BI4016 Research Project Thesis 2024/2025 School of Biological Sciences

Post-release distribution of pheasants in Scotland: distance-decay or uniform occupancy?

Student ID: 52210454

Work completed in partial fulfilment of the requirements for the degree BSc Honours Zoology

Word count (excluding references and legends): 4527

Abstract

Mass releases of captive-reared common pheasants for hunting represent one of the most significant introductions of vertebrate biomass in Europe. Pheasants have been shown to exert negative pressure on biodiversity through direct predation of reptiles, indirect predator subsidisation and disease transmission into wild populations. In the UK, current regulation prohibits releases within 500 m of protected areas, based on the assumption that pheasant occurrence declines with increasing distance from release pens. Here, we evaluate whether pheasant occupancy declines with distance from release sites (distance-decay hypothesis) or instead reflects a wider, persistent landscape reservoir formed through overwinter survival and multiyear reinforcement (landscape reservoir hypothesis). We deployed 45 camera traps across known pheasant release areas in the north-east of Scotland to collect detection–non-detection data. We used a single-season occupancy model, fitted in a Bayesian framework, to estimate pheasant occupancy. Occupancy was modelled as a function of distance from the nearest release pen. The effect of distance was estimated at 0.22, but the 95% credible interval ranged from -0.83 to 1.26, suggesting a lack of any directional relationship. This absence of a distance–decay relationship indicates that pheasants persist across the broader landscape rather than remaining concentrated around release sites, implying that a fixed 500 m buffer may not provide sufficient protection for native biodiversity, and highlighting the need for policy approaches that account for landscape-scale pheasant persistence when assessing ecological risk.

1 | Introduction

Human-mediated introductions of invasive alien species are a major driver of biodiversity loss, repeatedly causing population crashes, local extinctions, and genetic erosion in native wildlife (Simberloff, 1997). A prominent example is the deliberate release of non-native game birds for traditional sports, such as shooting. In Europe alone, 24.3% of mammals and 30.2% of birds introduced over the past 100 years were released for shooting purposes (Carpio et al., 2016). For decades, the common pheasant *(Phasianus colchicus)* has been released in large numbers across Europe to support traditional sporting practices. Across the UK, an estimated 35-46 million pheasants have been released annually over the last three years (Sage et al., 2025). In the Cairngorms National Park, releases reported over the same period by 10 participating shoots (representing 45% of identified shoots) indicate a median of 50,900 pheasants per year, however, these figures reflect only the surveyed shoots and not the entirety of shoots within the Park (Fletcher *et al.*, 2025). These releases make up one of the largest deliberate introductions of vertebrate biomass in Europe. Pheasants account for 74-81% of all alien biomass in the UK, and the biomass of the autumn release alone surpasses that of the entire native breeding bird assemblage in the UK (Blackburn & Gaston, 2018).

The ecological consequences of these introductions can be severe. In Belgium, reptile populations were absent from six sites subject to mass pheasant releases but present in all 261 control sites, with an average of 3.2 species recorded (Graitson & Taymans, 2022). The complete absence of the slow worm *(Anguis fragilis)*, typically present at several hundred individuals per hectare, highlights the intensity of predation pressure. Additionally, an isolated population of European adder *(Vipera berus)* failed to recolonise a previously inhabited site a decade after releases ceased, suggesting the introduction of pheasants led to the local extinction of this population (Graitson & Taymans, 2022). However, for parts of the UK, the evidence base for understanding the ecological impacts of pheasant releases remains limited, and recent assessments highlight the need for further research (Fletcher *et al.*, 2025).

Not only do pheasants act as predators, but they have also been suggested to indirectly increase predation pressure on native species. In a natural system, predator density is limited by prey abundance (Pringle *et al.*, 2019), but by releasing millions of captive-bred naive birds, predators are provided with an easy food source that has unusually low predator evasion ability (Madden and Perkins, 2017), effectively removing the density dependence of natural predator-prey systems. In 2019, a study found that in 1-km squares with high gamebird abundance, avian generalist predators exhibited significantly higher inter-annual population growth rates (Pringle *et al.*, 2019). Moreover, adult pheasants are not randomly distributed in the landscape, during brooding, females concentrate in insect-rich weedy and rough grass areas (Hill, 1985), which are structurally similar to the unmanaged herb-rich swards commonly maintained within protected areas. Therefore, predation pressure may not only be elevated overall but spatially biased towards some of the very sites that hold the highest conservation value and support many of the most threatened species.

Beyond altering predator-prey dynamics, captive-bred pheasants may also act as bridge hosts for avian pathogens, facilitating disease transmission between gamebirds and wild bird populations (Dwight et al., 2021). Pheasants are captive-reared in high densities, which directly facilitates disease transmission from both close contact and stress-mediated immune suppression, effectively creating incubation environments for pathogens (Scott, 1988). In 2021, this pathway was realised in Finland, where highly pathogenic avian influenza (HPAI) caused mass mortality events in released pheasants, with thousands of dead pheasants across four shooting estates (Tammiranta *et al.*, 2023). The mass mortalities created ideal conditions for spill-over events to mammals as predators scavenged infected carcasses and preyed upon debilitated birds, resulting in infections in red foxes (*Vulpes vulpes)*, a Eurasian otter *(Lutra lutra)* and the first recorded Eurasian lynx *(Lynx lynx)* infection (Tammiranta *et al.*, 2023).

