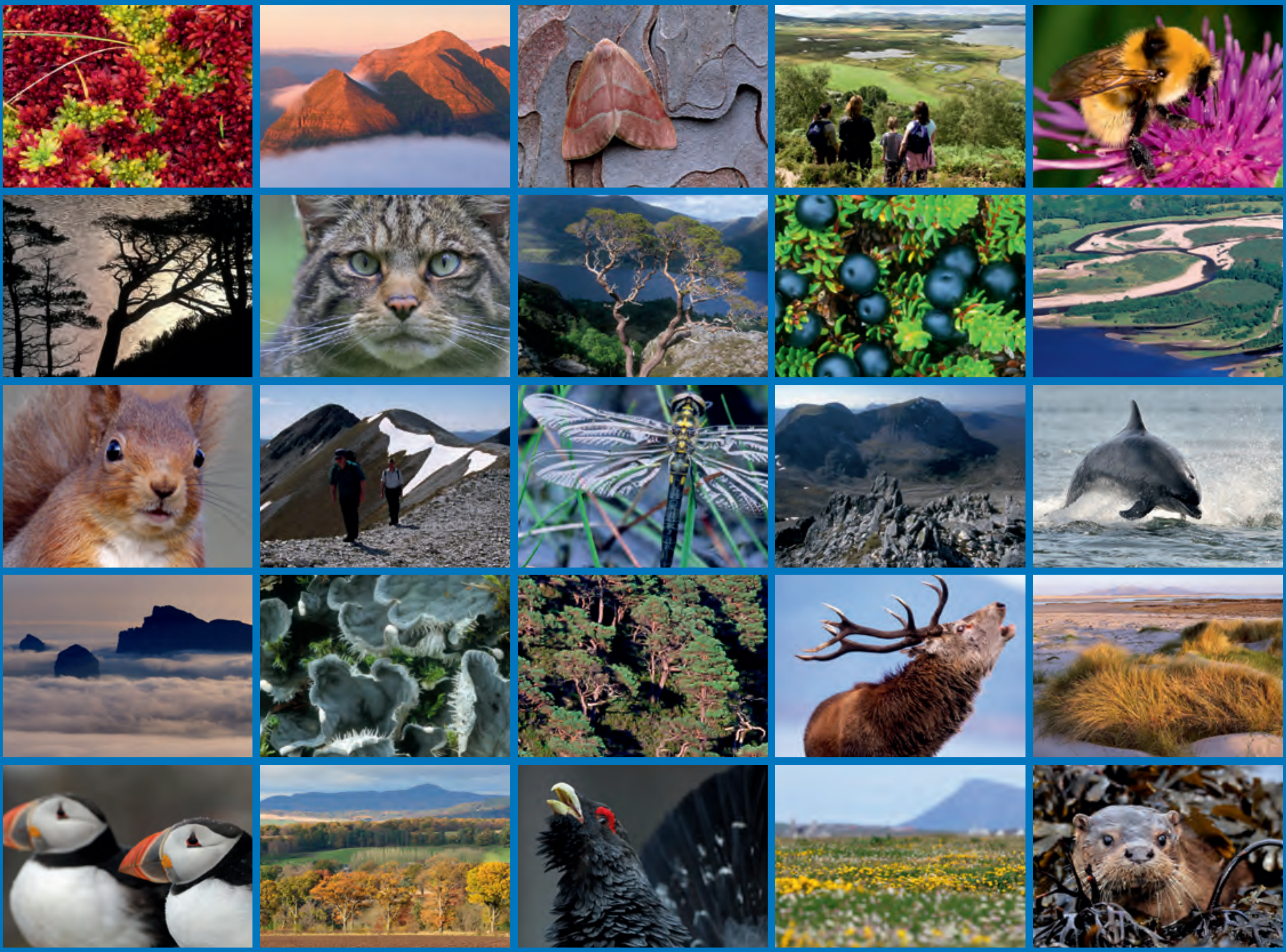
Scottish Natural Heritage
Commissioned Report No. 756

Status and population viability of Svalbard barnacle geese in Scotland



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Status and population viability of Svalbard barnacle geese in Scotland

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COMMISSIONED REPORT

Summary

Status and population viability of Svalbard barnacle geese in Scotland

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Keywords

Solway; *Branta leucopsis*; harvest; MARK; survival analysis; population viability analysis; management.

Background

This report provides analysis and predictive modelling of the Svalbard barnacle goose population which winters in Scotland.

Main findings

- The Svalbard barnacle goose population which winters almost exclusively on the Solway numbered around 34,000 in winter 2011-12.
- Analysis of population data (1959-2011) found no evidence for trends in survival or reproduction. Average parameters were estimated and were used to develop a population model.
- The model includes prediction of the effects of controlled shooting on the population.
- Annually, the population grew by an average of 4.6% between 1985 and 2011. The baseline population model (with no shooting) predicted a growth rate close to this (4.9%).
- For any given percentage harvest rate, the model simulations predicted that the average population growth rate would decrease by approximately the same amount (e.g. if 1% of the population are shot, the growth rate declines by 1%).
- On average, the population growth rate was predicted to remain positive until the annual harvest exceeded 5%, while in 95% of simulations it remained positive until the harvest exceeded 2.5%.

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Table of Contents Page

Acknowledgements

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1. INTRODUCTION

Two populations of barnacle goose (*Branta leucopsis*) are recognised in the UK. One breeds on the east coast of Greenland and winters across a wide area extending from north-west Scotland to west Ireland. The other breeds on the Svalbard archipelago and winters almost exclusively around the Solway Firth. This latter population has grown from a few hundred individuals in the middle of the last century, to a population numbering approximately 34,000 in 2011.

This report updates the previous Population Viability Analysis (PVA) reported in 2005 (Trinder *et al.*, 2005). In this, two models were developed to predict how the population would grow; one included density-dependent population regulation and the other did not. Comparison of each model's predictions with the observed population indicates that the density-independent model produced a closer match to how the population has grown since. Here we present a revised PVA based on updated analysis of demographic data. Using the outputs from this PVA, predictions of future population trends are presented and discussed, together with the potential impacts of future management changes.

2. DATA AND METHODS

2.1 Population counts and demographic trends

The Svalbard-breeding population of barnacle geese winters almost exclusively around the coast of the Solway Firth in south-west Scotland. A few tens of birds may remain for the winter at major autumn staging posts on the British east coast: Lindisfarne and the Loch of Strathbeg. The population is monitored by the Wildfowl and Wetlands Trust, and annual estimates of the population size, proportion of juveniles and mean brood size are available from the late 1950s.

The population grew steadily from around 1,350 in 1958 to a peak of 35,900 in 2010 (the most recent estimate in 2011 was slightly lower at 33,900; Figure 1). These numbers are the adopted counts, which are calculated as the average of those winter counts within 10% of the maximum count (for details see: http://monitoring.wwt.org.uk/our-work/goose-swan-monitoring-programme/species-accounts/svalbard-barnacle-goose/). The population has been protected from shooting on its wintering grounds since the 1950s, and throughout its range since the mid-1970s.

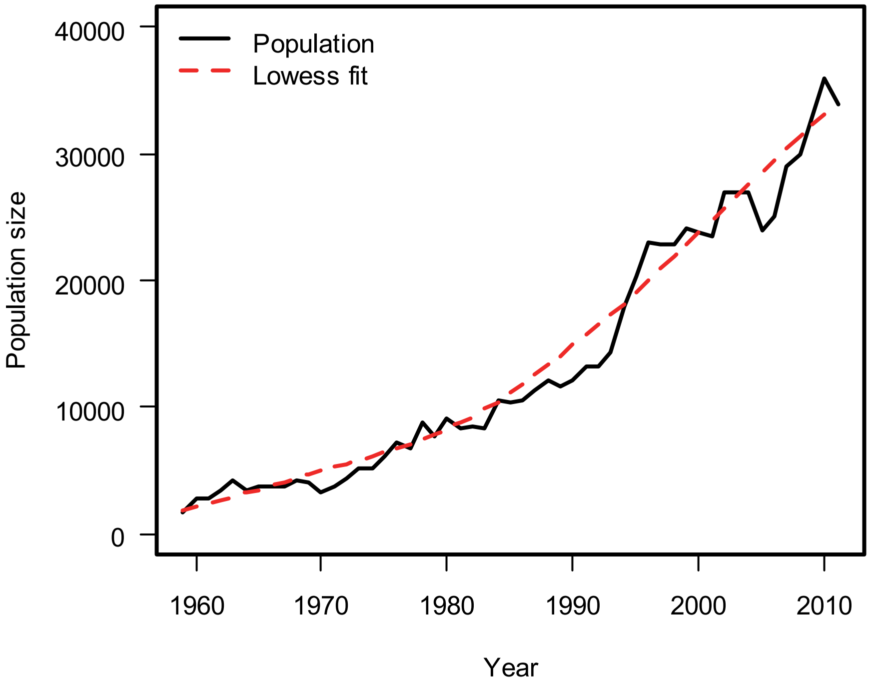

Figure 1. Winter population estimates for Svalbard breeding barnacle geese (black line), with a locally smoothed regression (Lowess) fitted to the counts (dashed red line).

A sample of the flocks has been studied each winter to determine the proportion of juveniles (PJ; 'birds of the year') and the mean brood size (BS); with a typical sample size of more than 10,000 individuals in recent years. From these data, together with the population estimate (N), it is possible to estimate values for the rate of survival (CS) and the minimum proportion of adults (aged three or older) which bred in the preceding summer (PB).

Preliminary scrutiny of the data indicated that there was greater variation during the first half of the time series, so the first and second halves of the data were analysed separately. Relationships between demographic parameters and time were investigated using linear models (Figure 2). Across the entire 53 years, the proportion of juveniles has declined from more than 0.2 to around 0.1, the proportion of breeding birds has declined from around 0.4 to 0.15, survival has increased from around 0.87 to 0.93 and the mean brood size has declined from approximately 2.5 to around 2.0 (Figure 2; Annex 1, Table A1.1). Reproductive rates appear to have changed more during the first period (1959-1985), and this split was even more pronounced when plotted against population size (Annex 1, Figure A1). The only significant relationship found was, however, in the decline in the proportion of breeding birds between 1959 and 1985 (GLM t=-2.02, *p*=0.05; Annex 1, Table A1.2). While reproductive rates have declined historically, the absence of significant trends in relation to the more recent increases in population size (i.e. since 1986) indicates that density dependent regulation has not been acting to regulate population growth (or at least not to an extent which can be detected at the population level).

