quantifiable predictions of population growth in scenarios with mortality equivalent to that recorded in the Ross-shire event, as well as scenarios without such mortality.

In addition, it is known from both Scotland and continental Europe that red kites collide with the rotor blades of wind turbines, and the outcome of such collisions is normally fatal (Duffy & Steward, 2008; Schaub, 2012). There are an increasing number of operational and proposed wind farms within the distribution of the North Scotland red kite population. Recently, a number of planning applications to build wind farms near the core area of the North Scotland red kite population have been submitted to the Scottish planning system. There are concerns that any additional mortality caused by potential new wind farms could lead to a population decline of red kites in the north of Scotland. The best method to assess whether new wind farms in the core breeding area will constrain the red kite population is to model the population growth with and without the expected annual additional mortality. In the autumn of 2014, SNH estimated the cumulative risk from active wind farms around the core North Scotland kite breeding area (built, consented and submitted to the planning system) at that time would equate to approximately two to three fatal bird collisions per year, assuming an avoidance rate of 98%. This has been used as the starting level for wind farm mortality modelling.

1.1 Objectives

This report has five main objectives:

1. Report trends in breeding population size in the four Scottish populations of red kites, including the years after the data collection by Smart *et al.* (2010) stopped (i.e. 2007 to 2014).
2. Report trends in breeding performance in the four Scottish populations of red kites. This includes "breeding success" (i.e. whether a nest produced at least one fledgling) and "breeding productivity" (i.e. how many fledglings successful nests produced).
3. Update the Smart *et al.* (2010) paper to test whether illegal killing still limits the population growth of red kites in North Scotland.
4. Evaluate the likely short- and long-term impacts of the Ross-shire event on the North Scotland red kite population.
5. Investigate the impact additional cumulative mortality caused by collisions with wind turbines may have on the North Scotland population.

2 METHODS

2.1 Monitoring of red kites

RSPB and SNH have monitored red kites annually since the re-introductions began in each of the four Scottish populations: North Scotland (1989), Aberdeenshire (2007), Central Scotland (1996) and Dumfries & Galloway (2001). In the last few years, an increasing proportion of the annual monitoring effort, especially in the Dumfries & Galloway and the Central Scotland populations, has been led by the Scottish Raptor Study Group (SRSG). The monitoring includes identification of occupied territories, assessment of whether the adult birds are marked with wing-tags and locating active nests. Once nests are found, the outcome of the breeding attempt (i.e. the number of fledglings produced) is monitored following best practice guidelines (Hardey *et al.*, 2013). All nest visits are carried out by fieldworkers with the required Schedule 1 licences. In all four red kite populations, almost every nesting attempt was located and monitored during the first 10 years after the re-introduction. Thereafter, population growth meant that a small number of pairs might have been missed each year. RSPB has maintained the database since the re-introduction began and has shared the relevant data with the Scottish Raptor Monitoring Scheme (SRMS), e.g. Challis *et al.* (2014).

2.2 Population trends of red kites in Scotland

The number of breeding pairs in each population was used as the unit for population size and as a response variable in many of the analyses in this report. In almost all cases, red kite pairs occupying a territory built a nest, which was found by the fieldworker. Territories occupied by single individuals are rare in comparison to other raptors (e.g. peregrines *Falco peregrinus* and golden eagles *Aquila chrysaetos*). In addition, re-nesting following a loss of a clutch is very rare in red kites (Evans *et al.*, 1999), so second nesting attempts within territories were not included in the analyses.

To answer the first objective of this study (i.e. report trends in population size of red kites), the population size of red kites in the four geographically separated Scottish populations in 2014 was compared. Also, the population size of red kites in the Chilterns in England, which was established in the same year as the North Scotland red kite population, using the same number of founder individuals, was compared with the North Scotland population.

The population growth rates since the start of the releases in each population were compared both visually and analytically. To calculate the population growth rate for each Scottish population, the following formula was used:

$$r = \frac{\log\left(\frac{N}{N_0}\right)}{t} \quad (\text{Eqn. 1})$$

where r = the rate of population change as a function of time (i.e. a 1% increase is expressed as 0.01 and a 1% decrease is expressed as -0.01), N_0 = the number of pairs in the year when red kites first bred in the regional population, N = The number of pairs in 2014, and t = the time (in years) between the first recorded breeding and 2014.

2.3 Trends in breeding performance of red kites in Scotland

To answer the second objective of this study (i.e. report trends in breeding performance of red kites), data on average breeding performance per year was compiled for the four Scottish red kite populations. Breeding performance was then compared across the four populations to assess whether it was particularly poor in North Scotland relative to other areas, as might be expected if breeding performance was limiting population growth.

Breeding performance is a composite measure of the "breeding success" (i.e. a binomial variable that depicts whether a nest was successful [producing ≥ 1 fledgling] or not) and "breeding productivity" (i.e. the number of fledglings produced per successful nest).

Variation in breeding success between populations was modelled using Generalised Linear Models (GLMs) with a binomial error structure. Breeding success was modelled as a Bernoulli distribution, i.e. "Successes / Totals", which takes the form "Successes / Failures" in the software R, weighted by the total number of pairs in each population in each year. All models included "number of years since the start of the introduction" as a continuous explanatory variable and "population" as a factorial explanatory variable.

Variation in breeding productivity between populations was modelled using Generalised Linear Models (GLMs) with a normal (i.e. Gaussian) error structure. Productivity was modelled as the number of fledglings produced per successful nest. All models included "Number of years since the start of the introduction" as a continuous explanatory variable and "Region" as a factorial explanatory variable. Pair-wise post-hoc tests were carried out to determine any significant differences between individual populations in success rate and chicks fledged per successful attempt.

To examine whether the breeding success and productivity had changed over time in the North Scotland population, Generalized Linear Mixed Models (GLMMs) were used. As there was a possibility that these measures differed between territories (e.g. due to variation in prey abundance), territory was modelled as a random effect. Year was modelled as a continuous linear and quadratic explanatory variable, which tests for any non-linear changes in breeding success over time. A binomial error structure was used when modelling breeding success (successes/failures) and a Poisson error structure was used when modelling breeding productivity (number of chicks fledged per pair and number of chicks fledged per successful pair). Additional GLMMs were run using time as a categorical variable with two levels ("Up to and including 2006" and "After 2006") to specifically test for any differences in breeding performance between the time period covered by Smart *et al.* (2010) and subsequent years. All analyses were carried out in R version 3.2.1 (R Development Core Team, 2011).

2.4 Quantifying causes of death

To answer the third objective of this study (i.e. to update the Smart *et al.* (2010) paper to inform whether illegal killing is still limiting the population growth of red kites in North Scotland), the causes of death of all wing-tagged, ringed or radio-tagged red kites were compiled. This work used all birds found dead during the time frame covered by Smart *et al.* (2010), and also all birds found dead between 2007 and 2014. Causes of death were determined according to post-mortem examinations carried out by independent veterinarians at the Scottish Rural College (SRUC; formerly the Scottish Agricultural College, SAC) and toxicology tests carried out by independent eco-toxicologists at SASA. Categories used were: natural death, electrocution, collision (including road, rail and wind turbine collisions), shooting, poisoning, suspected illegal killing (i.e. for a small number of cases where evidence strongly suggested illegal activities, such as no carcass recovered but radio tag and harness found which showed that the harness had been cut with a knife), other (for cases where cause of death was not covered by any of these categories) and unknown (for cases where cause of death could not be determined, for example when a very old carcass was found).

When toxicology tests conducted by SASA revealed that a red kite had died of poisoning, deaths were categorised as being caused by illegal pesticides (i.e. any of the eight substances banned under the Possession of Pesticides (Scotland) Order 2005, i.e. Strychnine, Aldicarb, Alphachloralose (but note that under certain circumstances this

substance might held legally), Aluminium phosphide, Bendiocarb, Carbofuran, Mevinphos and Sodium cyanide, or any other pesticide, such as legally-held first and second generation anti-coagulant rodenticides. The categories used by SASA when describing poisoning incidents (listed in Appendix 1) have been used as much as possible in this report. However, in many places in this report, SASA's category "Pesticide abuse" (i.e. a pesticide that has been deliberately used in an illegal manner to poison, or try to poison, animals), has been combined with shooting and trapping of red kites to form the category "illegal killing". Thus, the term "illegal killing" in this report refers to the deliberate abuse of a pesticide in an illegal manner as well as shooting and trapping that has caused the death of a red kite.

To test if changes in rates of illegal killing had changed over time, a Generalized Linear Model with a binomial error structure were used, with the proportion of total deaths caused by illegal killing as the response variable (i.e. using a Bernoulli distribution with "number of illegally killed red kites"/"Total number of dead red kites" for each year) and year as a continuous or factorial explanatory variable. Another test was carried out to study whether the rate of illegal killing differed between the time period 1989-2006 (the end year of data collection used in the Smart *et al.* (2010) study) and 2007-2014. This was done by means of a GLM with the same structure as above, but using Time period (i.e. "up to 2006" and "after 2006") as explanatory variable instead of year. The proportion of deaths caused by illegal killing, other causes or unknown causes was determined for each of three age classes (0-1, 1-2 and 2+ years), and also split by time period. Only individuals known to have fledged were included in these analyses.

Pre-fledging deaths were quantified separately and classified as being caused by natural factors (e.g. starvation, inclement weather and predation), rodenticides (including suspected cases), poison (by banned pesticides; see above), other reasons or unknown.

2.5 Updated estimation of survival rates

To answer objective 4 and 5 of this study (i.e. to evaluate the impacts of the Ross-shire event and investigate impact of additional mortality caused by collisions with wind turbines, respectively), survival rates of red kites had to be estimated. After scrutinising the data, only two additional resightings of the 93 kites released between 1989 and 1993 were found in the years after 2006. Therefore, the survival rate estimates for released birds presented in the original study (Smart *et al.*, 2010) were not updated.

The survival rates of wild-hatched birds were estimated by using a large dataset containing all wild-hatched individually marked red kites in North Scotland between 1993 and 2014 and their subsequent reported resightings. The marking consisted of year-specific coloured wing-tags with an individual alpha-numeric inscription. Birds were classified as being alive in a given year based at least on resighting in that year. Following Smart *et al.* (2010), each year started on 1 July, the approximate fledging time for kites, and ended 30 June. As resighting data from 2013 and 2014 were very sparse (Figs 2 & 3), data from these years were excluded from analysis to avoid any possible underestimation of survival.

In total, this dataset consisted of a sample of 1,055 resighting histories from individually marked birds across 20 cohorts. The resighting data were of a mixture of sighting from pairs on breeding territories and at winter roosts. Only data coming from a visual sighting of a bird were included, with "resightings" using radio or satellite tag fixes excluded.

Following Smart *et al.* (2010), the effect of age class (0-1, 1-2 and 2+) and the interactions between time and each age class (using year as a category) on survival were modelled. Due to the decrease in resightings over time (Fig. 2) and the variation in resighting rates between cohorts (Fig. 3), the continuous variables time and cohort, as well as the interaction between

time and cohort, were included in models of resighting rate together with the two age classes used in Smart *et al.* (2010), (0-1, 1+). Therefore, the global model was specified as:

$$Phi (1styr:t + 2ndyr:t + Ad:t) + S (Ageclass + cohort:t) \quad (Eqn. 2)$$

where *Phi* is the survival probability, *1styr* is the 1st-year, *2ndyr* is the 2nd-year, *Ad* is the adult (and as multiplied by *Phi*, these variables refer to survival probability for these age classes), *t* is the time (i.e. in year), *S* is the resighting probability, *Ageclass* is a binomial variable indicating whether the bird was a 1st-year bird or older, and *cohort* is the year a bird was hatched.

All models nested within the global model were tested, including models where the resighting rate was modelled as the additive effect of cohort + time, and models where the resighting rate was modelled as the multiplicative effect of cohort * time. Modelling therefore tested for any between age-class differences in survival rates of wild-hatched birds from 1993-2012 and for any changes over time within age classes in survival rate.

Models were fitted using RMark (version 2.1.13; Laake, 2013), using the Cormack-Jolly-Seber model. However, the goodness of fit (GOF) of the global model (\hat{c}) was assessed by means of a bootstrap GOF test conducted in Program MARK (version 6.1; White & Burnham 1999). As the GOF test suggested a small overdispersion of the data, all models were adjusted using a \hat{c} of 1.23, as well as for small sample size (Burnham & Anderson, 2002). Therefore, model selection was based on QAICc values. Models with the lowest QAICc values (i.e. a $\Delta QAICc \leq 7.00$) were retained as candidate models with a good fit to the data. Models with a higher $\Delta QAICc$ were considered having a poorer fit to the data and were not considered further (Burnham & Anderson, 2002). As several candidate models had a $\Delta QAICc$ of less than 7.0, model averaging was done across these models to estimate age and year specific survival rates.

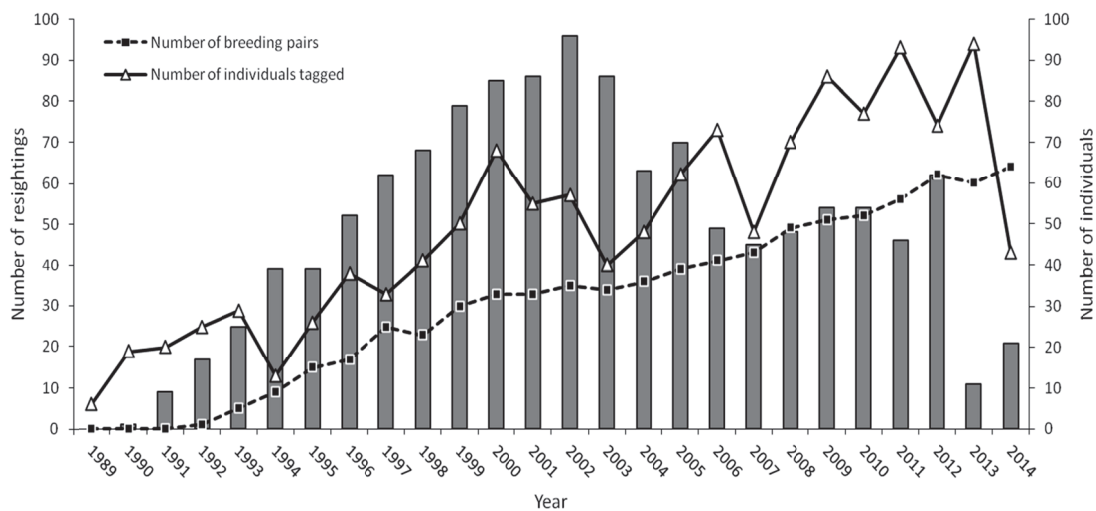


Figure 2. The number of resightings of live wing-tagged red kites in North Scotland in each year (grey bars; a year starts 1 July and ends 30 June). Also shown is the number of large red kite chicks tagged in each year (solid line, open triangles) and the observed number of breeding pairs (dashed line, filled quadrats).

