Task 5 Technical Report: A Population Viability Analysis (PVA) to inform reinforcement planning and decision-making

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1 Executive Summary

1. Using a previously fitted Bayesian state-space model for the Capercaillie population in Strathspey, Scotland, we developed a stochastic, age- and region-structured tool for Population Viability Analysis.
2. The objective was to examine the efficacy of different translocation strategies of imported birds from external populations, under different biological assumptions. We used a time horizon of 50 years.
3. A translocation strategy comprised 1) the total number of imported birds, 2) the timing and duration of the translocation window, and 3) the distribution of translocations across the translocation window and the different subpopulations (using the mating lek as a unit).
4. We derived baseline PVA projections under the status-quo (a steeply declining trajectory) and under the possibility of management actions (an increasing trajectory as a result of fence

removal and diversionary feeding of predators).

5. We discuss various quantitative measures of risk (e.g. extirpation probabilities and extirpation time horizons), but we mainly focus on deriving a single, scenario-specific measure of risk based on the economic concept of risk-discounting for long-term, large-population futures.
6. We used the stochastic nature of our PVA and the two baseline scenarios to quantify this novel measure of risk, which describes how much of the efficacy of the other interventions can be achieved by different translocation schemes.
7. Using 5-year reintroduction windows, we find that translocations reduce risk in a manner directly proportional to the number of individuals introduced. Maximum numbers of 300 individuals over 5 years from the present can only achieve a modest 12% of the risk reduction achievable through fence removal and diversionary feeding.
8. Examination of the influence of density dependence on these results indicates measurable differences which nevertheless do not alter the overall efficacy of translocations, compared to the density independence scenario.
9. Examination of possible delays of translocation by 8 or 13 years from the present do not result in large losses of efficacy in the already modest performance of translocations.
10. We found no indication that translocations clustered by lek or year offer greater gains in risk reductions.
11. We conclude that improving base demographic rates (by landscape and predator management) is, under the most conservative scenarios, approximately eight times more effective than direct population subsidies by translocations.
12. Equivalently, based on our modelling and allowing for a 40% survival of imported individuals, we would estimate that we would need to translocate ~2,500 birds to achieve the same results as other forms of management.

Keywords: Diversionary feeding, Fence removal, Population subsidies, Population Viability Analysis, Risk of extirpation, State-space model, Stochastic modelling, Strathspey, Tetrao urogallus, Translocation experiments

2 Background to modelling tasks

The overarching objective of this study was to explore the utility of capercaillie translocations in terms of the population's long-term viability. In particular, we have followed the ITT as closely as current methodology allows, and also tried to go beyond the strict three objectives of the ITT, by exploring the following questions:

1) How does the beneficial effect of translocations compare with the status-quo?
2) How does the beneficial effect of translocations compare with a regime of impactful interventions such as fence removal and diversionary feeding of predators?
3) Are there optimal spatial patterns of introduction, i.e., is there any benefit to clustering the translocations by lek?
4) How does the benefit of reintroductions vary with the total number of birds translocated?
5) Is there any difference in the predictions under assumptions of weak or strong density dependence (even if carrying capacities are not known regionally yet)?
6) What is the anticipated difference between immediate and postponed action in terms of translocation?
7) How do we quantify risk in a way relevant to the capercaillie population, without ignoring the systematic stochasticities arising from the ecology, and the parameter uncertainties arising from statistical model fitting?

These questions require a reliable tool for Population Viability Analysis (PVA), a relevant definition of risk and a well-defined set of environmental and management scenarios. In this report, we orient our derivation of these three ingredients by the requirements of the above seven questions. We first set out the strengths and limitations of the existing modelling approach and its recasting here as a PVA. We then outline novel definitions for risk and optimisation of translocations. Finally, we

explore benefits and long-term impacts under scenarios of density dependence and delayed action.

3 Biological assumptions of the model

The Strathspey capercaillie **PVA modelling framework** (Niven et al. 2026), as adopted by the Capercaillie Emergency Plan (CEP), allows us to investigate formally the effect of aggregated or staged reintroductions, into any given lek, or forest subregion in Strathspey. The model comprises both process and observation components. The process component (Figure 1 a), operates at the level of a single subpopulation (best identified with the neighbourhood of a lek) and comprises stages for chicks, juveniles and adults, distinguishing between males and females. Demographic processes, such as fecundity, survival and recruitment are age/sex-specific and covariate-dependent. In Niven et al. (2026), the effect of covariates was considered extensively (e.g., effects of fences, predation, climate trends, annual weather), and these types of investigation remain possible, but for future projections, when covariate trends are not available, there was also a version of the population model that encompassed environmental variation (both across leks and years) by using random effects. The observation model (Figure 1 b) makes use of data on observed lek size, brood size and total population size. These data were used to fit the model identifying all parameters (e.g., baseline demography and magnitude of random effects) and population states (e.g., initial population values, fluctuations in sex ratio). A distinction is made between the *model fitting* stage (increasingly known in the machine learning era as model training) and the *model prediction* stage (conducted by simulating forward from the fully fitted/trained model).

Our intention for this investigation was to explore the added long-term benefits conferred by each additional translocation into each lek. We focused on how many birds should be reintroduced, and at which locations and times, to maximise positive impact (it may be that the optimal strategy

is to introduce all birds at the same time and place, or that a more distributed, staged approach is better). The desired end-product of the PVA modelling is a recommended (optimal) scheme of reintroductions (i.e., numbers of animals per sub-region and year) that achieve a predetermined acceptable target reduction in the probability of extinction of the study population.

