Burrowing Seabird Survey 2023/2024:

Population estimates of sooty shearwaters and white-chinned petrels on Kidney Island and Top Island, Falkland Islands

Report to Falkland Islands Government

Authors: A. Kuepfer[1], B. Lee[2], A. Stanworth[1]

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[1] Falklands Conservation, 41 Ross Road, Stanley, Falkand Islands

[2] Faculty of Life and Environmental Sciences, University of Iceland, Reykjavik, Iceland

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Falklands Conservation: Jubilee Villas, 41 Ross Road, Stanley, Falkland Islands.

Corresponding author: Amanda Kuepfer: SE@conservation.org.fk

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Registered Office: 2nd Floor, Regis House, 45 King William Street, London, EC4R 9AN

Telephone: +44 (0) 1767 693710, info@conservation.org.fk

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Summary

Background

The FIG Environment Department contracted Falklands Conservation to survey burrowing seabirds on Kidney, Cochon, Top and Bottom Islands during the breeding seasons of 2023/2024 and 2024/2025 to estimate the population size of sooty shearwaters (*Ardenna grisea*) and white-chinned petrels (*Procellaria aequinoctialis*), and to assess population trends where possible. The surveys aim to inform the review of the Falkland Islands Government (FIG) Stanley Tussac Grass Islands Management Plan 2018–2023, and fulfil international commitments under the Agreement on the Conservation of Albatrosses and Petrels (ACAP) to monitor and manage ACAP breeding sites. The current report outlines the method and results for the surveys undertaken on Kidney and Top Islands in 2023/2024, and provides associated management recommendations.

Method

To estimate the number of breeding pairs of sooty shearwater and white-chinned petrels, and assess their distribution patterns, we applied Bayesian hierarchical spatial-temporal models on occupancy corrected burrow counts.

Results

- **Kidney Island 2023/2024**
  - **Sooty shearwater** population size = **131,000 (95% CI: 95,000 – 176,000) breeding pairs**. This estimate is not notably different from 2016 (**123,000; 95% CI: 87,000 – 167,000 breeding pairs**), and gives no strong indication of an important increase or decrease in breeding pairs at this site.
  - **White-chinned petrel** population size = **331 (95% CI: 52 – 1043) breeding pairs**. This represents a baseline estimate using a model-based approach.
- **Top Island 2023/2024**
  - **Sooty shearwater** population size = **12,000 (95% CI: 7,000 – 19,000) breeding pairs**. This represents a baseline estimate.
  - **White-chinned petrel** population size = **199 (95% CI: 33 – 594) breeding pairs**. This represents a baseline estimate.

Management recommendations

Sooty shearwaters were found in greater numbers in areas with dense tussac over peat, while white-chinned petrels were most numerous in areas of tussac peat with higher moisture content. Additionally, like most burrowing seabirds, these species are susceptible to invasive mammals and other non-native species due to predation and habitat loss. Effective site management for sooty shearwater and white-chinned petrels should focus on preserving healthy tussac habitat, maintaining suitable peat condition for burrowing, and ensuring the continued absence of invasive mammals. Biosecurity efforts should further aim to minimise the risk of introducing invasive plants, invertebrates and pathogens. Additional monitoring efforts to further refine population estimates and track habitat availability could be considered.

Contents

Introduction

Kidney, Cochon, Top and Bottom Islands are four small, rodent-free islands located near Stanley, East Falklands (**Figure 1**). These islands support areas of dense tussac grass (*Poa flabellata*), and with invasive rodents eradicated in 2001 (Top and Bottom), they represent crucial habitats for native fauna, including internationally threatened sooty shearwaters (*Ardenna grisea*, IUCN Red List – *Near threatened*) and white-chinned petrels (*Procellaria aequinoctialis,* IUCN Red List – *Vulnerable*) (BirdLife International 2024a, b).

Kidney and Cochon Islands have been designated as a National Nature Reserve (NNR) and an Important Bird Area (IBA), with Kidney Island additionally recognised as a Key Biodiversity Area (KBA). Furthermore, Kidney, Top and Bottom Islands, represent breeding sites No. 80 and 81 under the Agreement on the Conservation of Albatrosses and Petrels (ACAP), to which the Falkland Islands Government (FIG) is a signatory party.

The four islands are crown-owned and managed by the FIG Environment Department, as detailed in the Stanley Tussac Grass Islands Management Plan 2018–2023. As part of the Management Plan's review, FIG contracted Falklands Conservation to provide updated and baseline population estimates of sooty shearwaters and white-chinned petrels on these islands. Given the international and national importance of these sites, and significance of the burrowing birds present, strategic monitoring is crucial to support ACAP commitments and has high conservation and management value.

The specific objectives of the survey were to: (1) Obtain baseline population abundance and distribution estimates of sooty shearwaters and white-chinned petrels on the islands where such data do not yet exist. (2) Assess population trends where previous comparable data exist. (3) Provide a species list of other birds seen during fieldwork. (4) Record non-native and invasive fauna and flora. (5) Provide management recommendations.

To ensure suitability for a long-term monitoring programme, the survey and analytical methods need to be repeatable, sufficiently flexible to withstand unforeseen circumstances and, where possible, provide data comparable to previous surveys (**Table 1A**). To increase flexibility and robustness against weather and boat availability, the surveys are spread across two seasons. The 2023/2024 season was the first season of this work, focusing on Kidney and Top Islands.

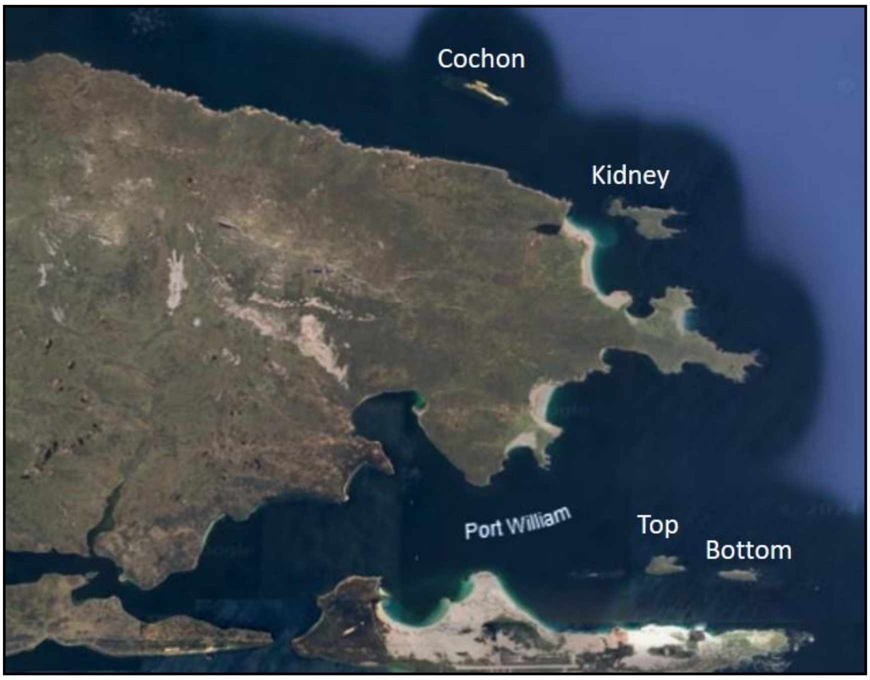

Figure 1 Location of the four "Tussac" islands (Cochon, Kidney, Top and Bottom) near Stanley, Falkland Islands. Source: Google Earth imagery.

Materials and methods

Data collection

All research was undertaken under FIG Research Licence No: R22/2023.

Seabird data

Data on sooty shearwaters and white-chinned petrel burrows were collected during surveys on Kidney and Top Islands conducted during the breeding season of 2023/2024 (hereafter 2023; **Table 1**).

Burrow density (the number of burrows per unit area) and burrow occupancy (whether or not a burrow is occupied by a breeding bird) are rarely consistent among colonies or areas (Whitehead et al. 2014), and therefore representative sampling is important to inform the model. As such, our sampling approach involved (1) a randomly projected grid of sampling plots (2.5-m radius; planar area = 19.63 m^2) onto the vegetated area of each island (after Clark et al. 2019), and (2) additional plots of the same size along the perimeter of the islands, specifically to capture the preferred habitat of more patchily distributed white-chinned petrels (after Poncet et al. 2012; **Table 1**, **Figure 2**, **Figure 3**).

Assessments of burrow occupancy are ideally undertaken at the end of the main laying period when most breeding birds are present and few nesting failures have occurred (sooty shearwaters: ~24 November on Kidney Island (Hedd et al. 2014); white-chinned petrels: ~22 November in South Georgia (Hall 1987)). The majority of fieldwork was conducted during early to mid-incubation, between 26 November and 10 December 2023 (**Table 1**). Additional density plots (burrow counts only, which are less time-sensitive) were surveyed on Kidney Island during mid-chick rearing (mid-February). The islands were accessed using Sullivan launches with permission from FIG. Key constraints for site access included weather conditions and launch availability.

The approximate (±3.65 m) location of each pre-defined plot was found using a GPS (Model Garmin GPSMAP 64s). To assess burrow density for each species, we counted the number of species-specific burrows in each plot. This involved two or more fieldworkers systematically searching for burrows in peat-covered ground by moving in a circular fashion around the plot, using a 2.5-m cord tied to a cane at the centre to delimit the outer circumference of the plot. Burrows measuring less than a forearm's length were not included, as these were not considered as viable burrows for either species (see also Clark et al. 2019). We did not assess burrow detection probability; however, given the relatively small size of the plot and intensity of search effort, we assumed that all burrow entrances were detected.

To derive a population estimate from burrow counts, it is imperative to account for burrow occupancy rates. For sooty shearwaters, burrow occupancy was assessed at a subset of survey plots, where the occupation status of five random nests was determined as "occupied" (1) or "unoccupied" (0). For white-chinned petrels, where burrow density is much lower and patchier, the occupancy status was assessed at all burrows found within study plots. Burrow occupancy was determined using audio-playback (successful only with white-chinned petrels) and burrowscopes (Teslong Endoscope NTS430).

To directly compare Kidney Island sooty shearwater population estimates from 2023 with those from 2016/2017 within a single analytical framework, T.J. Clark and E. Wakefield kindly provided the raw data collected during that period (hereafter 2016; Wakefield et al. 2017, Clark et al. 2019; see Discussion for details).

Table 1 Summary of data collection on Kidney and Top Island in 2023/2024.

	Kidney Island	Top Island
Survey dates	26/11/2023, 27/11/2023; 16/02/2024[1]	09/12/2023, 10/12/2023
Total area	0.32 km^2	0.12 km^2
Vegetated area	~0.25 km^2	~0.09 km^2
SSW density; occupancy plots (n)	65; 18	86; 31
WCP density; occupancy plots (n)	94; 94	86; 86
Survey effort	168 hrs (2.5 days, 4 fieldworkers + 1 day, 8 fieldworkers)	80 hrs (2 days, 4 fieldworkers)

[1] Burrow density counts only. SSW = Sooty shearwater; WCP = White-chinned petrel.

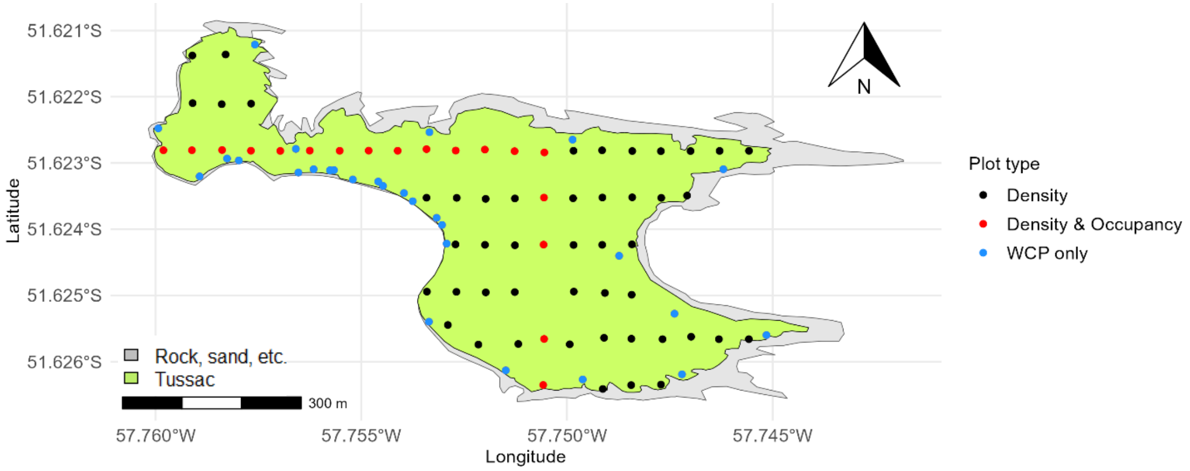

Figure 2 The position of the 2023/2024 survey plots on Kidney Island. Sooty shearwater (SSW) burrows were surveyed only at red and black plots, and represent the approximate area where they were surveyed by Clark et al. 2019 (see Figure 1A). White-chinned petrels (WCP) were surveyed in all SSW plots, as well as in additional plots along the island perimeter (in blue). For WCP, occupancy was assessed in all plots where burrows were found.

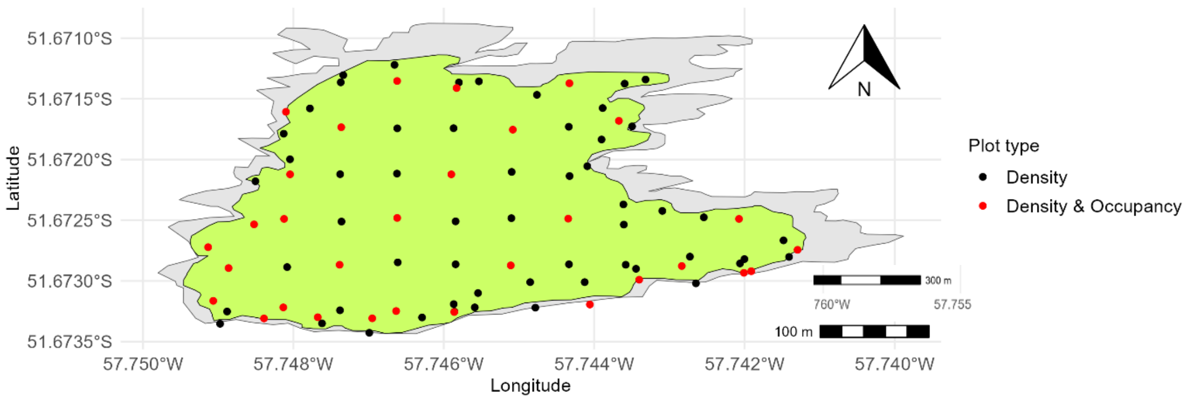

Figure 3 The position of the 2023/2024 survey plots on Top Island. Both sooty shearwaters and white-chinned petrels were surveyed at all plots. For WCP, occupancy was assessed in all plots where burrows were found.

Habitat indices

Suitable breeding habitat can be limited by factors such as vegetation, terrain and competition with other species. Therefore, quantifying habitat availability is crucial for informing model predictions. For each plot, we collected habitat indices likely to affect the breeding preferences of the two species (see Clark et al. 2019; **Table 2A**).

In situ, we quantified soil moisture (1 – 4, from dry to standing water), average tussac height (cm), tussac density (%), presence of bare rock (1/0), and presence of other fauna (1/0), specifically referring to pinnipeds and other birds.