Given the threat of direct predation, predator subsidisation, and pathogen transmission, it is reasonable to assume that the mass annual release of pheasants would be subject to stringent safeguards, particularly around sites hosting sensitive native species. In England, this protection is implemented through Natural England's general licence (WML-GL43), which prohibits the release of pheasants and red-legged partridges on or within 500 m of protected areas which fall under European site boundaries (e.g. SSSI) unless authorised under licence (Natural England, 2024). This policy treats 500 m as a pragmatic and biologically meaningful threshold beyond which ecological impacts are presumed to decline to negligible levels. Natural England justifies this threshold by "There is moderate but consistent evidence from a series of radio tracking studies to indicate that the majority of released gamebirds do not, on average, disperse further than 500m from their point of release, with numbers declining with distance (Sage et al., 2020)" (Natural England, 2024). However, the Sage et al. (2020) review does not provide a scientific foundation for any spatial buffer, let alone one as specific as 500 m. The review is a rapid narrative synthesis, not a spatial analysis, and it explicitly acknowledges that most underlying studies are non-experimental, limited in scale, and subject to methodological bias. Even the dispersal evidence used to support the buffer is selectively drawn from a radio-tracking study in which 334 pheasants were monitored for approximately 100 days following release. During this short period, the mean maximum distance reached by individuals was 913 m (± 82 m), with movements ranging from 32 m to 4,685 m (Turner, 2007); almost double the current threshold. Under such assumptions, we may expect to see occupancy of pheasants decline notably, ca. 900 m from the nearest release pen, directly contradicting the 500 m threshold.

Yet even this contradiction is not the central issue. The fundamental limitation is that, by relying on short-term telemetry to assess ecological risk, Natural England treats dispersal as a proxy for long-term occupancy, effectively assuming that the short-term movements recorded by telemetry indicate where pheasants will ultimately occur at any point thereafter. This interpretation treats pheasant releasing as a form of point-source pollution, akin to a chemical spill in which impact decays predictably with distance from a fixed origin. However, unlike a chemical contaminant, a pheasant is a mobile biological vector that actively seeks suitable habitat and may cluster in sensitive sites far beyond its average post-release dispersal distance. The failure of this point-source interpretation becomes evident when we observe that approximately 15% of released pheasants survive each shooting season (Sage et al., 2025). Annual releases of tens of millions of birds, combined with clutch sizes of up to 15 eggs (Shick, 1947), mean that these survivors can establish reservoirs of naturalised individuals capable of breeding in subsequent years. The repeated generational cycle of release, dispersal, overwinter survival, reproduction, and subsequent dispersal can establish a widespread landscape reservoir of pheasants that persist independently of their proximity to active release sites, extending ecological effects far beyond the immediate shadow of any individual pen. On this basis, our *a priori* expectation is that pheasant occupancy should show no relationship with distance to release pens, even if newly released birds were limited to dispersing 500 m on average. It is this landscape reservoir, rather than short-term post-release movements, that should be considered when evaluating broad-scale ecological effects, yet it remains absent from the literature and thus from the evidence used to justify the current regulatory buffer.

Within this paper, we test the core assumption underpinning Natural England's 500 m buffer, that pheasant occurrence declines with increasing distance from release pens. However, under the landscape reservoir hypothesis, long-term patterns of occurrence are expected to be independent of proximity to active release pens, reflecting the multi-year accumulation of surviving and breeding individuals rather than short-term post-release movements. Therefore, we hypothesise that either occupancy will decline between 500 to 900 m from the nearest release pen, supporting the distance-decay hypothesis, or alternatively that occupancy will show no relationship with distance to release sites, supporting the landscape reservoir hypothesis. By quantifying pheasant occupancy relative to release sites and habitat structure, this study provides the first spatially explicit assessment of whether pheasant presence remains concentrated around individual release pens or instead represents a broader landscape reservoir that is inconsistent with the assumptions underlying the current 500 m regulatory boundary as well as having relevance to current discussions about pheasant releases in the Cairngorms National Park (Fletcher *et al.*, 2025).

2 | Materials and Methods

2.1 | Study area

Fieldwork was conducted between 16th of June 2025 to the 20th of August 2025 in Aberdeenshire, north-east Scotland (57.06°–57.36° N, –2.94°–2.02° W, Figure 1). The region includes extensive gamebird management, alongside protected areas (two National Nature Reserves (NNRs)). The landscape is predominantly intensively managed grass and arable systems, hedgerow networks, and semi-natural woodland, providing a strong analogue for many pheasant hunting landscapes across the UK. Six pheasant release pens were identified through a combination of NatureScot records and local landowner knowledge. All but two of these release pens were active, with pheasants having been released October 2024 (Figure 1). We were able to obtain release information for two of the active release pens, which reported releasing 100 and 300 pheasants in October 2024.