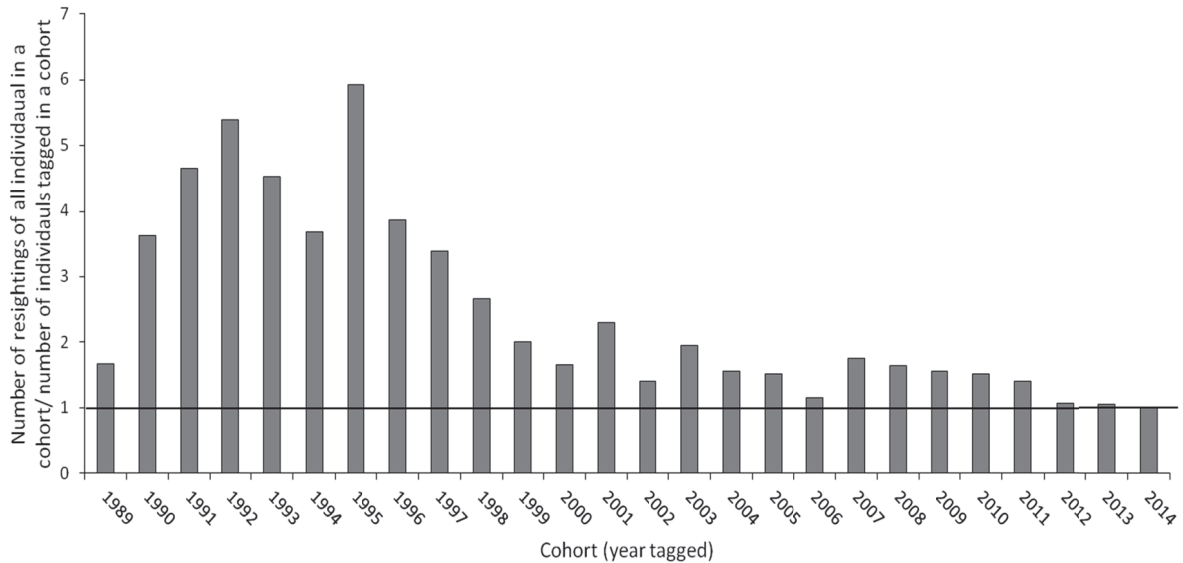


Figure 3. Ratio between the number of resightings of red kites in north Scotland and the total number of birds tagged in each cohort. By definition, all birds were observed alive in the year in which they were tagged (i.e. during the tagging event). A value of 1.0 means that no resightings were made of birds from that cohort after the tagging year. On average, birds from the cohort with most resightings (i.e. 1995) have been observed around six times in the subsequent years.

For population modelling, a mean survival rate, with an associated standard deviation was calculated for each age class using the survival estimates from all six top models. As a first step, when models contained time-dependent estimates within an age class, variance component estimation within Program MARK was used to calculate the mean overall survival estimate for that age class for that particular model (White *et al.*, 2001). This provided a more robust mean value of survival across multiple, year-specific estimates compared with an arithmetic mean (White *et al.*, 2001). For models where the survival rate was a constant (i.e. the survival of an age class did not change over time), the single value of survival probability, from the RMark model output for that age class was used. The overall mean age-specific survival estimates were then calculated as a weighted mean of the six top model survival estimates, weighted by the QAICc weight of each model.

A mean of the standard deviation, weighted by model QAICc weight, was calculated in the same way, with variance component estimation used to derive standard deviation for models containing time-dependent effects and a standard deviation of zero being assumed for models with a constant survival rate across years.

2.6 Population modelling

The population size of the north Scotland red kite population was modelled by using data on the number of released individuals, their age and origin-specific survival (updated from Smart *et al.*, 2010; see section 2.5) and year-specific mean breeding productivity (i.e. mean number of fledglings produced per pair each year; see section 2.3). The modelling was carried out using the software Vortex (Lacy *et al.*, 2005), which is a powerful individually-based Population Viability Analysis (PVA) software that can account for environmental stochasticity (e.g. variation in survival and reproduction due to e.g. weather). The software allows the user to set very detailed specifications regarding the study organism in order to examine the risk of extinction, but also to predict future population size. In essence, the Vortex follows the fate of individuals (e.g. survival, breeding success and dispersal) over a

specified amount of time. By running multiple iterations (normally >500, and this study used 1,000 iterations) with the same settings, which all will result in slightly different population trajectories due to the environmental stochasticity, the mean time to extinction and mean final population size (and many other variables) can be estimated.

As with many other modelling approaches, the results are only relevant if the scenarios are based on realistic input values. For this study, input values were based on the Scottish populations and, whenever possible, on values obtained from the North Scotland population. For example, it was assumed that red kites are monogamous, with 80% and 20% of the individuals starting breeding at the age of two and three years, respectively, and that they can breed once per year throughout their adult life, producing between zero and four fledglings per breeding attempt. It was also assumed that of the birds that had reached a breeding age (i.e. 80% of the 2nd-year birds), 100% of both males and females attempted to breed. This means that the number of individuals of the sex with fewest individuals was used as a proxy of the number of breeding pairs in each year, i.e. the same approach as used in similar studies of raptors (Evans *et al.*, 2009; Smart *et al.*, 2010). The maximum age of individuals was set to 20 years. The initial population size was zero and the number of individuals released during the first five years of the re-introduction programme was used as founders. However, the few birds that died very quickly after being released were subtracted from the number of released individuals. Also incorporated was the "harvest" of red kite chicks from the North Scotland population to start other re-introduction programmes in other parts of the UK (i.e. 12, 20 and 20 chicks were removed from their nests in 2001, 2002 and 2003, respectively). Finally, an equal sex ratio of released and wild-hatched birds was assumed, as well as a notional carrying capacity of 10,000 individuals. This removed any impact of density-dependence on modelled trajectories, as to date there is no evidence of density-dependent reproduction and survival of red kites in the UK.

Updated estimates of age-specific survival of wild-hatched red kites were combined with estimates of age-specific survival of released red kites from Smart *et al.* (2010) to model population growth in a PVA. To realistically capture the gradual disappearance of the original released cohorts of red kites (i.e. due to mortality and the end of releases after five years) the North Scotland population was modelled as a metapopulation consisting of two sub-populations. The first sub-population consisted of the released individuals, which had a 100% probability of "dispersing" into the other sub-population, which consisted of the wild-hatched individuals. There was no dispersal from the "wild" subpopulation to the "released" subpopulation. This ensures that the population modelling is done correctly within Vortex, and has been successfully implemented in other PVAs of released raptor populations (e.g. Evans *et al.*, 2009; Smart *et al.*, 2010). When a released individual dispersed from "released" to "wild", it still maintained the origin-specific survival rates for the remainder of its life.

The PVA modelling exercise started by running models between the years 1989 and 2014 to ensure that the input values resulted in a population trajectory that replicated the observed number of pairs recorded in the North Scotland population during this time period. This scenario is referred to as "Baseline 1989-2014". The potential population growth rate was then modelled using survival rates adjusted to remove poisoning deaths caused by abuse of illegal pesticides, shooting and trapping of red kites. For this scenario ("Without illegal killing 1989-2014"), the updated wild survival estimates, and new estimates of the levels of illegal killing in the North Scotland population were used to estimate age-specific survival rates of wild-hatched red kites in the absence of illegal killing. This approach was the same as that used by Smart *et al.* (2010). In summary, the new survival estimates in the absence of illegal killing was calculated as:

$$S_{xk} = 1 - ((M_{est} \times pM_{nk}) \times (M_{est} \times pM_k) + (M_{est} \times pM_{nk})) \quad (\text{Eqn. 3})$$

Where S_{xk} is the survival without illegal killing, M_{est} is the estimated mortality (i.e. 1 - the estimated survival rates given in Table 6); pM_{nk} is the proportion of deaths through other means than illegal killing (including those where the cause of death was unknown) and pM_k is the proportion of deaths through illegal killing (see section 3.3).

These estimates, together with those of age-specific survival of released red kites in the absence of illegal killing (see Table 3 in Smart *et al.* (2010)), were used to predict what the number of breeding pairs of red kites in the North Scotland population would have been in the absence of illegal killing (Table 6). To follow Smart *et al.* (2010), a comparison between the scenario "Without illegal killing 1989-2014" and the population growth of the red kite population in the Chilterns in England (re-introduced at the same time as the North Scotland population using the same number of founders) is provided. In addition, to extend the work by Smart *et al.* (2010), the population growth rate was modelled retaining illegal killing of fledged birds but removing the effects of losses of chicks before fledging through rodenticide poisoning. This was done by calculating the number of chicks fledged per breeding attempt in the absence of rodenticide chick mortality (i.e. "adding back" the chicks that died back into the population in Vortex). In the few cases where the exact number of chicks involved in an in-nest poisoning event was unknown, only one "extra" chick was added back into the fledging success rate. This was a conservative approach, as most rodenticide instances affecting broods with comprehensive knowledge of clutch size and the number of eggs hatched resulted in the whole brood dying. Therefore, the models presented here might underestimate the true negative effect of rodenticides poisoning of unfledged red kites on population growth.

Thereafter, the "Baseline 1989-2014" model was extended over a further 30 years (i.e. 2015 to 2044) to predict the number of breeding pairs over this period. For this forecasting, the updated year-specific reproductive rates were used up to 2014, but for the following years, the mean (\pm S.D) number of fledglings produced over the previous 10 years (i.e. between 2005 and 2014) of 1.755 (\pm 0.35) was used, with the assumptions that there were no changes in these values, in survival rates (i.e. modelling a constant level of illegal killing) and carrying capacity over time. This first model is referred to as the "Baseline Predictive Model" (BPM).

To address the fourth objective of this study (i.e. the short- and long-term effects of the Ross-shire event), the BPM was modified to include a one-off "harvest" in the PVA, involving the number of birds of each sex and age-class found dead in the Ross-shire event. Of the 16 red kites found dead, the cause of death for 12 birds was confirmed as poisoning by an illegally held substance. The four remaining birds were too scavenged and/or decomposed to be analysed by SASA. However, for the purpose of examining the effects of a large-scale harvest, the confirmed cause of death is of less importance as it is the number of birds dead that matters for population growth. Thus, to estimate the effect of the Ross-shire event, this study used a harvest of 16 red kites in the PVA models. It should be noted that the updated age-specific survival rates, including the effects of illegal killing and natural deaths, were used in this scenario.

Six of the 16 red kites found dead were not ringed or wing-tagged, but based on plumage characteristics, four of these birds were hatched in 2013. Their sex could not be determined and therefore only age-class, based on plumage features, could be allocated to these individuals. Thus, for the purpose of this study, two of the four unsexed 1st-year kites were assumed to have been females and two were assumed to have been males. Similarly, of the two unsexed adult birds, one was assumed to have been male and the other female. In addition, as the settings in Vortex (v. 9.99) only allowed harvesting of 2nd-year and adult birds, the 1st-year birds killed in the Ross-shire event had to be included in the harvest of 2nd-year birds. However, in the absence of the Ross-shire event, some of these birds would have died during their first year anyway. The probability of 1st-year birds surviving into their

2nd year is 0.41 (Table 6) thus the number of killed 1st-year birds in the Ross-shire event had to be multiplied by the probability of surviving to their 2nd year. To mimic the large-scale death of red kites in the Ross-shire event, this study used the following age and sex-specific deaths: 4.04 2nd-year males, 1.23 2nd-year females, two adult males and four adult females. The predicted number of breeding pairs at one, two, three, five, 10, 20 and 30 years after the event was compiled and compared with the BPM.

To address the fifth objective of the study (i.e. the short- and long-term effects of additional mortality caused by collisions with wind turbines), the BPM was modified to include an annual "harvest" of two and three individuals. This modelled the wind farm-related mortality estimated by SNH for all wind farms in the area that were built, consented and submitted to the planning system by the autumn 2014. Four scenarios were modelled using the two different harvest rates, and for two scenarios including increased mortality affecting five and 10 red kites, respectively;

- i) all mortalities affected sub-adult individuals ("sub-adult scenario"; one of each sex and two males and one female, respectively);
- ii) all mortalities affected adult breeding birds ("adult scenario"; one of each sex and two males and one female, respectively);
- iii) the mortality affecting both sub-adult and adult birds ("mixed scenario"; a) "2": one sub-adult male and one adult female, b) "3": one sub-adult male, one adult male and one sub-adult female, c) "5": two sub-adult males and females and one adult female, and d) "10": three sub-adult males and females as well as two adult males and females.

The predicted number of breeding pairs at one, two, three, five, 10, 20 and 30 years after 2014 (i.e. assuming that a wind farm became operational in early 2015) were evaluated and compared with the BPM values. Again, the updated age-specific survival rates, including the effects of illegal killing and natural deaths, were used in this scenario. It should be noted that this study assumed that the core area of the North Scotland red kite population was saturated with breeding pairs (i.e. no further pairs could settle in this area). This meant that there was no need to estimate density-dependent turbine collision risk, as new pairs would settle outside the core red kite areas and the proposed wind farm. This was the observed settlement pattern of red kites in Central Scotland when the Braes of Doune wind farm was constructed (Duffy & Urquhart, 2014). Finally, the effects of the Ross-shire event (i.e. 16 birds found dead in 2014) were combined with the additional mortality of wind farm collisions. Firstly, the mortality of two to three adult birds (one to two males and one female) annually, and secondly five and 10 birds killed annually (for these higher impact scenarios, a mix of sub-adult and adult kites were assumed to die from collisions with wind turbines as it is unlikely that just adults would be killed by turbines). The predicted number of breeding pairs at one, two, three, five, 10, 20 and 30 years after 2014 (i.e. assuming that a wind farm became operational in early 2015) were evaluated and compared with the BPM values, the Ross-shire event scenario and the wind farm scenario.