Additionally, we calculated six remotely-sensed habitat indices using the R statistical software ("raster" R package, Hijmans 2023) or QGIS (v 3.34.00): Distance to shore, elevation, aspect, slope, the Normalised Difference Vegetation Index (NDVI) and the Normalised Difference Moisture Index (NDMI).

Distance to shore was calculated at 1-m resolution using a shapefile polygon outlining each island. Elevation, aspect and slope were derived from a digital elevation model (DEM) at a 1 arc second resolution (~53-m resolution in the Falkland Islands) retrieved from the NASA Earthdata search portal (https://search.earthdata.nasa.gov/).

For ground classification, we used a level 1C multi-spectral image captured by the MSI instrument on board the Sentinel 2B satellite, corresponding to the nearest survey dates without cloud cover (17/09/2016; 26/09/2023), and obtained from the Copernicus Open Access Hub (https://dataspace.copernicus.eu/). NDVI was calculated at 10-m resolution (after Stokes et al. 2021), and NDMI at 20-m resolution (see Appendix for details). NDVI, a proxy for photosynthetically active vegetation cover, is commonly used as an indicator of vegetation health and growth (Huang et al. 2021). NDMI is used to determine vegetation water content (USGS 2024).

Other fauna and flora

During field days, we opportunistically recorded other bird species and pinnipeds seen outside study plots, as well as signs of invasive plants and mammals. However, it is important to note that we did not systematically monitor for these.

Data analysis

Unless specified otherwise, all data processing and analyses were conducted using R software (R Core Team 2023). Explanatory variables were explored for outliers, and the presence of collinearity assessed visually, as well as through Pearson correlation coefficients and variance inflation factors (VIF) (Zuur et al. 2010; see Appendix). Evidence of collinearity was present among NDVI, NDMI and tussac density, as well as between elevation and distance to shore (see Appendix). As such only one could be used within each model. Distance to Shore, although generally found to be a variable of importance, showed a relationship with the spatial random field (SRF) in Models 2 and 3, and was therefore excluded from these models. To reduce numerical issues and to compare effect size of explanatory covariates, continuous variables were standardised prior to model fitting (Zuur et al. 2017). To maximise our dataset, models were run on all available data, with Island and Year included as explanatory variables as appropriate.

To estimate the breeding pair numbers of sooty shearwaters and white-chinned petrels, we applied Bayesian hierarchical modelling using the integrated nested Laplace approximation (INLA) approach, implemented using the R-INLA package (http://www.r_inla.org; Martins et al. 2013). INLA is a flexible, computationally efficient method for a large class of latent Gaussian models (including spatial-temporal models) in a Bayesian framework (Rue et al. 2009).

Starting from the full model that contained all explanatory variables, the most parsimonious model was selected based on the lowest Deviance Information Criterion (DIC; Spiegelhalter 2002) and Widely Applicable Information Criterion (WAIC; Watanabe 2010). For model validation, we conducted a simulation study in which 10,000 sets of fitted values were generated from our model's posterior distribution and compared to the observed data (Zuur et al. 2017, Zuur & Ieno 2018), as outlined by Lee et al. (2021). The fit was analysed by comparing the observed versus fitted values, examining patterns in residuals, evaluating model dispersion, and checking for zero-inflation (see Appendix). Remaining auto-correlation was assessed using variograms. Credible intervals (CI) were set at 95% throughout. For annual and site-based predictions, continuous explanatory variables were held at their mean and Fauna presence was set to 0.

Sooty shearwater breeding pair estimates

For sooty shearwaters, where only a subset of burrows was assessed for occupancy, the first step was to estimate occupancy rates, to correct burrow counts accordingly. We used a Bernoulli distribution to analyse burrow occupancy, where the response variable Occ_i, indicates whether burrow *i* is occupied (1) or not (0) (Model 1). To account for clustering within individual plots, we included Plot Id as a random effect (**Table 2**). Model 1 is formulated as follows:

$$Occ_i \sim Bernoulli\ (\pi_i)$$

$$E\ (Occ_i) = \pi_i \quad \text{and} \quad var(Occ_i) = \pi_i \times (1 - \pi_i)$$

$$logit(\pi_i) = \beta_1 + \beta_2 \times Covariate1_i + \beta_3 \times Covariate2_i + \cdots + \beta_{n+1} \times Covariate_{ni} + u_{Plot\_id}$$

Here, Occ_i is assumed to be Bernoulli distributed with probability π_i. The mean and variance of Occ_i are π_i and $\pi_i \times (1 - \pi_i)$, respectively. We model π_i as a function of covariates. The parameter β_1 represents the intercept, and subsequent coefficients (β_2, β_3,…, β_{n+1}) quantify the effect of the explanatory covariates (see **Table 2**). To ensure that the fitted probabilities are always between 0 and 1 we use the logistic link function (Zuur et al. 2010). As occupancy data were collected differently in 2016 (Clark et al. 2019; see Discussion), we assigned a 1 (occupied) to all burrows with a ≥50% probability of being occupied, and a 0 (unoccupied) to all burrows with <50% probability of being occupied. Overall, model validation showed that 69% of occupancy was predicted correctly (correct predictions: 0s: 22%, 1s: 92%). The final model was used to predict a probability of occupancy for each burrow counted.

To estimate sooty shearwater breeding pair numbers, we applied a Bayesian species distribution model (BSDM; Model 2; **Table 2**). This approach allows the spatial and temporal components of the data to be incorporated as random variables, reducing the influence of these on the effects of other variables (Martinez-Minaya et al. 2018). The method has been used elsewhere to calculate reliable estimates of seabird abundance and associated uncertainty (e.g. Soriano-Redondo et al. 2019, Vilela et al. 2021), and to assess their spatial distribution for the purpose of management (e.g. Sadykova et al. 2017, Sarzo et al. 2023).

The breeding pair count, *Num*, for each plot was determined by rounding the product of the total number of burrows counted at the plot and the estimated proportion of occupied burrow (π) derived from Model 1. Spatial correlated random effect terms were included in the modelling framework using a progressive Stochastic Partial Differential Equations (SPDE) approach (Lindgren et al. 2011). In this approach, a continuously indexed Gaussian field was approximated with a Matérn covariance function by a Gaussian Markov random field (GMRF). A Delaunay triangulation which consists of a dense triangular grid is constructed on the spatial domain. This grid (mesh) forms the structure on which the representation of the field is based (see **Figure 5A**). To avoid increased variance occurring at the borders of the latent field (an edge effect due to sampling plots occurring at the edge of the island), the mesh was extended to the entire island beyond the boundary of the tussac-covered area (Zuur et al. 2017). The expected values of bird abundance (*Num*) at each sampling plot *i*, during the defined time period *t* were related to covariates based on the zero-inflated Poisson (ZIP) model equation:

$$Num_{it} \sim ZIP\ (\mu_{it}, \pi)$$

$$log(\mu_{it}) = \beta_1 + \beta_2 \times Covariate1_{it} + \beta_3 \times Covariate2_{it} + \cdots + \beta_{n+1} \times Covariate_{nit} + f(Covariate_{it}) + w_{it}$$

$$log(\pi) = \gamma_0$$

where π is the probability of 'false' zeros, and μ is the mean abundance of sooty shearwater breeding pairs (including the 'true zeros'). The parameter β_1 represents the intercept, and the following betas represent the coefficients that quantify the effect of the explanatory covariates (**Table 2**). Smooth term *f*() was fitted to nonlinear explanatory covariates represented by a cubic regression spline with 4 knots to describe biologically realistic response terms. The term *w* represents the spatial (temporal) correlated random effects (SRF) for the model. This term is a latent variable that changes in space and time, thus capturing any spatial and temporal patterns that are not already explained by the covariates (Zuur et al. 2017, Zuur & Ieno 2018). In this model, the extent of correlation in the SRF is reflected in the continuous temporal correlation hyperparameter ρ, with values extending from 0 to 1. This allows for the quantification of changes in the distribution among seasons as independent (ρ = 0 – 0.33), intermediate (ρ = 0.34 – 0.67) or persistent (ρ = 0.67 – 1).

Penalised complexity priors (PC priors) were imposed on the model range and marginal standard deviation hyperparameters of the SRF (Fuglstad et al. 2019). Specifically, these were set when defining the SPDE as follows: the probability that the spatial effect range was smaller than 100 m was 0.01, and the probability that the spatial effect standard deviation was greater than 0.2 was <0.001. Default priors were assigned for all fixed effect parameters, which are approximations of non-informative priors designed to have little influence on the posterior distribution.

White-chinned petrel breeding pair estimates

For estimates of white-chinned petrel breeding pairs, given all burrows were assessed for occupancy, we applied a BSDM directly to the number of occupied burrows encountered (*Num*) (Model 3). We used the same mesh as for Model 2 but defined a persistent SPDE, as no annual comparisons was conducted for this species. The expected values of breeding pair numbers (*Num*) at each sampling plot *i* were related to covariates based on the zero-inflated Poisson (ZIP) model equation:

$$Num_i \sim ZIP\ (\mu_i, \pi)$$

$$log(\mu_i) = \beta_1 + \beta_2 \times Covariate1_i + \beta_3 \times Covariate2_i + \cdots + \beta_{n+1} \times Covariate_{ni} + w_i$$

$$log(\pi) = \gamma_0$$

where π is the probability of 'false' zeros, and μ is the mean abundance of white-chinned petrel breeding pairs (including the 'true zeros'). The parameter β_1 represents the intercept, and the following betas represent the coefficients that quantify the effect of the explanatory covariates (**Table 2**). The term *w* represents the SRF for the model.

PC priors defining the SPDE specified that the probability that the spatial effect range was smaller than 100 m was 0.01, and the probability that the spatial effect standard deviation was greater than 1 was <0.001. Default priors were assigned for all fixed effect parameters.

Results

Kidney Island

Sooty shearwaters

Burrow counts and occupancy rate

Observed mean (± SE) burrow density of sooty shearwaters on Kidney Island in 2023 was 0.93 ± 0.08 burrows/m^2 (range: 0 – 2.45 burrows/m^2, n = 65; **Figure 4**) which is substantially higher compared to observed burrow counts in 2016 (0.60 ± 0.06 burrows/m^2, range: 0 – 1.73 burrows/m^2, n = 66). Burrows were found predominantly in the northwest and along the northeast coast (**Figure 4**).

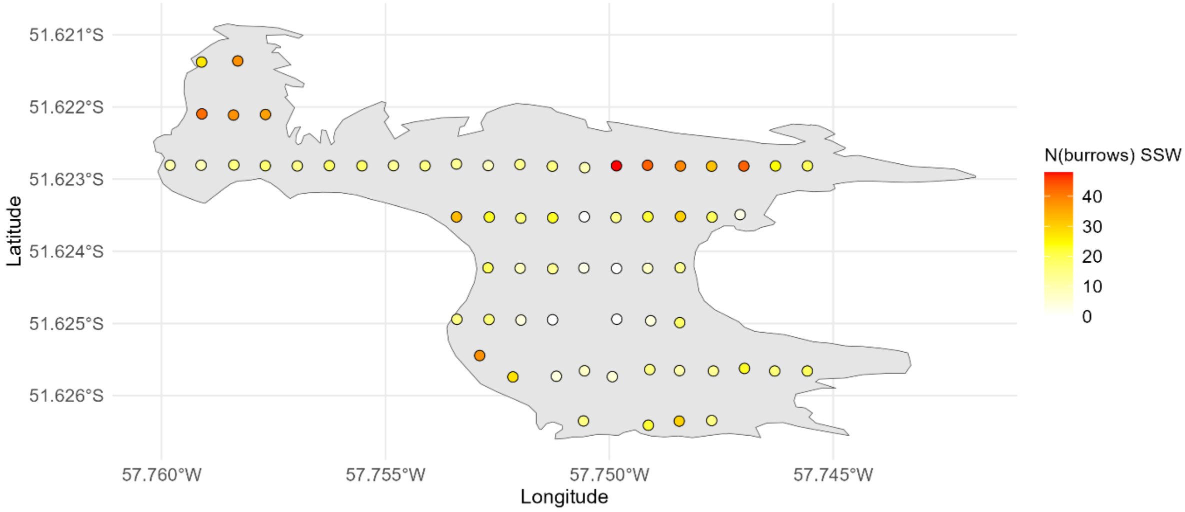

Figure 4 Sooty shearwater burrow numbers at each survey plot at Kidney Island in 2023.

Observed occupancy rate was 0.83 ± 0.05 in 2016 and 0.65 ± 0.06 in 2023. Predicted occupancy rates, estimated using Model 1, found Year to be a variable of importance with lower rates in 2023 (0.63 ± 0.01) compared to 2016 (0.80 ± 0.01; **Table 2**). Some caution is warranted with this result, given the difference in data collection approach for occupancy between years. Fauna was also identified as a variable of importance, with lower burrow occupancy predicted in areas where other birds or pinnipeds are present (**Table 2**). Other birds and pinnipeds were recorded in 27/131 plots across the two years, and typically involved South American sea lions (*Otaria flavescens,* 86% of plots where other fauna was recorded) but also Magellanic penguins (*Spheniscus magelanicus,* 7%) and white-chinned petrels (3.5%).

Breeding pair estimate

A total of 122,566 (CI: 87,154 – 166,991) sooty shearwater breeding pairs were estimated to occur on Kidney Island in 2016, and 130,999 (CI: 94,938 – 175,735) breeding pairs in 2023 (Model 2). These results do not suggest a substantial change in the number of breeding pairs at this site between these two breeding seasons (**Figure 5**; **Table 2**).

Breeding pair numbers were positively influenced by tussac density (**Table 2**). Slope and elevation had a non-linear effect on breeding pair numbers with a distinct positive effect at 8-12 m above sea level and over a moderate to high slope (10-20 degrees; **Figure 6**). Fauna was also a variable of importance, with lower breeding pair numbers predicted in areas used by other birds or pinnipeds (**Table 2**).

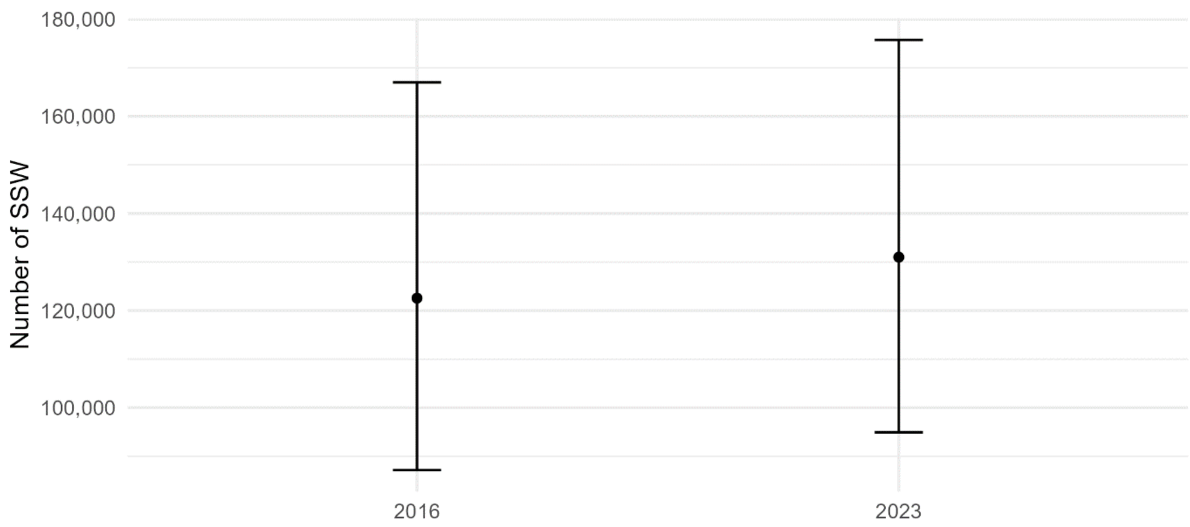

Figure 5 Estimated breeding pair numbers (mean ± 95% CI) of sooty shearwater (SSW) at Kidney Island in the seasons of 2016/2017 and 2023/2024.