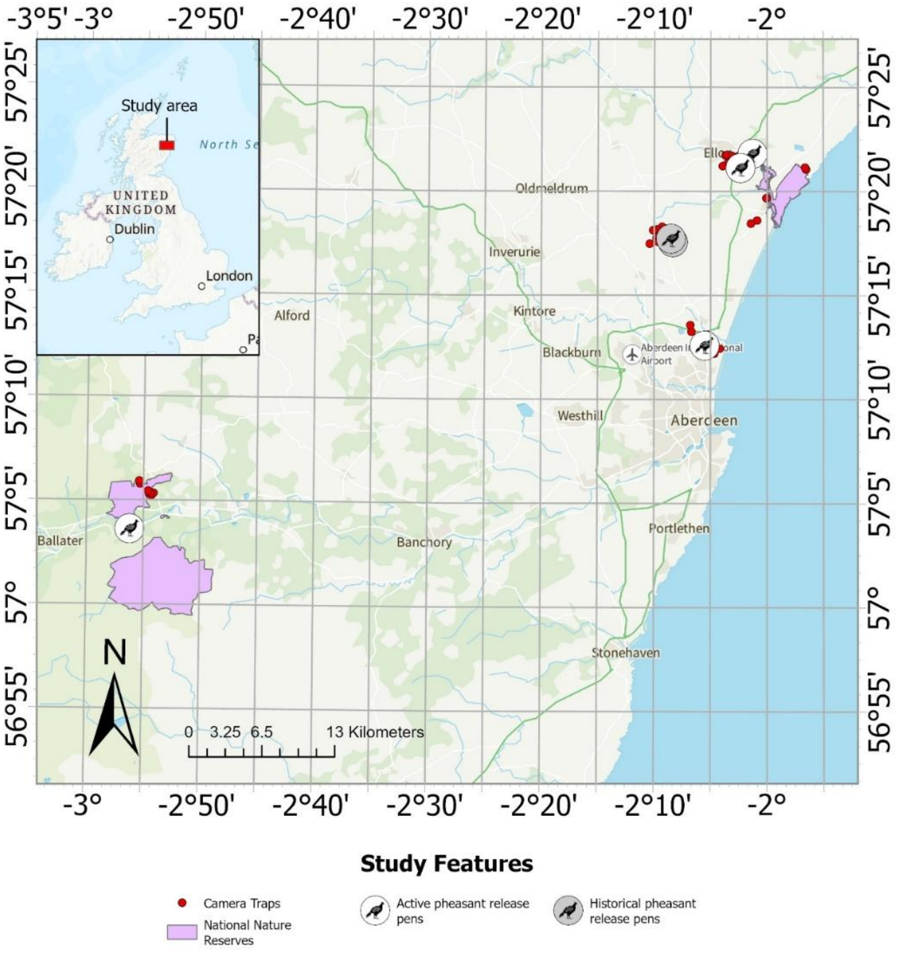

Figure 1. Study area, National Nature Reserves, camera traps and the six pheasant release pens.

A total of 45 camera traps (a mix of models but predominantly Browning BTC-8A) were installed across the study area to capture pheasants. Landowner permission was requested prior to any work, with camera locations selected based on landscape features likely to facilitate pheasant detection if present, i.e. woodland edges, hedgerows, dense shrub cover, and natural choke points. Site selection was further guided by field signs such as tracks, trampled vegetation, calls, or direct sightings, combined with local knowledge of pheasant activity. Camera traps were positioned 1.2 km apart on average (range: 43 m – 1.7 km, Figure 1). Each camera was mounted approximately 40 cm above ground level, angled to capture any pheasant movement, oriented away from the south to avoid glare, programmed to take four images in quick succession with a 30-second interval, and operated continuously for 14 days.

At each site, covariates expected to influence pheasant occupancy (ψ) through cover availability, predator avoidance (Kenward et al., 2001), the availability of seeds and arthropod-rich foraging habitat, such as weedy margins and rough grass (Hill, 1985), were recorded.

These included distance from the nearest pen (m), canopy cover (%), shrub density (%), woodland density (%), and arable area (%) (Figure 2). Shrub density, woodland density, and arable area were quantified within a 300 m buffer around each camera trap, representing the typical daily home range of pheasants. Shrub density was estimated using the Copernicus Land Monitoring Service, Small woody features 2018 dataset (100m resolution) (CLMS, 2025), while woodland density (%) and arable area (%) areas were derived from the UK Centre for Ecology and Hydrology (UKCEH) Land Cover Map 2024 (Morton *et al.*, 2024). Distance from release pen (m) was calculated as the Euclidean distance between each camera and the nearest pen. Canopy cover (%) was visually estimated in the field during camera deployment. Spatial covariates were processed using ArcGIS Pro.