The generation of a PVA tool from the fitted population model required us to recast the Bayesian fitting code into stochastic simulation code, using the full complement of joint posterior parameter estimates, and retaining the uncertainty in the predictions. Since the objective of the translocation project was horizon scanning and forecasting for management, we did not use the version of the model with covariates, but rather, the model with random effects. This therefore also allowed the propagation of uncertainty and stochasticity relating to environmental fluctuations, as previously quantified by (Niven et al. 2026) during model fitting. The PVA is implemented under certain simplifying assumptions, partly inherited from (Niven et al. 2026) which we set out first, to allow us to contextualise the predictions, later in the report.

3.1 Density independence

We recognise that the data for Strathspey track a period of the Scottish capercaillie population during which every subpopulation unit is known to be in decline and well-below carrying capacity. Therefore, the model of Niven et al. (2026) did not attempt to estimate the strength of density dependence, as we have done in other modelling efforts (Jeglinski et al. 2023). Instead, a very high value of carrying capacity was placed on the maximum number of males that could be accommodated by each lek. This was assumed to be 50 observed cocks, which, when scaled by the detection probability (estimated at 30%), scales to ~167 cocks. The absence of density dependence in models of capercaillie follows precedent (Augustine et al. 2020), however, ignoring density dependence in PVA scenarios can lead to overoptimistic predictions. Even if the majority of forecasts from the PVA

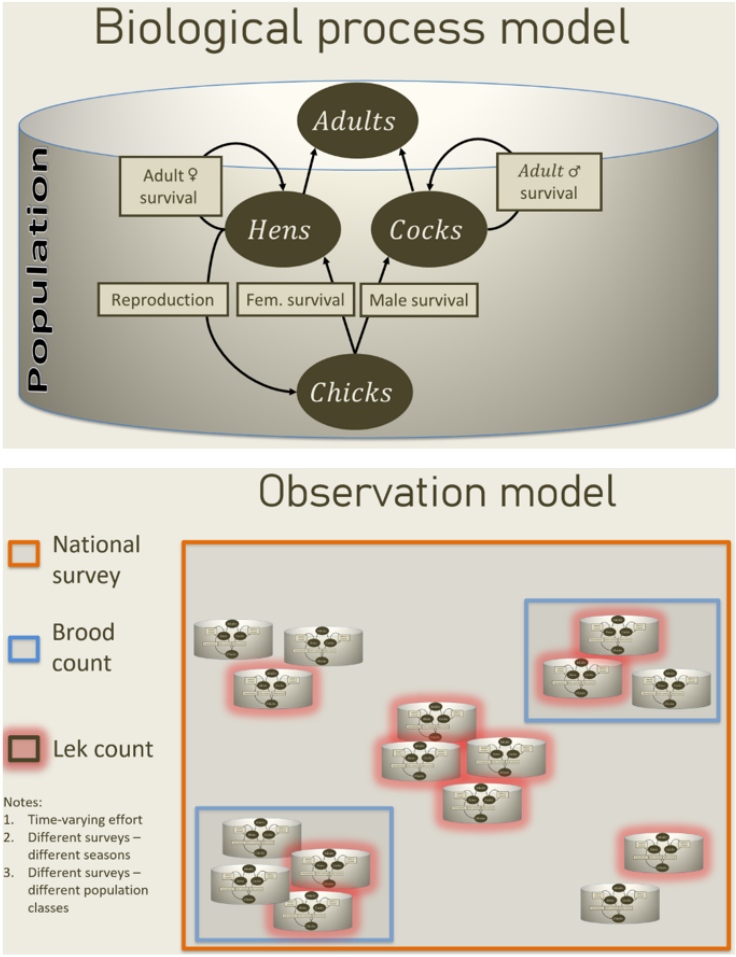

Figure 1: a) The Caparcaillie population model consists of a subpopulation process model, that keeps track of population classes and the demographic processes that connect them. b) This process model is replicated for each lek-subpopulation, and these units of population structure are clustered, first into regional/forest groupings, and second into the complete population of Strathspey. Data were available for different levels of that hierarchy and the information in these was used during model fitting (see Niven et al. (2026)) to arrive at a fully parameterised population model for the whole of Strathspey.

suggest a declining population, there may be occasional outlier trajectories that increase indefinitely as a result of chance. Hence, for investigations later in the report where we sought to explore the effects of a stricter level of density dependence, we brought this number down to 20 observed cocks (Moss and Picozzi 1994), corresponding to a likely total of ~67 cocks per lek. It is worth noting that even this number is likely to be high, but it represents a stronger constraint, allowing us to explore if density dependence makes a difference for future projections. A final point regarding density dependence is that, in theory, if the subpopulations in Strathspey reached some notional carrying capacity, it is likely that the population would keep on growing, by the establishment of new leks, the reactivation of historical leks, or the expansion back into the species historical range. These questions will need to be properly addressed, if a recovery begins to be achieved, with the aid of full analyses of habitat composition and caper habitat preferences.

3.2 Connectivity and rescue effects

There was little information in the data to allow estimation of subpopulation connectivity in this model, and even though it is almost certain that individuals are able to disperse between forests and leks, the model of Niven et al. (2026) prioritised estimation of the dominant demographic processes. However, to account for the fact that dispersal can manifest as a local rescue effect for subpopulations that become extinct, the original model allowed random rescue effects to occur. We retained this feature here. In future extensions of these models, it will be useful to locate the leks geographically, so that we can explicitly model the likelihood of particular, centrally located leks to operate as sources for more peripheral subpopulations.

3.3 Survival of translocated individuals [Needs additional text and references]

There is little evidence for or against the existence of secondary effects in breeding and survival of reintroduced birds. Hence, we did not explore the possibility of second year, or second generation persistent effects. However, we assumed a conservatively low estimate of survival of 40% for the new introductions.