Table 2 Summary of model outputs. CI = Credible intervals. ZPP = zero-probability parameter for the zero-inflated Poisson models. Fixed effects variables of importance are in **bold.** Partial effects of smooth functions are presented in **Figure 6**.

Model	Effects	Variables	Mean	Stdev	25% CI	97.5% CI
Model 1	Fixed	Intercept	1.841	0.314	1.225	2.457
		fYear2023	**-1.102**	**0.395**	**-1.877**	**-0.326**
		IslandTop	-0.356	0.378	-1.098	0.385
		Fauna1	**-0.766**	**0.333**	**-1.418**	**-0.114**
		Dist to Shore	-0.265	0.151	-0.561	0.030
		Tuss_D	-0.240	0.144	-0.521	0.042
	Random	Sigma for Plot_id	0.010			
Model 2	Fixed	Intercept	2.309	0.161	1.989	2.623
		fYear2023	0.005	0.177	-0.347	0.356
		IslandTop	**-1.443**	**0.292**	**-2.021**	**-0.869**
		Fauna1	**-0.421**	**0.095**	**-0.607**	**-0.234**
		Tuss_D	**0.233**	**0.052**	**0.131**	**0.335**
	Random	ZPP	0.075	0.023	0.037	0.128
		Range	0.185	0.026	0.139	0.242
		Stdev for w2	0.326	0.034	0.265	0.398
		GroupRho for w2	0.382	0.179	-0.002	0.692
Model 3	Fixed effects	Intercept	-3.921	0.761	-5.49	-2.495
		IslandTop	0.552	0.859	-1.146	2.268
		Soil	**0.536**	**0.159**	**0.224**	**0.850**
		DEM	**-1.876**	**0.508**	**-2.900**	**-0.900**
	Random	ZPP	0.061	0.016	0.036	0.098
		Range for w1	0.146	0.041	0.084	0.244
		Stdev for w1	0.871	0.169	0.579	1.241

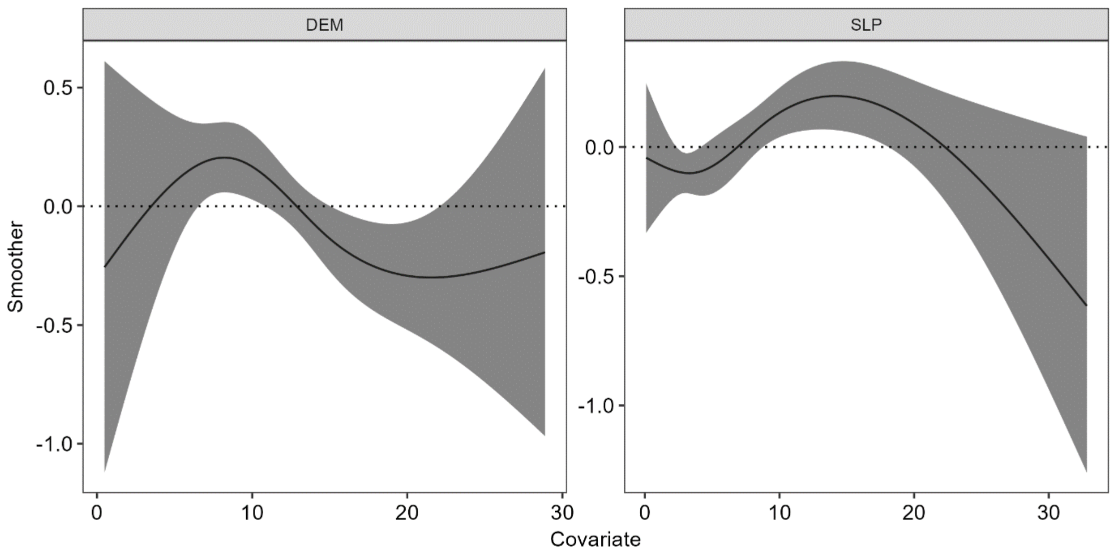

Figure 6 Partial effect of elevation (DEM) and slope (SLP) on sooty shearwater breeding pair abundance, as estimated in Model 2. The estimated effect is presented as smooth functions (cubic regression spline) with 95% credible intervals.

The SRF consistently highlights the northwest of the island as an area of high sooty shearwater breeding pair density over the two years, although some shifts in distribution are apparent (**Figure 7**). This was reflected in the temporal correlation parameter ρ, suggesting intermediate levels of change in the distribution between the two seasons (ρ = 0.38; **Table 2**). Notably, an area in the northeast has shown an increase in the predicted sooty shearwater distribution in 2023. A persistent area of lower abundance that occurs in the central hump of the southern extent of Kidney Island has become constrained in 2023 relative to 2016. Minor areas of increased breeding pair abundance (hotspots) are identified along the southern and south-western coast of Kidney Island during 2023.

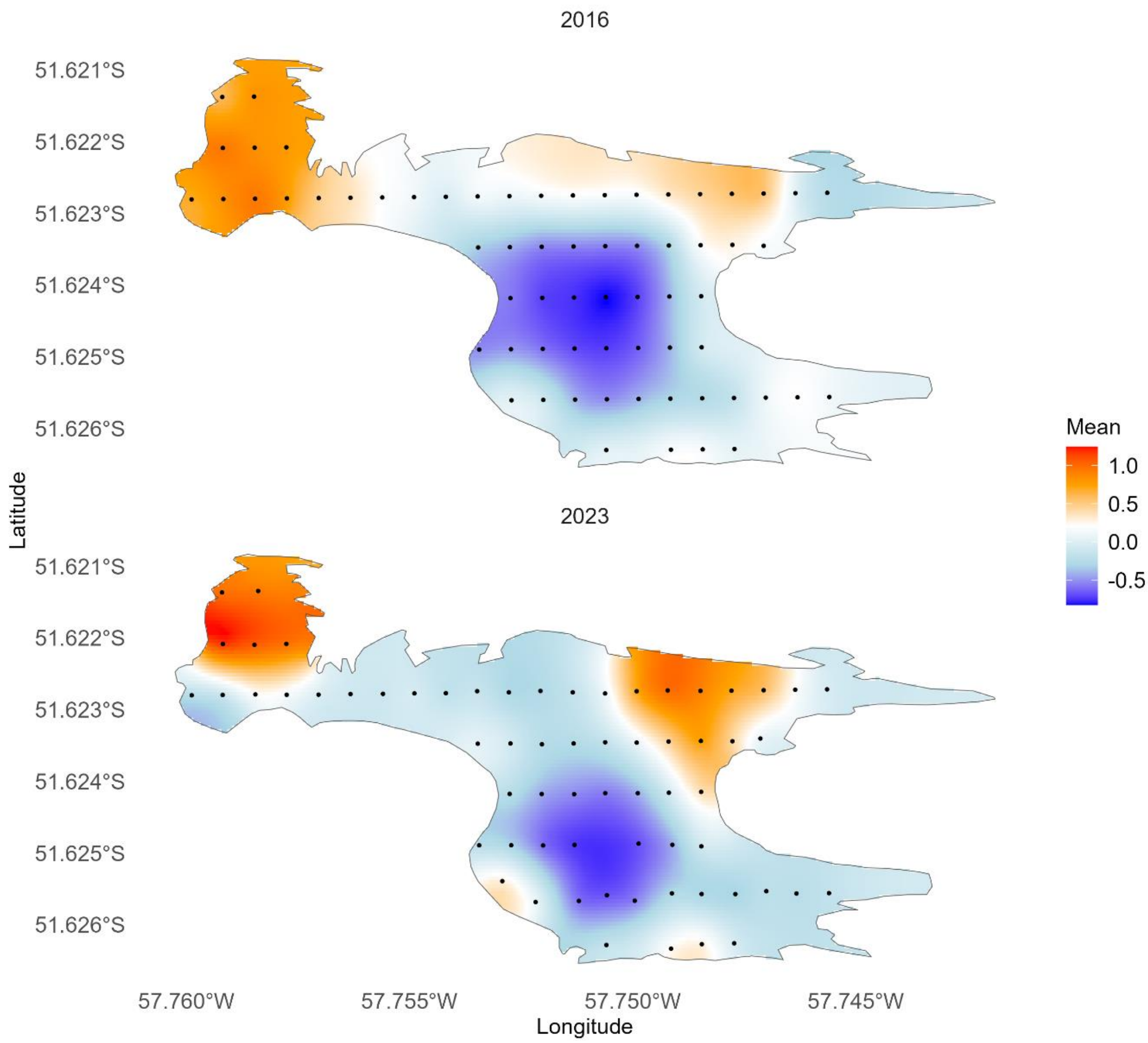

Figure 7 The posterior mean for the spatial random field of Model 2 with progressive spatial-temporal correlation for Year, demonstrating the partial effect of space on number of sooty shearwater breeding pairs across Kidney Island.

White-chinned petrels

Burrow counts and occupancy rate

We encountered 35 burrows within 95 plots, of which 24 (69%) were occupied. This equates to a mean (± SE) density of 0.01 ± 0.004 occupied burrows/m^2 (range: 0 – 0.20). Burrows were predominantly concentrated along the landing bay, although three unoccupied burrows were found in the southeast of the island in February during sooty shearwater burrow density counts (**Figure 8**).

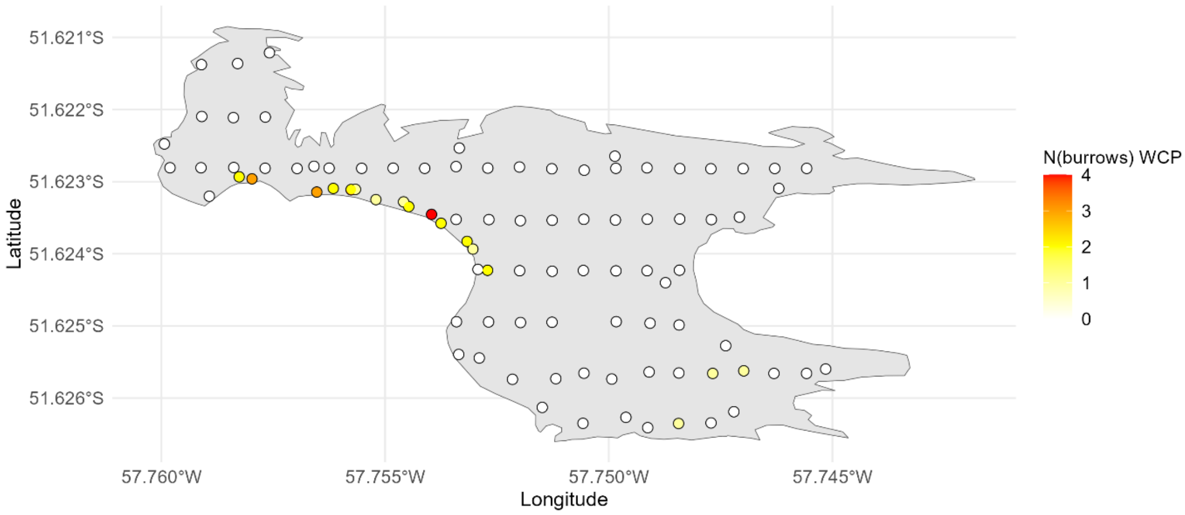

Figure 8 White-chinned petrel burrow numbers at each survey plot at Kidney Island in 2023.

Breeding pair estimate

A total of 331 (CI: 52 – 1043) white-chinned petrel breeding pairs were estimated on Kidney Island in 2023 (Model 3). Variables of importance were soil moisture and elevation, with higher bird abundance predicted in wetter areas and at lower elevation (**Table 2**). The SRF highlights the highest bird distribution along the landing bay. Model predictions indicated relatively high range and standard deviation parameters for the SRF (**Table 2**). This suggests the need for a higher resolution in survey effort within the area of burrow occurrence, to better inform model predictions.

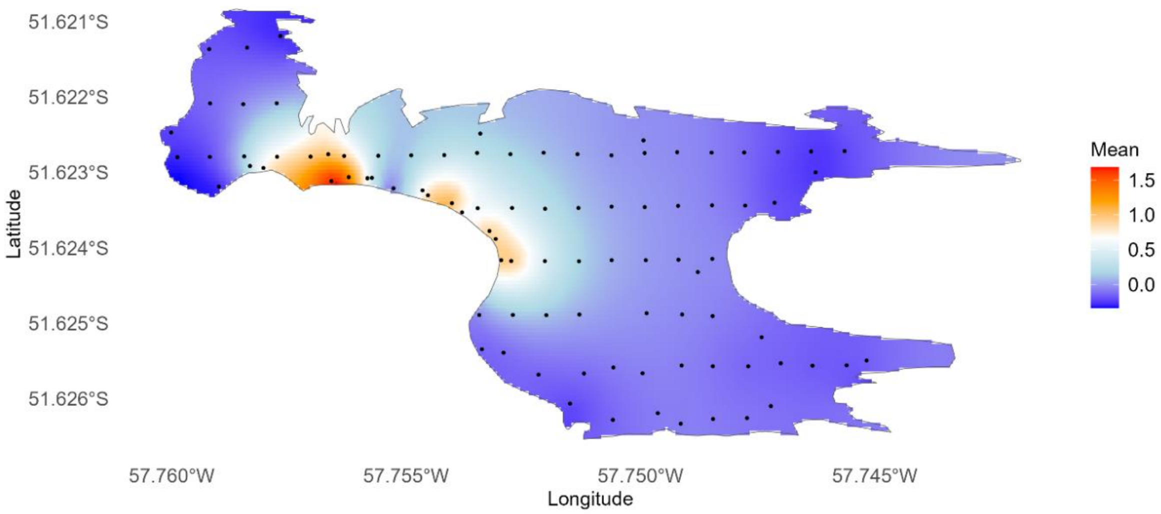

Figure 9 The posterior mean for the spatial random field of Model 3 with persistent spatial-temporal correlation, demonstrating the partial effect of space on number of white-chinned petrel breeding pairs across Kidney Island.

Other fauna and flora

Additional birds and pinnipeds recorded during surveys are summarised in **Table 3**. No invasive fauna or flora were detected at Kidney Island, although we did not systematically search for these. While not considered invasive, some persistent non-native prickly sow-thistles (*Sonchus asper*) have previously been recorded on the north coast of Kidney in a few places along and between the edges of the southern rockhopper colony (S. Poncet pers. comm.).

Table 3 Birds and pinnipeds recorded at Kidney and Top Islands during burrowing seabird surveys in 2023/2024.

Species	Latin name	Kidney	Top	Comment
Cobb’s wren	*Troglodytes cobbi*	✓		
Falkland steamer duck	*Tachyeres brachypterus*	✓	✓	
Great shearwater	*Puffinus gravis*		✓	Found in burrow
Grey-backed storm petrel	*Garrodia nereis*	✓		Carcass found
Kelp goose	*Chloephaga hybrida*		✓	Confirmed breeding
Magellanic oystercatcher	*Haematopus leucopodus*	✓	✓	
Magellanic penguin	*Spheniscus magellanicus*		✓	Confirmed breeding
Rock shag	*Phalacrocorax magellanicus*	✓		Juvenile
Sooty shearwater	*Ardenna grisea*	✓	✓	Confirmed breeding
Southern rockhopper penguin	*Eudyptes chrysocome*	✓		Confirmed breeding
Turkey vulture	*Cathartes aura*		✓	
Tussac bird	*Cinclodes antarcticus*	✓	✓	
Unidentified storm petrel			✓	Found in burrow

White-chinned petrel	*Procellaria aequinoctialis*	✓	✓	Confirmed breeding
South American sea lion	*Otaria flavescens*	✓	✓	

Top Island

Sooty shearwaters

Burrow counts and occupancy rate

Observed mean (±SE) burrow density of sooty shearwaters was 0.2 ± 0.02 burrows/m^2 (range: 0 – 0.92 burrows/m^2, n = 86). Almost no burrows were found within the central area of the island where the tussac was sparsely scattered and small fern were abundant (**Figure 10**).