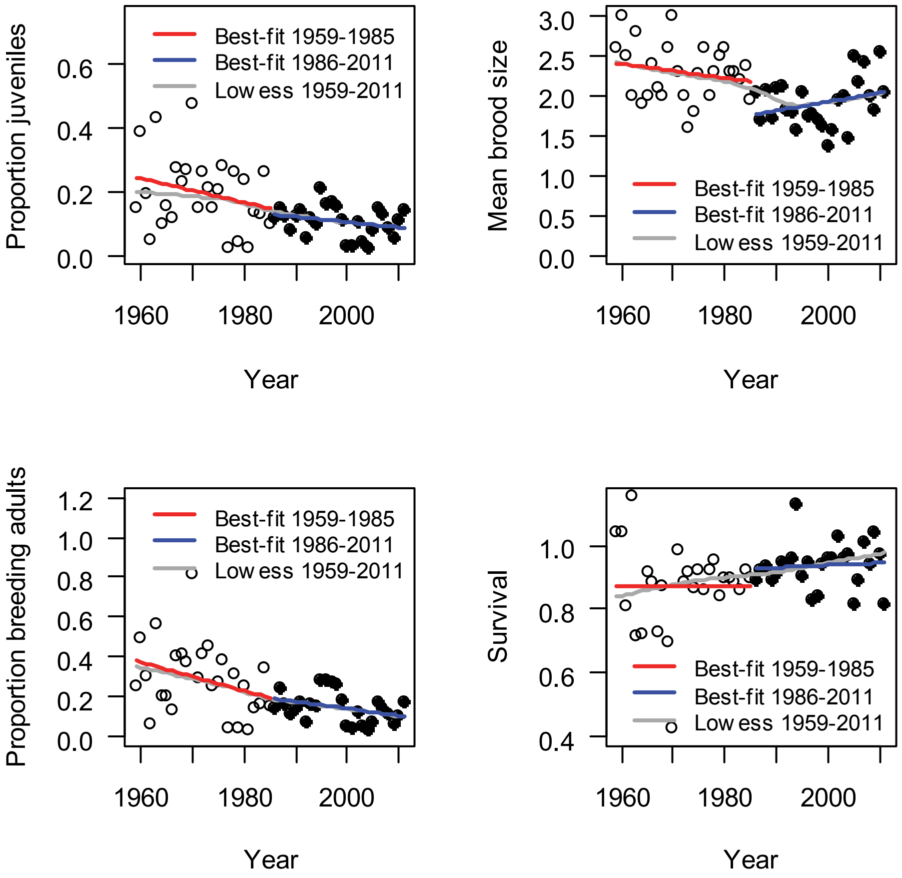

Figure 2. Trend analysis of Svalbard barnacle goose reproduction and survival rates since 1958. Open symbols used for 1959 to 1985, filled symbols for 1986 to 2011. Solid lines are from linear model fits obtained for parameter vs. year (red: 1959-1985; blue: 1986-2011) and grey lines are for locally smoothed regressions (Lowess) to illustrate the suitability of the linear model fits.

2.2 Estimation of demographic parameters

Average demographic parameters for use in the population model were estimated using data collected since 1986 (see above for rationale). They were derived from:

- Population counts (N);
- Age ratios (proportion of juveniles in the population; PJ); and,
- Mean brood size (BS).

The methods for calculating rates of survival and reproduction are summarised as follows. In the following equations subscript 't' is used to denote the current year and 't-1' the previous year.

1. The mean brood size was calculated as the average across all years since 1986;
2. Using the age ratio, the number of juveniles (J) was calculated in each year;

$$J_t = PJ_t * N_t$$

3. Survival (CS) was estimated as the number of adults in year t divided by the total population in year t-1;

$$CS_t = \frac{N_t - J_t}{N_{t-1}}$$

4. The number of second year birds (N2) in year t was calculated as the product of number of juveniles in year t-1 and the survival rate from year t-1 to t;

$$N2_t = J_{t-1} * CS_t$$

5. From this the number of adults in their third year or older (N3) was found by subtraction from the counts;

$$N3_t = N_t - (J_t + N2_t)$$

6. The minimum proportion of breeding adults (PB) was calculated as the number of juveniles divided by the mean brood size to get the number of family units, which was divided by half of the number of third year or older birds.

$$PB_t = \frac{J_t / BS_t}{0.5 * N3_t}$$

Estimating survival from population counts tends to inflate inter-annual variation due to errors in the counts themselves. These errors tend to cancel each other out across a time series (i.e. high and low years balance one another). Whilst this minimises bias in the average survival rate, the variance is exaggerated.

To reduce the estimated variance, survival for the period 1985 to 2011 was re-calculated after substituting running mean population counts (of 3, 5 and 7 years) for the annual counts. This smoothing reduces the magnitude of variation in the counts, and thereby reduces the variance in survival. The mean and standard deviation of survival calculated using population counts averaged over 1, 3, 5 and 7 years were plotted (Figure 3).

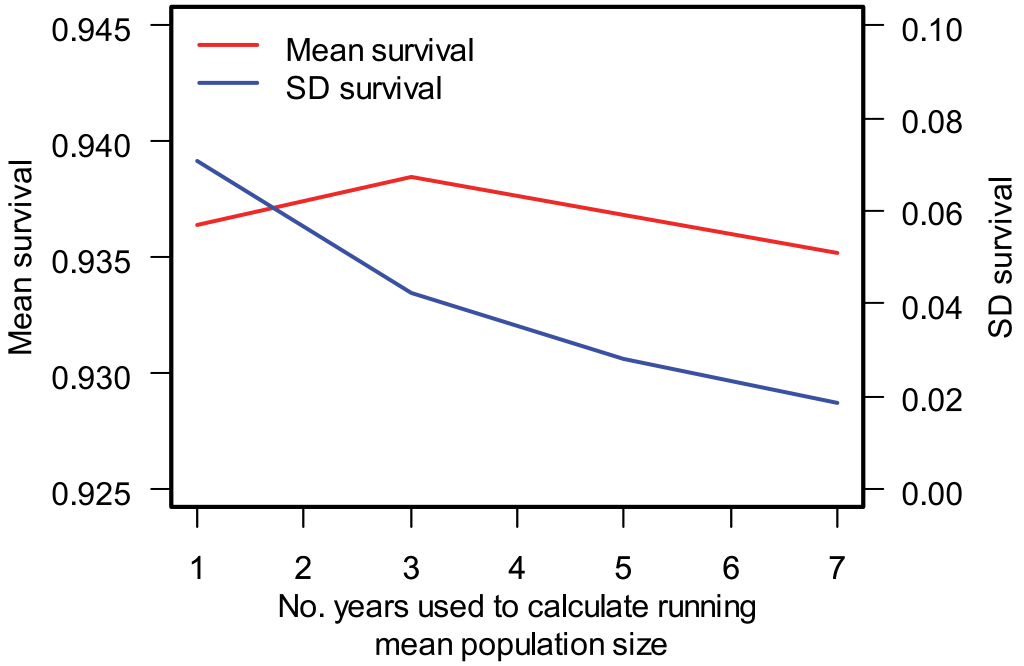

Figure 3. Mean and standard deviation (SD) of survival of Svalbard barnacle geese plotted against the number of years of population counts used to calculate the running mean.

With each incremental increase in the number of years used to calculate the running mean population size, the gain in precision became smaller (i.e. the slope of the SD line in Figure 3 became flatter), while the estimated survival rate remained more or less constant. A running mean of five years was considered to provide a reasonable balance between improved precision (i.e. minimising the SD) while minimising the number of years averaged relative to the overall period. Compared with using the annual counts the standard deviation more than halved (from 0.071 to 0.028), while mean survival remained the same.

2.3 Survival analysis

Since the 1960s, barnacle geese on the Solway have been caught and fitted with individual leg rings readable by telescope. Over 10,000 individuals had been marked in this study by 2011 which has generated over 400,000 re-sighting records. Among the analyses possible using these individual-based data is the estimation of individual-based survival rates using capture-mark-recapture (or more accurately capture – mark - re-sighting) methods (Lebreton *et al.*, 1992).

An attempt to estimate survival rates using these data was conducted as part of the previous PVA of this population (Trinder *et al.*, 2005). However, the presence of systematic biases in the data meant this attempt did not yield reliable outputs. The problems were predominantly associated with estimating re-sighting probabilities, which were found to vary in relation to an individual's recent re-sighting history. This effect is referred to as 'trap-dependence' as it resembles the behaviour of individuals in classic capture-mark-recapture studies (for example of small mammals). In such studies, individuals may either preferentially re-enter traps (trap-happy, e.g. if the trap offers food and shelter) or avoid them (trap-shy). Either behaviour leads to systematic bias in recapture probability, violating the underlying assumption of the statistical models used for data analysis (which assume random recapture probabilities).

In the case of geese, apparent trap dependence does not refer to traps *per se*, since geese are typically re-sighted using telescopes as they forage undisturbed in fields around the Solway. It is thus not clear why this effect is present in the data, although it seems likely that some form of availability bias is present, whereby certain individuals are more likely to be observed than others.

A comprehensive re-analysis of the full dataset, exploring these potential sources of bias and generating robust survival estimates was beyond the scope of the current project. However, it was considered worth undertaking a preliminary analysis on the basis of some simplifying assumptions, whilst acknowledging that the results would probably not be wholly reliable.

The first stage in survival analysis is estimating the fit of a general model to the data, from which further refinements can be made. During goodness-of-fit testing the problem shows up as a significant lack of fit of the data to the standard survival model (Cormack-Jolly-Seber; CJS), with approximately 80% of the lack of fit due to apparent trap-dependence. This effect can be accommodated through the use of a modified re-sighting structure. This separately estimates re-sighting rates for individuals observed at the previous time step and those not seen at the previous time step (in the following model descriptions this structure is termed '*td*' for trap-dependence). A general model including this term then becomes the starting point for analysis.