3.4 Effect of fence management and diversionary feeding

Having fitted the population model to the pooled survey data Niven et al. (2026) explored the implications for viability of fence removal/marking and diversionary feeding of capercaillie predators, during years of low alternative prey (voles). The range of estimates were obtained from the literature: Survival benefits of fence marking (1.37-15.3%, Baines and Andrew 2003), fence removal (2.92-19.8%, Moss et al. 2000) and breeding success benefits of diversionary feeding (44.1-371%, Bamber et al. 2024), were then used to explore stochastic viability of the population. It was found that even under the most conservative estimates, a combination of fence removal and diversionary feeding could reverse the capercaillie decline. Hence, these combined interventions represent a best-case scenario for capercaillie management, even if they only work to the minimum of their estimated function. We therefore defined an intervention scenario in which fence removal led to 2.92% increase in survival and diversionary feeding led to 44.1% increase in breeding success. We used this highly conservative scenario as a baseline for comparison, to evaluate the effectiveness of translocations (see next section).

4 Evaluating management performance: Definition of risk

Previous studies on capercaillie have focused on characterising minimum viable population sizes, often based on generic PVA tools (Marshall and Edwards-Jones 1998; Grimm and Storch 2000). Here, we take a more holistic and tailored approach to risk, by connecting the performance of each translocation scenario to a minimum and maximum baseline of risk.

The maximum risk baseline takes the form of future projections under maintenance of the status-quo (i.e., do-nothing scenario). For Strathspey, this is provided by the forecast of the model in Niven et al. (2026) and visualised in the top panel of Figure 2. The colour shading indicates the likelihood $f(P, T)$ that the population size P corresponding to the y-axis is achieved in the year T indicated by the x-axis. The grey curves show median projections and the dashed grey curves show their associated 75% credible intervals. The PVA generating these results is fully stochastic, accounting for both parameter uncertainty and demographic stochasticity.

The minimum risk baseline is defined as an increasing population that benefits from the combination of fence removal and diversionary feeding (see previous section). This is shown in the top panel of Figure 3.

Measures of future population viability can be expressed in terms of three ingredients. The probability of extirpation p, the time horizons of interest T, and the population threshold P_E below which we consider the population to have become condemned. We introduce two measures of viability (which have been plotted at the bottom half of Figure 2 and Figure 3):

Extirpation probability $p(P_E, T)$: The probability that the population has dropped below P_E individuals T years from today. In practice this is estimated by obtaining the proportion of stochastic projections (out of a large number, here, 10,000) that are below the threshold P_E in year T from

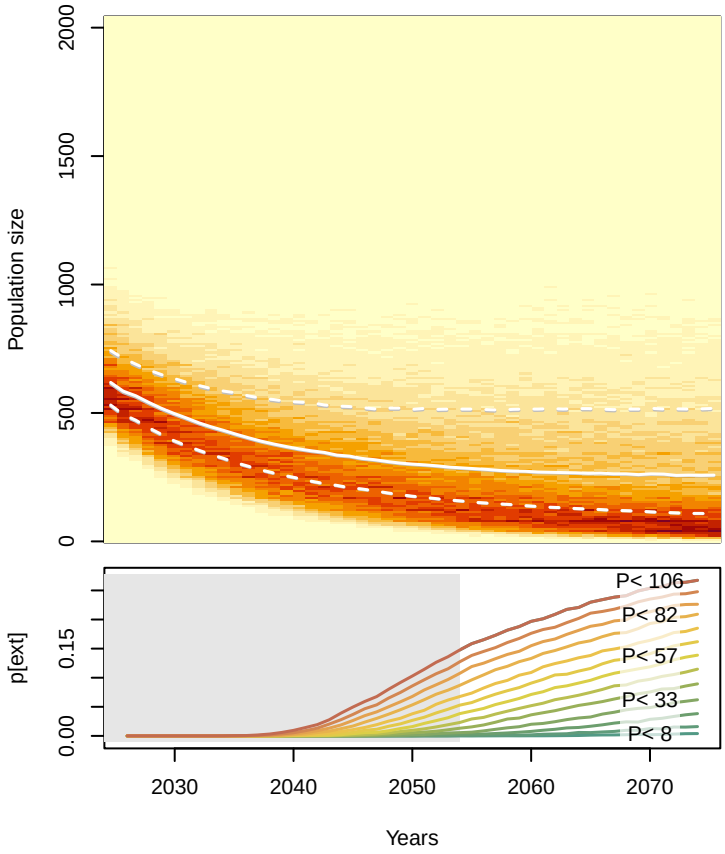

Figure 2: Current projections for the population for a time horizon of 50 years. The top panel shows population projections via the median population size (solid grey curve) with 75% credible intervals (dashed grey curves). The colours represent the probability density of the population being a particular size in a particular year (redder colours are more likely sizes). The bottom panel shows curves of the probability of extirpation (p[ext]). Each curve is the probability that the population is below a certain level in a particular year. The population thresholds are annotated on the curves. The grey part of the plot represents the extirpation time horizon, in this case, the time point at which 10% of trajectories have dropped below 100 individuals.