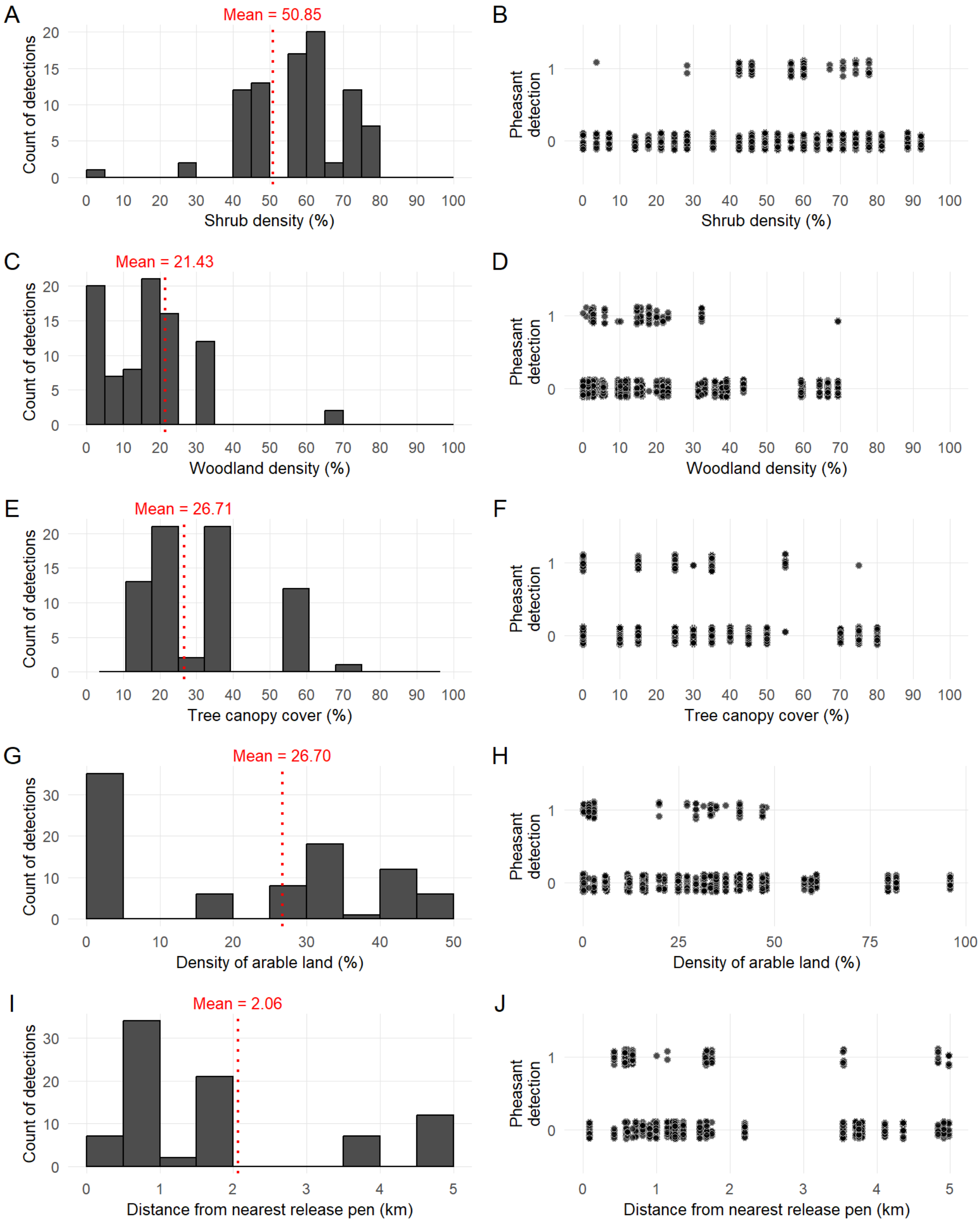

Figure 2. Histograms show the frequency of detections across the range of each occupancy covariate (red dashed line = mean covariate value). Scatter plots show detection outcomes across the same covariate ranges for all surveys.

To account for variation in detection probability (p) covariates suspected to influence ground nesting bird activity and detection were included for each survey day (Moynahan et al., 2007). These covariates included vegetation height (cm), mean daily temperature ($^{\circ}C$), daily rainfall (mm), open or closed camera view (open view was defined as camera placements where the field of view was largely unobstructed, such as open grassland, or other areas without substantial vegetation or structural barriers, closed view referred to sites where visibility was restricted by dense vegetation, shrubs, trees, or other features that could obstruct detection), and the number of days since the project began (Figure 3). Weather data was retrieved from Visual Crossing Weather Service (Visual Crossing Corporation, 2025), all other detection covariates were visually estimated in the field during camera deployment. Each image was timestamped, allowing detection histories to be constructed for each site across the 14 survey days. Each day of sampling was treated as a survey occasion, providing sufficient replication and satisfying the closure assumption required for occupancy modelling.