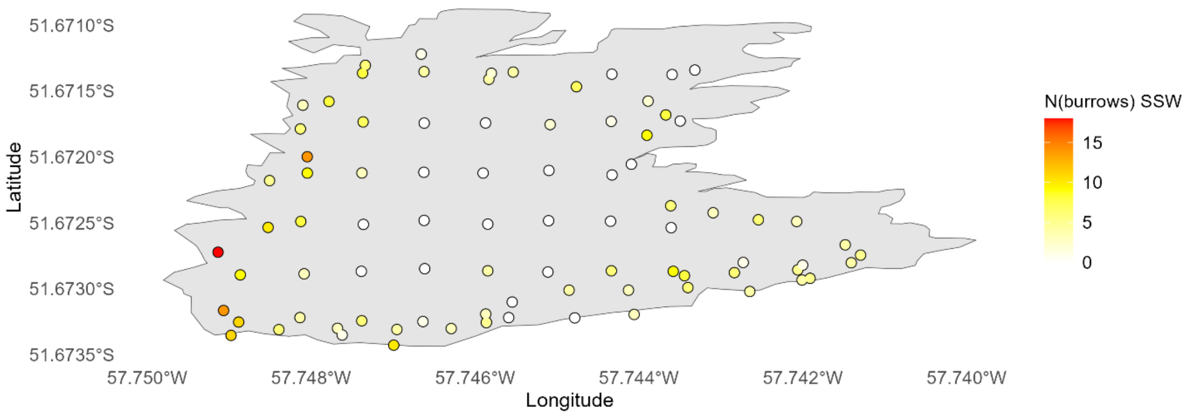

Figure 10 Sooty shearwater burrow numbers at each survey plot at Top Island in 2023.

Mean (± SE) observed occupancy was 0.54 ± 0.05, and the estimated occupancy across the island based on Model 1 was 0.57 ± 0.01. As for Kidney Island, predicted burrow occupancy was lower in areas used by other birds and pinnipeds (**Table 2**). Other birds and pinnipeds were recorded in 48/86 plots and predominantly involved sea lions (56% of plots where other fauna was recorded), Magellanic penguins (35%), white-chinned petrels (23%), as well as kelp geese (*Chloephaga hybrida*) and turkey vultures (*Cathartes aura*) (12%).

Breeding pair estimate

Model 2 estimated 11,886 (CI: 6,906 – 19,053) sooty shearwater breeding pairs on Top Island in 2023 (**Figure 11**). As on Kidney, predicted bird numbers were higher in areas of increased tussac density, and lower in areas used by other birds and pinnipeds (**Table 2**). The spatial random field indicates that the highest predicted distribution of sooty shearwaters occurs along the west of the island (**Figure 11**). Isolated areas of high breeding pair abundance also occur as hotspots along the south coast.

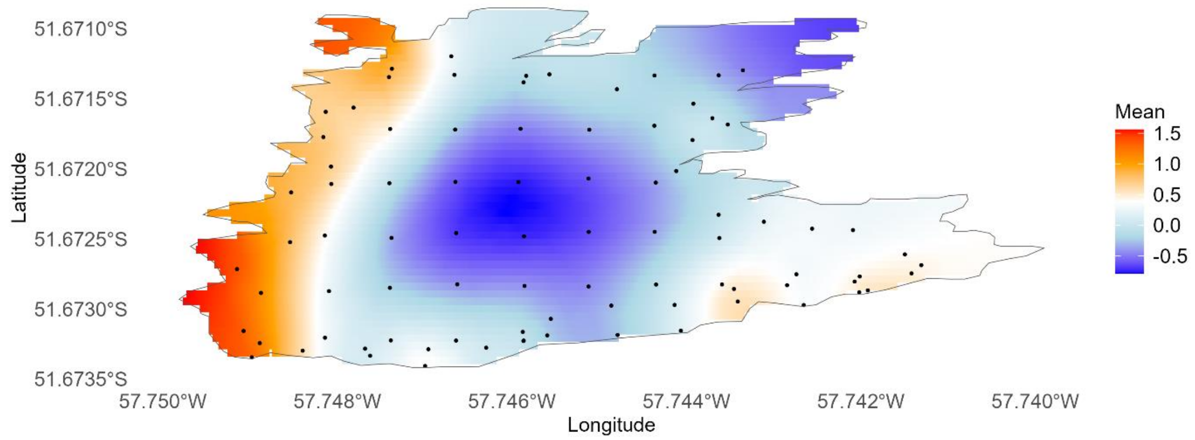

Figure 11 The posterior mean for the spatial random field of Model 2, demonstrating the partial effect of space on number of sooty shearwater breeding pairs across Top Island.

White-chinned petrels

Burrow counts and occupancy rate

At Top Island, we found 37 white-chinned petrel burrows within 86 plots, of which 24 (65%) were occupied. This represents 0.01 ± 0.005 occupied white-chinned petrel burrows/m^2 (range: 0 – 0.20 burrows/m^2). All burrows were located along the southern shore of the island.

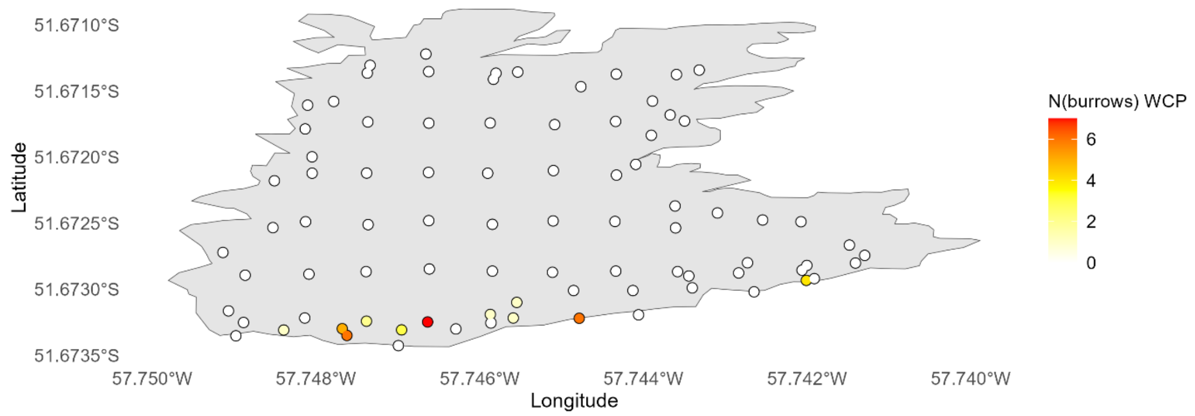

Figure 12 White-chinned petrel burrow numbers at each survey plot at Top Island in 2023.

Breeding pair estimate

Model 3 predicted 199 (CI: 33 – 594) white-chinned petrel breeding pairs at Top Island in 2023. The SRF highlights the highest predicted distribution in the south of this island, although the relatively large range and standard deviation suggest the need for a higher resolution of survey effort along this area to better inform model predictions (**Table 2**, **Figure 13**).

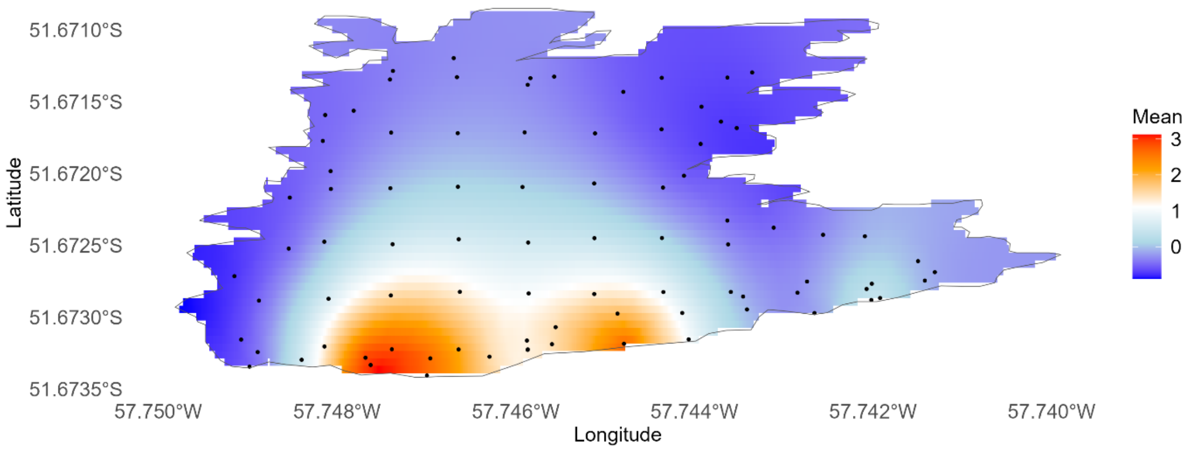

Figure 13 The posterior mean for the spatial random field of Model 3 with persistent spatial-temporal correlation, demonstrating the partial effect of space on number of white-chinned petrel breeding pairs across Top Island.

Other fauna and flora

Additional birds and pinnipeds recorded during surveys are summarised in **Table 3**. No invasive fauna or flora were detected at Top Island, although we did not systematically search for these. While not considered invasive, several non-native grasses (Meadow-grass (*Poa pratensis*), bent (*Agrostis* sp.), sheep's sorrel (*Rumex acetosella*), heath groundsel (*Senecio sylvaticus*), common groundsel (*S. vulgaris*) and chickweed (*Stellaria media*)) have previously been recorded on Top Island (S. Poncet pers. comm.).

Discussion

The current study provides up-to-date population estimates for sooty shearwaters and white-chinned petrels on Kidney and Top Islands using a spatial-temporal modelling approach. This approach identified significant patterns in the data, allowing us to make and compare species abundance predictions in space and time. The results serve to inform the management of these sites for these species in line with national and international commitments. It is imperative that comparisons of our findings with previous estimates are made with due consideration of any methodological differences (**Table 1A**; discussed below). In the following sections, we present our findings, highlight key considerations, and provide management recommendations.

Survey findings

Sooty shearwaters

Our 2016 and 2023 breeding pair estimates of sooty shearwater at **Kidney Island (122,566 (CI: 87,154 – 166,991) and 130,999 (CI: 94,938 – 175,735)**, respectively), do not provide evidence of a substantial change in numbers at this site. The breeding pair estimate for 2016 differs slightly from Clark et al. (2019) (140,000; 95% CI: 90,000 – 210,000) due to analytical differences and additional data informing model predictions. Population estimates in the region of 100,000 breeding pairs were reported from a design-based survey in 2006 (Woods & Woods 2006), although it is unclear whether this estimate accounted for occupancy rate.

At **Top Island**, our estimate of **11,886 (CI: 6,906 – 19,053)** sooty shearwater breeding pairs represents a baseline value; no previous occupancy-corrected population estimates exist for comparison. We found fewer burrows per square metre (0.2 ± 0.02 burrows/m^2) compared to Poncet et al. (2012) who reported 0.30 ± 0.05 burrows/m^2 using transect lines. However, their estimate included all burrowing species combined, and excluded the central area of the island, over which SSWs are largely absent. These factors likely contributed to an overestimation of burrow density.

Burrow density was highest in coastal regions, consistent with previous findings (Poncet et al. 2012, Clark et al. 2019). Within this area, hotspots were evident predominantly in the northwest and northeast (Kidney Island) and along the west coast (Top Island) (**Figure 7**, **Figure 11**). Hotspots identified at Kidney Island were not persistent, showing intermediate level of change between seasons (**Figure 7**). The ground in these hotspot areas was extremely fragile due to the high number of burrows, requiring great care to avoid causing physical damage.

Our models showed that tussac density was an important driver of increased burrow occupancy and bird abundance at both islands, corroborating findings by Clark et al. (2019). Tussac density serves as a proxy for suitable habitat, as tussac provides shelter and retains and produces peat favourable for burrowing (see also Clark et al. 2019). Soil characteristics (depth and softness) are important for sooty shearwaters elsewhere (Jones et al. 2008, Charleton et al. 2009), and the loss of suitable breeding habitat is considered a contributing factor to declines in New Zealand (McKechnie et al. 2008). In the Falkland Islands, the conservation of tussac is critical for preserving suitable nesting substrate for this species.

We found lower sooty shearwater burrow occupancy and density in areas where other birds and pinnipeds were present. This may indicate that these areas provide unsuitable habitat for sooty shearwaters, or suggest competition between species (Rodríguez et al. 2019, Devincenzi et al. 2023). Understanding the numbers and distribution of sea lions across these islands could provide further insight into the spatial-temporal trends of burrowing seabird abundance.

White-chinned petrels

Breeding white-chinned petrels were found at both Kidney and Top Islands, validating their continued importance as ACAP breeding sites. Our breeding pair estimates provide a baseline for **Top Island** (**199; CI: 33 – 594**), and the first model-based estimate for this species at **Kidney Island** (**331; CI: 52 – 1043**). At Kidney Island, our estimate is not substantially different from previous design-based estimates by A. Stanworth unpubl. data (432 breeding pair, December 2015), and fall within the 100 – 1000 breeding pairs estimated by Woods & Woods (1997). Contrarily, using a full-census approach, Reid et al. (2007) reported only 23 pairs in 2005 and 27 pairs in 2006.

White-chinned petrel burrows were highly clustered, occurring mostly within 25 m from the shore along southern facing slopes in wet tussac peat. At Kidney Island, burrows were predominantly found along the landing bay – an area previously identified as a hotspot (Reid et al. 2007, A. Stanworth unpubl. data). In this area, Reid et al. (2007) reported only 6 and 9 occupied burrows in 2005 and 2006, respectively, using a full-census approach. A. Stanworth unpubl. data found 47 occupied burrows within a transect belt. In comparison, we located 24 occupied burrows along the landing bay using a sample-based approach. Reid et al. 2006 also found occupied burrows along the southern shore of Kidney Island where we located three unoccupied burrows in February. Given the timing, we cannot exclude the possibility of a temporal bias in the occupancy status of these burrows.

At Top Island, the main hotspot occurred in the western part of the southern shore. Poncet et al. (2012) also found most of the confirmed white-chinned petrel burrows along the southern side of the island, although a single confirmed burrow was also found in the north.

The white-chinned petrels' preference for damper soil, as shown by Model 3, is consistent with previous findings (Reid et al. 2007, Poncet et al. 2012, A. Stanworth unpubl. data). A changing climate is expected to decrease soil moisture content and impact soil health in the Falkland Islands, leading to increased erosion risks (Upson et al. 2016). Peatlands are particularly susceptible to this (Upson et al. 2016), posing a threat to suitable breeding habitat for white-chinned petrels (and other species).

Non-native fauna and flora

We found no evidence of invasive species on the islands during surveys, although we did not systematically search for these. Several non-native grasses had previously been recorded on both islands, although none are considered invasive (S. Poncet pers. comm.). The absence of rats and mice at Kidney and Top Islands has been confirmed annually over the last few years by South Atlantic Detection Dogs. Invasive mammals such as mice, rats, cats, rabbits and grazing livestock can be detrimental to burrowing petrel populations (including sooty shearwaters and white-chinned petrels) through predation (Jones 2000, Jones et al. 2008, Dilley et al. 2017) and habitat modification (Rodríguez et al. 2019). Invasive invertebrates and plants, as well as new pathogens, can present further threats (Rodríguez et al. 2019). Strict biosecurity can help minimise harmful impacts from non-native species to burrowing seabirds.