If, however, subsequent model selection (by AIC) identifies an interaction term between trap dependence and time (*td x time*) as being the most parsimonious re-sighting model (i.e. the model which best describes the data), then the resulting re-sighting estimates are considered to be unreliable. This effect results from propagation of the jointly unidentifiable survival and re-sighting estimates for the final period back through all the estimates. The problem can only be side-stepped by using models with constrained relationships between time and trap dependence (e.g. a constant rate at all time-steps, or an additive effect between the two terms rather than an interaction). Use of such constrained models can, however, only be considered valid if they are well supported by the data.

Nonetheless, the potential value of survival rates estimated from the re-sighting data was considered to be sufficiently high for it to be worth taking a pragmatic approach to this problem. Therefore, efforts were made firstly to reduce the expected magnitude of bias and if necessary use a constrained re-sighting model. While the survival rates estimated using this approach cannot be considered completely robust, they should provide a reasonable guide.

The first step was to attempt to reduce the magnitude of the re-sighting rate bias. A first analysis used observations:

- Collected between 1973 and 2011;
- From locations in the immediate vicinity of WWT Caerlaverock (Eastpark Farm and Newfield; on the advice of Eileen Rees and Larry Griffin); and,
- Made between November and March (inclusive).

Three alternative models for the re-sighting rate were considered:

- *td x time* (full interaction between time and trap-dependence)
- *td + time* (time varying model with an additive trap-dependent effect)
- *td* (no time variation, but retained a constant trap-dependent effect)

In each case the inclusion of *td* generates two re-sighting rates, for individuals seen in the previous year and those not seen in the previous year. Inclusion of the time term permits estimation of a different rate in each year. In all cases survival was modelled using two age classes, with independent constant rates for first year juveniles and adults (termed *a2*).

This dataset comprised 10,322 individuals. Goodness-of-fit tests were conducted using U-Care to test for preliminary model fit and to identify the most appropriate approach for modelling the data. These tests revealed the same magnitude of lack of fit as found during previous analyses (e.g. Trinder *et al.*, 2005). A moderate lack of fit for survival analysis models is considered acceptable, accommodated with the use of a correction factor ĉ (termed 'c-hat') which accounts for over-dispersion in the data. The value for ĉ is calculated from the data and thus generates a data specific correction. As a rule, if ĉ is less than 3 then analysis can proceed (Lebreton *et al.*, 1992). Without the inclusion of the trap-dependent term, ĉ was more than 10, but this was reduced to 3.38 with the *td* term included. While this remained larger than the recommended maximum of 3, here it is considered to be partly a reflection of the large dataset available and hence the greater likelihood of classical tests identifying a 'significant' result. Consequently, modelling proceeded using this ĉ correction.

The re-sighting model which received the most support (by a considerable margin) was the interaction model (*td x time*; Table 1). The next best model had a delta QAIC of 393 (models are considered equivalent with a delta QAIC score of ≤ 2), which means the interaction model was almost 400 times better than the additive one.

Table 1. Initial re-sighting model fits to data between 1973 - 2011. Note that AIC has become Quasi-AIC due to ĉ adjustment.

Re-sighting model	No. re-sighting parameters	QAICc	DeltaQAICc	Model weight	QDeviance	ĉ
time x td	76	34309	0	1	34153	3.38
time + td	39	34702	393	0	34620	3.38
td	2	35796	1487	0	35788	3.38

Examination of the re-sighting rates estimated by the *td x time* re-sighting model indicated that from around 1985 onwards the two sets of re-sighting rates (for individuals either seen or not seen in the previous year) behaved in a similar manner, falling and rising synchronously (Figure 4).

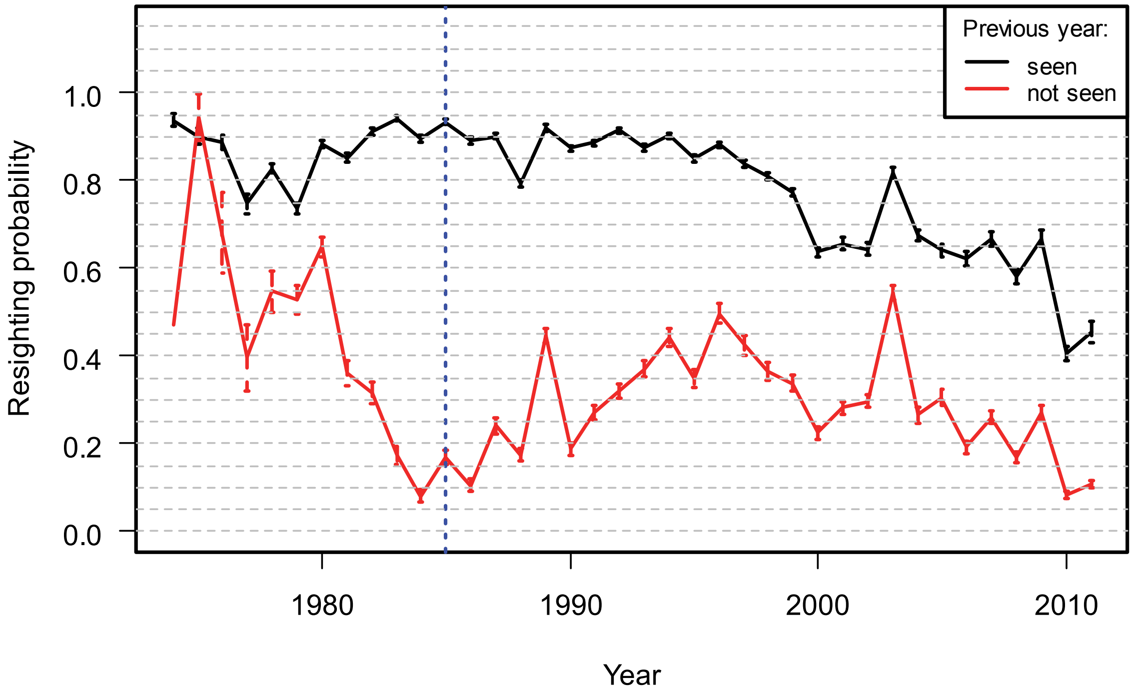

Figure 4. Re-sighting rates from model td x time*. The blue vertical dotted line at 1985 indicates the point at which the two lines were considered to vary synchronously. See text for details.*

On the basis of this observation, a second analysis was undertaken, restricted to data collected after 1985. This dataset comprised 9,017 individuals. Goodness-of-fit testing provided a revised ĉ estimate for these data of 2.84 (after inclusion of the *td* term). The same suite of re-sighting models was compared with this reduced span of years and the same order of model fits was obtained (Table 2). However, the comparative fit of the additive model was much closer (delta QAIC of 92.8), thus while the interaction model remained the best supported, the restriction in years improved the relative fit of the additive model.

Table 2. Re-sighting model fits to a reduced span of years (1985-2011). Note that AIC has become Quasi-AIC due to ĉ adjustment.

Re-sighting model	No. re-sighting parameters	QAICc	DeltaQAICc	Model weight	QDeviance	ĉ
time x td	53	32736	0	1	32628	2.84
time + td	27	32829	92.8	0	32771	2.84
td	2	33989	1253.2	0	33981	2.84

Plotting the estimated values for both the interaction and additive models revealed that the two models actually generated very similar parameter estimates (Figure 5). Thus, retention of the additive model (*td* + *time*) for subsequent survival rate modelling was considered to be a pragmatic approach, even if this was not strictly supported by the estimated relative model fits. Because this is a constrained model it avoids the problems caused by unidentifiable parameters. The additive re-sighting model (*td* + *time*) was therefore retained for subsequent refinement of the survival model.

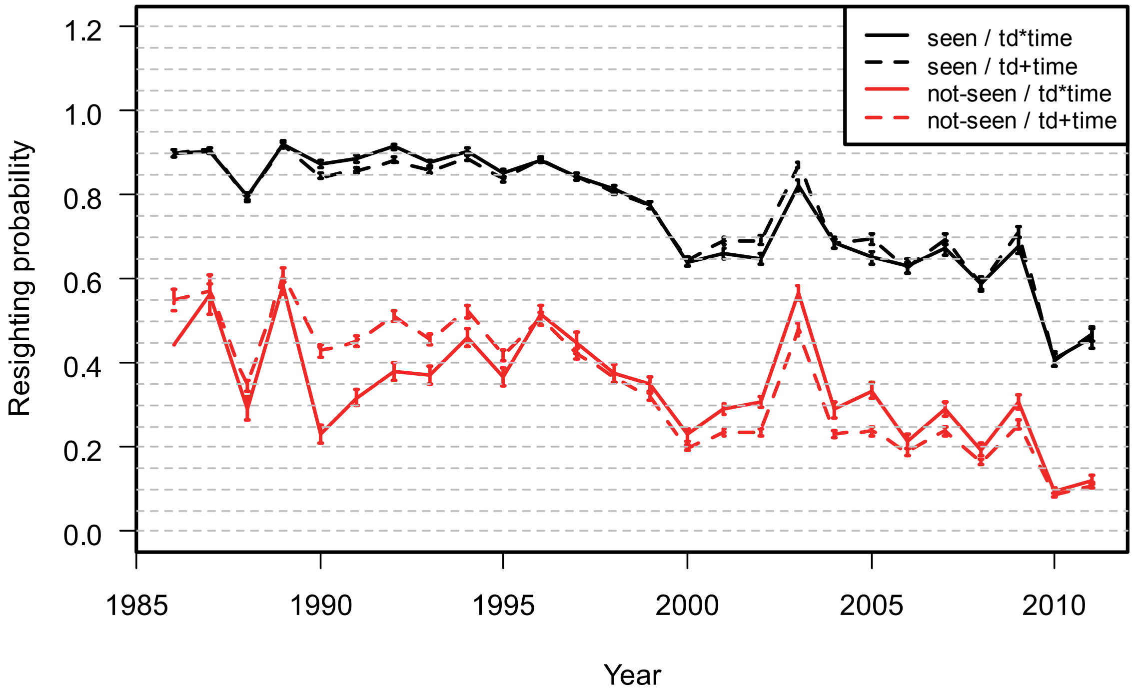

Figure 5. Comparison of re-sighting rates from models td *x* time *and* td + time*.*

Survival rates were modelled as combinations of age (juvenile and adult), sex and time using the following model structures:

- *time* (time varying survival, no age or sex effects)
- *sex x time* (time varying survival estimated independently for males and females)
- *age x time* (time varying survival estimated independently for each age class)
- *age x sex x time* (time varying survival estimated independently for all combinations of sex and age class)
- *age* (constant rate for each age class)
- *age x sex* (constant rates for each combination of sex and age class)
- *sex* (constant rate for each sex)

The best supported of these survival models included only time (Table 3), with no support for variation in survival between age classes or males and females.