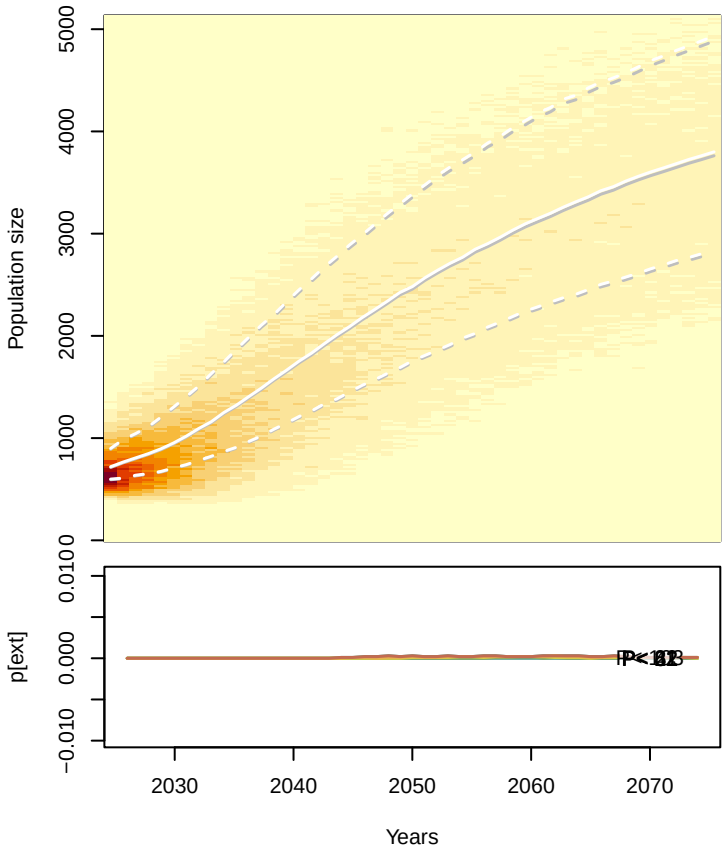

Figure 3: Projections for the population for a time horizon of 50 years under the assumption that survival increases by 2.92% from fence removal and breeding success increases by 44.1% from diversionary feeding. The top panel shows population projections via the median population size (solid white curve) with 75% credible intervals (dashed white curves). The colours represent the probability density of the population being a particular size in a particular year (redder colours are more likely sizes). The bottom panel shows curves of the probability of extirpation (p[ext]). Each curve is the probability that the population is below a certain level in a particular year. The population thresholds are annotated on the curves. The grey part of the plot represents the extirpation time horizon, in this case, the time point at which 10% of trajectories have dropped below 100 individuals.

now. In the bottom panel of Figure 2 we plot the probability of extirpation at different times in the future, for different population thresholds. Hence, a curve annotated as $P < 10$ represents the probability that the population drops below $P_E = 10$ individuals. For this report, we will use $p(100, 20)$ (i.e., probability of fewer than 100 individuals in 20 years) as our baseline extirpation probability. Under current projections this is $p(100, 20) = 10^{-4}$.

Extirpation time horizon $T_E(P_E, p)$: The time it will take for the population to drop below P_E individuals in a given proportion p of future trajectories. Here, we use $T_E(100, 0.1)$, which for the baseline scenario (under continuation of the status-quo) is estimated to be ∞. In the bottom panel of Figure 2 we represent this time horizon by a light grey band.

Both of these measures use a population threshold to quantify risk, and while they are informative visually, for the optimisation and reporting stages of the report (below) we need a single numerical summary of each scenario outlining risk as discerned from the full 10,000 population projections of each scenario. Our definition of risk borrows from the concepts of the above measures but avoids thresholding, instead using the concept of *discounting*. Two forms of discounting are considered:

Temporal discounting: Risk is higher if the population becomes small today, than in 5 years' time. Therefore, the more we can postpone small population sizes, the better a conservation strategy is. In economics, this is known as future discounting (a situation where investors will take a smaller return today than risk losing a higher return in the future).

Population discounting: Addition of an individual to a small population can make a big difference for risk, whereas when the population is already large, further increases represent diminishing returns for risk-reduction. Conversely, our evaluation of risk should be very sensitive to population changes at small population sizes. This is a form of population discounting.

We define risk as the weighted average of the likelihood $f(P, T)$ over the entire $P - T$ plane

$$R = \int_P \int_T f(p,t)w(p,t)dpdt \tag{1}$$

The weights in Equation 1 represent the two forms of discounting described above. To proceed with the applied investigations for capercaillie we need to be more specific about formulations for these weights. We can initially assume that there is no dependence between population and temporal discounting. This means that our perception of risk along the time axis does not alter on the basis of population size, and vice-versa, that we will not evaluate risk along the population axis differently for the present and the future. We can therefore write

$$w(p,t) = w_P(p)w_T(t) \tag{2}$$

We can define discounting as an exponential function in each of those directions, a convenient default used to model discounting in economics.

$$w(p,t) = exp(\theta_0 + \theta_1 p + \theta_2 t), \qquad \theta_1, \theta_2 < 0 \tag{3}$$

In a scenario where the population is known to be extinct in the present, this would give a projection in which all likelihood was concentrated at the origin of the $P-T$ plane, such that $f(0,0) = 1$. Equally, in that scenario, we would have $w(0,0) = exp(\theta_0)$. Setting these into Equation 1 implies $R = exp(\theta_0)$. We can define risk in that case to be equal to 1, implying $\theta_0 = 0$. This leaves two parameters, θ_1 and θ_2 that specify the rate of discounting along the population and time dimensions respectively. Because these are management-related parameters, they will always be characterised by subjectivity, and their impacts on the results will always be important. We define the following

two quantities that can be specified by managers:

- $T_{0.5}$ The time horizon at which our management concern for the population drops to 50% of the present. Here, we set this to 20 years from the present. It is clear from the remit of conservation plans for capercaillie that evaluation of risk needs to happen at a decadal, rather than annual scale, hence the choice of this value.

- $P_{0.5}$ The population size at which our management concern drops to 50%, compared to a very small population. Here, we set this to 500 individuals. This is the approximate size of the population currently, and it is considered a very low size by historical standards, so setting the parameter to this value indicates that any declines from this point should be accompanied by rapid increases in our evaluation of risk.