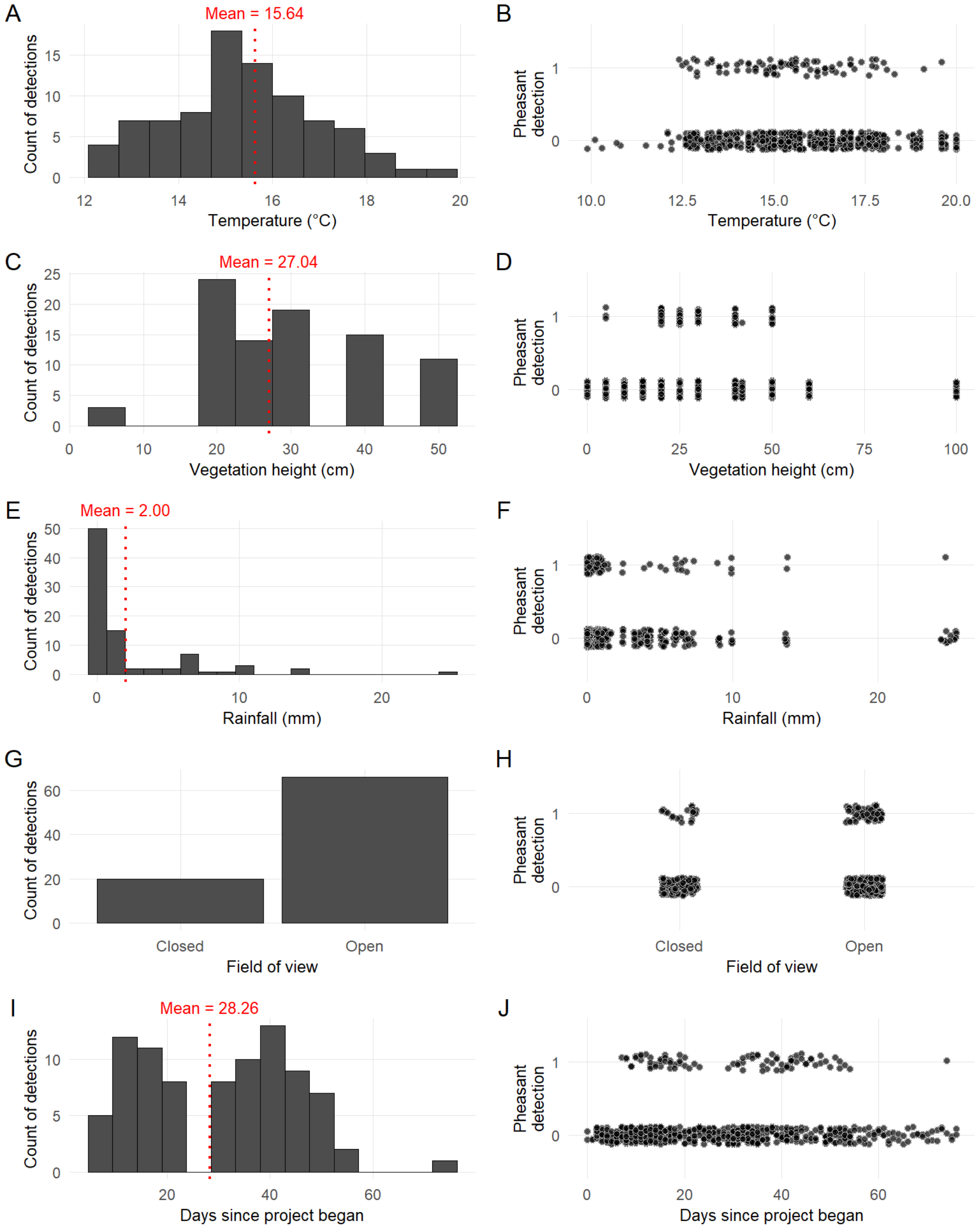

Figure 3. Histograms show the frequency of detections across the range of each detection covariate (red dashed line = mean covariate value). Scatter plots show detection outcomes across the same covariate ranges for all surveys.

2.2 | Statistical analysis

To evaluate which of the two competing hypotheses, the distance–decay hypothesis or the landscape reservoir hypothesis, was better supported by the data, we used a Bayesian hierarchical single-season spatial occupancy model to estimate pheasant occurrence in relation to distance from release pens and habitat structure. Occupancy modelling is well suited to this problem because it captures patterns of site use and persistence rather than short-term post-release movements, providing a biologically realistic means of evaluating how pheasants distribute themselves across landscapes with a long history of releases. We fitted the model to the detection–non-detection data collected from the camera traps, modelling occupancy as a function of distance to release sites and habitat covariates. Because species may be present but go unrecorded, occupancy models separate the ecological process (true occupancy) from the observation process (detection given presence), thereby providing unbiased estimates of both occupancy and detection probability (MacKenzie et al., 2002). Imperfect detection was accounted for using survey-level predictors. The occupancy model was specified as follows:

$$z_i \sim \text{Bernoulli}(\psi_i)$$

$$logit(\psi_i) = \beta_0 + \beta_1\,\text{arable}_i + \beta_2\,\text{dist}_i + \beta_3\,\text{shrub}_i + \beta_4\,\text{wood}_i + \beta_5\,\text{canopy}_i + w_i$$

$$y_{i,j} \sim \text{Bernoulli}(p_{i,j} \times z_i)$$

$$logit(p_{i,j}) = \alpha_0 + \alpha_1\,\text{days}_{i,j} + \alpha_2\,\text{rain}_{i,j} + \alpha_3\,\text{temp}_{i,j} + \alpha_4\,\text{veg}_{i,j} + \alpha_5\,\text{open}_{i,j}$$

$$w_i \sim \text{MVN}(0, \Sigma)$$

$$\Sigma_{ij} = \sigma^2 \times C(d_{ij})$$

where z_i is the true occupancy state at site i and ψ_i is the probability that site i is occupied. The detection probability at site i on survey day j is represented by $p_{i,j}$. β_0 is the intercept and $\beta_1 \ldots \beta_5$ are the slopes for the occupancy covariates. α_0 is the intercept, $\alpha_1 \ldots \alpha_4$ are slopes for the detection covariates, and α_5 is the difference of open view relative to closed. Spatial dependence was represented via a latent site-level spatial random effect w_i, modelled as arising from a multivariate normal distribution, where Σ is the covariance matrix defined as $\Sigma_{ij} = \sigma^2 C(d_{ij})$, and $C(d_{ij})$ represents the Matérn correlation function of the pairwise distance between sites, with smoothness ν and range ϕ. A Uniform(0.5, 2.5) prior was placed on ϕ as it was assumed spatial autocorrelation in occurrence would not extend beyond 2.5 km given a maximum dispersal of 913 m (± 82 m) (Turner, 2007). Uninformative priors $Normal(0, 2.72)$ were placed on occupancy and detection coefficients ($\beta_0, \ldots, \beta_5$ and $\alpha_0, \ldots, \alpha_5$) because there was insufficient prior knowledge regarding the plausible magnitude or direction of covariate effects on pheasant occupancy.