Table 3. Survival model fits to a reduced span of years (1985-2011). Note that AIC has become Quasi-AIC due to ĉ adjustment.

Survival model	No. survival parameters	QAICc	Delta QAICc	Model weight	QDeviance	ĉ
time	26	32732	0.0	1.00e+00	32626	2.84
time x *sex*	52	32759	26.7	1.59e-06	32601	2.84
a2 x *time*	52	32790	57.5	3.30e-13	32631	2.84
a2 x *time* x *sex*	105	32802	70.2	0.00e+00	32540	2.84
a2	2	32829	96.3	0.00e+00	32771	2.84
a2 x *sex*	4	32831	98.4	0.00e+00	32769	2.84
sex	2	32839	106.7	0.00e+00	32781	2.84

Examination of survival rates from the time model indicated that survival remained more or less constant between 0.93 and 0.9 from 1985 to 2003, but thereafter became both more variable and was also estimated with less precision (Figure 6). However, this end of period (last 5-7 intervals) variation and decline in survival is a recognised trait from analysis of such data as a result of capture heterogeneity (Buckland, 1982). Therefore, the last few estimates are not considered to reflect a real effect.

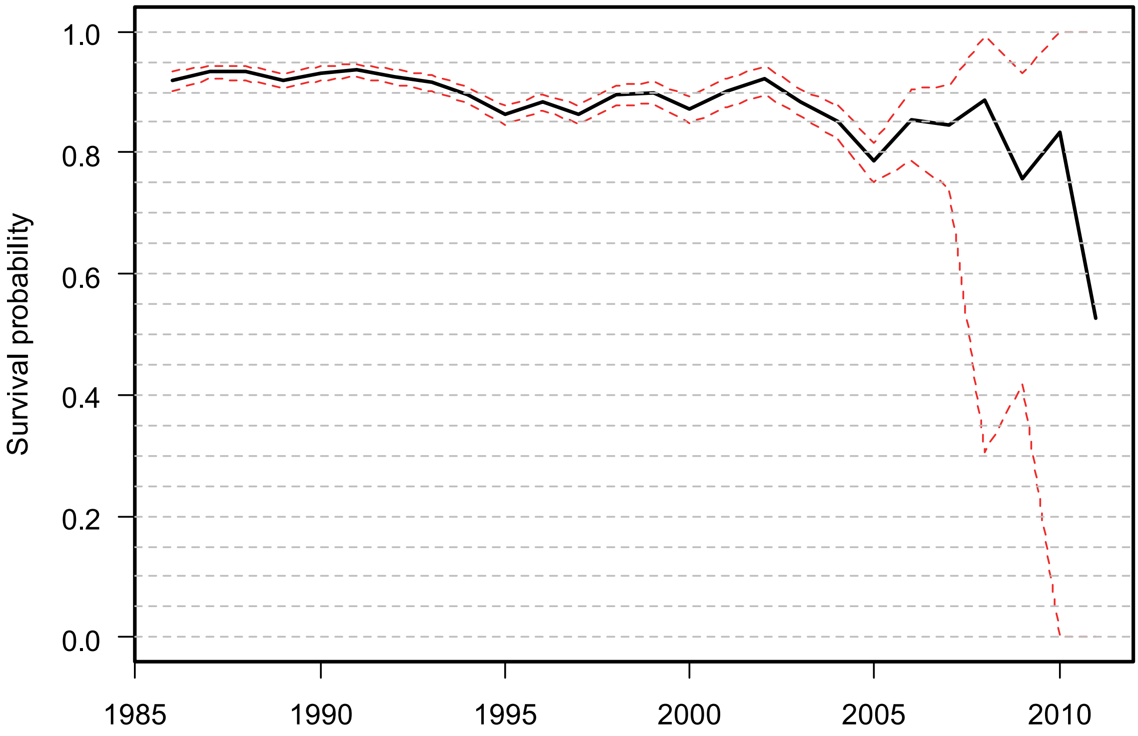

Figure 6. Svalbard barnacle goose survival (and 95% confidence range) between 1985 and 2011 from best fit survival model with only time effects.

As an indication of potential differences in survival between males and females of each age class, the constant rates estimated by the model *a2 x sex* are provided in Table 4. Note that this model was not well supported (Table 3), so these rates are provided for illustration only.

Table 4. Constant rate model estimates for each age and sex generated by survival rate model a2 x sex.

Age group	Sex	Survival rate	SE
Juvenile	Female	0.848	0.01
	Male	0.873	0.01
Adult	Female	0.90	0.002
	Male	0.90	0.002

The average survival rate between 1985 and 2003 from the best-fit time model was 0.90, with a standard deviation of 0.028. The parameter values estimated by the best-fit model are provided in Annex 2.

2.4 Comparison of count based and re-sighting based survival estimates

Due to several complications with the data analysis it was not possible to estimate all the demographic parameters needed for the population model in a completely robust manner. As discussed above, count-based survival rates are based on an assumption that there are no age influences on survival (after birds are six months old), and that sampling errors in the population counts 'even out' over time. Furthermore, even if these assumptions are reasonable, survival calculated from counts and age ratios is very likely to overestimate inter-annual variations. There is therefore a need to consider how to account for this variance inflation.

Count-based survival estimates are also dependent on estimated annual productivity. While these estimates are derived from observations of large sample flocks in autumn and winter, and thus should be representative of the entire population, it is difficult to test this assumption.

Survival rates derived from re-sightings of marked individuals are also subject to potential bias, relating to the representativeness of the marked population. If particular subsets of the population have a greater likelihood of being caught and fitted with leg rings, or of subsequent observation, then estimated survival rates will reflect those population subsets. As detailed above, there is unexplained variation among the marked birds with regards to their individual likelihood of being re-sighted in a year following a positive re-sighting. The presence of such bias may indicate some form of population structure in the wintering flock which is not currently understood, and it is therefore difficult to account for this.

Consequently, in an effort to remove some of the potential sources of heterogeneity, survival analysis was conducted on a reduced dataset. This was only partially successful at removing the re-sighting structure and the resulting survival rates remain subject to uncertainty regarding their robustness.

The survival and reproductive rates calculated using the various datasets are listed in Table 5.

Table 5. Demographic parameters estimated for the Svalbard barnacle goose population.

Demographic parameter (1985-2011)	Average	Standard deviation
Survival (re-sighting based)	0.903	0.028
Survival (annual counts)	0.936	0.071
Survival (estimated using 5 year running mean of counts)	0.937	0.028
Mean brood size	1.912	0.296
Proportion of breeding adults	0.145	0.075

It was not possible to determine which survival rate was more appropriate as the basis for predicting future growth, so both the re-sighting derived (0.903) and the count-based (0.936) rates were used in preliminary runs of the population model. By running the model from the 1985 count, it was possible to compare model outputs generated using the two survival rates against the actual counts. In this manner the population model could be used to guide the decision on which survival rate was more appropriate for generating predictions. In both cases the standard deviation on survival was 0.028, as derived from both the re-sighting analysis and the five-year running mean survival rate. The reproductive rates were kept the same throughout (Table 5).

2.5 Population model

A stochastic, density-independent population model was developed, based on the demographic parameters in Table 5. Initial model runs used both survival rates (re-sighting and count-based) with the standard deviation estimated from the five-year running mean and re-sighting analysis (0.028). The same survival rate was used for all age classes; although the re-sighting analysis indicated that juveniles may have lower survival than adults, the survival models including age were not well-supported by the data. This combined age survival rate would be expected to over-estimate survival of birds in their first year as measured from fledging onwards, since younger birds would be expected to experience higher mortality in late summer than adult birds. However, since the mean brood size value (which is a component of the reproductive rate) is calculated from observations made on the wintering grounds, this includes any mortality which occurs between fledging and winter (i.e. a period of approximately four months). Therefore, the lower survival of young birds is incorporated into the reproductive rate. From their first winter, juveniles are then expected to have a similar survival rate to older birds. Reproduction was modelled using the mean and standard deviations for the proportion of adults which breed each year (0.145, SD= 0.075) and the mean and standard deviation for the mean brood size (1.912, SD= 0.296).

Initial model runs were used to identify which survival rate generated back-dated predictions closest to the observed population trend. Once this determination was made, subsequent model simulations used the survival rate thus identified.

The model has three age classes and simulates an annual time step, with individuals moving from the 0-1 year (juvenile) group to the 1-2 (immature) year group and finally to the 2+ (adult) age group, with a probability defined by the survival rate. Only the adult age group breeds, with the number of juveniles being a product of survival (only surviving birds breed), the proportion of breeding birds and the mean brood size. Both sexes are modelled together, thus in the above calculation the mean brood size is halved prior to calculation. Survival rates and the proportion of breeders were modelled using beta distributions (survival for each age class was modelled independently, although the mean and variance were the same) while the mean brood size was modelled using a stretched beta distribution

(Morris & Doak, 2002). A stretched beta distribution is simply a rescaled beta distribution with a user-defined maximum and minimum.

As illustrated in the analysis of demographic parameters (Section 2.1 and Annex 2) there is no evidence that the population has been regulated by density-dependent feedback during the last 25 years. Although density-dependent regulation may be expected to exert an influence on population growth in the future, without knowing how this might operate it is not possible to produce reliable density-dependent population predictions. On several previous occasions the population has been predicted to be approaching its carrying capacity (e.g. Owen & Norderhaug, 1977; Owen, 1984; Rowcliffe *et al.*, 1995), but each time these predictions have proved incorrect. This illustrates the challenges involved in robustly identifying density-dependent responses in this population. Consequently it is considered more appropriate to use a density-independent model.