Although the particular values chosen above are important, their exact values within our 50-year horizon do not alter the conclusions of the rest of the work. Having defined these quantities, we can then write

$$\theta_1 = \frac{\ln 0.5}{P_{0.5}}, \qquad \theta_2 = \frac{\ln 0.5}{T_{0.5}} \tag{4}$$

This definition of risk (based on Equation 1, with the parameters of Equation 4), can be applied to the likelihood projections from any given scenario. Therefore, it can be calculated for our minimum and maximum baseline scenarios, as well as for any of our translocation scenarios. To allow an intuitive comparison between these three options, we introduce the following risk-reduction performance indicator:

$$\pi = \frac{R_{Sq} - R_{Tr}}{R_{Sq} - R_{In}} \times 100 \tag{5}$$

where, R_{Sq}, R_{Tr}, R_{In} are the values of risk, respectively under the status-quo, under a particular translocation scenario, and under the maximum baseline of fence and diversionary feeding interventions. This metric indicates what percentage of the performance improvements of a conservative fence and predator intervention scenario can be achieved by a particular scheme of translocation.

5 Optimising reintroduction schedule

For a given number N of reintroduced birds, we are interested in creating a template of reintroductions that will maximise the impact on viability. We define a schedule of reintroduction as an $M \times T$ matrix $\mathbf{R}$ with as many rows (M) as the subpopulations in the model (the leks being the highest level of disaggregation) and as many columns as the number of years into the future that the management strategy is required to apply. Hence, a particular entry $R_{m,t}$ is the number of translocated individuals introduced into lek m in t years from now. We call the complete matrix $\mathbf{R}$, the *translocation schedule*, or *schedule*, for short.

In the original ITT we were encouraged to consider the problem in terms of trigger points which we proposed to define as the parts of the population trajectory, in the present and future model projections, when reintroduction of an additional single individual causes the greatest long-term reductions in extirpation probability. In our PVA investigations such trigger points never became apparent, but we have nevertheless maintained the spirit of the approach.

Specifically, we considered the plausible case of a 5-year translocation program. The scenarios specified therefore looked at optimal times within this window, optimal positions of that time window within the broader future timeline and also, optimal allocation of translocations to different leks. Since there is very little difference estimated between leks in the fitted model (and this is likely to remain the case until variable habitat suitability is brought into the model), the optimisation

could only be relied on to advise whether translocations should be spatio-temporally clustered or dispersed across leks and years.

Our optimisation algorithm had the following steps:

1. Generate projections for intervention scenario (minimum risk baseline).
2. Calculate risk (R_{In}) associated with minimum baseline.
3. Using zero translocations, generate projection for status-quo (maximum risk baseline).
4. Calculate risk (R_{Sq}) associated with status-quo, and set risk reduction measure to zero.
5. Take next increment in the number of translocated individuals (we explored increments of 5 individuals in each iteration)
6. Place these individuals randomly by lek and year.
7. Rearrange a small number of individuals (5 in our implementation), by shifting them to a proximate lek ID and year.
8. Apply translocation mortality stochastically on these.
9. Generate population projections.
10. Calculate risk (R_{Tr}) associated with this translocation scheme.
11. Is the risk reduction associated with this scheme an improvement on previous best? If so, retain scheme and update risk reduction score. If not, then check how many iterations have occurred without improvement.
12. If iterations without improvement are more than 100, then go to step 5.
13. Otherwise, go to step 7.
14. Terminate when the algorithm has examined schemes for up to 300 translocated individuals.

6 Management under no density dependence

In this section, we followed the precedents of Niven et al. (2026) and Augustine et al. (2020) in omitting density dependent effects. The exhaustive exploration of translocation schemes and risk-reduction performance (Figure 4 a) shows a near-linear pattern of risk reduction. There are no clear inflection points in the curve that might be used to determine best value for cost of per-capita translocations. Additionally, the % values on the y-axis indicate modest gains via risk reductions (only up to 12% of the benefits that could be obtained by fence removal and supplementary feeding. Each of the values plotted in (Figure 4 a) corresponds to an optimised schedule of reintroductions. We have plotted the schedule corresponding to the maximum examined number of 300 translocations in (Figure 4 b). We see that, beyond the effects of stochastic placements, there is no recommendation to release animals in a particular year (within the 5-year program window), or a particular lek. Therefore, the schedule optimisation appears to recommend spreading the translocations across space and time.

7 Management under assumption of density dependence

It is currently difficult on the basis of available evidence, to quantify environmental carrying capacities for capercaillie (in the local or regional scales). However, even arguing from basic principles we may postulate that the addition of some level of density dependence is likely to lead to comparatively 1) less optimistic projections about future viability, and 2) less clustered recommended introductions by lek. Less optimistic projections, compared to the density independent case would come from the exclusion of stochastic outliers: On rare occasions, even a predominantly declining population will benefit from a confluence of positive environmental random effects and drift upwards. In the absence of a carrying capacity, these rare positive outcomes will prejudice a pessimistic scenario