Models were fitted using the spOccupancy package (Doser et al., 2022), within R (v4.5.1, R core team, 2025) which fits Bayesian hierarchical occupancy models via Markov Chain Monte Carlo (MCMC) sampling. All continuous covariates were mean centered and standardised prior to analysis. The model was run with 4 MCMC chains, each with 200 batches of 50 iterations (10,000 iterations total) with 200 burn-in iterations. Convergence was assessed using traceplots, the Gelman-Rubin statistic ($\hat{R}$), and the effective sample size (ESS).

Packages used throughout included dplyr (Wickham *et al.*, 2023), ggplot2 (Wickham, 2016), patchwork (Pedersen, 2025), readr (Wickham *et al.*, 2024), and spOccupancy (Doser *et al.*, 2022).

3 | Results

From the 16th of June 2025 to the 20th of August 2025, pheasants were detected on 86 site–survey occasions across the 45 camera trap sites, with pheasants recorded at 37.78% of sites (17/45). All camera traps were within 5.5 km from release pens (Figure 2I).

The relationship of primary interest, distance from the nearest release pen, had an estimated posterior mean indicating a mild positive effect $\beta_2 = 0.22$ [-0.83, 1.26] on pheasant occupancy, though we found no clear relationship given the wide credible intervals (Table 1, Fig 4A). This result appears to support the landscape reservoir hypothesis, and not the distance-decay hypothesis.

Table 1. Statistical summary of posterior distributions from the spatial occupancy model applied to the detection-non-detection data from the 45 camera traps. Parameter estimates for occurrence and detection probability are on the logit scale.

Occurrence				
Parameter	Mean	2.50%	50.00%	97.50%
β_0 (Intercept)	-0.74	-1.79	-0.72	0.14
β_1 (Arable area (%))	-0.50	-1.71	-0.49	0.61
β_2 (Distance from release pen (m))	0.22	-0.83	0.22	1.26
β_3 (Shrub density (%))	0.78	-0.26	0.76	1.97
β_4 (Woodland density (%))	-1.26	-2.70	-1.23	-0.03
β_5 (Canopy cover (%))	-0.26	-1.33	-0.24	0.67
Detection				
α_0 (Intercept)	-1.05	-2.02	-1.04	-0.12
α_1 (Days since project began)	0.21	-0.10	0.21	0.53
α_2 (Rain (mm))	0.08	-0.19	0.08	0.34
α_3 (Temperature (˚C))	-0.05	-0.34	-0.05	0.24
α_4 (Vegetation height (cm))	0.51	-0.24	0.51	1.27
α_5 (Open View)	0.54	-0.53	0.53	1.63
Spatial Covariance				
σ^2 (Spatial Variance)	2.92	0.21	0.98	19.68
ϕ (Spatial decay)	1.50	0.55	1.50	2.44
ν (Spatial smoothness)	1.49	0.55	1.49	2.44

Woodland density was the only site-level covariate where the 95% credible intervals did not overlap zero, however, the credible intervals were wide, and uncertainty remained high. All other site-level covariates had 95% credible intervals that overlapped zero, indicating no clear or directional effects on occupancy (Table 1). While trends were observed, these relationships were weak, uncertain or both. The predicted relationships for each site level covariate have been reproduced on the unstandardised covariate scales and ranges in Figure 4.

Predicted effects for detection covariates are omitted for brevity and instead included in the supplementary material (Figure 5.)

Model diagnostics support the reliability of the parameter estimates and indicate successful convergence of MCMC chains (see Supp Mat Table 2. for details). The $\hat{R}$ statistic was <1.05 (median = 1.00) for all parameters. The effective sample sizes across all chains (ESS) were all above 550 (median = 11,136, of 39,200 total retained iterations), indicating that the model produced a sufficient number of independent samples.