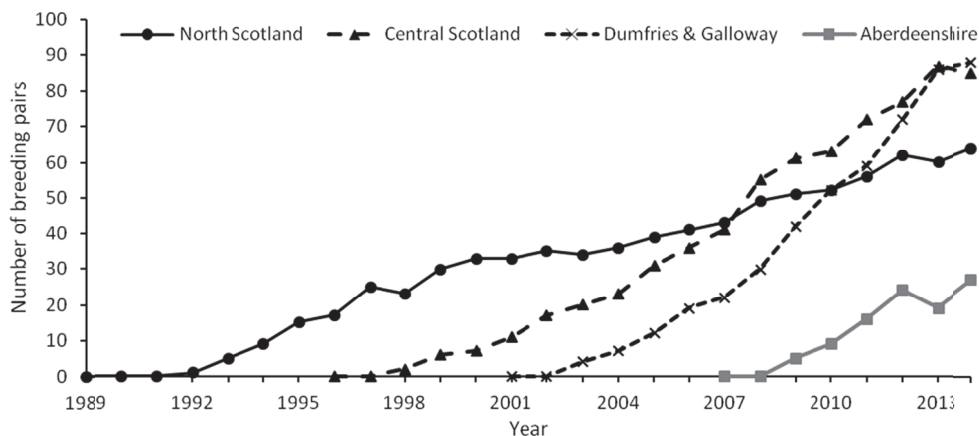
3. RESULTS

3.1 Population trends 1989-2014

Since the Smart *et al.* (2010) paper, which included data collected up to and including 2006, the North Scotland red kite population has continued to grow slowly (Fig. 4a). In comparison with the other three Scottish populations, it is clear that the North Scotland population has grown at a very low rate (Fig. 4b, Table 1), especially when visualised as number of breeding pairs in relation to the number of years since the re-introduction started.

Overall, the population trends reported in Smart *et al.* (2010) are still valid. Specifically, there is strong population growth in Dumfries & Galloway, intermediate growth in Central Scotland and slow growth in North Scotland. It is still relatively early to assess the Aberdeenshire population growth rate with any level of certainty, but the observed population growth suggests that it mimics the population growth of the two southern Scottish populations more than the North Scotland population (Fig. 4b).

a)



b)

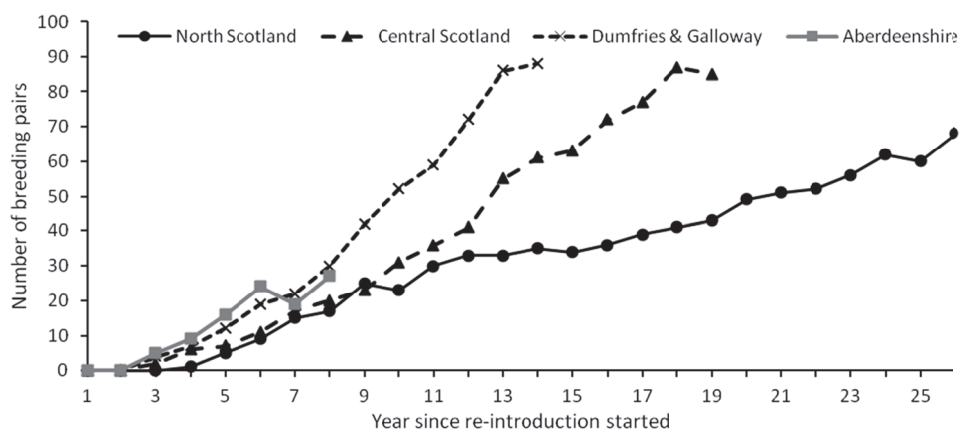


Figure 4. The number of breeding pairs in the four re-introduced red kite populations in Scotland. In a), the number of pairs in each year is shown, whereas in b) the number of pairs in each year since the re-introduction begun for each population is shown.

Table 1. The number of released individuals, years of re-introduction programme, year of first breeding, number of pairs in 2014 and the observed rate of population change (r) between the year of first recorded breeding attempt and 2014 in the four red kite populations in Scotland.

Population	No. released	Years of re-introduction	First breeding	No. of breeding pairs in 2014	r
North Scotland	93	1989-1993	1992	64	0.082
Central Scotland	103	1996-2001	1998	86	0.121
Dumfries & Galloway	104	2001-2005	2003	88	0.177
Aberdeenshire	101	2007-2009	2009	28	0.289

3.2 Breeding performance 1989-2014

Overall, the mean proportion of successful red kite nests differed significantly between populations (Table 2), with lowest breeding success in Central Scotland (Fig. 5). Pair-wise post-hoc tests showed that North Scotland had significantly higher breeding success than Central Scotland ($p < 0.001$), but not significantly different from other populations. The proportion of successful nests had not changed since the start of each re-introduction (Table 2). Similarly, the trend in proportion of successful breeding attempts over time did not vary significantly between populations, since the re-introductions began (i.e. the interaction between Population and Year since start of re-introduction was not significant; Table 2).

Table 2. Results of a binomial GLM testing for differences in red kite breeding success between populations, in terms of number of successful pairs/ total number of pairs (weighted by total number of breeding attempts).

Variable	χ^2	df	p
Population	50.22	3	<0.001
Year since start of re-introduction	1.54	1	0.21
Population * Year since start of re-introduction	2.57	3	0.46

The number of chicks fledged per successful attempt differed significantly between populations ($p = 0.001$; Table 3). Post-hoc pair-wise tests showed that North Scotland had significantly higher breeding productivity than Dumfries & Galloway ($p = 0.026$), but did not differ significantly from the other populations (Fig. 6). Different populations did not vary significantly, in terms of change in breeding productivity over time, since the re-introductions began (i.e. the interaction between Population and Year since start of re-introduction was not significant; Table 3).

Table 3. Results of a GLM with normal (Gaussian) error structure testing for differences between populations and time since the start of the re-introduction in the number of fledglings produced per successful breeding attempt.

Variable	χ^2	df	p
Population	16.61	3	0.001
Year since start of introduction	2.45	1	0.12
Population * Year since start of introduction	1.53	3	0.68

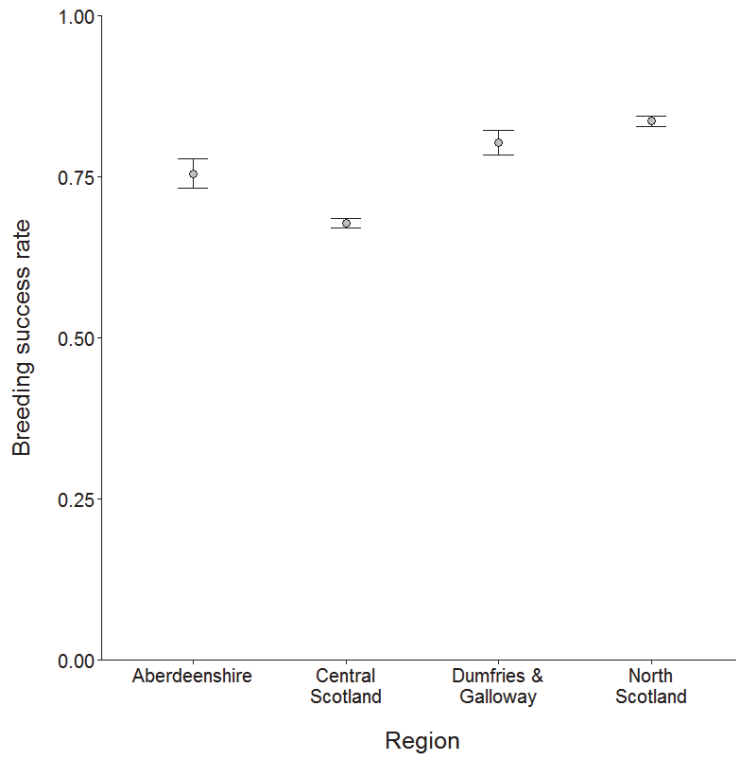


Figure 5. Mean breeding success rate (\pm S.E.; bootstrapped over 1,000 iterations) of red kite breeding attempts with known outcome across years in each Scottish population (weighted by the number of breeding attempts per year and population).

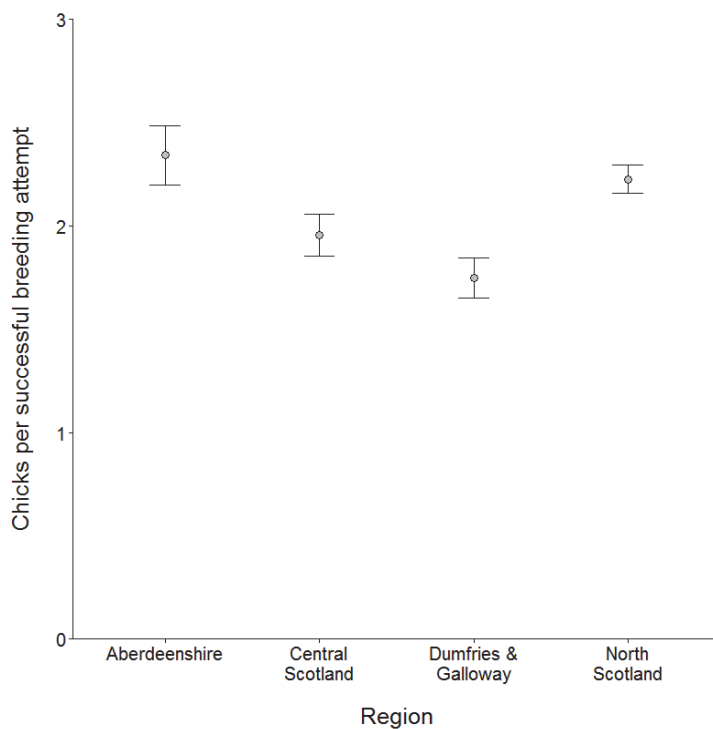


Figure 6. Breeding productivity (\pm S.E.; i.e. the number of fledglings produced per successful breeding attempt) of red kites in each Scottish population.

Specifically for North Scotland, the breeding success rate of red kite did not change significantly over time (Effect of Year: estimate = -0.098 ± 0.11 , $\chi^2 = 1.32$, d.f. = 2, $p = 0.517$). However, the number of fledglings produced per breeding attempt in North Scotland has varied significantly over time in a non-linear manner (effect of year²: $\chi^2 = 5.962$, d.f.= 2, $p = 0.041$). Initially, the number of fledglings produced increased slightly, but in recent years productivity has declined.

When restricting the analyses to successful nests only, the number of chicks fledged per successful attempt showed a nearly significant quadratic change over time, with some evidence of a decline in breeding productivity in recent years (effect of year²: Estimate = -0.041 ± 0.03 , $\chi^2 = 6.38$, d.f. = 2, $p = 0.051$; Fig. 7).

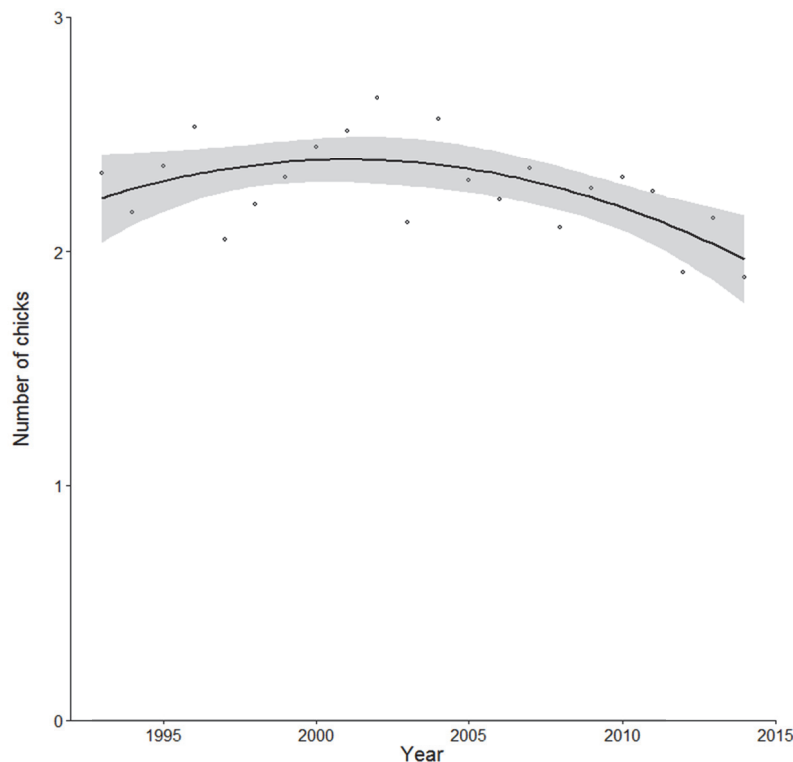


Figure 7. The number of fledglings produced per successful breeding pair in the North Scotland red kite population between 1992 and 2014. To aid the interpretation of the relationships, the non-linear trend is shown as a solid line with standard errors shaded in grey.

To compare the breeding performance of red kites in North Scotland in the years covered by the paper by (Smart *et al.*, 2010) and subsequent years (i.e. 2007-2014), a number of comparisons were made. The results showed that breeding success rate did not differ between the two time periods (Table 4). However, the mean number of chicks fledged per pair was significantly lower for the period 2007-2014 (mean \pm SE: 1.68 ± 0.06) than in the period included in the study by Smart *et al.* (2010); mean \pm SE: 1.86 ± 0.11 (Table 3). When restricting the analysis to successful breeding attempts, there was a near-significant difference between the two time periods, with slightly lower number of fledglings per successful nest for the period 2007-2014 (mean \pm SE: 2.20 ± 0.14) than in the period covered by Smart *et al.* (2010); mean \pm SE: 2.32 ± 0.19 (Table 3).

Table 4. Change in North Scotland red kite breeding success comparing the period 2007-2014 with the period 1992-2006, using GLM models of each measure of breeding success.

Response variable	Estimate \pm SE	d.f	χ^2	p
Success rate	0.300 \pm 0.221	1	1.847	0.174
Chick per breeding attempt	0.135 \pm 0.054	1	6.273	0.012
Chicks per successful breeding attempt	0.097 \pm 0.052	1	3.492	0.062

3.3 Rates of illegal killing between 1989 and 2014

The tagged (including wing-tagged and ringed with BTO issued rings) and recovered red kites died for a number of reasons (Fig. 8). In total, 160 red kites were recovered between 1989 and 2014. Overall, 54 red kites were confirmed to have been illegally killed, and another 10 were suspected to have been illegally killed. Thus, 40.0% of all dead red kites were confirmed or suspected to have been illegally killed between 1989 and 2014. The most frequently identified causes of death were pesticide abuse (31.9%), collisions with man-made objects (e.g. cars and trains; 22.5%), suspected illegal killing (6.3%) and natural causes (6.3%; Fig. 8). For 23.1% of all recovered red kites the cause of death was unknown. The illegal killings were mainly made up of poisoning abuse (N=51), whereas shooting and trapping was relatively rare (N=3). The high number of poisoning incidents in 2014 was a result of the Ross-shire event.