Survey considerations

Comparability with previous studies

This study aimed to establish population baselines and changes of sooty shearwaters and white-chinned petrels, and to provide a repeatable approach for longer-term monitoring. We therefore applied a comparable approach to previous studies (**Table 1A**), but introduced adaptations we felt would benefit a robust long-term monitoring programme. For example, previous studies did not always account for burrow occupancy (e.g. Poncet et al. 2012). Occupancy rates can differ in space and time (Bird et al. 2022, Rexer-Huber et al. 2023), and excluding this information can therefore bias population estimates and obscure population changes. Furthermore, previous studies for white-chinned petrels on Kidney Island were design-based (i.e. abundance was counted within a sampled area and extrapolated to the wider area of perceived suitable habitat). If certain habitats are over- or under-sampled, this approach can lead to biases (Mercker et al. 2021). We instead opted for a model-based approach, which relies on the relationship between species abundance and spatial and environmental covariates to infer abundance in space and/or time (Bird et al. 2022). Model-based analyses provide an improved understanding of the uncertainty around population estimates, quantifying habitat availability more realistically, and providing useful spatial representation of population distribution. Further, these approaches can quantify environmental drivers of population size and distribution, which is useful for management purpose (Clark et al. 2019, Bird et al. 2022).

Our chosen approach closely followed Clark et al. (2019) for sooty-shearwaters, but was adapted for white-chinned petrels given their lower numbers and patchy distribution (Bird et al. 2022). Nonetheless, there were two key differences in the field approach between our study and that of Clark et al. (2019) for sooty shearwaters.

Firstly, our data were collected in late November/early December (early incubation), whereas the 2016/2017 data were collected in January (late incubation/early chick-hatching). This timing difference may have introduced a small but unquantified bias due to the higher risk of failed breeders later in the season. Targeting the early incubation period is preferable to reduce this risk.

Secondly, occupancy was assessed differently. Whereas we assigned individual burrows as 1 (occupied) or 0 (unoccupied), Clark et al. (2019) determined occupancy using an indirect disturbance method which required repeat visits to individual burrows over a period of 10 days. While this method allows for modelling the probability of correctly or incorrectly detecting occupancy at each burrow, the data can still be subject to bias if, for example, disturbance is caused by prospecting birds. In addition, the need for repeated visits to assess occupancy is not conducive to a long-term programme, especially considering the difficulties of accessing the sites.

The differences in field approach between our study and that of Clark et al. (2019) likely have a minor, but in our opinion, insignificant effect on comparability. We further enhanced comparability by integrating the 2016 and 2023 data within a single analytical framework, thus eliminating the need to account for any analytical discrepancies.

Understanding uncertainty

Uncertainties associated with estimated breeding pairs are highlighted by the relatively wide credible intervals around the mean estimates. This is a common issue in burrowing petrel studies due to the difficulties in meeting the data requirements for monitoring these birds – specifically, occupancy and density observations – in challenging environments (Moller et al. 2009, Clark et al. 2019, Bird et al. 2022). While large credible intervals may impede the detection of small-scale population changes, they do not prevent the identification of broader patterns, temporal trends, and distribution shifts, which can serve as early indicators of small-scale fluctuations.

Our initial plan was to establish permanently marked survey plots within which all burrows would be counted and assessed for occupancy. Over time, this approach would theoretically allow us to work with a population index with reduced confidence intervals, making it easier to detect fine-scale population changes compared to using whole-island estimates (Hopkins & Kennedy 2004). However, due to the limited time window and the difficulties in determining burrow occupancy (see below), it was deemed unfeasible to deploy sufficient plots to cover a representative area of the habitat. Targeted searches that are non-representative of the habitat may superficially appear to perform better, but they carry an unquantifiable bias, undermining their use for detecting change (Bird et al. 2022). Although future advances in burrow occupancy assessment tools (e.g. detection dogs) may address this issue, maintaining permanently marked plots in loose tussac peat remains inherently challenging. Additionally, pinniped activity on these islands increases the risk of markers becoming dislodged.

For our current approach, adding more occupancy plots could enhance predictions and reduce uncertainty for sooty shearwaters. For the more patchily distributed white-chinned petrels, increasing survey effort in areas where they have previously been located would better inform the 'range' hyperparameter in the spatial model. Other studies on white-chinned petrels also recommend incorporating a mark-and-recapture element to better understand and inform population trends (Rexer-Huber et al. 2023), though this would require additional resources.

Accurate burrow identification

Population size estimates rely on accurate burrow identification. It can sometimes be difficult to distinguish between white-chinned petrel burrows and similar-sized Magellanic penguin burrows (see also Poncet et al. 2012). We mitigated this by assessing occupancy visually and audibly at each suitable burrow. Sooty shearwater burrows can be confused with those of other smaller petrels. While common diving petrels (*Pelecanoides urinatrix*), Wilson's storm petrels (*Oceanites oceanicus)* and grey-backed storm-petrels (*Garrodia nereis*) generally burrow higher in tussac pedestals (Woods 1970; although see current study), greater shearwaters (*Puffinus gravis*) and thin-billed prions (*Pachyptila belcheri*) also burrow into the ground (Stokes et al. 2021; this study). Our occupancy data indicates that only a minor proportion of burrows identified as sooty shearwater burrows actually housed a different species: Out of 115 burrows assessed, one contained a greater shearwater, and one held an unidentified storm petrel).

Accurate occupancy assessment

Model 1 highlighted challenges in estimating burrow occupancy, specifically with estimating the zero values. This difficulty could be due to (1) false zeros, (2) insufficient data, or (3) insufficient explanatory power of the tested variables. Sooty shearwater burrows, in particular, are long and convoluted, rendering occupancy assessment difficult and time-consuming. There is a possibility that complex burrows were on occasions mistakenly deemed completely searched and empty, when a bird was actually present. This could have introduced a small bias in the true occupancy. The use of detection dogs might offer a viable solution for enhancing occupancy data in future surveys.

Environmental considerations

The season of 2023/2024 may have been an atypical year for breeding seabirds. Firstly, the Highly Pathogenic Avian Influenza (HPAI) virus arrived in the Falkland Islands in October 2023 (Falkland Islands Government, unpubl. data). Although we did not detect signs of HPAI on Kidney or Top Island, it is unclear how surveyed species may have been impacted by the virus away from their breeding colony. In addition, an El Niño was recorded during the austral summer of 2023/2024 (NOAA 2024), which may have negatively impacted the breeding probability of surveyed species (see e.g. McKechnie et al. 2020).

Management recommendations

Mitigating threats

Sooty shearwaters were found in greater numbers in areas with dense tussac over peat, while white-chinned petrels preferred tussac peat with higher moisture content. Additionally, like most burrowing seabirds, sooty shearwaters and white-chinned petrels are vulnerable to invasive species. Effective site management for sooty shearwaters and white-chinned petrels should therefore focus on preserving healthy tussac habitat, maintaining suitable peat condition for burrowing, and ensuring the (continued) absence of invasive species. Specifically, we recommend the following actions:

- **Biosecurity** – Non-native, invasive mammals such as cats, rats and mice can significantly and negatively impact burrowing seabirds through predation. Introduced rabbits and livestock, as well as plants and invertebrates, can pose further threats by altering the habitat and rendering it unsuitable for breeding. Maintaining strict biosecurity is essential to prevent the unintentional (re-) introduction of smaller mammals, plants and insects, while intentional (re-) introduction of grazing livestock should be prohibited. Biosecurity measures should further aim to minimise the risk of human-induced spread of pathogens.
- **Visitor management** – Areas with higher burrow density (hotspots) are extremely vulnerable to footfall, and even with extreme care, burrow damage can occur. To minimise potential damage from visitors, access to these hotspots should be restricted. Guides accompanying visitors unfamiliar with the site should further minimise impact by both limiting access to hotspots and educating visitors about the area's sensitivity.
- **Tussac islands fire response plan** – A changing climate is predicted to increase the frequency and intensity of extreme weather such as drought and severe storms (Easterling et al. 2000), including in the Falkland Islands (see e.g. Upson et al. 2016, Ventura et al. 2023). A drying environment and increased occurrence of electrical storms increases the risk of wild fires, which can cause direct mortality of nesting birds, long-term habitat loss and loss of nesting substrate. This could have significant population-level impacts at a national scale for these sites. A fire response plan would minimise the impact from such fires.

Monitoring

Continued monitoring is necessary to understand the health of the sooty shearwaters and white-chinned petrel populations, inform management and meet ACAP commitments. Our study highlights the inherent challenges associated with estimating population size of burrowing birds and accessing Stanley's tussac islands during the narrow time window. Going forward, we recommend the following:

- **Sooty shearwaters:** To increase certainty in estimates, the number of occupancy plots surveyed would ideally be increased, and distributed to capture the full habitat variability, as done on Top Island. If necessary, the number of burrows checked per plot could be reduced from 5 to 3.
- **White-chinned petrels:** Increased survey effort within areas where burrows have been located would help improve estimates and spatial predictions. Additional areas should continue to be surveyed as per the approach detailed in the current study. Studies elsewhere recommend incorporating a mark-and-recapture element to better understand and inform white-chinned petrel population trends, although this would require further resources.
- **Survey frequency:** A repeat survey with additional effort within a year or two would allow these data to refine the initial baseline models. Subsequent surveys could then be conducted less frequently (~every 4 – 5 years).
- **Other environmental variables:** Monitoring additional dynamic factors that influence sooty shearwater and white-chinned petrel habitat availability/suitability (e.g. vegetation and soil characteristics, and number and distribution of other birds and pinnipeds) could provide further insight into spatial-temporal trends of these species.
- **Resourcing**: In order to implement the above, additional on-the-ground resources would be required to maintain data quality, and to retain flexibility in relation to weather windows and transport availability.

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Appendix

Past and present surveys

Table 1A Summary of surveys that have estimated numbers of sooty shearwaters (SSW) and white-chinned petrels (WCP) on Kidney and Top Islands. Occ. corr = occupancy corrected.

Island	Spp.	Season	Pop. estimate	Approach taken	Reference
Kidney	SSW	1960	2000	Design-based, unclear if occ. corr. estimate	Woods 1970
		1996	10,000 – 20,000	Design-based, unclear if occ. corr. estimate	Woods and Woods 1996
		2006	100,000	Design-based, unclear if occ. corr. estimate	Woods and Woods 2006
		2016/17	140,000 (95% CI: 90,000 – 210,000)	Model-based; fieldwork early to mid-Jan; occ. corr. estimates	Wakefield et al. 2017; Clark et al. 2019
		2016/17	122,566 (95% CI: 87,154 – 166,991)	Model-based; fieldwork early to mid-Jan; occ. corr. estimates	This study
		2023/24	130,999 (95% CI: 94,938 – 175,735)	Model-based; fieldwork late Nov; occ. corr. estimates	This study
	WCP	1960 – 1962	100s	Mark and recapture	Woods 1975
		1960s	50 – 60	Mark and recapture	I. Strange (pers. comm. in Reid et al. 2007
		1992	100-1000	Design-based; fieldwork Dec	Woods and Woods 1997
		2005/06 2006/07	23 pairs (2005) 27 pairs (2006)	Counts of occupied burrows where birds were observed to land; fieldwork mid-Jan.	Reid et al. 2007
		2015/16	432; 0.02 pairs/m^2	Design-based; fieldwork mid-Dec; occ. corr. estimates	A. Stanworth unpubl. data
		2023/24	331 (95% CI: 52.2 – 1043.0); 0.01 pairs/m^2	Model-based; fieldwork late Nov; occ. corr. estimates	This study
Top	SSW	2012/13	0.30 ± 0.05 burrows/m^2 (20,000 burrows (15,100 – 25,400) for all spp.	Design-based; fieldwork Sep – Oct. Estimates not occ. corr.	Poncet et al. 2012
		2023/24	11,886 (95% CI: 6,906 – 19,053)	Model-based; fieldwork, early Dec; occ. corr. estimates.	This study
	WCP	2012/13	“10s”; 0.30 ± 0.05 burrows/m^2 (20,000 burrows (15,100 – 25,400) for all spp.	Design-based; fieldwork Sep – Oct. Estimates not occ. corr.	Poncet et al. 2012
		2023/24	199 (95% CI: 33.0 – 594); 0.01 pairs/m^2	Model-based; fieldwork, early Dec; occ. corr. estimates.	This study

Design based approach = Simple extrapolation of raw numbers within known area to total area of perceived suitable habitat.

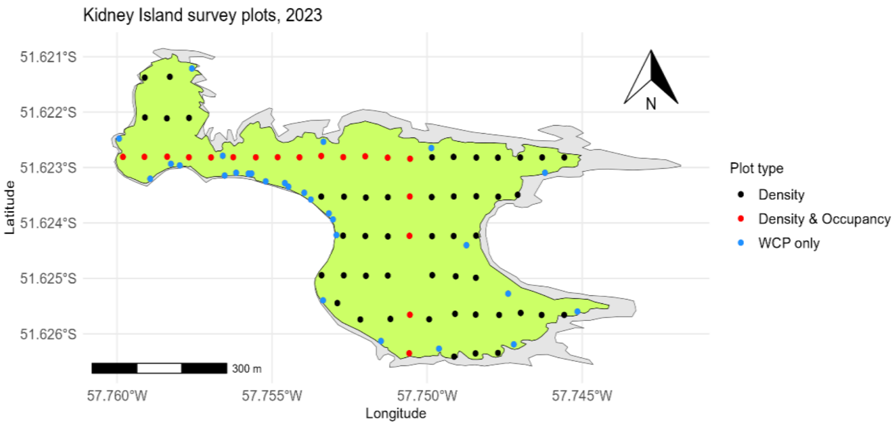


Figure 1A: The position of the 2016/2017 survey plots on Kidney Island. Source: Wakefield et al. 2017.

Environmental covariates: Additional details

NDMI values typically range from ~0.1 – 0.2 for bare soil to 0.3 – 0.8 for vegetation (Rouse Jr. et al. 1974). NDVI was calculated as

$$\mathrm{NDVI} = \frac{(R_{NIR} - R_{VIR})}{(R_{NIR} + R_{VIR})} = \frac{(band\ 8 - band\ 4)}{(band\ 8 + band\ 4)}$$

where R_{NIR} and R_{VIR} denote the reflectance measured in the near-infrared and visible red bands, respectively, while band 8 and band 4 refer to the corresponding Sentinel-2 bands.

NDMI was calculated as:

$$\mathrm{NDMI} = \frac{(R_{VNIR} - R_{SWIR})}{(R_{VNIR} + R_{SWIR})} = \frac{(band\ 8A - band\ 11)}{(band\ 8A + band\ 11)}$$

where R_{VNIR} and R_{SWIR} denote the reflectance measured in the red visible near-infrared and short-waved infrared bands, respectively, while band 8A and band 11 refer to the corresponding Sentinel-2 bands.

Table 2A Summary of covariates considered in the various models (✓) (See **Table 2**). Use of smoothers are noted as s_{cr} = cubic regression spline.