The effect of annual shooting on population growth was simulated as a harvest rate (i.e. a percentage of the population is removed each year). The rate was fixed for the duration of a simulation, with the number removed in year *t* calculated as a percentage of the population in year *t*-1, thereby introducing a small lag. This was considered to match how shooting could be used to limit population growth. There was no stochastic variation around the percentage used, thus the number shot was in exact proportion with the population size in the previous year.

The starting year for simulations was 2011, the year of the most recent available count data. Three alternative initial population sizes were considered for the modelling; the most recent count (33,900), the 10-year average count (29,140) and the fitted value for 2011 obtained from a linear model of the most recent 10 counts against year (33,951). Preliminary model runs were used to estimate how sensitive the model predictions were to the choice of initial population size. A comparison of the outputs obtained using the largest and smallest count values found that the probability of a 10% population decline was around 0.4% higher when the lowest starting size was used compared with the highest. Given the similarity between the fitted value from the linear model and the 2011 count, the 2011 count was considered to be the most suitable for predictive purposes.

2.6 Model validation

Using the re-sighting based survival rate of 0.903, the model produced a poor fit to the count data since 1985 (Figure 7). The model generated a mean population growth rate of 1.013 compared with the observed rate across the period 1985 to 2011 of 1.046.

Using the count-based survival rate of 0.936, the model produced a close fit to the count data since 1985 (Figure 8). The model generated a mean population growth rate of 1.049 which was close to the mean observed rate between 1985 and 2011 of 1.046.

The count-based survival rate was expected to perform well when compared with the counts (since they are derived from the same data), and on the basis of the above comparison it was considered more likely to generate reliable predictions than the re-sighting based survival rate.

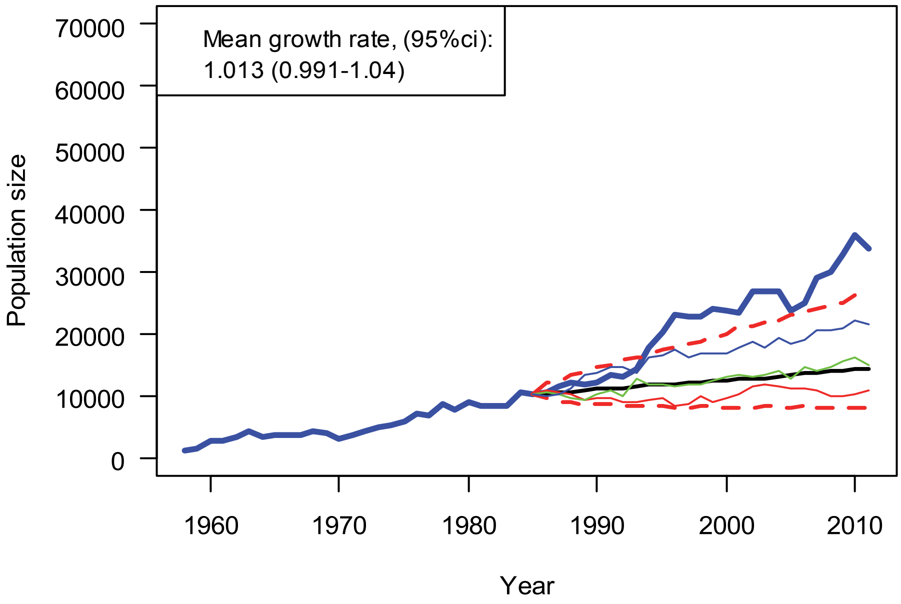

Figure 7. Svalbard barnacle goose population model outputs derived using the re-sighting based survival estimate: comparison between median model prediction (black line) and 95% confidence range (red dashed lines) and the observed population trend (heavy blue line). Randomly selected individual trajectories (thin red, green and blue lines) are included for illustration.

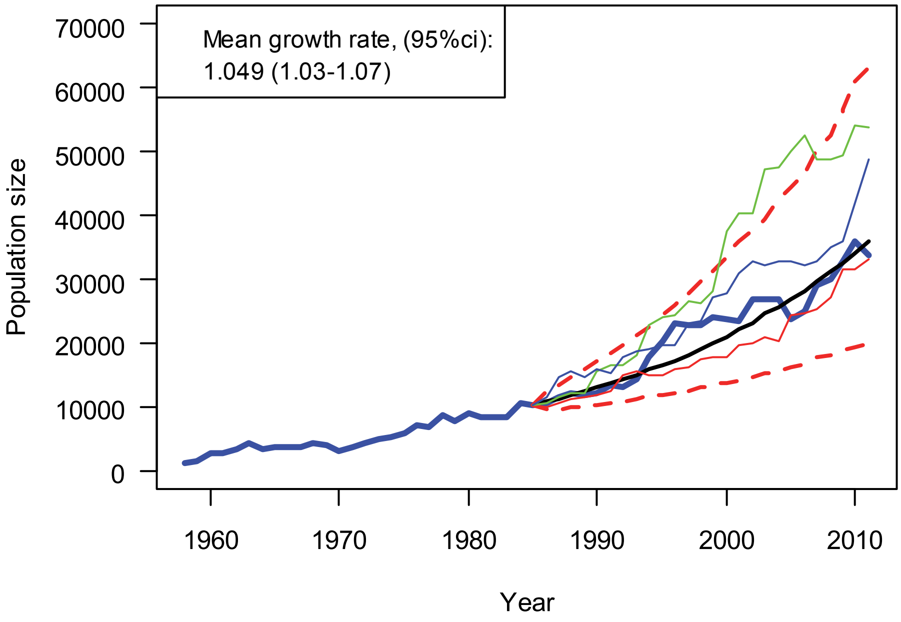

Figure 8. Svalbard barnacle goose population model outputs derived using the count-based survival estimate: comparison between median model prediction (black line) and 95% confidence range (red dashed lines) and the observed population trend (heavy blue line). Randomly selected individual trajectories (thin red, green and blue lines) are included for illustration.

3. SIMULATION RESULTS

Projecting the population forward from the 2011 count, the model predicts an annual population growth rate of 4.9% (Figure 9, top-left). When additional mortality of between 0.5% and 7.5% of the population was applied each year (as may occur through shooting such proportions of the population), the average growth rate declined by approximately the same magnitude as the percentage shot (i.e. shooting 1% of the population led to a 1% reduction in the growth rate; Figure 9; Annex 3, Table A3.1).

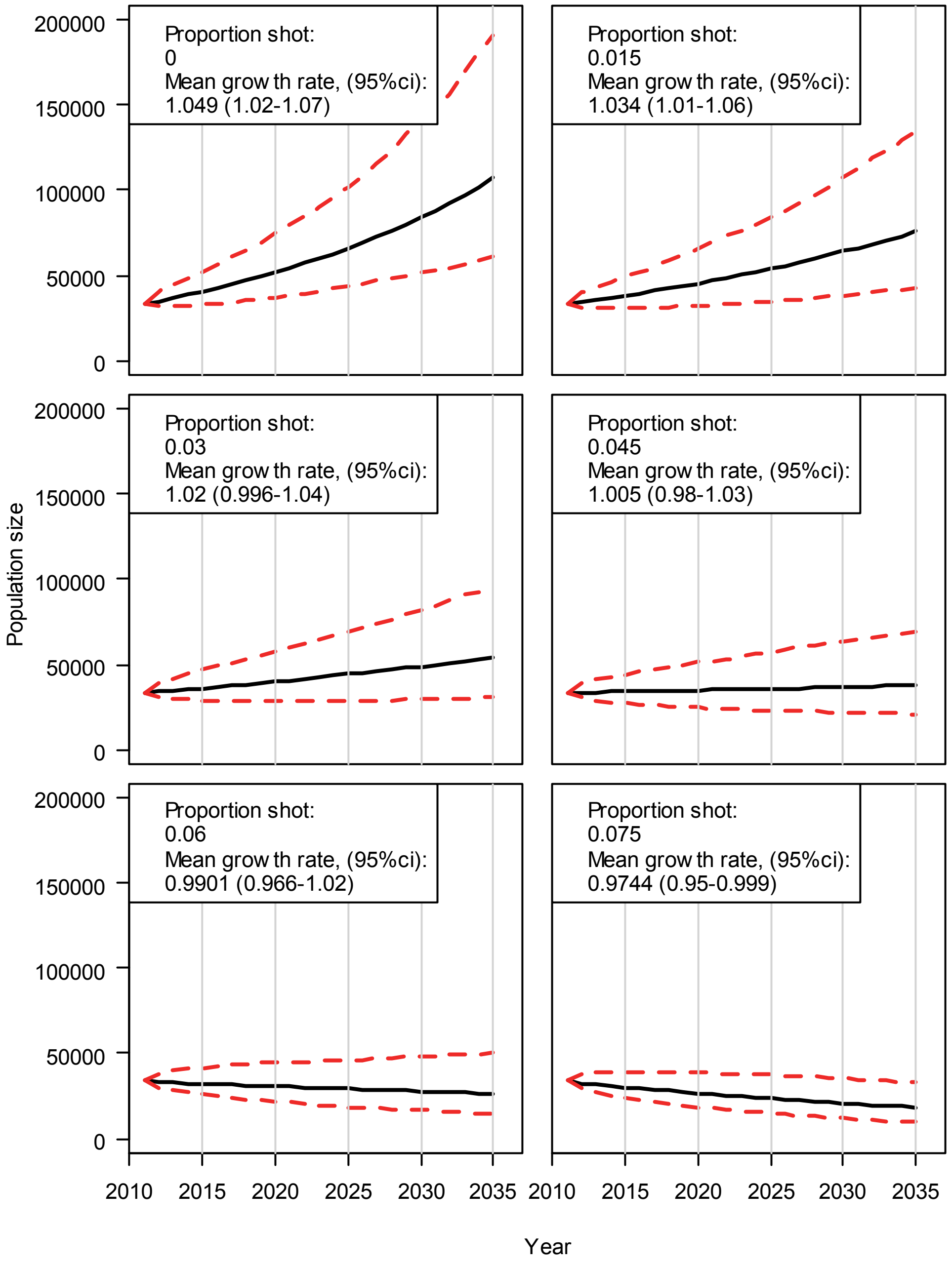

Figure 9. Simulation of the predicted changes in the Svalbard barnacle goose population over 25 years with increasing rates of shooting (proportion shot range: 0 – 0.075).