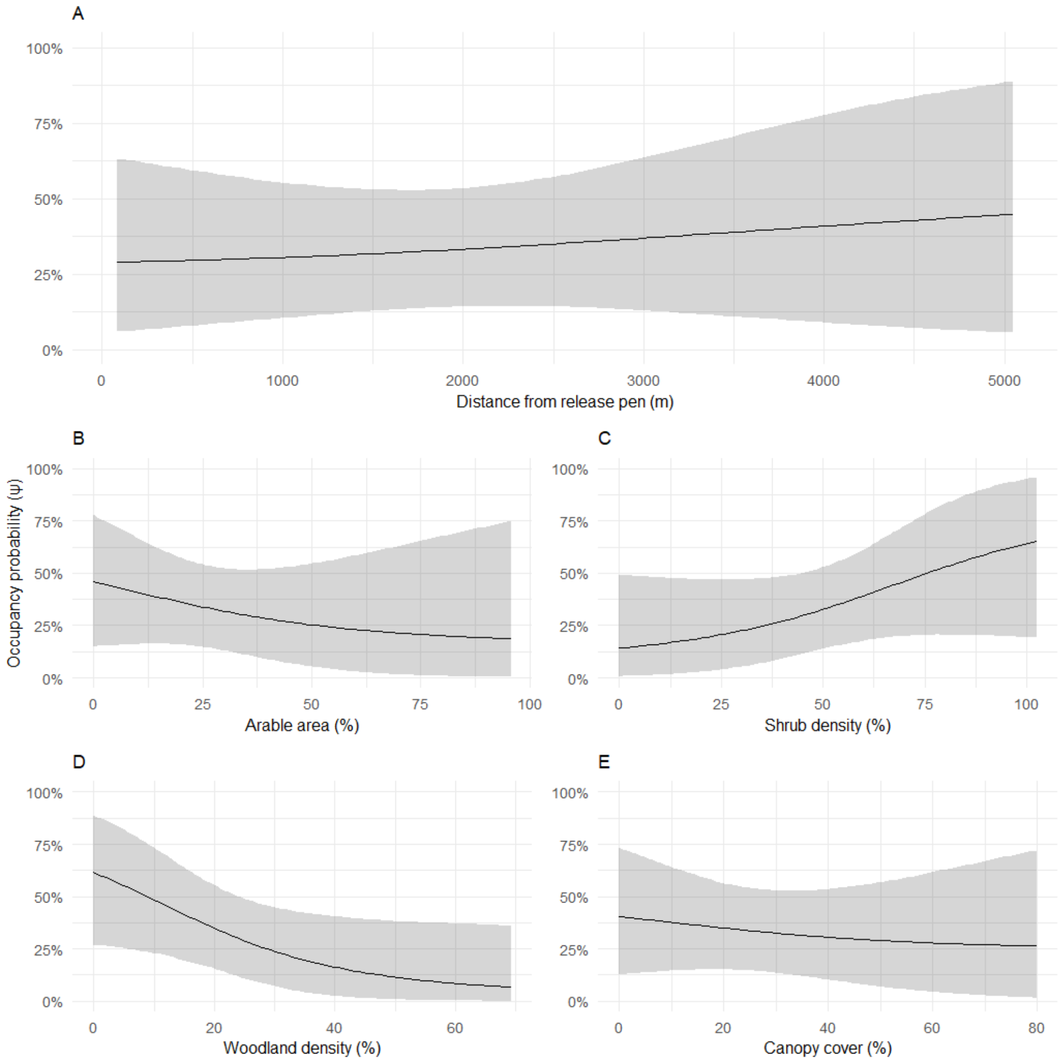

Figure 4. Predicted pheasant occupancy probability (ψ) across site-level covariates based on the fitted model. In each multiple, successive relationships are shown with all other covariates held at their mean values. Covariates are **A** distance from the nearest known release pen (m), **B** percentage of land attributed to arable area within the 300 m (28.27-ha) buffer around each camera, **C** percentage of land attributed as shrubland within the 300 m (28.27-ha) buffer, **D** percentage of land attributed to woodland within the 300 m (28.27-ha) buffer, and **E** canopy cover at the camera site.

This study is not free from limitations. Our sample size was relatively small, as only 45 camera traps were deployed over a limited field window, and placement was further constrained by where landowner permission could be obtained, limiting the spatial coverage and randomness of the sampling design. Covariates derived from remote-sensing data reflect habitat conditions at the time of data collection rather than during the camera-trap deployment period, meaning they may not fully capture habitat structure at the time camera traps were deployed. Several covariates estimated in the field were recorded by multiple surveyors, and inconsistencies between surveyors may have introduced additional variation. These constraints should be considered when drawing inferences from our results.

4 | Discussion

Our results provide no evidence that pheasant occupancy declines with increasing distance from release pens, offering little support for the distance-decay hypothesis. The estimated effect of distance was weakly positive ($\beta_2 = 0.22$), but the 95% credible interval ranged widely from -0.83 to 1.26, indicating a lack of any meaningful directional relationship. In practical terms, pheasants were just as likely to occur near release pens as at the farthest surveyed locations (ca. 5 km), with the model unable to detect any change in occupancy probability across the distance gradient. The observed spatial uniformity is inconsistent with the assumption that occupancy diminishes significantly beyond 500 m from the release pen and instead aligns with the landscape reservoir hypothesis: pheasant distribution appears shaped by overwinter persistence, reproduction, and multi-year accumulation rather than by short-term dispersal around release sites. Our model therefore suggests a broadly saturated landscape, contradicting the assumption that short-term post-release dispersal can be used as a proxy for ultimate pheasant occupancy.

Our findings are broadly consistent with previous estimates of site use. Sage et al. (2025) compared pheasant densities across sites categorised as ‘release’ and ‘non-release’. They reported that pheasants were present on land classed as non-release, situated several hundred metres to more than 2 km from release pens, at densities of 20–60 pheasants per km^2 in spring and <20 per km^2 in early summer, suggesting that pheasants may persist at distances exceeding the 500 m regulatory buffer. However, these estimates relied on a crude, non-model correction for detectability, which likely resulted in conservative abundance estimates (Sage et al. 2025). Even so, their reported densities remain substantially higher than natural galliform baselines, for example, non-invasive DNA sampling in a more natural Norwegian system estimated black grouse *(Lyrurus tetrix)* at 1.6–2.3 per km^2 and capercaillie *(Tetrao urogallus)* at 0.7 per km^2 (Brøseth, Kleven and Bevanger, 2025), suggesting that even the ‘low’ pheasant densities recorded on non-release land exceeded natural galliform baselines by an order of magnitude.

The particular setup of our study may provide estimates that more closely reflect the true spatial extent of pheasant occurrence. By accounting for imperfect detection and modelling distance from release pens as a continuous predictor, our analysis provides a more statistically robust assessment of pheasant site use than short-term telemetry studies (Turner, 2007), or methods relying on raw detections, categorical distance bands, and crude corrections (Sage et al., 2025). Our study area received substantially lower pheasant inputs than those examined by Sage et al. (2025), yet our model still estimated broadly uniform occupancy across the landscape, suggesting that pheasants can persist under comparatively modest reinforcement. However, because our model estimates the probability of site use rather than abundance, our results should not be interpreted as a measure of pheasant pressure, and it remains possible that the apparent persistence suggested by our model reflects ecologically insignificant pheasant densities.