When considering all types of poisoning incidents, 87.9% of all poisonings were caused by pesticide abuse (i.e. illegally held substances, such as Carbofuran and Aldicarb), and the remaining incidents caused by pesticide misuse (i.e. legally held substances, such as various types of rodenticides and one incident involving a pesticide used to kill ticks on sheep).

There was no evidence that the proportion of deaths caused by confirmed or suspected illegal killing had changed over time (effect of year: Estimate = 0.033 \pm 0.025, χ^2 = 1.94, d.f. = 1, p = 0.18; Fig. 9). Similarly, the mean proportion of deaths caused by illegal killing did not differ significantly between the time period covered by Smart *et al.* (2010) (mean \pm SE: 0.42 \pm 0.10), and in the period 2007-2014 (0.38 \pm 0.07; χ^2 = 0.68, p = 0.35).

Illegal killing within the different age classes showed some difference when the period between 1989 and 2006 was compared with the time period 2007-2014 (Fig. 10). Smart *et al.* (2010) showed that 41.7% of 1st-year birds were confirmed illegally killed between 1989 and 2006, whereas for the period 2007-2014 the equivalent figure was 44.8%. For 2nd-year birds, the percentage confirmed illegally killed red kites had increased from 25.0% for the period 1989-2006 to 66.7% in the period 2007-2014. Finally, Smart *et al.* (2010) reported that 45% of adults were illegally killed in the period 1989-2006, and for the time period 2007-2014, the equivalent number was 30.0% (Fig. 10). The percentage of unknown mortality causes were broadly similar between the two time periods, apart from 2nd-year birds, where the percentage fell from 50.0% to 6.7% (Fig. 10).

Red kite chicks in North Scotland have been killed by pesticide misuse (i.e. killed by legally held second generation anti-coagulant rodenticides) before fledging (Fig. 11). There is no obvious pattern indicating a change over time in deaths of chicks from rodenticides in this way. However, in recent years, especially in 2010, high numbers of chicks have died due to rodenticide poisoning.

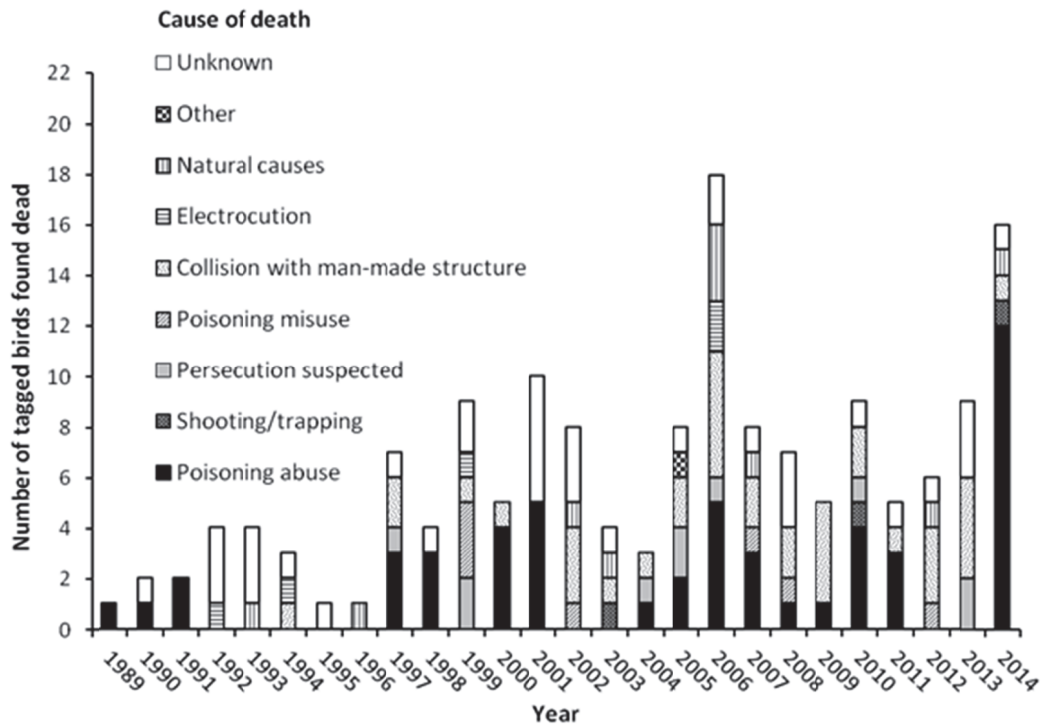


Figure 8. The causes of death of tagged red kites in North Scotland between 1989 and 2014.

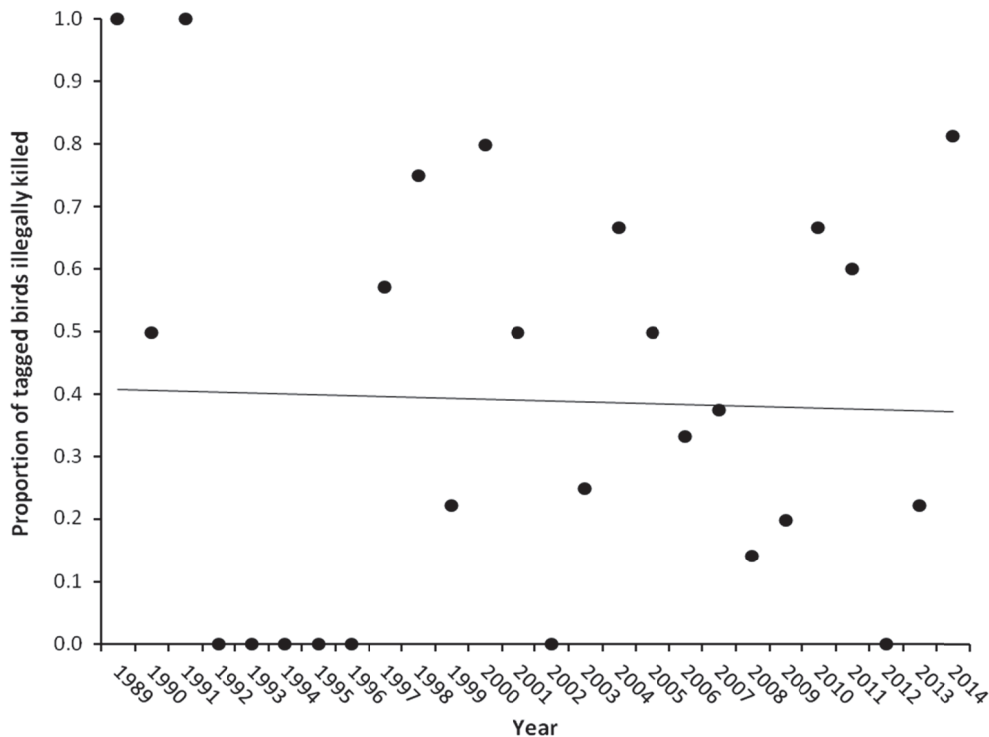


Figure 9. The proportion of red kites in North Scotland found dead in each year where the cause of death was confirmed or suspected illegal killing (see Methods for definitions). The solid line is the linear trend fitted for illustration of change over time (non-significant, $p=0.18$).

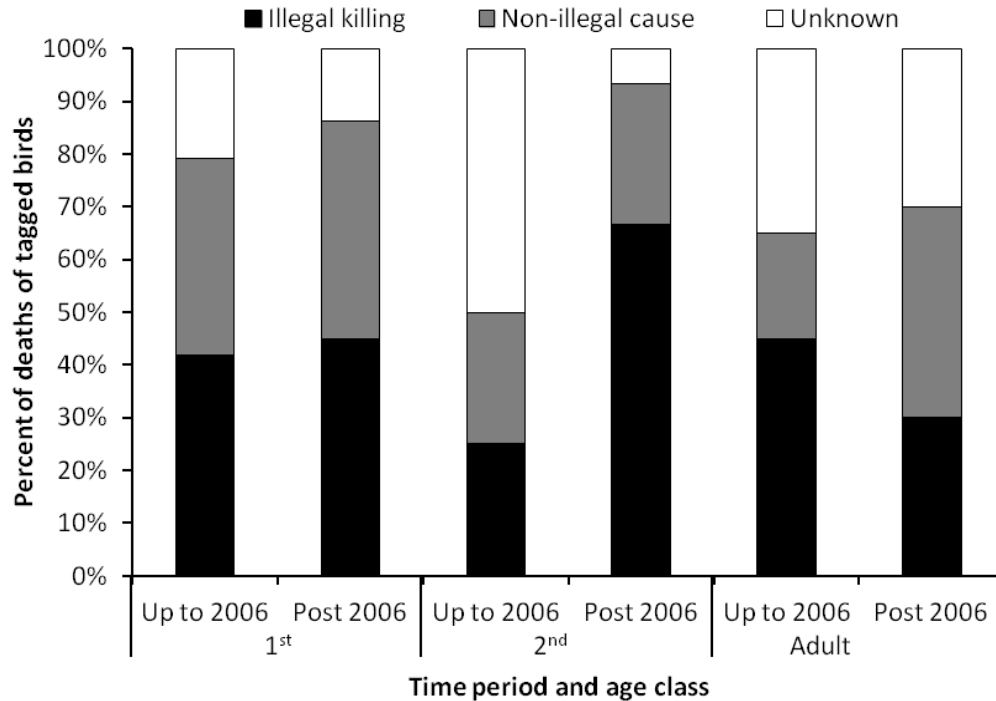


Figure 10. The percentage of the total number of deaths of red kites in North Scotland caused by confirmed illegal killing relative to other known and unknown causes for the different age classes separated into two time periods (i.e. the period 1989-2006, which was reported in Smart et al. (2010), and the later time period 2007-2014).

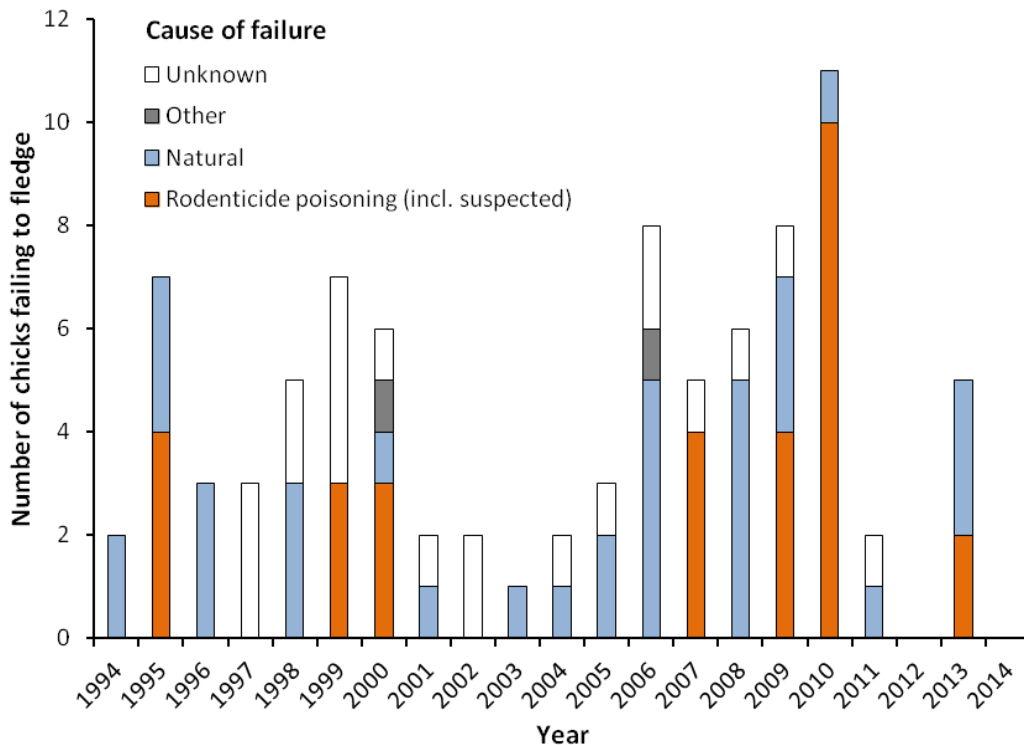


Figure 11. The number of red kite chicks failing to fledge in each year since 1994 by cause. Where only information on breeding success was available (i.e. successful or failed) and the number of large chicks was unknown, a conservative approach was taken whereby only one chick was assumed to have died from any of the listed causes.

3.4 Model validation 1989-2014

3.4.1 Updated survival estimates

The updated global model of survival of wild red kites in North Scotland was only slightly overdispersed ($\hat{c}= 1.23$), providing an adequate fit to the data. After adjusting for this slight overdispersion, six models had some support, all with a ΔQAICc of <7 . The cumulative QAICc weight of these models was 0.983 (Table 5).

All models contained the interaction between cohort and time for resighting rates (p), which suggests that resighting rates varied differently over time between cohorts (Table 5). In addition, the variable age class was also important in determining resighting probability in three of six top models.

Survival (Φ) of juveniles (1st-year) was only time-dependent in one of the six top models. This model had a QAICc weight of only 0.023, which suggests that there was little evidence for a year-specific effect on survival for juvenile red kites, and much stronger support for a constant survival rate over time. Survival was found to be time-dependent for sub-adults (2nd-year) in four of the six top models, including the two highest ranked models, lending strong support to time-varying survival in this age class. Support for time-dependent survival in adults was less strong, although three of the six top models contained this term (Table 5).

Table 5. Model results from the red kite survival analysis of wild-hatched birds wing-tagged in North Scotland between 1993 and 2012. Models were adjusted by a \hat{c} of 1.23 and ranked by QAICc (i.e. the AIC values adjusted for the overdispersion and for small sample size). Only the top 6 models, all with a ΔQAICc of <7 and a cumulative AICc weight of 0.983, are shown. All models shown were used to produce model averaged estimates of survival. Φ = survival p =probability of resighting, "time"= a year-specific estimate, "Time"=a trend over time, age= two age class 0-1 year and 1+ years.