The change in population growth against the total proportion shot can be seen in Figure 10. Population growth remained positive on average until the number removed exceeded 0.05, while the lower 95% confidence interval remained positive until the total proportion shot exceeded 0.025.

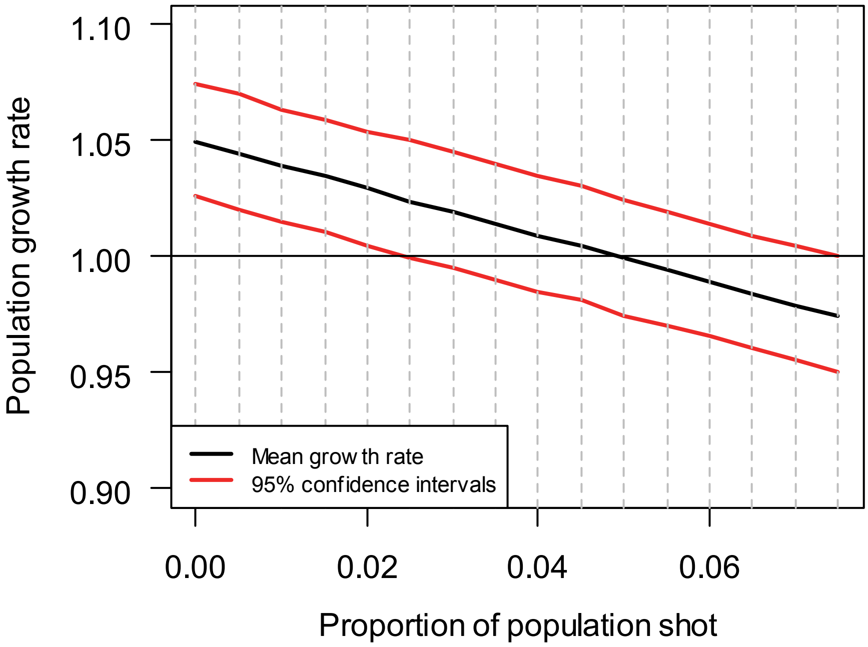

Figure 10. Population growth rate of Svalbard barnacle geese predicted with increasing proportion shot.

The probability of any population decline after 10 years in response to shooting increased from 0.4% with no shooting, to 52% with 5% of the population shot each year and 89% with 7.5% of the population shot each year (Figure 11, red line). The equivalent risks of a population reduction below 80% of the initial size increased from <0.01% to 11% when 5% were shot, and 53% when 7.5% were shot (Figure 11, purple line). Tabulated model results are provided in Appendix 3.

The probability of any population decline after 25 years was initially low (<0.01%) with no shooting, and increased to 53% with 5% of the population shot and 97% when 7.5% of the population was shot (Figure 12, red line). Similarly, the equivalent risk of a population reduction below 80% of the initial size increased from <0.01%, to 25% when 5% were shot and 90% when 7.5% were shot (Figure 12, purple line).

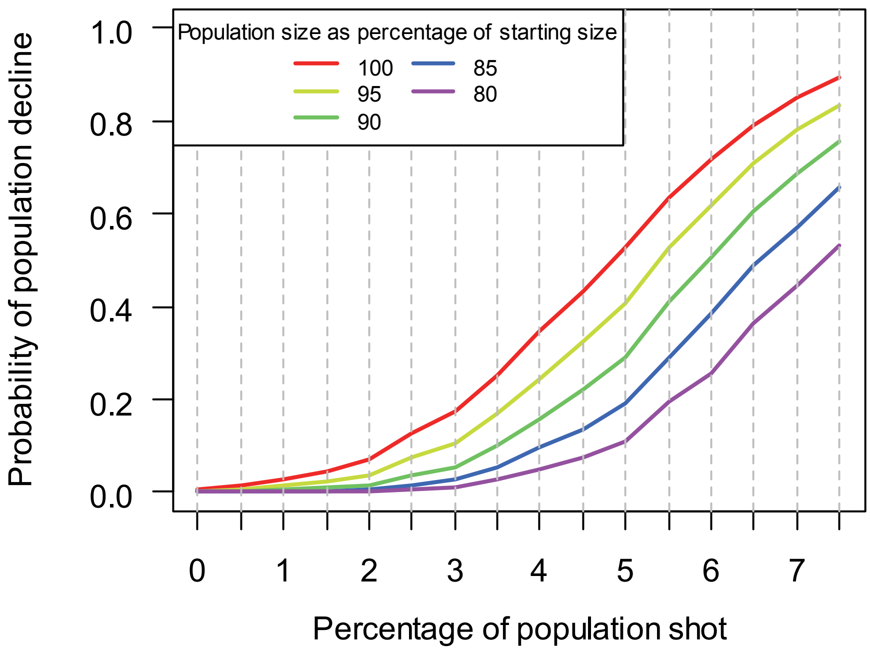

Figure 11. Probability of the Svalbard barnacle goose population declining below percentage thresholds of the initial size after 10 years, with increasing percentages of the population shot. Figures given in Annex 3, Table A3.2.

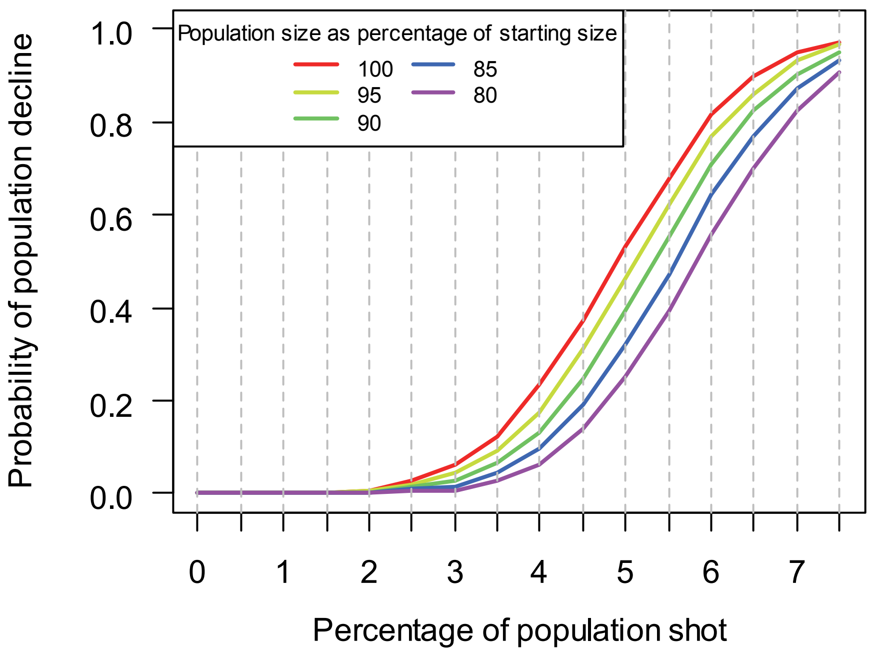

Figure 12. Probability of the Svalbard barnacle goose population declining below percentage thresholds of the initial size after 25 years, with increasing percentages of the population shot. Figures given in Annex 3, Table A3.3.

4. DISCUSSION

The Svalbard barnacle goose population has continued to grow since the previous PVA for this population was conducted (Trinder *et al.*, 2005). The count for the last year used in the previous assessment (2003/4) was 27,000, while the equivalent for the current study was 33,900 (2011/12). The average growth rate over this recent period has been approximately 3%, which is lower than the 4.9% estimated since 1985. This may indicate the start of density-dependent population regulation. However, analysis of the count and reproductive data found no evidence for such population regulation during the last 27 years. This is similar to results obtained for a number of other Arctic-breeding goose populations (e.g. Morrissette *et al.*, 2010), although detecting density-dependent regulation is challenging in slow-breeding species such as geese. Nevertheless, there is currently little evidence to suggest that the goose population has been regulated to date by competition for resources. This makes it very difficult to predict robustly how density-dependent regulation could affect this population, and consequently the population has been modelled as density-independent. At some point it is reasonable to suppose the population will attain a size at which resource competition will limit further growth, and it is possible this may occur within the 25 year projections presented in this analysis. Indeed, within such a time frame it is very likely that other environmental factors may begin to exert an influence on the population. Consequently, the 10 year predictions should be considered as more reliable than the 25 year ones, particularly in relation to management considerations. The longer-term predictions are provided as an indication of potential growth rather than robust evidence on which to base management decisions.

As well as modelling the population as density-independent, another assumption made relates to survival. We have limited information on age-related survival, thus the adult rate has been used for all age classes. While this is a simplification, it is unlikely to have an important effect on the model results since the model structure is such that juveniles are actually several months old and have survived their first autumn migration before they are included. Their survival rate is probably quite similar to older age classes from this point.

The results of the additional mortality simulations suggest that the average population growth rate, which in the baseline form is predicted to be around 5% per year, would be reduced in equivalent percentage terms as the proportion shot each year. Thus if 1% of the population is shot (equivalent to approximately 340 individuals in 2011), a 1% reduction in the growth rate would be predicted. Any such population management in Scotland would occur under licence and therefore could be rapidly adapted (in terms of harvest levels) to reflect changes in the population size. The impact of shooting on the population would also be affected if the population does attain an equilibrium size (i.e. density-dependence limits growth). It is obviously difficult to predict how the conclusions of this analysis would be modified under such circumstances. However, harvesting a population which is already fluctuating around an equilibrium size would be expected to reduce competition within the remaining population rather than further reducing the equilibrium population size itself.

Survival analysis

The Svalbard barnacle goose dataset represents an invaluable resource which could be used to generate robust estimates of survival for the last four decades. However, there appear to be aspects of these data which prevent straightforward analysis. Preliminary analysis of the Svalbard barnacle goose re-sighting data confirmed the presence of a systematic bias in the individual re-sighting rates which complicates the selection of a robust model for estimating re-sighting probabilities. This effect replicates the phenomenon referred to as 'trap-dependence' (when marked individuals become habituated to, and take advantage of, live traps). Over 80% of the lack of fit of the standard CJS model to the data was due to this effect. An alternative model structure can be used to account for such behaviour, by estimating re-sighting probabilities separately for individuals seen in the previous year from those not seen in the previous year. Any remaining lack of fit can then be accommodated using an over-dispersion correction factor. However, this approach is predicated on the assumption that the re-sighting rate model cannot include an interaction term between the trap-dependence structure and time, as this results in unidentifiable parameter estimates.