While our results indicate that pheasant populations may persist in the wider landscape (ca. 5 km) surrounding release sites, the ecological relevance of our occupancy estimates remains limited without knowing the abundance of the persisting population, the threshold density at which ecological impacts begin to occur, and the releasing effort necessary to sustain the persisting population below that impact threshold. Until these knowledge gaps are addressed, we cannot determine whether reptiles distributed across the wider landscape are vulnerable to direct predation (Graitson & Taymans, 2022), whether predator–prey dynamics are altered through predator subsidisation (Pringle et al., 2019), or whether persistent pheasant populations act as a host network facilitating pathogen transmission into native bird communities or scavenging mammals far from release points (Tammiranta et al., 2023). Ecological impacts may emerge only at high pheasant densities, or even low densities may be sufficient to influence sensitive taxa or key ecological processes. We suggest that, until such thresholds are identified, the ecological risk associated with pheasant releasing cannot be fully understood.

In the absence of clear evidence on how such populations affect native biodiversity, a precautionary approach to releases near protected areas containing sensitive or threatened species is therefore warranted. A notable example is the Cairngorms National Park, where pheasant releasing occurs at multiple shoots and at levels exceeding those in our study area (Fletcher *et al.*, 2025). Given that our model suggested continuous occupancy under substantially lower release pressure, such levels of releasing in the Cairngorms suggest that pheasant presence may not remain localised around release pens. Several taxa within the park, including ground-nesting birds and native reptiles, could plausibly be susceptible to pheasant-mediated ecological effects.

However, the species of greatest concern within the Cairngorms National Park is the capercaillie *(Tetrao urogallus)*. The Scottish population is on the verge of extinction, with the Cairngorms representing a vital stronghold for the remaining birds (Moss, 2001). Capercaillie are already subject to multiple interacting pressures, and chick survival is tightly dependent on the availability of invertebrate prey (Picozzi, Moss and Kortland, 1999). Any pheasant-driven effects, such as predator subsidisation, pathogen transmission, or reductions in woodland invertebrate abundance documented for other Galliformes (Molnár et al., 2025), could therefore represent an additional pressure on a population already at critical risk. The acute vulnerability of the species means that even low-probability risks warrant serious consideration.

Identifying an effective and proportionate response to the ecological risks associated with pheasant releasing is inherently complex, in part because pheasant-driven effects on biodiversity operate across different spatial scales and through distinct mechanisms. Although our results suggest that fixed-distance buffers are unlikely to be effective at mitigating pheasant-driven ecological impacts, they do not point to a straightforward alternative. Even if pheasant presence could be entirely removed from sensitive areas that support critically vulnerable species, such measures would be expected to prevent only direct impacts. If pheasants persist at high densities in the surrounding landscape, indirect effects such as predator subsidisation and pathogen transmission are unlikely to be eliminated. As pheasant numbers decline, subsidised predators may switch to alternative prey, potentially extending indirect predation pressure into sensitive areas (Randa et al., 2009), while pathogens and parasites can be spread over long distances by mobile scavengers or infected birds, with the potential to expose sensitive or threatened species in adjacent landscapes (Tammiranta et al., 2023), regardless of immediate pheasant presence.

It remains possible that the only measure that would unequivocally eliminate pheasant-driven ecological impacts would be the cessation of pheasant releasing altogether. However, given the entrenched socio-economic and cultural value of pheasant shooting to stakeholders, such an approach is unlikely to be feasible in the near term (Crow and Baysha, 2013). In conservation conflicts, complete resolution is rarely achievable, and management efforts are more likely to succeed when they focus on mitigating impacts rather than attempting to eliminate conflict altogether (Redpath et al., 2013). While we cannot offer a definitive solution, our results suggest that future research should focus on quantifying the density and persistence of the landscape reservoir population implied by our model. This could be achieved using conventional abundance estimation methods, such as Random Encounter Model (REM), Camera Trap Distance Sampling (CTDS) (Palencia *et al.*, 2021) or Spatially Explicit Capture Recapture models (SECR) (Efford and Fewster, 2013). Integrating these estimates with density–impact models would facilitate the identification of thresholds at which pheasants begin to exert negative ecological effects and would reveal whether such impacts increase non-linearly with density, as is common in invasive species (Yokomizo et al., 2009). We suggest that improved evidence on the density and impacts of the landscape reservoir population could provide a clearer basis for management decisions than proximity-based restrictions alone. When considered alongside stakeholder interests, such evidence may support the development of mitigation strategies that reduce ecological risk while remaining acceptable to all affected parties.

5 | Conclusion

Our results suggest that pheasant occupancy does not follow a predictable decay with distance from release pens, but rather a surviving population persists across the wider landscape. This suggests the current 500 m buffer may be ineffective in safeguarding native biodiversity. We emphasise the need to move beyond proximity-based risk assessments and better establish the links between release magnitude and overwinter survival. Failure to do so risks exposing sensitive native species to direct predation, increased predation pressure from generalist predators, and exposure of wild populations to pathogens. Recognising pheasants as a persistent reservoir population rather than a localised release effect will be critical if future policy is to provide meaningful protection for vulnerable species and habitats. We acknowledge that adjusting policy to preserve biodiversity will not be straightforward, yet the complexity of the challenge must be weighed against the risk of inaction.

Acknowledgements

I would like to thank Deon Roos for his continuous support and guidance.

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