Covariate Name	Covariate type	Potential effect on burrowing sea-birds burrow numbers/occupancy	M1	M2	M3
Year	Factor, fixed effect	Defines time	✓	✓	
Island	Factor, fixed effect	Defines space	✓	✓	✓
Plot id	Factor, random effect	Defines space	✓		
Fauna	Factor, fixed effect	Competition	✓	✓	✓
Soil	Continuous, fixed effect	Affects ease of burrow construction			✓
Dist. to Shore	Continuous, fixed effect	Affects take-off	✓		
Elevation	Continuous, fixed effect	Affects take-off		✓(s_{cr})	✓
Aspect	Continuous, fixed effect	Affects take-off			
Slope	Continuous, fixed effect	Affects take-off		✓(s_{cr})	
Tussac Density	Continuous, fixed effect	Provides protection from weather and predators, improves drainage, produces and retains peat	✓	✓	
Tussac Height	Continuous, fixed effect	Provides protection from weather and predators		✓	
NDVI	Continuous, fixed effect	Provides protection from weather and predators, improves drainage			
NDMI	Continuous, fixed effect	Affects ease of burrow construction			

Model 1: Additional details

 # Data exploration

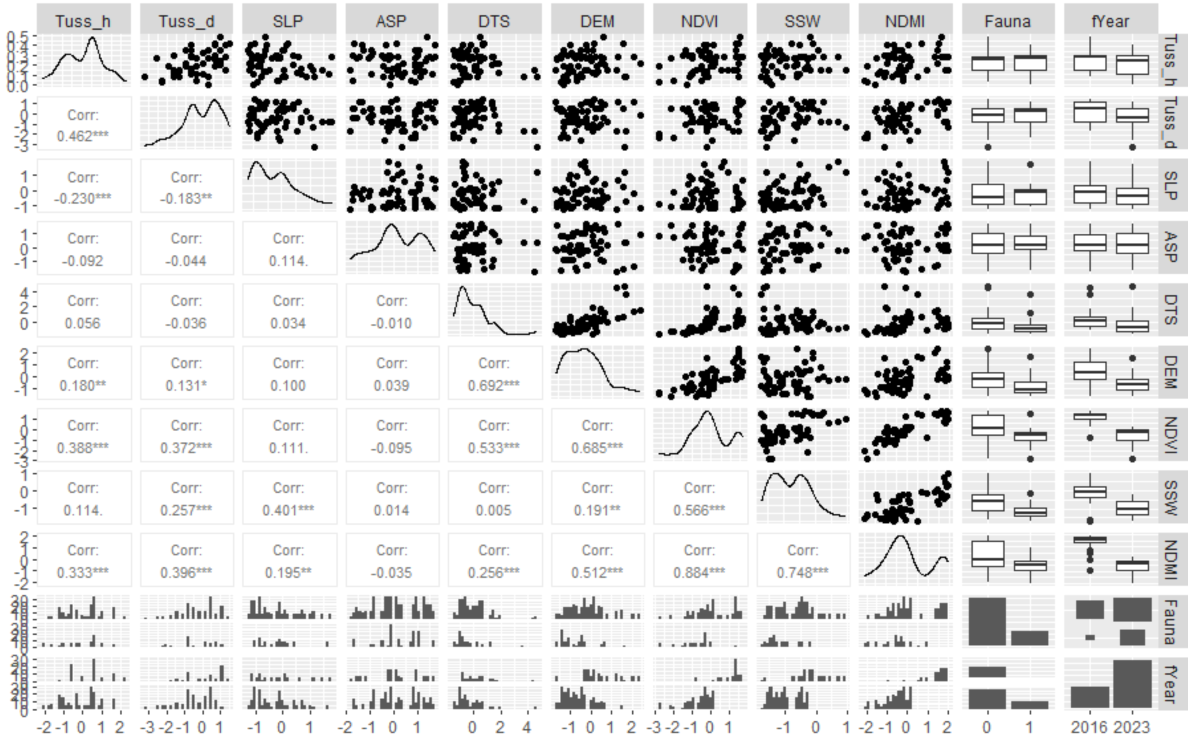


Figure 2A: Correlation matrix of covariates considered for Model 1.

Model validation*

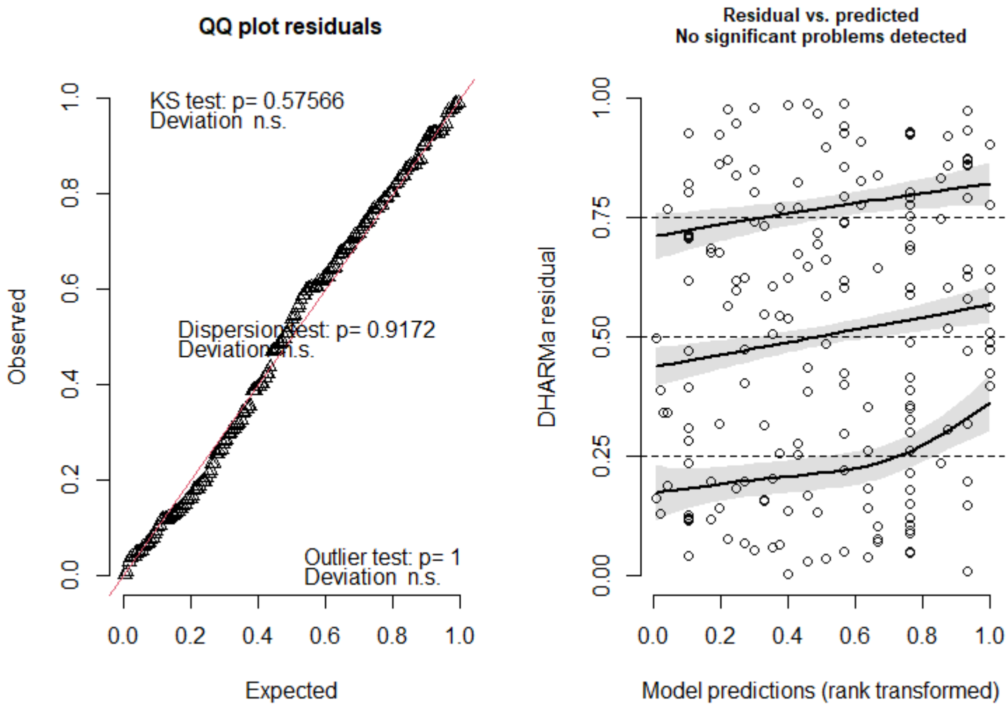


Figure 3A: Distribution of the Pearson's residuals of continuous variables in Model 1.

Model 2: Additional details

 # Data exploration

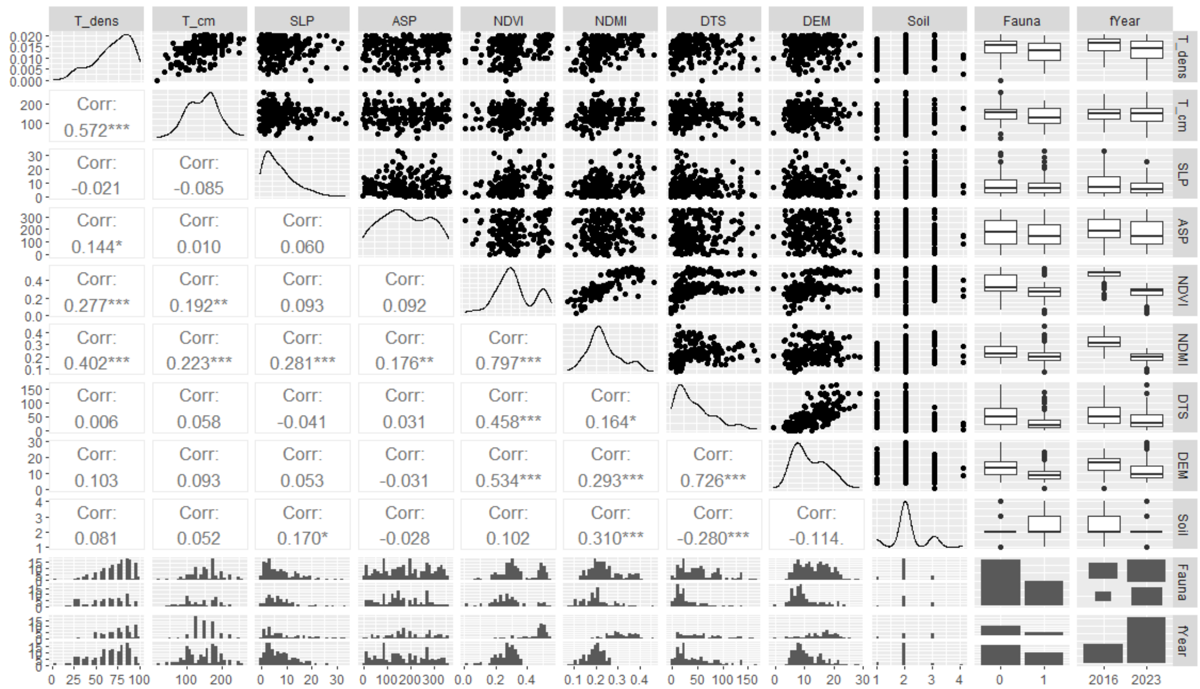
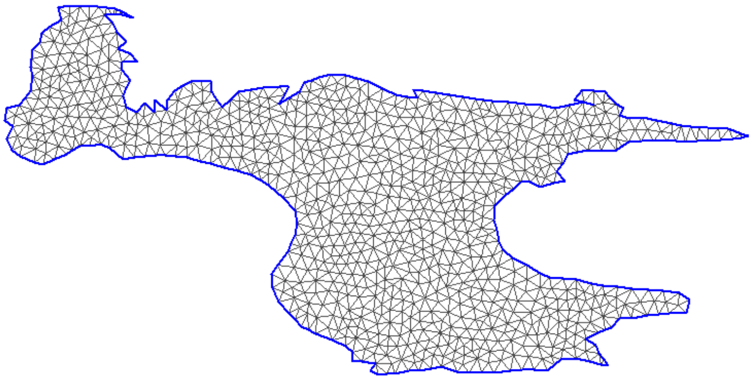


Figure 4A: Correlation matrix of covariates considered for Model 2.

Spatial Random Field

A



B

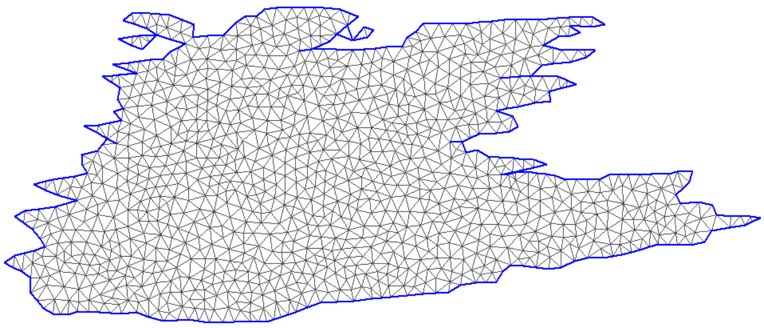


Figure 5A: Mesh used for the spatial random field in Models 2 and 3 for Kidney Island (A) and Top Island (B). Range = 0.15.

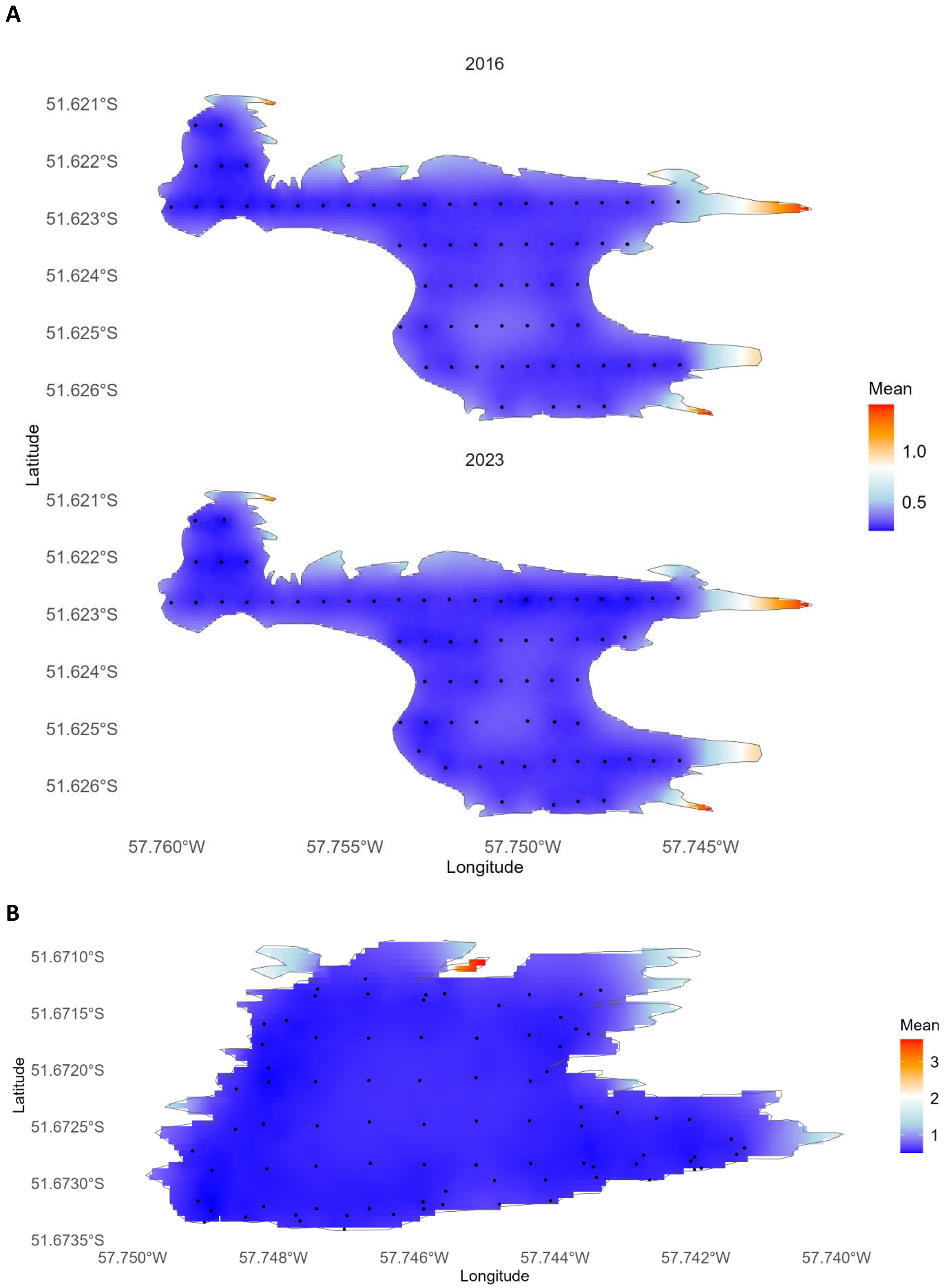


Figure 6A: Standard deviation of the spatial random field fitted for Model 2 at Kidney Island (A) and Top Island (B).