In the current analysis the best-fit re-sighting model did include this interaction, with the consequence that the survival estimates from this model cannot be considered robust. The recommended solution is to undertake further investigations to identify the source of the trap-dependence effect and take steps to accommodate this through modified model structure or data filtering. It was not possible within the current project to undertake such detailed analysis, and thus an alternative approach was adopted, whereby the second best fitting re-sighting model (which had an additive relationship between trap dependence and time rather than an interaction) was used for further analysis. Because re-sighting rates are constrained in this model structure, it does not generate unidentifiable parameters and hence survival rates estimated using this model are more robust. However, these have necessarily been derived from a model which is not the best-supported one. It is difficult to determine what the implications of this are, and how reliable the resulting survival estimates are. The similarity between the re-sighting rates obtained from the interaction and additive models suggests that the differences in survival are likely to be small, but the unidentifiable nature of the outputs of the interaction model mean that this conclusion should be treated with caution.

The solutions to the problems identified for the analysis of these data are likely to include efforts to understand how the data have been collected and how goose use of the Solway has changed over the course of the study, in order to derive suitable modelling approaches.

Population management

Shooting at a harvest rate represents a simple means to minimise the risks of either under- or over-exploiting a population. Given the close monitoring of this population, setting harvest limits on the basis of removing a percentage of the population could be readily achieved and hence this approach would have a low risk of inadvertently removing a larger number than intended.

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ANNEX 1: SVALBARD BARNACLE GOOSE POPULATION DATA AND ANALYSIS

Table A1.1. Svalbard barnacle goose counts and demographic data. Details of methods for counts, age ratios and mean brood sizes from http://monitoring.wwt.org.uk/our-work/goose-swan-monitoring-programme/species-accounts/svalbard-barnacle-goose/

Year	Total count	Proportion juveniles	Mean brood size	Proportion breeding adults	Survival
1958	1350	0.489	2.70	NA	NA
1959	1650	0.145	2.60	0.256	1.044
1960	2800	0.386	3.00	0.490	1.042
1961	2800	0.189	2.50	0.304	0.811
1962	3400	0.050	2.00	0.065	1.154
1963	4250	0.428	2.80	0.563	0.715
1964	3400	0.097	1.90	0.198	0.722
1965	3700	0.157	2.00	0.206	0.918
1966	3700	0.119	2.40	0.133	0.881
1967	3700	0.273	2.10	0.406	0.727
1968	4200	0.231	2.00	0.413	0.873
1969	4000	0.270	2.60	0.370	0.695
1970	3200	0.472	3.00	0.816	0.423
1971	3700	0.149	2.30	0.287	0.984
1972	4400	0.259	2.00	0.411	0.881
1973	5100	0.210	1.60	0.448	0.916
1974	5200	0.150	1.80	0.248	0.867
1975	6050	0.207	2.27	0.270	0.923
1976	7200	0.281	2.59	0.380	0.856
1977	6800	0.025	2.00	0.036	0.921
1978	8800	0.261	2.30	0.316	0.956
1979	7700	0.038	2.50	0.042	0.842
1980	9050	0.236	2.60	0.247	0.898
1981	8300	0.023	2.30	0.027	0.896
1982	8500	0.135	2.30	0.139	0.886
1983	8400	0.132	2.20	0.160	0.858
1984	10500	0.262	2.38	0.344	0.923
1985	10400	0.096	1.96	0.147	0.895
1986	10500	0.119	2.05	0.146	0.889
1987	11400	0.151	1.70	0.237	0.922
1988	12100	0.121	2.07	0.157	0.933
1989	11700	0.080	1.71	0.116	0.889
1990	12100	0.120	2.09	0.142	0.910

1991	13300	0.140	2.12	0.174	0.945
1992	13200	0.051	1.82	0.069	0.942
1993	14350	0.118	1.80	0.157	0.959
1994	17900	0.097	1.56	0.156	1.126
1995	20400	0.209	2.05	0.285	0.901
1996	23000	0.161	1.74	0.279	0.946
1997	22900	0.168	1.76	0.273	0.828
1998	22800	0.156	1.69	0.263	0.840
1999	24100	0.109	1.62	0.179	0.942
2000	23800	0.030	1.37	0.051	0.958
2001	23550	0.031	1.58	0.042	0.959
2002	27000	0.104	1.96	0.122	1.027
2003	27000	0.041	2.00	0.048	0.959
2004	26900	0.021	1.46	0.031	0.975
2005	23900	0.079	2.50	0.070	0.818
2006	25000	0.146	2.18	0.170	0.893
2007	29000	0.128	2.42	0.142	1.012
2008	29900	0.087	1.99	0.110	0.941
2009	32900	0.051	1.82	0.065	1.044
2010	35900	0.108	2.54	0.100	0.973
2011	33900	0.139	2.06	0.176	0.813

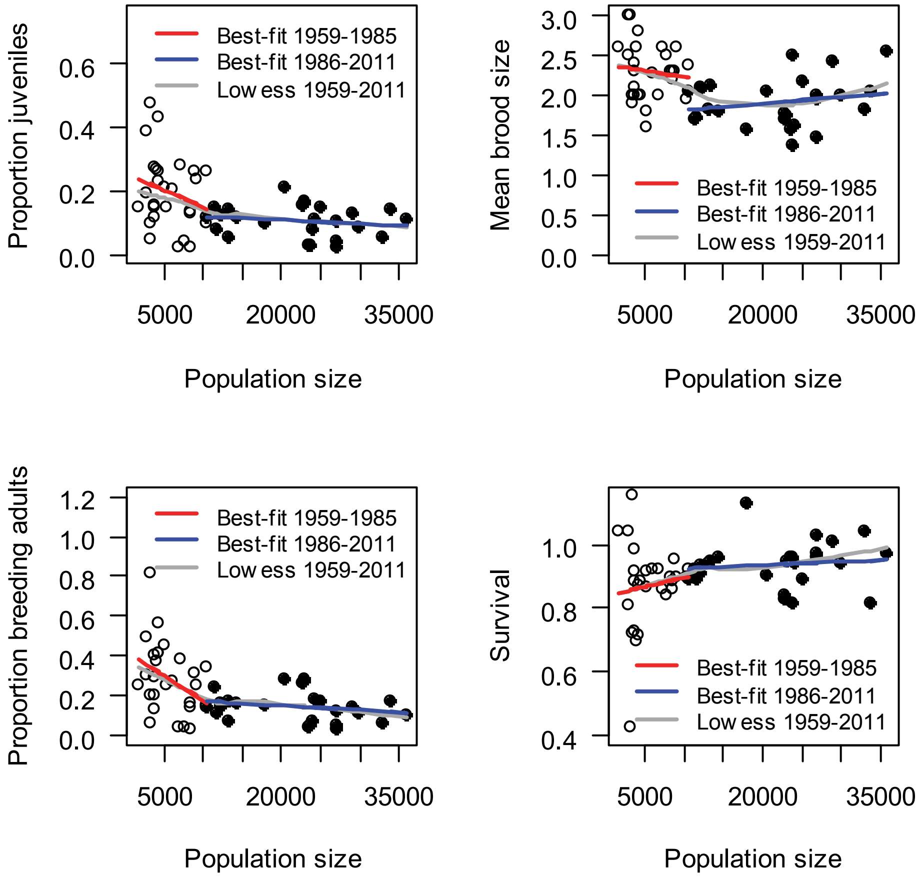

Figure A1. Trend analysis of Svalbard barnacle goose population, reproduction and survival rates since 1959 in relation to population size. Open symbols are used for 1959 to 1985, filled symbols for 1986 to 2011. Solid lines are from linear model fits obtained for parameter vs. population size (red: 1959-1985; blue: 1986-2011) and grey lines are for locally smoothed regressions (Lowess) to illustrate the suitability of the linear model fits. See Table A1.2 for model details.

Table A1.2. Outputs of a Generalised Linear Model of Svalbard barnacle goose reproduction and survival. Significant trends (at p<0.05) are highlighted.

Explanatory variable	Dependent variable	Years	Deviance	df		t	*p*	Slope	S.E.
Year	Prop. Young	1959-1985	11,361	1	25	-1.62	0.12	-0.03	0.02
	Mean brood size	1959-1985	133	1	25	-1.01	0.32	-0.004	0.004
	Prop. Breeding	1959-1985	11,779	1	25	-2.02	0.05	-0.04	0.02
	Survival	1959-1985	0.865	1	25	0.20	0.84	<0.001	0.005
Population	Prop. Young	1959-1985	11,749	1	25	-1.30	0.20	-0.4	0.30
	Mean brood size	1959-1985	134	1	25	-1.00	0.33	-0.06	0.06
	Prop. Breeding	1959-1985	11,963	1	25	-1.89	0.07	-0.69	0.37
	Survival	1959-1985	0.855	1	25	0.58	0.57	0.04	0.08
Year	Prop. Young	1986-2011	14,899	1	24	-1.18	0.25	-0.02	0.01
	Mean brood size	1986-2011	109	1	24	1.45	0.16	0.006	0.004
	Prop. Breeding	1986-2011	19,439	1	24	-1.94	0.06	-0.03	0.02
	Survival	1986-2011	0.141	1	24	0.31	0.76	<0.001	0.002
Population	Prop. Young	1986-2011	15,291	1	24	-0.80	0.40	-0.30	0.30
	Mean brood size	1986-2011	116	1	24	0.72	0.48	0.06	0.08
	Prop. Breeding	1986-2011	21,080	1	24	-1.20	0.20	-0.50	0.40
	Survival	1986-2011	0.14	1	24	0.53	0.60	0.02	0.04

Generalised Linear Models were used with the following error structures:

- Prop. Young – quasibinomial (response variable was number of juveniles : number of adults & sub-adults)
- Mean brood size – quasipoisson (response variable was mean brood size multiplied by 100 to transform to an integer)
- Prop. Breeding – quasibinomial (response variable was number of breeding adults : number of non-breeding adults)
- Survival – Gaussian (response was log transformed)

In all cases model residuals were tested for normality using the Shapiro-Wilks test. All tests returned non-significant results indicating that selected models were appropriate.