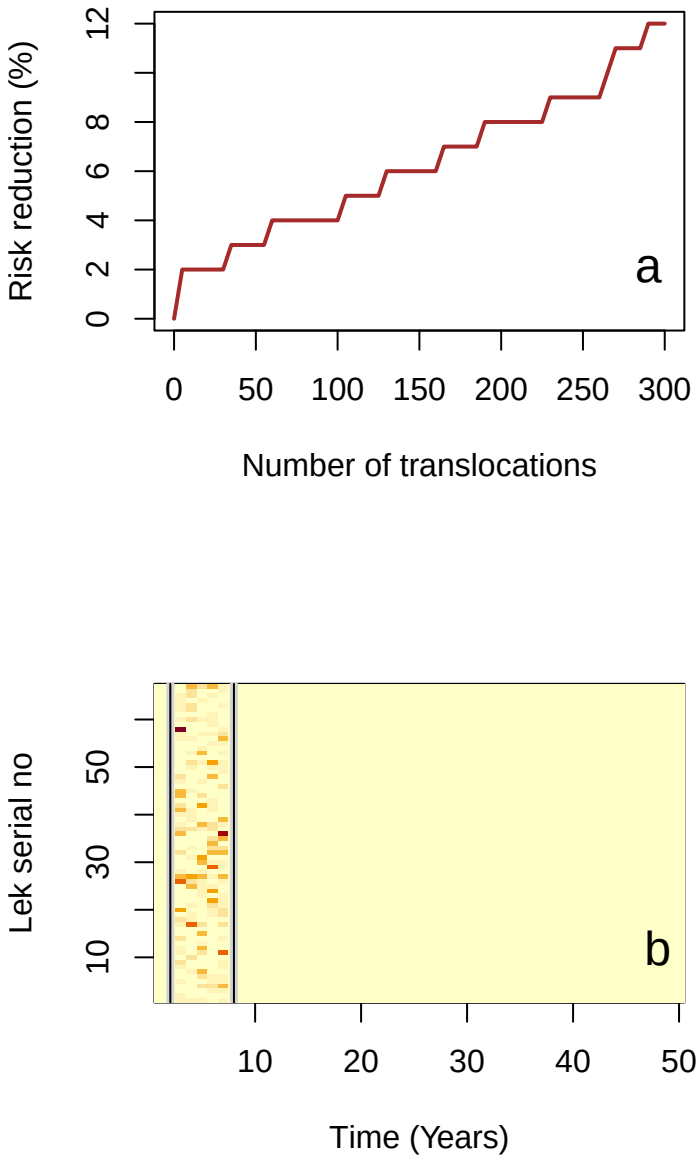

Figure 4: Investigation of risk reduction performance (as a percentage defined in Equation 5) obtained from optimal translocation schemes. In the top diagram, the maximum risk reduction achieved is plotted against the number of translocations. In the bottom diagram, the optimal arrangement of reintroductions across leks and the translocation time window is shown by the colour intensities.

upwardly. Similarly, concentrated translocations at single leks may be deemed sub-optimal by the schedule design algorithm because increasing translocations at the same lek will have diminishing returns in the presence of a carrying capacity.

We explored these hypotheses by introducing a more stringent limit to the lek carrying capacity, from 50 down to 20 observed cocks (Moss and Picozzi 1994). Once this observed number is scaled up by our estimated detection probability of 0.3, it corresponding to a likely total of ~67 cocks per lek.

We first generated the two baseline scenarios corresponding to the management status-quo and to the scenario of fence removal and diversionary feeding. The status-quo scenario with density dependence (Figure 5) gives only marginally less optimistic projections than the equivalent density independent scenario (Figure 2). The median trajectory and its credible intervals become more aligned with the probability densities (shown in heat colours), which means that median projections are less affected by optimistic outliers. The extirpation time horizon is not affected, but extirpation probabilities become slightly higher in later years.

A stronger effect is observed in the baseline scenario with interventions (Figure 6), because in this case interventions are expected to lead to an increasing population and therefore the carrying capacity is realised in several leks in the future years of the PVA. The population is projected to settle to a size of ~2000 individuals (both sexes). Of course, this only refers to the extant leks in Strathspey, so this optimistic scenario would project much higher numbers if it was augmented by a habitat suitability model that allowed the creation of new leks pushing back into the species' historic range across Scotland.

Using these two baseline scenarios, as we did in the case of density independence, we explored the efficacy of optimal translocation programs for different numbers of individuals (Figure 7). We found

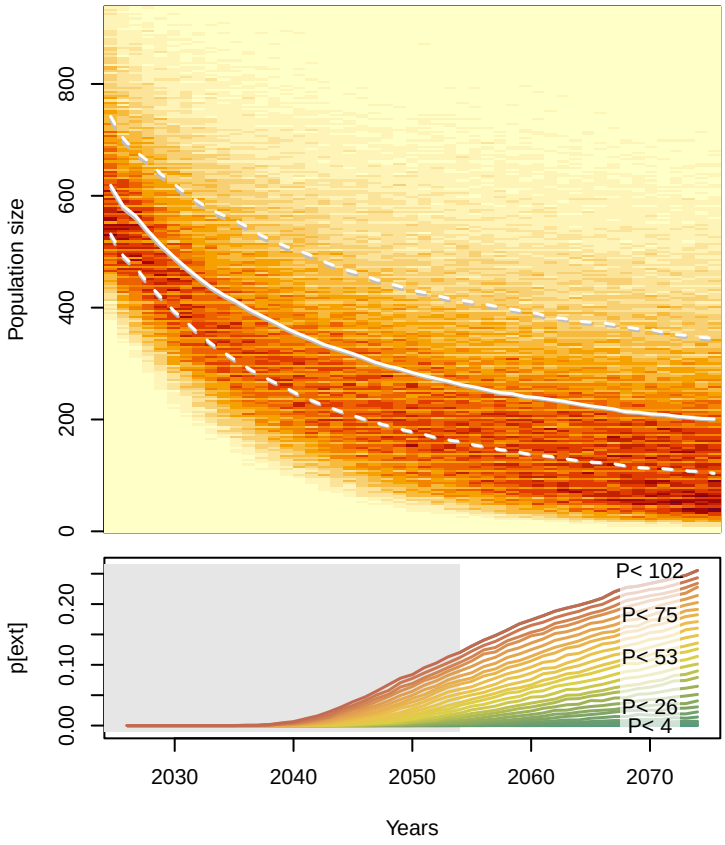

Figure 5: Baseline scenario of density dependence with no interventions. Current projections for the population for a time horizon of 50 years, under a scenario of density dependence. The top panel shows population projections via the median population size (solid white curve) with 75% credible intervals (dashed white curves). The colours represent the probability density of the population being a particular size in a particular year (redder colours are more likely sizes). The bottom panel shows curves of the probability of extirpation (p[ext]). Each curve is the probability that the population is below a certain level in a particular year. The population thresholds are annotated on the curves. The grey part of the plot represents the extirpation time horizon, in this case, the time point at which 10% of trajectories have dropped below 100 individuals.