Model	QAICc	ΔQAICc	Weight	Parameters	Deviance
$\Phi \sim (1^{\text{st}}\text{yr} + 2^{\text{nd}}\text{yr} * \text{time} + \text{adult})$ $p \sim (\text{cohort} * \text{Time})$	2576.6	0.00	0.497	59	632.8
$\Phi \sim (1^{\text{st}}\text{yr} + 2^{\text{nd}}\text{yr} * \text{time} + \text{adult})$ $p \sim (\text{age} + \text{cohort} * \text{Time})$	2577.9	1.24	0.268	60	631.9
$\Phi \sim (1^{\text{st}}\text{yr} + 2^{\text{nd}}\text{yr} * \text{time} + \text{adult} * \text{time})$ $p \sim (\text{cohort} * \text{Time})$	2579.5	2.86	0.119	75	601.2
$\Phi \sim (1^{\text{st}}\text{yr} + 2^{\text{nd}}\text{yr} * \text{time} + \text{adult} * \text{time})$ $p(\sim \text{age} + \text{cohort} * \text{Time})$	2581.6	4.96	0.042	76	601.1
$\Phi \sim (1^{\text{st}}\text{yr} + 2^{\text{nd}}\text{yr} + \text{adult} * \text{time})$ $p \sim (\text{cohort} * \text{Time})$	2582.0	5.35	0.034	58	640.3
$\Phi \sim (1^{\text{st}}\text{yr} * \text{time} + 2^{\text{nd}}\text{yr} + \text{adult})$ $p(\sim \text{age} + \text{cohort} * \text{Time})$	2582.8	6.14	0.023	61	634.7

The year-specific survival estimates for 2nd-year birds, based on all six models with a QAICc less than seven, showed high variation between years, with wide standard errors (Fig. 12). For this age-group, the survival estimates in four of the five last years are very low and below the overall average (Fig. 12). In comparison, the estimates of adult survival, based on model averaging of the six top models, vary far less (Fig. 13).

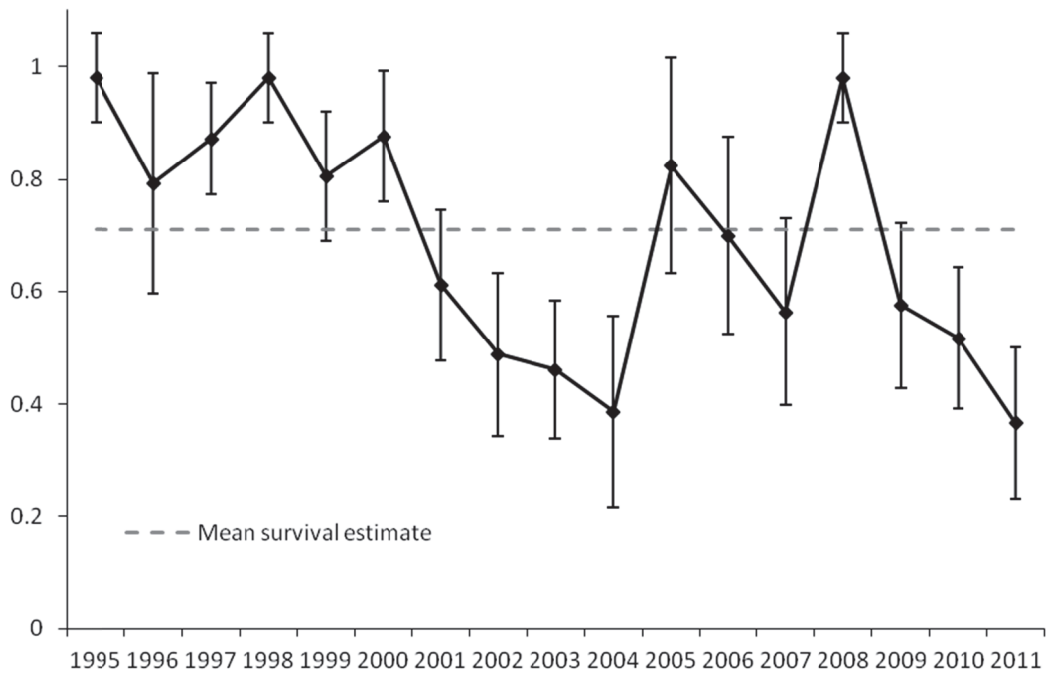


Figure 12. Annual survival probabilities (\pm SE) of 2nd-year wild red kites in North Scotland based on model averaged values of the models shown in Table 5. The dashed line shows the mean survival estimate, calculated as a weighted mean, using the variance component estimation for time-dependent models and MARK-derived values from models with a constant survival rate. The latter values used the QAICc weight as a weighting factor.

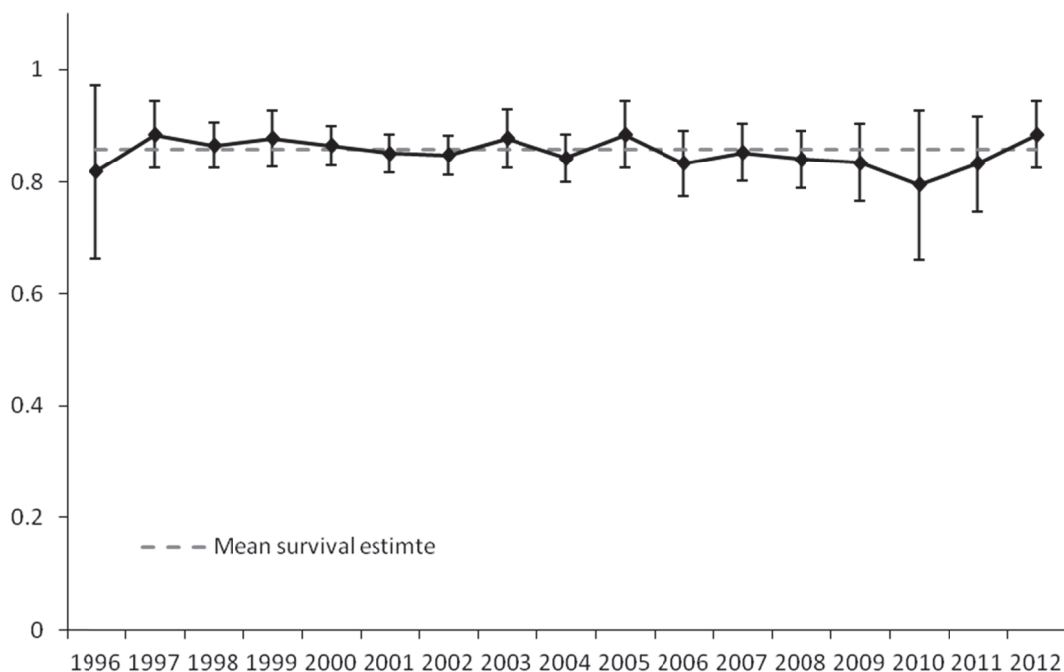


Figure 13. Annual survival probabilities (\pm SE) of adult wild red kites in North Scotland, based on model averaged values of the models shown in Table 5. Dashed line shows the mean survival estimate calculated as a weighted mean, using the variance component estimation for time-dependent models and MARK-derived values from models with a constant survival rate. The latter values used the QAICc weight as a weighting factor.

The updated age-specific mean estimates for survival of wild-hatched birds, using data between 1989 and 2012, show very little change from the survival rates estimated by Smart *et al.* (2010; Table 6). Estimates for age-specific survival for wild birds in the absence of illegal killing were also updated using these survival rates and all information available on the cause of death for tagged red kites up to 2014.

*Table 6. Estimates of age and origin-specific survival rates (\pm SD) for the North Scotland red kite population as well as the estimated survival rates in the absence of illegal killing. For 2nd-year and adult survival, mean values were calculated from year-specific estimates, using variance component estimation in Program MARK (White *et al.* 2001). For the Vortex PVA models, updated survival estimates for wild-hatched red kites and survival estimates for released red kites from Smart *et al.* (2010) were used. For the "No illegal killing" scenario, the same standard deviations as the "Baseline" scenario were used.*

Scenario	Age	Released	Wild-hatched	Wild-hatched
		(Smart <i>et al.</i> 2010)	(Smart <i>et al.</i> 2010)	(Updated)
		Survival	Survival	Survival
Baseline	1 st year	0.52 \pm 0.00	0.37 \pm 0.00	0.41 \pm 0.00
	2 nd year	0.87 \pm 0.16	0.72 \pm 0.25	0.71 \pm 0.20
	Adult	0.85 \pm 0.20	0.87 \pm 0.12	0.86 \pm 0.02
No illegal killing	1 st year	0.66 \pm 0.00	0.54 \pm 0.00	0.58 \pm 0.00
	2 nd year	0.90 \pm 0.16	0.78 \pm 0.25	0.82 \pm 0.20
	Adult	0.91 \pm 0.20	0.92 \pm 0.12	0.91 \pm 0.02

3.4.2 Updated models of population growth

The initial model validation showed that the models used could predict the population trajectory of red kites in North Scotland very accurately (Table 7, Fig. 14). This is in line with the results presented by Smart *et al.* (2010). Running these models with survival rates adjusted to remove deaths caused by illegal killing (Table 7) predicted a much higher population growth rate, which is remarkably similar to the observed population growth in the red kite population in the Chilterns (Fig. 15).

Table 7. Mean (\pm SD) stochastic population growth rate (r) as well as observed and predicted number of breeding pairs of red kites using survival estimates for juvenile, sub-adult and adult red kites specific for the North Scotland population in scenarios with and without illegal killing between 1989 and 2014.

Scenario	Mean $r \pm$ SD	Mean number of breeding pairs		
		2006	2010	2014
Observed		41	52	64
Smart <i>et al.</i> (2010)	0.138 \pm 0.301	39	50	62
Updated wild survival	0.150 \pm 0.278	38	51	65
No illegal killing	0.281 \pm 0.273	309	685	1481

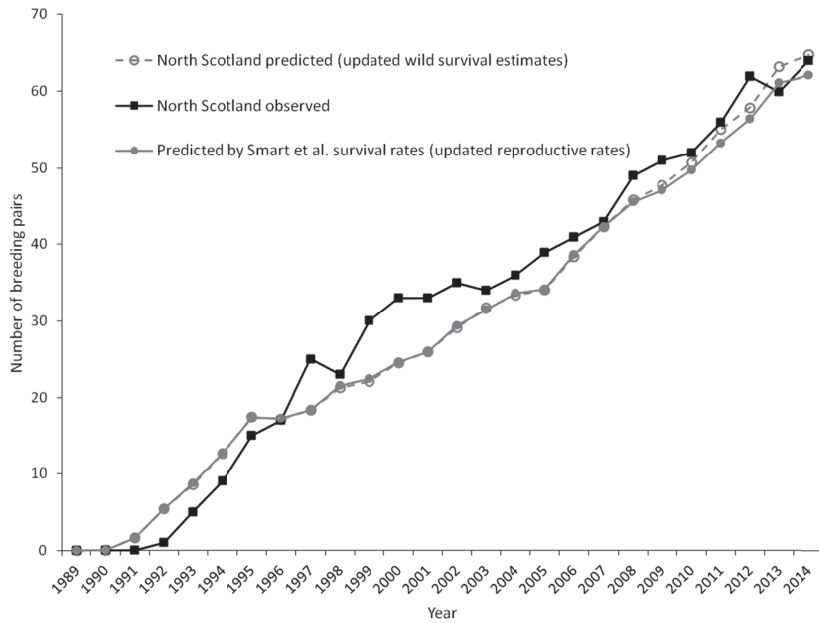


Figure 14. The number of observed pairs of breeding red kites in North Scotland between 1989 and 2014 (solid black line, filled squares). In addition, the predicted number of breeding pairs using the age-specific survival rates of released and wild-hatched birds presented in Smart et al. (2010), with the updated reproductive rates (solid grey line, filled circles) is shown. Finally, the predicted number of breeding pairs using the updated age-specific survival rates of wild-hatched birds and updated reproductive rates (this study; dashed grey line, unfilled circles) in the North Scotland population is shown. The modelled population trajectories presented are the means for each year based on 1,000 iterations in program Vortex.

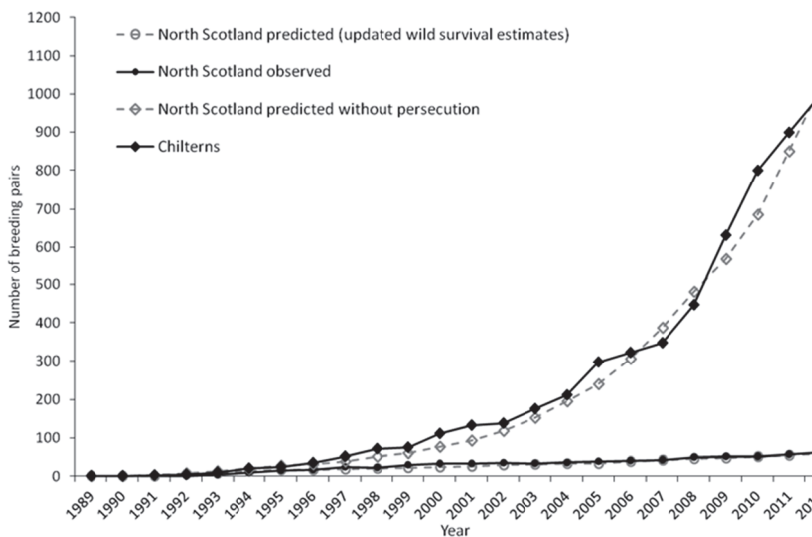


Figure 15. Comparison between the observed (solid lines) and predicted (dashed lines) population growth of red kites in North Scotland and the Chilterns in England. When excluding mortality caused by illegal killing in the North Scotland population, the predicted population growth (open diamonds) is similar to the population growth observed in the Chilterns (black diamonds), which is not subjected to illegal killing. The modelled population trajectories presented are the means for each year based on 1,000 iterations. The expansion of the population in the Chilterns means that for values 2010-2012 may not be as accurate as previous estimates and it is difficult to get reliable estimates of the number of pairs after 2012.

The effect of rodenticide-induced deaths of red kite nestlings on the breeding population size was relatively small. For example, the predicted number of breeding pairs from a model that assumed that no red kite nestlings died of rodenticide poisoning showed close resemblance with the observed population growth (Fig. 16). If all red kite nestlings that were confirmed dying from rodenticides we added back into the population, there would only be 69 pairs of red kites in 2014 (i.e. five more than in fact observed in 2014).