ANNEX 2: INDIVIDUAL BASED PARAMETER ESTIMATES

Demographic rates for the Svalbard barnacle goose marked population derived from the best-fit CJS model [Phi(~time)p(~time + TD)]. For details see section 2.3 of the text.

Parameter		Period from	Period to	Estimate (S.E.)	Confidence interval (95%) Lower - Upper
Phi		1985	1986	0.918 (0.008)	0.901 - 0.933
Phi		1986	1987	0.935 (0.005)	0.923 - 0.944
Phi		1987	1988	0.933 (0.006)	0.919 - 0.944
Phi		1988	1989	0.92 (0.007)	0.906 - 0.932
Phi		1989	1990	0.931 (0.006)	0.918 - 0.942
Phi		1990	1991	0.936 (0.006)	0.924 - 0.947
Phi		1991	1992	0.926 (0.006)	0.913 - 0.937
Phi		1992	1993	0.917 (0.007)	0.903 - 0.929
Phi		1993	1994	0.896 (0.007)	0.882 - 0.909
Phi		1994	1995	0.862 (0.008)	0.846 - 0.877
Phi		1995	1996	0.884 (0.007)	0.87 - 0.897
Phi		1996	1997	0.864 (0.008)	0.848 - 0.878
Phi		1997	1998	0.894 (0.008)	0.878 - 0.909
Phi		1998	1999	0.899 (0.009)	0.88 - 0.915
Phi		1999	2000	0.872 (0.011)	0.849 - 0.892
Phi		2000	2001	0.902 (0.012)	0.876 - 0.923
Phi		2001	2002	0.922 (0.012)	0.894 - 0.943
Phi		2002	2003	0.884 (0.011)	0.86 - 0.904
Phi		2003	2004	0.852 (0.015)	0.821 - 0.878
Phi		2004	2005	0.786 (0.017)	0.751 - 0.817
Phi		2005	2006	0.854 (0.03)	0.786 - 0.903
Phi		2006	2007	0.844 (0.043)	0.741 - 0.912
Phi		2007	2008	0.886 (0.148)	0.305 - 0.993
Phi		2008	2009	0.758 (0.138)	0.418 - 0.932
Phi		2009	2010	0.835 (1.322)	0 - 1
Phi		2010	2011	0.527 (0.795)	0.002 - 0.998
p	seen t-1	-	1986	0.9 (0.009)	0.88 - 0.916
p	seen t-1	-	1987	0.906 (0.006)	0.895 - 0.917
p	seen t-1	-	1988	0.794 (0.008)	0.778 - 0.809
p	seen t-1	-	1989	0.919 (0.005)	0.909 - 0.928
p	seen t-1	-	1990	0.846 (0.007)	0.832 - 0.859
p	seen t-1	-	1991	0.857 (0.006)	0.845 - 0.869
p	seen t-1	-	1992	0.884 (0.006)	0.873 - 0.894
p	seen t-1	-	1993	0.859 (0.006)	0.846 - 0.871

Parameter		Period		Estimate (S.E.)	Confidence interval (95%)
		from	to		Lower - Upper
p	seen t-1	-	1994	0.889 (0.005)	0.878 - 0.899
p	seen t-1	-	1995	0.839 (0.007)	0.825 - 0.853
p	seen t-1	-	1996	0.882 (0.006)	0.871 - 0.893
p	seen t-1	-	1997	0.843 (0.007)	0.829 - 0.856
p	seen t-1	-	1998	0.807 (0.008)	0.792 - 0.822
p	seen t-1	-	1999	0.775 (0.008)	0.759 - 0.79
p	seen t-1	-	2000	0.643 (0.01)	0.623 - 0.662
p	seen t-1	-	2001	0.691 (0.01)	0.671 - 0.71
p	seen t-1	-	2002	0.693 (0.01)	0.673 - 0.713
p	seen t-1	-	2003	0.871 (0.006)	0.858 - 0.883
p	seen t-1	-	2004	0.686 (0.011)	0.664 - 0.707
p	seen t-1	-	2005	0.696 (0.011)	0.673 - 0.718
p	seen t-1	-	2006	0.627 (0.013)	0.602 - 0.651
p	seen t-1	-	2007	0.696 (0.012)	0.672 - 0.719
p	seen t-1	-	2008	0.59 (0.013)	0.565 - 0.615
p	seen t-1	-	2009	0.713 (0.012)	0.689 - 0.736
p	seen t-1	-	2010	0.407 (0.014)	0.38 - 0.435
p	seen t-1	-	2011	0.471 (0.017)	0.438 - 0.504
p	not seen t-1	-	1986	0.552 (0.025)	0.502 - 0.601
p	not seen t-1	-	1987	0.571 (0.017)	0.537 - 0.605
p	not seen t-1	-	1988	0.346 (0.012)	0.323 - 0.37
p	not seen t-1	-	1989	0.61 (0.015)	0.58 - 0.639
p	not seen t-1	-	1990	0.43 (0.014)	0.403 - 0.457
p	not seen t-1	-	1991	0.453 (0.013)	0.427 - 0.479
p	not seen t-1	-	1992	0.511 (0.014)	0.484 - 0.539
p	not seen t-1	-	1993	0.456 (0.013)	0.429 - 0.482
p	not seen t-1	-	1994	0.524 (0.014)	0.496 - 0.551
p	not seen t-1	-	1995	0.418 (0.014)	0.392 - 0.445
p	not seen t-1	-	1996	0.507 (0.014)	0.479 - 0.535
p	not seen t-1	-	1997	0.424 (0.014)	0.398 - 0.451
p	not seen t-1	-	1998	0.366 (0.012)	0.342 - 0.39
p	not seen t-1	-	1999	0.321 (0.011)	0.3 - 0.343
p	not seen t-1	-	2000	0.199 (0.008)	0.184 - 0.214
p	not seen t-1	-	2001	0.235 (0.009)	0.219 - 0.253
p	not seen t-1	-	2002	0.237 (0.009)	0.22 - 0.255
p	not seen t-1	-	2003	0.482 (0.014)	0.454 - 0.509
p	not seen t-1	-	2004	0.231 (0.01)	0.212 - 0.251

Parameter		Period		Estimate (S.E.)	Confidence interval (95%)
		from	to		Lower - Upper
p	not seen t-1	-	2005	0.24 (0.011)	0.219 - 0.261
p	not seen t-1	-	2006	0.188 (0.009)	0.171 - 0.206
p	not seen t-1	-	2007	0.239 (0.011)	0.219 - 0.261
p	not seen t-1	-	2008	0.165 (0.008)	0.15 - 0.181
p	not seen t-1	-	2009	0.255 (0.011)	0.233 - 0.278
p	not seen t-1	-	2010	0.086 (0.005)	0.077 - 0.097
p	not seen t-1	-	2011	0.109 (0.007)	0.097 - 0.123

ANNEX 3: TABULATED MODEL PREDICTIONS

Table A3.1. Population growth rates.

Proportion shot	Mean population growth rate	95% confidence intervals	
		Lower	Upper
0	1.0496	1.0260	1.0749
0.005	1.0445	1.0201	1.0703
0.01	1.0391	1.0150	1.0635
0.015	1.0347	1.0107	1.0591
0.02	1.0293	1.0049	1.0540
0.025	1.0240	0.9994	1.0505
0.03	1.0197	0.9952	1.0455
0.035	1.0144	0.9904	1.0399
0.04	1.0093	0.9851	1.0346
0.045	1.0043	0.9810	1.0302
0.05	0.9992	0.9747	1.0243
0.055	0.9942	0.9703	1.0193
0.06	0.9891	0.9654	1.0143
0.065	0.9842	0.9604	1.0091
0.07	0.9791	0.9552	1.0044
0.075	0.9745	0.9505	1.0004

Table A3.2. Probability of decline below population thresholds within 10 years.

Proportion shot	Population reduction thresholds				
	0	0.05	0.1	0.15	0.2
0	0.0042	0.0024	0.0010	0.0002	0.0000
0.005	0.0150	0.0058	0.0022	0.0006	0.0002
0.01	0.0266	0.0148	0.0048	0.0014	0.0006
0.015	0.0452	0.0236	0.0104	0.0028	0.0010
0.02	0.0726	0.0360	0.0158	0.0054	0.0022
0.025	0.1284	0.0734	0.0360	0.0154	0.0074
0.03	0.1724	0.1042	0.0556	0.0284	0.0104
0.035	0.2508	0.1682	0.0992	0.0546	0.0258
0.04	0.3454	0.2416	0.1582	0.0950	0.0488
0.045	0.4346	0.3254	0.2234	0.1372	0.0730
0.05	0.5256	0.4082	0.2904	0.1904	0.1112
0.055	0.6366	0.5290	0.4108	0.2892	0.1940
0.06	0.7156	0.6176	0.5060	0.3838	0.2574
0.065	0.7894	0.7072	0.6056	0.4892	0.3624

0.07	0.8498	0.7818	0.6882	0.5728	0.4464
0.075	0.8922	0.8348	0.7566	0.6574	0.5314

Table A3.3. Probability of decline below population thresholds within 25 years.

Proportion shot	Population reduction thresholds				
	0	0.05	0.1	0.15	0.2
0	0.0000	0.0000	0.0000	0.0000	0.0000
0.005	0.0002	0.0000	0.0000	0.0000	0.0000
0.01	0.0002	0.0000	0.0000	0.0000	0.0000
0.015	0.0018	0.0008	0.0004	0.0000	0.0000
0.02	0.0074	0.0040	0.0014	0.0010	0.0008
0.025	0.0276	0.0184	0.0126	0.0088	0.0046
0.03	0.0610	0.0428	0.0278	0.0156	0.0080
0.035	0.1240	0.0920	0.0648	0.0442	0.0264
0.04	0.2324	0.1760	0.1294	0.0950	0.0642
0.045	0.3742	0.3102	0.2488	0.1918	0.1402
0.05	0.5332	0.4642	0.3936	0.3218	0.2500
0.055	0.6780	0.6218	0.5552	0.4726	0.3952
0.06	0.8162	0.7692	0.7092	0.6426	0.5596
0.065	0.8978	0.8610	0.8232	0.7688	0.7014
0.07	0.9502	0.9318	0.9044	0.8702	0.8228
0.075	0.9734	0.9652	0.9518	0.9324	0.9050

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