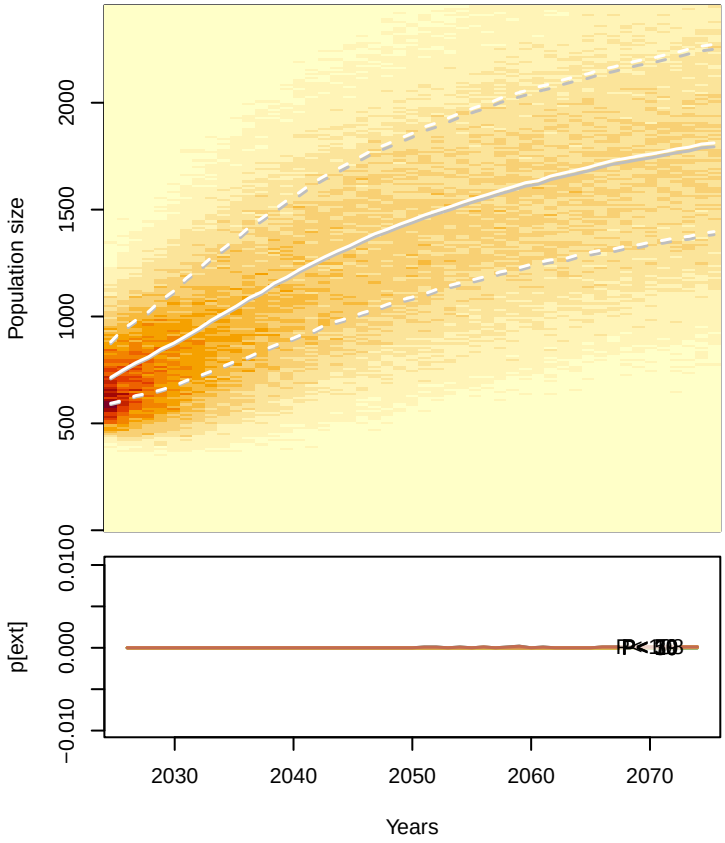

Figure 6: Baseline scenario of density dependence with interventions. Projections for the population for a time horizon of 50 years under the assumption that survival increases by 2.92% from fence removal and breeding success increases by 44.1% from diversionary feeding. The top panel shows population projections via the median population size (solid white curve) with 75% credible intervals (dashed white curves). The colours represent the probability density of the population being a particular size in a particular year (redder colours are more likely sizes). The bottom panel shows curves of the probability of extirpation (p[ext]). Each curve is the probability that the population is below a certain level in a particular year. The population thresholds are annotated on the curves. The grey part of the plot represents the extirpation time horizon, in this case, the time point at which 10% of trajectories have dropped below 100 individuals.

almost no difference either in the overall risk reduction (compare Figure 7 a with Figure 4 a) or the general pattern of clustering across leks and years (compare Figure 7 b with Figure 4 b). So, even though density dependence makes an incremental difference on individual projection scenarios, we expect that this will not have a great bearing on any decisions about translocation strategies.

8 Delayed translocations

A key question discussed under the Capercaillie Emergency Plan, is whether translocations should be expedited, or whether we can afford, to treat the strategy as a solution of last resort, in case the other types of interventions cannot be implemented successfully (e.g., in the instance of no agreement between stakeholders to implement fence and predator diversionary management). To investigate this question, we re-run the analysis of the previous section twice more, for five-year translocation programs that were initiated further into the future. Our PVA runs up until this point had assumed that translocations would be initiated in 2027 (i.e., a two-year delay after 2025). In the new runs, we considered initiation times of 2032 and 2037. We have summarised these results in Figure 8. Although there is a small loss in the efficacy of risk reduction (Figure 8 a) between the three time windows, this is of the order of 2%, an impact that is likely to be undetectable in a real system. Similarly, there was no difference in the recommended schedule of translocations between the three time windows (Figure 8 b, c and d). We therefore conclude that the small benefit likely to be conferred by translocations is not going to be lost by postponing them, until more powerful interventions have been examined and attempted.

9 Conclusions

The analyses adopted in this report lead to a concise set of conclusions:

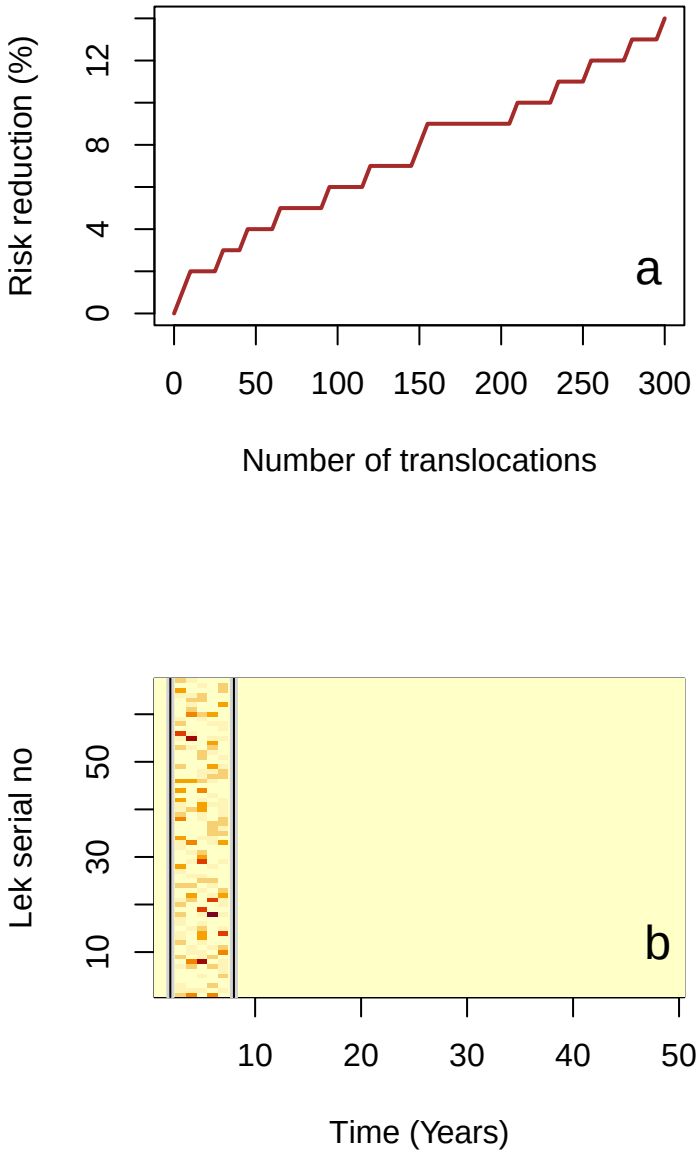

Figure 7: Current projections for the population for a time horizon of 50 years. The top panel shows population projections via the median population size (solid white curve) with 75% credible intervals (dashed white curves). The colours represent the probability density of the population being a particular size in a particular year (redder colours are more likely sizes). The bottom panel shows curves of the probability of extirpation (p[ext]). Each curve is the probability that the population is below a certain level in a particular year. The population thresholds are annotated on the curves. The grey part of the plot represents the extirpation time horizon, in this case, the time point at which 10% of trajectories have dropped below 100 individuals.

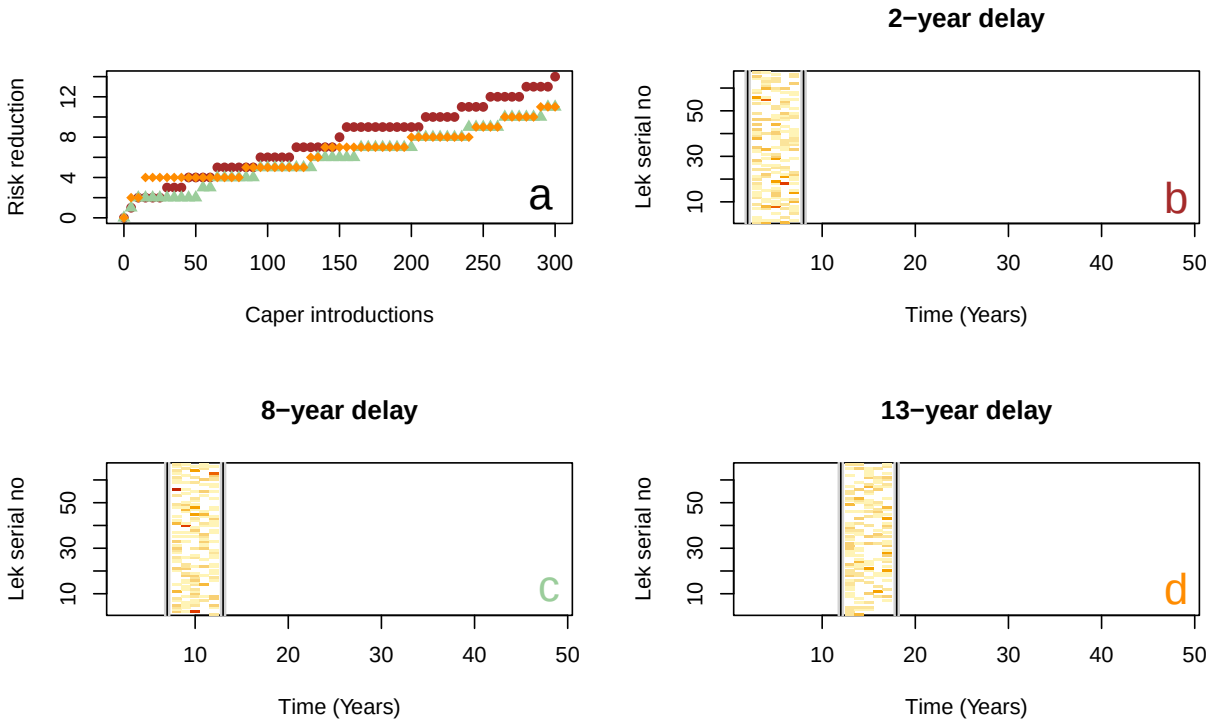

Figure 8: Summary of results from delayed translocation programs. Efficiency of the programs, as previously is represented by the ability of translocations to achieve risk reductions, as a proportional position between the status-quo and optimistic intervention scenario. We present three scenarios of translocations corresponding to 5-year schedules but with increasing delay times of 2 years (brown colour in plate a, and optimal 300 individual schedule shown in b), 8 years (sage colour in plate a, and optimal 300 individual schedule shown in c) and 13 years (orange colour in plate a, and optimal 300 individual schedule shown in d).

- Currently, the population is projected to decline.
- Even if the twin interventions of fence removal and predator diversionary feeding operate to the least of their anticipated efficacy, the declining projections stand to be reversed.
- Translocations of up to 300 individuals, implemented in the near future, over a 5-year time window can only achieve approx 12% of the expected improvements from other interventions.
- The pattern of improvements (i.e., risk reductions) increases linearly with increasing numbers of translocated individuals.
- These conclusions are not sensitive to modest increases in the strength of density dependence (i.e., the assumed carrying capacity of individual leks being reduced from 50 to 20 observed cocks).
- The incremental benefits from translocations are not sensitive to delays in their initiation. We examined delays of up to 13 years.
- Overall, improving base demographic rates (via landscape and predator management) is, even under the most conservative scenarios, approximately eight times more effective than direct population subsidies by translocations.
- Equivalently, based on our modelling and allowing for a 40% survival of imported individuals, we would estimate that we would need to translocate ~2,500 birds to achieve the same results as other forms of management.
- For comparison, this would imply importing close to four times the estimated current population in Strathspey.

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