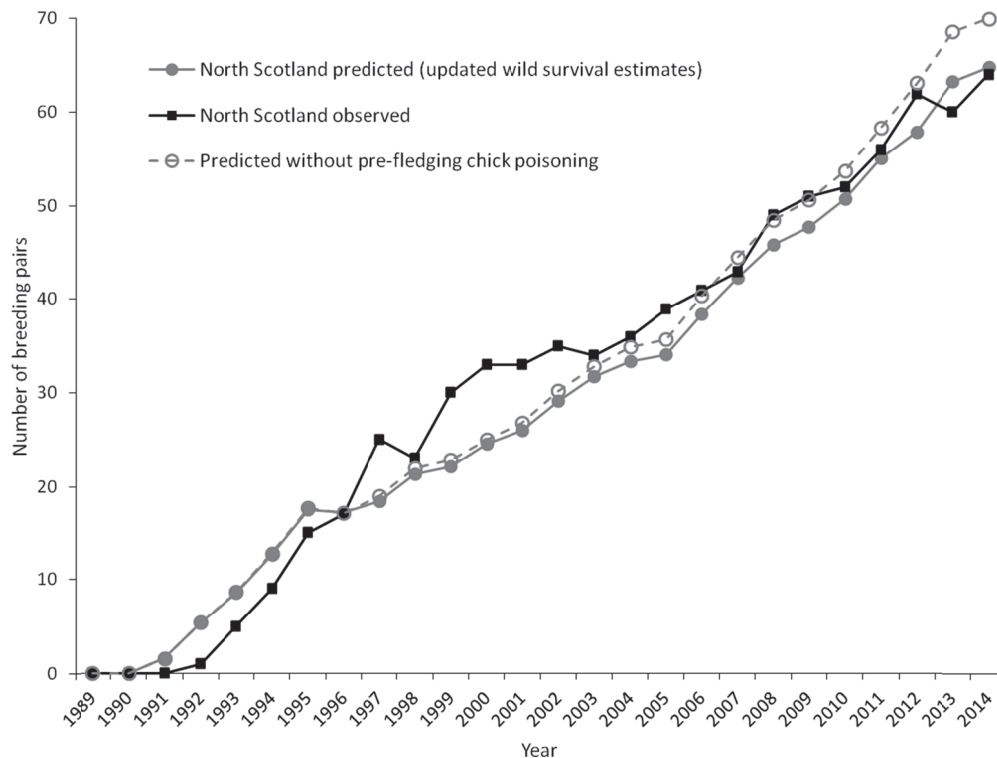


Figure 16. The number of observed breeding pairs of red kites in the North Scotland population between 1989 and 2014 (solid black line, filled squares) contrasted with the predicted number of red kite pairs with (solid grey line, filled circles) and without (dashed grey line, open circles) losses of broods through rodenticides poisoning. The modelled population trajectories presented are the means for each year based on 1,000 iterations.

3.5 Predicting the future population growth

3.5.1 Short and long-term effects of the Ross-shire event

The future population growth rate of the North Scotland red kite population was predicted initially by running a baseline scenario that did not include mortality caused by the Ross-shire event. However, it did include the mortality caused by illegal killing at levels recorded during the years 1989-2014 (Table 6). Under these circumstances, the population was expected to have an annual mean (\pm SD) stochastic population growth rate (r) of $0.101 (\pm 0.197)$. After accounting for losses during the Ross-shire event, the population was estimated to grow with a mean annual stochastic rate of $0.099 (\pm 0.200)$. The effect of the Ross-shire event is likely to have a small impact on the population growth rate in the short term (Table 8), but a more noticeable long-term effect. For example, in 2018, the effects of the Ross-shire event are expected to result in five fewer breeding pairs (i.e. 81 ± 5 pairs) compared with a scenario where the Ross-shire event had not taken place (i.e. 86 ± 2 pairs; Table 8). When predicting further into the future, the estimates become less accurate. Nevertheless, in 2044, the baseline scenario resulted in a population of 550 ± 15 breeding pairs, compared with $513 (\pm 15)$ pairs than if the Ross-shire event had not taken place. Thus,

the Ross-shire event is likely to result in a population size that is 37 (6.7%) pairs fewer than predicted under the scenario where no poisoning event had occurred (Table 8; Fig. 17). Eventually the population will reach 550 pairs, but the Ross-shire event has delayed this to happen.

3.5.2 The effects of additional cumulative mortality caused by wind turbine collisions

The effect of additional cumulative mortality of red kites caused by collisions with wind turbines was modelled using scenarios that tried to create mortality predicted which would be equivalent to 2-3 fatalities per year. The results suggested that there were small differences in population growth rate between scenarios where the mortality affected sub-adult, adult or a mix of sub-adult and adult birds. In addition, there was little variation in population growth rate between scenarios where two or three birds were killed by wind farm collisions (e.g. Fig. 18). For example, all modelled scenarios using two fatalities had a mean annual stochastic population growth rate of between 0.094 and 0.096 and a predicted number of breeding pairs between 444 and 489 (Table 8). Similarly, all modelled scenarios using three fatalities had a mean annual stochastic population growth rate of between 0.090 and 0.092 and a predicted number of breeding pairs between 407 and 454 (Table 8).

When increasing the number of fatalities to five and 10 birds, the population growth declined, and the resulting population size in 2044 are clearly much lower than other scenarios (Table 8; Fig. 19).

3.5.3 The effects of the Ross-shire event and cumulative mortality caused by wind turbine collisions

The results of the cumulative impacts of illegal killing, the Ross-shire event and the additional annual mortality of between two and 10 birds, using the same settings as in previous models suggested that these scenarios predict a lower mean stochastic growth rate than the BPM (i.e. between 0.071 and 0.092; Table 8). In addition, the cumulative impacts of illegal killing, the Ross-shire event and the mortality caused by wind turbine collisions would result in a lower number of breeding pairs of red kites in the North Scotland population, specifically when more than five birds die annually due to collisions with wind turbines (Table 8; Fig. 20). For example, the number of breeding red kite pairs in North Scotland in 2044 is expected to be 340 (± 13) and 244 (± 13) when five and 10 birds are killed annually, respectively (Table 8). This is 38.2% and 55.6% lower than in a scenario without mortality caused by the Ross-shire event and collisions with wind turbines.

Table 8. Mean (\pm SE) stochastic population growth rate (r) as well as predicted number of breeding pairs of red kites in future years using survival estimates for juvenile, sub-adult and adult red kites specific for the North Scotland population under different scenarios. The Baseline Predictive Model (BMP) for the years 1989-2044 includes observed levels of illegal killing but not the mortality caused by the Ross-shire event. However, the mortality caused by the Ross-shire event is included in the scenario with that name. The expected mortality from wind farms near the core of the breeding area is modelled in scenarios under Wind farm mortality, and the combined effects of the Ross-shire event and the expected additional mortality from turbine collisions is modelled in scenarios under Ross-shire and wind farm. See text for details of the various scenarios.

Scenario	Mean $r \pm$ SD	Mean \pm SE breeding pairs in year						
		2014	2015	2016	2018	2024	2034	2044
BPM 1989-2044	0.101 \pm 0.197	65 \pm 1	70 \pm 2	75 \pm 2	86 \pm 2	131 \pm 2	265 \pm 7	550 \pm 15
Ross-shire event	0.099 \pm 0.200	63 \pm 1	67 \pm 2	71 \pm 2	81 \pm 2	124 \pm 3	251 \pm 7	513 \pm 15
<u>Wind farm mortality</u>								
2 Sub-adult per year	0.096 \pm 0.199	64 \pm 1	69 \pm 2	73 \pm 2	83 \pm 2	121 \pm 3	235 \pm 7	478 \pm 14
3 Sub-adults per year	0.092 \pm 0.203	64 \pm 1	69 \pm 2	73 \pm 2	82 \pm 2	118 \pm 3	226 \pm 7	454 \pm 14
2 Adults per year	0.094 \pm 0.203	64 \pm 1	69 \pm 2	72 \pm 2	81 \pm 2	116 \pm 3	222 \pm 6	444 \pm 13
3 Adults per year	0.090 \pm 0.208	62 \pm 1	67 \pm 2	69 \pm 2	78 \pm 2	109 \pm 3	207 \pm 7	407 \pm 15
2 Mixed per year	0.096 \pm 0.202	65 \pm 1	70 \pm 2	74 \pm 2	82 \pm 2	123 \pm 3	240 \pm 7	489 \pm 15
3 Mixed per year	0.091 \pm 0.205	64 \pm 1	69 \pm 2	73 \pm 2	80 \pm 2	114 \pm 3	212 \pm 7	427 \pm 15
5 Mixed per year	0.088 \pm 0.210	64 \pm 1	69 \pm 2	73 \pm 2	79 \pm 2	109 \pm 3	202 \pm 6	400 \pm 14
10 Mixed per year	0.074 \pm 0.231	64 \pm 1	69 \pm 2	72 \pm 2	76 \pm 2	92 \pm 3	149 \pm 6	270 \pm 12
<u>Ross-shire and wind farm</u>								
Ross-shire & 2 Adults per year	0.092 \pm 0.209	62 \pm 1	66 \pm 2	69 \pm 2	78 \pm 2	111 \pm 3	214 \pm 7	432 \pm 14
Ross-shire & 3 Adults per year	0.088 \pm 0.214	53 \pm 1	56 \pm 2	59 \pm 2	66 \pm 2	95 \pm 3	184 \pm 7	365 \pm 14
Ross-shire & 5 Mixed per year*	0.083 \pm 0.215	60 \pm 1	63 \pm 2	67 \pm 2	72 \pm 2	98 \pm 3	177 \pm 6	340 \pm 13
Ross-shire & 10 Mixed per year*	0.071 \pm 0.236	55 \pm 1	57 \pm 2	60 \pm 2	63 \pm 2	78 \pm 3	131 \pm 6	244 \pm 13

*5 Mixed was: 2 female sub-adults, 1 female adult; 1 male sub-adult, 1 male adult.

*10 Mixed was: 3 female sub-adults, 2 female adults; 3 male sub-adults, 2 male adults.

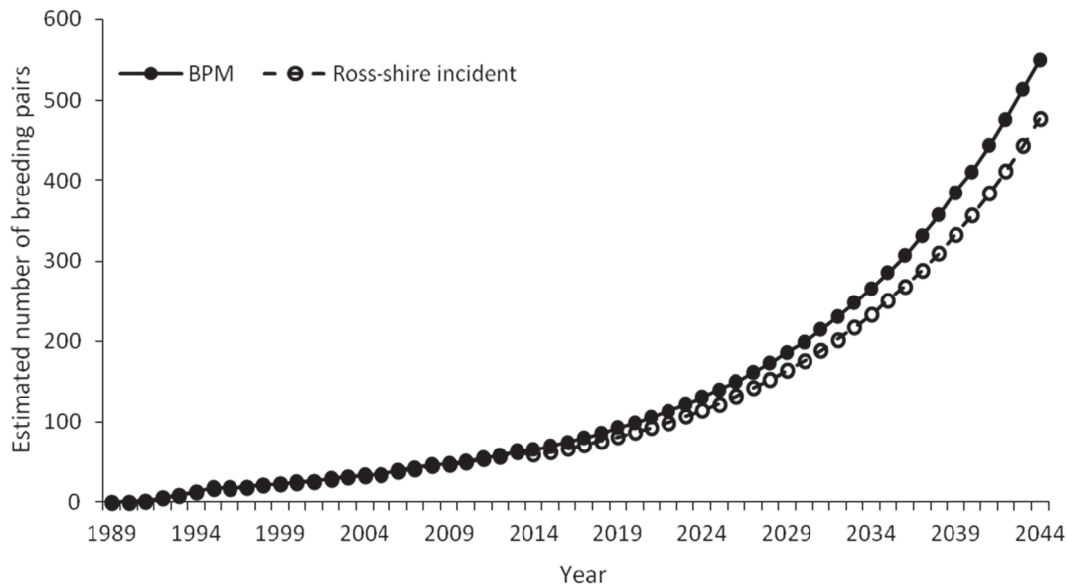


Figure 17. Comparison between two scenarios of the predicted population growth of red kites in North Scotland. In the first (Baseline Predictive Model, BPM; solid line and circles) the mortality recorded during the Ross-shire event was excluded. In the second scenario (Ross-shire event; dashed line and open circles), this mortality was included as a single event that caused the death of 16 red kites. The mortality was equally spread between juvenile males and females and adult males and females. Both model included illegal killing at levels published in this report. The modelled population trajectories presented are the means for each year based on 1,000 iterations.

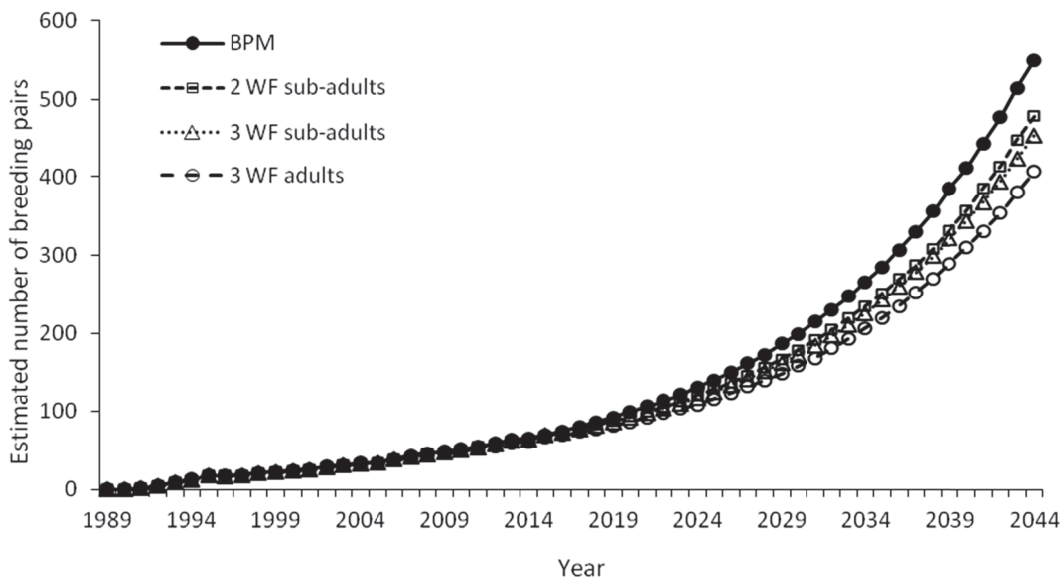


Figure 18. Comparison between four scenarios of the predicted population growth of red kites in North Scotland. In the first (Baseline Predictive Model, BPM; solid line, solid circles), there was no wind farm-related mortality. However, this mortality was included as an annual mortality of two (dashed line, open squares) and three (dotted lined, open triangles) sub-adult birds, respectively, as well as three adult birds (dashed line, open circles). All four scenarios included illegal killing at levels published in this report. The modelled population trajectories presented are the means for each year based on 1,000 iterations.