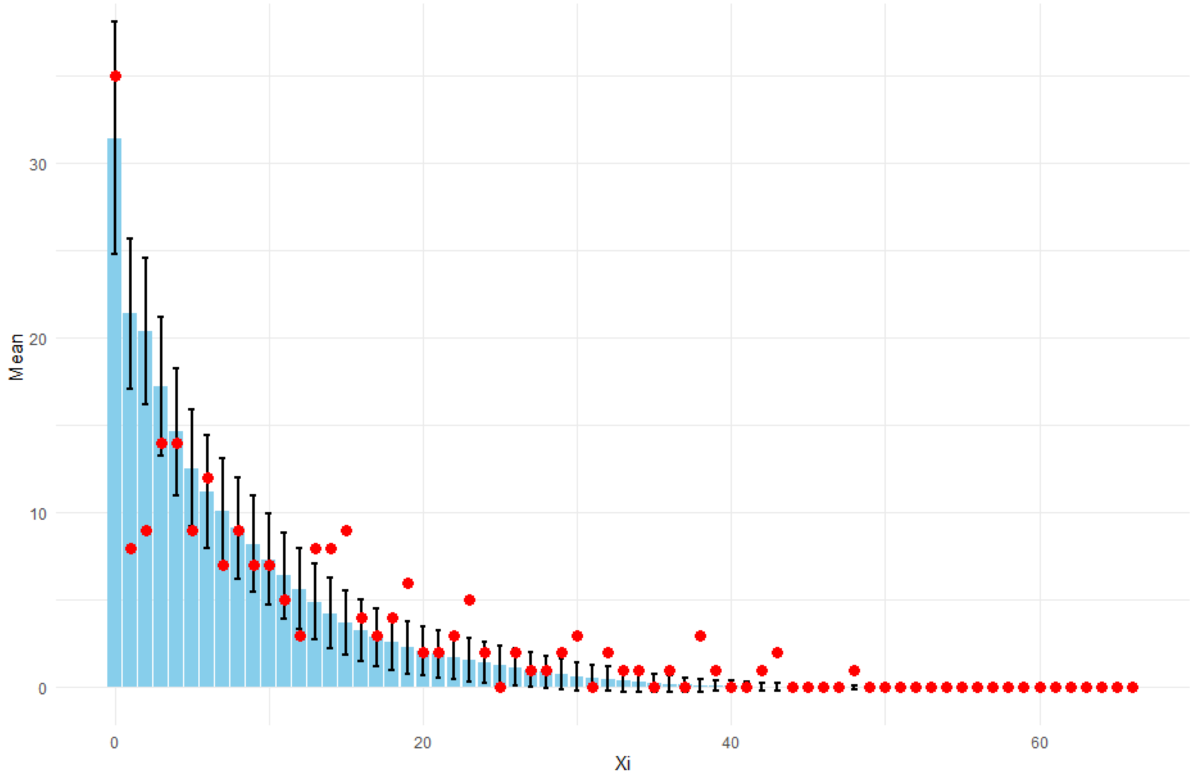


Figure 7A: Observed values (red) fitted against simulated data (blue bars with credible intervals) in the 10,000 datasets that were fitted from Model 2.

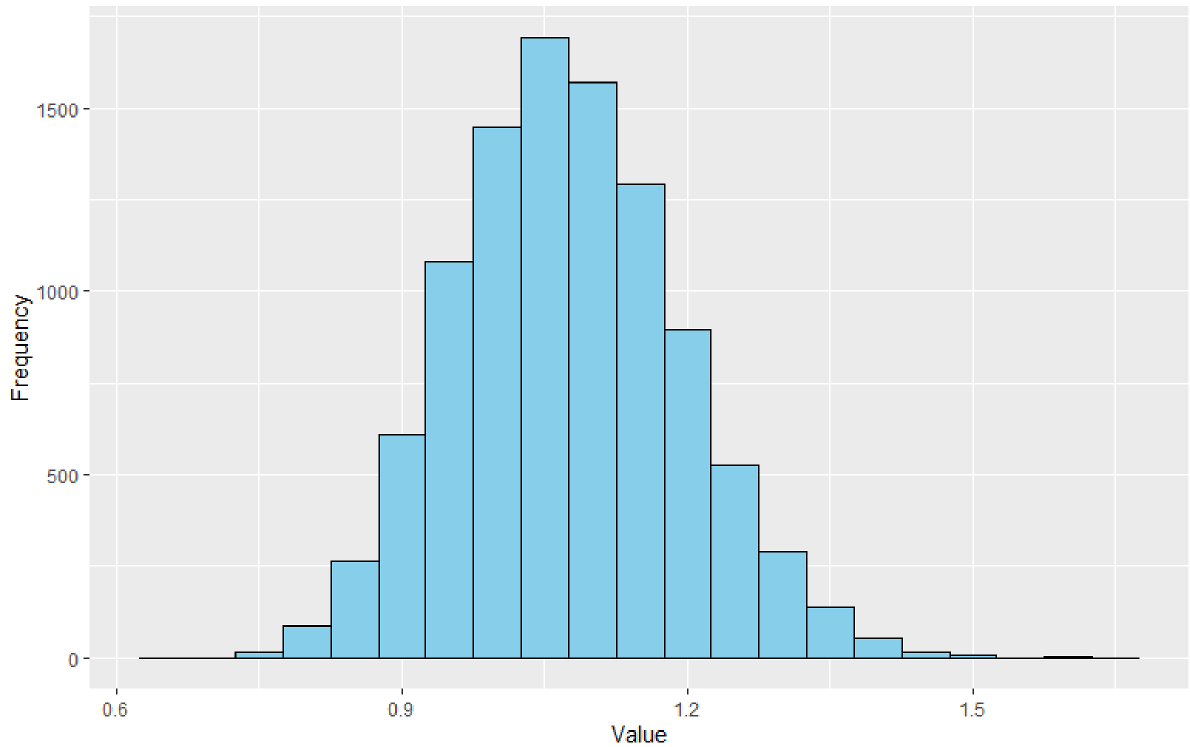


Figure 8A: Frequency distribution of the dispersion statistic in the 10,000 simulated datasets that were fitted from Model 2.

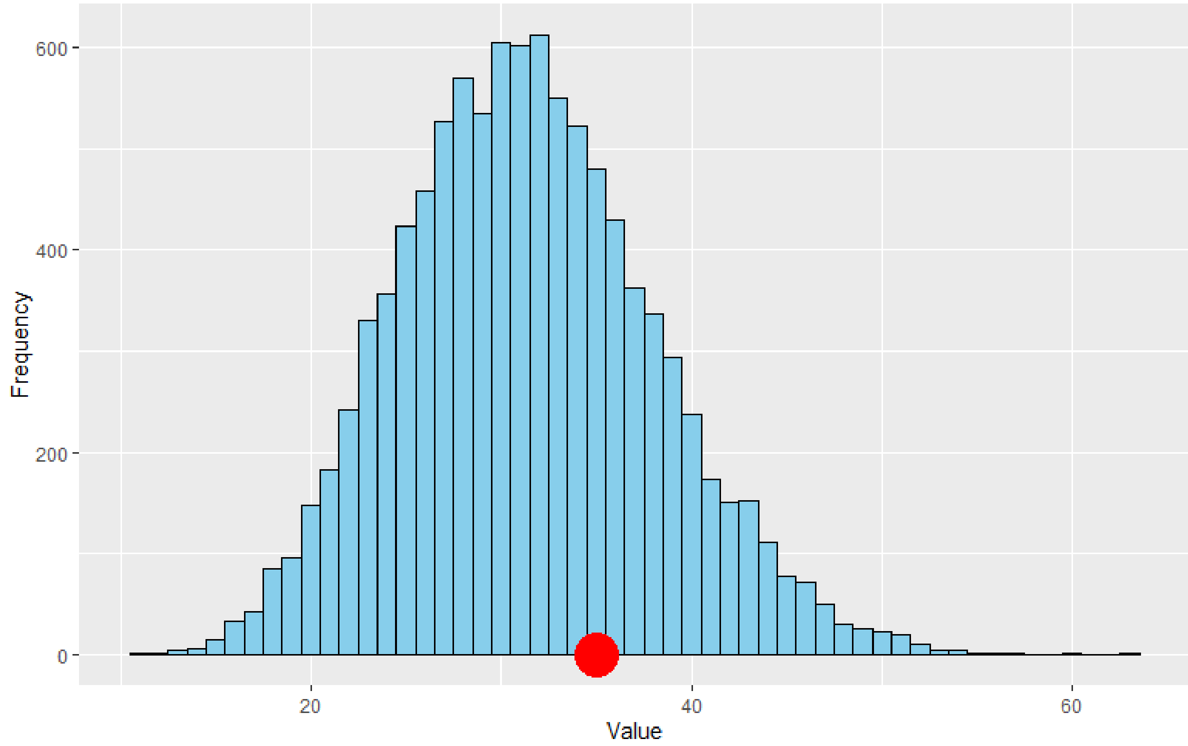


Figure 9A: Simulation study showing the observed number of zeros (red dot) and the simulated number of zeros in 10,000 data sets that were fitted from Model 2.

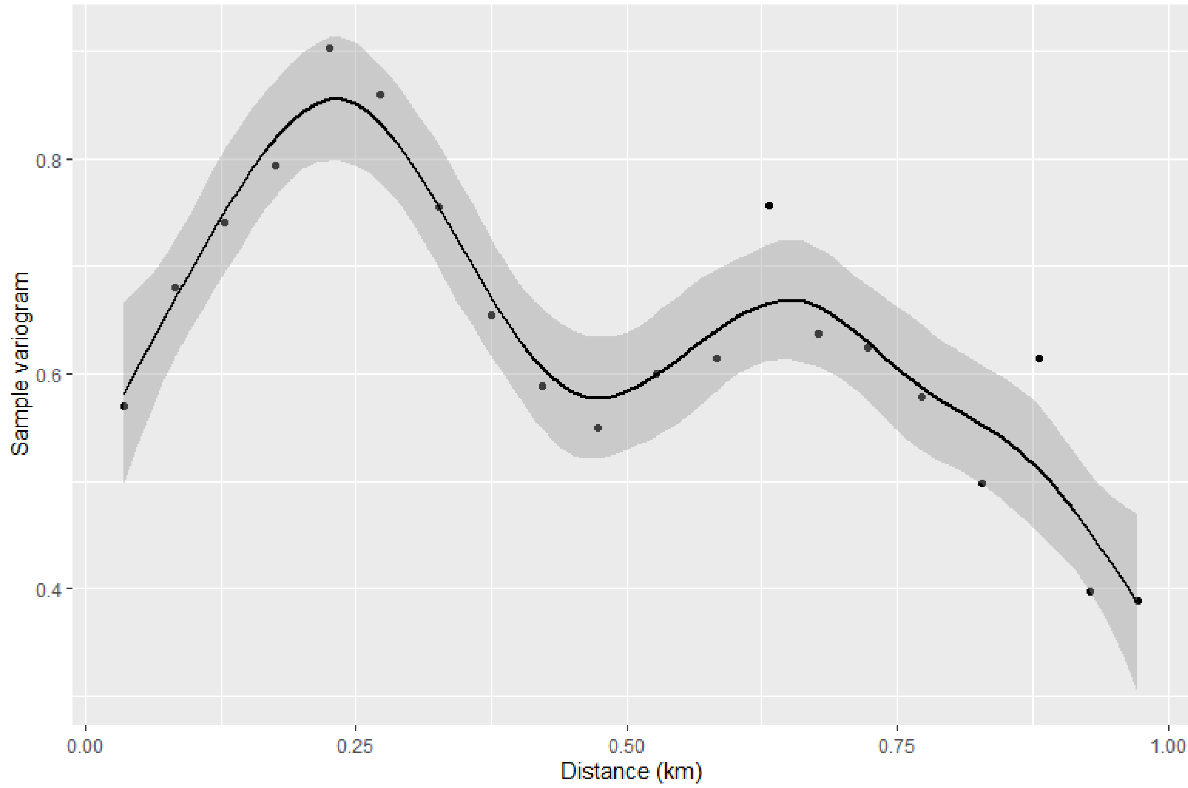


Figure 10A: Sample variogram of the residuals to identify outstanding correlation in the residuals of Model 2.

Model 3: Additional details

 # Data exploration

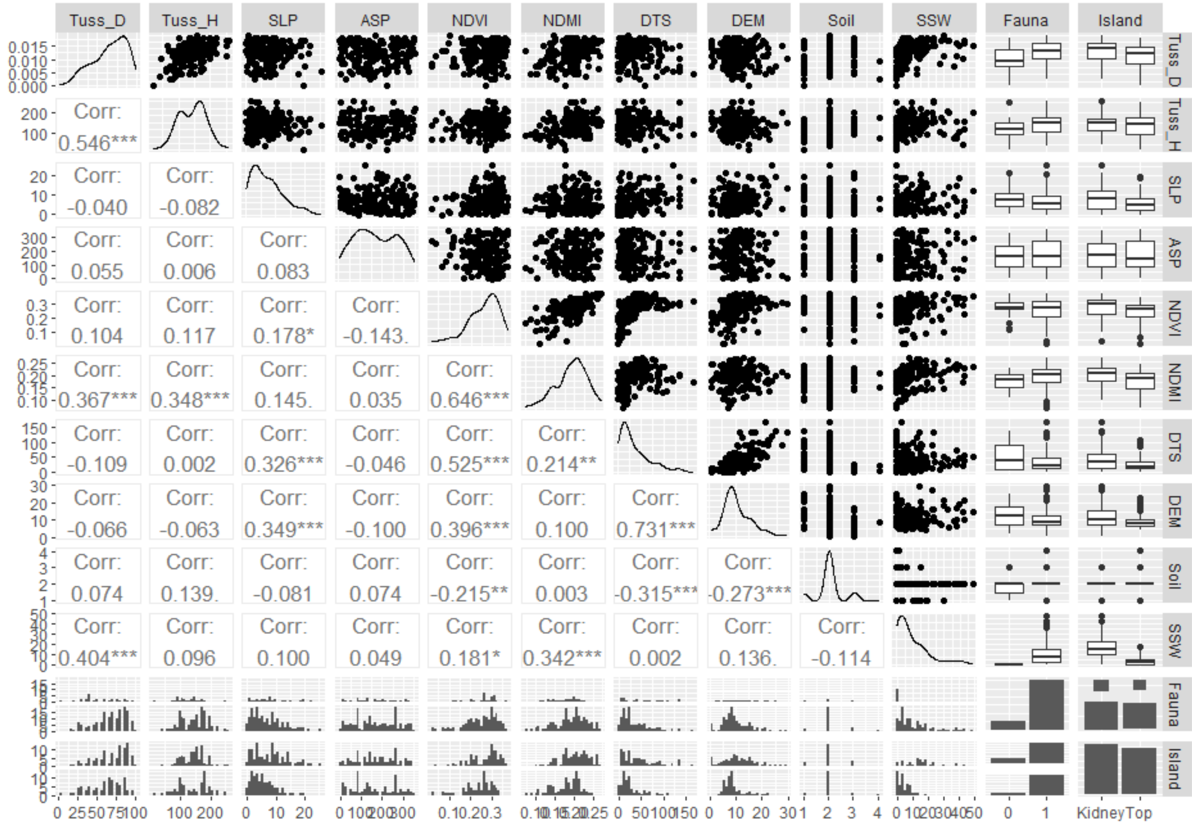
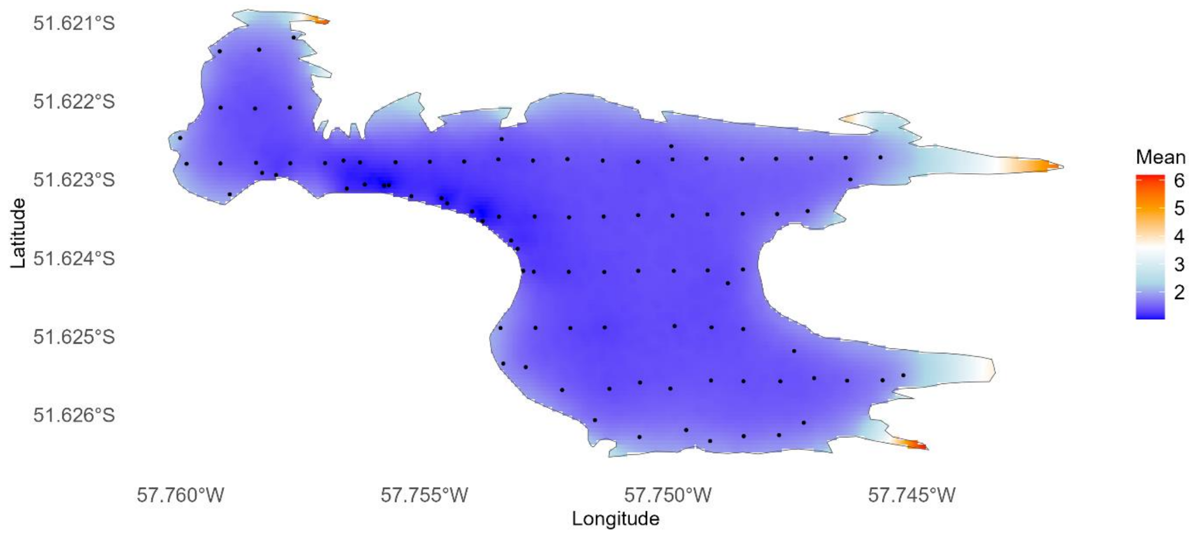


Figure 11A*:* Correlation matrix of covariates considered for Model 3.