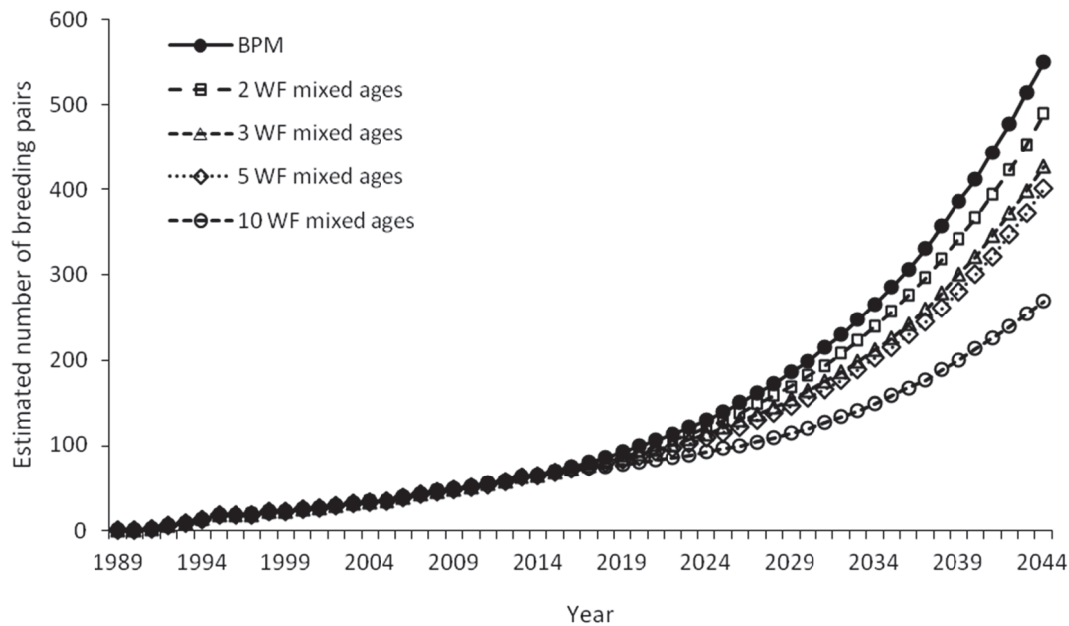


Figure 19. Comparison between five scenarios of the predicted population growth of red kites in North Scotland. In the first (Baseline Predictive Model, BPM; solid line, solid circles), there was no wind farm-related mortality. However, this mortality was included as an annual mortality of 2 to 10 birds of different ages. All five scenarios included illegal killing at levels published in this report. The modelled population trajectories presented are the means for each year based on 1,000 iterations.

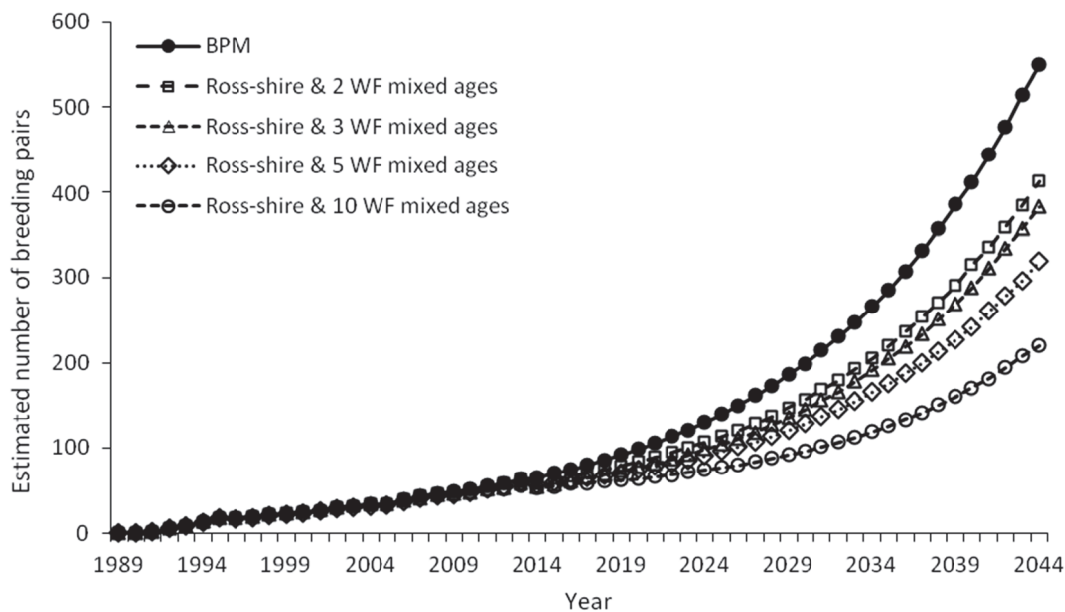


Figure 20. Comparison between five scenarios of the predicted population growth of red kites in North Scotland. In the first (Baseline Predictive Model, BPM; solid line) the mortality recorded during the Ross-shire event was excluded. The other scenarios included mortality levels recorded at the Ross-shire event (i.e. as a single event where 16 red kites were found dead) as well as between 2 and 10 fatalities annually caused by collisions with wind turbines. All models included mortality caused by illegal killing at levels published by in this report. The modelled population trajectories presented are the means for each year based on 1,000 iterations.

4. DISCUSSION

4.1 Observed population growth and reproduction in North Scotland 1989-2014

The monitoring of the four Scottish red kite populations has provided very detailed data on number of breeding pairs and breeding performance. The observed population growth is much lower in the North Scotland population than the three other red kite populations in Scotland (e.g. Fig. 4). The low population growth in North Scotland (i.e. Fig. 4b) is unlikely to be explained by poor breeding performance. Breeding success and the number of fledglings produced per successful nest in North Scotland was highest and second highest, respectively, among the four Scottish red kite populations (Figs 5 & 6). In fact, the productivity recorded in the North Scotland population is one of the highest in the UK (Newton *et al.*, 1994; Evans *et al.*, 1999; Smart *et al.*, 2010). Yet, the North Scotland population grew much more slowly compared with the other populations with lower breeding performance (e.g. Dumfries & Galloway and Central Scotland). There was some evidence suggesting a recent decline in breeding productivity in the North Scotland population. Currently, the causes of this decline are unknown. There were significantly fewer fledglings per breeding attempt from 2007 onwards compared with the period covered in Smart *et al.* (2010). However, the mean breeding success over the last 10 years of 1.755 chicks per pair is still relatively high, suggesting that density-dependent reproduction is not at a level that would limit population growth. Finally, there is very limited movement of individual kites from the North Scotland population to join other geographically separated red kite populations (Smart *et al.*, 2010), and it is therefore unlikely that undetected dispersal into any other Scottish or English populations is causing the slow population growth.

4.2 Evaluation of population growth model accuracy

An early assessment of the demographic rates of the red kite populations in the Chilterns and in North Scotland up to and including 1995 suggested that the populations would exceed 100 breeding pairs in 1998 and 2007, respectively (Evans *et al.*, 1999). The population in the Chilterns passed 100 breeding pairs in year 2000, just two years after the prediction. The modelling work presented here suggests that with the estimates of illegal killing and the mortality recorded during the Ross-shire event, the North Scotland population will not exceed 100 breeding pairs until 2023. The contrast between the red kite populations in the Chilterns, which now is well above 1,000 breeding pairs, and North Scotland is stark. The strong population growth in England and Wales has meant that the Rare Breeding Bird Panel (RBBP) has now dropped red kite in their annual report. This is a testament to the overall conservation success of the re-introduction programme.

In line with the findings from the study by Smart *et al.* (2010), the baseline model using updated estimates of survival in the North Scotland population accurately predicted the number of breeding pairs of red kites between 1989 and 2014 (Fig. 14). In addition, the models employed here showed that if the illegal killing did not happen in North Scotland, the population would have grown in a manner that shows strong resemblance with the red kite population in the Chilterns, which is not subjected to significant illegal killing, up to 2012, when the monitoring in the Chilterns was reduced.

4.3 Illegal killing

There was no significant trend in the level of illegal killing over time or when comparing the two time periods 1989-2006 and 2007-2014 (e.g. Fig. 9). There were some apparent increases in the percentage of deaths caused by illegal killing in 2nd-year birds (Fig. 10). However, there was also a decrease in the percentage of cases where the cause of death was unknown. The analytical methods used in the toxicological tests have improved since 2007 and the awareness of wildlife crime among the general public may have increased, leading to a higher reporting rate of dead birds. It cannot be ruled out that more cases than

previously can be positively identified as being caused by illegal killing. Thus, based on this comparison alone, it is difficult to say whether or not there has been a real increase in illegal killing affecting sub-adult red kites. However, supporting evidence that there has been little change in the illegal killing levels comes from the survival analyses using mark-resighting data between 1989 and 2012. These analyses suggested that the age-specific survival estimates for wild-hatched red kites were very similar to those reported in Smart *et al.* (2010), with a very small decline in 2nd-year and adult survival and a small increase in 1st-year survival.

In addition to illegal killing of fledged birds, some red kite chicks died due to rodenticide poisoning (i.e. pesticide misuse). This can affect relatively large numbers of chicks in some years (e.g. N=10 in 2010; Fig. 11). However, when modelling what the effect these deaths have had on population growth rates, only a small effect on population growth was observed (Fig. 16).

4.4 The short- and long-term effects of the Ross-shire event on red kites

The models presented here suggested that the Ross-shire poisoning event that affected the North Scotland red kite population in 2014 will only have a relatively small effect on the number of breeding pairs in the short term (i.e. next five years). However, these effects will become more apparent in the long term (i.e. between 10 and 30 years; Table 8). The interruption in population growth caused by the event is likely to result in approximately 37 fewer pairs by 2044. What is not known is how the *frequency* of such events might affect the population growth. The Ross-shire event was the first detected mass-poisoning event since the re-introduction of red kites started in 1989, but if there were similar events in the future, it is likely that the loss of individuals will have a significant effect on the population growth. For example, in France and Spain, farmers have responded to large increases of water voles (*Arvicola terrestris*), which might affect the yield of certain agricultural crops negatively, by dispensing high volumes of rodenticides on their fields. This has resulted in large numbers of raptors, including red kites, being killed when they have eaten the dead and dying water voles (Coeurdassier *et al.*, 2012, 2014). Such large-scale indirect poisoning has reduced the number of red kites in these countries, and the Ross-shire event could, if repeated, potentially have similar effects on the red kite population in North Scotland. Further modelling work exploring how the frequency (and severity, i.e. the number of individuals killed) of similar events could be done if comparable events occurred again.

4.5 The effects of wind turbine mortality on red kites

Red kites and other raptors are at risk of colliding with wind turbines and associated overhead wires linking the wind farms to the main electricity grid (Bevanger 1994; Barrios & Rodriguez, 2004; Duffy & Steward, 2008). Given the increased number of operational, consented and planned wind farms in Scotland in the last decade, there has been a fear that the red kite populations in Scotland would decline in numbers. For example, the wind farm Braes of Doune, which is located close to the core of the centre of the Central Scotland population, was expected to cause a number of red kite fatalities per year, and carcass searches under the turbines have confirmed that this has been the case (Duffy & Steward, 2008). However, in spite of this additional mortality, which has remained constantly low over the years (Duffy & Urquhart, 2014), the red kite population in Central Scotland has continued to increase at a higher rate than the North Scotland population (e.g. Fig. 4b). This has primarily been achieved by range expansion out from the core area near the Braes of Doune wind farm (Duffy & Urquhart, 2014).

In this study, the predictive models specific to the North Scotland population used a baseline mortality estimate provided by SNH of two to three red kites being killed annually by collisions with wind turbines. These models predicted that this relatively low level of mortality

would have a negligible effect on the population growth of red kites in North Scotland (e.g. Fig. 18). However, when the number of fatalities increased to 10 per year, the population growth rate was lowered from 0.101 (\pm 0.197) in the baseline scenario without wind farm-related mortality to 0.074 (\pm 0.231). This was predicted to reduce the population size by approximately 280 pairs by 2044 (Table 8, Fig. 19).

4.6 Cumulative effects of illegal killing, wind turbines mortality and the Ross-shire event

Cumulative effects of several unrelated sources of mortality and breeding performance, such as illegal killing and afforestation, have been shown to negatively affect the territory occupancy of golden eagles in Scotland (Whitfield *et al.*, 2004a, 2007, 2008), and it is likely that red kites could show similar changes in territory occupancy if additional deaths by wind turbine collisions were sufficiently frequent. In this study, the predictions show how the cumulative effects of illegal killing, the Ross-shire poisoning event and increased mortality due to collisions with wind turbines would affect the North Scotland red kite population. The results suggest that when the wind turbine-related mortality exceeds five fatalities per year, such mortality could further limit the population growth rate. In the most extreme scenario modelled here, which included illegal killing, the Ross-shire event mortality and 10 birds killed by wind turbines per annum, the mean stochastic population growth rate (\pm SD) would be as low as 0.071 (\pm 0.236), compared with 0.101 (\pm 0.197) in the baseline scenario without extra mortality, which would lead to a predicted population size which is over 300 pairs lower than the baseline scenario by 2044.

In a wider context, the models predicting the future population growth in the North Scotland population under different scenarios suggest that the major factor limiting population growth is illegal killing. Neither the impact of the Ross-shire event nor the predicted additional mortality caused by wind turbine collision are close to causing similar levels of impact on population growth as the levels of illegal killing reported here and by Smart *et al.* (2010). This is best visualised by Figure 15, where in the absence of illegal killing, the population trajectory shows strong resemblance with the observed population growth in the Chilterns. In comparison, Figures 17-20 suggest that the cumulative effects of the Ross-shire event and wind farms have far lesser impact on population growth.

Overall it is clear that the combination of detailed data from intense population monitoring and powerful analytical tools such as the individual-based stochastic PVA software Vortex (Lacy *et al.*, 2005) can provide realistic insights of how the red kite populations are likely to grow in the future.

4.7 Red kites and illegal killing

There are a number of reasons why the red kite is vulnerable to illegal killing, especially in terms of being killed by poison. Most importantly, kites often scavenge carrion, so they are commonly the first victims of illegal poisoned meat baits. Scientific studies have shown that red kites constitute little threat to gamebirds (Davis & Davis, 1981; Coeurdassier *et al.*, 2012). However, this view may not be shared by all land managers, who might target red kites using baits laced with pesticides. In addition, it is possible that illegal baits are aimed at other predators, such as other birds of prey, red fox (*Vulpes vulpes*) and corvids, which many game managers regard as being a more potent threat to gamebirds than red kites, but the kites become victims. Regardless of the intention of the person using the poison, the indiscriminate nature of poisoned baits means that scavengers such as red kites are likely victims. In addition, such baits are not only a threat to wildlife, but also to pets and potentially humans too (RSPB, 2013; RSPB Scotland, 2013). Another reason for kites being vulnerable to poisoned baits is that they often disperse at the end of their first year and into their second year. In this exploratory phase of their life, they are likely to cover large areas in search for

suitable habitats, and they may encounter poisoned baits. The North Scotland population, which breed in predominantly lowland farmland areas interspersed with woodlands, are surrounded by active grouse moors (Anderson *et al.*, 2009), which are associated with high levels of persecution of raptors (Whitfield *et al.*, 2004a; Whitfield *et al.*, 2008; Fielding *et al.*, 2011). For example, for the years 1989 to 2006, many illegally killed North Scotland red kites were found within 50 km from the core breeding area outside Inverness (Smart *et al.*, 2010), suggesting that local and regional high levels of illegal killing can limit the population growth of red kites. In this context, it is important to point out that the Ross-shire event occurred on land that was not managed for red grouse. It therefore constitutes an atypical event in a local area not historically associated with high levels of illegal killing.