 # Spatial Random Field

See Model 2.

A



B

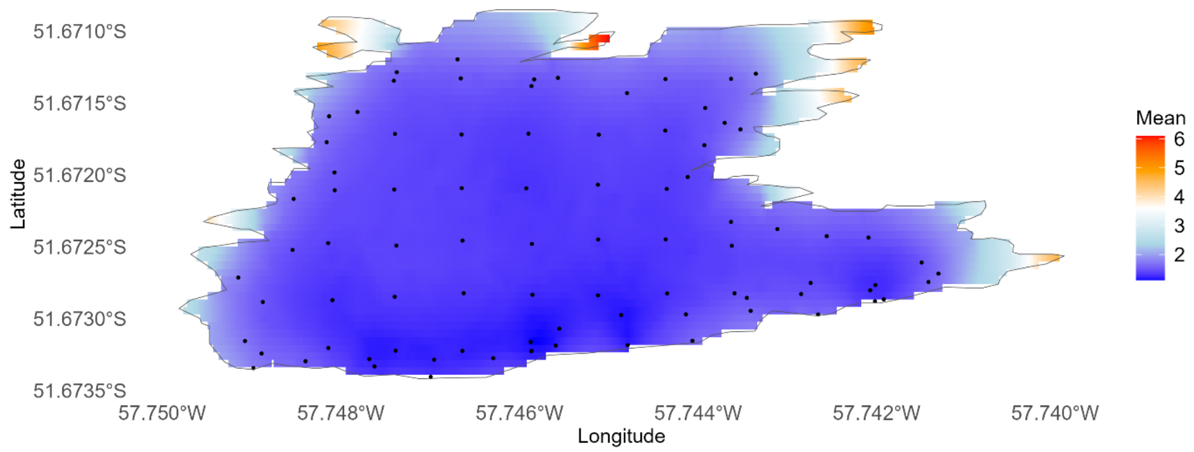


Figure 12A: Standard deviation of the spatial random field fitted for Model 3 at Kidney Island (A) and Top Island (B).

Model validation

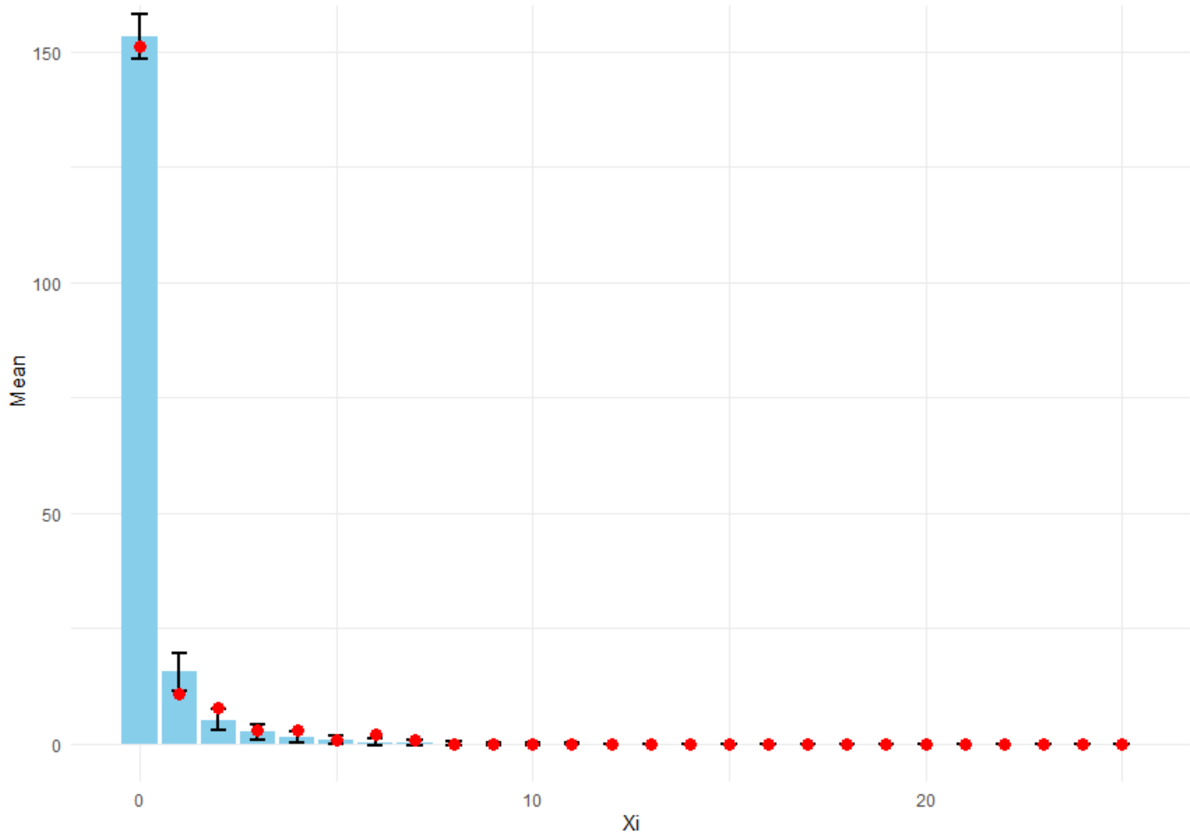


Figure 13A: Observed values (red) fitted against simulated data (blue bars with credible intervals) in the 10,000 datasets that were fitted from Model 3.

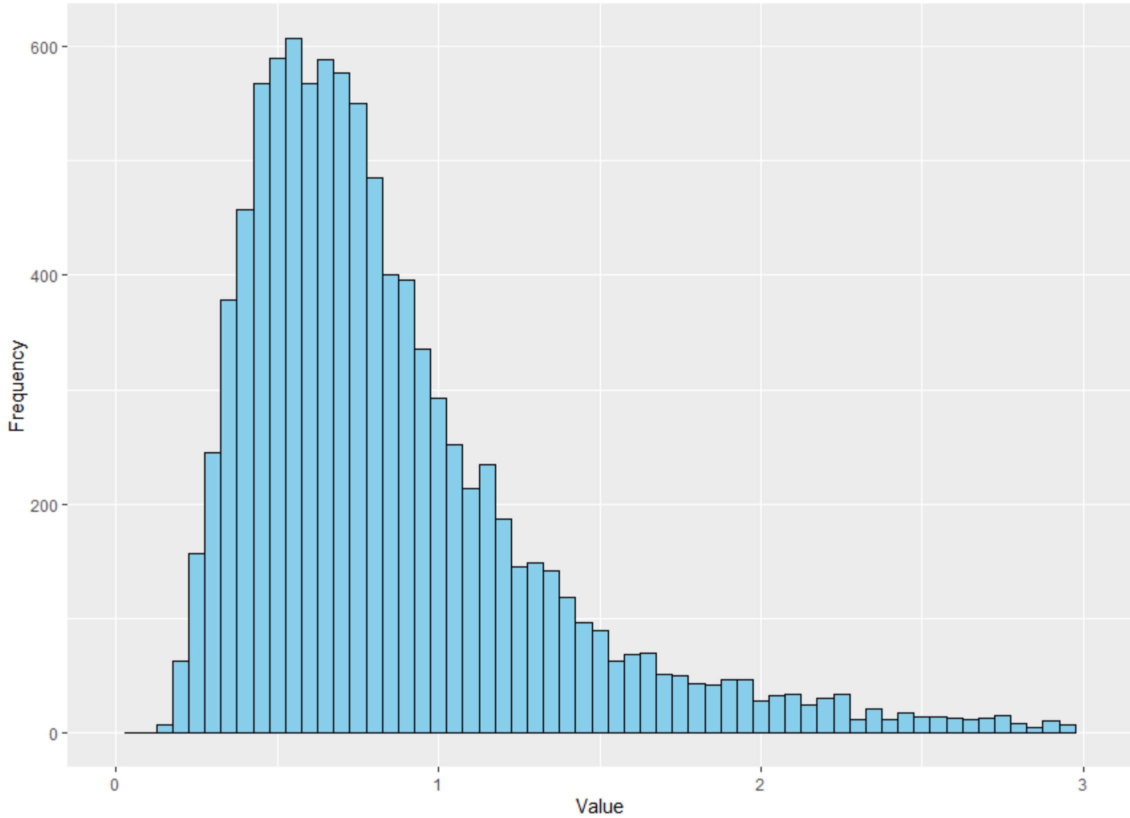


Figure 14A: Frequency distribution of the dispersion statistic in the 10,000 simulated datasets that were fitted from Model 3.

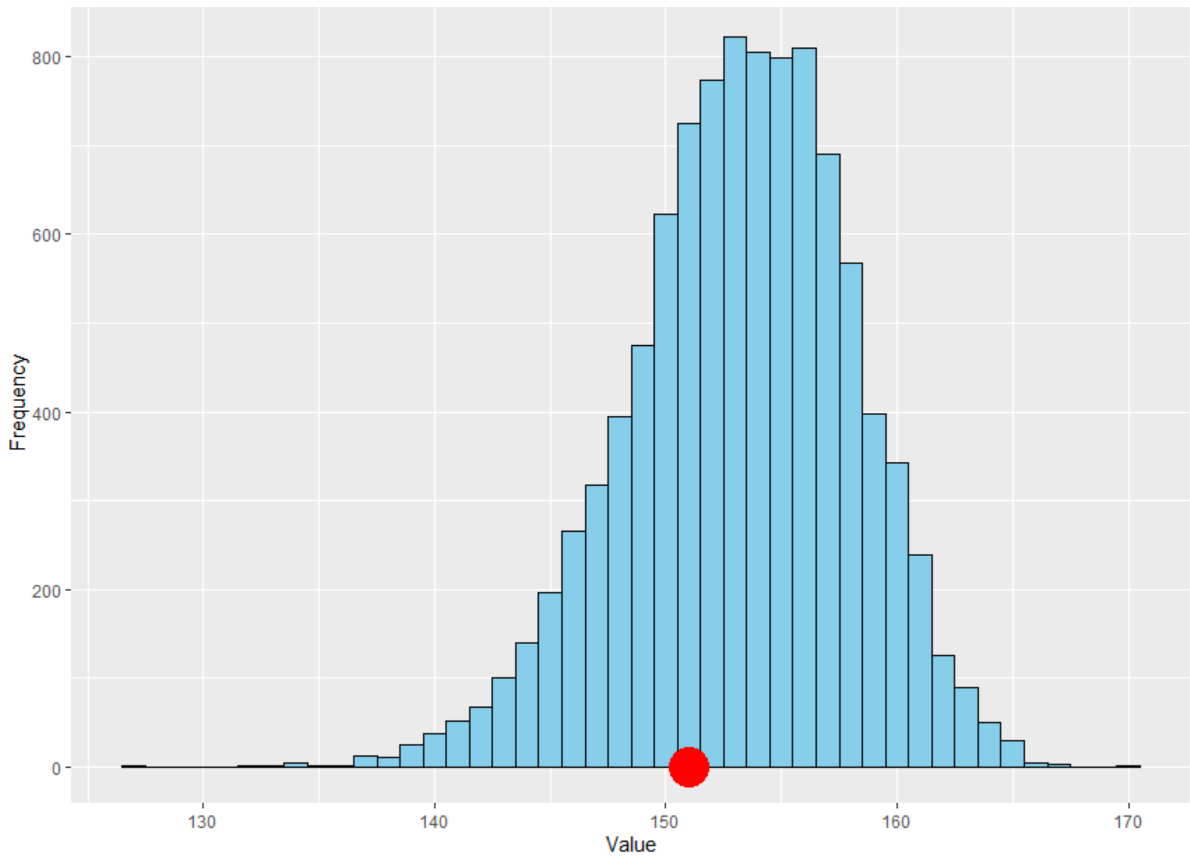


Figure 15A: Simulation study showing the observed number of zeros (red dot) and the simulated number of zeros in 10,000 data sets that were fitted from Model 3.

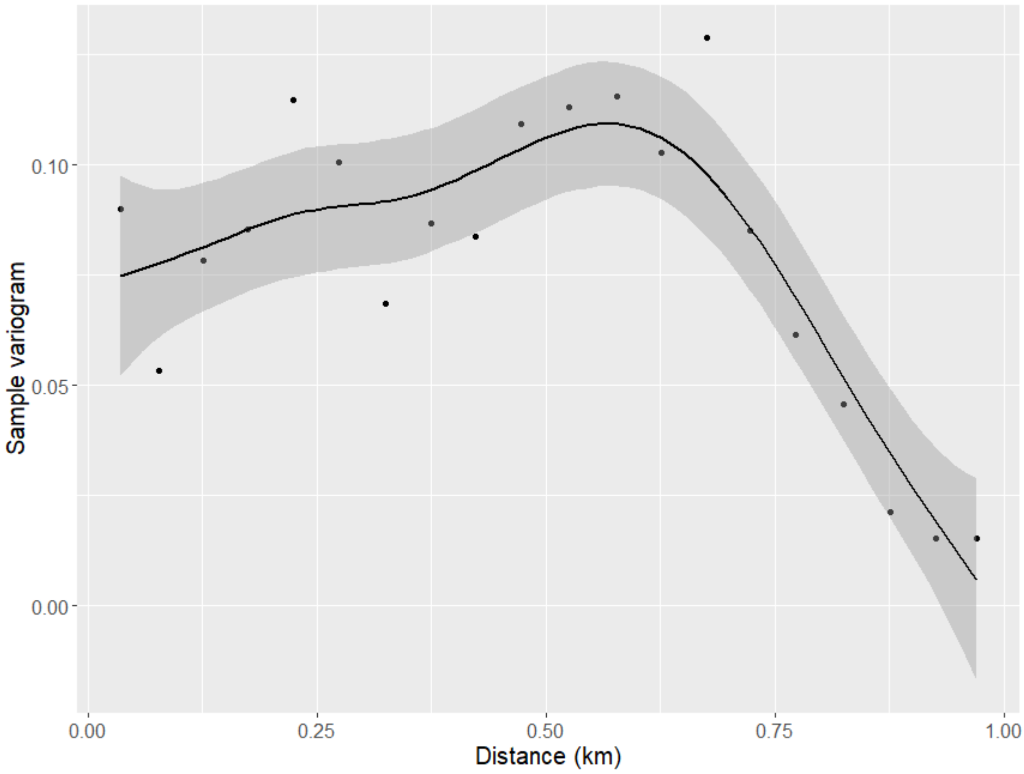


Figure 16A: Sample variogram of the residuals to identify outstanding correlation in the residuals of Model 3.

Appendix References

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