Overall, this study has shown that the population growth of red kites in North Scotland is not limited by low breeding performance. In addition, it has shown that illegal killing constitutes a larger constraint on population growth than a single poisoning event killing at least 12 (but possibly 16) red kites and the expected mortality caused by collisions with wind turbines. However, it is still very important to evaluate the potential effects of wind turbine related mortality before applications to build wind farms are approved.

The two main partners in the re-introduction programme (RSPB and SNH) have been involved in a large number of initiatives to reduce the illegal killing of birds of prey in general and red kites in particular. For example, both SNH and RSPB are partners in "The Partnership for Action Against Wildlife Crime Scotland" (PAW Scotland), which represents a wide range of organisations concerned with the prevention and tackling of crimes against wildlife. As illegal killing is still ongoing wildlife crime prevention initiatives based on education and awareness-raising, better policing and stronger sentences are still required and the messages of these reinforced.

4.8 Technical aspects of the ecological modelling in this study

The resighting data from 2002 onwards were relatively sparse and declined over time (Figs 2 & 3). The decrease in available resighting data reached the point in 2013 and 2014 where the resighting rate was so low that it was not possible to accurately estimate year-specific survival for these years. This was unfortunate, as a major objective of this study was to compare survival rates for the period analysed by Smart *et al.* (2010) and later years (i.e. 1989-2006 vs. 2007-2014). Fortunately, the resighting data up to 2012 were of high enough quality to make the age-specific survival estimates robust. Thus, the comparison between the two time periods could be completed, but without resighting data for the years 2013 and 2014. By including time and cohort in the models, most of the decreases in resighting rate were accounted for. Interestingly, these factors were clearly important in determining probability of resighting, as they were included in all the top models.

The Variance Component Estimation in Program MARK is a little known method to calculate robust survival estimates over years (White *et al.*, 2001). This method takes the errors around each year-specific estimate into account to produce the most robust overall mean for a specific age-class. It is therefore similar to a weighted mean that takes the sample size of different categories into account when calculating a mean value. This means that the overall average values for survival by age class are likely to be influenced by the years with the largest samples of data, which also tends to have the smaller errors around the mean. For this study, the consequence might be that any recent change in survival rates may have gone undetected, as later years had lower resighting rates, which resulted in slightly larger errors around the year-specific estimates from 2004 onwards (cf. Fig. 12).

4.9 Suggestions of future work

This study has shown that a combination of dedicated fieldwork aimed at nest monitoring and identification of individuals combined with state-of-the-art ecological modelling can accurately predict population growth of a raptor of conservation concern. This can give stakeholders important insights into how change in land-use patterns might affect the population's ability to grow and indeed its viability.

It is clear that the nest monitoring of red kites in the North Scotland population is of extremely high quality, and that there are few breeding pairs that go undetected. However, since 2003, the number of resightings per individual has dropped, despite the fact that there are many more wing-tagged red kites available to be resighted now than in the years 1989-2002 (Fig. 2). To better estimate year-specific survival so that any changes in survival can be detected, it is important to quickly increase the effort that goes in to fieldwork dedicated to resighting wing-tagged red kites in North Scotland. Ideally, this should be done in two time periods; during the breeding season when older birds are likely to be associated with a nest, and during the winter, when most 1st-year birds and many adult birds form winter roosts. It would also be desirable if all bird monitoring staff at wind farms (i.e. pre and post-construction, normally undertaken by ecological consultancies) were required to submit records of wing-tagged red kites. This would undoubtedly increase the number of records and would benefit red kite conservation.

If such an increase in resighting effort was made, it is possible that year-specific estimates could be calculated for the years 2013 and 2014. In the light of the evidence presented in this report that there is no decline in levels of illegal killing, it is important to consider a rapid response to increase the resighting rate of red kites in North Scotland.

5. CONCLUSIONS

The observed population growth in the North Scotland red kite population continues to be slow, but the breeding performance is still high. Thus, other constraints are acting on this population. It is likely that illegal killing is the main reason for the observed population growth. In line with this, the predictive models using published data on confirmed illegal killing of red kites could accurately replicate the observed number of breeding pairs in the North Scotland population as well as a reference population in the Chilterns in England that is not subjected to illegal killing.

The predictive models presented in this report suggest that the Ross-shire poisoning event that affected the North Scotland red kite population in 2014 will have a relatively minor effect on the number of breeding pairs in the short term (i.e. next five years), and a slightly more noticeably effect in the long term (i.e. between 10 and 30 years).

Another set of predictive models suggested that a low level of mortality caused by collisions with wind turbines (i.e. two to three birds per year) would have limited effects on the population growth of red kites in North Scotland. However, when the number of fatalities increased to above five per year the predicted population size could be over 200 pairs fewer in 2044 than the baseline scenario that excludes wind turbine-related mortality. Therefore wind farm proposals in the breeding areas of the North Scotland red kite population will continue to be carefully assessed due to potential cumulative impacts on the population.

The cumulative effects of illegal killing, the Ross-shire event and increased mortality due to collisions with wind turbines can severely slow down the population growth rate of red kites in North Scotland. Put together, these unrelated sources of mortality could lead to a predicted population size in 2044 that is up to 306 breeding pairs lower than in the baseline scenario, which excludes the mortality caused by the poisoning event and the wind turbine-related mortality. However, in a wider perspective, it is clear that ongoing illegal killing is still the major factor limiting the population growth of red kites in North Scotland. For example, the population size in North Scotland in 2015, in the absence of illegal killing, was predicted to be around 1,500 breeding pairs in the absence of any natural ecological constraints.

Finally, this study has shown that the combination of long-term monitoring of the number of breeding pairs, breeding performance and powerful analytical tools can aid the decision-making regarding planning as well as conservation and management of species of high conservation concern. For these tools to continue to be effective in the future, it is important that high-quality resighting data of tagged red kites need to be collected.

6. REFERENCES

- Anderson, B. J., Arroyo, B. E., Collingham, Y. C., Etheridge, B., Fernandez-De-Simon, J., Gillings, S., Gregory, R. D., Leckie, F. M., Sim, I. M. W., Thomas, C. D., Travis, J. M. J. & Redpath, S. M. 2009. Using distribution models to test alternative hypotheses about a species' environmental limits and recovery prospects. *Biological Conservation*, 142, 488-499.
- Barrios, L. & Rodriguez, A. 2004. Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. *Journal of Applied Ecology*, 41, 72-81.
- Bevanger, K. 1994. Bird interactions with utility structures: collision and electrocution, causes and mitigating measures. *Ibis*, 136, 412-425.
- Breitenmoser, U. 1998. Large predators in the Alps: the fall and rise of man's competitors. *Biological Conservation*, 83, 279-289.
- Burgman, M. A., Ferson, S. & Akçakaya, H. R. 1993. *Risk assessment in conservation biology*. Chapman & Hall, London.
- Burnham, K. P. & Anderson, D. R. 2002. *Model Selection and Multi-Model Inference: A Practical Information-Theoretic Approach*. Springer.
- Challis, A., Holling, M., Stevenson, A., Roos, S., Stirling-Aird, P. & Wilson, M. W. 2014. *Scottish Raptor Monitoring Scheme Report 2013*. 48 pp. Stirling, UK.
- Coeurdassier, M., Poirson, C., Paul, J. P., Rieffel, D., Michelat, D., Reymond, D., Legay, P., Giraudoux, P. & Scheifler, R. 2012. The diet of migrant Red Kites *Milvus milvus* during a Water Vole *Arvicola terrestris* outbreak in eastern France and the associated risk of secondary poisoning by the rodenticide bromadiolone. *Ibis*, 154, 136-146.
- Coeurdassier, M., Riols, R., Decors, A., Mionnet, A., David, F., Quintaine, T., Truchetet, D., Scheifler, R. & Giraudoux, P. 2014. Unintentional wildlife poisoning and proposals for sustainable management of rodents. *Conservation Biology*, 28, 315-321.
- Davis, P. E. & Davis, J. E. 1981. The food of the Red Kite in Wales. *Bird Study*, 28, 33-40.
- Deinet, S., Leronymidou, C., McRae, L., Burfield, I. J., Foppen, R. P., Collen, B. & Böhm, M. 2013. *Wildlife comeback in Europe: The recovery of selected mammal and bird species*. Final report to Rewilding Europe by ZSL, BirdLife International and the European Bird Census Council. 308 pp. Rewilding Europe, London, UK.
- Duffy, K. & Steward, M. 2008. *Turbine search methods and carcass removal trials at the Braes of Doune windfarm*. Natural Research information Note 4. 7 pp. Natural Research Banchory, UK.
- Duffy, K. & Urquhart, B. 2014. Braes of Doune Wind Farm; Report on red kite studies 2004-2012. Prepared by Natural Research Projects Ltd on behalf of the Braes of Doune Ornithological Steering Group. 49 pp. Natural Research Banchory UK.
- Etheridge, B., Riley, H., Wernham, C., Holling, M. & Stevenson, A. 2013. *Scottish Raptor Monitoring Scheme: Report 2011*. 44 pp.

- Evans, I. M., Summers, R. W., O'Toole, L., Orr-Ewing, D. C., Evans, R., Snell, N. & Smith, J. 1999. Evaluating the success of translocating Red Kites *Milvus milvus* to the UK. *Bird Study*, 46, 129–144.
- 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.
- Fielding, A., Haworth, P., Whitfield, P., D., M. & Riley, H. 2011. *A Conservation Framework for Hen Harriers in the United Kingdom*. 90 pp. JNCC, Peterborough.
- Godfray, H. C. J. & Crawley, M. J. 1998. Introductions. *In*: Sutherland, W. J. (ed.) *Conservation Science and Action*. 39-65 pp. Blackwell Science, Oxford.
- 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.
- IUCN. 1998. *Guidelines for re-introductions*. Gland, Switzerland and Cambridge, UK.
- IUCN/SSC. 2013. Guidelines for reintroductions and other conservation translocations. 57 pp. IUCN Species Survival Commission, Gland, Switzerland.
- Laake, J. L. 2013. RMark: *An R interface for analysis of capture-recapture data with MARK*. Alaska Fish. Sci. Cent., NOAA, Natl. Mar. Fish. Serv., 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.
- Liberg, O., Chapron, G., Wabakken, P., Pedersen, H. C., Hobbs, N. T. & Sand, H. 2012. Shoot, shovel and shut up: cryptic poaching slows restoration of a large carnivore in Europe. *Proceedings of the Royal Society B-Biological Sciences*, 279, 910-915.
- Meffe, G. K. & Carroll, C. R. eds. 1994. *Principles of Conservation Biology*. Sinauer Associates, Sunderland, MA.
- Newton, I., Davis, P. E. & Moss, D. 1994. Philopatry and population growth of Red Kites, *Milvus milvus*, in Wales. *Proceedings of the Royal Society of London B: Biological Sciences*, 257, 317-323.
- R Development Core Team. 2011. R: A language and environment for statistical computing. 2.13.1, <http://www.R-project.org>, Vienna. Austria.
- Ritchie, E. G., Elmhagen, B., Glen, A. S., Letnic, M., Ludwig, G. & McDonald, R. A. 2012. Ecosystem restoration with teeth: what role for predators? *Trends in Ecology & Evolution*, 27, 265-271.
- 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.
- Schaub, M. 2012. Spatial distribution of wind turbines is crucial for the survival of red kite populations. *Biological Conservation*, 155, 111-118.

- 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.
- White, G. C. & Burnham, K. P. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study*, 46, 120-139.
- White, G. C., Burnham, K. P. & Anderson, D. R. 2001. Advanced features of Program MARK. *Wildlife, Land, and People: Priorities for the 21st Century*. Bethesda, Maryland, USA: 368-377.
- 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. 2008. *A conservation framework for golden eagles: implications for their conservation and management in Scotland*. 163 pp. Scottish Natural Heritage, Edinburgh.
- Whitfield, D. P., Fielding, A. H., McLeod, D. R. A., Morton, K., Stirling-Aird, P. & Eaton, M. A. 2007. Factors constraining the distribution of Golden Eagles *Aquila chrysaetos* in Scotland. *Bird Study*, 54, 199-211.
- Wotton, S. R., Carter, I., Cross, A. V., Etheridge, B., Snell, N., Duffy, K., Thorpe, R. & Gregory, R. D. 2002. Breeding status of the Red Kite *Milvus milvus* in Britain in 2000. *Bird Study*, 49, 278-286.

APPENDIX 1: DEFINITIONS OF CATEGORIES USED BY SCIENCE AND ADVICE FOR SCOTTISH AGRICULTURE (SASA)

As this report builds on red kite population data collected between 1989 and 2014, we used the categories used by SASA in the Wildlife Incident Investigation Scheme (WIIS) up to and including 2012. The slightly different categories and lengthier explanation used by SASA from 2013 onwards have no implications for the results of this report.

Approved use: a pesticide is used in accordance with its conditions of authorisation.

Misuse: the product has not been used according to the conditions of its authorisation, but often just carelessly or accidentally, without the intention of harming animals.

Abuse: a pesticide has been deliberately used in an illegal manner to poison, or to try to poison animals.

Background residues: residues of pesticides found but levels are probably not indicative of lethal poisoning.

Unspecified use: the cause of death was uncertain or the incident could not be classed as being in one of the other categories.

Veterinary products: Incidents arising from veterinary use are not included in the scheme.

www.snh.gov.uk

© Scottish Natural Heritage 2016
ISBN: 978-1-78391-377-0

Policy and Advice Directorate, Great Glen House,
Leachkin Road, Inverness IV3 8NW
T: 01463 725000

You can download a copy of this publication from the SNH website.



Scottish Natural Heritage
Dualchas Nàdair na h-Alba

All of nature for all of Scotland
Nàdar air fad airson Alba air fad